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Probabilistic risk assessment of residential exposure to metal(loid)s in a mining impacted community

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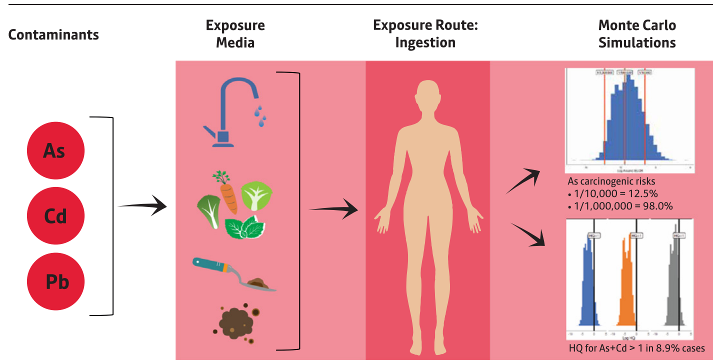
HIGHLIGHTS

- Community-based participatory research methods used to engage "Gold Country" CA females concerned about breast cancer
- 40 participants completed surveys and collected a total of 354 water, soil, home-grown foods, and dust samples
- Arsenic exposure from water (63.5%) and home-grown food (33.3%) poses carcinogenic risk above an EPA recommended limit
- Cadmium exposure results mainly from home grown food consumption (83.7%)
- A broader range of sources contribute to lead exposure

GRAPHICAL ABSTRACT



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ABSTRACT

The "Gold Country" region of California is impacted by legacy and active gold mines. Concomitantly, Gold Country has an increased rate of female breast cancer relative to the state average. Using community-based participatory research methods, 40 participants completed surveys and collected a total of 354 water, soil, home-grown foods, and dust samples from their homes, which we compared to state, federal, and international contamination standards for arsenic, cadmium, and lead. All soil samples exceeded U.S. EPA and California EPA soil standards for arsenic. When comparing other media to state, federal and international standards for arsenic, cadmium, and lead, 15 additional exceedances for indoor/outdoor dust, drinking water, and/or vegetable were documented. A probabilistic risk assessment was conducted to determine an adult female's exposure to arsenic, cadmium, and lead and estimated risk. Arsenic exposure, due largely to water (63.5 %) and homegrown food (33.3 %), presents carcinogenic risks in excess of the EPA recommended upper limit for contaminated sites (1×10^{-4}) in 12.5 % of scenarios, and exceeds a risk level of 1×10^{-6} in 98.0 % of cases. Cadmium exposure results mainly from homegrown food consumption (83.7 %), and lead exposure results from a broader range of sources. This research indicates that rural areas in Gold Country face environmental exposures different than in urban areas. Exposure to arsenic in the female population of Gold Country may be driven by consumption of home-grown foods and water, and exposure to cadmium is driven by home-grown food intake. Since mining sites are of concern internationally, this risk assessment process and associated findings are significant and can be used to inform and tailor public health interventions. The weight of the evidence suggests that the arsenic exposure identified in this study could contribute to increases in the cancer rate among those living in Gold Country, California.

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1. Introduction

Activities such as mining and smelting generate large amounts of metal (loid)-contaminated wastes. Contaminants in wastes such as slag and mine tailings are subject to windblown transport of contaminated particles, uptake by plants and movement through ecosystems, as well as leaching or runoff of metal(loids) into the surrounding soil, surface water, or groundwater (Singh et al., 2018). Communities within or neighboring resource extraction activities often experience negative health outcomes (Goldenberg et al., 2010). Exposure pathways may vary based on the metal(loid) and individual and community activities. Chronic exposure to metal(loid)s such as arsenic, cadmium, and lead increases the likelihood of negative health outcomes such as arsenical keratoses, kidney failure, and heart disease. Both arsenic and cadmium are known to have carcinogenic effects and are suspected to contribute to the etiology of and are associated with breast cancer specifically (Rahimzadeh et al., 2017; Ratnaike, 2003; Khanjani et al., 2017; Gallagher et al., 2010). It is critical to understand the risks posed by metal(loid) exposure to protect the health of mining-impacted communities. Using a Markov Chain Monte Carlo-based probabilistic site-specific risk assessment method, the objective of this study was to estimate the exposure to arsenic, cadmium, and lead from active and legacy mining in "Gold Country" California and determine whether this is leading to increased community health risks.

The "Gold Country" region of California is located in the foothills of the Sierra Nevada Mountain range in eastern California. This region was dominated by intensive mining operations starting in 1848 during the California gold rush and mostly ending around 1964 (Craig and Rimstidt, 1998), however mining is still occurring today (Fig. 1). These mines generated wastes contaminated with metal(loid)s occurring naturally with gold deposits, such as arsenic and cadmium, which have, to an unknown extent, been released into the environment. Previous studies have assessed the state of environmental exposure to, and body burden of, these contaminants for those living in the area, and results have prompted further study (Manjón et al., 2020; Von Behren et al., 2019).

Community residents have long been concerned about possible exposures to legacy mining contaminants, especially in light of the high breast cancer incidence rates observed. From 2015 to 2019, Placer and Nevada counties represented the 2nd and 6th highest rates of breast cancer out of California's 58 counties at 143.4 and 139.3 cases per 100,000 females, respectively. These rates are higher than the average California cancer rate at 124.1 per 100,000 (California Cancer Registry, 2022) and United States 2018 breast cancer rate at 126.8 per 100,000 females (U.S. Cancer Statistics Working Group, 2021).

To address this community concern and potential exposure to arsenic, cadmium, and lead, community-based participatory research (CBPR) and community science approaches were used to co-produce a population and site-specific risk assessment for females living in the Gold Country area. In this paper, we describe the environmental monitoring and risk assessment process conducted in the Gold Country region of California. Using the Markov Chain Monte Carlo framework, we present the estimated cumulative exposure to arsenic, cadmium, and lead from soil, water, homegrown foods, and dust as well as the carcinogenic and non-carcinogenic risks posed by arsenic and cadmium.

2. Methods

This exposure assessment study, a CBPR partnership between the University of California San Francisco, University of Arizona *Gardenroots*, and Sierra Streams Institute (SSI), a non-governmental watershed monitoring, research, and restoration group based in Nevada City, California, was the third in a series of related studies in this community and is referred to hereafter as CHIME 3. The study goal was to determine whether arsenic and cadmium exposures are associated with the consumption of local foods, gardening, and trail use in areas with mining residue.

2.1. Site description

CHIME 3 sampling was conducted at participant's homes and farms distributed throughout Gold Country, and concentrated in Nevada County. Participant demographics are given in Supplemental Table 1.

2.2. Recruitment

To recruit study participants, flyers were distributed in English and Spanish throughout communities in Nevada, Placer, and El Dorado

