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# Diurnal and Seasonal Variations of NO, NO<sub>2</sub> and PM<sub>2.5</sub> Mass as a Function of Traffic Volumes Alongside an Urban Arterial

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1 **Diurnal and seasonal variations of NO, NO<sub>2</sub> and PM<sub>2.5</sub> mass as a function of traffic volumes**  
2 **alongside an urban arterial**

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## 35 Abstract

36 Urban arterial corridors are landscapes that give rise to short and long-term exposures to  
37 transportation-related pollution. With high traffic volumes and a wide mix of road users, urban  
38 arterial environments are important targets for improved exposure assessment to traffic-related  
39 pollution. A common method to estimate exposure is to use traffic volumes as a proxy. The study  
40 presented here analyzes a unique yearlong dataset of simultaneous roadside air quality and traffic  
41 observations for a U.S. arterial to assess the reliability of using traffic volumes as a proxy for  
42 traffic-related exposure. Results show how the relationships of traffic volumes with NO and  
43 NO<sub>2</sub> vary not only by time of day and season but also by time aggregation. At short-term  
44 aggregations (15 minutes) nitrogen oxides were found to have a significant linear relationship  
45 with traffic volumes during morning hours for all seasons although variability was still high ( $r^2 =$   
46  $0.1 - 0.45$  NO,  $r^2 = 0.14 - 0.27$  NO<sub>2</sub>), and little to no relationship during evening periods ( $r^2 < 0.01 -$   
47  $0.03$  NO,  $r^2 < 0.01 - 0.05$  NO<sub>2</sub>). Comparisons with coarse annual results validate the use of traffic  
48 volumes to estimate annual exposure concentrations for morning periods ( $r^2 = 0.89$  NO,  $r^2 = 0.87$   
49 NO<sub>2</sub>) and evening NO<sub>2</sub> ( $r^2 = 0.46$ ). Traffic volumes are a weak or poor predictor for annual  
50 evening NO ( $r^2 = -0.09$ ) and short-term 15 minute aggregations. Seasonal and diurnal  
51 characterizations show that roadside PM<sub>2.5</sub> (mass) measurements do not have a relationship with  
52 local traffic volumes, leading us to conclude that PM<sub>2.5</sub> mass is more tied to regional sources and  
53 meteorological conditions. As epidemiology and personal exposure assessment research aims to  
54 study health impacts and pollutant levels encountered by pedestrians, bicyclists, those waiting for  
55 transit, and other road users, these results show when traffic volumes alone can be a reliable  
56 proxy for exposure and when this approach is not warranted.

57  
58 **Keywords:** Urban, arterial, roadside, PM<sub>2.5</sub>, nitrogen oxides, exposure assessment

## 59 1. Introduction

61 Transportation-related emissions are a significant component of poor air quality  
62 exposures within urban areas. Increasing urbanization worldwide (United Nations 2014) and  
63 growth in high density development (Schneider and Woodcock, 2008) will both lead to more  
64 people who reside, work, attend school, and commute within a near-road environment. More  
65 than four-fifths of the United States population currently resides in metropolitan areas and from  
66 2000 to 2010 population growth rates for metropolitan areas were greater than the national rate  
67 (U.S. Census Bureau 2012). Elevated concentrations of traffic-related pollution such as nitric  
68 oxide (NO), nitrogen dioxide (NO<sub>2</sub>), and particle number concentrations (PNC) surrounding  
69 major roadways has been documented (Karner et al., 2010; Rao et al., 2014; Zhou and Levy,  
70 2007). Adverse respiratory and cardiovascular effects for populations living within this near  
71 roadway environment have been shown through epidemiology and toxicology (Brugge et al.,  
72 2007; Health Effects Institute, 2010; Kim et al., 2008). Short-term exposures as experienced by  
73 drivers, vehicle occupants, or pedestrians are also associated with short-term morbidity and

74 negative health responses (McCreanor et al., 2007; Peters et al., 2013, 2004).

75 The United States Environmental Protection Agency (U.S. EPA), in recognition of the  
76 heterogeneous pattern of traffic-related pollution and health impacts, has made the one hour NO<sub>2</sub>  
77 National Ambient Air Quality Standard more stringent and mandated roadside monitoring of  
78 NO<sub>2</sub> in large population centers (CFR, 2010). This new roadway monitoring network is  
79 primarily sited near major highways and will also include measurements of PM<sub>2.5</sub> mass and CO  
80 (Watkins and Baldauf, 2012). Major urban arterial roadways represent 10% of total U.S. road  
81 mileage, but account for 48% of vehicle miles traveled (US DOT, 1998). Urban arterial systems  
82 have high traffic volumes, but unlike highways, accommodate a high compositional mix of road  
83 users such as those waiting for transit, pedestrians, and bicyclists. A variety of land uses  
84 including residential, school, park, or commercial are located directly at the road edge in these  
85 transportation microenvironments. Close proximity to traffic results in higher exposure  
86 concentrations to traffic-related pollution for all road users (Kaur et al., 2007; Kendrick et al.,  
87 2011; Thai et al., 2008). There is also potential for increased uptake of traffic-related pollution  
88 for bicyclists and pedestrians due to increased respiration rates or longer travel times (McNabola  
89 et al. 2008; Kaur et al. 2007; Bigazzi et al. 2014). The urban arterial microenvironment is an  
90 important feature of cities and a key location for increased exposures to traffic-related pollution  
91 but remains an under-measured microenvironment.

