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A Comparative Study Characterizing Traffic Related Air Pollutant Concentrations At Near-Road Communities In El Paso, Texas

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A COMPARATIVE STUDY CHARACTERIZING TRAFFIC RELATED AIR POLLUTANT CONCENTRATIONS AT NEAR-ROAD COMMUNITIES IN EL PASO, TEXAS

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by

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ABSTRACT

Numerous scientific studies have demonstrated the influence of traffic related air pollution at near-road communities and the associated health risks for these populations. This study uses on-site air quality monitors to characterize air pollutants at near-road schools in El Paso, TX to understand children's exposure to traffic-related air pollutants. Ambient air monitoring stations were installed at Coldwell Elementary, Bliss Elementary, and a residential house located in close proximity to major inter-state roadways. In this study, air quality data for $PM_{2.5}$, PM_{10}, NO_2, and O_3 was collected over a period of nine weeks in Fall 2018. The spatial and temporal variability in the pollutant concentrations in this region was assessed by comparing air quality data obtained from the study with central ambient monitoring sites. All on-site monitors recorded similar trends in measured pollutant across all examined sites. Higher concentrations of PM_{10} were recorded at Bliss Elementary. Spearman correlations, coefficient of divergence, and diurnal graphs help to characterize the differences in the pollutant levels between these sites. The results suggest a spatial and temporal variability between the sites examined and available CAMS sites. It is recommended that studies performed in El Paso employ on-site measurements to avoid exposure misclassification and erroneous estimations from using distant CAMS sites.

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CHAPTER 1: INTRODUCTION

1.1. Effects of Air Pollution

Numerous epidemiologic studies have demonstrated the health risk air pollution poses for underserved communities living near highways (Sharma et al., 2009; McConnell et al., 2006; Janssen et al., 2003). Air pollution from near-road traffic poses a significant threat to human health contributing to the disease and asthma burden in children (Perez et al., 2013). The elderly and children with asthma are at higher risk to complications that arise from these chronic traffic exposures. Exposure to particulate matter (PM), anthropogenic (human produced) or geogenic (naturally occurring by Earth processes), has been acknowledged to increase risks for human morbidity and mortality (Du et al., 2016). Within El Paso County in Texas, an estimated 60% of lifetime cancer risk is attributed to on road sources (Collins et al., 2011). A study in El Paso determined Hispanics to be more sensitive to $PM_{2.5}$ than other groups (Grineski et al., 2015). The effects of air quality can deleteriously impact different aspects and quality of life. Schools located near areas with high air pollution been associated with lower attendance records and higher proportion of students failing to meet state educational testing standards (Mohai et al., 2011). The alarming effects of prolonged exposure warrants better understanding of the extent of contribution on air quality from highways on near-road communities.

1.2. Influence of Near-road Emissions on Air Quality

Near-road is attributed to being within 50m from any road with a high Annual Average Daily Traffic (40 CFR Part 58, Appendix E). The influence of traffic generated air pollution at near-road communities has been widely documented (Karner et al., 2010; Padró-Martínez et al., 2012). Emissions from motor vehicles are major contributors to a city's air pollution. They emit nitrogen oxides, carbon monoxide, particulate matter, and other hazardous pollutants. Major roads where traffic is the densest will observe its residents exposed to elevated emissions. The impact of mobile sources from highways is observed to be within the spatial extent of a few hundred meters (Zhou, 2007). A near-road study in Somerville, Massachusetts found the highest pollutant

concentrations to be within 50 m of Interstate 93 (Padró-Martínez et al., 2012). Quantification of traffic generated air pollution is problematic due to the variable geography, meteorology, and time (Karner et al., 2010). Regression models attributed the variation of NO_2 measurements in El Paso to the proximity to highways and elevation (Gonzales et al., 2005). Wind Speed and mixing layer growth rate were revealed to influence NO_x dispersion in El Paso (MacDonald et al., 2001). Keeping other factors constant, wind direction has been found to play a vital role in the variation of roadway emissions (Venkatram et al., 2013).

1.3. Air Quality Measurements

Different approaches have been explored to measure the extent of air pollution at near-road schools and communities. Central ambient monitoring sites operated by Texas Commission on Environmental Quality (TCEQ), ground level monitors, statistical models, land-use regression models, and emission-based air quality models have been used extensively to quantify the impact (Chang et al., 2015; Mejia et al., 2010).

A typical way to assess air quality data is through remote measurements. Data is obtained through central ambient monitoring stations and generalized for the communities surrounding them (Physick et al., 2011; Gauderman et al., 2007; Kim et al., 2005). This method is a cost-effective method of obtaining air quality data since many cities already have central monitoring stations in place. However, high spatial variability and distance from monitoring stations makes this method less accurate than on-site measurements. This creates a problem in exposure studies where air pollution is over or underestimated at sites located at varying distances from the monitoring stations. A study in France found success with this method as onsite measurements at schoolyards produced comparable results with city monitoring stations (Annesi-Maesano et al., 2007).

Computer models are used extensively to assess high exposure sites due to traffic-related air pollution. The Community Line Source (C-LINE) modeling system is one such example. It can compare different scenarios to predict traffic-related air pollution for roadways in the United States

(Barzyk et al., 2015). This screening model has been used in a variety of near-road monitoring studies to help identify areas that could potentially be at risk for elevated air pollution levels. A Southern California Children's Health Study used a line source dispersion model to determine traffic-related air pollution (McConnell et al., 2010). Some of the preferred air quality dispersion models include: AERMOD, CALPUFF, and CALINE3. The availability of quality input data for models such as traffic volume estimates is the main concern in order to obtain accurate results. Wen and company documented dispersion models can predict air pollution near roadways when emission factors, meteorological, and traffic data is available near-road (Wen et al., 2017).

Different spatial scales have been used for exposure estimates. For exposure studies, it is vital to have the most accurate data. On-site measurements provide the most accurate data compared to the remote measurements and computer models. Caution is recommended when determining the location of the monitoring station. Monitoring sites too close to major sources of emissions may overestimate air pollutants at schools. The expense of purchasing and maintaining monitoring equipment makes this method of obtaining air quality data less feasible for some institutions and researchers. The quantity of air monitors in possession limits the locations that can be monitored simultaneously.

1.4. Air Quality Near Schools

The following are examples of studies conducted at schools in North America, Europe, and Asia where air pollution was examined near schools:

- Ambient measurements of $PM_{2.5}$ and PM_{10} were collected for two-week for four seasons at two elementary schools in Detroit, Michigan. Increased levels were found in the southwest community during winter with $PM_{2.5}$ at 20.6 ± 8.1 $\mu g/m^3$, and PM_{10} at 30.8 ± 12.3 $\mu g/m^3$ possibly due to nearby highway and industrial sources (Keeler et al., 2002).
- Concentrations of NO_2 and NO_x were measured outside of 10 elementary schools of varied proximity to freeways in California. Concentrations increased with decreasing downwind

distance for sites within 350 m. Average outdoor NO_2 concentrations at school sites ranged from 19 to 30 ppb (Singer et al., 2004).

- A study conducted in the San Francisco Bay Area recorded concentrations for traffic pollutants at ten schools. Average pollutant levels ranged from 11 to 15 $\mu g/m^3$ for $PM_{2.5}$, 29 to 32 $\mu g/m^3$ for PM_{10}, and 19 to 31 ppb for NO_2 (Kim et al., 2004).
- Particle number counts were recorded at four elementary schools and one university soccer field. Number counts of particles increased the nearer the sites were to the highway. Mean concentration of NO_2 was below 100 ppb, and average ozone levels were 106 ± 47 ppb (Rundell et al., 2006).
- Indoor and outdoor air quality was measured at 39 schools in Barcelona, Spain. Outdoor $PM_{2.5}$ was found to be 1.7 times higher than background levels. $PM_{2.5}$ values ranged from 10 to 111 $\mu g/m^3$. NO_2 concentrations at schools were reported 1.2 times higher than urban background levels. NO_2 values ranged from 14 to 98 $\mu g/m^3$. Higher levels of traffic related air pollutants were reported in the center of the city than in the outskirts (Amato et al., 2014).
- Concentrations of $PM_{2.5}$ and NO_2 were assessed in schoolyards and city monitoring stations in three French cities. Schoolyards and city level concentrations had similar valued for both measured pollutants. $PM_{2.5}$ values ranged from 12.5 to 54 $\mu g/m^3$ and NO_2 values ranged from 31.1 to 70.4 $\mu g/m^3$ (Annesi-Maesano et al., 2007).
- An asthma study measured air pollution at seven elementary schools in South Korea. The three schools identified near traffic zones had average pollutants near the main entrance at approximately 24 to 45 $\mu g/m^3$ for PM_{10}, 11 to 48 ppb for NO_2, and 2 to 35 ppb for ozone (Kim et al., 2016).
- Winter air pollution at three schools in southern England was examined to study the effects on respiration in children. The 24-hr averages ranged from 18.4 to 22.7 $\mu g/m^3$ for PM_{10}, 17.1 to 19.2 ppb for NO_2. The 8-hr maximum moving average for ozone ranged from 19 to 21.6 ppb (Peacock et el., 2003).

- A study in Ciudad Juarez, Chihuahua, Mexico measured air pollutants at schools in a respiratory health study. Two-week NO_2 and 48-hr average concentrations of $PM_{2.5}$ were recorded as mean (SD) of 18.2 ppb (±9.6 ppb) and 17.5 μg/m³ (±8.9 μg/m³) respectively (Holguin et al., 2007).
- A network of passive NO_2 monitors were installed at twenty elementary schools during winter to examine the spatial variation at the El Paso school district. The Chamizal monitoring station experienced the highest concentration (37.7 ppb). Schools in the northwest and northeast recorded the lowest concentrations, 13 ppb and 11 ppb respectively (Gonzales et al., 2005).
- Indoor and outdoor concentrations of PM, black carbon, and NO_2 were measured at four schools in El Paso, Texas, USA, and Ciudad Juarez, Chihuahua, Mexico for 16 weeks. El Paso low and high traffic schools recorded ambient outdoor 96-hr average NO_2 concentrations at 4.5 ppb (±3.5) and 14.2 ppb (±3.2), 48-hr concentrations of $PM_{2.5}$ at 8.3 μg/m³ (±4.1) and 14.5 μg/m³ (±7.8), and 48-hr concentrations of PM_{10} at 18.4 μg/m³ (±8.8) and 39 μg/m³ (±17.6). The air pollutants concentrations were higher at both Ciudad Juarez schools (Raysoni et al., 2011).
- Three schools in 'high-exposure' zones and one school in 'low-exposure' zones were selected to record ambient air quality data. The mean outdoor concentrations at the high exposure schools recorded 48-hr concentrations of $PM_{2.5}$ at 13 to 14 μg/m³, PM_{10} at 35 μg/m³ (±17), and 96-hr average NO_2 concentrations at 9.47 to 10.69 ppb. (Raysoni et al., 2013).

1.5. Air Pollution in El Paso, Texas

National Ambient Air Quality Standards (NAAQS) were established in 1990 by the Environmental Protection Agency (EPA) per the Clean Air Act for criteria pollutants deemed harmful to public health or the environment. El Paso currently meets the National Ambient Air Quality Standards for nitrogen dioxide, $PM_{2.5}$, and ozone (U.S. EPA, 2018). However, El Paso's

desert setting makes attainment of PM_{10} difficult and has led to its nonattainment classification. The Paso del Norte (PdN) air basin is shared by El Paso, Texas, Ciudad Juarez, Chihuahua and Sunland Park, New Mexico. Traffic emissions from the El Paso-Ciudad Juarez border crossings make up a sizable portion of the mobile vehicle emissions in El Paso.

Previous studies have attempted to characterize the air pollution trends in the Paso del Norte air basin. Industrial sources, meteorological conditions, and topography were determined to cause variation on the concentration of air pollutants in the region (Noble et al., 2003). Li and colleagues performed gravitational and chemical analysis to characterize the temporal and spatial variations, along with the composition of particulate matter. PM_{10} and $PM_{2.5}$ were found to increase during the winter months. Particulate matter in arid regions is influenced more by geological sources than by traffic emissions due to the terrain and abundance of unpaved roads (Li et al., 2001). A study conducted from March 1st through June 4th, 2010 across 4 schools found that PM_{10} was greater in the area encompassed by I-10 and the El Paso/Ciudad Juarez border highway (Raysoni et al., 2011).

