frontiers | Frontiers in Ecology and Evolution

REVIEW
published: 03 June 2022
doi: 10.3389/fevo.2022.873542

Legacy Lead in Urban Garden Soils: Communicating Risk and Limiting Exposure

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Specialty section:
This article was submitted to
Urban Ecology,
a section of the journal
Frontiers in Ecology and Evolution

Received: 10 February 2022
Accepted: 10 May 2022
Published: 03 June 2022

Citation:
Paltseva AA, Cheng Z,
McBride M, Deeb M, Egendorf SP
and Groffman PM (2022) Legacy
Lead in Urban Garden Soils:
Communicating Risk and Limiting
Exposure.
Front. Ecol. Evol. 10:873542.
doi: 10.3389/fevo.2022.873542

Lead (Pb) exposure has long been recognized as a hazard to human health. Urban garden soils often contain elevated levels of Pb, mainly from legacy sources, which is a main barrier for urban gardening. The capacity of gardeners to access, understand, and act on scientific data related to soil contamination is also variable. This synthesis paper briefly summarizes the current scientific knowledge on soil Pb in urban gardens. Our objective is to produce clear recommendations about assessing actual risks and limiting exposure. First, we synthesize the nature and extent of soil contamination with Pb, and then describe how the bioavailability and risk of this contamination to humans is assessed. We then go on to potential exposure pathway through plants and remediation methods to improve soil health and reduce human exposure. We have developed best management practices for practitioners that include: (1) urban soil testing should be prioritized because of the high probability of Pb contamination, and urban gardening should not begin until thorough testing or remediation has been done; (2) documentation of land-use history should be required in all property transactions so that the potential for soil (and other) contamination can be clearly identified; (3) amendments cannot be relied upon as a treatment for contaminated soils to reduce risk to gardeners because they do not always make contaminants less harmful; (4) certain crops (such as fruiting vegetables) are much less susceptible to contamination than others and thus should be prioritized in urban gardens; (5) wherever feasible, raised beds filled with upcycled local mineral and organic materials are the preferred substrate for urban gardening. Further monitoring of potentially contaminated and remediated soils as well as effective communication with the public are necessary to ensure human safety.

Keywords: urban garden, risk, best management practices, lead, soil contamination

HIGHLIGHTS

- Urban soil testing should be prioritized.
- Documentation of land-use history should be required and made available.
- Amendments cannot be relied upon as treatment for contaminated soils.
- Certain crops are much less susceptible to contamination than others.
- Raised beds filled with upcycled local mineral and organics are preferred.

INTRODUCTION

One of the earliest metals discovered and smelted from ore by humans, lead (Pb) has been used for over 5,000 years (Lewis, 1985). Lead was used by the ancient Romans to make water pipes and drinking vessels and to line baths. During the Middle Ages, alchemists considered Pb as a key constituent used in techniques thought to generate gold from baser metals. Uses in paint and fuel increased the amount and the spread of Pb in the environment (Mielke, 2016). By the twentieth century, the U.S. had emerged as the world's leading producer and consumer of refined Pb. According to the National Academy of Science's report on Lead in the Human Environment, by 1980, the U.S. was consuming about 1.3 million tons of Pb per year (Lewis, 1985). Lead paint was banned in the US in 1978 and leaded gasoline was only banned worldwide in 2021 (Palmer, 2021).

An enduring concern about Pb is in urban soils, where it is often present at alarmingly high levels (Cheng et al., 2015; Gupta et al., 2019; Rai et al., 2019). According to the US EPA (USEPA, 1998), three primary sources contribute to elevated soil-lead levels: (1) lead-based paint, (2) point source emitters (e.g., smelters, refuse incinerators, dumpsites for lead-acid batteries), and (3) leaded gasoline emissions. It is often difficult to identify specific sources responsible for elevated soil-lead concentrations at any particular location (USEPA, 1998).

The presence of Pb in urban soils create risks for urban agriculture and gardening. Urban (and peri-urban) agriculture is estimated to produce 15–20% of the global food supply (Lal, 2020), with over 800 million people actively engaging in urban agriculture worldwide (Hamilton et al., 2014). Urban cropland covers over 67 million ha or more than 5% of the world's cropland area. About 266 million households may be engaged in urban crop production across developing countries (Hamilton et al., 2014). Fruit and vegetable cultivation can potentially yield up to 50 kg m^{-2} $year^{-1}$ of edible produce (Eigenbrod and Gruda, 2015).

From 2008 to 2013, the number of home gardens in the US increased by 4 –37 million households, while community gardens tripled from 1 to 3 million, a 200 percent increase (National Gardening Association, 2014). There are more than 18,000 community gardens throughout the United States and Canada (Lawson and Drake, 2013). In New York City, there are over 750 community-sponsored open space garden project sites (Lawson and Drake, 2013) covering more than 90 acres (Eizenberg, 2013). There are over 745 school gardens, over 100 gardens in land trusts, and over 700 gardens at public housing developments on city property (Eizenberg, 2013).

MISCOMMUNICATION AND CONFUSION ABOUT SOIL CONTAMINATION AND RISK

Unfortunately, trace metal contamination is common in urban soils and is a major constraint on urban agriculture (Gupta et al., 2019). In addition to Pb, contaminants found in urban soils include pesticides, petroleum products, asbestos, creosote, and radon (Alloway, 2004). Contaminated soils have the potential to produce contaminated plants, which can then be consumed by humans (Rai et al., 2019). There are many publications on the topic of trace element pollution (Hough et al., 2004; Pelfrêne et al., 2013, 2019; Izquierdo et al., 2015; Bidar et al., 2020) and on how to mitigate the health risk associated with gardening and consumption of garden produce. However, awareness and understanding by the general public of the actual risks related to urban gardening are often lacking (Burghardt et al., 2015; Hunter et al., 2019; Bidar et al., 2020).

Gardeners respond to and act on scientific data related to soil contamination in variable ways (Harms et al., 2013; Kim et al., 2014; Balotin et al., 2020). Communities are often unaware of soil contamination risks and remediation strategies (Whitzling et al., 2010). Around the world, many newspapers and websites encouraged urban gardening during the COVID-19 pandemic, but few highlighted potential contaminations in urban soils and the associated risk for human health. This is an unfortunate omission, and general interest publications such as newspapers are more accessible to community gardeners than scientific literature. In addition, property owners have great temptation to ignore contamination issues, as they may negatively impact property values (MacNair, 2004).

Even when gardeners are aware of contamination issues and mitigation strategies, they may not be able to apply these strategies properly or be aware of the extent of contamination and the routes of exposure (Harms et al., 2013; Kim et al., 2014; Balotin et al., 2020). There is also high variation among groups of gardeners, e.g., one study found that home gardeners were more aware of the potential health risks associated with soil trace elements than community gardeners (Balotin et al., 2020).

