

Arsenic, As^(III), and tungsten in Nevada County's private water supplies

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ABSTRACT

In parts of the western United States groundwater used for drinking water contains high concentrations of metals, including arsenic. In a rural county in Nevada, USA, we measured concentrations of arsenic and tungsten and the proportion of arsenic occurring in trivalent form (As^(III)) in tap water samples from private domestic wells in 307 households. The proportion of arsenic occurring as As^(III) ranged from 0 to 100% (ave. 21%, median 1%). Tungsten concentrations ranged from 0 to 610 $\mu\text{g l}^{-1}$ (ave. 26 $\mu\text{g l}^{-1}$, median 2 $\mu\text{g l}^{-1}$). Among 253 respondents who consumed water: (a) 177/253 (70%) of tap water samples contained more than 10 $\mu\text{g l}^{-1}$ total inorganic arsenic (ave. 66 $\mu\text{g l}^{-1}$, median 20 $\mu\text{g l}^{-1}$); (b) As^(III) occurred as a small proportion of total arsenic in most samples (ave. 22%, median 3%); and (c) tungsten occurred in concentrations ranging from below the detection limit (3 $\mu\text{g l}^{-1}$) to a maximum of 610 $\mu\text{g l}^{-1}$ (ave. 30 $\mu\text{g l}^{-1}$, median 3 $\mu\text{g l}^{-1}$). Log₁₀ concentrations of tungsten and total arsenic in consumed water were positively correlated ($\log_{10}[W] = -0.400 + 0.703(\log_{10}[\text{As}^T])$, $p = 0.000+$, adj. $r^2 = 0.53$). This suggests that householders in this area were likely to be exposed to both metals simultaneously, given that 253/307 of the respondents (82%) reported consuming tap water.

Key words | arsenic, groundwater, private wells, public health, tungsten

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INTRODUCTION

Approximately 15% of the population of the United States is served by private drinking water supplies; the remainder is served by public supplies (USEPA 2004). Private domestic wells are unregulated drinking water sources that usually serve a single household. Standards for chemical, microbiological and physical characteristics of water codified in the Safe Drinking Water Act and amendments apply to public water supplies and are not enforceable for private water supplies.

Arsenic in US groundwater

The current standard for arsenic (10 $\mu\text{g l}^{-1}$) became enforceable for public water supplies in 2006. High arsenic concentrations in groundwater (>10 $\mu\text{g l}^{-1}$) have been documented in many areas of the United States

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(Welch *et al.* 2000). No estimates exist of the proportion of private water supplies with arsenic concentrations that exceed 10 $\mu\text{g l}^{-1}$. Although arsenic can occur in methylated forms and in several valence states, the primary form of arsenic in groundwater is considered to be inorganic (Bissen & Frimmel 2003), in the most oxidized, pentavalent state (As^(V)) and in the reduced trivalent state (As^(III)) (NRC 1999).

Significance of trivalent arsenic (As^(III))

The valence form of arsenic may be significant with respect to health effects; As^(III) is 2–3 times more acutely toxic than As^(V) (ATSDR 2000; Hughes 2002). Research with cultured human cells (skin, urinary bladder and lung) demonstrated that 24 hour exposure to inorganic As^(III)

had significant effects on cellular ability to methylate As^(III) at concentrations of approximately 300, 749 and 1,498 $\mu\text{g l}^{-1}$ (Styblo *et al.* 2000). A review of arsenic carcinogenicity reported results of lifetime duration experiments with laboratory animals exposed to high concentrations of arsenite (As^(III)) (5,000 $\mu\text{g l}^{-1}$) administered in drinking water, with no carcinogenic effect (Hughes 2002). Although concentrations in the highest doses used for these experiments have been observed in groundwater in Churchill County, Nevada (e.g. 2,100 $\mu\text{g l}^{-1}$) (Walker *et al.* 2005), the treatments at which effects were reported ($\geq 300 \mu\text{g l}^{-1}$) represent unusually high concentrations of arsenic in groundwater.

The effects of chronic exposure among humans to As^(III) have not been established with any statistical significance because, in retrospective studies from which correlations between human health effects and exposure are drawn, the valence form of arsenic to which people have been exposed is often unclear (ATSDR 2000). It is thought that if concentrations of As^(III) are high, or methylation of As in the body is not complete, ingested inorganic As^(III) may react with and be retained by tissues for longer periods of time than might be expected for equivalent concentrations of methylated forms of arsenic (NRC 1999).