Nitrogen dioxide has been found to predominate in central El Paso with lower values in East and West part of the city. A winter pilot study witnessed significant variability in NO_2 concentrations across El Paso where NO_2 concentrations decreased as elevation increased (Gonzalez et al., 2005). Land use regression studies further corroborated this and added that concentrations of NO_2 increase during the winter season (Gonzalez et al., 2012; Smith et al. 2006).

Ozone is a secondary pollutant produced from photochemical reactions involving sunlight and gases such as CO, NO_x and VOCs (volatile organic compounds). A 1996 study showed toluene dominated the VOC fraction in El Paso with the central area responsible for majority of total VOC emissions (Fujita, 2001; Funk et al., 2001). A combination of high surface temperatures, strong sunlight, high precursor concentrations, and low convective boundary layer growth rate were the cause of high ozone concentrations (MacDonald et al., 2001). A more recent study performed in 2010 found improvement in the air quality in the region. A comparison between toluene and benzene concentrations in the Lower Valley region revealed a 2 to 3 times decrease in

concentrations compared to the 1999 study (Raysoni et al., 2017; Mukerjee et al., 2004). Ozone is consistent throughout the region. Although the ozone level has been declining, the 8-hour design value was exceeded by 3 central ambient monitoring stations in 2017 (Li et al., 2017).

1.6. Problem Statement

El Paso has twelve central ambient monitoring stations (CAMS). However, few are equipped to measure criteria pollutants ($PM_{2.5}$, PM_{10}, NO_2, CO, O_3). Ozone is widely monitored at El Paso CAMS sites. However, only four CAMS sites in the city of El Paso measure $PM_{2.5}$, three measure CO, three measure NO_2, and two measure PM_{10}. Many of the sites that measure criteria pollutants are located near the U.S.- Mexico international border. The quantification of air pollutant concentration at near-road communities in El Paso is limited. Previous studies in El Paso have focused on areas surrounding I-10 (Raysoni et al., 2011 & 2013). Near-road studies for areas farther north from the border are scarce, and TCEQ monitoring sites are limited. The lack of central monitoring stations near schools and major traffic sources warrants an in-depth inquiry of the levels of air pollution in their neighborhood.

Highways and roadways are major sources of vehicular traffic air emissions. The contribution from these highway emissions on the air quality in the surrounding communities varies at the neighborhood levels. The use of ambient monitoring stations in areas farther from children's area of exposure increases the risk of exposure misclassification. Park and Kwan (2017) stressed the importance of finer spatiotemporal scales to more accurately determine personal exposure. It is imperative that on-site ambient monitors be used to accurately capture air pollution within study areas. This research characterizes air pollutants in three near-road communities.

1.7. Objectives

This research demonstrates the use of on-site air quality monitors to characterize air pollutants in near-road schools and communities to understand children's exposure to traffic-related air pollutants. Ambient monitoring of air pollutants at schools is part of a larger health

study to understand children's exposure to traffic-related air pollutants that included conducting respiratory health measurements in a cohort of children at schools in the selected community

The tasks comprised of installing ambient air monitoring stations at selected schools, collecting ambient air quality data for $PM_{2.5}$, PM_{10}, NO_2, and O_3, collecting meteorological data, and assessing the spatial and temporal variability in the region by comparing air quality obtained from the study at the central ambient monitoring sites.

1.8. Significance of the Work

On-site monitoring of air pollution at near-road schools was performed to accurately capture high resolution variations in air quality. In the past, 48-hr pollutant data was measured from school monitoring studies near I-10 and the El Paso-Ciudad Juarez, Mexico international border highway. By increasing the time resolution to 1-hr and choosing sites not located near central ambient air quality monitors, we can validate previous claims that intra-urban spatial variation exists in the PdN region and examine the diurnal variations from regions outside of the CAMS sites. Determining the influence vehicular emissions from highways and roadways have on the air quality of the surrounding communities can help raise awareness on the adverse effects underserved communities face when living near highways. This nuanced understanding could help policy makers make informed decisions when selecting the location of future schools.

1.9. Data Analysis

Air quality data was plotted and analyzed using Microsoft Excel 2016 and the Python programming language. Wind roses were constructed with Lakes Environmental WRPLOT View software.

CHAPTER 2: STUDY DESIGN

This study was conducted in El Paso, Texas, between October 10 and December 20, 2017 to measure particulate matter ($PM_{2.5}$ & PM_{10}), nitrogen dioxide (NO_2), and ozone (O_3). Three air monitoring stations in El Paso were installed at two schools and a near-road residential home next to U.S. Highway 54. Continuous samples were recorded at 5-min intervals for all pollutants. Using the nearest TCEQ CAMS sites, meteorological and pollutant data were extracted for comparison. Spatial variations were examined, and diurnal patterns were constructed for the measured sites and available CAMS sites. Figure 2.1 below shows the location of the two schools and the residential community sampled in this study along with the CAMS stations.

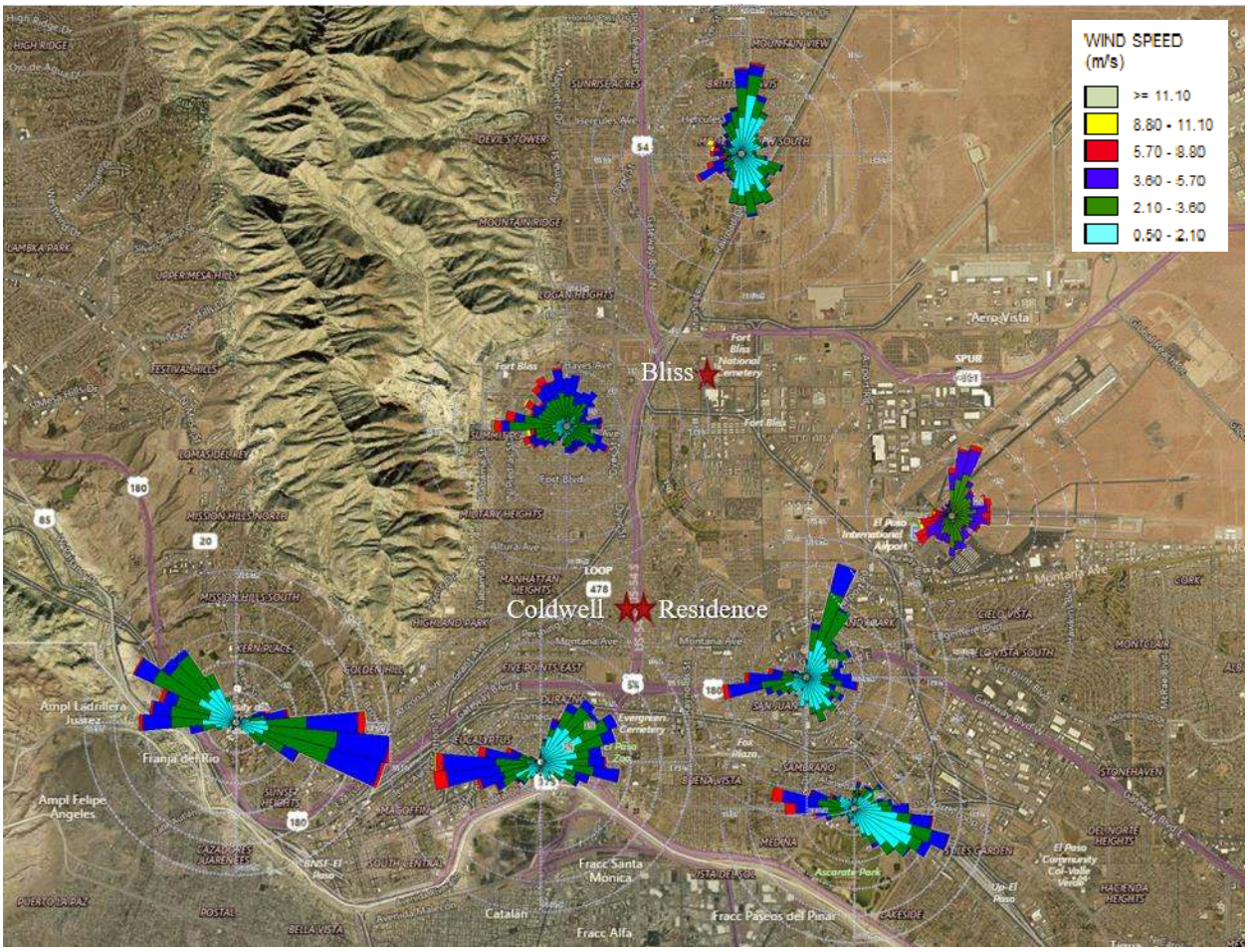


Figure 2.1- Map of sampling sites and wind roses across El Paso using TCEQ CAMS and airport meteorological data.

2.1. Site Selection

Criteria for school selection depended on Annual Average Daily Traffic (AADT), the proximity to the highway, the direction of the prevailing winds, and the number of asthmatic students at the schools. Two schools from the El Paso Independent School District were identified after meeting the required criteria. The proximity to U.S. Highway 54 made Bliss and Coldwell Elementary prime choices for conducting the study. US54 traffic counts are estimated at approximately 107,237 AADT with westerly winds characterizing this region. School Principals were contacted, and necessary protocols were submitted to start the study. Figure 2.1 depicts the sampling sites and wind roses in the surrounding areas during the study period. Wind roses were plotted using software from Lakes Environmental, Inc. CA. A wind rose is a graphical representation of the joint frequency distribution of wind speed and wind direction at a location. Wind speed and direction were obtained from the nearest TCEQ CAMS sites and El Paso Airport for the duration of the study period. Winds during the study period differ significantly by site due to the Franklin Mountains.

The first station was installed at Coldwell Elementary (School CW), located 190ft west of US54. This school had approximately 526 students enrolled. This site was located in a residential area with the school wall on the west, and predominantly paved roads in the immediate surroundings. The second station was installed at Bliss Elementary, located inside of Fort Bliss, 460ft east of US54. Fort Bliss is the second largest United States Army post that houses military personnel and their families. Bliss Elementary had approximately 655 students enrolled. The site was characterized as a large open space next to unpaved grounds with the nearby railway located parallel to US54. A third site, a residential house, was selected opposite of US54 from Coldwell Elementary. This site, located 275ft west of US54, mirrored the Coldwell site with the house wall on the east, and predominantly paved roads in the immediate surroundings. The residential and the Coldwell locations were carefully selected in accordance to the upwind-downwind configuration relative to the prevailing wind direction and the orientation of the highway.

2.2. Set-up

Each monitoring station consisted of three instruments. GRIMM Technologies Aerosol Spectrometer 11-A was used to measure particulate matter ($PM_{2.5}$ and PM_{10}), NO_2 was obtained from 2B Technologies Model 405 $NO_2/NO/NO_x$, and ozone was measured using 2B Technologies Model 202. At both schools, instruments were arranged inside a sheltered perforated cabinet. The monitoring station at the residential house was kept under the front porch, open to the environment. Inflow PTFE (Teflon®) tubing from the monitors to a sampling height of 5ft was maintained with the aid of a retort stand tripod. The tubing was faced down at the ends to limit the influence of high winds and non-air pollutants from entering through the inlet. Inlets were covered by a metal dish, and monitoring stations were kept open to prevent overheating with a table acting as shade and weather protection. Temperature ranges in El Paso during the study period fell well within the acceptable operating temperature ranges for the air monitors, requiring no additional forms of climate control. Data was downloaded twice a week (Tuesday and Friday), unless school holidays prevented it. Weather was routinely monitored to ensure the safe operation of the monitoring stations. Figures 2.2-2.4 depicts the monitoring stations during installment at Bliss Elementary, the residential house, and Coldwell Elementary.

Figure 2.2- Monitoring station at Bliss Elementary.

Figure 2.3- Monitoring station at the residential house.

Figure 2.4- Monitoring station at Coldwell Elementary.

