Evaluation of Near-Road Air Pollution at Environmental Justice (EJ) and Non-EJ Communities in the Greater Detroit Area

Final report to the Michigan Department of Environment, Great Lakes, and Energy (EGLE)

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Executive Summary

Overview

The Michigan Department of Environment, Great Lakes and Energy (EGLE) monitored several pollutants next to Highways I-96 and I-275 during 2016-2017. The pollutants were black carbon (BC), ultrafine particles (UFP), toxic carbonyls, toxic metals such as arsenic and chromium, fine particles ($PM_{2.5}$), carbon monoxide (CO), nitrogen dioxide (NO_2), and benzene, toluene, ethylbenzene, and xylene (BTEX). There were two monitoring locations: one in the environmental justice (EJ) community of Brightmoor, at Eliza Howell Park, and one in the non-EJ area of Livonia, 8 miles away.[1]

This suite of measurements will help us

(1) Quantify near-road air toxics concentrations in an environmental justice (EJ) area of Detroit next to I-96 (Eliza Howell Park), and a non-EJ area eight miles away next to I-275 (Livonia);

(2) Compare near-road toxics concentrations to urban "background" concentrations, to national concentrations, and to health benchmarks; and

(3) Evaluate concentrations under different traffic conditions: 138,000 vehicles per day at Eliza Howell (5,600 of which are commercial trucks) versus 200,000 per day at Livonia (12,200 of which are commercial trucks).

Why Near-Road?

Air pollution near roadways is typically higher than elsewhere in urban areas. As described on EPA's Near Road Research Page[2]:

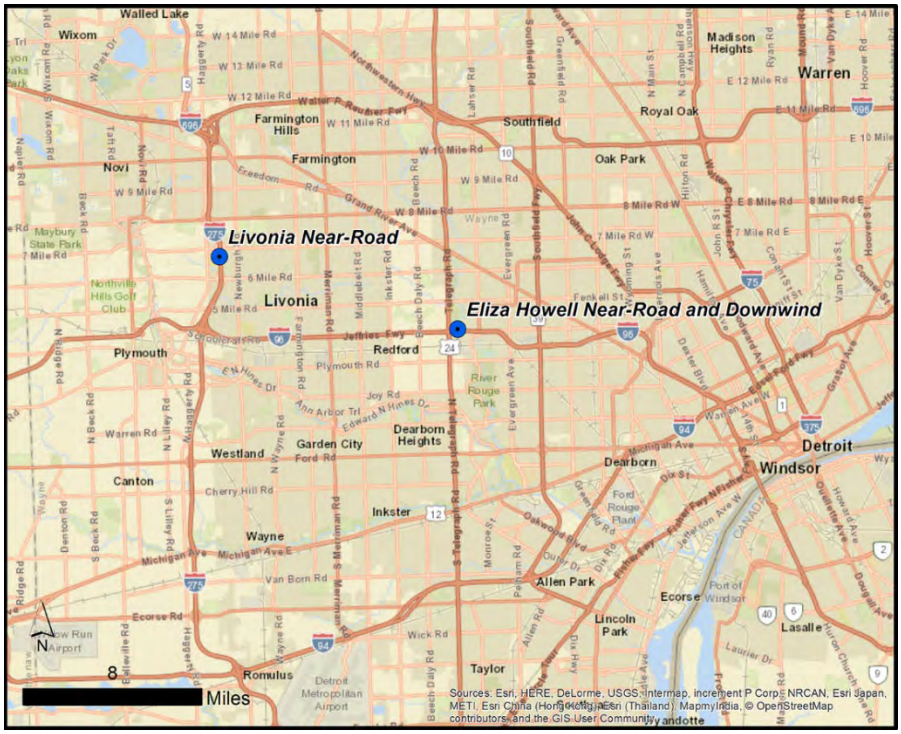
"With more than 45 million people in the U.S. living within 300 feet of a major transportation facility or infrastructure, notably busy roads, there is concern about the potential health impacts from air pollutants emitted from cars, trucks and other vehicles. Research has demonstrated that exposure to pollutants emitted from motor vehicles can cause lung and heart problems and premature death."

[1] "Environmental justice" is meant to ensure that economically disadvantaged communities are not unfairly exposed to more risk from pollution than other communities. This study intentionally compares a disadvantaged EJ community to a non-EJ community while also studying pollution near major roads.

[2] https://www.epa.gov/air-research/research-near-roadway-and-other-near-source-air-pollution

Communities

- The Eliza Howell Park monitoring site is located in the Brightmoor EJ community of Detroit. Poverty in census tracts around the site can be in excess of 70%, and excess cancer risk is relatively high (for the United States), at about 80 excess cases per million.
- The Livonia monitoring site is in the city of Livonia, a non-EJ area that includes low-density suburban subdivisions, shopping malls, and other land uses.



Map of monitoring site locations.

Findings

- Concentrations of most pollutants are lower than during a prior study by EPA/FHWA in 2010-2011, meaning that pollution levels next to the roadway tended to decrease over time; only BC was higher than in 2010-2011.
- Concentrations are highest 10 meters (m) from the roadway, and they drop off substantially 50 m to 100 m away.
- Concentrations of toxic carbonyls and metals, such as manganese, chromium, and nickel, were highest 10 m from the roadway at Eliza Howell but were significantly lower at Livonia; these concentrations were generally similar to the range seen across the United States.
- Concentrations of these toxic carbonyls and metals were below one-in-a-million cancer risk levels, with the exception of arsenic; however, arsenic concentrations were typical of other Midwestern cities and were not unusually high.

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Abstract

According to the U.S. Environmental Protection Agency (EPA), more than 45 million people live within 300 feet of a major roadway. Poor air quality in such near-road environments contributes to adverse health outcomes as well as health disparities due to race and income between communities and populations. During 2010 and 2011, the U.S. Federal Highway Administration (FHWA) and the EPA measured mobile source air toxics (MSATs) adjacent to Interstate Highway 96 (I-96) in an environmental justice (EJ) community, Eliza Howell Park, in Detroit, Michigan. Our study, conducted six years later in 2016 and 2017, collected measurements of carbon monoxide (CO), nitrogen oxide (NO), nitrogen dioxide (NO_2), oxides of nitrogen (NO_x), black carbon (BC), particulate matter less than 2.5 microns ($PM_{2.5}$), ultra-fine particles (UFP), total suspended particulate (TSP) metals, and carbonyls in Eliza Howell Park (EH-NR) and in Livonia, a non-EJ near-road location adjacent to I-275 (LIV-NR) 8 miles away. Hourly average concentrations of BC, $PM_{2.5}$, CO, and $NO/NO_2/NO_x$ at the Eliza Howell near-road site are from 1.4 times greater (CO) to 4.6 times greater (NO) than at the Livonia near-road site and from 1.3 times greater (CO) to 4.5 times greater (NO) than at a downwind monitoring site at Eliza Howell Park (EH-DW). Average hourly $PM_{2.5}$ concentrations at the Eliza Howell near-road site were 12.0 $\mu g/m^3$, compared to 7.5 $\mu g/m^3$ at Livonia, while hourly average BC concentrations at Eliza Howell were 1.2 $\mu g/m^3$ compared to 0.68 $\mu g/m^3$ at Livonia. Concentrations of NO, NO_2, and NO_x were 8-28% lower during our study than in the EPA/FHWA study, and BC was higher, even though overall traffic volume was similar (AADT of 154,500 in 2016 versus 165,300 in 2010). Twenty-four hour carbonyl concentrations were also lower than in the EPA/FHWA study; acetaldehyde in particular was 77%-80% lower (2.05-2.67 ppb average in the EPA/FHWA study versus 0.46-0.53 ppb here), while formaldehyde concentrations were 29%-30% lower (2.6-3.7 ppb average in the EPA/FHWA study versus 1.8-2.6 ppb here), suggesting that roadway emissions of these toxics have significantly decreased over approximately the past six years. These results serve to document multi-year concentration trends and establish baseline data for future-year comparisons, which can be used to evaluate control program impacts and allow EPA to demonstrate the effectiveness of new vehicle emissions requirements.

Introduction

Background on Near-Road Air Quality

Numerous studies have shown links between adverse health outcomes and exposure to traffic-related air pollutants, especially for those in the areas adjacent to busy roadways with large volumes of truck traffic. Adverse effects include reduced lung function, asthma, cardiovascular disease and premature death (Health Effects Institute, 2010; Batterman, 2013; Brandt et al., 2014; Ghosh et al., 2016; Kan et al., 2007; Kim et al., 2004; 2018; Rowangould, 2013; Ritz et al., 2014; Rioux et al., 2010). Concentrations of air pollutants such as nitrogen dioxide (NO_2), black carbon (BC), and carbon monoxide (CO) are typically higher near roadways than elsewhere in the urban environment, while concentrations of particulate matter (PM) are modestly higher (Baldauf et al., 2008; Karner et al., 2010; Zhang et al., 2004; Zhu et al., 2002a; 2009; Jeong et al., 2019; Brown et al., 2014; Hagler et al., 2010). Pollutant levels decrease with increasing distance from the roadway and typically decrease to urban levels within a few hundred meters (approximately 500 feet), though the rate of decrease varies by pollutant and meteorological conditions (Karner et al., 2010; Health Effects Institute, 2010).

Ultrafine particles (UFP), which are typically defined as particles that are less than 100 nanometers in diameter, dominate total particle number counts in ambient air but contribute little to $PM_{2.5}$ mass overall (de Jesus et al., 2019; Health Effects Institute, 2013). In urban areas, UFP are predominantly from mobile source emissions (Riddle et al., 2008; Marmur et al., 2006; Kleeman et al., 2009; Kulmala et al., 2004; Kumar et al., 2014; Salma et al., 2014; Harrison et al., 2011), though other sources such as airports, natural gas combustion, and cooking can also contribute (Venecek et al., 2019; Kumar et al., 2014; Ham and Kleeman, 2011). UFP can be emitted directly from combustion, or formed in the atmosphere via nucleation or condensation on pre-existing particles (Németh et al., 2018; Kulmala et al., 2004; Holmes, 2007; Brines et al., 2015; Salma et al., 2014; Canagaratna et al., 2010; Robinson et al., 2007).

In urban areas, UFP concentrations can vary widely, even while particulate matter less than 2.5 microns ($PM_{2.5}$) is more homogenous (de Jesus et al., 2019). UFP concentrations are typically highest next to roadways, especially those with a higher number of heavy-duty vehicles (HDVs) (Harrison et al., 2011), and rapidly fall off within 100-300 m of the roadway (Karner et al., 2010; Zhu et al., 2002b), and with strong spatial gradients within cities (Venecek et al., 2019). Concentrations at urban locations in the 2000s were 7-11 x 10^3 particles/cm^3 for urban areas and 35-48 x 10^3 particles/cm^3 for a range of roadside or street canyon monitoring sites (Health Effects Institute, 2013; Morawska et al., 2008), though lower levels were reported by de Jesus et al. (roughly 15-20 x 10^3 particles/cm^3 for roadside sites), using data from cities across the world (de Jesus et al., 2019).

Background on This Study and Prior Work in the Same Locations

According to the U.S. Environmental Protection Agency (EPA), more than 45 million people live within 300 feet of a major roadway while another recent study found that nearly 20 percent of the U.S. population lives near high-volume roadways and that increased traffic density is associated with

larger disparities in race and income (U.S. Environmental Protection Agency, 2018; Rowangould, 2013). The objective of this study was to compare concentrations of traffic-related air pollutants and their variability due to distance from the roadway, traffic conditions (e.g., traffic speed, volume, and vehicle types), and meteorological conditions, between two areas east of Detroit: Eliza Howell, an EJ area in the Brightmoor community, and Livonia, a non-EJ community. Census blocks in the Brightmoor community rank in the highest 80th percentile EJ index for cancer risk and $PM_{2.5}$, according to the EPA EJ Screen (U.S. Environmental Protection Agency, 2017). This report focuses on measurements of CO, nitrogen oxide (NO), NO_2, oxides of nitrogen (NO_x), BC, particulate matter less than 2.5 microns ($PM_{2.5}$), UFP, total suspended particulate (TSP) metals, and carbonyls at the Eliza Howell and Livonia monitoring sites; the differences between measurements at the two sites; and comparisons of these concentrations to national ranges.

In order to assess how varying traffic conditions contribute to air toxics in an area where industrial sources are also significant contributors to local air toxics concentrations, the study leverages prior work by state and federal agencies. From 2010 to 2011, the U.S. Federal Highway Administration (FHWA) and the EPA measured mobile source air toxics (MSATs) adjacent to I-96 as part of their joint National Near-Road Study (Kimbrough et al., 2013) and Near-Road Exposures and Effects of Urban Pollutants Study (NEXUS) (Henry et al., 2011; Isakov et al., 2014; Vette et al., 2013; Kimbrough et al., 2013). Our study uses the same monitoring locations at Eliza Howell, five years after the FHWA-EPA effort. In addition, our work builds on past efforts by the EGLE to assess toxics. The Detroit Air Toxics Initiative (DATI-1) measured toxics in 2001 and 2002, and EGLE completed follow-up measurements in 2006 and 2007 (DATI-2) (EGLE, 2010). By leveraging past state and federal work, our findings will document multi-year concentration trends and establish baseline data for future-year comparisons. Such trends analyses will document the impacts of control programs and allow EPA to demonstrate the effectiveness of new vehicle emissions requirements.

Methods

Sampling Site Locations. Air pollution was measured at three monitoring sites in two different communities in Detroit, Michigan, during 2016 and 2017. The measurement campaign included two monitoring sites in Eliza Howell Park (EH), an EJ area of Detroit next to I-96, and a third near-road monitoring site eight miles away in Livonia, a non-EJ area next to I-275 (**Figure 1**). The Eliza Howell near-road site (EH-NR) was located at grade and 8.5 m from I-96, and the Eliza Howell downwind site (EH-DW) was located at grade and 100 m from I-96. Both locations were used during the FHWA-EPA near-road study (Kimbrough et al., 2013) and are in the Brightmoor EJ community of Detroit. According to EPA's EJ Screen (U.S. Environmental Protection Agency, 2017), poverty in census tracks around the Eliza Howell monitoring site can exceed 70%, and excess cancer risk is relatively high (for the United States), at about 80 per million. According to the Michigan Department of Transportation, annual average daily traffic (AADT) on this stretch of I-96 is 138,000 vehicles, 5,400 of which are commercial trucks. Our third site (LIV-NR) is located 9 meters above grade and 49 m from I-275 in Livonia, in one of the highest-traffic-count areas in Detroit. AADT there is nearly 200,000 vehicles,

12,200 of which are commercial trucks. This non-EJ area includes low-density suburban subdivisions, shopping malls, and other land uses.

