Title

Primary prevention of outdoor lead (Pb) exposure on residential properties in Rochester, NY, and potential of a sustainable remediation solution involving the reuse of drinking water treatment residual (WTR) waste generated daily by the city.

The Authors

Padmini Das[1], Ph.D.; Stephanie M. Zamule[1], Ph.D.; Deanna R. Bolduc[1]; Michelle J. Patton[1]; Meghan L. Mendola[1]; Julia M. Penoyer[1], B.S.; Benjamin E. Lyon[1]; Hanna M. Chittenden[1], B.S.; Alexander C. Hoyt[1] B.S.; Cassandra E. Dupre[1], B.S.; Jack L. Wessel[1], B.S.; Ivan Gergi[1], B.S.; Charlotte V. Buechi[1], Jane A. Shebert[1], M.S.; Mandeep Chauhan[2], M.S.; David Giacherio[1], Ph.D.; and Jacob C. Phouthavong-Murphy[3], OMS-III.

Affiliations

[1]Nazareth College of Rochester. 4245 East Ave, Rochester, NY 14618, USA**;** [2]Northeastern University, 360 Huntington Ave, Boston, MA 02115, USA; [3]Touro College of Osteopathic Medicine 230 West 125th Street, New York, NY 10027

Corresponding Author

Padmini Das, PhD. Email: pdas8@naz.edu, phone: 5855000424; 585-389-2552, fax: 585-389-2672; Address: 4245 East Ave, Rochester, NY 14618, USA.

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Abstract

Background

Prevention of lead (Pb) poisoning is imperative for public health and environmental justice. This study presents both assessment and affordable mitigation of Pb exposure risks at individual yard scales, which are critically important to homeowners to attain primary prevention, but not yet explored thoroughly.

Objectives and Methods

In phase I, the potential risk of Pb exposure from soil, surface-dust, and vegetables grown in yard-gardens were measured along with secondary estimation of predicted blood Pb levels in children in seven urban and six suburban residential properties in and around Rochester, NY, U.S. In phase II, aluminum-based water treatment residuals (Al-WTR), a waste sludge generated by the city's drinking water treatment plant, was investigated for its potential to immobilize Pb as a remedial measure.

Results

All tested properties with pre-1978 built structures were found to contain Pb in exterior paint at varying depth, soil, surface dust, and garden vegetables. Soil Pb levels (SLL) were heterogeneous (11 to 173,500 mg/kg) and higher in urban than suburban properties (mean 3,178 and 461 mg/kg respectively), underscoring the inequitable risk. 71% of vegetables collected across four property gardens exceeded the European Union's health-based Pb guidelines and the median Pb in plants was 72 times higher than the U.S. market-basket values. Very strong sorption (>90%), minimal desorption (<2%), effective prevention (>85%) of Pb leaching from high Pb-containing exterior paint chips and estimated Freundlich and Langmuir parameters suggest practically irreversible Pb-WTR binding.

Discussion

Primary prevention tools were developed to communicate risk with homeowners through GIS-maps that illustrate risk-categories and prediction models to calculate SLL and predicted Pb in vegetables using distance from the house. Al-WTR showed very high Pb-immobilizing ability,

suggesting the implementation of this no-cost, zero-waste sorbent as a soil amendment would provide sustainable primary prevention for low-income urban communities where Pb exposure prevails.

1. Introduction

Lead (Pb) exposure continues to be the single most critical environmental health issue affecting American children (CDC 2002), despite the ban on lead-based paint (LBP) and the phasing out of leaded gasoline that took place four decades ago (USEPA 2011; Wagner and Langley-Turnbaugh 2007). Lead is a potent neurotoxicant particularly detrimental to developing fetuses and children under the age of six, as it causes cognitive and behavioral impairments, learning disabilities, attention deficit, juvenile delinquency, and violent or even criminal behavior (CDC 2002). The number of children with elevated blood lead levels (EBLL) has been reduced due to the implementation of secondary interventions such as testing BLL in children followed by tracking and mitigating the hazard source. However, the problem remains unaddressed in two critical areas.

Firstly, it is increasingly evident from recent research that although EBLL-based secondary prevention has been effective in reducing exposure to some extent, it comes at a cost to children who have already been exposed, and who pay a steep price in irreparable systemic damage through cumulative exposure (Crinnion and Pizzorno 2019). 80% of the total body Pb in children, and 90% in adults, is stored in bone; and while the half-life of Pb is only thirty-five days in blood, it is eight to forty years in bone (Hu *et al.* 2007). Pregnant and postmenopausal women undergo increased bone turnover, causing Pb poisoning of the fetus and Pb-related age-associated problems in adult women (Gulson *et al.* 2007; Nash *et al.* 2004). Cumulative exposure through bone-Pb turnover continues throughout life causing neuroinflammatory diseases like Alzheimer's and Parkinsonism at variable BLL (Bakulski *et al.* 2012; Weisskopf *et al.* 2010); increased cardiovascular mortality in BLL as low as 3.62 µg/dL (Menke *et al.* 2006); and anxiety and depressive disorders in young adults with BLL as low as 1.7 µg/dL (Bouchard *et al.* 2009). The BLL action limit for children has recently been reduced from 10 µg/dL to 5 µg/dL by U.S. Centers for Disease Control and Prevention, despite their acknowledgment that there is no safe limit for BLL in children; however, only 18 states (not including NY) have adopted this standard thus far (CDC 2018). Moreover, USEPA regulatory limits for residential soil Pb levels (SLL) (400 and 1200 mg/kg for bare soil in children's play areas and the remainder of the yard, respectively; USEPA 2001) are associated with BLL higher than 5 µg/dL according to two prediction models that independently showed a steep increase in BLL at lower soil Pb levels (Mielke et al. 2007; Johnson and Bretsch 2002). These guidelines, which are substantially higher than the soil Pb standards set by some European nations (40 to 150 mg/kg) and Canada (140 mg/kg), have remained unchanged for decades, despite recommendations by researchers to lower them to safer limits (Mielke et al. 2008; Reagan and Silderbard 1989; Bell et al. 2010).

Secondly, the current state of residential Pb exposure poses critical questions related to equality and social justice. Despite a substantial drop in the national average, EBLL still prevails in densely populated urban communities, meaning that children from lower-income minority families suffer most from Pb poisoning (Mielke 1999; Geltman *et al.* 2001). Fillippelli *et al.* 2005 reported that 15% of children living in urban communities exhibit BLL's above 10 µg/dL, compared to only 2.2% of children nationally. Similar trends were observed in western New York state cities, like Rochester and Buffalo, where most frequent Pb exposure was found to occur in low-income urban neighborhoods, where people of color are highly concentrated (Magavern 2018). The established correlation between Pb exposure and crime in teens and young adults in low-income urban communities (Nevin et al. 2007) underscores the socioeconomic injustice of this issue and the importance of developing a sustainable intervention that will address the environmental, social, and economic aspects of this problem equitably.

Primary prevention, which directly measures and corrects hazards before an exposure happens (CDC 2005), is the only process that can provide an equitable solution to the Pb exposure issue without vulnerable populations being disproportionately affected as collateral damage. Sustainable measures of primary prevention are not easy to attain, as they involve a number of multifaceted tasks and challenges. A complete assessment of SLL distribution, outdoor surface dust, and property garden vegetables at the individual property scale is critically important for homeowners to understand the contamination pattern, associated health risks, and potential remedial strategies (Clark and Knudsen 2013). Primary contributors of EBLL in children are Pb in soil and house dust that consists of 20-80% resuspended Pb from the soil, along with other dietary sources. (Mielke et al. 2007; Wu *et al.* 2008; Bugadalski *et al.* 2013). One-third of U.S. residents grow at least one vegetable in their yard for intended consumption (National Gardening Association 2011), heightening the risk of Pb exposure from contaminated yard soils (Clark and Knudsen 2013). This percentage is expected to increase due to the recent COVID-19 pandemic given the disrupted access to fresh and nutritious foods (Lal 2020). Further, an effective and affordable remediation solution must be provided to the families of lower socioeconomic standing to prevent Pb exposure at identified hot spots. Surface soils act as a sink for accumulating Pb over years and can remain so for centuries in the absence of proper remediation and intervention (LaBelle *et al.* 1987; Datko-Williams et al. 2014). Although the excavation of the surface soil with high Pb is the best practice to remove the risk (Mielke et al. 2011), this process is not economically feasible for homeowners in lower socioeconomic urban neighborhoods. The lack of access to risk assessment measures and economically-feasible alternatives for Pb removal contributes to the ongoing inequitable Pb poisoning in children.

To address primary prevention, we investigated thirteen residential properties in and around Rochester, NY, United States, utilizing three unique approaches that differ from earlier studies. As opposed to evaluating commercial and residential locations on a city-wide scale, we focused on a site-specific scale of individual properties with structures built prior to 1978 as the principal outdoor Pb sources. Furthermore, we determined the spatial distribution of SLL, dividing every property into broader risk categories such as: safe areas with minimal to no risk; low-risk areas safer for children to play, but not safe enough to grow edible produce; and high-risk areas with Pb hot spots, which should be avoided and considered for remedial actions. Using this data, primary prevention tools including mathematical prediction models and Geographical Information System (GIS) maps were developed to provide homeowners with a thorough understanding of the risk distribution on their properties. Finally, to address the need for a cost-effective remediation strategy to reduce exposure risk from the Pb hot spots, we explored the potential reuse of a waste sludge as a soil amendment to chemically immobilize Pb. This water treatment residual (WTR) is generated as a waste product on a continual basis by the city's largest drinking water treatment plants. Successful implementation of this technology will provide the city with a sustainable solution for reducing costs of waste management as well as provide homeowners with a low-cost solution to reduce the risk of Pb exposure from high-risk property hot spots. To the best of our knowledge, this study is the first to focus on both aspects of primary prevention by directly measuring the outdoor Pb levels at individual yard scales in urban and suburban residential properties and proposing a low-cost, zero-waste solution for the homeowners by reusing a waste that the city generates every day.

2. Methods

2.1. Phase I: Assessment

2.1.1. Field Site Descriptions

Seven urban and six suburban residential properties were selected based on varying characteristics including the age of the property's structures; socioeconomic categorizations of the homeowners such as race, education, and household income; and the willingness of the homeowners to participate in this study. Tableau data visualization software (version 2020.1) was used to create site location maps with respect to the number of houses built prior to 1978 and the socioeconomic standing of the corresponding areas of the sub-county (figure 1). The software was used to develop a geographical map of Monroe and Ontario counties utilizing latitude and longitude coordinates for each sub-county boundary. An urban location was defined by a geographical location within the city limits of Rochester, NY, United States. A suburban location was defined as a geographical location outside the city limits and within the county limits of Monroe or Ontario counties. The socioeconomic categorization data was collected from the Census Bureau for each respective sub-county.

2.1.2. In-situ measurement of soil Pb using XRF

Soil lead concentrations were measured on-site using a USHUD certified Thermo Scientific™ Niton™ XLp 300 series x-ray fluorescence analyzer (XRF). The XRF was set to "standard bulk" mode while an uncovered soil surface was chosen to conduct the measurement in each location (n = 20). The soil surface locations included: i) alongside structures including corners and beneath doors, front stairs, and windows; ii) along a perpendicular line starting from a high soil Pb point adjacent to the structure towards the property line; iii) adjacent to sidewalks and near structures of neighboring properties. Additional parameters including variation in slopes and presence of loose paint chips were noted if observed. Once recorded, each measurement was marked with flags with different colors (figure S.1), a process used to communicate with the homeowners about the distribution of soil Pb on their property and potential risk associated with it.

2.1.3. Measurement of Pb in paint and outdoor dust

Exterior paint was measured *in-situ* using the "paint mode" of the XRF on outdoor surfaces such as siding, doors, steps, window sills, pillars, and old housing materials stored outside and inside a barn, (site E only). Calibration of the XRF was performed according to the HUD calibration protocol. Levels of Pb in the paint on a given surface area and a depth index indicating how close the Pb was to the surface were obtained as part of each measurement.

Outdoor dust collection was completed using Fisher Scientific™ dust wipes (catalog number: NC0309992) on any surface which registered an XRF lead reading. To ensure maximum collection, the dust wipes were unfolded for the initial collection, then folded in half and used within the same surface area. Pb levels on each dust wipe were measured in the laboratory using the "dust wipe" mode of the XRF by placing each wipe in a Niton Portable Test Stand and scanning for 80 seconds total (20 seconds for each position of the holder).

2.1.4. Collection of produce/plant samples and preparations

Four assessed properties (A, H, C, M) had home gardens. Produce and plant samples were collected with the permission of the homeowners. Plant samples were cleaned and weighed using a Fisher Science Education analytical balance, dried in a ThermoScientific Heratherm OGS100 oven at 105°C for 24 hours (or until completely dry), and then weighed again to calculate moisture content. A dry weight (maximum of 0.5 g or based on the available mass of produce) of plant sample was digested in Fisher Scientific Trace Metal Grade HNO_3 following the US EPA 3051 protocol in a CEM MARS6 digestion microwave unit. Once digested, samples were filtered through Thermo scientific 0.45 µm PTFE-L Luer-Lock syringe filters, diluted to 50 mL with RO water from a Thermofisher Scientific Barnstead Nanopure purifier, and centrifuged at room temperature in an Eppendorf 5424R for 15 minutes at 4,000 rpm.

