Exposure to arsenic and lead of children living near a copper-smelter in San Luis Potosi, Mexico: Importance of soil contamination for exposure of children

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Abstract

The objective of this study was to assess the levels of soil contamination and child exposure in areas next to a primary smelter (arsenic–copper metallurgical) located in the community of Morales in San Luis Potosi, Mexico. In Morales, 90% of the soil samples studied in this work were above 400 mg/kg of lead, and above 100 mg/kg of arsenic, which are guidelines recommended by the United States Environmental Protection Agency (EPA). Bioaccessibility of these metals was studied in vitro in 10 soil samples; the median values of bioaccessibility obtained in these samples were 46.5% and 32.5% for arsenic and lead. Since the concentrations of arsenic and lead in soil were above normal values, and taking into account the bioaccessibility results, exposure to these metals was evaluated in children. Regarding lead, children aged 3–6 years had the highest mean blood lead levels; furthermore, 90% of them had concentrations above 10 mg/dl (CDC’s action level). Total urinary arsenic was higher in children aged 8–9 yr; however, the percentage of children with concentrations above 50 mg/g creatinine (CDC’s action level) or 100 mg/g creatinine (World Health Organization [WHO] action level) was similar among different age groups. Using the EPAs integrated exposure uptake biokinetic model for lead in children (IEUBK), we estimated that 87% of the total lead in blood is obtained from the soil/dust pathway. The exposure dose to arsenic, estimated for the children living in Morales using Monte Carlo analysis and the arsenic concentrations found in soil, was above the EPA’s reference dose. With all these results, it is evident that studies are needed in order to identify adverse health effects in children living in Morales; nevertheless, it is more important to develop a risk reduction program as soon as possible.

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1. Introduction

For soil contamination, the most important sources of metals worldwide are mine tailings, smelter emissions, and atmospheric fallout (Nriagu and Pacyna, 1988). In fact, metal concentration in urban areas located in the vicinity of smelters has been shown to be higher than background levels. Arsenic, cadmium, and lead contamination have been reported in smelter areas located in different countries, among them Poland (Dunnette et al., 1994), Russia (Bustueva et al., 1994), the United States (Hwang et al., 1997), Mexico (Benin et al., 1999; Díaz-Barriga et al., 1993), Bolivia (Díaz-Barriga et al., 1997a), Chile (Rivara et al., 1997), and Peru (Ramírez, 1986).

Considering that the main pathways of human exposure to metals in metallurgical areas are soil and air contamination, and taking into account the general toxicokinetics of these pollutants, it is easy to assume that children and pregnant women are the groups at highest risk in those areas. For children, soil contamination is an important pathway; thus, bioavailability of metals in soil is a factor that deserves special attention. It has been determined that lead acetate and lead oxide are more bioavailable than lead sulfide or lead ore concentrate (Dieter et al., 1993). These data are relevant, as it has been determined that in dust,
lead oxide species are the dominant species associated with smelter operations (Spear et al., 1998). Furthermore, it has been reported that blood lead concentrations in exposed individuals are higher in smelter areas than in mining communities, where mine wastes are composed primarily of lead sulfide (Steele et al., 1990).

In the case of arsenic, bioavailability in smelter areas has been studied in monkeys (Freeman et al., 1995). The bioavailability of soil and house dust arsenic relative to soluble arsenic (by gavage) was between 10% and 30%, depending upon whether urinary or blood values were used (Freeman et al., 1995). This result is somewhat in agreement with a study done in vitro, where the bioaccessibility of arsenic in soil was reported to be in the range of 44–50% (Ruby et al., 1996).

Human exposure to metals has been reported in the vicinity of smelters. For example, high levels of urinary arsenic were found in children living around metallurgical plants located in Belgium (Buchet et al., 1980), Mexico (Diaz-Barriga et al., 1993), and the United States (Hwang et al., 1997; Milham and Strong, 1974; Morse et al., 1979), while blood lead levels above normal values have been reported for children and women living in smelter areas of Yugoslavia (Graziano et al., 1990; Wasserman et al., 1997), Mexico (Calderon-Salinas et al., 1996; Diaz-Barriga et al., 1997b), the United States (Landrigan and Baker, 1981), the Czech Republic (Cikrt et al., 1997), and Canada (Hilts et al., 1998), among others. Considering the toxicity of these metals (ATSDR, 1999; ATSDR, 2000), the levels of exposure suggest health risks that have justified remediation programs in some smelter areas (Goulet et al., 1996; Hilts et al., 1998; Langlois et al., 1996).