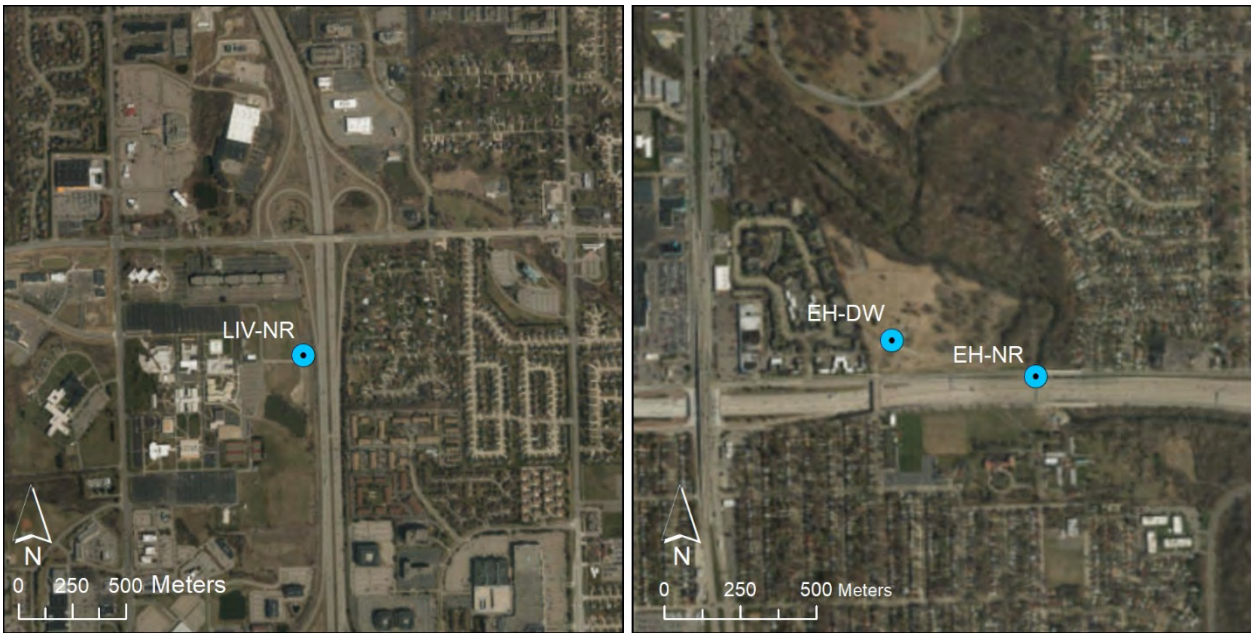

Figure 1. Google Earth views of (right) EH study area, next to I-96 in the Brightmoor community of Detroit; and (left) Livonia study area, next to I-275. The Livonia location represents suburban areas east and west of the freeway.

Continuous Measurements (Air Quality and Meteorology). Long-term routine measurements at the three monitoring sites were augmented for two years with continuous measurements of BC using the Magee Scientific AE33 Aethalometer. The AE33 collects ambient aerosol onto a filter tape, and the absorbance of the material deposited on the tape is measured at seven wavelengths, ranging from 370 to 960 nm; the BC concentration reported is from the 880 nm channel. Raw 1-minute BC measurements were quality-controlled for instrument alarms and averaged to 5-minute and hourly resolution, requiring 75% completeness for each aggregate value. A Teledyne API Ultrafine Particle 651 monitor was used to collect total particle counts at the EH-NR site. Flow rates were verified upon setup and monthly thereafter. Data were collected in the native 1-second time interval of the instrument and averaged up to 1-minute, 5-minute, and hourly values, requiring 75% completeness for each aggregate value. The BC and UFP measurements complemented existing hourly measurements of CO, $NO/NO_2/NO_x$, $PM_{2.5}$, and wind speed and direction at all three sites; these measurements were obtained from EPA's Air Quality System (AQS) database.

Integrated measurements of Metals and Carbonyl. During a three-month intensive study in May through July 2017, EGLE collected 24-hr averaged TSP samples on glass fiber filers every other day at all three sites. Filters were analyzed for a standard set of metals by inductively coupled plasma mass spectrometry (ICP-MS) Federal Equivalent Method (FEM) (40 CFR Part 50 Appendix G and Part 58 Appendix A), consistent with other TSP metal measurements in the area. Twenty-four-hour

2,4-dinitrophenylhydrazine (DNPH) cartridge samples were collected and analyzed via EPA Method TO-11a for carbonyls. Site visits occurred at least weekly during the two years and included flow verifications during sample set-up and take-down. During the three-month intensive period, EGLE staff visited the sites multiple times weekly to change filters and cartridges.

National concentrations of TSP metals for the May-July 2017 time period were downloaded from AQS API in July 2019. Sites with at least six measurements over the time period were aggregated to get quarterly average concentrations. Each of these sites was used to create a national distribution of average concentrations for comparison to the near-road Michigan sites.

Traffic. Traffic data were collected by EGLE at the Eliza Howell near-road location from November 1, 2016, through January 17, 2017. Vehicle counts were recorded for all lanes every five minutes for six lengths of vehicles: less than 10 feet, 10-30 feet, 30-40 feet, 40-50 feet, 50-60 feet, and greater than 60 feet. Vehicles greater than 40 feet were binned as "trucks" as in prior work (Cahill et al., 2016). Data were then aggregated to the total count of cars and trucks each hour.

Data Validation and Analysis. All data were statistically and visually screened to identify outliers, instrument drift, flow issues, or other malfunctions. Of the data obtained from AQS, the CO data were limited to January-October 2016 because of baseline shifts that were observed during late 2016 and 2017 at the Livonia site. Results presented here are restricted to the time period in 2016 when all data at all three monitoring locations did not have baseline shifts. BTEX data were limited to hours when the o-xylene/m,p-xylene ratio was 0.3 ±20%. This is the typical ratio seen in the atmosphere and at other sites in Detroit.

The measurements collected by site, as well as hourly summary statistics for the study time period, are provided in **Table 1**. The summary reflects statistics for the entire record of validated available data. Analyses where we compare concentrations among sites were limited to coincident data across all sites and are noted in the figure and table captions.

Table 1. Summary of measurements collected by site as well as hourly summary statistics for the study time period.

Site	Measurement (unit)	Instrument	Start Date (2016)	End Date (2017)	N	Min.	Max.	Mean	Med.	St. Dev.
EH-DW	BC ($\mu g/m^3$)	Magee Scientific AE33	3-Feb	25-Aug	12345	-0.8	6.7	0.7	0.5	0.6
EH-NR	BC ($\mu g/m^3$)	Magee Scientific AE33	3-Feb	25-Aug	11320	0.0	15.4	1.2	0.9	1.0
LIV-NR	BC ($\mu g/m^3$)	Magee Scientific AE33	17-Aug	25-Aug	7763	0.0	6.3	0.7	0.5	0.7
EH-DW	CO (ppb)	Thermo 48i	1-Jan	20-Oct*	6705	0.0	2.2	0.4	0.3	0.2
EH-NR	CO (ppb)	Thermo 48i	1-Jan	19-Oct*	6482	0.0	2.2	0.5	0.4	0.2
LIV-NR	CO (ppb)	Thermo 48i	1-Jan	20-Oct*	6720	0.0	1.3	0.3	0.3	0.1
EH-DW	NO (ppb)	TECO 42C	1-Jan	31-Dec	16346	-0.3	192.0	4.8	1.0	11.6
EH-NR	NO (ppb)	TECO 42C	1-Jan	31-Dec	16026	-0.3	195.0	20.9	12.8	22.8
LIV-NR	NO (ppb)	TECO 42C	1-Jan	31-Dec	16326	-0.3	172.0	4.4	1.0	9.9
EH-DW	NO_2 (ppb)	TECO 42C	1-Jan	31-Dec	16348	-0.6	62.0	11.2	9.2	7.9
EH-NR	NO_2 (ppb)	TECO 42C	1-Jan	31-Dec	16383	-0.6	58.0	16.1	15.0	9.1
LIV-NR	NO_2 (ppb)	TECO 42C	1-Jan	31-Dec	16327	-1.1	51.0	9.4	6.9	8.0
EH-DW	NO_X (ppb)	TECO 42C	1-Jan	31-Dec	16337	0.6	227.0	16.0	11.0	17.1
EH-NR	NO_X (ppb)	TECO 42C	1-Jan	31-Dec	16030	0.9	233.6	37.1	29.0	29.7
LIV-NR	NO_X (ppb)	TECO 42C	1-Jan	31-Dec	16287	-1.3	223.0	13.9	8.0	16.4
EH-NR	$PM_{2.5}$ ($\mu g/m^3$)	Met One BAM 1020	21-May	31-Dec	13404	-2.7	278.6	11.7	10.1	7.6
LIV-NR	$PM_{2.5}$ ($\mu g/m^3$)	Met One BAM 1020	24-Aug	31-Dec	10178	-3.8	183.1	7.5	6.0	6.2
EH-NR	UFP (particles/cm^3)	Teledyne API 651	28-Oct	25-Aug	5080	446	92704	27645	22488	19637

* Dates are for 2016 rather than 2017.

Results and Discussion

Near-Road Air Pollution at Eliza Howell and at Livonia

As shown in Table 1, hourly average concentrations of BC, CO, NO/NO_2/NO_x, and $PM_{2.5}$ are greater at the Eliza Howell near-road site than at the Livonia near-road site or the Eliza Howell downwind monitoring site (Table 1 and **Supplemental Figures**).

Based on all coincident measurements, average hourly BC concentrations at the Eliza Howell near-road site were 1.2 µg/m^3, compared to 0.68 µg/m^3 at Livonia and 0.67 µg/m^3 at the downwind location (**Figure 2**). This is 1.8 times greater at EH-NR than at LIV-NR and EH-DW. Mean BC concentrations were similar at LIV-NR and EH-DW, even though I-275 has a higher AADT and more commercial trucks than I-96, and the LIV-NR site is closer to the roadway than the EH-DW site (50 m compared to 100 m). In addition, the LIV-NR monitoring site is not at grade and may not represent conditions at grade with the road. Based on all coincident measurements, average hourly $PM_{2.5}$ concentrations at the Eliza Howell near-road site were 12.0 µg/m^3 compared to 7.5 µg/m^3 at Livonia (**Figure 3**). This is 1.6 times greater at EH-NR than at LIV-NR. Following what was seen in the BC data, it may be that background $PM_{2.5}$ concentrations impacting EH-NR are higher, and that the influence of roadway emissions is stronger at EH-NR than at LIV-NR.

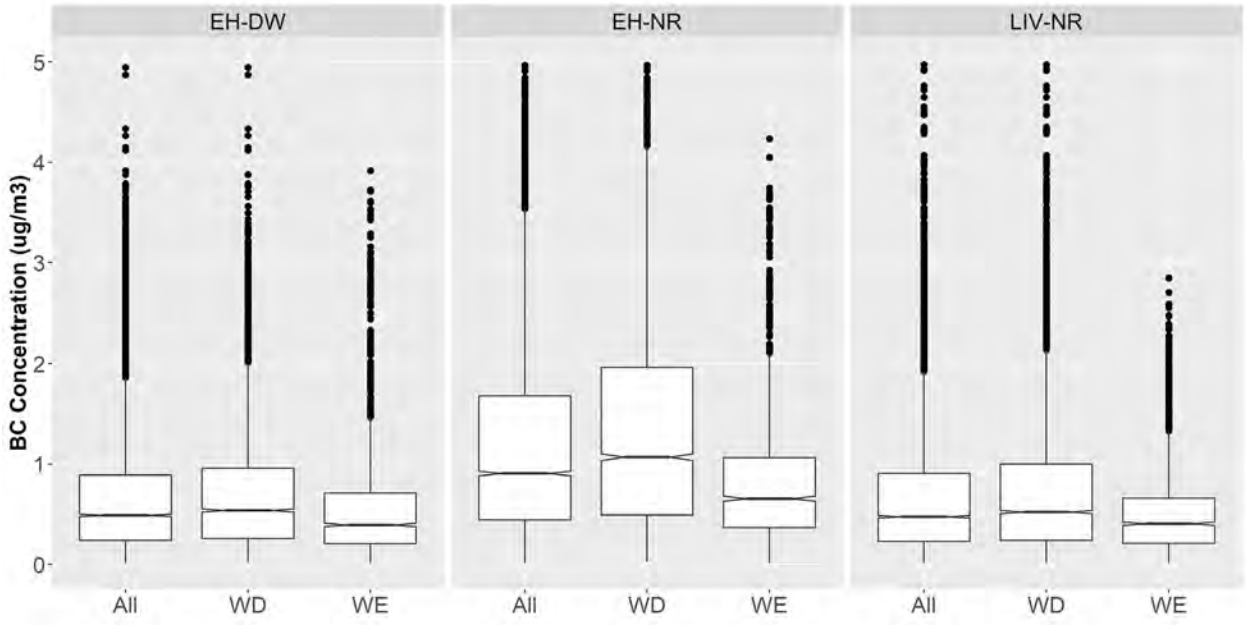

Figure 2. Distribution of hourly black carbon concentrations at three near-road sites in Detroit, Michigan, from August 17, 2016, through August 25, 2017; coincident measurements for the time period are compared to coincident measurements on weekdays (WD) and weekends (WE). In box-whisker plots, the box shows the 25th, 50th (median), and 75th percentiles. The whiskers have a maximum length equal to 1.5 times the length of the box (the interquartile range, IQR). The boxes are notched at the median and return to full width at the 95% lower and upper confidence interval values.

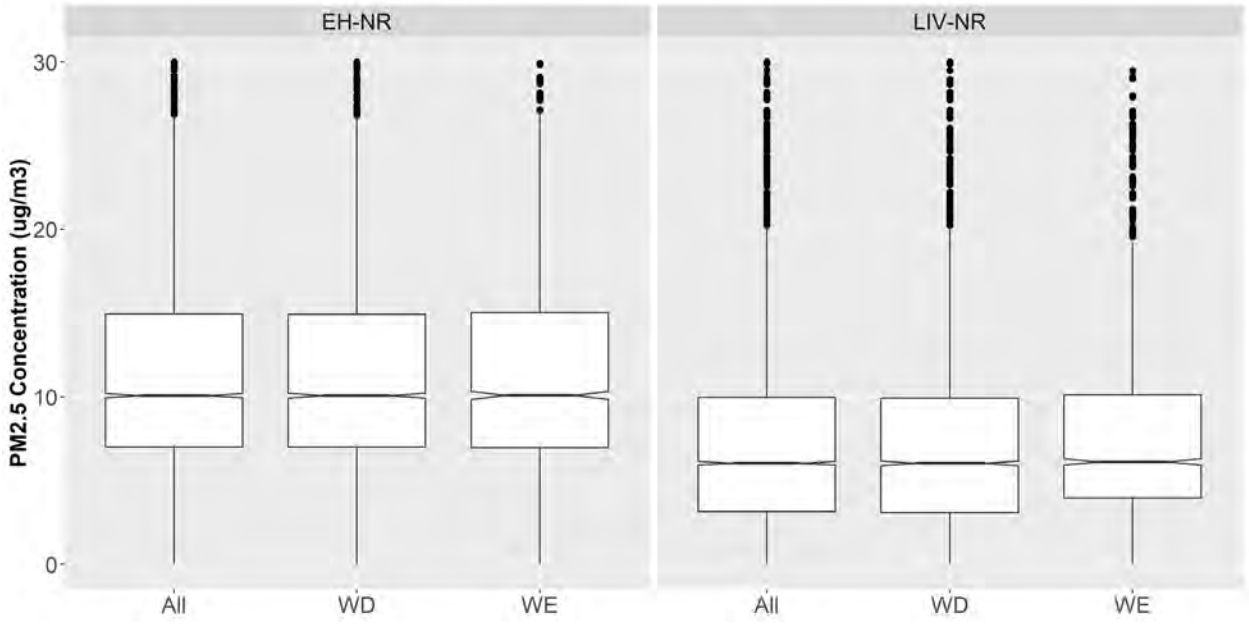

Figure 3. Distribution of hourly $PM_{2.5}$ concentrations at two near-road sites in Detroit, Michigan, from August 24, 2016, through December 31, 2017; coincident measurements for the time period are compared to coincident measurements on weekdays (WD) and weekends (WE).

Analysis of Roadway Influence on Black Carbon

Concentrations of BC were greater on weekdays than on weekends at all three sites (Figure 2). A larger weekday/weekend ratio in concentrations was observed at the EH-NR site (1.7) on weekdays versus weekends compared to the weekday/weekend ratio at LIV-NR (1.4) or EH-DW (1.2). This result is similar for CO and $NO/NO_2/NO_x$ (see **Supplemental Figures**). $PM_{2.5}$ concentrations were not significantly different between weekdays and weekends at either EH-NR or EH-LIV (Figure 3). Typical of urban monitoring locations, the influence of the roadway is relatively small (10% or less) compared to the regional urban background quantity of $PM_{2.5}$ (Karner et al., 2010; Seagram et al., 2019; DeWinter et al., 2018), and a relatively small increase is therefore observed during peak traffic conditions on weekdays.

