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A community noise survey in Southwest Detroit and the value of supplemental metrics for truck noise

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ABSTRACT

Noise exposure can affect sleep, health and cognitive performance, and it disproportionately affects communities of color. This study has the objective of evaluating both conventional and supplemental noise metrics in a community noise survey examining Southwest Detroit, Michigan, a densely populated and industrialized area with extensive truck traffic on residential streets. Sound pressure level (SPL) monitors were deployed at 21 residential sites within 900 m of a major interstate highway. With assistance from youth volunteers, continuous SPL measurements were obtained for 1.5–7 days at each site, and short-term vehicle counts on local roads were recorded. We calculated conventional noise metrics, including the day-evening-night average sound level L_{DEN} and the 90th percentile 1-hr maximum $L_{10}(h)$, and evaluated the effect of distance from highways, traffic volume, time-of-day, and other factors. Supplemental metrics potentially appropriate for intermittent traffic noise were calculated, including fraction of time over specific SPL thresholds using a new metric called F_{DEN}, which is the fraction of time over 60, 65 and 70 dB during night, evening and daytime periods, respectively, and a peak noise metric called $L_{2P}(h)$, which utilizes the 98th percentile SPL within time blocks to increase robustness. The conventional metrics indicated five sites that exceeded 70 dB, and the highest noise levels were found within ~50 m of truck routes, arterials and freeway ramps. The estimated impact of truck traffic ranged up to 17 dB for hourly averages and to 33 dB for 1-s peaks. The conventional metrics did not always capture short-term noise exposures, which may be especially important to annoyance and sleep issues. In addition to showing widespread exposure to traffic noise in the study community that warrants consideration of noise abatement strategies, the study demonstrates the benefits of supplemental noise metrics and community engagement in noise assessment.

1. Introduction

1.1. Background

Exposure to noise is a public health concern due to its potentially deleterious effects on sleep, cardiovascular and psychosocial health, and cognitive performance (Halperin, 2014; Héroux et al., 2020; World Health Organization, 2009). Chronic noise exposure disproportionately affects communities of color (Casey et al., 2017), thus preventing and reducing noise exposure are important goals for environmental health justice. In urban areas, road traffic noise is the dominant source of chronic exposure (McAlexander et al., 2015; Paunović et al., 2009)· affecting an estimated 18 million people in the U.S. (Basner et al., 2014; Corbisier, 2003)· Motor vehicle noise is the most common noise source identified by residents classifying their neighborhood as noisy (U.S. Environmental Protection Agency, 1974). Noise from construction, rail, industry, aircraft, neighbors and other sources also contribute to community noise levels and can induce noise annoyance, speech interference, and sleep disruption (Onchang and Hawker, 2018). An estimated 104 million Americans experience continuous noise exposure levels exceeding 70 dB and are at risk for hearing loss and other noise-related health effects (Basner et al., 2014; Hammer et al., 2014). Unlike Europe where comprehensive national-level studies have mapped community noise levels (Bluhm and Eriksson, 2011; Niemann et al., 2006), noise levels in U.S. cities and most other countries have not been well

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Abbreviations

$F_{65}(m)$	fraction of minute over 65 dB
$F_{65}(h)$	fraction of hour over 65 dB
$F_{65}(d)$	fraction of day over 65 dB
F_{DN}	fraction of day over 60 and 70 dB for night and daytime periods, respectively
F_{DEN}	fraction of day over 60, 65 and 70 dB for night, evening and daytime periods, respectively
$L_{ave}(h)$	arithmetic average of sound pressure level for the hour (dB)
L_{DN}	day-night sound level
L_{DEN}	day-evening-night sound level (Community Noise Equivalent Level)
$L_{eq}(h)$	equivalent continuous sound pressure intensity for the hour (dB)
L_{NIGHT}	arithmetic average sound pressure level for nighttime hours
$L_{10}(m)$	noise level exceeded 10% of the time of the measurement duration, using 1-min arithmetic averages and the A-weighting
$L_{10}(h)$	noise level exceeded 10% of the time of the measurement duration, using 1-hr arithmetic averages and the A-weighting
$L_{P5}(m)$	noise level exceeded 5% of the time in a 1-min period, using 1-s measurements, i.e., 3rd highest 1-s SPL average in the minute
$L_{P5}(h)$	$L_{p5}(m)$ value exceeded in 5% of the minutes in a 1-hr period, i.e., 3rd highest $L_{P5}(m)$ over the hour
$L_{P5}(d)$	$L_{p5}(h)$ value exceeded in 5% of the hours in a 1-day period, highest $L_{P5}(h)$ over the day
Z	perceived psychoacoustic loudness or volume

characterized (Hammer et al., 2014). Such assessments are needed to understand exposure and guide mitigation actions to reduce individual and community exposure, e.g., noise walls, buffers, alternative traffic routes and truck routing, limits on noise sources and vibration levels, and sound proofing (Committee on Technology for a Quieter America, 2010; Waitz et al., 2007).

Noise indicators or metrics that characterize noise exposure can be tailored for specific or general applications, including determining impacts and setting community limits. In the U.S., federal noise legislation was established in 1972 and 1978 (e.g., Noise Control Act and Quiet Communities Act), but funding for enforcement was terminated in 1982, and state and local governments effectively are responsible for regulating community noise (U.S. Environmental Protection Agency, 2017). In 1974, EPA set a 55 dB level stated to be requisite to protect public health and welfare in residential and other outdoor areas (U.S. Environmental Protection Agency, 1974) using the Day-Night Sound Level (DNL or L_{DN}), defined as the 24-h equivalent continuous sound pressure level (SPL) obtained after adding a 10 dB penalty for the night time period (e.g., 10 p.m. to 7 a.m. local time). For highway projects, the U.S. Federal Highway Administration (FHWA) uses noise abatement criteria (NAC) that depend on the land use. The recommended metric is $L_{10}(h)$, the 90th percentile sound pressure level, using the 1-hr equivalent SPL and the A-weighting, and the recommended outdoor criteria are 60 dB for areas where "serenity and quiet are of extraordinary significance," 70 dB for picnic areas, recreation areas, playgrounds, active sports areas, parks, residences, motels, hotels, schools, churches, libraries and hospitals, and 75 dB for other developed lands (Federal Highway Administration, 2017a). If projected noise levels exceed 65 dB, FHWA recommends the use of sound walls, which can reduce noise by 5–10 dB (Federal Highway Administration, 2017a). U.S. states typically follow the FHWA criteria, e.g., the Michigan Department of Transportation (MDOT) recommends an $L_{10}(h)$ limit of 70 dB for residential noise (Jaeckel and Mazur, 2019; Michigan Department of Transportation, 2011). In practice, such criteria are used to assess the need for mitigation measures (e.g., sound walls), and such measures may be recommended if 65–70 dB is exceeded, a sufficient number of people are affected, and mitigation can make a "substantial" impact, e.g., 5–10 dB change (Federal Highway Administration, 2017a). In addition, FHWA regulations limit individual truck noise to 80 dB measured 50 ft from the centerline of travel for newly manufactured trucks with gross vehicle weight ratings (GVWR) exceeding 10,000 lbs (Federal Highway Administration, 2017b). For aircraft noise, the U.S. Federal Aviation Administration (FAA) uses the L_{DN} metric to screen aircraft routes and determine whether noise in affected communities increases by ~1 dB, in which case more detailed analysis or actions may be undertaken with targets of limiting L_{DN} increases to 1.5, 3 or 5 dB in areas where L_{DN} exceeds 65, 60 or 45 dB, respectively (Federal Aviation Administration, 2015; Guski et al., 2017).

For general applications, e.g., reporting, mapping and planning, the European Union, California and others use the day-evening-night sound level (L_{DEN}), also called the Community Noise Equivalent Level (CNEL), which adds an additional 5 dB penalty for evening hours (e.g., 7 p.m.–10 p.m.) to L_{DN} (the 10 dB penalty for nighttime remains) (Van den Berg et al., 2000). Further recommendations for this metric include the use of long term average A-weighted incident sound pressure levels, determinations at 4 m above the ground, use of the most exposed façade, exclusion of reflections from buildings, and inclusion of all sources, ground conditions and screens (most relevant for modeling.) The World Health Organization uses L_{DEN} in stating that road traffic noise levels exceeding 53 dB are associated with negative health outcomes (Guski et al., 2017; Héroux et al., 2020; Organization, 1999), and recommends a night noise guideline of 40 dB and interim target of 55 dB using the average equivalent SPL for nighttime hours (L_{NIGHT}) to judge adverse effects on sleep (World Health Organization, 2009). Noise outside dwellings often is considered the most sensitive indicator; other important locations include schools, hospitals, and indoor locations (Grelat et al., 2016).

Measures like L_{NIGHT}, L_{DN} and L_{DEN} reflect the increased sensitivity to noise at night and during (normally) quiet periods. The annoyance and sleep impairment due to noise can be driven by noise characteristics that are averaged out or not reflected in the SPL average, e.g., the tonal or impulsive content, low frequency noise, sound pressure level fluctuations, and prominent noise events (National Academies of Sciences Engineering and Medicine, 2014). Thus, a number of supplemental metrics have been proposed, including the time above 65 dB or other threshold, number of events above 70 dB, the "intermittency ratio" that reflects changes in SPLs, and community noise tolerance levels (Brambilla et al., 2019; International Institute Of Noise Control Engineering, 2015; Miedema and Vos, 1998; Onchang and Hawker, 2018; Van den Berg et al., 2000; Wunderli et al., 2016).

1.2. Study objectives

This study applied and evaluated several approaches and metrics to characterize community noise levels. We conducted a community survey using with youth volunteers to monitor sound levels using cell phones and record short-term vehicle counts at residential sites, and deployed SPL instrumentation at these and other sites for up to 1-week long periods. In addition to using conventional noise metrics, we introduce several new metrics that may be appropriate for intermittent noise, including truck noise common in the study area. Effects associated with measurement location, obstructions, distance from local and major roads, time-of-day and day-of-week are investigated. In addition to evaluating several supplemental metrics, the study was intended to inform decisions regarding future monitoring and noise mitigation actions associated with potential increases in truck traffic.

