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Childhood Lead Exposure in an Industrial Town in China: Coupling Stable Isotope Ratios with Bioaccessible Lead

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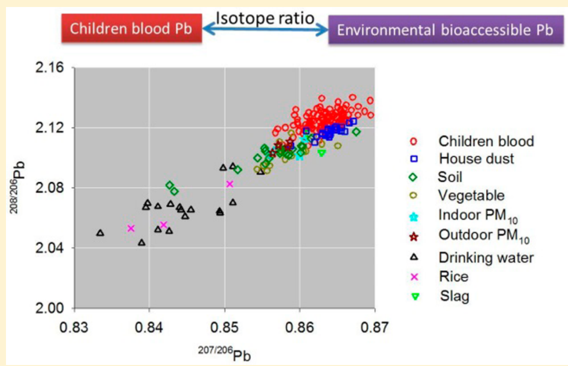
Supporting Information

ABSTRACT: Fingerprinting based on stable isotopes of lead (Pb) in blood and environmental media helps to identify Pb exposure pathways in children. However, previous studies used stable isotopes of total Pb in media. In this study, a wire rope production town in China (Zhuhang) was selected for investigating the effectiveness of using isotope ratios in bioaccessible Pb to identify childhood Pb exposure pathways. Blood Pb levels of 115 children in Zhuhang were 1.7−20.4 μg dL^{-1}, averaging 6.1 ± 3.2 μg dL^{-1} (mean ± standard deviation), and were ~1.6 times the national average in China (3.9 ± 1.8 μg dL^{-1}). Among different environmental media (housedust, soil, PM_{10}, vegetables, rice, and drinking water), housedust (695 ± 495 mg kg^{-1}) and vegetables [0.36 ± 0.40 mg (kg of fresh weight)$^{-1}$] contained elevated Pb concentrations. The isotope ratios (^{207}Pb/^{206}Pb and ^{208}Pb/^{206}Pb) of total Pb were the highest in housedust (0.8587 ± 0.0039 and 2.1049 ± 0.0087) but lower than blood Pb ratios (0.8634 ± 0.0027 and 2.1244 ± 0.0061). When using bioaccessible Pb in housedust (0.8639 ± 0.0018 and 2.1171 ± 0.0036), the isotope ratios overlapped with blood Pb ratios, suggesting that incidental ingestion of housedust was the predominant contributor to children's blood Pb. Coupling the stable isotope technique with bioaccessible Pb is more reliable for identifying Pb exposure pathways than total Pb determinations.



INTRODUCTION

Lead (Pb) exposure is a serious hazard for children, adversely affecting their cognitive and behavioral development.[1,2] Recent reports suggest that neurological damage occurs in children at low blood lead levels (BLLs; 2−10 μg dL^{-1}), indicating there are no "safe" BLLs.[3] Although a dramatic decline in BLLs is observed worldwide after leaded gasoline has been phased out, children's exposure to Pb is still common in China because of industrial expansion.[4,5] To reduce exposure to Pb for children living near industrial areas, it is important to identify predominant Pb exposure pathways, which include both nondietary (ingestion of dust/soil and inhalation of particulates) and dietary pathways (consumption of food and drinking water).[6−8] However, identifying major exposure pathways is often compromised by the lack of paired data of blood Pb with environmental Pb for a given cohort.

Fingerprinting based on stable Pb isotope ratios (^{207}Pb/^{206}Pb and ^{208}Pb/^{206}Pb) of environmental and blood samples helps to identify Pb exposure pathways.[9] In general, natural sources of Pb have ratios lower than those of anthropogenic sources.[10] As a result, when environmental Pb sources are isotopically distinct from one another, Pb exposure pathways may be identified by comparing the isotopic composition of blood Pb with environmental Pb.[11] Over the years, this technique has been successful in identifying pathways of childhood Pb exposure. For example, by comparing the ^{208}Pb/^{206}Pb isotope ratio in environmental media (e.g., soil, dust, paint, air particulates, and gasoline) with that in blood samples of children having elevated BLLs (28−43 μg dL^{-1}), Yaffe et al.[12] found that blood isotope ratios were close to those of soil and housedust, suggesting ingestion of soil and housedust was the source of blood Pb. Liang et al.[13] found that the Pb isotopic composition of children's blood in Shanghai, China, matched those in PM_{10} and coal combustion ash, suggesting inhalation of air particulates contaminated with coal combustion ash was an important blood Pb contributor. Cao et al.[14] found close agreement between Pb isotope ratios of children's blood and vegetables, wheat, drinking water, and air particulates, while blood Pb ratios were different from those of soil and housedust, suggesting that dietary and inhalation pathways were the most likely sources of children's blood Pb. Similarly, Gulson et al.[15,16]

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and Delves and Campbell[17] successfully used Pb isotopes to identify housedust and drinking water as predominant blood Pb sources in children.