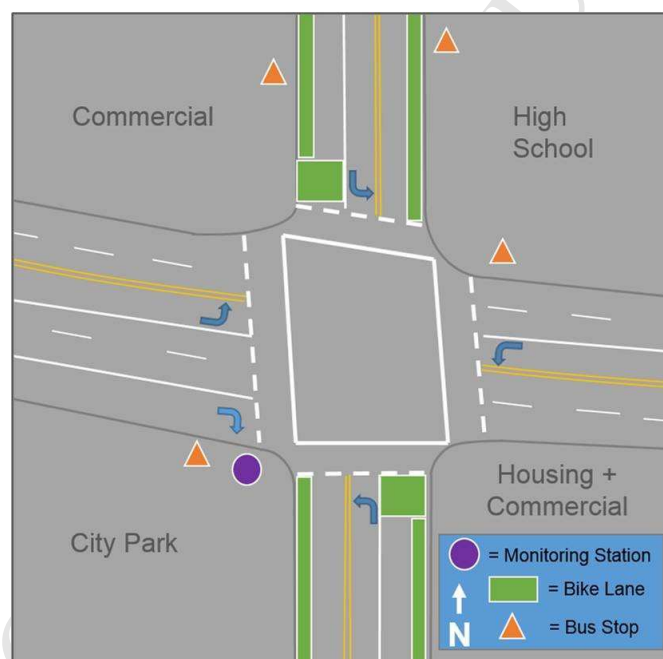
92 To our knowledge, the only near road air quality studies using a long-term curbside  
93 station have been conducted outside of North America and primarily within European Union  
94 countries with different fuel use and vehicle fleets than U.S. cities. Near-road air quality research  
95 in the U.S has provided critical data about the spatial extent of traffic-related pollution  
96 surrounding highways, emission factors under real-world driving conditions, and the contribution  
97 of in-car exposure to total exposure using mobile measurements made within the roadway (Cho  
98 et al., 2009; Clements et al., 2009; Fruin et al., 2008; Kimbrough et al., 2013; Richmond-Bryant  
99 et al., 2009; Zhang et al., 2004; Zhu et al., 2004). This study uses long-term curbside monitoring  
100 (1 year) to assess the magnitude and temporal characteristics of roadside traffic-related pollution  
101 alongside a heavily-trafficked arterial that accommodates pedestrians, transit users, cyclists,  
102 schools, businesses, and homes at the road edge of a major arterial in Portland, Oregon. The  
103 traffic data collection was facilitated through a partnership with the City of Portland, Portland  
104 Bureau of Transportation.

105 The aim of this project is to use roadside measurements to characterize diurnal and  
106 seasonal variations of NO, NO<sub>2</sub>, and PM<sub>2.5</sub> as a function of traffic volumes. Such  
107 characterizations directly improve understanding of ambient surrogates of traffic for the U.S.  
108 urban arterial microenvironment. Characterization by time of day and seasons is important as  
109 epidemiological studies incorporate time activity patterns within cohort studies to understand  
110 exposure beyond annual mean concentrations (Adar and Kaufman, 2007). The promotion of  
111 active transportation modes such as bicycling and walking also increases the need for  
112 understanding short-term concentrations to improve exposure evaluation for these travel modes

113 (de Nazelle et al., 2011; Tiwary et al., 2011). Lastly, our analysis has implications for using  
114 traffic volumes as a proxy for transportation-related emissions. For example, land use regression  
115 (LUR) typically uses proximity to road type or traffic volumes to estimate exposure (Hoek et al.  
116 2008). As LUR models are developed to predict more temporally resolved exposure estimates  
117 and health studies move towards shorter term exposure and impact assessments, the limitations  
118 of traffic volumes as a proxy highlighted here are important to note.

## 119 2. Methods

120 An air quality monitoring station was established at a high traffic intersection that  
121 includes a diverse mix of road users (freight, public transit, pedestrians, cyclists and passenger  
122 vehicles) in Portland, Oregon USA (southwest corner of SE Powell Boulevard and SE 26<sup>th</sup>  
123 Avenue, Figure 1). SE Powell Boulevard is a major arterial roadway that runs east/west with  
124 peak hourly traffic volumes of 2,800 vehicles and 28,000 Average Annual Daily Traffic  
125 (AADT), including 6% trucks on weekdays.



126

127 **Figure 1 Schematic of study intersection. Air quality station (purple circle) is located on the**  
128 **SW corner of SE Powell Blvd which runs east/west and SE 26<sup>th</sup> Ave runs north/south**

129

130 Pedestrian activity through the intersection is high due to the school, housing, a city park,  
131 and access to bus stops for major north-south and east-west bus routes. A snapshot traffic survey  
132 was conducted by the City of Portland for two hour periods each in the morning (including  
133 school start) and evening on one day in February 2012 (PortlandMaps 2014). Total pedestrians  
134 crossing the intersection, not including individuals waiting at bus stops, was 225 and 157 for

135 morning and evening two hour period. The two hour morning bicycle count for all crossings was  
136 111 bikes and the evening count was 106 bikes. City bicycle data shows summer biking levels to  
137 range from 1.8-2.4 times higher than bike counts for the month of February which is cold and  
138 wet for the Portland area (Bike Barometer: Hawthorne Bridge 2014).

139 Traffic volumes in fifteen minute bins for each lane in the intersection were collected  
140 using inductive loop detectors and the Sydney Coordinated Adaptive Traffic System (SCATS)  
141 infrastructure. SCATS is an adaptive signal system that operates on a 6 km stretch of Powell  
142 Boulevard including the study intersection (Kendrick et al., 2014). NO and NO<sub>2</sub> were monitored  
143 using a Teledyne T200 chemiluminescence analyzer and PM<sub>2.5</sub> using a TSI DRX DustTrak  
144 monitor. Equipment is housed in a pole mounted traffic signal cabinet with a sampling inlet  
145 placed 2.5 m above the sidewalk and connected to the inlet with non-reactive sampling lines. The  
146 height of the inlet ensures that intakes are out of reach of disturbance from the street (due to  
147 nearby busy bus stop) but still captures road emissions at the street level. Prior to installation,  
148 needed lengths of tubing for each instrument were tested to confirm that there was no significant  
149 loss in pollutants. Calibration of the NO<sub>x</sub> analyzer was performed throughout the study period  
150 using certified standard gases and calibration of the DustTrak was conducted through factory  
151 calibration with zero and flow checks performed on site, with zero checks performed  
152 approximately every 14 days.

153 Additional data collected at the intersection includes wind speed and direction (RM  
154 Young 3D Sonic Anemometers Model 81000), temperature and relative humidity (RM Young  
155 Probe Model 41382VC). Wind data was also supplemented with measurements from the Oregon  
156 Department of Environmental Quality's (DEQ) regional monitoring station. The DEQ site is  
157 located 3.4 km east of the roadside station and 0.09 km south of Powell Boulevard. Continuous  
158 air quality (NO, NO<sub>2</sub>, PM<sub>2.5</sub>) and meteorology data from this background station is collected,  
159 stored and accessed through Horizons database (George et al., 2005). The DEQ site instruments  
160 and calibration procedures follow federal monitoring guidelines. All data referred to as "urban  
161 background" is from this DEQ station.