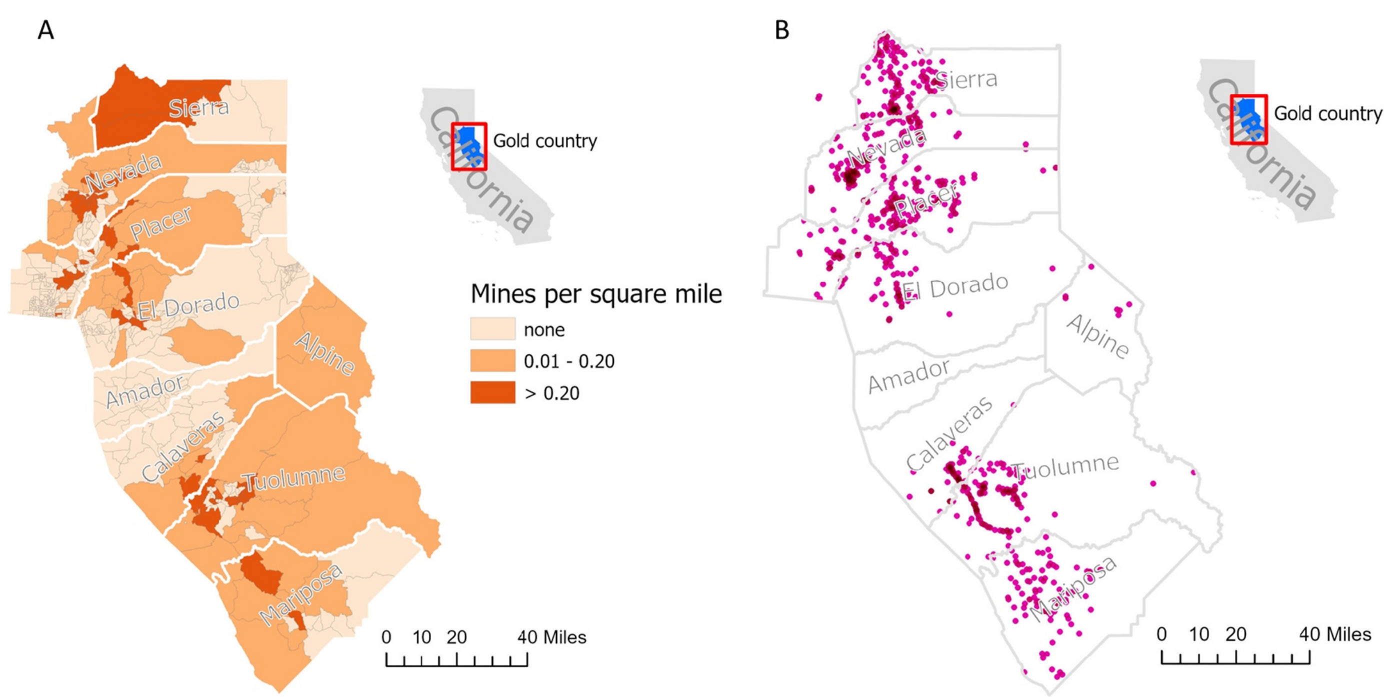

Fig. 1. (A) Mine density by block group for Gold Country and (B) Mine locations based on selected data from the Principal Areas of Mine Pollution dataset from California Department of Conservation.

counties. Flyers were posted at grocery stores, local markets, bars, salons, cafes, gas stations, churches, parks, trailheads, and retail stores. Before the COVID-19 pandemic and associated lockdowns, we visited several very small communities to meet with locals and discuss the study. Local newspapers and radio stations were contacted via email and phone to make announcements to communities. Flyers were placed on SSI's website, social media pages, and newsletter as well as a blog posts to recruit study volunteers. Several local women's organizations and environmental groups shared our posts to expand recruitment. We also relied on word of mouth (snowball recruiting) in small, rural communities where there is a large distrust of outsiders and especially environmental, science/research-oriented organizations. Farmers were also recruited via word of mouth, phone calls, and emails by a SSI staff member who is also a local farmer. Unfortunately, due to the onset of COVID-19 mitigation measures in early March of 2020, we were unable to conduct in-person outreach or distribute flyers in Sierra, Amador, Alpine, Calaveras, Tuolumne, and Mariposa counties. Following the implementation of quarantine measures, recruitment efforts were conducted online and through outreach in local newspapers and radio stations (including Spanish speaking radio stations) in the nine targeted counties.

2.3. Survey administration

Before participation in the study, each participant electronically signed a consent form through RedCap, an online survey and eConsent platform (Harris et al., 2009; Harris et al., 2019). The study protocol was reviewed and approved by the University of California, San Francisco Institutional Review Board (IRB).

Participants were asked to complete two surveys: (1) Food Frequency (FF) and (2) Exposure Assessment online via Qualtrics (Version: August 2020. Copyright © 2020, Qualtrics). The FF survey collected information on the rate of consumption of different foods previously identified as accumulators of arsenic, lead, and/or cadmium and/or commonly consumed in the partnering community (Manjón and Ramírez-Andreotta, 2020). The Exposure Assessment survey gathered demographic information as well as information about factors that might be predictive of a participant's exposure to metal(loid)s. Due to inconsistency in reporting, participant FF survey data was not used in the exposure and risk calculations, see section "Intake Rate Simulation." Many exposure assessment survey questions allowed for input of additional information as text, selection of multiple options, or were entirely open ended. To reduce dimensionality, select open-ended survey responses were coded into categorical variables. Questions used for modeling, along with the codebook are provided in Supplemental Table 2.

2.4. Community science design and community trainings

Participants were trained and given sample collection kits, which included the needed materials and an instruction manual with steps for proper sample collection. Environmental monitoring sampling kits and instruction booklets where prepared at the UArizona and shipped to each participant's home. Once sampling kits were received by participants, we provided three "live online" community trainings via Zoom (version 4.4.6619) for participants. Each training lasted approximately 2 h and covered all the necessary sampling protocols and techniques. One of the trainings was recorded with participant consent and made available to those participants who were unable to attend the live trainings. Participants were also provided contact information for university researchers and SSI staff to ask any questions. Participants then completed the field sampling procedures outlined below. To view the sampling instructions and protocols, visit: https://gardenroots.arizona.edu/community-status. Participants were instructed to store samples in their refrigerator until the designated drop off dates and time with SSI staff at designated outdoor locations. Once received by SSI staff, samples were immediately placed in a refrigerated container and a chain of custody form was completed by both the participant and SSI staff member. SSI staff then shipped participant samples via USPS in chilled coolers to the Integrated Environmental Science and Human Health Risk Laboratory at the UArizona. A select number of participants were sent sampling kits with ice packs and cooler liners and sent samples to our lab directly. A total of 100 households were recruited, 62 were trained and received a kit, and 40 households returned samples for analysis.

2.5. Sample collection, intake, preparation, and analysis

2.5.1. Water

Following proper protocols (wearing gloves, sample labeling, etc.), participants collected water, soil, settled dust, and plant samples. Participants collected one 50 ml water sample in a trace metal-free 50-mL tube (VWR, Cat. Number 89049-17) after running the water for 3 min prior to sample collection. In addition, the participant collected a 50 ml field blank sample near the water source also using trace metal-free 50-mL tubes. Nanopure water (>18 MΩ deionized Nanopure water, Millipore) was provided to participants as field blank water, which was carefully transferred into a clean and empty sample tube.

Upon arrival to the lab, samples were refrigerated. To measure dissolved metals, we followed the U.S Environmental Protection Agency (EPA) Method 200.8 (U.S. EPA, 1994a). A 20 g sub sample of each water sample was filtered using 40 um filters (Environmental Express), acidified with 0.2 g of 2 Molar Nitric Acid (VWR Analytic) for shelf stability, and analyzed using Inductively Coupled Plasma Mass Spectroscopy (7700×, Agilent, Santa Clara CA).

2.5.2. Soil

Participants were instructed to collect a composite soil sample from their yard (unamended) and garden soils (amended). The participants selected six spots in a grid-like pattern in both their yard and garden areas and collected the top 15 cm of soil from each spot. Participants then composited and mixed the soil samples thoroughly (bulk sampling) in two plastic buckets, one designated for yard soil and the other for garden soil. If participants did not have a garden, they were instructed to follow the protocol above and collect two soil samples at their discretion.

All soils were air dried upon arrival to the lab. Following air drying, a sub sample of soil was reserved for pH and electrical conductivity (EC) analysis. The remainder of the soil was passed through a 2 mm sieve, and a portion was oven dried at 60C° to constant mass. Dried soil samples were then milled for 5 min (Spex Sample Prep Dual Mixer Mill 8000D), digested with concentrated (70 %) nitric acid, and analyzed for arsenic, cadmium, and lead via ICP-MS. Soil texture, pH, and EC are not reported in this communication and can be retrieved in Huerta et al. (2021).

2.5.3. Settled dust

Participants were instructed to collect settled dust samples from three different locations on their property using Housing and Urban Development (HUD) Guidelines (HUD, 2012). Using Ghostwipes (Environmental Monitoring Systems), participants collected a sample from: (1) a hard surface floor in a non-entryway area of their house, (2) their outdoor porch, or if they had no porch, another hard surface outdoors, and (3) from one entire indoor windowsill. For the floor and porch samples, we provided participants with a 30x30cm sampling frame to standardize sampling area across the study. Participants were asked to record the area of their windowsill and to attach photos of all sampling areas. For all samples, participants were instructed to wipe first in an "S" shaped motion over the whole area, then in the same "S" shaped motion sideways, and finally to wipe the inside perimeter of their sampling area. Participants were also instructed to collect a blank ghost wipe sample at each of their sampling locations by removing a ghost wipe from its packaging, completely unfolding it, refolding it, and placing it in a sample container.

Ghostwipe samples were immediately refrigerated upon arrival to the UArizona lab. All ghostwipe samples were weighed, acid digested with 70 % nitric acid and analyzed using ICP-MS. The total amount of each element in the sample was converted to units of ug m^{-2} using the area of the sampling frame for indoor floor and outdoor porch samples, and the measurements provided by participants for indoor windowsill samples.

