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A study of urban housing demolition as a source of lead in ambient dust on sidewalks, streets, and alleys☆

Mark R. Farfel[a,b,*], Anna O. Orlova[b], Peter S.J. Lees[c], Charles Rohde[d], Peter J. Ashley[e], J. Julian Chisolm Jr.[a,†]

[a]Kennedy Krieger Research Institute, 707 North Broadway, Baltimore, MD 21205, USA
[b]Department of Health Policy and Management, The Johns Hopkins University Bloomberg School of Public Health, 624 N. Broadway, Hampton House 380, Baltimore, MD 21205, USA
[c]Department of Environmental Health Sciences, The Johns Hopkins University Bloomberg School of Public Health, 615 N. Wolfe Street, Baltimore, MD 21205, USA
[d]Department of Biostatistics, The Johns Hopkins University Bloomberg School of Public Health, 615 N. Wolfe Street, Baltimore, MD 21205, USA
[e]US Department of Housing and Urban Development, 451 7th St., SW, Room P3206, Washington, DC 20410, USA

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Abstract

We examined changes in ambient dust lead (Pb) levels associated with the demolition of older row houses containing lead paint in Baltimore, MD, USA. Our previous paper describes the three study sites, the demolition processes, and increases in the Pb dustfall rate during demolition (>40-fold) and debris removal (>6-fold) within 10 m of sites where wetting was of limited effectiveness. This paper presents the analysis of settled dust collected using a cyclone device from streets, sidewalks, and alleys within 100 m of study sites before, immediately after, and 1 month after demolition. We found acute increases in Pb loadings and dust loadings after demolition and debris removal that are of public health concern. Streets and alleys had the greatest increases in Pb loadings and the highest levels overall. At one site, geometric mean (GM) Pb loadings immediately after demolition increased 200% for streets to 8080 μg/ft^2, 138% for alleys to 6020 μg/ft^2, and 26% for sidewalks to 2170 μg/ft^2. One month after demolition, the GM Pb loadings for streets, alleys, and sidewalks were reduced on average by 41–67% from postdemolition levels and were below baseline levels for alleys and sidewalks. The other main site had smaller increases in GM Pb loadings immediately after demolition—18% for alleys to 1740 μg/ft^2 and 18% for sidewalks to 2050 μg/ft^2—and a decrease of 29% for streets to 2730 μg/ft^2. Exterior dust is a public health concern because it is a pathway of ambient Pb exposure and a potential source of residential exposure via tracking and reaerosolization and redeposition. Our findings highlight the need to control demolition-related Pb deposition and to educate planners, contractors, and health and housing agencies. This is particularly important given the large numbers of aging US dwellings that will be razed as part of future urban redevelopment efforts.

Keywords: Lead sources; Demolition; Dust; Urban housing; Street dust

1. Introduction

Urban redevelopment efforts include the demolition of aging housing and other buildings. An estimated 1.8 million US housing units will be razed this decade alone (President's Task Force, 2000), many of which will likely contain lead (Pb) in paint and dust (Jacobs et al., 2002). Redevelopment can reduce the number of high-risk housing units with lead-based paint hazards and

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*Corresponding author. Department of Health Policy and Management, The Johns Hopkins University Bloomberg School of Public Health, 624 N. Broadway, Hampton House 380, Baltimore, MD 21205, USA. Fax: +1 410 614 3097.
E-mail address: mfarfel@jhsph.edu (M.R. Farfel).
†Deceased.

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increase the stock of new housing free of lead-based paint. On the other hand, demolition mobilizes a potentially large source of Pb that is dispersed into urban neighborhoods where children are already at a high risk of lead poisoning (US CDC, 2000). Little research has been conducted on demolition-related changes in ambient and residential Pb levels. In a small study, Diorio (1999) found increased dust Pb loadings in neighboring houses, particularly when demolition was performed without wetting.

To assess demolition-related changes in ambient and residential Pb levels, we conducted a field study of three residential demolition sites in Baltimore, MD in 1999–2001. We measured ambient dustfall, exterior settled dust, and interior residential settled dust collected in the vicinity of the study sites. Our previous paper (Farfel et al., 2003) provided a detailed description and photographs of the study sites and the demolition processes, a summary of Pb paint testing prior to demolition, and the Pb dustfall results. It also discussed demolition-related community concerns and the need for a new protective approach to residential demolition.

As previously reported, we found acute increases in Pb dustfall during residential demolition activities. The geometric mean (GM) Pb dustfall rate within 10 m of study sites increased by >40-fold during residential demolition to 2700 $\mu g\,Pb/m^2$ per work day (equivalent to 251 $\mu g\,Pb/ft^2$ per work day) and by >6-fold during debris removal to 440 $\mu g\,Pb/m^2$ per work day (equivalent to 41 $\mu g\,Pb/ft^2$ per work day) compared to predemolition baseline values (Farfel et al., 2003). Dustfall Pb concentration also increased during demolition (GM = 2600 mg/kg) and debris removal (GM = 1500 mg/kg) compared to baseline (GM = 950 mg/kg). Pb in dustfall is a public health concern because it settles and becomes a pathway of ambient Pb exposure and a potential source of household exposure from tracking and reaerosolization and redeposition of exterior dust.

