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Impact of transient truck and train traffic on ambient air and noise levels in underserved communities

Inkyu Han[a,b,*], **Lara Samarneh**[c], **Thomas H. Stock**[a,b], **Elaine Symanski**[a,b]

[a]Department of Epidemiology, Human Genetics, and Environmental Sciences, University of Texas Health Science Center School of Public Health, 1200 Pressler Street, Houston, TX 77030, United States

[b]Southwest Center for Occupational and Environmental Health, 1200 Pressler Street, Houston, TX 77030, United States

[c]Department of Biomedical Engineering, University of Texas at Austin, 110 Inner Campus Drive, Austin, TX 78705, United States

Abstract

Traffic-related air and noise pollution on or near major roadways have been examined but these pollutants have not been extensively investigated away from major roadways in residential communities, especially in the United States. To evaluate the impact of trucks and trains passing nearby on air and noise pollution in residential areas during non-rush hours, we simultaneously measured concentrations of size-resolved airborne particulate matter (PM) and sound pressure levels as A-weighted equivalent (dBA) with frequencies in three underserved communities adjacent to industrial facilities in Houston, TX. We found that median concentrations for PM_1 (particle size ≤ 1 μm) and PM_{10} (particle size ≤ 10 μm) were highest when trucks passed by at sampling locations, followed by periods when trains passed by. PM_1 and PM_{10} concentrations were lowest at background (defined when there was no truck or train traffic near the monitoring location). Median $PM_{2.5}$ (particle size ≤ 2.5 μm) mass concentrations were 19.8 μg/m^3 (trains), 16.5 μg/m^3 (trucks), and 13.9 μg/m^3 (background). Short-term increases in noise were attributed to trains and trucks passing nearby as well. The median noise levels were the highest when trains passed by (66.7 dBA) followed by periods when trucks were in the vicinity of the monitoring locations (62.5 dBA); background levels were 58.2 dBA. The overall Spearman correlation coefficients between air and noise pollution were between 0.09 and 0.46. Hence, we recommend that both air pollutant and noise levels be concurrently evaluated for accurate exposure assessment related to traffic sources in residential communities.

Keywords

particulate matter (PM); Size-fractionated PM; Noise; Frequency; Traffic; Disparities

*Corresponding author: Inkyu.Han@uth.tmc.edu (I. Han).

1. Introduction

People living in urban environments may be concurrently exposed to air and noise pollution from various sources. This pollution is generated from passenger cars and trucks on roadways, industrial activities, and trains transporting goods to or from industrial facilities. Traffic-related airborne particulate matter (PM) levels are the highest on roadways and are elevated at locations near heavily trafficked roadways (Shu et al., 2014; Zhu et al., 2002). Moreover, ambient air PM is easily dispersed by wind and the levels of PM far (> 300 m) from major roadways have been measured to be 90% of those measured near the roadway (Zhu et al., 2002). Industrial sources are another important contributor to air pollution in urban areas. For example, Han et al. (2017) found that more than half of $PM_{2.5}$ in Baton Rouge, Louisiana was attributed to secondary sulfate and industrial emission sources. Another study determined that road dust and oil-refinery sources contributed more than 63% of PM_{10} in Houston, Texas (Bozlaker et al., 2013).

Environmental noise in urban residential areas is positively associated with total traffic volume on roadways and the density of industrial facilities. For instance, King et al. (2016) found that the noise level measured on a sidewalk near a roadway in New York City was 41% higher than at a background site. Another study estimated that a traffic count of 40 vehicles/min increased the noise level by 20 dBA (A-weighted equivalent level for 16 h (7 AM to 11 PM)), compared with no traffic (Zuo et al., 2014). Elevations in traffic-related noise is heavily influenced by the number of trucks and buses (Ross et al., 2011). Furthermore, industrial sources also significantly increased noise levels in residential areas. In Toronto, Canada, an industrial facility area of 0.015 km^2 within 100 m of a residential area increased the measured noise level by approximately 10 dBA compared with similar communities without any industrial facilities within a 100-m radius (Zuo et al., 2014). Thus, communities near industrial sites may be seriously affected by traffic-related air and noise pollution which can contribute to adverse health outcomes.

Environmental exposure to air and noise pollution is an important risk factor for adverse cardiovascular outcomes. For example, cardiovascular disease (CVD) mortality and hospitalization are strongly associated with exposure to airborne PM and its constituents (Ito et al., 2011; Niu et al., 2013; Thurston et al., 2016). Ito et al. (2011) reported that $PM_{2.5}$ mass was significantly associated with an increase in CVD mortality of 1.0–2.0%. It is known that inhaled PM and its constituents induce systemic inflammation and oxidative stress resulting in a wide range of CVD effects (Gong et al., 2014; Roy et al., 2014; Weichenthal et al., 2014). Recent epidemiological studies about noise pollution in urban areas suggest that traffic-related noise pollution can increase the risk of cardiovascular disease (Barceló et al., 2016; Halonen et al., 2017; Vienneau et al., 2015). Barcelo et al. (2016) reported that noise exposure occurring throughout the day, evening, and nighttime was associated with a 2% increase in myocardial infarction mortality for males in Barcelona, Spain. Other studies showed that an increase of 5 dB in daytime traffic noise was significantly associated with a 3% increase in hypertension (Babisch, 2014; Babisch et al., 2012). The European Union established that noise pollution causes almost 8 million sleep disturbances and over 900,000 cases of hypertension each year in Europe (European Environment Agency, 2014).

Although noise consists of a wide range of frequencies, the impact of exposure to different noise frequencies is not well understood. When low frequency noise was the dominant component of sources, dBA measurements did not accurately represent the perceived loudness of noise (Leventhall, 2009). Another study found that perceived noise annoyance was associated with exposure to low-frequency noise (Torija and Flindell, 2015). Chang et al. (2011) observed that associations between noise exposure and risk for hypertension were strongest for low frequencies. Walker et al. (2016) also found that low frequency noise reduced the heart rate variability among 10 males in a controlled audio test room. Trucks and trains primarily generate low-frequency noise, which comes from their engines, contact with paved roads or rails when they are moving. Several studies have reported moderate correlation between traffic-related air pollutants and noise near major roadways (Allen et al., 2009; Shu et al., 2014) or street canyons (Can et al., 2011).

Urban residents typically spend more than 70% of their time in their homes (Peters et al., 2017; Su et al., 2015). Although air and noise pollution has been assessed adjacent to major roadways, the combined exposure to air and noise pollution in residential areas has not been well characterized. In Houston, where there is no formal zoning code, there are residential communities, which are located near industrial facilities and rail lines that generate truck and train traffic. These communities are often low-income and minority, and, hence share an unequal burden of air and noise pollution in the city. As articulated by the U.S. Environmental Protection Agency, environmental justice is achieved when all communities and people experience the same degree of protection from exposure to hazards (US EPA, 2018). Yet, a growing body of evidence suggests that health disparities among US communities have widened (Casey et al., 2017a; Grineski et al., 2017) and that underserved communities (defined as low-income, minority dominant, or less access to social infrastructure) are facing poorer environmental quality (Chakraborty et al., 2017; Gee and Payne-Sturges, 2004). Bell and Ebisu (2012) found that Non-white races have higher exposure to specific constituents of $PM_{2.5}$ and disparities in noise exposure have been reported as well (Carrier et al., 2016; Casey et al., 2017b). Houston neighborhoods widely vary in terms of sociodemographic, economic, and environmental quality because of different land-use patterns in and surrounding residential areas (Chakraborty et al., 2017; Collins et al., 2017). Thus, the objective of this study was to measure air and noise pollution in communities of mixed land use and to evaluate the transient impact of train and truck traffic on air and noise pollution in underserved communities.

