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Research article

Exposure and health risk assessment from consumption of Pb contaminated water in Addis Ababa, Ethiopia

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ABSTRACT

Exposure to lead (Pb) through drinking water has been linked to adverse health outcomes. Children are particularly susceptible. This study was designed to measure Pb contamination level in drinking water of the Ethiopian city Addis Ababa and assess the associated health risks. Eighty-eight fully-flushed drinking water samples were collected from all ten sub-cities of Addis Ababa. Pb concentration was measured using an Inductively Coupled Plasma Mass Spectrometer (ICP-MS). The chronic daily intake (CDI), the hazard quotient (HQ), and the cancer risk (CR) of Pb were determined to assess exposure levels and health effects. Blood lead level (B–Pb) for children was modelled using the integrated exposure uptake biokinetic model (IEUBK). The mean concentration of Pb in the drinking water was 17.8 μg/l, where >50% of the samples exceeded WHO's 10 μg/l guideline. Significant spatial variation of Pb was noticed among sub-cities. The mean CDI was 1.43 and 0.59 μg/kg/day for children and adults, respectively. The HQ showed that 8% of children and 2.3% of adults exceeded the safe limit. The predicted geometric mean of B–Pb ranged from 3.23 to 14.65 μg/dl. The risk of a child having a B–Pb level >5 μg/dl at the median water Pb concentration (10.5 μg/l) was estimated at 13.4%. Based on the 95th percentile Pb concentration (75.1 μg/l), 89.6% of children would have B–Pb levels above the 5 μg/dl threshold. The estimated CR was found in the range of 1×10^{-7} to 9.9×10^{-5}; hence cancer risks are not a concern. The study concluded that Addis Ababa's drinking water is likely to be a source of lead exposure where consumers at specific city locations are at risk of numerous non-cancer health effects. The impacts are expected to be severe in the Ethiopian context; hence further investigations and coordinated interventions are required.

1. Introduction

Lead (Pb) is a naturally occurring element abundantly found throughout the earth (NIH 2020). It is one of the heavy metals with no known physiologically relevant role in the body (Wani et al., 2015). However, toxicants can cause adverse health effects, given excessive exposure and accumulation in the body (Shefa and Héroux 2017; Zhou et al., 2020). The poisoning effects of Pb have been recognized for several centuries (ATSDR 2020). Many body organs and systems are potential targets for Pb. These include effects on blood production, the nervous system, the kidneys and reproduction, bladder, cardiovascular, hepatic, endocrinal, and gastrointestinal effects (Awadalla et al., 2020; Devóz et al., 2021). Pb exposure during pregnancy has also been associated with gestational hypertension, miscarriage, premature birth and low birth weight (Committee on Obstetric Practice, 2012; Custodio et al., 2020). The main target for Pb toxicity is the nervous system, both in adults and children (Lim et al., 2013). It can create irreversible intellectual impairment in infants and young children, even at blood lead (B–Pb) levels below 10 μg/dl (Naranjo et al., 2020).

There is no known safe blood lead level. Nevertheless, it is known that, as lead exposure increases, the magnitude and severity of symptoms and effects also increases. CDC uses a B–Pb reference value of 5 μg per deciliter (μg/dl) to identify children with B–Pb levels that are much higher than most children's levels (CDC 2020a). The International Agency for Research on Cancer has also determined that inorganic Pb and its compounds are likely carcinogenic in humans, classified under "group 2A" (IARC 2006). Although human exposure to Pb can occur via food, water, air, soil, and dust (SCHER 2009), a more significant proportion of

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the current Pb burden is attributable to Pb in drinking water (Ravenscroft et al., 2018). Many studies detected elevated waterborne Pb levels exceeding the guideline value (Sarvestani and Aghasi 2019; Redmon et al., 2020). In addition, severe outbreaks and elevated B–Pb levels from Pb-contaminated drinking water have also been reported from different parts of the world (Edwards et al., 2009; Zahran et al., 2020). The European Food Safety Authority (EFSA) estimated that drinking water contributes up to 20% of an adult's total exposure and overall lead burden (EFSA 2010). In contrast, infants who consume primarily formula prepared with tap water can receive 40–60% of their Pb exposure from drinking water (US EPA 2016a).

Lead may enter into a drinking water system either from environmental contaminations or arise from the distribution system itself (Kim et al., 2011; L. et al., 2021). Primary sources for drinking water are from household plumbing systems in which the pipes fixtures, solder, or service connections to homes may contain Pb (WHO 2019; Water Research Center 2020). WHO recommends a provisional guideline value of 10 μg/l for drinking water to minimize human exposure to Pb (WHO 2017). Ethiopia also adopted this value as a compulsory standard (ESA 2013).

Prevention of lead poisoning requires improved methods to identify and provide interventions to groups at high risk (Meyer et al., 2003). Exposure and health risk assessment is widely used to measure the level of exposure and the nature and probability of adverse health effects from waterborne Pb (ILA 2017). Simulation tools like the US EPA Integrated Exposure Uptake Biokinetic Model (IEUBK) can further provide more site-specific data in the risk assessment process.

Ethiopia is one of the countries where Pb is a public health concern (WHO 2015). Few biomonitoring studies conducted in Ethiopia documented high B–Pb levels with different exposure pathways, such as air pollution (Chercos and Moges 2016) and occupational exposure (Gebrie et al., 2014). One study also reported high Pb deposits on children's teeth in Addis Ababa (Tvinnereim et al., 2011). Even though drinking water is an important source of Pb exposure (Ravenscroft et al., 2018), the relative contribution of drinking water to the overall Pb-related disease burden in Addis Ababa has not yet been assessed. Meanwhile, many Ethiopian cities, including the capital Addis Ababa, are believed to have old drinking water distribution systems and pipelines that may contain Pb. Therefore, this study aimed to evaluate the compliance status of waterborne Pb against the standard and quantify the degree of exposure and the associated health risks from the consumption of Pb contaminated drinking water in Addis Ababa. The findings will help to develop a guideline and establish regulations on Pb level in drinking waters to intervene in the toxic and adverse health effects of Pb.