2. Methods

2.1. Study area and site selection

The study was conducted in Southwest Detroit, Michigan, a densely populated and industrialized area that has many truck routes that pass through largely residential neighborhoods. Truck traffic has been a long standing environmental and safety concern in this community (Sampson et al., 2020). An additional motivation for selecting this area is the construction of a major new U.S. to Canada crossing, the Gordie Howe International Bridge (GHIB), which may increase local truck traffic when completed in 2024. This new bridge is being constructed ~2 km southwest of the existing Ambassador Bridge, crossed daily by 7000–9000 heavy duty vehicles and ~12,000 light duty vehicles (Bridge and Tunnel Operators Association (BTOA), 2020). The 24.4 km^2 study area is bordered to the southeast by the Detroit River and to the northwest by highways I-94 and MI-12 (Fig. 1). The area's population of ~60,000 (Data Driven Detroit, 2013; Google Maps, 2020) is mostly racial and ethnic minorities (57.2% Hispanic and 23.5% African American), and most (77.3%) households have an income below $50, 000 (Data Driven Detroit, 2013). Most homes are one- or two-family structures with small front and back yards. Residential areas are interspersed among areas of heavy industry (e.g., refinery, steel and coke facilities, auto assembly plants), surface arterial routes and highways, the border crossing, and many logistical, port and intermodal facilities. A major highway, I-75, traverses the area; near the bridge, this 6-10 lane freeway has an average annual daily traffic (AADT) volume of 110,000, including 15,000 commercial vehicles (Michigan Department of Transportation, 2018). I-75 is elevated in the SW and NE portions of the study area, and below grade in the center of the area. The major airport in the area is ~28 km from the study area; aircraft noise is rarely apparent in the study area.

Homes for SPL measurements were recruited by our community partner, a local community organization, via community meetings, flyers and word-of-mouth in both English and Spanish. Measurement sites fell into two groups, called S and G sites. Selection of the S sites, used for short term (~1.5 days) monitoring of traffic counts and SPLs by community volunteers, was based on the proximity to I-75 and the desire to both include and exclude local truck routes. Selection of the G sites, used for longer term (~5 days) monitoring, considered proximity to I-75, site security, and the ability to obtain electrical power. All sites were within 900 m of I-75. S007 and G115 used the same residence, however, instrumentation was placed on the elevated porch at S007 and in the backyard at G115. For Spanish speaking households, a professional translator accompanied our technicians. Recruitment and all other study aspects complied with the University of Michigan Institutional Review Board requirements.

2.2. SPL measurements and vehicle counts

SPL measurements and vehicle counts at the S sites were obtained in a community science initiative that engaged 14 high school youth volunteers and 7 adult supervisors. Youth were trained by our community partner prior to the field study. At each site, an iPhone using the app DecibelX (SkyPaw, Hanoi, Viet Nam) obtained continuous 0.2-s data, and a SPL meter (REED SD-4023, Newmarket, Ontario, Canada) obtained and logged continuous 1-s data. The instruments used external microphones and were calibrated to 114 dB before and after sampling. Phones and sound meters were mounted on secured tripods at ~1.5 m height placed on the front porch or in the front yard, away from walls, furniture or foliage. Foam boards were placed behind the phone and against the house wall to reduce reflections. The volunteers, who also

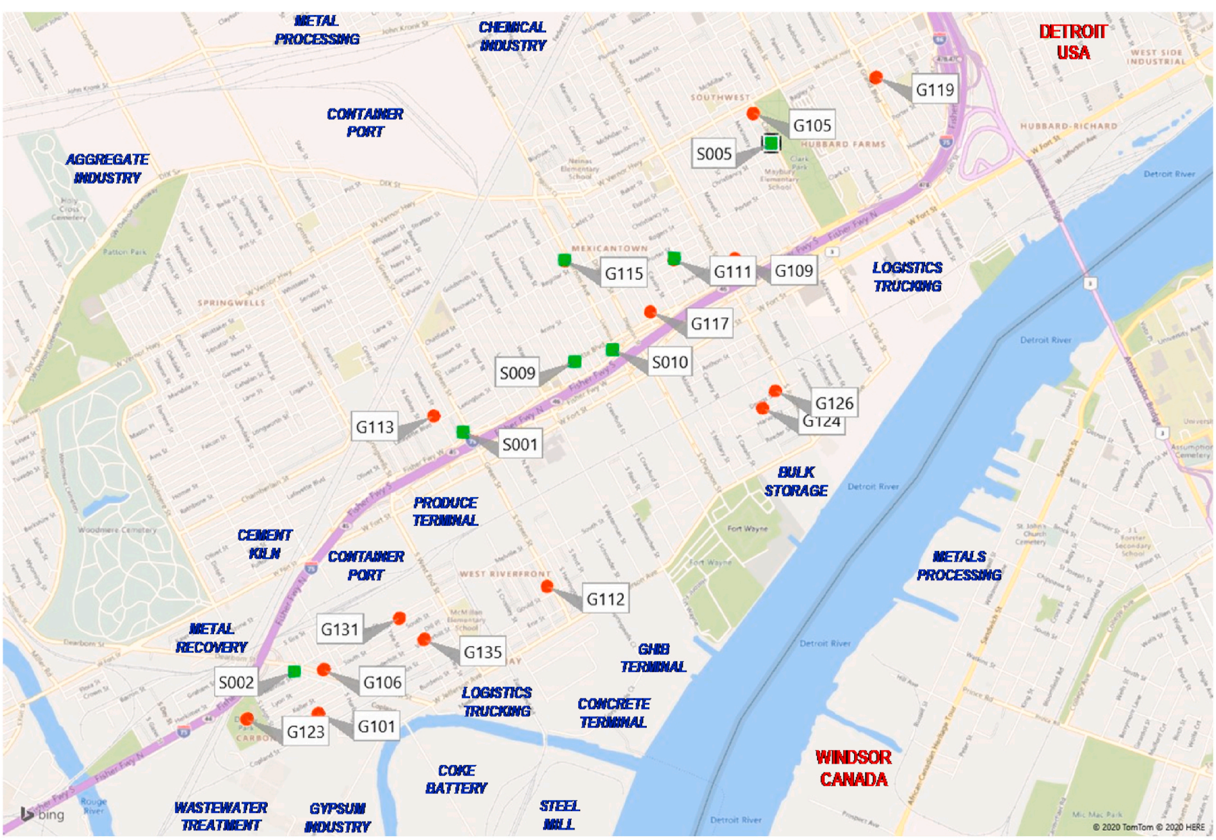

Fig. 1. Site map indicating locations of monitoring sites with S sites are in red and G sites in green. S003 is also G111, and S07 is G115. Also shows location of larger industries. GHIB = construction area for new bridge. Map spans 5.4 × 4.5 km region. Map from Bing (2020). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

assisted with vehicle counts (see below), sat quietly on the opposite side of the porch or elsewhere in the yard. Unfortunately, a technical issue in the phone app was encountered that limited the data collected, which ultimately was not used. (The app on many of the youths' iPhones did not enable the "Prevent Application Sleep" setting, which would allow recording while the phone was locked or "asleep", thus data collection ended prematurely when the phone automatically locked.) This also precluded our evaluation of the congruence of the app to the SPL monitors.

At each S site, two volunteers monitored vehicle counts (one per lane) on the local road simultaneously with sound monitoring. Each vehicle passing the site was tallied on a form that denoted vehicle type (e.g., car, motorcycle, truck) and time. The iPhone and traffic monitoring was conducted during 6-hr daytime periods, mostly between 11:00 and 17:00, on two weekdays (August 7–8, 2019). Traffic counts were performed in three 2-hr consecutive shifts each day. The SPL meter was used continuously throughout this period, obtaining measurements for the two daytime periods as well as the evening and night. Data were downloaded and saved in the evening; other information (e.g., operator name, start/stop times, a hand drawn map) were recorded on a standard form. The duration of sampling at the S sites was 30 ± 2 h (average ± standard deviation).

At the 15 G sites, the same SPL meter type was deployed and retrieved by our professional technicians. The instrument was installed in a weatherproof case and the microphone was placed in an open and weather protected shelter. Auxiliary power was provided to ensure continuous operation. In most cases, the monitor was placed in the backyard due to security concerns. Data was downloaded after each sampling event. Data collection occurred for ~5-day periods in two seasons: season 1 was June 27 to September 13, 2019, and season 2 was October 4 to December 13, 2019. 10 sites were monitored in season 1 and 9 in season 2, and 3 sites were monitored in both seasons (G101, G112, G119). The sampling duration at the G sites was 153 ± 56 h (including the sites monitored in two seasons). (Table S1 provides the sampling period at each site.).

2.3. Truck, highway, and site information

A unified dataset was constructed containing information for each site using photographs, a local inventory, Google Maps, and other data sources. The dataset contained site details (e.g., instrument location, latitude, longitude), obstructions (e.g., buildings, trees), neighborhood characteristics (e.g., residential, commercial), distances and directions to the nearest local road and I-75, AADT and commercial AADT (CADT) for the portion of I-75 nearest each site (Michigan Department of Transportation, 2018), and potential noise sources within 500 m of each site (e.g., wastewater facilities, gas stations, factories) identified visually during site visits and using Google Maps, the Toxic Release Inventory (TRI), and the State of Michigan Air Emissions Reporting System. Table S2 presents selected site details, including distance to roads and traffic volumes.

2.4. Noise metrics and data analyses

Using the 1-s sound intensity data, we calculated the average energetic level for each minute that contained at least 45 1-s measurements, and 1-hr averages for hours that contained at least 45 1-min measurements, denoted as $L_{eq}(m)$ and $L_{eq}(h)$, respectively. There was little missing data. We also computed arithmetic averages of the SPL, $L_{ave}(m)$ and $L_{ave}(h)$, which were used in only selected comparisons. Descriptive statistics, including tests of normality and lognormality (Shapiro-Wilk and D'Agosino's K-squared tests) for the entire dataset and by site showed departures from normality at most sites ($p < 0.05$) for the 1-min and 1-h data. We calculated $L_{10}(h)$, the 90th percentile 1-h value over the day, as well as L_{DN}, L_{DEN}, and daily average $L_{eq}(d)$.

Using Grubbs's test, 88 potential outliers were detected among $L_{eq}(h)$. At several sites, $L_{eq}(d)$ was substantially elevated over $L_{ave}(d)$, also suggesting the influence of outliers. Typically, the highest 1-s measurements occurred as 1–3 s long peaks with SPLs from 100 to 126 dB; these peaks occurred mostly at daytime and at a subset of sites. The effect of omitting 1-s data that exceeded thresholds from 95 to 120 dB was evaluated. We selected a threshold of 105 dB as relatively few measurements (134 1-s observations, 0.0015% of the total) were excluded, but $L_{eq}(d)$ and $L_{10}(h)$ were substantially lowered at several sites, most notably S005 and S007 where these metrics decreased by 9.1–11.1 dB. While half of the sites had no change (no omitted data), this threshold eliminated 9 outliers and $L_{eq}(d)$ and $L_{10}(h)$ were lowered by an average of 1.5 and 0.9 dB across the sites, respectively. While results at several sites may be sensitive to outliers, the 105 dB threshold represents a balance between censoring data that is not known to be invalid and the SPLs that may be encountered in community settings, e.g., truck or motorcycle exhaust, and horns. Any substantial changes caused by omitting data exceeding 105 dB are noted in the text. Future studies might employ video or audio recording to help determine causes of high measurements and whether they are valid measurements.

