

Site-specific Data Confirm Arsenic Exposure Predicted by the U.S. Environmental Protection Agency

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The EPA uses an exposure assessment model to estimate daily intake to chemicals of potential concern. At the Anaconda Superfund site in Montana, the EPA exposure assessment model was used to predict total and speciated urinary arsenic concentrations. Predicted concentrations were then compared to concentrations measured in children living near the site. When site-specific information on concentrations of arsenic in soil, interior dust, and diet, site-specific ingestion rates, and arsenic absorption rates were used, measured and predicted urinary arsenic concentrations were in reasonable agreement. The central tendency exposure assessment model successfully described the measured urinary arsenic concentration for the majority of children at the site. The reasonable maximum exposure assessment model successfully identified the uppermost exposed population. While the agreement between measured and predicted urinary arsenic is good, it is not exact. The variables that were identified which influenced agreement included soil and dust sample collection methodology, daily urinary volume, soil ingestion rate, and the ability to define the exposure unit. The concentration of arsenic in food affected agreement between measured and predicted total urinary arsenic, but was not considered when comparing measured and predicted speciated urinary arsenic. Speciated urinary arsenic is the recommended biomarker for recent inorganic arsenic exposure. By using site-specific data in the exposure assessment model, predicted risks from exposure to arsenic were less than predicted risks would have been if the EPA's default values had been used in the exposure assessment model. This difference resulted in reduced magnitude and cost of remediation while still protecting human health. *Key words:* arsenic bioavailability, arsenic biomarkers, exposure assessment, soil and dust ingestion, urinary arsenic concentrations. *Environ Health Perspect* 106:133–139 (1998). [Online 3 February 1998] <http://ehpnet1.niehs.nih.gov/docs/1998/106p133-139walker/abstract.html>

The mandate of the Superfund program established under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 as amended in 1986 is to protect human health and the environment from current and potential threats posed by uncontrolled hazardous substance releases. To that end, the EPA developed a framework for evaluating and documenting public health threats at Superfund sites (1). An integral step of this framework in evaluating and documenting public health threats is an exposure assessment. The exposure assessment is conducted to estimate the magnitude of actual or potential human exposure (1). The model the EPA uses to estimate potential human exposure is presented in *Risk Assessment Guidance for Superfund* (1). In 1992, the EPA's Risk Assessment Council advocated that all risk assessments

address or provide descriptions of 1) individual risk to include the central tendency and high end portions of the risk distribution, 2) important subgroups of the population such as highly exposed or highly susceptible groups or individuals, if known, and 3) population risk (2).

To address these population descriptions, risk assessors develop a central tendency exposure (CTE) estimate and a reasonable maximum exposure (RME) estimate for populations potentially at risk.

The methodology used by the EPA to estimate risk has been criticized as being too conservative, i.e., the methodology overpredicts potential risks associated with chemicals present in environmental media at Superfund sites. The criticism is usually directed on the development of toxicity values, particularly the development of the cancer slope factors or the definition of acceptable risk (3–5); however, the assumptions used in the exposure assessment model have also been criticized as too conservative (6).

While the EPA supports the use of site-specific data in the exposure assessment model, scientifically sound site-specific data are rarely available to actually compare the assessment model results to measured biomarkers of exposure. However, at the Anaconda National Priorities List Site in Montana, the Atlantic Richfield Company (ARCO), the owner of the site, collected data that have contributed significantly to the characterization of human exposure. These data provide a unique opportunity to compare measured exposure data to modeled exposure predictions.

The Anaconda site is located in southwestern Montana at the southern end of Deer Lodge Valley. The population of the town is approximately 12,000. Copper concentrating and smelting operations

began in the area around 1884 and continued until 1980. Smelting activities resulted in aerial deposition of material released from stacks and from waste piles in the vicinity of the smelter. Arsenic, lead, cadmium, copper, and zinc were released as smokestack particulates and fugitive dust emissions, contaminating the surrounding soil.

At the Anaconda, Montana, Superfund site, multiple studies have been conducted to assess site-specific exposure to arsenic. Hwang et al. (7) measured arsenic levels in soil, interior dust, drinking water, and urine of children between the ages of 8 and 76 months. While the subjects in the urinary arsenic study were children, children are not considered a sensitive subpopulation to the adverse effects of arsenic. Children were selected because their exposure pathways could be more easily defined than the exposure pathways of adults. Freeman and co-workers experimentally determined the bioavailability of arsenic in soil and interior dust collected in Anaconda in both rabbits (8) and monkeys (9). E.J. Calabrese (unpublished data) measured incidental soil ingestion in children living in Anaconda and arsenic concentrations in food, fecal samples, soil, and dust.

