INTERNATIONAL PERSPECTIVES

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Measurements of Arsenic in the Urine and Nails of Individuals Exposed to Low Concentrations of Arsenic in Drinking Water From Private Wells in a Rural Region of Québec, Canada

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Abstract **Chronic exposure to inorganic arsenic leads to an increased risk of cancer. A biological measurement was conducted in 153 private well owners and their families consuming water contaminated by inorganic arsenic at concentrations that straddle 10 µg/L. The relationship between the external dose indicators (concentration of inorganic arsenic in wells and daily well water inorganic arsenic intake) and the internal doses (urinary arsenic—sum of As^{III}, DMA, and MMA, adjusted for creatinine—and total arsenic in toenails) was evaluated using multiple linear regressions, controlling for age, gender, dietary sources of arsenic, and number of cigarettes smoked. It showed that urinary arsenic was associated with concentration of inorganic arsenic in wells ($p < .001$) and daily well water inorganic arsenic intake ($p < .001$) in adults, and with daily well water inorganic arsenic intake ($p = .017$) and rice consumption ($p = .022$) in children ($n = 43$). The authors' study reinforces the drinking-water quality guidelines for inorganic arsenic.**

Introduction

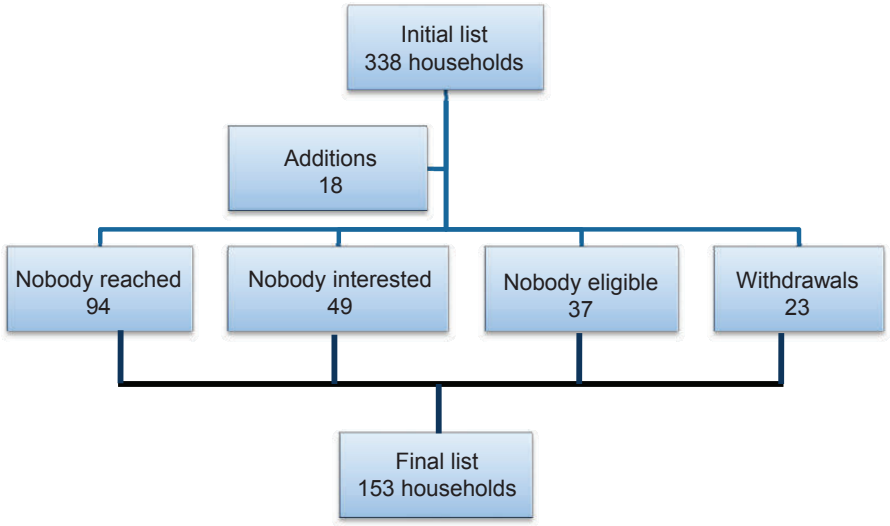
Exposure to arsenic primarily occurs via food, drinking water, soil, and air (Kendall, Bens, & Cobb, 2003). Cigarette smoking is also a source of inorganic arsenic (Cui, Kobayashi, Akashi, & Okayasu, 2008). In the case of populations dwelling in the vicinity of an important geological source of inorganic arsenic, drinking water could represent the most significant source of inorganic arsenic exposure (Health Canada, 2006a).

Chronic exposure to inorganic arsenic in drinking water can lead to an increased risk of cutaneous, pulmonary, bladder, and hepatic cancer (Ferreccio et al., 2000; Morales, Ryan, Kuo, Wu, & Chen, 2000; Rossman, Uddin, & Burns, 2004; Steinmaus, Yuan, Bates, & Smith, 2003). To limit such cancer risks, a guideline or a standard of 10 µg/L for public drinking water supplies has been adopted by different organizations (Health Canada, 2006b; U.S. Environmental Protection Agency, 2001; World Health Organization, 2003). This level is not enforceable, however, for private wells in the province of Québec, Canada (Gouvernement du Québec, 2008).