This paper presents findings related to changes in Pb levels in settled exterior dust collected from streets, sidewalks, and alleys within a 100 m radius of study sites. Demolition activities studied in this research were planned and performed by other entities as part of ongoing redevelopment efforts in East Baltimore.

2. Materials and methods

2.1. Study sites and demolition methods

As previously described (Farfel et al., 2003), demolition was performed using typical practices at three study sites in a low-income residential section of East Baltimore built prior to 1950. The area had no industrial sources of Pb exposure. Houses at Sites 1 and 2 were tested prior to demolition using X-ray fluorescence methods and were found to have had lead-based paint on interior and exterior surfaces.

Site 1 was one block of a 3.5 m-wide alley street with 26 two-story row houses demolished between October 27 and November 8, 1999. Site 2 was one block of 27 two-story row houses on a 3.5 m-wide alley street where all 13 houses on the east side and 5 houses on opposite ends of the west side were demolished during April 19–26, 2000. At Site 3, partial block demolitions were performed during April 1–12, 2000, on a total of 20 row houses located on adjacent blocks within 100 m of Site 2.

Demolition and debris removal were performed using a track-mounted backhoe with either a "claw" bucket or a material handler as described earlier (see photographs, Farfel et al., 2003). Wetting of the houses and debris where the demolition work was occurring was performed using a 3-in hose (Site 1) or a 1-in hose (Sites 2 and 3). Whole-block demolition was typically completed in a day, but debris removal took 1–2 weeks per site and entailed loading and removing hundreds of trucks or roll-off bins placed near the work site. For example, at Site 1 approximately 400 roll-off bins were loaded and removed (approximately 15 roll-off bins with a capacity of 15.3 m^3 (20 cubic yards) per house). Sometimes hoses were used for wetting during debris removal. When only two or three houses were demolished at a time (e.g., one side of Site 2) demolition and debris removal was completed the same day. The sites were then either backfilled with soil (<200 mg Pb/kg) from a remote location or covered with gravel.

2.2. Dust collection

We collected street, sidewalk, and alley dust within a 100-m radius of Sites 1 and 2 before demolition and immediately after demolition and debris removal. For logistical reasons, sample collection took more than a week at each site for each sampling period. Baseline samples at Site 1 were collected during a 4-week period starting June 24, 1999; additional baseline samples were collected just north of Site 1 between the end of September and mid-October of 1999 because demolition was expected to occur there also. At Site 2, baseline samples were collected at two different times (November 8–18, 1999, and March 4–31, 2000) due to delays in the implementation of demolition. The postdemolition samples at Sites 1 and 2 were collected during a 1–2-week period after the completion of demolition and debris removal activities. At Site 2, exterior dust samples were collected again starting 1 month after demolition and debris removal. At Site 3, only sidewalk dust was collected before, immediately after, and 1 month after demolition.

Settled dust was collected in tared 50-ml Teflon microwave digestion liners using a high-volume cyclone device attached to a hand-held Dirt Devil® vacuum as described elsewhere (Farfel, 1994a). The cyclone device's relatively high and consistent sample collection efficiency across a range of particle sizes and sample masses and its ability to be used for the collection of composite samples (Farfel, 1994a) made it suitable for this study. The cyclone device yields estimates of dust Pb loading (mass of Pb per unit area), dust Pb concentration (mass of Pb per gram of dust), and dust loading (mass of dust per unit area) that enabled calculation of several dust endpoints.

Composite dust samples were collected from the sidewalks, alleys, and streets at locations at both ends and in the middle of each block in the study area. Sidewalk dust was collected as a composite sample from five 1-ft^2 subareas (total area sampled was 5 ft^2) in front of each study house or other study location when houses were not present. Samples were collected in a standard pattern using disposable 1-ft^2 cardboard templates. At each location one subsample was collected from the sidewalk immediately in front of the front door or steps, a second was collected near the curb perpendicular to the door, and a third was collected halfway between these two subsamples. The remaining two subsamples were collected along the axis of the sidewalk parallel to the street at a distance of 1–2 m from the midsidewalk subsample. Alley dust was collected from alleys behind study houses using a pattern similar to that described above. Street dust was collected at the curb in front of each house as a composite of two 0.5-ft^2 subareas (total area sampled was 1 ft^2) separated by a distance of 1–2 m. A total of 648 street, alley, and sidewalk samples were collected. Four samples were voided in the laboratory, and the remaining 644 samples were analyzed for Pb.

2.3. Sample preparation and laboratory analysis

The microwave digestion liner was placed in a drying oven at 100 °C for a minimum of 4 h prior to taring. Tared and loaded weights were measured using a Mettler AM100 analytical balance (instrumental detection limit (IDL) = 0.002 g). Closed-vessel nitric acid digestion in a CEM Model 2100 microwave digestion system was used as the primary digestion method for dust samples according to SW 846 Method 3051 (US EPA, 1986a). When the amount of dust in the sample exceeded 2 g, nitric acid hot-plate digestion was performed according to SW 846 Method 3050 (US EPA, 1986b) on a 1-g aliquot of the dust sample. The following reagents were used: nitric acid (trace metal grade, concentrated, 69.9–70%), hydrogen peroxide (30% reagent ACS), and deionized water. The digestates were analyzed for Pb by inductively coupled plasma-atomic emission spectroscopy (Perkin–Elmer (PE) Plasma 1000) according to US Environmental Protection Agency (EPA) Method 6010 (US EPA, 1986c). Standard solutions for calibration were as follows: 0.25, 0.5, 1.0, 5.0, 10.0, and 20.0 mg/kg. Standard solutions were prepared in 10% nitric acid with GFS Chemicals Pb standard solution (1000 mg Pb/kg).