2. Material and methods

2.1. Study area

The study neighborhoods were previously selected for a community-based participatory research (CBPR) project supported by the National Institute of Environmental Health Sciences (NIEHS) whose purpose is to measure and evaluate risks due to levels of metal air pollutants in four disadvantaged neighborhoods in Houston and apply public action plans to improve environmental health literacy and environmental quality in these communities. In our ancillary study, we conducted air and noise measurements in three of the four neighborhoods where trucks and trains routinely transport materials or commercial goods to and from the industrial facilities.

Within these neighborhoods, we selected sampling locations following these criteria: a residential area mostly consisting of private homes; at least one industrial facility within 400 m of where air and noise monitoring would occur; a rail track close by (within 100 m) but at least 500 m away from major highways. Fig. 1(a) shows the map of the three sampling locations selected for this study.

The indicators of demographics and environmental quality in these neighborhoods were obtained using the United States Census Bureau American Community Survey 2011–2015 Data (US Census Bureau, 2016) and the United States Environmental Protection Agency Environmental Justice Screening and Mapping Tool (EJSCREEN) Version 2016 (US EPA, 2016). EJSCREEN is an environmental justice and screening tool developed by US EPA using combined data from the United States Census Bureau and other governmental institutes. It provides population characteristics and potential environmental quality issues for locations within the U.S. The demographic and environmental data for our study neighborhoods are summarized in Table 1.

2.2. Air and noise pollution measurement

Noise and air sampling were conducted on 11 days between November 18, 2015 and June 16, 2016. To avoid the potential impact of rush-hour traffic on air quality and noise measurements in the three communities, monitoring at each site occurred between 09:00 and 13:00. We measured ambient air PM mass concentrations by size (0.25–30 μm) using a Grimm 11-R Universal Aerosol Spectrometer (Grimm Technologies, Douglasville, GA) and A-weighted noise level as dBA (L_{eq}, 1 min) with frequencies using a SoundTrack LXT sound level meter (Larson-Davis, Depew, NY) on each sample day. We used dBA to evaluate environmental noise because this scale is considered to be an expression of the relative loudness of sounds in air as perceived by the human ear. The noise frequencies ranged from 31.5 Hz to 8000 Hz. Prior to deployment of each instrument, we synchronized date and time with a laptop computer using software provided by both manufacturers. On each sampling day, we used a HEPA filter for zero air to check potential contamination of the inlet port of the Grimm 11-R. Prior to monitoring noise levels on each sampling day, we also performed calibration of the SoundTrack LXT following the manufacturer's recommendations.

A portable table was set up approximately 8–14 m from railways where residential houses are located. Although these houses are not adjacent to major freeways, diesel-powered trucks occasionally passed by during our field measurements because of the transportation of materials and goods to and from the industrial facilities near these houses in the communities. Air sampler was placed on the portable table and the sound level meter was mounted on a tripod. The height of sampling inlet or microphone was approximately 1.2 m above the ground level (see Table 1 and Fig. 1). We simultaneously turned on the instruments to monitor PM mass concentrations by size and sound pressure levels with frequencies at one minute intervals. After completion of sampling, we downloaded the PM data using Grimm Spectrometer 1.178 software (Grimm Technologies, Douglasville, GA) and the sound level data using SLM Utility G3 (Larson-Davis, Depew, NY) to a laptop computer for further data analyses. On each sampling day, a field investigator stayed at the sampling location. During the sampling period, an investigator recorded the time and duration as a truck or a train passed by the sampling location. We classified three environments while sampling occurred: “truck”, “train”, and “background”. (1) We defined the background environment when there was no truck or train traffic. (2) The truck environment was defined when a 18-wheeler truck (we refer hereafter as “truck”) passed by the sampling location. When a truck passed by the sampling location, air and noise levels increased and reached peak concentrations and then decreased to background levels. (3) The train environment was defined when a train passed by the sampling location.

2.3. Data analysis

To examine the effects of trucks and trains on air and noise pollution in three communities, we calculated 1-min average concentrations of PM mass by different particle size modes or noise levels using the raw data from the Grimm 11-R and the SoundTrack LXT, respectively. One investigator recorded information about time, duration, and type of mode of transportation (truck or train) and these data were added to the database of noise measurements. Another investigator independently reviewed the database and flagged observations if the time or duration did not match with the measured data. Both investigators reviewed the flagged observations by checking the original raw data and the records written on the sampling logbook. After validation, we finalized the database for data analyses.

We used the GRIMM 11-R PM mass concentrations for PM_1 (particle size ≤ 1 μm in aerodynamic diameter), $PM_{2.5}$ (particle size ≤ 2.5 μm), and PM_{10} (particle size ≤ 10 μm). For sound level data, we used A-weighted sound pressure level as dBA with frequencies. We calculated the average sound pressure level for 1-min following the equation:

$$L_T = 10\log\left(\sum_{i=1}^{n} 10^{L_i/10}\right)$$

where, L_T = average sound pressure level (dBA),

n = number of measured sound pressure levels,

L_i = measured individual sound pressure level at time = *i*,

i = time.

Prior to the data analysis, we examined the normality of the data for PM_1, $PM_{2.5}$, PM_{10}, sound pressure levels (dBA) and frequencies (Hz). We found that none of these data was normally distributed; thus, we used log-transformed data for the descriptive statistical analyses: selected percentiles of the data and minimum and maximum values. We also calculated arithmetic means and standard deviations for the measured air pollutants and sound pressure levels. Additionally, we performed the same analysis categorized by the three different scenarios (background, truck, and train). We used the Kruskal Wallis non-parametric test to compare the median difference among the three periods. Lastly, we examined the Spearman correlations between particle size distributions and frequencies by the three different periods.

3. Results

3.1. Descriptive analysis

The three sampling sites in this study were 0.77–2.08 km away from the nearest major roadway (at least 4 lanes in each direction) and had 10–17 industrial facilities within a 800 m radius from the sampling locations (Table 1). Each sampling location was close to active rail lines. The distance from each sampling location to the closest railroad track ranged from 8.0 m to 13.7 m. Table 1 also provides sociodemographic characteristic of residents living in these neighborhoods. Hispanics or Hispanics and blacks comprised 90 to 98 percent of the total population and had low per capita income ($11,674–$14,840).

