Lead (Pb) Contamination in Low- and Middle-Income Countries: Exposures, Outcomes and Mitigation

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Cover image: informal used lead acid battery recycling, Dhaka, Bangladesh

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Statement of Originality

This work has not previously been submitted for a degree or diploma in any university. To the best of my knowledge and belief, the thesis contains no material previously published or written by another person except where due reference is made in the thesis itself.

Bret Ericson 30 June 2019

Abstract

Human lead exposure contributes to a number of adverse health and social outcomes. The most significant historical source of exposure was the addition to tetraethyl lead (TEL) to petrol throughout the 20th century, which resulted in widespread contamination of human and natural environments. By the early 21st century nearly all countries had taken steps to phase out the use of TEL. Subsequent biological assessments of humans have universally shown significant and rapid declines in lead concentrations, typically measured in blood. In high income countries, the declines were particularly substantial. In the United States, for example, mean blood lead levels (BLLs) are currently < 1 µg/dL compared to > 15 µg/dL before the 1976 phase out. In contrast BLLs in low- and middle-income countries (LMICs) have remained elevated, with national mean BLLs above 5 µg/dL in multiple locations. Despite this, little is known about the continued nature and extent of lead exposure in LMICs.

This thesis sought to characterize sources of human lead exposure in LMICs, quantify their impact, and propose methods of mitigation. It is comprised of 12 studies arranged in three chapters along these themes. In the first chapter, sources of human lead exposure in LMICs were evaluated through a systematic literature review and *in situ* assessment. In the second chapter, the attributable disease burden of some of those sources was quantified using common public health metrics, primarily Disability Adjusted Life Years (DALYs). Finally, in the third chapter, mitigation case studies were executed in three different countries and evaluated for their efficacy. The thesis contends that lead exposure in LMICs is primarily industrial in nature and results in a larger disease burden than is currently estimated. In addition it argues that cost effective measures exist to mitigate some of the more significant sources of exposure.

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Thesis Composition

The thesis is comprised of twelve distinct studies organized into three chapters and bookended with an introduction and conclusion. Eleven of the 12 studies have been published in the peer-reviewed literature. Paper 1 has not yet been published but has been submitted to Environmental Research for review.

Chapter 1: Introduction provides context for the thesis. Background information on the nature and extent of the issue is presented. The three central chapters examine the scale and severity of lead exposure in low- and middle-income countries (LMICs) and investigate methods of mitigating some of those exposures. The first of the original research chapters, *Chapter 2: Exposures*, identifies and quantifies the relative impact of key sources of lead exposure in LMICs. *Chapter 3: Outcomes* calculates the burden of disease attributable to some of the exposures described in Chapter 2. Finally, *Chapter 4: Mitigation* presents a series of case studies to describe methods employed to mitigate exposures in LMICs. Finally *Chapter 5: Synthesis of Research* integrates the main findings of the thesis and discusses their implications for research and policy. Altogether, this thesis endeavours to present a coherent approach of assessing, quantifying and mitigating lead exposure in LMICs.

Much of the research described was carried out in tandem with work for the USA-based non-governmental organization Pure Earth where I serve as Chief Operating Officer. This position has afforded me a unique opportunity to conduct primary research in multiple countries jointly with our staff and partners, who have been included where appropriate as co-authors in research outcomes. My role has also enabled me to execute several risk mitigation projects, some of which are documented in this thesis.

Chapter 2: Exposures

Ericson, B. (45 %), Nash, E. (35 %) Sinitsky, J. (5 %), Masek, V (5 %) Ferraro, G (5%) & Taylor, M.P. (5 %) (2018). Sources of lead exposure and resulting blood lead levels in low- and middle-income countries: a systematic review and meta-analysis. Submitted to Environmental Research.

- The study was conceived, designed and executed by BE. BE conducted the literature review, data extraction and synthesis, and served as the primary author of the manuscript. Nash repeated the literature review, extracted data from a randomly selected subset of articles, and contributed to the writing of the manuscript. Taylor contributed to the writing of the manuscript and provided guidance on the approach.

Dowling, R. (30 %), Caravanos, J. (10 %), Grigsby, P. (15 %), Rivera, A. (5 %), **Ericson, B. (10 %),** Amoyaw-Osei, Y. (5 %), Akuffo, B. (20 %) & Fuller, R. (5 %) (2016). Estimating the prevalence of toxic waste sites in low-and middle-income countries. *Annals of Global Health*, *82*(5), 700-710

- The study was led by Dowling who directed all major activities. Grigsby conducted the statistical analysis. Akuffo conducted the field work. Caravanos and BE assisted in the design, execution, and authorship of the project. Fuller and Amoyaw-Osei assisted in the project direction.

Ericson, B. (40 %), Hariojati, N. (25 %), Susilorini, B. (5 %), Crampe, L. F. (5 %), Fuller, R. (5 %), Taylor, M. P. (10 %), & Caravanos, J. (10 %) (2019). Assessment of the prevalence of lead-based paint exposure risk in Jakarta, Indonesia. *Science of The Total Environment*, *657*, 1382-1388.

- BE designed the project, directed data collection, conducted statistical analysis, and authored this study. Hariojati collected data. Caravanos provided guidance on sample collection. Taylor assisted with the writing of the manuscript. Susilorini, Crampe, and Fuller assisted with project direction.

Ericson, B. (60 %), Otieno, V.O. (15 %), Nganga, C. (5 %), St. Fort, J. (5 %) and Taylor, M.P. (15 %) (2019). Assessment of the Presence of Soil Lead Contamination Near a Former Lead Smelter in Mombasa, Kenya, *Journal of Health and Pollution*, *9(21), 190307.*

- BE conducted the fieldwork, the statistical analysis and modelling, and wrote the manuscript. Otieno assisted with fieldwork. Taylor assisted with the writing of the manuscript. St. Fort and Nganga assisted with the direction of the project.

Bose-O'Reilly (40 %), S., Yabe, J. (20 %), Makumba, J. (10 %), Schutzmeier, P. (10 %), **Ericson, B. (10 %),** & Caravanos, J. (10 %) (2018). Lead intoxicated children in Kabwe, Zambia. *Environmental Research*, *165*, 420-424.

- Bose-O'Reilly lead the overall authoring of the project assisted by Schutzmeir. Primary data collection was conducted by BE, Makumba, Yabe, and Caravanos.

Chapter 3: Outcomes

Ericson, B. (65 %), Landrigan, P. (20 %), Taylor, M. P. (2.5 %), Frostad, J. (5 %), Caravanos, J. (2.5 %), Keith, J. (2.5 %), & Fuller, R. (2.5 %) (2016). The global burden of lead toxicity attributable to informal used lead-acid battery sites. *Annals of Global Health*, *82*(5), 686-699.

- BE designed the study, developed the statistical model, and wrote the manuscript. Landrigan assisted with authorship of the manuscript. Frostad assisted with the calculation of and provided data for Disability Adjusted Life Years. Taylor, Caravanos, Keith, and Fuller contributed to the conceptual development of the study.

Estrada-Sánchez, D. (40 %), **Ericson, B. (40 %),** Juárez-Pérez, C. A. (2.5%), Aguilar-Madrid, G. (2.5 %), Hernández, L. (5 %), Gualtero, S. (5 %), & Caravanos, J. (5 %) (2017). Pérdida de coeficiente intelectual en hijos de alfareros mexicanos. *Revista Médica del Instituto Mexicano del Seguro Social*, *55*(3), 292-299.

- Estrada-Sánchez and BE jointly carried out the fieldwork, conducted the modelling, and wrote the manuscript. BE translated the manuscript, originally published in Spanish, to English for inclusion in this thesis. Juárez-Pérez, Aguilar-Madrid, Gualtero and Hernández assisted with writing. Caravanos assisted with modelling.

Ericson, B. (50 %), Dowling, R. (12.5 %), Dey, S. (2.5 %), Caravanos, J. (2.5 %), Mishra, N. (2.5 %), Fisher, S. (5 %), Ramirez, M. (5 %), Sharma, P. (2.5 %), McCartor, A. (2.5 %), Guin, P. (2.5 %), Fuller, R. (2.5 %) & Taylor, M. P. (10 %) (2018). A meta-analysis of blood lead levels in India and the attributable burden of disease. *Environment International*, *121*, 461-470.

- BE conducted the literature review, data extraction and synthesis, and wrote the manuscript. Dowling assisted with writing the manuscript, conducted the literature review and data extraction. Taylor assisted with authorship of the manuscript. Ramirez and Fisher conducted the literature review and data extraction. Dey, Mishra, Sharma, McCartor, Guin, and Fuller conceived of the study and assisted with authoring of the manuscript. Caravanos assisted with study design and authoring of the manuscript.

Caravanos, J. (15 %), Carrelli, J. (50 %), Dowling, R. (15 %), Pavilonis, B. (5 %), **Ericson, B. (10 %),** & Fuller, R. (5 %) (2016). Burden of disease resulting from lead exposure at toxic waste sites in Argentina, Mexico and Uruguay. *Environmental Health*, *15*(1), 72.

- Carrelli designed the study, conducted the statistical analysis, and authored the manuscript. Caravanos, Dowling, Pavilonis, and BE assisted with the study design and authoring of the manuscript. Fuller assisted with overall direction.

Chapter 4: Mitigation

Ericson, B. (55 %), Duong, T. T. (2.5 %), Keith, J. (2.5 %), Nguyen, T. C. (2.5 %), Havens, D. (2.5 %), Daniell, W. (5 %), Karr, C.J. (5 %), Ngoc Hai, D. (2.5 %), Van Tung, L. (2.5 %), Thi Nhi Ha, T. (2.5 %), Wilson, B. (2.5 %), Hanrahan, D. (2.5 %), Croteau, G. (2.5 %), Taylor, M.P. (10 %), (2018). Improving human health outcomes with a low-cost intervention to reduce exposures from lead acid battery recycling: Dong Mai, Vietnam. *Environmental Research*, *161*, 181-187.

- BE led the design and execution of the intervention, conducted statistical analysis, and wrote the manuscript. Daniell, Karr, and Taylor assisted with the authoring of the manuscript. The remaining authors played substantive roles in the execution of the project.

Heacock, M. (40 %), Trottier, B., Adhikary, S., Asante, K. A., Basu, N., Brune, M. N., … **Ericson, B (5 %)** ... & Chen, A. (2018). Prevention-intervention strategies to reduce exposure to e-waste. *Reviews on Environmental Health*, *33*(2), 219-228.

- Heacock wrote the manuscript. The remaining authors (n=30) contributed to either the primary research or conceptual design. Of the five projects described as case studies, two were managed by BE.

Ericson, B. (70 %), Caravanos, J. (5 %), Depratt, C. (2.5 %), Santos, C. (5 %), Cabral, M. G. (5 %), Fuller, R. (2.5 %), & Taylor, M. P. (10 %) (2018). Cost Effectiveness of Environmental Lead Risk Mitigation in Low-and Middle-Income Countries. *GeoHealth*, 2(2), 87-101.

- BE conceived of the research, carried out fieldwork, managed the intervention, conducted statistical analysis, and authored the manuscript. Caravanos, Depratt, Santos and Cabral carried out fieldwork. Taylor assisted with authoring the manuscript. Fuller assisted with overall direction of the project.

Acronyms

BLLs – Blood lead levels
CDC – United States Centres for Disease Control and Prevention
CEA – Country Environmental Assessment
DALY – Disability adjusted life year
GAELP – Global Alliance to Eliminate Lead Paint
GBD – Global Burden of Disease
GEF – Global Environment Facility
IHME – Institute for Health Metrics and Evaluation
IMSS – Mexican Social Security Institute
LMICs – Low- and middle-income countries
MEA – Multilateral environmental agreement
NGO – Non governmental organization
NIOEH – National Institute of Occupational and Environmental Health (Vietnam)
μg/dL – microgram per decilitre
OECD – Organization for Economic Cooperation and Development
POP – Persistent Organic Pollutants
SAICM – Strategic Approach for International Chemicals Management
TEL – Tetraethyl Lead
UNEP – United Nations Environment Programme
USD – United States Dollar
USEPA – United States Environmental Protection Agency
WHO – World Health Organization

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1. Chapter 1: Introduction

1.1. Galena

Ore is a term used in mining to refer to a material imbedded with something valuable. In mineral extraction, ore is processed through mechanical and chemical methods leaving a small amount of something valuable, and a large amount of something else. In the case of gold, one of the highest grade mines in the world (Fosterville, Victoria, Australia) contains ore that is up to 0.000215 % gold, and 99.999785 % something else.[1,2] The ore is extracted from the ground, pulverized, and processed to concentrate the gold. What was once mostly rock and only 0.000215 % gold is now perhaps > 99 % pure. That small amount of material attracts a price sufficient to support one of the more complex mining operations in human history. The something else is left in above-ground piles next to the mine.

All metals are extracted from different ores. Aluminium ore is called bauxite, named after a village in Provence, France, and containing about 15–25 % aluminium. Polished bauxite is visually distinct; a lattice of various shades of orange and brown. Mercury ore is called Cinnabar. It has a deep red lustre and is composed of mercury bound up with sulphur. The Chinese decoratively carved Cinnabar for millennia; deep red marbleized vases, tobacco snuff jars. Other ores are less romantic: iron ore is called iron ore and looks like a common stone (and sometimes is). Titanium ore is called titanium ore, and so on.

The primary ore from which lead is extracted is called Galena. Galena is 3–8 % lead bound up with sulphur and sometimes contains silver. It is both dull in parts and glossy in others. It is angular and heavy. It feels expensive.

Figure 1. Galena. The primary ore from which lead is extracted.

Lead has been used by humans since perhaps 7000 BCE.[3] There are or have been lead mines on every continent except Antarctica. The ore is extracted, pulverized, combined with pine oil and frothed in water. The oil wraps itself around the lead and floats on the surface of the froth. Then the lead, the second most dense metal on the planet, is skimmed off the top. It is heated, but not melted, to remove residual sulphur and form a solid lump of material. The lump, called sinter, is then roasted at 1,300–1,400 °C in a furnace and combined with a source of carbon (i.e. coke). At this high temperature, oxygen is removed from the sinter by combining with the carbon in the coke, forming carbon dioxide or carbon monoxide. The same occurs with the sulphur, forming sulphur dioxide gas. This is immediately discernable when in the presence of a lead furnace with inadequate ventilation. Eyes and lungs are uncomfortably irritated. The molten lead, now > 95 % pure, is tapped from the bottom of the furnace and taken for more processing (i.e. more refining to remove residual impurities) before being cast into ingots, cooled, and sold.

1.2. The father of all metals

Freed from its ore, elemental lead is malleable, dense, heavy, corrosion resistant, and highly conductive of electricity. It can be sweet to the taste. When melted in a ceramic glaze it provides a lustrous waterproof coating. Ground up in a powder it can be a striking skin cosmetic, hair dye, or paint pigment. It is easily melted down and reformed, which can be done an indefinite number of times. Lead from batteries can be formed into fishing weights, which can be melted into bullets, which can be flattened into roofing tiles, which can be used again in batteries, and so on. After steel, lead is the most highly recycled material in the world; more than aluminium; more than copper; more than paper.[4]

Lead is also highly toxic. Lead exposure most commonly occurs when small particles are ingested. In children, perhaps 50 % of ingested lead is absorbed; mistaken by the body as calcium, it crosses the blood brain barrier and impedes the development of grey matter. Lead exposed children are more likely to have lower IQs, be more violent, less educated, and earn less money.[5–8] Lead can cause gout and hearing loss in adults.[9] Lead is also a killer: acute exposures killed nearly 1,000 children in Nigeria in 2012; chronic exposures kill 1 million adults every year, accounting for 2 % of global deaths.[10,11]

It is this duality, lead's utility and toxicity, that defines the history of our interaction with it. It is both irreplaceable and deadly in ways that perhaps no other material we use is. Lead was referred to by ancient Romans as the 'father of all metals' and by French during the Renaissance as a *poudre de la succession* (succession powder), owing to its use as a poison.[12]

Lead was not widely employed until classical antiquity in Rome around 700 BCE, though its first use was likely millennia earlier. Roman applications included plumbing (a term derived from the Latin name plumbum, hence the atomic notation Pb), cosmetics, food storage and preservation, and wine making. Processed sugar was not common in ancient Rome. Thus lead acetate, which is very sweet, provided an obvious utility.[13,14] Lead's toxicity was first documented by Hippocrates in the third century BCE who described colic in highly exposed people.[14,15] Famously, Nriagu attributed the collapse of the Roman empire to adverse neurological outcomes associated with lead's toxicity, though this claim has been elsewhere disputed.[16,17]

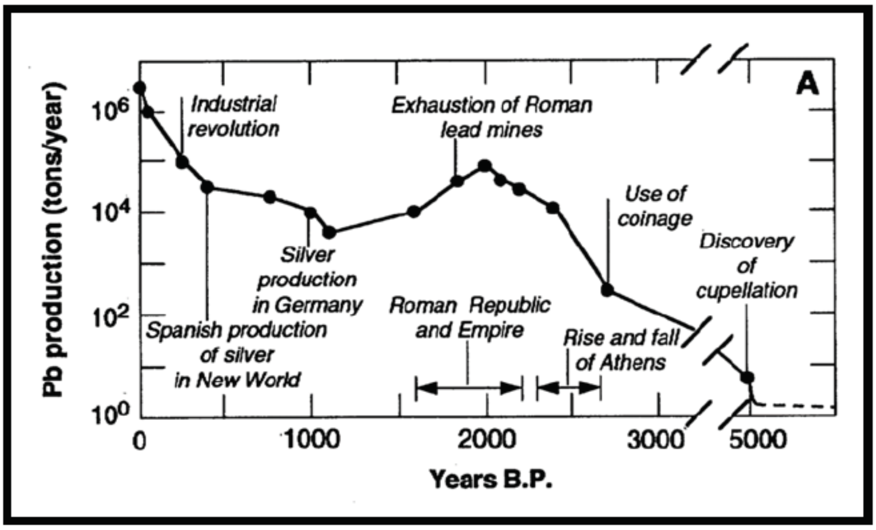

Figure 2. *Historical lead production in 'years before present,' from Hong et al (1994).*[20]

When lead is smelted, or combusted in engines, very small particles known as aerosols are dispersed into the atmosphere where they can be distributed and deposited globally. These global depositions are not necessarily high enough in concentration to impact human health, but they can provide a useful measurement of how much lead has been released into the atmosphere over time. This is particularly the case in the poles, where annual freeze-thaw cycles result in a tree ring-like structure in ice dating back hundreds of thousands of years. Thus it is possible to determine the tonnes of lead smelted in ancient Rome 2,500 years ago by evaluating the nanograms of lead 500 meters down in the artic ice today. Murozumi, et al. (1969) were the first to do this with regard to lead. Comparing contemporaneous measurements with pre-Roman deposition, the authors found a 300-fold increase in the amount of lead bound up in the ice.[18] Recent studies have been able to achieve incredible granularity in the data, tracking large increases in lead emissions to the Roman Empire and subsequent decreases associated with reduced production during plagues in the 1st and 3rd centuries ACE, for example.[19]

1.3. Twentieth Century releases of lead and resulting exposures

The story of humans' interaction with lead changes most dramatically in the 20th century as production increased by a factor of > 10. Where previous production of lead peaked in ancient Rome at ~80,000 tonnes/ year, global production by 1970 was over 4.6 million tonnes/ year.[20,21] Importantly, this does not necessarily indicate a proportionate increase in human exposure. While some level of contamination is present at nearly all lead smelters, exposure is often determined more by the application once lead leaves the facility. For instance, lead used in food containers clearly presents a greater exposure risk than lead used to shield dentists from X-rays.

In December 1921, a team led by a 32-year old mechanical engineer named Thomas Midgley initiated one of the largest chemical poisoning of humans ever documented. The team was working for General Motors to solve the problem of engine knock. In short, engine knock is the combustion of fuel outside of its intended time window. In the internal combustion engine, a piston compresses fuel and air within a cylinder. At maximum compression (i.e. top-dead-centre), a spark plug ignites, combusting the fuel and driving the piston downward. This forces a separate piston upward to compress fuel and air in a different cylinder. Another spark plug in that cylinder causes combustion, in turn driving the cylinder down, and so on. Combustion outside of this window, either early or late, can result in a pinging sound in mild cases, to engine failure in severe ones. Thomas Midgley and his team solved this problem by adding tetraethyl lead (TEL) to the fuel-air mixture, which suppressed combustion outside of top-dead-centre and eliminated engine knock.[22]

Tetraethyl lead (TEL) is organic, meaning it contains a carbon atom. Unlike metallic lead, it is readily absorbed by the skin. It is fat soluble and moves through the body more easily than metallic lead [23]. By handling the material in a laboratory setting, Midgely himself was overcome with symptoms of lead poisoning and forced to cancel several speaking engagements after having worked with TEL for just 12 months. Ironically, these engagements were in honour of his TEL discovery.[22] A year later in Bayway, NJ, USA more than 300 workers began to suffer from convulsions and hallucinations at one of the first facilities to industrially produce TEL. Five workers died from the exposure, which lasted only a few months.[24]

From TEL's first use as a petrol additive in Midgely's laboratory to its eventual global phase-out in the early 21st century, more than 10 million tonnes of lead were combusted and emitted from car engines around the world [25]. Because of their comparably larger cars, high rates of ownership, and long distances driven, Americans were the most heavily impacted, accounting for consumption of more than 80 % of the leaded petrol used before 1980.[16] TEL was eventually phased out in the US with the introduction of the catalytic convertor, which was rendered useless by the lead additive. Lead was found to bond with the various catalysts in the converter, inhibiting contact with the exhaust.[26] Thus contrary to intuition, TEL was not phased out for public health reasons in the US, though the benefits were significant.

From 1976 to 1980 mean blood lead levels (BLLs) in the US fell 37 %, from 15.8 μg/dL to 10 μg/dL.[27] Mean BLLs continued to decline rapidly through the 80s and 90s and are today < 1 μg/dL.[28] For context, the current US Centers for Disease Control and Prevention (CDC) reference level is 5 μg/dL, when CDC recommends public health interventions.[29] Other phase-outs of TEL followed globally. Rich countries did so voluntarily; most low- and middle income-countries (LMICs) were compelled to do so through conditional loans from the World Bank.[30,31] At present, only Algeria still adds lead to automobile petrol.[32]

The economic and social implications of TEL were staggering; increased violent crime, billions of lost IQ points, perhaps more deaths in the US than those caused by tobacco use.[33–35] The CDC estimated that children born each year after the phase-out would add an additional USD 110–319 billion more to the American economy over their lifetimes than children born before the phase-out.[7] A similar analysis globally found an additional USD 3.4 trillion added annually to the global economy, owing to phase-outs elsewhere.[36]

Lead-based residential paint too had a disproportionate impact on the United States. Banned in much of Europe and Australia, lead-based paint was used widely in the US until it was phased out beginning in 1971.[37] In the absence of bans, the availability of cheaper alternatives like titanium dioxide reduced lead use in the US beginning in the 1930s, though in total perhaps as much lead was used in paint as petrol in the 20th century.[38] Exposure to lead-based paint occurs most commonly when coatings deteriorate and become ingested as dust.[39] Thus, as coatings deteriorated decades after application, exposure likely increased. This coincided with the development of new analytical approaches in the 1960s like atomic absorption spectrometry which facilitated blood lead monitoring.[14] New York began limiting the use of lead-based paints in 1955, while Chicago began the nation's first BLL monitoring program in 1966.[37,40] Given its widespread application and high cost of removal, lead-based paint remained present in most homes for decades. A study in 2002 found that 40 % of American homes still had lead-based paint somewhere in the building.[41]

Episodic outbreaks of lead poisoning still occur in high income countries. A water contamination issue in Flint, Michigan garnered international attention. Lead exposure in public housing in New York City triggered a partial takeover by the Federal government.[42] Broken Hill and Port Pirie still experience extremely high levels of lead exposure in children, with ~50% of children < 5 years having BLLs > 5μg/dl.[43,44] Though, broadly speaking, societal lead poisoning in high income countries has been effectively eliminated. Blood lead levels in Flint were not commonly reported as mean values; but as the percentage above the CDC reference dose of 5 µg/dL. Approximately 5 per cent of area residents exceeded this level at the peak of the crisis in 2015, compared with 2.5 % before.[45] By way of comparison, in 1976 as leaded petrol was beginning to be phased out, 99.2 % of all Americans exceeded 5 µg/dL.[46]

1.4. New Sources of Lead Exposure

Despite the cessation of these major sources of exposure, lead production is increasing at a faster rate now than at any time in past 100 years. In 2013, 10.6 million metric tonnes of lead were produced globally – more than twice the amount in 1975 when leaded petrol began to be phased out.[21,47] Most of this lead – 85 % – is used in battery production, and most of those batteries are being made, bought and sold in LMICs.[21]

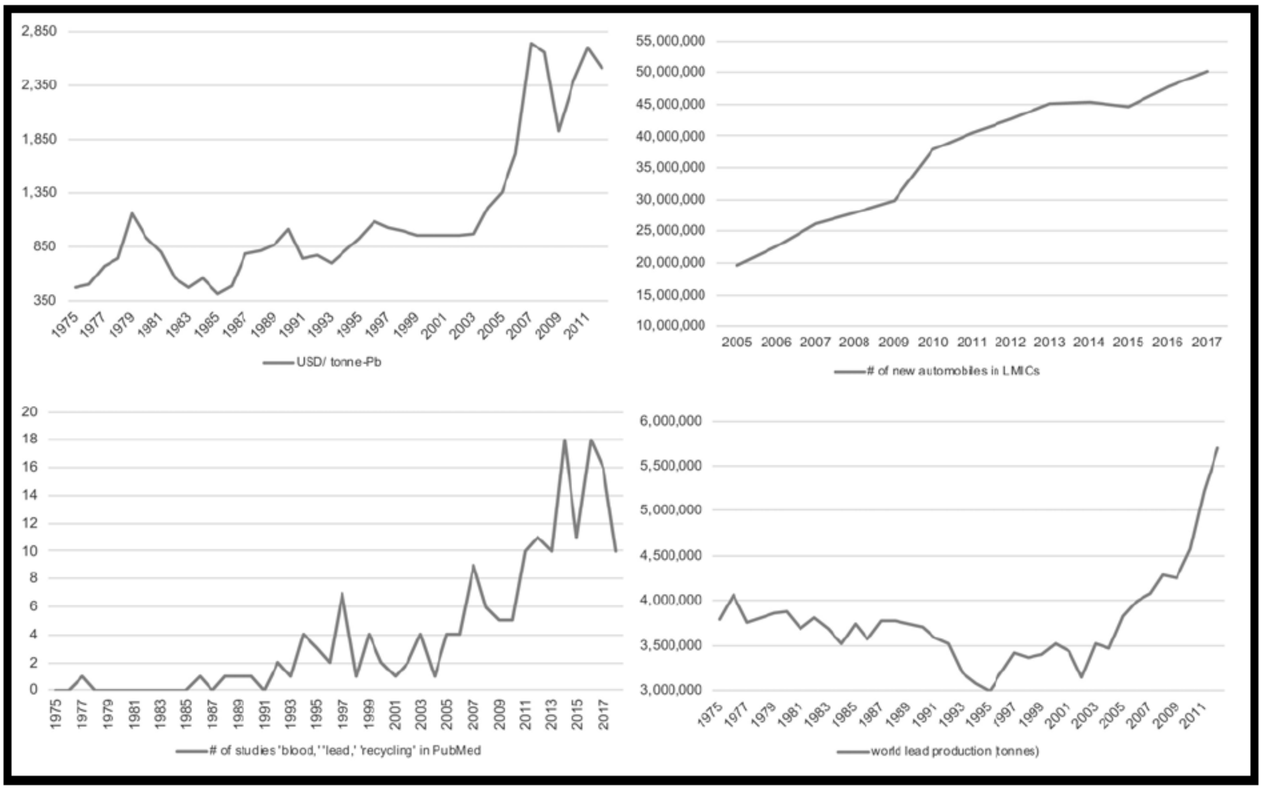

Figure 3. *Increasing interest in lead. Clockwise from top left: Price/ tonne-Pb on the London Metals Exchange;[48] number of new car registrants in select LMICs;[49] 'blood,' 'lead,' 'recycling' articles in PubMed;[47] world lead production (tonnes)[46]*

The impacts of this industrial transition are not well understood. Anecdotal reports of acute community-wide lead poisoning are beginning to appear in the literature with greater frequency. From the decade 1990–2000, 10 articles are listed in PubMed against the terms 'blood,' 'lead,' and 'recycling.' From 2000–2010, there are 38. In the most recent decade, 110 articles are listed. By contrast, only a single study listed in PubMed from the 1970s meets these criteria.[48]

There has also been a sharp increase is the price of lead, closely tracking the commodity boom.[49] Unlike other commodities however, lead has remained expensive, driven by real demand for automotive batteries. The number of new automobiles being sold in LMICs more than tripled from 2000 to the present.[50] Batteries have shorter lifespans in the warm climates of LMICs, which tend to cluster around the equator.[51,52] Local replacement batteries can have poor build quality and even shorter lifespans. In general, a lead acid battery in an LMIC may last about 2 years. Electric bikes, backup power supplies, deep-cycle batteries in solar arrays – all are increasing in lockstep, collectively forming a massive reservoir of accessible, valuable lead.

1.4.1. Informal Sector Industry

The term 'informal sector' is generally attributed to the British anthropologist Keith Hart who developed it in response to the complex nature of work he encountered in Ghana in the 1970s.[53] Broadly, it defines the portion of a given economy that is not regulated by the state. It is an oddly comprehensive term, essentially describing what businesses do not do: informal sector businesses generally do not pay taxes; they do not provide or receive benefits; comply with relevant social, environmental, or health and safety laws. Illicit drug dealers are members of the informal economy, as are 'off-the-books' day-care centres and street hawkers. The informal sector comprises nearly 20 % of global GDP and more than 30 % of global employment. The size of the informal economy in a given country is strongly associated with its income; poor countries generally have larger informal sectors. OECD countries are ~15 % informal; Africa is > 40 % informal.[54]

'Informal' does not apply exclusively to work. Housing, too, can be informal. More than 30 % of the world's urban population lives in slums – illegal housing typically without tenure. In LMICs, more than 60 % of urban residents live in slums.[55] Informal housing is not zoned, or at least the zoning laws are not enforced. Slums are built in the margins - underutilized land between industrial estates, on river beds, in swamps and along train tracks and highways. Slums are heterogeneous and vibrant; entire cities symbiotically coexisting within their formal counterparts. There are churches, bakeries, schools and playgrounds.

In rare cases, informal industry is allotted land where it can operate away from residential areas.[56,57] Separating industrial activity from residential areas is of course common in high income countries. Cities are zoned to separate heavy from light industry, light industry from commercial, commercial from residential, and so on. Environmental standards stem from this zoning. The USEPA screening level for lead in industrial soil is 1,200 mg/kg, for instance, three times that for residential play areas.[58] When not allotted a space for their work, informal businesses invariably find one. Slums provide an obvious relief from regulatory oversight, can be affordable and convenient. Thus, much of informal industry occurs in residential areas; in backyards, living rooms, and converted flats.[59–62]

1.4.2. Used Lead Acid Batteries

Figure 4. *Informal lead smelter, Bogor, Indonesia*

Informal lead recycling is uncomplicated. At its most basic an informal smelter is comprised of a hole in the ground, a limited number of instruments for ladling and casting molten lead, and a machete. Nearly all feed stock for secondary lead smelters is comprised of used lead-acid batteries (ULABs). Lead-acid batteries have a range of applications with minor differences made to the basic design. Starting, lighting, and ignition batteries provide high amperage in a short period for the purpose of turning over an engine. Deep cycle batteries hold a charge a long time and deliver far fewer amps, making them ideal for solar arrays. Truck batteries are big, motorcycle batteries are small.

All lead-acid batteries follow the same basic design developed in France in the mid-19th century. Alternating lead (negatively charged) and lead oxide (positively charged) plates reside in a sulphuric acid bath, separated by dividers. When the positive and negative terminals are connected, electrons move through the bath from one plate to the other. This transfer of electrons forms an amorphous sulphate coating on the outside of the plates, eventually limiting any further chemical reactions. When the battery is recharged, the reverse occurs. The amorphous lead sulphate is dissolved and the plates are converted back to lead oxide and lead. Over multiple discharge and charge cycles, lead sulphate crystals form on both plates, rendering them useless.

In an informal smelter, batteries are cracked open with a hand axe or machete and the sulfuric acid solution is released onto the ground. The plastic dividers are peeled from the lead plates and deposited in waste piles near the site or burned as fuel. The hole in the ground, about 30 centimetres deep, is filled with charcoal and ignited. Lead plates are then placed in the hole, melted and stirred. Because lead is so dense, impurities such as the lead sulphate crystals, float to the top of molten mixture and are skimmed off. The molten lead is then ladled into ingot moulds, cooled, and sold. In most countries, these ingots are bought by formal sector actors, though informal battery production is common in others.

Figure 5. *Informal ULAB processing in Dhaka, Bangladesh*

A typical car battery contains ~10 kg of lead, worth about USD 20 in 2018. Informal recyclers purchase used batteries from a network of brokers, mechanics, and end users. Because informal operators avoid the some of the more significant costs associated with running a lead smelter—pollution controls and worker health and safety—their profit per unit is greater than their formal counterparts. They can therefore afford to pay a higher price for individual batteries. It follows then that a perennial complaint of formal smelters is the lack of batteries to meet full capacity.[63,64] However, given that many smelters also manufacture batteries, purchasing lead from the informal sector can function as a regulatory back door, allowing these companies to sidestep some of the costs associated with environmental compliance.

Within informal recycling facilities, the exposures are horrific. Pulverized lead powder is inadvertently ingested and vapours are inhaled. The ground is uniformly darkened by sulphuric acid. Workers become coated in the material; carrying it home on their skin, their clothes and in their hair. As they walk through their communities they deposit micrograms of lead onto area surfaces. In their homes, they deposit the balance. Children of smelter workers in LMICs almost invariably have dangerously high levels of lead in their bodies.[65–67] In the most severe cases, lead is processed within the home. One notorious example resulted in the deaths of 20 children in Senegal in 2012, though the practice is not uncommon.[68]

Figure 6. *Semiformal ULAB processing in Dong Mai, Vietnam*

Aerosols arising from the rudimentary smelters quickly cool and fall to the ground. Soil concentrations are highest at the site and taper off radially for about 100 m. That deposited lead resides in the top 2.5 cm of soil more or less indefinitely.[69–71] The lead-rich particles are transported as windblown dust and inhaled. Lead dust is typically too large to penetrate the lungs and is moved by cilia to the throat where it is ingested. Once there, it is partially dissolved and taken into the blood. Lead is stored in bone. Broken bones in adults can lead to a spike in BLLs from a childhood exposure decades earlier.[9]

The Indian Lead Zinc Development Association estimates that informal recyclers process more than 50 % of secondary lead in that country.[64] A paper published as part of this thesis extends this number to apply to all secondary lead processed globally, based on interviews with industry experts and automobile production data (Paper 6). These estimates would not be captured in official statistics given above and therefore imply that in reality much more lead is being recycled than is reported. Whatever the amount being processed in the informal sector actually is, it is certainly very large. Most of that lead is reused in batteries, though there are leakages to other applications. Lead from battery recycling is used as an adulterant in spices and illicit drugs, as a glaze in roofing tiles and traditional pots, and in fishing weights.[72–75] It is also used in metal cookware.[76] Lead has a low specific heat, meaning it stays hot for a long time. Aluminium is the opposite. A scourge of informal pot makers is the rapid congealing of molten aluminium before it fully penetrates the cast. The addition of lead keeps the molten aluminium hotter longer and eliminates this issue.

Figure 7. *Semiformal ULAB smelting in Tegal, Indonesia*

1.4.3. The international response

The steep increase in production and use of lead in LMICs is distinct in its reach and exposure risk. Human beings have simply never processed lead in their homes at such a scale before. However, current efforts to monitor and mitigate the issue are limited. The Institute for Health Metrics and Evaluation (IHME), based at the University of Washington (Seattle, USA) is the global leader in the collection and presentation of health data. In 2018, IHME signed a memorandum of understanding with the World Health Organization (WHO) to cooperate in the development of annual global burden of disease estimates, effectively making IHME the official source of these statistics.[77] Due to of the dearth of health reporting globally – only 33 % of global deaths are certificated – IHME supplements existing data with complex modelling.[78] In the case of lead, their model begins with a literature review of published blood lead data. The literature review from the most recent analysis included 553 studies spanning the years 1970–2016. Looking at the period 2010–2017, IHME utilized 88 studies from 27 countries.[79] On its surface, the review does not seem to be comprehensive. As context, a systematic review included in Paper 1 of this thesis extracted data from 477 articles from 50 countries for the period 2010–2018.

The results of IHME's literature review are then used with four other covariates to estimate BLLs for all countries. These are then used as the basis of calculations of the attributable disease burden. One of the covariates is gender; women have fewer red blood cells (where lead resides) than men and accordingly lower BLLs.[80] Significantly, all three other covariates relate to the influence of a single source: leaded petrol. At present, no other environmental inputs are considered beyond what are inherently included in the literature review. This is important because leaded petrol was phased out in most countries more than a decade ago, though other sources are common, particularly in LMICs.

International efforts to mitigate lead exposure, like those to accurately quantify its health impacts, are somewhat limited and poorly resourced. The most recent WHO report on lead, which focuses on used lead acid battery recycling, was published in 2017. The report that preceded it was released 2010, is 5 pages long, and lists petrol as a key source of exposure. The United Nations Environment Program's lead department currently consists of two researchers and an intern, whom also work in other areas. The only international funding mechanism to mitigate lead exposure is the Global Environment Facility (GEF), which is by mandate restricted to funding multilateral environmental agreements (MEAs). Given that an MEA does not exist for lead, the GEF had to make an exception under the Persistent Organic Pollutants window at the urging of the Global Alliance to Eliminate Lead Paint. In the most recent funding cycle (2014) the GEF distributed USD 1.8 million in grants to mitigate lead-based paint exposures, of a USD 4.4 billion budget.

1.5. Methods

It is difficult to know the size of something we are not measuring, or to deal with a problem that we know little about. This thesis seeks to begin to quantify the exposure risk presented by lead in LMICs and to explore methods to mitigate the most severe cases. It does this through five chapters. The first is the present introductory chapter, which provides context to the overall thesis. In Chapter 2 (Exposures) three risk assessments are grouped with a systematic review and a modelling exercise. The purpose of the chapter is to elucidate the sources of exposure and to understand their reach and severity. Lead acid batteries are examined, as is lead-based paint, mines, and smelters. The chapter includes two studies of issues that received significant international attention, though presented a limited exposure risk. The intention here is to underline the importance of data in formulating a response.

Chapter 3 (Outcomes) attempts to quantify the disease burden attributable to these exposures. This is done primarily through the calculation of Disability Adjusted Life Years (DALYs) and comparisons with IHME's results. When the data are amenable to direct comparison, the results highlight illuminating differences in the methods. In the India meta-analysis conducted as part of Paper 9 for instance, the DALY estimates for adults are within the margin of error. By contrast, our estimates for Indian children highlight a near absence of a lead-attributable disease burden in IHME's report on this population. This has been partly corrected for in their most recent analysis, though is still likely too conservative.[11]

Chapter 4 (Mitigation) details risk mitigation work in LMICs through different projects BE has managed. The projects are associated with informal ULAB recycling and e-waste and were executed in four different countries. Three of the projects described resulted in significant declines in the biological burden of lead in affected children. Given the near absence of lead risk mitigation work in LMICs, this chapter has the twofold intent of providing guidance for other implementers and making the case that the problem can be cost effectively mitigated.

The major results are presented again in a concluding chapter (Chapter 5). Here, the disparate discussions from preceding chapters are synthesized into a more holistic argument. There are several themes woven throughout the chapter. The three overarching arguments are that the lead-attributable burden of disease in LMICs is very large, that the sources are distinct from high income countries, and that cost effective, locally appropriate solutions can be applied. These arguments are rooted in empirical evidence from environmental and health sciences. Together they are intended to provide some indication of a way forward. The fourth argument is that the current international response to the issue of lead exposure in LMICs does not necessarily reflect the *de facto* nature and severity of those exposures. This argument is based on an evaluation of current disease burden estimates and a review of existing relevant programs. It is made throughout the studies presented in the thesis, and synthesized to conclude Chapter 5.

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2. Chapter Two: Exposures

Chapter 2 consists of the following 5 studies:

2.1 **Ericson, B**., Nash, E. Sinitsky, J., Masek, V, Ferraro, G & Taylor, M.P. (2018). Sources of lead exposure and resulting blood lead levels in low- and middle-income countries: a systematic review and meta-analysis. Submitted to Environmental Research.

2.2 Dowling, R., Caravanos, J., Grigsby, P., Rivera, A., **Ericson, B.,** Amoyaw-Osei, Y., ... & Fuller, R. (2016). Estimating the prevalence of toxic waste sites in low-and middle-income countries. *Annals of Global Health*, *82*(5), 700-710

2.3 **Ericson, B.,** Hariojati, N., Susilorini, B., Crampe, L. F., Fuller, R., Taylor, M. P., & Caravanos, J. (2019). Assessment of the prevalence of lead-based paint exposure risk in Jakarta, Indonesia. *Science of The Total Environment*, *657*, 1382-1388.

2.4 **Ericson, B.,** Otieno, V.O., Nganga, C., St. Fort, J. and Taylor, M.P. (2019). Assessment of the Presence of Soil Lead Contamination Near a Former Lead Smelter in Mombasa, Kenya, *Journal of Health and Pollution*, *9(21), 190307.*

2.5 Bose-O'Reilly, S., Yabe, J., Makumba, J., Schutzmeier, P., **Ericson, B.,** & Caravanos, J. (2018). Lead intoxicated children in Kabwe, Zambia. *Environmental Research*, *165*, 420-424.

Chapter 2 provides the basis for the subsequent thesis chapters, which address the quantification and mitigation of exposures in LMICs. The chapter presents context for the research aims of this thesis. Specifically, blood lead levels in LMICs remain above international thresholds, yet the sources of exposure are both poorly understood and distinct from historical sources in high income countries

The five studies that comprise Chapter Two evaluate sources of lead exposure in low- and middle-income countries (LMICs). The first study presents a systematic review and meta-analysis of recently published literature on BLLs. Major sources of lead contamination and their resulting biological burden are quantified. The second study in the chapter examines one such source—contaminated land resulting from informal industry—and estimates its scale across a single country based on a smaller subset of environmental samples. This paper presents a possible method for dealing with the existing data gaps on the presence of contamination sources in LMICs. The three remaining studies in Chapter 2 present data from *in-situ* assessments, evaluating routes of exposure to detail how lead behaves in human environments.

2.1. Paper One

Ericson, B., Nash, E. Sinitsky, J., Masek, V, Ferraro, G & Taylor, M.P. (2018). Sources of lead exposure and resulting blood lead levels in low- and middle-income countries: a systematic review and meta-analysis. To be submitted to Environmental Health Perspectives.

This study completed the most extensive systematic review of the peer-reviewed literature on lead exposure and its sources in LMICs to date. The review accessed the PubMed database by searching with the keywords 'blood,' 'lead,' and '[country name]' for each of the 137 countries in World Bank LMIC country groupings, yielding 10,093 articles. The articles were then screened for pre-defined inclusion criteria relating to human exposure, which returned 477 papers containing 979 sampled populations (i.e. subsamples) from 50 countries comprising 702,069 individuals who had been sampled as part of these studies. Data on the source of exposure and biological concentration of each sample were extracted and pooled to develop national BLL estimates.

This study examined a central theme of Chapter 2, which is that sources of lead exposure in LMICs are distinct from historical sources in high income countries. The results enabled a quantitative assessment of the established sources of exposure. In addition, the aggregation and publication of the extracted data provides a valuable source for other researchers who wish to evaluate lead exposure in LMICs, extending the benefit and life of the study beyond the confines of this thesis.

Paper one is currently undergoing final QA/QC and will be submitted to Environmental Research.

Sources of lead exposure and resulting blood lead levels in low- and middle-income countries: a systematic review and meta-analysis

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lead exposure; LMICs; blood lead levels; meta-analysis; children

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Abstract
Few studies have systematically evaluated the academic literature to assess blood lead levels (BLLs) in low- and middle-income countries (LMICs). Fewer have begun to evaluate the attributable exposures. This review accessed the PubMed database by searching with the keywords ‘blood,’ ‘lead,’ and ‘[country name]’ for each of the 137 countries in World Bank LMIC country groupings, yielding 10,164 articles. The articles were then screened for pre-defined inclusion criteria relating to human exposure, which returned 477 papers containing 979 sampled populations (i.e. subsamples) from 50 countries comprising 702,069 individuals who had been sampled as part of these studies. Data on biological concentrations were extracted and pooled to calculate national BLL estimates. Data on sources of exposure were also extracted when available. Background BLLs could be pooled for children in 32 countries and for adults in 36 countries. Altogether more than 670 million children were calculated to have BLLs exceeding the CDC reference dose of 5 µg/dL in the 32 countries included.

1. Introduction

Naturally occurring concentrations of lead in the earth’s crust are below applicable human health guidelines, with exposure resulting primarily from anthropogenic contamination (Alloway, 2013; Environment Canada, 1999; Smith et al., 2013; USEPA, 1998). Lead was first widely employed by humans during the Roman Empire primarily in plumbing, food storage and as a sweetener and preservative in wine (Needleman, 1999; Retief and Cilliers, 2010). Incidence of lead poisoning in the Romans is well documented, with some researchers associating exposure with the Empire’s decline (Gilfillan and Gillifan, 1965; Nriagu, 1983; Woolley, 1984). Lead continued to be used in food storage, plumbing and other applications for centuries. Studies of Ludwig van Beethoven's hair carried out using x-ray fluorescence spectrometry have implicated lead exposure from wine containers in causing his hearing loss and death, though these conclusions have been elsewhere disputed (Barron, 2010; Mai, 2006; Stevens et al., 2013).

The addition of tetraethyl lead to petrol as an anti-knocking agent throughout the 20th century resulted in the widespread distribution of lead onto surface soils globally (Ng and Patterson, 1981; Nriagu, 1990). More than seven million tons of lead were released by petrol in the United States from 1926 when it was introduced until 1985 when it was largely phased out, with at least an additional 2 million tons being released in other countries (Nriagu, 1990). Largely as a result of those emissions, present global surface soil concentrations in cities regularly exceed 200 mg/kg, approximately 10-20 times natural background levels (Ajmone-Marsan and Biasioli, 2010; Alloway, 2013; Smith et al., 2013).

In high-income countries, a separate important contribution to human lead exposure was made by lead-based enamel paints in residential housing, particularly in the first half of the 20th century in the United States (US) (Jacobs et al., 2002; Needleman, 2004; Pirkle et al., 1998; Schwartz and Levin, 1991). As painted surfaces deteriorated years after application, incidental ingestion of lead dust resulted in elevated blood concentrations in children (Dixon et al., 2009). Several other sources of human lead exposure in high income countries have included the use of lead in water distribution systems, traditional medicines, mining and smelting complexes, and other industrial processes (Brown and Margolis, 2012; Kristensen et al., 2017; Levin et al., 2008; Spalinger et al., 2007).

Biological monitoring of human lead exposure is most commonly done through blood lead level (BLL) measurements, though can also be quantified through assessments of bone. Other biological media have not been evaluated to the same extent (US EPA, 2014). The half-life of lead in whole blood is approximately 30 days, thus BLLs capture the level of exposure proximal to the time of extraction (ATSDR, 2007; Chamberlain et al., 1978; Rabinowitz et al., 1976). This contrasts with bone lead measurements which are better suited for assessments of chronic exposure; lead is cumulative in bone, where it can reside for decades (Rabinowitz, 1991). Elevated BLLs are associated with cognitive deficits, cardiovascular disease, liver and kidney disease, hearing loss, gout, and multiple other adverse health impacts (US EPA, 2013).

In high-income countries, BLLs and sources of exposure for the general population are relatively well understood. In the most rigorous case, the US Health Resources and Services Administration adopted the American Academy of Pediatrics recommendation to conduct regular BLL testing and risk assessment of children in 1998 (Raymond et al., 2014). At present, these recommendations call for risk assessment by primary care physicians of all children at 9 months followed by BLL screening at 12 and 24 months of age (Hagan et al., 2017). This data collection is supplemented by a more comprehensive biomonitoring regime in a representative sample of the US population through the National Health and Nutrition Examination Survey (NHANES), which has been carried out since the 1960s and examines approximately 5,000 people each year (Center for Health Statistics, 2017). European countries monitor BLLs to different extents, with the most comprehensive being Germany (Rudnai, 2007). Canada collects a representative sample from the population rather than census level coverage (Canada, 2013).

There have been several efforts to systematically review BLLs in low- and middle-income countries (LMICs). In the absence of large government datasets like those available in high-income countries, researchers have generally relied on reports in the peer-reviewed literature. Horton, *et al* (2013) conducted a systematic review of English language studies published between 2000–2012 containing children's (≤ 18 years) blood and urine metals concentrations in 10 emerging market countries. The authors presented the results of 76 studies, but did not endeavor to synthesize them into a nationally representative value for each country. In most cases, the BLLs identified exceeded US reference levels (Horton et al., 2013). Olympio, *et al* (2017) reviewed papers published from 2000–2014 with children's (≤ 18 years) BLLs in all Latin American and Caribbean countries. In total, the authors extracted results from 56 papers in English, Portuguese and Spanish. Like Horton, *et al* (2013), Olympio, *et al* (2017) presented the results for each study independently and did not endeavor to synthesize the results into nationally representative values (Olympio et al., 2017). A third study, completed by Attina and Trasande (2013), calculated the economic costs associated with pediatric lead exposure in LMICs. As a basis for their calculation, the authors extracted data from 68 articles published from 2000–2012 containing BLLs for children. One sample used in their analysis included BLLs of individuals ≤ 20 years, while the balance represented children ≤ 14 years. In cases where their samples spanned the implementation of leaded petrol bans, the authors adjusted BLLs downward to reflect the removal of that source. Where more than one sample was available in each country, the authors calculated sample size-weighted means using methods developed by Fewtrell, *et al* (2003; 2004). They further used pooled sub-regional levels to estimate BLLs in countries without any available data (Attina and Trasande, 2013).

Finally, the Institute for Health Metrics and Evaluation (IHME) conducts a literature review of BLLs as the basis for their annual Global Burden of Disease calculations. Their most recent analysis included 553 studies spanning the years 1970–2016, with 345 being from LMICs. From the period 2010–2017, 88 studies from 27 LMICs were utilized (IHME, 2018). The published BLLs are pooled and then adjusted for covariates (i.e. traffic, urbanicity, gender, leaded petrol phase-out date) to calculate national BLLs and the attributable disease burden in Disability Adjusted Life Years (DALYs) and deaths.

Despite these efforts to systematically review the available data, it is possible that a number of studies have not yet been included. Moreover, while previous studies have calculated national BLL estimates for the purpose of calculating the attributable disease burden, no study has yet to publish these estimates. The present effort endeavored to both expand the scope of the literature review to a larger set of studies as well as to publish nationally representative BLLs and supporting data. The purpose is in part to aggregate the data in a format amenable to use by other researchers.

Additionally, no systematic review of BLLs in LMICs has yet to characterize sources of exposure. This study reviewed publications for probable sources and provides the results of that effort here. Finally, the study included only those data published after leaded petrol phase outs, thereby somewhat facilitating an assessment of exposure levels from non-petrol sources.

2. Methods

2.1 Literature Review

A PubMed search was conducted in August 2018 using the search terms '[country name]' (all fields, Medical Subject Headings [MeSH] terms, abstract text) 'blood' (subheading, all fields, MeSH terms), and 'lead' (all fields, MeSH terms) for publications dated between 1 January 2010 and 30 June 2018 (National Library of Medicine (US), 1946). Low- and middle-income country names (n=137) were taken from World Bank groupings (World Bank, 2016). The review was listed with the PROSPERO International prospective register of systematic reviews maintained by the UK National Institute for Health Research on 13 September 2018 (Ericson and Nash, 2018).

Studies were reviewed for the following inclusion criteria:

1) contained BLL data from human populations residing in [country];
2) comprised at least 30 participants;
3) presented BLL data derived from venous, capillary, or umbilical cord samples of whole blood (serum and plasma samples were excluded);
4) data must have been collected after 2005;
5) published in English.

Two reviewers (BE, EN) independently searched PubMed with the defined search terms. In cases where searches returned 50 or fewer results, titles were reviewed in the PubMed online interface. Titles that indicated that the study was relevant to the effort were exported to a comma separated values (CSV) file. In cases where searches returned more than 50 results, all titles were downloaded as a spreadsheet (CSV) for review. Following a review of each title in the spreadsheet for relevance, keywords were then used to identify any titles missed in the initial review. The following 18 terms were used as keywords: trace metals; Pb; e-waste; e waste; battery; workers; paint; metal; pipes; smelter; mine; ceramic; candy; cosmetic; exposure; kohl; surma; heavy metals. Due to the large number of studies examined and excluded at this stage (n > 9,000), exclusion reasons were not noted. In the case of one country, India, the literature review was conducted by a single reviewer only (EN). The results of this review were then compared with the results of a recent systematic review conducted by BE (Ericson et al., 2018).

The abstract of each selected title was reviewed for relevance. When a study did not meet the inclusion criteria, it was excluded and a justification was provided by the reviewer. Each reviewer independently completed all steps in the review process, with all unique studies being combined in a comprehensive list.

2.2 Data Extraction

In cases where the abstract indicated that the study met all the inclusion criteria, a single reviewer (BE) extracted the following information where available: title; author; year; location; population characteristics (gender, age); BLL statistics (central tendency, dispersion, sample size); sources of exposure; and analysis method. In addition, the nature of the exposure (i.e. background, occupational, non-occupational) was captured.

Subsamples were coded to different subgroups based on the subsample's characteristics. Subsamples with an abnormally high exposure (e.g. living near a known hazardous waste site, applying surma/ kohl) were coded to the subgroup 'non-occupational.' Those drawn from general populations or used as controls in case-control studies were coded to the subgroup 'background.' Subsamples drawn from worker populations with exposure to lead were coded to the subgroup 'occupational.'

Where possible, BLLs were separated into adult/ child subsamples, and coded to the appropriate adult or child subgroup. Where this was not possible, the subsample was coded to subgroup 'both.' For the purpose of this study a child was defined as ≤ 18 years of age, consistent with the UN Convention on the Rights of the Child (United Nations General Assembly, 1989). Subsamples were also disaggregated by occupation and source of exposure where those data were available. Subsamples were disaggregated by gender only when a sample with both genders was unavailable. If gender was not specified, subsamples were coded to subgroup 'both.'

With regard to the identification of exposure sources, these were coded based on the weight of the evidence provided. Studies with probable sources of exposure such as communities living near identified contaminated hotspots or adjacent to industrial areas, occupational exposures, or those with statistically significant associations between BLLs and an environmental assessment or questionnaire were coded to subgroup 'probable.' Studies that evaluated BLLs in a given population and that provided possible sources of exposure based on a review of the literature rather than an *in situ* assessment of exposure were coded to subgroup 'possible.' Studies that did not define an exposure source were coded to subgroup 'undefined.' Studies were not systematically evaluated for bias. Data from all studies meeting the inclusion criteria were included.

As noted above, a single reviewer (BE) extracted data from all studies included. As a quality control measure, the same reviewer then confirmed the descriptive statistics for a single extracted subsample from each study. Where errors were identified, all subsamples from the affected study were reviewed and corrected where necessary.

A team of reviewers (EN, JS, VM, GF) then divided and reviewed the entire extracted dataset equally, correcting immaterial discrepancies independently, and reviewing material findings with BE and EN.

2.3 Imputing Missing Data

Multiple studies did not provide sufficient statistical information for the pooling of data. Specifically, arithmetic mean, standard error and standard deviation in micrograms per deciliter (µg/dL) were required to pool data in the methods utilized here. Where these data were unavailable a number of methods were employed to impute these values. The selection of the appropriate approach was guided by recommendations set out in Weir et al., (2018), where widely used methods for imputing missing data were evaluated. These are described below.

2.3.1 Unit Conversions

The analysis was conducted using units in µg/dL, however studies presented results in a range of different units. Where data were presented in µg/dL, no conversions were made. Where data were presented in µg/L or mg/dL, the order of magnitude was adjusted appropriately to make the units consistent. Some data were presented in µmol/L. In these cases, the atomic weight of Pb^{208} (the most common isotope, i.e. 207.966525) was used to make the conversion to the appropriate units (CRC Handbook, 2018).

2.3.2 Arithmetic Means

Arithmetic means were imputed using a number of methods. These are presented here in order of preference, with the most preferred presented first and the least preferred presented last.

If the arithmetic mean was not available for a given subsample, the geometric mean was used. If the geometric mean and arithmetic mean were not available, three separate methods were employed to impute the arithmetic mean and are presented below. The first two were taken from Wan, *et al* (2014) and improve upon widely utilized approaches developed by Hozo, *et al* (2005). Of the methods presented by Wan, *et al.* (2014) the two utilized here provide the least biased results when dealing with skewed data, as is typically the case with BLLs. In all cases:

a = the minimum value,
q_1 = the first quartile,
m = the median,
q_3 = the third quartile,
b = the maximum value,
n = the sample size.

In the most preferred scenario (1) the following data were available for the subsample: q_1, m, and q_3. In this scenario, the following calculation was used to estimate the arithmetic mean:

$$\bar{X} = \frac{q_1 + m + q_3}{3} \tag{1}$$

Where:
$\bar{X}$=Subsample mean

In the second most preferred scenario (2) where the following data were available for the subsample: a, m, and b. In this scenario, the following calculation was used to estimate the arithmetic mean:

$$\bar{X} = \frac{a + 2m + b}{4} \tag{2}$$

Finally, if only a median value was available, this value was taken to be equal to the mean.

2.3.3 Standard Deviation

In cases where the standard deviation was not presented for a given subsample, it was calculated by multiplying the standard error (if available) by the square root of the sample size in a manner consistent with the *Cochrane Handbook* (Higgins and Green, 2008). If neither the standard deviation nor standard error were presented, a series of methods were employed to impute the standard deviation of the subsample, as above. Similarly, these are presented in order of preference, with the most preferred presented first and the least preferred presented last.

In the most preferred scenario (3) the following statistics were available for the subsample: a, q_1, m, q_3, b, and n. In this scenario, the following calculation from Wan, *et al* (2014) was used to impute the standard deviation:

$$s \approx 1 + \frac{b-a}{4\Phi^{-1}\left(\frac{n-0.375}{n+0.25}\right)} + \frac{q_3-q_1}{4\Phi^{-1}\left(\frac{0.75n-0.125}{n+0.25}\right)} \tag{3}$$

Where:
s=Standard Deviation of the subsample
$\Phi^{-1}(z)$=the upper zth percentile of the standard normal distribution

In the second most preferred scenario (4), the following statistics were available for the subsample: q_1, m, and q_3. In this scenario, the following calculation from Wan, *et al* (2014) was used to impute the standard deviation:

$$s \approx \frac{q_3-q_1}{2\Phi^{-1}\left(\frac{0.75n-0.125}{n+0.25}\right)} \tag{4}$$

In the third most preferred scenario (5), the following statistics were available for the subsample: a, m, and b. In this scenario, the following calculation from Wan, *et al* (2014) was used to impute the standard deviation:

$$s \approx \frac{b-a}{2\Phi^{-1}\left(\frac{n-0.375}{n+0.25}\right)} \tag{5}$$

Finally, in the least preferred (6), where only the sample size and mean were available, missing standard deviations were imputed based on the weighted average of variances reported in other studies, following Ma, Liu, Hunter, & Zhang (2008). Doing so relies on the assumption that samples are randomly selected from the same population. Because of the large difference between background BLLs, and those of non-occupationally and occupationally exposed groups, missing SEMs for each group were calculated separately based on the reported and imputed values from their respective group only. Standard errors (SEM) were imputed first then used to calculate the standard deviation by multiplying the SEM by the square root of the sample size, as described above. The following equation describes the process for imputing missing SEMs from reported SEMs:

$$SEM_j \approx \frac{\sum_i^k SEM_i\sqrt{n_i}}{k\sqrt{n_j}} \tag{6}$$

Where:
j = studies without published standard deviations
k = studies with published standard deviations

2.3.4 Standard Error

In cases where the SEM was not available it was imputed by taking the standard deviation and dividing it by the square root of the sample size in a manner consistent with the *Cochrane Handbook* (Higgins and Green, 2008).

2.4 Pooling Sample Means

Mean background BLLs were calculated at the country level for both children and adults using two different approaches. The first approach follows Ericson, *et al* (2018) employing a random effects meta-analysis model using the metan tool in Stata 15.1 (StataCorp. LP, 2017). This tool utilizes the method presented by DerSimonian & Laird (1986), weighting each sample mean by its SEM. Using this tool, a pooled mean and 95% confidence interval were calculated for each country where data were available.

The second approach (7) follows Trasande and Attina (2013) utilizing a method presented by Fewtrell (2003). In this method, means and standard deviations were transformed into their natural logarithms before being weighted by sample size. Specifically, the log-transformed sample means are multiplied by their respective sample sizes and summed. The sum was then divided by the sum of the sample size to attain the average. Finally, the natural antilogarithm of the average was taken. Thus, using this method the population arithmetic mean for each country was determined as follows:

$$\mu = e^{\frac{\sum \ln(X) n}{\sum n}} \tag{7}$$

To calculate the population standard deviation, the variance was first calculated by squaring each sample standard deviation. The natural logarithm of the variance was then taken and multiplied by the sample size, which was in turn divided by the sum of the sample sizes to calculate the average. Finally, the natural antilogarithm of the square root of the average was taken. The resulting equation (8) is as follows:

$$\sigma = e^{\sqrt{\frac{\sum \ln(s^2) n}{\sum n}}} \tag{8}$$

Mean BLLs by exposure setting and source were calculated using the second approach only. Specifically, mean BLLs for background exposure, non-occupational, and occupational settings for children and adults were calculated for each identified exposure source.

Finally, if only one subsample was available for a given subgroup in a given country, the values of that subsample were taken as representative of the country as a whole.

2.5 Calculating the Number of Children Above Threshold BLL Values

Once a population mean BLL and standard deviation were determined for each country, the number of children above certain threshold values was calculated. The United States Centers for Disease Control and Prevention (CDC) currently utilizes a reference level of 5 µg/dL for lead in children (Centers for Disease Control and Prevention, 2017). This level represents the 97.5th percentile of NHANES evaluations and was lowered from the previous reference level of 10 µg/dL in 2012 (Centers for Disease Control and Prevention, 2017). In this study, the portion of children (0–14 years old) above each of these thresholds was calculated using country specific age estimates provided by the Institute for Health Metrics and Evaluation (IHME) and the mean and standard deviations calculated as described above (Forouzanfar et al., 2016). In the case of one country, Kosovo, IHME does not provide population estimates. Here the age distribution information from Serbia was used in combination with a total population estimate provided by the World Bank (World Bank, 2019). Given that BLLs tend to be log-normally distributed, the distribution of a population above a threshold was calculated in Microsoft Excel 2018 using the following syntax (Microsoft Corporation, 2018):

=1-(NORM.DIST(LN([threshold]),LN([mean]),LN([sd]),TRUE)).

2.6 Sensitivity Testing

Sensitivity testing was done to assess the relative influence of imputation methods using the leave one out approach. In this case, imputation methods were reviewed sequentially, with subsamples associated with each method successively removed or reinserted in each run. Thus subsamples with arithmetic means imputed with eq. 1 were removed in the first run, though reinserted in the second, when those imputed with eq. 2 were removed and so on. Variability in the results was assessed through a series of paired t-tests against the full dataset. Sensitivity testing was carried out on adult and child background BLL subsamples only.

2.7 Statistical Analysis

Data were aggregated, organized, and analyzed in Microsoft Excel 2018 (Microsoft Corporation, 2018). The pooling of BLLs using method 1, from Ericson, *et al* (2018), and t-tests were done using Stata 15.1 statistical software (StataCorp. LP, 2017).

3. Results

3.1 Results of the Literature Review

The PubMed search involved the names of 137 unique countries and returned 10,164 article listings. Of these, 9,189 were excluded during the title review of PubMed listings, leaving 950 studies. During abstract review and data extraction an additional 473 studies were excluded. The final data set for analysis therefore included 477 studies with 979 sampled populations (i.e. subsamples) with a total population of 702,069 people from 50 countries. Table 1 summarizes the results of the literature review by country.

As mentioned above, 473 studies were excluded during the abstract review and data extraction stage. Of these, 335 were excluded based on the criteria given above and here:

1) Did not contain BLL data from human populations residing in [country] (n=72);

    - 46 animal studies;
    - 24 studies were carried out on human populations residing in high income countries (e.g. South Asian populations living in Toronto; search for China returned Taiwan, a high-income country);
    - 2 studies were carried out in a different LMIC than the one searched. Both were conducted in Pakistan but were returned for searches in both China and Pakistan. Those returned for China, were coded as excluded. Those returned for Pakistan were included.
2) Was not comprised of at least 30 participants (n=26);
3) Did not present BLL data derived from venous, capillary, or umbilical cord samples of whole blood (serum and plasma samples were excluded; n=159);
    - 59 studies did not contain blood samples of any kind;
    - 63 studies analyzed samples of serum or plasma;
    - 37 studies did not contain analysis of lead concentrations.
4) Data were not collected after 2005 (n=23);
5) Not published in English (n=55).

In addition to those criteria listed above, a further 138 studies were excluded for reasons other than those stated in the initial inclusion criteria. Of these, 59 reported subsamples previously reported elsewhere (e.g. meta-analysis; reuse of data for further analysis). An additional 36 provided insufficient descriptive statistics, most commonly with no indication of the central tendency of the data (e.g. number or percent of samples above a threshold) or no indication of sample size. Seventeen titles were retrieved via PubMed that could not be located or accessed by the reviewers in full online. Six studies were excluded because the units could not be readily converted to µg/dL (e.g. reported as mass per mass) and 18 studies were excluded because of serious quality control concerns with data analysis and presentation. Finally, 18 studies presented data on subgroups selected on the basis of a biological predisposition to higher BLLs, such as renal failure patients on hemodialysis. In 16 of these cases, the affected subsample was excluded and an acceptable subsample from the same study was included. In 2 cases, the studies were excluded entirely.

With regard to the study language criteria (inclusion criteria # 5), PubMed returned all abstracts meeting the specified search terms including those translated from other languages. Given that the English language was not specified as part of the search, PubMed returned a number of translated abstracts in addition to studies published entirely in English. In an effort to include a maximal number of studies, translated abstracts with sufficient detail to meet the other criteria were included. In the case of one country, Colombia, 5 titles were returned in total, with 3 being in English and 2 being in Spanish. In this case, the subsamples presented by the Spanish language articles nearly doubled the size of the background sample for this country. Given the relatively small number of available studies for Colombia and proportional benefit of using this data, the Spanish language studies were included. A similar situation was potentially presented with studies from one other country, Russia. Here, 3 studies were amenable to inclusion in the present analysis, though 8 additional studies published in Russian were returned by PubMed. In this case, the full articles could not be accessed online and were thus excluded.

3.2 Results of Data Extraction

In total, suitable data was extracted from 477 studies with 979 subsamples from 50 countries. Of these, studies from 32 countries with 295 subsamples presented background BLLs for children, while 36 with 305 subsamples presented background BLLs for adults. Studies of non-occupationally exposed children were available in 20 countries (117 subsamples), while studies of non-occupationally exposed adults were available in 16 countries (97 subsamples). Finally, studies of occupationally exposed persons, including 6 child subsamples from Egypt and one from Pakistan, were available in 26 countries (179 subsamples).

'Probable' sources of exposure were identified for 467 subsamples. An additional 83 subsamples had subsamples associated with 'possible' sources of exposure. Of the 968 subsamples, 44 % (n=429) did not identify a source of exposure. Only probable sources of exposure were used in the analysis of attributable sources here.

Following data extraction, probable sources were grouped into 19 different categories (Table 2). The category 'other' was comprised of a wide range of sources, including cosmetics, opium use, and traffic. The category 'diet' was comprised of sources from contaminated foodstuffs, spices and water, as well as contamination from non-ceramic cookware or containers. Table 2 summarizes the number of subsamples reporting each probable exposure source. Exposure sources sorted by country are available in the supplementary materials.

3.3 Results of Estimates for Missing Data

Of the 979 subsamples 728 published arithmetic mean BLLs. In 137 cases, the arithmetic mean was not available and geometric mean was used. Where neither arithmetic or geometric means were reported the arithmetic mean was imputed using the methods described above from eq. 1 (n=68) or eq. 2 (n=33). Finally, when the available data were insufficient to apply any of the above equations to impute the mean values, the median value was used in 13 cases.

With regard to populating the standard deviation, 623 of 953 subsamples reported standard deviations. In 45 subsamples, where the standard deviation was not published, but the SEM was available, the SEM was multiplied by the square root of the sample size to calculate standard deviation (Higgins & Green, 2008). To calculate the remaining 311 standard deviations, the following methods were used: eq. 3 (n=6); eq. 4 (n=74); or eq. 5 (n=59). Finally, when the data required to impute the standard deviation using any of the above methods were not available, the sample weighted SEM from other studies (eq.6) was used in 172 cases.

3.4 Pooled Mean Results

Background BLLs for children and adults were pooled using two different methods. In the first method a random effects meta-analysis weighting by standard error was conducted resulting in a pooled mean and 95% confidence intervals for each country. In the second, presented by Fewtrell, *et al.* (2003), sample-weighted mean BLLs and standard deviations were calculated for each subgroup. In both cases there was not a significant difference between the results of the two methods ($p < 0.05$). Table 3 presents the results using Fewtrell *et al.'s* (2003) method organized by country. The results of first method, the random effects meta-analysis, are available in the supplementary materials. Figure 1 presents pooled mean background BLLs and standard deviations for children in the 32 countries where data were available. Results for non-occupational and occupational subgroups are available in the supplementary materials as are the results of the random effects method. Blood lead levels were also pooled by exposure setting and type (background, non-occupational, occupational) for both children and adults (Table 4).

3.4 Children above threshold values

Assuming a log-linear distribution of BLLs in a given population, approximately 670 million children age 0–14 years old were estimated to have BLLs exceeding the CDC reference level of 5 µg/dL. Of those, 436 million were estimated to exceed the previous reference level of 10 µg/dL. Table 5 presents the results by country.

3.4 Results of Sensitivity Testing

Paired t-tests of the full dataset and versions with sequentially removed imputed values did not result in significant differences in the findings ($p > 0.05$).

4. Discussion

4.1 Significant findings

This systematic review is the largest yet carried out on BLLs in LMICs. Previously, the most comprehensive study was that conducted by IHME as part of their annual disease burden estimates. In their most recent disease burden calculation 88 studies from 2010 to the 2017 were included for the purpose of calculating BLLs (IHME, 2018). By contrast, this study extended this period by six months to June 2018 and reviewed more than five times the number of studies (n=477). The inclusion of more studies allows for both a more comprehensive assessment of exposure sources as well as likely more informed national BLL estimates.

Two findings of the review stand out as most significant. The first is that BLLs remain elevated in LMICs, most importantly in children. Lead exposure can result in a number of adverse health outcomes, including neurological decrement, cardiovascular disease, kidney disease, decreased immunological resistance, and adverse developmental outcomes, among other effects (US EPA, 2013). Humans are most commonly exposed to lead through ingestion. Inhaled lead particles are typically too large to penetrate the lungs and migrate via the mucociliary elevator to the esophagus where they are ultimately ingested (ATSDR, 2007). In children up to 50% of ingested lead in absorbed into the blood, confused by the body as the calcium ion Ca^{2+} (ATSDR, 2007; Flora et al., 2012). In the blood, lead can pass the blood-brain barrier and impede brain growth (ATSDR, 2007). Accordingly, adults exposed to lead as children tend to have less grey matter, the part of the brain associated with decision making, than their non-exposed counterparts (Cecil et al., 2008).

Even low levels of pediatric lead exposure are associated with Intellectual Quotient (IQ) decrement (Lanphear et al., 2005; Schwartz, 1994; US EPA, 2014). A 2013 international pooled analysis found the loss of a single IQ point at blood lead concentrations as low as 0.1–1 µg/dL in school age children (5–10 years) (Budtz-Jørgensen et al., 2013). Further, the study found that neurological impacts continued through higher levels of exposure, though with proportional impacts waning as concentrations increase (Budtz-Jørgensen et al., 2013). There are also indications that adverse neurological consequences may continue to accrue through adulthood. A 2017 study of New Zealanders found that elevated BLLs at age 11 were associated with IQ decreases later in life when the same children were assessed 27 years later as adults (Reuben et al., 2017).

Lead-attributable neurological decrement impacts the life outcomes of individuals. Lead-exposed children are less likely to pursue advanced education, and more likely to have an unwed pregnancy or commit violent crime (Gould, 2009; Mielke and Zahran, 2012; Nevin, 2000). At a societal level these impacts can be significant. Evaluating the economic impact of lead exposure, a 2002 CDC study of the US found that an additional USD 110–319 billion would be added to the lifetime earnings for each cohort of two-year-old children born after the leaded petrol phase-out, compared with children born before the phase-out (Grosse et al., 2002). In LMICs, Attina and Trasande (2013) found that nearly USD 1 trillion in lifetime earnings were lost as a result of lead exposure (1.88–4% of GDP in the countries studied).

In adults, cardiovascular disease is the largest lead attributable health outcome in burden of disease estimates. IHME estimates that perhaps 1 million deaths each year result from heart disease attributable to lead exposure globally (IHME, 2018). Most of these deaths (> 80 %) occur in people above age 60 years (IHME, 2018). Nawrot, et. al (2002) found that < 40 µg/dL, a doubling in BLL was associated with increases in systolic and diastolic blood pressure of 1.0 mmHg and 0.6 mmHg, respectively. Thus, similar to neurological decrement, the highest proportional impacts appear to be at lower levels of exposure.

In this study more 670 million children were estimated to have BLLs exceeding CDC reference level of 5 µg/dL, itself well above concentrations associated with lifelong adverse health outcomes. A further 436 were estimated to have BLLs above 10 µg/dL. Importantly, these estimates were calculated based on subsamples coded as 'background' exposure scenarios. In most cases these subsamples were chosen in studies as control populations specifically because they represented an absence of exposure. By excluding non-occupational exposures in the pooling of BLLs, this study therefore presents a more conservative indication of lead toxicity. Future studies might use the results presented here to develop more robust estimates of the lead attributable social, economic and health outcomes.

The second main finding is that the sources of exposure appear to be distinct from those in high income countries. Elevated BLLs in high income countries have been most commonly attributed to tetraethyl lead used in petrol or lead-based enamel paints used in residential settings (Jacobs et al., 2002; Needleman, 2004; Nriagu, 1990). In both high and low income countries, leaded petrol has been phased out, with few exceptions (UNEP, 2016). The last country in this analysis to phase out the use of leaded petrol was the Democratic Republic of Congo, which did so in 2009, preceded by Indonesia in 2006 (ACFA, 2007; Tuakuila et al., 2013). Other African countries had largely phased out leaded petrol by the close of 2005 (Lean, 2006). Phase-outs in the remaining countries generally occurred several years earlier. China and India phased out leaded petrol in 2000, for instance (Singh and Singh, 2006; Wan et al., 2014). Following leaded petrol phase-outs, BLLs decline precipitously, owing in part to the short half-life of lead in blood of about 30 days (Annest et al., 1983; ATSDR, 2007; Singh and Singh, 2006). Fewtrell, et al. (2004) note that population wide BLL decreases following phase outs range from 30–48 %. Thus, this study's use of BLLs assessed after 2005 controlled somewhat for the influence of leaded petrol.

Importantly, residual lead in surface soils resulting from aerial deposition from petrol is highly recalcitrant (Mielke et al., 2010; Semlali et al., 2004; Zahran et al., 2013). Therefore this inclusion criterion is only intended to control for exposure to airborne lead exhausted from vehicles.

With regard to lead-based paint, relatively few LMICs have legislatively banned this source of exposure for use in residential settings (UNEP, 2017). However, despite the absence of bans in most countries, lead-based paint does not appear to be a major source of lead exposure in LMICs. Indeed lead exposure in LMICs seems to be dominated by industrial sources of lead, accounting for > 65 % of exposure sources identified in background subsamples and > 64 % in non-occupational subsamples. This conclusion is similarly supported by a recent assessment of Jakarta homes and preschools that revealed a low exposure risk despite lead-based paint being readily available in stores (Ericson et al., 2019).

The results of the study indicate a pressing need for interventions to reduce lead exposure in LMICs. Several possible recommendations follow. The first such recommendation is to improve BLL surveillance. Nationally representative studies to determine background BLLs could greatly improve the accuracy of the estimates presented here. Support could be provided to execute efforts akin to the US NHANES program. In the absence of these more comprehensive efforts, academic studies could provide additional insight. Of the 137 countries included in this study, data on children's background BLLs were only available in 32. Thus researchers could be supported in the non-represented countries to carry out background BLL assessments, particularly of children. Further these studies, as well as those currently being executed, could be leveraged into a more coherent approach. An international BLL registry could be established and housed in either an academic institution such as IHME or international organization, like the World Health Organization. Researchers could be encouraged to register the anonymized results of their studies into the database and follow basic quality assurance and control methods. Given that the vast majority of BLL testing is conducted by individual academics, such an effort could greatly improve current knowledge.

A second recommendation relates to the international response to the issue. First, existing resources should be augmented to deal with lead exposure in LMICs. At present the limited international response is incongruent with the enormity of the problem. Indeed in the most recent funding cycle (2014) the Global Environment Facility, the largest multi-lateral environmental donor, distributed just USD 1.8 million in grants to mitigate lead-based paint exposures, of a USD 4.4 billion budget (GEF, 2014a, 2014b). Second, interventions should be tailored to the better suit the *de facto* nature of exposure sources in LMICs. Current lead exposure mitigation efforts, such as the Global Alliance to Eliminate Lead Paint, tend to focus on the exposure risk of lead-based paint, which does not seem to be as significant a contributor to BLLs as other sources (WHO, 2017). International donors and executing agencies should be encouraged to develop capacity and funding streams to mitigate the sources with a confirmed exposure risk, such as informal lead recycling.

4.2 Limitations

The study has a number of limitations. Most significantly, only one reviewer (BE) extracted data from all studies. While all studies were reviewed by at least one other reviewer, no single reviewer duplicated the review in its entirety. This introduces the possibility that errors may have been included in the final dataset.

A second limitation is that studies were not assessed for bias. The large volume of studies both diminished the relative statistical influence of any single report and made a systematic review of bias logistically challenging. Nevertheless, failing to include an assessment of bias could adversely affect the results.