2. Materials and methods

2.1. Study area

This study was conducted in Addis Ababa, the capital of Ethiopia and the seat of the African Union, geographically positioned between 9°1′48″N latitude and 38°44′24″E longitude. The city lies at an average elevation of 2,500 m above sea level and occupies a total area of 540 km^2. Addis Ababa has been structured into ten boroughs, called sub-cities (Addis Ababa City Government 2017), and is home to about 5 million people (World Population Review 2021). The city provides its water supply from both surface water and groundwater sources through three main drinking water supply sub-systems: Legedadi, Gefersa, and the Akaki sub-system (Solomon 2014).

2.2. Sampling

Water samples were collected from 88 private taps located in the ten sub-cities of Addis Ababa between the 20th of November 2019 and the 10th of January 2020 (Figure 1). Based on the WHO recommendation on sampling locations for actual human exposure assessment (WHO 2003), all water samples in this study were taken from taps inside residential houses rather than from drinking-water sources. Two types of water samples were taken from the taps: the first drops of cold water after an overnight stagnation (first-flush sample), and the fully-flushed (second-flush) sample, where samples were taken after flushing for 2 min. However, Pb concentration data and risk calculations in this paper are based on the fully-flushed samples only.

The samples were collected in 500 ml capacity high-density polyethylene (HDPE) bottles, which were thoroughly cleaned and soaked in 10% nitric acid for two days and then rinsed with distilled water before use. All sample bottles were labeled correctly and stored at 4 °C until the laboratory analysis took place. The sampling procedure followed the US EPA's Quick Guide to Drinking Water Sample Collection (US EPA 2016b).

2.3. Water sample analysis

Conductivity, pH, and water temperature were measured on-site directly after sampling using Oakton Portable meter kit 600 series. Analysis for total Pb was done using an Inductively Coupled Plasma Mass Spectrometer (ICP-MS, PerkinElmer NexION system with Syngistix software) at the Laboratory of Analytical Chemistry and Applied Ecochemistry of Ghent University, Belgium. The analytical procedure followed the USEPA's "protocol for determining trace elements in waters and wastes by ICP-MS" (US EPA 2019).

All quality control workflows of the ICP-MS instrument and Syngistix software were maintained to ensure the analysis's data integrity, basically including instrument calibration and running quality control samples. Calibration solutions were prepared with a multi-element standard solution in 1% pico-pure nitric acid and run between samples during the analysis. Spikes were prepared at a 10 and 50 μg/l concentration by mixing 2 ml of sample with 2 ml standard solution. Blanks and internal standards were also analyzed alongside the samples.

2.4. Exposure and health risk assessment

A human health risk assessment is the process to predict the nature and probability of undesirable health effects in individuals who may be exposed to chemicals in contaminated environmental media, now or in the future (US EPA 2014). The risk assessment is a very important tool in decision making and in the process of risk management (ILA 2017).

2.4.1. Exposure through ingestion

The human exposure to Pb through drinking water was calculated as a chronic daily intake (CDI), using Eq. (1), which is adopted from the United States of America Environmental Protection Agency (US EPA 1989). In addition, the method of determination was used as mentioned by Giri and Singh (2015) and Mirzabeygi et al. (2017).

$$\mathrm{CDI\ (mg/kg/day)} = (\mathrm{CW} \times \mathrm{IR} \times \mathrm{EF} \times \mathrm{ED}) / (\mathrm{BW} \times \mathrm{AT}) \tag{1}$$

For exposure and risk calculation in this study, an average body weight of 60 kg estimated for African adults (Walpole et al., 2012) was used. This was also supported by available body weight data in Ethiopia, ranging from 54.1 kg (EPHI 2016) to 64.65 kg (Sinaga et al., 2019). For children aged 7 and below, an average body weight of 12.5 kg was used based on average weight reports of 11.1 kg (Pollock et al., 2007) and 13.9 kg (Stefano and De Angelis 2009). Details of the parameters applied in the exposure assessment are given in Table 1.

2.4.2. Risk characterization

The non-cancer hazard quotient (HQ), given in Eq. (2), assumes that there is a level of exposure (i.e., RfD) below which it is unlikely for even sensitive populations to experience adverse health effects. If the exposure level (chronic daily intake) exceeds this threshold (i.e., if CDI/RfD exceeds unity), there may be a concern for potential non-cancer effects. As a rule, the greater the value of CDI/RfD above unity, the greater the level of concern (US EPA 2015a).