The valence form of inorganic arsenic affects treatment efficiency, especially in the types of system commonly used for small-volume household applications, including reverse osmosis. Removal efficiency with reverse osmosis units has been shown to be lower when arsenic is present as As^(III) (Chen *et al.* 1999; USEPA 2000; Walker *et al.* 2008).

Significance of tungsten in drinking water

Consumption of dissolved tungsten has not been studied extensively, but it is not considered a cause of adverse health effects as a result of chronic exposure through water (ATSDR 2000). However, the US Centers for Disease Control urged in 2002 that tungsten be studied by the National Toxicology Program of the National Institutes of Health (currently under consideration, see National Toxicology Program website: http://ntp-server.niehs.nih.gov/ntp/htdocs/Chem_Background/ExSumPdf/tungsten.pdf, accessed November 2008). Tungsten appears to pass through the body quickly, though it may be retained in

high concentrations in slow-growing tissues such as bone (Koutsospyros *et al.* 2006). Although it may have an acute effect at high concentrations ($> 1,500,000 \mu\text{g l}^{-1}$ (Marquet *et al.* 1996, 1997)), consequences of chronic exposure to low concentrations (in the $\mu\text{g l}^{-1}$ range) are unclear. Following single exposures to sodium tungstate dehydrate, concentrations of tungsten in internal organ and bone tissues initially rose in experimental rats and mice, but were substantially reduced after 24 h (McDonald *et al.* 2007). The lowest doses administered orally (1 mg per kg animal body weight) were the equivalent of dissolved tungsten concentrations of approximately 35,000 $\mu\text{g l}^{-1}$, based on the assumption of a 70 kg human body mass and 2 litre consumption of water per day. Thus, the experiments represent doses that are far higher than would be expected in groundwater, but well below a concentration ($> 1,500,000 \mu\text{g l}^{-1}$) that was observed to lead to acute health effects (Marquet *et al.* 1996).

Compilations of urine sampling results reported that the geometric means of samples taken from the US population more than 6 years of age were 0.093 $\mu\text{g l}^{-1}$ and 0.082 $\mu\text{g l}^{-1}$ ($n = 2,338$ in 1999–2000, 95% confidence interval: 0.087–0.100 $\mu\text{g l}^{-1}$; $n = 2,652$ in 2001–2002, 95% CI: 0.073–0.092, respectively) (CDC 2005). In some locations the reported concentrations were considerably higher. For example, the lower bound of the 95% confidence interval for the geometric mean concentration of tungsten in urine samples from subjects in Churchill County, Nevada (1.19 $\mu\text{g l}^{-1}$, 95% CI: 0.89–1.59 $\mu\text{g l}^{-1}$), exceeded the upper bounds of 95% CI reported for the 95th percentiles from the 1999–2000 and 2001–2002 sampling results (0.550, 0.560 $\mu\text{g l}^{-1}$, respectively) (Rubin *et al.* 2007).

Subdermally implanted tungsten/nickel/cobalt alloy (used as a projectile for weapons) led to metastatic rhabdomyosarcoma in rats (Kalinich *et al.* 2005). Sheppard *et al.* (2006) discussed the possibility of a link between exposure to tungsten via inhalation of airborne particulates and cases of acute lymphoblastic leukemia in children in Churchill County, Nevada. Debate about research related to airborne tungsten (Seiler *et al.* 2006; Sheppard *et al.* 2006) suggested that the hypothesis merited rigorous testing (Kalinich 2005; Sheppard *et al.* 2006). However, the significance of these results is unclear with respect to chronic ingestion of dissolved tungsten.

Churchill County, Nevada

The recent occurrence of a childhood leukemia cluster in Churchill County (CDC 2003) prompted extensive investigation of groundwater quality to identify naturally occurring and anthropogenic chemicals that could be causes (Seiler 2004; Seiler *et al.* 2005). Churchill County is the terminus of the Carson River, which flows from the Sierra Mountains in California to the Carson Sink. The Carson Sink is physiographically typical of the basin and range province in the Great Basin, bounded by southwest to northeast trending mountain ranges with broad expanses of relatively flat valley floor.