2.4. Quality control

At least one field blank was collected each day of sampling at each site. Laboratory quality control (QC) samples, including a reagent blank, were included in each analytical batch. QC samples were prepared by spiking reagents with known amounts of standard reference materials (SRM): 0.25 g of NIST SRM 2582 (Lead-Based Paint, nominal 200 mg Pb/kg); 0.25 g of certified reference material (CRM) 014-050 (Baghouse Dust, nominal 1914 mg Pb/kg); 0.5 g of NIST SRM 2710 (Montana Soil, nominal 5532 mg Pb/kg), and 0.5 g NIST SRM 2711 (Montana Soil, nominal 1162 mg Pb/kg). Stock solution spike and spike duplicate samples were prepared by spiking reagents with 0.5 mL of PE pure atomic spectroscopy standard (1000 mg Pb/kg).

For all cyclone dust samples in this study, the mean percentage recovery for the various QC samples was as follows: 94% ($n = 92$, SD = 4.3% and 5.2%, respectively) for stock solution spikes and for spike duplicates; 109% ($n = 69$, SD = 5.7%) for SRM 2582; 94% ($n = 74$, SD = 7.3%) for CRM-014-050; 89% ($n = 15$, SD = 9.6%) for SRM 2710; and 87% ($n = 15$, SD = 4.0%) for SRM 2711. For reagent blanks and field blanks, no evidence of systematic Pb contamination was found. Reagent blanks ($n = 89$) and field blanks ($n = 129$) had low mean Pb concentrations (0.12 mg/L, SD = 0.14 and 0.10 mg/L, SD = 0.10, respectively) that were below the limit of quantitation (0.55 mg/L, PE Plasma 1000) and close to the IDL (0.11 mg/L, PE Plasma 1000).

2.5. Data analysis

Data analysis included the calculation of the following metrics: Pb loading (μg Pb/ft^2), Pb concentration (mg Pb/kg of dust), and dust loading (g/ft^2). The calculated limits of detection were 5.5 μg Pb/ft^2 for Pb loading and 5.5 mg/kg for Pb concentration. For data analysis purposes, samples with values less than the IDL were recorded as IDL divided by the square root of two (Hornung and Reed, 1990).

Dust Pb loadings and concentrations and dust loadings were transformed using the natural logarithm (ln) prior to data analysis. Regression analysis was performed using generalized estimating equations (GEE) to account for correlation over time. For each surface type (street, sidewalk, alley), the regression analysis was performed separately by site due to differences in dust Pb concentrations across sites. The regression model included phase (before demolition, immediately after demolition, and 1 month after demolition) and a variable that represented distance from the demolition site.

Sidewalk dust data from Sites 2 and 3 were combined for data analysis and data display purposes. Predemolition dust samples were collected in November 1999 and again in March 2000 at Site 2 because of delays in the start of demolition work. The two baselines were not statistically different and were pooled for subsequent statistical analysis purposes. The figures are based on GIS methods as previously described (Farfel et al., 2003).

In order to make comparisons to EPA wipe-based standards for Pb loading in residential dust (US EPA, 2001), we estimated wipe-equivalent Pb loadings for the cyclone-based Pb loadings. In the absence of a conversion factor for exterior dust samples, we estimated wipe equivalents of cyclone-based Pb loadings using empirical data from a study of side-by-side wipe and cyclone samples collected inside houses (Farfel et al., 1994b). In that study, GM Pb loadings based on the cyclone device exceeded GM Pb loadings based on wipes by factors of 4.2 for floors and 10.4 for window wells. These two factors were used to calculate an estimated range of wipe-equivalent Pb loadings for individual and GM cyclone Pb loading values.

3. Results

Results are presented separately for the three dust endpoints. Because Pb loading is a function of dust loading (μg/ft^2) and Pb concentration (μg/g), the results for these endpoints are presented first. Overall, the pattern of results showed an increase in the amount of dust and Pb on exterior surfaces (i.e., an increase in dust loadings and Pb loadings) but little change in Pb concentration.

3.1. Dust loadings

Table 1 displays descriptive statistics on dust loadings over time by surface type and by site. In general, individual dust loading values were highly variable. GM values before demolition ranged from 0.68 to 5.2 g/ft^2 across surface types and sites. Before demolition, dust loadings were highest on streets, lowest on sidewalks, and intermediate on alleys. GM dust loadings were higher after demolition on streets, alleys, and sidewalks at both sites except for alleys at Site 1, and the changes were statistically significant for each surface type at Site 2/3 but not at Site 1. At 1 month, GM dust loadings at Site 2/3 were intermediate between the pre- and postdemolition values. For streets, dust loadings at 1 month were statistically significantly higher than predemolition values.