Using the censored data, we recalculated all statistics. The 1-hr data was evaluated by hour, day (7:00 to 18:59), evening (19:00 to 22:59), and night (23:00 to 6:59) periods, and by weekday and weekend periods. We examined the range, skewness and kurtosis of the distribution of 1-min and 1-hr averages. $L_{eq}(d)$ and $L_{10}(h)$ metrics were used to assess potential associations with the distance to nearby roads and highways, highway grade, traffic volume (AADT and CADT), type of neighborhood (i.e., residential or mixed), instrument location, nearby noise sources, degree of sheltering, and time of day using scatterplots, linear and nonlinear regressions, correlation coefficients, and parametric and non-parametric tests, e.g., t-tests and Mann-Whitney (M-W) tests. Given the non-normality of most of the data, we report the non-parametric results. Then, linear and nonlinear regressions were used to identify factors potentially affecting noise levels, with interaction terms for traffic volume and (inverse) proximity, and distance to road and sheltering of that road. Variable selection in the final models were selected using step-wise regression and weekday data.

Changes in the perceived psychoacoustic loudness or volume Z were calculated as:

$$Z = 2^{\Delta L / L_{ref}} \tag{1}$$

where ΔL = change in $L_{eq}(h)$ or other SPL metric (dB), and L_{ref} ranged from 6 to 10 dB, the SPL change associated with a doubling of the perceived loudness (Warren, 1973).

Several metrics were developed to account for noise in community settings, including intermittent traffic noise that may not greatly alter hourly or daily averages. First, we calculated intermittency ratios (IR) using the 1-s data and thresholds C of 3, 5 and 10 dB; results using C = 5 dB provided a good range of values and are described in the text. Next, metrics quantifying the fraction of time over SPL thresholds were developed using 1-s data and thresholds that matched recommendations for community noise limits, which mostly range from 53 to 70 dB, as discussed earlier. We calculated as the fraction of each minute over 50, 60, 65 and 70 dB, called $F_{50}(m)$, $F_{60}(m)$, $F_{65}(m)$ and $F_{70}(m)$, respectively, and then for complete hours (containing at least 45 1-min measurements), 1-hr averages, e.g., $F_{70}(h)$, and then daily averages from hourly averages, weighted to give each hour the same influence, e.g., $F_{70}(d)$. The metrics have a straightforward interpretation, e.g., $F_{70}(d)$ is the fraction of the day exceeding 70 dB. To account for increased sensitivity during evening and nighttime periods, we incorporated 5 and 10 dB penalties (as in L_{DN} and L_{DEN} metrics) in daily measures called F_{DN} and F_{DEN}, which were calculated as a weighted sum of $F_{60}(h)$, $F_{65}(h)$ and $F_{70}(h)$ metrics for night, evening and daytime periods. F_{DEN} (dimensionless) was calculated as:

$$F_{DEN} = 9/24\, \Sigma_{night}\, F_{60}(night) + 3/24\, \Sigma_{eve}\, F_{65}(eve) + 12/24\, \Sigma_{day}\, F_{70}(day) \tag{2}$$

where night = 9 h from 10 p.m. to 7 a.m., eve = 3 h from 7 to 10 p.m., and day = 12 h from 7 a.m. to 6 p.m. F_{DN} was similarly calculated using F_{60}(night) and expanding F_{70}(day) to 15 h (7 a.m.–10 p.m.).

The third type of metric was designed to quantify short duration noise. Using the 1-s measurements, we calculated 90th, 98th, 99th percentile and maximum SPL for each minute, called L_{10}(m), L_2(m), L_1(m) and L_0(m), respectively. Then, two types of 1-hr and 24-hr measures were derived. These included hourly and daily averages, e.g., L_{10}(h) is the hourly average of the 1-min 90th percentile SPL, and L_{10}(d) is the daily average of the hourly averages. These metrics represent the average 90th percentile envelope of the 1-min and 1-hr data, respectively. This formulation "averages out" infrequent and short peaks and thus represents an upper confidence bound for the average SPL; consequently, this metric closely tracks the average SPL and obscures peak information. We emphasize an alternative metric using upper percentile measures calculated for each hour and percentile, e.g., the 90th percentile of the 1-min data L_{10}(m), called L_{P10}(h), and the 90th percentile of the 1-hr L_{10}(h) data, called L_{p10}(d). Similar calculations were performed for the 98th, 99th and 100th (maximum) percentiles. This approach provides "blocking" that accounts for peaks occurring over 1-min and 1-hr periods, and the peak percentile metrics may be considered as that percentile of the 1-s SPL envelope calculated by minute, hour or day. For example, consider a 60-s duration high noise event that exceeds other SPL measurements over that day. This would be reflected in the top 1.67% of the 3600 measurements collected for the hour of the noise event, and the top 0.028% of the 86,400 1-s measurements collected over the day. Thus, this event would be reflected in the 99th percentile 1-s measurement for that hour and the 99.93rd percentile 1-s measurement for the day – but not in lower percentile measures. In contrast, the F_{P1}(h) and F_{P1}(d) metrics would reflect this noise event. These peak measures are robust since obtaining a high value requires multiple peaks over the period considered, except for L_{P0}(h) and L_{P0}(d), which reflect the single highest 1-s measurement over the hour and day, respectively.

Six sites with a range of noise exposure were selected to illustrate trends of the metrics, using a single but complete 24-hr weekday period. Then all sites are examined and contrasted with L_{eq}(d) and other metrics.

3. Results

3.1. Site descriptions

Sites locations are mapped in Fig. 1 and site characteristics are summarized in Table 1. The 21 unique sites spanned a 5.4 km long corridor along I-75 that was ~2 km wide. Most sites (N = 16, 76%) were in residential areas; the others (N = 5, 24%) were in mixed residential/commercial/industrial areas. All roads immediately adjacent to the residences had two lanes (either one- or two-way) with parking on both sides. Monitors were set up in the front yard or front porch facing the local road at 7 S sites and 5 15 G sites; at the remaining 10 G sites, monitors were in the side or back yard. Instrumentation was placed 23 ± 8 m (average ± standard deviation) from the center of the local road (range: 20–25 m for S sites; 9–35 m for G sites). Few of the street-facing sites had significant obstructions or vegetative screening between the monitor and the local road. Photos and local maps for six sites are in the supplemental materials (Figs. S1-6).

Sites averaged 376 ± 249 m from I-75 (range: 64–608 m for S sites; 93–811 m for G sites). AADT on I-75 nearest each site averaged 101,734 ± 5188 vehicles/day; CADT was 12,947 ± 505 vehicles/day. Two sites (S001, S010) were particularly close to I-75 (64–65 m) and nearly adjacent to its service drive, which has both car and truck traffic. While our truck counts did not fully distinguish truck size or weight, most trucks were heavy duty diesel vehicles (e.g., 5 axle tractor trailers). Most sites (15 of 21) had buildings between the monitor and I-75 (Table 1). Additional noise sources within 500 m were identified at 10 of 21 sites, e.g., gas stations (G105, S002), wastewater treatment plant and other industry (G101, G124, G126), rail lines (G106, G115, G123, G126, G131), and large trucking operations (G131).

3.2. Sound pressure levels

3.2.1. Summary across sites

Weekday SPLs and other noise metrics are displayed in Fig. 2 with sites ranked by L_{eq}(d). Noise metrics (including stratification by weekday and weekend periods) are summarized in Table 2. Across the 21 sites, weekday L_{eq}(d) averaged 62.1 dB (range: 55.3–73.8 dB), L_{10}(h) averaged 64.3 dB (57.7–76.7 dB), L_{DN} averaged 66.8 dB (60.4–76.8 dB), and L_{DEN} averaged 67.1 dB (60.7–77.1 dB). L_{DN} and L_{DEN} metrics exceeded the 70 dB guideline at S007, S002, S010, S001, S005 and G123, and G106 and G113 were within 2 dB of this criterion. No site met

Table 1
Site features and characteristics separated by site type (S and G sites).

Feature/Road	Category	Unit	All Sites		G Sites	S Sites
			Count/Ave	Std.Dev.	Count/Ave.	Count/Ave.
Neighborhood	Residential	(count)	17	–	12	5
	Commercial	(count)	4	–	3	1
Monitor Site	Front or side yard	(count)	11	–	5	6
Location	Back yard	(count)	10	–	10	0
Monitoring duration	Average	(hours)	118	74	153	30
Nearby Sources	None	(count)	6	6	5	1
Potential	Low	(count)	5	5	4	1
	Medium	(count)	9	9	6	3
	High	(count)	1	1	0	1
Local Road	No obstructions	(count)	12	–	8	4
	Vegetation obstructions	(count)	7	–	5	2
	Building Option	(count)	2	–	2	0
	Average Distance	(m)	23.4	8.3	23.9	22.2
	Average AADT	(vehicles/day)	3420	1795	–	3571
	Average CADT	(vehicles/day)	809	633	–	809
I75	No obstructions	(count)	3	–	2	1
	Vegetation obstructions	(count)	4	–	1	3
	Building Option	(count)	14	–	12	2
	Distance	(m)	382	254	427	271
	AADT	(vehicles/day)	100,519	5215	98,829	104,746
	CADT	(vehicles/day)	12,922	503	12,755	13,341

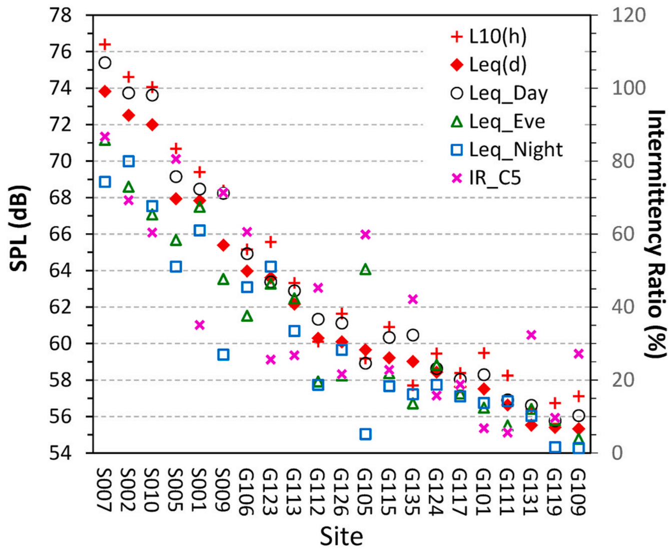

Fig. 2. Selected noise metrics at the 21 sites, ranked by L_{eq}(d). Shows weekday L_{eq} for 24-hr average, day (7 a.m.–6 p.m.), evening (7 p.m.–10 p.m.) and night periods (11 p.m.–6 a.m.), L_{10}(h), and intermittency ratio for 5 dB threshold (IR_C5).

the 53 and 55 dB targets suggested by WHO and EPA, respectively (although 4 sites were close, below 57 dB), and only two sites met WHO's 55 dB target for L_{NIGHT}(h). The WHO target values apply to the most exposed façade, and since 10 of 21 sites had monitors in shielded back or side yards, these statistics underestimate the true exposure. (As noted later in Section 3.3, backyard placements lowered L_{eq}(d) and L_{10}(h) by ~7 dB) Intermittency ratios (IRs) had moderate correlation with L_{eq}(d) (R^2 = 0.61) with several sites departing from the overall trend, e.g., sites S001 and G105.