As is so frequently the case in complex societal issues, the flow of information from science to society regarding soil contamination is incomplete and inefficient, and scientific results are often misinterpreted. For example, in 2014, the New York Post published two articles on "toxic soils" and "toxic vegetables" grown in the Sterling Community Garden in Brooklyn (Buiso, 2014a,b). The stories were misinterpretations of the data published by McBride et al. (2014) and used sensational phrases such as "disturbing new state data," "the numbers are startling," and "this is insane." Further research showed that only a small area in the garden was contaminated and that risks could be readily mitigated (Paltseva et al., 2020). Despite decades of research on soil contamination, there is still a clear need to synthesize information and produce clear, generally accessible, and understandable recommendations for safe urban gardening practices. Thus, the objectives for this synthesis paper are to briefly summarize the current scientific knowledge on soil Pb in urban gardens and to produce clear recommendations about assessing actual risk and limiting exposures. To write this paper we used online sources Web of Science, Google Scholar, Research Gate and purposefully searched or browsed for the articles related to our topic that we have been working on for several years. We also used reference lists in published articles to help us narrow down searches to the most recent and relevant materials. The following road map lays out the outline of the paper (**Figure 1**). In the sections below, we first review the nature and extent of soil contamination with Pb, and then describe risks to humans. We then discuss potential exposure pathways through plants, how Pb bioavailability is assessed, and remediation methods to improve soil health and reduce human exposure. We conclude with the best management practices for practitioners and future perspectives.

URBAN AND SUBURBAN SOIL CONTAMINATION

Levels and Distribution of Trace Elements in Urban Garden Soils

The first step to promote the many benefits of urban gardening is to assess the risk from trace element exposure by characterizing the nature and extent of Pb levels in gardens. For example, in NYC, high median Pb concentrations (n = 2,322) were found in a wide range of neighborhoods cutting across density and socioeconomic gradients (Paltseva and Cheng, 2019a). Similar results have been found in other cities (**Table 1**) and have been compiled in a number of review papers (Ajmone-Marsan and Biasioli, 2010; Wei and Yang, 2010; Alekseenko and Alekseenko, 2014; Cheng et al., 2014; Antoniadis et al., 2019; Hanfi and Yarmoshenko, 2020). We suggest that a clear and useful conclusion and recommendation would be to assume that urban soils of any industrial or postindustrial country have a high probability for Pb contamination and that urban gardening should not begin until thorough testing or prophylactic remediation (e.g., construction of raised beds with clean soil) has been done.

Suburban Soil Contamination

A topic that has received less attention is that soil contamination by trace elements extends beyond city boundaries and is an issue for the much larger suburban areas around cities (**Table 2**). For example, trace metal concentrations in suburban Christchurch, New Zealand, garden soils were higher than background soil concentrations (Ashrafzadeh et al., 2018). Neighborhoods in Christchurch developed before the 1950s were affected by leaded paint and gasoline and had a mean Pb concentration of 282 mg kg^{-1} in their garden soils. Areas that experienced demolition and reconstruction work following earthquakes were vulnerable to redeposition of contaminants associated with this disturbance (Ashrafzadeh et al., 2018).

In suburbs of Tianjin City, China, Pb exceeded the "Soil Environmental Quality Risk Control Standard for Soil Contamination of Agricultural Land (GB 15618-2018)" with

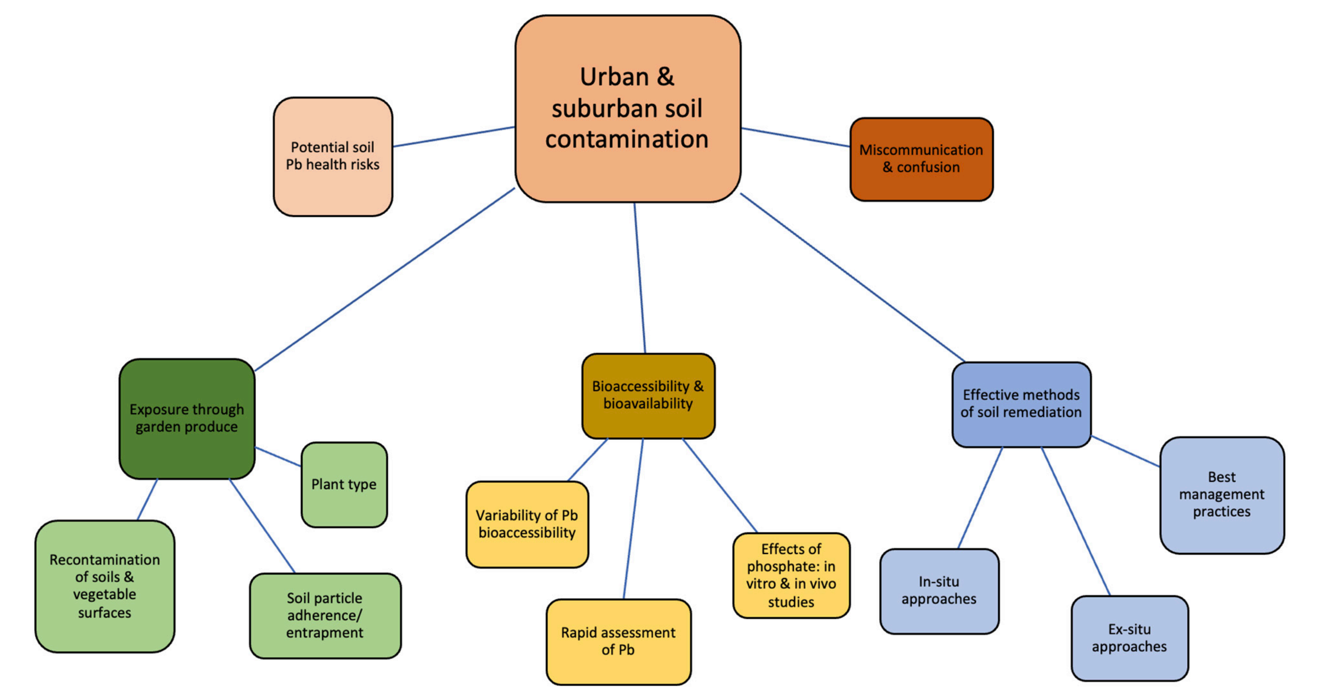

FIGURE 1 | "Road map" to the paper highlighting main sections.

TABLE 1 | Comparison of mean trace elements concentrations in urban soils in selected cities around the world (in mg kg^{-1}).