3.2. Dust Pb concentrations

In general, Pb concentrations were highly variable for each exterior surface type and site (Table 2) and higher for sidewalk dust compared to street dust. At Site 1, the GM sidewalk Pb concentration was higher than the GM alley Pb concentration. Before demolition, GM Pb concentrations by surface type and by site ranged from 518 to 2550 mg/kg. Street and sidewalk Pb concentrations before demolition were statistically significantly higher at Site 1 compared to Site 2/3 in the GEE model; the alley dust Pb concentration at Site 1 was statistically significantly lower than the corresponding value at Site 2/3. Based on the regression analysis for each site, Pb concentrations after demolition were not statistically different from those before demolition. At 1 month postdemolition, the GM Pb concentrations were lower than the pre- and postdemolition values for each surface type, but the changes were not statistically significant except for the change from predemolition to 1 month postdemolition for sidewalks.

3.3. Dust Pb loadings

Table 3 provides descriptive statistics on Pb loadings by site, surface type, and phase. Figs. 1 and 2 display Pb

Table 1
Descriptive statistics for dust loadings (g/ft^2) by site, surface type, and phase

Site	Phase	Sidewalk						Street						Alley					
		n	GM	Min	Max	Mean on ln scale	GSD	*n*	GM	Min	Max	Mean on ln scale	GSD	*n*	GM	Min	Max	Mean on ln scale	GSD
1	Baseline	38	0.679	0.025	7.92	−0.387	1.41	38	4.21	0.392	28.6	1.44	1.02	36	2.31	0.067	18.2	0.837	1.42
	After demolition	32	1.01	0.121	9.05	0.010	0.983	32	4.43	0.337	18.5	1.49	0.993	33	2.30	0.380	17.7	0.833	1.01
	1 month after	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
2/3	Baseline[a]	78	1.17	0.012	20.2	0.157	1.32	60	5.20	0.254	71.5	1.65	1.22	41	1.30	0.075	17.3	0.262	1.45
	After demolition	54	2.03	0.037	17.2	0.708	1.40	40	11.1	1.07	70.8	2.41	1.08	21	3.69	0.035	17.0	1.30	1.61
	1 month after	59	1.41	0.066	14.6	0.344	1.33	51	8.91	0.365	90.8	2.19	1.23	28	2.02	0.092	9.82	0.703	1.21

GSD, geometric standard deviation; ln, natural logarithm.
[a]Based on pooled data from both baselines (November 1999 and March 2000).

Table 2
Descriptive statistics for Pb concentrations (mg/kg) by site, surface type, and phase

Site	Phase	Sidewalk						Street						Alley					
		n	GM	Min	Max	Mean on ln scale	GSD	*n*	GM	Min	Max	Mean on ln scale	GSD	*n*	GM	Min	Max	Mean on ln scale	GSD
1	Baseline	38	2550	188	36,800	7.84	1.20	38	908	137	42,400	6.81	1.13	36	642	53	12,100	6.46	1.55
	After demolition	32	2020	126	29,500	7.61	1.27	32	617	16	128,000	6.42	1.33	33	756	38	13,600	6.63	1.44
	1 month after	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
2/3	Baseline[a]	78	1470	64	54,900	7.29	1.30	60	518	BDL	8050	6.25	1.37	41	1950	89	198,000	7.58	1.85
	After demolition	54	1070	153	96,300	6.98	1.64	40	727	74	47,200	6.59	1.34	21	1630	216	92,500	7.40	1.93
	1 month after	59	901	86	16,700	6.80	1.11	51	479	102	30,700	6.21	1.35	28	986	173	33,000	6.89	1.38

GSD, geometric standard deviation; ln, natural logarithm; BDL, below detection limit.
[a]Based on pooled data from both baselines (November 1999 and March 2000).

Table 3
Descriptive statistics for Pb loadings (μg/ft^2) by site, surface type, and phase

Site	Phase	Sidewalk						Street						Alley					
		n	GM	Min	Max	Mean on ln scale	GSD	*n*	GM	Min	Max	Mean on ln scale	GSD	*n*	GM	Min	Max	Mean on ln scale	GSD
1	Baseline	38	1740	67	96,100	7.46	1.76	38	3820	185	318,000	8.25	1.55	36	1480	10	134,000	7.30	2.03
	After demolition	32	2050	149	30,000	7.63	1.43	32	2730	5	1,300,000	7.91	1.89	33	1740	117	40,900	7.46	1.49
	1 month after	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
2/3	Baseline[a]	78	1720	9	88,700	7.45	1.83	60	2700	38	183,000	7.90	1.62	41	2530	56	187,000	7.84	2.12
	After demolition	54	2170	18	176,000	7.68	2.04	40	8080	221	1,750,000	9.00	1.82	21	6020	116	528,000	8.70	2.10
	1 month after	59	1270	38	24,800	7.15	1.67	51	4430	247	650,000	8.40	1.60	28	1980	160	96,900	7.59	1.59

GSD, geometric standard deviation; ln, natural logarithm.
[a]Based on pooled data from both baselines (November 1999 and March 2000).

loadings by surface type before demolition and changes in Pb loadings after demolition for Sites 1 and 2/3, respectively. Fig. 1 includes baseline data collected during the summer and fall of 1999. Fig. 2 also shows changes in Pb loadings 1 month after demolition compared to levels before demolition. When samples were collected from the same locations at both predemolition sampling periods at Site 2, the average value is displayed in Fig. 2. For this reason, the number of Site 2 baseline observations displayed in Fig. 2 is less than the corresponding number of pooled baseline observations for Site 2 in Table 3. Additionally, for each display in Figs. 1 and 2, a small number of observations are hidden from view or are beyond the map display.

