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The spatial relationship between traffic-generated air pollution and noise in 2 US cities[☆]

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Abstract

Traffic-generated air pollution and noise have both been linked to cardiovascular morbidity. Since traffic is a shared source, there is potential for correlated exposures that may lead to confounding in epidemiologic studies. As part of the Multi-Ethnic Study of Atherosclerosis and Air Pollution (MESA Air), 2-week NO and NO_2 concentrations were measured at up to 105 locations, selected primarily to characterize gradients near major roads, in each of 9 US communities. We measured 5-min A-weighted equivalent continuous sound pressure levels (L_{eq}) and ultrafine particle (UFP) counts at a subset of these NO/NO_2 monitoring locations in Chicago, IL (N = 69 in December 2006; N = 36 in April 2007) and Riverside County, CA (N = 46 in April 2007). L_{eq} and UFP were measured during non-"rush hour" periods (10:00–16:00) to maximize comparability between measurements. We evaluated roadway proximity exposure surrogates in relation to the measured levels, estimated noise–air pollution correlation coefficients, and evaluated the impact of regional-scale pollution gradients, wind direction, and roadway proximity on the correlations. Five-minute L_{eq} measurements in December 2006 and April 2007 were highly correlated (r = 0.84), and measurements made at different times of day were similar (coefficients of variation: 0.5–13%), indicating that 5-min measurements are representative of long-term L_{eq}. Binary and continuous roadway proximity metrics characterized L_{eq} as well or better than NO or NO_2. We found strong regional-scale gradients in NO and NO_2, particularly in Chicago, but only weak regional-scale gradients in L_{eq} and UFP. L_{eq} was most consistently correlated with NO, but the correlations were moderate (0.20–0.60). After removing the influence of regional-scale gradients the correlations generally increased (L_{eq}–NO: r = 0.49–0.62), and correlations downwind of major roads (L_{eq}–NO: r = 0.53–0.74) were consistently higher than those upwind (0.35–0.65). There was not a consistent effect of roadway proximity on the correlations. In conclusion, roadway proximity variables are not unique exposure surrogates in studies of endpoints hypothesized to be related to both air pollution and noise. Moderate correlations between traffic-generated air pollution and noise suggest the possibility of confounding, which might be minimized by considering regional pollution gradients and/or prevailing wind direction(s) in epidemiologic studies.

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Keywords

Air pollution; Noise; Traffic; Confounding; Cardiovascular

1. Introduction

Researchers have reported associations between chronic exposure to traffic and adverse cardiovascular health effects including hypertension, myocardial infarction, stroke, atherosclerosis, heart disease, and mortality. These associations have been attributed to traffic-generated air pollution (Finkelstein et al., 2004; Hoek et al., 2002; Hoffmann et al., 2006, 2007; Maheswaran and Elliott, 2003; Tonne et al., 2007) or road noise (Babisch, 2006; Babisch et al., 2005; Bluhm et al., 2007; de Kluizenaar et al., 2007; Selander et al., 2009; van Kempen et al., 2002). If air pollution and noise are both linked to cardiovascular effects, the fact that traffic is a major shared source suggests the potential for correlated exposures that may lead to confounding in epidemiologic studies (Schwela et al., 2005).

The potential for confounding is increased by the exposure assessment approaches that are commonly used in epidemiologic studies, in which it is not feasible to measure exposure for every participant. As an alternative to measurements, investigations of road noise and health generally use models to estimate noise exposures (Babisch et al., 2005; Beelen et al., 2008; Bluhm et al., 2007; Calixto et al., 2003; de Kluizenaar et al., 2007; van Kempen et al., 2002). Similarly, some studies of traffic-generated air pollution use dispersion and/or "land use regression" models to estimate concentrations (Ainslie et al., 2008; Jerrett et al., 2005a; Su et al., 2008). However, these air pollution models often require spatially dense monitoring or extensive data on emissions and meteorology prior to model development. As a result, simple roadway proximity-based metrics are commonly used as exposure surrogates, in part because they are easily implemented using readily available data and do not require any air pollution measurements (Adar and Kaufman, 2007; Finkelstein et al., 2004; Hoek et al., 2002; Hoffmann et al., 2007, [2006]; Jerrett et al., 2005a, b; Maheswaran and Elliott, 2003; Tonne et al., 2007). These surrogate measures are based on correlations between roadway proximity and measured levels of traffic-generated air pollutants (Beckerman et al., 2008; Gilbert et al., 2007, 2003; Pleijel et al., 2004; Roorda-Knape et al., 1998; Zhu et al., 2002). However, interpretation of epidemiologic studies that use the proximity approach is complicated by the fact that noise levels are also related to roadway proximity (Hothersall and Chandler-Wilde, 1987). The ability of these roadway proximity metrics to predict measured levels of air pollution and noise has not been directly compared.