The influence of the roadway is also observable in BC concentrations when the monitoring sites are downwind of the freeway. Due to the east-west orientation of I-96 relative to the monitoring sites, both EH-NR and EH-DW were classified as "downwind" when winds originated from 90-270 degrees (Figure 1). LIV-NR was classified as "downwind" when winds originated from 0-180 degrees, due to the north-south orientation of I-275 (Figure 1). In **Figure 4**, the pollution rose shows an increase in BC concentrations when winds are from the south at EH-DW and EH-NR, and when winds are from

the east at LIV-NR. Downwind BC concentrations are significantly greater than upwind concentrations at all three sites (Figure 4). Mean downwind BC at EH-NR (1.7 $\mu g/m^3$) was 2.4 times greater than mean upwind BC (0.7 $\mu g/m^3$). A similar ratio (2.2) is observed at LIV-NR, where mean downwind BC was 1.1 $\mu g/m^3$ and mean upwind BC was 0.5 $\mu g/m^3$. The roadway influence was less pronounced at EH-DW, located 100 m from I-96, where mean downwind BC (0.8 $\mu g/m^3$) was 1.5 times greater than mean upwind BC (0.5 $\mu g/m^3$). Downwind $PM_{2.5}$ concentrations are also significantly greater than upwind concentrations at EH-NR and LIV-NR (**Figure 5**). Mean downwind $PM_{2.5}$ at EH-NR (13.6 $\mu g/m^3$) was 1.3 times greater than mean upwind $PM_{2.5}$ (10.1 $\mu g/m^3$). A similar ratio (1.2) is observed at LIV-NR, where mean downwind $PM_{2.5}$ was 8.6 $\mu g/m^3$ and mean upwind $PM_{2.5}$ was 6.9 $\mu g/m^3$.

A moderate relationship (R^2 = 0.56) is observed in BC between EH-NR and EH-DW during downwind conditions (**Figure 6**). As shown in Figure 4, meteorology was generally consistent across all three sites. However, the proximity of EH-NR to nearby trees to the northeast and southeast likely contributes to a greater frequency of winds from the west and southwest ("channeling effect"), which is not observed at the EH-DW site. It is possible this effect contributes to the differences in BC concentrations between EH-NR and EH-DW.

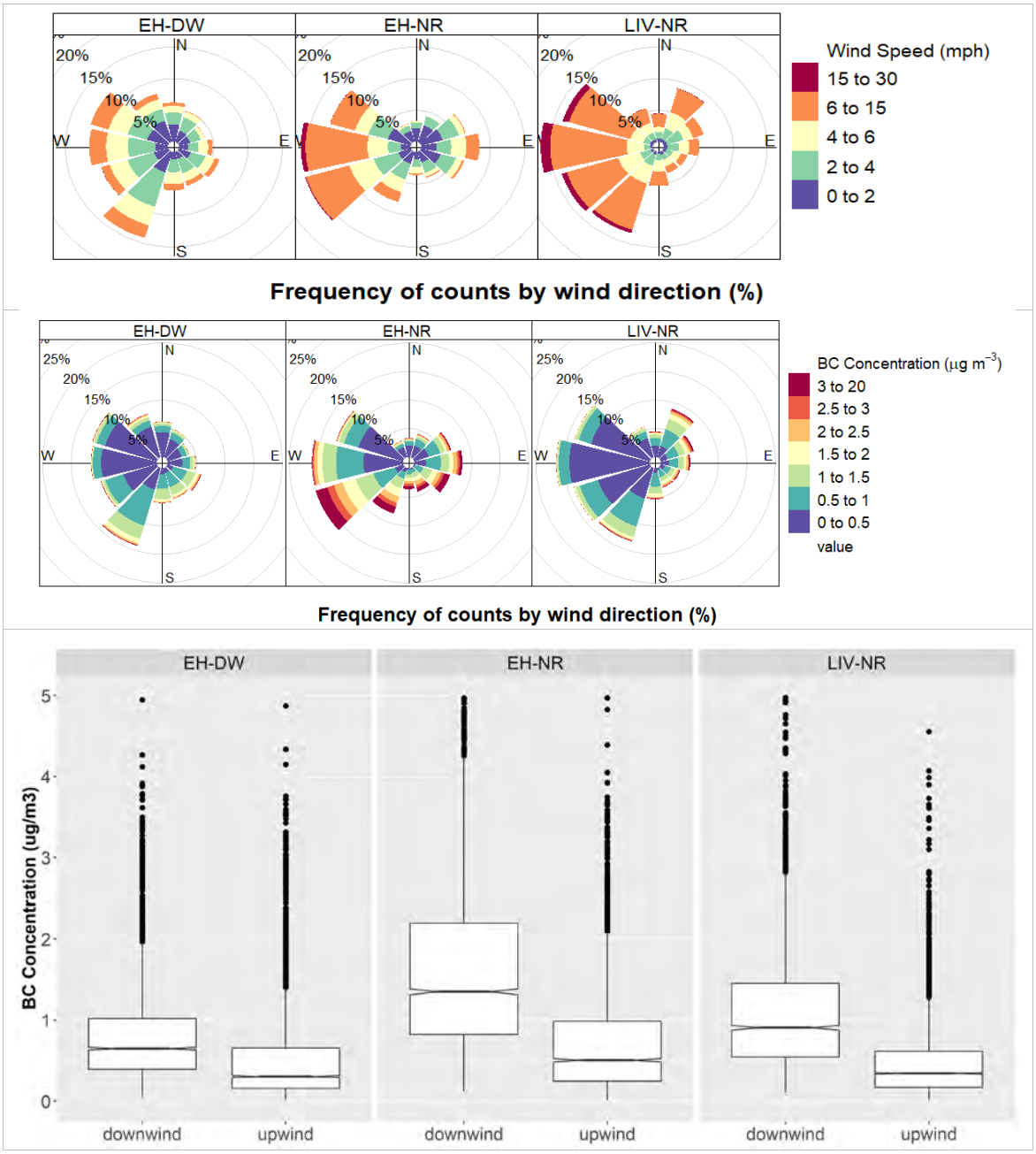

Figure 4. Distribution of black carbon concentrations at three near-road sites in Detroit, Michigan, from August 17, 2016, through August 25, 2017; coincident measurements for the time period are compared for all wind directions (middle) and during downwind versus upwind conditions (bottom). The top row displays the wind pattern during the same time period. The wind roses give a depiction of how wind speed and direction are typically distributed at a particular location. Presented in a circular format, the length of each "spoke" is related to the frequency of time that wind blows from that direction. The color of each spoke indicates differences in wind speed.

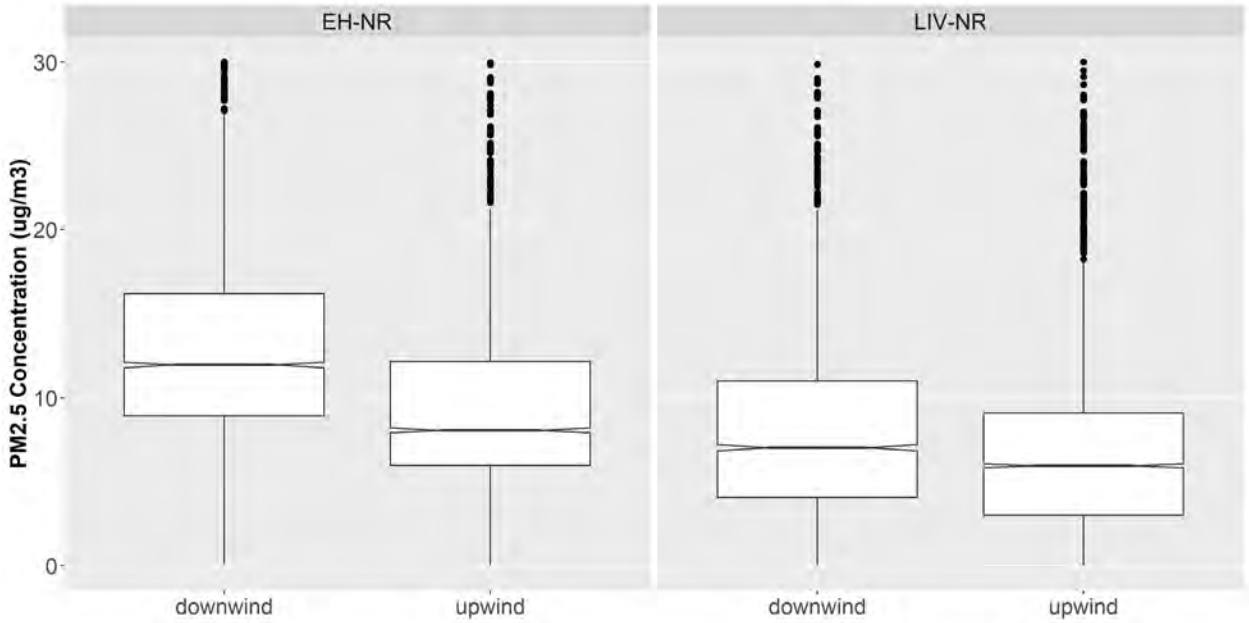

Figure 5. Hourly $PM_{2.5}$ concentrations at Eliza Howell (EH-NR) and Livonia (LIV-NR) during downwind and upwind conditions.

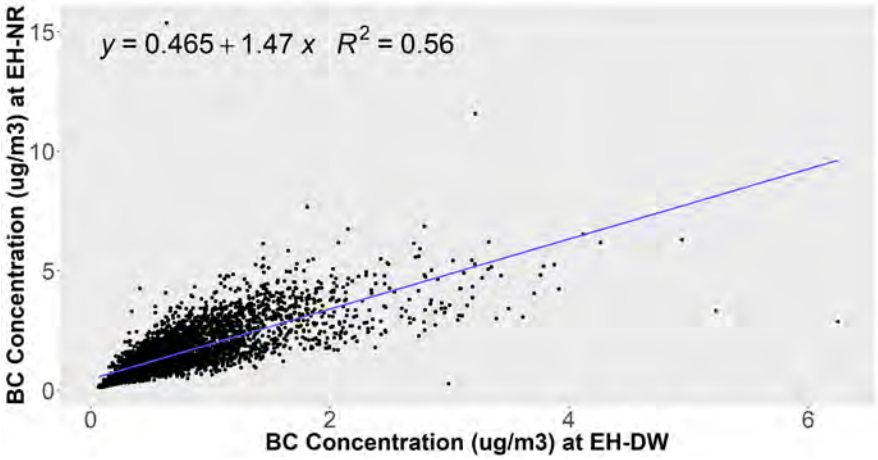

Figure 6. Comparison of hourly black carbon concentrations during downwind conditions at EH-NR and EH-DW.

Analysis of Traffic Patterns on BC Concentrations at Eliza Howell

During November-January, the average daily traffic total was 154,509 vehicles, including 4.4% heavy-duty trucks and 95.6% cars/light-duty trucks. During the week, the average daily traffic total was 170,807 vehicles, including 4.9% heavy-duty trucks and 95.1% cars/light-duty trucks. The average daily total of vehicles decreased over the weekend (113,026), with the typical fraction of heavy-duty

trucks decreasing to 2.0%. The reduction in truck volume on weekends is consistent with prior studies (Cahill et al., 2016). Overall, the average hourly speed was 67.5 mph (46.7-71.9 mph). This pattern was highly consistent throughout the three-month time period with the exception of the Thanksgiving and Christmas holidays, which had lower traffic volumes.

The diurnal pattern of traffic volume for cars/light-duty trucks and heavy-duty trucks is shown in **Figure 7**. Consistent with urban commute patterns, a strong peak in cars/light-duty trucks is observed in the morning and evening on weekdays; this pattern is less evident in heavy-duty trucks, which are consistent throughout the weekday.

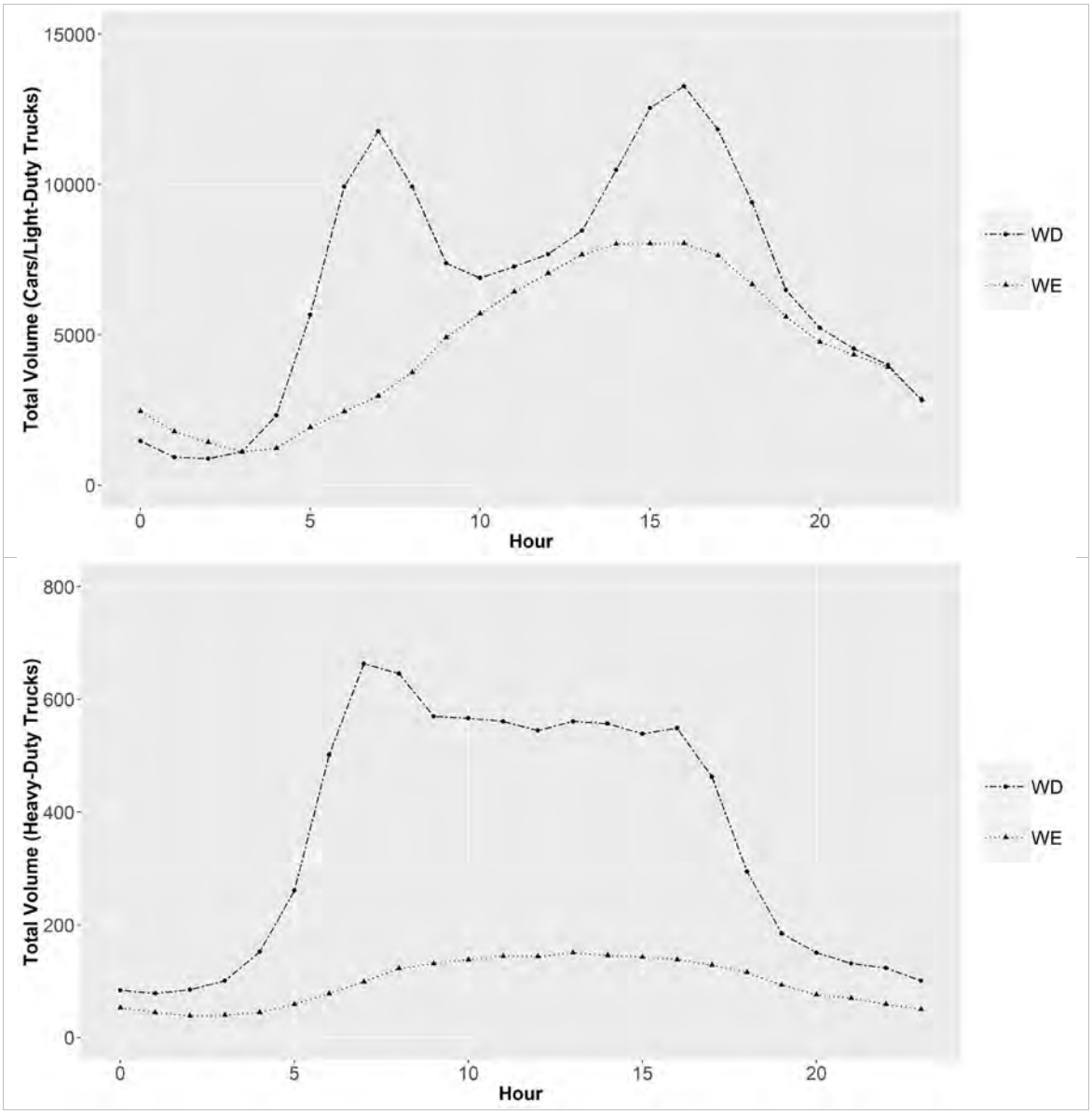

Figure 7. Diurnal traffic patterns for cars and light-duty trucks (top) and heavy-duty trucks (bottom) during weekdays (WD) versus the weekends (WE).

The relationship between BC and the volume of cars/light-duty trucks and heavy-duty trucks is shown in **Figure 8**. A weak relationship (R^2=0.24) is observed between BC and the volume of cars/light-duty trucks during downwind conditions. Consistent with prior findings, a modest relationship is observed between BC and the volume of heavy-duty trucks during downwind conditions (R^2=0.35) compared to upwind conditions (R^2=0.04) and all time periods (R^2=0.2, not shown). After limiting the data to weekdays during downwind conditions, when variations in BC concentrations should best correspond to variations in traffic volumes, we observe a similar relationship (R^2=0.32, not shown) between BC and the volume of heavy-duty trucks. The relationship is consistent controlling for only daytime (5:00 a.m. – 6:00 p.m.) hours (R^2=0.11), as shown in Figure 8, or controlling for variations in wind speeds (**Supplemental Figure S-6**).

