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Influence of solid noise barriers on near-road and on-road air quality

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HIGHLIGHTS

- Mobile monitoring measured near-road air quality impacts of a solid, noise barrier.
- Downwind concentration reductions of up to 50% occurred behind the barrier.
- Downwind reductions were highest within the first 50 m from the road.
- Reductions extended as far as 300 m from the road.
- On-road levels did not increase in front of barrier, contrary to model predictions.

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ABSTRACT

Public health concerns regarding adverse health effects for populations spending significant amounts of time near high traffic roadways has increased substantially in recent years. Roadside features, including solid noise barriers, have been investigated as potential methods that can be implemented in a relatively short time period to reduce air pollution exposures from nearby traffic. A field study was conducted to determine the influence of noise barriers on both on-road and downwind pollutant concentrations near a large highway in Phoenix, Arizona, USA. Concentrations of nitrogen dioxide, carbon monoxide, ultrafine particles, and black carbon were measured using a mobile platform and fixed sites along two limited-access stretches of highway that contained a section of noise barrier and a section with no noise barrier at-grade with the surrounding terrain. Results of the study showed that pollutant concentrations behind the roadside barriers were significantly lower relative to those measured in the absence of barriers. The reductions ranged from 50% within 50 m from the barrier to about 30% as far as 300 m from the barrier. Reductions in pollutant concentrations generally began within the first 50 m of the barrier edge; however, concentrations were highly variable due to vehicle activity behind the barrier and along nearby urban arterial roadways. The concentrations on the highway, upwind of the barrier, varied depending on wind direction. Overall, the on-road concentrations in front of the noise barrier were similar to those measured in the absence of the barrier, contradicting previous modeling results that suggested roadside barriers increase pollutant levels on the road. Thus, this study suggests that noise barriers do reduce potential pollutant exposures for populations downwind of the road, and do not likely increase exposures to traffic-related pollutants for vehicle passengers on the highway.

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1. Introduction

A growing number of health studies have identified that populations spending significant amounts of time near high traffic roadways experience increased risks of a number of adverse effects

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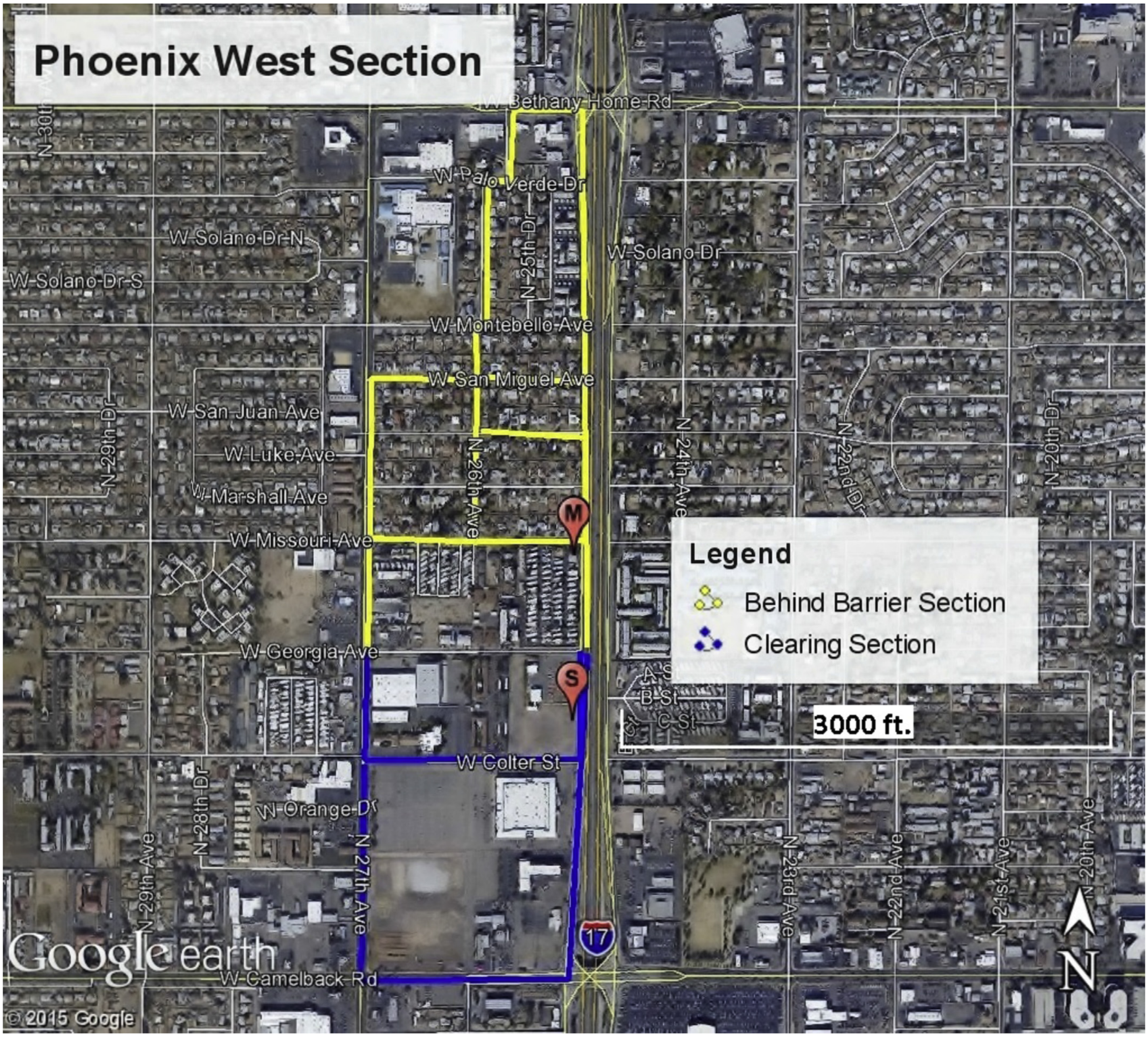

Fig. 1. Mobile and fixed-site monitoring locations. The top figure shows the western section and the bottom the eastern section. The lines represent the mobile driving route (blue lines for the clearing, red lines along the noise barrier). The pins show the fixed-site SUV (S) and portable meteorological and BC instrument (M) locations. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

(e.g. summary by HEI, 2009). These effects have been attributed to increased exposure to particulate matter, gaseous criteria pollutants, and air toxics emitted by vehicular traffic. This significant impact of traffic emissions on populations all over the world has increased interest on identifying methods to reduce exposures to these pollutants. While emission control technologies and programs to directly reduce air pollution emissions are vital components of air quality management, roadway design techniques that can reduce population exposures may also provide an important role in public health protection.

