ASSESSING CHILDREN'S SPATIOTEMPORAL EXPOSURES TO TRANSPORTATION POLLUTANTS IN NEAR-ROAD COMMUNITIES

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Final Report

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16. Abstract

Traffic-related air pollution has a profound impact on human health especially for residents living in near-road communities which are constantly exposed these air pollutants. A near-road community is expected to observe significant spatial and temporal variations in pollutant concentrations, as air pollution resulting from emissions from major highways decreases rapidly from the highway. This research conducted on-site traffic and air quality measurements on four critical transportations related air pollutants, $PM_{2.5}$, PM_{10}, NO_2, O_3, as well as emission and air dispersion modeling of transportation emission impacts in a near-road community. Using numerical models provided by the EPA, integrated with field measurements of both traffic and air quality, this research developed spatial and temporal pollutant concentration variation patterns in a near-road community using MOVES and AERMOD, EPA emissions and dispersion models. It was observed that modeled-to-monitored comparisons show that air quality impact in near-road communities resulting from traffic-related emissions are dominated by regional background concentrations. Additionally, the AERMOD predictions rendered highest concentration estimates at locations where the traffic volume is the highest and downwind of the prevailing winds. However, impacts of the traffic emissions on the air quality subside rapidly with increasing distance away from the highway, at around 200 meters. This research also apportioned the differences in exposure concentrations to background concentrations and those contributed from major highways. In the near-road community studied, traffic emissions from the highway were 4.8 times higher than the contributions made by local arterial roads. For better transportation air quality impact assessments, higher quality traffic data such as time-specific traffic volume and fleet information as well as meteorological data such as site-specific surface meteorological could help yield more accurate concentration predictions.

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Chapter 1: Introduction

1.1 Background and Motivation

Traffic-related air pollution has a profound impact on human health especially for communities located in close proximity to highways. Transportation sources are the dominant source of various pollutants such as particulate matter, nitrogen oxides, carbon monoxide, hydrocarbons, oxides of sulfur and lead. Many of these emissions also contribute to the formation of secondary pollutants such as ozone (O_3) and secondary particulate matter (Abu-Allaban et al. 2007). In cities where high levels of human activities are around transportation corridors, there exists a high incidence of health problems in the community (Sharma, Massey, and Taneja 2009; Cyrys et al. 2003). Long-term exposure experienced by the near-road population has been shown to produce various adverse health issues (HEI 2010; Baldauf et al. 2008). Numerous epidemiologic studies have shown an association between ambient air particulates and increased illness and mortality (Du et al. 2016). Exposure to traffic-related air pollutants near highways is associated with adverse health effects including cardiopulmonary disease, asthma and reduced lung function (Brugge, Durant, and Rioux 2007; Janssen et al. 2001; Gauderman et al. 2007; McConnell et al. 2010; Krzyżanowski, Kuna-Dibbert, and Schneider 2005). These conclusions have motivated research to understand and quantify the types and amounts of pollutants in near-highway environments.

Epidemiologic work conducted over several years has suggested that long-term residence in communities with elevated ambient levels of air pollution from combustion sources is associated with increased mortality. Exposure to ambient Nitrogen Dioxide (NO_2) may increase the risk of respiratory tract infections through the pollutant's interaction with the immune system (Chen et al. 2007b). Ground-level ozone (O_3) has been shown to cause decreased lung function and has been associated with other important respiratory health effects (Chen et al. 2007a). A special report by

the Health Effects Institute concluded that exposure to particulate matter (PM) leads to respiratory and cardio-vascular diseases (Lin et al. 2002).

As one the six criteria pollutants, the Clean Air Act requires the U.S. Environmental Protection Agency (EPA) to set national air quality standards for PM (Mccarthy, Parker, and Schierow 2011). In 2006, the EPA published a final ruling requiring transportation conformity analysis of project-level PM for projects of air quality concern in nonattainment areas. The EPA developed the "Transportation Conformity Guidance for Quantitative Hot-spot Analyses in $PM_{2.5}$ and PM_{10} Nonattainment and Maintenance Areas" to describe transportation conformity requirements for hot-spot analyses, and provide technical guidance on estimating project emissions with EPA's emissions and dispersion models such as the MOVES model and AERMOD, among others (U.S. EPA 2010b).

PM can be generally classified into two groups, coarser particles with sizes ranging up to 10 μm (PM_{10}) and finer particles with sizes up to 2.5 μm ($PM_{2.5}$). PM_{10} is mainly created from industrial sources, windblown soil and dust, vehicle brake and wire wear. $PM_{2.5}$ is mainly created from vehicle combustion, burning plants, smelting and processing metals (Almeida et al. 2006; Chow et al. 1996). $PM_{2.5}$ is often found to be of higher detriment to human health, as this size particle can travel further into the respiratory system and $PM_{2.5}$ exposure continues to pose a significant risk to public health (Fann et al. 2012).

These traffic-related air pollution problems are compounded in the Paso del Norte (PdN) border region, which comprises the cities of El Paso, Texas, Ciudad Juarez, Chihuahua, and Sunland Park, New Mexico where there is rapid economic growth, and a substantial number of people living in close vicinity of major roadways (Raysoni et al. 2011; Zora et al. 2013; Raysoni et al. 2017; Li et al. 2001). The rapidly worsening air quality seen in populations along the U.S.–

Mexico border is partly due to high rates of urbanization and industrial development (Pennington et al. 2004). The problems persist in El Paso due to the topographic situation. Prior studies have documented the adverse health effects of traffic-related air pollution on humans (Carlsten et al. 2008). Various studies have suggested that exposure to traffic-related air pollution may be associated with increased risk of asthma and other reduced lung function ailments in schoolchildren (Janssen et al. 2001; Branco et al. 2014; H. H. Kim et al. 2016).

One major source of air pollution in urban areas is traffic. Studies have shown the association between exposure to air pollution and adverse health effects on humans, especially children (Hasunuma et al. 2018; Sarnat et al. 2012). Various studies have been conducted to quantify the emissions from mobile sources in urban areas near highways (Farrell et al. 2016; Karner, Eisinger, and Niemeier 2010; Patton et al. 2014; Zavala et al. 2006). Concerns for the health of populations exposed to traffic-related emissions of particles and gases have led the U.S EPA to establish a near-road ambient monitoring program, carried out by the Texas Commission on Environmental Quality (TCEQ) as part of their Annual Monitoring Network Plan (AMNP) since 2014 (TCEQ 2018). Determining the effect of the emissions from vehicles on air quality of the near-road communities is important as vehicle emissions are a main contributor to urban air pollution. Despite all the field studies and experimental findings, health researchers are in need of improved assessment of exposure to the vehicle emissions to better quantify the health impacts on the community and to support more definitive findings about causality (Adar and Kaufman 2007; HEI 2010). Because of the adverse effects of traffic-related air pollution on human health, various policies have been implemented to monitor worsening air quality. With the AMNP, the EPA encourages states to measure the criteria pollutants, meteorology, and traffic volume. Currently, there are six near-road monitoring stations in Texas, all located in major urban areas. These

monitoring sites record data on ambient air concentration of select pollutants and meteorological conditions.

Along with incorporating air quality effects into transportation planning, there has also been an increase in integrating health considerations into transportation planning and policy-making. In recent years, the Federal Highway Administration (FHWA) has been examining how MPOs in the U.S. can effectively combine the health outcome analyses with their transportation analyses in order to create healthy communities (Schreffler et al. 2012). For these kinds of transportation polices to succeed, it is necessary to accurately estimate emissions and pollutant concentrations to include air quality and public health considerations.

1.2 Research Objectives

It is critical to accurately capture the distribution and impact of these pollutants on air quality and human health at a finer resolution, as a coarser resolution analysis will be unsuccessful in capturing the temporal and spatial variations at a local scale near these critical roadways. This study will first address two assumptions. The first is that urban near-road communities are exposed primarily to regional background air pollution and traffic emissions in the communities while the contribution of the traffic emissions to the total exposure concentrations is of limited fraction. The second is that only near-road receptors are affected by the traffic emissions from major highways while spatial and temporal variations of pollutant concentrations in near-road communities are dominated by local traffic.

Using numerical models provided by the EPA, integrated with field measurements of both traffic and air quality, one objective of this study is to develop spatial and temporal pollutant concentration variation patterns in a near-road community. The modeling framework begins with the travel demand model, used to estimate traffic volumes in the area. Combined with field measurements of traffic volumes, factors related to vehicle fleet information, roadway

characteristics, and fuel and weather conditions, this information is used to provide emissions factors estimates for the roadways in the study area. A dispersion model is then used to calculate the dispersion of these emissions in the atmosphere based on fate and transport properties of the pollutants, meteorological conditions, and land use characteristics. A second objective of this study is to apportion the differences in exposure concentrations to background concentrations and that contributed from major highways. This includes and analysis of air quality estimates considering emissions resulting solely from the major highway and those from the arterial roads confined in the study area.

1.3 Significance of Research

This study assesses traffic-related emissions and dispersions at a micro-scale level using higher spatial and temporal resolution at an hourly level, providing further clarity to the temporal and spatial variation of these pollutants in urban areas. This analysis also provides further insight on the correlations and accuracy between modeled estimates and field measurements, both provided using the most up-to-date research methods. The assessments provided by this study can be used to create the relevant policy considerations in future transportation and urban planning projects. The move towards combining higher temporal resolution of pollutant dispersion will also contribute to more accurate health outcome studies which can provide a better representation of the associations between air pollution and the health of the communities affected.

Chapter 2: Background Literature Review

2.1 Introduction

This chapter provides an overview of literature pertaining to the models used in the pollutant dispersion analysis, followed by a literature review on previous near-road exposure studies. While Section 2.2 provides an overview of the transportation models used to provide data for this study, only the results from such models were used in this study. Section 2.3 discusses the history of the emissions models leading to the latest version used in the study, Section 2.4 gives and overview of the air dispersion models available from the EPA, and Section 2.5 provides a literature review of previous studies related to air pollution in communities located near major highways.

2.2 Transportation Planning Models used in Study

Transportation planning models are used to forecast the future travel demand for the transportation infrastructure. These models combine information on current conditions of traffic, economic growth, population, and land to predict the travel demand for the existing situation. Based on future information on population and land use, the model predicts travel demand for future conditions. Traditional travel demand models are based on a four-step methodology of trip generation, trip distribution, mode choice and trip assignment. Trip generation stage produces the total number of trips generated from each zone in the study domain based on socioeconomic characteristics of people and households. Linear regression, cross-classification and trip rate models are the three major approaches to calculate trip generation rates expressed as a function of one or more explanatory variables based on socioeconomic characteristics of people.

2.3 Emission Models

Studies on exposure to near-road communities must begin with correct and adequate assessments of the levels of air pollution emitted for the area. Emission estimation is typically conducted through emission models which provide link-based emission rates or total emission inventory. A number of emission models were developed over the past decades to estimate emissions and energy consumption from mobile sources. Typically, all these models take into account the various factors affecting emissions, although they differ in their modeling approach, modeling structure, and in the data used to develop them (Grote et al. 2018). The following sections discuss the two main mobile source emissions models, MOBILE and MOVES, followed by an in-depth history of MOVES, and a detailed review of the modeling process using the MOVES model.

2.3.1 Overview of Mobile Source Emission Models

Emission rates required for air dispersion modeling are obtained through the use of mobile source emission models. The development of these models was due to the Clean Air Act, which requires the EPA to regularly update its mobile source emission models (Mccarthy, Parker, and Schierow 2011). EPA continuously collects data and measures vehicle emissions to make sure the best possible understanding of mobile source emissions is obtained.

The development of these models began with the MOBILE model, first developed as MOBILE1 in the 1970s. This model been intermittently updated with more accurate data, changes in technologies, changes in regulations and standards, and general improved understanding of emission levels and the factors that affect them (CRC 2004). MOBILE calculates various pollutant emissions from passenger cars, motorcycles, light- and heavy-duty trucks; these include hydrocarbons (HC), oxides of nitrogen (NOx) and carbon monoxide (CO). MOBILE is based on emissions testing of tens of thousands of vehicles. The model accounts for the emission impacts

of factors such as changes in vehicle emission standards, changes in vehicle populations and activity, and variation in local conditions such as temperature, humidity and fuel quality.

The newest model, EPA's MOtor Vehicle Emission Simulator (MOVES) replaces EPA's previous mobile source emissions model MOBILE (U.S. EPA 2015a). MOVES contains a significant expansion of capabilities compared to MOBILE. MOVES is an emission modeling system that estimates total emissions and energy use from all on-road sources including cars, trucks, buses, and motorcycles. These emissions can be measured at the national, county, and project level for criteria pollutants, greenhouse gases, and air toxics.

Additionally, since MOVES' debut in 2010, there have been several improvements to the model. MOVES2014 is a major new revision to EPA's mobile source emission model and it replaces MOVES2010 and its minor revisions (MOVES2010a and MOVES2010b). MOVES2014a, released in December 2015, is the latest version of MOVES. It incorporates significant improvements in calculating on-road and non-road equipment emissions. MOVES2014a does not significantly change the criteria pollutant emissions results of MOVES2014 and therefore is not considered a new model for SIP and transportation conformity purposes (U.S. EPA 2015b). However, MOVES2014a was used in this research because of its updated defaults and improvements in calculating emissions.

2.3.2 Comparing MOVES to MOBILE

The input structure MOVES provides is more flexible than its predecessor is. It includes a graphical user interface (GUI), while MOBILE required text input and output files. MOVES uses MySQL software and Java operating in Windows rather than MOBILE FORTRAN software and operating in DOS. MOVES has a relational database structure to store data in tables that allows updates without requiring changes to the model code (Vallamsundar and Lin 2012).

In terms of outputs, MOVES provides an estimate on a total emission inventory as well as emission rates, supplanting the need for extensive external post-processing. The output is also easily customizable with varying levels of aggregation and disaggregation.

The temporal and geographical reach of MOVES far exceeds the capabilities of MOBILE. MOVES can provide emission estimates at national, county, and project level, rather than MOBILE's regional scale with no geographical specificity. MOVES can also generate estimates by hour, weekday, weekend, month or year. MOVES emissions are based on "operating modes" such as acceleration, cruising, and deceleration as well as average speed, but MOBILE is only based on aggregate driving cycles accounting only for differences in average speed.

MOVES includes the ability to estimate emissions of criteria pollutants, greenhouse gases and air toxics, while MOBILE only calculates emissions of hydrocarbons, oxides of nitrogen and carbon monoxide from passenger cars, motorcycles, light and heavy duty trucks (Sturtz et al. 2014).

MOVES consist of a larger data set including in-use data on light duty vehicles, PM data for light duty vehicles with temperature effects, data for heavy-duty vehicles including speed effects and crankcase, start, and extended idle emissions (Fujita 2001). MOBILE used certification data rather than in-use and did not provide for various speed and temperature effects (Granell and Street 2004). MOVES adopts a much more sophisticated, modal-based estimation procedure than the simplistic fuel economy approach in MOBILE for computing transportation energy consumption and Green House Gas (GHG) emissions (Vallamsundar 2012). MOVES was therefore used as the model chosen to calculate the emission rates for this study.

2.3.3 Previous MOVES Versions

MOVES is used as a post-processor to determine the air quality impact of vehicle emissions. MOVES2014a is the latest version of the processor, preceded by MOVES2004, MOVES-HVI (released in 2007), MOVES2010, MOVES2010a, and MOVES2010b.

MOVES2004, released in 2004, was the first installment of the new generation of mobile source modeling framework that could be used to estimate and project national inventories at the county level for nitrous oxide, methane and carbon dioxide from highway vehicles.

MOVES-HVI, released in 2007 was a demonstration version of MOVES that is the Highway Vehicle Implementation of EPA's model. This version's only added features were to estimate criteria pollutant emissions such as gaseous hydrocarbons, carbon monoxide, oxides of nitrogen and particulate matter from highway vehicles, but results were not to be considered realistically (Bai, Eisinger, and Niemeier 2008).

A draft version of MOVES was released in 2009 to the public mainly for users' review and comments and was not intended for official use. The first emissions model was designed to work with databases to accommodate for newly available data. The model also included a "default" database that summarized emission relevant information for the United States. This data comes from EPA research studies, Census Bureau vehicle surveys, Federal Highway Administration travel data, and other federal, state local, industry and academic sources. A finalized version was released in December 2009 as MOVES2010 (U.S. EPA 2010a). Previous versions of official MOVES include MOVES2010, MOVES2010a, and MOVES2010b. MOVES2010 was the first of the EPA's processors for estimating emissions from highway vehicles.

MOVES2010a, released in August 2010, is a minor revision to MOVES2010. This version allows users to account for emissions under new car and light truck energy and greenhouse gas

standards affecting model years for 2012 and later and updates effects. MOVES2010b includes corrections to database as well as several improvements to network operations.

MOVES2014 is the first major revision to the MOVES series since the original release of MOVES2010. MOVES2014 incorporates new emissions test data, the impacts of new emissions standards, new features, and other functional improvements, all of which contribute to improved estimates of criteria pollutant emissions compared to MOVES2010 (U.S. EPA 2015b).

MOVES2014 allows users to benefit from new regulations promulgated since the release of MOVES2010b and incorporates new and up-to-date emissions data, and has improved functionality compared to MOVES2010b. MOVES2014 also has added the capability to model non-highway mobile sources by incorporating EPA's NONROAD2008 model (U.S. EPA 2014).

2.3.4 Review of MOVES2014a

MOVES2014a is a computer model designed by the EPA to estimate emissions from cars, trucks, buses and motorcycles. This model can be used to estimate emissions from transportation projects that include roadways intersections, highways, transit projects and parking lots. MOVES is designed to allow for the estimation of motor vehicle emissions at multiple scales, from national to county to project-level, using different levels of input data. Additionally, the model can be used to complete project-level hot-spot analyses for transportation conformity determinations, modeling project-level emissions for state implementation plans, and completing environmental assessments and environmental impact statements as required by the National Environmental Policy Act (NEPA).

There are several decisions to be made before conducting a project level analysis as required by this research. A general overview of the EPA's guidance manuals, "Transportation Conformity Guidance for Quantitative Hot-spot Analyses in $PM_{2.5}$ and PM_{10} Nonattainment and Maintenance Areas" and "MOVES2014a User Guide" is intended to help evaluating and choosing

models and the associated methods and assumptions before conducting the analysis (U.S. EPA 2015b). The following sections describe the inputs necessary to conduct the emission factor generation needed for dispersion analysis.

2.3.4.1 RunSpec Parameters

This section describes the inputs necessary for the three different types of analysis. In order to process the RunSpec a description must be entered as well as a selection of the scale of the analysis. A time frame must be selected for the analysis to include the year, month, day, and hour. At the project level, each MOVES run represents one specific hour. The user may select either "weekday" or "weekend" but for most analytical purposes "weekday" is the appropriate choice. The project scale also allows the user to define the specific a single county where the project takes place. The user is able to specify the vehicle types that are included in the run, of which there are 13 "source use types" to select from. In addition to the vehicle type, the user must identify fuel/source type combinations. Fuel types Gasoline, diesel, ethanol, and compressed natural gas should always be selected. MOVES includes five different road types users can choose which include rural restricted access, rural unrestricted access, urban restricted access, urban unrestricted access, and off-network. MOVES utilizes the road types to determine the default drive cycle on a particular link. Pollutants and processes are chosen at the same time due to some pollutants/processes being chained and calculated as ratios to others. Finally, output details must be selected to specify the level of detail desired in the output data.

The MOVES model allows for three different levels of analysis. Using the national scale analysis, the model can be used to model the entire country, one or more states, or one or more counties. This scale allows the user to use the information in the MOVES default database, but still provides the option to input local data and override the default data.

The county scale analysis can be used to model an individual county or a Custom Domain made up of several counties. This scale is required for use in State Implementation Plans and conformity analyses. The user must enter county-specific data for the input database. While there is some access to default data, local data is necessary for most inputs. The project scale analysis provides link level modeling of specific transportation projects including highways, intersections, interchanges, transit projects and parking lots. The user must enter project-specific data for the input database. For each of the three levels of analysis, a RunSpec must be created. The RunSpec specifies the scale, location, time period, alternate data, and output preference of the MOVES run. A description panel allows for the inclusion of details in the form of text. The scale panel indicates the scale of the analysis. Calculation type can be either Inventory or Emission Rates. Using both can give equivalent results but post-processing errors are more common if using emission rates calculation type.

Time spans panel allows a time aggregation to be chosen from year, month, day, or hour. National level allows for choosing multiple years, months, days, and hours. County level runs can choose all hours and months but only a single year. Project level allows for choosing only one year, one month, one hour, and either weekend days or weekdays.

The geographic bounds panel allows the user to choose the county in which the analysis is in; this accesses the available default data stored for that county. The Vehicles/Equipment panel defines the types of vehicles to be analyzed. For most analyses, all valid gasoline, diesel, ethanol and CNG vehicle combinations are used. Table 1 displays the MOVES Source Types and HPMS Vehicle Types. The user can indicate which road type to include in the analysis. Table 2 provides descriptions for the available road types.

Table 1 MOVES Vehicle Source Types

sourceTypeID	sourceTypeName	HPMSVtypeID	HPMSVtypeName
11	Motorcycle	10	Motorcycles
21	Passenger Car	25	Light Duty Vehicles
31	Passenger Truck	25	Light Duty Vehicles
32	Light Commercial Truck	25	Light Duty Vehicles
41	Intercity Bus	40	Buses
42	Transit Bus	40	Buses
43	School Bus	40	Buses
51	Refuse Truck	50	Single Unit Trucks
52	Single Unit Short-haul Truck	50	Single Unit Trucks
53	Single Unit Long-haul Truck	50	Single Unit Trucks
54	Motor Home	50	Single Unit Trucks
61	Combination Short-haul Truck	60	Combination Trucks
62	Combination Long-haul Truck	60	Combination Trucks

Table 2 MOVES Road Types

Road Type	Description
Off-Network	Captures emissions that occur while vehicles are not moving, i.e., start, extended idle (hoteling of long haul combination trucks), and resting evaporative emissions. Idle emissions that occur during normal running operation, such as at signalized intersections, is captured in the other road types.
Rural Restricted Access	Captures running emissions, including running evaporative emissions. Restricted indicates restricted vehicle access via ramps, such as freeways and interstates.
Rural Unrestricted Access	Captures running emissions, including running evaporative. Un-Restricted indicates all other rural roads not included in Restricted.
Urban Restricted Access	Captures running emissions, including running evaporative. Restricted indicates restricted vehicle access via ramps, such as freeways and interstates.
Urban Unrestricted Access	Captures running emissions, including running evaporative. Un-Restricted indicates all other urban roads not included in Restricted.

The pollutants and processes panel allows the choosing of the pollutant and process combinations required for the analysis. Some pollutants/processes are chained and are calculated as ratios to others. MOVES calculates emissions of criteria pollutants, greenhouse gases, and selected air toxics associated with motor vehicle operation. MOVES also calculates energy consumption for onroad and fuel consumption in terms of mass fuel per day (i.e., grams fuel per

day) for nonroad. For many pollutants, the emissions calculation is based on the prior calculation of another pollutant emission. The Pollutant/Process will display an error message if the user selects a dependent pollutant but not the base pollutant. In MOVES2014, the option to automatically select all prerequisite pollutants is available. There are fewer pollutants available for nonroad equipment, but the prerequisites are the same as for onroad and all of the buttons in this window operate identically for nonroad.

In MOVES, Processes refers to the mechanism by which emissions are created. Engine operation creates Running Emissions Exhaust, Start Emissions Exhaust (the addition to running emissions caused by the engine start), and Extended Idle Emissions Exhaust (i.e., hotelling emissions from a combination, long-haul truck). MOVES Onroad emission processes also distinguish Crankcase Running Exhaust, Crankcase Start Exhaust, and Crankcase Extended Idle Exhaust to describe the exhaust gases that escape around the piston rings and enter the crankcase during normal operation. For nonroad equipment, start and running emissions are both included in "Running Exhaust." The Crankcase Running process is available in nonroad but only for the total hydrocarbon pollutant. Evaporative emissions occur when unburned fuel escapes the vehicle's fuel system. For onroad vehicles, MOVES models these emissions through the following processes: Evaporative Fuel Vapor Venting, Evaporative Permeation, Evaporative Fuel Leaks, Refueling Displacement Vapor Loss and Liquid Spillage Loss.

For nonroad equipment, MOVES models evaporative emissions separately by the following processes: Crankcase Running Exhaust (which is actually Evaporative, not Exhaust), Refueling Displacement Vapor Loss, Refueling Spillage Loss, Evap Tank Permeation, Evap Hose Permeation, Diurnal Fuel Vapor Venting, Hot Soak Fuel Vapor Venting, and Running Loss Fuel Vapor Venting.

For Onroad vehicles only, Brakewear and Tirewear describe the non-exhaust particulate emissions that result from brake use and tire wear.

In general, the MOVES data importers, such as the Data Importer, Nonroad Data Importer, County Domain Manager, and the Project Data Manager, should be used to enter data rather than the Manage Input Data Sets panel. It is highly recommended to use the MOVES data importers and managers because they provide advantages such as checking the data for errors, creating input templates, and exporting default data filtered to be consistent with other RunSpec settings. However, MOVES allows the user to select Manage Input Data Sets on the Navigation Panel to specify specialized user-supplied data to be read by the model during execution.

Output databases allow the user to choose what output data to be displayed and calculated for units, activity, and output emission details. The units available for the mass are kilograms, grams, pounds, or U.S. tons. Available energy units are joules, kilojoules, or million BTUs (British Thermal Units). The available distance units are miles or kilometers. Only one choice can be made for each unit. The activity that can be displayed in outputs includes distance traveled, source hours, hoteling hours, source hours operating, source hours parked, population, and starts.

2.3.4.2 Data Manager Inputs

Data is entered using the County Data Manager (CDM) or the Project Data Manager (PDM). Setting the descriptions for the RunSpec first allows the data manager to filter default data for relevant information. The data manager also conducts error-checks on the user imported data to make sure there are no conflicts with description entered in initial RunSpec.

The meteorology data importer allows the user to import temperature and humidity data for months, zones counties, and hours that are included in the RunSpec. The MOVES default database contains 10-year average temperature and humidity data for the period from 2001 to 2011 for each county, month, and hour.

The importer also allows for the specification of Source Type Population by inputting the number of onroad vehicles for each source type in the geographic area.

The user can also enter data that provides the distribution of vehicle counts by age for each calendar year and vehicle type as a fraction adding to one for each vehicle type and year.

The vehicle type VMT importer is used to enter vehicle miles traveled data and VMT time allocation fractions into MOVES. VMT may be entered by HPMS typed according to the Federal Highway administration or by MOVES source types as annual or daily VMT.

The user can input average speed data specific to vehicle type, road type, and time of day. MOVES defines 16 speed bins, which describe the average driving speed on a road type or link. The fraction of driving time in each speed bin for each hour/day type, vehicle type, road type, and average speed, must be entered, where the fractions sum to one for each combination of vehicle type, road type, and hour/day type specified in the RunSpec.

The ramp fraction allows the user to modify the fraction of time driving on ramps on selected road types.

The fuel tab of the importer includes four different aspects of fuel data that can be specified. The fuel formulation property allows the selection of an existing fuel in the MOVES database and the option to change its properties, or create a new fuel formulation with different fuel properties. Fuel supply assigns existing fuels to fuel regions, months and years and an associated market share for each fuel. Fuel usage refers to the fraction of E-85 capable vehicles using E-85 compared to conventional gasoline. The Alternative Vehicle and Fuel Technologies allows to specify the mix of fuel types in the model, specifically the fleet distribution fraction by fuel type, source type, model year, and engine technology.

The hotelling importer is used to import information on combination truck hotelling activity. In MOVES2014, hotelling can be divided into three operating modes: Extended Idle, Diesel Auxiliary power (APU), and APU-Off. Extended Idle is defined as long-duration idling with more load than standard idle and a different idle speed. It is used to account for emissions during hotelling operation when a truck's engine is used to support loads such as heaters, air conditioners, microwave ovens, etc. Diesel Auxiliary power refers to use of auxiliary power units that allow for heating/cooling/power for the cab without running the truck's engine. APU-Off refers to hotelling when the truck's engine is off and an APU is not being used. This could include hotelling resulting from truck-stop electrification. All hotelling processes only apply to long-haul combination trucks.

Specific to the Project Data Manager, the link source types importer is used to enter the fraction of the link traffic volume, which is driven by each source type. It is not used to enter off-network data, and is not required if the Project contains only an off-network link. For each link ID, the source type hour fraction must sum to one across all source types. If you enter data for source types that are not selected in the RunSpec, MOVES will ignore that data. The Project level calculator will not re-normalize the fractions to omit the contribution of source types that are not selected in the RunSpec.

Also specific to the PDM, the operating mode distribution importer allows the import of operating mode fraction data for source types, hour/day combinations, roadway links and pollutant/process combinations that are included in the RunSpec and Project domain. This data is entered as a distribution across operating modes. Operating modes are modes of vehicle activity that have a distinct emission rate. Running activity for light duty vehicles has modes that are distinguished by their Vehicle Specific Power and instantaneous speed. Start activity has modes

that are distinguished by the time the vehicle has been parked prior to the start. The start process has eight operating modes that require data and tire wear has sixteen operating modes. It is optional for modeling 'running emission' processes. However, if chosen, data for all twenty-three running exhaust operating modes must be entered.

The Link Drive Schedules Importer is used only in the PDM. It defines the precise speed and grade as a function of time, in seconds, on a particular roadway link. The time domain is entered in units of seconds, the speed variable in miles per hour and the grade variable in percent grade. This importer is used only when modeling 'running emission' processes when the Link Drive Schedules Importer is used. For a given roadway link, an operating mode distribution input will take calculation precedence over an imported drive schedule. An imported drive schedule will take calculation precedence over an average link speed input when more than one is entered for a given link. However, at least one of three, an operating mode distribution, a link drive schedule or a link average speed, must be entered for each of the defined roadway links.

The off-network importer used in project-level scales provides information about vehicles that are not driving on the project links, but still contribute to the project emissions. For each source type in the RunSpec, vehicle population is the average number of off-network vehicles during the hour being modeled. The start fraction field is a number from zero to 1.0, which specifies the fraction of this population that has a 'start' operation in the given hour.

Finally, the Inspection and Maintenance (I/M) programs importer specifies the level of compliance and general effectiveness of the I/M program design being used. The compliance factor input is a multiplicative factor that encompasses I/M program performance metrics such as waiver rates, exemptions, special training programs and general effectiveness. It can range from 0 percent (a program that has no effectiveness or merit) to 100 percent (highest possible success).

The compliance factor is entered as a function of pollutant-process, location, source type, model year range, fuel type and specific I/M test types.

2.3.4.3 MOVES2014a Outputs

MOVES allows for two types of outputs, emission rates or inventory. Specifying for emissions rates provides output as a set of emission rates per mile or per vehicle. This output can be post-processed by multiplying rates by vehicle activity data to get inventory. MOVES produces three sets of rates: rate per distance, rate per vehicle, and rate per profile. The table of emission rates is further organized by varying temperature, speed, road type, and fuel type. Rates can be applied to multiple counties and multiple days with the same fuels and Inspection and Maintenance Programs. Emission rate output should be used when modeling many counties as well as to model a wide range of temperatures. The user can apply rates on a link basis for a link-based inventory.

The inventory output delivers emissions in units of mass in the form of grams, kilograms, pounds, and tons. MOVES processes results, rates multiplied by activity, to yield total mass of emissions. Inventory output can be used to model a project over a limited time period and when it is necessary to minimize post-processing and avoid calculation errors. The output format can then be converted from "grams per link" to the necessary units in the dispersion modeling process.

2.4 Air Dispersion Models

Air dispersion models are used to determine how pollutants are dispersed in the atmosphere and how their concentrations might dilute over distance as well as time. The main types of atmospheric dispersion modeling can be categorized as follows: Gaussian plume dispersion model, atmospheric box model, Gaussian puff model, and complex numerical models that include diagnostic and prognostic analysis (Hall and Hall 1997). The most commonly used dispersion models are steady-state Gaussian-plume models, which are at the core of most regulatory models. These models operate on the assumptions that plume spread occurs primarily by turbulent

diffusion, and that horizontal and vertical pollutant concentrations in the plume are normally distributed. The pollutant concentrations additionally in this model account for the rate of the plume dispersion, reflections from the ground and the plume rise (Turner 1994). Because of the simplistic description of the dispersion process and the fundamental assumption, this type of model may not accurately reflect reality. The concentration estimates are based on four factors: 1) emission rate, 2) downwind distance in direction x, 3) distance from the plume centerline in the horizontal direction (y), and 4) distance from the plume centerline in the vertical direction (z) (De Nevers 2000). The basic complete Gaussian plume equation is shown in Equation 1 below.

$$C(x,y,z;H) = \frac{Q}{2\pi u\sigma_y\sigma_z} exp\left[-\frac{y^2}{2\sigma_y^2}\right]\left\{exp\left[-\frac{(H-z)^2}{2\sigma_z^2}\right] + exp\left[-\frac{(H+z)^2}{2\sigma_z^2}\right]\right\} \qquad (1)$$

where,
C = Air pollutant concentration in mass per volume (g/m^3)
Q = pollutant emission rate in mass per time (g/s)
u = wind speed at the point of release, (m/s)
σ_y = Standard deviation of the concentration distribution in the horizontal direction at the downwind direction x
σ_z = Standard deviation of the concentration distribution in the vertical direction at the downwind direction x
H = the effective height of the centerline of the pollutant plume

Air dispersion modeling is performed with computer programs that contain the algorithms derived in the type of model being used. There are numerous proprietary or open-domain air dispersion models available in the market for various kinds of purpose. The EPA's Air Quality Modeling Group (AQMG), which is in the EPA's Office of Air and Radiation (OAR), is in charge of directing a full range of air quality models used in assessing control strategies. The EPA's first issue of the Guideline on Air Quality Models in 1978, which has been periodically updated, provided consistency and equivalence in the use of modeling for air quality management. Starting in 1980, regulatory modeling was accomplished with the Industrial Source Complex Model (ISC), which employs in steady-state Gaussian plume model. The updated Industrial Source Complex-

Short Term, Version3 (ISCST3) is the EPA approved and recommended dispersion modeling program that is being used by most state air pollution regulatory agencies. ISCST3 includes a set of Gaussian plume-based models that can be used to predict downwind concentrations from point, line, and area sources.

A similar model developed in the era, CALPUFF, is an advanced non-steady-state meteorological and air quality modeling system developed by the Sigma Research Corporation, sponsored by the California Air Resources Board (CARB). It is a multi-layer, non-steady-state Lagrangian puff dispersion model, modeling dispersion as discrete "puffs" of pollutants emitted from sources (Scire et al. 2000).

The California Line Source Dispersion Model (CALINE) was developed in 1972 in response to the California Clean Air Act and other EPA air dispersion models. This microscale model is used to assess air quality impacts near transportation facilities through analysis of source emissions strength, meteorology, site geometry, and modeling site characteristics (Benson, 1979). The CALINE model series includes its various successors CALINE 3, CAL3QHC and CAL3QHCR.

AERMOD, the American Meteorological Society & Environmental Protection Agency Regulatory Model Improvement Committee Dispersion Model, was developed based on the ISC model following updates to modeling techniques, regarding dispersion in the convective and stable boundary layers (Turner and Schulze 2007). Among other dispersion models, the AERMOD model is considered the most versatile and is widely used by the industries as well as regulatory agencies. AERMOD is the EPA's leading air dispersion model among the other dispersion models which include BLP, CTDMPLUS, and OCD.

The following sections provide an overview of CALPUFF, the CALINE models, and AERMOD. Because this study focuses on the use of the AERMOD model, the more detailed overview is provided for this model followed by a model performance review provided by previous literature.

2.4.1 Review of CALPUFF

CALPUFF is listed by the EPA as an alternate model for assessing long-range transport of pollutants and their impacts and for studies involving complex meteorological conditions (Scire et al. 2000). CALPUFF operates with a preprocessor CALMET, and a post processor CALPOST. CALMET, the first component of this model, develops the hourly wind and temperature fields in a three dimensional modeling domain with diagnostic and prognostic wind field generators, which includes mixing height, surface characteristics, and dispersion properties. The CALPUFF model operates with a Gaussian puff dispersion model, with non-continuous characteristics of the air dispersion plume, which tends to be a more accurate representation of ambient air properties. The model incorporates wet and dry deposition, complex terrain algorithms, and plume fumigation. The model provides four different source types: point, line, volume, and area source using an integrated puff formulation incorporating the effects of partial penetration, buoyant/momentum plume rise, and building downwash effects. CALPOST provides a summary of the hourly concentrations or dourly deposition fluxes at the selected receptor locations.

2.4.2 Review of CALINE Models

CALINE is a line source air quality model developed by the California Department of Transportation (Caltrans). The model is based on the Gaussian diffusion equation and employs a mixing zone concept to characterize pollutant dispersion over the roadway. The benefits of using this model is the relatively minimal input from the user, as the model does not require spatial and temporal arrays of wind direction. With improvements to the original CALINE model, CALINE3

was developed in 1980 by the EPA to be used for non-reactive pollutants near the highway (Eckhoff 1995). Several enhancements were made on CALINE3 model, resulting in CAL3QHC, CAL3QHCR, and CALINE4 models to be developed. These models are collectively known as the CALINE3 series and have been recognized as appropriate for regulatory use in specific roadway applications for CO and PM analyses. CALINE4 is the newest version of the CALINE model series, released in 1984, requires more input parameters but remains one of the less complicated dispersion model. However, it is approved by the EPA for use only in the state of California.

CALINE3 divides individual highway links into a series of elements and sums the incremental concentration from each element. However, it does not permit the direct estimation of the contribution of emissions from idling vehicles (Eckhoff 1995). CAL3QHC enhances CALINE3 by incorporating methods for estimating queue lengths and the contribution of emissions from idling vehicles. The model permits the estimation of total air pollution concentrations from both moving and idling vehicles. CAL3QHCR uses the same basic algorithm as the CAL3QHC model. A major change between the CAL3QHC and CAL3QHCR models includes CAL3QHCR's ability to process up to a year of hourly meteorological data which allow for a yearly analysis of vehicular emissions, traffic volume, and signalization data in one run, whereas CAL3QHC was designed to process one hour of meteorological, emissions, traffic, and signalization data in a single run. The meteorological file for CAL3QHCR must include wind vector (degrees), wind speed (meters/sec), ambient temperature (K), stability class, and mixing heights. These files can be created using available EPA auxiliary meteorological processors and downloaded meteorological data. CAL3QHCR incorporates various concentration-averaging algorithms (1-hour, 8-hour, 24-hour, and annual concentrations), compared with the maximum hourly average algorithm in CAL3QHC. CAL3QHCR has some built-in assumptions, mostly

related to the model application. Wind speed should be at least one meter per second (m/s), and speeds below 1 m/s have not been validated for the model. According to the EPA, AERMOD is the recommended model for dispersion analysis because of the following factors: 1) AERMOD can represent sources in various configurations compared to CALINE models representing all sources as "line sources", 2) AERMOD is able to process a much higher number of receptors and sources simultaneously, 3) AERMOD employs the most current atmospheric science when treating dispersion in the lower atmosphere.

2.4.3 Review of AERMOD

AERMOD, a steady-state dispersion model, was developed as a replacement for the EPA's ISC Model and incorporates the planetary boundary layer (PBL) (Perry et al. 2005). AERMOD addresses improvements on PBL characterizations, plume interaction with terrain, surface releases, building downwash, and urban dispersion. AERMOD includes the effects on dispersion from vertical variations in the PBL. The concentration distribution in the stable boundary layer (SBL) is Gaussian in both the vertical and horizontal orientations. While the horizontal distribution in the convective boundary layer (CBL) is Gaussian, the vertical concentration distribution is described as being a bi-Gaussian probability density function (PDF) (Willis and Deardorff 1981). The model considers the effect of building wakes and augments the vertical turbulence in nighttime urban areas to account for the "convective like" boundary layer conditions (Paine et al. 1998; Cimorelli et al. 2005).

The AERMOD modeling process also involves the use of various pre-processors. There are two input data processors that are regulatory components of the AERMOD modeling system: AERMET, a meteorological data preprocessor that incorporates air dispersion based on planetary boundary layer turbulence structure and scaling concepts, and AERMAP, a terrain data preprocessor that incorporates complex terrain using USGS Digital Elevation Data. Other non-

regulatory components of this system include: AERSCREEN, a screening version of AERMOD; AERSURFACE, a surface characteristics preprocessor, and BPIPPRIM, a multi-building dimensions' program incorporating the GEP technical procedures for PRIME applications.

AERMET arranges and processes the meteorological data and estimates the boundary layer parameters necessary for dispersion calculations in AERMOD. The structure of the PBL is calculated by AERMOD based on surface characteristic such as surface roughness, albedo, and information on surface moisture, which drive the fluxes of heat and momentum in the PBL. AERMET requires inputs on surface characteristics, temperature, cloud cover, a morning upper-air temperature sounding, and wind speed and wind direction. AERMET can then calculate the friction velocity, Monin-Obukhov length, convective velocity scale, temperature scale, mixing height, and surface heat flux (U.S. EPA 2004). AERMET also characterizes the state of the PBL by first estimating the sensible heat flux (H) with an energy balance approach and then calculates the friction velocity (u*) and the Monin-Obukhov length (L); with these variables, the model can estimate the mixing height and the convective velocity scale. Among the surface characteristics calculated by AERMET are the surface roughness, the albedo, and the Bowen ratio. The surface roughness length is related to the height of obstacles to the wind flow and is the height at which the mean horizontal wind speed is zero based on a logarithmic velocity profile. The surface roughness length influences the surface shear stress and is an important factor in determining the magnitude of mechanical turbulence and the stability of the boundary layer. The albedo is the fraction of total incident solar radiation reflected by the surface back to space without absorption. The daytime Bowen ratio, an indicator of surface moisture, is the ratio of sensible heat flux to latent heat flux and, together with albedo and other meteorological observations, is used for

determining planetary boundary layer parameters for convective conditions driven by the surface sensible heat flux (Cimorelli et al. 2005).

AERMOD then uses these parameters and uses the shape of the similar profiles to interpolate between adjacent vertical measurements, which consider the effects from vertical variations in wind, temperature and turbulence (Cimorelli et al. 2005).

2.4.4 Review of AERMOD Model Inputs

Running the AERMOD model requires a "runstream" setup file containing the selected modeling options and parameters, the source locations, receptor locations, meteorological data file specifications, and output options. The modeling options for an analysis using urban sources include population estimates which are used to estimate the urban heat island effect.

The AERMOD model provides pollutant concentration estimates for PM2.5, CO, or NOx. It can predict concentrations using source configurations of point, area, and volume sources (U.S. EPA 2018). Line sources such as roadway links can be modeled as area sources with the roadway length and width, or as multiple volume sources. Input of line sources requires beginning and ending coordinates (meters), elevation (meters), emission rate ($g/s/m^2$), release height (meters), width (meters), initial vertical dimension (meters) and emission factor ($g/s/m^2$) if desired. Volume sources require x and y coordinate (m), representing the center of each source, elevation (meters), emission rate (g/s), release height (meters), initial vertical dimension (meters) and emission factor in terms of g/s. The width of each volume source is necessary to calculate the initial lateral dimension, which is not a parameter utilized in area source analysis. The amount of volume sources which will represent the roadway link is found by creating multiple volume sources which add up to the total link length, with each volume source being less than 8 m in width. Volume source representation of an emission source requires characterization of the initial horizontal and vertical dispersion caused by the near-wake turbulence, induced by the physical presence of a bluff body

(i.e., various types of on-road vehicles). This additional initial dispersion characterization would result in a wider spread of the pollution and consequently a lower concentration estimates at near-source locations. However, the EPA generally recommends the use of area source characterization over volume source (U.S. EPA 2010b).

For the area source characterization, an initial vertical dispersion height is used to account for vehicle induced turbulence and is estimated to be 1.7 times the average vehicle height. The source release height is used to account for the height at which wind begins to affect the concentration plume and is estimated from the midpoint of the initial vertical dispersion.

Receptors in the model are selected to develop pollutant concentration estimates at various geographic points and can be placed in large grid formats or at discrete locations of importance to the analysis. Receptor locations are typically positioned at ground level or at the average human breathing height, around 1.5 meters.

Meteorological files necessary for input are processed through the meteorological preprocessor (AERMET), and a terrain data preprocessor (AERMAP). Meteorological data refers to upper and surface air data specific to the study area. Upper air data provides information of the atmospheric conditions aloft that change with height in the atmosphere. Variables include pressure, temperature, geopotential height, relative humidity, dew point depression, wind direction and speed. The surface data refers to data that characterizes the atmospheric conditions of lower layers of the atmosphere. Two additional EPA regulatory processors are used to create the input files needed in AERMET. The first of these processors is AERMINUTE. NWS meteorological data is typically used in AERMINUTE. A potential concern related to the use of NWS meteorological data for dispersion modeling is the often-high incidence of calms and variable wind conditions reported for the Automated Surface Observing Stations (ASOS) in use at most NWS stations. The

AERMOD model currently cannot estimate dispersion under calm or missing wind conditions. To reduce the number of calms and missing winds in the surface data, AERMINUTE is used to process archived 1-minute winds for the ASOS stations to calculate hourly average wind speed and directions, which are used to supplement the standard archive of hourly observed winds processed in AERMET (U.S. EPA 2004).