However, there have also been studies that failed to identify blood Pb sources in humans. Manton et al.[18] suggested that the success rate of blood Pb source identification using Pb isotopes in the United States is ~20%. One reason is the overlap of isotopic compositions of different environment samples with one another. For example, Soto-Jiménez and Flegal[19] found that the Pb isotope ratios of dust, soil, and aerosols from a smelting area of México were indistinguishable from each other, all being similar to blood Pb ratios. Another issue associated with previous Pb isotope studies is that isotope ratios are usually determined for total Pb in environmental media. However, it is recognized that following ingestion or inhalation of soil, dust, air, and food matrices, only the bioavailable fraction of Pb is absorbed into the systemic circulation.[20] Conceivably, comparison of Pb isotope ratios for blood and total Pb in environmental matrices may not provide an accurate estimate of exposure sources when isotope ratios of bioavailable Pb and nonbioavailable Pb differ.[21]

In this study, BLLs were assessed in a cohort of children who resided in Zhuhang, China, a town characterized by intensive wire rope production. Potential Pb exposure sources were assessed by determining total and bioaccessible Pb concentrations, and stable Pb isotope ratios in children's blood and environmental media. We hypothesized that comparing Pb isotopic data between children's blood and bioaccessible Pb in environmental media would provide an approach for identifying Pb exposure pathways more robust than total Pb determinations, especially when isotope ratios vary significantly between total and bioaccessible Pb.

■ MATERIALS AND METHODS

Geographic Location and Child Blood Sampling. Zhuhang, located in Nantong, Jiangsu Province, China, is an industrial town characterized by intensive wire rope production, with 122 small wire rope factories (WRFs) within ~31 km^2 producing ~20% of the wire rope in China.[22] Many WRFs are surrounded by residential districts with the distance to housing, in some cases, being <50 m. During wire rope manufacturing, wire is heated at ~950 °C and then rapidly quenched in molten Pb at ~500 °C.[23] The emission of Pb-containing smoke and dust during the process has impacted the surrounding environment and community. In spring 2012, several children who suffered from hyperactivity and irritability in a nearby residential district were diagnosed with elevated BLLs (>10 μg dL^{-1}). Further blood monitoring determined that 28% of individuals tested (184 of 650) had BLLs of >10 μg dL^{-1}.[22] Lead emissions from WRFs were presumed to be the primary source resulting in elevated BLLs; however, the source–pathway continuum was not identified.

A neighborhood in Zhuhang was chosen as a representative study area for this study. The neighborhood of ~8000 people was built between 2004 and 2005 and is surrounded by WRFs (Figure S1 of the Supporting Information). The study population comprised 115 children from 91 families, ranging in age from 2 to 14 years of age (median of 6 years of age). In July 2013, individual's venous blood (~2 mL) was collected in Pb-free heparinized vacutainers at the local health center by registered physicians for Pb quantification and stable Pb isotope analysis.

Sampling of Dust, Soil, PM$_{10}$, Food, and Water. Elevated BLLs in children may result from Pb exposure arising from different sources and pathways. To determine the potential influence of different sources on blood Pb concentration, 91 housedust, 30 soil, 8 PM$_{10}$, 26 vegetable, 9 rice, and 21 drinking water samples were collected in November 2013 and analyzed for total and bioaccessible Pb in addition to stable Pb isotope ratios. Housedusts were collected with plastic brushes from indoor surfaces (floors, windowsills, and furniture) and sieved (<150 μm) for Pb analysis. Surface soils (depth of 0–5 cm) were collected from locations that were frequently used by children and sieved (<150 μm) for Pb analysis.[24] Four paired indoor and outdoor PM$_{10}$ samples were obtained using air samplers and Teflon membrane filters. Polished rice that was being consumed locally in randomly selected homes was collected and ground to a powder. In addition, commonly consumed vegetables comprising 7 species (spinach, pakchoi, chives, asparagus lettuce, crown daisy, hyacinth bean, and sweet potato) were obtained, with the edible portion being washed, freeze-dried, and ground to a powder prior to analysis without cooking. Tap water first-flush samples were collected from homes, filtered, acidified, and stored at 4 °C prior to Pb analysis. Details regarding sample collection are provided in the Supporting Information.

Assessment of Bioaccessible Pb in Environmental Samples. To determine Pb isotope ratios in bioaccessible Pb fractions of environmental media, an *in vitro* gastrointestinal extraction method [U.S. Environmental Protection Agency (EPA) Method 9200.2-86] was used to assess bioaccessible Pb in selected 24 dust (337–2352 mg kg^{-1}) and 22 soil subsamples (17.7–1020 mg kg^{-1}). Though the method may overestimate Pb bioavailability based on a previous study,[25] it showed a strong correlation with *in vivo* oral Pb bioavailability in contaminated soil and dust.[26,27] In this study, U.S. EPA Method 9200.2-86 was also applied to measure Pb bioaccessibility in food matrices (25 vegetable and 9 rice samples). In addition, an *in vitro* method using simulated human lung fluid [artificial lysosomal fluid (ALF)] was applied to analyze bioaccessible Pb following inhalation of PM$_{10}$.[28] Details of the bioaccessibility analysis are provided in the Supporting Information. For QA/QC, a standard reference material SRM NIST Montana Soil 2711a (National Institute of Standards and Technology) was included. Lead bioaccessibility in the SRM was 84.0 ± 1.1% (n = 3), in agreement with the recommended mean value of 85.7%.[25]

