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Case studies and evidence-based approaches to addressing urban soil lead contamination☆

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A B S T R A C T

Urban soils in many communities in the United States and internationally have been contaminated by lead (Pb) from past use of lead additives in gasoline, deterioration of exterior paint, emissions from Pb smelters and battery recycling and other industries. Exposure to Pb in soil and related dust is widespread in many inner city areas. Up to 20–40% of urban children in some neighborhoods have blood lead levels (BLLs) equal to or above 5 μg per decilitre, the reference level of health concern by the U.S. Centers for Disease Control. Given the widespread nature of Pb contamination in urban soils it has proven a challenge to reduce exposure. In order to prevent this exposure, an evidence-based approach is required to isolate or remediate the soils and prevent children and adult's ongoing exposure. To date, the majority of community soil Pb remediation efforts have been focused in mining towns or in discrete neighborhoods where Pb smelters have impacted communities. These efforts have usually entailed very expensive dig and dump soil Pb remediation techniques, funded by the point source polluters. Remediating widespread non-point source urban soil contamination using this approach is neither economical nor feasible from a practical standpoint. Despite the need to remediate/isolate urban soils in inner city areas, no deliberate, large scale, cost effective Pb remediation schemes have been implemented to isolate inner city soils impacted from sources other than mines and smelters. However, a city-wide natural experiment of flooding in New Orleans by Hurricane Katrina demonstrated that declines in soil Pb resulted in major BLL reductions. Also a growing body of literature of smaller scale pilot studies and programs does exist regarding low cost efforts to isolate Pb contaminated urban soils. This paper reviews the literature regarding the effectiveness of soil Pb remediation for reducing Pb exposure and BLL in children, and suggests best practices for addressing the epidemics of low-level Pb poisoning occurring in many inner city areas.

1. Background

1.1. Urban soil lead patterns and sources

Surface soils of larger cities, especially inner cities, in the United States and the internationally have been contaminated by lead (Pb) from past use of Pb additives in gasoline, deterioration of exterior paint, emissions from smelters, battery recycling, and other industries. The spatial pattern of contamination depends on the history of development of roadways, housing, and industries, and is associated with inequitable chronic Pb exposure, along with social and environmental justice issues (Leech et al., 2016). While few comprehensive city-wide studies of soil Pb have been conducted, the few existing studies indicate neighbourhood-scale regions of high soil Pb are in communities of older housing, heavily trafficked roads, and point source Pb emissions (Filippelli and Laidlaw, 2010; Leech et al., 2016).

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A reservoir of highly bioaccessible Pb in urban soil and dust derived from that soil is concentrated near the surface where it is available to be resuspended in the air during dry periods (Filippelli et al., 2005; Laidlaw, 2016, Laidlaw et al., 2012, 2014; Mielke et al., 2014; Zahran et al., 2013a).

1.2. Ingestion and inhalation exposure pathways

The exposure pathway most commonly stressed for humans, especially children, is uptake of Pb via inadvertent ingestion of Pb -contaminated soils and dust, and subsequent absorption in the intestines (Manton et al., 2000). The multiple exposure pathways of soil and dust Pb include track-in via soil particles attached to shoes (Johnson, 2008; Hunt et al., 2006; Hunt and Johnson, 2012), pets tracking Pb indoors on their fur (Brunekreef et al., 1983), and direct contact with soils when the weather is favorable and children play outdoors. Exposure and uptake are age dependent; toddlers and small children ingest much more dirt than adults relative to their body mass. (USEPA, 2006), and their developing intestines absorb as much as 50% of the Pb they inadvertently ingest (Roberts et al., 2001). The potential for exposure to soil was underscored by a hand-wipe study of children at daycare centers in New Orleans (Viverette et al., 1996). The children's hands were wiped two times when they crossed the door threshold. The children's hands picked up more Pb when they played outdoors than when they played indoors (Viverette et al., 1996).

While it is common to assert that the main exposure pathway is ingestion via hand-to-mouth behavior, available empirical data on the association between air Pb and BLL indicates that inhalation is an underappreciated pathway of Pb exposure. Evidence is from the US national experience of the phase-down of tetraethyl Pb additives in fuel took place: when air lead decreased there was a concurrent decline of children's Pb (Annest et al., 1983; Bridbord and Hanson, 2009). Likewise, when the Bunker Hill Pb smelter in Idaho closed and Pb aerosols ceased, then the BLL of the children underwent a sharp decline (Schoof et al., 2016; Von Lindern et al., 2003a). Also, a common observation in urban studies is that children's BLL fluctuates seasonally. The BLL response includes summer peaks and winter minimums in synchrony with rises in air Pb during the summer and declines of air Pb during winters; air Pb is associated with soil resuspension (Laidlaw et al., 2005, 2012, 2014; Zahran et al., 2013a). It should be noted however that very little experimental *in-vitro* or *in-vivo* work has been conducted to confirm this pathway, and hence there is a great research need to understand pulmonary particulate geochemistry.

One misunderstanding about soil Pb arises from the fact that measurement units used for outside soils are not equivalent with measurement units used for interior floor surfaces. Soil that meets the current US EPA lead content standard of 400 μg/g is associated with a Pb loading value of about 16,200 $\mu g/m^2$ (1500 $\mu g/ft^2$) or 38 times more Pb than the interior floor Pb standard of 430 $\mu g/m^2$ or 40 $\mu g/ft^2$ (Mielke et al., 2007; US EPA, 2015). The major point is that even moderately Pb-contaminated soils contain a substantial surface reservoir of Pb dust and this reservoir of lead dust becomes available when soils become dry in late summer resulting in a seasonal exposure spikes (Mielke et al., 2014; Zahran et al., 2013a). Evaluating sources and pathways of exposures requires at least the use of equivalent measurement units. Soil lead content is not the same as lead loading of the soil surface. This issue persists as a common misunderstanding regarding the quantity of surface Pb available from the soil reservoir. In addition to soil Pb accumulated on the soil surface, the particle size is important.

Particle size is an especially important consideration regarding the impact of soil Pb on inhalation. The past use of tetraethyl lead (TEL) by vehicles in the urban environment was associated with the exhaust of particle sizes mostly < 0.1 μm diameter (Little and Wiffen, 1978). Accounting for Pb in New Orleans revealed that at least ten times more Pb came from traffic flows than from paint (Mielke et al., 2011a,b). Research of Pb aerosols leaving the South Coast Air Basin of California reveals that conventional industrial sources of Pb accounts for about 13% of the Pb aerosols; the former use of TEL that accumulated in the soil undergoes resuspension and accounts for most of the remaining 87% of the Pb aerosols leaving the basin (Harris and Davidson, 2005).

1.3. Bioaccessibility and bioavailability

Because of the assumption that ingestion is the most important pathway of Pb exposure the topics of bioaccessibility and bioavailability are recognized as important keys to risk assessment. At the basis of the issue is the fact that humans display hand-to-mouth behavior in the womb and during very early childhood (Desmurget et al., 2014). This observed behavior is assumed to link children directly with soils and sediments. As Chaney et al. (1989) stated: “Because the worst case child may have pica for soil, the ultimate soil Pb limit will depend more on the bioavailability of soil Pb than on the amount of soil ingested.” Experimental techniques use either animal models, such as rats or miniature pigs, or *in-vitro* models. The *in-vitro* models use an artificial stomach and synthetic pH-adjusted fluids along with an artificial intestinal tract with synthetic pH-adjusted fluids. As indicated by a small sample of studies, the efforts to determine bioaccessibility have involved enormous resources on the part of researchers accompanied by mixed results (Farmer et al., 2011; Luo et al., 2012; Wragg et al., 2011). The results linking contaminants in soil and sediments prove to be extremely complicated (Semple et al., 2004). Assuming that hand-to-mouth behavior is the link between the environment and exposure, the focus of the studies has been on the ingestion pathway of exposure. However, as discussed in the previous section, inhalation is an underestimated pathway of exposure. The ingestion and the inhalation pathways of exposure must be included to assess soil bioaccessibility and bioavailability.

1.4. Associations between soil lead and blood lead

In New Orleans, Louisiana, Zahran et al. (2013b) combined two extensive data sets: (i) 5467 surface soil Pb samples collected from 286 census tracts; (ii) geo-referenced BLL data for 55,551 children in metropolitan New Orleans, USA. The results indicated that children's BLL are spatially associated with soil Pb levels. About 67% of the variation in children's BLL was explained by the four independent soil Pb sample location variables—house-side, residential street-side, busy street-side, and open-space. Importantly, this study adds to a very small number of large sample studies that have examined the spatial relationship between soil Pb and BLL (Bickel, 2010; Johnson and Bretsch, 2002; Zahran et al., 2011). The dose response relationship between soil Pb and BLL using the United States Environmental Protection Agency's (USEPA) Integrated Exposure Uptake Biokinetic (IEUBK) model was calculated. A chart showing these four empirically based soil Pb dose response relationships is Fig. 2. Note the trends for some cities for a curvilinear BLL response at the lower soil Pb contents. This indicates that there is a steeper rise in BLL per unit increase in soil Pb at low soil Pb than at higher soil Pb quantities.

1.5. Health effects and the margin of safety issue

Elevated Pb in children is associated with lower Intelligence Quotient (IQ), learning disorders, and behavioural problems (Nevin, 2000; Nigg et al., 2008). These impacts are also largely age dependent, as neural development is most rapid in young children, and the presence of Pb in blood during this time interferes with proper neuron formation (Lucchini et al., 2014). Furthermore, much of the Pb initially present in blood is stored for years in bone, which becomes a long-term chronic source of Pb back into the blood stream. Even after a child is removed from a Pb -rich environment there is even some indication of decreased cognitive function in elderly osteoporotic patients from the increased release of Pb into the bloodstream due to rapid bone loss (Needleman, 2004). A review of the impacts of low level Pb concentrations in children has been compiled by the United States Department of Health and Human Services National Toxicology Program (USDH-NTP, 2012), United States Environmental Protection Agency and the Agency Toxic Substances and Disease Registry (ATDSR, 2016). Recently, progress has been made in understanding numerous chronic health effects associated with Pb exposure. For example, there are strong associations between Pb and a multitude of diseases such as motor neuron disease (Laidlaw et al., 2015; Santurtún et al., 2016), autism (Rossignol et al., 2014; Gorini et al., 2014; Mostafa et al., 2016; Kim et al., 2016) preeclampsia (Kennedy et al., 2012) developmental delays in children (Earl et al., 2015), heart disease (Navas-Acien et al., 2007), ADHD (Goodlad et al., 2013), dementia (Genuis and Kelln, 2015), mental illness (Tomás et al., 2012) and brain cancer (Wu et al., 2012). Indeed, even as more exposure-disease relationships are uncovered, there are new findings which point to yet another outcome previously unrecognized and missed (Bellinger, 2011).