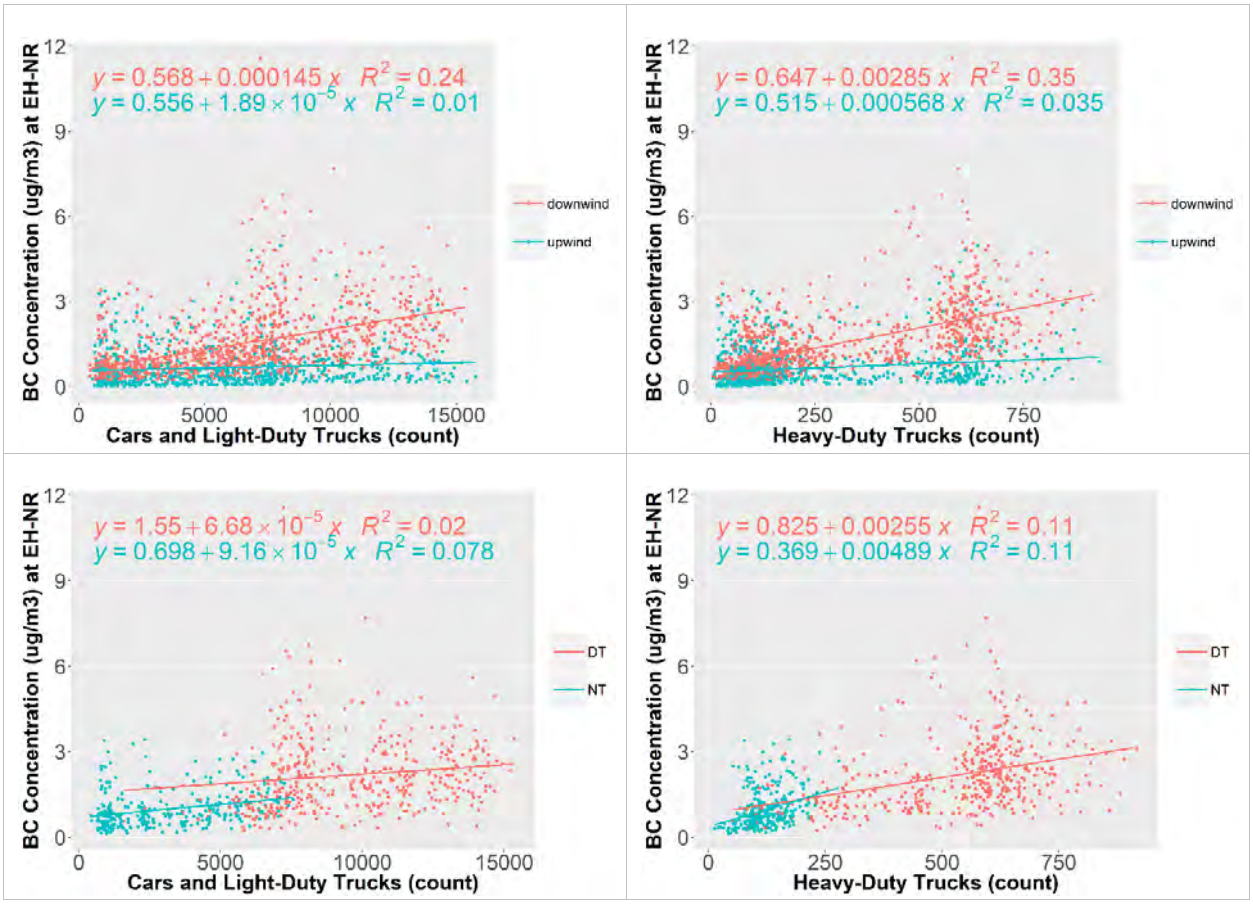

Figure 8. Comparison of hourly BC concentrations with varying volumes of cars and light-duty trucks (upper left) and with volume of heavy-duty trucks (upper right), during all days of the week downwind (red) versus all days of the week upwind (blue). Comparison of BC concentrations with varying volumes of cars and light-duty trucks (lower left), and with heavy-duty trucks (lower right), during downwind weekday daytime (red, 5:00 a.m. – 6:00 p.m.) versus downwind weekday overnight (blue, 7:00 p.m. – 4:00 a.m.).

Carbonyls and TSP Metals

Formaldehyde concentrations were highest at the Eliza Howell near-road site; concentrations at the Livonia near-road and Eliza Howell downwind sites were significantly lower than at EH-NR. **Figure 9** shows the comparison of concentration distributions for formaldehyde and acetaldehyde at the three Detroit sites compared to the average national concentration distribution for May–July 2017. Concentrations of acetaldehyde were significantly lower at most national sites than at Livonia near-road. Concentrations of formaldehyde were significantly lower at most national sites than at Eliza Howell near-road. Median concentrations at other sites were not statistically distinguishable from national median concentrations. It is unclear why formaldehyde concentrations are higher at EH-NR and acetaldehyde concentrations are higher at LIV-NR. Typically, these two pollutants come from similar photochemical and combustion processes and would be expected to have relatively consistent relationships across an urban area unless there are specific emissions sources of the individual pollutants.

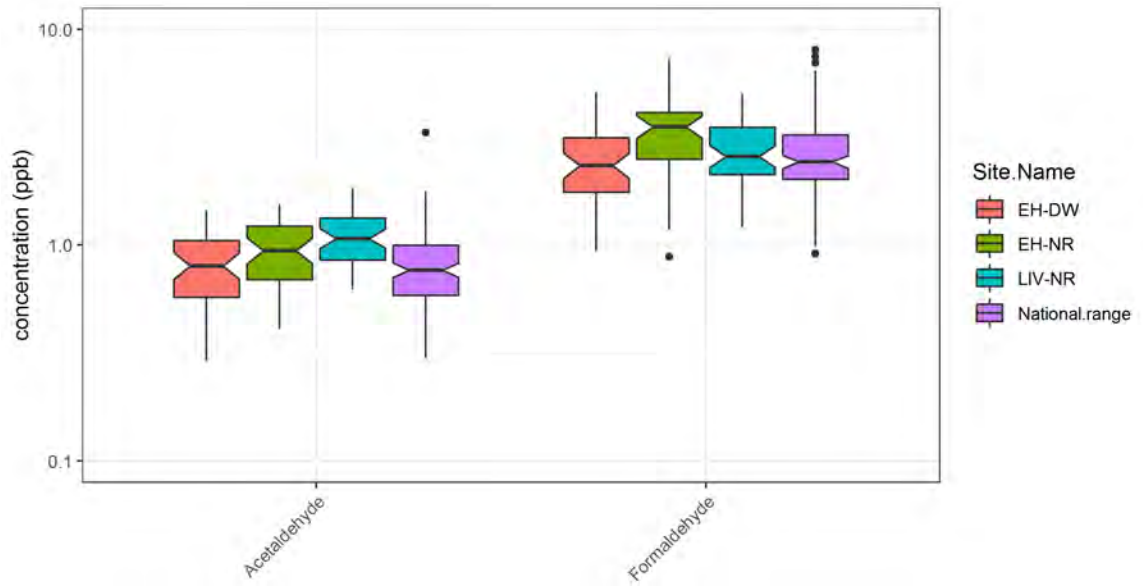

Figure 9. Distribution of 24-hour concentrations of acetaldehyde and formaldehyde (ppb) in Detroit, Michigan, sites and at all national sites from May through July 2017.

The relationship between formaldehyde and acetaldehyde across the three Detroit sites suggests a relatively straightforward correlation, although some striation in the sites may suggest different source influences (**Figure 10**). The uncertainty in the median concentrations (shown as the notches in the waist of the box-notch whisker plots in Figure 9) of formaldehyde and acetaldehyde at the two Eliza Howell sites overlap, but the Livonia site appears to have a higher acetaldehyde concentration for any given formaldehyde concentration, with a constant offset of approximately 0.3 ppb relative to the Eliza Howell sites. This offset is large enough that the difference of the fit over the observed

range is statistically significant. It is unclear what local source of acetaldehyde may be influencing the Livonia site.

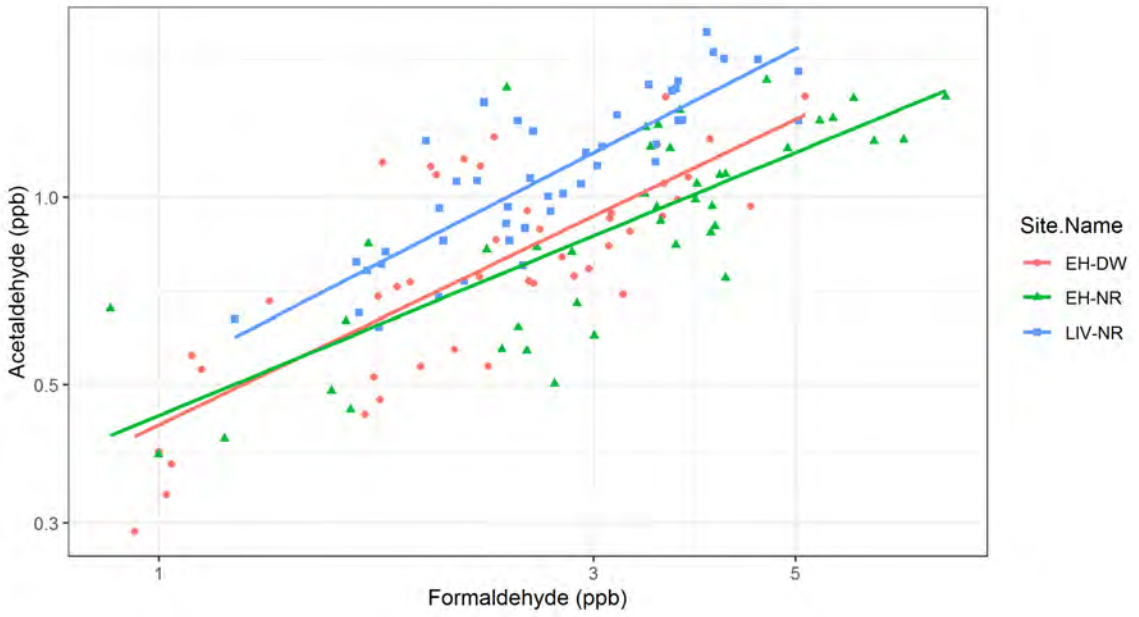

Figure 10. Scatter plot of daily average concentrations of acetaldehyde and formaldehyde (ppb) at the Detroit, Michigan, sites in summer 2017. Lines indicate the linear least squares fit to the data for each site.

A comparison of the TSP metals with nationally collected metals data over the same time period shows that the metals concentrations for most pollutants were similar to or lower than national average ranges observed at other sites over the same time period (**Figure 11**). Concentrations of lead and manganese are slightly higher than national averages at the Eliza Howell Park near-road site. All other metals observed were at or below national range levels.

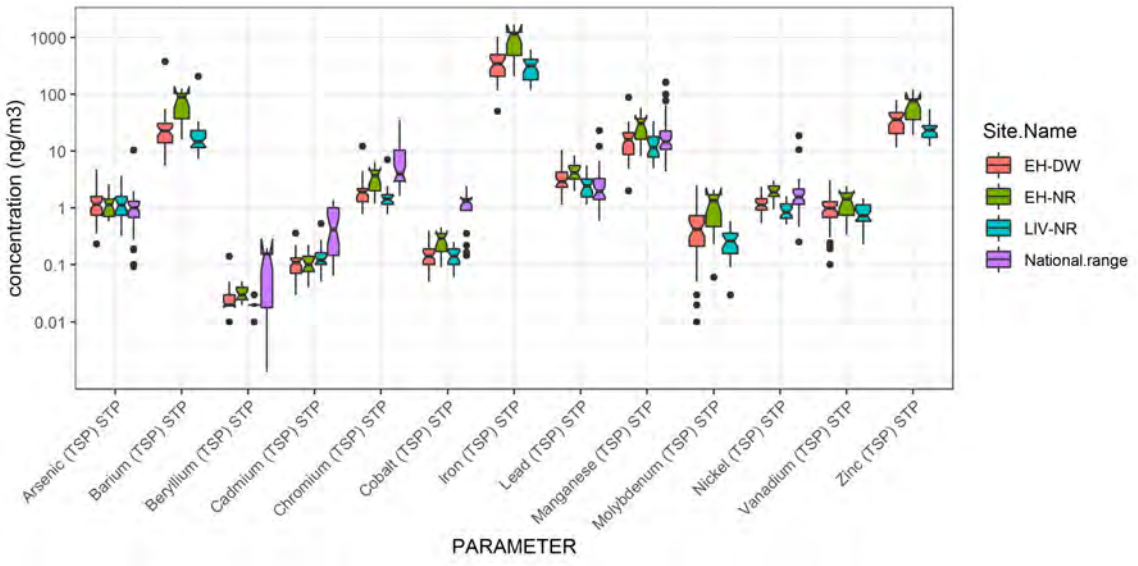

Figure 11. Distribution of concentrations of TSP metals (ng/m^3) in Detroit, Michigan, sites and at all national sites from May through July 2017. Not all metals were available in the AQS dataset as of July 2019—those with no national range shown had no available data.

Table 2 shows the mean concentration (µg/m^3) measured at each site for the May–July 2017 period. Additional chronic exposure threshold benchmarks from EPA's Office of Air Quality Planning and Standards (OAQPS) are provided for context.[3] Note that the three-month averages from this study are not representative of an annual mean concentration, but are likely within a factor of 2-3 of the true annual mean (McCarthy et al., 2007). Pollutants without benchmarks (NA) have no EPA OAQPS dose-response threshold for comparison.

Concentrations of acetaldehyde and formaldehyde are comparable to those seen in other urban areas. None of the other aldehydes is above health benchmark levels. Concentrations of arsenic are above one-in-a-million cancer risk levels, which is typical for U.S. cities east of the Mississippi River (McCarthy et al., 2009); other metals are all below cancer and noncancer risk thresholds. However, this comparison should be considered only as a screening level analysis when comparing to an annual exposure level, given the small sample size and limited seasonality.

[3] www.epa.gov/sites/production/files/2018-06/chronicfinaloutput_6_18_2018_10-09-26_am_1.xlsx, downloaded August 25, 2019.

Table 2. Mean concentrations (µg/m^3) measured at each of the three Detroit sites from May-July 2017 compared to EPA OAQPS dose-response chronic thresholds (noncancer reference concentrations (RfC) and 1-in-a-million excess cancer risk).