Lead Concentration and Stable Isotope Ratio Analysis. Blood and environmental samples (housedust, soil, PM$_{10}$, rice, and vegetables) were digested according to U.S. EPA Method 3050B using a Hot Block digestion system (Environmental Express, Mt. Pleasant, SC). Lead concentrations in digests (total Pb), drinking water samples, and simulated gastric/lung fluid extractions (bioaccessible Pb) were measured using inductively coupled plasma mass spectrometry (ICP-MS, NexION300X, PerkinElmer) after dilution with 0.1 M HNO_3. During ICP-MS analysis, an indium isotope (^{114}In) was used as an internal standard. For QA/QC, the total Pb in SRM NIST 2711a was determined. The measured Pb concentration of 1309 ± 105 mg kg^{-1} (n = 3) was in agreement with the certified value of 1400 mg kg^{-1}. During Pb determination, duplicate analysis, check and spiked samples (1 μg L^{-1}) were included. The relative standard deviation (RSD) for triplicate analysis was 3.2% (1.1–5.6%), the average check recovery (n =

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Table 1. Children's BLLs and Total Pb Concentrations in Housedust, Soil, Indoor and Outdoor PM_{10}, Rice, Vegetables, and Drinking Water Collected from a Wire Rope Production Neighborhood in Nantong City, China

	Pb concentration			
sample	range	mean	median	standard deviation
children blood (n = 115, μg dL^{-1})	1.72−20.4	6.10	5.40	3.20
housedust (n = 91, mg kg^{-1})	58.9−3545	695	655	495
soil (n = 30, mg kg^{-1})	14.9−1020	105	50.0	186
indoor PM_{10} (n = 4, mg kg^{-1})	1034−1372	1,168	1,134	146
outdoor PM_{10} (n = 4, mg kg^{-1})	973−1226	1,097	1,094	103
rice (n = 9, μg kg^{-1})	11.9−35.0	17.6	16.2	6.86
vegetables [n = 26, mg (kg of fresh weight)$^{-1}$]	0.02−1.91	0.36	0.23	0.40
drinking water (n = 21, μg L^{-1})	0.02−4.46	0.72	0.29	1.12

20) was 99.1% (92.1−107%), and spike sample recoveries (n = 25) ranged from 92.8 to 109% (average of 102%).

Stable isotope ratios ($^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$) of children's blood Pb and total and bioaccessible Pb in environmental media were determined using ICP-MS following dilution of digests and extracts to concentrations of 1−15 μg of Pb L^{-1} using 0.1 M high-purity HNO_3.[27] Drinking water samples were concentrated via evaporation to achieve a similar Pb concentration range. Pb isotopic standard NIST 981 was measured every five samples to determine and update the ratio correction factors. Each sample was measured in 10 replicates with the RSD generally being <0.5% for $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ ratios. The measured values for $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ ratios of NIST 981 (0.9160 ± 0.0013 and 2.1696 ± 0.0028, respectively; n = 6) were in agreement with the certified values of 0.9146 and 2.1681, respectively. Instrument parameters for stable isotope ratio measurements are provided in the Supporting Information.

Statistical Analysis. One-way analysis of variance was performed to determine significant differences in Pb isotope ratios among environmental media and between total and bioaccessible Pb based on the Tukey's post hoc test. All analyses were performed using SAS (version 9.1.3), while figures were created using SigmaPlot (version 10.0, Systat Software Inc., San Jose, CA).

RESULTS AND DISCUSSION

Blood Pb Levels in Children. Within the cohort of 115 children, BLLs were 1.7−20.4 μg dL^{-1}, averaging 6.1 ± 3.2 μg dL^{-1} (Table 1). Eleven percent of children had BLLs of >10 μg dL^{-1} (Figure 1), the international guideline value above which intervention is recommended by the U.S. Centers for Disease Control and Prevention (CDC).[29] With a growing body of evidence of intellectual and behavioral deficits occurring in children with BLLs of <10 μg dL^{-1},[3] the CDC recently reduced the guideline value to 5 μg dL^{-1}.[30] Using the reference value of 5 μg dL^{-1}, the number of children in the Zhuhang cohort with elevated BLLs was 57%. Males tended to have BLLs (6.5 ± 3.4 μg dL^{-1}) higher than those of females (5.9 ± 3.1 μg dL^{-1}); however, the difference was insignificant (p > 0.05) (Figure S2A of the Supporting Information). For children from 2 to 6 years of age, an obvious increase in BLLs with age was observed (Figure S2B of the Supporting Information), which mirrors the rise of BLLs with the increase in age for children 0−6 years of age.[31−33]

The observed mean BLLs for children in Zhuhang were ∼1.6 times the Chinese national average (3.9 ± 1.8 μg dL^{-1})[34] and ∼5 times the geometric mean (1.3 μg dL^{-1}) for U.S. children