In total 640 of 979 subsamples were published with arithmetic mean and either standard deviation or SEM. Missing data were imputed for the remaining 339 subsamples. While the utilized methods have been subject to robust sensitivity testing elsewhere, and have performed reasonably well on skewed data like that presented by BLLs, the reliance on imputation introduces some uncertainty into the results.

A separate limitation is the study's reliance on the English language literature. While English continues to be used in the majority of peer-reviewed publications, the exclusion of other languages here likely results in significant gaps, as has been observed elsewhere (Amano et al., 2016).

A final set of limitations relates to the pooling of studies to calculate a nationally representative BLL value. The data presented in the reviewed studies were in most cases not collected for this purpose and typically represent a small geographic area or subsection of the population. The relative influence of the results of any once study is somewhat diminished when pooled with others, however in several cases only a single subsample was available for each subgroup in a given country. In 13 of the 33 countries for which data were available on children for instance, only a single subsample could be identified. Thus the nationally representative BLL for those countries presented here was derived from a single study in each country. A related limitation concerns a similar if opposite issue. In China, India, and Mexico the larger number (142, 40 and 32, respectively) and broad geographic representation of studies could have facilitated more nuanced and geographically granular estimates. Such an effort was not attempted here.

5. Conclusion

This study conducted the most comprehensive literature review of BLLs in LMICs to date. The review of 477 studies revealed that BLLs remain elevated in LMICs despite leaded petrol phase-outs. The primary sources of exposure appear to be industrial in nature. Approximately 670 million children were found to have BLLs exceeding the CDC reference dose of 5 μg/dL in the studied countries. Given the lifelong adverse impacts of lead on the developing brain and other biological systems, urgent attention is required to mitigate the worst exposures. To better monitor BLLs in LMICs an international registry should be established to house the anonymized results of testing carried out by researchers.

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Table 1. Results of Literature Review

Country	World Bank income group	n Titles Returned by PubMed	n Abstracts Reviewed	n Studies Included in Analysis	n Subsamples Included in Analysis	Total population of reviewed subsamples
China	UMIC	3254	317	142	266	381,992
Mexico	UMIC	335	79	42	87	23,193
India	UMIC	1130	59	40	93	229,744
Brazil	LMIC	892	62	34	55	13,057
Pakistan	LMIC	202	56	31	106	10,597
Iran	UMIC	701	47	26	48	4,696
Turkey	UMIC	784	49	16	36	2,533
Nigeria	LMIC	133	46	16	27	3,574
Egypt	LMIC	237	26	15	39	2,424
Bangladesh	LMIC	55	19	11	17	4,613
Armenia; Benin; Bolivia; Bosnia and Herzegovina; Cameroon; Colombia; Congo, Dem Rep; Ecuador; Ethiopia; Ghana; Grenada; Indonesia; Iraq; Jamaica; Jordan; Kenya; Kosovo; Lebanon; Macedonia, FYR; Malaysia; Mongolia; Morocco; Nepal; Paraguay; Peru; The Philippines; Romania; Russian Federation; Senegal; Serbia; South Africa; Sri Lanka; Sudan; Tanzania; Thailand; Uganda; Ukraine; Vietnam; West Bank and Gaza Zambia		1,644	184	104	195	25,646
Afghanistan; Albania; Algeria; American Samoa; Angola; Azerbaijan; Belarus; Belize; Bhutan; Botswana; Bulgaria; Burkina Faso; Burundi; Cabo Verde; Cambodia; Central African Republic; Chad; Comoros; Congo, Rep ; Costa Rica; Cuba; Djibouti; Dominica; Dominican Republic; El Salvador; Equatorial Guinea; Eritrea; Fiji; Gabon; Gambia, The; Georgia; Guatemala; Guinea; Guinea-Bissau; Guyana; Haiti; Honduras; Ivory Coast; Kazakhstan; Kiribati; Korea, Dem People's Rep ; Kyrgyz Republic; Lao PDR; Lesotho; Liberia; Libya; Madagascar; Malawi; Maldives; Mali; Marshall Islands; Mauritania; Mauritius; Micronesia, Fed Sts ; Moldova; Montenegro; Mozambique; Myanmar; Namibia; Nauru; Nicaragua; Niger; Papua New Guinea; Rwanda; Samoa; São Tomé and Principe; Seychelles; Sierra Leone; Solomon Islands; Somalia; South Sudan; St Lucia; St Vincent and the Grenadines; Suriname; Swaziland; Syrian Arab; Republic; Tajikistan; Timor-Leste; Togo; Tonga; Tunisia; Turkmenistan; Tuvalu; Uzbekistan; Vanuatu; Venezuela, RB; Yemen, Rep ; Zimbabwe		797	6	0	0	0
		10,164	950	477	969	702,069

LMIC: Lower Middle Income Country; UMIC: Upper Middle Income Country

***Table 2**. Sources of lead (Pb) exposure in background, non-occupational and occupational settings reported in the 467 subsamples with probable sources of exposure.*

Exposure source	n subsamples	Total sample size	n background subsamples	n non-occupational subsamples	n occupational subsamples
automobile repair	18	895	2	2	14
battery manufacture or recycling	116	11,863	24	17	76
bullets	6	1,166	2	3	1
ceramics	13	1,485	5	5	4
contaminated site	6	3,796	0	6	0
diet	16	2,963	11	5	0
dumpsite	6	487	2	3	2
ewaste	38	10,548	6	24	8
industry (lead)	8	836	0	4	4
industry (other)	47	3,327	23	6	18
lead-based paint	7	940	4	0	3
mining	35	5,650	1	33	6
other	25	6,257	9	9	6
petrol	11	1,527	8	0	3
smelting	56	22803	13	22	22

***Table 3**. Pooled mean BLLs for background adults and children in the 44 LMICs where data were available.*

Country	Background children				Background adults		
	n subsamples	total sample size	pooled mean BLL (sd) (μg/dL)		n subsamples	total sample size	pooled mean BLL (sd) (μg/dL)

Bangladesh	9	3,058	7.66 (6.2)		4	1,290	3.9 (3.53)
Benin	2	888	5.18 (5.47)		2	287	4.95 (4.54)
Brazil	10	3,821	1.49 (2.16)		21	6,657	3.18 (5.08)
Cameroon	1	147	8.7 (3.9)		0	0	--
China	103	301,448	5.22 (3.72)		67	57,504	3.73 (4.79)
Colombia	3	713	4.94 (4.55)		3	833	2.49 (6.15)
Democratic Rep. of the Congo	6	432	8.08 (5.74)		1	157	9.1 (8.74)
Ecuador	1	69	3.17 (4.24)		0	0	--
Egypt	13	1,425	9.07 (5.23)		15	757	17.53 (5.3)
Ethiopia	1	132	1.66 (4.24)		3	208	3.92 (5.56)
Ghana	0	0	--		4	110	1.45 (4.58)
Grenada	0	0	--		1	52	1.17 (4.24)
India	19	5,641	5.49 (5.8)		35	221,713	4.43 (5.46)
Indonesia	1	108	6.4 (2)		0	0	--
Iran	7	828	3.65 (3.85)		13	1,159	6.04 (3.81)
Iraq	1	207	5.3 (1.9)		2	666	8.47 (4.53)
Jamaica	3	325	2.6 (4.09)		2	130	0.83 (2.81)
Jordan	0	0	--		3	163	3.61 (3.7)
Kosovo	3	250	2.52 (2.4)		2	165	4.79 (2.89)
Lebanon	0	0	--		1	55	3.57 (0.25)
Macedonia, FYR	0	0	--		2	119	2.41 (1.72)
Malaysia	0	0	--		1	136	2.6 (2.1)
Mexico	42	4,196	3.54 (4.06)		22	5,769	3.3 (3.95)
Mongolia	0	0	--		1	100	3.1 (4.24)
Morocco	12	770	4.57 (4.8)		3	107	4.88 (6.33)
Nepal	1	312	6.69 (4.22)		5	173	4.44 (5.08)
Nigeria	5	818	8.99 (6.22)		8	485	8.1 (1.7)
Pakistan	33	2,461	10.14 (3.76)		45	4,656	11.73 (4.52)
West Bank and Gaza	1	178	3.2 (2.4)		0	0	--
The Philippines	1	2,860	6.4 (8.57)		0	0	--
Romania	1	84	3.77 (1.96)		0	0	--
Russia	3	599	3.18 (4.13)		2	30	1.56 (0.52)
Senegal	1	32	8.22 (3.16)		1	52	6.51 (2.92)
Serbia	0	0	--		3	119	1.27 (0.82)
South Africa	3	1,760	7.21 (5.03)		4	1,522	1.55 (2.67)
Sri Lanka	0	0	--		2	100	8.71 (5.47)
Sudan	0	0	--		1	15	0.39 (1.25)
Tanzania	1	43	2.26 (0.96)		1	24	4.71 (1.62)
Thailand	4	1,502	5.79 (4.24)		6	2,025	5.26 (4.34)
Turkey	1	224	4.23 (1.31)		16	781	2.87 (2.86)
Uganda	2	263	6.68 (5.23)		0	0	--
Ukraine	0	0	--		2	102	3.28 (3.61)
Vietnam	1	311	4.97 (5.5)		1	51	3.9 (2.2)

***Figure 1**. Pooled mean background BLLs for children in the 32 countries where data were available.*

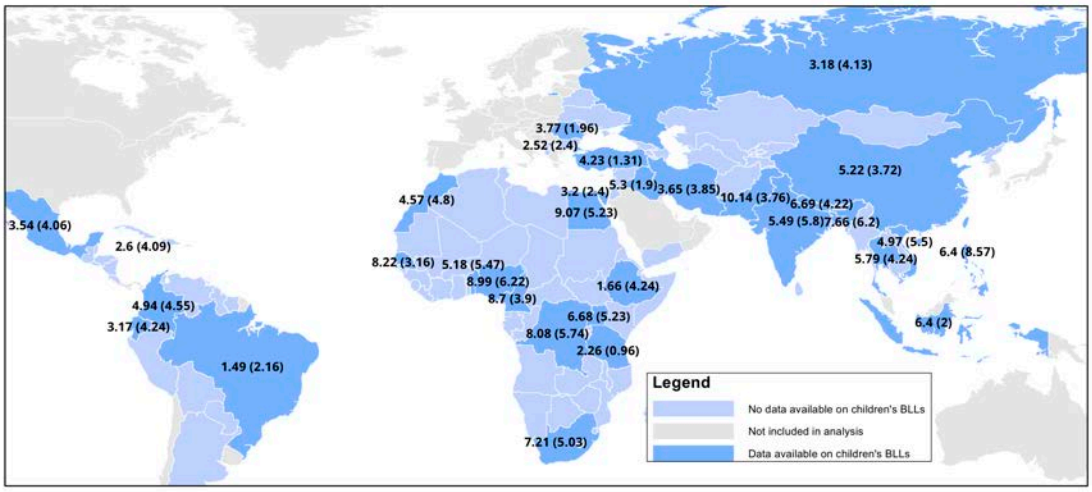

***Table 4**. Pooled mean BLLs (standard deviation) (µg/dL) by probable exposure source and setting in the 477 studies reviewed.*

Exposure source	Background children	Non-occupational children	Background adults	Non-occupational adults	Occupational
automobile repair	--	32.35 (5.1)	17.41 (5.72)	--	21.57 (7.64)
battery manufacture or recycling	9.21 (4.02)	11.6 (6.65)	14.01 (7.46)	22.8 (8.97)	27.39 (8.09)
ammunition (shooting ranges and bullet wounds)	4.66 (4.24)	--	5.51 (4.24)	8.73 (8)	6.2 (2.28)
ceramics	5.03 (4.49)	32.68 (12.34)	3.93 (3.88)	--	28.73 (9.74)
contaminated site	--	7.47 (4.78)	--	10.8 (7.84)	--
diet	3.68 (5.89)	24.3 (12.71)	4.84 (4.22)	18 (9.01)	--
dumpsite	6.68 (5.23)	17.46 (7.33)	--	10.27 (7.29)	9.24 (3.96)
ewaste	6.28 (4.2)	7.88 (5.39)	5.5 (4.3)	16.9 (8.72)	19.86 (8.38)
industry (lead)	--	7.82 (7.66)	--	7.29 (8.16)	25.47 (4.33)
industry (other)	6.26 (4.5)	9.97 (6.32)	5.09 (4.43)	--	11.93 (5.11)
lead-based paint	6.67 (5.84)	--	--	--	10.79 (4.45)
mining	--	22.08 (8.53)	1.7 (0.7)	17.71 (7.04)	15 (6.28)
other	3.63 (5.32)	30.64 (9.71)	31.77 (7.49)	19.52 (8)	6.74 (6.95)
petrol	6.12 (5.2)	--	9.1 (8.74)	--	17.32 (6.29)
smelting	6.35 (7.27)	5.28 (7.61)	5.15 (6.25)	15.59 (8.6)	41.01 (10.59)
tobacco products	3.03 (5.09)	10.77 (3.72)	6.35 (4.07)	22.78 (2.41)	--

***Table 5**. Estimated number of children (age 0–14 years) exceeding 5 and 10 µg/dL in countries covered by the analysis.*

Country	Pooled BLL (µg/dL)	Children > 5 µg/dL	Children > 10 µg/dL
Bangladesh	7.66 (6.2)	28,045,623	20,922,029
Benin	5.18 (5.47)	2,349,421	1,614,950
Brazil	1.49 (2.16)	2,798,944	327,108
Cameroon	8.7 (3.9)	6,574,477	4,588,727
China	5.22 (3.72)	119,634,420	72,345,333
Colombia	4.94 (4.55)	5,833,789	3,767,446
Democratic Rep. of the Congo	8.08 (5.74)	21,698,682	16,106,832
Ecuador	3.17 (4.24)	1,767,414	1,001,718
Egypt	9.07 (5.23)	19,198,203	14,278,524
Ethiopia	1.66 (4.24)	9,184,389	4,410,256
India	5.49 (5.8)	197,328,888	138,773,472
Indonesia	6.4 (2)	45,511,423	18,502,284

Iran	3.65 (3.85)	7,574,440	4,222,892
Iraq	5.3 (1.9)	8,001,759	2,407,230
Jamaica	2.6 (4.09)	223,444	117,815
Kosovo	2.52 (2.4)	63,591	16,899
Mexico	3.54 (4.06)	14,116,167	8,038,249
Morocco	4.57 (4.8)	4,462,849	2,887,454
Nepal	6.69 (4.22)	5,425,152	3,648,335
Nigeria	8.99 (6.22)	50,354,198	38,359,290
Pakistan	10.14 (3.76)	46,652,240	33,446,094
West Bank and Gaza	3.2 (2.4)	574,214	181,692
The Philippines	6.4 (8.57)	17,638,390	13,500,546
Romania	3.77 (1.96)	1,025,305	223,837
Russia	3.18 (4.13)	8,485,954	4,744,149
Senegal	8.22 (3.16)	4,402,364	2,853,029
South Africa	7.21 (5.03)	8,582,920	6,109,888
Thailand	5.79 (4.24)	6,461,120	4,214,807
Turkey	4.23 (1.31)	5,291,849	14,236
Uganda	6.68 (5.23)	10,760,426	7,627,385
Vietnam	4.97 (5.5)	10,768,832	7,361,977
Total		**670,790,886**	**436,614,484**

2.2. Paper Two

Dowling, R., Caravanos, J., Grigsby, P., Rivera, A., **Ericson, B.,** Amoyaw-Osei, Y., ... & Fuller, R. (2016). Estimating the prevalence of toxic waste sites in low-and middle-income countries. *Annals of Global Health*, *82*(5), 700-710

This study utilized a limited dataset on contaminated sites in Ghana to calculate an estimate of the total number sites across the whole country. Overall, there are limited data on lead exposures in LMICs, though the contribution of informal industrial sources is expected to be significant. Given that lead has low mobility in the environment, sites that have become contaminated can potentially pose a persistent risk to nearby populations. Thus, the contribution of contaminated sites, or ‘hotspots,’ is expected to comprise a major source of human lead exposure in LMICs.

This study uses cluster random sampling methods to extrapolate data from a limited set of geographical areas (quadrats) across Ghana as a whole. The result is a method that could be deployed elsewhere to estimate the number of sites in areas where limited resources inhibit site inventories. Such an exercise was conducted in Chapter 3 (Paper 6) in calculating the number of informal battery processing locations in LMICs and attributable burden of disease.

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ORIGINAL RESEARCH

Estimating the Prevalence of Toxic Waste Sites in Low- and Middle-Income Countries

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Abstract

BACKGROUND Exposure to heavy metals at contaminated industrial and mining sites, known also as hot spots, is a significant source of toxic exposure and adverse health outcomes in countries around the world. The Toxic Sites Identification Program (TSIP) developed by Pure Earth, a New York—based nongovernmental organization, is the only systematic effort to catalogue contaminated sites globally. To date, TSIP has identified and catalogued 3282 sites in low- and middle-income countries. The TSIP methodology is not designed to survey all contaminated sites in a country. Rather sites are prioritized based on their perceived impact on human health, and only a limited number of the most highly hazardous sites are surveyed. The total number of contaminated sites globally and the fraction of contaminated sites captured by TSIP is not known.

OBJECTIVE To determine the TSIP site capture rate, the fraction of contaminated sites in a country catalogued by TSIP.

METHODS Ghana was selected for this analysis because it is a rapidly industrializing lower middle income country with a heterogeneous industrial base, a highly urban population (51%), and good public records systems. To develop an estimate of the fraction of sites in Ghana captured by TSIP, assessors targeted randomly selected geographic quadrats for comprehensive assessment using area and population statistics from the Ghana Statistical Service. Investigators physically walked all accessible streets in each quadrat to visually identify all sites. Visual identification was supplemented by field-based confirmation with portable x-ray fluorescence instruments to test soils for metals. To extrapolate from survey findings to develop a range of estimates for the entire country, the investigators used 2 methodologies: a "bottom-up" approach that first estimated the number of waste sites in each region and then summed these regional subtotals to develop a total national estimate; and a "top-down" method that estimated the total number of sites in Ghana and then allocated these sites to each region. Both methods used cluster random sampling principles.

FINDINGS The investigators identified 72 sites in the sampled quadrats. Extrapolating from these findings to the entire country, the first methodology estimated that there are 1561 sites contaminated by heavy metals in Ghana (confidence interval [CI]: 1134-1987), whereas the second estimated 1944 sites (CI: 812-3075). The estimated total number of contaminated sites in Ghana is thus 7-9 times the number of sites captured through TSIP. On a population basis, it was estimated that there are between 31 and 115 contaminated sites per million inhabitants in Ghana.

CONCLUSIONS The findings of this study indicate that the TSIP methodology provides a sound statistical basis for policy formulation. The statistical approaches used in this study can be replicated in other countries to improve estimates of the prevalence of contaminated sites. This information provides

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important input to calculations of the global burden of disease attributable to hazardous exposures at contaminated sites.

KEY WORDS contaminated sites, pollution, global burden of disease, heavy metals, field survey, Toxic Sites Identification Program (TSIP)

INTRODUCTION

Pollution is critically linked to poverty and produces disproportionate health effects on low-income communities worldwide.[1] On a global scale, low- and middle-income countries face barriers to infrastructural and inclusive economic development as a result of challenges borne from pollution and associated environmental health issues.[2,3] Rapid industrialization, population growth, and exploitation of natural resources have potentially resulted in significant environmental degradation in many low- and middle-income countries (LMICs).[4] These countries often have limited governmental capacity and few incentives to formally regulate environmentally damaging industries or address contaminated sites. Even in cases where regulations exist, the capacity to manage or enforce laws can be limited.

Contamination by toxic chemicals and heavy metals presents a unique and ongoing problem. Adverse health effects from chemical contamination, through processes such as informal lead-acid battery recycling, natural resource extraction, and electronic waste recovery and disposal, often go unnoticed in part because of the latency and chronic nature of environmental toxicants.[5,6] Limited regulatory policies as well as the ubiquity of small-scale informal practices make identifying active and legacy contaminated sites a challenge for intervening policy and health care professionals.

Lack of best practices in unregulated and small-scale industries often leads to an increased risk of exposure to toxicants. Artisanal small-scale gold (ASGM) mining, for example, typically involves panning gold-containing alluvial soils or crushed ores with elemental mercury (Hg). This mercury-gold amalgam is then heated, which drives off the mercury as a vapor and leaves behind both gold and some residual mercury.[7] Large amounts of Hg vapor recondense and deposit locally and can be re-emitted from water and soil surfaces or can be methylated, bioaccumulate, and biomagnify in food chains.[8,9] Surface soils, water bodies, and sediments are the major biospheric sinks for Hg.[10] In short, artisanal gold mining using elemental mercury poses a significant risk to human health because mercury is a potent neurotoxin and systemic toxin.[11]

In addition to mercury, other metals, including lead (Pb), cadmium, hexavalent chromium, cobalt, and manganese resulting from informal industry can be hazardous to human health. Lead exposure, for example, can lead to cognitive impairment, anemia, hypertension, kidney damage, and, in extreme cases, death.[12] Historically, the source of lead exposure is often traced to gasoline, paint, air, water, interior dust, soil, and food.[13] In areas within the vicinity of mining and industrial establishments, ingestion of soil and dust contaminated with heavy metals is a primary source of lead exposure.[14,15] Lead can also be ingested via drinking water when soluble forms are present in surface or groundwater.[16]

Epidemics of metal poisoning resulting from informal industry have been documented in multiple LMICs, including Senegal, Ghana, Indonesia, and the Philippines.[17 20] A particularly severe case was the recent (2010) tragedy in the Nigerian state of Zamfara resulting from ASGM with lead-laden ore.[21] Acute pediatric exposures there resulted in the deaths of at least 400 children, nearly 25% of who were younger than age 5.[22,23] Two-thirds of households reported processing lead-contaminated gold ore inside family compounds. Soil lead levels in 85% of family compounds exceeded the US Environmental Protection Agency (EPA) action level for areas of bare soil where children play (400 mg/kg).[24]

Despite the significant health burden posed by informal industry, little documentation of sites and exposures exists. Several factors likely inhibit programs, including a lack of available government

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resources in LMICs and a possible institutional reluctance to identify a Pandora's box of problems that can be difficult to resolve.[25]

The only systematic effort to catalogue contaminated sites globally is the Toxic Sites Identification Program (TSIP) developed by Pure Earth, a New York based nongovernmental organization.[1] TSIP uses a rapid risk assessment tool modeled after the EPA's Hazard Ranking System.[26] The protocol, the Initial Site Screening (ISS), requires a site visit and sample collection and relies on the source-pathway-receptor model. The EPA uses a similar, albeit more robust, model to assess contaminated sites in the United States. The EPA Hazard Ranking System requires additional information on spatial attenuation, pollutant persistence, migration potential, and likelihood of future release. By contrast the ISS is indented to be implemented quickly and at low cost. As of December 2015, 3282 waste sites had been identified globally. Of those, an ISS was carried out at 2434 sites in 51 countries. Contaminated sites are identified through several methods, including knowledge and expertise of local staff, investigation of previously identified legacy sites, collaboration with local governments or research organizations, and online nominations. Once a site has been identified, a specially trained investigator conducts an ISS. The completed ISS is entered into an online database and reviewed by the New York office for quality assurance and control.

The TSIP effort has been useful for documenting hazardous waste sites, though it has been somewhat limited by time and resources. As a result, likely only a fraction of the total eligible sites in a given country are being captured. Moreover, the TSIP program likely suffers from selective inclusion and may not be an accurate representation of the distribution of waste sites in any particular country because it is not designed to survey all contaminated sites in a country. Rather sites are prioritized based on their perceived impact on human health, and only a limited number of the most highly hazardous sites are surveyed in each country. Sites are selected through interviews with relevant government agencies, academics, and local leaders.

We undertook the present study to assess the TSIP capture rate, the fraction of all contaminated sites in the country catalogued in TSIP. The West African country of Ghana was selected for the study because of a partnership between Pure Earth and a highly qualified nongovernmental organization in Ghana and a strong relationship with both the Ghana Environmental Protection Agency and Ghana Health Service. Additionally, Ghana regularly collects data and has good public records systems.

Ghana (area 238,535 km^2) is a lower middle income country that is home to 26.8 million residents.[27] The population density is highest in the southern half of the country, where urban centers such as Kumasi and Accra attract more economic opportunities than the rural areas of the north. Chemical production and metals smelting and processing are among the largest contributors to the formal industrial economy. Automotive battery recycling, ASGM using mercury, and scrap metal and electronic waste recycling are the most abundant industries of the informal sector.[28,29]

METHODS

The statistical approaches used in this study seek to estimate the total number of heavy metals contaminated sites throughout Ghana. This estimate is then compared with the number of sites currently catalogued in the TSIP database to obtain a capture rate.

This study makes use of the site identification protocol designed under TSIP to focus on site assessments throughout Ghana's administrative districts. Given that the majority of waste sites in TSIP involve heavy metals such as Hg, lead, chromium, and cadmium, this study focused on estimating sites containing heavy metal contamination.

This study used 2 surrogate methodologies to estimate the total number of heavy metals contaminated waste sites in Ghana. Although individual cases of heavy metal polluting industries within Ghana are well documented (eg, small-scale gold mining and used lead-acid battery recycling), research on the countrywide extent of pollution and potential number of contaminated sites is deficient. Surrogate methods were used because of a lack of resources, limited amount of data, and no clearly appropriate methodology for extrapolating the data to the country. For these reasons, we pursued 2 methodologies in our extrapolation.

The first method (Regional) estimated the regional number of waste sites first and then summed to find a total in Ghana within a particular confidence interval. The second method (Countrywide) estimated the total number of sites in Ghana, then "allocated" them to each region. Both methods use cluster random sampling analytical principles in determining the estimated number of waste sites.

Although this methodology is well known, this is the first instance where it has been applied to waste site estimation.[30]

District Selection, Site Screening, and Inclusion of Toxicants. Ghana is divided into 10 regions that are further subdivided into 216 administrative districts. Two regions, Upper East and Upper West, each represent less than 5% of the country's total population and were not included in data collection. One district from each of the remaining 8 regions was randomly selected for inclusion in the study. The information collected from each district was then used to estimate the number of toxic sites found per region and for the country as a whole. The following 8 districts were randomly selected for inclusion into this study: Amansie West, Tano South, Abura/Asebu/Kwamankese, Afram Plains South, Ningo Prampram, Yendi, Ho, and Juabeso. Districts selected for screening are highlighted in Figure 1.

All population and area data were collected from the Ghana Statistical Service (GSS).[31] GSS conducts censuses every 10 years and presents data in aggregate level by district and region. The data is freely available online.

An onsite waste site identification protocol based on visual assessment was established to effectively identify contaminated sites. Although this methodology targets the sampling in place of randomized techniques, it is important to note the unique conditions in which most toxic sites emerge. Informal mining and recycling of metals and scraps often occur with little regulatory oversight from the government in low-income areas and informal settlements. In addition, legacy sites in industrial centers that pose no risk to human health can stand for long periods without risk management. Randomly sampling throughout a district to confirm toxic site status is not time or cost effective because certain areas such as high-income residential neighborhoods would produce few sites if any. Additionally, randomly sampling in areas where populations are not exposed would not reveal sites within our scope. The visible site identification protocol was developed with these conditions in mind.

The protocol defined 2 levels of toxic site identifiers: primary and secondary. The identifiers were given a scoring rubric including 2 points for primary identifiers and 1 point for secondary identifiers. In order to assess a potentially toxic site, investigators were instructed to review all available identifiers and only begin site assessment on achieving a minimum required score of 6. Site assessments used x-ray fluorescence spectrometry (Innov-x Systems Alpha Series 6500 Handheld XRF Analyzer) to evaluate the level of contamination of various heavy metals.

The list of primary identifiers was composed of the following: documentation of abandoned or legacy site status (ie, confirmed by community or public records), tailings piles, visible particle emissions, and confirmed industry activity. Confirmed industry activity for the purposes of this study could be an active or legacy site used for mining or ore processing, battery recycling, chemical manufacturing, dye industry, electronic waste recycling, heavy industry, industrial or municipal dumpsites, lead mines, lead smelting, pesticide manufacturing, petrochemical industries, product manufacturing, recycling, ship breaking, or tannery operations. The list of secondary identifiers was composed of the following: visibly stressed vegetation, mechanical tools or supplies (including baghouses and filtering equipment), industrial equipment in disrepair, organic olfactory clues, petroleum or other stained soil, and visibly discolored surface water or puddles.

A local team of trained environmental site investigators was assigned to locate and assess various potentially toxic sites throughout the 8 administrative districts. Trained site investigators obtained street maps for each district and worked with local officials to strategize and plan the most effective route for mapping. Individual towns, villages, and neighborhoods were targeted. Highways and main roads between populated areas were overlooked. The purpose of the exercise was to assess contaminated sites that had an apparent pathway for human exposure.

Estimates of the number of sites found per district within these regions were determined after data collection. Statistical analysis was performed with variance and a 95% confidence interval determined for the mean number of toxic sites per district sampled. Standard statistical techniques were used to extrapolate the mean and variance of the sample to the entire country of Ghana.

Although statistical analysis relies on random sampling techniques, the sampling was targeted, which has likely biased our estimates of the number of toxic sites in Ghana away from zero. The size of this bias is difficult to ascertain, but we have chosen this methodology to include the widest possible corridor in our 95% confidence interval to reflect both the variation in the sampling results and the uncertainty regarding the assumption of independence between the samples.

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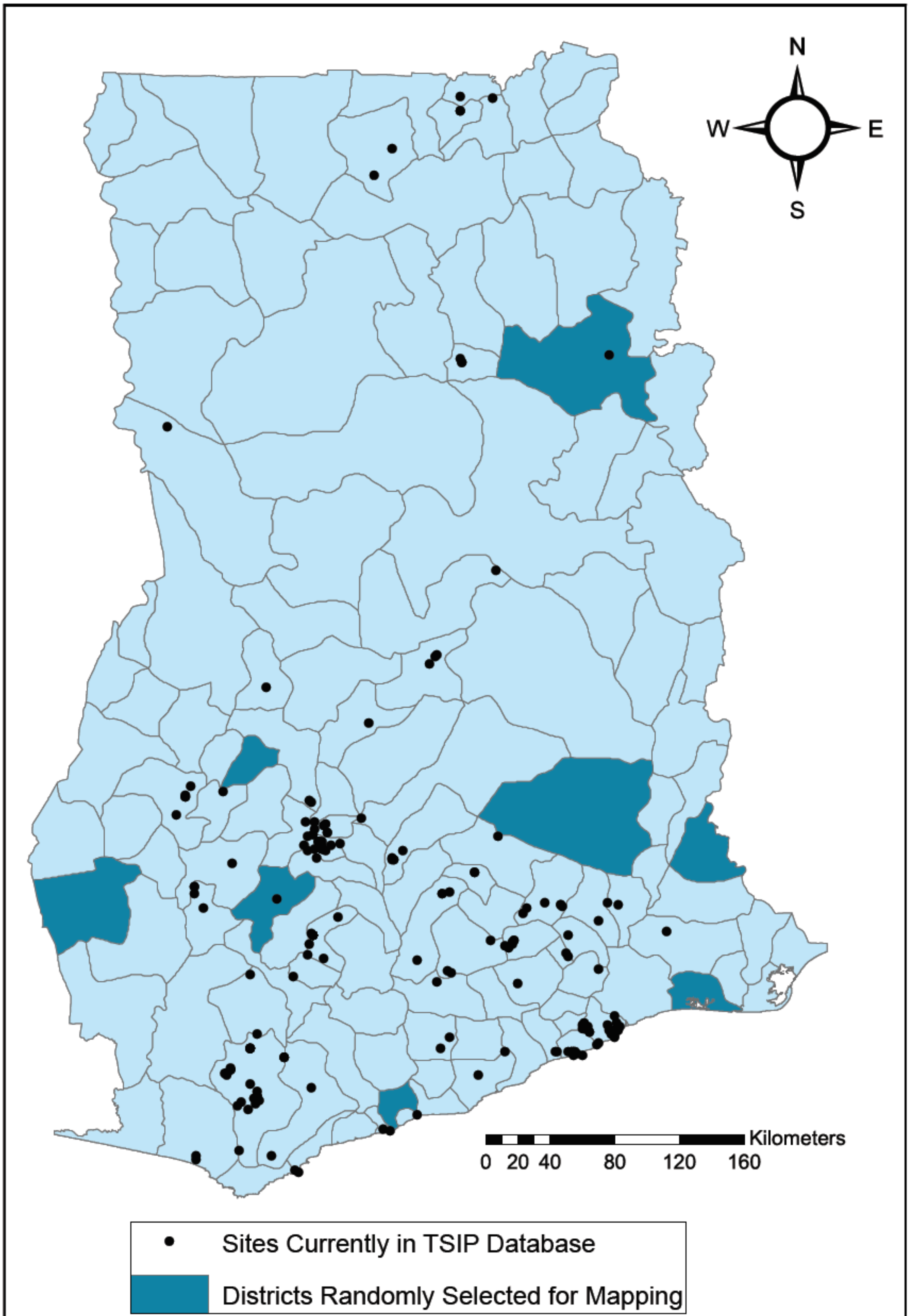

Figure 1. **Map of Ghana. Abbreviations: TSIP, Toxic Sites Identification Program.**

The collected data set was truncated to include only sites where the key pollutant was a heavy metal with credible health impacts and sites where the level of contamination exceeded US EPA standards or equivalent for exposure. Raw data were compared against US EPA standards 3 times to ensure quality control. These criteria narrowed the focus of the dataset to include the toxicants lead, chromium, arsenic, cobalt, and manganese at concentrations that would pose a threat to human health. The 5 heavy metals are each heavily cited in the literature as causing adverse health effects from exposure and intake at relatively low quantities. Mercury, also a key toxicant, was not included in the final data capture because of measuring limitations in the equipment used. The focus on heavy metals also limits the scope of the analysis. Therefore, the results are not an estimate of all contamination countrywide, but rather an estimate solely of heavy metals contamination.

Regional Analysis Methodology. In this approach, the data collected in the 8 randomly chosen districts were extrapolated to estimate the number of toxic sites per region using regional characteristics of area and population. Site estimates by area were calculated using the number of sites found in a sampled district *(x_i)* divided by the total area of the district *($Area\ Dst_i$)* and then multiplied by the total area of the region to which the district belongs *($Area\ Rgn_i$)*. Estimates based on population were calculated in the same manner. The 2 results were averaged to estimate the number of toxic sites expected in each region; see the formula for u later. Table 1 contains information for area and population statistics by region.

Because of budgetary and time constraints, 2 regions of Ghana were not sampled and were selected for exclusion by population weighting. The number of contaminated sites in these regions was estimated using data collected and described earlier for the 8 mapped and sampled regions. An average number of toxic sites in the sampled area of Ghana was found by area and population. These averages were assumed to apply proportionally to the unsampled regions. The area and population-based results were then averaged to estimate the number of toxic sites expected in each region; see the formula for v later.

A total estimate of toxic sites countrywide *(T)* is equal to the sum of the sampled and not sampled extrapolations. A 95% confidence interval around this estimate was determined using techniques applicable to cluster random sampling, treating each region as a cluster.

Regional Calculation Formulas

$$T = u + v$$

$$where\ u = \frac{1}{2}\left[\sum_{i=1}^{N}\left(\frac{x_i}{Area\ Dst_i}\right) \times Area\ Rgn_i + \sum_{i=1}^{N}\left(\frac{x_i}{Pop\ Dst_i}\right) \times Pop\ Rgn_i\right]$$

$$and\ v = \frac{1}{2}\left[\sum_{i=1}^{M}\left(Avg\ Sites\ Per\ Area\ Sampled\right) \times Area\ Rgn_i + \sum_{i=1}^{M}\left(Avg\ Sites\ Per\ Pop\ Sampled\right) \times Pop\ Rgn_i\right]$$

where

T is the total estimate of toxic sites countrywide.
u is the sampled district extrapolation.
v is the not sampled district extrapolation.
x_i is the number of sites found in district i.
N is the number of regions sampled.
M is the number of regions not sampled.
Area Dst_i is the area of district i.
Area Rgn_i is the area of region i.
Pop Dst_i is the population of district i.
Pop Rgn_i is the population of region i.

Countrywide Analysis Methodology. A second analysis estimated the total number of toxic sites in Ghana assuming the randomly selected districts where toxic site analysis was performed were a representative sampling of districts in Ghana. The total number of sites in the county was then determined to be the average number of sites found in sampled districts ($\overline{x}$) multiplied by the number of districts in Ghana *(b)*; see the formula for *T* below. The 95% confidence interval around this estimate was also determined assuming random sampling techniques.

From this total estimate, the number of toxic sites per region *(R_i)* was estimated using an equally weighted measure of 3 categories: number of sites

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found in the region, area of the region, and population of the region; see the formula for R_i later.

Countrywide Calculation Formula

$$T = \bar{x} \times b = \sum_{i=1}^{P} R_i$$

$$\text{where } R_i = \frac{T}{3}\left[\left(\frac{x_i}{\sum_{i=1}^{N} x_i}\right) + \left(\frac{Area\ Rgn_i}{Total\ Country\ Area}\right) + \left(\frac{Pop\ Rgn_i}{Total\ Country\ Pop}\right)\right]$$

where

T is the total estimate of toxic sites countrywide.

N is the number of regions sampled.

$\bar{x}$ is the average number of sites found in sampled districts.

R_i is the estimated number of contaminated sites in region i.

x_i is the number of sites found in district i.

b is the number of districts.

P is the total number of regions.

RESULTS

The team conducted data analyses between July and August 2015. In total, 72 toxic sites were confirmed (Table 2). For the purpose of this study, a confirmed toxic site was defined as having a soil sample containing one of the predefined heavy metals at a concentration above US EPA standards (or equivalent) through instrumental XRF analysis. The breakdown of contaminated sites by pollutant can be seen in Figure 2. The geometric mean (GM) lead level of the samples taken at sites classified as contaminated was 2723 ppm (n = 27); the US EPA action level for lead in residential soil is 400 ppm. The highest lead reading in a mixed-use industrial/residential area was 97,835 ppm more than 244 times the residential standard and 81 times the industrial standard. The GM of chromium samples taken at contaminated sites was 628 ppm (n = 20). The highest chromium sample recorded in the study was 4216 ppm more than 19 times the residential standard of 220 ppm. The GM of arsenic samples taken at contaminated sites was 101 ppm (n = 16). The highest sample recorded was 5661 ppm more than 470 times the standard of 12 ppm arsenic in either residential or industrial soil. Cobalt (n = 7) and manganese (n = 2) samples had geometric means of 846 ppm and 687 ppm, respectively.

Results Using Regional Analysis Methodology. Analysis using the Regional Analysis technique (bottom-up approach) for toxic site assessment yielded 1521 and 1601 contaminated sites for estimates based on area (km^2) and population, respectively (Table 3). Assuming equal weighting, the mean number of contaminated sites was 1561 (95%

Table 1. Regional Characteristics and District Selection

Region	No. of Districts	Area (km^2)	Population (2010 census)	Population Weighting (%)	Population Density (persons/km^2)	No. of Districts Chosen
Ashanti	30	24,889	4,780,380	19.39	192.1	1
Brong Ahafo	27	39,557	2,310,983	9.37	58.4	1
Greater Accra	16	3245	4,010,054	16.26	241.8	1
Central	20	9826	2,201,863	8.93	136.3	1
Eastern	26	19,323	2,633,154	10.68	1235.8	1
Northern	26	70,384	2,479,461	10.06	35.2	1
Western	22	23,921	2,376,021	9.64	103.0	1
Upper East	13	8842	1,046,545	4.24	99.3	0
Upper West	11	18,476	702,110	2.85	118.4	0
Volta	25	20,570	2,118,252	8.59	38.0	1
Total	**216**	**239,033**	**24,658,823**	**100**		**8**

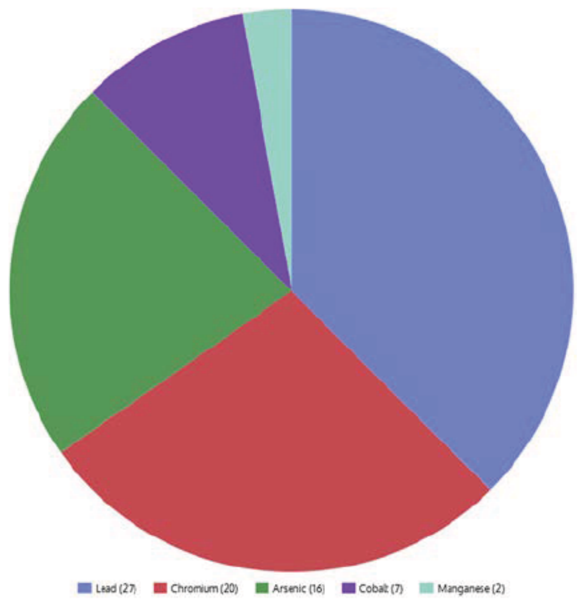

Figure 2. **Key pollutants at confirmed contaminated sites.**

confidence interval [CI]: 1135-1987), with a range of sites between 8 and 311 per region.

Results Using Countrywide Analysis Methodology. The Countrywide Analysis technique (top-down approach) estimated 1944 existing contaminated sites (95% CI: 812-3075) in Ghana. At the regional level the range of estimated toxic sites is 51-444. The countrywide extrapolation statistics and number of site estimates per region can be found in Tables 4 and 5.

DISCUSSION

Significance of Results. The estimation and extrapolation techniques outlined in this study are a preliminary attempt to estimate the scale of heavy metals contaminated sites countrywide based on district-level data. Such estimation has not previously been reported. The current extrapolation indicates that there are between 1561 and 1944 heavy metal contaminated sites in Ghana (CI 812-3075). The number of confirmed toxic waste sites in Ghana according to the 2015 TSIP database is 215. These estimates indicate that the total number of contaminated sites in Ghana is approximately 7 to 9 times the number of contaminated sites previously enumerated through the TSIP methodology. When including only those sites that are contaminated by lead, arsenic, chromium, cobalt, and manganese, the extrapolations reported by both methods are between 16 and 20 times the currently enumerated

Table 2. **Number of Contaminated Sites by District**

District	Area (km^2)	Population	Population Density (Inhabitants/km^2)	No. of Confirmed Contaminated Sites
Amansie West	1197	134,331	112.2	7
Tano South	699	78,129	111.7	7
Abura/Asebu/Kwamankese	368	117,185	318.4	6
Afram Plains South	4882	218,235	44.7	1
Ningo Prampram	1553	122,836	79.1	7
Yendi	4090	199,592	48.8	21
Ho	978	271,881	278.1	5
Juabeso	2050	111,749	54.5	18
Total	**72**			

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Table 3. Summed Regional Extrapolation (Methodology 1)*

Region	No. of Confirmed Contaminated Sites	Regional Extrapolation by Area	Regional Extrapolation by Population	Final Extrapolation (50% Area, 50% Population)
Ashanti	7	145.52	249.11	197
Brong Ahafo	7	395.88	207.05	301
Greater Accra	6	160.19	121.65	141
Central	1	3.96	12.07	8
Eastern	7	14.63	228.52	122
Northern	21	361.38	260.88	311
Western	18	105.20	38.96	72
Upper East	0	209.99	382.72	296
Upper West	0	40.25	60.09	50
Volta	5	84.10	40.31	62
Total	**72**	**1521.10**	**1601.35**	**1561**

* Sample variance: 32,496; 95% confidence interval (CI; lower bound): 1135; 95% CI (upper bound): 1987.

number of sites in Ghana. Word of mouth and footwork are the most effective tools for toxic site identification.

This extrapolation offers insight into the potential number of sites contaminated by heavy metals nationwide in Ghana. Reliable toxic site estimates and potential exposure based on thorough data via targeted sampling could greatly improve environmental burden of disease estimates.[1] Sites identified via the TSIP protocol can help fill critical knowledge gaps and play an important role in policymaking.

Site identification efforts carried out through the Toxic Sites Identification Program remain a cost effective and technically simple yet sound way to identify and rank contaminated sites. Going forward, additional efforts should be made to verify the population basis estimates calculated in this study and determine country-specific capture rates and underestimation factors. Verifying and narrowing these estimates will greatly assist policymakers and practitioners of public health as they continue to address toxic sites and chemical exposures in low- and middle-income countries. In those countries where toxic site data is extremely limited or completely unavailable, completing a mapping exercise within a representative sample of quadrats using the methods outlined in this study has the potential to greatly inform health and environmental policy.

Ghana has made notable efforts in curtailing impacts from heavy metal exposures to both workers and the general population. The central government has banned lead in consumer paint and in 2003 successfully phased out lead in gasoline. In the same year, Ghana ratified the Basel Convention in support of controlling the transboundary movement of hazardous wastes and waste disposal. Yet, like many LMICs, ensuring capacity to address the scale of contamination problems requires policy enforcement, funding, and thorough transitioning to best practices where necessary. Information regarding existing numbers of toxic sites and estimated projections are useful in determining a stepwise plan toward further site identification and eventual mitigation. Such estimates are useful in prioritizing the limited remediation funds available from developmental organizations.

Study Limitations. Though the selection of districts where site assessments took place was randomized, sampling was not. Therefore, statistical techniques to estimate the country total are not based on simple random sampling, largely because of the time-consuming and costly nature of randomly

Table 4. Countrywide Extrapolation (Methodology 2)

Basis	Countrywide Extrapolation
Districts sampled	8.00
Mean sites per sample	9.00
Standard deviation of the samples	6.82
Standard error for mean	2.37
t table value (5%)	2.36
95% CI (lower)	3.39
95% CI (upper)	14.61
Total districts	216
Total sites estimated	1944
Variance of total	228,852
95% CI (lower bound)	812.8
95% CI (upper bound)	3075.2

CI, confidence interval.

Table 5. Regional Site Estimates From Countrywide Extrapolation (Methodology 2)*

Region	No. of Districts	Estimated Sites Weighting	Area Weighting (km^2)	Population Weighting	Weighted Regional Estimate
Ashanti	30	63	67	125	255.21
Brong Ahafo	27	63	107	60	230.54
Central	20	54	27	62	142.64
Eastern	26	9	52	69	130.09
Greater Accra	16	63	9	105	176.44
Northern	26	189	191	65	444.51
Volta	25	45	56	55	156.04
Western	22	162	65	62	288.85
Upper East	13	0	24	27	51.28
Upper West	11	0	50	18	68.41
Total	**1,944.00**				

* Equal weights (33.333%) were given to each factor.

sampling throughout several thousand square kilometers. The lack of randomization likely biases the results slightly. However, because of the lack of such a methodology in the scientific published reports, there appears to be no "gold standard" for a method.

In addition, because of analytical limitations of the XRF sampling equipment, mercury, a highly toxic transition metal, was not included in the sampling analysis. The Agency for Toxic Substances and Disease Registry suggests a health safety limit of 1 ppm for elemental mercury; however, the sampling equipment was unable to detect mercury levels less than 15 ppm. Ghana's rich gold ore deposits make small-scale mining a viable economic source for many low-income families. In 2013, the country's total gold output stood at 97.8 tons.[32] One-third of Ghana's annual gold production likely comes from the artisanal and small-scale industry and almost all gold is exported.[33] Presently there are 77 mercury-contaminated sites, largely the result of gold mining, in the TSIP database. If the same extrapolation found for heavy metals included in this study is applied to mercury sites, we estimate there are between 539 and 847 mercury-contaminated sites throughout the country. However, because of a lack of comprehensive identification and sampling within a representative section of the country, these values cannot be confirmed with any certainty. Had mercury sites been included in the assessment, the estimated number of toxic waste sites would have increased substantially.

CONCLUSIONS

Our current extrapolation indicates that there may be an estimated 1561-1944 heavy metal contaminated sites in Ghana, excluding mercury-contaminated sites. This is approximately between 7 and 9 times the total number of contaminated sites in Ghana previously documented in the TSIP. On a population basis, it was estimated that there are between 31 and 115 contaminated sites per million inhabitants in Ghana.

Identification of contaminated sites allows health and environmental ministries to better allocate limited resources for improved health surveillance and remediation. Because of the costly and time-consuming nature of toxic site identification and analysis, contaminated site extrapolations prove to be a valuable tool. The statistical approaches used in this study can be replicated in other countries to better understand the prevalence of contaminated sites and formulate policy. Additional efforts should be made to verify the population basis estimates calculated in this study and determine country-specific capture rates and underestimation factors.

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2.3. Paper Three

Ericson, B., Hariojati, N., Susilorini, B., Crampe, L. F., Fuller, R., Taylor, M. P., & Caravanos, J. (2019). Assessment of the prevalence of lead-based paint exposure risk in Jakarta, Indonesia. *Science of The Total Environment*, *657*, 1382-1388.

This objective of this study was to evaluate the extent of lead-based paint use and exposure in the Indonesian capital city of Jakarta. A subset of homes and preschools were selected in different neighborhoods across the city. In total, 103 homes and 19 preschools were assessed for the presence of lead-based paint and lead dust. The results indicated both that lead-based paint did not seem be prevalent in Jakarta area homes or to present an apparent exposure risk.

While similar lead exposure assessments of housing have been carried out in high income countries, the authors of the study are not aware of any such effort in LMICs. The study contributes to a central theme of the overall thesis, which is that sources of exposure in LMICs are distinct from those in high income countries. At present, the only existing international effort to mitigate lead exposure in LMICs deals with the exposure risk presented by lead-based paint.[1] In the context of the other papers presented in this thesis, this study challenges the assumptions at the center of that approach.

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Assessment of the prevalence of lead-based paint exposure risk in Jakarta, Indonesia

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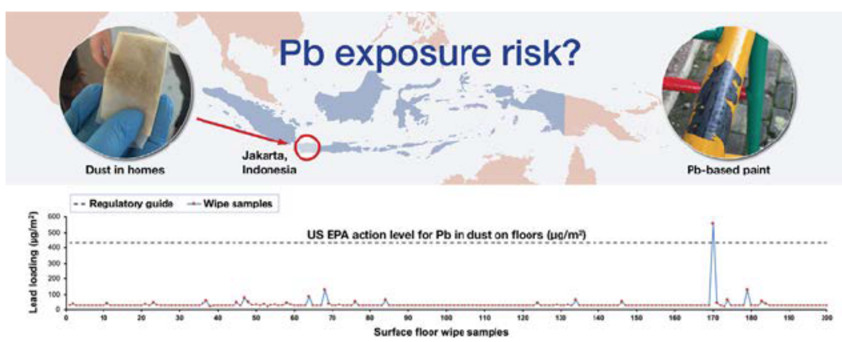
HIGHLIGHTS

- Lead based paint use and exposure appear to be limited in Jakarta homes and preschools.
- 2.7% of pXRF measurements and 0.05% of dust wipes exceeded relevant guidelines.
- 11% of homes and 26% of preschools had at least one positive pXRF measurement.
- Older structures were more likely to have lead based paint.

GRAPHICAL ABSTRACT



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ABSTRACT

While lead based paint has been banned for use in residential settings in most high income countries, it remains commonly available in many low and middle income countries (LMICs). Despite its continued availability, little is known about the specific exposure risk posed by lead based paint in LMICs. To address this knowledge gap, an assessment of home and preschool dust and paint was carried out in Greater Jakarta, Indonesia. A team of investigators used field portable X ray Fluorescence (pXRF) to measure 1574 painted surfaces for the presence of lead (mg/cm^2) and collected 222 surface dust wipe samples for lead loading (µg/m^2) from 103 homes and 19 preschools across 13 different neighborhoods of Jakarta. The assessment found that 2.7% ($n = 42$) of pXRF measurements and 0.05% ($n = 1$) of dust wipe samples exceeded the commonly applied USEPA guideline values for paint (1 mg/cm^2) and dust (floors: 431 µg/m^2; window sills: 2691 µg/m^2). Thus, contrary to expectations the locations analyzed in Greater Jakarta showed that exposure risk to lead based paint appears low. Further study is required in other settings to confirm the findings here. Precautionary measures, such as the proposed ban on lead based paint, should be taken to prevent the significant social and economic costs associated with lead exposure.

1. Introduction

Lead is a known neurotoxicant with a range of adverse health outcomes, including IQ decrement in children and cardiovascular effects in adults (ATSDR, 2007). There is no known safe level of lead exposure

Abbreviations: LMICs, Low- and Middle-Income Countries; pXRF, Portable X-ray Fluorescence Instrument; MoEF, Ministry of Environment and Forestry.

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(Centers for Disease Control and Prevention, 2017). With regard to lead based paint, exposure can occur through direct ingestion of paint chips or more commonly through incidental ingestion of lead dust (ATSDR, 2007; USEPA, 2008). Household lead dust is typically too large to pene trate the lungs when inhaled and accordingly migrates via the mucociliary elevator to the esophagus where it is ingested (ATSDR, 2007). Estimates of children's daily ingestion rates vary, with recent studies in Bangladesh indicating they may be as high as 550 mg/day (Kwong et al., 2017). The United States Environmental Protection Agency's (USEPA) guidance on children's lead exposure utilizes the widely accepted value of 85 135 mg/day based on studies carried out in Massachusetts, USA in the 1990s (OSWER, 1994; Stanek and Calabrese, 1995). Dermal exposure has not been demonstrated to be a significant pathway for the biological uptake of lead (ATSDR, 2007). Given children's significant hand to mouth behavior, rapid biological uptake of ingested lead, and developing central nervous systems, they are at greater risk of adverse health outcomes from exposures (Abrahams, 2002; ATSDR, 2007; Mielke and Reagan, 1998). In serious cases, exposure to lead based paint can result in very high blood lead levels (Gulson et al., 1995; Marino et al., 1990).

The prohibition of lead based paint use for residential purposes first occurred in 1920 in Brisbane, Australia (Needleman, 2004). Throughout the 20th century most high income countries enacted similar bans (Needleman, 2004; UNEP, 2016). By contrast, most low and middle income countries (LMICs) did not (UNEP, 2016). A recent study by the United Nations Environment Program reported that only 70 of 196 countries surveyed had existing laws regulating the use of lead in deco rative paints (UNEP, 2016).

A large number of lead based paint exposure assessments have been undertaken in high income countries with a marked dearth of similar assessments in LMICs despite its continued availability (Jacobs et al., 2002; Pirkle et al., 1998; Schwartz and Levin, 1991). The authors are not aware of any study assessing exposure to lead based paint in Indonesia. Studies that have examined lead based paints in LMICs have focused primarily on its available in the market, rather than on ex posure (Clark et al., 2006, 2009; Montgomery and Mathee, 2005). A 2014 study by the International POPs Elimination Network (IPEN) of 7 South and Southeast Asian countries found that 76% of 803 oil based enamel paints contained >90 mg/kg lead (Brosché et al., 2014). The value of 90 mg/kg is a widely accepted threshold used by the United States Consumer Product Safety Commission, among others (CPSC, 2008; UNEP, 2016). In Indonesia, the IPEN study found that 77% of 78 paints sampled had >90 mg/kg. The average concentration of analyzed paints was 17,300 mg/kg. For colored paints specifically, the average concentration was 27,500 mg/kg, while 80% contained >90 mg/kg (Brosché et al., 2014). In 2015 Indonesia passed a voluntary standard for solvent based decorative paints with a maximum concentration of 600 mg/kg (SNI 8011 2014) (National Standardization Agency of Indonesia, 2014). The Ministry of Environment and Forests (MoEF) has expressed its intention to phase out lead based paints entirely in 2018, though as of November 2018 the phase out had not occurred (BaliFokus, 2017).

The objective of this study is to address the paucity of published lead based paint exposure assessments in LMICs by characterizing and evaluating the extent of the problem in Greater Jakarta. This study de tails the outcomes of environmental assessments conducted in Jakarta homes and preschools between September 2016 and August 2017. It was designed to assess both the extent to which lead based paint was present in Jakarta homes and preschools and whether environmental lead levels in household and preschool dust exceeded relevant exposure threshold values. An effort was made to select a set of structures that would be geographically and economically representative of the city as a whole and unlikely to be affected by informal lead smelting. Assess ments were conducted in a manner that was broadly consistent with similar exposure assessments conducted elsewhere (US HUD, 2012; USEPA, 1995). The study utilized field portable X ray Fluorescence (pXRF) instrumentation to assess the presence of lead based paint. Lead dust was measured via acid digestion and analysis of surface wipes collected in accordance with the method ASTM E 1728 16 (ASTM, 2016).

2. Methods

2.1. Setting

Jakarta is the capital city of Indonesia, situated on the northwestern coast of the island of Java. In 2015, it had a population of approximately 10 million people having grown from 5 million in 1975 (United Nations, 2014). Jakarta city itself is divided among 6 different regencies (Central, East, North, South and West Jakarta, and Thousand Islands). The Greater Jakarta area is generally considered to include Jakarta proper and 4 more neighboring regencies (Bogor, Depok, Tangerang, and Bekasi), to gether comprising the second largest metropolitan area in the world after Tokyo. Greater Jakarta had a population of approximately 28 mil lion people in 2010, growing at a rate of 2.6% per year (Pravitasari et al., 2015).

2.2. Structures assessed

In situ assessments of 103 homes and 19 preschools in 13 differ ent Jakarta neighborhoods were conducted during the study period. The 13 neighborhoods were located in 10 Jakarta area regencies (Bekasi City, Central Jakarta, Depok, East Jakarta, Kota Bogor, North Jakarta, South Jakarta, South Tangerang, Tangerang and West Ja karta). Fig. 1 depicts the spatial distribution and number of struc tures assessed across Greater Jakarta. Neighborhood selection was based on three criteria: geographic representation, household in come, and the absence of any known lead smelter. With regard to ge ography and income, heterogeneity was prioritized to facilitate the collection of samples from households with different incomes throughout the city. With regard to the presence of lead smelters, in formal lead smelting is common in Jakarta (Haryanto, 2016). Neigh borhoods were therefore selected to limit the influence that smelter emissions might have on the study.

Socioeconomic data were not available at the neighborhood level. The most granular level for which this data could be identified was re gency (Central Bureau of Statistics, 2015, 2017). Accordingly the sample could not be evaluated to determine economic heterogeneity. As an al ternative, neighborhoods that were presumed to be economically het erogeneous were selected jointly by the study authors and the Ministry of Environment and Forestry (MoEF). Individual homes representing low, middle, and high incomes were then identified in tan dem with local government officials (i.e. sub district office) in each neighborhood. This selection was guided by the Indonesian Ministry of Housing regulation 11/PERMEN/M/2008 which lays out characteris tics of low, middle and high income housing, including descriptions of size, condition and cost (Ministry of Housing, 2011).

Home and preschool owners were required to sign consent forms before the study began. During the assessment, occupants completed a questionnaire consisting of approximately 27 questions for preschools and 44 for homes. The questionnaire collected data describing the struc ture itself, its use, and its occupants. Of the 103 homes assessed, house hold income level was determined in all but one. Of these, 25 were high income, 38 were middle income and 39 were low income. Of the 122 total structures assessed, 3 were built between 1970 and 1980; 5 were built between 1981 and 1990; 25 were built between 1991 and 2000; 68 were built between 2001 and 2010; and 20 were built between 2011 and 2015. Age was not determined for one structure. The majority of homes ($n = 53$ or 52%) and all preschools had occupants under 6 years of age.

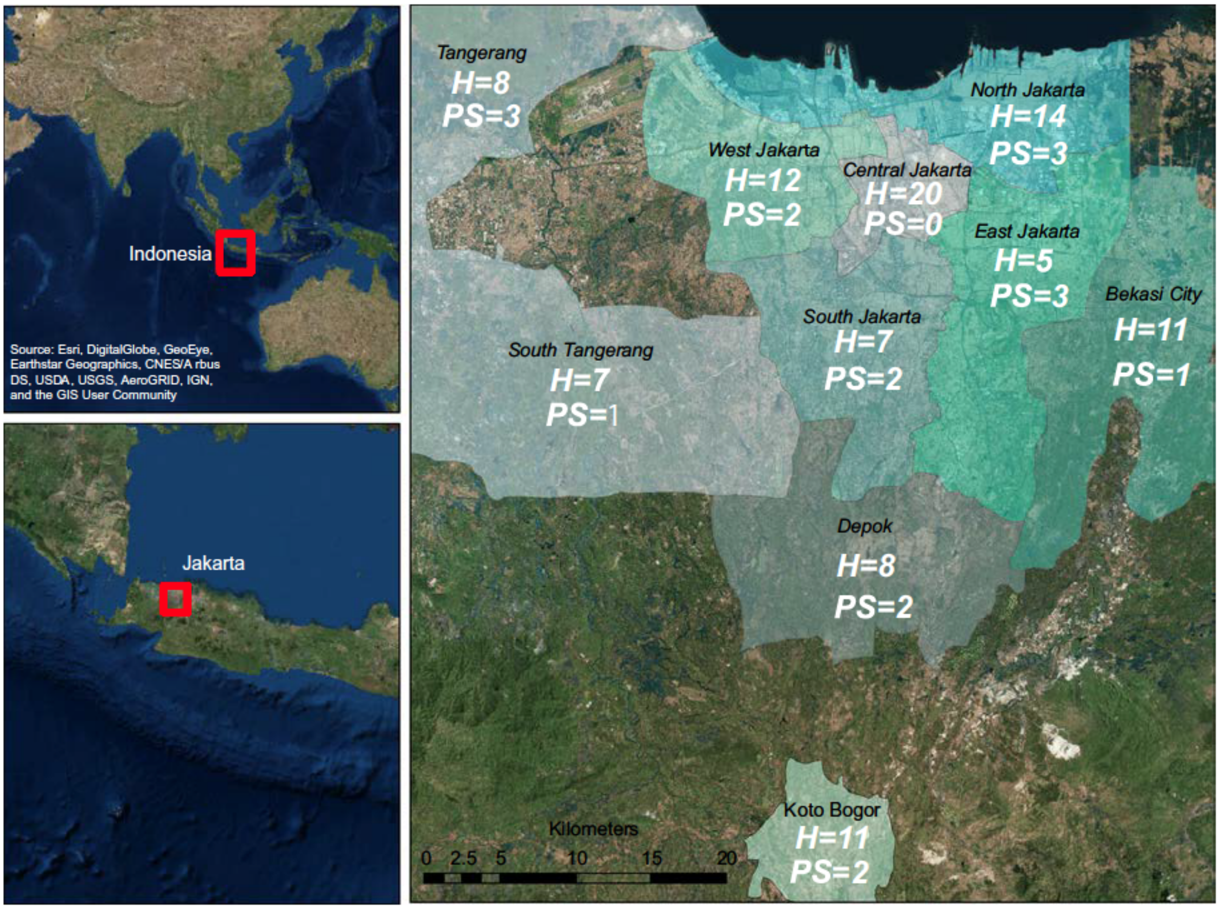

Fig. 1. Study location depicting the number of houses (H) and preschools (PS) in each regency assessed for the presence of Pb-based paint and lead in dust.

2.3. X ray fluorescence measurements

Each wall of each assessed room was checked for lead based paint with pXRF instrumentation with lower instrument detection limits for lead in paint of <0.02 mg/cm^2 after a 3 s test (Olympus NDT, 2017; Thermo Scientific, 2017). A Niton XL3t and an Olympus Delta Classic Plus instrument were used for the assessment. Both instruments were calibrated according to their respective manufacturers' instructions be fore use. Given that Indonesia does not have a comparable standard for painted surfaces, the USEPA threshold of 1 mg/cm^2 was used for assess ment purposes (USEPA, 1998, 2008). Thus, painted surfaces with a lead concentration of ≥1 mg/cm^2 were classified as positive for lead based paint. In total, 1574 measurements of painted surfaces were compiled for inclusion in this study. The surfaces tested comprised: 1359 walls, 114 window frames, 52 from playground equipment, and 49 pieces of furniture. The study took an average of 12 pXRF paint measurements per house (range: 4 26) and 18 pXRF paint measurements per pre school (range: 8 58).

2.4. Dust wipes

A total of 222 dust wipe samples were collected using Ghost Wipes®, nonwoven polyvinyl alcohol fiber wipes moistened with de ionized water. Of these, 205 dust wipes were collected from floors and 17 from window tracks. An average of 1.7 (range: 1 8) wipes were taken at each house and an average of 2.3 (range: 1 7) wipes were taken at each preschool. In total 179 wipes were collected from homes and 43 from preschools. Each floor sample consisted of a wipe over a 60 cm × 60 cm area as per the method set out in ASTM E 1728 16 (ASTM, 2016). Sampled dust wipes were placed in individually labeled and sealed Nasco Whirl Pak® polyethylene bags. Information about each wipe, including sample ID, wiped surface area and location was re corded in the field. Used wipes were couriered to a certified laboratory in New Jersey, USA (EMSL Analytical, Inc.) for analysis of acid digestible lead using flame atomic absorption spectrophotometry (modified USEPA Method 700B) (A2LA, 2017; USEPA, 2007). The laboratory limit of reporting based on the lowest standard analyzed in its calibration was <10 μg (Pb)/wipe.

In an effort to provide a conservative estimate of lead dust loading, floor wipe samples were focused on areas expected to have higher dust accumulation. Sampling the center of rooms was avoided because it was anticipated that dust concentrations would be lower due to cleaning and foot traffic. With regard to windowsills, these areas have been identified as an important source of lead based paint exposure in the US (Dixon et al., 2009; Lanphear et al., 1996; USEPA, 2010). How ever, windowsills are not common in Indonesian households and as a result were not available for sampling in the homes examined for this study. As an alternative, a limited number of dust wipe samples were collected from the track of horizontally oriented sliding windows. Al though this study relies on US regulatory standards to guide interpreta tion of the lead in surface wipes, all data is reported in metric using the National Institute of Standards and Technology conversion of 0.09290304 ft^2 per m^2 (Butcher et al., 2006).

2.5. Statistical analysis

Where the data were amenable, regression analysis was conducted to assess the relationship between the results of the questionnaire and those of the environmental assessment. For the purpose of regression analysis, pXRF measurements were converted to dichotomous variables with samples above the 1 mg/cm^2 USEPA guidance level coded as positive for lead based paint. Dust wipes were considered positive if the results showed detectable lead within the laboratory reporting limits. Dust wipe results were also assessed as continuous variables to determine the relative influence of multiple independent variables on lead loading.

The results of environmental analysis were regressed against the following questionnaire categories: structure age (continuous in years), presence of air conditioning (absence = 0, presence = 1), cleaning frequency (0 3 times per day), presence of a garden (absence = 0, presence = 1), renovation within preceding 12 months (no = 0, yes = 1), and presence of pets (no = 0, yes = 1). Multiple logistic regression was used when the dependent variable was dichotomous with multiple linear regression being applied when the dependent variable was continuous. Income data were not collected for preschools. To assess whether household income influenced the presence of lead based paint or lead dust, a simple logistic regression model was run for homes only with the inclusion of the independent variable income (low = 1, middle = 2, high = 3). Finally, simple logistic regression was conducted to assess the relationship between positive paint and dust samples. Statistical analysis was conducted using Stata 15.1 (StataCorp. LP, 2017).

3. Results

The majority of samples analyzed fell below the instrument and laboratory reporting limits of the relevant analysis method. The results are summarized below for both pXRF and dust wipes. Table 1 presents the results of pXRF measurements against subgroups. Table 2 presents the results of the dust wipe analysis.

Table 1
Prevalence of lead-based paint in structures assessed in Jakarta.

	Structures assessed with pXRF		Structures with ≥1 positive pXRF reading (≥1 mg/cm^2)	
	Number	Percent of sample	Number	Percent of subgroup
Total	122	100.0%	16	13.1%
Structure Type				
Preschools	19	15.6%	5	26.3%
Homes	103	84.4%	11	10.7%
Construction Year				
1970–1980	3	2.5%	3	100%
1981–1990	5	4.1%	2	40%
1991–2000	25	20.5%	2	8.0%
2001–2010	68	55.7%	7	10.3%
2011–2015	20	16.4%	2	10%
Unknown	1	0.8%	0	0%
Income (Homes only)				
High Income	25	24.3%	3	12.00%
Low Income	39	37.9%	5	12.8%
Middle Income	38	36.9%	2	5.3%
Unknown Income	1	1%	1	100%
Regency				
Bekasi	12	9.8%	2	16.7%
Bogor	13	10.7%	0	0%
Central Jakarta	20	16.4%	0	0%
Depok	10	8.2%	0	0%
East Jakarta	8	6.6%	2	25%
North Jakarta	17	13.9%	2	11.8%
South Jakarta	9	7.4%	6	66.7%
South Tangerang	8	6.6%	3	37.5%
Tangerang	11	9%	1	9.1%
West Jakarta	14	11.5%	0	0%

3.1. Portable XRF results

Of the 1574 paint samples analyzed with a pXRF, 2.7% ($n = 42$) were positive, meaning they exceeded the USEPA lead based paint guideline of 1 mg/cm^2 (range: 1.16 6.42; median = 2). The balance ($n = 1532$; 97.3%) of the paint samples were negative, meaning they were below the USEPA guideline. Seventy eight percent ($n = 1232$) of pXRF measurements were below the instrument limit of detection (0.02 mg/cm^2). Among the 42 positive measurements 19 were in homes and 23 were in preschools. Of the 23 positive measurements in preschools, 4 were from walls, 13 were from playground equipment and 6 were from classroom furniture. Within homes, 17 of the 19 positive measurements were from walls, while 2 were from windows.

Eleven homes (10.7% of all homes assessed) and 5 preschools (26.3% of preschools assessed) had at least one positive pXRF paint reading. Approximately 7% ($n = 23$) of the 330 pXRF measurements taken in preschools and 1.5% ($n = 19$) of the 1244 pXRF measurements taken in homes were positive.

3.2. Dust wipe results

Of the 205 dust wipes taken from floor surfaces, a single sample (0.05% of analyzed wipes) exceeded USEPA action level for floor surfaces of 431 μg/m^2 (40 μg/ft^2) (USEPA, 1998). This wipe was taken from the floor of a preschool and had a loading of 555 μg/m^2. A further twenty three wipes (11.2%) had loadings within the reporting range of the laboratory but below the USEPA action level. These wipes returned a median reading of 50 μg/m^2 (IQR: 42 66 μg/m^2).

Of the seventeen window track wipes taken, none had loadings exceeding the USEPA action level of 2691 μg/m^2 (250 μg/ft^2) for window sills (USEPA, 1998). Fourteen samples (82.3%) with a median loading of 522 μg/m^2 (IQR = 312 1184) were above the reporting limit but below the USEPA action level for windowsills. The highest dust wipe loading (1829 μg/m^2) came from a window track.

In total 20 structures had at least one dust wipe with results within the reporting limits. Of these 16 were from homes (15.5% of all homes assessed) and 4 were from preschools (21% of all preschools assessed).

3.3. Results of statistical analysis

An evaluation of structures with at least one positive pXRF measurement against the results of the questionnaire found a statistically significant association with the age of the structure only ($p < 0.05$; $z = 2.36$). Older structures were more likely to have lead based paint than newer structures.

In structures where at least one dust wipe was within the laboratory reporting limits, a statistically significant association was identified with the presence of an air conditioner only ($p < 0.05$; $z = 2.22$). Structures with air conditioning were more likely to have detectable lead dust than those without air conditioning.

In assessing factors contributing to lead loading, a statistically significant relationship was identified with a recent home renovation only ($p < 0.01$). With regard to household income, this variable was not a statistically significant predictor of the presence of lead based paint or lead dust. Finally, a statistically significant relationship was identified between structures with at least one positive pXRF measurement and a dust wipe within the laboratory reporting limits ($p < 0.01$; $z = 3.03$).

4. Discussion

This study of 103 homes and 19 preschools in Greater Jakarta found that lead based paint exposure risk appears to be low. A small number of surfaces in this study were found to contain lead based paint; only 42 of 1574 of pXRF measurements (2.7%) across a range of surfaces in homes and preschools were found to contain lead based paint. Further, lead based paint was found in a minority of structures; 10.7% of homes

Table 2
Analysis of dust wipes from floors and window tracks.

Surface	Total samples	Samples within laboratory reporting limits				USEPA action level	Number of samples above action level (% of total)
		Number within reporting limits (% of total)	Mean μg/m^2 (SD)	Median μg/m^2 (IQR)	Range μg/m^2		
Floors	205	38 (18.5%)	58 (86)	36 (IQR: 28–56)	28–555	431 μg/m^2	1 (0.048%)
Window tracks	17	14 (82%)	738 (600)	522 (IQR = 312–1184)	107–1829	2691 μg/m^2	0 (0%)

and 26.3% of preschools contained at least one surface with lead based paint. For individuals to be at risk from lead based paint hazards, they must first become exposed. This typically occurs through ingestion of deteriorated lead based paint as dust (Dixon et al., 2009). Analysis of the surface wipes collected in this study revealed that only one sample (0.05%) of 222 analyzed presented above the USEPA action level of 431 μg/m^2 (40 μg/ft^2). For context, in 1990, 12 years after lead based paint was phased out in the US, the USEPA conducted a national lead based paint survey that evaluated 381 housing units (USEPA, 1995). The study found that 83% of homes contained lead based paint somewhere in the building (USEPA, 1995). Dust wipes in the corresponding homes found lead dust above the contemporaneous guidelines (2152 μg/m^2 for floors, 5381 μg/m^2 for windows) in 12 29% of samples (USEPA, 1995). Thus by comparison, the results of the present study, conducted before a legislative phase out in Indonesia, seem to show a much lower risk. Of note, a 2002 study led by the US Department of Housing and Urban Development with a larger sample size and more robust assessment method than the 1990 effort found a much lower prevalence (40%) of American homes with lead based paint somewhere in the building (Jacobs et al., 2002).

It is important to discuss the difference between the results of this study and those of the 2014 IPEN study or its precursor conducted in 2013 (Brosché et al., 2014; Ismawati et al., 2013). The IPEN effort found that 77% of store bought enamel paints, selected at random, contained >90 mg/kg lead, meaning they were lead based. By contrast, this study found that only 2.7% of all samples of both enamel and non enamel paints could be classified as lead based paint. Thus while lead based paints seem to be widely available on the market, they do not seem to be prevalent in Jakarta homes or preschools. One possible reason for this might be the limited use of enamel paints more generally in these settings. Enamel paints are evidently more prevalent in preschools where brightly colored coatings are common, especially on playground equipment. Indeed, given that this study found a higher prevalence of lead based paints in preschools than schools (7% of all samples compared to 1.5%, respectively) seems to be consistent with this anecdotal observation. However, even in these settings few measurements of enamel paints were positive for lead based paint. Importantly, these results are not immediately comparable with IPEN's as their study presents concentrations (mass per mass or mg/kg) while this study utilizes loadings (mass per area or μg/m^2). Relevantly, with regard to exposure assessment, lead loading has been shown to be the more significant predictor of children's BLLs (Davies et al., 1990; Gulson and Taylor, 2017; Lanphear et al., 1995).

As noted above, exposure to lead based paint occurs most significantly through the ingestion of deteriorated lead based paint as dust (Dixon et al., 2009). A number of factors can contribute to the peeling or chipping of enamel paints, including the presence of air conditioners and home renovation work. Home renovation work can play a direct role in mobilizing lead dust as coverings are chipped or sanded (Marino et al., 1990; Rabinowitz et al., 1985). High ambient humidity, like that present in Jakarta, can contribute to peeling paint. Thus the presence of air conditioners, which dehumidify the ambient environment, has been associated with decreased blood lead levels (Jacobs et al., 2009; Kimbrough et al., 1995; Othman et al., 2015). In this study, the presence of air conditioners inside homes was associated with an increased likelihood of detectable lead dust, which is in contrast to studies conducted elsewhere. The reasons for this were not immediately clear, though future studies might endeavor to further evaluate this result, including possible interactions with outdoor air.

There is some limited evidence that lead dust exposure can be mitigated through regular cleaning, although the efficacy of education and home interventions has elsewhere been questioned (Ettinger et al., 2002; Nussbaumer Streit et al., 2016). In this study, 100% of survey respondents reported sweeping and mopping at least once daily. Of these 60% ($n = 74$) reported sweeping and mopping at least twice daily and 3% ($n = 4$) reported sweeping and mopping three times daily. There was no statistically significant relationship identified between cleaning once or three times daily, however it is possible that low lead loading is attributable in part to diligent cleaning practices. Future efforts might more thoroughly evaluate the influence of cleaning than was attempted here.

The Indonesian government ostensibly intends to ban the sale of lead based paint imminently (BaliFokus, 2017). This study suggests that even in the absence of such a ban, lead based paint exposure in Jakarta may be decreasing. Newer structures in this study were less likely to have lead based paint than older ones ($p < 0.05$). In this context, the phase out can be seen as precautionary in nature rather than as a reaction to a confirmed public health risk. This measure could potentially mitigate both human health risks and economic costs associated with abatement. For example, a 2009 study in the United States found that the costs of mitigating lead based paint exposure ranged from USD 1200 10,800 per household (in 2006 USD) (Gould, 2009). Extrapolating for the country as a whole, the same study found that mitigating exposures in the 1.02 million homes at significant risk would cost USD 1.2 11 billion (Gould, 2009).

This study has a number of limitations. Among them is the reliance on different levels of government to identify households for inclusion in the study. This introduced the possibility of a sampling bias, meaning that homes may have been included or excluded in an effort to influence the results unbeknown to the research team. However, given the limited knowledge of the use and risk associated with lead based paints in Indonesia, such a scenario would be unlikely. A second related limitation is the income designations used in the study. Income data at the household level were not available to randomly select structures for inclusion. Thus, the study relied on local (sub district) officials to identify households in each income group. This again introduces the possibility of sampling bias and inaccurate assessment. Anecdotal evidence obtained by site investigators indicated that income designations were accurate, however this information was not further evaluated by other means, such as the questionnaire. Nevertheless, because income could potentially influence the use of lead based paints, this represents a study limitation. Finally, the small sample size limited the number of analyses that could be conducted. The relative impact of neighborhood or regency location, for example, could not be properly evaluated due to the sample size.

5. Conclusion

Exposure to lead from lead based paint in Greater Jakarta appears to be limited. Although lead based enamel paints are widely available for sale, this study found that they do not seem to be highly prevalent in settings where humans are most at risk or significantly contribute to lead dust loadings. Further study is required to confirm the findings here. While the current lead based paint exposure risk appears low, the anticipated regulatory ban is an important precautionary measure to prevent the high economic and social costs associated with lead based paint exposure as experienced in high income countries.

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Conflict of interest

Pure Earth executes risk mitigation projects at contaminated sites in LMICs including those related to informal car battery recycling.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.12.154.

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2.4. Paper Four

Ericson, B., Otieno, V.O., Nganga, C., St. Fort, J. and Taylor, M.P. (2019). Assessment of the Presence of Soil Lead Contamination Near a Former Lead Smelter in Mombasa, Kenya, *Journal of Health and Pollution*, 9(21), 190307.