Before demolition, there was a wide range of dust Pb loadings, and dust Pb loadings were higher on streets than on alleys and sidewalks (Table 3). After demolition, overall GM Pb loadings were higher than corresponding baseline values for each surface type and site, except for streets at Site 1. The maximum dust Pb loading was also higher immediately after demolition across sites and surface types, except for alleys and sidewalks at Site 1. Individual Pb loading measurements for streets, alleys, and sidewalks showed a mixed pattern of large and small increases and decreases after demolition (Figs. 1 and 2).

Based on the regression model in the presence of site and sampling phase, the increase in dust Pb loadings after demolition was statistically significant for streets and alleys but not for sidewalks. Based on the regression analysis by site, the increase in dust Pb loadings after demolition was statistically significant only for streets at Site 2. For each surface type at Site 2/3, the GM Pb loading at 1 month after demolition was lower than the immediate postdemolition value. For sidewalks and alleys at Site 2/3, the GM Pb loading at 1 month was lower than the predemolition value, but the difference was not statistically significant. Distance from the demolition site was not significant in any of the analyses in the presence of sampling phase.

3.4. *Comparisons to the EPA standard for Pb in residential floor dust*

Before demolition at Site 1, the GM cyclone-based Pb loadings exceeded the EPA floor standard (40 μg/ft^2) by 37-fold (estimated wipe equivalent of 3.6–8.8-fold) for

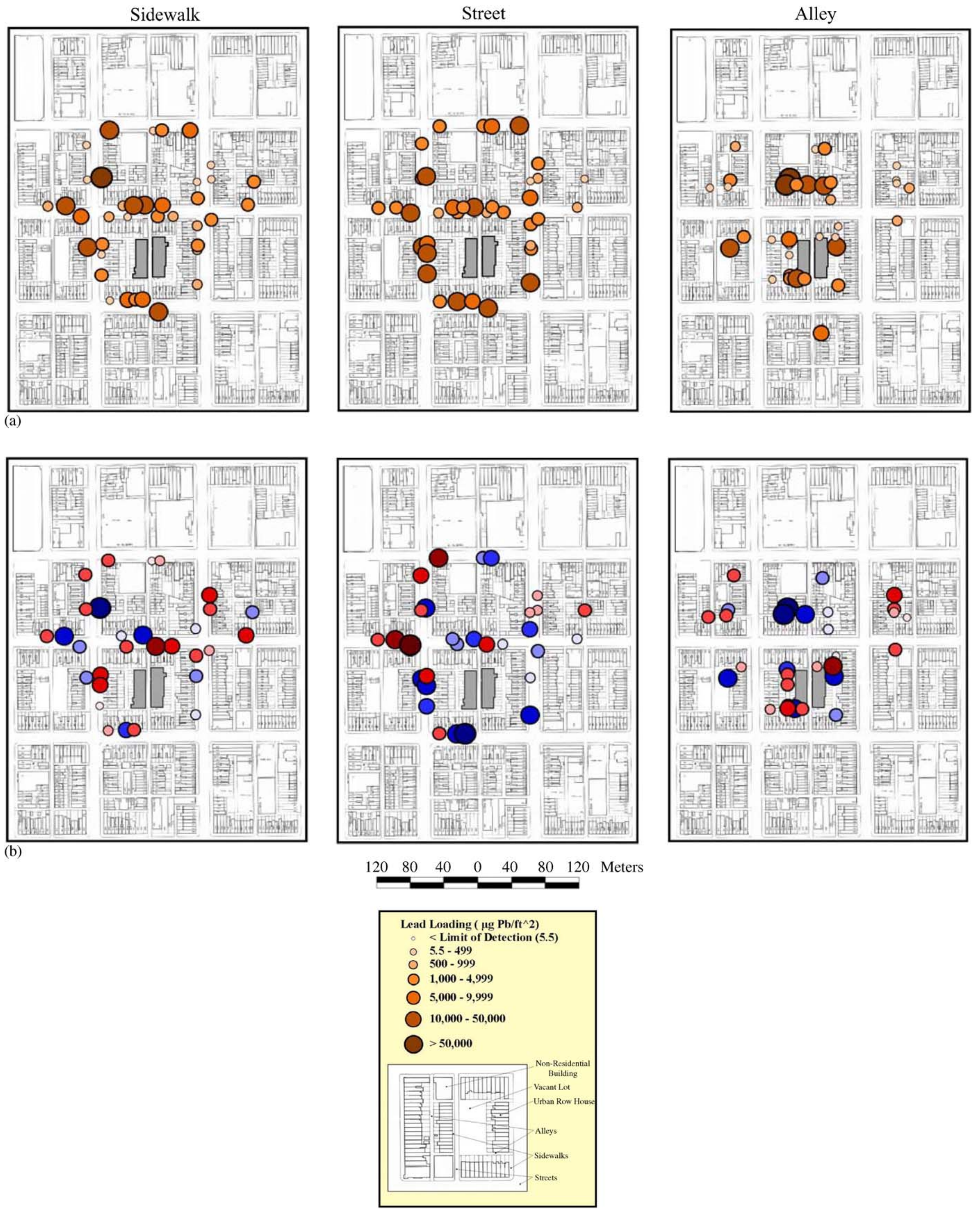

Fig. 1. Site 1 Pb loadings for sidewalk, street, and alley dust: (a) before demolition, and (b) changes from baseline immediately after demolition.