The concentrations of arsenic in the county have been reported in newspapers, public meetings and radio and television broadcasts (Benson 2003) and many homeowners with private wells as water sources have installed systems to treat drinking water. A recent study showed that 33% of respondents were highly concerned about arsenic and had installed household RO systems (Walker *et al.* 2006). All drinking water in Churchill County is from groundwater.

Previous studies of occurrence of arsenic indicate that concentrations increase along the groundwater flow path from the western margins of the basin to the eastern margins. However, arsenic concentrations vary substantially even in wells in close proximity (Fitzgerald 2004). Several sampling surveys found that concentrations of arsenic in groundwater often exceeded the 10 µg l⁻¹ maximum contaminant level set by the Safe Drinking Water Act for drinking water (Fitzgerald 2004). Tungsten in groundwater is thought to be from natural sources such as geothermal waters and tungsten-bearing sediments in subsurface and surface formations (Seiler *et al.* 2005).

Four aquifers are found in the county, including a basalt aquifer and three sedimentary aquifers. Households in rural areas obtained drinking water from private wells in basin-fill sedimentary aquifers (Lico & Seiler 1994). The three sedimentary aquifer systems (of fluvial, pluvial, alluvial, eolian and lacustrine origin) are designated as shallow (0–50 ft [0–15 m]), intermediate (50–500 ft [15–152 m]) and deep (500 + ft [152 + m]) (Lico & Seiler 1994). The primary basis for these designations is water chemistry,

particularly water hardness (Glancy 1986). The principal source of recharge to the shallow basin-fill aquifers is infiltration of surface water from irrigation, supplied by diversion from the Carson and Truckee Rivers.

Soils in the Great Basin are highly alkaline and often covered with a crust of evapoconcentrated salts and minerals. A compilation and analysis of water chemistry information from western Nevada reported that groundwater in the region had variable but in many cases high concentrations of arsenic (with an average of 79 µg l⁻¹ from 356 wells (Thundiyil *et al.* 2007)).

The Great Basin includes much of northern Nevada and portions of western Utah and southeastern Oregon. In 2006, the US Census Bureau reported that approximately 25% of Nevada's population (total: 2,495,529 people) lived in the 12 counties that are part of the Great Basin, with as few as 1,480 people in Eureka County and 396,428 people in Washoe County (www.census.gov, last accessed August 2008). The overall growth rate in these 12 counties (7.25%) was lower than that reported for the state (24.9%) and populations of some of the counties decreased between 2000 and 2006 (www.census.gov).

Purpose of study

This study explored the levels of tungsten and the proportions of arsenic occurring as As^(III) in household tap water from private wells in Churchill County, Nevada, and the potential for simultaneous exposure to both through household tap water. Groundwater in the area contains naturally occurring arsenic concentrations, which commonly exceed 100 µg l⁻¹ in groundwater (Fitzgerald 2004; Seiler 2004). In 2001, arsenic concentrations in 89 of 100 wells sampled in Churchill County exceeded 10 µg l⁻¹ (Seiler 2004). The median arsenic concentration in eight municipal wells sampled in the valley was 103 µg l⁻¹ (range 90–110 µg l⁻¹) and was 61 µg l⁻¹ (range 2–2,910 µg l⁻¹) in the 92 domestic wells. Arsenic concentrations exceeding 1,000 µg l⁻¹ have been documented by several authors (Welch *et al.* 2000; Seiler 2004; Shaw *et al.* 2005; Walker *et al.* 2006), and a child in a household served by a well with 2,750 µg l⁻¹ of arsenic was reported to have been poisoned in the late 1960s (Donahue 2001).

METHODS

Population of study area

Approximately 24,000 people resided in the county when this study was carried out (2002) and an estimated 11,500 were served by 16 public water supplies. The study area was approximately 225 square miles (583 km²) and excluded the service districts of public water supplies, the largest of which served the city of Fallon (the county seat) (Figure 1). In 2002 an estimated 5,500 households (approximately 10,750 people), relied on private domestic water supplies (personal communication, Churchill County Planning Department, 2002).

