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The Urban Lead (Pb) Burden in Humans, Animals and the Natural Environment

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Abstract

Centuries of human activities, particularly housing and transportation practices from the late 19th century through the 1980's, dispersed hundreds of millions of tons of lead into our urban areas.

The urban lead burden is evident among humans, wild and domesticated animals, and plants. Animal lead exposures closely mirror and often exceed the lead exposure patterns of their human partners. Some examples: Pigeons in New York City neighborhoods mimicked the lead exposures of neighborhood children, with more contaminated areas associated with higher exposures in both species. Also, immediately following the lead in drinking water crisis in Flint MI in 2015, blood lead levels in pet dogs in Flint were 4 times higher than in surrounding towns. And combining

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lead's neurotoxicity with urban stress results in well-characterized aggressive behaviors across multiple species.

Lead pollution is not distributed evenly across urban areas. Although average US pediatric lead exposures have declined by 90% since the 1970s, there remain well defined neighborhoods where children continue to have toxic lead exposures; animals are poisoned there, too. Those neighborhoods tend to have disproportionate commercial and industrial lead activity; a history of dense traffic; older and deteriorating housing; past and operating landfills, dumps and hazardous waste sites; and often lead contaminated drinking water. The population there tends to be low income and minority. Urban wild and domesticated animals bear that same lead burden.

Soil, buildings, dust and even trees constitute huge lead repositories throughout urban areas.

Until and unless we begin to address the lead repositories in our cities, the urban lead burden will continue to impose enormous costs distributed disproportionately across the domains of the natural environment. Evidence-based research has shown the efficacy and cost-effectiveness of some US public policies to prevent or reduce these exposures. We end with a series of recommendations to manage lead-safe urban environments.

Keywords

Lead; urban exposures; Environmental Justice; One Health; urban environmental policy

1. Introduction

The parallels are striking, with wildlife and our pets closely mirroring the urban lead exposure patterns of their human partners.

> A year after the Flint MI lead-in-water crisis splashed across US newspapers, pet dogs in Flint showed a fourfold increase in lead exposure compared to control dogs from a nearby, unaffected metropolitan area (Langlois et al., 2017).
>
> In August 1990, Millie, the First Dog under US President George H. W. Bush, was lead poisoned while living in the White House during exterior renovations (Radcliffe, 1990).
>
> Near a closed smelter, 10% of the humans and > 30% of the pets had elevated lead exposures. The authors recommended using the exposure of indoor pets to indicate risks in young children (Berny et al., 1995).
>
> The most common source of elevated lead exposure in cats is paint, and contaminated soil is second (Knight and Kumar, 2003).
>
> Mockingbirds living in New Orleans neighborhoods with high lead exposures exhibited heightened territorial aggression demonstrating a behavior similar to that observed in lead-exposed humans and laboratory animals (McClelland et al., 2019).

In this review, we show that across species, domestic and wild, lead exposures are higher in urban areas than non-urban. Indeed, animal exposures often exceed those of proximate human populations.

Lead contamination is not smoothly distributed even within urban areas. Distinct and predictable human populations have higher lead exposures (Frostenson and Kliff, 2016). Similar geographic distributions and exposure patterns are evident in pets, feral animals, urban wildlife, and plants. For instance, pigeons in New York City neighborhoods mirrored blood lead levels (BLLs) in neighborhood children, with more contaminated areas associated with higher BLLs in both species (Cai and Calisi, 2016); earlier studies showed similar results (Ohi et al., 1981; Tansy and Roth, 1970).

The urban lead burden is commonly acknowledged (Levin et al., 2008; Pirkle et al., 1994) but the common paradigm is incomplete. Vast lead repositories exist within cities, related to past and current lead sources, housing, business and transportation activities, that differentially impact inhabitants. An ecosystem-wide profile is necessary to characterize urban lead contamination. Without that, our ignorance will undermine our efforts to protect our urban children and the other species that share environments with us. One Health is an emerging discipline that enables cross-species and environmental data from our shared ecosystem to be integrated into a coherent whole. We used the One Health approach to array the data on the urban lead burden.

We conclude with recommendations based on evidence-based research that shows cost-effective policies exist to reduce urban lead burdens.

2. Materials and methods

2.1. EPA Integrated Science Assessment for Lead

Under the Clean Air Act (CAA), the Environmental Protection Agency (EPA) sets national ambient air quality standards (NAAQS) for lead and five other pollutants considered harmful to public health and the environment. Also required under the CAA, EPA must periodically review those standards to ensure they provide adequate health and environmental protection, and to update those standards as necessary. EPA does this by developing successive Integrated Science Assessments (ISAs); the ISAs were previously called ‘NAAQS Criteria Documents’ or just ‘Criteria Documents’. The ISA provides a concise review, synthesis, and evaluation of the science to serve as a scientific foundation to review NAAQS and includes a framework for evaluating weight of evidence and drawing scientific conclusions and causal judgments. The general process for NAAQS reviews is described at http://www.epa.gov/ttn/naaqs/review.html.

To identify all relevant scientific studies published since the last review of the NAAQS, EPA maintains an ongoing literature search process, including both general and specialized searches, which is iteratively reviewed and updated. EPA also issues a call for information in the Federal Register that invites the public to provide information relevant to the assessment, such as new or recent publications, studies, or applications. All relevant epidemiologic, controlled human exposure, toxicological, and ecological and welfare effects studies published since the last review are considered for inclusion in the ISA and are added to the Health and Environmental Research Online (HERO) database developed by EPA (http://hero.epa.gov). As of 6/1/20, there were over 104,000 entries in HERO related to lead.

The ISAs have supported numerous EPA regulations over several decades.

Hill (Hill, 1965) serves as EPA's metric to investigate 'causality' and distinguishes between associations and causality. Hill's criteria are refined by the ecological parameters of causality adapted by Mielke and Reagan (Mielke and Reagan, 1998) to investigate the relationship among variables and factors.

This evaluation of the urban lead burden rests upon EPA's successive ISAs, updated most recently in 2006 and 2013 (EPA, 2006; USEPA, 2013). Procedurally, each successive ISA collapses the findings from the previous assessments into areas of scientific consistency, which are cited through the previous ISA. Only discordant and new studies are cited by author in the most recent edition. We have followed this procedure and generally cite results and findings to the most recent ISA; however, where only a few studies present evidence, for areas not examined by EPA and for studies published after ISA 2013, we cite the publication by author.

Methodologically, we searched the ISAs for 'urban' and related terms (e.g., 'population density', 'city', etc.) for inclusion in this analysis. We used very little 'evaluative' judgement; for the purposes of this analysis the literature is remarkably consistent. The very few results that were inconsistent are noted within the text.

We supplemented the ISA with literature relevant to a One Health consolidated analysis of lead and to differential lead loadings within cities; the intent of the ISA is to support human health findings nationally, so not all studies related to animals or plants or to variability within cities were included. We searched the animal and veterinary literature for studies omitted from the ISAs and suspect that most of the omissions are not intentional. For instance, we found that within the animal/veterinary literature, high BLLs are referred to as 'lead toxicosis' while within the human health field, these are 'elevated' or 'poisonings'. Again, we used very little 'evaluative' judgement; again, the literature is remarkably consistent. Where inconsistencies exist, we have noted the discrepancies.

2.2 The One Health approach

One Health is an emerging interdisciplinary field that integrates the study of health across the triad of human health, animal health, and environmental health. We reviewed literature across the One Health triad according to the framework for One Health research (Lebov et al., 2017) and the checklist for One Health Epidemiological Reporting of Evidence (Davis et al., 2017). Evaluating lead contamination of air, water and soil and its adverse impacts on public health, wildlife, domestic animals, and environmental parameters provides a unique opportunity to integrate across disciplines and consider holistic solutions to an ancient problem (Levin et al., 2020; Pokras and Kneeland, 2008).

Again, we used the ecological parameters of causality and association adapted from Hill (Hill, 1965) by Mielke and Reagan (Mielke and Reagan, 1998) to investigate the commonalities in exposure patterns across the One Health triad.

2.3 Review focus and benchmarks

The focus of this review is primarily the US, but the findings are applicable throughout the industrialized world.

We have used existing federal statutory or regulatory benchmarks for describing environmental contamination levels and human exposures. These standards are not based strictly upon health risks. Most incorporate significant considerations of costs, enforcement and monitoring convenience, technological feasibility, and other priorities. None consider that no evidence of a threshold for lead toxicity has been found. One example is EPA's Lead and Copper Rule (LCR) regulating lead (and copper) in US public drinking water. Under the LCR, the Action Level (AL) is 15 ug/l (ppb) at the 90^{th} percentile of samples (U. S. EPA, 2019). Exceeding the AL is not a violation of the regulation (EPA, 2019). Public water systems violate the LCR only when they fail to perform additional actions required by the regulations after the AL is exceeded. EPA's guidance for drinking water in schools is higher -- 20 ug/l, even though schools are child-occupied facilities; it has no enforcement authority. Both of the water-lead levels were based upon feasibility judgements, not health.

A listing of EPA's lead standards is available at https://www.epa.gov/lead/lead-regulations. A listing that includes other federal lead standards, such as food and occupational exposures, is available at https://www.atsdr.cdc.gov/csem/csem.asp?csem=34&po=8.

3. Results

3.1. The development of US cities

The Victorian legacy.—The migration of workers from the countryside to the cities during the Industrial Revolution resulted in successive housing booms beginning in the 1850s and the building of millions of houses, many painted inside and out with lead-based paint (LBP). LBP continued to be used, albeit with diminishing frequency, until it was banned from residential use or in products for children in 1978. An estimated 38 million US housing units still had LBP in 2000 (Jacobs et al., 2002). LBP still accounts for up to 70% of the elevated pediatric BLLs in the US (Levin et al., 2008). (The often quoted '70% of EBLs are due to LBP' may be an overestimate. It results from a common protocol that if LBP is present in the child's environment that is the attributed cause of the EBL. There is little assessment of other potential sources and if present, what proportion of BLL is attributable to each source. Further, without isotopic analysis it is impossible to distinguish whether the soil or dust was contaminated by LBP, gasoline or other sources.)

The Victorians also installed more-expensive lead water pipes in those burgeoning residential districts because they were anticipated to last for 50 years or more. In 1900, over 70% of US large cities (populations exceeding 30,000) used lead pipes (Rabin, 2008). Installation of new lead service lines was not prohibited in the United States until the Safe Drinking Water Acts Amendment of 1986 (Rabin, 2008). In 2016, EPA estimated that between 5.5 and 10 million lead service lines (LSLs) still provide water to an estimated 15 to 22 million Americans (Cornwell et al., 2016; U. S. EPA, 2016).

The biggest contributor to environmental lead contamination, however, was tetraethyl lead added to gasoline to prevent knocking. Since its introduction in the early 1920's, leaded gasoline quickly surpassed other fuels; it also became the principal lead exposure source of the US population (EPA, 1986). The highest US emissions from leaded gasoline were during the 1960's and 1970's and coincide with the peak BLLs (US EPA, 1996). Lead use in gasoline decreased by 99.8% between 1976 and 1990 (EPA, 1991), as did BLLs. Where leaded gasoline has been banned, average population BLLs have declined rapidly (Meyer et al., 2008).