One roadway design option that has received increased interest is the construction of roadside structures, such as noise barriers (also often referred to as sound walls), which dilute nearby vehicle emissions to reduce pollutant concentrations for the highest exposed populations. This technique can provide planners and developers with an option to reduce concentrations of air pollutants in a relatively short time frame. The addition of noise barriers for near-road air quality improvement can also complement existing pollution control programs or provide measures to reduce impacts from sources difficult to mitigate.

Air quality in areas adjacent to major roadways is influenced by primary emissions from on-road vehicles. Concentrations of primary pollutants (e.g., particulate matter (PM), nitrogen oxides (NO_x), carbon monoxide (CO), mobile source air toxics (MSATs)) in vehicle tailpipes, just before they are emitted into the atmosphere, are often two to four orders of magnitude higher than concentrations measured in ambient air (Zhang et al., 2004). Motor vehicles also emit and re-suspend PM constituents from brake, tire, and pavement wear during operation on the road. Key factors contributing to observed near-road pollutant concentrations

Fig. 1. *(Continued).*

include source strength (e.g., number and types of vehicles and their emission rates); distance from a road to a receptor location (e.g., nearby homes, schools, and other places representing potential points of exposure); and meteorological conditions (wind speed and direction and mixing height). These combined influences result in both diurnal and seasonal variations in near-road pollutant concentrations. The scale of the near-road environment is of the order of 100 m, which means that the time frame pollutant transport from the source to the receptor is on the order of seconds to minutes. At these time scales, dispersion by atmospheric turbulence is the primary process for reducing concentrations. However, some physical transformations (e.g., nucleation of hot tailpipe gases in Wang and Zhang (2012)) and chemical transformation (e.g., the titration of existing ozone with NO to form NO_2) can also be important (Wang et al., 2011; Yang et al., 2015).

Previous field measurement studies have examined the impacts of noise barriers on near-road air quality (Baldauf et al., 2008; Lidman, 1985; McNabola et al., 2008; Naser et al., 2009; Ning et al., 2010; Nokes and Benson, 1984). Generally, all of these studies have shown that downwind air pollution levels can decrease in the presence of a noise barrier. The earlier studies by Nokes and Benson (1984) and Lidman (1985) showed decreases immediately downwind of the barrier, with this reduction attributed to lofting of the traffic emission plume over the solid structure of the barrier. Later studies (Baldauf et al., 2008; Ning et al., 2010) showed that concentrations behind the barrier generally remained lower than concentrations at a clearing out to several hundred meters, with reductions as high as 50% behind the barrier. The study by Baldauf et al. (2008) also showed that noise barriers can increase pollutant concentrations on the upwind side of the wall, and pollutants can meander around the edges of the barrier wall. Ning et al. (2010) also measured higher pollutant concentrations further from the noise barrier compared to concentrations at a similar perpendicular distance with no barrier present. Naser et al. (2009) demonstrated that noise barriers can reduce the high, peak concentrations often found near roads. While most field studies have investigated noise barriers at heights of 4–6 m above the road surface, McNabola et al. (2008) showed that a low boundary wall of approximately 2 m in height also reduced exposures to traffic pollutants for pedestrians on a boardwalk next to a road.

Wind tunnel simulations have also investigated the impact of noise barriers on near-road air pollution (Heist et al., 2009; Hölscher et al., 1993; Lidman, 1985). These simulations generally show reduced concentrations downwind of the barrier due to the development of a vortex formed behind the wall with low pollutant concentrations as air and the traffic emission plume from the roadway is lofted above and over the structure. Heist et al. (2009) demonstrated that downwind pollutant concentrations gradients behind the noise barrier were lower out to hundreds of meters from the barrier.

Finn et al. (2010) conducted a field study using a tracer gas release to evaluate the effect of a noise barrier on air pollutant transport and dispersion under varying atmospheric stability conditions. This study showed 80% mean reductions in tracer concentrations downwind of a 6 m noise barrier up to 15 wall-heights (90 m) horizontally downwind of the barrier compared with collocated measurements without a noise barrier. Reductions in pollutant concentrations were reported for all atmospheric stability conditions. Gradients and absolute concentrations varied by atmospheric conditions, with the steepest gradients occurring under unstable conditions and the highest concentrations under stable conditions.

Modeling studies have also been used to simulate noise barrier impacts on near-road air quality (Bowker et al., 2007; Hagler et al., 2011; Janssen and De Maerschalck, 2009; McNabola et al., 2008; Steffens et al., 2014, 2013, 2012). These studies used computational fluid dynamics (CFD) models to evaluate noise barrier characteristics on pollutant transport and dispersion. Schulte et al. (2014) used a semi-empirical dispersion model to describe observations from the wind tunnel (Heist et al., 2009) and field (Finn et al., 2010). These modeling results show that noise barriers can reduce downwind pollutant concentrations through the lofting of the traffic plume over the solid structure followed by enhanced dispersion downwind of the barrier. The concentrations in the immediate downwind vicinity of the barrier correspond to the entrainment of the material from the lofted plume into the cavity behind the barrier.

Results from the modeling studies evaluate measurements made during neutral conditions, with winds relatively high and the surface heat flux close to zero. However, current models perform poorly when the boundary layer departs from neutrality. For example, the model by Schulte et al. (2014) did not simulate the large reduction in concentration seen in Finn et al. (2010) during unstable conditions. The model also underestimates concentrations measured during very stable conditions, with this discrepancy attributed to the plume being carried around the barrier edge by very stable flow that tends to remain horizontal rather than go over the barrier vertically. The model assumes that the plume only goes over the top of the barrier.

Because vehicle emissions occur primarily during daylight hours when the surface boundary layer can be unstable, understanding the mitigation effects of roadside barriers during these conditions is important. This paper describes field study results quantifying the impact of roadside barriers under real-world conditions in Phoenix, Arizona, USA. The results from this study were used to evaluate and modify existing dispersion models that simulate roadside barrier effects. The evaluation and refinement of the model is presented in Part II of this paper.