CHAPTER 3: CALIBRATION

Calibration of instruments was performed prior and post the study sampling session. During calibration all instruments were positioned next to TCEQ CAMS 12. Prior to the study, instruments were positioned inside a van parked next to CAMS 12 as shown in Figure 3.1. Tubes ran from the end of the monitors to the top of the van through pre-opened gaps. After the study, monitors were arranged inside a sheltered cabinet next to CAMS 12. Instrument 1-hr averages were compared with data collected from CAMS 12's FRM and FEM devices and with each other to determine precision and accuracy. The readings from both calibrations were lumped together to determine a best fit curve.

Figure 3.1- Instrument calibration set-up at CAMS 12.

3.1. Precision

Precision is defined as the closeness of two of the same instruments. Five-minute measurements were obtained and converted to hourly averages. Instrument 1-hr averages were used for comparison. The linear regression and correlation between two of the same instruments

was calculated to determine the precision of the instruments. Table 3.1 summarizes the results obtained of $PM_{2.5}$ and PM_{10} from GRIMM Technologies Aerosol Spectrometer 11-A, NO_2 from 2B Technologies Model 405, and O_3 from 2B Technologies Model Ozone.

Table 3.1- Linear regression and correlation between instruments.

X	*Y*	*Linear Regression (R^2)*
GRIMM-$PM_{2.5}$ 1	GRIMM-$PM_{2.5}$ 2	y = 0.9048x + 0.1511 (0.997)
GRIMM-$PM_{2.5}$ 1	GRIMM-$PM_{2.5}$ 3	y = 0.9896x – 0.0067 (0.999) [A]
GRIMM-$PM_{2.5}$ 2	GRIMM-$PM_{2.5}$ 3	y = 0.9778x + 0.0501 (0.999) [A]
GRIMM-PM_{10} 1	GRIMM-PM_{10} 2	y = 0.7824x + 0.5804(0.985)
GRIMM-PM_{10} 1	GRIMM-PM_{10} 3	y = 1.1895x – 0.4676 (0.997) [A]
GRIMM-PM_{10} 2	GRIMM-PM_{10} 3	y = 1.1703x – 0.2454 (0.998) [A]
2B Tech-405 1	2B Tech-405 2	y = 1.3628x + 13.341 (0.729) [B]
2B Tech-405 1	2B Tech-405 3	y = 1.1272x + 0.4275 (0.890) [B]
2B Tech-405 2	2B Tech-405 3	y = 0.6599x – 5.1537 (0.794)
2B Tech-Ozone 1	2B Tech-Ozone 2	y = 1.0366x – 0.1236 (0.996) [A]
2B Tech-Ozone 1	2B Tech-Ozone 3	y = 1.0305x – 0.3499 (0.986)
2B Tech-Ozone 2	2B Tech-Ozone 3	y = 1.0107x + 0.9002 (0.996) [A]

[A] *Pre-calibration comparison only*
[B] *Post-calibration comparison only*

3.1.1. Ozone

As shown in Figure 3.2, instrument 1 was relocated from the House to Bliss to replace instrument 2 malfunctioning in November. Instruments 1 and 3 operated for the complete duration of the study. No post-calibration was performed on instrument 2 due to the fact that the instrument was sent back to the manufacturer for checks after it malfunctioned. All 3 instruments demonstrate a close one-to-one linear relationship and strong correlation with each other.

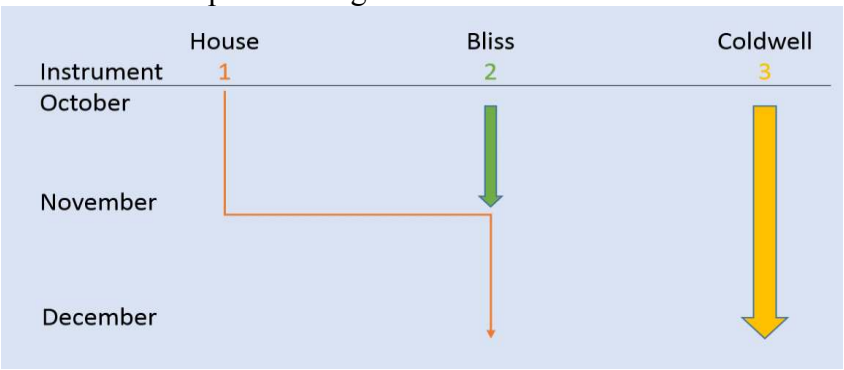


Figure 3.2- Ozone monitor locations during the study period.

3.1.2. Nitrogen Dioxide

Instruments 2 and 3 operated for the complete duration of the study. Unforeseen complications with instrument 1 prevented us from obtaining measurements at the house until November 21st. All three instruments show a high correlation with each other, but the linear regression between instrument 2 and the other instruments varies.

3.1.3. Particulate Matter

Table 3.2 reveals the slopes between GRIMM instruments. Debris in the inlet of instrument 2 arose during post-calibration and prompted us to return to Bliss and CAMS 12 to re-run the GRIMM instruments again 2 months later to assess the validity of the data. Similar results were obtained from side by side comparisons at Bliss and CAMS 12 (Post-1 and Post-2). Instruments 1 and 2 remained consistent throughout the study, but a drift from pre and post study is observed for instrument 3. As seen in Table 3.1, for $PM_{2.5}$ all instruments demonstrate a close linear regression with each other and strong correlation. The linear regression between the instruments for PM_{10} is similar to $PM_{2.5}$.

Table 3.2- Slope between GRIMM instruments.

Unit	*PM*	*Pre*	*Post-1*	*Bliss*	*Post-2*
1~2	2.5	1.011	-	0.931	0.903
	10	1.016	-	0.823	0.771
1~3	2.5	0.978	0.616	0.591	0.593
	10	1.189	0.518	0.491	0.466

3.2. Accuracy

Accuracy is defined as the closeness of measured values from an instrument to a standard value. The accuracy of the instruments was evaluated by computing the linear regression and correlation of each instrument with TCEQ CAMS 12. The instruments at central ambient monitoring stations use the U.S. EPA approved Federal Reference Method (U.S. EPA, 2017). By calibrating our instruments to a CAMS site, comparisons with other CAMS sites can be established from the results obtained at the study sites. Five-minute measurements were obtained and

converted to hourly averages. Table 3.3 summarizes the results obtained of $PM_{2.5}$ and PM_{10} from GRIMM, NO_2 from 2B Technologies Model 405, and O_3 from 2B Technologies Model Ozone.

Table 3.3- Linear regression and correlation between instruments and TCEQ CAMS12.

X	*Y*	*Linear Regression (R^2)*
GRIMM-$PM_{2.5}$ 1	CAMS12	y = 0.6649x + 2.3405 (0.856)
GRIMM-$PM_{2.5}$ 2	CAMS12	y = 0.6703x + 2.7425 (0.836)*
GRIMM-$PM_{2.5}$ 3	CAMS12	y = 1.0749x + 2.1609 (0.835)
GRIMM-PM_{10} 1	CAMS12	y = 1.2395x + 9.8322 (0.905)
GRIMM-PM_{10} 2	CAMS12	y = 2.4181x + 5.7214 (0.857)*
GRIMM-PM_{10} 3	CAMS12	y = 1.9944x + 11.160 (0.762)
2B Tech-405 1	CAMS12	y = 1.0880x + 1.3371 (0.895)
2B Tech-405 2	CAMS12	y = 0.6083x – 3.3454 (0.706)
2B Tech-405 3	CAMS12	y = 0.8601x + 2.1692 (0.777)
2B Tech-Ozone 1	CAMS12	y = 1.1650x – 3.1970 (0.889)
2B Tech-Ozone 2	CAMS12	y = 0.9268x + 2.9831 (0.751)
2B Tech-Ozone 3	CAMS12	y = 1.1253x – 2.6891 (0.892)

**Post-calibration was performed 2 months after other instruments.*

3.2.1. Ozone

The 2B Technologies Ozone instruments performed the best in contrast to the other instruments. As seen in Figure 3.3, all ozone instruments show a high correlation and a linear regression close to one-to-one.

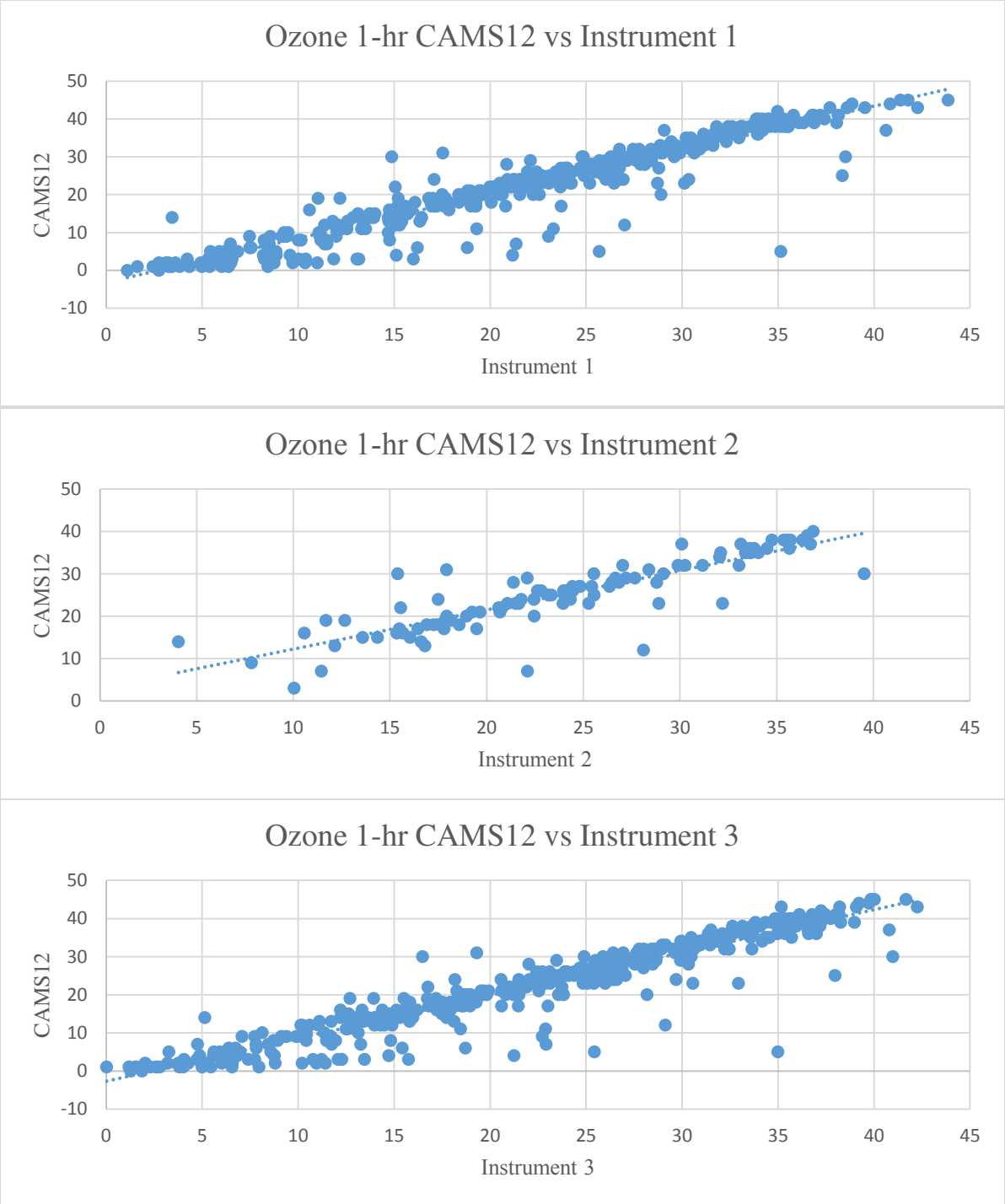


Figure 3.3- Ozone 1-hr CAMS12 vs instrument calibration.

3.2.2. Nitrogen Dioxide

As seen in Figure 3.4, the NO_2 instruments vary in both correlation and linear regression. Instrument 1 has a strong correlation and linear regression close to one-to-one. Instrument 3

performed the second best. Instrument 2 is the least reliable with a linear regression slope of 0.6; although, still showing a high correlation (0.706).