The published data on the relationship between noise and air pollution are also very limited and somewhat inconsistent. A study in Madrid evaluated the relationship between 1096 daily measurements of noise (measured at 6 locations) and NO_2/NO_x (measured at 24 locations) (Tobias et al., 2001). The authors reported noise-NO_2 and noise-NO_x correlation coefficients of 0.32 and 0.35, respectively. However, while informative for daily time-series studies, these temporal relationships are of limited value in interpreting epidemiologic studies of chronic exposures in which the spatial exposure contrast is of interest. More relevant to chronic effects studies are the findings of Klaeboe et al. (2000), who modeled 24-h L_{eq} and 3-month average NO_2 concentrations based on traffic volumes at approximately 1000 locations in Oslo and reported a modest relationship ($r = 0.46$). In a study of chronic noise exposure and hypertension in the Netherlands, de Kluizenaar et al. (2007) reported a correlation coefficient of 0.72 between modeled noise and modeled annual average PM_{10}. In a study in Germany, Ising et al. (2004) reported a strong correlation ($r = 0.84$) between measurements of nighttime (0:00–6:00) traffic noise and 58–93 h measurements of NO_2 at 25 locations. A recent study in Vancouver, BC, calculated correlations between 5-min noise and 2-week NO_2 and NO_x measured at 103 locations. They reported noise-NO_x and noise-NO_2 correlations of 0.64 and 0.53, respectively (Davies et al., 2009). A study in the Netherlands found a relatively poor correlation between yearly modeled noise and background black smoke ($r = 0.24$) (Beelen et al., 2008). Most recently, Selander et al. (2009) reported a correlation coefficient of 0.6 between long-term modeled estimates of L_{eq} and NO_2 in Sweden. Although these studies suggest the potential for confounding, all but the Vancouver work were conducted in Europe where differences in the vehicle fleet, roadway configuration, fuel composition, and urban design, as well as these studies' frequent reliance on models, may limit the generalizabilty of the results to other settings. In summary, little is presently known about the spatial relationship between traffic-generated air pollution and noise in North America.

Here we present the results from a pilot investigation of the relationship between traffic-generated air pollution and noise in Chicago, IL, and Riverside County, CA. Our primary objective was to assess the potential for confounding in epidemiologic studies of chronic health effects by evaluating the correlations between noise and 3 markers of traffic-generated air pollution: NO, NO_2, and ultrafine particles (UFPs). A secondary objective was to evaluate and compare the ability of simple roadway proximity metrics to predict measured levels of air pollution and noise.

2. Methods

2.1. Pollution measurements

This work leveraged off of the Multi-Ethnic Study of Atherosclerosis and Air Pollution ("MESA Air"). MESA Air is an ongoing investigation of chronic exposure to fine particulate matter ($PM_{2.5}$) and other air pollutants in relation to the progression of subclinical atherosclerosis in 9 US communities. As part of its air pollution exposure assessment efforts, MESA Air collected simultaneous measurements of NO_2 and NO_x at up to 105 locations in each study community. Time-integrated NO_2/NO_x sampling was conducted over 2-week periods using passive Ogawa samplers attached to utility poles at approximately 2.5 m above the ground level. Three such 2-week sampling sessions were conducted in each study community to capture seasonal variations in spatial patterns.

MESA Air NO_2/NO_x sampling locations were selected to ensure spatial coverage around study participants and provide information in regions of high spatial variability of pollution, with emphasis on major roadways (i.e., those in census feature classification code [CFCC] categories A1, A2, and A3). Concentration gradients near roads were captured by deploying 6 samplers along a trajectory perpendicular to major "target" roadways. At each of these "gradient sites" 3 samplers were placed on both sides of the target roadway between 0 and 50, 50 and 100, and 100 and 350 m from the target roadway's edge on small (CFCC category A4) streets or alleys. Samplers were also deployed in non-residential areas to capture the influence of multiple land uses, and additional samples (approximately 10% of the total) were collected at random locations within the study areas. NO concentrations were calculated by subtracting NO_2 from NO_x.

To assess the relationship between traffic-generated air pollution and noise we measured 5-min equivalent continuous sound pressure levels (L_{eq}), reported in units of A-weighted decibels (dB_A), at MESA Air NO_x/NO_2 sampling locations during 2 sampling sessions in Chicago, IL (December 5–8, 2006 and April 10–13, 2007) and 1 sampling session in Riverside, CA (April 17–19, 2007). L_{eq} measurements in Chicago were made using a Larson Davis 870B sound level meter (SLM) (Larson Davis, Depew, New York), and in Riverside we used a Larson Davis 820 SLM. The SLMs were calibrated each morning prior to data collection. For sampling, the SLM was placed on a tripod approximately 1 m above ground level and as close as possible to the utility pole supporting the Ogawa sampler. Five-minute L_{eq} measurements were collected under the strong assumption that these “grab samples” would be representative of long-term noise levels (Davies et al., 2009). During each 5-min L_{eq} sample technicians recorded roadway characteristics and the presence of sporadic sources of noise (e.g., barking dogs, music, etc.). A single SLM was used for each sampling session and technicians moved from site to site, making a measurement at up to 20 locations per day. Noise sampling was conducted between 10:00 and 16:00 to avoid the influence of “rush hour” traffic and to maximize comparability between measurements. We did not sample during rain or snow due to concerns about the generalizability of noise levels measured under such conditions. There was some ice and snow on the ground in Chicago during the 2006 sampling session, and during all monitoring in both cities the field technicians recorded the road conditions (e.g., dry, wet, snow, ice, etc.) at the sampling location.