This paper, which presents a comparison of results of these studies to the results of the exposure assessment model, confirms that measured urinary arsenic concentrations reasonably agree with the urinary arsenic concentrations predicted by the EPA's CTE assessment and that the RME assessment model identifies the upper percentiles of the exposed population. In addition, we discuss results from two sample collection methodologies for soil and interior dust and describe why one methodology is preferable to the other for data to be used in the EPA exposure assessment model.

This paper presents a summary of the methods used, the results of the comparison of measured urinary arsenic concentrations to predicted urinary arsenic concentrations, and a discussion of the importance of scientifically sound site-specific data in evaluating human exposure at Superfund sites.

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Methods

This section identifies the assumptions and methodology of the comparison of measured urinary arsenic concentrations to predicted urinary arsenic concentrations. These comparisons include the urinary and environmental arsenic study by Hwang et al. (7); the *in vivo* bioavailability study by Freeman et al. (9); the soil ingestion study and the arsenic concentration in food, fecal material, soil, and dust study by Calabrese and Stanek (10); the EPA exposure assessment model for arsenic; and the statistics used for comparison.

Two of the three studies collected multiple data used in this comparison. Table 1 identifies the multiple data measured in these studies.

The comparison methodology assumes that the amount of arsenic excreted in urine is equal to the amount of arsenic absorbed from food, water, dust, and soil. Excretion of absorbed arsenic is mainly via the urine. Studies have shown that between 6 and 9% of ingested arsenic is eliminated in the feces, whereas 80% is excreted via urine in about 3 days (11).

Urinary arsenic concentrations are indicative of recent arsenic exposure. The biologic half-life of ingested inorganic arsenic is about 10 hr, and that of methylated arsenic has been estimated at 30 hr. The biotransformation of inorganic arsenic to methylated species is thought to occur predominately in the liver. Kidney and lung tissue are also thought to contribute to the biotransformation of arsenic (12). Total urinary arsenic is the measure of intake of all forms of arsenic, including organoarsenic compounds. Organoarsenicals are found in many foods, particularly seafood (13). They are thought to be nontoxic and are generally excreted without metabolic transformation (14). Speciated urinary arsenic is the measure of inorganic and biotransformed arsenic species and represents the total intake of inorganic arsenic. It is the recommended biomarker for recent inorganic arsenic exposure. Species of arsenic detected by this methodology are the inorganic arsenic species, arsenate and arsenite, and the biotransformed species, monomethylarsonate and dimethylarsinate. While the biological activity of these species varies, the sum of the species represents the total exposure to arsenic. The comparison methodology assumes that absorbed inorganic arsenic remains unchanged or is biotransformed to methylated arsenic species (15).

Hwang study data collection methodology. From the summer of 1992 to the summer of 1993, urinary arsenic concentrations and environmental arsenic concentrations were collected from 364 children (7). Children studied by E.J. Calabrese (unpublished data) are a subset of the Hwang study

population. First morning voided urine samples were collected two consecutive mornings and the results were averaged. Results were reported as micrograms of arsenic per liter. In addition, 24-hr urine output was measured in a subset of 25 children. Creatinine, total urinary arsenic, and speciated urinary arsenic were measured. Total urinary arsenic includes inorganic arsenic, methylated arsenic, arsenobetaine, and arsenocholine. Speciated urinary arsenic contains As^{+3} , As^{+5} , monomethylarsonic acid, and dimethylarsinic acid, which are the major arsenic species and/or metabolites found after exposure to inorganic arsenic.

Creatinine was measured with a colorimetric method using the Sigma Diagnostics Creatinine Kit (Sigma Chemical Co., St. Louis, MO). Urinary arsenic was measured using a hydride generation system attached to a flame atomic absorption spectrometer. Total arsenic analysis was performed on acid-digested urine samples, while speciated urinary arsenic analysis was based on direct analysis of the urine sample. Twenty-four hour urinary volume was estimated by averaging the urinary volume for children less than 36 months of age, between 36 and 60 months of age, and greater than 60 months of age.