Sampling survey design

The study focused on 307 households, which were a subset of 351 households included in a 2002 sampling survey (Benson 2003; Shaw *et al.* 2005; Walker *et al.* 2005). A surveyor visited each household, administered a questionnaire and collected a tap water sample, as described

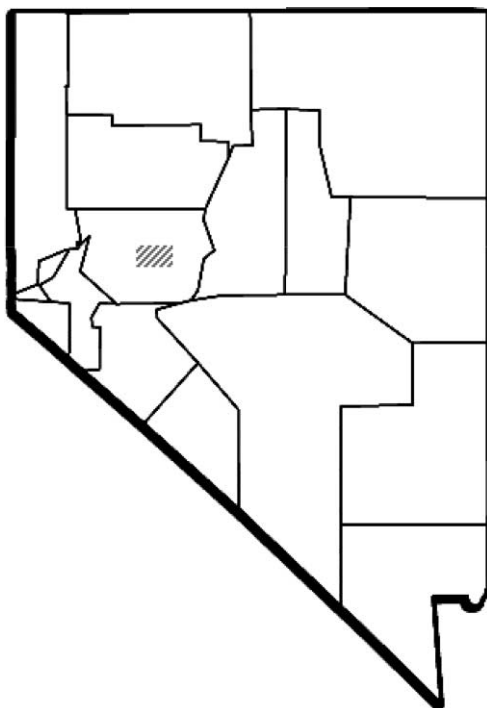


Figure 1 | Location of the study area in Churchill County, Nevada, USA.

below. The 307 household subset of the 351 households were further analyzed for As^(III) and W. Households that participated in the survey, including the 307 from which these samples were taken, were recruited by direct solicitation by a research team and by flyers directly distributed to businesses in Churchill County and homes. The recruitment technique was chosen to achieve a balanced spatial distribution of sampling points in the county and to represent county socio-economic characteristics as best as possible (Benson 2003). Approximately 31.6% of the participants responded to flyers left in their mailboxes and the remainder were recruited using direct intercept by a survey recruiter. The intercept survey approach is commonly used to administer surveys about recreational use and has advantages and disadvantages compared with other sampling methods (Krysan *et al.* 1994). As in all household surveys where participation is voluntary, the final results cannot be assumed to represent households that did not participate. As a check of the representativeness of the survey, we compared sample statistics (e.g. age distribution, home ownership, income and education levels) with those obtained by the US Census Bureau in 2000 and 2004. Discrepancies between the sample and the county included a potential bias in median income (\$50,000/year (sample) vs. \$46,000/year (for 2004, from US Bureau of Census <http://quickfacts.census.gov>, accessed January 2008)), age (median age (years) 58 (sample) vs. 35 (population)), proportion of homeowners/renters (91%/9% vs. 66%/34% (sample, population respectively)), and sex ratio (M/F) (41%/59% vs. 50%/50% (sample, population respectively)) (Walker *et al.* 2006).

Sampling protocols and sample preservation

All 307 samples collected were analyzed for total arsenic, tungsten and trivalent arsenic (As^(III)). Tap water samples were collected from the point of most frequent use identified by the respondent—in most cases a kitchen tap. Collection involved minimal purging (approximately five seconds of flow prior to collection). All samples were unfiltered. Comparison of arsenic concentrations in filtered and unfiltered samples of groundwater in Churchill

Table 1 | Results of 307 tap water sample analyses for tungsten from Churchill County, Nevada, USA

| Quartile | Concentrations, all samples ($\mu\text{g l}^{-1}$) | Concentrations, consumed water ($\mu\text{g l}^{-1}$) | Concentrations, treated water ($\mu\text{g l}^{-1}$) | Concentrations, untreated water ($\mu\text{g l}^{-1}$) |
|----------|---|--|---|---|
| 1st | ND* | ND* | ND* | 1 |
| Median | 2 | 2 | 1 | 4 |
| 3rd | 11 | 9 | 4 | 17 |
| Maximum | 610 | 510 | 510 | 610 |
| Average | 26 | 11 | 12 | 34 |
| <i>n</i> | 307 | 253 | 114 | 193 |

*ND: not detected, with detection limit of $1 \mu\text{g l}^{-1}$.

County indicated no significant differences in concentrations (Seiler 2004).