2. Methods

2.1. Site description and sampling schedule

Two highway segments along Interstate-17 (I-17) in Phoenix, Arizona, USA were chosen to evaluate the impacts of noise barriers on near-road air quality. One segment was located on the west side of the highway and one segment on the east side of the highway, within 1 km but not directly across from one another. This configuration maximized the ability to evaluate the variability of air pollutant concentrations during downwind conditions. Each segment was approximately 2 km (km) in length and 500 m (m) in width and primarily residential. For the segment, the highway was at-grade with the surrounding terrain and contained a section with and without a noise barrier along the same stretch of limited-access highway. This study design ensured that traffic volumes and vehicle operations were consistent along the segment. The choice of monitoring either the east or west segment each day was based on predicted wind directions for the sampling period in order to maximize the probability of sampling primarily downwind of the highway. Fig. 1 shows the location of each segment chosen for the study. The noise barriers along each segment were approximately 4.5 m in height, less than 1 m in thickness, approximately 3 m from the nearest travel lane of I-17, and had an access road immediately behind the wall.

A combination of mobile and fixed sampling instrumentation provided data on air quality on and nearby I-17. The field study was conducted for approximately one month, from late October until late November 2013. Sampling occurred for approximately 3 h each day during either morning or afternoon time periods. Data collected included traffic counts and speed, meteorology, and air quality for multiple pollutants.

The Maricopa Association of Governments (MAG) collected and reported traffic volume data for I-17 along the segments evaluated in this study as shown in Fig. 1. MAG provided 15-min average count and speed data for each day of monitoring. This information was obtained from the Arizona Department of Transportation's (ADOT) Freeway Management System (FMS) database ADOT (2009). The FMS contained total vehicle count data for both directions of the highway separately. Estimates of the percent of heavy-duty trucks were obtained from a truck travel survey conducted previously by MAG (2007). No traffic data was available for the access road.

Meteorological measurements using three-dimensional (3-D) sonic anemometers provided information required to interpret the concentration measurements and construct inputs for the dispersion model evaluated in this study. Measurements were made approximately 20 m from the nearest travel lane of the highway at both the clearing and behind noise barrier sites. The behind noise barrier site was at least 100 m from either endpoint of the barrier. The behind barrier site also contained three levels of sonic anemometers at 2, 4, and 5 m above the highway and access road surface.

Air quality measurements consisted of multiple portable and mobile measurement platforms. At the clearing and behind barrier sites, portable aethalometers were placed alongside each sonic anemometer to provide corresponding black carbon (BC) measurements. Thus, there was one BC measurement at the clearing approximately 2 m above ground, and three BC measurements behind the noise barrier at 2, 4, and 5 m above ground. A sport utility vehicle (SUV) was parked at the clearing site (approximately 20 m from the highway) and collected air quality measurements, including CO, NO, NO_2, and PM. Batteries located within the vehicle powered these samplers, so the vehicle's engines could be turned off during all sampling periods. Mobile monitoring was also conducted using an electric vehicle equipped with air quality analyzers to provide spatially-resolved air quality data by driving a route on low traffic side roads located on both the east and west sides of I-17, as well as driving on I-17. As shown in Fig. 1, this driving route measured pollutant concentrations along nearby low-volume roads both behind the noise barrier and along the clearing section. These low-volume roads were parallel and perpendicular to I-17 to allow mapping of air quality concentrations in the area. The mobile monitoring routes also included the locations of the fixed-site sampling to allow comparisons of both measurement methods for quality assurance purposes. Measurements conducted on-board the mobile platform included location (with a global positioning system) and CO, NO_2, PM, and BC. All air quality instruments used in mobile monitoring measured at 1-s sampling intervals, resulting in a spatial resolution of approximately 10 m at typical side road driving speeds (e.g., 30 km/h) and 25 m at highway driving speeds (e.g., 100 km/h). Hagler et al. (2010) describes the vehicle set up, instrumentation, and sampling methods used in the mobile monitoring platform.

No other major sources of air pollution were identified within a 5 km area around the study sites. The main source of pollutants that could confound the measurements of impacts from the highway were vehicles operating on the side access roads and traffic on the arterial roads which bordered the study area. In previous studies, to aid in identifying the influence of side road traffic, major short-term spikes in concentrations in the second-by-second mobile monitoring measurements were identified using the procedure described by Hagler et al. (2010). This procedure utilized short-term fluctuations in either CO or ultrafine particles (UFPs) concentrations as exhaust indicators. For most of the study area, these brief concentration spikes indicating localized impacts from side road vehicles were either nonexistent or rare in nature. Exceptions were the measurements collected along 27th Avenue and Camelback Road for the west segment and 23rd Avenue for the east section, which experienced relatively consistent and higher traffic volumes during the study periods. As a result, all data collected along these roads, and within 20 m of these road along the transect routes, were removed from analysis. Another exception was the measurements collected along the access roads immediately behind the barriers for both sections. Since pollutants could remain elevated behind the noise barrier for extended time periods, the Hagler et al. (2010) method often could not identify the influence of local, access road traffic emissions on the measurements. Thus, these behind-barrier measurements likely reflect some influence of local traffic, and these measurements would have been lower if the access road was not present.

Land use in the study area consisted primarily of single-family, detached homes and sparse vegetation, along with some commercial areas. As shown in Fig. 1, land use for the east section was generally similar, with mostly residential dwellings. The west section had more residential land use behind the barrier, while the open, no barrier section consisted of more commercial buildings along with parking lots, potentially influencing air flow differently. In addition, the southern boundary of the west section driving route (Camelback Road) contained a busy commercial district with higher traffic volumes as previously described.

2.2. Analytical instruments and methods

Air quality measurements consisted of black carbon (BC), ultrafine particles (UFP), nitrogen dioxide (NO_2), and carbon monoxide (CO) using mobile and fixed site monitoring. Mobile monitoring followed the routes shown in Fig. 1 (blue and red lines). This figure also shows the locations of the fixed site monitoring: one location adjacent to the clearing segment used a SUV equipped with BC, UFP, NO_2, and CO instruments operated from separate batteries and a second location behind the noise barrier with three portable BC instruments mounted on a pole and placed at heights of 2,4, and 5 m above ground. A sonic anemometer was placed at each of these heights to determine wind flow in the area. A fourth sonic anemometer was placed adjacent to the SUV to obtain wind flows at the clearing location. Traffic data will be confirmed through video cameras mounted on the SUV (video recording traffic volumes on I-17 and the access road along the highway) and electric vehicle (video of traffic along the driving route). A summary of the measurements collected during the study is shown in Table 1.