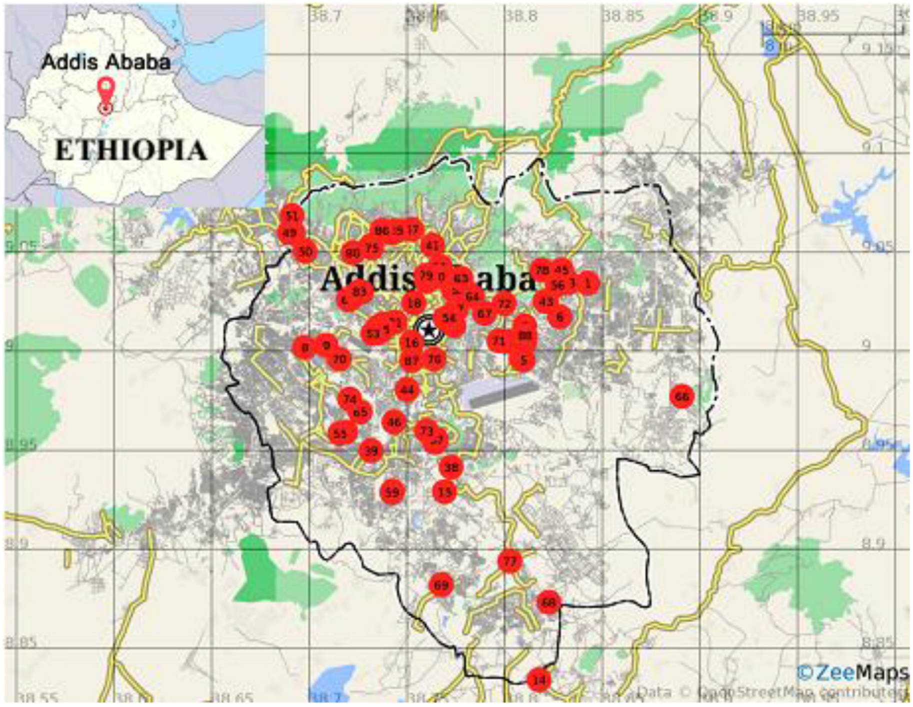

Figure 1. Map of water sampling sites in Addis Ababa city.

The Oral RfD in this study was assumed to be 0.0035 mg/kg/day, based on the Provisional Weekly Tolerable Intake (PWTI) of 25 μg of Pb per kg of body weight which corresponds to 3.5 μg/kg/day, as established by the Joint FAO/WHO Expert Committee on Food Additives (JECFA) (WHO 1993, 2003). This level of RfD was also used in other studies (Lim et al., 2013; Wongsasuluk et al., 2014; Mirzabeygi et al., 2017; Liang et al., 2017).

$$HQ = CDI/RfD \tag{2}$$

Results from the hazard quotient in Eq. (2) were interpreted into three levels of health risks:

Table 1. Input parameters used to characterize exposure and risk.

Parameter	Description	Unit	Value used	References
CW	Site-specific concentration of the chemical (Pb) in water	mg/l	Variable, ranged from 3.6×10^{-4} to 0.146	
IR	Ingestion rate	Liters/day	Adults = 2L	US EPA (2015a)
			Children = 1L	
EF	Exposure frequency	Days/year	365	
ED	Exposure duration	Years	70	
BW	Bodyweight	Kg	Adults = 60	Walpole et al. (2012)
			Children = 12.5	Pollock et al. (2007); Stefano and De Angelis (2009)
AT	Average time over which exposure is averaged	Days	$70 \times 365 =$ 25,550	US EPA (2015a)
RfD	Reference dose	mg/kg/day	0.0035	WHO (1993, 2003)
CSF	Cancer slop factor	mg/kg/day	0.0085	OEHHA (2015)

- HQ < 1 L: low, HQ = 1 M: medium, and HQ > 1 H: high.

The cancer risk (CR) estimation of Pb from ingestion of drinking water (Eq. 3) was calculated according to the US EPA's risk assessment manual (US EPA 1989). CR is estimated as the incremental probability of an individual developing cancer over a lifetime. The typical range of cancer risk value targeted by the US EPA for risk management is 10^{-6} to 10^{-4}, which is thought to be public-health protective (US EPA 2005). However, there is no consensus CSF for inorganic Pb (US EPA 2015b). Hence, the CSF in this study was based on the California Office of Environmental Health Hazard Assessment (OEHHA 2015). This same CSF value was also used in recent studies (Mohammadi et al., 2019; Sarvestani and Aghasi 2019; Khalid et al., 2020).

$$CR = CDI \times CSF \tag{3}$$

2.5. Estimation of B–Pb level

As most of the available information on human exposure and the health effects of Pb is based on B–Pb (EFSA 2010), it was rational to incorporate blood data in this study. B–Pb for children under seven years of age were estimated using the US EPA's Integrated Exposure Uptake Biokinetic (IEUBK) model (US EPA 2007), in the same way as applied in other related studies (Deshommes et al., 2013; Jarvis et al., 2018). The IEUBKwin1_1Build11 version was run under 'Batch Mode' option to address the different media concentrations for each specific location where the tap-waters were sampled. For the water data input, a variable Pb concentration in drinking water from all sampling locations ranging from 0.36 to 146.1 μg/l was used as the primary site-specific input in the modelling process. The IEUBK default outdoor lead concentration (0.1 μg/m^3) was applied for the air data input, which was also supported by available air quality data for Addis Ababa (Etyemezian et al., 2005; IQAir 2019). Two options were assumed regarding soil/dust input data: the IEUBK default value, and soil data reported by Endale et al. (2012). According to the study report, the average concentrations of Pb in the roadside soils in Addis Ababa were 418.6 ± 3.4 μg/g (Endale et al.,

Table 2. Summary of the studied parameters (N = 88).

		Pb (μg/l)	pH	Conductivity (μS/cm)	Temperature (°C)
Mean		17.8	7.8	283.1	19.2
SD		23.4	0.3	191.8	1.6
Minimum		0.36	6.9	86.8	15.8
Maximum		146.1	8.9	603	24
Percentiles	25	3.6	7.5	130.1	18.4
	50	10.5	7.8	136.2	19
	75	24.1	7.9	490.2	19.9
	95	75.2	8.3	595.5	22.8

2012). Because the soil/dust samples and the water data did not share exactly the same sampling location, the B–Pb estimation using this data was done separately, as a scenario, shifting with the IEUBK default value. The CDC current reference value of 5 μg/dl (CDC 2020a) and the typical 10 μg/dl level of concern used in the IEUBK model (US EPA 2007) were used as cut-off points for B–Pb level to analyze the proportion of exceedance.

2.6. Data analysis

One-way analysis of variance (ANOVA) with Tukey's post hoc test was computed to compare Pb concentration between groups. Pearson correlation analysis was also performed to evaluate the measured parameters' relationship. All statistical analyses were done using IBM SPSS version 20 (SSPS Inc., Chicago, IL, USA). Kernel interpolation in ArcGIS (Version 10.1) was used to construct the contour map for spatial analysis of health risks. Water sampling sites were mapped using Zee Maps.

