Characterizing Waterborne Lead in Private Water Systems

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ABSTRACT

Lead is a common additive in plumbing components despite its known adverse health effects. Recent research has attributed cases of elevated blood lead levels in children and even fetal death with the consumption of drinking water containing high levels of lead. Although the federal Environmental Protection Agency (USEPA) strives to minimize lead exposure from water utilities through the Lead and Copper Rule (LCR), an estimated 47 million U.S. residents reliant on private unregulated water systems (generally individual and rural) are not protected. Detection, evaluation, and mitigation of lead in private systems is challenging due to lack of monitoring data, appropriate sampling protocols, and entities to fund research.

Through a statewide sampling survey, over 2,000 homeowners submitted water samples for analysis. This survey documented that 19% of households had lead concentrations in the first draw sample (i.e., 250 mL sample collected after 6+ hours of stagnation) above the EPA action level of 15 µg/L, with concentrations as high as 24,740 µg/L. Due to the high incidence observed, this research focused on identifying system and household characteristics that increased a homeowner's susceptibility of lead in water. However, 1% of households had elevated lead concentrations after flushing for five minutes, which highlighted potential sources of lead release beyond the faucet. Therefore, a follow-up study was conducted to investigate sources and locations of lead release throughout the entire plumbing network. Using profiling techniques (i.e., sequential and time series sampling), three patterns of waterborne lead release were identified: no elevated lead or lead elevated in the first draw of water only (Type I), erratic spikes of particulate lead mobilized from plumbing during periods of water use (Type II), and sustained detectable lead concentrations (>1 µg/L) even with extensive flushing (Type III). Lastly, emphasis was given to understand potential lead leaching from NSF Standard 61 Section 9 certified "lead-free" plumbing components as the synthetic test water is not representative of water quality observed in private water systems. Overall, this dissertation research provides insight into a population that is outside the jurisdiction of many federal agencies.

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AUTHOR'S PREFACE

This dissertation comprises five separate manuscripts (Chapters 2 through 6) bracketed by an introductory chapter and a final chapter of concluding remarks and recommendations for future research. Chapters 2 through 5 are in journal format, while Chapter 6 comprises additional interesting household survey data that may yield publications in the future, and are worthy of mention here. The student is the primary author for the four articles.

Chapter 2 is the first identified investigation of corrosion in private systems in Virginia. Through the Virginia Household Water Quality Program (VAHWQP), a Virginia Cooperative Extension program based at Virginia Tech, over 2,000 homeowners submitted drinking water samples for analysis. This effort aimed to quantify waterborne lead concentrations and examine system and household characteristics that increased the likelihood of lead in water, which can be communicated to homeowners to encourage testing. In addition, this study has provided substantial justification for further investigations, as approximately 1 in 5 homeowners in Virginia provided water samples with lead concentrations in excess of the USEPA recommended action level of 15 µg/L. Chapter 1 has been accepted for publication in the *Journal of Water and Health*.

Chapter 3 examines the influence of geology and system construction methods on private water quality. Although previous literature has evaluated geology or system construction, this statistical analysis simultaneously evaluated both variables, which are often interdependent (e.g. certain systems designs are more common and/or appropriate for specific underlying geologic regions). This investigation highlights the importance of a holistic approach to private water systems. Chapter 2 is currently under review for publication in the *Journal of Environmental Health* (submitted, March 2015).

Chapter 4 assesses patterns of waterborne lead release in private water systems. A profiling technique was developed specifically tailored to private systems and tested in 15 private systems from Albemarle County, VA. The results of this effort identified three patterns of waterborne lead exposure specific private systems; perhaps most interestingly and unexpected, sustained detectable lead concentrations were observed in five systems, which was attributed to corrosion of components in the well. Chapter 4 is currently under preparation for submission to *Environmental Science and Technology (ES & T).*

Chapter 5 evaluates NSF certified "lead-free" plumbing under aggressive (i.e. low pH) water conditions representative of private water systems. NSF International (NSF) Standard 61evaluates and certifies lead leaching potentials from lead-free plumbing components, but the synthetic water used to assess lead leaching is nonaggressive (i.e. non-corrosive, high pH) and may not be representative of water quality in private systems, where pH often is below the USEPA recommended minimum of 6.5. This study illustrated that lead-free plumbing components may release significant lead concentrations under aggressive water conditions (50-700 µg/L) and highlights potential gaps in the NSF protocol. Chapter 5 is currently under preparation for submission to *Journal American Water Works Association (JAWWA).*

Chapter 6 describes the water quality and homeowner survey data collected during the follow-up study in Albemarle County, VA. This effort incorporates data from the profiling effort of

Chapter 4, in addition to data from 29 households that initially submitted samples with low lead concentrations.

CHAPTER 1. INTRODUCTION

Lead poisoning is defined as a wholly preventable disease; however, more than half a million children in the United States under the age of five have elevated blood lead levels (BLLs; >5 micrograms of lead per deciliter of blood) (Center for Disease Control and Prevention, 2010). Monitoring efforts indicate that the percent of children with elevated BLL is declining as the third National Health and Nutrition Examination Survey (NHANES; 1988-1994) estimated that 8.9% of children between the age one and five had BLLs greater than 10 μg/dL (Pirkle et al., 1994), and the fourth NHANES (1999-2010) determined that only 2.6% of children between the age one and five had BLLs greater than 5 μg/dL (Wheeler and Brown, 2013). Although the CDC has established 5 μg/dL as a current reference threshold for lead poisoning in children (Advisory Committee on Childhood Lead Poisoning Prevention, 2012), there is no safe long term exposure level (Wheeler and Brown, 2013).