In San Luis Potosi, Mexico, a smelter producing copper, lead, and arsenic is located within the urban area. Metal levels in air have decreased since a study of our group showed high levels of urinary arsenic in children (Diaz-Barriga et al., 1993), furthermore, the industry is working at 50% of its capacity. However, the historical accumulation of metals in soil (the industry has been operating since 1890) may be a risk for children living in the area. In this study we present data about the bioaccessibility of metals in soil and levels of urinary arsenic and lead in blood of children living next to this copper–arsenic smelter. We used the term “bioaccessibility” instead of “bioavailability,” as the first is the percentage of the total metal concentration that is extracted in gastric or intestinal phases in an “in vitro” system (described in methods), whereas bioavailability refers to studies done in different “in vivo” (animal or human) models.

2. Methods

2.1. Study areas

The site, which is known as “Morales,” is an urban area located within the city of San Luis Potosi, Mexico, 1860 m above sea level. The study area was within 1.5 km of a smelter complex consisting of a copper–arsenic smelter (which also is producing lead as a by-product) and an electrolytic zinc refinery. As a result of our first study in the area (Diaz-Barriga et al., 1993), during 1990, the streets of Morales were paved and the contaminated soil in some areas was removed. Also, at that time, the smelter introduced control devices that decreased the emissions of metals. The source of drinking water is an aquifer located 100 m below ground level; arsenic and lead levels in this aquifer are within national guidelines. The mean temperature is 24.4°C, and the annual precipitation is 490 mm. About 19,000 inhabitants live in the study area, including 7000 children.

2.2. Environmental monitoring

A systematic sampling was undertaken within a 200-m grid in the area presumably impacted by the emissions of the smelter, located within 1.5 km of the stack. Areas repeatedly used by children were also included in the collection. Samples of surface soil (1–5 cm in depth) were obtained with a stainless steel scoop on an approximately 1 m² surface area and stored in polyethylene bags.

2.3. Bioaccessibility of metals in soil

The physiologically based extraction test (PBET) was used to determine the As and Pb bioaccessible fraction in 10 soil samples. This method establishes conditions simulating the gastric and intestinal phases of the digestion process by incorporating values of pH, temperature, chemical composition, and residence time (Ruby et al., 1993, 1996). The amount of metal extracted from soil under these conditions is similar to the amount of the metal in the ingested soil that would be available for absorption by the body after digestion.

Because of the prolonged time required to reach pH 7.0 at the beginning of the intestinal phase according to the original procedure (2–3 h), in this work, the neutralization was performed by adding NaHCO₃ saturated solution as showed by Rodriguez et al. (1999). The gastric solution pH was 1.3 and the solution of this phase was sampled an hour after the test started; intestinal solution was sampled 3 h after neutralization. Arsenic and lead chemical analyses were made from both gastric and intestinal solution samples. The bioaccessibility is expressed as the percentage of the total concentration that was extracted in gastric or intestinal phases.

2.4. Gastric phase

The gastric solution was prepared dissolving by 1.25 g porcine pepsin, 0.50 g sodium citrate, 0.50 g sodium malate, 420 μL lactic acid, and 500 μL of acetic acid in 1 L of deionized water; pH was adjusted to 1.3 using concentrated HCl. Fifty milliliters of gastric solution was placed in a 250-mL polyethylene funnel to which 0.5 g of soil sample was
added. The funnel was kept submerged in a water bath with controlled temperature at 37 °C. The mixture stayed at rest for the first 10 min before beginning the agitation by constant flow of 1 L argon/min at the bottom of the funnel. Five minutes after the initiation of agitation and then every 10 min, pH was adjusted to 1.3 by adding concentrated HCL when necessary. One hour after the initiation the gastric phase, the agitation stopped and a 10-mL solution sample was extracted, which was then filtered with 11-μm pore diameter Whatman filter paper. Sample was preserved by adding concentrated HNO₃ and maintained under refrigeration until chemical analysis. The sample solution volume was compensated adding an equal amount of original gastric solution.

2.5. Intestinal phase

To continue with the intestinal phase, a NaHCO₃ saturated solution was added slowly to gastric solution until reaching pH of 7.0. Then 87.5 mg biliary salts and 25.0 mg pancreatina were added. At this point, agitation was reinitiated and maintained until the end of the extraction 3 h later, recovering at this time the rest of the solution under the same procedure used in the gastric solution.