3. Result and discussion

3.1. Concentration of Pb in drinking water

This study extensively assessed the level of Pb contamination in the drinking water supply system of Addis Ababa. The parameters measured in the water samples were Pb, pH, conductivity, and temperature (Table 2).

Considerable Pb concentration was recorded in the sampled tap waters, ranging from the lowest 0.36 to the highest 146.1 μg/l, with a mean concentration of 17.8 μg/l. As illustrated in Figure 2, more than 50% of the samples surpassed the 10 μg/l provisional guideline set for drinking water. The US EPA claimed that if the Pb concentration exceeded the action level in more than 10% of customer taps sampled, then the entire water system is considered to have exceeded the action level (US EPA 2015c; Masten et al., 2019). The present data, therefore, showed that the water supply system in Addis Ababa is a concern for its high Pb level.

Table 3. Pb concentration in drinking water reported in Ethiopia and Africa.

Water Pb level (μg/l)	Location of the study	Reference
0.36–146.1(range), 17.8(mean)	Addis Ababa, Ethiopia	This study
24.5(mean)	Jimma, Ethiopia	Getaneh et al. (2014)
19.5(mean)	Addis Ababa, Ethiopia	Fite (2008)
152.4–220.2(range)	Jimma, Ethiopia	Siraj and Kitte (2013)
400.0(mean)	Arba Minch, Ethiopia	Guadie et al. (2020)
69–106(range)	Mekelle, Ethiopia	Mebrahtu and Zerabruk (2011)
52–1347(range)	IndaSelasie, Ethiopia	
250.0 and 31.0(mean)	Ghana	Cobbina et al. (2015)
600.0(mean)	South Africa	Elumalai et al. (2017)
0.4–28.6(range)	Cairo	Moawad et al. (2016)

A previous study conducted in Addis Ababa ten years ago recorded a mean Pb concentration of 19.5 μg/l, with a higher proportion of exceedance (85% > 10 μg/l) (Fite 2008). This may indicate the existence of chronic contamination of Pb in the city's water supply system. According to the literature, chronic toxicity of Pb is of most concern when considering the potential risk to human health because of its long half-life in the body (EFSA 2010). Excess Pb level in drinking water sources was also detected in many other parts of the country. A study in Jimma town, Ethiopia, reported a mean concentration of 24.5 μg/l in tap water samples (Getaneh et al., 2014) while data on groundwater sources in the same town showed concentration ranging from 152.4 to 220.2 μg/l. A much higher value was reported from IndaSelasie, another town in Ethiopia, where the maximum concentration in drinking water was 1347 μg/l. The data from different studies reflect that Pb contamination in drinking water sources is a major problem in Ethiopia. A summary of drinking water Pb levels reported by studies in Ethiopia and other African countries is given in Table 3.

3.2. Spatial distribution of Pb in tap water

This study revealed a significant spatial variation of Pb between sub-cities where the water samples were taken, as determined by a one-way ANOVA (F (9, 78) = 3.386, p = 0.001). The highest Pb level was recorded in Bole sub-city, followed by Kolfe Keraniyo and Gulele (Figure 3).

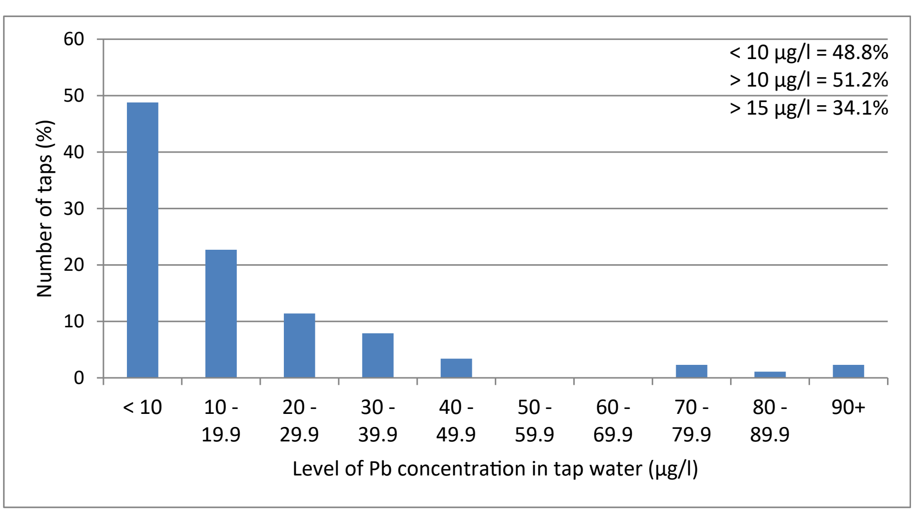

Figure 2. The proportion of private taps per level of Pb contamination in Addis Ababa.