Soil samples were composite samples taken from the first 2 cm of soil at each residential parcel. The areas sampled were perimeter soil, bare-area soil, garden-area soil, play-area soil, and if present, hard pack driving- or parking-area soil. All soil samples were sieved, and only soil passing through a 250- μ m mesh sieve was analyzed. Arsenic was analyzed using energy dispersive X-ray spectrometry. Interior dust samples were collected using a low volume air sampler from three locations: an area adjacent to the main entrance, a floor area in the room used most by the subject child, and a floor area in the child's bedroom. Interior dust arsenic was analyzed using graphite furnace atomic absorption spectrometry (GFAAS). First-draw water samples were collected from the kitchen faucet for those residences with domestic wells and analyzed with GFAAS. Most of the subjects used municipal water.

Freeman study data collection methodology. Freeman et al. (9) used soil

and dust collected from Anaconda to measure bioavailability in cynomolgus monkeys. The soil was sieved through a 250- μ m screen. Geometric mean particle sizes used for dosing were 25.2 μ m for soil and 30.8 μ m for dust. Three female monkeys were cycled through four dosing regimes. The four treatments included a single intravenous treatment [0.62 mg As/kg body weight (bw)], gavage treatment (0.62 mg As/kg bw), an oral administration of encapsulated soil (0.62 mg As/kg bw), and an oral administration of house dust (0.26 mg As/kg bw). Urine and feces were collected from each animal for 168 hr after dosing. Blood was also collected at predetermined times for 168 hr after dosing. Arsenic was measured in these samples using GFAAS.

The EPA obtained estimates of absolute bioavailability from Sigma Plot (Version 2.0 for Windows; Jandel Scientific, San Rafael, CA) by evaluating the area under the curve of the plotted urinary arsenic concentrations using the trapezoidal rule. The EPA did not normalize the curve for less than 100% from urinary arsenic from the intravenous exposure group.

Calabrese study data collection methodology. E.J. Calabrese (unpublished data; available in the Administrative Record for the Anaconda Superfund Site) collected food and fecal samples from 64 children for 7 days in September of 1992. The fecal sample collection began 1 day after food sample collection and ended 1 day later. Of the 448 samples, three food samples were lost or not submitted for analysis. Seven families reported missing fecal samples for children in the study. All children in this study were also in the Hwang et al. study (7).

Calabrese (unpublished data) also measured arsenic in soil and dust collected from 26 of the 64 children's homes. Soil samples were collected as composite samples using a proportional estimate of play to determine the proportion of soil from each area to be composited. Interior dust samples were collected with a commercial vacuum cleaner. Soil and dust samples were ground in a mixer mill for 10 min before analysis. Because Hwang et al. (7) collected soil and dust samples from the same 26

Table 1. Data measured in studies used for comparison

Study	Bioavailability	Soil ingestion rate	Total urinary arsenic	Speciated urinary arsenic	Soil arsenic	Dust arsenic	Arsenic in drinking water	Food arsenic
Hwang ^a			✓	✓	✓	✓	✓	
Freeman ^b	✓							
Calabrese ^c		✓			✓	✓		✓

^aData from Hwang et al. (7).

^bData from Freeman et al. (9).

^cData from Calabrese and Stanek (10).

homes during approximately the same time period using a different methodology, the methodologies could be compared.

The trace elements measured in food, fecal material, soil, and dust were aluminum, silicon, titanium, chromium, yttrium, zirconium, lanthanum, cerium, and neodymium. Arsenic was also measured in each sample. Inductively coupled plasma–atomic emission spectrometry was used for analysis of silicon, titanium, aluminum, yttrium, and zirconium. Inductively coupled plasma–mass spectrometry was used for analysis of chromium, lanthanum, cerium, neodymium, yttrium, and arsenic.

Two methodologies were used to estimate incidental soil ingestion. The first method, known as the “best single tracer methodology,” identifies the trace element with the lowest food/soil ratio for each child and then uses that tracer to estimate soil ingestion (10). The second method uses the median of four best tracers with the lowest food/soil ratio for each child to estimate soil ingestion.

EPA exposure assessment model. The EPA exposure assessment model was used

to estimate daily absorption of arsenic from soil, interior dust, water, and food (when estimating total arsenic exposure). Absorption was estimated as milligrams of arsenic per day. Site-specific data were used in the model when they were available. Where site-specific data were unavailable, standard EPA default assumptions were used to quantify intakes (1,16). The formula used to estimate daily absorption is

$$ABS = \frac{C_s \times IR_s \times CF_s \times EF \times BAF_s}{AT} + \frac{C_w \times CF_w \times IR_w \times EF \times BAF_w}{AT}, \quad (1)$$

where C_s is the As concentration of ingested soil and dust; IR_s is the ingestion rate of combined soil and dust; CF_s is the conversion factor for soil; EF = exposure frequency; BAF_s is the bioavailability of soil and dust; C_w is the As concentration of ingested water; CF_w is the conversion factor for water; IR_w is the ingestion rate of water; BAF_w is the bioavailability of water; and AT is the averaging time. See Table 2 for

further information and the assumed values used to estimate both CTE and RME daily absorption. Site-specific data used in the exposure assessment model are indicated.