2.3. Collocated sampling

At the end of each sampling day, all analyzers (mobile and fixed-

Table 1
Measurement methods used at each monitoring site.

Measurement location	Measurement Parameter	Sampling approach	Instrument make/ Model	Sampling height (above ground)	Sample type and frequency
Clearing (SUV) Site (at-grade section; 20 m from highway)	Carbon Monoxide (CO)	nondispersive infrared	Model 48i-TLE Thermo Scientific	2 m	Continuous, 1-s averages
	Oxides of nitrogen ($NO/NO_2/NO_x$)	chemiluminescence	Model 200A API		
	Particle number concentration (size range 5.6–560 nm, 32 channels)	Engine Exhaust Particle Sizer	TSI, Inc., Model 3090		
	Black Carbon	Micro-Aethalometer	Magee Scientific, AE-51		
	Wind speed and direction	sonic anemometer	RM Young 81000		
Mobile Monitoring Vehicle	Carbon Monoxide (CO)	Quantum Cascade Laser	Aerodyne Research, Inc.	1.5 m	Continuous, 1-s averages
	Nitrogen Dioxide (NO_2)	Cavity Attenuation Phase Shift	Aerodyne Research, Inc.		
	Particle number concentration (size range 5.6–560 nm, 32 channels)	Engine Exhaust Particle Sizer	TSI, Inc., Model 3090		
	Black Carbon	Single-channel Aethalometer	Magee Scientific, AE-42		
	Longitude and Latitude	Global positioning system	Crescent R100, Hemisphere GPS		
	Video	Webcam			
Behind Barrier Portable Monitoring Tower Site (20 m from highway)	Black Carbon	Micro-Aethalometer	Magee Scientific, AE-51	2,4,5 m	Continuous, 1-s averages
	Wind speed and direction	sonic anemometer	RM Young 81000	2,4,5 m	

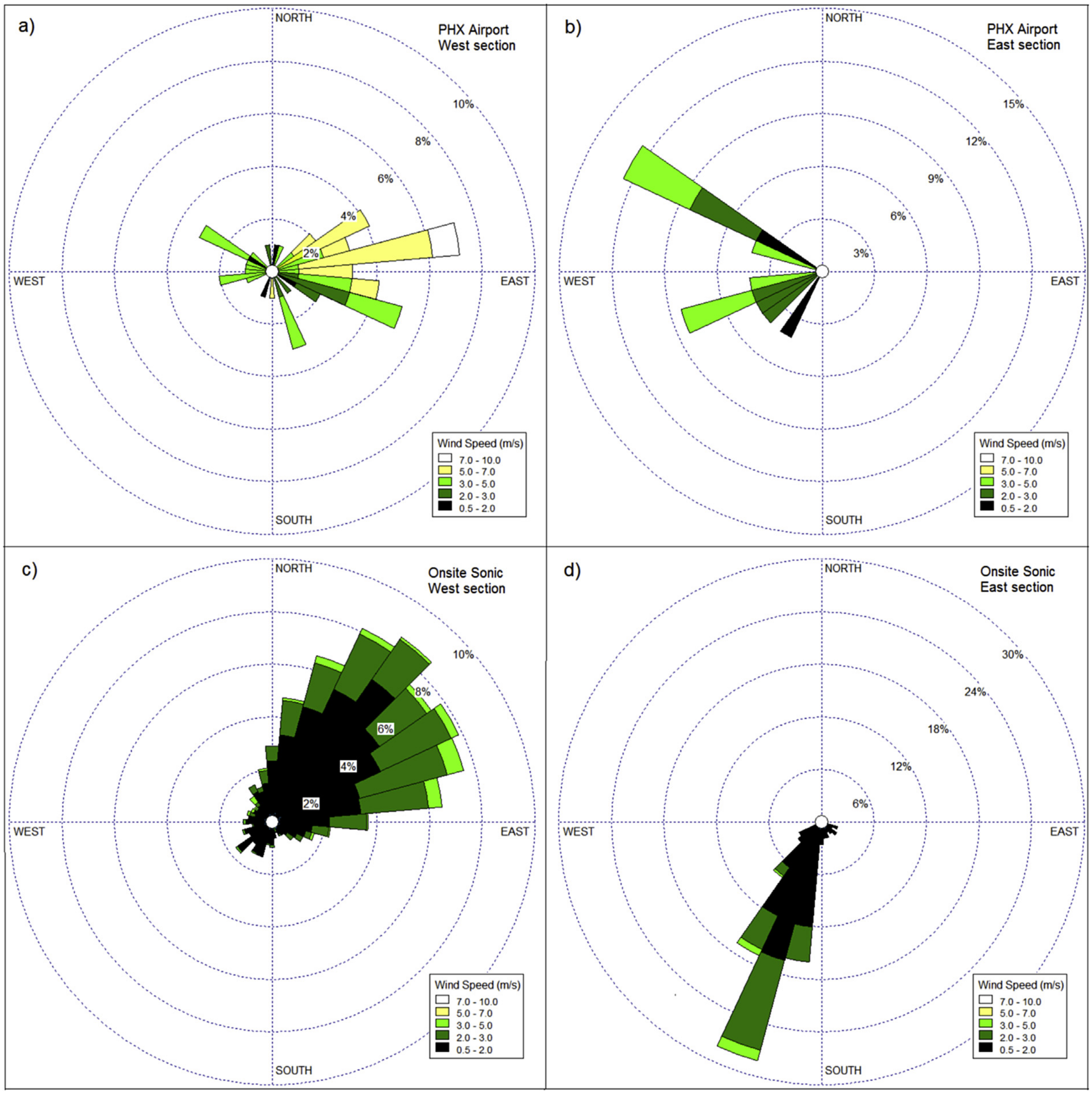

Fig. 2. Wind roses for Phoenix 2013 field study based on a) hourly observations from Phoenix Sky Harbor airport during measurements in west sampling section; b) hourly observations from Phoenix Sky Harbor airport during measurements in east sampling section; c) 10-min average data from onsite sonic measurements in the western sampling section; and d) and 10-min average data from onsite sonic measurements in the eastern sampling section.

sites) were moved to the SUV location along the clearing for 10–30 min to conduct side-by-side sampling to estimate the precision of the samplers. The collocated measurements of black carbon (BC) concentration and particle number (PN) concentration by bin were assessed at 1, 2, 3, 5, and 10 min averaging intervals through visual inspection and least-squares regression. A fixed time base version of the aethalometer optimized noise-reduction algorithm (ONA) (Hagler et al., 2011) was applied to the BC data set. Any data that showed insufficient change in the light attenuation signal over the target averaging period (i.e. low signal-to-noise ratio) were removed from analysis.