2.6. Biological monitoring

Children attending schools in the area were selected at random from those families that agreed to participate. Healthy children who had at least 2 yr of residence in their particular area (within 1.5 km of the smelters) were considered for the study. All parents filled out an exposure questionnaire. Blood was obtained by venous puncture using lead-free tubes containing EDTA as anticoagulant. First void urine samples were collected, stored in plastic bottles, and kept frozen until analysis.

2.7. Analytical methods

Lead in blood was analyzed with matrix modifier (diammonium hydrogenphosphate–Triton X-100 in the presence of 0.2% nitric acid) according to Subramanian (1987) and the samples were analyzed with a Perkin–Elmer 3110 atomic absorption spectrophotometer using a graphite furnace. An aliquot of urine (0.5–3.0 mL) was wet digested with nitric, sulfuric, and perchloric acids according to Cox (1980). Soil samples were oven dried at 30 °C for 1 to 2 days. The <600-μm fraction was separated with a 28 mesh Tyler Series sieve. Soil samples were acid digested (HNO₃ 25%) for 30 min under 80 psi pressure using a CEM MDS-2000 microwave extraction system. After digestion, the extracted solution was filtered through Whatman filter paper with 11 μm pore diameter. Analyses for lead in soil were carried out by flame atomic absorption spectrometry using a Varian Spectra AA 220 atomic absorption spectrometer; arsenic (either in urinary or soil samples) was analyzed by flame atomic absorption spectrometry using a Perkin–Elmer Aanalyst 100 atomic absorption spectrometer coupled to a Perkin–Elmer FIAS 100 hydride generation system.

2.8. Quality control considerations

Analysis of primary standard reference material in each run was conducted as an internal quality control. For soil, NIST-SRM 2710 (Montana soil) was used with recovery of 98% for lead and 97% for arsenic. At the time of the study, our laboratory was participating in the blood lead proficiency testing program of the Centers for Disease Control (CDC), and in the Interlaboratory Comparison Program organized by the Centre de Toxicologie du Québec for urinary arsenic and lead in blood. Each sample was analyzed in duplicate. Distilled–deionized water was used for all analytical work, and glassware and other materials were soaked in 10% nitric acid, rinsed with doubly distilled water, and dried before use.

2.9. Estimates of lead exposure

EPA’s integrated exposure uptake biokinetic model for lead in children (IEUBK) was used to estimate lead exposure. Results using IEUBK were obtained under the following conditions: Air: Pb in air 0.76 μg/m³ (data from Morales according to the Mexican government). Time spent outdoors 4 h/day. Diet: 9.3 μg/dia (Batres et al., 1995). Water: Pb in water 6.4 μg/L and 1 L/day of water ingestion. Soil: Pb in soil 1122 mg/kg (median value of the results depicted in Table 1), Pb in dust 861 mg/kg (data given by the IEUBK in the multiple source analysis option), and 200 mg/day of soil ingestion (ATSDR, 2001). Bioavailability: 30% (default value in the IEUBK).

<table>
<thead>
<tr>
<th>Table 1</th>
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<tbody>
<tr>
<td>Arsenic and lead levels in surface soil (mg/kg)</td>
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<tr>
<td></td>
</tr>
<tr>
<td>Lead</td>
</tr>
<tr>
<td>Arsenic</td>
</tr>
</tbody>
</table>

Note: Results are depicted as the arithmetic mean (SD) standard deviation. The minimum (min) and maximum (max) concentrations found in the samples are also shown. In the last two columns the percentages of samples above the corresponding reference value are presented.
We did not include exposure to lead-glazed pottery, as 70% of the children in the smelter area did not consume food cooked in this pottery.

2.10. Statistical analysis

The distribution of blood lead levels (PbB) and urinary arsenic concentrations (AsU) were skewed; therefore, values were log transformed for statistical analysis. Differences in PbB between age groups according to use of lead-glazed pottery were examined by the Student \( t \)-test. Probabilistic risk assessment was performed by Monte Carlo analysis using Crystal Ball v5.5.