2.2. Phase II: Remediation

2.2.1. WTR Collection, Characterization, and TCLP

Two batches of WTR were collected from two sludge lagoons of Monroe County Water Authority's drinking water treatment plant in Greece, NY, U.S. in 2017 and 2018. These lagoons contained the post-flocculation waste sludge from Lake Ontario source water after being treated with alum (aluminum sulfate). WTR samples were then thoroughly mixed, air-dried, ground into powder, and passed through a 2 mm sieve. Triplicate samples collected from each batch of WTR were used for determining physicochemical properties, including pH, EC, moisture content, and organic matter content. Toxicity characteristic leaching procedures (TCLP) were performed using EPA SW-846 methods 1311.

2.2.2. Sorption and Desorption of Pb by WTR

Kinetic and equilibrium sorption and desorption experiments were carried out in triplicate using a 1:20 solid to solution (m/v) ratio (SSR) of WTR to a lead-spiked solution made from lead nitrate and 0.01M KCl as a background electrolyte. Kinetic sorption/desorption experiments were conducted at four initial Pb concentrations (0, 25, 200, and 1000 mg/L) for 24h with six intermediate sample collections after 0, 1, 2, 5, 10, and 24 h during end-over-end mixing on an Analytical Testing Model DC-20 Rotary Agitator shaker at 250 rpm. Similarly, equilibrium sorption experiments were conducted at six initial Pb concentrations (0, 200, 400, 600, 800, and 1000 mg/L) for 10 h using the same sampling method. Three subsequent desorption cycles (24h each) were carried out to determine the extent of the potential release of pre-adsorbed Pb from WTR. In addition to shaking time, the WTR remained in contact with the Pb solution for approximately 8 min (0.13 h) during the sample preparation. This time in combination with the shaking time was represented as the contact time for kinetic data. After the respective sorption or desorption processes, samples were centrifuged using an Eppendorf 5804R for 15 minutes at 4000 rpm, filtered through Thermo scientific 0.45 µm PTFE-L Luer-Lock syringe filters, and the supernatants were analyzed for Pb and Al.

2.2.3. Optimizing WTR to prevent paint chip Pb leaching

Loose paint chips were collected from the yard of site A both before and after the home was renovated, in order to conduct two experiments. First, a TCLP test was conducted to estimate the Pb leaching potential from these scattered paint chips in simulated landfill conditions following the EPA SW-846 methods 1311. Second, a batch desorption experiment in the presence of varying levels of WTR was performed to determine the optimal amount of WTR necessary to effectively prevent Pb leaching. Four WTR-to-paint chips (m/m) ratios (10, 20, 50, and 100) were added to a

0.01 M KCl solution while maintaining a 1:20 SSR. Samples were equilibrated for 7 d with two desorption cycles (1 and 6 d) using end-over-end mixing. The samples were centrifuged at 4000 rpm for 15 mins, filtered through a Thermo scientific 0.45 µm PTFE-L Luer-Lock syringe filter, and the supernatants were analyzed for Pb.

2.3. Pb Analysis

In phase I, Plant samples were analyzed for lead content in triplicate using an Agilent Technologies 4210 MP-AES. The MP-AES was calibrated with standards ranging from 2500 to 10000 µg/L prepared from a stock solution of Agilent Technologies 1000 µg/mL Pb in 5% HNO3. Method analysis settings were as follows: pump speed of 25 rpm, sample uptake time of 15 seconds, stabilization time of 15 seconds, read time of 30 seconds, and nebulizer flow of 0.8 L/min.

In phase II, Pb in 5% of *in-situ* soil samples (following USEPA method 6200), Pb and Al from the batch sorption/desorption, and Pb from leaching experiments were measured using a Perkin Elmer 400 graphite flame Atomic Absorption Spectrophotometer (AAS). After TCLP, the eight RCRA metals (Arsenic, Barium, Cadmium, Chromium, Lead, Mercury, Selenium, and Silver) were measured using a Perkin Elmer Optima 8000 inductively coupled plasma optical emission spectroscopy (ICP-OES) to attain data at lower detection limits. Calibration coefficient limits were set to 0.995, with an allowed calibration error of 10%, and quality control samples were accepted within $\pm$5% error margins.

2.4. Data analyses, Modeling, and Mapping

Statistical analyses were carried out using JMP IN version 13.0 (Sall et al. 2005). One-way and Two-way ANOVA were performed as required, followed by mean comparisons using the Tukey–Kramer honest significant difference (HSD) test. Multivariate correlation analyses of Pb in surface dust were conducted with Pb in paint and depth index; bivariate correlation analyses of Pb in edible produce or plant parts were conducted with the adjacent soil Pb levels. Two types of prediction models were developed using the relationship between i) soil Pb over distance from the structure(s) at each site and ii) Pb in produce/plants grown in the yard gardens with the adjacent soil Pb concentrations. Sorption data were fit to both Freundlich and Langmuir isotherm models to gain insight on possible mechanisms of sorption between Pb and WTR and the reversibility of this binding. XRF measurements of SLL were taken along with the latitude and longitude of each sampling location and the collected readings were imported into ArcGIS Pro 2.5.2 and portrayed with proportional symbols, color categories, and heat maps.

3. Results

3.1. Outdoor Pb-exposure in urban and suburban residential properties

As described in fig. 1 and table S.1, seven urban and six suburban residential properties built between 1830 and 2003 were tested for outdoor Pb exposure sources. Among those, the house at site F was the only structure built after 1978. As expected, there was no Pb found in exterior paint, dust, or natural soil at this property. Interestingly, low soil Pb levels (27.8 $\pm$ 0.98 mg/kg) were found at two sites in the garden area, both of which were covered with store-bought potting soil and mulch, suggesting that either may have contributed Pb to the garden. Two suburban sites (I and J) were found to have Pb in exterior paint and soil; however, the SLL in these two properties were below 400 mg/kg. All seven urban (A, B, C, G, H, K, M) sites and one suburban (L) site were found to have SLL higher than 400 mg/kg. Table S.1 shows the Pb in paint, its depth index, and the maximum concentration of Pb in soil and dust.

3.1.1. Distribution of Pb in soils

The distribution of SLL was extremely heterogeneous in all properties assessed and thus was explored further in eight properties that surpassed the federal limit. Three factors were found to be controlling this distribution; i) distance from the most prominent source of LBP, ii) slopes and iii) the presence of a secondary source at close proximity. Among these, the distance from the primary source of LBP was found to be the dominant factor controlling the distribution of SLL. The maximum SLL were found alongside the source structures at all sites, indicating the most likely primary source was LBP that leached into the soil over the years (figure 2). The data shows a continual decrease of soil Pb over distance, dropping below 100 mg/kg in all properties at variable distances from the houses. However, a few exceptions were noted in A_2, L_1, G_1, G_2, and K. These variations in SLL against the effect of distance were observed due to the presence of loose paint chips at close proximity to the measured locations in A_2 and slopes in L_1. Both sites G and K were inner-city residential properties with smaller yards where the structures of the neighboring property influenced the distribution of SLL by contributing as secondary sources of Pb (figure 3).

In each yard, SLL showed a biphasic pattern: initially steep and then a gradual decrease over distance from the source structures. This trend was utilized to create site-specific and overall regression models. Site-specific models based on polynomial cubic regression equations are presented for seven yards in table 1, which exhibited both significance as well as strength (R^2 ranging from 0.84 to 0.99). In contrast, the overall bivariate regression models across all sites were significant for linear ($p = 0.0011$) as well as nonlinear polynomial ($n = 2$ to 6) fit (table S.2), but neither were acceptable to be used, as the regression coefficients were too low (no more than 0.24).

3.1.2. Pb in outdoor surface dust

Figure 4 shows the multivariate correlation of surface dust collected from different surfaces with Pb in paint and depth index. All surfaces showed a significant ($p < 0.05$) correlation with both Pb in paint and depth index. Interestingly, Pb was found in all of these surfaces at a low depth index, suggesting the presence of Pb closer to the surface that impacted its loading in the dust. Figure S.3 shows a similar multivariate correlation for surfaces with a higher depth index and the surface dust only showed a significant correlation with Pb in paint but not with depth index. Surface dust measurements in all urban properties assessed (with the exception of site C) exceeded HUD's action level of 800 µg/ft^2 for exterior concrete. In contrast, surface dust measurements were within this limit in all suburban properties tested, with the exception of a barn at site E which showed a very high risk of potential Pb exposure through inhalation of dust. The assessment of Pb was conducted inside the barn built in 1830; however, the barn is considered an outdoor source as it was not located adjacent to the house built in 1966. The barn is currently used to store materials from a previously demolished structure and seldom used as a play area for visiting grandchildren of the current property owners. All 19 objects tested in this barn (including the wall, doors, wall decor, and various old materials) contained Pb in the paint with an average depth index of 1.66 $\pm$ 0.62, showing most of the Pb is on the surface. All 19 dust samples collected from this site were found to contain Pb averaging 1,551 µg/ft^2, almost twice the level of HUD's action level. The highest concentration of Pb-containing surface dust reported in this study (site E) was 8,612 µg/ft^2, which is 11 times higher than the federal guideline, showing the extent of the hidden risk of potential Pb exposure through inhalation of dust.

3.1.3. Pb in produce/plants in outdoor home gardens

Table 2 shows the plant Pb concentrations (mg/kg) and the Pb levels of the adjacent soil in four yard-gardens at sites A, C, H, and M. Tomatoes collected from the garden of Nazareth College of Rochester, NY, which had adjacent SLL below detection limits, were used to calculate the background plant Pb levels of plants from the property sites. Pb concentrations in tomatoes from the Nazareth College garden were subtracted from the Pb levels found in plants from the properties tested to focus on the effect of SLL on the Pb uptake and accumulation in plant tissues. Tomato plants were found in three gardens (sites A, C, and M), and the lead accumulation in tomatoes significantly ($p<0.0001$; F ratio=750.7) increased in a linear manner ($R^2 = 0.99$) with increasing SLL. Site A had several garden beds filled with potting soil which contained soil Pb, but at much lower levels than 400 mg/kg. Currently, there is no existing guideline defining the permissible level of Pb in the soil for gardening. All plants collected from property garden beds, including arugula, lettuce, radish, garlic, tomato, potato, and broccoli, were found to contain high concentrations of Pb, ranging from 1.25 mg/kg in garlic to 4.67 mg/kg in radish (by dry weight). Sites H and M had gardens in locations with higher SLL levels compared to other locations in their respective properties. Consequently, the tomatoes from site M and elephant foot yam from site H exhibited a very high accumulation of Pb. Overall, as shown in figure S.4, the increase in accumulated Pb in plants collected from all four yard-gardens over adjacent SLL followed a polynomial square fit. Equations 1 and 2 present two prediction models that were derived based on the strong correlation between the soil Pb and Pb in plant tissues per dry weight (d.w.) and fresh weight (f.w.) respectively.

$$y = 2.331 - 0.005 * x + 5.74e^{-7} * (x - 812.1)^2 \text{ --------Equation 1}$$

$$y = 2.338 - 0.0008 * x + 6.99\, e^{-8} * (x - 812.1)^2 \text{ --------Equation 2}$$

Where y is Pb concentration in (mg/kg) in plant tissues and x is Pb concentration (mg/kg) in the adjacent soils. Both models were significant ($p < 0.0001$; F ratio = 120.5 and 47.7 respectively) with strong regression coefficients ($R^2 = 0.85$ and 0.89 respectively).

3.2. Chemical immobilization of Pb by WTR

3.2.1. Physico-chemical and TCLP characteristics of WTR

Table 3 presents the relative physicochemical properties and the results of the TCLP of the two batches of WTR. Both exhibited almost neutral pH, high moisture content, and low organic matter content. TCLP levels of the RCRA metals revealed a complete absence of lead and mercury and the presence of other metals in substantially lower concentrations than the criteria set by the EPA. Both physicochemical properties and TCLP data for these two batches of WTR were statistically similar, which was evident in $p > 0.1$ and low F ratio.

3.2.2. Pb kinetic and equilibrium sorption/desorption by WTR

Kinetic sorption experiments showed that the binding of Pb by WTR was rapid and varied with the initial Pb concentrations in solution, while the Pb levels in the control solutions with no WTR remained unchanged (fig. 5). Equilibrium times could not be determined in lower Pb concentrations because of WTR's extremely rapid and strong sorption for Pb, as the sorption reached 100% instantaneously (within 8 minutes of contact time with no shaking) for 25 mg/L and within 5.13 h contact time in 200 mg/L initial Pb concentrations. At 1000 mg/L initial Pb in solution, the Pb sorption by WTR reached equilibrium within 5.13 h contact time and did not exhibit any significant change in sorbed Pb up to 24.13 h contact time. Absolutely no desorption occurred in any of these initial concentrations within 24 h.