Site locations and conditions at the three "quietest" and three "noisiest" sites are detailed in the supplemental materials (Fig. S2-S6). In brief, the noisy sites (S007, S002, S001) had weekday L_{eq}(d) from 72.0 to 73.8 dB and L_{10}(h) from 74.6 to 76.7 dB. These sites were in residential neighborhoods, near or along a busy surface road with occasional or frequent truck traffic, with few obstructions or vegetative screening near the street-facing monitors. These sites can be characterized as highly traffic impacted locations. At the three quiet sites (G109, G119, G131), L_{eq}(d) ranged from 55.3 to 55.5 dB. While one of these sites was relatively close (92 m) to I-75 (below grade at this location), these sites were at least 1.5 blocks from a major road and monitors were in backyards with only indirect exposure to road noise. Thus, these sites represent highly shielded locations. Other sites fell between the extremes of the traffic-impacted and shielded sites. Overall, the 21 sites represent a wide range of conditions, e.g., sites were at various distances from arterials and highways, several had nearby noise sources (e.g., highway ramps, railways, construction, gas stations, trucking depots), and the degree of shielding varied considerably.

3.2.2. *Diurnal patterns*

On weekdays, the highest SPLs occurred mostly from 7 to 10 a.m. and 2 a.m. to 5 p.m. (Fig. 3). The diurnal variation, ΔL_{eq}(h), defined as the difference between the noisiest and quietest hours of the day at a site, varied by site and site type. The median ΔL_{eq}(h) was 9.1 dB for the 11 road-facing sites and 2.7 dB for the 10 side/backyard sites (N = 10). At the three noisiest sites (S007, S002, S001), L_{eq}(h) exceeded 70 dB from 7 a.m. through 5 p.m., levels were fairly consistent throughout the day and unimodal, and ΔL_{eq}(h) exceeded 11 dB (Fig. S7A). ΔL_{eq}(h) was particularly large at sites S005, S007 and S009 (19.6–20.4 dB), reflecting a large (4–10 fold) change in perceived loudness. At these sites, the quietest hours were 2 to 3 a.m., and the noisiest hours tended to span daytime hours. In contrast, at the sheltered sites, L_{eq}(h) was mostly below 60 dB, and the afternoon peak lasted through the evening (Fig. 3C). At the three quietest sites, noise levels were highest in the early evening (Fig. S7B), suggesting local sources, e.g., air conditioners, which is consistent with the hot and humid weather on the sampling days in July and August 2019. For the peak noise metrics, diurnal trends again depended on site, but most sites showed higher levels from 8 a.m.

Table 2
Summary of noise metrics for all sites (G and S sites), and for weekday and weekend periods at G sites. Intermittency ratios (IR) use C = 5 dB.

Measure	Period	Weekday All (N = 21)			Weekday G Sites (N = 15)			Weekend G Sites (N = 15)		
		Ave	Min	Max	Ave	Min	Max	Ave	Min	Max
SPL, averaged and 1-hr 90th percentile										
L_{eq}(h)	24-h	**62.1**	55.3	73.8	**59.0**	55.3	64.0	**57.6**	53.3	62.2
L_{eq}(h)	Day	**63.0**	55.8	75.4	**59.6**	55.8	64.9	**57.5**	52.9	62.3
L_{eq}(h)	Evening	**61.0**	54.8	71.2	**58.5**	54.8	64.1	**58.1**	53.7	64.9
L_{eq}(h)	Night	**60.2**	54.3	70.0	**57.9**	54.3	64.2	**57.3**	53.6	62.7
L_{10}(h)	–	**64.3**	57.7	76.7	**61.1**	57.7	68.6	**58.9**	54.5	64.5
L_{DN}	24-h	**66.8**	60.4	76.8	**64.1**	60.4	70.1	**63.8**	59.9	69.0
L_{DEN}	24-h	**67.1**	60.7	77.1	**64.4**	60.7	70.3	**64.1**	60.2	69.4
Intermittency Ratio										
IR(5)	24-h	**39.3**	5.6	86.7	**28.1**	5.6	60.6	**23.9**	1.6	54.0
IR(5)	Day	**42.7**	6.3	85.8	**32.6**	6.3	71.0	**22.7**	2.1	63.9
IR(5)	Evening	**31.1**	3.3	85.6	**18.9**	3.3	76.4	**15.1**	1.8	46.2
IR(5)	Night	**26.9**	1.9	89.4	**14.6**	1.9	45.2	**12.8**	0.4	31.3
Time fraction over SPL threshold										
F50(h)	24-h	**0.96**	0.71	1.00	**0.98**	0.87	1.00	**0.97**	0.89	1.00
F60(h)	24-h	**0.32**	0.02	0.98	**0.20**	0.02	0.72	**0.16**	0.00	0.57
F65(h)	24-h	**0.12**	0.00	0.64	**0.03**	0.00	0.14	**0.02**	0.00	0.05
F70(h)	24-h	**0.03**	0.00	0.20	**0.01**	0.00	0.03	**0.00**	0.00	0.02
Peak SPL, averaged										
Lp10(h)_ave	24-h	**60.9**	55.3	71.0	**58.5**	55.3	63.4	**57.8**	53.9	62.4
Lp2(h)_ave	24-h	**63.0**	56.2	75.4	**59.8**	56.2	65.2	**59.0**	55.0	64.5
Lp1(h)_ave	24-h	**63.3**	56.4	76.0	**60.0**	56.4	65.4	**59.2**	55.2	64.8
Lp0(h)_ave	24-h	**63.6**	56.5	76.4	**60.2**	56.5	65.7	**59.4**	55.3	65.0
Peak SPL, by percentile										
Lp10(h)_p10	–	**67.5**	60.7	80.7	**64.3**	60.7	69.1	**62.6**	58.2	68.5
Lp2(h)_p2	–	**80.3**	67.7	98.7	**75.0**	67.7	85.9	**73.6**	63.5	82.6
Lp1(h)_p1	–	**85.0**	70.4	102.1	**79.9**	70.4	94.5	**78.5**	65.3	87.0
Lp0(h)_p0	–	**93.6**	72.9	105.0	**89.3**	72.9	104.0	**88.8**	69.4	103.6

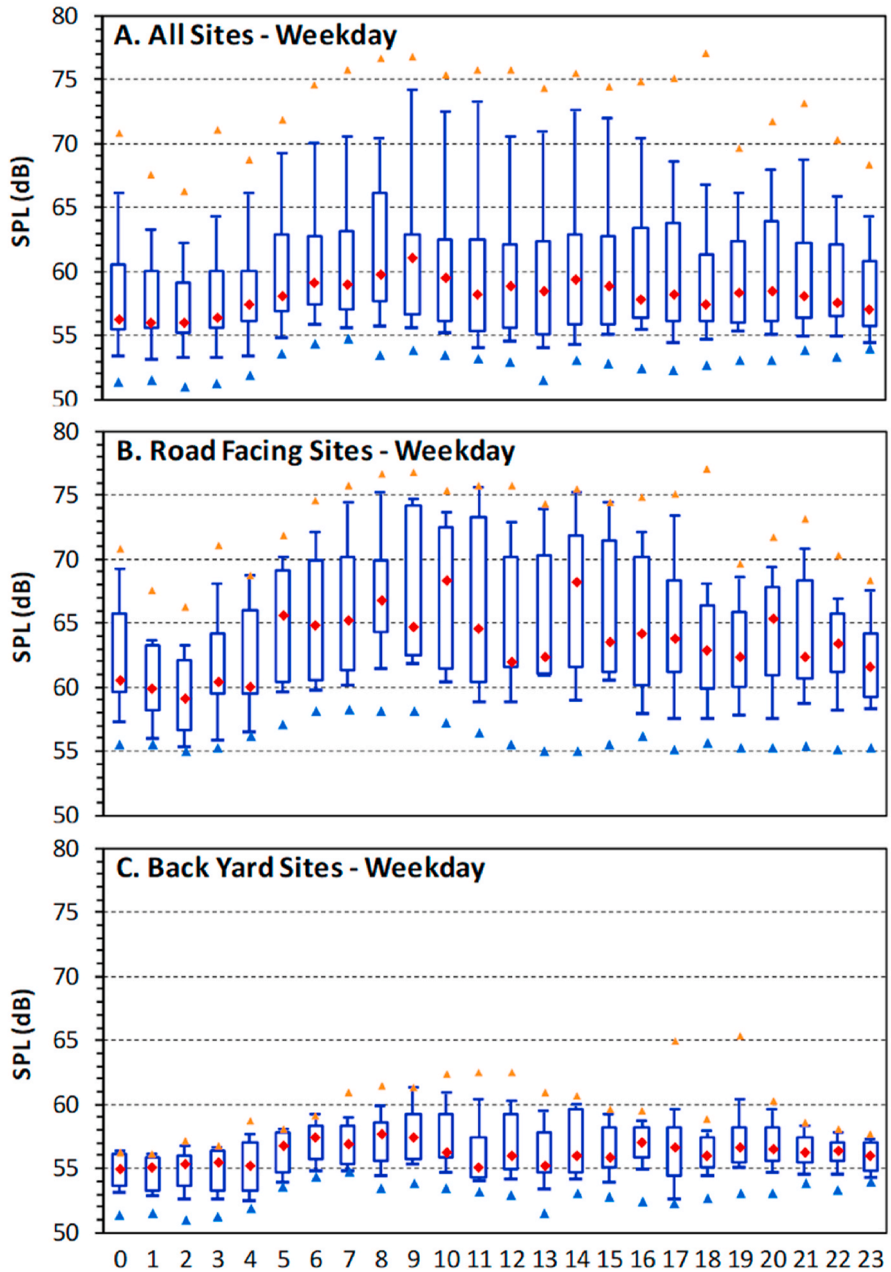

Fig. 3. SPLs by time of day for all sites (N = 21), road-facing sites (N = 11), and sites located in back or side yards (N = 10). Uses 1-hr SPL. Plots show maximum, 90th, 75th, median (as red diamond), 25th, 10th percentile and minimum 1-hr L_{eq} across the sites. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

to 4 or 5 p.m. (Fig. S8A), particularly at sites near truck routes (Fig. S8B).