In addition to raw meteorological data, AERMET requires surface characteristic information which can be provided by processing land use data using another EPA regulatory software, AERSURFACE. When applying the AERMET meteorological processor to process meteorological data for the AERMOD model, appropriate values for three surface characteristics must be calculated: surface roughness length, albedo, and Bowen ratio; these parameters are produced by AERSURFACE. Finally, two output files are produced by the AERMET processing, the surface file and the profile file. The surface file contains boundary layer parameters used for scaling and include reference-height winds and temperature. The profile file contains levels of winds, temperature and the standards deviation of the wind speed and wind direction, and typically would represent the site-specific data if included in the analysis (U.S. EPA 2008).

Conducting the AERMOD run, pollutant concentration estimates are provided at each receptor for varying averaging time period, hourly, 24-hours, or annual/period average, and can also provide the maximum concentration for each time period specified.

2.4.5 Literature Review of AERMOD Model Performance

There have been various studies assessing the performance of AERMOD through sensitivity testing of the parameters influencing dispersion results. Before conducting any modeling, the modeling protocol should identify the specific model, modeling options and input data such as, meteorology, emission source parameters, among others, to be used for a particular application. These modeling options are critical to results as the performance of AERMOD might

be sensitive to the representation of vehicle emissions as either volume, area or line sources (Askariyeh et al. 2017). Some studies have found that AERMOD has predicted higher concentrations of PM when emission sources were characterized as area sources as opposed to being characterized as a series of volume sources (Claggett 2014). In contrast, Schewe (2009) reported 1.8 to 3.8 times higher concentration predictions by AERMOD for highways configured as volume sources compared with those configured as area sources (Schewe, Smith, and Consultants 2009). The study recommends that careful source characterization be done when considering volume sources in AERMOD; in addition, the study found that volume sources were very sensitive to changes in surface roughness. The study found that in general, for both area and volume sources, larger source sizes produced lower concentration estimates. Differences between these two studies is evident in the source characterization and the sensitivity analysis. Table 3 shows the main differences between these two studies.

Table 3 Differences in Two Key Studies (Claggett, 2014; Schewe, 2011)

Parameter	**Claggett, 2014**	**Schewe, 2011**
Release Height	1.3 m	3.96 m
Source Elevation	0 m	0 m
Initial Vertical Dispersion	1.2 m	3.68 m
Initial Horizontal Dispersion	7.44 m	2.3-46.5 m
Receptor Elevation	1.5 m	AERMAP
Variations in Sensitivity Analysis	Discrete Wind Angles, Atmospheric Stability	Number/Size of Sources, Land Use

Observing these differences in source characterization might be helpful in evaluating discrepancies in future studies. It is evident that more studies are needed to further evaluate the performance of AERMOD for near-road predictions using different model configurations.

Other studies have evaluated sensitivity related to meteorological conditions. Long et al. (2004) found AERMOD results to be highly sensitive to surface roughness compared to solar radiation, cloud cover, albedo, ambient temperature, and urban population as well as varying by

source type used (Long, Cordova, and Tanrikulu 2004). Faulkner et al., (2008) found pollutant concentrations from AERMOD to be sensitive to surface roughness (very sensitive to values below 0.4m), wind speed (very sensitive to values below 10m/s), temperature, albedo and cloud cover (Faulkner, Shaw, and Grosch 2008). Schroeder et al., (2009) found the location and type of land use around meteorological data location to significantly affect the concentration estimates (Schroeder and Schewe 2009). Grosch et al., (1999) found the pollutant concentrations to change by factors of 1.4, 2.6 and 160 to changes in albedo, Bowen's ratio, and surface roughness length, respectively (Grosch et al. 1999). Kesarkara et al., (2007) found PM10 concentrations from AERMOD to be lower than the observed concentrations in a case study in Pune, India. These output comparisons between modeled and observed concentrations did not include background concentrations (Kesarkar et al. 2007). The authors note that the model performance can be based on comparing the similarity in day-today variation pattern between observed and modeled concentrations, especially when do adequate background concentrations are available. Additionally, the authors consider that the difference in the concentration results can be ascribed to the lack of reliable emission data, and hourly traffic data. These findings further illustrate the importance of obtaining accurate traffic conditions data. The importance of on-site meteorological data to lead to adequate estimates of observed concentrations in urban areas was illustrated by Venkatram et al. (2004) (Venkatram et al. 2004).

When comparing AERMOD and the model CALPUFF, Jittra et al. (2015) found that AERMOD provided more accurate estimates than the CALPUFF model for NO_2 and SO_2 concentrations (Jittra, Pinthong, and Thepanondh 2015). While both models did not perform well for prediction low SO_2 concentrations, AERMOD provided the best results when estimating extreme high-end concentrations.

Compared to other air dispersion models, Gokhale and Raokhande (2008) found CAL3QHC to perform better than AERMOD for all wind speeds greater than 1m/s (Gokhale and Raokhande 2008). Tavares et al. (2009) found CAL3QHCR to underestimate $PM_{2.5}$ and PM_{10} concentration results compared to measured concentrations, attributing this difference to EFs which may not accurately represent the area's actual traffic conditions (Tavares et al. 2010). Kim (2010) concluded that while both AERMOD and CALPUFF were able to reproduce the early morning high benzene concentration, AERMOD and CALPUFF failed to produce accurate predictions where the observed field data indicated elevated high benzene concentration, this mostly occurring under strong downwind conditions (H. S. Kim 2010). Gulia et al. (2012) found the AERMOD, ADMS-Urban and ISCST3 models perform satisfactory when compared to CALINE4, DFLSM and GFLSM for predicting CO concentrations (Gulia, Nagendra, and Khare 2017). This study also found all three models to perform "satisfactorily" for $PM_{2.5}$ concentration predictions, relative to each other. Isakov, et al., (2013), conducted a model inter-comparison based on data from two field studies that had known emissions of inert sulphur hexaflouride gas (SF6) tracers (Isakov et al. 2014). The models included AERMOD, CALINE3 and CALINE4, and measured four model performance statistics: fractional bias (FB), normalized mean square error (NMSE), the correlation (R), and the fraction of estimates within a factor of two of the measured value (FAC2). This study found that AERMOD predominantly performed better than CALINE3 and CALINE4, with a NMSE of 0.31 compared to 2.26 and 0.86 respectively.

A model performance comparison between CAL3QHCR, and three other models (ISCST3, AERMOD, and CALPUFF) shows the varying predictions by the four models (Radonjic, Chambers, and Kirkaldy 2003). The authors used CAL3QHCR as a reference model using a hypothetical road segment and examined different averaging periods and land use conditions; this

model has been widely validated against field observations around roadway sources. Although the line source algorithm used in CALPUFF was not designed for modeling road sources, by selecting parameters to limit the buoyancy of the line source plume, the algorithm can be made to approximate results obtained from using line sources in ISC or AERMOD. The study found that CALPUFF buoyant source best approximates CAL3QHCR followed by ISCST3. They found the PM concentrations from AERMOD to be higher than those from CAL3QHCR by factors ranging from one to six, depending on the averaging period and surface roughness. According to the authors recommendations, there is a need to incorporate a line source algorithm in ISCST3 and AERMOD to produce more reliable results.

In general, AERMOD has been cited as the most up-to-date dispersion model. According to the EPA's AQMG, the dispersion modeling science used in CALINE3 is obsolete compared to AERMOD, RLINE and other state-of-the-science dispersion models (U.S. EPA 2018). CALINE3 is based on the same dispersion science underlying the ISCTS3 model, which EPA replaced with AERMOD in 2005 as the ideal regulatory dispersion model for inert pollutants.

2.5 Previous Near-Road Studies

A number of studies have shown correlations between decay relationships of pollutants near busy roadways (Beckerman et al. 2008; Brugge, Durant, and Rioux 2007; Durant et al. 2010; Padró-Martínez et al. 2012). These studies observed the associations between distance, from highways or high traffic areas, and ambient concentrations of pollutants. These studies showed that various pollutant concentrations are elevated near highways and the decrease within certain distances as a result of dilution. Therefore, it is necessary to research the amounts and kind of pollutants created from mobile sources, especially when traffic corridors are adjacent to areas of high human activity. Various studies also indicate difficulties in predicting near-road emissions. Dhyani et al. (2017) in analyzing the CALINE4 model, found that many factors affecting

predictions were either not considered by the model or have little influence on model's prediction capabilities and therefore considered the model predictions to be unsatisfactory for prediction of $PM_{2.5}$ concentrations (Dhyani, Sharma, and Maity 2017). Hu et al. (2009) in conducting a near-road study of mobile source pollutant concentrations near a highway in Southern California, found that concentration levels measured after sunrise reached background levels at approximately 300 meters from the freeway, which is typically found in most studies. The authors found strong correlation between measured concentration levels and traffic counts on the freeway, and associated the higher observed concentration levels downwind of the freeway during pre-sunrise conditions to nocturnal surface temperature inversion, low wind speeds, and high relative humidity (Hu et al. 2009). Contreras (2015) found that $PM_{2.5}$ concentrations drop off quickly, reaching relatively low concentrations between 300 m to 400 m from the center line of high traffic volume roads. However, during stable atmospheric conditions such as nighttime and winter season, concentrations remain elevated at distances up to 1,000 m from roadway centerlines (Contreras 2015). This is typical in various other near-road studies of pollutant decay after 300-400m, especially of $PM_{2.5}$ concentrations (Patton et al. 2014; Weinstock 2013; Yazdi, Delavarrafiee, and Arhami 2015; Karner, Eisinger, and Niemeier 2010).

Chapter 3: Methodology and Study Design

This study was implemented in five phases in order to assess the exposure of the community living near a major highway. Figure 1 shows a summary of the five phases and the flow of results. The five phases of the study are:

Phase 1: Traffic data collection

Phase 2: Emission modeling

Phase 3: Air pollution measurements

Phase 4: Air dispersion modeling

Phase 5: Data processing and reporting

An area of 1 mile by 1 mile was selected in the northeast part of the City of El Paso. Figure 2 shows the study area of 1 mile by 1 mile. The area was selected based on the traffic conditions, proximity to the highway, and the direction of the prevailing winds. A community near Coldwell Elementary School along the U.S. Highway 54 was selected based on the known high Annual Average Daily Traffic volume (AADT) of 107,237 on U.S. 54 and the low-income status of the community. Traffic data was collected via tube counters located at the major roads found within the study area. Additional traffic data was obtained from a Texas Department of Transportation (TxDOT) highway camera located at the Pershing exit of U.S. 54.

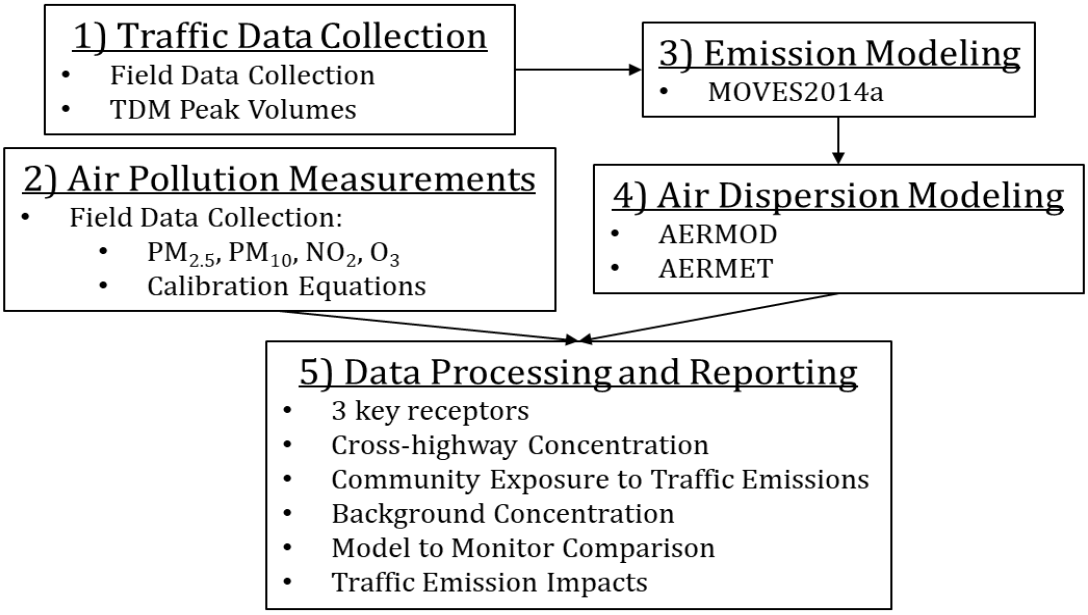

Figure 1 Project Phases and Overall Framework Flow of Results

The map in Figure 2 shows the locations of all the collections sites of traffic and air quality data. Shown in the map are three windroses providing wind speed and wind direction information for key meteorological data reporting sites in the area (El Paso International Airport, UTEP, and Womble). The two near-road sites, House and Coldwell, are both located within 8 and 6 meters from the frontage road alongside U.S. 54, respectively. The third air quality monitoring site, Radford, is located approximately 300 meters away from the frontage road of U.S. 54. The locations of the three tube counters are shown on the map, located at three major arterial roads in the study area. Using video data from TxDOT operated traffic cameras, the locations are shown in the figure, additional traffic volume data was collected for U.S. 54. Finally, the location of the study area, relative to the state of Texas, is highlighted in the figure.

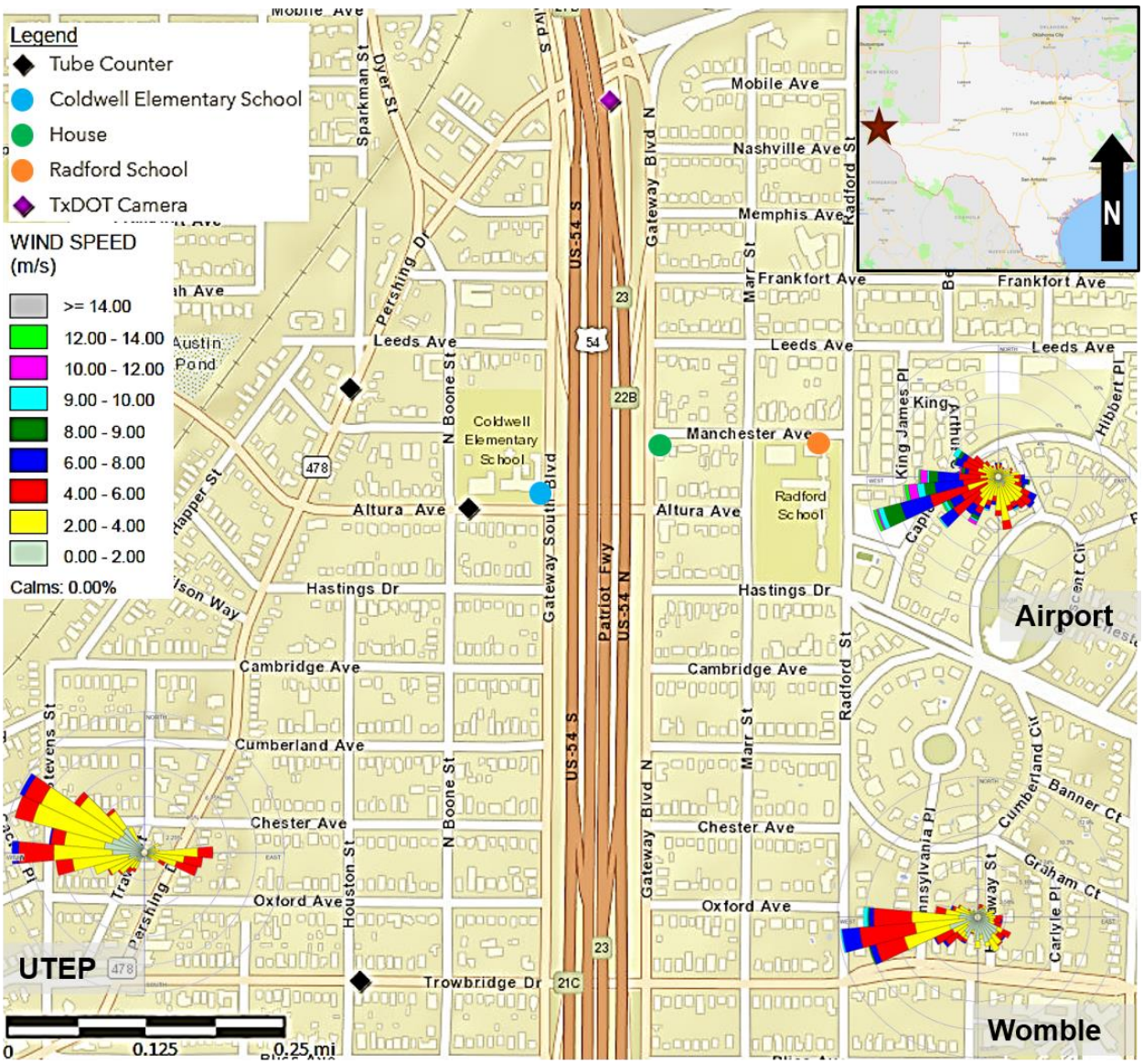

Figure 2 Map of Study Area

For emission factor generations and air dispersion modeling, a general modeling framework, based on the EPA's guidance manual, "Transportation Conformity Guidance for Quantitative Hot-spot Analyses in $PM_{2.5}$ and PM_{10} Nonattainment and Maintenance Areas", was adopted. This manual designates MOVES and EMFAC in California as the official mobile emission models; the official air quality models are AERMOD and CAL3QHCR. This study thus employs the use of MOVES and AERMOD for the modeling portion of the analysis. This will ensure the most accurate results from modeling, as designated by the EPA's guidelines. Details of

the MOVES emission factor generation, input data preparation for both MOVES and AERMOD, and detailed post-processing of AERMOD results are presented in the following chapters.

3.1 Phase 1: Traffic Data Collection

Limited traffic data was collected at 3 locations and at U.S. 54 in the study domain. Vehicle volume counts were recorded using the TRAX Apollyon Counter/Classifier (JAMAR Technologies 2010) at 3 arterial roads in the study area. An example of the tube counter sites is shown in Figure 3. A set of two counters was placed at each of the three different locations, which were chosen for their higher impact of traffic. Each counter included two tubes placed two feet apart; this method provides volume data and vehicle speed data for a two-way street. The vehicle volume was recorded for each hour of the day. The data was used to supplement and calibrate the traffic data previously collected by the City of El Paso Transportation Department at different times and different locations in the study domain. Traffic data for U.S. 54 was obtained by counting vehicles from the video traffic camera footage recorded by the Texas Department of Transportation El Paso District). Hourly vehicle class and number were manually counted by 3 researchers operating independently at different times to avoid human errors and ensure high data quality.

Figure 3 Tube Counters On-Site (Pershing Location)

Traffic volume data at the signalized intersections in the study area was retrieved for the study domain. The City of El Paso Department of Transportation routinely conducts and stores traffic counts at different intersections throughout the years for updating of traffic signal timing plans. This set of traffic volume data was limited to the hours from 7:00 a.m. to 6:00 p.m. and was provided for 9 signalized intersections. In order to utilize this set of data to develop emission estimates from the streets, vehicle class fractions are needed for this study, the vehicle class fractions for the State of Texas were obtained from state vehicle class distributions provided by the Texas A&M Transportation Institute (TTI) for their previous work with the El Paso MPO on the Travel Demand Model (TDM) analysis (EP MPO 2013).

Traffic data for U.S. 54 was obtained by counting vehicles from the video traffic camera footage recorded by the TxDOT El Paso District. This task was jointly conducted by researchers from the UTEP's Border Intelligent Transportation Lab and the Air Quality Research Lab using hand counters and repeated viewing of the video footage with a digital video recorder. As with the tube counting data, vehicle class and volume data was obtained hourly. A sample of the video counting images is shown in Figure 4.

Figure 4 Traffic Camera Video Sample

3.2 Air Quality Data Collection

Air quality data was collected using three different monitoring instruments at each of the three sites. The pollutants analyzed in this study were nitrogen dioxide, (NO_2), particulate matter ($PM_{2.5}$, PM_{10}), and Ozone (O_3). Nitrogen dioxide was measured using 2B Technologies $NO_2/NO/NO_x$ Monitor™ (2B Technologies 2017a). Ozone was measured using 2B Technologies Model 202 Ozone Monitor™ (2B Technologies 2017b). Particulate matter was measured using GRIMM Portable Laser Aerosolspectrometer and Dust Monitor (GRIMM 2010). The $PM_{2.5}$ sensors also provide particle counts for different particle size ranges which provides additional information for the understanding of the PM health effects. Ozone is an EPA regulated criteria pollutant, although not directly emitted from the vehicles but is a photochemical product involving another critical traffic pollutant, NO_2. Placement of the air quality monitors required protection from wind and rain, as well as a housing unit to provide shade. Calibration of the instruments was done in the week before and after the study period; this procedure is described in the next chapter.

Placement of the air quality monitors required protection from wind and rain, as well as a housing unit to provide shade. The figures below show the set-up used for each of the monitoring sites. Figures 5 and 6 show the monitoring sites chosen to be less than 10 m from the frontage road adjacent to the highway, with one monitor on each side of the highway. Figure 7 shows the set-up of the monitoring site chosen to represent the community exposure to the highway's pollution, with the site being around 300 meters away from U.S. 54.

Figure 5 Air Quality Monitor Set-Up: Coldwell Elementary School

Figure 6 Air Quality Monitor Set-Up: Near-Road Home

Figure 7 Air Quality Monitor Set-Up: Radford School

3.3 Phase 3: Emission Modeling

The traffic data generated from field traffic counts at arterial roads as well as the video counting of U.S. 54 traffic were used to generate vehicle emissions factors for AERMOD air exposure concentration estimates. The MOVES emission model was used to generate emissions estimates for all interstate/national highway, arterial roads, and frequently traveled surface roads in the model domain. Temperature, humidity, vehicle speed, vehicle volume, and vehicle fleet mix information were all considered as variables in the MOVES modeling. Each model run corresponds to one hour during each of the four weekday time periods (morning peak, midday, evening peak and overnight) for a representative month during the analysis year. The four weekday time periods are:

- Morning peak emissions based on data 7 a.m. to 9 a.m.
- Midday emissions based on data from 10 a.m. to 3 p.m.
- Evening peak emissions based on data from 4 p.m. to 7 p.m.
- Overnight emissions based on data from 8 p.m. to 6 a.m.

A specific hour within each of the four time periods was modeled and the results were extrapolated to cover the entire day. Because TDM estimates provide the average hourly traffic volume for each peak time period, this method was used to obtain the hourly traffic estimates for emissions modeling. The time span covered is the month of May and the distinct time periods are morning, midday, evening, and overnight. Emissions Factors (EFs) were calculated for a typical weekday, Saturday, and Sunday during the month. A total of 12 MOVES runs were conducted according to all the parameters of the study for each scenario. The speed range is from 20 mph to 60 mph based on posted speed limits in the study link sources.

The EFs produced by MOVES are in terms of grams/hour for each peak time period and included separate EFs for running exhaust emissions and brake wear and tire wear. EFs for re-

entrained dust were calculated for the different types of roads in the study and added to MOVES generated EFs. Re-suspended dust can be quantified using EPA's AP-42 method (U.S. EPA 2010b).

3.4 Phase 4: AERMOD Dispersion Modeling

The AERMOD modeling system includes the use of two regulatory components, a meteorological preprocessor (AERMET), and an air dispersion processor (AERMOD). Meteorological data is needed not only for AERMOD but also for MOVES modeling. Land use data was downloaded from the United States Geological Survey and both hourly surface meteorological data from the El Paso International Airport and upper air soundings and minute data from the regional Santa Teresa Airport were used in AERMET to generate the on-site meteorological data for this study. The following modeling parameters and options were used in AERMOD:

- Passive Pollutant
- Line source, characterized by 180 links, representation for the U.S. 54 highway section
- Urban environment
- Flat Terrain
- Ground-level Release
- Ground-level Receptor
- Initial Horizontal and Vertical Dispersion
- Site-specific Meteorology

Microscale concentration surfaces were established and concentrations at discrete receptor locations were quantified to study the total exposures of near-road communities using the AERMOD air dispersion model. Pollutant air concentrations were used to apportion the

contributions of emissions from the interstate highway as well as arterial roads. The figure below illustrates the flow of data in the AERMOD modeling process.

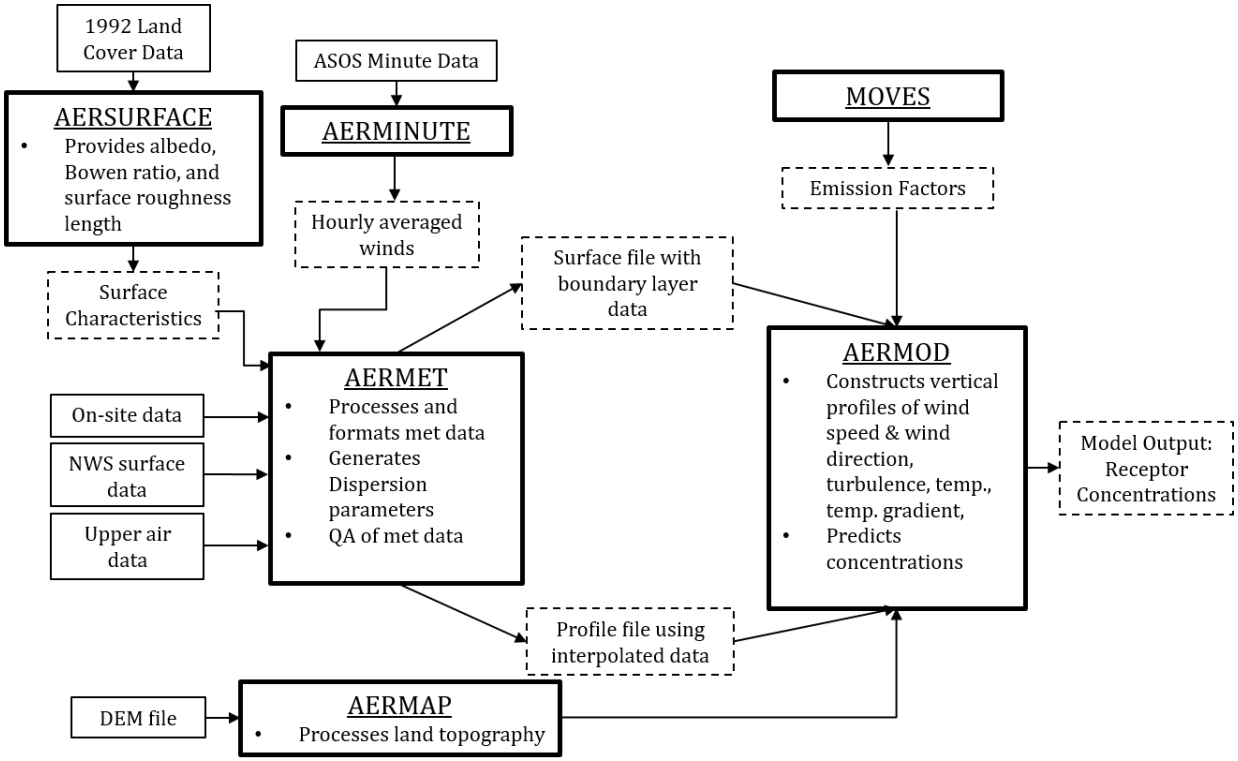

Figure 8 AERMOD Model Data Flow

AERMOD includes the use of two regulatory components, a meteorological preprocessor (AERMET), and a terrain data preprocessor (AERMAP). Meteorological data is needed for AERMOD and MOVES modeling and refers to upper air and surface data specific to the study area monitoring station locations. Upper air data provides information to measure the characteristics that change with height in the atmosphere, such as temperature. The surface data refers to data that measures the characteristic of lower layers of the atmosphere. As shown in the data flow chart, two additional EPA regulatory processors are used to create the input files needed in AERMET. The first of these processors is AERMINUTE. A potential concern related to the use of NWS meteorological data for dispersion modeling is the often-high incidence of calms and variable wind conditions reported for the Automated Surface Observing Stations (ASOS) in use at

most NWS stations. The AERMOD model currently cannot estimate dispersion under calm or missing wind conditions. To reduce the number of calms and missing winds in the surface data, AERMINUTE is used to process archived 1-minute winds for the ASOS stations to calculate hourly average wind speed and directions, which are used to supplement the standard archive of hourly observed winds processed in AERMET (U.S. EPA 2004).

In addition to raw meteorological data, AERMET requires surface characteristic information which can be provided by processing land use data using another EPA regulatory software, AERSURFACE. When applying the AERMET meteorological processor to process meteorological data for the AERMOD model, appropriate values for three surface characteristics must be calculated: surface roughness length, albedo, and Bowen ratio. The surface roughness length is related to the height of obstacles to the wind flow and is the height at which the mean horizontal wind speed is zero based on a logarithmic profile. The surface roughness length influences the surface shear stress and is an important factor in determining the magnitude of mechanical turbulence and the stability of the boundary layer. The albedo is the fraction of total incident solar radiation reflected by the surface back to space without absorption. The daytime Bowen ratio, an indicator of surface moisture, is the ratio of sensible heat flux to latent heat flux and, together with albedo and other meteorological observations, is used for determining planetary boundary layer parameters for convective conditions driven by the surface sensible heat flux (Cimorelli et al. 2005).

The meteorological files and emission factors produced by MOVES are used to develop a range of scenarios for dispersion modeling in AERMOD. The emission factors (EFs) produced by MOVES are converted into a format compatible with area source characterization in AERMOD. The BREEZE AERMOD and BREEZE ROADS models, commercial propriety software

developed by Trinity Consultants Inc. which provides an unaltered, user-friendly, window-based version of the EPA-approved AERMOD model with pre- and post-processors, is used to help with the source and receptor coding with AERMOD. Further details regarding the MOVES processing of EFs and the AERMOD model set up is discussed in following chapters.

Chapter 4: Calibration Data for Air Monitors

The study period for collecting air quality data was May 8th through May 25th, 2018. Pre-calibration was conducted in the week before the field study; post-calibration was conducted the week after the field study. All monitoring instruments were placed alongside the continuous air monitoring station (CAMS 12) operated by the Texas Commission on Environmental Quality (TCEQ) located on the UTEP campus. Figure 9 shows the placement and set-up of the study's air quality monitoring instruments located next to CAMS 12. . This set-up remained identical during the study period to reduce any variance caused by the housing of the units. Table 4 shows the calibration equations and how well the monitor data correlates with measured and validated CAMS data.

Figure 9 Air Monitoring Instrument Calibration Set-Up

The CAMS 12 data are recorded by using EPA-approved FRM devices. The data has the highest accuracy and precision and is accepted for regulatory compliance study. It was used to check the accuracy of the values reported by the air monitoring instruments used in our study and

develop calibration constants accordingly. The following table shows the calibration equations and R-values for the correlations of CAMS data with the instruments using the pre-calibration and post-calibration data.

Table 4 Calibration Data

$PM_{2.5}$			
Instrument	1	2	3
Equation	y = 0.757x + 3.0454	y = 0.7288x + 2.2831	y = 1.2163 + 2.7014
r-value	R^2 = 0.9524	R^2 = 0.9623	R^2 = 0.9585
PM_{10}			
Instrument	1	2	3
Equation	y = 2.9905x + 9.1655	y = 0.9254x + 9.4285	y = 3.7404x + 10.315
r-value	R^2 = 0.7958	R^2 = 0.8254	R^2 = 0.8853
NO_2			
Instrument	1	2	3
Equation	y = 1.0025x + 0.173	y = 0.6x - 3.009	y = 0.9018x - 6.2989
r-value	R^2 = 0.9743	R^2 = 0.7681	R^2 = 0.8741
Ozone			
Instrument	1	2	3
	Calibration	Calibration	Calibration
Equation	y = 1.1521x - 2.0866	y = 1.012x + 6.598	y = 1.0086x + 1.0995
r-value	R^2 = 0.9609	R^2 = 0.8745	R^2 = 0.9368

The calibration equations show how well the monitor data correlates with measured and validated CAMS data. All instruments show great accuracy with high R2 values (0.95-0.96 for PM2.5, 0.80-0.88 for PM10, 0.77-0.97 for NO_2, and 0.87-0.96 for ozone). Ozone monitors show the most accurate correlation with r-values between the three instruments averaging at 0.95. Calibration of all instruments used in this study is necessary since all our instruments measure pollutant concentrations using optical principles of the pollutants different from the principles used in EPA FRM devices. The calibration equations developed in this phase of study were used to correct the air quality data collected from the near-road study, as is discussed in the following chapter.

Chapter 5: MOVES Emission Factors Generation

This chapter discusses the process necessary in generating the EFs to be used in this study's analysis. The traffic data generated from field traffic counts at arterials as well as digital data record recounting of U.S. 54 traffic were used to generate vehicle emissions rates. This was done with the EPA's Motor Vehicle Emissions Simulator (MOVES). Temperature, wind speed, and wind direction were all considered as variables in the MOVES modeling. The MOVES emission model was used to generate emissions estimates for all interstate highway, arterial roads, and frequently traveled surface roads in the model domain.

5.1 MOVES Model Inputs

In order to produce the emissions data required by the dispersion model AERMOD, MOVES must first use traffic and vehicle fleet data to calculate emissions rates or inventory values of pollutants. Figure 23 illustrates the flow of data during the MOVES modeling process.

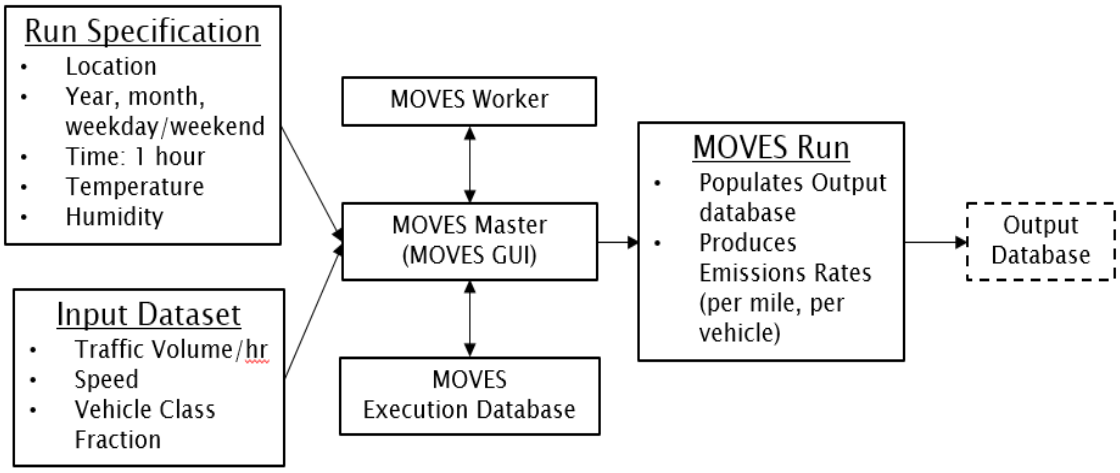

Figure 10 MOVES Model Data Flow

The MOVES model includes six road types: off-network, rural restricted, rural unrestricted, urban restricted, urban restricted. For the purpose of this emission estimation, freeways and interstates are classified as "urban restricted" roads. All other urban roads in the network are classified as "urban unrestricted" roads.

Model-to-monitor evaluation based on PM hot-spot process are based on the temporal attributes as required by the EPA hot-spot guidance (U.S. EPA 2010b). Depending on the level of sophistication required for the activity data for a given project, the emission estimates to be generated may range from a daily average-hour and peak-hour value to hourly estimates for all days of the year. The EPA recommends a minimum of 16 MOVES runs necessary for a yearly PM Hot-Spot analysis to capture changes in emission rates due to changes in ambient conditions. These 16 model runs correspond to four weekday time periods (morning peak, midday, evening peak and overnight) for four representative months (January [winter season)], April [spring], July [summer] and October [fall]). This study will instead only model the representative days of field collection, calculating emissions rates for a typical weekday, typical Saturday, and a typical Sunday. The following approach is suggested by the EPA for an analysis. The emission factor generation framework uses the peak-hour, or average-hour traffic volume for a typical weekday during the following four daily peak periods, established by the TDM:

- Morning peak emissions based on data 7 a.m. to 9 a.m.
- Midday emissions based on data from 10 a.m. to 4 p.m.
- Evening peak emissions based on data from 4 p.m. to 7 p.m.
- Overnight emissions based on data from 7 p.m. to 7 a.m.

A specific hour within each of the four time periods is modeled and the results are extrapolated to cover the entire day. The average of the hours during each time period is modeled for four different hours in MOVES2014a.

Macroscopic models such as TDMs are routinely used to estimate total base and forecast year traffic volume, vehicle miles traveled (VMT), and average speeds used in developing regional emissions inventories. The historical data for these parameters from the El Paso Metropolitan Organization (MPO), along with on-site vehicle data collected during the study period, were used as inputs to MOVES to generate emissions rates (EP MPO 2013). Classification, speed, and

volume are quantified and demonstrated in each link (road section) included in the MOVES analysis.

5.2 $PM_{2.5}$ Emission Factor Generation for Study Area

A total of 12 MOVES2014a runs were conducted according to all the parameters of the study for all scenarios. The time span covered is the month of May and the distinct time periods are morning, midday, evening, and overnight. Emissions factors (EFs) were calculated for a typical weekday, Saturday, and Sunday during May of 2018. All input data for MOVES2014a can be set up in two main steps. The first step is setting up the RunSpec input parameters discussed in Chapter 2. The details of this study's RunSpec inputs are summarized in Table 5.

Table 5 MOVES2014a RunSpec Inputs

Parameters	Specification for Run
Scale	Project-Level
Time Span	May 13-24 2018, Weekday, Weekend
Geographic Bounds	El Paso County
Vehicles/Equipment	Motorcycle, Passenger Car, Passenger Truck, Light Commercial Truck, Intercity Bus, Transit Bus, School Bus, Refuse Truck, Single Unit Short-haul Truck, Single Unit Long-haul Truck, Motor Home, Combination Short-haul Truck, Combination Long-haul Truck
Road Type	Urban Unrestricted
Pollutants and Processes	$PM_{2.5}$
Output	Inventory (grams/link)

The second step consists of preparing MOVES input data through the MOVES Project Data Manager (PDM) user interface. In general, there are two types of data required for project-level MOVES2014a runs:

- Site-specific traffic information, including traffic volumes, and speed.
- Local-specific inputs, including regional-level vehicle age, source distribution, meteorology, fuel supply, and I/M program parameters.

The following sections detail the input values necessary for generating the $PM_{2.5}$ EFs for the roadways in the study area using MOVES2014a.

5.2.1 Site-Specific Traffic Information

As discussed in Chapter 3, traffic data was collected for 3 arterial roads in the study area as well as for U.S. 54 during the study period in May 2018. The following section details how the traffic information for all the roadways in the study area was obtained. This site-specific data was used in conjunction with TDM estimates provided by the El Paso MPO as part of their Horizon 2040 Metropolitan Transportation plan.

5.2.1.1 TDM Adjustments with Traffic Data

The three sets of tube counter and video traffic data were used in conjunction with TDM estimates to supplement traffic data for all arterials and highway sections in the study area. As part of its Air Quality Conformity Analysis, the MPO utilized a TransCAD TDM to estimate future travel demand and traffic conditions for the city. The TDM has a validated 2007 base year with forecast network years of 2010, 2020, 2030 and 2040. The model is a 24-hour model, validated using 24-hr traffic counts. The time of day periods were generated by using time of day factors developed from the 2009 National House Hold Travel Survey (Federal Highway Administration 2010). Because of limited input data, the model does not provide hourly values but rather peak time period averages for the roadways modeled. These roadways in the TDM are those which are defined as being regionally significant. TDM estimates also provide posted speed data, which is necessary for EF generation. The speed range is from 20 mph to 60 mph based on posted speed limits in the study link sources. The roadway network links and associated traffic data were extracted as shown in Figure 11, highlighting the links used for obtaining ratios from the observed on-site traffic data from May 2018.

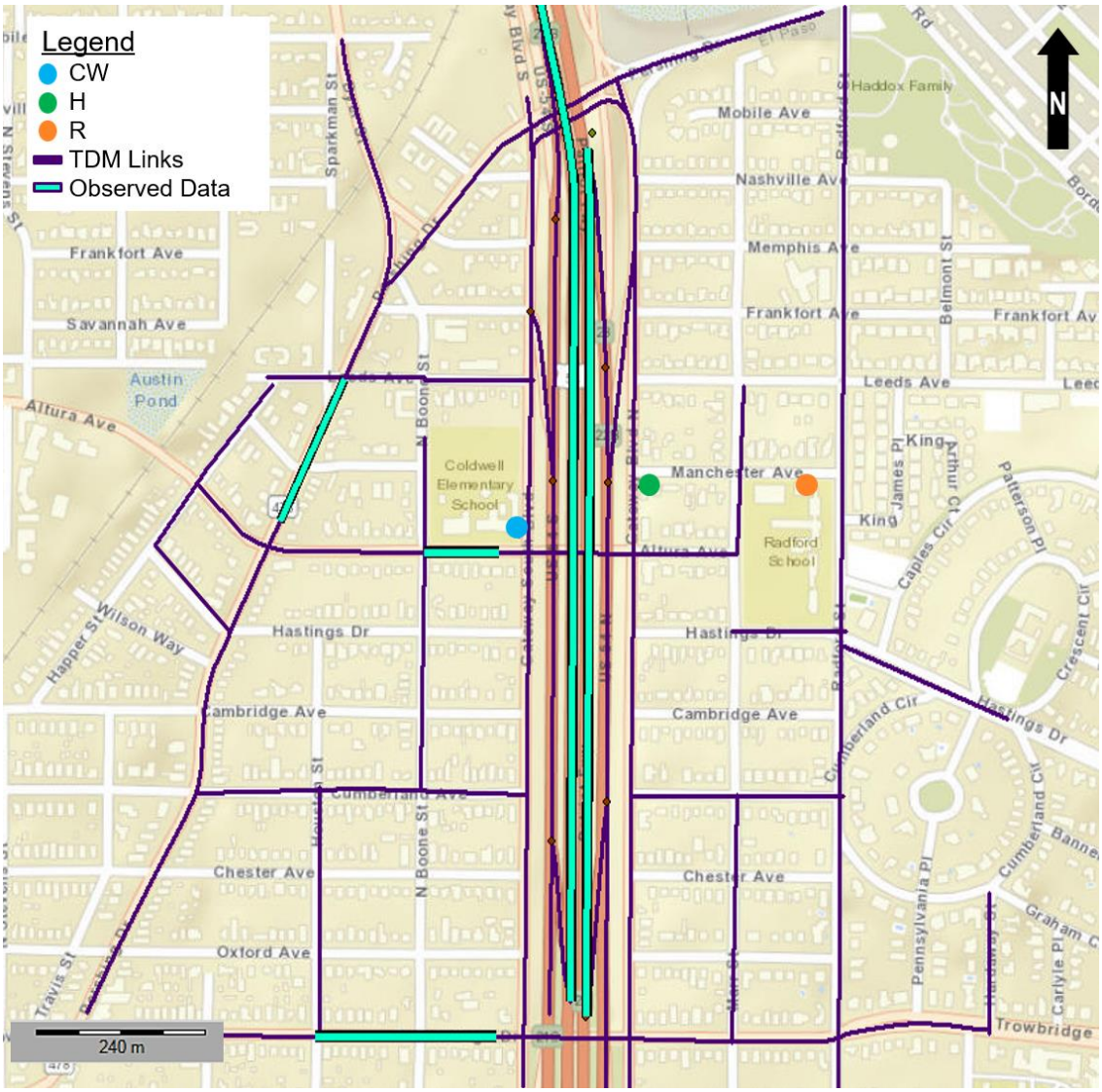

Figure 11 Roadway network Links Extracted from TDM for El Paso

TDM estimates only provide daily volume estimates for four time periods of the day, not distinguishable between weekday and weekend values. Using the traffic data measured during the May study period, ratios were created for the corresponding links from the TDM to provide greater resolution. The ratios were computed by dividing the TDM estimate by the measured data for the links that have both TDM and measured traffic data. A new adjusted weekday hourly estimate was created for each peak hour in the time period for all roads by multiplying the TDM values by the ratio of the same type of road, i.e. the ratio found from the Altura street was used to adjust TDM

estimates from similar small arterial streets. The same process was repeated to create weekend hourly estimates. Table 6 shows the corresponding adjustment ratios created from the observed data that are used for the rest of the links in the study area. Observed traffic data is used for the links with collected data from May 2018.