In addition to measuring noise, we also measured 5-min average UFP using the P-Trak (TSI, Shoreview, MN) during the 2007 sampling sessions in both cities. The P-Trak was zeroed daily prior to sampling. During sampling the instrument was placed on a tripod approximately 1 m above ground level and inside a sound-insulated toolbox near the SLM and the pole to which the Ogawa sampler was attached. The P-Trak has been validated based on laboratory comparisons with other condensation particle counters (CPCs), although a recent comparison in multiple microenvironments reported that the agreement between the P-Trak, which detects particles larger than approximately 30 nm, and a standard CPC depended on particle size and found that the P-Trak significantly underestimated UFP near a highway (Zhu et al., 2006b).

Repeated measurements of 5-min L_{eq} and UFP were collected at selected locations to assess within-session and between-session variability in our measurements. Hourly wind speed at Chicago O’Hare and Riverside Community airports were obtained from the National Oceanic and Atmospheric Administration, and technicians recorded the wind direction during the 5-min noise and UFP measurements in both 2007 sampling sessions.

2.2. Data analysis

The representativeness of our 5-min measurements was assessed by evaluating the variability of measurements made at the same location at different times of day on different days within the same sampling session (both cities), and in different seasons (Chicago only). This was important since we collected 5-min noise measurements but our interest was in long-term levels. For our analyses, we computed averages of noise or UFP at locations with multiple measurements. In addition, to ensure comparability of results across the different pollutants, we included only locations with valid measures of L_{eq}, NO, NO_2, and UFP (if measured).

Because simple roadway proximity measures have become a common exposure surrogate in air pollution epidemiologic studies (Allen et al., 2009; Finkelstein et al., 2004; Hoek et al., 2002; Hoffmann et al., 2007, 2006; Maheswaran and Elliott, 2003; Tonne et al., 2007) we evaluated the relationships between our noise and air pollution measurements and roadway proximity variables (both binary and continuous) computed using the Dynamap 2000 road network (TeleAtlas, Lebanon, NH) in ArcGIS 9.2 (ESRI, Redlands, CA). First we calculated correlation coefficients between measured levels and the logarithm of the distance to 2 roadway categories: nearest major road (defined as CFCC classes A1, A2, and A3) and nearest highway (defined as CFCC classes A1 and A2). We also evaluated the relationships between our measured values and a binary indicator of roadway proximity defined as <100 m from a highway (A1 or A2) or <50 m from a major arterial (A3). This specific exposure surrogate has been used in at least 3 previous air pollution epidemiology studies (Allen et al., 2009; Finkelstein et al., 2004; Hoek et al., 2002). In addition to comparing the near roads and far from roads distributions, we also explored whether this binary variable was successful in stratifying the samples into high- and low-level groups, as defined by being above or below the median levels of noise and air pollution. We report the results by pollutant as the percent of observations correctly classified.

Finally, we calculated Pearson's correlation coefficients between 5-min L_{eq} and each of the traffic pollution indicators: NO (2-week average), NO_2 (2-week average), and UFP (5-min average). In order to disentangle the regional (i.e., 10s to 100s of kilometers) and local-scale (i.e., 10s to 100s of meters) gradients of each pollutant, we first estimated the regional-scale gradients in air pollution and noise by conducting session-specific stepwise regressions (p<0.05 to enter and p<0.05 to remain in the model) of our measured values on variables expected to capture these large-scale gradients: distance to the city center in Chicago and location (i.e., distance north and distance east) within the study area in both cities. The variability from local sources was then estimated as the difference between our measurements and these modeled regional gradient surfaces. We calculated noise–air pollution correlation coefficients based on both the unadjusted measurements and the measurements after removing the influence of regional gradients. In addition, we calculated the correlation coefficients between noise and air pollution at gradient monitoring sites after stratifying measurements into locations upwind or downwind of the target roadway and less than or more than 100 m from the target roadway. For correlations involving NO or NO_2, we categorized gradient monitoring locations into upwind or downwind based on the target roadway configuration and the dominant wind direction(s) measured at the airports over the 2-week sampling period (Fig. 1). The correlations between 5-min L_{eq} and UFP measurements were categorized based on target roadway configuration and wind directions measured at O'Hare airport (2006 sampling session) or observed by field technicians during the 5-min measurements (2007 sampling sessions).

3. Results

We sampled noise at 74 of 103 locations with NO_2/NO_x measurements during the first sampling session in Chicago (Table 1). Due to equipment problems and a snow storm the 2007 sampling session in Chicago included only 37 locations for L_{eq} and 50 for UFP. In Riverside, we obtained valid noise and UFP measurements at 49 of 50 NO_2/NO_x measurement sites (fewer NO_2/NO_x samplers were deployed in Riverside because the MESA Air study area is smaller). Restricting our analyses to only those sites for which all measured pollutants were available resulted in 69 sites (covering approximately 2300 km^2) for 2006 in Chicago, 36 (covering approximately 800 km^2) for 2007 in Chicago, and 46 (covering approximately 400 km^2) for Riverside.