The diurnal noise pattern at most sites was consistent with traffic volume patterns on local highways. On weekdays, non-commercial vehicles on Detroit freeways have a bi-modal pattern, with peaks from 6 to 9 am and from 4 to 6 pm, reflecting commuting periods; commercial vehicles (mostly trucks) have a unimodal pattern with a broad flat peak between 7 a.m.–4 p.m. (Batterman et al., 2015). The total traffic-related source intensity would reflect both non-commercial vehicles and the fewer but generally noisier commercial vehicles. Patterns shift on weekends: non-commercial vehicles have a (single) broad peak in the late morning on Saturday and Sunday, and very low volume on Sunday evenings; commercial vehicles have a single broad peak from ~7 a.m.–2 p.m. on Saturday, and low but consistent volumes from ~10 a.m. to 10 p.m. on Sunday (Batterman et al., 2015).

Weekday/weekend differences depended on site. L_{eq}(d) fell an average of 1.4 dB on weekends, but changes during evening and night periods were negligible (Table 2). However, weekday/weekend trends differed by site: ten sites decreased on weekends (average of 1.3 dB, up to 2.9 dB at site G105), while five sites increased (average of 1.1 dB, up to 2.1 dB at site G131). The fraction of time over threshold and peak SPL metrics showed mostly small decreases on weekends. (The uncensored data showed several of the loudest 1-s measurement on weekends, e.g., 125 dB at G117 and 116 dB at G113; comparable measurements (110–126 dB) occurred on weekdays at 7 sites.)

Diurnal patterns on weekends differed by day: Saturday appeared bimodal, with SPL peaks in early morning (5–8 am) and late afternoon to evening (3–10 p.m.), and variation from site-to-site was considerable (shown by large interquartile ranges on Fig. S9C). Sunday showed increases in the early morning, but levels were otherwise similar throughout the day, again with considerable variability among sites

(Fig. S9B). To an extent, this matches the local traffic patterns described above. The Saturday afternoon-evening increase may reflect occupant activities; however, a definitive analysis requires documentation of site activities and a larger number of sampled weekends.

3.3. Influences on SPLs

Several factors were significantly associated with noise levels (Table 3). Street-facing monitors (N = 11) had $L_{eq}(d)$ and $L_{10}(h)$ medians that exceeded those of backyard monitors (N = 10) by 8.6 and 9.1 dB, respectively ($P < 0.001$). The front porch and backyard sites (S007 and G115) at the same residence provide a dramatic illustration of building shielding: daytime $L_{eq}(d)$ and $L_{10}(h)$ were 14.6 and 15.6 dB higher at the front, street-facing site. However, this difference may not be representative since it is based on a single site and measurements were collected on different days. Effects due to screening and shielding using a qualitative estimate of coverage by vegetation and buildings showed more modest effects, lowering $L_{eq}(d)$ and $L_{10}(h)$ by 3.4 and 4.2 dB, respectively; only the latter difference was statistically significant ($P = 0.04$).

Higher noise levels were associated with the presence of a nearby truck route or highway within 30 m compared to larger distances, which increased $L_{eq}(d)$ and $L_{10}(h)$ by 5.7 and 6.9 dB, respectively (Table 3). Truck routes at longer distances were associated with noise increases, although changes were not statistically significant. Only weak associations ($R^2 \leq 0.04$) were found between $L_{eq}(d)$ with distance to either the freeway or the nearest road, Moderately strong associations ($0.35 \leq R^2 \leq 0.68$) were shown with AADT or CADT on I-75 or adjacent roads (Fig. S10). The change in grade of I-75 (below grade at 13 sites, at surface at 2 sites, and above grade at 6 sites) may affect results: sites within 100–325 m of elevated portions of I-75 had median levels of $L_{eq}(d)$ and $L_{10}(h)$ that were 5.7 and 9.0 dB higher, respectively, than levels at 6 sites with similar distances where the highway was below or at grade, however, these results were not statistically significant ($P = 0.14$, $P = 0.07$), possibly due to interactions by distance (sites where the highway was elevated often had greater distances) and the limited sample size. The effect of proximity to truck traffic on either I-75 or a nearby arterial road used as a truck route is shown on Fig. 4A and fitted to weekday $L_{eq}(d)$ using a reciprocal relationship with distance and an intercept to account for background levels ($R^2 = 0.45$). Fit improved using only street-facing sites ($R^2 = 0.68$; Fig. 4B). Despite the considerable scatter, these plots suggest that as the distance to a truck route decreased from 320 to 20 m, $L_{eq}(d)$ increased from ~58 to ~74 dB and $L_{10}(h)$ increased from 59 to 76 dB.

Significant difference in noise metrics were not found between residential and mixed neighborhoods, but this classification may not be very meaningful since some residences in residential areas were near major surface roads and highways, and several bordered commercial and industrial areas. The road type (residential or arterial) adjacent to the monitoring site, and the nearest larger road within 500 m, made only small and statistically insignificant differences, as did proximity to nearby potential noise sources, e.g., factories and gas stations within 500 m of monitoring sites. Our analysis is limited since sources were visually identified, sound intensity of potential sources was not measured, and few such sources were identified near monitoring sites.

After testing various linear and non-linear models to account for multiple influences on $L_{eq}(d)$ and $L_{10}(h)$, we found two simple and similar models that explained most of the variation:

$$L_{eq}(d) = 6.27\ \text{StreetFacing} + 144.2\ \text{Distance}^{-1} + 55.5\ (R^2 = 0.71) \tag{3}$$

$$L_{10}(h) = 7.04\ \text{StreetFacing} + 155.5\ \text{Distance}^{-1} + 57.1\ (R^2 = 0.76) \tag{4}$$

where StreetFacing = indicator variable for monitoring site (0 = back or side yard; 1 = street facing) and Distance = distance to nearest truck route (m). Coefficients of the models (all significant at $P < 0.003$) suggest a 6–7 dB increase for street facing locations and a 7–8 dB increase near a truck route (Distance coefficient divided by 20 m, the shortest distance in the study). Model fit could be increased to $R^2 \approx 0.80$ by including variables for traffic volume, shielding, highway grade, freeway proximity, considering only daytime hours and adjusting the reciprocal relationship (e.g., subtracting 15 m from the Distance term), however, these additions did not attain statistical significance. Improved estimates of traffic volume and other information might better explain

Table 3
Evaluation of potential influences on $L_{ave}(d)$ and $L_{10}(h)$ at study sites. Shows mean, standard deviation (SD), median, interquartile range (IQR) and sample site (N) for two groups. P-values for t tests based on 2-sided tests and unequal variances.

	Group1						Group 2						P-values	
Comparison	Type	Mean	SD	Median	IQR	N	Type	Mean	SD	Median	IQR	N	*t*-test	MW test
Monitor location														
Leq(d)	Street-facing	65.6	5.3	65.6	7.1	11	Backyard	57.0	1.4	57.5	2.0	10	0.000	**0.000**
L10(h)	Street-facing	68.7	5.6	68.6	7.6	11	Backyard	59.5	1.6	59.2	3.0	10	0.000	**0.000**
Neighborhood														
Leq(d)	Residential	61.2	5.9	59.1	8.5	17	Mixed	62.6	6.6	60.5	7.6	4	0.724	0.574
L10(h)	Residential	64.0	6.2	62.1	9.7	17	Mixed	65.6	7.1	63.9	8.5	4	0.694	0.698
Screening														
Leq(d)	Little	62.9	5.4	62.3	9.7	12	Medium-High	59.5	6.2	57.5	3.8	9	0.207	0.095
L10(h)	Little	66.1	5.7	65.4	8.0	12	Medium-High	61.9	6.4	58.8	3.2	9	0.137	**0.041**
Road 1 Type (adjacent road)														
Leq(d)	Residential	61.3	6.2	57.9	9.2	19	Arterial, I-75	62.9	0.5	62.9	0.4	2	0.304	0.467
L10(h)	Residential	64.0	6.4	61.1	9.6	19	Arterial, I-75	67.8	1.1	67.8	0.8	2	0.043	0.400
Road 2 Type (next nearest larger road)														
Leq(d)	Residential	70.9	3.1	71.7	3.1	3	Arterial, I-75	59.9	4.6	57.8	5.2	18	0.008	**0.006**
L10(h)	Residential	74.4	2.7	75.1	2.7	3	Arterial, I-75	62.6	4.9	61.1	7.3	18	0.003	**0.003**
Leq(d)	Beyond	60.4	5.8	57.7	5.0	17	Within	66.1	4.0	65.5	5.6	4	0.055	**0.052**
L10(h)	Beyond	63.0	6.1	61.1	4.9	17	Within	69.9	3.3	68.9	2.4	4	0.014	**0.052**
Leq(d)	Beyond	60.5	6.1	57.7	5.8	15	Within	64.0	4.9	62.9	4.4	6	0.194	0.178
L10(h)	Beyond	63.2	6.4	61.1	6.3	15	Within	67.0	5.3	67.8	4.5	6	0.197	0.178
Leq(d)	Beyond	60.7	6.4	57.8	4.6	12	Within	62.4	5.3	62.5	8.0	9	0.519	0.508
L10(h)	Beyond	63.4	6.9	61.0	5.6	12	Within	65.5	5.4	67.0	6.4	9	0.436	0.345
I-75 Grade														
Leq(d)	Surface or Below	61.9	6.4	59.1	10.3	13	Elevated	60.7	5.2	58.9	5.0	8	0.642	0.916
L10(h)	Surface or Below	64.6	6.6	62.1	10.2	13	Elevated	63.8	6.0	61.6	7.2	8	0.786	0.860
I-75 Grade for sites within 100–325 m of I-75														
Leq(d)	Surface or Below	59.3	4.4	57.5	5.7	5	Elevated	65.8	5.1	63.2	4.6	3	0.146	0.143
L10(h)	Surface or Below	61.6	4.1	59.6	4.7	5	Elevated	70.2	4.3	68.6	4.1	3	0.048	**0.071**

