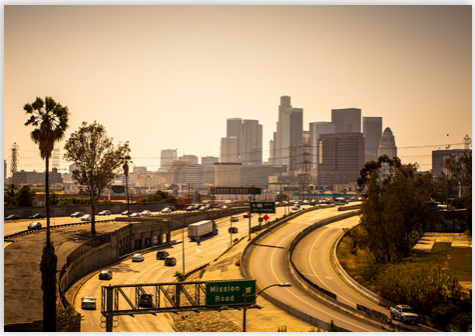
Analysis of Modeled and Measured Near-Road $PM_{2.5}$ Concentrations in Indianapolis and Providence During 2015 and 2016



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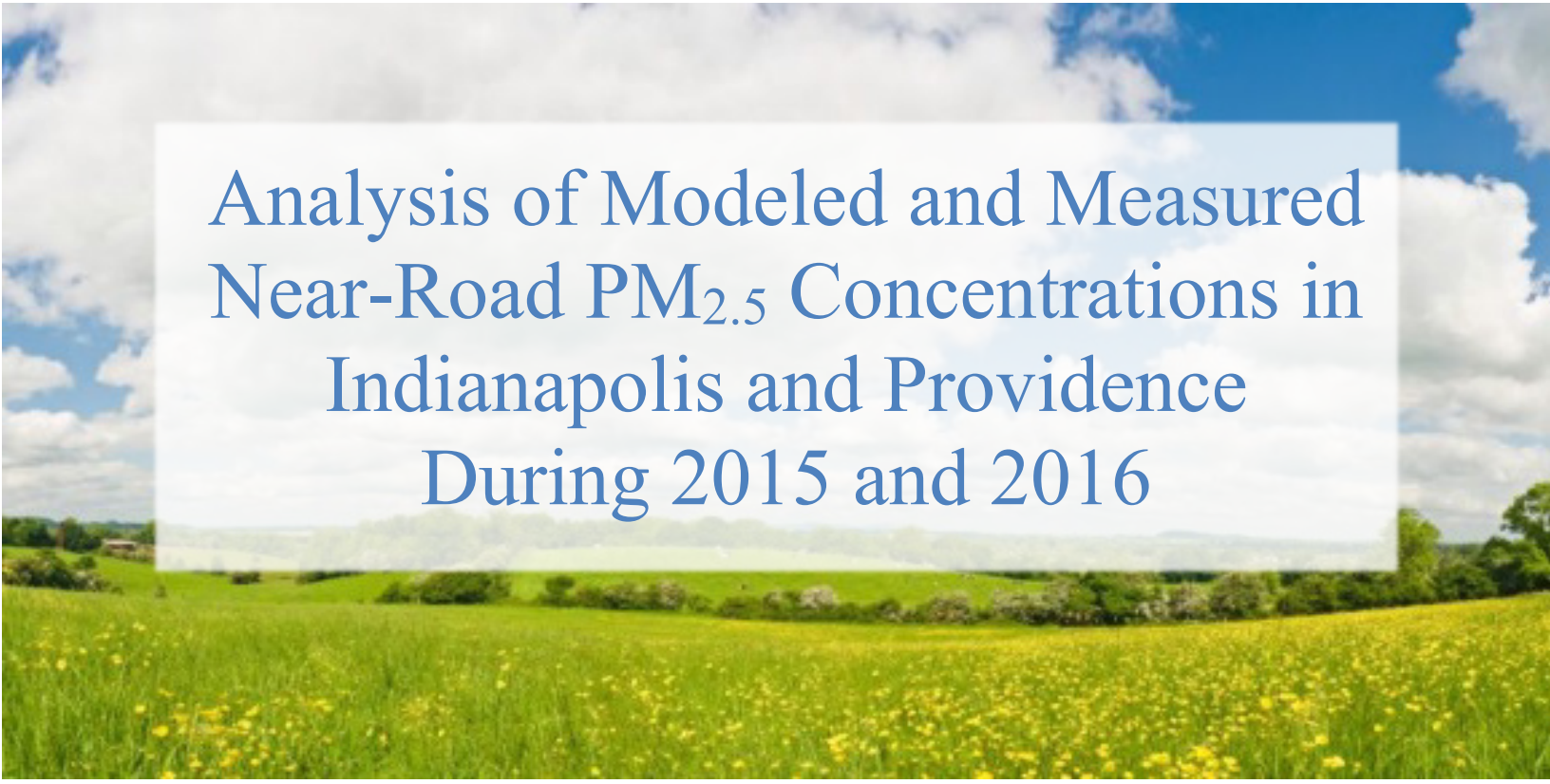

Analysis of Modeled and Measured Near-Road $PM_{2.5}$ Concentrations in Indianapolis and Providence During 2015 and 2016

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Abstract. This work was completed as part of the Near-Road Air Quality Research Pooled Fund TPF-5(284), under the U.S. Federal Highway Administration Transportation Pooled Fund Program. The lead agency for TPF-5(284) is the Washington State Department of Transportation. Other participants that funded this work include the U.S. Federal Highway Administration (FHWA) and the Arizona, California, Colorado, Ohio, Texas, and Virginia Departments of Transportation. Sonoma Technology, Inc., provides TPF-5(284) participants with technical, planning, facilitation, and website support.

Background. Transportation projects in particulate matter (PM) nonattainment and maintenance areas are required to undergo a quantitative transportation conformity hot-spot analysis if they are identified as projects of local air quality concern (POAQC) due to their potential to cause PM hot spots. In this work, we developed two dispersion modeling analyses to (1) evaluate near-road $PM_{2.5}$ concentrations predicted by the AERMOD dispersion model under real-world conditions, and (2) assess the sensitivity of modeled results to the choice of model (AERMOD or CAL3QHCR), meteorological data, and travel data processing approach. These analyses provide valuable information to practitioners to understand potential sources of uncertainty in the near-road modeling process.

Methods. In the primary analysis, we evaluate a $PM_{2.5}$ monitoring site near a major freeway in Indianapolis, Indiana, for 2016. In the secondary analysis, we evaluate a site near a major freeway in Providence, Rhode Island, for 2015-2016. The modeling analyses are built upon bottom-up estimates of resolved roadway $PM_{2.5}$ emissions based on detailed traffic monitoring data and current emission factor databases for the local vehicle fleet characterization. Dispersion model simulations are driven by local meteorological data collected at or close to the near-road monitoring sites. We estimated the difference between $PM_{2.5}$ concentrations at the near-road monitor and at nearby urban air quality monitoring sites (the measured near-road "increment") and the uncertainty associated with these estimates, and compared modeled results to the measured increments. Based on monitoring data, estimates of multi-day-averaged near-road $PM_{2.5}$ increments were 0.9 ± 0.6 μg/m^3 at Indianapolis and 1.4 ± 0.2 μg/m^3 at Providence (uncertainty represents the 95% confidence interval on the mean value), and were comparable to measured $PM_{2.5}$ increments at these sites in the near-road literature.

Results. Modeled roadway contributions to multi-day-averaged near-road concentrations substantially exceeded measured values based on the near-road monitoring data. The average near-road $PM_{2.5}$ increment modeled with AERMOD was more than 300% (factor of four) larger than the measured increment at Indianapolis, and more than 500% (factor of six) larger than the measured increment at Providence. These biases reflect cumulative uncertainty throughout the near-road $PM_{2.5}$ modeling chain, which includes travel activity data processing, emissions modeling, and air quality dispersion modeling. The emissions modeling part of the analysis may have contributed to the differences between modeled and measured concentrations in two ways. First, in both analyses, the relative contribution of modeled non-exhaust emissions ($PM_{2.5}$ brake wear, tire wear, and re-suspended road dust) compared to tailpipe exhaust emissions was higher than what has been documented in several published studies. Second, other research findings indicate that the U.S. EPA MOVES2014 model may over-predict tailpipe $PM_{2.5}$ exhaust. Together, these findings indicate uncertainty in the vehicle emissions estimates. The air quality modeling part of the analysis may have also contributed to differences between the modeled and measured concentrations. For example, when local meteorological data were used, AERMOD results were relatively insensitive to wind direction; as a result, modeled concentrations exceeded measured values regardless of whether the near-road monitor was upwind or downwind of the roadway.

1 Introduction

The impact of motor vehicle traffic on ambient concentrations of particulate matter with an aerodynamic diameter smaller than 2.5 μm ($PM_{2.5}$) can be difficult to quantify since total $PM_{2.5}$ concentrations are the result of primary particulate emissions such as black carbon (BC) from exhaust, metals from brake and tire wear, and re-suspended road dust, as well as secondarily formed particles from gas-phase emissions (Amato et al., 2011; Brown et al., 2010; Canagaratna et al., 2010; Durant et al., 2010; Kumar et al., 2008). Measured near-road $PM_{2.5}$ concentrations are heavily influenced by the fleet mix of vehicles traveling on the adjacent roadway, as diesel-fueled vehicles are an important source of BC in the near-road environment (Dallmann et al., 2014; Ban-Weiss et al., 2008). Numerous monitoring studies have found a 10-15% increase in $PM_{2.5}$ above urban concentrations next to the roadway (Guerreiro et al., 2011; Keuken et al., 2013; Sarnat et al., 2018; Karner et al., 2010; Hu et al., 2009; Kimbrough et al., 2018; Saha et al., 2018; Sofowote et al., 2018; Ginzburg et al., 2015; Baldauf et al., 2008; Zhou and Levy, 2007; Brown et al., 2014), and model results suggest that concentrations of $PM_{2.5}$ may decrease by 20% within 200-500 m of the roadway (Chang et al., 2015). There is significant interest in monitoring and modeling near-road pollution because more than 11 million people in the United States live within 150 m of a major highway (Boehmer et al., 2013), and increased exposure to pollution in the near-road environment has been associated with adverse health effects such as reduced lung function, low birth weights, and increased asthma and risk of heart failure (Health Effects Institute, 2010; McConnell et al., 2010; Padula et al., 2012).

Because of the potential for localized hot spots of ground-level PM concentrations near major roadways, quantitative hot-spot analyses are required for federally funded highway and transit projects that are identified as projects of local air quality concern (POAQC) in PM nonattainment and maintenance areas (U.S. Environmental Protection Agency, 2015b). In the transportation conformity context, a POAQC typically includes projects with substantial diesel truck or bus activity. Project-level assessments involve the use of steady-state Gaussian dispersion models such the American Meteorological Society – U.S. Environmental Protection Agency (EPA) Regulatory Model (AERMOD) or CAL3QHCR (based on the CALINE3 line source model) to simulate the effects of hourly emissions and meteorology on near-road PM concentrations. Accurate model predictions are important as the modeled incremental PM contribution of the transportation project must be added to a representative background concentration, and the combination (increment plus background) must be compared to the National Ambient Air Quality Standards (NAAQS). Other dispersion models, such as the line source models RLINE (Snyder et al., 2013) and Atmospheric Dispersion Modeling System (ADMS-Roads) (McHugh et al., 1997), and the street-canyon model SIRANE (Soulhac et al., 2011), have also been used to estimate near-road pollutant concentrations (Heist et al., 2013; Wang et al., 2016; Fallah-Shorshani et

al., 2017). As of 2019, AERMOD and CAL3QHCR could be used for U.S. regulatory project-level PM hot-spot analyses; after January 17, 2020, AERMOD is required as the sole dispersion model for these U.S. analyses (U.S. Environmental Protection Agency, 2017b).

Project-level modeling of near-road pollution involves three key steps: 1) travel activity data processing; 2) emissions modeling; and 3) air quality dispersion modeling. These steps constitute a modeling chain for predicting near-road PM concentrations. In a transportation conformity regulatory context, the models, methods, and assumptions used in completing these steps are determined through an interagency consultation process involving federal, state, and local transportation and air quality agencies. Each step has potential sources of uncertainty that can impact modeled near-road PM concentrations. These include uncertainties associated with estimating vehicle travel activity (volume, speed, and fleet composition); vehicle fleet characteristics (e.g., age distribution, engine technology, operating characteristics, and fuel properties); and vehicle emission factors (the mass of emissions per vehicle mile traveled [VMT]) for exhaust, brake and tire wear, and re-suspended road dust. Dispersion model results near major roadways have been shown to be sensitive to vehicle fleet mix, emission factors, and meteorological inputs (Snyder et al., 2014; Milando and Batterman, 2018a). Uncertainties are also associated with internal dispersion model parameters and formulations (Venkatram et al., 2013). For AERMOD, important practical choices such as whether to characterize roadway emissions as area sources or volume sources can significantly change modeled near-road PM concentrations (Claggett and Bai, 2012; Claggett, 2014), as the lateral plume meander formulation for AERMOD volume sources tends to reduce the higher concentrations as the lateral plume spread is enhanced (Heist et al., 2013).

Quantitative evaluation of dispersion models in the near-road environment is limited. Heist et al. (2013) evaluated several dispersion models based on data collected during the California Department of Transportation (Caltrans) Highway 99 (Benson, 1989) and the Idaho Falls (Finn et al., 2010) tracer experiments, while Askariyeh et al. (2017) evaluated the performance of various emission source representations in AERMOD based on data collected during the General Motors Sulfate Dispersion Experiments (Cadle et al., 1977). Though all field measurements involve some uncertainty, the use of metered emissions of an inert tracer species can largely eliminate uncertainty in emissions inputs. However, as tracer experiments are expensive and complex, and as a result, tracer data are only available for a limited number of days, meteorological conditions, and roadway configurations. While Heist et al. (2013) showed that AERMOD performed better than other dispersion models, such as CAL3QHCR, some evaluations have shown that AERMOD does not always perform as well as expected (Chen et al., 2009). Milando and Batterman (2018b) used ambient air quality observations located near high traffic roads in Detroit to evaluate dispersion model results for the Near-Road Exposure and Effects of Urban Air Pollutants Study (NEXUS) (Vette et al., 2013), but the $PM_{2.5}$ evaluation was inconclusive due to high

background concentrations and uncertainties in secondary aerosol formation and non-vehicle emissions. Additional evaluations based on $PM_{2.5}$ observations near major roadways are needed.

In 2010, the EPA mandated air quality monitoring next to major roadways in the United States. This near-road monitoring network initially focused on NO_2 as the mandate coincided with EPA's update to the annual NO_2 NAAQS and addition of a new 1-hr NO_2 NAAQS (U.S. Environmental Protection Agency, 2010), but $PM_{2.5}$ measurements have been added over time (U.S. Environmental Protection Agency, 2013). The national near-road monitoring network generally does not include measurements of coarse particulate matter (PM_{10}). Data from the U.S. near-road monitoring program have been used to estimate contributions of roadway emissions to near-road $PM_{2.5}$ concentrations at a national scale under diverse meteorological conditions, traffic conditions (annual average daily traffic [AADT], vehicle fleet mix), and roadway characteristics (e.g., distance from monitor to roadway, orientation) (DeWinter et al., 2018; Seagram et al., 2019). U.S. near-road data have also been used to examine contributions of roadway $PM_{2.5}$ emissions in Denver and Indianapolis (Brown et al., 2019), as well as Houston and Ft. Worth (Li et al., 2019). The U.S. near-road data provide a unique opportunity to evaluate near-road $PM_{2.5}$ concentrations predicted by dispersion models.

In this work, we developed two dispersion modeling analyses for the years 2015 and 2016 to 1) evaluate near-road $PM_{2.5}$ concentrations predicted by the AERMOD dispersion model under real-world conditions, and 2) to assess the sensitivity of modeled results to the choice of model (AERMOD or CAL3QHCR), meteorological data, and travel data processing approach. In the primary analysis, we evaluate a $PM_{2.5}$ monitoring site near a major freeway in Indianapolis, Indiana, for 2016. In the secondary analysis, we evaluate a site in close proximity to a major freeway in Providence, Rhode Island, for 2015-2016. The modeling analyses are built upon bottom-up estimates of temporally and spatially resolved roadway $PM_{2.5}$ emissions based on detailed traffic monitoring data and current emission factor databases for the local vehicle fleet characterization. Dispersion model simulations are driven by local meteorological data collected at the near-road monitoring sites. We estimated the difference between $PM_{2.5}$ concentrations at the near-road monitor and at nearby urban air quality monitoring sites (the measured "increment") and the uncertainty associated with these estimates, and compared modeled results to the measured increments. This work provides a unique evaluation of near-road $PM_{2.5}$ concentrations predicted by dispersion models, and provides valuable information to practitioners to further understand potential sources of uncertainty in the near-road modeling chain.