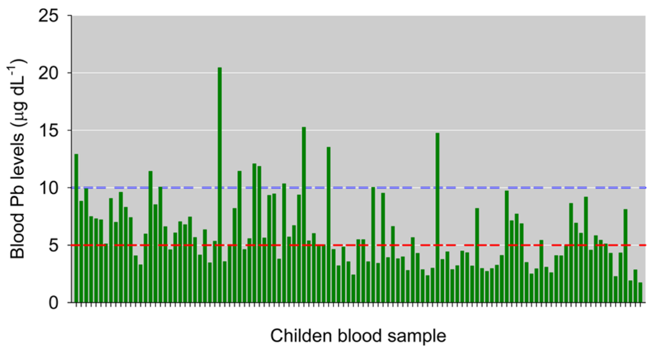

Figure 1. Blood Pb concentration of children (n = 115) living in Zhuhang, a wire rope production town, Nantong City, China. The blue line indicates the former blood Pb guideline value (10 μg dL^{-1}) above which intervention is recommended by the U.S. Centers for Disease Control and Prevention, while the red line indicates the new guideline value of 5 μg dL^{-1}.[29,30]

(1−5 years of age).[35] However, BLLs in the study cohort were lower than those in 165 children from Guiyu, an electronic waste (e-waste) recycling town in China (range of 4.4−32.7 μg dL^{-1}; mean of 15.3 ± 5.8 μg dL^{-1}).[36]

Pb in Environmental Matrices. Exposure to Pb may occur via both nondietary and dietary pathways. To ascertain the contribution of each pathway to BLLs in the study cohort, Pb concentrations in various environmental media (housedust, soil, PM_{10}, vegetables, rice, and drinking water) were determined.

Lead concentrations in housedust samples were 59−3545 mg kg^{-1}, averaging 695 ± 495 mg kg^{-1} (Table 1). Among the 91 housedust samples, 63% had Pb concentrations of >500 mg kg^{-1} and 19% concentrations of >1000 mg kg^{-1}. Compared to the mean Pb concentration in Chinese rural housedust [208 ± 328 mg kg^{-1} (n = 122)][37] and urban housedust in Canada [210 ± 446 mg kg^{-1} (n = 1025)],[38] significantly higher Pb concentrations (∼2.5 times) were observed in Zhuhang.

Compared to that of housedust, significantly lower Pb concentrations (range of 15−1020 mg kg^{-1}; mean of 105 ± 186 mg kg^{-1}) were observed in soil samples from Zhuhang. Only 1 of 30 soils had a Pb concentration exceeding the Chinese Pb soil guideline value of 300 mg kg^{-1} for residential soils (Figure S3 of the Supporting Information). Previous studies in Canada showed that Pb in housedust was >5 times the concentration in outdoor soil, primarily because of the contribution of Pb-based paint.[39] However, Pb-based paints were not used in the dwellings of the study cohort. The data suggested that the elevated Pb concentration in housedust arose from sources other than soil such as fine airborne

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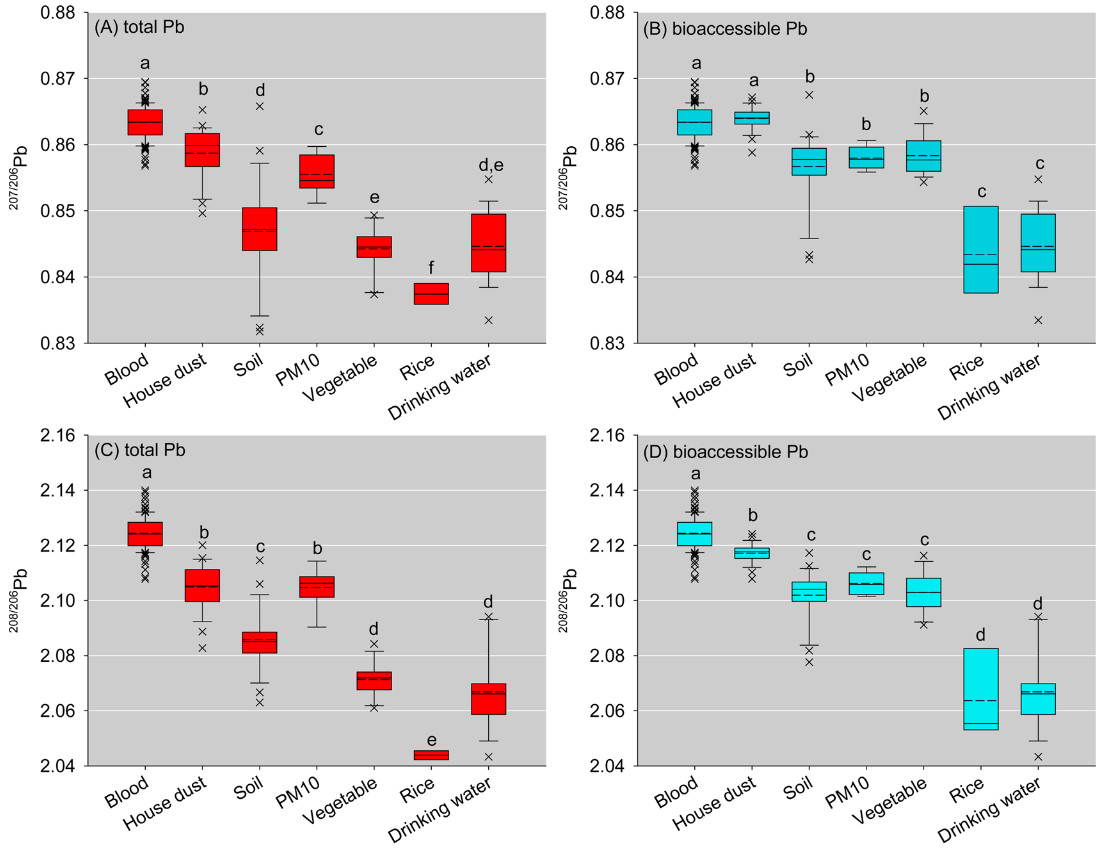