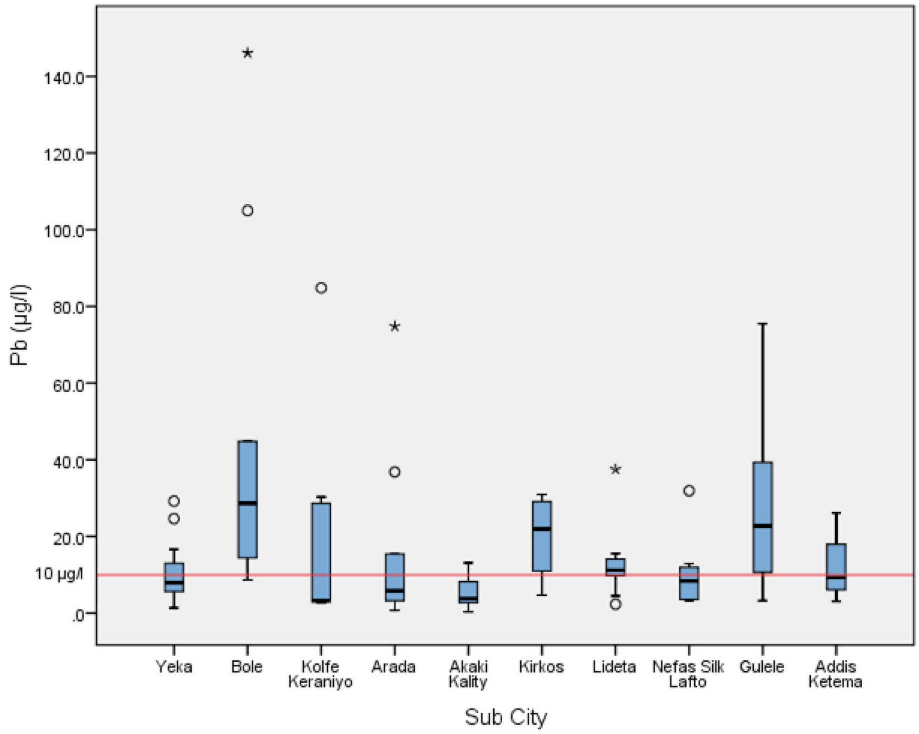

Figure 3. Spatial distribution of Pb (µg/l) in drinking water samples in Addis Ababa.

Table 4. Correlation matrix of the measured parameters (N = 88).

	Pb	°C	EC	pH
Pb	1.00	-0.12	-0.28**	-0.19
°C		1.00	0.28**	-0.02
EC			1.00	0.31**
pH				1.00

** Correlation is significant at the 0.01 level (2-tailed).

Table 5. Level of exposure (CDI), the non-cancer health risk (HQ) and cancer risk (CR) from waterborne Pb in Addis Ababa.

	CDI(mg/kg.day)		HQ		CR	
	Adults	Children	Adults	Children	Adults	Children
Mean	5.95E-04	1.43E-03	0.17	0.41	5.05E-06	1.21E-05
SD	7.82E-04	1.88E-03	0.22	0.54	6.64E-06	1.59E-05
Minimum	1.20E-05	2.88E-05	0.003	0.008	1E-07	2E-07
25th percentile	1.20E04	2.88E-04	0.034	0.08	1.02E-06	2.45E-06
Median	3.48E-04	8.36E-04	0.099	0.24	2.96E-06	7.11E-06
75th percentile	8.04E-04	1.93E-03	0.23	0.55	6.84E-06	1.64E-05
Maximum	4.87E-03	1.17E-02	1.39	3.34	4.14E-05	9.93E-05

The lowest concentration was seen in the Akaki Kality sub-city (0.36 µg/l), where all samples were in the acceptable range, except for two values. The Akakai Kality sub-city is among Addis Ababa's locations where the surface water sources were claimed to have a high pollution status due to extensive industrial discharges (Aschale et al., 2016). However, this was not consistent with the pollution status of the drinking water supply system, as discovered in the present study. It seems that the groundwater sources where the sub-city is supplied from are protected from the industrial effluents, or the existence of cross-contamination of Pb between the surface water and the drinking water supply systems is unlikely. This spatial variation of Pb might be attributed to the types of water sources and reservoirs where these sub-cities get water from, besides other possible factors, including cross-contamination from the nearby environment. Similarly, Sadeghi et al. (2020) reported spatial variability of Pb between surface and groundwater sources. There is no study addressing the specific sources of lead contamination in the drinking water system of Addis Ababa. However, literature documented Pb may enter the drinking water from different possible sources, including household plumbing, home service connections, and environmental contamination (WHO 2019; Water Research Center 2020; L. et al., 2021). Hence, identifying the source of Pb in the water system is a precedence research area.

3.3. Quality indicator parameters (conductivity, pH, and temperature)

The water quality indicator parameters pH, conductivity, and temperature were found within a typical drinking water quality level (Table 2). It was noted that these parameters, except for temperature, considerably vary between sub-cities (Conductivity $F(9, 78) = 32.471$, $p = 0.000$; pH $(9, 78) = 5.593$, $p = 0.000$). Pb exhibited a negative correlation with conductivity, as summarized in Table 4. This inverse correlation between Pb and conductivity could be attributed to the variation of these parameters by the drinking water sources. The groundwater sources in this study showed lower Pb concentration and higher conductivity compared to the surface water sources.

3.4. Exposure and risk characterization

This study is the first to assess the exposure and health risks from waterborne Pb in Addis Ababa. Based on the WHO recommendation on human exposure estimation (WHO 2003), the water Pb level here considered only the consumers' tap concentration, rather than at the