2 Methods

The modeling chain for predicting near-road $PM_{2.5}$ concentrations in this study consisted of: 1) travel activity developed from traffic monitor data; 2) emissions modeling with MOVES (for vehicle exhaust, tire wear, and brake wear emissions) and use of AP-42 methods for re-suspended road dust emissions; and 3) air quality dispersion modeling with AERMOD or CAL3QHCR. The data processing and modeling methods for the Indianapolis analysis are described below in Sections 2.1 through 2.7. Methods specific to the Providence analysis are shown separately in Section 2.8.

2.1 Site and Episode Selection

Dispersion modeling simulations were conducted to evaluate near-road concentrations at the Indianapolis near-road site (EPA Air Quality System [AQS] ID 18-097-0087), which is 3.1 km northeast of downtown Indianapolis and 24.5 m south of Interstate 70 (I-70) (Figure 1). I-70 is a major freeway with AADT in 2016 of 165,672, with 14% heavy duty trucks. Additional information about the Indianapolis near-road site is shown in Table 1. The project area (white circle in Figure 1) is centered on the near-road monitor and radially extends 1.5 km to include the major roadways that may affect $PM_{2.5}$ concentrations at the near-road monitor. The project area includes a freeway interchange about 1 km southwest of the near-road monitor.

The Indianapolis near-road site was one of several sites in the national near-road monitoring network in 2016 with $PM_{2.5}$ data and coincident nearby hourly traffic volume, vehicle speed, and fleet mix data (42 sites in the network collected $PM_{2.5}$ data in 2016). The Indianapolis near-road site had coincident $PM_{2.5}$ measurements and travel activity data for 152 days (non-consecutive) during 2016 to support modeling analysis. The Indianapolis site also had co-located hourly meteorological data for temperature, wind speed, and wind direction. One added benefit was the presence of two co-located $PM_{2.5}$ monitors at Indianapolis: a Federal Equivalent Method (FEM) monitor with continuous 1-hour duration $PM_{2.5}$ measurements, and a filter-based Federal Reference Monitor (FRM) with 1-in-3 day 24-hour measurements. Among the sites in the near-road monitoring network, the Indianapolis site was one of the most straightforward to model in that the local terrain was relatively flat, the roadway was at-grade, and there were no nearby roadside barriers, vegetation, or other obstructions (see Figure 1) that could influence near-road pollutant concentrations (Baldauf et al., 2016; Steffens et al., 2014; Brantley et al., 2014; Deshmukh et al., 2019; Venkatram et al., 2016) in ways that cannot be reasonably simulated in AERMOD and CAL3QHCR.

The Indianapolis near-road monitor is located in a mixed commercial area. There are two rail lines as close as 100 m from the near-road monitor in the opposite direction of I-70. These rail lines experience moderate free-flow train traffic with no idling trains. We examined BC measurements from

the Indianapolis near-road site and found that hourly BC concentrations were highest when the near-road monitor was downwind of I-70 and upwind of the rail lines. We found no noticeable increase in hourly BC concentrations when the monitor was downwind of the rail lines.

Figure 1. Indianapolis modeling project area with 1500-m radius (white circle) centered on the Indianapolis near-road air quality monitor (NR site, and pictured below), with available traffic monitors (labeled dots) and roadway links that were included in the modeling (black lines). Imagery source: Google Earth.

Table 1. Summary of data for the Indianapolis near-road site for 2016.[a]

Attribute	Value
AQS ID	18-097-0087
Coordinates	39.7879N 86.1309W
$PM_{2.5}$ instruments	FEM: Beta Attenuation Monitor (BAM 1020), continuous FRM: R&P Seq VSCC, every 3rd day
Number of lanes (I-70)	10
AADT	165,672
Heavy-duty truck fraction	14%
FE-AADT	374,419
Distance to road	24.5 m
Maximum 24-hour $PM_{2.5}$	39 μg/m^3
Annual mean $PM_{2.5}$	9.9 μg/m^3
Co-located meteorology	Wind speed, wind direction, temperature

[a] Fleet-equivalent traffic volume (FE-AADT) is a metric that considers both total traffic volume and fleet mix (number of heavy-duty vehicles) to obtain a single emissions-weighted traffic volume. The AADT and FE-AADT data were calculated from 2016 traffic data. Other data were obtained from EPA in May 2017 (U.S. Environmental Protection Agency, 2017a).

2.2 Travel Activity

For this study, we use monitored travel data from the Indiana Department of Transportation (INDOT) traffic data repository (indot.ms2soft.com) as the basis for estimating vehicle emissions. Traffic data for 40 roadways (approximately 20 miles) in the modeling project area were obtained from 49 traffic monitors. These roadways include all freeways and arterial roadways within the project area and represent the vast majority of VMT and diesel truck traffic in the project area. A key uncertainty for modeling near-road $PM_{2.5}$ concentrations is representing the travel activity. In regulatory PM hot-spot analyses, travel activity is typically based on estimates from a travel demand model developed for transportation planning purposes.

Among the 49 traffic monitors there were one permanent monitor and 48 temporary monitors. The permanent traffic monitor (labeled 951315 in Figure 1) was located on I-70, 0.9 km from the Indianapolis near-road air quality monitor. This traffic monitor was used to characterize hourly volume, speed, and fleet mix on I-70 where vehicle emissions had the greatest influence on pollutant concentrations at the near-road monitor. As discussed in Section 3.2.1, I-70 traffic accounted for 93% of the traffic-related $PM_{2.5}$ emissions that were modeled in the project area. This permanent traffic monitor

operated for 152 days in 2016, including weekdays and weekends, and these were the days included in the dispersion modeling analysis.

The 48 temporary traffic monitors were operated for approximately 2 days in 2014 or 2 days in 2016. For temporary traffic monitors that operated in 2014 (but not 2016), we estimate the 2016 volume data using the ratio of estimated AADT at these monitors between 2014 and 2016 as an adjustment factor. These estimated AADT values were available from the INDOT traffic data repository. Since the temporary monitors operated only on weekdays, weekend traffic volumes at these monitors were calculated using the ratio between weekday and weekend traffic volume data from the permanent monitor as an adjustment factor. The traffic data developed from the temporary monitors were then repeated to cover the analysis days.

Five arterial roadways that were included in the project area did not have traffic monitors installed. To generate volume data for these roadways, we combined the traffic volume from nearby monitors. A “mass-balance” approach was used, in which the traffic volume from a downstream monitor was subtracted from traffic volumes from an upstream monitor to determine the volume for the roadway without a monitor.

The INDOT traffic monitors recorded traffic data based on U.S. Federal Highway Administration (FHWA) vehicle class Scheme F. The vehicle class and speed data were mapped to the class and speed bins used in the EPA Motor Vehicle Emission Simulator (MOVES) model to support the emissions calculation. The vehicle class mapping is shown in Table 2. For each vehicle type, the distribution of MOVES source type was determined by the table in MOVES that describes the population by source type (table name: sourcetypeyear). This distribution was then applied to the sum of the traffic volumes from the FHWA Scheme F class under the same vehicle type. After the FHWA vehicle class was mapped to the MOVES source type, vehicle speed was further mapped to MOVES speed bins using the MOVES speed distribution table (table name: avgspeeddistribution).

Table 2. Vehicle mapping between the FHWA Scheme F and MOVES source types.

Vehicle type	FHWA Scheme F ID	FHWA Scheme F description	MOVES source type ID	MOVES source type ID description
Motorcycles	1	Motorcycles	11	Motorcycles
Passenger Cars	2	Passenger Cars	21	Passenger Cars
Light-duty trucks	3	Other Two-Axle Four-Tire Single-Unit Vehicles	31	Passenger Trucks
			32	Light Commercial Trucks
Single-unit trucks	5	Two-Axle, Six-Tire, Single-Unit Trucks	51	Refuse Trucks
	6	Three-Axle Single-Unit Trucks	52	Single Unit Short-Haul Trucks
	7	Four or More Axle Single-Unit Trucks	53	Single Unit Long-Haul Trucks
			54	Motor Homes
Buses	4	Buses	41	Intercity Buses
			42	Transit Buses
			43	School Buses
Combination trucks	8	Four or Fewer Axle Single-Trailer Trucks	61	Combination Short-Haul Trucks
	9	Five-Axle Single-Trailer Trucks	62	Combination Long-Haul Trucks
	10	Six or More Axle Single-Trailer Trucks		
	11	Five or Fewer Axle Multi-Trailer Trucks		
	12	Six-Axle Multi-Trailer Trucks		
	13	Seven or More Axle Multi-Trailer Trucks		

2.3 Emissions

We modeled $PM_{2.5}$ emissions using the EPA MOVES2014a model (U.S. Environmental Protection Agency, 2015a) for running exhaust, tire wear, and break wear, and followed AP-42 (U.S. Environmental Protection Agency, 2011) to calculate re-suspended road dust. For running exhaust, tire wear, and brake wear, MOVES was run in emission rate mode for the calendar year 2016 to generate the emission rate tables. These tables contain emission factors in gram per mile by vehicle speed, road type,

and vehicle source type. Local vehicle fleet characterization data obtained from the Indianapolis Metropolitan Planning Organization was used as additional input for MOVES. These local data provided information about the vehicle age distribution, vehicle technology, and fuel types that are important for estimating emission factors with MOVES but are not available from the traffic monitor data. Meteorological tables in MOVES were populated from meteorological data collected at the Indianapolis near-road air quality monitor. All roadway links were assumed to be free-flow links because freeways represent the vast majority of VMT, diesel truck traffic, and traffic-related PM emissions likely to impact the near-road air quality monitor; therefore, emissions were not estimated from idling or vehicle starts.

Traffic activity data for each roadway were merged with the emission rate tables by year, month, day, hour, source type, road type, and speed. The hourly emissions of a specific roadway can be calculated as $E=V\times L\times EF$ where E is the emission in grams, V is the traffic volume, L is the road length in miles, and EF is the emission factor in grams/mile. In AERMOD, each roadway was further divided into equal-sized small volume sources, yielding a total of 4,004 volume sources for the 40 roadways (approximately 20 miles). In a regulatory PM hot-spot analysis, only roads that are substantially affected by a proposed transportation project are modeled. In this study, all freeway, ramp, and arterial roadway links within 1.5 km of the near-road monitor were modeled so that virtually all potential sources of $PM_{2.5}$ emissions from vehicles were accounted for when comparing modeled concentrations to the observed near-road increment. As shown later, the ramp and arterial links account for less than 10% of the modeled $PM_{2.5}$ concentrations.

MOVES does not model re-suspended road dust. Therefore, we used the method prescribed in EPA's AP-42 emission factors handbook Section 13.2.1 (Paved Roads) to calculate hourly $PM_{2.5}$ emissions based on VMT, average vehicle weight, and prescribed silt loading factors that vary by road type. The default AP-42 silt loading factors were used as follows: 0.015 g/m^2 for freeway links with more than 10,000 AADT; 0.030 g/m^2 for arterial links with more than 10,000 AADT; and 0.060 g/m^2 for arterial links with AADT between 5,000 and 10,000. AP-42 adjustments for liquid precipitation (which suppresses road dust) and frozen precipitation (which increases road dust due to application of antiskid material) were also accounted for on an hourly basis. No road dust emission control or mitigation programs were included.

The inclusion of $PM_{2.5}$ road dust in this modeling analysis is somewhat unusual given that exhaust, break wear, and tire wear emissions typically dominate vehicle contributions to $PM_{2.5}$ concentrations near major roadways (Pant and Harrison, 2013). Road dust emission factors are highly uncertain (Venkatram, 2000) and the AP-42 methodology has received limited updates over the years. Typically, regulatory PM hot-spot analyses do not include $PM_{2.5}$ road dust emissions, as these emissions are only included when they are determined to be a significant contribution to the $PM_{2.5}$ air quality

problem in a given nonattainment or maintenance area. Most important, however, was the consideration that the monitoring data includes contributions from road dust; therefore, $PM_{2.5}$ road dust emissions were included in this analysis so that all potential sources of $PM_{2.5}$ emissions from vehicles were accounted for when comparing modeled concentrations to the observed near-road increment.

2.4 Meteorology

Hourly meteorological data for 2016 were processed using AERMET (version 18081), the meteorological preprocessor for AERMOD. Local wind speed, wind direction, and temperature data were obtained for the Indianapolis near-road monitoring site from AQS. The completeness of the meteorological data was 99% for wind speed and direction, and 100% for temperature. Other parameters needed for AERMET, such as cloud cover, were obtained from the National Center for Environmental Information (NCEI) for the Indianapolis International Airport meteorological station, located 14.5 km southwest of the Indianapolis near-road monitor. Upper-air data from the Wilmington, Ohio, airport were also used, consistent with current dispersion modeling guidance from the Indiana Department of Environmental Management (IDEM). Default regulatory AERMET options were used, including the low wind speed surface friction velocity adjustment and a low wind speed threshold of 0.5 m/s. The AERSURFACE program was used to calculate surface characteristics (albedo, Bowen ratio, and surface roughness length) around the meteorological tower at the near-road site based on land use files from the United States Geological Survey (USGS). Precipitation and snow cover data provided by IDEM were used to assign appropriate monthly albedo and Bowen ratio values for 2016. To support the AERMOD sensitivity simulation with alternative meteorological data, AERMOD-ready meteorological data files from Indianapolis International Airport were acquired from IDEM. These files were also developed using the default regulatory low wind speed surface friction velocity adjustment in AERMET.

To support modeling with CAL3QHCR, the AERMOD-ready meteorological data were translated into CAL3QHCR meteorological data format. We started with the AERMET-processed data to ensure consistent meteorological data inputs between the two models. AERMET-processed winds were rotated 180 degrees (wind blowing “to” instead of “from”), and methods from Golder (1972) were used to relate hourly Monin-Obukhov lengths in the AERMET file to Pasquill-Gifford stability classes needed by CAL3QHCR. Hourly urban mixing layer heights for CAL3QHCR were determined based on the maximum of the convective or mechanical boundary layer values in the AERMET file, and population of the urban area.

2.5 $PM_{2.5}$ Monitoring Data

The near-road $PM_{2.5}$ "increment" is the difference in concentration between the near-road monitor and a nearby urban background monitor. A key challenge is that the near-road $PM_{2.5}$ increment is relatively small compared to the urban background concentration. The choice of which nearby monitoring site(s) to use, and what approach to use to calculate the near-road increment, is important as there is no perfect approach to estimating the urban background concentration. DeWinter et al. (2018) and Seagram et al. (2019) found that there was good agreement among near-road $PM_{2.5}$ increments calculated using various approaches involving one or more nearby monitors. Given this consistency, and given the close proximity of potential background monitors to the Indianapolis near-road site, we estimate near-road increments based on data from individual (as opposed to combinations of) nearby background monitors.

To estimate near-road increments and characterize uncertainty, we analyzed $PM_{2.5}$ data from the Indianapolis near-road monitor and two nearby ambient monitors: Washington Park, 3 km northeast of the near-road site, and E. Michigan St., 1.6 km southeast of the near-road site (Figure 2). We calculated separate increments from each nearby monitor to understand the sensitivity of the near-road increment to the choice of background monitor. The Indianapolis near-road site had two co-located $PM_{2.5}$ monitors: a FEM monitor with continuous hourly average measurements and an FRM monitor with 24-hour measurements every third day. The near-road FEM monitor provides more frequent 24-hr data than the co-located FRM monitor, but each individual measurement is less precise; the precision of the FRM monitor is ± 7%, compared to ± 22% for the FEM monitor (U.S. Environmental Protection Agency, 2015d). The nearby sites had FRM monitors with daily measurement frequency at Washington Park and 1-in-3 day frequency at E. Michigan St.