The fourth paper in Chapter 2 describes the extent and severity of surface soil lead contamination in an informal settlement abutting a former lead smelter in Mombasa, Kenya. The smelter had been the subject of significant international scrutiny following an apparent lead poisoning event.[1] A community organizer and former smelter employee was awarded the Goldman prize for her role in shuttering the facility, while the Kenyan Ministry of Environment and Forests (MoEF) was sued by the community for permitting the smelter's development. The research was conducted at the invitation of MoEF for the purpose of developing a risk mitigation plan.

This study is the second in Chapter 1 to find limited exposure risk associated with high profile sources. The first such study focused on lead-based paint (Paper 3). In that case, the absence of exposure risk challenged assumptions at the center of international efforts. Here, the heightened profile of the site was also based on limited data. The paper thus supports the importance of evidence-based decision making. In-situ environmental data collection is combined with multiple modeling approaches to evaluate aerial deposition, residual contamination and attributable BLL increases.

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Assessment of the Presence of Soil Lead Contamination Near a Former Lead Smelter in Mombasa, Kenya

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Background. The informal settlement of Owino Uhuru near an abandoned lead smelter attracted international attention due to an apparent lead poisoning event. Despite this attention, the environmental data collected to date do not indicate high levels of residual contamination.

Objectives. To further confirm previous findings and determine any necessary risk mitigation measures, an assessment of surface soil lead concentrations was conducted in the community.

Methods. Investigators carried out an assessment of the soil in a ~12,000 m^2 section of the Owino Uhuru neighborhood over the course of a single day in June 2017 with the assistance of community leaders. Fifty-nine *in situ* soil measurements were taken using an Innov-X tube-based (40 kV) alpha X-ray fluorescence instrument (pXRF).

Results. The assessment found that mean surface soil lead concentrations in areas conducive to exposure were 110 mg/kg (95% CI: 54–168); below United States Environmental Protection Agency and the Environment Canada screening levels of 400 mg/kg and 140 mg/kg, respectively.

Conclusions. There is likely no current need for risk mitigation activities in the community. These results could inform discussions on the allocation of public health spending.

Competing Interests. The authors declare no competing financial interests. BE, VOO, CN and JSF are employees of Pure Earth. MPT sits on the Editorial Board of the Journal of Health and Pollution.

Keywords. lead exposure, LMICs, informal settlements

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Introduction

Lead smelting is a significant source of global soil contamination.[1-4] Surface soil lead concentrations in residential areas can result in exposure to humans through pica (hand-to-mouth) behavior and the inhalation and ingestion of contaminated soil and dust.[5,6] Accordingly, several studies have demonstrated that blood lead levels (BLLs) are strongly associated with soil lead concentrations.[6-11] There is currently no known safe level of exposure to lead, although the United States Centers for Disease Control (CDC) utilizes a reference dose of 5 µg/dL.[12,13] Childhood lead exposure can result in intelligence quotient decrement, decreased lifetime earnings and higher rates of aggravated assault, among other adverse outcomes.[14-18] Early life exposures have also been shown to not remit with age.[17,19] Although rare, extreme acute lead exposure can result in encephalopathy and death.[20] Lead exposure in adults most significantly results in increased incidence of heart disease, even at low levels of exposure.[20,21] The Institute for Health Metrics and Evaluation (IHME) estimates that lead exposure accounted for 540,000 deaths globally in 2016.[22]

Over the course of the 20th century, environmental lead contamination was most strongly associated with the use of tetraethyl lead additives in gasoline.[23-26] In the United States, lead-based paint was also a significant source of exposure.[27,28] Following the cessation of the use of lead in these two common products, blood levels have fallen significantly. For example, in the United States, average childhood blood lead declined from around 15 µg/dL in the mid-1970s, falling to < 1 µg/dL today.[27,29] Elsewhere in high-income countries, persistently elevated environmental and biological lead levels continue to be documented around mining and smelting locations.[30,31]

In low- and middle-income countries (LMIC), current major sources of lead exposure include traditional ceramic glazes and the manufacture and recycling of lead-acid batteries, particularly when conducted in an informal setting.[32-35] Discrete lead poisoning events have also been identified at mining and smelting locations.[2,36-38]

In Kenya, elevated environmental

and blood lead concentrations have been documented in occupational settings.[39] However, there is a paucity of information about exposures in the home or in residential areas. Data collected as part of the Pure Earth (New York, NY, USA) Toxic Sites Identification Program indicate that domestic lead exposure may be significant.[40] The program has identified and conducted rapid assessments of 60 discrete lead contaminated sites located in residential areas in Kenya since 2009.

A lead poisoning event in 2014 centered around a lead smelter in the coastal city of Mombasa garnered international attention.[41-45] News reports documented elevated BLLs and three deaths in the worker population at the refinery.[46] Limited reports of elevated BLLs were also later found in Owino Uhuru, an informal settlement of approximately 3,000 residents with an area of 28 000 m^2 bordering the northern wall of the facility (*Figure 1*). A 2010 study of three children found BLLs of 12, 17 and 23 µg/dL, while a later report of a separate child found a BLL of 32 µg/dL.[43,46] A community leader in Owino Uhuru and former employee of the smelter was awarded the prestigious Goldman Environmental Prize for her work in raising public awareness that would eventually lead to the closure of the facility in 2014.[45,47] The community subsequently filed a lawsuit against the national government for USD 1.5 million in compensation for its failure to monitor emissions from the smelter.[48]

Following the closure of the facility in 2015, a study of environmental lead concentrations and community BLLs was carried out jointly by the Kenya Ministry of Health and the CDC. The study found a geometric mean surface soil lead concentration in the community of 146.5 mg/kg (geometric

Abbreviations

BLLs	Blood lead levels	*LMIC*	Low- and middle-income countries
CDC	Centers for Disease Control and Prevention	*pXRF*	Alpha X-ray fluorescence instrument
GSD	Geometric standard deviation	*USEPA*	United States Environmental Protection Agency
IEUBK	Integrated Exposure Uptake Biokinetic model for children		

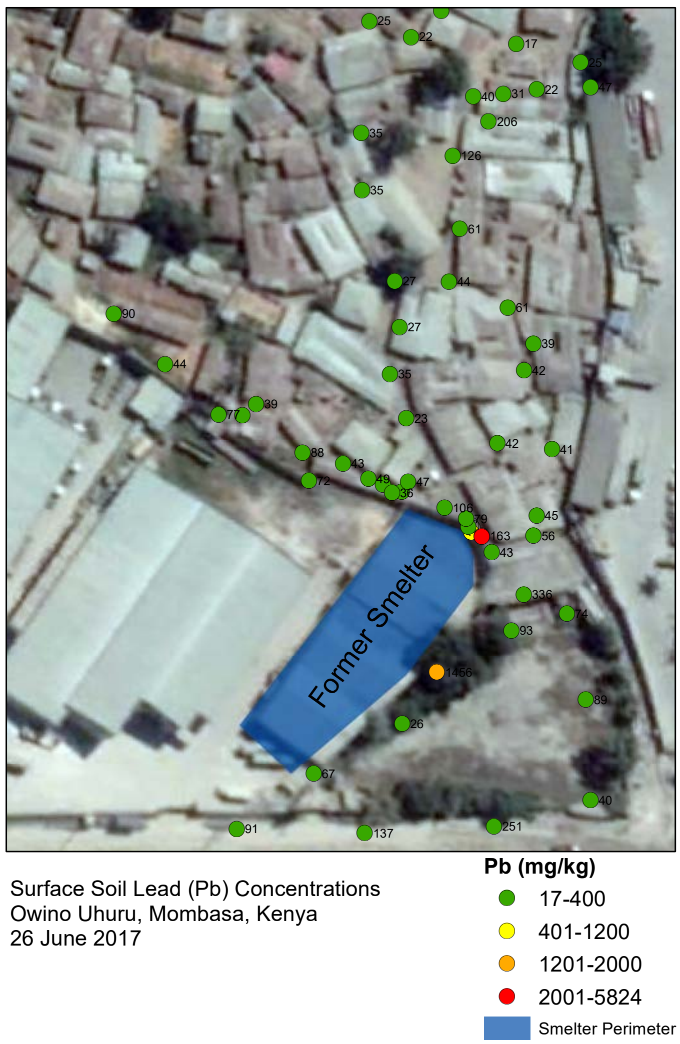

Figure 1 — Location and concentration of in-situ X-ray fluorescence instrument surface soil lead measurements in Owino Uhuru

standard deviation (GSD): 5.2) and dust lead loadings in homes of 1.5 μg/ft^2 (GSD: 12.3).[49] These results were significantly below applicable United States Environmental Protection Agency (USEPA) screening levels for residential soil and household dust of 400 mg/kg and 40 μg/ft^2, respectively.[50] In contrast with the low environmental levels, the study found a geometric mean BLL in children of 7.4 μg/dL (GSD: 1.9) in the community, exceeding both the CDC reference dose of 5 μg/dL and the BLLs found in a control neighborhood away from the facility of 3.7 μg/dL (GSD: 1.9).[13,49]

Importantly, while the BLLs found in Owino Uhuru exceeded CDC guidelines, they were somewhat lower than those reported in communities near lead smelters elsewhere in LMICs. Researchers working near lead smelters in the Dominican Republic, Mexico, Senegal and Vietnam, for example, have reported population-wide BLLs in children of 71 μg/dL (arithmetic mean), 27.6 μg/dL (median), 40.4 μg/dL (median), and 129.5 μg/dL (mean), respectively.[2,51-54]

Despite the absence of evidence of residual contamination, the site continues to attract national and international attention.[55,56] To further assess the veracity of the claims of contamination and provide the basis for necessary human health intervention strategies, an investigation of surface soil lead concentrations was carried out in Owino Uhuru in June 2017. This type of assessment forms the evidence-based rationale for any subsequent actions required to mitigate potential risk of harm arising from soil and dust contamination. Moreover, the results of the assessment in the context of its high profile potentially provide insight into the setting of public health priorities.

The results of this assessment are presented along with a simple air deposition model developed to estimate likely surface soil lead concentrations resulting from smelter emissions during its operation. The results of these models were used to estimate BLLs in children and adults in the absence of population data on exposures.

Methods

Investigators carried out an assessment over the course of a single day in June 2017 with the assistance of community leaders. Fifty-nine *in situ* soil measurements were taken using an Innov-X tube-based (40 kV) alpha X-ray fluorescence instrument (pXRF) over a ~12 000 m^2 section of the Owino Uhuru neighborhood that is adjacent to the facility. The pXRF has a lower detection limit of 5 mg/kg.[57] Fifty-seven measurements were taken directly from surface soil, while two were taken at a depth of 10 cm. Two of the surface soil measurements were taken from an area within the perimeter wall of the smelter, which are unlikely to be accessed by humans and thus are not indicative of community exposure. The pXRF was calibrated before the assessment using an alloy-grade 316 steel clip and measurement accuracy was evaluated by assessment of a National Institute of Standards and Technology (NIST) standard (2702: Inorganics in Marine Sediment) during the assessment.[58] The NIST reference material contains a known value for lead of 132.8 mg/kg. The pXRF measurement of this material found a value of 137 mg/kg (+/-10) for lead and was thus within acceptable range. The inside of the facility was not accessible and was not assessed.

Spatial and statistical analysis

Latitude and longitude for each sample point were collected using World Geodetic System 1984 format using a Garmin eTrex 10 with an accuracy of < 3 meters.[59] Spatial and statistical analyses were performed using ArcMap 10.5 and Stata 15.[60,61] Basic descriptive statistics of the data were generated to assess exposure. In addition, simple linear regression was conducted to assess any relationship between lead concentration and proximity to the smelter.

Aerial deposition model

To determine soil lead concentrations resulting from aerial emissions, a simple algorithm was developed based on known deposition rates of lead smelters in different settings. To determine the spatial extent of lead deposition, a Gaussian plume model was used to estimate the likely distributions in the Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model developed by the United States Oceanic and Atmospheric Administration.[62] Using contemporaneous meteorological data and the inputs set out below, the HYSPLIT model indicated that deposition was uniform within a range of 750 m of the smelter in all directions. The entire residential area of the Owino Uhuru falls within 300 m of the smelter stack. Thus, deposition was assumed to be uniform across the community.

A number of studies indicate that the accumulation of lead in soil is additive, meaning that concentrations increase proportionate to deposition.[63-65] Alloway sets out a simple mass balance equation where background concentrations are increased by the accumulation of metals from various sources with any reductions occurring due to crop removal, leaching, volatilization or erosion. In the case of Owino Uhuru, as no agriculture is present in the assessed areas, crop removal would not be a relevant factor. Similarly, lead is highly immobile in soil and is not volatile (1.77 mm mercury at 1,000°C), thus leaching and volatilization would not appreciably affect the net concentration of lead in soil.[66,67] Equation 1 sums the total estimated inputs and only accounts for limited migration.

Equation 1

$$SC_{mg/kg} = \left(\frac{D_{mg/m^2}}{SV_{m^3}\ x\ SM_{kg/m^3}}\ x\ O_d\ x\ O_y \right) + BC_{mg/kg}$$

where

SCmg/kg equals surface soil lead concentration

D_{mg/m^2} equals deposition rate of lead in mg/m^2 per day

O_d equals number of days of operation per year

O_y equals number of years of operation

SV_{m^3} equals volume of soil in m^3

SM_{kg/m^3} equals mass in kilograms of one unit of soil

$BC_{mg/kg}$ equals background soil lead concentration

Deposition rate inputs were based on van Alphen's study of an area surrounding a lead smelter in Port Pirie, New South Wales, Australia. This study found a mean deposition rate of 18.8 mg/m^2/day within 600 m of the smelter with maximum deposition rate of 299 mg/m^2/day.[68] These values are higher than those documented elsewhere. Studies at lead smelters in Arnhem (the Netherlands), El Paso (Texas, USA), Hoboken (Belgium), Missouri (USA), and Port Pirie (Australia) found deposition rates ranging from 0.1–17.5 mg/m^2/day, for example.[30,68-72] There are no known studies of deposition rates at rudimentary smelters like the one operated by the Mombasa facility, thus the selection of the highest deposition rates identified in the literature is intended to best approximate the poor conditions present at the facility. Similarly, a conservative stack height of 10 m was used for the purposes of modeling deposition.

Days of operation were assumed to be 260 days per year based on 5 days of operation per week for 52 weeks. The time period of the smelter operation was set at 10 years, based on news reports, which were the only available data.[46]

To determine the relevant mass of soil, a volume was calculated based on a likely penetration of deposited lead to a maximum depth of 2 cm. The 2 cm value is based on studies of the isotopic composition of soil lead at a smelter in Mount Isa, Australia.[73,74] Mackay et al. found that lead found below this depth tended to be associated with naturally occurring deposits, rather than aerial deposition from the smelter.[73] The soil type at the site is Haplic Lixisol with an approximate clay, silt and sand content of 18%, 27% and 55%, respectively and a mass of roughly 1.4 grams/cm^3.[75] These soils are conducive to metals mobility more generally, as discussed below. For the purpose of the sensitivity analysis the model was also run with a depth of 5 cm. Studies at a smelter in Boolaroo, Australia found deposited lead at a maximum depth of 5 cm with 80% less lead in the lower 2.5 cm than the top 2.5 cm.[76]

Background lead concentrations for the study area were not available. As an alternative, the mean background lead concentration for the earth's crust of 17 mg/kg was used.[77] For context, crustal lead concentrations average 25.8 mg/kg in the United States and range from 8.4–40 mg/kg in Europe and the United Kingdom.[65,78]

Blood lead level assessment

To estimate BLLs for children, the USEPA Integrated Exposure Uptake Biokinetic model for children (IEUBK) was used.[7] Following IEUBK guidelines, default values for lead from all exposure pathways were used and measured *in situ* soil concentrations were entered.[79] Default ingestion rates were then adjusted upward to account for higher ingestion rates in LMICs (250–400 mg/day).[10,80] Results were also calculated using the default values (85–135 mg/day). For adults, the USEPA Adult Lead Methodology was used.[7] Again, results were calculated using both default and augmented ingestion rates to account for increased exposure in LMICs (50–200 mg/day). Additionally, exposure duration was increased to account for a residential setting, as the Adult Lead Methodology's default values were intended for occupational exposures.

The IEUBK model was also used to estimate likely environmental Pb levels in air and soil required for a hypothetical 2-year-old child to have a BLL of 20 µg/dL. The IEUBK model assigns this age a higher BLL than younger or older age groups. It was selected to provide the most sensitivity to environmental levels.

Results

The results of both the *in situ* surface soil measurements and aerial deposition modeling indicate that environmental lead levels in Owino Uhuru are within or slightly above US regulatory screening levels and generally consistent with urban areas globally.

Surface soil assessment

The mean surface soil lead concentration in the areas assessed with the pXRF was 224 mg/kg (95% CI: 15–434). The median value was 47 mg/kg. Table 1 presents the summary results of the surface soil assessment. Four samples (7%) tested above 400

No. of measurements	59		
	Mean (SD)	Median (IQR)	Range
Concentration (mg/kg)	219 (776)	47 (36,91)	17–5824

Abbreviations: SD, Standard deviation; IQR, Interquartile range.

Table 1 — Descriptive Statistics of Surface Soil Concentrations in Owino Uhuru

mg/kg, the USEPA screening level for bare soil where children play.[81] Those four samples had the following concentrations: 582, 871, 1456 and 5824 mg/kg. The highest and third highest samples (5,824 and 871) were taken from an enclave in the perimeter wall of the smelter that is unlikely to be regularly accessed by humans and thus are not indicative of community exposure. Removing these from the likely exposure scenario results in a mean surface soil concentration of 110 mg/kg (95% CI: 54–168), below USEPA and Environment Canada screening levels of 400 mg/kg and 140 mg/kg, respectively.[81,82] Kenya has not yet developed its own guidance values for soil metal concentrations, including lead. Two samples were collected at a depth of 10 cm adjacent to the highest reading (5,824 mg/kg) for the purpose of assessing possible migration. These readings were 43 and 72 mg/kg, indicating insignificant down profile migration of lead from surface soils, which is consistent with studies of atmospherically deposited smelter soil lead contamination elsewhere.[73,74]

Within the targeted 12,000 m^2 sample area, soil lead measurements were spaced an average of 9.4 m apart (95% CI: 7.8–11.2) (*Figure 1*). There was no statistically significant association between proximity to the smelter and soil lead concentrations ($p<0.05$). The mean soil lead concentration of the eight samples taken within 3 m of the facility perimeter wall was 1,026 mg/kg (95% CI: 611–2663). The mean for the eight samples taken from 3 m to 10 m was 231 mg/kg (95% CI: 183–646) and the mean for the 43 samples taken beyond 10 m was 66 mg/kg (95% CI: 47–86). The sample taken closest to the smelter site was at the base of the perimeter wall, while the furthest was taken at a distance of 130 meters.

Aerial deposition model results

Using van Alphen's mean deposition rate of 18 mg/m^2/day and a 2 cm estimate for the likely maximum penetration of lead into surface soil resulted in an additional accumulation of 0.64 mg/kg/day within 750 meters of the facility while it was operating. Using the same deposition rate and the less conservative surface soil penetration estimate of 5 cm results in an additional accumulation of 0.26 mg/kg/day. These rates would have resulted in a surface soil concentration of 686–1,688 mg/kg after ten years of operation. To arrive at the mean value identified in pXRF sampling of 110 mg/kg, a daily deposition rate of 1–2.51 mg/m^2/day (0.036 mg/kg/day) would be required.

Blood lead level assessment

Current BLLs for 0- to 7-year-olds were estimated to be from 1.4–2.4 μg/dL using the default ingestion values in the IEUBK and 2.7–5.1 μg/dL using the augmented values. Current BLLs of adults were estimated to be 1.8–2.6 μg/dL, depending on the ingestion rate used. For surface soil exposure to result in a BLL of 20 μg/dL in 2-year-olds, an approximate surface soil concentration of ~2,500 mg/kg would be required with default ingestion values and ~850 mg/kg with the augmented values. With regard to air concentrations, a level of ~24 μg/m^3 would be required for a 2-year-old to have a BLL of 20 μg/dL, assuming a soil lead concentration of 110 mg/kg.

Discussion

Soil lead levels in Uhuru Owino seem to fall within internationally accepted screening levels and are at or below mean values in other cities globally. Abuja (Nigeria), Boston (USA), Brisbane (Australia), Glasgow (UK) and Stockholm (Sweden), for instance, have all been reported as having average city-wide soil concentrations exceeding 200 mg/kg.[83] In Owino Uhuru, the average soil lead concentration in accessible areas was 110 mg/kg.

The surface soil lead levels currently present in Owino Uhuru are unlikely to produce elevated BLLs, although elevated BLLs were reported at the site during operation or shortly after.[7,43,49] One possible explanation for the discrepancy could be a decline in surface level lead concentrations over time due to migration, meaning that current surface soil lead concentrations are not representative of soil lead concentrations while the smelter was operating. Lead is generally immobile in most soils, taking perhaps 700 years to halve in concentration in certain soil types.[66] A number of factors can influence its mobility, including pH, cation exchange capacity and texture.[84] Low cation exchange capacity (<2 cmol(+) kg), low pH, and a sandy texture are all associated with increasing lead mobility.[85,86]

However, even in locations where all or some of these conditions are present, very limited mobility of lead through soil profiles has been reported.[87-89] Teutsch et al., for example, found similar lead contamination profiles at the same location 15 years apart, with measurable but very small amounts of lead migrating at a rate of up to 1 cm/year.[89] The Haplic Lixisol soil in Owino Uhuru has an approximate cation exchange capacity of 1.8 cmol(+) kg and a pH of 5.9.[75] Its clay, silt and sand content are roughly 18%, 27% and 55%, respectively.[75] Thus, while these soils are more amenable to migration than others, migration is unlikely to account for any significant difference in surface soil concentration over the 10-year period between the opening of the smelter and the execution of this study.

A second possible explanation for the discrepancy between past BLLs and the current environmental levels present is that the primary exposure pathway to the community while the smelter was operating was the inhalation of airborne lead. Air lead concentrations would have declined immediately following the closing of the smelter. Reducing airborne sources of lead exposure near smelters has been strongly associated with declines in BLLs.[76,90,91] No data are available on airborne concentrations at the site, either currently or while the smelter was operating. Given the limited data on the smelter operations, it was beyond the scope the current study to model those concentrations. Elsewhere, air lead levels exceeding 1 μg/m^3 have been associated with BLL measurements above 10 μg/dL in children.[92] In Port Pirie, air lead concentrations were recorded at levels up to 21.44 μg/m^3.[93]

A separate likely source of exposure is associated with the workplace. Several residents with elevated BLLs were reported to have worked at the smelter, and anecdotal evidence indicates that children spent time in the facility during work hours. Additionally, take-home risk, or workers' inadvertent transporting of material on their person from the workplace to the home environment, could have also played a significant role as has been documented in multiple settings.[94-96] Interior surfaces were not assessed as part of this study, although those assessed as part of the Ministry of Health/CDC effort shortly after the closing of the facility found very low lead dust loadings.[49]

Given that the key sources of exposure were most likely associated with the operation of the facility, the need for mitigation work in Owino Uhuru may not be as pressing as presented elsewhere. The former facility very probably contains high levels of lead on site, which should be appropriately considered in any future land use plans.

Informal housing

There is a significant shortage of housing in urban areas in Kenya, with approximately 56% of urban dwellers living in informal settlements.[97] These settlements are often located on marginal lands, including areas prone to flooding or landslides.[97] In the case of Owino Uhuru, the settlement is an area that is characterized by industrial uses.[98] It is bordered on two of three sides by industrial facilities and is accessed primarily through industrial land. A master plan for the city currently under development envisions the reclamation of Owino Uhuru for industrial ends.[98] Industrial activity can play a critical role in economic development in LMICs, while siting residential areas sufficiently distanced from heavy industry, including lead smelting, could mitigate the most significant exposures.[34,99] Here, inhibiting the likely illegal occupation of industrial land may have mitigated much of the adverse impacts.

Setting public health priorities

A number of factors potentially influence the allocation of public health resources. These could include political forces, societal values and economic justifications, among others.[100-102] In response to the various challenges implicit in allocating resources in public health, there has been a general coalescence in recent decades around rational and transparent approaches. Foremost among these is the notion of evidence-based decision making, commonly articulated in burden of disease, cost effectiveness, or equity analysis approaches.[102] These approaches seek to make the best use of finite resources through "the conscientious, explicit, judicious and reasonable use of modern, best evidence in making decisions."[103,104]

Within global public health, it is arguable that the issue of environmental lead poisoning receives proportionately less attention relative to its impact than other public health risks. IHME, for instance, calculated that lead exposure attributed to 13.9 million disability-adjusted life years and 540,000 deaths globally 2016.[22] As context, this amounts to about 0.6 % of all disability-adjusted life years and about 1% of all deaths globally in the same year.[22] In addition, there are indications that this value may be an underestimation.[33] Major sources of environmental lead poisoning globally include industrial mining and smelting operations, lead-based ceramic glazes, and the recycling of used lead acid batteries.[105] Lead exposure also results in a range of adverse societal impacts, including increased rates of violence and decreased economic output, that are not captured in disease burden approaches.[15,106,107] Despite this considerable impact, there is currently no international convention or multi-lateral funding mechanism to support work related to lead contamination, resulting in a resource-poor environment for potential implementers.

In the context of disproportionately limited funding for lead interventions, the importance of evidence-based decision making is augmented. Put differently, the burden of proof to justify interventions should be high. The residents of Owino Uhuru are subject to myriad environmental health and livelihood risks common to life in informal settlements.[108] However, this study has found that those related to lead exposure were likely mitigated with the closing of the facility. The lead poisoning event in Owino Uhuru continues to receive national and international media attention despite the absence of current information regarding the health impact.[56,109,110] In evaluating the relative importance of an intervention in Owino Uhuru versus similar projects elsewhere, policy makers should be encouraged to utilize evidence like that presented here to form the basis of their decisions.

Limitations

A key limitation in the present study is the heavy reliance on pXRF measurements. The instrument was calibrated, and accuracy was confirmed in accordance with the manufacturer's instructions. When conducted in sufficient quantity, pXRF measurements have been shown to closely approximate wet chemistry techniques.[111] However, no samples from this study were sent for laboratory analysis. Measurements of the NIST reference material during the assessment found that the pXRF was functioning within acceptable standards.

A similar limitation relates to the spatial distribution of the surface soil measurements taken in the present study. The assessment, which focused on the residential area of Owino Uhuru, did not determine concentrations in the industrial areas to the west, south, and east of the facility. Moreover, within Owino Uhuru, significant assessment gaps are evident in Figure 1, particularly to the northwest of the facility. While this area is unlikely to present elevated concentrations resulting from aerial deposition, it is possible that the area could be contaminated by other means such as manual deposition of waste. However, interviews with community members did not indicate that this was the case. Nevertheless, that this area was not assessed here represents a limitation.

A separate limitation of the study is its reliance on the IEUBK model to estimate BLLs. A previous study carried out at the site found comparable environmental levels, as well as a statistically significant increase in BLLs associated with residence in Owino Uhuru compared to a control neighborhood.[49] Likewise, studies carried out elsewhere have found elevated BLLs resulting from similar environmental exposures.[112]

Conclusions

Surface soil lead concentrations within Owino Uhuru were within international screening levels at the time of the study, and lower than many urban areas globally. As noted above, a soil concentration of 850–2,500 mg/kg would be required for a 2-year-old child to have a BLL of 20 μg/dL. It is unlikely that this concentration was ever present in Owino Uhuru. Accordingly, there did not appear to be a compelling need for off-site exposure mitigation work at the time of the assessment. Proper occupational and engineering controls and better siting of the residential area would likely have mitigated the lead poisoning event. Future use of the land should consider probable high contamination levels onsite. The results of this study could be used to inform the discussion on public health spending and to inform any future intervention at the site.

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2.5. Paper Five

Bose-O'Reilly, S., Yabe, J., Makumba, J., Schutzmeier, P., **Ericson, B.,** & Caravanos, J. (2018). Lead intoxicated children in Kabwe, Zambia. *Environmental Research*, *165*, 420-424.

This short paper summarizes the results of a number of recent studies carried out to assess the extent of lead contamination in the city of Kabwe, Zambia. Kabwe city is centered around a century-old lead-zinc mine and smelter complex and represents one of the more severe global examples cases of population-wide lead poisoning. Mean BLLs in the population of 200,000 range from 35–82 µg/dL, with multiple individuals exceeding 100 µg/dL.

The author of this thesis organized the collection, analysis and presentation of data from Kabwe that ultimately contributed to the development of a USD 60 million World Bank loan intended in part to remediate the city. The study presented here relates to the overall thesis by documenting a site of historical importance with regard to human lead exposure. The severity and breadth of lead poisoning is Kabwe is unparalleled in the literature. The results of this study are discussed in Chapter 5 to highlight possible inaccuracies with burden of disease calculations.

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Lead intoxicated children in Kabwe, Zambia

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ABSTRACT

Kabwe is a lead contaminated mining town in Zambia. Kabwe has extensive lead contaminated soil and children in Kabwe ingest and inhale high quantities of this toxic dust. The aim of this paper is to analyze the health impact of this exposure for children. Health data from three existing studies were re-analyzed. Over 95% of children living in the most affected townships had high blood lead levels (BLLs) > 10 µg/dL. Approximately 50% of those children had BLLs ≥ 45 µg/dL. The existing data clearly establishes the presence of a severe environmental health crisis in Kabwe which warrants immediate attention.

1. Introduction

Kabwe is the fourth biggest town and capital of the central province of Zambia. The town has a long history of mining, which operated for more than 90 years and produced large quantities of lead (Pb) and zinc (Zn) until closure in 1994.

Lead is a toxic substance and chronic exposure causes serious ad verse health effects. The pathways of exposure are mainly ingestion of Pb contaminated soil and dust, but inhalation as a route of entry can also be significant. Pb can cause acute and chronic intoxication. High exposure can cause severe colic like abdominal pains, neurological symptoms, seizures, encephalopathy and finally death (World Health Organization, 2010).

Infants are at higher risk due to specific risk behaviors such as playing on bare soil, relevant hand to mouth activity and thus their oral uptake is greater compared with adults (World Health Organization, 2010). While high blood Pb levels (BLLs) have been associated with extensive adverse effects, evidence of low BLLs causing serious negative health effects is extensive and conclusive. The negative effect of Pb exposure during pregnancy to the fetus and during early childhood on the regular development of the brain has enormous adverse implica tions (Advisory Committee on Childhood Lead Poisoning Prevention, 2016; Needleman et al., 1990).

The CDC Reference Level for Pb is 5 µg Pb/dL blood (https://www.cdc.gov/nceh/lead/acclpp/blood lead levels.htm). Between 5 and 44 µg Pb/dL, actions to lower the body burden are recommended In the former "Kabwe lead poisoning management protocol" a Pb level of 20 µg Pb/dL was considered as minimum level for individual follow up (Project Technical Committee Zambia Consolidated Copper Mines Investments Holdings, 2006). The medical intervention level for chil dren is 45 µg Pb/dL. Children with confirmed Pb encephalopathy need to be hospitalized and treated individually (Advisory Committee on Childhood Lead Poisoning Prevention, 2016; Thurtle et al., 2014). Data from a large treatment survey in Nigeria indicates that oral chelation treatment with Chemet® (succimer, DMSA) is both safe and effective (Thurtle et al., 2014). However, chelation therapy without environ mental intervention may prove futile since re exposure will likely occur.

2. Environmental assessments

Lead contaminated soils in Kabwe pose a serious environmental hazard. In 2003 2006, the "Copperbelt Environment Project" analyzed over 1000 soil samples for Pb in various townships. The results showed,

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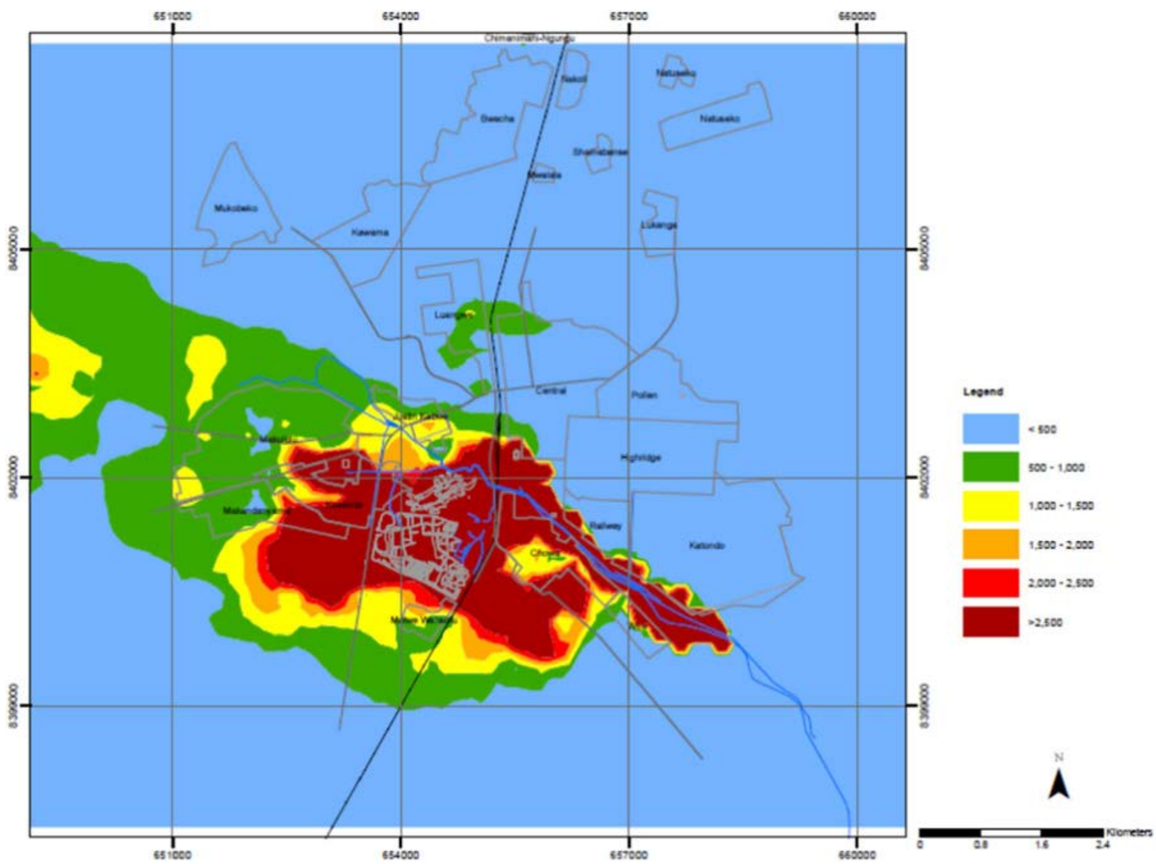

Fig. 1. Interpolated distribution of Pb in soil in the KSDS survey area based on district and township survey (Water Management Consultants Ltd, 2006).

that the soil "over a substantial area is highly contaminated with the metal". "Median Pb concentrations of soil in townships in the vicinity of the mine inducing Kasanda (3008 mg/kg), Makandanyama (1613 mg/kg), Chowa (1233 mg/kg), Mutwe Wansofu (1148 mg/kg), Makululu (870 mg/kg) and Luangwa (507 mg/kg) were recorded. All exceeded levels generally regarded as acceptable by international authorities with respect to residential areas" (Water Management Consultants Ltd, 2006).

The most affected townships are immediately adjacent to the former Kabwe mining complex and homes downwind from the smelter and the tailings (see Fig. 1).

Regrettably, the situation appears to have changed little in recent years as shown from work done by Pure Earth (formerly Blacksmith Institute) in 2014. Data shows townships close to the mining area are still polluted with Pb levels in soil well above recommended levels for residential areas. Soil samples analyzed with an Innov x Delta series X ray fluorescence by Pure Earth found median soil concentrations of 3212 mg/kg in Chowa, 6162 mg/kg in Kasanda, and 2286 mg/kg in Makululu (Caravanos et al., 2014; Office of Solid Waste and Emergency Response, 1994; Pure Earth, 2015). Citywide, surface soil Pb con centrations ranged from 139 mg/kg to 62,142 mg/kg, with a geometric mean concentration of 1470 mg/kg. Of the 339 soil tests, 86 readings (25.4%) were > 400 mg/kg.

This overview of results shows that the Pb contamination of soils in Kabwe is serious with townships close to the mining area being highly contaminated. Lead is not the only contaminant of concern in Kabwe; the different assessments showed high levels of cadmium (Cd) and Zn in the surrounding mining area and adjacent townships (Tembo et al., 2006; Water Management Consultants Ltd, 2006).

The aim of this paper is to analyze whether the high lead exposure has a health impact on children in Kabwe.

3. Health assessments

Presently, there are three information sources on childhood BLLs in Kabwe; (1) data from the Copperbelt Environment Project; (2) data from projects of Pure Earth and (3) data from a University of Zambia with collaborators from Hokkaido University, Japan. A summary of BLL data is provided below.

3.1. Copperbelt environment project

Commissioned by the Government of the Republic of Zambia, funded by World Bank, the "Copperbelt Environment Project" per formed the Kabwe Scoping and Design Study (KSDS) from 2003 to 2006. One of the aims of the KSDS was to update health and environ ment data for Kabwe. A specific aim was to reduce the geometric mean of BLLs substantially below 25 µg/dL for children in Kabwe. In the KSDS report, data from approximately 2500 participants are presented. Children were recruited by study nurses from the different townships. BLLs were especially elevated in children 0 7 years old. Nearly all children were above the reference level of 5 µg/dL and in some highly exposed townships over 50% of the children had BLLs at which medical treatment was warranted (see Table 1 (Water Management Consultants Ltd., 2006)).

The survey showed that the geometric mean BLLs in townships closer to the mining sites were higher. As shown in Table 1, the geo metric mean BLLs of children aged 0 7 years in the surrounding townships were: Chowa 31.7 µg/dL, Kasanda 32.8 µg/dL, Makanda nyama 38.2 µg/dL and Makululu 31.3 µg/dL (Water Management Consultants Ltd, 2006). The BLLs were in the range, where negative health effects are likely. Children aged 8 16 years and adults as well had increased BLLs, although lower than levels in children aged 0 7 years.

Table 1
Lead levels by township and year in μg Pb/dL blood.

Township	Year	N	Mean BLL	Median BLL	Geom. Mean BLL	Minimum BLL	Maximum BLL
Kasanda[1]	2003–2006	119			32.8	5.7	185
1	2007	216			29.7	4.4	121.8
1	2009	189			30.4	5.6	128
1	2012	189			34.5	2.1	121.8
4	2014	31	51.6	55.2	49.7	26.7	65.0+
2	2015	100	82.2	74.9		5.4	427.8
Makandanyama[1]	2003–2006	59			38.2	13.6	84.6
1	2007	66			36.6	3.3	119.4
1	2009	96			39.1	14.2	90.4
1	2012	291			33.9	15.1	112
Makululu[1]	2003–2006	408			31.3	3.4	114
1	2007	388			37.8	5.8	120
1	2009	493			25.1	3.6	121.8
1	2012	1134			33.9	2.1	128.4
4	2014	39	49.2	48,7	47,3	20,9	65,0+
3	2015	559	36.4			8.5	99.2
2	2015	129	57.1	51.1		9.4	388.7
Railways[1]	2003–2006	101			17.5	3.5	86.9
1	2009	4			15.8	9.3	98.1
1	2012	44			18.1	6.4	48.5
Waya[1]	2003–2006	37			26.1	6.5	58.9
1	2009	8			23	7.5	51
1	2012	139			27.2	6.3	121
Luangwa[1]	2003–2006	45			18.4	5	44.4
Chowa[1]	2007	43			26.6	5	121.8
1	2009	82			22.1	5.6	128
1	2012	349			29.2	4.8	100.2
4	2014	38	49.8	50.4	47.1	14.5	65.0+
2	2015	17	39	39.3		15.6	79.7
Katondo[1]	2007	4			30	7.8	46.4
1	2012	24			22.2	5	44.2
Mutwe Wa Nsofu[1]	2012	29			31.8		
4	2014	12	48.3	41.0	46.2	30.7	65.0+
Riverside[1]	2012	64			24.1	8.2	120
4	2014	7	49.9	50.4	48.3	29.8	65.0+
Kabwe[4]	2014	196	48.3	52.3	45.8	13.6	65.0+
Chililalila[3]	2015	244	35.5			0	98.3
4	2014	15	48.5	43.3	46.4	29.9	65.0+
Moomba[3]	2015	186	38.8			3.1	98.2
4	2014	17	45.6	44.4	43.1	22.3	65.0+
Zambezi[3]	2015	177	37.7			8.7	93.4
4	2014	30	40.3	37.0	37.2	13.6	65.0+

BLL = Blood lead level in μg/dl.
[1] data from Misenge projects.
[2] (Yabe et al., 2015).
[3] (Mbewe et al., 2015).
[4] (Caravanos et al., 2014).

The KSDS report shows that a high percentage of inhabitants had BLLs above the "predetermined level of concern" of 25 μg/dL. In detail it was extrapolated that 12378 children aged 0 7, 7919 children aged 8 16 and 3973 adults would have BLLs > 25 μg/dL (see Table 1) (Water Management Consultants Ltd, 2006).

Misenge Environmental and Technical Services Ltd (METS) performed several blood sampling sessions in various townships. As is shown in supplement 1, the geometric mean levels varied over time. There was no clear trend in the findings but differences in the geometric means among the townships were seen. The results showed that increased BLLs in Kabwe were widespread, especially in Makululu, Makandanyama and Kasanda where BLLs of children were constantly very high.

Recent data from METS was published in 2015 by the University of Zambia, Department of Public Health. In that study, BLLs in 1166 children under five years of age from Makululu township were measured with a LeadCare II™ portable blood instrument ((Magellan Diagnostics, Inc., N. Billerica, Massachusetts). The reported BLLs in Makululu were: (Mbewe et al., 2015):

- 99.4% of the children > 10 μg/dL
- 73.2% of the children > 25 μg/dL
- Geometric mean was 32.6 μg/dL (see Table 1)

Mbewe et al. (2015) determined a statistically significant association between age and BLLs. The authors concluded that affected areas would "require an expansive and integrated program of Pb exposure prevention"; the program should be "implemented by government and co operating partners" and that the program should also "take into account environmental management and disease surveillance" (Mbewe et al., 2015).

3.2. Pure earth projects

The previous findings were independently confirmed by BLL sampling performed in 2014 by Pure Earth (Caravanos et al., 2014). For the Pure Earth study, 196 children aged 2 8 years were selected from highly exposed townships. Their blood was tested with a LeadCare II™ analyzer. The mean BLL was 48.3 μg/dL and more than 50% of children would have needed medical treatment (see Table 1) (Blacksmith Institute, 2014; Caravanos et al., 2014). A substantial number (26.5%) of childhood BLLs exceeded the 65 μg/dL upper detection limit of the

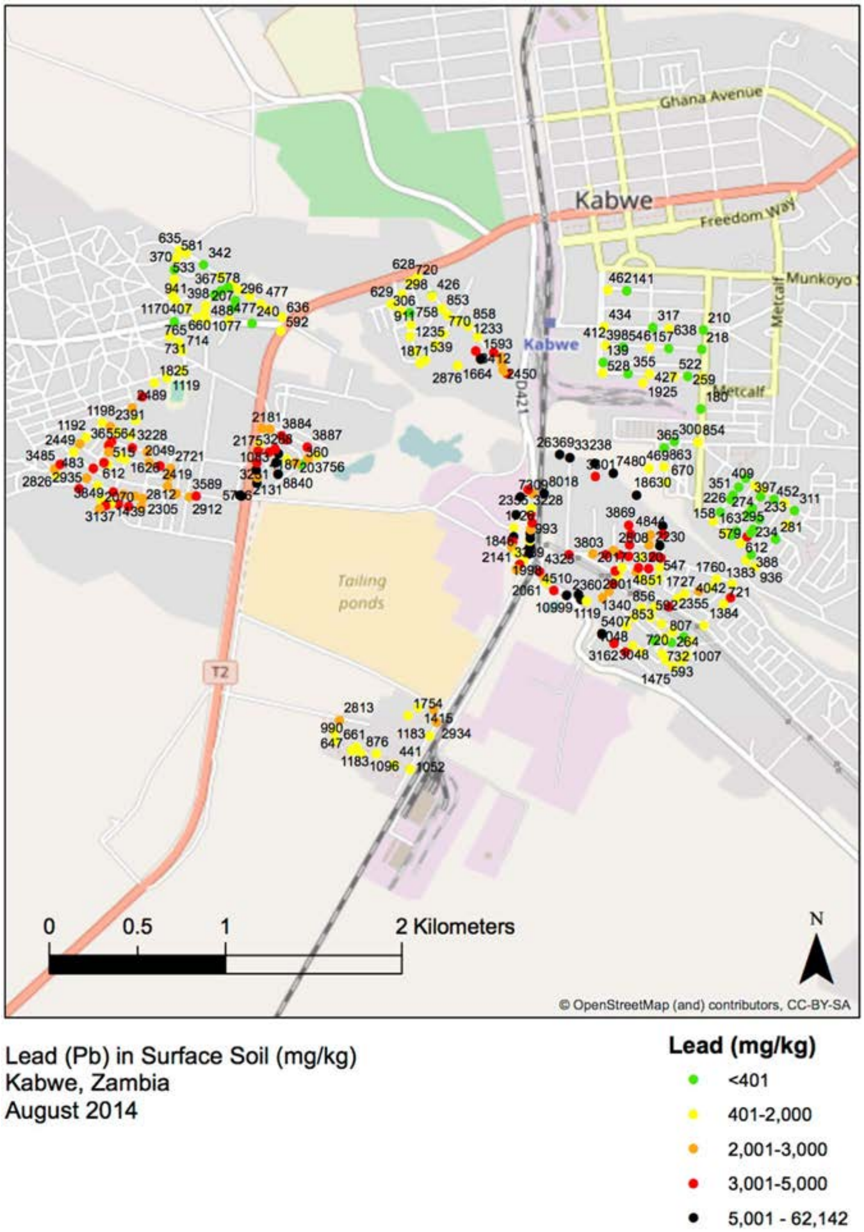

Fig. 2. Concentrations of lead in surface soil (August 2014).

LeadCare II™ instrument, which indicates that the poisoning was likely more severe than observed.

3.3. University of Zambia / JICA projects

Most recently a study in the three contaminated townships was conducted by the University of Zambia in collaboration with Hokkaido University. Children from the severely contaminated townships of Chowa, Kasanda and Makukulu were assessed. Blood lead levels were analyzed with ICP MS analyzer (7700 series, Agilent technologies, Tokyo, Japan) in 246 children under the age of seven (Yabe et al., 2015). The results showed that 100% of the sampled children had BLLs > 5 µg/dL. Moreover, high BLLs of ≥ 65 µg/dL were recorded in: Chowa 18%, Kasanda 57% and Makululu 25%. Eight children in Ka sanda and two children in Makululu had very BLLs of > 150 µg/dL, four children had extremely high BLLs of ≥ 300 µg/dL (for details see Table 1 and supplement 2).

Findings from Yabe et al. (2015) revealed extensive Pb poisoning in children in Kabwe and recommended chelation therapy, especially in children with BLL exceeding ≥ 45 µg/dL. This was critical so as to curtail the pernicious health impact of Pb poisoning in children. Moreover, through an individualized follow up plan of the affected children at their homes and regular health check ups at local health centers, it was observed that the quality of life of the affected children could be improved. The authors also observed that early detection and intervention could be useful to minimize overt cases of Pb poisoning such as persistent seizures, mental retardation and death. This would require enhanced community awareness of Pb poisoning and urgent interventions in the affected areas, with the full participation of local townships and various stakeholders for the sustenance of remedial measures.

Children with increased BLLs can develop clinical signs and symp toms of Pb intoxication. The data presented in this paper offers the most current and comprehensive reporting of BLLs in this highly con taminated city in Zambia. Three distinct groups of researchers have presented what appears to be continuing Pb poisoning on a major scale.

There is only one set of data set from UNZA / JICA containing some information about clinical findings from children in Kabwe. Data cap tured in questionnaires showed that 10 or more percent of children observed in 2012 had reported signs and symptoms, typical for chronic

Pb intoxication, such as anemia, intermittent abdominal pains, limb pains, memory problems, headaches, weakness in hands and feet, and seizures or convulsions (unpublished data). The nurses employed by METS caring for 200 children with increased BLLs reported verbally that they observed similar symptoms in children. Nurses and health care workers from Makululu, Kasanda and Chowa health centers report that children have severe intermittent abdominal pains, seizures, are smaller and do not develop well compared with other children. These are first indicators that the children in Kabwe suffer from Pb poisoning severely.

4. Discussion

The existing data clearly establishes the severity of Pb exposure in Kabwe. Environmental data also support the observation that in certain housing areas of Kabwe, the recommended tolerable soil Pb level of 400 mg/kg is clearly exceeded (see Fig. 1 and Fig. 2). The soil levels measured by Pure Earth 2014 are in general higher compared to those reported by KSDS in 2006. We suspect this results from enhanced granularity now available with a portable XRF (rather than an actual increase in concentrations).

Nearly all children in the townships close to the former mining area had BLLs ≥ 20 μg/dL, which is a value where urgent action is demanded to decrease the exposure immediately according to the KSDS study protocol. Since soil is the major pathway of exposure in Kabwe the most important task is to decrease exposure for all children by immediate and appropriate remediation.

In the most affected townships, approximately 50% of the children have BLLs ≥ 45 μg/dL, which is the threshold value above which medical antidote treatment is recommended. The former interventions of the KSDS project did not show any long term effect in reducing the environmental pollution nor the BLLs of children. In Makululu, Makandanyama and Kasanda, BLLs of children are even higher compared with other townships. Large proportions of children are not only highly exposed, but have high to extremely high levels of Pb in their bodies. From the KSDS survey it can be estimated that approximately 60% of children with BLLs ≥ 25 μg/dL might have BLLs of levels ≥ 45 μg/dL. According to threshold levels and international recommendation, the exposure has to be considerably reduced and the children need medical treatment. Children with BLLs ≥ 45 μg/dL should receive medical treatment (Agency for Toxic Substances and Disease Registry, 2010). However medical treatment without environmental and educational intervention will not be enough to sustain a safe and healthy population. A World Band funded project (ZMERIP) will soon enable the Government of Zambia to take action in Kabwe, in cuding reduction of exposure and treatment for children.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.envres.2017.10.024.

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3. Chapter Three: Outcomes

Chapter 3 consists of the following 4 studies:

3.1 **Ericson, B.,** Landrigan, P., Taylor, M. P., Frostad, J., Caravanos, J., Keith, J., & Fuller, R. (2016). The global burden of lead toxicity attributable to informal used lead-acid battery sites. *Annals of Global Health*, *82*(5), 686-699.

3.2 Estrada-Sánchez, D., **Ericson, B.,** Juárez-Pérez, C. A., Aguilar-Madrid, G., Hernández, L., Gualtero, S., & Caravanos, J. (2017). Pérdida de coeficiente intelectual en hijos de alfareros mexicanos. *Revista Médica del Instituto Mexicano del Seguro Social*, *55*(3), 292-299.

3.3 **Ericson, B.,** Dowling, R., Dey, S., Caravanos, J., Mishra, N., Fisher, S., ... & Taylor, M. P. (2018). A meta-analysis of blood lead levels in India and the attributable burden of disease. *Environment International*, *121*, 461-470.

3.4 Caravanos, J., Carrelli, J., Dowling, R., Pavilonis, **B., Ericson,** B., & Fuller, R. (2016). Burden of disease resulting from lead exposure at toxic waste sites in Argentina, Mexico and Uruguay. *Environmental Health*, *15*(1), 72.

Chapter 3 builds on the exposure estimates presented in Chapter 2 by quantifying those exposures in immediately legible metrics. It forms a central part of this thesis by demonstrating that lead exposure results in a large and likely underestimated disease burden.

Chapter 3 includes four papers that each take slightly different approaches to calculating the disease burden attributable to lead exposure. In all cases the findings indicate that the burden is greater than the existing literature might indicate. Underlying this difference are two primary drivers. The first is that at present the biological burden of lead in LMICs is likely underestimated. Specifically, due in part to a dearth of studies on BLLs in LMICs, global burden of disease (GBD) calculations rely heavily on regression models with a limited number of independent variables, and exclude some of the more significant inputs, such as informal battery processing. A second reason is that the primary pediatric health outcome of lead exposure, impeded brain growth, may not be adequately captured in GBD estimates.

Chapter 3 utilizes two common metrics. The first, intellectual quotient (IQ) decrement captures the amount of intelligence effectively lost, or foregone, because of pediatric lead exposure. Lead is mistaken by the body as calcium, absorbed into the blood, and passes the blood brain barrier where it impedes brain growth. The most significant proportionate losses are likely at low BLLs with damage continuing, though tapering, as exposures increase. The second metric utilized in Chapter 3 is the Disability Adjusted Life Year (DALY). This is a metric that captures both morbidity and mortality and was developed by the World Bank to compare disparate health outcomes. Each paper in Chapter 3 uses a different approach to calculate the inputs used in the burden of disease calculations. By doing so the Chapter implicitly tests the sensitivity of key parameters, thereby building a more robust overall argument about the underestimation of current calculations.

3.1. Paper Six

Ericson, B., Landrigan, P., Taylor, M.P., Frostad, J., Caravanos, J., Keith, J., & Fuller, R. (2016). The global burden of lead toxicity attributable to informal used lead-acid battery sites. *Annals of Global Health*, *82*(5), 686-699.

Paper 6 presents the first large-scale estimate of the scale of the informal used lead acid battery (ULAB) recycling industry across 90 LMICs. The total amount of lead extracted from ULABs annually is calculated and an estimate of the number and size of informal smelters is developed across different sub-regions. Finally, based on assessments of informal smelters carried out at 28 sites in 12 different countries, exposure scenarios are used to calculate an attributable disease burden. The results of the paper formed the basis of the lead exposure assessment in *The Lancet's* 2016 *Commission on Pollution and Health* and were used recently by the World Bank in their development of Country Environmental Assessments in Bangladesh and Pakistan.[1,2]

The study provides the fundamental arithmetic underpinning the argument that ULAB recycling poses a significant exposure risk to populations in LMICs. In short, while nearly 100 % of batteries are recycled, only 50 % can be accounted for by the formal sector. Thus, the implication is that the balance is recycled by the informal sector, often in residential areas, posing a critical risk to human health.

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ORIGINAL RESEARCH

The Global Burden of Lead Toxicity Attributable to Informal Used Lead-Acid Battery Sites

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Abstract

BACKGROUND Prior calculations of the burden of disease from environmental lead exposure in low- and middle-income countries (LMICs) have not included estimates of the burden from lead-contaminated sites because of a lack of exposure data, resulting in an underestimation of a serious public health problem.

OBJECTIVE We used publicly available statistics and detailed site assessment data to model the number of informal used lead-acid battery (ULAB) recyclers and the resulting exposures in 90 LMICs. We estimated blood lead levels (BLLs) using the US Environment Protection Agency's Integrated Exposure Uptake Biokinetic Model for Lead in Children and Adult Lead Model. Finally, we used data and algorithms generated by the World Health Organization to calculate the number of attributable disability adjusted life years (DALYs).

RESULTS We estimated that there are 10,599 to 29,241 informal ULAB processing sites where human health is at risk in the 90 countries we reviewed. We further estimated that 6 to 16.8 million people are exposed at these sites and calculate a geometric mean BLL for exposed children (0-4 years of age) of 31.15 µg/dL and a geometric mean BLL for adults of 21.2 µg/dL. We calculated that these exposures resulted in 127,248 to 1,612,476 DALYs in 2013.

CONCLUSIONS Informal ULAB processing is currently causing widespread lead poisoning in LMICs. There is an urgent need to identify and mitigate exposures at existing sites and to develop appropriate policy responses to minimize the creation of new sites.

KEY WORDS informal economy, lead poisoning, low- and middle-income countries, soil pollution, disability adjusted life years, recycling

INTRODUCTION

Contaminated soils at polluted "hot spots" active and abandoned mines, smelters, industrial facilities, and chemical waste sites threaten the environment and human health globally. In high-income countries (HICs), substantial progress has been made toward identifying and remediating hazardous waste sites and thus in reducing exposures and disease. In low- and medium-income countries (LMICs), by contrast, the extent and severity of soil contamination at these sites has not been adequately mapped or quantified.[1] Information on the burden of disease attributable to hazardous exposures at contaminated

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sites has not previously been available for inclusion in estimates of the global burden of disease by either the Institute for Health Metrics and Evaluation (IHME) or the World Health Organization (WHO).[2,3]

Toxic Sites Identification Program. To close the information gap regarding soil pollution at industrial hot spots in LMICs and its effects on human health, Pure Earth (PE; formerly Blacksmith Institute) launched the Toxic Sites Identification Program (TSIP) in 2008.[4]

The central element of TSIP is a protocol for rapid field-based identification and assessment of hazardous waste sites in LMICs. This protocol has been adapted from the standard source-pathway-receptor model for field assessment of toxic sites in use by the US Environmental Protection Agency (USEPA) and was specifically adjusted by PE to accommodate field application in LMICs by nonprofessional local investigators trained through the program.[5,6] At each site, environmental samples are collected, photographs are taken, and key characteristics are documented. Completed assessments are entered into an online database. To date 2591 sites in 49 countries have been assessed.

The basic approach is to identify a single key pollutant at each site. The key pollutant is the dominant contaminant at a site whose concentrations are most significantly elevated above relevant environmental standards. Lead, mercury, hexavalent chromium, and highly toxic environmentally persistent pesticides have been the key pollutants most commonly identified. Estimates of environmental contamination and disease burden are attributed solely to the key contaminant and not to any other hazardous materials that may be present at a site. This bias likely underestimates the true environmental burden of disease.

PE has collaborated with local partners and secured support from national and international funders to remediate some of the most hazardous sites identified through TSIP. In these remediation efforts, extensive site mapping is undertaken to characterize the spatial distribution of contaminants, consultations are held with local area residents to help design the most effective and appropriate interventions, and biological samples of residents are taken, as appropriate, before and after remediation.

From 2012 to 2014, PE and researchers from the Mount Sinai School of Medicine in New York City, used data collected as part of the TSIP to calculate the burden of disease resulting from exposures at contaminated sites. The effort focused on 3 countries in Southeast Asia: India, Indonesia, and the Philippines. Chatham-Stephens et al.[7] used sampling data from individual sites and existing demographic information to conduct an exposure assessment. Dose-response was calculated using USEPA reference doses and slope factors[8] for noncarcinogenic and carcinogenic chemicals, respectively. An estimated incidence of disease was thus determined for various age groups. Finally, deaths (years of life lost [YLL]) were determined, and appropriate WHO disability weights (DW), discounting, and age weights were applied to calculate resulting years lost due to disability (YLD). YLD and YLL were summed at all sites and presented in the aggregate. The authors found that 828,722 disability adjusted life years (DALYs) resulted from contaminated sites in those 3 countries alone. A second paper[9] applied this same method to 3 additional countries in Latin America.

Sites Captured by TSIP. The TSIP method is not designed to survey all contaminated sites in a country. Rather sites are prioritized based on their perceived effect on human health. Moreover, finite resources limit the number of site visits. Relevant government agencies, academics, and others are interviewed to assist investigators to identify sites, although a systematic identification process similar to the National Priorities List of USEPA[10] is not in place. Underestimation of the number of sites and the attributable burden of disease therefore results.

To obtain data on the number of sites captured by TSIP relative to the total number of contaminated sites in a country and thus to assess the degree of underestimation, PE conducted a systematic census of hazardous waste sites in Ghana in 2014-2015.[11] Ghana was selected for this analysis because it is an LMIC with a heterogeneous industrial base and a highly urban population (51%).[12] Assessors targeted a randomly selected number of geographic quadrats for comprehensive assessment. The investigators physically walked all accessible streets in each quadrat to visually identify sites. Visual identification was supplemented by field-based sampling with portable x-ray fluorescence instruments to test soils for metals (InnovX Delta Series, Olympus, Bridgeport, CT, USA). The investigators identified 72 sites in the sampled quadrats. They then extrapolated these data to the country as a whole using 1 of 2 methods. The first method (regional), which used cluster random

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sampling, estimated the regional number of waste sites first and then summed them to find a total in Ghana within a particular confidence interval. The second method (countrywide) estimated the total number of sites in Ghana, then "allocated" them to each region.[11]

Based on this extrapolation, the investigators estimated there were between 812 and 3075 contaminated sites in Ghana. By comparison, TSIP had screened 215 sites in Ghana through 2014. It was therefore calculated that the percentage of sites captured by TSIP was between 7% and 26%. On a population basis, it was estimated from these data that in Ghana there are between 31 and 115 contaminated sites per 1 million population. Of the sites identified in Ghana, 37% presented lead as the key pollutant.[11]

Informal ULAB Processing. The principal global use of lead today is in manufacture of plates and components for lead-acid storage, lighting, and ignition (SLI) batteries, which accounts for 85% of global lead consumption.[13] The lead used in batteries is derived from 2 main sources: primary lead mining and smelting (ie, newly mined from lead containing ore) and secondary lead smelting from used SLI batteries and other sources. Formal industrial lead smelters (both secondary and primary) are necessarily subject to the regulatory regimes of their host countries. Although many continue to present significant occupational hazards,[14 17] formal smelters at a minimum tend to be located in industrially zoned areas and have in place some sort of emissions and discharge control infrastructure.

Informal industry by contrast is characterized by lack of adherence to regulation, including zoning and pollution controls.[18,19] Accordingly, a higher number of informal industrial sites are located within residential areas than their formal counterparts and pollute more per unit than their formal counterparts. The TSIP program has encountered significant difficulty in assessing these sites, as operators have tended to be uncooperative. Little systematic research other than TSIP investigations has been undertaken to identify and map ULAB sites globally. Informal ULAB processers tend to be much smaller in scale and more widely distributed than their formal counterparts and are prone to changes in location in response to regulatory efforts or other forces.[20] Consequently it is a challenging exercise to develop a truly accurate assessment of their location and number.

Disability Adjusted Life Years. DALYs are a standard metric for burden of disease calculations. DALYs are used, most significantly, by the WHO and the IHME in their respective periodic updates of the global burden of disease (GBD). DALYs are calculated for a range of health outcomes including infectious and chronic diseases; however, their use with regard to contaminated sites remains limited.[2,21,22]

Environmental lead exposure can result in a number of health outcomes including renal effects and cardiovascular disease (CVD) in adults, and neurologic effects in children.[23] In their most recent GBD report, IHME modeled lead exposures resulting from aerial deposition of leaded gasoline.[2] Due in part to a lack of information on hotspots globally, IHME did not endeavor to estimate DALYs resulting these exposures. Based on exposure to leaded gasoline deposition only, IHME calculated between 4,199,925 and 15,594,412 DALYs for 2015.[2]

For the purpose of their analysis, IHME developed a complex algorithm that uses a range of required covariates.[2] Importantly, the IHME algorithm relies on both blood and bone lead estimates. Lead is a calcium analog and can be absorbed by bone as a result of chronic exposure.[24] IHME uses bone lead estimates to model both CVD and resulting deaths in older age groups. BLLs are only used by the IHME algorithm for the purpose of computing DALYs resulting from pediatric exposure. WHO by contrast presents a series of publically available tools that are more amenable to determining the attributable risk of lead exposure at specific sites using blood lead levels (BLLs) only.[25]

We endeavored to calculate the prevalence and resulting burden of disease of informal ULAB processors in LMICs.

METHODS

In the absence of a comprehensive dataset on ULAB sites, we used relevant existing statistics to infer an estimate of the number of sites and resulting exposures. We calculated the annual amount of lead entering the secondary market in each country and subtracted the amount of lead known to be recycled formally. We assumed the difference between these 2 numbers was the total amount of lead recycled informally.

Geographic Scope. Our analysis was restricted to LMICs only, defined by the World Bank as those countries with an annual per capita gross national income below US $12,475 per year.[26] Countries were further excluded for any one of the following reasons: Data were unavailable or could not be reasonably estimated; active countrywide conflict (eg, Syria) where models were unlikely to apply; Balkan countries, which although within the income guidelines, are unlikely to have widespread informal ULAB processing due to their proximate location to richer European neighbors. The remaining number of countries covered by the analysis presented herein is 90.

Estimating the Tonnage of Lead Entering the Informal Sector. To calculate the amount of lead entering the secondary market, we summed the metric tons generated annually from 6 different sources: automobile starter batteries (cars, trucks, buses, and other automobiles); motorbike starter batteries; uninterrupted power supplies (UPS); other transport vehicles (eg, forklifts); electric bicycles; and other vehicles and applications, such as green energy storage. Data provided by the Organisation Internationale des Constructeurs d'Automobiles (OICA) was used to determine the number of cars and trucks (including buses and other large automobiles) in use.[27] The number of motorbikes in use was provided by the ministries of transport for several Southeast and South Asian countries, and estimated elsewhere. We used a widely quoted estimate of 200 million electric bicycles for China.[28] We assumed zero electric bicycles for all other countries.

We used a model developed by the International Lead Association (ILA) to determine the amount of lead generated annually by each type of vehicle. After accounting for the weight of other materials in each battery, the calculator assumes a weight of 8.4 kg of lead for car batteries, 15 kg of lead for truck batteries, and 4 kg for both bicycle and motorbike batteries. Batteries expire more quickly in warm climates.[29,30] As most countries covered by the analysis fit within this category a life span of 2 years is assumed. Because trucks typically have 2 batteries, they are expected to produce $\geq$1 ULAB annually.

The remaining metric tons from motorbikes as well as the contribution from other sources is estimated using their relative proportions of market share provided by the Industrial Technology Research Institute (ITRI).[31] Car and truck batteries, for which we were provided values by the OICA, were assumed to comprise 52% of the total lead tonnage used in batteries. The remaining 48% was distributed among the following 5 categories: motorbike starter batteries (6%); UPS (10%); other transport vehicles (eg, forklifts) (9%); electric bicycles (9%); and other vehicles and applications (14%).

The total amount of lead formally processed in secondary smelters in each country (provided by the US Geological Service)[32] was subtracted from the total amount generated. The difference between these 2 values was assumed to enter the informal sector. This approach to calculating informal lead production as described is generally consistent with those used elsewhere.[33,34] The statistical calculation was set out as:

$$\text{Total Pb}_i = 1/52 * \left[\frac{\text{Cars}_i * .0084}{2} + \frac{\text{Trucks}_i * .015}{1}\right]$$

$$\text{Total Pb}_i - \text{FormRcyl}_i = \text{InformRcyl}_i$$

where:

Cars_i is the number of cars in use in country i
Trucks_i is the number of cars in use in country i
TotalPb_i is the total lead in metric tons entering the secondary market in country i
FormRcyl_i is metric tons of lead processed in the formal sector i
InformRcyl_i is metric tons of lead processed in the informal sector i

Estimating the Number of Informal ULAB Sites. We used 2 separate methods for estimating the number of informal ULAB processing locations. In the first approach (method 1), we used the tonnage estimates from the model described previously and estimated yearly output of informal secondary smelters. This method is described in detail later. In the second approach (method 2), we used the Ghana extrapolation to determine the total number of informal ULAB sites on a population basis.[11] The Ghana effort found 31 to 112 contaminated sites for every 1 million residents in the country, with 37% of all sites being contaminated with lead from informal ULAB processors. Here we assumed (as a lower minimum estimate) 31 sites, and accordingly 11.47 informal ULAB sites, for every 1 million residents in a given country. We applied the weights described below to this initial number.

In the first approach (method 1), we assumed that 100% of lead not entering the formal sector is recycled at informal ULAB processing sites. We defined these as informal workshops typically located in residential areas where battery casings are ruptured, often manually. Plates are extracted and ingots are formed through rudimentary smelting techniques with little protective equipment or emissions controls. We modeled 3 sizes of informal processors (small, medium, and large) each with a

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set annual volume (100, 500, and 1000 metric tons, respectively). We divided the overall stream of lead into these 3 groups, assuming 50% flowing to small operators, 35% flowing to medium operators, and 15% flowing to large operators. In the absence of an existing body of research on the prevalence and throughput of informal sector operators, we based the above estimates on limited PE site assessment data and consultations with experts. This calculation provided us with an initial estimate of the total informal processing sites in a country:

$$\frac{\text{InformRcyl}_i * .5}{100_{sm}} + \frac{\text{InformRcyl}_i * .35}{500_{sm}} + \frac{\text{InformRcyl}_i * .15}{1000_{1g}} = \text{IntTotalSites}_i$$

where:

InformRcyl_i is the metric tons of lead processed in the informal sector in country i
IntTotalSites_i is the initial estimate of the total number of informal lead processing sites in country i
sm is small processors
md is medium processors
lg is large processors

To account for the influence of other factors on the total number of sites, we further weighted our initial estimate by 4 values: per capita gross domestic product at purchasing power parity (GDP PPP); percent of the country living in urban areas; percent of the overall economy derived from the informal sector; and percent of the overall economy derived from mining, manufacturing and utilities (MMU).

The rationale for inclusion of these elements is as follows:

1. GDP PPP: In the context of LMICs, the relatively high price of lead affords a profit margin for recyclers even at low volumes. As the income of countries increases, this margin diminishes and larger volumes are required to sustain profit, limiting the feasibility of informal operation. We use values provided by the World Bank for 2013.[35]
2. Percent of the economy based in the informal sector: A higher rate of informality for the economy as a whole was taken to indicate a higher rate of informality in the recycling sector. We used values presented by Schneider et al on the size of each country's informal sector in 2006 and 2007.[36]
3. Percent of the population living in urban areas: People are more likely to come into contact with informal processing sites in urban areas than rural areas. We used urban percentage rates provided by the World Bank for 2013.[12]
4. Percent of the economy based in MMU This parameter, provided by the United Nations Statistics Division (UNSD),[37] was the best indicator that we could identify of the overall level of industrial activity in a country. There are linkages between the formal and informal economies in LMICs, with growth in former potentially associated with growth in the latter.[38] We assumed this was the case regarding ULABs and used MMU data provided by UNSD for 2013.

Of the 4 parameters, GDP PPP was the most heavily weighted (0.75). No other parameter was more closely associated with the presence of informal battery processing sites than income. In HICs, for instance, these sites are essentially nonexistent, whereas they proliferate in LMICs. Thus this parameter is inversely weighted to increased income. GDP PPP is followed by percent urban (0.1) and equal weightings for percent MMU and percent informal (0.075). The marginally higher weighting for percent urban is intended to reflect the relative importance of population density on increasing risk of exposure. Weights affect the result based on their relative difference from values taken for the country of Ghana. Given that Ghana is the only country for which we have a reasonable country estimate, we assumed that countries that are poorer, more urban, more reliant on industry, and had higher rates of informality are more likely to have informal battery processing sites. We assumed the opposite for countries that are richer, less urban, less industrial, and less informal. The equation is as follows:

$$\text{IntTotalSites}_i * \left(\text{GDPWt}_i + \text{InfWt}_i + \text{UrbWt}_i + \text{MMUWt}_i\right) = \text{TotalSites}_i$$

where:

IntTotalSites_i is the initial estimate of the total number of informal lead processing sites in country i
TotalSites_i is the final estimate of the total number of informal lead processing sites in country i
GDP Wt_i is the weighted value for income i
Inf Wt_i is the weighted value for informality i
Urb Wt_i is the weighted value for urbanization i
MMU Wt_i is the weighted value for industrialization i

Estimates were provided on a subregional level to account for limited transboundary movement of

ULABs. ULABs were categorized as hazardous waste under the Basel Convention and accordingly documentation should be kept of transboundary import and export transactions. An assessment of these records was outside of the scope of the present effort and would be unlikely to affect the most significant results. By way of example, about 5.1 million metric tons of lead are formally recycled globally each year.[32] The countries covered by this study account for more than 2.3 million tons, leaving the balance to be recycled in HICs. The United States is the second largest secondary processor of lead in the world after China, recycling 1.1 million tons annually, however, high car ownership rates preclude the United States as a reliant on an overseas stream of lead.[32] The situation is similar in other industrially developed countries with significant secondary lead smelting (eg, Canada, Germany, Spain, Japan, Italy), thus we excluded these countries as relying on an overseas supply of batteries.

Estimating Exposures at Informal ULAB Sites. ULAB sites present a heterogeneous set of conditions. To estimate population exposures to lead at these sites we developed a series of exposure scenarios, reflecting first the relative size of populations exposed and second the severity of exposure.

To capture number of exposed people we modeled operations of 3 different sizes: those where 200 people were exposed (small), those where 500 people were exposed (medium), and those where 2000 people were exposed (large). As previously noted, we assumed that small operations account for 50% of ULAB recyclers globally; that medium account for 35%, and that large account for 15% of all operations.

To capture the severity of exposure we divided the populations at each site into 3 exposure scenarios (low, medium, and high). Low represents soil concentrations of 850 mg/kg, medium represents concentrations of 2500 mg/kg, and high represents concentrations of 5000 mg/kg. All sites (small, medium, and large) were divided identically, with 50% of populations falling in the lower group (850 mg/kg), 35% falling in the medium exposure group (2500 mg/kg), and 15% falling in the high-exposure group (15%).

Exposure scenarios were based on the results of 28 assessments carried out by PE at informal ULAB sites in the following 12 countries: Argentina (n = 4), Bangladesh (n = 2), the Dominican Republic (n = 1), Ghana (n = 1), India (n = 4), Indonesia (n = 2), Kenya (n = 2), Peru (n = 3), the Philippines (n = 4), Senegal (n = 1), Uruguay (n = 5), and Vietnam (n = 1). Eight of these assessments were comprised of >20 target soil samples (range, 20-553 samples, analyzed with a handheld InnovX Delta Series x-ray fluorescence instrument) and were considered "detailed" assessments for our purposes. A second group, considered "limited," was comprised of 10 assessments with ≥6 composite samples each, and 10 assessments with ≥3 composite samples each. All assessments were divided into tertiles based on the relative severity of contamination. Tertiles were then averaged across all assessments equally resulting in values for the low, medium, and high exposure groups (810, 2626, and 6993 mg/kg, respectively). Detailed assessments revealed much higher values for all exposure groups (1536, 6465, and 16,257 mg/kg, respectively) when compared with limited assessments (509, 1150, and 2937 mg/kg, respectively). In an effort to remain conservative, we capped the soil concentrations for each group to the values given above (850, 2500, and 5000 mg/kg, respectively).

We used the spatial extent of contamination from detailed assessments only to guide the development of our estimates of exposed populations. We drew concentric circles with radii of 100 m, 200 m and 300 m from the center of each site to determine three site sizes (small, medium, large). We then applied NASA Socioeconomic Data and Applications Center gridded population density data[39] to the actual extent of each site to determine the number of exposed people. In one case (Vietnam) we deferred to PE population estimates from the detailed assessment. We averaged the sites in each group to determine the relative population sizes (251, 1053, 2029) and adjusted these to the slightly more conservative values of 200, 1000, and 2000 for small, medium and large sites, respectively. To determine the proportion of each population in each exposure scenario (lower, medium, high) we drew three equidistant concentric circles from the center of each site. We assumed increased severity with increased proximity. The resulting rings for each scenario made up 11%, 33% and 55% of each defined area. We adjusted these slightly to 15%, 35% and 50% to account for the diffuse nature of sources at a ULAB sites and assumed even population distribution across the whole area.

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Calculating Blood Lead Levels. Humans can become exposed to lead in soil either directly or in dust form through ingestion, inhalation, and dermal contact, with direct ingestion being the most common dominant pathway. Most inhaled soil, for instance, is trapped and ingested before entering the lungs.[40] Efforts have been made in HICs to estimate the amount of soil children ingest daily,[41,42] although comparatively little work has been done in this area in the LMIC context. Those studies that do exist indicate daily intakes of perhaps 4 to 5 times those in the United States.[43,44] In addition to incidental ingestion, the deliberate ingestion of soil, or geophagia, remains a common practice in LMICs.[45]

We used 2 separate methods to calculate the BLLs of exposed groups. To calculate the BLL for children, we used the USEPA's Integrated Exposure Uptake Biokinetic Model for Lead in Children, Windows version (IEUBK).[46] We adjusted the default intake values for soil from 85 to 135 mg/day to 250 to 400 mg/day to account for regional differences in soil ingestion.[43,44] All values for all other pathways were adjusted to zero. To calculate BLLs for adults, we used the USEPA Adult Lead Methodology (ALM).[46] We adjusted default intake values for soil from 50 to 200 mg/day and allocated an exposure frequency of 365 days instead of the occupationally adjusted 219.

An increase proportional to that of children would have resulted in an intake value of 148 mg/day, however Harris and Harper (2004) estimate intakes of 400 mg/day for adults as well as children. We opted for a more conservative approach and arbitrarily selected 200 mg/day.

Both the IEUBK and the ALM rely on age-specific exposures. To estimate the proportion of an exposed population within a given age group, we used unpublished age distribution values provided by the IHME and used in their most recent GBD calculations.[2] Using this method, we calculated BLLs for each of 17 different age groups in each of our exposure scenarios (low, middle, high).

Calculating Attributable Disability Adjusted Life Years. To calculate the DALYs resulting from lead exposure at informal ULAB recycling sites we applied publicly available WHO methods to the exposure estimates above. We did so for both resulting CVD and intellectual disability. We further augmented these calculations with a separate calculation for YLD for pediatric IQ decrement, following Chatham-Stephens and not used by the WHO.[7]

DALYs from Cardiovascular Disease. We used the Microsoft Excel-based calculator developed by the WHO to determine the attributable fraction of the CVD burden from lead exposure.[25] We input the geometric mean BLL of all adult exposures at all sites (21.2 μg/dL) to determine the sex-disaggregated attributable fraction of ischemic, cerebrovascular, hypertensive, and other heart diseases. We then applied these values to the most recent WHO GBD report country estimates.[21] Because the WHO provides values at the country level, we first scaled their results to our numbers of the exposed populations at each site for both low- and high-exposure estimates. As a result, the CVD rate at ULAB sites is assumed to be the same as the national level. No effort was made to estimate any increase in the overall CVD rate at sites as a result of disproportionate lead exposure. We did not adjust for comorbidity.

DALYs From INTELLECTUAL Disability. To determine the number of DALYs attributable to intellectual disability from lead exposure, we used the WHO Microsoft Excel-based calculator. We input the geometric mean BLL for all children (ages 0-4) at all sites (31.15 μg/dL) and entered the appropriate regional adjustment. This calculation resulted in an incidence rate of 6.49 to 8.34 cases of attributable mild mental retardation (MMR) from pediatric exposure for every 1000 people, and prevalence of 32.45 to 41.72 cases per 1000 people. The calculation also resulted in an incidence rate of 32.45 to 41.72 cases per 1000 for prenatal exposure, depending on region. For prenatal exposure, incidence and prevalence were taken to be the same.