Children with elevated BLLs are more likely to experience mental deficiencies such as lower IQ scores and shortened attention spans (Godwin, 2009). Detection of lead poisoning at early stages can be challenging as health symptoms of elevated BLLs can be nonspecific at concentrations below 70 μg/dL, including behaviors such as loss of appetite, reduced attention span or irritability (Edwards, 2008; Jones et al., 2009; U.S. Environmental Protection Agency, 2012a). Until recently, public health interventions to reduce BLLs have focused heavily on minimizing exposure from lead-based paint (Triantafyllidou and Edwards, 2012). The CDC has estimated that 24 million homes have deteriorating lead-based paint and/or lead-contaminated dust, and that more than four million children reside in these homes (Center for Disease Control and Prevention, 2009). However, removal of lead-based paints is not wholly sufficient to prevent lead poisoning, as the CDC also states that approximately 30% of children with EBLLs have not been exposed to leaded-paint source (Levin et al., 2008). Exposure from secondary sources (i.e. non-paint sources) is difficult to quantify as typical doses and pathways are not as well understood, and therefore likely underestimated (Levin et al., 2008). According to the latest estimates available from the U.S. Environmental Protection Agency (USEPA), the ingestion of waterborne lead may contribute up to 20% of an individual’s total lead exposure, and up to 40 to 60% of an infant reliant on formula’s exposure (U.S. Environmental Protection Agency, 2006). To minimize waterborne lead exposure, the USEPA has established and enforced drinking water regulations that focus on identifying and controlling corrosion, the primary mechanism for lead release from pipes into water, in plumbing networks connected to public (municipal) systems to prevent waterborne lead exposure.

In 1991, the Lead and Copper Rule (LCR) established an “action level” of 15 μg/L lead as measured at the point of use (POU, i.e. consumers’ taps) following a minimum of six hours of stagnation (U.S. Environmental Protection Agency, 2002). It is important to recognize however, that while this sampling protocol is designed to measure lead contamination at the POU to incorporate contamination from premise plumbing, the action level refers to community-wide rates of contamination (Triantafyllidou and Edwards, 2012), i.e. a water utility is required to implement corrosion control measures and inform the public only if more than 10% of the households sampled exceeded 15 μg/L (U.S. Environmental Protection Agency, 2002). Under the LCR, the EPA also established a maximum contaminant level goal (MCLG) of zero μg/L. This non-enforceable goal represents the concentration of lead in drinking water that will not result in adverse health effects following human consumption, which reflects the scientific consensus that there is no “safe” concentration of lead in drinking water.

Private drinking water systems

Approximately 15% of U.S. households are reliant on unregulated private water systems (e.g. wells, springs), which are generally in rural communities where municipal water service is not available (Knobeloch et al., 2013; Swistock et al., 2013; U.S. Environmental Protection Agency, 2012b). Private water systems, i.e. systems with less than 15 service connections and serve less than 25 individuals for at least 60 days per year, do not fall under the regulatory authority of the EPA and are therefore not subject to drinking water standards enforced by the EPA (U.S. Environmental Protection Agency, 2012b). Without characterizing lead exposure in private water systems, which serve more than 47 million residents, it will be impossible to gage progress towards goals such as Healthy People 2020’s objective of eliminating elevated BLLs in children (U.S. Department of Health and Human Services, 2014).

Occurrence of lead in private systems

There is general consensus that lead in drinking water overwhelmingly originates from the corrosion of lead-bearing plumbing components within the distribution and/or premise plumbing networks. Surveys of lead at the point of entry, considered representative of groundwater (i.e. source water), are generally very low or non-detectable (DeSimone, 2009; Knobeloch et al., 2013). The issue of lead in private systems is therefore in some ways analogous to that of homes dependent on municipal water, as studies of treatment plant effluent have observed low or no lead in finished water entering the distribution system; observed lead levels are likely nearly entirely the result of premise plumbing corrosion (Agency for Toxic Substances and Disease Registry, 1988; Center for Disease Control and Prevention, 1991; Levin, 1987; Triantafyllidou and Edwards, 2012).

To date, there have only been four available published studies that investigate elevated lead concentrations in first draw samples collected from private systems. The USEPA conducted a comprehensive national survey of rural water supplies in the late 1970s and early 1980s and observed that 9.2% of the 265 households sampled had lead concentration in excess of 50 µg/L (Francis et al., 1982). In Pennsylvania, 19% of private systems sampled between 1989 and 1992 exceeded 15 µg/L and 12% of samples exceeded 15 µg/L during a follow-up study approximately ten years later (Swistock et al., 2013; Swistock et al., 1993). Elevated concentrations were also observed in North Carolina in the late 1980s, as 34% of samples had lead greater than 10 µg/L (Maas & Patch 1990). These results indicate that in the past, elevated waterborne lead concentrations are not uncommon in private systems.

The research presented in this dissertation advances the knowledge of corrosion control and public health protection in private water systems through a variety of bench and field scale research efforts. The results will lead to procedures, protocols, and insights regarding lead in water in private water systems. The specific research objectives (O1-O4) include:

O1. Quantify dissolved and particulate waterborne lead concentrations in private water systems across Virginia;

O2. Examine potential predictive relationships between waterborne lead and environmental conditions, system construction characteristics, and household demographics;

O3. Identify patterns of waterborne lead release in private systems and evaluate the plumbing source materials for the elevated lead; and

O4. Evaluate lead release from NSF certified plumbing components when exposed to water representative of private water systems.