To allow comparison of estimated daily absorption in milligrams per day to measured urinary arsenic levels in micrograms per liter, the following formula was used:

$$EXC = \frac{ABS \times CF_{abs}}{RATE \times CF_{exc}} \quad (2)$$

EXC is the urinary AS excreted; CF_{abs} is the conversion factor for micrograms to milligrams; $RATE$ is the estimate urinary output per day for a given age (months); and CF_{exc} is the conversion factor from milliliters to liters.

Statistics. Arithmetic and geometric means and standard deviations (SDs) were calculated for all comparisons. The Student's t -test was used to compare paired measured speciated and total urinary arsenic to predicted speciated and total urinary arsenic. The Kruskal-Wallis test was used to determine if the populations were the same when the data were not

Table 2. Symbol, definition, and assumed values used for the EPA Exposure Assessment Model

Symbol	Definition	Value used	
		Central tendency	Reasonable maximum
C_s	Arsenic concentration of ingested soil and dust (mg/kg)	55% sampled interior dust arsenic ^a 45% sampled average yard arsenic ^a	55% sampled interior dust arsenic ^a 45% sampled average yard arsenic ^a
IR_s	Ingestion rate of combined soil and dust (mg/day)	100 mg soil/day	200 mg soil/day
CF_s	Conversion factor (kg/mg)	10^{-6} kg/mg	10^{-6} kg/mg
EF	Exposure frequency (days)	350 days	350 days
AT	Averaging time (days)	365 days	365 days
C_w	Arsenic concentration of ingested water (µg/l)	Measured water arsenic concentration (or one half the detection limit; 0.5 µg/l) ^a	Measured water arsenic concentration (or one half the detection limit; 0.5 µg/l)
CF_w	Conversion factor (mg/µg)	10^{-3} mg/µg	10^{-3} mg/µg
IR_w	Ingestion rate of water (l/day)	0.7 l water/day	1 l water/day
BAF_s	Bioavailability of soil and dust	25.8% for dust ^a 18.3% for soil ^a	25.8% for dust ^a 18.3% for soil ^a
BAF_w	Bioavailability of water	100%	100%
ABS	Estimated absorbed arsenic per day for each individual child (mg/day)	Calculated for each individual child	Calculated for each individual child
EXC	Urinary arsenic excreted (mg/day)	Calculated for each individual child	Calculated for each individual child
CF_{abs}	Conversion factor for micrograms to milligrams	103 µg/mg	103 µg/mg
CF_{exc}	Conversion factor for milliliters to liters (l/ml)	10^{-3} l/ml	10^{-3} l/ml
$RATE$	The estimated urinary output per day for a given age in months	Children <36 months of age excrete 240 ml urine/day ^a Children between 36 and 60 months of age excrete 355 ml urine/day ^a Children >60 months of age excrete 432 ml urine/day ^a	Children <36 months of age excrete 240 ml urine/day ^a Children between 36 and 60 months of age excrete 355 ml urine/day ^a Children greater >60 months of age excrete 432 ml urine/day ^a

^aIndicates site-specific data used in the exposure assessment model.

paired. Nonparametric confidence limits for quantiles were estimated using the method described by Gilbert (17).

Results

While the site-specific studies used in the comparison of measured to predicted urinary arsenic have been published, data from the individual were evaluated independently by the EPA for consideration in the exposure assessment model. This section presents the results of the studies by Hwang et al. (7), Freeman et al. (9), and Calabrese et al. (10), the results of the exposure assessment, and the comparison of measured to predicted urinary arsenic concentrations.

Hwang study results. To measure urinary arsenic, Hwang et al. (7) collected first morning void for consecutive days and averaged the results for each child. The arithmetic and geometric means for measured average total and speciated urinary arsenic are presented in Table 3. Speciated urinary arsenic was measured in 366 children and total urinary arsenic was measured in 364 children.

For the 25 children for which urinary output was measured over a 24-hr period, the estimated urinary output per day was 240 ml/day for children less than 36 months of age, 355 ml/day for children between 36 and 60 months of age, and 432 ml/day for children greater than 60 months of age.