Pollutant	EH-DW	EH-NR	LIV-NR	RfC Annual (µg/m^3)	1-in-a-million risk (µg/m^3)
Acetaldehyde	0.82	0.95	1.2	9	0.45
Acetone	0.91	0.94	1	NA	NA
Acrolein	0.039	0.055	0.041	0.35	NA
Benzaldehyde	0.046	0.059	0.049	NA	NA
Formaldehyde	2.4	3.4	2.9	9	0.077
Hexaldehyde	0.041	0.042	0.09	NA	NA
Methyl ethyl ketone and butanone	0.15	0.16	0.21	5000	NA
Methacrolein	0.063	0.066	0.068	NA	NA
Propionaldehyde	0.12	0.14	0.16	8	NA
Valeraldehyde	0.041	0.045	0.072	NA	NA
Arsenic (TSP)	0.0015	0.0012	0.0013	0.015	0.00023
Barium (TSP)	0.03	0.076	0.023	NA	NA
Beryllium (TSP)	0.000024	0.000033	0.000019	0.02	0.000417
Cadmium (TSP)	0.00011	0.00011	0.00014	0.01	0.00055
Chromium (TSP)	0.002	0.0035	0.0016	0.1	0.000083
Cobalt (TSP)	0.00015	0.00027	0.00014	0.1	NA
Iron (TSP)	0.38	0.99	0.31	NA	NA
Lead (TSP)	0.0034	0.0046	0.0025	0.15	NA
Manganese (TSP)	0.017	0.03	0.014	0.3	NA
Molybdenum (TSP)	0.00058	0.0011	0.00027	NA	NA
Nickel (TSP)	0.0012	0.0019	0.00091	0.09	0.00208
Vanadium (TSP)	0.001	0.0013	0.00079	NA	NA
Zinc (TSP)	0.035	0.069	0.025	NA	NA

UFP

Average hourly UFP concentrations at EH-NR were 2.76 x 10^4 particles/cm^3. This average is typical of roadside concentrations, which are historically between 3 x 10^4 and 5 x 10^4 particles/cm^3 (Health Effects Institute, 2013; Morawska et al., 2008). These concentrations are higher than average concentrations adjacent to roadways in Chicago (Xiang et al., 2018) and are similar to concentrations seen in near-road studies elsewhere in the world (generally around 2.0 x 10^4 particles/cm^3) (de Jesus et al., 2019). There is a clear diurnal pattern, with higher concentrations in the daytime compared to overnight. Overnight (10:00 p.m. to 5:00 a.m.) concentrations are similar on weekdays and weekends;

in contrast, daytime (roughly 6:00 a.m. to 7:00 p.m.) concentrations are typically twice as high on weekdays as on weekends (**Figure 12**).

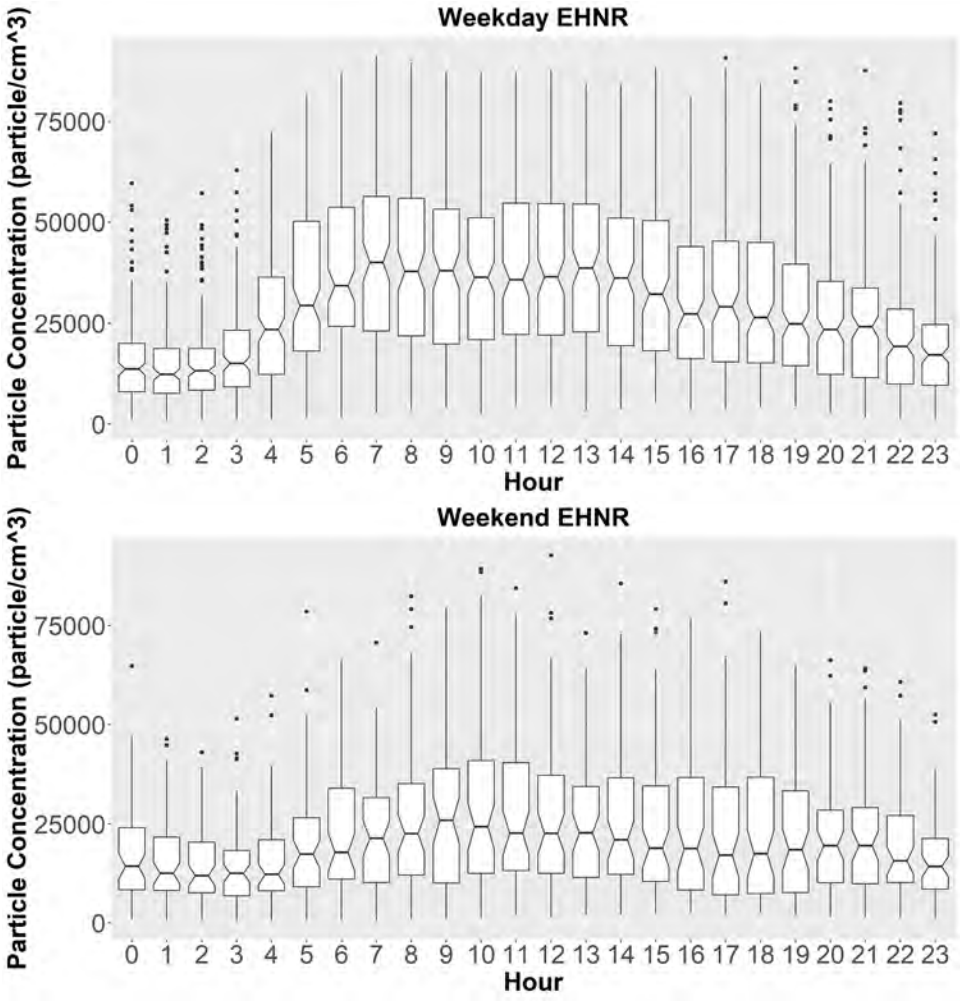

Figure 12. Diurnal pattern in UFP concentrations during the week (top) compared to the weekend (bottom).

UFP concentrations had little correlation with other pollutants such as BC (R^2 = 0.32) or with wind speed (R^2 < 0.01) or relative humidity (R^2 < 0.01). However, there is a clear and significant difference in concentrations when the monitoring site is upwind or downwind of the freeway that is evident regardless of wind speed, relative humidity, or other factors. In **Figure 13**, the polar plot shows a clear distinction in concentration levels across all wind speeds between upwind (270-90 degrees) and downwind (90-270 degrees), with higher concentrations when winds are perpendicular or near perpendicular to the freeway (i.e., from the south). High concentrations of UFP are seen across a wide range of RH conditions, when winds are downwind of the freeway, and when winds are generally less than 10 m/s (see Figure 13 bottom).

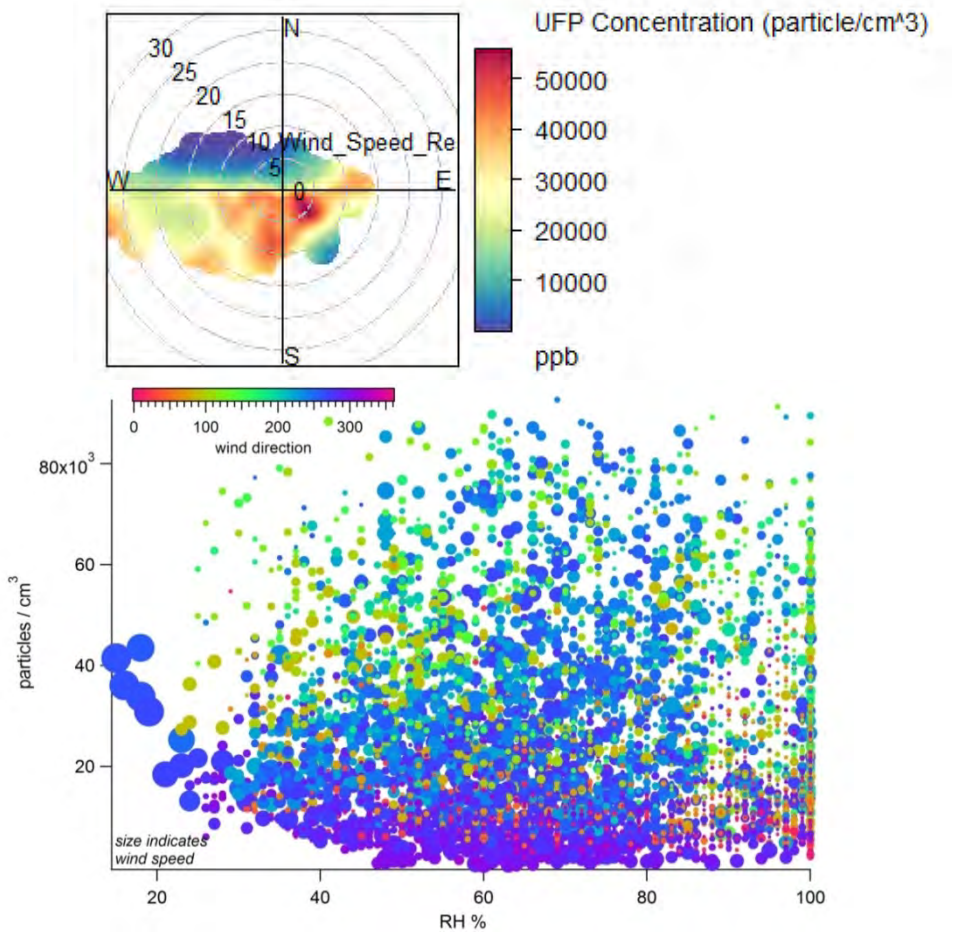

Figure 13. (top) Polar plot for hourly UFP concentrations (particles/cm^3) at EH-NR; and (bottom) UFP concentrations (particles/cm^3) compared with RH, colored by wind direction (downwind is ~90 to 270 degrees) and sized by wind speed (points with lower wind speed have smaller markers). In the polar plot (top), hotter colors (orange, red) indicate higher concentrations; concentric circles indicate the wind speed (m/s); and the location of the color indicates the direction the wind is coming from. In the bottom plot, higher UFP concentrations occur under a range of RH conditions (y axis), but only during downwind conditions (green and blue dots) and typically under lower wind speeds (smaller dots).

Benzene, Toluene, Ethylbenzene, and Xylene (BTEX) Results

Concentrations of benzene, toluene, ethylbenzene, and xylene (BTEX) were all significantly higher at the EH-NR site compared to EH-DW and Livonia; **Figure 14** shows a box plot of hourly average concentrations by site. Benzene was, on average, 0.32 µg/m^3 at EH-NR, equal to 0.1 ppb. This is lower than the 0.46 ppb reported in the EPA/FHWA study, though that study had intermittent collection of 1-hour VOC samples at peak traffic times in the morning and evening, so may not be strictly comparable. Concentrations of benzene and toluene were significantly higher on weekdays

compared to Sundays at EH-NR, but there was no significant difference in weekdays and Sundays for BTEX species at either EH-DW or Livonia.

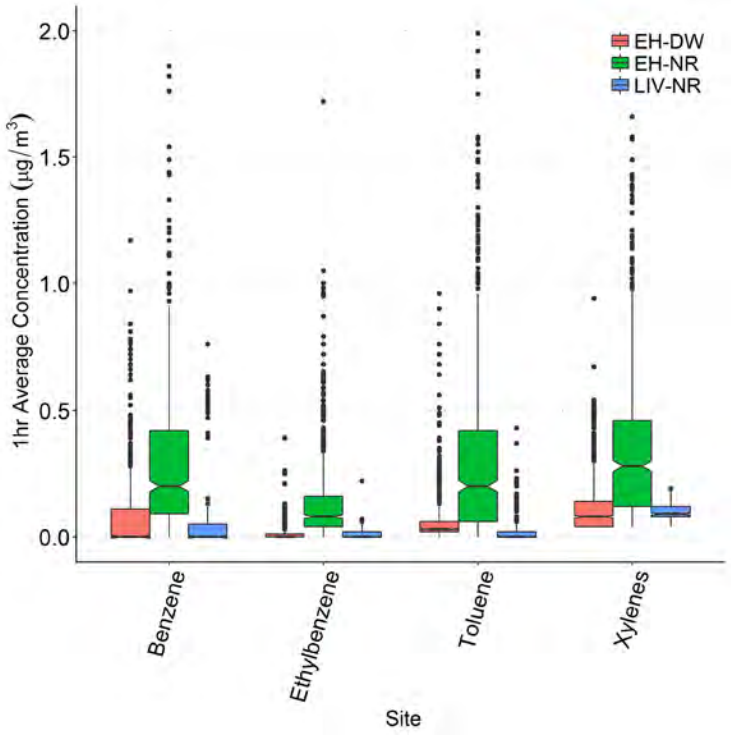

Figure 14. Notched box plot of benzene, ethylbenzene, toluene, and total xylenes concentrations (µg/m^3) by site.

Comparison to Prior Studies

Prior work at the Eliza Howell sites has been conducted by EPA, FHWA, and others as part of the EPA/FHWA Detroit Near-Road Study and the Near-Road Exposures and Effects of Urban Pollutants Study (NEXUS) (Henry et al., 2011; Isakov et al., 2014; Vette et al., 2013; Kimbrough et al., 2013). In the EPA/FHWA study, BC, NO/NO_2/NO_x, $PM_{2.5}$, CO, particles with diameter greater than 20 nm, and carbonyl concentrations were measured from September 29, 2010, to June 15, 2011, at the same near-road and downwind sites as in our study (Kimbrough et al., 2013). Carbonyls were measured for 1 hour, twice a day at peak traffic times, five days every quarter (N=30). In the NEXUS study, BC, $PM_{2.5}$, NO/NO_2/NO_x and other pollutants were measured at the Eliza Howell downwind location during September–October 2010 (Vette et al., 2013). Here, we focus our comparison on the EPA/FHWA study, since it had measurements over the course of many months versus roughly a month of measurements in NEXUS.

Concentrations of NO, NO_2, and NO_x were lower during our study than in the EPA/FHWA study, even though overall traffic volume was similar (AADT of 154,500 in 2016 versus 165,300 in 2010). NO_2 was 8% lower at the near-road site (17.4 ppb to 16.1 ppb) and 26% lower at the downwind site (15.2 ppb to 11.2 ppb), meaning the ratio of concentrations at the downwind to near-road sites went from 0.87 to 0.70. NO_x concentrations were 22% lower at the near-road site (48 ppb to 37 ppb) and 34% lower at the downwind site (24 ppb to 16 ppb), though the ratio of concentrations at the downwind site to near-road site decreased slightly (0.50 to 0.43). Thus, both NO_x and NO_2 have a steeper gradient between the downwind and near-road site in 2016-2017 than in 2010. BC concentrations at both the near-road and downwind sites increased by 35%-40%; at near-road sites, BC increased from 0.86 $\mu g/m^3$ to 1.2 $\mu g/m^3$ and at downwind sites, BC increased from 0.52 $\mu g/m^3$ to 0.7 $\mu g/m^3$. With a consistent increase at both sites, the ratio between sites was similar in both studies (0.58 and 0.60), meaning that while NO_2 and NO_x have a steeper near-road gradient and lower concentrations, BC had a similar gradient between the studies with higher concentrations. The source of this increase in concentrations is unclear, since traffic and trucks were similar in both studies. The average concentration when the near-road and downwind sites are downwind of the freeway (i.e., winds from the south) also increased, from 1.28 $\mu g/m^3$ to 1.86 $\mu g/m^3$ at the near-road site, and from 0.70 $\mu g/m^3$ to 0.90 $\mu g/m^3$ at the downwind site, further suggesting that increases in traffic or changes in traffic mix led to the increased BC concentrations. Alternatively, it may be that differences in instrumentation could be the cause; the EPA/FHWA study used an AE22 model Aethalometer, while here we used the newer AE33 model Aethalometer.

Carbonyl concentrations were lower than in the EPA/FHWA study at both the near-road and downwind locations, though sampling methods were quite different (1-hr rush-hour samples in EPA/FHWA versus 24-hour samples here). Acetaldehyde in particular was 77%-80% lower (2.05-2.67 ppb average in the EPA/FHWA study versus 0.46-0.53 ppb here), with a smaller difference in the ratio between the downwind and near-road site (0.77 ppb in the EPA/FHWA study versus 0.86 ppb here), suggesting that roadway emissions of acetaldehyde have significantly decreased over the past six years. Formaldehyde concentrations were 29%-30% lower (2.6-3.7 ppb for the EPA/FHWA study versus 1.8-2.6 ppb average here, with the same ratio in downwind to near-road concentrations as before (~0.70).

Conclusions

Our study offers an updated reporting of MSAT concentrations in the EJ area of Eliza Howell between 2010-2011 and 2016-2017. In comparing the EH-NR and LIV-NR sites, our study found significant differences in concentrations of BC, UFP, $PM_{2.5}$, and CO/NO/NO_2/NO_x as a result of vehicle emissions and proximity to freeways. The greatest impact from on-road emissions is at the EH-NR site, which is closer to the roadway than either the EH-DW or LIV-NR site. The differences between the EJ community of Eliza Howell and the non-EJ community of Livonia are unclear, as the concentrations observed at the EH-DW and LIV-NR locations were similar. Similar to prior work, the relationship between BC and vehicle emissions could not be well established using only the metrics of traffic counts for cars or trucks.