CHAPTER 2. INCIDENCE OF WATERBORNE LEAD IN PRIVATE DRINKING WATER SYSTEMS IN VIRGINIA

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Abstract

Although recent studies suggest contamination by bacteria and nitrate in private drinking water systems is of increasing concern, data describing contaminants associated with the corrosion of onsite plumbing are scarce. This study reports on the analysis of 2,146 samples submitted by private system homeowners. Almost 20% of first draw samples submitted contained lead concentrations above the USEPA action level of 15 μg/L, suggesting that corrosion may be a significant public health problem. Correlations between lead, copper, and zinc suggested brass components as a likely lead source, and dug/bored wells had significantly higher lead concentrations as compared to drilled wells. A random subset of samples selected to quantify particulate lead indicated that, on average, 47% of lead in the first draws was in the particulate form, although the occurrence was highly variable. While flushing the tap reduced lead below 15 μg/L for most systems, some systems experienced an increase, perhaps attributable to particulate lead or lead-bearing components upstream of the faucet (e.g. valves, pumps). Results suggest that without including a focus on private as well as municipal systems it will be very difficult to meet the existing national public health goal to eliminate elevated blood lead levels in children.

Introduction

The United States' Centers for Disease Control and Prevention (CDC) and the World Health Organization (WHO) classify lead poisoning as a preventable disease as exposure can be prevented through the direct removal of sources of lead and/or implementation of measures aimed at controlling lead releases. Since the 1970s, various federal regulations, including the phase-out of lead in gasoline and the ban of lead-based paint, have significantly reduced blood lead levels in the United States (Pirkle *et al.* 1994; Jones *et al.* 2009). However, recent reports suggest that exposures from secondary sources (i.e. non-paint sources) have likely been underestimated (Levin *et al.* 2008), and that these exposures can result in elevated blood lead levels in children and perhaps even fetal death (Edwards *et al.* 2009; Brown *et al.* 2011; Edwards 2014). The United States Environmental Protection Agency (USEPA) has estimated that waterborne lead may contribute up to 20% of an adult's total lead exposure and up to 60% of total lead exposure for an infant consuming formula (U.S. Environmental Protection Agency 1993); however, as other sources of exposure are removed from the environment, the relative percentage of exposure related to water is likely rising (Levin *et al.* 2008).

In 1991, the USEPA promulgated the Lead and Copper Rule (LCR) to minimize waterborne lead exposure through the identification and control of corrosion, which is the primary mechanism for lead release in municipal drinking water systems. The LCR states that if more than 10% of high risk households sampled at the point of use (POU; i.e. consumers' taps) have lead concentrations greater than the "action level" of 15 μg/L, the utility must take actions to control corrosion, and educate the public regarding risks and water-lead-avoidance strategies (U.S. Environmental Protection Agency 1993).

Approximately 10 to 15% of U.S. households are dependent on private drinking water systems (i.e. systems that serve an average of less than 25 individuals for at least 60 days per year and have less than 15 service connections), which by definition are not regulated by the USEPA under the Safe Drinking Water Act (SDWA) or subject to LCR requirements (U.S. Environmental Protection Agency 1993, 2012, 2013). Although some states have regulations related to the initial siting of private wells, these regulations do not address long-term water quality monitoring to ensure safe drinking water. As the monitoring and maintenance of private systems are solely the responsibility of the homeowner, water quality testing is often inconsistent and systematic reporting of water quality at the community or state level is uncommon (Francis *et al.* 1982; Knobeloch *et al.* 2013; Swistock *et al.* 2013). Past studies report that 20 to 50% of private systems exceed at least one SDWA health-based standard, most often for coliform bacteria (DeSimone 2009; Knobeloch *et al.* 2013; Swistock *et al.* 2013). Contamination due to the corrosion of internal plumbing is rarely assessed, but results from the few studies that measured lead concentrations in "first draw" samples (i.e. sample collected from the POU following a minimum of six hours of stagnation) suggest that corrosion may be a concern in private systems.

The first comprehensive national survey of rural water supplies in the United States (e.g. individual wells and small community systems) conducted in the late 1970s and early 1980s reported that 9.2% of the 2,654 household samples contained lead concentrations in excess of 50 μg/L (Francis *et al.* 1982). During a 1985 investigation of 55 private wells in rural Nova Scotia, 20% of first draws exceeded 50 μg/L (Maessen *et al.* 1985). In 1988, 34% of samples from 605 private systems in North Carolina had lead concentrations greater than 10 μg/L (Maas & Patch 1990); the authors noted that private system homeowners were at greater risk of lead exposure than homeowners connected to municipal systems. A study of over 1,500 private systems in Pennsylvania conducted between 1989 and 1992 reported that 19% of household samples exceeded 15 μg/L lead, with concentrations as high as 2,800 μg/L (Swistock *et al.* 1993). Roughly a decade later, the same group in Pennsylvania observed lead concentrations greater than 15 μg/L in 12% of 251 submitted samples (Swistock *et al.* 2013). Although these studies are limited in scope, they support the need for further, more detailed, investigation of corrosion in private systems.

To more effectively prevent lead release within plumbing networks, recent studies of municipal systems have distinguished and quantified the physical states of lead (i.e. soluble vs. particulate lead). Soluble lead is operationally defined as the concentration passing through a 0.45 µm pore size filter, and was previously the sole target of most sampling efforts, often resulting in systematic underestimates of actual lead levels and the potential for human exposure (McNeill & Edwards 2004; Triantafyllidou *et al.* 2007). The incidence of particulate lead (i.e. lead retained by a 0.45 µm filter) during sample collection is considered highly variable and is often associated with higher flow rates and/or hydraulic disturbances (Schock, 1990; Triantafyllidou *et al.* 2007; Deshommes *et al.* 2010; Triantafyllidou *et al.* 2013). Studies consistently emphasize that the characterization of particulate lead levels is critical when estimating human exposure and designing appropriate remediation efforts (Triantafyllidou *et al.* 2007; Deshommes *et al.* 2012; Triantafyllidou *et al.* 2013; Clark *et al.* 2014). To date, only one identified study has evaluated particulate lead in private drinking water systems (Swistock *et al.* 1993). The authors concluded that solely quantifying soluble lead may have underestimated lead concentrations by 6 to 18 µg/L in their 126 samples, emphasizing the importance of designing appropriate sampling strategies to ensure accurate measurement.