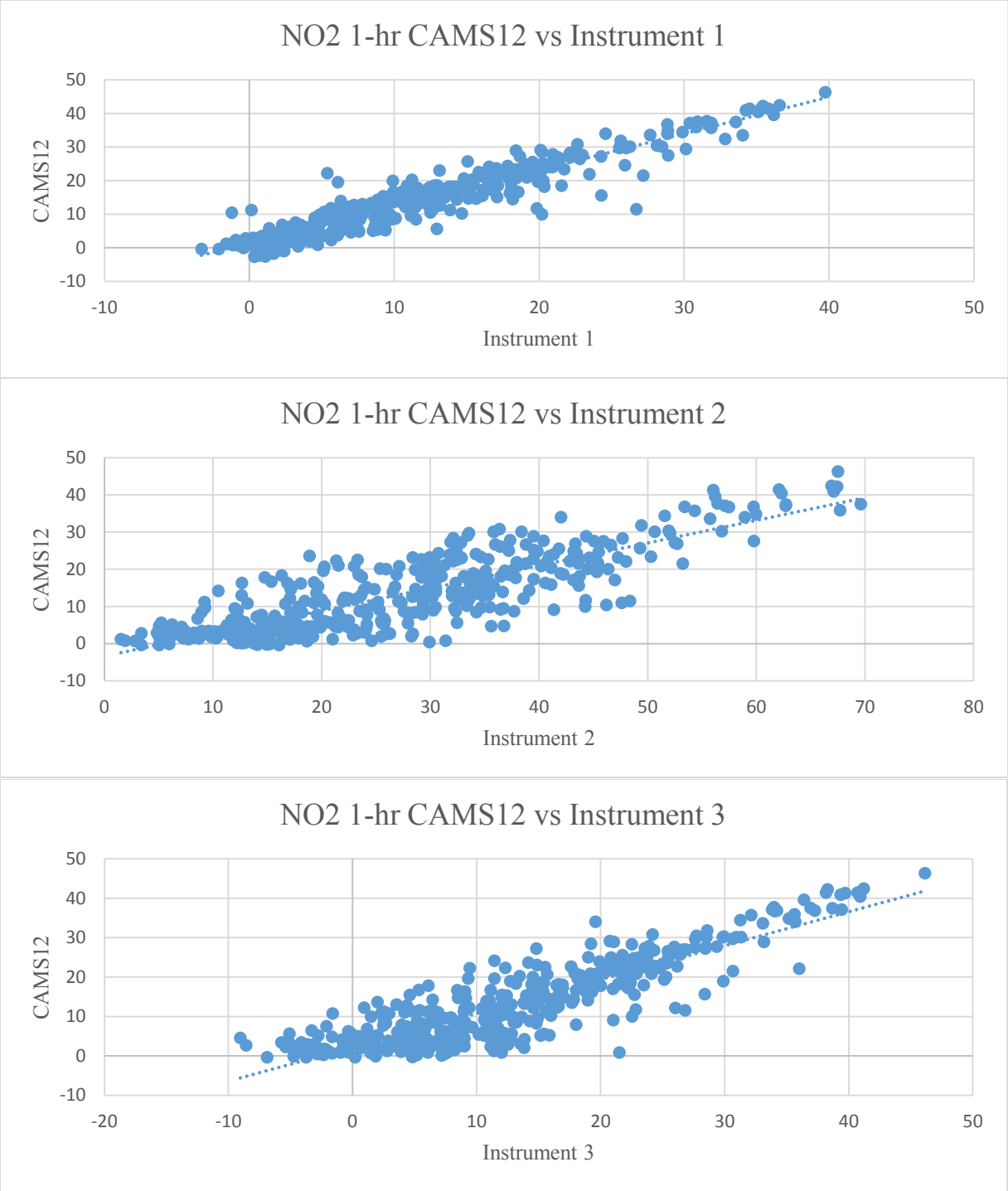


Figure 3.4- NO_2 1-hr CAMS12 vs instrument calibration.

3.2.3. Particulate Matter

For $PM_{2.5}$, all the instruments show a high correlation. Instrument 3 recorded similar values as CAMS 12, but instruments 1 and 2 over-recorded. For PM_{10}, the linear regression varies significantly from the $PM_{2.5}$ regression line obtained from the same instruments. PM_{10} is under-recorded in all instruments. A shown in Table 3.4, a drift from pre and post study is observed for all GRIMM instruments.

Table 3.4- Slope of GRIMM instruments vs CAMS12.

Unit	PM	Pre	Field	Post-1	Post-2
1	2.5	0.576	Pre + Post-1	0.667	0.597
	10	1.214		1.245	2.010
2	2.5	0.567	Pre + Post-2	-	0.661
	10	1.188		-	2.610
3	2.5	0.581	Pre + Post-1	1.08	0.996
	10	1.017		2.363	4.315

3.3. Data Adjustments

The downtime from when data was being downloaded from the instruments left an hour of missing data. To avoid having additional missing data, the hour was interpolated. Negative values indicate a below detection limit and were assigned a value of 0.5 $\mu g/m^3$ for $PM_{2.5}$ and 0.5 ppb for ozone and NO_2, half of the detection limit.

CHAPTER 4: RESULTS AND DISCUSSION

4.1. Summary of Data

Hourly averages for each pollutant were calculated from the 5-minute readings and adjusted according to the linear regression equations from Table 3.3. Figures 4.1-4.4 depict the hourly time series data from each of the monitoring stations. While all 3 monitoring stations demonstrated a similar trend throughout the study period for PM, the station at Bliss Elementary consistently logged the highest readings. For $PM_{2.5}$, Bliss recorded an average value of 17.8 μg/m³, Coldwell 11.6 μg/m³, and house 8.5 μg/m³. For PM_{10}, Bliss recorded an average value of 55.7 μg/m³, Coldwell 42.9 μg/m³, and house 30.4 μg/m³. In contrast, for NO_2 the station at Coldwell Elementary exhibited the highest values. Bliss recorded an average value of 14.9 ppb, Coldwell 18.4 ppb, and house 16.1 ppb. As seen in Table 4.1, ozone values were the most consistent across the sites. Ozone values for the 3 monitoring stations were nearly identical for Coldwell Elementary and the residential house. Bliss Elementary exhibited the same general trend, but recorded slightly higher values. An examination of the max 8-hr ozone continuous averages supports the claim that ozone is higher at Bliss. Ozone is a secondary pollutant with precursors including NO_x and VOCs. The difference between non-recorded precursor emissions from Coldwell and Bliss could potentially play a role in the creation of ozone.

Nitrogen dioxide and $PM_{2.5}$ are acknowledged to be good indicators for emissions originating from traffic. The distance to the highway is an important variable when determining near-road impact of traffic pollutants. The station at Bliss Elementary is located the farthest from the highway and has the lowest readings for NO_2. To prove that closer proximity to US-54 increases NO_2 would require measuring the background concentrations in the study region. This station at Bliss was also in an area with predominantly unpaved grounds. Particulate matter in arid regions is influenced more by geological sources than by traffic emissions. It is plausible that the surroundings have a greater influence on particulate matter than traffic emissions coming from US54. Boxplots were plotted to illustrate the variation in pollutants across the 3 sites. As

demonstrated in Figure 4.5, Bliss elementary varied the greatest for particulate matter, PM_{10} and $PM_{2.5}$. The house varied the greatest for NO_2, and ozone was near identical at all three sites.

Pollutants concentrations may vary by season. Winter season pollutant concentrations in El Paso may be higher for PM and NO_2. Average pollutant concentrations for PM and NO_2 during the study period, seen in Table 4.1, may be a conservative representation for the year. However, the higher ozone concentrations occur during the summer months, and thus average O_3 concentrations during the study may be lower than other seasons.

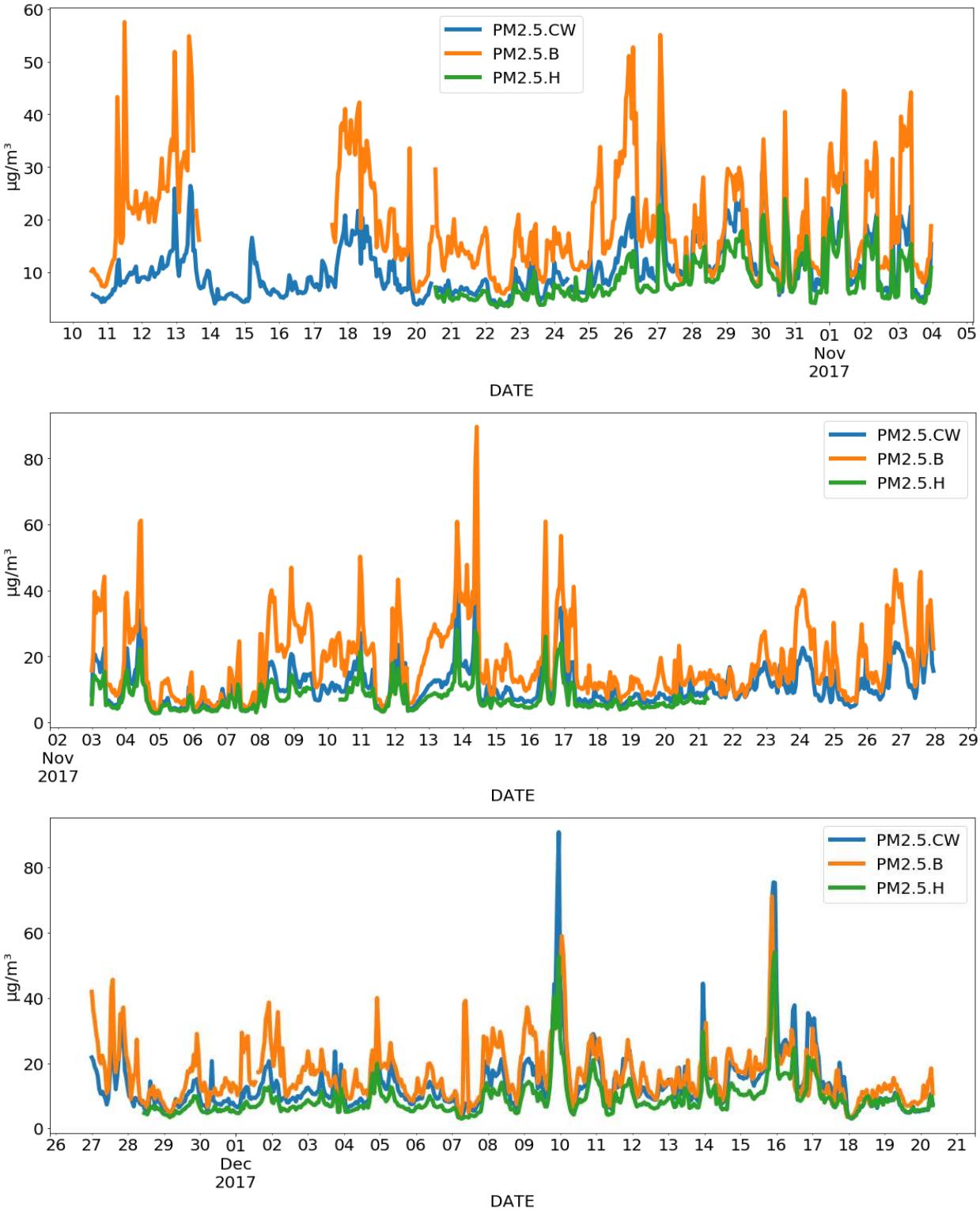


Figure 4.1- Time series of 1-hr averages of $PM_{2.5}$ for CW (Coldwell Elementary), B (Bliss Elementary), and H (residential house) during the study period.