The extent of equilibrium sorption of Pb by WTR increased with increasing Pb concentration in solution (fig. 6a), but the percent sorption showed a reverse trend. However, very high sorption affinities (> 90%) were exhibited in all initial loads within 10h (fig. 6d). The data followed an L-type curve (fig. 6a) and showed strong fit to the linearized Freundlich (Eqn. 2; fig. 6b; $R^2 = 0.99$) as well as (Eqn. 3; fig. 6c; $R^2 = 0.98$) Langmuir equations and the associated parameters were calculated accordingly.

$$log\ Q = log\ K_F + (1 / n)\ log\ C_{eq}\text{-------Equation 3}$$

Where K_F (mg/kg) and n are the Freundlich equilibrium constants representing the sorption capacity and adsorption intensity respectively; Q (mg/kg) is the adsorbed amount of Pb by WTR at equilibrium; Ceq (mg/L) is the equilibrium concentration of Pb in solution. The value of K_F was determined as 1.6 mg/g, which exhibited a very high sorption capacity of Pb by WTR. The n value determined in this study was above unity (1.82), indicating favorable adsorption of Pb onto WTR through physical processes (McKay 1980; Özer and Pirinççi 2006).

$$(\mathrm{Ce} / \mathrm{Q}) = (1 / K_L Qm) + (\mathrm{Ce} / Qm)\text{ --------Equation 4}$$

Where Qm (mg/kg) is the maximum adsorption capacity and K_L (L/mg) represents the Langmuir constant related to the energy of adsorption. A good fit to the Langmuir isotherm indicates the monolayer coverage and homogeneous distribution of Pb on the active binding sites of the WTR surfaces with an adsorption maximum of 25 mg/g, that exhibits a very high sorption capacity. Additionally, another Langmuir constant named separation factor (R_L) was calculated following equation 4.

$$R_L = 1 / 1 + K_L C_0\text{--------Equation 5}$$

Where C_0 (mg/L) is the initial Pb concentration. The R_L values indicates the nature and the feasibility of the adsorption process, such as unfavorable sorption if $R_L > 1$, linear sorption if $R_L = 1$, favorable sorption if $0 < R_L < 1$, or irreversible sorption if $R_L = 0$ (Meroufel et al. 2013). The value of R_L calculated for all tested initial concentrations of Pb in this study ranged from 0.03 to 0.15, which were between zero to unity, suggesting favorable sorption of Pb on WTR. In addition, these values were much closer to zero than one, indicating more irreversibility in adsorption of Pb on WTR. This was evident in the equilibrium desorption of Pb from WTR. Figure 6e shows the total desorbed Pb after three desorption cycles (24 h each). No desorption was noted in 72 h up to 600 mg/L initial Pb concentration, indicating complete hysteresis or complete irreversible binding of Pb by WTR at these initial loads. Although with the further increase of initial Pb, the extent of desorption increased; it remained restricted to < 2% of the sorbed Pb in WTR even after 72 h of shaking.

3.2.3. WTR concentration effects on paint chip Pb leaching

As evident from figure S.1, the numerous loose paint chips found in the yard at site A both before and after the renovation of the structure resulted in an extremely high SLL. Locations with buried paint chips exhibited an average of 12 and 5 times higher SLL as compared to the surrounding soil with no paint chips before and after renovation, respectively. The surface soil with both loose and buried paint chips posed a very high risk of Pb exposure through dust loading as well as leaching into the stormwater runoff. A TCLP assessment of these paint chips collected from the yard revealed that they released 47.9 $\pm$ 1.71 mg/L (n=6) of Pb in solution, which is 9.6 times higher than the EPA permissible Pb levels for TCLP. This data highlights the need for determining potential Pb leaching from these paint chips in a stormwater solution and provided us with an ideal opportunity to test the effectiveness of WTR to prevent the mobilization of Pb from the paint chips with extremely high Pb-leaching potential.

The results were highly promising as WTR concentrations significantly ($p<0.0001$ and F ratio = 90.7) reduced Pb leaching from loose paint chips. 43, 61, 84, and 93% prevention of Pb leaching was achieved by 10, 20, 50, and 100% WTR to paint chips (m/m) application after two desorption cycles of 1 and 6 d. Effect of time was masked ($p=0.2$) by WTR's very strong affinity for Pb; no significant difference between the higher WTR treatments suggests the application of WTR in a 1 to 2 ratio to paint chips would be enough to achieve the optimum prevention. As these paint chips exhibited the maximum Pb concentrations and acted as the major contributor of SLL in site A, a lower dose of WTR would be enough for achieving optimum prevention of Pb leaching in soils at lower SSR.

4. Discussion

4.1 Field Assessment

Although the City of Rochester succeeded in reducing the number of children with EBLL (compared to neighboring cities like Buffalo and Syracuse) through its adoption of an aggressive secondary intervention approach, the city continues to face significant issues related to Pb exposure (Magavern 2018). This is consistent with our findings showing that the overall SLL medians across all tested urban and suburban properties with structures built prior to 1978 were

413 and 155 mg/kg, respectively, with mean SLL as high as 3,178 and 461 mg/kg, respectively. This study also documented the highest maximum residential SLL (173,500 mg/kg), as compared to those reported over the last two decades in urban and suburban areas of thirty-one U.S. cities (data compared with 36 research articles; not shown). Risk of Pb exposure through soil Pb, potential dust loading, and produce grown in property gardens was found to be much higher in the urban residential properties where residents already face significant socioeconomic challenges. According to the 2010 Rochester census data, urban neighborhoods in the City of Rochester have the highest percentage of residents living below the poverty line, the lowest percentage of those with above-average income, and the broadest distribution of race with the highest percentage of African American residents when compared to suburban sub-counties. The unjust burden of Pb exposure in urban neighborhoods in which many low-income minority families reside has been documented in twenty studies conducted over the last four decades using data from sixteen U.S. cities (Datko-Williams *et al.* 2014).

The prevalence and correlation of SLL and BLL are well established in the literature (Bickel 2010; Zahran *et al.* 2010; Wu *et al.* 2010). The overall median SLL in urban properties exceeding the federal standard clearly underscores this risk. However, the lower median SLL of suburban properties does not represent a risk-free condition either. Adverse health effects of Pb are associated with BLL as low as 2 mg/dL, which is associated with SLL < 100 mg/kg (Canfield et al. 2003; Schwarz et al. 2012). Two existing prediction models by Mielke et al. (2007) and Johnson and Bretsch (2002) independently showed a steep increase in BLL at SLL <100 mg/kg and a gradual rise at SLL >300 mg/kg. To assess the exposure risk more specifically, we used both of these models (equations 5 and 6) to predict the potential BLL if residents (especially children) are being exposed at tested residential properties.

$BLL = 2.038 + 0.172 * (SLL\ median)^{0.5}$ -------Equation 6 (Mielke *et al.* 2007)

$BLL = 2.097 * Ln(SLL\ median) - 3.6026$ -------Equation 7 (Johnson and Bretsch 2002)

SLL in all but one suburban property was predicted to result in BLL > 5 µg/dL according to equation 7, whereas five out of the seven urban properties and one out of the five suburban properties were predicted to result in BLL > 5 µg/dL according to equation 6 (table 4). We observed that the predicted BLL based on the median SLL of the entire property underestimates the predicted risk for properties with high variation in SLL throughout the property. For instance, as discussed earlier, site A showed a high variation in SLL throughout the property, both before and after renovation. The predicted BLL of the front yard after the renovation was as high as 20.9 µg/dL (equation 6) and 16.1 µg/dL (equation 7), both of which are much higher than the predicted BLL based on the median SLL of the entire property. In such cases, yard-specific predictions would be a more appropriate measure of risk assessment. Racial and socioeconomic distribution in the City of Rochester, along with the SLL and estimated BLL documented in this study, echo the findings of Mielke *et al.* (1999) and Geltman *et al.* (2001), who reported that higher SLL in urban soils result in inequitable EBLL in children of lower-income families, minorities, and recent immigrants. This is also evident from the data shown by Korfmacher (2007) that low-income families in the crescent area in urban Rochester characterized by high poverty, crime, and pre-1950 rental housing correlated with 23.6% of children under 6 years of age having blood lead levels greater than 10 µg/dL.

In accordance with this study, extreme heterogeneous distribution of soil Pb has been documented by other researchers, and has been observed to decrease with increasing distance from three Pb source categories: i) industrial sites or brownfields with residues of historic Pb use (Marshall and Thornton 1993; Wu et al. 2010), ii) areas near highways or roads with intense traffic density (Mielke et al. 2008; Laidlaw 2001; Filippelli et.al. 2005), and iii) aged structures with exterior LBP (USEPA 2000; Litt et al. 2002; and Clark and Knudson 2013). Some studies have reported that the combustion of leaded gasoline is the primary source of Pb in larger urban settings, as noted in Baltimore (Mielke et al. 1983), New Orleans (Mielke et al. 2008), and Indianapolis (Laidlaw 2001; Filippelli et.al. 2005). Clark and Knudson (2013) argue that exterior LBP dominates the distribution of soil Pb in smaller urban communities like Appleton, WI. Our study showed that even in a medium-sized urban community like Rochester, with a history of past industrialized use of heavy metals, the primary source of Pb in all tested sites was exterior LBP from structures built prior to 1978, as none of these sites are located in neighborhoods with high traffic density or at close proximity of any brownfield or superfund site. The building structures can also act as barriers and facilitate atmospheric deposition of Pb (Mielke et al. 1983), leaving a pattern of higher Pb trapping in the street side of the front yard as compared to the back or side yard. A combined effect of preferential trapping and LBP results in a U-shaped pattern of higher SLL near the house and the road, and lower SLL in the middle of the yard (Olszowy *et al.* 1995). Lack of these types of spatial patterns negates the effect of preferential trapping on properties we assessed. Based on the findings of our current study, we argue that even in medium urban communities, leaded exterior paint can be the dominant source of soil Pb if the property is located at a certain distance from the industrial sites, freeways, or dense traffic area. The location and age of the building structure are the two most important parameters that influence which source will dominate the distribution of soil Pb and to what extent in any given community, irrespective of its size.

Nationwide studies conducted by U.S. Housing and Urban Development report that 76% of homes built prior to 1960 have Pb in exterior paint and 83% of pre-1980 housing stock contains Pb in the paint (USHUD 1990, 1995). This study found Pb in exterior paint at varying depths for all tested urban and suburban properties built prior to 1978, supporting the observation of Clark and Knudson (2013) that all communities of similar age and site throughout the US show a similar trend of contamination. SLL can exist around all sides of the structures and extend far into the yard (Clark and Knudson 2013; USEPA 2000, Litt et al. 2002). In all properties, the SLL was observed to be very high alongside the structures and decreased with increasing distance from the homes. In addition, we also noted that in inner-city properties like site A, G, and K, the neighboring structures, adjacent to the tested yards, also acted as prominent sources of Pb. Buried or scattered paint chips in the soil of site A increased the SLL in some locations against the influence of distance.

Pb contamination in residential soil is so widespread that it can be found in unapparent locations, increasing the threat of continual risk of exposure. In accordance with previous studies, we have observed three such unforeseen scenarios: i) high soil Pb in a suburban house with a well-maintained exterior (Clark and Knudson 2013) was observed in site L, which exhibited SLL as high as 3000 mg/kg; ii) high soil Pb in a property with an unpainted exterior was noted in site B, where the SLL surpassed the federal guideline only beneath the windows, a trend also observed by Linton *et al.* (1980); and iii) increase in SLL after renovation and repainting of a house (Clark and Knudson 2013), which is duly noted in site A. Table S.3 presents a comparative SLL, estimated BLL, and potential dust loading values in the three yards of site A that illustrates the detrimental effects of renovation activity on soil Pb.

Dust is a major source of concern for children, as inhalation is a major route of Pb exposure. The respiratory tract is responsible for the absorption of lead particles into the systemic circulation, and respiration and metabolic rates are faster in children, resulting in increased absorption of lead via dust (Menezes-Filho 2018). Two tools were used in this study to estimate the risk of Pb exposure from outdoor dust: i) direct measurement of accumulated dust on the outdoor surfaces, and ii) secondary estimation of potential lead on play surfaces (PLOPS). Overall, accumulated Pb in dust showed significant ($p<0.001$) positive correlation with Pb in paint and negative correlation with depth index, suggesting flaking LBP on surfaces such as window sills, doors, steps, and old building materials can potentially contribute Pb to dust.

Lead loading on exterior surfaces has been found to be a continuous process, primarily due to resuspension of soil Pb (Caravanos *et al.* 2006; Mielke *et al.* 2006). Researchers identified the resuspension of soil Pb during dry seasons as the major driving force behind the seasonally EBLL in children in New Orleans, Syracuse, and Indianapolis (Laidlaw *et al.* 2005). Mielke et al. (2006) developed a new tool for measuring the potential Pb surface loading per area (mg/ft^2) of the soil. Potential Pb on play surfaces, a measure of the Pb concentration of a soil surface per surface area as a source of Pb dust for children who would play on that surface, was calculated based on a prediction model created by Mielke et al. (2006) (equation 8).

$$PLOPS = -43.74 + 24.85 * (SLL\ median)^{0.69} \text{ -------Equation 8}$$

The PLOPS values for all urban and suburban properties are presented in table 5 to provide insight into the overall potential of Pb exposure from these surfaces, both directly through hand-to-mouth activity, or indirectly through resuspension of Pb dust. Mielke et al. (2006) also showed a large drop in PLOPS in 136 properties and vacant lots in New Orleans after the contaminated surfaces were treated with a soil cover. Similar measures should be considered for the properties assessed in our study, as the large PLOPS values indicate a high potential risk of Pb exposure to children if they play in these areas.