Though regulatory mechanisms such as the SDWA and LCR ensure the provision of safe drinking water to homes connected to municipal systems, recent studies have highlighted the need to more deeply examine private drinking water systems where exposure to waterborne contaminants of human health concern are increasingly recognized (Wallender *et al.* 2014). In particular, without characterizing waterborne lead exposure in these systems, achieving the federal goal to eliminate elevated blood lead levels in children by 2020 will be challenging (U.S. Department of Health and Human Services, 2014). To better characterize the incidence of lead in samples from these systems, this study: 1) documents the occurrence of lead in water samples collected from the point of use of households dependent on private drinking water systems; 2) quantifies the relative amounts of dissolved and particulate lead in these samples; 3) identifies major system and environmental characteristics associated with high lead concentrations; and 4) evaluates associations between homeowner perception of water quality and the presence of high lead concentrations.

Methods

Samples were collected via a collaboration with the Virginia Household Water Quality Program (VAHWQP; www.wellwater.vt.edu), a long-standing state-sponsored Cooperative Extension program based at Virginia Tech. Although private water supply systems provide water for several domestic purposes, this research focused on drinking water quality (e.g. samples at the point of use) and will refer to these systems as *private drinking water systems*. As lead was only added to those target contaminants included in the VAHWQP testing package in January of 2012, this partnership represents the first identified investigation of lead concentrations in private drinking water systems in the Commonwealth of Virginia.

Sample collection

Through VAHWQP's county-based drinking water clinics, 2146 water samples were collected between March 2012 and November 2013. Participation in the drinking water clinics was wholly voluntary and participants were therefore self-selected. Homeowners who wished to participate purchased a water sampling kit that included: (i) sampling instructions on how to properly collect each sample; (ii) a questionnaire about system characteristics, perception of water quality, and household demographics; and (iii) sampling bottles pre-prepared per Standard Methods (e.g. autoclaved, acid washed) for the collection of water for different analyses. Per the instructions, homeowners collected samples from a non-swivel faucet on a predetermined morning. After a minimum of six hours of stagnation, homeowners removed the aerator and collected 250 mL of water at a pencil-thin flow (*first draw)*. The system was then flushed for a minimum of five minutes, and three additional samples (two 250 mL and one 100 mL) were collected at a pencil-thin flow (*flushed samples)*. Two bottles were used to evaluate bacteria and physicochemical parameters, while the third bottle (250 mL) was evaluated for flushed metal concentrations. After collection, participants brought the samples to a designated location for transportation on ice to Virginia Tech. Samples were processed within 8 to 12 hours after collection.

Water quality analyses

Samples were processed in the Virginia Tech Biological Systems Engineering (BSE) Water Quality Laboratory for pH, conductivity (proxy for total dissolved solids), nitrate-N, sulfate, and fluoride per methods 4500-H+, 2510, and 4110C (American Public Health Association, American Water Works Association & Water Environment Federation 1998). Bacteria (i.e. total coliform and *Escherichia coli*) were quantified using the IDEXX Colilert 2000 method (www.idexx.com, Westbrook, MN, USA). Metals (aluminum, arsenic, cadmium, calcium, chloride, chromium, copper, iron, lead, magnesium, manganese, nickel, silver, sodium, tin, and zinc) were analyzed in the Virginia Tech Environmental and Water Resources Engineering (EWRE) Water Quality Laboratory using Inductively Coupled Plasma – Mass Spectrometry (ICP-MS) per method 3125 B (American Public Health Association, American Water Works Association & Water Environment Federation 1998). Hardness was calculated based on the concentrations of calcium and magnesium. For data quality assurance and quality control (QA/QC), blanks and spikes of known concentrations were measured every 10-15 samples for all analyses, except in the case of Colilert, where a positive and negative control were included for each clinic's sample set.

Acidification for metals analysis

During the 2012 clinics, metal concentrations were quantified via 10 mL aliquots of the first draws and flushed samples; this practice of sample transfer was previously used by extension to quantify metals and has been used by several municipalities to quantify lead (Freud 2012; Triantafyllidou *et al.* 2013; Etchevers *et al.* 2015). The first draw and flushed aliquots were acidified with 2% nitric acid and digested for a minimum of 16 hours. Due to the unexpectedly high levels of lead observed during the first year of sampling, the 2013 sampling protocol was modified to permit acidification in the original sampling bottle in keeping with the USEPA LCR protocol.

Evaluating systematic error and quantifying particulate lead

To evaluate systematic error associated with acidifying a sample aliquot (2012 method) rather than the full bottle (2013 method) and to quantify particulate lead, 20% of the first draws and flushed samples submitted during 2013 were selected for additional analyses. Samples with visible particles, discoloration and/or odor were preselected for this analysis, with a remaining number of samples randomly selected to reach a total of 20% of samples submitted. Two 10 mL aliquots were pipetted from each sample: the first aliquot was used to evaluate the 2012 method, and the other aliquot was filtered through a 0.45 μm pore size Whatman nylon filter to quantify dissolved lead. Both aliquots and the remaining sample (~230 mL) were acidified with 2% nitric acid and digested for a minimum of 16 hours prior to ICP-MS analysis. Note: total lead concentrations were used exclusively throughout this study, except for the "Occurrence of particulate lead" section, which discussed both particulate and total lead concentrations.

Statistical analysis

All statistical analyses were conducted in R version 3.0.2 (R Development Core Team, 2012) assuming an alpha of 0.05 as an indication of significance. Due to the non-normal distribution of the lead data (Shapiro-Wilk; $p<0.05$), nonparametric statistics were used throughout this study. Potential associations between lead concentrations and the other water quality parameters (e.g. copper, zinc, and tin) were evaluated using the Spearman's rank correlation (ρ), while the Wilcoxon signed-rank test and Kruskal-Wallis test compared lead concentrations based on categorical household characteristics (e.g. income, education, piping material) and homeowner perception of water quality (e.g. taste, odor). Odds ratios were calculated to determine the odds of having elevated lead concentrations based on homeowners' perceptions of water quality.