We performed data analyses for three environments: (1) truck(s) passing nearby, (2) train(s) passing nearby, and (3) background, where there was no truck or train traffic passing near sampling locations. Because we sampled during non-rush hour periods (9:00 AM to 1:00 PM), most of the sampling times were classified as background. There were no significant additional sources of air pollution and noise except lawn mowing activities. As such, we excluded measurements that were collected when we observed that lawn mowing was taking place during monitoring. In this study, we had 20 occasions of truck traffic, 28 occasions of train traffic, and 2 occasions of truck and train traffic. Due to the limited occurrences of simultaneous train and truck traffic, we excluded these data from our analyses.

Over the 11 sampling days in the three different neighborhoods, the median concentrations of airborne PM were the highest for PM_1 and PM_{10} as trucks passed by, followed by train and background periods (Fig. 2). Fig. 2(a) shows that median concentrations of PM_1 were 9.0 μg/m^3 during periods when trucks passed by. Compared to periods when trucks were coming and going, the median concentrations of PM_1 were 27% and 48% lower for the train (6.6 μg/m^3) and background periods (4.7 μg/m^3), respectively. For $PM_{2.5}$ mass concentrations, median concentrations of $PM_{2.5}$ for trains (19.8 μg/m^3) were 17% and 30% higher than truck and background periods (Fig. 2(b)). Fig. 2(c) showed that median concentrations of PM_{10} with both train and truck periods were significantly higher than background periods. For sound pressure levels, Fig. 2(d) shows that the median noise levels were the highest for the train periods (66.7 dBA) followed by the truck (62.5 dBA) and background (58.2 dBA) periods.

Mean concentrations of $PM_{2.5}$ and PM_{10} were significantly increased when trains or trucks were passing by the sites A and B as compared to measured background levels (Table 2). Site A did not show increases in mean concentrations of PM_1 when trains passed by but Sites B and C showed increases in mean concentrations of PM_1 when trains passed by. We observed the mean levels of noise were significantly higher during train or truck periods than background periods at all locations.

Fig. 3 shows the overall correlations among airborne PM size fractions and noise levels. The log-transformed PM_1 had the highest correlation with log-transformed $PM_{2.5}$ (Spearman correlation coefficient, $r_s = 0.79$), and a lower correlation with PM_{10} ($r_s = 0.29$). A moderate correlation was observed between $PM_{2.5}$ and PM_{10} ($r_s = 0.67$). Correlations of noise levels were relatively low with PM_1 ($r_s = 0.09$), $PM_{2.5}$ ($r_s = 0.35$), asnd PM_{10} ($r_s = 0.46$).

3.2. PM mass by different particle size mode

We observed that concentrations of PM mass varied by different particle size modes for each of the environmental settings (truck, train and background). Table 3 summarizes the median PM mass concentrations with interquartile ranges for different particle size modes. For particle sizes < 0.5 µm, we observed that the median PM mass concentrations were the highest for the truck periods followed by the train and background periods. For particle size modes from 0.5 µm to 4 µm, we found that the median PM mass concentrations for the train periods were significantly higher than for both truck and background periods, except for particle size 2.00 µm. For larger particle sizes (> 5 µm), we observed that the median PM mass concentrations were the highest for truck periods, followed by train and background periods.

Fig. 4 shows cumulative frequency percentiles of mass concentrations of selected particle size modes. In general, particle mass concentrations of both truck and train were higher than those of background. Particle sizes less than 2.5 µm (Fig. 4(a)–(c)) show that mass concentrations smaller than median values (50th percentiles) were not greatly different among truck, train, and background. However, the differences of particle mass concentrations among truck, train, and background were greater as the cumulative frequency probability was larger than median values (50th percentiles). For large particle size (Fig. 4(d)), truck and train always showed larger particle mass concentrations than background levels.

3.3. Noise pollution and frequencies

We found that median noise levels for each frequency were significantly higher as trains or trucks passed by, compared to background (Table 4). The differences were the largest for the lowest frequency (31.5 Hz). The median noise level of the train periods at this low frequency range (31.5–250 Hz) was 43% higher than background and the median noise level of the train periods at the frequency of 8000 Hz was 18% higher than background ($p < 0.001$). Similarly, the median noise level of the truck periods at the frequency of 31.5 Hz was 27% higher than background and the median noise level of the truck periods at the frequency of 8000 Hz was 13% higher than background ($p < 0.001$).

Fig. 5 shows the cumulative frequency distributions of noise at 65 Hz (Fig. 5(a)) and 4000 Hz (Fig. 5(b)). At low frequency of 65 Hz, noise levels were always higher for both train and truck than background. Especially, periods of train traffic showed approximately 10 dB higher than background levels across all ranges of data. At high frequency of 4000 Hz, noise levels were higher for both train and truck traffic than background. However, the magnitude of the difference was smaller at a frequency of 4000 Hz than at a frequency of 65 Hz. Small differences were mostly observed in percentiles below the median.

3.4. Association of air pollution and noise pollution by particle size modes and frequencies

Fig. 6 shows the heat map of Spearman correlation analyses between particle size modes (0.25–10 μm) and frequencies (31.5–8000 Hz). It shows associations between each particle size mode and individual frequencies. In general, there was no association between small particle size (< 0.8 μm) and noise frequencies for all three environments. The association between fine particle size (1–3 μm) was observed to increase across environments. For background, we observed that Spearman correlation coefficients (r_s) ranged from 0.30 to 0.45 between the fine particle size modes (1–2.5 μm) and the lowest frequency (31.5 Hz) (Fig. 4(a)). For both train and truck environments, we found that the Spearman correlation coefficients were higher (r_s: 0.41–0.73) between the fine particle size modes (1–2.5 μm) and the lowest frequency (31.5 Hz) compared to the background sites (Fig. 3(b) and (c)). In coarse particle size mode (> 3 μm), we found that the Spearman correlation coefficients (r_s) ranged from 0.36 to 0.71 with low frequencies (31.5–250 Hz) for both truck and train environments, whereas the Spearman correlation coefficients were lower (r_s: −0.05 to 0.30) with low frequencies for the background. The results did not show significant correlation coefficients between any particle size modes (0.25–10 μm) and medium to high frequencies (500 Hz to 4000 Hz) for all three environments.

4. Discussion

We examined traffic-related air and noise pollution in residential areas of three underserved communities that were not in close proximity to busy roadways but located near industrial facilities in Houston, TX. We observed that median mass concentrations of PM_1 and PM_{10} were always the highest during periods when trucks passed nearby, followed by periods when trains passed nearby, with the lowest levels occurring during background periods, defined as when neither a truck nor a train passed nearby the monitoring locations. Median concentrations of $PM_{2.5}$ were the highest when trains passed by compared to the periods of trucks and background. Similar to ambient $PM_{2.5}$ results, A-weighted noise levels (dBA) were the highest for the train periods, followed by the truck and background periods. The overall results suggest that short-term environmental air and noise pollution during non-rush hours in residential communities are significantly affected by passing trains and trucks related to neighboring industrial activities.