Table 6 Ratios for Adjusting TDM Estimates

		TDM Estimate	Observed (May 2018)			Ratio		
			Weekday (avg)	Saturday	Sunday	Weekday (avg)	Saturday	Sunday
U.S. 54 SB	AM	2111	5433	2669	2004	2.57	1.26	0.95
	MD	1207	2925	3360	2522	2.42	2.78	2.09
	PM	1153	3649	2900	2177	3.17	2.52	1.89
	NT	450	647	1103	828	1.44	2.45	1.84
U.S. 54 NB	AM	1811	2725	1356	1018	1.50	0.75	0.56
	MD	1587	2466	2647	1987	1.55	1.67	1.25
	PM	2101	4374	2466	1851	2.08	1.17	0.88
	NT	457	763	1269	953	1.67	2.78	2.08
Pershing	AM	2145	441	185	125	0.21	0.09	0.06
	MD	1009	498	475	413	0.49	0.47	0.41
	PM	1468	596	415	335	0.41	0.28	0.23
	NT	139	123	177	104	0.89	1.28	0.75
Altura	AM	1122	218	79	45	0.19	0.07	0.04
	MD	425	165	143	149	0.39	0.34	0.35
	PM	718	238	122	120	0.33	0.17	0.17
	NT	48	36	54	36	0.75	1.14	0.76
Trowbridge	AM	433	411	149	111	0.95	0.34	0.26
	MD	64	390	416	415	6.09	6.50	6.48
	PM	336	621	380	289	1.85	1.13	0.86
	NT	10	107	137	90	10.77	13.84	9.09

This adjustment was necessary to provide the most accurate traffic estimates for the EF generation. In addition to providing more accurate estimates from the TDM modeled links, EFs created from these estimates were used to represent other similar roadways in the study area which amounted to 180 links total. All of the roadways modeled in MOVES2014a and AERMOD are shown in Figure 12.

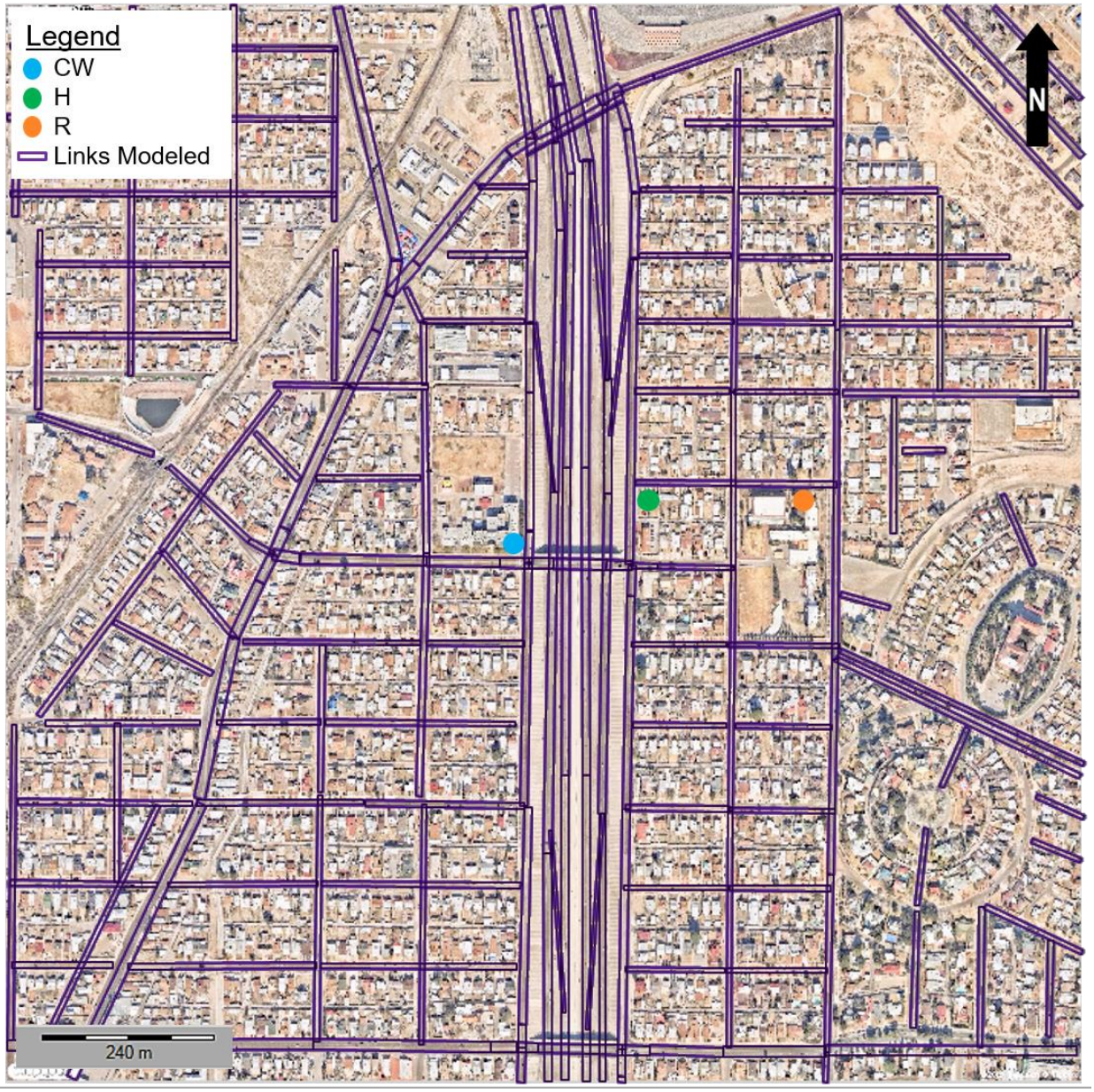

Figure 12 Total Links Modeled in MOVES2014a

The detailed AERMOD source characterization of these 180 links is defined further in Chapter 7.

5.2.2 Local-Specific Inputs

Local-specific inputs generally include regional-level vehicle age, source type distribution, fuel supply, and meteorology. The meteorology data, which consists of hourly temperature and humidity, was obtained from the El Paso Airport Site which was chosen to represent the

meteorology for the study area. The meteorological data used to create the files necessary for air dispersion modeling in AERMOD are discussed in Chapter 6. The other local-specific inputs, pertaining to vehicle fleet information, were provided by TTI and the El Paso MPO (EP MPO 2013).

5.2.3 Post-Processing of MOVES2014a Outputs

The EFs produced by MOVES are in terms of grams/hour for each peak time period and include separate EFs for running exhaust emissions and break wear and tire wear. Conducting AERMOD dispersion modeling using the area characterization for sources requires a combined EF in grams/sec/m^2 so further calculations were conducted to prepare the EFs for use in AERMOD. Additionally, EFs for re-entrained dust were calculated for the different types of roads in the study and added to MOVES generated EFs. Re-suspended dust can be quantified using EPA's AP-42 method or alternative local methods. AP-42 is EPA's compilation of data and methods for estimating average emission rates from a variety of activities and sources from various sectors (U.S. EPA 2010b).

Chapter 6: Meteorological Data

AERMOD requires two meteorological input files for developing concentration estimates; these are a surface and profile file, both created using the U.S. EPA-approved AERMET meteorological model. The following chapter outlines a detailed overview of the meteorological data processing for dispersion modeling in AERMOD.

6.1 Meteorological Data Processing for AERMOD

Meteorological conditions strongly impact the pollutant dispersion in the atmosphere. Three types of data are required for processing the meteorological data, namely, surface data that measure characteristics of lower layers of the atmosphere, upper air data that measure characteristics that change with height in the atmosphere (such as temperature), and land use data that represent surface characteristics. For this study, the raw meteorological and land use data were obtained from the following sources:

- Automated Surface Observing Stations (ASOS).
- National Weather Station databases (NWS).
- U.S. Geological survey land use database (USGS).

The ASOS and NWS databases are owned and maintained by NCDC and National Oceanic and Atmospheric Administration (NOAA) under the U.S. Department of Commerce (NOAA, 2018). USGS land use database is a national archive for remotely sensed images of Earth's land surface maintained by the U.S. Department of the Interior (USGS 2018). Figure 13 shows the process of meteorological data processing for AERMOD. The raw data are processed using meteorological preprocessors namely, AERMINUTE, AERMET, and AERSURFACE to produce data in a format compatible for AERMOD. Flow components with dashed outlines indicate files produced as outputs by the different pre-processors.

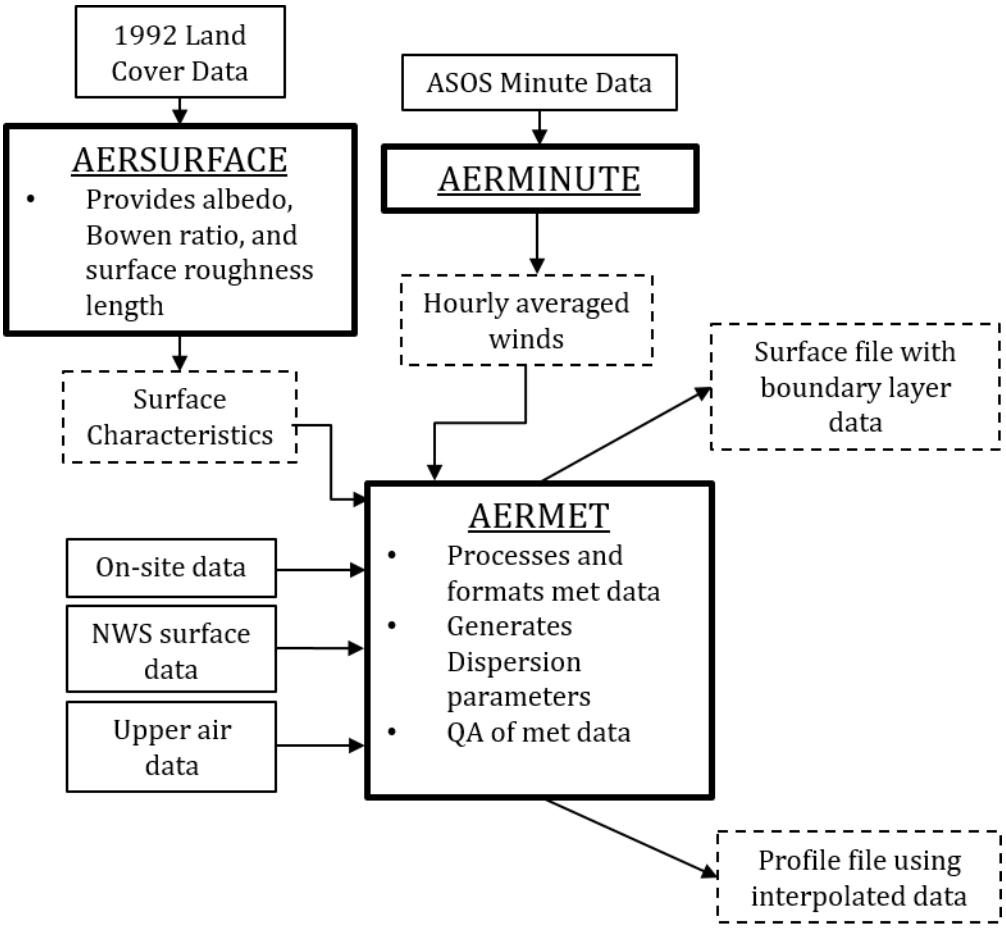

Figure 13 Meteorological and Land Use Data Processing for AERMOD

High resolution wind data are processed by the AERMINUTE preprocessor. In order to obtain supplemental hourly averages for surface meteorological data, the AERMINUTE tool uses 1-minute average wind speeds for each minute of the hour for most ASOS stations to find hourly averages. These values help supplement any missing hours of data from the surface and on-site meteorological data files. One of the main concerns in using NWS surface data directly for AERMOD is the presence of high incidence of calm and missing wind data. AERMOD cannot accurately simulate dispersion with calm/missing winds. To reduce this, NCDC started archiving raw one-minute data logged by automated stations. AERMINUTE is used to process the one-minute data to produce hourly wind speed and direction averages to improve the quality of surface data obtained from the NWS.

The AERSURFACE pre-processor helps modelers obtain realistic and reproducible surface characteristics for input to AERMET. These surface characteristics relate to the following parameters:

• Albedo: fraction of total incident solar radiation reflected back to space without absorption.

• Bowen ratio: indicates how much heat the ground imparts to the air instead of evaporating moisture at the surface (amount of surface moisture conditions).

• Surface roughness length: indicates how much the surface features at a given site interrupt a smooth-flowing wind (height of obstacles to the wind flow).

This data can be obtained from a national archive for remote sensor images of Earth's land surface maintained by the U.S. Department of the Interior. National Land Cover Data from 1992 (NLCD 1992) is obtained for use in this tool from the USGS. These databases contain archived data measured by surface and upper air stations throughout the country.

Finally, AERMET incorporates surface and upper data from the NWS database and combines them with the hourly wind speed and direction averages produced by AERMINUTE and land cover surface data (albedo, surface roughness, and Bowen's ratio) from AERSURFACE to produce output files for AERMOD. The two files produced by AERMET consist of a boundary layer parameter (surface) file that includes turbulence parameters, mixing height, and friction velocity. The second file (profile) contains the vertical profile of winds, temperature, and standard deviation of the fluctuating components of the wind. These two files are directly incorporated into AERMOD. According to EPA (U.S. EPA 2004), AERMET shall be used to preprocess all meteorological data, be it observed or prognostic, for use with AERMOD in regulatory

applications, and the AERMINUTE processor, in most cases, should be used to process 1-minute ASOS wind data for input to AERMET when processing NWS ASOS sites in AERMET.

6.2 Data Processing: Meteorological Files required by AERMET

The following section details the input files required by the pre-processor AERMET to create the necessary meteorological files for air dispersion modeling in this study.

6.2.1 Surface/On-Site Data Input

Meteorological data, including measurements of wind speed, wind direction, ambient temperature, barometric pressure, peak wind gust and precipitation, observed at ambient monitoring stations is used in this study. The surface input file is acquired from NCDC of NOAA. Because of the lack of an available on-site meteorological station for this study, the surface data and on-site data was obtained from the same site. The meteorological site chosen to represent the on-site meteorology was the El Paso International Airport., 3.75 miles from the study area location and is owned by the National Oceanic and Atmospheric Administration (NOAA).

The airport site was chosen for its proximity to the study area as well as having the most similar topographic characteristics. Because this site is operated by the NOAA, it also provides the most complete and accurate data compared to other meteorological sites in the area. Figure 14 shows the windrose depiction of wind speed and wind direction during the study period from all available meteorological stations in El Paso. It can be seen that the predominant wind direction in the area is from southwest to northeast.

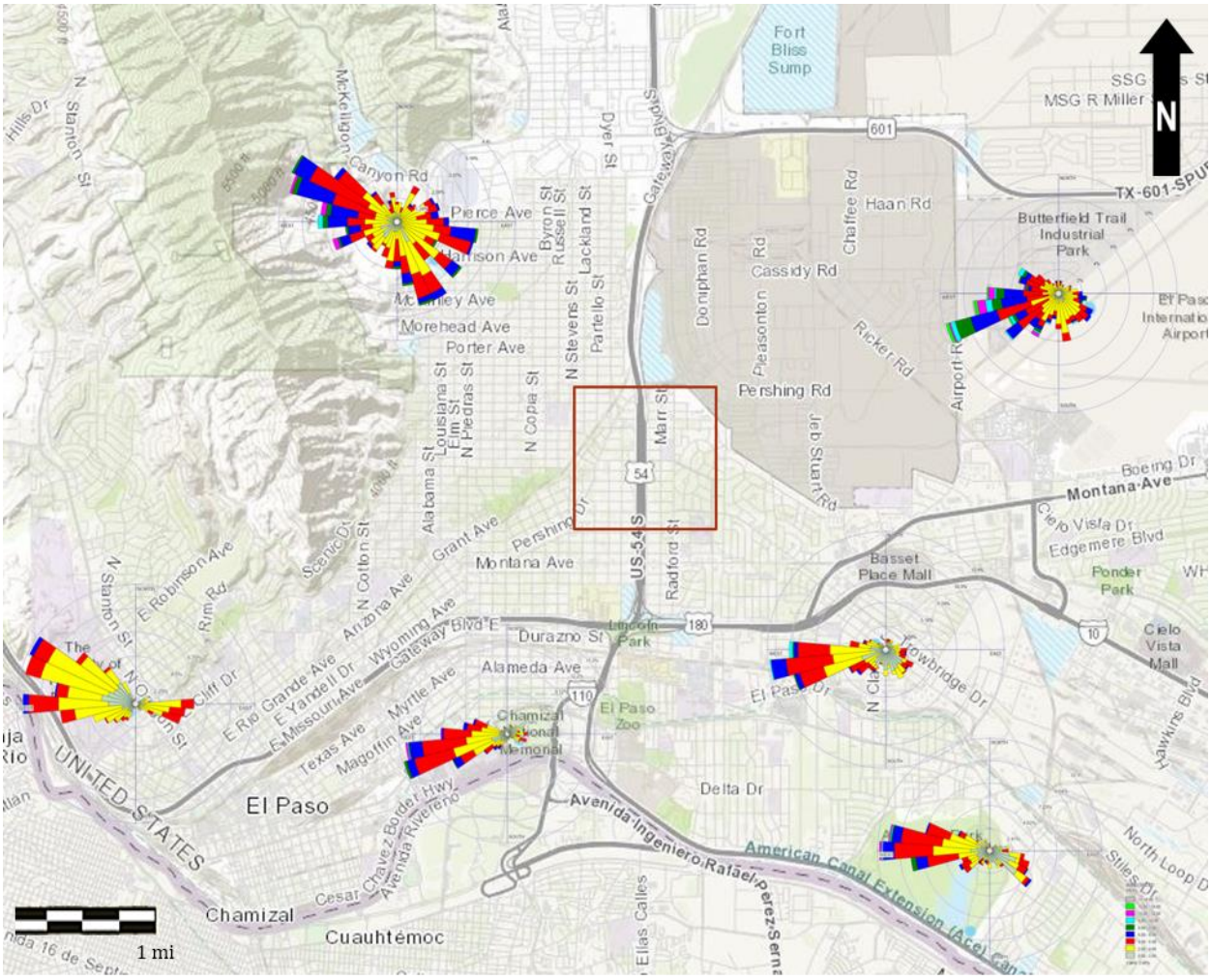

Figure 14 Meteorological Data for El Paso

6.2.2 Upper Air Data Input

Upper air data are recorded at unevenly, sparsely distributed locations throughout the United States. The NOAA stations provide twice-daily upper air soundings and data, which can be retrieved at the NOAA's Radiosonde Database (NOAA 2018). Selection of the closest upper air data for use in air dispersion modeling requires special attention as only certain stations record data at a certain time so the closest upper air station to the point of interest can be far away from the modeling domain. This study obtained data from the upper-air station in Santa Teresa, NM, as it is the closest station for modeling done in the El Paso area.

6.2.3 AERMINUTE Input

A potential concern related to the use of ISD meteorological data for air dispersion modeling is the often-high incidence of calms and variable wind conditions. In the reporting of surface weather data, a calm wind is defined as a wind speed less than 3 knots and is assigned a value of 0 knots. In addition, the wind direction may be reported as missing if the wind direction varies more than 60 degrees during the 2-minute averaging period for the observation (O'Donnell et al. 2011). To reduce the number of calms and missing winds in the surface data, the 1-minute ASOS wind data are used to calculate hourly average wind speed and directions, which are used to backfill the missing data and calms in the ISD data. This ASOS minute data can be found in the NCDC database, from the same database as the surface data (El Paso Airport). The ASOS data contain both TD 6405 and TD 6406 formatted files. For the purpose of creating a meteorological file, the data start with 6405 followed by the desired year were used. As the ASOS minute files are unusually large, they need to be downloaded separately based on the months required.

6.2.4 AERSURFACE Input

The AERSURFACE processor is developed to compute surface characteristic values such as albedo, Bowen ratio, and surface roughness length, in a modeling domain for use in AERMET (U.S. EPA 2008). Similar to AERMINUTE, data from AERSURFACE can be created or simplified by dividing the area of study into different sectors and giving each sector an albedo, Bowen ratio, and surface roughness. For this project, the AERSURFACE program was run using National Land Cover Data from 1992 (NLCD 92) from the United States Geological Survey (MRLC 2018).

6.3 Meteorological files for use in AERMOD

Once the surface file and profile files have been created, they can be used as input into AERMOD for air dispersion modeling. It is important to note that in the treatment of calm condition, AERMET assigns zero values and defaults the wind direction to 0 degree for all wind speeds of less than or equal to 1 m/sec. In addition, the model sets the concentration values to zero for hours with calm wind or missing meteorological data and calculates the average by summing each valid (non-calm) 1-hour average concentration and dividing by the total number of non-calm hours or 75 percent of the total number of hours in the period, whichever is greater (U.S. EPA 2004). The total percentage of missing data for the month of May was found to be 5.6%, or 42 hours, and correspond to missing upper air data that cannot be adjusted.

Chapter 7: AERMOD Dispersion Model Set Up

Air dispersion models such as AERMOD are used by regulatory agencies to illustrate that federally supported transportation projects will not have a significant effect on the human environment. Recognizing the important role of these models in the transportation conformity project level hot-spot process, a model-to-monitor evaluation approach is used based on hot-spot analyses. Hot-spot analysis, as defined in 40 CFR Part 93.101, is an estimation of likely future, localized pollutant concentrations and their comparison to the NAAQS. Hot-spot analyses are a part of the conformity requirements for pollutants that have localized impacts, such as particulate matter (PM). They are generally required for projects identified as being of air quality concern, in the respective PM nonattainment or maintenance areas. Using this method can help maintain an adequate comparison of monitored data with modeled data. Steps to be followed in the evaluation and implementation of the modeling process are further illustrated and summarized in Table 7.

Table 7 Steps in Modeling Approach

1. Calculate a representative average daily traffic with hourly variations corresponding to the study time period, May 2018.
2. Based on the average data traffic, calculate $PM_{2.5}$ emission rates corresponding to exhaust emissions, brake and tire wear using the MOVES model
3. Develop 1-year of onsite meteorological data based on ambient parameters measured at the nearest continuous air monitoring stations for year 2018 combined with the nearest representative upper air and surface stations (El Paso Airport Data)
4. Set-up AERMOD with source and receptor characterization of the study area
5. Calculate modeled concentrations corresponding to 1-hr maximum, 24-hr maximum, and annual averaging period
6. Calculate the background concentration corresponding to year 2018 from representative ambient monitors surrounding the study area using a normalized inverse distance method or other appropriate method
7. Calculate the near-road increment from the near-road monitored evaluations and background concentration corresponding to 1-hr maximum, 24-hr maximum, and annual averaging period
8. Compare the modeled estimates with the near-road increment corresponding to 1-hr maximum, 24-hr maximum, and annual averaging period
9. Assess the model-to-monitor comparison, for modeled and modeled + background estimates

7.1 Modeling Setup

After compiling the necessary data related to meteorology, land use, and emission factors, the parameters for the dispersion modeling must be defined. Base imagery can be obtained from sources such as Google Earth, ArcMap, or the Input map feature in BREEZE AERMOD graphic user interface. The model domain is defined as a 1-mile by 1-mile area in the Coldwell Elementary School area, as shown in Figure 2.

7.1.1 Model Parameters

For this study, the dispersions model was set to estimate the pollutant PM2.5, with no depositions and settling. Concentration estimates were calculated for hourly, maximum hour, 24-hour, and all-period (or 1-month in our study) averages.

AERMOD allows for two different designations for land use: urban, and rural site. If at least 50% of the land use within a 3-kilometer (km) radius of the model domain is of an urban type, the source is designated urban, and rural if otherwise (U.S. EPA 2018). For urban areas, the model activates the urban heat effect, a term used to describe urban areas that are hotter than nearby rural areas, especially at night, mainly as a result of heat retention by urban materials. Because of this heat retention, the vertical motion of the air is increased through convection, thereby leading to the increased dispersion of pollutants. AERMOD accounts for urban dispersion effects and also requires the urban area population to determine the degree of urban heat island effect occurring in a specific urban area. In this study, the modeling domain is classified as “urban”.

7.1.2 Source Characterization and Dispersion Parameters

AERMOD can model roadway line source as a series of volume or area sources. EPA guidance recommends modeling roadway links as area or volume sources for PM hot-spot analysis. In our study, roadway emissions are modeled as a series of area sources, which are

defined as flat, two-dimensional spaces from which emissions originate. They are appropriate for near ground level sources with no plume rise.

- Source Characterization

Area sources model emissions with a uniform distribution along the roadway link and are not distributed beyond the edge of a defined roadway link. In AERMOD, a series of area sources can be modeled as a "line" source with specified width and length for simulating roadway emissions simulation. This source characterization also allows for a lower number of sources, reducing run times. Therefore, "Line Source' is selected to characterize the source configuration of each road link. Each source is defined by the travel activity, physical dimensions, and orientation of the roadway link it is representing.

- Initial dispersion characterization

To simulate the initial dispersion on highway due to the additional turbulent mixing of the winds behind and around the vehicle due to the physical presence of the vehicles, AERMOD allows the users to characterize the wake effect around the vehicles by defining an initial horizontal dispersion coefficient and a vertical dispersion coefficient. According to EPA hot-spot guidance, the initial vertical dimension for roadway emissions is assumed to be about 1.7 times the average vehicle height, to account for the effects of vehicle-induced turbulence. For light-duty vehicles, this height is about 2.6 m, using an average vehicle height of 1.53 m, or 5 ft. For heavy-duty vehicles, this height is about 6.8 m, using an average vehicle height of 4.0 m. The AERMOD User's Guide recommends that the initial vertical dispersion coefficient (σzo) to be estimated for a surface-based area/volume source by dividing the initial vertical dimension by 2.15. For typical light-duty vehicles, this figure corresponds to a σzo of 1.2 m. For typical heavy-duty vehicles, this figure corresponds to a σzo of 3.2 m. For roadway links having a combination of light-duty and

heavy-duty traffic, the guidance recommends the coefficient to be calculated as a combination of their respective σzo values by using a traffic volume-weighted or emissions-weighted average. Initial lateral dispersion is only required for modeling volume sources.

- Source Release Height

The source release height is the height at which winds begins to affect the plume. It is estimated from the midpoint of the initial vertical dimension. The source release height is used to account for the height at which wind begins to affect the concentration plume and is estimated from the midpoint of the initial vertical dispersion. Similar to σzo, the source release height for roadways with a combination of light duty and heavy-duty vehicles is calculated using a traffic volume-weighted or emissions-weighted average. In this study, the source release height is calculated to be 1.45 meters.

- Emission Rates from MOVES

Characterizing emission sources consists of defining their area and assigning the rate at which emissions are produced by the source. Emission rates from MOVES2014a are converted into the appropriate unit compatible with area source characterization as used by the AERMOD model. Emission factors for area source characterization must be input into AERMOD in units of "grams/sec/m^2".

- Line Source Representation

Sources are characterized by their corresponding links from the MOVES emissions calculations. The area of each source is designated using the length and number of lanes in each road segment. The area source characterization for the entire study are was modeled using 180 area sources. The study aims to model as many roadways as possible and therefore all roadways in the study area are modeled as sources.

- Receptor Selection

Receptors are points at which the AERMOD model provides concentration estimates for the pollutant modeled. Receptors for the study area placed at an elevation equal to the meteorological site, i.e. ground-level, at the three monitor locations from the May 2018 study. A grid of 2,500 receptors is also placed to capture concentration estimates throughout the entire study area. Figure 15 shows the model set-up with the 180 sources and the grid and discrete receptors used for creating concentration surface maps.

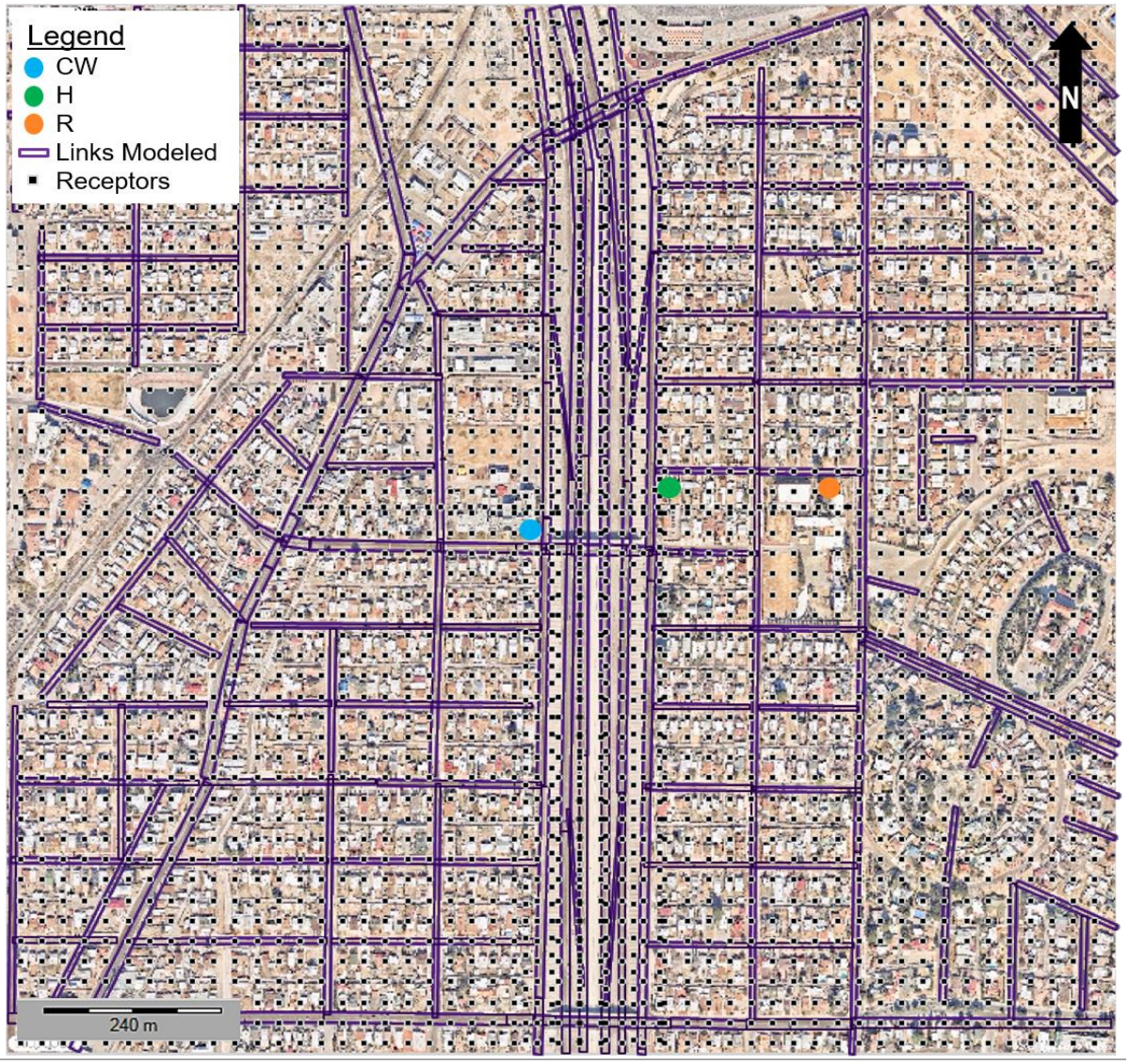

Figure 15 AERMOD Area Source and Receptor Model Set-up

7.2 Background $PM_{2.5}$ Emissions

Air pollutant concentrations near busy highways are composed of the incremental concentrations resulting from traffic emissions and the background concentrations resulting from emissions from other area, mobile, and point sources. Background concentrations should be as representative as possible for the area where the project site is located. Studies have shown that $PM_{2.5}$ measured at near-road air quality monitors is only moderately impacted by traffic emissions. More than 85% of the roadside $PM_{2.5}$ concentrations are believed to be regional urban-scale background concentrations which are primarily caused by ubiquitous urban emission sources (DeWinter et al. 2018).

For an area surrounded by multiple background ambient $PM_{2.5}$ monitors, EPA recommended that the data should be analyzed by statistical or mapping methods to develop an appropriate background concentration estimate for use in the analysis. Li et al. (2019) reevaluated EPA's recommendations and suggested that background concentrations developed by normalized distance-weighted averaging of the data available from all urban-scale background monitors appear to perform better than non-normalized methods with higher accuracy; these findings are shown in Appendix A and Appendix B (Li, Jeon, et al. 2019; Li, Chavez, et al. 2019). Unfortunately, background $PM_{2.5}$ data were only available at 2 sites, UTEP and Ascarate, for this study. While these two sites are equidistant to the study area and could be used to create a background concentration estimate, the Ascarate site is located near a major highway as well as a border crossing, which would not provide a background estimate representative of the area. Therefore, data recorded at the UTEP monitor during the study period was selected to be the hourly background concentrations. Figure 16 displays the $PM_{2.5}$ hourly concentrations of the on-site monitors compared to the two El Paso CAM stations. It can be seen in this figure that the Ascarate site reports much higher $PM_{2.5}$ concentrations than the study sites and the UTEP site.

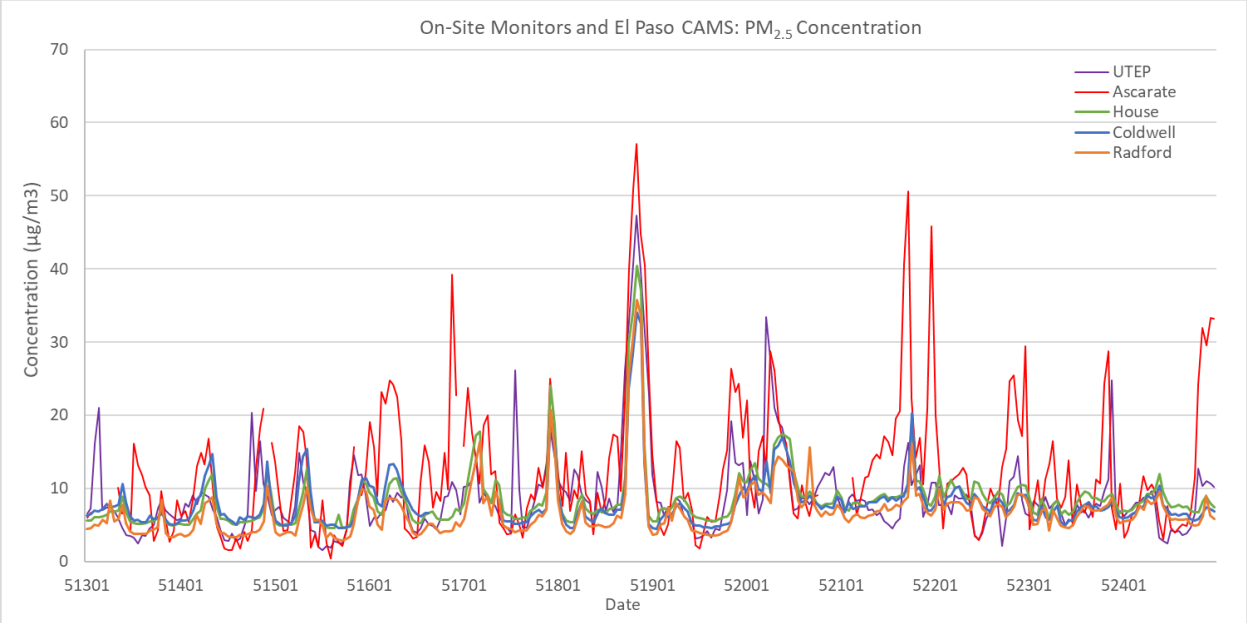

Figure 16 On-site Monitors and El Paso CAMS: $PM_{2.5}$ Hourly Concentrations

Chapter 8: Traffic and Air Quality Results

The following chapter discusses the results of the traffic and air quality data collected during the study. The traffic data was used in conjunction with data from TDM estimates to conduct emission modeling. Observed traffic data was used to calibrate TDM estimates for arterial roads in the study area. Observed pollutant air concentrations were used for comparison with the dispersion model estimates and to apportion the contributions of emissions from the interstate highway as well as the arterial roads.

8.1 Traffic Data Results

8.1.1 Arterial Roads and Local Streets

Traffic volume and vehicle class data was retrieved from the tube counters at three different counting locations in the study area. The locations, as shown in the site map in Chapter 3, are in front of Coldwell Elementary (CW), on Trowbridge Drive (TB), and at Pershing Drive (PS). The devices allow for classification of 13 classes of vehicles, as defined by the Federal Highway Administration (JAMAR Technologies 2011). These classes are also defined by MOVES2014a and are used in calculations for emissions rates at each link. Figure 17 displays the diurnal trends of weekday and weekend traffic volume during the study period at the three counter locations on the arterial roads. It is seen in the figure that the weekday traffic peaked in the morning, at around 7 a.m., and late afternoon around 5 p.m., while the weekend traffic peaked in the early afternoon. The trends at arterial roads, such as PS and TB, agree well with the normalized diurnal traffic pattern reported by Batterman et al. (2015) based on the traffic data from 14 sites over a period of 4 years (Batterman 2015). At a less traveled road near the elementary school, the traffic pattern at the CW site showed significantly lowered traffic than that observed at the other two sites although the peaks are seen to occur consistently in the morning and afternoon rush hours during weekdays and around noon time on weekends.

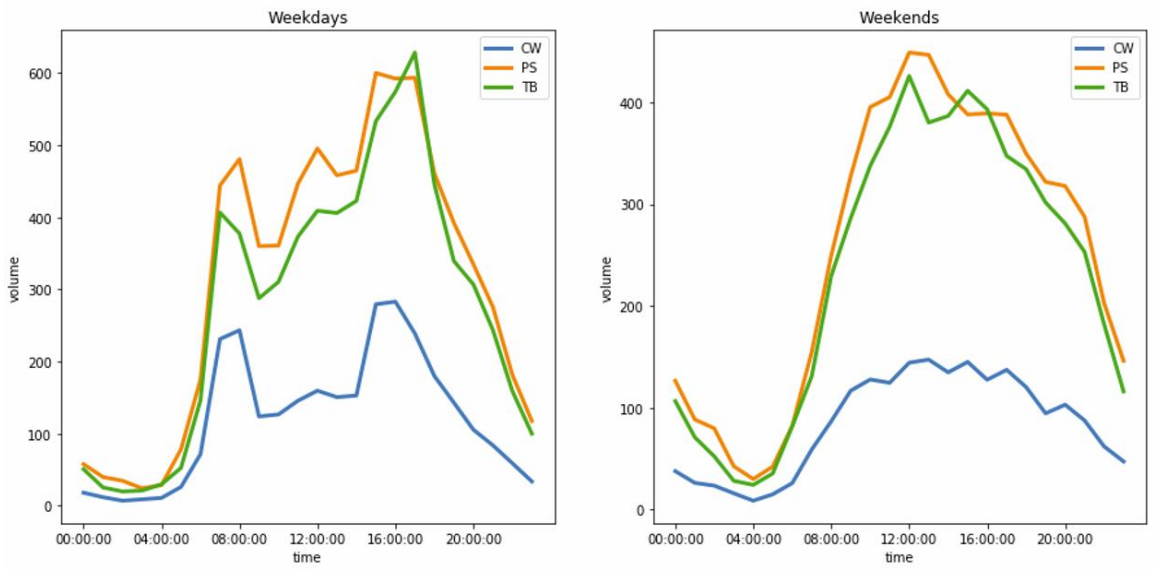

Figure 17 Hourly Average Weekday and Weekend Traffic Volume (number of vehicles)

8.1.2 Interstate Highway

Traffic data for U.S. 54 was obtained through the use of TxDOT videos. Figure 18 displays the diurnal trends of weekday and weekend traffic volume during the study period using the vehicle counts obtained from the highway video recordings. Traffic volume is shown for the northbound (NB) and southbound (SB) lanes. Because of the limitations of the video source, counting was only conducted for the lanes of the highway and not on the frontage roads. Similar diurnal patterns are seen in highway traffic, peaking in the morning hours and evening hours on weekdays at 7 a.m. and 5 p.m., respectively. It is also notable that during the morning peak on weekdays, southbound traffic is higher than northbound traffic and this trend is reversed in the evening peak hours. It is seen in the figure that the southbound traffic during the morning peak is about 50higher than the northbound traffic, but approximately 30% lower in the evening peak.

Finally, it is most important to note that the southbound lanes experience considerably higher traffic volume during most hours on weekdays and weekends, compared to traffic volumes on the northbound lanes. This correlates well with traffic volume estimates provided by the TDM, which also predict higher traffic volume occurring on southbound highway lanes and southbound

frontage roads compared to those northbound lanes. It is important to note that these higher traffic volume estimates occur near the Coldwell Elementary site, located adjacent to the southbound frontage road.

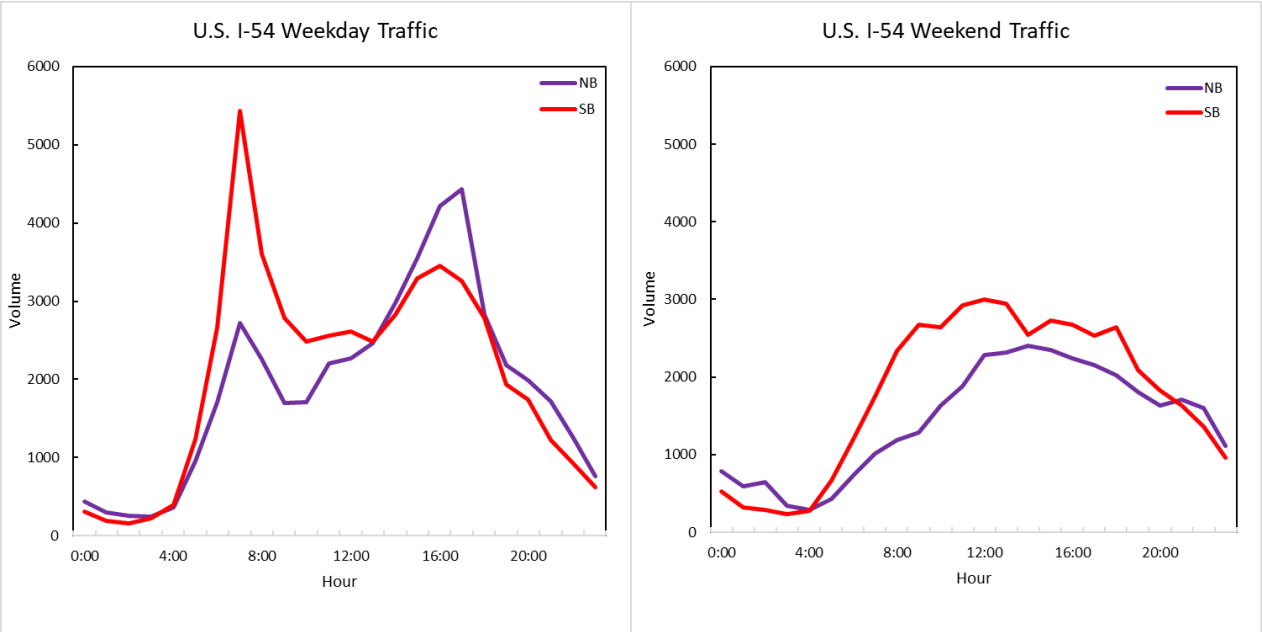

Figure 18 Hourly Average Weekday and Weekend Traffic Volume on U.S. 54

8.2 Air Quality Data Results

The air pollution data collected during this study was processed for comparison of traffic-related air pollution at near-road receptors and in a near-road community. Data was first examined for detectability and completeness to ensure and validate the quality of the data. Values reported by any of the monitors as negative, due to being below the monitors' method detection, were corrected. The reported concentrations can be negative due to zero drift in the electronic instrument output, data logger channel, or calibration adjustments to the data. Slightly negative values were automatically set to 0.5 (i.e., 1/2 of the detection limit of 1 $\mu g/m^3$ for PM or 1 ppb for NO_2 and Ozone), unless the negative values were more than three consecutive values; these were considered missing data. An hour of missing data resulted from the process of downloading the data from the monitors, three times a week. This hour of data was estimated by averaging the two adjacent

values, before and after the missing hour. The finalized air pollution data was also adjusted using the calibration equation for each instrument found from a combination of the pre-and post-calibration data. The detailed analysis and completeness of each set of pollutant data is detailed along with max 1-hr, max 24-hr, and period average in Section 8.2.2.

8.2.1 Monitor-Specific Adjustments for a Period of Time for PM Concentrations

As previously mentioned, the GRIMM Portable Laser Aerosolspectrometer and Dust Monitor was used to read different concentrations of particulate matter. The study used three identical monitors purchased at the same time. During the study, Instrument 3 (CW), located at Coldwell Elementary, began reading values significantly higher than the other two monitors from May 10th to the 17th. The abnormal readings were noted on May 14th during a day of downloading data from all the monitors. Consulting with the monitor manual and GRIMM Technical Support Staff, the high readings were thought to be caused by rotating particles in the laser chamber, resulting in multiple readings of particle counts. The monitor was cleaned with an air duster and set to continue its collection and returned to sensible readings matching the nearby monitors.

Offsets can occur over time even with sophisticated instruments as they are prone to be sensitive. The effects of such offsets can be missed until there are dramatic changes in the instrument readings or changes in correlation with the other instruments. The magnitude of the offset in this case was high but showed a pattern consistent with the other two monitors, indicating a ratio could be found to correct the offset data.