We then used 2 separate methods for the calculation of DALYs from intellectual disability. The first (incidence) was developed and used by the WHO for GBD calculations from 1990 to 2010. We calculated the number of childhood and prenatal exposures at each site by applying IHME population distributions and World Bank birth-rate estimates to exposed populations.[2,47] We assumed the same demographic profile at the site as at the country level. We then applied the population-adjusted incidence values above to those estimates to determine the number of cases. Finally, we adapted the DALY spreadsheet and calculation described in Prüss-Üstün et al[48] to facilitate the calculation of YLD for a large data set. We used the contemporaneous disability weight of 0.361 for MMR from pediatric lead exposure and did not disaggregate by sex. The equation as used (for a single disabling event) is as follows (adapted from Prüss-Üstün et al):

$$\mathrm{YLD} = \mathrm{DW}\left\{\frac{\mathrm{KCe}^{\mathrm{ra}}}{\mathrm{r}+\beta^2}\left[\mathrm{e}^{\ (\mathrm{r}+\beta)(\mathrm{L}+\mathrm{a})}\left[-(\mathrm{r}+\beta)(\mathrm{L}+\mathrm{a})-1\right] - \mathrm{e}^{\ (\mathrm{r}+\beta)\mathrm{a}}\left[-(\mathrm{r}+\beta)\mathrm{a}-1\right]\right] + \frac{1-\mathrm{K}}{\mathrm{r}}\left(1-\mathrm{e}^{\mathrm{rL}}\right)\right\}$$

where:

a = age of death (years)
r = discount rate (3%)
ß = age weighting constant (eg, ß = 0.04).
K = age-weighting modulation constant (eg, K = 1).
C = adjustment constant for age weights (eg, C = 0.1658).
L = duration of disability (years)
DW = disability weight

The second method for calculating DALYs (prevalence) was developed and used by the IHME for the 2010 GBD report and is now the standard approach of both the IHME and the WHO. This method does not account for future disease resulting from exposures and as such does not employ discounting or age weights for the calculation of YLD. Rather, YLD are calculated in a straightforward multiplicative fashion shown in the following equation (adapted from the WHO[49]):

$$\mathrm{YLD}_i = \mathrm{DW}_i \times p_i$$

where:

YLD = years lost due to disability
DW = disability
p = prevalence

We used the prevalence values from the aforementioned WHO spreadsheet for MMR and applied them to the entire exposed population. Because the MMR DW for pediatric lead exposure (0.361) is no longer used by the WHO, we utilized the closest analog, mild intellectual disability with a DW of 0.1270. We further applied the DWs for borderline (0.0034), moderate (0.2930), severe (0.3830), and profound (0.4440) intellectual disability to the prevalence of each sequela.[49] To calculate prevalence of the borderline, moderate, severe, and profound intellectual disability, we used relative distribution percentages provided by the WHO for LMICs and extrapolated from cases of MMR.[49] We summed all resulting YLDs to determine attributable DALYs. Neither the incidence nor the prevalence method resulted in a value for YLL. DALYs for intellectual disability were therefore the result of YLD equations only. We did not adjust for comorbidity.

DALYs From IQ Decrement Not Resulting in MMR. Following Chatham-Stephens et al, we applied an additional DW for IQ decrement not resulting in MMR. All children at all sites covered by this analysis were expected to suffer lifelong impairment from IQ decrement resulting from lead exposure. Using the WHO calculator referred to above, we estimated that the vast majority (863 per 1000 population) were expected to have a decrement of ≥3.5 IQ points. An additional 135 children

Table 1. Estimated Number of Informal ULAB Processing Sites and Exposed Populations by Utilized Subregion

Region	Number of Sites (Method 1)	Population Exposed (Method 1)	Number of Sites (Method 2)	Population Exposed (Method 2)
Central Africa	126	72,532	157	90,205
East Africa	161	92,486	2702	1,553,638
North Africa	498	286,503	857	492,667
Southern Africa	529	303,962	1578	907,299
West Africa	351	201,849	2780	1,598,733
East Asia and Indochina	2959	1,701,232	6736	3,873,174
Former Soviet Union	564	324,511	776	446,387
Caribbean	90	51,536	213	122,293
Central and North America	569	327,108	819	471,096
South America	1352	777,252	1495	859,806
Middle East	438	251,836	741	426,153
South Asia	1369	787,023	8538	4,909,218
Southeast Asia (excluding Indochina)	1594	916,633	1849	1,063,431
Total	10,599	6,094,463	29,241	16,814,100

ULAB, used lead-acid battery.

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Table 2. DALYs by Health Outcome at Informal ULAB Processing Sites (Incidence Method)

Sequela	Low	High
Mild mental retardation	49,009	160,070
Total cardiovascular diseases	35,017	191,208
Hypertensive	4125	21,683
Ischemic	12,381	65,285
Cerebrovascular	14,224	80,575
Other cardiovascular diseases	4287	23,665
IQ decrement (other)	463,448	1,261,198
Total DALYs	547,474	1,612,476

DALY, disability adjusted life year; ULAB, used lead-acid battery.

per 1000 were calculated to have a decrement of 1.95 to 3.25 IQ points. IQ decrement resulting from pediatric lead exposure has long been acknowledged to result in decreased lifetime earnings and other adverse lifelong effects.[50 52] Despite this linkage, no DW currently existed that captured this impairment. As an alternative, we applied the DW for anemia (0.024).

In the incidence approach, we calculated IQ decrement for all non-MMR fetuses and children (ages 0-4) for all years of life. We used the applicable WHO DALY calculation (incidence) with full age weighting. We did not disaggregate by sex. In the prevalence approach, we calculated IQ decrement for fetuses, children, and adolescents (ages 0-14). We assumed 100% prevalence for all groups for whom we had not already calculated intellectual disability. We excluded possible IQ decrement for adults because we assumed an age of 15 years for all ULAB sites.

Table 3. DALYs by Health Outcome at Informal ULAB Processing Sites (Prevalence Method)

Sequela		Low	High
Total intellectual disability		65,883	182,149
Intellectual disability	borderline	602	1652
Intellectual disability	mild	27,569	75,700
Intellectual disability	moderate	23,482	64,485
Intellectual disability	severe	10,233	28,097
Intellectual disability	profound	4448	12,215
Total cardiovascular diseases		35,017	191,208
Hypertensive		4125	21,683
Ischemic		12,381	65,285
Cerebrovascular		14,224	80,575
Other cardiovascular diseases		4287	23,665
IQ decrement (other)		25,896	88,114
Total DALYs		127,248	461,470

DALY, disability adjusted life year; ULAB, used lead-acid battery.

RESULTS

We estimate that there are 10,599 to 29,241 informal ULAB processing sites where human health is at risk in the 90 countries that we reviewed. We further estimate that 6 to 16.8 million people are exposed to lead at these sites, resulting in exposures to 557,000 to 1.8 million children (ages 0-4). Summary results of the number of sites and exposed populations are presented in Table 1.

We calculated a geometric mean BLL for children at ULAB sites of 31.15 μg/dL (range: 19.5-55.8 μg/dL) and a geometric mean BLL for adults of 21.2 μg/dL (range: 9.7-49.5 μg/dL). We estimated that these combined exposures resulted in 127,248 to 1,612,476 DALYs in 2013. Table 2 presents the results of the incidence method. Table 3 presents the results of the prevalence method. Table 4 presents a summary of all major findings organized by WHO region.

DISCUSSION

The main findings of the present study are that there are between 10,599 and 29,241 informal ULAB processing sites where human health is at risk in the 90 countries we reviewed; that battery recycling operations at these sites result in exposures to between 6,094,463 and 16,814,100 million people; and that these exposures to lead resulted in 127,248 to 1,612,476 DALYs in 2013.

Because global data on lead exposure at ULAB sites have not previously been available, none of the findings from our analysis were included in previous analyses of the GBD. In their most recent estimate of the GBD, the IHME modeled lead exposures resulting from aerial deposition of leaded gasoline.[2] They used these estimates to determine that 4,199,925 and 15,594,412 DALYs resulted globally in 2015 from exposure to lead from gasoline.

In the 90 countries covered by the present analysis, IHME estimates that approximately 3,092,760 and 7,017,424 DALYs resulted in 2015 from exposure to lead from gasoline. When that estimated burden of disease is added to the findings from this analysis, we calculated that the total burden of disease resulting from lead exposure in these 90 countries amounts to 3.2 million to 8.6 million DALYs in 2013. This finding indicates that lead is one of the largest environmental contributors to the GBD.

Table 4. Informal ULAB Sites, Exposed Populations, and Attributable DALYs by WHO Region

Region	Number of Sites (Method 1)	Population Exposed (Method 1)	Number of Sites (Method 2)	Population Exposed (Method 2)	DALYs (Low)	DALYs (High)
Africa	1348	775,058	7152	4,112,339	17,570	627,017
Europe	564	324,511	776	446,387	9128	46,978
Southeast Asia	3615	2,078,530	9625	5,534,601	42,825	395,526
Americas	2010	1,155,897	2527	1,453,195	21,629	118,157
Western Pacific	2222	1,277,492	6489	3,731,000	25,497	234,906
Eastern Mediterranean	840	482,975	2672	1,536,578	10,600	189,891
Total	10,599	6,094,463	29,241	16,814,100	127,248	1,612,476

DALY, disability adjusted life year; ULAB, used lead-acid battery; WHO, World Health Organization.

Throughout the present analysis, we tried to err in the direction of conservatism and to base our analysis on low-bound estimates. We noted a series of limitations in our approach, some of which resulted from this tendency to down-weight the numbers. First, our estimated geometric mean BLLs for both children and adults (31.15 and 21.2 µg/dL, respectively) are well below the levels that we have encountered in our own experience at ULAB sites in LMICs and are also below the levels reported in previously published studies. Thus, our calculations with regard to BLLs and DALYs likely resulted in a significant underestimate of the full magnitude of the disease burden due to lead exposure at ULAB sites globally. Comparable studies in China, for example, have found population-wide BLLs >100 µg/dL[53,54] and studies from other countries regularly report BLLs >40 µg/dL.[55 57] Our low estimates may reflect our reliance on environmental exposure estimates and on the IEUBK and ALM algorithms that were designed for a US context. We have tried to correct for this limitation by adjusting upward for the higher levels of ingestion that have been reported in LMICs. An alternative approach might be to apply a calculation for BLLs based on a meta-analysis of relevant literature to exposed populations.

Another potential shortcoming in the present analysis is that the algorithms we developed to estimate the number of contaminated sites in each region relied on a relatively small number of parameters and on a series of assumptions. Possibly incorrect assumptions included our use of a 100% recycling rate for ULABs and our exclusion of transboundary movement (beyond the individual subregions). Other potential shortcomings include the fact that national statistics in LMICs are potentially inaccurate. However, absent an alternative, we relied wholly on these statistics in many cases and elsewhere we extrapolate from them to fill gaps. We used Dowling et al. as a guide for developing these estimates and hedge toward the more conservative option.[11] Future, more detailed assessment at ULAB sites globally will result in refinement and very likely in increases in the estimates that we presented here.

With regard to our DALY calculations, we followed WHO methods closely on the calculation of DALYs resulting from intellectual disability and CVD with the exception of failing to account for comorbidity. We acknowledge that this may have resulted in a slight overestimate for the attributable fraction of lead exposure to these sequelae. More significantly we provided an additional DW for lifetime IQ decrement resulting from pediatric exposure. The result of this calculation disproportionately contributes to the overall disease burden presented. Pediatric lead exposure and associated IQ decrement result in a lifetime of adverse consequences, including decreased earnings, poorer educational and health outcomes, and higher rates of societal violence.[50,52,58,59] The existing list of WHO DWs simply does not account for these outcomes and as such falls woefully short of documenting the disease burden suffered by leaded children. We disaggregated all DALYs by sequelae and provided detailed accounting for our calculations, thus enabling the reader to elect to include this additional weighting or not.

We calculated DALYs for intellectual disability using 2 separate approaches. The first, prevalence, resulted in a range of 127,248 to 461,470 DALYs, whereas the second, incidence, resulted in the higher values of 547,474 to 1,612,476 DALYs. The prevalence approach is the most common method currently used in DALY calculations. Accordingly, these values are immediately

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comparable with those developed elsewhere. The incidence approach was used by the WHO until 2013 and is potentially more sensitive to certain characteristics of ULAB sites. Among them, this approach tends to more heavily weight nonfatal disease in children compared with adults.[49] We assumed an age of 15 years for all sites in our analysis, indicating that the future disease burden will be more significant than current population-wide prevalence suggests. Given that the incidence approach accounts for future morbidity, this is reflected in the results, whereas it is absent from the prevalence approach. Others have observed that for similar reasons the incidence approach may be preferable in cost-effectiveness analysis of interventions.[60] The different DALY values presented here do not reflect an actual difference in the amount of disease resulting from informal ULAB sites. They are simply 2 ways of presenting the same data.

The approach that we took here to assessing lead exposure at ULAB sites could be modified and applied to assess the disease burden resulting from exposures to other poorly quantified toxic chemicals at contaminated sites globally. Among the more prominent contaminants in the TSIP database are hexavalent chromium, obsolete pesticides, mercury, and radionuclides. It is likely that if analyses were undertaken of the disease burden resulting from exposures to these pollutants, the GBD attributable to pollution would increase still further.

The contamination resulting from informal ULAB sites presents a unique set of challenges, including its persistence, prevalence, and severity. Lead is highly immobile in the environment and is unlikely to migrate on its own from surface to subsurface soils where exposures would be reduced.[61] Accordingly, without intervention, ULAB sites will pose a risk to populations well into the future. By way of example, the phase down of leaded gasoline in the United States began in 1975, resulting in significant reductions over the next 2 decades and a total ban in 1995.[62] Soil lead concentrations in US urban areas, however, have remained elevated, contributing to 256,704 DALYs in 2015.[2,62,63] The case is similar globally. Most HICs phased down leaded gasoline along the same timeline as the United States.[64] At present, leaded gasoline is only sold in 2 countries: Algeria, Iraq, and Yemen.[65] Despite the absence of an ongoing source of leaded gasoline emissions, 4,199,925 and 15,594,412 DALYs result globally from legacy depostion.[66] In the case of informal ULAB sites, contaminated soils similarly could not be expected to naturally attenuate. To mitigate these long-term exposures, sites will need to be identified and characterized and site-specific remediation plans developed and executed.

Informal ULAB sites are diffuse and common. Haryanto, for instance, documented 71 distinct smelting locations in the Jakarta metro area.[20] These sites are disparately located in nearly all major areas of the city.[20] Similar spatial diffusion has been identified in those countries where TSIP has been conducted. ULAB operations are of low capital intensity and require little skill. They are therefore somewhat amenable to relocation; requiring a different policy response than their formal counterparts. In some cases, informal operators have been aggregated into industrial clusters that are spatially segregated from residential areas.[56,57] Elsewhere, sites have been shuttered by authorities only to relocate. Much has been written on adapting regulatory frameworks to the informal sector.[67,68] There is a need to evaluate informal ULAB sites in the context of this literature and develop appropriate regulatory responses.

ULAB sites present a uniquely severe risk to area residents. Indeed, the predicted pediatric BLLs in the present study are extremely rare in HICs, and then only seen in adults in very high-exposure occupational settings.[69] Surface soil lead concentrations resulting from ULAB sites are at least an order of magnitude higher than the highest set of roadside levels in HICs.[55,62] At its peak, leaded gasoline in the United States resulted in a mean BLL of 14.6 μg/dL for the population as a whole and 13.6 μg/dL in children.[70,71] In the present study, we predicted geometric mean BLLs of 31.15 μg/dL in children and 21.2 μg/dL in adults living near ULAB sites. We further stated that these predictions are much lower than actual the BLLs found by researchers at sites in the field[53 57] and that our predictions are very likely underestimates.

CONCLUSIONS

We estimate that in 2013 informal ULAB processing sites put the health of 6,094,463 to 16,814,100 million people at risk and contributed to 127,248 to 1,612,476 DALYs in the 90 countries we evaluated. These estimates indicate that this industry is currently causing widespread lead poisoning in

LMICs. Informal ULAB sites pose a unique threat to area residents in their persistence, prevalence, and severity. There is an urgent need to identify, characterize, and mitigate exposures at existing sites. There also is a need to identify appropriate policy responses and practical and effective long-term management approaches to minimize the creation of new sites.

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3.2. Paper Seven

Estrada-Sánchez, D., **Ericson, B.,** Juárez-Pérez, C. A., Aguilar-Madrid, G., Hernández, L., Gualtero, S., & Caravanos, J. (2017). Pérdida de coeficiente intelectual en hijos de alfareros mexicanos. *Revista Médica del Instituto Mexicano del Seguro Social*, *55*(3), 292-299.

Lead oxide has been applied as a glaze in traditional ceramics for centuries. Although the practice persists in very few countries, it remains common in Mexico with an estimated 10,000 ceramic workshops using lead oxide glazes. These workshops typically either adjoin the home or are contained within it, thereby presenting a significant exposure risk. This study attempts to characterize the risk associated with the ongoing use of lead oxides in pottery glazes via analysis of environmental sampling from 19 different Mexican ceramics workshops and calculation of the resulting attributable disease burden in terms of IQ point decrement. This paper applied two well-established models of lead attributable IQ decrement to pediatric exposures. It is the first such study to conduct this type of analysis in this context. The paper was originally written in Spanish (Appendix A) with co-authors from a major government health agency (IMSS) and published in a government journal. The overall the intention of this approach was to have maximal public health impact in Mexico. The paper sits equally well in both Chapters 2 and 3 as it describes a unique though ubiquitous source of lead poisoning in Mexico and the resulting disease burden as measured in IQ decrement.

Intelligence quotient loss in the children of Mexican ceramicists*

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*Article translated rom the original Spanish to English by Bret Ericson or submission with this thesis Formatting o the journal generally maintained

Intelligence quotient loss in the children of Mexican ceramicists

Background: In Mex co, art sans frequent y use ead ox de or greta n order to produce utens s, wh ch are dest ned to preparat on and storage of food and dr nks. Add t ona y, the r sk of ead po son ng of art sans and the r fam es s greater than n genera popu at on, and w th n these fam es, ch dren are the most suscept b e to ead po son ng. The a m of th s study was to est mate IQ oss n Mex can ch dren from potter fam es exposed to ead.

Methods: Lead concentrat ons n so were determ ned n 19 potter's homes that funct oned as pottery workshops n seven Mex can states between 2009 and 2012. Th s nformat on was used to est mate b ood ead eve s through the ntegrated exposure uptake b ok net c (IEUBK) mode . The oss of IQ po nts was then est mated accord ng to the Lan phear and Schwartz mode s.

Results: The mean ead concentrat on found n the workshops' so was 1098.4 ppm. B ood ead eve s est mated n ch dren under 8 years o d were 26.4 µg/dL and the oss of IQ po nts compr sed from 7.13 to 8.84 po nts depend ng on the mode .

Conclusions: It s poss b e that 11 ch dren from fam es of art sans n Mex co may be os ng between 7.13 to 8.84 IQ po nts, due to ead expo sure n the r houses workshops. Th s oss n IQ po nts cou d have mpor tant hea th, econom c and soc a mpacts.

The use of lead-based ceramic glazes dates back to ancient Greece and has continued relatively unchanged until only recently.[1] Although their use has decreased worldwide, lead glazes are currently employed to produce handicrafts in several countries in Latin America, North Africa and the Middle East. In the Americas, the Spanish introduced lead-based glazes as an alternative to burnished ceramics in the sixteenth century.[1,2,3] Nearly five centuries later, lead glaze continues to be used in at least 20 Mexican states.[2] The US Food and Drug Administration (FDA) utilizes a maximum permitted level of lead in ceramics based on use, with a range of 0.5 to 3 µg /mL.[4] In Mexico, *NOM-004-SSA1-2013* states that the use of lead-based ceramic glazes should be avoided.[5] However, both in the formal and informal sector, ceramicists continue the regular use of lead-based glazes for pottery used in food preparation and storage.[6] Such ceramics are used throughout the country, predominantly among the poor; however, for cultural and traditional reasons, they are also used by higher income Mexicans. The majority of pottery in Mexico is produced in low temperature ovens (between 850 and 1000°C). They therefore do not reach a sufficient temperature to form insoluble lead silicates resistant to chemical attack by food and liquids acids. Under these conditions, lead is more bioavailable and can be easily released into food and drinks in contact with the glaze.[7] Food acids such from tomato, coffee, chili and lemon juice accelerate the lead leaching process.[8]

In 1991, Ávila *et al.* determined that 58% of the risk attributable to blood lead levels of Mexican women was due to the use of pottery glazed with lead to prepare, serve, and store food and beverages.9 In a recent publication by Caravanos et al. (2014) average blood lead levels were estimated to be 8.85 µg/dL in urban areas and 22.24 µg/dL in rural areas. The study identified pottery as the primary source of exposure, in addition to other sources such as mining and smelting.[10]

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Keywords	Palabras clave
Ceram cs	Cerám ca
Lead	P omo
Occupat ona exposure	Expos c ón ocupac ona
Child	N ño
Env ronmenta hea th	Sa ud amb enta

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Resumen

Introducción: en Méx co, os a fareros cont núan usando frecuentemente e óx do de p omo o greta para produc r utens os, os cua es se dest nan a a pre parac ón y a macenam ento de a mentos y beb das. Ad c ona mente, e r esgo de ntox cac ón por p omo de os a fareros y sus fam as es mayor que en a pob a c ón genera , y en ta es fam as, os n ños son os más suscept b es a a ntox cac ón por p omo. E objet vo de estud o fue est mar a pérd da de puntos de coef c ente nte ectua (CI) en h jos de a fareros mex canos expuestos a p omo.

Métodos: durante e per odo de 2009 a 2012 se deter m naron as concentrac ones de p omo en sue o de 19 casas ta eres de a fareros en s ete estados mex ca nos. Esta nformac ón se ut zó para est mar e n ve de p omo en sangre, por med o de mode o b oc nét co ntegrado de absorc ón por expos c ón (IEUBK, por sus s g as en ng és). Poster ormente, se ca cu aron os puntos perd dos de CI según os mode os de Schwartz y Lanphear.

Resultados: a concentrac ón promed o de p omo en sue o fue de 1098.4 ppm. Se est mó un n ve de p omo en sangre de 26.4 µg/dL para menores de 8 años. La pérd da de puntos de CI est mada fue 7.13 y 8.84, según e mode o ut zado.

Conclusión: es pos b e que a menos 11 n ños de fam as a fareras mex canas estén perd endo entre 7.13 y 8.84 puntos de CI, deb do a a expos c ón a p omo en sus casas ta eres, o que supone mportan tes mpactos económ cos, soc a es y de sa ud.

Other studies have documented the impact of glazed pottery production on the health of adults and children in Mexico, including reports of blood lead concentrations greater than 20 µg/dL in families of ceramicists in Tzintzuntzan, Michoacán, and more than 30 µg/dL in a pottery community of Veracruz state.[7,11,12] Children are more susceptible to lead poisoning than adults.[13] Moreover, the risk of lead poisoning to ceramicists and their families is greater for a series of reasons. These include elevated environmental exposure, foods cooked in glazed pottery with lead, malnutrition, the shared use of spaces for work and living, bad practices in the handling of lead oxide, and the fact that that lead becomes an endogenous source of exposure.[12,14,15]

The relationship between the level of lead in blood and alterations in the IQ has been widely studied.[16] A meta-analysis by Schwartz[17] found that lead affects neuro-transmissions and is associated with a decrease of 2.6 IQ points with levels of between 10 and 20 µg/dL of lead in blood. Several authors have studied the association between elevated blood lead levels and environmental exposure to lead in soil.[15,18,19,20,21] From the reviewed literature no studies were found concerning the impact of lead on the IQ of ceramicists' children, which represents the population most vulnerable to the use of lead in in this industry. Therefore, the present study estimated the loss of IQ points from lead levels in soil for children living in ceramicists' homes.

Methods

A cross-sectional study was carried out between November 2009 and March 2012 in which Pure Earth (formerly Blacksmith Institute) conducted lead soil measurements in 25 ceramicists' homes in seven states of Mexico (Colima, State of Mexico, Jalisco, Michoacán, Oaxaca, Puebla and Tlaxcala), as part of site evaluations intended to facilitate remediation.[22,23] Heads of households that had previously participated in programs provided by the National Fund for the Promotion of Handicrafts (FONART) were invited to participate. The selection of the sample was by convenience, as only those households who were interested in knowing the levels of lead in the soil of their home-workshops were included. Of the 25 home-workshops that participated via FONART, 19 were selected for the study. In all 19 cases the application of glazes was carried out inside the home, thus reducing the influence of atmospheric variables that could significantly decrease the levels of lead in the soil. All the families that participated in the study belonged to social stratum D (very low income) and subsisted mainly from the production of pottery. In all cases it was observed that the spaces for production and lead glaze application were used as common areas within the home.

Blood lead levels were estimated with the US EPA Integrated Environmental Uptake Biokinetic win v1.1 build 11 (IEUBK) model based on soil measurements taken in home-workshops. Estimated blood concentrations were then used to estimate the number of IQ points lost in individual children.[21,24]

Environmental Lead Levels

The investigation analyzed the soil in and around the home of each family of artisans. An average of 10 samples were taken in each workshop. A portable X-ray fluorescence instrument (XRF) was used to measure concentration of lead in the surface soil (Equipment: INNOV-X Alpha 6500, Woburn, Massachusetts). This instrument instantly measures 21 metals in solid and semi-solid soil matrices. Study participants were informed about the risks associated with environmental and occupational exposure to lead, as well as on strategies to minimize their exposure.

Soil lead measurements were shared with the homeowners during the assessments, owing to the 15-30 second response time of the XRF. In cases where surface soil lead concentrations exceeded the United States Environmental Protection Agency (EPA) recommended level of 400 ppm workers were advised to cease using lead oxide.

Six home-workshops were excluded from the study because their operations carried out exclusively outdoors, thus the soil was subject to the dispersion or entrainment of lead by climatic influences. Consequently, the analysis of burden of disease was based on 149 readings of 19 Pottery workshops (Table

Estimated Lead Levels

The IEUBK win v1.1 version build 11, was used for the estimation of blood lead levels. This toxicokinetic model can be obtained for free and is widely used to predict the blood lead levels of children from between six months to seven years of age, based on the environmental levels of lead.[25,26] The IEUBK model was developed by the US EPA to estimate the effect of emissions from lead smelters and to evaluate remediation activities at Superfund sites.[27] It is used in various applications and scientific research and, therefore, has been reviewed by experts.

The IEUBK model uses standard biokinetic absorption factors as default values, though allows the user to edit parameters related to exposure. Thus the user can edit inputs from lead in air, soil, water and food sources, as well as the amount of water ingested, the volume of inhaled air and gastrointestinal absorption by age group.[28] It also allows the user to enter the level of lead in the mother's blood. In this study the model default values of zero (0) for air, water, food, blood and maternal exposure were used. Additionally, it was assumed that lead deposits on the exterior floor and dust inside they were the same. The default value of soil ingestion (100 mg/day) was increased to 400 mg/day to account for dustier conditions resulting from earthen floors, and unpaved streets and sidewalks, which increase exposure.

Table I Home-workshops of participants (*n* = 19), States and population (from November 2009 to March 2012)

Estado	Town	Population	Workshops in the town (and state)	Studied workshops
Colima	Colima	24 939	5 (5)	1
Jalisco	El Grullo	21 825	10 (597)	2
México	San José del Arenal	875	(1189)	2
México	Tecomatepec	1549	(1189)	1
Michoacán	Capula	5086	600 (3435)	4
Michoacán	Santa Fé de la Laguna	4046	1000 (3435)	1
Oaxaca	Oaxaca	258 008	(2500)	1
Oaxaca	Santa María Atzompa	16 855	(2500)	2
Puebla	Zautla	18 567	1837 (1931)	2
Tlaxcala	Tenexyecac	2863	(480)	3
	Total	339 982	10 586 (México)	19

To determine the population of affected children in the 19 workshops, it was assumed that all households were composed of four people, which is the average number of members per family in Mexico. Additionally, with 2010 data from the National Institute of Statistics and Geography (INEGI), it was estimated that 15.2 % of the population is between 0 and 7 years old, which means that around 11 children between 0 and 7 years old lived in the 19 workshops.[30] As the study was based on single visits focused on environmental concentrations, family heads were not questioned about the number of children residing in the home. The field experience of the researchers indicates that the use of INEGI estimates is likely conservative.

Estimation of IQ points lost

Once the blood lead level was estimated, it was utilized to calculate its impact on the IQ.
Two meta-analyzes were used: the Schwartz study of 1994[17] and that of Lanphear *et al.* from 2005.[31] Schwartz's work was based on eight studies that monitored lead levels of 2,702 children from different socioeconomic strata with blood lead levels between 10 and 20 µg/dL. The meta-analysis found that an increase of 10 to 20 µg/dL of lead in blood resulted in a loss of 2.57 IQ points. With this investigation, a linear model to predict the impact in that range, so the loss of IQ can be calculated by multiplying the blood lead level by 0.257.[16]

To estimate the effect of lead on IQ in children with less than 10 µg/dL of lead in the blood, the Lanphear *et al.* metaanalysis was used. This study analyzed cohorts that included 1,333 children.[31] The investigation took into account multiple variables that could influence the relationship between lead exposure and IQ, such as the child’s gender, birth order, birth weight, maternal education, maternal age, marital status, prenatal exposure to alcohol, prenatal exposure to tobacco, as well as an index that reflects the quality and quantity of emotional and cognitive stimulation at home, measured by the program of the Home Observation Measurement of the Environment (HOME). The study found that a logarithmic-linear model predicted the impact on the IQ. The equation of this model is loss of IQ = beta * ln (concurrent level of blood lead/cutoff point), with a beta of -2.70 and a cutoff point of 1 µg/dL. The level of blood lead estimated for each home-workshop was used in both equations to predict the decrease in IQ.[31]

Results

Soil lead levels in home-workshops of artisans

149 readings were taken in 19 workshops. The average lead concentration (geometric mean) in the soil inside home-workshops was 1,098.4 ppm, with a confidence interval (CI) of 95% of 898-1,343.5 ppm (Table II). More than 50% of the workshops had maximum readings that exceeded 5,000 ppm (Table II). In cases where the concentrations of lead identified were above the levels recommended by the EPA (400 ppm), owners were advised to immediately stop the use of lead oxide.

In addition to lead, the XRF instrument also determines concentrations of 38 other elements. Of these, none of the readings showed high levels of arsenic, cadmium, mercury or any other relevant toxic materials.

Estimation of blood lead levels

Lead levels in blood were calculated with the IEUBK model, based on default zero (0) inputs for air, water, food and maternal blood and using the total geometric mean soil lead levels of 1,098.4 ppm (Table II), with the aforementioned 95% CI of 898-1343.5 ppm. This yielded an average blood lead level of 26.4 µg/dL in children under 8 years of age, with a CI at 95% of 23.2-29.8 µg/dL. This average result is five times greater than the one recommended by the US Centers of Disease Control (CDC;5 µg / dL).[32]

Estimation of IQ points lost

Using the Lanphear *et al.* and Swhwartz models, it was estimated that the 11 children under eight years of age, who hypothetically would be living in the 19 houses-workshops, would probably have a loss between 7.13 (Schwartz) and 8.84 (Lanphear *et al.*) IQ points as a result of this type of exposure to lead (table III).

Discussion

During the last decades, several studies have shown high levels of blood lead in communities of ceramicists in Jalisco, Michoacán and Veracruz.[11,12,14] These studies have indicated that the increase in blood lead levels is associated with factors such as the location of workshops inside homes, cooking in glazed earthen ceramics, being a child, being a woman and having a dirt floor.

This study estimates an average loss of 7.13 to 8.84 IQ points for children living in the 19 pottery home-workshops. In Mexico there are 10,586 registered workshops in the FONART ceramicists' census. Based on the average Mexican family size it can be estimated that there are at least 42,344 people who live in ceramicists' home-workshops in Mexico. If the conditions found in this study were present in all of these, it could be estimated that approximately 6,436 children under the age of eight (15.2% of the ceramicist family population) would be at risk of decreasing their IQ as a result of the exposure to high levels of lead in their home-workshops.[30] Depending on the choice of the model, this loss could vary between 7.13 and 8.84 points on average for each child.

This IQ decrease could also have a negative social impact. In 1998, Gottfredson described IQ loss as the condition most deeply implicated in adverse social outcomes (poverty, poor well-being, delinquency and school failure).[33] Health impacts caused by pediatric lead exposure also have a negative impact on the economy. The impact can be quantified in terms of public health costs, the need to additional educational resources, and low productivity due to lost IQ. Ceramicists working with ceramic glazes present the highest levels of lead in blood, compared to other populations that are occupationally and environmentally exposed.[34,35]

Table III So ead eve s, b ood ead eve s and IQ decrement of ceram c sts' ch dren that ve n houses w th ead g azes

So ead eve s (ppm)	B ood ead eve µg/dL (< 8 years of age)	IQ decrement est mate (Schwartz)	IQ decrement est mate (Lanphear)
1098.4 (geometr c mean)	26.4	−7.13	−8.84
898 (95% CI ow)	23.2	−6.26	−8.49
1343.5 (95% h gh)	29.8	−8.05	−9.17

A study conducted in 2002 by Landrigan *et al.* examined the financial costs of four diseases associated with exposing children to different environmental factors: lead poisoning, cancer, asthma and developmental disorders.[36] The study found that environmental exposure related to these diseases generates approximately 2.8% of the total annual health care costs in the United States, which amounted to a cost of USD 54.9 billion. This study is important to understand the risk presented by the use of lead oxide for ceramicists and their families.

Study Limitations

This study is based on the evaluation of lead concentration on the floors of 19 workshops and utilized convenience sampling. It is not a representative sample of the total number of workshops that exist in the participating states, or all of Mexico. Therefore the results cannot be generalized to the pottery workshops throughout the country. The researchers noticed several elements that can influence lead exposure risk in home-workshops, which can vary significantly between workshops. Therefore, the estimates made in this study could vary. These elements include and in the number of inhabitants, the percentage of children in each home-workshop, the volume of production, the distribution of the work/residential space, the flooring material of the home-workshop, work roles, glaze management practices, hygiene habits, the number of children not in the family circle with access to the contaminated areas, and the use of glazed pottery to cook. It is likely that the results of the study underestimate the magnitude and scope of the problem as the calculations only include one route of exposure to lead (soil) and did not take into account other important exposure pathways, such as contaminated food intake and the endogenous bone source of the mother to the child *in utero*. Additionally, this study utilized a single assessment of the concentrations of lead in soil, which is not representative of exposure throughout life in each home-workshop. Further it did not analyze other variables that could confound the relationship of lead exposure with the loss of IQ points, such as the ages of children, schooling, nutritional status, among others.

Conclusion

The present study focused on the intellectual quotient points lost in ceramicists' children and can serve as a baseline to expand the knowledge of the impact of the lead on IQ due to pottery production. Lead may also have other adverse effects on multiple systems of the human body, such as the neurological, hematological, gastrointestinal, cardiovascular, and renal systems.[36] Thus ceramicists' children are a population at significant risk. An epidemiological surveillance program should be developed to monitor their health, along with that of the ceramicists themselves. It is necessary to improve knowledge of the social, economic and health costs of lead-based glazes in Mexico. It is also necessary to apply existing laws to more effectively eliminate this type of market product and to make consumers aware of the importance of buying and using lead free pottery. Therefore, it is necessary that a permanent and effective governmental policy encourage replacement of lead glazes with non-toxic salts.

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Declaration of conflict of interest: the authors have comp eted and sent the trans ated form (Span sh)of the dec arat on of potent a conf cts of nterest to the Internat ona Comm ttee of Med ca Journa Ed tors, and a conf ct was not reported.

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3.3. Paper Eight

Ericson, B., Dowling, R., Dey, S., Caravanos, J., Mishra, N., Fisher, S., ... & Taylor, M. P. (2018). A meta-analysis of blood lead levels in India and the attributable burden of disease. *Environment International*, *121*, 461-470.

Of the studies in Chapter 3, Paper Eight most directly contributes to the aims of the Chapter. Recent published studies (2010–2017) on BLLs in India were reviewed to calculate pooled mean BLLs for different subgroups based on age, gender and the nature of the exposure (i.e. occupational or non occupational). Pooled means were then used to estimate the attributable disease burden in DALYs and IQ decrement and compared with disease burden estimates calculated elsewhere. In contrast with other papers in the chapter that model BLLs based on environmental data, this study utilized actual BLLs collected from patients. It thus removes some uncertainty from the overall approach.

The DALY calculations include robust sensitivity testing of model parameters. In all cases, the DALY estimates are significantly larger than have been calculated elsewhere, underlying the central theme of this thesis that the lead-attributable disease burden is underestimated.

The study was prepared with the involvement of the Public Health Foundation of India (PHFI), a leading Indian public health organization. Their involvement increased the likelihood that the study might have relevance for policy makers within India. Like the preceding chapter, this decision was strategic in that its aims were to encourage a policy response to mitigate exposures.

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A meta-analysis of blood lead levels in India and the attributable burden of disease

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ABSTRACT

Multiple studies in India have found elevated blood lead levels (BLLs) in target populations. However the data have not yet been evaluated to understand population-wide exposure levels. We used arithmetic mean blood lead data published from 2010 to 2018 on Indian populations to calculate the average BLLs for multiple subgroups. We then calculated the attributable disease burden in IQ decrement and Disability Adjusted Life Years (DALYs). Our Pubmed search yielded 1066 articles. Of these, 31 studies representing the BLLs of 5472 people in 9 states met our study criteria. Evaluating these, we found a mean BLL of 6.86 μg/dL (95% CI: 4.38–9.35) in children and 7.52 μg/dL (95% CI: 5.28–9.76) in non-occupationally exposed adults. We calculated that these exposures resulted in 4.9 million DALYs (95% CI: 3.9–5.6) in the states we evaluated. Population-wide BLLs in India remain elevated despite regulatory action to eliminate leaded petrol, the most significant historical source. The estimated attributable disease burden is larger than previously calculated, particularly with regard to associated intellectual disability outcomes in children. Larger population-wide BLL studies are required to inform future calculations. Policy responses need to be developed to mitigate the worst exposures.

1. Introduction

Lead is a naturally occurring metal with a range of industrial applications and well documented adverse health effects when human exposure occurs (ATSDR, 2007). Its widespread use has resulted in significant contamination of natural and human environments (Needleman, 2004; Prüss Üstün et al., 2010). Chronic lead exposure, even at very low levels, is associated with cognitive impairment, cardiovascular effects, anemia and low birth weight, among other adverse health outcomes (Budtz Jørgensen et al., 2013; Lanphear, 2015; National Toxicology Program, 2012; United Nations Environment Programme, 2010). Lead exposure has been associated with decreased economic output, lower life expectancy and increased societal violence (Demayo et al., 1982; Landrigan and Goldman, 2011; Mielke and Zahran, 2012; Prüss Üstün et al., 2010; Taylor et al., 2016).

The 2016 Global Burden of Disease, Injuries and Risk Factors Study by the Institute for Health Metrics and Evaluation (IHME) estimated that lead exposure resulted in 13.9 million Disability Adjusted Life Years (DALYs) and 540,000 deaths in 2016 globally. The DALY metric is used in quantifying the burden of disease and is intended to capture morbidity and mortality attributable to a given disease or risk factor in a population (World Health Organization, 2016). In India alone, IHME found 4.6 million lead attributable DALYs and nearly 165,000 deaths (IHME, 2017a).

The most significant historic source of global lead exposure was the use of tetraethyl lead in petrol in the 20th century (Bollhöfer and Rosman, 2001, 2000; Flegal et al., 1984; McConnell et al., 2015; Schwikowski et al., 2004; Véron et al., 1999). In cities where it was used, leaded petrol accounted for 80 to 90% of airborne lead pollution (Lovei, 1999). High income countries began banning the use of lead in most fuels, as well as in paints, in the 1970s, resulting in significant declines in societal blood lead levels (BLLs) (Needleman, 2004). Leaded

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petrol was phased out in India from 1996 to 2000 and was similarly followed by BLL declines (Singh and Singh, 2006). Nichani et al. (2006), for instance, documented a 60% decrease in BLLs among re sidents of Mumbai from 1997 to 2002, following the full adoption of unleaded petrol. Similarly, Singh and Singh (2006) found a mean BLL decrease of 33% following the leaded petrol phase out in the urban centers of Mumbai, Chennai, Bangalore, Amritsar and Lucknow.

Despite these substantial improvements in exposure reduction, studies conducted more than a decade after the Indian phase out of leaded petrol continue to report elevated BLLs, often associated with proximity to lead smelting sites (Bellinger et al., 2005; Ghose et al., 2005; Sharma et al., 2005). Other sources of lead exposure to the Indian public have included ayurvedic medicine, cosmetics (kohl/surma) and contaminated foodstuffs (Goswami, 2013; Raviraja et al., 2010; Singh et al., 2010; Singhal, 2016). In some cases these exposures have found severely elevated levels in both occupational and non occupational settings (Ghanwat et al., 2016; Goswami, 2013). Studies of environ mental media have reported elevated lead concentrations in tube wells, rivers, and soil, among other media (Borah et al., 2010; Chatham Stephens et al., 2013; Lokhande et al., 2012). With regard to lead based paint, India currently maintains one of the stricter global limits of 90 ppm soluble lead (UNEP, 2017). However a 2015 study that assessed store bought cans of enamel paint found that 46% of those tested contained > 10,000 ppm lead (Toxics Link, 2015). Additionally, some studies have posited lead based paint as a possibly significant source of exposure (Ahamed et al., 2009; Khan et al., 2010).

Few studies have attempted to calculate population wide mean BLLs in low and middle income countries (LMICs), with most focusing on discrete cohorts of exposed individuals (Olympio et al., 2017). Caravanos et al. (2014) conducted a meta analysis of BLLs in Mexico, finding a mean concentration of 5.36 μg/dL in urban areas after the phase out of leaded petrol. A 2009 study of Chinese BLLs reviewed published papers and found a mean BLL of 7.93 μg/dL for male children and 7.69 μg/dL for female children living in urban areas (He et al., 2009).

In this assessment we reviewed existing studies on BLLs to infer broader conclusions about the population of a subset of India. We first conducted a literature review and meta analysis of Indian BLLs pub lished between 2010 and 2018. We then used the results to quantify the disease burden in terms of IQ decrement and attributable DALYs. The objective of this study was to quantify the potential public health im pacts of lead exposure in India and to stimulate policies, education, and, where appropriate, remediation of contaminated sites.

2. Methods and approach

2.1. Literature review and data selection

We conducted a PubMed search in April 2018 using the terms blood (subheading, all fields, MeSH terms) lead (all fields, MeSH terms), and India (all fields, MeSH terms, abstract text) between 1 January 2010 and 1 January 2018 (National Library of Medicine (US), 1946). We then assessed each article by a the following 6 criteria: 1) the study pub lished BLL data from human populations residing in India; 2) the study included at least 30 participants; 3) BLL data were derived from venous, capillary, or umbilical cord samples (bone, organ or tissue samples were excluded); 4) the utilized data were collected after 2005; 5) the study was published in English; 6) the study contained a statistical mean and standard deviation (SD) or standard error (SE) for the original data set. Articles that did not meet one or more of the above criteria were ex cluded from the meta analysis.

2.2. Subgroup rational

The BLL data for each study were analyzed by certain demographic categories following the literature review. Where possible samples were disaggregated by the following four subgroups: gender, age, urbanicity, and occupation.

Age categories were defined using United Nations Children's Fund's parameters outlined in the Convention on the Rights of the Child. An individual was considered a “child” if he or she was at or below the age of 17 at the time of the original study, and an “adult” if he or she was identified as at or above 18 (United Nations General Assembly, 1989). Gender was stratified into four different categories: female, male, both and unspecified. Urbanicity was determined by a review of studies for ‘urban’ or ‘rural’ keywords. If this was not indicated in the article, the study location was used to make this determination. The Census of India classification of 400 people per square kilometer was used as the threshold for an urban area (India, 2011). Finally samples were coded as occupational if the relevant occupation substantively involved lead and therefore a higher risk of elevated BLLs. Samples comprised of battery recyclers for instance were coded as occupational, while studies of teachers were coded as non occupational.

2.3. Identification and use of sample means

Where possible, the mean and SD/SE were derived for our specific subgroups. In cases where the subgroups used by study were incon gruous with our own, the mean and SD/SE were taken for a larger subset, such as the study population.

If the same population was assessed multiple times, and treatment was not provided in between assessments, the mean for all analyses was used. In cases where the mean for all analyses could not be taken, the most conservative value (i.e. lowest) value was used. If treatment was provided to the patients with the intent of lowering BLLs, pre treatment values were used.

Three studies assessed the BLLs of the same large cohort of un treated children at different points (Palaniappan et al., 2011; Roy et al., 2013, 2009). In this case, one study had a slightly larger sample size than the other two and all presented similar overall results with regard to BLLs. The study with the largest population was thus included and the other two were excluded.

In one case, BLLs were assessed at the same point using multiple methods having different results (Reddy et al., 2014). In this case we selected the most conservative (i.e. lowest) value.

Some studies segregated the sample exclusively based on the results of the BLL test (e.g. high and low subgroups). In these cases we took the pooled BLL for the study. In one study (Ravibabu et al., 2015), the pooled mean was not available. We therefore used both subgroups as discrete samples. Other studies disaggregated the sample by health outcome. Tiwari et al. (2012), for instance analyzed BLLs for three groups of anemic women (mild, moderate, severe) and one control group. A pooled mean was not available for the study as a whole, so we used the means for each subgroup and presented them as discrete samples.

Two studies, Goswami et al. (2013) and Chaudhary et al. (2017), found exceptionally high BLLs in children. While the exposures that result in these BLLs were not occupational they do represent an acute scenario that is not representative of the general population, thus jus tifying their exclusion. Goswami et al. (2013) looked at children that apply surma (kohl) as a cosmetic, which has long been identified as an acute source of lead exposure, and a control group of children that do not apply surma (Ali et al., 1978; Gogte et al., 1991). In this case a study mean was not available, so we utilized the control group and excluded the exposed group. Chaudhary et al. (2017) assessed the BLLs of 260 children (age 6 months to 12 years) attending the pediatrics outpatient department at a hospital in Lucknow, Uttar Pradesh. The study reported a mean BLL of 55.7 μg/dL (SD: 227.38). We were unable to identify a comparably high value of a general population in the literature. Other studies in Lucknow have found much lower levels. Ahamed et al. (2011) assessed the BLLs of 68 children (age 3 12 years) in Lucknow, finding BLLs of 4.23 9.86 μg/dL. An earlier study by the same authors evaluated 200 children (age 3 12 years) finding a mean BLL of 9.3 μg/dL (range: 1.0 27.9 μg/dL) (Ahamed et al., 2009). A separate study of study of 500 pregnant women in 1996 before the phase out of leaded petrol found an average BLL of 14.3 μg/dL (Awasthi et al., 1996). In addition to being significantly higher than other studies of Lucknow populations, the BLLs identified by Chaudhary et al. (2017) are in consistent with the other studies we assessed from elsewhere in the country. The next highest BLL for a sample of children was 11.8 μg/dL, while half of the studies of occupationally exposed adults found lower mean BLLs. Thus this study was assumed to represent an acute exposure and was excluded from the pooled studies of children's BLLs.

The following data were extracted for each identified sample: study authors; publication date; sex of sample; state of India; age category (adult/child); urbanicity; arithmetic mean BLL; subgroup rationale (e.g. surma users, mechanics); number of participants; SD or SE; age if children.

2.4. Meta analysis of BLLs

We pooled the arithmetic mean BLLs from each study using a Random Effects (RE) meta analysis model. Meta analyses are typically conducted using either an RE or Fixed Effects (FE) model. Fixed effects models are concerned with within study variability only and do not account for variability between studies. An FE model is appropriate when the effect size for all studies is assumed to have one true value and any variance that occurs is due to sampling error (Borenstein et al., 2010). Random effects models, by contrast, assume that studies re present a random sampling of different populations within a larger 'super' population (DerSimonian and Kacker, 2007; Hedges, 1992). Thus in an RE model variance observed in the evaluated studies is as sumed to be due in part to true variance between the sampled groups (Borenstein et al., 2010). Effectively this method weights each sample's effect size by its inverse variance in pooling effect sizes and confidence intervals.

In the present effort we evaluated studies drawn from discrete po pulations across India; each with different lifestyles and exposure sce narios. We therefore assumed that variance reflected in the samples was due, at least in part, to true differences in mean BLL concentrations. Accordingly we took the mean BLL and standard error from each sample and pooled them using a RE model. We used the metan tool in Stata 15.1 for the analysis (StataCorp. LP, 2017). The metan tool uti lizes the DerSimonian and Laird method (1986) for RE and a method taken from Mantel and Haenszel (1959) to assess heterogeneity (Sterne, 2009). In addition to a pooled effect size and confidence intervals, the metan tool generates q, I^2, and tau^2 statistics. We present these in the relevant figures below. In all but one of our evaluated subgroups the p value of the q statistic is below 0.000, confirming heterogeneity and further indicating that an RE model was appropriate.

2.5. Calculating IQ decrement

We calculated IQ decrement resulting from pediatric lead exposure using the log linear model described in Budtz Jørgensen et al. (2013). The authors used internationally pooled data from seven cohorts of children to calculate a benchmark dose of 0.1 1.0 μg/dL for the loss of a single IQ point. Budtz Jørgensen et al. (2013) re evaluated the data and approach of Lanphear et al.'s (2005) study of low level environmental lead exposure (defined as < 7.5 μg/dL) and its impact on the devel oping brain. The cohorts used in both studies are comprised of school age children (age 5 10 years) with chronically elevated BLLs. Verbal and performance tests were conducted to determine the extent of in tellectual impairment and those results are compared with BLL mea surements from the following four periods: early childhood (age 6 24 months); average lifetime; maximum lifetime; and concurrent (at the time of the IQ test). Lanphear et al. (2005) found that concurrent BLL measurements had the strongest relationship with IQ decrement. Budtz Jørgensen et al. (2013) accordingly applied concurrent geo metric mean BLLs to both log linear and two piece linear models, and found the log linear to be the best fit. Here, we used the log linear model presented and input the arithmetic mean BLL for the subgroup children in India to determine IQ points lost for children age 10 years and under.

2.6. Calculating DALYs

We used the meta analysis results for non occupationally exposed adults and children to calculate DALYs. DALYs are a metric intended to represent the disease burden in a given population and the relative contribution of disparate health outcomes to it. They are the sum of two other metrics, Years of Life Lost (YLL) and Years Lived with Disability (YLD). YLL represents early attributable mortality while YLD represents the severity and duration of a given health outcome (World Health Organization, 2016). DALYs are employed most notably by the World Health Organization (WHO) and IHME in their respective periodic global burden of disease reports (Forouzanfar et al., 2016; World Health Organization, 2016).

Lead exposure results in a number of quantifiable adverse health outcomes, however methods for integrating those outcomes into DALY calculations, as with other chemical exposures, are somewhat limited (Grandjean and Bellanger, 2017). We therefore calculated DALYs for two sequelae only: cardiovascular disease (CVD) and intellectual dis ability. We followed the approach outlined by Ericson et al. (2016, 2018a) and described below. DALY calculations for both sequelae uti lized the total population of the all states where the individual studies were conducted.

To calculate DALYs resulting from cardiovascular disease in 2013 we used a prevalence rate calculator developed by the WHO for BLLs (Fewtrell et al., 2003). The calculator requires the geometric mean BLL and standard deviation for a given population to determine the lead attributable fraction of CVD in that population. Values are returned for four classifications of CVD: ischemic, cerebrovascular, hypertensive, and other heart diseases. In the absence of a population wide geometric mean, we input the pooled arithmetic mean and standard deviation for non occupational exposures for adults to determine the attributable fraction for each case. We then applied these attributable fractions to the most recent (2013) WHO CVD DALY estimates for India to de termine the number of DALYs and deaths attributable to lead exposure (WHO, 2014). We further proportionately reduced the national number of DALYs for India to the population of those states from which studies were drawn.

To calculate DALYs resulting from lead induced intellectual dis ability in 2012 we used the WHO calculator described above to de termine a prevalence of Mild Mental Retardation (MMR) in a given population of 0 4 year olds with a given geometric mean BLL. We again input the arithmetic mean and standard deviation for children in our study (all non occupational). As above, we used the total population of the states where studies were conducted rather the national population to develop our estimates.

The WHO calculator was developed in 2003 and uses the now an tiquated classification of MMR and its associated disability weight. These values have since been revised to more accurately capture a gradient of intellectual disability. While MMR was previously quanti fied with a disability weight of 0.361, the revised disability weights for intellectual disability are as follows: borderline (0.0034), mild (0.1270), moderate (0.30), severe (0.3830) and profound (0.4440) in tellectual disability (Colin et al., 2004; WHO, 2013). To determine the proportional composition of these subgroups, we assumed MMR was analogous to mild intellectual disability and calculated the prevalence of the remaining subgroups by extrapolating from that value. To do so, we used relative proportions provided by the WHO (2013). We then determined the number of DALYs attributable to each sequalae using the following equation:

Table 1
Mean values and relevant statistics of Indian blood lead levels by subgroup.

Subgroup	Number of samples	Mean BLL (μg/dL)	LCI	UCI	CHI^2	p	I^2	TAU^2
Occupational adults								
Female	0	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Male	14	48.65	38.74	58.56	12,593.47	0.000	99.9%	336.62
Non occupational								
Adult female	9	4.32	3.41	5.23	413.71	0.000	98.1%	1.62
Adult male	8	7.23	4.52	9.94	375.19	0.000	98.1%	14.66
All adults	28	7.52	5.28	9.76	29,841.92	0.000	99.9%	23.21
Unspecified	3	9.62	6.25	12.96	17.13	0.000	88.3%	7.54
Children	17	6.86	4.38	9.35	7135.25	0.000	99.8%	26.89
Urban adults	19	6.69	4.89	8.48	5642.36	0.000	99.7%	15.50
Rural adults	1	10.90	9.34	12.46	0.00	N/A	N/A	N/A
Urban children	16	6.92	4.35	9.50	7051.37	0.010	99.8%	27.19
Rural children	1	5.90	4.82	6.98	0.00	N/A	N/A	N/A

$$YLD = DW \times p$$

where:

p = prevalence
DW = disability weight

Adapted from WHO (2013).

2.7. Sensitivity analysis

To assess the sensitivity of the meta analysis, we employed two distinct approaches. In the first we simply ran the analysis using a FE model (Bown and Sutton, 2010). In the second we utilized "leave one out" cross validation. In this approach, we ran the RE model succes sively removing a single study from the sample in each run (Arlot and Celisse, 2010). We took the squared error for each run (actual minus predicted value) and calculated the mean squared error for all runs.

To assess the sensitivity of the DALY model parameters, we calcu lated DALYs using IHME disability weights for intellectual disability. In our study we utilized the following WHO weights: borderline (0.0034), mild (0.1270), moderate (0.30), severe (0.3830) and profound (0.4440) (WHO, 2013). In our sensitivity analysis we used the following IHME weights: borderline (0.011), mild (0.043), moderate (N/A), severe (0.16) and profound (0.2) (Global Burden of Disease Collaborative Network, 2017).

3. Results

3.1. Results

Our PubMed search yielded 1066 studies. Of these 979 did not contain BLL data on human populations within India and were ex cluded. A further 56 studies did not meet one or more of the remaining criteria and were excluded. The remaining 31 studies contained 67 samples for use in our study (marked with an asterisk in the references). The 67 samples represented a population of 5472 people in 9 different Indian states (Andhra Pradesh, Karnataka, Maharashtra, Punjab, Rajasthan, Tamil Nadu, Telangana, Uttar Pradesh, and West Bengal). These states had an approximate population of 717 million people re presenting 56% of India's national population in 2011 (India, 2011).

3.2. Blood lead levels

Seventeen of the 67 samples were comprised of children re presenting 2009 individuals in 6 different states. These states had an estimated population of 560,190,596 at the time of the most recent census (India, 2011). All childhood exposures were identified as non occupational. The samples utilized in this study were normally dis tributed as assessed with the Shapiro Wilk test ($p > 0.6$). The pooled arithmetic mean for all children was 6.86 μg/dL (95% CI: 4.38 9.35).

All children in all studies were < 14 years of age. Most studies included age ranges covering multiple years and provided limited detail on the composition of the those groups. Thus, a mean age for the study population could not be determined. Of those children included in the samples, at least 24% (n = 486) were ≤2 years of age, at least 66% (n = 1318) were ≤7 years of age, at least 80% (n = 1618) were ≤10 years of age, and at least 90% (n = 1814) were ≤12 years of age. Samples comprised exclusively of children ≤2 years of age had a mean BLL of 8.49 μg/dL (95% CI: 6.18 10.8), those ≤7 years of age had a mean BLL of 6.9 μg/dL (95% CI: 2.70 10.67), those ≤10 years of age had mean BLL of 6.52 μg/dL (95% CI: 3.24 9.8), and those ≤12 years of age had a mean BLL of 6.73 μg/dL (95% CI: 4.17 9.28). It therefore appears that a younger age was associated with a higher BLL, though given the lack of specificity in the data this observation cannot be properly evaluated.

A forest plot of all samples of children is presented as Fig. 2. Mean BLLs and related statistics (confidence interval, p value, q, I^2, and tau^2) for subgroups are presented below in Table 1. A forest plot of all samples and studies used in the analysis are presented in Fig. 1.

Fifty of the samples were comprised of adults representing 3463 individuals in nine states. These states had an estimated population of 717,577,668 at the time of the most recent census (India, 2011). Of these 22 samples were made up of 1499 individuals with occupational exposures, while the balance (n = 28) were comprised of 1964 in dividuals with non occupational exposures. The pooled arithmetic mean for non occupationally exposed adults in the study was 7.52 μg/dL (95% CI: 5.28 9.76). The samples utilized were normally distributed as assessed with the Shapiro Wilk test ($p > 0.05$). A forest plot of all samples of adults used in the analysis is presented as Fig. 3.

With regard to subgroups, occupationally exposed adults had sig nificantly higher BLLs than any other group. Fourteen samples (all men) represented individuals from this group and had a mean BLL of 48.65 μg/dL (95% CI: 38.74 58.56). By contrast mean BLLs for all other subgroups ranged from 4.32 to 10.9 μg/dL.

3.3. IQ decrement

Within the study population, children had an arithmetic mean BLL of 6.86 μg/dL (95% CI: 4.38 9.35). Using the log linear model de scribed in Budtz Jørgensen et al. (2013) we determined that this BLL would result in an average decrement of 4 IQ points (95% CI: 2.5 4.7) for children under age 10.

3.4. Disability Adjusted Life Years (DALYs)

We calculated that cardiovascular disease attributable to lead ex posure resulted in 2.7 million DALYs (95% CI: 2.3 3) in 2012 in the 9 states we reviewed. We further found that intellectual disability in

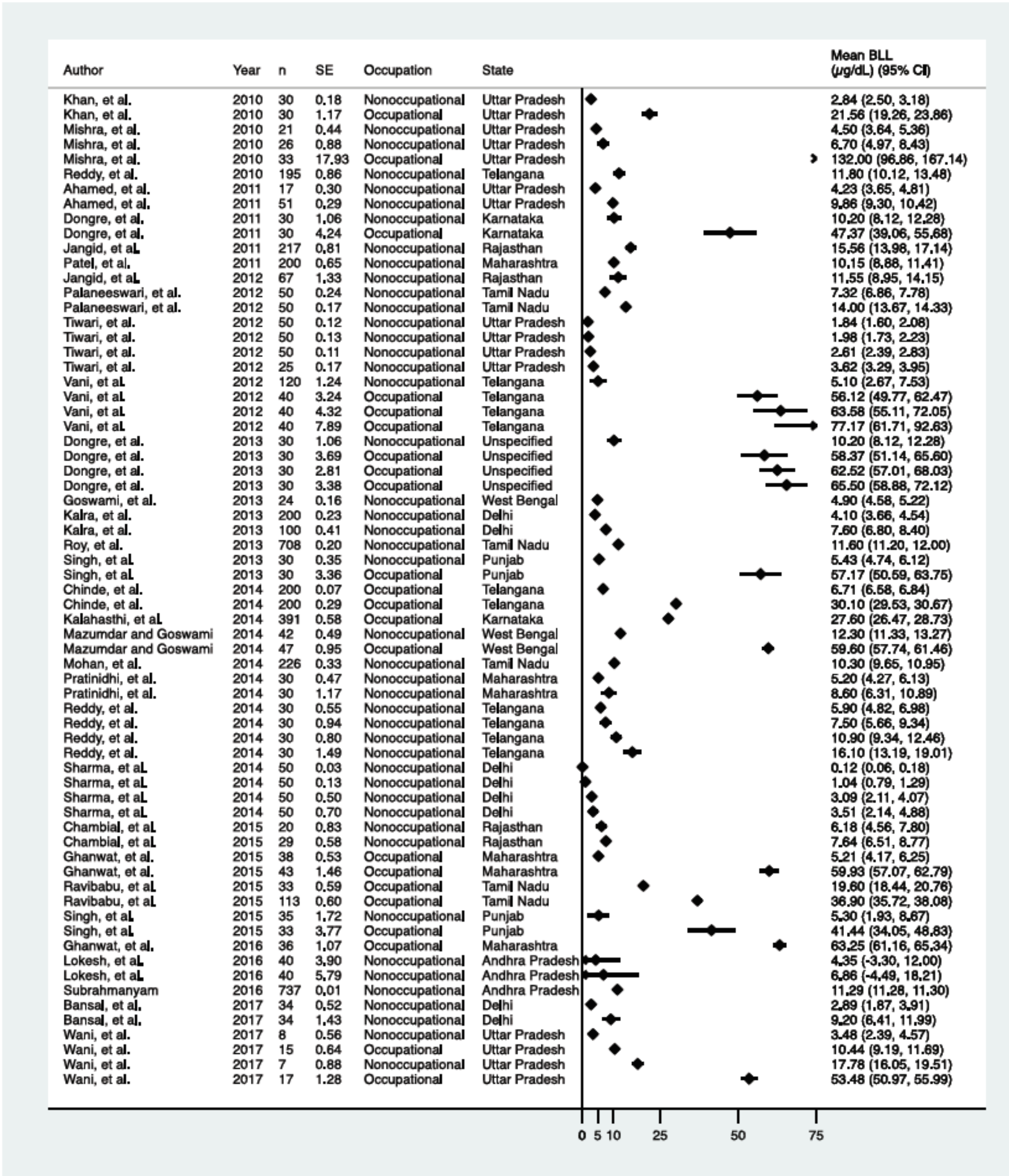

Fig. 1. Forest plot of all samples and studies used in the analysis.

children (age 0 4 years) attributable to lead exposure resulted in 2.2 million DALYs (95% CI: 1.6 2.6) in the same year in the 6 states we reviewed. Altogether we calculate 4.9 million DALYs (95% CI: 3.9 5.6) attributable to lead exposure in the geography we reviewed in 2012. Tables 2 4 summarize attributable DALYs by sequelae and calculated BLLs.

3.5. Results of the sensitivity analysis

Conducting the analysis with an FE model yielded a mean BLL of 0.87 µg/dL (95% CI: 0.81 0.93) in children and a mean BLL of 11.18 µg/dL (95% CI: 11.17 11.2) in non occupationally exposed adults. In the case of children, a single study with both a low mean BLL (0.12 µg/dL) and a low standard error (0.126) disproportionately in fluenced the effect size with a weighting of 88.92%. Similarly in non occupationally exposed adults, a single study with a mean BLL of 11.29 µg/dL and low standard error (0.0058) received a weighting of 98.58%. The RE method used here accounts for heterogeneity between samples, while the FE model does not. Thus to some extent our ap proach mitigates this issue.

Using the leave one out approach, we found that children's BLLs ranged from 6.55 7.26 µg/dL and had a mean squared error of 0.042. With regard to adults, we found that BLLs ranged from 7.13 7.77 µg/dL and had a mean squared error of 0.031.

Replacing the WHO disability weights with those used by IHME, we calculated 695,068 DALYs (95% CI: 522,191 822,872) for children in our geographic subgroup in 2012, indicating that disability weighting has a significant influence over the results.

4. Discussion

Our analysis of studies of BLLs from a geographic subgroup within India found that 4.9 million DALYs (95% CI: 3.9 5.6) were attributable to lead exposure in 2012. This is somewhat greater than the disease

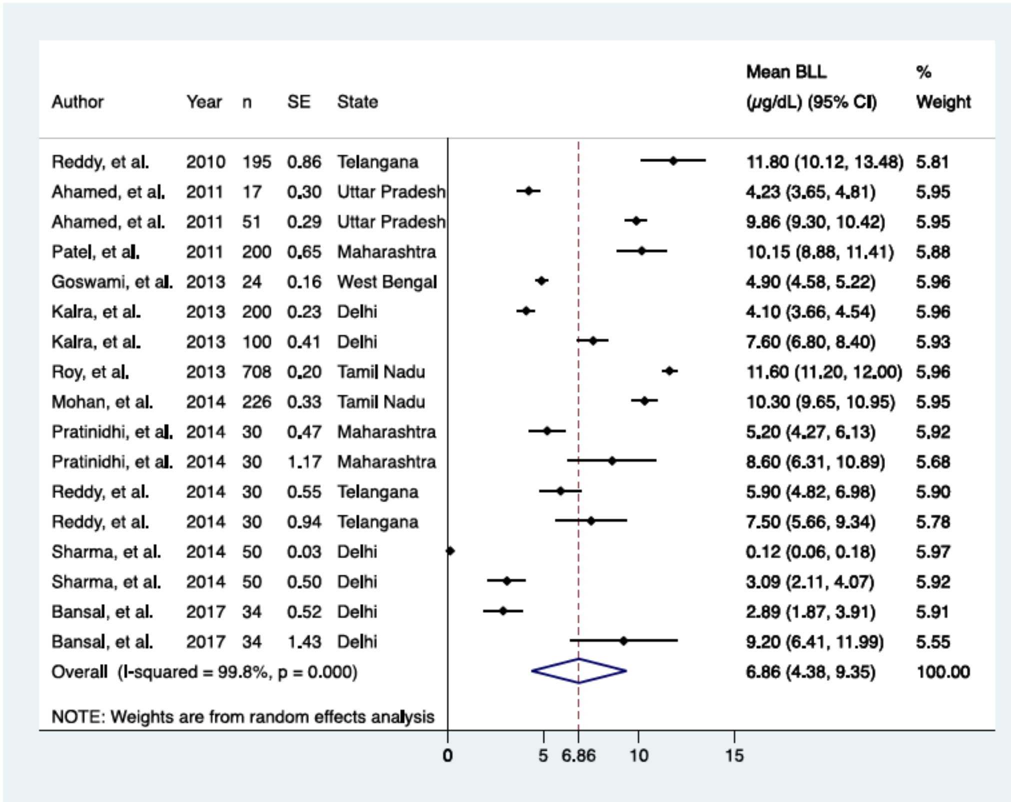

Fig. 2. Forest plot of all samples of children used in the analysis.

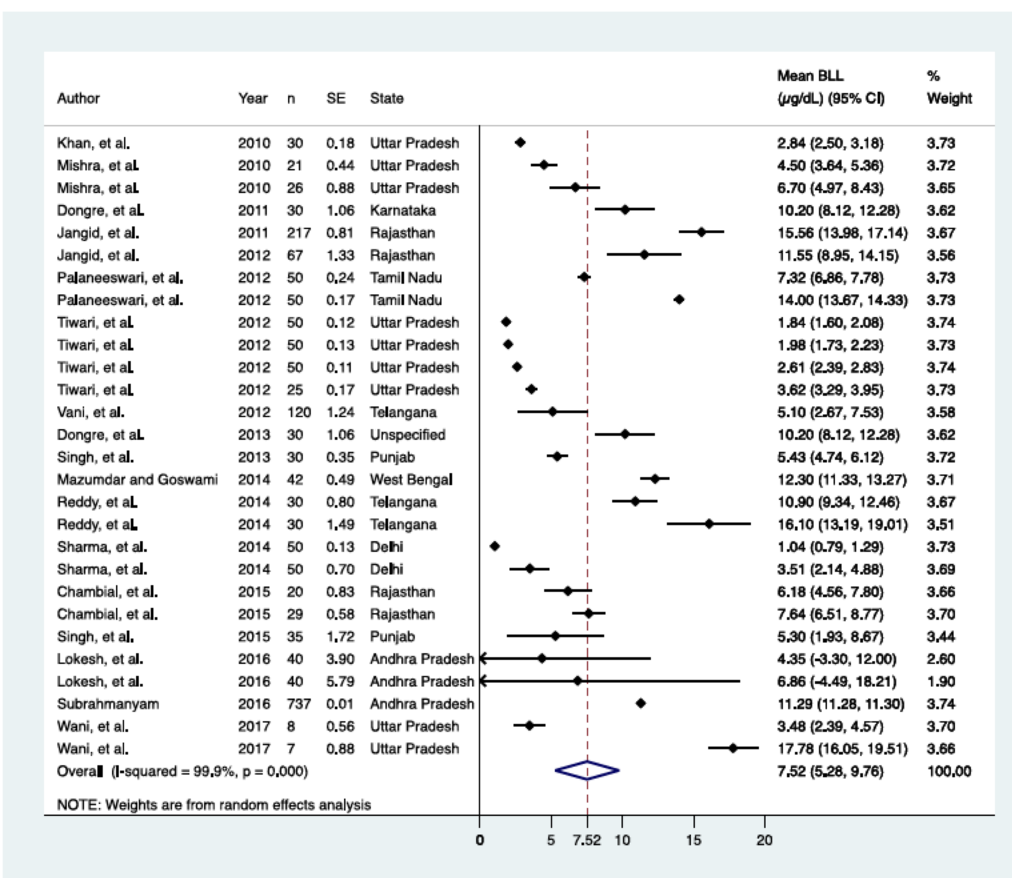

Fig. 3. Forest plot of all samples of non-occupationally exposed adults used in the analysis.

Table 2
DALYs from cardiovascular disease attributable to lead exposure in 9 Indian states in 2012.

	Hypertension	Ischemic	Cerebrovascular	Other CVD	Total DALYs (CVD)
Pooled mean (7.52 µg/dL)	93,082	1,146,922	1,199,438	286,028	2,725,470
LCI (5.28 µg/dL)	79,573	983,168	1,033,822	244,882	2,341,444
UCI (9.75 µg/dL)	103,241	1,269,450	1,322,146	316,886	3,011,723

Table 3
DALYs from intellectual disability attributable to pediatric lead exposure in 6 Indian states in 2012.

	Borderline	Mild	Moderate	Severe	Profound	Total DALYS (intellectual disability)
Pooled mean (6.86 µg/dL)	19,935	913,231	777,933	338,963	147,356	2,197,418
LCI (4.38 µg/dL)	14,973	685,936	584,312	254,598	110,680	1,650,500
UCI (9.35 µg/dL)	23,599	1,081,053	920,892	401,253	174,435	2,601,233

Table 4
Calculated DALYs (all sequalae) attributable to lead exposures in the reviewed states in India in 2012.

	Total DALYs (CVD)	Total DALYS (intellectual disability)	Total DALYs (all sequalae)
Pooled mean	2,725,470	2,197,418	4,922,889
LCI	2,341,444	1,650,500	3,991,944
UCI	3,011,723	2,601,233	5,612,955

burden calculated by IHME of 4.6 million DALYs (95% CI: 2.9 6.5) for the country as a whole in 2016 (IHME, 2017b). The discrepancy is most pronounced in children (age 0 4 years) who accounted for 33,264 DALYs (95% CI: 12,428 33,264) in IHME's analysis. In the 6 states included in our review of children's BLLs we found this group incurred more than 2.2 million DALYs (95% CI: 1.6 2.6). This discrepancy is in part due to differences in how IHME weights sequelae related to intellectual disability and how we do so here. In our sensitivity analysis we calculated DALYs using the IHME weights, finding 695,068 DALYs (95% CI: 522,191 822,872). Thus while disability weighting significantly influences the results, it alone insufficiently accounts for the discrepancy.

Looking at ages 15 years and above only, IHME calculates 4.3 million DALYs (95% CI: 2.6 6.3) attributable to lead exposure compared with the 2.7 million DALYs (95% CI: 2.3 3) found by this study. The 9 states covered by this study represent approximately 56% of the national population. Scaling IHME's values to a population of comparable size results in 2.4 million DALYs (95% CI: 1.4 3.5). Thus the results are similar for adults.

It is possible that the 2016 IHME GBD report underestimates the pediatric disease burden from lead exposure in India. In this study, we calculated a mean BLL of 6.86 µg/dL (95% CI: 4.38 9.35) for all children in our geographic subgroup. We further calculated that, using our method, a national mean BLL of < 1 µg/dL would be required to arrive at the 33,264 DALYs (95% CI: 12,428 61,466) estimated by IHME for 2016. While this value has been achieved in the United States, it would seem inconsistent with the recent blood lead exposure data examined here (Center for Health Statistics, 2017).

In the 2004 WHO global burden of disease estimate, average BLLs of 7.4 µg/dL for children and 9.8 µg/dL for adults were used to calculate the attributable burden (Prüss Üstün et al., 2010). The researchers found a prevalence of 5.5 cases of MMR per 1000 population attributable to lead exposure based on these estimates. This is somewhat less than we found in the present effort (~13 cases of MMR per 1000) however significantly more than the ~0.27 per 1000 prevalence that would be required to reach the 33,264 DALYs calculated by IHME (using our method). Few other studies have calculated the disease burden of chemicals either globally or on a national level for India (Chatham Stephens et al., 2013; Prüss Ustün et al., 2011). Therefore, there is a limited basis for assessing the relative accuracy of the estimates provided here and by WHO or IHME. Given the robust literature on the adverse effects of lead on neurological development and the likely elevated BLLs in children in India, the topic could clearly benefit from further study.

A 2015 study by Iyer et al. reports on the blood lead analysis of 222,668 individuals from multiple states in India. The study provides limited statistical information and was therefore not included in the present analysis. Specifically, neither SD nor SE was included with the sample mean. However given the exceptionally large sample size, the study provides useful context for our results. For children under 2 years of age (n = 119), the authors find a mean BLL of 4.91 µg/dL and for children 2 10 years of age (n = 688) the authors find a mean BLL of 4.2 µg/dL. In adults (n = 219,303), the authors find mean BLLs of different age groups ranging from 4.24 4.95 µg/dL. In all cases, the values reported by Iyer et al. (2015) are somewhat lower than our results. Of particular interest are the geographic differences in BLLs identified by Iyer et al. (2015). For instance, the authors define a 'high' BLL as 15 µg/dL and provide the percentage of blood samples from each state that exceed this threshold. In two states, Maharashtra and Bihar, this percentage exceeds 10, while in Gujarat it is 2.5. This indicates that significant differences in BLLs exist between states. Further review of the vast dataset utilized by Iyer et al. (2015) to better understand these differences could greatly benefit other researchers.

4.1. Contemporary sources of lead contamination

A number of possible environmental sources of lead exposure are present in India, including ayurvedic medicine, contaminated food and cosmetics (Goswami, 2013; Raviraja et al., 2010; Singh et al., 2010). Lead contamination as a food safety issue was recently brought to the fore when supplies of a popular noodle product, Maggi, were found to have elevated concentrations of lead (Singhal, 2016). Additionally, lead based enamel paint evidently remains widely available (Toxics Link, 2015).

Eighty five percent of global lead production is used in the manufacture of storage, lighting and ignition (SLI), or lead acid, batteries (International Lead Association, 2016). In India 700 750,000 metric tons of lead are recycled each year with perhaps 50% being recycled in the informal sector (Ericson et al., 2016; Pugazhenthy, 2017). Widespread informal used lead acid battery (ULAB) recycling is perhaps due in part to the confluence of a large informal economy and increased car ownership. Approximately 21% of India's GDP is generated in the informal sector while the number of automobiles in India nearly tripled from 55 million to 159.5 million from 2001 to 2012 (Schneider et al., 2010; Shukla et al., 2015).

Informal ULAB recycling is a prominent source of lead exposure in LMICs where primitive operations of unregulated backyard smelters cause widespread contamination (Daniell et al., 2015; Ericson et al., 2018a, b, 2016; Haefliger et al., 2009; Prajapati, 2016). One well documented example of an extreme case of poisoning resulting from informal ULAB recycling was the deaths of 18 children in Senegal linked to informal battery smelting (Haefliger et al., 2009). Informal smelters are by definition illegal, and are accordingly particularly vul nerable to regulatory intervention. In response, these low cost opera tions are often operated intermittently at different locations in different neighborhoods, resulting in the creation of new hotspots of con tamination (Shen et al., 2016). Due to lead's low mobility in the en vironment, contamination hotspots are likely to pose a risk indefinitely without remediation (Kabala and Singh, 2001). Lead deposited from smelters and other sources, such as leaded petrol, in surface soils is readily re suspended as dust, presenting an ongoing exposure risk (Laidlaw et al., 2012).

In addition to lead contamination, smelting operations can generate elevated concentrations of other toxic trace metals including arsenic, cadmium and mercury (Roussel et al., 2010; Stafilov et al., 2010). There are limited published studies detailing effective approaches to miti gating the health risks posed at informal ULAB sites. One recent ex ample from Vietnam describes the construction of an industrial zone for informal workers located 1 km from residential areas. The relocation, coupled with community education and soil lead abatement work, re sulted in median BLL declines of 67% in children (< 6 years of age) within one year of the intervention (Ericson et al., 2018b).

In India, the product life cycle of lead acid batteries is regulated under India's *Batteries Management and Handling Rules of 2001*, amended in 2010 (Ministry of Environment and Forests (India), 2001, 2010). The Rules create a deposit refund system in which retailers collect used lead acid batteries from consumers when they purchase new batteries and offer a rebate for the new purchase. ULABs in turn are required to be sold only to registered recyclers, who transport, handle and recycle the used batteries responsibly. Despite this existing legislation, informal (unregulated) ULAB recycling is widespread. One study found that among major battery manufacturers, few were able to collect > 40% of the used batteries they had produced (Prajapati, 2016).

4.2. Study limitations

The study is most significantly limited by its reliance on a relatively small number of studies (n = 31). As a result, values are inferred for a population of 717 million from the BLL results of only 3973 non oc cupationally exposed people. Future studies might endeavor to collect more comprehensive biological data from a more representative cross section of the country. It should be noted that the US National Health and Nutrition Examination Survey (NHANES) is slightly larger in size, with data collected from approximately 5000 individuals annually, and done with the specific intention of inferring results for the population as a whole (Center for Health Statistics, 2017).