In general, we found that both passing trucks and trains significantly increased PM mass concentrations for small size bins (0.25–1 μm) compared with background. It is well documented that PM mass emissions from diesel combustion mostly consists of particle sizes smaller than 1 μm. The results are similar to the findings from previous studies. For example, Yoon et al. (2015) examined that the ratio of $PM_1/PM_{2.5}$ mass emissions by heavy-duty diesel trucks in an ambient dilution tunnel was 0.96, indicating that the $PM_{2.5}$ mass consists mostly of PM_1. Our study showed that median $PM_1/PM_{2.5}$ ratio for train, truck, and background environments were 0.48, 0.48, and 0.75, respectively. Our study showed lower ratios of $PM_1/PM_{2.5}$ than results from previous studies. The discrepancies may be related to the different environmental settings. Higher ratios of $PM_1/PM_{2.5}$ from Yoon et al. (2015) were observed in tunnel environments where the emitted PM from diesel engines was not rapidly dispersed or diluted by ambient air. However, our study locations are open environments in residential areas where the emitted PM was rapidly diluted by ambient air.

The contribution of both truck and train traffic to coarse PM ($PM_{10–2.5}$) in residential neighborhoods has not been fully examined. In our study, we examined the impact of passing trains and trucks on ambient air coarse PM. We observed that concentrations of the coarse particle mode (2.5–10 μm) was 1.5–2 times higher when trains were passing nearby and 2–4 times higher when trucks were passing nearby, compared with background. The elevation of coarse PM levels is due to passing trains and trucks resuspending dust on the surface of railways or streets. We found that median ratios of $PM_{10–2.5}/PM_{10}$ were 0.66 (trains), 0.75 (trucks), and 0.58 (background). This indicates that trains and trucks passing nearby contribute 13% and 29% more, respectively, to coarse PM concentrations in the background air of the residential communities in our study. It is well known that sources of coarse PM in urban areas are classified into two categories: natural sources such as sea spray, dust storm, and surface soils; and anthropogenic sources such as industrial activities, construction, and traffic-induced resuspended coarse PM (Charron and Harrison, 2005; Harrison et al., 2001). The authors found that the ratios of coarse PM to PM_{10} ranged from 0.18 to 0.53 in London, UK. Other studies conducted in the North America and Australia reported that the proportion of coarse PM were between 0.31 and 0.62 (Chan et al., 1997; Chow et al., 2006; Clements et al., 2014). Thus, the increases in 0.08 (trains) and 0.17 (trucks) for coarse PM in this study are likely attributed to the resuspension of dust when trains and trucks passed by the sampling locations.

We noted that noise levels were elevated by passing trains or trucks in the residential areas. The passing trains and trucks increased noise levels by 15% and 7%, respectively, relative to baseline levels (median = 58.2 dBA). Davies et al. (2009) measured noise pollution in Vancouver, BC and found that a light density of heavy-truck traffic elevated noise levels by 17% compared with no heavy truck traffic (mean = 55.3 dBA). A recent study examined the contribution of railways and road traffic to assess environmental noise exposure using a land-use regression (LUR) model. The authors determined that the railways and road traffic (all vehicles) increased environmental noise levels by 9% and 14%, respectively, compared with no major noise sources (mean = 57.4 dBA) (Ragettli et al., 2016). Consistent with earlier studies, our results confirm that trains and trucks contribute significantly to noise pollution in residential areas. The noise ordinance in the City of Houston (COH) in residential areas has established maximum permissible sound levels of 65 dBA and 58 dBA during daytime hours (7:00 AM – 10:00 PM) and nighttime hours (10:00 PM – 7:00 AM), respectively. The median A-weighted noise level from trains (66.7 dBA) during daytime in our study exceeded 65 dBA. However, we measured noise levels during transient periods. Nonetheless, our results indicate a need for noise monitoring over daytime and nighttime hours, particularly in neighborhoods in close proximity to railways. Should noise levels exceed permissible sound levels, mitigation measures might include restrictions when train engineers blow their horns.

Low-frequency noise is common in urban environments and its effects are of particular concern due to its pervasiveness, causing annoyance and possibly adverse health outcomes

(Chang et al., 2014; Leventhall, 2004). Thus, we also examined the effects of passing trains and trucks on changes of noise frequency. We noted that low to medium frequencies of noise (31.5 – 500 Hz) were primarily attributed to passing trains whereas medium to high frequencies of noise (> 1,000 Hz) were elevated due to passing trucks. Notably, the differences between train periods and background for 63 Hz (low frequency) was the largest (11.1 dB) while the differences between truck periods and background for 63 Hz was only 3.5 dB. For high frequency (4,000 Hz), the median difference between train periods and background was 5 dB but the median difference between truck periods and background was greater (7.4 dB). This shows that elevated noise levels measured as L_{eq} are similar for both trains (12% increase) and trucks (10% increase) but the mechanism of noise generation is different. Terlich (2013) measured sound pressure levels with frequencies for diesel locomotive trains and found that the highest noise levels were detected at low frequencies (31.5 and 63 Hz). The major frequencies of truck noise are known to fall between 100 and 1000 Hz (Chang et al., 2014; Hostmad et al., 2015; Liu et al., 2016; Ross et al., 2011).

Many investigators have examined the association between traffic-related particulate matter and noise pollution in urban areas. Weber and Litschke (2008) found that the correlation between noise and PM_1 ranged from 0.03 to 0.53 in urban areas. Other studies reported that the correlation between noise and $PM_{2.5}$ varied between −0.17 and 0.39 (Boogaard et al., 2009; Gan et al., 2012; Ross et al., 2011). Our results also showed that the correlations between noise and PM mass concentrations by different sizes were low ($r_s = 0.09$) to moderate ($r_s = 0.46$). This suggests that sources of PM and noise in urban residential areas vary widely rather than being due to a common origin. Although we did not measure the emissions of PM and noise from other industrial facilities near our sampling locations, there are many industrial facilities in operation within the 800 m buffer zones. For example, one of sampling locations was surrounded by 17 industrial facilities within the 800 m buffer zone. Due to the absence of zoning in Houston, a residence can be located near multiple industrial facilities.

The strengths of our study include the simultaneous measurement of size-resolved PM mass concentrations and sound pressure levels with frequencies in different environmental conditions in underserved communities. In addition, we assessed community exposures to air and noise pollution in residential areas adjacent to railways and mixed industrial facilities that have not been previously studied. Most existing studies have focused on road-traffic noise and air pollution near heavily-trafficked roadways (King et al., 2016; Ross et al., 2011; Shu et al., 2014).

A major limitation of our study is that we did not measure size-resolved PM mass concentrations and sound pressure levels for 24 h or longer, which did not allow us to evaluate the association between air and noise pollution during the periods of morning and evening rush hours, as well as extended daytime or overnight periods. Thus, the short-term measurements of air and noise pollution in this study should not be used to assess daily or long-term exposure assessment for air and noise pollution. Another limitation is that we did not simultaneously measure meteorological parameters (e.g., wind speed and wind direction) at the sampling locations although we frequently checked the weather conditions using publicly available weather information (e.g., wundeground.com) at the nearest weather station from the sampling locations, especially wind direction, during the sampling periods. However, these qualitative hourly reported meteorological parameters were not used in this study because we were not able to match with the 1-min log data of airborne PM and noise. In future research, this limitation could be addressed using newly developed sensors providing real-time information about the exposition of traffic noise and air traffic-related pollutants in urban areas (Kheirbek et al., 2014; Mydlarz et al., 2017). These sensors are also equipped with a weather station measuring temperature, wind, and humidity simultaneously.