The relationship between Pb in produce and the adjacent soil Pb is equivocal in the literature; with some studies reporting maximum Pb levels in vegetables associated with the highest SLL (Bielinska 2009; Huang *et al.* 2012; Moir and Thornton 1989) and other studies finding only weak correlations between crop Pb levels and the adjacent soil Pb, due to unverified reasons (Hough *et* *al.* 2004; Murray *et al.* 2011; McBride *et al.* 2014). Our study found strong and significant ($r = 0.92$ based on fresh weight and $r = 0.85$ based on dry weight; $p < 0.001$) correlations between the plant-accumulated Pb and SLL. Ten out of fourteen edibles tested exceeded the health-based guidance values set by the European Union (EU) (EC 2006). There are no such standards currently in the U. S., but the median Pb in edible plants found in this study was 72 times higher than the median market basket concentrations of Pb reported by the U. S. Food and Drug Administration's (FDA) Total Diet Study. (USFDA 2010). The FDA recommended the maximum ingestion lead level for a child is 3 µg/day; eating between four and five grams of contaminated produce would reach that limit (Frank 2019). Interestingly, Pb levels exceeding the EU guideline standards were found in plants grown in both indigenous soils (sites H and M) of these properties as well as potting soils (site C and side yard of site A). The median Pb in plants grown in indigenous soils (1.05 mg/kg) was higher than those grown in the potting soil (0.21 mg/kg) and showed a stronger correlation with the Pb concentrations in adjacent soils ($r = 0.91$ and 0.69 for indigenous and potting soils respectively). However, it is deeply concerning that the edible plants grown in potting soils containing Pb lower than 60 mg/kg were found to contain Pb levels exceeding the health guidelines. The federal standard of 400 mg/kg soil Pb provides a misconception of safety for soils containing lower Pb concentrations; there are no current levels set by either U.S. Environmental Protection Agency (EPA) or HUD for urban gardening. Moreover, each species of produce has different biological properties that determine their uptake, absorption, and bioavailability of Pb, demonstrated by the varying lead concentrations in them (USFDA 2010). Differential uptake of Pb by different vegetables at similar SLL was noted in the current study (site A) in Rochester as well as by McBride *et al.* (2014) in New York City and Buffalo, NY; and Misenheimer et al. (2018) in Puerto Rico. These findings strongly support the development of new plant-specific federal guidelines for soil Pb to promote safer gardening practices.

4.2 Safer use of yards by primary prevention tools

Based on results of the field assessment, we developed three primary prevention tools to help homeowners understand the Pb exposure risks throughout their properties as the first step of primary prevention: i) a prediction model to calculate soil Pb using the measured distance from the structure; ii) a prediction model to estimate Pb in edible produce grown in yard gardens from the adjacent soil Pb that can be calculated from the distance from the built structure, iii) GIS maps showing Pb hot spots with high risk, low risk, and safe areas in the yards. Prediction models and GIS maps presented by prior research, covering a larger city area, are crucial for bringing necessary changes in the policy (Wu et al. 2010), but this is an ongoing, long-term process. Considering the continual Pb exposure and elevated BLL in families with lower socioeconomic standing, it is of utmost importance to develop an easier process for the homeowners to understand the risk to be able to avoid it. To the best of our knowledge, the present study is the first attempt to develop site-specific prediction models and precise GIS maps illustrating the potential risk through mapping the SLL in the exact latitude and longitude of the sampling locations on individual residential properties. The GIS maps also mark the safer locations to create gardens based on the two prediction models we developed. These measures would provide the homeowners with an opportunity to predict soil Pb levels to plan safer uses of their yards.

City-wide larger models to calculate soil Pb based on distance from urban centers of big cities or roads with high traffic density considered the atmospheric deposition of Pb as the dominant source (Yaylah-Abanuz 2019; Brown et al. 2008; Zhang et al. 2015; Wu et al. 2010). However, residential soil Pb with exterior LBP as the major source varies widely from one property to another, depending on the age of the structure(s), presence and types of siding, past land use, and renovation processes. Site-specific prediction models would provide homeowners with an opportunity to predict soil Pb levels using the distance from a structure to utilize their yards more safely. For instance, a play area should be planned in soil that contains less than 400 mg/kg Pb, according to federal guidelines. Based on our site-specific models, SLL drops from 400 mg/kg to < 100 mg/kg if the play area is located 2 m further from the house. Considering the steep increase of BLL between 100 to 300 mg/kg SLL, simply moving a play area a few meters further from the structure would increase the chance of primary prevention at no cost. Although we highly recommend at least a one-time assessment of soil Pb for each property, it is likely that one prediction model would be valid for one block or neighborhood if the homes were built at a similar time; however, further investigation of this assumption is necessary.

This study also presents prediction models exhibiting a strong correlation ($r = 0.85$ and 0.92) between Pb accumulation (per d.w. and f.w.) in produce/plant parts and the Pb levels in the adjacent soil. One limitation of these prediction models is that they will not be able to provide any insight into the plant-specific differential Pb uptake, as the pronounced effect of soil Pb may have masked the effects of plant-specific uptake in the current study. However, utilization of these equations would be able to provide a rough estimate of the potential Pb accumulation into vegetables from the adjacent SLL to estimate a safer location for a garden. For instance, the vegetable gardens in sites M and H were located in the areas with the highest SLL, resulting in median Pb levels in edible parts of 1.58 and 0.56 mg/kg, respectively, which exceed the EU guideline by 15.8 and 5.6 folds, respectively (EC 2006). Both of these properties have areas with much lower soil Pb further into the yard. Simply moving these gardens to such locations would be a useful measure of intervention to reduce the Pb exposure risk through the consumption of Pb-contaminated vegetables from the property gardens. We estimated an SLL value associated with the EU guideline of 0.1 mg/kg Pb per plant f.w. using a reverse fit of equation 2 (R^2 = 0.97, $p<0.0001$) which associates with a SLL of 64.7 mg/kg; a slightly lower value of 0.09 mg/kg Pb per plant f.w. associates with an SLL of 58.2 mg/kg. Using the reverse fit of the site-specific models, we calculated the associated distance with 58.2 mg/kg soil Pb in site M and site H, and these distances are found to be 6.44 and 3.85 m respectively. Thus, our combined models indicate that to attain a minimum requirement for the primary prevention, a home garden should be set up at least 6.44 m away in site M and 3.85 m away in site H from their respective structures. Similarly, a safer distance can be calculated for any property garden to avoid consumption of Pb through homegrown vegetables.

Figure 8 presents two reference GIS maps of site A showing the Pb exposure risk associated with soil Pb distribution in three areas of the property before and after the renovation of the home. To simply and clearly communicate risk levels to homeowners, we developed a visual tool utilizing a graded color scheme of green to black representing risk levels. Table S.4 presents a guideline created for the homeowners to explain each color category, associated risk, and the recommended use of that location on their properties. In addition to using the federal standards for assigning colors to the range of Pb concentrations, we have also used our prediction models to mark areas of the property where it would be safer to create a garden based on the EU health guidelines for edible produce. The areas marked in red to black colors represent hotspots with very high to extreme risk of potential Pb exposure and thus must be avoided and should be considered for remedial actions.

4.3 Sustainable Remediation

The second and final step of primary prevention is to correct the hazards through remedial actions, which led us to investigate an affordable process that would provide substantial risk reduction without economically burdening the homeowners. WTR collected from Rochester's largest drinking water treatment plant not only exhibited a very high Pb-binding ability but was also found to be safe when used as a soil amendment according to the federal TCLP guidelines. Past research documented varying effectiveness of different sorbents to immobilize lead, including activated carbon (Moyo et al. 2013), natural and engineered clay-like zeolite (Wingenfelder et al. 2005), mordenite (Turkyilmaz et al. 2014), montmorillonite (Zhang and Hou 2008), bentonite (Inglezakis et al. 2007), palygorskite, and sepiolite (Shirvani et al. 2006). Although these technologies provide *in-situ* alternatives to the current expensive *ex-situ* practices, the cost of these natural or engineered sorbents in bulk could become cost-prohibitive to low-income urban families. WTR is a no-cost sorbent which would provide these homeowners with an affordable remediation alternative to be utilized as soil cover or amendment in the most concerning areas of their properties. The only required cost is associated with its implementation, which could be further reduced by conducting a proper assessment to mark the risk hotspots (as shown in the GIS maps for site A), where the remediation process is absolutely needed. Very few studies have investigated the effectiveness of Pb adsorption by Fe and Al-based WTR collected from different parts of the world. Pb sorption by Al-WTR showed a similar pattern of L type sorption isotherm (Castaldi et al. 2015; Zhou et al. 2011). Adsorption maximum was reported to be 17.55, 15.66, and 15.13 mg/g by WTRs collected from Miyamachi and Nishino in Japan and Italy, respectively, as compared to 24.4 mg/g found in this study (Putra and Tanaka 2011). Furthermore, Soleimanifer et al. (2019) reported the unique approach of reducing metal contamination in urban stormwater with WTR coated mulch; a similar approach could be further investigated for its potential to reduce plant-Pb uptake if used in home gardens. The sorption desorption data along with the calculated Freundlich and Langmuir isotherm parameters suggests very strong, favorable, and more irreversible adsorption of Pb on WTR through a physical process, promising an effective option for reusing this waste byproduct that the city generates every day to immobilize Pb in the high-risk soils of many contaminated residential properties.

Using WTR as a soil amendment will prevent the runoff and leaching of soil Pb and also decrease the Pb available to be accumulated in garden vegetables. However, the implementation of only WTR will not prevent the resuspension of Pb into dust. Our goal is to implement a dual-process remediation system using WTR as a soil amendment with the addition of a plant cover to decrease the dust loading of Pb. Several studies have shown the phytoremediation potential of Pb (Jagetiya and Kumar 2020), but it is of utmost importance to use a non-edible plant in the Pb hot spot areas to prevent its further entry into the food chain. Vetiver, a perennial non-edible and non-invasive grass has been shown to be highly effective for Pb removal (Andra et al. 2009); however, the harsh winters of western NY could be a serious impediment for its implementation. Our group has tested 17 native inedible plant species for their potential tolerance of high Pb concentration, but all plants showed severe phytotoxic effects of Pb in hydroponic media, where 100% Pb was available for plant uptake (data not shown). This further underscores the utility of WTR for Pb removal, which according to the results of this study would retain more than 90% of total soil Pb, making only a small fraction available for the plants to tolerate. The ongoing studies in our greenhouse are characterizing this dual process technique using WTR as an amendment to the soils from the backyard of site A, with an SLL median of 2900 mg/kg, and switchgrass (*Panicum virgatum*), high biomass, perennial, fast-growing, inedible grass, which is also native to western NY and which was previously studied for its ability to remove other environmental contaminants (Phouthavong-Murphy 2020). This combined remediation process would provide the homeowners with an effective and affordable primary prevention alternative to intervene with the Pb exposure from the risk hotspots of their property.

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Author contributions

P.D. designed the research, field assessment, and experiments. P.D., D.R.B., M.J.P, B.E.L., A.C.T., J.M.P., H.M.C., J.L.W., I.G., and D.G. conducted the field assessment; B.E.L. and A.C.T analyzed Pb in surface dust, J. S., C.V.B., M.L.M., D. R. B., M.J.P., C.E.D. analyzed Pb in plants with the supervision of S.M.Z. and P.D.; J.L.W., I.G., J.P.M., J.M.P., H.M.C., A.C.T. conducted phase II experiments with the supervision of S.M.Z. and P.D; P.D. conducted the secondary analyses, isotherm modeling, statistical data analyses, and developed the prediction models. M.C. created the GIS maps and D.R.B. created the Tableau maps. P.D. wrote the manuscript with S.M.Z. and J.P.M. with contributions and feedback from all other authors.

Abbreviations:

Pb: lead; WTR: drinking water treatment residual; XRF: x-ray Fluorescence Analyzer; Al-WTR: aluminum-based water treatment residual; SLL: soil lead level; mg/kg: milligram per kilogram; WTR: water treatment residual; GIS: geographical information systems; LBP: lead based paint; EBLL: elevated blood lead level; BLL: blood lead level; $\mu g/dL$: micrograms per deciliter; USEPA: United States Environmental Protection Agency; USHUD: United States Department of Housing and Urban Development; n: number of samples; HUD: Department of Housing and Urban Development; TCLP: toxicity characteristic leaching procedures; SSR: solid to solution ratio; KCl: potassium chloride; mg/L: milligrams per liter; HNO3: nitric acid; AAS: atomic absorption spectrophotometer; RCRA metals: 8 metals (Arsenic, Barium, Cadmium, Chromium, Lead, Mercury, Selenium, Silver) listed under Resource Conservation and Recovery Act; ICP-OES: individually coupled plasma optical emission spectroscopy; Tukey-Kramer HSD: Tukey Kramer honest significant difference.

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Table 1: Polynomial cubic fit of lead concentrations (mg/kg) in residential soil by distance from potential source of Pb based paint. The nonlinear correlation fit, regression equation, and parameter estimates were conducted using JMP (version 15). Parameters of the regression equations were estimated and the significance levels of the parameters were expressed as * ($p < 0.05$), ** ($p < 0.01$), *** ($p < 0.001$), **** ($p < 0.0001$).