	# Samples	Cr	Ni	Cu	Zn	As	Cd	Pb	Type of samples	References
Cities with over 1 million people										
New York city	1,652	49	28	77	248	10	1.2	355	Garden soil	Cheng et al., 2015
Bangkok	30	25	23	27	38	–	0.15	29	Topsoil	Wilcke et al., 1998
Hong Kong	236	17	4	10	78	–	0.33	71	Urban	Lee et al., 2006
Beijing	~770	34	27	21	62	7.8	0.12	27	Topsoil, mean	Chen et al., 2005
Hangzhou	80	47.5	24.1	41	148	–	1.3	75.7	Urban topsoil	Zhang and Ke, 2004
London—Richmond	214	–	–	30	108	–	<0.2	158	Topsoil	Kelly et al., 1996
London—Wolverhampton	295	–	–	62	231	–	0.80	106	Topsoil	Kelly et al., 1996
Vienna	96	80	–	18	97	8	0.2	65	Urban, 0–20 cm	Simon et al., 2013
Warsaw	nd	32	12	31	166	–	0.73	57	nd	Czarnowska, 1980
Madrid	55	75	14	72	210	–	–	161	0–20 cm	De Miguel et al., 1998
Moscow	224	–	17	30	105	4	0.5	30	Public areas, 0–20 cm	Romzaykina et al., 2021
Berlin	2,182	25	8	31	129	3.9	0.35	77	0–20 cm	Birke and Rauch, 2000
Damascus	51	51	35	30	84	–	–	10	Topsoil, agriculture	Möller et al., 2005
Manila	286	114	21	99	440	–	0.57	214	Metropolitan area, 0–5 cm	Pfeiffer et al., 1988
Dublin	1,058	44	41	51	248	15.5	1.8	123	0–10 cm	Glennon et al., 2014
Melbourne	39	17	15	40	218	8	–	102	0–5 cm, community gardens	Laidlaw et al., 2018
Melbourne	395	50	16	49	334	9	–	204	0–2 cm, Residential gardens	Laidlaw et al., 2018
Torino	123	129	153	71	147	–	–	94	1–10 cm	Biasioli et al., 2007
Coruna	15	39	28	60	206	–	0.3	309	Garden soil, 0–5 cm	Cal-Prieto et al., 2001
Quezon city	64	370	150	445	1,540	–	–	594	Metropolitan area, 0–15 cm	Navarrete et al., 2017
Mumbai (Bombay)	30	79	144.5	147	–	–	1.3	42.5	Urban soil	Ratha and Sahu, 1993
Cities with less than 1 million people										
Zagreb	331	–	49	18	70	–	0.5	23	Agricultural soil	Romic and Romic, 2003
Baltimore	422	–	2.8	17	92	–	0.56	100	Vegetable garden soil	Mielke et al., 1983
New Orleans*	4,388	2	7	16	146	–	2	100	Topsoil 0–2.5 cm	Mielke et al., 2005
Oslo	~300	29	24	24	130	4.5	0.34	34	Topsoil	Tijhuis et al., 2002
Detroit	30	212	39.6	–	–	26	3.4	256	Residential, 0–15 cm	Howard et al., 2019
Galway	166	–	22	27	85	8	–	58	Topsoil	Zhang, 2006
Lisbon	18	63.6	93.7	31.5	86.6	–	<LOD	66.1	Topsoil 0–20 cm	Bechet et al., 2018
Nantes	29	33.3	30.1	42.3	86.4	–	2.1	60.6	Topsoil 0–20 cm	Bechet et al., 2018

**Median values.*

TABLE 2 | Comparison of mean trace elements concentrations in suburban soils in selected cities around the world (in mg kg^{-1}).

	# Samples	Hg	Cr	Ni	Cu	Zn	As	Cd	Pb	Type of samples	References
Christchurch, New Zealand	50	0.29	22.7	11.4	34.6	189	12.1	1.2	137	Residential gardens	Ashrafzadeh et al., 2018
Tianjin city, China	86	0.97	101	–	67	100.6	9.5	0.49	52.5	Suburban	Shi et al., 2010
Wien, Austria	96	–	86	–	16	73	6.9	0.2	46	Flood-plain forest, 0–20 cm	Simon et al., 2013
Wien, Austria	96	–	52	–	21	105	5.2	0.3	70	Oak–hornbeam forest, 0–20 cm	Simon et al., 2013
Hanoi, Vietnam		–	175	60	196	204	–	4	131	Irrigated vegetable farms	Huong et al., 2010
Hong Kong	31	–	20.8	3.54	9.72	67.9	–	0.37	57.8	Suburban, 0–15 cm	Lee et al., 2006
New Orleans, United States*	19	–	1	5.5	5	33	–	1.6	12	Suburban (top 2.5 cm)	Mielke et al., 2004

**Median values.*

exceedance rates of 3.06% (Zhang et al., 2019). Wastewater and sludge irrigation and air deposition were the primary contaminant sources (Shi et al., 2010). In Vienna, Austria, Pb concentrations were significantly higher in urban than suburban and rural areas (Simon et al., 2013), but the differences were not statistically significant between suburban and rural areas. Vienna's most important pollution sources are numerous roads and motorways with heavy traffic (Simon et al., 2013). Agricultural surface soils (0–20 cm) in Thanh Tri District of Tam Hiep Commune in Hanoi City, Vietnam, had concentrations of Pb (145 mg kg^{-1}) exceeding the Vietnamese standard for agricultural soil (Huong et al., 2010). The primary source of contamination is the application of polluted irrigation water to the soil. Vegetable heavy metal concentrations also exceeded the Vietnamese standards for Pb (Huong et al., 2010).

At a location in the suburbs of New York City (Duke Farm, New Jersey) where lead-arsenate pesticides were used to control gypsy moths in the 1920s, vegetables had Pb levels above health and safety standards, especially root and leafy green vegetables (Paltseva et al., 2018). This contamination appears to have come from soil particles adhering to the vegetables; i.e., these crops did not appear to be taking up significant amounts of metals into their tissues *via* roots. Nevertheless, tomatoes were the only vegetables entirely safe for consumption. Phosphate-bearing amendments added to the soil reduced extractable Pb but increased extractable As in soils. Compost additions significantly improved soil quality, plant health, and yield and decreased soil and vegetable contamination levels. Lead-arsenate pesticides were widely used in orchards (McBride, 2013; Gamble et al., 2018) and have left a widespread legacy of contamination similar to that observed at Duke Farm.

There is a strong need to expand awareness of trace element problems beyond dense urban areas. While the contaminant levels found in suburban soils are commonly lower than densely populated urban areas, they are often elevated compared to background soils and exceed thresholds set forth for human health. We suggest that a clear and useful recommendation is to include land-use history in all property transactions so that the potential for soil (and other) contamination can be clearly identified.