5. Conclusion

This study showed that air and noise pollution in underserved communities were impacted by passing train and truck traffic related to industrial activities. The passing trains and trucks significantly increased PM_1, $PM_{2.5}$, and coarse PM ($PM_{10–2.5}$) from background levels. The trains and trucks also elevated noise levels. We found that passing trains contributed increased low-frequency noise, whereas passing trucks primarily increased mid ranges of frequency noise in this study. Levels of air pollutants and noise were not strongly correlated, suggesting that the sources of air and noise pollution vary widely, although passing trains and trucks substantially added to background levels. This study suggests that air and noise pollution should be measured simultaneously for improved exposure assessment of traffic-related pollution.

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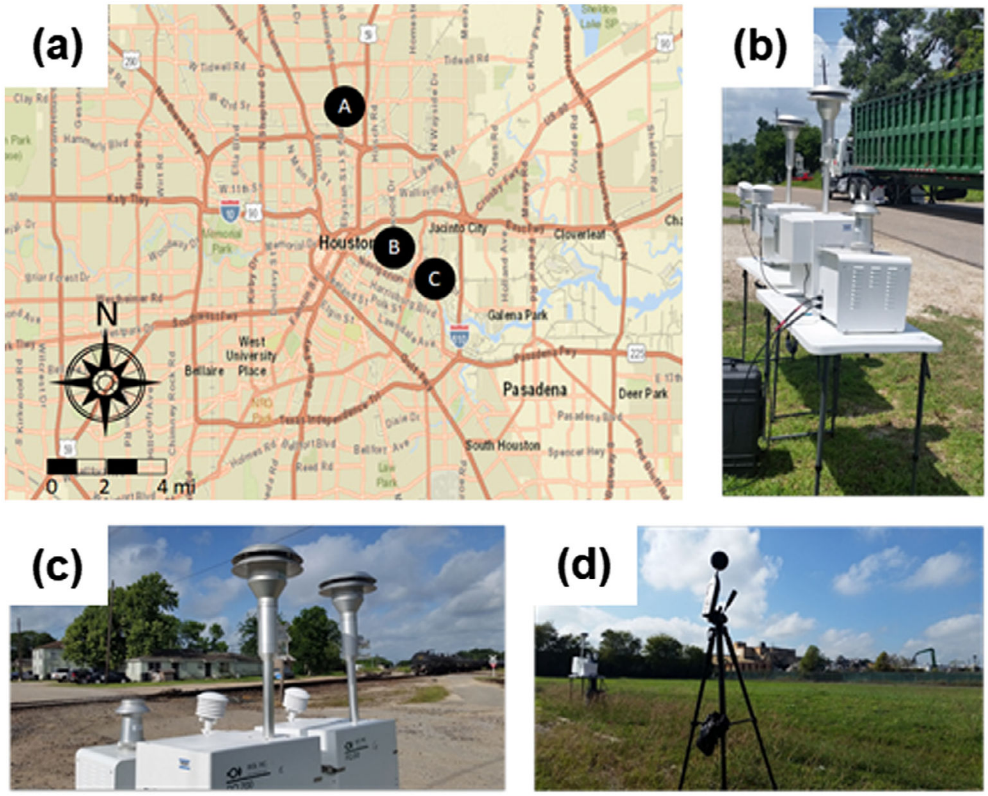

Fig. 1.

Locations of 3 neighborhoods where air and noise sampling devices were deployed (Fig. 1a); an example of a truck passing by near sampling location (Fig. 1b); an example of a train passing by in residential area (Fig. 1c); and an example of sampler set-up at a sampling location (Fig. 1d).

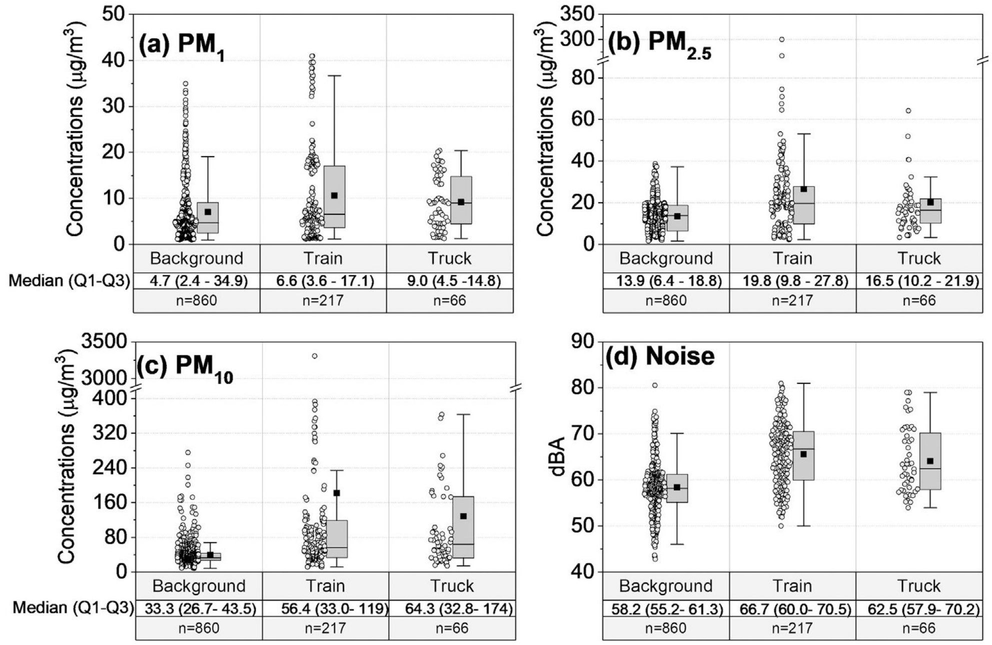

Fig. 2.

Boxplots of PM_1 (Fig. 1a), $PM_{2.5}$ (Fig. 1b), PM_{10} (Fig. 1c), and A-weighted noise level (Fig. 1d) among three different environments. Median levels and interquartile ranges (Q1: 25th percentile and Q3: 75th percentile) are shown for PM and noise levels. Number of observation represents each data point with 1-min log interval.