162 Data analysis presented here is based on one year of monitoring including a total of 362  
163 days of NO<sub>x</sub> and 289 days of PM<sub>2.5</sub> measurements at 30-second intervals (January 01, 2013-  
164 December 31, 2013). Total sampling points for PM<sub>2.5</sub> were limited by flow and data storage  
165 issues as well as time off-site for factory calibration. For quality assurance/quality control  
166 (QA/QC), measurements made outside of each instrument's operating temperature range based  
167 on continuous monitoring of the cabinet temperature were excluded from further analysis.  
168 Photometers such as the DustTrak can overestimate ambient particulate matter concentrations  
169 due to particle composition, density, morphology, and relative humidity due to hygroscopicity. A  
170 correction factor of 0.5 was applied to the DustTrak measurements based on a comparison study  
171 in Portland between DustTrak and gravimetric measurements of ambient PM<sub>2.5</sub> (Zhu et al.,  
172 2011). Data presented is aggregated to 15 minutes to allow for direct comparisons with 15  
173 minute binned traffic volumes. Analysis uses measurements only when the wind direction was

174 coming from the road. This will be referred to as the road wind direction bin which includes 270°  
175 (W) to 365° (N) and 0° (N) to 125° (SE) in order to capture arterial traffic influences. Data  
176 analysis was conducted using R statistical language and the lattice and quantreg packages (R  
177 Core Team 2013, Sarkar 2008, Koenker 2015).

178

### 179 **3. Results and Discussion**

#### 180 **3.1 Traffic and Pollutant Relationships**

181 Mean, median, 5<sup>th</sup> and 95<sup>th</sup> percentile values of roadside NO, NO<sub>2</sub>, PM<sub>2.5</sub> and traffic  
182 volumes are presented in Table 1 to show distributions of measurements and compare to urban  
183 background concentrations. Data is separated by weekdays and weekends. For each pollutant,  
184 roadside measurements show elevated peak concentrations (95<sup>th</sup> percentiles) compared to  
185 background levels, demonstrating an increase in measured pollutant levels for the roadside over  
186 urban background. Over the entire monitoring period, roadside weekday and weekend means are  
187 significantly greater than urban background means for all three pollutants (t-values ranging from  
188 4.4 to 32, all p-values <0.01 using a data subset in which serial correlation has been reduced).

189

190 Roadside NO<sub>2</sub> and PM<sub>2.5</sub> show smaller differences from urban background levels than  
191 roadside NO. This is not unexpected for NO<sub>2</sub> since the rate of secondary formation of NO<sub>2</sub>  
192 directly at the roadside would vary depending on mechanical turbulence, dispersion by wind, and  
193 existing concentrations of NO, ozone (O<sub>3</sub>, primary oxidant for NO to NO<sub>2</sub> conversion) and other  
194 chemical species. Additionally, the urban background site is located within the 300m buffer that  
195 NO and NO<sub>2</sub> are typically elevated above background concentrations from the major road  
196 (Karner et al., 2010). The average NO/NO<sub>2</sub> ratio for the urban background site is 0.7 while the  
197 average NO/NO<sub>2</sub> ratio at the roadside site is 1.6 showing a higher proportion of NO or freshly  
198 emitted pollutants as expected directly at a roadside.

199

200 Mean weekday and weekend PM<sub>2.5</sub> roadside concentrations were found to be  
201 significantly higher than background levels by a mean of the differences equal to only 2 µg/m<sup>3</sup>  
202 (weekdays) and 1 µg/m<sup>3</sup> (weekends). The DustTrak has a resolution of 1 µg/m<sup>3</sup> so the statistically  
203 significant difference is just above or at instrument resolution. Weekend mean and 95<sup>th</sup> percentile  
204 PM<sub>2.5</sub> concentration for both the roadside and urban background site are slightly higher than  
205 weekdays despite higher traffic volumes on weekdays. Table 1 does not show a strong local  
206 signal for roadside PM<sub>2.5</sub> measurements compared to urban background.

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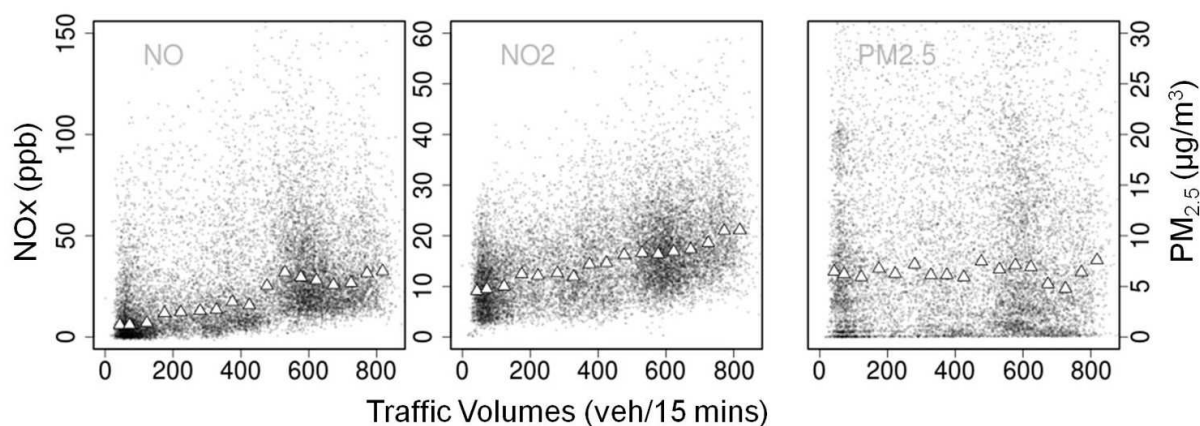
213  
214  
215**Table 1 Roadside and urban background comparisons for weekdays and weekends, spanning Jan 2013- Dec 2014 at 15 minute aggregations.**

Pollutant	Weekdays		Weekends	
	Roadside Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)	Urban Background Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)	Roadside Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)	Urban Background Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)
NO (ppb)	28 20 (1.3- 79)	9 2 (0.4 - 45)	18 10 (1 – 59)	8 2 (0.4- 38)
NO <sub>2</sub> (ppb)	16 15 (5 - 31)	11 9 (3 – 24)	11 10 (4 – 23)	9 7 (2 -22)
PM <sub>2.5</sub> (µg/m <sup>3</sup> )	9 6 (0 – 28)	7 5 (2-20)	10 5 (0 – 32)	8 5 (2 – 22)
Traffic Volumes/ 15 mins	493 554 (65-888)		395 407 (74- 735)	

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To further investigate the roadway impact, the relationship of each pollutant versus traffic volumes for all weekdays and road wind direction bin is shown in Figure 2. The rest of the data analysis presented will focus on weekdays only when traffic volumes and exposure concentrations are highest and also because weekends show separate diurnal patterns. The median relationship between each pollutant concentration and vehicle volumes binned by fifty is highlighted by triangles in Figure 2. These relationships show increasing NO and NO<sub>2</sub> with increasing traffic volumes and little change in PM<sub>2.5</sub> mass as a function of traffic at the intersection. Roadside NO and NO<sub>2</sub> are markers of the increased roadway emissions while roadside PM<sub>2.5</sub> mass is not responsive to local traffic volumes. For all three pollutants, variance is high as highlighted in Figure 2. How this variability changes by season and time of day is investigated next using seasonal and diurnal characterizations.