As expected based on the roadway gradient site selection strategy in MESA Air, we captured a substantial amount of variability in NO, NO_2, UFP, and L_{eq} (Table 1). The range in L_{eq} was approximately 25 dB_A during all 3 monitoring sessions. There were some important differences in the monitoring locations between the 3 monitoring sessions. A lower proportion of measurements were made near highways (CFCC categories A1 or A2) during the first session in Chicago (10%) than during the 2007 sessions in Chicago (22%) or Riverside (26%) (Table 1). In addition, the proportion of gradient monitoring sites focused on highways was lower in Chicago (5 of 13 groups of samples in 2006 and 4 of 8 in 2007) than in Riverside (6 of 7). Average hourly wind speeds during noise measurements were higher in Chicago (6.4–6.7 m/s) than in Riverside (3.8 m/s) (Fig. 1).

We assessed temporal variation in our 5-min measurements and their ability to represent longer averaging times by comparing measurements made at different times of day within our 10:00–16:00 window (Table 2). Noise measurements were stable over time, with coefficients of variation (CV) at individual locations ranging between 0.5% and 12.9%. In contrast the UFP measurements were much more variable; the UFP CVs ranged between 58% and 67%, indicating that our 5-min UFP measurements do not provide a reliable estimate of longer-term concentrations.

We also evaluated the long-term variability in 5-min L_{eq} measurements by comparing measurements from the December 2006 and April 2007 sampling sessions in Chicago. The 22 locations sampled during both sessions were highly correlated (r = 0.84; Fig. 2). The absolute value differences in repeat L_{eq} measurements between the 2 seasons ranged between 0.3 and 9.0 dB_A, with a median of 2.3 dB_A. These results suggest that 5-min measurements made between 10:00 and 16:00 are stable across seasons and are representative of long-term average noise levels during those times of day. The between-season correlations for NO and NO_2 at this subset of locations with noise measurements were moderate, with correlation coefficients of 0.59 and 0.42, respectively. Across all sites with NO/NO_2 data in both seasons (N = 72) the between-season correlations for NO and NO_2 were 0.65 and 0.56, respectively.

Stepwise regression of our measured values on distance to city and location within the study area revealed some strong regional gradients, particularly for NO and NO_2 (Table 3). In Chicago, combinations of these predictors explained 49–54% of the variability in NO and 70–82% of the variability in NO_2. Distance to city alone explained 78% of the December, 2006 NO_2 variance in Chicago (with decreasing concentrations at increasing distances from the city). The air pollution gradients were less pronounced in Riverside, although distance from the southern edge of the study area explained 17% and 27% of the variability in NO and NO_2, respectively. As expected, our L_{eq} and UFP measurements showed very weak regional-scale spatial patterns.

Our gradient sampling sites captured substantial variation in the measured pollutants in relation to distance from the “target” roadway (Fig. 3). When comparing regional gradient-adjusted concentrations <100 m vs. > 100 m from the target road, levels of NO and L_{eq} were significantly (p<0.05) higher near the target road during all sessions, while NO_2 was only elevated within 100 m during both of the 2007 sampling sessions (potentially due to the greater emphasis on highways in those sessions). UFP did not differ by road proximity, although this result is complicated by the temporal variation in UFP measurements described above. NO was most sensitive to wind direction, with significant differences by road proximity only on the downwind side, and consistently higher correlations with logarithmic distance to the target road on the downwind side.

We evaluated the relationships between our measurements, adjusted for regional gradients to focus on local sources of variability, and simple surrogates of exposure based on roadway proximity (Table 4). L_{eq}, NO, and NO_2 were generally correlated with both distance metrics. The logarithm of distance to nearest major road was slightly better correlated with L_{eq} (r = −0.41 to −0.52) than NO (r = −0.30 to −0.46) or NO_2 (r = −0.22 to −0.40). Conversely, modest correlations with logarithmic distance to the nearest highway did not demonstrate consistent trends for NO (r = −0.43 to −0.53), NO_2 (r = −0.39 to −0.59), or L_{eq} (−0.18 to −0.57).

We also explored a binary roadway proximity variable (where “near roads” was defined as <100 m from an A1 or A2 road or <50 m from an A3 road) in relation to regional gradient-adjusted pollutant measurements. We compared the ability of this proximity variable to correctly identify levels of noise and air pollution by calculating the number of observations that were “correctly classified,” which was defined as measurements made near roads that were above the median level or measurements made distant from roads that were below the median. This binary roadway proximity variable consistently provided a better characterization of noise (65–87% of measurements classified correctly) than of NO (59–72%), NO_2 (61–70%), or UFP (44–61%).

The overall correlations between measured L_{eq} and traffic-generated air pollution were moderate during both of the 2007 sampling sessions (left half of Table 5), with correlation coefficients of 0.40–0.60 for NO and 0.38–0.46 for NO_2 (all $p<0.05$). In contrast, the correlations during the 2006 sampling session in Chicago were weak for NO ($r = 0.20$) and NO_2 ($r = -0.08$). After removing the influence of the regional variation primarily in NO and NO_2, correlations between noise and air pollution generally increased (right half of Table 5). The most dramatic increase was in the L_{eq}–NO correlation for 2006 in Chicago, which increased from 0.20 ($p<0.10$) to 0.49 ($p<0.01$). Wind speed was correlated with noise only during the December 2006 monitoring session in Chicago, although adjusting for wind speed did not influence the noise–air pollution correlations. Other noise sources recorded by technicians were not associated with measured noise levels (results not shown).