POTENTIAL HEALTH RISKS OF SOIL LEAD: THE EFFECTS OF ENVIRONMENTAL VARIABLES AND EXPOSURE PATHWAYS ON HEALTH OUTCOMES

While Pb is estimated to cause 412,000 premature deaths per year and has been deemed a "leading risk factor for premature death in the United States" (Lanphear et al., 2018), the relative importance of exposure to Pb in soil, a key concern in urban gardening, is not apparent (Gailey et al., 2020). Zahran et al. (2013b) reported that children's blood lead levels (BLLs) were spatially correlated with neighborhood soil Pb levels in New Orleans. Research in Syracuse, NY, Minneapolis, and St. Paul, MN, Detroit, MI arrived at similar conclusions (Aschengrau et al., 1994; Johnson and Bretsch, 2002; Zahran et al., 2011; Mielke et al., 2016, 2019; Laidlaw et al., 2017). Despite these observations in numerous cities, along with data on soil Pb in New York City (NYC) (Nussbaumer-Streit et al., 2016), the NYC Health Department (DOHMH) does not officially recognize the impact of soil contamination on children's Pb exposure and BLLs (**Figure 2**). Former NYC Health commissioner stated at a New York City Council Oversight Hearing, "Soil is not, I repeat not, a significant source of lead exposure for children in New York City" (Werth, 2019). Soil Pb concentrations in NYC gardens (mostly private home gardens) range up to 9,000 mg kg^{-1} with a median of 355 mg kg^{-1} (Cheng et al., 2015). Among all the garden samples tested (n = 1,646), 48% exceeded the US EPA standard of 400 mg kg^{-1} for children's play areas (USEPA, 2014).

To achieve NYC's goal to eliminate childhood Pb exposure, more research is needed on the degree to which soil Pb exposure affects children's blood Pb levels. Exterior environmental sources of Pb dust can relocate more Pb to children's hands (Viverette et al., 1996; Hunt et al., 2006; Laidlaw and Filippelli, 2008) than interior sources. Soil directly contributes to Pb found on children's hands after outdoor play (Viverette et al., 1996), which occurs in private backyards and playgrounds that have been sinks for atmospheric Pb and tend to have higher Pb concentrations than public spaces (Cheng et al., 2015; Landes et al., 2019). Thus, it is crucial to investigate various sources of Pb exposure to assess their relative contribution as exposure pathways. Primary factors affecting the magnitude of sources include type and age of housing, home maintenance methods, proximity to major roads and industrial zones, socioeconomic factors, soil Pb, and water Pb. Studies relating to soil cover condition and garden Pb content in public areas, behavioral factors relating to children (children's age, mouthing habits, time spent on playgrounds), and socioeconomic factors relating to the community (income, ethnicity, education, occupation) are also very important for the assessment of exposure pathways. Primary prevention is crucial for managing the associated risks of soil Pb in an urban environment; thus, public participation is necessary to lower childhood exposure (Mielke, 2015; LeadFreeNYC, 2019). Informed residents with basic remediation skills can directly reduce their children's risk.

Identification of Pb exposure pathways affecting BLLs has to be done in collaboration with medical researchers and public health agencies to assess the relationship of environmental variables to actual health outcomes (i.e., lowered BLLs in children). Topics that need to be addressed include: the effectiveness of soil remediation in a neighborhood (e.g., importing clean topsoil) to lower Pb exposure in children, the impact of gardening activities on blood Pb of gardeners and their families, the impact of overall diet (e.g., calcium intake) of individuals on blood Pb, and the correlation between Pb concentrations in house dust and soil outside the house. Addressing these topics will be critical in reducing confusion around soil Pb contamination in urban gardening communities.

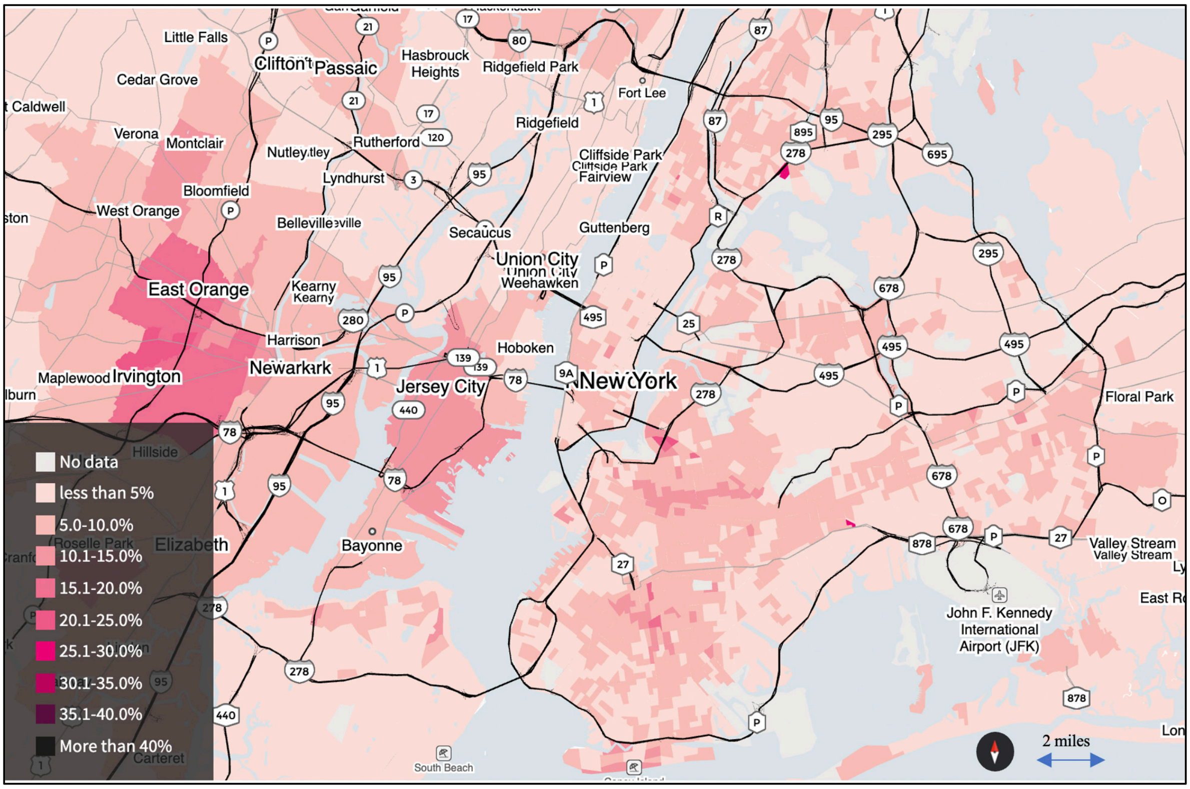

FIGURE 2 | Map of children's blood lead (BLL) data per census tract (2005–2014) in NYC created by Reuters. The CDC estimates that nationwide, around 2.5% of children ages 0–6 have an elevated lead level, defined as 5 mg/dL or higher (Scheneryer, 2017).

EXPOSURE THROUGH GARDEN PRODUCE

There is great concern about the potential transfer of soil contaminants into vegetable crops when Pb and other toxicants contaminate garden soils. Contamination of vegetables grown in contaminated soil in a greenhouse has been shown to be dependent on soil total Pb concentrations, soil pH, and plant species (McBride, 2013). Exposure to contaminated produce can come *via* ingestion of shoots, roots, fruits, or adhered soil particles to any of these plant parts. This complexity and extent of produce contamination have been a major source of confusion for urban gardening stakeholder groups.