A second limitation is our reliance on an older method for calcu lating the attributable disease burden. The prevalence rate calculator we used was developed by WHO in 2003 and has been validated, though to the best of the authors' knowledge has not been modified in the intervening years. Significantly, the WHO calculator estimates the prevalence of MMR using an older linear IQ decrement model devel oped by Schwartz (Schwartz, 1994). Replacing those values with the more recent Budtz Jørgensen et al. (2013) log linear model would likely result in a higher estimate of the prevalence of intellectual dis ability, and thus a higher disease burden (Ericson et al., 2018a). We do not endeavor to do so here.

5. Conclusion

Population wide BLLs in India remain elevated despite regulatory action to mitigate the most significant sources. The attributable disease burden may be larger than previously calculated, particularly with regard to intellectual disability in children. Larger population wide BLL studies are required to inform future calculations. Major traditional sources of lead exposure based on leaded petrol emissions and de positions are insufficient to account for the results here. Therefore, the attributable portion of disease associated with lead exposure must in volve other sources, with the most likely suspect being ULAB proces sing. Lead exposure can result in a number of lifelong outcomes with adverse implications for individuals as well as the broader society. Consequently, there are clear societal benefits that could be accrued from more targeted investment in remediation, mitigation and policy development to mitigate the worst exposures.

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Competing interest

BE, RD, JC, SF, MR, PS, AS and RF were employed by Pure Earth while working on this manuscript. Pure Earth is a charity that works on pollution issues in low and middle income countries, including India.

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3.4. Paper Nine

Caravanos, J., Carrelli, J., Dowling, R., Pavilonis, **B., Ericson,** B., & Fuller, R. (2016). Burden of disease resulting from lead exposure at toxic waste sites in Argentina, Mexico and Uruguay. *Environmental Health*, *15*(1), 72.

This study relied on data collected on hazardous waste sites in Latin America as part of an international risk assessment program managed by Pure Earth. This study, which focused exclusively on lead exposures, was the most recent in a series of papers that calculated the disease burden attributable to hazardous waste sites in LMICs.[1–4] The overarching goal of this series was to quantify exposures from a limited dataset in an effort to draw conclusions more broadly about the impact of pollution.

This paper utilized a well-accepted WHO method to calculate lead-attributable DALYs at hazardous waste sites in the region. Like papers 6 and 7, this study relies on the USEPA Integrated Environmental Uptake Biokinetic risk assessment tool to model BLLs, which form the most significant input parameter.

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Environmental Health

RESEARCH Open Access

Burden of disease resulting from lead exposure at toxic waste sites in Argentina, Mexico and Uruguay

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Abstract

Background: Though lead contaminated waste sites have been widely researched in many high income countries, their prevalence and associated health outcomes have not been well documented in low and middle income countries.

Methods: Using the well established health metric disability adjusted life year (DALY) and an exposure assessment method developed by Chatham Stephens et al., we estimated the burden of disease resulting from exposure to lead at toxic waste sites in three Latin American countries in 2012: Argentina, Mexico and Uruguay. Toxic waste sites identified through Pure Earth's Toxic Sites Identification Program (TSIP) were screened for lead in both biological and environmental sample media. Estimates of cardiovascular disease incidence and other outcomes resulting from exposure to lead were utilized to estimate DALYs for each population at risk.

Results: Approximately 316,703 persons in three countries were at risk of exposure to pollutants at 129 unique sites identified through the TSIP database. Exposure to lead was estimated to result in between 51,432 and 115,042 DALYs, depending on the weighting factor used. The estimated burden of disease caused by exposure to lead in this analysis is comparable to that estimated for Parkinson's disease and bladder cancer in these countries.

Conclusions: Lead continues to pose a significant public health risk in Argentina, Mexico, and Uruguay. The burden of disease in these three countries is comparable with other widely recognized public health challenges. Knowledge of the relatively high number of DALYs associated with lead exposure may be used to generate support and funding for the remediation of toxic waste sites in these countries and others.

Keywords: Latin America, Burden of disease, Disability adjusted life year, Chemical exposure, Toxic waste sites, Lead poisoning

Background

Environmental exposure to pollution from hazardous waste sites is an understudied contributor to the global burden of disease [1]. Increasing industrial development, urbanization and socioeconomic forces in Latin America have contributed to an increase in environmental pollution and the negative health effects resulting from exposure [2]. The disability-adjusted life year (DALY), which takes into consideration the burden of disease resulting from illness, injury and death, is a standard metric for estimating the burden of disease resulting from exposure to environmental toxicants, among other risk factors. A previous study by Prüss-Ustün estimated that exposure to various chemicals accounts for 5.7 % of total global DALYs and 8.3 % of global deaths [3]. Another study estimated that 0.22 % of the total estimated DALYs from all causes were attributed to pollutants found at hazardous waste sites in India, Indonesia and the Philippines [4].

It is estimated that 94 % of the burden of disease resulting from pollution falls on low- and middle-income countries as defined by the World Bank (LMICs) [5]. While much of the developed world has made significant progress in eliminating the burden of disease caused by infectious diseases, chronic illnesses increasingly affect a great population [6]. Chronic illnesses, such as cardiovascular disease, neurodevelopmental disorders and cancers are

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often linked to environmental exposures, yet enumerating the specific burden of disease impacts from environmental agents has proven difficult [7].

There is a need to better understand linkages between contaminated sites and health outcomes in LMICs. Accurate DALY models enumerated by contaminant, exposure pathway, and affected population offer one possible approach. Summary measures may then be used during the policymaking process to discern what public health threats are of greatest concern and what policies are most effective [8].

Heavy metals are still widely used in the production of consumer goods [9]. In LMICs, inadequate regulation, informality of many industries, poor surveillance, and improper disposal of contaminants can result in dangerous exposures to nearby residents. Of particular concern is lead (Pb). Although the risk of disease resulting from exposure to lead is widely known, widespread use continues [10]. For example, lead is still used to glaze artisanal ceramics despite the availability of less hazardous alternatives. Elevated levels of lead in water and soil can then expose adjacent populations, putting them at risk of a number of adverse health outcomes.

Lead is a bluish-gray metal with many desirable qualities such as electrical conductivity, malleability, density and low-corrosivity and has been mined for centuries, often combined with other metals to form alloys [10]. Anthropogenic sources of lead in the environment include smelting, mining, used lead-acid battery (ULAB) recycling and ceramic pottery making [11, 12]. Compared to adults, children absorb more lead and are therefore more vulnerable to the adverse effects of lead. Early childhood exposure to lead can be particularly harmful and has been shown to cause behavioral problems in adolescence [13], IQ decrements [14], cognitive impairment [15], and decreased visuospatial skills [16]. Adults are typically exposed occupationally and experience higher rates of hypertension than the general population, leading to an increased incidence of cardiovascular disease [17]. Low-level chronic exposure to lead may result in low sperm count or impotence in males. In females, it can result in miscarriage and low birth weight of offspring, as lead may be transferred through the placenta to the fetus [16].

This research aims to accurately quantify the burden of disease caused by lead found at toxic waste sites in Argentina, Mexico and Uruguay. Earlier work done by Caravanos et al. described the pediatric burden of Pb and other heavy metals exposure in several Asian countries [18]. This analysis seeks to elucidate the impact of lead on human health in a different part of the world. The resulting analysis aims to provide a basis for public health intervention and environmental remediation at both the national and regional level, as well as to inform strategies for continued site investigation of contaminated sites.

Methods

Site identification

Environmental and biological exposure data were obtained from the Toxic Sites Identification Program (TSIP). The TSIP is an effort implemented by the New York-based non-profit Pure Earth (formerly Blacksmith Institute) and has been supported by the United Nations Industrial Development Organization (UNIDO), the European Commission (EC), the World Bank, and the Asian Development Bank, among others. The TSIP identifies active and abandoned hazardous waste sites resulting from both formal and informal industrial activities in LMICs. Informal activities include but are not limited to electronic waste or scrap metal recycling, used lead-acid battery recycling, small-scale gold mining, leather tanning, and ceramic pottery making. There are currently more than 3200 sites in the TSIP database, of which 2300 have been visited onsite by a trained TSIP investigator. A majority of the locations screened are abandoned (legacy) sites, including former tanneries and small-scale artisanal sites such as ULAB recycling and artisanal gold mining [4]. The TSIP does not include exposure data from non-point sources such as vehicle traffic or sewage contaminated water. As part of a TSIP investigation, a “key pollutant” is identified and analyzed. Heavy metals are the most commonly occurring key pollutant, with ingestion of contaminated soils being the most commonly occurring route of exposure listed in the TSIP database [19].

Three countries were utilized for this analysis: Argentina, Mexico and Uruguay (Table 1). These countries were chosen primarily on the basis of the availability of data after the inclusion and exclusion criteria outlined below were applied to the raw data. Sites that did not meet all five criteria were not included in the analysis. No additional data collection was conducted for this paper.

In order for a hazardous waste site to be included in the analysis, five criteria must have been met: there must be a credible pathway of human exposure; a biological or

Table 1 Country demographics[a]

Country	Total population (in millions)	Population density (inhabitants per km^{2})	GDP per capita (USD)	Infant mortality (per 1,000)	Life Expectancy (years)
Argentina	41.45	14.4	22375	11	76.01
Mexico	122.3	57	10174	11	77.14
Uruguay	3.4	18.9	16996	9	76.91

[a]The World Bank, 2014

environmental sample had to be present; a population at risk had to be specified; the location of the site was represented by GPS coordinates; and a description of the activities leading to contamination were outlined. The TSIP database contained 23 site surveys in Argentina, 62 in Mexico, and 44 in Uruguay that met the inclusion criteria of this study. A total of 129 sites analyzed with data from 164 environmental lead samples and 75 blood lead level measurements were included in the analysis. It should be noted that while numerous sites contain both blood lead and soil lead data, there are also many sites that contain data from only one sample medium.

Exposure assessment

Local site investigators in the field collected environmental samples with the guidance of a sampling protocol provided by Pure Earth. Biological samples were made available through collections by local health offices and ministries. An independent ethics committee determined that the study was exempt from further review as categorized by the US Department of Health and Human Services Policy for Protection of Human Research Subjects. Lead concentrations in soil were measured in the field using an Innov-X handheld X-ray fluorescence (XRF) spectrometer (4000 Alpha Series; Auburndale/Newton, MA). XRFs are calibrated accordingly prior to soil sample analysis. When an XRF was unavailable, samples were sent to a local laboratory for analysis. Exposure pathways in the analysis included inhalation of dust and ingestion of lead contaminated soil. All Pb exposure was estimated through blood lead levels (BLLs) ($n = 75$) or soil concentrations ($n = 164$). In areas suspected of lead contamination, BLLs were prioritized, as they are the standard marker of human exposure [20]. The U.S. Centers for Disease Control and Prevention (CDC) sets an upper limit of 5 μg/dL for children under the age of 6 years [21]. While the "actionable" reference BLL was lowered from 10 μg/dL to 5 μg/dL in 2012 by the CDC, 10 μg/dL is still the standard reference BLL in most countries [22]. When a site contained less than 5 biological samples, environmental sample data such as lead in soil was used to calculate burden of disease estimates.

Population estimates and age distribution

An age distribution of the population must be used when calculating the burden of disease. It has been well documented that children are more susceptible to negative health effects caused by exposure to toxic pollution than adults [23, 24]. Hazardous chemicals are ingested and inhaled into the body of children at a much higher rate than in adults. Furthermore, toxicants can affect children during critical windows of development when children's bodies and neurological function are most at risk [24]. Children also engage in more high-risk behaviors when compared to adults—they are lower to the ground and tend to have more unwashed hand to mouth contact [18]. As age distribution was not recorded as part of TSIP protocol, province-specific age distributions from the respective countries' census institutions were used in disease estimates [4, 25–28].

A local country specific investigator develops a Conceptual Site Model (CSM) for each site assessed as part of the TSIP. The CSM allows the investigator to determine key sources, migration routes, and chemical exposure pathways. Additionally the investigator using the CSM determines the estimated "population at risk." A population count is then generated from residences and communities adjacent to all sources of exposure using reported housing densities (number of persons per household). High-resolution aerial imagery is also used to confirm population estimates by reviewing the number of people residing within the affected area (defined as having a radius of 50 m). For the purpose of this analysis, population at risk estimates were reviewed against similar sites in the TSIP database.

Risk estimates

Risk was calculated for non-carcinogenic health endpoints based on lead toxicity [4]. Disease incidence and burden for lead were calculated separately using the USEPA's Integrated Exposure, Uptake and Biokinetic (IEUBK) model and tools developed by the World Health Organization (WHO) [29, 30]. The IEUBK model is used to estimate BLLs in children resulting from Pb exposure via soil, air, water, food, and maternal blood lead [20].

The IEUBK Model is a validated tool that estimates the geometric mean of BLL from exposure to multiple sources of lead. However, for this analysis we limited the model to soil lead exposure from each site. We entered these values into the model and calculated mean BLLs for each site. Exposure intakes for air lead levels, dietary intake of lead, water lead levels, maternal BLL and alternate sources of lead were set to "zero" so that the resultant estimated BLL is attributable solely to soil lead exposure. The IEUBK EPA model is specific to children so in estimating adult blood lead levels we applied the USEPA's Adult Lead Methodology (ALM) exposure model [31]. As with IEUBK, only lead in soil inputs were used in the model with all other sources set to "zero".

Incidence of disease

Blood lead levels from exposure to environmental soil and dust lead levels were estimated using the US EPA's IEUBK model. Exposure estimates were calibrated upward to account for the typically dustier conditions of low-income areas in LMICs. Values used elsewhere for indigenous populations were utilized here [32–34]. DALYs resulting from measured blood lead level samples and estimated blood lead levels were calculated separately using disease incidence and spreadsheets created by the WHO [35]. Using these spreadsheets, both incidence of mild mental retardation (MMR) in children and cardiovascular disease in adults were calculated for lead [4].

Burden of disease calculation

The DALY is a time-based measure of health that combines indices of years lived with disability (YLD) and years of life lost (YLL). YLD and YLL were calculated based on exposure estimates collected in the field. YLD is the product of years lived with a disability and a specific disability weight (DW). A DW is scaled between zero and one, with zero representing perfect health and one representing the worst possible state of health (equivalent to death) [36]. For example, mild mental retardation attributable to lead exposure has a DW of 0.36 while metastatic lung cancer has a DW of 0.75 [37].

In order to calculate YLD the relevant type of non-cancer health effect was matched with the corresponding DW (i.e., neurological effects) [4]. Years lived with disability resulted from an estimation of life expectancy multiplied by the appropriate disability weight for the exposure scenario [4]. Years of life lost were calculated only for exposure to carcinogens; as a result, lead exposure did not contribute to YLL [4]. This is the standard method for calculating lead induced MMR, as lead exposure very rarely results in death.

DALYs resulting from cardiovascular disease were transformed into percentages to show a distribution across age groups for each country using the WHO's Global Health Estimates Summary Tables [29]. The percentage of DALYs attributable to ischemic heart disease, cerebrovascular disease, hypertensive disease and all other cardiac diseases were calculated for each country individually. By using BLLs and this percentage of DALYs attributable to cardiovascular disease in a WHO spreadsheet, DALYs attributable to lead exposure were calculated [4, 38].

Age weighting factors, along with a discount rate, were applied to both YLD and YLL to provide a range of DALY estimates. Age weights are applied to burden of disease estimates in an effort to reflect the relative population distribution, while discount rates are often employed in burden of disease studies to account for intergenerational differences in health benefits reaped from public health interventions and a decrease disease incidence [4, 39]. Both the age weights and discount rates are signified in the notation $\mathrm{DALYs}_{(r,K)}$, where r is the discount rate and K is the age weight. Results expressed as $\mathrm{DALYs}_{(3,1)}$ represent a 3 % discount rate (recommended by the U.S. Panel on Cost-Effectiveness in Health and Medicine and utilized by the WHO) and full age weighting, while those expressed as $\mathrm{DALYs}_{(3,0)}$ include only the discount rate [40]. $\mathrm{DALYs}_{(0,0)}$ represent a burden of disease estimate without weighting.

Sensitivity analysis

A range of estimates was also created through a sensitivity analysis by adjusting the size of the population at risk. This analysis calculates the effect of lead on a population plus and minus 25 % of the current estimate to account for possible fluctuations in the population.

Results

Exposure data were collected from a total of 129 hazardous waste sites distributed across Argentina ($n = 23$), Mexico ($n = 62$), and Uruguay ($n = 44$). The geographical distributions of these sites are shown in Figs. 1 and 2. The estimated population at risk of exposure was 316,703 individuals (mean = 2455; median = 250 per site), which is approximately 0.19 % of the total population of all three countries. Of this population, it was estimated that 80,021 were women of childbearing age (15–49 years of age), and 122,084 individuals were younger than 18 years of age (Table 2). Of the exposed population, the proportion of women of childbearing age was relatively equal across the three countries.

Biological ($n = 75$) and environmental measurements ($n = 164$) were used to calculate risk. An arithmetic mean was calculated for environmental or biological samples at each site, unless the site's test results differed more than one order of magnitude. In these cases, a geometric mean was used as outlined in the sampling protocol provided by Pure Earth. Mean blood lead levels (BLLs) in Mexico ($n = 56$) and Uruguay ($n = 19$) were found to be 19.63 μg/dL and 13.3 μg/dL respectively (Table 3). Lead concentrations in soil ($n = 164$) were the highest and relatively uniform across the three countries (Table 4). Mean soil lead concentrations in Mexico (2748 mg/kg) were the greatest, followed by Uruguay (2559 mg/kg) and Argentina (1730 mg/kg).

BLLs were used to estimate DALYs in exposed populations in Mexico (79,196) and Uruguay (5859). As BLLs were not collected in Argentina, DALYs based on that exposure measurement could not be calculated. Elevated BLLs were responsible for 23,421 DALYs in Mexico, representing 52 % of the disease burden estimated as a result of lead exposure. Elevated BLLs were responsible for 942 DALYs in Uruguay, representing 46 % of the total disease burden for lead exposure.

An estimated 27,069 DALYs resulted from exposure to lead in soil. Combined with an estimated 24,363 DALYs based on BLL, overall lead exposure accounted for a total of 51,432 YLDs. The estimated population at risk for exposure to lead was 316,703, largely derived in sites from Mexico (189,593) and Argentina (112,208). An estimated 0.31 $\mathrm{DALYs}_{(3,1)}$ per person resulted from lead exposure at 129 unique toxic waste sites screened in Argentina, Mexico and Uruguay.

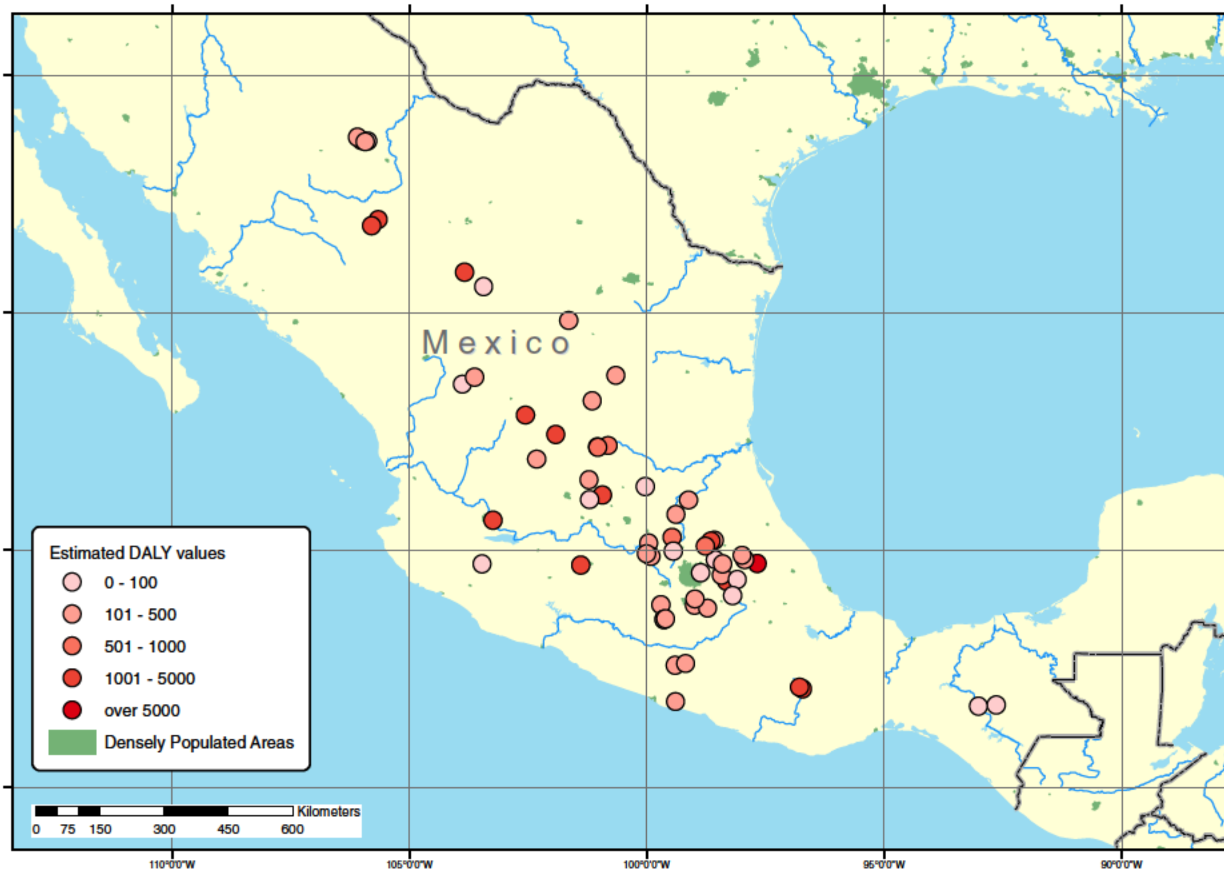

Fig. 1 Geographical distribution of TSIP sites in Mexico with DALYs resulting from lead exposure

Without age weights, 45,492 DALYs $_{(3,0)}$ resulted, while removing both age weight and discount rate resulted in 115,042 DALYs $_{(0,0)}$. To present a range of estimates, exposed population was adjusted to 25 % less than the original estimate, resulting in 38,581 DALYs$_{(3,1)}$. If the exposed population was adjusted to 25 % greater than the original estimate, the resulting DALYs$_{(3,1)}$ were 64,266. A remediation scenario where lead levels were adjusted below international standards resulted in 7,078 DALYs$_{(3,1)}$ (Table 5).

Discussion

This study sought to characterize the number of years lost due to illness, disability, or early death from lead exposure in Argentina, Mexico, and Uruguay. Environmental levels of lead were characterized in those countries and DALYs were calculated based on estimated exposure. In total, an estimated 51,432 DALYs from a total of 316,703 people exposed to lead at 129 toxic waste sites were located throughout the study region. This translates to approximately .31 DALYs$_{(3,1)}$ per person. The estimated burden of disease as a result of exposure to lead was approximately 0.12 % of DALYs for all causes as estimated by the WHO in Argentina, Mexico and Uruguay [29].

By quantifying disease burden from lead pollution through a DALY-based method developed by Chatham-Stephens et al., comparisons can be made to other public health threats and illnesses. The modeled burden of disease estimated for exposure to lead at screened sites is comparable to the burden resulting from more widely recognized public health issues such as Parkinson's disease (52,800 DALYs), Acute Hepatitis B and C combined (43,300) and bladder cancer (59,500 DALYs) in the three countries analyzed [28, 40]. The estimated burden of disease due to lead exposure is also greater than estimates for all childhood-cluster diseases including pertussis, diphtheria, measles, and tetanus (9,100 DALYs) and multiple sclerosis (27,500 DALYs) [29]. A comparison of DALYs from lead exposure and other health outcomes in Argentina, Mexico and Uruguay can be seen in Table 6, though it must be reiterated that DALYs from lead exposure are estimated rather than empirical.

The ingestion and inhalation of lead contaminated soil and dust was the main exposure pathway in the data analyzed. Biomarkers (blood lead levels) were used in the

Fig. 2 Geographical distribution of TSIP sites in Argentina and Uruguay with DALYs resulting from lead exposure

calculation of disease burden for lead exposure in Mexico and Uruguay, accounting for 47.4 % of the DALYs estimated in those countries.

Mexico is the fourth-largest producer of lead worldwide, with 222,000 metric tons generated in 2012 and a continually increasing output [41]. Sites in Mexico included in the lead exposure analysis were currently or previously involved with production of earthenware with leaded glaze ($n = 31$), mining operations ($n = 22$), smelting activities ($n = 3$), used lead-acid battery recycling ($n = 1$) and manufacturing ($n = 5$).

If BLLs were adjusted to below the "actionable" limit recommended by the CDC (5 ug/dL), an estimated 24,281 DALYs in Mexico (23,342 DALYs) and Uruguay (939 DALYs) could be eliminated. Such interventions include the introduction of lead-free glaze in ceramic ware, legislation to regulate battery-recycling, reduction of lead dust in homes, education about the health effects resulting from Pb exposure, as well as continued monitoring of BLLs. In comparison with the initial $DALYs_{(3,1)}$ estimate, 44,354 $DALYs_{(3,1)}$ could be eliminated if these sites were remediated. Despite producing lead in smaller quantities, exposure contributed significantly to disease burden in both Argentina (83,700 metric tons from primary and secondary lead smelting in 2013, 4,061 $DALYs_{(3,1)}$) and Uruguay (no lead production data available, 2,051 $DALYs_{(3,1)}$) [42].

A previous burden of disease study by Chatham-Stephens et al. found 54,432 DALYs attributable to lead

Table 2 Exposed population by age and childbearing status

Country	Total Exposed Population (Population)	Women of Childbearing Age 15 49 years old (Population)	All Genders <18 Years old (Population)	Remaining Age Groups (Population)
Argentina	112208	27852	36659	47697
Mexico	189593	48770	80904	59919
Uruguay	14902	3399	4521	6982
Total	316703	80021	122084	114598

Table 3 Blood Lead Level (BLL) Data by Country

Country	N	Mean (μg/dL)	S.D. (μg/dL)	Range
Argentina	N/A	N/A	N/A	N/A
Mexico	56	19.63	9.9	7.7 41.1
Uruguay	19	13.3	9.21	5.1 47.35

exposure in India ($n = 24$), 78,982 DALYs in Indonesia ($n = 28$) and 394,084 DALYs in the Philippines ($n = 27$). While these estimates are larger than the estimated 45,321 DALYs attributable to lead-contaminated sites in Mexico ($n = 62$), an estimated 0.41 DALYs per person resulted from lead exposure at these sites, higher than previous estimates for India (0.21 DALYs per person), Indonesia (0.21 DALYs per person) and the Philippines (0.30 DALYs per person). In the same study, 0.10 DALYs per person were estimated for exposure to eight chemicals in India, Indonesia, and the Philippines (mean population at risk of exposure per site = 23,079) [4], while an estimated .31 $DALYs_{(3,1)}$ per person occurred due to exposure to lead in Argentina, Mexico and Uruguay (mean population at risk of exposure per site = 2455). This higher average DALY per person was likely a result of a smaller population at risk and higher lead concentrations found in the three countries in this review.

A number of limitations for the calculation of disease burden should be noted. One such limitation has to do with extrapolation from a limited number of samples. The TSIP assessment process relies on minimal environmental sampling, composed of targeted and composited samples. The methodology was developed for screening purposes and is insufficient to fully characterize health risks at a site. As a result, the estimates here are necessarily indicative rather than definitive in nature.

A second significant limitation has to do with the limited number of sites captured by the TSIP. The number utilized here, 129, is very likely a significant undercount of the total number. Future efforts might endeavor to document additional sites or develop a robust methodology for modeling what that number might be.

A final limitation is the singular focus on lead. TSIP site investigators collect data for a range of pollutants including arsenic, hexavalent chromium, mercury, pesticides and particulate matter contributing to air pollution. However, these analyzed samples were too few in number to generate an accurate burden of disease estimate. In order for this exercise to be repeated with other pollutants, both data collection and site identification need to be improved.

Table 4 Environmental Lead Sample Data

	Argentina		Mexico		Uruguay	
	N	Mean[a]	N	Mean[a]	N	Mean[a]
Soil Samples (mg/kg)	48	1730	59	2748	57	2559

[a]Value shown is either the mean or single measurement

Table 5 Sensitivity analysis estimates

Scenario	Total DALYs
Primary estimate of screened sites	51432 $DALYs_{(3,1)}$
Estimate without age weights	45492 $DALYs_{(3,0)}$
Estimate without age weights or discount rate	115042 $DALYs_{(0,0)}$
Remediation scenario	7078 $DALYs_{(3,1)}$
If actual exposed population is 25 % less	38581 $DALYs_{(3,1)}$
If actual exposed population is 25 % greater	64266 $DALYs_{(3,1)}$

The use of mercury in artisanal small-scale gold mining (ASGM), for example, is a known threat to public health in Latin America, and future site investigations must continue to identify sites of mercury exposure [43]. As the analysis was solely focused on exposure to lead, it is likely that the burden of disease resulting from exposure to toxic pollution is largely underestimated.

Conclusion

Intervention and remediation programs must focus on lead-contaminated sites in Argentina, Mexico and Uruguay as exposure to lead continues to contribute a significant disease burden for the population in these countries. An estimated 316,703 persons are subject to lead exposure at screened sites in these countries, resulting in 51,432 $DALYs_{(3,1)}$. However, site investigations and efforts to estimate the burden of disease caused by pollution must continue to incorporate threats from exposure to mercury, arsenic, hexavalent chromium, pesticides, air pollution and

Table 6 DALY Comparisons by Health Outcome

Selected Outcomes and Exposures	Total DALYs
Leishmaniasis	2800
Childhood cluster Diseases[a]	9100
Multiple Sclerosis	27500
Chlamydia	39100
Acute Hepatitis B and C	43300
Lead Exposure (Modeled)	**51432**
Parkinson's Disease	52800
Bladder Cancer	59500
Melanoma and Skin Cancers	63800
Tuberculosis	141500
Asthma	295700
Diarrheal Disease	375100
HIV	469100
Respiratory Infections[b]	153500
Diabetes Mellitus	3102600

DALYs 2012 estimates (WHO, 2014)
Bold data are based on our modeled estimate using a method developed by Chatham Stephens et al
[a]Childhood cluster diseases include pertussis, diphtheria, measles, and tetanus
[b]Respiratory infections includes lower respiratory infections, upper respiratory infections, and otitis media

other contaminants. Future studies should attempt to extrapolate these estimates to unscreened sites in an effort to approximate a more accurate burden of disease. This larger estimate is likely to be comparable with the burden of disease resulting from myriad chronic illnesses, and may be used as a tool to generate support and funding for the remediation of toxic waste sites in these countries and others. While the three countries of study have protocols in place to monitor children's BLLs and reduce lead exposure, programs to regulate ULAB recycling exist only in Argentina, and regulations limiting the content of residential paint exist in only Argentina and Uruguay. Efforts to reduce the burden of disease resulting from lead exposure such as these and others must be implemented in all countries to adequately reduce the burden of disease from lead exposure.

Abbreviations

ALM, Adult lead methodology; ASGM, Artisanal small scale gold mining; BLL, Blood lead level; CDC, U.S. Centers for Disease Control; CSM, Conceptual Site Model; DALY, Disability adjusted life year; DW, Disability weight; EC, European Commission; IEUBK, Integrated Exposure, Uptake and Biokinetic model; LMICs, Low and middle income countries; MMR, Mild mental retardation; TSIP, Toxic Sites Identification Program; ULAB, Used lead acid battery; UNIDO, United Nations Development Organization; USEPA, United States Environmental Protection Agency; WHO, World Health Organization; YLD, Years lived with disability; YLL, Years of life lost

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Authors' contributions

BE, JoC and RD conceived this study based on an exposure assessment method developed by Chatham Stephens et al. JoC carried out the burden of disease analysis and drafted the manuscript. RD, BP, JaC, BE and RF reviewed the paper and provided comments, edits and senior level guidance. All authors read and approved the final manuscript.

Competing interests

The authors declare that they have no competing interests.

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4. Chapter Four: Mitigation

Chapter 4 consists of the following 3 studies:

4.1 **Ericson, B.,** Duong, T. T., Keith, J., Nguyen, T. C., Havens, D., Daniell, W., ... & Wilson, B. (2018). Improving human health outcomes with a low-cost intervention to reduce exposures from lead acid battery recycling: Dong Mai, Vietnam. *Environmental Research*, *161*, 181-187.

4.2 Heacock, M., Trottier, B., Adhikary, S., Asante, K. A., Basu, N., Brune, M. N., … **Ericson, B** ... & Chen, A. (2018). Prevention-intervention strategies to reduce exposure to e-waste. *Reviews on Environmental Health*, *33*(2), 219-228.

4.3 **Ericson, B.,** Caravanos, J., Depratt, C., Santos, C., Cabral, M. G., Fuller, R., & Taylor, M. P. (2018). Cost Effectiveness of Environmental Lead Risk Mitigation in Low-and Middle-Income Countries. *GeoHealth*, 2(2), 87-101.

Chapter 4 of this thesis presents methods to mitigate lead exposures described in earlier Chapters. All three studies presented in this Chapter evaluate the results of remediation projects at lead contaminated sites in LMICs. An overarching finding of this thesis is that lead contaminated sites, particularly those resulting from informal ULAB recycling, pose an important and under recognized exposure risk. This Chapter explores the efficacy of some solutions to this difficult problem that have been used and tested as part of this thesis.

This Chapter covers four distinct lead risk mitigation projects in four different LMICs (The Dominican Republic, Ghana, Uruguay, and Vietnam). Each intervention was designed to be cost-effective and tailored for the affected community based on the available resources. In the context of limited resources in LMICs to mitigate exposures at lead contaminated sites, cost-effective and locally employable solutions are required. Paper 10 describes a project in detail in part to provide guidance to implementers elsewhere. Paper 11 provides summaries of several projects and includes recommendations for implementers.

Finally Paper 12 addresses this issue of cost effectiveness by reviewing a single project in the Dominican Republic. To complete this analysis, the study relies on data collected *in-situ* before and after the intervention to assess the potential extant exposure risk, calculate the outcomes, and describe the intervention. The study bridges the themes of the three central Chapters of this thesis (Exposures, Outcomes, and Mitigation), thereby providing a conclusion.

4.1. Paper Ten

Ericson, B., Duong, T. T., Keith, J., Nguyen, T. C., Havens, D., Daniell, W., ... & Wilson, B. (2018). Improving human health outcomes with a low-cost intervention to reduce exposures from lead acid battery recycling: Dong Mai, Vietnam. *Environmental Research*, *161*, 181-187.

Paper ten directly addresses the mitigation theme of Chapter 4. This study details and evaluates an environmental and social intervention executed at a lead contaminated village in Vietnam from 2013–2015. Many of the village's population of approximately 3,000 people were engaged in backyard lead-acid battery recycling operations. The activities resulted in widespread contamination of the environment and severely elevated BLLs. This paper describes a low-cost intervention targeting the most significant exposure risks. The efficacy of the intervention was benchmarked using a range of metrics, in particular pre- and post-intervention childhood BLLs. The demonstrable benefits of the intervention approach applied resulted in its adoption for use in other similarly industrialized villages in the Vietnam.

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Improving human health outcomes with a low-cost intervention to reduce exposures from lead acid battery recycling: Dong Mai, Vietnam

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ABSTRACT

This study details the first comprehensive evaluation of the efficacy of a soil lead mitigation project in Dong Mai village, Vietnam. The village's population had been subject to severe lead poisoning for at least a decade as a result of informal Used Lead Acid Battery (ULAB) recycling. Between July 2013 to February 2015, Pure Earth and the Centre for Environment and Community Development (Hanoi, Vietnam) implemented a multi-faceted environmental and human health intervention. The intervention consisted of a series of institutional and low-cost engineering controls including the capping of lead contaminated surface soils, cleaning of home interiors, an education campaign and the construction of a work-clothes changing and bathing facility. The mitigation project resulted in substantial declines in human and environmental lead levels. Remediated home yard and garden areas decreased from an average surface soil concentration of 3940 mg/kg to < 100 mg/kg. One year after the intervention, blood lead levels in children (< 6 years old) were reduced by an average of 67%—from a median of 40.4 µg/dL to 13.3 µg/dL. The Dong Mai project resulted in significantly decreased environmental and biological lead levels demonstrating that low-cost, rapid and well-coordinated interventions could be readily applied elsewhere to significantly reduce preventable human health harm.

1. Introduction

In high income countries (HICs), regulatory controls, notably bans on lead in widely used and available products (e.g. residential paint, gasoline), have resulted in significant lowering of population blood lead levels (Kristensen et al., 2017; Needleman, 2004; Schwartz and Pitcher, 1989). In Low and Middle Income Countries (LMICs) key sources of environmental lead exposure include mining (both legacy and active), used lead acid battery (ULAB) processing, and lead based ceramic glazes, among other sources (Farías et al., 2014; Lo et al., 2012; Meyer et al., 2008; Yabe et al., 2015). Used lead acid battery processing in particular is known to cause significant environmental contamination and human health exposure (Farías et al., 2014; Lo et al., 2012; Meyer et al., 2008; Yabe et al., 2015). In the context of limited regulatory oversight the extent and severity of lead poisoning in LMICs is less well documented but is suspected to be prevalent (Braithwaite, 2006; Chatham Stephens et al., 2013; Dowling et al., 2016; Ericson et al., 2013). Lead is a known neurotoxicant and can result in an IQ decrement in children and cardiovascular disease in adults, among other adverse health outcomes (ATSDR, 2007). On a societal level, lead exposure has been associated with increased levels of aggravated assault and de creased economic output (Gould, 2009; Mielke and Zahran, 2012; Prüss Üstün et al., 2010).

Multiple studies have documented very high blood lead levels (BLLs) in communities where informal ULAB processing occurs (Daniell et al., 2015; Haefliger et al., 2009; Matte et al., 1991). This activity is typically undertaken in residential areas and is characterized by poor or no hazard control and migration of material offsite (Shen et al., 2016).

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A recent study has estimated that between 10,599 to 29,241 such sites exist in 90 different LMICs, placing the health of 6 16 million people at risk (Ericson et al., 2016). A limited number of projects have been executed at comparable lead contaminated sites to mitigate exposures, with most utilizing a combination of in situ and ex situ engineering controls coupled with community education (Laidlaw et al., 2016; Pure Earth, 2017). However, there is an absence of peer reviewed studies evaluating the efficacy of remediation strategies at informal ULAB and remediation more broadly in LMICs.

1.1. Study location and background

In Vietnam, informal industry including ULAB recycling is commonly centered in 'craft villages'. These small to medium sized areas produce a range of consumer and industrial goods and are typically characterized by inadequate waste management practices (Mahanty and Dang, 2013). Craft village industrial activities are also characteristically household based, with individual homeowners often working in concert to complete sequential tasks in the production of a single good (Mahanty and Dang, 2013). This research focuses on the outcomes of a multi faceted intervention to reduce lead exposures in the Dong Mai village, Chi Dao Commune, Hung Yen Province. Dong Mai village has been involved in ULAB recycling activities since 1978 (Tung, 2011).

The recycling process involved the collection of automotive batteries from outside the village, battery breaking for lead plate removal, smelting to form new lead ingots, and manually recovering and processing the resulting waste tinker and slag for re smelting. These processes were replicated across the village with the majority of households participating in some stage of the recycling process. Consequently, the impact of the recycling activities resulted in pervasive contamination across the whole village.

Several previous studies have described the contamination in Dong Mai (Daniell et al., 2015; Noguchi et al., 2014; Tung, 2011). A 2006 National Institute of Occupational and Environmental Health (NIOEH) study found elevated lead concentrations in air, wastewater and soil (Tung, 2011). A separate effort by a Japanese research team in 2011 found severely elevated blood and urine lead levels due to occupational exposure in adults. The study evaluated 93 individuals, including 23 children, and found an average BLL of 34 μg/dL (Noguchi et al., 2014).

The most comprehensive study of exposures to date was completed by a joint effort of the NIOEH and the University of Washington School of Public Health (USA) (referred to hereafter as 'the University of Washington'). In these assessments, lead in surface soil and dust in 11 different homes along with capillary blood samples of 109 children (age 0 10 years) were evaluated. The results revealed BLLs ranged between 12 > 65 μg/dL (the detection limit of the LeadCare® II analytical equipment) and extensive soil contamination with two thirds of samples above the USEPA reference level for bare soil in children's play areas of 400 mg/kg (Daniell et al., 2015; EPA, 2015). For context, the equivalent Vietnamese standard is 70 mg/kg (Vietnam, 2015). Of the 109 children evaluated 33% had BLLs of 10 29.9 μg/dL; 37% 30 44.9 μg/dL; 16% 45 64.9 μg/dL, and 14% > 65 μg/dL (Daniell et al., 2015).

Following the 2006 NIOEH study (Tung, 2011) the People's Committee of Hung Yen province established an industrial park for Chi Dao Commune to consolidate ULAB recycling activities and extricate the activity from Dong Mai village. The new industrial park covered 200,000 m^2 and was located about 1 km south of the village's residential areas. Subsequently, from around 2012, the majority of ULAB industrial activity was relocated to the industrial area, although a minority continued recycling activities in the village. About 66,000 m^2 of the new industrial park is currently in use.

In 2012, Pure Earth, an international non profit organization dedicated to solving pollution problems in low and middle income countries where human health is at risk, became aware of the site through its Toxic Sites Identification Program (Ericson et al., 2013). From 2013 2015 Pure Earth worked jointly with the Vietnamese Centre for Environment and Community Development (referred to hereafter as 'CECoD') to assess the extent and severity of contamination and execute a targeted intervention. The intervention was supported by national and international organizations: the Centre for Environmental Consultancy and Technology of the Vietnam Environment Administration (referred to hereafter as the 'Environmental Administration') (Hanoi, Vietnam), the International Lead Management Centre (Research Triangle Park, USA), the University of Washington, and local community and industry partners. This study details the intervention and assesses its efficacy and potential for wider application in similarly impacted LMICs, particularly where financial resources are limited.

Assessment of the key sources of lead exposure in Dong Mai included contaminated indoor dust and soils in outdoor residential and public spaces. Soil has long been identified as a significant pathway of lead exposure (Mielke and Reagan, 1998). Children in particular ingest high levels of dust from soil and soil itself (Abrahams, 2002; Stanek and Calabrese, 1995).

At the study site, a primary source of contamination was identified as legacy waste from previous recycling activities, which because of its scale had resulted in exposures across the village. In addition to the legacy waste sources, ongoing recycling activities were also identified as a secondary source of contemporary contamination. These recyclers were engaged in manual tinker separation, battery breaking and smelter operations. Lead dust contained on all worker's clothing was also an important source of exposure to the workers and their families (Daniell, 2015). A further source of contamination involved the loss of primary lead material during its transport through the village on motorbikes and small trucks to the formal industrial area for processing.

2. Materials and methods

The approach employed in Dong Mai involved a series of increasingly detailed environmental assessments to guide targeted interventions. Environmental and human exposure monitoring were used to assess their efficacy.

2.1. Environmental assessment

Assessments of environmental contamination were carried out on an ongoing basis, beginning in December 2012 with a rapid qualitative assessment of contamination sources and exposure pathways by CECoD investigators. The rapid assessment confirmed earlier reports of poor work practices including breaking and open smelting of batteries in residential areas, lack of personal protective equipment and the storage and transport of hazardous material in the village.

A detailed assessment of the extent of soil contamination was conducted by Pure Earth in May 2013. In situ surface soil lead measurements at 235 sites were completed over two days in residential areas and public spaces using a handheld portable InnovX Delta series X ray fluorescence instrument (pXRF) with a lower detection limit for lead of 5 mg/kg. The instrument was calibrated twice daily with a 316 steel clip provided by the manufacturer. Following calibration, certified reference materials produced by the National Institute of Standards and Technology (2702: Inorganics in Marine Sediment) were used to confirm the accuracy of lead detection (Gonzalez et al., 2016). In situ pXRF analysis has been shown to be consistent with more commonly used wet chemistry techniques (e.g. Inductively Coupled Plasma Atomic Emission Spectrometry and Inductively Coupled Plasma Mass Spectrometry), particularly in the context of high density sampling like that carried out here (Rouillon et al., 2017; Rouillon and Taylor, 2016). Water samples were not collected by the assessment team as previous data indicated this lead exposure pathway was low risk (Anh, 2008).

From September 2013 to February 2014 the Environmental Administration conducted a qualitative assessment of every village residential yard (n = 546). At that time 269 households were identified as having earthen yards or gardens. Investigators were unable to access 14 of these yards. The balance (n = 255) of yards were assessed in detail with > 5 surface samples per yard analyzed using the pXRF. Analysis of lead in soil across individual yards was evenly spaced in order to capture acurately contamination levels in areas accessed by residents. To assess the depth of contamination in the soils, a limited number (n = 10) of samples from contaminated yards were assessed to a maximum depth of 1 m.

Results of the surface soil lead testing were compiled using ESRI ArcMap 10.5 and analyzed for proximity to roadways (ESRI, 2015). Statistical analysis was conducted with Stata 14 (StataCorp LP, 2015).

2.2. Lead contamination mitigation

The primary mitigation work was conducted over a 4 month period from December 2013 to March 2014. An additional phase conducting work in 11 homes and lasting one month was executed in January 2015. The soil mitigation strategy consisted of covering contaminated earthen yards with a geotextile fabric followed by 20 cm of compacted clean soil and was explicitly based on the method described by Mielke et al. (2011). Alluvial soils were procured for the capping and tested for lead content with a pXRF and in all cases were confirmed to be < 50 mg/kg. In cases where residents opted to pay the incremental cost of capital improvement, other materials were used in addition to the compacted clean soil. These included paver stones and concrete. Overall, the interventions had the same intent isolating the con taminated soils from causing potential human exposures. Capping with a clean layer of soil ≥ 10 cm is an internationally accepted approach to dealing with soil lead contamination and has been used in the United States and Australia, for example (Mielke et al., 2011; Yang and Cattle, 2017). By contrast, capping with soil ≤ 5 cm has been ineffective at mitigating potential exposures (Harvey et al., 2016).

Due to budget constraints, mitigation work was prioritized based on the severity of contamination. Three thresholds were set: 400 mg/kg 800 mg/kg (low priority); 800 1200 mg/kg (medium priority); and > 1200 mg/kg (high priority). The lowest value, 400 mg/kg, represents the USEPA guideline level for bare soil in children's play areas, while 1200 mg/kg represents the USEPA guideline for bare soil where chil dren do not play (EPA, 2015). A soil lead level of 800 mg/kg was chosen to further stratify the homes.

Out of the 255 yards assessed in detail, the project team undertook mitigation in 49. An additional 47 homeowners mitigated their own yards utilizing the project's protocol. Thus mitigation work was exe cuted in a total of 96 yards. The total area of land capped during the project was 11,370 m^2. Twelve homes identified with average surface soil concentrations between 400 800 mg/kg were not mitigated (Fig. 1).

Home interiors were cleaned using a Pure Earth protocol developed on previous lead exposure mitigation projects (Keith and Ericson, 2013). The protocol was translated into the Vietnamese language and photographs of local residents were used to demonstrate cleaning ap proaches. The project team utilized High Efficiency Particulate Air (HEPA) vacuums in the 49 homes targeted by the project. The re maining homes, including the 47 that capped their own yards, were internally cleaned using the HEPA vacuums followed with spot checks of dust by the project team with a pXRF.

In parallel with the Pure Earth intervention, a Vietnamese rural development scheme was engaged by residents to provide co financing for the concreting of streets in the village. As a result more than 80% of secondary roads in Dong Mai were covered with concrete after the project compared with none before the project.

2.3. Community education

A community education and engagement campaign covering health risks and exposure pathways was undertaken by CECoD. The campaign

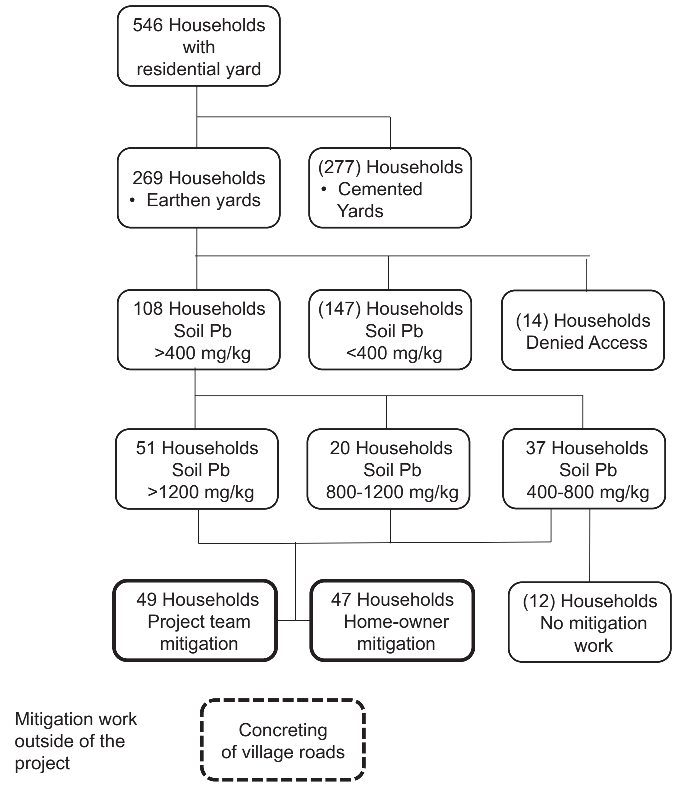

Fig. 1. Flow chart of assessment and mitigation activities.

included a three day workshop where a USD $10 stipend was provided, resulting in the attendance of 540 households (from a total of 546). In addition a door to door education campaign was conducted and regular announcements were made on the village's loudspeaker system.

2.4. Improving ULAB workshop work practices

At the beginning of the project 34 informal processors were iden tified in the residential area of the village. These processors were in cluded in stakeholder workshops and were prioritized for community education activities. Other than this, they were not engaged differently than the community as a whole.

An International Lead Management Centre representative (ilmc.org) conducted an assessment of the formal facility and industrial zone in September 2013 and made multiple recommendations on improving workplace practices (Wilson, 2013). One significant outcome of the assessment was the design and building of a 'clean in, clean out' changing room for workers. The purpose of the clean in clean out changing room was to mitigate the migration of material offsite on workers' clothing. The facility was co financed by a private smelter owner who provided USD$15,000 in labor and a site for the new building. The PE project provided USD$12,500 and the building design. The importance of using the facility and a number of recommendations for improved hygiene were covered in the community education workshops discussed above.

2.5. Blood lead level (BLL) assessment

The University of Washington and NIOEH conducted pre and post intervention blood lead testing in December of 2013 and September of 2014 using a LeadCare® II instrument and capillary blood samples. Two hundred and four children (< 6 years old) were sampled at both time points. Using US Census data for foreign countries we calculated that 10.8% of the village's estimated population of 2600, or 281 children, fell within this age group, indicating that the study captured BLLs for

73% of the population under 6 years of age (US Census Bureau, 2016). The age distribution of the 2013 2014 cohort was as follows: 7% ≤12 months, 23% 12 24 months, 20% 24 36 months, 14% 36 49 months, 17% 48 60 months, and 19% 60 72 months at the time of the initial test. Institutional Review Board approval was given by the Human Subjects Division of the University of Washington and by the NIOEH Institutional Review Committee. In May 2015 NIOEH collected a further 196 intravenous blood samples from village children (< 6 years old), which were assessed for lead concentration using atomic absorption spectrometry (NIOEH, 2015). The age distribution of the 2015 cohort was as follows: 7% ≤12 months, 18% 12 24 months, 28% 24 36 months, 17% 36 49 months, 12% 48 60 months, and 18% 60 72 months at the time of the test.

3. Results

3.1. Soil Lead Levels

Average soil concentrations in Dong Mai before the intervention were 3940 mg/kg (95% CI: 1567 6312 mg/kg; median = 648 mg/kg). Yards assessed in detail are represented by their average concentration and are labeled as 'Composite' (Fig. 2). The contamination resulted from multiple hotspots occurring sporadically throughout the village rather than a single source. No association could be identified between surface level lead concentrations and proximity to roads. Samples taken within 3 m of a roadway averaged 3774 mg/kg; samples between 3 and 10 m of a roadway averaged 5391 mg/kg, while those more than 10 m from a road averaged 2505 mg/kg. Contamination was evident in deeper soils and in some cases increased with depth. This finding was consistent with anecdotal evidence that much of the village had been constructed upon slag infill.

Detailed household assessment work undertaken by the Environmental Administration showed that 34 of the 546 homes in the village had active ULAB processing before the project. Just over 50% of all homes (269) had earthen yards and approximately 25% (108) had average surface soil concentrations below 400 mg/kg. A smaller number of homes exceeded the following surface soil concentrations: 400 800 mg/kg (37 homes); 800 1200 mg/kg (20 homes); > 1200 mg/kg (51 homes).

At the close of the project, average surface soil lead concentrations in all 96 targeted yards were confirmed to be below the cleanup threshold value of 100 mg/kg. A 2016 pXRF survey of 20 randomly selected points throughout the village was unable to identify lead in surface soils above the instrument detection limit of 5 mg/kg. A detailed survey of yards after the project was not undertaken.

3.2. Improving ULAB workshop work practices

At the start of the project 34 homes were actively processing batteries in the residential area. A 2016 survey revealed that all processors had either closed or had relocated to the new industrial area. Unfortunately, the same survey also revealed under utilization of the changing room, with most workers preferring to change and bathe at home.

3.3. Blood Lead Levels

The BLLs of 204 Dong Mai children aged 0 6 years were obtained at the beginning of the intervention in December 2013 and six months after its completion in September 2014 with the LeadCare® II. Assessment of BLLs before and after the intervention revealed median blood lead concentrations that decreased by 37%, from 40.35 µg/dL (IQR = 30 59.2) to 25.35 µg/dL (IQR = 19.05 36.85), indicating the intervention helped lower children's lead exposures. Of the 204 children tested, 86% had decreased BLLs (n = 176), 3% showed no change (n = 7), and 14% had increased BLLs during this time period (n = 21).

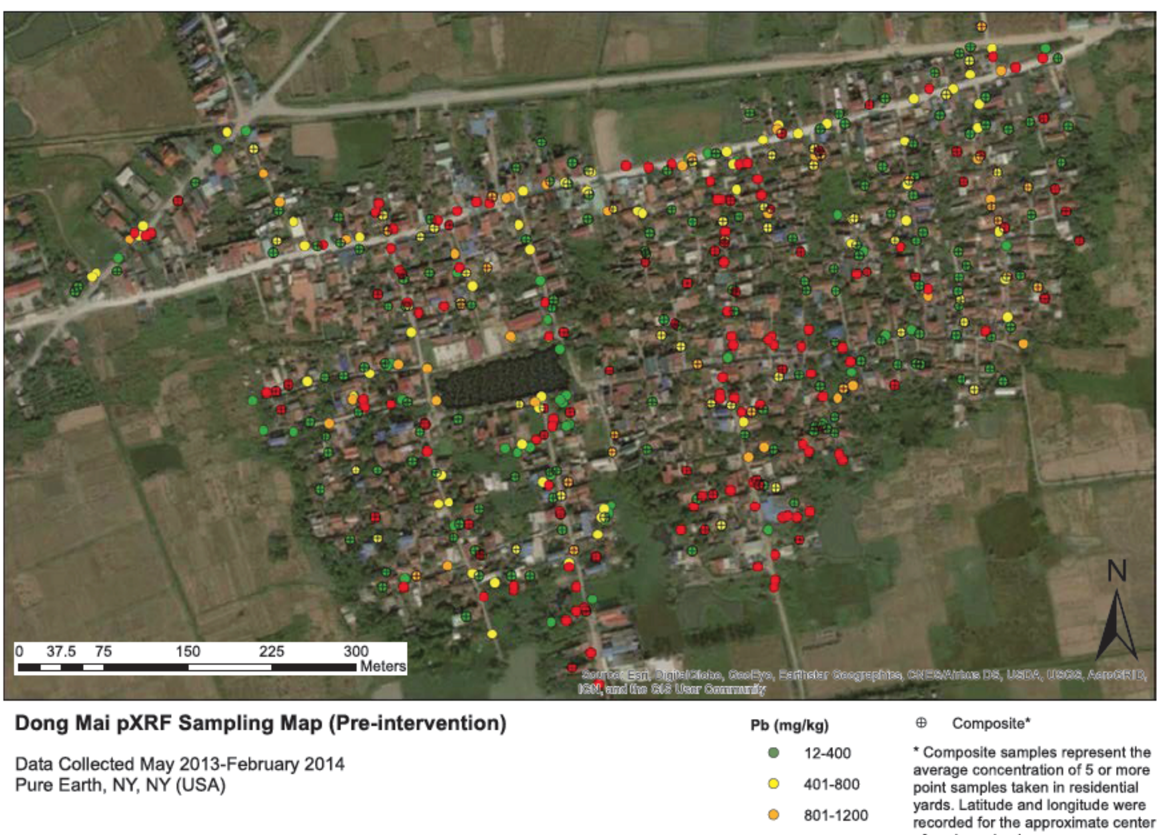

Fig. 2. Surface soil lead concentrations before the intervention. Samples with crosshairs indicate residential yard averages consisting of 5 or more point samples.

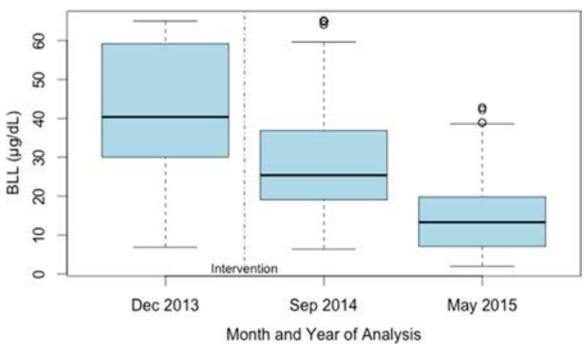

Fig. 3. Results of blood lead levels (BLL) analysis of Dong Mai children (< 6 years) before and after the intervention. Assessment of post remediation BLLs in May 2015 showed exposures were statistically lower than pre-remediation values collected in December 2013 ($t(192) = 21.3511$, $p < .0005$).

The effectiveness of the intervention was benchmarked by additional blood lead analysis in 2015 that comprised 196 children aged 0 6 years, which found further declines in exposures. The median of samples analyzed in 2015 was 13.3 µg/dL (IQR = 7.1 19.8), down from 40.35 µg/dL in December 2013 and 26.35 µg/dL in September 2014. Importantly, when comparing the 2015 analysis to the results 2013 and 2014 data, it is worth noting that the most recent data were derived from analysis of blood using atomic absorption. The 2013 and 2014 analyses were conducted using a LeadCare® II. Fig. 3 summarizes graphically the stepped reduction in blood lead levels from assessments between 2013 and 2015.

Forty nine (24%) of the children assessed in 2013 had BLLs exceeding the upper detection limit of the LeadCare® II device (65 µg/dL). This number decreased to 11 (5%) in 2014 and 0 in 2015. Because of the large number of children above the upper detection limit in 2013 the median presented for that year is likely much lower than the actual population wide median.

4. Discussion

It is well established that lead exposure can result in a number of lifelong health and socioeconomic outcomes including cardiovascular disease, intellectual disability and reduced lifetime earnings (ATSDR, 2007; Lanphear, 2015; Reuben et al., 2017). Other studies have found an association between environmental lead exposure and societal violence (Mielke and Zahran, 2012). In NIOEH's (2015) report on blood lead levels in Dong Mai, they indicate that 16.7% of the children tested showed signs of mental illness, defined as mental retardation, attention deficit hyperactivity, low development of language skills or having difficulties with studying. Pediatric lead exposure is a well established risk factor for these important childhood morbidities (e.g. Baghurst et al., 1992; David, 1976; David et al., 1972; Dietrich et al., 1993; Grandjean and Landrigan, 2006; Lanphear et al., 2005).

In the most severe cases, acute lead poisoning can be fatal. A major lead contamination event in Senegal in 2010 from informal recycling of car batteries resulted in the death of 20 children (Haefliger et al., 2009). A separate event in the Zamfara state of Nigeria in 2012 resulted in the death of more than 500 children in 2012 (Lo et al., 2012). While there were no reported deaths in Dong Mai attributable to lead pollution, chronic lead poisoning can also result in increased morbidity and mortality from a range health effects including cardiovascular disease (Fewtrell et al., 2003). Aerial deposition of lead from previous leaded gasoline emissions globally has been estimated to be responsible for nearly 500,000 excess deaths annually (Forouzanfar et al., 2016). Lead is highly immobile in the environment and likely to remain in surface soils where exposure can continue to occur (Komárek et al., 2008; Kabala and Singh, 2001; Semlali et al., 2004). Therefore without physical intervention, lead contaminated sites in residential environments will continue to pose a risk to human health indefinitely. In the case of informal ULAB sites in LMICs, there is a need to develop cost effective intervention strategies that can be employed using local resources. The magnitude of contamination from ULAB is a major global problem with an estimated 10,599 to 29,241 sites across 90 different LMICs, posing a risk of harm to the health of between 6 and 16 million people (Ericson et al., 2016).

Resources to execute lead exposure mitigation projects in LMICs are typically limited. Gross National Incomes per person in this income group are below USD $12,475 and no existing multilateral agreement to facilitate funding of lead remediation exists (European Commission, 2017; World Bank, 2016a). It is therefore necessary to both prioritize the most severe sites and identify cost effective approaches to mitigating exposures. In the case of Dong Mai the total project cost was USD $118,750. Future efforts might endeavor to evaluate this project's cost effectiveness relative to other public health interventions (cf. Gould, 2009).

The mitigation method employed by the project offers a number of advantages in the resource poor context of LMICs when compared to other in situ and ex situ approaches (Laidlaw et al., 2016). There is a general dearth of hazardous waste repositories in LMICs, including in Vietnam (Mmereki et al., 2016; Thai, 2009). One facility in the region located some 40 km from Dong Mai agreed to store the material and quoted a disposal cost to the project of $ 400 per m^3. Excavation and disposal of 11,370 m^2 of contaminated surface soils versus capping with clean soils would have had significant budget implications. Assuming excavation of the top 20 cm of soil only, the cost of disposal (before labor, supplies, and transport) would have been approximately USD $909,600, or USD$9475 per house; 4 times the Vietnamese GDP per person (World Bank, 2016b). Thus, the cost of such an approach would have been prohibitive. By contrast, capping a single yard as part of the project incurred total costs of less than USD $1000 and has been shown to be effective at reducing exposure (Laidlaw et al., 2016; Mielke et al., 2011).

Moreover, as has been observed elsewhere, ex situ methods also potentially carry risks associated with re mobilizing and removing contaminated soil (Kuppusamy et al., 2016; Wuana and Okieimen, 2011). In this project, Pure Earth identified this as a potential high risk. Factors contributing to this assessment included the likely spillage of material by local haulage trucks of varying quality that would be needed to transfer it through the narrow streets of the village to the waste facility more than 40 km away.

Other in situ options were also explored, including soil amendments and capping with an impermeable surface. Soil amendments can be blended with lead contaminated soils to reduce the percentage of bioavailable lead, primarily through decreasing its solubility. Phosphorus, bone char, and compost, among others, have all been shown to be effective with decreases of bioavailable lead of up to 66% in laboratory conditions and typically 50% or below in field tests (Brown et al., 2004; Chen et al., 2006; Farfel et al., 2005; Hettiarachchi et al., 2001). Given the high concentrations of lead in soil (average 3940 mg/kg at Dong Mai), amendments would have insufficiently reduced bioavailable lead in this case. Furthermore, soil amendments also carry risks including increased eutrophication and groundwater contamination (Kilgour et al., 2008; Wuana and Okieimen, 2011).

Capping with a more permanent covering of contaminated yards with concrete, tile, paver stones, or other impermeable surface was also explored. Based on concerns of possible perverse incentives, the project opted not to pay for these improvements. Specifically, it was considered possible that the prospect of a capital improvement could inadvertently encourage a homeowner to intentionally contaminate his or her own yard. Soil capping sidesteps this issue providing no capital improvement to earthen yards. In a limited number of cases, homeowners opted to invest in these capital improvements themselves.

This project placed a high value on developing a rapid response. By way of example an early visit resulted in 235 geo referenced surface samples. A map projecting these samples across the village was provided to the village leadership within two days of the site visit. Any blood lead levels taken within the context of the project were provided to the patient immediately by their local clinician. The duration of the first phase of the project from assessment to completion was 11 months. In addition to urgency, the project emphasized cross sectional representation of the community in decision making, accepting the public as a legitimate partner. The importance of this in promoting effective cleanup in lead contaminated environments has been observed previously by the US EPA (Covello and Allen, 1992).

A number of endogenous factors benefitted this project and contributed to the subsequent lowering of blood lead levels in the community. Among these is the fact that the Vietnamese government funded the construction of a new industrial zone before any project intervention work was initiated. In addition, the works undertaken by the Vietnam rural development program to cover every major road contributed to reducing the recirculation of contaminated road dust, which has been identified elsewhere as an the potential source of lead exposure (Zahran et al., 2013). A number of cultural and social factors including near universal literacy, which helped to augment the community education program, are also likely contributors to the lowering of blood lead levels. While these measures are difficult to formally quantify, they formed part of a multi faceted approach.

Collectively, the direct involvement and ownership of the problem and solution by community was a likely contributor to the broader socialization of project efforts. Similar outcomes in environmental projects have been identified elsewhere (Lester and Temple, 2006). In the case of Dong Mai, the community engagement and ownership is exampled by the road improvement program, self financed mitigation and the self relocation of the majority of the informal ULAB recycling activities out of the village. Forward momentum in maintaining and further lowering BLLs in the community is likely to be achieved by continued media attention (e.g. Vietnam Television, 2014) and ongoing government support.

Despite the effectiveness of the intervention, BLLs at Dong Mai remain elevated above acceptable international levels with the most recent median blood leads for children (< 6 years) at 13.3 μg/dL. The CDC recommends that public health action be initiated when BLLs exceed 5 μg/dL (Centers for Disease Control and Prevention, 2017). A significant limitation that remains unaddressed is the ongoing underutilization of the clean in, clean out changing room, which would limit exposure pathways from the processing plant back to residential environments. In order to increase its use, deeper and more committed engagement with the recyclers is required, particularly with respect to worker education and hygiene.

This project utilized the USEPA soil screening levels for the purpose of prioritizing yards for intervention. These have recently come under criticism for being insufficiently protective (Jackson, 2009). California for example, maintains a current screening level of 80 mg/kg for residential soils as risk assessments indicate that this level of soil lead equates to a increased BLL < 1 μg/dL, which in turn is < 1 point IQ decrement (Integrated Risk Assessment Branch Office of Environmental Health Hazard Assessment, 2009).

Given the chronic and severe nature of the exposure in the community preceding the intervention, it is likely that skeletal bone in Dong Mai residents will remain a significant reservoir of lead exposure, which poses life long risks to the community (Silbergeld, 1991; Silbergeld et al., 1992). The half life of lead in bone is estimated to be from 5 to 19 years, indicating that BLLs will likely remain elevated for some time (Rabinowitz, 1991). Future investigations could target children born after 2015 to better evaluate the efficacy of the intervention and lingering sources so as to target remediation to effect a long term reduction in adverse exposures.

5. Conclusion

Informal ULAB processing sites in LMICs can produce high environmental lead levels and significant human health exposures. The Dong Mai project resulted in significantly decreased environmental and childhood lead levels. This success resulted from a rapid, well coordinated response and a focus on a cost effective execution that could be replicated elsewhere.

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4.2. Paper Eleven

Heacock, M., Trottier, B., Adhikary, S., Asante, K. A., Basu, N., Brune, M. N., … **Ericson, B** ... & Chen, A. (2018). Prevention-intervention strategies to reduce exposure to e-waste. *Reviews on Environmental Health*, *33*(2), 219-228.

This study is an output of an electronic waste (e-waste) workshop organized by the US National Institute for Environmental Health Sciences in Jakarta, Indonesia in 2015. The purpose of this article is to provide general contextual information on e-waste recycling challenges in LMICs. The study details five separate interventions designed to mitigate exposures in four countries. Two of the interventions described contribute to this thesis and were managed by the author.

In the first example, the paper describes informal cable burning in Uruguay. The plastic sheathing of cables is commonly burned away by recyclers to recover the encased copper wire. Because the plastic sheathing contains high concentrations of lead, the burning results in high levels of lead contamination in area soils. The *ex situ* intervention (meaning excavation and disposal offsite) described was conducted jointly with the municipality of Montevideo. The second intervention described included the installation of cable strippers in Ghana to deal with a similar issue. The intention in the latter case provide a sustainable alternative to cable burning. In both cases, the efforts resulted in scaling up following the interventions. In Montevideo, the municipality began a citywide lead remediation program, while in Ghana largescale investment from the German aid agency GIZ contributed to the construction of a larger wire stripping facility.

The paper addresses the purpose of Chapter 4 in that it describes different interventions to mitigate lead exposure at e-waste sites. Because this Chapter is primarily comprised of different risk mitigation case studies, this study presents a valuable contribution by demonstrating a multiplicity of approaches. The intervention is described in detail and biological and environmental results of that intervention are presented.

Short Communication

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Prevention-intervention strategies to reduce exposure to e-waste

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Abstract: As one of the largest waste streams, electronic waste (e-waste) production continues to grow in response to global demand for consumer electronics. This waste is often shipped to developing countries where it is disassembled and recycled. In many cases, e-waste recycling activities are conducted in informal settings with very few controls or protections in place for workers. These activities involve exposure to hazardous substances such as cadmium, lead, and brominated flame retardants and are frequently performed by women and children. Although recycling practices and exposures vary by scale and geographic region, we present case studies of e-waste recycling scenarios and intervention approaches to reduce or prevent exposures to the hazardous substances in e-waste that may be broadly applicable to diverse situations. Drawing on parallels identified in these cases, we discuss the future prevention and intervention strategies that recognize the difficult economic realities of informal e-waste recycling.

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Keywords: electronic recycling; electronic waste (e-waste); environmental health; global health; intervention; prevention.

Introduction

Electronics are an increasingly large part of daily life, and millions of electronic devices are discarded every year in countries around the world. An estimated 65 million tons of electronic waste (e-waste) was created globally in 2017, with further increase projected in the years ahead (1). Due to the great expense of proper disassembly and disposal of electronics, e-waste is frequently shipped to developing countries (2).

In this commentary, we focus on informal e-waste recycling sites in Asia, South America and West Africa, where the work is often performed by women and children, with few occupational or environmental protections, and with little or no public health infrastructure (1, 2). We present case studies to illustrate the range of activities and conditions at these sites and the health hazards associated with them. In addition, we describe the intervention approaches that may be broadly applicable to diverse scenarios. We also discuss the future prevention and intervention strategies while recognizing the difficult realities of the informal e-waste recycling economy.

This commentary arose out of discussions held at a workshop on exposure to e-waste convened jointly by the U.S. National Institute of Environmental Health Sciences (NIEHS) and the World Health Organization (WHO), in collaboration with the Chulabhorn Research Institute of Thailand, the Children's Health and Environment Program (The University of Queensland), and Pure Earth. The workshop received financial support from NIEHS. It was held immediately following the 16th Annual Conference of the Pacific Basin Consortium on August 14, 2015, in Depok, West Java, Indonesia.

E-waste and informal recycling

In developing countries, e-waste is predominantly recycled informally in rural communities, in urban or non-urban neighborhoods, and in small family workshops rather than at dedicated facilities (3, 4). Informal recycling sites can range from small, microscale operations in homes or neighborhoods, to sprawling sites as large as entire towns. Informal recycling often uses uncontrolled methods and employs practices that can produce byproducts with considerable negative impacts on the environment and human health. Although recycling practices and exposures vary by geographic region, e-waste workers often do not wear personal protective equipment, and may be engaged in similar activities to dismantle and recycle electronics (5). These activities involve extracting the valuable components, such as gold, copper and silver, from electronic products, including cell phones, computers, DVD players, game stations, televisions, refrigerators and washing machines (5).

Economic considerations

E-waste contains not only hazardous substances, but also valuable materials such as copper, palladium and gold, which are driving the recycling process. Extracting these commodities provides a much-needed living for people in developing countries with limited alternative sources of income. Recycling and other informal activities represent the largest source of financial support for many economically disadvantaged families. For example, in West Africa, workers can make between $16 and $52 USD per 10–12 h workday, far higher than the national values (6). Unfortunately, these informal recycling entrepreneurs often endanger their own health, the health of their families and of people in their communities in their quest for a livelihood, pointing to the need for interventions that reduce health risks while recognizing these economic realities (3, 4).

Potential hazards

E-waste contains a mixture of hazardous substances released during the recycling process. These include metals (e.g. lead, mercury, cadmium); brominated flame retardants; and chemicals found in plastics (e.g. phthalates). When the materials are burned during recycling, toxic and carcinogenic substances are produced and released (e.g. dioxins, furans and polycyclic aromatic hydrocarbons) (7).

As a result, significantly elevated levels of such contaminants can be found in soil, road dust, air and water, in residential, school and park areas near recycling sites (8, 9).

Increased levels of some of these contaminants have been measured in the blood of exposed workers in the informal e-waste recycling industry and in children living in nearby contaminated areas. Exposure to these contaminants are associated with adverse health effects. For example, a systematic review (10) pointed to associations between exposure to polybrominated diphenyl ethers (PBDEs) in e-waste and alterations in thyroid function and higher levels of thyroid stimulating hormone leading to hypothyroidism. In addition, children whose mothers were exposed to higher levels of perfluorooctanoic acid showed increased risk of slowed neonatal physical development and adverse birth outcomes such as premature delivery, low birth weight and stillbirth compared to children whose mothers were not exposed (11).

Lead exposure is also a significant concern in nearly all informal recycling areas, which in some cases comprise entire towns. Studies have linked lead and other heavy metal exposures in children in e-waste recycling areas to attention-deficit/hyperactivity disorder and other neurodevelopmental disorders (12, 13). Children represent a population uniquely vulnerable to the exposure of environmental chemicals. They breathe more air and consume more food than adults per surface area of the respiratory tract and pound body weight. They are still growing and developing, and at certain stages of development, exposure to environmental chemicals can lead to irreversible damage (14). This together with their frequent hand-to-mouth behaviors, can increase their exposures. As children often work directly in informal recycling operations, they may be vulnerable to long-term adverse health effects resulting from exposures to toxicants in e-waste released from its recycling processes. These are just a few examples of documented negative health effects linked to e-waste recycling. A major concern is that the full scope of the problem is not well characterized, as workers in the informal sector are not screened or monitored for blood lead levels or other toxic exposures.

E-waste case studies

Although family and informal e-waste recycling practices and exposures vary by geographic region and scale of operations, our examination of case studies from several different countries show some parallels that may be useful to consider for sites that are not well characterized. While solutions to reduce exposure and protect human health must be locally tailored, we can learn valuable lessons from work that has been done to reduce exposures and protect health in the case studies presented.

Uruguay

Description of site

Informal e-waste recycling sites in Uruguay are largely located in Montevideo, where they are scattered throughout suburban residential neighborhoods, particularly in those with higher social and economic vulnerability. It is estimated that there are more than 550 such urban settlements with more than 165,000 inhabitants, although not all settlements are involved in e-waste activities (15). Typically workers will dismantle electronic products manually and burn cables to extract copper, without any proper personal protective equipment (16). These microscale recycling activities often occur near homes and where children often play, and children participate in these activities by gathering metals.

Exposure information

As noted, participating in recycling activities and living and playing around recycling-contaminated sites increase children's exposure to lead (16). Elevated blood lead levels have been measured in children exposed to lead through the burning of cables in or around the home, through soil, or through lead-based paint. In one study in Uruguay, even though some activities, such as gathering metals, were not associated with increased blood lead levels, the average blood levels among the children at the first consultation were substantially higher (mean 9.19 μg/dL) than the U.S. Centers for Disease Control and Prevention's (CDC) current reference level of 5 μg/dL, suggesting the need for primary prevention (16, 17). The highest lead levels were seen in the youngest children (16).

Intervention approaches

In response to measured elevated blood lead levels among children living in an e-waste recycling area in Uruguay, researchers implemented several intervention approaches. These included family education, home visits and outreach and communication with community members. In addition, the non-profit organization Pure Earth conducted indoor and outdoor remediation, such as excavating and replacing contaminated soil, to reduce exposure (16, 18). As a result of the various intervention approaches, blood lead levels were found to be decreased by a mean of 6.96 μg/dL (16, 18). These reductions were paralleled by decreases in lead measured in soil after remediation. The researchers suggest that educational interventions for families that focus on environmental hygiene and nutrition may be useful to reduce children's exposures as part of a multi-pronged approach, though direct evidence is limited (16).

Ghana

Description of site

Agbogbloshie, centrally located in the capital city of Accra and home to about 40,000 people, is one of the largest and best-studied e-waste sites on the African continent. The waste is processed in Agbogbloshie by recyclers working out of small sheds and out in the open, scattered among residences and Accra's largest food market (6). Common e-waste recycling practices at Agbogbloshie include scavenging for electronics, manual dismantling of electronic equipment and open burning to isolate valuable metals (3, 4, 6). Non-valuable materials are dumped out in the open (6). In most cases, workers at these informal facilities do not use any personal protective equipment (PPE) (6).

Exposure information

Plumes of smoke from this site can be seen from afar due to open burning of cables during recycling. Not surprisingly, the main environmental exposure has been estimated to be from the burning process, although contaminated food and soil are also of concern (3, 4, 6). Informal e-waste workers who were studied at this site were found to have significantly higher concentrations of blood lead compared to a control population that lived in a suburb of Accra not involved in e-waste processing (19). Studies have also shown higher blood levels of polychlorinated dibenzo-p-dioxins and dibenzofurans, which are produced during the burning process, among workers (6). Elevated levels of polycyclic aromatic hydrocarbon (PAH) metabolites, lead, nickel, arsenic and cobalt have also been measured in the urine of e-waste workers compared to controls (6).

Intervention approaches

A model intervention implemented by Pure Earth in Ghana has had some success. A new e-waste recycling center is helping to reduce toxic exposures by providing electric-powered, automated wire-stripping machines (20). In the initial stages, the machines provided were not well suited to the small wires and cables being dismantled. These machines were later replaced with ones that were more practical for the workers who used them (20). In the most recent stage, mechanized equipment to handle larger devices (e.g. motors, capacitors, rotors) were added, and workers are being trained on their use.

Pure Earth incorporated community feedback, that led to new machines that worked better for the recyclers, highlighting the importance of engaging stakeholder needs in the intervention process. Although burning and other unsafe practices have not been eliminated, there is more community support for the project as the tools and technologies more closely align with their needs. These tools are now providing an alternative to open burning and are offering greater safety to workers (20).