In order to analyze the proper factor to apply, measurements for the hours before and beginning in the offset data, as well as before and beginning with the sensible data after cleaning, were used to determine the ratios for comparisons. By calculating the 1-hour average from the 5-minute averages within the first and last hours of the measurements we received a ratio of 0.02 before the offset, and 0.05 after the ratio for PM_{10}. For $PM_{2.5}$ there was a ratio of 0.2 before the

offset, and 0.5 after the offset. After these ratios were determined, they were applied to the raw original off-set data in order to adjust the values to reasonable concentrations more closely related to the other instruments. An adjustment factor was chosen based on the correlations before and after the offset. Figure 19 and Figure 20 show the PM data before the adjustment whereas Figure 21 and Figure 22 show the adjusted data after the ratio was applied to the offset data. As can be seen in the graphs, the adjusted data follows the same trends for the other instruments once the ratio was applied. Another convincing factor to support the use of the adjusting ratio, includes the observation that instrument 3 (CW) reported consistently lower readings compared to the other instruments throughout the study which can be seen in the adjusted figures.

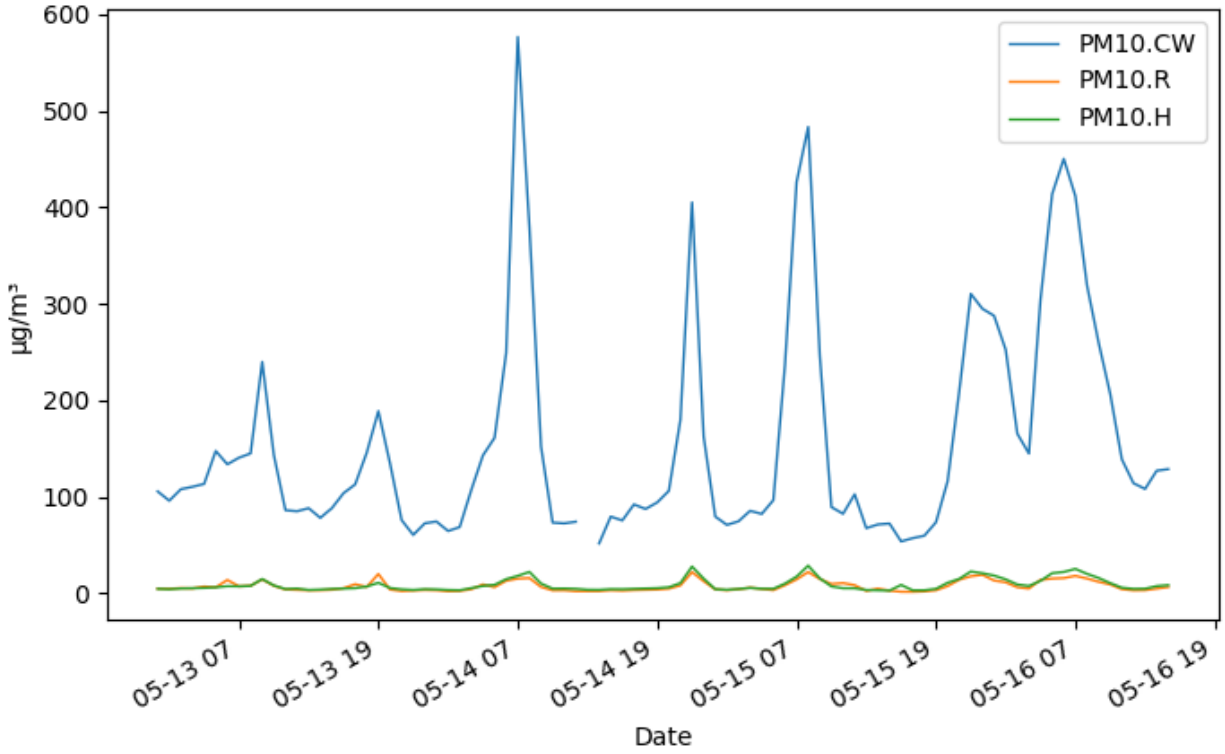

Figure 19 PM_{10} Original Data May 10-14

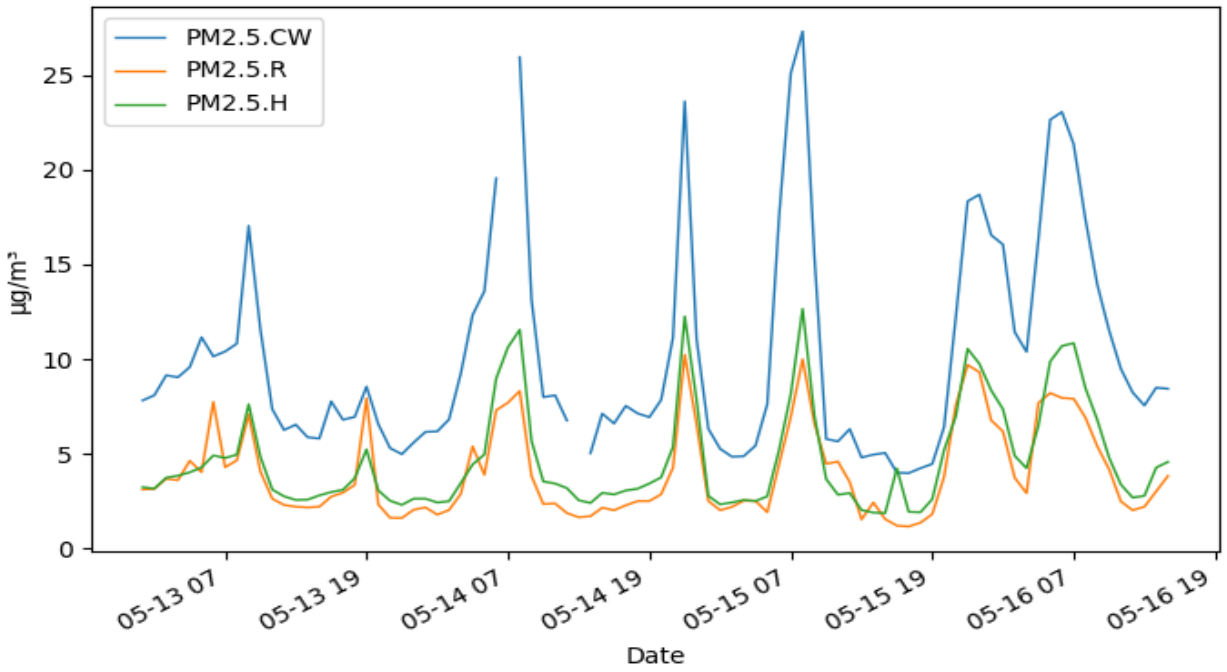

Figure 20 $PM_{2.5}$ Original Data May 10-14

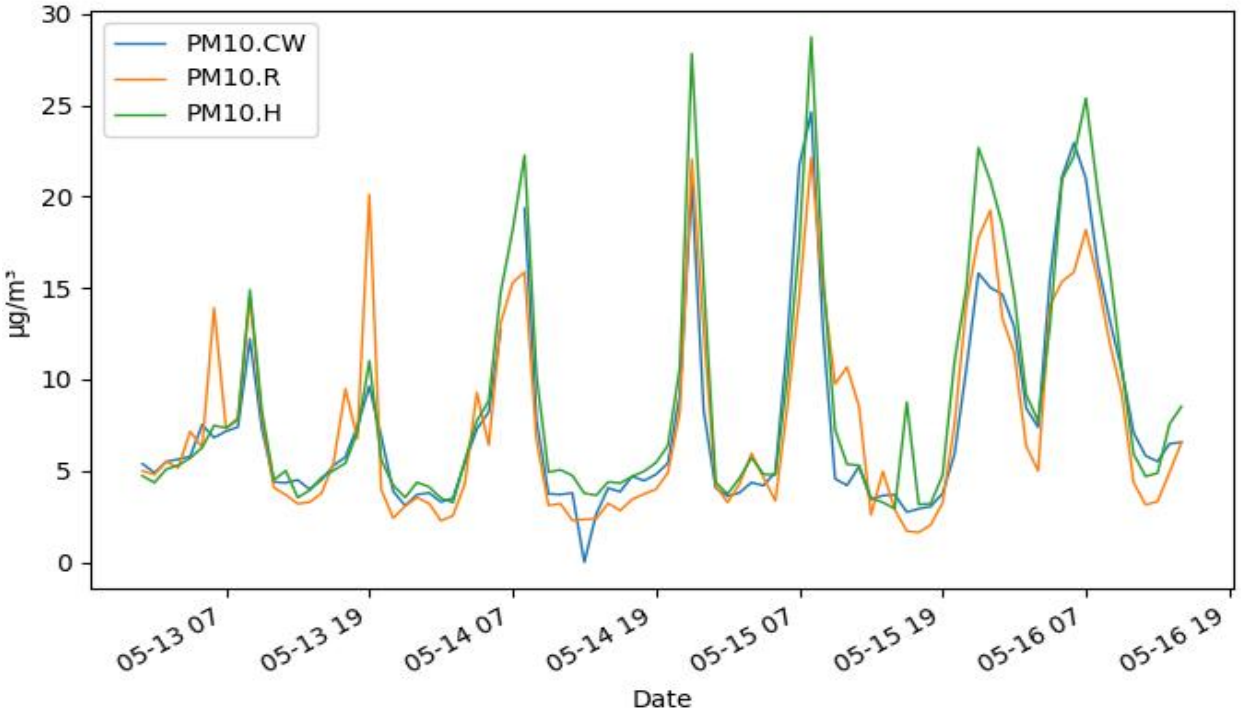

Figure 21 PM_{10} Adjusted Data on Instrument 3 May 10-14

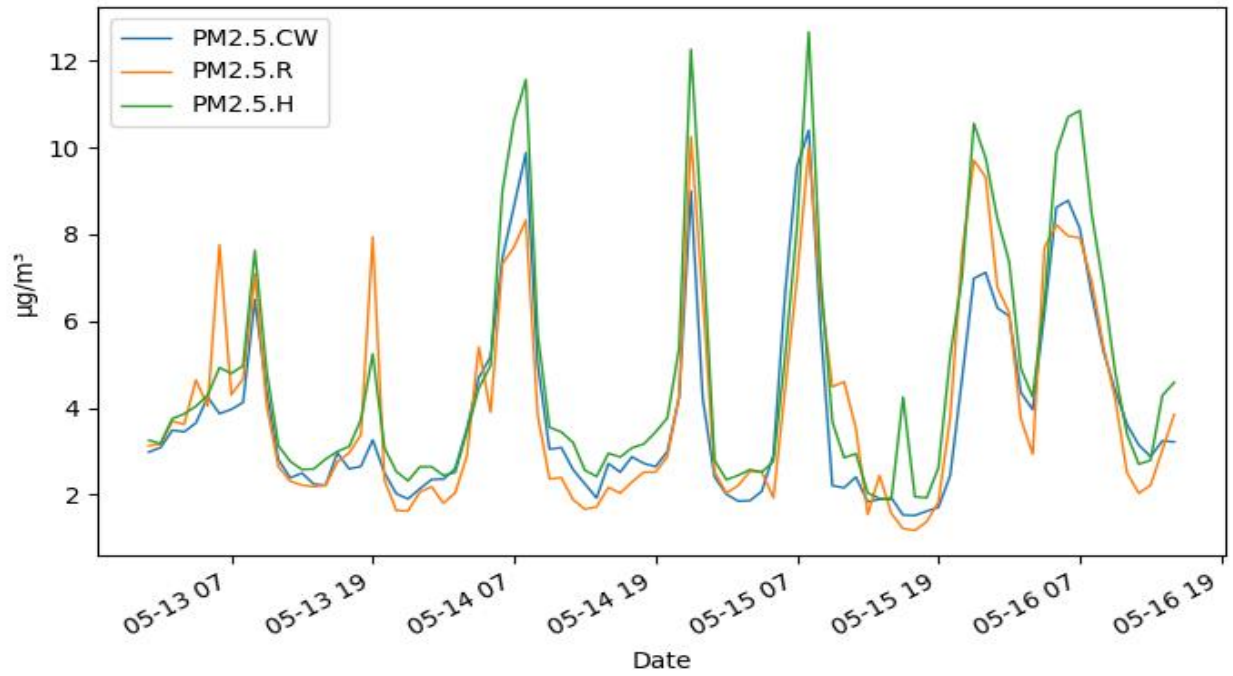

Figure 22 $PM_{2.5}$ Adjusted Data on Instrument 3 May 10-14

8.2.2 Final Air Quality Data Results

Following the adjustment of the offset data for instrument 3 (CW), adjustments were made to all instruments with instrument-specific calibration equations, as well as missing data and negative data adjustments mentioned previously. The air pollution data collected during this study was validated for accuracy and completeness. Values reported by any of the monitors as negative, due to being below the monitors' method detection, were corrected. The reported concentrations can be negative due to zero drift in the electronic instrument output, data logger channel, or calibration adjustments to the data. Slightly negative values were automatically set to 0.5 (i.e., 1/2 of the detection limit), unless the negative values were more than four consecutive values; these were considered missing data. An hour of missing data resulted from the process of downloading the data from the monitors, three times a week. This hour of data was estimated by averaging the two adjacent values, before and after the missing hour. The finalized air pollution data was also adjusted using the calibration equation for each instrument found from a combination of the pre-

and post-calibration data. Details of monitoring results of each pollutant measured during the study are shown in the following sections.

8.2.2.1 $PM_{2.5}$

PM sampling provided continuous and integrated measurements for particle matter. This section details the analysis of observed $PM_{2.5.}$ Continuous measurements provided information on the relationship of vehicle activity and environmental conditions with near-road PM concentrations and characteristics. Figure 23 depicts the hourly time series data from the three monitoring stations for the pollutant $PM_{2.5.}$

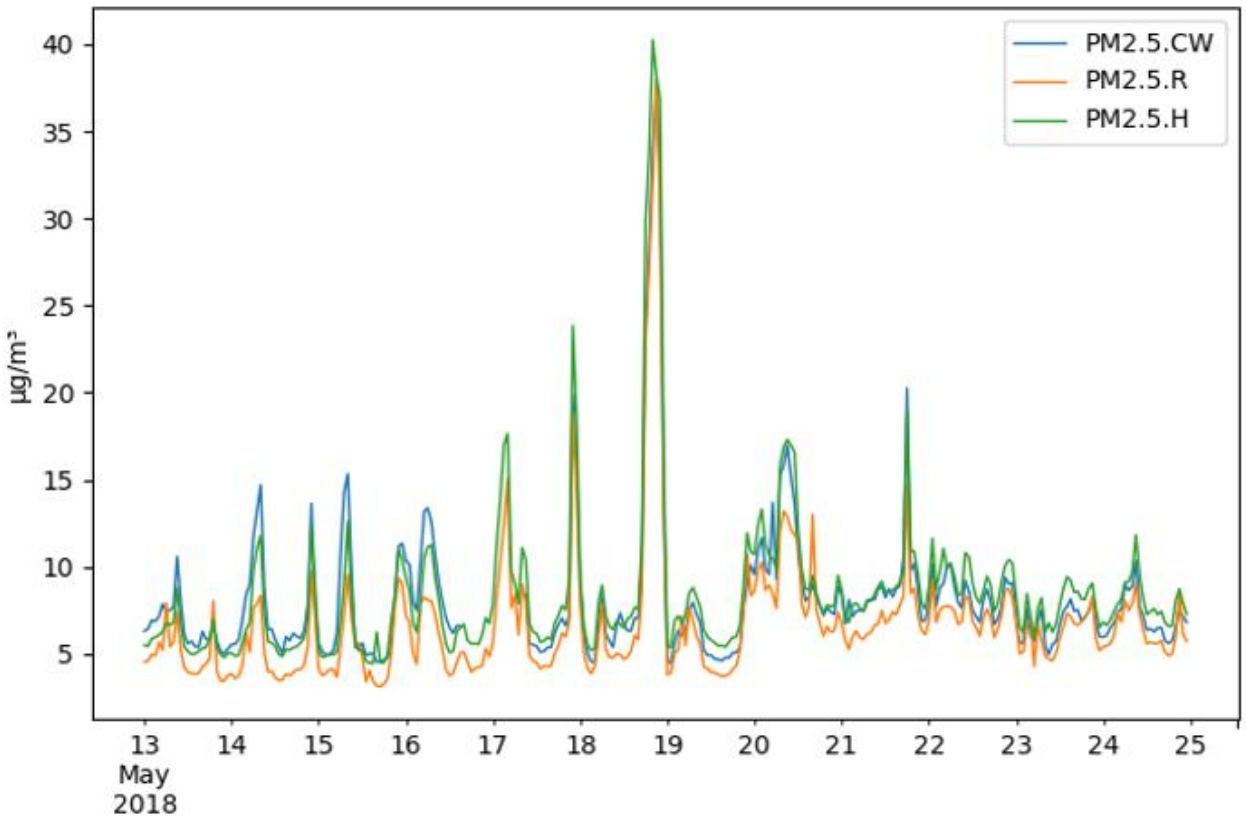

Figure 23 $PM_{2.5}$ Time Series May 13-24

It is noted that the spike of $PM_{2.5}$ observed on May 18th, which occurs at around midnight, could have been caused by a "Motorcycle Run" event wherein a large group of motorcyclists drove through the City of El Paso earlier that day. It is noted that $PM_{2.5}$ shows great temporal variability,

with obvious peaks. In general, it can be seen that the Radford site measures $PM_{2.5}$ values consistently lower than the two near-road sites.

The diurnal patterns of $PM_{2.5}$ pollution data for weekdays and weekends during the study period are shown in Figure 24.

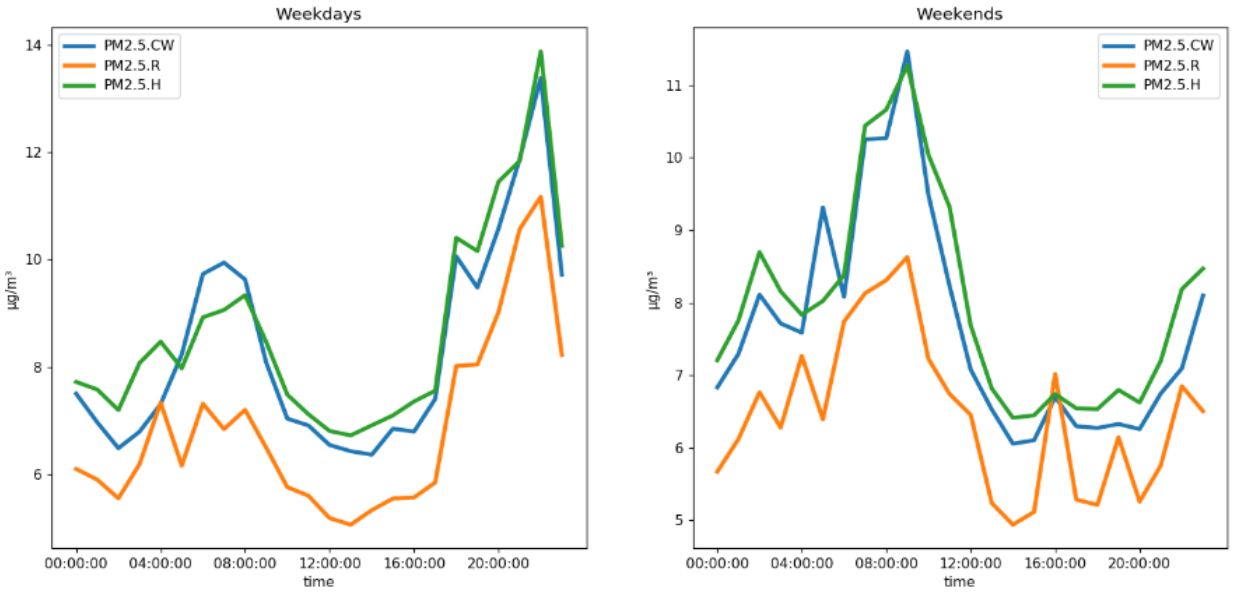

Figure 24 Hourly Average $PM_{2.5}$ Weekday/Weekend

$PM_{2.5}$ has been observed to peak in the morning as well as in the afternoon in El Paso, Texas (Li et al. 2001; 2003). For this near-road community, the morning $PM_{2.5}$ peak coincided well with the morning traffic (Figure 17) but deviated from the early afternoon traffic peak occurring around 4 p.m. The early afternoon traffic peak appears to correlate well with the off-school traffic during weekdays whereas the $PM_{2.5}$ appears to be more correlated to the regional air pollution, indicating that the regional air pollution is likely to be more prevalent for the near-road community, even at locations that are immediately adjacent to an interstate highway. It is also observed that $PM_{2.5}$ values peak in the late-night hours, especially peaking overnight due to reduced atmospheric mixing. As is seen in the time series figure, a $PM_{2.5}$ peak occurs in the early hours of May 20th for the two near-road stations. This occurs on a Sunday during the study period

and therefore results in a peak for 5 a.m. in the diurnal pattern of $PM_{2.5}$ on weekends for the Coldwell site. Examination of the video record shows that a construction rerouting was occurring near the southbound lanes of U.S. 54, closest to the Coldwell site.

Monitored pollutant data for $PM_{2.5}$ is presented in this section, separate from the modeled results. Comparisons with modeled $PM_{2.5}$ results from AERMOD are discussed in Chapter 9. Table 8 shows the maximum 1-hr, maximum 24-hr, and all-period average of $PM_{2.5}$ concentrations monitored at the three sites. The completeness of data for the House, Coldwell, and Radford sites is 100%, 94%, and 100%, respectively. Additionally, the values for $PM_{2.5}$ concentrations at CAMS 12 are also shown in the table.

Table 8 $PM_{2.5}$ Max 1-hr, Max 24-Hr, and Period Average for Monitors (in µg/m3)

$PM_{2.5}$	**Max 1-hr**	**Max 24-hr**	**All Period Average**	**Completeness**
House	40.3	13.5	8.5	100%
Coldwell	37.8	12.1	8.1	94%
Radford	38.0	11.0	6.7	100%
CAMS 12	47.3	16.4	8.8	100%

It is interesting to compare the data observed at the two near-road monitors, Coldwell and House. Coldwell site was 6 meters from the frontage road and approximately 38 meters from the closest lane of the southbound highway whereas the House site was about 8 meters from the frontage road and approximately 42 meters from the closest lane of the northbound highway. Data for the two locations exhibit the characteristics of near-road monitors. Table 8 shows that the difference in $PM_{2.5}$ between the two monitor locations are well within 12%, specifically, the differences are 7%, 12%, and 5% for the maximum 1-hr, maximum 24-hr, and all-period average, respectively. The difference could very well be caused by the direction-varying traffic volume, and time-varying emissions and meteorological conditions. Yet, the difference is practically minimal if one considers all possible uncertainties including upwind-downwind configuration, instrument sensitivity, uncontrollable emission episodes such as emissions from older, poorly

maintained vehicles, cooking, barbeque, among other unreported emissions. Furthermore, these maximum 1-hr, maximum 24-hr, and all-period averages were all indistinguishable from the data measured at the regional monitor, CAMS 12 located at UTEP. For the residential location at Radford, that is 300 meters away from the highway, the maximum 1-hr, maximum 24-hr, and all-period PM2.5 averages are consistently lower than the near-road monitor House by 6%, 21%, and 23%, based on the limited size of the data collected in the study.

8.2.2.2 PM_{10}

Figure 25 depicts the hourly time series data from the three monitoring stations for the pollutant PM_{10}.

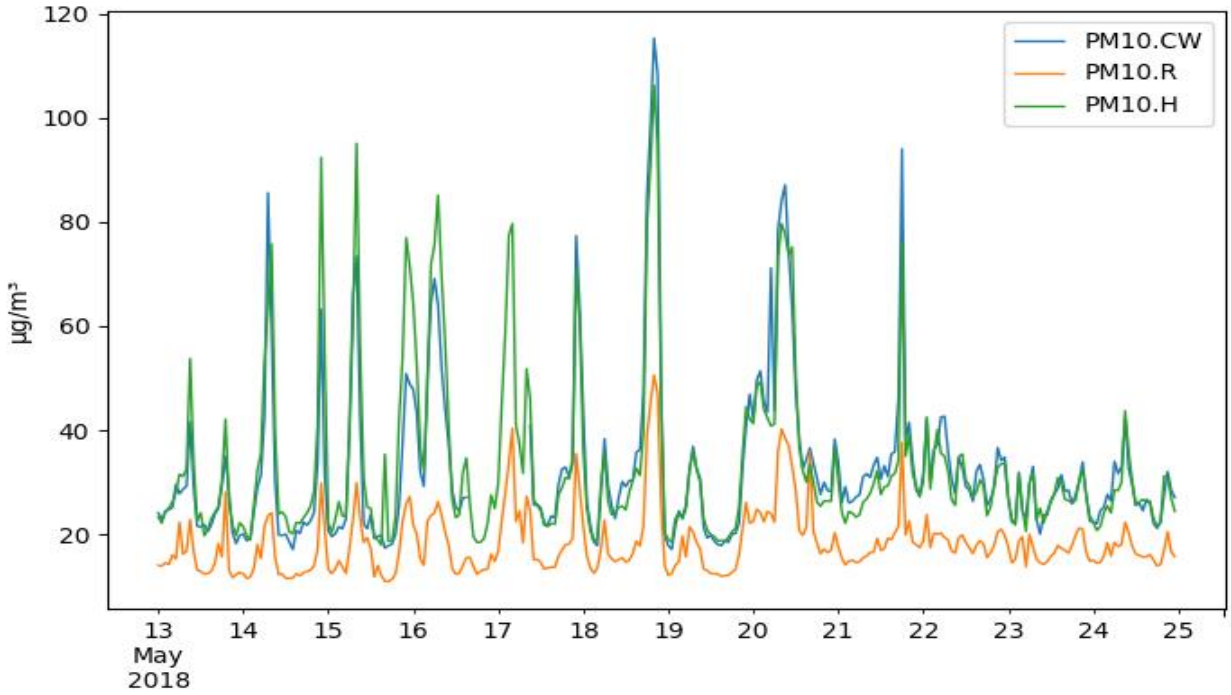

Figure 25 PM_{10} Time Series May 13-24

It is observed that the two near-road sites measure nearly identical concentrations of PM_{10}, while the community air monitor located 300 meters away from the highway, measures concentrations around half as much. This may be in some part due to the actual site set-up. The Radford site was located below a tree and behind a concrete wall, on the school's campus; this

might provide some insulation from high wind patterns which can increase PM_{10} concentrations. The site's proximity to many residential homes in the area may also provide additional insulation from high wind patterns. Similar to the $PM_{2.5}$ data, a comparable peak of PM_{10} is observed during early morning May 20th for the two near-road sites. This could again be possibly due to the higher density traffic observed during the construction rerouting occurring on the highway, specifically the southbound lanes.

Figure 26 shows the diurnal pattern of PM_{10} data for weekdays and weekends during the study period from May 13-May 24.

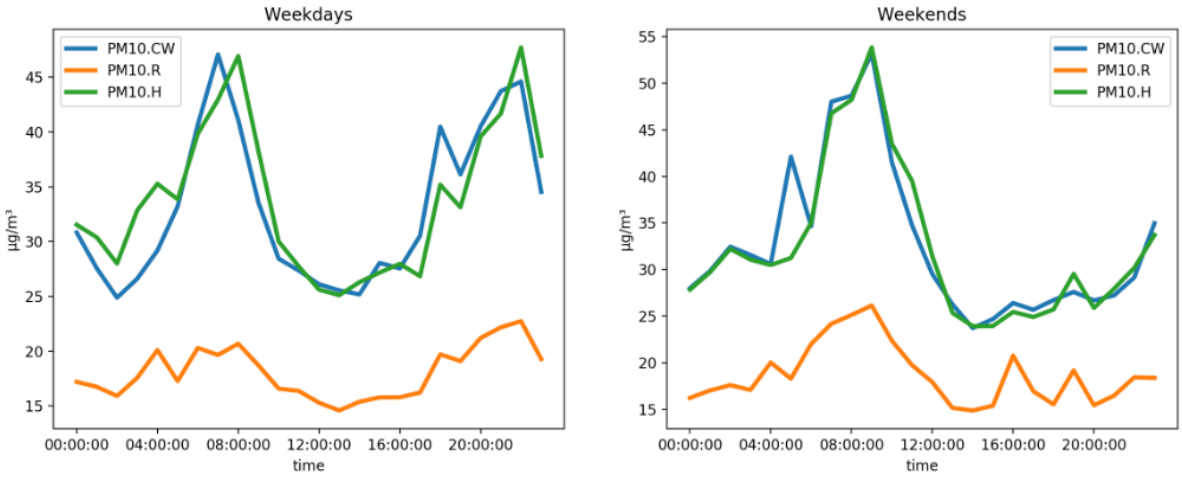

Figure 26 Hourly Average PM_{10} Weekday/Weekend

Similar to the hourly time series data, it can be seen that the community monitor at Radford recorded significantly lower values of PM_{10} than the near-road monitors. All three monitors continue to record PM_{10} at similar weekday patterns, peaking in the morning and evening rush hours. On weekends, it is seen that PM_{10} peaks around 9 a.m. and decreases and remains at lower concentrations the rest of the day. As with $PM_{2.5}$, the higher density traffic peak observed on May 20th at 5 a.m. affects the weekend hourly average and shows a peak at this hour for the Coldwell site.

Table 9 shows the maximum 1-hr, maximum 24-hr, and all-period average of PM_{10} concentrations monitored at the three sites. The completeness of data for the House, Coldwell, and Radford sites is 100%, 94%, and 100%, respectively.

Table 9 PM_{10} Max 1-hr, Max 24-Hr, and Period Average for Monitor (in µg/m3)

PM_{10}	Max 1-hr	Max 24-hr	All Period Average	Completeness
House	106.3	44.5	33.5	100%
Coldwell	115.3	47.5	32.8	94%
Radford	50.6	25.0	18.3	100%

The two near-road monitors show similar values of max 1-hr, max 24-hr, and period average PM_{10}, at only an 8% difference, 7% difference, and 2% difference, respectively. For the residential location at Radford, the maximum 1-hr, maximum 24-hr, and all-period PM_{10} averages are consistently lower than the near-road monitor Coldwell by 78%, 62%, and 57%, respectively.

8.2.2.3. NO_2

Figure 27 depicts the hourly time series data from the three monitoring stations for the pollutant NO_2. As previously mentioned, reported concentrations can be negative due to zero drift in the electronic instrument output, data logger channel, or calibration adjustments to the data, and are thus adjusted to 0.5 (i.e., 1/2 of the detection limit). It can be seen from this time series that the three monitoring sites report similar trends for NO_2 concentrations.

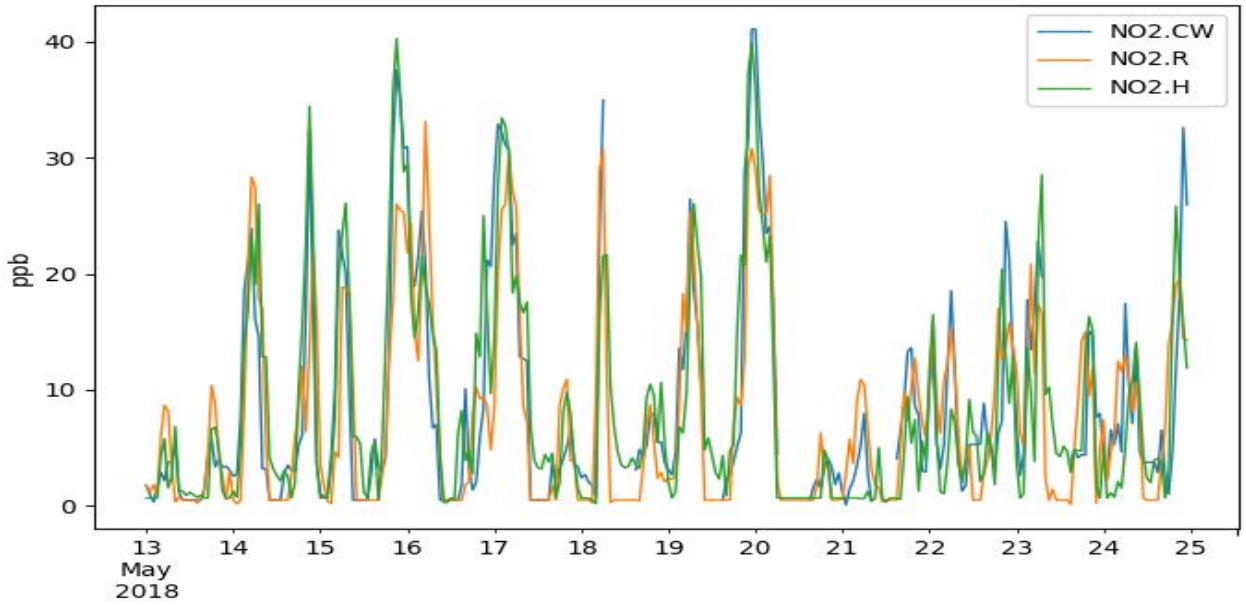

Figure 27 NO_2 Time Series May 13-24

Figure 28 shows the diurnal pattern of NO_2 pollution data for weekdays and weekend during the study period. NO_2 seems to peak in the early morning at around 6 a.m., and in the late evening at around 8 p.m., during weekdays. It is seen that this similar peak pattern occurs on weekends, with more variance seen per hour between the three sites.

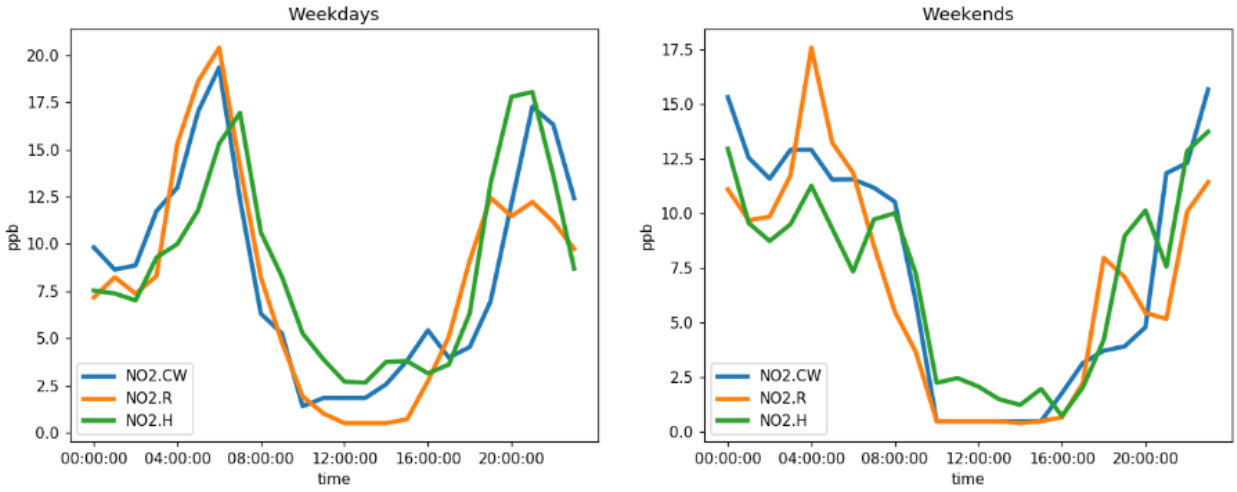

Figure 28 Hourly Average NO_2 Weekday/Weekend

In this study, there seems to be little to no correlation between traffic volume in Figure 17 and the hourly average NO_2 concentrations. According to Kendrick et al. (2015), relationships of traffic volumes and NO_2 vary not only by time of day but also by time aggregation (Kendrick,

Koonce, and George 2015). However, it can be seen that NO_2 levels have a somewhat opposite peak pattern to O_3. This is due to the photochemical reaction between O_3 and nitrogen oxide (NO) reacting readily to create NO_2.

Table 10 below shows the maximum 1-hr, maximum 24-hr, and all-period average of NO_2 concentrations monitored at the three sites. The completeness of data for the House, Coldwell, and Radford sites is 93%, 87%, and 90%, respectively.

Table 10 NO_2 Max 1-hr, Max 24-Hr, and Period Average for Monitor (in ppb)

NO_2	Max 1-hr	Max 24-hr	All Period Average	Completeness
House	40.2	13.0	8.9	93%
Coldwell	41.1	12.9	9.1	87%
Radford	33.1	11.1	8.4	90%

The two near-road monitors show similar values of max 1-hr, max 24-hr, and period average NO_2, at only a 2% difference, 0.4% difference, and 2% difference, respectively. For the residential location at Radford, the maximum 1-hr, maximum 24-hr, and all-period NO_2 averages are consistently lower than the near-road monitor Coldwell by 21%, 15%, and 8%, respectively.

8.2.2.4. Ozone

Figure 29 depicts the hourly time series data from the three monitoring stations for the pollutant O_3. Monitored O_3 values were the most consistent across the sites. Ozone values for the three monitoring stations were nearly identical. Ozone is a secondary pollutant with precursors including NOx and VOCs. Included in this figure are the O_3 concentrations observed at CAMS12.

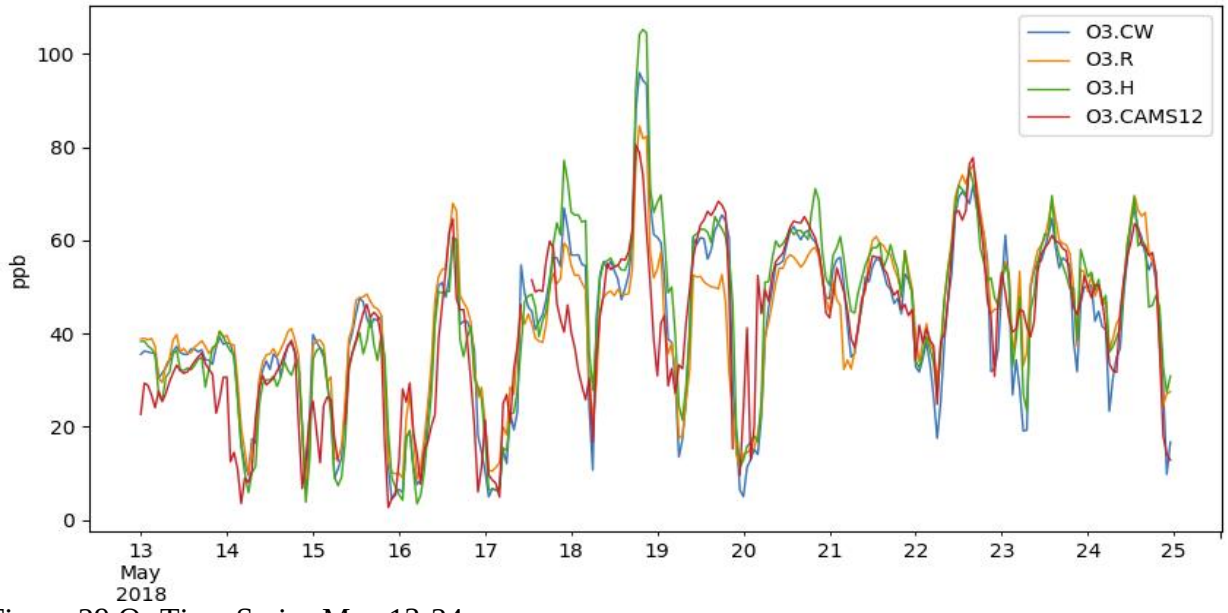

Figure 29 O_3 Time Series May 13-24

The diurnal patterns of O_3 pollution data for weekdays and weekends are shown during the study period Figure 30. Ozone pollutant concentrations correlate very well at the three sites. This indicates that ozone is a more homogenous and ubiquitous pollutant throughout the city, with not much variation regarding distance to high-traffic sources. It can also be seen that the measurements at CAMS 12 also trend closely to the O_3 concentrations observed at the three monitors.

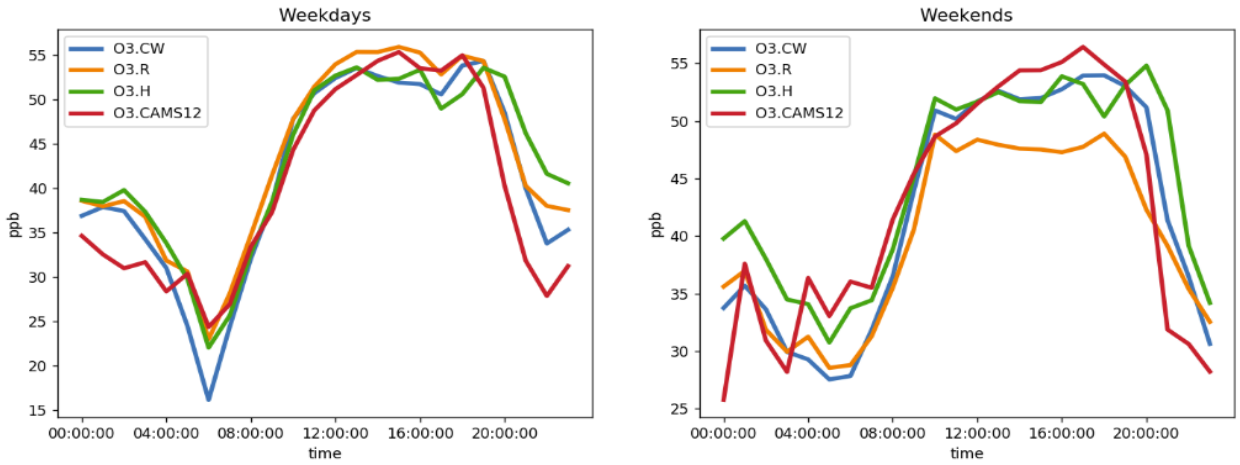

Figure 30 Hourly Average O_3 Weekday/Weekend

Ozone begins to peak slowly as the morning sun rises, but continues throughout the day peaking during the daytime. This is due to the photochemical formation of O_3. The levels of O_3

are influenced by prevailing levels of precursors like NOx. Similar to the time series plot of O_3, the diurnal patter of the concentrations observed at CAMS 12 match well with the diurnal trend of concentrations observed at the three monitoring sites.

Table 11 shows the maximum 1-hr, maximum 24-hr, and all-period averages of O_3 concentrations monitored at the three sites. The completeness of data for the three sites was 100%. Also included in this table are the values for O_3 measured at CAMS 12.

Table 11 O_3 Max 1-hr, Max 24-Hr, and Period Average for Monitor (in ppb)

O_3	Max 1-hr	Max 24-hr	All Period Average	Completeness
House	105.2	63.5	43.4	100%
Coldwell	95.9	57.9	41.5	100%
Radford	84.5	53.5	42.6	100%
CAMS 12	80.4	52.1	40.7	100%

The two near-road monitors show similar values of max 1-hr, max 24-hr, and period average O_3, at only a 9% difference, 9% difference, and 5% difference, respectively. For the residential location at Radford, the maximum 1-hr, maximum 24-hr, and all-period NO_2 averages are consistently lower than the near-road monitor House by 22%, 17%, and 2%, respectively. For this pollutant, it is seen that the all period average between the three sites remains the most consistent, in addition to matching well with values at CAMS 12.

Chapter 9: Results and Discussion

This chapter discusses the $PM_{2.5}$ concentration estimates provided from dispersion modeling using AERMOD and the MOVES emissions factors. The model estimates are then combined with background $PM_{2.5}$ concentrations to create total modeled estimates. These model estimates are compared to the monitored data for $PM_{2.5}$, presented in the previous chapter.

9.1 AERMOD Model Predictions

$PM_{2.5}$ concentration estimates resulting from traffic emissions from U.S. 54 were generated using AERMOD. Concentration surfaces were generated using discrete receptors as well as grid receptors in order to evaluate the impacts of traffic emissions on the community using the AERMOD concentration estimates.

9.1.1 Near-Road Receptors and Off-Highway Receptor

The $PM_{2.5}$ concentrations predicted by AERMOD for the maximum 1-hr, maximum 24-hr, and all-period averaged $PM_{2.5}$ concentrations at the three monitor sites are listed in Table 12 (Columns 4, 7, and 10). The magnitudes of the model prediction for the all-period average do not appear to be dominated by the prevailing westerly winds (see the windroses in Figure 2). Instead, the upwind Coldwell site shows higher concentrations than the downwind House site. This is likely due to the higher traffic estimates for the southbound gateway and highway. The detailed temporal variability can be observed in the highway traffic volume data shown in Figure 18 in Chapter 8. Observing the total hours measured in the study period, the northbound lanes experience an average volume of 1,760 while the southbound highway experience an average volume of 1,949. During the study period, the northbound highway experienced a total volume of 292,095 vehicles, while the southbound highway experienced a total volume of 323,574 vehicles. An approximately 65% decrease in the all-period averaged $PM_{2.5}$ concentration predictions is observed between the House

site and the Radford site which is situated on the same side of the highway as the House site, but 300 meters off the highway.

9.1.1.1 Time-series Prediction of $PM_{2.5}$ Concentrations

The $PM_{2.5}$ concentration time series estimates for the three sites can be seen in Figure 31. It is observed that for the time period between May 20th at 7 a.m. and May 21st at 7 p.m., $PM_{2.5}$ concentrations estimates were consistently lower at the House and the Radford receptors. These estimates are likely due to the high easterly winds during these hours.