Given the difficulties of quantifying the pathways of exposure, bioavailability, bioaccessibilty, and understanding the full extent of health effects, it is essential to include a ‘margin of safety’ to protect the health of the most vulnerable individuals, especially children. Soil ingestion is common in humans especially in children (Starks and Slabach, 2012). According to U.S. research guidelines, if ingestion is involved then a factor of 10 must be applied as a margin of safety (US DHHS, 2005). Soil standards are poorly developed and the most commonly cited soil Pb standard, the U.S. EPA 400 mg/kg value, is not a health-based standard. There are many soil lead standards ranging from 20 mg/kg to over 1000 mg/kg (Jennings, 2008). Most of the soil Pb regulatory guidance values are aligned with the EPA value. Several studies and reviews proposed health-based soil Pb guidelines. For example, Madhavan et al. (1989) proposed a maximum permissible soil Pb of 600 mg/kg assuming that the exposure would increase blood lead by 5 μg/dL (now considered the reference blood lead guideline by CDC). Also notable is a review which proposed a guideline of <100 mg/kg based on evidence-based data from several studies, with an added margin of safety, and the assumption that 10 μg/dL is safe (now known to be at odds with clinical studies) (Reagan and Silbergeld, 1990; Xintaras, 1992). A taskforce formed by the Society of Environmental Geochemistry and Health (SEGH) suggested guidelines that took into account multiple factors including: “… a soil or dust guideline, a blood lead target concentration in micrograms of lead per deciliter of whole blood, a geometric standard deviation of the blood lead distribution, the baseline concentration of blood lead from multiple sources, the degree of protection required for the population at risk, and the slope of the relationship of blood lead to lead in soil and dust were on an exposure guideline (Wixson and Davies, 1994).” The result of the SEGH deliberation is a wide range of soil lead guidelines depending on many variables.

A 1999 evidence-based study on soil and blood lead in communities of New Orleans indicated that if the goal is to prevent Pb exposure ≥10 μg/dL for children, then the soil lead standard should be ~80 mg/kg (Mielke et al., 1999). A repeat of the evidence-based soil and blood lead study 10 years after Hurricane Katrina revised the soil lead guidance value. If the goal is to prevent exposure at the revised CDC BLL reference value of ≥5 μg/dL, then the soil Pb standard should be ≤ 40 mg/kg to ensure that most children are protected from the risks of inadvertent exposure by environmental sources of soil and dust Pb (Mielke et al., 2016). The precautionary lesson is that given the ever-changing understanding of life-long health issues associated with exposure to lead, a ‘margin of safety’ is essential to prevent lead exposure by the youngest members of society (Bellinger, 2011).

1.6. Primary prevention versus secondary prevention

Currently the United States screens a relatively small percentage of children's BLL in a community and follows up with home visits and remedial actions typically focused on Pb paint and dust *if* funding is available. The prevalence of BLL screening varies widely within individual states and is often conducted as part of government mandated testing in low-income communities. This screening method uses children as “canaries in a coal mine” and identifies children after they have been exposed to Pb and the harm has been done. Unfortunately, these types of programs that focus on Pb paint hazard education and dust clean-up as the interventions to reduce Pb poisoning are not particularly effective on a community-wide scale, at least from an environmental health perspective (Kennedy et al., 2016; Yeoh et al., 2014). This *ad hoc* approach to community health protection is currently the dominant Pb poisoning prevention method even though the locations in inner city areas where the majority of the Pb poisoning cases occur are well known and re-occur (Leech et al., 2016). The community wide and multi-generational pattern of Pb poisoning profoundly impacts the health and livelihood of neighborhoods, leading to lower educational outcomes (Zahran et al., 2009; Zhang et al., 2013a,b; Evens et al., 2015), lower incomes (Needleman, 2004; Attina and Trasande, 2013) and higher incarcerations rates (Needleman et al., 2002; Nevin, 2007, 2009; Mielke and Zahran, 2012). The Pb poisoning situation internationally may be worse than the U.S. because typically, there are scant to no public health Pb screening efforts, and because chronic low level Pb poisoning exhibits few to no symptoms that are diagnosable during a standard clinical visit.

Primary prevention of Pb poisoning means identifying and then isolating or remediating all potential sources of environmental Pb exposure prior to allowing children into the environment. The best primary prevention practice uses a holistic approach of eliminating all major Pb hazards (lead sources in water, air and dust/soil) before a child moves into a residential environment.

1.7. Criteria of effective soil lead interventions

The city-wide scale of Pb contaminated soil is massive and includes many residential communities (see Fig. 1). Practical methods of intervention/remediation must be inexpensive and should rely on low Pb materials that are locally abundant in large quantities. If these criteria are met, then there is a chance for extensive intervention at the scale of the community. The material must provide a margin of safety for the most vulnerable population living within communities. Empirical research indicates that soil lead must be a tenth of the current U.S. soil Pb standard of 400 mg/kg or ~40 mg/kg or even less (see Fig. 2). The sensitivity of children is confounded by the fact that the routes of exposure includes ingestion and inhalation (especially during seasonally dry periods of the year) as described in section 1.2.

2. Case studies of interventions that focus directly on soil lead

Despite the widespread need for soil Pb remediation in urban

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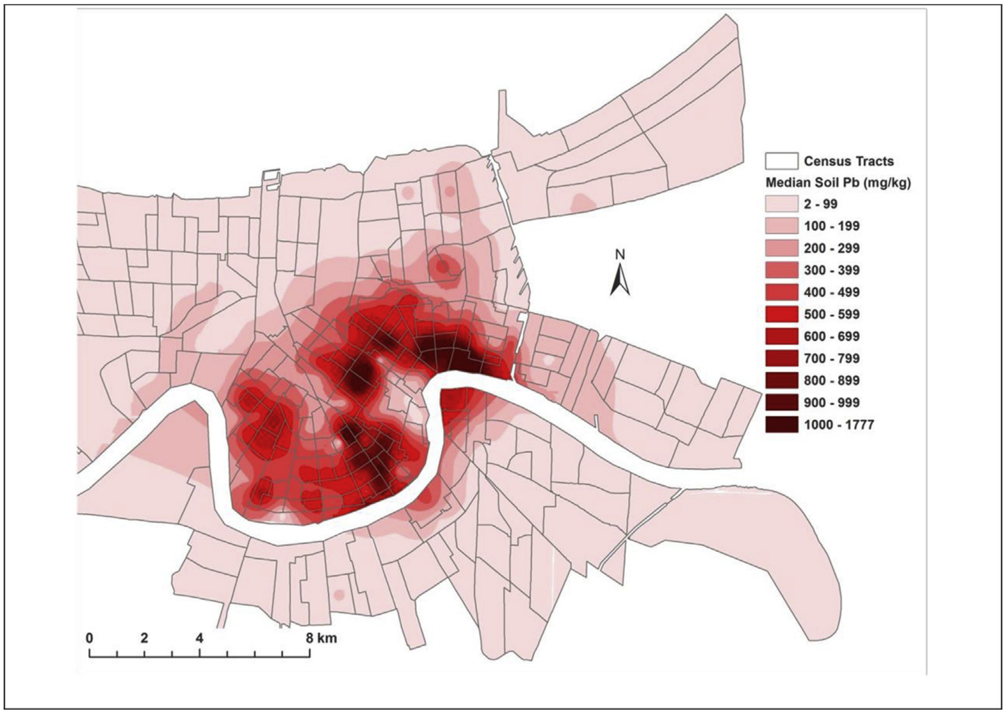

Fig. 1. Typical urban inner city soil lead distribution as shown in pre-Hurricane Katrina New Orleans, Louisiana, USA. The low lead soil area surrounded by soil with extreme lead values is the Central Business District where surface soil is covered by pavement and not readily available (Gonzales CR, Powell ET, Mielke HW, unpublished original for this review).

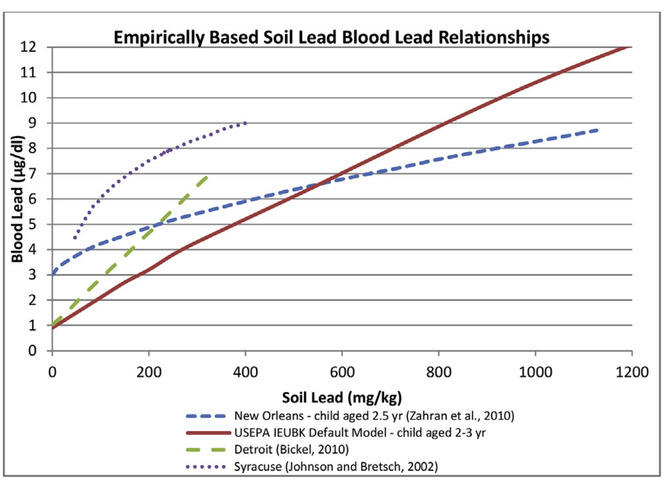

Fig. 2. Empirically based soil lead blood lead relationships.

communities, there are relatively few peer reviewed studies in the literature that can serve as a base of evidence to evaluate the effectiveness of soil Pb interventions. The studies listed in Table 1 demonstrated soil Pb interventions that resulted in declines in children's BLL levels. This review has not included soil intervention studies where ongoing Pb deposition from an existing source, such as a smelter, was operating during the soil intervention period. Some of the soil intervention studies conducted at operating Pb smelter sites are presented in Lorenzana et al. (2003).