2.5.4. Plants

Participants collected four replicates of three edible plants in their garden and stored them separately in provided Whirl-Pak® bags. The selection of plant was left to the participant's discretion. We instructed participants to only collect plants the day before sample drop off and to store plants in their refrigerator to minimize spoilage.

Upon arrival to the lab, samples were refrigerated. Inedible or rarely eaten portions of plant samples were removed using a knife and all plant samples were washed with deionized water for 30 s to mimic a typical plant washing process in the home. Following this, all plant samples were weighed and dried in an oven at 60 °C to constant mass. Plant samples were homogenized using a mortar and pestle, acid digested, and analyzed using ICP-MS.

The method detection limit (instrument detection limit times dilution factor) by media and element are given in Supplemental Table 3. When a value was non-detect, the MLOD was divided by √2.

2.6. Bioconcentration factor calculation

To analyze patterns in plant uptake of metal(loid)s, we calculated plant Bioconcentration Factors (BCF's) using paired soil and plant data and the following equation:

$$Bioconcentration\ factor = \frac{Dry\ plant\ concentration\ of\ Metal(loid)}{Dry\ soil\ concentration\ of\ Metal(loid)}$$

Using participant sampling notes, we excluded 28 samples from calculation of BCF values, as they were not grown in the same soil that the participant sampled, but instead were purchased or grown in containers.

2.7. Exposure estimation

To simulate the Average Daily Dose (ADD, milligrams per kilogram of body weight per day), and Lifetime Average Daily Dose (LADD) of arsenic, cadmium, and lead from water, soil, locally grown plants, and settled indoor dust, we performed an exposure assessment using Monte Carlo Simulations in Oracle Crystal Ball (11.1.2.4, 2021). Inhalation of airborne dust was not considered, as the environmental data to support such assessment was not available, and inhalation of airborne dust in previous work was not identified as a major contributor to arsenic, cadmium, and lead exposure (Manjón et al., 2020). A Monte Carlo simulation was used to generate distributions of ingestion rates, metal(loid) concentrations, and other exposure factors. When possible, we selected population specific values and distributions using participant demographics reported in the Exposure Assessment Survey. From these simulated values, ADD and LADD were calculated using the following equations:

$$ADD\ \left[mg\ kg^{-1} day^{-1}\right] = \frac{C_{p,s,d,w} \times IR \times BAF \times CF \times EF \times ED}{BW \times AT - NC}$$

$$LADD\ \left[mg\ kg^{-1} day^{-1}\right] = \frac{C_{p,s,d,w} \times IR \times BAF \times CF \times EF \times ED}{BW \times AT - C}$$

where IR = ingestion rate, BAF = bioaccessibility factor, CF = conversion factor (0.001 mg μg^{-1}), EF = exposure frequency, ED = exposure duration, BW = body weight, AT-C = averaging time (cancer) and AT-NC = averaging time (non-cancer). Specific values used are provided in Supplemental Material Table 3.

ADD and LADD were calculated for each metal(loid) from each specific source and cumulatively.

2.7.1. Simulation of concentrations of arsenic, cadmium and lead

We simulated concentrations of arsenic, cadmium, and lead in environmental media using distributions ranked by the Anderson Darling test. A significance level of $\alpha = 0.01$ was selected for these tests due to the high number of distributions fit to the data (n = 15). All concentration distributions were modeled with a lower cutoff of 0. To account for correlations between concentrations of metals(loid)s, we simulated distributions using a correlation matrix between all metal(loid) concentrations. No distribution was rejected by Anderson Darling tests at $\alpha = 0.01$. All distributions but one were best fit with a lognormal distribution. The exception, arsenic in water, was best fit with a Gamma distribution, which is a right skewed, continuous distribution similar to a lognormal distribution.

For settled dust, the participant's indoor floor ghost wipe sample data were used to fit the distribution of contaminants in indoor dust. For water, the participant's submitted water sample data were used. Based on the U.S. EPA Exposure Factors Handbook (EFH) (U.S. EPA, 2011), participant plant samples were categorized as either: exposed vegetables (vegetables eaten without removal of exterior) exposed fruits (fruits eaten without removal of exterior), and root vegetables (vegetables whose roots are eaten) and the associated concentrations by metal(loid) were fit to distributions separately. For participants submitting two or more samples of the same plant type, (n = 30), concentrations among plant types were averaged and considered as 1 sample for the purpose of distribution fitting. Due to the low number of “root vegetables” (n = 5), the concentration of metal(loid)s in root vegetables was simulated as a point value which was the mean of all root vegetable samples. Plants submitted from stores and not grown locally were excluded from consideration.

We used participants' yard soil sample concentrations to fit the distribution of metal(loid)s in soil. This decision was made since garden soil is representative of a smaller area, i.e., their garden and the EFH recommended value for soil ingestion is inclusive of outdoor dust as well as soil (U.S. EPA, 2011), making the yard the better representative sample. For participants who submitted two yard samples (n = 4), concentrations were averaged and considered as one sample for the purpose of distribution fitting.

2.7.2. Intake rate simulation

Intake rates for plants, soil, dust, and water were drawn from the US EPA's EFH and are in Supplemental Table 4. The intake rates of plants, water, and dust were modeled as normal distributions from summary statistics. The intake rate of soil was provided and modeled as a point value. Plant consumption was calculated by simulating both a random consumption rate and whether or not a person consumed each of the three plant categories using proportions provided by the EFH.

2.7.3. Bioaccessibility factors

Bioaccessibility factors were used in the plants and soil exposure calculation. Due to our sampling method's inability to distinguish between dust aerodynamic diameters, and the wide variety of bioaccessibility values observed for dust (Kastury et al., 2017), we used a conservative assumption of 100 % metal(loid) bioaccessibility in dust. Plant As bioaccessibility was based on assumptions in previous work (e.g., Ramirez-Andreotta et al., 2013). Soil As and Pb bioaccessibility were drawn from U.S. EPA recommendations (U.S. EPA, 2012; U.S. EPA, 1994b).

To provide conservative estimates where recommended values were unavailable, bioaccessibility values for plant Cd and Pb, as well as soil Cd were drawn from the highest reported value in selected peer reviewed bioaccessibility studies. Bioaccessibility values and sources are provided in Supplemental Table 4.

2.7.4. Body weight, exposure duration, and average time

Body weight reported for females ages 50–60 was used (U.S. EPA, 2011). Exposure duration was assumed to be 35 years, which is the 95th percentile value for female residential occupancy period (U.S. EPA, 2011). Average time (noncancer) was assumed to be 35 years and average time (cancer) was assumed to be 80 years, which is the mean female lifespan (U.S. EPA, 2011). Please note that exposure duration and average time account solely for the period of exposure.

2.8. Risk characterization

The Increased Excess Lifetime Cancer Risk (IELCR) from exposure to arsenic was calculated for each simulation using the expression:

$$IELCR = LADD * CSF$$

where CSF is the cancer slope factor for arsenic, 1.5 units risk per mg $kg^{-1}$$day^{-1}$ (U.S. EPA, 2006). IELCR was calculated for total exposure to arsenic, as well as by environmental media collected in this study. The Hazard Quotient for exposure to arsenic and cadmium was calculated for each simulation using the expression:

$$HQ = \frac{ADD}{RfD}$$

where RfD is the oral reference dose of either arsenic or cadmium, given in Supplemental Table 4 (U.S. EPA, 2006). HQ values were calculated for total exposure, as well as by source. Neither a reference dose, nor a cancer slope factor for lead were available from the US EPA's IRIS. Consequently, risk from lead exposure is not quantitatively accounted for in this study.

2.9. Statistical analysis and modeling

2.9.1. Plant concentration and bioconcentration by family

We asked participants with gardens to list plants they grow in their gardens. We then categorized these responses, as well as actual submitted samples by family and plant type as defined by the United Nation Food and Agricultural Organization's Codex Alimentarius.

We built two additional linear models of: (1) log transformed metal(loid) concentrations on a wet weight basis and (2) log transformed bioconcentration factors on a dry weight basis, using data from submitted plant and soil samples. We used plant family, metal(loid), and the interaction effect of these two variables as predictive terms in each model. For consistency, both models exclude plants for which BCF's were not calculated due to lack of associated soil data. We removed 2 observations (*Curcurbitacea* and *Rutacea*) due to submission of a single sample. All model analysis was performed in Rstudio (RStudio Team, 2020). Following modeling, we conducted Tukey tests on any variable in a model significant at $\alpha = 0.05$.