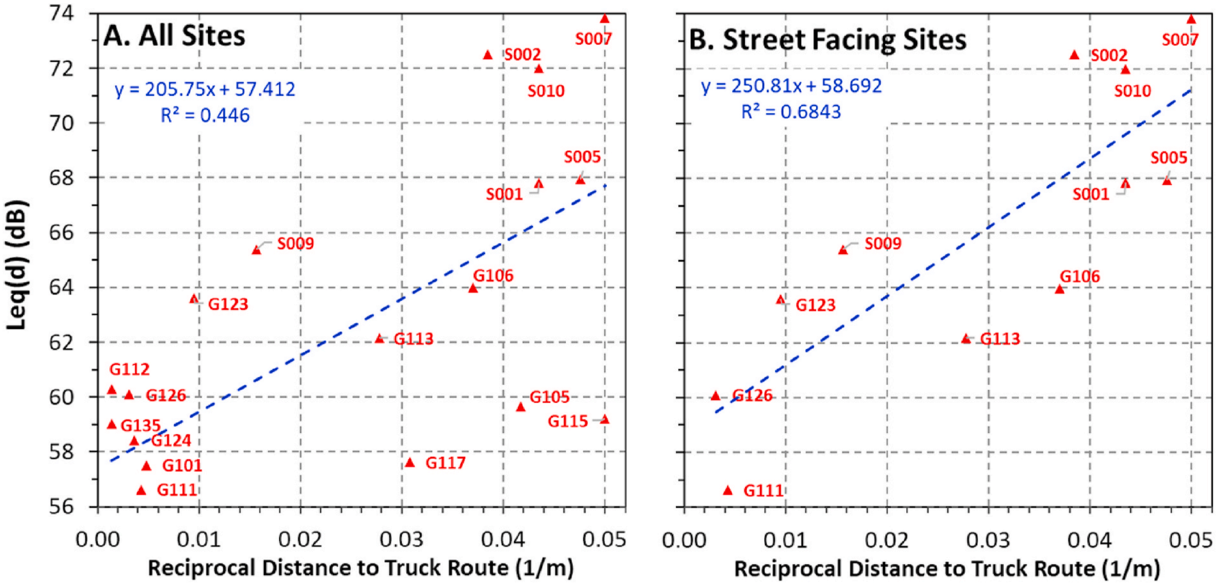

Fig. 4. 24-h average SPL by site versus distance from truck routes for all sites (A) and street facing sites (B). Blue line uses reciprocal relationship with distance. Distances range from 20 to 730 m for all sites, and 20–320 m for street facing sites. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

traffic influence. Still, our results suggest that proximity to trucking routes accounts for much of the noise, comparable to the magnitude of diurnal variation, much of which also is traffic driven. While effect sizes are smaller than seen earlier, this analysis helps confirm that both building shielding and proximity to truck routes are key determinants of noise levels.

3.4. Comparison of metrics

3.4.1. Diurnal trends

Trends over a 24-h period at the three "quietest" sites (G109, G131, G119) and three "noisiest" sites (S007, S002, S010) illustrate the behavior of the various metrics (Fig. 5). At the noisy sites, $F_{50}(h)$ was nearly always 1.0, indicating that the sound pressure level consistently exceeded 50 dB threshold. F_{60} was nearly always 1.0 during the day, but lower (0.10–0.60) in the late evening and morning, but still frequently exceeded the 60 dB nighttime criterion (incorporated in L_{DN} and L_{DEN}). Conversely, at the three quiet sites, $F_{65}(h)$ and $F_{70}(h)$ were nearly always zero, indicating few excursions over 65 and 70 dB. The $F_{60}(h)$ trend at these sites was bimodal, increasing in early morning and evening, the pattern shown by the 1-hr average, $L_{eq}(h)$. The $F_{65}(h)$ metric showed the greatest change over the day: levels were highest during the day and especially the morning, and low to moderate (0.1–0.5) in the evening, thus, the 65 dB evening criterion also was regularly exceeded. As expected (and shown later), the fraction over threshold metrics, for any given threshold, tended to increase with $L_{eq}(d)$. The temporal correlation among $L_{eq}(d)$ and fraction over threshold metrics mostly ranged from 0.2 to 0.7. In most cases, the fraction over threshold metrics were not normally distributed (based on S-W and D-P tests, and Q-Q plots).

The peak metrics (dashed lines in Fig. 5) had trends that differed from other metrics at some sites. The 90th percentile metric $L_{P10}(h)$ ranged between 65.2 and 82.2 dB at the three noisy sites with diurnal (hourly) variation between 12.5 and 14.1 dB; at the three quiet sites, $L_{p10}(h)$ was 53.9–61.7 dB with hourly variation from 4.0 to 7.8 dB. The higher percentile metrics showed several peaks at most sites; the three noisiest sites had peaks at 8 to 9 a.m. for $L_{P2}(h)$, $L_{P1}(h)$ and $L_{P0}(h)$, and 1-s maxima that reached or exceeded 100 dB. These metrics were somewhat sensitive to data cleaning, e.g., without data exclusions, 1-s maxima reached 123 and 126 dB at sites S010 and S002, respectively. The 98th percentile metric $L_{P2}(h)$ was selected as a compromise between the 90th percentile measure, which was less sensitive to changes, and the maxima, which was sometimes influenced by outliers and thus not robust. $L_{P2}(h)$, the 2nd highest 1-s SPL in the 2nd highest minute of the hour, reached 94.5–97.9 dB at the noisy sites and 66.3–68.8 at the quiet sites, and varied considerably over the day, especially at the noisiest sites (e.g., 22 dB at S007). Over the day, $L_{P2}(h)$ was moderately correlated with $L_{eq}(h)$ ($R = 0.84$–0.92), except at site G131 ($R = -0.07$). These peak measures did not have normal distributions in most cases.

3.4.2. Comparisons across sites

Fig. 6A-E plot the various metrics versus $L_{eq}(d)$ at the 21 sites using weekday data. These data are tabulated in a heat map in Table S3. Given their formulation, $L_{eq}(d)$, L_{DEN} and L_{DN} were closely correlated ($R^2 > 0.96$). As noted earlier, three sites exceeded the 70 dB guideline for $L_{eq}(d)$, and five sites for L_{DN} and L_{DEN}. $L_{10}(h)$ also was highly correlated with $L_{eq}(d)$ (Fig. 6A, $R^2 = 0.97$). We did not find consistent correlation between these metrics and the range, skewness or kurtosis of the distribution of 1-min and 1-hr averages (Table S4).

The fraction over threshold metrics generally increased with $L_{eq}(d)$, but depended on the selection of the threshold, as noted earlier (Fig. 6B). The time-weighted metrics F_{DN} and F_{DEN} were moderately correlated to $L_{eq}(d)$ ($R^2 = 0.58$), and individual sites varied from the trend line, especially at high noise levels (Fig. 6C). F_{DEN} had low to moderate correlation with other noise metrics, including supplemental noise metrics like the IR ($R^2 = 0.13$). At site S010, F_{DN} and F_{DEN} were 0.50 and 0.52, respectively, showing that 60, 65 and 70 dB criteria for night, evening and day periods were exceeded half of the time. F_{DEN} exceeded 0.3 at three other sites (S001, S002, G123), and F_{DEN} exceeded 0.1 at 10 of 21 sites, suggesting widespread potential for noise disturbance.

The peak metrics had slightly curvilinear relationships with $L_{eq}(d)$, especially $L_{P0}(h)$ and $L_{P1}(h)$, and variability increased with peak percentile (Fig. 6D). We focus on the 98th percentile measure $L_{P2}(h)$ for reasons noted earlier, which is shown in Fig. 6E along with the daily averages of the hourly values. $L_{P2}(h)$ ranged from 91 to 99 dB at sites S007, S002, S010, S005 and S009; 82–87 dB at S001, G106, and G123; and 70–74 dB for the lower half of sites. While most sites did not exceed the 70 dB criteria for $L_{eq}(d)$, $L_{10}(h)$, L_{DEN} or L_{DN}, several sites with high noise peaks were identified by other metrics. As examples: site S005 ($L_{P2}(h) = 95.2$ dB) is on a residential street with frequent truck traffic (AADT = 5,326, AADT = 901); S009 is on a wider boulevard with