The arithmetic mean of reported arsenic soil values for each yard was used as the soil concentration (C_s) of arsenic to predict soil exposure for each child for the EPA exposure assessment model. Interior dust samples were collected as a composite from the three locations within the house. Reported values could be used as the C_s to predict interior dust exposure for each child in the CTE exposure assessment model. For the total number of samples of soil and interior dust collected in Anaconda, the arithmetic

means \pm SDs were 172.3 \pm 74.4 and 86.5 \pm 53.4 mg/kg, respectively. For the subset of households where soil and dust were measured by both Hwang et al. (7) and E.J. Calabrese (unpublished data), the arithmetic means for soil and dust were 192 mg/kg and 75 mg/kg, respectively, in the Hwang study and 75 mg/kg and 29 mg/kg, respectively, in the Calabrese study. Examples of the differences between individual household soil and dust arsenic concentrations are presented in Table 4. Table 4 also presents the average soil and dust concentrations for each study.

Forty-six of the children included in the study used domestic wells as the household water supply. Arsenic concentrations in well water ranged from 1.1 to 9.9 μ g/l. Arsenic was not detected in the municipal water supply. The detection limit was 1.0 μ g/l.

Freeman study results. Bioavailability is defined as the fraction of the amount of arsenic in the system after oral dosing compared to the amount of arsenic in the system after intravenous administration adjusted for the difference in the size of the dose. The mean absolute bioavailability of arsenic estimated from urine arsenic concentrations in cynomolgus monkeys was 91%, 18.3%, and 25.8% for gavage, soil, and dust, respectively. The study demonstrated that the absorption of arsenic from soil and dust was significantly less than the absorption of soluble arsenic from water and provided support for site-specific adjustments in arsenic bioavailability at the Anaconda site.

Calabrese study results. Using the best single tracer methodology, the median soil and dust ingestion rate for 64 children living in Anaconda was 51 mg/day, the mean ingestion rate was 117 mg/day, and the 90th percentile was 277 mg/day. The range was 0–899 mg/day. The “four best tracers”

methodology resulted in a median ingestion rate of 39 mg/day, a mean ingestion rate of 83 mg/day, and a 90th percentile rate of 273 mg/day. The range was 0–515 mg/day. The findings of the Anaconda soil and dust ingestion study support the Superfund Program’s default assumption of 100 mg/day as the CTE assumption for children 0–6 years of age.

The ingestion rate includes ingestion of both soil and interior dust. The EPA assumption for relative contribution of the two media are 55% from indoor dust and 45% from soil. The average arsenic soil and interior dust concentrations are presented in Table 4.

Arsenic in the daily food samples from the subset of 30 children was 0.00705 \pm 0.0065 mg/day (mean \pm SD). All children over 18 months of age were assumed to eat solid food.

Exposure assessment results. Speciated and total urinary arsenic concentrations were predicted for each child using the EPA exposure assessment equation, the CTE and RME assumptions presented in Table 2, arsenic concentrations from soil and interior dust for each individual child’s exposure unit, the site-specific bioavailability for soil and dust, and the estimated daily urinary output. The soil and interior dust concentrations used were from Hwang et al. (7).

Comparison of predicted and measured urinary arsenic concentrations. Table 3 presents the median, the arithmetic and geometric means, and the arithmetic and geometric SDs for both total and speciated measured urinary arsenic concentrations and the total and speciated urinary arsenic concentrations predicted using the CTE assessment model. Figure 1 presents the measured speciated urinary arsenic concentrations for 366 children between the

Table 3. Comparison of measured and CTE predicted urinary arsenic concentrations

Statistic	Urinary arsenic concentration (μ g/l)	
	Measured	Predicted
Total urinary arsenic		
Median	19.5	26.6
Arithmetic mean \pm SD	26.1 \pm 25.9	28.7 \pm 7.4
Geometric mean (SD)	20.8 (1.8)	27.8 (1.3)
Speciated urinary arsenic		
Median	9	8
Arithmetic mean \pm SD	10.9 \pm 7.1	9.4 \pm 1.7
Geometric mean (SD)	9.0 (4.4)	8.1 (1.5)

Speciated urinary arsenic, the recommended biomarker for recent inorganic arsenic exposure, measures arsenate, arsenite, monomethylarsenate, and dimethylarsinate. Abbreviations: SD, standard deviation; GSD, geometric standard deviation.