2.1. Soil interventions that assessed the reduction in blood lead levels

Schoof et al. (2016) examined blood lead level (BLL) trends in children from 2003 to 2010 in the Pb contaminated mining town of Butte, Montana and compared the BLL trends to a reference dataset matched for similar demographic characteristics over the same period. A program was initiated that identified the sources of lead, including sampling for indoor dust, outdoor soil, indoor and outdoor paint, and lead in drinking water (from plumbing). A total of 2340 of the 3646 properties (64%) were screened. Between 2003 and 2010, 545 abatements were conducted (294 for soil and 251 for paint) while no renovations were identified as being necessary for plumbing. For 2003–2004, 2005–2006, and 2007–2008, geometric mean BLLs were higher for Butte than for the reference dataset. By 2009–2010, adjusting for all other variables, the geometric mean BLL for Butte was no longer significantly higher. The Butte geometric mean BLLs declined by 24% per 2-year increment from 2003 to 2010, while the reference dataset BLLs showed a significantly slower decline of 9% per 2-year increment ($p < 0.0001$). Over the full study period, the percentage of BLLs >10 µg/dL declined from 3.5% in 2003 to 1.5% in 2010, while BLLs >5 µg/dL declined from 33.6% in 2003 to 9.5% in 2010. The report concluded that abatement programs that included home evaluations and assistance in addressing multiple sources of Pb exposure can be an important complement to community-wide soil remediation activities.

The Bunker Hill Superfund Site (BHSS) is a historic mining and smelting district located in a 21 square mile in the Coeur d'Alene Basin in northern Idaho. Five residential communities of Pinehurst, Kellogg, Smelterville, Wardner, and Page are located within the BHSS (Sheldrake and Stifelman, 2003; Von Lindern et al., 2003a). Between 1990 and 2001, through a community wide clean-up program of establishing a clean soil Pb barrier, the geometric mean soil Pb levels in Kellogg were reduced from approximately 700 mg/kg to 175 mg/kg, while the soil Pb levels in Smelterville were reduced from approximately 750 mg/kg to approximately 175 mg/kg (Sheldrake and Stifelman, 2003). Similarly, between 1990 and 2001, the vacuum bag house dust Pb concentration in

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Kellogg declined from approximately 1200 mg/kg to 390 mg/kg and from 1750 mg/kg to approximately 325 mg/kg in Smelterville (Sheldrake and Stifelman, 2003). Children's BLLs >10 μg/dL in Kellogg and Smelterville declined from approximately 46% in 1988 to approximately 3% in 2001 (Sheldrake and Stifelman, 2003). Von Lindern et al. (2003a) also discuss the soil Pb remediation program at the BHSS in further detail.

In the BHSS, Maisonet et al. (1996) conducted a pair-matched, case-control study to identify if risk factors or behaviors suspected to affect childhood BLLs were more prevalent among children with elevated BLLs living in the vicinity of the BHSS. The study observed that for all the variables examined, yard soil remediation was found to be a protective factor for elevated BLLs in children (odds ratio, 0.28; confidence interval, 0.08–0.92).

Soils and dusts in the Tar Creek Oklahoma Superfund communities of Picher and Cardin were contaminated from Pb due to lead and zinc mining activities. In 1997, 21.5% of children living near Tar Creek exhibited BLLs exceeding 10 μg/dL (USEPA, 2014). Following USEPA, state, and Native American clean-up activities, Pb-contaminated soil was removed from 2887 residential yards and public properties in the area (USEPA, 2014). In 2014 none of the children in Picher or Cardin had BLLs exceeding 10 μg/dL, which suggests that the soil Pb remediation efforts were successful (USEPA, 2014).

Between about 2009 and 2010 artisanal gold mining caused widespread contamination resulting in the deaths of more than 400 children in Zamfara State, Nigeria. Initial village surveys estimated that >90% of ongoing Pb exposure was due to ingestion of lead-contaminated soils and food (Von Lindern et al., 2011; Dooyema et al., 2012). Primary sources of Pb were contaminated surface soils (generally <5 cm depth) within the compounds, dusts on surfaces and soft materials (e.g., sleeping mats), and food preparation utensils used in ore processing. Between 2010 and 2013 more than 27,000 m^3 of contaminated soils and mining waste were removed from 820 residences and ore processing areas in eight villages. Excavated areas were capped with clean soils (<25 mg/kg Pb), decreasing soil Pb concentrations by 89%. Following intervention and chelation of many of the children, geometric mean BLLs for children ≤5 years old decreased from 149 μg/dL to 15 μg/dL over the four-year remedial program.

Lanphear et al. (2003) evaluated the effect of soil abatement on children's BLLs in a subset of children living near a former smelting and milling operation in Midvale, Utah. From 1993 to 1996, soil abatement was conducted around homes with average soil Pb concentration >500 mg/kg. The soil abatement entailed excavation and disposal of the top 46 cm (18 inches) of soil. Results indicated that Pb in interior dust of homes that underwent soil abatement declined significantly compared to unabated homes ($p < 0.05$).

Table 1
Summary of global soil lead interventions and impacts on children's blood lead levels.

Case Study	Scale	Intervention Approach	Average Soil Lead Concentration Before Remediation	Soil Pb Reduction	Blood Pb Reduction and Time Frame	Reference
Butte, Montana (USA)	City	excavation and/or capping (unknown depth)	unknown	unknown	% BLL > 5 μg/dL from 33.6% to 9.5% in 7 years	Schoof et al. (2016)
Coeur d'Alene Basin, Idaho (USA)	Multiple Cities	15-30 cm soil removal or 30–60 cm soil cap	Kellogg - 700 mg/kg; Smelterville 750 mg/kg	approximately 500 mg/kg	% BLL > 10 μg/dL from 46% to 3% in 13 years	Sheldrake and Stifelman (2003)
Tar Creek, Oklahoma (USA)	City	excavation (unknown depth)	unknown	unknown	% BLL > 10 μg/dL from 21.5% to 0% in 16 years	USEPA (2014)
Zamfara State (Nigeria)	8 Villages	5 cm soil removal; >8 cm soil cap	300 mg/kg to 4,143 mg/kg	soil Pb greater than 400 mg/kg capped or excavated	Avg. BLL from 149 μg/dL to 15 μg/dL in three years[b]	Tirima et al. (2016)
Midvale, Utah (USA)	Individual Homes	46 cm soil removal; clean soil backfill	542 mg/kg (geometric mean)	approximately 500 mg/kg	BLL declined 42.8% faster in abated houses; 3.5 μg/dL reduction per 1000 mg/kg lead removal	Lanphear et al. (2003)
St-Jean-sur-Richelieu, Quebec (Canada)	Neighborhood	10 - 30 cm soil removal	1756 mg/kg (median)	soil lead removed if > 400 or 500 mg/kg	% BLL > 15 μg/dl from 21.3% to 0% and geometric mean BLL decreased 48% in 3 years	Goulet et al. (1996)
Boston, Massachusetts (USA)	Individual Homes	15 cm soil removal; 20 cm clean soil cap	2075 mg/kg (median)	1790 mg/kg	Average BLL reduction of 2.44 μg/dL in 11 months	Weitzman et al. (1993; Aschengrau et al. (1994)
Boston, Massachusetts (USA)	Individual Homes	15 cm soil removal; 20 cm clean soil cap	2075 mg/kg (median)	1790 mg/kg	Net difference of 5.6 μg/dL compared to no soil treatment in 9 months time[a]	Aschengrau et al. (1997)
Dong Mai Village, (Vietnam)	Village	Geotextile; 5 cm sand cover; 20 cm misc. Cap	42 of 261 gardens >800 mg/kg;	soil lead >800 mg/kg remediated	Avg. BLL from 39.1 μg/dL to 25.35 μg/dl (35% reduction) in 6 months	Ericson (2014)
Bauru, São Paulo (Brazil)	1000 m radius	5 cm soil removal	350 mg/kg (maximum)	Maximume soil lead before remediation = 350 mg/kg	Median BLL from 15.0 μg/dL to 8 μg/dL (46.23% reduction) in 1.5 Years	de Freitas et al. (2007)
Baltimore, Maryland (USA)	Neighborhood	15 cm soil removal; 15 clean soil cap	501 (trimean)	Avg reduction of trimean from 501 to 34 mg/kg	No reduction in BLL	Farrell et al. (1998)
New Orleans, Louisiana (USA)	City	Hurricane Katrina Sediment	280 mg/kg (median)	280 to 132 mg/kg	Median BLL from 5.0 to 1.8 μg/dL in 10 years	Mielke et al. (2016)

[a] Lead-based-paint remediation alone was associated with statistically significant blood lead increase of 6.5 μg/dL over the subsequent 9 months, but with an increase of only 0.9 μg/dL when combined with soil abatement.
[b] Note – In addition to the soil intervention, 2349 children simultaneously received chelation treatment.

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Lanphear et al. (2003) stated that "After adjustment for potential confounders, the blood lead concentration in children ages 6–72 months who lived in soil-abated housing declined 42.8% faster than children who lived in unabated housing ($p = 0.14$). In children ages 6–36 months, the decline was 45.4% faster ($p = 0.03$). The estimated reduction in blood lead for children ages 6–36 months was 3.5 μg/dL for every 1000 ppm reduction in soil lead concentration (95% confidence interval [CI]=2.4 μg/dL, 4.6 μg/dL)." Lanphear et al. (2003) concluded that soil abatement was associated with a significant decline in children's BLLs and indoor environmental levels of Pb.

Goulet et al. (1996) reported the results of a public health program to monitor pediatric BLLs in St-Jean-sur-Richelieu, Quebec, Canada. In 1989, it was determined that soils near a residential area near a battery reclamation plant were contaminated with Pb. Remedial activities included a variety of activities including street dust removal and removal of the top 10–30 cm of soil from all accessible surfaces depending on soil Pb concentrations and type of surface (exposed soil, grass or gravel covered). Professional housecleaning, including high-efficiency particulate air filter (HEPA) vacuuming, was conducted in 115 homes in which a child had a BLL ≥15 μg/dL. Blood Pb surveys of the children living in the area were sampled before (1989) and after the soil Pb intervention (1991). Results indicated that the geometric mean BLL of children 6 months to 10 yr of age decreased 48% and for children 6 months to 5 years of age the geometric mean BLL level decreased 44%, a result that was statistically significant. In addition, 21.3% of the children included prior to the intervention had BLLs exceeding ≥15 μg/dL, and none of the children had a BLL ≥15 μg/dL after the intervention.