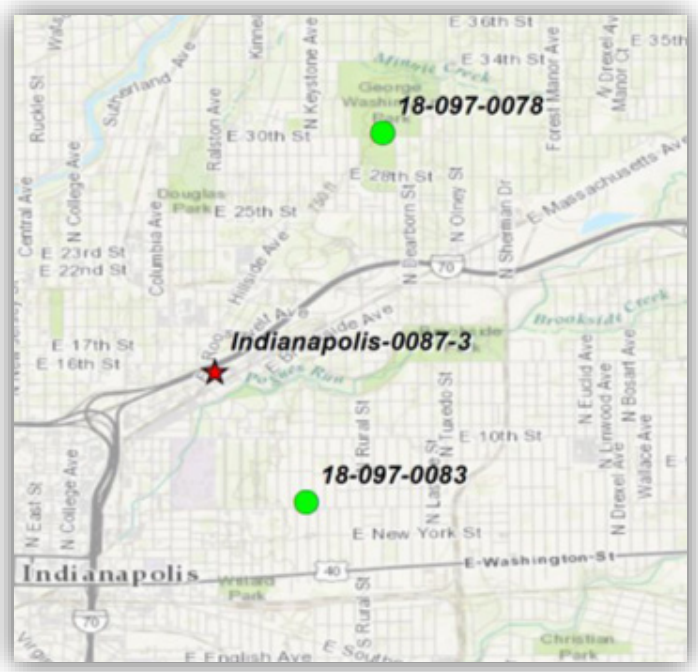

Figure 2. Location of the Indianapolis near-road $PM_{2.5}$ FRM and FEM monitors (star) and nearby $PM_{2.5}$ FRM monitors (green dots) at Washington Park (AQS ID 18-097-0078) and E. Michigan St. (AQS ID 18-097-0083).

Monitoring data were retrieved from EPA's AQS. Data from AQS are quality-controlled by the submitting tribal, state, or local air monitoring agency, and certified May 1 of each year. We retrieved data from AQS after the certification date. All data were further quality-assured by excluding any values with a specified data qualifier, including those flagged for quality assurance errors, natural events, or exceptional events (where air quality is affected by an unusual or naturally occurring event that cannot be reasonably controlled by tribal, state, or local air agencies, such as high $PM_{2.5}$ concentrations due to smoke from wildland fires). Unless otherwise noted with a data qualifier, concentrations equal to zero or negative concentrations (which may occur as a result of zero drift in the instrument or calibration adjustments) were treated as valid.

2.6 Dispersion Model

The Gaussian dispersion model AERMOD (Cimorelli et al., 2004; 2005; Perry et al., 2005; U.S. Environmental Protection Agency, 2016) was used for all modeling scenarios except for one sensitivity case. Link-level $PM_{2.5}$ emissions derived from MOVES and AP-42 were used as input for the dispersion modeling. Line-volume sources were used to characterize the roadway links in the AERMOD

simulations. An hourly emission file was created for all roadway links, for every hour of each analysis day. Hourly emissions were mapped to traffic and emission modeling data discussed previously. The hourly emission file includes the hourly emission rate in g/s for each volume source, along with hourly source release heights and initial dispersion parameters. The release heights and initial dispersion parameters were calculated based on current regulatory modeling guidance and change on an hourly basis depending on the fleet mix (percent cars and percent trucks) for each hour using a traffic volume weighted average approach. Approximately 20 miles of roads within the project area were modeled. AERMOD was run with urban dispersion, flat terrain, and no particle deposition (U.S. Environmental Protection Agency, 2015b).

Modeling receptors are locations within the dispersion model where pollutant concentrations are calculated. One receptor was placed at the location of the Indianapolis near-road monitor, 24.5 m from the edge of the I-70 east-bound travel lanes. Based on photographs of the near-road site, the air intake inlet for the $PM_{2.5}$ sensors was estimated at 4 m above ground level (AGL). Therefore, a receptor height of 4 m AGL was used in the model, which is higher than a typical receptor height for regulatory analyses (1.8 m AGL). Model sensitivity as a function of height was not evaluated here, but AERMOD performance has been shown to be sensitive to receptor heights ranging from 0.5 to 9.5 m AGL (Askariyeh et al., 2017).

Roadway emissions in AERMOD were represented by a series of adjacent volume sources (also known as line-volume sources). Volume sources were chosen because tracer evaluations have shown that AERMOD performs slightly better when volume sources, rather than area sources, are used to characterize roadway emissions (Heist et al., 2013). The arrangement of AERMOD volume sources in the immediate vicinity of the near-road monitor and receptor location is shown in Figure 3. Two sets of line-volume sources, one eastbound and one westbound, were used to characterize emissions from the multi-lane I-70. This arrangement is consistent with dispersion modeling guidance and best practices, allows for independent specification of emissions from eastbound and westbound traffic, and ensures that the modeling receptor remains outside any volume source exclusion zones (see circles in Figure 3) where AERMOD results are not considered reliable. Freeway ramps and other arterials were represented by a single set of line-volume sources representing traffic in both directions.

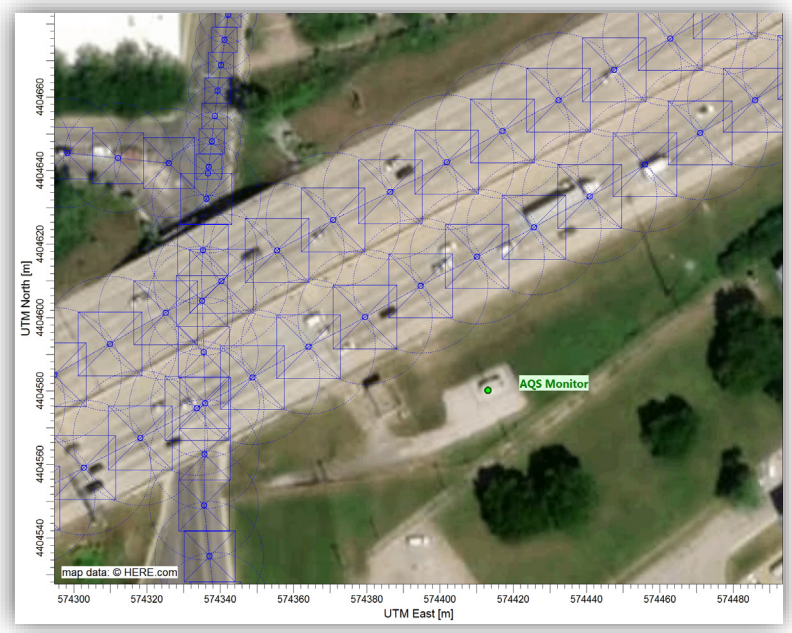

Figure 3. AERMOD line-volume source layout in the immediate vicinity of the Indianapolis near-road monitor (AQS Monitor). Volume sources are shown as squares, and their exclusions zones are shown as circles. Exclusion zones represent areas where concentrations cannot be estimated by AERMOD.

2.7 Modeling Scenarios

Four dispersion modeling simulations were conducted to compare modeled concentrations from roadway emissions to the measured near-road $PM_{2.5}$ increment at Indianapolis and to examine the sensitivity of selected processes to the near-road modeling results (Table 3). The base-case AERMOD scenario was used as the best estimate of modeled near-road $PM_{2.5}$ concentrations for comparisons with monitored near-road increments. The base-case scenario used hourly traffic and emission data from the project area and local meteorological data at the Indianapolis near-road site. Estimates of the monitored near-road $PM_{2.5}$ increment and the associated uncertainty are discussed in Section 3.1, and results from the base-case AERMOD modeling scenario are discussed in Section 3.2. Results from three sensitivity simulations were compared to the base-case results and to the measured near-road $PM_{2.5}$ increments at Indianapolis. These sensitivity scenarios are described below, and results are discussed in Section 3.3.

Table 3. Dispersion modeling scenarios conducted with different combinations of dispersion models, traffic data, and meteorological inputs.

Simulation	Dispersion Model	Inputs
Base case	AERMOD	Hourly traffic data Local near-road meteorology
AltTraff	AERMOD	Aggregated traffic data (e.g., by peak and off-peak periods) Local near-road meteorology
AltMet	AERMOD	Hourly traffic data Non-local NWS meteorology (from airport)
CAL3	CAL3QHCR	Hourly traffic data Local near-road meteorology

The alternative traffic (AltTraff) scenario was conducted to assess the sensitivity of modeled near-road $PM_{2.5}$ concentrations to the level of temporal detail provided in the emissions inputs. The hourly travel activity data used in the base-case simulation are not typically available to practitioners, and the preparation of these detailed day- and hour-specific data can be tedious. For regulatory applications, travel activity is typically based on estimates from a travel demand model used to support transportation planning decisions. Travel demand modeling data are less detailed and provide travel volumes for 4-5 time periods during the day (e.g., morning rush hour, afternoon, evening rush hour, and evening off-peak), and these same volumes are assumed for all days of the year. For the AltTraff simulation, the hourly traffic data for each roadway link were aggregated into a typical-day diurnal pattern, and emissions were calculated based on these aggregated traffic data. The diurnal profile was created using all analysis days, including weekdays and weekends, but weekday and weekend differences were not accounted for in the AltTraff scenario since travel demand models do not always differentiate weekends from weekdays. Other modeling inputs in the AltTraff scenario are identical to the base case.

The alternative meteorology (AltMet) scenario was developed to assess the sensitivity of modeled near-road concentrations to the meteorological inputs. In regulatory applications, representative meteorological data are typically selected from a nearby National Weather Service (NWS) airport observation site. The use of non-NWS data is less common in regulatory applications due to the lack of

available local data and the time and expense of collecting local data at the project site. Meteorological data from the Indianapolis International Airport was used in the AltMet simulation for 2016. Other modeling inputs were identical to the base case. The biggest differences between the near-road site and the Indianapolis airport are the wind speeds, exposure to nearby buildings, and surface roughness around the sites. Annual wind roses from the near-road site and the NWS airport site (Figure 4) show that the wind directions were similar at the two sites, but the wind speeds were higher at the NWS airport site.

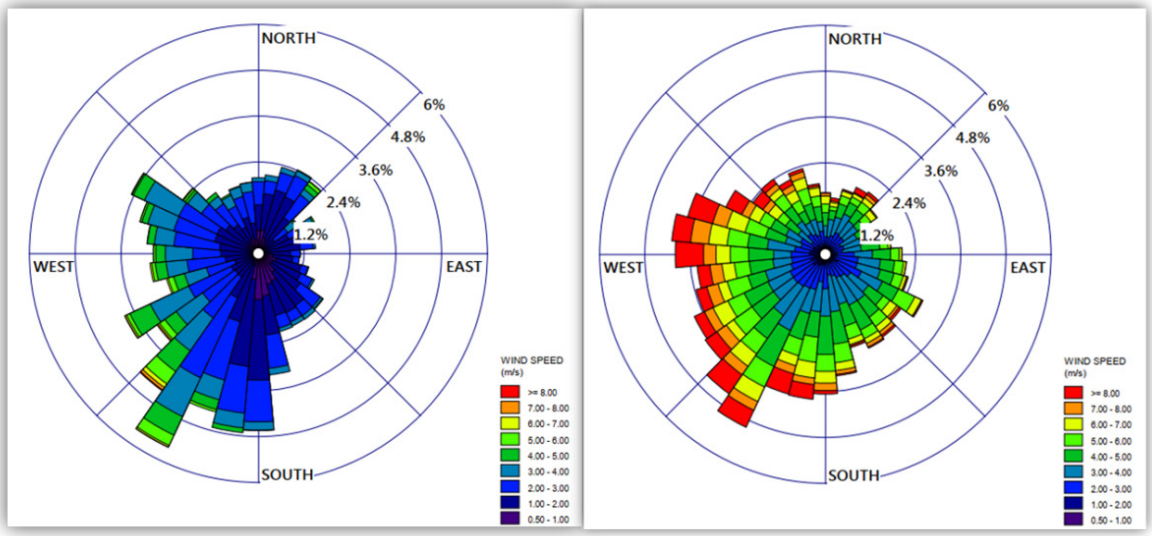

Figure 4. 2016 annual wind rose at the Indianapolis near-road monitor (left) and the Indianapolis International Airport NWS site (right).

The CAL3QHCR (Cal3) scenario was developed to assess the sensitivity of modeled near-road concentrations to the choice of dispersion model. The CAL3QHCR model (Benson, 1989; Eckhoff and Braverman, 1995) has historically been an approved dispersion model for regulatory applications. Through a formal rulemaking process, EPA determined that CAL3QHCR should be replaced by AERMOD as the required dispersion model, based in part on findings from the Caltrans Highway 99 and Idaho Falls near-road tracer study evaluations (U.S. Environmental Protection Agency, 2015c, 2017b). However, CAL3QHCR was approved for regulatory use until January 17, 2020, and there is still significant interest in the transportation community for understanding how AERMOD results compare to CAL3QHCR in real-world applications. For the Cal3 scenario, CAL3QHCR was used with the same hourly emissions and meteorological inputs as in the AERMOD base case. Free-flow, at-grade line sources were used to represent the roadway emissions. The local meteorological data from the base case were processed into CAL3QHCR format. A subset of 40 analysis days from the base case were modeled with CAL3QHCR, as CAL3QHCR does not have the ability to output concentrations for each specific

day with differing emissions inputs. We chose days with high modeled daily average concentrations from the base-case AERMOD simulation for both the summer (20 days) and winter (20 days) seasons. These 40 days had varying average daily wind directions.

2.8 Providence Analysis

2.8.1 Site and Episode Selection

To provide additional context for the Indianapolis modeling results, a dispersion modeling simulation was conducted to evaluate near-road concentrations at a Providence near-road site (AQS ID 44-007-0030), which is 1 km north-northwest of downtown Providence and 5 m east of Interstate 95 (I-95) (Figure 5) in a highly urbanized area. I-95 is a major freeway with AADT in 2016 of 233,036 with 7% heavy duty trucks. Compared to the Indianapolis analysis, the Providence freeway AADT is higher but the truck percentage is lower. Additional information about the Providence near-road site is shown in Table 4. The project area (yellow circle in Figure 5) is centered on the near-road monitor and radially extends 1 km from the monitor to include the major roadways that may affect concentrations at the near-road monitor. The project area includes a freeway interchange about 400 m south of the near-road monitor. The Providence near-road site had an FEM monitor that collected hourly $PM_{2.5}$ data, and also had coincident nearby hourly traffic volume, vehicle speed, and fleet mix data. $PM_{2.5}$ and travel activity data were sufficiently complete for 382 days (non-consecutive) during 2015-2016 to support modeling analysis. Because the Providence near-road site is located in a highly urbanized area, there are buildings adjacent to the site that could influence near-road pollutant concentrations.