US cities after World War II—US veterans returning from World War II produced the 'Baby Boom' of the late 1940's and 1950's. To meet the need for new housing, the G.I. Bill (officially the Servicemen's Readjustment Act of 1944) provided benefits for those veterans, most notably low-cost mortgages and educational payments. Mass–produced housing in preplanned suburban communities was cheaper than existing housing. One-fifth of all single-family homes built in the 20 years following World War II were financed with help from the GI Bill's loan guarantee program; even working-class Americans could afford to own homes in the suburbs.

But overt racism by several members of the US Congress, under the guidance of Mississippi Congressman John Rankin, blocked black veterans from receiving many of those benefits, especially the housing loan guarantees (Rothstein, 2017). Less than 2% of loans from the Federal Housing Administration (FHA) issued between 1947 and 1959 went to African Americans (Rothstein, 2014). So, the suburbs became and remained bastions for whites, with blacks largely contained in older urban centers.

This established a racial lead-exposure pattern that has continued and intensified. Black populations in cities increased and African Americans began moving into formerly all-white neighborhoods with older housing (Fredrickson, 2016). Panicked white homeowners sold their homes and moved away, a pattern known as 'white flight'. 'Blockbusters' used the threat of integration to get white homeowners to sell for low prices. Much of the housing then became rental housing, often that was poorly maintained.

During the post-World War II period, US transportation policy also spurred suburbanization. The 1956 Interstate Highway Act authorized billions of dollars to complete over 40,000 miles of highways, half of which were to go through cities (Fredrickson, 2016). The purpose was to provide transportation corridors from the developing suburbs for suburbanites to work in the city. The cycle was self-reinforcing: The proliferation of interstates and automobiles made downtowns increasingly obsolete, encouraging further movement to the suburbs. Suburban commuting contributed directly to urban lead poisoning; deposition of leaded particles from commuters who converged on cities daily contaminated urban homes and neighborhoods.

The suburbs were not lead free, however. They depended upon automobiles for access, and from the 1950's and into the 1970's, leaded gasoline contaminated suburban yards and homes. There were other local and regional lead sources, but they did not approach the contamination level of older urban areas.

3.2. Past lead sources and current environmental loadings

Both environmental and anthropogenic factors account for more intensive lead loadings in urban areas (EPA, 2006; USEPA, 2013). High modification of the urban environment by human activity, including buildings, pavements, and high-density motorized transportation, influences the distribution and redistribution of lead particulate matter, in some cases, spreading localized contamination over a wider area and in others, maintaining high lead loadings in localized areas (USEPA, 2013). Another difference in urban areas is the shorter transport distance for soil or dust to penetrate indoors.

Paint, gasoline and plumbing were the dominant sources to contaminate US urban areas, but dozens of other industries and incinerators also emitted, disgorged and loaded lead onto workers, near-by residents, consumers and the environment. Historically, commercial and industrial activities occurred conterminously to habitation; consequently, older cities are more contaminated than newer (EPA, 2006).

Past lead sources have created current lead repositories.

3.2.1 Lead in environmental media

3.2.1.1 Air.: **A**ir-lead emissions declined by 98% from 1970 to 1995 and then by an additional 77% from 1995 to 2008 (EPA, 2006, 1986; USEPA, 2013). Nonetheless, urban atmospheres still have more airborne lead than nonurban, contributing to increased exposures by inhaling air and consuming contaminated dust (Strosnider et al., 2017). Long-distance transport of lead occurs, but most of the lead emitted in urban areas remains there (Flegal et al., 1989).

Leaded gasoline was the dominant source of airborne lead in the 20th century until it was restricted and then banned during the 1970's and 1980's; higher traffic density in urban areas resulted in higher lead loadings. The typically very small lead combustion-derived spherical particles associated with vehicle exhaust emissions may not only be easily absorbed through the respiratory tracts of children, they may also be directly absorbed into the brains of children via the olfactory tract (Maher et al., 2008; Menon et al., 2016). After the peak of leaded gasoline usage in the 1960's and 1970's, the 99.8% decrease in use was matched lockstep by the decrease in BLLs.

3.2.1.2 Soil and dust.: Soil is the major reservoir for lead previously deposited from past activities, near and distant, of which leaded gasoline and lead-based paint (LBP) remain the principal sources, with gasoline more widespread (Mielke et al., 2011). EPA estimates that about 6 million metric tons of tetraethyl lead was added to gasoline (EPA, 2006).

About equal amounts of lead were used in LBP pigment between 1884 and 1989 as in leaded gasoline between 1929 and 1989 (National Research Council 1993). LBP contaminates soil and interior and exterior dust (Clark et al., 1991). Figure 1 overlays US production of LBP with tetraethyl lead from gasoline. LBP and leaded gasoline were used in both urban and non-urban environments, but due to traffic and housing density, environmental loadings were more concentrated in urban areas. Homes in urban areas are generally older, and while LBP was not banned until the 1970s, its use decreased since the 1930's: 87% of housing built before 1940 contains LBP, 69% of housing built between 1940 and 1959 contains LBP, and 24% of housing built between 1960 and 1977 contains LBP(EPA, n.d.). Suburban commuters disgorged leaded gasoline particles in their daily commutes into the city to work, contributing to higher lead environmental loadings.

The retention time for lead in soil is longer than in air and soils are active exposure repositories through periodic re-suspension of fine lead particulates (or aerosols). Resuspended airborne lead and soil-lead concentrations are proportional to the size of the city and amount of vehicle traffic. Larger cities present higher amounts of Pb than small towns, and concentrations also relate to population density (Datko-Williams et al., 2014; Mielke et al., 2011).

Soil lead concentrations can vary greatly even within small areas (Filippelli et al., 2018). In residential areas, soil near the dripline from the roof typically has higher lead values on a given property, with this pattern exacerbated in properties nearer the urban core (Filippelli et al., 2018). Soil lead concentrations tend to be highest in old downtown urban areas and decrease with increasing distance from the city center (Filippelli et al., 2018). Older studies found soil-lead levels highest near roadways (EPA, 2006); urban soil-lead levels may now be declining (H. W. Mielke et al., 2019). Urban areas with smelting and similar industrial histories are likely to have very high lead loadings in soil as well as outdoor and indoor dust (Simon, 2013; Taylor et al., 2013).

Dust contaminated by anthropogenic Pb is more toxic and potent than naturally occurring Pb dust (Laidlaw et al., 2005).

Throughout the US, soil- and dust-lead concentrations, including in playgrounds, correlate with BLLs, income and race/ethnicity (Filippelli GM and Laidlaw MAS, 2010;Mielke and Reagan, 1998). The association holds even in newer cities such as Phoenix, AZ (Zhuo et al., 2012).

In addition, there is a seasonal pattern of BLLs that increases during summer and fall and decreases during winter and early spring (Levin et al., 2020; Zahran SA, Laidlaw MA, McElmurry SP, Filippelli GM, 2013). The seasonality of BLLs has been linked to environmental acidification, which may increase lead bioavailability and mobility from environmental media (air, water and soil) to biological tissues (Levin et al., 2020). Global warming would intensify this pattern.

Outdoor and street dust is contaminated by proximate lead in soil, air, housing, etc, and cycles with interior dust. BLLs correlate with transportation corridors (both in and outside of urban centers), where lead from gasoline accumulated along with pulverized brake pads, tires, and leaded wheel weights (Aelion and Davis, 2019; EPA, 2006; Macey et al., 2001). EPA estimated that 1.6 million pounds of lead tire weights fall off and are pulverized by traffic annually (EPA, 2009; Root, 2000). In addition, road paint can legally contain lead and contaminate road dust (Grue et al., 1986; Murakami et al., 2007).

Vacant lots can be convenient dumping grounds for any material. They can also serve as play areas for local children, habitats for animals or urban gardens.

Sources of lead in **indoor dust** include lead paint, tracked or resuspended soils from outside and take-home lead brought in on the clothing and shoes of lead-exposed workers. Indoor dust lead correlates tightly with children's BLLs (Filippelli et al., 2018; Lanphear et al., 1998b; Mielke et al., 2011; O'Connor et al., 2018). Dust loadings and particularly dust-lead concentrations are associated with the age of the housing (Rasmussen et al., 2013; Whitehead et al., 2014). Young children usually are exposed through normal, age-appropriate hand-to-mouth activity, when lead-contaminated dust settles on their toys and the floor (Lanphear et al., 1998b; Mielke et al., 2011).

3.2.1.3 Water.: Legacy lead sources, including industries and leaded gasoline, contributed to higher lead loadings in nearby surface waters and sediments (Hutchinson and Rothwell, 2008). EPA has identified atmospheric deposition, urban runoff/snowmelt and industrial discharges as major lead sources in surface waters,(EPA, 2006) even in the absence of specific identified emission sources (Nowell et al., 2013).

Lead can appear in the water column, but exposures are generally transient through that pathway as lead dispersal in waterways occurs relatively rapidly (EPA, 2006). Concentrations in surface waters are highest close to point sources then significantly reduced through flushing, dilution, and sedimentation.

Lead is relatively stable in sediments, with limited mobility and long residence times (EPA, 2006). Areas near past lead-emitting industries and urbanized areas tend to have higher sediment lead loadings than areas farther from current or past sources (Callender and Rice, 2000; Chalmers et al., 2007). There is an order of magnitude difference between lead sediment in urban vs rural water bodies (Chalmers et al., 2007).

Rain runoff or snowmelt can redistribute lead within an urban area (EPA, 2006). Urban roof runoff and rainwater tanks have caused lead contamination of adjacent water bodies; the lead sources included flashing on the roofs and possibly lead-stabilized PVC drainpipes (USEPA, 2013). Severe contamination of near-by ecosystems from urban runoff and domestic wastewater discharge has been documented (Soto-Jiménez and Flegal, 2009). An estimated 3,000 tons/year of lead is flushed by storm runoff from residential areas of US cities with populations > 100,000 (EPA, 1990).

Drinking water.: Lead contamination of tap water occurs primarily after the water leaves the treatment plant and passes through pipes, solder, faucets, valves and other plumbing fixtures containing lead (U. S. EPA, 2019). Lead pipes are the component most likely to leach significant lead into the drinking water. Although lead pipe installation decreased in the 1900's and virtually ceased after World War II, the city of Chicago, for instance, mandated them until they were banned by Congress in 1986 (U. S. EPA, n.d.). Galvanized pipe was generally installed prior to the 1960s, and lead solder was ubiquitous until 1986, when it was also banned nationally. As a result, cities, especially older cities, have higher water lead levels (WLLs) (Etchevers et al., 2015; Levin et al., 2008; Renner, 2010). Brass- and chrome-plated fixtures can also leach lead into the drinking water.