Figure 2. Variation in $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ isotopic ratios of total (A and C) and bioaccessible Pb (B and D) in different environmental media [housedust, soil, PM_{10}, vegetables, rice, and drinking water] collected from Zhuhang Town, Nantong City, China. Boxes represent the 25th to 75th percentiles. Solid and dashed lines in boxes are the median and mean values, respectively. Error bars represent the 5th and 95th percentiles, and time signs show outliers. Means marked with different letters indicate significant ($p < 0.05$) differences.

particulate matter (PM_{10}) or tracking in of Pb dust from wire rope production workers.

A strong association between BLLs and Pb concentrations in air has been identified.[40] In Zhuhang, Pb concentrations in indoor and outdoor PM_{10} particles (1168 ± 146 and 1097 ± 103 mg kg^{-1}, respectively) were similar (Table 1), suggesting their analogous Pb source. However, Pb loading was significantly lower in indoor air (range of 35−108 ng m^{-3}; mean of 70 ± 32 ng m^{-3}) than in outdoor air (range of 66−252 ng m^{-3}; mean of 159 ± 103 ng m^{-3}) (Figure S3 of the Supporting Information). While limited information about indoor air PM_{10} Pb values in China is available, outdoor PM_{10} Pb loadings in Zhuhang were higher than that reported in other Chinese cities such as Beijing (110 ng m^{-3})[41] and within the range reported for an e-waste recycling site.[42] This suggests the contribution of anthropogenic activities (i.e., wire rope production) to elevated Pb loading.

In addition to nondietary pathways, children may be exposed to Pb through dietary pathways.[43,44] A survey of readily consumed vegetables in Zhuhang showed that Pb concentrations in edible portions were 0.02−1.9 mg [kg of fresh weight (FW)]$^{-1}$ {0.1−27.3 mg [kg of dry weight (DW)]$^{-1}$} (Table 1). Among the samples analyzed, 58% had Pb concentrations above the maximal allowable level of 0.2 mg (kg of FW)$^{-1}$ proposed by the Chinese Ministry of Health, including spinach [0.64 ± 0.85 mg (kg of FW)$^{-1}$ (n = 4)], pakchoi [0.35 ± 0.29 mg (kg of FW)$^{-1}$ (n = 15)], and chives [0.42 ± 0.22 mg (kg of FW)$^{-1}$ (n = 3)] (Figure S4 of the Supporting Information).[45] Leafy green vegetables have previously been shown to contain elevated concentrations of metals.[46,47] Accumulation of Pb may occur in vegetables grown in farmlands surrounded by WRFs via root or foliar uptake resulting from impacted soils or atmospheric deposition, although foliar uptake may be the predominant pathway because of the low Pb concentrations in soils.

In contrast to vegetables, low Pb concentrations (11.9−35.0 μg kg^{-1}) were detected in 9 rice samples randomly collected from Zhuhang households, the staple food for the study cohort (Table 1). The observed values were significantly lower than the limit of 0.2 mg kg^{-1} proposed by the Chinese Food Standards Agency.[48]

The concentrations of Pb in 21 first-flush tap water samples from randomly selected households (0.02−4.46 μg L^{-1}; mean of 0.72 ± 1.12 μg L^{-1}) (Table 1) were well below the Chinese limit of 10 μg L^{-1} but higher than those in tap water from Shanghai [0.01−0.49 μg L^{-1} (n = 14)].[13] Housing in Zhuhang was recently (2004−2005) constructed with Pb-free water servicing lines. Therefore, drinking water contributed little to the Pb budget in children of Zhuhang compared to that seen for older houses in other locations (e.g., United States, up to 910 μg L^{-1}).[44,49]

Pb Bioaccessibility in Environmental Matrices. When considering exposure of the Zhuhang cohort to Pb, the magnitude of exposure will be influenced by the exposure frequency and duration in addition to the concentration and bioavailability of Pb in various environmental media. Data from total Pb concentrations may be used to calculate daily Pb intake values (Table S1 of the Supporting Information); however,