Results and discussion

Overview of participating systems

Between March 2012 and November 2013, 2146 water samples from 61 of the 95 counties in Virginia were submitted for analysis through the VAHWQP Cooperative Extension program. As participation in clinics was dependent on local interest and awareness, the number of samples submitted per county varied substantially. Of the samples submitted, 94.5% were collected from a private well (n=2029), 3.6% from a spring (n=78), less than 0.1% from a cistern (n=2), and the remaining 1.7% of samples were either from a municipal system or the source was not identified (n=37). Of the 2029 wells, 79% were "drilled", 12% were "dug/bored", and the remaining 9% were "unknown". Response rates regarding the year the well was constructed and the depth of the well were somewhat low (69% and 59% responding, respectively). The average year of construction reported was 1988, though responses ranged from 1850 to 2013. Reported well depths varied from 1.2 to 381 m, with a mean and median depth of 77.7 and 67.0 m, respectively. Roughly half (54%) of homeowners indicated that their system included some sort of treatment, with the most common type of treatment being a water softener (23%) or a sediment filter (29%).

Although previous studies report infrequent rates of water quality testing (Francis *et al.* 1982; Swistock *et al.* 1993), 55% of homeowners submitting samples to this program indicated that they had previously tested their water. Reported household demographics suggested that homeowners who participated in the VAHWQP drinking water clinics were largely older, more affluent, "White or Caucasian", and had a college or graduate-level education. Based on self-reported household income (n=1796), 51% of homeowners reported an annual household income greater than $65000, and only 8% reported an income in the lowest category (i.e. below $24000). With respect to education (n=1983), 69% of homeowners received a college degree with 37% having obtained a post college degree (i.e. masters or doctoral). The population sampled also tended to be older (n=2024) as 81% were older than 50 and 59% of homeowners were older than 60. During the 2013 clinics, self-reported race was added to the questionnaire (n=1115), and 89% of homeowners self-identified as being "White or Caucasian". In addition to evaluating the population participating in this extension water quality testing, these results are important in the context of lead exposure as they highlight the low rate of participation in this water quality testing amongst young adults (<50 years old) most likely to have children under the age of six, who are at greatest risk for detrimental health effects following lead exposure.

Overall prevalence of contamination

Using the SDWA standards for municipal systems as guidelines, 58% of submitted samples (n=1250) exceeded at least one Maximum Contaminant Level (MCL) (Table 2-1). Bacterial contamination was the most common issue, with 46% of systems testing positive for total coliform. Using the action level for lead and copper as a threshold, 19% of systems had elevated lead concentrations (≥15 µg/L) and 12% had elevated copper concentrations (≥1.3 mg/L) in the first draw. Twenty six percent of households (n=553) had a pH value outside the recommended 6.5 to 8.5 range, and 89% of these pH values (n=491) were below 6.5. Values exceeding other water quality constituents targeted by VAHWQP were relatively rare.

Table 2-1. Summary statistics of water quality parameters in samples collected from private drinking water systems

Target water quality constituent	**Standard**		**Number of observations**	**Mean**	**Median**	**90th Percentile**	**Percent Exceeding**
[‡]Arsenic, in mg/L	MCL	0.010 mg/L	2146	BDL	BDL	BDL	0.1%
[†]Cadmium, in mg/L		0.005 mg/L	2144	BDL	BDL	BDL	0.6%
[†]Chromium, in mg/L		0.1 mg/L	2144	BDL	BDL	0.001	0.0%
[‡]Fluoride, in mg/L		4.0 mg/L	2146	BDL	BDL	BDL	0.4%
[‡]Nitrate, mg/L		10 mg/L	2146	1.5	BDL	4.2	1.3%
[‡]Total coliform, in cfu		ABSENT	2143	-	-	-	46%
[‡]*E. coli*, in cfu		ABSENT	2143	-	-	-	10%
[†]Copper, in mg/L	Action Level	1.3 mg/L	2144	0.677	0.153	1.727	12%
[†]Lead, in mg/L		0.015 mg/L	2144	0.022	0.004	0.027	19%
[†]Aluminum, in mg/L	SMCL	0.05-0.2 mg/L	2144	0.047	0.003	0.061	3.8% (12%[*])
[‡]Chloride, in mg/L		250 mg/L	1378	14.3	5.4	31.4	0.2%
[‡]Copper, in mg/L		1.0 mg/L	2144	0.677	0.153	1.727	15%
[‡]Fluoride, in mg/L		2.0 mg/L	2146	BDL	BDL	BDL	2.7%
[‡]Iron, in mg/L		0.3 mg/L	2146	0.142	0.008	0.234	8%
[‡]Manganese, in mg/L		0.05 mg/L	2146	0.032	0.002	0.049	10%
[‡]pH, in standard units		6.5-8.5	2146	7.1	7.2	8.0 (6.0[§])	26% (23%[**])
[†]Silver, in mg/L		0.1 mg/L	2144	BDL	BDL	BDL	0.0%
[‡]Sulfate, in mg/L		250 mg/L	2146	45.1	8.0	52.3	2.4%
[‡]TDS, in mg/L		500 mg/L	2146	237.1	179.6	501.1	10%
[†]Zinc, in mg/L		5 mg/L	2144	0.868	0.231	2.247	3.1%

MCL: Maximum Contaminant Levels – associated with risk to human health
SMCL: Secondary Maximum Contaminant Levels – associated with aesthetic considerations
BDL: Below detection limit
cfu: colony forming units
[†] Total concentration measured in the first draw
[‡] Total concentration measured in the flushed sample
[§] 10th percentile
[*] Percent above 0.05 mg/L
[**] Percent below pH of 6.5

Lead concentrations

Compliance with the LCR requires that no more than 10% of samples exceed the action level. As each private system is an independent distribution system and only one first draw sample was collected during this study, for comparison purposes, the action level was used as a threshold. Lead concentrations in the first draws ranged from below detection (<1 µg/L) to 24,740 µg/L (Figure 2-1a). Eighty percent of households had detectable lead concentrations (≥1 µg/L) and 19% of samples exceeded the lead action level of 15 µg/L.