Plant Type

Crop contamination varies markedly with plant species and type. Zhou et al. (2016) found that crop contamination decreased as follows: leafy vegetables > stalk vegetables/root vegetables/solanaceous vegetables > legume vegetables/melon vegetables. Paltseva et al. (2020) found that onion > kale > eggplant > cabbage > tomato. Finster et al. (2004) found that most Pb was concentrated in the roots with some translocation into the shoots and that Pb was not concentrated in the edible parts of fruiting vegetable plants (beans, tomatoes, zucchini, peppers). Rather, edible leaf crops (Swiss chard, collard greens), herbs (mint, cilantro), and edible roots (carrot, onion, radish) had the highest levels of Pb (Finster et al., 2004). Studies by Paltseva et al. (2018, 2020) were consistent with previous work showing that crop type is more important than soil amendment type.

Contribution of Soil Particle Adherence/Entrapment to Vegetable Contamination

Plant tissues strongly retain soil solids, even after rigorous washing with soil-dispersing agents. This retention can significantly increase the Pb content of produce because soil usually has orders of magnitude higher Pb concentrations than plant tissues. McBride et al. (2014) found that vegetable Pb concentrations were correlated with Al concentrations and interpreted this observation as evidence that soil particle adherence was more important in determining vegetable contamination than root Pb uptake. This interpretation is based on the idea that because there is no biological need for Al, any Al in produce must be an indicator of soil particle adherence when soil is pH neutral or near-neutral. Sheppard and Evenden (1992) also determined that soil adhesion was a dominant pathway for plant accumulation of uranium, thorium, and Pb using plant/soil concentration ratios. Similar results were observed in case studies in New York and New Jersey (United States), where plant tissue Pb was correlated with plant tissue Al (Paltseva et al., 2018; Egendorf et al., 2021).

Certain crops, in particular, fruiting vegetables (e.g., tomato, eggplants, cucumbers), are much less susceptible to contamination than others. These crops should be prioritized in urban gardens. There should be a clear recommendation that crops more susceptible to contamination should be avoided unless thorough testing or prophylactic measures (e.g., construction of raised beds with clean soil) have been done.

Recontamination of Remediated Soils and Vegetable Surfaces

Although highly Pb-contaminated sites can be remediated by importing clean soil to replace or cover the existing surface soil, the environment is very dynamic, and recontamination can occur, particularly from aerial deposition (Engel-Di Mauro, 2021). Given that plant uptake of Pb from the soil is generally low, contamination by atmospheric sources has been detectable in various crops (Uzu et al., 2014; Amato-Lourenco et al., 2016; Shahid et al., 2017; Ma et al., 2019). Research has documented how contaminants can be lodged in tissues (Schreck et al., 2012) and absorbed by leaves (Uzu et al., 2010). However, more research is needed to effectively distinguish the role of metal speciation in foliar uptake, toxicity, compartmentation, and detoxification inside plants, especially in urban agriculture (Shahid et al., 2017).

Resuspension of contaminated soils have been shown to contribute to atmospheric Pb in Birmingham, Alabama, Chicago, Illinois, Detroit, Michigan, and Pittsburgh, Pennsylvania (Laidlaw et al., 2012), which has been linked to seasonal fluxes in elevated Blood Lead Levels (BLLs) of children (Havlena et al., 2009; Zahran et al., 2013a). Even though recontamination of remediated soils is expected to be a prolonged process, the possibility that clean soils imported into gardens could become Pb-contaminated again over decades warrants more study. As stated by Binns et al. (2004), "the long-term sustainability of intervention applications is unclear." In addition, the effects of bioturbation and evapotranspiration of soil water on soil Pb translocation in capped and covered soils after remediation are still unknown. Thus, mitigating current contamination and preventing potential recontamination can be achieved by building raised beds, reducing dust, keeping soil moist, for example, with mulch. Adding compost also improves moisture retention, dilutes the overall contamination and can bind metals.

BIOACCESSIBILITY AND BIOAVAILABILITY

Variability of Lead Bioaccessibility in Urban Soils

A great source of complication in trace element contamination in soils is multiple forms of metals in soil. These forms vary significantly in their potential to cause harm in animals, including humans, as they are ingested or inhaled. "Knowledge of lead bioavailability is important because the amount of lead that actually enters the blood and body tissues from an ingested medium depends on the physical chemical properties of the lead and of the medium. For example, lead in soil may exist, at least in part, as poorly water-soluble minerals, and may also exist inside particles of inert matrices such as rock or slag of variable size, shape, and association. These chemical and physical properties may tend to influence (usually decrease) the absorption (bioavailability) of lead when ingested. Thus, equal ingested doses of different forms of lead in different media may not be of equal health concern" (USEPA, 2013).

Exposure may be assessed *in vivo* using Pb relative bioavailability (RBA; the fraction of soil-borne Pb absorbed into the systemic circulation following comparison to Pb absorbed from a Pb acetate reference dose) or by *in vitro* Pb bioaccessibility (IVBA) (the fraction of soil-borne Pb that is dissolved in simulated gastrointestinal [GI] solutions) (USEPA Oswer, 2007; USEPA, 2012; Kastury et al., 2019b). *In vitro* GI extraction tests have been developed to measure bioaccessible Pb concentrations as a predictor of bioavailable Pb concentrations. The Relative Bioaccessibility Leaching Procedure (RBALP) (Kelly et al., 2002; Drexler and Brattin, 2007), the Solubility/Bioavailability Research Consortium (SBRC) *in vitro*-gastric and *in vitro*-intestinal assays (Juhasz et al., 2009), EPA Method 1,340 are all based on a glycine solution extract using pH 1.5, or 2.5 in some cases. Other *in vitro* methods for measuring bioaccessibility have also been correlated with *in vivo* measurements, such as the PBET (Ruby et al., 1996), OSU IVG (Schroder et al., 2004), UBM, and BARGE (BARGE – INERIS, 2010; Wragg et al., 2011), RIVM (Oomen et al., 2003), USBLT (Chaney et al., 2011), and W-PBET (Furman et al., 2006) methods.

The mean values of Pb bioaccessibility in contaminated soils range widely from 3 to 59% (n = 140), according to the review conducted by Li et al. (2020). The authors found that SBRC produces the highest, followed by UBM, IVG, and DIN [German standard bioaccessibility methodology (DIN, 2000)] assays, with the PBET assay producing the lowest bioaccessibility values. The variability of results obtained by different methods leads to confusion when attempting to assess potential human exposure (especially children) by comparing results among tests.