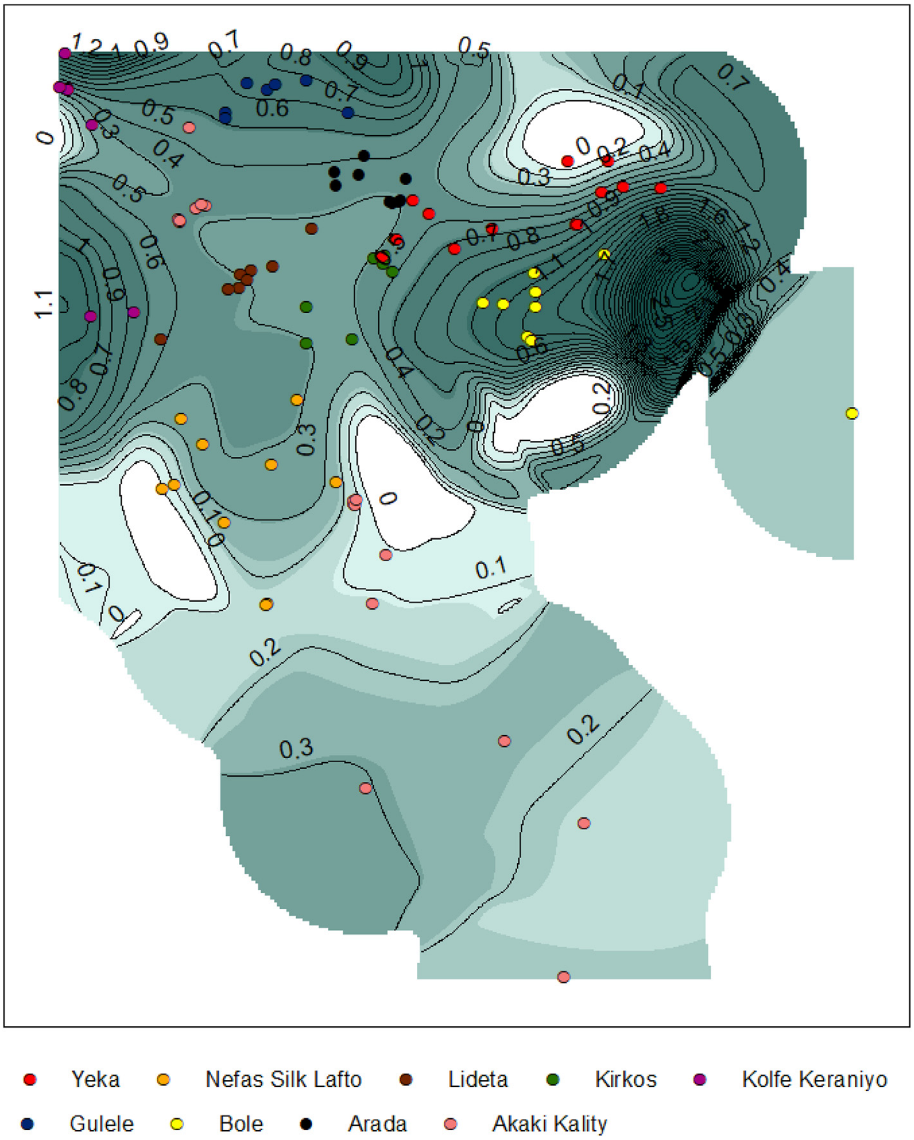

Figure 4. Contour map showing the magnitude and spatial distribution of the HQ value in sub-cities of Addis Ababa.

Table 6. GM of B–Pb levels and proportion of exceedance from IEUBK simulation (for children aged below 7 years).

SCENARIO	Pb in water (μg/l)	A. Simulation with water data alone			B. Simulation with water, soil/dust, and air data		
		Modeled B–Pb (μg/dl)	Exceedance of 10 μg/dl (%)	Exceedance of 5 μg/dl (%)	Modeled B–Pb (μg/dl)	Exceedance of 10 μg/dl (%)	Exceedance of 5 μg/dl (%)
Minimum	0.36	3.2	0.8	17.7	4.47	4.3	40.6
25thpercentile	3.6	3.6	1.57	24.9	4.92	6.5	48.6
Median	10.45	4.3	3.5	36.8	5.59	10.7	59.3
75th percentile	24.13	5.3	9.12	55.6	6.52	18.1	71.4
95th percentile	75.21	9.1	41.71	89.7	10.03	50.2	93
Maximum	146.1	13.8	75.2	98.4	14.65	79.2	98.9

drinking-water sources. Since exposure and risk are site-specific, both measurements were calculated for each of the 88 samples, separately for adults and children (Table 5).

The mean CDI value was calculated to be 0.59 and 1.43 μg/kg/day for adults and children, respectively. These values reflect the mean daily Pb consumption through drinking water. The HQ ranged from 0.003 to 1.4 for adults and 0.008 to 3.34 for children. A higher proportion of unacceptable non-cancer HQ was calculated for children with 8% of exceedance above unity, while adults' proportion was 2.3%.

A comparable result was recorded in a risk assessment study in Ghana with a maximum HQ of 3.6 and 1.7 for children and adults, respectively (Ofosu-Asiedu et al., 2013). A much higher HQ (12.1 for children and 8 for adults) was reported for South Africa (Elumalai et al., 2017). As depicted in Figure 4, sub-cities with higher Pb concentration also