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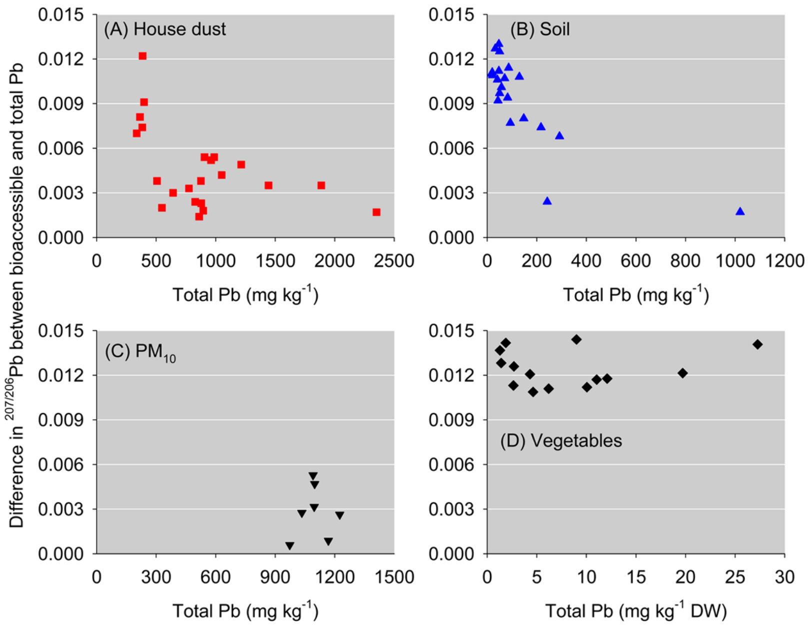

Figure 3. Differences in the $^{207}Pb/^{206}Pb$ isotope ratio between bioaccessible and total Pb vs total Pb concentrations in (A) housedust, (B) soil, (C) PM_{10}, and (D) vegetable samples collected from Zhuhang, China.

these calculations do not consider the influence of Pb bioavailability, which is influenced by physiochemical properties of the matrix in addition to physiological parameters of the individual.[26] In an attempt to refine exposure, *in vitro* methods were used as a surrogate to estimate Pb bioavailability in 24 housedust (337–2352 mg kg^{-1}), 22 soil (17.7–1020 mg kg^{-1}), 8 PM_{10} (973–1372 mg kg^{-1}), 25 vegetable [0.57–27.3 mg (kg of DW)$^{-1}$], and 9 rice (11.9–35.0 μg kg^{-1}) subsamples. Small variations in Pb bioaccessibility were found among different housedust and soil samples (Figure S5 of the Supporting Information). The average Pb bioaccessibilities were 81.4 ± 5.8 and 70.7 ± 8.8% for Pb in housedust and soil, respectively. Similarly, the mean Pb bioaccessibilities in PM_{10}, vegetables, and rice were 86.1 ± 6.0, 69.6 ± 9.2, and 65.0 ± 3.6%, respectively. These values fall within the range that has been measured in housedust (46–99%),[27] soils (34–99%),[26] PM_{10} (9–98%),[50] and vegetables (20–68%),[51] although there is a dearth of information about the bioaccessibility of Pb in rice.

Variation of Stable Pb Isotope Ratios among Environmental Media and Children's Blood. Understanding stable Pb isotopic data in environmental matrices and children's blood may facilitate the identification of major Pb exposure pathways for children.[9] Total Pb isotope ratios ($^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$) were significantly ($p < 0.05$) higher in housedust (0.8587 ± 0.0039 and 2.1049 ± 0.0087) than in soil (0.8469 ± 0.0075 and 2.0858 ± 0.0108), PM_{10} (0.8523 ± 0.0018 and 2.0919 ± 0.0041), vegetables (0.8442 ± 0.0033 and 2.0713 ± 0.0063), rice (0.8374 ± 0.0022 and 2.0439 ± 0.0023), or drinking water (0.8446 ± 0.0054 and 2.0668 ± 0.0141) (Figure 2A,C and Table S2 of the Supporting Information). Lead isotopic ratios were also determined for bioaccessible Pb in environmental media. Compared to total Pb, $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ isotope ratios were significantly ($p < 0.05$) higher for bioaccessible Pb in housedust, soil, vegetables, and rice (Figure 2B,D and Table S2 of the Supporting Information). The difference in the $^{207}Pb/^{206}Pb$ isotope ratio between bioaccessible and total Pb was greater in housedust and soil having low Pb concentrations, while it did not vary with Pb concentrations in PM_{10} and vegetables (Figure 3). Lead isotope ratios for bioaccessible Pb in PM_{10} were also elevated compared to the total Pb concentration; however, the difference was insignificant ($p > 0.05$) (Figure 2 and Table S2 of the Supporting Information).

Previous studies have demonstrated that $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ ratios for extractable Pb were higher than ratios determined for total Pb in contaminated soil and dust.[21,27,52] This was attributed to differences in the extractability of Pb from anthropogenic and geogenic source, resulting in the differences in Pb isotope ratios. Compared to naturally derived Pb in crystal structures, anthropogenic Pb present as Pb carbonate, Pb oxide, and adsorbed to Fe/Al oxides exhibits higher $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ ratios and are more soluble in acidic solutions such as those found in gastric fluids.[10,53] As housedust and soil samples contained a mixture of geogenic and anthropogenic Pb, total Pb showed a mixed isotopic composition, while bioaccessible Pb mainly represented anthropogenic Pb inputs. For housedust and soil having higher Pb concentrations, the contribution of geogenic Pb to total Pb became smaller; therefore, little difference in isotope ratios between bioaccessible and total Pb was observed (Figure 3). The small difference in isotope ratios between total and bioaccessible Pb for PM_{10} was also caused by the high Pb concentrations in this medium (973–1372 mg kg^{-1}) as a result of the predominance of anthropogenically-derived Pb as well as high Pb bioaccessibility (86.1 ± 6.0%).

