Longitudinal Investigation of Exposure to Arsenic, Cadmium, and Lead in Drinking Water

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Arsenic, cadmium, and lead have been associated with various forms of cancer, nephrotoxicity, central nervous system effects, and cardiovascular disease in humans. Drinking water is a well-recognized pathway of exposure to these metals. To improve understanding of the temporal dimension of exposure to As, Cd, and Pb in drinking water, we obtained 381 samples of tap and/or tap/filtered water and self-reported rates of drinking water consumption from 73 members of a stratified random sample in Maryland. Data were collected at approximately 2-month intervals from September 1995 through September 1996. Concentrations of As (range < 0.2–13.8 µg/L) and Pb (< 0.1–13.4 µg/L) were within the ranges reported for the United States, as were the rates of drinking water consumption (median < 0.1–4.1 L/day). Cd was present at a detectable level in only 8.1% of the water samples. Mean log-transformed concentrations and exposures for As and Pb varied significantly among sampling cycles and among respondents, as did rates of drinking water consumption, according to a generalized linear model that accounted for potential correlation among repeated measures from the same respondent. We used the intraclass correlation coefficient of reliability to attribute the total variance observed for each exposure metric to between-person and within-person variability. Between-person variability was estimated to account for 67, 81, and 55% of the total variance in drinking water consumption, As exposure (micrograms per day), and Pb exposure (micrograms per day), respectively. We discuss these results with respect to their implications for future exposure assessment research, quantitative risk assessment, and environmental epidemiology. *Key words*: arsenic, cadmium, chronic exposure, drinking water, lead.
Environ Health Perspect 108:731–735 (2000). [Online 26 June 2000]
http://ehpnet1.niehs.nih.gov/docs/2000/108p731-735ryan/abstract.html

Numerous metals have received attention as both environmental contaminants and potential toxicologic hazards. For example, arsenic, cadmium, and lead are extensively distributed in the environment (*1–3*). Human activities have substantially altered the natural distribution of these metals in the environment, leading to potentially elevated concentrations of these metals in many environmental media. The occurrence of As, Cd, and Pb in drinking water is considered an important pathway of potential exposure for citizens of the United States and many other nations (*4–6*).

The temporal dimension of exposure to these elements in drinking water is of interest because their toxicologic effects, such as cancer, kidney disease, and impaired cognitive function, are considered to result from chronic exposure rather than short-term exposure (*7–9*). Improved understanding of chronic exposure to As, Cd, and Pb in drinking water can be expected to increase the precision and accuracy of health risk assessments performed for these metals.

Our objectives were to measure short-term and prolonged (1-year) average exposure to As, Cd, and Pb from ingestion of drinking water; to characterize the temporal variability of short-term exposure measures; and to evaluate the reliability of short-term measures of exposure to assess long-term average exposure. We assessed exposure to the metals from repeated measurements of the analytes in drinking water samples collected from each member of the study population. These data are one component of a longitudinal investigation of chemicals in multiple media: The National Human Exposure Assessment Survey in Maryland (NHEXAS-MD).

Methods

Study design. A stratified random sample of 80 individuals selected from four contiguous counties in Maryland was enrolled in the study in September 1995. The NHEXAS-MD sampling strategy was designed to ensure adequate representation of urban, suburban, and rural residences as well as the racial diversity of the metropolitan Baltimore area. The area represented is the Baltimore metropolitan statistical area, which consists of Anne Arundel County, Baltimore County, Baltimore City, and Queen Anne's County. We included an additional contiguous county, Talbot County, to ensure adequate representation of the rural stratum. Within each of these geographic regions, U.S. Census block groups were defined as urban (population density > 10,000/square mile) predominantly (> 50%) white; urban predominantly nonwhite; suburban (1,000/square mile < population density < 10,000/square mile) predominantly white; suburban predominantly nonwhite; and rural (population density < 1,000/square mile). There was no racial differentiation for the rural block groups. We selected five block groups within each stratum, with probability of selection based on the number of residents, for a total of 25 block groups. Our sampling frame was a list of residential telephone numbers obtained from a commercial vendor. Errors are introduced using this sampling frame, but these are considered small in comparison to other potential errors dictated by the final sample size. Our desired final sample size, based primarily on logistical limitations, was 50 households, or two per each block group. Given expected dropout rates, this required initial monitoring of 80 residences, or just over three per block group. We selected 20 homes per block group (500 total telephone numbers) to ensure a sufficient sampling frame. Telephone numbers were called randomly within each block group until the desired number within each block group was reached. We determined sample weights through reflection of the sampling design, with appropriate weights reflecting differential probability of selection from the initial population for each stratum. Actual sampling weights were fairly consistent across the 12 strata, suggesting that weighted and unweighted results should be similar. Specific weights for each participant and cycle combination can be obtained from the authors.