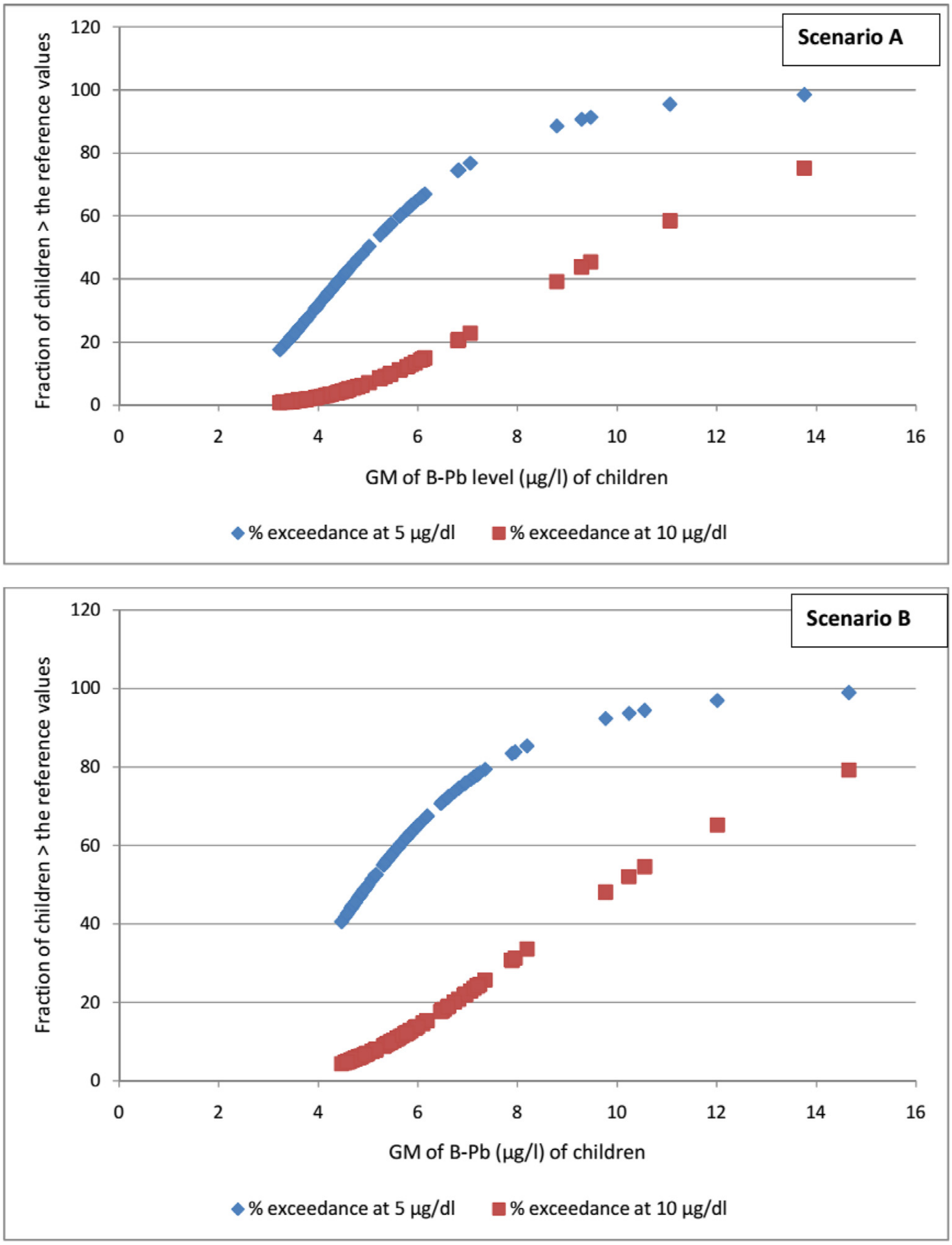

Figure 5. Percentage exceeding the 5 and 10 µg/dl thresholds as function of GM of B–Pb in scenario A and B.

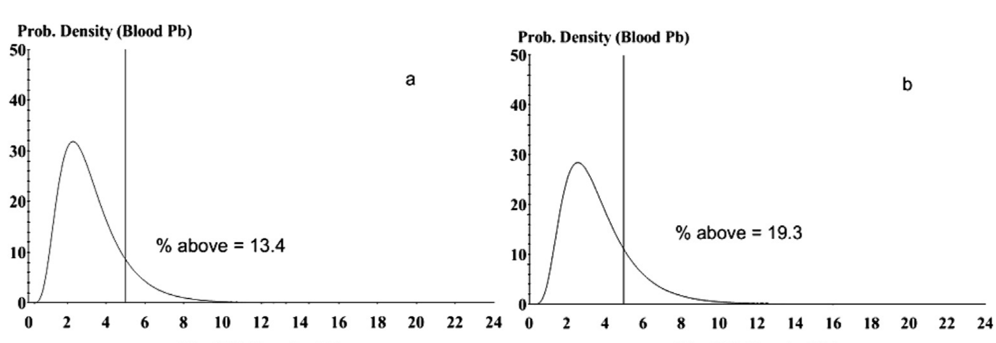

Figure 6. Probability for a child with a risk of exceeding 5 µg/dl (a) at 50th, (b) at 75th percentile Pb concentration (modelling with fully-flushed water data).

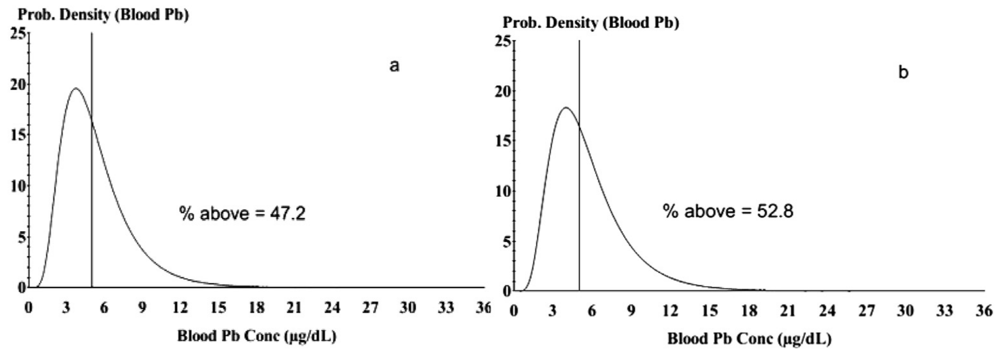

Figure 7. Probability for a child with a risk of exceeding 5 μg/dl (a) at 50th, (b) at 75th percentile Pb concentration (modelling with fully-flushed water data and soil data).

experienced higher HQ values; hence it would be reasonable to give these sites front priority during the design of mitigation measures.

The possible non-cancer health risks that correspond to the observed high-level HQ in this study may include, but are not limited to, cardiovascular effects, increased blood pressure and incidence of hypertension, decreased kidney function, and reproductive problems (US EPA 2016a). As the HQ go far above one, the range and severity of effects also increases (WHO 2019). However, the adverse health effects might also occur even at the low HQ values since there is no safe level of Pb. Infants who drink formulae prepared with lead-contaminated water are considered to be at a higher risk of exposure because of the large amount of water they consume relative to their body size (CDC 2020b).

The estimated CR from the consumption of Pb through drinking water was found in the range of 1×10^{-7} to 9.9×10^{-5}. According to the US EPA scale for carcinogenic health risks (US EPA 2005), 86% of the calculated CR falls in the acceptable level while the rest, 14%, was considered insignificant; hence, the cancer risk can be neglected. Therefore, with the observed level of exposure, carcinogenic health risks from waterborne Pb in Addis Ababa's water supply system should not be considered as a concern.

3.5. Simulated B–Pb level for children aged below 7 years

Results from the IEUBK (Table6) depict the geometric mean (GM) of B–Pb and the proportion of children exceeding the US EPA's and CDC's reference values versus the corresponding Pb concentrations. To give a broad picture of the prediction, results are presented in different scenarios. In Scenario A, only the water data were used while other default values of the IEUBK remain unchanged. Scenario B incorporated available soil/dust and air data besides the water data.