Site	R^2	F ratio	p value	Regression Equations	Significant Parameters
A1	0.9994	632.9	0.0292	$y = 3684 - 575.9 * x + 89.45 * (x - 4.321)^2 - 2.277 * (x - 4.321)^3$	Intercept**, x^{2**}
A3	0.9366	14.78	0.0266	$y = 2141 - 1307 * x + 8370 * (x - 1.045)^2 - 5519 * (x - 1.045)^3$	x^{2*}
A4	0.9868	50.02	0.0197	$y = 2226 - 282.6 * x + 126.5 * (x - 3.569)^2 - 28.10 * (x - 3.569)^3$	x^{2*}
L1	0.9567	22.13	0.0151	$y = 388.3 - 249.5 * x + 497.1 * (x - 2.199)^2 - 108.1 * (x - 2.199)^3$	x^{2*}, x^{3*}
L2	0.8593	18.32	0.0004	$y = 420.3 - 170.1 * x + 277.5 * (x - 2.110)^2 - 61.83 * (x - 2.110)^3$	x^{2***}, x^{3*}
M	0.8402	21.03	<0.0001	$y = 394.3 - 68.31 * x + 135.8 * (x - 3.029)^2 - 28.03 * (x - 3.029)^3$	x^{2***}, x^{3**}
H	0.9820	218.5	<0.0001	$y = 752.9 - 220.8 * x + 127.8 * (x - 2.286)^2 - 22.73 * (x - 2.286)^3$	Intercept****, x^{****}, x^{2****}, x^{3*}
C1	0.9607	32.63	0.0029	$y = 232.8 - 183.5 * x + 486.1 * (x - 1.181)^2 - 246.8 * (x - 1.181)^3$	x^{2**}
C2	0.9868	150.4	<0.0001	$y = 153.3 - 71.01 * x + 362.7 * (x - 1.371)^2 - 268.6 * (x - 1.371)^3$	x^{2****}, x^{3**}
B	0.9865	97.77	0.0003	$y = 1175 - 713.6 * x + 105.9 * (x - 1.124)^2 - 150.5 * (x - 1.124)^3$	Intercept***, x^{***}, x^{3*}
G1	0.8887	13.31	0.0081	$y = 361.8 - 71.36 * x + 97.78 * (x - 1.439)^2 - 41.25 * (x - 1.439)^3$	Intercept*, x^{2*}
G2	0.9848	86.41	0.0004	$y = 508.9 - 71.63 * x + 13.23 * (x - 3.458)^2 - 0.4656 * (x - 3.458)^3$	Intercept***, x^{**}, x^{2**}

Table 2. Pb concentrations in produce or parts of plants (mg/kg) grown in the gardens of the tested residential properties. Data are expressed as mean (n=3) $\pm$ one standard deviation. Mean comparison is conducted by one-way ANOVA (F ratio = 121.4; $p < 0.0001$) followed by Tukey-Kramer honest significant difference test. Mean comparison letters are expressed as superscript and levels not connected by same letters are significantly different from one another.

Produce/Plants	Parts of plants analyzed	Site	Pb concentrati on in soil (mg/kg)	Pb concentration in plants by dry weight (mg/kg)	Pb concentration in plants by fresh weight (mg/kg)	Guidance Value (EC, 2006)
Tomato (*Solanum lycopersicum*)	Tomato (fruit)	Garden bed in C	8.5	0 ± 0[G]	0.02 ± 0.009[I]	0.1
Arugula (*Eruca vesicaria*)	Leaves	Garden bed 1 in A	11.2	3.52 ± 0.22 [DE]	0.19 ± 0.016[H]	0.3
Lettuce (*Lactuca sativa*)	Leaves	Garden bed 8 in A	28.3	2.98 ± 0.10 [DE]	0.01 ± 0.005[I]	0.3
Radish (*Raphanus sativus*)	Radish (root)	Garden bed 8 in A	28.3	4.67 ± 0.33 [DE]	0.08 ± 0.014[I]	0.1
Garlic (*Allium sativum*)	Garlic (bulb)	Garden bed 7 in A	30.4	1.25 ± 0.02 [F]	0.34 ± 0.004[G]	0.3
Tomato (*Solanum lycopersicum*)	Stem	Garden bed 7 in A	30.4	1.97 ± 0.04 [F]	0.37 ± 0.006[G]	0.3
Potato (*Solanum tuberosum*)	Root	Garden bed 2 in A	32.6	1.88 ± 0.07 [F]	0.23 ± 0.008[H]	0.1
Broccoli (*Brassica oleracea* var. *italica*)	Leaves	Garden bed 5 in A	52.8	1.79 ± 0.07 [F]	0.48 ± 0.012[EF]	0.3
Ground Ivy (*Glechoma hederacea*)	Whole plant	Front yard in A	7400	16.07 ± 0.37 [A]	3.35 + 0.59[A]	N/A
Elephant Foot Yam (*Amorphophallus paeoniifolius*)	Whole plant	Front yard garden in H	358.7	3.42 ± 0.13 [E]	0.34 ± 0.013[G]	0.1
Elephant Foot Yam (*Amorphophallus paeoniifolius*)	Whole plant	Front yard garden in H	511.4	4.42 ± 0.01 [DE]	1.05 ± 0.003[D]	0.1
Elephant Foot Yam (*Amorphophallus paeoniifolius*)	Whole plant	Front yard garden in H	663.3	4.89 ± 0.08[D]	0.55 ± 0.008[E]	0.1
Elephant Foot Yam (*Amorphophallus paeoniifolius*)	Whole plant	Front yard garden in H	998.9	4.54 ± 0.14[DE]	0.40 ± 0.013F[G]	0.1
Elephant Foot Yam (*Amorphophallus paeoniifolius*)	Whole plant	Front yard garden in H	1042	10.89 ± 0.56[B]	1.22 ± 0.054[C]	0.1
Tomato (*Solanum lycopersicum*)	Tomato (fruit)	Back yard garden in M	985.1	8.40 ± 0.30[C]	1.59 ± 0.056[B]	0.1

Table 3. Physicochemical characteristics and TCLP of WTR collected from lagoon 1 in 2017 and lagoon 2 in 2018. Data are expressed as mean (n = 3) ± one standard deviation. Mean comparison is conducted between WTR collected from different space and time.

					TCLP RCRA Metals (mg/L)							
	Moisture Content (%)	**pH**	**Electrical Conductivity (μs/cm)**	**Organic matter content (%)**	**Ba**	**Ag**	**As**	**Cr**	**Pb**	**Cd**	**Se**	**Hg**
WTR from Lagoon 1 collected in 2017	48.32 ± 1.997	6.99 ± 0.064	371.8 ± 38.09	2.195 ± 0.166	1.157 ± 0.0102	0.001 ± 0.000	0.086 ± 0.023	0.033 ± 0.006	0 ± 0	0.003 ± 0.001	0.036 ± 0.006	0 ± 0
WTR from Lagoon 2 collected in 2018	47.99 ± 1.962	6.98 ± 0.035	450.3 ± 52.50	2.259 ± 0.157	1.047 ± 0.021	0.001 ± 0.000	0.098 ± 0.035	0.04 ± 0.001	0 ± 0	0.002 ± 0.001	0.036 ± 0.006	0 ± 0
EPA permissible Limit					100	5	5	5	5	1	1	0.2
F ratio	0.042	0.155	4.398	0.235	3.381	NA	0.273	3.811	NA	0.500	0.018	NA
p value	0.848	0.714	0.104	0.653	0.139	NA	0.629	0.123	NA	0.519	0.899	NA

Table 4. Percentile, mean, standard deviation (SD) of soil lead level (SLL) (mg/kg) and potential blood lead level (BLL) (μg/dL) in children and potential lead on play surfaces (PLOPS) (μg /ft^2) using three prediction models developed by Mielke et al., (2007), Mielke et al., (2006), and Johnson and Bretsch (2002) respectively.

	Percentile SLL (mg/kg)					**SLL Statistics (mg/kg)**			**Potential BLL (μg/dL)**		**PLOPS (μg /ft^2)**
	Max	75%	**Median**	25%	Min	Mean	SD	N	Equation 6	Equation 7	Equation 8
Urban											
A after Renovation	173500	7346	**1665**	455	57	9619	25045	79	**9.1**	**12**	4107
A before Renovation	50000	6811	**963**	257	82	6857	11768	42	**7.4**	**10.8**	2800
C	1553	992	**578**	144	11	619	469	58	**6.2**	**9.7**	1956
M	3743	990	**442**	253	58	707	759	36	**5.7**	**9.2**	1617
H	2378	530	**357**	181	112	502	490	45	**5.3**	**8.7**	1389
G	1200	427	**276**	198	75	328	206	54	**4.9**	**8.2**	1156
B	1077	504	**187**	131	65	329	326	16	**4.4**	**7.4**	874
K	847.7	248	**109**	82	32	190	180	43	**3.8**	**6.2**	589
Suburban											
L	3088	1099	**558**	201	34	753	720	45	**6.1**	**9.7**	1909
I	1295	150	**116**	80	65	202	301	16	**3.9**	**6.4**	615
J	115.7	97	**82**	47	37	76	28	8	**3.6**	**5.6**	478
E	248.4	63	**45**	35	25	66	61	15	**3.2**	**4.4**	301
Overall											
Urban	173500	1073	**413**	185	11	3178	12768	373	**5.5**	**9.0**	1542
Suburban	3088	711	**155**	66	25	461	627	84	**4.2**	**7.0**	763

Figure 1. Field assessment site maps with respect to number of pre-1978 built structures (a), racial diversity (b), education (c), and annual household income (d) of the sub-counties. These maps were created using Tableau software, which developed a geographical map of Monroe and Ontario County utilizing latitude and longitude coordinates for each sub-county boundary. In subfigure 1a, the size of the bubble depicts the number of pre-1978 houses found within that sub-county, and the color of the bubble denotes the respective sub-county. In subfigure 1b, 1c, and 1d, the distribution of racial diversity, education, and annual income data in their respective categories was represented by overlaying pie-charts into each sub-county.

Figure 2. Effect of distance (m) from potential source of lead-based paint on the lead concentrations (mg/kg) of residential soils in the urban or suburban neighborhoods of Rochester, NY; Data are expressed as mean (n = 20) $\pm$ one standard deviation. Each subfigure expressed data for one residential property except site A, which was expressed in two subfigures (a, b). The subfigures are arranged left to right in rows based on their decreasing Y axis values, indicating the magnitude of soil Pb, of which the maximum was recorded in the front yard of site A after renovation (a) followed by the back yard of site A (b) before and after renovation; site L (c); site M (d); site H (e); site C (f), site B (g); site G (h); and site K (i).

Figure 3. SLL measurements in the backyard of site G showing two assessment lines; G1 (pink line), which was measured from the corner of the garage at site G, showed a decrease in soil Pb with distance except on one location (X), which was found to be horizontally aligned to the corner of the neighbor's house, indicating it to be an additional source of LBP (fig. S.2). The line G2 was measured from a point adjacent to the property line between these two houses and was horizontally aligned to the corner of the neighbor's house. G2 also showed a continual decrease with distance in the soil Pb concentration, except the increase at the last measured location point, which is the closest to the main house in site G, suggesting an additional source of LBP. The photograph is included in the manuscript with the permission of the homeowner.

Figure 4. Multivariate correlation among Pb in outdoor dust, paint and the depth index indicating how deep Pb is in the paint from a window (a), door (b), and front stairs (c) of site G; discarded old house materials from site A (d); pillars from the front porch in site H (e); and discarded old house materials left in a barn at site E (f).

Figure 5. Kinetic sorption (%) of Pb in presence or absence of WTR. Data are expressed as mean (n = 3) $\pm$ one standard deviation. Mean comparison is conducted by two-way ANOVA followed by Tukey-Kramer honest significant difference test. It is conducted separately for each initial Pb concentration. Levels not connected by same letters are significantly different.

Figure 6. Equilibrium sorption and desorption of Pb in 5% WTR. Linear (a), Freundlich (b), and Langmuir (C) isotherms of Pb-sorption by WTR in 24h. Data are expressed as mean (n = 3) $\pm$ one standard deviation. % sorption (d) and % total desorption (e) of Pb by WTR in three desorption cycles of 24h each. Desorption of Al (mg/kg) from WTR during sorption of Pb (f). Mean comparison is conducted by one-way ANOVA followed by Tukey-Kramer honest significant difference test. Levels not connected by same letters are significantly different.

Figure 7. Effect of WTR concentrations on the leaching of lead (µg/L) from the paint chips collected from site A. Data are expressed as mean (n = 3). Mean comparison is conducted by two-way ANOVA followed by Tukey-Kramer honest significant difference test. Levels not connected by same letters are significantly different.

Figure 8: GIS maps showing distribution of SLL (mg/kg) in site A before (8a) and after (8b) the restoration/repainting of the built structure. These maps were created using ArcGIS Pro 2.5.2 by plotting the SLL (mg/kg) measurements into their respective latitude and longitude. An approximate property-line (dotted line) was overlaid to show the boundaries of the property segregating it from the side walk in south and west and three built structures of neighboring property in north and east. Based on different guidelines for safety, 7 sizes and color categories were used, each representing a range of SLL.