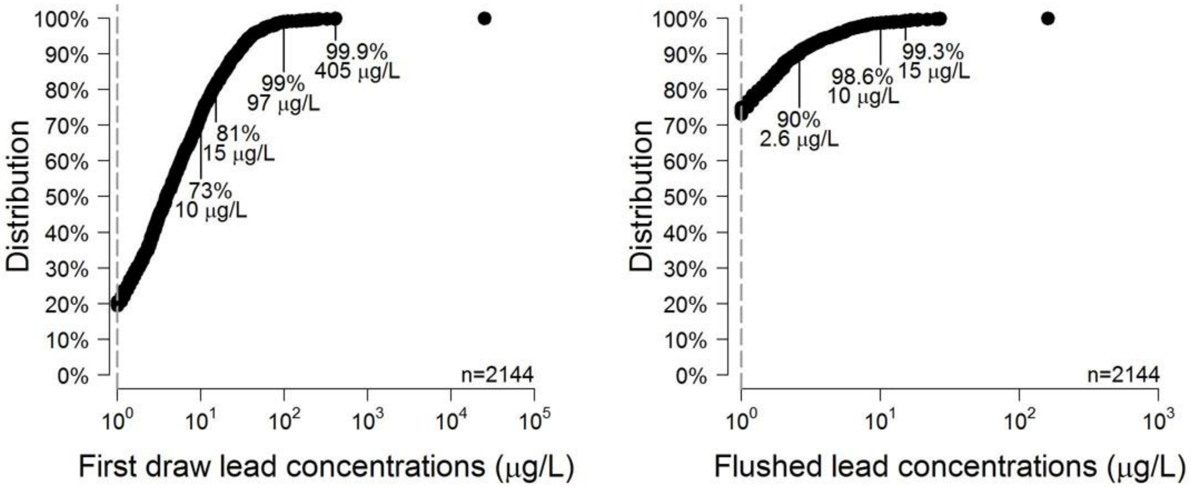

Figure 2-1. Lead concentrations in the (a) first draws and (b) flushed samples collected during the 2012 and 2013 VAHWQP drinking water clinics. Dashed lines represent the detection limit (1 μg/L).

Flushing the system for five minutes appeared to reduce lead concentrations to the recommended concentration (i.e. below 15 μg/L) for most households in this study; less than 1% of flushed samples exceeded 15 μg/L, and 74% of samples had non-detectable (<1μg/L) concentrations (Figure 2-1b). However, 2% of the households (n=46) had higher lead concentrations in the flushed sample relative to the first draw (Figure 2-2; i.e. above the 1:1 line). For most of these 46 households, flushed concentrations increased by a relatively small amount (mean and median increase of 5.2 and 1.0 μg/L, respectively), but in eight homes increases were >5 μg/L (max observed increase = 130 μg/L). In households with higher flushed concentrations, there may be components upstream of the faucet that are responsible for lead release such as brass valves (Sandvig *et al.* 2008) or submersible pumps (Maas *et al.* 1998), and/or particulate lead may have been mobilized (Schock 1990).

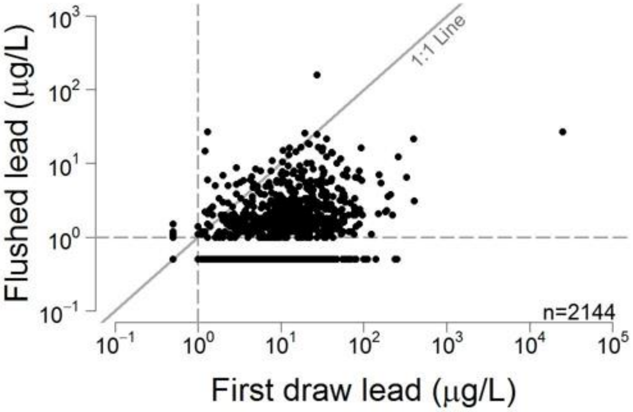

Figure 2-2. First draw lead concentrations compared to lead concentrations after five minutes of flushing. Dashed lines represent the detection limit (1 μg/L). Samples below the detection limit were set to 0.5 μg/L.

Occurrence of particulate lead

To better understand patterns and mechanisms of lead release, 20% of first draw and corresponding flushed samples submitted to the 2013 drinking water clinics (n=253) were filtered through a 0.45 µm filter to quantify dissolved lead. As previously mentioned, samples with noticeable characteristics (e.g. odor, flakes; n=69) were preselected for this analysis, which therefore may represent a worst-case scenario for estimation of particulate lead. The remaining quantity (n=184) were randomly selected. Four samples were removed from this analysis as these samples did not meet associated QA/QC checks. Particulate lead was calculated as the difference between total lead and dissolved lead.

On average, for the preselected first draws with detectable total lead concentrations (≥ 1µg/L, n=55), 75% of the total lead was in the particulate form, although some samples were entirely composed of particulate lead (Figure 2-3a; i.e. >99%). For the randomly selected first draws with detectable total lead concentrations (≥ 1µg/L, n=143), 47% of the total lead was in the particulate form, and again, some samples were composed entirely of particulate lead (i.e. >99%). The median lead concentration for preselected first draws (n=67, 5.2 µg/L) was significantly higher (Wilcoxon signed-rank tests, $p<0.05$) than randomly selected first draws (n=183, 1.1 µg/L). Only 30% of the corresponding flushed samples (n=76) measured detectable lead concentrations, but 62% of the total lead was in the particulate form (Figure 2-3b).