Numerous studies have shown that WLL remains poorly monitored and addressed at all levels in the US, local, state and federal (Leonnig et al., 2004; Olson and Pullen Fedinick, 2016; Triantafyllidou and Edwards, 2012). EPA's regulatory program reports that for public water supplies, larger cities are generally associated with fewer lead drinking water violations than are smaller systems (Strosnider et al., 2017). According to EPA at least 15 water systems serving over 100,000 people had lead violations in 2015 (Olson and Pullen Fedinick, 2016). However, several national studies show much higher contamination levels across the US. They document that actual violations in large water systems are likely severely underreported in EPA's data, and that millions of Americans likely have elevated WLLs (Leonnig et al., 2004; Young and Nichols, 2016). High WLLs are not always regulatory violations, however. Even these data probably underestimate actual occurrence significantly (Olson and Pullen Fedinick, 2016). An estimated 25% of domestic dwellings in the European Union have lead pipes, potentially putting 120 million people at risk from lead in drinking water (Hayes and Skubala, 2009).

High lead contamination of drinking water is disproportionately reported or uncovered in low income and minority communities (Butler et al., 2016; Gostin, 2016). Flint MI is the best known, but not exceptional. It typifies the disparity in lead exposures from drinking water by race and income.

The particular use patterns in schools (long periods of disuse overnight, weekends and vacations) foster high WLLs. Detectable concentrations of lead in school water systems have been documented in at least 38 US states and the District of Columbia, including several exceedances of the federal drinking water standard (Lambrinidou et al., 2010; Olson and Pullen Fedinick, 2016; Young and Nichols, 2016). Urban schools, which tend to be older than suburban schools, are also more likely to have plumbing components containing lead. Resulting contamination can be high: for instance, 57.4% of Philadelphia's public-school buildings had WLLs exceeding EPA's school lead action level of 20 ppb, and 28.7% had water with mean lead levels over 50 ppb (Bryant, 2004).

3.2.2 Lead in buildings—Buildings are full of lead: paint, varnishes, caulking, electronics, plumbing, additives in brass/alloys, gutters, roof flashing, glazing, cable and wire casing, mortar in brick and stonework in older buildings, structural steel primer, etc. as well as lead-contaminated dust and soil both inside and out. Other than paint and some new plumbing components in residences, lead uses remain unregulated. Buildings thus constitute large lead repositories. Except in urban renewal areas, buildings in cities are generally older than in non-urban areas.

Housing.: The average age of housing has likely increased in the US in the past 30 years (Zhao, 2017). Until it was banned in 1978, LBP was often used for both interior and exterior surfaces. After the ban LBP did not disappear. An estimated 76% of the housing units built after 1960 contain some LBP as do 92% of houses built between 1940 and 1959 (Clickner et al., 2001). EPA and HUD estimated that 38 million US housing units still had LBP in 2000 (Jacobs et al., 2002; Jacobs and Nevin, 2006).

The older the housing, the higher the association with elevated BLLs in children (Sargent et al., 1995; Vivier et al., 2011). The combination of older housing, rental property and a larger city is particularly significant (Pirkle et al., 1998).

Income is a determinant of housing condition, and deteriorated housing presents the highest risk of high BLLs in residents (Jacobs et al., 2002; Landrigan et al., 2010). Deteriorated housing is often clustered in low income and minority areas.

Renovations and repairs to older housing are a risk factor both for elevated exposures of those near-by, especially children, as well as localized environmental contamination (Bellinger et al., 1986; Jacobs, 1995)(Spanier et al., 2013). Pets are not immune; under President HW Bush, the first dog, Millie, was lead poisoned when the portico of the White House was renovated (Radcliffe, 1990). Demolition or burning old buildings will also contaminate both soil and dust (EPA, 2006; Farfel et al., 2003; Rabito et al., 2007),(Farfel et al., 2005).

Public and commercial buildings contain all of the lead components of housing and are not subject to the ban on LBP. Paints and primers containing lead are commonly used to protect steel structures from corrosion. Further, LBP on furniture, fixtures such as handrails, doors, windows, indoor and outdoor equipment, etc. along with plumbing, electronics, roofing, etc. has continued to be used legally in the US. Due to fewer protections, lead loadings in public and commercial urban buildings may be high.

Past practices have heavily loaded some buildings with lead. For instance, after the devastating Notre-Dame Cathedral fire in 2019, honey samples from the downwind area had higher lead concentrations than from elsewhere in central Paris (Smith et al., 2020). Children were lead-poisoned there, too.

Schools have some features of housing and some of public and commercial buildings. The LBP ban applied to some school building components but not all. For instance, offices, commercial recreational venues, play fields, etc, could contain equipment, paint, and consumer goods containing lead. As with housing, soil and dust in schools may entrap lead particles from air and other sources (Rodrigues et al., 2018).

3.2.3. Lead use in human activities—**Off-road vehicles** and other uses of leaded gasoline were exempt from EPA’s ban on lead in gasoline (US EPA, 1996). Local airports are a particular exposure source. Currently the largest source of Pb in EPA’s National Emission Inventory is emissions from leaded aviation gasoline (av gas) in piston-engine aircraft, constituting 45% of US lead emissions (EPA, 2020; USEPA, 2013) (Figure 2). There are about 13,000–20,000 municipal and general airports where av gas is used, most of which are within or immediately adjacent to densely populated urban centers (EPA, 2020). Airborne lead concentrations are higher there, exposing both proximate and downwind communities as well as workers; children, who inhale more air based on body mass than adults, are particularly vulnerable (Chen and Eisenberg, 2013; EPA, 2010; Miranda et al., 2011; Zahran et al., 2017). The EPA estimated that 16 million people live within 1 kilometer of an airport servicing piston-engine aircraft (EPA, 2010). This may slightly underestimate

actual lead exposure from av gas due to the at least hypothetical contamination of unleaded gasoline if distributed through pipes that also deliver av gas.

Auto repair shops.: In the US and worldwide, the largest single use for lead is in automobile and other vehicular storage batteries (Gearhart et al., 2003). Overall, the automobile industry is the greatest lead consumer in the US currently, accounting for the release or transfer each year of more than 300 million pounds (136,508 metric tons) of lead (Gearhart et al., 2003). LBP is still used in industrial paints including car paint. Lead is also used in car electronics and radiators (as solder), wheel weights and numerous other applications. As a result, auto body and auto repair shops can significantly expose workers, their families and near-by residents, and occupational lead poisoning is not uncommon (E-News, 2006). Most states regulate car repair operations to some extent, but because of the decentralized nature of the industry, compliance appears to be low (Levin et al., 2008). Unregulated and pop-up auto repair facilities are common in urban areas, especially in low income and minority areas.

Municipal landfills.: Many materials containing lead, including construction debris and consumer goods, are disposed of in both regulated and makeshift waste accumulation sites. Municipal solid waste landfills (MSWLFs) receive household waste, and may also receive other types of nonhazardous wastes, such as commercial solid waste, nonhazardous sludge, conditionally exempt small quantity generator waste, and industrial nonhazardous solid waste. The Resource Conservation and Recovery Act (RCRA) requires waste that exceeds the lead toxicity characteristic limit of 5 mg/L be managed as hazardous waste (EPA, 1998). However, EPA specified that home renovation debris, including LBP debris, may be disposed of as household waste and is exempt from the regulation (EPA, 2000). Furthermore, it is permissible to dispose of LBP debris in construction and demolition debris landfills. Lead has been detected in leachate from MSWLFs and in construction and demolition debris landfills in the US (Jang and Townsend, 2003; Wadanambi et al., 2008).

An integrated review found pediatric BLLs correlated with landfill areas (Kim and Williams, 2017). Numerous studies both in the US and in Europe have found landfills disproportionately sited in low income and minority neighborhoods (Martuzzi et al., 2010; Mohai and Saha, 2006).

A **scrapyard** (also called a junk yard) is a recycling center that buys and sells scrap metal. There are more than 8,200 registered companies in the US (Huffman, 2014), most in or immediately adjacent to urban areas. Scrap or junk cars are a large part of the business along with electronics and appliances, which all contain lead; for scale, about 12.6 million cars are recycled annually in the US (Huffman, 2014). Contamination of the surrounding area by lead and other contaminants (PCBs, mercury, arsenic, etc) is common. Few states regulate salvage yards to limit local risks; VT is one (Huffman, 2014).

Lead Smelters.: Peak primary lead production in the US occurred in the 1920s, but only 3 lead smelters and refineries were still operating in the US in 1999; some of the areas around those defunct primary smelters are now more densely populated (Rabinowitz, 2005). Secondary lead production (recovery of lead from scrap, primarily for lead-acid batteries) expanded during the early 20th century and because batteries are heavy and expensive to transport, secondary smelters developed near the largest sources of scrap, viz. large cities. The secondary industry now accounts for most lead produced globally, and in the US more than 80% of lead comes from secondary production (Group, n.d.; Survey, n.d.; US Geological Survey, n.d.). The sites of many historic lead smelting operations remain unidentified. Eckel et al (2001) used historical records to identify 639 lead smelter sites, of which, 430 were absent from US EPA or state records; 8 defunct secondary smelters within the cities of Philadelphia and Baltimore had highly contaminated soil within or close to residential or commercial districts (Eckel et al., 2001). Active and defunct secondary smelters are disproportionately located in low-income and minority areas (Incorporated, n.d.).

Brownfields are land previously used for industrial or commercial purposes with known or suspected pollution including soil contamination due to hazardous waste (U. S. EPA, 2018). There are an estimated 450,000 Brownfield sites in US urban areas, which may be an underestimate (Simonds, 1998). EPA estimates that about 5% of US children under age 5 and about 15 million Americans overall live within 1 mile of a registered Superfund site (U. S. EPA, n.d.). Superfund and Brownfield sites are disproportionately sited proximate to low income and minority areas (Miller et al., 2011).

Urban agriculture.: EPA's maximum allowable lead concentration in residential soil is 400 mg/kg. A study in New York found that 44% of gardens tested had at least one soil sample exceeding 400 mg/kg lead, with a maximum value of 2450 mg/kg (Mitchell et al., 2014). Soils in vacant lots in Cleveland and Columbus, Ohio and Terre Haute, Indiana ranged from 116 to 1246 mg/kg (Kaiser et al., 2015; Latimer et al., 2016) and soil lead concentrations in Milwaukee, Wisconsin were higher (Johnson et al., 2016). Similar results are found in Australia and Europe (Entwistle et al., 2019; Pelfrêne et al., 2019; Rouillon et al., 2017).

While some plants uptake lead directly from air, the primary source of lead in most produce is contaminated soil (Brown et al., 2016; Xiong et al., 2016). Plant uptake of lead from soil is usually low, and is species specific (Entwistle et al., 2019; McBride et al., 2014). Atmospheric deposition is likely the major source of lead in leafy vegetables like lettuce (Dala-Paula et al., 2018).

Although some had modest elevations, vegetables grown in urban gardens did not generally have higher lead concentrations compared to commercial produce (Kohrman and Chamberlain, 2014; McBride et al., 2014). Thorough washing decreased lead in produce grown in contaminated soil, but root vegetables, such as radishes and carrots, retained higher lead concentrations (Brown et al., 2016; Goldstein, 2016). Using compost can reduce the potential of transferring soil Pb to humans via vegetable consumption and direct soil ingestion (Attanayake et al., 2014), but lead contamination of municipal compost has also occurred (Pfeiffer and Jolicoeur, 2012). Urban gardening may provide some soil-lead remediation: some established urban gardens exhibited lower soil lead concentrations than newly established gardens and vacant lots (Kaiser et al., 2015; Mitchell et al., 2014). A few US and European studies found higher lead levels associated with urban gardens than commercially obtained produce, especially in areas with previous lead histories (Pelfrêne et al., 2019; Ruderman et al., 2017; Warming et al., 2015).