At the Eliza Howell Park near-road site, concentrations of acetaldehyde and formaldehyde were comparable to those seen in other urban areas, and none of the other aldehydes were above health benchmark levels. Concentrations of TSP metals were similar to or lower than data collected nationally; concentrations of lead and manganese were only slightly higher than national averages. While concentrations of $NO/NO_2/NO_x$ have decreased between our study and the prior results from EPA/FHWA in 2010-2010, BC concentrations at both the near-road and downwind sites increased by 35%-40%. The reason for the increase is unclear, although differences in instrumentation could be the cause. At both the near-road and downwind locations, carbonyl concentrations have decreased since the EPA/FHWA study. The pollutants reported here are significant for their health impacts, and the multi-year documentation of concentrations can be used by EGLE to evaluate control program impacts and allow EPA to demonstrate the effectiveness of new vehicle emissions requirements.

References

Baldauf R., Thoma E., Hays M., Shores R., Kinsey J.S., Gullet B., Kimbrough S., Isakov V., Long T., Snow R., Khlystov A., Weinstein J., Chen F.-L., Seila R., Olson D., Gilmour I., Cho S.-H., Watkins N., Rowley P., and Bang J. (2008) Traffic and meteorological impacts on near-road air quality: summary of methods and trends from the Raleigh near-road study. *J. Air Waste Manage.*, 58, 865-878, July.

Batterman S. (2013) The near-road ambient monitoring network and exposure estimates for health studies. *Air & Waste Manag. Assoc.'s EM*, 2013(7), 24-30. Available at http://www.ncbi.nlm.nih.gov/pmc/articles/PMC4335686/.

Brandt S., Perez L., Künzli N., Lurmann F., Wilson J., Pastor M., and McConnell R. (2014) Cost of near-roadway and regional air pollution–attributable childhood asthma in Los Angeles County. *Journal of Allergy and Clinical Immunology*, 134(5), 1028-1035, doi: 10.1016/j.jaci.2014.09.029, November. Available at http://www.sciencedirect.com/science/article/pii/S0091674914013645.

Brines M., Dall'Osto M., Beddows D.C.S., Harrison R.M., Gómez-Moreno F., Núñez L., Artíñano B., and others (2015) Traffic and nucleation events as main sources of ultrafine particles in high-insolation developed world cities. *Atmospheric Chemistry & Physics*, 15, 5929-5945, doi: 10.5194/acp-15-5929-2015, May 28. Available at http://www.atmos-chem-phys.net/15/5929/2015/acp-15-5929-2015.pdf.

Brown S.G., McCarthy M.C., DeWinter J.L., Vaughn D.L., and Roberts P.T. (2014) Changes in air quality at near-roadway schools after a major freeway expansion in Las Vegas, Nevada. *J. Air Waste Manage.*, 64(9), 1002-1012, doi: 10.1080/10962247.2014.907217 (STI-3889).

Cahill T.A., Barnes D.E., Lawton J.A., Miller R., Spada N., Willis R.D., and Kimbrough S. (2016) Transition metals in coarse, fine, very fine and ultra-fine particles from an interstate highway transect near Detroit. *Atmos. Environ.*, 145, 158-175, November. Available at http://www.sciencedirect.com/science/article/pii/S1352231016307233.

Canagaratna M.R., Onasch T.B., Wood E.C., Herndon S.C., Jayne J.T., Cross E.S., Miake-Lye R.C., Kolb C.E., and Worsnop D.R. (2010) Evolution of vehicle exhaust particles in the atmosphere. *J. Air Waste Manage.*, 60(10), doi: 10.3155/1047-3289.60.10.1192.

de Jesus A.L., Rahman M.M., Mazaheri M., Thompson H., Knibbs L.D., Jeong C., Evans G., and others (2019) Ultrafine particles and $PM_{2.5}$ in the air of cities around the world: are they representative of each other? *Environment International*, 129, 118-135, doi: 10.1016/j.envint.2019.05.021. Available at http://www.sciencedirect.com/science/article/pii/S0160412019311110.

DeWinter J.L., Brown S.G., Seagram A.F., Landsberg K., and Eisinger D.S. (2018) A national-scale review of air pollutant concentrations measured in the U.S. near-road monitoring network during 2014 and 2015. *Atmos. Environ.*, 183, 94-105, doi: 10.1016/j.atmosenv.2018.04.003 (STI-6777), June.

EGLE (2010) Detroit air toxics initiative: risk assessment update. By Michigan Department of Environment, Great Lakes, and Energy, December 22. Available at http://www.michigan.gov/documents/deq/deq-aqd-dati-riskassessmentupdate_340942_7.pdf.

Ghosh R., Lurmann F., Perez L., Penfold B., Brandt S., Wilson J., Milet M., Künzli N., and McConnell R. (2016) Near-roadway air pollution and coronary heart disease: burden of disease and potential impact of greenhouse gas reduction strategy in southern California. *Environ. Health Persp.*, 124(2), 193-200, doi: 10.1289/ehp.1408865, February. Available at http://ehp.niehs.nih.gov/1408865/.

Hagler G.S.W., Thoma E.D., and Baldauf R.W. (2010) High-resolution mobile monitoring of carbon monoxide and ultrafine particle concentrations in a near-road environment. *Journal of Air and Waste Management Association*, 60(3), 328-336, March.

Ham W.A. and Kleeman M.J. (2011) Size-resolved source apportionment of carbonaceous particulate matter in urban and rural sites in central California. *Atmos. Environ.*, 45(24), 3988-3995, doi: 10.1016/j.atmosenv.2011.04.063, August. Available at http://www.sciencedirect.com/science/article/pii/S1352231011004523.

Harrison R.M., Beddows D.C., and DallOsto M. (2011) Source apportionment by PMF of wide range particle size spectra. *Geophysical Research Abstracts*, 13(EGU2011-7031).

Health Effects Institute (2010) Traffic-related air pollution: a critical review of the literature on emissions, exposure, and health effects. Report prepared by the Health Effects Institute, Boston, MA, Special Report 17, January. Available at https://www.healtheffects.org/publication/traffic-related-air-pollution-critical-review-literature-emissions-exposure-and-health.

Health Effects Institute (2013) Understanding the health effects of ambient ultrafine particles. Report prepared by the HEI Review Panel on Ultrafine Particles, Health Effects Institute, Boston, MA, January. Available at http://pubs.healtheffects.org/getfile.php?u=893.

Henry R., Vette A., Norris G., Vedantham R., Kimbrough S., and Shores R. (2011) Separating the air quality impact of a major highway and nearby sources by nonparametric trajectory analysis. *Environ Sci Technol*, 45, 10471-10476, doi: 10.1021/es202070k, November 16. Available at https://www.ncbi.nlm.nih.gov/pubmed/22044064.

Holmes N.S. (2007) A review of particle formation events and growth in the atmosphere in the various environments and discussion of mechanistic implications. *Atmos. Environ.*, 41(10), 2183-2201, doi: 10.1016/j.atmosenv.2006.10.058. Available at http://www.sciencedirect.com/science/article/pii/S1352231006010934.

Isakov V., Arunachalam S., Batterman S., Bereznicki S., Burke J., Dionisio K., Garcia V., Heist D., Perry S., Snyder M., and Vette A. (2014) Air quality modeling in support of the Near-Road Exposures and Effects of Urban Air Pollutants Study (NEXUS). *International Journal of Environmental Research and Public Health*, 11(9), 8777, doi: 10.3390/ijerph110908777. Available at http://www.mdpi.com/1660-4601/11/9/8777.

Jeong C.-H., Wang J.M., Hilker N., Debosz J., Sofowote U., Su Y., Noble M., Healy R.M., Munoz T., Dabek-Zlotorzynska E., Celo V., White L., Audette C., Herod D., and Evans G.J. (2019) Temporal and spatial variability of traffic-related $PM_{2.5}$ sources: comparison of exhaust and non-exhaust emissions. *Atmos. Environ.*, 198, 55-69, doi: 10.1016/j.atmosenv.2018.10.038, 2019/02/01/. Available at http://www.sciencedirect.com/science/article/pii/S1352231018307283.

Kan H., Heiss G., Rose K.M., Whitsel E., Lurmann F., and London S.J. (2007) Traffic exposure and lung function in adults: the Atherosclerosis Risk in Communities study. *Thorax*, 62, 873–879 doi: 2010.1136/thx.2006.073015 (STI-3162), April 18.

Karner A., Eisinger D.S., and Niemeier D. (2010) Near-roadway air quality: synthesizing the findings from real-world data. *Environ. Sci. Technol.*, 44, 5334-5344, doi: 10.1021/es100008x (STI-3923). Available at http://pubs.acs.org/doi/abs/10.1021/es100008x.

Kim J.J., Smorodinsky S., Lipsett M., Singer B.C., Hodgson A.T., and Ostro B. (2004) Traffic-related air pollution near busy roads: the East Bay Childrens Respiratory Health Study. *Am. J. Resp. Crit. Care*, 170(5), 520-526, Sep 1.

Kim J.S., Alderete T.L., Chen Z., Lurmann F., Rappaport E., Habre R., Berhane K., and Gilliland F.D. (2018) Longitudinal associations of in utero and early life near-roadway air pollution with trajectories of childhood body mass index. *Environmental Health*, 17(1), 64, September 14. Available at https://doi.org/10.1186/s12940-018-0409-7.

Kimbrough S., Shores R.C., and Whitaker D.A. (2013) FHWA and EPA National Near-Road Study, Detroit, MI. Near-road pollutant concentration monitoring report for June 1, 2007, to September 30, 2011, prepared for the Office of Natural Environment, Federal Highway Administration, Washington, DC, by the U.S. Environmental Protection Agency, Research Triangle Park, NC, DTFH61-07-X-30015, May. Available at http://www.fhwa.dot.gov/environment/air_quality/air_toxics/research_and_analysis/near_road_study/detroit_michigan/detroit06112013.pdf.

Kleeman M.J., Riddle S.G., Robert M.A., Jakober C.A., Fine P.M., Hays M.D., Schauer J.J., and Hannigan M.P. (2009) Source apportionment of fine ($PM_{1.8}$) and ultrafine ($PM_{0.1}$) airborne particulate matter during a severe winter pollution episode. *Environ. Sci. Technol.*, 43(2), 272-279, doi: 10.1021/es800400m, January. Available at http://dx.doi.org/10.1021/es800400m.

Kulmala M., Vehkamaki H., Petäjä T., Dal Maso M., Lauri A., Kerminen V.M., Birmili W., and McMurry P.H. (2004) Formation and growth rates of ultrafine atmospheric particles: a review of observations. *Journal of Aerosol Science*, 35(2), 143-176, doi: 10.1016/j.jaerosci.2003.10.003, Feb. Available at http://www.sciencedirect.com/science/article/pii/S0021850203004373.

Kumar P., Morawska L., Birmili W., Paasonen P., Hu M., Kulmala M., Harrison R.M., Norford L., and Britter R. (2014) Ultrafine particles in cities. *Environment International*, 66, 1-10, doi:

548 10.1016/j.envint.2014.01.013, May. Available at
549 http://www.sciencedirect.com/science/article/pii/S016041201400018X?np=y.

550 Marmur A., Park S.K., Mulholland J.A., Tolbert P.E., and Russell A.G. (2006) Source apportionment of $PM_{2.5}$
551 in the southeastern United States using receptor and emissions-based models: conceptual
552 differences and implications for time-series health studies. *Atmos. Environ.*, 40(14), 2533-2551,
553 May.

554 McCarthy M.C., Hafner H.R., Chinkin L.R., and Charrier J.G. (2007) Temporal variability of selected air toxics
555 in the United States. *Atmos. Environ.*, 41(34), 7180-7194, (STI-2894). Available at
556 http://dx.doi.org/10.1016/j.atmosenv.2007.05.037.

557 McCarthy M.C., O'Brien T.E., Charrier J.G., and Hafner H.R. (2009) Characterization of the chronic risk and
558 hazard of hazardous air pollutants in the United States using ambient monitoring data. *Environ.*
559 *Health Persp.*, 117(5), 790-796, doi: 10.1289/ehp.11861 (STI-3267), May. Available at
560 http://www.ncbi.nlm.nih.gov/pubmed/19479023.

561 Morawska L., Ristovski Z., Jayaratne E.R., Keogh D.U., and Ling X. (2008) Ambient nano and ultrafine
562 particles from motor vehicle emissions: characteristics, ambient processing and implications on
563 human exposure. *Atmos. Environ.*, 42, 8113-8138, November. Available at
564 http://www.sciencedirect.com/science/article/pii/S1352231008006961.

565 Németh Z., Rosati B., Zíková N., Salma I., Bozó L., Dameto de España C., Schwarz J., Ždímal V., and
566 Wonaschütz A. (2018) Comparison of atmospheric new particle formation events in three Central
567 European cities. *Atmos. Environ.*, 178, 191-197, doi: 10.1016/j.atmosenv.2018.01.035.

568 Riddle S.G., Robert M.A., Jakober C.A., Hannigan M.P., and Kleeman M.J. (2008) Size-resolved source
569 apportionment of airborne particle mass in a roadside environment. *Environ. Sci. Technol.*, 42(17),
570 6580-6586, doi: 10.1021/es702827h.

571 Rioux C.L., Tucker K.L., Mwamburi M., Gute D.M., Cohen S.A., and Brugge D. (2010) Residential traffic
572 exposure, pulse pressure, and C-reactive protein: consistency and contrast among exposure
573 characterization methods. *Environ. Health Persp.*, 118(6), 803-811, doi: 10.1289/ehp.0901182.
574 Available at http://www.ncbi.nlm.nih.gov/pubmed/20123638.

575 Ritz B., Qiu J., Lee P.-C., Lurmann F., Penfold B., Weiss R.E., McConnell R., Arora C., Hobel C., and Wilhelm
576 M. (2014) Prenatal air pollution exposure and ultrasound measures of fetal growth in Los Angeles,
577 California. *Environmental Research*, 130, 7-13, doi: 10.1016/j.envres.2014.01.006, April. Available at
578 http://www.sciencedirect.com/science/article/pii/S0013935114000103.

579 Robinson A.L., Donahue N.M., Shrivastava M.K., Weitkamp E.A., Sage A.M., Grieshop A.P., Lane T.E., Pierce
580 J.R., and Pandis S.N. (2007) Rethinking organic aerosols: semivolatile emissions and photochemical
581 aging. *Science*, 315, 1259.

582 Rowangould G.M. (2013) A census of the US near-roadway population: public health and environmental
583 justice considerations. *Transportation Research Part D: Transport and Environment*, 25, 59-67.

584 Salma I., Borsós T., Németh Z., Weidinger T., Aalto P., and Kulmala M. (2014) Comparative study of
585 ultrafine atmospheric aerosol within a city. *Atmos. Environ.*, 92, 154-161, doi:

586 10.1016/j.atmosenv.2014.04.020, August. Available at
587 http://www.sciencedirect.com/science/article/pii/S1352231014002921.

588 Seagram A.F., Brown S.G., Huang S., Landsberg K., and Eisinger D.S. (2019) National assessment of near-
589 road air quality in 2016: multi-year pollutant trends and estimation of near-road $PM_{2.5}$ increment.
590 *Transportation Research Record*, doi: 10.1177/0361198119825538 (STI-6963), January 30. Available
591 at https://journals.sagepub.com/doi/10.1177/0361198119825538.

592 U.S. Environmental Protection Agency (2017) EJSCREEN environmental justice mapping and screening
593 tool: technical documentation. August.

594 U.S. Environmental Protection Agency (2018) Research on near roadway and other near source air
595 pollution. Available at https://www.epa.gov/air-research/research-near-roadway-and-other-near-
596 source-air-pollution.

597 Venecek M.A., Yu X., and Kleeman M.J. (2019) Predicted ultrafine particulate matter source contribution
598 across the continental United States during summertime air pollution events. *Atmos. Chem. Phys.*,
599 19(14), 9399-9412, doi: 10.5194/acp-19-9399-2019. Available at https://www.atmos-chem-
600 phys.net/19/9399/2019/.