Levels of arsenic and its metabolites in urine and nails have been used as biomarkers of short-term (Agency for Toxic Substances and Disease Registry, 2007) and long-term arsenic exposure (Orloff, Mistry, & Metcalf, 2009), respectively. Various studies have demonstrated a correlation between these biomarkers and the inorganic arsenic exposure via contaminated water (Agusa et al., 2009; Caceres et al., 2005; Gault et al., 2008; Hinwood et al., 2003; Kendall et al., 2003; Mandal, Ogra, & Suzuki, 2003; Meza, Kopplin, Burgess, & Gandolfi, 2004; Sun et al., 2007; Uchino, Roychowdhury, Ando, & Tokunaga, 2006). Almost all of these correlations were established, however, using median well water inorganic arsenic levels exceeding 50 µg/L. Few studies have established a clear relationship between lower levels and the consumer's internal dose after controlling dietary sources; solid food is a major contributor to urine arsenic levels (Clayton, Pellizzari, Whitmore, Perritt, & Quackenboss, 1999).

FIGURE **1**

Flow Chart of Household Selection



The objectives of our project were as follows:

1. Conduct a biological measurement study of inorganic arsenic exposure in a population living in Abitibi-Témiscamingue, Québec, a rural area where groundwater is affected by a natural geological source of arsenic.
2. Evaluate variations in the internal doses of inorganic arsenic in terms of the various well contamination levels, the external dose, and other potential sources of arsenic (primarily diet).

Materials and Methods

Study Population

Our study concerns a population of private well owners and their families living in the Abitibi-Témiscamingue region of Québec, Canada.

Recruitment of Subjects

On the basis of a previous screening campaign three groups of private wells were established: group 1 with inorganic arsenic levels <10 µg/L, group 2 with inorganic arsenic levels 10–20 µg/L, and group 3 with inorganic arsenic levels ≥20 µg/L (Poissant, 1997). Of the 400 households initially available for the first group, 150 were randomly selected. For the second group, all of the 67 potential households were contacted. The 121 potential households from the third group and 18 new households were all included.

To be eligible, interested individuals had to live in a home supplied by a private well, had to drink or use this water for preparing beverages and food for at least one month, and had to be seven years of age or older. Those who had an arsenic treatment system, who were occupationally exposed, or who consumed homeopathic medications or herbal dietary supplements were excluded. Pregnancy, kidney or liver diseases, and active cancer were also considered as exclusion criteria. Participants were recruited via telephone.

Variables and Data Collection

During a first home visit, a water sample of 125 mL was collected from the most frequently used tap in an opaque amber bottle containing 1.25 mL of ethylenediaminetetraacetic acid (EDTA) 0.25M. The samples were then kept at 4°C until analysis.

Two questionnaires regarding participant lifestyle and dietary habits (Bouchard, Normandin, Levallois, & Ayotte, 2007a; Bouchard et al., 2007b) were distributed to each participant.

In the lifestyle questionnaire, participants were asked to estimate their usual daily well water consumption in the past year, including all types of beverages prepared with well water (i.e., number of servings equivalent to 250 mL). Age, gender, and smoking habits were then recorded.

The diet questionnaire was self administered during the two days prior to biological sampling. This questionnaire was designed to quantify the portions of inorganic arsenic–contaminated (rice, breakfast cereals, and pasta; based on the equivalent portion presented for each), and organic arsenic–contaminated foods (chicken, fish, seaweed, shellfish, and mushrooms) (Hughes, 2006). The participants were also asked to record their well water consumption (i.e., number of servings), including water alone or beverages prepared with well water.

These two questionnaires, the first morning void urine, and nail clippings from the big toes were collected from the participant during a second visit. Urine samples were then frozen (-20°C) and toenails refrigerated (4°C) until analysis.

Data Analysis

Water samples were analyzed with a high performance liquid chromatography with inductively coupled argon plasma mass spectroscopy (Centre d'Expertise en Analyse Environnementale du Québec, 2008; Garbarino, Bednar, & Burkhardt, 2002). All chemicals were American Chemical Society–certified quality. The determination of trivalent arsenic (As^{III}) and pentavalent arsenic (As^{V}) was done to look for varying toxicological properties of these forms in further analysis. This speciation raised the limit of detection (LD) to 5 µg/L for each chemical forms, an effect that was controlled by a sensitivity analysis (see Discussion).