Finally, we further examined the noise–air pollution correlations by evaluating the impact of wind direction and road proximity on the measurements made at roadway gradient monitoring sites (Table 6). The L_{eq}–NO correlations were consistently higher downwind of major target roads ($r = 0.53$–0.74; all $p<0.05$) than they were upwind ($r = 0.35$–0.65). Although there was not a significant relationship between L_{eq} and NO_2 in either wind direction during the 2006 sampling session, the L_{eq}–NO_2 correlations during both 2007 sampling sessions were also higher downwind ($r = 0.57$ and 0.71) than upwind ($r = 0.37$ and 0.60). The effect of road proximity was not consistent across cities. In Chicago, higher correlations were noted within 100 m of the target road, while in Riverside we generally found higher correlations for sites greater than 100 m from the target road.

4. Discussion

To our knowledge, this is the first investigation of the relationship between traffic-related air pollution and noise in the US. The temporal variability of noise was found to be much lower than that of NO or NO_2 in Chicago, perhaps due to the greater impact of meteorology on air pollution concentrations. In fact, 5-min grab samples of noise repeated at the same location were found to be quite stable over time (between-season $r = 0.84$). This stability was extremely important for this study since we used single instruments to obtain 5-min grab samples due to the costs and security concerns associated with deploying multiple monitors. Furthermore, these results suggest that short-term noise measurements may provide a useful indicator of long-term averages of community noise in the absence of extensive noise monitoring or data-intensive noise models. Alberola and colleagues (2005) analyzed hourly noise measurements (6:00–19:00) collected over 2 weeks at 50 locations impacted primarily by road noise and found that variability in hourly measurements increased with decreasing noise level. This suggests that our 5-min measurements may have been most representative of longer-term averages at the loudest locations, i.e., near major roads. In contrast to our noise measurements, we did not observe temporal stability for UFP, which demonstrated large differences in concentrations across time. This instability is likely due to the fact that as primary pollutants with short atmospheric lifetimes, UFPs are highly sensitive to the type of vehicles and the local wind conditions (Zhu et al., 2006a). Because of this variability and the fact that 10–26% of our measurements were made within 100 m of a highway, where the P-Trak has been shown to significantly underestimate UFP concentrations, we place greatest emphasis on the results for the other traffic-related air pollutants, NO and NO_2.

Logarithmic distances to nearby major roads were only moderately (r≈0.4–0.6) correlated with L_{eq}, NO, and NO_2. These correlations are lower than in previous studies of the relationship between NO_2 and logarithmic distance to roads, which have reported correlations of 0.83 in the Netherlands (Roorda-Knape et al., 1998), 0.94 in Montreal (Gilbert et al., 2003), and 0.97 in Sweden (Pleijel et al., 2004). One possible explanation for this discrepancy is that our sampling was conducted on a variety of target roads, including major arterials, to estimate concentrations in areas near MESA Air residences, while previous studies focused primarily on highways. The fact that simple proximity measures were found to predict noise at least as well as air pollution suggests that these exposure surrogates are not a unique identifier in epidemiologic studies of health outcomes for which noise and air pollution are both hypothesized causes.

NO and NO_2 were moderately correlated with noise levels in both cities. These correlations were generally similar to the noise-NO_2 and/or noise-NO_x correlations reported in Oslo, Stockholm, and Vancouver (Davies et al., 2009; Klaeboe et al., 2000; Selander et al., 2009), but lower than the noise-PM10 (r = 0.72) and noise-NO_2 (r = 0.84) correlations reported in the Netherlands and Germany, respectively (de Kluizenaar et al., 2007; Ising et al., 2004).

Since both NO and NO_2 had clear gradients across the study areas (noise and UFP had very weak regional-scale spatial patterns), these larger trends were found to impact our noise–air pollution correlations. Large-scale pollution gradients may also explain the relatively low correlations between noise and background black smoke (r = 0.24) observed in a recent study in the Netherlands (Beelen et al., 2008). These findings imply that epidemiologic investigations exploiting the influence of local roadways might be more vulnerable to confounding than studies focused on larger regional differences. However, the importance of this finding is difficult to quantify as the relative importance of localized vs. regional pollution gradients has not yet been determined. Numerous studies have reported adverse cardiovascular health effects among individuals residing in close proximity to major roads (Finkelstein et al., 2004; Hoek et al., 2002; Hoffmann et al., 2007, 2006; Maheswaran and Elliott, 2003; Tonne et al., 2007), suggesting an important role for more localized pollution gradients, although 2 recent studies have reported elevated risks in relation to urban scale fine particulate matter air pollution gradients across the Los Angeles urban area (Jerrett et al., 2005b; Kunzli et al., 2005).