2.9.2. Comparison of homegrown vegetables to store bought foods

We compared the concentrations observed in the homegrown plant samples to the average concentrations of arsenic, cadmium, and lead measured in the U.S. Food and Drug Administration's total diet study from the years 2003–2017 (U.S. FDA, 2021). First, we summarized average concentrations for food groupings analyzed in the total diet study. Study samples were then matched to market basket sample types for which data were available and the mean for each sample type was determined (see Supplemental Table 5). The average of these means was then calculated and compared for each metal(loid). This was done to reduce sampling bias due to variability in the number of samples of certain vegetable types included in the total diet study.

3. Results

3.1. Participant sociodemographic data

Just over 97 % of the participants who submitted samples were white and ranged in age from 32 to 76 years of age, with a mean age of 57.15 years. Participants reported four different levels of educational attainment, with 10 % (24) reporting completion of some college, 42.5 % (17) reporting completion of a bachelor's degree, 40 % (16) reporting completion of a post graduate degree, and 7.5 % (3) reporting completion of a trade/technical program. Participant income levels were broken down using 2018 Housing and Urban Development (HUD) income limits for Nevada County. 10 % (4) participants reported being below HUD's Extremely Low Income limit, 17.5 % (7) reported being below HUD's Very Low income limit, 15 % (6) participants reported being below HUD's Low Income limit, 55 % (22) reported income above all income limits, and 2.5 % (1) choose not to answer. Further details are provided in Supplemental Table 1.

3.2. Concentrations and comparison to standards

A total of 40 sampling kits were received, however in some cases, participants did not submit all requested samples. Supplemental Table 6 details how many of each sample type were submitted, and by how many total participants. Sample values were compared to soil and water standards from the U.S. EPA (U.S. EPA, 2021a) and/or California Environmental Protection Agency (CAL/EPA, 2020), plant standards from the United Nations Food and Agricultural Organization's Codex Alimentarius (2019), and settled dust standards from the U.S. Department of Housing and Urban Development (U.S. HUD, 2017).

Mean concentrations of arsenic, cadmium, and lead in water samples were 1.97 μg L^{-1}, 4.13×10^{-2} μg L^{-1}, and 0.299 μg L^{-1} respectively (Table 1, Supplemental Fig. 1). Of these, two samples exceeded the U.S. EPA's primary drinking water standard for arsenic of 10 μg L^{-1} and none exceeded the standards for lead or cadmium. The mean concentration of arsenic, cadmium, and lead in garden soil samples was 7.4 mg kg^{-1}, 0.38 mg kg^{-1} and 74.3 mg kg^{-1} respectively (Table 1, Supplemental Fig. 2). In yard samples, the respective concentrations were 11.6 mg kg^{-1}, 0.29 mg kg^{-1} and 85.6 mg kg^{-1} (Table 1, Supplemental Fig. 2). Of these samples, five exceeded the U.S. EPA's Regional Screening Level (RSL) for non-carcinogenic risk from lead in residential soil of 400 mg kg^{-1} and eight exceeded the same standard from the California EPA of 80 mg kg^{-1}. All yard and garden soil samples ($N = 79$) exceeded the U.S. EPA and California EPA's RSLs for carcinogenic risk from arsenic of 0.68 and 0.11 mg kg^{-1} respectively. No samples exceeded the EPA or California EPA soil standards for cadmium. The mean concentrations of arsenic, cadmium, and lead in plant samples by fresh weight were 2.28×10^{-2} mg kg^{-1}, 1.35×10^{-2} mg kg^{-1}, and 2.01×10^{-2} mg kg^{-1} respectively (Table 1, Supplemental Fig. 3). Of these, one sample exceeded the Codex Alimentarius standard for cadmium and one exceeded the standard for lead. Standards set by Codex Alimentarius vary based on plant type. No Codex Alimentarius standard for arsenic is available for comparison to study samples. The mean concentrations of arsenic, cadmium, and lead in settled dust samples were 7.39 μg m^{-2}, 1.70 μg m^{-2}, and 26.9 μg m^{-2} for indoor floor samples; 7.41 μg m^{-2}, 72.9 μg m^{-2}, and 36.9 μg m^{-2} for indoor windowsill samples; and 20.6 μg m^{-2}, 1.22 μg m^{-2}, and 68.1 μg m^{-2} for outdoor porch

Table 1
Concentrations by environmental media and analyte. Column A = mean ± standard deviation and column B = median (low-high) range of metal(loid).

Environmental Media (N = 354)	As		Cd		Pb	
	A	B	A	B	A	B
Plant (mg kg^{-1}) n = 116	$2.28 \times 10^{-2} \pm 5.44 \times 10^{-2}$	8.742×10^{-3} (1.36×10^{-4}–0.539)	$1.35 \times 10^{-2} \pm 2.08 \times 10^{-2}$	5.829×10^{-3} (5.04×10^{-4}–0.156)	$2.01 \times 10^{-2} \pm 5.09 \times 10^{-2}$	6.304×10^{-3} (6.72×10^{-4}– 0.506)
Yard Soil (mg kg^{-1}) n = 40	11.6 ± 13.8	7.66 (0.78–87.27)	0.29 ± 0.24	0.20 (3.06×10^{-2}–0.93)	85.6 ± 252	14.33 (2.91–1275)
Garden Soil (mg kg^{-1}) n = 40	7.4 ± 4.81	5.27 (2.95–20.15)	0.38 ± 0.20	0.36 (0.10–0.83)	74.3 ± 218	13.88 (2.85–1157)
Indoor Floor Dust (μg m^{-1}) n = 40	7.39 ± 8.02	4.78 (2.53–48.76)	1.70 ± 6.45	0.266 (4.70×10^{-2}–40.58)	26.9 ± 74.5	6.02 (0.970–451.4)
Indoor Windowsill Dust (μg m^{-1}) n = 40	7.41 ± 5.7	4.97 (1.55–24.72)	72.9 ± 390	0.561 (3.38×10^{-2}–2461)	36.9 ± 85.9	12.0 (0.478–7.99)
Porch Floor Dust (μg m^{-1}) n = 40	20.6 ± 16.3	17.9 (3.51–89.0)	1.22 ± 1.56	0.609 (5.55×10^{-2}–2461)	68.1 ± 121	29.9 (2.36–633)
Water (μg L^{-1}) n = 39	1.97 ± 7.38	0.167 (3.12×10^{-2}–38.3)	$4.13 \times 10^{-2} \pm 1.78 \times 10^{-2}$	3.53×10^{-2} (1.69×10^{-2}–0.100)	0.299 ± 0.411	0.157 (2.89×10^{-2}–2.42)

samples (Table 1, Supplemental Fig. 4). The only U.S. standards available for metal(loid)s in settled dust are U.S. HUD lead dust action levels used to clear homes during lead abatement (U.S. HUD, 2017). Two indoor floor samples exceeded the HUD standard of 107 μg m^{-2} (10 μg $ft.^{-2}$) for indoor floor wipes and one sample exceeded the HUD standard of 430 μg m^{-2} (40 μg $ft.^{-2}$) for outdoor porch wipes.

3.3. Lifetime average daily dose (cancer) and average daily dose (noncancer)

The mean LADD values for arsenic, cadmium, and lead were: 3.41×10^{-5} mg kg^{-1} day^{-1}, 9.27×10^{-6} mg kg^{-1} day^{-1} and 1.50×10^{-5} mg kg^{-1} day^{-1} respectively. The mean ADD values for arsenic cadmium and lead were 9.81×10^{-5} mg kg^{-1} day^{-1}, 2.47×10^{-5} mg kg^{-1} day^{-1} and 4.38×10^{-5} mg kg^{-1} day^{-1} respectively. Contributions to total arsenic, cadmium, and lead exposure is given in Fig. 2. In general, arsenic exposure resulted mainly from plant (33.3 %) and water (64.5 %) consumption. Daily cadmium exposure was mostly due to plant consumption (83.7 %), with some exposure from ingestion of dust (11.2 %) and water (4.9 %). Daily lead exposures were driven by a wider variety of sources, with exposures due to plant consumption (47.3 %), soil ingestion (26.1 %), water consumption (20.2 %), and dust ingestion (6.3 %).