As for the biological samples, the two forms of urinary inorganic arsenic (As^{III} and As^{V}) and their metabolites (MMA and DMA), as well as arsenocholine and arsenobetaine were all measured (Calderon, Hudgens, Schreinemachers, & Thomas, 1999). Total arsenic concentrations were measured in the toenail samples. Gas chromatography extraction and inductively coupled plasma-mass spectrometry (ICP-MS) identification procedures

(Belanger & Dumas, 2010) were used for the analysis of the biological samples. The LDs for urinary arsenic and toenail arsenic were 0.7 µg/L and 0.1 µg/g, respectively. Urinary creatinine was measured using colorimetry (Jaffé, 1886).

An imputed value of LD/√(2) was used for water or biological samples with levels below the LD. The concentration of inorganic arsenic in wells was determined by adding the As^{III} and As^{V} concentrations. The internal doses were estimated by urinary and ungual arsenic concentrations. Urinary arsenic concentration was calculated by adding As^{III}, DMA, and MMA, adjusted for creatinine. External doses were estimated by the concentration of inorganic arsenic in wells and the short- and long-term inorganic arsenic daily intakes from well water that were calculated by multiplying inorganic arsenic in wells (µg/L) with the mean daily well water consumption (L) in the two days prior to sampling, in the first case, and the usual daily well water consumption (L) on an annual basis, in the second case.

Descriptive statistics, including Chi squared distribution analysis and Fisher's test, were used to compare sociodemographic characteristics in the three groups. Analysis of variance was performed to compare the geometric mean internal dose among the three exposure groups. *T*-tests were used to compare variation of these levels depending on water uses (i.e., drinking or using this water for preparing beverages vs. using this water only for preparing food) in the two days prior to sampling or the average daily well water consumption in the past year.

The relationship between the external dose indicators and the internal doses was evaluated using multiple linear regressions, controlling for potential confounding variables (age, gender, dietary sources of arsenic, number of cigarettes smoked). The short-term daily well water inorganic arsenic intake was used as an independent variable in models with urinary arsenic and the long-term daily well water inorganic arsenic intake in models with toenail arsenic. The multiple linear regression analyses performed with urinary arsenic were also repeated excluding those participants with significant organic arsenic exposure, as estimated by the presence of detectable arsenocholine and arsenobetaine in their urine.

TABLE **1**

Sociodemographic Characteristics of Participants (Adults and Children) by Exposure Group

Groups[a]	Adults (*n* = 261)			Children (*n* = 43)		
	Group 1	Group 2	Group 3	Group 1	Group 2	Group 3
n	126	55	80	20	12	11
Age groups (%)						
7–12 years	–	–	–	65	33	36
13–17 years	–	–	–	35	67	64
18–44 years	31	33	31	–	–	–
45–64 years	55	6	53	–	–	–
65 and over	14	7	16	–	–	–
p-Value[b]	.649			.140		
Gender (%)						
Male	42	53	41	50	42	27
Female	58	47	59	50	58	73
p-Value	.344[b]			.486[c]		
Last year of education completed (%)						
Primary	14	11	20	75	33	45
Secondary	47	49	52	25	67	55
Junior college or vocational	18	25	23	–	–	–
University	21	15	5	–	–	–
Data missing	1	–	1	–	–	–
p-Value[b]	.072			.049		
Active smoker (%)						
Yes (*n*)	21 (26)	16 (9)	23 (18)	0 (0)	0 (0)	18 (2)
No (*n*)	79 (100)	84 (46)	78 (62)	100 (20)	100 (12)	82 (9)
p-Value[b]	.679			.061		

[a]Group 1: <10 µg arsenic/L water; group 2: 10–20 µg arsenic/L water; group 3: ≥20 µg arsenic/L water.
[b]Determined using Pearson Chi-square test.
[c]Determined using Fisher's test.