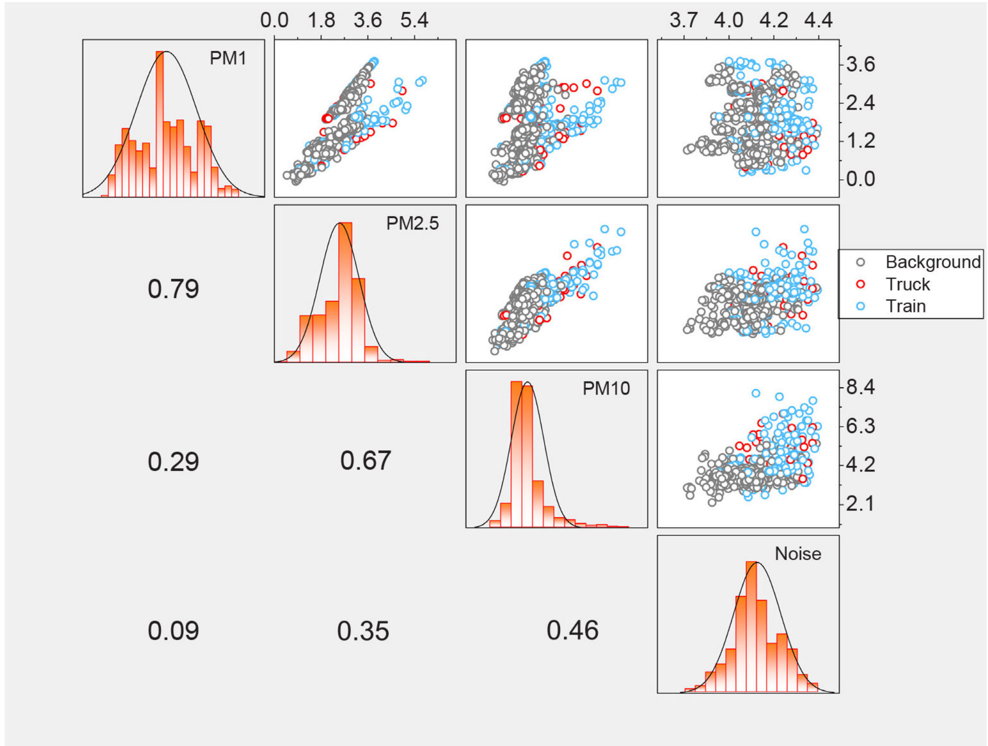

Fig. 3.

Correlation matrix among PM_1, $PM_{2.5}$, PM_{10}, and noise. All raw data were transformed as natural log values. The numbers on the left bottom of panel are Spearman correlation coefficients using the log-transformed data.

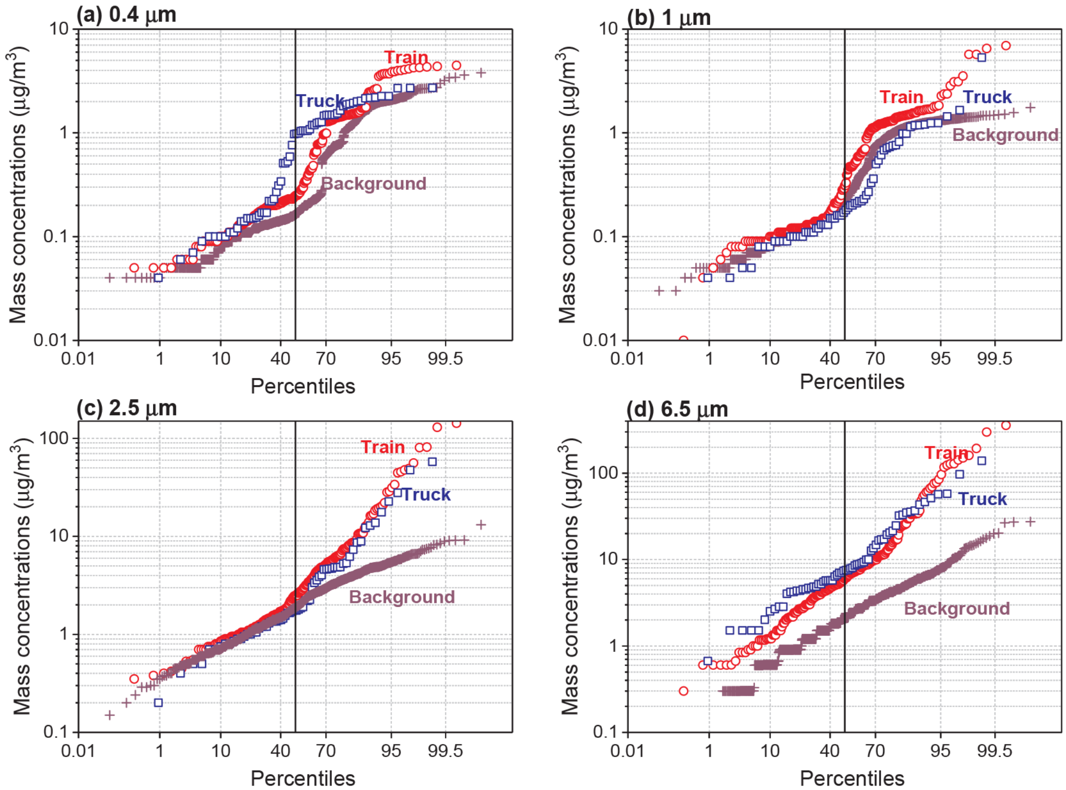

Fig. 4.

Probability plots of size-fractionated particle mass concentrations for train, truck, and background with the selected particle sizes of 0.4 µm (Fig. 3(a)), 1 µm (Fig. 3(b), 2.5 µm (Fig. 3(c), and 6.5 µm (Fig. 3(d)). Open circle (○) shows train, open square (□) shows truck, and cross (+) shows background environments. The solid line on the X-axis represents 50th percentiles.

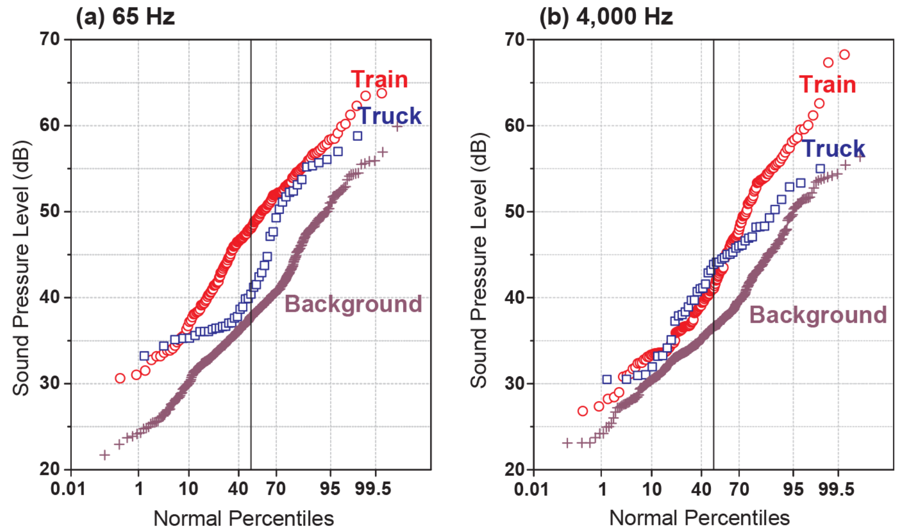

Fig. 5.

Probability plots of sound pressure levels for train, truck, and background at the frequency of 65 Hz (Fig. 4(a)) and 4000 Hz (Fig. 4(b)). Open circle (○) shows train, open square (□) shows truck, and cross (+) shows background environments. The solid line on the X-axis represents 50th percentiles.