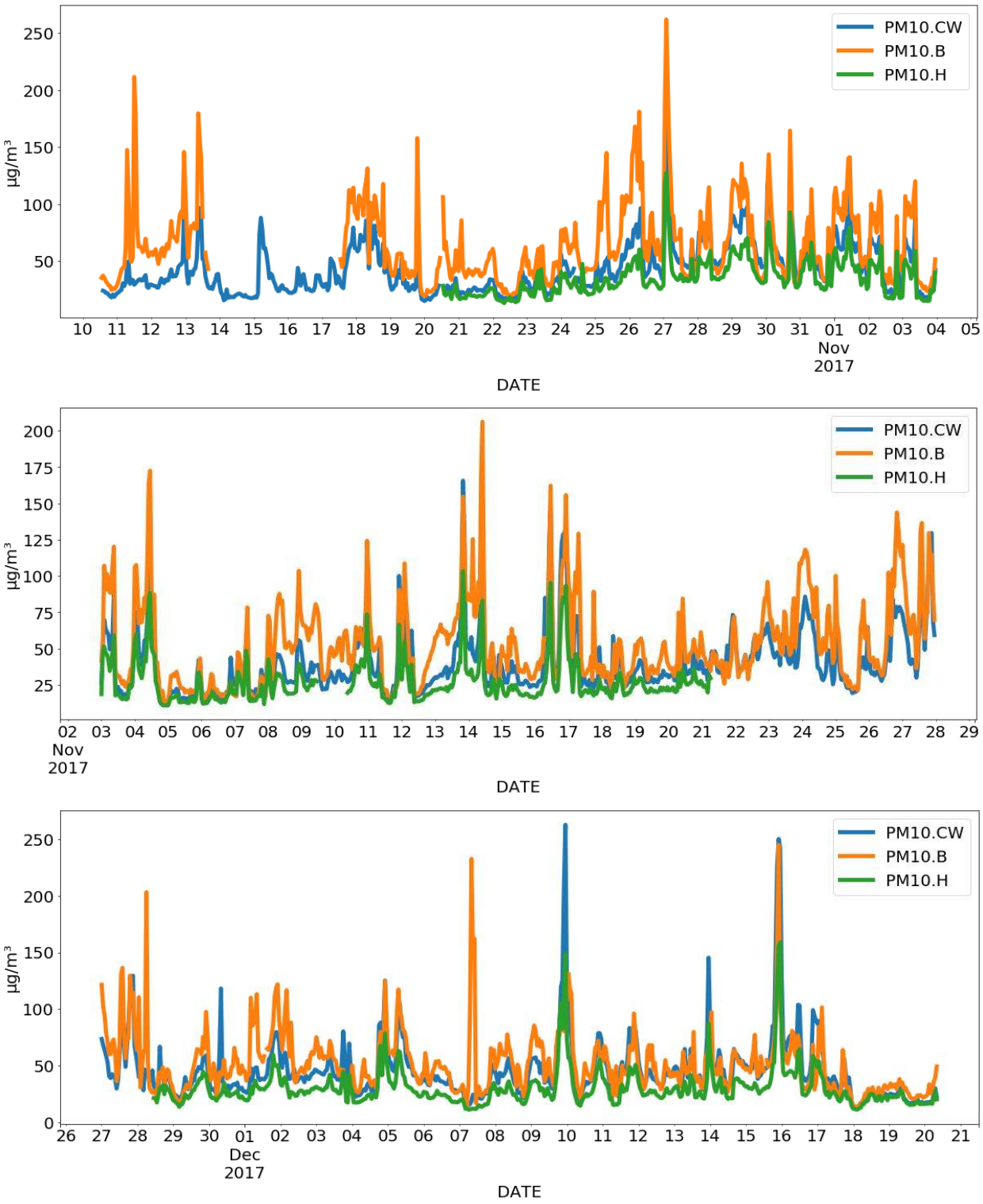


Figure 4.2- Time series of 1-hr averages of PM_{10} for CW (Coldwell Elementary), B (Bliss Elementary), and H (residential house) during the study period.

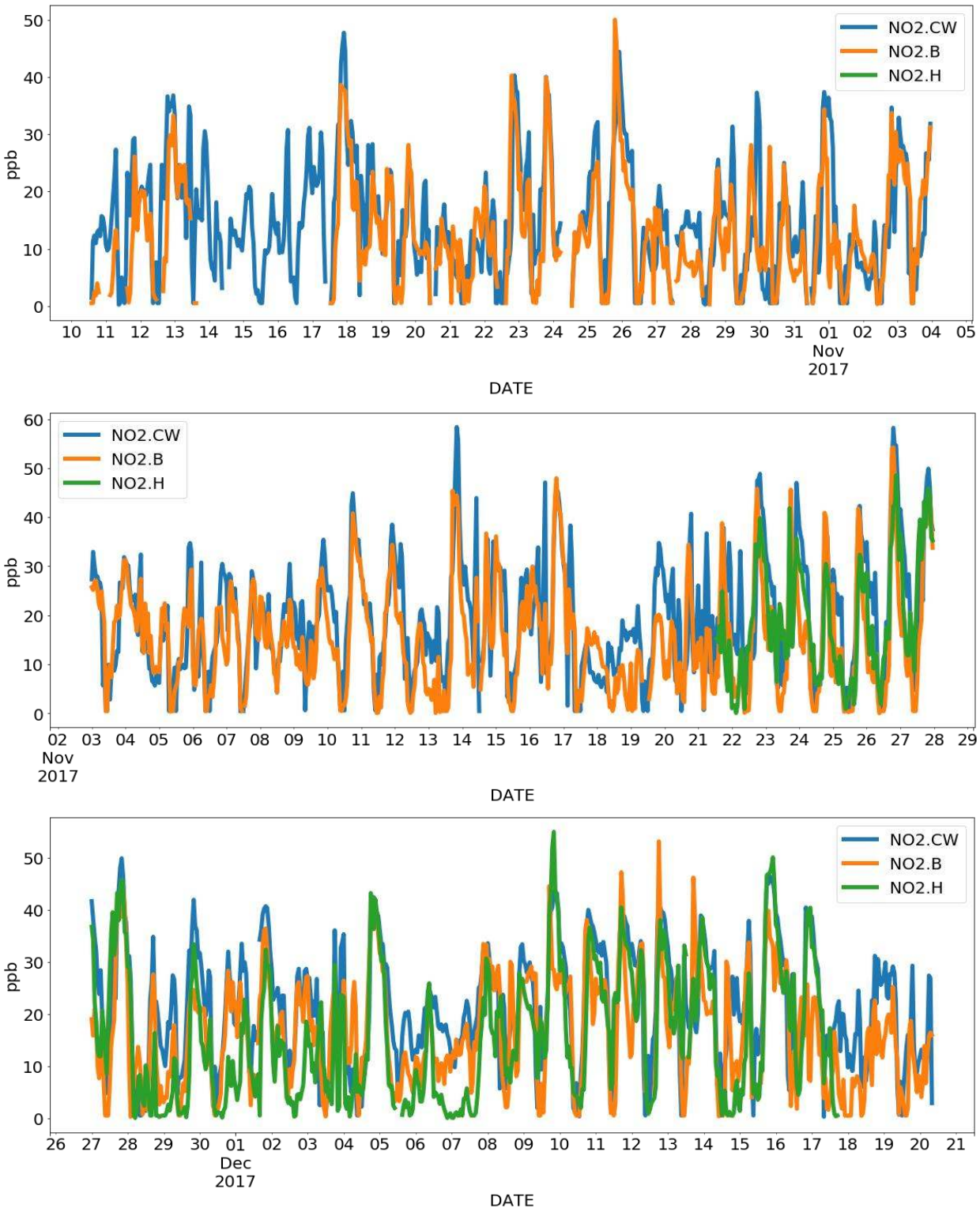


Figure 4.3- Time series of 1-hr averages of NO_2 for CW (Coldwell Elementary), B (Bliss Elementary), and H (residential house) during the study period.

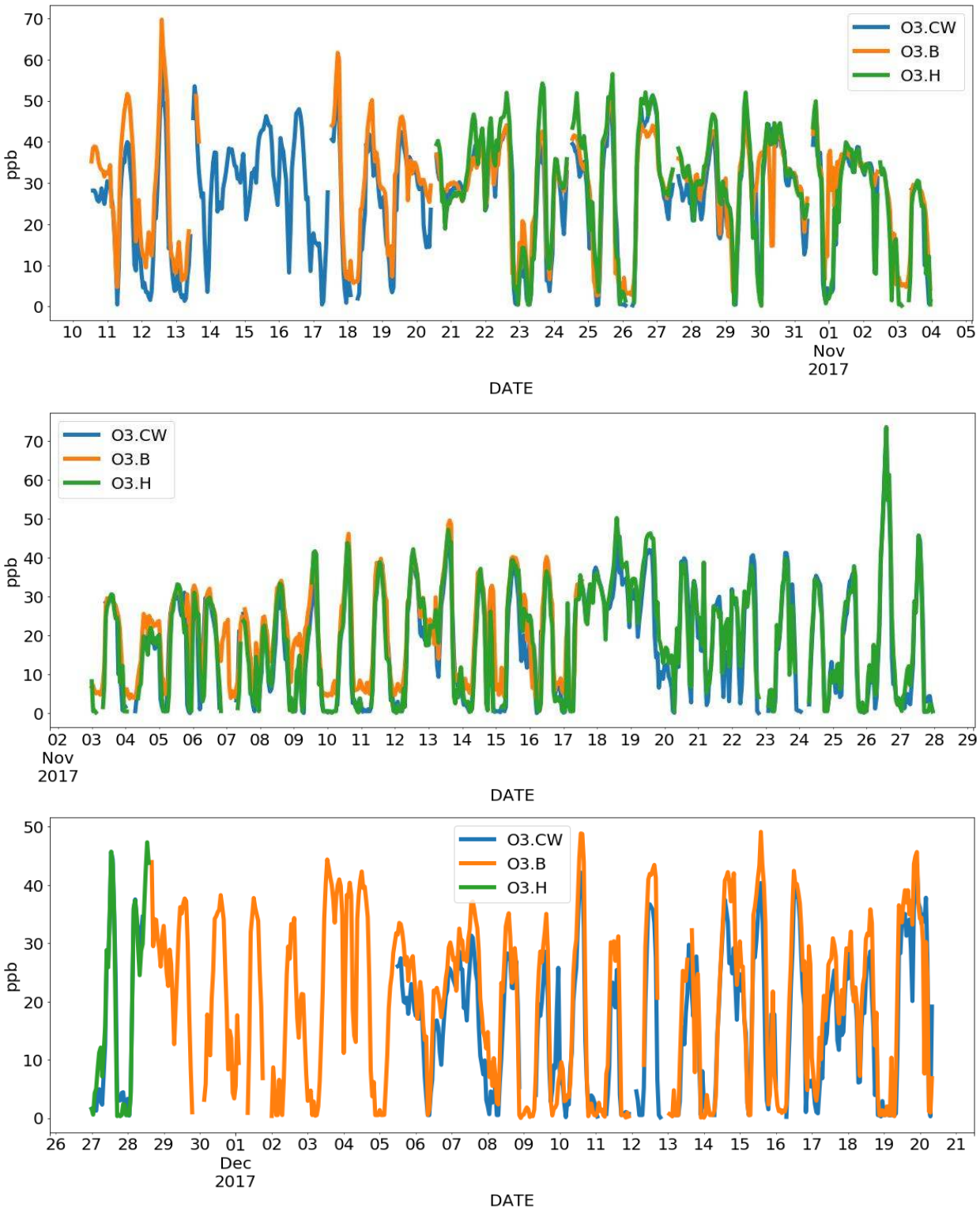


Figure 4.4- Time series of 1-hr averages of O_3 for CW (Coldwell Elementary), B (Bliss Elementary), and H (residential house) during the study period.

Table 4.1- Statistics of time series data for 1-hr pollutant concentrations across monitoring sites.