This strategy has been proposed as the best way to control the influence of organic arsenicals of marine origin (arsenosugars and arsenolipids) on urine DMA (Navas-Acien, Silbergeld, Pastor-Barriuso, & Guallar, 2009). The logarithmic conversion of variables was carried out as required in the regression analysis. Results were considered statistically significant when $p < .05$. Data were analyzed using SPSS version 17.0.

Ethical Considerations

Our project was approved by Health Canada's research ethics board and by the human research ethics board at the Centre Hospitalier Universitaire de Sherbrooke.

Results

Of the 356 households initially available, 153 were included in our study (see flow chart in Figure 1). In terms of participants, 489 individuals were available for the study in the households contacted, but 71 were uninterested, 69 were ineligible, and 54 withdrew because they changed their mind during the interval between the phone call and the home visit. In the end, 304 participants were recruited, including 43 children.

TABLE **2**

Distribution of Urinary and Toenail Arsenic (As) Among Participants

Distribution	Urinary As[a] (µg/L)		Urinary Arsenocholine + arsenobetaine (µg/L)		Toenail As (µg/g)	
	Adults *n* = 261	Children *n* = 43	Adults *n* = 261	Children *n* = 43	Adults *n* = 261	Children *n* = 43
5th percentile	2.49	3.23	0.50	0.50	0.07	0.14
95th percentile	34.23	22.70	81.6	18.60	0.89	1.37
Arithmetic mean	12.66	8.93	26.55	3.33	0.28	0.48
Standard deviation	26.23	5.64	281.39	7.92	0.30	0.37
Geometric mean	8.06	7.53	1.89	0.97	0.20	0.37
Median	7.70	7.39	1.00	0.50	0.18	0.30
Missing data (*n*)	1	0	1	0	5	1
Data below limit of detection (*n* [%][b])	–	–	113 (43.5%)	30 (69.8%)	55 (21.2%)	1 (2.4%)

[a]Urinary As = Σ (MMA + DMA + As^{III}).
[b]Percentage indicated: number of values below the limit of detection/total number of samples analyzed.

The three exposure groups were similar in terms of age group, male-to-female ratio, and smoking status (Table 1). The average length of residence of participants at their current address was 17.2 years (range: 1–24).

External Doses

A difference existed between the usual daily well water consumption (L/day) on an annual basis among the three adult groups (group 1: 1.24; group 2: 1.41; group 3: 0.67; $p < .001$). A difference also occurred between the mean daily well water consumption in the two days prior to sampling among the three adult groups and the three children groups.

The concentration of inorganic arsenic in wells had a log-normal distribution with an arithmetic mean of 22.77 µg/L ($n = 153$), a geometric mean of 14.18 µg/L, and a median of 10.54 µg/L. Data were below the LD in 70 wells (46% of samples) for both As^{III} and As^{V}, and the highest concentration was 193.54 µg/L.

Internal Doses

Urinary arsenic and toenail arsenic distributions are described in Table 2. Practically all the values in the form of DMA were above the LD in adults (98.5%) and children (97.7%). By contrast, practically all the values for As^{V} were below the LD in adults (98.1%) and children (100%).

In total, 21.1% of adults and 2.4% of children had toenail arsenic concentrations below the LD.

Relationships Between Internal and External Doses

Bivariate Analysis

A statistically significant difference between the geometric mean urinary arsenic (µg/g creatinine) among the three exposure groups was observed in adults (group 1: 5.83; group 2: 8.93; group 3: 11.67; $p < .001$) but not in children (group 1: 6.12; group 2: 6.52; Group 3: 7.59; $p = .585$). A statistically significant difference was observed in urinary arsenic between adults who drank well water in the two days prior to sampling and those who only use this water for preparing food in groups 2 and 3 ($p = .042$ and $p < .001$, respectively).

With regard to toenail arsenic (µg/g), a statistically significant difference occurred among the three groups for both adults (group 1: 0.130; group 2: 0.210; group 3: 0.358; $p < .001$) and children (group 1: 0.305; group 2: 0.308; group 3: 0.656; $p = .034$, respectively). For adults, only group 3 showed toenail arsenic concentrations that were significantly higher for those who normally drink well water compared with those who did not ($p = .013$).