Within the three plant categories included in our exposure assessment, *exposed* vegetable intake contributed the most to intake of all three metal(loid)s, making up 88 % of arsenic from plants, 65.9 % of cadmium from plants, and 70 % of lead from plants. The contribution to metal(loid) consumption from each plant category is shown in Fig. 3.

3.4. Increased excess lifetime cancer risk (IELCR) estimation

The mean of all IELCR values was 5.13×10^{-5}, or approximately one in 20,000. In 98 % of simulations, the IELCR from arsenic exposure exceeded a risk level of one-in-a-million, which is considered the lower limit of the EPA's acceptable risk range for carcinogenic risks (U.S. EPA, 2021b). In 12.5 % of simulations, the IELCR from arsenic exceeded a risk level of one-in-ten-thousand, which the EPA maintains as their upper limit of their target risk range for carcinogenic effects.

3.5. Hazard quotient (HQ)

The mean of total hazard quotient from both arsenic and cadmium (also called the Hazard Index) was 0.377. The hazard quotient exceeded a value of one from arsenic and cadmium exposure only in 7.7 % and 0.08 % of cases, respectively. When combining arsenic and cadmium, the HQ of one was exceeded in 8.9 % of cases. All HQ values, even total HQ, were <3, which the EPA considers a reasonable upper limit of their target risk range for non-carcinogenic effects. (U.S. EPA, 2021b).

3.6. Plant metal(loid) concentration and bioconcentration modeling

3.6.1. Plant concentration and bioconcentration by family

In our exposure assessment survey, participants reported growing plants from 26 plant families, with 11 families making up 90 % of reported plants. Plant family and plant type (as defined by the Codex Alimentarius) for plants grown by all participants who responded to the exposure assessment survey (n = 63) are presented in Supplemental Figs. 5 and 6. Participants submitted plants from 12 families total, with 7 families making up 90 % of submitted samples. Family and plant type for participant submitted plants are presented in Supplemental Figs. 7 and 8.

Models of the concentration and bioconcentration of metal(loid)s observed significant correlations of plant family, metal(loid), and their interaction effect at $\alpha = 0.05$ within the model. Tukey tests on contrasts of metal(loid)-family combinations found a number of significant contrasts at $\alpha = 0.05$ using adjusted p values. Mean plant concentrations and bioconcentration factors by family and associated significance are presented in Figs. 4 and 5. Supplemental Fig. 9 highlights metal(loid) concentrations by the following generic plant categories: fruiting, leafy, and root. Supplemental Tables 7 and 8 give a summary of the concentration and bioconcentration models and individual contrasts are reported in Supplemental Tables 9 and 10.

3.6.2. Comparison of plant results to total diet study

To compare the homegrown plants to what a household could purchase at an average grocery store, we compared our plant metal(loid) concentrations

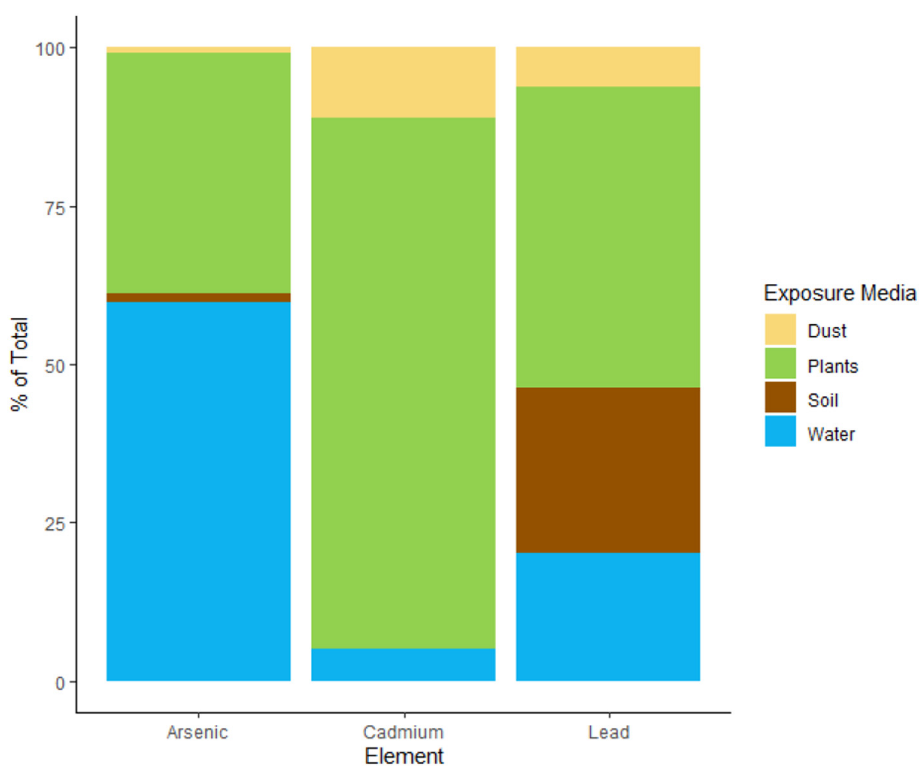

Fig. 2. Relative (percent) of arsenic, cadmium, and lead contribution to average daily dose by measured environmental media in Gold Country California.

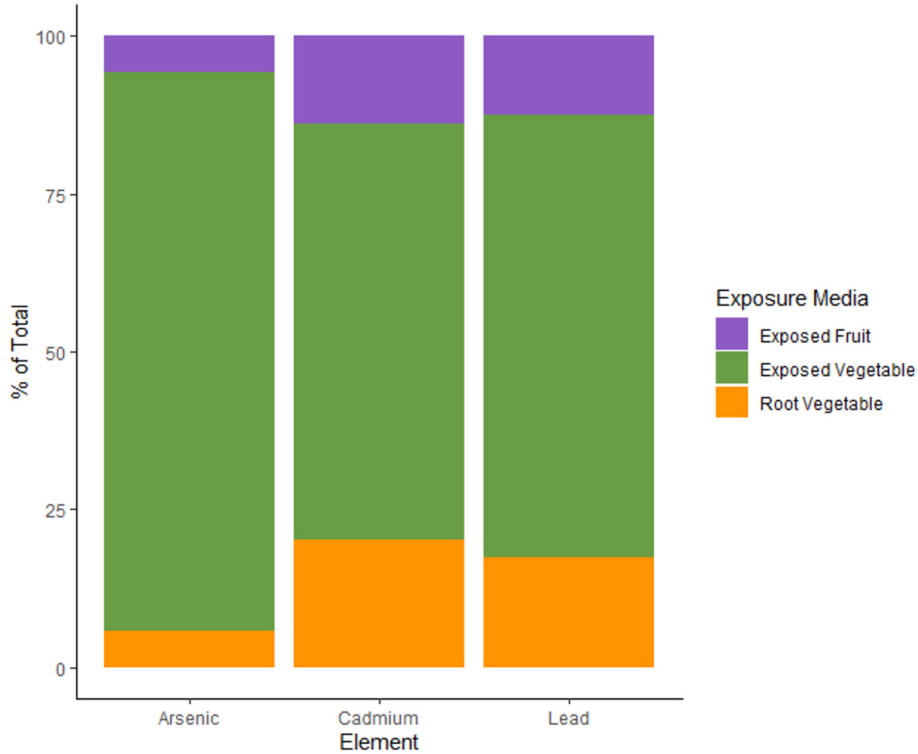

Fig. 3. Relative (percent) of arsenic, cadmium, and lead exposure by homegrown plant category in Gold Country, California.

to the U.S. FDA Total Diet Study. We observed that the average concentration of arsenic, cadmium and lead in the US FDA total diet study data for plant types contained in this study to be 7.15×10^{-3} mg kg^{-1}, 2.52×10^{-2} mg kg^{-1}, and 5.30×10^{-3} mg kg^{-1}. Our samples had average values of 8.35×10^{-3} mg kg^{-1}, 1.55×10^{-2} mg kg^{-1}, and 8.83×10^{-3} mg kg^{-1}. Arsenic and lead levels were therefore above what was observed in the total diet study, while cadmium concentrations were below.