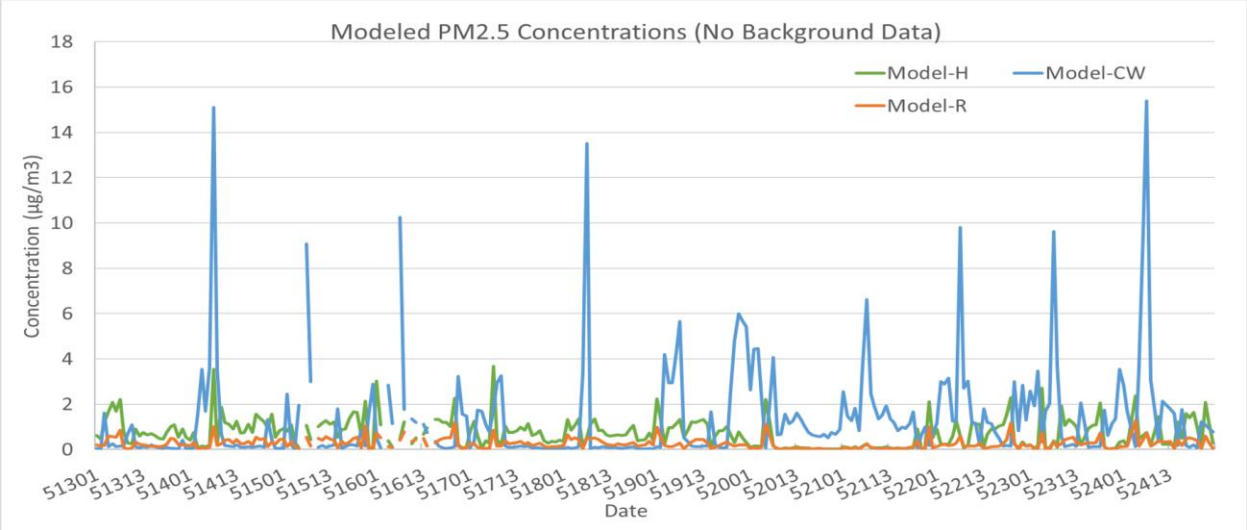

Figure 31 Modeled $PM_{2.5}$ Concentration

Observing the meteorological conditions during the peaks hours estimated for the Coldwell site, it can be seen that for all predicted estimates of $PM_{2.5}$ greater than 3 µg/m^3 (32 hours), wind speeds are less than 2.7 m/s. More importantly, it is also noted that wind direction during these hours is east to west, positioning the Coldwell site downwind of the highway. These meteorological factors, combined with the higher ERs found on the southbound lanes of the highway, yield these higher estimates at the Coldwell site compared to the other near-road site.

Higher $PM_{2.5}$ concentrations at Coldwell were consistently predicted than at the other two sites, due to the previously mentioned high traffic volume occurring on the southbound highway.

Additionally, this site is located 6 meters from the southbound frontage road. As previously mentioned, these southbound lanes experience higher traffic volume than the other near-road site near the northbound highway lanes, and this is most significant during the morning peak hours.

It is also observed that many of the highest estimates at the Coldwell site occurred at 7 a.m. The higher concentration estimates obtained during this hour, in spite similar traffic volumes in the following hours, is due to the vertical temperature profile in the early morning hours (Turner 1994). The urban option within AERMOD was modified, beginning with version 11059, to address potential issues associated with the transition from the nighttime urban boundary layer to the daytime convective boundary layer. Prior to version 11059, the enhanced dispersion due to the urban heat island during nighttime stable conditions was ignored once the rural boundary layer became convective. This could result in an unrealistic drop in the mixing height for urban sources during the morning transition to a convective boundary layer, which could contribute to overly conservative concentrations for low-level sources under such conditions (U.S. EPA 2004). This correction to avoid overly conservative concentrations could possibly result in overestimating values at the hour of the transition from the nighttime urban boundary layer to the daytime convective boundary layer, which in the case of El Paso occurs at hour 7.

When examining the diurnal patterns of the modeled results, it can be seen that certain patterns occur between these predictions and observed results of the $PM_{2.5}$ concentrations collected at the three sites. Figure 32 depicts the hourly average of $PM_{2.5}$ concentrations modeled by AERMOD shown with the monitored results at the House site. Model results are measured on the left side axis and the monitored results are measure on the right side of each graph. While modeled results are largely affected by changing wind directions and wind speed at each hour, the morning peaks and midday to afternoon lows with evening peaks, are observed in the modeled results, which are

similar to the monitored results for weekdays. Weekend model results show similar patterns to monitored weekend results, with peaks in the late morning and gradual decreasing trend for the rest of the day. The R^2 value for the House site compared to the monitored results is 0.0279.

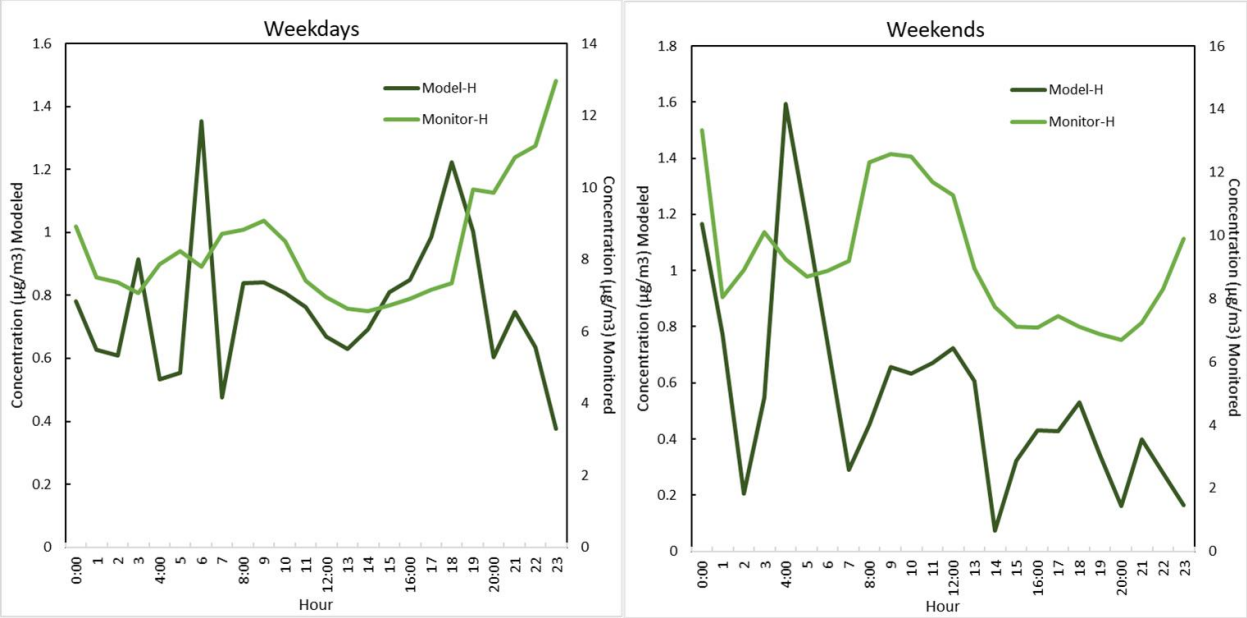

Figure 32 Hourly Average $PM_{2.5}$ Weekday/Weekend at House: AERMOD results and Monitored Concentrations

Figure 33 depicts the hourly average of $PM_{2.5}$ concentrations modeled by AERMOD shown with the monitored results at the Coldwell site. Model results are measured on the left side axis and the monitored results are measure on the right side of each graph. It can be seen from the modeled results that the model is able to capture some of the diurnal patterns observed in monitored data. During weekdays, modeled results show similar patterns of peaks overnight with a particular peak value occurring at 10 p.m. As previously mentioned, high values are observed in the modeled results for the Coldwell site at 7 a.m. due to high easterly winds during these hours. The R^2 value for this site compared to the monitored results is 0.006. This indicates very low correlation with monitored results.

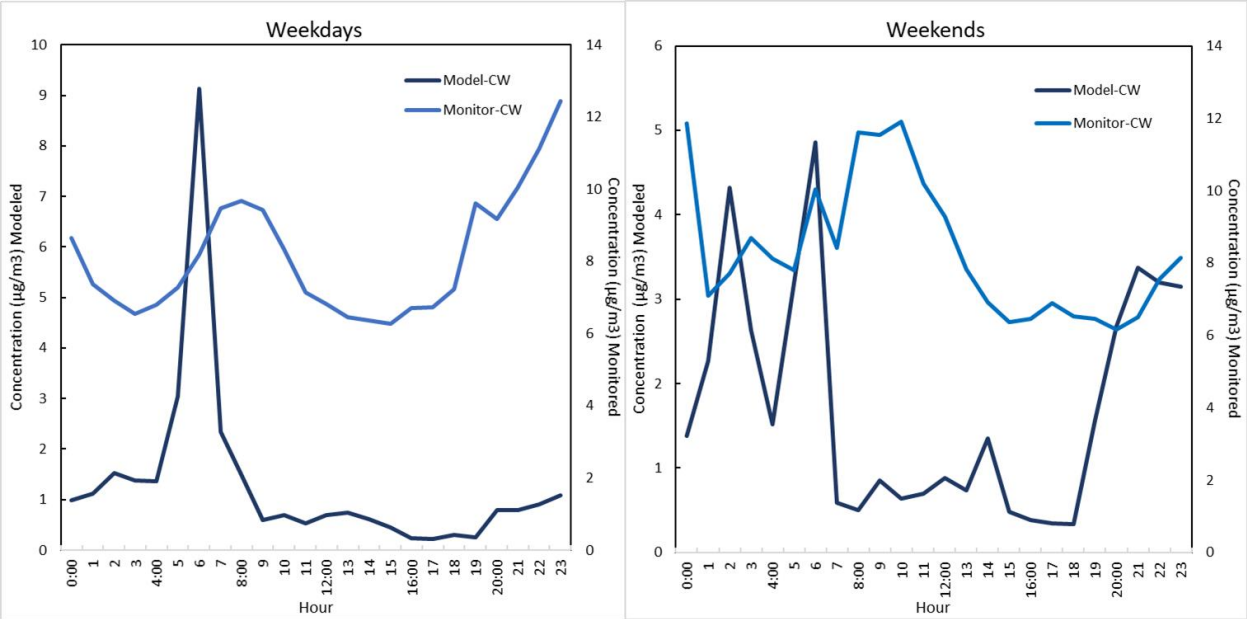

Figure 33 Hourly Average $PM_{2.5}$ Weekday/Weekend at Coldwell: AERMOD results and Monitored Concentrations

Figure 34 depicts the hourly average of $PM_{2.5}$ concentrations modeled by AERMOD shown with the monitored results at the Radford site. Model results are measured on the left side axis and the monitored results are measure on the right side of each graph. Weekday modeled results follow similar patterns as the monitored results, peaking in the morning around 7 a.m., dipping around 12 p.m. and peaking again in the evening starting at 5 p.m. The R^2 value for this site compared to the monitored results is 0.0145.

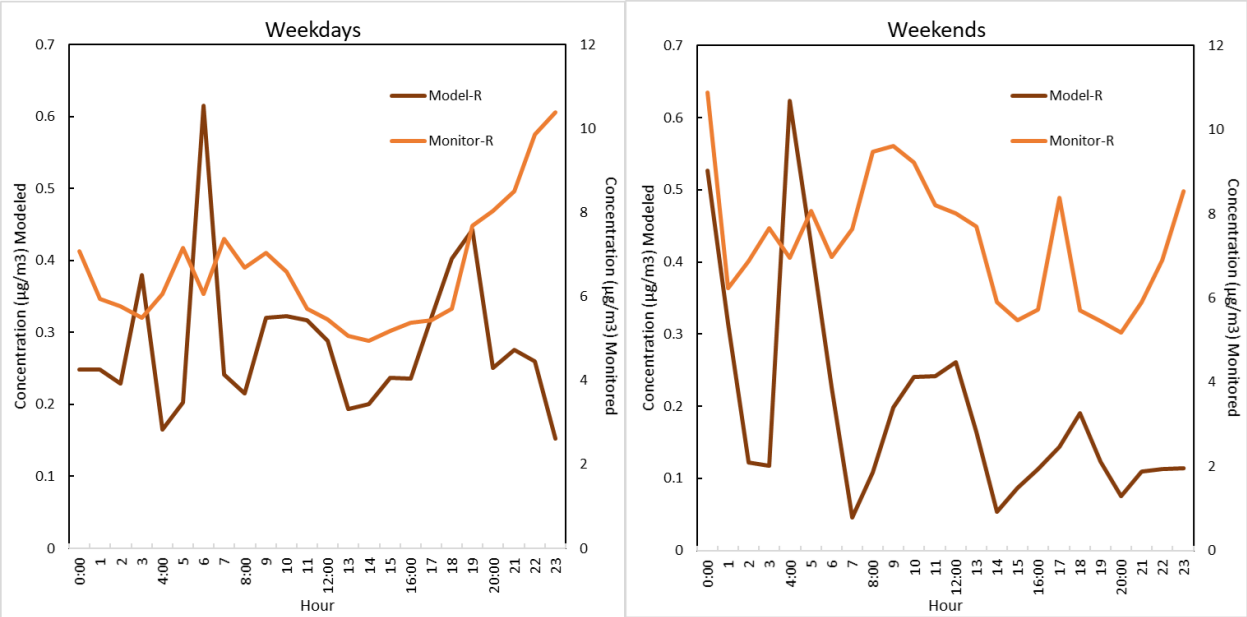

Figure 34 Hourly Average $PM_{2.5}$ Weekday/Weekend at Radford: AERMOD results and Monitored Concentrations

Because the monitored $PM_{2.5}$ concentrations are largely driven by background levels in the environment, not captured by the model, it is therefore acceptable that model results will not follow the same diurnal patterns. Additionally, the modeled results are driven by the wind speed and wind direction at each hour, it is clear the reasons for differences in $PM_{2.5}$ concentration estimates between the two near-road sites which are located on opposite sides of the highway.

9.1.1.2 Maximum 1-hr Concentration Predictions

Table 12 shows the maximum 1-hr, maximum 24-hr, and all-period averaged $PM_{2.5}$ concentrations monitored at the three sites (Columns 2, 5, and 8) and predicted by the AERMOD model (Columns 3, 7, 10). The column labeled "Modeled +BG" depicts the maximum 1-hr, maximum 24-hr, and all-period averaged $PM_{2.5}$ concentrations of the model estimates with the added hourly background values obtained from CAMS 12 at UTEP.

Table 12 $PM_{2.5}$ Max 1-hr, Max 24-Hr, and Period Average for Monitor, Model+Background, and Model Results (in $\mu g/m^3$)

	Max 1-hr			Max 24-hr			All Period Average		
	Monitored	Modeled +BG	Modeled	Monitored	Modeled +BG	Modeled	Monitored	Modeled +BG	Modeled
House	40.3	47.7	3.7	13.5	17.1	1.0	8.5	9.5	0.7
Coldwell	37.8	47.3	15.4	12.1	17.1	2.2	8.1	10.1	1.4
Radford	38.0	47.5	1.3	11.0	16.7	0.4	6.7	9.1	0.3

In examining the maximum 1-hr concentration predictions, it can be seen that the model predicts this value for the Coldwell site significantly higher than that at the House site. It is important to notice that these maximum 1-hr values (observed and predicted at the same site) do not necessarily occurred concurrently. This is unfortunate but realistic due to the uncertainties such as local episodic emissions, upset meteorological conditions, unexpected/unusual traffic congestion, that could not be effectively modeled in a computer simulation. A maximum 1-hr concentration should be viewed as a possible worst-case exposure concentration that could occur under the worst-case meteorological condition but under a routinely predictable emission scenario. It may serve well as a guideline value in regulatory compliance or policy making but may not correctly reflect the actual maximum concentration occur at a specific time in a community. It is also interesting to observe that the max 1-hr value is almost the same for all three sites, with a 1% difference, when the regional background value was added to the modeled value.

9.1.1.3 Maximum 24-hr Concentration Predictions

Table 12 shows that the maximum 24-hr concentrations for all 3 sites decrease significantly from the maximum 1-hr concentrations. The modeled concentration at the Coldwell site is seen to be approximately twice higher than that modeled for the other near-road House site. As discussed previously, higher southbound traffic in the morning, closer location to the interstate highway, and overall higher emission rate during the day all contribute to this discrepancy. The regional background concentration continued to prevail in the community where, on average, the

background concentration for the respective day when the maximum 24-hr concentration was predicted at a site was higher than the value measured in the community, whether near-road or in the residential area, by 2.6 to 5.3 μg/m^3 which practically obscured the pollution contribution from the traffic emissions in the community.

9.1.1.4 All-period average

Background concentrations for the region during the study period appear to be closer to that observed in the community, to be within 0.3 to 2.1 μg/m^3 difference. The predicted concentration in the residential area is seen to be much lower than that observed near busy highway. An almost 2-fold difference in the all-period average for the 2 near-road sites is seen in Table 12. Contribution of emissions from traffic, distance to the nearest highway, and atmospheric stability and low-wind conditions during high emission hours appear to be more critical than the prevailing wind direction in determining the pollution concentration at the near-road sites. Furthermore, emissions from the interstate highway as well as the local arterial roads contribute only less than 14% to the overall prediction of the near-road concentration, or less than 17% of the monitored concentration. The traffic emission contribution decreases further away from the highway, Table 12 shows that traffic emissions contribute to only 3 % of the predicted value in the residential area located approximately 300 m off the highway, or less than 4.5% of the monitored concentration. Differences between the modeled total (modeled + BG) and monitored concentration decreases significantly as the averaging time increases. The modeled total concentration over predicts the actual monitored data by 7.4 ~ 9.0 μg/m^3 for the maximum 1-hr average but converges and slightly over predicts the actual value by only 1.0 ~ 2.4 μg/m^3.

9.1.2 Cross-highway Concentration Distribution

The dispersion of $PM_{2.5}$ concentrations from the highway can also be analyzed with the placement of receptors at increasing distances from the highway, specifically, in the direction

perpendicular to the highway. A general rapidly decreasing trend of the predicted $PM_{2.5}$ concentrations with increasing distance from the nearby highway, was observed. Figure 35 shows the dispersion of the pollutant $PM_{2.5}$ away from the highway, where the concentration of airborne particles was characterized as a function of distance from U.S. 54, with negative values representing the distance increasing to the west of the highway. These results suggest that the vast majority of dispersion occurs within 200 meters of the highway. A secondary minor peak appearing to the west of the highway (Figure 35) is attributed to an arterial road running parallel to the highway, which can be seen modeled in the concentration maps. This road is adjacent to Coldwell Elementary at around 400 meters away from the highway. The extra emissions contributed from the traffic on this arterial road contribute to the small peak seen west of the highway at around 400 meters.

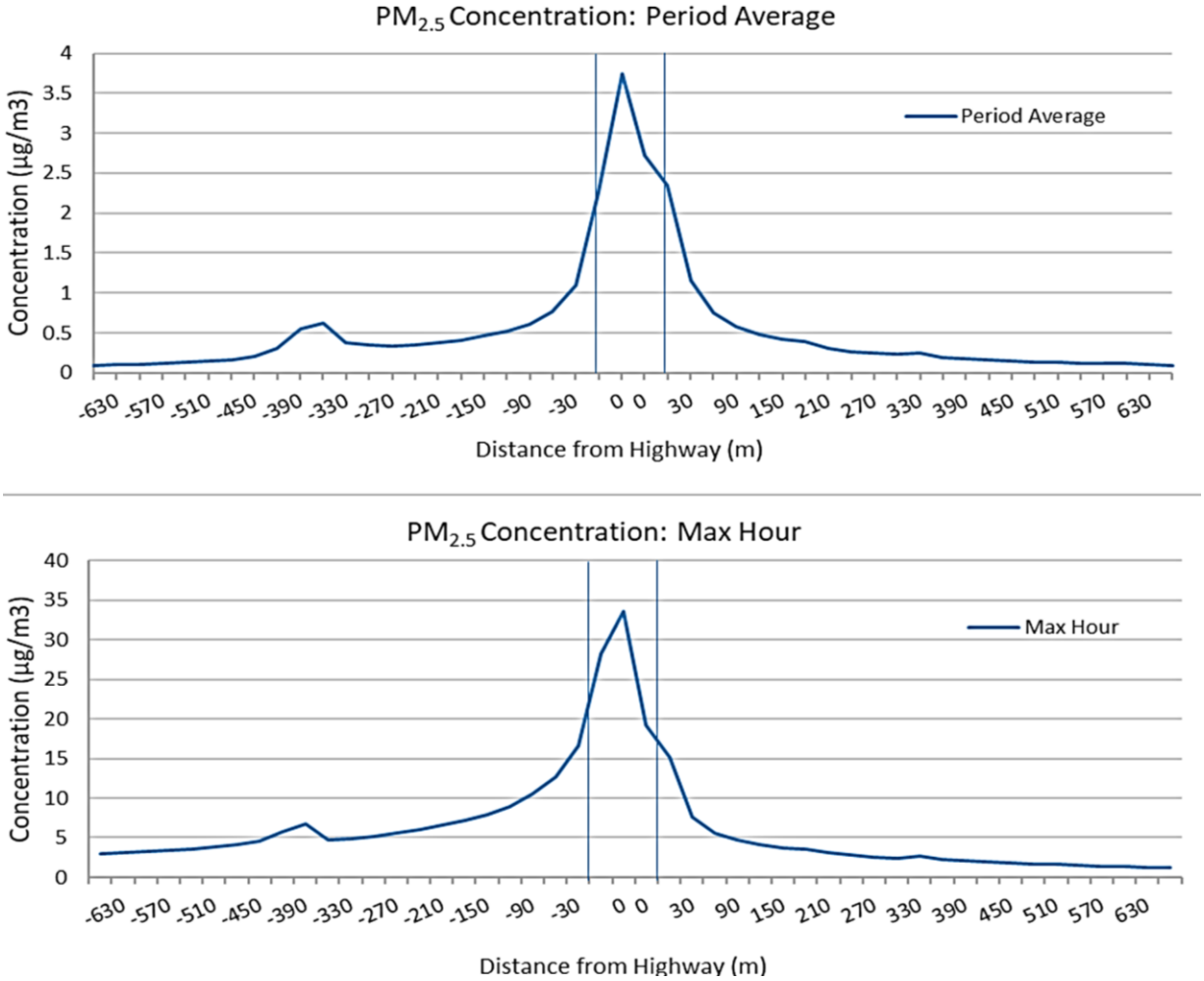

Figure 35 $PM_{2.5}$ Dispersion as a Function of Distance from the Highway

Karner and coworkers (2010) analyzed 41 roadside monitoring studies between 1978 and 2008 and concluded that almost all pollutants decay to background levels at a distance 115 m to 570 m from the edge of the road and the decay rate varies from one pollutant to another except $PM_{2.5}$ which achieved the background level by 990 m without any trend of rapid decrease from the road edge (Karner et al. 2010). However, Venkatram et al (2013) showed that the concentration of an inert pollutant decays rapidly to less than 1/5 of its initial strength in 100 m in the direction normal to the roadway (Venkatram et al. 2013). The discrepancy in $PM_{2.5}$ distribution off a highway could be attributed to many uncontrollable factors, such as the existence of sound walls

for at-grade freeways, elevated or filled section of a freeway, canopy vegetation, classification of atmospheric stability condition, existing local and regional point sources, among others. The decay rates observed in our current study correlate well with analysis and estimates from previous studies (Yazdi, Delavarrafiee, and Arhami 2015; Zhu et al. 2002; Clements et al. 2009). These results could be useful in determining a buffer area around highways to not include residential buildings and business activities on highway adjacent.

9.1.3 Community Exposure to Traffic Emissions

It is observed that the links with greater traffic volumes produce the greatest concentrations of $PM_{2.5}$, especially the southbound lanes on U.S. 54. The spatial distributions of $PM_{2.5}$ concentrations in the community at the maximum 1-hour, maximum 24-hour average, and all-period averages are shown in Figure 36. These figures provide a clearer illustration of the $PM_{2.5}$ exposure in the community due to the traffic emissions in the study area. All three time-averaged $PM_{2.5}$ concentrations decrease rapidly from the roadway towards the residential community. Arterial roads with higher traffic volume, such as Pershing and Trowbridge also display higher estimates of $PM_{2.5}$ concentration. The actual $PM_{2.5}$ concentrations near these arterial roads may be higher, but could not be shown, than what are presented in the figures because the grid receptors are spaced at an increment that does not provide the necessary resolution in the concentration surfaces. Nevertheless, the rapid decrease of $PM_{2.5}$ concentrations off the arterial roadway is expected to be similar to what has been observed along the busier interstate highway U.S. 54. It is also noted that the concentration surfaces for maximum 1-hr as well as maximum 24-hr averaged PM2.5 concentration represent only the maximum concentrations occurred at a location and these short-term time-averaged maximums at different locations may not occur at the same time.

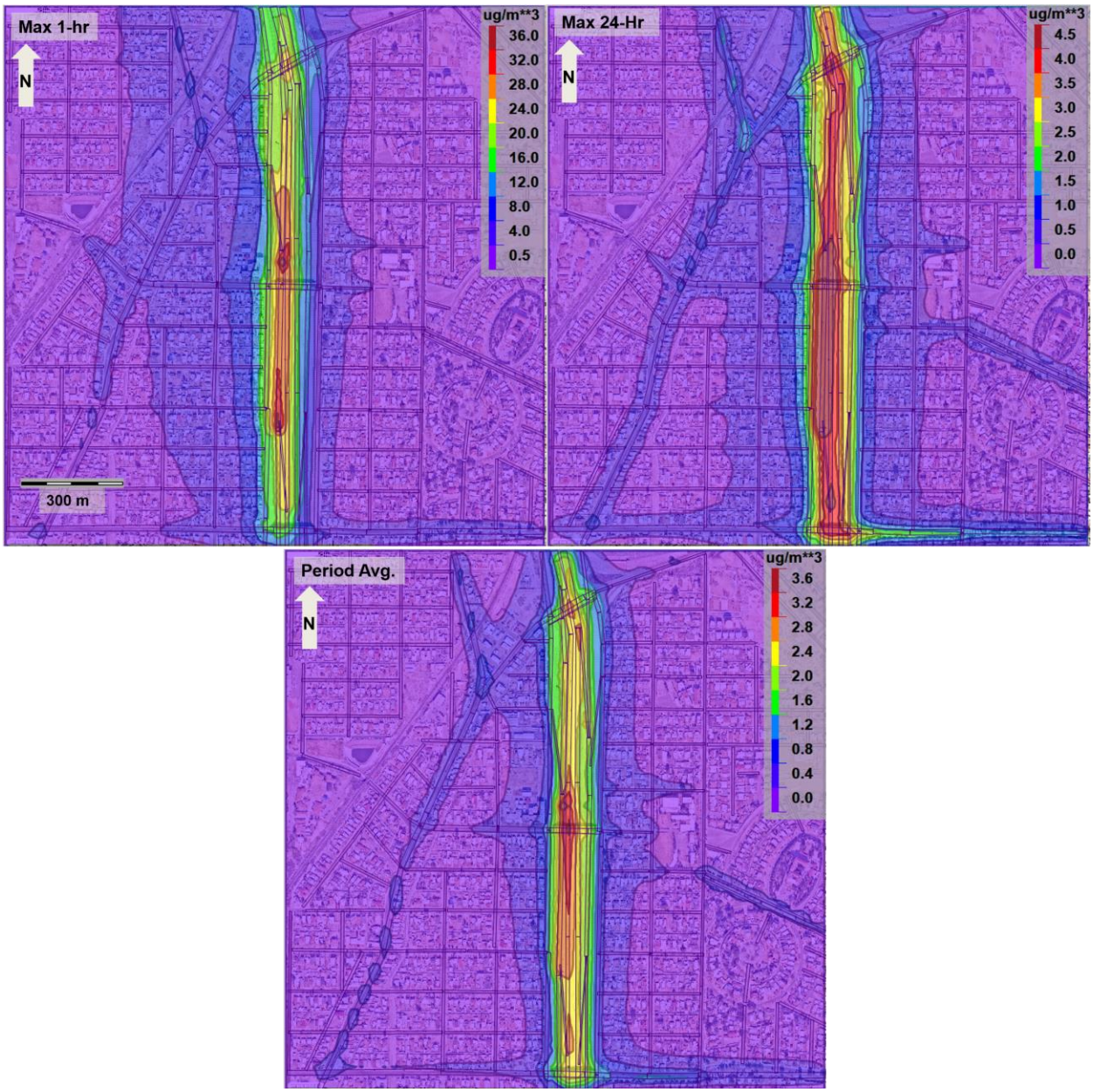

Figure 36 Max 1-hr, Max 24-Hr, and Period Average $PM_{2.5}$ Concentration Estimates

Figure 37 shows the time-evolving $PM_{2.5}$ concentrations modeled by AERMOD at four different peak hours, shown clockwise they represent 12 a.m., 7 a.m., 1 p.m., and 5 p.m. on Friday May 18, 2018. It is observed that $PM_{2.5}$ concentrations are higher during times of higher traffic volume, occurring at 7 a.m. and 5 p.m. The prevailing wind directions during these peak hours (and most of the day) are from the west to east or west to south east; the wind speed range throughout this particular day is from 5.8 to 9.8 m/s. Table 13 shows the predicted $PM_{2.5}$

concentrations at the three sites for these peak hours with the measured wind speed and wind direction at each hour. Here it can be seen that the high wind speeds during these hours correspond well with the dispersion seen in the concentration maps. The modeled estimates correspond well with these wind conditions, as the bulk of the emissions are observed to occur to the east of the highway.

Table 13 Modeled estimates at three sites for different peak hours

Hour	House	Coldwell	Radford	Wind Speed	Wind Direction
12 a.m.	0.35	0.02	0.16	7.2	290
7 a.m.	1.13	0.05	0.48	5.8	290
1 p.m.	0.61	0.09	0.19	9.8	240
5 p.m.	0.86	0.10	0.27	7.6	260

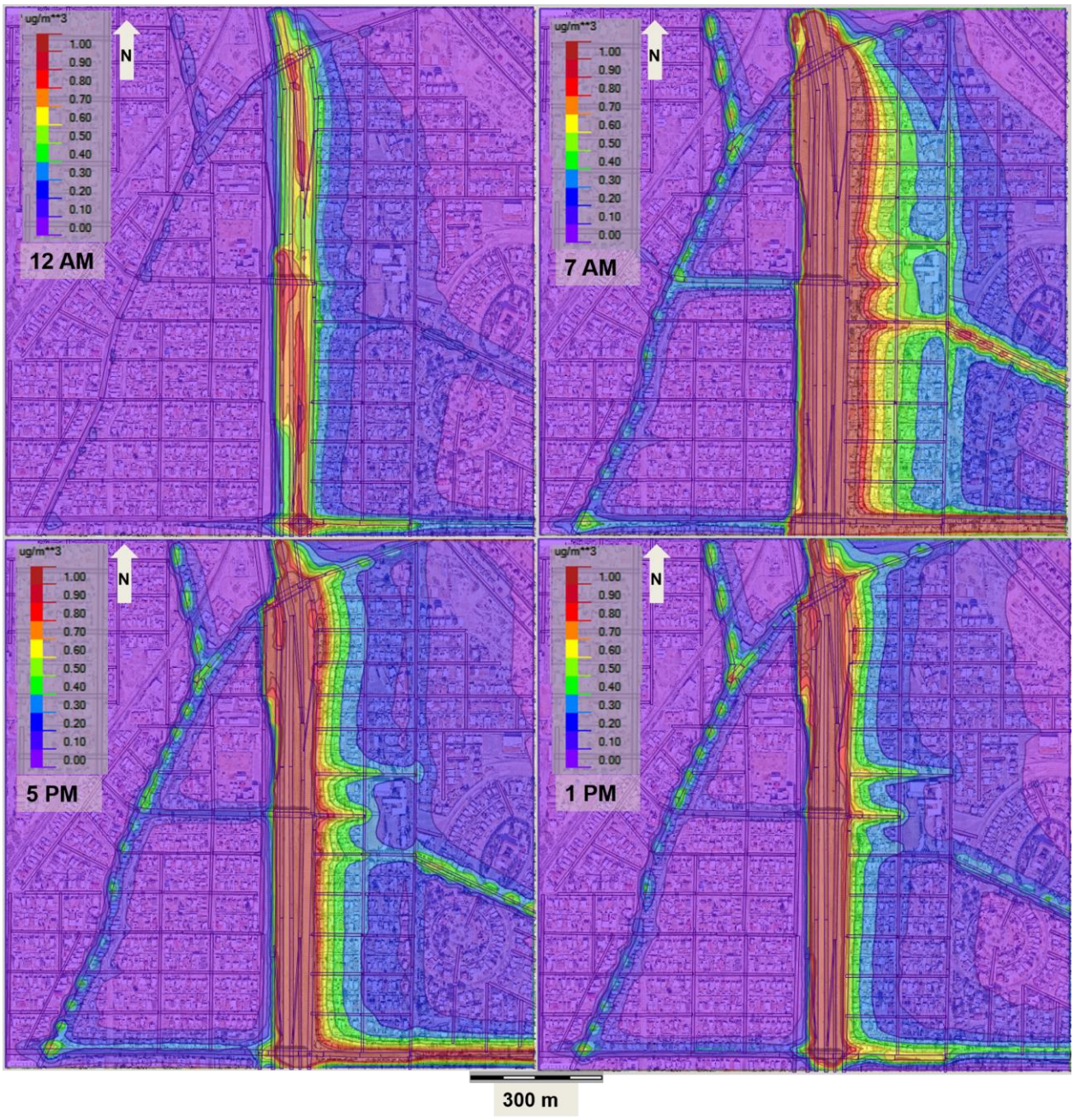

Figure 37 $PM_{2.5}$ Hourly Concentrations at Different Peak Hours, Friday May 18th

Wind speeds during the AM and PM peak period hours are 5.8 and 7.6, respectively, with west to east wind directions. The dispersion of $PM_{2.5}$ is therefore shown to be mostly on the right, easterly side of the highway. The levels of $PM_{2.5}$ are also observed to be higher at these hours, coinciding with higher ERs from higher traffic volume during these hours. These figures further emphasize the findings that AERMOD results are driven by hourly wind direction and wind speed.

9.2 Background Concentration

Air pollutant concentrations near busy highways are composed of the incremental concentrations resulting from traffic emissions and the background concentrations resulting from emissions from other area, mobile, and point sources. Background concentrations should be as representative as possible for the area where the project site is located. Studies have shown that $PM_{2.5}$ measured at near-road air quality monitors is only moderately impacted by traffic emissions. More than 85% of the roadside $PM_{2.5}$ concentrations are believed to be regional urban-scale background concentrations which are primarily caused by ubiquitous urban emission sources (DeWinter et al. 2018).

As previously mentioned the data recorded at the UTEP CAM site is used to represent the hourly background concentrations. Background $PM_{2.5}$ concentrations should be as representative as possible for the area where the study site is located. Ideal background concentrations for a near-road site without the influence of traffic emissions are rarely available. For an area surrounded by multiple background ambient $PM_{2.5}$ monitors, the EPA recommended that the data should be analyzed by statistical or mapping methods to develop a background concentration for use in the hot-spot analysis. In most cases, the simplest approach will be to use data from the monitor closest to and upwind of the project area with the following considerations (U.S. EPA 2010b):

- Similar characteristics between the background site and the study area
- Distance between the study area and the background site
- Meteorological conditions between the study area and the background site

The UTEP site was selected based on the above considerations. However, the site is 5 miles off the study area and may possess different topologic characteristics and inevitably adds unquantifiable uncertainties to this study.

9.3 Modeled-to-Monitored Comparison

The total $PM_{2.5}$ exposure in the community was assessed by adding the AERMOD modeled concentration estimates to the selected background concentrations. The modeled predictions were first compared to the $PM_{2.5}$ pollutant data measured at the three locations in Table 12. It appears that the model over-estimates the maximum 1-hr and 24-hr $PM_{2.5}$ at the near-road sites and the off-highway residence by at least 16% and 21%, respectively. The model accuracy improves for longer term average. It is important to note that this comparison involves the addition of the hourly background concentrations obtained from the UTEP CAM site. Furthermore, it is seen that this "background" value is often higher than even the observed concentrations at the two near-road sites.

Table 14 shows the maximum 1-hr and 24-hr $PM_{2.5}$ comparisons between the model results and the monitored values, examining according to when these values occur for the AERMOD results without added background. For example, at the House site, the model predicts the highest maximum 1-hr $PM_{2.5}$ concentration as 3.7, occurring on May 17th at 7 a.m., then the background and monitored values for this hour are used to examine the ratios between the model results and the modeled + background results (Column 7) and the modeled + background results and monitored values (Column 8). Finally, the percent difference between modeled + background results to monitored concentrations, is presented in Column 9, which in this example is 46%.

The maximum 1-hr AERMOD prediction at Coldwell occurs on May 24th at 7 a.m., however this estimate alone is almost twice the background value of 8.7. This results in an observed percent difference of the modeled + background estimate to the monitored value at Coldwell to be 168%.

Table 14 $PM_{2.5}$ Max 1-hr, Max 24-Hr, and Period Average for Monitor, Model+Background, and Model Results (in $\mu g/m^3$) in accordance to AERMOD (Modeled) results

	Max 1-hr							
	Modeled	**Date**	**BG**	**Modeled +BG (Total)**	**Monitored**	**Modeled / Modeled +BG**	**Modeled +BG / Monitored**	**% Diff Modeled +BG: Monitored**
House	3.7	51707	9.4	13.1	9.0	28%	146%	46%
Coldwell	15.4	52407	8.7	24.10	9.0	64%	268%	168%
Radford	1.3	52404	7.3	8.5	6.1	15%	139%	39%
	24-Hr							
House	1.0	514	7.6	8.5	6.7	12%	127%	27%
Coldwell	2.2	524	6.8	7.6	7.1	29%	107%	7%
Radford	0.4	516	7.0	7.3	5.5	5%	133%	33%
	All Period							
House	0.7	All Period	8.8	9.5	8.5	8%	112%	12%
Coldwell	1.4	All Period	8.8	10.1	8.1	13%	125%	25%
Radford	0.3	All Period	8.8	9.1	6.7	3%	135%	35%

Table 15 shows the maximum 1-hr and 24-hr $PM_{2.5}$ comparisons between the model results and the monitored values, examining according to when these values occur for the AERMOD results with added background values. For example, the maximum 1-hr concentration predicted by the model plus the background value as 47.7 for the House site occurring on May 18th at 10 p.m. Examining this hour, it is seen that the background concentration amounts to 99% of the total concentration prediction. Here the percent difference between the modeled + background concentration and the monitored value is 25%, for all three sites.

Table 15 $PM_{2.5}$ Max 1-hr, Max 24-Hr, and Period Average for Monitor, Model+Background, and Model Results (in $\mu g/m^3$) in accordance to Total Modeled Results (Modeled +BG)

	Max 1-hr							
	Modeled +BG	**Date**	**BG**	**Modeled**	**Monitored**	**Modeled / Modeled +BG**	**Modeled +BG / Monitored**	**% Diff Modeled +BG: Monitored**
House	47.7	51822	47.3	0.4	38.2	1%	125%	25%
Coldwell	47.3	51822	47.3	0.02	37.8	0%	125%	25%
Radford	47.5	51822	47.3	0.2	38.0	0%	125%	25%
	24-Hr							
House	17.1	518	16.4	0.8	13.5	4%	126%	26%
Coldwell	17.1	518	16.4	0.8	12.1	4%	142%	42%
Radford	16.7	518	16.4	0.3	11.0	2%	152%	52%

From these two tables, it is seen that when considering maximum values in accordance to the AERMOD estimates, without the added background hourly values, the percent differences between the modeled + background estimates and the monitored values are generally less, except for the maximum 1-hr modeled at Coldwell.

Figure 38 shows the modeled-to-monitored time series comparisons of $PM_{2.5}$ emissions during the study period. The figures are divided into two different weekly periods starting at Sunday May 13th through May 19th, followed by Sunday May 20th through May 24th. The elements labeled beginning with "Model" are those modeled through AERMOD; i.e. "Model-H" are the AERMOD modeled results for the receptor located at the House. The modeled results include the background concentration estimates provided by the El Paso CAM station at UTEP, located about 4 miles away from the study area.

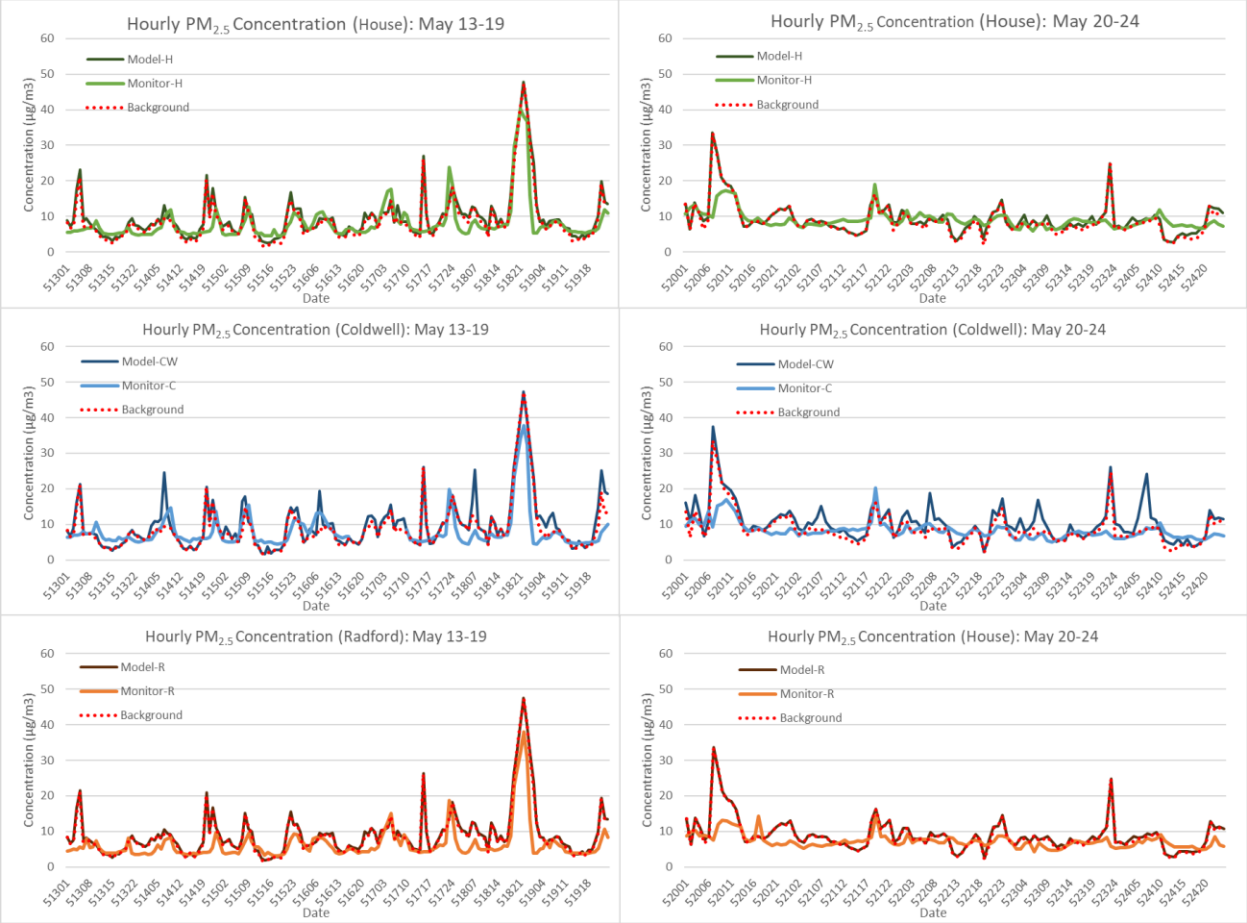

Figure 38 Comparison of Model Results and On-Site Monitoring: Hourly $PM_{2.5}$ Concentrations

As previously discussed, background concentrations account for a significantly portion of the $PM_{2.5}$ exposure near or off highway. Local traffic impacts account only approximately 10% of the total exposure. That is to say, the modeled results shown in the figures are driven largely by the regional background concentrations. It is noted that the spike observed on May 18th, which occurs at around midnight, could have been caused by a "Motorcycle Run" event wherein a large group of motorcyclists drove through the City of El Paso earlier that day.

9.4 Considering the Community Monitor (Radford) as Background

Because background values observed at CAMS12 are repeatedly higher than the "near-road" monitors, other avenues of estimating or obtaining background estimates are deliberated. While this monitor is located 700 m from the closest interstate highway I-10, it is located at a busy

intersection observing high volume traffic from the University and other nearby businesses. Therefore, it is possible to consider that a community monitor near the study area can be representative as a background monitor. Figure 39 shows the comparison of modeled results with the added background concentrations (considering Radford a background monitor), compared with the monitored results at the House site. Here it is easier to see where the model "under predicts" particularly from May 20th to May 21st.