The correlation between urinary arsenic and toenail concentration in adults was 0.34 ($p < .001$).

Multivariate Analyses

An analysis of residuals of regression demonstrated diverging observations that were apparently due to lobster consumption in the two days prior to sampling. Each serving of lobster increased urinary arsenic by 133.7 µg/L on average (see coefficient; model A; Table 3). Consumption of one serving of crab in the two days prior to sampling also increased urinary arsenic concentration by 25.7 µg/L ($p = .022$) (data not shown).

A second regression model was generated by excluding the seven participants (2.7% of the total sample) who consumed lobster or crab in the two days prior to sampling (model B). As in the case of the first model, the short-term daily well water inorganic arsenic intake was the only independent variable having a significant impact on the urinary arsenic concentration. At this level, however, it was the effect of fish that was more obvious, with one serving of fish increasing urinary arsenic by 7.6 µg/L ($p = .001$).

A final model was developed by withdrawing all 146 participants (56% of total sample) who had detectable arsenocholine or arsenobetaine in their urine. The short-term daily intake remained related to urinary arsenic ($p < .001$; model C). The tight control over dietary sources of arsenic in this model allowed us to isolate the direct influence of inorganic arsenic in wells on urinary arsenic ($p < .001$; model C). Women had about 4 µg/L more urinary arsenic than men on average ($p < .001$).

In children, the basic regression model accounted for 35.9% of the total variation of urinary arsenic ($p = .002$) in part owing to the contribution of the short-term daily well water inorganic arsenic intake ($p = .017$; model D, Table 3) and food containing arsenic; one serving of rice increased urinary arsenic by 8.5 µg/L on average. Each additional year of age decreased urinary arsenic by a little less than 1 µg/L.

With regard to toenail arsenic, the basic regression model accounted for 45.0% of the total toenail arsenic variation in adults after logarithmic transformation of toenail arsenic concentrations ($p < .0001$; model E, Table 4).

After excluding 12 extreme values, a restricted model accounted for 61.9% of the

total toenail arsenic variation ($p < .001$; model F). As in the former model, the concentration of inorganic arsenic in wells and long-term daily intake were still statistically significant for explaining variability in the total toenail arsenic concentration ($p < .001$ and $p = .002$, respectively). Total toenail arsenic decreased significantly with age ($p < .001$).

In children, the regression model using logarithmically transformed variables accounted for 29.1% of the total toenail arsenic variation ($p = .004$; model G; Table 4). Inorganic arsenic in wells was the only independent variable statistically significant for explaining this variability ($p = .001$). The older the child, the lower the toenail arsenic concentration ($p = .042$).

TABLE **3**

Multiple Linear Regression Models Between Urinary Arsenic[a] and External Dose Indicators[b] by Age Group

Variables	Nonstandardized Coefficient	*p*-Value
Model A in adults: $R^2 = .729$, $p < .0001$, $n = 260$		
Short-term daily intake (µg/d)	0.300	<.001
Gender (men = 0; women = 1)	4.097	.001
Age	0.067	.135
Mean lobster consumption in the two days prior to sampling (servings/day)	133.687	<.001
Model B in adults: $R^2 = .567$, $p < .0001$, $n = 253$		
Short-term daily intake (µg/d)	0.301	<.001
Gender (men = 0; women = 1)	4.153	<.001
Age	0.007	.837
Mean fish consumption in the two days prior to sampling (servings/day)	7.642	.001
Model C[c] in adults: $R^2 = .855$, $p < .0001$, $n = 114$		
Short-term daily intake (µg/d)	0.379	<.001
Arsenic concentration in well (µg/L)	0.087	<.001
Gender (men = 0; women = 1)	3.550	.001
Age	-0.029	.423
Model D in children: $R^2 = .359$, $p = .002$, $n = 43$		
Short-term daily intake (µg/d)	0.294	.017
Gender (boys = 0; girls = 1)	2.749	.056
Age	-0.853	.002
Mean rice consumption in the two days prior to sampling (servings/day)	8.504	.022

[a]Σ (MMA + DMA + AsIII) corrected for urinary creatinine (µg/g Cr*L).
[b]Inorganic arsenic concentration in well water or short-term daily well water inorganic arsenic intake.
[c]After withdrawing all 146 participants (56% of total sample) having detectable arsenocholine or arsenobetaine in their urine.