Figure 1:

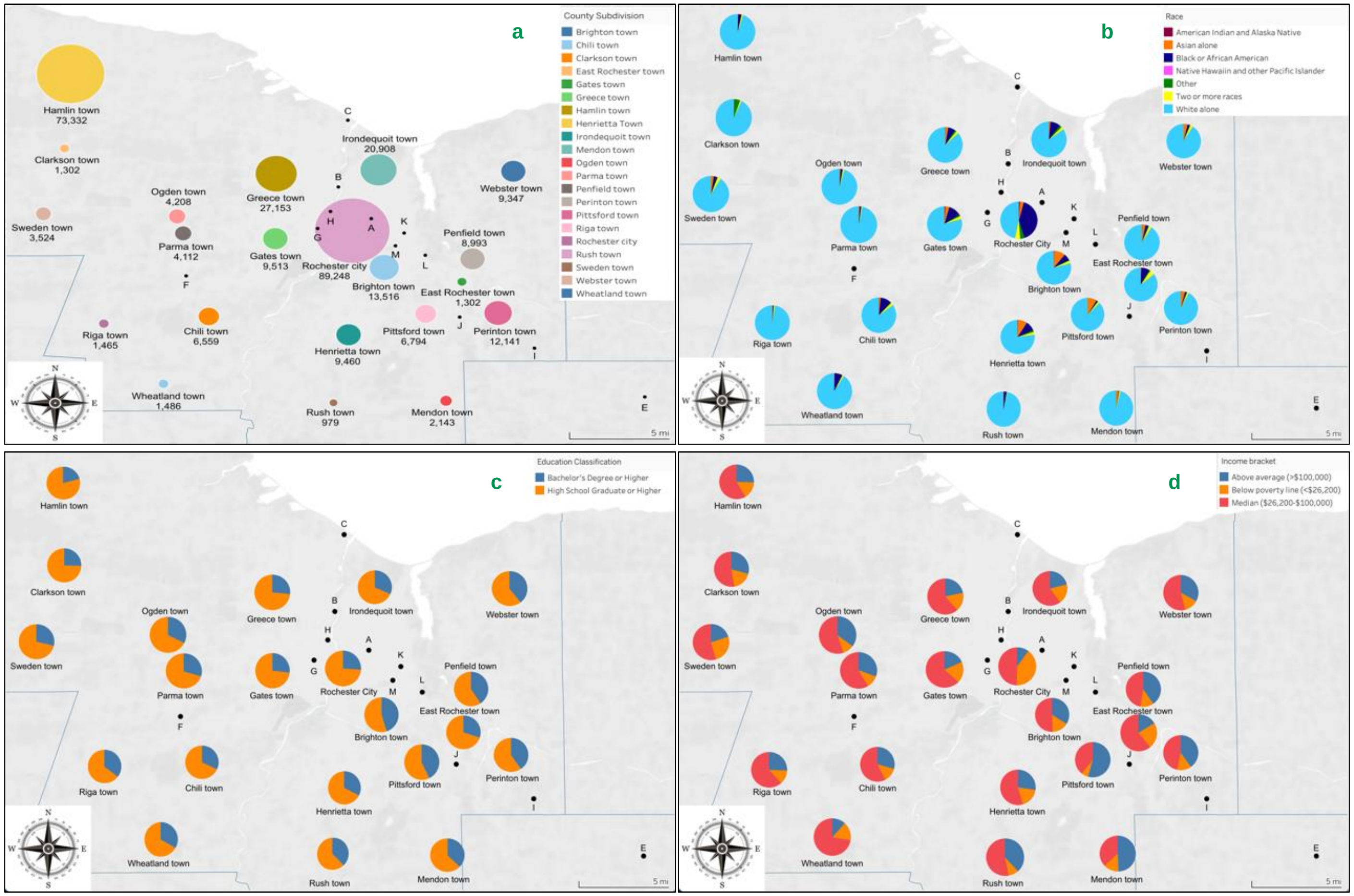


Figure 2:

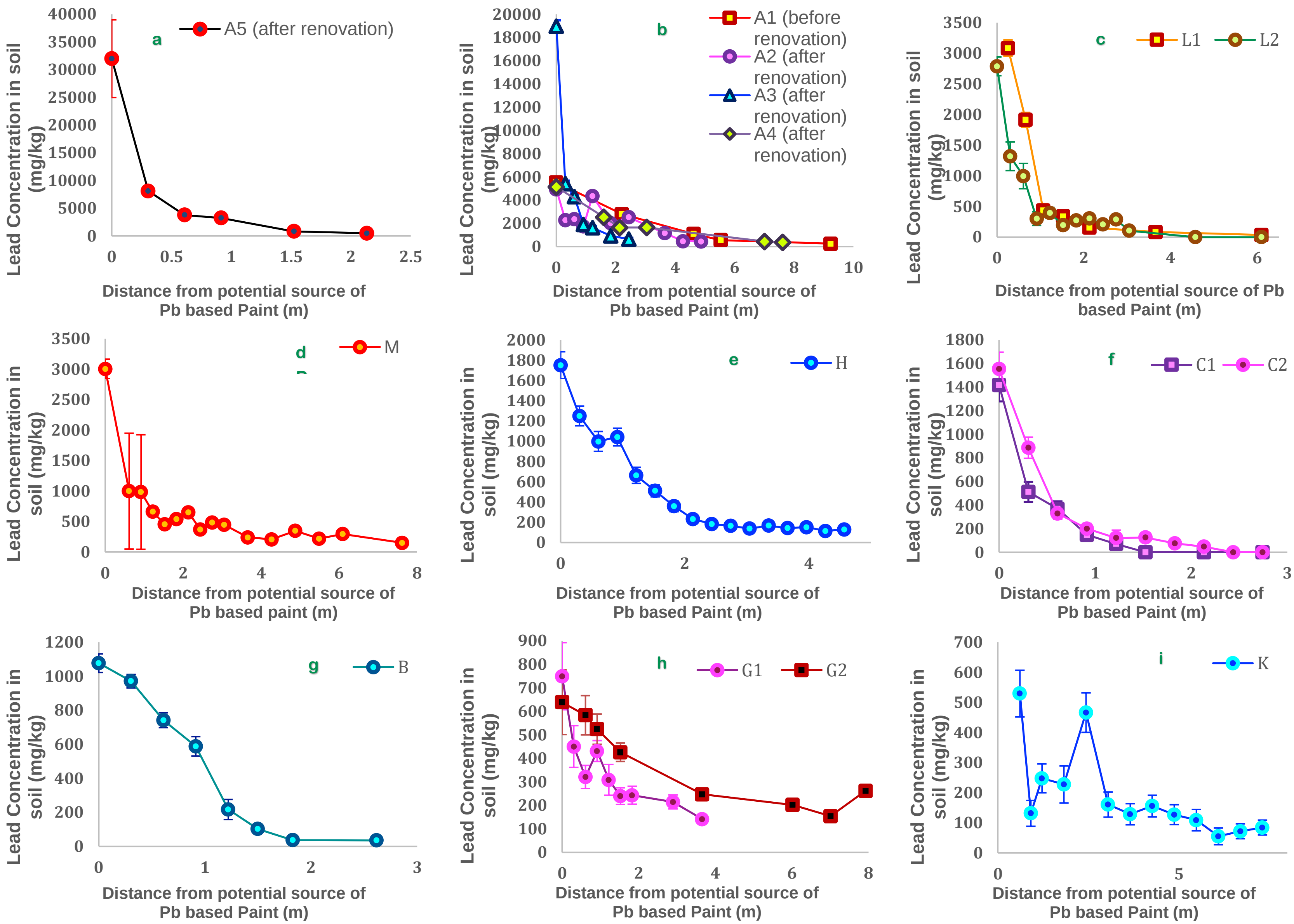
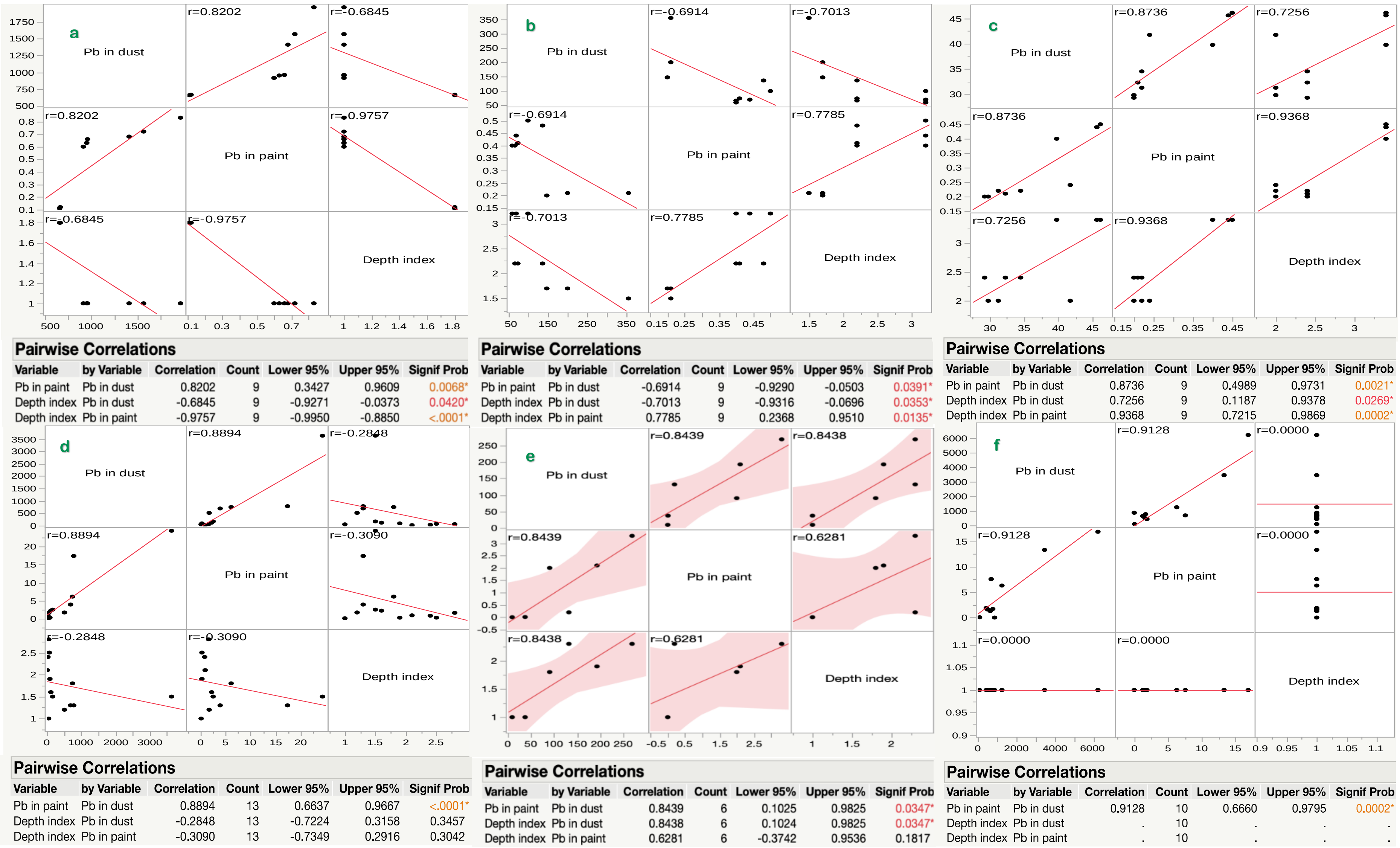

a
A5 (after renovation)
Lead Concentration in soil (mg/kg)
Distance from potential source of Pb based Paint (m)
b
A1 (before renovation)
A2 (after renovation)
A3 (after renovation)
A4 (after renovation)
c
L1
L2
Distance from potential source of Pb based Paint (m)
d
M
Distance from potential source of Pb based paint (m)
e
H
f
C1
C2
g
B
h
G1
G2
i
K

Figure 3:

Figure 4:



a — Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.8202	9	0.3427	0.9609	0.0068*
Depth index	Pb in dust	-0.6845	9	-0.9271	-0.0373	0.0420*
Depth index	Pb in paint	-0.9757	9	-0.9950	-0.8850	<.0001*

b — Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	-0.6914	9	-0.9290	-0.0503	0.0391*
Depth index	Pb in dust	-0.7013	9	-0.9316	-0.0696	0.0353*
Depth index	Pb in paint	0.7785	9	0.2368	0.9510	0.0135*

c — Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.8736	9	0.4989	0.9731	0.0021*
Depth index	Pb in dust	0.7256	9	0.1187	0.9378	0.0269*
Depth index	Pb in paint	0.9368	9	0.7215	0.9869	0.0002*

d — Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.8894	13	0.6637	0.9667	<.0001*
Depth index	Pb in dust	-0.2848	13	-0.7224	0.3158	0.3457
Depth index	Pb in paint	-0.3090	13	-0.7349	0.2916	0.3042

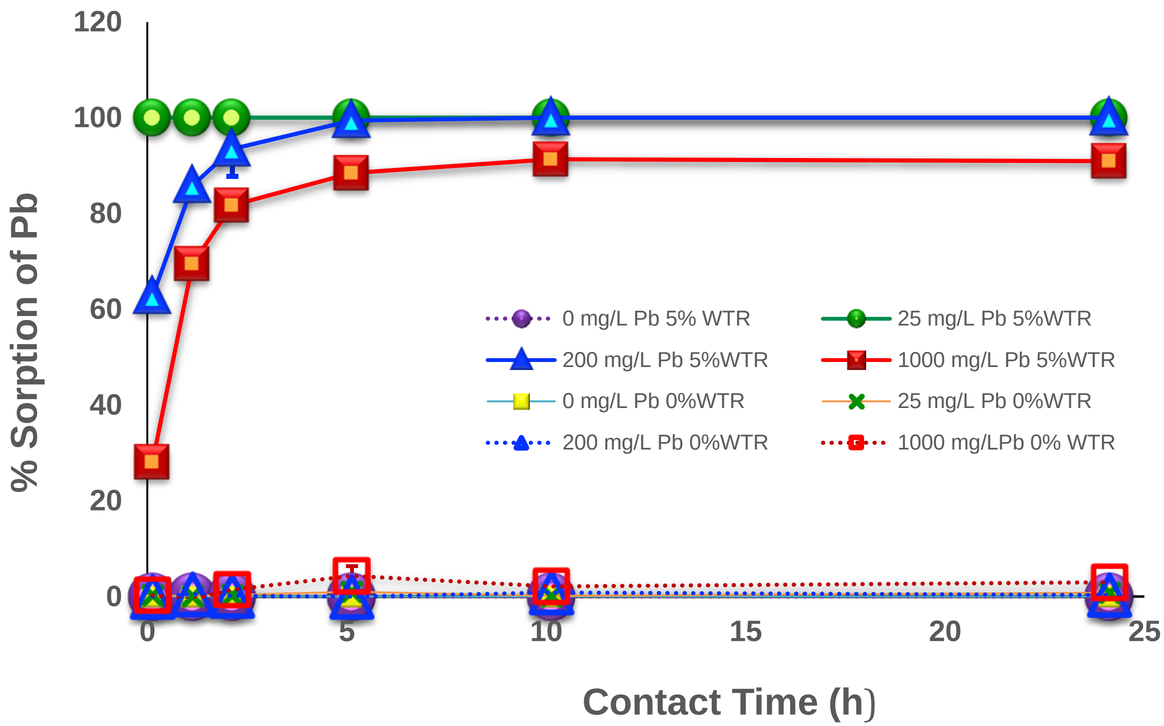
e — Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.8439	6	0.1025	0.9825	0.0347*
Depth index	Pb in dust	0.8438	6	0.1024	0.9825	0.0347*
Depth index	Pb in paint	0.6281	6	-0.3742	0.9536	0.1817

f — Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.9128	10	0.6660	0.9795	0.0002*
Depth index	Pb in dust	.	10	.	.	.
Depth index	Pb in paint	.	10	.	.	.