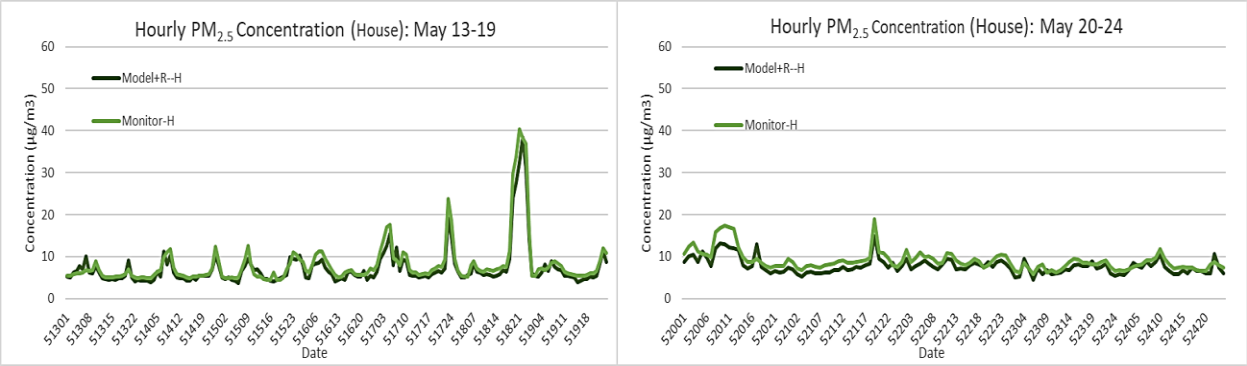

Figure 39 Comparison of Model Results with alternate BG and On-Site Monitoring: Hourly $PM_{2.5}$ Concentrations

Figure 40 depicts the comparison of modeled results with the added background concentrations (considering Radford a background monitor), compared with the monitored results at the Coldwell site. Because this new background estimates amount to less than the monitored results at the near-road sites, it is noticeable where the model estimates for the Coldwell site amount to higher hourly concentrations, for example starting on May 22nd to May 24th at 10 a.m.

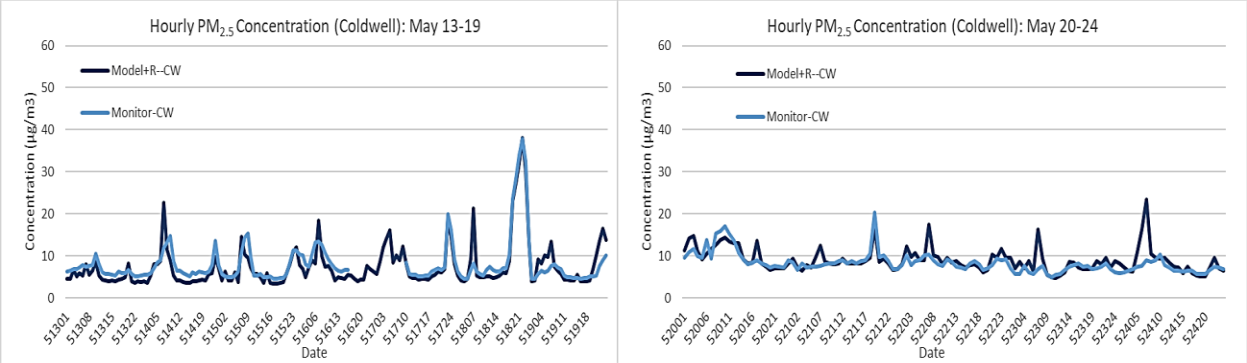

Figure 40 Comparison of Model Results with alternate BG and On-Site Monitoring: Hourly $PM_{2.5}$ Concentrations

Using a background estimate that is lower than the near-road concentrations can be more realistic than using a background monitor such as CAMS 12, which is located in a high traffic area. This comparison indicates the need for establishing a more adequate background monitor, especially for studying near-road concentration exposures.

9.5 Traffic Emission Impacts to the Community

Included in the analysis using AERMOD, each source was placed into three different "source groups" which allow the model to consider the impact of each source group on the receptors. These groups were "Arterial", "Gateway" and "Highway". Table 16 shows the percent of contribution to $PM_{2.5}$ by each source group on the three receptors.

Table 16 $PM_{2.5}$ Contribution to Receptors by Type of Source

	House	Coldwell	Radford
Arterial	11.6%	13.4%	49.4%
Gateway	1.4%	4.6%	2.0%
Highway	87.1%	82.1%	48.5%

This study observed that for the two near-highway receptors, the contribution to $PM_{2.5}$ concentrations was greater than 80%, whereas contribution from the highway was around 50% on the Radford receptor, located 300 meters away from the highway. The receptor at Coldwell received around 5% of the emissions contributed from the gateway, which is due to the higher traffic volumes on the southbound gateway links; it also received a greater contribution from

arterial roads than the other near-highway receptor due to the arterial roads near the school experiencing higher traffic volumes. Traffic emission impacts to the community are illustrated in detail in Figure 41 and Figure 42. Figure 41 shows the exposure impacts resulting from arterial roads in the community whereas Figure 42 shows the contribution of only the interstate highway emissions to the community.

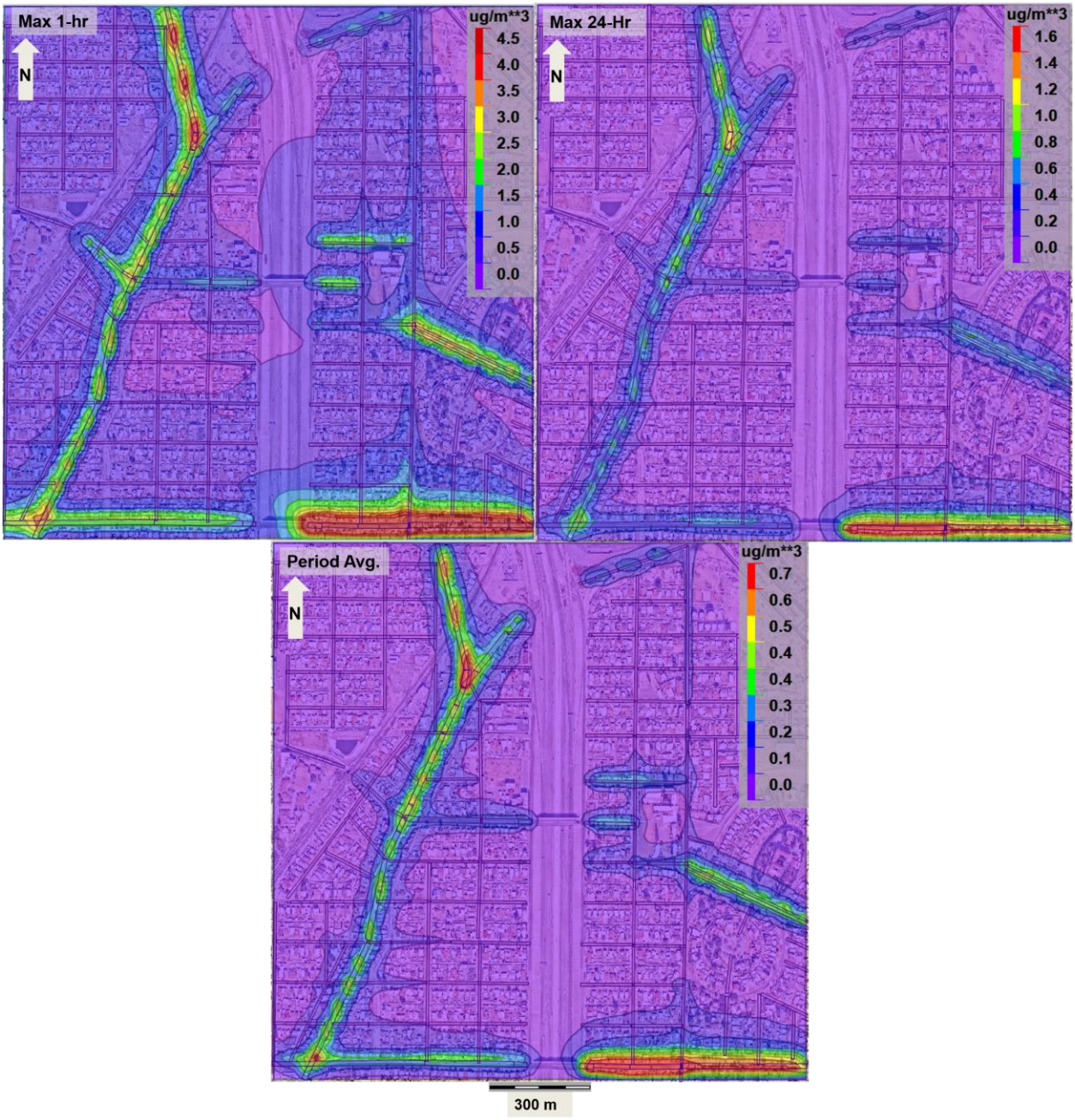

Figure 41 Exposure Impacts from Arterial roads in the community

It is noted that $PM_{2.5}$ concentrations are observed more diversely throughout the community, once the high concentrations from the highway are removed. That is to say community exposure that is directly from the arterial roads, is more clearly seen in Figure 41.

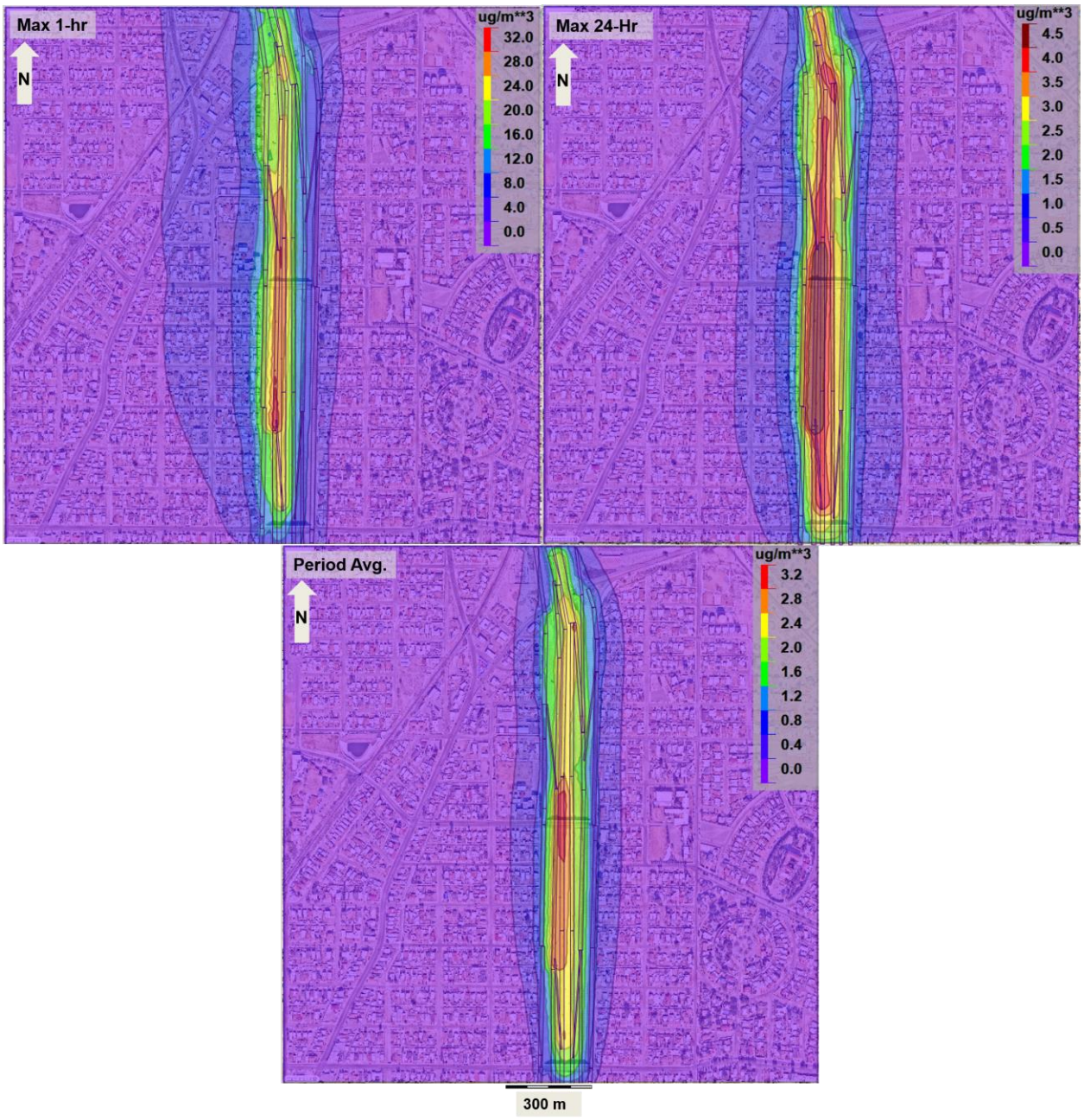

Figure 42 Exposure Impacts of U.S. 54 emissions to the community

While arterial roads show impact to the immediate areas, highway contributions occur at a much higher rate, up to 32 $\mu g/m^3$ estimated for the maximum 1-hr $PM_{2.5}$ concentration. In

observing the range of modeled $PM_{2.5}$ concentrations, it can be seen that the impact to the surrounding community is largely influenced by the traffic volumes found on the highway links.

Chapter 10: Conclusions

This study addresses the spatial and temporal concentration variations of $PM_{2.5}$ in a near-road community resulting from traffic emissions on a microscale. It appears that there is a divergence between the concentrations predicted by AERMOD and the monitored data. The following chapter details the summary of the objectives accomplished in this research, followed by recommendations for future research, and general conclusions of this study.

10.1 Objectives Summary

One goal of this research was to capture the distribution and impact of these pollutants on air quality and human health at a finer resolution, capturing the temporal and spatial variations at a local scale near these critical roadways. This research developed spatial and temporal pollutant concentration variation patterns for $PM_{2.5}$ in a near-road community. Traffic inputs were obtained from the travel demand model, field measurements of traffic volumes, and combined with factors related to vehicle fleet information, roadway characteristics, and fuel and weather conditions to create emissions factors estimates for the roadways in the study area. A dispersion model was used to calculate the dispersion of these emissions in the atmosphere based on fate and transport properties of the pollutants, meteorological conditions, and land use characteristics. The results of this modeling framework were combined with air quality results obtained through field measurements. The total $PM_{2.5}$ exposure in the community was assessed by adding the AERMOD modeled concentration estimates to the selected background concentrations. It appears that the model over-estimates the maximum 1-hr, maximum 24-hr $PM_{2.5}$ and All Period Average, at the near-road sites and the off-highway residence by at least 25%, 26%, and 12%, respectively. It is also apparent that the model accuracy improves for longer term average. It is important to note that this comparison involves the addition of the hourly background concentrations obtained from the UTEP CAM site. Furthermore, this “background” value is often higher than even the observed

concentrations at the two near-road sites. Additionally, the model is sensitive to wind speed, wind direction, and ERs. This results in higher "maximum" estimates for the Coldwell site (located east of highway, near higher traffic roads), compared to the other near road site at the House. Monitored results also show these "maximums" are much closer for the two near-road sites (6-10% difference). This indicates that max values captured by AERMOD may be obscured in real-life exposure due to the ubiquity of background $PM_{2.5}$ concentrations in urban areas.

The second objective of this research was accomplished by apportioning the differences in exposure concentrations between background concentrations and those contributed from major highways. Using the modeling framework, the dispersion model is used to assess percentage and distribution of emissions from highways and arterials in the study area. The model results show that the House and Coldwell, the two near-road sites, experience 87% and 89% of $PM_{2.5}$ contribution from the Highway, respectively. The community monitor, Radford, experiences 50% of $PM_{2.5}$ contribution from the highway. Monitored values show exposure to $PM_{2.5}$ emissions is largely due to the background concentrations in the urban area. Considering $PM_{2.5}$ stations as background estimates requires evaluation of the traffic and emissions rates of nearby arterials. Using a "community" monitor can be helpful for background estimates. Using the Radford monitor to represent background estimates results in more clarity in modeled results comparison to monitored values.

10.2 Recommendations

Recommendations for future studies include establishing and using field monitor for background estimate and using meteorological station on-site for more accurate model predictions. Additionally, evaluating different pollutants that are more closely correlated with traffic emissions can help assess the effects of traffic on community exposures. For accurate emission factors

generation, fleet information proves to be most difficult to obtain when most traffic counters provide only a broader classification of vehicles than what is required by the MOVES model.

Additional recommendations for improvement on the research are to run more detailed sensitivity analysis using the dispersion model AERMOD, such as source characterization, meteorological conditions, and land use parameters.

10.3 General Conclusions

On-site monitoring of air pollution at near road schools is able to capture high resolutions variations in air quality. The results from this study provide information needed in the field of vehicle emissions exposure to near-road communities. Determining the influence of mobile emissions from highways on the air quality of the surrounding communities can help raise awareness to underserved communities living near highways and help policy makers make informed decisions based on this knowledge. While it is shown through this study that highway emissions drop considerably after around 200 meters, communities would benefit from avoiding designation of residential and school facilities within these zones and could result in less exposure to harmful vehicle emissions.

This study addresses the spatial and temporal concentration variations in a near-road community resulting from traffic emissions on a microscale. It appears that there is a clear divergence between the concentrations predicted by AERMOD and the monitored data. The AERMOD predictions rendered highest concentration estimates at locations where the traffic volume is the highest and downwind of the prevailing winds. However, impacts of the traffic emissions on the air quality subside rapidly with increasing distance away from the highway. In the near-road community studied, traffic emissions from the highway were 4.8 times higher than the contributions made by local arterial roads. Model estimates are highly sensitive to meteorological conditions and source characterization, and additionally, higher quality of upper

air data could yield more accurate meteorological parameters from the AERMET preprocessor. Finally, obtaining accurate background data from the study area can help provide better modeled-to-monitored comparison, as background concentrations have been shown to be of greater impact in urban areas and contributes to around 85% of measured $PM_{2.5}$ concentrations.

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Appendix A

Paper Number: 19-01459

Title: CONTRIBUTION OF TRAFFIC EMISSIONS TO NEAR-ROAD $PM_{2.5}$ AIR CONCENTRATIONS AS IMPLIED BY URBAN-SCALE BACKGROUND MONITORING

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Introduction

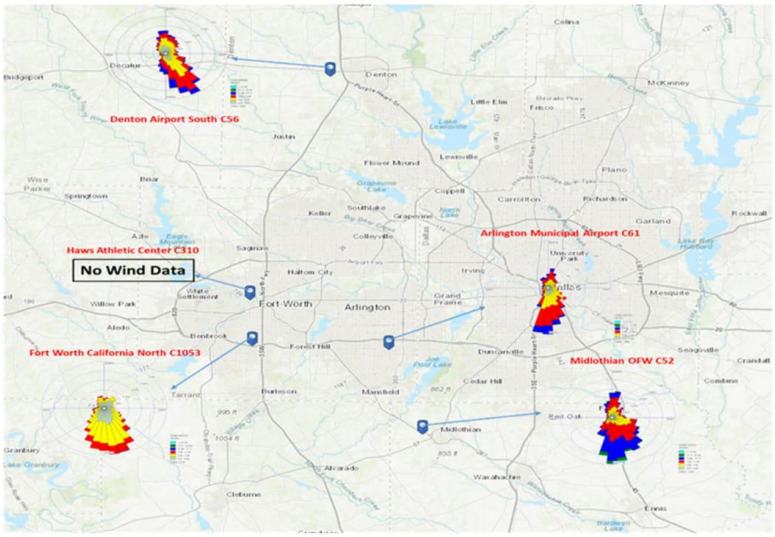
Traffic-related air pollution has the most profound impact on human health because of the quantity of pollutants emitted and the relatively close proximity between the source and the population. Prior studies have documented the adverse impacts of traffic-related air pollution on cardiovascular health in adults (*1, 2, 3*). Recent studies have concluded from reviews of near-road air monitoring data that only PM (PM_{10} or $PM_{2.5}$) in the near-road environment may exceed the annual or 24-hr average NAAQS (*4, 5*). It was also concluded that the contribution of traffic-related emissions to the near-road PM pollution is less than 15% (*4, 6, 7*) and near-road PM pollution does not decrease as rapidly as other pollutants off the highway (*8*). Indeed, traffic $PM_{2.5}$ pollution was reported to dilute slowly to the background level in approximately 1 km (*8*) or remain essentially undiluted at distances well beyond 200 m (*9*). These studies may not seem to agree well with the estimates derived from a typical Gaussian line source model. For instance, Venkatram et al (*10*) showed that the concentration of an inert pollutant decays rapidly to less than 1/5 of its initial strength in 100 m in the direction normal to the roadway. The discrepancy could be attributed to many uncontrollable factors, such as the existence of sound walls for at-grade freeways, elevated or filled section of a freeway, canopy vegetation, and classification of atmospheric stability condition. Nevertheless, this gross mismatch between the downwind concentrations and the model estimates shows the need for further model improvement.

A vast amount of effort has been focused on how to improve the accuracy of vehicle emissions and air dispersion models and how to address the sensitivities of various parameters (traffic, emissions, meteorological, topographic, behavioral, etc.) in the models. Unfortunately, model validation requires a good agreement between concentration estimates and data observed at a near-road monitor where more than 85% of the PM is attributed to background emissions from sources other than the road segment immediately adjacent to the monitor. Thus, success in selecting the best emission and dispersion models for a transportation conformity analysis is pinned to success in developing appropriate background concentrations for a near-road site.

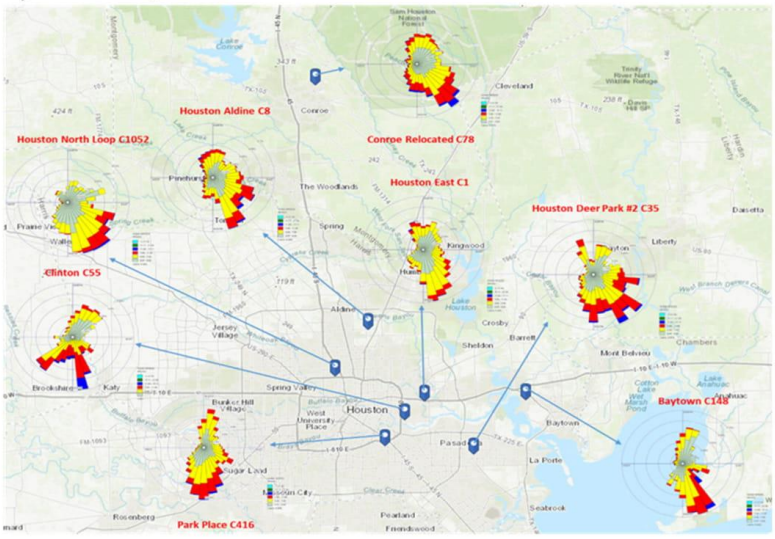
In a study to determine the contribution of traffic emissions to the near-road pollution, we compared the background $PM_{2.5}$ concentration estimates, developed from a modified U.S. EPA method with multiple urban-scale ambient air monitors in the same city (*11*), to that reported at two near-road monitors. We attempted to validate the background annual, 24-hour, and highest 20 24-hour average $PM_{2.5}$ concentrations using the observed and predicted values in conjunction with surface meteorological conditions. This paper describes the procedures, sheds light on how to develop the appropriate background concentrations using the regional and onsite concentrations and wind statistics, and determines the percentage of contribution in near-road $PM_{2.5}$ that could be attributed to adjacent roadway emissions.

METHODOLOGIES

The locations and wind statistics for the project sites (C1052 and C1053) and regional background stations are included in Figure 1. Integrated 24-hour filter-based $PM_{2.5}$ data (FRM data) as well as continuous hourly data (FEM data) from the background stations were used to develop the background concentration data for the project sites using methods recommended by EPA (12) and modified by Li et al (11). The background $PM_{2.5}$ concentration estimates developed for the near-road sites were compared to the observations in terms of annual and 24-hour averages to interpret the contribution of traffic emissions on near-road monitors.



a) Fort Worth, 2015



b) Houston, 2015

FIGURE 1 Locations and wind statistics (2015 only) for all sites (*11*)

Findings

It was found that background concentration estimates for the near-road sites trend well with data reported for other urban background sites in each city due to uncharacteristic terrain features and

predominant wind directions in both cities. The 24-hour $PM_{2.5}$ FRM data appear, at the first glance, to randomly scatter about the line representing the predicted background concentrations due to different sampling schedules employed by FRM and FEM sampling. A subset of the data sorted by the days FRM data were available is presented in Figure 2. It is displayed in this figure that the background concentration estimates trend very well with the FRM observations at the same site. In general, the background concentrations appear to be at levels comparable to that observed under the influence of near-road traffic emissions.

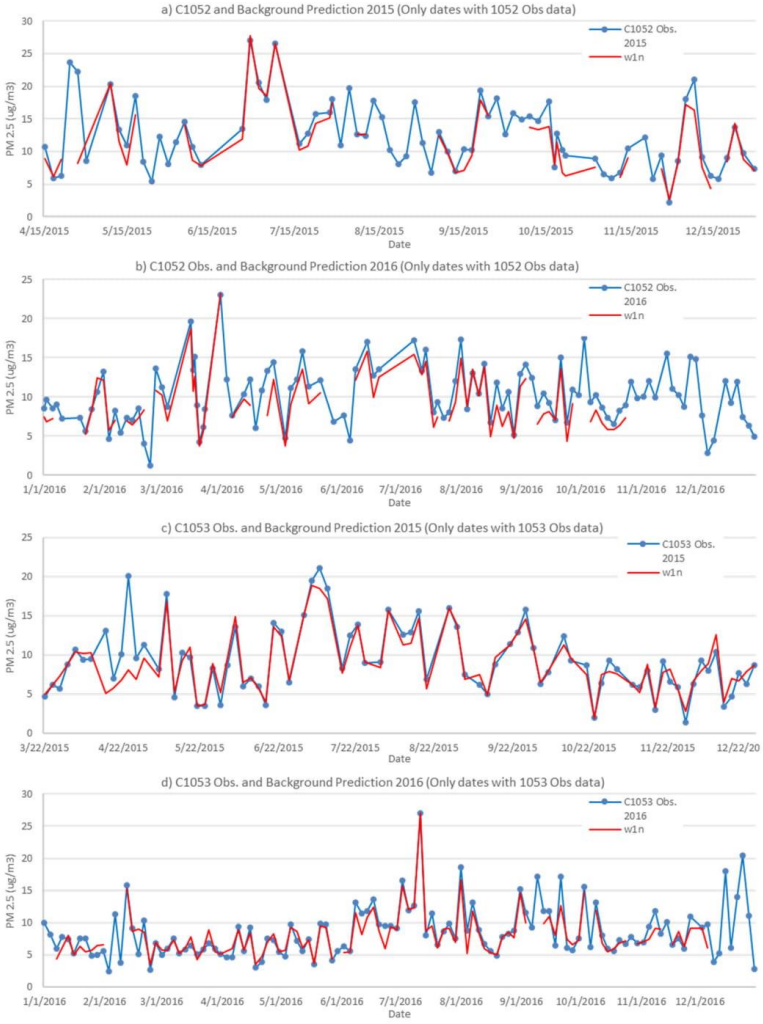


FIGURE 2 Comparison of background concentration estimates at C1052 and C1053 sorted by FRM 24-hour average data

Annual Average $PM_{2.5}$ Concentrations

Table 1 compares the annual average $PM_{2.5}$ concentrations and background concentration estimates for the two near-road sites to that measured at regional background sites. The regional annual average background concentration varied from 7.8 to 11.4 [9.6, 0.9] $\mu g/m^3$ for C1052, and 7.8 to 8.8 [8.2, 0.2] $\mu g/m^3$ for C1053, while the near-road monitors observed slightly higher annual averages at 11.3 and 8.9 $\mu g/m^3$, respectively, for 2015 and 2016. The annual averages at the background sites varied from -13% to +4% with a mean of 6.4%, compared to -19% to -6.4% with a mean of 12.7% at the near-road sites. Paired t-tests were performed for both near-road sites for 2015, 2016, and combined years to examine whether the average concentrations are statistically different from the background concentrations. The results indicated the annual averages observed at near-road monitors were significantly different from those estimated for all datasets. The contribution from traffic emissions to the annual average $PM_{2.5}$ concentrations was found to be approximately 14% and 6.7 % at C1052 and C1053, respectively.

TABLE 1 Summary of annual average $PM_{2.5}$ concentrations for the studied sites

	Annual Average PM2.5 Concenrations, µg/m3								
Year	CAMS 1052	BG Est.	**Houston**	BG 1	BG 2	BG 3	BG 4	BG 5	BG 6
2015	12.5	10.2		9.5	11.4	10.6	10.3	8.7	9.6
2016	10.1	9.2		8.7	10.0	10.0	9.0	7.8	9.4
Average	11.3	9.7		9.1	10.7	10.3	9.7	8.2	9.5
Year	CAMS 1053	BG Est.	**Fort Worth**	BG 1	BG 2	BG 3	BG 4		
2015	9.3	8.6		8.8	7.8	8.4	8.0		
2016	8.5	8.1		8.0	8.1	8.0	7.9		
Average	8.9	8.3		8.4	8.0	8.2	8.0		

Twenty-four-hour Average $PM_{2.5}$ Concentrations

Table 2 summarizes the 24-hour average $PM_{2.5}$ concentrations ranked by the highest 20 background concentrations. Data for a number of days were missing in the table due to the once every 3rd day monitoring schedule for FRM air monitoring network, instrument maintenance, malfunction, and other unavoidable circumstances. It is observed that the roadside $PM_{2.5}$ concentrations were obliterated by the high regional background concentrations. The average mean bias error (MBE) between the near-road observations and the estimated background concentrations varies from -0.3 to 0.6 $\mu g/m^3$ for C1052 and 0.2 to 0.4 $\mu g/m^3$ for C1053, representing an average of -1.1 % to 3.6% of normalized mean bias error (NMBE). Further evaluation of Table 2a reveals that:

1. Both sites experienced high urban-scale $PM_{2.5}$ pollution in the summer between May and September, particularly around the July 4th Independence Day holidays where more than 50% of top 5 background concentration days occurred between July 3 and July 15 (4 out of 10 at C1052 and 7 out of 10 at C1053).
2. The 98th percentile value for the background concentration is represented by the 7th or 8th highest value of a year, depending on the size of the dataset. If one limits the comparison to the highest 10 background concentration values in a year in Table 2a one would realize that there is basically no difference between the near-road and background concentrations,

although only 10 pairs of data points are available, when pollution is dominated by the urban-sale background concentrations.

Table 2b presents the same data but sorted by the highest concentrations reported at the near-road sites. High $PM_{2.5}$ pollution does not necessarily occur concurrently on high background concentration days. Indeed, the difference in $PM_{2.5}$ between the roadside monitors and the background concentrations is more distinguishable. The average deviation from the mean for the highest 20 values at C1052 is 2.9 and 1.5 $\mu g/m^3$ (or 13.6 to 9.3%) for 2015 and 2016, respectively, compared to 1.1 and 0.9 $\mu g/m^3$ (or 6.3 to 6.1%) for C1053. The range narrows to 1.3 to 1.5 $\mu g/m^3$ (or 11 to 9.3%) at C1052 and 0.5 to 0.9 $\mu g/m^3$ (or 3.6 to 6.1 %) at C1053 if two outliers were removed from the comparison. These values agree well with the field study, as reported by Dewinter et al (*7*), and modeling results, as reported by Vallamsundar and Lin (*6*).

TABLE 2 Twenty highest 24-hour average $PM_{2.5}$ data at the studied sites

a) Twenty highest 24-hr average PM2.5 concentrations ranked by background concentrations

Rank	Date	Conc. µg/m^3 BC	Conc. µg/m^3 Obs	Wind Dir.	Date	Conc. BC	Conc. Obs.	Wind Dir.	Obs-BC µg/m^3 2015	Obs-BC µg/m^3 2016	(Obs-BC)/Obs*100 % 2015	(Obs-BC)/Obs*100 % 2016
	Houston C1052								**2015**	**2016**	**2015**	**2016**
1	06/29/15	27.7	27.0	DW	07/10/16	27.0			-0.7		-2.7	
2	07/08/15	26.6	26.6	DW	07/11/16	26.7			0.0		0.1	
3	06/18/15	24.9			03/15/16	25.1						
4	05/10/15	23.4			03/31/16	23.3	23.0			-0.3		-1.1
5	06/21/15	22.0			07/15/16	21.2						
6	07/06/15	21.1			06/15/16	19.8						
7	07/09/15	21.0			03/16/16	18.7	19.6			0.9		4.7
8	07/07/15	21.0			05/10/16	16.8						
9	05/09/15	20.4	20.3	DW	06/14/16	15.8	17.0	DW	-0.1	1.2	-0.7	7.1
10	07/10/15	20.2			04/28/16	15.6						
11	07/03/15	20.1			07/08/16	15.4	17.2	DW		1.8		10.5
12	07/02/15	19.7	20.6	DW	07/02/16	15.1			0.9		4.3	
13	05/16/15	19.7			08/01/16	15.0	17.3	DW		2.3		13.4
14	06/19/15	19.5			04/29/16	14.6						
15	12/24/15	19.5			07/14/16	14.6	16.0	DW		1.4		8.9
16	06/30/15	19.4			07/09/16	14.4						
17	08/08/15	19.3			02/01/16	14.3						
18	07/05/15	18.5	17.9	DW	05/13/16	14.0			-0.6		-3.2	
19	05/17/15	18.2			08/06/16	13.9						
20	09/21/15	17.9	19.3		08/13/16	13.8	14.2	DW	1.4	0.4	7.1	2.7
							Average		-0.3	0.6	-1.1	3.6

Rank	Date	Conc. µg/m^3 BC	Conc. µg/m^3 Obs	Wind Dir.	Date	Conc. BC	Conc. Obs.	Wind Dir.	Obs-BC µg/m^3 2015	Obs-BC µg/m^3 2016	(Obs-BC)/Obs*100 % 2015	(Obs-BC)/Obs*100 % 2016
	Fort Worth C1053								**2015**	**2016**	**2015**	**2016**
1	07/09/15	26.2			07/11/16	27.0	27.0	DW		0.0		0.1
2	07/06/15	21.9			04/29/16	19.3						
3	07/10/15	21.0			07/15/16	19.2						
4	07/03/15	20.4			06/15/16	18.5						
5	07/05/15	18.9	19.5	DW	05/10/16	18.4			0.6		3.3	
6	07/08/15	18.5	21.1	DW	07/16/16	18.0			2.6		12.1	
7	07/30/15	18.3			08/01/16	16.7	18.6	DW		1.9		10.5
8	12/15/15	18.2			09/22/16	16.6						
9	06/21/15	17.5			05/25/16	16.4						
10	07/14/15	17.4			07/10/16	16.2						
11	06/22/15	17.2			06/16/16	15.8						
12	07/11/15	17.2	18.5	DW	07/02/16	15.7	16.6	DW	1.3	0.9	6.9	5.4
13	06/06/15	17.1			02/12/16	15.3	15.8	UP		0.5		3.3
14	05/08/15	16.9			07/12/16	15.3						
15	05/09/15	16.8	17.8	DW	07/03/16	15.1			1.0		5.6	
16	07/07/15	16.7			10/03/16	14.7	15.6	DW		0.9		5.9
17	10/08/15	16.3			08/31/16	14.5	15.2	UP		0.7		4.3
18	08/28/15	16.0	16.0	DW	08/02/16	14.2			0.0		0.2	
19	08/04/15	15.7	15.8	DW	02/13/16	14.1			0.1		0.8	
20	07/02/15	15.2	15.1	DW	06/30/16	13.4			-0.1		-0.6	
							Average		0.8	0.8	4.0	4.9

b) Twenty highest 24-hr average PM2.5 concentrations ranked by highest observations

Rank	Date	Conc. µg/m^3 Obs.	Conc. µg/m^3 BC	Wind Dir.	Date	Conc. Obs.	Conc. BC	Wind Dir.	Obs-BC µg/m^3 2015	Obs-BC µg/m^3 2016	(Obs-BC)/Obs*100 % 2015	(Obs-BC)/Obs*100 % 2016
	Houston C1052								**2015**	**2016**	**2015**	**2016**
1	06/29/15	27.0	27.7	DW	03/31/16	23.0	23.3		-0.7	-0.3	-2.7	-1.1
2	07/08/15	26.6	26.6	DW	03/16/16	19.6	18.7		0.0	0.9	0.1	4.7
3	04/24/15	23.7			10/03/16	17.5						
4	04/27/15	22.2	8.1	UP	08/01/16	17.3	15.0	DW	14.1	2.3	63.3	13.4
5	12/08/15	21.0	16.4		07/08/16	17.2	15.4	DW	4.6	1.8	21.9	10.5
6	07/02/15	20.6	19.7	DW	06/14/16	17.0	15.8	DW	0.9	1.2	4.3	7.1
7	05/09/15	20.3	20.4	DW	07/14/16	16.0	14.6	DW	-0.1	1.4	-0.7	8.9
8	08/04/15	19.7		DW	05/12/16	15.8	13.6			2.2		14.2
9	09/21/15	19.3	17.9		11/14/16	15.5			1.4		7.1	
10	05/18/15	18.5	15.7	DW	03/18/16	15.1	12.6		2.8	2.5	15.3	16.6
11	09/27/15	18.2		UP	11/26/16	15.1						
12	07/29/15	18.0	17.5	DW	09/21/16	15.0	13.6	UP	0.5	1.4	2.9	9.3
13	12/05/15	18.0	17.2	UP	11/29/16	14.8			0.8		4.6	
14	07/05/15	17.9	18.5	DW	04/27/16	14.4	12.2	DW	-0.6	2.2	-3.2	15.2
15	08/13/15	17.8		DW	08/13/16	14.2	13.8	DW		0.4		2.7
16	10/16/15	17.7	13.9	UP	09/03/16	14.1	12.4	UP	3.8	1.7	21.5	12.3
17	08/28/15	17.5			02/27/16	13.6	10.8	DW		2.8		20.8
18	07/28/15	16.0	15.1	DW	06/20/16	13.5	12.5		0.9	1.0	5.6	7.5
19	10/03/15	15.9		UP	06/08/16	13.5	12.2			1.3		10.0
20	07/23/15	15.7	14.3	DW	03/17/16	13.4	10.7	DW	1.4	2.7	9.1	20.3
							Average		2.9	1.5	13.6	9.3

Rank	Date	Conc. µg/m^3 Obs.	Conc. µg/m^3 BC	Wind Dir.	Date	Conc. Obs.	Conc. BC	Wind Dir.	Obs-BC µg/m^3 2015	Obs-BC µg/m^3 2016	(Obs-BC)/Obs*100 % 2015	(Obs-BC)/Obs*100 % 2016
	Fort Worth C1053								**2015**	**2016**	**2015**	**2016**
1	07/08/15	21.1	18.5	DW	07/11/16	27.0	27.0	DW	2.6	0.0	12.1	0.1
2	04/24/15	20.1	8.0		12/23/16	20.4			12.1		60.0	
3	07/05/15	19.5	18.9	DW	08/01/16	18.6	16.7	DW	0.6	1.9	3.3	10.5
4	07/11/15	18.5	17.2	DW	12/14/16	18.0		UP	1.3		6.9	
5	05/09/15	17.8	16.8	DW	09/21/16	17.2	12.6	DW	1.0	4.6	5.6	26.8
6	08/28/15	16.0	16.0	DW	09/09/16	17.2		DW	0.0		0.2	
7	08/04/15	15.8	15.7	DW	07/02/16	16.6	15.7	DW	0.1	0.9	0.8	5.4
8	09/27/15	15.8	14.6		02/12/16	15.8	15.3	UP	1.2	0.5	7.8	3.3
9	08/16/15	15.6	14.7	DW	10/03/16	15.6	14.7	DW	0.9	0.9	6.0	5.9
10	07/02/15	15.1	15.2	DW	08/31/16	15.2	14.5	UP	-0.1	0.7	-0.6	4.3
11	06/20/15	14.1	13.6	DW	12/20/16	14.0		DW	0.5		3.5	
12	07/23/15	13.9	13.8	DW	06/17/16	13.6	12.4	DW	0.1	1.2	0.8	8.9
13	06/05/15	13.6	14.9	DW	10/09/16	13.2	12.0		-1.3	1.2	-9.3	9.4
14	08/31/15	13.6	13.9	DW	08/07/16	13.1	11.7	DW	-0.3	1.4	-2.5	10.3
15	04/15/15	13.1	5.1		06/08/16	13.1	11.5	DW	8.0	1.6	61.2	12.2
16	06/23/15	13.0	12.4	DW	07/08/16	12.6	12.3	DW	0.6	0.3	4.5	2.8
17	09/24/15	12.9	12.9		07/05/16	11.9	12.0	DW	0.0	-0.1	0.1	-1.2
18	08/13/15	12.9	11.5		09/15/16	11.8	10.9	DW	1.4	0.9	11.0	7.6
19	08/10/15	12.6	11.3	DW	06/14/16	11.8	10.9	DW	1.3	0.9	10.3	7.9
20	07/20/15	12.5	11.0	DW	09/12/16	11.8	9.9	DW	1.5	1.9	11.9	16.3
							Average		1.6	1.2	9.7	8.2

Conclusion

It was found that the near-road $PM_{2.5}$ levels are comparable to the regional background levels and that the concentration increment resulting from transportation related emissions was relatively small and likely to be obliterated by the regional background concentrations during high $PM_{2.5}$ days. The upwind-downwind configuration between the air monitor and the adjacent highway section does not show dominance of highway emissions on near-road $PM_{2.5}$ pollution. Given the many uncertainties involved in air quality monitoring, emission modeling, and air dispersion modeling in assessing near-road pollution, the small concentration increment due to transportation enhanced emissions may be difficult to be verified by air quality measurements. Further research to expand the methodology employed in this study to other metropolises will help quantify the impact of traffic emissions on near-road air quality and select the best emission and air dispersion models for the near-road air pollution study.

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Appendix B

DETERMINATION OF BACKGROUND $PM_{2.5}$ CONCENTRATIONS FOR A POTENTIAL TRANSPORTATION PROJECT SITE

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ABSTRACT

Studies have shown that $PM_{2.5}$ measured at near-road air quality monitors is only moderately impacted by traffic emissions. More than 85% of the roadside $PM_{2.5}$ concentrations are believed to be regional urban-scale background concentrations which are primarily caused by ubiquitous urban emission sources. The U.S. EPA has established guidance on quantitative $PM_{2.5}$ hot spot analysis to ensure transportation projects do not worsen the existing air quality. Determination of the regional $PM_{2.5}$ background concentrations thus becomes important in the transportation air quality analysis as the background concentration is combined with project-specific incremental concentration to determine compliance with the NAAQS.

Seven background $PM_{2.5}$ concentration estimation methods, 4 of them suggested by the U.S. EPA, were evaluated in this paper using 2 years of hourly urban-scale background air monitoring data available at 11 sites in 2 Texas cities. Performance of the methods was assessed by comparing the observations at one site to that estimated from the surrounding sites. A performance metric consisted of three parameters (NMB, NME, and NRMSE) was used in the evaluation and random sampling with replacement by bootstrapping was performed to assess the sampling variability. This paper provides a methodology of developing the best estimates for background $PM_{2.5}$ concentrations at transportation project locations where background concentrations may not be available. It was found in this work that the 24-hour and annual average background $PM_{2.5}$ concentrations at a site can be best estimated by the normalized inverse squared distance weighted average of the concentrations measured at surrounding background sites.

Key Words: PM2.5, Near-road, Background Concentration, Air Pollution, Hot-spot Analysis

INTRODUCTION

Near-road air pollution has gained increased attention since 2010 when the U.S. EPA promulgated new minimum requirements for NO_2 monitoring and required the state and local air monitoring agencies to install near-road NO_2 monitors. According to EPA (40 CFR Part 58, Appendix D), near-road monitoring is defined as monitoring at locations within 50 meters of a major roadway with high annual average daily traffic (AADT) count. Initially, the monitoring priority focused on NO_2 as the primary pollutant of traffic emissions, but the priority quickly extended to cover two other criteria pollutants, $PM_{2.5}$ and CO, which are both strong markers of traffic emissions. Pollutant concentrations at near-road monitoring sites are affected by a number of factors related to transportation (such as traffic volume, vehicle fleet, vehicle age, speed, and inspection and maintenance), local meteorology (such as wind direction, wind speed, temperature, pressure), and terrain topography (such as roadway-receptor configuration, surface roughness, road condition, source and receptor elevations). A U.S. EPA initiated near-road pilot study concluded that near-road NO_2 concentrations are likely to be highest at locations near the roadway with highest traffic (*1*). The study also discovered that near-road NO_2 concentrations in 5 studied cities were all less than the 1-hr NAAQS and that the average near-road NO_2 concentrations were higher than the background concentrations observed at non near-road sites. The State of Maryland conducted a 3½-year study at a Maryland State Highway Administration monitoring site (*2*) that found no exceedances of the 24-hr or annual NAAQS for $PM_{2.5}$ during the studied period and that the near-road $PM_{2.5}$ concentrations were consistently higher than that measured at background locations.