4. Discussion

4.1. Major findings and consistency with previous studies

Using a Markov Chain Monte Carlo-based probabilistic risk assessment method, the objective of this study was to determine whether exposure to arsenic, cadmium, and lead from active and legacy mining in “Gold Country” California is leading to increased community health risks. The study results are relevant to the partnering community and indicate that based on arsenic levels from the measured media, there is an increased excess lifetime cancer risk (IELCR) for females living in Gold Country. Lifetime exposures to arsenic were found to exceed a IELCR value of one-in-a-million, the lower end of the EPA's acceptable risk threshold, in 98 % of cases. IELCR values exceeded one-in-ten-thousand, the upper end of the EPA's acceptable risk range, in 12.5 % of cases. When applied, these risk assessment results indicate the potential need for cleanup at a contaminated site. Although the mean IELCR value of 5.13×10^{-5} – which does not exceed the threshold of 1×10^{-4} that the EPA uses to calculate their regional removal levels – would indicate compliance with EPA risk standards, this was not the case for all participants. The mean ADDs of arsenic and cadmium were 9.82×10^{-5} mg kg^{-1} day^{-1} and 2.47×10^{-5} mg kg^{-1} day^{-1}, respectively. Calculated HQs for arsenic and combined exposures have a notable chance (7.7 % for arsenic, 8.9 % when combined with the cadmium HQ) to exceed the reference dose for non-carcinogenic effects as well, though they do not exceed the upper limit HQ of 3, which the EPA uses to calculate removal management levels, indicating compliance with EPA risk standards.

The ADD of arsenic determined in another legacy mining impacted community ranged from 9.19×10^{-5} mg kg^{-1} day^{-1} to 2.72 10^{-2} mg kg^{-1} day^{-1}, with a mean exposure of 2.39×10^{-3} mg kg^{-1} day^{-1} (Ramirez-Andreotta et al., 2013). This average ADD value is greater than the average determined for this study, 9.81×10^{-5} mg kg^{-1} day^{-1}. A study of arsenic in Columbian groundwater determined a LADD range of 1.0×10^{-4} mg kg^{-1} day^{-1} to 8.0×10^{-4} mg kg^{-1} day^{-1} (González-Martínez et al., 2018). Even the lower LADD estimate from González-Martínez et al. (2018) was greater than the mean LADD of arsenic for Gold Country (3.41×10^{-5} mg kg^{-1} day^{-1}), and did not include plant ingestion, which was found to be a significant contributor to total exposure in this study. While these results indicate that our determined intake of arsenic was somewhat lower than those found for other mining impacted communities, arsenic still presents substantial exposures in excess of EPA's acceptable risk levels.

It is important to note that the U.S. EPA IRIS Cancer Slope factor used to calculate the IELCR is based on linear prediction of the likelihood of developing skin cancer at different LADDs of arsenic. With this understanding, the rate of skin cancer among females (excluding basal and squamous cancers) in Nevada County from 2014 to 2017, where most of our participants reside, was 57.12/100,000 (California Cancer Registry, 2021). The mean of all IELCR values in this study was 5.13E-5, or 5.13 in 100,000. This suggests that the exposures to arsenic identified in this study could contribute around 9 % of skin cancers in the area. However, the EPA IRIS dose response data is meant to predict skin cancer. Unlike skin cancers, the link between arsenic and breast cancer is not as well established and a direct increased risk calculation for breast cancer specifically cannot be conducted.

A previous biomonitoring study in the Gold Country area observed an increased body burden of arsenic in females that ate homegrown produce more frequently than those who reported less frequent or no consumption of home-grown produce (Von Behren et al., 2019). Von Behren et al. (2019) also reports that participants'urinary arsenic concentrations (8.81 μg L^{-1}) were greater than the national average; in contrast, urinary cadmium concentrations (0.21 μg L^{-1} creatinine) were in line with the national average. This is consistent with our finding that residential exposure to arsenic in the female population of Gold Country may be driven by

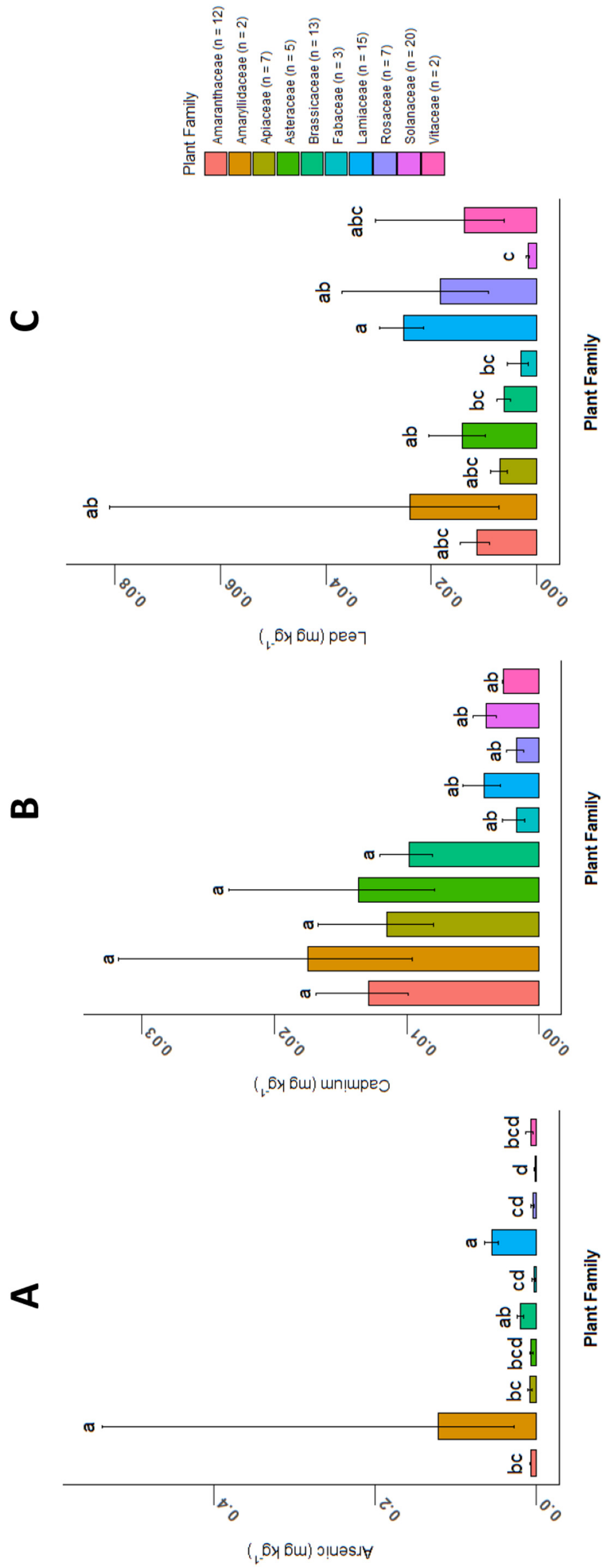

Fig. 4. Mean (A) arsenic, (B) cadmium, and (C) lead concentrations by plant family. Error bars indicate standard deviation. Based on the results of the Tukey test ($\alpha = 0.05$), shared letters indicate no significant difference between families (N = 86).