It is important to note that due to the variation in bioaccessibility, bioaccessible Pb may not correlate with total Pb concentrations (i.e., some samples with low concentrations of total Pb may have a high fraction of bioaccessible Pb). Although Roussel et al. (2010) found significant positive correlations between Pb bioaccessibility (UBM model) and total Pb concentration in 27 urban contaminated soils, no correlations were found between total Pb concentration and its bioaccessibility in 20 soils from various sources based on the RBALP model (Morman et al., 2009), in 90 Dutch soils based on RIVM model (Hagens et al., 2009), in 28 soils polluted with various Pb sources (Walraven et al., 2015), or in 49 urban soils from NYC based on EPA Method 1340 at pH 2.5 (Paltseva et al., 2018). Regarding the effects of particle size of soil fractions on bioaccessibility, Li et al. (2021)

established a trend of higher Pb bioaccessibility in the finer fractions based on the review of multiple studies. This trend is related to total metal concentration, its chemical form, and anthropogenic influence.

An obvious conclusion that emerges from the literature is that although many methods have been developed and used to determine the bioaccessibility of soil Pb, the results have been widely inconsistent. There is no consensus as to which method is more accurate than others. The US EPA Standard Method 1340 is also recognized as inappropriate for urban soils, which has highly variable Pb bioaccessibility (USEPA, 2017). This is a barrier to the clear transmission of information about risk to non-scientific audiences. To protect public health, a more conservative approach should be adopted, i.e., to assume a high bioaccessibility value before a reliable method can be found. Indeed, the failure of the ability of science to produce a well-defined and transparent assessment methodology underlies our recommendation to assume that any urban soil has a high potential for Pb contamination and associated risks.

Effects of Phosphate: *In vitro* and *in vivo* Studies on Urban Soils Amended With Phosphates

A complication in developing methods for assessing soil Pb bioavailability is complex interactions between phosphate and Pb (Miretzky and Fernandez-Cirelli, 2008). It has been generally understood that phosphate interacts with Pb to form insoluble pyromorphite minerals (Scheckel et al., 2013). These minerals stay largely insoluble after ingestion, and therefore, there is great interest in the ability of phosphate additions to decrease Pb bioavailability and bioaccessibility. However, the formation of Pb phosphates in typical urban soils with near-neutral pH and high organic matter content is likely very limited or slow even when soluble phosphate is added to the soil (Chrysochoou et al., 2007). In addition, lead is very water-insoluble in most urban soils and reacts extremely slowly with the added phosphate (Cai et al., 2017).

Phosphate treatment efficacy to reduce Pb bioavailability has been shown to depend on phosphate source, soil pH, Pb concentration, and competing ions based on the wide range of treatment effect ratio (TER) values (0.03–1.16) found in the literature (Scheckel et al., 2013; Juhasz et al., 2014). The application of liquified bone meal soil amendment to residential soils across Detroit, Michigan resulted in a 9.8% decrease in IVBA (Good, 2020). Greater immobilization efficacy (lower human bioaccessibility) of Pb (up to 19%) was observed for granular (2–4 mm diameter) compared to ground (< 0.5 mm) struvite (phosphate mineral) applied to urban agriculture sites in Chicago, Illinois (Gu et al., 2020).

Juhasz et al. (2014) demonstrated the possibility of pyromorphite formation *in vivo* (i.e., in the small intestines of mice) following solubilization of Pb and P in the stomach when phosphate amendments and Pb-contaminated soil were gavaged sequentially. They concluded that an initially soluble source of phosphate is more effective in immobilizing Pb than the much less soluble rock phosphate, which is not reflected by the *in vitro* assay. Kastury et al. (2019a) called for more extensive studies with different animal models and endpoints (e.g., Pb concentration in liver, kidney, and bones) to better predict Pb RBA in phosphate amended soils.

While phosphate has been shown to immobilize Pb, it can also mobilize As by increasing the solubility of soil As through competitive anion exchange (Peryea, 1991). Such complex interactions between Pb, phosphates, and As can bring unintended adverse consequences of phosphate-bearing amendments. Overall, from existing research, it can be concluded that it is often difficult to predict the effectiveness of many amendments in reducing Pb bioaccessibility, especially when information about other soil properties are not available. Urban soils are very heterogeneous, which makes the prediction of Pb bioaccessibility especially challenging. This is one of the major impediments to transmitting clear and useful information about urban gardening risks to stakeholders. A clear and useful recommendation is that in general, phosphate amendments cannot be relied upon to reduce risk to gardeners, and they should not be recommended blindly as a treatment for contaminated urban soils.

Rapid Assessment of Lead Bioaccessibility and Leachability With Field Portable X-Ray Fluorescence and Soil Lead Kits

One solution to the challenge of complex and confusing bioaccessibility testing protocols is the development of simple tools and methods to perform diagnostic screening to detect hotspots of contamination and associated risks. As discussed above, current methods for analyzing soil bioaccessibility in the lab are time-consuming and costly (Spliethoff et al., 2016). Urban soils, which are notable for their spatial heterogeneity, differ greatly in total and bioaccessible Pb concentrations over even small lateral or vertical distances, creating a need for large numbers of analyses.

One approach for low cost and rapid assessment of soil Pb bioaccessibility is based on portable X-Ray Fluorescence (pXRF) analyzers (Landes et al., 2019; Paltseva and Cheng, 2019b). These devices are relatively expensive (> $20,000) but are easy to use and can be made available to community groups with brief training. The procedure includes the following steps: (1) air-drying of soil samples, (2) extraction (0.4 M glycine) with a soil:extract ratio of 1:10 for 1 h at room temperature, (3) filtration of the extract, and (4) direct measurement of Pb in the liquid extract in cups by pXRF. The pXRF and ICP-MS measurements of the same extractants were highly correlated ($y = 1.16x-2$, $R^2 = 0.994$). This procedure can also be used to estimate IVBA as described in Drexler and Brattin (2007) and EPA Method 1340 (USEPA, 2013). This estimation would involve measuring Pb in the field-procedure extract solution by XRF or ICP-MS and then estimating the Pb extracted by the Drexler and Brattin (DB) method as $Pb_{DB} = (Pb_{Field\_procedure} - 89)/0.34$ (Landes et al., 2019).

An additional approach to rapid assessments is the use of inexpensive test kits. Landes et al. (2019) developed a soil bioaccessibility kit (∼$5/test) to measure hazardous levels of Pb in soil. The procedure for this kit was derived from the IVBA method of Drexler and Brattin (2007) and EPA Method 1340 but uses a higher soil-to-solution ratio. Currently, there are two commercially available tests for Pb that detect Pb in different media, including soil. For example, Lead Soil Check test strips (∼$4.6–$6.3/test) estimate 100, 200, 300, and 400 ppm (mg/L) of bioaccessible Pb in less than 10 min. However, these tests may not be reliable due to interferences with zinc and other trace elements in urban soils and should be used with caution.