Wind direction was an important modifier of the relationships between air pollution and noise. NO concentrations and distance decay relationships were most sensitive to wind direction. In contrast, noise had similar distance decay relationships upwind, and the similarity of 5-min noise measurements made at 22 locations in different seasons (with different wind characteristics) provides further evidence that noise was minimally impacted by wind direction. These differential wind direction effects impacted the noise–air pollution correlations; the strongest, most consistent correlations were observed between noise and NO downwind of major roads (r = 0.53–0.74).

Some important limitations of this study should be considered. First, our noise (5-min) and NO/NO_2 (2-week) measurements were conducted over different time periods. Although we cannot determine the impact of this difference on the observed correlations, one might speculate that measurements made over the same duration would be better correlated than those with significantly different durations, suggesting that the correlations presented here are underestimates. In addition, since our “grab sample” approach to characterizing spatial patterns also involved a temporal component, we limited our noise measurements to non-rush hour periods to minimize the influence of temporal differences on our assessment of spatial patterns. However, this approach may have also biased the observed correlations with air pollution, or may not represent the most biologically relevant correlations if nighttime noise exposures are most detrimental to health (Babisch, 2006). An additional limitation to this study design was our dependence on wind speed and direction measured at a single location in each community.

While the primary focus of this paper has been confounding in cardiovascular epidemiology, consideration of the interplay between noise and air pollution may also be important for respiratory health research. Several investigations have reported associations between traffic-generated air pollution and the development and exacerbation of asthma (Brauer et al., 2007; Gauderman et al., 2005; Gordian et al., 2006), and recent research suggests that stress may modify these effects (Chen et al., 2008; Clougherty et al., 2007). Noise is thought to act on cardiovascular health through repeated noise-induced stress responses (Babisch, 2002; Maschke et al., 2000) suggesting the potential need to also assess noise exposure in studies of traffic-related air pollution and asthma.

In conclusion, moderate correlations suggest the potential for confounded results if both noise and air pollution are not accurately assessed in epidemiological studies of traffic and health. Although very few epidemiologic studies have included both air pollution and noise in health effects models (Beelen et al., 2008; de Kluizenaar et al., 2007; Schwela et al., 2005; Selander et al., 2009), imperfect correlations between these exposures present opportunities for disentangling their impacts on health, and methods for analyzing correlated environmental exposures in health effects studies continue to emerge (Dominici et al., 2008; MacLehose et al., 2007; Thomas, 2007). Future studies of endpoints such as MI, for which there are hypothesized physiological mechanisms and preliminary epidemiological evidence implicating both noise (Babisch et al., 2005; Selander et al., 2009) and air pollution (Tonne et al., 2007), may require more sophisticated exposure assessments involving measurements and/or models of both pollutants. Consideration of prevailing wind direction(s) and/or regional-scale air pollution gradients may allow investigators to minimize the potential for confounding.

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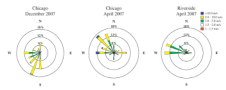

Fig. 1.
Hourly wind roses during the three 2-week NO_2/NO_x sampling sessions.

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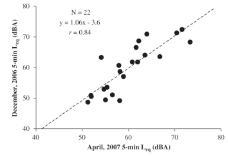

Fig. 2.
Relationship between repeated 5-min L_{eq} measurements in Chicago in December 2006 and April 2007.

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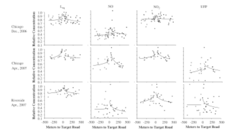

Fig. 3.

Relative concentrations adjusted for regional trends at monitoring sites upwind (white dots) and downwind (black dots) of targeted major roadway. Lines are logarithmic functions fit to the upwind and downwind data.

Table 1

Summary statistics of sampling locations' proximities to major roads and pollution measurements by city and sampling session.

		Chicago, December 2006 (*N* = 69)	**Chicago, April 2007 (*N* = 36)**	**Riverside, April 2007 (*N* = 46)**
Meters to the nearest A1 road	Mean±SD (Median)	1405±1310 (968)	639±671 (390)	842±1157 (270)
	Min–Max	37–4721	34–3390	34–3517
	% <100 m	4.3%	19.4%	26.1%
Meters to the nearest A2 road	Mean±SD (Median)	1985±1579 (1862)	2434±1733 (2074)	11,453±4367 (9991)
	Min–Max	20–5750	59–5750	5651–19,777
	% <100 m	5.7%	2.8%	0%
Meters to the nearest A3 road	Mean±SD (Median)	193±216 (108)	155±131 (117)	323±282 (261)
	Min–Max	4–921	4–405	1–846
	% <100 m	44.9%	47.2%	30.4%
Meters to the nearest major road (A1–A3)	Mean±SD (Median)	129±113 (94)	105±97 (62)	149±161 (95)
	Min–Max	4–409	4–310	1–666
	% <100 m	53.6%	63.9%	52.2%
NO (ppb)	Mean±SD (Median)	25.4±10.2 (23.6)	22.5±6.9 (21.3)	20.1±12.1 (18.8)
	Min–Max	11.2–71.5	9.7–42.1	6.1–68.1
NO_2 (ppb)	Mean±SD (Median)	17.7±4.7 (18.2)	20.4±4.5 (20.9)	18.2±5.4 (16.9)
	Min–Max	7.6–27.9	10.4–29.3	11.3–33.7
NO/NO_2 ratio (unitless)	Mean±SD (Median)	1.4±0.4 (1.3)	1.1±0.2 (1.1)	1.1±0.4 (1.1)
	Min–Max	0.7–2.9	0.7–1.4	0.4–2.0
UFP[a] (1000 × p/cm^3)	Mean±SD (Median)	-	25.6±18.4 (23.3)	11.8±5.2 (10.7)
	Min–Max	-	7.8–113.5	4.7–25.5
L_{eq} (dB$_A$)	Mean±SD (Median)	59.3±7.0 (59.1)	60.4±6.2 (59.3)	61.5±6.8 (62.2)
	Min–Max	48.1–72.4	51.3–73.4	48.3–74.4
Wind speed during noise measurements (m/s)[b]	Mean±SD (Median)	6.4±1.1 (6.7)	6.7±1.3 (7.2)	3.8±1.5 (3.6)
	Min–Max	3.6–8.9	4.0–8.0	0.0–6.7