Analyses were carried out by the Nevada State Health Laboratory, a certified drinking water analysis facility. Arsenic concentrations were determined by EPA method 200.8 (ICP-MS) for samples with turbidity less than 1 NTU and ASTM method D2972-93B (hydride generation AA) for samples with turbidity greater than or equal to 1 NTU (limit of detection for both methods = $3 \mu\text{g l}^{-1}$). Tungsten analyses were carried out using USEPA method 200.8 (ICP-MS, Methods for the Determination of Metals in Environmental Samples - Supplement 1, EPA-600/R-94-111, May 1994; limit of detection = $1 \mu\text{g l}^{-1}$).

The proportion of arsenic occurring as As^(III) was determined using atomic absorption (Varian Spectra AA) with graphite furnace. Samples were preserved upon collection with 0.7 ml EDTA/100 ml sample in opaque HDPE bottles (Bednar *et al.* 2002). We separated As^(III) from As^(V) and other inorganic arsenic species using an anion exchange resin (Supelco Supelclean LC-SAX SPE; Sigma-Aldrich cat# 57203) (Bednar *et al.* 2002). Sample analyses included one

analytic blank per 10 samples, with one duplicate and one standard ($20 \mu\text{g l}^{-1}$ As) per 20 samples. As a further quality assurance check, we used atomic absorption to determine the concentration of As^(T) in each sample, to compare against results obtained from the Nevada State Health Laboratory, which we treated as a standard. If comparison showed greater than 20% difference, we examined data produced in our laboratory and, if necessary, re-analyzed samples. In three cases, results remained outside of the 20% difference criterion and these samples were excluded from the results reported in this paper.

RESULTS AND DISCUSSION

The distributions of sampling results for arsenic, tungsten and the proportion of arsenic occurring as As^(III) are reported in Tables 1, 2 and 3. The results are reported for all tap water samples ($n = 307$), and for subsets of these results according to whether tap water was consumed in the household ($n = 253$). Consumption was defined as drinking

Table 2 | Results of 307 tap water sample analyses for arsenic from Churchill County, Nevada (detection limit of $3 \mu\text{g l}^{-1}$)

| Quartile | Concentrations, all samples ($\mu\text{g l}^{-1}$) | Concentrations, consumed water ($\mu\text{g l}^{-1}$) | Concentrations, treated water ($\mu\text{g l}^{-1}$) | Concentrations, untreated water ($\mu\text{g l}^{-1}$) |
|----------|---|--|---|---|
| 1st | 8 | 7 | 3 | 16 |
| Median | 27 | 22 | 13 | 38 |
| 3rd | 93 | 72 | 47 | 128 |
| Maximum | 2,196 | 909 | 844 | 2,196 |
| Average | 93 | 69 | 62 | 112 |
| <i>n</i> | 307 | 253 | 114 | 193 |

Table 3 | Results of 307 tap water sampling for [As^(III)]/[As^(T)] in Churchill County, Nevada

| Quartile | Proportion of As ^(III) , all samples | Proportion of As ^(III) , consumed water | Proportion of As ^(III) , treated water ($\mu\text{g l}^{-1}$) | Proportion of As ^(III) , untreated water ($\mu\text{g l}^{-1}$) |
|----------|--|---|---|---|
| 1st | ND* | ND* | ND* | ND* |
| Median | 0.01 | ND* | 0.04 | ND* |
| 3rd | 0.33 | 0.29 | 0.50 | 0.28 |
| Maximum | 1 | 1 | 1 | 1 |
| Average | 0.20 | 0.19 | 0.25 | 0.18 |
| <i>n</i> | 307 | 253 | 114 | 193 |

*ND: Not detected, at detection limit of $3 \mu\text{g l}^{-1}$.

water or using it to mix beverages (Benson 2003). The quantity consumed was not considered. Treatments applied included ion exchange, carbon filtration, filtration, reverse osmosis, distillation and greensand filtration (Benson 2003).

The distribution of tap water sampling results for tungsten and arsenic has a pronounced right skew, with maximum concentrations of 610 and $2,196 \mu\text{g l}^{-1}$, respectively. The distribution of concentrations of arsenic in tap water samples suggests a high potential (217/307 wells in this study) for those who depend upon private wells in this county to have concentrations of arsenic that exceed the $10 \mu\text{g l}^{-1}$ standard for public water supplies, as previously reported (Shaw *et al.* 2005; Steinmaus *et al.* 2005; Walker *et al.* 2006). The distributions of tungsten results are similar to those obtained for arsenic.