Gardeners also risk Pb exposure through inhalation and inadvertent ingestion of soil through hand-to-mouth contact (EPA, 2011; Spliethoff et al., 2016), although a study in Europe found no difference in BLLs between adult urban gardeners and non-gardeners (Entwistle et al., 2019). Children are often actively involved in family gardens, with 89–100% of gardeners reporting that children assisted in garden activities (Johnson et al., 2016).

There are at least 1500 community gardens in New York City (Mitchell et al., 2014).

The USDA estimated in 2010 that approximately 1% of urban dwellers in 4 major US cities kept poultry (USDA, 2010). Chickens, due to their natural foraging behavior, are susceptible to ingesting lead-contaminated particles (Grace and MacFarlane, 2016; Sobhakumari et al., 2019). Egg lead concentrations tend to correlate with soil Pb concentrations where the chickens are housed and lead contamination in the eggs may present a risk to consumers, especially children (Bautista et al., 2014; Grace and MacFarlane, 2016; Leibler et al., 2018; Mordarski et al., 2018).

After poultry, honeybees are the most commonly kept livestock on urban farms (Oberholtzer et al., 2016). Several European studies found lead levels ranging from detectable to significant in apiculture (Gajger et al., 2019; Jovetić et al., 2018; Lambert et al., 2012; Sadowska et al., 2019). Lead exposure has also been documented in urban livestock, including small ruminants like goats and sheep, and lead has been detected in milk from grazing animals (Patra et al., 2008; Swarup et al., 2005).

Diet.: Dietary sources of lead may be natural or anthropogenic (Levin et al., 2008). As discussed above, urban gardening may present an exposure risk both to the gardeners, especially children, and through the yield. In addition to produce, this includes milk from animals that graze in urban areas (Patra et al., 2008; Swarup et al., 2005) and eggs from backyard chickens (Leibler et al., 2018). Although no systematic surveys are available, for urban populations who participate in local subsistence fishing, lead dietary exposure may be higher than in non-urban areas (USEPA, 2013). Lead is sometimes found in imported spices, teas and tonics and can be a dominant lead exposure source in subpopulations (Lin et al., 2010).

Marinas.: Fuel for boats is also exempt from EPA's ban on lead in gasoline (U. S. EPA, 2016), and maritime paint and other components are exempt from the ban on LBP. In addition, lead is used in on-board electronics, plumbing, consumer goods, building components, solder and myriad other applications. As a result, both fresh- and salt-water marinas can be contaminated with lead (Hinkey et al., 2005; McMahon, 1989). The protection marinas offer to vessels from wind, currents, and waves may concentrate contaminants in sediments (Hinkey et al., 2005). Contamination contributed to the water can be significant; one study estimated that daily net flux out of the marina was on the order of hundreds of grams of metals (Crecelius et al., 1989).

Consumer products.: In 1978, LBP was banned in the US for children's products, including furniture, toys, and other articles. But over the past few decades, paint production and consumption have been steadily growing in developing countries, many of which lack restrictions on LBP (Clark et al., 2014; Kessler, 2014). Much of this paint contains extremely high lead concentrations. And it has entered the US (Levin et al., 2008; O'Connor et al., 2018).

Lead-contaminated consumer products, including ceramic cookware, traditional cosmetics, and jewelry, are widely available (Etchevers et al., 2015; Farley, 1998; Flegal and Odigie, 2020; O'Connor et al., 2018; Sharmer et al., 2010). Many goods made of polyvinyl chloride (PVC) including housewares, shower curtains, raincoats, toys, school supplies, food packaging, shoes, window blinds, Christmas trees, etc, also contain unsafe lead concentrations (Greenway and Gerstenberger, 2010; Turner, 2018). Between 1977 and 2014, the Consumer Product Safety Commission (CPSC) issued 350 recalls of more than 200 million consumer items for lead exposure risks (Dignam et al., 2019). An influx of lead-contaminated goods intended for children entered the US in the early 2000's (Levin et al., 2008), and in 2007 and 2008, CPSC recalled over 10 million children's toys in the "Year of the Recall" (Allen, 2008; Reports, 2007). In response, in 2008 Congress enacted the Consumer Product Safety Improvement Act, which mandated reducing the lead limit in these products to 0.009% by weight (US Consumer Product Safety Commission MD, 2008). Policy limits have not ensured success, and lead-contaminated household goods remain common (O'Connor et al., 2018; Shen et al., 2018).

Much of this material remains in homes, schools and outdoors. The higher density of urban living suggests there may be more consumer products potentially containing lead there.

While **occupational lead exposures** can occur anywhere, certain commercial activities with potential lead exposures are more common in urban and especially low-income communities, including auto and auto body repair, housing renovation, building construction and demolition, secondary smelting, electrical and plumbing repairs, domestic and commercial cleaning, etc (Dalton et al., 1997; Lipscomb et al., 2006; Nunez et al., 1993). Lead exposures are multiplied through contamination of the local environment, high exposures of those directly involved and take-home exposures to families and communities (Administration, n.d.; Newman et al., 2015). Low income and minority workers are over-represented in these businesses. Unfortunately, many of them have few employees and fall below regulatory or reporting levels and so are poorly controlled.

Leaded ammunition.: Currently, 3% of US lead production is used for ammunition (Group, n.d.; Survey, n.d.; US Geological Survey, n.d.). Indoor shooting ranges can present extremely high exposures to participants and workers; high BLLs among police officers and soldiers are common (Laidlaw et al., 2017). EPA does not consider lead shot/bullets of a designated firing range as 'hazardous waste'; EPA's reasoning is that the bullets are not 'discarded' and can be recovered or reclaimed on a regular basis (EPA, n.d.). Among the cited benefits of operating an indoor shooting range are profitability and convenience to high traffic areas (Port, 2020).

3.3. Evidence of the urban lead burden in receptors

US urban areas can contain high lead concentrations as discussed in sections 3.1 and 3.2, the older the city, the more likely to be highly contaminated. Past loadings constitute repositories and recurring exposures for those who live there, including humans, animals and plants. The odds ratio doubles for the increased risk of elevated BLL for living in an urban area (Lanphear et al., 1998a).

Human urban lead burden—The discrepancy in lead exposure of urban and rural children has been evident for decades (Annest et al., 1983; Pirkle et al., 1994;Lin-Fu, 1972). Young children and infants are more susceptible to lead due to normal hand-to-mouth activity and because their higher calcium demand increases the proportion of dietary lead absorbed (DC, 2004). Children have higher BLLs, peaking at ages 1–2; BLLs are higher in Black children than in white and correlate negatively with income (Lanphear et al., 1996; Lin-Fu, 1992; Mahaffey et al., 1982). Immigrant children also often have higher BLLs than children born in the US, but at least some of that higher exposure may occur after they arrive in the US (Kaplowitz et al., 2016; Tehranifar et al., 2008).

Children from low-income families living in poor and inner-city neighborhoods are 5 times more likely to have elevated BLLs (Bashir, 2002). BLLs of urban adults are also higher (Lee et al., 2005). This discrepancy has remained through the decrease of over 90% of in US children’s BLLs in the past 40 years, although the scale of the burden may be declining (Aelion and Davis, 2019; Gulson et al., 2018; Morrison et al., 2013).

Lead urban burden in animals—Domestic animals and wildlife share indoor and outdoor environments with humans, and lead exposures in pets mirror human exposures and environmental contamination levels (Backer et al., 2001). Following the publicized water-lead contamination event in Flint MI, for instance, BLLs in local dogs were fourfold higher compared to control dogs from a nearby, unaffected metropolitan area (Langlois et al., 2017). Another study, near a closed smelter in IL, found that human exposures were highly correlated to exposures in pet dogs and cats; the authors recommended using the BLLs of indoor pets to indicate exposure risks in young children (Berny et al., 1995). Most notable was the fact that when there was a pet with a high BLL in a house, the likelihood of finding a person with a BLL above 10 ug/dl was significantly increased (Berny et al., 1995). Domestic animals were more likely to have elevated blood lead concentrations than people in the same households (Berny et al., 1995).

Of domestic companion animals, canine lead exposures best reflect their human counterparts (Hamir and Handson, 1981; Hamir et al., 1986; Zook, 1973). BLLs are higher in urban dogs than rural (Balagangatharathilagar et al., 2006; Mañay et al., 2008; Swarup et al., 2000) but lower than in smelting and other metal processing areas (Koh and Babidge, 1986). A study from the 1980s, when leaded gasoline was still used, found canine BLLs highly correlated with traffic flow (Kucera, 1998). Elevated lead exposures in indoor pets, particularly dogs, were associated with an increased likelihood of higher blood lead concentrations in younger children in the home (Berny et al., 1995). Dogs can be poisoned when their homes that contain lead paint are renovated, even if that house is the White House (Radcliffe, 1990).

Monitoring lead levels in dogs has been recommended as a sentinel for protecting children (Hamir and Handson, 1981; Hamir et al., 1986).

Lead exposure patterns in pets mimic human exposure patterns. The most common source of elevated lead exposure in cats is paint, and contaminated soil is second (Knight and Kumar, 2003). Paint is the most common source for household dogs (Morgan et al., 1991; Radcliffe, 1990). Lead poisoning associated with paint ingestion has been reported in house rabbits and pet psittacine birds (Morgan et al., 1991; Walter et al., 2017).

Elevated Pb in urban wildlife and feral animals is common, particularly among the birds that inhabit most cities, whose normal foraging and grit-seeking behavior results in chronic lead exposures (Behmke et al., 2015; Williams et al., 2018). Feral pigeons are ubiquitous worldwide in urban settings, with normal foraging behavior that increases the likelihood of ingesting lead particles. Pigeons living in urban and industrial areas in Virginia, Pennsylvania, Paris, Japan and Korea have higher lead concentrations than rural pigeons (Frantz et al., 2012; Kendall and Scanlon, 1982; Nam and Lee, 2006; Ohi et al., 1974; Tansy and Roth, 1970). By neighborhood, BLLs in New York City pigeons mirrored BLLs in local children with more contaminated areas associated with higher BLLs in both species (Cai and Calisi, 2016); an earlier study had shown similar results in Japan (Ohi et al., 1981). Lead toxicosis has been associated with pigeons foraging on urban pavements and in road gutters (Lumeij, 1985).

House sparrows have also adapted well to living with humans in urban areas worldwide, with resulting global evidence of lead exposures (Andrew et al., 2019; Cid et al., 2018; Kekkonen et al., 2012; Swaileh and Sansur, 2006). Urban sparrows have higher BLLs than rural, and the more contaminated the urban area, the more prevalent higher leads in birds (Bichet et al., 2013; Millaku et al., 2015).

A cross species study found lead higher in both urban common blackbirds and earthworms (Scheifler et al., 2006).

Urban rodents in Argentina (Tripodi et al., 2018), Texas (Way and Schroder, 1982), Milan (Ceruti et al., 2002), and Cairo (Sures et al., 2003) have higher lead exposures. Other wild mammalian species also exhibit lead's urban burden, including the raccoon, which frequents urban and semi-urban habitats, often scavenging human garbage and litter. Raccoon exposures have ranged from only elevated to clinical Pb toxicosis (Diters and Nielsen, 1978) when compared to less urban exposures (Hamir et al., 1994; Khan et al., 1995; Sanderson and Thomas, 1961).

