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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 μg/ft^2 (95%CI: 32–97) and 9.7 μg/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 μg/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).

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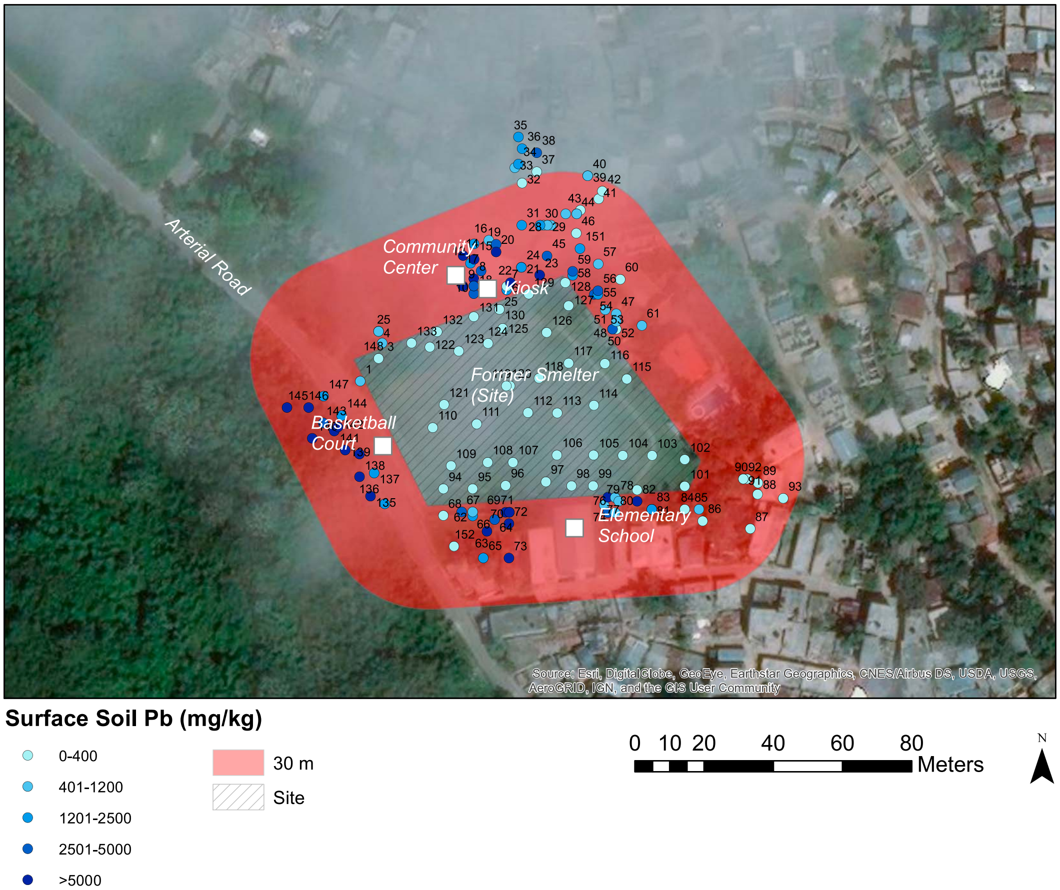

Figure 1. Overhead map of the former site of the MetaloXsa Smelter (Paraiso de Dios, Haina, Dominican Republic) and exposure scenarios used in this analysis. Soil lead concentrations displayed here were analyzed after the onsite remediation and before the offsite work.

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.

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):

$$\text{dust lead concentration}, \mu g/g = 50.96 \times \left(\text{dust lead loading}, \mu g/ft^2\right)^{0.6553}$$

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).

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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		
Population		Soil Pb (mg/kg)	Dust Pb (μg/g)	BLL μg/dL	20 years	35 years	50 years	20 years	35 years	50 years
Attributable 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)
			Subtotal (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)
Attributable 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)
			Subtotal (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)
			Total 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		
Population		Soil Pb (mg/kg)	Dust Pb (μg/g)	BLL μg/dL	20 years	35 years	50 years	20 years	35 years	50 years
Attributable to exposures from the site										
Group 1	13	8,964	4174	28.94 (28.25–29.64)	60.94 (60.61–61.24)	87.96 (87.48–88.4)	105.3 (104.73–105.83)	81.76 (81.32–82.17)	143.08 (142.3–143.8)	204.4 (203.29–205.43)
Group 2	8	4,409	780	20.59 (19.93–21.25)	29.51 (29.13–29.88)	42.6 (42.05–43.12)	51 (50.34–51.63)	39.6 (39.09–40.08)	69.3 (68.4–70.15)	99 (97.72–100.21)
Group 3	52	6,337	3818	24.79 (24.16–25.42)	232.19 (230.4–233.83)	335.14 (332.56–337.51)	401.22 (398.13–404.06)	311.52 (309.12–313.73)	545.16 (540.96–549.02)	778.8 (772.8–784.32)
Group 5	10	7,869	4026	28.88 (27.58–30.19)	0.66 (0.65–0.66)	0.95 (0.93–0.96)	1.13 (1.12–1.15)	0.88 (0.87–0.89)	1.54 (1.52–1.56)	2.2 (2.17–2.23)
Group 6	83	2,950	582	11.77 (11.24–12.29)	3.36 (3.22–3.48)	4.84 (4.65–5.03)	5.8 (5.57–6.02)	4.5 (4.32–4.67)	7.88 (7.57–8.18)	11.25 (10.81–11.69)
Group 7	10	5,242	3670	19.74 (18.44–21.04)	0.57 (0.55–0.59)	0.83 (0.8–0.86)	0.99 (0.96–1.02)	0.77 (0.74–0.79)	1.35 (1.3–1.39)	1.93 (1.86–1.99)
		Subtotal (site)			327.22 (324.56–329.69)	472.31 (468.47–475.88)	565.44 (560.85–569.71)	439.03 (435.46–442.34)	768.3 (762.05–774.1)	1097.57 (1088.65–1105.86)
Attributable to background exposures										
Group 4	74	323	226	3.48 (3.3–3.65)	0.01 (0–0.06)	0.01 (0–0.08)	0.01 (0–0.1)	0.01 (0–0.08)	0.02 (0–0.13)	0.03 (0–0.19)
Group 8	103	323	226	2.62 (2.17–3.08)	0.4 (0.26–0.58)	0.58 (0.37–0.83)	0.7 (0.44–1)	0.54 (0.34–0.77)	0.95 (0.6–1.36)	1.36 (0.86–1.94)
		Subtotal (offsite)			0.41 (0.26–0.63)	0.6 (0.37–0.92)	0.71 (0.44–1.1)	0.55 (0.34–0.85)	0.97 (0.6–1.49)	1.38 (0.86–2.13)
		Total DALYs averted			326.81 (324.3–329.06)	471.72 (468.11–474.96)	564.73 (560.4–568.62)	438.48 (435.12–441.49)	767.33 (761.45–772.61)	1096.19 (1087.79–1103.73)