Figure 5.



Effect Tests Initial Pb: 25 mg/L

Source	Nparm	DF	Sum of Squares	F Ratio	Prob > F
Contact time (h)	5	5	3.786	1.3827	0.2659
%WTR	1	1	14908.141	27222.30	<.0001*
%WTR*Contact time (h)	5	5	1.893	0.6913	0.6349

Effect Tests Initial Pb: 220 mg/L

Source	Nparm	DF	Sum of Squares	F Ratio	Prob > F
Contact time (h)	5	5	1.6720	0.0924	0.9926
%WTR	1	1	5905.4378	1632.648	<.0001*
%WTR*Contact time (h)	5	5	1544.7630	85.4146	<.0001*

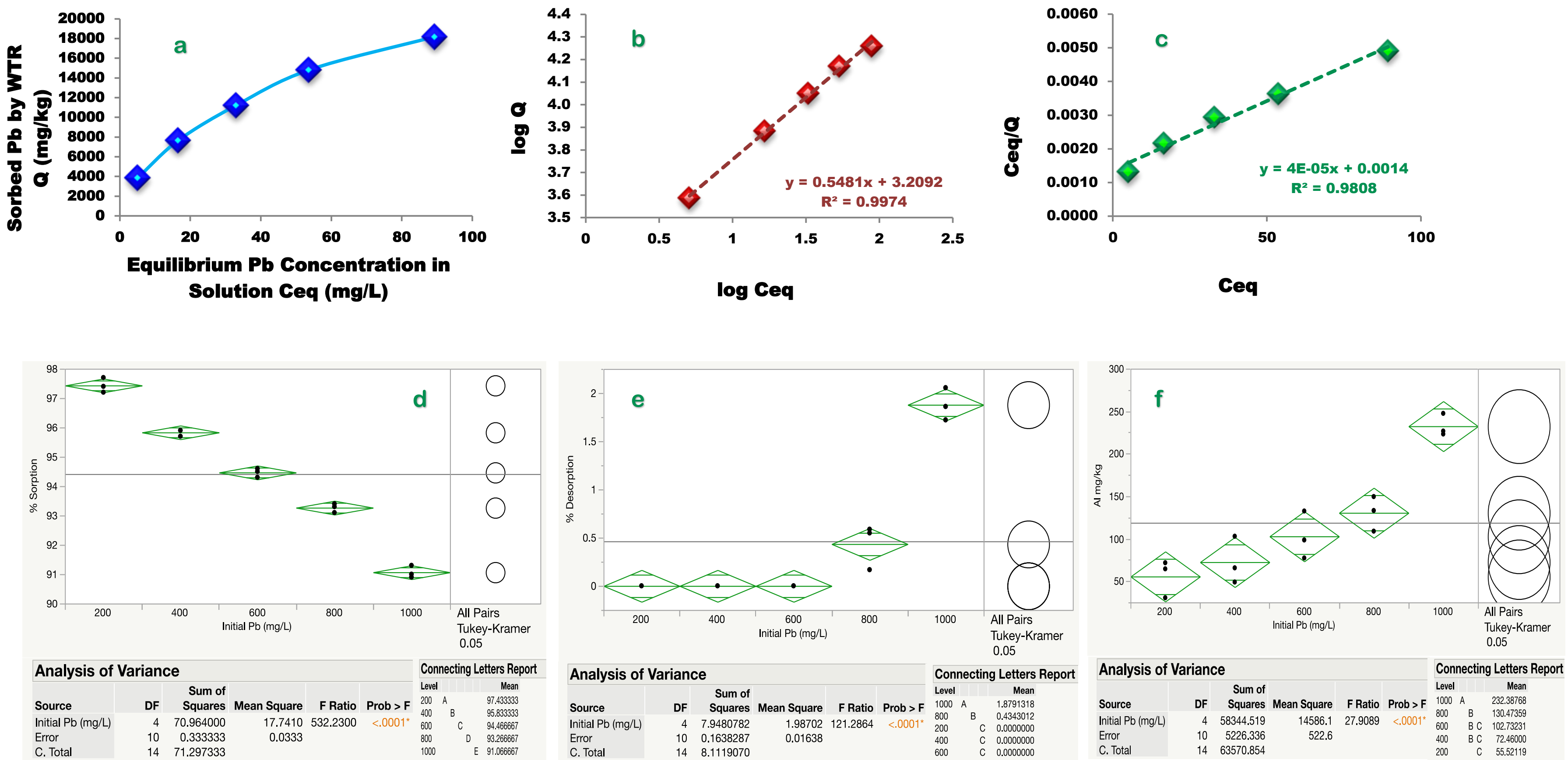
Effect Tests Initial Pb: 1000 mg/L

Source	Nparm	DF	Sum of Squares	F Ratio	Prob > F
Contact time (h)	5	5	48.3608	6.6813	0.0005*
%WTR	1	1	1158.3109	800.1372	<.0001*
%WTR*Contact time (h)	5	5	4072.6829	562.6650	<.0001*

Tukey-Kramer HSD
% WTR*Contact time (h)

Initial Pb (mg/L)	Contact time (h)	WTR (%)	Mean Comparison Letters
25	0.13	5	A
25	1.13	5	A
25	2.13	5	A
25	5.13	5	A
25	10.13	5	A
25	24.13	5	A
25	0.13	0	B
25	1.13	0	B
25	2.13	0	B
25	5.13	0	B
25	10.13	0	B
25	24.13	0	B
220	0.13	5	D
220	1.13	5	C
220	2.13	5	B
220	5.13	5	A
220	10.13	5	A
220	24.13	5	A
220	0.13	0	E
220	1.13	0	E
220	2.13	0	E
220	5.13	0	E
220	10.13	0	E
220	24.13	0	E
1025	0.13	5	D
1025	1.13	5	C
1025	2.13	5	B
1025	5.13	5	A
1025	10.13	5	A
1025	24.13	5	A
1025	0.13	0	FG
1025	1.13	0	G
1025	2.13	0	EFG
1025	5.13	0	E
1025	10.13	0	EFG
1025	24.13	0	EF

Figure 6.



Analysis of Variance (d)

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Initial Pb (mg/L)	4	70.964000	17.7410	532.2300	<.0001*
Error	10	0.333333	0.0333		
C. Total	14	71.297333			

Connecting Letters Report (d)

Level		Mean
200	A	97.433333
400	B	95.833333
600	C	94.466667
800	D	93.266667
1000	E	91.066667

Analysis of Variance (e)

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Initial Pb (mg/L)	4	7.9480782	1.98702	121.2864	<.0001*
Error	10	0.1638287	0.01638		
C. Total	14	8.1119070			

Connecting Letters Report (e)

Level		Mean
1000	A	1.8791318
800	B	0.4343012
200	C	0.0000000
400	C	0.0000000
600	C	0.0000000

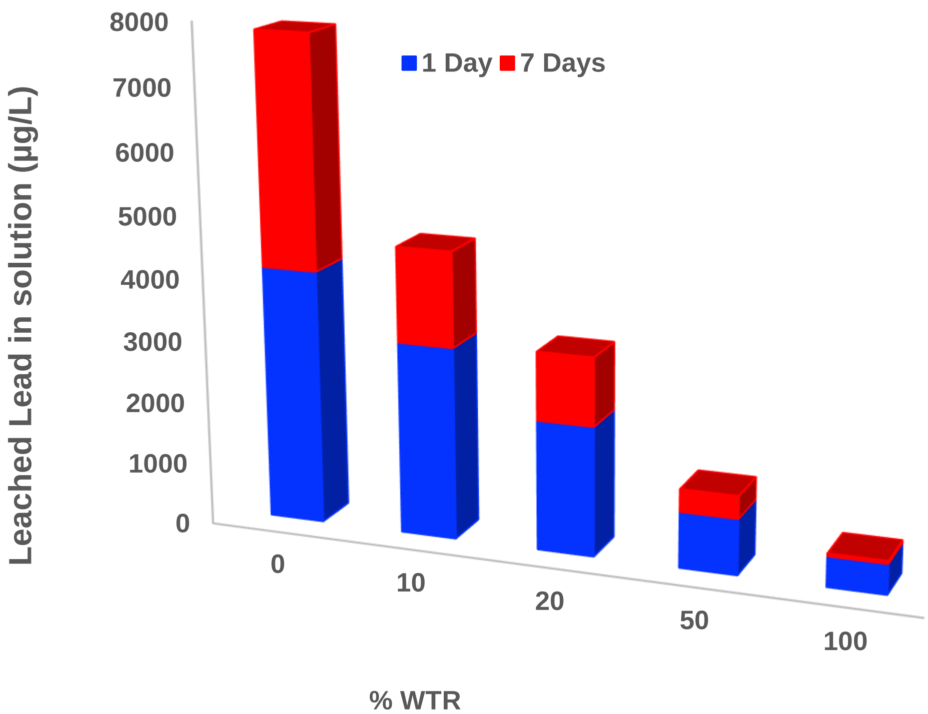
Analysis of Variance (f)

Source	DF	Sum of Squares	Mean Square	F Ratio	Prob > F
Initial Pb (mg/L)	4	58344.519	14586.1	27.9089	<.0001*
Error	10	5226.336	522.6		
C. Total	14	63570.854			

Connecting Letters Report (f)

Level		Mean
1000	A	232.38768
800	B	130.47359
600	B C	102.73231
400	B C	72.46000
200	C	55.52119

Figure 7.



Tukey-Kramer HSD (WTR % * Time in days)		
WTR %	Time (days)	Mean Comparison Letters
0	1	A
10	1	AB
20	1	BC
50	1	DE
100	1	DE
0	7	A
10	7	CD
20	7	CDE
50	7	E
100	7	E

Figure 8:

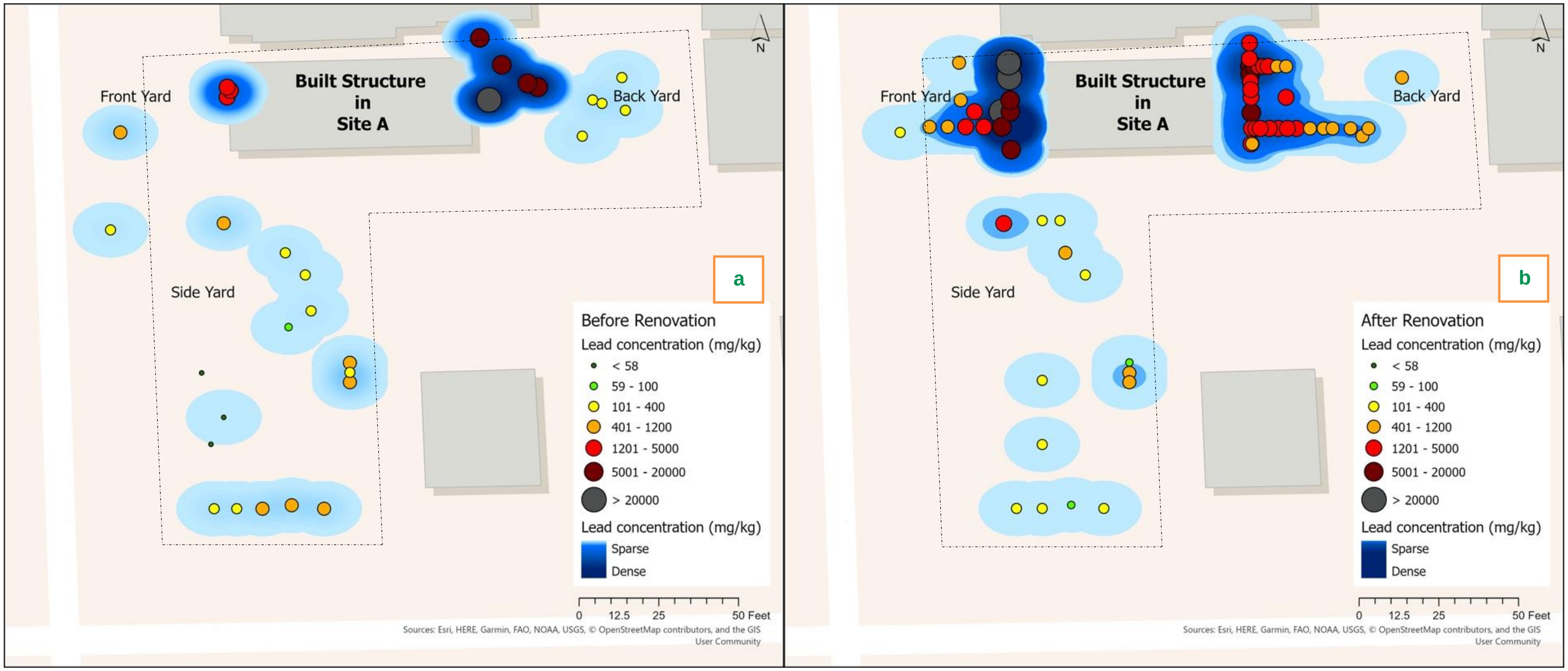


Figure 8 Footnote: Table S.4 includes a guideline created for the homeowners to explain each color category used in these maps, associated risk, and the recommended use of that location in their yards.