[a]UFP statistics in 2007 monitoring session in Chicago were influenced by one extreme observation. After excluding this observation the Mean ±SD, Median, and Min–Max for the remaining 35 observations were 23.1±10.7, 23.2, and 7.7–44.5 1000 p/cm^3, respectively.

[b]Based on hourly measurements at Chicago O'Hare and Riverside Community airports.

Table 2

Summary statistics for repeated measurements made on different days and at different times of day during specific monitoring sessions.

Sampling session	Location	L_{eq} (dB_A)				UFP (× 1000 p/cm^3)			
		N	Mean	SD	CV[a] (%)	*N*	Mean	SD	CV (%)
Chicago, December 2006	A	7	69.0	3.1	4.5	0	-	-	-
	B	7	62.2	3.4	5.4	0	-	-	-
Chicago, April 2007	B	5	62.5	1.5	2.3	5	37.0	21.3	57.6
	C	4	71.6	0.4	0.5	4	36.8	23.0	62.3
Riverside, April 2007	A	8	53.9	7.0	12.9	8	11.7	7.8	66.5
	B	8	72.0	3.0	4.1	8	13.7	9.2	67.4

[a] CV = coefficient of variation (SD/Mean).

Table 3

Stepwise regression equations used to model regional gradients.

City	Monitoring session	Pollutant	Final model	Model R^2
Chicago	December 2006 (N = 69)	L_{eq} (dB_A)	53.68+(0.12 km north)	0.08
		NO (ppb)	37.05−(0.57 km to City)	0.49
		NO_2 (ppb)	22.42−(0.37 km to City)+(0.06 km north)	0.82
	April 2007 (N = 36)	L_{eq} (dB_A)[a]	-	-
		NO (ppb)	81.15−(0.61 km to City)−(0.44 km north)−(1.02 km east)	0.54
		NO_2 (ppb)	54.86−(0.35 km to City)−(0.34 km north)−(0.51 km east)	0.70
		UFP (× 1000 p/cm^3)	38.33−(0.96 km north)	0.12
Riverside	April, 2007 (N = 46)	L_{eq} (dB_A)[a]	-	-
		NO (ppb)	11.06+(1.05 km north)	0.17
		NO_2 (ppb)	12.83+(0.59 km north)	0.27
		UFP (× 1000 p/cm^3)a	-	-

[a]No variables met criteria for entry into the stepwise regression model.

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Table 4

Pearson's correlation coefficients between pollutants and logarithm of roadway proximity after adjusting for regional gradients.

Metric	Session	L_{eq}	NO	NO_2	UFP[a]
Logarithm of distance to nearest major road[b]	Chicago 2006 (*N* = 69)	**−0.41*** ***	−0.40***	−0.22*	-
	Chicago 2007 (*N* = 36)	**−0.51** **	−0.46***	−0.40**	0.16 (0.04)
	Riverside 2007 (*N* = 46)	**−0.52*** ***	−0.30**	−0.31**	−0.51***
Logarithm of distance to nearest highway[c]	Chicago 2006 (*N* = 69)	−0.18	**−0.51*** ***	−0.50***	-
	Chicago 2007 (*N* = 36)	**−0.57*** ***	−0.43***	−0.39**	−0.16 (−0.16)
	Riverside, 2007 (*N* = 46)	−0.42***	−0.53***	**−0.59*** ***	0.01

The highest correlated pollutant is shown in bold.

Correlations after adjusting measurements using regional gradient modeling equations in Table 3 (see text).

[a] Value in parentheses is the correlation after removing one extreme UFP value from the 2007 monitoring session in Chicago.

[b] Major road defined as census feature classification code A1, A2, or A3.

[c] Highway defined as census feature classification code A1 or A2.

* $p < 0.10$.

** $p < 0.05$.

*** $p < 0.01$.

Table 5

Pearson's correlation coefficients between L_{eq} and air pollutants.