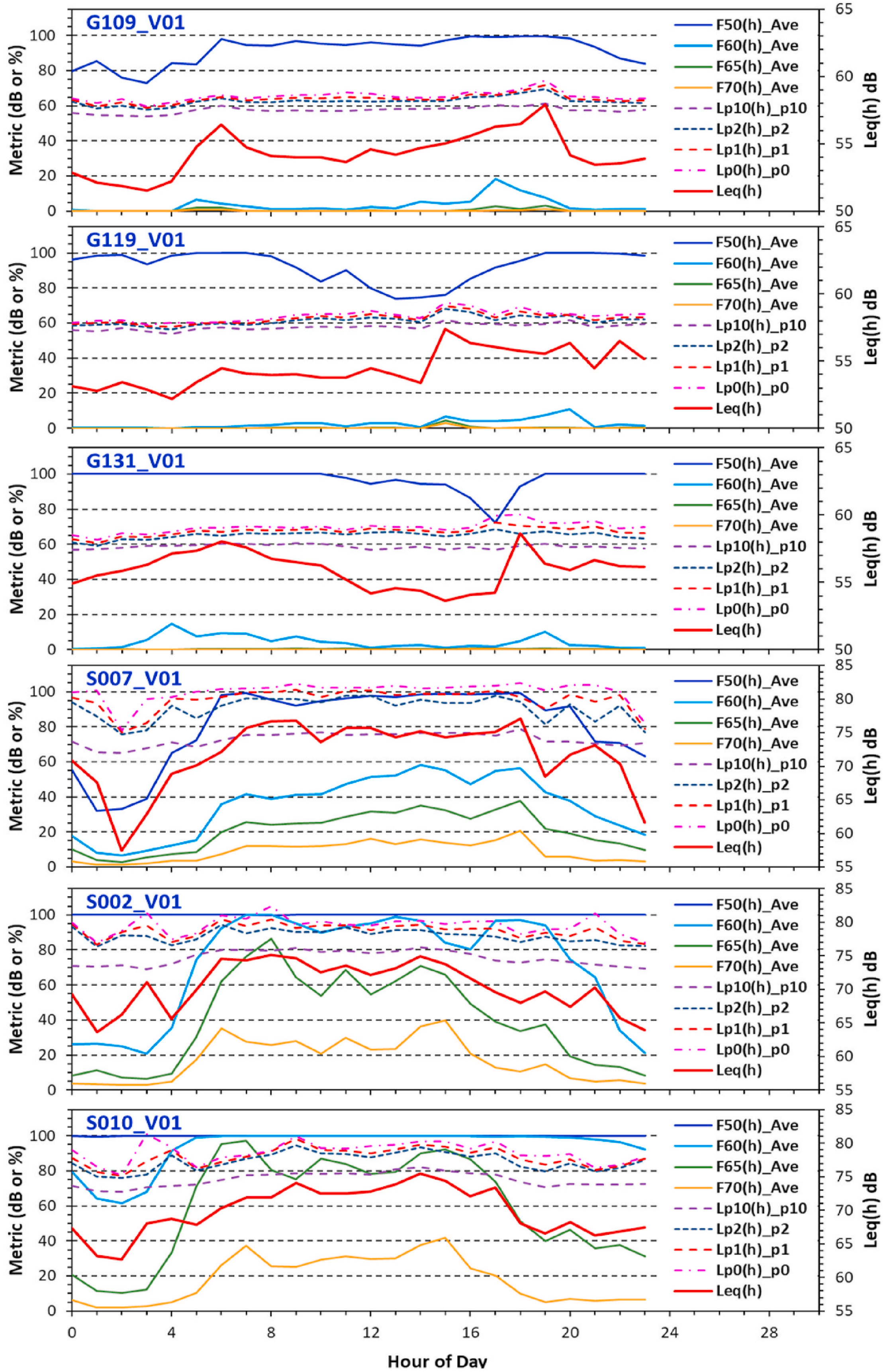

Fig. 5. 1 day of hourly noise metrics at the three quietest (G109, G131, G119) and noisiest sites (S010, S001, S002). SPL average $L_{ave}(h)$ uses right-hand scale. Scales change for the noisy sites.

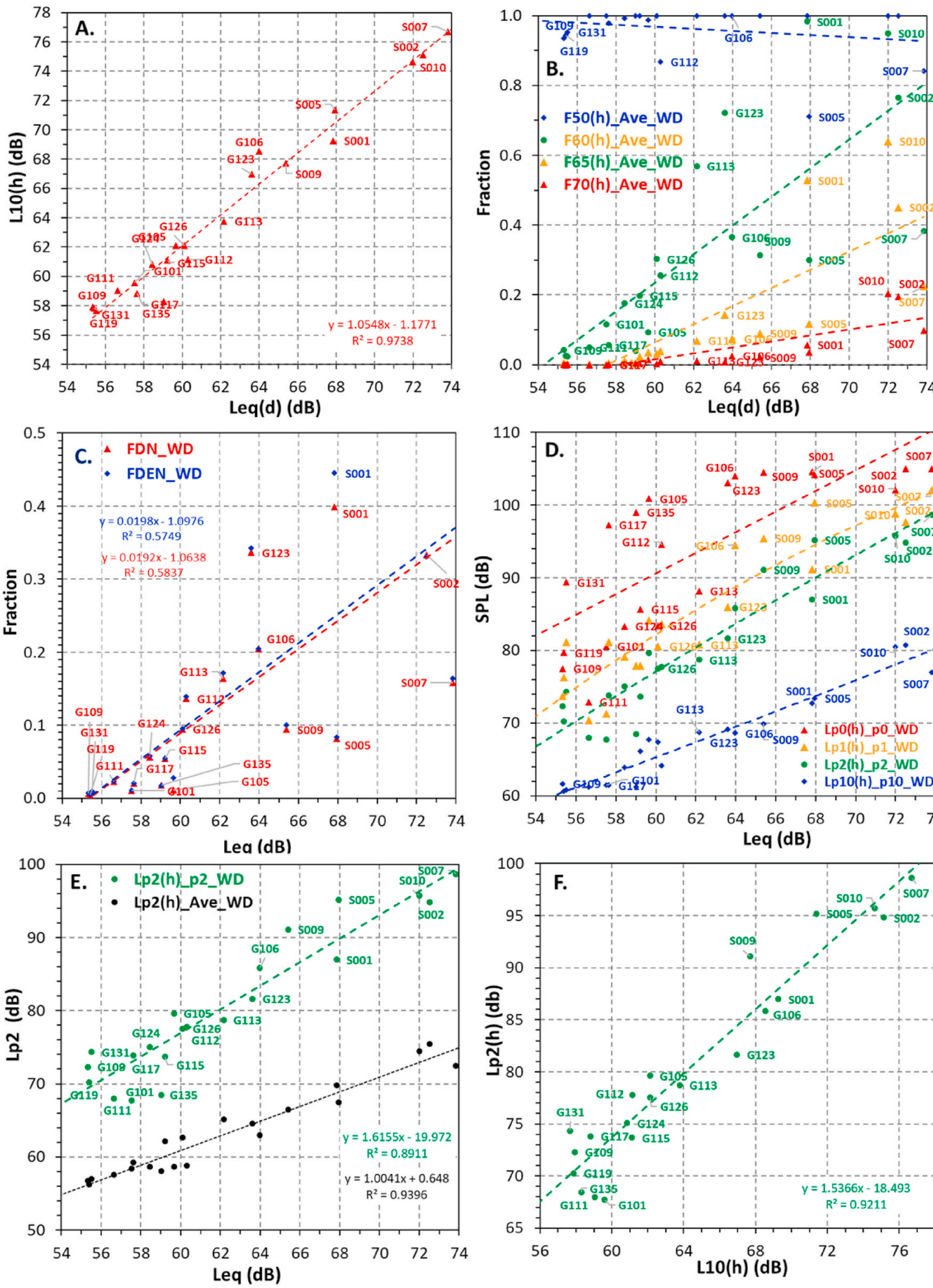

Fig. 6. Relationship between peak and fraction measures with average equivalent SPL $L_{eq}(d)$ (A–E), and 98th percentile peak and 98th percentile SPL metric $L_{p2}(h)$ (E). All plots use weekday data (N = 21).

limited truck traffic (AADT = 2,497, CADT = 84), 64 m from the I-75 service road and 104 m from I-75 (elevated at here), three houses separate the site from the service road, but the site has line-of-sight of a ramp from the I-75 service road; and G106 ($L_{P2}(h)$ = 85.9 dB) is on a surface arterial with frequent truck traffic (AADT = 2,514, CADT=NA), 321 m from I-75 (elevated here) and 220 m from ramps leading to I-75.

Time of day analyses for the noisiest six sites showed a minimum $L_{P2}(h)$ of 74 dB from 1 to 3 am, and a maximum of 90–93 dB from 9 to 12 a.m. (Fig. S8B), suggesting up to a 19 dB increase in short-term noise associated with traffic; sites S005 and S007 showed a 30–33 dB range. Importantly, this pattern may not be reflected in noise metrics using 1-hr or 24-hr averages, e.g., L_{DN}, L_{DEN} and $L_{10}(h)$, or in fraction over threshold measures, e.g., F_{DEN} and F_{DN}. The IRs for sites S005 and G106 were high (80.6 and 60.5%, respectively, for C = 5 dB), however, the IR does not indicate the SPL magnitude.

4. Discussion

4.1. Significance of traffic noise

Traffic noise is a defining feature of urban life. Exposure to road traffic noise has been associated with health, emotional, and behavior problems (Schubert et al., 2019), and community noise exposure often is an environmental justice issue (Khan et al., 2018). In addition, traffic and truck corridors are associated with higher levels of traffic-related air pollutants (e.g., nitrogen oxides and diesel particulate matter), lower income, poorer housing quality, and higher levels of mold and noise (Fecht et al., 2016; Kamal et al., 2014; Khan et al., 2018; Martenies et al., 2017), and the cumulative impact of these factors may exacerbate health disparities.

Noise exposure in the studied community was widespread. All sites had 24-h average equivalent SPLs that exceeded WHO and EPA recommendations (53 and 55 dB, respectively); eight sites had 1-hr averages (e.g., L_{10}(h)) that exceeded 65 dB for at least 1 h of the day; and based on L_{eq}(d), L_{10}(h), L_{DN} and L_{DEN}, 5 of 21 sites exceeded the 70 dB guideline used by FHWA and Michigan. Additional residences would have exceeded 70 dB had monitoring been conducted on the most exposed façade. The conventional noise metrics, e.g., L_{eq}(d), L_{10}(h), L_{DN} and L_{DEN}, incompletely reflect the intermittent and short-term noise associated with truck traffic on residential streets that can cause annoyance and interfere with speech. L_{DN} and L_{DEN} only partially reflect the greater sensitivity to noise in the evening or nighttime since they sum (weighted) nighttime and evening levels with (unweighted) levels from the rest of the day, potentially obscuring the adverse effects of the weighted periods (National Academies of Sciences Engineering and Medicine, 2014), while L_{10}(h) does not consider time of day. As discussed below, these conventional measures can be complemented with supplemental metrics that do capture these effects.

Our data indicated that trucks passing through otherwise quiet residential neighborhoods can routinely impair ordinary conversation, and we identified freeways and surface roads with moderate to high truck traffic as important noise sources. This was based on strong associations between noise metrics and proximity to freeways, arterials and ramps, as well as the diurnal and weekday/weekend patterns that matched truck traffic on freeways. Proximity to truck routes was associated with 7–17 dB impact on L_{eq}(d) or L_{10}(h), suggesting a 1.7–7.1 fold increase in perceived noise. Levels of short-term and intermittent noise depended on many factors but increases up to 33 dB were found for street-facing sites near truck routes and highways.

4.2. Supplemental metrics

Supplemental metrics can reflect characteristics of community noise that may not be captured in conventional metrics and provide useful interpretation of noise exposure, e.g., the extent of annoyance, sleep disturbance and other health issues attributable to traffic and other noise types (Gädeke et al., 1969; Li et al., 2011)· Another benefit of (certain) supplemental measures is their ability to communicate information in community settings (International Institute Of Noise Control Engineering, 2015). The IR is a commonly used supplemental metric, which reflects the contribution of peaks to average SPL (Brambilla et al., 2019; Wunderli et al., 2016), thus, the IR will increase at sites with occasional or intermittent (but not continuous) truck traffic. However, the IR requires selecting a parameter to define peaks, and it does not indicate the magnitude, frequency or timing of peaks. Annoyance or community noise tolerance (CNT) levels that transform SPLs into predictions of the number of people likely to be annoyed is a relevant indicator (Taraldsen et al., 2016), but noise-tolerance relationships can depend on noise types (Miedema and Vos, 1998) and calibration datasets may not be available or appropriate for local conditions. In the study area, for example, few houses have air conditioning and transmission of truck noise into living spaces through open windows may be greater than in other settings.