The collocated comparison for instruments of the same make and model showed varying estimates of precision for the instruments used in this study. For the UFP measurements with the EEPS, collocated sampling resulted in a r^2 value of 0.88 (best fit line slope of 1.3; intercept close to zero). The portable microaethalomters also showed relatively high precision with r^2 values greater than 0.9 for all units. Comparison of the NO_2 measurements revealed a r^2 value of 0.58 (best fit line slope of 0.88; intercept of 2.9 ppb). For CO, a r^2 value of 0.53 (best fit line slope of 0.62; intercept 120 ppb) was obtained from the collocated sampling. The difference in measurement technique and sample time-averaging between the two sampling methods compared for NO_2 and CO likely contributed to this lower precision for these pollutants.

3. Results and discussion

A total of twenty-two (22) valid sampling periods were collected during the study. Each monitoring route was driven for 2–3 h each day, allowing for between 5 and 9 laps of the driving routes shown in Fig. 1 per sampling period. The majority (18) of the sampling occurred along the western section of I-17 to capture downwind

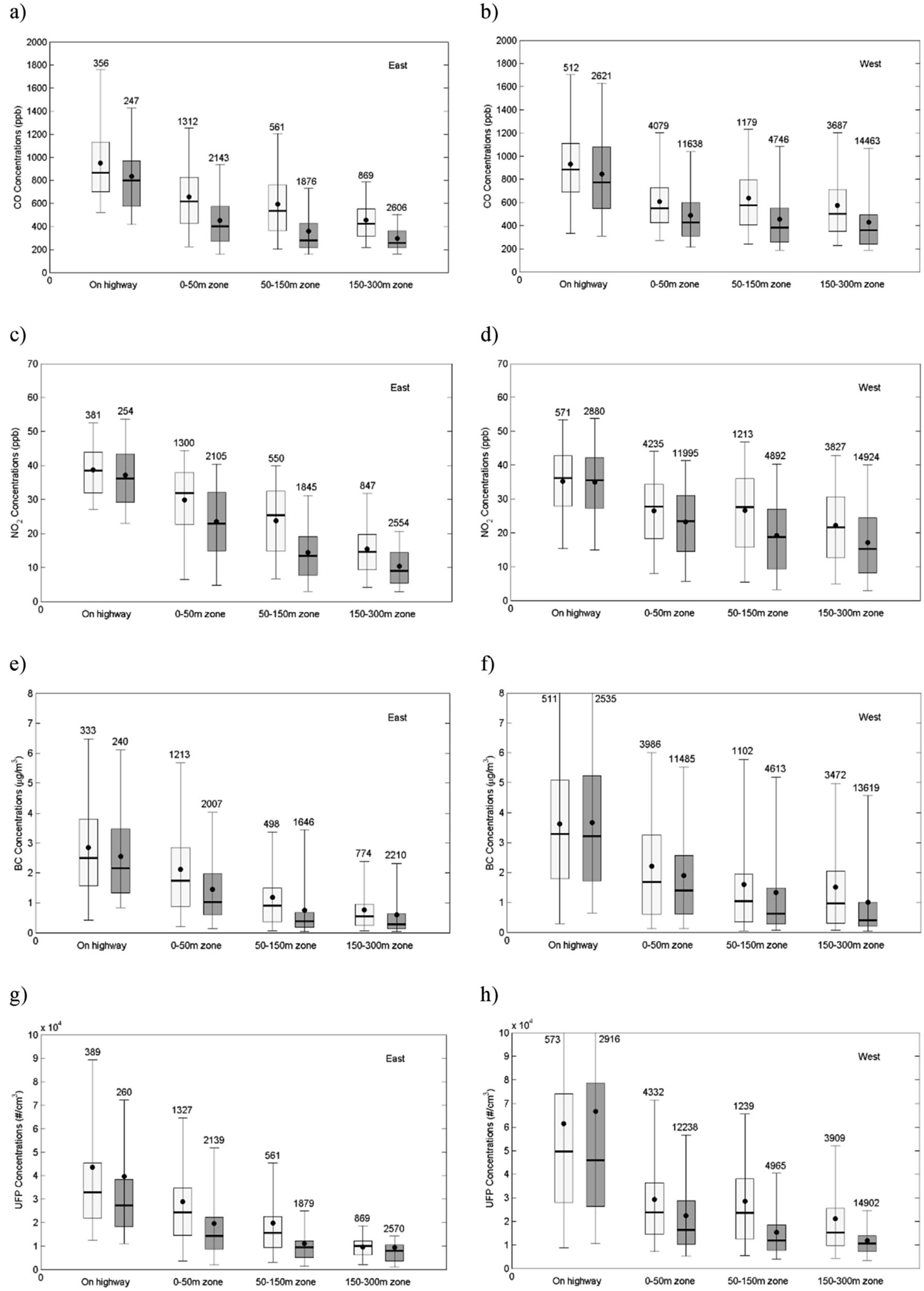

Fig. 3. Distributions of observed CO (a–b), NO_2 (c–d), BC (e–f), and UFP (g–h) concentrations from all mobile monitoring measurements in east (a,c,e,g) and west (b,d,f,h) sections behind the noise barrier (gray) and in the clearing with no barrier (white) at different distances from the highway (onroad, 0–50 m, 50–150 m, 150–300 m). Each distribution is based on n observations (shown above the upper whiskers). The middle line represents the median, the box the 25th and 75th percentiles, and the whiskers the 5th and 95th percentiles. The point represents the mean value of the distribution.

pollutant concentrations during wind events from the east, which typically occurred during the morning hours between 9:00AM and 12:00PM. The remaining sampling periods (4) occurred along the eastern section of the highway, typically during the afternoon hours of 2:00PM to 5:00PM. Fig. 2 shows the wind speed and directions measured at the Phoenix Sky Harbor and the on-site sampling location during the study period. As shown, wind conditions measured at these sites were similar during the study, although the on-site measurement suggested more variable winds with a northerly influence in the morning (western section sampling) and southerly influence in the afternoon (eastern section sampling), respectively.