Discussion

Compared with median exposures that were measured in the 2003–2004 National Health and Nutrition Examination Survey (NHANES) study in the U.S. (Navas-Acien, Silbergeld, Pastor-Barriuso, & Guallar, 2008), the participants in our study were more exposed to inorganic arsenic than the general U.S. population based on the results obtained in adults for comparable forms (for DMA, 5.70 µg/L vs. 3.0 µg/L, respectively). By contrast, our participants were less exposed to arsenic than Native American adults living in communities of Arizona served by public water systems with inorganic arsenic levels from less than 10 to 61 µg/L (median of urinary total arsenic concentrations: 7.70 µg/L vs. 18.6 µg/L, respectively; Gribble et al., 2012). For adults in our study, a statistically significant difference existed in the mean urinary arsenic concentration across the three groups of wells. This difference was also seen in long-term exposure, as measured in toenail concentrations in both adult and child participants.

For the multiple linear regressions, it is noteworthy that lobster, crab, and fish influenced urinary arsenic in adults. Excluding participants with detectable urinary arsenocholine or arsenobetaine increased the strength of the association between urinary arsenic and short-term daily well water inorganic arsenic intake while highlighting the direct effect of the concentration of inorganic arsenic in wells. These two independent variables mutually accounted for 85.5% of the total variation in the model ($p < .001$) once combined with age and gender (model C). While it could be suggested that the independent influence of gender in the regression models could be caused by gender differences in arsenic metabolism, in fact, it cannot be an explanation here considering that urinary arsenic is the sum of inorganic arsenic and methylated arsenic species in urine. Smoking did not contribute to measured internal doses as seen in other studies (Karagas et al., 2000, Marano et al., 2012).

In adults, the model for toenail arsenic shows the concentration of inorganic arsenic in wells and the long-term daily well water inorganic arsenic intake as independent variables ($R^2 = .619$; model F). This high R^2 is consistent with studies suggesting that toenail arsenic is a useful marker for low concentrations of inorganic arsenic (Karagas et al., 2000; Slotnick & Nriagu, 2006, Slotnick, Meliker, AvRuskin, Ghosh, & Nriagu, 2007). The good relationship between toenail arsenic and inorganic arsenic in wells is also consistent with a study reporting that toenail arsenic levels are primarily inorganic arsenic (Mandal, Ogra, & Suzuki, 2001). Although no speciation occurred in toenail arsenic, it is reasonable to expect that the excluded participants' exposure was in fact dietary: the inorganic arsenic in their well water was below the LD; and even if fish, seafood, or seaweed are especially known for their high arsenic content in organic forms, they can nevertheless contain an appreciable amount of inorganic arsenic (up to 10% in the case of mollusks; Food and Drug Admin-

TABLE **4**

Multiple Linear Regression Models Between Toenail Arsenic[a] and External Dose Indicators[b] by Age Group

Variables	Nonstandardized Coefficient	*p*-Value
Model E in adults: R^2 = .450, p < .0001, n = 256		
Arsenic concentration in well (log [µg/L])	0.577	<.001
Long-term daily intake (log [µg/d])	0.071	.014
Gender (men = 0; women = 1)	0.030	.372
Age	-0.006	<.001
Model F[c] in adults: R^2 = .619, p < .0001, n = 244		
Arsenic concentration in well (log [µg/L])	0.636	<.001
Long-term daily intake (log [µg/d])	0.069	.002
Gender (men = 0; women = 1)	0.014	.602
Age	-0.006	<.001
Model G in children: R^2 = .291, p = .004, n = 42		
Arsenic concentration in well (log [µg/L])	0.435	.001
Gender (boys = 0; girls = 1)	-0.058	.511
Age	-0.033	.042

[a](log [µg/g]).
[b]Inorganic arsenic concentration in well water or long-term daily well water inorganic arsenic intake.
[c]After withdrawing 12 participants (4.7% of total sample) with extreme values.

istration, 1993; Gagnon, Tremblay, Rouette, & Cartier, 2004).