The distribution of proportions of arsenic occurring as As^(III) is also right skewed, with a range of 0–100% for both all and consumed tap water samples. A factor found significant in predicting the proportion of As^(III) in Churchill County was the aquifer used for private water supply (Fitzgerald 2004). This is because of the reducing conditions in alluvium in the water-bearing formations supplying wells finished 50–500 feet (15–150 m) below land surface. In a comparison between intermediate and shallow aquifers carried out in a separate study, 51 wells of intermediate depth had a median proportion of 0.40 [As^(III)]/[total As] in water samples, as opposed to a median of 0.04 for shallow wells (Fitzgerald 2004). Figure 2 displays the average proportions of [As^(III)]/[As^(T)] of classes of arsenic concentrations found in tap water samples, for all samples and for households in which survey respondents

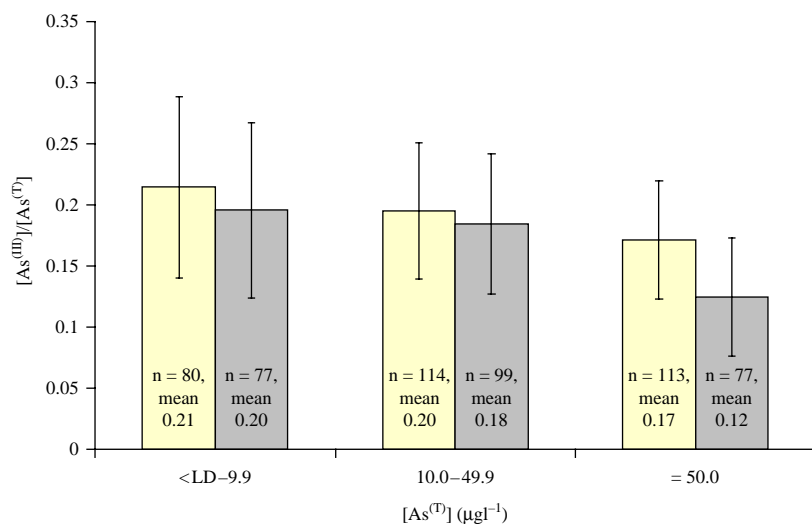


Figure 2 | Mean proportions of As^(III) in all tap water samples (open bars) and samples taken from households that reported consuming tap water (filled bars), by class of total arsenic concentration [As^(T)] with 95% confidence intervals for the mean proportion, labelled with numbers of samples in each class and mean proportions of [As^(III)]/[As^(T)].

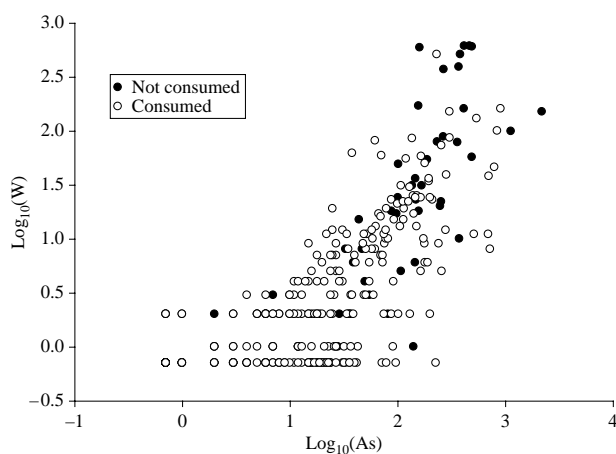


Figure 3 | Tungsten and arsenic (As^(T)) concentrations in samples of tap water reported to be consumed or not consumed.

reported consuming the water. This suggests a potential for exposure to As^(III), though the significance of exposure, especially in tap water that contained $< 10 \mu\text{g l}^{-1}$, is unclear.