During the study, average weekday traffic volumes along the monitored segments ranged from 120,000 to 140,000 vehicles per day, with weekend volumes around 100,000 vehicles per day. The average speeds varied depending on time of day and highway direction. For the northbound direction of I-17 adjacent to the eastern segment, speeds averaged around 65 miles per hour (mph) during the morning (represented by 9:00AM traffic conditions) with minimums around 61 mph and averaged approximately 55 mph during the afternoon (represented by 4:00PM traffic conditions) with a minimum of 30 mph. For the southbound direction of I-17 adjacent to the western segment, speeds averaged around 57 mph during the morning with a minimum of approximately 34 mph and averaged approximately 55 mph during the afternoon with a minimum of 40 mph. These traffic patterns reflected commuting toward the city center on the southbound lanes during the morning and commuting away from the city center during the afternoon on the northbound lanes. Based on measurements by MAG in 2011, the truck volume was estimated at 10 percent of the total traffic volume, equally distributed on the northbound and southbound lanes. While no data was available for the access road, study personnel estimated approximately 3000 vehicles per day used these roads with only a small fraction of that volume being trucks.

Comparison of measurements behind the barrier and along the clearing section revealed a consistent reduction in air pollutant concentrations for all pollutants analyzed. Fig. 3 shows the distribution of pollutant concentrations over all samples collected by mobile monitoring along the eastern and western sampling sections, respectively. These plots show concentration measurements under all meteorological and temporal conditions. The bin ranges chosen for this figure highlight microenvironments of interest for near-road exposures and provided sufficient sample numbers per bin for comparison.

The results indicate that the barrier reduced pollutant concentrations away from the road for all pollutants measured. The largest reductions occurred in the distances between 0–50 and 50–150 m from the road, although reductions were also seen as far as 300 m Table 2 shows the reduction in near-road concentrations by pollutant and distance bin for the median and mean values. These calculations represent the mean and median reductions for all data collected in each distance bin as compared to the previous distance bin closer to the road. For example, the reduction in the 0–50 m distance bin represents the difference between the on-road and 0–50 m measurements. For each distance bin (on-road, 0–50 m, etc.), the differences in the means of the concentrations corresponding to the barrier and the no-barrier measurements were more than twice the standard error of the differences, indicating a less than 2.5 percent probability of the results in Table 2 occurring by chance.

In general, higher pollutant reductions occurred in the eastern section than the western section within the first 150 m, while the western section experienced similar reductions across the entire sampling distance away from the road. Since the barrier characteristics, terrain, and land use were similar for both sections, these results may indicate that the barrier effects depend on atmospheric stability since the western section measurements typically occurred during morning, more stable, conditions.

The measurement results shown in Fig. 3 and Table 2 did not find evidence of a higher concentration approximately 150 m from the road behind the barrier compared with the clearing measurements as reported by Ning et al. (2010). In many instances, the highest pollutant reductions occurred within this distance from the highway. As a result, this study also did not indicate that the greatest reductions always occurred closest to the barrier wall as estimated by tracer gas and wind tunnel experiments in Finn et al. (2010) and Heist et al. (2009), respectively. Since these distributions reflect all wind directions and speeds, instances occurred with

Table 2
Median and mean reduction in near-road pollutant concentrations for all pollutants measured under all meteorological and temporal conditions.

Pollutant	Sampling section	Distance range (meters)	Median reduction (percent)	Mean reduction (percent)
CO	East	0–50	50	46
		50–150	31	21
		150–300	9	18
	West	0–50	45	42
		50–150	11	7
		150–300	6	5
NO_2	East	0–50	37	37
		50–150	41	39
		150–300	33	28
	West	0–50	34	34
		50–150	20	17
		150–300	19	11
BC	East	0–50	53	43
		50–150	63	49
		150–300	26	18
	West	0–50	57	48
		50–150	55	30
		150–300	37	24
UFP	East	0–50	48	50
		50–150	34	44
		150–300	16	15
	West	0–50	54	66
		50–150	27	31
		150–300	12	23