	COUNT	MEAN	STD	MIN	25%	50%	75%	MAX	UNITS
PM2.5.CW	1695	11.6	7.3	2.73	7.07	9.60	14.35	90.88	µg/m³
PM2.5.B	1605	17.8	10.2	3.15	10.62	14.82	22.66	89.63	µg/m³
PM2.5.H	1268	8.5	5.1	2.71	5.33	7.02	10.16	54.13	µg/m³
PM10.CW	1695	42.9	24.4	12.34	27.15	37.10	51.44	262.57	µg/m³
PM10.B	1605	55.7	30.8	13.15	34.78	47.90	67.38	262.03	µg/m³
PM10.H	1268	30.4	16.3	10.68	20.28	26.37	35.19	158.80	µg/m³
NO2.CW	1675	18.4	11.3	0.22	9.64	16.71	26.64	58.37	ppb
NO2.B	1547	14.9	10.5	0.04	7.03	13.36	21.65	54.21	ppb
NO2.H	624	16.1	12.4	0.02	5.40	13.61	25.05	54.95	ppb
O3.CW	1410	21.3	13.9	0.01	8.10	22.62	31.97	73.30	ppb
O3.B	1290	23.2	13.7	0.01	9.91	25.44	33.82	69.68	ppb
O3.H	874	23.4	14.8	0.05	10.69	25.44	34.41	73.59	ppb

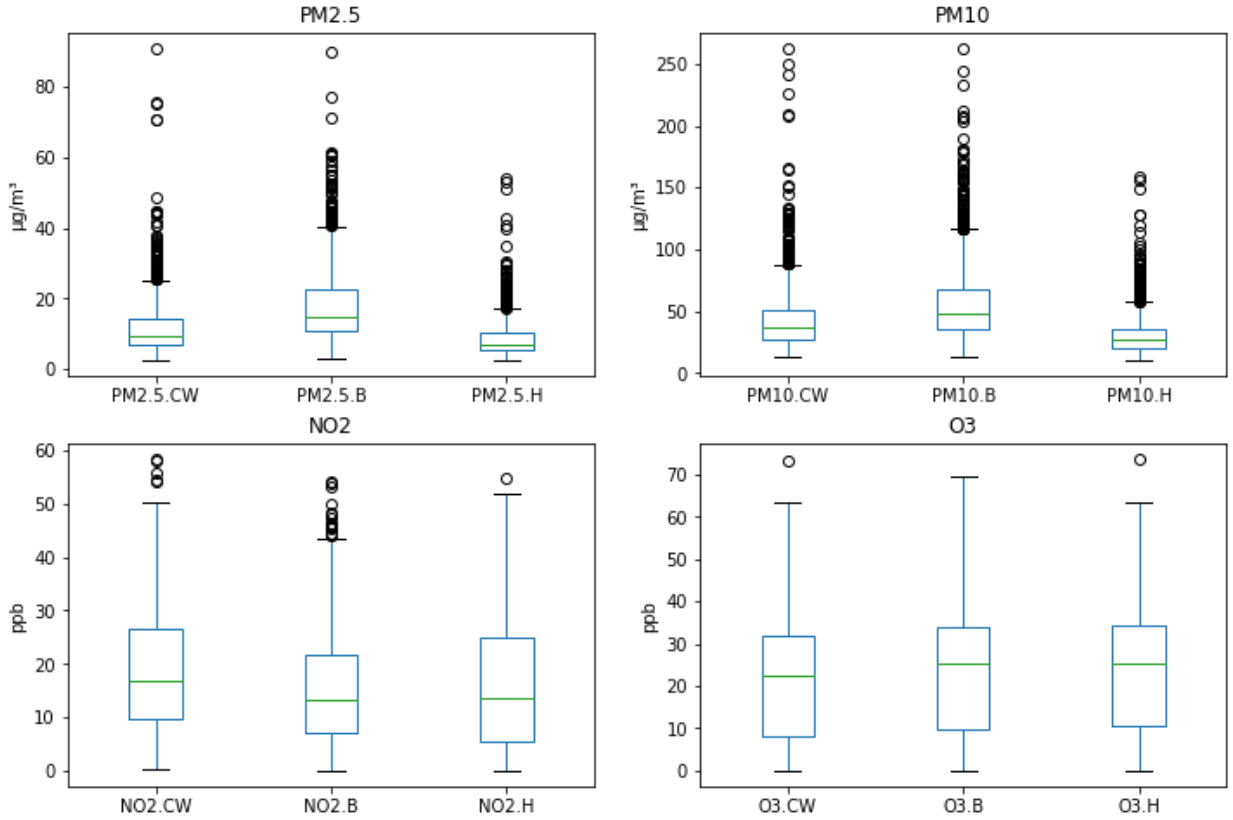


Figure 4.5- Boxplots of 1-hr averages of $PM_{2.5}$, PM_{10}, NO_2, and Ozone for CW (Coldwell Elementary), B (Bliss Elementary), and H (residential house) during the study period.

4.2. Pollutant Correlations Across the Study Sites

Spatiotemporal variation within the El Paso region can be attributed to meteorological conditions, sources of emissions, and topography. The Franklin Mountains dividing El Paso adds to the spatio-variability of air pollution in the region. Vehicular traffic patterns and wind vary throughout El Paso. Higher vehicle activity is observed near the border crossings. Pollutants from vehicle exhaust like PM and NO_2 are of special interest. The neighborhoods at the study sites do not have nearby CAMS sites to gauge the background concentrations of pollutants within the community. It is important that different communities have knowledge of the spatial variability in the region.

4.3. Inter-pollutant Correlations

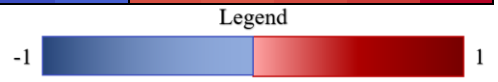
Inter-pollutant and intra-pollutant correlations were calculated based on hourly concentrations with the intent to demonstrate similarity of paired sites (Patton et al., 2014; Raysoni, 2011; Pinto et al., 2004; Wilson et al., 2005, Physick, 2011). Table 4.2 shows the Spearman correlation coefficients for the pollutants at the 3 sites and available CAMS sites. The 3 monitored sites exhibited strong correlations for all pollutants. This suggests that similar temporal trends can be observed in the communities surrounding these sites for all measured pollutants. Inter-pollutant Spearman correlations varied by site. In general, correlations were lower at Bliss Elementary. At every site Spearman correlations were highest among particulate pollutants. $PM_{2.5}$ correlations with NO_2 were moderate for both Coldwell ($r = 0.48$, p-value < 0.001) and Bliss ($r = 0.38$, p-value < 0.001). A previous study showed higher Spearman correlations between gases (NO, NO_x, CO) (Patton et al., 2014). We were not able to confirm this due to our study protocol only measuring NO_2. However, the residential house had high correlation between NO_2 and $PM_{2.5}$ ($r = 0\ .7$, p-value < 0.001). Ozone had negative correlations with NO_2 at Bliss ($r = -0.62$, p-value < 0.001) and Coldwell ($r = -0.67$, p-value < 0.001).

The $PM_{2.5}$ and PM_{10} correlations between Coldwell and the residential home exhibited the highest correlation, both at ($r = 0.96$, p-value < 0.001). It is expected that the two sites have the

highest correlation between sites due to their close proximity to one another. The correlation between the particulate species was also strong between Coldwell and Bliss ($r = 0.8$, p-value < 0.001).

Table 4.2- Intra-site and inter-pollutant Spearman's correlation coefficients.

		$PM_{2.5}$							PM_{10}					NO_2					O_3				
	Site	**CW**	**B**	**H**	**C12**	**C37**	**C41**	**C49**	**CW**	**B**	**H**	**C12**	**C41**	**CW**	**B**	**H**	**C12**	**C37**	**CW**	**B**	**H**	**C37**	**C41**
$PM_{2.5}$	**CW**	1	0.8	0.96	0.54	0.57	0.54	0.4	0.93	0.78	0.88	0.47	0.46	0.48	0.37	0.7	0.47	0.42	-0.4	-0.4	-0.3	-0.4	-0.4
	B	0.8	1	0.77	0.45	0.35	0.33	0.32	0.71	0.92	0.7	0.36	0.32	0.42	0.38	0.63	0.41	0.37	-0.3	-0.4	-0.4	-0.3	-0.4
	H	0.96	0.77	1	0.51	0.53	0.51	0.37	0.86	0.72	0.88	0.37	0.39	0.4	0.31	0.7	0.38	0.38	-0.3	-0.3	-0.3	-0.3	-0.4
	C12	0.54	0.45	0.51	1	0.56	0.51	0.57	0.51	0.42	0.5	0.77	0.49	0.52	0.52	0.57	0.73	0.55	-0.4	-0.4	-0.4	-0.5	-0.6
	C37	0.57	0.35	0.53	0.56	1	0.62	0.59	0.52	0.32	0.45	0.43	0.49	0.41	0.42	0.57	0.53	0.65	-0.4	-0.3	-0.3	-0.5	-0.4
	C41	0.54	0.33	0.51	0.51	0.62	1	0.41	0.51	0.33	0.46	0.42	0.56	0.45	0.4	0.61	0.5	0.51	-0.3	-0.3	-0.3	-0.4	-0.5
	C49	0.4	0.32	0.37	0.57	0.59	0.41	1	0.42	0.31	0.4	0.43	0.4	0.41	0.47	0.55	0.42	0.59	-0.3	-0.3	-0.4	-0.5	-0.4
PM_{10}	**CW**	0.93	0.71	0.86	0.51	0.52	0.51	0.42	1	0.8	0.96	0.57	0.54	0.41	0.32	0.67	0.37	0.34	-0.2	-0.2	-0.2	-0.2	-0.3
	B	0.78	0.92	0.72	0.42	0.32	0.33	0.31	0.8	1	0.8	0.46	0.37	0.35	0.31	0.55	0.31	0.27	-0.2	-0.2	-0.2	-0.2	-0.3
	H	0.88	0.7	0.88	0.5	0.45	0.46	0.4	0.96	0.8	1	0.54	0.51	0.35	0.27	0.62	0.29	0.29	-0.2	-0.2	-0.2	-0.2	-0.3
	C12	0.47	0.36	0.37	0.77	0.43	0.42	0.43	0.57	0.46	0.54	1	0.62	0.3	0.29	0.42	0.48	0.29	-0.2	-0.1	-0.1	-0.1	-0.3
	C41	0.46	0.32	0.39	0.49	0.49	0.56	0.4	0.54	0.37	0.51	0.62	1	0.28	0.25	0.51	0.38	0.36	-0.2	-0.2	-0.2	-0.2	-0.3
NO_2	**CW**	0.48	0.42	0.4	0.52	0.41	0.45	0.41	0.41	0.35	0.35	0.3	0.28	1	0.74	0.75	0.63	0.6	-0.7	-0.6	-0.6	-0.6	-0.7
	B	0.37	0.38	0.31	0.52	0.42	0.4	0.47	0.32	0.31	0.27	0.29	0.25	0.74	1	0.64	0.6	0.68	-0.6	-0.6	-0.5	-0.7	-0.6
	H	0.7	0.63	0.7	0.57	0.57	0.61	0.55	0.67	0.55	0.62	0.42	0.51	0.75	0.64	1	0.75	0.83	-0.6	-0.7	-0.6	-0.7	-0.7
	C12	0.47	0.41	0.38	0.73	0.53	0.5	0.42	0.37	0.31	0.29	0.48	0.38	0.63	0.6	0.75	1	0.75	-0.6	-0.6	-0.6	-0.7	-0.8
	C37	0.42	0.37	0.38	0.55	0.65	0.51	0.59	0.34	0.27	0.29	0.29	0.36	0.6	0.68	0.83	0.75	1	-0.6	-0.6	-0.6	-0.8	-0.7
O_3	**CW**	-0.4	-0.3	-0.3	-0.4	-0.4	-0.3	-0.3	-0.2	-0.2	-0.2	-0.2	-0.2	-0.7	-0.6	-0.6	-0.6	-0.6	1	0.91	0.95	0.79	0.84
	B	-0.4	-0.4	-0.3	-0.4	-0.3	-0.3	-0.3	-0.2	-0.2	-0.2	-0.1	-0.2	-0.6	-0.6	-0.7	-0.6	-0.6	0.91	1	0.91	0.8	0.81
	H	-0.3	-0.4	-0.3	-0.4	-0.3	-0.3	-0.4	-0.2	-0.2	-0.2	-0.1	-0.2	-0.6	-0.5	-0.6	-0.6	-0.6	0.95	0.91	1	0.83	0.85
	C37	-0.4	-0.3	-0.3	-0.5	-0.5	-0.4	-0.5	-0.2	-0.2	-0.2	-0.1	-0.2	-0.6	-0.7	-0.7	-0.7	-0.8	0.79	0.8	0.83	1	0.89
	C41	-0.4	-0.4	-0.4	-0.6	-0.4	-0.5	-0.4	-0.3	-0.3	-0.3	-0.3	-0.3	-0.7	-0.6	-0.7	-0.8	-0.7	0.84	0.81	0.85	0.89	1



4.4. Spatial Variation in Pollutant Concentrations

Continuous Air Monitoring Stations (CAMS) measure ambient air quality and meteorological data throughout the city, courtesy of TCEQ. A CAMS site near the study area was not available to compare with data collected. However, comparison with stations in different areas of El Paso would reveal if intra-urban spatial variability in air pollution levels exists. The location of CAMS 12, 37, 41, and 49 used for comparison are given in Table 4.3. CAMS 12 is located next to the University of Texas at El Paso campus. CAMS 37 is located inside Ascarate Park, north of the US-Mexico Border Highway. CAMS 41 is located near the Chamizal National Memorial, north of Bridge of the Americas. CAMS 49 is located is located in the lower valley of El Paso.