Red foxes captured within urban habitats in Switzerland had increased Pb per kilogram over foxes from suburban and rural habitats (Dip et al., 2001).

Grazing mammals have higher lead exposures in urban areas. Sheep in London that grazed along a motorway had higher BLLs (Ward and Savage, 1994). Urban sheep in Egypt that grazed on street garbage had higher BLLs compared to sheep that were housed indoors at the university veterinary teaching hospital. Sheep, horses and cattle in India, China and Kazakhstan show similar exposures (Dwivedi et al., 1995; Farmer and Farmer, 2000; Liu, 2003; Patra et al., 2008; Swarup et al., 2005). Furthermore, livestock are used for food. Some of these studies demonstrated a significant correlation between BLLs and milk lead levels, increasing potential human exposures. The urban lead burden is evident also in honey (Smith et al., 2020, 2019).

Even on the subclinical level, lead exposure is not without other physiological consequences. Lead's same pathological damages in humans are evident in animals, including hepatic and renal damage, immunosuppression, oxidative balance, and neurotoxic effects; these have effects on long-term survival in wild birds (Bichet et al., 2013; Sprowles et al., 2018). Avian reproductive toxicity related to lead exposure is also evident (Vallverdú-Coll et al., 2016). Studies have also examined the relationship between lead exposures and the prevalence of avian malaria (Bichet et al., 2013), blood chemistry alterations (Cid et al., 2018), protective differences in gene expression in house sparrows (Andrew et al., 2019) and behavioral changes, including increased aggression in urban songbirds (McClelland et al., 2019). Increased mortality in whooping cranes is likely related to lead exposure, through overt poisoning and as a contributing factor to predation or trauma (Cole et al., 2009). Both lead exposure and urbanicity are also associated with decreased species richness (Melles et al., 2003; US EPA, 2001; USEPA, 2013).

Lead urban burden in plants—In plants, the adverse effects of lead relate to its competition with calcium in cell activities, as it does in humans and animals (Sharma and Dubey, 2005). Symptoms of lead toxicity include stunted growth, chlorosis and blackening of the root system. Pb inhibits photosynthesis, upsets mineral nutrition and water balance, changes hormonal status and affects membrane structure and permeability (USEPA, 2013). Plants exhibit evidence of oxidative stress when exposed to lead, with reactive oxygen species increased in some plants and related systemic effects (USEPA, 2013). Several taxa grown in lead-contaminated soil responded to increasing exposure with increased antioxidant activity (USEPA, 2013). Reduced growth is sometimes found, as well as genotoxicity, decreased germination, and pollen sterility (USEPA, 2013).

Community and ecosystem-level research of lead has focused on soil microbial communities, showing alterations in both composition and activity (USEPA, 2013). Generally, soil microbial activity was diminished.

High exposure areas, including highly contaminated urban areas, show decreased species diversity, changes in plant community composition, and decreasing vigor of terrestrial vegetation (EPA, 2006). Generally, although plants themselves concentrate relatively little lead, trophic Pb transfer is pervasive. It is unclear whether there is trophic Pb magnification (EPA, 2006). Tree rings show discernible urban lead loadings, even in the absence of stationary sources. In some areas, ecosystem disturbance from urban lead contamination is comparable to that from mining, industrial and Superfund sites (EPA, 2006; USEPA, 2013). Perhaps the decreased vegetation common in urban areas is also related to lead effects.

Plant uptake of lead increases with lower soil pH (Vitousek et al., 1997). Climate change will likely further increase the bioavailability and toxicity of lead by increasing soil acidification and soil mobility (Levin et al., 2020; Vitousek et al., 1997).

Because plants, and especially trees, absorb lead they become repositories for past lead contamination. For instance, recent fires in pristine areas of California attest to remobilization of past lead depositions, with ambient air and soil lead levels rising afterwards (Odigie and Flegal, 2011). Isotopic analysis traces the lead to anthropogenic industrial activities. Climate change will likely increase the frequency and intensity of wildfires.

Phytoremediation and rhizofiltration technologies suggest the potential for cleaning Pb-contaminated soils (Sharma and Dubey, 2005). With phytostabilization, organic materials are added, and the lead is sequestered, diluted, and prevented from becoming bioavailable. However, moving these restoration strategies from the laboratory to contaminated sites has proved disappointing (R, 2017).

3.4. Special populations/communities at risk

Demographics and other characteristics of urban areas are changing, resulting in increased population density and increased socio-economic diversity. Worldwide, urban populations are growing; 68% of the human population is projected to reside in cities by 2050 (*World Urban. Prospect. 2018 Revis.*, 2019). In 2010, 81% of Americans (about 266 million people) lived in urban areas, up from 79% in 2000 (US Census Bureau, 2016). US urban counties are no longer predominately white: in 2000, 51% of urban US counties were predominately non-Hispanic white; in 2012–2014, only 44% were (Parker et al., 2018). The percentage of immigrants in US urban areas has increased, from 20% in 2000 to 22% in 2012–2014 (Morley et al., 2017). About 62.4 million children live in US urban areas, of whom more than 22% live below the poverty line (US Census Bureau, 2016).

Stress in urban areas.—Living in urban areas is stressful: denser housing and traffic, higher noise levels, increased stimulation (including noise and light), less control over the local environment, exposure to crime and physical deprivation, etc. all provoke stress reactions (Burton, 1990). Urban residents have higher serum concentrations of cortisol, the ‘stress hormone’. Both human and animal studies suggest that an over-reactive immune system accompanies stress-associated disorders and may even be causally involved in their pathogenesis (Böbel et al., 2018). Consequently, over time, elevated serum cortisol suppresses the immune system and contributes to other illness.

In addition, some mental illnesses (e.g., anxiety, psychotic, mood, or addictive disorders) may be more prevalent in cities (Peen et al., 2010), with the risk of schizophrenia particularly increased (Pedersen and Mortensen, 2001). But correlation is not causation. People with infirmities (physical or mental) and/or with limited means are likely to gravitate to cities, introducing confounding factors (Tunstall et al., 2015).

Psychosocial stressors and lead affect overlapping biological pathways and impact the body similarly (Cowell and Wright, 2017). Furthermore, increasing evidence indicates stress-induced changes to mothers may prime their children’s rapidly developing physiological systems for disruption by concurrent or subsequent exposure to lead and other environmental chemicals (Cowell and Wright, 2017). Both pre- and post-natal stress exposures increase lead’s neurotoxic effects (Gump et al., 2008; Tamayo y Ortiz et al., 2017). Similarly, there is evidence linking the urban environment to social stress processing (Lederbogen et al., 2011).

Independent of stress, higher childhood BLLs are associated with greater psychopathology across the individual's life course including difficult adult personality traits (Lederbogen et al., 2011). Thus, the stress of urban living may compound the toxicity of lead's urban burden exposure.

Morbidity and mortality in urban areas.—Mortality rates are higher in nonmetropolitan areas (Garcia et al., 2019) but chronic disease rates, including mental illness, are higher in urban areas (Lederbogen et al., 2011). Higher urban air pollution levels increase urban morbidity. Other health categories prevalent in urban areas include allergic, auto-immune, inflammatory, lifestyle and infectious disease (Lederbogen et al., 2011), especially among minorities (Friis and Sellers, 2004). Urban and especially low-income urban populations have exposures to pesticides at home, school and work (Berkowitz et al., 2003; Landrigan et al., 1999). Low income is an additive risk for increased morbidity and higher mortality rates, compounded by limited access to health services.

Compounding vulnerabilities of low income and minority populations.—BLLs have declined over 45 years, but low-income residents, minority populations and those living in older housing remain at greatest risk for higher lead exposures (Gasana and Chamorro, 2002; Jones et al., 2009; Lebrón et al., 2019; Marshall et al., 2020).

Locally undesirable land uses of all sorts, including businesses such as scrapyards and auto repair shops, vacant lots and brownfields, as well as landfills, are more common in low income and minority neighborhoods (Mohai and Saha, 2006). Bare soil tends to have higher lead concentrations than soil with vegetation, and the lead in the soil is more easily accessible to children (Mielke et al., 2011). Bare soil is more common in poor neighborhoods (Egerer et al., 2018).

Vitamin D deficiency is associated with urban living (Mousavi et al., 2019) and is more common in minorities, including blacks (Gutiérrez et al., 2011; Harris, 2006), Asians (Darling et al., 2013; Nimitphong and Holick, 2013) and Hispanics (Taksler et al., 2015). These groups also tend to have higher BLLs (Brown and Longoria, 2010; Cassidy-Bushrow et al., 2017; Hore et al., 2017). In the human body, BLLs and serum vitamin D concentrations exhibit the same seasonality (rising in warmer weather), and are related, although the relationship is not exactly clear (Levin et al., 2020). It is also unclear if innate serum Vitamin D levels differ among races (Gutiérrez et al., 2011).

Obesity is also more prevalent in low-income and minority populations and in U.S. cities (Fitzpatrick et al., 2018). Obesity and poor diet are important risk factors for chronic diseases in the USA (Murray et al., 2013). High fat, low mineral, low or high protein diets increase Pb-absorption in rats (Barltrop and Khoo, 1975) and are associated with higher BLLs in children (Gallicchio et al., 2002).

Both local and national studies have found a disproportionate pollution burden in low income and minority communities (O'Neill et al., 2003). Disadvantaged communities are also likely to be closer to Superfund and Brownfield sites and to landfills (U. S. EPA, n.d.; Office, 1983). Lead may be the most common contaminant at Superfund and Brownfield sites (U. S. EPA, n.d., n.d.).

Co-exposure to stress and Pb during early development may increase the risk of central nervous system dysfunction in both humans and rodents (Sprowles et al., 2018). Independently, low socio-economic status is a known risk factor for various diseases and dysfunctions, effects often ascribed to chronic stress and cascading alterations (Cory-Slechta et al., 2004). Indeed, the compounding and interrelated factors raise questions about whether lead or other assaults studied in isolation from other relevant risk factors can adequately identify neurotoxic hazards (Virgolini et al., 2005).

4. Discussion

What emerged from arraying the data on the urban lead burden across the 3 domains of One Health is the need for comprehensive and integrated remediation strategies. The higher human lead exposures in older urban areas that were evident 40 years ago (Annest et al., 1983; Levin et al., 2008; Pirkle et al., 1994) have been loaded also onto urban wild and domesticated animals and the natural environment (Mordarski et al., 2018), as well. These burdens are not distributed equally through the urban environment. Double-, triple- and even quadruple jeopardies fall on low income, minority, immigrant and other vulnerable populations within cities. Animals, wild and domesticated, reflect the exposures of their human environmental partners.

Lead is extremely useful. Either elemental or in alloys, lead now has thousands of commercial applications in electronics, hardware, paints and pigments, many types of glass, ceramic glazes and coatings, cable sheaths, machinery and manufacturing, light industry, radiation protection, housing and other construction, post and telecommunications, metallurgy, chemical production, transportation (rail, automobile and aviation), construction, weapons, aerospace, oil, plastics and petrochemicals, plumbing, pesticides, hair products, etc.