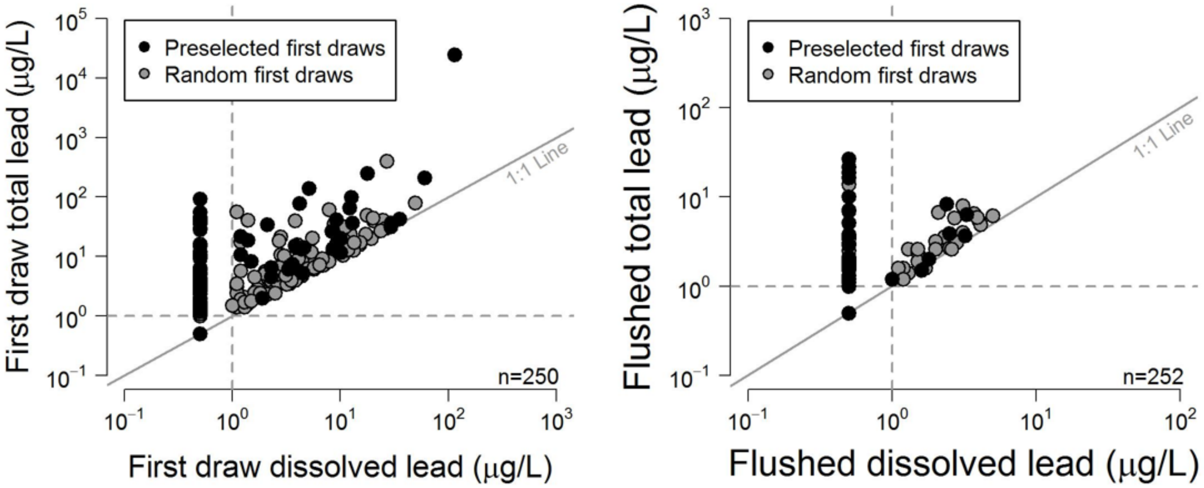

Figure 2-3. Presence of particulate lead in (a) first draws and (b) flushed samples. The presence of samples along the 1:1 line indicated that 100% of the lead was in the dissolved form (i.e. total and dissolved concentrations were equal. Dashed lines represent the detection limit (1 µg/L). Samples below the detection limit were set to 0.5 µg/L.

Overall, the fraction of particulate lead in the first draw did not increase with higher total lead concentrations. Households with elevated total lead concentrations had varying fractions of particulate lead ranging from non-detect to 99% composition (Figure 2-3a). This is consistent with prior work indicating that the occurrence of particulate lead is variable and unpredictable in public water systems, even when sampling is rigorously controlled (Triantafyllidou *et al.* 2007; Sandvig *et al.* 2008; Triantafyllidou *et al.* 2013; Clark *et al.* 2014).

When considering the systematic error associated with acidifying a sample aliquot (2012 method) rather than acidification in the bottle (2013 method), it is important to note that differences between subsample aliquot and total lead values were less than 1 µg/L for 64% of first draws and 92% of flushed samples. The first draw subsamples that under or overestimated total lead (n=92) were primarily composed of particulate lead (mean 70% particulate form), and had a mean absolute error of 11.0 µg/L (max under and overestimate of 96 and 39 µg/L, respectively). Flushed subsamples that under or overestimated total lead (n=20) were also primarily composed of particulate lead (mean 75% particulate form), and had a mean absolute error of 3.6 µg/L. Given the unpredictability of particulate lead release and the potential high associated error noted in both this study and previous studies of municipal water (Freud 2012; Triantafyllidou *et al.* 2013), acidification in bottle is highly recommended to ensure an accurate measure of lead concentration.

Potential factors predictive of elevated lead concentrations in first draw samples

System characteristics

The median lead concentration for dug/bored wells (n=248, 9.4 µg/L) was significantly higher (Kruskal-Wallis test, $p<0.05$) than drilled wells (n=1607, 3.6 µg/L) and springs (n=77, 3.5 µg/L). While 32% of samples from dug/bored wells exceeded the 15 µg/L action level, only 17 and 14% of drilled wells and springs exceeded this value, respectively. Swistock *et al.* (1993) observed higher lead concentrations in springs and dug wells compared to drilled wells, which the authors attributed to the acidity of shallow groundwater. This study did not observe a correlation between self-reported well depths and lead concentrations (n=1204, $\rho= -0.25$), but lead concentrations were negatively correlated with pH values ($\rho= -0.42$, $p<0.05$). pH varied significantly (Kruskal-Wallis test, $p<0.05$) based on system type. The median pH for dug/bored wells (pH 6.2) was significantly lower than springs (pH 6.9), which was significantly lower than drilled wells (pH 7.2). Not surprisingly, 63% of dug/bored wells, 29% of springs, and 17% of drilled wells had a pH below 6.5. There were no correlations between lead concentrations and fluoride, hardness, nitrate, sodium, sulfate or total dissolved solids (TDS).

Participants who indicated the use of a water treatment device did not have significantly lower median lead concentrations (Wilcoxon signed-rank tests, $p=0.84$). Consistent with the literature, most systems that included some form of water treatment had a sediment filter (29%) or water softener (23%). Reverse osmosis units and acid neutralizers are recommended treatments for households with corrosion, but only 2.6% and 4.8% of homeowners, respectively, indicated having installed these treatment devices. As this study did not independently verify or evaluate the maintenance of these treatment units, determining the influence of specific treatment types on lead concentrations was not possible. Although past studies have observed that bacterial contamination in private systems is more prevalent in lower income and education households (Francis *et al.* 1982; Smith *et al.* 2014), in the present study there was no significant difference in lead concentrations based on household income ($p=0.57$), education ($p=0.55$), race ($p=0.21$) or age ($p=0.23$).