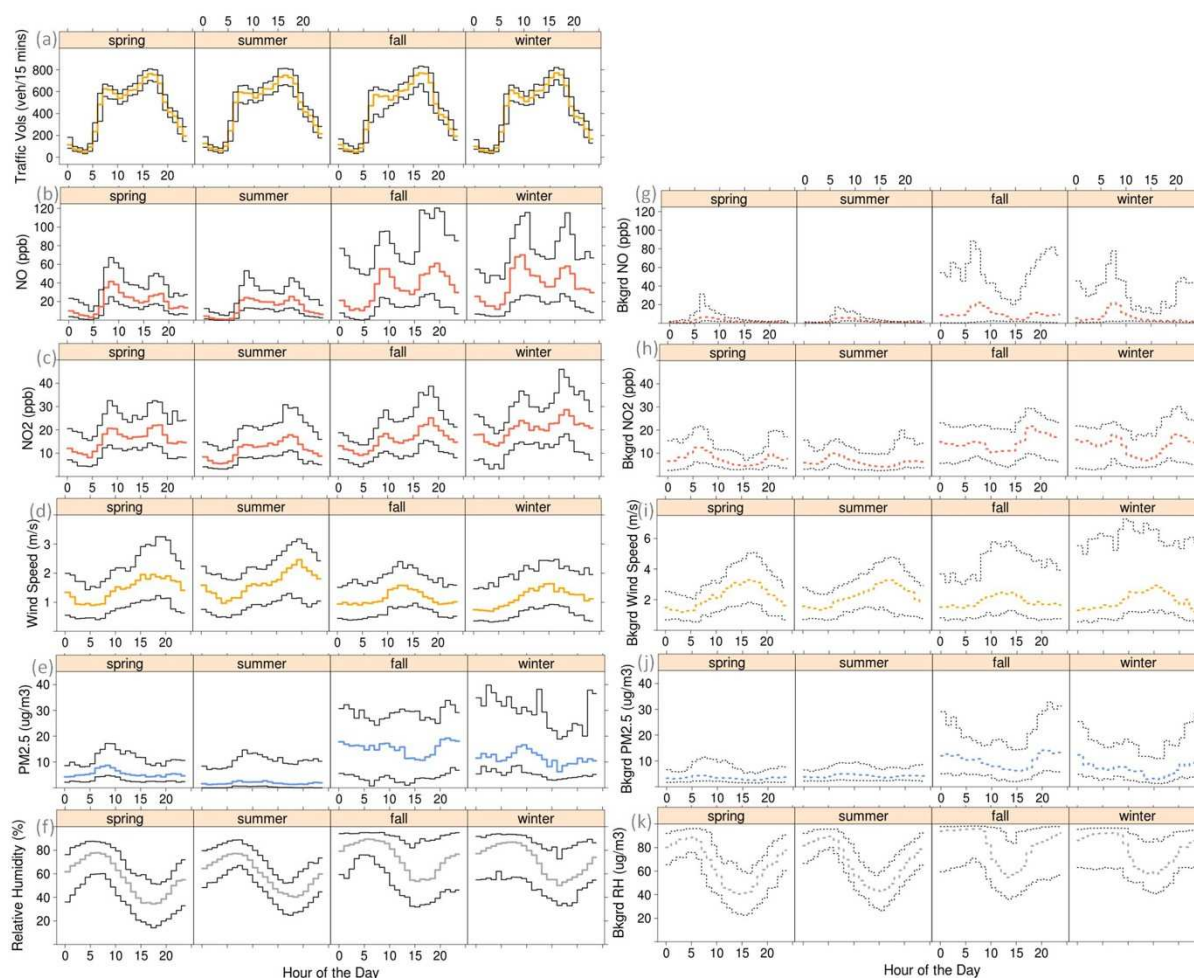




230  
231 **Figure 2 Weekday roadside pollutant measurements versus traffic volumes. White**  
232 **triangles show the median relationship of each pollutant with traffic volume bins of fifty.**

233  
234 **3.2 Weekday, Seasonal and Diurnal Characterization of Roadside Pollutant Concentrations**

235 Seasonal trends and diurnal patterns of traffic volumes and both roadside and urban  
236 background pollutant measurements are presented in Figure 3 with black lines representing the  
237 5<sup>th</sup> and 95<sup>th</sup> percentiles and the colored line is the median. Total traffic volumes and peak  
238 morning and evening traffic volumes do not change much seasonally for the study intersection  
239 (Figure 3a). Daily peak and median concentrations for all three pollutants are greatest in the fall  
240 and winter for roadside (Figure 3b, c, e) and urban background measurements (Figure 3g, h, j)  
241 showing the potential for higher peak exposures to occur in the fall and winter for urban  
242 residents.



243  
 244 **Figure 3 Weekday, seasonal diurnal distributions for (a) traffic volumes, (b-f) roadside NO,**  
 245 **NO<sub>2</sub>, wind speed, PM<sub>2.5</sub>, and relative humidity, and (g-k) urban background NO, NO<sub>2</sub>, wind**  
 246 **speed, PM<sub>2.5</sub>, and relative humidity.**

247

### 248 3.2.1 Diurnal NO and NO<sub>2</sub>

249 Roadside NO and NO<sub>2</sub> have a bi-modal diurnal distribution for weekdays reflecting the  
 250 morning and evening rush hours (Figure 3a, b, c). The NO and NO<sub>2</sub> bi-modal pattern shows a  
 251 seasonal trend. The evening peaks of NO and NO<sub>2</sub> are more pronounced in fall and winter. The  
 252 fall/winter and spring/summer differences in bi-modal distributions can be attributed to seasonal  
 253 boundary layer height conditions and subsequent lower median wind speeds in fall and winter,  
 254 and a change in the diurnal wind pattern by season (Figure 3d). As the height of the boundary  
 255 layer increases in spring and summer, wind speed increases at a faster rate throughout the day,  
 256 increasing dispersion. Urban background NO and NO<sub>2</sub> (Figure 3g and 3h) show similar seasonal  
 257 trends with higher concentrations in fall and winter and a more dominant peak in morning with  
 258 little to no evening peak in spring and summer.