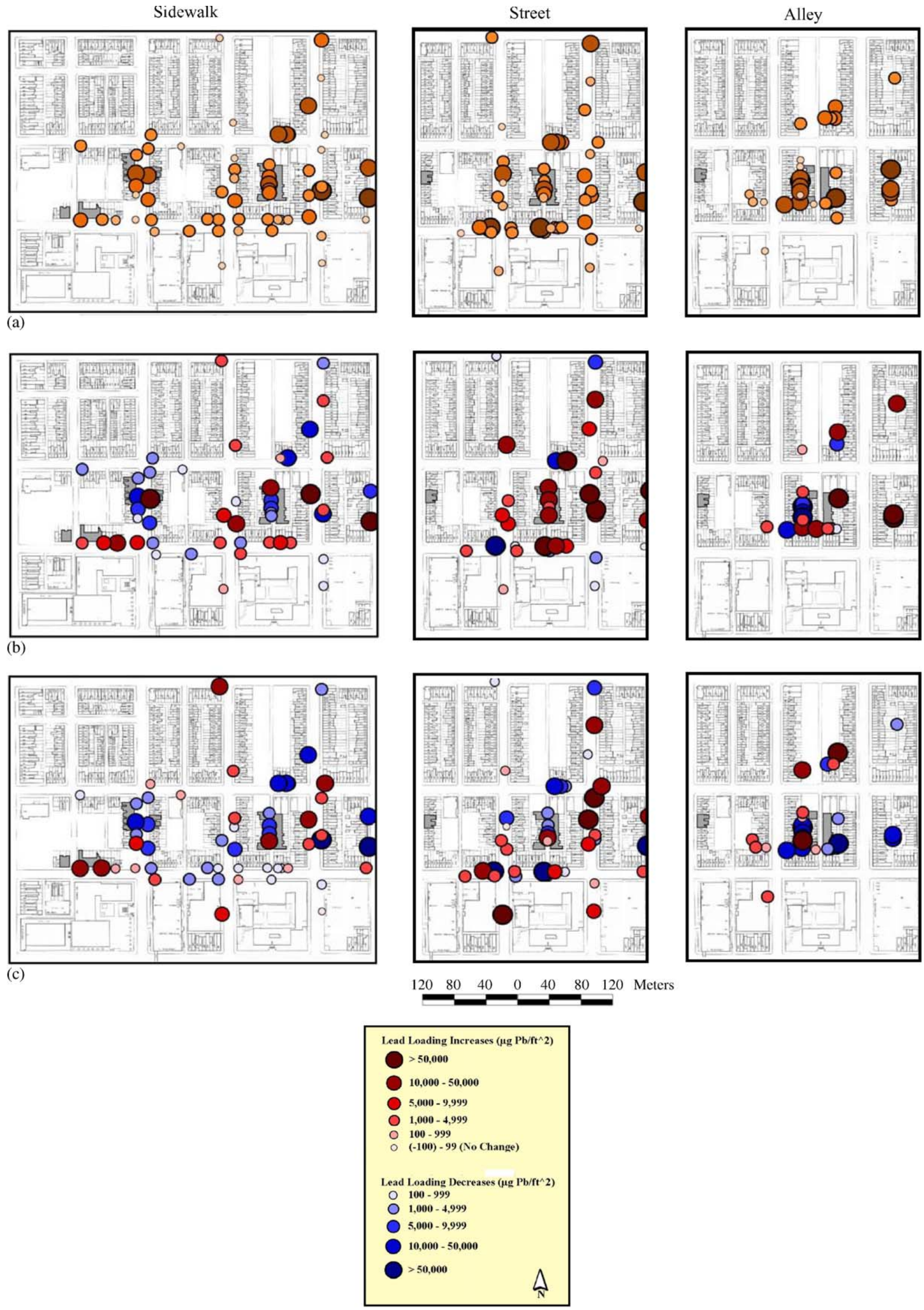

Fig. 2. Site 2/3 Pb loadings for sidewalk, street, and alley dust: (a) before demolition, (b) changes from baseline immediately after demolition, and (c) changes from baseline 1 month after demolition.

alleys, 43-fold (estimated wipe equivalent of 4.2–10.3-fold) for sidewalks, and 95-fold (estimated wipe equivalent of 9.2–22.7-fold) for streets. Before demolition at Site 2, GM cyclone-based Pb loadings exceeded the EPA floor standard by 63-fold (estimated wipe equivalent of 6.1–15-fold) for alleys, 43-fold (estimated wipe equivalent of 4.1–10.2-fold) for sidewalks, and 67-fold (estimated wipe equivalent of 6.5–16-fold) for streets. On an individual sample basis, exterior cyclone dust street samples before demolition at Site 1 were as much as 7950-fold (wipe equivalent of 764–1893-fold) higher than the EPA floor standard.