The children in our study seem to be less exposed than children aged 6–12 years old from agricultural areas in Mexico (sum of urinary As^{III}, As^{V}, MMA, and DMA: arithmetic mean of 8.93 µg/L vs. 30.9 µg/L, respectively; Meza-Montenegro et al., 2013). In the Mexican children, food consumption or dust inhalation may be more important routes of arsenic exposure than drinking water (Roberge et al., 2012). For children in the present study, the total variation in urinary arsenic and toenail arsenic determined through modeling was lower than in adults (R^2 = .359 and p = .002 for model D in Table 3; R^2 = .291 for model G in Table 4). A better methylation capacity in children may explain this for toenails (Sun et al., 2007). It is also difficult for them to estimate their water consumption. A larger proportion of water intake by children may also come from outside their home (school, neighborhood).

With regard to the child participants, it was interesting to observe that age was inversely related to internal dose as measured in urine or toenails. This could be explained by greater inhalation of inorganic arsenic contaminated dust by younger children; the soil of the region is naturally rich in inorganic arsenic. With regard to urine, the inverse relationship could also be caused by correction measurements, since urinary creatinine is higher in adolescents than in children owing to their greater muscle mass. In adults, the inverse relationship observed between age and toenail internal dose may well be due to lower consumption of fish and seafood by older people.

Generally, internal arsenic doses were associated with indicators of external doses whose inorganic arsenic concentrations were measured in the participants' wells even at concentrations that straddle 10 µg/L. In group 1, no significant difference in urinary arsenic concentration was observed in people who drank well water in the two days prior to sampling and those who only used this water for preparing food. Overall, we could consider as nonclinically significant the difference between the mean concentration of urinary arsenic in participants of the first group of wells and a comparable group of citizens (n = 328) in another sector of the same area served by a noncontaminated water supply system (6.9 µg/L vs. 4.6 µg/L; Gagné, 2007). This suggests that for users of wells with arsenic concentrations less than 10 µg/L, consuming well water represented a negligible contribution in comparison with all other sources of exposure to inorganic arsenic, and it is not an argument for an even more restrictive guideline. Moreover, a recent study has shown that mean aggregate inorganic arsenic intake among subjects living in homes with tap water arsenic ≤ 10 µg/L, 5 µg/L, and 3 µg/L was similar, with >54% of this exposure from food (Kurzius-Spencer et al., 2014).

The strengths of our study are the random selection of households in group 1 and the exhaustive recruitment of households in groups 2 and 3, the high participation rate (74.9%, or 373 of the 489 potential participants before exclusion of ineligible individuals), and the fact that our study considered not only drinking water but also water used to prepare food and beverages. The 71 uninterested individuals did not differ from participants in terms of gender, age group, education, and historical inorganic arsenic level in their well (p > .05).

The percentage of values below the LD (particularly for water samples) had a very small effect on the results. First, the comparisons between the geometric mean of internal doses among the three exposure groups were not influenced by the LD. Second, for the regression models, a sensitivity analysis was done after excluding participants with well water results below the LD; for all these additional analyses, the coefficients were similar and the results remained statistically significant (data not shown). Third, the relative weight of wells with arsenic below the LD for the exposure charge in the models is relatively modest (i.e., 13% after applying the contamination level of LD/√(2) to the 70 wells concerned, the arithmetic mean of 23 µg/L to the other wells, and assuming a uniform distribution of participants among the 153 households.

Conclusion

The results of our study confirm that concentrations of inorganic arsenic in wells contribute significantly to human exposure to inorganic arsenic and hence reinforce the recommendation for drinking water issued by the U.S. Environmental Protection Agency, the World Health Organization, and Health Canada regarding this contaminant.

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