259 Diurnal trends documented in other roadside monitoring studies are attributed to a  
260 combination of meteorology and changing traffic volumes throughout the year or measurements  
261 occur over a small segment of the year so the seasonal variation in diurnal trends is not captured.  
262 Summertime roadside measurements made in Raleigh, North Carolina near a highway showed  
263 only a morning peak in traffic-related pollutants (NO and CO) attributed to variable winds  
264 (Baldauf et al., 2008; Thoma et al., 2008). Springtime urban arterial roadside measurements  
265 made in Copenhagen, Denmark showed two peaks for NO<sub>x</sub>, but the afternoon peak was less  
266 distinct and attributed to lower afternoon traffic volumes (Wang et al., 2010). Urban arterial  
267 roadside measurements made in Athens, Greece over several years showed seasonal, diurnal  
268 variation in roadside NO with the highest evening peak occurring in winter attributed to a  
269 combination of stable atmospheric conditions and higher traffic volumes compared to other  
270 months (Mavroidis and Ilia, 2012). Additionally, Mavroidis and Ilia (2012) showed very little  
271 differences in NO<sub>2</sub> diurnal trends across seasons with winter concentrations being the lowest.

272 In contrast, we show distinct seasonal, diurnal differences that are due primarily to  
273 meteorology as the diurnal traffic volumes for this study road are uniform throughout the whole  
274 year and data presented is for wind directions coming from the road only. The influence of  
275 boundary layer height and meteorology is supported by the urban background concentration  
276 trends while roadside measurements show higher concentrations and the distinct bi-modal  
277 pattern following local traffic volumes for fall and winter. A strong boundary layer height effect  
278 on NO<sub>2</sub> with highest concentrations in winter despite the lower presence of oxidants is also  
279 shown. Urban background NO<sub>2</sub> concentrations present this same seasonal pattern. Traffic  
280 simulation and emissions modeling for planning and transportation projects typically use evening  
281 peak traffic volumes. Usually these models have a target of estimating maximum emission  
282 levels; evening peak traffic volumes tend to be greater than morning levels. However, if a  
283 transportation project requires estimates of peak pollutant concentrations and not just peak  
284 emissions, the results presented here show that morning periods are also important. The morning  
285 period could be used as a more consistent input across seasons for exposure concentration  
286 estimates depending on the objective of the modeling or exposure assessment project.

### 287 **3.2.2 Diurnal PM<sub>2.5</sub>**

288 PM<sub>2.5</sub> concentrations do not show a bi-modal distribution correlated with traffic (Fig 3e).  
289 The lack of this pattern in seasonal, diurnal PM<sub>2.5</sub> distributions is consistent with the lack of a  
290 strong relationship between local traffic volumes and roadside PM<sub>2.5</sub> mass. An increase in the  
291 morning can be seen for spring and winter. The median morning increase is 4 μg/m<sup>3</sup> for spring  
292 occurring between 8-9am and 6 μg/m<sup>3</sup> for winter occurring between 9-10am. These morning  
293 increases show some response to increasing traffic volumes, but the response is not consistent  
294 throughout the year. Concentrations are highest in fall and winter for both roadside and urban  
295 background PM<sub>2.5</sub> when boundary layer height is lower (Figure 3j). The diurnal pattern for these  
296 seasons is dominated by high relative humidity reflected by the elevated PM<sub>2.5</sub> levels in early  
297 morning and nighttime which follow the seasonal, diurnal relative humidity distributions (Figure

298 3f and 3k). The cooler temperatures and higher relative humidity increase condensation and  
299 coagulation leading to increased  $PM_{2.5}$  (Jamriska et al., 2008). Annual and seasonal, diurnal  
300 distributions for measured  $PM_{2.5}$  do not consistently reflect the local traffic volumes well and are  
301 most characteristic of meteorological conditions such as relative humidity and boundary layer  
302 height also reflected in urban background  $PM_{2.5}$ .

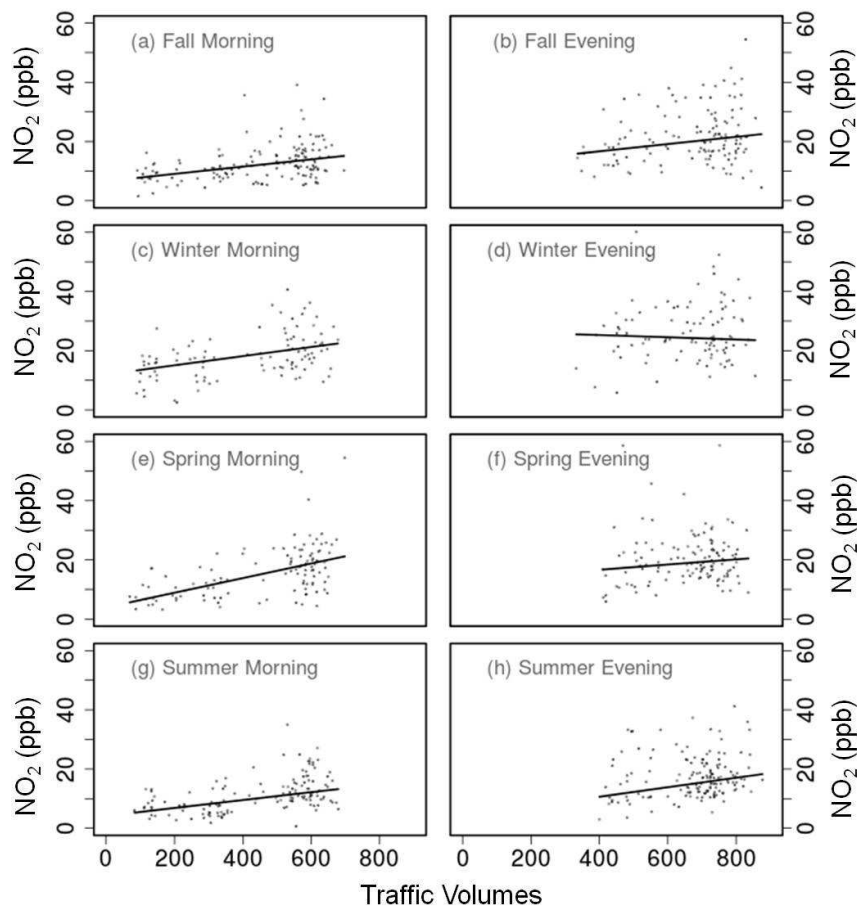
303 Particulate matter is a component of traffic-related emissions but mass measurements do  
304 not always capture this influence. Karner et al.'s (2010) meta-analysis based on over fifteen  
305 studies with  $PM_{2.5}$  measurements around roadways found mixed results regarding the spatial  
306 distribution of fine particles. Using a background normalization method,  $PM_{2.5}$  was found to  
307 have no trend around roadways so was not elevated at the roadside compared to background.  
308 However, using an edge of the roadway normalization method,  $PM_{2.5}$  was found to have a  
309 gradual decay away from roads. For an area such as Portland, OR that does not have as much  
310 diesel traffic as a freeway in California or a European roadway, the increase in  $PM_{2.5}$  mass  
311 directly from transportation-related emissions may be too small of a signal above the noise and  
312 secondary formation of urban background pollution.