Weitzman et al. (1993) reported the results of the first phase of the Boston Lead-in-Soil Demonstration Project. This study performed a randomized controlled trial of the effects of Pb-contaminated soil abatement on BLLs of children followed up for approximately 1 year after the intervention. A 15-cm layer of topsoil was removed from the entire yard and replaced with 20 cm of clean soil. A water-permeable geotextile fabric barrier was laid directly on top of the exposed subsurface immediately following removal of topsoil and prior to placement of clean topsoil and then covered with either sod, grass seeding, bark, or mulch. The average reduction in soil Pb concentrations was 1790 mg/kg. Results indicated that the mean decline in BLLs from before abatement and 11 months after abatement was 2.44 μg/dL in the study group ($P = 0.001$). The authors concluded that lead-contaminated soil contributes to the lead burden of urban children and that abatement of Pb-contaminated soil around homes resulted in a modest decline in BLLs.

Aschengrau et al. (1994) reported the results of a portion of the Phase II of the Boston Lead-In-Soil Demonstration Project. Aschengrau indicated that Pb-contaminated soil abatement of approximately 2060 ppm was associated with a 2.25–2.70 μg/dL decline in children's BLLs. Low levels of soil recontamination 1–2 years following abatement indicated that the intervention was persistent, at least over the short-term. Aschengrau et al. (1997) reported the results of a portion of the Phase II of the Boston Lead-in-Soil Demonstration Project. Soil abatement was offered to two groups and residential Pb-based-paint hazard remediation was offered to all three groups. Results indicated that Pb-based-paint remediation alone was associated with statistically significant BLL *increase* of 6.5 μg/dL over the subsequent 9 months, but with an increase of only 0.9 μg/dL when combined with soil abatement. The authors concluded that the beneficial impact of soil abatement may account for the smaller increase when both interventions were conducted.

The non-profit company Pure Earth has effectively used the geotextile and clean soil cap method employed by Mielke et al. (2011a,b) in the Dong Mai village in Vietnam where residents contaminated the soil as a result of extracting Pb from automotive batteries (Ericson, 2014). Remediation was enacted at 42 gardens where the soil Pb concentration exceeded 800 ppm. The soil intervention method included first covering contaminated soil with 5 cm of sand and a geotextile layer, with further layers of either 20 cm of sand, 20 cm of compacted clean soil, pavers (bricks), or concrete. Following clean soil emplacement, soils exhibited concentrations less than 50 mg/kg. Following one year after soil Pb remediation, BLLs fell from an average of 39 μg/dL to an average of 25 μg/dL (35%). Declines were seen in all age groups and at all levels of exposure.

The soils surrounding a former battery recycling plant located in Bauru, West of the state of Sao Paulo, Brazil, were contaminated with Pb, a maximum Pb concentration of 350 mg/kg (0–2 cm deep sample; de Freitas et al., 2007). The roads surrounding the plant were unpaved. Children's BLLs were measured in the area within 1000 m of the battery recycling plant. Environmental controls consisting of indoor vacuum-cleaning of households, removal of a 5 cm deep layer of soil around the houses, paving of nearby streets, and establishment of specific hygiene and educational programs targeted to the local population. A new BLL evaluation was performed 1.5 years later following implementation of the environmental controls. Results indicated that BLLs after remediation declined a median of 15 μg/dL to 8 μg/dL, a reduction of 46% ($p < 0.01$).

Farrell et al. (1998) conducted a study based in Baltimore, Maryland that revealed no BLL decline after soil remediation. In two Baltimore neighbourhoods (study and control) Farrell et al. (1998) implemented a prospective longitudinal study to test a hypothesis that a reduction of 1000 parts per million of soil Pb would reduce children's BLLs by 3–6 μg/dL. The study included monitoring BLL in 187 children. In the study area, contaminated soil was replaced with clean soil. Exterior paint was also stabilized. The results indicated that soil Pb abatement did not lower children's BLL.

Finally, Mielke et al. (2016) evaluated a unique natural experiment caused by Hurricane Katrina flooding of 80% of New Orleans. At the scale of an entire city this study evaluated the changes of soil and BLL pre- and ten years post-Katrina and considered the effectiveness of low Pb soil intervention on children's blood lead. Soil and BLL were mapped prior to Katrina. On 29 August 2005 Hurricane Katrina disrupted habitation in New Orleans. Beginning in 2015, New Orleans soil and BLL was remapped and the post-Katrina data were stratified by the same census tracts ($n = 176$) as pre-Katrina. On the same 176 census tracts the data-sets included soil Pb ($n = 3314$ and 3320, pre-vs. post-Katrina), BLL ($n = 39{,}620$ and 17,739, pre-vs. post-Katrina), distance, and changes in percent pre-1940 housing. Statistical analysis was by permutation procedures and Fisher's Exact Tests. Pre-vs. ten years post-Katrina soil Pb median concentrations decreased from 280 mg/kg to 132 mg/kg, while median BLLs decreased from 5 μg/dL to 1.8 μg/dL. The percent pre-1940 housing did not change significantly (P-value = 0.674). With the exception of age-of-housing results, all other P-values were extremely small ($<10^{-12}$). From the perspective of an entire city all of the variables, including age-of-housing, soil Pb and BLLs, decreased substantially ($<10^{-12}$) with distance from the center of New Orleans. The reduction of lead on soil surfaces throughout the city was associated with children's interaction with lead dust, thus underscoring soil Pb as a critical reservoir of exposure (Mielke et al., 2016).

2.2. Soil isolation interventions that reduce soil lead without accompanying blood lead results

The studies in Table 2 demonstrated soil Pb interventions that

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Table 2
Soil lead isolation interventions without blood lead exposure data.

Case Study	Scale	Intervention Approach	Soil Pb Reduction	Reference
New Orleans, Louisiana (USA)	9 Day Care Centres	Geotextile; 15 cm soil cover	Avg reduction of 554 mg/kg	Mielke et al. (2011b)
Broken Hill, NSW (Australia)	Individual Homes	excavation and/or capping (unknown depth)	unknown	Boreland and Lyle (2006)
Kabwe (Zambia)	78 Homes	Geotextile; 10–15 cm clean soil cap	Typically, 2000–4000 mg/kg	Ericson and Dowling (2016)
Copenhagen (Denmark)	2 Kinder-gardens	15 cm soil removal; Clean Fill	90-190 mg/kg	Nielsen and Kristiansen (2005)
Chicago, Illinois (USA)	62 Homes	Various yard treatments	478 to 698 mg/kg	Binns et al. (2004)
Cleveland, Ohio (USA)	200 Homes	Various yard treatments	unknown	Clark et al. (2011)
Boston, Massachusetts (USA)	Homes	Various yard treatments	unknown	Dixon et al. (2006)
Boston, Massachusetts (USA)	43 Homes	Various yard treatments	unknown	Hynes et al. (2001)
Boolaro, NSW (Australia)	>2000 homes	Various yard treatments	unknown	Harvey et al. (2015)

were intended to reduce soil Pb exposures but did not measure children's BLLs.

Multiple studies demonstrated declines in soil Pb as a result of intervention. For example, Mielke et al. (2011b) enacted a low cost solution to isolate Pb contaminated soils at 10 day care centers in New Orleans, Louisiana. Mielke et al. (2011b) tested the feasibility of reducing children's exposure to Pb polluted soil in New Orleans. The soil emplacement included first spreading out a bright orange, water pervious polypropylene geotextile material to cover the contaminated soil in the play area. Geotextile has a service life of 100 years (TEPPFA, 2014). On top of the geotextile, at least 15 cm (6 inches) of Pb-safe soil (median concentration of 5 mg/kg) from the Bonnet Carré Spillway, located up-river from New Orleans, and placed on top of the geotextile fabric and planted with grass. The initial 558 mg/kg median soil Pb (range 14–3692 mg/kg) decreased to median 4.1 mg/kg (range 2.2–26.1 mg/kg) after intervention with geotextile covered by 15 cm of river alluvium. Lead loading decreased from a median of 4887 $\mu g/m^2$ (454 $\mu g/ft^2$) range 603–56650 $\mu g/m^2$ (56–5263 $\mu g/ft^2$) to 398 $\mu g/m^2$ (37 $\mu g/ft^2$) range 86–980 $\mu g/m^2$ (8–91 $\mu g/ft^2$).

In Kabwe, Zambia, a city built around a now closed but formerly quite productive Pb mine and smelting operation, Pure Earth also used the polypropylene geotextile covered with a clean soil cap method to isolate soils at 78 homes selected for encapsulation and cleaning in 2015, in an effort to reduce extremely high soil Pb concentrations (routinely exceeding 1000 mg/kg (Ericson and Dowling. 2016). Ericson and Dowling (2016) stated that "The teams first laid a geotextile barrier cloth (Fibertex F25) at each home where appropriate, avoiding areas where lead is already encapsulated (i.e. paved or gravel driveways, stone walkways, densely packed grass areas, etc.). This geotextile was then covered with approximately 10–15 cm of clean laterite soil. In addition to the yards, a frequently used and highly contaminated pathway between the project area and nearby roads, schools, and churches was covered with clean soil." Clean laterite soil (<25 mg/kg) was sourced locally.