3. Results

As is shown in Table 1, high concentration levels of lead and arsenic in soil samples were found. Ninety percent of the samples were above 400 mg/kg of lead, and above 100 mg/kg of arsenic, which are intervention guidelines recommended by the United States Environmental Protection Agency (USEPA) (EPA, 1990, 2001). A distribution of the concentrations of arsenic and lead in the urban area can be observed in Figs. 1 and 2. It is important to note that the concentration of arsenic and lead in soil samples was three times higher in a soil particle size < 50 \( \mu \)m, when compared to particles size < 600 \( \mu \)m (data not shown).

Fig. 1. Contour maps corresponding to spatial distribution for arsenic in soil according to 200 m grid surface soil sampling. Mapping by Surfer 7.0 (Golden Software, Inc., USA) using distance inverse to a power interpolation method. Results depicted in Table 1 were used for the construction of this map.
Bioaccessibility was assessed in vitro as stated in Methods. In Table 2 the results of the intestinal phase in 10 samples are presented. It can be observed that arsenic and lead are bioaccessible, and in general, similar percentages of bioaccessibility were observed in the samples. The median values for the percentage of bioaccessibility obtained in these samples were 46.5 and 32.5 for arsenic and lead, respectively. In Figs. 3 and 4 we present the percentages obtained in both the gastric and the intestinal phases. For arsenic, both phases gave similar results; however, more lead was extracted in the gastric phase. The amount of lead or arsenic extracted, in either the gastric or the intestinal phases, and the total concentration of the metal in the original soil sample, presented a high correlation coefficient ($r = 0.875$); thus, the percentage of bioaccessibility for both elements was independent of the total concentration in the soil samples (Figs. 3 and 4). The percentage of bioaccessibility was also independent of the soil particle size (2000 μm vs. 600 μm, data not shown).

Since the concentrations of arsenic and lead in soil were above normal values, and taking into account the bioaccessibility results, exposure to these metals was assessed in children. Regarding lead, children aged 3–6 yr had the highest mean blood lead levels (Table 3); furthermore, 90% of them had concentrations above 10 μg/dl (CDCs action level) (CDC, 1991), and 27% were above 20 μg/dl. In 141 of the studied children 65% reported that they were not exposed to lead-glazed pottery and they had a mean lead level of 9.5 μg/dl, a level significantly lower.
than those found in children who were exposed to this kind of pottery (10.9 μg/dl). Total urinary arsenic was higher in children aged 8–9 yr (Table 4); however, the percentage of children with concentrations above the guidelines of 50 μg/g creatinine (Belson et al., 2005) or 100 μg/g creatinine (Hwang et al., 1997) was similar among age groups (Table 4).

In order to learn more about soil as a source of lead for children living in Morales, we used the IEUBK. Using values shown in Table 5, we calculated blood lead levels for children aged 3–6 yr. In the same table, we present a comparison between the blood lead levels calculated by the model and the real blood lead levels found in children living in the vicinity of the smelter. It can be observed that a good correlation was obtained between them. According to IEUBK’s data, 87% of the total lead in blood is obtained from the soil/dust pathway. As soil was important for lead, through Monte Carlo analysis we assessed the exposure to arsenic using the formulae and considerations depicted in Table 6. It can be observed that the exposure dose in one-third of the children is five times higher than the reference dose; furthermore, 10% of the population has a risk quotient of 9.
4. Discussion

Taking into account the risk associated with smelter emissions, a health assessment was performed in the urban area located in the vicinity of a copper–arsenic smelter in Morales, San Luis Potosi, Mexico. In this work we found that in Morales, lead and arsenic concentrations were above normal values in surface soil. In several smelter sites, including Morales (Diaz-Barriga et al., 1993), an inverse correlation has been reported between metal levels in soil and distance from the smelter site (Benin et al., 1999; Cikrt et al., 1997; Hwang et al., 1997; Milham and Strong, 1974). In this work, we showed that in Morales, at the same distance from the smelter stack, the northern area was more contaminated than the southern area (Figs. 1 and 2). This difference can be explained by the prevailing wind direction. Furthermore, when compared to other smelter sites, arsenic levels in soil in north Morales were higher than EPA’s guideline and than those found in a former copper smelter site (Hwang et al., 1997) or in an active arsenic smelter site (Polissar et al., 1990). In relation to lead levels, the mean concentration in North Morales was similar to the mean concentration found in the vicinity of lead smelters (Benin et al., 1999) and it was higher than the EPA guideline.