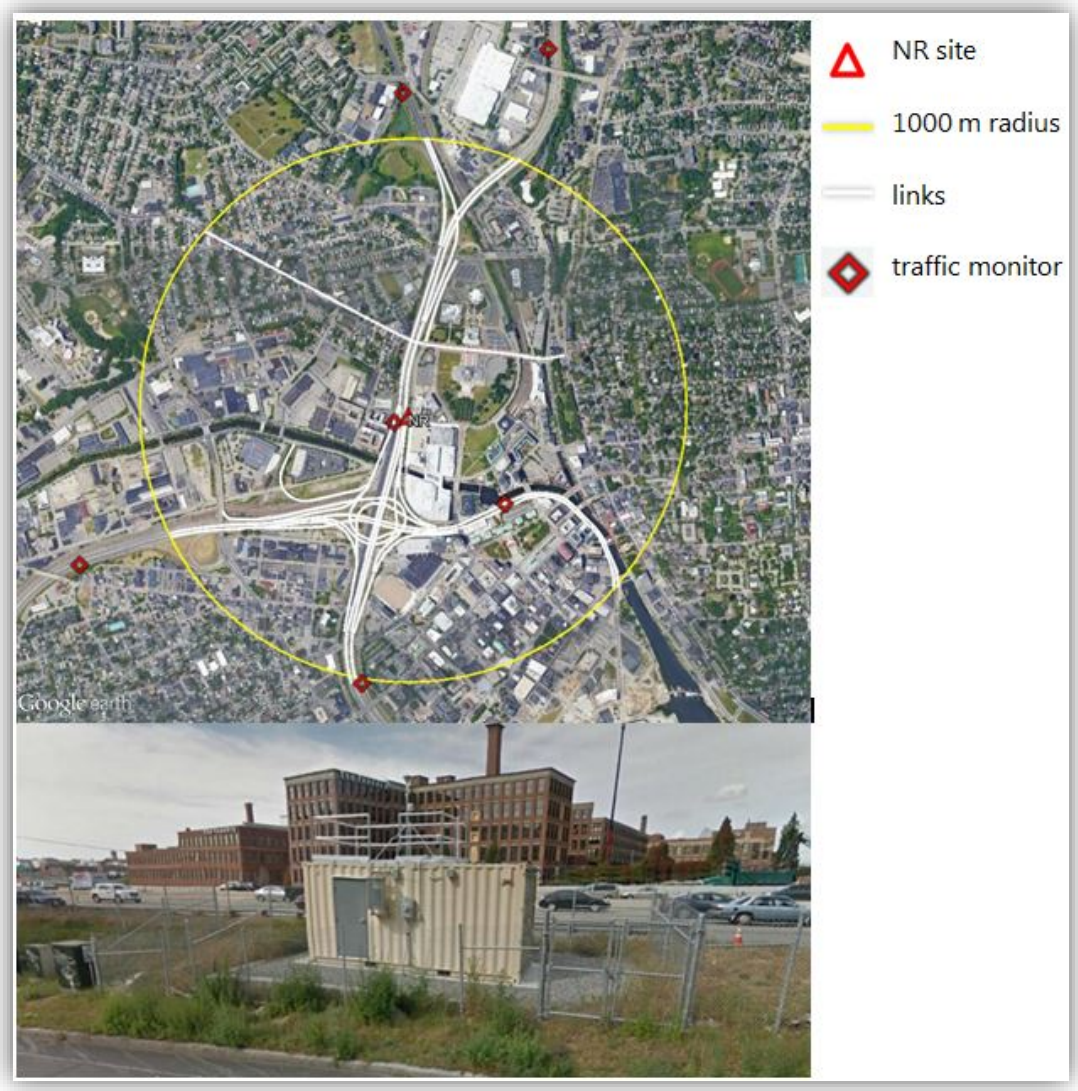

Figure 5. Providence modeling project area with 1000-m radius (yellow circle) centered on the Providence near-road air quality monitor (NR site, and pictured below), with available traffic monitors (◊) and roadway links that were included in the modeling (white lines). Imagery source: Google Earth.

Table 4. Summary of data for the Providence near-road site for 2016. The AADT and FE-AADT data were calculated from 2016 traffic data. Other data were obtained from EPA in May 2017 (U.S. Environmental Protection Agency, 2017a).

Attribute	Value
AQS ID	44-007-0030
Coordinates	41.8295N 71.7176W
$PM_{2.5}$ instruments	FEM: Beta Attenuation Monitor (BAM 1020)
Number of lanes (I-95)	8
AADT	233,036
Heavy-duty truck fraction	7%
FE-AADT	363,549
Distance to road	5 m
Maximum 24-hour $PM_{2.5}$	24.5 $\mu g/m^3$
Annual mean $PM_{2.5}$	9.3 $\mu g/m^3$
Co-located meteorology	None (nearest meteorological site at AQS 44-007-0022 2.4 km to the south)

During the analysis period, there was a freeway construction project (known as the Providence Viaduct project) and other non-freeway construction projects near the Providence near-road site. The Viaduct construction activity occurred primarily 200-600 m from the near-road site, while the smaller non-freeway construction projects were 50-100 m from the near-road site. Based on Rhode Island Department of Transportation (RIDOT) construction log data and Google Earth satellite imagery, 27 days (out of the 382 days with available data) were confirmed to have no nearby construction activity. These non-construction days were mostly weekends and holidays. During the other 355 days in 2015-2016, there was nearby construction activity. Therefore, the Providence modeling analysis was conducted separately for the 27 days without construction and 355 days with construction. Construction activity emissions were not modeled in this study.

2.8.2 Travel Activity and Emissions

We use hourly monitored travel data near the Providence near-road site as the basis for developing a bottom-up estimate of roadway $PM_{2.5}$ emissions. Traffic data were acquired from RIDOT for 6 traffic monitors (see Figure 5) that recorded traffic volume by RIDOT vehicle class and vehicle speed. Among the 6 traffic monitors there were five permanent monitors and one temporary monitor. One

of the permanent traffic monitors was located on I-95, about 50 m from the Providence near-road air quality monitor. This traffic monitor was used to characterize hourly vehicle volume, speed, and fleet mix on I-95 where vehicle emissions should have the greatest influence on pollutant concentrations at the near-road air quality monitor. The other four permanent traffic monitors were used to characterize traffic for other freeway segments in the modeling project area. The freeways characterized by the five permanent traffic monitors represent the majority of VMT and diesel truck traffic in the project area.

There are 50 roadways (approximately 9 miles) in the modeling project area. Forty of those roadways did not have traffic monitors installed. These roadways include arterial roadways, and the ramps at the I-95/U.S. 6 interchange south of the near-road air quality monitor. To generate hourly volume data for these roadways, we combined the available hourly monitored traffic data with additional AADT data provided by RIDOT and AADT data estimated from the State of Rhode Island Division of Statewide Planning travel demand model. In general, we used a "mass-balance" approach in which the traffic volume from a downstream monitor was subtracted from traffic volume from an upstream monitor to determine the volume for the roadway without monitors. For roadways without an upstream or downstream permanent monitor, the AADT data were either used as scaling factors to scale the hourly volume from the permanent monitors, or disaggregated to the hourly level based on the average diurnal pattern across the permanent monitors. Fleet mix was determined based on a nearby permanent traffic monitor, or based on the temporary traffic monitor at Memorial Blvd. (just east of the I-95/U.S. 6 interchange).

The RIDOT traffic monitors recorded traffic data based on RIDOT vehicle class. The vehicle class and speed from the monitors were mapped to the class and speed used in the EPA MOVES model to support the emissions calculation. The vehicle class mapping is shown in Table 5. For each vehicle type, the distribution of MOVES source type was determined by the table in MOVES that describes the population by source type. This distribution was then applied to the traffic volumes from the RIDOT vehicle class.

Table 5. Vehicle mapping between the RIDOT class and MOVES source types.

RIDOT class	RIDOT description	MOVES source type ID	MOVES source type ID description
1	Cars	11	Motorcycles
		21	Passenger Cars
		31	Passenger Trucks
		32	Light Commercial Trucks
2	Small Trucks	51	Refuse Trucks
		52	Single Unit Short-Haul Trucks
		53	Single Unit Long-Haul Trucks
		54	Motor Homes
3	Trucks	41	Intercity Buses
4	Double Trailer Trucks	42	Transit Buses
		43	School Buses
		61	Combination Short-Haul Trucks
		62	Combination Long-Haul Trucks

As with the Indianapolis analysis, we modeled $PM_{2.5}$ emissions using the EPA MOVES2014a model for running exhaust, tire wear, and break wear, and followed AP-42 to calculate re-suspended road dust. For running exhaust, tire wear, and brake wear, MOVES was run in emission rate mode for the 2015 and 2016 calendar years to generate the emission rate tables. Local vehicle fleet characterization data obtained from EPA (ftp://newftp.epa.gov/air/nei/2014/doc/2014v2_supportingdata/onroad) were used as additional input for MOVES. Meteorological tables in MOVES were populated from meteorological data collected at AQS site 44-007-0022 located 2.4 km south of the near-road monitor. Traffic activity data for each roadway were merged with the emission rate tables by year, month, day, hour, source type, road type, and speed, and hourly emissions were calculated for each roadway link. In AERMOD, each roadway was further divided into equal-sized small volume sources, yielding a total of 3586 volume sources for the 50 roadways. Portions of I-95 north-bound closest to the near-road monitor, as well as nearby arterial roads, were modeled lane-by-lane to avoid placing the near-road monitor within an

AERMOD volume source exclusion zone. Hourly $PM_{2.5}$ emissions from road dust were calculated using the method prescribed in EPA's AP-42 emission factors handbook Section 13.2.1 (Paved Roads) with default AP-42 silt loading factors, along with hourly precipitation adjustments and winter silt-loading factors when frozen precipitation occurred.

2.8.3 Meteorology

The Providence near-road site did not have co-located meteorological data; therefore, representative meteorological data from the nearest air quality monitoring station (Urban League, South Providence, AQS ID 44-007-0022), located 2.4 km to the south, was used. Hourly meteorological data for 2015-2016 were processed using AERMET (version 18081). Wind speed, wind direction, and temperature data from the Urban League site were used. Other parameters needed for AERMET, such as cloud cover, were obtained from NCEI for the T.F. Green Airport meteorological station, located 12 km south of the Providence near-road monitor. Upper-air data from the Chatham Municipal Airport in Massachusetts were also used. Default regulatory AERMET options were used, including the low wind speed surface friction velocity adjustment and a low wind speed threshold of 0.5 m/s. The AERSURFACE program was used to calculate surface characteristics around the meteorological tower based on land use files from the USGS.

2.8.4 $PM_{2.5}$ Monitoring Data

The near-road $PM_{2.5}$ increment was estimated between the Providence near-road monitor and the Urban League monitor in 2015-2016 for days when monitoring data was at least 75% complete. The Urban League South Providence monitor is 2.4 km south of the near-road monitor, and is generally upwind of the near-road monitor. The other monitor considered as a potential background monitor is the Francis School East Providence monitor (Figure 6), 4.8 km east northeast of the near-road monitor. We calculated separate increments from each nearby monitor to understand the sensitivity of the Providence near-road increment to the choice of background monitor. The Providence near-road site had a $PM_{2.5}$ FEM monitor with hourly measurements. Both nearby sites had co-located FEM and FRM monitors with hourly and 1-in-3 day measurement frequency, respectively. Monitoring data were retrieved from EPA's AQS.

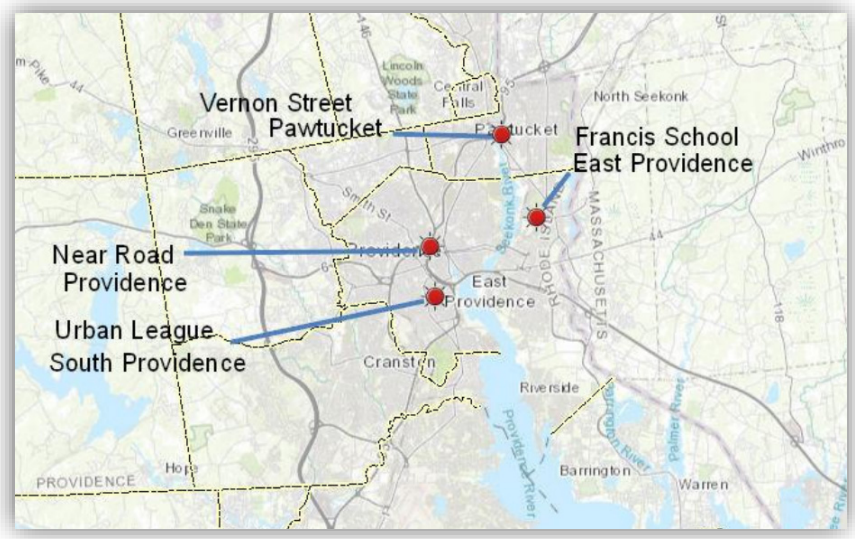

Figure 6. Location of the Providence near-road $PM_{2.5}$ FEM monitor and nearby $PM_{2.5}$ monitors (FEM and FRM co-located monitors). Data from Urban League South Providence (AQS ID 18-097-0078) and Francis School East Providence (AQS ID 18-097-0083) were considered in the Providence increment analysis.

2.8.5 Dispersion Modeling

The AERMOD modeling approach for the Providence analysis was similar to the Indianapolis analysis. AERMOD was run with urban dispersion, flat terrain, and no particle deposition. One modeling receptor was placed at the location of the Providence near-road monitor, 5 m from the edge of the I-95 north-bound travel lanes. Based on photographs of the near-road site, the air intake inlet for the $PM_{2.5}$ sensors was estimated at 3.6 m AGL. Therefore, a receptor height of 3.6 m AGL was used in the model.

Roadway emissions in AERMOD for the Providence case were represented as line-volume sources along approximately 9 miles of roadway within the project area. The arrangement of AERMOD volume sources and the receptor location is shown in Figure 7. Several sets of line-volume sources were created to characterize lane-by-lane emissions for north-bound I-95 in the immediate vicinity of the near-road monitor to avoid placing the receptor in volume source exclusion zones (represented as circles in Figure 7, where concentrations cannot be estimated by AERMOD). One arterial road adjacent to the near-road monitor was also represented as lane-by-lane line-volume sources. South-bound I-95 was represented as one set of volume sources, as were freeway ramps and other arterial roadways.

One AERMOD dispersion modeling simulation was conducted for the Providence analysis. The results from the Providence analysis are discussed in Section 3.4.

Figure 7. AERMOD line-volume source layout in the immediate vicinity of the Providence near-road monitor (yellow triangle). Volume sources are shown as squares, and their exclusions zones are shown as circles.

3 Results

The modeling chain for predicting near-road $PM_{2.5}$ concentrations consists of travel activity data processing, emissions modeling, and air quality dispersion modeling. Throughout this study, AERMOD results refer to predictions produced by the near-road modeling chain that involved the AERMOD dispersion model. Likewise, CAL3QHCR results refer to predictions produced by the near-road modeling chain that involved the CAL3QHCR dispersion model.

3.1 Estimating the Monitored $PM_{2.5}$ Near-Road Increment

We analyzed $PM_{2.5}$ data for year 2016 from the Indianapolis near-road monitor and nearby ambient monitors to estimate the near-road $PM_{2.5}$ increment and characterize its uncertainty. We calculated the near-road increment using various combinations of the two co-located FEM and FRM instruments at the near-road monitoring site, and two nearby background monitors (Washington Park and E. Michigan St., both of which are FRM instruments). Multiple increments were calculated to characterize how the increment varied based on the choice of background monitor (Washington Park or E. Michigan St.) and near-road measurement method (FEM or FRM).

The average daily near-road $PM_{2.5}$ increments calculated with four combinations of monitors are summarized in Figure 8. Data are shown for various daily average wind directions, when the Indianapolis near-road monitor was downwind (wind blowing from 274°-360° and 0°-33°), upwind (94°-213°), and parallel (34°-93° and 214°-273°) to I-70. Wind directions are 24-hour vector averages calculated from the hourly wind data. Increments involving the 1-in-3 day near-road FRM monitor (NR FRM in Figure 8) were calculated based on 46 days with coincident data at the Washington Park monitor, and 47 days with coincident data at E. Michigan St. Increments involving the near-road FEM monitor (NR FEM in Figure 8) were calculated based on 144 days with coincident data at Washington Park, and 47 days with coincident data at E. Michigan St. All days analyzed are subsets of the 152 days that were modeled.