MANAGEMENT OF SOIL CONTAMINATION

Given that contaminants can pose health risks through direct ingestion of soil or dust or the consumption of contaminant-bearing produce, it is prudent to reduce children's exposure to urban soils and gardens. However, playing outside, touching soil, observing soil and plant biodiversity, and growing food are practices that have been shown to have a significant positive effect on the intellectual and psychological development of a child (Koller et al., 2004; Moser and Martinsen, 2010; Chawla, 2015), especially for those in urban areas. Thus, there is great interest in the management and remediation of soil contamination.

Effective Methods of Soil Remediation

Ex situ Approaches

Remediation of metal-contaminated soils has been achieved by *ex situ* (i.e., removal of contaminated soil) or *in situ* (i.e., reduction of contaminant mobility by altering physical or chemical characteristics of the contaminated soil) processes. *Ex situ* treatments are often based on excavation and disposal in a landfill (**Table 3**) and replacing clean soil from another site (Deeb et al., 2020). An alternative approach is to "cap" contaminated soil to provide a physical barrier restricting access to contaminated soil and/or inhibit surface water infiltration from preventing release of contaminants to the surrounding areas or groundwater (Evanko and Dzombak, 1997). Removing contaminated soils is expensive and importing established topsoil reduces soil resources in one location to improve resources at another, which is ultimately unsustainable (Deeb et al., 2020).

Recent research has focused on mitigating exposure by building raised beds filled with constructed soils, or Technosols, which are defined as mixtures of organic and inorganic wastes and byproducts constructed to meet specific requirements (Egendorf et al., 2018; Deeb et al., 2020). In New York City, the Clean Soil Bank (CSB) program uses thoroughly tested and excavated sediments (deep subsoils) from construction projects to provide clean soils for municipal and community uses throughout the city. The program also produces constructed "topsoil" by amending sediments with compost particularly for urban gardens and farms (Walsh et al., 2018). Ensuring compost is tested for contaminants prior to use is an important consideration, as is assessing compost viability for supplying plant nutrients and monitoring for potential deposition or recontamination over time (Egendorf et al., 2018).

In New Orleans, Louisiana, fresh low-Pb alluvial deposits from the Mississippi River provide a great source of native soil materials. Projects using a geotextile soil cover (capping) showed the efficiency of remediation efforts at childcare centers in areas of the city where Pb concentrations exceed 400 mg kg^{-1} (Mielke et al., 2011). These projects demonstrated a relatively low cost and rapid method (only a few hours to complete) for reducing soil Pb on children's play areas (Mielke, 2016).

While the use of Technosols in raised beds is quite promising, large-scale implementation and long-term studies of this method have not been done (Erdem and Nassauer, 2006), and some key issues need to be resolved. Raised beds do not remove existing underlying contamination, so barriers separating contaminated materials below must be maintained over time. Continued monitoring for a breakdown of barriers between the raised bed and contaminated subsoil is recommended. Soil caps may require periodic maintenance when breaches appear, which means continued costs over time. Planting crops with deep roots can also lead to root penetration into contaminated soils, which may present limited risks of uptake but may further degrade physical barriers. Warning signs indicating subsoil contamination should be posted on sites utilizing this method to inform future gardeners, urban planners, and landscape designers (Cooper et al., 2020).

An additional potential challenge with constructed soils is that the organic amendments such as composts used in their construction may have elevated concentrations of metals. Metals can accumulate and reach a higher level over time in soil after repeated use of these amendments. Hornick et al. (1980) noted

TABLE 3 | Remedial options for metal-contaminated soil (adapted from Karna et al., 2017).

	Immobilization		Extraction			Isolation	
	Ex situ	***In situ***	***Ex situ***	***In situ***		***Ex situ***	***In situ***
Treatment technologies	• Solidification/ • Stabilization • Vitrification • Chemical red/Ox	• Solidification/ • Stabilization • Vitrification • Chemical red/Ox • Phytostabilization • Biological stabilization	• Soil washing • Physical separation • Chemical extraction • Thermal treatments • Electrokinetics	• Electrokinetics • Phytoextraction • Soil flushing	**Contaminant disposal technologies**	• Treatment followed by disposal (landfilling) or beneficial use of the material	• Capping

high Pb levels (300 mg kg^{-1}) in leaf compost, while Egendorf et al. (2018) found up to 226 mg Pb kg^{-1} in food waste and wood chip compost. Manure is usually a low-Pb organic amendment. However, near cities with smelters, manures may be heavily contaminated and should not be used (Chaney et al., 1984). Some modern swine and poultry feeding programs incorporate high Cu and/or Zn levels to increase feed use efficiency, often generating manure with as much Cu and Zn as sewage sludge (Mullins et al., 1982).

In situ Approaches

Remediation of metal-contaminated soil through *in situ* amendment applications can be uncertain because of various factors that influence geochemical properties and the specific metal immobilization mechanisms. For example, pH, conductivity, oxidation-reduction potential, and mineralogy can all influence the effectiveness of the amendments. Immobilization of metals mainly occurs through the addition of inorganic (e.g., fly ash, slag, zeolites), organic (e.g., biosolids, manures, paper pulp), or a combination of both inorganic and organic byproduct amendment types, often *via* the modification of soil pH and the precipitation of metal hydroxides (USEPA Office of Superfund Remediation and Technology Information, 2013).

Currently, organic byproducts (e.g., biosolids or food waste) are available in large volumes that can be used for remediation with such benefits as low cost, the proximity of the source to the remediation site, and/or a reduction in the use of virgin materials. The major environmental advantages of using "waste" byproducts as amendments are that wastes are diverted from landfills or surface impoundments, and the need to mine or synthetically produce a similar material is reduced.

Using biosolids (i.e., nutrient-rich organic materials derived from sewage sludge in the wastewater treatment process) (Palansooriya et al., 2020) and biosolids-based products, especially those high in Fe, have been recommended to reduce the bioavailability of Pb and other trace elements, thus, lessening human and ecological exposure (McIvor et al., 2012). However, because biosolids are a municipal byproduct, they contain many different synthetic organic and metal contaminants. Only a few of the contaminants are regulated, specifically nine metals and no synthetic organic pollutants, according to USEPA Office of Water (1995). A recent discovery of the fact that a substantial proportion of biosolids in the United States are contaminated by the non-degradable perfluoroalkyl substances (PFAS) (Stoiber et al., 2020) raises further concern about the use of these materials. However, the relative benefits and risks of biosolids use in urban agriculture should be assessed comprehensively and objectively in the context of climate change, waste management, food security, and public health.