Considering the environmental concentrations that were found around the smelter studied in this work, bioavailability became an important issue to explore. In the literature there are several reports about methodologies for the analysis of this subject (Oomen et al., 2002). We selected the one described by Ruby et al. (1993, 1996), as this method establishes conditions that simulate gastric and intestinal phases of the digestion process, by incorporating values of pH, temperature, chemical composition, and residence time. In this work, we are reporting a lead bioaccessibility of 13–64% and 39–69% for arsenic. These results are in agreement with those obtained by different authors in soils collected in smelter areas. For example, Rieuwerts et al. (2000) have reported a lead bioaccessibility of 35–68%, whereas Ruby et al. (1996) reported one of 29–54%. In regard to arsenic, Ruby et al. (1996) reported 44–50%. However, it is important to consider that in a primate model the bioavailability of arsenic in a smelter soil relative to soluble As (by gavage) was between 10% and 30%, depending upon whether urinary or blood values were used (Freeman et al., 1995). Thus, this bioavailability was somewhat lower than the bioaccessibility measured in this work. Without knowing the exact composition of the arsenical particles in the soils used in the studies, it is difficult to obtain an explanation of these findings.

Exposure was assessed using biomarkers. With regard to lead, levels found in children’s blood were higher than the CDC guideline and the Mexican background levels

Table 5
Blood lead levels in children living in Morales: comparison between the real values and those obtained with the IEUBK (µg/dl)

<table>
<thead>
<tr>
<th>Age</th>
<th>Real values</th>
<th>IEUBK</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>G mean</td>
<td>% &gt;10</td>
</tr>
<tr>
<td>3–6</td>
<td>14.8</td>
<td>90</td>
</tr>
</tbody>
</table>

Note: Results are depicted as the geometric mean. In the next two columns the percentage of children with levels above the reference value in µg/dl are presented. Real values were obtained from those shown in Table 3. IEUBK information was obtained as stated in Methods.

Table 6
Exposure dose estimated using arsenic levels in soil samples

<table>
<thead>
<tr>
<th>General formula</th>
<th>Exposure dose estimated using arsenic levels in soil samples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Exposure dose =</td>
<td>Soil concentration × ingestion rate × exposure factor × bioavailability</td>
</tr>
<tr>
<td>Body weight × conversion factor</td>
<td></td>
</tr>
</tbody>
</table>

Risk quotient = Exposure dose / Reference dose

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Comments</th>
<th>Distribution</th>
<th>n</th>
<th>Minimum</th>
<th>Maximum</th>
<th>Mean</th>
<th>SD</th>
</tr>
</thead>
<tbody>
<tr>
<td>As in soil (mg/kg)</td>
<td>Table 1</td>
<td>Normal</td>
<td>95</td>
<td>17.4</td>
<td>4424.2</td>
<td>791.5</td>
<td>798.26</td>
</tr>
<tr>
<td>Bioavailability</td>
<td>Table 2</td>
<td>Log normal</td>
<td>10</td>
<td>0.39</td>
<td>0.66</td>
<td>0.48</td>
<td>0.07</td>
</tr>
<tr>
<td>Soil ingestion (mg/day)</td>
<td>EPA (2002)</td>
<td>Triangle</td>
<td>100</td>
<td>400</td>
<td>200.0</td>
<td>4.82</td>
<td></td>
</tr>
<tr>
<td>Body weight (kg)</td>
<td>a</td>
<td>Log Normal</td>
<td>54</td>
<td>15.0</td>
<td>46.0</td>
<td>22.3</td>
<td>0.27</td>
</tr>
<tr>
<td>Exposure factor</td>
<td>b</td>
<td>Single</td>
<td></td>
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<tr>
<td>RfD as oral (mg/kg-day)</td>
<td>EPA (2005)</td>
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<td></td>
<td></td>
<td></td>
<td>0.0003</td>
<td></td>
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<tr>
<td>Percentage of children at risk</td>
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<tr>
<td>Risk quotient</td>
<td>% children</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>&gt;1</td>
<td>74.8</td>
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<td></td>
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<tr>
<td>&gt;2</td>
<td>63.7</td>
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<tr>
<td>&gt;5</td>
<td>31.8</td>
<td></td>
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<tr>
<td>&gt;9</td>
<td>10.0</td>
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</table>