China

Description of site

Until recently, Guiyu, a town in Shantou, China, was one of the largest e-waste recycling and dismantling communities in the world, with an estimated 1.7 million tons dismantled there annually (11, 21). In 2015, more than 6000 small, family-run workshops were reported to be participating in e-waste dismantling and recycling activities (8). In 2014, researchers observed that 80 percent of families in Guiyu were engaged with individual recycling workshops, nearly 160,000 workers. Recycling activities were scattered throughout many villages and communities in Guiyu.

Common practices include baking printed circuit boards, soaking parts in acid baths, open burning to extract metals and manually stripping plastic materials from electronic products and crudely classifying them (e.g. sorting by burning smell) (22–24). Researchers have documented workers wearing no protective equipment while participating in these activities.

Exposure information

Due to the large amount of open burning, contaminated air is a large contributor to environmental exposures. Levels of fine particulate matter (PM2.5), cadmium and lead in the ambient air were found to be much higher in Guiyu than in a reference area (9).

Children living near an e-waste recycling area in Guiyu have been shown to have significantly higher blood lead levels (22, 25). Elevated levels of other metals such as cadmium and mercury have been reported (12, 26, 27), as have increased levels of polybrominated diphenyl ethers (PBDEs), PAHs, polychlorinated biphenyls (PCBs), perfluorooctanoic acid, phthalate esters, and bisphenol A in blood, urine and other samples (21, 23, 26, 28–31).

Intervention approaches

Guiyu has seen major, rapid changes in its e-waste recycling practices following a December 2015 decree from the Chinese government that required all informal e-waste recycling in residences to shut down and move to a new industrial park where protective measures are in place (32, 33). In addition, new domestic and industrial sewage treatment plants were constructed by the government. These approaches combined with a series of educational outreaches on topics including heavy metal detection, health risk assessment and medical services have contributed to a reduction in blood lead levels (24, 34).

India

Description of site

While there is a great deal of information on larger, formal registered e-waste dismantlers and recyclers in India (largely concentrated in the southern state of Karnataka [52 facilities], Maharashtra [22 facilities] and Haryana [13 facilities]), like other countries, the scale of informal recycling activities are not well documented (35). This is concerning as it is estimated that more than 95 percent of e-waste ends up in the informal sector (35). As one example, many such operations exist in and around Delhi, including Seelampur, the largest subdivision of the North-East District of Delhi (36). Over 30,000 people participate in e-waste recycling in Seelampur (37), which is known as the largest scrap market in the country. Typical activities include manual dismantling of electronics, use of acid baths, baking circuit boards and burning wires (38). Workers, many of whom are children, are often not aware of the dangers of the chemicals and acids they handle without protective gloves and breathe without protective masks (38).

Exposure information

Similar to other countries, recycling activities in India release toxic fumes and contaminate water when e-waste is dumped into streams. In addition, metals and other contaminants have been measured at elevated levels in soils and sediments (39).

High levels of blood lead and urinary chromium have been found in workers from the informal e-waste sector in Delhi, India (5). For lead, values ranged from 8 to 58 μg/dL; these values are well above the CDC's current reference level of 5 μg/dL (5).

Intervention approaches

The Centre for Occupational Health at New Delhi is working with the University of Cincinnati to initiate a major project in India to study health outcomes of e-waste recycling (5). In addition, the Indian government proposed laws in 2011 that were later expanded upon in 2016 to regulate e-waste management and trade (35). The more recent regulations are more broad and cover a wider range of materials and industrial stakeholders (35). A major remaining challenge is the large number of informal workers who, unlike larger companies, are not covered by these rules (38), and the rules do not incorporate health and safety measures to protect workers and the environment (35). A key need in India is regulations that protect workers' interests, particularly those of vulnerable populations and children, and that cover the large informal recycling industry (5).

Philippines

Description of site

E-waste recycling in Manila, Philippines, is scattered in many different communities throughout the metropolitan area and surrounding suburban areas. Thousands of self-organized recyclers carry out microscale recycling activities in front of homes, on the streets, in backyards or along the river in this densely populated area (40). These recyclers are connected to more than 2000 junkshops that collect recycled materials (40). Recycling activities include manual dismantling and crushing, burning power cords and heating circuit boards. Recyclers often work with bare hands, wear flip-flops and do not wear personal protective equipment (40).

Exposure information

Limited research in informal e-waste recycling sites in metro Manila suggests increased levels of cadmium, cobalt, copper, manganese, nickel, lead and zinc in soil samples, similar to other large recycling sites in Asia (41–44). In addition, open burning of e-waste led to increased PAH exposure in soil samples (44). While not directly tested at this site, PAH levels were likely high in the air similar to what has been tested at other Asian sites (41–44). Very few studies have measured chemical exposures in recycling communities in the Philippines. In one of the few such studies, women living near a metro Manila e-waste dumping site presented slightly higher PBDE concentrations in breast milk compared with a control site (45).

Intervention approaches

In Manila, researchers found that going door-to door and interacting on an individual level with the residents was the best way to gain the trust of the community and assess their needs. Through these personalized interactions, they found that most workers had little understanding of the potential health risks associated with e-waste recycling and the particular vulnerability of children and pregnant women (40). The community perceived e-waste dismantling as an easy source of income, with their major health concern focused on limited access to health care (40). The needs assessment also showed that even health center physicians in Manila were unaware of the e-waste dismantling occurring in their community and its health hazards (40).

A pilot outreach intervention followed a risk-reduction approach focusing on the decreasing of exposures, community organizing and development and access to quality health care. Advocacy and sensitizing activities cut across each of the components. Outreach activities included using posters to educate workers about the need for protections, distributing protective equipment, providing tours of formal recycling facilities where protections are used and educating local healthcare workers about the health effects from exposure to e-waste recycling (40). In educating workers and their families, the goal was to reduce risks to health while providing a message easily understood by the audience. A needs assessment found that many of the workers were young boys, so this goal was accomplished by using a graphic cartoon featuring a character named “E-boy” to demonstrate safe recycling practices (40).

Emerging themes and needs: the way forward

The case studies presented here illustrate how local conditions and context for e-waste recycling can vary widely. Solutions to reduce exposure and protect human health must be locally tailored and take into consideration the large differences in the scale of recycling sites, which range from vast facilities to tiny family operations. Acknowledging these differences, we identified several overarching themes and common needs from these studies and experiences.

Economic considerations

E-waste recycling work is often conducted by informal workers who are focused on the urgent need to provide for their families, not the long-term health effects from exposure to e-waste. Thus, interventions to reduce the health threat of e-waste must recognize that informal e-waste recycling provides a living for many people with limited sources of income (46). While preventing children and pregnant women from working in informal e-waste recycling is a priority (4), banning all workers from participating in the practice is currently not a viable option because of this need for a livelihood (46).

It is critical to make the economic case for improvements in practices, conditions and preventive measures as economic incentives are strong motivators to encourage adoption of safer methods and technologies (34). Approaches may include discussing the economic consequences of exposures in light of disease burden outcomes and proving through business cases that profitability can increase with newer technologies that maximize recovery and minimize exposure (34).

Culturally appropriate communication

In any intervention, messages that pertain to e-waste recycling should be tailored to communities or regions based on insights gained from listening to the group’s concerns. This process is necessary to help gain the community’s trust and to learn about their needs. When community members feel that they are heard and understood, there is less room for miscommunication or mistrust of outsiders. Building relationships with the community and listening to their concerns is vital to the success of any community intervention or prevention initiative.

Stakeholders also have a large role to play in evaluating interventions. Iterative and multidirectional approaches are important, as stakeholders provide feedback to help determine which technological solutions are best suited to local cultural needs and expectations (40, 47). It is important to include community perspectives and involve a wide variety of stakeholders such as healthcare providers, local authorities, regulatory agencies and site community organizers (5, 40). Doing this in multiple stages of the intervention not only helps refine the approaches and tools to be more relevant for the target audience, but it may also improve buy-in from the community and promote continued success.

Cultural considerations, as well as those of age and gender, should be taken into account in outreach approaches (e.g. word choice, media and graphical messaging). While no single message will be appropriate or successful in all contexts, key factors for effective messaging include simplicity and accuracy (48). Communication tools that may be useful include posters, brochures, radio messages, presentations, videos and social media, either alone or combined (48).

Better exposure measurement

One of the first steps to understanding the potential impacts and designing intervention approaches for communities engaged in e-waste recycling is to quantify exposure levels and dominant exposure routes. While there are some commonalities between sites, the case studies illustrate that exposure can vary depending on the materials being recycled and the specific methods employed. Given the health effects observed from these exposures, environmental, biological, occupational and health monitoring is therefore important (34). Measuring environmental indicators of contamination left behind and collecting and archiving environmental samples should be a priority. Levels of contamination in soil, water and air should be measured. Monitoring should include PM2.5, metals, persistent organic pollutants (POPs) and PAHs.

This information can inform the scope of personal exposure monitoring in workers and residents, which carries more ethical concerns and is more expensive than measuring environmental indicators. For instance, if environmental monitoring does not detect POPs in soil, water or air, there may be no need to monitor their presence in the local population (34). Evaluating intervention approaches requires both baseline and post-intervention exposure data (48). These evaluations must include population monitoring in addition to environmental samples, to truly determine whether a prevention or intervention initiative has been successful (34). As baseline samples are not available in many cases, it is critical to collect samples before interventions begin. Technologies to more accurately measure personal exposures and population exposures will be needed to accomplish this (48).

Linking exposure to health outcomes is challenging, and in most areas affected by e-waste, publicly collected population data are not available. Thus, it will be important to monitor at the local level and identify and follow a set of defined health and exposure measures. Following trends over time will also be important to better understand the link between exposures and health outcomes (48).

Reducing exposures

There is a need to pilot test new technologies and approaches to reduce exposure (34). While these solutions must be locally tailored, technological and non-technological approaches, such as engineering controls, remediation tools and education, are critically important to decrease direct and indirect exposure.

Improvements in technologies to reduce exposures are necessary to allow clean-up of existing sites and establishment of better recycling practices. In places where e-waste recycling is performed informally by individuals in public spaces and in homes, remediation of contaminated sites is necessary to prevent additional exposure (4, 16, 49).

Education is also a priority. There is a need to highlight the importance of PPE availability and to train workers on its use (34). Likewise, there is a need to improve health education for medical doctors and nurses who work at the community level. Health education programs should include both community workers and traditional healers, who are the front-line health professionals in many areas (34).

Similarly, educational programs about e-waste exposures should be appropriate for and promoted among the most vulnerable populations, including children, pregnant women and workers (34).

Regulatory and policy considerations

Regional and national regulations regarding e-waste management must be reviewed and updated, including those mandating the use of PPE in the formal and informal e-waste recycling sectors. These measures will require strengthened interactions between policy makers and the business sector (34). Educating and protecting workers will require other stakeholders from many different arenas to work together to develop multi-sectoral e-waste regulations and policies that address environmental, economic, social and health aspects of e-waste recycling (4).

Conclusion

One of the key challenges of prevention and intervention studies is addressing the disconnect between the long-term risk from exposure to contaminants because of e-waste recycling activities, and the immediate, acute economic needs of the communities involved in these practices. E-waste recycling is often conducted by informal workers, who are more concerned about feeding their families than preventing later-life health effects from exposure to e-waste. These difficult realities must help inform how intervention and prevention approaches are designed and presented to communities. Their input can help researchers determine appropriate messaging that will resonate with their intended audience and help make the case for economic benefits that can be tied to improvements in practices, conditions and preventive measures, as well as benefits to human health.

Another key challenge is the fact that the e-waste problem has been growing for decades. Even after primary exposure is reduced or mitigated, the legacy of contamination will remain. Some of the hazardous substances in e-waste are persistent in the environment and can bioaccumulate or biomagnify in plants and animals. This means long after primary exposure has been reduced or removed, people in the community can continue to be impacted by chemicals that remain in their soil, water and food sources. Methods that remove these legacy sources of exposure, such as excavating contaminated soil, may be necessary and appropriate in some cases. There is still a great need for more research on effective remediation technologies to protect people from legacy exposures.

As e-waste recycling can be beneficial and sustainable, for recovering valuable resources, it is important to develop and incentivize business models that encourage safe, sustainable and efficient recycling practices. For example, by facilitating research and interactions among stakeholders in academic institutions, industry, governments and international organizations, the Solving the E-waste Problem (StEP) Initiative has been a leader in global management and development of environmentally, economically and ethically sound e-waste recovery, re-use and prevention (50).

As informal e-waste practices and interventions continue to evolve in various regions, continued collaboration and exchange of ideas among the various stakeholders will be vital to sustained progress toward making e-waste recycling a safer way to make a living. Continued and expanded research is needed, including improved design of electronics, safer extraction practices and advancements in remediation technologies. Developing an open-access catalogue of current e-waste research and resources describing state-of-the-art best practices that are affordable, usable and realistic for different recycling operations is also necessary to improve intervention and prevention approaches.

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4.3. Paper Twelve

Ericson, B., Caravanos, J., Depratt, C., Santos, C., Cabral, M. G., Fuller, R., & Taylor, M.P. (2018). Cost Effectiveness of Environmental Lead Risk Mitigation in Low‐ and Middle‐ Income Countries. *GeoHealth*, 2(2), 87-101.

This study developed a model to calculate the cost effectiveness of a risk mitigation project executed in the Dominican Republic. Environmental data (soil concentrations) and dust lead loadings were utilized to estimate the biological lead burden in the surrounding community and to calculate attributable DALYs. The benefits of the intervention were characterized by calculating the cost per DALY-averted calculated against the expenses incurred during the project, enabling comparison with other public health interventions.

This study is placed in Chapter 4 because it describes an intervention, even though it also, by necessity, involves aspects of exposure (Chapter 2), as it utilizes a sensitive and robust exposure assessment, and outcomes (Chapter 3), as DALYs are a central aspect of the study's method. Consequently, this final study captures the major aspects of each chapter into a single closing argument; that lead-exposure in LMICs presents a severe public health risk that can be mitigated in a cost-effective manner.

AGU PUBLICATIONS

GeoHealth

RESEARCH ARTICLE
10.1002/2017GH000109

Key Points:
- Pollution remediation in low and middle income countries has yet to be evaluated for its cost effectiveness
- We calculate DALYs averted by the lead remediation in Paraiso de Dios, Haina, the Dominican Republic completed in 2010
- Pollution remediation is cost effective according to WHO thresholds

Supporting Information:
- Supporting Information S1

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Cost Effectiveness of Environmental Lead Risk Mitigation in Low- and Middle-Income Countries

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Abstract Environmental remediation efforts in low- and middle-income countries have yet to be evaluated for their cost effectiveness. To address this gap we calculate a cost per Disability Adjusted Life Year (DALY) averted following the environmental remediation of the former lead smelter and adjoining residential areas in Paraiso de Dios, Haina, the Dominican Republic, executed from 2009 to 2010. The remediation had the effect of lowering surface soil lead concentrations to below 100 mg/kg and measured geometric mean blood lead levels (BLLs) from 20.6 μg/dL to 5.34 ug/dL. Because BLLs for the entire impacted population were not available, we use environmental data to calculate the resulting disease burden. We find that before the intervention 176 people were exposed to elevated environmental lead levels at Paraiso de Dios resulting in mean BLLs of 24.97 (95% CI: 24.45–25.5) in children (0–7 years old) and 13.98 μg/dL (95% CI: 13.03–15) in adults. We calculate that without the intervention these exposures would have resulted in 133 to 1,096 DALYs and that all of these were averted at a cost of USD 392 to 3,238, depending on assumptions made. We use a societal perspective, meaning that we include all costs regardless of by whom they were incurred and estimate costs in 2009 USD. Lead remediation in low- and middle-income countries is cost effective according to World Health Organization thresholds. Further research is required to compare the approach detailed here with other public health interventions.

Plain Language Summary We review the cost effectiveness of the remediation of a lead contaminated site in the Dominican Republic that posed a health risk to the surrounding community. We find that the project reduced a significant health burden for an acceptable cost according to thresholds established by the World Health Organization. Pollution poses a credible health risk to a large number of people; thus, it is important to identify cost effective methods of dealing with the worst sites.

1. Introduction

Cost effectiveness analysis is a potentially informative tool in the debate on resource allocation (Murray et al., 2000). In the case of public health interventions the Disability Adjusted Life Year (DALY) enables comparison between different health outcomes in terms of morbidity and mortality (Murray & Lopez, 2013). The DALY approach is a robust and globally accepted method used by the World Health Organization (WHO) and the Institute for Health Metrics Evaluation (IHME) for estimating the Global Burden of Disease (GBD) from an extensive list of health outcomes (Forouzanfar et al., 2016; World Health Organization (WHO), 2013). A DALY is the sum of two metrics: a year of life lost (YLL) and a year lost to disability (YLD). The former captures years lost due to premature death, while the latter captures the relative severity and duration of various adverse health outcomes (World Health Organization, 2016).

GBD estimates consider a multitude of risk factors and health outcomes. Several researchers, for example, have reviewed the DALY contribution of health risks such as smoking or urban air quality (Cohen et al., 2005; Zaher et al., 2004). Others, notably Prüss-Ustün et al. (2016, 2011) have calculated DALYs resulting from chemical exposures, including pesticides and naturally occurring arsenic in groundwater. An important gap in the literature, however, is the contribution of hazardous waste sites to the burden of disease, due in part to a lack of global information on their occurrence and related health risks.

Chatham-Stephens et al. (2013) examined this knowledge gap by utilizing data collected as part of Pure Earth's Toxic Sites Identification Program (TSIP). Pure Earth maintains a database of contaminated sites with estimates of exposed populations and the results of environmental sampling and analysis, among other parameters (Ericson et al., 2013). In reviewing TSIP data on hazardous waste sites in India, Indonesia, and the Philippines, the authors found that the burden of disease from toxic contaminant exposures in these countries was comparable to other significant environmental health risks, such as malaria (Chatham-Stephens et al., 2013).

In high-income countries, a number of studies have employed cost effectiveness analysis in evaluating pollution remediation projects. For example, Hamilton and Viscusi (1997) reviewed data from 150 Superfund sites in the USA to determine the cost per cancer averted. In the United Kingdom, a comparable effort evaluated the cost per life year gained from radon remediation efforts (Kennedy et al., 1999). In low- and middle-income countries (LMICs), a limited set of papers have reviewed health benefits of pollution remediation (Ericson et al., 2018; Jones et al., 2013; Ludlow & Roux, 2012; Tirima et al., 2016). However, evaluations of the cost effectiveness of these sorts of interventions in LMICs are, to the best of our knowledge, nonexistent.

This paper endeavors to begin to fill this gap by calculating the generalized cost effectiveness of remediating lead contaminated soil in the Dominican Republic community of Paraiso de Dios, Haina. The attributable disease burden from the site is modeled, and the cost effectiveness of the intervention is evaluated from a societal perspective. The purpose of this effort is to inform the discussion on resource allocation.

The source of contamination was a poorly managed secondary lead smelter that operated until the late 1990s, when high blood lead levels (BLLs) were documented in the surrounding community and the smelter was forced to close (Kaul et al., 1999; Kaul & Mukerjee, 1999; Wilson, 2002). Studies conducted before and after the closure found considerable decreases in the BLLs of area children. Mean BLLs declined from 72 μg/dL ($n = 116$) before the closure to 32 μg/dL ($n = 146$) 6 months after, indicating that the most significant exposures were associated with the smelter's operation (Kaul et al., 1999). The closure included a rudimentary repository for high-level material that failed shortly thereafter. Following the closure, the site was characterized by uncontrolled piles of battery waste and heavily lead-laden material.

The Haina soil lead intervention project, which was designed and overseen by TerraGraphics, Inc., was carried out in two distinct phases: the removal or capping of onsite contaminated soil in December 2009 and the mitigation of offsite exposures through soil removal and construction in August 2010 (Blacksmith Institute, 2010). Soil abatement has been shown to effectively mitigate exposures at lead contaminated sites (Ericson et al., 2018; Lanphear et al., 2003). Community education efforts were carried out in parallel. The site now serves as a city park.

A number of environmental assessments were carried out before and after the intervention. Hunter College and Pure Earth assessed concentrations of lead in surface soils and house dust in 2007. Samples were collected onsite and analyzed in New York, USA, using atomic absorption spectrometry (Caravanos, Fuller, & Nieves, 2007). This study reported median lead in soil concentrations of 55,420 mg/kg (IQR: 17,960–305,045) onsite and 11,225 mg/kg (IQR: 3,185–27,335) in residential yards based on 12 samples from each area (Caravanos et al., 2007). Median values were reported due to the log normal distribution of the data. Mean values for industrial and residential areas were 65,735 mg/kg (95%CI: 25,986–166,285) and 17,648 mg/kg (95%CI: 5436–29,859), respectively (Caravanos et al., 2007).

Caravanos et al. (2007) also collected dust samples in both Paraiso de Dios and, for the purpose of a control, the city center of Haina. The mean dust loadings inside of 13 sampled homes in Haina was 64.4 $\mu g/ft^2$ (95%CI: 32–97) and 9.7 $\mu g/ft^2$ (95%CI: 3.4–16.1) inside of 12 sampled homes in the city center. For context, the relevant U.S. Environmental Protection Agency (USEPA) reference levels for residential and industrial soils are 400 and 1,200 mg/kg, respectively (U.S. Environmental Protection Agency (USEPA), 1998). The relevant USEPA reference level for household dust is 40 $\mu g/ft^2$ (USEPA, 1998) (USEPA guidance is provided in US customary units. A square foot can be converted to a square meter by multiplying by 0.092903.). The Dominican Republic has not yet developed its own reference levels.

Pure Earth and TerraGraphics, Inc. collected 152 additional in situ surface soil lead measurements in 2010 between the onsite and offsite interventions. Analysis was conducted with a portable InnovX Delta series X-Ray fluorescence instrument (pXRF) with a lower detection limit for lead of 5 mg/kg (InnovX, 2016).

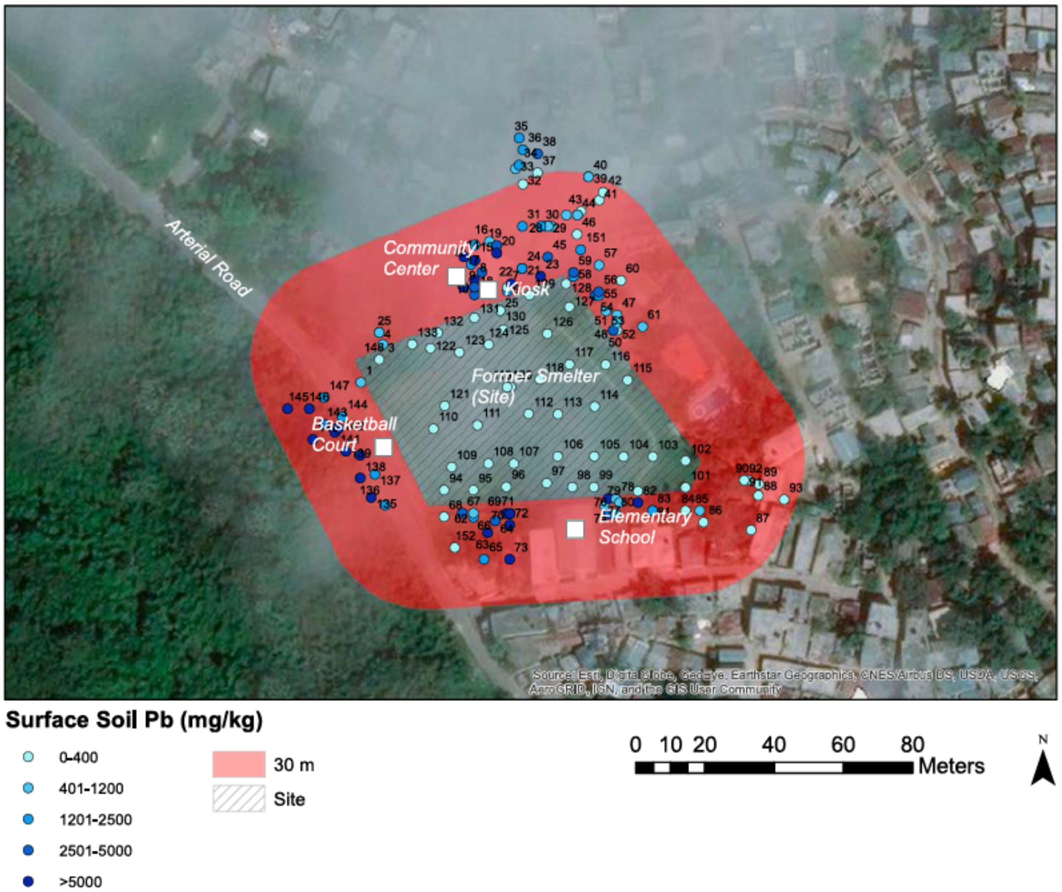

Figure 1. Overhead map of the former site of the MetaloXsa Smelter (Paraiso de Dios, Haina, Dominican Republic) and exposure scenarios used in this analysis. Soil lead concentrations displayed here were analyzed after the onsite remediation and before the offsite work.

Thirty-eight of the 152 measurements were taken from the former site having a mean lead concentration of 17 mg/kg (95%CI: 4–30), reflecting the effectiveness of the first phase of the intervention. One hundred and three measurements were taken within 30 m the site having a mean lead concentration of 4,410 mg/kg (95% CI: 3,175–5,645). These results are projected in Figure 1 and attached in the supporting information. Analysis of the comparability of in situ versus ex situ pXRF measurements using an Olympus InnovX Delta was carried out by Rouillon et al. (2017), who showed that in-field sampling was suitably robust (Rouillon et al., 2017).

2. Materials and Methods

2.1. Summary of Approach

To determine the cost per DALY averted as a result of the Haina intervention, we construct a number of models of the attributable disease burden. Key parameters of each model include: total exposed population; behavior of the population; soil lead content; soil ingestion rate; and lifetime of the project. The totals of all groups are summed to determine the total DALYs-averted.

2.2. Sources of Data for DALY Calculations

We utilize environmental data collected by TerraGraphics Inc., Pure Earth and Hunter College (Blacksmith Institute, 2010; Caravanos et al., 2007). Blood lead data are used to assess the effectiveness of the intervention and to guide the estimation of DALYs. Blood lead assessment in the population was undertaken by Pure Earth, the Autonomous University of Santo Domingo, and the Mount Sinai School of Medicine. Financial data are used to assess the overall cost effectiveness of the intervention and were acquired from a range of sources including government and multilateral websites, as detailed below. Where financial data could not be acquired, estimates are presented.

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Table 1
Blood Lead Tests Stratified by Year and Age Group

Age range blood lead tested in years	9 May	10 May	10 Dec	14 Feb
0 6	20	9	6	8
7 10	25	14	14	21
11 15	16	17	14	22
>15	17	11	10	31
Not recorded	1	1	1	
Total	79	52	45	82

2.3. Blood Lead Assessment

Blood lead levels (BLLs) were assessed on three instances between 2009 and 2010 by Pure Earth and the Autonomous University of Santo Domingo (UASD). Community BLLs were again assessed by Pure Earth in 2014 with the Mount Sinai School of Medicine (New York, USA). Samples were extracted and analyzed by physicians in a manner consistent with Institutional Review Board ethical guidelines using Magellan Diagnostics LeadCare I (May 2009) and Leadcare II (May 2010, December 2010, and February 2014) analyzers. The LeadCare II instrument, which is used widely for lead screening, has lower and upper detection limits of 3.3 μg/dL and 65 μg/dL (Magellan Diagnostics, 2015). Its predecessor, the Leadcare I had a lower detection limit of 1.4 μg/dL and the same upper detection limit of 65 μg/dL (ESA Biosciences, 1997).

Blood samples were collected and analyzed for 79 residents in May 2009, 52 in May 2010, 45 in December 2010, and 82 in February 2014. Participant residential addresses were not collected. Age information was collected in 2009 and 2014 only and was derived from this information for 2010 sampling events. Ages could be deduced for all but one sampled resident in 2009 and both 2010 sampling events. Age distributions of sampled residents are given in Table 1.

Twenty-five residents had their blood analyzed in both 2009 and 2014. In 2009, 4 were aged 0–6 years, 11 were aged 7–10 years, 5 were aged 11–14 years, and 5 were aged 15 years and above. The median age in 2009 was 10, while the median age in 2014 was 15.

2.4. Blood Lead Modeling

Blood lead data for the impacted population were required to calculate the attributable disease burden. However, because BLLs were only available for a subset of the population, we calculate BLLs using the Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK) and Adult Lead Methodology (ALM) (US EPA, OSWER, 2016). We model exposure scenarios for eight different groups of people. For all groups we determined the age and sex of the exposed population by applying unpublished national level age and sex distribution information from the latest IHME GBD study (Forouzanfar et al., 2016). The IHME GBD tables provide estimates of the relative percentage of the population across 20 different age groups and two sex categories, with more granularity at younger ages than could be identified elsewhere. The tables present estimates for five age groups under 10 years in comparison with two presented by the Dominican national census or the U.S. Census for foreign populations, for instance (National Statistics Office, 2012; US Census Bureau, 2016). All reviewed data included only one age group for ages 5–9 years. Because ages 5–7 years are of particular importance to this study, we estimated the number of 5–7 year olds by multiplying the 5–9 year old population by 0.6. We assume that the demographic composition of the site is identical to the national composition.

The relative severity and duration of each group's exposure is determined by their use of three distinct distal areas: the site, the site perimeter (defined as within 30 m from the border of the site), and offsite (defined as any area beyond the 30 m perimeter). For each distal area we assign uniform soil and dust lead concentrations. With regard to soil contamination in the onsite area, we use the median surface soil concentration from Caravanos et al. (2007) of 55,240 mg/kg. The small sample size and high variability of concentrations indicate that the mean (65,735 mg/kg) would not likely be an appropriate metric. For the perimeter we use the mean soil concentration of 4,410 mg/kg from the 103 measurements taken during 2010 Pure Earth/ TerraGraphics, Inc. assessment.

There is a dearth of available data on soil lead concentrations in Haina or the Dominican Republic more broadly beyond the site. Thus, to estimate a soil concentration offsite, we use house dust loadings from Caravanos et al. (2007) and convert these values to likely soil lead concentrations using the default IEUBK soil-to-dust coefficient of 0.7 finding a mean concentration of 323 mg/kg (OSWER, 1994). This value is similar to average soil lead concentrations in a number of cities globally (Ajmone-Marsan & Biasioli, 2010).

To calculate dust concentrations onsite we use the default IEUBK soil-to-dust coefficient of 0.7 finding a median concentration of 38,794 μg/g (OSWER, 1994). To determine the dust lead concentrations in homes in the perimeter, we adjust default IEUBK values downward to more accurately represent likely exposures.

Table 2
Number of Receptors, Relevant Soil Concentrations (Pb) and Durations of Exposure at Paraiso de Dios Before the Intervention

Group	Composition	Pop.	Hours onsite/week (work or play)	Hours in perimeter/week (home or school)	Hours offsite/week (work or other)	Time weighted dust (μg/g)	Time weighted soil (mg/kg)
			(soil = 55,420 mg/kg; dust = 38,794 μg/g)	(soil = 4,410 mg/kg; dust = 780 μg/g)	(soil = 323 mg/kg; dust = 226 μg/g)		
1	Children age 5 7 years living in perimeter	13	10	102	0	4,174	8,964
2	Children age 0 4 years living in perimeter	8	0	84	0	780	4,410
3	Children age 5 7 living offsite, attending school in perimeter	52	10	30	72	3,818	6,337
4	Children 0 7 living and attending school offsite (control)	74	0	0	112	226	323
5	Adults age > 14 years living in perimeter, working onsite	10	10	72	30	4,026	7,869
6	Adults age > 14 years living in perimeter and working offsite	83	0	72	40	582	2,950
7	Adults age > 14 years living offsite, working onsite	10	10	0	102	3,670	5,242
8	Adults age > 14 years living and working offsite (control)	103	0	0	112	226	323

Caravanos et al. (2007) analyzed 36 dust wipes collected from 13 homes within 30 m of the site and 12 collected from a control area. Wipes from within 30 m had mean lead loadings of 64.3 μg/ft^2 compared with 9.7 μg/ft^2 in the control area. The IEUBK utilizes dust lead concentrations rather than loadings. While noting that conversions between the two are prone to significant error, we nevertheless utilize the equation set out by the USEPA Office of Pollution Prevention and Toxics (2010) and provided below (OSWER, 2003):

$$\text{dust lead concentration}, \mu g/g = 50.96 \times \left(\text{dust lead loading}, \mu g/\text{ft}^2\right)^{0.6553}$$

Using this method we calculate a mean dust concentration of 780 μg/g for structures in the perimeter and 226 μg/g for structures offsite. Leaving the IEUBK dust conversions intact, we calculate 3,087 μg/g for the perimeter. Thus, our use of the Caravanos et al. (2007) values is conservative.

To determine the use of each distal area (site, perimeter, and offsite), we make a number of assumptions based on observations before and during the intervention. Where possible we complement these observations with information derived from census data and satellite imagery. Based on the use of each distal area, we time-weight soil concentrations as set out in the relevant IEUBK guidance material (OSWER, 2003). The usage patterns of our eight different exposed groups are given in Table 2. Two of the eight groups are used as controls, as they do not have regular contact with the site. Major assumptions underpinning the other usage patterns are as follows:

1. An elementary school in the perimeter has 150 pupils aged 5–13 years. We assume 60 of these pupils are aged 5–7 years, with all children in this age group living in the perimeter attending school here and the balance living offsite;
2. All children aged 5–7 years living in the perimeter or attending the school play 2 h/d onsite;
3. Twenty scavengers (adults >14 years old) access the site for 10 h each week. Ten live in the perimeter and 10 live offsite;
4. The perimeter has 133 residents. We use satellite imagery to count the households in this area and assume 2.5 to 4 residents per household based on census data, resulting in 102 to 164 residents. We take the average (133) as the population. (National Statistics Office, 2012).

2.5. Calculating Blood Lead Levels

To calculate BLLs for the exposure groups, the USEPA IEUBK and ALM were used. Both models were developed by the USEPA for the American context, and thus rely on default parameters that may not accurately reflect exposure scenarios in informal settlements in the Dominican Republic. One such parameter, ingestion rates, was evaluated here. Some studies, notably Sun and Meinhold (1997) and Harris and Harper (2004), have found higher rates of soil ingestion in LMICs and traditional societies resulting from comparatively dustier conditions than those in high-income countries. The IEUBK default intake values for soil ingestion are 85–135 mg/d, depending on age. By contrast Sun and Meinhold (1997) suggest a value of 500 mg/d, while Harris and Harper (2004) use 400 mg/d. Two recent studies also find large variances in daily soil intake rates. In the American state of Idaho, von Lindern et al. (2016) evaluated the relationship between historical BLL data and bioavailable lead in household and yard dust near a former lead mining and smelting complex, finding intake rates below 100 mg/d for children (<8 years old). Conversely, Kwong et al. (2017) observed the soil ingestion habits of a cohort of Bangladeshi children (<4 years old), finding intake rates from nearly 300 mg/d to 550 mg/d.

To determine which ingestion rate best reflect the study site conditions, we conduct three different batch runs in the IEUBK adjusting the soil ingestion inputs to low, medium, and high values. We then compare the results with the 2009 preintervention BLLs. We use the IEUBK default values of 85–135 mg/d as the low value and 400 mg/d as the high. We then proportionately scale back from this value to a range of 250–400 mg/d. Our medium value is the average of these two, 168–267 mg/d. We leave all other default IEUBK parameters intact. We then conduct three paired *t* tests of actual and predicted BLLs for groups 1 and 2 using Stata 15 and find that the default ingestion rates are the best fit (StataCorp. LP, 2017). We therefore use only these values (85–135) for our DALY calculations.

For adults, we adjust the ALM default soil ingestion values to reflect low, medium, and high rates and adjust exposure frequency to 365 days from 219 to account for residential, rather than occupational exposure. We use the default 50 mg/d as the low intake value, 72.5 mg/d as the medium, and 200 mg/d as the high. The latter two values are set out in the guidance material for reasonable medium and maximum exposure scenarios for occupational settings (OSWER, 2003). The ALM provides a single-point estimate for the geometric mean BLL of an adult worker. We take the low, medium, and high estimates for groups 5 and 6 and compare them to the actual geometric BLLs for adults >14 years old, finding the medium ingestion rate (72.5 mg/d) to be the best fit. We therefore use this value in our DALY calculations. The ALM is not intended for use in residential settings as IEUBK results are meant to determine remediation goals. The ALM is used here as no other comparable method for estimating adult BLLs is known.

2.6. Calculating DALYs

We calculate attributable DALYs from cardiovascular disease and intellectual disability resulting from lead exposure based on values for the year 2013. We do so in a manner consistent with current WHO and IHME approaches and described in WHO (2013).

2.6.1. Cardiovascular Disease

To calculate DALYs from cardiovascular disease (CVD) we utilize a prevalence rate calculator developed by WHO for determining the attributable fraction of CVD due to lead exposure (Fewtrell, Kaufmann, & Prüss-Üstün, 2003). The geometric mean BLL for adults in each group is used to determine the attributable fraction for ischemic, cerebrovascular, hypertensive, and other heart diseases. We then scale the most recent WHO DALY (2013) values for cardiovascular disease in the Dominican Republic to the population in each exposure scenario and apply the attributable fraction to the scaled value (WHO, 2014). The national prevalence of CVD is assumed to be representative of the site and is not calibrated upward to account for possible increases due to lead exposure. We use the most recent WHO DALY values (WHO, 2013) for the Dominican Republic because contemporaneous DALY calculations were done in a method that is no longer utilized. Thus, we assume that the 2013 values are representative of 2009 DALYs.

2.6.2. Intellectual Disability

We use the WHO prevalence rate calculator referred to above and input the geometric mean BLL for exposed children to calculate the prevalence of mild mental retardation (MMR) in children 7 years of age and younger. We assume that mild intellectual disability is analogous to MMR and extrapolate from this value to determine prevalence of borderline, severe and profound intellectual disability. To guide this extrapolation we use values provided by the WHO for the relative prevalence of each sequelae (WHO, 2013). YLD is then calculated with the straightforward multiplicative method below.

$$\mathbf{YLD_i} = \mathbf{DW_i} \times \boldsymbol{P_i},$$

where p = prevalence and DW = disability weight, adapted from WHO (2013).

We use the following WHO disability weights for each gradient of intellectual disability: borderline (0.0034), mild (0.1270), moderate (0.2930), severe (0.3830), and profound (0.4440) (WHO, 2013). We do not attempt to calculate YLL for children.

The WHO prevalence rate calculator utilizes values from Schwartz's (1994) meta-analysis of IQ decrement, which found a 2.6 reduction in IQ for a BLL increase from 10 to 20 μg/dL (Fewtrell et al., 2004). More recent meta-analyses have found higher levels of IQ decrement with a reduction of more than seven IQ points in this range (Budtz-Jørgensen et al., 2013; Lanphear et al., 2005). Thus, in addition to using the default Schwartz (1994) values, we also calculate DALYs from Intellectual Disability using values derived from the log-linear model presented in Budtz-Jørgensen et al. (2013).

2.6.3. Uncertainty

In addition to our best estimate, we calculate DALYs using the lower and upper confidence intervals for each groups BLLs in an approach outlined by WHO (Fewtrell et al., 2003). We also provide estimates of undiscounted DALYs, following Edejer et al. (2003). A more robust statistical analysis might model uncertainty using Monte Carlo analysis. As we do not attempt such an analysis, our results should be considered indications rather than uncertainty values, as such (Fewtrell et al., 2003).

2.7. Lifespan of Project Location

The project involved two major engineering controls to mitigate exposures. The first was the excavation and removal of ~3,000 m^3 of high-level waste and the in situ encapsulation of ~2,500 m^3 of contaminated soil under a cap with a minimum depth of 0.6–1.0 m. The second was the encapsulation of contaminated soil with the construction of a graded and reinforced concrete road. The failure of engineered repositories such as the one at this site is rare. Similar, albeit much larger, repositories around the city of Kellogg, Idaho (USA), for instance, are expected to last at least hundreds of years (EPA Region 10, 2016). Lead is highly immobile in the environment, requiring significant time (~700 years) to meaningfully migrate between horizons (Kabala & Singh, 2001; Semlali et al., 2004). Therefore, we assume that the risk of remobilization of the material is low and place a likely lifespan on the project of 35 years. In addition we model a conservative estimate of 20 years and an optimistic lifespan of 50 years.

2.8. Costs

We use a societal perspective, meaning that we incorporate all costs regardless of by whom they were incurred (Sanders et al., 2016). We reference all costs to 2009, the year the project was initiated. We estimate that the mitigation measures implemented at the site incurred a total cost of USD 430,684. The Ministry of Environment's (MoE's) financial statements and relevant detailed expense reports for FY 2009 are not accessible on the Internet and were not shared with coimplementers. However, an end of financial year report of MoE 2009 budget by program states that 8.9 million Dominican Pesos were allocated to the Remediation of Lead Contaminated Areas in Paraiso de Dios, Haina (Proyecto Presupuesto de Ingresos y Ley de Gastos Publicos 2009, 2009). We convert this amount to USD 247,222 using an exchange rate of 36.0000083198 from 1 December 2009 USD, the approximate date the project began (XE Currency Table: DOP—Dominican Peso, 2017). We assume that this amount covered all costs associated with the onsite phase of the project, including disposal. The Inter American Development Bank project allocated USD 85,000 to this component and Pure Earth reported USD 71,612 in cofinancing expenses, mostly related to project coordination among executing agencies.

TerraGraphics contributed significant in-kind costs to the design and management of the onsite intervention. We do not have access to their final accounting, but a budget in the original project proposal for both onsite and offsite work estimated USD 41,447 for all project design and oversight. We deduct the amount for offsite work paid by the Inter American Development Bank and estimate an in-kind expenditure of USD 26,850. Finally, we account for an outlay of USD 1,000 per year for operation and maintenance (O&M) and discount costs in future years by 3% (Department of Environmental Protection, 2014). Table 3 presents the costs incurred by each party.

Table 3
Costing for Environmental Intervention at Paraiso de Dios by Funding Agency (USD)

Funding source	Amount (USD)	Source of data
Ministry of Environment and Natural Resources	247,222	Relevant line item Ministry of Environment FY 2009 Budget
Inter American Development Bank	85,000	Inter American Development Bank project document
Pure Earth	71,612	Cofinancing letter submitted to IADB
TerraGraphics, Inc. (in kind)	26,850	Proposal budget
Operation and maintenance	1,000/year (Discounted 3%/yr)	
Total	430,684 (not including O&M)	

2.9. Calculating Cost Effectiveness Ratio

To calculate the cost effectiveness ratio (cost per DALY averted) we divide the total cost of the intervention plus the discounted O&M costs by the total DALYs averted.

We assume zero morbidity attributable to the site following the intervention. We further assume that this lack of attributable morbidity remains zero during the entire lifespan of all three scenarios (20, 35, and 50 years). The exposures at the site were the result of legacy contamination rather than ongoing emissions. The engineering controls implemented removed these exposures entirely. Thus, while lifelong disability and socioeconomic effects are likely, they would be the result of exposures from before the intervention and would thus be attributable to that time period (Reuben et al., 2017). Therefore, any attributable morbidity after the intervention would be due to a curtailed lifespan of the repository.

To determine the number of DALYs averted, we calculate the number of DALYs attributable to site and subtract those attributable to lead exposures not related to the site. To calculate the latter value we use background levels identified by Caravanos et al. (2007).

In our counterfactual scenario, the total number of DALYs is the annual amount existing preintervention (2009) multiplied by the total years in each modeled lifespan. We discount future attributable DALYs by 3% to account for societal preference for benefits in the present time (Edejer et al., 2003). That is, we assume a value of 1 for each DALY averted in the first year after the project (2010), while we assume a discounted value for subsequent DALYs averted as a result of the project. Therefore, one DALY in 2010 receives a value of 1, while a DALY in 2030 receives a value of 0.553 and a value of 0.228 in 2060.

We do not discount remediation costs, as all expenses were incurred in the first year. As noted above, we discount operation and maintenance costs at a rate of 3% year. Finally, we follow Edejer et al. (2003) and calculate undiscounted DALYs as well.

3. Results

Using our model, we find that before the intervention 176 people were exposed to elevated environmental lead levels at Paraiso de Dios resulting in mean BLLs of 24.97 (95% CI: 24.45–25.5) in children (0–7 years old) and 13.98 μg/dL (95%CI: 13.03–15) in adults. Measurements taken in the field in 2009 found geometric mean BLLs of 21.3 μg/dL (95% CI: 16.2–28) for children and 21.44 (95% CI: 14.59–31.49) for adults. Using the default values, from Schwartz (1994), for IQ decrement we calculate that without the intervention these exposures would have resulted in 133 to 444 DALYs, depending on the lifespan of the project. Using the revised values for IQ decrement, from Budtz-Jørgensen et al. (2013), we calculate 327 to 1096 DALYs attributable to the site. In all cases children accounted for the vast majority of DALYs (>95%).

With regard to cost effectiveness, using default IQ decrement values we find that one DALY was averted for USD 968 to 3,328, depending on lifespan of the project and discount rate. Using the revised values for IQ decrement, we find a cost per DALY averted of USD 392 to 1,317. Table 4 presents the main results using the default IQ decrement values, while Table 5 presents the main results using the revised values.

Using WHO cost effectiveness criteria, the Haina intervention was very cost effective (Edejer et al., 2003). Notably, this means only that the cost per DALY averted is less than the national annual gross domestic product (GDP) per capita of the Dominican Republic. In 2009, the Dominican GDP per capita was USD 5,099 (in 2010 USD), thus this threshold was easily met (World Bank, 2017).

Table 4
Estimation of DALYs at Paraiso de Dios Organized by Exposure Scenario With Default Values for IQ Decrement (95% CI)

					DALYs averted, discounted 3%			DALYs averted, not discounted		
Popu ation		Soi Pb (mg/kg)	Dust Pb (µg/g)	BLL µg/dL	20 years	35 years	50 years	20 years	35 years	50 years
Attributab e to exposures from the site										
Group 1	13	8,964	4,174	28.94	24.23	34.98	41.87	32.51	56.89	81.28
				(28.25–29.64)	(24.08–24.37)	(34.76–35.17)	(41.62–42.11)	(32.31–32.69)	(56.55–57.22)	(80.78–81.74)
Group 2	8	4,409	780	20.59	11.26	16.25	19.45	15.1	26.43	37.76
				(19.93–21.25)	(11.09–11.42)	(16–16.49)	(19.16–19.74)	(14.87–15.32)	(26.03–26.82)	(37.19–38.31)
Group 3	52	6,337	3,818	24.79	92.56	133.6	159.94	124.18	217.32	310.46
				(24.16–25.42)	(91.76–93.29)	(132.45–134.65)	(158.56–161.2)	(123.11–125.16)	(215.45–219.03)	(307.79–312.9)
Group 5	10	7,869	4,026	28.88	0.66	0.95	1.13	0.88	1.54	2.2
				(27.58–30.19)	(0.65–0.66)	(0.93–0.96)	(1.12–1.15)	(0.87–0.89)	(1.52–1.56)	(2.17–2.23)
Group 6	83	2,950	582	11.77	3.36	4.84	5.8	4.5	7.88	11.25
				(11.24–12.29)	(3.22–3.48)	(4.65–5.03)	(5.57–6.02)	(4.32–4.67)	(7.57–8.18)	(10.81–11.69)
Group 7	10	5,242	3,670	19.74	0.57	0.83	0.99	0.77	1.35	1.93
				(18.44–21.04)	(0.55–0.59)	(0.8–0.86)	(0.96–1.02)	(0.74–0.79)	(1.3–1.39)	(1.86–1.99)
			Subtota (site)		132.63	191.44	229.19	177.95	311.42	444.88
					(131.35–133.82)	(189.6–193.15)	(226.98–231.24)	(176.24–179.54)	(308.41–314.19)	(440.59–448.85)
Attributab e to background exposures										
Group 4	74	323	226	3.48	0	0	0	0	0	0.01
				(3.3–3.65)	(0–0.01)	(0–0.02)	(0–0.02)	(0–0.01)	(0–0.03)	(0–0.04)
Group 8	103	323	226	2.62	0.4	0.58	0.7	0.54	0.95	1.36
				(2.17–3.08)	(0.26–0.58)	(0.37–0.83)	(0.44–1)	(0.34–0.77)	(0.6–1.36)	(0.86–1.94)
			Subtota (offsite)		0.41	0.59	0.7	0.55	0.95	1.36
					(0.26–0.59)	(0.37–0.85)	(0.44–1.02)	(0.34–0.79)	(0.6–1.38)	(0.86–1.97)
			Tota DALYs averted		132.23	190.86	228.49	177.41	310.46	443.52
					(131.1–133.23)	(189.23–192.3)	(226.54–230.22)	(175.89–178.75)	(307.81–312.81)	(439.73–446.87)

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Table 5
Estimation of DALYs at Paraiso de Dios Organized by Exposure Scenario With Revised Values for IQ Decrement (95% CI)

					DALYs averted, discounted 3%			DALYs averted, not discounted		
Popu ation		Soi Pb (mg/kg)	Dust Pb (μg/g)	BLL μg/dL	20 years	35 years	50 years	20 years	35 years	50 years
Attributab e to exposures from the site										
Group 1	13	8,964	4174	28.94	60.94	87.96	105.3	81.76	143.08	204.4
				(28.25–29.64)	(60.61–61.24)	(87.48–88.4)	(104.73–105.83)	(81.32–82.17)	(142.3–143.8)	(203.29–205.43)
Group 2	8	4,409	780	20.59	29.51	42.6	51	39.6	69.3	99
				(19.93–21.25)	(29.13–29.88)	(42.05–43.12)	(50.34–51.63)	(39.09–40.08)	(68.4–70.15)	(97.72–100.21)
Group 3	52	6,337	3818	24.79	232.19	335.14	401.22	311.52	545.16	778.8
				(24.16–25.42)	(230.4–233.83)	(332.56–337.51)	(398.13–404.06)	(309.12–313.73)	(540.96–549.02)	(772.8–784.32)
Group 5	10	7,869	4026	28.88	0.66	0.95	1.13	0.88	1.54	2.2
				(27.58–30.19)	(0.65–0.66)	(0.93–0.96)	(1.12–1.15)	(0.87–0.89)	(1.52–1.56)	(2.17–2.23)
Group 6	83	2,950	582	11.77	3.36	4.84	5.8	4.5	7.88	11.25
				(11.24–12.29)	(3.22–3.48)	(4.65–5.03)	(5.57–6.02)	(4.32–4.67)	(7.57–8.18)	(10.81–11.69)
Group 7	10	5,242	3670	19.74	0.57	0.83	0.99	0.77	1.35	1.93
				(18.44–21.04)	(0.55–0.59)	(0.8–0.86)	(0.96–1.02)	(0.74–0.79)	(1.3–1.39)	(1.86–1.99)
		Subtota (site)			327.22	472.31	565.44	439.03	768.3	1097.57
					(324.56–329.69)	(468.47–475.88)	(560.85–569.71)	(435.46–442.34)	(762.05–774.1)	(1088.65–1105.86)
Attributab e to background exposures										
Group 4	74	323	226	3.48	0.01	0.01	0.01	0.01	0.02	0.03
				(3.3–3.65)	(0–0.06)	(0–0.08)	(0–0.1)	(0–0.08)	(0–0.13)	(0–0.19)
Group 8	103	323	226	2.62	0.4	0.58	0.7	0.54	0.95	1.36
				(2.17–3.08)	(0.26–0.58)	(0.37–0.83)	(0.44–1)	(0.34–0.77)	(0.6–1.36)	(0.86–1.94)
		Subtota (offsite)			0.41	0.6	0.71	0.55	0.97	1.38
					(0.26–0.63)	(0.37–0.92)	(0.44–1.1)	(0.34–0.85)	(0.6–1.49)	(0.86–2.13)
		Tota DALYs averted			326.81	471.72	564.73	438.48	767.33	1096.19
					(324.3–329.06)	(468.11–474.96)	(560.4–568.62)	(435.12–441.49)	(761.45–772.61)	(1087.79–1103.73)

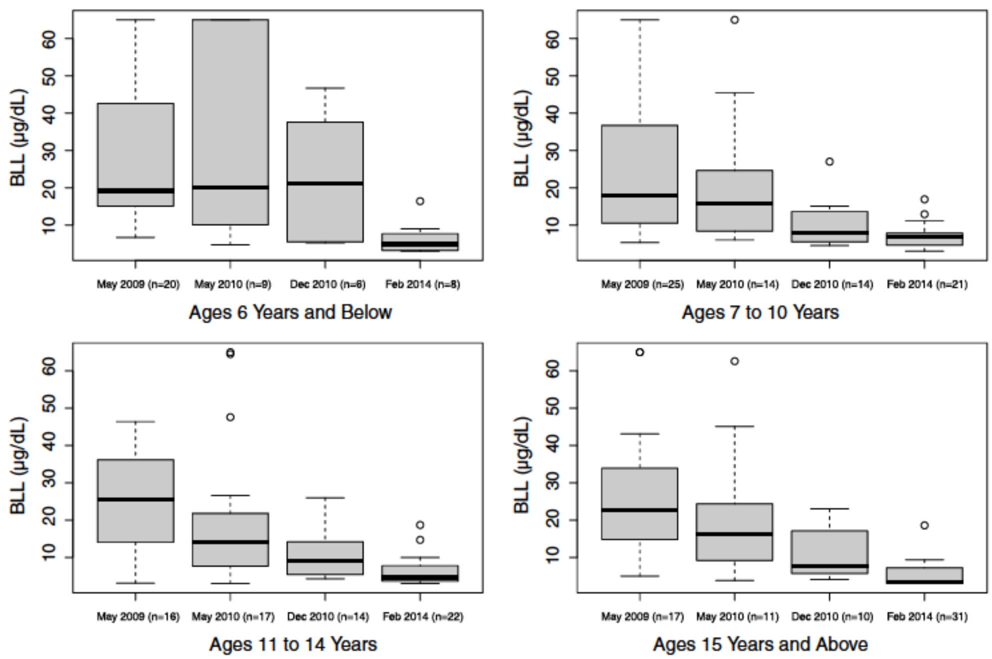

Figure 2. Blood lead level analysis results (median, IQR, and outliers) from 2009 to 2014 stratified according to age group.

Analyzed BLLs for all sampled residents decreased from a geometric mean of 20.6 μg/dL (95% CI: 17.5–24.3; GSD = 2.08) in May 2009 to 5.34 μg/dL (95% CI: 4.8–6; GSD = 1.67) in February 2014. In an unpaired *t* test the decrease was statistically significant ($p < 0.05$). Median and IQR values for BLLs are presented in Figure 2 stratified by age group. Twenty-five residents were tested in both 2009 and 2014. Over this period, the BLLs of these residents declined from a geometric mean of 22 μg/dL (95% CI: 15.8–30.7; GSD = 2.23) to 5.7 (95% CI: 4.4–7.3; GSD = 1.84). In a paired *t* test the decrease was statistically significant ($p < 0.05$) Figure 3.

Arithmetic mean BLLs for all residents decreased from 26.2 μg/dL (95% CI: 22.3–30; SD = 17.3) to 6.2 μg/dL (95% CI: 5.3–7; SD = 3.7). The BLLs of 20 individuals in the 2014 study were at or below the lower detection limit of the equipment (3.3 μg/dL), indicating that actual BLLs following the intervention are lower than we report here.

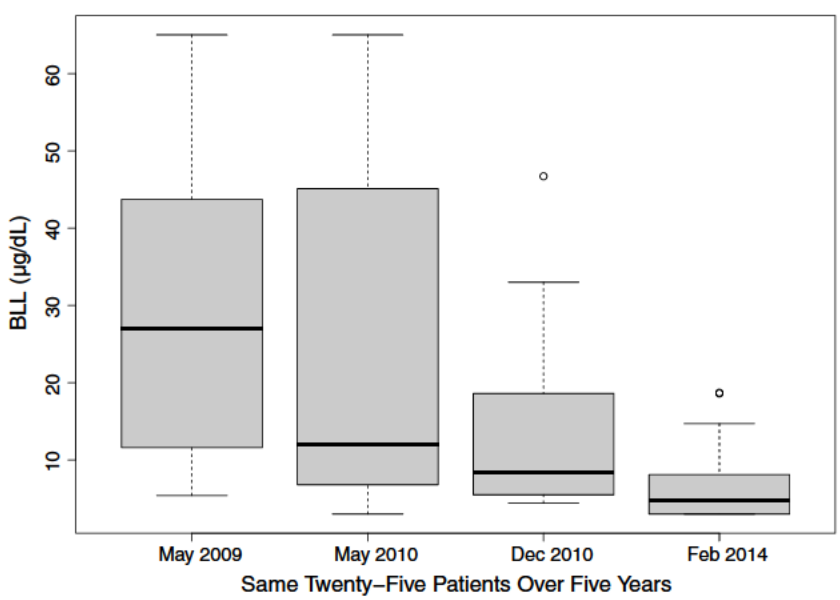

Figure 3. Blood lead levels (median, IQR, and outliers) for the same 25 residents over the 5 year period evaluated in this study (Median age in 2009 = 10 years).

4. Discussion

4.1. Cost per DALY Comparison

There is a dearth of studies on the topic of environmental remediation in the LMIC context, and we are aware of no such study that reviews an intervention through cost effectiveness analysis. As such, our results are of limited utility in assessing the project's cost effectiveness through the narrow lens of environmental remediation. A number of studies utilize cost effectiveness analysis to compare interventions targeting the same set of risk factors or a specific disease; however, those conducting a cross-sectorial analysis are limited (Murray et al., 2000). Additionally, the utilization of DALYs alone to calculate the lead attributable disease burden has been criticized for failing to capture a range of adverse societal and health impacts (Grandjean & Bellanger, 2017).

A related but separate approach, cost-benefit analysis, has been applied elsewhere to quantify the social and economic benefits of public health interventions. Gould (2009), for instance, found that each USD invested in lead paint hazard control in the USA yields USD 17–221 in returns, exceeding the cost-benefit of vaccinations, calculated at between USD 5.30 and 16.50. Future efforts might endeavor to calibrate these findings for the LMIC context.

The Institute for Health Metric Evaluation (IHME) recently estimated that between 4.2 and 15.6 million DALYs resulted from lead exposure in 2015 (Forouzanfar et al., 2016). Their analysis relies largely on the impact of exposures resulting from aerial deposition from leaded gasoline. Due in part to a lack of information, exposure at hazardous waste sites like Paraiso de Dios are not included. A number of recent publications provide evidence that the disease burden from these sites may be significantly larger than previously thought. These include case studies of lead poisoning epidemics recently reported in Nigeria, Senegal, Vietnam, and Zambia (Ajumobi et al., 2014; Caravanos et al., 2014; Haefliger et al., 2009; Noguchi et al., 2014) as well as estimations of the prevalence and nature of contaminated sites. Ericson et al. (2016), for instance, estimate the existence of between 10,599 to 29,241 lead contaminated sites result from the informal recycling of car batteries in LMICs adversely affecting the health of up to 16 million people. Separately, Dowling et al. (2016) conducted an extrapolation exercise in one country (Ghana) estimating the presence of between 812 and 3,075 contaminated sites with an apparent human exposure pathway. Of these, more than one third were contaminated with lead.

Studies from both LMICs and high-income countries have demonstrated the effectiveness of environmental remediation in mitigating exposures to toxic substances. Significant declines in elevated BLLs, for example, following soil abatement have been observed in a number of sites (Ericson et al., 2018; Lanphear et al., 2003). In the resource-poor environment of LMICs, there is a pressing need to include cost in the analysis of effectiveness; this paper offers one possible method.

A separate, possibly significant, finding of this paper is the likely underestimate of previous WHO estimates of the pediatric disease burden from lead exposure. The WHO prevalence rate calculator utilizes default values from an older study (Schwartz, 1994). More recent and robust analyses (Budtz-Jørgensen et al., 2013; Lanphear et al., 2005) find larger neurological deficits.

4.2. Limitations

Our analysis offers a potentially useful first assessment of environmental intervention; that of cost effectiveness analysis as regard contaminated site remediation in LMICs. The analysis has inherent limitations. The most significant of these is the reliance on limited environmental and biological data. The environmental and biological data initially collected at Paraiso de Dios were not necessarily intended for this sort of analysis, and consequently, the blood lead data lack extensive age or household information, limiting our ability to determine its spatial distribution in the community. While the existing environmental data are fairly comprehensive within 30 m of the site, data density tapers off significantly beyond this area. The result is that the disease burden has been underestimated in this study and the calculations should be viewed as a conservative evaluation.

A related limitation is the relatively small number of analyzed BLLs. While additional data on each age group would improve the statistical power of the analysis, the existing data show that BLLs in the final sample (February 2014) are significantly lower ($p < 0.05$) than the first sample prior to remediation in May 2009.

Regarding the future disease burden, we argue that the intervention resulted in zero prevalence of disease attributable to the site in subsequent years. We then calculate DALYs averted over three different time frames (20, 35, and 50 years) deducting background exposures. We are unaware of this approach being used elsewhere.

We use national population profiles and CVD rates for the study population. Studies in LMICs have found that both the relative proportion of children and CVD rates are higher in low-income areas like Haina (Gaziano et al., 2010). Our analysis calculates DALYs from IQ decrement in children and CVD rates in adults only, thus these demographic assumptions likely result in an underestimate of the disease burden.

Regarding the cost of the intervention, much of the historical data were inaccessible or commercial in confidence. We take submitted estimates at their face value, absent a reasonable alternative. It is likely that the most significant assumption with regard to cost is that the USD 247,222 MoE budget line item is inclusive of all disposal costs. In reality the value of the excavated material hauled to the new smelter likely vastly exceeds that of the overall project. It is possible, though has not been established, that this material was reprocessed and that the excavation costs were reimbursed to MoE. If this were the case, the societal cost of the intervention would be greatly reduced. However, due to the confidential nature of these arrangements, we conservatively include the full USD 247,222 in our costs. Additionally, we have not endeavored to calculate benefits accrued through related increased taxable income of the smelter or local project executors.

Finally, we depart from WHO guidance for the prevalence rate calculator in two significant ways. First, we calculate prevalence of MMR for ages 5–7 years, while it was intended for ages 0–4 years only. We argue there is sufficient justification in the literature to support the assumption that neurological decrement continues beyond age 4. Budtz-Jørgensen et al. (2013) and Lanphear et al. (2005), for instance, both found the strongest association of IQ decrement with BLLs in concurrent measurements of school age children aged 5–10 years. Reuben et al. (2017) evaluated the relationship between cognitive function at age 38 with BLLs taken at age 11 in 1,037 New Zealanders. In addition to identifying a strong association between lower socioeconomic status and childhood BLLs, Reuben et al. (2017) found that each 5 μg/dL increase in BLL at age 11 was associated with a decrease in 1.61 IQ points by age 38, indicating a continued loss of IQ attributable to childhood BLL. Second, we augment the estimated prevalence of MMR in the population with the use of the log-linear model from Budtz-Jørgensen et al. (2013). We argue that these results are much more robust than those presented by Schwartz (1994) and present both values in Tables 4 and 5.

5. Conclusions

On its face, the intervention at Haina would appear to have significant utility at similarly impacted environments. However, further research of equivalent type projects is required to determine its true cost effectiveness relative to other public health interventions and also to ensure scarce resources are allocated efficiently within and between projects.

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Erratum

In the originally published version of this article, a donor was omitted from the acknowledgments. The acknowledgments have since been updated, and this version may be considered the authoritative version of record.

5. Chapter 5: Synthesis of Research

5.1. Historical Context

Twentieth century lead emissions paralleled industrial expansion in high income countries and resulted in extensive contamination of human and natural environments. The associated chemical attributable disease burden was unprecedented. Because of lead's recalcitrance in the environment, much of what was released from car engines and old paint decades ago continues to be re-suspended and ingested by humans.[1] Laidlaw et al. (2016) for instance showed that in the case of the lead poisoning event in Flint, Michigan (USA), the majority of the underlying biological lead burden was not attributable to the switch in water supply as was popularly believed, but to residual contamination on the ground.[1] This point provides important context for the present discussion. Specifically, relatively low concentrations of lead deposited 40 years ago continue to pose an unacceptable risk. Put differently, on human scale, lead pollution in the environment is permanent.

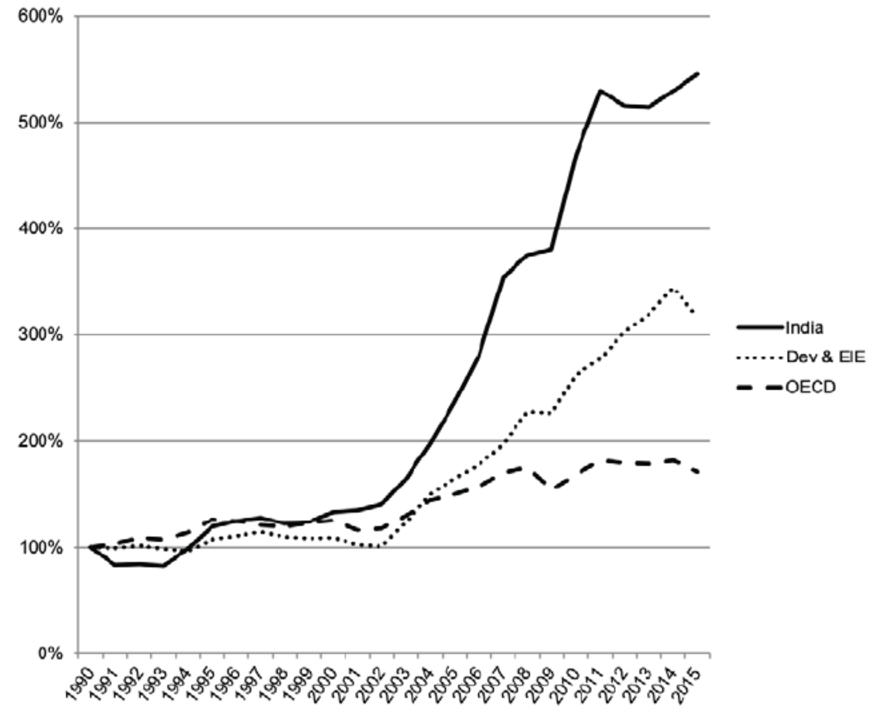

Figure 1. *Percent growth of manufacturing. OECD: Organization of Economic Cooperation and Development countries; Dev & EIE: Developing and Emerging Industry Economies.*[3]

A similar significant industrial expansion is occurring in LMICs. For example, from 1990 to 2015, the Indian manufacturing sector grew by more than five-fold. Across all low- and middle-income countries (LMICs). more generally, the expansion was closer to three-fold during this period. In China, the manufacturing sector grew by a

[1] Issues consequent to the water supply change were associated with an incremental increase in the percentage of the population with BLLs above the CDC threshold of 5 µg/dL.[46] However, BLLs were also elevated in Flint before the switch and were associated with seasonality, indicating that contaminated soil was the primary exposure pathway.[2,47]

remarkable twenty-five times from 1990 to 2015.[3]

The expansion draws on a range of commodities, including lead, much of which is being refined in LMICs. By way of example, in 2013, the US, the world's largest historical lead producer, refined 1.3 million tons, that country's most productive year on record. In the same year, China refined more than 3 times that amount (4.5 million tons).[4]

5.2. Summary of Chapter 2

This thesis explored the implications of this industrial expansion in terms of human lead exposure and proposed and evaluated methods of mitigation. Chapter 2 was concerned with the extent and severity of exposures. The first study in Chapter 2, Paper 1, provided important context for the thesis. The study is the largest systematic analysis of blood lead levels (BLLs) in LMICs carried out to date. For comparison, the Institute for Health Metrics and Evaluation's (IHME) most recent disease burden calculation included 88 studies from 2010 to the 2017.

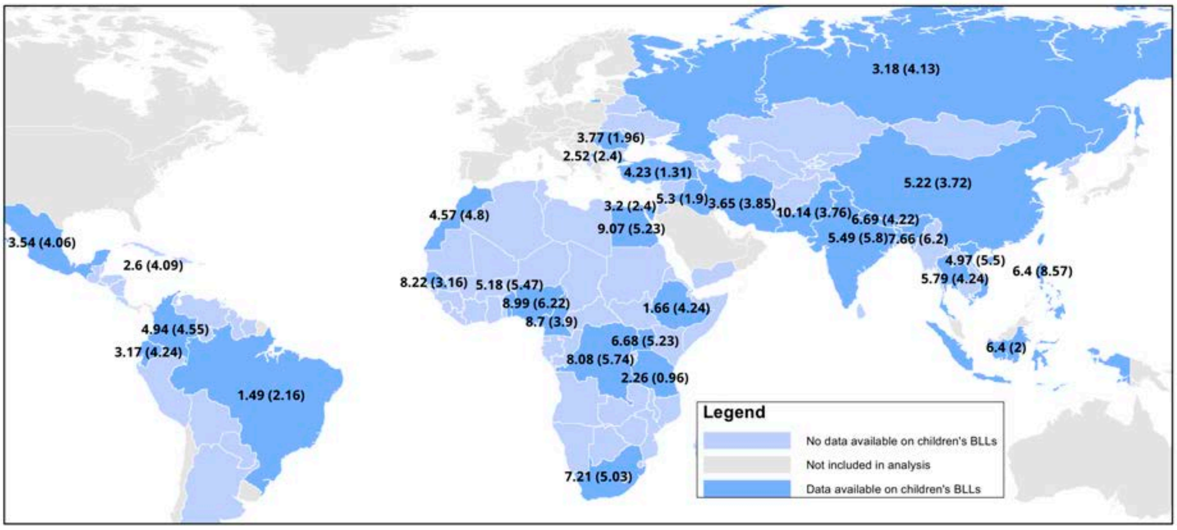

Figure 2. *Pooled mean BLLs (µg/dL) and standard deviations of children in multiple countries (Paper 1).*

Paper 1 extended this period by six months to June 2018 and reviewed more than five times the number of studies (n=477). The importance of expanding the review was not simply to refine the precision of national BLL estimates. Doing so also illuminated important differences between historical sources of exposure in high income countries and present-day exposures in LMICs. In LMICs industrial sources seem to dominate. Indeed, only 4 of the 110 background subsamples with confirmed sources of exposure listed lead-based paint as the source. Only one provided plumbing as a source of exposure. By contrast, 65 % (n=71) of subsamples had an industrial source of exposure, with 34 % (n=37) being associated with lead recycling or smelting.

Paper no.	Title	Chapter	Key contributions to the thesis
1	Sources of ead exposure and resu t ng b ood ead eve s n ow and m dd e ncome countr es: a systemat c rev ew and meta ana ys s.	2	• BLLs rema n e evated n LMICs • Sources of ead exposure n LMICs tend to be ndustr a
2	Est mat ng the preva ence of tox c waste s tes n ow and m dd e ncome countr es.	2	• Presents a method for est mat ng the number of contam nated s tes n a country
3	Assessment of the preva ence of ead based pa nt exposure r sk n Jakarta, Indones a.	2	• Lead based pa nt does not appear to present a s gn f cant exposure r sk n th s case
4	Assessment of the Presence of So Lead Contam nat on Near a Former Lead Sme ter n Mombasa, Kenya	2	• Emphas zes the mportance of ev dence based dec s on mak ng
5	Lead ntox cated ch dren n Kabwe, Zamb a	2	• H stor ca y mportant case of c tyw de ead po son ng that, n the context of other stud es n th s thes s, underscores naccurac es n GBD est mates
6	The g oba burden of ead tox c ty attr butab e to nforma used ead ac d battery s tes	3	• Quant f es the emerg ng pub c hea th threat of nforma ULAB recyc ng for the f rst t me
7	Pérd da de coef c ente nte ectua en h jos de a fareros mex canos	3	• Quant f es hea th burden from a preva ent trad t ona ndustry n Mex co
8	A meta ana ys s of b ood ead eve s n Ind a and the attr butab e burden of d sease	3	• Ca cu ates DALYs attr butab e to ead exposure n Ind a • H gh ghts poss b e under est mate n IHME ca cu at on
9	Burden of d sease resu t ng from ead exposure at tox c waste s tes n Argent na, Mex co and Uruguay	3	• Quant f es hea th burden of ead exposure at hazardous waste s tes n se ect Lat n Amer can countr es
10	Improv ng human hea th outcomes w th a ow cost ntervent on to reduce exposures from ead ac d battery recyc ng: Dong Ma , V etnam	4	• Affordab e methods to m t gate ead exposure can be effect ve y adapted for the LMIC context
11	Prevent on ntervent on strateg es to reduce exposure to e waste.	4	• Affordab e methods to m t gate ead exposure can be effect ve y adapted for the LMIC context
12	Cost Effect veness of Env ronmenta Lead R sk M t gat on n Low and M dd e Income Countr es	4	• Lead remed at on n LMICs s cost effect ve

Table 1. *List of studies included in this thesis, associated chapters, and major contribution to the thesis*

A separate significant finding of Paper 1 was that BLLs in LMICs remain elevated. Sixteen of the 32 countries from which BLLs could be extracted for children were found to have pooled mean BLLs above 5 μg/dL for that age group. Importantly, these numbers represent only the control populations in each study, meaning that in most cases they were specifically selected for an absence of exposure. They do not capture the more severe exposures, such as near hazardous waste sites, and therefore likely represent as a conservative indication of the scale of lead toxicity in each country. Taken together, these points, that population-wide BLLs remain elevated and that the sources appear to be industrial, provide an important basis for the balance of this thesis.

Two studies presented in Chapter 2 identified the absence of a relationship between a risk factor and an adverse outcome. In the first (Paper 4) a lead smelter abutting an informal settlement in Kenya had drawn international condemnation for contamination linked to its operations. The initial purpose of the investigation was to respond to the apparent emergency by designing an intervention. However, soil measurements taken as part of this investigation consistently found concentrations below international screening levels, indicating that no intervention was likely required. The soil results were later confirmed by a separate US CDC study.[5] Despite the international attention paid to this site, a number of others within Kenya likely present a much higher risk. For example, the study highlighted the existence of at least 63 other sites assessed by Pure Earth in Kenya with markedly higher environmental contamination and attributable exposure risk.[6,7] Thus, the study highlights the importance of prioritizing interventions based on evidence, particularly in the context of limited funding for such interventions.

The second study to find an absence of risk (Paper 3) makes a similar argument in evaluating the extent of lead-based paint exposure in residential settings in Jakarta. Lead-based paint has formed the focus of international health funding related to lead since the phaseout of leaded petrol. Further, the only existing multi-lateral effort to mitigate lead exposure is the World Health Organization/ UN Environment Program Global Alliance to Eliminate Lead Paint (GAELP). However, this approach seems to rely on the *a priori* assumption that lead-based paint presents an exposure risk in LMICs. Yet a defining characteristic of lead-based paint exposure assessments in LMICs seems to be their near absence. To better characterize the potential exposure risk from lead-based paint, this study assessed randomly selected homes (n= 103) preschools (n=19) across Jakarta, Indonesia. Of the 1,574 paint and 222 dust measurements taken, only 2.7 % of the paint and 0.05 % of the dust exceeded applicable guidelines. A similar study conducted in the US nearly 30 years after the lead-based paint phase out in that country found that 40 % of structures contained lead-based paint.[8] Thus, the exposure risk in Jakarta appears low by comparison.

Chapter 2 includes two additional studies. The first (Paper 2) utilizes *in situ* data collected throughout the country of Ghana to estimate the total number of contaminated sites in that country. The results of this exercise are potentially useful in quantifying exposures elsewhere. Indeed, in Chapter 3 this approach was one of two methods utilized to calculate the global number of informal battery processing sites (Paper 6). The second paper (Paper 5) summarizes recent research on a citywide case of lead contamination. The city of Kabwe, Zambia (population ~200,000) was developed as a lead-zinc mining town (Broken Hill) by the British in the early part of the 20th century. Emissions related to the mine have resulted in arguably the most severe example of lead poisoning of a population in the literature. The results are arresting. Mean BLLs in children are > 35 µg/dL across the city and regularly reach above 100 µg/dL in some individuals. However what is perhaps most disconcerting about the results is their striking similarity to a study executed at the same site in 1975 by then MPH student A.R.L. Clark.[9] The unpublished thesis for the London School of Tropical Hygiene presents BLL results from individual neighbourhoods throughout Kabwe city, enabling direct comparison with the contemporary results aggregated in Paper 5. In the neighbourhood of Kasanda, near the mine, Clark found mean BLLs of 42–104 µg/dL depending on the age analysed. The contemporary means presented in Paper 5 ranged from 51.6–82.2 µg/dL. In Chowa, another neighbourhood near the mine, contemporary mean BLLs range from 39–49.8 µg/dL, compared to 30–70 µg/dL in 1975. These two studies bridge the mine's peak operation in the 1970s with a period 20 years after its closure. The implication being that exposure has not substantially decreased and, absent concerted intervention, never will.

No other case contrasts the severe nature of lead exposures in LMICs with the absence of international response more so than Kabwe. The first major effort to mitigate exposures here did not occur until nearly 30 years after the Clark study and was largely ineffective.[10] A new effort began in 2016, triggered in part by the research presented in this thesis and, like the first, funded by the World Bank.[11] If this project is successful, it would be the first major effort to reduce lead exposures in the 117 years since the mine opened, and 44 years since the first formal assessment of the mine's impact on public health. The compounded adverse health, social, and economic implications of 117 years of exposure are necessarily profound.

A final study was intended for Chapter 2 that sought to evaluate the efficacy of remote sensing technologies to identify lead contaminated sites from satellites. The proposition was to utilize multispectral imagery coupled with *in situ* assessments from Kabwe to identify possible associations in reflectance and soil lead concentrations. There is some precedent for this.[12] The manuscript is currently in revision leading to submission. The most recent draft is attached as Appendix B. Following on this study, BE co-organized a half day workshop at the

World Bank in December 2018 to explore this approach. The workshop resulted in a funded USAID contract for MIT's Lincoln Laboratories to further assess this approach. The concept note for the workshop is attached as Appendix C. While the results of this work are not yet published, the effort itself grew out of this thesis and is therefore included here.

5.3. Summary of Chapter 3

Chapter 3 was concerned with evaluating the exposure sources identified in Chapter 2 and quantifying their disease burden in public health metrics. This was done most commonly through the calculation of Disability Adjusted Life Years (DALYs). In short DALYs are the sum of mortality and morbidity in a given population. For an individual, the absence of disease receives a value of zero, while the absence of health (i.e. death), receives a value of one. Various diseases fall along this spectrum with a disability's weight increasing with its severity. Thus, metastatic cancer has a higher disability weight than a broken arm which has a higher disability weight than a headache, and so on. Duration of the ailment is included as a percentage of the year evaluated. If an individual has a disease for six months of a given year, the applicable disability weight is halved. On a population level, these numbers are summed. Causes of disabilities or deaths are attributed to different risks. Lead exposure, for example, is categorized as a risk associated with the disability causes of cardiovascular disease and intellectual disability.