The predicted GM B–Pb level for children aged below 7 years in the study ranged from 3.32 to 13.8 μg/dl in scenario A, and 4.47–14.65 μg/dl in scenario B. It is observed that shifting the IEUBK model from scenario A to B would lift the B–Pb only by an average of 1.23 μg/dl.

Although no actual B–Pb data was found for children in Addis Ababa to evaluate this finding, studies conducted abroad showed observed and predicted GM B–Pb levels not deviating more than 1.0 μg/dl from the values we reported (Zhong et al., 2017). The estimated B–Pb in this study is lower than the actual B–Pb measures in Kenya (median B–Pb = 5.40 μg/dl, range = 3.30–24.70 μg/dl) (Olewe et al., 2009) and Kinshasa (median B–Pb = 11.5 μg/dl, range = 3.0–37.8 μg/dl) (Tuakuila et al., 2012).

For the 95th percentile Pb concentration (75.1 μg/l, scenario A), it was estimated that 41.7% of children would have a B–Pb level above 10 μg/dl while 89.7% would be above 5 μg/dl. At the median Pb concentration of 10.45 μg/l, the proportion of children with B–Pb above 5 μg/dl would be 36.8%, and this would rise to 59.3% if the soil data are considered (scenario B). A study by Jarvis et al. (2018) documented 46% and 98% of exceedance at 5 and 1.8 B–Pb reference values, respectively; while Deshommes et al. (2013) reported 59% of exceedance at 5 reference value. The proportion of the population exceeding the reference values of 5 and 10 μg/dl is depicted in Figure 5.

In addition to the data presented in Table 6, the IEUBK generated a probability density curve (Figures 6 and 7) depicting the B–Pb level most likely to occur for a hypothetical child at a specific water Pb concentration. Accordingly, a child's risk to have B–Pb level >5 μg/dl at the median water Pb concentration is 13%, while the risk would be 19% at the 75th percentile. Incorporating the soil data (418.6 μg/g) would raise the risk to 47.2 and 52.8% at the median and the 75th percentiles, respectively. This could be interpreted for all children with the same age group who share the same exposure scenario.

This study revealed a significant discrepancy in health risk predictions between the IEUBK and the HQ for the same water Pb level. According to the IEUBK, a much larger proportion of the population seems to be at risk compared to the HQ results. The observed discrepancy might be associated with the RfD value used in the HQ calculation. Although the oral RfD value in this study is based on the value set given by JECFA (WHO 1993, 2003) and is still used in many related studies (Yi et al., 2011; Cheshmazar et al., 2018; Ugbede et al., 2020; Abeer et al., 2020; Zakir et al., 2020; Maleki and Jari 2021), some other studies are reserved to use this value (Hou et al., 2013; Mingkhwan and Worakhunpiset 2018; Atangana and Oberholster 2021). Furthermore, the US EPA also has no consensus oral RfD for inorganic Pb because of the difficulty in identifying the classic "threshold" needed to develop an RfD (US EPA 2015b).

3.6. Expected health impacts for the modelled B–Pb

Children with the observed B–Pb level in this study are subjected to numerous health risks, including risks related to functioning of hematologic, gastrointestinal, cardiovascular, and renal systems. However, they would be particularly vulnerable to the neurotoxic effects; even relatively low levels of exposure can induce severe and, in some cases, irreversible neurological damage (WHO 2021). Prenatal exposure of the growing fetus would also be a potential concern (SCHER 2011).

EFSA determined a benchmark dose level (BMDL) of 1.2 μg Pb/dl as a reference point for the risk characterization of Pb when assessing the risk of intellectual deficits in children. This corresponds to a daily intake of 0.5 μg/kg/bw (EFSA 2010), significantly below the mean daily intake of Pb calculated for children in this study (1.43 μg/kg). Ninety-six % of the estimated B–Pb in our study falls in the range 2–10 μg/dl where, according to Gould (2009), children are subjected to average IQ point loss of 0.513 per 1 μg/dl of B–Pb. This IQ decrement is considered crucial when interpreted as a reduction of the IQ of the population rather than that of individual children (WHO 2003). In a study wherein Ethiopia was included, the annual lost lifetime economic productivity (LEP) in East Africa due to IQ loss from B–Pb was estimated at 23.1 billion USD (Attina and Trasande 2013). In general, the discussed health and economic impacts might even be higher in Ethiopia, where resources are limited and other health challenges emerge also at the front line. Reducing children's exposure to Pb in their environment would be the best intervention strategy.

4. Conclusion

This study demonstrated that drinking water in Addis Ababa is likely to be an important source of Pb exposure. The spatial analysis indicated that certain sub-cities and specific locations are more prone to varied non-cancer health risks because of the elevated waterborne Pb. The health and economic impacts from the observed HQ and B–Pb level are expected to be intense and long-lasting in the Ethiopian context; thus, interventions are mandatory to minimize effects. Therefore, future studies should also focus on identifying the sources of the Pb contamination in the study area. Moreover, we conclude that health risk evaluation based on HQ calculation alone may underestimate the actual risk from waterborne Pb. Hence, HQ results of Pb are better to be used in conjunction with additional risk evaluation tools, like the IEUBK.

Declarations

Author contribution statement

Yohannes Tesfaye Endale, Bernd Mees: Conceived and designed the experiments; Performed the experiments; Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Argaw Ambelu, Gijs Du Laing: Conceived and designed the experiments; Analyzed and interpreted the data; Contributed reagents, materials, analysis tools or data; Wrote the paper.

Geremew Sahilu G.: Conceived and designed the experiments; Contributed reagents, materials, analysis tools or data; Wrote the paper.

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Data availability statement

Data will be made available on request.

Declaration of interests statement

The authors declare no conflict of interest.

Additional information

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