a Body weight of 54 children living in the studied area.
b Exposure factor = 6 h per day Monday to Friday and 8 h per day during the weekend = 46/168 = 0.27.
blood lead concentrations, even those below 10 μg per deciliter, have been shown to be inversely associated with girls at concentrations lower than 10 μg/dl (Selevan et al., 2003), our results denote a considerable health risk for children living in the vicinity of the smelter. Urinary arsenic was also quantified in children living less than 1.5 km from the smelters. Total arsenic was determined in urine, taking into account that seafood is not an important dietary component in the communities that were studied. It has been claimed that total arsenic measurements are less valid as a biological indicator of exposure to inorganic arsenic than urinary arsenic speciation (Polissar et al., 1990). However, this assumption may be applicable only to communities where fish consumption, a source of methylated arsenic, is an important component of the diet. We have previously shown that, for the smelter site in Morales, total arsenic in urine was a good indicator of exposure, as an inverse correlation was found between urinary arsenic and distance to the smelter (Díaz-Barriga et al., 1993). In this work, results showed that children are exposed to arsenic in the studied area (Table 4). It is important to note that recently our group has reported increased DNA damage in children exposed to arsenic (Díaz-Barriga et al., 2004), whereas arsenic-induced effects on neurobehavioral development have been reported in adolescents (Tsai et al., 2003).

We have shown that children in Morales are simultaneously exposed to arsenic and lead, and although no attempts to systematically measure health effects were made, the data in the present work suggest that if exposures continue, children could suffer adverse health effects. Whether or not the interaction between lead and arsenic may have toxicological relevance for the exposed children is a matter that deserves further research; however, we have previously shown (Mejía et al., 1997), in rats treated with the mixture arsenic+lead, a 38% decrease of norepinephrine in the hippocampus and increases of serotonin in midbrain and frontal cortex (100% and 90%, respectively) over control values. These alterations were not elicited by either metal alone. Furthermore, preliminary results of our group have shown neuropsychological effects associated with arsenic and lead exposures in children living in Morales (Calderón et al., 2001); whereas, in smelter towns, associations between lead exposure and neurological endpoints have been reported (Baghurst et al., 1992; Baghurst et al., 1995; Factor-Litvak et al., 1999; Wasserman et al., 1997).

It is evident that studies are needed to identify adverse health effects in children living in Morales; nevertheless, it is more important to develop risk reduction programs. For example, in Morales, the reduction of metal concentrations in soil is urgently needed, as in this work we showed that soil is an important pathway of exposure for children living in the smelter area. Risk-reduction programs have to include the collection of street dust, as preliminary results with nine street dust samples showed a mean arsenic level of 387 mg/kg and a mean lead level of 911 mg/kg; interestingly, the bioaccessibility of these metals in these samples was similar to that found for soil samples (data not shown). Small particles, such as those found in street-dust, are easily inhaled or ingested by hand-to-mouth activity. Hand-to-mouth activity is more pronounced in young children and this may be an explanation for the observation that the highest mean blood lead was found in young children, 3–6 yr of age. Regarding arsenic in urine, no such relationship could be demonstrated; a possible explanation may be that according to data published by the Mexican government, arsenic air concentrations in the smelter area are higher than lead levels. Thus, inhalation, an age-independent pathway, can be a more important route of exposure for arsenic than it is for lead.

Children are different from adults, both in patterns of exposure to environmental risk and in their responses. With the recognition of the special vulnerability of children, it is better to prevent environmental diseases in children than to treat them. In developed countries, it has been reported that when more than 14% of children had elevated blood lead levels, the economic benefits of universal screening exceeded the costs (CDC, 1997). Among the economic benefits of reducing blood lead levels, the authors of that study included three main categories: (A) improvements in lifetime earnings attributable to reductions in lead-induced problems with intelligence or behavior; (B) reduction in lead-related special-education costs; and (C) economic benefits of identifying and fixing dangerous housing so that other people are not exposed to lead. The only difference in developing countries would be to ask whether it is convenient to invest in remediating the overexposure to metals in a community where a greater beneficial impact on health might be achieved by investment in general sanitation. In our opinion, the answer to this question is affirmative, and studies such as the present one are important, as they identify the critical pathways of exposure, and at the same time, they give social justification for investment in risk reduction programs. In 1999, a surveillance program was developed, and negotiations to start a remediation program in the area are imminent. Remediation actions will include playgrounds, school yards, and other places where children play outdoors. Contaminated soil will be removed or covered by un-contaminated soil; furthermore, street dust will be collected.

Acknowledgments

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