Spatial variability across the measured sites and CAMS for measured pollutants was evaluated using coefficients of divergence (COD). COD values indicate the differences between the concentrations of pollutants at simultaneously sampled monitoring sites (Raysoni, 2011; Pinto et al., 2004; Krudysz et al., 2008). The COD is calculated from simultaneously sampled sites, j and k by:

$$COD_{i,k} = \frac{1}{p}\sum_{i-1}^{p}\left[\frac{x_{i,j} - x_{i,k}}{x_{i,j} + x_{i,k}}\right]^2$$

Table 4.3- Location of TCEQ CAMS sites in El Paso.

TCEQ Site	Latitude	Longitude	Address
CAMS 12 (UTEP)	31˚46' 6" N	-106˚30' 5" W	250 Rim Road
CAMS 37 (Ascarate)	31˚44' 48" N	-106˚24' 10" W	650 R E Thomason Loop
CAMS 41 (Chamizal)	31˚45' 56" N	-106˚27' 19" W	800 S.San Marcial Street
CAMS 49 (Socorro)	31˚40' 3" N	-106˚17' 17" W	320 Old Hueco Tanks Road

COD values were calculated from simultaneous on-site measurements and CAMS data for hourly and 24-hr averages in Tables 4.4 and 4.5. A COD value less than or equal to 0.2 implies homogeneity in the pollutant concentration between two sites. This is observed for $PM_{2.5}$ between Coldwell and the residential house (0.17). For PM_{10}, homogeneity is inferred between Coldwell and the residential house (0.18), and Coldwell and Bliss (0.18). A COD value close to 1 implies non-uniformity between two sites. Based on the hourly ambient air pollution concentrations, there exists slight spatial heterogeneity between Coldwell and CAMS 12 for $PM_{2.5}$ (0.27) and PM_{10} (0.24). Moderate to high spatial heterogeneity can be implied for the three measured sites and CAMS sites.

Table 4.4- Coefficient of divergence values based on hourly concentrations.

Pollutant	Site	B	H	UTEP CAMS12	Ascarate CAMS37	Chamizal CAMS41	Socorro CAMS49
$PM_{2.5}$	CW	0.25	0.17	0.27	0.32	0.45	0.36
	B		0.35	0.40	0.42	0.53	0.48
	H			0.24	0.36	0.41	0.32
	CAMS12				0.34	0.43	0.30
	CAMS37					0.44	0.36
	CAMS41						0.45
PM_{10}	CW	0.18	0.18	0.24		0.56	
	B		0.29	0.30		0.59	
	H			0.25		0.54	
	CAMS12					0.54	
NO_2	CW	0.36	0.44	0.43	0.41		
	B		0.45	0.44	0.39		
	H			0.39	0.37		
	CAMS12				0.53		
O_3	CW	0.33	0.24	0.34	0.37	0.34	
	B		0.33	0.34	0.41	0.37	
	H			0.35	0.37	0.34	
	CAMS12				0.35	0.28	
	CAMS37					0.27	

Finer time resolutions reveal greater variability in pollutant concentrations. Examining the COD from 24-hr averages, shown in Table 4.5, reveals more homogeneity. Ozone and NO_2

between most sites is assumed to be homogeneous for this larger time average. Heterogeneity in PM is still pronounced at this time scale. Bliss Elementary shows the highest COD values for PM; implying greater heterogeneity between this site and the rest of El Paso.

Table 4.5- Coefficient of divergence values based on 24-hr average concentrations.

Pollutant	Site	B	H	CAMS12	CAMS37	CAMS41	CAMS49
$PM_{2.5}$	CW	0.23	0.17	0.16	0.23	0.24	0.25
	B		0.35	0.33	0.32	0.38	0.40
	H			0.13	0.32	0.23	0.18
	CAMS12				0.30	0.22	0.18
	CAMS37					0.27	0.29
	CAMS41						0.26
PM_{10}	CW	0.16	0.18	0.14		0.23	
	B		0.29	0.22		0.33	
	H			0.18		0.21	
	CAMS12					0.23	
NO_2	CW	0.14	0.24	0.20	0.18		
	B		0.17	0.17	0.17		
	H			0.19	0.18		
	CAMS12				0.17		
O_3	CW	0.10	0.07	0.08	0.13	0.06	
	B		0.09	0.10	0.19	0.12	
	H			0.08	0.15	0.07	
	CAMS12				0.16	0.08	
	CAMS37					0.12	

4.5. Temporal Variation

The 3 monitoring sites exhibited the same general trend for PM throughout the day. Figures 4.6 and 4.7 depict the weekday and weekend diurnal averages for $PM_{2.5}$ and PM_{10} during the study period. During the weekdays, PM experienced peaks in the morning around 7:00. The 7:00 peak can be attributed to the morning rush hour, but the midnight peak could be the result of stable atmospheric conditions. In contrast, the weekends see a spike in the nighttime hours, specifically from 24:00 to 2:00. It is interesting to observe the inconsistency between Bliss and Coldwell during the weekend nighttime hours. An increase in PM occurs between 24:00 to 2:00 at Bliss, while there is a decrease at Coldwell. Comparison with CAMS sites reveals varying trends. For $PM_{2.5}$, CAMS

12, 41, and 49 follow similar trends with each other, comparable with Coldwell and the residential house. However, CAMS 37 follows a unique trend. The variations at CAMS 37 change at a steeper rate.

Figure 4.8 shows the diurnal weekday and weekend trends of NO_2. The 3 monitored sites and the CAMS sites follow similar trends. Weekday concentrations of NO_2 reaches a morning maximum around 7:00. There is a steady decrease until 15:00-16:00, apart from Coldwell. NO_2 concentrations increase solely at Coldwell around 8:30-11:00. Nitrogen dioxide concentrations at Coldwell begin to increase earlier around 12:00. Weekend concentrations are similar to the weekday concentrations except there is not a morning peak during the weekend. Overall, concentrations of NO_2 are highest after sunset. This is expected because of the ozone photochemistry where first NO_2 is split into NO and an oxygen atom by sunlight.

Figure 4.9 shows the diurnal weekday and weekend ozone cycle recorded. The monitored sites and CAMS sites recorded similar trends with little variation. As expected, ozone increases when sunlight is the greatest. Ozone begins to increase in the morning until reaching a maximum concentration for both weekdays and weekends at approximately 14:00. Slight variations exist in the trends comparing weekdays and weekends, but weekends recorded higher average peak concentrations.

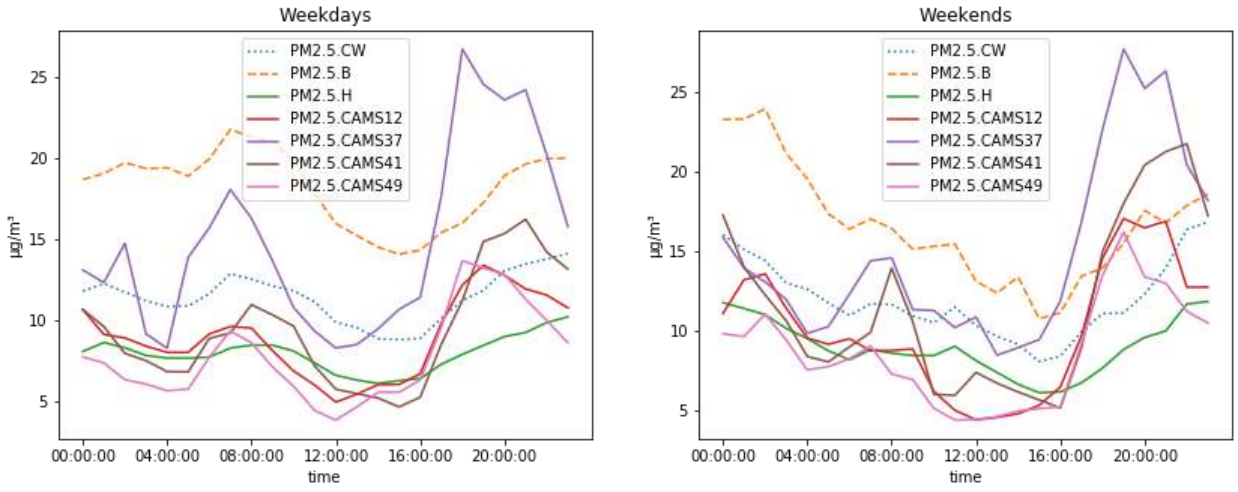


Figure 4.6- Diurnal averages of $PM_{2.5}$ for measured sites and CAMS during the study period.

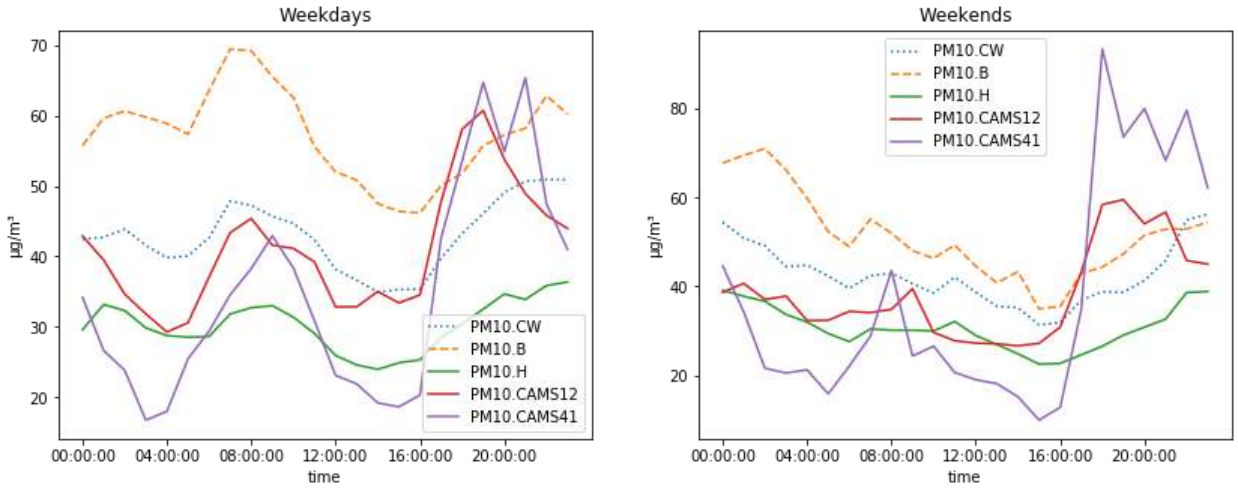


Figure 4.7- Diurnal averages of PM_{10} for measured sites and CAMS during the study period.