In the mining town of Broken Hill, Australia, Boreland and Lyle (2006) initiated a holistic home remediation program to reduce children's exposure to Pb by interrupting all identified Pb exposure pathways in 117 family homes, 103 of which had children with BLLs between 15 and 30 mg/dL. Boreland and Lyle (2006) indicated that the remediation activities included dust removal "….from ceilings using a vacuum cleaner fitted with a HEPA filter; gaps and cracks in wall and ceiling linings and around windows and doors were sealed with silicon sealer; flaking or powdery lead paint inside or outside the home was stabilized by encapsulation or removal; and yard soils with high lead levels were capped or removed. Structural works (for example, replacing a ceiling or window) were also carried out if this was considered necessary." The results indicated that the benefits of remediation were unevenly distributed among homes, with those having the highest pre-remediation Pb loadings benefiting the most and those with the lowest pre-remediation loadings benefiting only slightly or not at all. Overall, Boreland and Lyle (2006) found that remediation reduced Pb levels in homes with mean post-remediation Pb loading about half the pre-remediation loading, and remained low for the duration of the 10-month follow-up period. Boreland and Lyle (2006) concluded that holistic remediation is a useful strategy for reducing Pb risk for children living in homes with high indoor Pb levels.

Nielsen and Kristiansen (2005) conducted a soil intervention that consisted of replacement of contaminated top soil from the most intensively used playground areas and coverage of bare soil with wood chips or grass. Nielsen and Kristiansen (2005) included children from three kindergartens: one with very low levels of Pb in soil and two kindergartens with an average Pb concentration in soil of 100–200 mg/kg. Measurements of lead in soil 5–7 weeks after interventions in two kindergartens verified that the interventions had effectively reduced the potential exposure to Pb from the most intensively used areas of the playgrounds. The average Pb concentration in soil after intervention was below 10 mg/kg. Nielsen and Kristiansen (2005) found a good agreement between the average concentration of Pb in soil and the amount of Pb on the hands of the children. They concluded that the soil Pb intervention was a cost effective method to reduce the amounts of Pb on the hands of children.

Binns et al. (2004) evaluated the effectiveness of the application of phytostabilization techniques designed to reduce soil lead exposures in 62 homes in Chicago with high baseline soil Pb contamination. Design strategies varied by property. If lawn areas were in poor condition, the existing vegetation was killed using Roundup and the soil was tilled. Clean topsoil was added if the soil surface was below grade. Soils were reseeded and new grass was grown. Binns et al. (2004) stated that "Shady areas or areas too small to mow or not conducive to grass growth were prepared and planted with a perennial groundcover (vinca); then hardwood mulch was applied. Foundation perimeter soil areas and areas under porches with lead content of >2000 ppm were prepared and covered with a landscape fabric and a plastic edging placed before application of 2 in of rotten granite (a red-colored compactable stone). An alternative strategy for areas under porches was to make these areas inaccessible using lattice barriers. Foundation perimeter soil and area under porches with lead content >2000 ppm received treatments similar to the rest of the property. Garden areas received a hardwood mulch covering." The study found that after a 1 year period, the soil treatments lowered the acute hazard soil Pb level and reduced track-in of Pb onto floor mats by 50%. The results of the study indicated that the success of the various techniques to reduce Pb hazards required continued maintenance.

Clark et al. (2004) aimed to understand the contribution of exterior dust/soil Pb to post intervention interior dust Pb in a subset of housing from the HUD Lead-Based Paint Hazard Control Grant Program Evaluation that included housing from 12 state and local governments. Clark et al. (2004) observed that housing where soil Pb hazard control activities had been performed had lower post intervention exterior entry, interior entry floor, windowsills, and other floor dust loading levels. Statistical analysis revealed that exterior strategy influenced soil Pb concentration, and soil Pb concentration influenced street dust Pb loading.

Clark et al. (2011) conducted various soil Pb exposure reduction treatments in 200 homes in Cleveland and assessed the longevity of the treatments 7 years later. The treatments generally consisted of re-seeding the soil or covering the soil with plastic sheeting covered with mulch/wood chips or covering the soil with gravel. Fourteen factors that could possibly contribute to longevity of treatment integrity were assessed. Hazard control method and the presence/absence of shade were the only factors found to significantly affect visual failure rates. Of the three most commonly used control measures, the lowest visual failure rate was for re-seeding, 29.1% after a mean of 7.3 years; for non-shaded areas, which had been re-seeded, the failure rate was 22.2% compared to 35.7% for shaded areas. Geometric mean soil Pb concentrations were statistically significantly higher in the areas observed to have a visual treatment failure ($p < 0.003$). Duration of the treatment did not have a significant impact on visual failure of treatment.

Dixon et al. (2006) completed a project, "*The Boston lead safe yards low cost lead in soil treatment, demonstration and evaluation*". The purpose of the study was to evaluate the effectiveness of low-cost soil treatments to reduce soil and dust Pb hazards. These treatments included establishing dripline boxes, stepping stone paths, grass sod/seeding (lawn improvement), wood-raised garden plot, wood-raised play/picnic area and establishing gravel drive/path gravel parking areas. This study was conducted at properties where the buildings had been previously abated or de-leaded to Massachusetts's standards in the 5 years before enrolment. The hypothesis was that reducing soil lead levels and/or improving the condition of the yard surface cover may lessen direct and indirect exposure to soil Pb. The treatment cost ranged from $1095 to $5643 with an average cost of $2798. Dixon et al. (2006) reported that "Most of the barrier treatments continued to block access to the Pb -contaminated soil at 1 year. At the follow-up, few properties with grass treatment had areas that were completely bare, but 28% had more than a small amount of bare areas. Treatments were effective in reducing entryway dust Pb in the rear of the building if the residents reported they had maintained the yard treatments. Each additional yard work activity reported was predicted to lower 1-yr floor dust Pb loading at the rear common/main and dwelling unit entries by about 20%. Each additional 100 ft^2 [9.3 m^2] of yard treated was predicted to lower 1-year floor dust loading at the rear dwelling unit entry by 19%. Treatments did not show a dust Pb effect at 1 yr in the front entryway of the building, but the investigators suggested that this may be due to the effect of resident cleaning overshadowing the treatment."

Hynes et al. (2001) developed a series of *in-situ*, low-cost, low-technology measures that worked to reduce the exposure to lead-contaminated soils in 43 homes in the Dorchester neighbourhood of Boston, Massachusetts. Approximately 10% of the children in the Dorchester neighborhood exhibited BLLs exceeding 10 μg/dL. The project design was guided by the aims of the USEPA EMPACT (Environmental Monitoring for Public Access and Community Tracking) program (USEPA, 2001). Various yard treatments were used based upon soil Pb concentrations. The treatments included: semi-permanent barriers including wood-framed drip-edge boxes with a perforated landscape cloth or plastic material underlayment to create a permeable soil cover and filled with gravel or another material, such as mulch or wood chips (10–15 cm deep); installation of a path of walking stones for areas of high foot traffic; seeding and fertilizer treatment of grassy areas or covering with mulch or wood chips if not a suitable site for grass. Recommendations were also made for relocating play and picnic areas away from areas with high soil lead. No follow-up measurements were made to evaluate whether the program actually reduced Pb exposures.

Harvey et al. (2015) evaluated a soil Pb abatement strategy performed at a large number of properties near a former smelter located in Boolaroo, New South Wales, Australia. They evaluated the soil Pb concentration on a subset of the remediated properties before and after remediation. Apparently, confirmatory surface soil samples (soil validation) were not collected following the soil Pb treatments in the soil Pb abatement program. The results indicated that there was no difference in the soil Pb concentration before and after intervention. Harvey et al. (2015) concluded that the strategies employed by the Pb abatement strategy of shallow application (sprinkling) of soil through the existing grass cover, mulching or grass covering were insufficient to mitigate the risk of Pb exposure in the long term.

2.3. Limitations in evaluating soil lead interventions

One of the limitations for evaluating the effectiveness of soil Pb interventions on BLL is storage of Pb in bones; Pb stores can continue to contribute to BLL in the absence of further exposure. Bone lead may explain the seemingly contradictory fact that slow rates of decline in BLLs occur following Pb hazard interventions (Rust et al., 1999). Another factor that may give the appearance that soil lead intervention has been ineffective is that most studies focus on interventions of individual homes and not to surrounding properties. The problem is that this strategy neglects the pathways of soil resuspended from other properties in the surrounding community. In the Bunker Hill superfund site, structural equations models indicated that from 40 to 50% of the BLL absorbed from soils and dusts is through house dust with approximately 30% directly from community-wide soils and 30% from the home yard and immediate neighborhood (von Lindern et al., 2003b). For example, in Bunker Hill the homes with the highest risk were remediated first and this led to a reduction of dust Pb exposure from the soils at an individual home. However, when the surrounding homes underwent remediation, the Pb in the dust and the children's BLLs all gradually fell as neighbourhood dust Pb declined. The poor results from the soil Pb interventions in Boston (Aschengrau et al., 1994; Weitzman et al., 1993) and Baltimore (Farrell et al., 1998) may have resulted from the very high soil Pb levels at surrounding properties that resulted in a high rate of soil dust recontamination in the homes from the un-remediated properties. This same observation was reported by Clark et al. (2008) in Roxbury and Dorchester, Massachusetts, USA. It is possible that this contradictory result could arise in areas with very high soil Pb levels (>1000 mg/kg).

3. Other soil lead remediation strategies

3.1. Phytoremediation

Phytoremediation covers a range of plant-based technologies for use in the remediation of contaminated soils and water (Rahman et al., 2015). Of particular relevance to Pb-contaminated soils are phytoextraction (use of plants to absorb and translocate lead to above ground tissues which are harvested and removed) and phytostabilization (use of plants to stabilize and safely manage Pb *in-situ* so as to reduce the risk to human health and the environment).