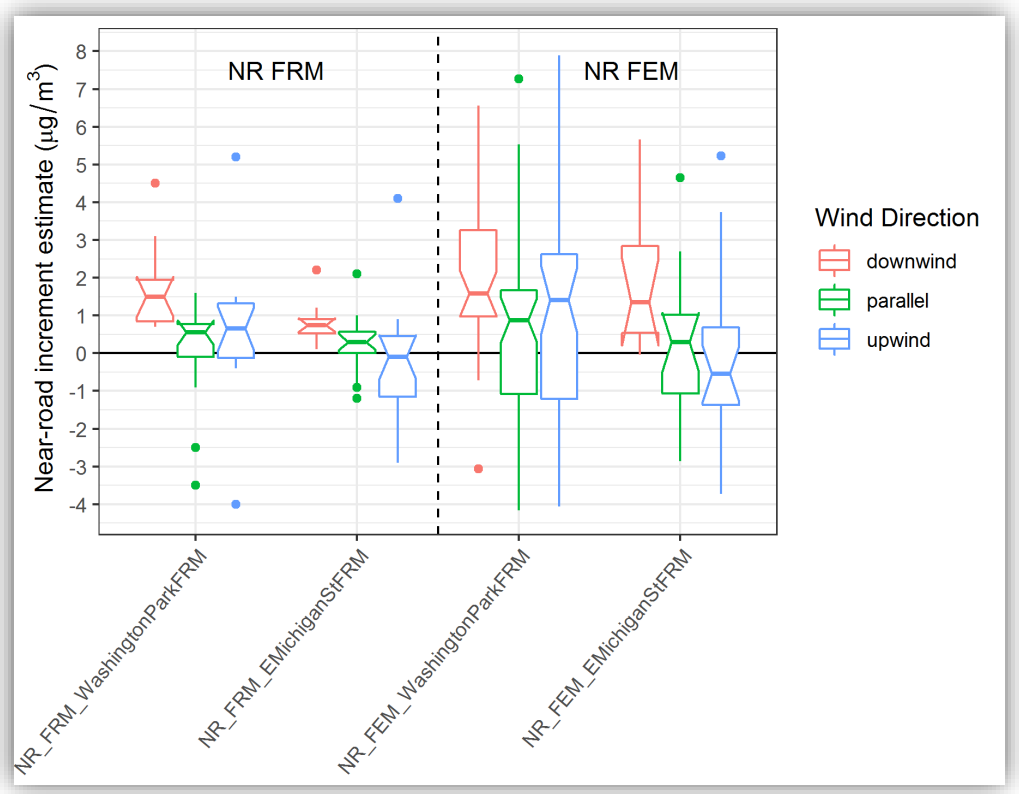

Figure 8. Indianapolis near-road $PM_{2.5}$ increments for four different combinations of near-road (NR) monitor (NR FRM and NR FEM) and nearby FRM $PM_{2.5}$ monitors at Washington Park and E. Michigan St. when the Indianapolis near-road monitor was downwind, upwind, and parallel to I-70. The horizontal line at the box notch indicates the median, box extents indicate the interquartile range (IQR), and whiskers indicate 1.5 times the IQR.

In all four comparisons in Figure 8, the average near-road $PM_{2.5}$ increment was highest when the near-road monitor was downwind of I-70, and lowest when the near-road monitor was upwind of I-70.

This result contrasts with AERMOD modeling results presented later. The median measured increment in downwind conditions was positive in all four comparisons, whereas the median increment in upwind conditions was negative when the E. Michigan St. monitor was used in the calculation. For a given combination of monitors, the median increment at Indianapolis varied by about 1 μg/m^3 as a function of wind direction. The variance of daily increments was considerably smaller when the FRM near-road monitor was used in the calculation because the FRM measurements are more precise than the FEM measurements.

The measured near-road increment concentration varied depending on the choice of background monitor. When considering just the FRM near-road monitor, the median increment differed by 0.4 μg/m^3 depending on the choice of background monitor. This result is due to variability in $PM_{2.5}$ concentrations across a relatively short distance (4 km) within the urban area. This intra-urban $PM_{2.5}$ variability has a modest effect on the near-road increment measurement.

An important point is that there can be a bias between different types of instruments, and this bias can affect the increment. The bias is distinct from the instrument precision discussed earlier. In this context, bias refers to a systematic difference between measurements from two different instruments, whereas precision refers to the random measurement error above and below the correct value from a single instrument. At the Indianapolis near-road site, $PM_{2.5}$ concentrations from the FRM monitor are systematically biased lower than concentrations from the co-located FEM monitor. As a result, increments based on the FRM near-road measurement are smaller. When considering just the Washington Park background monitor, the mean increment differed by 0.2 to 0.3 μg/m^3 depending on the choice of measurement method (FRM or FEM) at the near-road site, as shown in Table 6. Calculating increments from measurements that are based on the same monitoring method provides more consistent results and eliminates uncertainty associated with the bias between monitoring methods. Therefore, it is preferable to compare modeled near-road increments at Indianapolis to FRM-based increment measurements, since Washington Park and E. Michigan St. are both FRM monitors, and because FRM measurements are more precise than FEM measurements. We still show comparisons to FEM-based increment measurements for completeness.

We also estimated the uncertainty in the mean of near-road $PM_{2.5}$ increment measurements. Uncertainty in the mean, which arises from the variance in the distribution of daily increment measurements about the mean, was calculated as the 95% confidence interval around the mean value. Average daily near-road increments and their estimated uncertainty for the four combinations of near-road monitor and nearby background monitor are shown in Table 6. The uncertainty in the averaged increment is as large as 0.6 μg/m^3, which is similar to the variability due to the choice of background monitor (0.4 μg/m^3) and smaller than the variability across wind directions (about 1 μg/m^3). When a

sufficient number of days are considered, the uncertainty in the averaged increment measurement is sufficiently small to support reliable comparisons between modeled and measured near-road $PM_{2.5}$ increments. When considering the Washington Park background monitor, the absolute uncertainty in the FRM-based and FEM-based increments is similar because better precision in the FRM measurement offsets the reduced number of days that near-road FRM data are available.

Table 6. Summary of measured near-road $PM_{2.5}$ increments at Indianapolis. Uncertainty is calculated as the 95th percentile confidence interval in the monitored near-road increment.

Near-Road Monitor	Background Monitor	Mean Daily Near-Road Increment [μg/m^3]	Number of Observations
FRM	Washington Park	0.9 ± 0.6	46
FEM	Washington Park	1.2 ± 0.5	144
FRM	E. Michigan St.	0.2 ± 0.3	47
FEM	E. Michigan St.	0.4 ± 0.6	47

The uncertainty in individual daily measurements was not quantified. Given the large variability in daily measured increments, we expect comparisons of modeled and measured increments to be more uncertain when considering individual days. The maximum and 98th percentile of the distribution of modeled and measured increments were also calculated since these are relevant for regulatory analyses involving the 24-hour $PM_{2.5}$ NAAQS, but these comparisons should be considered less reliable than comparisons involving increments that have been averaged over many days.

For subsequent comparisons with modeled increments at Indianapolis, we selected the near-road increment based on measurements from the FRM near-road monitor and the Washington Park FRM monitor. This selection was based on: 1) the FRM-based measurement is more precise than the FEM-based measurement, and therefore the daily FRM-based near-road increments had less uncertainty; 2) the near-road and background concentration measurements are based on the same monitoring method (FRM), which eliminates uncertainty associated with differences in monitoring methods; 3) there were fewer negative daily increments when the Washington Park monitor was used; and 4) Washington Park was the upwind monitor when the near-road monitor was downwind of I-70 (see Figure 2). This best available near-road $PM_{2.5}$ increment estimate for the 2016 modeling analysis period was 0.9 ± 0.6 μg/m^3 averaged over all wind conditions (Table 6). This estimated near-road increment for Indianapolis is comparable to results from other studies. The annual average increment for Indianapolis was 0.7 to 1.7 μg/m^3 during

2014-2016 depending on the approach and the monitoring sites used to calculate the increment (DeWinter et al., 2018; Brown et al., 2019; Seagram et al., 2019).

3.2 Base-Case Simulation

3.2.1 Emissions

A summary of average daily $PM_{2.5}$ emissions calculated from vehicle activity for the 20 miles of roadways that were modeled within the Indianapolis modeling project area are shown in Table 7. The averaged daily emissions across all 152 modeling analysis days in 2016 were 70 lb/day. The average daily emissions were 76 lb/day on weekdays and 54 lb/day on weekends, reflecting the higher travel volumes observed on weekdays. In our assessment, road dust $PM_{2.5}$ was the biggest source of emissions (53% of total emissions), followed by exhaust (40%), brake wear, and tire wear. Non-exhaust emissions (tire wear, brake wear, and road dust) represented 60% of the total vehicle emissions. For context, the on-road $PM_{2.5}$ emissions reported in the 2014 National Emissions Inventory for Marion County, Indiana (which contains most of Indianapolis), was 4.1 tons/day, with 50% of emissions from paved road dust (U.S. Environmental Protection Agency, 2018).

Table 7. Summary of modeled $PM_{2.5}$ vehicle emissions for the Indianapolis project area.

Process	Average Daily $PM_{2.5}$ Emissions [lb/day]	% of Total
Road dust (AP-42)	37	53
Running exhaust	28	40
Brake wear	3	5
Tire wear	2	2
Total	70	100

The relative fraction of exhaust to non-exhaust $PM_{2.5}$ emissions from vehicles will vary by road type, fleet mix, fuel characteristics of the vehicle fleet, surface silt loading, and other factors. Past measurement studies indicate that exhaust emissions are typically responsible for the majority of the traffic-related $PM_{2.5}$ emissions near major roadways (Pant and Harrison, 2013). For example, source apportionment analysis of speciated $PM_{2.5}$ measurements next to a busy freeway in Toronto showed that about 35% of traffic-related $PM_{2.5}$ was due to non-exhaust emissions, and 12-20% was due to road dust (Jeong et al., 2019), while Ginzburg et al. (2015) estimated that road dust was 7-16% of the traffic-related

$PM_{2.5}$ 150 m away from a freeway near Baltimore. Based on this literature, the non-exhaust emissions in Table 7 may be overestimated, or the exhaust emissions may be underestimated. It is also possible that both the exhaust and non-exhaust emissions are overestimated.

AP-42 road dust estimates can be highly uncertain (Venkatram, 2000). For example, the AP-42 approach assumes an infinite reservoir of suspendible road dust, which may not be a valid assumption for heavily traveled paved highways. Brake and tire wear emissions estimates are also uncertain; $PM_{2.5}$ emission factors for these processes in the MOVES model are significantly lower than those in the California Emissions Factor (EMFAC) model (Reid et al., 2016). There is limited literature on the impacts of including road dust $PM_{2.5}$ emissions in near-road dispersion modeling applications, but in a project focused at the EPA near-road monitor in Ft. Worth, Texas, adding road dust $PM_{2.5}$ emissions increased the modeled $PM_{2.5}$ roadway emissions by 16-19% for highways and 139-208% for arterials, depending on season and time-of-day (Askariyeh et al., 2019). The inclusion of road dust $PM_{2.5}$ substantially affects the modeled $PM_{2.5}$ emission estimates, and will also affect modeled near-road concentrations.

3.2.2 AERMOD Results

AERMOD was executed for 152 analysis days in 2016 for the Indianapolis project area. The average modeled $PM_{2.5}$ concentrations (i.e., the modeled $PM_{2.5}$ near-road increment) for these days were compared to the monitored near-road $PM_{2.5}$ increments. The base-case AERMOD modeling results are compared with measured increments in Figure 9; summary statistics are shown in Table 8. Based on these results, AERMOD over-predicted the average near-road $PM_{2.5}$ increment. The average modeled increment (3.7 $\mu g/m^3$) was a factor of four larger than the measured FRM-based increment (0.9 $\mu g/m^3$), and a factor of three larger than the measured FEM-based increment (1.2 $\mu g/m^3$). The resulting bias (2.8 $\mu g/m^3$, or 311% of the FRM-based increment) for the averaged modeled increment is substantially larger than the estimated uncertainty of the measured near-road increment, and is also larger than the variability in the measured increment associated with the choice of background monitor. The FRM-based near-road increment was calculated from a 46-day subset of the 152 modeled days, but the results are similar when comparing to the FEM-based near-road increment, which was calculated from 144 of the modeled days. Therefore, the bias in the AERMOD result must be attributable to other factors.

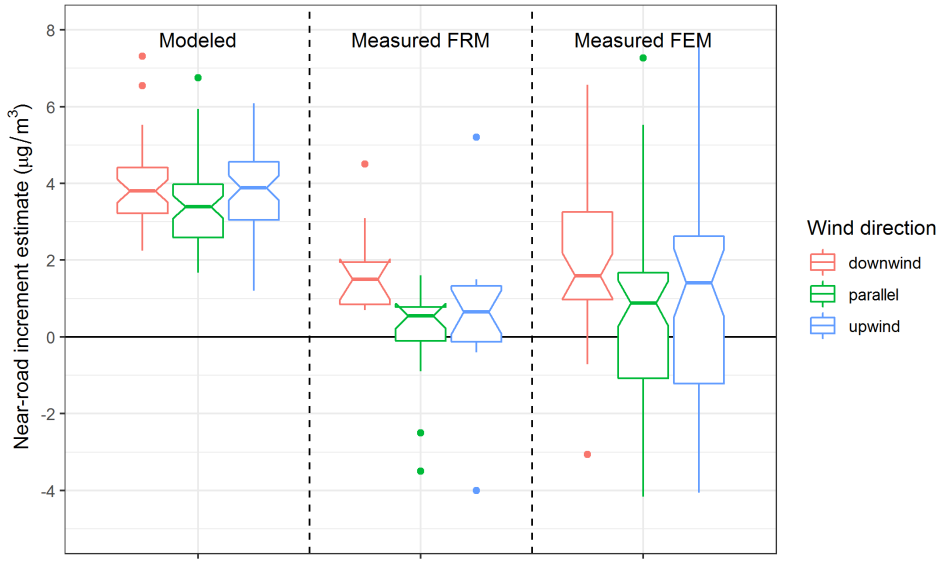

Figure 9. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured near-road $PM_{2.5}$ increments at the Indianapolis near-road monitoring site during 2016 for three wind conditions (near-road monitor downwind, upwind, and parallel to I-70).

Table 8. Summary of modeled and measured near-road $PM_{2.5}$ increments (µg/m3) at the Indianapolis near-road monitoring site. Monitored increments are calculated based on the Washington Park background monitor.

$PM_{2.5}$ **24-hour Increment**	**AERMOD Base-Case Increment (n=152)**	**FRM Monitored Near-Road Increment (n=46)**	**FEM Monitored Near-Road Increment (n=144)**
Average	3.7	0.9 ± 0.6	1.2 ± 0.5
Maximum	7.3	10.2	14.5
98th Percentile	6.1	5.7	7.9

Road dust $PM_{2.5}$ was the biggest source of modeled emissions at Indianapolis (53% of total emissions), and non-exhaust emissions represented 60% of the total vehicle emissions. Based on the literature, which includes measurement-based estimates that road dust is 7-20% of traffic-related $PM_{2.5}$ near major roadways (Jeong et al., 2019; Pant and Harrison, 2013), the relative contribution of non-exhaust emissions is likely overestimated. Because AERMOD is a chemically inert model, we expect

time-averaged concentrations from AERMOD to scale roughly linearly with total $PM_{2.5}$ emissions. Thus, road dust contributes to about half (1.8 μg/m^3) of the modeled near-road $PM_{2.5}$ increment. If road dust were not included in the simulation, the bias in the averaged modeled increment would have been about a factor of two instead of a factor of 4. Askariyeh et al. (2019) found that the inclusion of road dust increased modeled near-road $PM_{2.5}$ concentrations by 49-74% depending on time-of-day and season. These results highlight the need for further study on re-suspended $PM_{2.5}$ road dust emissions, particularly because non-exhaust emission components do not benefit from tailpipe emission control technologies.

The average modeled $PM_{2.5}$ increment was 30% lower on weekends (2.8 μg/m^3) than on weekdays (4.0 μg/m^3) due to reduced weekend traffic volumes, which resulted in lower modeled $PM_{2.5}$ emissions. The AERMOD results were therefore very sensitive to the traffic volume. However, there was no statistically significant difference in the measured near-road increment at Indianapolis between weekdays and weekends. Across the national near-road monitoring network, past work has shown that traffic volume is not a strong predictor for near-road $PM_{2.5}$, and that the relative contribution of roadway emissions is also likely driven by other local emissions and meteorology (DeWinter et al., 2018; Seagram et al., 2019). In this case, near-road $PM_{2.5}$ concentrations modeled by AERMOD were more sensitive to nearby traffic volumes compared to the measurement data results, an outcome consistent with findings based on the entire national near-road network.