After demolition at Site 1, GM cyclone dust Pb loadings exceeded the EPA floor standard by greater amounts than before demolition in all cases, except for streets, as follows: by 43-fold (wipe equivalent of 4.2–10.4-fold) for alleys, 51-fold (estimated wipe equivalent of 4.9–12.2-fold) for sidewalks, and 68-fold (estimated wipe equivalent of 6.6–16.3-fold) for streets. At Site 2, GM cyclone Pb loadings exceed the EPA floor standard by 151-fold (estimated wipe equivalent of 14.5–35.8-fold) for alleys, 54-fold (wipe equivalent of 5.2–12.9-fold) for sidewalks, and 202-fold (wipe equivalent of 19.4–48.1-fold) for streets. On an individual sample basis, cyclone dust street samples at Site 2 were as much as 43,750-fold (estimated wipe equivalent of 4210–10,400-fold) higher than the EPA floor standard.

4. Discussion

By design, dust samples were collected within 100 m (approximately a two-block radius) of study sites to maximize the ability to detect demolition-related changes. This approach was important in a pilot study in which Pb loading in ambient settled dust was expected to be relatively high and variable (Table 3). Overall, the background ambient Pb levels reflect the fact that the study was conducted in older neighborhoods (median period of construction, 1939–1946) with a high prevalence of residential lead-based paint on interior and exterior surfaces. Other sources of Pb in ambient dust include the historic deposition of Pb additives in gasoline and the deterioration of exterior lead-based paint over time.

This pilot study examined several measures of changes in ambient Pb contamination associated with demolition of older row houses. We found increases in Pb loadings and dust loadings on exterior surfaces that were of potential public health concern after demolition and debris removal activities. Immediately after demolition and debris removal, increases in cyclone-based GM Pb loadings (Table 3) compared to predemolition baseline loadings ranged from 260 $\mu g/ft^2$ for Site 1 alleys to 5380 $\mu g/ft^2$ for Site 2 streets. In some cases, the increases were statistically significant. As explained below, the increases in the GM and individual sample Pb loadings, expressed in terms of wipe-equivalent lead loadings, substantially exceeded the EPA's wipe-based residential floor standard and in some cases the US Department of Housing and Urban Development (HUD)'s higher clearance guidance for exterior concrete. It is important to emphasize that our findings are associated with activities where site wetting was of limited effectiveness based on visual observations of large amounts of dust emissions during demolition and where site wetting was limited or absent during debris removal (see photographs, Farfel et al., 2003). The findings are likely to be generalizable to other neighborhoods and cities where row houses built before 1950 are demolished using similar practices.

The increase in exterior dust Pb loadings is primarily attributable to higher dust loadings after demolition (Table 1) rather than changes in Pb concentrations (Table 2). After demolition, increases in GM dust loadings (Table 1) compared to predemolition baseline loadings ranged from 0.22 g/ft^2 for streets and 0.33 g/ft^2 for sidewalks at Site 1 to 5.90 g/ft^2 for streets and 0.86 g/ft^2 for sidewalks at Site 2. The increases in exterior dust loading and Pb loading are consistent with the increased rates of deposition of dustfall and Pb in dustfall measured at study sites during demolition and debris removal (Farfel et al., 2003). They also reflect the fact that Pb in dustfall settles onto exterior surfaces and becomes a pathway of ambient Pb exposure, highlighting the need for effective dust suppression and control during and after demolition activities.

We also found differences between sites in the magnitude of the overall increase in Pb loadings. At Site 2, GM Pb loadings immediately after demolition were 200%, 138%, and 26% higher than baseline levels for streets, alleys, and sidewalks, respectively. Compared to postdemolition, GM Pb loadings 1 month after demolition at Site 2 for streets, alleys, and sidewalks were reduced on average by 45%, 67%, and 41%, respectively, and were below baseline levels in the cases of alleys and sidewalks. At Site 1, GM Pb loading immediately after demolition increased by a smaller amount (18%) for both alleys and sidewalks and decreased by 29% for streets. We also found a mixed pattern of small to large increases and decreases in Pb loadings immediately after demolition across sampling locations at both sites (Figs. 1 and 2).

The study was not designed to assess factors influencing ambient Pb and dust such as weather, traffic volume, traffic patterns, and street-cleaning activity. It did not assess spatial variability (via side-by-side samples) or temporal (e.g., daily, weekly) variability of Pb in exterior settled dust before and after demolition. For these reason, differences within and between sites and the patterns of changes in dust endpoints for individual samples cannot be readily explained.

In addition to direct deposition of Pb in dust during demolition activities, the potential exists for indirect deposition of Pb in dust when debris is transported offsite. Transportation of hundreds of debris-laden roll-off bins and trucks from the demolition site potentially disperses Pb and dust into the neighborhood beyond the immediate vicinity of the site. At the time of the study, Maryland law required trucks to cover their loads. The findings that streets and alleys had the greatest increases in dust Pb loadings and the highest levels overall suggest that transportation of debris did deposit Pb dust into the neighborhood. Information on street cleaning during and after demolition and debris removal is not available. Sampling distance from the site was not significant in any of the statistical analyses, perhaps due to the limited (100 m) sampling radius in this study.