313 Particles in the  $PM_{2.5}$  range have a variety of anthropogenic sources in urban areas  
314 including other primary combustion sources (industry, large-scale cooking, liquid and solid fuel  
315 heating), non-road sources (construction equipment, railways), and secondary formation via gas  
316 to particle conversions, coagulation and condensation of smaller particles (Monn, 2001). These  
317 sources and processes lead to a more spatially homogenous pattern of  $PM_{2.5}$  mass. While  $PM_{2.5}$   
318 mass does not always show the impact of local traffic emissions, other measurements of  
319 particulates can. Particle composition has shown to vary spatially near roadways while  $PM_{2.5}$   
320 mass had low spatial variation (Martuzevicius et al., 2004). A review of indoor, outdoor, and  
321 personal exposure to particulates found  $PM_{2.5}$  spatial variation to be much smaller compared to  
322 particle number concentrations and larger size fractions (Monn et al. 2001). Our results and the  
323 variability in  $PM_{2.5}$  mass responses to traffic across studies suggests that measurements of  $PM_{2.5}$   
324 mass as a proxy to assess the impact of roadway pollution may need location-specific validation.

### 325 **3.3 Estimating Roadside NO<sub>x</sub> as a Function of Traffic Volumes**

326 Measured roadside NO and NO<sub>2</sub> is responsive to local traffic volumes, but with distinct  
327 seasonal and diurnal trends on the fifteen minute time scale. If traffic volumes are used as a  
328 proxy in exposure assessment focused on shorter time scales such as the time a pedestrian or  
329 bicyclist spends in the roadside microenvironment, traffic volumes alone may not correlate with  
330 roadside exposure concentrations. Quantile regression for the median is used to assess roadside  
331 NO<sub>x</sub> as a function of traffic volumes for short-term aggregations (15 minutes) for morning  
332 periods (5-10am) and evening periods (3-8pm) for each season. Time periods were chosen based  
333 on the previous diurnal, seasonal analysis to help control for meteorological differences and  
334 traffic peaks in order to assess traffic volumes alone as a predictor variable.

335 NO<sub>x</sub> and traffic volume measurements showed autocorrelation with high correlation  
336 coefficients for each value and the value at a lag of 1. In order to address this serial correlation  
337 while still allowing the use of high resolution measurements, three data points were randomly  
338 sampled from each morning and evening period per day. This sampling method reduced  
339 autocorrelation between NO and NO<sub>lag1</sub> from 0.9 to 0.52, NO<sub>2</sub> and NO<sub>2lag1</sub> from 0.71 to 0.41, and  
340 traffic volumes and traffic volumes<sub>lag1</sub> from 0.96 to 0.38 (total number of sample points used to  
341 build the models reduced from 13,739 to 1,197). Figure 4 shows the median regression  
342 relationships of NO<sub>2</sub> as a function of traffic volumes for morning and evening periods by season.  
343 Table 2 shows the following values for NO and NO<sub>2</sub> models built using both the randomly  
344 selected data subset and all data: model coefficients per 100 vehicles in a 15 minute period,  
345 standard errors for the coefficients per 100 vehicles, and  $r^2$  values. Quantile regression was used  
346 due to the right skewed nature of the data and because quantile regression can capture when a  
347 change in the independent variable exerts both a change in mean and variance in the dependent  
348 variable so heteroskedasticity is not a major concern (Davino et al., 2013).



349

350 **Figure 4 Median regression relationships of roadside NO<sub>2</sub> and traffic volumes for morning**  
351 **and evening periods by season.**

352 **Table 2 Median regression model results for roadside NO<sub>x</sub> as a function of traffic volumes**  
 353 **using randomly sampled subset (black) and all serially correlated data points (grey).**  
 354 **(\*\*significance at <0.01 and \*significance at <0.05)**

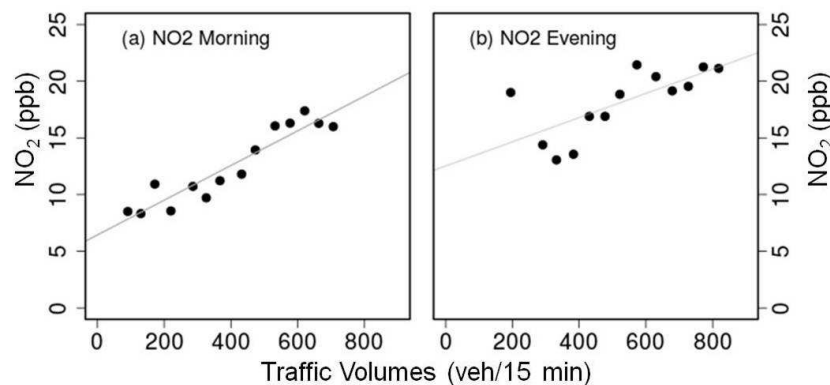
Season	Time Period	Coefficient NO per 100 vehicles per 15 mins	Standard Error of Coefficient NO per 100 vehicles per 15 mins	Adjusted r <sup>2</sup>	Coefficient NO <sub>2</sub> per 100 vehicles per 15 mins	Standard Error of Coefficient NO <sub>2</sub> per 100 vehicle per 15 mins	Adjusted r <sup>2</sup>
Fall	Morning	6.3** (7.9)**	1.4 (0.4)	0.1 (0.16)	1.2** (1.6)**	0.2 (0.1)	0.14 (0.24)
Winter	Morning	9.4** (11.2)**	1.4 (0.7)	0.14 (0.24)	1.5** (1.9)**	0.4 (0.2)	0.17 (0.26)
Spring	Morning	6.3** (6.7)**	0.8 (0.3)	0.41 (0.43)	2.5** (2.3)**	0.3 (0.2)	0.27 (0.28)
Summer	Morning	4.6** (4.4)**	(0.4) (0.2)	(0.45) (0.37)	1.3** (1.6)**	0.3 (0.1)	0.25 (0.23)
Fall	Evening	-1.3 (-1.6)	2.5 (1.1)	<0.001 (0.005)	1.2* (0.9)	0.6 (0.2)	0.05 (0.03)
Winter	Evening	-0.04 (0.09)	2.6 (1.3)	0.002 (0.001)	-0.4 (-0.08)	0.8 (0.4)	0.05 (0.03)
Spring	Evening	2.1 (2.2)**	1.1 (0.5)	0.03 (0.04)	0.9 (0.9)**	(0.8) (0.3)	0.007 (0.02)
Summer	Evening	2.1* (2.9)**	0.8 (0.3)	0.02 (0.07)	1.6* (1.9)**	0.7 (0.2)	0.03 (0.05)