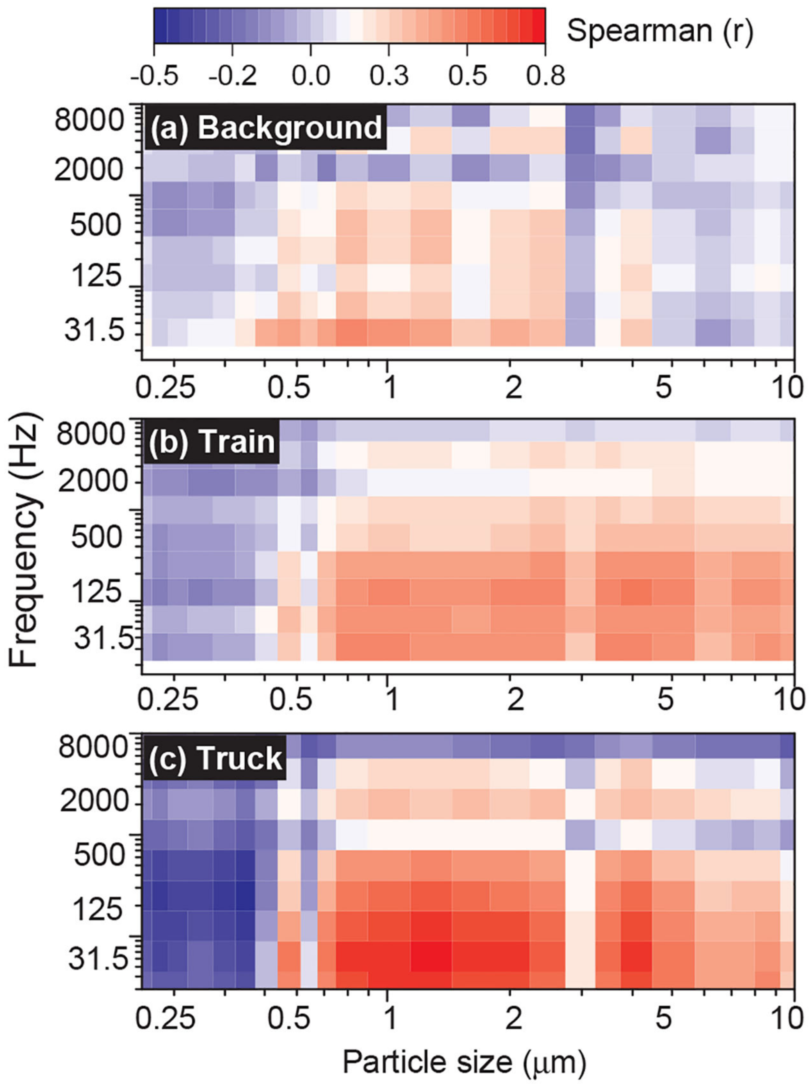

Fig. 6.

Correlations between each particle size mode (0.25–10 μm) and individual frequencies (31.5–8000 Hz) for background, train, and truck periods. The heat map was determined with Spearman correlation coefficient for each variable.

Table 1

Environmental and demographic indicators of sampling locations and residential communities.

Environmental and demographic Indicators		Site 1	Site 2	Site 3	USA Average
Air	Particulate matter ($PM_{2.5}$)[a]	11.9 μg/m^3	11.7 μg/m^3	11.8 μg/m^3	9.32 μg/m^3
Traffic	Traffic proximity and volume (daily traffic count/distance to road)[a]	2600	280	180	590
	Distance to nearest major roadway from the sampling location[b]	1.62 km (1.01 mile)	0.77 km (0.48 mile)	2.08 km (1.29 mile)	N/A
	Distance to nearest railroad from sampling location[b]	13.7 m	10.5m	8.0 m	N/A
Facility	RMP proximity (facility count/km distance)[a]	0.98	1.7	0.44	0.43
	No. of industrial facilities within 0.5 mile buffer from sampling location[b]	17	10	15	N/A
Demographic[c]	Total population[d]	3385	3522	4327	316,515,021
	Hispanic (%)	2253 (67%)	3256 (92%)	3789 (88%)	54,232,205 (17%)
	Black (%)	910 (27%)	196 (6%)	95 (2%)	38,785,726 (12%)
	White (%)	201 (6%)	58 (2%)	400 (9%)	197,258,278 (62%)
	Per capita income	$ 11,674	$ 14,840	$ 13,191	$ 28,930
	Not employed	1143 (46%)	971 (36%)	1218 (38%)	91,308,201 (36%)
	Speak English less than well	601 (19%)	1014 (32%)	1455 (37%)	N/A

[a] US EPA EJSCREEN Report with 0.5 mile buffer at each community site. Traffic proximity and volume determined by US EPA as count of vehicles per day at major roads within 500 m or (nearest one beyond 500 m), divided by distance in meters. RMP is defined by US EPA as potential chemical accident management plan facilities within 5 km or nearest one beyond 5 km, each divided by distance in km.

[b] The distance to nearest major roadway and railroad from each sampling location was measured using geo coordinates. Number of industrial facilities within 0.5 mile buffer from sampling location was obtained from City of Houston.

[c] The demographic information at each site with 0.5 mile buffer was obtained by US Census Bureau, American Community Survey (ACS) 2010–2014.

[d] Other race (e.g., Asian, Native Hawaiian and other Pacific Islander) is not included.

Table 2

Size selective airborne particulate matter and noise levels at three sampling locations.

Site	Environment	Occasion	No. of observations	Pollutants	Mean ± SD	Median (Q1, Q3)
Site A	Background	N/A	333	PM_1	6.85 ± 6.32	4.52 (2.14, 10.01)
			333	$PM_{2.5}$	13.07 ± 7.65	12.87 (5.31, 17.59)
			333	PM_{10}	42.86 ± 33.85	31.26 (27.35, 44.02)
			333	dBA	61.19 ± 4.58	59.92 (57.95, 63.26)
	Train	15	114	PM_1	5.38 ± 3.88	5.05 (2.51, 6.32)
			114	$PM_{2.5}$	32.58 ± 45.31	20.06 (7.62, 31.51)
			114	PM_{10}	304.1 ± 514.9	116.7 (50.94, 319.3)
			114	dBA	67.78 ± 6.27	67.93 (62.91, 72.15)
	Truck	6	25	PM_1	4.12 ± 2.94	3.93 (2.21, 4.66)
			25	$PM_{2.5}$	27.70 ± 31.27	18.77 (7.93, 28.53)
			25	PM_{10}	194.2 ± 234.7	92.95 (68.40, 242.0)
			25	dBA	68.48 ± 7.00	70.23 (63.43, 73.32)
Site B	Background	N/A	324	PM_1	8.32 ± 7.36	7.52 (1.85, 14.68)
			324	$PM_{2.5}$	10.73 ± 7.39	9.29 (4.26, 16.44)
			324	PM_{10}	34.51 ± 19.84	30.79 (21.42, 42.03)
			324	dBA	52.55 ± 4.20	52.97 (48.74, 55.38)
	Train	11	97	PM_1	16.86 ± 11.28	17.23 (8.08, 19.25)
			97	$PM_{2.5}$	19.60 ± 12.22	18.95 (9.79, 21.70)
			97	PM_{10}	46.58 ± 25.83	36.13 (29.24, 62.24)
			97	dBA	62.51 ± 6.75	62.01 (55.96, 68.47)
	Truck	14	41	PM_1	12.23 ± 4.90	13.05 (9.08, 16.36)
			41	$PM_{2.5}$	15.52 ± 7.19	15.49 (10.37, 18.73)
			41	PM_{10}	88.12 ± 116.1	49.67 (32.76, 77.18)
			41	dBA	59.70 ± 3.53	59.72 (57.01, 61.95)
Site C	Background	N/A	203	PM_1	5.38 ± 1.14	4.87 (4.55, 6.60)
			203	$PM_{2.5}$	18.50 ± 4.37	18.01 (15.23, 21.56)
			203	PM_{10}	39.79 ± 10.14	38.96 (33.33, 45.55)
			203	dBA	54.44 ± 4.52	53.00 (51.43, 57.03)
	Train	2	6	PM_1	7.24 ± 0.18	7.25 (7.14, 7.32)
			6	$PM_{2.5}$	24.88 ± 1.64	25.35 (23.15, 26.24)
			6	PM_{10}	48.32 ± 1.27	48.17 (47.16, 49.24)
			6	dBA	61.78 ± 6.98	63.89 (57.69, 65.58)
	Truck	N/A	Not measured	PM_1	N/A	N/A
				$PM_{2.5}$	N/A	N/A
				PM_{10}	N/A	N/A
				dBA	N/A	N/A