Lead is also toxic to almost every species investigated. Across taxa, most urban lead exposures are sub-lethal. Evidence suggests that with BLLs < 10 ug/dl no single source predominates, but even on the subclinical level, adverse effects of lead exposure cascade through human and animal species (Morley et al., 2017). Urban stress itself engenders psychopathological behaviors in both humans and animals (McClelland et al., 2019; Reuben et al., 2019). Urban wild animals, especially the typically adaptable generalist species that populate US cities, exhibit similar behavioral and population-level adaptations to urban living, including reduced wariness, and increased intraspecific aggression and population densities (Parker and Nilon, 2012). Urban noise increases territoriality in urban male Song Sparrows (Davies et al., 2018), and territoriality is stronger in urban common grey squirrels (Merrick et al., 2016). These behaviors mimic those described in the human health literature concerning aggression levels, ‘problem’ and ‘risky’ behaviors in children and adolescents, etc. associated with lead (Reuben et al., 2019; Reyes, 2015).

Some of these traits may increase survival in urban environments (Evans et al., 2010). But in circumstances where peak performance is necessary for survival, there is some evidence that lead exposures may soften that edge (Cole et al., 2009). In addition, urban residence increases the seasonality of lead, viz. higher blood lead levels during the summer, probably through increased exposures, especially to soil (Levin et al., 2020).

Pollution may be the biggest anthropogenic contribution to the natural world. Lead's ubiquity necessitates integrated responses across all levels of governance (federal, state and local), as well as participation by all stakeholders, that is, all of us. As urban populations and exposures are predicted to increase, integrated assessments and intervention strategies are needed to address the One Health impacts of urban lead exposure across species lines. Below, we apply the dual lenses of One Health and environmental justice to primary prevention to review existing federal standards and to suggest changes to incorporate the current state of the science and redress the equity issues that have created disparities in lead exposures.

4.1. Review of existing federal lead standards.

Federal standards are risk management tools and a compromise between health protection, technical capacity and other social priorities, viz. the costs and benefits of intervention vs. failing to respond. The federal standards for lead contamination and exposure have not kept up with the science on either health effects or technical improvements available to control or eliminate lead exposure. In addition, some of the standards have critical implementation obstacles.

An instructive case is EPA's regulation of lead in US public drinking water. As described above in section 2.3, the AL is not health based nor is the 20 ug/l standard for school drinking water (EPA, 2019). EPA also acknowledges implementation limitations with the LCR, including that the rule is complicated and difficult to enforce (EPA, 2016).

EPA has recently acknowledged fundamental shortcomings in their water-exposure modeling: it does not recognize the reductions in lead exposure sources that occurred during the last half century. As the legacy of leaded gasoline receded, drinking water is now a more dominant lead exposure. Indeed, under current realistic water use conditions in a public water system that is not violating the LCR, drinking water can supply up to 80% of US children's daily lead exposure (Stanek et al., 2020). EPA's lead model (SHEDS-IEUBK) also did not consider potential exposures in water systems that have elevated WLLs – again, not necessarily exceeding the LCR. (Zartarian et al., 2017) In addition, EPA's lead model continues to define high risk areas as those where a child has a 5% risk of a BLL ≥ 5μg/dL. Because there is no known safe BLL for children and because in 2016 1.3% of US children had BLLs ≥ 5μg/dL, these parameters are outdated (Control, 2019). The model as currently configured could allow lead exposures found at some Superfund sites.

Another deficit in EPA's modeling of water exposure is the application of a single WLL exposure coefficient. In fact, WLLs are characterized by high spatial and temporal variability, both across the water system and within a residence; the variability follows both predictable (LSL presence, season, sample protocol, etc) and random trajectories

(Schock et al., 1988; Schock and Lemieux, 2010). Indeed, inherent variability was part of EPA's justification for promulgating an AL over the statutorily required Maximum Contaminant Level. These systemic biases blunt EPA's ability to characterize compliance with its drinking water rules and especially obscure distributional distortions and burdens in service populations.

Finally, EPA has not issued a Health Advisory Level (concentration of drinking water contamination at which adverse health effects are not anticipated to occur over specific exposure durations, including 1 day, 10 day and lifetime) for lead. It is unclear whether EPA intends by the omission that *no* level is safe or that *any* level is safe (EPA, 2018).

The Occupational Safety and Hygiene Administration (OSHA) worker standards for lead exposure are a more egregious example of an outdated federal standard. These standards were set in 1978 and require worker protections such as removal from lead exposure that commence at BLLs of 40–60 ug/dl. They do not apply to industries with fewer than 10 employees or to contractors. Thus, many construction and remodeling, scrap metal recycling, auto body repair and other small industries with substantial lead exposure may be exempt from these standards. Again, these workers tend to be low-income and minority with a high percentage of immigrants, and the businesses are disproportionately located in urban EJ neighborhoods. The benefits of lowering those standards were estimated at $392 million (2014US$) per year just for the 10,000 or so U.S. workers with high occupational lead exposures who are reported under federal surveillance data (Levin, 2016). It is likely that the number of occupationally lead-exposed workers protected under OSHA is underestimated by 50% or more, and likely many times that (National Institute of Environmental Health Sciences, 2012).

These federal standards are especially problematic, but all federal lead standards are associated with some residual health risks. For example, Braun et al 2020 showed that the difference between EPA's current and potential new dust-lead standards is associated with a 26% reduction in lead exposure while failure to adopt the new standards results in doubling the risk of exceeding the Centers for Disease Control and Prevention's (CDC's) target maximum exposure of 5 ug/dl (Braun et al., 2020).

Policy recommendation: Under the guidance of EPA and CDC, a task force of federal agencies with responsibilities related to lead exposure or environmental contamination should convene a short-term effort to review all of the standards for lead. The panel should also include representatives of the related Federal Advisory Committees, recognized technical experts and members of the affected populations. The goal should be to standardize the approaches based upon best current health research and exposure science. Responsibility for updating the standards will remain with each agency. However, the results of the review should be presented to Congress within 90 days of its completion.

4.2 Policies to reduce urban lead levels across species: Using an Environmental Justice framework for primary prevention

Economic, cultural and social factors determine exposure to environmental hazards with ensuing health outcomes. Despite the 10-fold reduction in US pediatric BLLs since the

1970s, there remain too many urban areas where children continue to be exposed to toxic lead levels (Raymond and Brown, 2017); animals are poisoned there, too.

The exposures follow predictable patterns. Environmental Justice (EJ) seeks to redress the imbalance of exposures and damages (EPA, 2017; Whitehead, 2015). We apply the dual lenses of One Health and EJ to primary prevention to review federal activities and suggest changes to incorporate the current state of the science and redress the equity issues that have created disparities in lead exposures.

Evidence-based research has shown the efficacy of some US public policies to prevent or reduce these exposures (Brown and Falk, 2016; Wheeler and Brown, 2013) (Figures 3 and 4). Likely the most important federal action was reducing lead in gasoline, beginning in the early 1970's, accelerating in 1985 until a final ban in 1996; the exemption for off-road uses has remained (Needleman, 2000). These interventions are also demonstrably cost-effective. The Pew Charitable Trust estimated that preventing an annual birth cohort of children from ever experiencing a measurable BLL or higher produces benefits of about $84 billion over the lifetime of these children related to decreased cognitive damage and increased earnings (Morley et al., 2017). Other estimates of the benefits of lead abatement found that each dollar invested in lead paint hazard control results in a return of $17-$221 or a net savings of $181–269 billion (Gould, 2009). An earlier estimate of the benefits of corrosion control alone in drinking water systems found that the benefits greatly exceeded the costs (Levin, 1987). This research has reinforced that waiting for children or sentinel animals to reach some arbitrary established exposure criterion before instituting lead hazard control or elimination measures has severe costs.

A 'lead safe' environment does not mean that it is 'lead free'. Primary prevention includes resources for ongoing maintenance of lead safety, such as repairing damaged LBP, recapping lead-contaminated soil that was covered, maintaining drinking water corrosion control until all lead plumbing components have been replaced, etc. Data indicate that the expected life of these remediation strategies is, on average, 10 years (Wilson et al., 2006). A lead safe environment does require the existence and conformance with appropriate federal exposure standards and mitigation plans.

Lead monitoring, surveillance, and reporting.—Since at least the 1980's, the US has maintained a collaborative effort among the federal agencies related to lead use, regulation, health research and exposure assessment. This was also coordinated with BLL and environmental monitoring, surveillance, and reporting on the federal, state and local levels.

Monitoring of environmental lead loadings is necessary for assessing the progress of governmental policies and developing remediation plans at all levels, local, state and national. Monitoring can also enable policy makers and scientists to recognize problems and anticipate obstacles. Resources devoted to environmental monitoring and reporting have declined recently. EPA's environmental monitoring network can be enhanced using existing networks; for instance, if the form of the air-lead monitoring were altered from TSP (total suspended particulates) to ultrafine particles (PM 2.5), the fine particle speciation network could be used to augment the existing air-lead monitoring network.

Blood lead testing in children serves two distinct purposes. Because childhood lead poisoning typically has no distinct symptoms, blood lead testing allows the identification of unrecognized elevated BLLs in individual children. BLLs can also be used for surveillance of lead exposures at the community (local, state or province, or country) level. Ongoing analysis of surveillance data can identify community level risks permitting community-based interventions, again at the local, state or province, or national level.

Collaboration between federal, state, and local agencies, and the health care community has also weakened recently. CDC no longer routinely receives BLL surveillance data from all states or the most recent biomonitoring data. Some lead exposure sources have been eliminated but others have only been controlled and new sources of lead continue to emerge. In the absence of individual and environmental surveillance and monitoring, we cannot be confident that past successes in lead exposure reductions have been maintained.

This is evident from the apparent increase in US BLLs recently (Control, 2019). Lead consumption and production has also risen, both globally and in the US, over the past 20–30 years. In the US, this relates to higher demand for lead in starting-lighting-ignition (SLI) lead-acid storage batteries and in demand for lead in non-SLI battery applications (Group, n.d.; Survey, n.d.; US Geological Survey, n.d.). Lead waste from US production has also increased over the period 2007-2018 (EPA, n.d.).

Policy recommendation: Under the joint guidance of EPA and CDC, a government-wide interagency task force should rekindle lead monitoring, surveillance, and reporting collaboration across all levels of US government. This group should also evaluate current trends in US lead exposures, environmental contamination levels and loadings, and current and projected production/consumption to ensure that past successes are not lost. These data should be made available to the public and to Congress within one year of their collection.

Investment in public infrastructure.—Underinvestment in US public infrastructure on all levels (municipal, state and federal) has accelerated in the past half century (Engineers, 2017; McNichol, 2019). Investment patterns in public and especially environmental infrastructure mirror the structural inequities evident throughout US public policy. Low income and minority populations have received a disproportionately low fraction of the $120 billion over the last half century that the EPA has devoted to constructing wastewater and drinking water infrastructure (Coursen, 2020a). This is manifest: 2 million Americans, predominantly low-income and minority populations, lack indoor plumbing; 12% of U.S. households struggle to pay water bills; over 9 million homes, often in the poorest US cities, receive water through lead pipes; and over 18 million people receive drinking water that exceeds EPA standards(Coursen, 2020b;Olson and Pullen Fedinick, 2016).