All participants provided informed consent under protocols approved by an institutional review board. MacIntosh et al. (*10*) summarized the demographic characteristics of the respondents. Each individual participated in the study as many as 6 times over

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This research was supported by the U.S. Environmental Protection Agency under cooperative agreement CR822038-1, the U.S. Department of Agriculture Hatch Project GEO00843, and the University of Georgia Research Foundation.

Received 27 October 1999; accepted 11 April 2000.

the course of the following 12 months. The interval between sampling cycles for each participant was approximately 8 weeks. Dates of the six sampling periods were as follows:

- cycle 1, 20 September 1995–23 December 1995
- cycle 2, 15 January 1996–23 February 1996
- cycle 3, 27 February 1996–20 April 1996
- cycle 4, 22 April 1996–15 June 1996
- cycle 5, 18 June 1996–27 July 1996
- cycle 6, 30 July 1996–18 September 1996.

Field staff members collected samples of environmental and biologic media, including drinking water, during a consecutive 7-day period within a cycle. Study participants completed exposure-related questionnaires during each cycle.

Sample collection and analysis. On the first day of each sampling period, we obtained a water sample from the primary source of drinking water identified by the respondent (e.g., direct from the tap or effluent from a treatment system such as a water softener or charcoal filter). Beverage intake of metals is discussed elsewhere (*11*). One-liter water samples were drawn into precleaned high-density polyethylene containers and maintained at < 4°C during storage and transport. Before sample collection, tap water was flushed for 2 min. Samples were preserved at pH < 2 by the addition of several drops of 6 M nitric acid in the field. Aliquots of samples were directly injected for multielement analysis by inductively coupled plasma-mass spectrometry (ICP-MS). The target elements for the NHEXAS-MD investigations were As, Cd, chromium, and Pb. Results for other elements determined by ICP-MS are under study.

A questionnaire on daily time budgets and behavior patterns was administered to participants on each day of the 7-day sampling period for a given cycle. Responses to the question, "How many glasses of water did you drink today?" were used to obtain drinking water consumption rates. We used a nominal serving size of 0.296 L (10 fluid oz) per glass to convert from glasses of water to liters of water consumed per day. We computed exposure to the target analytes for each cycle as the product of the 7-day average drinking water consumption rate and the analyte concentrations measured in the water sample.

Quality assurance. To ensure creditability and accuracy of the data, we performed a series of quality assurance steps for concentration data on the target elements, As, Cd, Cr, and Pb, and for water intake. A chain-of-custody (COC) form followed each sample and questionnaire from the field to the laboratory and finally to the database manager. A water sample data point or questionnaire data point not accompanied by a completed COC was omitted from subsequent analysis.

We performed sample collection and analysis procedures in accordance with quality assurance measures prescribed by U.S. Environmental Protection Agency method 200.8 (*12*). Detection limits (DLs) and recovery efficiencies were determined for each metal species throughout the study. The DL was constant over time for As (0.2 µg/L), Cd (0.1 µg/L), and Pb (0.1 µg/L). Recovery efficiency as measured by fortified samples was 108.5% (n = 65, SD 4.9%) for As, 103.7% (n = 66, SD 4.5%) for Cd, and 103.5% (n = 66, SD 5.6%) for Pb. Fortified sample concentrations were typically 20.0 µg/L for As and 10.0 µg/L for Cd and Pb.

We obtained 16 field blanks and 24 duplicate samples throughout the study to evaluate potential contamination of samples during their collection and shipment. Arsenic, Cd, and Pb were not detected in any field blank samples. The mean absolute difference of metal concentrations between pairs of duplicate field samples was 0.09 (SD 0.19, median 0.02, interquartile range 0.05); 0.004 (0.02, 0.0, 0.0); and 0.70 (2.22, 0.07, 0.12) µg/L for As, Cd, and Pb, respectively. The ratio of the median absolute difference between duplicate pairs and the overall median concentration was 4% for As and 18% for Pb. For Cd, both values reflect the limit of detection (LOD) and the ratio is not meaningful.

Data analysis. To evaluate temporal variability of heavy metal exposure via drinking water, we restricted data reporting and analysis to participants from whom at least two sets of quality assured water sample and water consumption data were obtained. Metal concentrations below the DL were set to one-half the DL. Metal exposure via beverage consumption, including those exposures that could be derived from drinking water (e.g., coffee), was assessed separately (*11*).

We generated descriptive statistics for each variable for the complete data set and for each cycle. The observed concentration, water consumption, and exposure data exhibited positive skewness, whereas the corresponding natural log-transformed values were approximately normally distributed. Statistical analyses were therefore performed on natural log-transformed values. The observed water consumption data were approximately normally distributed; thus analyses of this variable were performed on the observed values. Based on procedures described in detail elsewhere (*13*), we used a mixed generalized linear model (GLM) to test for significant variability of mean metal concentrations (micrograms per liter), average daily water consumption rates (liters per day), and average daily exposure to each metal among sampling cycles (micrograms per day). In addition, we used a two-way GLM to test for significant interindividual variability for each exposure metric controlling for the effect of sampling cycle.

Reliability is a concept to describe the degree to which a single measure or set of measures of an event describes the mean of repeated measures of that event (*14*). We used three approaches to estimate the reliability of a short-term measure of heavy metal concentration in drinking water and the associated exposure. In the first approach, we computed the population mean intraindividual or within-person coefficient of variation for each exposure measure. The mean coefficient of variation for the population expresses the variability of exposure among sampling events for a typical individual compared to their annual average exposure. In the second approach, we used the PROC MIXED procedure in SAS 6.12 for Windows (SAS Institute, Cary, NC) to estimate the correlation coefficient between each pair of cycles for metal concentrations in water, water consumption, and exposure to metals in drinking water. The model includes linear terms for both population variability, using the household identification number (*HIN*) as a class variable representing individuals within the population, and temporal variability, using the sampling cycle variable (*cycle*) as a class variable:

$$\log X = \beta_0 + \beta_1\, HIN + \beta_2\, cycle$$

where X stands for metal concentration, tap water consumption, or metal intake. We report the mean correlation coefficient calculated from the correlation matrix for each exposure variable. Finally, we computed the intraclass correlation coefficient of reliability (R) according to the classic error model, where R is the ratio of between-person variance of exposure to the total variance of exposure observed in a repeated measure study (*14*). Total variance is the sum of between-person variance and within-person variance, which we estimated by a one-way GLM with an exposure metric (e.g., arsenic concentration in drinking water) as the dependent variable and respondent identification number as the categorical independent variable. R can range from 0 to 1, with values near zero indicating low reliability and values near one indicating high reliability.

Results

We obtained 403 valid water samples and 394 valid measures of drinking water consumption data. After we removed incomplete records (i.e., observations missing water sample or water consumption information) and the six households that participated in only one cycle, the database contained 381 observations from 73 participants. The number of observations per respondent was as follows: 38 respondents provided samples in all six cycles, 19 in five cycles, 10 in four cycles, and

6 in three cycles. The number of observations obtained in cycles one through six was 70, 66, 68, 66, 52, and 59, respectively.

Short- and long-term average metal concentrations and exposures. Arsenic, Cd, and Pb were present at detectable levels in 86.9, 8.1, and 92.4% of the water samples, respectively. The distributions of heavy metal concentrations and exposures are summarized in Table 1. The distribution of concentrations for each metal was skewed right and ranged over approximately 2 orders of magnitude. Cd was present at concentrations greater than the LOD only infrequently; for this reason, we did not perform further statistical analyses on the Cd data. The average daily drinking water consumption rate based on all 381 observations was approximately 1 L/day. One-half of the population reported average daily water ingestion between 0.51 and 1.31 L/day. These values are consistent with tap water consumption amounts obtained in other studies (*15*). Note that these short-term measures of exposure are based on analysis of a single drinking water sample and the average rate of drinking water consumption on 7 consecutive days.

We computed annual average drinking water metal concentrations and exposures using the 228 observations from the 38 respondents that participated in all six cycles (Tables 2 and 3). These long-term measures of exposure are based on six drinking water samples and the average rate of drinking water consumption on 42 days for each respondent collected over a 12-month period.

Temporal variability of metal concentrations and exposures. Median As concentrations ranged from 0.51 µg/L in cycle 6 to 0.72 µg/L in cycle 2 (Table 2). For Pb, median concentrations ranged over a factor of 3 among cycles. There was no apparent pattern to the timing at which the infrequent detection of Cd occurred. Mean log-transformed concentrations (Table 2) and exposures (Table 3) for As and Pb varied significantly among sampling cycles and among respondents. Mean log-transformed average daily drinking water consumption rates also varied significantly among cycles and participants. The least-squares means for water consumption obtained from the mixed model were as follows: 0.69, 0.86, 0.88, 0.91, 1.03, and 1.06 L/day for cycles one through six, respectively. Cycles five and six were not different from each other (p = 0.5223), although they were different from all other cycles ($p < 0.0169$).

Reliability of short-term exposure measures to assess long-term average exposure. The population mean within-person coefficient of variation for As concentration in drinking water was 0.20, whereas that for As exposure was 0.45 (Table 4). The correlation coefficients among repeated measures of As concentration (0.90) and exposure (0.82) as estimated by the mixed model procedure were greater than the corresponding correlation coefficients for Pb. In most cases, R was similar to the correlation among repeated measures estimated by the SAS procedure. Examination of Table 4 across analytes indicates that, as anticipated, the mean within-individual coefficient of variation is inversely related to R and to the mean between-cycle correlation coefficient.

Table 1. Distributions of exposure to three heavy metals in 2-min flush tap/filtered tap drinking water for 73 residences in Maryland between September 1995 and September 1996.

Analyte	No.	Mean[a]	SD[a]	Min	5%	25%	50%	75%	95%	Max
Concentration in drinking water (µg/L)										
As	381	0.77	1.11	< 0.20	< 0.20	0.48	0.61	0.72	2.58	13.80
Cd	381	0.07	0.05	< 0.10	< 0.10	< 0.10	< 0.10	< 0.10	0.14	0.30
Pb	381	1.08	2.01	< 0.10	< 0.10	0.19	0.37	0.84	3.63	13.40
Average consumption[b] (L/day)	381	0.91	0.60	0.04	0.10	0.51	0.89	1.31	2.07	4.10
Average exposure[c] (µg/day)										
As	381	0.78	1.78	0.004	0.03	0.21	0.52	0.83	2.65	26.83
Cd	381	0.06	0.06	0.002	0.005	0.03	0.05	0.07	0.12	0.40
Pb	381	1.06	2.50	0.005	0.02	0.13	0.33	0.75	2.78	18.13

[a]One-half the detection limit used for samples with concentrations less than limit of detection. [b]Daily water consumption. [c]Daily exposure through drinking water consumption.

Table 2. Summary statistics for heavy metals in 2-min flush tap/filtered tap drinking water samples for each cycle.

		Sample						
Measure	Statistic	1 (n = 70)	2 (n = 66)	3 (n = 68)	4 (n = 66)	5 (n = 52)	6 (n = 59)	Annual avg[a] (n = 38)
As, µg/L	% Detect	91.4%	87.9%	88.2%	86.4%	80.8%	84.7%	NA
(p = 0.0001)	Mean	0.82	0.83	0.84	0.74	0.59	0.76	0.60
MCL = 50 µg/L	SD	1.14	0.82	1.31	1.28	0.46	1.33	0.51
	Median	0.66	0.72	0.65	0.56	0.54	0.51	0.59
Cd, µg/L	% Detect	5.7%	6.1%	8.8%	10.6%	9.6%	8.5%	NA
(—)[b]	Mean	0.06	0.06	0.06	0.07	0.08	0.07	—
MCL = 5 µg/L	SD	0.02	0.03	0.04	0.05	0.07	0.06	—
	Median	0.05	0.05	0.05	0.05	0.05	0.05	—
Pb, µg/L	% Detect	91.4%	84.8%	89.7%	98.5%	92.3%	98.3%	NA
(p = 0.0001)	Mean	0.73	0.64	0.99	1.19	1.91	1.21	0.93
MCL = 5 µg/L	SD	0.92	0.88	2.33	2.16	3.43	1.36	1.21
	Median	0.36	0.24	0.24	0.41	0.72	0.66	0.57

Abbreviations: avg, average; % Detect, relative frequency of samples with detectable amount of analyte; MCL, maximum contaminant level; n, number of observations; NA, not applicable. p-Value for test of significant temporal variability of the log of the analyte concentrations is shown for each element. The MCL values are included for each element as a comparison value. [a]Computed from an average of six measurements for each respondent. [b]Statistical analysis not performed or summary statistic not computed because of censored data.

Table 3. Summary statistics for 2-min flush tap/filtered tap drinking water consumption and exposure to heavy metals via drinking water for each cycle.

		Sample						
Measure	Statistic	1 (n = 70)	2 (n = 66)	3 (n = 68)	4 (n = 66)	5 (n = 52)	6 (n = 59)	Annual avg[a] (n = 38)
Water	Mean	0.71	0.89	0.92	0.88	1.06	1.07	0.96
consumption,	SD	0.57	0.58	0.53	0.55	0.64	0.66	0.51
L/day	Median	0.66	0.85	0.95	0.89	1.12	1.10	1.00
(p = 0.0001)								
As intake,	Mean	0.64	0.80	0.87	0.75	0.69	0.93	0.55
µg/day	SD	1.30	1.22	2.05	2.19	1.16	2.36	0.49
(p = 0.0120)	Median	0.40	0.57	0.56	0.45	0.57	0.56	0.56
Cd intake,	Mean	0.04	0.05	0.06	0.06	0.08	0.07	—
µg/day	SD	0.03	0.03	0.04	0.05	0.07	0.06	—
(—)[b]	Median	0.03	0.04	0.05	0.04	0.06	0.05	—
Pb intake,	Mean	0.51	0.48	0.95	1.12	2.28	1.34	0.95
µg/day	SD	0.96	0.57	2.46	2.39	4.73	2.03	1.36
(p = 0.001)	Median	0.29	0.20	0.25	0.31	0.55	0.59	0.51

Abbreviations: avg, average; n, number of observations. p-Value for test of significant temporal variability of the log of the drinking water consumption and log of the heavy metal exposure is shown for each analyte.
[a]Computed from an average of six measurements from each respondent. [b]Statistical analysis not performed or summary statistic not computed because of censored data.

Table 4. Alternative measures of the reliability of short-term measures of exposure to metals in 2-min flush tap/filtered tap drinking water to assess long-term average exposure.

Measure	Concentration in drinking water (µg/L)		Drinking water intake (L/day)	Exposure via drinking water (µg/L)	
	As	Pb		As	Pb
CV[a]	0.20	0.76	0.40	0.45	0.90
Correlation[b]	0.90	0.55	0.69	0.82	0.59
R^c	0.88	0.50	0.67	0.81	0.55

Coefficient of variation (CV) results based on analysis of 381 samples obtained during a longitudinal survey of drinking water exposures to individuals in 73 residences in Maryland between September 1995 and September 1996.
[a]Mean within-person CV for the population. [b]Mean correlation coefficient among pairs of sampling cycles. [c]Intraclass correlation coefficient of reliability.

Discussion

Concentrations of As, Cd, and Pb in drinking water of the United States are typically < 10, 2, and 5 µg/L, respectively (*4,5,16*). Metal concentrations in the 381 water samples obtained from 73 members of the NHEXAS-MD population were within the ranges reported nationwide. The 31 water samples in this study with a Cd level greater than the DL were obtained from only 12 households. Observations of consistently measurable Cd concentrations at a residence could reflect leaching from plumbing materials (*17*). In future research, we will examine the potential covariates of metal concentrations in drinking water such as water source and age of the residential structure.

The results presented in this paper are based on measurements of total As, Cd, and Pb in the drinking water samples—we did not obtain information on valence states or compounds of the metals. Inorganic forms of As are more toxic than numerous organic As species (*9*). Inorganic forms of As account for essentially all of the As present in drinking water (*18*). Thus, the present findings for As could be generalized to exposure to inorganic As with some confidence.

Temporal variability of concentrations and exposures. A clear seasonal pattern for metal concentrations or exposures has not been established by previous studies; nevertheless temporal differences have been apparent. A 12-month investigation in Snohomish County, Washington, reported that As varied from 2- to 19-fold over time (*19*). Elevated values came from wells that previously did not contain high levels of As, suggesting a temporal variation in concentrations. A similar study found that monthly As concentrations varied between 0.20 and 0.60 ppm (*20*). A study in Oregon concluded that seasonal fluctuations in As levels of well water were inversely associated with rainfall (*21*). The degree to which these findings can be generalized to tap water drawn from surface water sources is difficult to assess because of the susceptibility of surface waters to nonpoint sources of pollution.

In the current study, mean log-transformed concentrations of As and Pb in drinking water samples and exposure to As and Pb by consumption of drinking water varied significantly across sampling cycles. To explore more fully this form of temporal variability, the mixed models described earlier were fit to the 228 observations obtained from the 38 subjects that participated in all six sampling cycles; i.e., a balanced data set. Analyses of the reduced data set also indicate significant variability among cycles of As and Pb concentrations in drinking water, drinking water consumption, and Pb exposure via drinking water. Thus, there is no indication that results from analyses of the unbalanced data set influenced the findings regarding temporal variability of these measures in a meaningful way. Mean log-transformed As exposure, however, did not vary significantly ($p = 0.3162$) among cycles in the reduced data set. Results from the reduced data set should be interpreted with caution. The reduced sample size increases the standard error estimates by a factor of nearly 1.5 over those for the full data set. The loss of power due to the reduction in sample size may be reflected in increased p-values for effects. Additionally, the remaining observations may not be representative of the full sample because of differential dropout rates with respect to certain variables. For example, the four participants that exhibited the 14 highest exposures in the full data set did not participate in all six cycles, suggesting that the removed data differ from those retained in a fundamental way. Finally, the mixed model approach attempts to account for the missing data in an unbalanced design using an optimum modeling strategy that includes relationships observed among the various units in the available data. It would not be surprising, therefore, if some parameter estimates differed between the full data set and the reduced data set.

Substantial within-person variability of exposure to copper in drinking water was reported for a cohort of children in Sweden (*22*). In that case, daily intake of copper varied several-fold among the 4 days on which data were obtained for each participant. We found a similar degree of intraindividual variability of exposure to Pb in the present study. For example, the SD of Pb exposures across cycles for an average person was 90% of their mean Pb exposure (Table 4), indicating that for many individuals in the NHEXAS-MD population a short-term measure of Pb exposure was at least 2-fold different than their long-term average exposure. In contrast, As concentrations in drinking water and exposures via drinking water were relatively constant across sampling cycles for individuals. This observation agrees with the findings from a study of As exposure for members of a community in Utah with As concentrations in drinking water ranging from 8 to 620 µg/L (*23*).

The results of this investigation suggest that exposure to As and Pb in drinking water varies significantly and perhaps substantively over time. However, the limited scope of the study requires that the findings be treated with caution. For example, it is difficult to generalize these findings to the sampling domain (i.e., metropolitan Baltimore and Annapolis, MD) because of limited representation of various subpopulations whose exposure could vary systematically from one and other. Similarly, the degree to which these findings can be generalized to other regions of the United States and other nations is not clear. Further, the longitudinal scope of the present study offers little information on the temporal aspects of exposure over periods of time shorter than 6–8 weeks and longer than 1 year.

Nevertheless, if concentrations of As and Pb in drinking water vary systematically by season for a population, then timing of data collection should be considered in exposure and risk assessments (*24*). Results of the NHEXAS-MD study suggest that the central tendency of As and Pb exposure via drinking water for a population may vary over time. However, the duration of the study was not sufficient to identify seasonal patterns. The value of additional information on systematic temporal differences in drinking water exposure to As and Pb should be evaluated in comparison to the cost of obtaining that information and the uncertainty about other inputs to the risk assessment procedure. Methods for performing value-of-information analyses are described elsewhere (*25*).

Implications of temporal variability. The temporal dimension of exposure to environmental contaminants is of interest because the nature and severity of a biologic response to an environmental challenge can be associated with the duration and timing of the exposure in addition to the magnitude of exposure. For example, the toxicologic effects of As and Pb, such as cancer and impaired cognitive function, are considered to result from chronic rather than short-term

exposure. Statistical models of the relationship between short-term and long-term exposures have been explored by several investigators (*26–29*). Additional analysis of repeated measure data sets is needed to identify appropriate methods for describing these relationships and incorporating them into research and regulatory efforts.

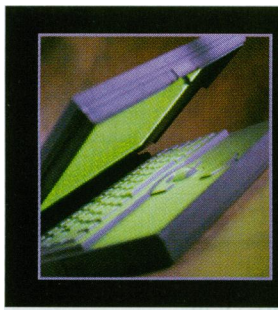
The findings from repeated measure studies have implications for tools such as epidemiology and quantitative risk assessment that are used to evaluate the potential effects of environmental contaminants on human health. The proportion of total variability in the observed data that can be apportioned to between-person and within-person variability is of particular relevance to epidemiology and risk assessment. We used the intraclass correlation coefficient of reliability (R) to apportion total variability in exposure measures between within-person and between-person variability. This approach is appropriate if the distribution of within-person errors is independent of long-term average exposure (*14*). Analysis of residuals from the GLMs described earlier demonstrated that the fundamental assumptions of the classic error model are met by this portion of the NHEXAS-MD data set. As described elsewhere (*30–32*), R can be used to evaluate the impacts of exposure measurement error due to temporal variability on *a*) misclassification in epidemiologic studies of disease and chronic environmental exposure; *b*) potential bias in estimates of the correlation coefficient, regression coefficient, relative risk, and other estimates of effect; and *c*) statistical power and sample size requirements.

Conclusions

Exposure assessment is a tool for assessing the potential harm a substance can elicit over time. Our results suggest that the timing of sample collection is a factor in measuring population exposure to As and Pb in drinking water. Repeated measures that span seasonal variation may provide more representative samples from which risk estimates and federal standards are derived. This work supports the hypothesis that observable within-person variability exists for As and Pb concentrations in drinking water, drinking water consumption, and As and Pb exposure. Intraindividual variability should be considered in the design, execution, and interpretation of environmental epidemiology studies of As and Pb. Additional research should be undertaken to improve understanding of temporal variability in exposure to As and Pb in drinking water and other potential exposure media. Such studies should include larger, more demographically diverse study populations and longer follow-up periods to allow for tests of periodicity of exposure.