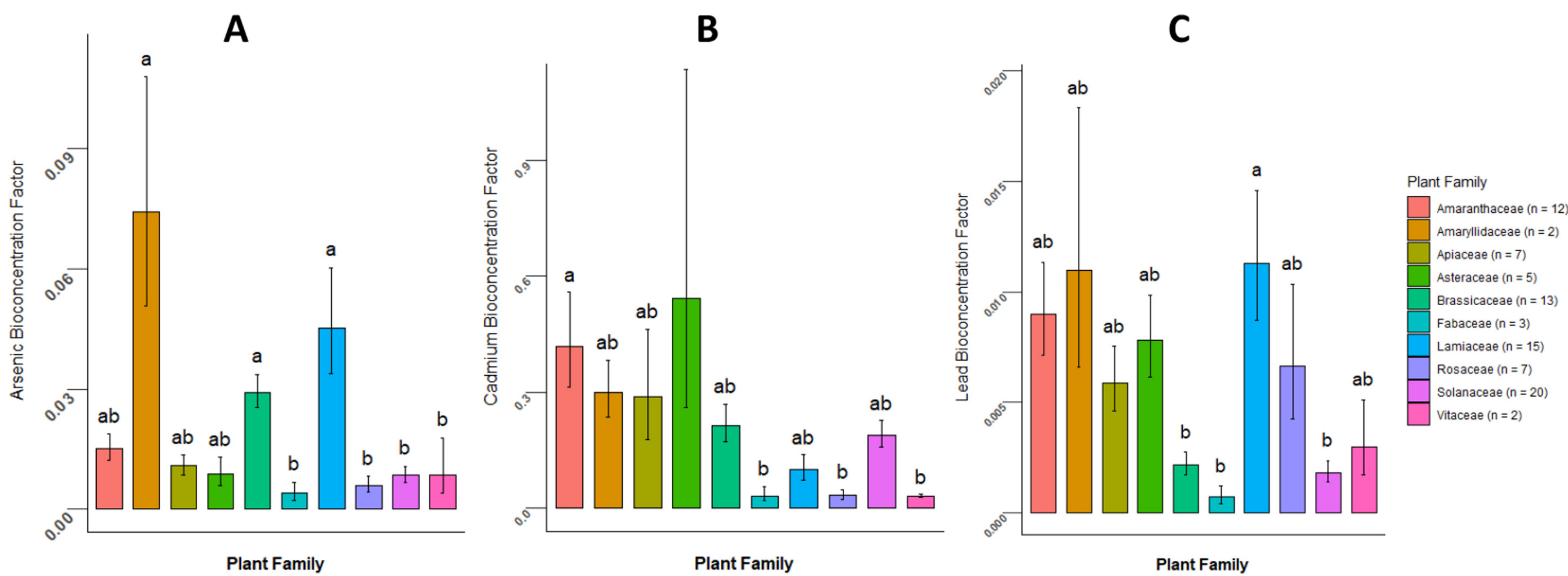

Fig. 5. Mean (A) arsenic, (B) cadmium, and (C) lead bioconcentration factors by plant family. Error bars indicate standard deviation. Based on the results of the Tukey test ($\alpha = 0.05$), shared letters indicate no significant difference between families ($N = 86$).

consumption of home-grown foods and water, while exposure to cadmium is driven by home-grown food intake. This suggests that interventions to reduce exposure via food and water would be more effective than those interventions targeting soil and household dust.

Water and plant consumption are known to be contributors to overall arsenic intake in rural and mining impacted communities, with water intake contributing significantly to total exposure (Díaz et al., 2004; Kurzius-Spencer et al., 2014; Chakraborti et al., 2013). Leafy greens in California's central valley in particular have been identified as a concern for cadmium exposure, which is consistent with our findings (McBride, 2003). Recently, the European Union has moved to implement stricter standards for cadmium in recognition of these concerns (Reuters, 2021). However, exposure to soil with elevated arsenic concentrations (ranging from 2.35 to 374 mg kg^{-1}), as well as to water and plants, has also been found to contribute significantly to arsenic exposure (Ramirez-Andreotta et al., 2013). This is inconsistent with our findings, and indicates variability in sources of these exposures.

Chakraborti et al. (2013) identify legacy gold mines, which are known to be widespread throughout Gold Country area, as a potential source of arsenic contamination of groundwater. However, this supposition is inconsistent with the lack of statistical significance between water source (public vs. private) and arsenic concentrations in our model of arsenic concentration in water samples. In this study 45 % of participants used public water, another 40 % used private water, and the rest had other sources or used a mix of sources. While the two greatest concentrations of arsenic in water were reported to be from private water sources, our linear model, which only used water source as a predictor, failed to indicate significance, most likely due to the small sample size. Further confirmation of water source used in the home, as well as examination of sources of groundwater contamination, is needed to properly examine arsenic exposure through groundwater in the Gold Country area.

Residential arsenic soil standards set by the U.S. EPA and California EPA were exceeded in all samples. However, these standards do not account for background levels in Gold Country. The mean arsenic background level among California soils in previous studies was found to be 3.5 mg kg^{-1} (Kearny Foundation of Soil Science, 1996), which is less than our mean values of 7.4 mg kg^{-1} in garden samples and 11.6 mg kg^{-1} in yard samples. An assessment of native soil in a California coastal area found a mean of 8.2 mg kg^{-1}, which is more similar to garden soil values in this study, but is still exceeded by the mean yard soil concentration (Behrsing et al., n.d.). This indicates at least slightly elevated levels of arsenic in our measured yard soils relative to values typical for California. While our exposure modeling indicated that direct exposure to arsenic from incidental soil ingestion does not contribute notably to arsenic intake, consumption of plants grown in the garden soils were found to contribute in large part to arsenic intake through ingestion, highlighting the need to assess metal(loid) uptake patterns of these plants.

Average concentrations of arsenic and lead in our samples were above the levels of arsenic and lead in comparable vegetable samples from the U.S. FDA's total diet study; however, the average cadmium concentrations were not. Analysis of plant concentration and bioconcentration data in this study identified several plant families that have statistically significant differences in their metal(loid) accumulation patterns. *Amaryllidaceae* (which includes onion and garlic) and *Lamiaceae* (which includes basil and rosemary) have arsenic concentrations significantly greater than a number of other families. *Rosaceae* (which includes apples and strawberries) has significantly lower arsenic concentrations and significantly decreased bioconcentration of cadmium when compared with many other families, and *Solanaceae* (which includes tomatoes, peppers, and potatoes) has significantly lower concentrations of lead than many other families.

Our finding of significant differences in metal(loid) bioaccumulation and bioconcentration between different families of plant is consistent with findings in the field of phytoremediation showing that certain species or families of plant have different abilities to bioaccumulate metal(loid)s (Yan et al., 2018; Roy and McDonald, 2015). Ramirez-Andreotta et al. (2013) and Manjón et al. (2020) report differences in metal(loid) accumulation patterns by plant family among several commonly grown plant families. Their research did not analyze the same plant families that ours did, so we are unable to directly compare results, however there was some overlap with plant families represented in this analysis, and no statistically significant difference found in their research contradicts the findings of our analysis.

This difference between plant families has significant implications for those in the area who garden or otherwise eat food produced locally. Those preferentially eating certain plant families such as *Amarylidaceae* due either to personal taste, cost/availability, or cultural preference, may also have increased arsenic consumption relative to those preferentially eating other families. For this reason, it is likely that both certain individuals, and certain socioeconomic groups face increased risks of cancer based on eating habits.

4.2. Strengths and limitations

Unlike many other site-specific risk assessments, this risk assessment identified and used exposure factors specific to the female population when calculating ADD and LADD. Specifically, this study used values for the female population ages 50–60, the most common age reported in our exposure assessment survey, whenever possible, and used home plant consumption data specific to home gardeners.

Our site-specific risk assessment is made more accurate through the use of these population specific exposure factors relative to risk assessments geared towards the general population. Additionally, allowing participants to collect their own samples from their homes ensures that our sampling methodology collects data that participants consider most relevant to their exposures while using minimally invasive methods.

While population specific exposure variables are used when possible in this study, our analysis is not necessarily devoid of sex and gender-based bias. Many exposure factors such as daily soil consumption used in this assessment were not disaggregated by sexual identity and may not reflect the patterns of the female population as well as those that were available by sex. Additionally, toxicological data from exposure studies, such as data used by the EPA in the development of reference doses and cancer slope factors used in this study, is often based on a largely or wholly male cohort and may be inaccurate when applied to female populations (Betansedi et al., 2018).

An additional limitation of this study is in the generalization of plant consumption to only three categories: exposed vegetables, exposed fruits, and root vegetables. While this generalization is necessary to make use of EPA home produced plant consumption data, our analysis of plant concentration data clearly shows that plant family has a significant effect on the concentrations of a metal(loid) present in home grown foods.

While dust was not found to be a predominant exposure route for the three metal(loid)s analyzed here, it is important to note that dust sampling was carried out subsequent to the 2020 California wildfires, and a small number (<5) of participants noted visibly apparent combustion byproducts such as ash and soot present in their dust sampling areas and in dust samples, especially in outdoor samples.

Based on HUD income limits classifications, income levels observed in our study appear to be slightly higher than the median household income for the area, however, this is based on 2018 classifications for only one county considered in the study and may not accurately frame the participant's income classifications. This may limit the applicability of results towards those with lower median incomes.

Aside from all participants being female, the most notable demographic characteristics of our sample were the skew towards older adults and the predominance of white participants. These skews mean our sample may fail to capture the exposures of younger females, who necessarily face increased harms from long term chronic exposure to the metal(loid)s considered here, as well as non-whites. Nevada County, however, (which 75 % of participants reported as their home county) is 93 % white, which closely matches the demographics of this study (U.S. Census Bureau, 2019).