Phytoextraction utilises standard agricultural approaches to grow 'crops' of plants that are harvested to remove the metals from the soil. For phytoextraction to be effective, a number of criteria need to be met including a combination of high accumulation of Pb in above ground tissues and high biomass to ensure adequate Pb

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removal from the soil and fast plant growth to reduce the overall remediation time (Rahman et al., 2015; Blaylock, 2000). In most circumstances, multiple crop rotations are required to remove enough metal from the soil to reach acceptable concentrations, therefore phytoextraction can take years to be completed (e.g. Blaylock, 2000; Salt et al., 1995). There are few candidate species for use in Pb phytoextraction due to the low bioavailability of Pb in soil restricting plant uptake. While 14 Pb hyperaccumulating species have been identified, there is concern the high tissue concentrations were due to growing under artificial conditions with high Pb solubility (hydroponics) or, in the case of field grown plants, that sample contamination from aerial deposition or soil splashing rather than true plant accumulation was the reason for the elevated Pb concentrations (van der Ent et al., 2013). Thus, confirmation of hyperaccumulator status is still required for these species. The use of chelates and other organic compounds to solubilize soil Pb, and thus, increase bioavailability and plant uptake has also been investigated. While this approach has proven successful in increasing Pb uptake by plants in field demonstrations (Blaylock, 2000; Butcher, 2009), there has been strong criticism due to the pollution potential from the solubilized Pb and chelates leaching into ground- and surface-waters (Evangelou et al., 2007; Chaney et al., 2007). In addition, the use of chelates at levels that adequately solubilize soil Pb is very expensive at the scales required for large-scale remediation of urban areas (Chaney et al., 2007). At this stage, it does not appear to be a feasible approach for remediating soils contaminated with Pb.

In comparison to phytoextraction, phytostabilization is a commonly used and accepted strategy for part or all of the Pb remediation approach (e.g. Binns et al., 2004). Ground cover serves as a physical barrier that reduces exposure via dermal, inhalation, and ingestion pathways by preventing erosion and dust resuspension from the Pb contaminated soil (Butcher, 2009). Phytostabilization provides a cost effective and low resource approach to Pb remediation but does require ongoing maintenance of plants and the soil to be effective (Butcher, 2009).

3.2. Phosphate treatment

The addition of phosphate in various forms has been suggested as a remedial solution to the soil Pb problem. Phosphate binds with lead to form the mineral pyromorphite which is the most insoluble form of Pb in soil and reduces the bioaccessibility of soil Pb (Hettiarachchi et al., 2000; Scheckel et al., 2013). One method of reacting phosphorus with Pb in contaminated soil involves rototilling pulverised fish bones with Pb contaminated soil. The USEPA conducted a pilot trial in the Prescott neighborhood of West Oakland California using this method in 40 homes with average soil Pb concentrations of 843 mg/kg (Freeman, 2012; USEPA, 2012). No studies have been identified that evaluated whether bioaccessibility of Pb was reduced in the West Oakland California soils. Phosphorus addition to soils converts soil Pb to pyromorphite more quickly if the soil pH is acidic (Ryan et al., 2004; Brown et al., 2004). The acidity solubilises the Pb and allows it to react with the phosphorus. A field trial in Joplin, MO found that the rate of pyromorphite formation was more rapid when phosphoric acid was added to soil in comparison to calcium phosphate (Ryan et al., 2004). This suggests that a two-stage remediation would be required - with the first stage to acidify soil and add the phosphorus and the second phase to add limestone to make the soil alkaline. This can be problematic and expensive. There is some work to suggest that if both phosphorus and Pb are present in ingested soil, pyromorphite will precipitate in the gastric system and lower the bioavailability of the ingested Pb (Scheckel and Ryan, 2003). Henry et al. (2015) reviewed many issues associated with the application of phosphate in urban Pb contaminated soils and concluded that more data are needed to better understand variability of bioavailability and using in-situ remediation for better risk-based decision making, particularly in urban areas.

One of the concerns of establishing high concentrations of phosphorus in soils across urban landscapes is that runoff water from soils may have the potential of contaminating nearby surface waters resulting in algal blooms and eutrophication (Conley et al., 2009). In addition, current phosphate application methods involve rototilling the phosphate into the soils which would require considerable time and effort if applied to vast tracts of urban Pb contaminated soils. Mielke (2016) estimated that the cost of applying fishbone phosphorus methods costs were approximately $\$194/m^2$ while placing a geotextile fabric covered with clean soil costs approximately $\$22/m^2$.

Phosphorus is essential for maintaining the productivity of agriculture and it is predicted to peak in 2030 and then decline (Cordell et al., 2009). Coupled to its value to food productivity and global food security, the suitability of phosphorus as a long term strategy for mitigating the soil lead risk is questionable. This was highlighted in the work of Chrysochoou et al. (2007) who suggested using phosphorus containing wastes as a more sustainable source for Pb immobilization than sources used in agriculture.

3.3. Biosolids and composts

In theory, the cycle of transport of agricultural produce to consumers, with associated wastes that are disposed of through solid waste or wastewater, can be cycled back to soils for both residual fertility and as a source of organic matter. This process can take place with both food waste and the solids produced from wastewater treatment. The latter, when appropriately treated, are referred to as biosolids. All cities generate large amounts of residual materials on a daily basis that can produce high quality soil amendments. In particular, each person generates about 30 kg annually of municipal biosolids as solid residual from wastewater treatment. Food scraps and yard waste, feedstocks for compost, are also generated in significant quantities (approximately 50 kg per capita annually) (King et al., 2011). Currently about 50% of the biosolids that are generated in the US are land applied, with only a small fraction of that used within the population centers where they are produced (North East Biosolids and Residuals Association, 2007). While about the same proportion of yard waste is diverted from landfills and composted, currently over 95% of all food scraps generated are landfilled. Soil products generated from these materials must meet standards for pathogen elimination and metal concentrations before being distributed to the general public. Because of dilution, personal care products and pharmaceuticals are a minor fraction of the waste compared with the large fraction of other organic material in biosolids. The Pb concentrations in biosolids have decreased over time as active sources of Pb underwent reductions. A study found median and 95th percentile concentrations of Pb in biosolids generated in Pennsylvania in 1996–7 of 65 and 202 mg kg^{-1}, respectively (Stehouwer et al., 2000). Many changes have taken place that has substantially reduced the amount of lead in biosolids as demonstrated in Tacoma, Washington, these include including banning of Pb in paints and gasoline and closing a local smelter in Tacoma (McIvor et al., 2012; Stehouwer et al., 2000).

Research has tested biosolids amendments for their ability to reduce the bioaccessibilty of soil Pb *in situ*. The studies have shown that in certain cases, adding biosolids composts to soil can change the mineral form of soil Pb reducing bioaccessibilty (Brown et al., 2003, 2004, 2012; Farfel et al., 2005). The observed reductions in bioaccessible Pb is related to the absorption of Pb on high surface area Fe oxides (Brown et al., 2012). Biosolids have been added to Pb contaminated soils in urban gardens, smelter contaminated soils, and mining wastes to restore a plant cover, dilute total Pb concentrations and reduce bioaccessible Pb. Composts, both biosolids and food and yard waste-based, have also been shown to reduce total soil Pb and Pb accessibility (Attanayake et al., 2014; Defoe et al., 2014).

Land application of biosolids is highly cost effective, often less expensive than alternative disposal options. Despite this, very few urban areas have taken advantage of this approach in efforts to reduce hazards associated with soil Pb contamination. There was a trial program in Baltimore where the high Fe biosolids composts were added to Pb contaminated soils in 9 home gardens, which reduced the bioaccessibility of lead in soil Previous work with these materials had shown them to be very effective in reducing Pb bioaccessibility (Brown et al., 2003, 2016; Farfel et al., 2005). Changes in Pb bioaccessibility have been observed in both lab tests and in vivo studies. The in vivo studies have been done with rats as the animal surrogate. Phosphorus amendments were also tested with the biosolids composts in the Joplin, MO soil feeding trail and similar reductions in bioaccessibility were observed between the biosolids compost and phosphorus. The P-based amendments were subsequently tested in trials involving swine and humans and demonstrated pronounced reductions in Pb bioavailabilty (Brown et al., 2003, 2004; Ryan et al., 2004)

Tacoma, Washington has a history of Pb and arsenic contamination from a metal smelter with a significant portion of the city listed on the U.S. EPA Superfund National Priority List. Biosolids from the city of Tacoma waste treatment plant yield a high organic matter (ranging from 25 to 50% for the general mixture, to potting soil, and topsoil) and low metal containing soil products known as Tagro (City of Tacoma Washington (2016)). The analytical results of Tagro indicate that it contains ~13 mg/kg lead and 1–3 mg/kg arsenic (see Tagro supplement). Tagro is the remediation material for the soil metal contaminated community within and surrounding the smelter. In addition, for several decades the biosolid-based soil products were available to home gardeners and used extensively. An urban agriculture program started in Tacoma provides biosolids soil products and materials to gardeners to construct raised bed gardens free of charge. Cardboard provides a barrier to the underlying contaminated soil with mulched wood waste diverted from the solid waste stream to cover the cardboard. Costs of supplying the biosolids and other materials are borne by the municipality. The Tacoma program has expanded from four to over 70 gardens.

3.4. Biochar

Biochar is found in archaeological digs that are hundreds of years old and was used by cultures living in the Amazon area to fertilise the agricultural lands or, “Terra Preta” soils, and they remain stable to the present (Lehmann et al., 2015). The modern counterpart of biochar is an engineered product of biomass heated in the absence of oxygen (pyrolysis). It has garnered attention as a way to manage waste, sequester carbon, amend soil and control pollution. The intended end use of a biochar governs the choice of feedstock and the heating conditions used.

Biochar has been used in many instances to remediate metal-polluted soils (Paz-Ferreiro et al., 2014), including Pb. Much of the research conducted in Pb polluted soils with biochar has been done at shooting ranges, where co-contamination with other metals, including tin and copper occur (Moon et al., 2013; Ahmad et al., 2014). As an example, Ahmad et al. (2014) found the organic bound Pb transformed into the more stable Pb-phosphate following amendment with oak biochar. Moon et al. (2013) applied different concentrations of a soybean stover (forage including stems and leaves) derived biochar to a shooting range soil. They found a reduction of over 90% in Pb leachability at the highest dose of amendment (20% biochar). Lead was immobilized as a consequence of a shift in its speciation. In particular, lead precipitated as Pb-hydroxide as a consequence of an increase in soil pH and was transformed into chloropyromorphite, one of the most stable Pb compounds in soil. Park et al. (2011) compared the performance of two biochars, derived from greenwaste and from manure, on Pb immobilization, monitoring NH_4NO_3 extractable metal concentration. They found chicken-manure biochar to be more effective, with a reduction of 93.5% in the concentration of extractable Pb when compared to the control (polluted soil). The temperature of pyrolysis is one of the main drivers in its ability to immobilize lead. Uchimiya et al. (2012) found chicken litter biochar prepared at low temperatures to immobilize larger amounts of Pb than biochars prepared at higher temperatures. This was attributed to a change in soluble phosphorus concentration.