Table 4. Examples of arsenic concentration in soil and dust samples collected by different methods from the same household, and data averages

Household number	Average soil arsenic concentration (mg/kg)		Average interior dust arsenic concentration (mg/kg)	
	Hwang ^a	Calabrese ^b	Hwang ^a	Calabrese ^b
11	161.9	76.572	25.10	15.774
15	197.22	20.547	91.70	7.943
20	122.83	96.073	40.70	12.568
31	257.71	77.799	54.30	26.489
43	127.14	123.424	76.10	45.591
50	187.11	112.509	110.80	34.467
65	86.2	70.501	83.40	33.570
72	79.25	18.595	38.70	13.347
123	172.33	94.700	84.2	45.930
Total number of samples from same household	26	25	25	26
Average arsenic concentration in samples from same household	192.36	74.67	75.14	29.03

^aData from Hwang et al. (7).

^bData from E.J. Calabrese (unpublished; available in the Administrative Record for the Anaconda Superfund Site).

ages of 8 and 76 months and the CTE predicted speciated urinary arsenic concentrations for 374 children who lived in residences where environmental media arsenic concentrations were available. Measured and predicted urinary arsenic concentrations were sorted separately and plotted from lowest to highest urinary arsenic concentration. Figure 1 shows reasonable agreement between the two populations.

Kruskal-Wallis one-way analysis of variance (ANOVA) demonstrated that the populations from which the measured and predicted data sets were drawn have the same mean. For measured urinary arsenic concentrations, as reflected in urinary arsenic values below 17 $\mu\text{g/l}$, the CTE-predicted estimates reproduce measured arsenic concentrations well, with a paired 2-tailed Student's *t*-test probability of 0.85 for the speciated urinary arsenic comparison. The remaining 50 children, or 13.6% of the population, have speciated urinary arsenic concentrations under-predicted by the CTE assessment model.

Table 5 presents measured, CTE-predicted, and RME-predicted urinary arsenic concentrations for different cumulative population percentiles. Concentrations predicted by the CTE assessment model were similar to measured urinary arsenic concentrations. The RME assessment model, designed to identify the 90th–98th cumulative percentile of the population, overpredicted urinary arsenic concentration for the majority of the population, but predicted the 98.9 cumulative percentile, demonstrating that the RME assessment model performed as designed.

Discussion

The results of this comparison demonstrate that the urinary arsenic concentrations predicted by the EPA CTE assessment model reasonably agree with measured urinary arsenic concentration (accepted biomarkers of recent arsenic exposure). The results also demonstrate that the most exposed or high-end population is identified using the

RME assessment model. This comparison indicates that the EPA exposure models can reasonably predict central tendency and high-end exposures, provided that site-specific data which adequately characterize the exposure pathways present at the site have been collected. For example, if the EPA's standard default assumption of 100% bioavailability of arsenic from soil and dust were used (18,19), the model estimates would have overpredicted exposure at the site by 200%.

In the comparison, variations were observed between measured and predicted urinary arsenic concentrations for each individual child; however, these variations were expected. Differences in behavior, the contaminated media contacted, and physiological parameters that influence toxicokinetics of arsenic are all plausible explanations for variability between measured and predicted urinary arsenic concentrations for an individual. Significant factors identified in this comparison that effect variability include soil ingestion rate, the exposure unit, dust versus soil ratio, urinary volumes, arsenic concentration in food, and the soil and dust data collection methods. Each factor is discussed below.

The findings of the site-specific soil ingestion rate data collected by Calabrese support the EPA default soil ingestion assumptions based on arithmetic means. There is some controversy concerning whether the arithmetic mean or median is the most appropriate statistical indicator of soil ingestion. The American Industrial Health Council uses the median soil ingestion value (20), whereas the EPA has made a policy decision to use the arithmetic mean (21). This comparison demonstrates that the arithmetic mean is appropriate for use as the predictor of exposure. If the median value were used for this comparison, predicted urinary arsenic concentrations would be approximately one-half the concentration presented in Table 4 and the modeled population would no longer be representative of the measured population.

While children are not a sensitive subpopulation to the toxic effects associated with exposure to arsenic, a child's exposure is assumed to be greater and more easily defined than an adult's exposure. By having measured urinary arsenic concentrations from children, comparison to urinary arsenic concentrations predicted from more easily defined exposures is likely to result in better agreement. It is much more difficult to model an adult's exposure. Because predicted and measured urinary arsenic concentrations reasonably agree for children, adult exposure can be modeled with greater confidence. For the EPA exposure assessment