Near-road air quality data has become more available in the U.S. since 2014 when state and local air pollution control agencies began to collect NO_2, CO, and $PM_{2.5}$ data and reported to the U.S. EPA's Air Quality System (AQS) database. A national-scale review of near-road air pollutant concentrations using the 2014-2015 AQS was conducted in 2016 (*3*). Concurrent state-reported AADT data of the major roads associated with each of the official near-road monitoring sites was evaluated to understand how concentrations varied by factors such as location, distance to roadway, and traffic parameters at near-road monitoring locations. It was discovered that all of the 1-hr CO values were well below the 1-hr NAAQS of 35 ppm or the 8-hr NAAQS of 9 ppm. NO_2 concentrations were observed to be well below the 1-hr NAAQS of 100 ppb (98^{th} percentile of 1-hr daily maximum concentrations averaged over 3 years). Only 2 occurrences of a daily 1-hr maximum NO_2 exceeded 100 ppb out of a total of 40 sites in 2014, and 5 occurrences (or 0.0015%) in 2015 from a total of 61 sites (*3*). $PM_{2.5}$ concentrations at near-road sites, however, behaved differently from the other two pollutants. Three out of 10 sites reported $PM_{2.5}$ concentrations of greater than the annual average NAAQS of 12 $\mu g/m^3$ in 2014, and 2 out of 36 sites in 2015. For the 24-hr average $PM_{2.5}$ concentrations, the NAAQS of 35 $\mu g/m^3$ was exceeded 15 times at 7 locations and 33 times at 12 locations in 2014 and 2015, respectively.

Pollutant concentrations measured at near-road monitors consist of the background concentration and an incremental concentration from the adjacent roadways. For example, DeWinter et al (*3*) reported that proximity to the high traffic roadway only results in small increment of $PM_{2.5}$ concentrations (an average of 1.2 $\mu g/m^3$ with a standard deviation of 0.2 $\mu g/m^3$) from the

background concentrations recorded at other urban scale locations. This increment represents, on average, a 13 to 15% increase depending on how close the near-road monitor is away from the roadway. This finding is consistent with that reported in the Maryland study (*2*) which also concluded that $PM_{2.5}$ impacts of traffic emissions are not immediately noticeable at a distance of 150 m (~0.1 mile) from the roadway and that only approximately 14% of $PM_{2.5}$ collected at the near-road site could be attributed to the roadway sources. The contribution of roadway emissions to the near-road $PM_{2.5}$ concentrations apparently vary significantly due to the uncertainties and variabilities involved in local meteorology, traffic count, vehicle fleet, source-receptor geometry, time, day and season of the year. For instance, Vallamsundar and Lin (*4)* estimated that only approximately 5% of the near-road $PM_{2.5}$ can be attributed to the emissions from the road segment, based on a project-level MOVES-AERMOD emission and air dispersion modeling analysis. A recent study conducted in Netherlands aiming at this effect further suggested that the urban $PM_{2.5}$ and PM_{10} concentrations are dominated by the regional background and that primary and secondary PM emission by urban sources contribute less than 15 % to the near-road sites (*5*).

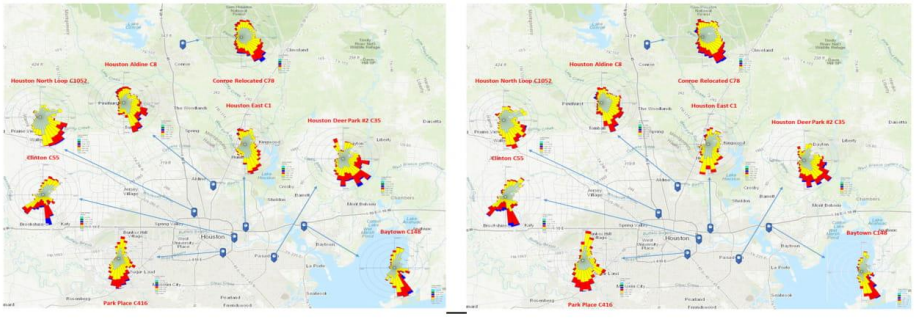
As required by EPA's transportation conformity hot-spot analysis, the analysis must calculate project specific contribution and background concentration using EPA's recommended procedures. The project specific contribution is estimated through MOVES emission and AERMOD air dispersion modeling (*6*). Compliance of the NAAQS is determined by comparing the design value or the sum of the modeled concentration from the project and the background concentration to the respective NAAQS. Obviously, the success of a compliance study depends on a reliable background concentration estimate. In a hot-spot analysis, an overestimated background concentration will inevitably result in overestimation of the air quality impacts and potentially jeopardize the implementation of a transportation project whereas an underestimation will underestimate the impacts and unintentionally increase risks to the public's health.

As recommended in EPA's guidance on hot-spot analysis, background $PM_{2.5}$ concentrations should be as representative as possible for the area where the project site is located. Ideal background concentrations for a near-road site without the influence of traffic emissions are rarely available. For an area surrounded by multiple background ambient $PM_{2.5}$ monitors, the EPA recommended that the data should be analyzed by statistical or mapping methods to develop a background concentration for use in the hot-spot analysis. Four methods, based on either a single station or multiple stations, are suggested by the EPA (*6*) for developing the background concentrations for a near-road location. However, no specific guidance was provided regarding which method is preferred for an area.

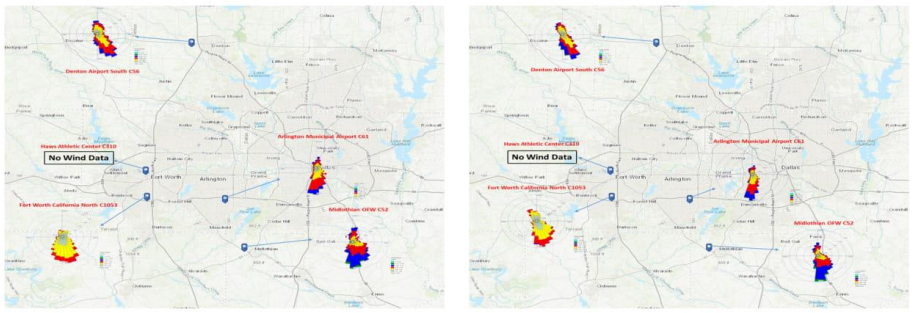
In this study, we selected 2 target project sites in Texas and evaluated 7 methods, with four of them suggested by EPA, for background concentration estimation using data from multiple urban-scale stations available in the project areas. We conducted statistical analyses and compared the performance of each method to determine the best approach for background concentration estimation for the project sites. This paper describes the methodology and selects the best model for background $PM_{2.5}$ concentration estimation.

METHODOLOGY

The two project sites identified in this study are locations near the North Loop (CAMS 1052) in Houston and California Parkway North (CAMS 1053) in Fort Worth, Texas. We collected hourly meteorology and pollutant data at all available background urban-scale air monitoring stations within 50 miles of the sites. In total, 7 TCEQ monitoring stations in Houston and 4 in Fort Worth were identified. Figure 1 shows the locations and wind statistics for all background and project sites in Houston and Fort Worth. We evaluated 7 background concentration estimation methodologies by comparing the $PM_{2.5}$ data observed at a specific site *i* to the concentration estimates developed from the remaining sites (e.g., 6 for Houston and 3 for Fort Worth) for site *i*. We then performed statistical analysis to select the best methodology. The selected methodology is believed to be applicable for providing best background concentration estimates for any transportation project in the vicinity of the study areas.



a) Houston, 2015 b) Houston, 2016



c) Fort Worth, 2015 d) Fort Worth, 2016

FIGURE 1 Locations and wind statistics (2015 an 2016) for all locations of interest in Houston and Fort Worth, Texas

Surface meteorological conditions are quite similar for all sites in Houston except CAMS 55 Clinton site, indicating that possibly a drainage northwest-southeast wind pattern towards the ocean exists between Houston metropolitan area and Galveston Bay by the Gulf of Mexico. At Fort Worth, wind patterns are quite similar between CAMS 1053 and surrounding sites, indicating that wind pattern in the great Dallas-Fort Worth area is quite similar. We collected hourly

meteorology and pollutant data at all these background urban-scale monitoring stations. Seven TCEQ air monitoring stations were found within a 50-mile radius from CAMS 1052, 4 for CAMS 1053.

The 7 methods (including 4 methods [Methods 1 through 4] recommended by EPA) for estimating the background concentrations at a near-road site were identified below:

1. Single station (based on distance, upwind location, and similar surface parameters)
2. Arithmetic mean from multiple stations
3. Inverse distance weighing from multiple stations
4. Inverse distance squared weighing form multiple stations
5. Normalized arithmetic mean from multiple stations
6. Normalized inverse distance weighing from multiple stations
7. Normalized inverse distance squared weighing from multiple stations

Statistical methods, including methods used for performance measures are used to determine the best model for interpolation of background annual and 24-hr averaged $PM_{2.5}$ concentrations.

Data Processing

Two years (2015 and 2016) of hourly $PM_{2.5}$ data was downloaded from the TCEQ TAMIS web interface. Missing hourly $PM_{2.5}$ data was replaced by the averages of the adjacent values, the previous and next hour of data, if less than 4 consecutive hours. Data missed for more than 4 hours are left untreated and flagged with missing data. Data of all background stations for the same hour with at least one flag was removed from the 1-hour database. For the 24-hour average database, data of all background stations for the same day was removed if more than 25% of the hourly data was missing from one or more stations (*7*).

Model Formulation

To test the methods, we designated one station as the target station and treated its data as observations while the data from other stations was used to developed background concentration estimates. In other words, the $PM_{2.5}$ concentration, $x_{i,j}$, represents a concentration observed at station i and at a time step j:

$x_{i,j}$ = $PM_{2.5}$ concentrations, $i = 1, \ldots, m$ and $j = 1, \ldots, n$

where

m: Number of stations

n: Number of data records

For any Houston site, we used the data from 6 of the 7 available sites to develop the concentration estimates for a time period (n = up to 365 days for 24-hr averages or up to 8760 hours for hourly averages). We then evaluated the performance of the 7 models to determine the best method for use in estimating the annual and 24-hr average background concentrations at the site. The same data processing procedure was applied to the Fort Worth site.

Method 1: Single Station Estimate, $y_{i,j}^{s}$,

$$y_{i,j}^{s} = x_{k,j} \ (k \neq i)$$

The single station approach looks for a station that best represents the background concentration for the project site. Factors to be considered in selecting the best representative site for the project site include distance to the project area, upwind-downwind location, similarity in topology, land-use, and meteorology, and mix of sources. These factors are listed in Table 1 for both sites, in addition to the surface wind statistics which are displayed as wind roses in Figure 1. Surface or boundary layer parameters such as surface albedo, Bowen ratio, and roughness for the sites were obtained from the AERSURFACE model (*8, 9, 10, 11*). The surface characteristics within 5 km of all sites are judged to be very similar. A close look of the land use distribution indicates that most of the sites can be described as residential communities of high and low intensity combined with moderate commercial, industrial, transportation facilities. Without a clear distinction in the topologic and meteorological conditions among these sites, the most representative single station was selected based only on the shortest distance to the project site (Table 1).

TABLE 1 Topologic and land use characteristics for the Houston background concentration sites

Station ID	**Reference**	HOU 1	HOU 2	HOU 3	HOU 4	HOU 5	HOU 6	HOU 7	**Reference**	FTW 1	FTW 2	FTW 3	FTW 4
Station Name	**Houston North Loop**	Houston Aldine	Clinton	Park Place	Houston East	Houston Deer Park #2	Baytown	Conroe Relocated	**Fort Worth California Parkway North**	Haws Athletic Center	Arlington Municipal Airport	Denton Airport South	Midlothian OFW
Topology													
- Dist. to C1052 (km)	**0**	11.28	15.45	16.73	16.9	29.77	34.76	59.55	**-**	-	-	-	-
- Dist. to C1053 (km)	-	-	-	-	-	-	-	-	**0**	35.57	23.5	62.93	35.55
- Dist. to Center Nearest Hwy (km)	**0.06**	1.21	0.8	0.42	0.27	0.71	1.19	0.92	**0.06**	0.08	2.35	1.26	0.51
- AADT	**198,460**	227,735	169,411	23,063	188,783	51,368	56,320	10,030	**169,890**	26,899	202,837	20,260	35,344
Surface Charc., R=1 km													
- Albedo	**0.17**	0.16	0.17	0.17	0.17	0.17	0.15	0.15	**0.17**	0.17	0.17	0.18	0.17
- Bowen Ratio	**1.08**	0.84	1.03	1.04	0.92	0.84	0.49	0.74	**0.95**	0.99	0.68	0.62	0.59
- Surface Roughness	**0.581**	0.252	0.371	0.461	0.392	0.251	0.238	0.145	**0.436**	0.174	0.145	0.107	0.099
Surface Charc., R=5 km													
- Albedo	**0.17**	0.16	0.17	0.17	0.17	0.17	0.15	0.15	**0.17**	0.17	0.17	0.18	0.17
- Bowen Ratio	**1.08**	0.84	1.03	1.04	0.92	0.84	0.49	0.74	**0.95**	0.99	0.68	0.62	0.59
- Surface Roughness	**0.509**	0.343	0.349	0.438	0.35	0.222	0.087	0.429	**0.364**	0.279	0.15	0.11	0.11
Land Use Charc. (%), 121 km^2													
- Open Water	**0.002**	0.145	2.319	0.892	3.264	0.208	23.216	0.379	**0.245**	2.460	3.936	0.784	1.486
- Perennial Ice/Snow	**0.000**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	**0.000**	0.000	0.000	0.000	0.000
- Low Intensity Residential	**36.613**	32.976	24.014	31.322	21.545	20.795	14.019	9.408	**37.157**	39.371	19.557	7.567	2.839
- High Intensity Residential	**34.005**	15.170	24.614	22.456	14.746	16.932	9.167	5.030	**20.337**	22.132	6.608	3.114	0.186
- Commercial/ Industrial/ Transportation	**13.990**	6.822	29.860	26.715	29.544	13.668	15.171	7.147	**13.012**	20.493	6.414	2.580	2.616
- Barren Rock/ Sand/ Clay	**0.007**	0.032	0.103	0.021	0.202	0.502	1.023	0.096	**0.010**	0.003	0.040	0.011	0.024
- Quarries/ Strip Mines/ Gravel Pits	**0.000**	0.000	0.000	0.001	0.000	0.010	0.000	0.085	**0.000**	0.000	0.000	0.000	1.110
- Transitional	**0.000**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	**0.000**	0.000	0.000	0.000	0.000
- Deciduous Forest	**0.521**	2.510	3.438	5.181	3.477	9.718	6.876	7.649	**1.001**	0.825	4.089	2.216	4.007
- Evergreen Forest	**0.305**	2.900	0.065	0.408	0.310	0.271	0.121	15.805	**1.061**	1.220	4.774	3.100	5.144
- Mixed Forest	**4.249**	16.561	2.411	0.914	10.449	1.003	3.474	40.039	**0.289**	0.304	1.471	1.228	1.853
- Shrubland	**0.000**	0.000	0.001	0.192	0.000	0.514	0.000	0.000	**0.112**	0.036	0.160	0.166	0.503
- Orchards/ Vineyards/ Other	**0.000**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	**0.000**	0.000	0.000	0.000	0.000
- Grassland/ Herbaceous	**0.000**	0.000	0.000	0.054	0.000	0.573	0.000	0.001	**3.854**	2.120	9.921	10.767	7.509
- Pasture/ Hay	**5.957**	14.724	6.292	4.406	9.550	23.517	17.865	12.100	**15.397**	5.693	31.949	55.929	50.829
- Row Crops	**0.000**	0.010	0.044	0.061	0.118	0.609	0.165	0.109	**4.639**	0.022	8.235	4.739	20.071
- Small Grains	**0.000**	0.000	0.000	0.000	0.000	0.000	0.006	0.000	**0.701**	0.031	0.978	6.716	1.096
- Fallow	**0.000**	0.000	0.000	0.000	0.000	0.000	0.000	0.000	**0.000**	0.000	0.000	0.000	0.000
- Urban/ Recreatinal Grasses	**4.327**	8.070	6.760	7.355	6.679	11.597	7.334	1.686	**2.185**	4.736	1.866	1.081	0.728
- Woody Wetlands	**0.000**	0.000	0.000	0.000	0.000	0.000	0.000	0.184	**0.000**	0.137	0.000	0.000	0.000
- Emergent Herbaceous Wetlands	**0.024**	0.079	0.079	0.023	0.117	0.083	1.562	0.281	**0.000**	0.418	0.000	0.001	0.000

Method 2: Arithmetic Mean Estimate, $y_{i,j}$,

The arithmetic mean approach provides an estimate by taking the average of concurrent data from all available background sites.

$$y_{i,j} = \frac{1}{m-1}[(\sum_{k=1}^{m} x_{k,j}) - x_{i,j}]$$

Method 3: Weighted Mean Estimate by Inverse Distance, $z_{i,j}$,

This method provides an average weighted by the inverse of the distance to the project site. Table 2 summarizes the distances between paired sites for the two project areas. The distance matrix is defined as $Distance_{i,k} = Distance\ between\ station\ i\ and\ k$ and the weighting factor $Weight1_{i,k}$ is

$$Weight1_{i,k} = \frac{\frac{1}{Distance_{i,k}}}{\sum_{\substack{k=1 \\ k\neq i}}^{m} \frac{1}{Distance_{i,k}}}, k \neq i$$

and $$Weight1_{i,k} = 0, \quad if\ k = i$$

The estimate $z_{i,j}$ is obtained as

$$z_{i,j} = \sum_{k=1}^{m} (x_{k,j} \cdot Weight1_{i,k})$$

TABLE 2 Distance Matrices for the background stations

	Houston: Distance Matrix (miles)						
Station	1	2	3	4	5	6	7
1	0	12.2	14.9	11.1	19.9	19.9	31.5
2	12.2	0	3.95	3.24	8.92	13.8	43.6
3	14.9	3.95	0	7.17	10.1	16.9	46.4
4	11.1	3.24	7.17	0	8.73	11.4	41.9
5	19.9	8.92	10.1	8.73	0	9.07	50.1
6	19.9	13.8	16.9	11.4	9.07	0	46.4
7	31.5	43.6	46.4	41.9	50.1	46.4	0

	Fort Worth: Distance Matrix (miles)			
Station	1	2	3	4
1	0	16.4	26.5	32.8
2	16.4	0	12.5	39.3
3	26.5	12.5	0	51.7
4	32.8	39.3	51.7	0

Method 4: Weighted Mean by Inverse Distance Squared, $w_{i,j}$,

This method is similar to Method 3 except the weighting factor is represented by the inverse of the distance to the square. The weighting factor $Weight2_{i,k}$ is defined as

$$Weight2_{i,k} = \frac{\frac{1}{(Distance_{i,k})^2}}{\sum_{\substack{k=1 \\ k\neq i}}^{m} \frac{1}{(Distance_{i,k})^2}}, k \neq i$$

and $$Weight2_{i,k} = 0, \quad if\ k = i$$

The estimate $w_{i,j}$ is

$$w_{i,j} = \sum_{k=1}^{m} (x_{k,j} \cdot Weight2_{i,k})$$

Method 5: Normalized Arithmetic Mean Estimate, $y_{i,j}^{N}$,

This method seeks to preserve the trend of the time series data at each station by normalizing the time series data at each station by its own annual average. The normalized data $X_{i,j}$ becomes

$$X_{i,j} = \frac{x_{i,j}}{\frac{1}{n}\sum_{j=1}^{n} x_{i,j}}, \qquad i = 1, \cdots, m, \qquad j = 1, \cdots, n$$

The normalized estimate becomes $Y_{i,j}$ and the estimate can be retrieved by multiplying $Y_{i,j}$ by the annual average of $y_{i,j}$.

$$Y_{i,j} = \frac{1}{m-1}[(\sum_{k=1}^{m} X_{k,j}) - X_{i,j}]$$

$$y_{i,j}^{N} = Y_{i,j} \cdot (\frac{1}{n} \cdot \sum_{1}^{n} y_{i,j})$$

Method 6: Normalized Inverse Distance Estimate, $z_{i,j}^{N}$,

Similar to Method 5, the normalized inverse distance estimate is $Z_{i,j}$ and the estimate can be retrieved by multiplying $Z_{i,j}$ by the annual average of $y_{i,j}$.

$$Z_{i,j} = \sum_{k=1}^{m}(X_{k,j} \cdot Weight1_{i,k})$$

$$z_{i,j}^{N} = Z_{i,j} \cdot (\frac{1}{n} \cdot \sum_{1}^{n} z_{i,j})$$

Method 7: Normalized Inverse Distance Squared Estimate, $w_{i,j}^{N}$,

The normalized inverse distance squared estimate $W_{i,j}$ and the estimate $w_{i,j}$ is:

$$W_{i,j} = \sum_{k=1}^{m}(X_{k,j} \cdot Weight2_{i,k})$$

$$w_{i,j}^{N} = W_{i,j} \cdot (\frac{1}{n} \cdot \sum_{1}^{n} w_{i,j})$$

PERFORMANCE EVALUATION

For each method, 7 sets of concentration estimates, each was computed from the other 6 sites, were computed for comparison to the observations for Houston sites, and 4 for Fort Worth sites. A total of 308 sets of data were obtained for 2 project areas, 2 years of data (2015 and 2016), and 2 averaging periods (hourly and 24-hr averages). Statistical analyses were conducted to evaluate the performance of each estimation methodology. It was found that the results for both years and both time periods are consistent and therefore only the results developed for the 24-hr averaging period at the two cities are presented in this paper.

Many statistical measures have been used to quantify the accuracy of prediction for actual observation (*12*). The degree of correspondence between the modeled and observed values is defined differently by characteristics of the performance measures. Some statistics measure average magnitude of the errors (e.g., mean bias error, mean error, and root mean squared error), while others quantify the amount of variation in a set of predictions or observations (e.g., standard deviation, variance). In this paper we limit our choice of metrics to those representing the magnitude of the errors, but normalized metrics by the corresponding mean of the observations, which allows for unitless measurements.

- *Normalized Mean Bias* (*NMB*) is a measure of the average deviation from actual observation (between -1 and ∞). The NMB represents the average model bias normalized by the mean of observations, with considering (positive and negative) direction of the errors.

$$NMB_i = \frac{\frac{1}{n}\sum_{j=1}^{n}(y_{i,j} - x_{i,j})}{\frac{1}{n}\sum_{j=1}^{n} x_{i,j}}$$

- *Normalized Mean Error* (*NME*) is a measure of the averaged absolute deviation without considering direction of differences between prediction and observation (between 0 and ∞). Contrary to the NMB, in the NME the absolute deviations are summed instead of the differences, and we have equal weight of underestimation and overestimation.

$$NME_i = \frac{\frac{1}{n}\sum_{j=1}^{n}|y_{i,j} - x_{i,j}|}{\frac{1}{n}\sum_{j=1}^{n} x_{i,j}}$$

- *Normalized Root Mean Square Error* (*NRMSE*) is a measure of the square root of the average of the squared differences between prediction and actual observation (between 0 and ∞). The root mean squared error (RMSE) represents standard deviation of the differences between predicted and observed values. In RMSE the squared differences are averaged, and the measure gives a relatively high weight to large errors compared with the mean error.

$$NRMSE_i = \frac{RMSE_i}{\frac{1}{n}\sum_{j=1}^{n} x_{i,j}}, \text{where } RMSE_i = \sqrt{\frac{\sum_{j=1}^{n}(y_{i,j} - x_{i,j})^2}{n}}$$

In practice, the statistics are based on finite samples of several sets of concentrations, and do not represent sampling variability. A statistical technique known as the *bootstrap* was used to account for the sampling variability in the predictions (*13*). The bootstrap is a resampling method to estimate standard error of a specific performance measure computed from resampled dataset. The bootstrap procedure follows the basic steps:

1) *Resample a given data set a specified number of times,* i.e., generate new estimates $x_{i,j}^{(1)}, x_{i,j}^{(2)}, \cdots, x_{i,j}^{(B)}$ where *B* is the bootstrap sample size (e.g., *B*=5,000).

2) *Calculate a specific statistic from each sample,* i.e., calculate 5,000 sets of the estimates for each method. For example, generate $y_{i,j}^{(1)}, y_{i,j}^{(2)}, \cdots, y_{i,j}^{(B)}$ and calculate 5,000 sets of NRMSE based on the prediction using inverse distance.

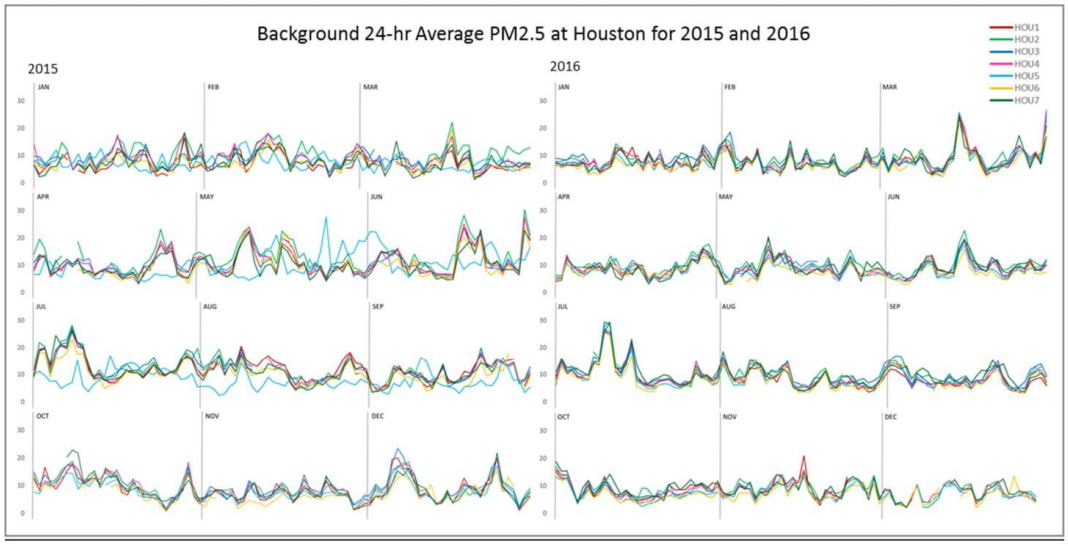
3) *Find the standard deviation of the distribution of that statistic.* Bootstrapped standard deviations are obtained to compare sampling variabilities of the statistics (i.e., NRMSE) between the 7 methods.

RESULTS AND DISCUSSION

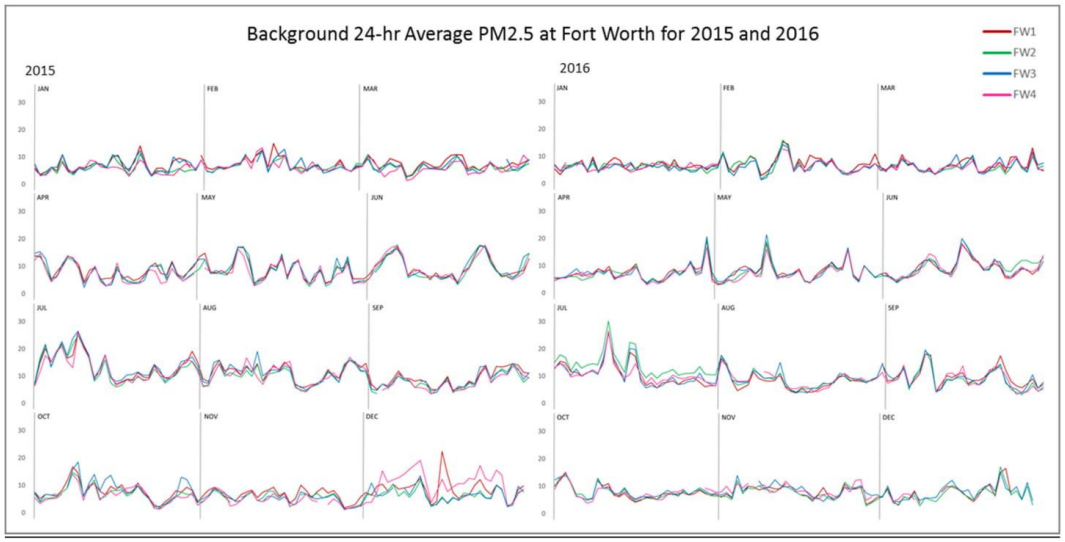
The TCEQ $PM_{2.5}$ network is designed to meet area, near-road, regional background, and regional transport requirements under the SLAMS network and NCore requirements for $PM_{2.5}$ (TCEQ 2017). All background stations selected in this study were designed to support compliance with NAAQS and research in air pollution studies (Title 40 CFR Part 58, Appendix D). They are located at a distance clear of highway emissions (AADT > 50,000) and are representative of the area's background concentration levels (Table 1). Figure 2 shows the 2-year quarterly time-series plots of the 24-hour average $PM_{2.5}$ concentrations measured in the Houston and Fort Worth areas. It is interesting to observe that some high $PM_{2.5}$ episodes recurred regardless of the year. For instance, high $PM_{2.5}$ days occurred around March 17, May 15, June 18, and July 7 each year for the Houston area, and around June 17, July 7, and Oct. 8 for the Fort Worth area. Furthermore, the time series data for the background concentrations appears to be strongly correlated with a well-defined trend for a project area. This is a clear indication of persistent meteorology and emission patterns prevailing in both areas. A seasonal $PM_{2.5}$ episode was observed in both cities around the Independence Day when high traffic, intense outdoor BBQ activities, and excessive fireworks (all considered major sources of $PM_{2.5}$ emission) took place.

Annual Average $PM_{2.5}$ Concentrations

The annual average concentration at each background station is listed in Table 3 for the Houston and Fort Worth areas. The annual average developed from the hourly dataset differs slightly from that developed from the 24-hr dataset due to the treatment of missing data in constructing the dataset, as described in the previous section. Because the normalized methods (Methods 5, 6, and 7) preserve the same annual averages as the non-normalized ones (Methods 2, 3, and 4) their mean values as well as the comparison statistics are the same as their respective counterparts. These values are therefore not included in the tables. Annual averages for the same year vary within a narrow range (±10% from the all site average) for both cities. Annual averages at the same site fluctuate from one year (2015) to another (2016) with a magnitude of up to 15%. Given the small increment (<15%) in $PM_{2.5}$ concentrations observed at near-road monitors that can be attributed to the traffic emissions (*3*), this magnitude of variation is significant especially when it is used for PM design value calculation that hinges predominantly on the background concentration.



a) Houston Area



b) Fort Worth

FIGURE 2 24-hr average $PM_{2.5}$ data for the Houston and Fort Worth Areas

375 **TABLE 3 Method comparison based on $PM_{2.5}$ annual averages at 7 sites in Houston**

			Annual Average, µg/m3					All sites	NMB	NME	RMSE	NRMSE
Method	1	2	3	4	5	6	7	Average	- - -	- - -	µg/m3	- - -
Hourly Data	Houston 2015											
Observation	**9.40**	**11.35**	**10.59**	**10.24**	**8.44**	**8.59**	**9.59**	**9.74**				
Method 1	10.24	10.24	11.35	11.35	10.24	8.44	9.40	**10.18**	0.04	0.09	1.00	0.10
Method 2, 5	9.80	9.47	9.60	9.66	9.96	9.93	9.77	**9.74**	**0.00**	0.10	1.14	0.12
Method 3, 6	10.02	9.87	10.14	10.17	10.09	9.86	9.78	**9.99**	0.03	0.08	1.01	0.10
Method 4, 7	10.23	10.16	10.67	10.72	10.14	9.69	9.78	**10.20**	0.05	**0.08**	**0.96**	**0.10**
24-hr Data												
Observation	**9.48**	**11.44**	**10.63**	**10.33**	**8.47**	**8.70**	**9.64**	**9.81**				
Method 1	10.33	10.33	11.44	11.44	10.33	8.47	9.48	**10.26**	0.05	0.09	1.03	0.10
Method 2, 5	9.87	9.54	9.68	9.73	10.04	10.00	9.84	**9.81**	**0.00**	0.10	1.15	0.12
Method 3, 6	10.10	9.94	10.22	10.24	10.17	9.92	9.85	**10.06**	0.03	0.08	1.02	0.10
Method 4, 7	10.30	10.23	10.75	10.79	10.22	9.75	9.85	**10.27**	0.05	**0.08**	**0.97**	**0.10**
Hourly Data	Houston 2016											
Observation	**8.62**	**9.87**	**9.86**	**8.88**	**8.19**	**7.72**	**9.33**	**8.93**				
Method 1	8.88	8.88	9.87	9.87	8.88	8.19	8.62	**9.03**	0.01	**0.07**	**0.68**	**0.08**
Method 2, 5	8.98	8.77	8.77	8.93	9.05	9.13	8.86	**8.93**	**0.00**	0.09	0.88	0.10
Method 3, 6	9.07	8.97	9.05	9.21	9.03	9.01	8.86	**9.03**	0.01	0.08	0.79	0.09
Method 4, 7	9.15	9.13	9.39	9.53	9.03	8.87	8.85	**9.14**	0.02	0.08	0.73	0.08
24-hr Data												
Observation	**8.69**	**9.96**	**9.96**	**8.97**	**8.25**	**7.79**	**9.41**	**9.00**				
Method 1	8.97	8.97	9.96	9.96	8.97	8.25	8.69	**9.11**	0.01	**0.07**	**0.69**	**0.08**
Method 2, 5	9.05	8.84	8.84	9.01	9.13	9.21	8.93	**9.00**	**0.00**	0.09	0.90	0.10
Method 3, 6	9.15	9.05	9.12	9.29	9.11	9.08	8.94	**9.11**	0.01	0.08	0.80	0.09
Method 4, 7	9.23	9.22	9.47	9.61	9.11	8.95	8.93	9.22	0.02	0.08	0.74	0.08
Hourly Data	Ft. Worth 2015											
Observation	8.91	7.86	8.46	8.08	---	---	---	**8.33**				
Method 1	8.08	8.08	7.86	7.86	---	---	---	**7.97**	-0.04	0.06	0.53	**0.06**
Method 2, 5	8.13	8.48	8.28	8.41	---	---	---	**8.33**	**0.00**	0.06	0.53	**0.06**
Method 3, 6	8.11	8.33	8.08	8.19	---	---	---	**8.18**	-0.02	**0.05**	**0.50**	**0.06**
Method 4, 7	8.09	8.26	7.96	8.03	---	---	---	**8.08**	-0.03	**0.05**	0.52	**0.06**
24-hr Data												
Observation	8.84	7.83	8.42	8.03	---	---	---	**8.28**				
Method 1	8.03	8.03	7.83	7.83	---	---	---	**7.93**	-0.04	**0.05**	0.52	**0.06**
Method 2, 5	8.10	8.43	8.24	8.37	---	---	---	**8.28**	**0.00**	0.06	0.52	**0.06**
Method 3, 6	8.07	8.29	8.04	8.15	---	---	---	**8.14**	-0.02	**0.05**	**0.49**	**0.06**
Method 4, 7	8.05	8.22	7.93	7.99	---	---	---	**8.05**	-0.03	**0.05**	0.51	**0.06**
Hourly Data	Ft. Worth 2016											
Observation	8.05	8.11	8.03	7.90	---	---	---	**8.02**				
Method 1	7.90	7.90	8.11	8.11	---	---	---	**8.01**	0.16	0.16	1.43	0.18
Method 2, 5	8.02	7.99	8.02	8.07	---	---	---	**8.02**	**0.00**	**0.01**	**0.11**	**0.01**
Method 3, 6	8.01	7.97	8.04	8.08	---	---	---	**8.03**	**0.00**	**0.01**	0.12	0.02
Method 4, 7	8.01	7.96	8.06	8.10	---	---	---	**8.03**	**0.00**	**0.01**	0.13	0.02
24-hr Data												
Observation	8.07	8.16	8.08	7.93	---	---	---	**8.06**				
Method 1	7.93	7.93	8.16	8.16	---	---	---	**8.04**	-0.01	0.02	0.18	0.02
Method 2, 5	8.06	8.03	8.05	8.11	---	---	---	**8.06**	**0.00**	**0.01**	**0.11**	**0.01**
Method 3, 6	8.05	8.01	8.08	8.12	---	---	---	**8.07**	**0.00**	**0.01**	0.13	0.02
Method 4, 7	8.04	7.99	8.10	8.14	---	---	---	**8.07**	**0.00**	**0.01**	0.13	0.02

376

377 Table 3 also shows the values for the performance metrics NMB, NME, RMSE, and NRMSE.
378 Smallest number in each column is highlighted in bold. The non-normalized and normalized

inverse distance squared methods appear to provide the best estimates except occasionally (in the case of Houston sites in 2016) Method 1, based on the shortest distance to the target site and, also provided good estimates.

24-hr Average $PM_{2.5}$ Concentrations

Overall sums of the performance measures across all sites for the 24-hr $PM_{2.5}$ concentration estimation are reported in Table 4 for both cities. NMB and NME for the 24-hr data represent the deviations of daily $PM_{2.5}$ concentrations from the observations. Normalized methods significantly reduce the bias from the observations, as seen in the tables where NMBs and NMEs for normalized methods (Methods 5-7) are much less than that reported for non-normalized methods (Methods 1-4). On the contrary, the RMSE and NRMSE show the spread of model predictions with respect to the 24-hr observations, where the smaller the value the better the prediction. All normalized methods perform better than the non-normalized methods, although the NRMSE values differ only slightly among the normalized methods. The method using normalized arithmetic mean performs well for both areas in year 2015, based on the RMSE and NRMSE calculations. Accuracy improved best (i.e., with having the smallest measure statistics) in the model predictions using normalized arithmetic mean for 2015 Houston, 2015 and 2016 Fort Worth data. The predictions using the normalized inverse distance method are most accurate in estimating 2016 Houston observations, based on either RMSE or NRMSE results. We observed significant tendency of overestimation in Houston Site 5, and moderate tendency in Houston Site 6 and Fort Worth Site 4 for year 2015. Table 4 shows the NRMSE values by stations for both project areas and years. The NRMSE value for any method for Site 5 in Houston 2015 data appears to be 2-3 times greater than the rest of sites, up to 2 times for Site 6 of Houston and Site 4 of Fort Worth. This poor accuracy can be attributed to a few outliers in the data or different pollution pattern caused by local sources. As seen in Figure 1, Houston Site 5 (Deer Park) is located approximately 6 miles from the Tabbs Bay and 20 miles from the city center, whereas Houston Site 6 (Baytown) is located at where the Buffalo Bayou enters the Galveston Bay and is approximately 30 miles from the city center. These locations are constantly downwind of the daily sea breeze and are close to many petroleum refinery facilities. Their geographical locations being away from the city center and unique local emission sources maybe the reasons for the significant deviation in $PM_{2.5}$ concentrations compared to other stations, although the deviation was less noticeable in 2016. In general, single station method (Method 1) performs worse than the multiple stations methods.

Bootstrapped standard deviations based on the NRMSE for assessing the sampling variabilities in between the methods are shown in Table 4. Although normalized arithmetic mean and inverse distance methods have more improved accuracy than non-normalized methods, as shown in the summary of NRMSEs, the standard deviations of our normalized methods do not vary significantly compared with non-normalized methods.

TABLE 4 Summary of performance measures and standard deviations (SD) using bootstrap resampling with bootstrap sample size, *B*=5000, at Houston (top) and Fort Worth (bottom).

	Year 2015					*Year 2016*				
Houston	NMB	NME	RMSE	NRMSE	SD	NMB	NME	RMSE	NRMSE	SD
Method 1	0.354	1.567	20.210	2.160	0.022	0.113	0.884	10.351	1.148	0.009
Method 2	0.082	1.355	17.086	1.798	0.018	0.061	0.918	10.342	1.155	0.008
Method 3	0.250	1.304	16.410	1.740	0.018	0.134	0.869	9.879	1.105	0.008
Method 4	0.388	1.310	16.558	1.761	0.018	0.214	0.872	9.948	1.112	0.008
Method 5	**0.000**	1.177	**1.579**	**1.579**	**0.014**	**0.000**	0.719	0.937	0.937	0.008
Method 6	0.000	**1.177**	1.582	1.582	0.014	0.000	**0.699**	**0.911**	**0.911**	**0.008**
Method 7	0.000	1.199	1.616	1.616	0.014	0.000	0.718	0.933	0.933	0.008
Ft. Worth										
Method 1	-0.162	0.654	7.957	0.964	0.016	-0.008	0.575	6.409	0.799	0.015
Method 2	0.012	0.548	6.617	0.800	0.016	0.000	0.458	5.040	0.628	0.011
Method 3	-0.059	0.542	6.575	0.796	0.015	0.002	0.471	5.205	0.649	0.011
Method 4	-0.104	0.543	6.632	0.803	0.015	0.004	0.491	5.450	0.680	0.012
Method 5	**0.000**	**0.519**	**0.764**	**0.764**	0.015	**0.000**	**0.457**	**0.622**	**0.622**	**0.010**
Method 6	0.000	0.521	0.766	0.766	0.015	0.000	0.470	0.641	0.641	0.011
Method 7	0.000	0.524	0.773	0.773	**0.015**	0.000	0.490	0.669	0.669	0.011

Highest Ten 24-hr Average $PM_{2.5}$ Concentrations

High 24-hour average $PM_{2.5}$ concentrations at near-road monitors are of particular interest to transportation engineers because a 24-hour $PM_{2.5}$ design value is defined by the U.S. EPA as the average of the 98th percentile values in a year over 3 consecutive years. In determining if a transportation project is in compliance with the NAAQS, the sum of the 98th background concentration and the 98th percentile of the $PM_{2.5}$ concentration estimate predicted from air dispersion modeling of the transportation project enhanced emissions is compared to the NAAQS, regardless whether the 98th percentile background concentration occurs concurrently to the 98th percentile traffic emission induced PM concentration. These values encompass the 98th percentile value of a year's $PM_{2.5}$ record, which is the background 24-hr average concentration used in developing the $PM_{2.5}$ design value in a hot-spot analysis. Accuracy in the estimation of the highest ten 24-hr average concentrations is evaluated separately in terms of NRMSE. The all-site averaged NRMSE for the highest 10 24-hr averages are shown below.

NRMSE				
Location		Houston	Fort Worth	
Method	2015	2016	2015	2016
1	0.22	0.11	0.17	0.17
2	0.20	0.11	0.16	0.11
3	0.18	0.11	0.16	0.12
4	**0.17**	**0.11**	**0.16**	**0.12**

5	0.21	0.11	0.16	0.11
6	0.19	0.11	0.16	0.12
7	**0.17**	**0.11**	**0.16**	**0.12**

All methods provide good estimates for the highest 10 values in a year except Method 1. Larger variability was observed in the calculated NRMSE values for Houston 2015 data due to poor quality of the data, as discussed previously, observed at 2 of the stations. Both Method 2 and Method 5 perform slightly better than other methods if one does not take the Houston 2015 data into consideration.

Figure 3 shows the comparison of observed and modeled $PM_{2.5}$ concentration estimates for Fort Worth Site 2 (figures for other sites are not presented in this paper). All methods capture the highest 5 concentrations (data above the 98^{th} percentile based on the available 252 days of data and circled in the figure) quite well while the worst method (Method 1) overestimated the peak values. Figure 4 provides illustrations of the comparison between predictions and observations at all sites for Methods 5 and 7. High concentrations observed at all sites are well captured by both methods. Although the differences between methods do not seem much (less than 20%) the impact on a hot-spot air quality conformity analysis could be significant because 1) the magnitude of this difference may be equivalent to or greater than the modeled concentration increment resulting from the project being analyzed for hot-spot analysis; and 2) the times of occurrences for the high background concentrations are predictable from the background stations such that the current application of a 98^{th} percentile background concentration from other stations as the background concentration at a target site, regardless of the time of occurrence, may be overly conservative and inaccurate.