In spite of some positive results, and a promising prospect for biochar in the abovementioned contexts, studies are lacking concerning lead immobilization by biochar in an urban setting. As mentioned before, most of the studies have been conducted in shooting ranges, where Pb is primarily in the form of hydrocerussite ($Pb_3(CO_3)_2(OH)_2$), cerussite ($PbCO_3$) and massicot (PbO) (Hardison et al., 2004).

Considering the important role of phosphorus in Pb immobilization, ash-rich biochars are prime candidates for Pb immobilization in polluted urban landscapes. Ash-rich biochars, as those prepared from manures have moderately high phosphorus contents when compared to other biochars. As an example, Cely et al. (2015) analyzed biochars prepared from different manures and found phosphorus contents ranging from 1.8 to 3.6 g kg^{-1}, an amount of phosphorus comparable to that in the feedstock. Alternatively, biochars prepared from other waste materials could be used to recover phosphorus from wastewater effluent (Sheperd et al., 2016) and then applied to the land. The potential for eutrophication of nearby surface waters from excess phosphorus runoff is another drawback of using traditional phosphates, a risk averted by using ash-rich biochars which act as a slow-release fertilizer (Wang et al., 2014).

As a means to manage and remediate Pb contaminated soil, biochar would bring further environmental improvements associated with carbon sequestration. On the other hand, biochar can contain PAHs and low concentrations of metals, but with thoughtful choice of feedstock this issue can be minimized or avoided. Moreover, the availability of metals tends to be less than in the feedstock (Paz-Ferreiro et al., 2014).

3.5. Nanoparticles

Metallic nanoparticles have been used to detoxify Pb in contaminated environments, however many metallic nanoparticle species are potential environmental pollutants (Maurer-Jones et al., 2013). Iron nanoparticles are currently used for site remediation of a number of different organic and inorganic pollutants including metals (Zhang, 2003). However, it is unlikely that iron nanoparticle complexes will be used to reduce the bioavailability of Pb in urban soils.

Iron, the fourth most abundant element in the earth's crust, is predominantly found in an iron oxide form (Skinner, 1979). Reactive nano-scale zero valent iron (nZVI) is the preferred form of iron for site remediation and is highly susceptible to transformation to iron oxides in aerobic environmental conditions. Therefore, its persistence and toxicity is seen as low and is suitable for groundwater remediation (Cook, 2009). However, for terrestrial environments where particles are not easily contained, the risks of exposure are elevated and the potential health implications are not currently understood. Although iron itself has not been considered toxic, nanoparticulate matter has sparked concerns with inducing reactive oxygen species and cancer within mammals (Apopa et al., 2009).

Nanoparticles can employ a number of mechanisms to transform metal pollutants. The route of detoxification/environmental stabilization is dependent on the standard redox potential (E^0) of the metal contaminant compared to that of iron. Iron's standard redox potential is −0.44 V. Absorption of metal pollutants to iron will occur if their E^0 is significantly lower than that of iron. Alternatively, significantly higher E^0 values will lead to the reduction and subsequent precipitation of the metal contaminant (O'Carroll et al., 2013).

Lead is a species which possesses an E^0 which is only slightly more positive than Fe^0 (−0.13 and −0.44 V respectively). In this case, it can be either transformed (reduced) or absorbed and subsequently converted into less toxic forms. Nano-scale zero valent iron rarely exists as pure Fe^0 and in most cases has an outer shell composed of oxides/hydroxides which aid in providing sites for chemical complexation of metals (Sh et al., 2011). Furthermore, iron oxides/hydroxides have been shown to co-precipitate with Pb preventing their solubility in soils. However, in highly acidic conditions and/or in the presence of organic ligands i.e. plant root exudates, Pb can dissociate from iron complexes and become bioavailable once more (Martinez and McBride, 2001).

At pH's below 4.5, Fe^0 starts to dissolve, iron nanoparticle/Pb interactions become weak and the Pb can dissociate and become bioavailable once more (Zhang et al., 2013a,b). However, above this threshold, Fe^0 is able to reduce Pb^{2+} to insoluble Pb^0 via the following pathway (Tehrani et al., 2015).

$$2Fe^0 + 3\,Pb^{2+} + 4H_2O \rightarrow 2FeOH + 3\,Pb^0 + 2H^+$$

A study by Gil-Díaz et al. (2014) investigated the ability of nZVI to immobilize Pb from acid soils (pH 5.3). Results showed that nZVI nanoparticles were capable of reducing Pb leaching from soils by 98% (Pb concentration of 9 mg/kg soil). Furthermore, Pb remained immobilized under conditions typical of 2-year annual rainfall from central Spain.

Liu and Zhao (2007) investigated phosphate stabilized iron nanoparticles (vivianite) to reduce Pb^{2+} leachability in acidic soil (pH 4.32), neutral soil (pH 6.93) and calcareous soil (pH 7.85). Findings showed that using a P:Pb molar ratio of 1.8–2.2 in acidic soil, neutral soil, and calcareous soil the Pb leachability was reduced by 56%, 30% and 26%, respectively,. Using a higher P:Pb molar ratio of 9.0–11 showed that in acidic soil, neutral soil, and calcareous soil the Pb leachability was reduced by 61%, 55% and 19%, respectively.

The introduction of iron nanoparticles has been shown to increase the pH of acidic soils and in turn promote Pb^{2+} absorption and precipitation. The increase in pH stimulated microbial respiration of the soil, the addition of nanoparticles also increased the soils water holding capacity while porosity was not significantly altered. Soil physiochemical properties were not significantly altered and may be seen as slightly enhanced (Liu and Zhao, 2007).

Although there is clear evidence that iron nanoparticles, particularly nZVI particles, have the capacity to detoxify free Pb ions from soil and solutions, the potential for this technology to be used *in situ* may not be viable (Karn et al., 2009). Site remediation of Pb by iron nanoparticles in groundwater has been estimated to be comparative in cost to current treatment methods (Adeleye et al., 2016); however Crane and Scott (2012) suggest that the cost of nZVI for site remediation needs to be reduced by at least 20% to give them the competitive edge required to be considered viable.

For the remediation of terrestrial environments, uncertainties in relation to health implications of nanoparticles on mammals raises concerns. In particular, the inhalation of airborne nanoparticles and their capacity to initiate cancers is of major concern (Apopa et al., 2009). Also, the potential for iron nanoparticles to enter aquatic environments via wind or erosion is of concern as nZVI have been implicated with impaired gill function, decreases in plasma protein levels and the production of reactive oxygen species in fish (Saravanan et al., 2015).

3.6. Soil resuspension control

One of the persistent trends observed in Pb poisoning is that children's BLLs peak in the dry summer months when Pb contaminated soils are resuspended into the air and minimized in the winter months when soil resuspension is reduced (Laidlaw et al., 2012, 2014, 2016; Zahran et al., 2013a,b). In Detroit Michigan, Zahran et al. (2013a,b) demonstrated that air lead, atmospheric soil and children's BLLs were highly correlated with seasonal shifts between dry summer months and wet winter months. Soil moisture is the dominant controlling factor regarding whether or not soils are resuspended (Chen et al., 1996; Clausnitzer and Singer, 1996, 2000; Nickovic et al., 2001; Cornelis and Gabriels, 2003; Hoffmann and Funk, 2015). Maintaining high soil moisture levels during the summer months via lawn watering in areas where soil is most lead contaminated, should minimize soil resuspension, and the summertime peaks of elevated BLLs reduced considerably. Soil moisture maintenance could be used as an interim exposure control method implemented prior to the installation of more permanent soil isolation strategies. In many cases watering systems are already part of the infrastructure of urban communities. A limitation of this strategy is that the direct soil contact pathway and the tracking of Pb contaminated soils on shoes and pets would not be eliminated. Also, a major consideration is the lack of water availability in some regions. No research trials have been conducted to evaluate this Pb exposure reduction strategy.

4. Discussion

The standard method of soil Pb reduction involves excavating the contaminated soil and replacing it with "clean" soil and growing grass or another cover crop on the surface. This method is extremely expensive and implementing it over large areas of residential communities in cities is prohibitive. The presentation of the case studies of the soil isolation/excavation programs above indicate several possibilities that soil isolation at the scale of the community can meet the criteria for effectively reducing soil Pb contamination including a margin of safety that prevents Pb exposure by children.

4.1. The clean soil remedy for lead contaminated soil

New Orleans provides a case study whereby a natural event occurred that decreased soil Pb and reduced the BLLs of children at the scale of an entire city (Mielke et al., 2016). Eighty percent of the near sea-level city was flooded by a storm surge from Hurricane Katrina that deposited large amounts of low Pb (~5 mg/kg) coastal sedimentary material into many communities. In addition, after the hurricane rebuilding the city required contractors to transport soil into the city to elevate the ground before constructing new buildings (Mielke et al., 2016). Ten years post-Katrina the soil and BLL of children demonstrated remarkable reductions compared to the situation before Hurricane Katrina. In this case, the addition of low Pb (5 mg/kg) soil only on the Pb contaminated surface appears to be sufficient alone to reduce the Pb dust available for ingestion and inhalation by children.

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The alluvium is available to the city because erosion occurring on the vast agricultural lands of the Mississippi watershed carries the soil as river sediments through New Orleans at an average rate of about 300 metric tons per minute (Mielke et al., 2006). Thus, there are enormous low Pb soil resources available to the city for continuing community-scale soil Pb intervention, especially in the inner city where unhealthy BLLs remain (Mielke et al., 2016).

Importantly, all U.S. cities have low Pb (geometric mean ~ 16 mg/kg) soil in areas outside the city, and the situation is not unique to New Orleans (Gustavsson et al., 2001). Because of this fact, every city has the possibility of changing, diluting, or isolating Pb contaminated soils caused by historical development of roadways, housing, and industries and the inequitable chronic Pb exposure suffered by communities of vulnerable citizens (Leech et al., 2016).