		Raw correlation			Adjusted for regional gradient[a]		
		NO	**NO_2**	**UFP[b]**	**NO**	**NO_2**	**UFP[b]**
Chicago, December 2006 (*N* = 69)	NO_2	0.71***			0.25**		
	UFP	-	-		-	-	
	L_{eq}	0.20*	−0.08	-	0.49***	0.16	-
Chicago April 2007 (*N* = 36)	NO_2	0.82***			0.65***		
	UFP[b]	0.45*** (0.57***)	0.45*** (0.51***)		0.17 (0.35**)	0.16 (0.16)	
	L_{eq}	0.60***	0.38**	0.26 (0.33*)	0.62***	0.41**	0.22 (0.31*)
Riverside April 2007 (*N* = 46)	NO_2	0.74***			0.68***		
	UFP	−0.05	0.11		−0.02	0.18	
	L_{eq}	0.40***	0.46***	0.41***	0.50***	0.62***	0.41***

[a]Correlations after adjusting measurements using regional gradient modeling equations in Table 3 (see text).

[b]Value in parentheses is the correlation after removing one extreme UFP value from the 2007 monitoring session in Chicago.

* $p<0.10$.

** $p<0.05$.

*** $p<0.01$.

Table 6

Pearson's correlation coefficients at roadway gradient monitoring sites after adjusting for regional gradients. The number of measurements used to calculate the correlation is in parentheses.

Session	Pollutants	Correlations								
		All gradient sites	Upwind	Downwind	> 100 m[a]	< 100 m[b]	Upwind		Downwind	
							> 100 m[a]	< 100 m[b]	> 100 m[a]	<
Chicago December, 2006	NO–NO_2	0.23^{*} (61)	0.18 (33)	0.31 (28)	0.09 (34)	0.51^{***} (27)	0.00 (18)	0.63^{**} (15)	0.20 (16)	0
	L_{eq}–NO	0.49^{***} (61)	0.35^{**} (33)	0.67^{***} (28)	0.32^{*} (34)	0.53^{***} (27)	0.30 (18)	0.40 (15)	0.44^{*} (16)	0
	L_{eq}–NO_2	0.14 (61)	0.18 (33)	0.08 (28)	0.06 (34)	0.37^{*} (27)	0.15 (34)	0.25 (15)	−0.16 (16)	0
Chicago, April 2007	NO–NO_2	0.60^{***} (32)	0.05 (16)	0.90^{***} (16)	0.29 (17)	0.43 (15)	−0.44 (7)	−0.25 (9)	0.75^{**} (10)	0
	NO–UFP	0.27 (32)	0.40 (16)	0.22 (16)	0.30 (17)	0.16 (15)	0.71^{*} (7)	0.31 (9)	−0.02 (10)	−
	NO_2–UFP	−0.04 (32)	−0.22 (16)	0.05 (16)	−0.25 (17)	−0.17 (15)	−0.42 (7)	−0.03 (9)	−0.18 (10)	−
	L_{eq}–UFP	0.31 (32)	0.46^{*} (16)	0.34 (16)	0.08 (15)	0.37 (15)	0.35 (8)	0.40 (8)	−0.10 (9)	0
	L_{eq}–NO	0.66^{***} (32)	0.65^{***} (16)	0.74^{***} (16)	0.22 (17)	0.62^{**} (15)	0.44 (7)	0.72^{**} (9)	0.02 (10)	0
	L_{eq}–NO_2	0.48^{***} (32)	0.37 (16)	0.57^{**} (16)	0.00 (17)	0.22 (15)	0.13 (7)	−0.01 (9)	0.03 (10)	0
Riverside, April 2007	NO–NO_2	0.65^{***} (36)	0.47^{**} (19)	0.82^{***} (17)	0.40^{*} (22)	0.71^{***} (14)	0.51 (11)	0.50 (8)	0.47 (11)	0
	NO–UFP	−0.02 (36)	−0.05 (19)	0.05 (17)	0.17 (22)	−0.24 (14)	0.09 (11)	−0.19 (8)	0.29 (11)	−
	NO_2–UFP	0.22 (36)	0.11 (19)	0.33 (17)	0.28 (22)	0.08 (14)	−0.06 (11)	0.24 (8)	0.53^{*} (11)	1
	L_{eq}–UFP	0.53^{***} (36)	0.68^{**} (13)	0.47^{**} (23)	0.50^{**} (22)	0.58^{**} (14)	0.70^{*} (7)	0.79^{*} (6)	0.41 (15)	0
	L_{eq}–NO	0.45^{***} (36)	0.36 (19)	0.53^{**} (19)	0.50^{**} (22)	0.26 (14)	0.54^{*} (11)	0.12 (8)	0.47 (11)	0
	L_{eq}–NO_2	0.59^{***} (36)	0.60^{***} (19)	0.71^{***} (19)	0.65^{***} (22)	0.31 (14)	0.55^{*} (11)	0.32 (8)	0.74^{***} (11)	0

Notes: Correlations after adjusting measurements using regional gradient modeling equations in Table 3 (see text).

For correlations with NO/NO_2, upwind and downwind are based on road configuration and 2-week NO/NO_2 sampling period wind roses shown in Fig. 1. For correlations between L_{eq} and UFP, upwind and downwind are based on technician observations during the 5-min sampling period.

[a] Sites < 100 m from the "target" roadway.

[b] Sites > 100 m from the "target" roadway.

* $p<0.10$.

** $p<0.05$.

*** $p<0.01$.