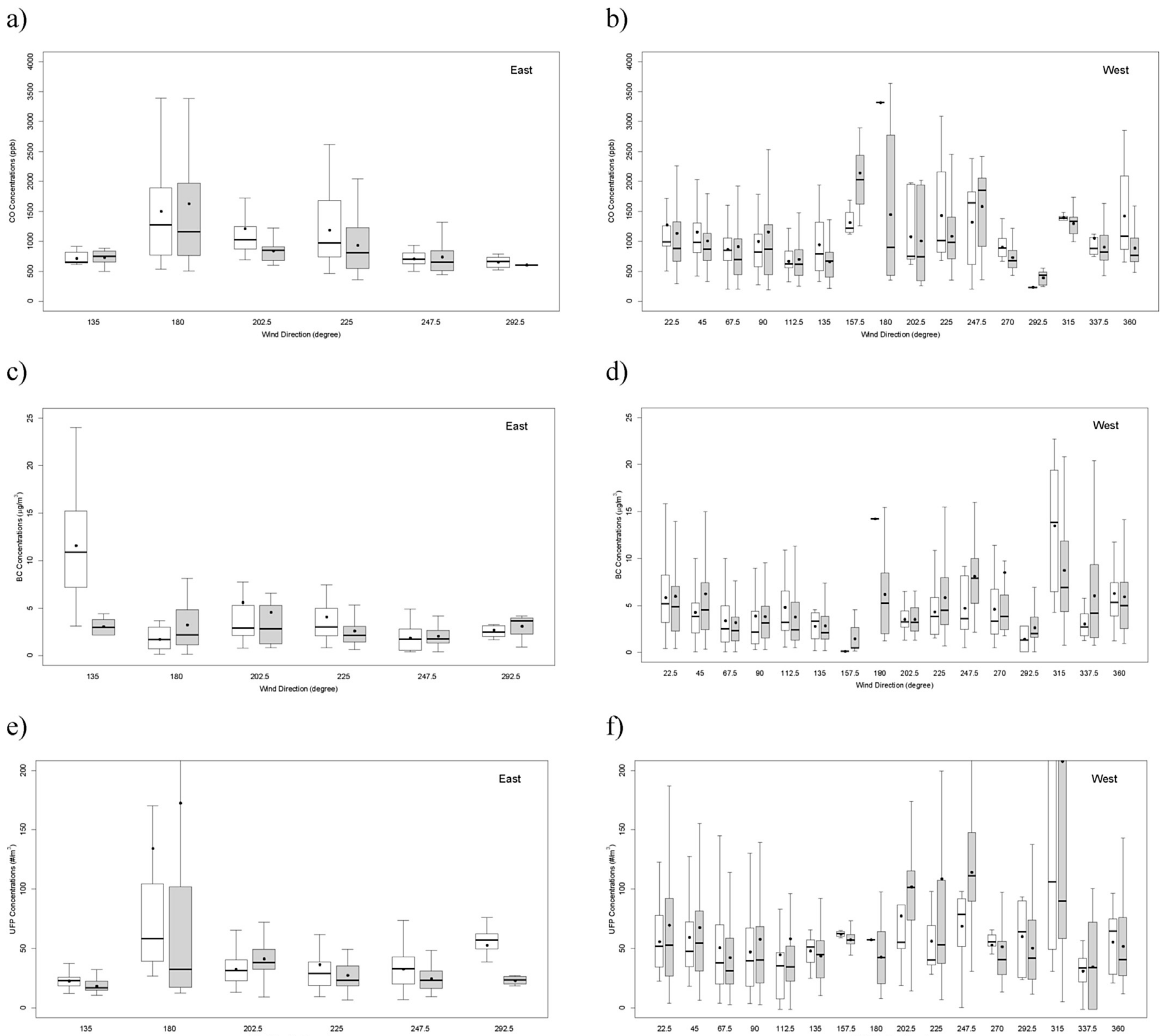

Fig. 4. Distributions of on-road CO (a–b), BC (c–d), and UFP (e–f) concentrations along the east (a, c, and e) and west (b, d and f) sections by wind direction. The data collected in front of the noise barrier are marked in gray and in front of the clearing with no barrier in white. The middle line represents the median, the box the 25th and 75th percentiles, and the whiskers the 5th and 95th percentiles. The data point represents the mean value of the distribution.

wind transport to the highway, which may have blocked upwind pollutant emissions at the barrier wall as reported by Baldauf et al. (2008). In addition, emissions from traffic activity along the access road likely contributed to some increase in concentrations in the 0–50 m distance bin as previously described. Thus, increased reductions would be anticipated if the access road was not present and only downwind conditions were evaluated. Modeling by Venkatram et al. (2016) suggested that traffic emissions from the arterial road that formed the southern border of the study area for the western section contributed to higher background pollutant concentrations along the no-barrier portion of the mobile monitoring route. Higher background concentrations along the no-barrier section, which was closer to this arterial road, compared with lower behind-barrier background concentrations due to this section being further from the arterial, would result in higher perceived concentration reductions from the barrier than might occur under ideal conditions. However, as shown in Fig. 1, the mobile monitoring route along the eastern study section did not have a high volume arterial road close to the no-barrier or behind-barrier sections, yet observed reductions were similar for this eastern section as the western section.

The potential for solid, roadside barriers to block air flow, and thus potentially increase pollutant levels in front of the structure, has raised concerns that these features can lead to increased pollution on the highway leading to increased exposures for drivers on the road (Baldauf et al., 2008; Hagler et al., 2010). The on-road measurements in this study did not support this concern. As shown in Fig. 3, comparison of the distribution of measurements collected on the highway in front of the noise barrier and along the open section did not show a difference in concentrations. To the

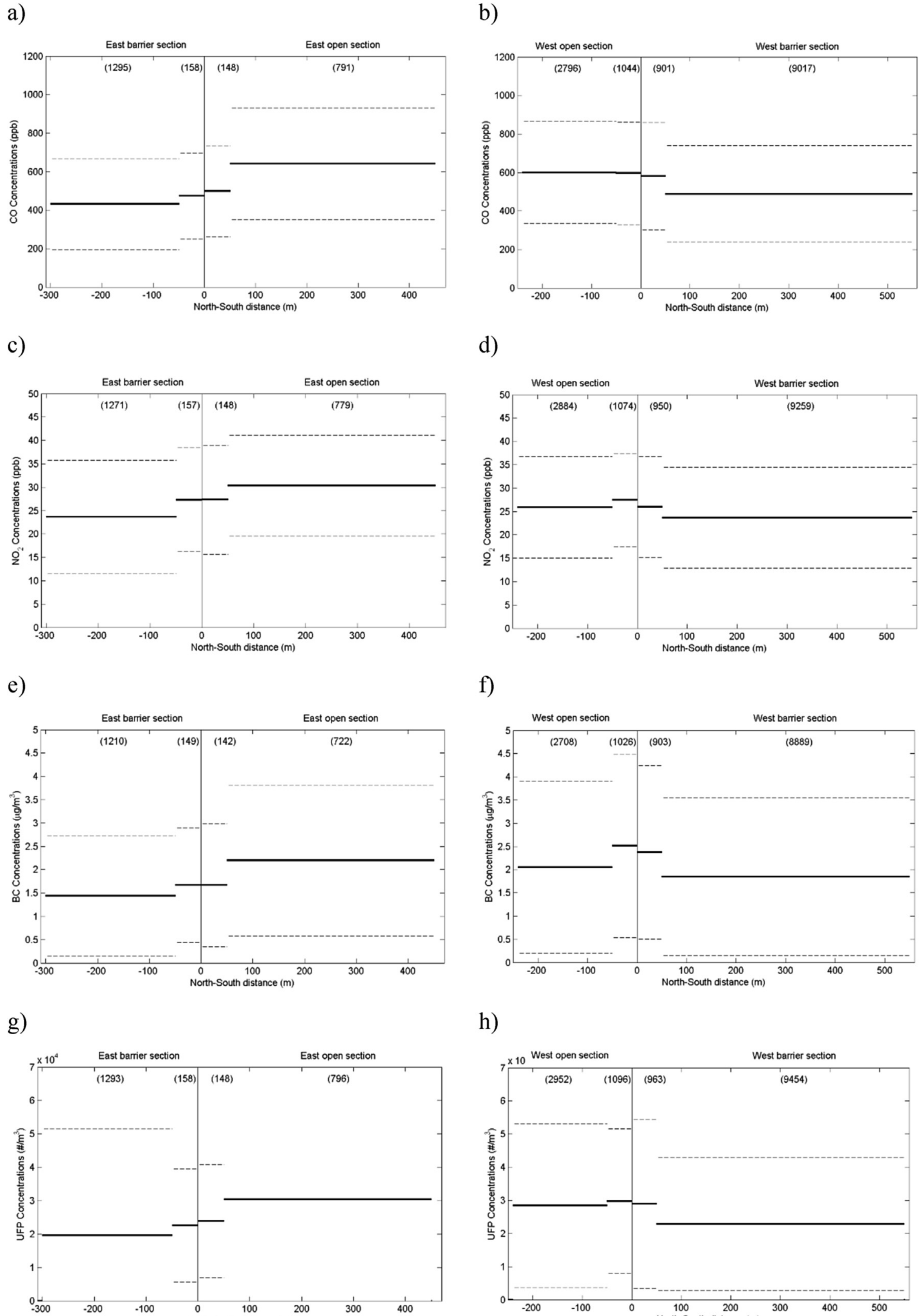

Fig. 5. Distributions of observed CO (a–b), NO_2 (c–d), BC (e–f), and UFP (g–h) concentrations parallel to the highway in east (left panels) and west (right panels) sections for the behind the noise barrier and no barrier conditions. Note: distributions are based on n observations (numbers shown above) within a 0–30 m zone. The solid line indicates mean values while the dashed lines indicate +1 standard deviation.