355  
 356 To determine if any significant changes occurred in the model coefficients built using the  
 357 full dataset versus the models built with the randomly sampled subset, coefficient values plus or  
 358 minus two standard errors were compared for all models. These comparisons showed the error  
 359 ranges overlapped for all models demonstrating no significant change in model coefficient  
 360 estimates. Randomly selecting points within our time periods and seasons to reduce  
 361 autocorrelation maintains the relationships found using all data. A change in significance is seen  
 362 for spring evening NO and NO<sub>2</sub> models using all data versus the subset, but coefficient values  
 363 plus or minus two standard errors still overlap for these models. Morning periods for all four  
 364 seasons show a significant linear relationship (p-values <0.01) between traffic volumes and NO  
 365 and NO<sub>2</sub>. Boundary layer height is low in the morning for all four seasons creating similar  
 366 meteorological conditions. As traffic volumes increase throughout the morning and boundary  
 367 layer height increases as well, a positive linear relationship is seen for traffic volumes and NO  
 368 and NO<sub>2</sub> on a fifteen minute scale. As meteorology introduces higher variability in the evening  
 369 due to a changing boundary layer and more variable winds, the linear relationship of traffic  
 370 volumes explains little to none of the variance in roadside 15 minute median NO<sub>x</sub> for this later  
 371 time period (r<sup>2</sup> range from <0.001 to 0.05).

372 A linear relationship between NO<sub>x</sub> and traffic volumes was found for free-flow  
 373 conditions only based on one month of roadside pollutant and hourly traffic measurements in  
 374 Leicester, UK (Agus et al., 2007). The linear relationship broke down during unstable and  
 375 congested traffic flow conditions with occupancy >10% (Agus et al., 2007). Results presented  
 376 here show that the linear relationship even at short-term aggregations holds true past free-flow

377 conditions and throughout the congested morning traffic period. We attribute the breakdown in  
 378 the linear relationship to changing meteorological and dispersion conditions.

379 While short-term NO<sub>x</sub> and traffic volume models were significant for all morning  
 380 periods, variance explained ranged from 0.1 to 0.45. Figure 5 shows the relationship of traffic  
 381 volumes and NO<sub>2</sub> for morning and evening periods at a coarser aggregation using traffic volume  
 382 bins of 50 vehicles and data across the entire year. Table 3 shows the annual regression values  
 383 for NO and NO<sub>2</sub>. There is still no significant effect of traffic volumes on annual NO for the  
 384 evening period. Using binned annual data, variance explained by traffic volumes increases to  
 385 0.89 (NO morning), 0.87 (NO<sub>2</sub> morning) and 0.46 (NO<sub>2</sub> evening). Summarized over one year, the  
 386 seasonal and some meteorological variability are averaged out and traffic volumes are a better  
 387 predictor of annually averaged roadside NO<sub>x</sub>. Even at a larger aggregation, the evening time  
 388 period remains more variable compared to mornings. To our knowledge, the data presented is  
 389 from the only long-term arterial roadside air quality monitoring station in the U.S., but  
 390 measurements from additional intersections would improve the robustness and application of the  
 391 relationships found here.



392

393 **Figure 5 Median roadside NO<sub>2</sub> as a function of traffic volumes binned by 50 vehicles across**  
 394 **the entire year.**

395

396 **Table 3 Regression results for median annual NO and NO<sub>2</sub> as a function of traffic volumes**

	Time Period	Coefficient NO per 100 vehicles per 15 mins	Standard Error of Coefficient NO per 100 vehicles per 15 mins	Adjusted r <sup>2</sup>	Coefficient NO <sub>2</sub> per 100 vehicles per 15 mins	Standard Error of Coefficient NO <sub>2</sub> per 100 vehicle per 15 mins	Adjusted r <sup>2</sup>
Annual	Morning	6.3**	0.6	0.89	1.5**	0.2	0.87
Annual	Evening	-0.2	1.4	-0.09	1.1**	0.3	0.46

397

#### 398 4. Conclusion

399 In this study, we characterized pollutant levels of an urban arterial roadside environment  
400 in a mid-sized U.S. city over the course of a year. Urban arterial roadside environments are  
401 under-sampled in the U.S. but have potential to produce high exposures due to the mix of road  
402 users and close proximity of urban residents. Roadside monitoring for a variety of roadway types  
403 including arterials is well established in European countries, though with different fleet mixes,  
404 fuel, vehicle and air quality standards. Such research has helped document long-term trends and  
405 important characteristics of traffic-related pollutants but has not focused on analysis of short-  
406 term changes as a function of traffic volume, time of day, and season (Bigi and Harrison, 2010;  
407 Carslaw, 2005; Mavroidis and Ilia, 2012). Here we extend the understanding of temporal  
408 variability of NO and NO<sub>2</sub> by showing how the relationships with traffic volumes change  
409 temporally and we provide model coefficients (Table 3) that are applicable to exposure  
410 assessment in North America.