Both the model and observations showed extremes beyond 1.5 times the Inter-Quartile Range (IQR) of daily near-road $PM_{2.5}$ increments. Measured near-road $PM_{2.5}$ increments were negative on some days, and these negative daily increments were included in averaging calculations. Negative 24-hr average increments are considered valid since, on average, the uncertainty of the measured increment should not be systematically biased positive or negative. As Mukherjee et al. (2019) noted, the measured increments for Indianapolis are free of confounding factors that could impact the calculated increment. For example: the I-70 freeway and the near-road monitor are at the same elevation, there are no roadside barriers nearby, and the land use between the near-road site and background sites is similar.

The maximum daily near-road increment modeled by AERMOD (7.3 μg/m^3) was smaller than the maximum measured increment (10.2 μg/m^3, unpaired in time with the observations). The 98th percentile of the distribution of the modeled $PM_{2.5}$ increments, relevant for regulatory modeling analyses involving the 24-hour $PM_{2.5}$ NAAQS, compared well to the 98th percentile of measured daily increments. It is important to emphasize that comparisons involving individual days in this study are more uncertain than comparisons involving increments that have been averaged over many days.

We also analyzed modeled $PM_{2.5}$ increments under various daily average wind directions, when the Indianapolis near-road monitor was downwind, upwind, and parallel (34°-93° and 214°-273°) to I-70. The AERMOD base-case results did not exhibit a large variation based on the wind direction. The

average measured near-road increment was larger (1.5 μg/m^3) when the near-road monitor was downwind of I-70, and smaller (0.7 μg/m^3) when the near-road monitor was upwind. In contrast, the modeled near-road $PM_{2.5}$ increment was actually larger when the near-road monitor was upwind of I-70. This counter-intuitive result can be explained by the plume meander treatment in AERMOD and is discussed further in Section 4.

Modeled near-road $PM_{2.5}$ contributions from the various types of roadways were tracked through the AERMOD source grouping function and are summarized in Table 9. The majority (93%) of modeled $PM_{2.5}$ came from the mainline I-70 links that are directly adjacent to the Indianapolis near-road monitor. These I-70 mainline links represent the vast majority of the traffic volume (including truck traffic) in the modeling project area. About 5% of modeled $PM_{2.5}$ was from arterial roads within the project area. Contributions from other links including on-ramps and off-ramps, I-65, and the I-65/I-70 interchange were small.

Table 9. Modeled near-road $PM_{2.5}$ increments (μg/m^3) at the Indianapolis near-road monitoring site location contributed from different road segment groups for 152 modeled days during 2016.

Road Segment Group	Contribution to Average Modeled Near-Road $PM_{2.5}$ Increment (μg/m^3)	Percent Contribution
Mainline I-70	3.4	93
Arterials	0.2	5
I-70 ramps and interchange, and I-65	0.1	2
All modeled road segments	**3.7**	**100**

3.3 Sensitivity Simulations

Results of the sensitivity modeling simulations for the Indianapolis analysis are summarized in Figure 10 and in Table 10. The modeled multi-day-average near-road $PM_{2.5}$ concentrations were higher than the observed near-road increment for all four dispersion modeling scenarios and wind direction bins except the Cal3 scenario for upwind conditions. The maximum modeled daily $PM_{2.5}$ increment was smaller than the observed values for all four dispersion modeling scenarios. The modeled 98th percentile of daily average increments was higher or lower than the observed value, depending on the modeling scenario and the near-road monitor used to calculate the measured increment. As noted earlier, comparisons involving individual days are more uncertain than comparisons involving increments that

have been averaged over many days. Results and implications from each sensitivity scenario are discussed below.

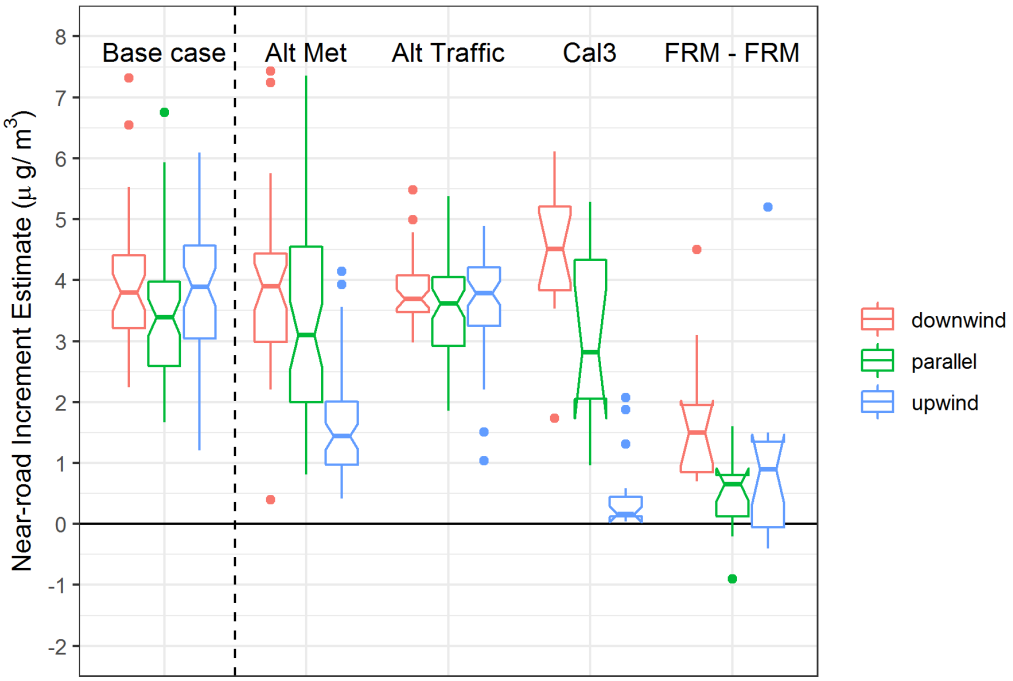

Figure 10. Distribution of modeled daily average $PM_{2.5}$ concentrations from the base case, Alt Met, Alt Traffic, and Cal3 scenarios, and measured FRM to FRM near-road $PM_{2.5}$ increments at the Indianapolis near-road monitoring site in 2016 during three wind conditions (near-road monitor downwind, upwind, and parallel to I-70). AERMOD results cover 152 days; Cal3 results cover 40 days.

Table 10. Summary statistics of modeled and measured near-road $PM_{2.5}$ increments (μg/m^3) at the Indianapolis near-road monitoring site. AERMOD results cover 152 days; Cal3 results cover 40 days.

$PM_{2.5}$ 24-hour Concentration	AERMOD Base Case	AltTraff	AltMet	Cal3	FRM Monitored Near-Road Increment	FEM Monitored Near-Road Increment
Average	3.7	3.7	2.8	2.6	0.9 ± 0.6	1.2 ± 0.5
Maximum	7.3	5.5	7.4	6.1	10.2	14.5
98th Percentile	6.1	5.3	6.3	5.6	5.7	7.9

3.3.1 Sensitivity to Travel Activity Data Processing Approach

The AltTraff scenario was conducted to assess the sensitivity of modeled near-road concentrations to the level of temporal detail provided in the emissions inputs. Compared to the base-case AERMOD results, the use of aggregated traffic data had little effect on the modeled average near-road $PM_{2.5}$ concentration, but reduced the maximum and 98th percentile of modeled daily average $PM_{2.5}$ concentrations and reduced the variance in the distribution of daily average concentrations. Aggregating the traffic data into a typical diurnal pattern averaged out extreme values for individual hours. The practical implication of this result is that using aggregated traffic data from travel demand models (which are typically available to practitioners in regulatory settings) will have little impact on period-average modeled concentrations, compared to using detailed day- and hour-specific data. Because of the high uncertainty in individual daily measured $PM_{2.5}$ increments, it is unclear whether using aggregated traffic data improves or degrades model performance for the highest predicted concentrations. The potential impact of using weekday diurnal traffic profiles rather than both weekday and weekend profiles was not investigated here, but could be important given that daily average traffic volumes and emissions in the model project area were about 30% lower on weekends than on weekdays.

3.3.2 Sensitivity to Meteorological Data Choice

The AltMet scenario was developed to assess the sensitivity of modeled near-road concentrations to the meteorological inputs. The base case used data at the near-road monitoring site, while the alternative meteorological data came from the NWS site at Indianapolis International Airport 14.5 km away. Compared to the base case, the use of NWS meteorology resulted in a lower average near-road $PM_{2.5}$ concentration by 0.9 $\mu g/m^3$ over the 152 analysis days. This resulted in a reduced bias with respect to the measured near-road increment, but the AltMet scenario still overestimated the average near-road $PM_{2.5}$ increment by more than a factor of 3. The maximum and 98th percentile of daily predicted $PM_{2.5}$ concentrations was similar between the AltMet and base-case simulations, suggesting that meteorological conditions leading to the highest AERMOD concentrations were similarly characterized in both the local near-road and NWS datasets.

The modeled $PM_{2.5}$ concentration in the AltMet case was higher during downwind conditions, and lower during upwind conditions. This contrasts with the base case in which the $PM_{2.5}$ concentration was higher when the near-road monitor was upwind of I-70. The emissions were identical between the two simulations, but the wind speeds were higher in the AltMet case. The reduction in upwind concentrations in the AltMet case suggests a reduced contribution from the AERMOD lateral plume meander algorithm. Lateral plume meander is more prominent during low wind speed conditions in the

base-case simulation, where, conversely, increased wind speeds in the AltMet case reduced the upwind plume meander contribution.

3.3.3 Sensitivity to Model Choice

The Cal3 scenario was developed to assess the sensitivity of modeled near-road concentrations to the choice of dispersion model. CAL3QHCR was modeled for a 40-day subset of the AERMOD analysis days with the highest base-case concentrations (20 in summer and 20 in winter). The average modeled near-road $PM_{2.5}$ concentration in the Cal3 simulation was lower than the AERMOD base case by about 1 $\mu g/m^3$. This resulted in a reduced bias with respect to the average measured near-road increment, but the Cal3 scenario still overestimated the near-road $PM_{2.5}$ increment by almost a factor of three.

The differences between the AERMOD base case and Cal3 simulations are also significant with respect to the wind direction sensitivity of the modeled concentrations. CAL3QHCR exhibited a large variation in concentrations with respect to the average daily wind direction, whereas the AERMOD results were relatively insensitive to wind direction. CAL3QHCR predicted a higher average $PM_{2.5}$ increment (4.9 $\mu g/m^3$) than AERMOD when the near-road monitor location was downwind of I-70, but a smaller increment during parallel wind conditions (2.8 $\mu g/m^3$), and an even smaller increment during upwind conditions (0.5 $\mu g/m^3$).

To ensure the differences found between the Cal3 and AERMOD findings were not just an artifact of the 40 days selected to be modeled, we compared the Cal3 results to the AERMOD results for the same 40-day sample. These findings are shown in Figure 11 and Table 11. When considering just the 40 analysis days modeled in the Cal3 scenario (see Figure 11 and Table 11), the measured increments are less certain (due to fewer number of measurement days), but the findings for the average increments are not substantially different from the findings shown in Figure 10 and Table 10. The maximum measured daily increment is substantially smaller when considering just the 40 analysis days modeled in the Cal3 scenario, and therefore there is much less discrepancy between the modeled and measured maximum daily increment compared to Table 10. This highlights the additional uncertainty in comparing individual daily increments in this study.

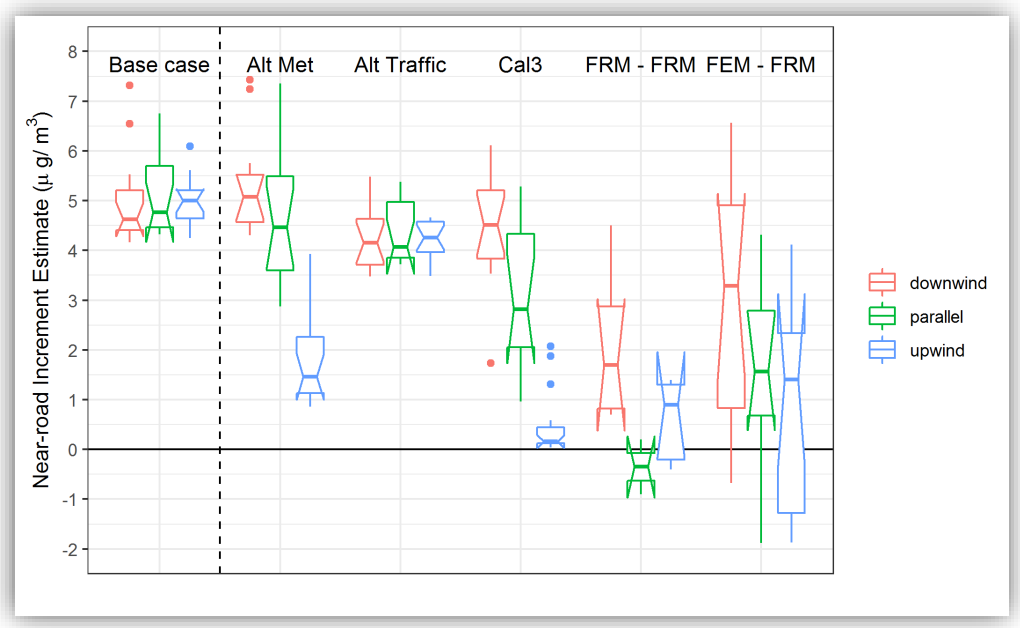

Figure 11. Distribution of modeled daily average $PM_{2.5}$ concentrations from the base case, Alt Met, Alt Traffic, and Cal3 scenarios, and measured FRM-based and FEM-based near-road $PM_{2.5}$ increments at the Indianapolis near-road monitoring site in 2016 during three wind conditions (near-road monitor downwind, upwind, and parallel to I-70), for days modeled in the Cal3 sensitivity simulation. AERMOD and Cal3 results cover the same 40 days.

Table 11. Summary statistics of modeled and measured near-road $PM_{2.5}$ increments (µg/m^3) at the Indianapolis near-road monitoring site for the analysis days modeled in the Cal3 sensitivity scenario. AERMOD and Cal3 results cover the same 40 days.

$PM_{2.5}$ **24-hour Concentration**	**AERMOD Base Case**	**Cal3**	**Monitored FRM-Based Near-Road Increment (n=14 days)**	**Monitored FEM-Based Near-Road Increment (n=38 days)**
Average	5.0	2.6	0.9 ± 0.9	0.9 ± 1.0
Maximum	7.3	6.1	4.5	6.6
98th Percentile	6.9	5.6	4.1	6.5

3.4 Providence Analysis

3.4.1 $PM_{2.5}$ Increment Analysis

We analyzed $PM_{2.5}$ data for years 2015-2016 from the Providence near-road monitor and nearby ambient monitors to estimate the Providence near-road increment and characterize its uncertainty. There were 426 days (398 construction days and 28 non-construction days) with coincident $PM_{2.5}$ data at the near-road monitor and at the nearby Urban League and Francis St. background monitors. Traffic data at Providence was not available for all 426 days, and therefore 382 days were modeled.

In contrast to the Indianapolis near-road site which had co-located FEM and FRM instruments that measured $PM_{2.5}$, the Providence near-road site had one FEM monitor. The nearby Urban League and Francis St. background sites have co-located FRM and FEM monitors. Like Indianapolis, there is a bias between co-located FRM and FEM measurements of $PM_{2.5}$ in Providence. Although the FRM monitors are more precise than the FEM monitors in Providence, we choose the FEM monitors to calculate increments for Providence to eliminate uncertainty associated with the bias between monitoring methods, as discussed in Section 3.1.

We calculated the near-road $PM_{2.5}$ increment at Providence using different combinations of the near-road FEM monitor and two nearby background FEM monitors (Urban League and Francis School) to characterize how the increment varied based on the background monitor choice. The average daily near-road FEM increments calculated for both the Urban League and Francis School monitors are summarized in Figure 12. Data are shown for various daily average wind directions, when the Providence near-road monitor was downwind (wind blowing from 220°-329°), upwind (40°-159°), and parallel (160°-219° and 340°-39°) to I-95. Wind directions are 24-hour vector averages calculated from the hourly

wind data. The median increment calculated between the Urban League monitor and the near-road monitor in 2015-2016 was 1.5 μg/m^3. The median increment calculated between the Francis School monitor and the near-road monitor was larger (2.4 μg/m^3). The 2015-2016 increments calculated here for Providence are smaller than the 2015 annual average $PM_{2.5}$ increments reported by De Winter et al. (2018) (2.7 to 3.4 μg/m^3 depending on the approach and monitoring sites used) but consistent with those reported by Mukherjee et al. (2019) (2.0 μg/m^3) when accounting for differences in monitoring methods (FRM vs. FEM) in the increment calculations.

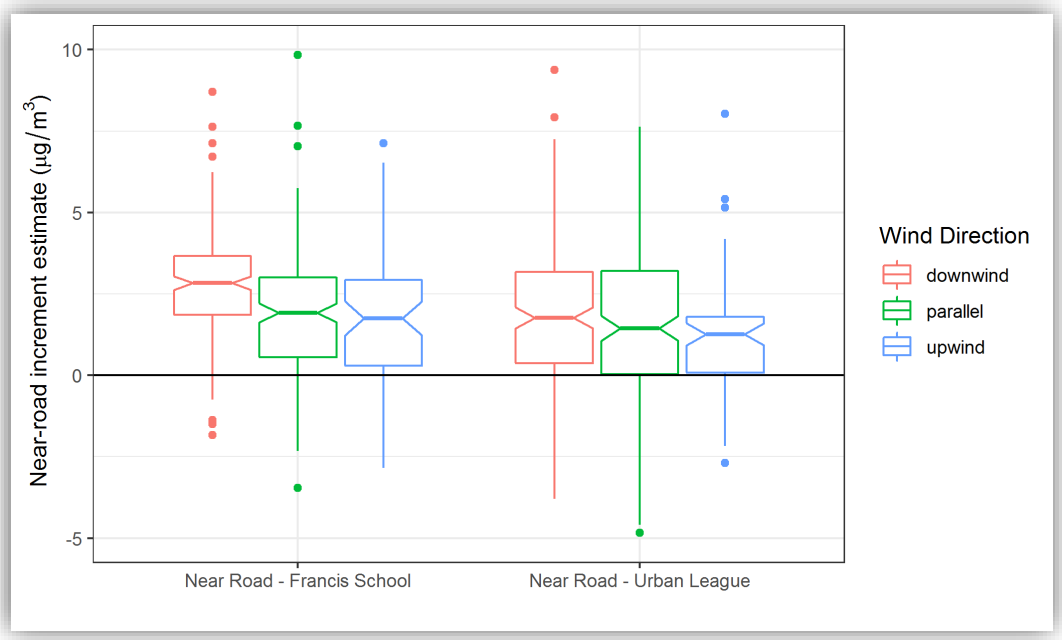

Figure 12. Providence near-road $PM_{2.5}$ increments between the near-road (NR) FEM monitor and nearby Urban League and Francis School FEM monitors when the Providence near-road monitor was downwind, upwind, and parallel to I-95.

We estimated the uncertainty of the near-road $PM_{2.5}$ increment measurements as the 95% confidence interval around the mean value. The uncertainty in the averaged increment at Providence is ± 0.2 μg/m^3 for days with nearby construction activity, ± 0.8 μg/m^3 for days without nearby construction activity, and ± 0.2 μg/m^3 for all analysis days. The uncertainty is larger for days without nearby construction activity because the sample size is smaller, but overall the uncertainty in the averaged increment measurement is sufficiently small to support reliable comparisons between modeled and measured near-road $PM_{2.5}$ increments. The uncertainty in individual daily measurements was not quantified, but given the large variability in daily measured increments, we expect comparisons of modeled and measured increments to be more uncertain when considering individual days.

For subsequent comparisons with modeled increments for Providence, we selected the near-road increment based on measurements from the near-road monitor and the Urban League FEM monitor (AQS ID 44-007-0022). This selection was based on: 1) the Urban League monitor is upwind of I-95 when the near road monitor is downwind of I-95; and 2) the highly urbanized land use around the Urban League monitor is more similar to the land use around the near-road monitor.

3.4.2 Emissions

A summary of average daily $PM_{2.5}$ emissions calculated from vehicle activity for the approximately 9 miles of roadways that were modeled within the Providence modeling project area are shown in Table 12. The average daily emissions across all 382 modeling analysis days in 2015-2016 were 54 lb/day. For the 27 non-construction days, the average daily emissions were 39 lb/day and for the 355 construction days, the average daily emissions were 55 lb/day. Construction equipment emissions were not modeled in this analysis; therefore, lower emissions on non-construction days are associated with lower traffic volumes on weekends and holidays. Although the total project emissions for Providence were smaller than for Indianapolis, the $PM_{2.5}$ emissions "per mile" of roadway was larger for Providence. This is explored further in Section 4. Exhaust emissions were 49% of total emissions, followed by road dust (44%), brake wear, and tire wear. Non-exhaust emissions (tire wear, brake wear, and road dust) represented 51% of the total vehicle emissions. Consistent with the Indianapolis analysis findings, the relative fraction of exhaust to non-exhaust $PM_{2.5}$ emissions for Providence was larger than those reported in the literature, and the inclusion of road dust $PM_{2.5}$ substantially affected the modeled emissions estimates.

Table 12. Summary of modeled $PM_{2.5}$ vehicle emissions for the Providence project area.

Process	Average Daily $PM_{2.5}$ Emissions [lb/day]	% of Total
Road dust (AP-42)	24	44
Running exhaust	26	49
Brake wear	3	5
Tire wear	1	2
Total	54	100

Emissions from the nearby construction activity were not modeled due to insufficient data regarding the construction vehicle activity. To estimate the potential uncertainty associated with this

construction activity, we used aerial drone footage provided by RIDOT for five different days during 2015 to characterize construction vehicle activity associated with the Providence Viaduct project (which occurred 200 to 600 m from the Providence near-road site), and then used the Caltrans construction emissions tool (Bai and Erdakos, 2018) to estimate hours of use and the $PM_{2.5}$ emissions from that activity. The estimated $PM_{2.5}$ emissions from the construction activity observed in the drone footage ranged from 0 to 1.5 lb/day. The observed equipment was idle on one of the drone footage days. The highest daily value (1.5 lb/day) is less than 5% of the modeled vehicle emissions for the Providence project area (54 lb/day). The construction activity is highly variable in terms of the types and number of vehicles, equipment operating load, emission control technology of the equipment, and spatial location of the activity, but based on these estimates, and the distances between construction work and the near-road monitor, this construction activity is expected to add less than about 5% uncertainty to the modeling analysis results.

3.4.3 Providence Scenario AERMOD Results

AERMOD modeling results for Providence are compared to measured increments for construction and non-construction days in Figure 13 and for all 382 analysis days combined in Figure 14. Summary statistics are shown in Table 13. Based on these results, AERMOD over-predicted the average near-road $PM_{2.5}$ increment at Providence. The over-prediction was more severe for the construction days than for the non-construction days. The average modeled $PM_{2.5}$ increment across all 382 analysis days (8.8 $\mu g/m^3$) was more than a factor of six (530%) larger than the average measured increment (1.4 $\mu g/m^3$). A high-bias in the modeled near-road increment was also found for the Indianapolis analysis, but the magnitude of the bias is larger in the Providence analysis (7.4 $\mu g/m^3$ compared to 2.8 $\mu g/m^3$). The averaged modeled increment is larger than the estimated uncertainty of the measured near-road increment ($\pm$ 0.2 $\mu g/m^3$) and larger than the variability in the increment due to the choice of background monitor (about 1.0 $\mu g/m^3$). On a relative basis, this bias is also substantially larger than the estimated uncertainty associated with the nearby construction activity (about 5%). Therefore, most of the uncertainty in the AERMOD result must be attributable to other factors.

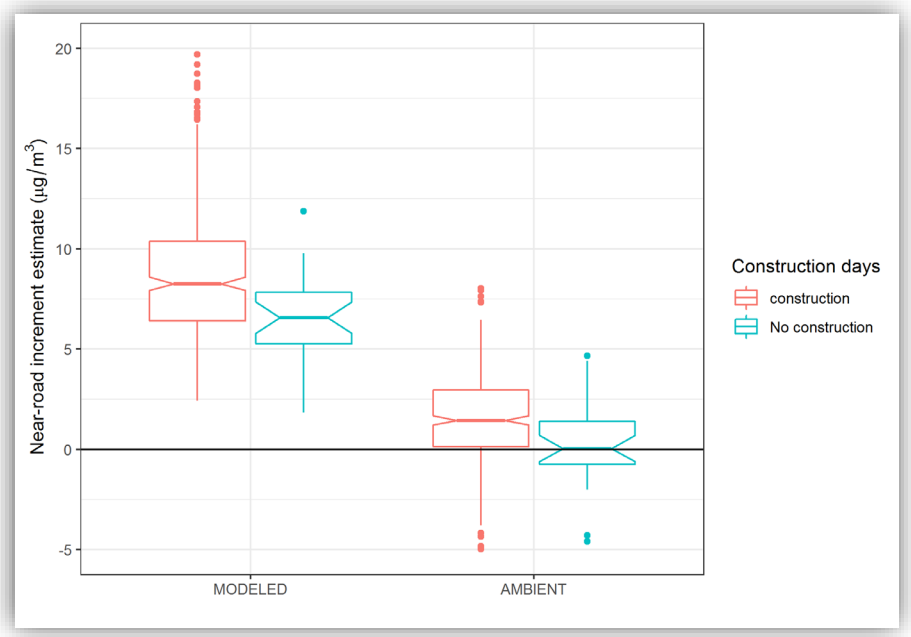

Figure 13. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured (ambient) near-road $PM_{2.5}$ increments at the Providence near-road monitoring site during 2015-2016 for days with construction (355 days) and with no construction (27 days). The horizontal line at the box notch indicates the median, box extents indicate the interquartile range (IQR), and whiskers indicate 1.5 times the IQR. Data beyond the whiskers are plotted individually as filled circles.

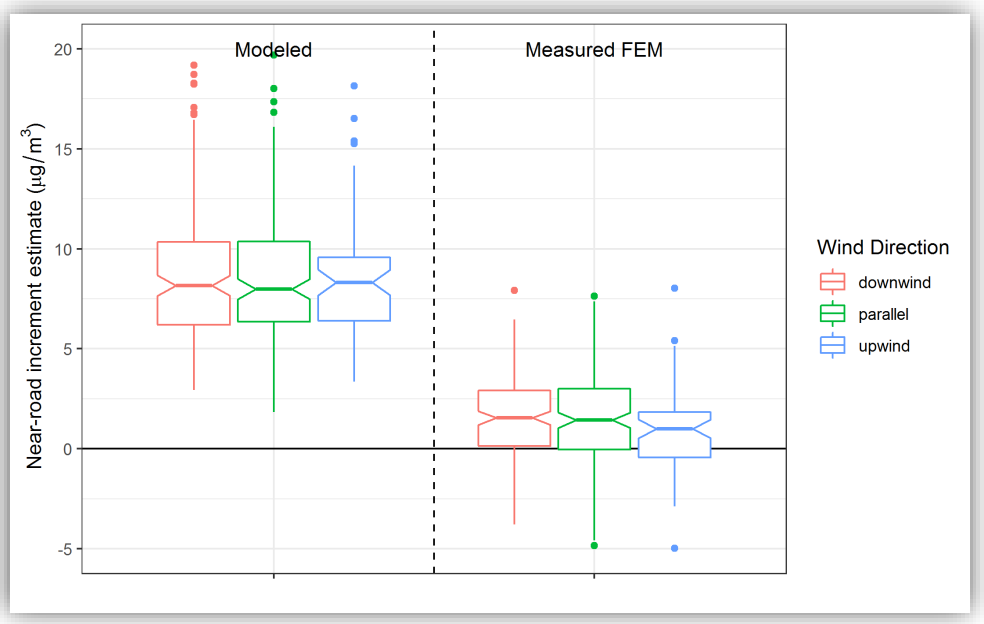

Figure 14. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured (ambient) near-road $PM_{2.5}$ increments at the Providence near-road monitoring site during 2015-2016 (382 days) for three wind conditions (near-road monitor downwind, upwind, and parallel to I-95).

Table 13. Summary of modeled and measured near-road $PM_{2.5}$ increments (µg/m^3) at the Providence near-road monitoring site.

$PM_{2.5}$ **24-hour Concentration**	**Days without nearby construction activity (n = 27)**		**Days with nearby construction activity (n = 355)**		**Days with and without nearby construction activity (n = 382)**	
	Modeled Increment	**Monitored Increment**	**Modeled Increment**	**Monitored Increment**	**Modeled Increment**	**Monitored Increment**
Average	6.5	0.3 ± 0.8	9.0	1.5 ± 0.2	8.8	1.4 ± 0.2
Maximum	11.9	4.7	22.0	8.0	22.0	8.0
98th Percentile	10.8	4.5	18.1	6.2	18.1	6.1

The modeled near-road increment was smaller on non-construction days (6.5 µg/m^3) compared to days with nearby construction activity (9.0 µg/m^3). Since construction equipment emissions were not modeled, this difference is due to the lower traffic volumes that were observed on weekends and holidays,

which resulted in lower modeled $PM_{2.5}$ emissions. This result further highlights the sensitivity of AERMOD results to traffic volume. The measured near-road increment was also smaller on the non-construction days, likely in response to the lower traffic volumes on weekends and holidays compared to weekdays when most construction days occurred. Note that any potential contributions from nearby construction activity cannot be quantified from the available air quality measurement data.

Both the model and observations showed extremes beyond 1.5 times the IQR of daily near-road $PM_{2.5}$ increments. As with Indianapolis, the monitored near-road $PM_{2.5}$ increments at Providence were negative on some days, and these negative daily increments were included in averaging calculations. The maximum daily near-road increment modeled by AERMOD (22.0 μg/m^3) did not compare well to the maximum measured increment (8.0 μg/m^3, unpaired in time with the observations). The 98th percentile of the distribution of the modeled $PM_{2.5}$ increments, relevant for regulatory modeling analyses, also did not compare well to the 98th percentile of measured daily increments, although comparisons involving individual days in this study are more uncertain than comparisons involving increments that have been averaged over many days.

Finally, we analyzed the modeled $PM_{2.5}$ increments under various daily average wind directions, when the Providence near-road monitor was downwind, upwind, and parallel to I-95. As with the Indianapolis AERMOD modeling results, the Providence AERMOD results did not exhibit a large variation based on the wind direction. The average measured near-road increment was larger (1.6 μg/m^3) when the near-road monitor was downwind of I-95, and smaller (0.8 μg/m^3) when the near-road monitor was upwind. Average daily modeled near-road $PM_{2.5}$ increments were similar for all wind directions (within 4%).

4 Discussion

4.1 Synthesis of Modeling Results

A synthesis of Indianapolis and Providence modeling results is shown in Table 14. The near-road increments modeled by AERMOD were larger than the measured near-road increment for both the Indianapolis and Providence analyses, but the modeled increment was more than a factor-of-two larger at Providence compared to Indianapolis. This difference is supported by two key differences between the two modeling analyses. First, the Providence near-road monitor is only 5.0 meters from the edge of the freeway, whereas the Indianapolis near-road monitor is 24.5 meters from the freeway edge. Pollutant concentrations decrease exponentially as downwind distance increases (Wen et al., 2017; Venkatram et al., 2013). When very close to the road, a 20 m difference in downwind distance can result in greater than 50% difference in the modeled concentrations. Second, although the modeled $PM_{2.5}$ emissions in the

Providence analysis were smaller than those modeled in the Indianapolis analysis, there were fewer miles of roads in the Providence project area (9 miles of roadways) compared to the Indianapolis project area (20 miles of roadways). Therefore, the $PM_{2.5}$ emissions source strength on a per-mile basis was larger at Providence. The stronger $PM_{2.5}$ emissions source strength for the Providence roadways is due to higher emission factors (on a grams per vehicle-mile basis) from the MOVES model, which is related to differences in local vehicle fleet characteristics, such as the age distribution, that are built into the local MOVES fleet data provided by INDOT and RIDOT. Although the AADT for Providence I-95 was about 41% higher than for Indianapolis I-70, the heavy-duty truck percentage at Providence was lower. As a result, the FE-AADT at both sites was similar, and therefore differences in traffic volumes do not explain the differences in modeled near-road increments.

Table 14. Model parameter comparison between the Indianapolis and Providence modeling analyses.

Parameter	Indianapolis	Providence
Modeling analysis year	2016 (n=152 days)	2015/2016 (n=382 days)
AADT (% Heavy Duty Truck)	165,672 (14%)	233,036 (7%)
FE-AADT	374,419	363,549
$PM_{2.5}$ average daily exhaust and non-exhaust emissions within project area [lb/day] (% road dust)	70 (53%)	54 (44%)
$PM_{2.5}$ average daily "per mile" exhaust and non-exhaust emissions within project area (lb/day/mile)	3.5	6.2
AERMOD average $PM_{2.5}$ increment ($\mu g/m^3$)	3.7	8.8
AERMOD peak 24-hr $PM_{2.5}$ increment ($\mu g/m^3$)	7.3	22.0
Average measured increment	0.9 ± 0.6	1.4 ± 0.2
Receptor distance to road	24.5 m	5.0 m

The modeling chain for predicting near-road PM concentrations in this study consisted of 1) travel activity developed from traffic monitor data; 2) emissions modeling with MOVES (for vehicle exhaust, tire wear, and brake wear emissions) and use of AP-42 methods for re-suspended road dust emissions; and 3) air quality dispersion modeling with AERMOD or CAL3QHCR. When AERMOD was

used, the average near-road $PM_{2.5}$ increment predicted by the modeling chain was more than 300% (factor of four) larger than the measured increment at Indianapolis, and more than 500% (factor of six) larger than the measured increment at Providence. When CAL3QHCR was used, the predicted near-road $PM_{2.5}$ increment was around 200% (factor of three) larger than the measured increment at Indianapolis. These biases are much larger than the uncertainty in the measured near-road $PM_{2.5}$ increment, and are also larger than the variability in the measured near-road increment due to the choice of background $PM_{2.5}$ monitor (0.4 $\mu g/m^3$ at Indianapolis and 0.9 $\mu g/m^3$ at Providence).

The biases in predicted near-road $PM_{2.5}$ increments reported in this study reflect cumulative uncertainty throughout the near-road PM modeling chain. Based on the analysis results from this study and insights from the scientific literature, the relative uncertainties that may be attributable to each step in the near-road $PM_{2.5}$ modeling chain are discussed below. For the analyses conducted in this study, the overall uncertainty in the modeling chain is likely dominated by uncertainties in the emissions and dispersion modeling components.

4.2 Uncertainties in the Near-Road $PM_{2.5}$ Modeling Chain

Use of the best available local traffic data helped minimize uncertainties associated with the travel activity data. The data processing approach is based on hourly measurements of traffic volume, truck percentage, and vehicle speed. Travel activity from freeways, which account for the vast majority of VMT and diesel truck traffic in the modeling project areas, were developed directly from monitored hourly traffic data. The travel activity analysis in this study involved substantially more rigor compared to a typical project-level regulatory (i.e., conformity) air quality analysis, which often relies on travel demand model output rather than a comprehensive assessment of traffic monitor data. Uncertainties associated with the travel activity data that dominated modeled and measured near-road $PM_{2.5}$ in this study are likely to be small compared to uncertainties associated with other components in the near-road PM modeling chain.

Uncertainty in the vehicle-related $PM_{2.5}$ emissions likely contributes to the modeling chain biases reported in this study. The relative contribution of non-exhaust emissions (compared to exhaust emissions) at Indianapolis and Providence was high compared to the near-road literature. The measurement data available at the Indianapolis and Providence sites are not sufficient to conclude whether the non-exhaust emissions were overestimated, the exhaust emissions were underestimated, or both the exhaust and non-exhaust emissions were overestimated. Although brake and tire wear emissions modeled with MOVES are uncertain, they were a relatively small percentage (7%) of the total modeled emissions; therefore uncertainties in brake and tire wear emissions cannot fully explain the modeled $PM_{2.5}$ concentration biases. As previously noted, if road dust $PM_{2.5}$ emissions were not included in the modeling

simulations, the bias in the averaged modeled increment would have been about a factor of two instead of four at Indianapolis, and about a factor of three instead of six at Providence. If the road dust emissions from AP-42 are overestimated, this alone would not fully explain the modeled $PM_{2.5}$ concentration biases. Recent real-world vehicle testing (Quiros et al., 2016; Dixit et al., 2017; Thiruvengadam et al., 2015) and findings from the MOVES Federal Advisory Committee Act (FACA) Review Workgroup (Sandhu and Sonntag, 2017) indicate that MOVES is overestimating exhaust $PM_{2.5}$ emission factors from heavy-duty trucks for model years 2010 and newer. EPA is planning to incorporate these recent vehicle test results into the next update of the MOVES model (Sandhu and Sonntag, 2019).

Uncertainty in the dispersion model itself may also contribute to the modeling chain biases reported in this study. The biases reported here are larger than the cited ±50% "irreducible" uncertainty associated with the physics and formulation of the Gaussian plume models (Hanna, 1982). For the Indianapolis analysis, the averaged near-road $PM_{2.5}$ increment concentration modeled by AERMOD was more than 1 $\mu g/m^3$ larger than the increment modeled by CAL3QHCR. AERMOD has shown a tendency to overestimate pollutant concentrations under low wind speed (< 2 m/s) conditions (Paine et al., 2015; Qian and Venkatram, 2011). AERMOD predicted the largest near-road $PM_{2.5}$ increment concentrations when the near-road monitor was upwind of the freeway. AERMOD incorporates a lateral plume meander algorithm (for volume sources) to account for the slow lateral shifting of plumes from non-diffusing eddies during lower wind speed conditions. The plume meander algorithm can produce non-zero concentrations upwind of emission sources. In a tracer study evaluation, Askariyeh et al. (2017) showed that AERMOD overestimated low concentrations at upwind locations when using volume sources (which incorporates plume meander) and underestimated upwind concentrations when using area sources (which does not treat plume meander). CAL3QHCR does not include a plume meander algorithm, and therefore there is no upwind dispersion in CAL3QHCR. The CAL3QHCR model underestimated the near-road increment during upwind conditions (Figures 10 and 11). EPA has been working to address potential issues related to plume meander in AERMOD, including the effect that plume meander has on concentration in low wind speed conditions and the degree to which upwind dispersion should be applied (U.S. Environmental Protection Agency, 2017c).

Meteorological input data are important for the dispersion model and the near-road PM modeling chain, and the Indianapolis modeling results showed that near-road $PM_{2.5}$ concentrations modeled by AERMOD are sensitive to the meteorological inputs. The reduction in the average $PM_{2.5}$ concentration in the AltMet case (which used non-local NWS airport data, 14.5 km from the near-road monitoring site) compared to the base case simulation (which used local data at the near-road monitoring site) can be explained by differences in wind speeds measured at the two meteorological data sites (see Figure 4). The average wind speed at the NWS airport site was 4.5 m/s, compared to 2.1 m/s at the near-road site. The

airport site is located in a wide open space unobstructed by buildings or vegetation, whereas the Indianapolis near-road site is located in a suburban area with trees and buildings within a few tens of meters from the site. This difference in exposure resulted in higher wind speeds at the NWS site, which increased dispersion, and thereby decreased modeled near-road $PM_{2.5}$ concentrations in the AltMet case.

Despite differences in modeled concentrations between the AltMet and base case simulations, AERMOD substantially overestimated the average near-road $PM_{2.5}$ concentrations in both simulations. Therefore, uncertainties in the meteorological data alone would not be sufficient to explain the modeling chain bias. The bias was somewhat less severe in the AltMet case, and it is notable that AERMOD compared more favorably to observations when using the non-local (nearby airport) meteorological data. In many dispersion modeling analyses, nearby airport data is considered representative for the surrounding region, but in this case, site-specific data were available and helped demonstrate that the nearby airport, with few surrounding obstacles, had higher wind speeds than those seen at the near-road site. The use of airport data in the AltMet case resulted in modeled increments that were lower, and therefore closer to the measured values compared to the base case; however, the modeled concentrations in the AltMet case were not derived from the most representative meteorological data available.

Most of the overall difference between the base case and AltMet case results was driven by differences in the predicted upwind concentrations. The modeled near-road concentrations during upwind conditions were much lower in the AltMet case compared to the base case, likely due to the higher wind speeds in the AltMet case. The modeled near-road concentrations in the AltMet case were also more sensitive to variations in wind direction compared to the base case. These results further highlight the sensitivity of AERMOD near-road modeling results to lateral plume meander during low wind speed conditions.

In the Indianapolis analysis, the majority (93%) of modeled $PM_{2.5}$ came from the mainline I-70 links that are directly adjacent to the near-road monitor. Modeled contributions from other links, including the I-65/I-70, interchange were small. The I-65/I-70 interchange is approximately 1 km from the Indianapolis near-road monitor, and was too far away to contribute significantly to the modeled $PM_{2.5}$ at the near-road monitor. In the Providence analysis, the vast majority of vehicle emissions were from the mainline I-95 links. In a regulatory PM hot-spot analysis, roadways that are substantially affected by a proposed transportation project must be modeled. Understanding the relative importance of vehicle emissions from different types of roadways at varying distances from a target project site can help guide practical decisions about what roadways should be included in a modeling analysis, to ensure that potential near-road PM hot spots are adequately captured.

4.3 Limitations

There are limitations of this study that warrant further research. First, the study conclusions are specific to the two high-volume roadways at the Indianapolis and Providence near-road air quality monitoring sites. The key findings are consistent across both study sites, but further analysis work is needed across different geographic settings, roadway types, and configurations. Further analysis work involving multiple methodologies (e.g., tracer evaluations) is also needed to help identify findings common to multiple settings, and to build a more complete picture of near-road modeling chain biases that should be addressed.

Second, the results and discussion provide important insights on components that appear to be driving the uncertainty in the near-road $PM_{2.5}$ modeling chain for the Indianapolis and Providence analyses, but additional research is needed to quantify the bias that can be attributed to individual modeling chain components. Measurements of speciated $PM_{2.5}$ components in the near-road environment would help constrain uncertainties associated with the relative contribution of exhaust and non-exhaust emissions. Studies that compare modeled and measured near-road carbon monoxide (CO) increments could provide additional insight on biases that could be attributed to the dispersion model component, since modeled CO emissions from vehicles are expected to be more certain than modeled $PM_{2.5}$ emissions.

Finally, the measurement uncertainty in the average near-road $PM_{2.5}$ increments was quantified, but uncertainty in individual daily increments was not. Comparisons of modeled and measured increments are more reliable when averaged over many days, and are less reliable when considering just a single day. Several years of measurement data are needed to develop a robust uncertainty estimate for individual daily increments and improve the confidence in comparisons involving the maximum and 98th percentile of measured increments. Analyses involving pollutants with small background concentrations, such as CO, could also reduce uncertainty associated with estimating near-road increments.

5 Conclusion

We developed a primary dispersion modeling analysis for year 2016 centered on a $PM_{2.5}$ monitoring site near the I-70 freeway in Indianapolis, Indiana, to compare near-road $PM_{2.5}$ concentrations predicted by the AERMOD dispersion model to measured near-road $PM_{2.5}$ increment concentrations. To provide additional context, a secondary dispersion modeling analysis was conducted for a $PM_{2.5}$ monitoring site near the I-95 freeway in Providence, Rhode Island. Predictions of near-road PM concentrations involved a modeling chain that included travel activity data processing, emissions modeling, and air quality dispersion modeling. These modeling analyses were built upon bottom-up

estimates of temporally and spatially resolved roadway $PM_{2.5}$ emissions based on detailed traffic monitoring data and current emission factor databases for the local vehicle fleet characterization. Dispersion model simulations were driven with local meteorological data collected at the Indianapolis near-road monitoring site, except for one sensitivity simulation that used alternative meteorological data. For Providence, representative meteorological data 2.4 km away were used.

We estimated the increment between $PM_{2.5}$ concentrations at the Indianapolis FRM near-road monitor and at the nearby Washington Park urban air quality monitor to be 0.9 ± 0.6 μg/m^3. Similarly for Providence, we estimated the increment between $PM_{2.5}$ concentrations at the Providence near-road FEM monitor and at the nearby Urban League air quality monitor to be 1.4 ± 0.2 μg/m^3. These multi-day-average values were within the range of increments that were estimated at these sites in prior studies in the literature. The relatively small uncertainty in the increment measurement supports reliable comparisons between modeled and measured near-road $PM_{2.5}$ increments.

Modeled multi-day-average roadway contributions to near-road concentrations substantially exceeded measured values based on the near-road monitoring data. The average near-road $PM_{2.5}$ increment for Indianapolis modeled with AERMOD was 3.7 μg/m^3, which was more than 300% (factor of four) larger than the measured increment (0.9 μg/m^3) for the 152-day analysis period. Road dust $PM_{2.5}$, calculated using the EPA AP-42 approach, was the biggest source of modeled emissions (53% of total emissions), and non-exhaust emissions represented 60% of the total vehicle emissions. For Providence, the average near-road $PM_{2.5}$ increment modeled with AERMOD was 8.8 μg/m^3, which was more than 500% (factor of six) larger than the measured increment for the 382-day analysis period. For both analyses, the relative contribution of non-exhaust emissions (compared to exhaust emissions) was high compared to the near-road literature. The biases in predicted near-road $PM_{2.5}$ increments reflect cumulative uncertainty throughout the near-road $PM_{2.5}$ modeling chain.

Additional dispersion modeling simulations for Indianapolis were also conducted to assess the sensitivity of modeled results to the choice of model (AERMOD or CAL3QHCR), meteorological data, and travel data processing approach. In all three sensitivity simulations, the modeled near-road $PM_{2.5}$ increment was substantially larger than the measured increment with model biases ranging from 191% to 312%. The CAL3QHCR results more closely tracked observed variations in near-road increment with wind direction compared to the AERMOD results. The AERMOD modeling results were sensitive to the choice of meteorological data input (local data vs. nearby NWS airport site). Although the airport data was less representative than the local meteorological data, the use of airport data reduced the overestimation in near-road increments modeled by AERMOD during upwind conditions. Aggregating the hourly travel data into a typical-day diurnal profile resulted in a modest reduction in the maximum modeled $PM_{2.5}$ increments, but had virtually no effect on the averaged modeled increment. Based on the

study results and insights from the scientific literature, the overall uncertainty in the modeling chain for the Indianapolis and Providence analyses was likely dominated by uncertainties in the emissions and dispersion modeling components.

The maximum modeled daily $PM_{2.5}$ increment was smaller than the observed values for the dispersion modeling scenarios assessed. The modeled 98th percentile of daily average increments was higher or lower than the observed value, depending on the modeling scenario and the near-road monitor used to calculate the measured increment. In this study, comparisons involving individual days are more uncertain than comparisons involving increments that have been averaged over many days; the findings from this work therefore emphasized the multi-day-averaged results.

This work provided a unique evaluation of near-road $PM_{2.5}$ concentrations predicted by the near-road PM modeling chain, and provides valuable information for practitioners to further understand potential sources of uncertainty in the near-road PM modeling process. Additional research related to $PM_{2.5}$ emission factors is needed to help constrain modeling uncertainties in real-world applications. In particular, additional research is needed to quantify uncertainties in the emissions and dispersion model components of the near-road PM modeling chain. Long-term near-road measurements of $PM_{2.5}$ species, including black carbon, metals, and crustal minerals, coupled with dispersion modeling analysis and evaluation, could provide additional insights and help constrain uncertainties associated with roadway $PM_{2.5}$ emissions. Further work is also recommended to assess AERMOD model performance, particularly for low-wind speed conditions and near-road settings within 50 m of the roadway edge. Further modeling and analysis work using other near-road settings with different roadway configurations, traffic patterns, and vehicle fleet characteristics, will help identify findings common to multiple settings.

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