The similarity of exterior dust Pb concentrations before and after demolition suggests that they share common source(s) of Pb (e.g., lead-based paint) and that past demolition-related dust deposition and/or exterior paint deterioration might be sources of the Pb in exterior dust measured before demolition. The similarity of Pb concentrations in street, alley, and sidewalk dust to Pb concentrations measured in dustfall at the study sites (Farfel et al., 2003) before demolition (GM = 950 mg/kg), during demolition (GM = 2600 mg/kg), and during debris removal (GM = 1500 mg/kg) also suggests that past demolition in the study area is a source of background ambient Pb.

The close proximity of the sidewalks to lead-painted row houses (Figs. 1 and 2) might account in part for the higher dust Pb concentrations on sidewalks compared to streets and alleys. In a recent study of exterior dust in 12 US cities, Clark et al. (2004) found higher Pb concentrations in entryway dust (GM = 1641 mg/kg) compared to street dust (GM = 431 mg/kg) and suggested that Pb in settled dust close to houses was a source of Pb in street dust. They also reported a GM Pb loading for street dust collected in Baltimore City (GM = ~1050 μg/ft^2) that was lower than the corresponding value we obtained before demolition and a GM Pb loading value for exterior entryway dust (GM = 2280 μg/ft^2) that was similar to our geometric mean values on sidewalks and alleys before demolition (Table 3).

4.1. Public health significance of the increased amount of Pb in settled exterior dust

Exterior dust is a residual source of Pb in dust in the urban environment. By increasing Pb in settled exterior dust, residential demolition and debris removal activities can increase the risk of Pb exposure beyond the acute work phase, especially among children. Lead-containing exterior dust is also of concern because it can be reaerosolized and deposited into houses or tracked inside on shoes (Adgate et al., 1998; Bornschein et al., 1986). This is important because the likelihood and frequency of children's exposure to Pb in dust is expected to be greater inside homes than from exterior surfaces because children typically spend more time indoors than outside (US EPA, 1997).

No health-based standards exist for Pb in exterior dust. For exterior concrete or other rough exterior surfaces, US HUD issued a postabatement clearance guidance level identical to its clearance level for window wells (troughs) (i.e., 800 μg/ft^2 based on wipe sampling, US HUD, 1995). This guidance level was not included in the EPA's Pb loading standards for residential dust (US EPA, 2001). To better understand the public health significance of the findings, we compared the estimated wipe-equivalent Pb loading results to the US EPA wipe-based standard for Pb in settled dust on residential floors (40 μg/ft^2). The rationale for this comparison is that dust on exterior surfaces is a pathway of Pb exposure for young children, via the hand-to-mouth route of ingestion, in and around the homes in the neighborhood surrounding the demolition site.

To assess whether the contribution of demolition to Pb in settled exterior dust is a public health concern, we compared the increase in GM and individual sample wipe-equivalent Pb loadings to the EPA's Pb loading standard for residential floors and HUD's clearance guidance for exterior concrete (800 μg/ft^2). We found that the increases in GM and individual sample wipe-equivalent Pb loadings can substantially exceed the EPA floor standard and in some cases exceed HUD's higher clearance guidance standard for exterior concrete. After demolition, increases in cyclone-based GM Pb loadings (Table 3) compared to predemolition baseline loadings ranged from 260 μg/ft^2 (estimated wipe equivalent of 25–62 μg/ft^2) for Site 1 alleys to 5380 μg/ft^2 (estimated wipe equivalent of 517–1280 μg/ft^2) for Site 2 streets. The absolute increases in cyclone-based GM values were 6.5 times (estimated wipe equivalent of 0.6–1.5 times) to 135 times (estimated wipe equivalent of 13–32 times) greater than the US EPA's floor dust Pb standard and up to 6.7 times (estimated wipe equivalent of 0.64–1.6 times) greater than the US HUD's clearance guidance standard for exterior concrete. The maximum Pb loading measured immediately after demolition was 1,750,000 μg/ft^2 (estimated wipe equivalent of 168,000–416,000 μg/ft^2).

The public health concern is particularly important in low-income and minority communities experiencing urban redevelopment that involves the demolition of multiple blocks of houses. Such communities are already at high risk of lead poisoning due to poor housing conditions and the age of housing (President's Task Force, 2000), and they have likely experienced cumulative increases in ambient Pb from multiple demolitions and other sources over time in the same neighborhood.

5. Conclusion

Residential demolition needs to be conducted in a manner that minimizes Pb release into the environment, which poses a potential source of exposure for residents and workers, so that the process of redevelopment does not increase risks of lead poisoning. The findings for dustfall and exterior dust highlight the need to identify and implement improved work practices and controls to minimize the dispersion of Pb in dust during and after demolition and debris removal and to minimize residents' contact with the site. Given the large numbers of older houses that will be demolished nationwide, it is essential that developers, contractors, urban planners, and public health agencies become more aware of demolition-related public health issues.

Future research should investigate Pb deposition and non-Pb endpoints associated with improved practices to control fugitive dust emissions during and after demolition, including practices to control the release of dust and Pb when debris is transported from the demolition site. Research is also needed on the effectiveness and longevity of various street and sidewalk cleaning methods for reducing dust loadings and Pb loadings. Future work should also assess changes in Pb in ambient dust at distances greater than 100 m from the demolition site and longer-term changes and patterns in ambient Pb.

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