4.3. Significance for rural health

Rural and urban populations are known to have disparities in several measures of health, including the mortality rate from cancers, self-reported health status, and life expectancy (Hartley, 2004; Matz et al., 2015). These disparities are often widest for older adults such as those included in this study (Cohen et al., 2018). This study did not aim to examine any effects of age or participant location. However, the study area is known to be largely rural, and our results are able to inform, in part, what is known about the specific challenges of rural health.

While the exposure routes identified as driving arsenic consumption in this study are not exclusive to rural areas, research has found place-based differences in the ability of people to grow a home garden, use of home gardening required by food deserts, and the proportion of people who rely on private wells (Morton et al., 2008; Miller et al., 2016; Bain et al., 2014). While our modeling did not find a significant effect of water source on arsenic concentrations, it is worth noting here that the two highest concentrations of arsenic in water by far were submitted by participants reporting the use of a private well for their drinking and cooking water.

Rural health disparities are thought to exist for a diverse set of reasons. Along with environmental contamination which we and previous studies have found evidence of in Gold Country (e.g., Manjón et al., 2020; Von Behren et al., 2019), contributors to rural health disparities include access to health care, cultural challenges, access to healthy foods, and a lack of time or motivation to take proper care of one's health (Beyer et al., 2011).

4.4. Significance for mining impacted communities

Legacy gold mining is known to present both ecological health risks as well as human health risks. Ngole-Jeme and Fantke (2017) found significantly elevated cancer risks from arsenic and nickel in soil located near legacy gold mining sites. Ngole-Jeme & Fantke did not address contamination of groundwater, but such contamination is known to occur (Singh et al., 2018), and may explain the results of this research. Risks from mining impact communities most directly adjacent to active or legacy mining operations but may be dispersed over large areas (Moya et al., 2019; Bird et al., 2010).

Harms from mining are not distributed equally. We found that concentrations of all contaminants were significantly lognormally, or in one case gamma (a distribution similar to a lognormal curve), distributed. Consequently, intake of these contaminants and distributions of risk from these contaminants are lognormally distributed, meaning that a small proportion of the population bears the largest amount of the risk. This information informs public health intervention design: blanket approaches may succeed in addressing the concerns of a majority of community members, but may not appropriately address the concerns of the smaller populations upon which health risks from environmental exposure are concentrated.

While the distribution of risk between those of differing socioeconomic status was not examined in this analysis, the portion of the population bearing the brunt of mining activity related exposures is often found to be of lower socioeconomic status (Moya et al., 2019; Lewis et al., 2017). This occurs due to discriminatory siting of mines, and the exclusion of marginalized communities from decision making regarding mining as well as other activities which may threaten their health (Moeng, 2019; Sicotte, 2008; Moreno Ramírez, 2020). This represents an environmental injustice that is likely to worsen already existing health inequalities among poor and minority communities.

4.5. Future directions

Questions remain regarding the nature of arsenic and cadmium exposure patterns in Gold Country. On its own, this ingestion-based risk assessment is only meant to address whether community members could be at greater risk of cancer based on the observed concentrations of arsenic in their water, soil, dust, and home-grown foods. We are not able to determine the source of these contaminants, or whether they account for the total observed increase in breast cancer rates in quantitative terms.

Site specific dust inhalation data would be useful both to refine this analysis and to address community concerns regarding inhalation. This requires determination of both metal(loid) air concentration, as well as participant activity and inhalation rate patterns. Sampling is needed both in and around participant homes, but also at work sites, recreation sites and in participant vehicles as well. Lastly and as stated in Section 2.3, unfortunately, due to inconsistent reporting by participants, the EPA's EFH data had to be used instead of the FF survey data, which would have provided participant-specific intake rates. Future efforts should budget for more time with participants to ensure survey data is completed successfully to account for participant-specific consumption and practices.

Analysis of plant consumption at the family level rather than at the classification level defined by the EPA EFH and used in this analysis would likely improve both the accuracy and precision of our estimates by accounting for variation in metal(loid) accumulation patterns at a more detailed level. This would require the generation of plant consumption data at this level, which was not implemented successfully here due to the wide variety of plant species and families received. This analysis would also be improved by sampling throughout the year to account for the seasonality of garden crop production and ingestion.

While cancer risk from arsenic was assessed in this research, lack of available toxicological data from EPA IRIS precluded examination of cadmium carcinogenicity, as well as any effects of lead exposure, and the resultant risk estimates. This is despite the well documented carcinogenicity of cadmium and hazardous effects of lead. There is no safe level of lead, especially for children (Vorvolakos et al., 2016), which precludes the development of a reference dose. Cadmium carcinogenicity information in particular would be necessary to fully address community concerns, as research has indicated a link between cadmium and breast cancer (Gallagher et al., 2010; Lin et al., 2016). Additional toxicological information that takes into account cumulative effects and the interaction between arsenic, cadmium, and lead would be useful for refining estimates of carcinogenic effects, and for producing estimates of cumulative non-carcinogenic effects beyond the total hazard quotient. Based on previous efforts in the area, this study focused on arsenic, cadmium, and lead; however future efforts should consider expanding and including other metal(loid)s that are associated with legacy and active mining. As highlighted in Section 4.4, harms from mining are not distributed equally and socioeconomic variables need to also be considered.

5. Conclusion

This CBPR study worked with a community-based organization to collect soil, water, home-grown food, and dust samples from 40 households. This study is one part of the larger CHIME project, which seeks to address community concerns regarding the rate of breast cancer in the Gold Country area relative to California's average (Von Behren et al., 2019; Manjón et al., 2020). The project ultimately aims to answer whether legacy mining may be contributing to increased levels of human exposure to carcinogenic metal(loid)s such as arsenic and cadmium, and whether these increased levels could be contributing to increased breast cancer rates. This research indicates that rural areas in Gold Country face environmental exposures via home-grown foods and water consumption that are likely different than in urban areas. This information can be used to tailor public health interventions designed to reduce these exposures. While modeling did not find a significant effect of water source on arsenic concentrations, the two highest concentrations of arsenic in water by far were submitted by participants reporting the use of a private well for their drinking and cooking water. The results of this study are consistent with previous efforts (Von Behren et al., 2019; Manjón et al., 2020) indicating that residential exposure to arsenic in the female population of Gold Country may be driven by consumption of home-grown foods and water, and exposure to cadmium is driven by home-grown food intake, suggesting that interventions to reduce exposure via food and water could be more effective than those interventions targeting soil and household dust. The weight of the evidence, both from this study and from what is known about the carcinogenicity of arsenic generally, suggests that the arsenic exposure identified in this study could contribute to increases in the cancer rate among those living in Gold Country, California.

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CRediT authorship contribution statement

Diego Huerta: Investigation, Formal analysis, Validation, Data curation, Visualization, Writing – original draft, Writing – review & editing. **Taylor Schobel:** Investigation, Resources, Writing – review & editing, Project administration. **Annika Alexander-Ozinskas:** Investigation, Resources, Writing – review & editing, Project administration. **Joanne Hild:** Writing – review & editing, Funding acquisition. **Jeff Lauder:** Writing – review & editing. **Peggy Reynolds:** Writing – review & editing, Project administration, Funding acquisition. **Julie Von Behren:** Writing – review & editing, Project administration, Funding acquisition. **Dan Meltzer:** Visualization, Writing – review & editing. **Mónica D. Ramírez-Andreotta:** Conceptualization, Methodology, Investigation, Resources, Visualization, Writing – review & editing, Supervision, Project administration, Funding acquisition.

Data availability

Data will be made available on request.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

We respectfully acknowledge that the University of Arizona is on the land and territories of Indigenous peoples. Today, Arizona is home to 22 federally recognized tribes, with Tucson being home to the O'odham and the Yaqui. At the University of California San Francisco (UCSF), we would also like to acknowledge the Ramaytush Ohlone people, who are the traditional custodians of this land. We pay our respects to the Ramaytush Ohlone elders, past, present, and future who call this place, the land that UCSF sits upon, their home. We are proud to continue their tradition of coming together and growing as a community. We thank the Ramaytush Ohlone community for their stewardship and support, and we look forward to strengthening our ties as we continue our relationship of mutual respect and understanding. Sierra Streams Institute is located in Nevada City, California, which is the ancestral homeland of the Nevada City Rancheria Nisenan Tribe. The Nisenan are still here and are fighting for federal recognition.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.162228.

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