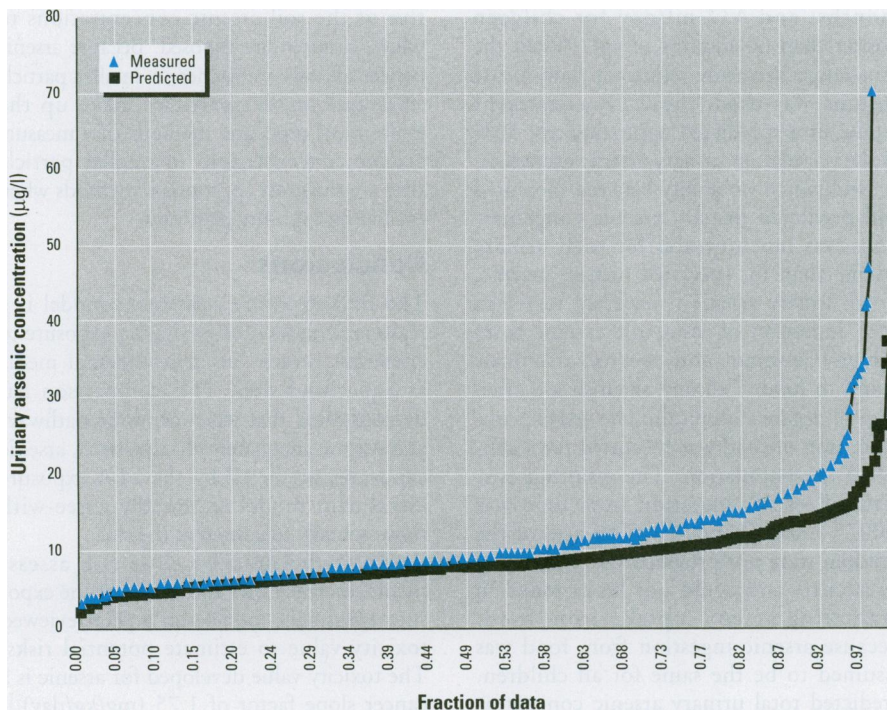


Figure 1. Measured and predicted speciated urinary arsenic concentrations.

Table 5. Measured and predicted speciated urinary arsenic concentrations at different population percentiles

Population percentile	Speciated urinary arsenic ($\mu\text{g/l}$)		
	Measured (CI)	CTE predicted (CI)	RME predicted
5	4.5 (4.0–5.0)	4.0 (3.7–4.3)	7.5
25	6.5 (6.0–7.0)	6.2 (5.7–6.4)	11.5
50	9.0 (8.5–9.5)	8.1 (7.7–8.6)	15.3
75	13.0 (12–14)	10.7 (10–11)	20.2
95	21.0 (20–28.5)	17 (15–21)	30.1
100	47	35	45.5

CI, 95% confidence interval; CTE, central tendency exposure; RME, reasonable maximum exposure.

model, each child's exposure is assumed to be from the soil in the child's yard and the dust in the child's home. Variability between measured and predicted urinary arsenic concentrations is still expected, even with children, because the model assumes 100% of the child's daily activity occurs at home; factors such as a child spending time at a day-care facility were not included in the model.

Individual variability is also associated with the ratio of exposure to arsenic in interior dust and soil. In the absence of site-specific data, the EPA assumes that the concentration of chemicals in interior dust is the same as the concentration of chemicals in soil. However, at the Anaconda site, measured concentrations of arsenic in interior dust samples were slightly less than one-half the measured concentrations in soil.

The estimated amount of interior dust ingested represents an estimate of the time an individual spends indoors. For the Anaconda site, interior dust was assumed to represent 55% of a child's ingestion exposure to environmental arsenic. Changing the ratio of interior dust to soil ingestion will change the predicted urinary arsenic concentration. Stanek and Calabrese (22) indicated that approximately 50% of residual fecal tracers in a soil ingestion study were of indoor origin. The Integrated Exposure Uptake Biokinetic Model for Lead in Children developed by the EPA uses 45% ingestion of dust and 55% ingestion of soil as the default assumption (23). The ratio of soil to dust intake is not considered proportional to the amount of time spent outdoors versus the amount of time spent indoors. It is estimated that children spend only 15–30% of their waking hours outdoors, but are more exposed to accessible particles and are less likely to wash their hands as often when outdoors. While the ratio of soil to dust ingestion has not been fully resolved in the scientific literature, between 45 and 55% dust ingestion is considered reasonable.

Individual variability between measured and predicted urinary arsenic concentrations may be related to spot measurements of urinary arsenic compared to predicted average yearly exposure. The EPA exposure assessment model is designed to predict the average yearly exposure of the population. Hwang et al. (7) measured spot urinary arsenic concentrations. Although not detailed in the present comparison, Hwang et al. (7) also examined seasonal variation in urinary arsenic concentrations. Urinary arsenic concentrations were highest in the late spring and summer, intermediate in the fall and early spring, and lowest in the winter. If urinary arsenic was measured daily and averaged over a year, there might

be less variability between measured and predicted urinary arsenic concentrations.

Hwang et al. (7) reported the urinary arsenic concentration for each child in micrograms per liter. The EPA exposure assessment model predicts exposure in milligrams per day. To allow comparison of the measured and modeled results, an estimate of daily urinary volume was required. Measured 24-hr urinary arsenic output was a site-specific variable increasing the agreement between measured and predicted urinary arsenic concentrations. Predicted urinary arsenic excretion is directly dependent on estimated urinary output per day. The reported volume of urine for children between the ages of 12 and 36 months is 500–600 ml/day, between the ages of 37 and 60 months is 600–700 ml/day, and between the ages of 61 and 96 months is 650–1,000 ml/day (24). The site-specific urinary arsenic study measured urine volume for a subset of 25 children at 240 ml/day for children under 36 months; 355 ml/day for children between 36 and 60 months; and 432 ml/day for children greater than 60 months of age. When the mid-range literature value for daily urine volume was used, the CTE assessment model underpredicted approximately 30% of the population urinary arsenic excretion.

Individual variability between measured and predicted urinary arsenic concentrations was much greater for total urinary arsenic than for speciated urinary arsenic. Total urinary arsenic is expected to reflect both ingestion of inorganic arsenic from the environment and organic arsenicals found in food. Fish and shellfish are often high in organic arsenicals. The arsenic concentration of food was measured for a subset of the population. The resulting estimate of arsenic ingestion from food was 0.0075 ± 0.0065 mg/day. The ages of the children were not provided by the primary researchers and could not be included in evaluating arsenic uptake from food. Because arsenic ingestion from food was assumed to be the same for all children, predicted total urinary arsenic concentrations had a much higher degree of variability when compared to measured total urinary arsenic concentrations than to measured and predicted speciated urinary arsenic concentrations.

For future exposure assessments to generate similar comparability between measured and predicted urinary arsenic concentrations, the soil and dust sample collection methods used by Hwang et al. (7) appear to be the collection methods of choice. For both soil and dust, arsenic concentrations were approximately 2.6 times greater in the Hwang study than in the Calabrese study.

The methodology used in the Hwang study generates reasonable comparability between measured and predicted urinary arsenic concentrations. Davis et al. (25) demonstrated that arsenic-bearing phases in soil and interior dust particles collected in Anaconda increase with decreasing size. Small particles have also been shown to more readily adhere to children's hands and are therefore more available for ingestion (26). For a small subset of the population, soil and dust samples were collected from the same residences using two different methodologies. The methodology of Hwang et al. (7) focused on collecting samples containing the smaller particle sizes. In the Hwang study, interior dust was collected using a low-volume air sampler (7), whereas dust samples in the Calabrese study were collected using a commercial vacuum cleaner (10). Low-volume air sampling is expected to collect smaller particles. The Hwang method of soil sample collection resulted in analysis of particles sieved through a 250- μ m screen, small particles that are expected to be more representative of the soil arsenic concentrations to which children are exposed. Because arsenic concentrations are higher in smaller particle sizes and smaller particles make up the majority of exposure, methods that measure arsenic concentrations in smaller particle sizes are the most appropriate methods when evaluating exposure pathways.

Conclusions

The EPA exposure assessment model is a reasonable model for predicting exposure to chemicals present in environmental media at Superfund sites. This comparison has demonstrated that when exposure pathways at a site are adequately characterized, arsenic exposures predicted by the EPA exposure assessment model reasonably agree with those actually measured at the site.

For Superfund baseline risk assessments, the EPA uses the results of the exposure assessment model and a peer-reviewed toxicity value to estimate potential risks. The toxicity value developed for arsenic is a cancer slope factor of 1.75 (mg/kg/day)⁻¹. If default values had been used in the CTE and RME exposure assessment models for this site, the resulting potential cancer risks would have been predicted to be 1.7×10^{-4} and 8.5×10^{-5} , respectively. Instead, by incorporating the site-specific data in the exposure assessment models, the resulting risks were 4.0×10^{-5} and 1.9×10^{-5} . The use of site-specific data in the risk assessment allowed focused cleanup in fewer areas of the community, resulting in a more cost efficient remediation while being protective of human health.

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