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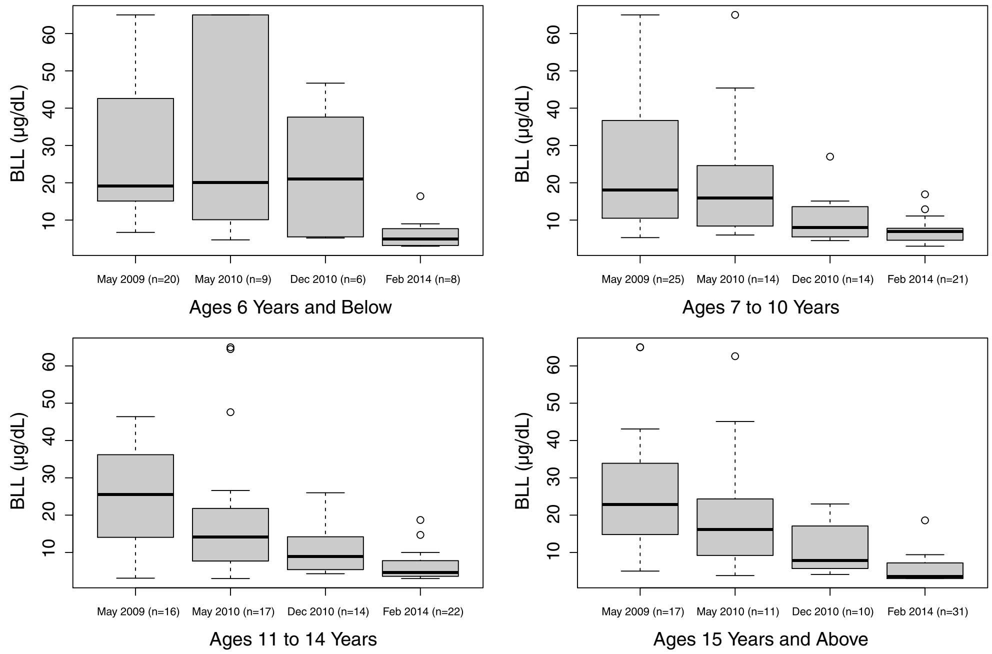

Figure 2. Blood lead level analysis results (median, IQR, and outliers) from 2009 to 2014 stratified according to age group.

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.

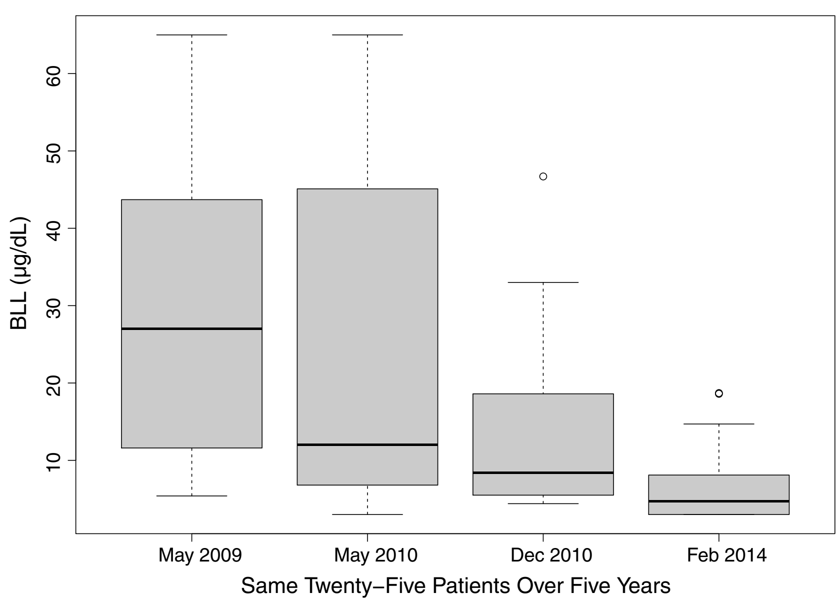

Figure 3. Blood lead levels (median, IQR, and outliers) for the same 25 residents over the 5 year period evaluated in this study (Median age in 2009 = 10 years).

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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