References and Notes

1. Chappell WR, Beck BD, Brown KG, Chaney R, Cothern CR, Irgolic KJ, North DW, Thornton I, Tsongas TA. Inorganic arsenic: a need and an opportunity to improve risk assessment. Environ Health Perspect 105:1060–1067 (1997).
2. Mislin H, Ravera O. Cadmium in the Environment. New York:Princeton Architectural Press, 1986.
3. Gordon C, Hutchinson T. Global Perspective on Lead, Mercury and Cadmium Cycling in the Environment. Elkins Park, PA:Franklin Book Company, 1994.
4. Elinder C. Cadmium: uses, occurrence, and intake. In: Cadmium and Health: A Toxicological and Epidemiological Appraisal, Vol 1 (Friberg L, Elinder C, Kjellstrom T, Nordberg G, eds). Boca Raton, FL:CRC Press, 1985;23–64.
5. Millstone E. Lead and Public Health: The Dangers for Children. Washington, DC:Taylor & Francis, 1997;97–100.
6. Borum D, Abernathy C. Human oral exposure to inorganic arsenic. In: Arsenic Exposure and Health (Chappell WR, Abernathy CO, Cothern CR, eds). Environmental Geochemistry and Health, Vol 16. Northwood, UK:Science and Technology Letters, 1994;21–30.
7. Nriagu J. Cadmium in the Environment: Health Effects: Pt. II. Melbourne, Australia:Krieger Publishing, 1986.
8. Mushak P, Crocetti A. The Nature and Extent of Lead Poisoning in Children of the U.S: A Report to Congress. Upland, CA:DIANE Publishing, 1996.
9. Abernathy CO, Liu Y-P, Longfellow D, Aposhian HV, Beck B, Fowler B, Goyer R, Menzer R, Rossman T, Thompson C, et al. Arsenic: health effects, mechanisms of actions, and research issues. Environ Health Perspect 107:593–597 (1999).
10. MacIntosh D, Needham L, Hammerstrom K, Ryan PB. A longitudinal investigation of selected pesticide metabolites in urine. J Expo Anal Environ Epidemiol 9:494–501 (1999).
11. MacIntosh D, Kabiru C, Scanlon K, Ryan PB. Longitudinal exposure to arsenic, cadmium, chromium and lead via beverage consumption. J Expo Anal Environ Epidemiol 10(2):196–200 (2000).
12. Long S, Martin T. EPA method 200.8: determination of trace elements in waters and wastes by inductively coupled plasma - mass spectrometry. In: Methods for the Determination of Metals in Environmental Samples. Cincinnati, OH:U.S. Environmental Protection Agency, 1991.
13. Scanlon K, MacIntosh D, Hammerstrom K, Ryan PB. A longitudinal investigation of dietary exposure to selected elements. J Expo Anal Environ Epidemiol 9:485–493 (1999).
14. Fleiss JL. Reliability of measurement. In: The Design and Analysis of Clinical Experiments. New York:John Wiley & Sons, 1986;1–32.
15. U.S. EPA. Exposure Factors Handbook. EPA/600/P-95/002Fa. Washington, DC:U.S. Environmental Protection Agency, 1997.
16. Davis M, Reich K, Tikkanen M. Nationwide and California arsenic occurrence studies. In: Arsenic Exposure and Health (Chappell WR, Abernathy CO, Cothern CR, eds). Environmental Geochemistry and Health, Vol 16. Northwood, UK:Science and Technology Letters, 1994;31–49.
17. Subramanian K, Connor J, Meranger J. Leaching of antimony, cadmium, copper, lead, silver, tin and zinc from copper piping with non-lead-based soldered joints. J Environ Sci Health Part A 26:911–929 (1991).
18. Irgolic K. Determination of total arsenic and arsenic compounds in drinking water. In: Arsenic Exposure and Health (Chappell WR, Abernathy CO, Cothern CR, eds). Environmental Geochemistry and Health, Vol 16. Northwood, UK:Science and Technology Letters 1994;51–60.
19. Frost F, Frank D, Pierson K, Woodruff L, Raasina B, Davis R, Davies J. A seasonal study of arsenic in groundwater, Snohomish County, Washington, USA. Environ Geochem Health 15:209–213 (1993).
20. Cebrian M, Albores A, Aguilar M, Blakely E. Chronic arsenic poisoning in the north of Mexico. Hum Toxicol 2:121–133 (1983).
21. Nadakavukaren J, Ingermann R, Jeddeloh G, Falkowski S. Seasonal variation of arsenic concentration in well water in Lane County, Oregon. Bull Environ Contam Toxicol 33:264–269 (1984).
22. Pettersson R, Rasmussen F. Daily intake of copper from drinking water among young children in Sweden. Environ Health Perspect 107:441–446 (1999).
23. Calderon RL, Hudgens E, Le XC, Schreinemachers D, Thomas DJ. Excretion of arsenic in urine as a function of exposure to arsenic in drinking water. Environ Health Perspect 107:663–667 (1999).
24. National Research Council. Variability. In: Science and Judgement in Risk Assessment (Council NR, ed). Washington, DC:National Academy Press, 1994;188–223.
25. Thompson K, Evans J. The value of improved national exposure information for perchloroethylene (PERC)—a case study for dry cleaners. Risk Anal 17:253–271 (1997).
26. Slob W. Modeling long-term exposure of the whole population to chemicals in food. Risk Anal 13:525–530 (1993).
27. Wallace LA, Duan NH, Ziegenfus R. Can long-term exposure distributions be predicted from short-term measurements. Risk Anal 14:75–85 (1994).
28. Buck R, Hammerstrom K, Ryan PB. Estimating long-term exposures from short-term measurements. J Expo Anal Environ Epidemiol 5:359–374 (1995).
29. Buck RJ, Hammerstrom K, Ryan PB. Bias in population estimates of long-term exposure from short-term measurements of individual exposure. Risk Anal 17:455–466 (1997).
30. Ozkaynak H, Ryan PB, Spengler JD, Laird NM. Bias due to misclassification of personal exposures in epidemiologic studies of indoor and outdoor air pollution. Environ Int 12:389–393 (1986).
31. Shy C, Kleinbaum D, Morgenstern H. The effect of misclassification of exposure status in epidemiological studies of air pollution. Bull New York Acad Med 54:1155–1165 (1979).
32. Hatch M, Thomas D. Measurement issues in environmental epidemiology. Environ Health Perspect 101(suppl 4):49–57 (1993).



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