contrary, the median and mean of the distribution of on-road measurements collected for most pollutants at both the eastern and western sections of the study area yielded the same or lower concentrations in front of the barrier as compared to the open section. Only UFP measurements along the western section showed higher average concentrations in front of the barrier; however, the median was lower in front of the barrier.

The influence of ambient wind direction on on-road concentrations of CO, BC and UFP was further explored in Fig. 4. For this analysis, NO_2 was not evaluated due to the potential for secondary reactions. For the perpendicular wind direction, i.e., 90° for the west section measurements (Fig. 4b, d and f), the median on-road concentrations of CO, BC and UFP are higher in front of the noise barrier than the corresponding values with no barriers. This trend is consistent with the findings from previous wind tunnel and modeling studies, which focused on perpendicular wind conditions (Hagler et al., 2010). For the east section analyses, the wind direction of 270° did not occur during the study according to the on-site sonic data (Fig. 2d). Thus, we are not able to examine whether the same trend persists using the data collected from the on-road measurements in the east section. For non-perpendicular wind directions during the east section measurements (Fig. 4a, c and e), the median on-road concentrations of CO, BC and UFP are generally lower in front of the noise barrier than the corresponding values when no barrier is present.

For non-perpendicular wind directions during the west section measurements (Fig. 4b, d and f), the trend was more complicated. For this analysis, the wind directions were grouped into sixteen segments, with each segment covering 22.5° ranges. Eleven out of the fifteen non-perpendicular wind segments experienced higher on-road CO concentrations with no barrier than those with the noise barrier present, while eight of the BC and nine of the UFP were higher, respectively. Thus, a slight majority of the west section measurements followed the same trend as observed at the east section.

The wind direction-resolved analysis described above indicated the strong influence of wind direction on the relative differences between on-road concentrations in front of a noise barrier. While the underlying physical mechanisms for these results cannot be verified from this study, these results suggest that the turbulence generated on the highway likely results in enough mixing to counteract the blocking effect of the barrier unless winds are perpendicular to the barrier. This vehicle-induced turbulence is often not included in modeling analyses, and the measurements by Baldauf et al. (2008) were upwind of the barrier, not on the highway-side, so would not be influenced by this turbulence. Under these conditions, pollutants would be swept along and away from the barrier to the open, non-barrier section of the highway. Sweeping of pollutants to these open sections would also result in the higher open section measurements and gradients observed in this study. Results from this study might not reflect results for highways with very long stretches of barriers or cut section designs where the pollutants cannot be swept away by vehicle-induced turbulence. However, these results suggest that concerns of increased pollutant concentrations for passengers on the road may not be the concern raised in previous measurement and modeling studies

In addition to understanding how solid noise barriers influence pollutant concentrations away from the highway, understanding how pollutant concentrations vary at the barrier edges is also important in ensuring maximum protection for near-road populations. Although many passes along the parallel access roads behind the noise barrier of the east and west monitoring segments occurred, changes in wind and traffic conditions on both the highway and the access road contributed to high variability in individual measurements. As a result, this study only allowed for a generalized assessment of trends in pollutant concentrations parallel to the highway to determine differences in pollutant concentrations with distance from the barrier edge. Fig. 5 shows mean and standard deviation plots of mobile measurements along the parallel access roads of the east and west sections, approximately 20–30 m from the highway including 10 m from the barrier. The figure shows average concentrations behind the barrier, along the open section, and 50 m on either side of the barrier edge. The 50 m distance was chosen to allow a sufficient number of data points and be consistent with the barrier edge effects reported in Baldauf et al. (2008). The figure also highlights the variability of these measurements through the one standard deviation plots, likely a result of local traffic emissions on the access road. The results of Fig. 5 suggest that reductions in behind-barrier pollutant concentrations generally occur within a 50 m distance of the barrier edge; however, this distance will likely vary depending on meteorological and highway traffic conditions.

4. Conclusions

This field study showed that roadside barriers led to reductions in concentrations of vehicle-emitted pollutants relative to those measured in the absence of barriers. The reductions ranged from 50% within 50 m downwind of the barrier to about 30% as far as 300 m from the barrier. These relatively large reductions are consistent with those seen in the tracer study (Finn et al., 2010) during unstable conditions. Venkatram et al. (2016) describes the comparison of the tracer study and Phoenix field study results on the predicted reductions from a dispersion model.

The results of this study do not show the increase in concentration at about 100–150 m from the wall, reported by Ning et al. (2010), to levels above those corresponding to the no barrier section of the road. This suggests that the dominant effect of roadside barriers is mitigation of exposure to vehicle-related pollutants. The magnitude and spatial extent of the reduction in concentrations behind the barrier is consistent with results from the tracer experiment (Finn et al., 2010) conducted during unstable conditions, even though the barrier heights were different (4.5 m in Phoenix compared to 6 m in the Idaho Falls tracer study) and the surrounding terrain more complex in Phoenix.

The concentrations on the highway upwind of the barrier were similar to those measured in the absence of the barrier, contradicting previous modeling results that suggest that roadside barriers might increase the exposure to vehicle-related pollutants for vehicle passengers on the road. This study result suggests that vehicle-induced turbulence likely counters any blocking effect of the barrier on the highway. Concentration variability at the edge of the barriers can be seen, with concentrations generally decreasing within 50 m of the barrier edge; however, individual concentration measurements were variable, highlighting that meteorology and traffic activity will affect this distance.

Disclaimer

The views expressed in this paper are those of the authors and do not necessarily represent the views or policies of the U.S. Environmental Protection Agency.

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