Policy Recommendation: EPA and the American Society of Civil Engineers (ASCE) should assess the EJ issues related to the condition of US environmental infrastructure, especially wastewater and drinking water. This effort should be correlated with the ASCE's Report Card on America's Infrastructure, updated every 4 years.

4.3 Policies to redress disproportionate urban lead exposures

Residential LBP hazards.—LBP and LBP-contaminated house dust are the most highly concentrated sources of lead available to US children with few exceptions (Laidlaw et al., 2005). In many cities, old poorly maintained housing is concentrated in well identified neighborhoods.

Pew estimated that eradicating lead paint hazards from older homes of children from low-income families would provide $3.5 billion in future benefits, or approximately $1.39 per dollar invested, and protect more than 311,000 children (Morley et al., 2017). The cost of controlling lead paint hazards is estimated at about $2.5 billion for the 2018 cohort of children.

Furthermore, ensuring contractor compliance with lead-safe work practices would protect about 211,000 children born in 2018 and provide future benefits of $4.5 billion, or about $3.10 per dollar spent (Morley et al., 2017). That effort is estimated to cost about $1.4 billion. In addition, lead hazards from other buildings that children frequent, such as day care centers, schools, etc should be removed.

Policy Recommendation: In housing with two or more residential units, all units in a building where a child with an elevated BLL resides should be inspected, cited for identified lead hazards and made lead safe.

Policy Recommendation: At property transfer, sale or rental of a housing unit, dust wipe sampling should be required to demonstrate that the unit continues to be lead safe. Local ordinances that protect families with young children from housing discrimination are necessary to ensure that families are not barred from rental units. Successful models for these strategies exist in Washington DC, Maryland and other communities.

Policy Recommendation: EPA's Renovation Repair and Painting (RRP) Rule should be implemented with strong enforcement measures. Local governments should require licensing and certification of contactors as a prerequisite for permitting home renovation projects.

Lead in residential soil.—LBP contamination of soil is common in neighborhoods with older housing, even when the properties are well maintained. Fortunately, existing policies have reduced both soil-lead and children's BLLs, but they remain too high in some communities (H. Mielke et al., 2019).

Policy Recommendation: Include soil sampling as part of a comprehensive inspection whenever a home is inspected for lead. Remedies include capping or covering contaminated soil or removing and replacing it with clean soil, and practices such as placing door mats at entry ways and planting bushes near foundations. Soil for gardening should be tested as well as using containers to grow vegetables in areas with older housing and/or where contaminated soil is found. Soil should be tested annually.

Policy Recommendation: For municipalities that permit urban agriculture, soil for gardening or where animals reside should be tested and demonstrate safe levels.

Lead in soil at active and former industrial sites, vacant lots, etc.—Secondary smelters are often sited in older residential areas. These smelters recycle lead acid batteries. Lead is also among the most common contaminants at Superfund and Brownfield sites. Vacant urban areas accommodate many wild animals, as well as providing play space for children.

Active Sites

Policy Recommendation: Scrap yards, metal recyclers and other industries that use or emit lead should be required to register with the local government. Practices to reduce worker exposure and 'take home' occupational exposures and to control lead emissions should be implemented. Regulations and enforcement strategies coupled with monitoring of air and worker BLLs help ensure consistent protections.

Policy Recommendation: Cradle to cradle tracking of used lead acid batteries will prevent batteries from being recycled improperly in the US or exported abroad for recycling. Manufacturers must be accountable for responsible recycling of their products at the end of their useful life.

Former Sites

Policy Recommendation: When metal-processing industrial facilities close, planning should be required to ensure that the site is safe for future use. As former lead sites are identified they should be subject to the same identification and abatement. Remediation of lead contamination is very site-specific and regardless of size, every site requires comprehensive testing of environmental media using validated sampling and analytic techniques. Oversight by trained risk assessors and remediation experts is necessary.

Policy Recommendation: Lead remains useful; formerly worthless wastes may now be valuable. New technologies that protect human health and preserve the environment are essential as companies try to re-mill old mining/smelting wastes or attempt landfill-mining to harvest metals from dumps.

Lead in waste streams.—Lead's ubiquity means it is widely present in both residential and commercial waste streams. Construction debris containing LBP is exempt from federal regulation.

Policy Recommendation: The EPA and state departments of environmental protection should monitor waste handling to prevent illegal dumping of hazardous waste containing lead.

Policy recommendation: Cradle to cradle tracking of used lead acid batteries will prevent batteries from being recycled improperly in the US or exported abroad for recycling. Manufacturers must be accountable for responsible recycling of their products at the end of their useful life.

Lead in drinking water.—Lead most often contaminates drinking water by leaching from the plumbing components of the public and especially residential plumbing systems. Lead pipes are the greatest contributors to high WLLs. No single reliable sampling method exists to predict WLL risk in a particular house. Pets can be poisoned as well as humans.

For large and medium size water systems, corrosion control measures to reduce lead leaching into water from plumbing fixtures are demonstrated, cost-effective ways to reduce WLLs (Levin, 1987). However, water treatment modifications, including disinfection practices, can substantially undermine lead corrosion control, and consistent management oversight is necessary.

In addition, the Pew Trust estimated that removing lead service lines from the homes of children born in 2018 would protect more than 350,000 children and yield $2.7 billion in future benefits, or about $1.33 per dollar invested. The cost of removing the pipes is estimated at about $2 billion (Morley et al., 2017).

Finally, EPA's LCR does not establish a health-based standard, does not provide guidance for testing the homes of children with high BLLs, and can be easily manipulated or 'gamed' (Olson and Pullen Fedinick, 2016). Further, EPA's enforcement data show both poor record keeping and poor compliance with US drinking water standards. EPA's Inspector General found that only 9–23% of reporting and monitoring violations of drinking water standards are reported to EPA by the state primacy agencies, and only 24% of health-based violations are reported(EPA, 2004). Even with such under-reporting of non-compliance, in 2015 EPA's enforcement data base documented that over 18 million people were served by 5,363 community water systems that violated the LCR (Olson and Pullen Fedinick, 2016).

Policy recommendation: EPA should work with water authorities and the states to identify implementation barriers to improving compliance with the LCR and to develop and implement an accurate water surveillance and compliance system. EPA should consider a single enforceable drinking water standard such as a Maximum Contamination Level for lead. These activities and the results should be reported to the public and Congress every 2 years.

Policy Recommendation: EPA should work with water authorities and the states to develop enforceable and actionable plans to address lead contamination of school drinking water, beginning with pre-schools and kindergartens, then middle schools and high schools.

Policy Recommendation: Water sampling at the consumer's tap should be conducted by water suppliers whenever there are changes in water treatment, quality or source. If water lead levels are elevated, alternative water sources should be provided immediately and continued until WLLs are reduced.

Policy Recommendation: Water authorities should be required to map the presence of lead service lines in their water district and develop plans to replace these lines in a timely manner.

Policy Recommendation: Partial replacement of lead service lines should be banned when a lead service line is replaced.

Policy Recommendation: Corrosion control treatment to reduce lead from leaching from interior water pipes and fixtures should only be discontinued after extensive testing of a community's tap water.

Policy Recommendation: The drinking water of every child with an elevated BLL should be tested under circumstances that will show evidence of the risk of high WLLs.

Lead in consumer and food-related products including pet food and products.—Lead continues to be used in numerous everyday household products and to contaminate the food chain (Flegal and Odigie, 2020; Fund, n.d.), including food and products for pets (Hankin et al., 1975; Riggs et al., 2002; "Walgreens Study, 2014," 2014), principally through imported goods.

Policy Recommendation: The federal government, through participation in the Codex committee, should encourage expedited reduction of international limits on lead in foods and consumer goods, particularly those that appeal to young children, babies or pets. Where local surveillance data indicate lead exposures from identifiable sources, state and local agencies should target at-risk neighborhoods.

Policy recommendation: Under guidance from EPA and CPSC, using formal and informal mechanisms, the US should explore reducing all uses of lead.

Urban agriculture.—The growth of urban agriculture in the US has the potential to increase food security, improve nutrition, advance sustainability and foster community cohesion. But lead-contamination of urban soils is common, resulting in high exposures in some backyard chickens and possibly produce. Surface contamination of produce as well as lead-contaminated compost has also been reported.

Policy recommendation: Testing soil is preferable but may be cost prohibitive. EPA recommends siting gardens away from old buildings and roads, adding uncontaminated organic matter/compost and topsoil, and covering with mulch (EPA, 2011). Produce should be washed thoroughly before consumption. Produce less likely to accumulate lead includes tomatoes, eggplants, tubers like potatoes, legumes, tree fruits and nuts, and berries.

Leaded fuel for off-road vehicles.—EPA exempted off-road vehicles, including piston-aircraft, small boat motors, lawn and farm equipment, etc. from the ban on leaded fuel. Some of these are known exposure sources for local residents and all constitute exposures to users, workers and the environment. The Pew Trust estimated that eliminating lead from airplane fuel would protect more than 226,000 children born in 2018 who live near airports, generate $262 million in future benefits, and remove roughly 450 tons of lead from the environment every year (Morley et al., 2017).

Policy recommendation: EPA should evaluate the risks and costs of leaded fuel for off-road equipment and vehicles, especially av gas, specifically to consider banning the use of tetraethyl lead as a petro-fuel additive. Contamination at marinas should also be considered.

Alternatives to lead.—Lead has numerous uses that can contribute acute and chronic lead exposures to local fauna and humans and that are easily substituted by other materials. These include lead roof flashing, which can poison rodents and birds; lead tire weights, which fall off and are pulverized and commonly eaten by birds; paint used on playing fields, which is an exposure source both for humans who play there and for the animals and plants that live there; leaded ammunition, fishing and other sporting goods, which can poison wildlife, participants and their families; outdoor equipment in public parks; etc.

Policy recommendation: With the US Fish and Wildlife Service, EPA and the Consumer Product Safety Commission should review the suitability of substitutes for lead in applications that have been documented to expose wildlife or humans, including building materials, sporting goods, etc.

4.4. Limitations

Among Medicaid provisions for low income Americans, all children under 6 must be tested for lead exposure. Many states do not comply with this requirement (Gasana and Chamorro, 2002), and possibly half of at-risk children may not be screened for lead (Neuwirth, 2018). Thus, lead exposures in US children may be underestimated significantly.

BLLs in humans have been decreasing since the elimination of leaded gasoline over 30 years ago. There are no longitudinal data assessing trends in animal or plant exposures.

Finally, while this analysis rests upon decades of research, this One Health assessment is a new cross-species integrated approach. The strengths of the commonalities remain to be tested.

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Highlights

- High lead burdens in urban areas expose humans, domesticated and wild animals, and plants.
- Higher exposures fall differentially on different populations even within urban areas.
- Urban lead repositories include soil, buildings, dust and even trees.
- Urban stress and lead exposure cause recognizable aggressive behaviors in both humans and animals.
- Evidence-based research shows cost-effective policies exist to reduce urban lead burdens.