When stable Pb isotope ratios were determined in blood samples for the study cohort, the ratios did not vary significantly, except several children with BLLs of <5 μg dL^{-1} had lower isotope ratios (Figure S6 of the Supporting Information). Stable $^{207}Pb/^{206}Pb$ and $^{208}Pb/^{206}Pb$ isotope ratios in 115 children blood samples were 0.8586–0.8695 and 2.1078–2.1399 averaging 0.8633 ± 0.0025 and 2.1244 ± 0.0061 (Table S2 of the Supporting Information). No correlation was found between Pb concentrations and isotope ratios of blood samples. A similar small variation of blood Pb isotope ratios (0.8597 ± 0.0035 and 2.1119 ± 0.0101) has been

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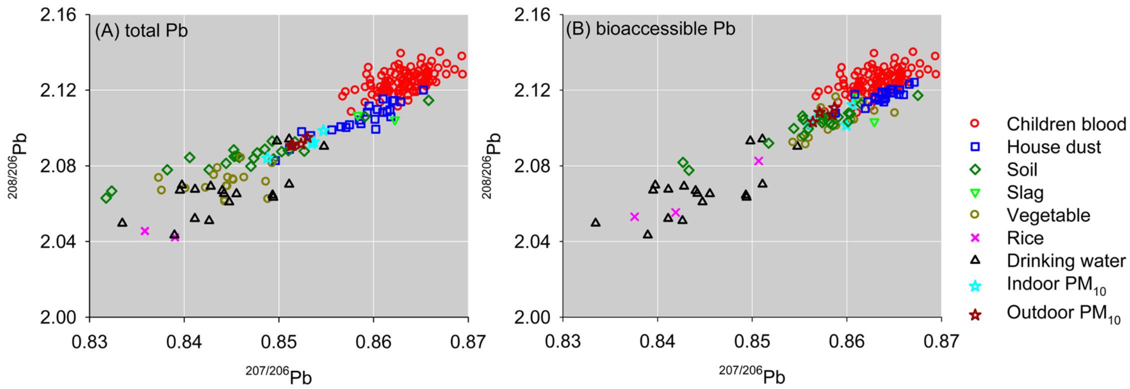

Figure 4. Comparison of isotopic composition ($^{208}Pb/^{206}Pb$ vs $^{207}Pb/^{206}Pb$) in children's blood Pb with that of total Pb (A) and bioaccessible Pb (B) in housedust, soil, airborne suspended particle matter (PM_{10}), drinking water, rice, vegetable, and wire rope production slag samples collected from Zhuhang Town, Nantong City, China.

reported for children from other Chinese cities such as Shanghai.[13] The lack of variability in stable Pb isotope ratios and the variability observed among environmental media suggest that children are exposed to a predominant Pb source.

Isotope Fingerprinting of Blood Pb Based on Total and Bioaccessible Pb in Environmental Media. Because Pb isotope ratios varied significantly between total and bioaccessible Pb in environmental media, source identification of children's blood Pb based on isotope ratio analyses was assessed using data derived for both approaches. Initially, the isotopic ratios ($^{208}Pb/^{206}Pb$ and $^{207}Pb/^{206}Pb$) of blood Pb were compared to those of total Pb in environmental media (Figure 4A). Among different environmental media, only isotopic ratios of housedust were close to blood Pb values, as housedust had the highest Pb isotope ratios among the environmental samples. However, there were still significant ($p < 0.05$) differences in the ratios between blood Pb (0.8634 ± 0.0027 and 2.1244 ± 0.0061) and housedust total Pb (0.8587 ± 0.0039 and 2.1049 ± 0.0087) (Figure 2A,C). Therefore, using this approach, we cannot identify a predominant Pb exposure pathway for the studied cohort.

When isotopic ratios of children's blood Pb were compared to those of bioaccessible Pb in environmental media, we found that children's blood Pb isotope ratios resembled those of bioaccessible Pb in housedust (0.8639 ± 0.0018 and 2.1171 ± 0.0036) (Figure 4B). However, they were different from those of bioaccessible Pb in other environmental media. There was no significant difference ($p > 0.05$) in $^{207}Pb/^{206}Pb$ ratios between blood Pb and housedust bioaccessible Pb (Figure 2B), although the $^{208}Pb/^{206}Pb$ ratio in the housedust was slightly lower than that of blood Pb (Figure 2D). This may suggest that an additional contributor of blood Pb with a higher $^{208}Pb/^{206}Pb$ ratio was possible. There were no significant differences in bioaccessible Pb isotope ratios between indoor and outdoor PM_{10}, and between soil (0.8566 ± 0.0055 and 2.1019 ± 0.0091) and PM_{10} (0.8580 ± 0.0017 and 2.1061 ± 0.0040) (Table S2 of the Supporting Information), suggesting an analogous anthropogenic Pb source in soil and PM_{10} in Zhuhang. However, Pb isotope ratios for soil and PM_{10} were significantly lower than blood Pb ratios, though they were elevated in bioaccessible Pb. Similarly, the isotope ratios of bioaccessible Pb in vegetables (0.8583 ± 0.0029 and 2.1029 ± 0.0071), rice (0.8434 ± 0.0067 and 2.0636 ± 0.0165), and drinking water (0.8446 ± 0.0054 and 2.0668 ± 0.0141) were different from blood Pb ratios, indicating that dietary pathways, incidental soil ingestion, and inhalation pathways were minor contributors for exposure of children to Pb in Zhuhang. This is consistent with the report that drinking water is a minor contributor to exposure of Chinese children to Pb.[13] Therefore, using bioaccessible Pb ratios, we successfully identified the predominant Pb exposure pathway for children in Zhuhang, i.e., incidental ingestion of housedust. This was consistent with severe Pb contamination of housedust compared to other environmental media in addition to its high Pb bioaccessibility.