Supplementary Information

Table S.1: Overview of outdoor Pb exposure in the urban and suburban residential properties in Rochester, NY.

Residential Properties	Neighborhood	Year	Pb in Paint (mg/cm^2)	Depth Index	Maximum Pb in soil (mg/kg)	Maximum Pb in outdoor dust ($\mu g/ft^2$)
A (Before Renovation)	Urban	Unknown	33.7	1.0	50000	Not Measured
A (After Renovation)	Urban	Unknown	20.3	6.3	173500	3635
B	Urban	1923	2.3	1.0	1077	Not Measured
C	Urban	1930	5.9	10	1553	43.39
E	Suburban barn	1830	14.1	1.6	NA	8612
F	Suburban	2001	0	1.0	0	0
G	Urban	1910	6.4	4.4	1200	1969
H	Urban	1883	27.7	5.5	2378	1479
1I	Suburban	1900	23.0	1.9	390.5	290.2
J	Suburban	1954	8.4	6.5	115.7	83.86
K	Urban	1930	19.4	9.0	847.7	156.4
L	Suburban	1923	33.8	10.0	3088	80.42
M	Urban	1910	38.2	4.7	3743	Not Measured

Table S.2: Parameters of linear and nonlinear bivariate regression fit of Pb concentrations (mg/kg) in residential soil by distance (m) from potential source of Pb based paint using data collected from all properties.

Degrees of Fit	R^2	F ratio	p value
1	0.074	11.13	0.0011
2	0.138	11.04	<0.0001
3	0.179	9.992	<0.0001
4	0.209	9.017	<0.0001
5	0.229	8.016	<0.0001
6	0.238	6.986	<0.0001

Table S.3: Distribution of soil Pb and associated risks in three yards of site A. Percentile, mean, standard deviation (SD) of soil lead level (SLL) (mg/kg) and potential blood lead level (BLL) (μg/dL) in children and potential lead on play surfaces (PLOPS) (μg /ft^2) using three prediction models developed by Mielke et al., (2007), Johnson and Bretsch (2002), and Mielke et al., (2006), respectively.

		Percentile SLL (mg/kg)				SLL Statistics (mg/kg)				**Potential BLL (μg/dL)**		PLOPS (μg/ft2)
		Max	75%	**Median**	25%	Min	Mean	Std Dev	N	Equation 6	Equation 7	Equation 8
Front yard	Before Renovation	4130	3516	**854**	392	211	1667	1568	7	**7.1**	**10.6**	2575
	After Renovation	173500	26350	**12050**	3660	373	26167	39086	26	**20.9**	**16.1**	16220
Side yard	Before Renovation	1044	621.3	**345.4**	201	82	410	286	11	**5.2**	**8.7**	1358
	After Renovation	740.6	550.95	**367.5**	121	61	362	237	9	**5.3**	**8.8**	1420
Back yard	Before Renovation	50000	19625	**2899.5**	645	232	11326	14045	24	**11.3**	**13.1**	6042
	After Renovation	19000	2534.75	**1644.5**	469	68	2340	3211	42	**9.0**	**11.9**	4071

Table S.4: A guideline created for the homeowners to explain each color category used in the GIS map (figure 8), associated risk, and the recommended use of that location in their yards.

Color categories	Based on SLL	Risk Categories	Recommended use
Dark Green	58 mg/kg, which associates with the European Union's advisory limit for Pb in vegetables, calculated according to the prediction models developed in this study.	Minimal to No Risk	Safer locations to set up yard gardens
Light Green	100 mg/kg, which is Minnesota's current soil Pb advisory and recommended federal standard by scientists	Very Low Risk	Safer area for the children to play Not safe for home gardens
Yellow	400 mg/kg, current federal limit for children's play area	Low Risk	Low exposure risk for the children to play Not safe for home gardens
Orange	1200 mg/kg, current federal limit for remaining yard	Moderate Risk	Not safe for the children to play Not safe for home gardens
Red Maroon Black	Three extents of high concentrations exceeding all regulatory threshold	Risk Hotspots	Should be completely avoided. Remedial actions are required.

Figure S.1: *In situ* measurements of soil Pb by XRF and marking the assessment with colored flags, a tool that was used to communicate with the homeowners to show them the distribution of soil Pb and risk hotspots instantly. Red, green, and blue flags were used to mark SLL > 400, 80 – 400, and < 80 mg/kg, respectively. The photograph is be included in the manuscript with the permission of the homeowner. Photo courtesy: Michael Fisher.

Figure S.2: Lead concentration (mg/kg) in soil at Location A before (a) and after (b) renovation in 2018 and 2019 respectively. Data are expressed as mean (n=20) and + one standard deviation.

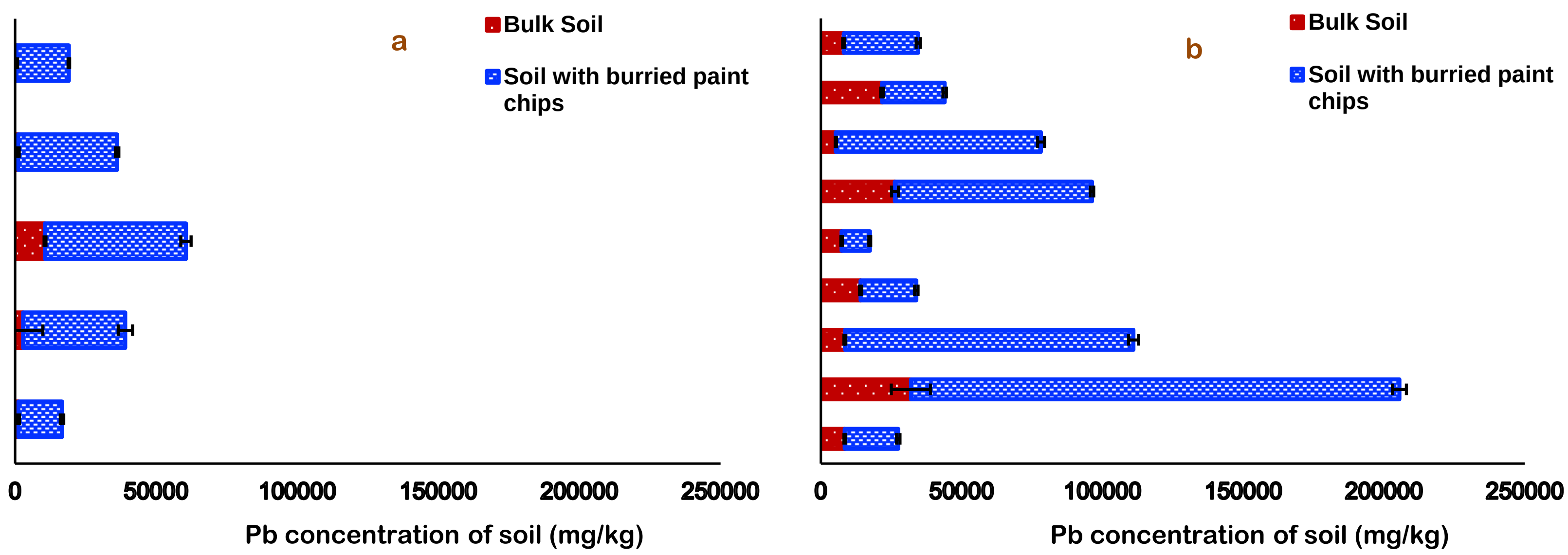
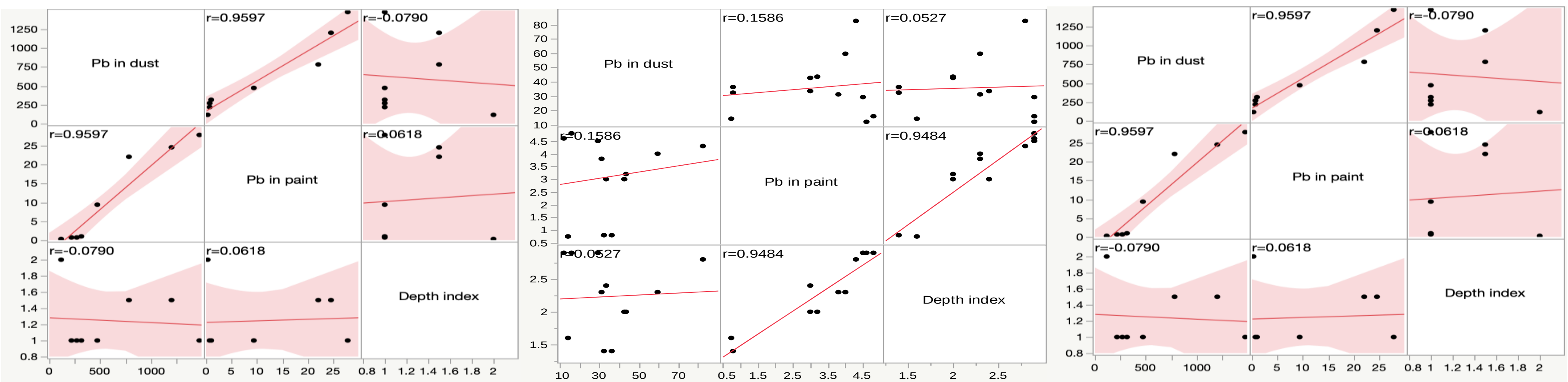


Figure S.3: Multivariate correlation among Pb in outdoor dust, paint and the depth index on discarded painted pieces of wood from site A (a), a window in site G (b), and a window in site H (c).



Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.9597	8	0.7879	0.9929	0.0002*
Depth index	Pb in dust	-0.0790	8	-0.7423	0.6626	0.8526
Depth index	Pb in paint	0.0618	8	-0.6722	0.7345	0.8845

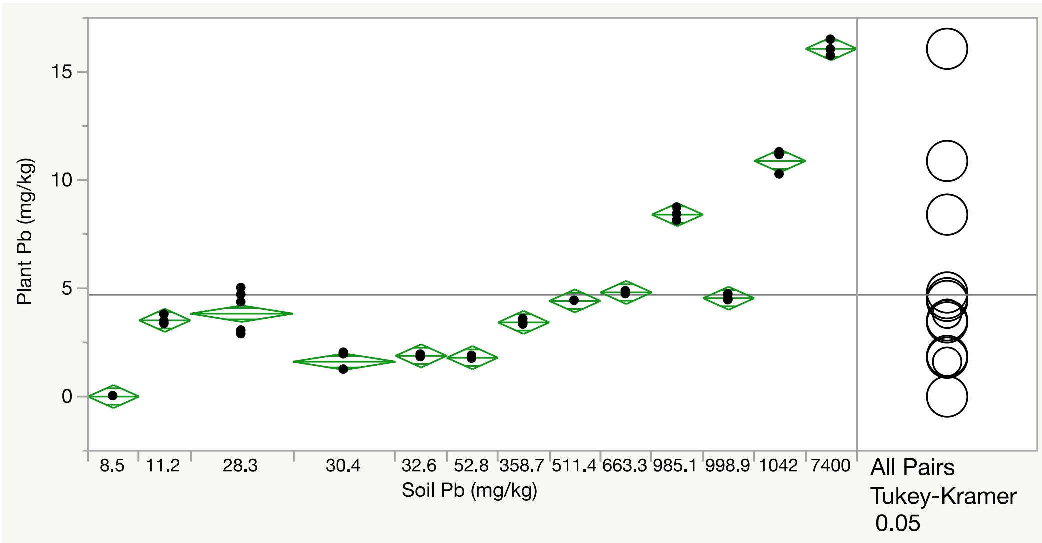
Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.1586	12	-0.4569	0.6714	0.6225
Depth index	Pb in dust	0.0527	12	-0.5374	0.6082	0.8707
Depth index	Pb in paint	0.9484	12	0.8219	0.9858	<.0001*

Pairwise Correlations

Variable	by Variable	Correlation	Count	Lower 95%	Upper 95%	Signif Prob
Pb in paint	Pb in dust	0.9597	8	0.7879	0.9929	0.0002*
Depth index	Pb in dust	-0.0790	8	-0.7423	0.6626	0.8526
Depth index	Pb in paint	0.0618	8	-0.6722	0.7345	0.8845

Figure S.4: Effect of soil Pb (mg/kg) on Pb concentration in plants (mg/kg) collected from all four home gardens of the tested residential properties and its bivariate regression with a polynomial square fit.



Polynomial Fit Degree=2

Plant Pb (mg/kg) = 2.3310181 + 0.0052245*Soil Pb (mg/kg) - 5.74e-7*(Soil Pb (mg/kg)-812.127)^2

Summary of Fit

RSquare	0.851557
RSquare Adj	0.844488
Root Mean Square Error	1.613987
Mean of Response	4.708192
Observations (or Sum Wgts)	45

Analysis of Variance

Source	DF	Sum of Squares	Mean Square	F Ratio
Model	2	627.63052	313.815	120.4686
Error	42	109.40809	2.605	**Prob > F**
C. Total	44	737.03862		<.0001*

Parameter Estimates

Term	Estimate	Std Error	t Ratio	Prob>|t|
Intercept	2.3310181	0.296985	7.85	<.0001*
Soil Pb (mg/kg)	0.0052245	0.000561	9.31	<.0001*
(Soil Pb (mg/kg)-812.127)^2	-5.74e-7	9.425e-8	-6.09	<.0001*