Plumbing components

There was no significant difference (Wilcoxon signed-rank tests, $p=0.21$) in lead concentrations between households that identified only having copper (n=514) versus plastic plumbing (n=915). Given that several studies estimate that the volume of water contained within the faucet and the connective piping is approximately 125 mL (Sandvig *et al.* 2008), and the first draw in this study was 250 mL, high concentrations in first draws may reflect leaching from brass within the faucet. Brass is an alloy composed mainly of copper (60-80%), zinc (4-32%), and lead (2-8%), but can contain other metals such as tin, nickel, aluminum, iron or cadmium (Lytle & Schock 1996). In keeping with this hypothesis, lead concentrations were strongly correlated ($p<0.05$) with zinc ($\rho=0.67$), copper ($\rho=0.61$), and nickel ($\rho=0.56$), and weakly correlated ($p<0.05$) with aluminum ($\rho=0.27$). Based on these findings, the interior piping material (e.g. copper, plastic) may not be as important in predicting lead concentrations as the type of fittings used in the plumbing network (e.g. brass).

Households constructed pre-1988 had a significantly higher ($p<0.05$) median lead concentration (5.4 μg/L, n=600) compared to households constructed post-1988 (3.3 μg/L, n=805). This is in keeping with Swistock *et al.* (2013), which observed that the majority of households with elevated lead concentrations had copper plumbing installed before 1991. This may reflect the 1986 Lead Ban, which required the use of "lead-free" plumbing components in the installation or repair of any municipal system or residential and nonresidential buildings connected to a municipal system after June 1988 (U.S. Environmental Protection Agency, 1993). While the Lead Ban was not directly aimed at private systems, these "lead-free" plumbing components are being installed in private systems and reducing the risk of lead release. It is important to note that as of January 2014, the Reduction of Lead in Drinking Water Act lowered the allowable lead content in "lead-free" plumbing components, which should reduce lead concentrations in newly constructed or repaired private drinking water systems (U.S. Environmental Protection Agency, 2014).

Perception

Although lead cannot be directly identified by smell or taste in drinking water (U.S. Environmental Protection Agency, 1993), previous studies suggest lead is associated with parameters (e.g. pH) that have discernable physical characteristics (e.g. metallic taste, blue-green stains, and obvious signs of corrosion) (Swistock *et al.* 1993). Odds ratios (OR) describing relationships between homeowner reported perceptions of water quality and observed lead concentration in the first draws analyzed in this study are illustrated in Figure 2-4.

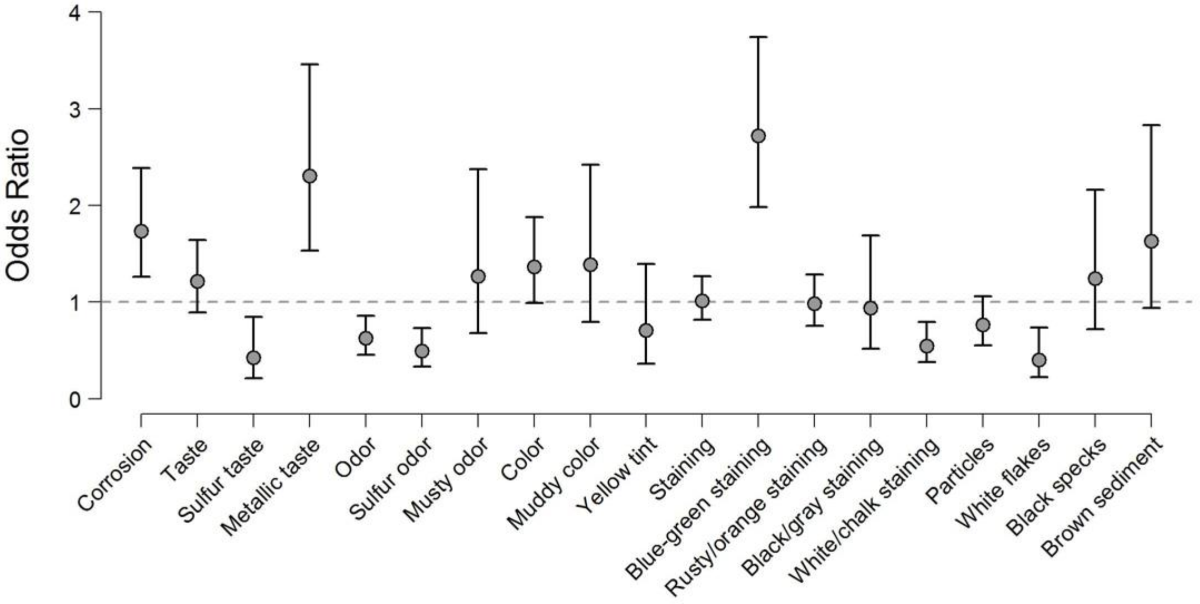

Figure 2-4. Odds ratio for physical characteristics identified by homeowners

Homeowners who identified obvious signs of corrosion (OR = 1.72), blue-green staining on plumbing fixtures (OR = 2.78) and/or described the taste of water as metallic (OR = 2.29) were 1.7 to 2.8 times more likely to have elevated lead concentrations compared to homeowners who did not identify these characteristics. Median lead concentrations for homeowners who identified these characteristics were also significantly higher (Wilcoxon signed-rank tests, $p<0.05$). As lead was correlated with pH and copper (most likely leaching from brass), these indicators are most likely linked to copper corrosion and lower pH values. