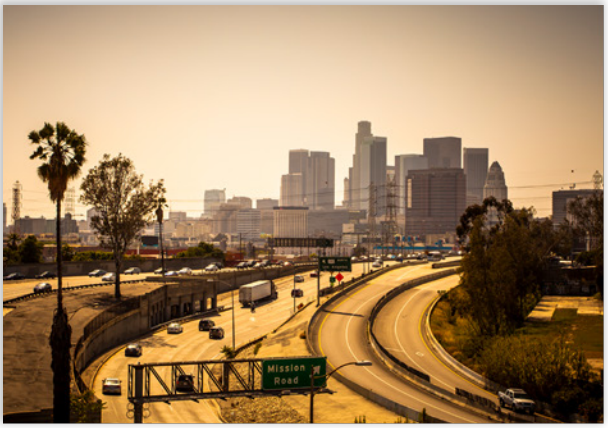
Summary Findings: Near-Road Air Quality Transportation Pooled Fund



Summary Report Prepared for

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Summary Findings: Near-Road Air Quality Transportation Pooled Fund

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Acknowledgments and Sources

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Peer-Reviewed Journal Articles

Brown S.G., Penfold B.M., Mukherjee A., Landsberg K., and Eisinger D. (2019) Conditions leading to elevated $PM_{2.5}$ at near-road monitoring sites: case studies in Denver and Indianapolis. *International Journal of Environmental Research and Public Health*, 16(9), 1634, doi: 10.3390/ijerph16091634 (STI-7047), May 10.

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

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

Reid S., Bai S., Du Y., Craig K., Erdakos G., Baringer L., Eisinger D., McCarthy M., and Landsberg K. (2016) Emissions modeling with MOVES and EMFAC to assess the potential for a transportation project to create particulate matter hot spots. *Transportation Research Record: Journal of the Transportation Research Board*, 2570, 12-20, doi: 10.3141/2570-02 (STI-6330).

Conference Presentations

Brown S., Craig K., Eisinger D., Landsberg K., Mukherjee A., Baringer L., Chang S.Y., DeWinter J., McCarthy M., and Huang S. (2019) National assessment of near-road (NR) air quality: requirements, trends, and analysis insights. Presented for the U.S. Transportation Research Board webinar series, September 30, by Sonoma Technology, Inc., and the Washington State Department of Transportation.

Craig K., Baringer L., Chang C., Bai S., Landsberg K., and Eisinger D. (2019) Near-road $PM_{2.5}$, modeled vs. monitored data comparison: Indianapolis case study. Presented at the Transportation Research Board 2019 Annual Meeting, Analysis Subcommittee, Transportation and Air Quality Committee, January 14, by Sonoma Technology, Inc., Petaluma, CA, and the Washington State Department of Transportation, Olympia, WA. STI-7020.

Seagram A., Huang S., Brown S., Landsberg K., DeWinter J., and Eisinger D. (2019) National assessment of near-road air quality: trends and estimation of near-road $PM_{2.5}$ increment. Presented at the Transportation Research Board 2019 Annual Meeting, Analysis Subcommittee, Transportation and Air Quality Committee, Washington, DC, January 14, by Sonoma Technology, Inc., Petaluma, CA. STI-7034.

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DeWinter J.L., Brown S.G., Graham A.R., Landsberg K., and Eisinger D.S. (2016) National-scale air quality data assessment: initial findings from the near-road NO_2 monitoring program. Poster presented at the *National Ambient Air Monitoring Conference, St. Louis, MO, August 8-11*. STI-6503.

Reid S., Bai S., Du Y., Craig K., Erdakos G., Baringer L., Eisinger D., and McCarthy M. (2016) Emissions modeling with MOVES and EMFAC to assess the potential for a transportation project to create particulate matter hot spots. Presentation given at the *2016 Transportation Research Board Annual Meeting, Washington, DC, January 13,* by Sonoma Technology, Inc., Petaluma, CA. STI-914202-6417.

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TPF Reports

Craig K., Baringer L., Chang S.Y., McCarthy M., Bai S., Ravi V., Eisinger D. (2019) Analysis of Modeled and Measured Near-Road $PM_{2.5}$ Concentrations in Indianapolis and Providence During 2015 and 2016. Final report prepared for the Washington State Department of Transportation, lead agency for the Near-Road Air Quality Research Transportation Pooled Fund, by Sonoma Technology, Inc., Petaluma, CA, STI-914202-7127. October 31.

Mukherjee A., McCarthy M.C., Brown S.G., Huang S., Landsberg K., and Eisinger D.S. (2019) Influence of roadway emissions on near-road $PM_{2.5}$: monitoring data analysis and implications for determining transportation projects of local air quality concern (POAQC). Final report prepared for the Washington State Department of Transportation, lead agency for the Near-Road Air Quality Research Transportation Pooled Fund, by Sonoma Technology, Inc., Petaluma, CA, STI-914203-7088, October.

DeWinter J., Brown S., and Eisinger D. (2019) Factors affecting estimates of background concentrations. Final report prepared for the Washington State Department of Transportation, Olympia, WA, by Sonoma Technology, Inc., Petaluma, CA, STI-914203-7078, May 2.

Seagram A., Brown S., Huang S., Penfold B., and Eisinger D. (2018) National near-road data assessment: report No. 3 with 2016 data. Final report prepared for the Washington State Department of Transportation, Olympia, WA, by Sonoma Technology, Inc., Petaluma, CA, STI-914203-6821, February 15.

Graham A., McCarthy M., Baringer L., Pavlovic N., Brown S., and Eisinger D. (2016) Effects of roadside barriers on near-road pollutant concentrations. Revised final report prepared for the Washington State Department of Transportation, Olympia, WA, by Sonoma Technology, Inc., Petaluma CA, STI-914205-6495, December.

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Reid S.B., Bai S., Eisinger D.S., Erdakos G.B., Du Y., and Baringer L. (2016) Scoping study to identify potential project types and situations that will not create PM hot spots. Final report prepared for the Washington State Department of Transportation, Seattle, WA, by Sonoma Technology, Inc., Petaluma, CA, STI-914202-6259-FR, February.

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Graham A.R. and Eisinger D.S. (2015) Case studies of truck replacement mitigation programs. Final report prepared for the Washington State Department of Transportation, Olympia, WA, by Sonoma Technology, Inc., Petaluma, CA, STI-914205-6229-FR, September.

DeWinter J.L., Brown S.G., Graham A.R., Landsberg K., and Eisinger D.S. (2015) National-scale air quality data assessment: initial findings from the near-road NO_2 monitoring program. Poster presented at the *AGU Fall Meeting, San Francisco, CA, December 14-18*. STI-6332.

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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 FHWA and the Arizona, California, Colorado, Ohio, Texas, and Virginia Departments of Transportation. Sonoma Technology, Inc., provided TPF-5(284) participants with technical, planning, facilitation, and website support.

Near-Road Air Quality Transportation Pooled Fund Major Findings

Background. Near-road air quality has been recognized for many years as an important environmental concern. Federal and state regulations require transportation agencies to complete air quality-related assessments for transportation projects, especially those with substantial truck activity. In addition, beginning 2014, U.S. EPA-mandated near-road air quality monitoring began operations across numerous U.S. metropolitan areas. In response, the U.S. Federal Highway Administration (FHWA) and seven state departments of transportation (DOTs) created a transportation pooled fund (TPF) to assess near-road air quality. TPF partners sought to understand near-road air quality data; improve near-road air quality (hot-spot) evaluations; implement effective mitigation; and more effectively respond to stakeholder information requests.

Methods. The TPF, managed by the Washington State DOT, established a five-year research program and selected Sonoma Technology, Inc. (STI) to complete analyses. Work included development of a strategic research plan; evaluation of near-road air quality data; emissions and air quality modeling to assess current and forecasted near-road conditions; completion of national-scale and case study evaluations; evaluation of potential mitigation; and dissemination of findings and resources via peer-reviewed articles, conference presentations, creation of near-road dataset resources to facilitate ongoing evaluations, and development and use of an information-exchange website.

Results. Research generated numerous insights of practical value to transportation practitioners.

> ***Emissions highlights***: As fleet turnover substantially reduces on-road exhaust emissions over the next several decades, the relative importance of non-exhaust particulate matter (PM) emissions grows. More work is needed to quantify non-exhaust PM, especially given disparities between EPA-based and California Air Resources Board-based emissions models.
>
> ***Measured concentration highlights***: Near-road measurements of carbon monoxide (CO) and nitrogen dioxide (NO_2) demonstrate that neither CO nor NO_2 concentrations adjacent to major U.S. roads exceed existing (as of 2019) National Ambient Air Quality Standards (NAAQS). $PM_{2.5}$ concentrations near major roads exceed the form of the daily and annual NAAQS in a limited number of locations, primarily in California. Multi-year data is

suggestive of declining near-road $PM_{2.5}$ concentrations, consistent with modeled emissions changes over time, although more data is needed to establish trends. Near-road concentrations are influenced by site characteristics that vary widely, including factors such as the distance between the road and air quality measurements, traffic conditions, wind direction and speed, roadway geometry, nearby structures and emissions sources, and (importantly) background concentrations. Given the diversity of factors that affect concentrations, near-road $PM_{2.5}$ concentrations are only weakly correlated with traffic conditions. The incremental $PM_{2.5}$ concentration contribution from major roads ranges from approximately 0.0 to 2.0 $\mu g/m^3$, based on recent (2017) U.S. data and background concentration calculation methods that place greater weight on background monitors located close to near-road monitoring sites. Findings from these assessments can assist interagency consultation regarding identification of projects of local air quality concern (POAQC), and determinations of whether detailed quantitative PM hot-spot modeling analyses are required in given project situations.

Modeled concentration highlights: An in-depth case study of Indianapolis, Indiana found that modeled near-road $PM_{2.5}$ concentrations uniformly exceeded measured concentrations. The findings were consistent across various sensitivity tests, including use of the AERMOD and CAL3QHCR dispersion models, and supplemental case study assessments for Providence, Rhode Island. Modeled concentrations exceeded measured concentrations by at least a factor of two to three for the case studies examined. More work is needed to assess underlying reasons behind over-predicted modeling outcomes, and to develop case studies that expand the site conditions used for comparison.

Mitigation highlights: The TPF completed exploratory work to examine two mitigation actions: truck replacements and use of near-road barriers such as sound walls or vegetative screens. A California truck replacement case study was used to assess potential to mitigate short-term project impacts. The case provided implementation lessons regarding identification of candidate trucks for replacement. Implementation can be aided by targeting trucks that operate near ports, where port entry/exit slips and/or radio-frequency identification (RFID) tag tracking may help track truck activity. The TPF also assessed use of near-road barriers as a potential approach to reduce roadway-related pollutant concentrations downwind of a road. The literature documented 20–60% concentration reductions within the first 100 meters of a road, assuming perpendicular wind conditions and a barrier of typical height (e.g., 6 m) set roughly at the edge of the road shoulder. Simplified dispersion modeling performed for the TPF indicated that concentrations downwind of a barrier are typically lower than they would be in the absence of a barrier and that barrier effectiveness increases with increasing barrier height, consistent with the literature. The modeling work used a simplified site illustration and a tool (R-LINE) under development by EPA at the time work was completed. Further research is needed to more fully assess the ability of modeling tools to simulate barrier effects and to examine how effects vary with site-specific conditions.

Executive Summary

Introduction

Beginning in 2014 and extending through 2019, eight federal and state agencies participated in a Transportation Pooled Fund (TPF) to address near-road air quality issues. The Near-Road Air Quality TPF research program included participants from: the U.S. Federal Highway Administration (FHWA) and the Arizona, California, Colorado, Ohio, Texas, Virginia, and Washington State Departments of Transportation (DOTs). Researchers from Sonoma Technology, Inc. (STI) provided the TPF with technical support. The Washington State DOT (WSDOT) served as the TPF lead agency and managed the overall research program.

During the five-year period the TPF was active, participants engaged in several jointly sponsored efforts. These included development of a strategic research plan; evaluation of near-road air quality data; emissions and air quality modeling to assess current and forecasted near-road conditions; completion of national-scale and case study evaluations; and dissemination of findings and resources via peer-reviewed articles, conference presentations, creation of near-road dataset resources to facilitate ongoing evaluations, and development and use of an information-exchange website.

This summary presents a digest of some of the major findings from the TPF effort regarding

- CO, NO_2, and $PM_{2.5}$ near-road pollutant concentrations
- Background concentration assessments
- $PM_{2.5}$ increments (background concentration subtracted from near-road concentration)
- Projections to address fleet turnover impacts on exhaust emissions
- Findings to assist with identification of Projects of Local Air Quality Concern (POAQC)
- Comparisons of modeled and measured $PM_{2.5}$

Other findings are included in the main report, as well as additional detail on the summary findings included here.

Near-Road Pollutant Concentration Findings

For research purposes, this discussion includes comparisons of measured data to NAAQS levels. These comparisons are provided for context and are not meant to assess attainment status. Attainment and nonattainment areas are designated by the EPA. This Executive Summary includes example findings for selected analysis years. Generally, pollutant-specific findings illustrated here were consistent across the analysis years covered by TPF assessments (2014-2017).

Carbon Monoxide (CO): The 8-hour CO concentrations measured at near-road monitoring sites in 2016 ranged from -0.40 ppm to 3.51 ppm (**Figure ES-1**). There were no exceedances of the 8-hour NAAQS threshold of 9 ppm.

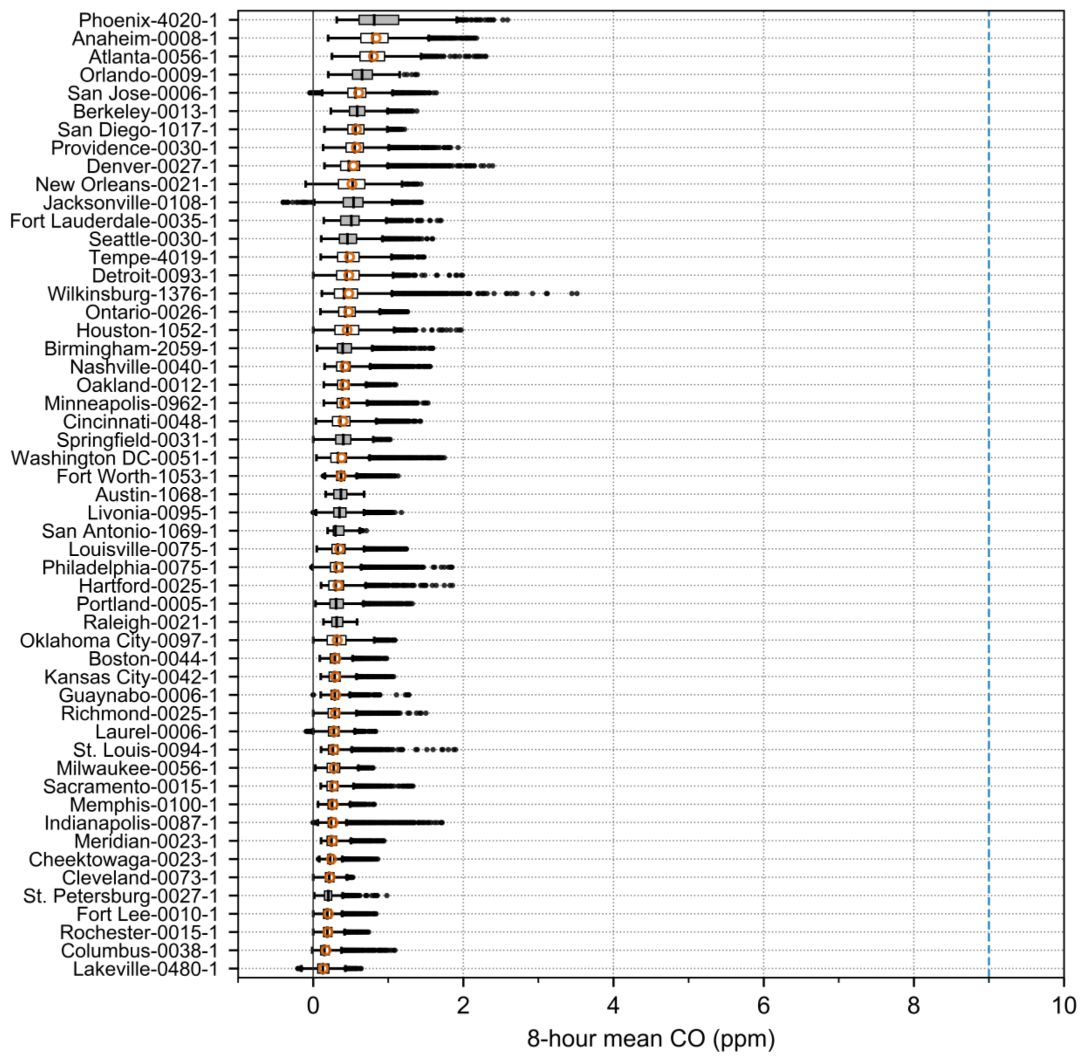

Figure ES-1. Distribution of 8-hour CO concentrations at near-road monitoring sites in 2016. Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) is displayed for monitors with complete annual datasets. The blue dashed lined denotes the 8-hour NAAQS threshold (9 ppm).

Nitrogen Dioxide (NO_2): Figure ES-2 summarizes the distributions of 1-hour NO_2 in 2016. Only four 1-hour NO_2 values exceeded 100 ppb across all sites.

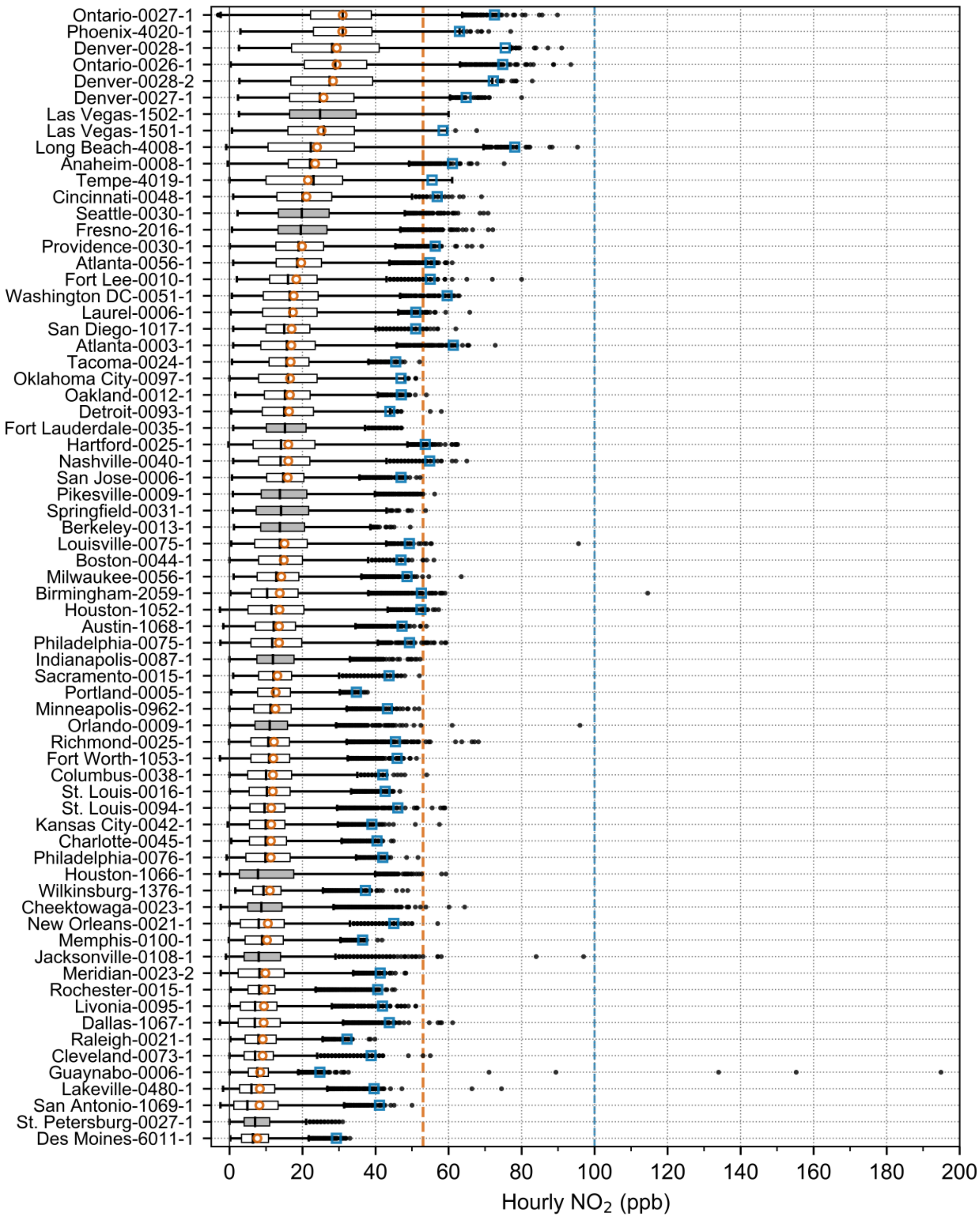

Figure ES-2. Distribution of 1-hour NO_2 concentrations at near-road monitoring sites in 2016.[1] Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) and 98th percentile of 1-hour daily maximum concentrations (blue square) are displayed for monitors with complete annual datasets. The orange dashed lined denotes the annual NAAQS threshold (53 ppb), and the blue dashed line the 1-hr NAAQS threshold (100 ppb).

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[1] Box extents indicate the 25th and 75th percentile of data. The horizontal line within the box indicates the 50th percentile (median). The whiskers have a maximum length equal to at most 1.5 times the length of the box (the interquartile range, IQR). Individual data points beyond the whiskers are plotted as individual points.

Particulate Matter Less Than or Equal to 2.5 Microns in Diameter ($PM_{2.5}$): In 2016, $PM_{2.5}$ was monitored at 42 sites, six of which had collocated monitors, resulting in 48 total $PM_{2.5}$ monitors. There were 29 monitors that measured $PM_{2.5}$ on an hourly basis, while the remaining 19 monitors sampled $PM_{2.5}$ locations on a 24-hour basis at various sampling frequencies (daily, 1-in-3, or 1-in-6). In this analysis, we aggregated all 1-hour $PM_{2.5}$ measurements to 24-hour mean $PM_{2.5}$ if the 75% completeness criterion for each day was met. Twenty-eight monitors had complete data for 2016.

Figure ES-3 presents 24-hour and annual mean $PM_{2.5}$ findings. There were 19 instances where 24-hour $PM_{2.5}$ concentrations exceeded 35 µg/m^3. The only site where the 98th percentile of 24-hour $PM_{2.5}$ concentrations exceeded 35 µg/m^3 in 2016 was at Ontario-0027-1.

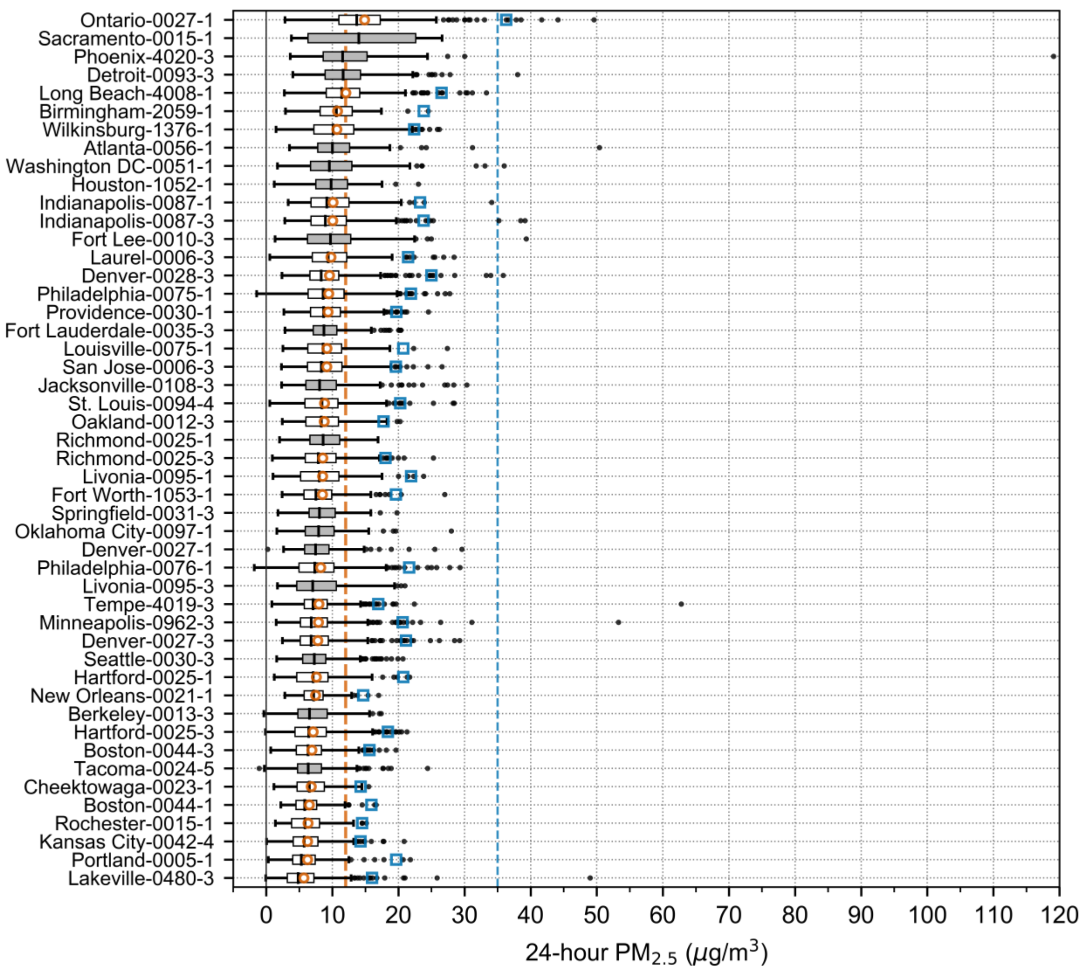


Figure ES-3. Distribution of 24-hour $PM_{2.5}$ concentrations at near-road monitoring sites in 2016. Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) and 98th percentile of 24-hour $PM_{2.5}$ concentrations (blue square) are displayed for monitors with complete annual datasets. The orange dashed lined denotes the annual NAAQS threshold (12 µg/m^3), and the blue dashed line denotes the 24-hour NAAQS threshold (35 µg/m^3).

Background Concentration Assessments

Numerous practical challenges can make background concentration estimation difficult for particular sites. STI assessed the range of $PM_{2.5}$ concentrations measured at monitors within each of the metropolitan areas for which a $PM_{2.5}$ near-road monitor was operational in 2017. The results provide quantitative bounds for how much uncertainty can be introduced in project analyses by the incorrect selection of a background monitor. $PM_{2.5}$ hot-spot analyses are performed by estimating both the 24-hour and annual forms of the $PM_{2.5}$ NAAQS (also referred to as "design values").[2] Both forms are calculated by using three years of measurements. Given the requirement to use three years of data for design value calculations, to complete the background concentration analyses discussed here we obtained and evaluated three years of data for monitors that might be used to derive background concentration values (data from 2015, 2016, 2017, obtained from the EPA Air Data web portal).

Overall, 45 core-based statistical areas (CBSAs) were evaluated in these analyses. As shown in **Figure ES-4**, in half of the CBSAs, the maximum estimated background concentration, using either the three-year average of annual means or three-year average of 98th percentiles of daily means, could be from 25% higher to 200% higher than the minimum estimated background concentration, depending on monitor choice. In the other half of CBSAs, monitor choice results in a more similar background value (i.e., less than a 25% difference in design value).

[2] See https://www.epa.gov/criteria-air-pollutants/naaqs-table.

Figure ES-4. Range of 3-year average annual mean (top) and range of 3-year average of the 98th percentile of daily mean (bottom) across monitors, by CBSA area; "n" indicates the number of monitors in each CBSA (excluding the near-road monitor).

$PM_{2.5}$ Increments Based on 2017 Data

A key TPF analysis objective was to improve understanding of the difference between pollutant concentrations observed adjacent to major roads and concentrations measured in surrounding areas. In simple terms, the roadway contribution, or increment, is calculated by subtracting a background concentration from a measured near-road concentration. If background is assessed correctly, the difference between the two values (near-road minus background) represents the incremental pollutant concentration contributed by the roadway.

Here, we present findings from an assessment of $PM_{2.5}$ increments using 2017 data. In this work, we identified and removed data that could potentially affect the increment analyses due to confounding factors. Removing sites with a noted confounding factor resulted in a sample of 20 near-road sites. Increments from the 20 sites are presented in Figure ES-5. The upper bound of $PM_{2.5}$ increments is 2.04 μg/m^3. Only three sites have an increment greater than 1.44 μg/m^3; monitors at each of these three sites are sited less than 10 meters from the roadway.

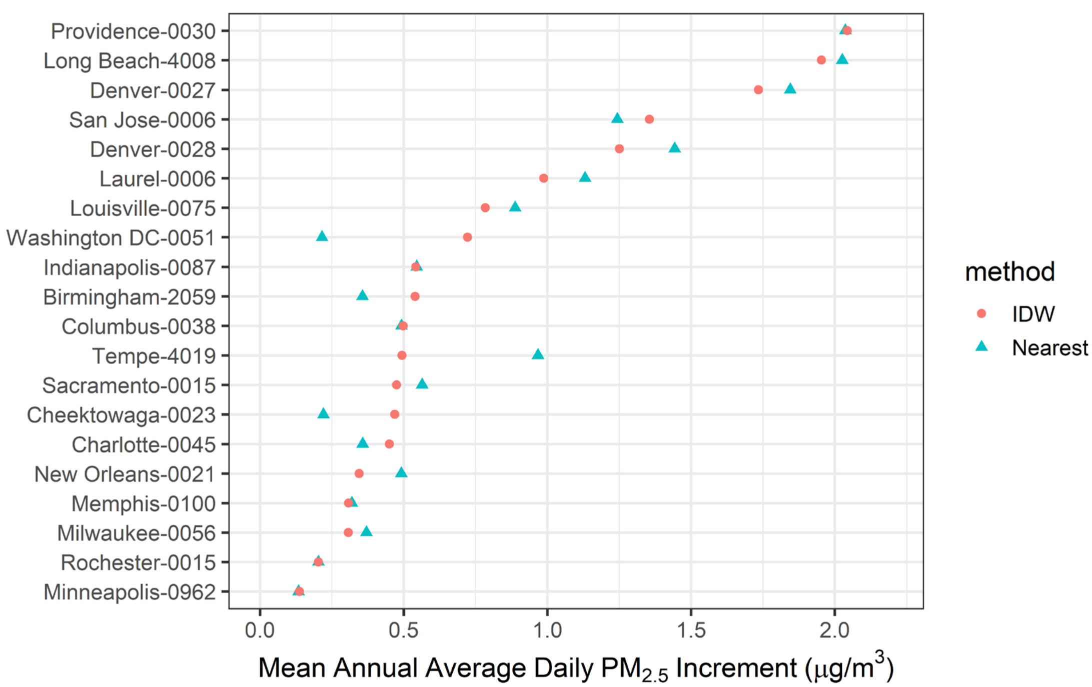


Figure ES-5. Distributions of annual average $PM_{2.5}$ increments computed using inverse distance weighting (IDW) and nearest monitor calculations (Nearest). Results for 20 sites are shown, controlling for confounding factors.

Using an initial case of 31 sites (to improve sample size), and a focused case of 20 increments (to remove confounding factors), we also used 2017 data to assess the relationship of near-road increments to variables representing meteorology, traffic, and site characteristics. Findings are illustrated in Figure ES-6.

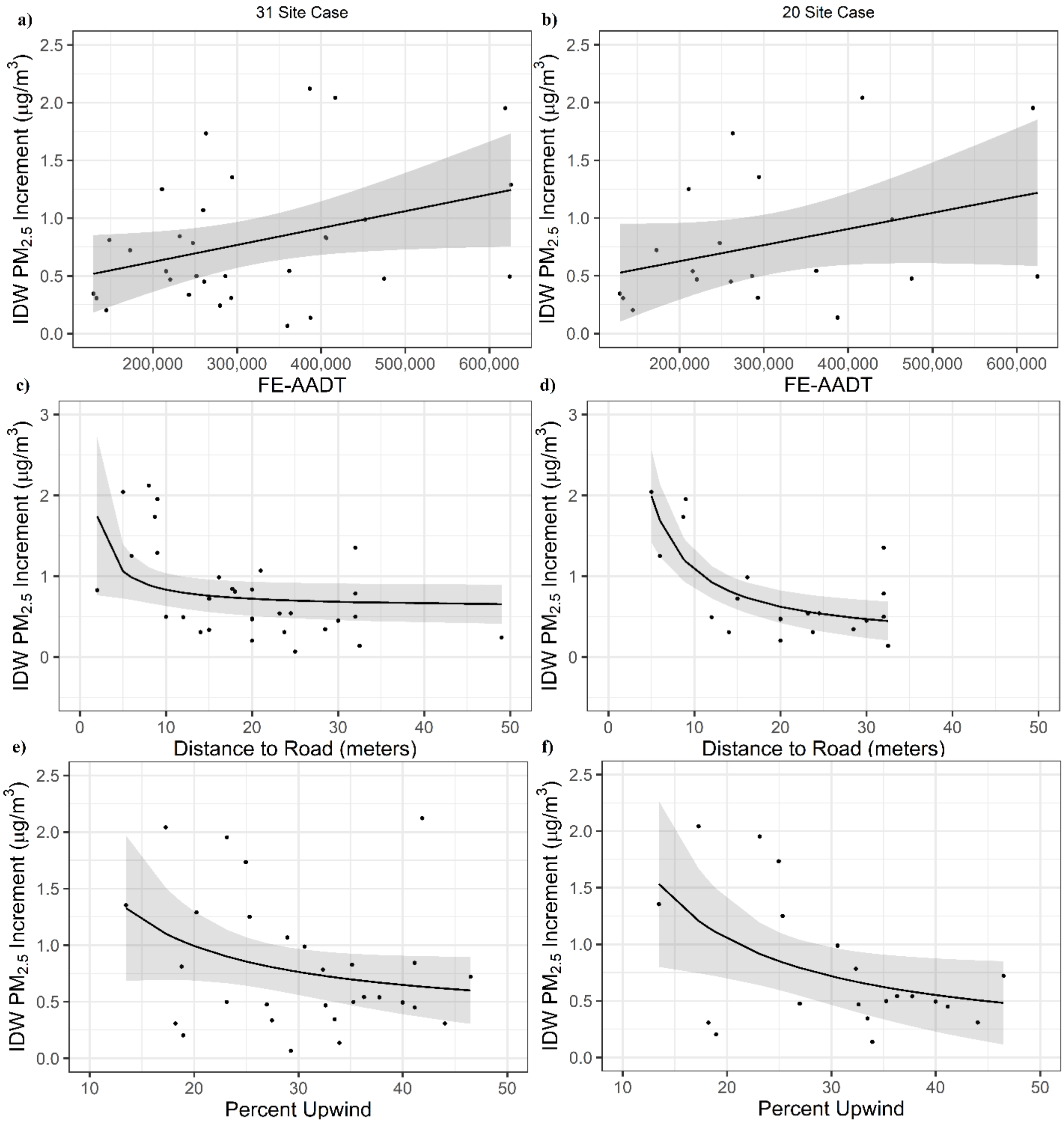


Figure ES-6. The relationship between the IDW $PM_{2.5}$ increment in comparison to FE-AADT (a and b), distance to road (c and d), and percent of time upwind (e and f). The initial case of 31 near-road sites is shown at left (a, c and e), and the focused case of 20 sites limiting confounding factors is shown at right (b, d and f). Regressions are shown in black, with the range of the standard error of the regression line shown in dark gray.

Projections to Address Future Fleet Turnover

The projected change in exhaust emissions for U.S. vehicles for the calendar years 2017 (baseline) to 2040 are shown in **Table ES-1.** Data in Table 9 show percent change in emissions relative to the year 2017, for heavy-duty diesel vehicles (HDDVs), light-duty vehicles (LDVs), and an average vehicle in a vehicle fleet composed of 8% HDDVs and 92% LDVs. A decrease of exhaust emissions is shown in all cases, due to fleet turnover. For example, the exhaust emissions of a roadway with 8% HDDV in San Francisco are projected to decrease by 88% by 2040 using EMFAC, assuming constant AADT and a constant HDDV fraction of 8%. The specific change in exhaust emissions for a given roadway will depend on changes in the regional vehicle fleet and traffic activity over time.

Table ES-1. Projected change of exhaust-only $PM_{2.5}$ emissions of vehicles for the 2018-2040 calendar years, relative to the baseline year of 2017. The percent of 2017 emissions is shown for heavy-duty vehicles (HDDV), light-duty vehicles (LDV), and a fleet mix with 8% HDDV and 92% LDV. Emissions are shown for a national average modeled using MOVES, and for San Francisco (SF) using EMFAC.

Year	MOVES HDDV	MOVES LDV	MOVE HDDV 8%	EMFAC SF HDDV	EMFAC SF LDV	EMFAC SF HDDV 8%
2017	100	100	100	100	100	100
2018	87	93	88	80	101	83
2019	75	87	79	63	102	70
2020	66	81	70	48	100	57
2025	34	62	42	7	80	20
2030	18	49	28	7	60	16
2035	13	42	21	7	46	14
2040	12	39	20	7	39	12

The upper bound of traffic-related $PM_{2.5}$ represented by the increment for near-road sites in 2017 can be combined with the projected change in exhaust emissions (Table ES-1), and an assumed fraction of traffic-related $PM_{2.5}$ due to exhaust, to forecast the upper bound of traffic-related $PM_{2.5}$ in the coming decades. In 2017, the upper bound of the observed annual average $PM_{2.5}$ traffic impact was 2.04 ± 0.16 $\mu g/m^3$, with an upper bound of 1.44 ± 0.17 $\mu g/m^3$, for sites with monitors 10 meters or more from the roadway (Figure ES-5). As an example case, assume 40% of total roadway-related $PM_{2.5}$ emissions are related to exhaust, and 60% are related to non-exhaust emissions such as brake wear, tire wear, and road dust. The change in the $PM_{2.5}$ traffic impact for a given project can then be calculated using the equation:

$PM_{2.5}$ increment (future year) μg/m^3 = 1.224 + 0.816 • [Percent change from 2017 to future year]

Where 1.224 μg/m^3 is the 60% of traffic-driven $PM_{2.5}$ that is attributed to non-exhaust factors and 0.816 μg/m^3 is the 40% of traffic-driven $PM_{2.5}$ that is attributed to exhaust, which is forecast to decrease. Likewise, for distances 10 meters or more from the roadway, we have the equation:

$PM_{2.5}$ increment, 10 meters or greater from roadway (future year) μg/m^3 = 0.864 + 0.576 • [Percent change from 2017 to future year]

The projected change of the $PM_{2.5}$ increment is illustrated in the following examples. These examples are based on the formulas above, use the MOVES national average emissions estimate, and assume a project with constant vehicle speeds and HDDV fraction of 8% for all years. The exhaust emissions are projected to be 42% of 2017 emissions by 2025 and 20% of 2017 emissions by 2040. Using the equations above, the $PM_{2.5}$ impact of a highway would fall from 2.04 ± 0.16 μg/m^3 in 2017 to 1.57 ± 0.16 μg/m^3 by 2025, and to 1.39 ± 0.16 μg/m^3 by 2040, if AADT, fleet mix, and vehicle speeds remain constant. Following the same method, for the domain greater than or equal to 10 meters from the roadway, the $PM_{2.5}$ impact of a highway would fall from 1.44 ± 0.17 μg/m^3 in 2017 to 1.11 ± 0.17 μg/m^3 by 2025 and 0.98 ± 0.17 μg/m^3 by 2040.

$PM_{2.5}$ Findings to Assist with Identification of Projects of Local Air Quality Concern (POAQC)

Traffic activity data developed by the EPA for a hypothetical project was adjusted to match the EPA POAQC example of 125,000 AADT and 8% diesel truck traffic. From this starting point, the team estimated PM_{10} and $PM_{2.5}$ emissions for the hypothetical project for a 2006 base year (to match the year of EPA's PM hot-spot rulemaking), for analysis years ranging from 2007 to 2035, and for a range of vehicle fleet compositions (i.e., percentage of diesel trucks).

MOVES-based $PM_{2.5}$ emissions estimates for the hypothetical project are cut approximately in half between 2006 and 2015 and are reduced by 92% between 2006 and 2035. EMFAC-based $PM_{2.5}$ estimates are also cut approximately in half between 2006 and 2015 and are reduced by about 70% between 2006 and 2035. EMFAC-based brake wear $PM_{2.5}$ emission estimates are consistently about eight times higher than MOVES-based estimates, limiting the overall reduction in project-level emissions (**Figure ES-7**). In addition, the contribution of brake wear and tire wear to the overall EMFAC-based $PM_{2.5}$ inventory rises from 30% in 2006 to 95% in 2035. For MOVES, the increase in the contribution of these processes to the overall inventory is less pronounced but also significant, rising from 4% in 2006 to 52% in 2035.

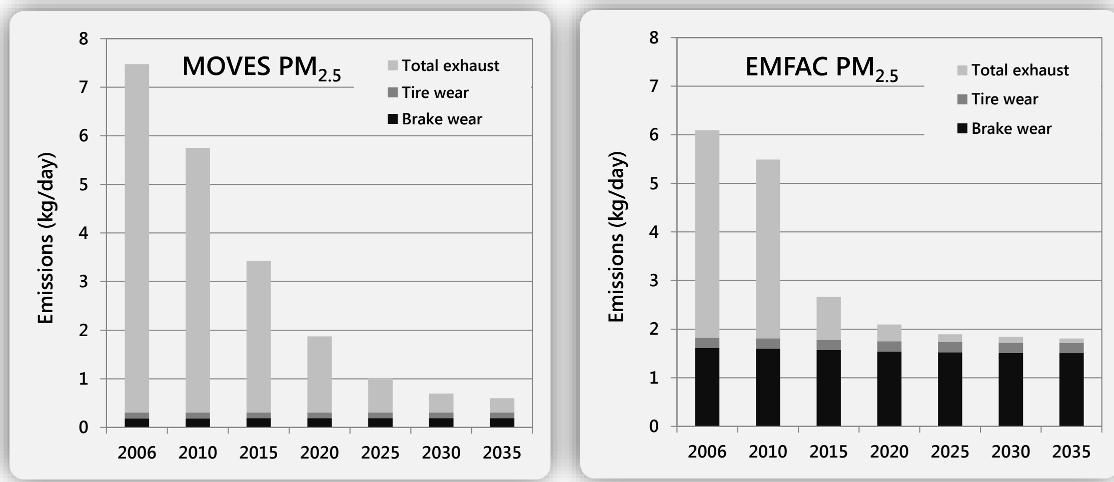


Figure ES-7. $PM_{2.5}$ emissions for a hypothetical freeway project with an AADT of 125,000 vehicles, 8% of which are diesel trucks.

Because of fleet turnover effects, producing emission levels equivalent to 2006 (125,000 AADT) requires higher traffic volumes in later years. To examine this effect, the team held the truck percentage constant at 8% and calculated AADT required to generate emissions totals in later years that are equivalent to the 2006 baseline emissions. For example, producing project-level PM_{10} emissions equivalent to those from the 2006 analysis year in 2020 would require traffic volumes of 180,000 vehicles for MOVES-based analyses and 167,000 vehicles for EMFAC-based analyses (Figure ES-8). By 2035, traffic volumes of 200,000 for MOVES-based analyses and 175,000 for EMFAC-based analyses would be required to match 2006 emission levels, increases of 60% and 40%, respectively.

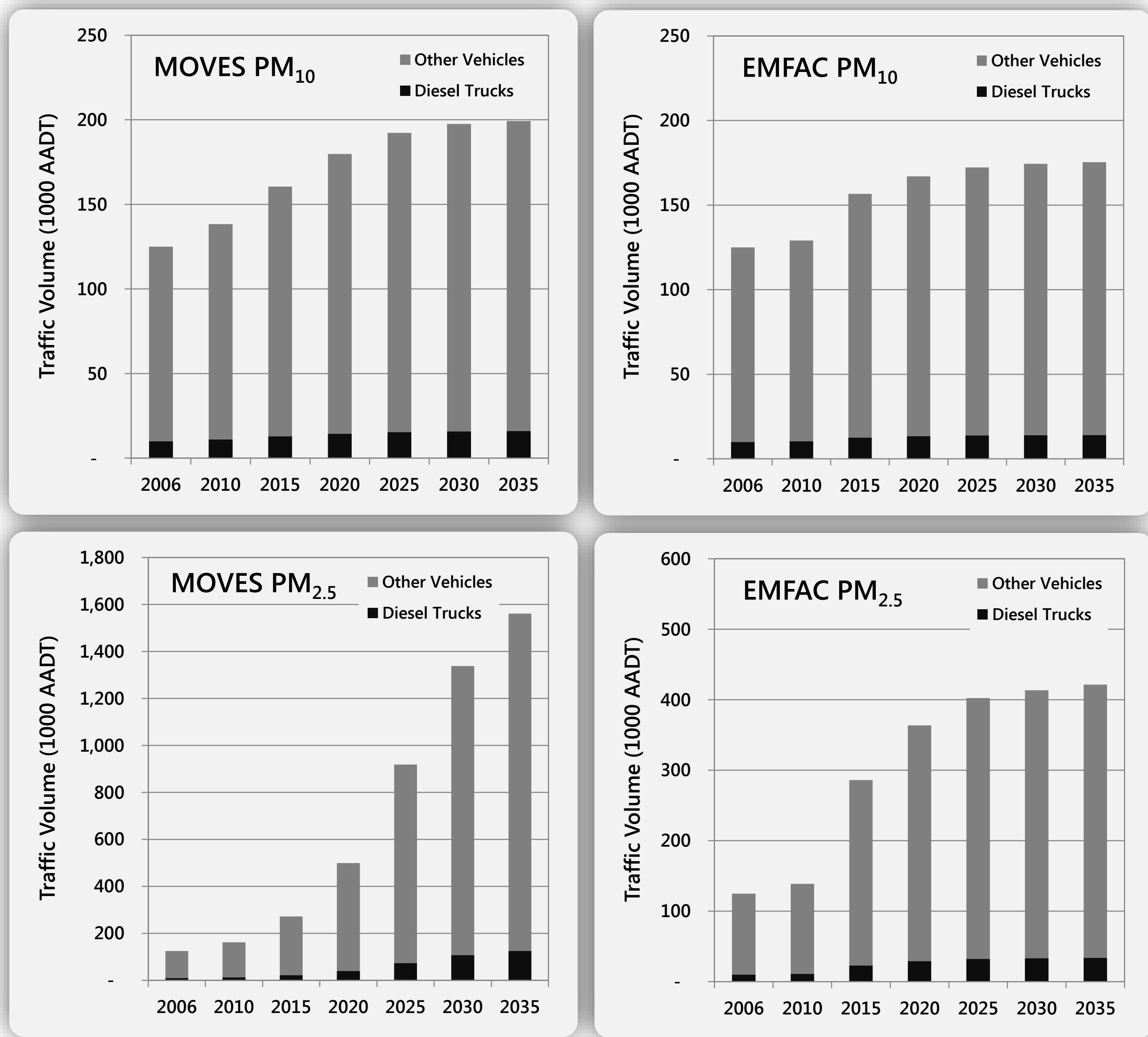


Figure ES-8. Projected traffic volumes needed to produce 2006-equivalent emissions (Scales are different for MOVES- and EMFAC-estimated $PM_{2.5}$ emissions).

Comparisons of Modeled and Measured $PM_{2.5}$

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) 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 evaluated a $PM_{2.5}$ monitoring site near a major freeway in Indianapolis, Indiana, for 2016. In the secondary analysis, we evaluated a site in close proximity to a major freeway in Providence, Rhode Island, for 2015-2016. The 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. 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 measured increments.

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 ES-9**. Based on these results, the AERMOD-based analysis over-predicted the average near-road $PM_{2.5}$ increment. The average modeled increment (3.7 µg/m^3) was a factor of four larger than the measured FRM-based increment (0.9 µg/m^3), and a factor of three larger than the measured FEM-based increment (1.2 µg/m^3).

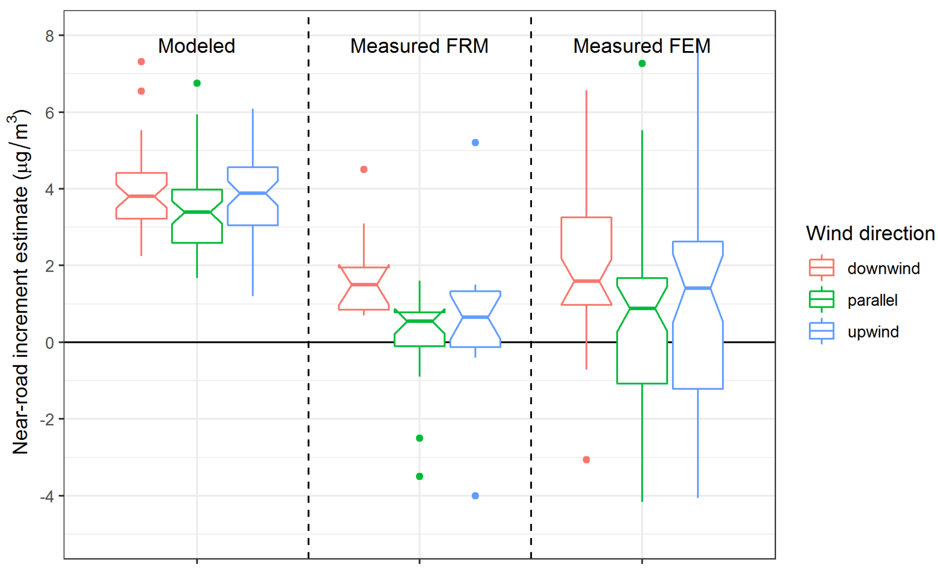


Figure ES-9. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured near-road $PM_{2.5}$ increments at the Indianapolis near-road monitoring site during 2016 for three wind conditions (near-road monitor downwind, upwind, and parallel to I-70).

AERMOD modeling results for Providence were compared to measured increments for 382 analysis days (**Figure ES-10**). The AERMOD-based analysis for Providence over-predicted the average measured near-road $PM_{2.5}$ increment. The average modeled $PM_{2.5}$ increment across all 382 analysis days (8.8 µg/m^3) was more than a factor of six (530%) larger than the average measured increment (1.4 µg/m^3).

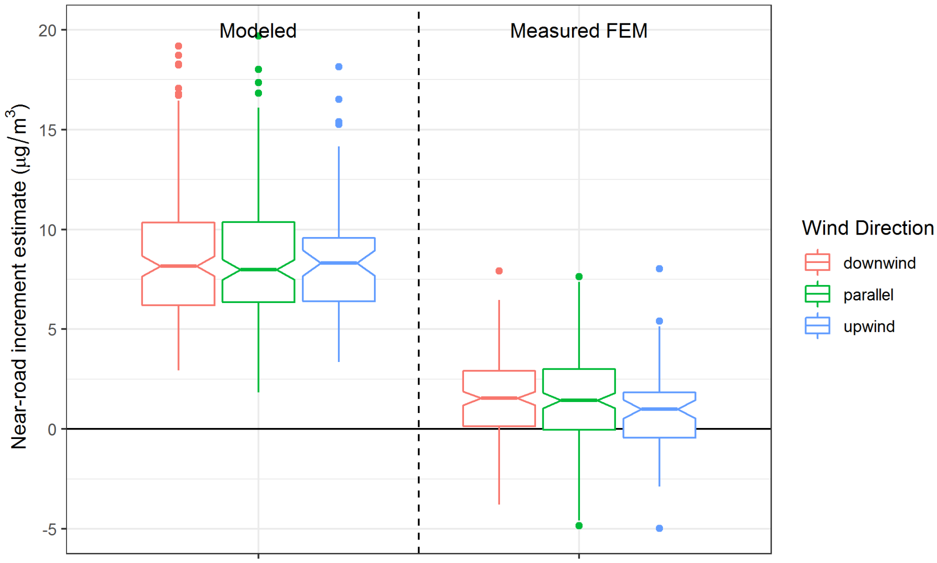


Figure ES-10. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured (ambient) near-road $PM_{2.5}$ increments at the Providence near-road monitoring site during 2015-2016 (382 days) for three wind conditions (near-road monitor downwind, upwind, and parallel to I-95).

Digest of Major Technical Findings

Highlights of some of the major findings from the five-year TPF program include the following:

1. Near-road concentrations of CO and NO_2 were not problematic when benchmarked against the existing (as of 2019) NAAQS.

2. Near-road $PM_{2.5}$ concentrations are likely trending downward; however, these findings are based on a limited number of sites that operated over the analysis years covered by this work. More data is becoming available each year to help establish multi-year trends across the entire network of near-road sites.

3. Relative to the total number of near-road sites measuring $PM_{2.5}$, a small number of locations exceed the 24-hr or annual $PM_{2.5}$ NAAQS.

4. Based on 2017 data, the upper bound of $PM_{2.5}$ increments, for 20 sites across the United States, was 2.04 µg/m^3. Only three sites had an increment greater than 1.44 µg/m^3; monitors at each of these three sites were sited less than 10 meters from the roadway.

5. Over time, an increasing number of $PM_{2.5}$ nonattainment areas is expected to achieve the NAAQS. Therefore, $PM_{2.5}$ hot-spot analyses will increasingly be completed in maintenance areas where a "buffer" exists between the background concentration and the NAAQS. The

work completed by the TPF provides a better understanding of the likely increment from proposed projects. The increment findings can contribute to interagency consultation and determinations as to whether projects are POAQC and should undergo quantitative hot-spot analysis.

6. There can be substantial differences between measured and modeled near-road $PM_{2.5}$ concentrations. Based on the case study work completed here, modeled concentrations over-predict measured values; this outcome was exaggerated the closer to the road the modeling was meant to represent. Further work is needed to assess differences between modeled and measured near-road concentrations.

Future Research Recommendations

Future work could further examine:

1. The relative contribution of exhaust and non-exhaust $PM_{2.5}$ emissions.
2. Modeled and measured near-road $PM_{2.5}$ concentrations across different geographic settings, roadway types, and configurations.
3. Modeled and measured near-road CO pollutant concentrations. Near-road CO concentrations are far below the NAAQS and not problematic; the purpose of this work would be to use CO to help further refine understanding of modeling chain biases contributing to differences between measured and modeled $PM_{2.5}$ concentrations.
4. Quantitative estimates of the effects of near-road barriers and roadway grade on near-road pollutant concentrations.

1. Introduction

Beginning in 2014 and extending through 2019, federal and state agencies participated in a Transportation Pooled Fund (TPF) to address near-road air quality issues. By its conclusion, the Near-Road Air Quality TPF research program included participants from eight government agencies including the U.S. Federal Highway Administration (FHWA) and the Arizona, California, Colorado, Ohio, Texas, Virginia, and Washington State Departments of Transportation (DOTs); researchers from Sonoma Technology, Inc. (STI) provided the TPF with technical support. The Washington State DOT (WSDOT) served as the TPF lead agency and managed the overall research program.

Key motivators for the formation of the Near-Road Air Quality TPF included recognition that motor vehicle-related air pollutant concentrations and health impacts can be elevated near heavily traveled roads; U.S. Environmental Protection Agency (EPA) requirements to quantitatively evaluate potential particulate matter (PM) hot-spots; EPA and FHWA requirements to assess mobile source air toxics (MSATs); and EPA requirements, implemented beginning 2014, to monitor air quality near heavily traveled roads (e.g., , 2010; Health Effects Institute, 2010; U.S. Environmental Protection Agency, 2010b, 2012; Federal Highway Administration, 2012).

During the five-year period the TPF was active, participants engaged in several jointly sponsored efforts. These included development of a strategic research plan; evaluation of near-road air quality data; emissions and air quality modeling to assess current and forecasted near-road conditions; completion of national-scale and case study evaluations; evaluation of potential mitigation; and dissemination of findings and resources via peer-reviewed articles, conference presentations, creation of near-road dataset resources to facilitate ongoing evaluations, and development and use of an information-exchange website.

This report presents a digest of the major findings from the TPF effort. The TPF was able to, for the first time, complete comprehensive assessments of U.S. near-road air quality based on data that became available from implementation of the EPA-mandated near-road monitoring network. By 2017, that network had grown to encompass approximately 70 U.S. metropolitan areas (**Figures 1 and 2**).

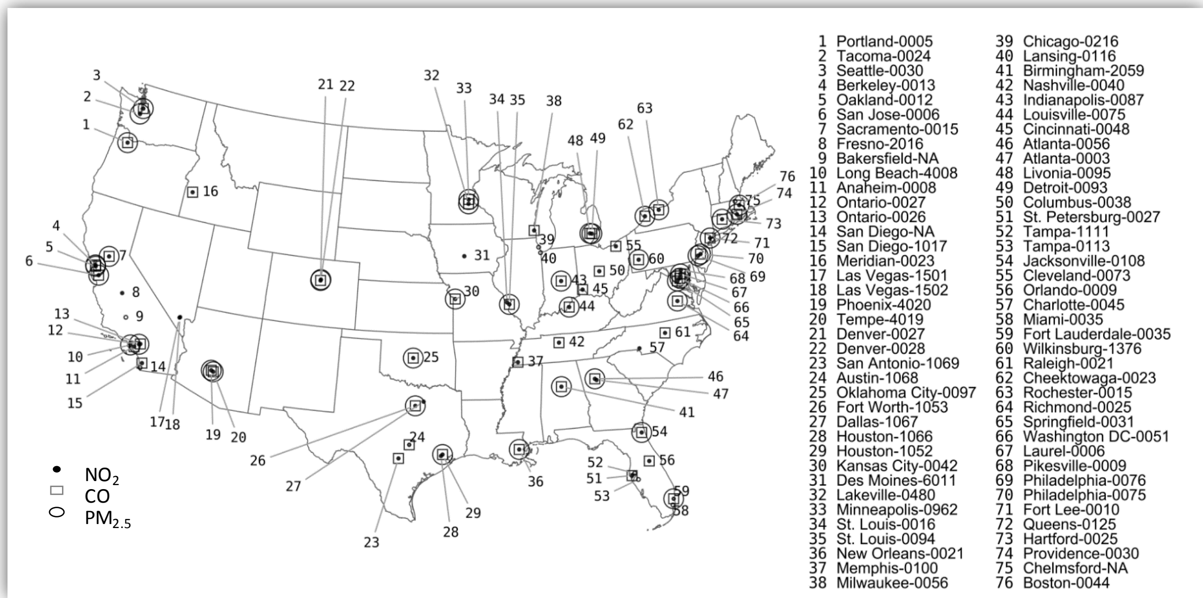


Figure 1. Near-road air quality monitoring site locations, 2017. The network included 68 NO_2, 53 CO, and 42 $PM_{2.5}$ monitoring locations. Sites are referred to by city name and AQS site ID code. The distance from road (in meters) is also indicated on map. Not shown is the site in Puerto Rico, Guaynabo 0006 (+18.42° latitude, -66.12° longitude); it monitored NO_2 and CO.

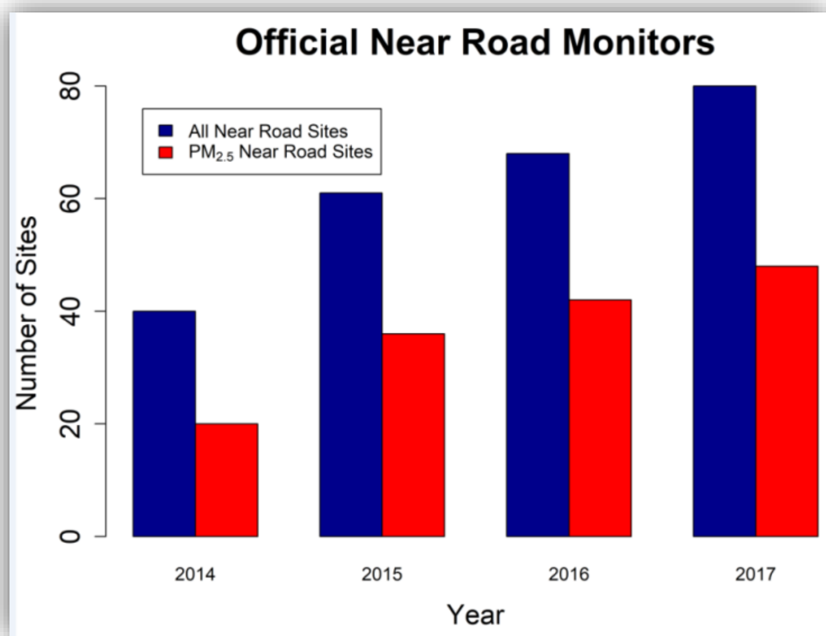


Figure 2. Number of U.S. near-road monitoring sites, 2014 to 2017. Data source: U.S. EPA.

In addition to evaluating data from the near-road monitoring network, the TPF completed computer modeling to investigate on-road emissions and their impact on near-road air quality. The studies completed involved use of multiple emissions and dispersion modeling tools, and involved hypothetical case analyses as well as simulations of actual site-specific episodes for which the study team had observed data, paired in time, for meteorological, traffic, and air quality parameters. These efforts produced new insights that will help transportation-air quality analysts better understand the opportunities and limitations of using modeled and measured data to characterize the near-road environment.

Finally, the TPF completed exploratory work to assess two potential ways to mitigate near-road air quality impacts. Research evaluated practical implementation experience reducing emissions through accelerated truck replacements, and pilot tested an EPA near-road modeling tool being developed to simulate how well near-road barriers may reduce exposure to air pollution in areas immediately downwind of major roads. These studies yielded important insights that will help practitioners assess and implement mitigation.

The remainder of this report is organized as follows:

Section 2 covers national-scale findings related to observed CO, NO_2, and $PM_{2.5}$ near-road concentration data.

Section 3 presents insights from case studies that examined high near-road $PM_{2.5}$ concentration events.

Section 4 examines the incremental pollutant contribution from major roads. The section focuses on $PM_{2.5}$ and assesses how near-road and background concentrations differ.

Section 5 develops screening insights related to POAQC determinations for $PM_{2.5}$ hot-spot analyses.

Section 6 compares monitored and modeled near-road $PM_{2.5}$ concentrations for two study sites, one in Indianapolis, Indiana, and another in Providence, Rhode Island.

Section 7 provides insights from exploratory work the TPF sponsored to assess potential mitigation options.

Section 8 highlights summary findings and future research needs.

Section 9 includes references.

2. Observed Near-Road Air Quality

2.1 Background

The EPA-mandated national near-road monitoring network became operative during the course of the TPF's work. The EPA promulgated near-road air quality monitoring requirements in 2010 at the same time that the agency revised the NAAQS[3] for NO_2 (U.S. Environmental Protection Agency, 2010a). Monitoring requirements were revised in 2013 to extend deadlines for initiating near-road NO_2 monitoring (U.S. Environmental Protection Agency, 2013b). Monitoring near major roadways in cities across the United States initially focused on NO_2, but additional CO and $PM_{2.5}$ requirements were added during NAAQS rulemakings (U.S. Environmental Protection Agency, 2013a). The EPA adopted a phased implementation plan, with the first set of monitoring sites to be operational by January 1, 2014; subsequent sites were added in later years. A small subset of the sites also measure air toxics, BC, and UFP; however, there are no requirements to monitor these compounds. As near-road data became available, it became apparent that CO and NO_2 concentrations were not exceeding the form of the NAAQS. In light of the data, EPA subsequently revised the near-road monitoring requirement for NO_2 by lifting the January 1, 2017, requirement for NO_2 monitoring in Core Based Statistical Areas (CBSAs) with populations between 500,000 and 1,000,000.

As illustrated by Figure 2, the number of near-road monitors deployed grew over the years, covering various locations and pollutants. TPF work involved obtaining and analyzing data from EPA for each calendar year beginning 2014. Data for a given calendar year were usually certified as final by EPA in May of the year following; for example, EPA certified calendar year 2015 data in May 2016. This discussion highlights major findings, by pollutant, using certified data for the calendar years presented. In general, this summary presentation highlights findings using the most recent data evaluated, since that data covers the greatest number of near-road sites. TPF analyses are available, however, for a broader range of calendar years.

For research purposes, this discussion includes comparisons of measured data to NAAQS levels; the NAAQS are shown in **Table 1**. These comparisons are provided for context and are not meant to assess attainment status; attainment and nonattainment areas are designated by the EPA.

[3] With its 2010 NO_2 NAAQS revision, the EPA augmented the existing annual standard of 53 ppb, calculated as the annual arithmetic mean, with a 1-hr NAAQS of 100 ppb. Compliance with the 1-hr NO_2 NAAQS is determined by calculating the 98th percentile of all the daily maximum 1-hr concentrations in a year, and then averaging three consecutive years of these 98th percentile values with the averaged value not to exceed 100 ppb.

Table 1. Primary NAAQS levels for CO, NO_2, and $PM_{2.5}$ (source: **epa.gov/criteria-air-pollutants**.)

Pollutant	Averaging Time	Level	Form
CO	8-hr	9 ppm	Not to be exceeded more than once per year
	1-hr	35 ppm	Not to be exceeded
NO_2	1-hr	100 ppb	98th percentile of 1-hr daily maximum concentrations, averaged over 3 years
	Annual	53 ppb	Annual mean
$PM_{2.5}$	24-hr	35 µg/m^3	98th percentile, averaged over 3 years
	Annual	12 µg/m^3	Annual mean, averaged over 3 years

Data summarized in this report show that near-road U.S. CO and NO_2 concentrations are not observed at levels above the form of the NAAQS. Near-road $PM_{2.5}$ concentrations, however, are in excess of the daily and annual NAAQS at some locations. Accordingly, the data highlights provided here include more detailed information regarding $PM_{2.5}$ compared to CO and NO_2, and later report sections examine more fully $PM_{2.5}$ background concentrations, roadway-related incremental concentration contributions, and modeled and measured near-road concentrations.

2.2 Carbon Monoxide (CO) Data Findings

2.2.1 1-Hour CO

The 1-hour CO concentrations measured at near-road monitoring sites in 2016 ranged from -0.40 ppm to 7.09 ppm, with a mean of 0.39 ppm (n = 392,087) and 75% of the values below 0.50 ppm. Given that the 1-hour NAAQS for CO is 35 ppm, these values are quite low.

In general, CO concentrations exhibit very small variation at each near-road monitoring site (**Figure 3**). The highest hourly concentrations were observed in Puerto Rico at Guaynabo-0006-1 (e.g., 7.09 ppm on October 3, 2016, at 08:00 LST; 4.40 ppm on March 31, 2016, at 08:00 LST; 4.2 ppm on March 4, 2016, at 08:00 LST)[4] and in Pennsylvania at Wilkinsburg-1376-1 (e.g., 4.75 ppm on September 14, 2016, at 07:00 LST; 4.54 ppm on July 21, 2016, at 00:00 LST; 4.51 ppm on September 14, 2016, at 06:00 LST). Whereas the largest outliers at Guaynabo-0006-1 tended to occur around peak morning traffic periods (approximately 06:00 LST to 09:00 LST), the highest 1-hour CO concentrations at Wilkinsburg-1376-1 occurred throughout the evening and early morning hours (not shown).

[4] The three highest CO concentrations that occurred at Guaynabo-0006-1 all occurred either before or after a zero/span check on the instrument, which could indicate erroneous data. However, since the data submitted to EPA's AQS were to be quality-assured by the submitting agency, we did not remove these data from the analysis.

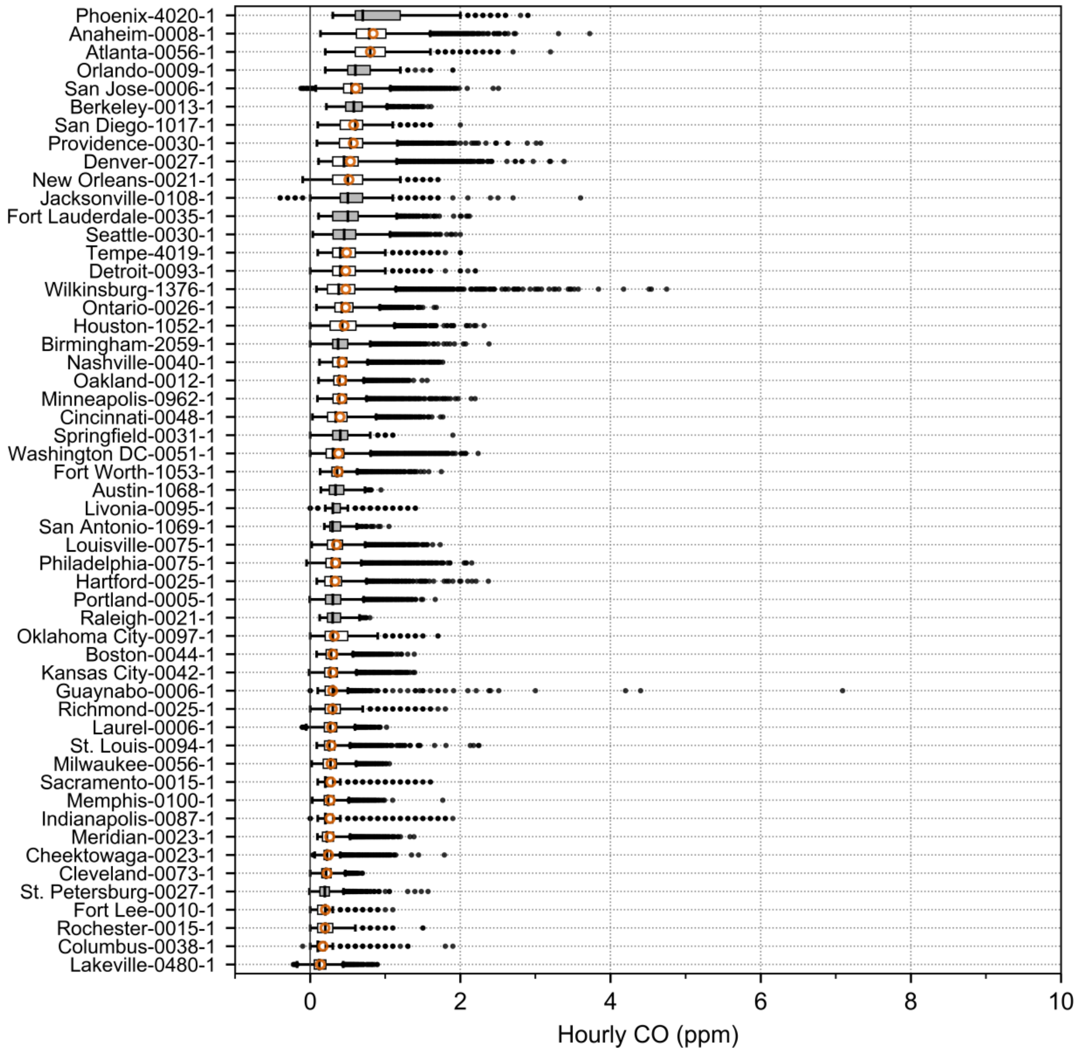


Figure 3. Distribution of 1-hour CO concentrations at near-road monitoring sites in 2016. Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) is displayed for monitors with complete annual datasets. The 1-hour NAAQS for CO is 35 ppm (not shown).

2.2.2 8-Hour CO

The 8-hour CO concentrations measured at near-road monitoring sites in 2016 ranged from -0.40 ppm to 3.51 ppm. The overall mean was 0.39 ppm (n = 387,310), where most of the values (based on the 75th percentile) were below 0.50 ppm. Similar to distributions of 1-hour CO concentrations, the distributions of 8-hour CO concentrations at near-road monitoring sites in 2016 were quite small, and all 8-hour CO concentrations were quite low (Figure 4). There were no exceedances of the 8-hour NAAQS threshold of 9 ppm.

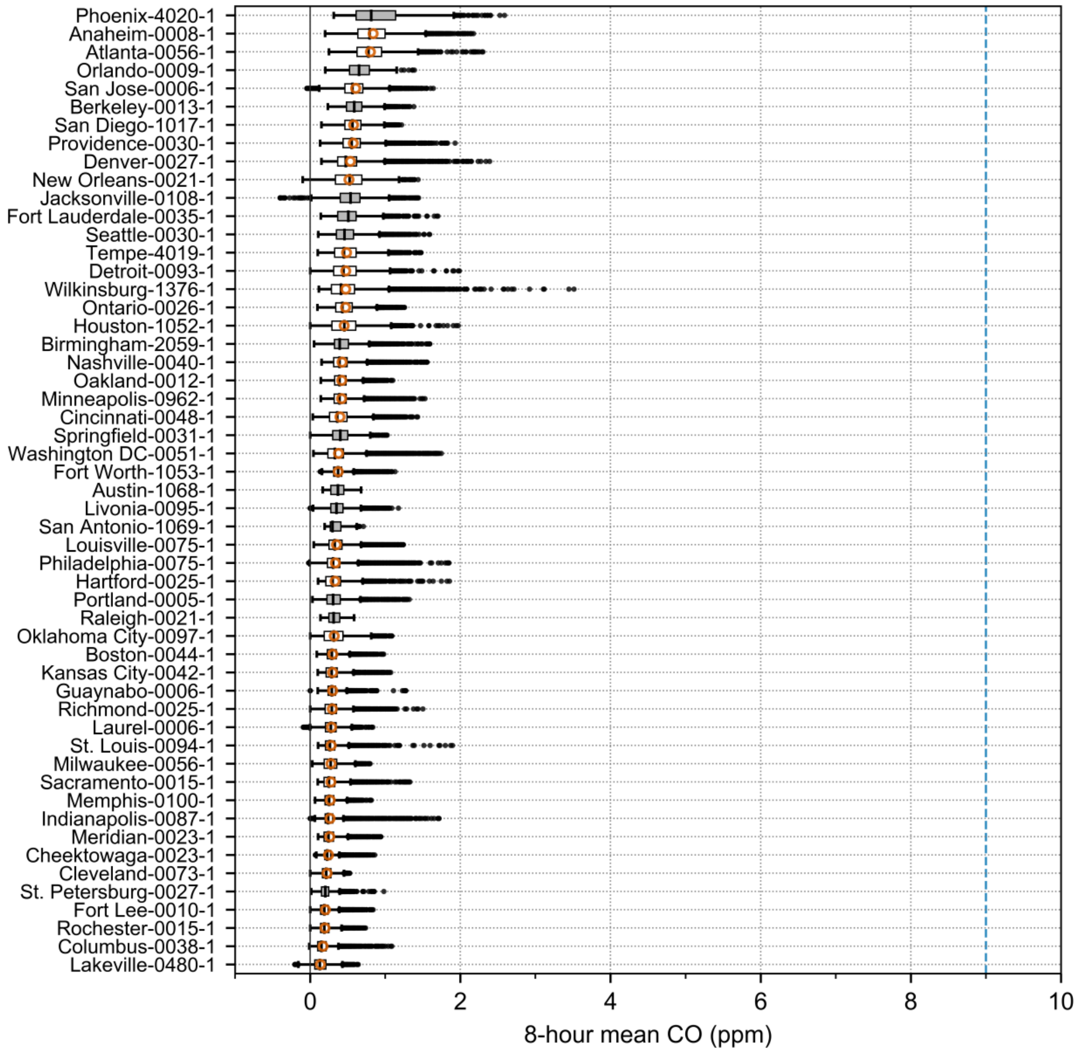


Figure 4. Distribution of 8-hour CO concentrations at near-road monitoring sites in 2016. Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) is displayed for monitors with complete annual datasets. The blue dashed lined denotes the 8-hour NAAQS threshold (9 ppm).

The 2016 CO data findings shown in Figures 3 and 4 are similar to findings from TPF evaluations of the 2014 and 2015 data. Figure 5 illustrates CO data findings from 2015.

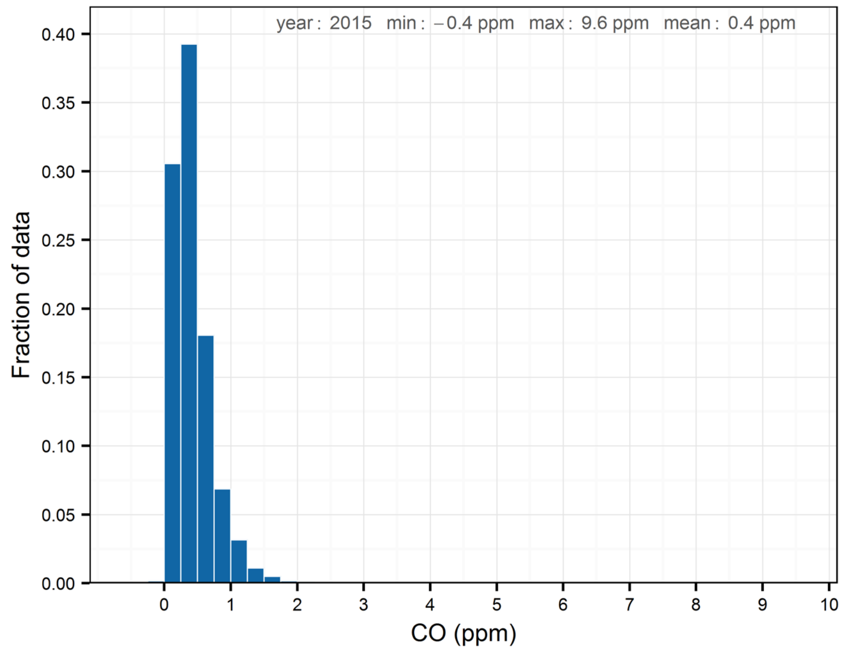


Figure 5. Distribution of all hourly CO concentrations at near-road monitors in 2015.

2.3 Nitrogen Dioxide (NO_2) Data Findings

2.3.1 1-Hour NO_2

Figure 6 summarizes the distributions of 1-hour NO_2 in 2016. Only four 1-hour NO_2 values exceeded 100 ppb across all sites: at Birmingham-2059-1 on October 25, 2016, at 09:00 LST (114.55 ppb), and at Guaynabo-0006-1 on July 6, 2016, at 12:00 LST (155.21 ppb), 14:00 LST (194.86 ppb), and 15:00 LST (133.98 ppb). The high hourly NO_2 concentrations at the Guaynabo-0006 near-road monitoring site may be due to the monitor's proximity to De Diego Highway (José de Diego Expressway) and Buchanan Toll Plaza, where vehicles slow down (resulting in a higher per-mile pollutant emission rate) and traffic may be congested.

Based on data from 2016 only, there were no exceedances[5] of the 1-hour or annual NO_2 NAAQS thresholds; that is, the annual mean NO_2 concentrations were below 53 ppb, and the 98th percentile of maximum 1-hour concentrations was below 100 ppb at all near-road sites in 2016 with complete datasets. The annual mean ranged from 7.6 ppb (at Des Moines-6011-1) to 31.0 ppb (at Ontario-0027-1), whereas the 98th percentile of maximum 1-hour concentrations range from 24.8 ppb (at Guaynabo-0006-1) to 78.2 ppb (at Long Beach-4008-1). Although the highest 1-hour

[5] In this report, "exceedance" refers to whether or not a value was greater than or equal to the NAAQS threshold, while a "violation" would be if a site exceeded the 3-year NAAQS calculation. These findings are for research purposes and should not be used for determining attainment status.

NO_2 concentrations occurred at Guaynabo-0006-1 in 2016, the annual mean and 98th percentile of daily 1-hour maximum concentrations of NO_2 at this monitor were among the lowest of all near-road sites. This indicates that the high 1-hour NO_2 outliers are a rare occurrence.

2.3.2 3-Year Statistics

The TPF used available data for 2014-2016 to assess multi-year trends for sites reporting three years of data; however, given the small number of sites and the limited number of years evaluated, no statistically significant trends in mean annual statistics were firmly established. For sites with three complete years of data, 3-year annual statistics following NAAQS guidelines can be examined to qualitatively assess whether there have been common changes in air quality at near-road monitoring sites over the three-year period. Generally, the mean annual mean 98th percentile of daily 1-hour maximum NO_2 concentrations across all near-road monitoring sites decreased from 2014 to 2016, but the maximum and range of values increased as more sites came online in 2015 and 2016 (**Figure 7, left**). Based on data from individual sites, the increased range can be likely attributed to the increased number of sites with complete data in 2015-2016, rather than generally worsening air quality conditions. There were 14 near-road monitoring sites with complete annual datasets of 1-hour NO_2 from 2014 to 2016. The annual mean 98th percentile of daily 1-hour maximum NO_2 concentrations generally decreased over this three-year period at most of these sites (**Figure 7, right**). Finally, the NAAQS three-year average of the 98th percentile of daily 1-hour maximum concentrations were calculated (**Table 2**). The values ranged from 32.8 ppb at Des Moines-6011-1 to 66.7 ppb at Denver-0027-1. All of the three-year averages were well below the NAAQS threshold of 100 ppb.

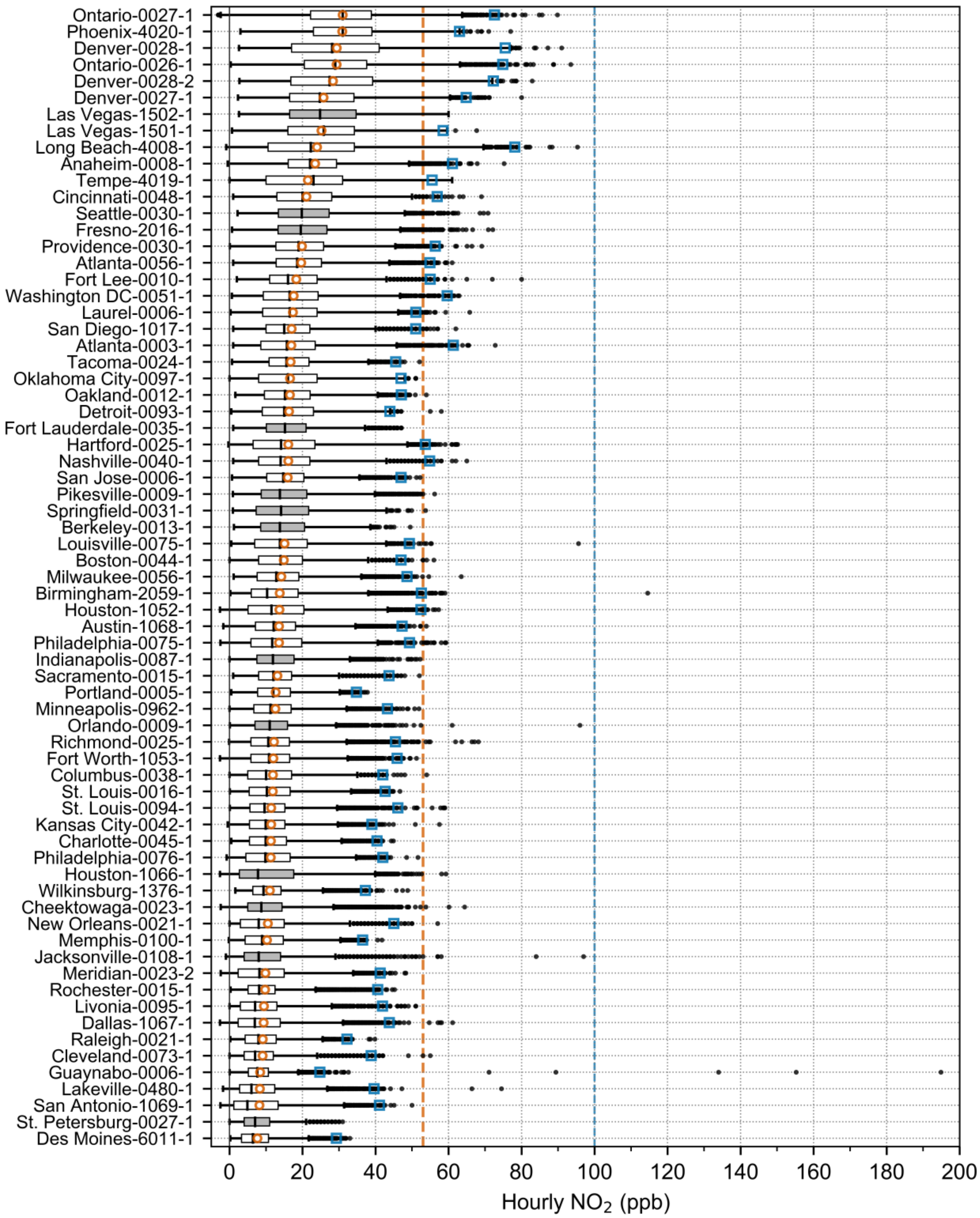


Figure 6. Distribution of 1-hour NO_2 concentrations at near-road monitoring sites in 2016.[6] Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) and 98th percentile of 1-hour daily maximum concentrations (blue square) are displayed for monitors with complete annual datasets. The orange dashed lined denotes the annual NAAQS threshold (53 ppb), and the blue dashed line the 1-hr NAAQS threshold (100 ppb).

[6] Box extents indicate the 25th and 75th percentile of data. The horizontal line within the box indicates the 50th percentile (median). The whiskers have a maximum length equal to at most 1.5 times the length of the box (the interquartile range, IQR). Individual data points beyond the whiskers are plotted as individual points.

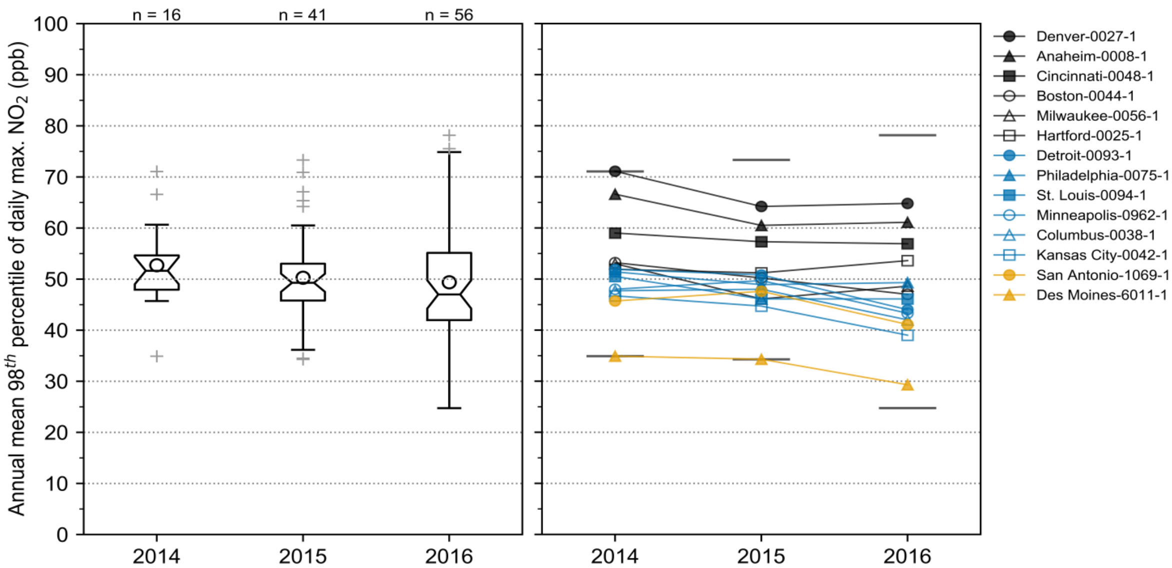


Figure 7. Annual mean 98th percentile of daily 1-hour maximum NO_2 concentration at near-road monitoring sites with complete data. Left: Distribution of values by year, where box extents indicate the 25th and 75th percentile.[7] The number of points (n sites) in each distribution is shown at the top. Right: Time series for near-road monitoring sites with complete NO_2 datasets in all years. Horizontal bars indicate the minimum and maximum values within each year.

Table 2. Mean 98th percentile of daily 1-hour maximum NO_2 concentrations averaged over three years (2014-2016) ("*p98*") at near-road monitoring sites with complete data in all three years.

AQS ID	Monitor	*p98* (ppb)
08-031-0027	Denver-0027-1	66.7
06-059-0008	Anaheim-0008-1	62.7
39-061-0048	Cincinnati-0048-1	57.7
09-003-0025	Hartford-0025-1	52.2
25-025-0044	Boston-0044-1	50.1
42-101-0075	Philadelphia-0075-1	49.9
55-079-0056	Milwaukee-0056-1	49.2
26-163-0093	Detroit-0093-1	48.9
29-510-0094	St. Louis-0094-1	47.6
27-053-0962	Minneapolis-0962-1	47.0
39-049-0038	Columbus-0038-1	45.9
48-029-1069	San Antonio-1069-1	44.8
29-095-0042	Kansas City-0042-1	43.5
19-153-6011	Des Moines-6011-1	32.8

[7] The horizontal line within the box indicates the 50th percentile (median), and the open circle indicates the mean. The whiskers have a maximum length equal to at most 1.5 times the length of the box (the interquartile range, IQR). Individual data points beyond the whiskers are plotted as individual points (crosshairs). Box notches indicate the 95% confidence interval of the median.

2.4 Particulate Matter 2.5 Microns or Less in Diameter ($PM_{2.5}$) Data Findings

2.4.1 24-Hour $PM_{2.5}$

In 2016, $PM_{2.5}$ was monitored at 42 sites, six of which had collocated monitors, resulting in 48 total $PM_{2.5}$ monitors. There were 29 monitors that measured $PM_{2.5}$ on an hourly basis, while the remaining 19 monitors sampled $PM_{2.5}$ locations on a 24-hour basis at various sampling frequencies (daily, 1-in-3, or 1-in-6). In this analysis, we aggregated all 1-hour $PM_{2.5}$ measurements to 24-hour mean $PM_{2.5}$ if the 75% completeness criterion for each day was met. Twenty-eight of the monitors had complete data for 2016.

Figure 8 presents 24-hour findings. There were 19 instances where 24-hour $PM_{2.5}$ concentrations exceeded 35 µg/m^3. Several of these instances occurred at Ontario-0027-1 (7 days), and at Indianapolis-0087-3 (3 days). Three of these high 24-hour $PM_{2.5}$ concentrations occurred on either January 1 (Phoenix-4020-3, Tempe-4019-3) or July 4 (Detroit-0093-3), which are days typically associated with increased $PM_{2.5}$ due to local firework display events.

At two sites, the annual mean 24-hour $PM_{2.5}$ concentration exceeded 12 µg/m^3: at Ontario-0027-1 (14.9 µg/m^3), and at Long Beach-4008-1 (12.0 µg/m^3). The annual mean $PM_{2.5}$ concentration at Detroit-0093-3, Sacramento-0015-1, and Phoenix-4020-3 also exceeded 12 µg/m^3; however, the datasets from these sites were not complete (there was only one quarter of data available from the latter two sites). The only site where the 98th percentile of 24-hour $PM_{2.5}$ concentrations exceeded 35 µg/m^3 in 2016 was at Ontario-0027-1.

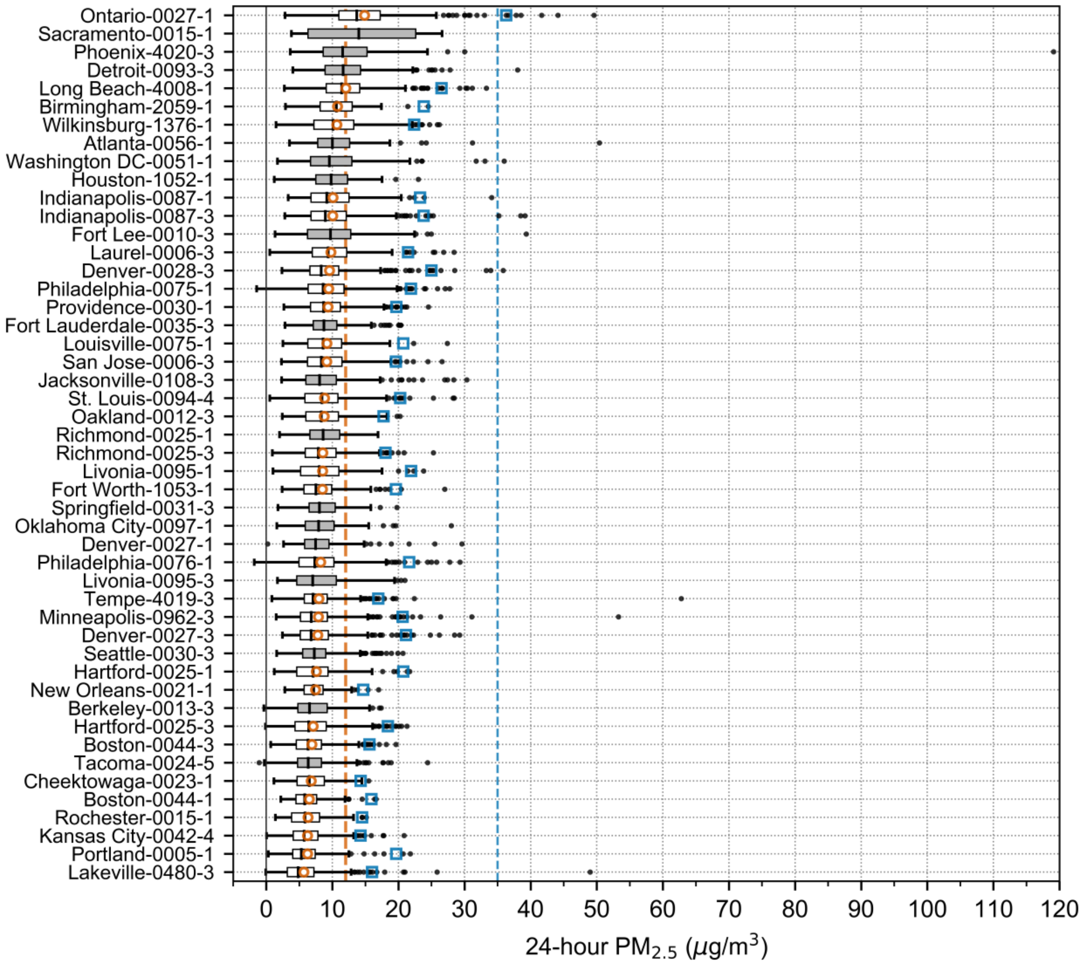


Figure 8. Distribution of 24-hour $PM_{2.5}$ concentrations at near-road monitoring sites in 2016. Monitors with datasets that did not meet the 75% completeness criteria are shaded in gray. The annual mean (orange circle) and 98th percentile of 24-hour $PM_{2.5}$ concentrations (blue square) are displayed for monitors with complete annual datasets. The orange dashed lined denotes the annual NAAQS threshold (12 µg/m^3), and the blue dashed line denotes the 24-hour NAAQS threshold (35 µg/m^3).

2.4.2 3-Year Statistics

Generally, the annual mean 24-hour $PM_{2.5}$ (**Figure 9**) and mean 98th percentile of 24-hour $PM_{2.5}$ concentrations (**Figure 10**) decreased from 2014 to 2016. As seen with the three-year statistics of NO_2 concentrations, the maximum values and the range of values increased while the means of these statistics decreased. Again, this is likely due to the increased number of sites with complete $PM_{2.5}$ datasets in more recent analysis years.

There were only eight near-road monitoring sites with complete annual datasets of 24-hour $PM_{2.5}$ from 2014 to 2016. Both the annual mean and 98th percentile of 24-hour $PM_{2.5}$ concentrations generally decreased over this three-year period at most of these sites. However, since data were not complete at all near-road monitoring sites, this trend may not be representative of annual $PM_{2.5}$ statistics at all near-road monitoring sites.

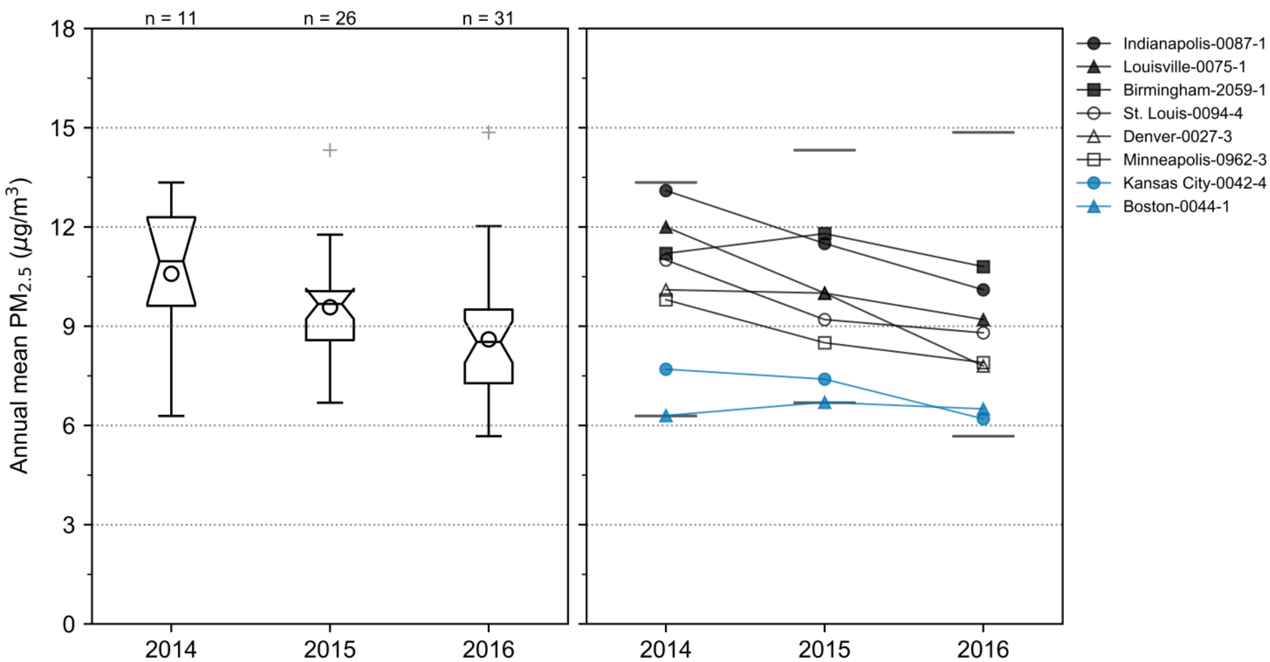


Figure 9. Annual mean 24-hour $PM_{2.5}$ concentrations at near-road monitoring sites with complete data. Left: Distribution of values by year; the number of points (n sites) in each distribution is shown at the top. Right: Time series plot; horizontal bars indicate the minimum and maximum values within each year.

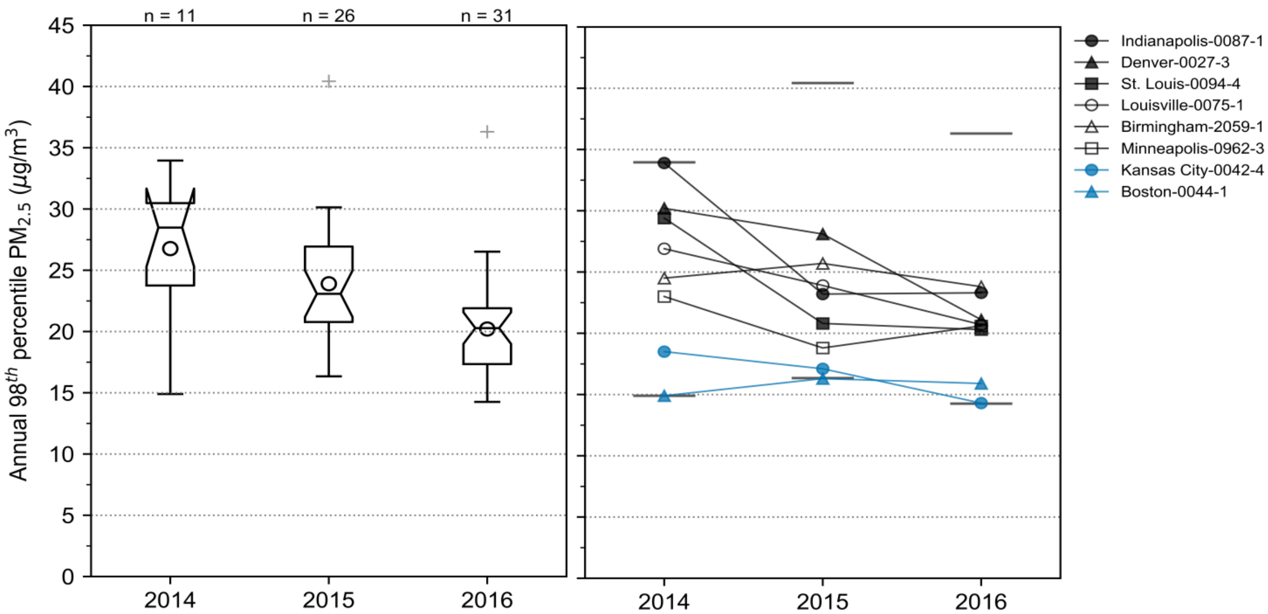


Figure 10. Annual 98th percentile 24-hour $PM_{2.5}$ concentrations at near-road monitoring sites with complete data. Left: Distribution of values by year; the number of points (n sites) in each distribution is shown at the top. Right: Time series plot; horizontal bars indicate the minimum and maximum values within each year.

The three-year averages of mean 24-hour $PM_{2.5}$ and 98th percentile of 24-hour $PM_{2.5}$ concentrations were calculated following NAAQS guidelines for 2014-2016. The highest value of both of these statistics occurred at the Indianapolis-0087-1 near-road monitoring site; the mean 24-hour $PM_{2.5}$ averaged over this period (11.6 $\mu g/m^3$) is close to but does not exceed the NAAQS standard of 12 $\mu g/m^3$.

3. Case Study Insights: High Near-Road $PM_{2.5}$ Concentration Events

3.1 Overview

In this section, we examine two cases—Denver, Colorado, and Indianapolis, Indiana—where some of the highest near-road $PM_{2.5}$ concentrations were observed. These in-depth case studies complement the national-scale data reviews in the prior two sections. The cases present a day-by-day look at $PM_{2.5}$ concentrations at two near-road sites in comparison to nearby sites, traffic, and meteorological conditions. The cases represent the highest measured U.S. near-road 24-hour $PM_{2.5}$ concentrations in 2014 (Denver), and the highest measured U.S. near-road 3-yr annual average $PM_{2.5}$ concentrations for 2014-2016 (Indianapolis). Figure 11 shows the case study sites.

Figure 11. Satellite (top) and ground-level (bottom) views via Google Earth of the near-road sites at Indianapolis (left) and Denver (right); the Denver monitoring site is somewhat obscured by cars in the image, so the blue box denotes its location.

The case study analyses examine the conditions under which $PM_{2.5}$ concentrations are higher at the near-road site, i.e., how changes in traffic or meteorological patterns impact $PM_{2.5}$ concentrations next to the roadway.

3.2 Denver

The near-road site in Denver is 9 m from the east side of I-25, with an annual average daily traffic (AADT) count of 249,000 and a fleet-equivalent AADT (FE-AADT, a measure that weights trucks more heavily for count purposes) of 263,118. Between February 2 and 12, 2014, four periods of relatively high $PM_{2.5}$ concentrations were measured at the Denver near-road site. Hourly concentrations were high overnight on February 3-4, during the morning and afternoon of February 4, at midday on February 7, and throughout February 9 and 10. The 24-hr $PM_{2.5}$ concentrations exceeded 35 µg/m^3 on three days: February 7 (35.4 µg/m^3), February 9 (44.4 µg/m^3), and February 10 (57.0 µg/m^3). The February 10 $PM_{2.5}$ concentration was the highest 24-hr $PM_{2.5}$ value measured in 2014 across all of the near-road monitoring sites in the U.S.

Figure 12 shows traffic volume and speed near the near-road site, hourly and 24-hr $PM_{2.5}$, hourly $NO/NO_2/NO_x$, wind speed, and whether the near-road monitor was upwind or downwind of the freeway. Hourly $PM_{2.5}$ concentrations do not exhibit a consistent diurnal cycle during this period (February 2−12, 2014). In some cases (e.g., February 2 and 7), hourly $PM_{2.5}$ concentrations peaked in the middle of the day, while in other cases (e.g., February 3), $PM_{2.5}$ concentrations were highest during the overnight hours. Between February 9 and 11, $PM_{2.5}$ concentrations were consistently elevated for multiple consecutive days.

In general, $PM_{2.5}$ concentrations were higher when wind speeds were lower. Although wind speeds vary from day to day, they are generally higher during midday and afternoon. When hourly $PM_{2.5}$ concentrations were elevated, wind speeds were typically low, and winds were often from the north (i.e., roughly parallel to the freeway), varying between upwind and downwind conditions.

In some cases (e.g., February 7), high hourly $PM_{2.5}$ concentrations coincided with high hourly NO_x concentrations; however, during other periods of high $PM_{2.5}$ concentrations (e.g., February 4), NO_x concentrations were not elevated. NO_2 concentrations were consistently low at the near-road site between February 2 and 12. There is no consistent relationship between $NO/NO_2/NO_x$ or temperature and high $PM_{2.5}$; when the temperature was below 20°F on February 5-6, $PM_{2.5}$ was relatively low.

Lastly, traffic volumes exhibit a typical diurnal activity pattern, with morning and afternoon peaks consistent with the morning and evening commute times. Traffic speeds are somewhat variable and do not track the diurnal signature of traffic volumes. Neither NO_x nor $PM_{2.5}$ concentrations were correlated with vehicle speeds or traffic volumes during February 2−12, 2014.

The Denver site has additional characteristics that likely influence the measured near-road concentrations. Two factors in particular are worth noting: the roadway alignment and the presence of buildings next to the monitor. The analysis here characterizes wind conditions that result in the near-road monitor being upwind or downwind of the road; these conditions are defined by the roadway alignment adjacent to the monitor. However, Figure 11 illustrates that I-25 does not continue in a straight line north and south of the monitor; the curvature of the road to the north and south results in a weak "C" shape visible in satellite imagery. When winds are roughly parallel to the road, sections of the road to the north and south may effectively be upwind of the monitor, albeit only slightly. In addition, the monitor is sited between I-25 and the rear side of buildings that create a nearly continuous wall facing I-25. These buildings may act as a solid barrier that prevents pollutant dispersion, leading to higher near-road pollutant concentrations than if the buildings were not present.

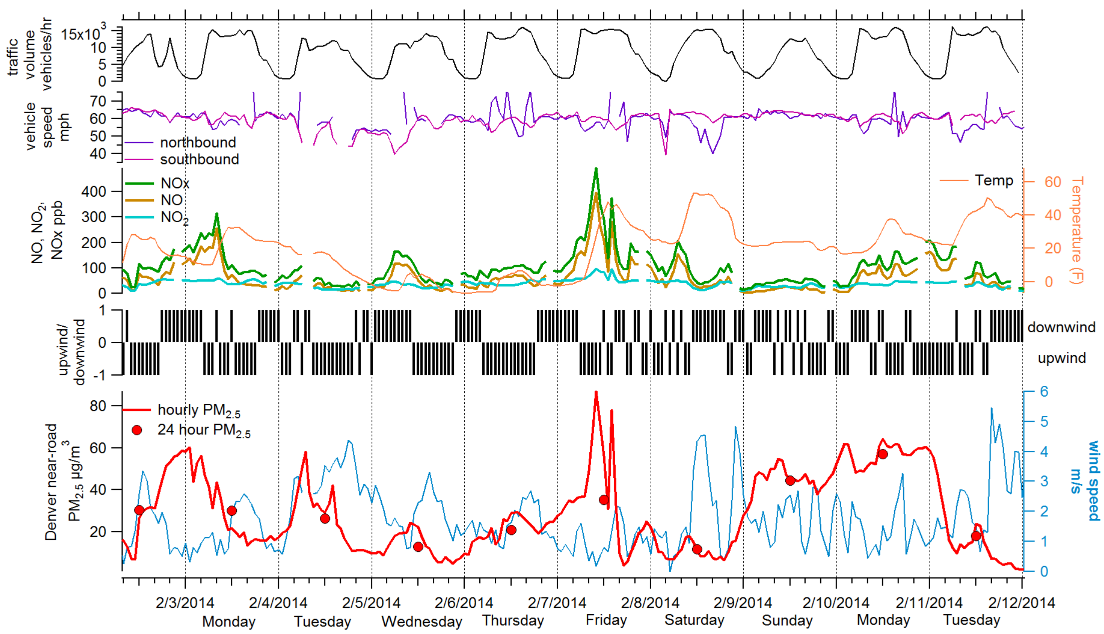


Figure 12. Denver case characterization. Hourly and 24-hr $PM_{2.5}$ concentrations (µg/m^3, red line), 24-hr $PM_{2.5}$ concentrations (µg/m^3, red dots), wind speed (m/s, blue line), whether the monitoring site was upwind (0 to 180 degrees) or downwind (180 to 360 degrees) of the freeway (black bars), NO_x concentrations (ppb, green line), NO concentrations (ppb, gold line), NO_2 concentrations (ppb, teal line), and temperature (°F, orange line) at the near-road site in Denver during February 2–12, 2014. Also shown are vehicle speeds on I-25 (northbound, purple line; southbound, dark pink line) and traffic volume on I-25 (black line). Traffic data are for I-25 at 6th Ave., approximately 700 m south of the monitoring site.

Next, we examined $PM_{2.5}$ data from nearby sites, to assess whether all sites varied together, indicating an urban-scale $PM_{2.5}$ signature. Hourly $PM_{2.5}$ concentrations at the near-road site were closely correlated with concentrations at nearby sites, rather than with whether the near-road site was upwind or downwind of the freeway. $PM_{2.5}$ concentrations were typically higher at the near-road site than at the regional sites. On all but one day of February 2014, 24-hr concentrations at the near-road site were the highest of any of the sites in Denver, regardless of wind speed or other factors. PM concentrations were also consistently higher at the near-road site than at other nearby sites throughout the year: on 79% of the 365 days in 2014, the near-road site measured the highest concentration of sites in Denver; on 10 out of 365 days (3% of days), the near-road site concentration was lower than the other sites' concentrations by at least 5 µg/m^3. Figure 13 shows a pollution rose of $PM_{2.5}$ in February 2014 at the near-road site, as well as a time series of $PM_{2.5}$ concentrations at the near-road site and at three nearby regional $PM_{2.5}$ monitoring sites.

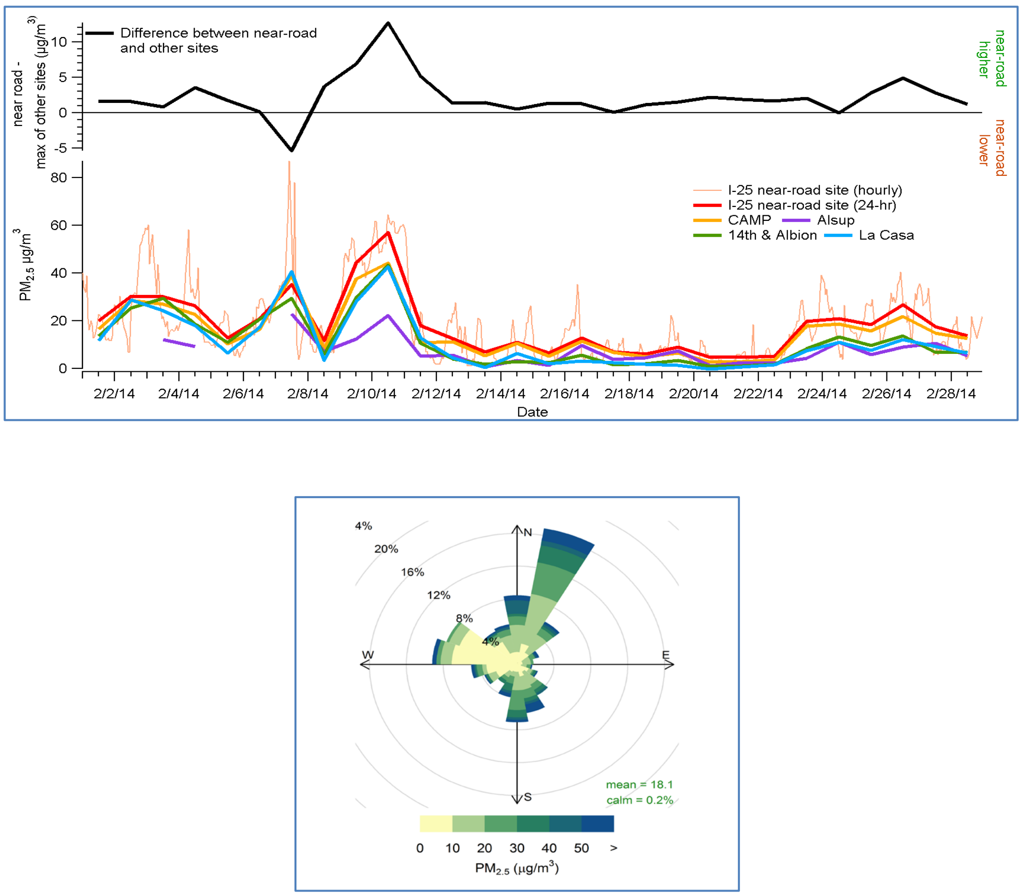


Figure 13. Denver near-road $PM_{2.5}$ increment and pollution rose. (a) Hourly (thin orange line) and 24-hr $PM_{2.5}$ concentrations (µg/m^3, thick colored lines) at the near-road and nearby sites during February 2014, and the difference between 24-hr $PM_{2.5}$ concentrations (µg/m^3) at the near-road site and the maximum 24-hr $PM_{2.5}$ concentration (µg/m^3) at the nearby sites (black line). (b) Pollution rose for hourly $PM_{2.5}$ concentrations (µg/m^3) at the Denver near-road site in February 2014. The size of each wedge indicates the frequency of wind direction. (For example, winds were out of the northeast nearly 25% of the time.) Color bands indicate the relative fraction of the time that concentrations occurred for each wind direction. For example, when winds were from the northeast, concentrations of 10-20 µg/m^3 were most frequent, followed by concentrations of 20-30 µg/m^3.

In summary, the Denver near-road site consistently had the highest $PM_{2.5}$ in the Denver area. During February 2014 and year-round in 2014, $PM_{2.5}$ concentrations at the near-road site were typically highest in the region regardless of meteorology and season. The high near-road concentration events during February 2014 do not appear to be caused by unusual traffic conditions next to the monitoring site; the high concentrations were largely driven by regional $PM_{2.5}$ conditions. Typically, including during the February 2014 $PM_{2.5}$ events, the near-road site had higher $PM_{2.5}$ concentrations than the other urban sites, indicating that emissions from the roadway contributed to the $PM_{2.5}$ at the near-road site, absent the presence of other sources, which we did not detect.

3.3 Indianapolis

The Indianapolis near-road site is 24.5 m south of the I-70 freeway, with an AADT of 189,760 and an FE-AADT of 362,110. The Indianapolis near-road site had the highest 3-year annual average (11.6 µg/m^3) of any U.S. near-road site where three years of complete data through 2016 were available. Between November 3 and 22, 2016, several periods of relatively high $PM_{2.5}$ concentrations were measured at the near-road site. $PM_{2.5}$ was above 35 µg/m^3 on three days, which are the focus of this case study: November 5 (38.5 µg/m^3), November 6 (39.1 µg/m^3), and November 13 (35.2 µg/m^3). These days were all on weekends. The highest $PM_{2.5}$ concentrations occurred during the evening and early morning hours (approximately 18:00 LST through 08:00 LST), when the wind directions were from the east and southeast, though winds from these directions were relatively infrequent. The orientation of I-70 with respect to the Indianapolis near-road site means that when winds are from the northwest, the site is downwind of the freeway. When winds were from the north/northwest, i.e., from the direction of I-70, $PM_{2.5}$ concentrations were typically lower than when winds were from other directions.

High hourly concentrations consistently occurred in the late evening, overnight, and in the early morning. There was not a constant diurnal cycle of 1-hour $PM_{2.5}$ concentrations during November 3–18, 2016. On most days, $PM_{2.5}$ concentrations were highest during the night. In general, $PM_{2.5}$ was higher when wind speeds were slower. Although wind speeds varied from day to day, they were generally faster during midday and afternoon periods. When hourly $PM_{2.5}$ concentrations were elevated, wind speeds were typically slow and the near-road monitor was more often upwind of I-70; this is true for November 5, 6, and 13, when 24-hour $PM_{2.5}$ concentrations exceeded 35 µg/m^3. **Figure 14** represents site conditions during these episodes.

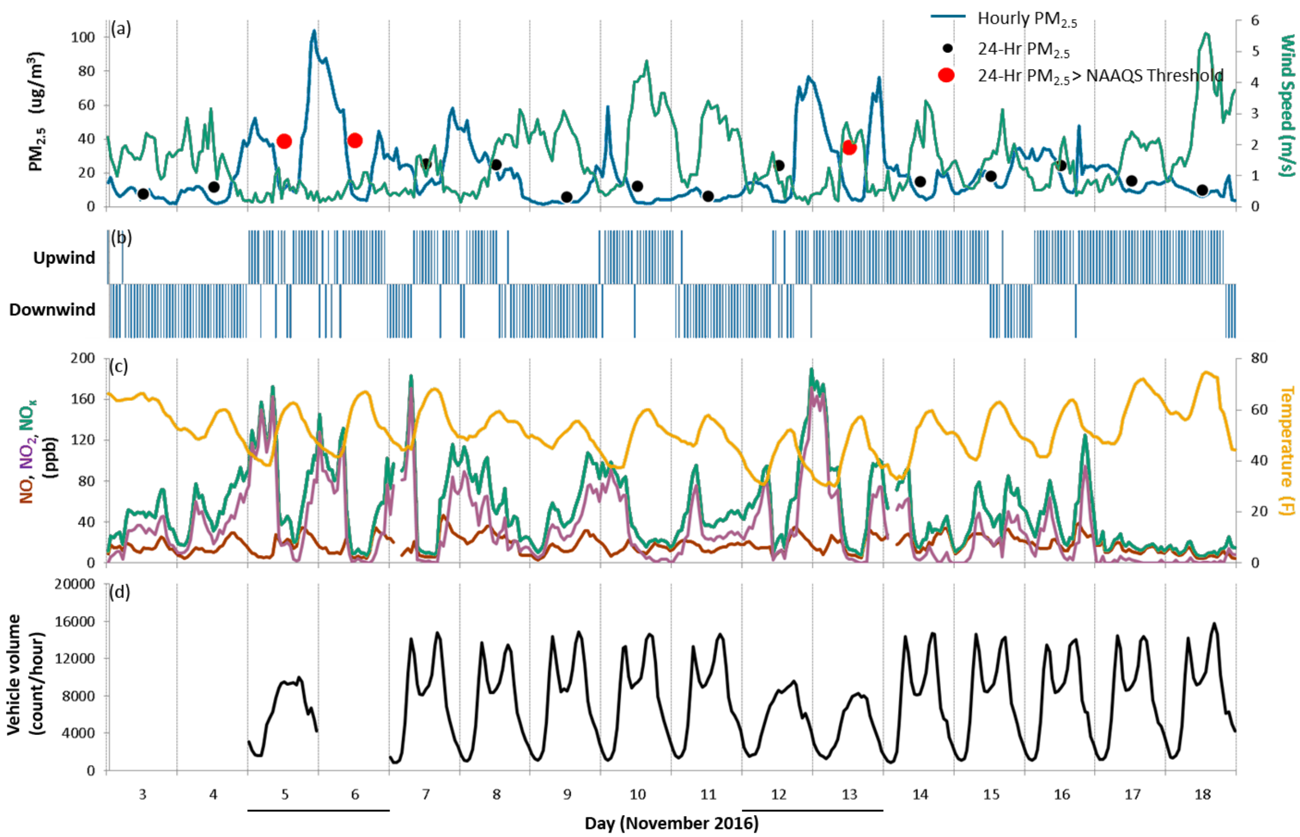


Figure 14. Indianapolis case characterization. Time series of meteorological, air quality, and traffic-related data in November 3–18, 2016, at the Indianapolis-0087 near-road monitoring site: (a) 1-hour $PM_{2.5}$ concentrations (blue line), 24-hr $PM_{2.5}$ concentrations (black and red dots, where red dots highlight 24-hour $PM_{2.5}$ concentrations that exceeded the 35 µg/m^3 NAAQS threshold), and wind speed (green line); (b) wind component (downwind means the near-road monitor was downwind of I-70, with winds originating from 240° to 60°); (c) NO (red line), NO_2 (purple line), and NO_x (green line) concentrations, and temperature (orange line); and (d) traffic volume (vehicle count per hour). Weekend days are underlined along the x-axis.

One indicator to help determine whether high levels of $PM_{2.5}$ are associated with emissions from mobile sources is to examine whether NO_2 concentrations are also elevated. During the period of November 3–18, 1-hour $PM_{2.5}$ concentrations coincided with high hourly NO_x and NO_2 concentrations (e.g., overnight peak on November 5) at some times, but not at others (e.g., overnight peak November 13). Traffic volumes exhibited a typical diurnal weekday activity pattern, with morning and afternoon peaks consistent with the morning and evening commute times. Traffic volumes on weekend days peaked near midday. NO_x concentrations and $PM_{2.5}$ concentrations were poorly correlated with traffic volumes during these periods. Days when $PM_{2.5}$ exceeded 35 µg/m^3 (November 5, 6, and 13) were all weekend days. While traffic measured on November 5 (Saturday) was higher than the mean traffic volume from October–December, $PM_{2.5}$ concentrations for November 5, 6, and 13 did not correlate well with traffic volumes. In addition, the near-road monitoring site was predominantly upwind of I-70 on these days.

Given the ambient air quality data and traffic volumes, the elevated period of $PM_{2.5}$ concentrations at the Indianapolis-0087 near-road monitoring site during November 2016 was likely not caused by on-road mobile emission sources. Rather, the diurnal pattern of 1-hour $PM_{2.5}$ may be indicative of the influence of residential wood burning (which generally causes elevated $PM_{2.5}$ concentration at night when temperatures are low), overnight temperature inversions in the lower boundary layer (which exacerbate air quality conditions overnight and in the early morning, especially when wind speeds are slow), or both.

We examined the temporal relationship of $PM_{2.5}$ concentrations from the near-road site and nearby sites (i.e., air quality monitoring sites within the same CBSA; n = 4 sites) during November 3–18. Homogeneous $PM_{2.5}$ concentrations would indicate a regional-scale influence on air quality, rather than impacts from local emission sources at the near-road site. During November 3–18, 24-hour $PM_{2.5}$ concentrations at the near-road site were closely correlated with those at nearby sites (**Figure 15**). $PM_{2.5}$ concentrations were typically higher at the near-road site than at the regional sites during this period.

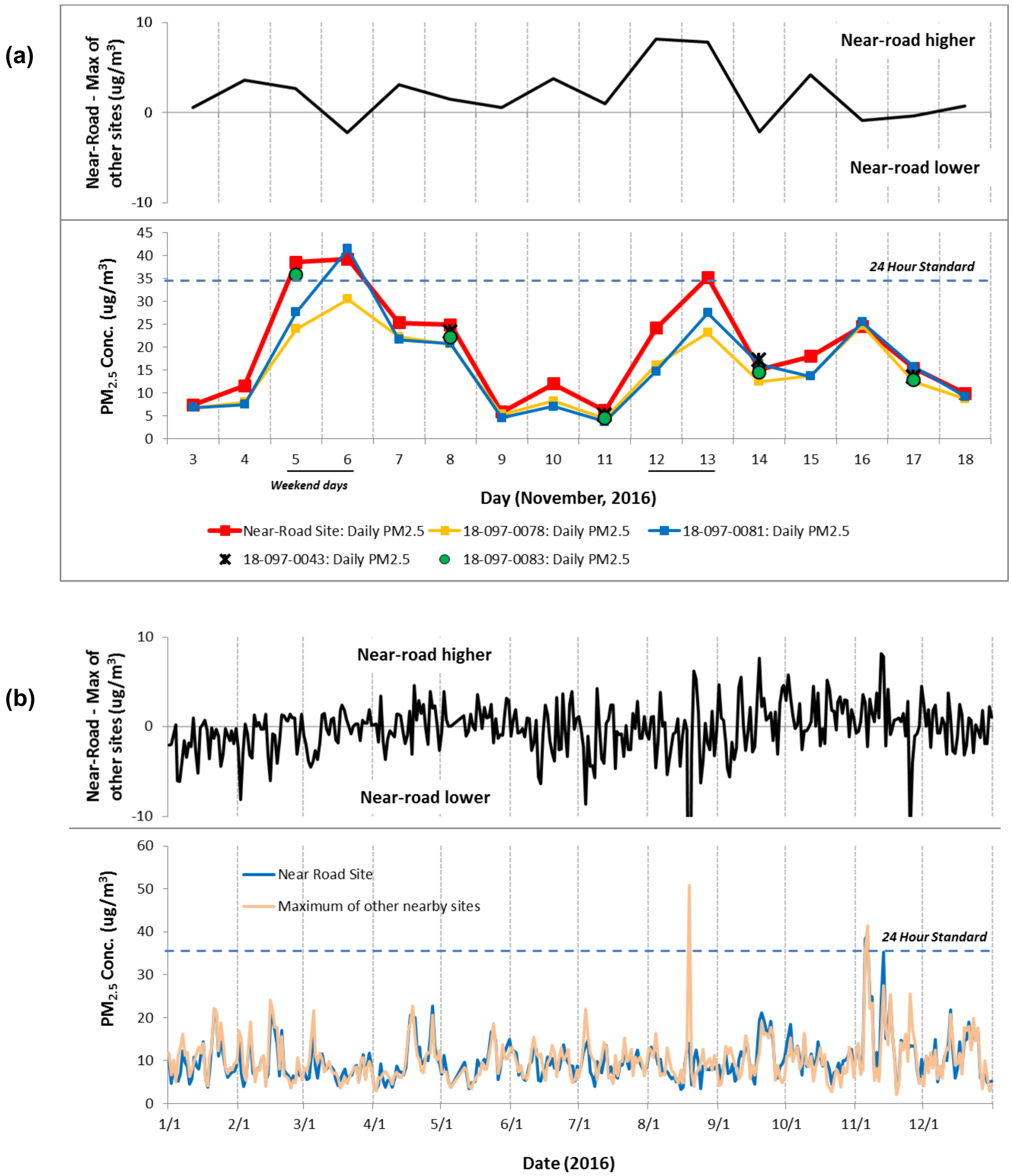


Figure 15. Indianapolis near-road $PM_{2.5}$ increment. (a) 24-hour $PM_{2.5}$ concentrations at the Indianapolis-0087 near-road monitoring site and nearby sites (denoted by AQS ID) in Indianapolis from November 3 to 18, 2016. The differences between 24-hour $PM_{2.5}$ concentrations at the near-road site and the maximum 24-hour $PM_{2.5}$ concentration at the nearby sites are also shown in the top panel. (b) 24-hour $PM_{2.5}$ concentration at Indianapolis-0087 and the maximum 24-hour $PM_{2.5}$ concentration of all other nearby sites in 2016. The difference between these measurements is shown in the top panel.

While 24-hour $PM_{2.5}$ concentrations were higher at the Indianapolis near-road site than at nearby sites on all but four days during November 3–18, 2016, this was not always the case in 2016. The 24-hour $PM_{2.5}$ concentrations measured at the near-road site were the highest among those measured in the urban area on 49% of days in 2016, and were higher than the two nearest monitoring sites on 70% of the days. Overall, as reported in Seagram et al. (2019), the average estimated near-road increment at the Indianapolis near-road site was between 0.7 and 0.9 $\mu g/m^3$; during November 3-18, the near-road site typically had the highest concentrations of any site in Indianapolis, with an average increment of 2 $\mu g/m^3$ when compared to the maximum of all sites in the CBSA, and 4.5 $\mu g/m^3$ when compared to the nearest site.

3.4 Case Study Insights: Measured Concentrations

These two case studies exhibit some of the highest U.S. near-road $PM_{2.5}$ concentrations observed in 2014-2016. In both cases, $PM_{2.5}$ concentrations at the near-road monitoring site were above NAAQS thresholds, and were higher than at nearby sites. However, in neither case was there a clear indicator in traffic data (congestion, increased number of vehicles or trucks) that would suggest that $PM_{2.5}$ should be higher at the near-road site than at other nearby sites. The near-road site was not downwind of the freeway more often than usual during these periods; an exception may be Denver, depending on the extent to which roadway alignment effectively broadened the wind conditions when the monitor was downwind of the road. Lastly, NO_2, NO, and NO_x varied widely during these periods and did not correspond to high $PM_{2.5}$. Thus, $PM_{2.5}$ concentrations were higher next to the road regardless of variations in daily traffic or meteorology.

No speciation data are available at these two near-road sites to further examine the $PM_{2.5}$ composition and the specific elements or particulate species that led to the $PM_{2.5}$ near-road increment. Jeong et al. reported similar results in Toronto, where a near-road site had higher concentrations than a nearby site; by apportioning the components of $PM_{2.5}$ through high-time-resolution measurements of $PM_{2.5}$ species, they found that traffic-related sources of $PM_{2.5}$ were two to three times higher next to the near-road site than at a nearby site (Jeong et al., 2019). The traffic-related sources included exhaust emissions plus brake/tire wear and re-suspended road dust, and the non-exhaust sources did correlate with the number of heavy-duty vehicles, though the total mass was small (roughly 5%-10% of $PM_{2.5}$). The Jeong et al. and our results are similar to those of other studies with a roadway/background site pair, where some organic compounds and black carbon are higher next to the roadway compared to a nearby or background site (Ntziachristos et al., 2007; Pant and Harrison, 2013a; Amato et al., 2011; Oakes et al., 2016). This combination may lead to higher near-road concentrations of trace metals or other species and a modest increase in total $PM_{2.5}$ mass. Over the course of 24 hours, and relying on total $PM_{2.5}$ mass as we have in our case studies here, any direct link between changes in traffic and the near-road increment of $PM_{2.5}$ mass may be masked. At the Denver and Indianapolis sites, the $PM_{2.5}$ increment is higher not when winds are directly perpendicular to the roadway, but at a slightly larger angle: near-downwind. Again, at both sites, there is asymmetry in the results, likely due to differences in wind speed and site characteristics such as the on-ramp next to the Denver site. These results support the concept that the combination of wind direction and distance is an important predictor of near-road concentrations, as seen in in modeling work elsewhere. In addition, examining hourly variations in other pollutants that have a larger near-road gradient than $PM_{2.5}$, such as black carbon or ultrafine particles, could find further evidence of how concentrations may be higher at angles other than directly downwind.

In summary, we combined the analysis of short-term $PM_{2.5}$ episodes at two near-road monitoring sites with high frequency data collected for an entire year at both sites to understand under what conditions near-road $PM_{2.5}$ is high. Near-road $PM_{2.5}$ was consistently higher at two near-road locations in Indianapolis and in Denver than at nearby locations. No direct link between changes in traffic was found to explain the day-by-day $PM_{2.5}$ increment between the near-road and nearby sites, and the increment was highest on an hourly basis when winds were near-perpendicular to the roadway, rather than directly perpendicular. The relatively small but consistent difference in near-road and nearby $PM_{2.5}$ concentrations is consistent with literature finding that road dust, break/tire wear, and black carbon are the main components resulting in higher $PM_{2.5}$ concentrations next to the roadway, and that near-road $PM_{2.5}$ concentrations averaged over the long-term are dominated by background concentrations. Future work could further examine near-road-speciated $PM_{2.5}$ concentration data, a broader array of sites with varying measured near-road increments, and the differences between measured and modeled concentration data in the near-road setting.

4. Increments: Near-Road vs. Regional-Scale Concentrations

4.1 Introduction

A key TPF analysis objective was to improve understanding of the difference between pollutant concentrations observed adjacent to major roads and concentrations measured in surrounding areas. In simple terms, the roadway contribution, or increment, is calculated by subtracting a background concentration from a measured near-road concentration. If background is assessed correctly, the difference between the two values (near-road minus background) represents the incremental pollutant concentration contributed by the roadway. Although the analysis approach is simple in concept, numerous practical challenges can make background concentration estimation difficult for particular sites.

The TPF examined background estimation and near-road increments from multiple perspectives. Our work included tasks such as surveying practitioners about the practical challenges encountered when estimating background; establishing and comparing results from various background estimation methods; [8] and, completing increasingly refined analyses to improve background and roadway increment estimation. The major analysis work comprised four phases:

- Phase One: use all available national near-road site data to estimate increments (completed with 2015 and 2016 data). This effort did not screen out data that may have been influenced by confounding factors.
- Phase Two: assess potential sources of uncertainty in background concentration estimation (completed with 2015, 2016, and 2017 data).
- Phase Three: refine the near-road data used to reduce or eliminate confounding factors that could bias estimation of the difference between background and roadway-related concentrations (completed with 2017 data, which included data from an increased number of near-road sites that became operational compared to prior years).
- Phase Four: provide insights into forecasting future-year increments.

In the first phase, our analyses established increment values by using data from all available sites within a certain radius of the near-road monitors to estimate background.[9] Analyses assessed $PM_{2.5}$

[8] The TPF developed background estimation methods based on EPA national guidance for completing PM hot-spot analyses; see Chapter 8 of the EPA PM hot-spot analysis guidance, available at https://www.epa.gov/state-and-local-transportation/project-level-conformity-and-hot-spot-analyses#pmguidance.

[9] Data were used provided monitoring locations met data "completeness" criteria. More detail on TPF data selection is available in other TPF reports and publications; see, for example, Seagram et al. (2019)

and NO_2 increments, and also assessed whether the increments were correlated with key factors such as distance to road, AADT, and FE-AADT (e.g., Seagram et al., 2019). One of the limitations of the first phase of work was that, by utilizing all available data, there was the possibility for confounding factors to influence increment values. For example, the first phase utilized data from all sites, regardless of land use differences between background monitor and near-road monitor locations. As an illustration of the limitations of the first phase findings, the analyses estimated negative near-road increments in some locations. Despite these shortcomings, since findings were aggregated over the entire country and were able to draw upon data from hundreds of monitors, first phase results helped identify increment disparities across background concentration analysis methods, metropolitan areas, traffic volumes, and roadway-to-near-road-monitor distances. Overall, the initial work established useful bounds on estimated near-road increments. We completed most of the first phase work with 2015 and 2016 data.

The second work phase examined uncertainty in background monitor selection. In this work, we demonstrated the range of potential background concentrations that could be used to represent a given metropolitan area, based on the monitoring data available. We also illustrated how background concentrations systematically differ by land use type: rural, suburban, and urban. The findings helped motivate the need for the phase three work effort.

Following completion of the first two work phases, the TPF completed more refined assessments of background concentrations and increments, focused on $PM_{2.5}$. The third phase of work focused on $PM_{2.5}$ since earlier TPF work had conclusively shown that CO and NO_2 were not present in the near-road environment at concentrations that violated the NAAQS (e.g., DeWinter et al., 2018).

In the third phase of analysis work, the TPF systematically refined the data set used to prepare background concentrations and thus establish increments. Our work involved in-depth comparisons between near-road sites and candidate background monitoring sites, and accounted for potentially confounding factors such as monitoring equipment differences, land use differences, unique site conditions such as nearby obstructions or monitor elevation issues, and large-scale natural features that could result in important meteorological differences between the near-road and background locations. The third phase used 2017 data, and was also able to take advantage of the growth in the near-road monitoring network that occurred over time. In 2016, for example, there were 29 near-road monitoring sites with complete $PM_{2.5}$ datasets; by 2017, there were 49 near-road monitoring sites with complete $PM_{2.5}$ datasets.

Finally, the fourth phase of our work examined how to forecast future-year increments, based on available near-road monitoring data. EPA guidance encourages analysts to calculate future-year background by scaling measured concentrations with results for future years from Chemical Transport Models (CTM). Modeling results are typically available in areas where EPA-approved state implementation plan (SIP) attainment demonstrations relied on future-year modeling analyses. However, modeling results are not available in all areas. Therefore, the TPF used two analysis approaches to help analysts assess future-year background concentrations in the absence of grid-based modeling results. The first approach compared “current” background concentrations for the metropolitan area (represented by EPA-calculated design values) to statutory attainment deadlines to achieve the NAAQS. Findings could potentially be used to develop year-by-year scaling factors to adjust background values over time. The second approach used vehicle fleet-average emission rate data embedded in the MOVES and EMFAC model. The approach illustrated how estimated changes in exhaust emissions over time could be used to as a scaling factor to derive future-year increments.

The remainder of this chapter includes the following discussion sections:

- **Methods.** Highlights methods developed by the TPF to calculate background and increments.
- **Phase One Increment Findings.** Shares summary results from the first phase increment analyses.
- **Phase Two Uncertainty Assessment.** Helps illustrate the need for more refined background concentration assessment by profiling the wide range of background concentrations that could be available for use in $PM_{2.5}$ nonattainment and maintenance areas.
- **Phase Three Refined Background and Increment Analysis: Removal of Potentially Confounding Factors.** Presents findings on 2017 $PM_{2.5}$ increments after accounting for confounding factors.
- **Phase Four Assessments to Represent Future Year Concentrations.** Presents two approaches to potentially scale current background and increment findings to represent future-year conditions.

4.2 Methods

This discussion describes the methods the TPF developed to estimate increments based on 2015 and 2016 data. Once the findings from these methods were available, and it was apparent that the methods were robust, we used core elements of these methods to also estimate increments based on 2017 data. The description included here focuses on our application to $PM_{2.5}$. To derive $PM_{2.5}$ 24-hour increment values, we subtracted the 24-hour background $PM_{2.5}$ concentrations from near-road concentrations. Daily increment values were then averaged over the entire year to obtain the annual mean increment. Three approaches were used to calculate the background and increment values and are described below.

Method 1. Distance/Correlation (DC). To determine which nearby sites should be used to calculate the background, the distance between the near-road monitor and nearby monitors, and the temporal correlation (based on the linear correlation coefficient, R^2) of their 24-hour $PM_{2.5}$ time series were used. This method follows the work of DeWinter et al. (2018) and is described briefly here: nearby sites with complete $PM_{2.5}$ data were selected based on a radius of 25 km, 50 km, and 100 km from the near-road site. Nearby sites were also selected within 100 km where the R^2 of the 24-hour $PM_{2.5}$ concentrations was greater than or equal to 0.50, 0.75, and 0.90. For each of these six selection approaches, daily $PM_{2.5}$ increments were calculated, then averaged over the entire year. The average increment at each site was calculated as the average of the annual mean daily increment from all six approaches.

Method 2. Inverse Distance Weighting (IDW). The EPA suggests that IDW interpolation can be used to calculate background concentrations (U.S. Environmental Protection Agency, 2015c). In this study, only sites within 40 km of the near-road site were selected. This radius was used to ensure that nearby sites were in the same general urban area as the near-road site (Prud'homme et al., 2013). Daily background $PM_{2.5}$ concentrations for each near-road site were calculated through IDW interpolation using the daily $PM_{2.5}$ data from up to four of the closest $PM_{2.5}$ monitoring sites. If a nearby site had collocated $PM_{2.5}$ monitors, only the data from the monitor with the most complete dataset were used. The daily differences were then averaged for each near-road site as the mean $PM_{2.5}$ increment.

Method 3. Upwind Monitor (WD). The background $PM_{2.5}$ concentration was based on data from a single nearby site that met the following criteria: (1) $PM_{2.5}$ and wind (direction and speed) data were complete, (2) the nearby site is within a similar setting (land use category) as the near-road site, (3) the site is within 40 km of the near-road site, (4) the site is located within the predominant upwind direction of the near-road site. These criteria are also similar to those suggested by EPA to calculate background concentrations (U.S. Environmental Protection Agency, 2015c). The predominant upwind direction was determined by identifying the 90° bin from which the wind was most frequently observed (excluding when the wind speed was 0 m/s). If a nearby upwind site had collocated $PM_{2.5}$ monitors, only the data from the monitor with the most complete dataset was used. The daily differences were then averaged for each near-road site as the mean $PM_{2.5}$ increment.

4.3 Phase One: Increments Based on all Near-Road Data (Confounding Factors Not Yet Addressed)

$PM_{2.5}$ increment calculations using all three methods (distance/correlation, IDW, upwind) for 2016 are shown in Figure 16. With the IDW method, the mean increment was 0.6 µg/m^3, which was 6% of the annual average $PM_{2.5}$ across all sites. With the WD approach, the mean increment was 1.1 µg/m^3. Thus, across all three approaches, the mean increment is 0.6-1.1 µg/m^3, and is consistent overall and for most individual sites.

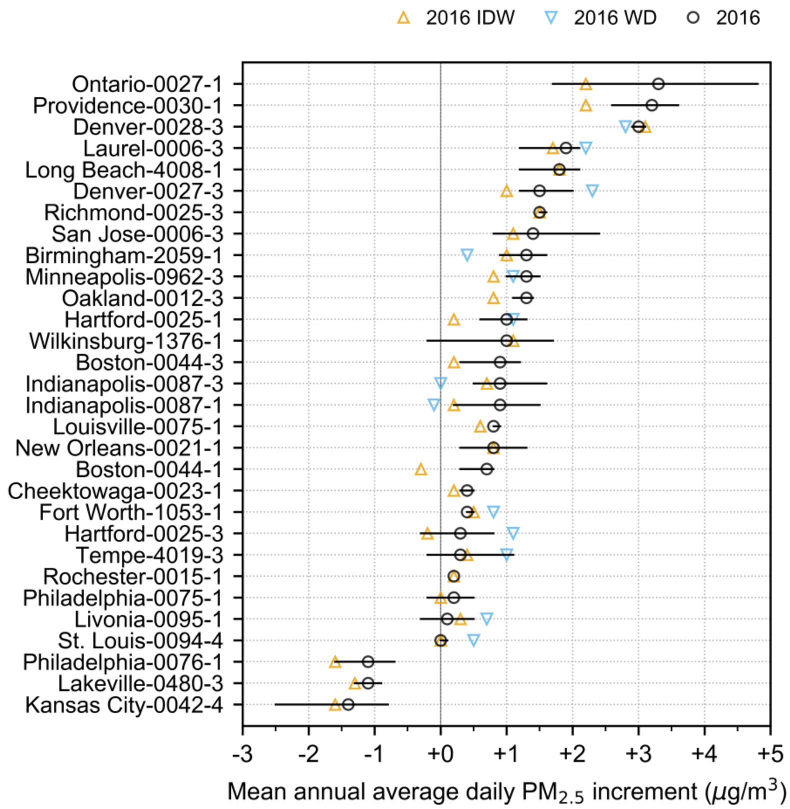


Figure 16. Mean annual average daily $PM_{2.5}$ increment in 2016, based on the method used to calculate the increment (IDW approach, upwind approach, and distance/correlation approach). Black circles indicate the mean of the distance/correlation method, horizontal lines indicate the range of $PM_{2.5}$ increment values obtained from the distance/correlation method.

It is important to realize there is year-to-year variability in measured $PM_{2.5}$ at near-road sites, and in the calculated increments. Figure 17 illustrates differences between 2015 and 2016 findings, for the distance/correlation method. In 2016, the average increment across the sites was 0.9 µg/m^3 (n = 30 sites), slightly less than the mean of 1.2 µg/m^3 in 2015 (n = 26 sites). The range of increments from individual sites was similar in both years, from -1.2 µg/m^3 to 3.2 µg/m^3 in 2015, and from -1.4 µg/m^3 to 3.3 µg/m^3 in 2016.

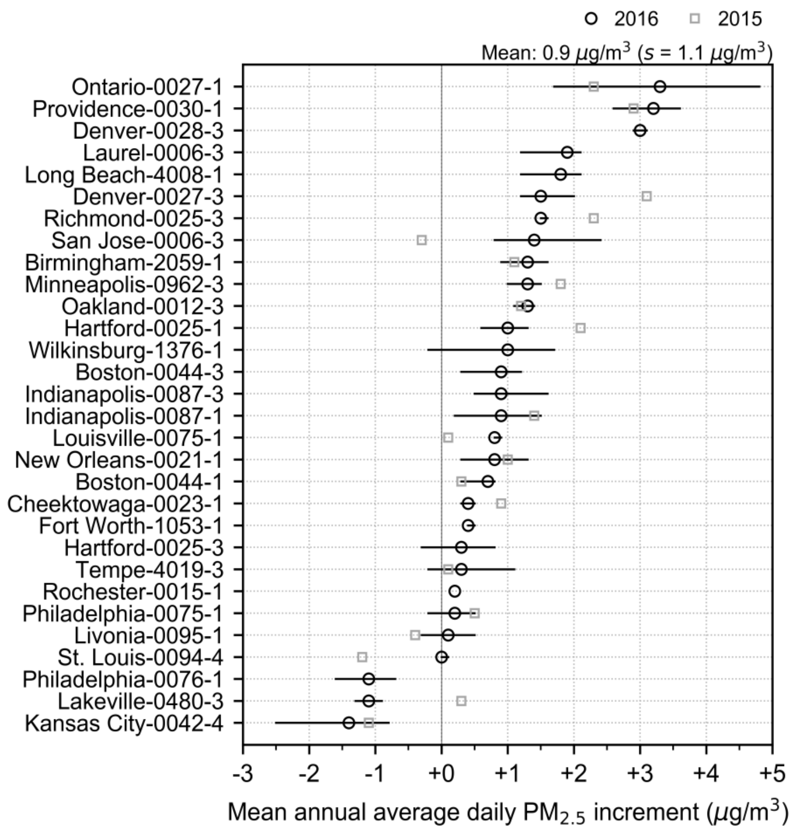


Figure 17. Annual average daily $PM_{2.5}$ increment at near-road monitoring sites in 2016 using the distance/correlation method (black circles). The range of $PM_{2.5}$ increment values obtained from this method is denoted (horizontal lines). The mean annual average daily $PM_{2.5}$ increment in 2015 is also plotted (gray squares); not all results are available in 2015 due to data completeness requirements.

The results presented in Figures 16 and 17 can be compared to the more refined analysis results presented later (Section 4.5), to illustrate how removal of potentially confounding factors affects the range of $PM_{2.5}$ increments observed throughout the United States.

In addition, the first phase analysis also assessed the relationship of estimated increments to distance from road (*d*) and traffic counts (Figure 18).

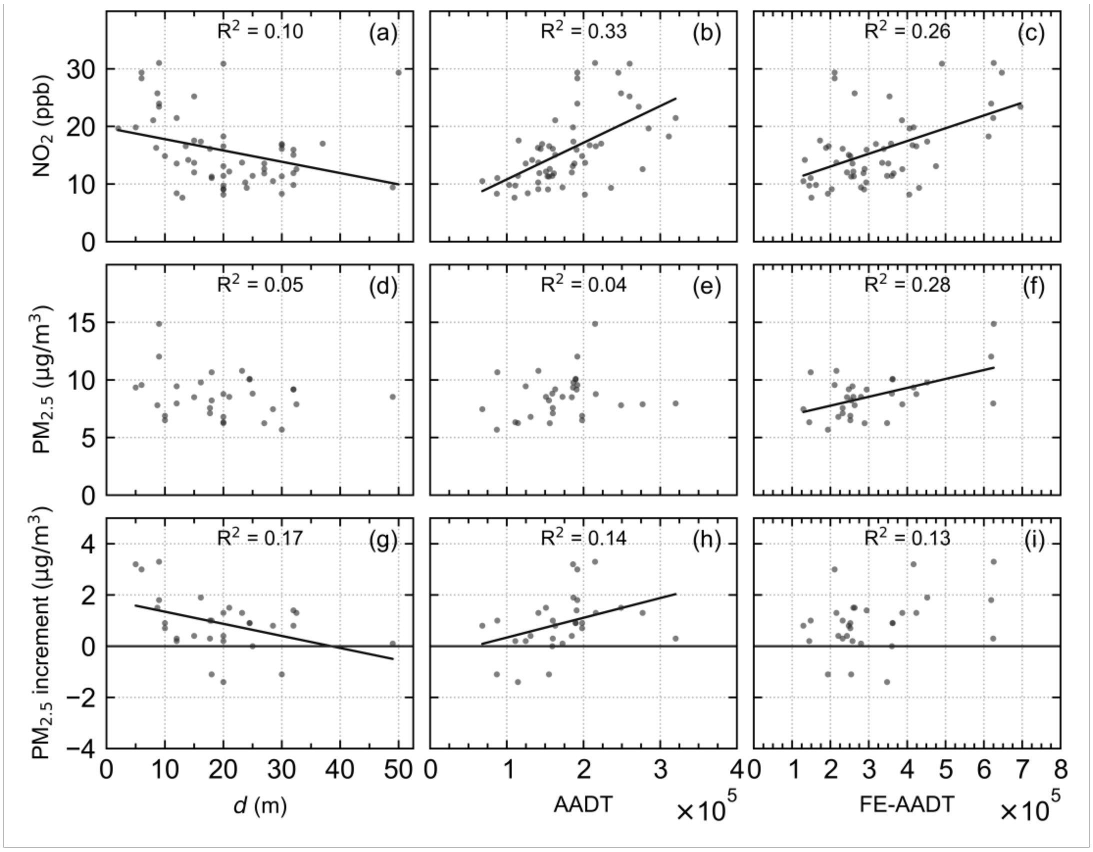


Figure 18. Relationship between annual mean NO_2, $PM_{2.5}$, and mean daily near-road $PM_{2.5}$ increment (using DC) in 2016 and distance to target road (d), AADT, and FE-AADT. The R^2 of a linear regression is displayed at the top of each panel. Linear models (solid lines) are displayed only for linear regressions with p-value < 0.05.

There was a statistically significant (*p*-value > 0.05) linear decrease of annual mean NO_2 with increasing *d* (Figure 18). There was no statistically significant linear relationship between annual average $PM_{2.5}$ and *d*. This is similar to what other studies have found (Roorda-Knape et al., 1998). However, there was a statistically significant relationship between the $PM_{2.5}$ increment and *d*. This relationship is not surprising, given that we expect that the closer the near-road monitoring site is to the roadway, the greater the difference would be between $PM_{2.5}$ concentrations at the near-road monitoring site and nearby (regional) monitoring sites, which are farther from near-road sites than near-road sites are from each other. Low R^2 values are somewhat expected, given that this analysis approach is not akin to a true gradient study, since near-road sites are subjected to different traffic patterns and volumes, and meteorological conditions over the same annual analysis period.

There was little correlation between annual mean pollutant concentrations with FE-AADT, though there is some evidence that annual mean NO_2 concentrations increase with increasing AADT and FE-AADT (Figure 18). While this relationship is to be expected based on previous research, FE-AADT is also based on NO_x emissions, so it may not be surprising that annual average $PM_{2.5}$ is not well correlated with this parameter.

4.4 Phase Two: Background Concentration Assessment Uncertainty

STI assessed the range of $PM_{2.5}$ concentrations measured at monitors within each of the metropolitan areas for which a $PM_{2.5}$ near-road monitor was operational in 2017. The results provide quantitative bounds for how much increment calculation uncertainty can be introduced by the incorrect selection of a background monitor. $PM_{2.5}$ hot-spot analyses are performed by estimating both the 24-hour and annual forms of the $PM_{2.5}$ NAAQS (also referred to as "design values").[10] Both forms are calculated by using three years of measurements. Therefore, we obtained annual summary data for the years 2015-2017 from the EPA Air Data web portal. The annual summary data set includes annual means calculated using both forms of the standard. From this dataset, the three-year average was calculated; only complete annual and three-year aggregates were included in subsequent analyses. Data that were categorized by EPA as resulting from exceptional events were excluded from the annual and three-year aggregates. Using these results, STI performed the following analyses.

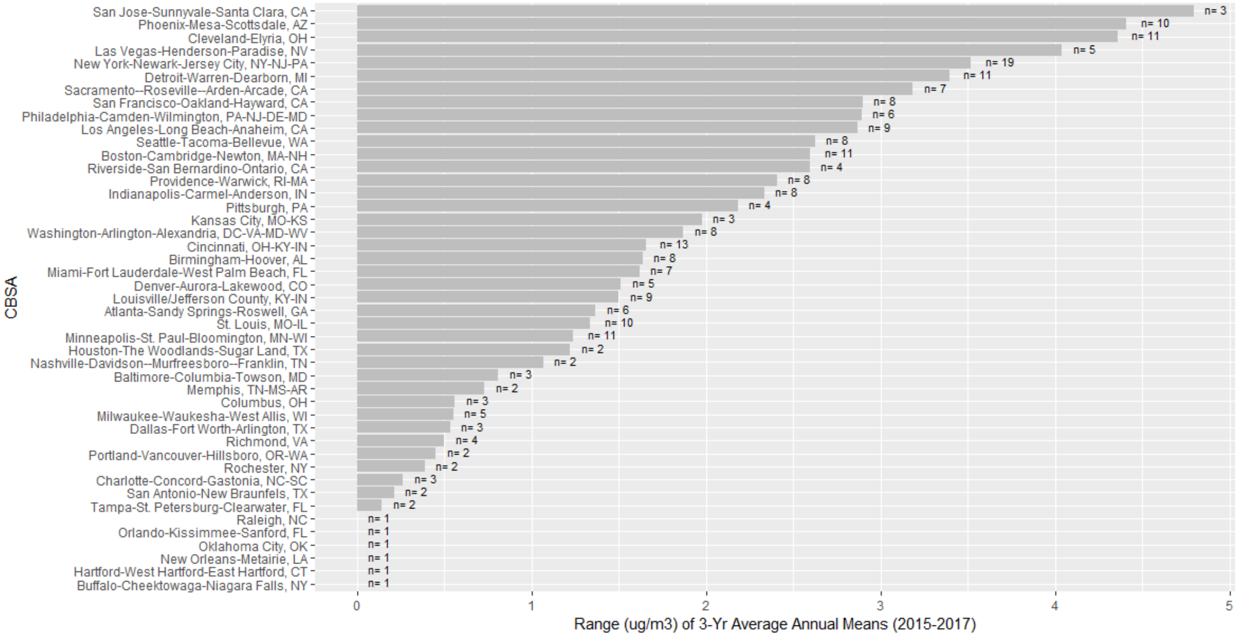
- **Three-year average annual mean.** We evaluated the range of three-year average annual mean data across $PM_{2.5}$ monitors in each metropolitan area with a near-road monitor in 2017. For example, assume Metropolitan Area A has four $PM_{2.5}$ monitors. For Area A, we calculated the three-year average of the annual means for each monitor using data from 2015-2017. We then developed graphics to display the range across all monitors in Area A, in order to help project analysts quantitatively understand the potential impact of incorrectly selecting the monitor(s) to represent background concentrations at their project site.
- **Three-year average of 98th percentile of 24-hr mean.** To complement the annual average data from the first analysis, we also calculated the three-year average of the 98th percentile of daily mean data for each monitor in the metropolitan areas that had an operational near-road $PM_{2.5}$ monitor in 2017. We then developed graphics to display the range across all monitors in each metropolitan area.
- **Nearest monitor comparison.** Finally, for each metropolitan area in which a $PM_{2.5}$ near-road monitor was operational in 2017, we evaluated the differences in three-year average annual means between monitor pairs by subtracting the value at the nearby monitor from the value at the near-road monitor. The point of this analysis is to illustrate the potential uncertainty or bias that could be introduced into a background concentration calculation by selecting a monitor *near* the "correct" representative monitor, rather than the correct representative monitor itself. We note that the increments calculated as part of this analysis may differ from previously reported results because they are calculated using three-year mean data, rather than daily data.

Overall, 45 core-based statistical areas (CBSAs) were evaluated in these analyses. As shown in Figure 19, in half of the CBSAs, the maximum estimated background concentration, using either the

[10] See https://www.epa.gov/criteria-air-pollutants/naaqs-table.

three-year average of annual means or three-year average of 98th percentiles of daily means, could be from 25% higher to 200% higher than the minimum estimated background concentration, depending on monitor choice. In the other half of CBSAs, monitor choice results in a more similar background value (i.e., less than a 25% difference in design value).

The number of monitors in a CBSA is not on its own an indicator of the potential range of background concentration values that might be available for use. However, as shown in **Figure 20**, monitor location classification (rural, suburban, city center) is an important factor. Design values at rural monitors are typically lower than the CBSA-wide mean; suburban or city center monitors are typically equal to or greater than the CBSA-wide mean. Many CBSAs cover a large geographic area that can be highly varied in terms of population density, land use, and number and type of emissions sources. In general, a larger range is observed across monitors in a CBSA when the 24-hour metric is used. This result is expected given the greater variability in 98th percentile values compared to annual mean values.



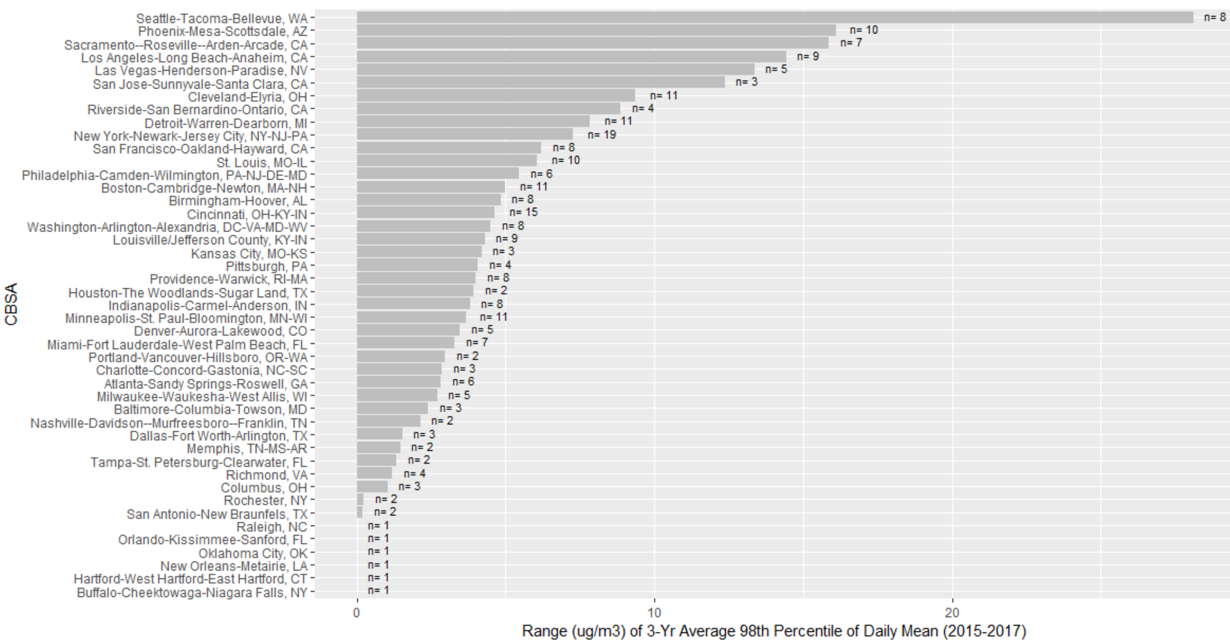


Figure 19. Range of 3-year average annual mean (top) and range of 3-year average of the 98th percentile of daily mean (bottom) across monitors, by CBSA area; "n" indicates the number of monitors in each CBSA (excluding the near-road monitor).

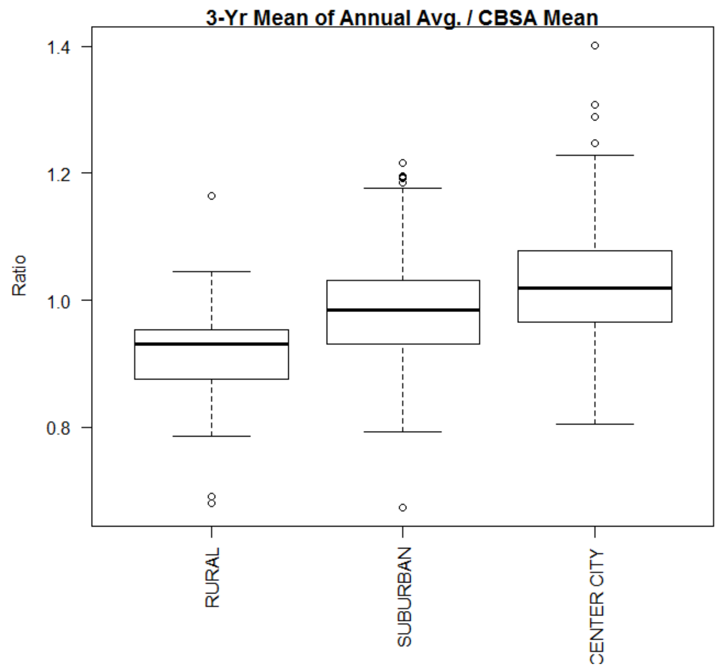


Figure 20. Ratio of three-year average annual means, by location type, to overall CBSA mean in the CBSA area.

Figure 21 provides the results of the nearby-monitor-pairs analysis that was conducted for 20 near-road monitors that had three years of complete $PM_{2.5}$ data, using three-year average annual mean data for complete pairs. The increment across pairs ranged from -2.5 μg/m^3 to 5 μg/m^3. A negative increment implies that if the background were estimated using that nearby monitor, it would overestimate true background concentrations at the specific near-road location. In general, increments increased (positively or negatively) with greater distance between monitor pairs.

Also shown in Figure 21 are the increments between the near-road monitor and the nearest upwind monitor. EPA recommends that background concentrations be estimated using the data from the nearest upwind monitor, if such a monitor is available nearby and not in an environment with other emissions sources that are different from those in the project area. Seven near-road monitors with a nearby upwind monitor were available for evaluation; upwind monitors were not available for the other near-road locations due to either the absence of an upwind monitor or incomplete data. Of these pairs, most increments were smaller than if the increment had been calculated using another monitor in the CBSA. For the two near-road monitors collocated in Denver, using the nearest upwind monitor would result in the lowest estimate of background, and thus the largest calculated increment. However, in these two locations, the "nearest" upwind monitor was 10-20 km away.

The findings presented from this uncertainty analysis helped demonstrate the need for more refined evaluation of background monitor selection when estimating increments. In the next section, we present results from the third phase work effort.

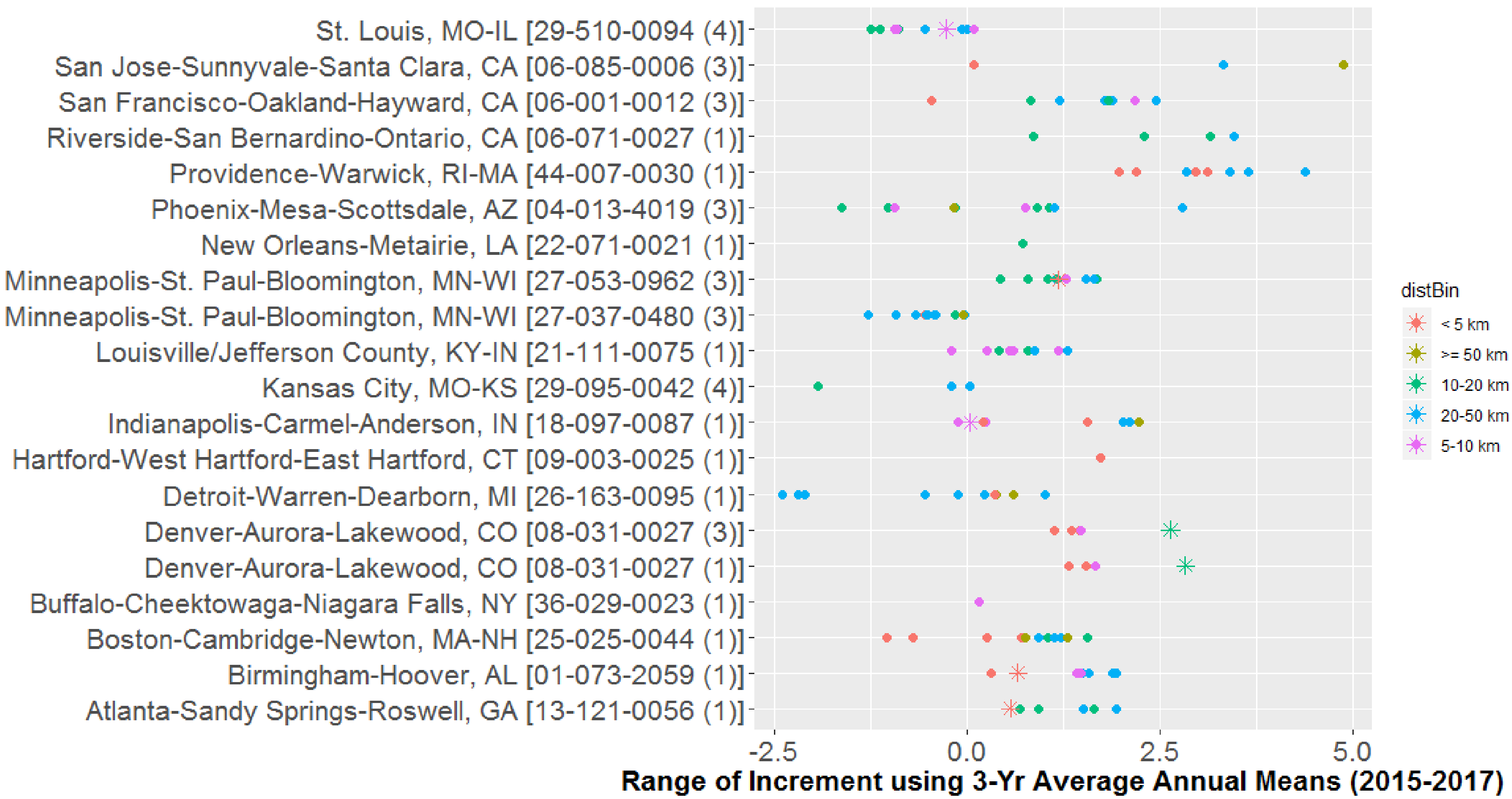


Figure 21. $PM_{2.5}$ increment between the 3-year average annual mean at the near-road monitor (y-axis) and each nearby monitor within the CBSA. The colors indicate the distance between each nearby monitor and the near-road monitor. The asterisk indicates the nearest upwind monitor to the near-road monitor, for the seven locations where prevailing winds could be used to establish an upwind site.

4.5 Phase Three: Refined Background and Increment Analysis (Confounding Factors Removed)

Here, we present the findings from a more refined analysis. In this work, based on the larger pool of near-road $PM_{2.5}$ monitors and data available from 2017 compared to 2016, we identified and removed data that could potentially affect the increment analyses due to confounding factors. First, we examined $PM_{2.5}$ measurements from all 49 near-road sites where measurements met EPA data completeness thresholds during 2017. Since our objective was to assess near-road $PM_{2.5}$ increments, we needed to then pair near-road site data with data from an ambient monitor. Of the 49 near-road sites, 48 sites had at least one ambient site within 40 km where $PM_{2.5}$ measurements met data completeness thresholds. For the 48 sites, total near-road $PM_{2.5}$ concentrations were compared to the NAAQS to establish a national-scale understanding of near-road $PM_{2.5}$. Next, to improve accuracy for increment assessments, we narrowed our sample to cover only those site pairings where the near-road and the ambient site measured $PM_{2.5}$ using identical monitoring instruments. This produced a sample of 40 near-road sites for which we estimated $PM_{2.5}$ increments. Among the 40 near-road sites, nine sites were estimated to have negative $PM_{2.5}$ increments, indicating that confounding factors skewed increment assessment at those sites (i.e., roadway emissions by definition add some incremental concentration to background; a negative increment implies incorrect representation of background concentrations at the near-road site). We therefore further narrowed our analysis sample to the 31 sites for which we estimated a positive near-road $PM_{2.5}$ increment. We refer to this sample as our “initial case” in findings presented later. Each of the 40 near-road sites were then evaluated for potential confounding factors including characteristics of the near-road environment such as site elevation or the presence of nearby barriers, commonality of land use between near-road and background monitors, and potential sea breeze effects that could skew findings. Removing sites with a noted confounding factor resulted in a sample of 20 near-road sites we refer to as the “focused case” which is also a subset of the initial case. In summary, we progressively narrowed our data sample to remove confounding factors and improve increment evaluation:

1. 49 sites: our starting sample of all sites reporting complete near-road $PM_{2.5}$ data in 2017
2. 48 sites: all of the near-road sites with an ambient monitor available for background
3. 40 sites: near-road sites paired with ambient sites using identical monitoring instruments
4. 31 sites: our “initial case” near-road site sample after removing negative increment sites
5. 20 sites: our “focused case” sample after addressing remaining confounding factors

The statistical relationships between the annual average $PM_{2.5}$ increment and traffic volumes, distance of the monitor to the roadway, and meteorological variables were assessed using pairwise correlation of determination (R^2), regression models, and a general additive model (GAM) from sets (4) and (5) of the near-road sites listed above.

4.5.1 Data Analysis Methods: Confounding Factors

We evaluated four confounding factors: the commonality of instrument method at near-road and ambient sites, characteristics of the near-road site environment, commonality of land use between near-road and nearby ambient sites, and the sea breeze effect.

Monitoring instruments that meet specific quality control and operational standards are designated by the EPA as Federal Reference or Equivalent Methods (FRM or FEM) provided they are calibrated and operated according to standardized procedures (U.S. Environmental Protection Agency, 2016). While these instruments have met rigorous standards, there still remain differences in instrumental precision and performance, which can depend on the chemical composition of the PM sampled, the instrument method, and environmental conditions. A national 3-year assessment from 2014-2016 found biases unique to each instrument were less than 10% for FRMs, and up to 22% for some FEM instruments (U.S. Environmental Protection Agency, 2015a). For this project, a case study of the Milwaukee-0056 site was used to illustrate the impact of choosing differing or identical instrument methods at the near-road and ambient sites. Based on EPA's work and the Milwaukee case study, increments were calculated using only identical instrument methods between sites; e.g., if the near-road site had an FRM instrument only data from the same FRM instrument at the nearby sites were used to calculate the increment.

The immediate environment of the near-road sites was examined to see if local topography or other environmental characteristics were a confounding factor in measuring near-road $PM_{2.5}$ for any sites. The immediate area of the near-road sites was examined using Google Earth and Google Maps Street View. Google Earth was used to quantify the elevation difference between the near-road monitor and the centroid of the target road. Street View was used to determine the presence or absence of any barriers such as sound walls, trees, or bushes between the monitor and the roadway or near the monitor. Many sites had complex local topography and/or a complex built environment, including nearby interchanges, depressed roadways, or nearby walls that could influence the $PM_{2.5}$ measurements.

The availability of ambient monitors that can provide an accurate representation of the background $PM_{2.5}$ was examined using land use data. $PM_{2.5}$ can vary significantly within a metropolitan domain; for example, variations of annual average $PM_{2.5}$ of about 25-30% were observed in Jefferson County in the years 2000-2009 (Superczynski and Christopher, 2011). Jefferson County includes the Birmingham-2059 near-road site and is roughly equivalent to our 40 km radius zone. The study of Jefferson County found significantly higher $PM_{2.5}$ near the urban center of Birmingham, driven by emissions from manufacturing, industry, and power generation (Superczynski and Christopher, 2011). We used population density, imperviousness, and derived urban intensity to evaluate site pairs of near-road and ambient sites to determine whether ambient monitors were representative of the background $PM_{2.5}$ at the near-road site. We binned near-road and ambient sites into one of four land uses: rural, suburban, urban or dense urban. If no ambient monitor in the same land use bin was within 20 km of the near-road site, we identified land use as a confounding factor. For this study, a case study of the near-road site, Cleveland-0073, was carried out to demonstrate how commonality of land use can impact the increment.

Another confounding factor that could skew representation of background $PM_{2.5}$ is the sea breeze effect. The sea breeze effect is the impact of meteorology on the environment of coastal domains and other areas near large bodies of water. Coastal communities often experience a diurnal pattern of winds flowing toward the land during the day and toward the ocean during the night, driven by the pressure gradient resulting from different rates of heating on land and on water. This diurnal wind circulation, and the absence of significant emissions from the water, leads to lower air pollutant concentrations alongside coastal regions. The sea breeze effect can lead to a positive or negative bias in calculating the increment. A case study of the near-road site, Berkeley-0013, was carried out to demonstrate how the sea breeze effect can impact the increment.

4.5.2 Results

$PM_{2.5}$ NAAQS Comparison for 48 Near-Road Sites

The distributions of 2017 daily average near-road $PM_{2.5}$ from the sample of 48 near-road sites with background data available are presented in **Figure 22**. The $PM_{2.5}$ NAAQS (annual and 98th percentile daily average) are also shown. Note that Figure 22 compares measured concentrations to the NAAQS for research purposes only; the analysis represents only one year of data and does not represent a calculation to determine attainment status. There are significant differences across the near-road monitoring sites, representing the range of $PM_{2.5}$ seen across the different metropolitan areas and the impact of local sources. Sites with multiple instruments (POCs) are plotted separately; different distributions are due to the differences in instrument method and in the sampling intervals. Two sites, Long Beach-4008 (POC 1 and POC 3), and Ontario-0027, exceeded the annual average $PM_{2.5}$ NAAQS value of 12 μg/m^3. Five sites exceeded the daily 98th percentile $PM_{2.5}$ value of 35 μg/m^3: Long Beach-4008 (POC 1 and POC 3), Ontario-0027, Oakland-0012, San Jose-0006, and Seattle-0030.

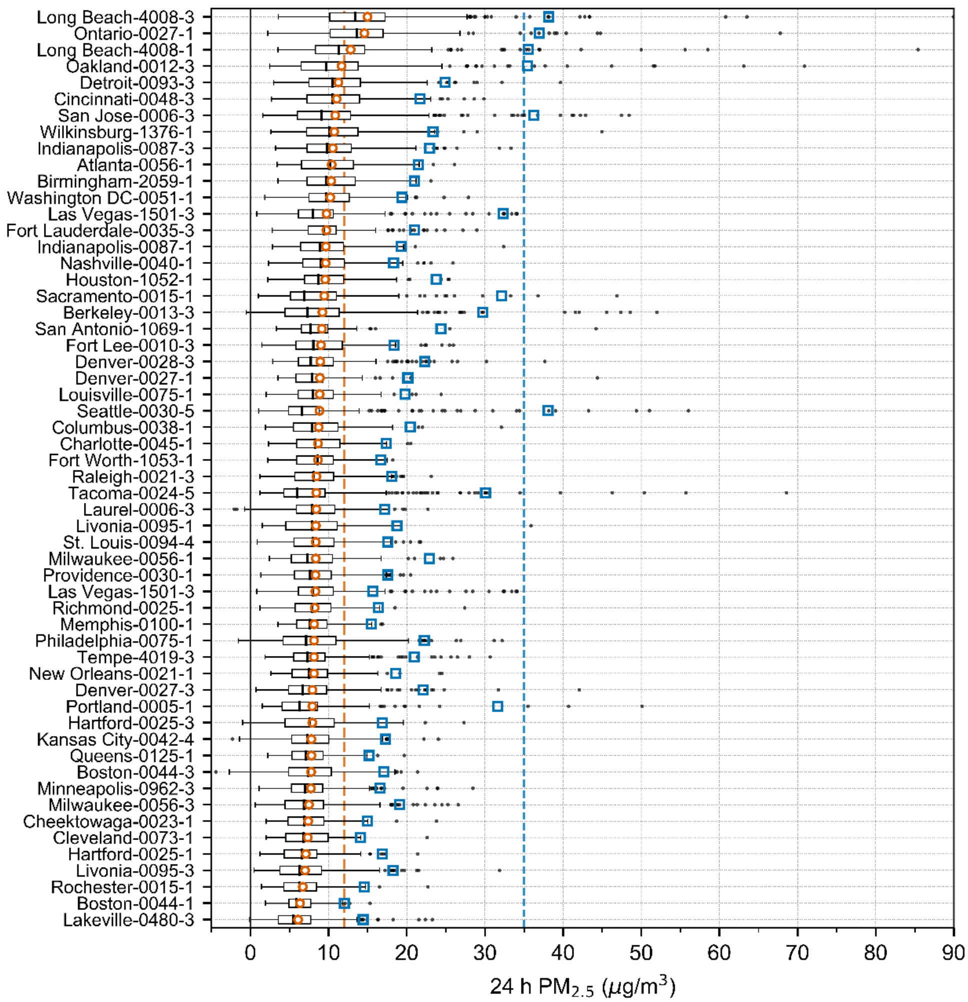


Figure 22. Distribution of daily average $PM_{2.5}$ at 48 near-road monitoring sites in 2017, sorted by annual mean. The annual mean (orange circles) and 98th percentile of 24 hr $PM_{2.5}$ concentrations (blue squares) are shown. The orange dashed lined denotes the annual NAAQS threshold (12 μg/m^3), and the blue dashed line denotes the daily average NAAQS threshold (35 μg/m^3).

Near-Road Site Characteristics and Confounding Factor Evaluations

Table 3 lists the 20 sites included after removing confounding factors. Increments from the IDW and nearest monitor methods are presented only where identical method comparisons were available. The meteorological parameters of average wind speed, and upwind vs. downwind conditions for the near-road sites are shown. The number of trucks was estimated from EPA's FE-AADT values assuming a scaling factor of 10 for HDDVs.

Table 3. Near-road site characteristics and 2017 increments from IDW and nearest monitor methods, for the case of 20 near-road sites where all sites with a noted confounding factor have been removed. Number of trucks was estimated from FE-AADT and AADT values assuming the default scaling factor of 10.

Site Name	AQS ID	Annual Average $PM_{2.5}$ (2017)	IDW $PM_{2.5}$ Increment	Nearest Neighbor $PM_{2.5}$ Increment	Average Wind Speed (m/s)	Percent Downwind	Percent Upwind	Percent Parallel	Distance to Road (meters)	FE-AADT	AADT	Number of Trucks	Method Type	N of Background Stations (IDW)
Birmingham-2059	01-073-2059	10.33	0.54	0.36	0.2	36	38	27	23	215,527	141,190	8,260	FRM	4
Charlotte-0045	37-119-0045	8.65	0.45	0.36	1.0	13	41	45	30	260,830	153,000	11,981	FRM	2
Cheektowaga-0023	36-029-0023	7.39	0.47	0.22	2.5	47	33	20	20	220,543	131,019	9,947	FRM	2
Columbus-0038	39-049-0038	8.72	0.5	0.49	1.0	37	35	28	32	286,050	142,361	15,965	FRM	3
Denver-0027	08-031-0027	7.93	1.73	1.84	1.0	24	25	52	9	263,118	249,000	1,569	FRM	6
Denver-0028	08-031-0028	8.92	1.25	1.44	1.1	33	25	42	6	210,835	192,000	2,093	FEM	3
Indianapolis-0087	18-097-0087	10.55	0.54	0.55	1.9	28	36	36	25	362,110	189,760	19,150	FRM	5
Laurel-0006	24-027-0006	8.49	0.99	1.13	1.0	25	31	44	16	452,309	186,750	29,507	FEM	4
Long Beach-4008	06-037-4008	14.9	1.95	2.03	1.1	52	23	25	9	619,008	192,000	47,445	FRM	2
Louisville-0075	21-111-0075	8.86	0.78	0.89	1.9	30	32	38	32	247,600	163,000	9,400	FRM	2
Memphis-0100	47-157-0100	8.14	0.31	0.32	2.1	23	44	33	24	292,968	140,850	16,902	FRM	2
Milwaukee-0056	55-079-0056	7.52	0.31	0.37	2.3	47	18	35	14	133,000	133,000	0	FRM	3
Minneapolis-0962	27-053-0962	7.72	0.14	0.13	1.7	30	34	36	33	387,250	277,000	12,250	FEM	6

Site Name	AQS ID	Annual Average $PM_{2.5}$ (2017)	IDW $PM_{2.5}$ Increment	Nearest Neighbor $PM_{2.5}$ Increment	Average Wind Speed (m/s)	Percent Downwind	Percent Upwind	Percent Parallel	Distance to Road (meters)	FE-AADT	AADT	Number of Trucks	Method Type	N of Background Stations (IDW)
New Orleans-0021	22-071-0021	8.07	0.35	0.49	3.2	41	33	25	29	129,229	68,015	6,802	FRM	2
Providence-0030	44-007-0030	8.31	2.04	2.04	2.2	44	17	39	5	416,790	186,300	25,610	FEM	5
Rochester-0015	36-055-0015	6.69	0.2	0.2	3.5	40	19	41	20	144,717	110,990	3,747	FRM	1
Sacramento-0015	06-067-0015	9.46	0.48	0.56	1.5	29	27	44	20	475,000	186,000	32,111	FRM	3
San Jose-0006	06-085-0006	10.81	1.36	1.24	2.2	33	13	53	32	294,140	191,000	11,460	FEM	4
Tempe-4019	04-013-4019	8.14	0.49	0.97	0.7	45	40	15	12	624,315	320,138	33,797	FEM	7
Washington DC-0051	11-001-0051	10.23	0.72	0.22	1.6	27	47	26	15	172,747	115,480	6,363	FEM	3

$PM_{2.5}$ Increments at 20 Sites, After Removal of Confounding Factors

Next we examine increments from the 20 near-road sites remaining after all sites with one or more of the previously discussed confounding factors have been removed. Increments from the 20 sites are presented in Figure 23. Once confounding factors are addressed, there are no sites with negative increments. The upper bound of $PM_{2.5}$ increments is 2.04 µg/m^3. Only three sites have an increment greater than 1.44 µg/m^3; monitors at each of these three sites are sited less than 10 meters from the roadway.

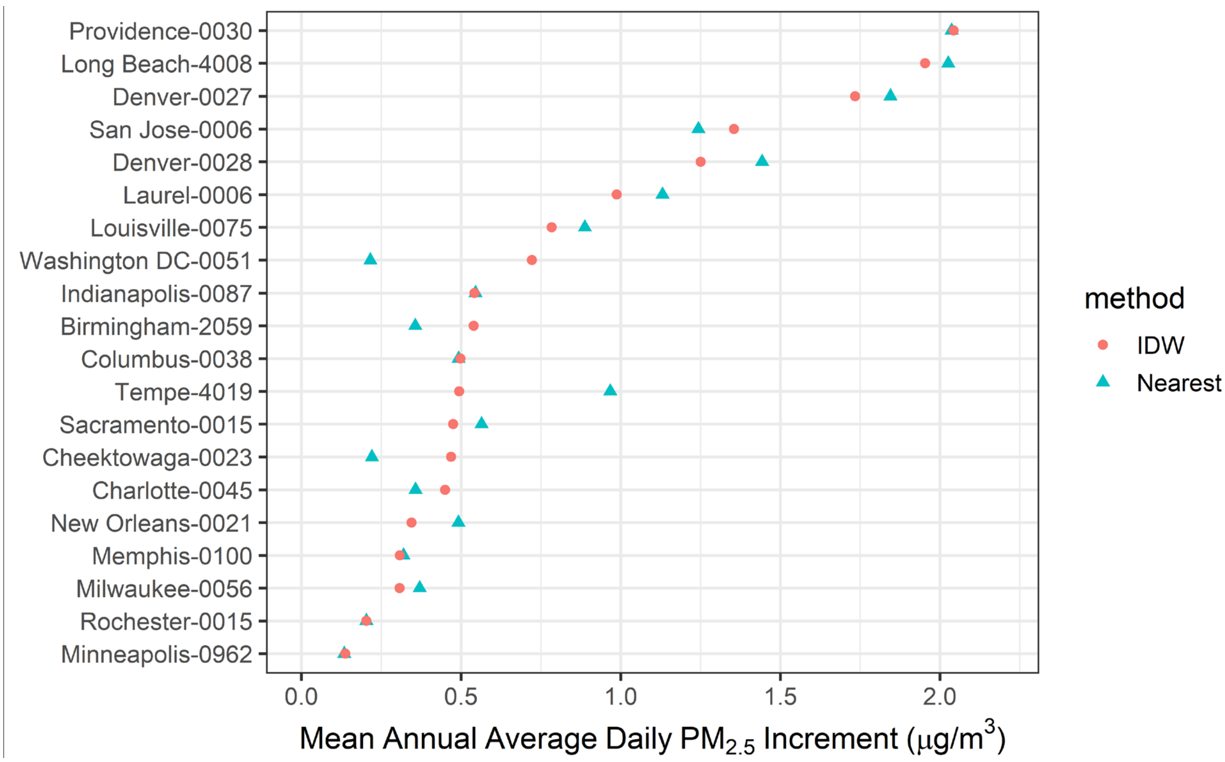


Figure 23. Distributions of annual average $PM_{2.5}$ increments computed using IDW and nearest monitor calculation (Nearest). Results for 20 sites are shown, controlling for confounding factors.

Comparison to Meteorology, Traffic, and Site Characteristic Variables

The initial case of 31 increments and the focused case of 20 increments, all using identical instrument comparisons, were used to assess the relationship of near-road increments to variables representing meteorology, traffic, and site characteristics. The initial case of 31 sites was included due to its higher sample size. The coefficient of determination (R^2 value) is presented for four cases: the sets of 31 sites and 20 sites, both with IDW and nearest monitor calculations of the increment (Table 4). Increments were compared with annual average wind speed; percent of time the near-road site was downwind, parallel, or upwind of the adjacent roadway; distance to road; FE-AADT; AADT; and estimated number of HDDVs. A positive correlation with the increment was seen for FE-AADT, AADT and the percent of time the near-road site was parallel or downwind of the road. A negative correlation was seen between the increment and percent of time upwind of the road, distance to road, and wind speed. For the focused case of 20 sites, and the IDW method, the largest correlation was for distance to road (R^2 = 0.34), followed by percent upwind (R^2 = 0.25) and FE-AADT (R^2 = 0.12). Weaker correlations were observed for number of HDDVs, percent of time parallel to the road, AADT and average wind speed. Almost no correlation was seen for percent of time downwind of the road. Correlations were similar between the set of 20 sites and the set of 31 sites, with some higher correlations seen in the 20 site case. FE-AADT and number of HDDVs show a higher correlation to the increment than AADT, likely indicating the importance of HDDVs in contributing to $PM_{2.5}$ emissions for these roadways. Increments calculated through the IDW method and the nearest monitor method showed very similar statistical relationships to the meteorology and traffic variables. These pairwise correlations are limited because they only consider one comparison at a time independently.

Table 4. Coefficient of determination (R^2 value) for IDW and nearest monitor increments. The comparisons are shown with the initial case of 31 sites with identical instrument methods and the focused case of 20 sites limiting confounding factors. Variables are rank-ordered by IDW Method, 20 site comparison R^2 values.

Variable	IDW Method (31 Sites)	Nearest Monitor (31 Sites)	IDW Method (20 Sites)	Nearest Monitor (20 Sites)
Distance to Road	0.30	0.26	0.34	0.37
Percent Upwind	0.08	0.10	0.25	0.28
FE-AADT	0.13	0.11	0.12	0.22
Number of HDDVs	0.13	0.09	0.10	0.16
Percent Parallel	0.14	0.09	0.09	0.08
AADT	0.03	0.06	0.07	0.17
Average Wind Speed	0.08	0.07	0.06	0.07
Percent Downwind	0.01	0.00	0.03	0.04

Regressions for the IDW-calculated increment and three other variables, distance to road, percent of time the monitor was upwind, and FE-AADT, are shown in Figure 24, for both the 31-site and the 20-site cases. A linear regression ($y = a \cdot x + b$) was used for FE-AADT and an inverse relationship ($y = a / x + b$) was used for distance to road and percent of time the monitor was upwind. The coefficients and p values of these regressions are shown in Table 5. The p values show that these modeled relationships are statistically significant. The modeled increment falls from approximately 2 μg/m^3 at 5 meters from the roadway to approximately 0.5 μg/m^3 at 30 meters from the roadway for the 20 site case (99.9% confidence that the relationship exists). The increment falls from 1.5 μg/m^3 when the receptor is upwind of the road 15% of the time, to 0.75 μg/m^3 when the receptor is upwind of the road 30% of the time for the 20 site case (97% confidence). The linear regression for FE-AADT predicts a relationship of 0.14 μg/m^3 higher $PM_{2.5}$ values for every 100,000 increase in FE-AADT.

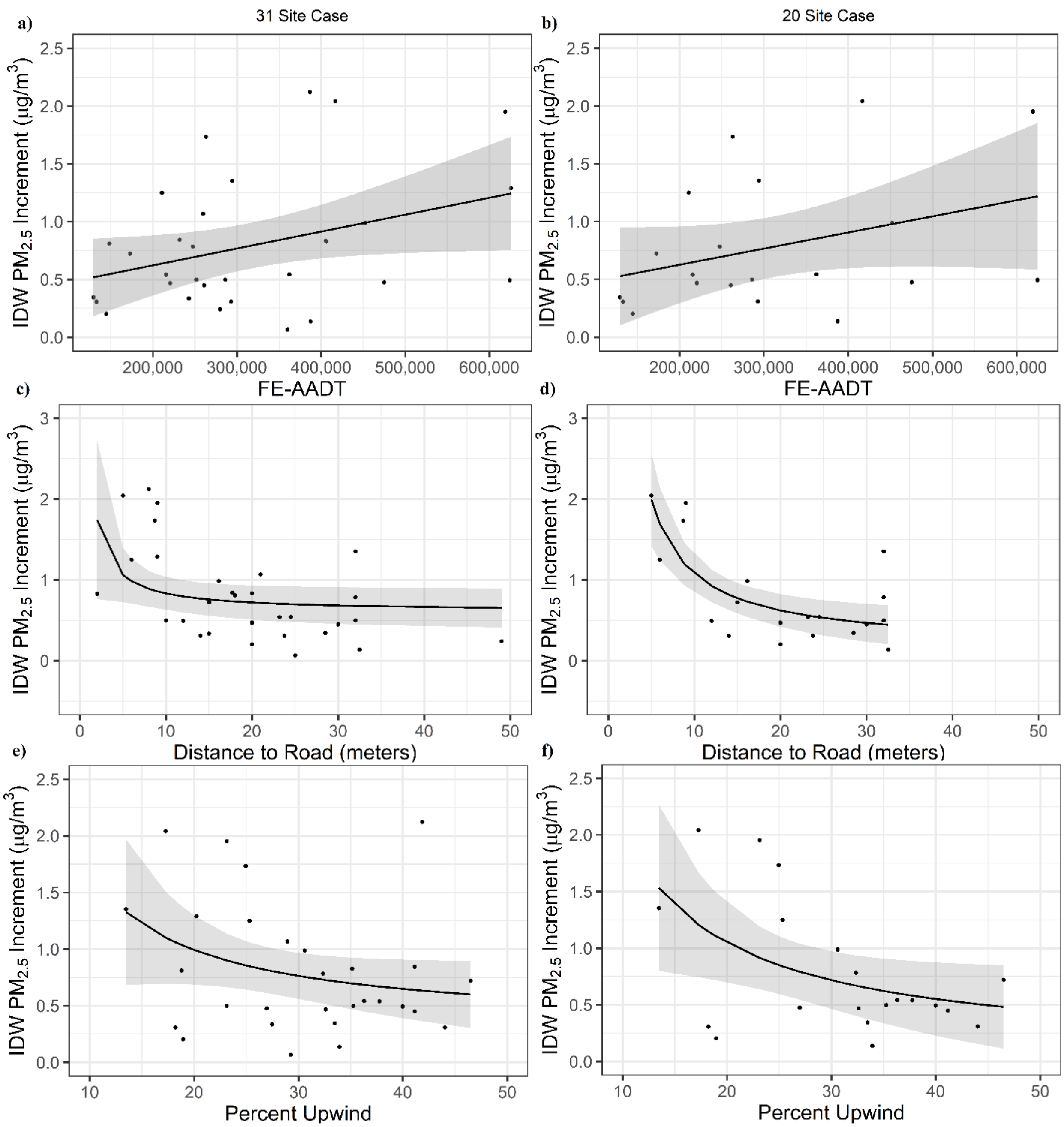


Figure 24. The relationship between the IDW $PM_{2.5}$ increment in comparison to FE-AADT (a and b), distance to road (c and d), and percent of time upwind (e and f). The initial case of 31 near-road sites is shown at left (a, c and e), and the focused case of 20 sites limiting confounding factors is shown at right (b, d and f). Regressions are shown in black, with the range of the standard error of the regression line shown in dark gray.

Table 5. The intercepts, slopes, p values and R^2 values for the regressions presented for six cases in Figure 24. For FE-AADT, a linear regression is used, of the form y = a • x + b. For distance to road and percent upwind, an inverse relationship is used, of the form y = a/x + b.

Regression Model	Distance to Road vs IDW Increment (20 Sites)	Distance to Road vs IDW Increment (31 Sites)	Percent Upwind vs IDW Increment (20 Sites)	Percent Upwind vs IDW Increment (31 Sites)	FE-AADT vs IDW Increment (20 Sites)	FE-AADT vs IDW Increment (31 Sites)
a	9.15	2.27	19.93	13.80	$1.40 \cdot 10^{-6}$	$1.46 \cdot 10^{-6}$
b	0.17	0.61	0.05	0.30	0.35	0.33
p value	0.000185[a]	0.0517 [a]	0.034 [a]	0.082	0.127	0.0483 [a]
R^2	0.34	0.30	0.25	0.08	0.12	0.13

[a] Statistically significant relationship at 95% or higher confidence.

In order to represent multiple explanatory variables at once, a GAM was used to predict the near-road IDW increment using distance to road, percent of time a site was upwind, and FE-AADT (the top three factors from Table 4). The model was run to optimize the restricted maximum likelihood estimation (REML), and each of the three predictor variables was given three degrees of freedom. The model was constructed for both the 31-site case and focused 20-site case. As before, the most important explanatory variables were distance to road, then percent upwind, and FE-AADT. For the 31 site case, the model had an adjusted R^2 value of 0.36, a modest improvement from the original R^2 value of 0.3 between the IDW increment and distance to road (shown in Table 5). For the focused case of 20 sites, distance to road had a p value equal to 0.00032, percent upwind had a p value equal to 0.15 and FE-AADT had a p value equal to 0.22. The model had an overall adjusted R^2 value of 0.63, predicting the majority of the variability in the IDW increment. Overall, the regression models shown in Figure 24, the correlations shown in Table 5, and the GAM model show a statistically significant correlation between distance to road, percent upwind, and the increment, and a modest correlation between FE-AADT and the increment.

4.6 Phase Four: Forecasted Background and Increments

Depending on the timeline for a given transportation project, the attainment status for the project area, and the relationship of the years to be modeled for the conformity analysis to the attainment deadlines, it may be appropriate to account for future changes in the estimate of the background concentration or the incremental near-road $PM_{2.5}$ contribution from a major road. This discussion highlights findings from TPF work that examined forecasting approaches to assist with background and increment calculations.

4.6.1 Future Changes in Background

Estimates of future-year background concentrations can be developed using Chemical Transport Models (CTM) combined with ambient concentrations, if model outputs are available for areas with monitoring sites considered for estimating background. The required attainment deadline for all nonattainment areas violating the 2012 $PM_{2.5}$ standard is 2021 (six years after the promulgation of the standard and area designations in 2015).[11] By 2021, areas in nonattainment as of 2015-2017 are expected to reduce concentrations to at least the levels of the NAAQS. Therefore, consideration of future-year background concentrations is an important topic to discuss during the interagency consultation process that guides conformity analyses, assuming areas are making the required progress toward meeting the attainment deadline.

Of the nine areas classified (as of 2018) as nonattainment (or maintenance) for $PM_{2.5}$ based on the 2012 annual standard (12 $\mu g/m^3$), six areas contained a near-road monitor in 2017 (**Table 6**). All of these areas are classified as moderate nonattainment. For some areas, the 2015-2017 Annual Design Value is below the NAAQS, although the area's official designation has not yet been updated to attainment or maintenance.

Table 7 summarizes the nonattainment or maintenance areas for the 2006 $PM_{2.5}$ 24-hour standard (35 $\mu g/m^3$) as well as the near-road monitors in those areas, attainment deadlines, and differences between 2017 concentrations and the NAAQS level. The attainment deadline for moderate nonattainment areas was December 2015, while the attainment deadline for serious nonattainment areas was December 2019;[12] the attainment deadlines in Table 7 for moderate nonattainment areas may no longer be accurate for areas that did not reach attainment by the deadline. For some areas, the 2015-2017 24-hour Design Value is below the NAAQS, but as of the time this work was completed, the official designation had not yet been updated to attainment or maintenance.

[11] See https://www.epa.gov/sites/production/files/2016-08/documents/pm25-sip-requirements-rule-webinar-august-16-2016.pdf.

[12] See https://www.epa.gov/pm-pollution/fine-particulate-naaqs-implementation-milestones.

Table 6. Nonattainment areas based on the 2012 annual $PM_{2.5}$ NAAQS; six of these areas had a near-road monitor in 2017.

Near-Road Monitor				Nonattainment Area			
CBSA	City	AQS Code	Target Road	Nonattainment Area	Attainment Deadline	2015-2017 Annual Design Value (µg/m³)[13]	% Above Annual Standard
Pittsburgh, PA	Wilkinsburg	42-003-1376	I-376	Allegheny County, PA	2021	13.0	8.3
Los Angeles-Long Beach-Anaheim, CA	Anaheim	06-059-0008	I-5	Los Angeles-South Coast Air Basin, CA	2021	14.7	22.5
Riverside-San Bernardino-Ontario, CA	Ontario	06-071-0026	I-10	Los Angeles-South Coast Air Basin, CA	2021	14.7	22.5
Fresno, CA	Fresno	06-019-2016	CA 99	San Joaquin Valley Air Basin, CA	2021	22.2	85.0
Bakersfield, CA	Bakersfield	Unknown**	CA 99	San Joaquin Valley Air Basin, CA	2021	22.2	85.0
Cleveland-Elyria, OH	Cleveland	39-035-0073	I-271	Cleveland, OH	2021	11.7	-2.5
Nonattainment Areas where a near-road monitor was not located in 2017							
				Delaware County, PA	2021	10.3	-14.2
				Imperial County, CA	2021	12.0	0.0
				Lebanon County, PA	2021	10.1	-15.8
				Plumas County, CA	2021	15.1	25.8
				West Silver Valley, ID	2021	12.4	3.3

**The Bakersfield near-road monitor is planned but not yet operational.

[13] https://www.epa.gov/air-trends/air-quality-design-values#report.

Table 7. Nonattainment areas based on the 2006 24-hour $PM_{2.5}$ NAAQS; 18 areas had a near-road monitor in 2017.

Near-Road Monitor				Nonattainment Area				
CBSA	City	AQS Code	Target Road	Nonattainment Area	Attainment Deadline	Designation Status	2015-2017 24-Hour Design Value (µg/m3)[14]	% Above 24-Hour Standard
Birmingham-Hoover, AL	Birmingham	01-073-2059	I-20	Birmingham, AL	2015	Maintenance	22	-37.1
Cleveland-Elyria, OH	Cleveland	39-035-0073	I-271	Cleveland-Akron-Lorain, OH	2015	Maintenance	25	-28.6
Detroit-Warren-Dearborn, MI	Detroit	26-163-0093	I-96	Detroit-Ann Arbor, MI	2015	Maintenance	28	-20.0
Detroit-Warren-Dearborn, MI	Livonia	26-163-0095	I-275	Detroit-Ann Arbor, MI	2015	Maintenance	28	-20.0
Fresno, CA	Fresno	06-019-2016	CA 99	San Joaquin Valley, CA	2019	Nonattainment	72	105.7
Bakersfield, CA	Bakersfield	Unknown**	CA 99	San Joaquin Valley, CA	2019	Nonattainment	72	105.7
Los Angeles-Long Beach-Anaheim, CA	Anaheim	06-059-0008	I-5	Los Angeles-South Coast Air Basin, CA	2019	Nonattainment	39	11.4
Riverside-San Bernardino-Ontario, CA	Ontario	06-071-0026	I-10	Los Angeles-South Coast Air Basin, CA	2019	Nonattainment	39	11.4
Milwaukee-Waukesha-West Allis, WI	Milwaukee	55-079-0056	I-94	Milwaukee-Racine, WI	2015	Maintenance	22	-37.1
New York-Newark-Jersey City, NY-NJ-PA	Fort Lee	34-003-0010	I-95/ US 1	New York-N. New Jersey-Long Island, NY-NJ-CT	2015	Maintenance	23	-34.3

[14] https://www.epa.gov/air-trends/air-quality-design-values#report

Near-Road Monitor				Nonattainment Area				
CBSA	City	AQS Code	Target Road	Nonattainment Area	Attainment Deadline	Designation Status	2015-2017 24-Hour Design Value (μg/m3)[14]	% Above 24-Hour Standard
New York-Newark-Jersey City, NY-NJ-PA	Queens	36-081-0125	I-495 (L.I.E.)	New York-N. New Jersey-Long Island, NY-NJ-CT	2015	Maintenance	23	-34.3
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	Philadelphia	42-101-0075	I-95	Philadelphia-Wilmington, PA-NJ-DE	2015	Maintenance	25	-28.6
Philadelphia-Camden-Wilmington, PA-NJ-DE-MD	Philadelphia	42-101-0076	I-76	Philadelphia-Wilmington, PA-NJ-DE	2015	Maintenance	25	-28.6
Pittsburgh, PA	Wilkinsburg	42-003-1376	I-376	Pittsburgh-Beaver Valley, PA	2015	Maintenance	24	-31.4
Sacramento--Roseville--Arden-Arcade, CA	Sacramento	06-067-0015	I-5	Sacramento, CA	2015	Nonattainment	34	-2.9
San Francisco-Oakland-Hayward, CA	Oakland	06-001-0012	I-880	San Francisco Bay Area, CA	2015	Nonattainment	35	0.0
San Francisco-Oakland-Hayward, CA	Berkeley	06-001-0013	I-80	San Francisco Bay Area, CA	2015	Nonattainment	35	0.0
San Jose-Sunnyvale-Santa Clara, CA	San Jose	06-085-0006	US 101	San Francisco Bay Area, CA	2015	Nonattainment	35	0.0

Near-Road Monitor				Nonattainment Area				
CBSA	City	AQS Code	Target Road	Nonattainment Area	Attainment Deadline	Designation Status	2015-2017 24-Hour Design Value (µg/m3)[14]	% Above 24-Hour Standard
Nonattainment Areas where a near-road monitor was not located in 2017								
				Allentown, PA	2015	Maintenance	24	-31.4
				Canton-Massillon, OH	2015	Maintenance	22	-37.1
				Charleston, WV	2015	Maintenance	17	-51.4
				Chico, CA	2015	Nonattainment	28	-20.0
				Fairbanks, AK	2019	Nonattainment	85	142.9
				Harrisburg-Lebanon-Carlisle-York, PA	2015	Maintenance	30	-14.3
				Imperial County, CA	2015	Nonattainment	31	-11.4
				Johnstown, PA	2015	Maintenance	25	-28.6
				Klamath Falls, OR	2015	Nonattainment	36	2.9
				Knoxville-Sevierville-La Follette, TN	2015	Maintenance	34	-2.9
				Lancaster, PA	2015	Maintenance	28	-20.0
				Liberty-Clairton, PA	2015	Nonattainment	37	5.7
				Logan, UT-ID	2015	Nonattainment	33	-5.7
				Nogales, AZ	2015	Nonattainment	28	-20.0
				Oakridge, OR	2015	Nonattainment	46	31.4
				Provo, UT	2019	Nonattainment	31	-11.4
				Salt Lake City, UT	2019	Nonattainment	37	5.7
				Steubenville-Weirton, OH-WV	2015	Maintenance	25	-28.6

Near-Road Monitor				Nonattainment Area				
CBSA	City	AQS Code	Target Road	Nonattainment Area	Attainment Deadline	Designation Status	2015-2017 24-Hour Design Value (µg/m3)[14]	% Above 24-Hour Standard
				Tacoma, WA	2015	Maintenance	31	-11.4
				West Central Pinal, AZ	2015	Nonattainment	32	-8.6
				Yuba City-Marysville, CA	2015	Maintenance	28	-20.0

**The Bakersfield near-road monitor is planned but not yet operational.

Interagency consultation participants could use the data in Tables 6 and 7 to discuss rates of reasonable further progress toward attainment, and estimation of background $PM_{2.5}$. For example, Allegheny County had a 2015-2017 annual $PM_{2.5}$ design value of 13 µg/m^3, with a requirement to attain 12 µg/m^3 by 2021. If interagency consultation determined that Allegheny County was on track to meet attainment, transportation project analyses could use 12 µg/m^3 as the assumed background concentration for project-level analysis years of 2021 or later, even in the absence of CTM-based data.

The material presented here does not assess the rate of progress or the ability of individual areas to reach attainment. Those factors are best addressed during interagency consultation; however, the material included here can inform interagency discussion about how to employ the NAAQS deadline, paired with a local understanding of rate of progress toward attainment, as a potential method of refining future-year background concentration estimates.

4.6.2 Future Changes in Roadway Contributions (Increments)

The Contribution of Exhaust Emissions to Traffic-Related $PM_{2.5}$

Traffic-related $PM_{2.5}$ emissions come from three sources: exhaust, re-suspended road dust, and brake and tire wear. **Table 8** shows results from three analyses about the relative contributions of $PM_{2.5}$ emissions from these sources. In a companion research effort to this one, also supported by the TPF, Craig et al., used the MOVES model to estimate site-specific emissions for two cases (Craig et al., 2019). For the case of Providence-0030, a target roadway with 186,300 AADT and 13.7% HDDV, the exhaust was found to contribute to 49% of traffic $PM_{2.5}$ emissions in 2015 and 2016. For the case of Indianapolis-0087, a target roadway with 189,760 AADT and 10.1% HDDV, the exhaust was found to contribute to 40% of traffic $PM_{2.5}$ emissions in 2016. A study by Jeong et al. (2019) examined the traffic emissions of a roadway in Toronto in 2016 with 400,000 AADT, using the chemical composition of $PM_{2.5}$ and source apportionment. They found that exhaust contributed to 65% of traffic-related $PM_{2.5}$ emissions, with a strong dependence on HDDV fraction (approximately 6-13% of the fleet mix). For comparison, a review article examining studies from roadways around the world found that exhaust emissions contributed to the majority of overall traffic $PM_{2.5}$ emissions in most of the published studies reviewed (Pant and Harrison, 2013a).

Table 8. Ratios of $PM_{2.5}$ traffic emissions by process from modeled studies of Providence-0030 and Indianapolis-0087, and a measurement campaign in Toronto.

Process	Craig et al. (2019) Modeled $PM_{2.5}$ Emissions (%): Providence	Craig et al. (2019) Modeled $PM_{2.5}$ Emissions (%): Indianapolis	Jeong et al. (2019) Measured $PM_{2.5}$ Emissions (%): Toronto
Running Exhaust	49	40	65
Road Dust	44	53	13
Brake and Tire Wear	7	7	22
Total	**100**	**100**	**100**

Projections of Exhaust Emissions from 2017 to 2040

The projected exhaust emissions for U.S. vehicles for the calendar years 2017 to 2040 are shown in Table 9. Emissions in grams per mile per average vehicle are shown for HDDVs, light-duty vehicles (LDVs), and an average vehicle in a vehicle fleet composed of 8% HDDVs and 92% LDVs. The 8% HDDV case is a weighted average using 8% HDDV emission rate and 92% LDV emission rate. National average emission estimates are presented using the MOVES 2014 model developed by the EPA, and for San Francisco using the EMFAC 2017 model, developed by the California Air Resources Board (CARB). The percent change of the emissions reductions relative to the year 2017 are shown in Table 10, using data from Table 9. A decrease of exhaust emissions is shown in all cases for both vehicle types, due to fleet turnover. For example, the exhaust emissions of a roadway with 8% HDDV in San Francisco are projected to decrease by 88% by 2040 using EMFAC, assuming constant AADT and a constant HDDV fraction of 8%. The specific change in exhaust emissions for a given roadway will depend on changes in the regional vehicle fleet and traffic activity over time.

Table 9. Projected exhaust-only $PM_{2.5}$ emissions of vehicles for the calendar years 2017-2040. Emissions are shown in grams per mile per average vehicle, for HDDV, LDV, and a fleet mix with 8% HDDV and 92% LDV. Emissions are shown for a national average using MOVES, and for San Francisco (SF) using EMFAC.

Year	MOVES HDDV	MOVES LDV	MOVES HDDV 8%	EMFAC SF HDDV	EMFAC SF LDV	EMFAC SF HDDV 8%
2017	0.22042	0.00807	0.02506	0.12152	0.00216	0.01171
2018	0.19095	0.00749	0.02216	0.09684	0.00218	0.00976
2019	0.16631	0.00698	0.01973	0.07686	0.00220	0.00818
2020	0.14486	0.00656	0.01762	0.05888	0.00217	0.00670
2025	0.07571	0.00497	0.01063	0.00902	0.00172	0.00231
2030	0.04051	0.00397	0.00690	0.00857	0.00130	0.00188
2035	0.02778	0.00341	0.00536	0.00830	0.00100	0.00158
2040	0.02579	0.00315	0.00497	0.00814	0.00084	0.00143

Table 10. Projected change of exhaust-only $PM_{2.5}$ emissions of vehicles for the 2018-2040 calendar years, relative to the baseline year of 2017. The percent of 2017 emissions from Table 9 is shown for heavy-duty vehicles (HDDV), light-duty vehicles (LDV), and a fleet mix with 8% HDDV and 92% LDV. Emissions are shown for a national average modeled using MOVES, and for San Francisco (SF) using EMFAC.

Year	MOVES HDDV	MOVES LDV	MOVE HDDV 8%	EMFAC SF HDDV	EMFAC SF LDV	EMFAC SF HDDV 8%
2017	100	100	100	100	100	100
2018	87	93	88	80	101	83
2019	75	87	79	63	102	70
2020	66	81	70	48	100	57
2025	34	62	42	7	80	20
2030	18	49	28	7	60	16
2035	13	42	21	7	46	14
2040	12	39	20	7	39	12

Projected Traffic Contribution to $PM_{2.5}$

The upper bound of traffic-related $PM_{2.5}$ represented by the increment for near-road sites in 2017 can be combined with the projected change in exhaust emissions (Table 10), and an assumed fraction of traffic-related $PM_{2.5}$ due to exhaust, to forecast the upper bound of traffic-related $PM_{2.5}$ in the coming decades. In 2017, the upper bound of the observed annual average $PM_{2.5}$ traffic impact was 2.04 ± 0.16 μg/m^3, with an upper bound of 1.44 ± 0.17 μg/m^3, for sites with monitors 10 meters or more from the roadway (Figure 23). Using an example conservative case, based on the modeled Indianapolis-0087 study, we can take the fraction of 40% to illustrate a lower end estimate of the contribution of exhaust emissions to total traffic-related emissions. The change in the $PM_{2.5}$ traffic impact for a given project can then be calculated using the equation:

$PM_{2.5}$ increment (future year) μg/m^3 = 1.224 + 0.816 • [Percent change from 2017 to future year]

Where 1.224 μg/m^3 is the 60% of traffic-driven $PM_{2.5}$ that is attributed to non-exhaust factors and 0.816 μg/m^3 is the 40% of traffic-driven $PM_{2.5}$ that is attributed to exhaust, which is forecast to decrease. Likewise, for distances 10 meters or more from the roadway, we have the equation:

$PM_{2.5}$ increment, 10 meters or greater from roadway (future year) μg/m^3 = 0.864 + 0.576 • [Percent change from 2017 to future year]

The projected change of the $PM_{2.5}$ increment is illustrated in the following examples. These examples are based on the formulas above, use the MOVES national average emissions estimate, and assume a project with constant vehicle speeds and HDDV fraction of 8% for all years. The exhaust emissions are projected to be 42% of 2017 emissions by 2025 and 20% of 2017 emissions by 2040. Using the equations above, the $PM_{2.5}$ impact of a highway would fall from 2.04 ± 0.16 μg/m^3 in 2017 to 1.57 ± 0.16 μg/m^3 by 2025, and to 1.39 ± 0.16 μg/m^3 by 2040, if AADT, fleet mix, and vehicle speeds remain constant. Following the same method, for the domain greater than or equal to 10 meters from the roadway, the $PM_{2.5}$ impact of a highway would fall from 1.44 ± 0.17 μg/m^3 in 2017 to 1.11 ± 0.17 μg/m^3 by 2025 and 0.98 ± 0.17 μg/m^3 by 2040. Using the EMFAC modeled emissions of San Francisco, a roadway with constant AADT, and 8% HDDV fleet mix could expect to have its $PM_{2.5}$ impact fall from 2.04 ± 0.16 μg/m^3 in 2017 to 1.32 ± 0.16 μg/m^3 by 2040. This procedure only accounts for changing exhaust emissions caused by fleet turnover, and does not account for the full range of factors in forecasting incremental $PM_{2.5}$ impacts that may be considered during interagency consultation.

5. Screening Insights Related to POAQC for $PM_{2.5}$ Hot-Spot Analyses

The TPF completed two phases of work to help inform interagency consultation regarding POAQC determinations for $PM_{2.5}$ hot-spot analyses. The first phase of work used the EMFAC and MOVES emissions modeling tools to forecast how fleet changes will effect on-road vehicle emissions. The second phase of work extended phase one findings by applying forecasted fleet emissions changes to anticipated near-road increments. Highlights of both work efforts are presented here.

5.1 EMFAC and MOVES Modeling to Forecast Fleet Emissions Changes

5.1.1 Overview

Because the steps involved in a POAQC analysis involve data collection efforts and complex modeling tasks, a complete analysis may take several months. Moreover, the proposed project and its build alternatives may be revised during the analysis process, requiring additional data collection and modeling work. Therefore, transportation project analysts need information to help identify projects that are not likely to be POAQCs. To help provide such information, the team performed scenario analyses for a hypothetical transportation project, which was roughly based on a hypothetical project developed by the EPA for PM hot-spot analysis training purposes. This hypothetical project features a new freeway with four mixed-flow lanes in each direction. For this work, traffic activity data developed by the EPA was adjusted to match the POAQC example of 125,000 AADT and 8% diesel truck traffic. From this starting point, the team estimated PM_{10} and $PM_{2.5}$ emissions for the hypothetical project for a 2006 base year (to match the year of EPA's rulemaking), for additional analysis years ranging from 2007 to 2035, and for a range of vehicle fleet compositions (i.e., percentage of diesel trucks).

The team then compared scenario-specific emission results with the 2006 baseline results to evaluate the impact of fleet turnover (i.e., the introduction of newer, cleaner vehicles into the fleet over time) and truck percentages on potential project-level air quality impacts. These analyses were designed to provide answers to questions such as:

- Given the 2006 rulemaking year, what PM_{10} and $PM_{2.5}$ emission levels might be expected of a 2006 project with AADT of 125,000 and at least 8% diesel truck traffic?
- How would a 2030 project with AADT of 125,000 and 8% trucks compare to the 2006 project in terms of PM emission levels and concentrations?

- What is the influence of diesel truck percentage on project-level air quality impacts, and how might fleet turnover effects offset those impacts?
- What is the contribution of non-exhaust emissions processes (e.g., re-entrained dust, tire wear, brake wear) to project-level air quality impacts, and how do those contributions vary for different analysis years?

Scenario analyses were performed using the MOVES2014 (database version 20141021) and EMFAC2014 (version 1.0.1) models to quantify exhaust, tire wear, and brake wear emissions, and methods from the EPA's AP-42 emission factors handbook (U.S. Environmental Protection Agency, 2007) were used to estimate emissions from re-entrained road dust.

5.1.2 Results: 2006 Baseline Emissions

For PM_{10}, total 2006 MOVES- and EMFAC-based emissions estimates for the hypothetical project are very similar, totaling 20.0 and 19.1 kg/day, respectively. For both sets of emissions, tire wear and re-entrained road dust emissions are about equal, with approximately half of the total PM_{10} emissions being associated with the AP-42-based road dust emissions estimates (Figure 25). However, MOVES exhaust emissions estimates for 2006 are about 70% higher than the EMFAC-based estimates, and EMFAC brake wear estimates are 2.5 times higher than the MOVES-based estimates.

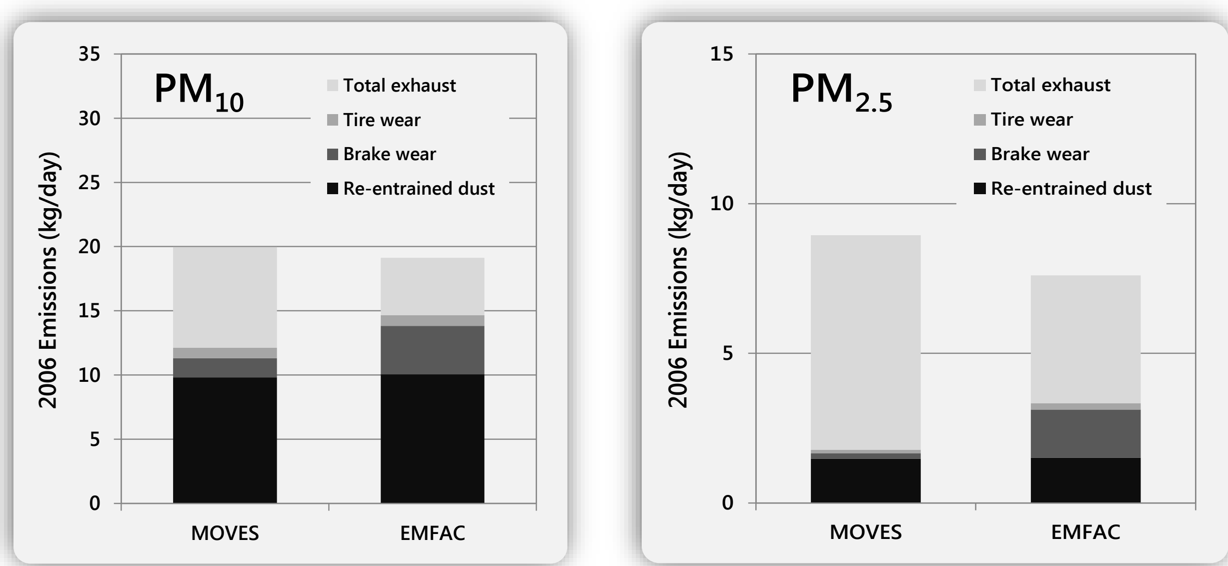


Figure 25. Baseline PM_{10} (left) and $PM_{2.5}$ (right) emissions for a hypothetical 2006 freeway project with an AADT of 125,000 vehicles, 8% of which are diesel trucks.

For $PM_{2.5}$, total MOVES- and EMFAC-based emissions for the 2006 hypothetical project are 8.9 and 7.6 kg/day, respectively. Because MOVES and EMFAC do not calculate re-entrained road dust emissions directly, the team used the AP-42 method to calculate emissions from this process, which were then added to MOVES and EMFAC emissions estimates for exhaust, tire wear, and brake wear. Road dust emissions would typically not be considered for a $PM_{2.5}$ hot-spot analysis unless road dust represented a significant $PM_{2.5}$ source in the project region. MOVES- and EMFAC-based $PM_{2.5}$ emissions without road dust total 7.5 and 6.1 kg/day, respectively. For subsequent analyses shown in this paper, $PM_{2.5}$ emissions will include only exhaust, brake wear, and tire wear components.

For $PM_{2.5}$, MOVES produces higher exhaust emissions than EMFAC, while EMFAC produces much higher brake wear emissions. In MOVES, a $PM_{10}/PM_{2.5}$ brake wear ratio of 8 is assumed, while EMFAC uses a $PM_{10}/PM_{2.5}$ ratio of 2.3; this difference results in EMFAC-based $PM_{2.5}$ brake wear emissions that are eight times higher than the MOVES-based estimate.

The re-entrained road dust emission levels shown in Figure 25 can vary by region, even if projects have similar traffic activity, because silt loading values are region-specific. However, these 2006 emission levels provide a useful baseline illustration for understanding impacts associated with the EPA's hypothetical highway project with 125,000 AADT and 10,000 diesel trucks, as well as evaluating traffic activity levels required to produce similar project-level emissions in other years.

5.1.3 Fleet Turnover Scenarios

Beyond 2006, vehicle exhaust emissions decrease significantly as a result of federal and California emissions standards. To examine the impacts of fleet turnover, the team estimated PM_{10} and $PM_{2.5}$ emissions for several analysis years from 2010 to 2035, holding the vehicle fleet constant at 125,000 AADT and 8% diesel trucks. For PM_{10}, MOVES-based emissions estimates for the hypothetical project decrease from 20.0 kg/day in 2006 to 12.5 kg/day in 2035, a reduction of about 37%. EMFAC-based PM_{10} estimates decrease from 19.1 kg/day to 13.6 kg/day, a reduction of about 29%. PM_{10} emissions reductions are associated with the exhaust portion of the emissions inventory, with emissions for tire wear, brake wear, and re-entrained road dust remaining nearly constant across all analysis years (**Figure 26**). For both the MOVES and EMFAC models, tire wear and brake wear emission rates change little over time, so the emissions from these processes do not decrease with fleet turnover, as is the case with exhaust emissions.

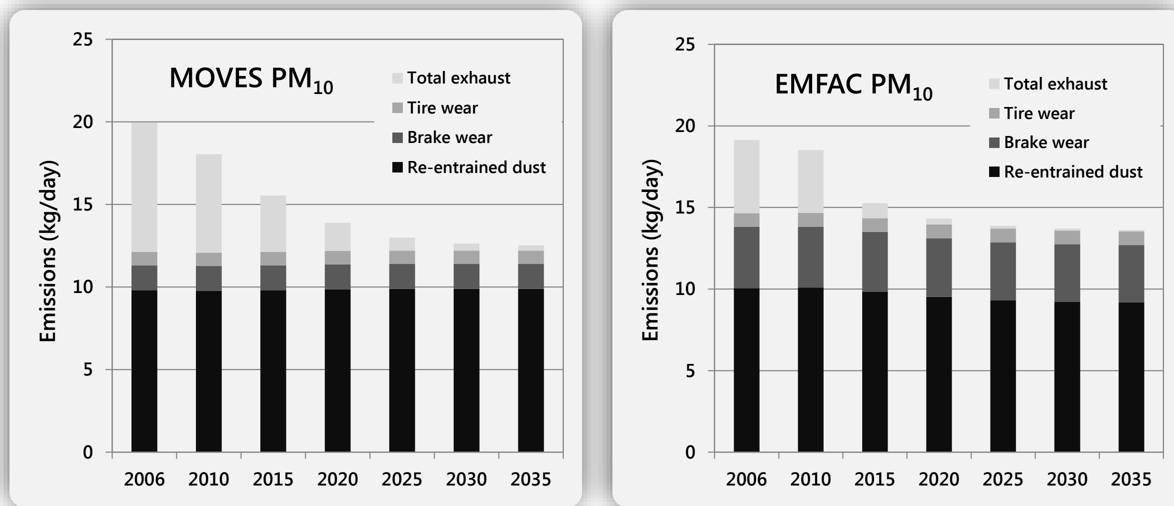


Figure 26. PM_{10} emissions for a hypothetical freeway project with an AADT of 125,000 vehicles, 8% of which are diesel trucks.

As a result of these trends, the contribution of the non-exhaust processes increases sharply over time, rising from 61% in 2006 to 97% in 2035 for MOVES-based estimates. Notably, the contribution of re-entrained road dust alone rises from 49% in 2006 to 79% in 2035. For EMFAC-based estimates, a sharp decrease in exhaust emissions occurs between 2010 and 2015 due to the impact of California diesel regulations. After 2015, further fleet turnover benefits are minimal, and project-level emissions remain nearly constant. For the MOVES-based estimates, decreases in exhaust emissions are more gradual over time; however, project-level PM_{10} emissions change little beyond 2020. These findings suggest that, for PM_{10}, fleet turnover benefits are largely limited to near-term years, and project-level emissions are increasingly dominated by non-exhaust processes (especially re-entrained road dust) over time.

For $PM_{2.5}$, re-entrained road dust is less frequently considered, and project-level emissions are more influenced by exhaust emissions than is the case with PM_{10}. MOVES-based $PM_{2.5}$ emissions estimates for the hypothetical project are cut approximately in half between 2006 and 2015 and are reduced by 92% between 2006 and 2035 (decreasing from 7.5 kg/day to 0.6 kg/day). EMFAC-based $PM_{2.5}$ estimates are also cut approximately in half between 2006 and 2015 and are reduced by about 70% between 2006 and 2035 (decreasing from 6.1 kg/day to 1.8 kg/day). EMFAC-based brake wear $PM_{2.5}$ emission estimates are consistently about eight times higher than MOVES-based estimates, limiting the overall reduction in project-level emissions (Figure 27). In addition, the contribution of brake wear and tire wear to the overall EMFAC-based $PM_{2.5}$ inventory rises from 30% in 2006 to 95% in 2035. For MOVES, the increase in the contribution of these processes to the overall inventory is less pronounced but also significant, rising from 4% in 2006 to 52% in 2035.

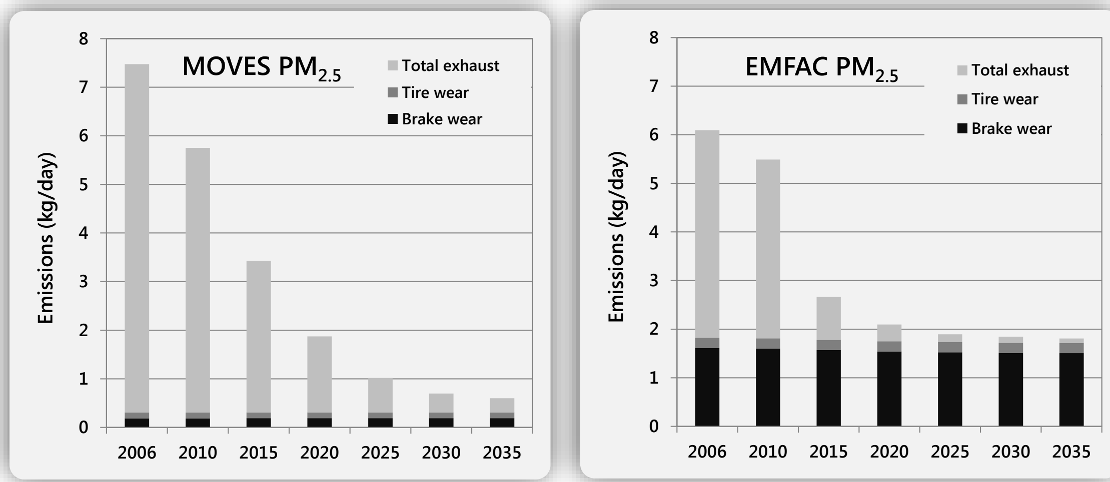


Figure 27. $PM_{2.5}$ emissions for a hypothetical freeway project with an AADT of 125,000 vehicles, 8% of which are diesel trucks.

As was the case for PM_{10}, EMFAC-based exhaust $PM_{2.5}$ emissions decrease sharply between 2010 and 2015, resulting in project-level emissions that remain nearly constant after 2015. For the MOVES-based estimates, decreases in exhaust emissions are more gradual over time, and significant fleet turnover benefits are observed in 2020 and 2025. These findings suggest that, for projects outside California, fleet turnover results in sharp $PM_{2.5}$ reductions over time for the hypothetical project with 125,000 AADT and 8% trucks. However, these fleet turnover benefits are somewhat limited for California projects due to the high brake wear emissions estimates produced by EMFAC and the modest decreases in exhaust emissions that occur after 2015.

Another important finding related to these PM_{10} and $PM_{2.5}$ emissions results is that the most uncertain aspects of the emissions inventories become more important over time. Although exhaust emissions have been researched extensively through engine-testing programs, relatively little research has focused on PM emissions from re-entrained road dust, tire wear, and brake wear. Tire and brake wear emission rates in the MOVES and EMFAC models are based on data from two published studies (Garg et al., 2000; Sanders et al., 2003), and the large differences in brake wear emissions estimates between the two models highlight the uncertainty associated with these estimates.

5.1.4 Increased AADT

The next set of scenarios modeled increased overall AADT while holding the truck percentage constant at 8%. For analysis years 2006 to 2035, the team evaluated emissions for overall traffic volumes ranging from 125,000 to 250,000 AADT, with truck volumes ranging from 10,000 (8% of 125,000) to 20,000 (8% of 250,000). For each scenario evaluated, PM_{10} emissions calculations include re-entrained road dust emissions, while $PM_{2.5}$ emissions do not include road dust. For a given fleet mix and analysis year, and constant travel speeds, a linear relationship exists between traffic volumes and emissions. Because of fleet turnover effects, producing emission levels equivalent to 2006 requires higher traffic volumes in later years. To examine this effect, the team held the truck percentage constant at 8% and calculated the overall AADT required to generate emissions totals in later years that are equivalent to the 2006 baseline emissions.

Producing project-level PM_{10} emissions equivalent to those from the 2006 analysis year in 2020 would require traffic volumes of 180,000 vehicles for MOVES-based analyses and 167,000 vehicles for EMFAC-based analyses (**Figure 28**). By 2035, traffic volumes of 200,000 for MOVES-based analyses and 175,000 for EMFAC-based analyses would be required to match 2006 emission levels, increases of 60% and 40%, respectively.

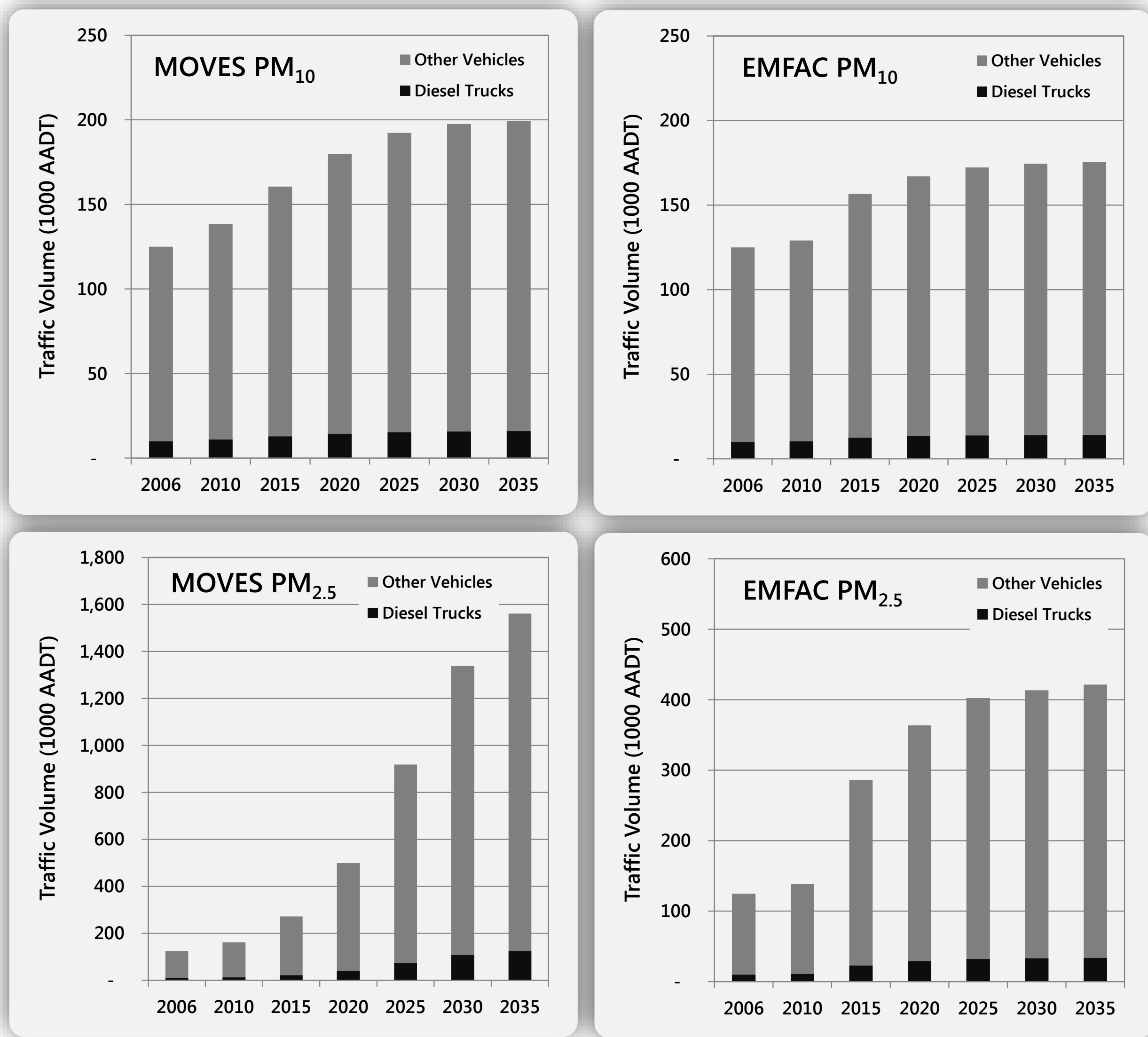


Figure 28. Projected traffic volumes needed to produce 2006-equivalent emissions (Scales are different for MOVES- and EMFAC-estimated $PM_{2.5}$ emissions).

For $PM_{2.5}$ emissions, which are dominated by exhaust emissions and, therefore, impacted to a greater extent by fleet turnover, the changes in traffic volumes are even more extreme. By 2020, MOVES-based $PM_{2.5}$ estimates would require an AADT of 500,000 vehicles to reach 2006 emission levels, while EMFAC-based $PM_{2.5}$ estimates would require an AADT of 360,000 to reach 2006 emission levels. Note that this analysis illustrates traffic volumes required to produce emissions equivalent to the 2006 baseline scenario, holding travel speeds constant. Actual project analyses would adjust traffic speeds to appropriately reflect roadway capacity and traffic volume changes specific to the project's characteristics.

By 2035, reaching 2006 emission levels requires an AADT of 1.6 million vehicles for MOVES-based analyses; this number is almost 13 times higher than the baseline volume of 125,000 vehicles. For EMFAC-based $PM_{2.5}$ estimates, a 2035 AADT of approximately 420,000 vehicles would be required to reach 2006 emission levels; this number is more than three times higher than the baseline volume. The differences in MOVES- and EMFAC-based traffic volumes are primarily driven by the higher brake wear emissions estimated by EMFAC, as brake wear emissions are not impacted by fleet turnover and change little by analysis year.

5.1.5 Increased Diesel Truck Traffic

The final set of scenarios modeled increased truck percentage for the baseline traffic volume of 125,000 vehicles. In these analysis scenarios, the truck category includes combination short-haul and long-haul trucks (for MOVES modeling) and medium heavy-duty and heavy heavy-duty trucks (for EMFAC modeling). For analysis years 2006, 2015, 2025, and 2035, the team evaluated emissions for truck percentages of 8%, 20%, and 40%. MOVES- and EMFAC-based results are calculated for PM_{10} (**Figure 29**). Because increased truck volumes significantly impact both exhaust and re-entrained road dust emissions, the overall PM_{10} emissions inventories increase sharply as the truck percentage increases. Increasing the truck percentage from 8% to 40% results in MOVES- predicted PM_{10} emissions increasing by a factor of 3 across all future projected years. In addition, for both MOVES and EMFAC, modeling a truck percentage of even 20% results in total PM_{10} emissions across all analysis years that are higher than the 2006 baseline of 20 kg/day (based on 8% trucks in 2006).

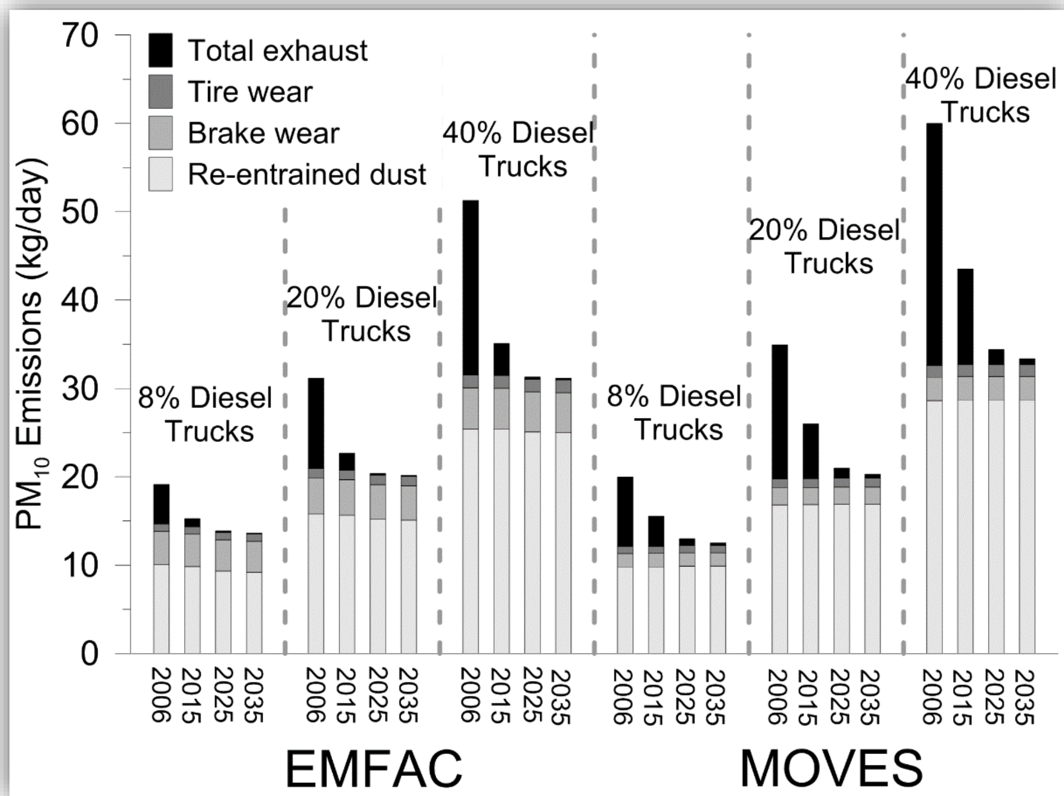


Figure 29. Projected PM_{10} emissions changes associated with increased truck volume in future year scenarios.

For $PM_{2.5}$, in contrast, the absence of re-entrained road dust emissions and the considerable decrease in exhaust emissions over time offsets the impact of increased truck traffic volumes. For example, in 2015, total MOVES-based $PM_{2.5}$ emissions for the 20% truck scenario are less than the 2006 baseline $PM_{2.5}$ emissions with 8% trucks. For the 40% truck scenario, by 2020, total MOVES-based $PM_{2.5}$ emissions are less than the 2006 baseline emissions (Figure 30). Similarly, for the EMFAC-based $PM_{2.5}$ results, by 2015, emissions for both the 20% and 40% truck scenarios are less than the 2006 baseline. These findings indicate that a current-year (2015) California transportation project with 125,000 AADT and 40% trucks has lower $PM_{2.5}$ impacts than the hypothetical 2006 POAQC with 125,000 AADT and 8% trucks.

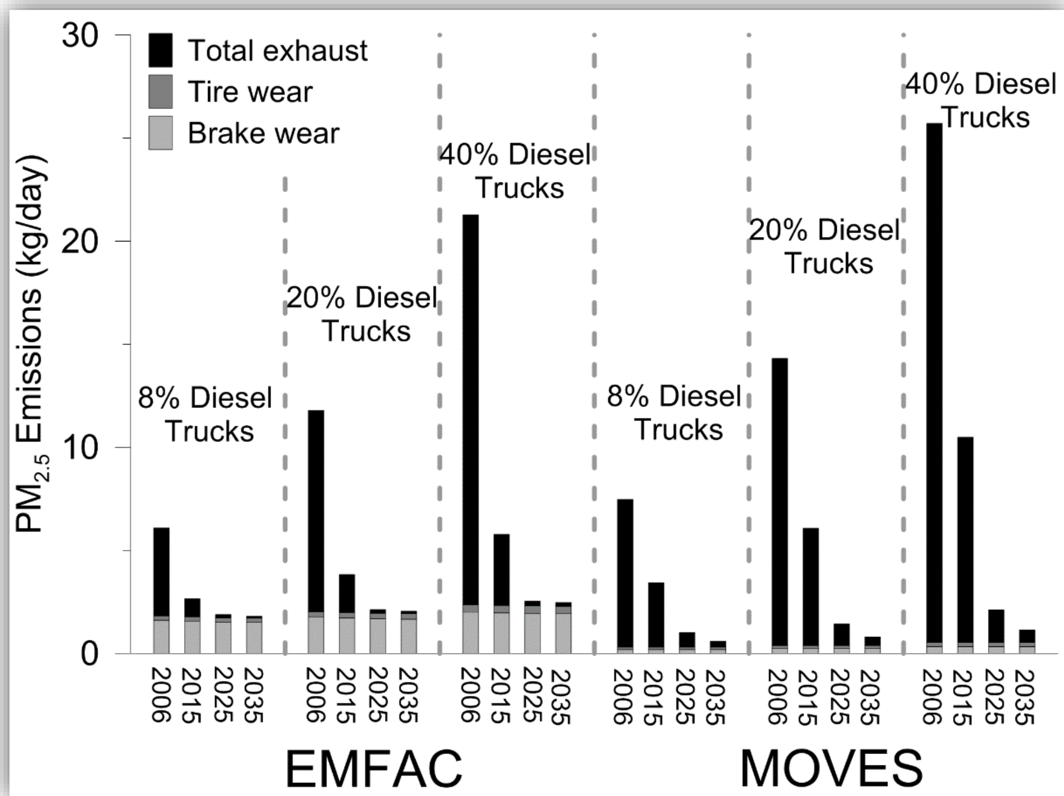


Figure 30. Projected $PM_{2.5}$ emissions changes associated with increased truck volume in future year scenarios.

5.1.6 Discussion

The results of this study yield a number of key insights that may be helpful in POAQC determinations. First, the results highlight the importance of project location and relevant NAAQS standards to POAQC determinations. For projects in PM_{10} nonattainment areas, re-entrained road dust emissions (and, to a lesser extent, tire wear and brake wear emissions) increasingly dominate project-level inventories over time, and these emissions vary little by analysis year. Therefore, fleet turnover effects and congestion relief will neither provide significant emissions reductions over time, nor allow build scenarios to compare favorably with no-build scenarios. On a national scale, the trend in monitored PM_{10} concentrations has shown an overall decrease during the last 25 years, (U.S. Environmental Protection Agency, 2015b) and additional research is needed to assess how PM_{10} levels in the near-road environment are changing compared to national and regional levels.

For projects in $PM_{2.5}$ nonattainment areas, the picture is very different. Exhaust emissions dominate the project-level inventory (especially for MOVES-based analyses), and for the year 2015, impacts from a highway project with 125,000 AADT and 8% trucks are already approximately 50% less than impacts from such a project in 2006. In addition, fleet turnover means that, by 2020 and beyond, even projects with 125,000 AADT and 40% (50,000) diesel trucks are likely to produce $PM_{2.5}$ emissions equivalent to or less than the emissions from the 2006 baseline project with 10,000 trucks. Additional research is required to assess the impact of congestion on $PM_{2.5}$ emissions at the project level, as changes in emissions with varying average travel speeds were not considered in this analysis.

Another important insight is the linear relationship between traffic activity and PM emissions (assuming a consistent vehicle fleet and travel speeds). This linear relationship, combined with the various scenarios analyzed for this study, may allow project analysts to quickly estimate PM impacts associated with their project and compare those impacts with the 2006 hypothetical project. For example, suppose an analyst is reviewing a highway project with a 2025 analysis year in a $PM_{2.5}$ nonattainment area. The analyst could use the data illustrated in the figures above to qualitatively assess where the project's traffic volumes for diesel and other vehicles are comparable, and whether the 2025 volumes would be expected to result in emissions substantially less than the 2006 baseline case developed here. The analyst could then use such a comparison during the conformity interagency consultation process to help determine whether the project was a POAQC that required the more rigorous evaluation steps established by the EPA.

Another important insight for POAQC determinations is that current emissions modeling techniques have limitations with regard to estimates of emissions from re-entrained road dust, tire wear, and brake wear; with time, these processes will become increasingly important at the project level. For example, our modeling scenarios identified substantial differences in brake wear emissions estimates between MOVES and EMFAC. For this emissions process, limited published data are available for deriving emission factors, and the two models rely on different assumptions regarding vehicle characteristics, roadway conditions, and PM_{10} to $PM_{2.5}$ ratios. To properly characterize brake wear emissions, emission factors must properly reflect vehicle activities (e.g., increased brake emission rates at higher travel speeds) and the impact of various brake materials on dust emission rates per brake application. Additional research is needed to refine the understanding of these emissions processes and to improve associated modeling techniques.

5.2 Increment-Based POAQC Insights

The results from the near-road monitoring sites showing the range of traffic-related $PM_{2.5}$ impacts (Figure 23) can be synthesized with our understanding of the projected change of vehicle emissions in the coming decades to yield insights pertinent to POAQC determinations. In Section 4.6.2 we examined (1) the relative contribution of exhaust and non-exhaust emissions to traffic-related $PM_{2.5}$, (2) how exhaust emissions of $PM_{2.5}$ will change over time with fleet turnover, and (3) how those findings could be used to forecast how the increments identified in Section 4.5.2 will change in the coming decades. This discussion presents how those findings can assist interagency consultation regarding POAQC determinations.

POAQC determinations are considered during interagency consultation on a case-by-case basis for proposed projects. Under the transportation conformity requirements, Section 93.123(b)(1)(i), consultation partners evaluate whether projects involve a significant number or increase in diesel vehicles. The analysis procedures presented here can dynamically link proposed projects to anticipated incremental air quality impacts over time, thus helping to define on a case-by-case basis whether projects involve significant changes in diesel vehicles and merit being identified a POAQC. The assessment process can reflect site-specific characteristics such as projected changes of fleet-average exhaust emissions, HDDV fraction, AADT, the distance between the roadway edge and sensitive receptor locations, and the persistence of prevailing wind direction.

The upper bound of the 2017 $PM_{2.5}$ increment was 2.04 ± 0.16 µg/m^3 for the focused case of 20 near-road sites, where all sites noted with possible confounding factors have been removed (Figure 23). The upper bound of the increment for near-road sites with monitors sited 10 meters or more from the roadway was 1.44 ± 0.17 µg/m^3. The implications for designating proposed roadway projects as POAQC are shown next, followed by the forecasts of the upper bound of traffic-related $PM_{2.5}$ for the coming decades using projected changes in emissions.

In 2006, EPA issued a final rule entitled "$PM_{2.5}$ and PM_{10} Hot-Spot Analysis in Project-Level Transportation Determinations for the New $PM_{2.5}$ and Existing PM_{10} National Ambient Air Quality Standards" (PM hot-spot rule).[15] The 2006 final rule states that if a project has a significant number of diesel vehicles, it is determined to be a POAQC; the preamble to the rule gives the example of a highway with an AADT greater than 125,000 and a diesel truck traffic of 8% or more. Our study presents results based on 2017 data, and a procedure to represent fleet turnover in the coming decades. The results presented here can inform interagency consultation on whether proposed highway projects have a significant number of diesel vehicles and are POAQC for conformity purposes. Suggested analysis steps include:

1. Compare proposed project characteristics to roadway characteristics evaluated in this study. If the proposed project's characteristics are covered by the data used in this study, proceed to the remaining steps.

2. Establish the current project site's background $PM_{2.5}$ concentration, and determine whether a "buffer" exists. For example, if the annual average $PM_{2.5}$ background concentration is 9 µg/m^3, and the NAAQS is 12 µg/m^3, the buffer is:
   12 µg/m^3 - 9 µg/m^3 = 3 µg/m^3.

3. If estimates are available of how background $PM_{2.5}$ is forecast to change for the conformity analysis years, calculate the adjusted buffer for those years (see Section 4.6.1).

4. Determine what receptors are of interest for the project and their distance to the roadway.

[15] 40 CFR 93.123(b)(1).

5. Identify the current maximum increment applicable to the project based on closest receptor: the current maximum increment is 2.04 ± 0.16 $\mu g/m^3$ within 10 meters of the roadway, 1.44 ± 0.17 $\mu g/m^3$ beyond 10 meters.

6. Forecast how the increment will change for the conformity analysis years, considering the projected fleet mix in future years (see Section 5.2).

7. If the increment is less than the buffer for the analysis years, and if no other unique project characteristics are expected to increase the $PM_{2.5}$ increment, it can be reasonably determined that the number of diesel vehicles is not significant and the project is not a POAQC.

8. If the increment is greater than the buffer, examine the forecasted project characteristics for FE-AADT, percent upwind, and receptor distance from roadway. Based on the statistically significant relationships shown in Figure 24, assess whether the maximum expected increment should be adjusted downward. Determine if any other known project characteristics warrant additional increment adjustments. Redo previous analysis step.

6. Comparison of Monitored and Modeled $PM_{2.5}$ Concentrations

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.

6.1 Indianapolis Site Description

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 31**) 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. Table 11 provides details about the site characteristics.

Figure 31. 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 11. 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, 2017).

6.2 Indianapolis 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 12). 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.

Table 12. 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

6.3 Indianapolis $PM_{2.5}$ 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 32**). 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, 2015a). The nearby sites had FRM monitors with daily measurement frequency at Washington Park and 1-in-3 day frequency at E. Michigan St.

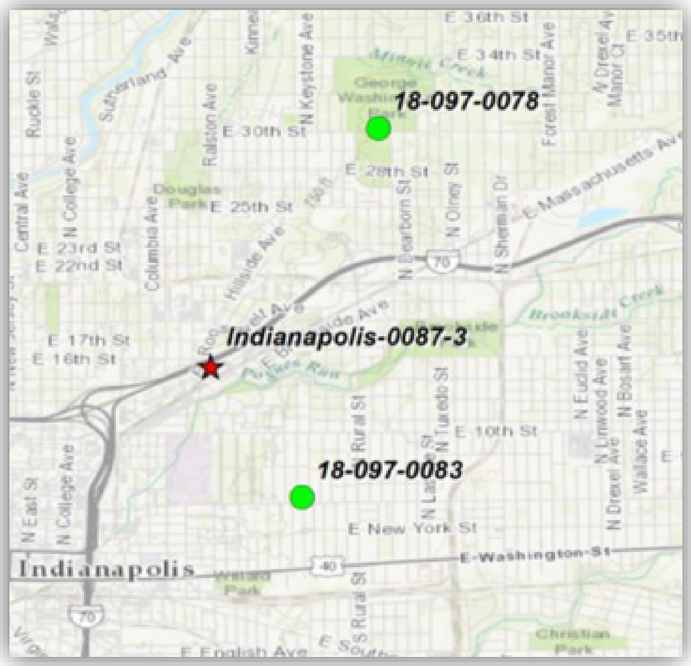


Figure 32. Location of the Indianapolis near-road $PM_{2.5}$ FRM and FEM monitors (star) and nearby $PM_{2.5}$ FRM monitors (green dots) at Washington Park (AQS ID 18-097-0078) and E. Michigan St. (AQS ID 18-097-0083).

6.4 Providence Analysis: Site Description

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 33) 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 13. The project area (yellow circle in Figure 33) 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.

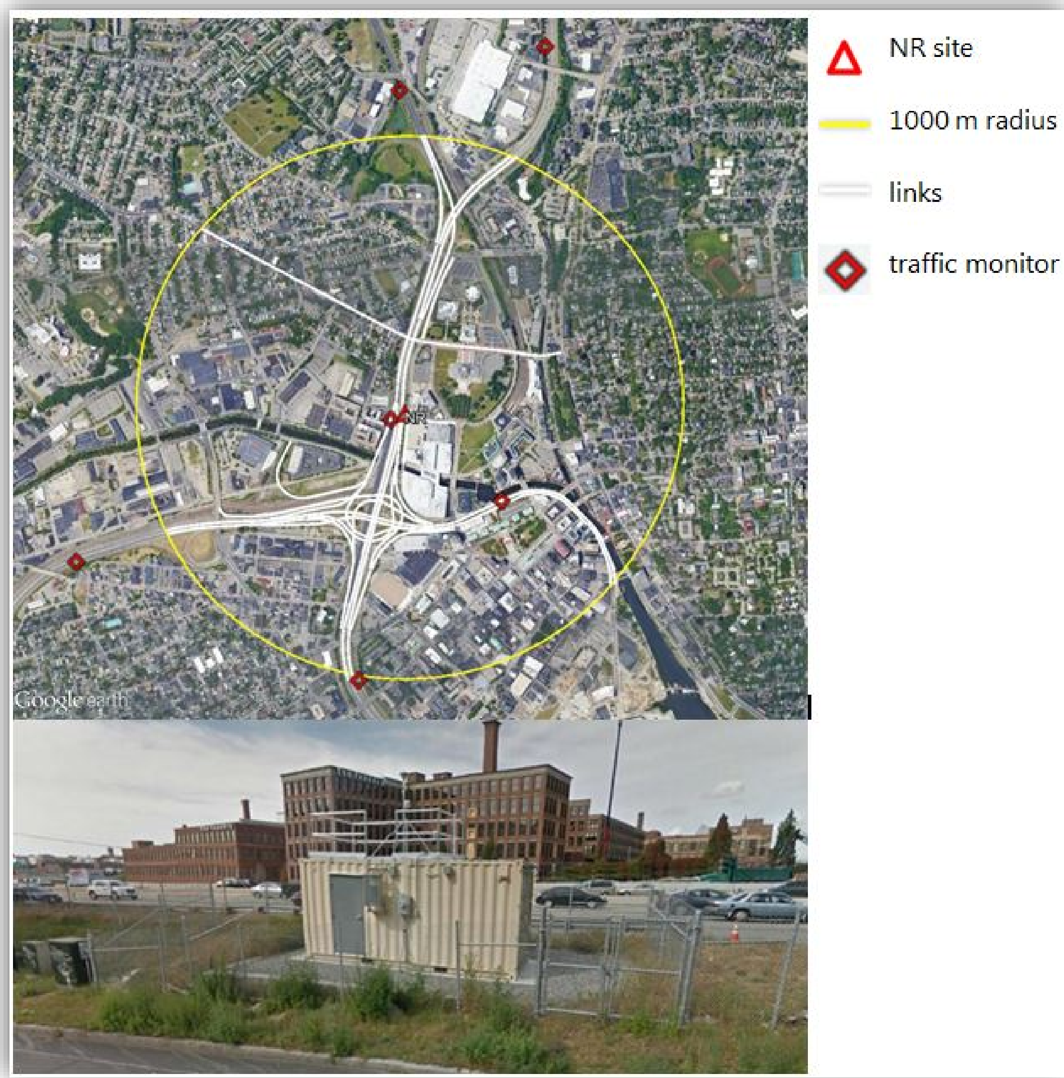


Figure 33. Providence modeling project area with 1000-m radius (yellow circle) centered on the Providence near-road air quality monitor (NR site, and pictured below), with available traffic monitors (◊) and roadway links that were included in the modeling (white lines). Imagery source: Google Earth.

Table 13. 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, 2017).

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 µg/m^3
Annual mean $PM_{2.5}$	9.3 µg/m^3
Co-located meteorology	None (nearest meteorological site at AQS 44-007-0022 2.4 km to the south)

6.5 Providence Monitoring Data and Background Sites

The near-road $PM_{2.5}$ increment was estimated between the Providence near-road monitor and the nearest air quality monitoring station (Urban League, South Providence, AQS ID 44-007-0022). The analysis covered 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 34), 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.

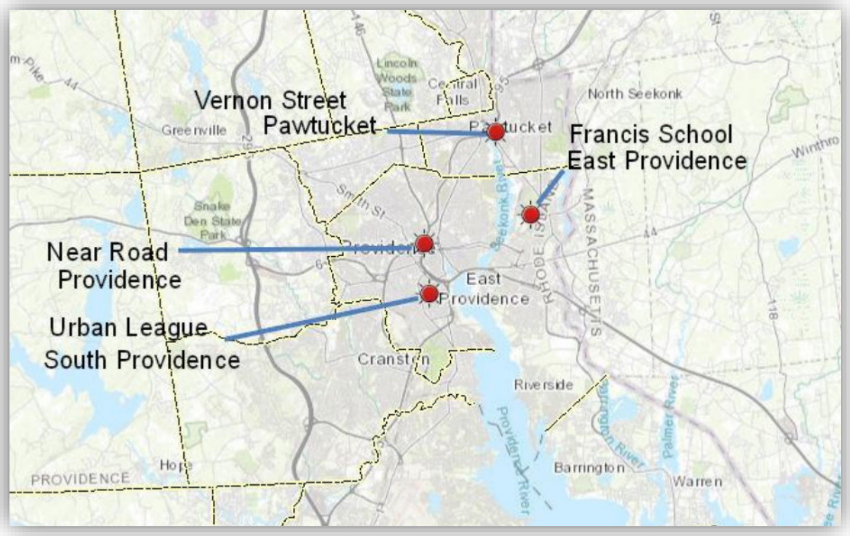


Figure 34. Location of the Providence near-road $PM_{2.5}$ FEM monitor and nearby $PM_{2.5}$ monitors (FEM and FRM co-located monitors). Data from Urban League South Providence (AQS ID 18-097-0078) and Francis School East Providence (AQS ID 18-097-0083) were considered in the Providence increment analysis.

6.6 Results: Indianapolis

6.6.1 Measured Increments

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 35. 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 35) 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 35) 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.

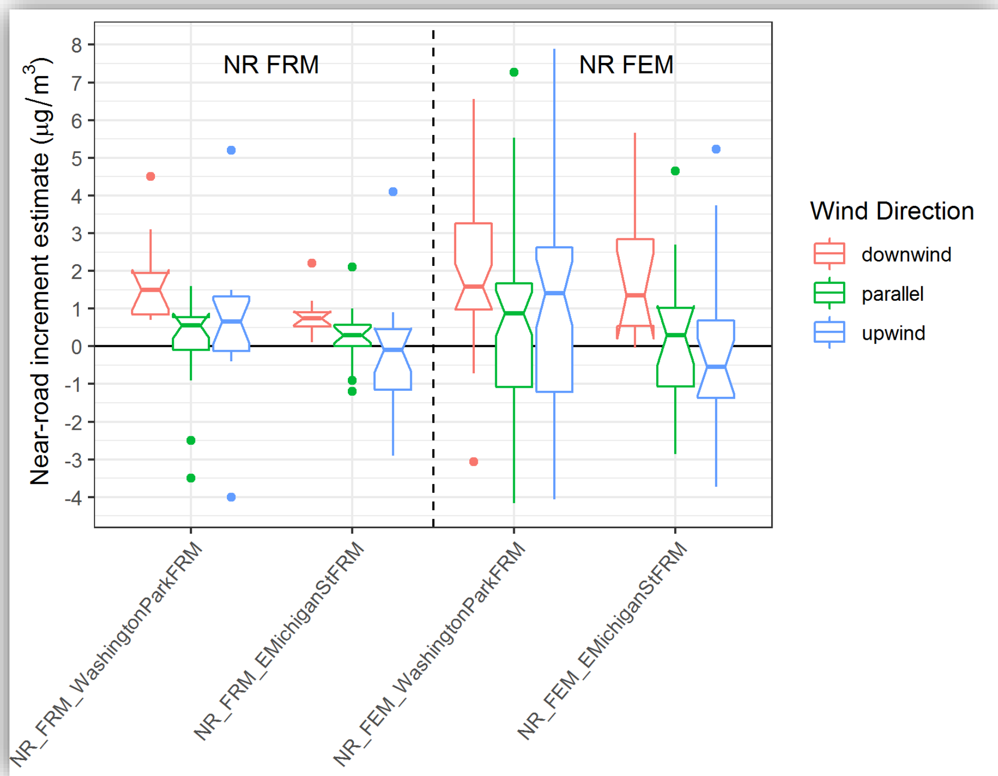


Figure 35. Indianapolis near-road $PM_{2.5}$ increments for four different combinations of near-road (NR) monitor (NR FRM and NR FEM) and nearby FRM $PM_{2.5}$ monitors at Washington Park and E. Michigan St. when the Indianapolis near-road monitor was downwind, upwind, and parallel to I-70. The horizontal 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 35, 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.

6.6.2 Modeled Increments

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 36; summary statistics are shown in Table 14. Based on these results, AERMOD over-predicted the average near-road $PM_{2.5}$ increment. The average modeled increment (3.7 µg/m^3) was a factor of four larger than the measured FRM-based increment (0.9 µg/m^3), and a factor of three larger than the measured FEM-based increment (1.2 µg/m^3). The resulting bias (2.8 µg/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.

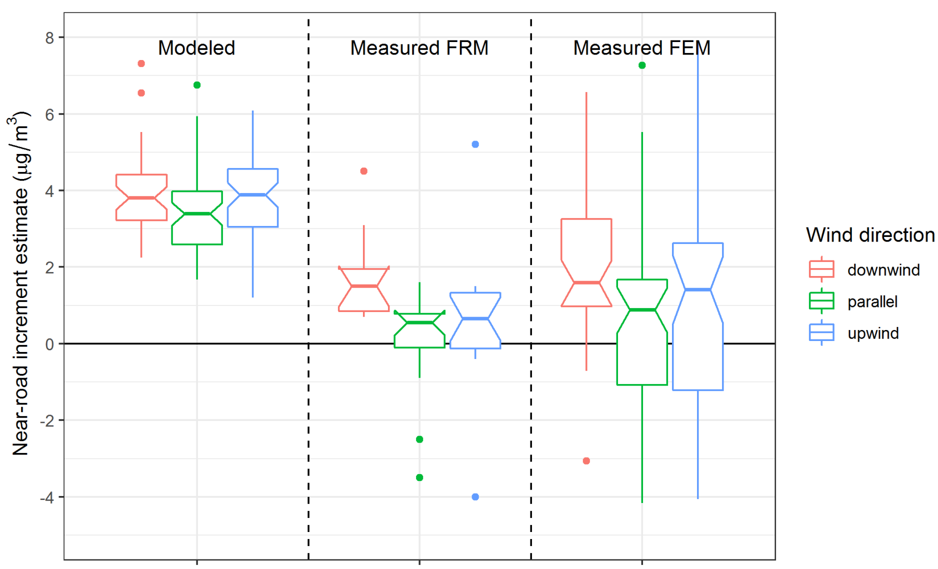


Figure 36. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured near-road $PM_{2.5}$ increments at the Indianapolis near-road monitoring site during 2016 for three wind conditions (near-road monitor downwind, upwind, and parallel to I-70).

Table 14. 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, 2013b), 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.

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 15. 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 15. 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

Results of the sensitivity modeling simulations for the Indianapolis analysis are summarized in Figure 37 and in Table 16. 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.

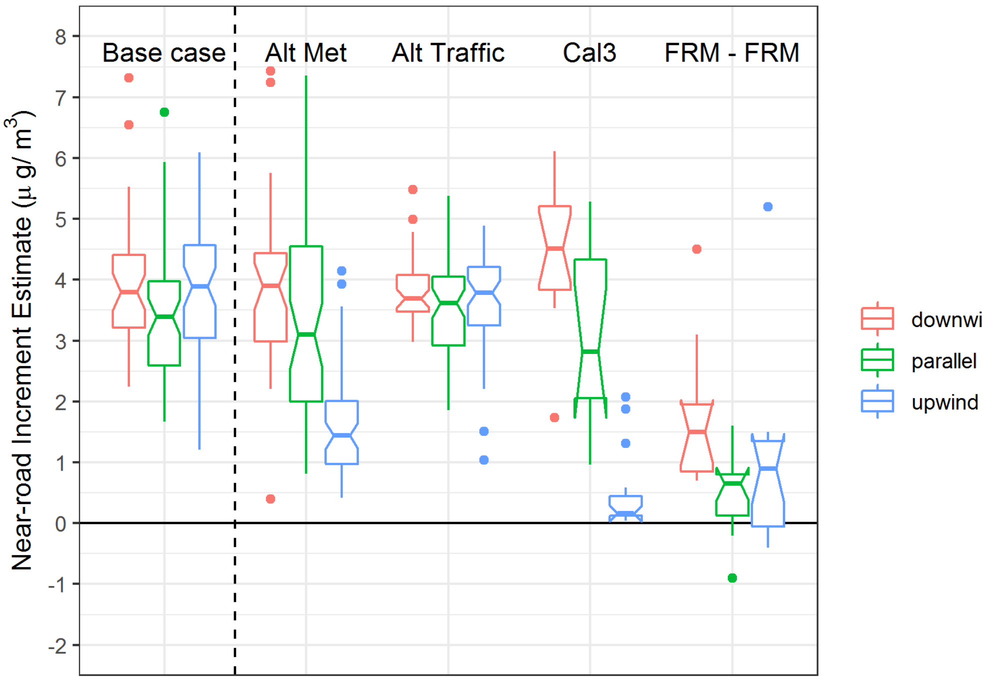


Figure 37. Distribution of modeled daily average $PM_{2.5}$ concentrations from the base case, Alt Met, Alt Traffic, and Cal3 scenarios, and measured FRM to FRM near-road $PM_{2.5}$ increments at the Indianapolis near-road monitoring site in 2016 during three wind conditions (near-road monitor downwind, upwind, and parallel to I-70). AERMOD results cover 152 days; Cal3 results cover 40 days.

Table 16. 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

6.7 Results: Providence

6.7.1 Measured Increment

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.

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 38**. 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 $\mu g/m^3$. The median increment calculated between the Francis School monitor and the near-road monitor was larger (2.4 $\mu g/m^3$). 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 $\mu g/m^3$ depending on the approach and monitoring sites used) but consistent with those reported by Mukherjee et al. (2019) (2.0 $\mu g/m^3$) when accounting for differences in monitoring methods (FRM vs. FEM) in the increment calculations.

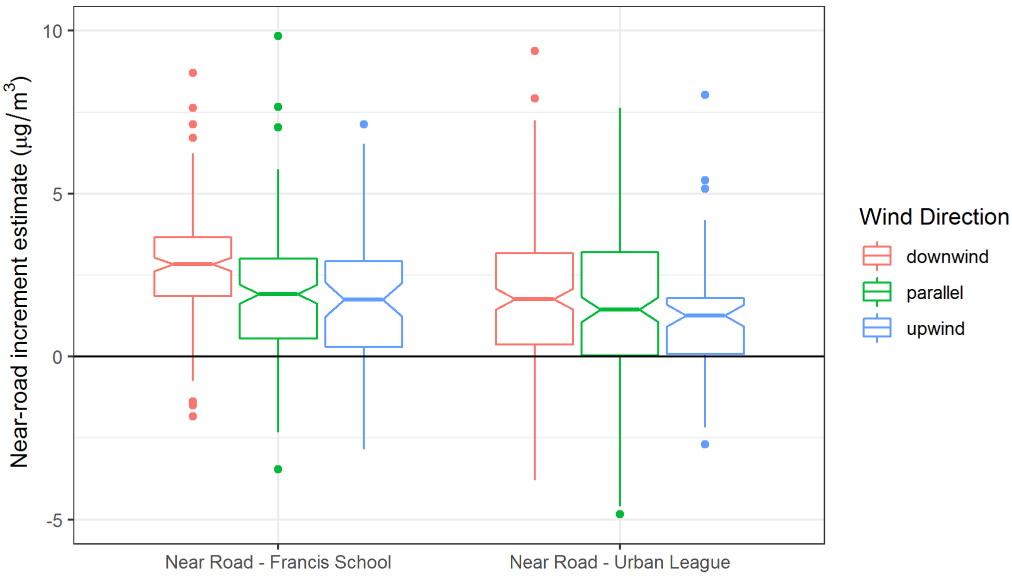


Figure 38. Providence near-road $PM_{2.5}$ increments between the near-road (NR) FEM monitor and nearby Urban League and Francis School FEM monitors when the Providence near-road monitor was downwind, upwind, and parallel to I-95.

6.7.2 Modeled Increment

AERMOD modeling results for Providence are compared to measured increments for all 382 analysis days in Figure 39. Based on these results, AERMOD over-predicted the average near-road $PM_{2.5}$ increment at Providence. The average modeled $PM_{2.5}$ increment across all 382 analysis days (8.8 µg/m^3) was more than a factor of six (530%) larger than the average measured increment (1.4 µg/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 µg/m^3 compared to 2.8 µg/m^3). The averaged modeled increment is larger than the estimated uncertainty of the measured near-road increment (± 0.2 µg/m^3) and larger than the variability in the increment due to the choice of background monitor (about 1.0 µg/m^3).

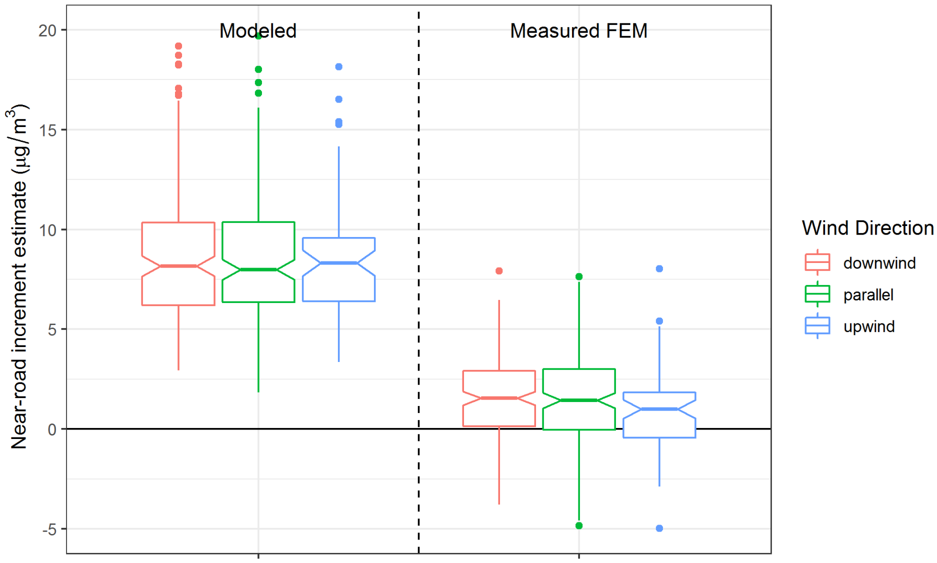


Figure 39. Distribution of AERMOD-modeled daily average $PM_{2.5}$ concentrations and measured (ambient) near-road $PM_{2.5}$ increments at the Providence near-road monitoring site during 2015-2016 (382 days) for three wind conditions (near-road monitor downwind, upwind, and parallel to I-95).

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%).

6.8 Synthesis of Findings: Indianapolis and Providence

A synthesis of Indianapolis and Providence modeling results is shown in **Table 17**. 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 17. 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 (µg/m^3)	3.7	8.8
AERMOD peak 24-hr $PM_{2.5}$ increment (µg/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 µg/m^3 at Indianapolis and 0.9 µg/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.

7. Mitigation Insights

Also, as part of the work completed by the TPF, the research program conducted two exploratory assessments of potential near-road mitigation. One project evaluated a California case study of a program to offset transportation project construction emissions with truck retrofits; a second project evaluated potential impacts from use of near-road barriers as a method to reduce concentrations downwind of a road. Highlights from both studies are presented here.

7.1 Truck Retrofit Case Study

This project produced findings from an examination of four truck replacement programs and offered lessons learned and considerations for mitigating project-level air quality impacts. Work focused on the Heim Bridge Replacement Mitigation Truck Program (Heim Truck Program), which provided support for the replacement of heavy-duty trucks operating in and around the Ports of Los Angeles and Long Beach (San Pedro Bay Ports). The program was developed as a mitigation measure to offset the increase in emissions resulting from marine vessel detours during construction of the Commodore Schuyler F. Heim Bridge in southern California. Related findings from three other truck replacement programs aiming to improve regional air quality were also evaluated (those findings are available in the original report).

The goal of these truck replacements was to mitigate NO_x impacts over a discrete time period by replacing older, higher-emitting diesel-powered trucks with newer vehicles. Since diesel trucks are important sources of NO_x and particulate matter (PM) emissions, implementation lessons from the case study profiled here are also applicable to truck-related PM emissions control. In addition, although the case study involves mitigation of construction impacts, the lessons learned may also apply to mitigating project-level operational emissions. In some cases, operational emissions may need to be mitigated in the near-term only, since vehicle fleet turnover, given sufficient time, will adequately reduce emissions over the long-term. The case study profiled here illustrates a near-term emissions control approach.

7.1.1 Heim Bridge Project Background

The Commodore Schuyler F. Heim Bridge was constructed in 1946 as a vertical-lift bridge spanning the Cerritos Channel (approximately three-quarters of a mile) along State Route 47 (SR-47) in southern California (**Figure 40**). Located within the City of Los Angeles on land owned by the Port of Long Beach, it serves as a major traffic route connecting Terminal Island within the San Pedro Bay Ports to the mainland cities of Long Beach and Los Angeles. During its operation, the vertical-lift bridge was typically raised several times per day to allow ship traffic to pass underneath.

Figure 40. The Commodore Schuyler F. Heim Bridge before (photograph on the left) and after (architect's rendition on the right) the replacement project.

The Schuyler Heim Bridge Replacement and SR-47 Expressway Project was composed of two major construction tasks:

1. Replacing the seismically unsafe lift-span portion of the Schuyler Heim Bridge over Cerritos Channel with a six-lane, fixed-span bridge along and east of the existing bridge alignment (the focus for the air pollution mitigation effort), and
2. Adding a four-lane elevated roadway that bypasses three signalized intersections and five railroad crossings, providing a high-capacity alternative route along the Alameda Corridor between Terminal Island and Alameda Street, south of Pacific Coast Highway (postponed as of August 2015).[16]

Figure 41 shows a map of the project relative to the ports. The replacement bridge was planned to be approximately 13 m wider than the former vertical-lift bridge to accommodate the addition of standard shoulders, and was designed to maintain a minimum vertical clearance of 14.3 m over the width of the navigable channel (approximately 55 m). The project cost was estimated to be $180 million, with construction occurring from 2011 to 2017.

[16] More project details are available from ACTA (acta.org/projects/projects_planning_SR47.asp) and Caltrans (dot.ca.gov/dist07/travel/projects/details.php?id=28).

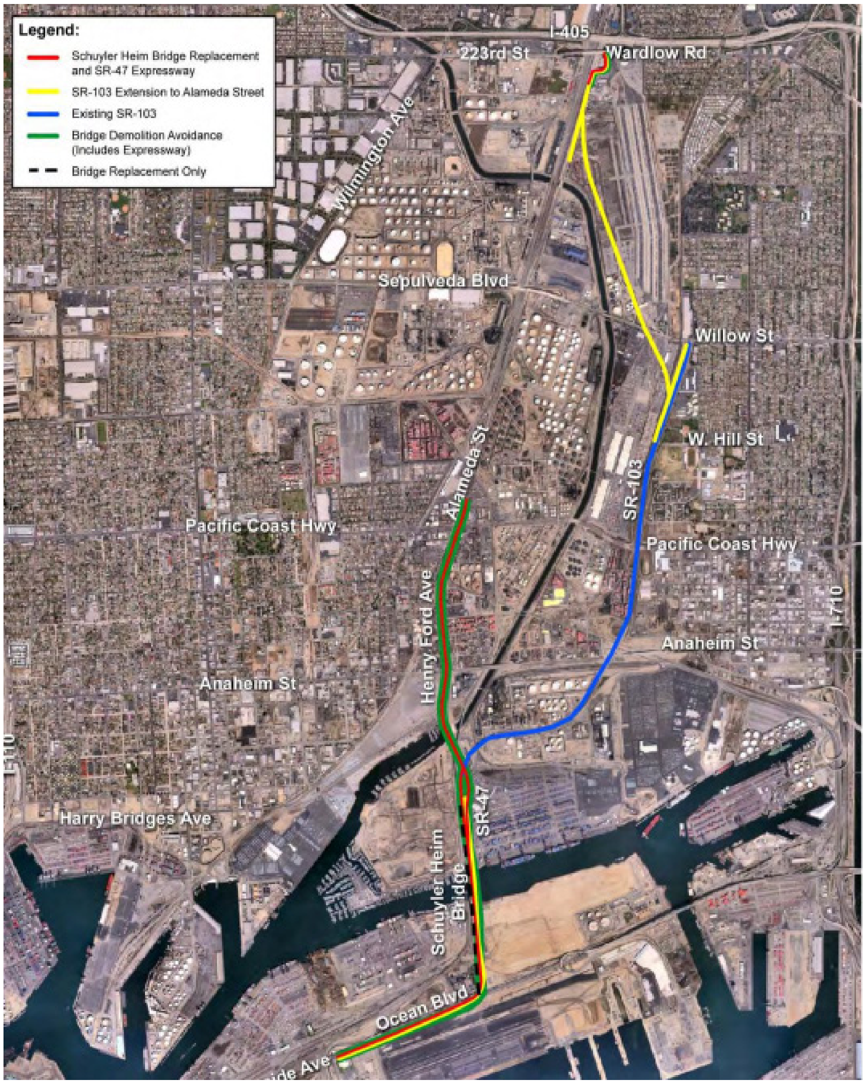


Figure 41. Map showing the locations of the Schuyler Heim Bridge Replacement and SR-47 Expressway Projects. Reproduced from "Schuyler Heim Bridge Replacement and SR-47 Expressway Project," available at futureports.org/events/sr47presentationhahnstaff.pdf.

In addition to seismic safety concerns, the Schuyler Heim Bridge Replacement Project was motivated by an increase in truck traffic volume and congestion around the San Pedro Bay Ports that had limited the movement of people, freight, and goods, particularly during traffic flow interruptions when the lift-span bridge was raised for marine traffic. The project was designed to relieve congestion on the Harbor and Long Beach freeways and to improve goods movement by providing alternative routes for port-related truck traffic to Terminal Island and local distribution centers and warehouse facilities in the area.

7.1.2 The Heim Bridge Replacement Mitigation Truck Program

During the environmental review process of the Schuyler Heim Bridge Replacement Project, emissions of nitrogen oxides (NO_x) were projected to exceed daily significance thresholds set by the South Coast Air Quality Management District (SCAQMD), in part due to marine vessel detours during construction. To offset air quality impacts during construction, the project committed to implementing several mitigation measures, including a heavy-duty truck buyback program known as the Heim Bridge Replacement Mitigation Truck Program (Heim Truck Program). The program offered $25,000 in grant funding to replace each of up to 15 heavy-duty trucks servicing the San Pedro Bay Ports with trucks equipped with newer, lower-emitting engine models. The Alameda Corridor Transportation Authority (ACTA) was responsible for implementing the Heim Truck Program on behalf of the California Department of Transportation (Caltrans).

The air quality technical study for the Schuyler Heim Bridge replacement estimated that each truck replacement would reduce NO_x and PM by approximately 0.55 and 0.12 tons per year, respectively. Prior to program implementation, the total program cost was estimated to be approximately $600,000; the estimate was based on the cost of previous truck replacement programs. This cost estimate included grant funding for 15 truck replacements and administrative costs. Emissions reductions from the Heim Truck Program were expected to continue for at least three to five years (exceeding the duration of the project construction phase), with the potential to mitigate truck emissions for a longer period of time if the cleaner replacement trucks continued to operate in and around the San Pedro Bay Ports. The cost-effectiveness of the program at reducing NO_x emissions, based on the cost-effectiveness of recent buyback programs, was projected at approximately $25,000 to $50,000 per ton of NO_x. In summary, the program offered grant funding for the replacement of on-road, Class 8 heavy-duty "exempt" drayage trucks, as defined by the California Air Resources Board (CARB) Drayage Truck Regulation Exemption.[17] Drayage trucks are trucks that transport goods over a short distance, often operating near a port. The CARB Drayage Truck Regulation required that Class 7 and 8 drayage trucks using model year 2006 and older engines be replaced with trucks using model year 2007 or newer engines by December 31, 2013. The regulation applied to trucks hauling cargo that originated from or was destined for rail yards and ports in California. Trucks that are exempted from the Drayage Truck Regulation and eligible under the Heim Truck Program included dedicated-use, uni-body vehicles such as fuel-delivery vehicles and scrap haulers, concrete mixers, logging trucks, and on-road mobile cranes.

Eligibility for the Heim Truck Program was largely related to three factors that govern the emissions reductions achieved by a truck replacement: (1) the engine model years of the existing and replacement trucks; (2) the annual vehicle miles traveled (VMT); and (3) the number of trips made to the San Pedro Bay Ports. The average annual VMT of the existing truck for the two years prior to replacement was used to establish the baseline emissions for the existing truck and the engine model needed for the new truck to meet the emissions reduction target of the project. All applicants

[17] arb.ca.gov/msprog/onroad/porttruck/exemption.htm

were required to submit truck replacement project applications electronically via a link on ACTA's website (acta.org/truckgrant/). Hard copies of grant applications were not accepted. Only one application per applicant was permitted; however, applicants could apply for the replacement of up to three trucks per application.

The truck to be replaced had to be an operational, insured, and registered Class 8 on-road vehicle. The truck had to be equipped with a heavy-duty diesel engine of model year 2009 or older, have a Gross Vehicle Weight Rating (GVWR) of more than 33,001 pounds, and have a history of operating near the San Pedro Bay Ports. Operational eligibility criteria for existing trucks included (1) an annual mileage requirement for the previous two years based on the engine years of the existing and replacement trucks; (2) a port trip requirement that the existing truck made at least 150 service trips to the San Pedro Bay Ports in each of the last two years; and (3) current registration with the California Department of Motor Vehicles (DMV) for the previous two years. The existing truck had to be scrapped with an approved, California state certified recycler, and an ACTA representative had to be present at the scrapping.

The replacement truck had to be a new or used diesel or alternative fuel Class 8 on-road vehicle with a GVWR of more than 33,001 pounds and had to be the same type of truck as the existing truck (for example, a car carrier must be replaced with a car carrier). The replacement truck had to be equipped with a heavy-duty engine that met or exceeded the model year 2007 California heavy-duty, diesel-fueled on-road emissions standards[18] and had to operate in the San Pedro Bay Ports for three consecutive years upon purchase. The truck had to be purchased by the grantee from a California licensed truck dealership, be registered in the state of California, be operational within 60 days of the effective date of the grant agreement, operate within California 100% of the time, and make no fewer than 150 service trips to the San Pedro Bay Ports per year of the agreement (450 port trips total over three years). The grantee was required to disclose the funding methods used to cover the remainder of the purchase price of the truck not covered by the Heim Truck Program grant. During the term of the agreement, grantees had to maintain the replacement truck in operating condition according to manufacturer's records and make the replacement truck available for inspection upon request.

7.1.3 Heim Truck Program Implementation and Lessons Learned

Table 18 provides a timeline of key events related to implementation of the Heim Truck Program.

[18] arb.ca.gov/regact/HDDE2007/hdde2007.htm.

Table 18. Timeline of key Heim Truck Program implementation activities.

Date	Activity
August 2009	Record of Decision (ROD) for the replacement of the Schuyler Heim Bridge is approved; truck program is listed as a mitigation measure to offset indirect construction emissions bridge replacement
October 2011	Construction work on the Schuyler Heim Bridge begins
October 2013	Heim Truck Program planning begins
March 2014	Outreach/recruitment for the truck program begins
April 2014	Grant solicitation is released
May 2014	Deadline to submit Phase 1 application for the truck program
October 2014	First truck is replaced
2017	Scheduled completion of construction of the Schuyler Heim Bridge

As the Heim Truck Program began, the San Pedro Bay Ports had reached the tail end of a port truck replacement program under the San Pedro Bay Ports Clean Air Action Plan (CAAP).[19] As a result, while the Heim Truck Program originally planned to replace heavy-duty diesel vehicles (HDDV), program administrators found that many of these vehicles had already been replaced or were in the process of being replaced under the CAAP. In response, facilitators identified a new pool of target trucks for replacement under the Heim Truck Program: heavy-duty diesel trucks that had been exempt from replacement under the CAAP. Vehicle types exempt under the CAAP include dedicated-use trucks such as car carriers, pneumatic tankers, and scrap haulers.

The new pool of trucks targeted for replacement by the Heim Truck Program was substantially smaller than the original pool, resulting in fewer applications than anticipated. Program administrators sent an estimated 7,000 emails to reach potential applicants and visited locations frequented by truck owner/operators (e.g., truck stops and union meetings) to post flyers to publicize the program. They identified eligible candidates by consulting a list of trucks that were exempt from the CARB Drayage Truck Regulation and reached out to the owner/operators of those trucks. The Heim Truck Program was advertised on the Caltrans and ACTA websites, and recruitment efforts and materials were made available in English and Spanish.

As of August 2015, four heavy-duty diesel trucks had been replaced and two applications for the replacement of three additional trucks were under review (one of the trucks under review was anticipated to be replaced in early September 2015). The trucks replaced were scrap haulers with

[19] The San Pedro Bay Ports Clean Air Action Plan was adopted in 2006 to reduce air pollution and health risks associated with port activity. A major component of the plan was a truck replacement program to phase out all older diesel trucks operating in the ports within five years by replacing the trucks with retrofitted or newer vehicles that operate more cleanly. The program established a progressive ban on highly polluting trucks between 2008 and 2012, ending with a ban on all trucks that did not meet the 2007 Federal Clean Truck Emissions Standards (http://epa.gov/otaq/hd-hwy.htm) by January 1, 2012.

engine model years in the 1980s. The total cost of the replacement trucks ranged from approximately $70,000 to $170,000, depending on whether a used or new truck was purchased. As of August 2015, truck replacements resulted in larger air quality benefits than originally anticipated on a per-truck basis because the trucks replaced had been particularly old, high-emitting vehicles. Program administrators offered a revised estimate of seven trucks needed to mitigate air quality impacts associated with the bridge construction. They intended to replace at least one truck more than was needed to meet program goals, so that if a participant defaulted on a contract, the target emissions reduction would still be met.

The planning phase of the Heim Truck Program required a high level of effort for program administrators. This was due in part to the need to identify a new pool of program applicants and to recruitment challenges, but was also due to the effort required to draft and revise the program contract to address the consequences of a default. Legal advice was sought to help craft, review, and revise contract language, a process that required additional time and increased administrative costs from what was originally anticipated for the planning phase. Once the planning phase was complete, day-to-day administrative costs decreased substantially. Administrators estimate that the staff time requirement for program administration decreased to approximately one-half full-time equivalent (FTE) and that once all participants had been recruited, program administration would decrease further to approximately one-quarter FTE. Despite the extended application and implementation phases beyond what was originally anticipated, the project remained on budget.

Heim Truck Program administrators offered the following implementation lessons learned:

- **Identifying eligible applicants was a challenge due to the success of previous truck retrofit/replacement programs.** Facilitators had to identify a new pool of target trucks and selected heavy-duty trucks that had been exempt from replacement under the CAAP. This new target truck type posed a challenge because only a small pool of these trucks met the program criteria. Thus, a large recruitment effort was required to identify eligible applicants. E2ManageTech visited truck stops, union meetings, and other facilities frequented by truck owner/operators to post flyers and sent out emails to approximately 7,000 possible applicants. The Heim Truck Program administrators anticipated receiving 500 or more applications; however, only about 15 to 20 applications were received in the first month, and many of the applicants were not qualified to participate. Because of the low number of qualified candidates, the open application process was repeated several times. As of August 2015, four trucks had been replaced, and applications for the replacement of an additional three trucks were under review.
- **Providing documentation to establish activity (VMT and San Pedro Bay Port trips) for previous years was a challenge for applicants.** Administrators had planned to use radio-frequency identification (RFID) tags that monitor port entry and exit, along with trip destination records, to validate port trips and VMT; however, while the types of heavy-duty trucks originally targeted by the program had RFID tags, the types of trucks that ended up being replaced by the project did not. This meant that port "trip slips" issued by the San

Pedro Bay Ports for each trip had to be used, and many applicants did not have complete records (i.e., port trip slips from each trip over the previous two years) to demonstrate that they met the requirements. As a solution, administrators allowed applicants who were unable to assemble all trip slips to submit the trip slips that they had in their possession along with an affidavit affirming that the activity requirements had been met in previous years.

- **Program implementation took longer than expected.** Even after an applicant qualified to participate in the program, it could take months to complete the steps and paperwork necessary for executing the grant agreement. For some applicants, additional time was needed to pay off a loan on the existing truck before it could be scrapped. Applicants also faced challenges and delays in securing financing to cover the cost of the replacement truck not covered by the $25,000 grant. Furthermore, a common challenge faced by many program applicants was the lack of time to complete the necessary paperwork. All but one of the program participants were individual owner/operators working long hours with little free time to complete the paperwork. Those owner/operators often solicited assistance from their wives or significant others to help complete the necessary paperwork.
- **There are pros and cons associated with online application and document management.** Program administrators acknowledged that online document management is particularly helpful when many applications are anticipated, because it can increase processing efficiency. A downside to online document management, however, is that it requires more effort to develop the process, and users who do not have much computer experience may require more support to complete applications and submit documentation online.
- **There are many nuances to program implementation, and it is important to be flexible to address unforeseen issues.** Program administrators emphasized that there are many nuances to program implementation and that while some lessons learned may be applicable to future programs, others may be specific to a project's area or region. For example, for the Heim Truck Program, local programs and regulations, including previous truck programs, impacted program implementation. Additionally, program facilitators found that eligible candidates were reluctant to make the financial commitment of purchasing a new truck because of concerns over the state of the economy.
- **Word of mouth can be valuable for recruitment.** Program administrators found that word of mouth was an important component of recruitment. Several program applicants indicated they learned about the program from other truck owner/operators. Flyers were also a useful outreach tool. Electronic media was not as useful as word of mouth or flyers.
- **Time and effort are required to ensure that candidates understand contract terms.** Program administrators stressed the importance of ensuring that applicants fully understand all of the terms of the contract. For the Heim Truck Program, these terms included the three-year term length and reporting requirements; the timeline for applying for the program, scrapping the old truck, and purchasing the replacement truck; and that truck maintenance is not covered by the program.

- **Contractor experience is valuable to program implementation.** Program administrators emphasized the value of experienced contract support during program implementation. A good candidate for contracting support has local outreach contacts with port terminals (in the case of a port truck replacement program), truck recycling companies, financing institutions, trucking unions, and the community.

Heim Truck Program administrators offered the following recommendations to parties interested in developing a similar truck retrofit or replacement program in the future:

- **Be aware of any state and/or regional-level efforts to replace/retrofit target trucks.** Research programs that are currently in place in the project area, as well as local issues and other requirements, may influence the design of a truck program. Knowledge of such programs early in the planning process will save time in the long run.
- **Be aware of target truck costs and select the award amount accordingly.** Replacement of more expensive trucks will likely require a higher award amount to encourage participation. For example, for the Heim Truck Program, the $25,000 award amount was more attractive to owner/operators of scrap haulers (less expensive trucks) than owner/operators of car carriers (more expensive trucks). Identify the types of trucks that will be targeted by the program early in the planning stage to ensure that award amounts are sufficient to cover a substantial fraction of the total truck cost and attract qualified applicants. Higher award amounts will likely draw more interest.
- **Streamline application and implementation processes as much as possible.** Implementation of a truck mitigation program can be time-consuming for a variety of reasons. The application process may take longer than expected if recruitment issues are encountered, and applicants may require more support along the way than anticipated. Furthermore, funding transfer delays can result in project delays. It is important to streamline the implementation process as much as possible to retain qualified applicants. A major concern for applicants is the amount of time the process takes, particularly the time between scrapping the old truck and receiving funding for the purchase of the new truck, because owner/operators rely on their trucks for their livelihood. In the case of the Heim Truck Program, program administrators attended the truck scrapping and handed the grant award check to the applicant immediately following the truck destruction to minimize the amount of time that the applicant was without a truck. Another consideration is that truck dealers will not necessarily hold the new truck targeted for purchase. If the target truck is sold, the applicant will be required to repeat much of the application process, resulting in additional delays.
- **Be prepared to provide assistance to applicants throughout the application process.** Applicants may not have computer experience and may need help with applications. Applicants may also face challenges related to language barriers; three out of five of the participants in the Heim Truck Program (as of August 2015) spoke very little or no English—offering information in Spanish was critical to those applicants. As discussed earlier, program administrators found that applicants have very little time to complete applications and

gather necessary documentation, particularly during normal business hours. The availability of technical support may need to be scheduled to accommodate applicants' schedules (e.g., periodically provided after normal business hours or on weekends).

7.2 Near-Road Barrier Evaluation

The intent of this work was to identify known near-road barrier effects on air quality, based on the literature, and to complete a simplified assessment of dispersion modeled barrier effects using the latest applicable U.S. Environmental Protection Agency (EPA) modeling tools. This work summarized pertinent research examining the effects of barriers on near-road air quality, including results from measurement, wind tunnel, and modeling studies; it also provided one illustration of modeled outcomes by presenting findings from a limited series of sensitivity analyses performed to model the effects of roadway and barrier configurations on near-road pollutant concentrations. These analyses used the EPA's R-LINE v2.0 dispersion model. The different model scenarios varied the barrier height, meteorological conditions, and vehicle emission release height. This study was conducted for research purposes and does not address model uncertainties or the potential for any unintended consequences associated with the placement of barriers near roadways. As discussed in prior sections, near-road pollutant concentrations are a function of ambient background concentrations plus the added incremental impact of on-road mobile sources. The potential mitigation benefit of a barrier, as discussed here, applies to the on-road mobile source increment and not to ambient background concentrations.

7.2.1 Literature Review Findings

The effects of barriers and elevation differences have been studied in three ways: monitoring studies, wind tunnel tests, and computational modeling studies. Most existing studies focus on the scenario in which winds are perpendicular to the roadway direction. Under these crosswind conditions, studies show that a barrier or a difference in the elevation of a roadway in relation to the surrounding terrain (i.e., an above-grade or below-grade road) obstructs air flow, thereby decreasing concentrations on the leeward side (i.e., downwind) of the road for a distance that is dependent on the height of the barrier or grade, the wind speed, and proximity to the edge of the barrier.

STI examined near-roadway studies that evaluated pollutant concentration gradients in the vicinity of barriers or other elevated/depressed roadway configurations, regardless of pollutants measured or modeled. The existing body of literature includes few studies that measure particulate matter (PM) concentrations. Studies of ultrafine particles (UFP), black carbon (BC), carbon monoxide, or nitrogen oxides are more common. The biggest difference between these other pollutants and PM is that background concentrations of PM are much higher; therefore, the relative gradient in PM concentrations near the roadway is much smaller.

Tables 19 and 20, which are updated versions of a table initially developed by McCarthy et al. (2011) for Caltrans, show the studies that were reviewed. Table 19 summarizes studies that assess barrier-related reductions relative to a no-barrier control case; Table 20 summarizes results from studies that assess barrier-related reductions relative to the roadway.

Table 19. Summary of literature review findings on the effects of a barrier on near-road air quality. These studies all assess the percent reductions due to the barrier relative to a section of road without a barrier. Updated from McCarthy et al. (2011).

Pollutant	Type of Study	Barrier Height (m)	Wind Data (Duration, Direction, Speed)	Downwind Distance from Barrier (m)	% Reduction Relative to No-Barrier Control [c]	Reference
CO	Monitoring	6	Integrated, but directionally from road	15, 45, 95, 295	15−50%; not quantified by distance [c]	Baldauf et al. (2008b)
Particle counts (20 nm; 75 nm)	Monitoring	6	Integrated, but directionally from road	15, 45, 95, 295	15% for the first 100 m [c] for 20 nm; 15% for the first 40 m; −10% at 80-120 m [c] for 75 nm	Baldauf et al. (2008b)
NO_2, NO_x	Monitoring	4	>1 m/s, 1.5 years of data with wind direction ±60° from perpendicular	5, 10, 28	1%, 14%, 7% [c] for NO_2 27%, 20%, 13% [c] for NO_x	Dutch Air Quality Innovation Programme (2009)
PM_{10}	Monitoring	4	>1 m/s, 1.5 years of data with wind direction ±60° from perpendicular	5, 10	20%, 34% [c]	Dutch Air Quality Innovation Programme (2009)

Pollutant	Type of Study	Barrier Height (m)	Wind Data (Duration, Direction, Speed)	Downwind Distance from Barrier (m)	% Reduction Relative to No-Barrier Control [c]	Reference
Black carbon, particle counts >0.5 µm	Monitoring	~10	Integrated over 28 days	30 m	BC – 12.4% during downwind conditions [c] Particle counts – 0% during downwind conditions [c]	Brantley et al. (2014)
BC, CO, NO_2, UFP	Monitoring	~4.5	Integrated over one month	0-50, 50-150, 150-300	BC – [0–50 m] 43–48%; [150–300 m] 18–24% [c] NO_2 – [0–50 m] 34–37%; [150–300 m] 11–28% [c]	Baldauf et al. (2016)
Particle counts	Monitoring	3.7, 5.2	3.1 m/s 0.9 m/s	15 to 400	60% at 15 m, 0% at 100 m, 30% at 200 m [c] 55% at 15 m, 0% at 75 m, 50% at 150 m [c]	Ning et al. (2010)
BC, CO, NO_2	Monitoring	3.7, 5.2	3.1 m/s 0.9 m/s	15 to 400	22-60% at 15 m; -50 to -125% at 100 m; -38 to -105% at 200 m [c] 23-49% at 15 m; -67 to -122% at 60 m; -65 to -150% at 150 m [c]	Ning et al. (2010)
SF_6 (tracer)	Field experiment	6	1.4, 1.6, 3.6, 5.5 m/s	18 to 180	80% up to 120 m [c] 60–80% at 180 m [c] depending on wind speed	Finn et al. (2010)
Inert tracer	Wind tunnel	4	Not reported	Up to 110	60% reduction at 40 m [c]	Hölscher et al. (1993)
Inert tracer	Wind tunnel	6	4.97 m/s	Up to 240	>50% reduction at distances <60 m, converging to base case as distance increases [c]	Heist et al. (2009)
Inert tracer	Modeling	6	2.25 m/s	Up to 450	>90% reduction at distance <50 m, less than 50% reduction at distances greater than 50 m, slight increase at distances at 200–250 m [c]	Bowker et al. (2007)

Pollutant	Type of Study	Barrier Height (m)	Wind Data (Duration, Direction, Speed)	Downwind Distance from Barrier (m)	% Reduction Relative to No-Barrier Control [c]	Reference
Inert tracer	Modeling	6	3.6, 7.4, 1.65 m/s	Up to 180	Model reproduction of the Finn et al. (2010) field study using the Comprehensive Turbulent Aerosol Dynamics and Gas Chemistry (CTAG) model. Reductions of 80% up to 120 m [c]	Steffens et al. (2013)
Inert tracer	Modeling	3, 6, 9, 18	4 m/s	Up to 450	15−61% reduction at 20 m [c]	Hagler et al. (2011)
Inert tracer	Modeling	6	1.7, 3.6, 5.0, 7.4 m/s	Up to 240	Model reproduction of the Finn et al. (2010) field study and Heist et al. (2009) wind tunnel study using source shift and mixed-wake dispersion models. Reductions of 30-90% up to 120 m [c]: biased low under stable and high under unstable conditions.	Schulte et al. (2014)
Inert tracer	Modeling	3, 6, 9	2.5, 5, 10 m/s	Up to 360	Model reproduction of the Heist et al. (2009) wind tunnel study with additional simulation scenarios. Reductions in inert tracer concentrations between 10-70% within the first 90 m of the barrier [c]	Steffens et al. (2014)
Inert tracer	Modeling	2-20	12 m/s	Up to 250	Model simulation of double noise barrier as a function of the ratio of the height of the barrier to the width of the road. Characterizes different flow regimes as isolated roughness, wake interface, and skimming flow. Reductions of 20−90% [c]	Jeong (2014)
UFP	Modeling	6, 9, 10	1, 2, 4 m/s	100	Increased reductions in particle number counts when wide vegetative barriers or a combination of solid and vegetative barriers are implemented. UFP reductions up to 60% [c].	Tong et al. (2016)

Table 20. Summary of literature review findings on the effects of a barrier on near-road air quality. These studies all assess the percent reduction in concentration as a function of distance from the roadway relative to on-road concentrations; the effects of the barrier are not directly quantified. Updated from McCarthy et al. (2011).

Pollutant	Type of Study	Barrier Height (m)	Winds (Duration, Direction, Speed)	Downwind Distance from Barrier (m)	% Reduction Relative to Roadway [r]	Reference
CO	Monitoring	2.44	Integrated over 6 weeks	Not described	20% [r]	Nokes and Benson (1984)
$PM_{2.5}$, VOCs	Monitoring	1	Variable	2	65% [r] for $PM_{2.5}$; – 43% [r] for VOCs	McNabola et al. (2008)
NO_x	Monitoring	NA	>1 m/s under neutral atmospheric stability from road	5, 35, 70, 150 (variable)	15% at 65 m, 22% at 90 m relative to 30 m [r] 15% at 30 m, 38% at 55 m relative to 15 m [r]	Naser et al. (2009)
Particle counts	Monitoring	3.7, 5.2	3.1 m/s 0.9 m/s	15 to 400	60% at 15 m, 0% at 100 m, 30% at 200 m [r] 55% at 15 m, 0% at 75 m, 50% at 150 m [r]	Ning et al. (2010)
BC, CO, NO_2	Monitoring	3.7, 5.2	3.1 m/s 0.9 m/s	15 to 400	65−80% at 15 m; 30−40% at 100 m; 50−60% at 200 m [r] 65−75% at 15 m; 30−45% at 60 m; 50−55% at 150 m [r]	Ning et al. (2010)

The literature consistently shows a reduction in pollutant concentrations behind (downwind of) a solid barrier. Most modeling studies suggest that solid barriers loft pollution farther downwind behind the barrier; some of the measurement literature notes that the presence of a barrier may lead to increased concentrations farther downwind as the plume reattaches to the surface (Ning et al., 2010). However, the reattachment plume phenomenon has not been demonstrated conclusively across measurement studies. Additionally, the potential spatial scale of the reattachment plume depends on many variables, including the roadway configuration, barrier height, wind speed, and atmospheric stability.

Pollutant concentrations directly behind a barrier are lower than both on-road concentrations and roadways with no barrier. This finding is qualitatively consistent across monitoring, model, and wind tunnel exercises. The magnitude of the pollution reduction due to the presence of a barrier varies widely across the different monitoring studies and pollutants of interest, with some studies reporting reductions as small as a few percent (Brantley et al., 2014; Dutch Air Quality Innovation Programme, 2009) and others reporting reductions as high as 75% (Ning et al., 2010; Finn et al., 2010). A majority of previous studies show reductions of approximately 20−60% within the first 100 m, and approximately 25−65% within the first 50 m, downwind of a barrier (Nokes and Benson, 1984; Baldauf et al., 2016; 2008a; 2008b; Tong et al., 2016; Brantley et al., 2014; Hagler et al., 2012; 2010; 2009; Bowker et al., 2007; Jeong, 2014). Several studies indicate that dispersion models are able to reproduce the reduction in concentrations due to the presence of barriers in the near-road environment under relatively stable atmospheric conditions (Steffens et al., 2014; 2013; 2012; Schulte et al., 2014). Under unstable conditions, models are less effective at representing downwind concentrations (Schulte et al., 2014). Models may also be less effective at representing edge effects; however, few measurement studies are available to validate models under these conditions.

Elevated and depressed roadways are also effective at reducing downwind pollutant concentrations. Mitigation of near-road air quality impacts by a solid barrier or elevated/depressed roadway configuration occurs by isolating the on-road pollution. For elevated roadways, the on-road pollution is lofted to a plume height of approximately 1.5 times the height of the barrier/grade. This plume may reattach to the surface at a distance of approximately 15 times the height of the barrier/grade (Ning et al., 2010; Heist et al., 2009; Steffens et al., 2014) and may or may not lead to a slight increase in concentrations at the point of reattachment compared to similar conditions where no barrier is present. The surface concentrations behind the barrier are reduced, assuming there are no additional pollution sources leeward of the barrier such as frontage or access roads that can emit within the recirculation zone.

Dispersion modeling tools indicate that the location and spatial extent of a pollution plume is a function of barrier height, roadway grade, wind speed and direction, and vehicle-induced turbulence (a function of traffic counts, speed, and fleet mix). Given that meteorological and traffic characteristics change over short time scales, the spatial location of the reattachment plume (if any) is also likely to change over short time scales. The distance at which the plume reattaches to the surface has been estimated by Heist et al. (2009) to be 10 to 30 times the height of the barrier, and by Finn et al. (2010) to be more than 30 times the height of the barrier. A more recent modeling study estimates a distance of approximately 15 times the height of the barrier (Steffens et al., 2014). The variable nature of these findings suggests that the location of plume reattachment (if it occurs) is highly variable depending on a large number of meteorological and roadway configuration factors.

A key modeling study that covers the largest range of near-road configurations is a study by Steffens et al. (2014), which investigated the following sets of roadway configuration variables:

- Barrier height (no barrier, 3 m, 6 m, 9 m)
- Barrier configuration (no barrier, barrier on the upwind side, barrier on the downwind side, and barriers on both sides)
- Roadway configuration (at grade, elevated with angled slope, depressed with no slope, depressed with angled slope)
- Wind speed (2.5 m/s, 5 m/s, and 10 m/s)
- Additive effects (combining differences in barriers and roadway configuration)
- Edge effects (what happens near the edge of the noise barrier)

Steffens et al. (2014) found that a single barrier, on either side of the road, is almost as effective as a double barrier (a barrier on each side of the road). The initial displacement of air around the barrier disrupts the flow of air over the road, resulting in lower downwind concentrations in both cases.

In the modeling literature, barrier height and roadway grade have the largest effect on downwind concentrations when winds are perpendicular to the roadway. The barrier lofts the plume of on-road pollution, resulting in possible dilution of the plume and a reduction in concentrations at downwind receptors compared to the no-barrier case. The effect is stronger for higher barriers, but the flow regime changes if the ratio of the height of the barrier to the width of the road exceeds 0.15 (Jeong, 2014). In an early paper examining this effect, researchers found that a height-to-width ratio of less than 0.3 results in an isolated roughness flow regime (Oke, 1988). In these cases, the movement of air across the barrier produces vortices like those seen in urban canyons, which can result in increased concentrations in the on-road environment.

Studies indicate that edge effects are an important factor governing the effectiveness of barriers at reducing pollutant concentrations downwind. The "edge" in this instance refers to the vertical edge at the end of the barrier; for example, in the case of a half-mile-long sound wall, the edge refers to the sound wall terminus at either end of the half-mile wall. Steffens et al. (2014) found that relative pollutant concentrations increase more than 10% within 150 meters of the edge of the barrier on the downwind side, because air flow wraps around the edge of the barrier and transports higher on-road pollution to the back (downwind) side of the barrier.

Computational fluid dynamics (CFD) modeling and wind tunnel studies suggest that the largest reductions in downwind concentrations would be achieved by combining a below-grade roadway with a barrier at the top of the grade. This is predicted to result in greater reductions in downwind concentrations than those from an equivalently sized barrier in an at-grade road scenario (Steffens et al., 2014; Heist et al., 2009).

In addition to the literature just discussed, in July 2016, EPA published Recommendations for Constructing Roadside Vegetation Barriers to Improve Near-Road Air Quality (Baldauf, 2016). The guidance includes important considerations and recommendations for designing a roadside vegetative barrier to mitigate air quality impacts from the roadway, based on characteristics that have been shown to effectively reduce near-road air pollutant concentrations. The report also provides links to additional resources for siting, designing, and maintaining roadside vegetative barriers. Although the focus of the guidance is on design characteristics for vegetative barriers, many of the considerations are also applicable to solid barriers and can be used to optimize their mitigation potential, such as

- Use a higher barrier to achieve greater downwind reductions in pollutant concentrations,
- Minimize gaps in the barrier that can lead to increased pollutant concentrations downwind, and
- Ensure that the barrier end is not located near sensitive receptors.

In summary, the literature identifies the following key parameters regarding barriers' effectiveness in reducing near-road concentrations when winds are perpendicular to the road:

- Barrier height
- Roadway configuration (above-grade, below-grade, at-grade)
- Barrier configuration (upwind, downwind, neither, or both; distance from road)
- Barrier length – edge effects
- Barrier type – solid or vegetative
- Meteorological conditions – primarily atmospheric stability/buoyancy, wind direction, and wind speed

On the scale of tens of meters to a few hundred meters from the roadway, studies suggest that barriers can effectively mitigate and dilute concentrations of mobile source-emitted pollutants. The magnitude of the reductions depends on the matrix of factors listed above. Both measurement and model studies indicate that the largest reductions occur within the first 50 m of the road, with the magnitude of the reduction typically decreasing by at least 20% after the first 50 m relative to the initial reduction. Within the first 100 m, typical modeled reductions are approximately 20-80%, with a majority of the reductions falling between 40-75%. Effects from the five measurement studies range from a 100% increase in concentrations to a 50% decrease in concentrations; a majority of the studies showed reductions between 0 and 40%. If we take the mid-points of the 0-40% range and the 40-75% range, we estimate reductions on the order of 20-60% within the first 100 meters of a road, assuming a barrier of typical height (e.g., 6 m) set roughly at the edge of the road shoulder, and assuming that winds are perpendicular to the roadway. Using a similar approach, we estimate reductions within the first 50 m to be on the order of 25-65%. Differences between modeling and measurement studies, as well as differences among findings from studies examining the same pollutant type, indicate there are substantial uncertainties in the quantitative effects of barriers. Furthermore, effects from an actual barrier will be a function of many site-specific factors.

7.2.2 Modeling Results

R-Line Model

The R-LINE model is a research-grade dispersion model developed by the EPA Office of Research and Development (ORD) for assessing air quality impacts near roadways. R-LINE is based on a steady-state Gaussian formulation and is designed to simulate emissions from line-type sources (e.g., mobile sources along roadways) by numerical integration of point source emissions. R-LINE models the dispersion of pollutants released near the surface using vertical and lateral dispersion rates based on field and wind tunnel study measurements. The model uses surface meteorology developed using the AERMET meteorological data preprocessor. R-LINE is optimized for flat roadways; however, beta-option algorithms are available for simulating the effects of complex roadway configurations, such as roadside barriers and depressed roadways, in the publicly available version 1.2. The performance of the base version of the R-LINE model (without beta-option barrier or depressed roadway configurations selected) has been evaluated by comparison with other Gaussian dispersion models (AERMOD, CALINE) and with near-road concentrations from independent field studies (e.g., Heist et al., 2013; Snyder and Heist, 2013). At the time this work was completed, R-LINE was not an EPA-approved model for PM hot-spot analyses.

At the time this work was performed, the publicly available version of R-LINE was version 1.2 (**https://www.cmascenter.org/r-line/**); however, version 2.0 was under active development. Model enhancements in Version 2.0 include

- The option to use hourly varying emission rates
- The option to model multiple pollutants
- Three optional NO_x chemistry algorithms
- An improved barrier algorithm to simulate increased mixing downwind of the barrier due to the change in flow patterns induced by the barrier, consistent with field study observations
- Updated input file formats
- Improved computational speed

Given the expected improvement in barrier algorithm performance over Version 1.2, EPA recommended that we use Version 2.0 for our modeling.[20] Thus, Version 2.0 was used for the analyses described below. At the time this study was performed, the depressed roadway algorithm was not yet available in R-LINE v2.0.

[20] Source: David Heist, Research Physical Scientist, U.S. Environmental Protection Agency; personal communication with Steven G. Brown, February 2016.

The R-LINE barrier algorithms were derived from Schulte et al. (2014), and are based on the observation of increased mixing downwind of the barrier due to the change in flow patterns induced by the barrier. Barriers affect dispersion of roadway emissions by: (1) increasing vertical dispersion through turbulence generated in the wake of the barrier, (2) inducing vertical mixing behind the barrier in the cavity region, and (3) lofting the emissions plume above the barrier (Schulte et al., 2014). The effects of the barrier and changes to vertical lofting were expected to be largest during perpendicular wind conditions; therefore, the underlying formulation of R-LINE barrier algorithms was based on experimental data sets and wind tunnel studies that focused on wind directions close to perpendicular to the road (Schulte et al., 2014). Prior work supported by EPA found that R-LINE performance was superior when winds were ±40 degrees of perpendicular to the road (Snyder and Heist, 2013; Schulte et al., 2014). In this study, we limited our analysis to perpendicular wind directions within 40 degrees of perpendicular, for consistency with previous work.

Model Setup

Model scenarios were applied to a 10-lane highway, with five 3.6 m lanes in either direction. The modeled roadway was 1.1 miles long and was configured in the southwest to northeast direction such that the predominant wind patterns in the meteorological data were aligned perpendicular to the roadway. A five-year AERMET (version 15181) meteorological data set for 2010−2014 processed by the San Joaquin Valley Air Pollution Control District[21] for the Fresno, California, airport was used.

Figure 42 illustrates the roadway setup for modeling. Figure 43 shows the side view of the roadway and barrier configuration, along with the predominant wind direction. A single barrier was located downwind of the road 3.05 meters from the roadway edge. There was no barrier on the upwind side of the road. The wind rose in Figure 44 shows wind speeds and directions in the five-year Fresno AERMET meteorological data set. The perpendicular wind direction was set to 315 degrees because the winds originate from the northwest a majority of the time in the data set. We limited our analysis to conditions when wind directions were within 40 degrees of perpendicular to the roadway line source (i.e., an 80 degree arc from 275 to 355 degrees in the Fresno meteorological data set).

For model scenarios including a barrier, a barrier the same length as the roadway (1.1 miles) was located 21.05 m southeast (downwind) of the roadway centerline, or 3.05 m from the edge of the road. This location was selected to represent an example of the minimum lateral clearance between a barrier and edge of the travel way allowable in state DOT sound wall design specifications, and therefore represents a practical implementation example (California Department of Transportation, 2006). A barrier extending the full length of the roadway was modeled to minimize potential influences from edge effects discussed in the literature.

Model receptors were located 1, 3, 5, 10, 15, 20, 25, 30, 35, 40, 45, 50, 75, 100, 150, 200, 250, and 300 meters downwind of the barrier, extending out from the center of the road, at 1.8 meters in height to

[21] http://www.valleyair.org/busind/pto/tox_resources/airqualitymonitoring.htm#met_data.

represent concentrations near ground level (U.S. Environmental Protection Agency, 2015c). Hourly average concentrations for each receptor were modeled.

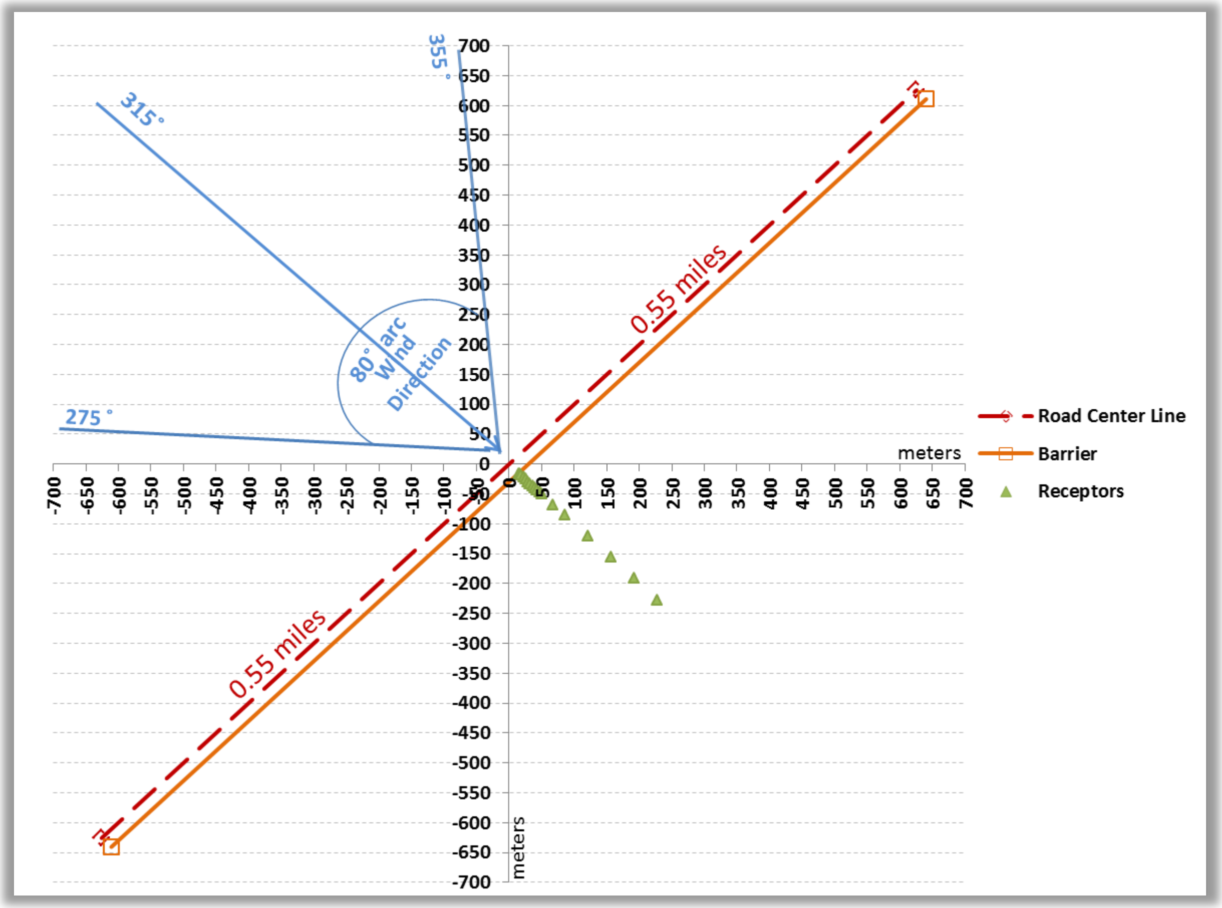


Figure 42. Aerial view illustrating the configuration of the modeled roadway and predominant wind direction relative to the barrier and receptors.

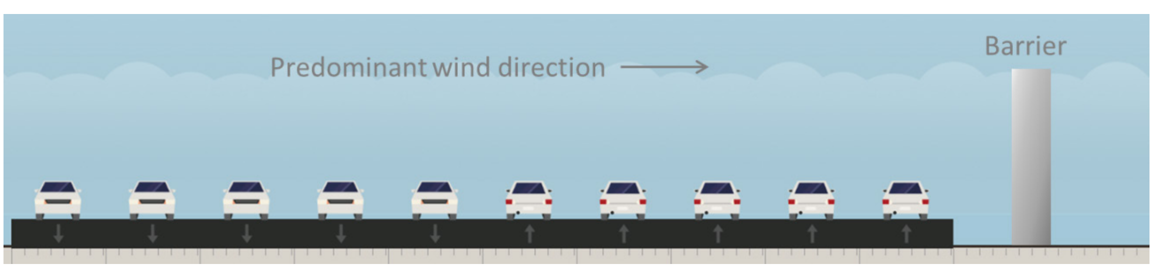


Figure 43. Side view illustrating the configuration of the modeled roadway and predominant wind direction relative to the barrier. Image created using STREETMIX (http://streetmix.net).

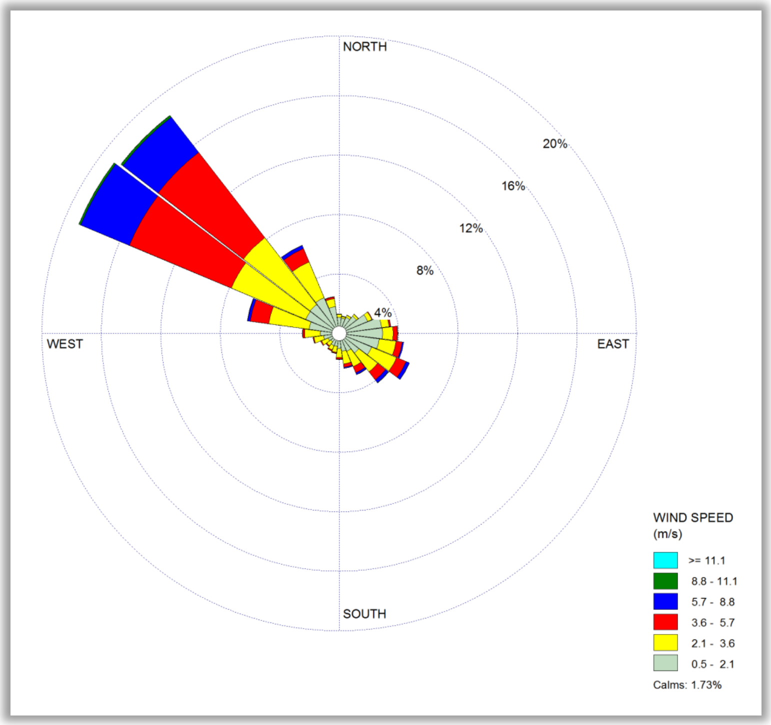


Figure 44. Wind rose illustrating predominant wind speeds and directions in the Fresno AERMET data set (2010–2014) used in this study.

The ratio of displacement height to roughness length (f-factor) was set to 7.7 using the average surface roughness in the AERMET file and the displacement height from Table 2 of the R-LINE v1.2 guidance document (Snyder and Heist, 2013). The numerical method option was employed for all scenarios, using the default total plume plus meander option. The average weighted vehicle height was based on 8% truck and 20% truck scenarios (1.73 m and 2.02 m respectively), based on the EPA 2015 Transportation Conformity Guidance for PM Hot-Spot Analyses, Appendix J (U.S. Environmental Protection Agency, 2015c). Corresponding initial plume dispersion (sigma-z) parameters were 1.37 m for the 8% truck scenarios (Scenarios 0–3) and 1.60 m for the 20% truck scenarios (Scenarios 4 and 5).

Six model scenarios were developed to examine the effects of barrier height and average vehicle height on pollutant concentrations in the near-road environment. For each of these scenarios, the influences of meteorological conditions were also examined. Table 21 summarizes model scenarios generated for this study.

Table 21. Model scenarios examined in this work.

Scenario	Modeling Factor	Barrier Height (m)	Average Vehicle Height (m)[a]
0	Base Case	No barrier	1.73
1	Barrier Height	2.5	1.73
2		5	1.73
3		7.5	1.73
4	Average Vehicle Height (Fleet Mix)	No barrier	2.02
5		5	2.02

[a]Average vehicle height of 1.73 m represents a fleet mix of 8% trucks and 92% cars. Average vehicle height of 2.02 m represents a fleet mix of 20% trucks and 80% cars.

Modeled Outcomes

Figure 45 shows the average normalized pollutant concentrations as a function of distance from the barrier when winds are within 40 degrees of perpendicular of the roadway with (1) no barrier, (2) a 2.5 m barrier, (3) a 5 m barrier, and (4) a 7.5 m barrier (Scenarios 0 through 3). Concentrations are normalized by the concentration for the no-barrier scenario at 1 m from the barrier location. Concentration gradients are shown at a 1.8 m receptor height, which EPA uses to represent a typical breathing height for an adult human. In the absence of a barrier, pollutant concentrations originating from the roadway decrease by 50% within the first 100 m of the road. The model predicts that the presence of a barrier reduces the maximum concentrations downwind of the roadway; the taller the barrier, the larger the reduction. In the presence of a 2.5 m barrier, concentrations are estimated to be reduced by approximately 17−24% relative to the no-barrier case.

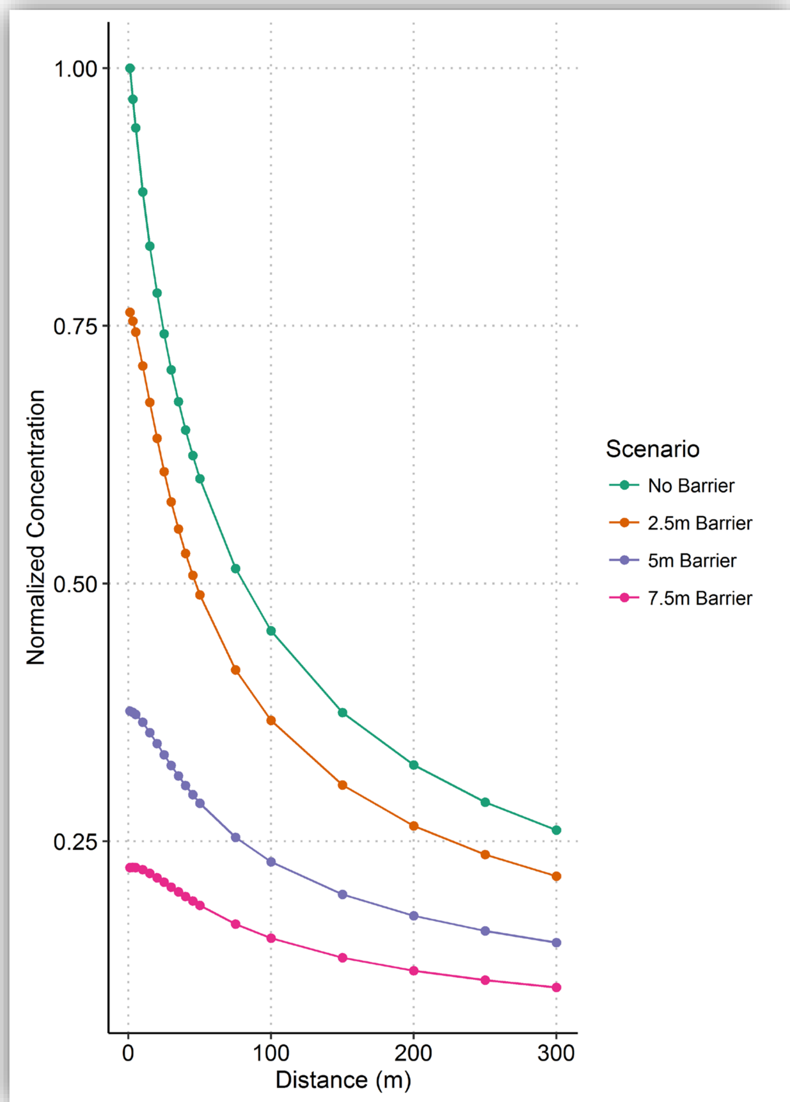


Figure 45. Average normalized concentration by receptor distance from the barrier for model scenarios with no barrier and with a 2.5 m, 5 m, and 7.5 m barrier, for winds within 40 degrees of perpendicular to the barrier (275 to 355 degrees). All concentrations are normalized relative to the modeled concentration for the no-barrier scenario at 1 meter from the barrier location. Distances are measured from the barrier's location at 3.05 m from the road edge.

Figure 46 shows the ratio of concentrations for each scenario with a barrier to concentrations for the no-barrier scenario (i.e., the decrease in modeled concentrations when a barrier is next to the roadway compared to when there is no barrier). Table 22 summarizes the modeled average percent concentration reduction by barrier height and distance from the barrier, compared to the no-barrier scenario. Concentrations at 1 m from the barrier are reduced by 24% for a 2.5 m barrier, by 62% for a 5 m barrier, and by 78% for a 7.5 m barrier, compared to the no-barrier case. For the 2.5 m barrier, the concentration reduction relative to the no-barrier case is greatest in the wake of the barrier, decreasing with increasing distance to approximately 25 meters, and then increasing by a few percent out to approximately 100 m before leveling off at a reduction of approximately 19%. The 5 m and 7.5 m barriers exhibit steeper percent reduction gradients for the first 50 m and then shift towards a more gradual slope of reduction relative to the no-barrier case.

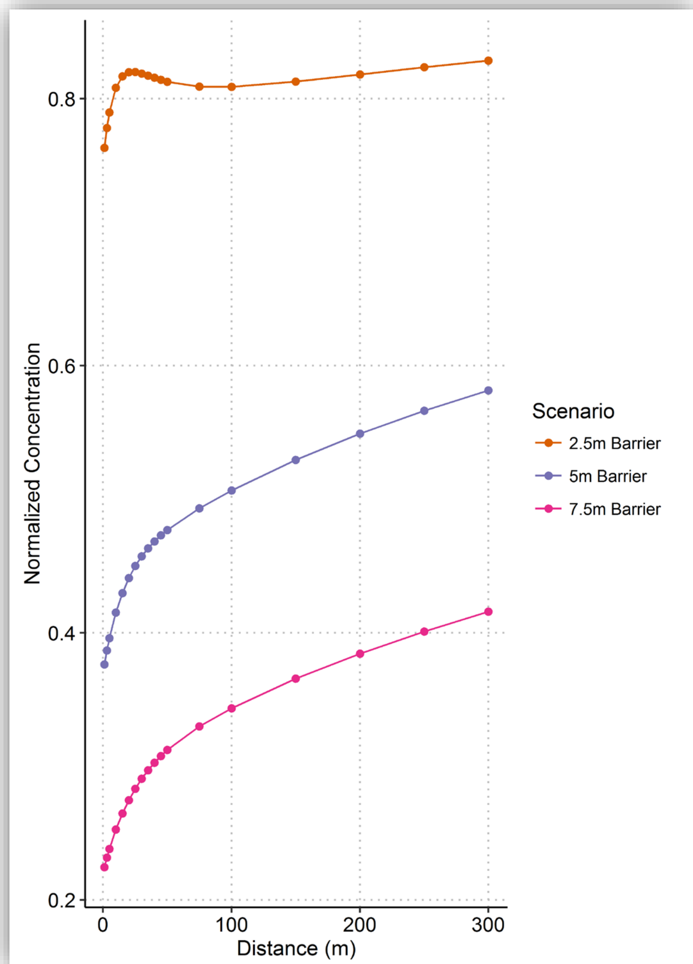


Figure 46. Ratio of modeled concentrations in the presence of a 2.5 m, 5 m, and 7.5 m barrier relative to the no-barrier case by receptor distance from the barrier, with winds within 40 degrees of perpendicular to the barrier (275 to 355 degrees).

Table 22. Modeled average percent reduction in concentrations at receptors due to a barrier relative to the no-barrier case, with winds within 40 degrees of perpendicular to the barrier (275−355 degrees).

Receptor Distance (m)	% Reduction Relative to No-Barrier Scenario		
	2.5 m Barrier	5 m Barrier	7.5 m Barrier
1	24	62	78
3	22	61	77
5	21	60	76
10	19	58	75
15	18	57	74
20	18	56	73
25	18	55	72
30	18	54	71
35	18	54	70
40	18	53	70
45	19	53	69
50	19	52	69
75	19	51	67
100	19	49	66
150	19	47	63
200	18	45	62
250	18	43	60
300	17	42	58

Figure 47 shows the ratio of results for a 1.73 m average vehicle height versus a 2.02 m average vehicle height, for the no-barrier and 5 m barrier cases (a ratio of less than one indicates lower concentrations for a 2.02 m vehicle height). On average, modeled concentrations for the 1.73 m and 2.02 m scenarios are very similar. Concentrations for the 2.02 m vehicle height were from 3% (5 m barrier) to 5% (no barrier) lower at 1 m from the roadway than concentrations for the 1.73 m average vehicle height. By 300 m from the road, the average difference in concentrations between the 2.02 m and 1.73 m vehicle heights is approximately 2% for both the no-barrier and 5 m barrier cases.

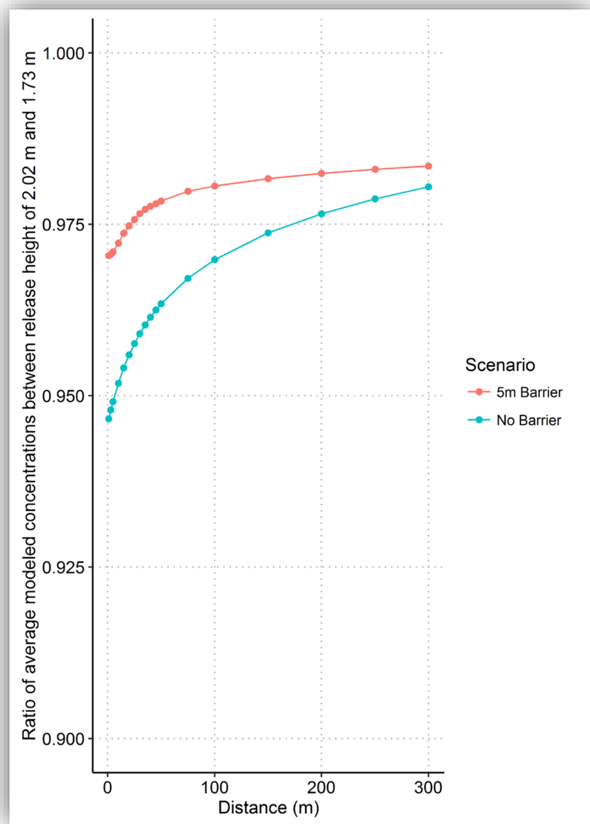


Figure 47. Ratio of average modeled concentrations for average vehicle heights of 2.02 m and 1.73 m by receptor distance for the no-barrier and 5 m barrier cases.

Conclusion: Literature Compared to Modeled Outcomes

The literature findings indicate that barriers typically reduce pollutant concentrations on the order of 20−60% within the first 100 m and approximately 25−65% within the first 50 m downwind of a 6 m barrier, although actual effects will be a function of many site-specific factors. The exploratory modeling results obtained here were for 2.5 m, 5 m, and 7.5 m barrier cases. The 5 m results from this study fall within the range of barrier reductions relative to no-barrier reported in the literature for similar barrier heights, at distances from the barrier ranging from approximately 25 m to 150 m. Modeled reductions for a 5 m barrier within approximately 25 m of the barrier were lower in this study (~55-60%) than in a majority of previous modeling studies (~75-80%), whereas modeled reductions at 200-250 m from this study were higher (~45%) than those from previous modeling studies (~10-20%).

8. Conclusions: Findings and Research Needs

8.1 Major Technical Findings

Highlights of the major findings from the five-year TPF program include the following:

1. Near-road concentrations of CO and NO_2 were not problematic when benchmarked against the existing NAAQS.

2. A small number of locations exceed the 24-hr or annual $PM_{2.5}$ NAAQS.

3. Near-road $PM_{2.5}$ concentrations are likely trending downward, however these findings are based on only eight sites operating over the multiple years covered by this work. More data is becoming available each year to help establish multi-year trends across the entire network of $PM_{2.5}$ sites.

4. The upper bound of $PM_{2.5}$ increments from the year-2017 data evaluated here, for 20 sites across the United States, was 2.04 $\mu g/m^3$. Only three sites had an increment greater than 1.44 $\mu g/m^3$; monitors at each of these three sites were sited less than 10 meters from the roadway.

5. Over time, an increasing number of $PM_{2.5}$ nonattainment areas is expected to achieve the NAAQS. Therefore, $PM_{2.5}$ hot-spot analyses will increasingly be completed in maintenance areas where a "buffer" exists between the background concentration and the NAAQS. The work completed by the TPF provides a better understanding of the likely increment from proposed projects. The increment findings can contribute to interagency consultation and determinations as to whether projects are POAQC and should undergo quantitative hot-spot analysis.

6. Findings illustrate the substantial forecasted impacts of fleet turnover. For example, for a constant truck percentage of 8%, an overall AADT of approximately 360,000 (EMFAC) to 500,000 (MOVES) vehicles in 2020 would have $PM_{2.5}$ impacts similar to 125,000 vehicles in 2006. For a constant truck percentage of 8%, an overall AADT of approximately 420,000 (EMFAC) to 1.6 million (MOVES) vehicles in 2035 would have $PM_{2.5}$ impacts similar to 125,000 vehicles in 2006.

7. There can be substantial differences between measured and modeled near-road $PM_{2.5}$ concentrations. Based on the case study work completed here, modeled concentrations over-predict measured values; this outcome was exaggerated the closer to the road the modeling was meant to represent.

8. Several insights were generated to assist with situations where mitigation is needed. Near-road barriers offer promise as a method of reducing concentrations in the geographic zones immediately downwind of major roads. Special attention needs to be placed on avoiding unintended consequences during barrier placement, such as increasing concentrations at receptors close to the edge of barriers. Truck replacements also offer promise as a method of accelerating fleet turnover at project locations such as ports where truck activities are concentrated. Truck replacement implementation needs to address potential challenges such as the availability and use of target vehicles, outreach to vehicle owners, and eligibility screening to ensure replaced trucks result in emissions reductions in the areas of concern.

8.2 Future Research Needs

Highlights of important recommended future research includes the following:

1. Future work is needed to assess the relative contribution of exhaust and non-exhaust $PM_{2.5}$ emissions, and to refine understanding of non-exhaust emission changes over time. Measurements of speciated $PM_{2.5}$ components (including black carbon, metals, and crustal minerals) in the near-road environment would help constrain uncertainties associated with the relative contribution of exhaust and non-exhaust emissions, and would help resolve related differences that exist between the EMFAC and MOVES modeling tools.
2. The key findings related to comparisons between modeled and measured near-road concentrations were consistent across both study sites examined (Indianapolis and Providence), 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.
3. Assessments are needed to compare and evaluate modeled and measured near-road inert pollutant concentrations, such as those for CO. Near-road CO concentrations are far below the NAAQS and not problematic; the purpose of this work would be to use CO to help further refine understanding of modeling chain biases contributing to differences between measured and modeled $PM_{2.5}$ concentrations. CO analyses could provide unique insights since modeled CO emissions from vehicles are expected to be more certain than modeled $PM_{2.5}$ emissions, given the $PM_{2.5}$ findings discussed in this report. This work would involve evaluating modeled and measured near-road CO increments across different geographic settings, roadway types, and configurations.
4. In the modeled compared to measured work done here, 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 98^{th} 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. Given the interest in and potential benefits of the use of barriers to reduce near-road air pollutant concentrations, several studies could be performed to build upon the TPF's exploratory work and improve the quantitative estimates of near-road barrier effects. Examples include:

    A. A field study could be designed and implemented to further examine and quantify edge effects. A study could examine areas spatially affected by barrier edges to better define how far a barrier would need to extend to minimize the influence of edge effects on downwind receptors and thereby safeguard against unintended pollutant concentration consequences from barrier use.

    B. The number of modeling scenarios that could be examined as part of this study was limited because the R-LINE model was still under active development at the time this work was underway. Additional modeling scenarios and sensitivity analyses could be completed to compare modeled outcomes of barrier effects with previous measurement, wind tunnel, and other modeling studies. Such analyses would enhance understanding of model performance and the potential effects of roadside barriers on downwind concentrations.

    C. Of particular interest are the effects of depressed or elevated roadway configurations. Given the potential indicated by previous studies for a mitigation enhancement associated with a depressed roadway configuration, future modeling could be performed to assess the effects of varying roadway depression depth.

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