4.2. Other approaches to intervene contaminated soil lead exposure

Based on the background evaluation of the children's Pb exposure situation and a review of the soil intervention literature there appear to be strategies that are similar or maybe even more protective than clean soil only for advancing effective, low-cost reduction of soil lead and children's exposure.

4.2.1. Biosolids

The use of biosolids to remediate metal contaminated soil has desirable characteristics. It uses materials that are continuously created and in ready supply to provide soil for covering soil and gardening. The Tacoma program has expanded from four to over 70 gardens. This is an example of a cost-effective program that uses materials readily available to all cities for reducing hazards associated with Pb contaminated urban soil and provides the municipality with an option for productively recycling its continuous output of biosolid residuals. However, it must be noted that there a danger that biosolids may contain other toxins that may not be desirable. In addition, some regions may have sewage/storm runoff with higher metal concentrations than others due to inputs from local industries or variable traffic density, therefore biosolids from different regions may have different contaminant levels that would have to be tested and carefully evaluated prior to usage.

4.2.2. Clean soil over geotextile

The strategy covers lead-contaminated soils with water pervious polypropylene geotextile fabric and place on the surface a clean soil cap approximately 15 cm (6 inches) thick. The method is remarkably rapid to employ and ideal for childcare centers and public playgrounds. The geotextile and soil cap method isolates the soil lead surface to prevent both ingestions and inhalation exposure pathways. This technique has been used by Mielke et al. (2011a,b) in New Orleans, Ericson (2014) in Vietnam and by Ericson and Dowling (2016) in Zambia. The geotextile and soil cap method does not require continual maintenance. Compared with dig and haul methods the cost of geotextile and soil cap is much lower at, ~\$22 per m^2 versus ~\$388 per m^2 (Mielke, 2016). There are limitations to the geotextile and cap technique. Future generations and residents may not be aware of the presence of Pb contaminated soil beneath the geotextile layer and some type of intergenerational education would be required to notify future residents about the risks of digging beneath the geotextile layer. The geotextile and cap method may require periodic maintenance when breaches appear in the protective layer for various reasons. Furthermore, large-scale implementation of this method in urban areas and long-term studies using this method has not been undertaken.

4.2.3. Interim yard treatments

Another relatively low cost strategy involves implementing various interim yard treatments (Hynes et al., 2001; Binns et al., 2004; Clark et al., 2004, 2011; Dixon et al., 2006). Their soil intervention consists of various yard treatments such as:

- reseeding or tilling old soil and establishing new grass cover;
- covering contaminated with a thin layer of low Pb soil on top of the existing grass cover;
- covering soils with plastic or landscape fabric and wood chips; covering soils with gravel;
- establishing dripline boxes to cover the Pb contamination in soils deposited from paints along the sides of homes;
- adding stepping stone paths to prevent the accumulation of soil on shoes and boots and grass sod/seeding (lawn improvement);
- creating raised-bed garden plots;
- emplacing a play/picnic area, and;
- establishing gravel drive/path gravel parking areas to prevent direct exposure to soil.

The main limitation of the various interim yard treatments is that they generally require continuous maintenance, which means continued cost over time. Also, by not including larger areas of contaminated land they offer only a partial reduction in soil lead exposure (Binns et al., 2004; USEPA, 2001). They are unlikely to reduce the “summer disease” increases in BLLs.

4.2.4. Phosphate treatment of contaminated soil

The soil lead remediation methods of applying phosphate amendments has limitations. An important limitation is that rototilling of vast tracks of land requires considerable time and cost. The method assumes that the formation of pyromorphites will reduce Pb exposure but the time for formation may be years. Another objection is that some of the sources of phosphorus such as fish bones are extremely malodorous and the solution is to add a layer of clean soil on the layer of fish bones. Furthermore, eutrophication is also an important consideration, especially with phosphate amendments, and phosphorus is in limited supply and needed for agricultural uses. Another limitation is that these three methods only partially reduce Pb bioaccessibility.

4.2.5. Nanoparticle treatment, phytoremediation, and soil moisture maintenance of contaminated soil

Nanoparticles are severely limited due to potential toxicity and cost; phytoextraction is questionable because currently there are no known plants that both hyperaccumulate Pb and simultaneously produces large enough amounts biomass to extract lead from soils (Rahman et al., 2016). Maintaining soil moisture in areas contaminated with high soil Pb levels during dry seasons may assist in reducing exposures associated with seasonal soil resuspension, but is limited by water shortages. Also it does not eliminate the home interior exposure pathway via track in of lead in soil dust from shoes and pets. The method leaves the contaminated soil in place. Large-scale study has not been undertaken.

4.3. Recommendations

Comprehensive urban soil Pb risk intervention requires understanding processes of contamination and mapping communities to show the risk of Pb exposure. The soil Pb mapping sampling protocols of Mielke et al. (2005) and the British Geological Survey (BGS) are well established (BGS, 2016a). Using the Mielke protocol, 19 soil samples are collected within each census tract for four types of residential sites where children could play: four soil samples are collected from within 1 m the public right-of-way of the busiest roads of the census tract; nine samples are collected on the public right-of-way within 1 m of ordinary residential streets; three samples are collected from open spaces away from both houses and streets; and three samples are collected within 1 m of house foundations (Mielke et al., 2005). In this collection protocol, the soil sample density per area is directly proportional to the density of the population (Mielke et al., 2005). The BGS protocol entails sampling on a grid with a density of four samples per square kilometre independent of the population density (Johnson, 2005). This method has been applied to 22 cities in the United Kingdom (BGS, 2016b). However, the Mielke and BGS protocols provide large amounts of information and are not excessively inexpensive even though they entail collecting, acid digestion of the soil samples, and analysis via Inductively Coupled Plasma – Atomic Emission Spectroscopy (ICP-AES). A lower cost method is using portable X-Ray Fluorescence (XRF) analysis of soil samples directly in the field (Rouillon and Taylor, 2016). In New Orleans, samples collected along residential street-sides correlated well with BLLs of children living in the same communities (Zahran et al., 2013b). With XRF large numbers of soil samples can be analyzed relatively accurately and rapidly and converted into soil lead risk maps that can be posted on a website to inform the public of potential Pb exposure risks of lead in the soil environment. However, it must be noted that one of the limitations of soil lead measurements using the XRF is that soil lead measurement may not be accurate when soil moisture levels are elevated. If XRF measurements are made in the laboratory under controlled conditions (sieved fractions, dried, etc.), then more accurate measurements can be made compared with XRF sampling in the field.

Empirical soil Pb and BLL dose response studies have been carried out by Bickel (2010) in Detroit and Zahran et al. (2011) in New Orleans and the IEUBK default model, see Fig. 2. These studies show that to maintain population BLLs below 5 μg/dL the exposure pathway for soil lead concentrations at the community scale would need to be less than 80 mg/kg per California guidelines (established prior to the 5 μg/dL reference value) or less than 40 mg/kg according to empirical data from New Orleans (California, 2009; Mielke et al., 2016). Validation (collecting confirmatory surface soil samples) would not be necessary if low Pb soil was emplaced across whole neighborhoods.

One of the challenges of a soil intervention program is the need to secure large deposits of soil to cover existing soils. In New Orleans Louisiana, Pb-safe soil is sourced from the Bonnet Carré Spillway, located up-river from New Orleans (U.S. Army Corps of Engineers). The alluvial soil, derived from the sediments of the Mississippi River at the Bonnet Carré Spillway, has a median Pb content of 5 mg/kg (Mielke et al., 2000). Other geographic areas will need to evaluate the availability of similar Pb-safe source materials. As described in sections 4.1 and 4.2 all cities have a reliable source of biosolids and every city also has sources of low Pb soil in the outskirts.

The behavior of residents can also impact exposures to soil Pb sources. The simple act of taking shoes off before entering homes can impact on the amount of Pb in soil dust that enters a home. If a soil Pb abatement program were introduced, it would also be necessary to establish a program of public education to inform residents of strategies that could help lower Pb exposures in homes.

A limiting factor of a soil Pb intervention program is cost. The cost of funding the holistic lead intervention schemes could be obtained from taxes on fuel and paints in each state (Farquhar, 1994). Given that the Pb in soils and home interiors originated from these sources this is appropriate. The taxes could be collected at the state level and the soil Pb remediation program could be run by individual states. The design of the holistic remediation programs could be developed by the federal government as a series of guidance documents which could be used by the states who are ultimately in charge of each program. In retrospect Pb exposure has undergone enormous reductions since the 1960s and 1970s. The policy developments focused on removing Pb from paint the vehicle fuels. Future progress in reducing exposure will need to be geared toward the difficult tasks of curtailing exposure to Pb residues that are the legacy of past uses of Pb.

5. Conclusions

Urban inner-city soils have been contaminated with lead primarily due to the past usage of lead additives in petrol and the deterioration of exterior lead based paints. The most vulnerable population to Pb residues is very young children and they are being exposed via ingestion and inhalation routes of exposure that is sourced from soil, water and air. A review of the literature indicates that soil Pb interventions consisting of various isolation techniques have effectively reduced BLLs in children at various scales such as the individual house, neighborhoods, villages, and cities. The literature also indicates that interventions that did not measure the effects of the intervention on BLLs reduced soil Pb to varying degrees. For projects that studied both soil Pb and BLL the most effective and lowest cost project consists of covering large areas of contaminated soil with a clean soil cap of about 15 cm thickness. Soils with low Pb levels should be available in the outskirts of every city. Engineering advances have been applied to developing low Pb biosolids that are abundant in the operation of every city and a cost effective source for soil Pb intervention and reducing BLLs. Other soil Pb remediation techniques such as adding phosphate, biochar or watering soils to prevent resuspension can reduce Pb in soil but they are limited and do not assist with providing the margin of safety necessary for primary prevention of Pb exposure. To eradicate the Pb poisoning problem requires a concerted effort to curtail Pb exposure from all major environmental components water, air and soil.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.apgeochem.2017.02.015.

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