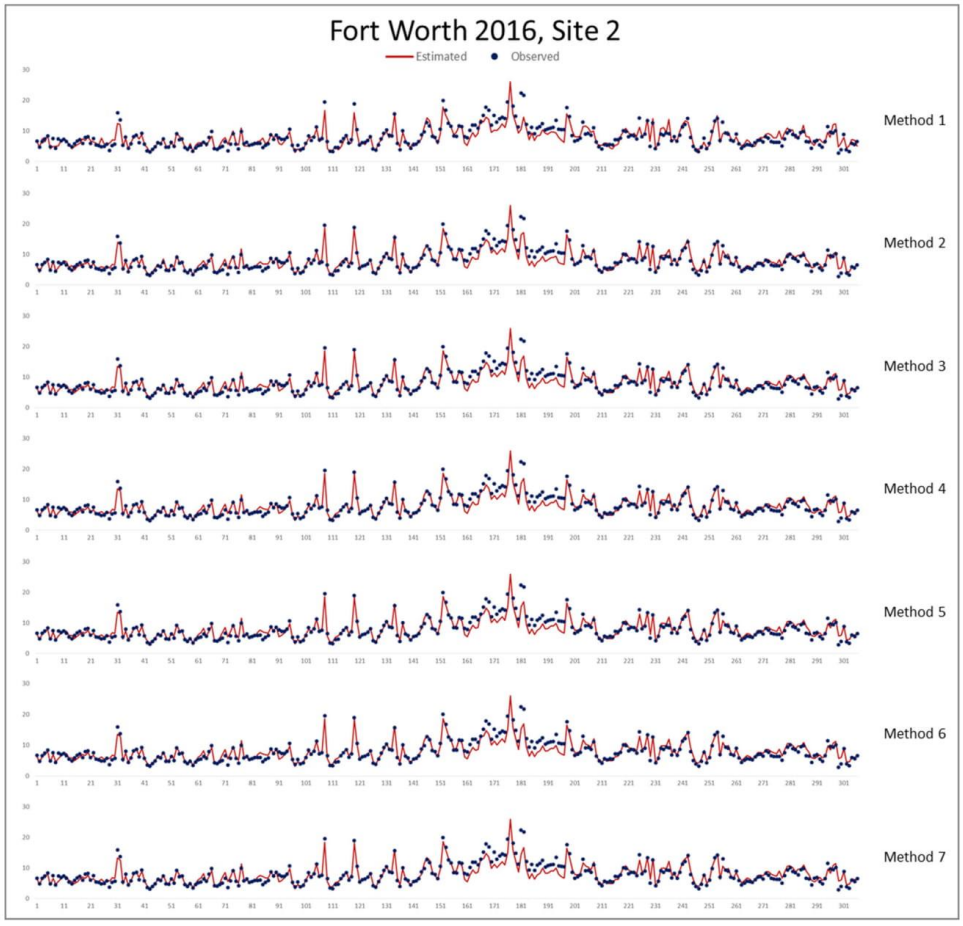
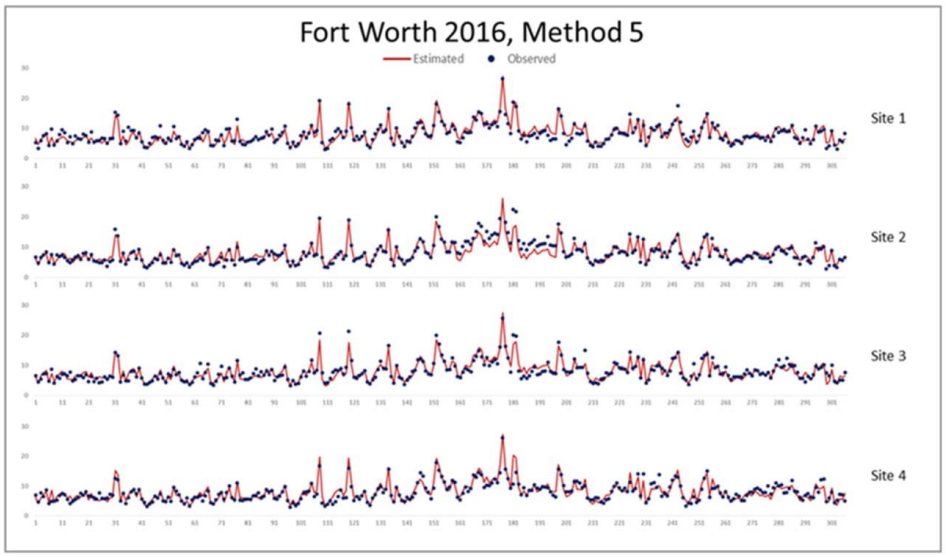
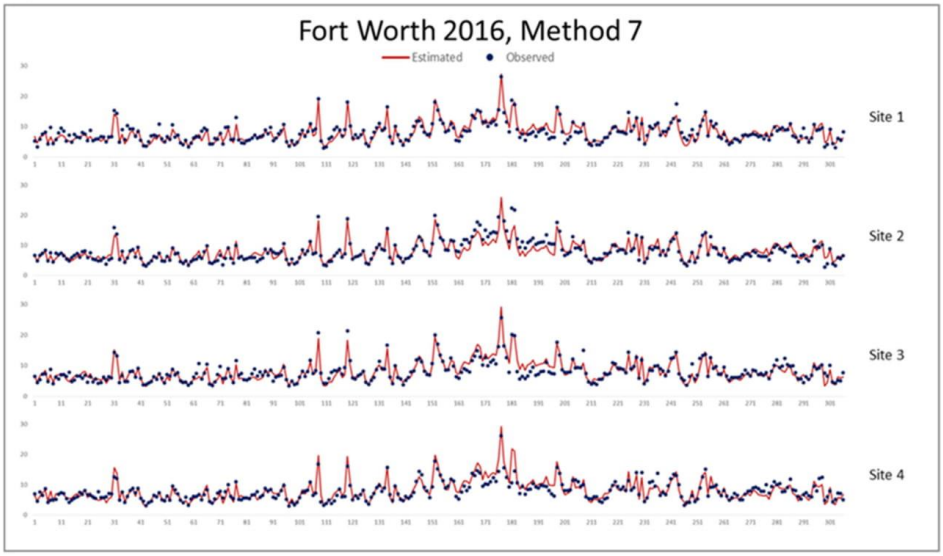


FIGURE 3 Comparison of predicted vs observed $PM_{2.5}$ concentrations by different methods



a) Method 5



b) Method 7

459

460 **FIGURE 4 Comparison of predicted vs observed $PM_{2.5}$ concentrations by Method 5 and**
461 **Method 7 for the Fort Worth sites**

462

CONCLUSION

Accurate estimation of background concentrations is a critical component for estimating the design value to show compliance with NAAQS for transportation conformity hot spot analysis. We evaluated 7 methods, with four of them suggested by EPA, for background concentration estimation using data from multiple background ambient monitoring stations and by resampling of the same dataset using bootstrapping technique. For the two identified project areas in Texas, we observed similar $PM_{2.5}$ pollution pattern at urban locations in the same city and found that the annual average $PM_{2.5}$ concentration developed from different methods are, in general, acceptable except the single station approach that is selected on the basis of on shortest distance to a target station. Normalized methods appear to perform better than non-normalized methods with higher accuracy. Among the normalized methods, predictions made by normalized inverse distance squared method appear to be slightly better than other models, based on the statistical metrics for annual, 24-hr, and highest 10 24-hr average $PM_{2.5}$ concentrations. Future research will include applying the methodology and evaluating how the methods perform against field observations at near-road sites.

AUTHOR CONTRIBUTION STATEMENT

The authors confirm contribution to the paper as follows: study conception and design: Li, Vallamsundar, Farzaneh; data collection: Chavez, Rangel, Urbina, Ramirez; analysis and interpretation of results: Li, Jeon, draft manuscript preparation: Li, Vallamsundar. All authors reviewed the results and approved the final version of the manuscript.

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Appendix C

Comparison of Modeled-to-Monitored $PM_{2.5}$ Exposure Concentrations Resulting from Transportation Emissions in a Near-Road Community

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ABSTRACT

Residents living in near-road communities are constantly exposed to traffic-related air pollutants. Their health could be adversely impacted by these pollutants both chronically and acutely. A near-road community is expected to observe significant spatial and temporal variations in pollutant concentrations, as air pollution resulting from emissions from major highways decreases rapidly from the highway.

This study conducted traffic and air quality measurements as well as emission and air dispersion modeling of transportation emission impacts in a near-road community. It was observed that a) $PM_{2.5}$ pollution in near-road communities is dominated by the regional background concentrations which account for more than 85% of the pollution; and b) only near-road receptors are affected by the traffic emissions from major highways while spatial and temporal variations of $PM_{2.5}$ concentrations in near-road communities are less influenced by local traffic. Modeled $PM_{2.5}$ concentrations were compared to monitored data. For a better transportation air quality impact assessment, higher quality traffic data such as time-specific traffic volume and fleet information as well as meteorological data such as site-specific surface meteorological and topographic conditions and higher quality upper air data could help yield more accurate concentration predictions. Modeled-to-monitored comparison shows that air quality impact in near-road communities resulting from traffic-related emissions are dominated by regional background concentrations.

Keywords: $PM_{2.5}$ Traffic Emissions, El Paso, AERMOD, MOVES

INTRODUCTION

Residents living in near-road communities are constantly exposed to traffic-related air pollutants and their health could be adversely impacted both chronically and acutely. Concerns for the health of populations exposed to traffic-related emissions of particles and gases have led the U.S. Environmental Protection Agency (EPA) to establish a near-road ambient monitoring program, carried out by the Texas Commission on Environmental Quality (TCEQ) as part of their Annual Monitoring Network Plan since 2014 (TCEQ, 2018). Exposure to the traffic pollutants in a near-road community could vary significantly spatially and temporally due to the various traffic emission sources and as the result of rapid dispersion from roadways. The time-resolved concentrations used in health outcome studies could mask the short-term effects of a pollutant on people's health. A temporal and spatial characterization of exposure concentrations would fill out the data gap between air pollution exposures and health outcome measurements for near-road communities.

This study is designed to test two hypotheses: 1) Urban near-road communities are exposed primarily to regional background air pollution and traffic emissions in the communities while the contribution of the traffic emissions to the total exposure concentrations is of limited fraction; 2) Only near-road receptors are affected by the traffic emissions from major highways while spatial and temporal variations of pollutant concentrations in near-road communities are dominated by local traffic. This study, thus, attempts to characterize community exposures for three traffic-related air pollutants ($PM_{2.5}$, NO_2, and ozone) with the objectives to 1) develop spatial and temporal pollutant concentration variation patterns, and 2) apportion the differences in exposure concentrations to background concentrations and that contributed from major highways.

METHODOLOGY AND STUDY DESIGN

This study was implemented in four phases in order to assess the exposure of the community living near a major highway An area of 1 mile by 1 mile was selected in the northeast part of the city. The area was selected based on the traffic conditions, proximity to the highway, and the direction of the prevailing winds. A community near Coldwell Elementary School along the Interstate Highway 54 was selected based on the known high annual average daily traffic volume (AADT) of 107,237 on I-54 and the low-income status of the community. Figure 1 shows the study domain of 1 mile by 1 mile.

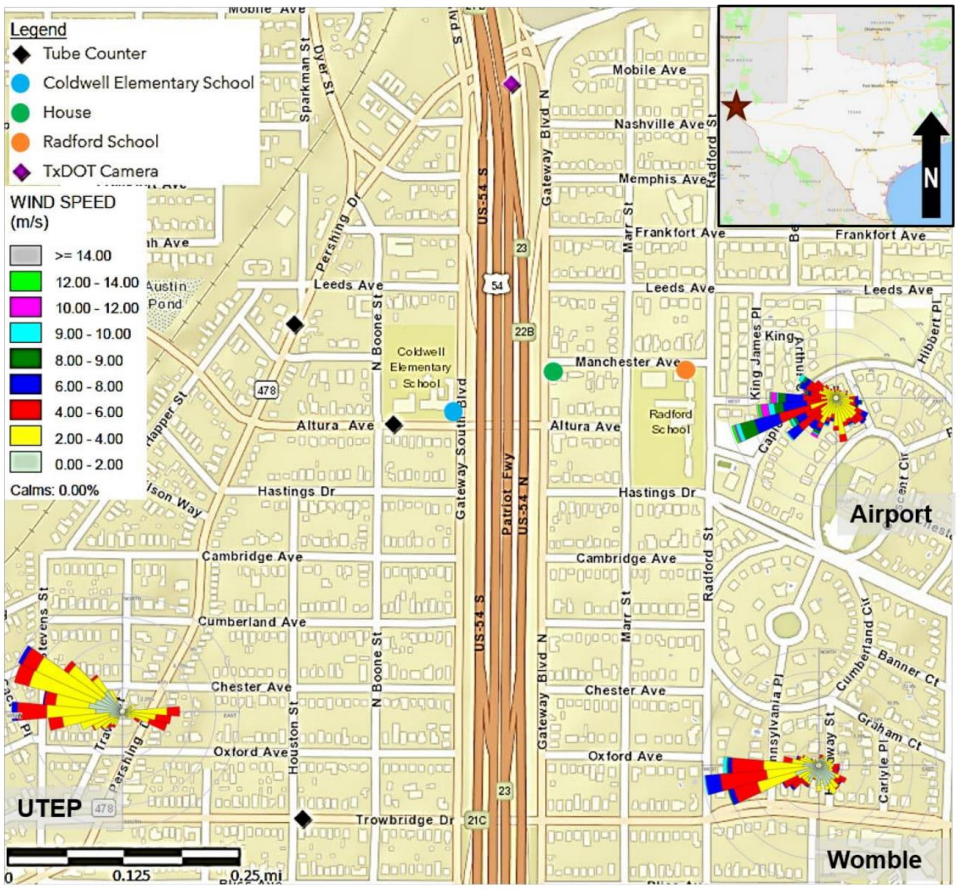


FIGURE 1 Map of Study Area with community Windrose Data

Phase 1: Traffic Data Collection

Limited traffic data was collected at 3 locations and at I-54 in the study domain. Vehicle volume counts were recorded using the TRAX Apollyon Counter/Classifier at 3 arterial roads in the study area (JAMAR Technologies, 2010). A set of two counters was placed at each of the three different locations, which were chosen for their higher impact of traffic. The two counter method provides volume data, vehicle classification data, and vehicle speed data. This data was recorded for each hour of the day. The data was used to supplement and calibrate the traffic data previously collected by the City of El Paso Transportation Department at different times and different locations in the study domain. Traffic data for I-54 was obtained by counting vehicles from the video traffic camera footage recorded by the Texas Department of Transportation (TxDOT). Hourly vehicle class and number were manually counted by 3 researchers operating independently at different times to avoid human errors and ensure high data quality.

Phase 2: Air Pollution Measurements

Air quality data was collected using three different monitoring instruments at each of the three sites. The pollutants analyzed in this study were nitrogen dioxide, (NO_2), particulate matter ($PM_{2.5}$, PM_{10}), and Ozone (O_3). Nitrogen dioxide was measured using 2B Technologies $NO_2/NO/NO_x$ Monitor™ (2B Technologies, 2017a). Ozone was measured using 2B Technologies Model 202 Ozone Monitor™ (2B Technologies, 2017b). Particulate matter was measured using GRIMM Portable Laser Aerosolspectrometer and Dust Monitor (GRIMM, 2010). The $PM_{2.5}$ sensors also provide particle counts for different particle size ranges which provides additional information for the understanding of the PM health effects. Ozone is an EPA regulated criteria pollutant, although not directly emitted from the vehicles but is a photochemical product involving another critical traffic pollutant, NO_2. Placement of the air quality monitors required protection from wind and rain, as well as a housing unit to provide shade. Only $PM_{2.5}$ data is presented and discussed in this paper.

Calibration of the instruments was done in the week before and after the study period. All monitoring instruments were placed alongside the continuous air monitoring station (CAMS 12) operated by the TCEQ located on the UTEP campus. The same process was used for pre-calibration as for post-calibration. This set-up remained identical during the study period to reduce any variance caused by the housing of the units. Table 1 shows the calibration equations and how well the monitor data correlates with measured and validated CAMS data. Calibration data was used to correct the air quality data collected from the near-road study, as is discussed in the following chapter.

TABLE 1 $PM_{2.5}$ Calibration Data

	$PM_{2.5}$		
Instrument	**1 (House)**	**2 (Radford)**	**3 (Coldwell)**
Calibration Equation	y = 0.757x + 3.0454	y = 0.7288x + 2.2831	y = 1.2163 + 2.7014
r-value	R^2 = 0.9524	R^2 = 0.9623	R^2 = 0.9585

Phase 3: Emission Modeling

The traffic data generated from field traffic counting at arterial roads as well as the digital data video counting of I-54 traffic were used to generate vehicle emissions factors for AERMOD air exposure concentration estimates. The MOVES emission model was used to generate emissions estimates for all interstate highway, arterial roads, and frequently traveled surface roads in the model domain. Temperature, humidity, vehicle speed, vehicle volume, and vehicle fleet mix information were all considered as variables in the MOVES modeling. Each model run corresponds to one hour during the four weekday time periods (morning peak, midday, evening peak and overnight) for a representative month during the analysis year. The four weekday time periods are:

- Morning peak emissions based on data 7 a.m. to 9 a.m.
- Midday emissions based on data from 10 a.m. to 3 p.m.
- Evening peak emissions based on data from 4 p.m. to 7 p.m.
- Overnight emissions based on data from 8 p.m. to 6 a.m.

A specific hour within each of the four time periods was modeled and the results were extrapolated to cover the entire day. The hour with the highest traffic activities within each time period is modeled for example, 7-8 a.m. during the morning peak time period. The time span covered is the month of May and the distinct time periods are morning, midday, evening, and

overnight. Emissions factors (EFs) were calculated for a typical weekday, Saturday, and Sunday during the month. A total of 12 MOVES runs were conducted according to all the parameters of the study for each scenario. The speed range is from 20 mph to 60 mph based on posted speed limits in the study link sources.

The EFs produced by MOVES are in terms of grams/hour for each peak time period and included separate EFs for running exhaust emissions and brake wear and tire wear. EFs for re-entrained dust were calculated for the different types of roads in the study and added to MOVES generated EFs. Re-suspended dust can be quantified using EPA's AP-42 method (EPA, 2015).

Phase 4: Air Dispersion Modeling

The AERMOD modeling system includes the use of two regulatory components, a meteorological preprocessor (AERMET), and an air dispersion processor (AERMOD). Meteorological data is needed not only for AERMOD but also for MOVES modeling. Land use data was downloaded from the United States Geological Survey and both hourly surface meteorological data from the El Paso International Airport and upper air soundings and minute data from the regional Santa Teresa Airport were used in AERMET to generate the on-site meteorological data for this study. The following modeling parameters and options were used in AERMOD:

- Passive Pollutant
- Line source, characterized by 180 links, representation for the I-54 highway section
- Urban environment
- Flat Terrain
- Ground-level Release
- Ground-level Receptor
- Initial Horizontal and Vertical Dispersion
- Site-specific Meteorology

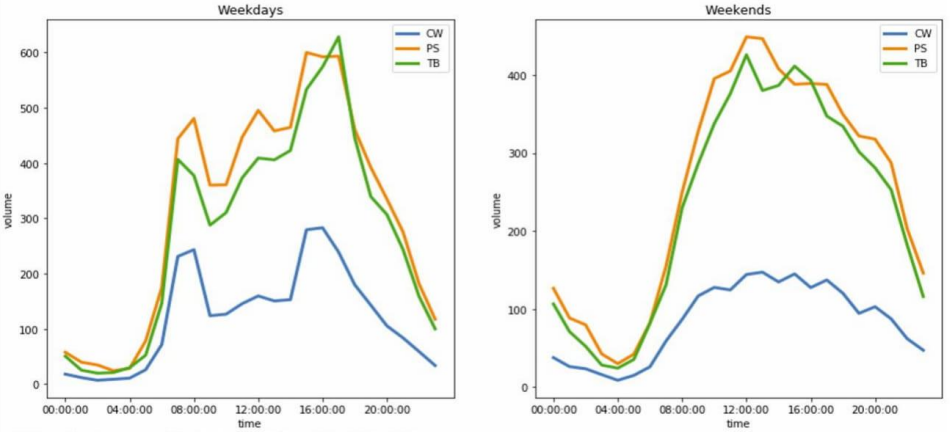
RESULTS AND DISCUSSION

Traffic Data

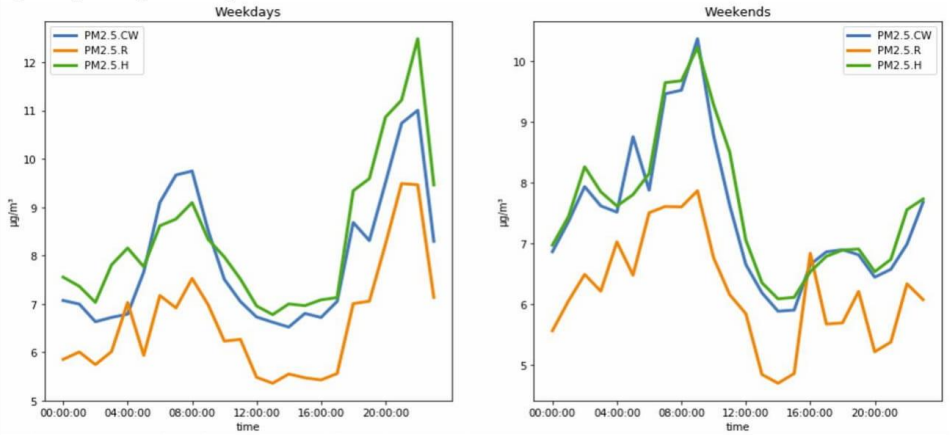
Traffic volume and vehicle class data was retrieved from the tube counters at the three different counting locations on arterial roads in the study area. The locations, as shown in the site map in Figure 1, are in front of Coldwell Elementary (CW), on Trowbridge Drive (TB), and at Pershing Drive (PS). The devices allow for download of classification of 13 classes of vehicles, as defined by the Federal Highway Administration (JAMAR Technologies, 2011). These classes are also defined by MOVES2014a and are used in calculations for emissions rates at each link. Figure 2a displays the diurnal trends of weekday and weekend traffic volume during the study period at the three counter locations. It is seen in the figure that the weekday traffic peaked in the morning and late afternoon around 5 p.m., while the weekend traffic peaked in the early afternoon. The trends agree well with the normalized diurnal traffic pattern reported by Batterman et al. (2015) based on the traffic data from 14 sites over a period of 4 years.

The three sets of tube counter and video traffic data were used in conjunction with El Paso Metropolitan Organization (MPO) data to estimate traffic data for all arterial roads and highway sections in the study area. The MPO utilized a Travel Demand Model (TDM) to estimate future travel demand and traffic conditions for the city. The TDM estimates provide traffic volume

estimates for four time periods of the day; these estimates are for daily values, not distinguishable between weekday and weekend values. Using the traffic data measured during the summer study period, ratios were created for the corresponding links from the TDM by dividing the TDM estimate by the measured data for the links that have both TDM and measured traffic data. A new adjusted weekday hourly estimate was created for each peak hour in the time period for all roads by multiplying the TDM values by the ratio of the same type of road. Because the study only collected traffic data at three locations along the arterial roads and at the I-54 highway, these ratios were used on corresponding and similar streets and roads among the other 180 link sources in the study area to create weekday/weekend emission factors. The same process was repeated to create weekend hourly estimates. Classification, speed, and volume are quantified and demonstrated in each link (road section) and were included in the MOVES2014a analysis.



a) Hourly Average Weekday/Weekend Traffic Volume



b) Hourly Average Weekday/Weekend $PM_{2.5}$ Concentrations

FIGURE 2 a) Hourly Average Weekday/Weekend Traffic Volume
b) Hourly Average Weekday/Weekend $PM_{2.5}$ Concentrations

Air Quality Data

The air pollution data collected during this study was processed for accuracy and completeness. Values reported by any of the monitors as negative, due to being below the monitors' method detection, were corrected. The reported concentrations can be negative due to zero drift in the electronic instrument output, data logger channel, or calibration adjustments to the data. Slightly negative values were automatically set to 0.5 (i.e., 1/2 of the detection limit), unless the negative values were more than three consecutive values; these were considered missing data. An hour of missing data resulted from the process of downloading the data from the monitors, three times a week. This hour of data was estimated by averaging the two adjacent values, before and after the missing hour. The finalized air pollution data was also adjusted using the calibration equation for each instrument found from a combination of the pre- and post-calibration data. The diurnal pattern of $PM_{2.5}$ data collected for weekdays and weekends during the study period are shown in Figure 2b.

$PM_{2.5}$ has been observed to peak in the morning as well as in the afternoon in El Paso, Texas (Li et al. 2001; 2005). For this near-road community, the morning $PM_{2.5}$ peak coincided well with the morning traffic (Figure 2a), but deviated from the early afternoon traffic peak occurring around 4 p.m. The early afternoon traffic peak appears to correlate well with the off-school traffic during weekdays whereas the $PM_{2.5}$ appears to be more correlated to the regional air pollution, indicating that the regional air pollution is likely to be more prevalent for the near-road community, even at locations that are immediately adjacent to an interstate highway.

Table 2 shows the maximum 1-hr, maximum 24-hr, and all-period averaged $PM_{2.5}$ concentrations monitored at the three sites (Columns 2, 5, and 8). It is interesting to compare the data observed at the two near-road monitors, Coldwell and House. Coldwell site was 6 meters from the frontage road and approximately 38 meters from the closest lane of the southbound highway whereas the House site was about 8 meters from the frontage road and approximately 42 meters from the closest lane of the northbound highway. Data for the two locations exhibit the characteristics of near-road monitors. Table 2 shows that the difference in $PM_{2.5}$ between the two monitor locations are well within 12%, specifically, the differences are 7%, 12%, and 5% for the maximum 1-hr, maximum 24-hr, and all-period average, respectively. The difference could very well be caused by the direction-varying traffic volume, and time-varying emissions and meteorological conditions. Yet, the difference is practically minimal if one considers all possible uncertainties including upwind-downwind configuration, instrument sensitivity, uncontrollable emission episodes such as emissions from older, poorly maintained vehicles, cooking, barbeque, among other unreported emissions. Furthermore, these maximum 1-hr, maximum 24-hr, and all-period averages were all indistinguishable from the data measured at the regional monitor, CAMS 12 located at UTEP. For the residential location at Radford, that is 300 meters away from the highway, the maximum 1-hr, maximum 24-hr, and all-period $PM_{2.5}$ averages are consistently lower than the near-road monitor House by 6%, 19%, and 21%, based on the limited size of the data collected in the study.

TABLE 2 $PM_{2.5}$ Max 1-hr, Max 24-Hr, and Period Average for Monitor, Model+Background, and Model Results (in µg/m^3)

	Max 1-hr			Max 24-hr			All Period Average		
	Monitored	Modeled +BG	Modeled	Monitored	Modeled +BG	Modeled	Monitored	Modeled +BG	Modeled
House	40.3	47.7	3.7	13.5	17.1	1.0	8.5	9.5	0.7
Coldwell	37.8	47.3	15.4	12.1	17.1	2.2	8.1	10.1	1.4
Radford	38.0	47.5	1.3	11.0	16.7	0.4	6.7	9.1	0.3

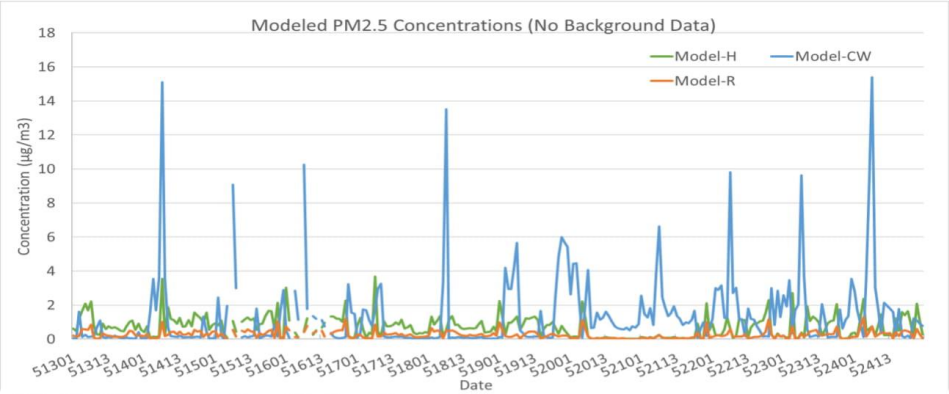
AERMOD Model Predictions

$PM_{2.5}$ concentration estimates resulting from traffic emissions from I-54 were generated using AERMOD. Concentration surfaces were generated using discrete receptors as well as grid receptors in order to evaluate the impacts of traffic emissions on the community using the AERMOD concentration estimates.

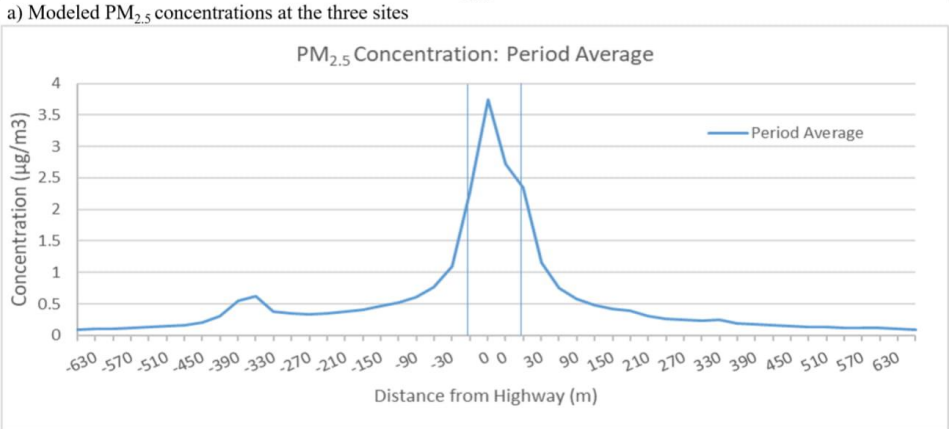
- *Near-road receptors and off-highway receptor*

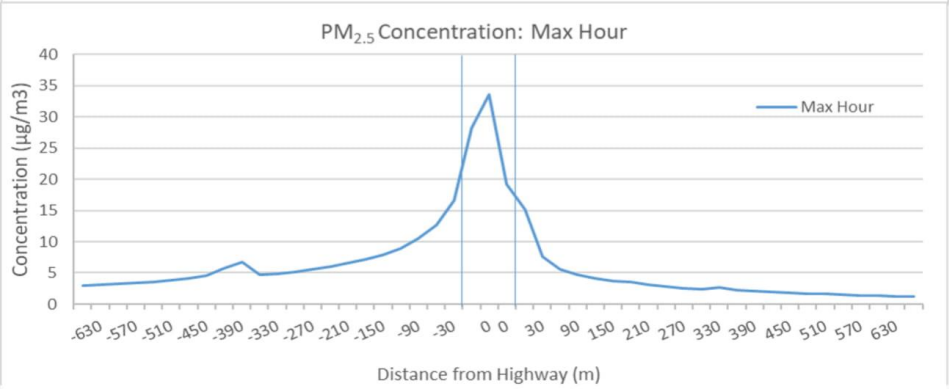
The $PM_{2.5}$ concentrations calculated by AERMOD for the maximum 1-hr, maximum 24-hr, and all-period averaged $PM_{2.5}$ concentrations at the three monitor sites are listed in Table 2 (Columns 4, 7, and 10). The magnitudes of the model prediction do not appear to be dominated by the prevailing westerly winds (see the windroses in Figure 1). Instead, the upwind Coldwell site shows higher concentrations than the downwind House site. This is likely due to the higher traffic estimates for the southbound gateway and highway. An approximately 65% decrease in the $PM_{2.5}$ concentration predictions is observed between the House site and the Radford site which is situated on the same side of the highway as the House site, but 300 meters off the highway. It is also observed that for the time period between May 20th at 7 a.m. and May 21st at 7 p.m., $PM_{2.5}$ concentrations estimates were consistently lower at the House and the Radford receptors. These estimates are likely due to the high easterly winds during these hours.

The $PM_{2.5}$ concentration time series estimates for the three sites can be seen in Figure 3a. Higher $PM_{2.5}$ concentrations at Coldwell were consistently predicted than at the other two sites, due to the previously mentioned high traffic volume occurring on the southbound highway. It is also observed that many of the highest estimates at the Coldwell site occurred at 7 a.m. The higher concentration estimates obtained during this hour, in spite similar traffic volumes in the following hours, is due to the vertical temperature profile in the early morning hours (Turner, 1994). The urban option within AERMOD was modified, beginning with version 11059, to address potential issues associated with the transition from the nighttime urban boundary layer to the daytime convective boundary layer. Prior to version 11059, the enhanced dispersion due to the urban heat island during nighttime stable conditions was ignored once the rural boundary layer became convective. This could result in an unrealistic drop in the mixing height for urban sources during the morning transition to a convective boundary layer, which could contribute to overly conservative concentrations for low-level sources under such conditions (EPA, 2004). This correction to avoid overly conservative concentrations could possibly result in overestimating values at the hour of the transition from the nighttime urban boundary layer to the daytime convective boundary layer, which in the case of El Paso occurs at hour 7.



a) Modeled $PM_{2.5}$ concentrations at the three sites





b) $PM_{2.5}$ Dispersion as a Function of Distance from the Highway

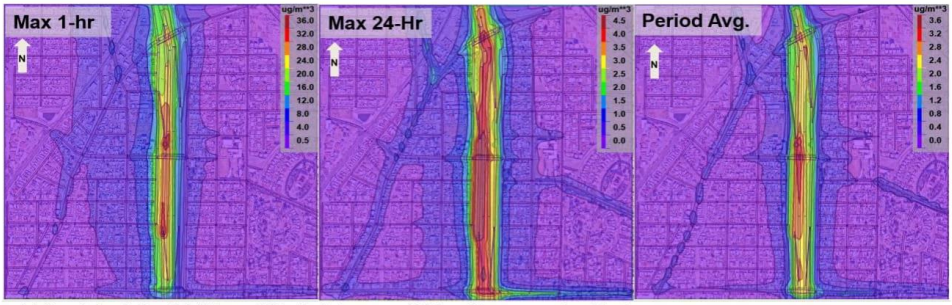
FIGURE 3 a) Modeled $PM_{2.5}$ Concentrations at the three sites
b) $PM_{2.5}$ Dispersion as a Function of Distance from the Highway

- *Cross-highway Concentration Distribution*

The dispersion of $PM_{2.5}$ concentrations from the highway can also be analyzed with the placement of receptors at increasing distances from the highway, specifically, in the direction perpendicular to the highway. A general rapidly decreasing trend of the predicted $PM_{2.5}$ concentrations with increasing distance from the nearby highway, was observed. Figure 3b shows the dispersion of the pollutant $PM_{2.5}$ away from the highway, where the concentration of airborne particles was characterized as a function of distance from I-54, with negative values representing the distance increasing to the west of the highway. These results suggest that the vast majority of dispersion occurs within 200 meters of the highway. A secondary minor peak appearing to the west of the highway (Figure 3b) is attributed to an arterial road running parallel to the highway, which can be seen modeled in the concentration maps.

- *Community Exposure to Traffic Emissions*

It is observed that the links with greater traffic volumes produce the greatest concentrations of $PM_{2.5}$, especially the southbound lanes on I-54. The spatial distributions of $PM_{2.5}$ concentrations in the community at the maximum 1-hour, the maximum 24-hour average, and the period average are shown in Figure 4a. These figures provide a clearer illustration of the $PM_{2.5}$ exposure in the community due to the traffic emissions in the study area. Arterial roads with higher traffic volume, such as Pershing and Trowbridge also account for higher estimates of $PM_{2.5}$ exposures to the community.



a) Max 1-hr, Max 24-Hr, and Period Average $PM_{2.5}$ Concentration Estimates

b) Max 1-hr, Max 24-Hr, and Period Average $PM_{2.5}$ Concentration Estimates

FIGURE 4 a) Max 1-hr, Max 24-Hr, and Period Average $PM_{2.5}$ Concentration Estimates b) $PM_{2.5}$ Hourly Concentrations at Different Peak Hours, Friday May 18th

Error! Reference source not found. Figure 4b shows the $PM_{2.5}$ concentrations modeled by AERMOD at four different peak hours, shown clockwise they represent 12 a.m., 7 a.m., 1 p.m., and 5 p.m. on Friday May 18, 2018. It is observed that during times of higher traffic volume, occurring at 7 a.m. and 5p.m., $PM_{2.5}$ concentrations are higher. The prevailing wind directions during these peak hours (and most of the day) are from the west to east or west to south east; the wind speed range throughout this particular day is from 1.3 to 10.3 m/s. The modeled estimates correspond well with these wind conditions, as the bulk of the emissions are observed to occur to the east of the highway.

- *Background Concentration*

Air pollutant concentrations near busy highways are composed of the incremental concentrations resulting from traffic emissions and the background concentrations resulting from emissions from other area, mobile, and point sources. Background concentrations should be as representative as possible for the area where the project site is located. Studies have shown that $PM_{2.5}$ measured at near-road air quality monitors is only moderately impacted by traffic emissions. More than 85% of the roadside $PM_{2.5}$ concentrations are believed to be regional urban-scale background concentrations which are primarily caused by ubiquitous urban emission sources (De Winter et al. 2018).

For an area surrounded by multiple background ambient $PM_{2.5}$ monitors, EPA recommended that the data should be analyzed by statistical or mapping methods to develop an appropriate background concentration estimate for use in the analysis. Li et al. (2019) reevaluated EPA's recommendations and suggested that background concentrations developed by distance-weighted averaging of the data available from all urban-scale background monitors appear to perform better than non-normalized methods with higher accuracy. Unfortunately, background $PM_{2.5}$ data were only available at 2 sites, UTEP and Ascarate, for this study. While these two sites are equidistant to the study area and could be used to create a background concentration estimate, the Ascarate site is located near a major highway as well as a border crossing, which would not provide a background estimate representative of the area. Therefore, data recorded at the UTEP monitor during the study period was selected to be the hourly background concentrations.

- *Modeled-to-Monitored Comparison*

The total $PM_{2.5}$ exposure in the community was assessed by adding the AERMOD modeled concentration estimates to the selected background concentrations. The modeled predictions were first compared to the $PM_{2.5}$ pollutant data measured at the three locations in Table 2. It appears that the model over-estimates the maximum 1-hr and 24-hr $PM_{2.5}$ at the near-road sites and the off-highway residence by at least 16% and 21%, respectively. The model accuracy improves for longer term average. For the all-period average in this study, the over-predictions continue to be significant at 14% and 33% for near-road sites and off-highway residence, respectively.

Figure 5 shows the modeled-to-monitored time series comparisons of $PM_{2.5}$ emissions during the study period. The figures are divided into two different weekly periods starting at Sunday May 13th through May 19th, followed by Sunday May 20th through May 24th. The elements labeled beginning with "Model" are those modeled through AERMOD; i.e. "Model-H" are the AERMOD modeled results for the receptor located at the House. The modeled results include the background concentration estimates provided by the El Paso CAM station at UTEP, located about 4 miles away from the study area.

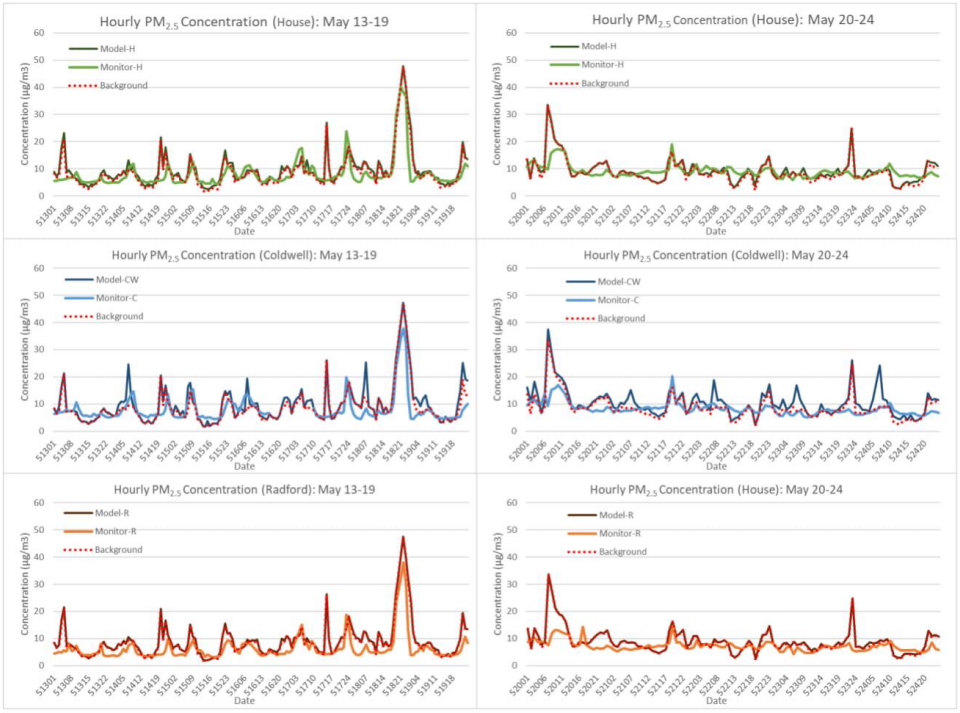


FIGURE 5 Comparison of Model Results and On-Site Monitoring: Hourly $PM_{2.5}$ Concentrations

As previously discussed, background concentrations account for a significantly portion of the $PM_{2.5}$ exposure near or off highway. Local traffic impacts account only approximately 10% of the total exposure. That is to say, the modeled results shown in the figures are driven largely by the regional background concentrations. It is noted that the spike observed on May 18th, which occurs at around midnight, could have been caused by a "Motorcycle Run" event wherein a large group of motorcyclists drove through the City of El Paso earlier that day.

- *Traffic Emission Impacts to the Community*

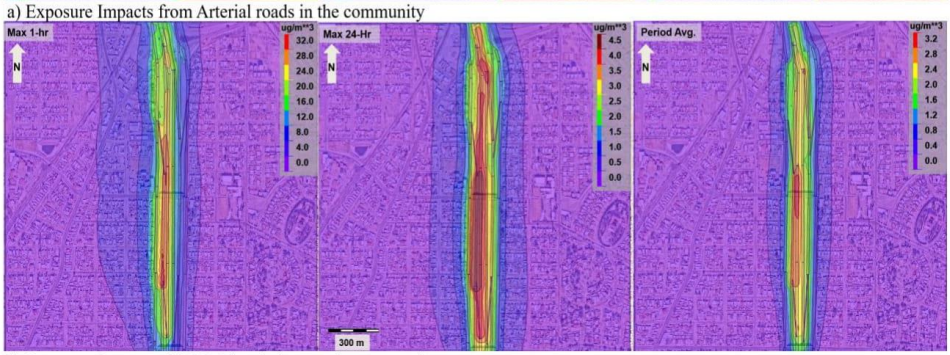
Included in the analysis using AERMOD, each source was placed into three different "source groups" which allow the model to consider the impact of each source group on the receptors. These groups were "Arterial", "Gateway" and "Highway". Table 3 shows the percent of contribution to $PM_{2.5}$ by each source group on the three receptors.

TABLE 3 $PM_{2.5}$ Contribution to Receptors by Type of Source

	House	Coldwell	Radford
Arterial	11.6%	13.4%	49.4%
Gateway	1.4%	4.6%	2.0%
Highway	87.1%	82.1%	48.5%

This study observed that for the two near-highway receptors, the contribution to $PM_{2.5}$ concentrations was greater than 80%, whereas contribution from the highway was around 50% on the Radford receptor, located 300 meters away from the highway. The receptor at Coldwell received around 5% of the emissions contributed from the gateway, which is due to the higher traffic volumes on the southbound gateway links; it also received a greater contribution from arterial roads than the other near-highway receptor due to the arterial roads near the school experiencing higher traffic volumes. Traffic emission impacts to the community are illustrated in detail in Figure 6. Figure 6a shows the exposure impacts resulting from arterial roads in the community whereas Figure 6b shows the contribution of only the interstate highway emissions to the community.

a) Exposure Impacts from Arterial roads in the community



b) Exposure Impacts of US I-54 emissions to the community

FIGURE 6 a) Exposure Impacts from Arterial roads in the community
b) Exposure Impacts of I-54 emissions to the community

CONCLUSION

This study addresses the spatial and temporal concentration variations in a near-road community resulting from traffic emissions on a microscale. It appears that there is a clear divergence between the concentrations predicted by AERMOD and the monitored data. The AERMOD rendered highest concentration estimates at locations where the traffic volume is the highest and downwind of the prevailing winds. However, impacts of the traffic emissions on the air quality subside rapidly with increasing distance away from the highway. In the near-road community studied, traffic emissions from the interstate highway were 4.8 times higher than the

contributions made by local arterial roads. Air quality in near-road communities is dominated by the regional background concentrations which account for more than 85% of the pollution. Challenges in modeling air quality impacts of transportation emissions are presented at each step of the process. For an accurate emissions modeling, fleet information proves to be most difficult to obtain when most traffic counters provide only a broader classification of vehicles than what is required by the MOVES model. In regards to dispersion modeling, model estimates are highly sensitive to meteorological conditions and source characterization, and additionally, higher quality of upper air data could yield more accurate meteorological parameters from the AERMET preprocessor. Finally, obtaining accurate background data from the study area can help provide better modeled-to-monitored comparison, as background concentrations have been shown to be of greater impact in urban areas and contributes to around 85% of measured $PM_{2.5}$ concentrations.

Future steps include performing sensitivity tests on AERMOD model performance with respect to the number of sources modeled that provide enough accuracy and low computation times, comparison between area and volume sources in AERMOD, and conducting analysis of other pollutants which are more closely related to vehicular emissions.

On-site monitoring of air pollution at near road schools is able to capture high resolutions variations in air quality. The results from this study could provide the information needed in the field of vehicle emissions exposure to near-road communities. Determining the influence of mobile emissions from highways on the air quality of the surrounding communities can help raise awareness to underserved communities living near highways and help policy makers make informed decisions based on this knowledge. While it is shown through this study that highway emissions drop considerably after around 200 meters, communities would benefit from avoiding designation of residential and school facilities within these zones and could result in less exposure to harmful vehicle emissions.

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