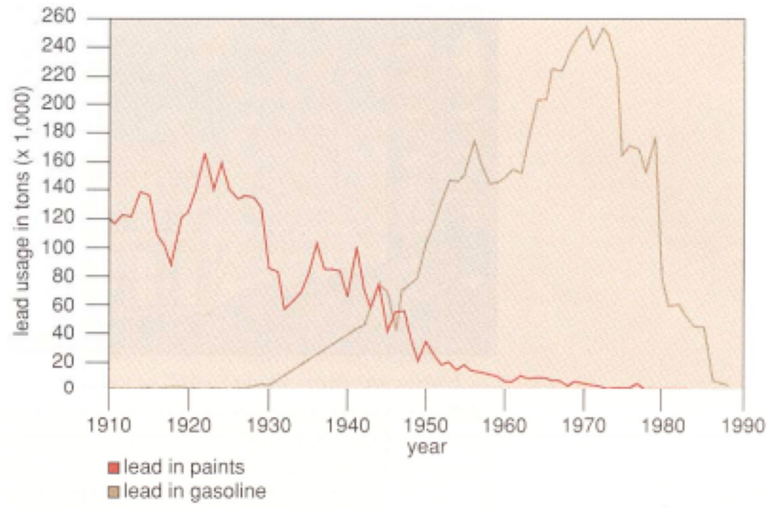

Figure 1:
Total US usage of lead in paint and gasoline, 1910–1990. Source: Mielke HW. 1999. Lead in the inner cities. American Scientist 87 (Jan-Feb): 62–73.

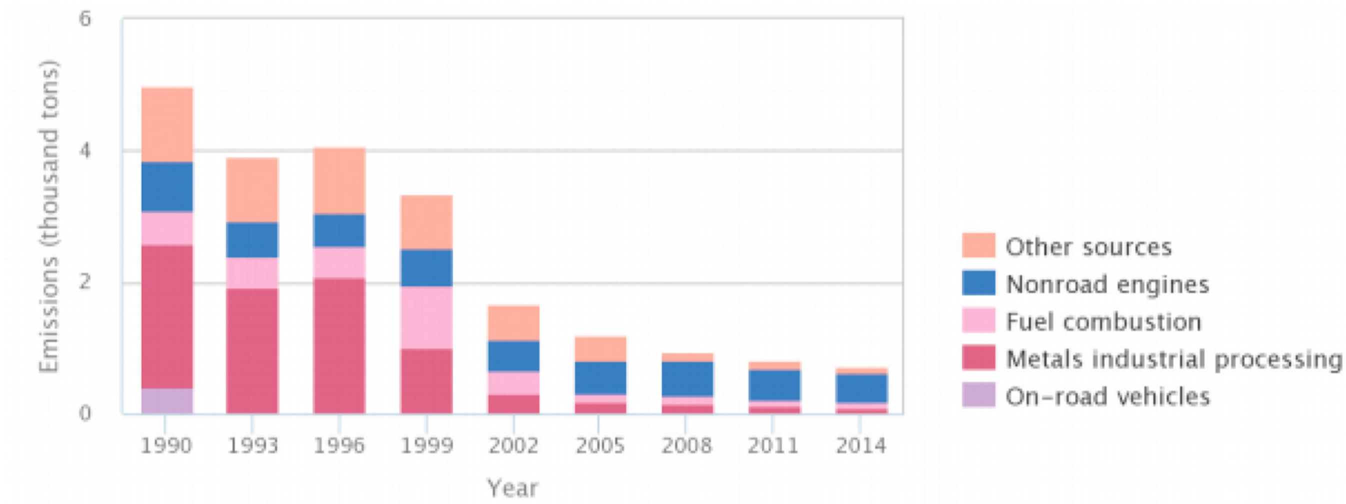

Figure 2.
Anthropogenic US lead emissions by source, 1990–2014 Source: EPA 2018 (U.S. EPA. 2018. Data from the 2014 National Emissions Inventory, Version 2. Accessed 8/12/20. https://www.epa.gov/air-emissions-inventories/2014-national-emissions-inventory-nei-data.)

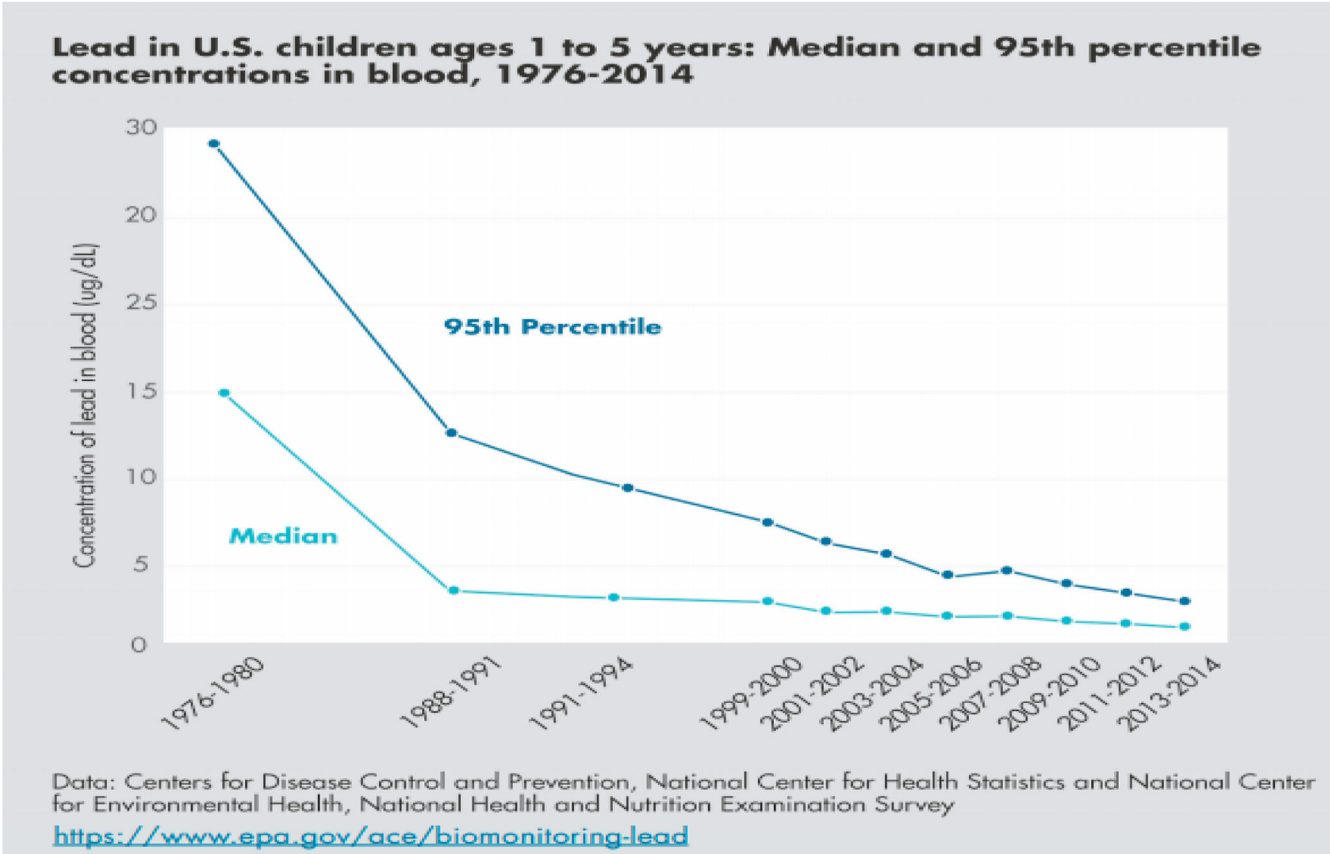

Figure 3.

Average US pediatric blood lead levels 1976–2014. https://ptfceh.niehs.nih.gov/features/assets/files/key_federal_programs_to_reduce_childhood_lead_exposures_and_eliminate_associated_health_impactspresidents_508.pdf

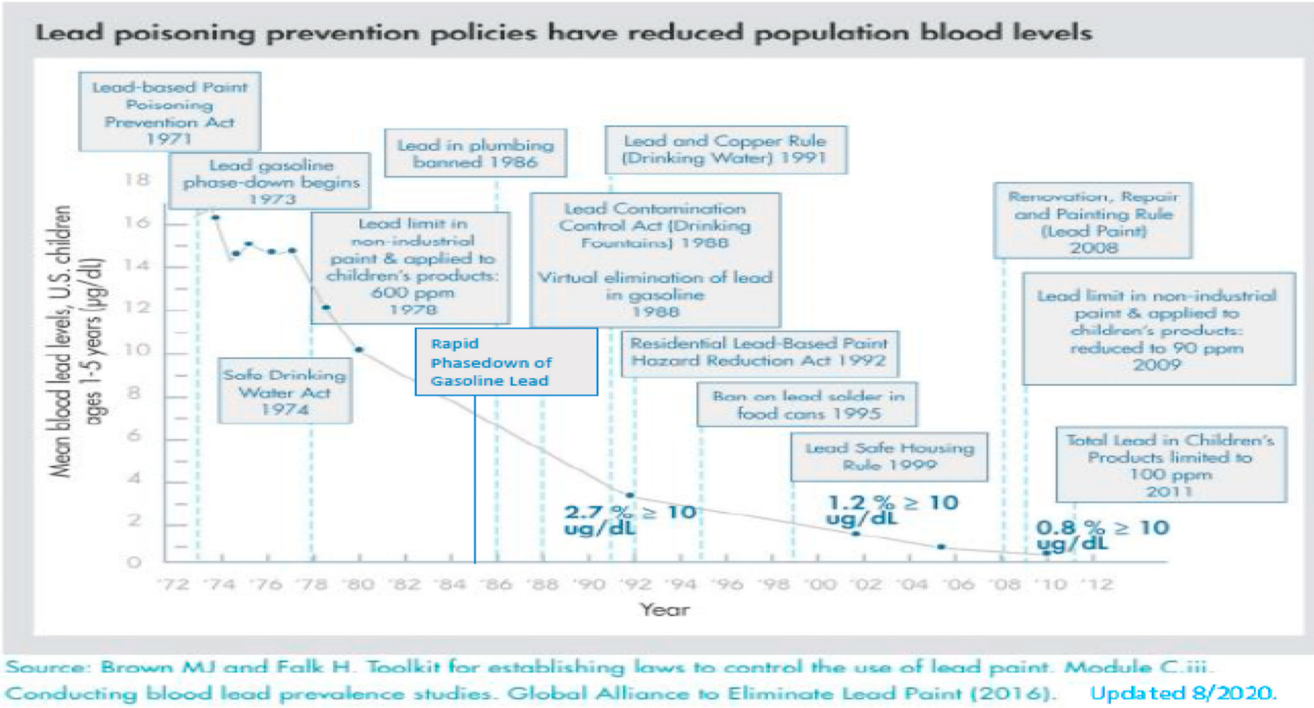

Figure 4.

Average US pediatric blood lead levels 1976–2014, overlaid with US federal policies and regulations. https://ptfceh.niehs.nih.gov/features/assets/files/key_federal_programs_to_reduce_childhood_lead_exposures_and_eliminate_associated_health_impactspresidents_508.pdf