Paper 6 transitions between Chapters 2 and 3, estimating exposures as well as the attributable disease burden. The study provides estimates of the number of informal used lead acid battery (ULAB) recycling sites globally, the size of the exposed populations, and the attributable disease burden. At the core of the study is a simple mass balance equation of lead being recycled. The total number of ULABs generated each year is calculated, and the amount recycled formally is subtracted from the total. It is then assumed that the balance is recycled informally. Exposure assessments from 28 ULAB recycling sites in 12 countries were used to model typical exposure scenarios. These were then applied to the calculated sites to model exposure. The study found that 10,599–29,241 informal ULAB recycling facilities existed in 2013, each putting the health of an average of 575 people at risk. The study was the source of the lead-attributable DALYs estimates in the recent *Lancet Commission on Pollution and Health*.[13] It was also used as the source of lead exposure estimates in the World Bank's Country Environmental Assessment (CEA) of Bangladesh and the forthcoming Pakistan CEA.[14] One way to conceptualize these sites is as thousands of smaller, hidden 'Kabwes,' occurring across 90 countries and yet to be documented. From this perspective the need to synthesize an approach to address the problem becomes particularly pressing.

Papers 7 and 9 calculated the lead-attributable disease burden using two common metrics. Paper 7 uses intellectual quotient (IQ) decrement resulting from lead exposure in the households of ceramics artisans. There are approximately 10,000 households producing traditional ceramics in Mexico using lead-oxide. As this activity is typically done in the home or adjoining workshop, the resulting exposures are significant. This study utilized soil concentrations from 19 different homes to calculate the resulting BLLs and IQ decrement. The paper was originally written in Spanish and published in a leading Mexican medical journal so as to have maximal public health impact in the affected country. This strategic decision was part of a larger campaign by Pure Earth to encourage concerted action on the issue. This effort resulted most recently in the first nationally representative BLL assessment in Mexico carried out by the Mexican Social Security Institute (IMSS), which is analogous to the CDC in the US. The results of that stud, which indicate both that BLLs are elevated and that ceramics are a key source of exposure, are currently being compiled as a manuscript for peer-reviewed publication with BE as a co-author.

Paper 9, which calculated lead-attributable DALYs in Latin America was, like Paper 7, part of series of studies.[15–18] The series utilized data collected by Pure Earth's Toxic Sites Identification Program to calculate the disease burden at hazardous waste sites.[6,7] The overall objective of this effort was to identify and quantify a previously unevaluated disease burden. In the case of Paper 9 the evaluation looked only at lead exposures present at these sites.

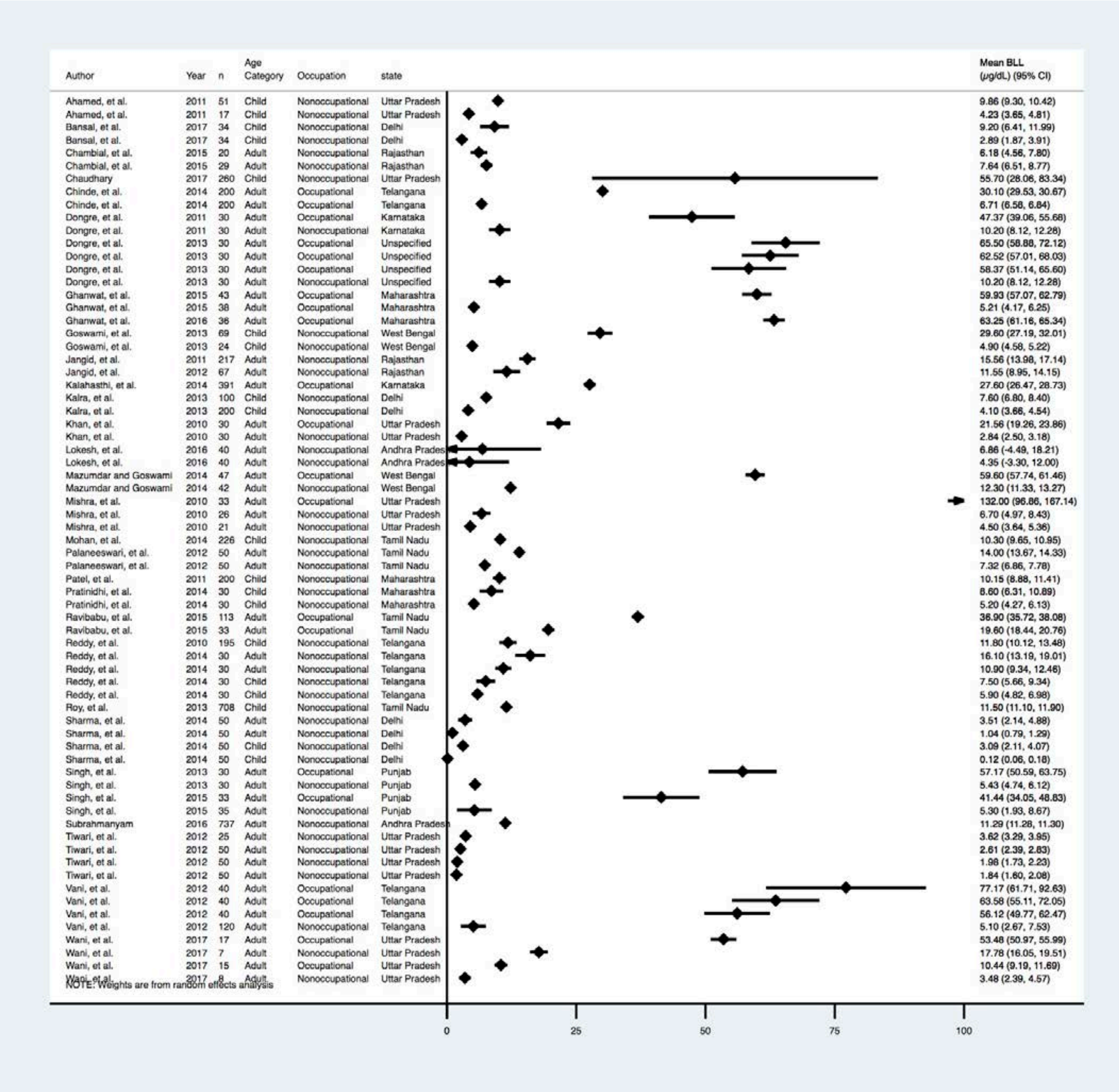

Figure 3. *Forest plot of BLL subsamples evaluated in Paper 9 (µg/dL).*

Paper 8 included a systematic review of published BLLs in India. Because very few national BLL surveys exist in LMICs, most BLL testing is done as part of academic research studies. These studies tend to evaluate an exposed population, typically occupational in nature, and a control population intended to represent the absence of exposure. By evaluating these control populations, as was done in Papers 1 and 8, one can undertake a reasonable assessment of background BLLs in a given country. In the case of Paper 8, studies with BLL data from one country, India, were utilized. Age and gender specific BLLs were extracted where available and pooled using a random effects model. Those BLLs were then used to calculate DALYs using WHO protocols. Where inputs of the IHME were published, they were used in the sensitivity analysis. The study highlighted a potentially significant undercounting of lead attributable DALYs in children (< 5 years) by IHME. Their 2016 Global Burden of Disease (GBD) report estimated about 33,000 DALYs for Indian children (< 5 years) attributable to lead. By contrast, Paper 8 found more than 2 million DALYs in children (< 5 years) in only the 6 states evaluated. Using the IHME inputs that are publicly available (disability weights) resulted in a reduction in attributable DALYs to a total of just more than 700,000, thus reducing the discrepancy somewhat. However, the ~33,000 DALYs calculated by IHME stands out as unusually low, particularly in comparison to other risks and outcomes. The 2016 estimates are no longer available in detail through the IHME website, though in 2017 IHME attributed ~1.3 million DALYs in adults to tension headaches in India. A further ~500,000 were attributed to bullying. Low fibre diets accounted for ~4.4 million DALYs.[19]

The research presented in Chapter 3 was part of a more prolonged effort to persuade IHME to re-evaluate the burden of disease attributable to lead and other environmental toxicants. That effort in part resulted in a workshop at IHME's offices in February 2018 with leading epidemiologists and environment health scientists to discuss the relative contribution of environmental toxicants to GBD. One direct result of that workshop was a commentary in *Environmental Health* by David Bellinger on the under-recognized role of neurotoxicants in disease burden calculations.[20]

A separate, likely related, result was the doubling of lead-attributable deaths from ~500,000 to ~1 million in 2017 GBD compared with the previous year. Lead-attributable DALYs increased from 14 million to 24 million, now representing more than 1 % of the global disease burden.[19] With regard to India specifically, IHME now attributes ~165,000 DALYs to lead exposure in children (< 5 years), a fivefold increase over the previous year. This brings the estimate closer in line to that presented in Paper 8. But it is also curious, as IHME's global number of lead-attributable DALYs for children (< 5 years) is only ~262,000, meaning that in their analysis, India comprises more than 62 % of the global lead-attributable disease burden in children (< 5 years). Further evaluation reveals only 7,836 lead attributable DALYs in this age group in China, the world's largest lead recycler. This seems to be in contrast with the extensive literature on lead-exposed children in China. Paper 1, for instance, utilized data from 142 different studies in China since 2010 finding a mean BLL of 5.22 µg/dL in non-exposed children. Of those 42 samples comprised of 7,852 exposed children had a geometric mean BLL of 9.23 µg/dL. Given that there are about 50 million children in this age group in China, this would seem to imply a considerable underestimate in IHME's report. Moreover, their underestimate is consistent across all countries, except for perhaps India.

5.4. Summary of Chapter 4

The last Chapter in this thesis to present published research, Chapter 4, is comprised of a series of mitigation case studies. Of the multiple cases presented, BE was involved with the execution of four of them. Two of the four interventions described are ULAB recycling sites; two are e-waste sites. The chapter is intended to both share experiences with other potential implementers and evaluate evidence about whether lead exposures can be cost effectively mitigated. The first study in the Chapter, Paper 10, describes an intervention in the Vietnamese village of Dong Mai, about 50 km east of Hanoi. The village had recycled lead in residential settings for decades. In 2006, a study carried out by the Vietnamese National Institute of Environmental and Occupational Health (NIOEH) identified elevated BLLs triggering the construction of an industrial area 1 km from the village by the Vietnam Environmental Agency and a government-wide effort to relocate the recyclers. The intervention described in the study began in 2013 with an investigation that revealed extensive lead contamination of area surfaces. In addition > 30 informal lead recyclers had not at that time been relocated to the industrial cluster. Over the next two years, an intervention was carried out to remediate residential areas and encourage relocation. The primary engineering control employed was the capping of contaminated soil with a permeable geotextile fabric and 20 cm of compacted clean soil. The research study details a 67 % decline in children's median BLLs from 40.4 µg/dL to 13.3 µg/dL in the same children (< 6 years old) over a one-year period following completion of the project. The project was executed for slightly over USD 100,000 and demonstrably impacted the health of more than 3,000 people. Since the intervention, BLLs have reportedly fallen below 10 µg/dL, though those data are not publicly available. The project came to national attention in Vietnam and provided the justification for more extensive BLL monitoring by NIOEH nationally. In addition, the general approach of relocating and remediation is currently informing approaches to similar sites in Bac Ninh province north of Hanoi. Paper 10 is likely the first study to document remediation at a ULAB site in an LMIC.

In the case of Dong Mai a number of factors contributed to the project's success. In the industrial zone, worker protections and environmental controls were introduced incrementally. This resulted in minimal impact on the livelihoods of workers and no incidence of industrial activity returning to the village. The smelters still present an unacceptably high risk to workers (Chapter 1; Figure 7), though that risk is reduced each year through incremental improvements. The cost efficient approach used in the village to remediate homes was affordable enough to be done by the residents themselves, facilitating broad community involvement. This resulted in the vast majority of remediation costs being voluntarily assumed by the residents. The project, for example, funded the remediation of 37 yards, while 67 were remediated by homeowners. Before the project began, few arterial roads in the village were paved. At present, nearly all roads have been paved with the community sharing costs with the Vietnamese government. This level of community investment was the result of a concerted effort to engage residents early and often in the design and execution of the project.

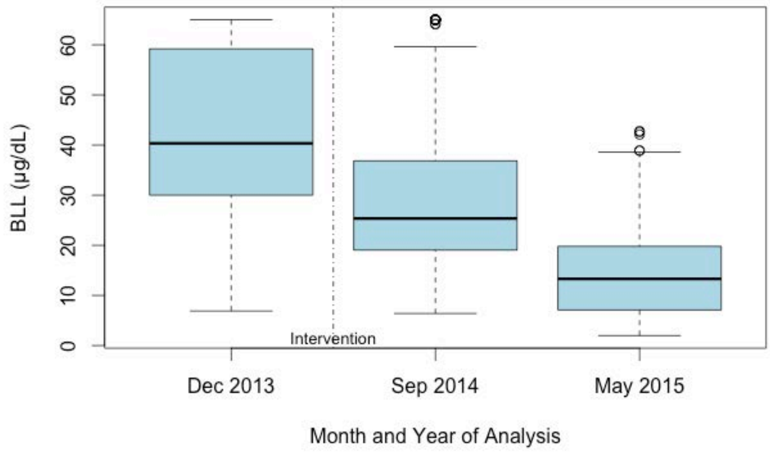

Figure 4. *Decreases in median BLLs of children within a year of the intervention in Dong Mai (Paper 10)*

Paper 11 describes 5 different e-waste interventions. Documentation of two of those was generated by work conducted as part of this thesis. In the first (Ghana), wire stripping equipment was installed at an informal e-waste processing centre in Accra. The industrial process at the centre of e-waste recycling is the burning of electronic cables to remove the sheaths and acquire the copper inside. That copper is then sold to formal sector operators for use in other products. This process creates a range of airborne hazards associated with burning plastics (e.g. PM, CO, dioxins). But it also, importantly, deposits significant quantities of lead onto the ground. This is because lead is still commonly added to cable sheathing as a stabilizer. Accordingly soil lead concentrations at e-waste burn sites are highly elevated. At the second case described in Paper 11 (Uruguay) these concentrations were found at up to 18,000 mg/kg in residential areas. As context, the US and Canadian residential screening levels are 400 mg/kg and 140 mg/kg, respectively.[21,22] Concentrations in Dong Mai and Kabwe ranged from 3,000 to 4,000 on average. While burn sites tend to be highly localized with lead contamination not spreading beyond a few meters, they present a very high dose to the community when exposure occurs. For the burners themselves, there is the additional risk of inhalation of small lead particles aerosolized when then cable is combusted.

In Montevideo, Uruguay, the BLLs in residents in the affected area before the intervention were nearly 9 µg/dL, and about 5 µg/dL afterwards. What was unique about the Montevideo case was the disparate nature of the problem. Dozens of e-waste burning locations had been detected around the city, but without portable analytical equipment, implementors were hamstrung by time consuming environmental analysis. The project loaned the municipality a portable X-ray fluorescence instrument to facilitate rapid data collection and keep up with e-waste burners. In addition, basic engineering guidance on soil replacement remediation approaches was provided by Pure Earth. This relatively small contribution resulted in extensive mapping of contamination and the environmental remediation of more than 10 sites to date.

The final study in the thesis critically evaluates the cost effectiveness of an intervention at a ULAB recycling site in the Dominican Republic in 2010. The paper bridges all three research chapters integrating innovative approaches to exposure assessment, disease burden quantification and risk mitigation. The site at the centre of the study was a former battery recycler shuttered by authorities in 1999 after a study by Kaul and Mukerjee (1999) revealed highly elevated BLLs in the surrounding community.[23,24] Following the closure of the facility, BLLs dropped by nearly half, to 32 µg/dL, but remained elevated until the remediation project began in 2009.[23,25] The project consisted of a series of engineering interventions including excavation, capping, and the construction of a large *in situ* repository. It had the effect of lowering mean BLLs in the community from 26 µg/dL to 6 µg/dL and cost approximately USD 430,684. In the study, contemporaneous environmental measurements were used to model BLLs and the attributable disease burden in DALYs was calculated. The DALYs were then projected forward over different lifespan scenarios, discounted to take into account uncertainty, and divided by the cost of the project. This process resulted in a cost (USD) per DALY estimate, which was then compared against WHO cost effectiveness thresholds.

Multiple aspects of Paper 12 were innovative. The study included granular estimates of exposure and BLL modelling, disaggregated at the household level by age and daily activity. Those estimates were then used in tandem with USEPA exposure modelling software to calculate BLLs. To improve accuracy, the model was run multiple times with changes made to key input parameters and the results statistically compared with actual BLLs from the site. The results were then used in a WHO prevalence rate calculator to determine DALYs. Here too, adjustments to default parameters were made based on more recent and robust analyses than the default assumptions in the calculator. Finally, and most significantly, the study presents the first cost effectiveness results of a remediation project in an LMIC.

5.5. The international response to lead exposure in LMICs

5.5.1. Lead-based paint

The research presented in this thesis describes widespread low-level lead poisoning of populations in LMICs with discrete contaminated areas that result in dangerously high BLLs in subsections of the population. The sources are defined as predominately industrial in nature. However due largely to the informal nature of these sources and their regular occurrence in residential areas, an historically distinct exposure scenario is presented. This characterization contrasts with the international response to the issue, which seems to both under-estimate the attributable disease burden and respond to sources of exposure that may not present a significant risk.

The Global Alliance to Eliminate Lead Paint (GAELP) is introduced in Chapter 1. This organization is co-managed by the UNEP and WHO and was formed in 2011. The GAELP works with various governments and non-governmental organizations (NGOs) around the world to encourage legislation to ban the use of lead in residential paints. Funding for the program is provided by the Strategic Approach for International Chemicals Management (SAICM) and the Global Environment Facility (GEF). SAICM is essentially a UNEP program working on a range of chemicals issues itself as well as distributing small grants to other organizations. The GEF is the largest multilateral grant maker to environmental causes, giving away USD 4.4 billion in the last cycle beginning in 2014. Operationally, these resources are distributed to UN agencies and one large NGO (World Wildlife Fund) who act as implementing agencies for various projects. Implementing agencies in-turn contract executing agencies, who in-turn can contract consultants. Governments often receive direct support from GEF grants via implementing agencies. To receive that support they must provide evidence of project co-financing, typically in-kind, with matching requirements tiered to income. GEF Executing agencies are often NGOs or other private sector actors.

The GEF's mandate is enormous, covering 6 thematic areas including climate change, biodiversity, international waters, and chemicals and waste. Its budget is also growing, from USD 1 billion in 1994, to USD 3 billion in 2000, to USD 4.4 billion in 2014. GEF funds are replenished every 4 years by member country governments. The largest historical donors are the United States (USD 1.7 billion), Germany (USD 1.5 billion) and the UK (USD 1 billion).[26] The largest single recipient in the last cycle was China (USD 1.2 billion), followed by Brazil and India, each receiving about USD 500 million. Most African countries received USD 50–100 million each.[27] Funding is allocated to programs addressing the indicators of multilateral environmental agreements (MEAs). The MEAs include, for example, the Stockholm convention on persistent organic pollutants (POPs), the Minamata convention on mercury, and the UN framework convention on climate change. Lead is not covered by any convention. Programs dealing with lead are therefore not eligible for funding through the GEF, with one exception. Lead-based paint receives a small amount of support through the POPs window. Thus, there is no concerted effort at the GEF to disallow lead programs; it is simply that no mechanism exists through which support could be provided.

Lead is commonly used in enamel paints as a pigment and to reduce corrosion and shorten drying time. Most European countries banned the use of lead in paints in the late 19th century or early 20th. Brisbane, Australia did the same in 1914.[28] The League of Nations recommended banning lead-based paint to its members in 1922.[29] The only high-income country not to ban lead-based paint at this time was the United States, which did not begin to officially phaseout the coating for another 50 years, in 1971. Despite this late regulatory action, the actual use of lead in residential paints decreased precipitously from the 1920s until the 1970s as cheaper alternatives like titanium oxide became available.[30] A 1990 study found that more than 83 % of homes in the US had lead-based paint somewhere in the building, with 12–29 % presenting an exposure risk, meaning lead was found in household dust.[31] It stands to reason, then, that lead-based paint poses an outsized risk in the United States.

However, outside of this geography there are remarkably few studies assessing the exposure risk, particularly in LMICs. Indeed, from 2010–2018 likely only three studies from two countries have been published in English identifying a statistical association between lead-based paint and elevated BLLs (Paper 1). These studies were drawn from India and Nepal and had a pooled mean of 6.67 µg/dL (SD: 5.84) in exposed children.

Significantly, the lead-based paint exposure risk in the US could not be meaningfully compared with that posed by tetraethyl lead at a societal level. The removal of lead from petrol resulted in a precipitous decline in American BLLs.[32] There was no such association between the use of lead-based paint and society-wide BLLs. Rather there have been episodic poisonings of discrete populations that tend to be poorer, more urban, and people of colour.[33]

A hazard is anything with potential to do harm. The edge of a cliff is a hazard, as is a blender, as is an icy sidewalk. Whether that hazard becomes a risk, depends on a number of mediating factors: proximity to the cliff, whether your hand is in the blender, how much time you spend outside in winter. Lead-based paint is a hazard. In the most recent survey 40 % of American homes contained lead-based paint somewhere in the building; perhaps more than any country on earth.[8] Importantly, though Americans also have some of the lowest societal BLLs measured in modern humans, currently with a median BLL below < 1 µg/dL.[34] Poorer, urban people of colour in the United States are more severely exposed not necessarily because they have more lead in their homes, but because a number of historical mediating factors place these people in substandard housing where exposure is more likely.

Paper 3, based on research executed in Jakarta, Indonesia, was the first survey of lead-based paint in housing in an LMIC of which the authors are aware. Previous studies found that lead-based paint was widely available on the market in Indonesia.[35,36] It was therefore anticipated that the coating would be prevalent in homes and preschools. However, remarkably only 2.7 % of samples taken within homes were positive for lead. Moreover, only 0.05 % of dust samples contained lead over applicable thresholds, thereby presenting a very low exposure risk.

At present only 37 % of countries have legislative limits on the use of lead in residential paints.[37] Given the low cost of transitioning manufacturers to lead-free alternatives, as well as those associated with adapting and passing local laws, the effort is an easy one to support. Preventing the use of lead-based paints in residential settings could prevent a potential time bomb of future exposure and high costs associated with mitigation. Yet the rationale is also hypothetical. It is verifiably not in response to a *de facto* crisis of exposure in LMICs, but is perhaps a projection of the American perception of their experience abroad. This is important because there really is a lead-poisoning epidemic unfolding in LMICs. It is not being controlled, it is getting worse, and the human and environmental outcomes are permanent. Moreover, its connection to lead-based paint is tenuous. Thus, while lead-based paints should, without question, be banned for use in residential settings globally, the singular focus on this relatively low source of present exposure seems misguided in the context of some of the evidence presented in this thesis.

5.5.2. The Burden of Disease

Disease burden estimates of IHME are used throughout this thesis. Nearly every study presents or critiques their estimates of the lead-attributable disease burden. The IHME was formed in 2007 in Seattle, USA with support of the Bill and Melinda Gates Foundation and has since become the global authority on burden of disease estimates. It has more than 450 staff and hundreds of collaborators around the world, including many of the world's top epidemiologists. In 2017 Bill Gates said that "[b]y using GBD, understanding where we're going year-by-year and course-correcting…gives us the best chance of saving 20 million [under-5] lives."[38] In 2018, WHO signed a memorandum of understanding with IHME to utilize their data in WHO's GBD estimates. The annual GBD estimates, published in *the Lancet*, are consistently the most highly cited articles for the publication. Their 2013 report, for instance, has been cited nearly 7,000 times.[39]

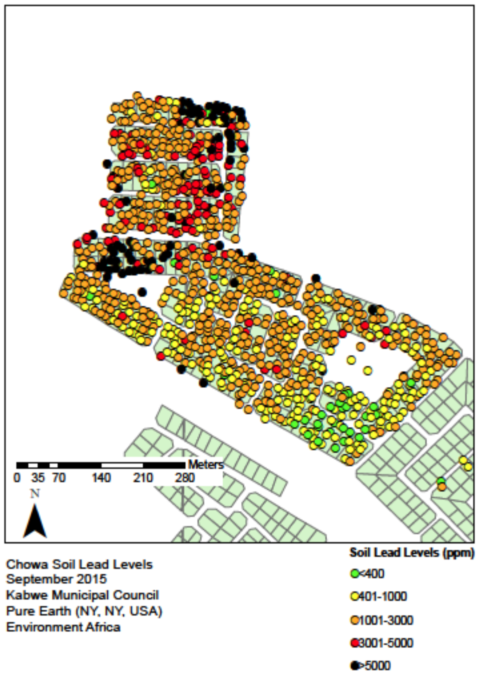

Figure 5. *Soil lead concentrations in the Chowa neighborhood of Kabwe, Zambia.*

A stated purpose of IHME's estimates is their relevance and application to policy. Their programs actively support the use of country estimates in priority setting. It is an historically unparalleled and enormous endeavour, to understand what makes people sick and how they die, and to use that information to respond on a global level. IHME is not without criticism, but they have undoubtedly vastly expanded human knowledge in this area. One small piece of their mandate, $1/79^{th}$ to be exact (IHME assesses 79 risk factors), is to assess the burden of environmental lead exposure on human health. Several papers in this thesis have tried to make the case that IHME's existing estimates are incongruous with the *de facto* situation in LMICs. Indeed, the current lead exposure model used by the organization includes a single environmental input, leaded petrol. Like efforts to deal with lead-based paint in LMICs, this focus has not been revised to account for actual sources of exposure in LMICs. One result is almost certainly an undercount of the attributable disease burden.

Age	**Effect**	**Blood lead[a] (μg/dL)**	**Bone lead[a] (g/g)**
Children	Depressed ALAD	<5	ND
Children	Neurodeve opmenta effects	<10	ND
Children	Sexua maturat on	<10	ND
Children	Depressed v tam n D	>15	ND
Children	E evated EP	>15	ND
Children	Depressed NCV	>30	ND
Children	Depressed hemog ob n	>40	ND
Children	Co c	>60	ND
Adults (elderly)	Neurobehav ora effects	>4	>30
Adults	Depressed ALAD	<5	ND
Adults	Depressed GFR	<10	>10
Adults	E evated b ood pressure	<10	>10
Adults	E evated EP (fema es)	>20	ND
Adults	Enzymur a/ prote nur a	>30	ND
Adults	Per phera neuropathy	>40	ND
Adults	Neurobehav ora effects	>40	ND
Adults	A tered thyro d hormone	>40	ND
Adults	Reduced fert ty	>40	ND
Adults	Depressed hemog ob n	>50	ND

A Concentration range associated with effect.

ALAD = δ-aminolevulinic acid dehydratase; EP = erythrocyte protoporphyrin; GFR = glomerular filtration rate; NCV = nerve conduction velocity; ND = no data

Table 2. *Blood and bone lead concentrations corresponding to adverse health effects. Adapted from ATSDR (2007).*[45]

The average BLL in the Chowa neighbourhood of Kabwe, Zambia is 50 µg/dL. There are about 500 homes here and likely more than 1,000 children. In 1975 cases of lead-induced encephalopathy resulting in convulsions and coma were common and were associated with seasonal dust recirculation, confirming that the soil was the source.[9] They continued to be common through 2012 (Paper 5). BLLs of less than 5 µg/dL result in an IQ decrement of up to 3 points in children. While the impact is log-linear, affecting children proportionately less at higher levels, increasing levels do result in increased decrement. At 50 µg/ dL there is likely a mean loss of more than 10 IQ points in children.[40,41] However, IQ loss, like most impacts of low-level lead poisoning, does not usually present clinically. Rather lead exposures manifest much later in life through anti-social behaviour, heart disease, and other adverse outcomes.[42–45] The most recent health survey of children in Kabwe found that 10 % of children suffered from clinical symptoms of lead poisoning. This is exceptionally high and implies that perhaps 2,000 children present with these symptoms each year across the city and 100 in the Chowa neighbourhood (Paper 5).

In the most recent GBD, IHME calculated only 207 lead-attributable DALYs for children (< 5 years) in Zambia as a whole. Estimates are not yet available at a more granular level, so it is not possible to calculate Kabwe's relative contribution to those 207 Zambian DALYs. Though because IHME's model does not include an input for exposures like those present here, the disease burden is very likely proportionate to Kabwe's population. Meaning that the 207 DALYs are likely more or less evenly spread across all Zambian of children (< 5 years). This would imply that IHME effectively calculates fewer than three lead-attributable DALYs for children (< 5 years) in Kabwe. Given the high rate of clinical symptoms of lead poisoning in Kabwe, it seems unlikely that such an estimate could be accurate.

In light of some of the information presented in this thesis it appears that IHME's method for calculating lead-attributable DALYs may result in significant underestimates, particularly with respect to severely impacted locations. Resolving discussions around disability weighting could mitigate part of the problem, as has been argued elsewhere.[20] But there is more likely a fundamental issue with their estimates of prevalence. Even with an exceptionally low disability weight, it seems unlikely, for instance, that Chinese children (< 5 years) have only 7,000 DALYs attributable to lead exposure.

5.6. Recommendations

Current approaches to quantify and mitigate lead exposures in LMICs do not seem to adequately take into account certain fundamental truths about the nature of those exposures. Rather they seem to implicitly rely on *a priori* assumptions based on the experience in high income countries. Shifting away from those inclinations could ultimately illuminate an undetected public health crisis associated with lead exposure in LMICs. Four recommendations stem from the research presented in this thesis:

- Blood lead testing registry. A global registry should be created and housed either with an academic institution or the WHO. Researchers collecting human BLLs should be encouraged to enter aggregate data about their studies in the registry. This could include basic descriptive statistics, sources of exposure, and analytical methods. Because academics carry out the majority of BLL analyses in LMICs such a registry would be a low hanging fruit that could vastly improve surveillance.
- Source identification. Blood lead levels remain elevated in LMICs. The sources could be better characterized. A more rigorous approach to quantifying the relative contribution of sources could improve the efficacy of intervention. Targeted studies in a small number of geographies should be supported and use a standard method. The US CDC protocol for such studies could provide the outlines of such a model.
- Biannual conference on cost effective lead mitigation. Country governments keen to execute projects should be financed to attend biannual conferences held in their regions with a primary purpose of knowledge sharing and capacity building. Mitigation methods should be presented and discussed. The conference should take an interdisciplinary approach, with experts from health, environmental and social sciences.
- Lead Convention. Ultimately any meaningful effort to deal with lead exposure may require a convention. Achieving the support of multiple countries presents an obvious hurdle. Given the likely size of the disease burden such an effort is seemingly pressing. Responsible formal industry operators should be engaged early as key partners, perhaps through the International Lead Association.

5.7. Conclusion

This thesis sought to make four broad arguments. The first related to exposures. Specifically, that the sources of exposure in LMICs are distinct from high income countries. This argument is made most robustly in Paper 1. The extensive literature review in this study provides a quantitative underpinning for the more country or site-specific assessments presented elsewhere in the chapter. Second, the thesis endeavoured to quantify lead exposures in LMICs using public health metrics. Two studies in the second chapter are worth highlighting. Paper 6 presented the first global quantification of informal ULAB facilities published in the literature. Estimates from this paper were used by the World Bank in their Country Environmental Assessments of Pakistan and Bangladesh. Because these assessments provide the basis for future work in these countries, the unprecedented inclusion of ULAB facilities is significant. The estimates were also used as the basis for the *Lancet Commission on Pollution and Health,* which has been cited nearly 400 times since being released in 2017 and received prominent media attention.[13] The second study worth noting here, Paper 8, is important because it uses a meticulous and rigorous method to point out a fundamental and significant inaccuracy in the disease burden estimates of IHME.

The third overarching argument made by the thesis is that the most severe lead exposures in LMICs can be mitigated through cost effective and locally appropriate solutions. Paper 12 addresses the cost effective side of this argument while Papers 10 and 11 describe adapting engineering and administrative controls to local contexts. Paper 10 in particular describes an innovative approach to mitigating soil contamination in residential environments that could be replicated at similar sites elsewhere.

Finally, the fourth argument posed that the international response to the issue of lead exposure in LMICs does not necessarily reflect the *de facto* nature and severity of those exposures. This argument is made implicitly throughout the thesis as sources are identified, evaluated and mitigated. It is then made most explicitly in this concluding chapter. International efforts to mitigate lead exposure in LMICs tend to focus on the potential risk posed by lead-based paint, despite robust evidence that the most significant sources of exposure are industrial in nature. Disease burden estimates meanwhile significantly underestimate the issue, likely due in part to a lack of adequate environmental inputs.

This thesis sought to begin to quantify an emerging public health risk attributable to lead exposure in LMICs and propose methods of mitigation. The toxicant in this case is well studied and understood, though the sources of exposure remain poorly characterized in LMICs. In the absence of more detailed information there has been an inclination to understand the issue through the lens of rich country experience. The consequence is an observation bias that results in missing a problem hiding in plain sight. Cost effective approaches exist to begin to deal with the problem. The issue becomes particularly pressing given lead's persistence in the environment and increasing usage. In this context, the research in this thesis might illicit a sense of urgency to act to protect some of the more vulnerable among us.

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Appendices

Appendix A – Paper 7 original version published in Spanish
Appendix B – Using existing field X-ray fluorescence data to calibrate multi-spectral images of contaminated sites for semi-quantitative remote sensing of lead contamination in soil: The test-case of legacy mining in Kabwe, Zambia
Appendix C – Remote sensing workshop concept note
Appendix D – Link to supplemental materials

Pérdida de coeficiente intelectual en hijos de alfareros mexicanos

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Intelligence quotient loss in Mexican pottery artisan's children

Background: In Mexico, artisans frequently use lead oxide or greta in order to produce utensils, which are destined to preparation and storage of food and drinks. Additionally, the risk of lead poisoning of artisans and their families is greater than in general population, and within these families, children are the most suscept ble to lead poisoning. The aim of this study was to estimate IQ loss in Mexican children from potter families exposed to lead.

Methods: Lead concentrations in soil were determined in 19 potter's homes that functioned as pottery workshops in seven Mexican states between 2009 and 2012. This information was used to estimate blood lead levels through the integrated exposure uptake biokinetic (IEUBK) model. The loss of IQ points was then estimated according to the Lanphear and Schwartz models.

Results: The mean lead concentration found in the workshops' soil was 1098.4 ppm. Blood lead levels estimated in children under 8 years old were 26.4 µg/dL and the loss of IQ points comprised from 7.13 to 8.84 points depending on the model.

Conclusions: It is possible that 11 children from families of artisans in Mexico may be losing between 7.13 to 8.84 IQ points, due to lead exposure in their houses-workshops. This loss in IQ points could have important health, economic and social impacts.

Keywords	Palabras clave
Ceramics	Cerámica
Lead	Plomo
Occupational exposure	Exposición ocupacional
Child	Niño
Environmental health	Salud ambiental

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Los orígenes y usos de esmaltes con contenido de óxido de plomo para vidriar la cerámica se remontan a la antigua Grecia y han seguido relativamente sin cambios hasta hace pocos años.[1] Aunque su uso ha disminuido a nivel mundial, actualmente son empleados para producir artesanías en varios países en Latinoamérica,[2] el Norte de África y Medio Oriente.[1] En América, los españoles introdujeron el vidriado con plomo en el siglo XVI, lo que ofreció una alternativa a la cerámica bruñida.[3] A casi cinco siglos de distancia, el vidriado con plomo se usa en al menos 20 estados de México.[2] La Administración de Medicamentos y Alimentos de los Estados Unidos (Food Drug Administration o FDA por sus siglas en inglés) determinó los niveles máximos permitidos de plomo en cerámica, en un rango de 0.5 a 3 µg/mL, según su uso.[4] En México, la *NOM-004-SSA1-2013* establece que se debe evitar el uso de plomo para producir alfarería vidriada utilitaria.[5] Sin embargo, tanto en el sector formal como en el informal, los alfareros continúan usando frecuentemente el óxido de plomo, o greta,[6] para producir utensilios, los cuales se destinan a la preparación y el almacenamiento de alimentos y bebidas. Estos utensilios son usados en todo el país, predominantemente entre la población pobre; sin embargo, por cuestiones de tradición, también son empleados por las clases media y alta mexicanas.

La mayoría de la alfarería en México se produce en hornos de baja temperatura (entre 850 y 1000 °C). Por lo tanto, estos no alcanzan los grados de fusión para formar silicatos de plomo insolubles que sean resistentes al ataque químico de alimentos y líquidos ácidos. Bajo estas condiciones, el plomo es biodisponible y puede ser fácilmente liberado en alimentos y bebidas al contacto con el vidriado.[7] Los alimentos ácidos como el jitomate, el café, el chile y el jugo de limón aceleran el proceso de lixiviación del plomo.[8] En 1991, Ávila *et al.*[9] determinaron que 58% del riesgo atribuible a niveles de plomo en la sangre de las mujeres mexicanas se debió al uso de la alfarería vidriada con plomo para preparar, servir y almacenar comida y bebidas. En una revision hecha por Caravanos *et al.* (2014) se estimaron niveles de plomo en

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Resumen

Introducción: en México, los alfareros continúan usando frecuentemente el óxido de plomo o greta para producir utensilios, los cuales se destinan a la preparación y almacenamiento de alimentos y bebidas. Adicionalmente, el riesgo de intoxicación por plomo de los alfareros y sus familias es mayor que en la población general, y en tales familias, los niños son los más susceptibles a la intoxicación por plomo. El objetivo del estudio fue estimar la pérdida de puntos de coeficiente intelectual (CI) en hijos de alfareros mexicanos expuestos al plomo.

Métodos: durante el periodo de 2009 a 2012 se determinaron las concentraciones de plomo en suelo de 19 casas-talleres de alfareros en siete estados mexicanos. Esta información se utilizó para estimar el nivel de plomo en sangre, por medio del modelo biocinético integrado de absorción por exposición (IEUBK, por sus siglas en inglés). Posteriormente, se calcularon los puntos perdidos de CI según los modelos de Schwartz y Lanphear.

Resultados: la concentración promedio de plomo en suelo fue de 1098.4 ppm. Se estimó un nivel de plomo en sangre de 26.4 µg/dL para menores de 8 años. La pérdida de puntos de CI estimada fue 7.13 y 8.84, según el modelo utilizado.

Conclusión: es posible que al menos 11 niños de familias alfareras mexicanas estén perdiendo entre 7.13 y 8.84 puntos de CI, debido a la exposición al plomo en sus casas-talleres, lo que supone importantes impactos económicos, sociales y de salud.

sangre de 8.85 µg/dL en áreas urbanas y 22.24 µg/dL en áreas rurales, y se identificó que la minería, la fundición y, en mayor medida, la alfarería eran fuentes principales de exposición.[10]

Otros estudios han documentado el impacto en la salud de adultos y niños en México, debido a la fabricación de alfarería vidriada[7] y han reportado concentraciones superiores a 20 µg/dL de plomo en sangre en familias de alfareros en Tzintzuntzan, Michoacán,[11] y de más de 30 µg/dL en una comunidad alfarera del estado de Veracruz.[12] Los niños son más susceptibles a la intoxicación por plomo que los adultos.[13] Adicionalmente, el riesgo de intoxicación por plomo de los alfareros y sus familias es mayor debido a la exposición ambiental al plomo, a los alimentos cocinados en alfarería vidriada con plomo, la malnutrición, el uso del espacio laboral para la vivienda, las malas prácticas en el manejo del óxido de plomo y al hecho de que el plomo se convierte en una fuente endógena de exposición.[12,14,15]

Asimismo, la relación entre el nivel de plomo en sangre y las alteraciones en el coeficiente intelectual ha sido ampliamente estudiada.[16] El metaanálisis de Schwartz[17] encontró que el plomo afecta las neurotransmisiones y se asocia con un decremento de 2.6 puntos de coeficiente intelectual (CI) con niveles de entre 10 y 20 µg/dL de plomo en sangre.

Varios autores han estudiado la asociación existente entre los niveles de plomo en suelo y en sangre,[15,18,19,20,21] debido a la exposición ambiental al tóxico.

De la literatura revisada no se encontraron estudios relativos al impacto del plomo en el CI de la comunidad infantil alfarera, la cual representa la población más vulnerable al uso de plomo en la alfarería del barro vidriado. Por ello, el presente estudio estimó la pérdida de puntos de CI, a partir de niveles de plomo en suelo para niños que viven en hogares de alfareros, la cual es una variable más que se debe considerar en la carga de la enfermedad de la población expuesta ambiental y ocupacionalmente al plomo.

Métodos

Se llevó a cabo un estudio transversal entre noviembre de 2009 y marzo de 2012 en el cual Pure Earth (antes Blacksmith Institute) realizó mediciones de plomo en suelo en 25 viviendas de alfareros, en siete estados de México (Colima, Estado de México, Jalisco, Michoacán, Oaxaca, Puebla y Tlaxcala), como parte de las actividades de identificación y clasificación de sitios contaminados, con la intención de remediar el problema.[22,23] Se contactó a los jefes de familia que trabajaron previamente en programas del Fondo Nacional para el Fomento de las Artesanías (FONART) y, por medio de dicha institución y Pure Earth, se les invitó a participar. La selección de la muestra fue por conveniencia, ya que solo participaron aquellos alfareros que lo consintieron y que estaban interesados en conocer los niveles de plomo en el suelo de sus casas-talleres. De los 25 talleres que participaron previamente con FONART, 19 fueron seleccionados para el estudio, debido a que las actividades de aplicación y traslado del óxido de plomo (greta) dentro de esas viviendas, se llevaban a cabo en un lugar cerrado para descartar variables atmosféricas que disminuyeran significativamente los niveles de plomo en el suelo exterior.

La totalidad de las familias que participaron en el estudio pertenecían al estrato social D y subsistían principalmente de la producción de alfarería. En todos los casos se observó que los espacios para la producción son compartidos con áreas de vivienda, por lo que el área donde se aplica el óxido de plomo es de uso común para la familia.

El proceso para determinar la pérdida de puntos de CI como resultado de la exposición al plomo en los talleres de los alfareros incluyó la evaluación de la concentración del plomo en el suelo en los talleres de alfarería, el cálculo de los niveles de plomo en sangre (con base en el modelo biocinético integrado de absorción por exposición —IEUBK, por sus siglas en inglés—) y, por último, la estimación de pérdida de puntos del coeficiente intelectual.[21,24]

Niveles de plomo ambiental

Esta investigación analizó el suelo dentro y alrededor de las casas-talleres de cada familia de artesanos, y se tomaron en promedio 10 muestras en cada taller. Un instrumento portátil de fluorescencia de rayos X (XRF, por sus siglas en inglés) fue usado para analizar la concentración de plomo del suelo superficial (Equipo: INNOV-X Alpha 6500, Woburn, Massachusetts). Este equipo mide de forma instantánea 21 metales en matrices sólidas y semisólidas de suelo.

A los propietarios de cada uno de los talleres participantes se les informó sobre los riesgos asociados a la exposición ambiental y ocupacional al plomo, así como sobre las estrategias para minimizar esta exposición. Los resultados fueron dados a conocer a los dueños de los talleres durante las evaluaciones, debido a que el tiempo de respuesta del XRF es de 15 a 30 segundos.

Como resultado de lo anterior, se aconsejó a los dueños dejar de usar óxido de plomo en todos los casos en los que existieron lecturas de plomo en suelo superiores a 400 ppm, nivel recomendado por la Agencia de Protección Ambiental de Estados Unidos (EPA, por sus siglas en inglés).

Por otra parte, se eliminaron seis casas-talleres del análisis, debido a que sus operaciones se llevaban a cabo exclusivamente en exteriores, por lo que el suelo estaba sujeto a la dispersión o arrastre del plomo por influencias climáticas. En consecuencia, el análisis de carga de enfermedad se basó en 149 lecturas de 19 talleres de alfarería (cuadro I).

Niveles de plomo estimados

Se utilizó el modelo biocinético integrado de absorción por exposición, en la versión IEUBK win v1.1 build 11. Este modelo tóxico-cinético se puede obtener de manera gratuita y es ampliamente usado para predecir los niveles de plomo en sangre de niños de entre seis meses y siete años de edad, con base en los niveles ambientales de plomo.[25,26] El modelo IEUBK fue desarrollado por la EPA para estimar el efecto de las emisiones de las fundidoras de plomo y para evaluar las actividades de remediación en los sitios del programa Superfund.[27] Es usado en diversas aplicaciones y en la investigación científica y, por lo tanto, ha sido revisado por expertos.

El modelo IEUBK utiliza factores biocinéticos estándar de absorción como valores predeterminados, pero permite editar los valores de absorción de plomo de fuentes de aire, suelo, agua y alimentación, así como la cantidad de agua ingerida, el volumen de aire inhalado y la absorción gastrointestinal por grupos de edad.[28] También permite introducir el nivel de plomo que hay en la sangre de la madre. En este caso, se usó el modelo empleando valores predeterminados de cero (0) para aire, agua, alimentos y sangre

Cuadro I Casas-talleres de cerámica artesanal participantes (*n* = 19), estados de la república y población (de noviembre de 2009 a marzo de 2012)

Estado	Municipio	Población[2]	Talleres en la población (y estado)[2]	Talleres estudiados
Colima	Colima	24 939	5 (5)	1
Jalisco	El Grullo	21 825	10 (597)	2
México	San José del Arenal	875	(1189)	2
México	Tecomatepec	1549	(1189)	1
Michoacán	Capula	5086	600 (3435)	4
Michoacán	Santa Fé de la Laguna	4046	1000 (3435)	1
Oaxaca	Oaxaca	258 008	(2500)	1
Oaxaca	Santa María Atzompa	16 855	(2500)	2
Puebla	Zautla	18 567	1837 (1931)	2
Tlaxcala	Tenexyecac	2863	(480)	3
	Total	339 982	10 586 (México)	19

materna. Adicionalmente, se asumió que las concentraciones de plomo en el suelo exterior y del polvo en el interior eran los mismos. Los niveles predeterminados de ingesta de suelo, como 100 mg/día en el modelo IEUBK, fueron elevados a 400 mg/día, pues se consideró la mayor exposición a plomo en los lugares visitados, donde se observó, en todos los casos, un alto contenido de polvo, debido a pisos de tierra, caminos de terracería y banquetas sin pavimentar, lo cual aumenta la exposición a tóxicos.[29]

Para determinar la población de niños afectados en los 19 talleres, se asumió que todos los hogares estaban compuestos por cuatro personas, que es el número promedio de integrantes por familia en México. Adicionalmente, con los datos del Instituto Nacional de Estadística y Geografía (INEGI) de 2010, se estimó que 15.2% de la población tiene entre 0 y 7 años de edad, lo que significa que alrededor de 11 niños entre 0 y 7 años viven en los 19 talleres.[30] Debido a la naturaleza del estudio y como un primer y único acercamiento con los alfareros, los jefes de familia no fueron cuestionados sobre el número de niños. La experiencia en campo de los investigadores indica que el estimado con datos del INEGI se aproxima a la realidad de forma conservadora y, por lo tanto, sirve como referente confiable.

Estimación de pérdida de puntos de CI

Una vez que se estimó el nivel de plomo en sangre, se procedió a calcular su impacto sobre el CI. Para ello, se utilizaron dos metaanálisis: el estudio de Schwartz de 1994[17] y el de Lanphear *et al.* de 2005.[31]

El trabajo de Schwartz se basó en ocho estudios de investigación que monitorearon los niveles de plomo de 2702 niños de distintos estratos socioeconómicos y con niveles de plomo en sangre entre 10 y 20 μg/dL. El metaanálisis encontró que un incremento de 10 a 20 μg/dL de plomo en sangre se traduce en una pérdida de 2.57 puntos de CI. Con esta investigación, se determinó un modelo lineal para predecir el impacto en dicho rango, por lo que la pérdida de CI se puede calcular multiplicando el nivel de plomo en sangre por 0.257.[16]

Para estimar el efecto del plomo sobre el CI en niños con menos de 10 μg/dL de plomo en sangre, el metaanálisis de Lanphear *et al.* analizó estudios prospectivos que incluían a 1333 niños.[31] La investigación tuvo en cuenta múltiples variables que podían confundir la relación de la exposición al plomo y el CI, como el género del menor, el orden de nacimiento, el peso al nacer, la educación materna, la edad materna, el estado civil, la exposición prenatal al alcohol, la exposición prenatal al tabaco, así como un índice que refleja la calidad y cantidad de estimulación emocional y cognitiva en el hogar, medido por el programa de Observación del Hogar para la Medición del Ambiente (HOME, por sus siglas en inglés). Se identificó que un modelo logarítmico-lineal predice el impacto en el CI. La ecuación de este modelo es pérdida de CI = beta * ln (nivel concurrente de plomo en sangre / punto de corte), con una beta de −2.70 y un punto de corte de 1 μg/dL. El nivel de plomo en sangre estimado para cada casa-taller fue sustituido en ambas ecuaciones para predecir la disminución en el CI.[31]

Resultados

Niveles de plomo en el suelo de casas-talleres de artesanos

Se tomaron 149 lecturas en 19 talleres. El nivel de plomo promedio (media geométrica) en el suelo dentro de las casas-talleres de los artesanos fue de 1098.4 ppm, con un intervalo de confianza (IC) al 95% de 898-1343.5 ppm (cuadro II). Más de 50% de los talleres tuvieron lecturas máximas que superaron las 5000 ppm (cuadro II). En los casos en los que las concentraciones de plomo identificadas estuvieron por encima de los niveles recomendados por la EPA (400 ppm), se les aconsejó a los propietarios detener inmediatamente el uso de óxido de plomo.

Además de plomo, el equipo XRF también determina concentraciones de otros 38 elementos. De estos, ninguna de las lecturas arrojó niveles elevados de arsénico, cadmio, mercurio o de ningún otro tóxico relevante.

Estimación de niveles de plomo en la sangre

Los niveles de plomo en la sangre se calcularon con el modelo IEUBK, con base en los valores preestablecidos de cero (0) para aire, agua, alimentos y sangre materna y utilizando la media geométrica total de los niveles de plomo en suelo de 1098.4 ppm (cuadro II), con el ya mencionado IC al 95% de 898-1343.5 ppm. Esto arrojó un nivel promedio de plomo en sangre de 26.4 μg/dL en niños menores de 8 años, con un IC al 95% de 23.2-29.8 μg/dL. Este resultado promedio es cinco veces mayor que el recomendado por el Centro de Control de Enfermedades (CDC, por sus siglas en inglés: Centers for Disease Control and Prevention, de los Estados Unidos (5 μg/dL).[32]

Estimación de la pérdida de CI

Con los modelos de Lanphear *et al.* y Swhwartz se estimó que los 11 niños menores de ocho años de edad, que hipotéticamente estarían viviendo en las 19 casas-talleres, probablemente tendrían una pérdida

Cuadro II Niveles de plomo en suelo (μg/g o ppm) y número de lecturas por casas-talleres de alfarería de barro vidriado con plomo (2009-2012)

No. de la casa-taller	Lecturas de plomo en suelo	Media geométrica (ppm)	IC95% (ppm)	Mediana (ppm)	Cuartiles 25 y 75 (ppm)
1	10	949.2	596.5-1510.3	847.5	415, 4485
2	8	2055.4	733.7-5757.6	847.5	258, 4485
3	22	895.3	551.9-1452.5	892.5	200, 13928
4	12	1224.4	649-2310	1151.5	266, 7536
5	8	3707.4	1664.6-8256.9	2788	1014, 26955
6	4	1100.9	119.9-10112	738	203, 25806
7	9	1415.8	833.1-2406.1	1192	486, 5269
8	6	3133.6	800.9-12259.8	1961	327, 25141
9	4	781.4	347.4-1757.5	733	322, 2274
10	9	1396.6	657.5-2966.4	1148	435, 8392
11	5	2330.5	858.2-6328.6	3672	379.5, 6350
12	3	643.5	332.8-1244.4	626	364.5, 1168
13	7	428.5	232.6-789.6	319	187, 1970
14	5	2241.5	306.3-16403.1	532	291, 27176
15	3	502.5	412.5-612.2	523.5	415, 584
16	18	447.1	292.5-683.5	378.5	120, 6227
17	3	302.6	97.9-936.0	212	140, 934
18	6	1494.5	732-3051.2	1872	495, 3694
19	7	1448.6	863.4-2430.4	1548	596, 4767
Total	149	1098.4	898-1343.5	934	120, 34385

IC95% = intervalo de confianza al 95%

entre 7.13 (Schwartz) y 8.84 (Lanphear *et al.*) puntos de CI como resultado de este tipo de exposición al plomo (cuadro III).

Discusión

Durante las últimas décadas, varios estudios han mostrado altos niveles de plomo en sangre en comunidades de alfareros en Jalisco, Michoacán y Veracruz.[11,12,14] Estos estudios han señalado que el incremento de los niveles de plomo en sangre está relacionado con tener el taller en el interior de la casa, cocinar en barro vidriado, ser niño, ser mujer y tener piso de tierra. El presente estudio estima una pérdida en promedio de 7.13 a 8.84 puntos de CI para hijos de alfareros de 19 talleres de alfarería.

En México existen 10 586 talleres registrados en el censo de alfareros del FONART.[2]

Con base en el tamaño promedio de una familia mexicana, se puede estimar que hay al menos 42 344 personas que viven en casas-talleres de alfareros en México. En caso de que las condiciones de este estudio fueran aplicables a todos los talleres registrados ante FONART, se podría estimar que aproximadamente 6436 niños menores de ocho años (15.2% del total de la población)[30] estarían en riesgo de disminuir su coeficiente intelectual como consecuencia de la exposición a niveles elevados de plomo en sus casas-talleres. Dependiendo de la elección del modelo, esta pérdida podría variar entre 7.13 y 8.84 puntos en promedio para cada niño.

Por otro lado, esa disminución puede tener un impacto social negativo. En 1998,[33] Gottfredson describió el impacto de la pérdida de puntos del CI como la circunstancia más profundamente implicada con los resultados sociales adversos (la pobreza, la falta de bienestar, la ilegitimidad y el fracaso escolar).

Los daños a la salud provocados por la exposición infantil al plomo también repercuten negativamente en la economía. El impacto se puede cuantificar a partir de los costos de salud pública, de la necesidad de recursos educativos adicionales y de la baja productividad debido al decremento en el CI. Los alfareros del barro vidriado presentan los niveles más altos de plomo en sangre, en comparación con otras poblaciones que están expuestas al plomo ocupacional y ambientalmente,[34,35] lo que incrementa el riesgo a su salud.

Cuadro III Niveles de plomo en suelo, niveles de plomo en la sangre y déficits del coeficiente intelectual en los niños que viven en las casas talleres de alfarería del barro vidriado con plomo

Niveles de plomo en suelo (ppm)	Niveles de plomo en la sangre µg/dL (< 8 años de edad)	Déficits de CI estimados (Schwartz)	Déficits de CI estimados (Lanphear)
1098.4 (media geométrica)	26.4	−7.13	−8.84
898 (95% más bajo)	23.2	−6.26	−8.49
1343.5 (95% superior)	29.8	−8.05	− 9.17

Un estudio realizado en 2002 por Landrigan *et al.*[36] examinó los costos financieros de cuatro enfermedades asociadas con la exposición de los niños a diferentes factores ambientales: la intoxicación por plomo, cáncer, asma y trastornos del desarrollo. El estudio encontró que la exposición ambiental relacionada con estas enfermedades genera aproximadamente 2.8% del total de los costos anuales de atención de salud en los Estados Unidos, lo que equivale a un costo de 54.9 mil millones de dólares.

Este estudio es importante para entender el riesgo que presenta el uso del óxido de plomo para los alfareros y sus familias.

Limitaciones del estudio

Este estudio está basado en la evaluación de niveles de plomo en suelo de 19 talleres y corresponde a una muestra por conveniencia, por lo que no es una muestra representativa del total de talleres que existen en los estados participantes, ni de todo México. Por lo tanto, los resultados no se pueden generalizar a los talleres de alfarería de todo el país.

Los investigadores notaron varios elementos que pueden influir en el riesgo a la exposición al plomo, los cuales pueden ser muy diferentes entre los talleres de los artesanos y, por lo tanto, las estimaciones realizadas en este estudio podrían variar. Estos elementos incluyen el número de habitantes, el porcentaje de niños en cada casa-taller, el volumen de producción, la distribución del espacio laboral-residencial, los materiales de los pisos de la casa-taller, las condiciones hidrográficas, los roles de trabajo, las prácticas de manejo de la greta, los hábitos de higiene, el número de niños que no están en el círculo familiar con acceso a los sitios contaminados y el uso de alfarería vidriada con greta para cocinar. Es probable que los resultados del estudio subestimen la magnitud y el alcance del problema, ya que los cálculos solo incluyen una vía de exposición al plomo (suelo) y no tuvieron en cuenta otras vías de exposición importantes, como la ingesta de alimentos contaminados con plomo y la fuente endógena ósea de la madre al producto *in utero*. Adicionalmente, este estudio analizó una sola determinación de las concentraciones de plomo en suelo, lo cual no es representativo de la exposición a lo largo de la vida en cada casa-taller y no analizó otras variables que pudieran confundir la relación de la exposición al plomo con la pérdida de puntos del CI, como las edades de los niños, la escolaridad, el estado nutricional, entre otras.

Conclusión

El presente estudio se enfocó en los puntos del coeficiente intelectual perdidos en hijos de alfareros y puede servir como una línea base para profundizar en el conocimiento del impacto del plomo en el CI, debido al oficio de la alfarería. Por otro lado, el plomo puede tener otros efectos adversos en múltiples sistemas del cuerpo humano, como el neurológico, el hematológico, el gastrointestinal, el cardiovascular y el renal.[37] Por ello los hijos de los alfareros del barro vidriado con plomo son una población en gran riesgo, por lo que debería desarrollarse con ellos un programa de vigilancia epidemiológica para cuidar de su salud, junto con la de los mismos alfareros.

Es necesario mejorar el conocimiento que se tiene sobre el costo social, económico y de salud del vidriado a base de plomo en México, aplicar las normas vigentes para eliminar del mercado este tipo de producto y concientizar a los consumidores sobre la importancia de comprar y utilizar únicamente alfarería libre de plomo; por lo tanto, es necesario que una política gubernamental permanente y efectiva logre sustituir la greta con plomo por sales no tóxicas.

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Using existing field X-ray fluorescence data to calibrate multi-spectral images of contaminated sites for semi-quantitative remote sensing of lead contamination in soil: The test-case of legacy mining in Kabwe, Zambia

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Abstract

We use existing field X-ray fluorometry (XRF) data collected by the non-governmental organization (NGO) Pure Earth (formerly Blacksmith Institute) in Kabwe, Zambia to derive a spectral index, which we apply to the semi-quantitative remote sensing of lead (Pb) contamination in soil as a novel image classification method. Kabwe, Zambia, a city heavily contaminated by nearly a century of unregulated Pb and zinc (Zn) mining, is a unique test-case for the applicability of this spectral index to predicting areas of high Pb contamination from satellite imagery. XRF measurements provide an immediate quantitative measure of Pb contamination in soil, which we used to calibrate imaging spectrometer data from the WorldView-2 (WV-2) instrument. Comparing image pixels associated with field XRF measurements of high Pb contamination ($[Pb]_{soil} \geq 1001$ ppm) with those associated with lower Pb contamination ($[Pb]_{soil} \leq 1000$ ppm) revealed a separable spectral trend that we applied as a spectral index to classify an image collected over Kabwe, Zambia in May 2014. Images classified using spectral index values correctly predict previously identified very high and low Pb areas with a 63.24% success rate for the most highly contaminated areas. The spectral index-based classification improves the predictive quality of image classification for Pb contamination by a factor of 1.87 compared to standard Spectral Angle Mapper (SAM) classification (33.82% effective) and by a factor of 2.87 compared to minimum distance classification (22.06% effective). The spectral index-derived classification method that we present is a first step toward fully quantitative methods for the remote detection of Pb contamination in soil. With further development, it may be possible to use field XRF or other quantitative data to provide real-time, *in-situ* calibration for imaging spectrometer data and produce a fully quantitative remote sensing method for the global detection and monitoring of Pb contamination.

1. Introduction and Background

1.1 Mine waste and Pb contamination effects on human health and the environment

Humans have exploited the earth for valuable materials since the dawn of civilization, but the environmental and human health impacts of mining activities have only recently begun to be fully grasped. Volumetrically, mine wastes are one of the world's largest waste streams (Hudson-Edwards et al., 2011). Mine wastes are also complex, containing both a huge amount of raw waste material (mine waste) and non-valuable processed waste (slag). Mine wastes also contain high concentrations of elements, such as lead (Pb), that can have severe negative effects on human health and the environment (Plumlee and Morman, 2011).

Many studies have shown that mining, smelting, or ongoing exposure to Pb-bearing ores leads to elevated blood-lead levels (BLL) among many of the people working with the ores or living nearby, especially children. (Gulson et al., 1994; Landrigan et al., 1975; Mackay et al., 2013; Yankel et al., 1977; Zhang et al., 2012). The risks are especially severe in developing countries (Nriagu, 1988). The Institute for Health Metrics and Evaluation (IHME) found in their Global Burden of Diseases, Injuries, and Risk Factors Study 2015 that 15.7% of men and 14.5% of women were exposed to Pb (Forouzanfar et al., 2016). Additional studies focusing on Pb contaminated sites, in particular used lead-acid battery (ULAB) recycling, have found that global estimates may be underestimating the burden of disease from environmental Pb exposure in low- and middle-income countries (LMICs). As many as 16.8 million people may be exposed to Pb at informal ULAB recycling sites alone, in more than 90 LMICs (Ericson et al., 2016).

In terms of its neurocognitive impacts on humans and its exposure mechanisms, Pb is perhaps the best understood metal toxicant commonly found in the environment (e.g., Filippelli et al., 2015). Symptoms of exposure range from the acute effects of blindness, renal failure, gastric distress, and death to chronic effects on neurodevelopment and behavior as well as, anemia and kidney disease (Sanders et al., 2009). Previously, the most important global sources of trace metals in soils, including Pb, were estimated as mine tailings, smelter emissions and atmospheric fallout (Nriagu and Pacyna, 1988). However, isotopic studies have proven that the largest historical source of dispersion and exposure for lead globally is actually tetraethyl Pb from leaded gasoline or petrol (Bollhöfer and Rosman, 2001, 2000; Flegal et al., 1984). However, proximity to Pb mining or smelting, especially artisanal and informal mining, is still a major source of risk for morbidity and mortality from Pb exposure, especially in LMICs

(Forouzanfar et al., 2016). Acute Pb poisoning episodes have been documented in Dakar, Senegal as a result of informal ULAB recycling (Haefliger et al., 2009) and in Nigeria as a result of artisanal gold mining (Lo et al., 2012). Children living near mines and smelters are especially at risk due to their relatively higher rates of soil/dust ingestion. For example, in one study of children living near a copper smelter in Mexico, it was estimated that 87% of the total Pb in their blood was accumulated via the soil/dust pathway, which involves the ingestion of contaminated soil and dust as a result of playing in or on contaminated soil and putting their hands or toys in their mouths (Carrizales et al., 2006). However, a later study at the Bunker Hill Mining and Metallurgical complex in Idaho found that actual ingestion rates at Bunker Hill were lower than those usually used in human health risk assessment and that likely a substantial source of exposure comes from beyond the immediate home environment (von Lindern et al., 2016). Thankfully, simple, low cost environmental remediation interventions have been shown to significantly improve human health outcomes in Pb poisoned areas (Ericson et al., 2018; Tirima et al., 2016).

Despite the risk of devastating health effects and the possibilities of low-cost remediation, many developing countries lack the resources to adequately protect their populations (especially children) from mine wastes and heavy metal contamination in soil (Lessler, 1988). The work presented here is a novel approach toward remote monitoring of Pb contamination using satellite imaging spectrometer data directly calibrated by field X-ray Fluorometry (XRF) to a semi-quantitative method for Pb detection. This approach, which does not require laboratory analyses, may provide a cheaper and quicker method for ongoing Pb contamination monitoring and exposure prevention in vulnerable populations.

1.2 Study site: Kabwe, Zambia

The post-colonial landscape of sub-Saharan Africa contains widespread environmental issues. Heavy metal contamination is one serious environmental issue that poses a major impediment to development. Pb exposure is a major public health concern in nearly all African countries, with poverty being a key underlying factor preventing decisive action and prevention (WHO Regional Office for Africa, 2015). Kabwe, Zambia, the provincial capital of Central Province located about 110 km north of the national capital of Lusaka (Figure 1) is one of the world's most contaminated places and is a classic example of widespread Pb contamination exposure as a result of poor mining regulation and historical negligence (Bose-O'Reilly et al., 2017; Caravanos et al., 2014; Walsh, 2007).

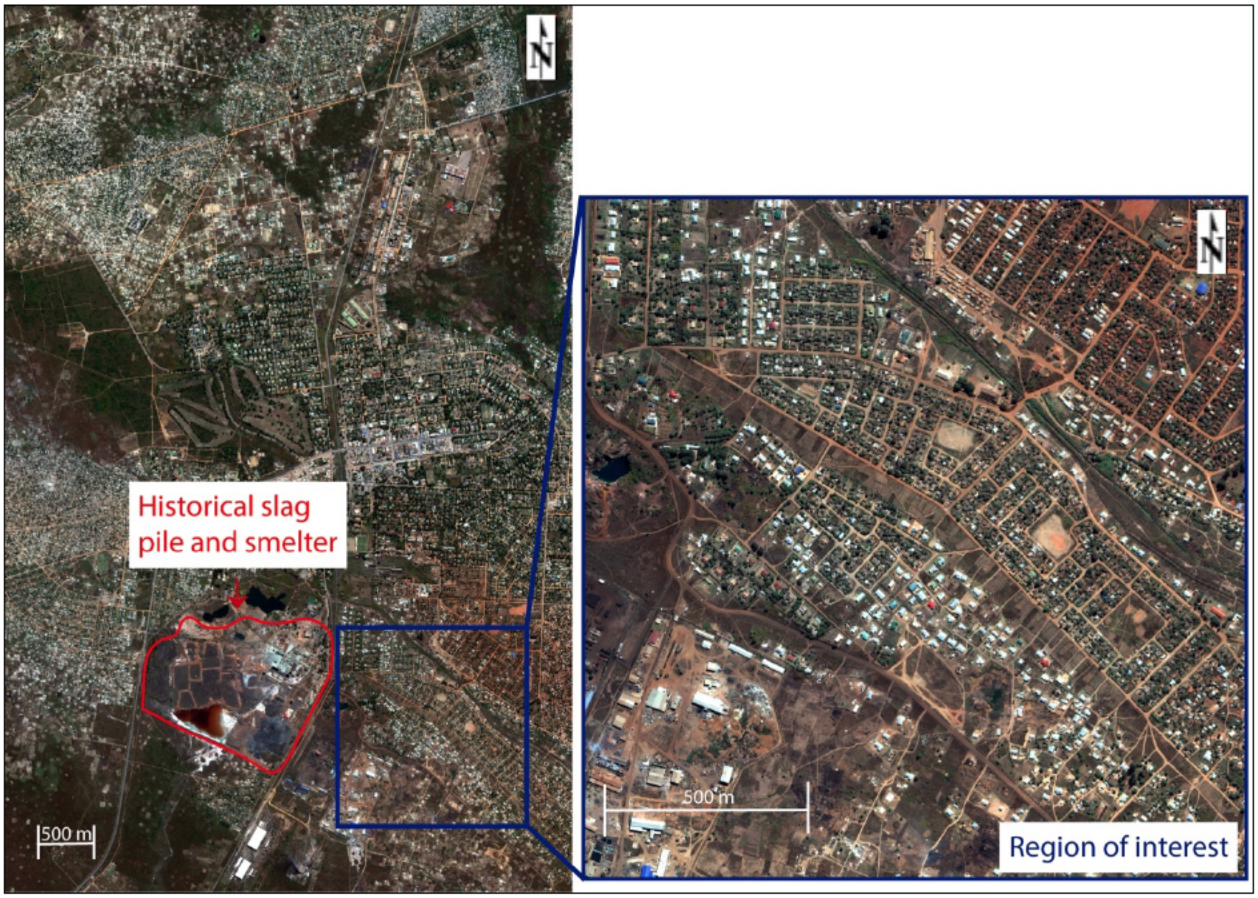

Figure 1. Orientation map of Kabwe, Zambia (654923.2, 8403030.0) showing the historical mine and slag pile location (red outline, labeled) and the area of one neighborhood in Chowa township (656069.5, 8400476.5) where 1013 field X-ray fluorescence (XRF) measurements were taken in August of 2014 to survey the extent of soil Pb contamination (blue outline, labeled). Center coordinates in WGS-84, UTM zone 35S (easting, northing).

The Broken Hill mine in Kabwe was Zambia's principal producer of Pb and zinc (Zn) from the early 1900's through 1994. During 88 years of underground mining, the Broken Hill mine produced 1.8 megatons (Mt) of Zn, 0.8 Mt of Pb, and minor amounts of silver (Ag), vanadium oxide and cadmium (Cd) (Kamona and Friedrich, 2007). Unregulated mining and smelting of Pb, Zn and Cd and the production of sulfuric acid were the original sources of pollution in Kabwe, Zambia (Sorensen et al., 2015; Tembo et al., 2006). Today, the mining site continues to be a major source of pollution despite the fact that it has been closed since 1994. Soils are contaminated with toxic metals over a substantial area (Bose-O'Reilly et al., 2017;

Nakayama et al., 2011; Tembo et al., 2006). An estimated 6 million metric tons (bmrplc.com) of mine wastes have been left on the surface (Figure 1, red outline). Smelting is still carried out in an official capacity by BMR Group (bmrplc.com), with the slag processed outside of Kabwe. However, many residents also informally mine the slag pile (Carrington, 2017).

The BMR Group previously commissioned a report of the mineral resources estimates for multiple tailings sources in the center of Kabwe, Zambia and found a combined Pb level of 15.92 wt%. Away from the tailings piles, Pure Earth has measured soils by field X-ray fluorometry (XRF) with Pb concentrations above 30,000 ppm (3%). The profound environmental contamination has led to detected BLL in children between 60 and 120 μg/dL (Nakayama et al., 2013) and more recent maximum BLL values as high as 427.8 μg/dL (Bose-O'Reilly et al., 2017). These high-Pb soils and BLL resulted in Kabwe repeatedly being named among the top ten most polluted places in the world (Carrington, 2017; Walsh, 2007). As a result of the large amount of surface Pb contamination and its widespread distribution throughout the area, Kabwe presents a useful study site for developing image analysis methods to detect Pb. There is both a clear source of contamination and a variety of affected groundcover types (e.g., bare soil, cultivated fields, residential areas, etc.). In addition, Kabwe's status as a highly contaminated town has led to several remediation efforts both past and ongoing (Caravanos et al., 2014; Walsh, 2007), which can provide sources of ground-truth data.

Kabwe also presents a compelling study site from a geochemical perspective. The soils in and around Kabwe are primarily red clays overlying rocks rich in ferromagnesian minerals with surrounding sandvelt soils, which are loamy sands or coarse grained sand with clay content increasing with depth (Surveyor General, 1967). As discussed below (section 1.3), clays and ferromagnesian minerals have both been associated with heavy metal identification by spectroscopy. Therefore, it is possible that pixels dominated by bare soil in and around Kabwe may contain enough Pb associated with spectrally active components (clay minerals and ferromagnesian minerals) that useful spectral information can be extracted directly from imaging spectrometer data of the area. The goal of this study was to determine whether that information could be extracted and associated with *in-situ* field XRF measurements to produce a quantitative image-analysis method for Pb detection via remote sensing without a need for laboratory analysis. Previous work investigating the detection of heavy metals in soil by laboratory spectroscopy guided our efforts.

1.3 Laboratory evidence for spectroscopic detection of heavy metals in soil

There is extensive laboratory research on the use of spectroscopic techniques to detect heavy metal contaminants in soil. There have been several excellent, fairly recent reviews of this work, which will not be fully reproduced here (e.g., Ben-Dor et al., 2009; Horta et al., 2015; Schwartz et al., 2011; Shi et al., 2014; Soriano-Disla et al., 2014; Stenberg et al., 2010). However, it is clear from the contradictory results presented by many of the studies cited in these reviews that generalizing the results of any one individual study is difficult because soil properties often differ distinctly from study to study. This is very important because spectroscopic detections of heavy metals in soil are generally attributed to the association of heavy metals with various spectrally active soil components, including clay minerals, organic content, and iron-and aluminum-oxides (Dupuy and Douay, 2001; Vohland et al., 2009; Wu et al., 2007; Wu et al., 2005a; Wu et al., 2005b). More specifically, Choe et al. (2008) suggested a surface complexation model in which heavy metals bind to hydroxylated mineral surfaces throughout soils, affecting the absorption peaks of various soil constituents especially, Fe-, Al-, and Mn-oxides (500-1000 nm), clay minerals (2000 nm), and organic matter (400-600 nm). They applied this theory to derive spectral parameters, which they associated with heavy metal concentrations. They found that Pb had one of the highest correlation coefficients (0.71) with the spectral parameter: ratio of the 610-nm reflectance band to the 500-nm reflectance band. In contrast, copper (Cu) showed weak correlations with all spectral parameters.

Previous research in chemistry using vibrational spectroscopy to study heavy metal complexes and, in particular, the structure and locations of adsorbed or complexed cations (e.g., Goldberg and Johnston, 2001; Piccolo and Stevenson, 1982), generally supports the idea that it should be possible to estimate the concentrations of heavy metals in soil using laboratory spectroscopy. Kemper and Sommer (2002) suggested that heavy metals in soil could be spectroscopically estimated by observing changes to the spectral features of spectroscopically active soil components altered by the presence of heavy metals. They found that the most important wavelengths for their analyses could be generally attributed to the absorption features of iron and iron oxides. In short, there is strong laboratory evidence that heavy metals can be detected spectroscopically via their interactions with other, spectrally active soil components. However, the results of laboratory efforts to calibrate the spectra of heavy metal contaminated soil and use these to quantitatively determine contaminant concentrations have been mixed.

Wu et al. (2007) showed that identifiable spectral features for heavy metals only appeared in the near infrared (NIR) spectra (500 – 2500 nm) of soil for transition elements (e.g., around 620 nm for Cr and 820 nm for Cu) at concentrations at or higher than 4000 mg/kg and suggested, more generally, that heavy metal elements cannot be detected with reflectance spectroscopy at concentrations less than or equal to 1000 ppm. This led later researchers (e.g., Kleinebecker et al., 2013) to recommend that reliable predictions of heavy metal concentrations by NIR be limited to samples with high concentrations. However, other researchers have suggested that visible-near infrared (VNIR) reflectance spectroscopy can be used to estimate heavy metal concentrations in soil at levels well below 100 ppm, if calibrated correctly (e.g., Wang et al., 2014). In addition, Pb concentrations have been successfully predicted in soil samples using combined visible near infrared-short wavelength infrared (VNIR-SWIR) reflectance spectroscopy between 400 – 2500 nm and partial least squares regression (PLSR) analysis at levels below 1000 ppm as long as the samples being compared are homogeneous (Al Maliki et al., 2014). Airborne urban particulate Pb contamination along a major roadway in Indianapolis has also been shown to be differentiable at levels as low as 100 – 400 ppm by VNIR reflectance spectroscopy combined with PLSR analysis (Pandit et al., 2010). Pandit, et al. (2010) also showed that Pb concentrations in soil have a linear bivariate relationship with reflectance values at the wavelengths 800 and 1300 nm.

Much of the most compelling evidence for the use of laboratory spectroscopy as a quantitative measure of heavy metal concentrations comes from studies of specific contaminated sites. Malley and Williams (1997) studied the heavy metal contaminant concentrations of freshwater sediment samples in Canada using NIR reflectance spectroscopy and PLSR analysis. After analyzing 169 samples, they successfully predicted the concentrations of Cu, Zn, Pb, Ni, Mn, and Fe in their samples and determined that heavy metal concentrations associated with protein, cellulose, and oil (e.g., organic matter). Similarly, Kooistra et al. (2001) found that VNIR absorbance spectroscopy could be used to predict the concentrations of Cd (0 – 6 mg/kg) and Zn (0 – 750 mg/kg) in 69 soil samples from the floodplains of the river Rhine in the Netherlands. They attributed the predictive success of their spectroscopic methods to the proxies of organic matter and clay content. Later studies have occasionally found differing predictive success rates with different heavy metals. For example, Bray et al. (2009) evaluated VNIR and mid-infrared (MIR) diffuse reflectance spectroscopy as diagnostic tools for predicting the concentrations of Cd, Cu, Pb, and Zn in soils above certain threshold concentrations. The thresholds were based on current Australian and New Zealand Environment Conservation Council (ANZECC) guidelines for each of the metals: Zn 200 mg/kg, Pb 300 mg/kg, Cu 60 mg/kg, and Cd 3 mg/kg. They found that Zn and Cu were well predicted (greater than 80% accuracy), while Cd and Pb were less well predicted (less than 70% accuracy). However, in a comprehensive review, Shi et al. (2014) found that predictive results between studies are inconsistent because the analysis is often based on a small number of calibration data. Nearly 40% of the studies cited by Shi et al. (2014) have fewer than 100 samples and many studies lack independent validation data. As a result, findings are unlikely to be generalizable. However, there have been attempts to carry out verification studies on large sample suites. One very large geochemical analysis by Soriano-Disla et al. (2013) used only diffuse reflectance MIR spectra, but analyzed 1000 randomly selected samples (out of a total sample suite of over 4000). Using this large sample suite, they developed a partial least squares (PLS) model associating the concentrations of 45 elements in their samples, as measured by laboratory XRF, with MIR diffuse reflectance features. They found that the elements: Ca, Mg, Al, Fe, Ga, Co, Ni, Sc, Ti, Li, Sr, K, Cr, Th, Be, S, B, Rb, V, Y, Zn, Zr, Nb, Ce, Cs, Na, In, Bi, Cu, and Mn can be well predicted by MIR diffuse reflectance spectroscopy ($R^2 > 0.6$), while the elements: As, Ba, La, Tl, P, U, Sb, Mo, Pb, Se, Cd, Sn, Hg, Ag and W were not well predicted ($R^2 < 0.5$).

In summary, there is strong evidence that heavy metal contaminants, especially Pb, can be identified in soil via indirect observations of their interactions with spectroscopically active soil components. However, it is difficult to use these relationships to derive calibrated, quantitative measures of heavy metal concentration in soil using laboratory spectroscopy and the quality of these results varies depending both on the spectroscopic technique used and the wavelength range investigated. Results are often inconsistent and difficult to generalize. Despite this, there have been several previous attempts to apply laboratory spectroscopic results to satellite imaging spectrometer datasets and to use image analysis techniques to attempt the remote sensing of heavy metal contaminants in soil. As described below, results for image or regional-scale analyses have been mixed and also rely heavily on extensive field spectroscopy or *a-priori* knowledge of the investigated site. Our goal was to determine if useful information could be gleaned through the direct association of field measurements of Pb concentrations in soil with image-scale spectral information from a high spatial resolution, multi-spectral (8-band) imaging dataset.