Concentrations of total As and W in all tap water samples were correlated on a \log_{10} scale (Pearson correlation coefficient = 0.743, $p = 0.000 +$; Figure 3). A linear model of the relationship between $\log_{10}[\text{As}]$ and $\log_{10}[\text{W}]$ derived by regression (Equation 1) was significant ($p = 0.000 +$), and explained slightly more than half of the variability in the data (indicated by an adjusted r^2 value of 55%). The coefficient for arsenic concentrations was significant at $p = 0.000 +$:

$$\log_{10}[\text{W}] = -0.407 + 0.707 \log_{10}[\text{As}^{(\text{T})}] \quad (1)$$

The relationship between $\log_{10}[\text{W}]$ and $\log_{10}[\text{As}^{(\text{T})}]$ (Equation 2) for consumed tap water was similar, with an adjusted r^2 value of 53%, and significant at $p = 0.000 +$:

$$\log_{10}[\text{W}] = -0.400 + 0.703 \log_{10}[\text{As}^{(\text{T})}] \quad (2)$$

The regressions indicate that concentrations of tungsten and arsenic will be positively correlated in tap water and that this relationship is nearly identical in households that reported consuming tap water.

CONCLUSIONS

Households that rely on private water supplies are responsible for checking and maintaining the quality of

tap water. Households served by this type of unregulated supply represent a minority of the US population, though in some areas a majority of the population may rely on such water sources. Because private water supplies are unregulated and untested very little information exists about the occurrence and concentration of contaminants and potential exposure to contaminants by those who consume water. This sampling survey was designed to assess levels of arsenic, tungsten and the proportion of arsenic occurring as As^(III) in tap water samples taken from residents who rely on private wells in a rural county in Nevada. It also examined potential exposure to both arsenic and tungsten within households that reported consuming water. Among the sampled population, concentrations of arsenic in tap water were very high, relative to the $10 \mu\text{g l}^{-1}$ standard set for public water, as previously reported (Shaw *et al.* 2005; Walker *et al.* 2006). Samples from 176 of the wells that provided drinking water for 253 households had concentrations of arsenic exceeding $10 \mu\text{g l}^{-1}$.

The proportions and concentrations of As^(III) were low compared with health effects studies carried out with experimental animals. Inefficiencies associated with valence may explain why distributions of $[\text{As}^{(\text{III})}]/[\text{As}^{\text{T}}]$ were higher for treated tap samples than for those from untreated sources on a quartile by quartile basis because of lower removal efficiencies of As^(III) relative to As^(V). However, the difference between average $[\text{As}^{(\text{III})}]/[\text{As}^{\text{T}}]$ for treated and untreated supplies was not statistically significant at $p = 0.089$. Walker *et al.* (2008) noted that when $[\text{As}^{(\text{III})}]/[\text{As}^{\text{T}}]$ exceeded 50%, reverse osmosis units in Churchill County were likely to be less than 50% efficient in reducing the concentration of arsenic in groundwater.

The model of the relationship between arsenic and tungsten concentrations (Equation 1) suggests that when arsenic concentrations in tap water samples are high, tungsten concentrations will also be high. Equation (2) demonstrates that the relationship is the same for tap water that is consumed. The correspondence between arsenic and tungsten is expected, based on previously observed associations between dissolved arsenic and tungsten concentrations in groundwater (Grimes *et al.* 1995) and ore deposition patterns (Kesler *et al.* 2003) in the Great Basin region of the United States. It also corresponds closely with observed correlations between As and W in groundwater in

this area (Seiler *et al.* 2005). Seiler *et al.* (2005) reported a correlation coefficient of 0.63 between As and W, based on analysis of 100 samples. The investigators concluded that similar depositional processes and geochemical conditions affected release and mobility of W and As in groundwater flow systems, reflected in the high degree of correlation.

Tungsten concentrations cannot be compared with a drinking water standard because tungsten is not a regulated contaminant. The high concentrations of tungsten observed in urine samples from Churchill County reported by Rubin *et al.* (2007) may reflect exposure to dissolved tungsten through several routes, including airborne particles and tap water. Concern about tungsten is recent and the significance of exposure through ingestion of dissolved tungsten is not clear. Ongoing debate (Seiler *et al.* 2006; Sheppard *et al.* 2006) about the effects of tungsten exposure in this county has focused on the potential link between inhalation of tungsten as a particulate and childhood leukemia. Others have suggested that groundwater be considered as a route of exposure as well (Koutsospyros *et al.* 2006). If review of the toxicology of tungsten provides grounds for further concern, it is worth noting that, among private well users in the US Great Basin, exposure levels may be high and positively correlated with arsenic concentrations that are well above 10 $\mu\text{g l}^{-1}$ in a majority of households that consume water from private domestic wells.

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