601 Vette A., Burke J., Norris G., Landis M., Batterman S., Breen M., Isakov V., and others (2013) The Near-Road
602 Exposures and Effects of Urban Air Pollutants Study (NEXUS): study design and methods. *Science*
603 *of The Total Environment*, 448, 38-47, doi: 10.1016/j.scitotenv.2012.10.072, March 15. Available at
604 https://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=254202&Lab=NHEERL.

605 Xiang S., hu Z., Zhai W., Wen D., and Noll K. (2018) Concentration of ultrafine particles near roadways in an
606 urban area in Chicago, Illinois. *Aerosol and Air Quality Research*, 18, 895-903, doi:
607 10.4209/aaqr.2017.09.0347.

608 Zhang K.M., Wexler A.S., Zhu Y.F., Hinds W.C., and Sioutas C. (2004) Evolution of particle number
609 distribution near roadways. Part II: The "road-to-ambient" process. *Atmos. Environ.*, 38, 6655-
610 6665.

611 Zhu Y., Hinds W.C., Kim S., Shen S., and Sioutas C. (2002a) Study of ultrafine particles near a major highway
612 with heavy-duty diesel traffic. *Atmos. Environ.*, 36(27), 4323-4335, doi: 10.1016/s1352-
613 2310(02)00354-0, September.

614 Zhu Y., Pudota J., Collins D., Allen D., Clements A., DenBleyker A., Fraser M., Jia Y., McDonald-Buller E., and
615 Michel E. (2009) Air pollutant concentrations near three Texas roadways, part I: ultrafine particles.
616 *Atmos. Environ.*, 43, 4513-4522, April 6.

617 Zhu Y.F., Hinds W.C., Kim S., and Sioutas C. (2002b) Concentration and size distribution of ultrafine
618 particles near a major highway. *J. Air Waste Manage.*, 52(9), 1032-1042, September.

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Supplemental Information

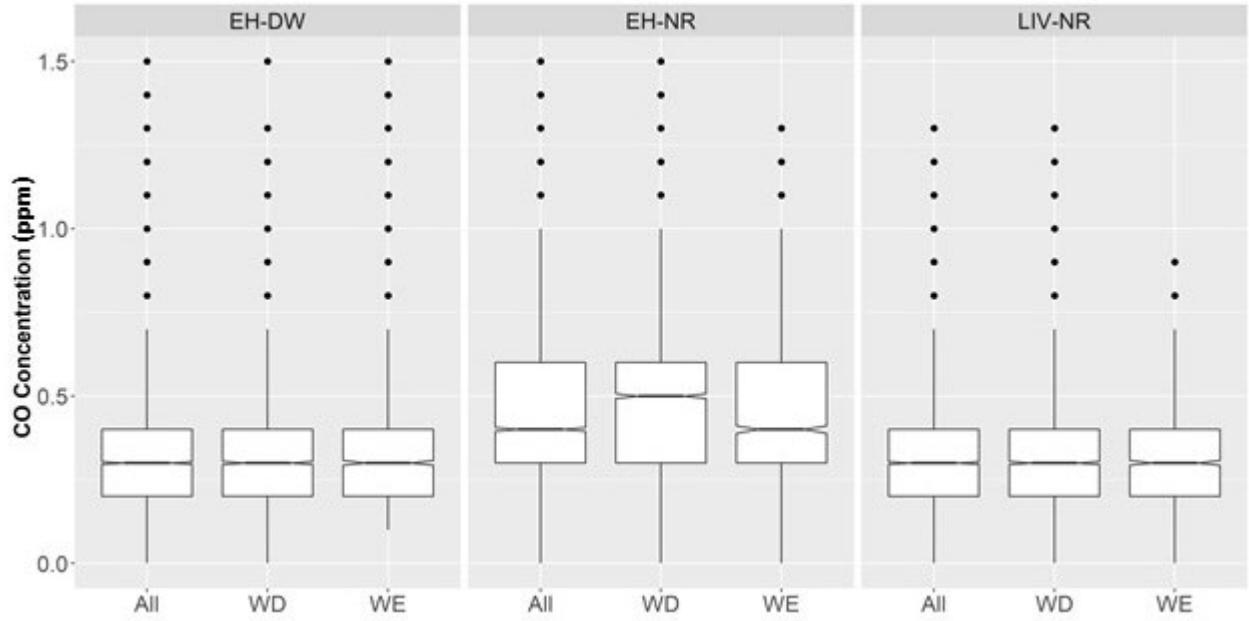


Figure S-1. Distribution of CO concentrations at three near-road sites in Detroit, Michigan, from January 1, 2016, through August 19, 2016; coincident measurements for the time period are compared to coincident measurements on weekdays (WD) and weekends (WE).

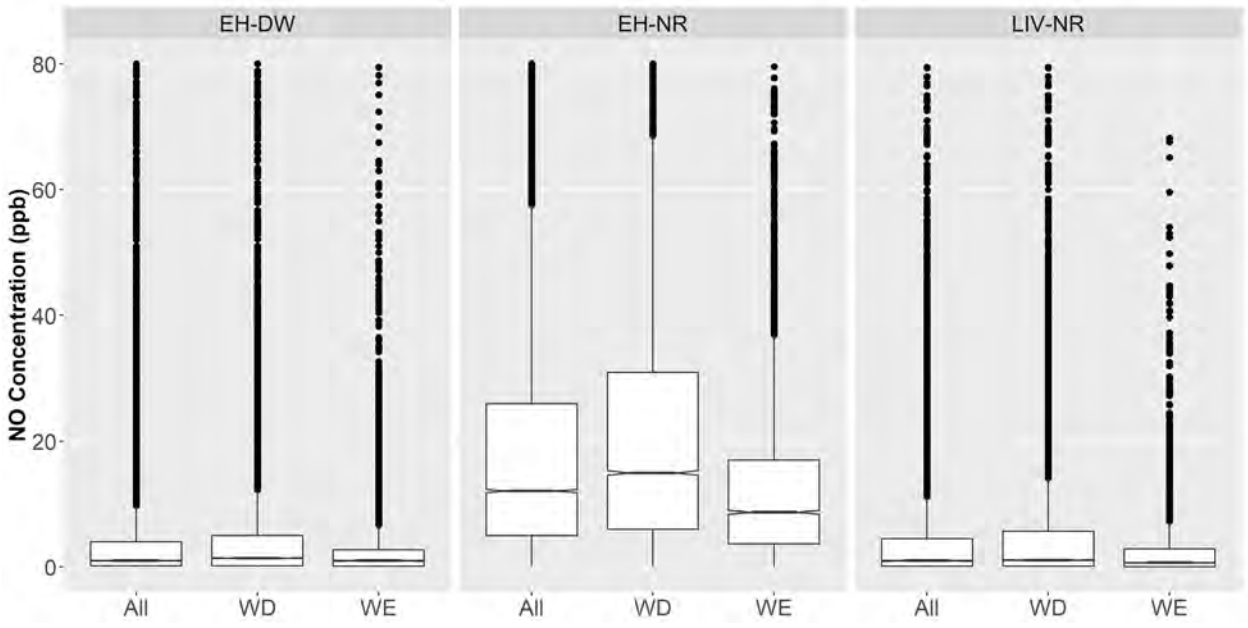
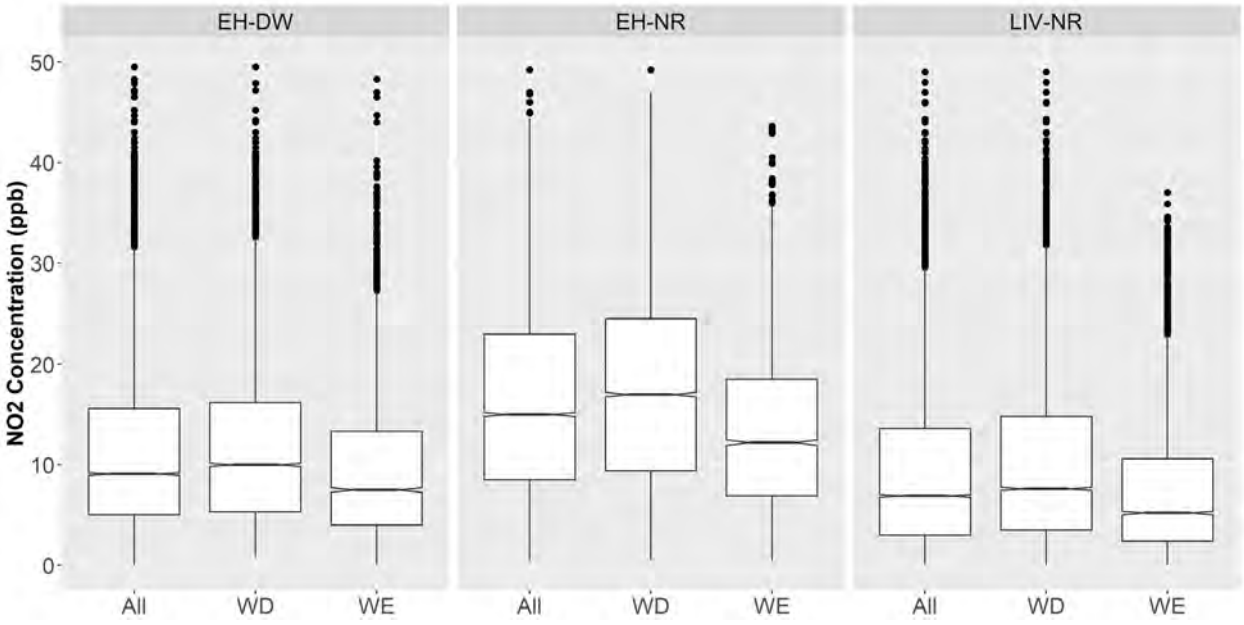


Figure S-2. Distribution of NO concentrations at three near-road sites in Detroit, Michigan, from January 1, 2016, through December 31, 2017; coincident measurements for the time period are compared to coincident measurements on weekdays (WD) and weekends (WE).



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634 **Figure S-3.** Distribution of NO_2 concentrations at three near-road sites in Detroit,
635 Michigan, from January 1, 2016, through December 31, 2017; coincident measurements
636 for the time period are compared to coincident measurements on weekdays (WD) and
637 weekends (WE).

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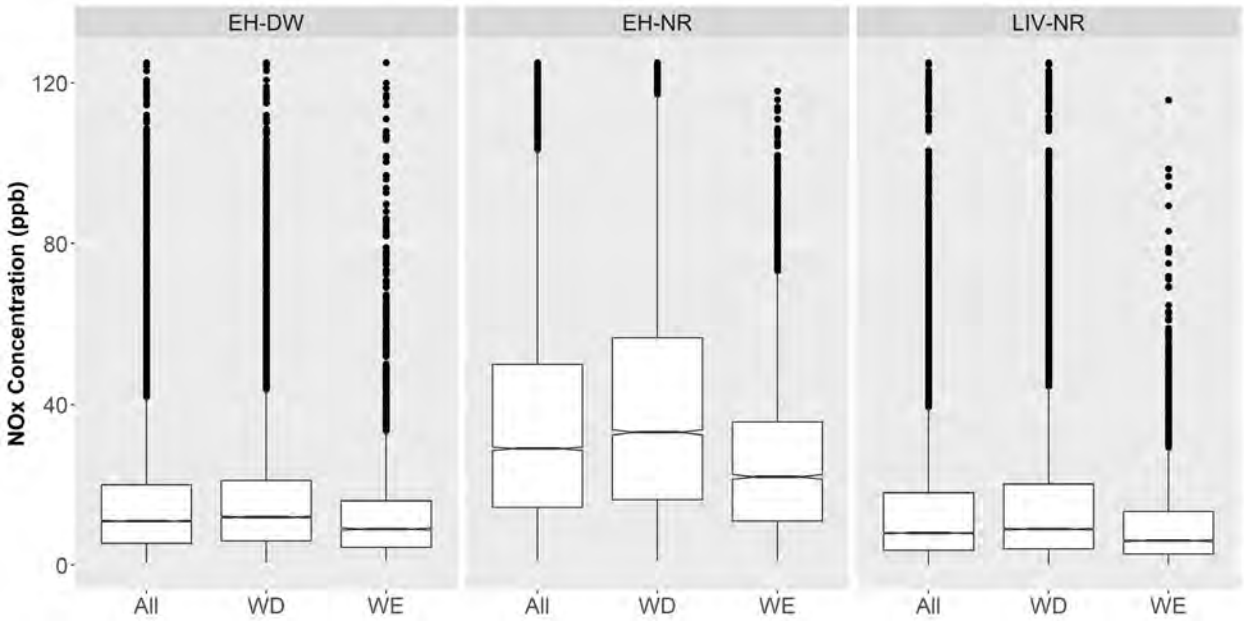

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640 **Figure S-4.** Distribution of NO_x concentrations at three near-road sites in Detroit,
641 Michigan, from January 1, 2016, through December 31, 2017; coincident measurements
642 for the time period are compared to coincident measurements on weekdays (WD) and
643 weekends (WE).

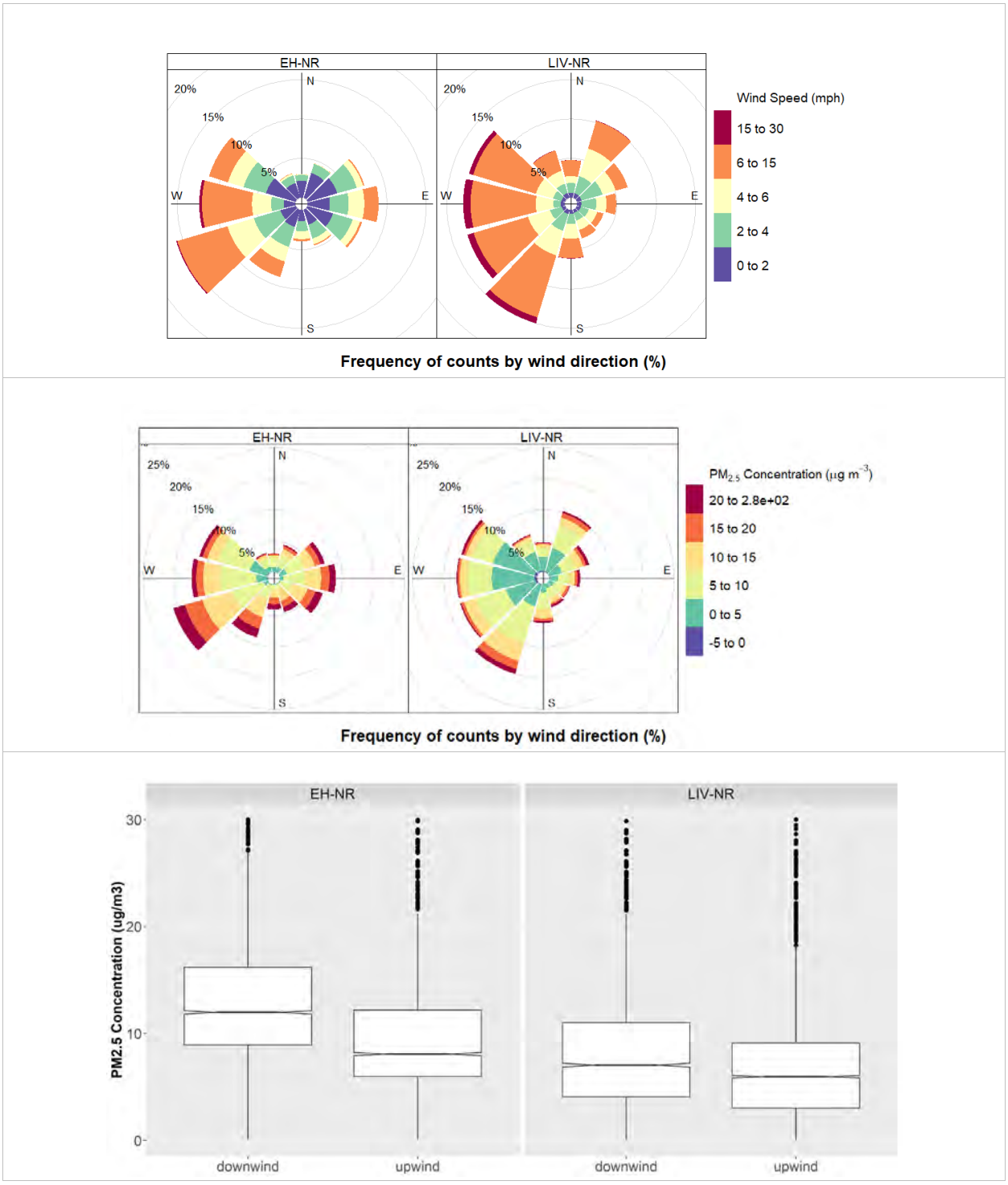

Figure S-5. Distribution of $PM_{2.5}$ concentrations at two near-road sites in Detroit, Michigan, from August 24, 2016, through December 31, 2017; coincident measurements for the time period are compared for all wind directions (middle) and during downwind versus upwind conditions (bottom). The top row displays the wind pattern during the same time period.