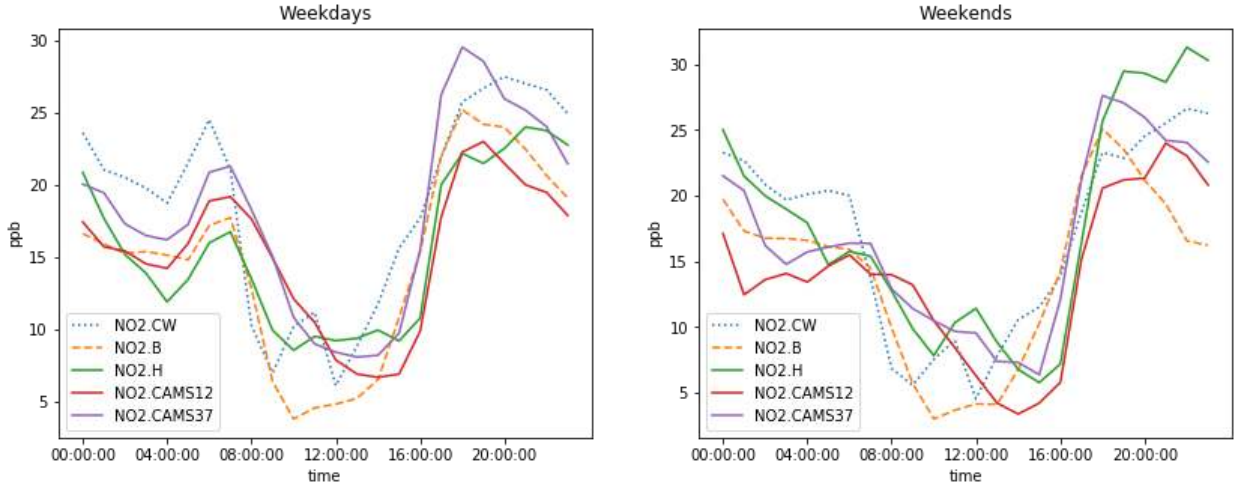


Figure 4.8- Diurnal averages of NO_2 for measured sites and CAMS during the study period.

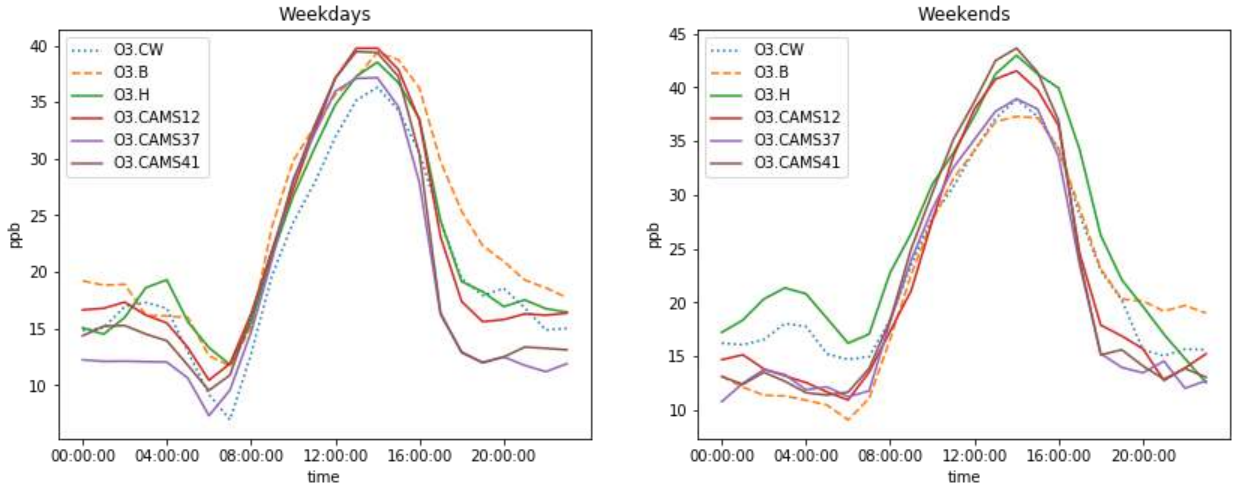


Figure 4.9- Diurnal averages of ozone for measured sites and CAMS during the study period.

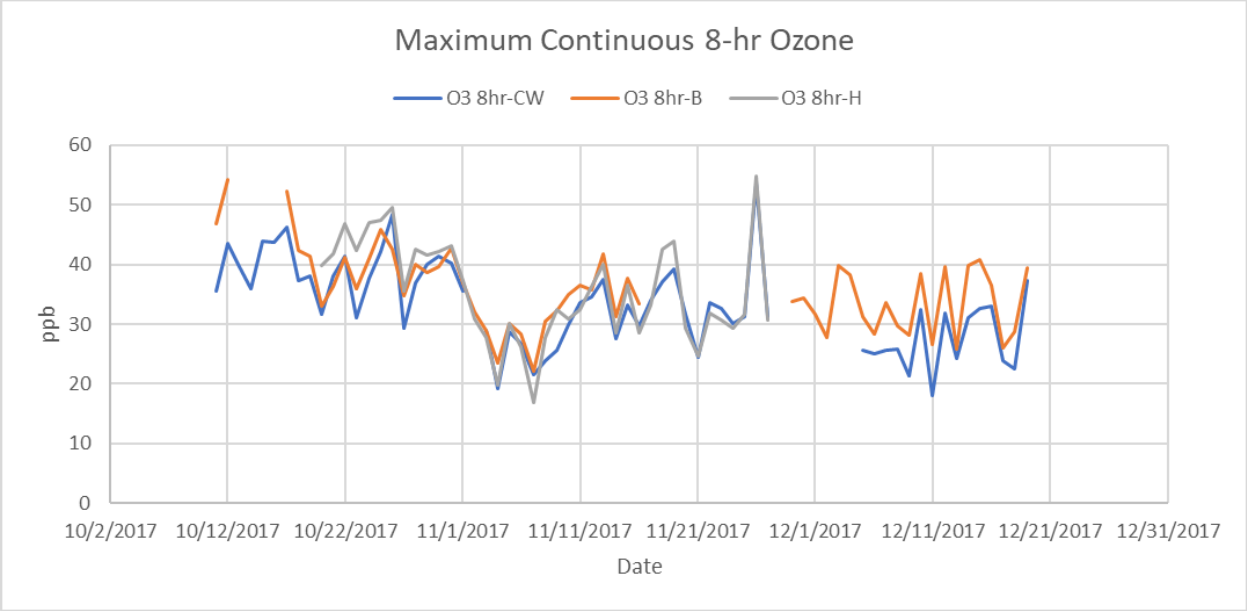


Figure 4.10.- Maximum daily continuous 8-hr O_3 average for CW (Coldwell Elementary), B (Bliss Elementary), and H (residential house) during the study period.

4.6. Comparison in Pollutant Concentrations between the Three Sites

The three measured sites revealed similar trends with one another. Coldwell Elementary and the residential house are located on opposite sides of the highway. The differences in local street traffic is a probable reason for the differences in air pollution levels. The pollutant data extracted from the nearest TCEQ CAMS for comparison revealed varying trends, apart for ozone.

Comparing the results from the study sites with literature reveals interesting observations. The average $PM_{2.5}$ concentrations for Bliss Elementary is consistent with that of past studies in the Paso del Norte air basin. The average PM_{10} concentrations at both schools were exceptionally high compared to other studies outside of the PdN region. The average NO_2 concentrations for Bliss Elementary is comparable to past studies in the PdN region, but Coldwell Elementary agrees more to studies outside of this region.

Table 4.6- Air quality comparison with on-site studies

Reference	Location	Pollutant	Average Concentration	Site Similarity
Keeler et al., 2002	Two Elementary schools in Detroit, Michigan	$PM_{2.5}$	20.6 μg/m^3	Bliss (17.81 μg/m^3)
		PM_{10}	30.8 μg/m^3	House (30.37 μg/m^3)
Singer et al., 2004	Ten Elementary schools in California	NO_2	19-30 ppb	Coldwell (18.37 ppb)
Kim et al., 2004	Ten schools in San Francisco Bay Area	$PM_{2.5}$	11-15 μg/m^3	Coldwell (11.6 μg/m^3), Bliss (17.81 μg/m^3)
		PM_{10}	29-32 μg/m^3	House (30.37 μg/m^3)
		NO_2	19-31 ppb	Coldwell (18.37 ppb)
Annesi-Maesano et al., 2007	Schoolyards in French cities	$PM_{2.5}$	20.7 μg/ m^3	Bliss (17.81 μg/m^3)
		NO_2	46.4 ppb	-
Kim et al., 2016	Seven elementary schools in South Korea	PM_{10}	24-45 μg/m^3	Coldwell (42.85 μg/m^3), House (30.37 μg/m^3)
		NO_2	11-48 ppb	ALL
		O_3	2-35 ppb	ALL
Peacock et el., 2003	Three schools in England	PM_{10}	18.4-22.7 μg/m^3	-
		NO_2	17.1-19.2 ppb	Coldwell (18.37 ppb)
		O_3	19-21.6 ppb (8-hr avg)	-
Gonzales et al., 2005	El Paso, TX	NO_2	11-13 ppb	Bliss (14.94 ppb)

Holguin et al., 2007	Ciudad Juarez, Chihuahua	$PM_{2.5}$	17.5 μg/m^3	Bliss (17.81 μg/m^3)
		NO_2	18.2 ppb	Coldwell (18.37 ppb)
Raysoni et al., 2011	El Paso & Ciudad Juarez	$PM_{2.5}$	14.5 μg/m^3	Coldwell (11.6 μg/m^3), Bliss (17.81 μg/m^3)
		PM_{10}	39 μg/m^3	Coldwell (42.85 μg/m^3),
		NO_2	14.2 ppb	Bliss (14.94 ppb)
Raysoni et al., 2013	El Paso, TX	$PM_{2.5}$	13-14 μg/m^3	Coldwell (11.6 μg/m3), Bliss (17.81 μg/m^3)
		PM_{10}	35 μg/m^3	House (30.37 μg/m^3)
		NO_2	9.47-10.69 ppb	-

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

5.1. Conclusions

This study demonstrates the use of on-site air quality monitors to characterize air pollutants in near-road schools to understand children's exposure to traffic-related air pollutants. Ambient air monitoring stations were installed at selected schools, ambient air quality data for $PM_{2.5}$, PM_{10}, NO_2, and ozone was collected, and comparisons between various CAMS throughout El Paso were produced to characterize air pollution in the surrounding schools and communities.

All monitors recorded similar trends per measured pollutant across all examined sites. The unexpected higher concentrations of PM_{10} at the Bliss Elementary is of concern. Considering that PM_{10} is characterized by natural sources, it is plausible that an increase of unpaved roads or dust from the Franklin Mountains could increase the concentration of PM_{10} around the selected sites. The three monitored sites exhibited strong Spearman correlations for all pollutants, especially among particulate pollutants. In general, correlations were lower at Bliss Elementary. Correlations between sites were moderate ($\geq$0.6) for NO_2. Coefficients of divergence helped assess the spatial variability across the measured sites and CAMS. At 1-hour time resolution, moderate to high spatial heterogeneity can be implied for the three measured sites to CAMS for all measured pollutants. At 24-hour time averages, O_3 and NO_2 between most sites is assumed to be homogeneous. However, heterogeneity in particulate matter is observed at both time resolutions. Bliss Elementary showed the highest COD values for particulate matter; implying greater heterogeneity between this site and the rest of El Paso.

Investigating the association between children's exposure and traffic and meteorological variables is challenging due to the numerous variables involved. Spearman correlations, coefficient of divergence, and diurnal graphs do not completely elucidate the differences in the pollutant levels between sites. The results suggest a spatial and temporal variation between the sites examined and available CAMS sites.

5.2. Recommendations

The study demonstrated that CAMS sites may not accurately reflect pollutant concentrations in neighborhoods with diverse traffic levels, meteorology, and emission sources. It is recommended that studies performed in El Paso employ on-site measurements to avoid exposure misclassification and erroneous estimations from using distant CAMS sites.

Average pollutant concentrations during the study period may not be an accurate representation for the year. In El Paso, higher ozone concentrations occur during the summer months, and higher PM and NO_2 is common during the winter season. Air pollution is sometimes generalized as one thing. Along with raising awareness of near-road air pollution, communities should be informed that there exist different types of air pollutants. Each different type of air pollutant affects the population differently, and exist at varying concentrations per community. Knowledge of individual responses to a particular pollutant and understanding of the air pollution characteristics of the region can mitigate the health concerns by helping individuals avoid areas detrimental to their health.

5.3. Future Work

Wind varies throughout the region due primarily to the Franklin Mountains. Due to the distance to CAMS sites, meteorology from CAMS sites was not used to characterize our study sites, so wind was not able to be used as a variable in this study. Future work could employ on-site meteorology stations to accurately measure meteorological data. Traffic related air pollutants are not limited to the air pollutants measured in this study. Measuring other pollutants such as carbon monoxide would aid in determining the influence of traffic related air pollutants in near-road communities. Finally, collecting traffic data would provide further insight to the quantity of air pollution that is attributed to traffic.

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