Biochar produced at high temperatures has been shown to reduce metal mobility in soil (O'Connor et al., 2018). The effectiveness of biochar immobilization of Pb varies with feedstock type and application rate, biochar mixing depth, crop type, soil properties, and meteorological factors. However, the aging effect of biochar may reduce the efficiency of metal immobilization over time (O'Connor et al., 2018). The conversion of biosolids to biochar would be a promising way to increase biosolids' use efficiency and safety. During the pyrolysis process, microbial and organic contaminants in biosolids are subjected to thermal breakdown, and metals such as and Hg tend to volatilize during the thermal conversion process, but other metals such as Pb may become more concentrated (Palansooriya et al., 2020).

Best Management Practices

To achieve the best results, it may be necessary to combine several soil remediation strategies and sustainable conservation practices to decrease contamination risk and prevent recontamination (**Figure 3**):

- Dust control measures, e.g., mulch on top of contaminated soil and drip irrigation (moist soil), result in minimum resuspension of contaminated soil particles into the air as well as reduced splash or flooding (compared to other watering techniques), decreasing potential recontamination of the soil surface and above-ground plant tissues.

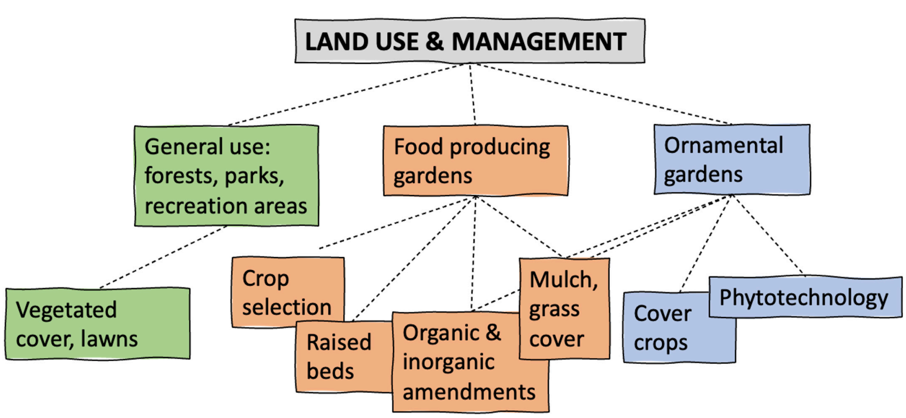

FIGURE 3 | Managing strategies required for different land uses (modified from Vasenev et al., 2017).

- The addition of organic matter (e.g., compost, biosolids, biochar) improves soil structure with more stable aggregates that adsorb contaminants and reduce erosion (Deeb et al., 2017), reducing the possibility of fine particle resuspension into the air.
- Conservation tillage practices that leave residue on the soil surface increase soil organic matter content that immobilizes metals by forming stable organo-metal complexes in the soil. The vegetation protects soil from raindrop impact and impedes overland flow, while root systems help bind the soil together to reduce runoff (Weil, 2015). These practices reduce topsoil erosion and decomposition rate of organic matter (Weil, 2015), enhance aggregate formation, diminish the possibility of particle resuspension into air or water, and prolong the effect of soil remediation by organic matter amendments.
- Liming increases soil pH, which reduces trace metal bioavailability and improves soil aggregation and structure (Lwin et al., 2018).
- Installing a garden at least 50 m away from heavy traffic avoids the accumulation of contaminants in the soil over time (Werkenthin et al., 2014).
- Climate change effects, especially temperature and precipitation extremes on soil contamination and metal mobility in soils, requires special attention. In the case of dry seasons, the addition of phosphate fertilizers, composts, and soil amendments may not form stable (and thus less bioavailable) forms of metals due to the lack of water content required for chemical reactions to occur (Paltseva and Neaman, 2020).

CONCLUSION

Gardening brings many benefits to urban residents such as nutritious food, grocery bill reduction, physical (diet, exercise) and mental (time spent in nature, pleasure) health benefits, a deeper connection to agriculture and urban nature, aesthetics, and promotion of biodiversity (Draper and Freedman, 2010; Clavin, 2011; Cameron et al., 2012; Chalmin-Pui et al., 2021). The importance of gardening to urban residents was made very apparent during the COVID-19 pandemic. It significantly contributed to human wellbeing by reducing anxiety, maintaining physical activities, and other effects (Sofo and Sofo, 2020; Chalmin-Pui et al., 2021). Raymond et al. (2019) identified emerging co-benefits of home gardening such as "the desire to leave a legacy for future generations, multiple sense of place benefits related to identity, family and friend bonding in place and the sheer enjoyment associated with incidental nature experiences like seeing a rare bird feed in their yard." In the future, gardening may decrease the need to import food, encourage the use of indigenous and traditional seeds, and motivate local exchange of fruits, vegetables, flowers, and seeds (Židak and Osmanagić Bedenik, 2019).

Our review suggests that while encouraging individual or community urban gardening is important, information about this activity must be communicated carefully with comprehensive safety guidelines to raise awareness about soil contamination. In-person classes, hands-on workshops, and physical and digital references for gardeners would be beneficial and an option for gardeners to ask experts questions on soil heavy metals contamination (Balotin et al., 2020).

Lead is often found at high levels in soils near current and old industrial sites, old orchards, high-traffic streets and highways, and older houses. Rapid screening methods estimating total soil Pb can locate the hotspots for target investigation. However, better, more reliable testing kits that can be accessible to a layperson are still needed along with biodegradable sensors for monitoring Pb over time. Despite the abundance of scientific literature discussing soil contamination, cases of elevated children BLLs are still widely found worldwide.

Despite decades of research on soil Pb contamination in urban area clear, consistent, generally accessible, and understandable recommendations for safe urban gardening practices need to be effectively shared with the public. The failure of science to produce a well-defined and transparent assessment methodology underlies our recommendation to assume that soil of any urban area has a high potential for Pb contamination. This paper suggests that urban gardening should not begin until thorough testing, mitigation, or remediation measures have been taken. Another recommendation would be to include land-use history in all property transactions so that the potential for soil contamination can be clearly identified in urban and suburban areas.

Certain crops such as fruiting vegetables, in particular, are much less susceptible to contamination than others. These crops should be prioritized in urban gardens. There should be a clear recommendation that crops that are more susceptible to contamination should be avoided unless thorough testing or *ex situ* remediation (e.g., construction of Technosols) has been done. There is significant uncertainty when soil amendments are used to reduce risk to gardeners. Compost has many beneficial effects on soil structure, metal solubility, and plant growth, reducing toxic metal concentrations in crops. However, ongoing monitoring of composts and soils is necessary to prevent accumulation of undesired trace elements in remediated soils over time.

AUTHOR CONTRIBUTIONS

AP, ZC, MD, and PG contributed to conception and design of the study. AP wrote the first draft of the manuscript. ZC, PG, MD, MM, and SPE contributed to manuscript revision, read, and approved the submitted version. All authors contributed to the article and approved the submitted version.

FUNDING

The present study was partially supported by the Strategic Leadership Program. AP was partially supported by the Russian Science Foundation project 19−77−30012. This research was supported in part by the Cornell Atkinson Center for Sustainability.

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