Isotope ratios of bioaccessible Pb in housedust were similar to those in slag produced at wire rope factories (Table S2 of the Supporting Information). Furthermore, isotope ratios of Pb in each environmental medium formed a linear distribution in the plot of $^{208}Pb/^{206}Pb$ versus $^{207}Pb/^{206}Pb$, with slag as an anthropogenic end member (Figure 4), confirming that environmental Pb contamination in Zhuhang primarily arose from emission of Pb from wire rope production. Atmospheric transport and deposition of Pb emitted from wire rope factories are an important source of Pb in housedust in Zhuhang. However, additional deposition routes may also be responsible for elevated Pb concentrations in housedust, including tracking in production dust, via work clothes from factory employees, because Pb concentrations in some housedust samples were higher than in indoor PM_{10} (1168 ± 146 mg kg^{-1}).

Fingerprinting based on Pb isotope ratios has the potential to identify Pb exposure sources and pathways. However, when measuring Pb isotope ratios is difficult, calculation of daily Pb intake from various pathways based on Pb concentration and bioaccessibility in environmental media can be used to assess predominant exposure pathways.[54] We calculated Pb intake for children, showing that housedust ingestion contributed 54.7% of the total Pb intake (Table S1 of the Supporting Information), supporting the stable Pb isotopic results. However, Pb isotopic fingerprinting was more powerful for discarding the minor contributors, because Pb intake calculations overestimated the contribution from vegetable consumption, i.e., 35.5%. Though some vegetables were contaminated with Pb in Zhuhang, their different isotope ratios from blood Pb showed that consumption of vegetables was not a major Pb contributor, possibly because of the limited amount of vegetables consumed by the study cohort or limited Pb uptake from vegetable consumption.[55]

Previous comparisons of stable Pb isotopic ratios of blood and environmental media have utilized isotope data based on total Pb in environmental samples without considering the bioaccessible fraction.[12−14] Using this approach, some studies successfully identified Pb exposure pathways.[12,13] Their success may be due to very high Pb concentrations in housedust (1200−3300 mg kg^{-1}), soil (1220−1370 mg kg^{-1}), and air particulate (3077 mg kg^{-1}) samples, which narrowed the differences in isotope ratios between bioaccessible and total Pb (Figure 3). These successful cases suggest that when environmental media are highly contaminated with Pb (e.g., >1000 mg kg^{-1}), total Pb isotope ratios are suitable for identifying Pb exposure pathways because of their small differences from bioaccessible Pb ratios. However, with low Pb concentrations in environmental samples, total Pb isotope determination is inaccurate in identifying blood Pb sources because of its large variation from bioaccessible Pb isotope ratios. For example, on the basis of the total Pb determination, Cao et al.[14] found that isotope ratios of Pb in soil and housedust were much lower than children blood Pb ratios. However, if isotope ratios were determined for bioaccessible Pb in soil and housedust of Cao et al.,[14] which had low Pb concentrations (39 and 24 mg kg^{-1}, respectively), isotope ratios much higher than those of total Pb and similar to that of blood Pb might be obtained.

In this study, we found that comparing isotope ratios of children blood Pb to those of total Pb in environmental matrices failed to identify Pb sources. However, coupling stable Pb isotope analysis of children's blood Pb to bioaccessible Pb in environmental media provided a more robust approach to identifying the predominant Pb exposure pathway, especially when the level of environmental Pb contamination was low and there were large differences in Pb isotope ratios between total and bioaccessible Pb because of their different sources. The Pb isotope ratios in bioaccessible Pb better reflected the contribution to blood Pb than total Pb in environmental media.[20] Using this strategy, incidental ingestion of housedust was identified as the major Pb exposure pathway for children in Zhuhang. Future Pb monitoring programs could use this coupled technique to determine whether Pb mitigation strategies aimed at reducing exposure via the housedust ingestion pathway are effective at reducing BLLs in children.

■ ASSOCIATED CONTENT

Ⓢ Supporting Information

Description of environmental sample collection and bioaccessible Pb and isotope ratio analysis, the study area, variation of BLLs with children's gender and age, total Pb concentrations in environmental matrices, Pb bioaccessibility in housedust and soil, variation of blood Pb isotope ratios with BLLs (Figures S1−S6), daily Pb intake calculation (Table S1), and isotope ratios of bioaccessible and total Pb (Table S2). This material is available free of charge via the Internet at http://pubs.acs.org.

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Notes

The authors declare no competing financial interest.

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