411 Direct evaluations with local traffic volumes showed NO and NO<sub>2</sub> are significantly  
412 affected by traffic volumes and PM<sub>2.5</sub> mass is not responsive to traffic volumes at this  
413 intersection. PM<sub>2.5</sub> mass appears to be more tied to regional sources. Seasonal and diurnal  
414 characterization of NO and NO<sub>2</sub> showed morning peaks to be consistent throughout all seasons,  
415 while a bi-modal distribution was most prominent in fall and winter. The consistent diurnal  
416 traffic volume trends allowed for the large impact of seasonal boundary layer height to be  
417 evident. Highest exposures would occur in fall and winter when boundary layer height conditions  
418 reduce dilution. Other measures of particulate matter may be needed to capture the increased  
419 health risk due to primary combustion particles. This is an important consideration as the U.S.  
420 EPA adds PM mass measurements to the roadway monitoring network. Identification of the type  
421 of environments that are more locally or regionally dominated by PM<sub>2.5</sub> mass could help pinpoint  
422 the type of monitoring needed to accurately assess exposure to particulates due to traffic-related  
423 emissions. Emissions and dispersion models for the study intersection will be run in the future to  
424 allow for comparisons of measured concentrations and model outputs. If modeled PM<sub>2.5</sub> mass  
425 concentrations are strongly tied to local traffic parameters, the models may be missing important  
426 processes related to urban particulate matter formation.

427 Our study shows the limitations of using traffic volumes alone as a proxy for traffic-  
428 related emissions. Traffic volumes explained little of the variance in roadside NO<sub>x</sub> for evening  
429 time periods across all seasons at 15 minutes aggregations. Morning periods at short-term  
430 aggregations showed significant relationships between NO<sub>x</sub> and traffic volumes for all seasons,  
431 but variability was still high. On an annual timescale, the variability in roadside NO and NO<sub>2</sub> due  
432 to seasonality and meteorology is reduced and traffic volumes alone are a better predictor of  
433 roadside concentrations. Even at this coarser temporal scale, the relationship between traffic  
434 volumes and NO and NO<sub>2</sub> in evening periods is not consistent. Using daily averaged data in  
435 exposures studies could potentially miss very different patterns of exposure (Bigi and Harrison,  
436 2010). As exposure assessment moves to account for time spent in transportation



437 microenvironments, traffic volume data will need to be combined with local meteorology in  
438 order to best represent the high variability in roadside pollutant concentrations.

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- 571

**Table 1 Roadside and urban background comparisons for weekdays and weekends, spanning Jan 2013- Dec 2014 at 15 minute aggregations.**

Pollutant	Weekdays		Weekends	
	Roadside Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)	Urban Background Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)	Roadside Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)	Urban Background Mean Median (5 <sup>th</sup> -95 <sup>th</sup> Percentiles)
NO (ppb)	28 20 (1.3 - 79)	9 2 (0.4 - 45)	18 10 (1 - 59)	8 2 (0.4 - 38)
NO <sub>2</sub> (ppb)	16 15 (5 - 31)	11 9 (3 - 24)	11 10 (4 - 23)	9 7 (2 -22)
PM <sub>2.5</sub> (µg/m <sup>3</sup> )	9 6 (0 - 28)	7 5 (2 - 20)	10 5 (0 - 32)	8 5 (2 - 22)
Traffic Volumes/ 15 mins	493 554 (65 - 888)		395 407 (74 - 735)	

**Table 2 Median regression model results for roadside NO<sub>x</sub> as a function of traffic volumes using randomly sampled subset (black) and all serially correlated data points (grey). (\*\*significance at <0.01 and \*significance at <0.05)**

Season	Time Period	Coefficient NO per 100 vehicles per 15 mins	Standard Error of Coefficient NO per 100 vehicles per 15 mins	Adjusted $r^2$	Coefficient NO <sub>2</sub> per 100 vehicles per 15 mins	Standard Error of Coefficient NO <sub>2</sub> per 100 vehicles per 15 mins	Adjusted $r^2$
Fall	Morning	6.3** (7.9)**	1.4 (0.4)	0.1 (0.16)	1.2** (1.6)**	0.2 (0.1)	0.14 (0.24)
Winter	Morning	9.4** (11.2)**	1.4 (0.7)	0.14 (0.24)	1.5** (1.9)**	0.4 (0.2)	0.17 (0.26)
Spring	Morning	6.3** (6.7)**	0.8 (0.3)	0.41 (0.43)	2.5** (2.3)**	0.3 (0.2)	0.27 (0.28)
Summer	Morning	4.6** (4.4)**	(0.4) (0.2)	(0.45) (0.37)	1.3** (1.6)**	0.3 (0.1)	0.25 (0.23)
Fall	Evening	-1.3 (-1.6)	2.5 (1.1)	<0.001 (0.005)	1.2* (0.9)	0.6 (0.2)	0.05 (0.03)
Winter	Evening	-0.04 (0.09)	2.6 (1.3)	0.002 (0.001)	-0.4 (-0.08)	0.8 (0.4)	0.05 (0.03)
Spring	Evening	2.1 (2.2)**	1.1 (0.5)	0.03 (0.04)	0.9 (0.9)**	(0.8) (0.3)	0.007 (0.02)
Summer	Evening	2.1* (2.9)**	0.8 (0.3)	0.02 (0.07)	1.6* (1.9)**	0.7 (0.2)	0.03 (0.05)

**Table 3 Regression results for median annual NO and NO<sub>2</sub> as a function of traffic volumes.**

	Time Period	Coefficient NO per 100 vehicles per 15 mins	Standard Error of Coefficient NO per 100 vehicles per 15 mins	Adjusted $r^2$	Coefficient NO <sub>2</sub> per 100 vehicles per 15 mins	Standard Error of Coefficient NO <sub>2</sub> per 100 vehicles per 15 mins	Adjusted $r^2$
Annual	Morning	6.3**	0.6	0.89	1.5**	0.2	0.87
Annual	Evening	-0.2	1.4	-0.09	1.1**	0.3	0.46

**Highlights****Diurnal and seasonal variations of NO, NO<sub>2</sub> and PM<sub>2.5</sub> mass as a function of traffic volumes alongside an urban arterial**

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- Arterial air quality has been monitored to assess temporal variability with traffic.
- Traffic volumes have a significant effect on NO<sub>x</sub> for morning periods in all seasons.
- Traffic volume explains little to no NO<sub>x</sub> variability in evening periods.
- Roadside PM<sub>2.5</sub> mass is not a consistent measure of traffic impact.
- Traffic volumes alone are of limited use to predict short-term roadside exposures.