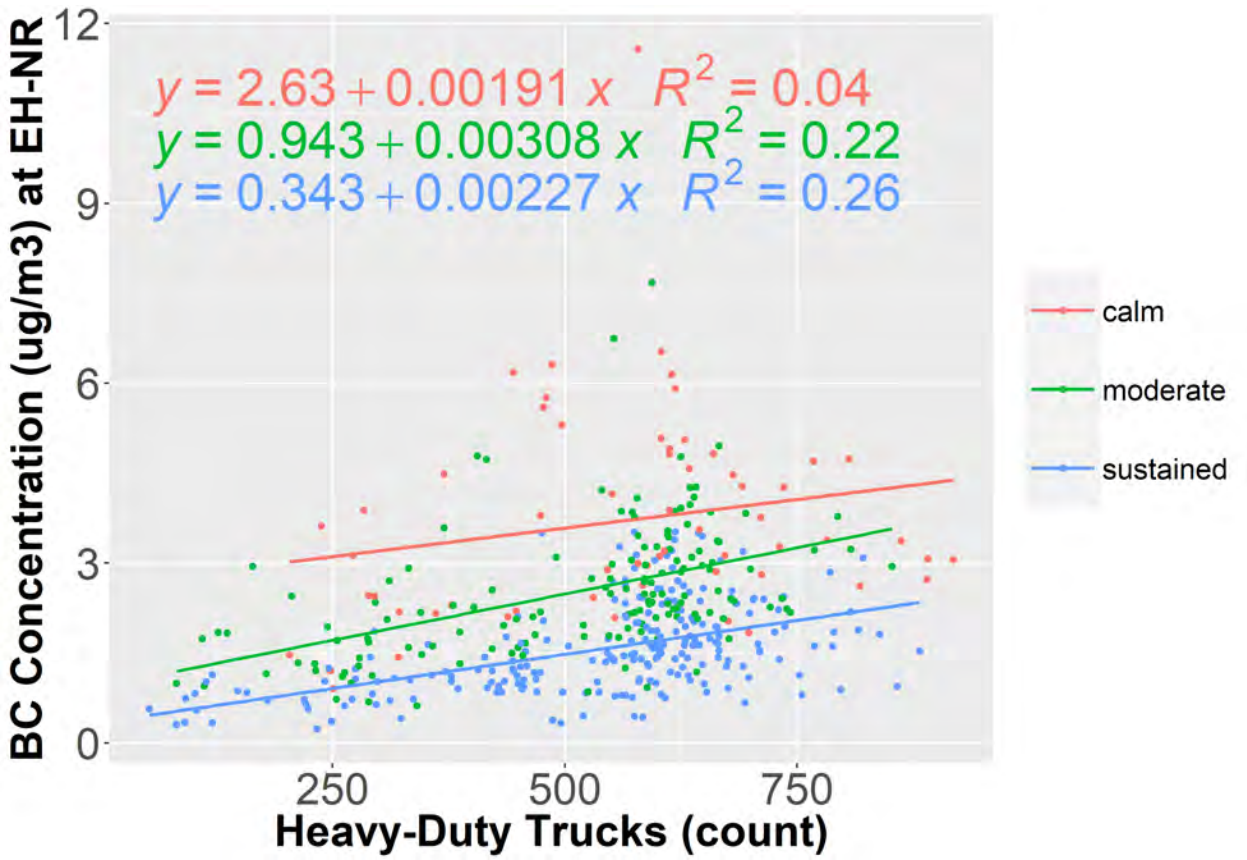

Figure S-6. Comparison of BC concentrations with heavy-duty trucks during downwind weekday daytime (5:00 a.m. – 6:00 p.m.) when winds are calm (less than 3 mph), moderate (3-6 mph), and sustained (greater than 6 mph).

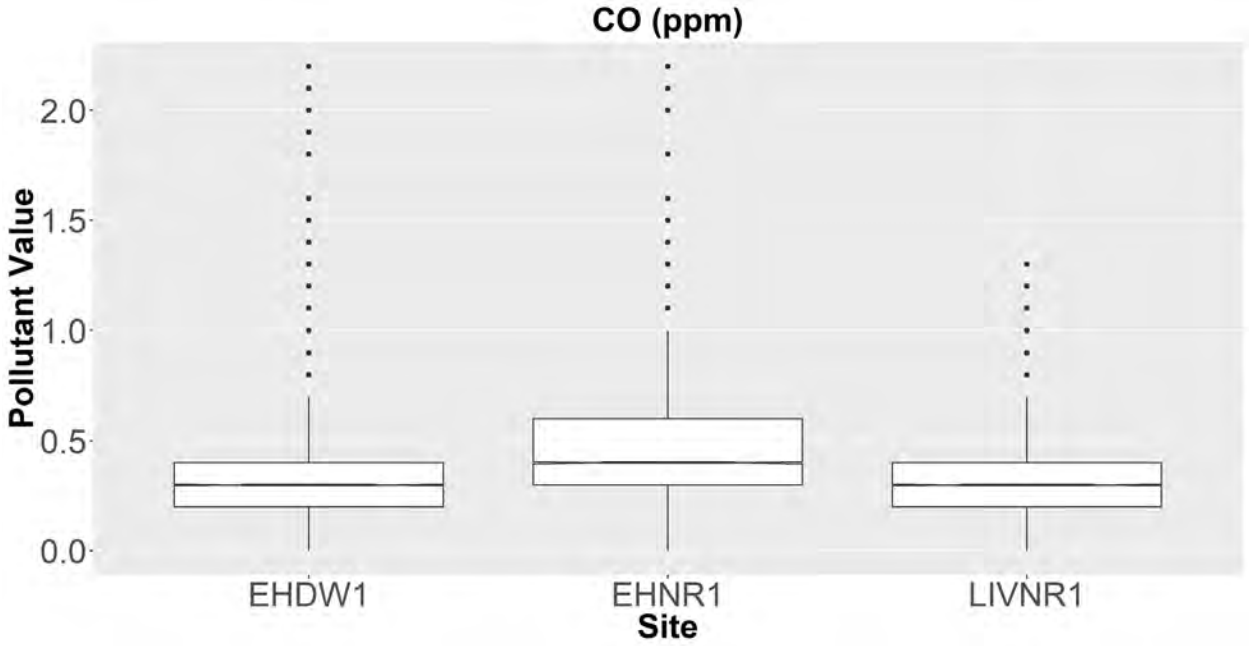

Figure S-7. Distribution of CO concentrations at three near-road sites in Detroit, Michigan, from January 1, 2016, through August 19, 2016.

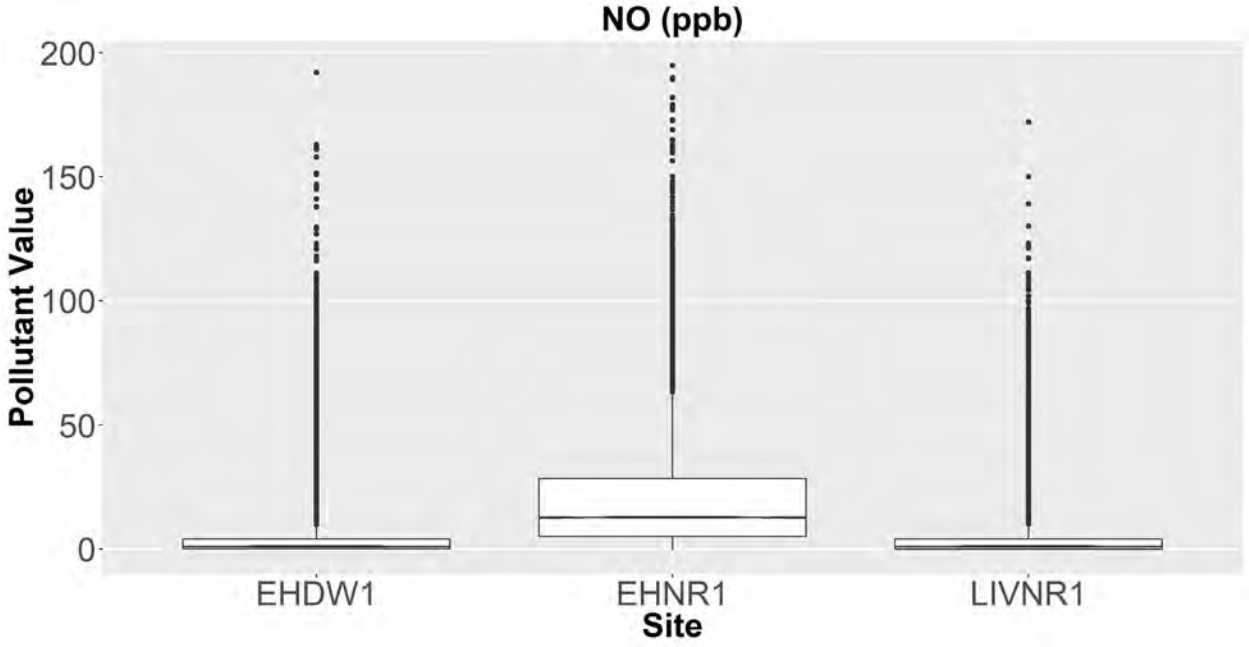

Figure S-8. Distribution of NO concentrations at three near-road sites in Detroit, Michigan, from January 1, 2016, through December 31, 2017.

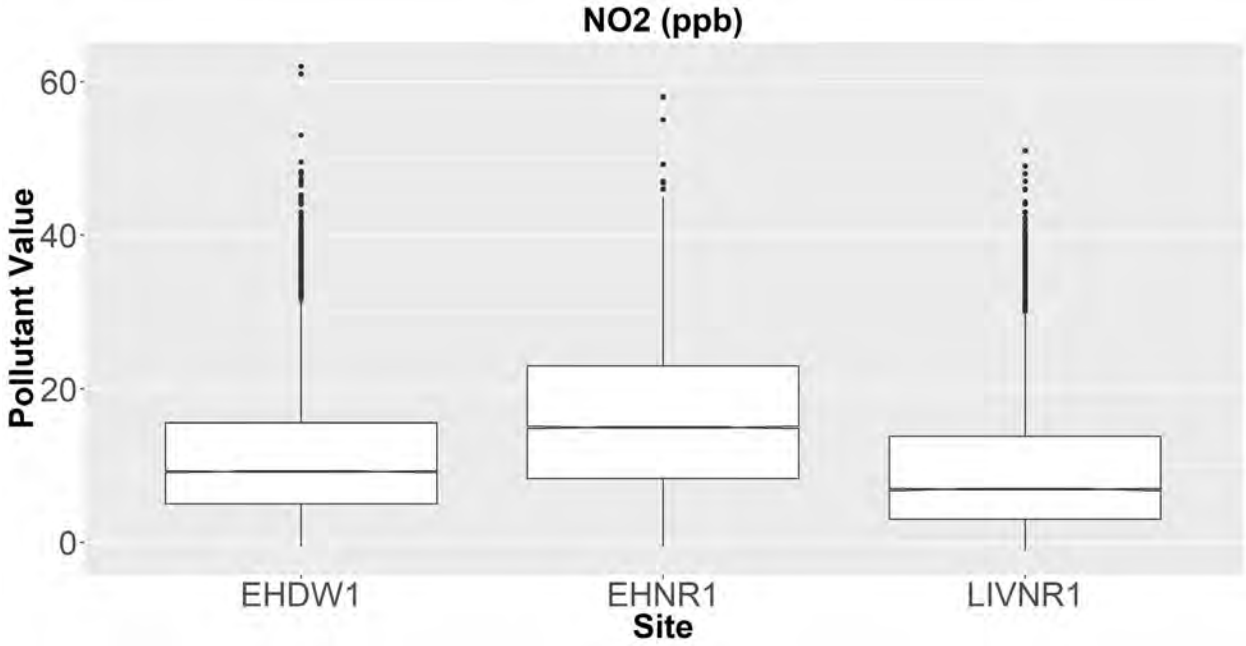

Figure S-9. Distribution of NO_2 concentrations at three near-road sites in Detroit, Michigan, from January 1, 2016, through December 31, 2017.

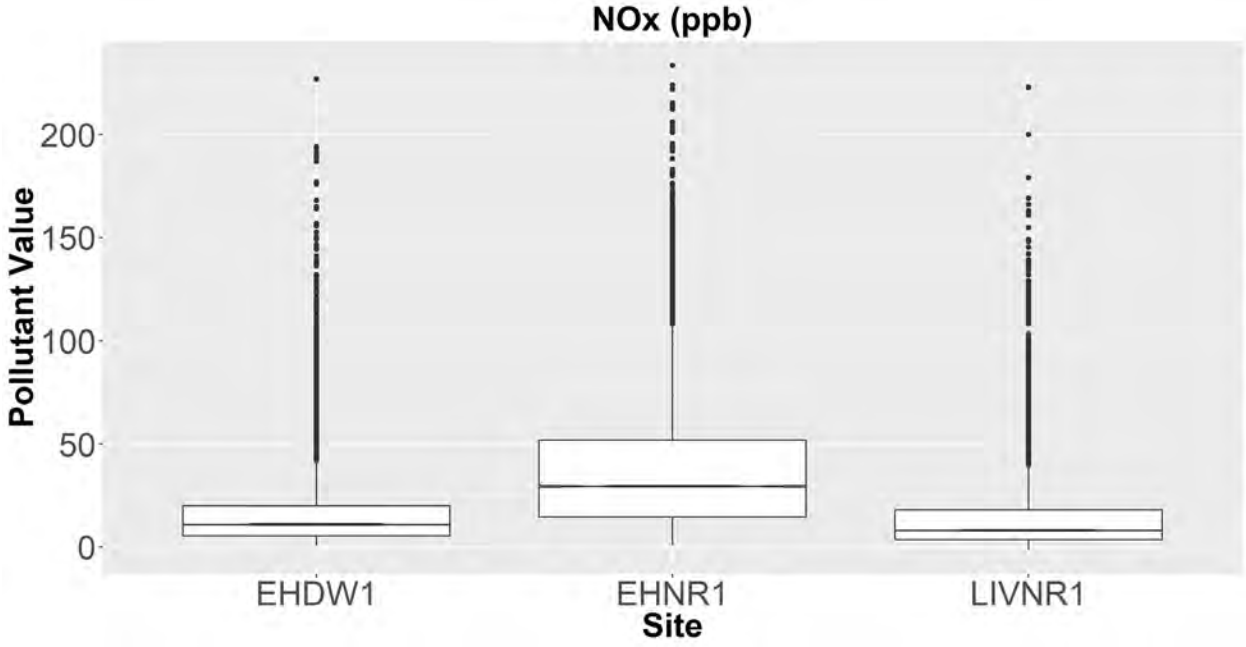

Figure S-10. Distribution of NO_x concentrations at three near-road sites in Detroit, Michigan, from January 1, 2016, through December 31, 2017.