1.4 Previous work on the remote sensing of heavy metal contamination

Field and laboratory spectroscopy has been combined with imaging spectrometer data by previous researchers with limited success and has produced qualitative contaminant mapping methods. The first step is generally to calibrate field or laboratory spectra and associate changing spectral features with heavy metal concentrations. Various multivariate statistical analyses have been applied to accomplish this, including multiple linear regression (MLR) and artificial neural networks (ANN) (Kemper and Sommer, 2002) and PLSR (Al Maliki et al., 2014; Pandit et al., 2010). As described above, heavy metals are detected in soils by association with the features of spectrally active soil components. Choe et al. (2008) collected reflectance spectra in the field in the Rodalquilar mining area of Spain. They then mapped the distributions of reflectance spectral feature changes in these spectra that they found associated with heavy metal interactions with iron-oxides and clay mineral hydroxyl groups. By comparing the spatial distributions of spectral feature changes to gradient maps derived from heavy metal concentrations that they measured in collected soil samples, they found qualitative similarities between the two maps. Based on this, they suggested that field-derived spectral feature changes could be used to develop quantitative contamination mapping methods. However, the results of their efforts to apply the spectral changes that they observed in their field spectroscopy results to HyMap images were largely equivocal. They found two features with weak correlations between the spectral parameter values from their field spectra and the same spectral parameter measured in imaging spectrometer pixels (R^2 values of 0.54 and 0.55 for the ratio of the 1344 to the 778 nm bands and the area of the 2200 nm absorption, respectively). The four remaining features had R^2 values <0.5. More problematically, their statistical tests comparing ground and imaging spectrometer data showed significant differences between the ground and imaging spectrometer datasets for three of the six spectral parameters. They suggest that differences of scale between their field spectrometer and imaging spectrometer datasets likely account for these spectral differences. HyMap imaging spectrometer data has a 4 x 4 m spatial resolution, while their field spectrometer had a spot size of 10 mm diameter. Choe et al. (2008) also do not identify or account for any groundcover types other than possibly contaminated soil in their analysis of HyMap imaging spectrometer data.

Later, more successful applications of imaging spectrometer data to the remote sensing of contamination include work by Pascucci et al. (2012) who used field and laboratory measurements of red dust (RD), as well as soil and water samples contaminated by RD from an aluminum processing plant in Montenegro to calibrate a Multispectral Infrared and Visible Imaging Spectrometer (MIVIS) image. They were thus able to map RD contamination in the area and RD suspended and transported by the river. Pascucci et al. (2012) make clear, however, that the method they developed is only a rapid spectral screening tool that can detect minerals relevant to the presence of RD surface contamination and cannot be used to do a quantitative analysis of the level of RD (or contamination) in soils. Their method works by spectrally fingerprinting RD samples using laboratory spectroscopy and using these lab spectra to identify spectrally similar regions in their MIVIS images. Because RD is itself generally very high in trace metals, it acts as a proxy for soil contamination in imaging spectrometer data. Similarly, heavy metal contamination has also been detected by remote sensing using vegetative stress as a proxy for contamination (e.g., Clevers et al., 2004; Kooistra et al., 2004).

One of the most detailed examples of applied field spectroscopy to hyperspectral imaging spectrometer data is the continuation of the work originally presented by Kemper and Sommer (2002). In a later paper (García-Haro et al., 2005), the authors used a Variable Multiple Endmember Spectral Mixture Analysis (VMESMA) strategy to un-mix imaging spectrometer data collected using the HyMap sensor concurrent with two field seasons in 1999 and 2000. They strategically applied various field spectra collected during each of the two field seasons as end members in their analysis. Their results showed that it is possible to apply field spectra to imaging spectrometer data and map contaminated areas to a reasonable degree of accuracy. However, they used extensive *a-priori* knowledge of the contaminated area to help guide their analysis. The collection of such *in-situ* data may not always be possible for every contaminated region. As summarized by Ben-Dor et al. (2009), the reflectance properties of soils enable the assessment of various contaminants in soil environments and imaging spectroscopy is a promising, but not yet fully proven, tool for this purpose. Currently, research in this area relies on the application of laboratory or field spectra to imaging spectrometer datasets. We were curious whether potentially predictive information could be gleaned directly from the distribution of known contaminant concentrations within a multispectral image. To the best of our knowledge, this is the first attempt to produce a remote sensing method for Pb detection in soil based directly on the analysis of satellite imaging spectrometer data. This approach combines two available, relatively low-cost data sets (field XRF and commercial imaging spectrometer data). If effective, this method will not require extensive sample collection and laboratory analysis. However, it still requires the collection of *in-situ* field XRF data.

1.5 In-situ X-ray Fluorometry (XRF)

Portable, hand-held XRF is rapidly becoming the field methodology of choice as a low-cost, quick way to measure the contamination levels of many heavy metals in soil (Carr et al., 2008; Kalnicky and Singhvi, 2001). Previous work on the precision and accuracy of hand-held XRF devices (e.g., Bernick et al., 1995) has shown that well calibrated instruments can be used to analyze Pb and other target metals in soils with detection limits well below EPA action levels, which are 400 - 1200 mg/kg of Pb depending on the context (U.S. EPA, 2001). Correctly used hand-held XRF instruments also display excellent measurement comparability to standard laboratory techniques, such as ICP-AES (Rouillon and Taylor, 2016). Hand-held XRF instruments are advantageous because they can be used to produce much denser grids of data than traditional sampling and laboratory analysis as a result of their low cost per analysis and quick analysis times; new instruments can achieve good accuracy in 30 seconds. This is useful for potential remote sensing applications because denser *in-situ* data provides a more accurate picture of contaminant concentration variability. In addition, NGO's, aid, and environmental agencies have begun acquiring more XRF instruments because they are inexpensive. Methods are currently being developed to make use of the advantages of XRF in combination with traditional sampling and laboratory methods (Rouillon et al., 2017). With commercially available global imaging spectrometer data and increasing acquisitions of field XRF data, combining these two datasets into a quantitative remote sensing method could greatly increase the utility of both methods. Further work may also enable the development of quantitative, fully remotely sensed methods for the detection of heavy metal contamination in soil.

2. Data and Methods

2.1 In-situ field data

The NGO Pure Earth, which has worked extensively in Kabwe to identify and mitigate Pb contamination from the historical mine at the center of the city, collected and provided the XRF field data for this study. In August of 2014, Pure Earth conducted a field XRF measurement campaign jointly with the Kabwe Municipal Council in preparation for remediation work in a neighborhood in the Chowa township of Kabwe. The area was prioritized based on an earlier citywide assessment conducted by Pure Earth in 2013 and comprised of 394 measurements. Pure Earth collected 1013 geo-located XRF measurements of soils in the area using an Innov-X α-6500 hand-held X-ray fluorometer (XRF) in preparation for the remediation of the topsoil in the neighborhood from September – December 2015 (Figure 1).

The geo-locations of the collected XRF data points were determined by hand-held global positional system (GPS) units with ±3 m precision. The XRF data points were initially provided in the standard coordinate system, World Geodetic System 1984 revision (WGS 84). We converted this to the more location-specific projection, WGS-84, Universal Transverse Mercator (UTM) zone 35S, to match the original coordinate system and projection of the WV-2 standard image data product. We then binned, colorized, and mapped the provided XRF data points.

We initially binned the provided XRF data by equal count quantiles, which we modified to reflect the current U.S. Environmental Protection Agency standard for soil Pb contamination; 400 ppm for bare soil in children's' play areas and 1200 ppm average for bare soil in the rest of the yard (U.S. EPA, 2001). This produced five original bins (Table 1): 50 – 550 ppm (88 total points), 551 – 1000 ppm (185 total points), 1001 – 2000 ppm (353 total points), 2001 – 3500 ppm (240 total points), and 3501 – 63000 ppm (147 total points), which became the basis for the selection of pixels for regions of interest (ROI's) used to classify the image and derive the novel spectral index described below.

2.2 Imaging spectrometer (WorldView-2) data

WorldView-2 (WV-2) is a commercial multi-spectral satellite sensor managed by Satellite Imaging Corporation (https://www.satimagingcorp.com/satellite-sensors/worldview-2/). WV-2 launched on October 8, 2009 and began operating at full capability on January 4, 2010. WV-2 collects images with one high spatial resolution panchromatic band (0.21 m^2/pixel) and 8 multispectral bands: coastal blue (λ = 427.3 nm), blue (λ = 477.9 nm), green (λ = 546.2 nm), yellow (λ = 607.8 nm), red (λ = 658.8 nm), red edge (λ = 723.7 nm), near infrared 1 (NIR1) (λ = 832.5 nm) and NIR2 (λ = 908 nm). The high spatial resolution of WorldView-2 images makes them good candidates for associating field and imaging spectrometer data and the 8 multi-spectral bands are comparable to other remote sensing instruments (i.e., the Landsat series). Analysis results achieved with WV-2 may be modified and generalized to images collected by other instruments.

The image used in this study was acquired over Kabwe on May 10, 2014. It is the usable image collected closest in time to Pure Earth's field season in August of 2014. When attempting to calibrate remote sensing methods by comparing field and image data, it is important that the image and the field data be collected as close to concurrently as possible to avoid major changes in environment or groundcover. We worked with standard image product, which was partially processed by Digital Globe and delivered already georeferenced to an appropriate cartographic projection (WGS-84, UTM zone 35S) and normalized for topographic relief. Standard imagery also features Atmospheric Compensation, which is an algorithm, developed by Digital Globe, that corrects for atmospheric conditions and effects on spectral reflectance, but is not a complete radiometric correction and does not convert the image data out of raw data number (DN). Before analysis, the data must still be radiometrically converted.

2.3 Image analysis pre-processing

The imaging spectrometer data that we used in this study was processed and analyzed using the ENVI 5.3/IDL 8.5 image processing and analysis software package. We used a standard pre-processing workflow for WV-2 multi-spectral imagery based on the processing chain described by Wolf (2010) for preparing WV-2 images for land mapping and feature extraction (see Figure 2.2-1 in Wolf, 2010). To complete the radiometric correction of the image data, we first ran a dark-pixel (e.g., dark object) subtraction using the band minimum subtraction method (Updike and Comp, 2010 and citations therein). The data were then converted to top-of-atmosphere spectral radiance using the gain and offset factors provided by Digital Globe and the built-in radiometric correction functionality of the ENVI 5.3/IDL 8.5 image processing software package. Conversion to radiance is consistent with previous uses of WV-2 imagery for spectral indexing and land cover determinations (e.g., Wolf, 2010). In addition, our masking procedure (section 2.4.1) removed pixels dominated by vegetation, rooftops, and water from our analysis thereby also minimizing shadow in the scene. There is also little significant topography in the region of interest. Therefore, albedo effects do not need to be separated from illumination effects. Future applications or applications to data from other sensors may require further processing.

Once pre-processed, we subset the image data to focus on the neighborhood where Pure Earth collected XRF measurements (Figure 1, blue outline). We selected the boundaries of the region of interest to include both the XRF data as well as nearby neighborhoods that were not investigated. The southwest boundary of the region of interest extends to capture part of the built-up areas immediately southeast of the smelting complex (http://www.bmrplc.com). This area is assumed to be heavily contaminated. Image subsetting greatly reduced the computation time for subsequent image processing and analysis.

The final pre-processing step that we applied was to use the high resolution WV-2 panchromatic band to sharpen the multi-spectral image and produce a high resolution multi-spectral image (Parente and Belfiore, 2015). Previous research (e.g., Li et al., 2017) has shown that the adaptive Gram-Schmidt pan-sharpening method (Aiazzi et al., 2007; Laben and Brower, 2000) is appropriate for processing WV-2 images not used specifically for measurements of vegetation and water bodies, and that this method causes minimal spectral distortion. The ENVI 5.3/IDL 8.5 package contains a built-in Gram-Schmidt pan-sharpening workflow (http://www.harrisgeospatial.com/docs/GramSchmidtSpectralSharpening.html).

Increasing the spatial resolution of the imaging spectrometer multi-spectral data is important for attempting to classify and spectrally analyze pixels by their overlap and/or proximity to geo-located ground measurements. Field XRF measurements are individual points, while the image is made up of pixels covering larger areas of the study site. Having the highest possible spatial resolution, which for WV-2 is 0.21 m^2/pixel, or the smallest possible pixel size (0.46 m per pixel edge), may help mitigate these differences in scale between field and imaging spectrometer data, as suggested by Choe et al. (2008).

2.4 Image analysis methods

2.4.1 Identify and mask pixels of known spectral character

Based on previous laboratory work (e.g., Choe et al., 2008; Goldberg and Johnston, 2001; Kemper and Sommer, 2002; Piccolo and Stevenson, 1982; Wu et al., 2007), we expected changes in the imaging spectrometer data related to Pb contamination to appear only in pixels dominated by bare soil. Although vegetation has previously been used as a proxy for the detection of heavy metal contamination (e.g., Clevers et al., 2004; Kooistra et al., 2004), that was not the purpose of this study. Thus, as described in Mulder et al. (2013), pixels having substantial cover of non-useful groundcover types, in their case: vegetation, clouds, snow, and water, were masked out of the final analysis to retain strong signal from soils. For this study, we masked pixels dominated by vegetation, water, and components of the built environment (e.g., houses and rooftops) using the methods described in detail below.

We identified pixels dominated by water using the Normalized Difference Water Index (NDWI), first developed from WV-2 imagery by Wolf (2010). The NDWI examines the spread of the differences between the radiance values of the coastal and NIR2 bands in examined WV-2 pixels (Equation 1) and is part of the suite of spectral indices included in the ENVI 5.3/IDL 8.5 software package.

$$\frac{Coastal\ Blue\ (427.3\ nm) - NIR2\ (908.0\ nm)}{Coastal\ Blue\ (427.3\ nm) + NIR2\ (908.0\ nm)} \qquad \text{(Equation 1)}$$

The suite of indices included in the ENVI 5.3/IDL 8.5 software package also contains the WorldView Improved Vegetation Index (WV-VI), which we used to identify and mask pixels dominated by the spectral signature of vegetation (Equation 2).

$$\frac{NIR2\ (908.0\ nm) - Red\ (658.8\ nm)}{NIR2\ (908.0\ nm) + Red\ (658.8\ nm)} \qquad \text{(Equation 2)}$$

Both the NDWI and WV-VI are normalized difference indices, which produce values ranging from -1 to 1. Large differences in pixel radiance response values produce extreme spectral index values, which, if the index produces good results, tend to cluster together in reasonable locations within the scene. The threshold values for spectral indices are determined based on the reasonableness of the locations where certain response values lie in a given scene. For example, high NDWI values would be expected where there are standing bodies of water. To identify standing bodies of water in our WV-2 image, we used a threshold value of 0.75. Any pixel with an index value of 0.75 and above was classified as water and masked from the final analysis. The normal range for green vegetation from the WV-VI is 0.2 – 0.8 (Wolf, 2010; http://www.harrisgeospatial.com/docs/BroadbandGreenness.html#WorldVie2). The image that we used was collected in May 2014, just after the rainy season and therefore contained a lot of healthy, green vegetation. We used the normal range to identify and mask pixels dominated by green vegetation in the scene.

Rooftops in Kabwe proved significantly more challenging than either vegetation or water. There are multiple spectral indices built into the ENVI 5.3/IDL 8.5 software package that have been designed to identify pixels dominated by the built environment and man-made structures. However, none of these proved effective for the rooftops of the residential parts of Kabwe. This was likely due to the variety of materials used for roofing in the residential and industrial parts of

Kabwe. For example, many of the roofs of the structures in the smelting complex were made of corrugated steel or painted corrugated steel. Plain corrugated steel roofs are highly reflective and registered as too bright to classify. Meanwhile, painted steel roofs produced radiance spectra dominated by the wavelengths that made up the color they had been painted. For example, green roofs reflected strongly in the green, sometimes resembling vegetation, but without a strong NIR radiance peak; while red or blue roofs produced radiance signatures dominated by wavelength regions corresponding to those colors. Further complicating the situation, the scene also contained thatched and clay roofs, which resembled dried vegetation and soil, respectively. Ultimately, we determined that manual digitization was the most efficient way to ensure accurate identification of buildings and rooftops without excluding too many pixels. In future work, we will work toward a more accurate method to automatically identify rooftops and classify rooftop pixels as this is an important step for the remote sensing method described here. Figure 2 shows the final classification of the analyzed scene identifying the three excluded groundcover classes.

Figure 2. Classified zoomed-in image with buildings, water, and vegetation identified. These regions were masked from later analyses.

We then used the free, open-source Geographic Information System (GIS) software package QGIS v2.18.9 to analyze the overlap between the final masks for each groundcover type and the original 1013 collected XRF data points. This masking removed 528 of the original XRF points from the later analysis (Table 1) because they were geographically associated with non-useful pixels. Thirty of the removed XRF points overlapped with more than one masked region. As a result, the total number of masked points in Table 1 sums to 558, rather than the actual number of masked points (528).

Table 1. Total number of XRF measurements masked by common groundcover types in the image of Kabwe, Zambia, separated by detected Pb concentration.

Pb concentration in soil by XRF (ppm)	**Original number of measurements (count)**	**Masked by water (count)**	**Masked by rooftops (count)**	**Masked by vegetation (count)**	**Final number of measurements (count)**
$50 \leq [Pb]_{soil} \leq 550$	88	1	10	35	46
$551 \leq [Pb]_{soil} \leq 1000$	185	0	14	82	93
$1001 \leq [Pb]_{soil} \leq 2000$	353	4	31	155	171
$2001 \leq [Pb]_{soil} \leq 3500$	240	6	31	110	104
$3501 \leq [Pb]_{soil} \leq 63000$	147	3	14	62	71
Total	1013	14	100	444	485

2.4.2 Established classification methods: Spectral Angle Mapper and minimum distance

As described by Kruse et al. (1993), the Spectral Angle Mapper (SAM) tool permits rapid mapping of the spectral similarity between image and reference spectra. SAM results are indifferent to illumination differences between pixels. We used image-derived end member spectra corresponding to the mean spectra of regions of interest (i.e. groups of pixels) associated with each of the Pb contamination level bins described in section 2.1 for our initial SAM classification. We used a maximum spectral angle of 0.1 radians to define good matches. Based on the mean spectra of the final classified regions, we observed clear spectral differences at the 1000 ppm threshold and repeated the analysis with Pb contamination measurements grouped according to that single threshold (greater or less than 1000 ppm). We also compared our SAM results with those for minimum distance classification (Richards, 1999) using the pre-set standard deviation threshold of 1.00. The minimum-distance classification method is sensitive to albedo and shadow. Thus, it is a good check on our SAM results.

2.4.3 Derivation of the novel spectral index

In the mean spectra of regions classified using SAM and minimum distance, we observed a separable spectral trend between the 658.8 nm and 477.9 nm spectral bands that was clear in the mean spectra of regions classified using the 1000 ppm threshold. We hypothesized that this trend could be applied as a spectral index, which might be correlated with Pb contamination and could then be used to classify the image by spectral index values, rather than image-derived reference spectra. We chose to use a normalized difference index similar to the WV-VI and NDWI indices (Equation 3) to reduce the effects of albedo and illumination differences.

$$\frac{Red(658.8\,nm) - Blue(477.9\,nm)}{Red\,(658.8\,nm) + Blue(477.9\,nm)} \qquad \text{(Equation 3)}$$

For the purposes of mapping the index results, we set high and low threshold values on the index of -0.6 to 0.3 normalized radiance units. This range captured 99.8% of the analyzed (not masked) pixels in the image, or 6,896,591 pixels out of a total of 6,910,412 analyzed pixels.

2.4.4 Comparison of spectral index results to other classification techniques

To test the predictive quality of the spectral index, we compared the regions identified with very low spectral index values (high likelihood of Pb contamination) to regions with good SAM matches to the mean spectrum of a region of interest defined within the leaching plant tailings pile. We used the change detection difference map function in ENVI 5.3/IDL 8.6 (http://www.harrisgeospatial.com/docs/ChangeDetectionAnalysis.html#Computin) to generate a difference map and compute the overlap between these two classified images. The Mineral Corporation Consultancy estimated in a 2010 report for BMR Group PLC, that the leaching plant tailings residue contains up to 8.71% Pb by weight (http://bmrplc.com/tailings_stockpiles.php). Pixels spectrally similar to this mean spectrum likely reflect contamination by components of the slag pile, which are believed to be the main point source of Pb contamination within Kabwe (Caravanos et al., 2014), and therefore are expected to have high levels of Pb contamination. This makes the locations of such pixels a reasonable verification of the efficacy of our spectral index-derived classification for detecting regions where there is a high probability of Pb contamination.

2.5 Statistical analysis

To investigate whether the novel spectral index could be used predictively to identify regions of high Pb contamination, we examined the correlation between the index values from image pixels and their associated log transformed Pb contamination concentrations determined by XRF. Using a bivariate linear correlation, we analyzed the statistical significance of any association between the spectral index and Pb contamination concentration in soil.

We also examined the predictive quality of the image classification based on our spectral index by exporting classified images as vectors and testing their overlap with binned XRF data points using the QGIS vector testing tool: “count points in polygon”. Finally, similarly to the approach of Pascucci et al. (2012), we derived confusion matrices (i.e. contingency matrices) using the pixels associated with XRF measurements of ground-truth Pb contamination to assess and compare the overall accuracies of the SAM, minimum distance, and spectral index classification results with respect to the highest and lowest Pb contamination classes.

3. Results and Discussion

3.1 Established classification methods do not accurately predict ground control data

Figure 3 shows the image classification results from SAM (A) and minimum distance (B) classification methods applied to mean spectra of ROIs derived from pixels associated with the locations of field XRF measurements, which were grouped according to Pb concentration as described in section 2.1 (above). There are some obvious qualitative similarities between these two classified images, such as good spectral matches to high Pb contamination in the southwest corner of the image and good spectral matches to low Pb contamination along the dirt roads of the image. These qualitative similarities may reflect a remotely detectable trend indicative either of Pb contamination or of the presence of contaminated material from the central slag pile (a proxy for Pb contamination). The qualitative similarities between these two classification results are also encouraging because one method (SAM) is indifferent to albedo and illumination effects, to which the second (minimum distance) is sensitive. The fact that both methods produce qualitatively similar class distributions, although the minimum distance classification classifies many fewer pixels, indicates that spectral trends associated with changing Pb concentration in soil are not simply an artifact of overall albedo or pixel illumination.

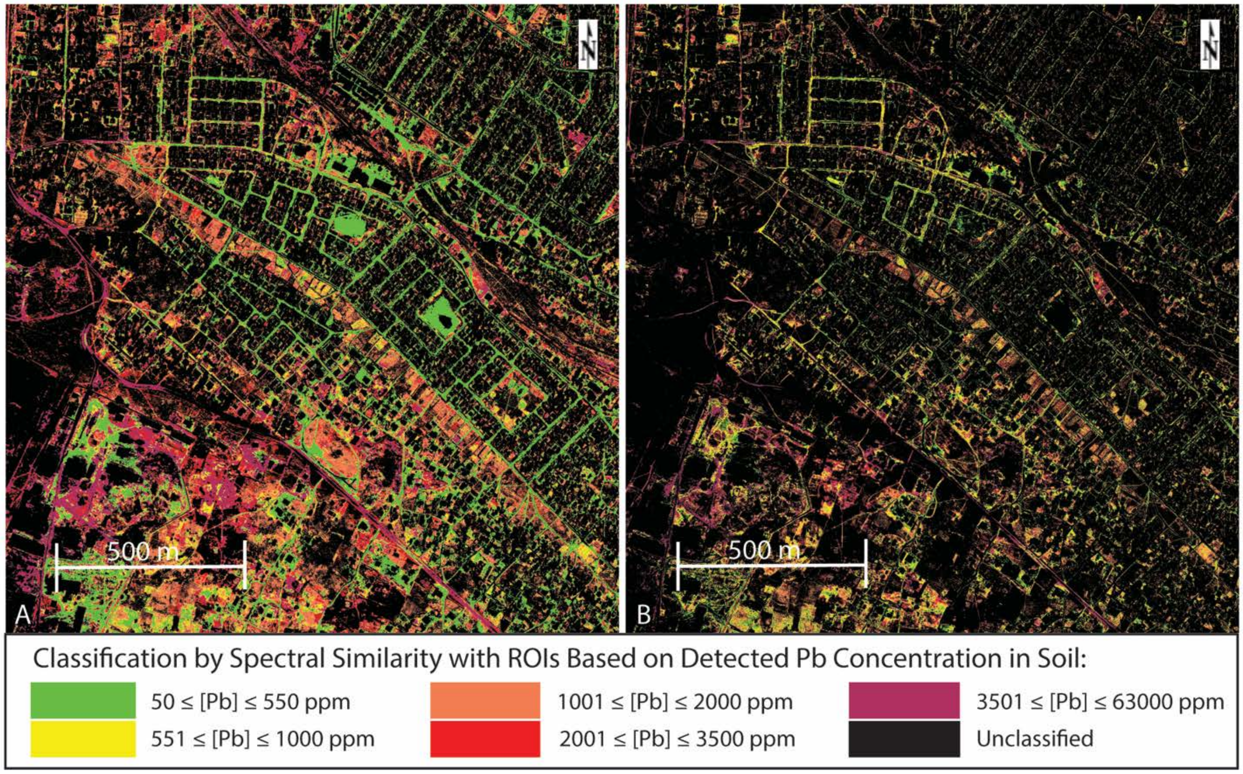

Figure 3. SAM (A) and minimum distance (B) classification results using ROI's based on binned Pb level detections by XRF (Table 1). The maximum angle specified for SAM was 0.10 radians and a standard deviation threshold of 1.00 was applied to the minimum distance classification.

Despite the encouraging similarity between these two established classification methods, however, the accuracy of these classifications when verified by ground-truth is not high. We examined the classification accuracy of SAM and minimum distance both through comparison to the overlap of classified regions with field XRF data points directly (Table 2) and by derivation of confusion matrices using initial ROI pixels as ground truth (Tables 3 and 4 for SAM and minimum distance, respectively). The overall accuracies computed during the generation of confusion matrices ranges from 16.64% (SAM) to 9.77% (minimum distance). More problematically, the accuracy of the SAM classification decreases with increasing measured Pb concentrations in soil (Table 3). The accuracy of the minimum distance classification also decreases after the lowest Pb concentration region of interest, but is mostly consistent across the higher four Pb concentration classes with the exception of the $2001 \leq [Pb]_{soil} \leq 3500$ ppm class, which has a ground-truth accuracy of 3.88%. For both methods, the number of ground-control XRF data points that are incorrectly captured is a factor of 3 larger than the number of XRF data points that are correctly captured. It is clear from these results that established classification methods do a poor job of accurately identifying regions of high Pb contamination when applied straightforwardly. However, as we describe below, a general spectral trend emerged from this analysis, which we first investigated as a simpler classification with only two Pb contamination classes and then applied as a spectral index to the region of interest.

Table 2. Distribution of XRF data points classified correctly by SAM and Minimum Distance classification methods.

Pb concentration in soil by XRF (ppm)	**XRF measurements correctly classified by SAM (%)**	**XRF measurements incorrectly classified by SAM (%)**	**XRF measurements unclassified by SAM (%)**	**XRF measurements correctly classified by Minimum Distance (%)**	**XRF measurements incorrectly classified by Minimum Distance (%)**	**XRF measurements unclassified by Minimum Distance (%)**
$50 \le [Pb]_{soil} \le 550$	50.00	23.91	26.09	21.74	10.86	67.40
$551 \le [Pb]_{soil} \le 1000$	10.75	67.73	21.52	9.68	23.66	66.66
$1001 \le [Pb]_{soil} \le 2000$	21.05	47.94	31.01	9.94	29.82	60.24
$2001 \le [Pb]_{soil} \le 3500$	8.65	67.3	24.05	10.58	39.43	50.01
$3501 \le [Pb]_{soil} \le 63000$	8.45	56.33	35.22	11.27	36.62	47.89
Total	17.32	54.84	27.84	9.90	31.34	58.76

Table 3. Confusion matrix (percent) showing predictive quality of the SAM classification compared to ground truth ROIs based on binned XRF data locations. The shaded cells denote the correct classification diagonal. The overall accuracy is 16.63%.

	Ground Truth (percent)						
Predicted	Class (Pb concentration ppm)	$50 \le [Pb]_{soil} \le 550$	$551 \le [Pb]_{soil} \le 1000$	$1001 \le [Pb]_{soil} \le 2000$	$2001 \le [Pb]_{soil} \le 3500$	$3501 \le [Pb]_{soil} \le 63000$	Total
	Unclassified	23.91	21.51	30.99	23.30	36.76	27.65
	$50 \le [Pb]_{soil} \le 550$	50.00	38.71	27.49	19.42	16.18	28.48
	$551 \le [Pb]_{soil} \le 1000$	2.17	10.75	6.43	9.71	13.24	8.52
	$1001 \le [Pb]_{soil} \le 2000$	8.70	12.90	19.88	19.42	20.59	17.46
	$2001 \le [Pb]_{soil} \le 3500$	4.35	4.30	5.26	8.74	7.35	6.03
	$3501 \le [Pb]_{soil} \le 63000$	10.87	11.83	9.94	19.42	5.88	11.85
	Total	100.00	100.00	100.00	100.00	100.00	100.00

Table 4. Confusion matrix (percent) showing predictive quality of the minimum distance classification compared to ground truth ROIs based on binned XRF data locations. The shaded cells denote the correct classification diagonal. The overall accuracy is 9.77%.

	Ground Truth (percent)						
Predicted	Class (Pb concentration ppm)	$50 \le [Pb]_{soil} \le 550$	$551 \le [Pb]_{soil} \le 1000$	$1001 \le [Pb]_{soil} \le 2000$	$2001 \le [Pb]_{soil} \le 3500$	$3501 \le [Pb]_{soil} \le 63000$	Total
	Unclassified	67.39	66.67	60.23	49.51	52.94	58.84
	$50 \le [Pb]_{soil} \le 550$	21.74	13.98	14.04	12.62	11.76	14.14
	$551 \le [Pb]_{soil} \le 1000$	6.52	9.68	11.70	10.68	13.24	10.81
	$1001 \le [Pb]_{soil} \le 2000$	2.17	2.15	9.36	10.68	8.82	7.48
	$2001 \le [Pb]_{soil} \le 3500$	0.00	2.15	1.17	3.88	1.47	1.87
	$3501 \le [Pb]_{soil} \le 63000$	2.17	5.38	3.51	12.62	11.76	6.86
	Total	100.00	100.00	100.00	100.00	100.00	100.00

The mean spectra of the ROIs used to derive the SAM and minimum distance classification targets (Figure 4) resemble modified bare soil spectra in that the radiance values generally peak in the RedEdge (723.7 nm), rather than the Red (658.8 nm) band region, and do not drop off toward the Blue or NIR ranges of the spectrum as quickly as normal soil. The spectra shown in Figure 4 also show a dramatic change in spectral character at the 1000 ppm Pb concentration level. The mean spectra of the three ROIs derived from pixels associated with XRF measurements with Pb levels greater than 1000 ppm are similar to one another; while those derived from pixels associated with XRF measurements with Pb levels below 1000 ppm are also similar to one another and distinct from the other three spectra. This result is consistent with previous laboratory work showing that 1000 ppm may be a threshold for the detection of heavy metals in soil by reflectance spectroscopy (Wu et al., 2005a; Wu et al., 2005b).

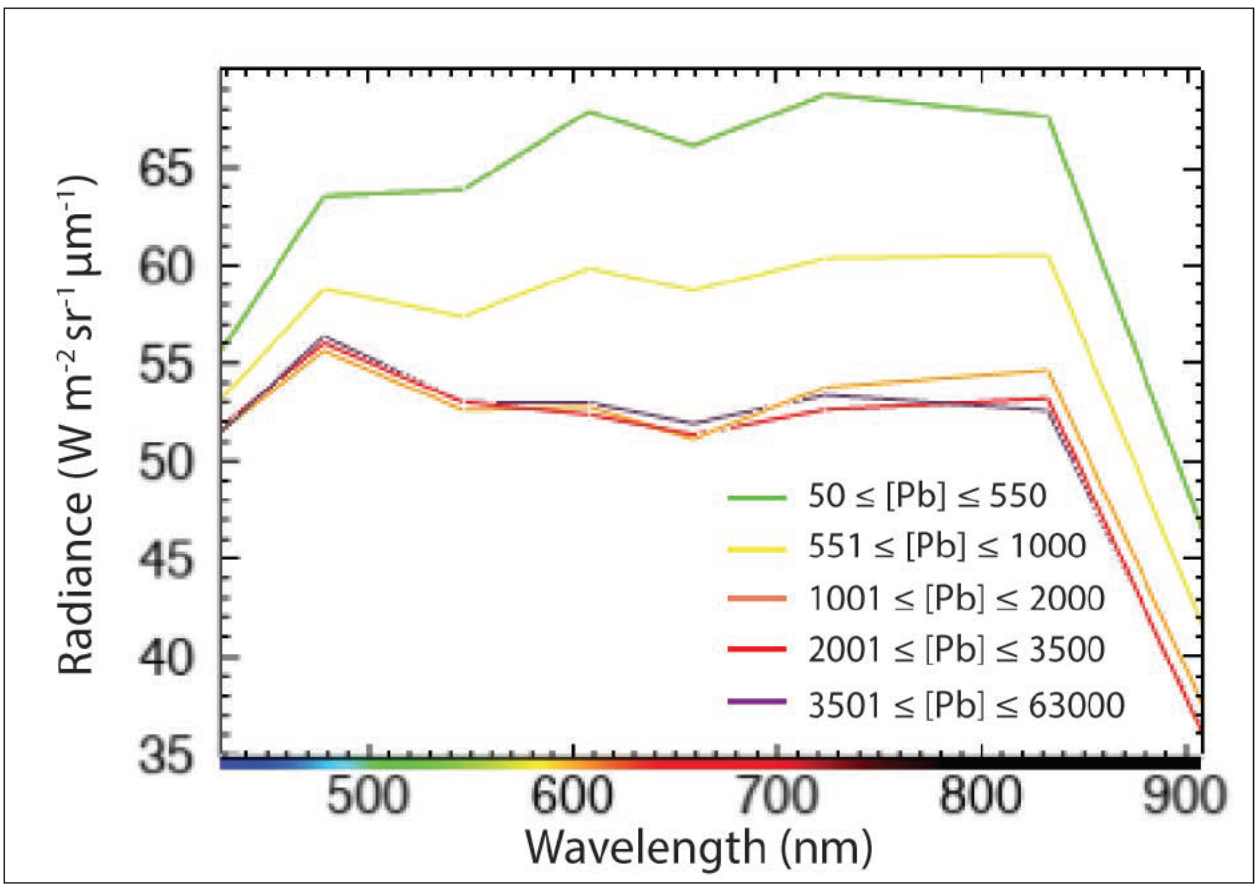

Figure 4. Mean spectra of regions of interest derived from XRF measurement locations, binned by Pb concentration in soil.

Based on this, we re-examined the classification of our imaging spectrometer data using SAM and minimum distance with only two XRF measurement classes: $[Pb]_{soil} \geq 1001$ ppm or $\leq$ 1000 ppm instead of the original five (Figure 5).

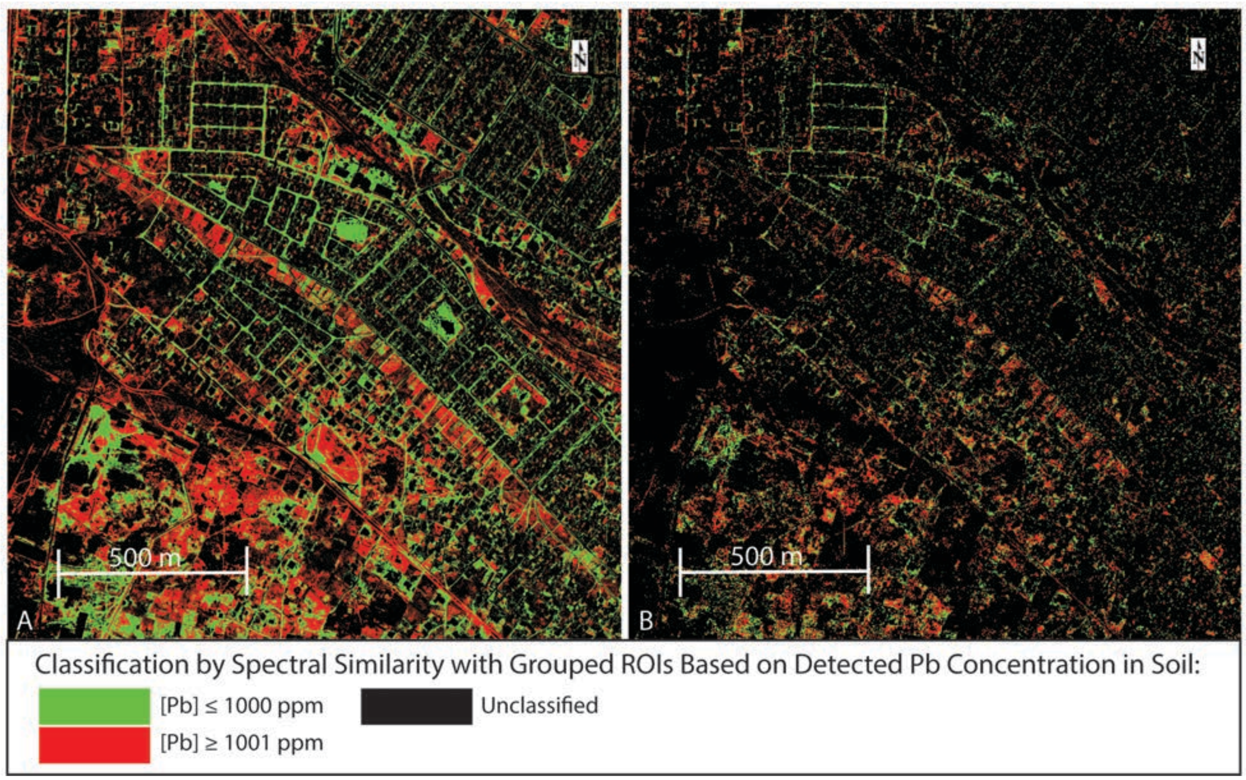

Figure 5. SAM (A) and minimum distance (B) classification results using ROI's based on less finely binned Pb level detections by XRF with only two classes: $[Pb]_{soil} \geq 1001$ ppm or ≤ 1000 ppm. The maximum angle specified for SAM was 0.10 radians and a standard deviation threshold of 1.00 was applied to the minimum distance classification.

There are qualitative similarities between these simpler classified images and those based on the original five XRF measurement classes. Unsurprisingly, the regions identified in Figure 3 as spectrally similar to the ROIs defined by the three Pb concentration classes with $[Pb]_{soil} \geq$ 1001 ppm, combine to make up the "high Pb" ($[Pb]_{soil} \geq 1001$ ppm) classified region in this simpler classification. The regions in Figure 3 that were identified as spectrally similar to the two classes with $[Pb]_{soil} \leq 1000$ ppm, combine to make up the "low Pb" ($[Pb]_{soil} \leq 1000$ ppm) classified region of this simpler classification. Simplifying the classification also improves the accuracy of the predictions derived from the classified image relative to ground-truth. After deriving the confusion matrices for the highest and lowest XRF-derived $[Pb]_{soil}$ classes, the overall accuracy for the SAM classification is 40.35% and the accuracy for the highest XRF-derived $[Pb]_{soil}$ class increases from 5.88% to 33.82% (Table 5). The accuracy for the lowest XRF-derived $[Pb]_{soil}$ class remains the same at 50.0%.

Table 5. Confusion matrix (percent) showing predictive quality of the SAM classification derived from XRF measurements with Pb concentrations in soil binned above and below 1000 ppm compared with ROIs derived from locations with very high and very low detected Pb concentrations. The shaded cells denote the correct classification diagonal. The overall accuracy is 40.35%.

	Ground Truth (percent)			
Predicted	Class (Pb concentration ppm)	$50 \leq [Pb]_{soil} \leq 550$	$3501 \leq [Pb]_{soil} \leq 63000$	Total
	Unclassified	32.61	38.24	35.96
	$[Pb]_{soil} \leq 1000$	50.00	27.94	36.84
	$[Pb]_{soil} \geq 1001$	17.39	33.82	27.19
	Total	100.00	100.00	100.00

Similarly, the simpler minimum distance classification shows an increased overall accuracy of 18.42%, relative to the classification with five $[Pb]_{soil}$ classes. In contrast to the SAM results, however, the accuracy for the lowest $[Pb]_{soil}$ class decreases with this simpler classification to 13.04%, while the accuracy for the highest $[Pb]_{soil}$ class increases to 22.06% (Table 6). The number of unclassified ground-truth pixels also increases for both SAM and minimum distance classifications using this simpler classification method.

Table 6. Confusion matrix (percent) showing predictive quality of the minimum classification derived from XRF measurements with Pb concentrations in soil binned above and below 1000 ppm compared with ROIs derived from locations with very high and very low detected Pb concentrations. The shaded cells denote the correct classification diagonal. The overall accuracy is 18.42%.

	Ground Truth (percent)			
Predicted	Class (Pb concentration ppm)	$50 \leq [Pb]_{soil} \leq 550$	$3501 \leq [Pb]_{soil} \leq 63000$	Total
	Unclassified	84.78	54.41	66.67
	$[Pb]_{soil} \leq 1000$	13.04	23.53	19.30
	$[Pb]_{soil} \geq 1001$	2.17	22.06	14.04
	Total	100.00	100.00	100.00

The spectral differences observed for the mean spectra of the five original XRF $[Pb]_{soil}$ classes are also observed for the larger regions classified by the simpler SAM and minimum distance methods using only two XRF $[Pb]_{soil}$ classes. Pixels associated with Pb contamination above 1000 ppm Pb in soil show a dip in radiance toward the Red (658.8 nm) band that pixels associated with Pb contamination below 1000 Pb in soil do not (Figure 6). This trend applies within one standard deviation from the mean spectrum of each region.

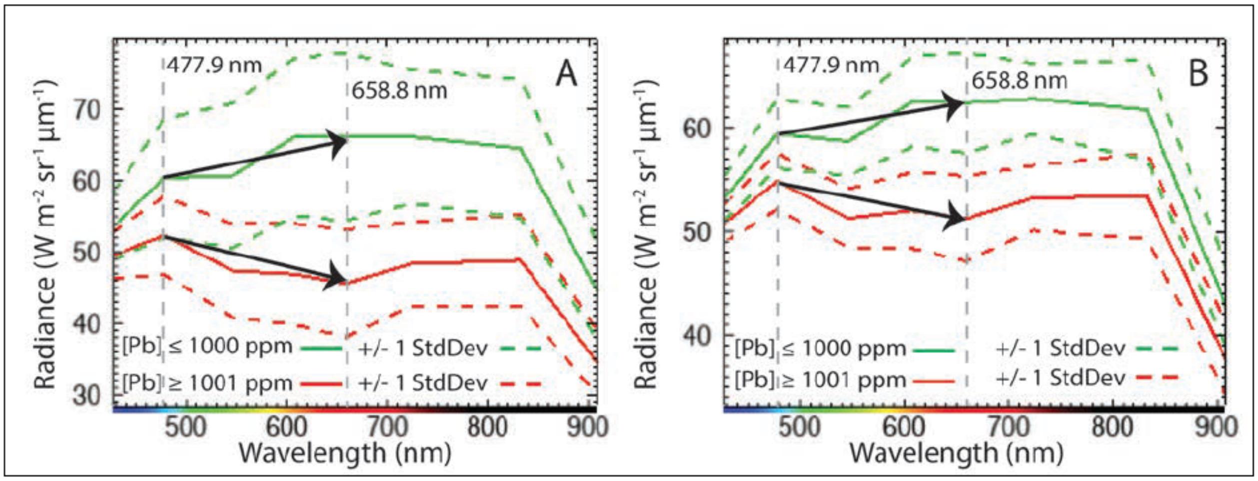

Figure 6. Mean spectra (+/- one standard deviation, dashed lines) of the regions classified by SAM (A) and minimum distance (B) using ROI's based on binned Pb level detections: $[Pb]_{soil} \geq$ 1001 ppm and $\leq$ 1000 ppm. A separable spectral trend between the radiance values of the Blue (477.9 nm) and Red (658.8 nm) bands that appeared to correlate with Pb levels in soil emerged from the examination of the mean spectra of these less finely binned ROI's.

Based on this observation, we derived a novel spectral index (Equation 4) to quantify the differences between these bands. By applying the spectral index values as a classification throughout the analyzed scene we were able to investigate correlations with Pb concentration that we used semi-quantitatively to identify regions of likely high Pb contamination. The index is most accurate for ground truth regions with $3501 \leq [Pb]_{soil} \leq 63000$ ppm and $50 \leq [Pb]_{soil} \leq 550$ ppm. With additional investigation, this approach may lead to a fully quantitative method for detecting regions of Pb contamination using existing multi-spectral imaging spectrometer data.

3.2 Regression between index values and XRF measurements of soil Pb concentration

To test the applicability of the spectral index derived from our SAM and minimum distance classification results to the prediction of Pb concentration in WV-2 imagery, we investigated whether there was any correlation between index values and XRF-measured Pb concentration in soil (ground truth). Figure 7 shows the final results of the regression performed in JMP v13.1 statistical analysis software.

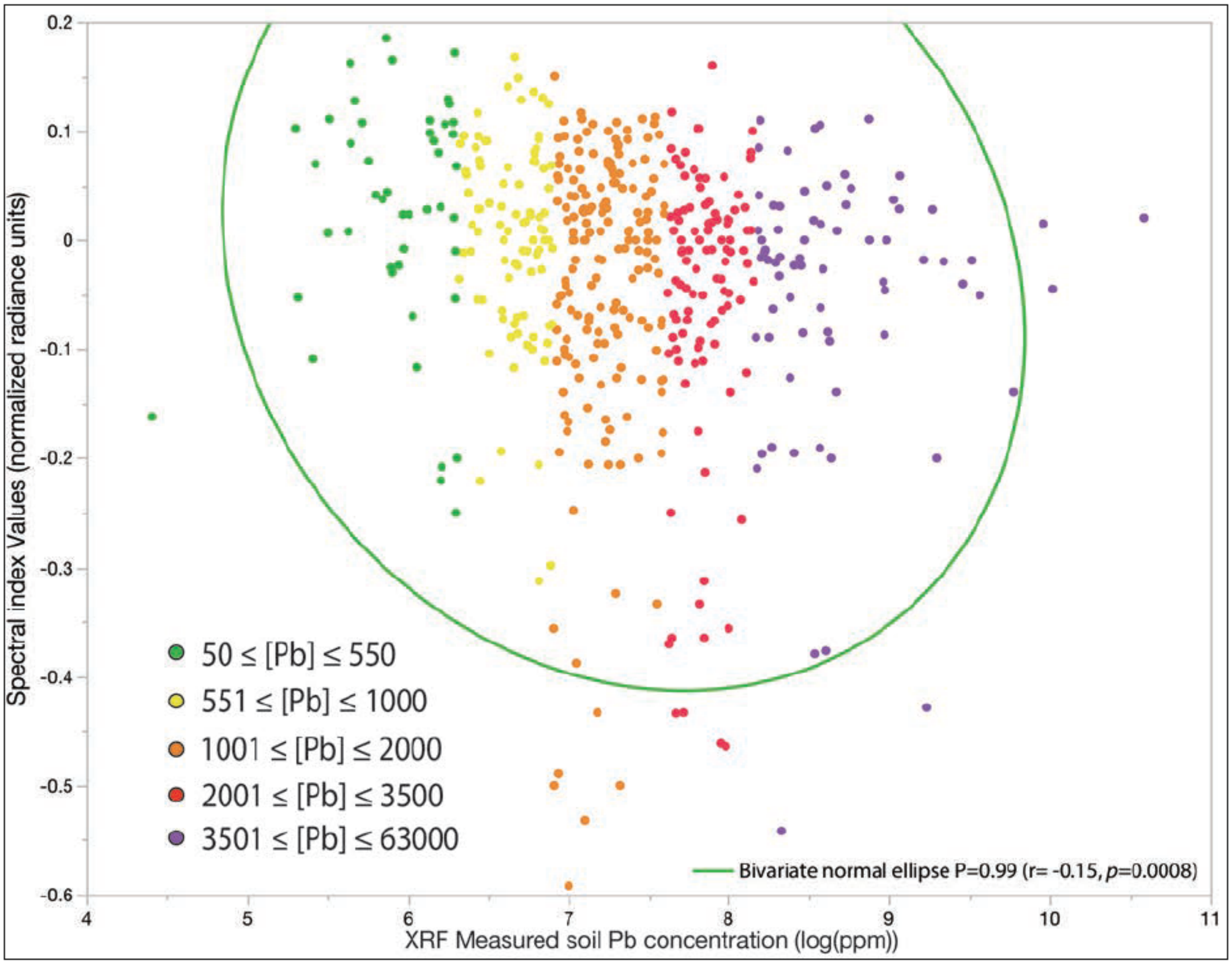

Figure 7. Regression between the novel spectral index developed for Pb contamination detection and log(Pb) concentrations in soils as measured by XRF. Data points are colorized according to the five original $[Pb]_{soil}$ classes. There is a weak (r=-0.15), but significant (*p*=0.0008) negative correlation between log (Pb concentration) and spectral index values.

The regression shows a weak (r=-0.15), but significant (*p*=0.0008) negative correlation between index values and Pb concentrations in soil as measured in the field using XRF. The low r value implies that Pb concentration alone explains less than 4% of the variance in the spectral index produced by pixels in the scene. This is not unexpected because Pb concentration only likely affects part of each bare soil pixel in the scene and it is possible that other components contribute to the spectral character. One part of the low amount of variance explained is likely the simple difference in scale between a 0.25 m^2 WV-2 pixel and an XRF data point from a handheld XRF instrument with a roughly 8 mm^2 footprint (Young et al., 2016). Similar explanations have been previously suggested to explain disconnects between laboratory spectra and imaging spectrometer datasets (e.g., Choe et al., 2008). Despite this, the significance of the correlation implies that the spectral index likely reflects a detectable signal from the limited 8-band satellite imaging spectrometer data, although that signal is small. Previous laboratory work has reported correlations between heavy metal concentrations and spectral features as significant if $p < 0.01$ when measured in the laboratory (e.g., Choe et al., 2009). Our p-value is $\ll 0.01$ even with the complications introduced by image-based, rather than more straightforward laboratory spectral analysis. Based on these results, we hypothesized that it should be possible to use spectral index values to produce a classified image that effectively maps probable areas of high Pb contamination. We discuss the results of this classification below.

3.3 Image classification using spectral index values

Figure 8 shows the final result of the classification of the WV-2 image of our region of interest in Kabwe by colorized spectral index values. For visualization purposes, the index values were truncated to the range -0.6 – 0.3 normalized radiance units. The full range of computed spectral index values spans from -1.62 to 0.91. The displayed range captures 99.8% of the analyzed pixels and eliminates outlying values. In the neighborhood where Pure Earth collected ground-truth XRF data, pixels showed the full range of displayed index values. The correlation between the spectral index and measured Pb concentration in soil is too weak to use the spectral index to fully quantitatively predict the level of Pb contamination in soil in a given region. It is clear, however, that the likelihood of moderate – high Pb contamination ($[Pb]_{soil} \geq 1001$ ppm) increases with smaller (more negative) spectral index values. This is supported by the results of three internal validation tests that we applied to our spectral index-derived image classification results. We therefore believe that the spectral index we developed here is a good indicator of probable Pb contamination and can be used to remotely identify contaminated regions, although it cannot be used to precisely predict the concentration of Pb in the soils of contaminated areas.

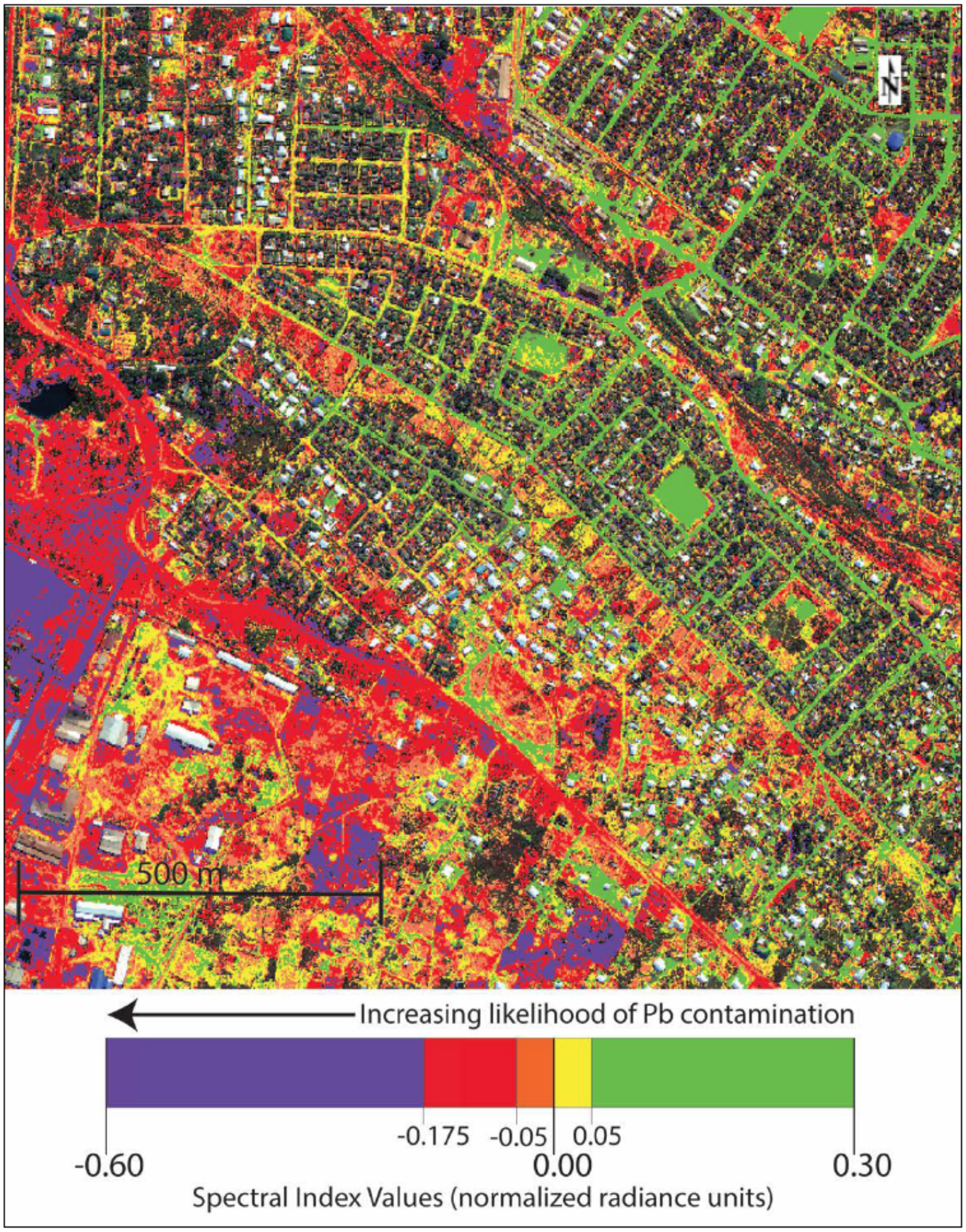

Figure 8. A classified map of the region of interest using colorized spectral index values to designate areas by increasing likelihood of Pb contamination. The classes are overlain on a true-color (RGB) composite of the pan-sharpened original WV2 image.

As a first internal measure of validation, we tested the predictive quality of the index by investigating how well regions classified as likely Pb contaminated (spectral index values < 0) or not (spectral index values > 0), based on the spectral index, captured relevant XRF data points (Table 7). Compared to established classification methods, the spectral index-derived image classification is much more effective and, usefully, it is most effective at predicting the locations of ground-truth points with very high ($[Pb]_{soil} \geq 3501$) or very low ($[Pb]_{soil} \leq 550$ ppm) detected levels of Pb contamination in soil.

Table 7. Predictive quality for binned XRF data points of the image classification based on the novel spectral index where index values > 0 are defined as low Pb and index values < 0 are defined as high Pb. The index improves the predictive quality of the image classification by a factor of 7.5, compared to SAM, and a factor of 5.6, compared to minimum distance, for the regions with the highest Pb contamination concentration.

XRF measurement soil Pb concentration range (ppm)	**Classified Correctly (%)**	**Classified incorrectly (%)**
$50 \leq [Pb]_{soil} \leq 550$	67.39	32.61
$551 \leq [Pb]_{soil} \leq 1000$	54.84	45.16
$1001 \leq [Pb]_{soil} \leq 2000$	49.12	50.87
$2001 \leq [Pb]_{soil} \leq 3500$	59.62	40.38
$3501 \leq [Pb]_{soil} \leq 63000$	63.38	36.62
Overall	56.29	43.71

The spectral index-derived image classification is most effective for the highest Pb ($3501 \leq [Pb] \leq 63000$ ppm) and lowest Pb ($50 \leq [Pb] \leq 550$ ppm) bins with 63.38% and 67.39%, respectively, of XRF data points in each correctly classified. It is least effective for the middle bin ($1001 \leq [Pb] \leq 2000$ ppm) with 50.87% of XRF data points incorrectly classified. This may be due to the fact that the majority of the points in this bin (65%) fall below the middle value (1500 ppm) and there are more points closer to 1000 ppm, which was the boundary of the initial image analysis through which we derived the spectral index. The correct classification of over sixty percent of the XRF data points in the highest and lowest Pb bins demonstrates the overall effectiveness of the novel spectral index. Although this method does not quantitatively identify soil Pb levels, it does provide a reliable guide for identifying very likely areas of high contamination and low contamination with a mean 65% accuracy (Table 7).

The second internal validation method that we used was to compare regions classified as likely Pb contaminated (spectral index values < 0) with the locations of pixels in the original regions of interest derived from the binned ground-truth XRF data (section 2.1). Because the index is not a fully quantitative image classification method, it can only be accurately used to derive two classes of image pixels (likely Pb contaminated and not). As a result, confusion matrices derived from image classification results based on the spectral index can only be compared to two ground-truth ranges at a time. We compared the highest detected Pb contamination ($3501 \leq [Pb] \leq 63000$ ppm) and the lowest detected Pb contamination ($50 \leq [Pb] \leq 550$ ppm) and found that the overall accuracy of the spectral index-derived classification for these Pb contamination ranges was 64.91% (Table 8). This is consistent with our previous internal validation test. We also derived a confusion matrix for regions of interest with intermediate ($2001 \leq [Pb] \leq 3500$ ppm) and low ($551 \leq [Pb] \leq 1000$ ppm) Pb contamination with an overall accuracy of 56.63% (Table 9), also consistent with our previous results.

Table 8. Confusion matrix (percent) showing predictive quality of the novel spectral index for ROIs derived from locations with very high and very low detected Pb concentrations in soil. The shaded cells denote the correct classification diagonal. The overall accuracy is 64.91%.

		Ground Truth (percent)		
Predicted		$50 \leq [Pb]_{soil} \leq 550$	$3501 \leq [Pb]_{soil} \leq 63000$	Total
	Background (unclassified)	0.00	1.47	0.88
	Likely uncontaminated Spectral index value > 0	67.39	35.29	48.25
	Likely contaminated with Pb Spectral index value < 0	32.61	63.24	50.88
	Total	100.00	100.00	100.00

Table 9. Confusion matrix (percent) showing predictive quality of the novel spectral index for ROIs derived from locations with moderately high and moderately low detected Pb concentrations in soil. The shaded cells denote the correct classification diagonal. The overall accuracy is 56.63%.

		Ground Truth (percent)		
Predicted		$551 \leq [Pb]_{soil} \leq 1000$	$2001 \leq [Pb]_{soil} \leq 3500$	Total
	Background (unclassified)	0.00	0.00	0.00
	Likely uncontaminated Spectral index value > 0	53.76	40.78	46.94
	Likely contaminated with Pb Spectral index value < 0	46.24	59.22	53.06
	Total	100.00	100.00	100.00

The spectral index-derived classification shows considerable improvement over established image classification methods. Overall accuracy improves from 16.63% (five ROIs based on Pb concentration) to 40.35% (two ROIs) to 64.91% (spectral index) for the SAM classification method; while it improves from 9.77% (five ROIs) to 18.42% (two ROIs) to 64.91% (spectral index) for minimum distance classification. The spectral index-derived classification matches ground truth for the highest and lowest $[Pb]_{soil}$ regions most accurately (64.95%), but is less accurate (56.63%) for intermediate levels of Pb contamination. We consider this result “semi-quantitative” because the spectral index cannot be used to predict Pb concentration in soil. However, the spectral index significantly improves the accuracy of remotely detected regions of probable Pb contamination. Such a spectral index would be useful for targeting future investigations and remediation efforts. We note here that it is not yet clear whether this spectral index can be accurately applied to the WV-2 images of other contaminated regions. It is possible that the index is sensitive to bulk soil composition or is detecting an interaction between Pb and soil components that are specific to Kabwe and may not be applicable to other contaminated regions. The index may also be sensitive to soil components that are correlated with Pb (e.g., clays, iron-oxides), rather than Pb itself. Future work will involve investigating the general applicability of the spectral index, as well as validation through the collection of additional ground-truth data and the examination of changes in the index-based mapping of the same area in different years, as well as testing in other areas with similar contamination. In particular, the concurrent collection of field measurements and imaging spectrometer data followed by sampling directed by the image analysis results to confirm them. However, we were able to validate our results without additional fieldwork by comparing the results of the spectral index-derived classification to the results of a SAM classification based on the mean spectrum of a region of interest defined within the central slag pile.

3.4 Comparison of spectral index classification with target detection from an area of extreme Pb contamination (the central slag pile)

As a further verification of the results of our spectral index-derived image classification, we compared the locations of the 6.86% of pixels with the lowest spectral index values (-0.60 to -0.175 normalized radiance units) with the locations of pixels that were good matches by SAM for the mean spectrum of a ROI derived from the central slag pile (Figure 9). The slag pile is both highly contaminated, with up to 8.71% Pb by weight and is also thought to be the likely source for much of the contamination found within the residential parts of Kabwe (Caravanos et al., 2014; http://bmrplc.com/tailings_stockpiles.php). As a result, we expect pixels with strong spectral matches to the slag pile to be contaminated and thus to provide verification of the effectiveness of the spectral index for detecting regions with a high probability of contamination in Kabwe.

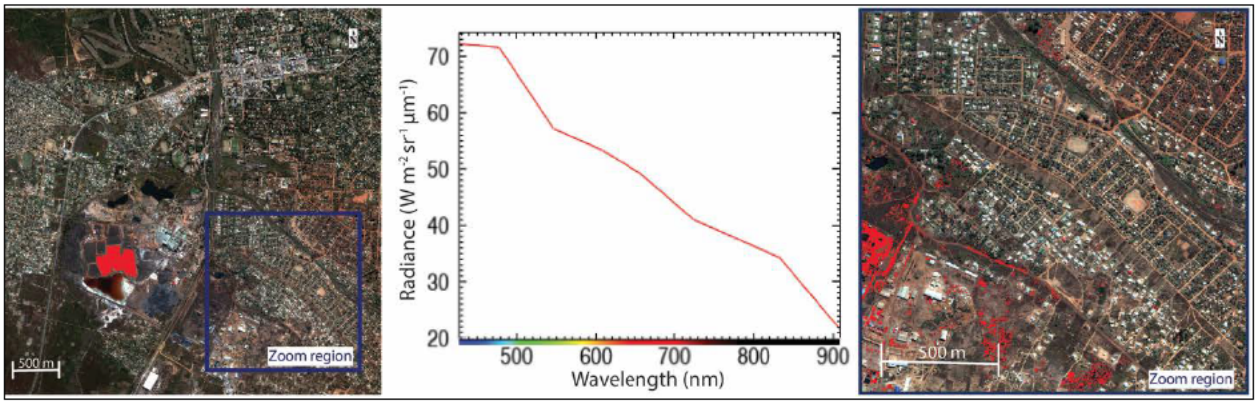

Figure 9. SAM classification of the region of interest image using a training spectrum (middle panel) derived from the mean spectrum of a second region of interest defined within the slag pile in the center of Kabwe. Maximum spectral angle set to 0.10 radians. Red colored regions in the zoomed in image are good spectral matches to the mean spectrum of the slag pile ROI. These areas are expected to be highly contaminated.

The results of the change detection difference map computation are shown in Figure 10. There is an 80.22% overlap between the two methods as determined by the number of pixels in the spectral index-derived classification that "change" to the SAM classification, compared to the total number of pixels in the SAM classification. The spectral index-derived classification classifies nearly twice (85.36%) as many pixels as the SAM classification result. Nearly all of these additional classified pixels are in the residential neighborhoods and their surroundings within the greater region of interest (Figure 10c). Although fieldwork in these areas has shown that there are contaminated places (e.g., the ground-truth data used in this study and provided by the NGO Pure Earth), these pixels are not classified by the SAM classification because they are spectrally distinct from the slag pile. In contrast, the spectral index-derived classification identifies both the highly contaminated areas nearby the slag pile with an 80.22% overlap with the SAM results (Figure 10c, white regions) and likely contaminated residential areas with 63.24% accuracy (Table 8). This shows, similar to the results discussed above, that the spectral index is a more effective classification method than other established methods (e.g., SAM). The strong overlap between the results of the two methods in the more limited area classified by SAM is also a good verification of the applicability of the spectral index classification to the identification of areas in and around Kabwe, Zambia with a high probability of being contaminated by Pb.

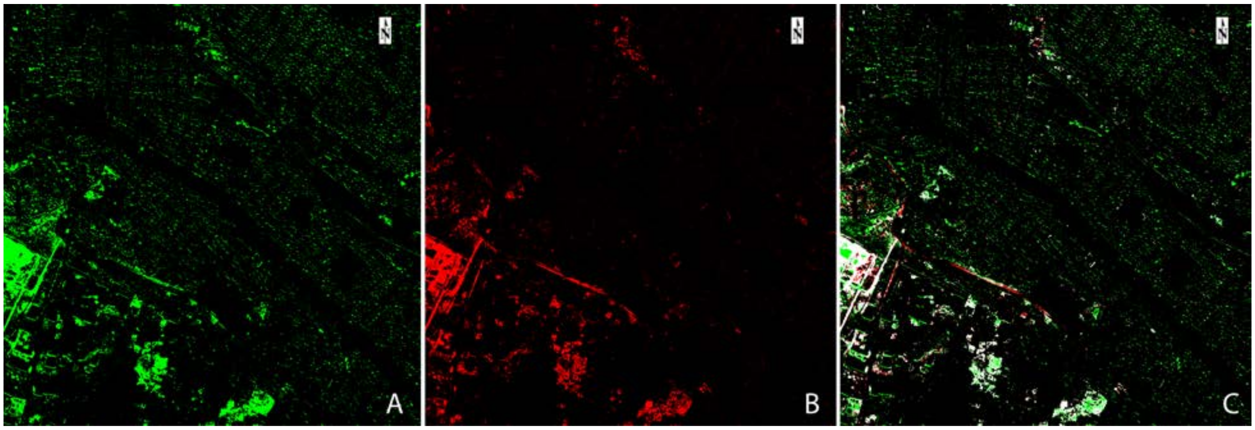

Figure 10. (A) A classified image displaying only pixels with spectral index values between: -0.60 and -0.175 normalized radiance units, 6.86% of the analyzed pixels (green). (B) Locations identified by SAM classification as good matches for the mean spectrum of the slag pile (red). (C) A comparison map showing both regions overlain on one another. White regions in (C) correspond to perfect overlap between the two methods (80.22% of pixels). Green and red regions in (C) correspond to areas where the spectral index (green) and SAM (red) classifications differ from one another.

Based on previous laboratory and remote sensing work (e.g., Pandit et al., 2010; Pascucci et al., 2012; Al Maliki et al., 2014), our initial hypothesis was that there would be a detectable, individual spectral feature or a change in the radiance of a single band associated with increasing Pb concentration in soil. However, like Choe et al. (2008), we find that broad spectral trends are more useful for investigating image-scale contaminant distributions. Three factors likely contribute to this result. The first, as pointed out in Choe et al. (2008), is likely differences in scale between field XRF point data and two-dimensional (on the ground) imaging spectrometer pixels containing components other than contaminated soil. This pixel-scale mixing changes the spectral signatures of individual pixels and complicates analysis at the image scale, even with very high spatial resolution data, such as WV-2. The second is the limited information about spectral feature changes (symmetry, band center shifts, etc.) that can be gleaned from multi-spectral (e.g., 8-band) imaging spectrometer data. The third factor is the groundcover similarity throughout the neighborhood investigated by Pure Earth using XRF. Trees and buildings dominate the spectral character of the areas of densest sampling in and around the surveyed neighborhood. This limited the number of useful sites not dominated by either vegetation or buildings. It may also limit the analysis, which is dominated by the spectral character of this particular area. With a larger sample encompassing a greater variety of groundcover types, it is possible that stronger differences arising from Pb contamination changes could be detected. In future field data collection seasons, NGOs could be advised to collect data with more consideration of possible remote sensing applications, such as collecting data from a variety of groundcover types and including background measurements some distance from the contaminated area for image analysis controls. Collection of this type of ground control data would help to advance the semi-quantitative result presented here to a more fully quantitative method for the remote detection of Pb contamination in soil.

4. Conclusion

We have examined the possibility of combining *in-situ* XRF data with satellite imaging spectrometer data to produce a calibrated remote sensing method for the detection of Pb contamination in a highly contaminated site in Zambia. Using established remote sensing methods (SAM and minimum distance classification), we were unable to accurately identify contaminated areas within the image, relative to ground-truth. However, these methods revealed a trend associating decreasing radiance at 658.8 nm, compared to steady radiance at 477.9 nm, that was associated with an apparent threshold at 1000 ppm Pb concentration in soil. Based on this spectral trend, we developed a spectral index that showed a statistically significant, but small correlation with Pb concentration in soil. Using this spectral index to classify the image based on whether the index values are positive or negative correctly classifies 63.38% of the highest Pb data points and 67.39% of the lowest Pb data points. Derived confusion matrices using regions of interest based on ground-truth XRF data, showed that the spectral index has an overall accuracy of 64.91% for areas with very high ($[Pb]_{soil} \geq 3501$ ppm) and very low ($50 \leq [Pb]_{soil} \leq 550$ ppm) Pb contamination levels; and an overall accuracy of 56.63% for intermediate ($2001 \leq [Pb] \leq 3500$ ppm) and low ($551 \leq [Pb] \leq 1000$ ppm) areas of Pb contamination. The spectral index-derived classification shows greatly increased overall accuracy relative to established classification methods. In the case of comparable SAM classification using only two regions of interest with a Pb concentration threshold at 1000 ppm in soil, the spectral index improves the overall accuracy by 24.56%. The spectral index-derived classification can identify probable areas of contamination with a 65% success rate and provides an accurate picture of the contamination distribution in Kabwe to that level of confidence. The accuracy of the spectral index-derived classification is further supported by comparison with the results of a SAM classification using the mean spectrum of the most highly contaminated area in Kabwe, the slag pile. There is an

80.22% overlap between the two methods, though the spectral index-derived classification is more effective because it can identify contamination in regions not spectrally similar to the slag pile, which the SAM method cannot. Although the spectral index is significantly associated with Pb concentration in soil, there is too much variance in the spectral image data to use the index to directly infer quantitative Pb concentration information for specific regions. However, the index-derived classification is effective at identifying regions with a high probability of being contaminated. The results of this work revealed concrete steps that can be taken in future research to acquire field data in ways that would help advance this analysis to a fully quantitative method for remotely detecting Pb contamination in soil. In particular, avoiding heavy vegetation, collecting ground-truth control points in uncontaminated locations where groundcover in similar to contaminated areas, and collecting ground-truth measurements on a wider variety of groundcover types would all help to improve future work similar to the analysis presented here. This work is an important first step toward the quantitative remote sensing of Pb contamination and a significant improvement over the results from established methods.

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Remote Sensing of Lead (Pb) Contaminated Surface Soil

LOCATION

14 December 2018
12 pm - 5 pm
World Bank, Room MC7-860
1818 H Street NW
Washington, DC

SUMMARY

This half-day workshop will assess the feasibility of utilizing remote sensing techniques to identify elevated surface soil lead (Pb) concentrations globally, in an effort to target strategies for mitigating this toxin and improving human health. This initial workshop is intended to be small in size with the purpose of identifying gaps in existing knowledge while formulating an approach to address those gaps.

AGENDA

- **12:00-12:45 PM**
  Welcome – Benoit Blarel
  Light Lunch and Introductions
- **12:45-1:00 PM**
  USAID's Support for Site Assessment – Katherine Swanson
  Lead Exposure Globally & Workshop Goals – Bret Ericson
- **1:00-1:15 PM**
  Current Approaches – Gabriel Filippelli
- **1:15-3:00 PM**
  Breakout Groups (Objectives & Roadblocks)
- **3:00-3:15 PM**
  Short Break
- **3:00- 4:30 PM**
  Report Back and Group Discussion
- **4:30 PM**
  Close of Workshop

LEAD EXPOSURE

Lead exposure can result in a number of adverse health outcomes including neurological decrement and increased risk of cardiovascular disease (ATSDR, 2007). Adverse social outcomes associated with lead exposure include decreased economic output and increased rates of aggravated assault (Attina and Trasande, 2013; Mielke and Zahran, 2012). The Institute for Health Metrics and Evaluation currently estimates a lead-attributable global disease burden of 14 million Disability Adjusted Life Years, however recent studies have indicated that this may be an underestimate (Bellinger, 2018; Grandjean and Landrigan, 2006; IHME, 2017; Lanphear et al., 2018). The most significant source of historical lead contamination has been the widespread use of leaded petrol, which has been phased out in nearly all countries (Bollhöfer and Rosman, 2001). In high income countries, particularly in the United States, the use of lead-based enamel paints also resulted in high biological burdens of lead (Schwartz and Levin, 1991). In low-and middle-income countries (LMICs), the sources and extent of lead contamination are much less well documented. Recent events of population-wide poisoning indicate that key sources of contamination include informal used lead acid battery (ULAB) recycling and both formal and informal mining and smelting operations (Bose-O'Reilly et al., 2017; Daniell et al., 2015; Haefliger et al., 2009; Lo et al., 2012; Pebe et al., 2008; Filippelli and Taylor, 2018).

IDENTIFICATION OF LEAD CONTAMINATED SITES

The extent and severity of lead contaminated sites in high-income countries is generally well understood (European Environment Agency., 1998; US EPA, OSWER, 2018). In LMICs, far less is known (Brandon, 2013). High priority locations in LMICs are typically identified in an ad hoc manner through press reports, civil society organizations, academic research or other methods. Few systemic efforts are in place. Likely the largest such effort is the Toxic Sites Identification Program (TSIP) implemented by Pure Earth and supported by the European Commission, USAID, and DFID through the World Bank's Pollution Management for Environmental Health Program, among others (Pure Earth, 2018). The effort is executed jointly with country governments and employs national investigators to identify and assess sites using a tailored protocol (Ericson et al., 2013). Since 2008, 1,056 lead contaminated sites in 51 countries have been visited and assessed as part of the TSIP. Collectively these sites place the health of an estimated 8 million people at risk. While the TSIP has made a significant contribution to understanding the disease burden related to contaminated sites, the results likely represent a fraction of the overall extent (Dowling et al., 2016).

Remote sensing is the acquisition of information about an area through the collection and interpretation of reflected radiation. Remote sensing approaches have been employed successfully by development organizations to understand a disparate set of phenomena, including flooding in Bangladesh, artisanal small-scale gold mining in the Brazilian Amazon, and informal settlement development in Manila (Blasco et al., 1992; Lobo et al., 2016; Singh and Gadgil, 2017). When used jointly with machine learning, remote sensing approaches offer a potentially powerful tool for rapid, cost-effective data collection and interpretation. In the case of understanding lead-contamination these approaches have had limited application. Perhaps most relevantly, Pandit, Filippelli and Li (2010) explored the use reflectance spectroscopy to assess soil lead concentrations in Indianapolis, USA. In measuring concentrations of lead, zinc, manganese and copper, with both reflectance spectrometry and inductively coupled plasma-atomic-emission spectrometry the authors found a correlation coefficient of 0.992 when evaluating lead.

STRUCTURE OF THE WORKSHOP

The five-hour workshop will focus specifically on identifying the steps required to employ remote sensing to catalog elevated surface soil lead concentrations globally. Gaps will be identified and proposals to address those gaps will be formulated. The workshop will last 5 hours and engage 10-15 relevant experts. Tentative attendees are listed below.

CO-HOSTS

- Bret Ericson, Chief Operating Officer, Pure Earth (bret@pureearth.org)
- Ernesto Sanchez Triana, Lead Environmental Specialist, World Bank
- Gabriel Filippelli, Professor of Earth Sciences and Director of the Center for Urban Health, Indiana University (gfilippe@iu.edu)

CONFIRMED ATTENDEES

- Alexander van Geen, Research Professor, Lamont-Doherty Earth Observatory, Columbia University
- Carol Sumkin, Chief Development Officer, Pure Earth
- Dave Luo, Anthropocene Labs
- Franziska Landes, PhD candidate, Columbia University

- Gregory Druschel, Professor of Earth Sciences, Indiana University
- Hamed Alemohammad, Lead Geospatial Data Scientist, Radiant Earth
- Jack Caravanos, Clinical Professor, School of Global Public Health, NYU
- Jonathan Pitts, John Aldridge, Mischa Shattuck, Adam Norige, MIT Lincoln Laboratory
- John Keith, Lead Technical Advisor, Pure Earth
- Katherine Swanson, Office of Energy and Infrastructure, USAID
- Mary Jean Brown, Adjunct Assistant Professor, School of Public Health, Harvard University
- Matthew Jelacic, Infrastructure Policy Advisor, USAID
- Victoria Gammino, Chief Science Officer, Radiant Earth

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Appendix D – Link to supplemental materials

Much of the supplemental material for the studies presented in this thesis is comprised of large datasets not amenable to inclusion in this format. Supplemental materials have therefore been stored in an online Dropbox folder, organized by publication, and accessible at the following link:

https://www.dropbox.com/sh/cmt17ps8q8595sa/AACY4N8lNldf1O5GpL3Hb897a?dl=0