Unit: μg/m^3 for PM_1, $PM_{2.5}$, and PM_{10}. dBA for noise level.

Q1 and Q3 represent interquartile ranges (Q1: 25th percentile and Q3: 75th percentile).

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Table 3

Median and interquartile ranges of particle mass concentration by each particle size mode for three different environments.

Particle size (μm)	Background	Train	Truck	p-value[a]
0.25	0.80 (0.49, 2.52)	0.81 (0.87, 4.62)	**2.14 (0.50. 2.73)**	0.079
0.28	0.55 (0.33, 2.00)	0.57 (0.36, 3.75)	**1.78 (0.32. 2.34)**	0.028
0.30	0.43 (0.22, 1.29)	0.48 (0.27, 2.46)	**1.22 (0.28. 1.71)**	0.001
0.35	0.31 (0.17, 1.01)	0.43 (0.23, 1.76)	**1.12 (0.26. 1.79)**	< 0.001
0.40	0.16 (0.12, 0.74)	0.25 (0.16, 1.38)	**0.98 (0.15. 1.49)**	< 0.001
0.45	0.13 (0.09, 0.44)	0.19 (0.12, 0.89)	**0.67 (0.12. 1.04)**	< 0.001
0.50	0.23 (0.12, 0.34)	**0.36 (0.20. 0.45)**	0.33 (0.19, 0.49)	< 0.001
0.58	0.19 (0.09, 0.32)	**0.27 (0.12. 0.43)**	0.22 (0.13, 0.31)	< 0.001
0.65	0.17 (0.09, 0.25)	**0.25 (0.14. 0.32)**	0.22 (0.15, 0.34)	< 0.001
0.70	0.15 (0.07, 0.34)	**0.22 (0.10. 0.51)**	0.18 (0.11, 0.30)	< 0.001
0.80	0.14 (0.06, 0.43)	**0.20 (0.06. 0.65)**	0.11 (0.07, 0.30)	0.012
1.00	0.23 (0.12, 0.88)	**0.31 (0.12. 1.23)**	0.18 (0.11, 0.68)	< 0.001
1.30	0.32 (0.17, 1.07)	**0.39 (0.19. 1.57)**	0.24 (0.17, 0.84)	< 0.001
1.60	0.54 (0.33, 2.80)	**0.78 (0.36. 4.26)**	0.49 (0.29, 2.80)	< 0.001
2.00	**1.05 (0.47. 3.21)**	1.03 (0.44, 4.93)	0.56 (0.38, 3.28)	0.039
2.50	1.87 (1.15, 3.29)	**2.49 (1.20. 6.02)**	1.75 (1.05, 4.81)	< 0.001
3.00	2.51 (1.24, 3.72)	**4.68 (2.71. 7.34)**	4.43 (2.81, 9.42)	< 0.001
3.50	1.81 (1.02, 2.74)	**2.60 (1.39. 6.82)**	2.28 (1.57, 6.03)	< 0.001
4.00	3.53 (1.95, 6.01)	**3.62 (1.72. 18.76)**	3.16 (1.86, 15.35)	0.003
5.00	3.52 (2.51, 5.18)	6.86 (4.01, 18.73)	**7.90 (4.73. 23.81)**	< 0.001
6.50	2.11 (1.21, 3.92)	6.19 (3.01, 11.47)	**7.45 (4.52. 17.20)**	< 0.001
7.50	1.51 (0.90, 3.02)	4.22 (1.81, 9.96)	**4.89 (2.72. 14.41)**	< 0.001
8.50	1.39 (0.70, 3.48)	4.18 (1.80, 12.53)	**6.06 (2.70. 17.40)**	< 0.001
10.0	2.50 (1.25, 4.18)	4.18 (2.09, 13.78)	**6.27 (2.78. 21.29)**	< 0.001

Unit: $\mu g/m^3$.

The largest values among three different environments were underlined with bold fonts.

[a]p-values were determined by the Kruskal Wallis non-parametric test to compare the median difference among the three groups.

Table 4

Median and interquartile ranges of noise frequencies among three different environments.

Frequency (Hz)	Background	Train	Truck	p-value[a]
31.5	25.7 (22.3, 30.1)	**36.7 (26.1, 40.9)**	32.0 (27.1, 41.7)	< 0.001
63	37.9 (33.9, 41.9)	**49.0 (42.9. 53.2)**	41.4 (36.6, 52.0)	< 0.001
125	44.0 (40.0, 47.5)	**53.0 (46.5. 57.8)**	47.6 (44.0, 58.0)	< 0.001
250	45.1 (42.1, 48.0)	**53.8 (46.7. 59.0)**	48.2 (45.1, 59.3)	< 0.001
500	45.9 (42.1, 49.0)	**52.4 (46.4. 57.8)**	48.8 (45.6, 58.0)	< 0.001
1000	48.1 (43.6, 50.9)	**50.5 (46.0. 57.2)**	50.0 (46.3, 55.0)	< 0.001
2000	42.5 (38.2, 46.7)	**48.5 (41.5. 54.5)**	46.9 (42.1, 50.3)	< 0.001
4000	36.6 (33.5, 41.2)	41.6 (36.5, 49.5)	**44.0 (38.2. 46.5)**	< 0.001
8000	28.4 (25.6, 32.1)	**33.5 (28.0. 39.5)**	32.0 (28.1, 34.4)	< 0.001

Unit: decibel (dB).

The largest values among three different environments were underlined with bold fonts.

[a]p-values were determined by the Kruskal Wallis non-parametric test to compare the median difference among the three groups.