Our application of fraction over threshold metrics showed strong evidence of traffic noise, particularly at the noisier and street-facing sites. The proposed F_{DN} and F_{DEN} metrics combine the F_{60}(h), F_{65}(h) and F_{70}(h) metrics to account for evening and nighttime periods of the day when sensitivity to noise is greater, providing an approach consistent with the rationale behind the widely used L_{DN} and L_{DEN} metrics. Possibly the nighttime noise threshold in these metrics should be dropped to WHO's interim night noise guideline target (55 dB), especially considering the desirability of keeping windows open for ventilation. Fraction over threshold measures have been criticized since they can be sensitive to small changes in monitor placement, calibration or instrument errors, particularly as noise levels approach the threshold (International Institute Of Noise Control Engineering, 2015), and thus may not be suitable for regulatory purposes. However, our novel construction of F_{DEN}, using 1-s averages and three SPL thresholds, resulted in a scaler metric with a wide range (0.01–0.52), and it identified sites with intermittent traffic noise that were not always indicated by the conventional metrics. Our results suggest that F_{DEN} provides new and potentially valuable information for community noise assessments, and this metric is readily communicated in community settings as simply the fraction of time that noise guidelines are exceeded.

We also proposed several metrics to capture intermittent peak noise in community settings. These metrics also gave information that was independent, to varying degrees, of the other metrics. The 98th percentile measure proposed, L_{P2}(h), appeared to balance sensitivity to high but brief noise events and statistical robustness, i.e., it was not unduly influenced by outliers. For example, L_{P2}(h) counts a "burst" of noise lasting 2–60 s as a single peak, and two or more such peaks are needed to elevate the 1-hr measure. L_{P2}(h) ranged from 70 to 99 dB, and several sites that had values above 85 dB, indicating loud but brief periods of noise that were not identified by the other metrics. Other percentiles and formulations might help to quantify intermittent noise, and accounting for acoustic properties and very short (impulse) noise might also be valuable. While we focused on truck traffic, other types of intermittent noise may be important, e.g., residents in portions of the study community reported occasional loud bangs, which appear to be shipping containers being stacked or loaded onto trucks.

While no single supplemental metric has found universal acceptance for community noise assessment, the benefit of such metrics is their potential significance in revealing information relevant to community noise assessment. However, no guideline levels yet exist for the supplemental metrics.

4.3. Community monitoring and mitigation actions

Community involvement in noise monitoring and participatory research in general can enhance the relevance of research questions, strengthen interventions within the cultural context, and increase trust and understanding between communities and institutions (Israel et al., 2005; Lercher et al., 2017). Moreover, encouraging communities of color to advocate for their health and equipping them with appropriate tools is a deeply rooted principle of environmental justice. In this study, engagement with 14 youth and 7 adult volunteers and 20 community members donating the use of their house was integral to the project's success and provided opportunities for education (specifically in science, technology, and mathematics areas or STEM), as well as future opportunities for engagement. Unfortunately, we encountered a technical issue in the phone app that significantly limited the data collected, which ultimately was not used in this analysis, and which precluded our evaluation of the congruence of the app to the SPL monitors. The SI provides additional discussion of community engagement in noise monitoring, including use of phone apps and non-technical issues. Still, we saw that residences near truck routes, major arterials, freeways and freeway ramps experienced substantial levels of traffic-related noise, and multiple sites exceeded noise guidelines.

In Southwest Detroit, efforts to mitigate noise have been limited. Potential infrastructure strategies to control noise include the use of noise walls, roadside vegetation and buffers (Kalansuriya et al., 2009; Watts and Godfrey, 1999). To be most effective, such barriers should be at least 3 m high and made of absorptive materials, e.g., crumb rubber blends (Han et al., 2008; Watts and Godfrey, 1999). Across the 47,432 miles of interstate highways in the U.S., only 2748 miles have noise barriers (Cielec, 2015; Federal Highway Adminstration, 2017), suggesting this approach is underutilized. Often, noise walls are put in place during the construction of new highways, although vegetated buffers and walls are finding increase use. Few noise walls have been used in Southwest Detroit, although some new noise walls and shrub and tree planting efforts along I-75 are planned. Source strategies include noise limits on vehicles, quieter tires, smoother pavement, and road repair and maintenance (Ohiduzzaman et al., 2016; Waitz et al., 2007). Administrative strategies include restricting trucks and/or other vehicle types to designated routes that avoid residential areas, restricting traffic to certain periods, imposing speed, truck weight and size limits, banning engine braking and horn use, anti-idling strategies, identifying, repairing and/or penalizing noisy vehicles (e.g., with broken mufflers), and strict zoning and urban planning policies to separate residential and other land uses (Seshagiri, 1998; Wier et al., 2009). Finally, indoor exposure can be reduced by modifying walls, windows and doors to reduce sound transmission through the building envelope (U.S. Department of Transportation, 1993). Approximately 200 homes in the study area near new construction on I-75 are undergoing such improvements, sponsored by a community benefits agreement associated with the new international bridge and customs/immigration plaza in Southwest Detroit. (Noise levels reported here represent conditions before the new bridge is completed and do not account for the possibility of increased truck traffic in the future.)

We are aware of few local or national initiatives in the U.S. to monitor and address urban noise levels other than those described earlier and city ordinances, which are usually poorly enforced. Several trends are reducing traffic noise intensity, e.g., improved aerodynamics and vehicle electrification, but these may be countered by increasing truck traffic (both heavy duty and local delivery vehicles), increasing vehicle-miles-traveled, and urban densification. The cumulative effect of noise, traffic-related air pollutants, and other environmental justice issues identified earlier gives impetus to such initiatives.

Solutions depend on local conditions. Priorities in Southwest Detroit might include noise walls and vegetated buffers along I-75 and major arterials, and designated and enforced truck routes that eliminate truck travel through or near residential areas. Truck routing should consider existing logistics (e.g., warehousing), repair, freight transfer, intermodal, and other facilities, and any new such facility might only be permitted if a suitable route is available. Such efforts would benefit from detailed truck maps, noise surveys and maps, and the identification of residential and other locations of concern, with input from the community, governmental and other organizations.

4.4. Study strengths and limitations

Community-engaged work can serve multiple purposes, but also imposes logistical and technical challenges. While we monitored at occupied residences, 21 sites for an area the size and complexity of Southwest Detroit cannot represent all conditions of interest, and this number of sites is insufficient for noise mapping. The study area is bisected by highways and results may reflect conditions near major roads, but they may not be representative of suburban areas that have lower density, or central cities that have greater density and high-rise buildings. It would have been preferable to monitor street-facing and backyard sites simultaneously at all sites to understand shielding effects, to sample for longer periods and in multiple seasons to increase representativeness, to sample simultaneously at all sites to account for seasonality and other time-varying factors, and to derive the acoustic properties that correlate with truck traffic. Still, we captured at least 1.5–4 weekdays at each site, and we were able to examine time-of-day and weekday-weekend differences. Standardizing monitor placement at a certain distance and height from the roadway would aid some interpretations, although it would not reflect resident exposure as well as porch or yard locations that residents frequent. Vehicle counts were collected at only the S sites, and their completeness and reliability may have been affected by a lack of adherence to the sampling schedule, confusion about counting traffic in the designated direction, and participant fatigue. We identified several factors that could influence noise measurements and noise sources visually; intensity measurements would have been useful. We utilized $L_{eq}(h)$ and $L_{10}(h)$ in analyses of potential influences on noise levels; other metrics might reveal information relevant to community noise exposure. While factors affecting noise levels were identified using several approaches, including multivariate models, additional sites and additional site-specific data could be useful, e.g., vehicle counts and classification using tube counters or photographic records. Lastly, some results may be sensitive to data outliers, however, our overall conclusions remain unchanged with or without data censoring.

5. Conclusion

Monitoring in Southwest Detroit suggests that exposure to traffic-related noise is widespread, and residents near truck routes, highway ramps, and highways are disproportionately exposed. Sound pressure levels tend to be higher on weekday morning, midday and evening periods, particularly at street-facing sites with little screening or shielding. Conventional noise metrics e.g., $L_{10}(h)$, L_{DN} and L_{DEN}, indicated five sites that exceeded the 70 dB guideline and eight sites over 65 dB. The conventional noise metrics incompletely accounted for the short-term and intermittent nature of noise from truck traffic on residential streets. We demonstrated several supplemental metrics that provided additional information on short-term and intermittent noise levels, and suggest two new metrics that can provide insight on traffic-related noise: a metric called F_{DEN} that reflects the fraction of time that sound pressure levels exceed 60, 65 and 70 dB during night, evening and day time periods, respectively, and a peak noise metric called $L_{P2}(h)$ that utilizes the 98th percentile SPL with time blocks that separate peaks over minute and hour periods to increase robustness. We found significantly higher noise levels at residences within ~50 m of truck routes, arterials and freeway ramps, and the estimated impact from truck noise ranges up to 17 dB for hourly averages and 33 dB for peak measures. These metrics also suggest locations where noise may cause annoyance, sleep and health issues. Community involvement in evaluating noise assessment and informing the selection of abatement measures is suggested, particularly in neighborhoods that experience exposure to intermittent or chronic noise. These might include noise mapping, community surveys of annoyance or sleep disturbance, and community preferences for noise abatement measures.

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Human subjects

This study did not involve human subjects, however, an allied portion of the study obtained some information from participants and provided financial incentives, in accordance with the University of Michigan Institutional Review Board under for the project "The Gordie Howe International Bridge (GHIB) Air Monitoring", (HUM00148745).

Author statement

Stuart Batterman: Conceptualization, Methodology, Resources, Writing – original draft, Supervision, Funding acquisition. Sydni C. Warner: Writing – original draft, Visualization, Formal analysis. Tian Xia: Investigation, Software. Simone Sagovac: Conceptualization, Methodology, Supervision, Project administration. Benjamin Roberts: Writing – review & editing. Bridget Vial: Investigation. Chris Godwin: Investigation, Data curation.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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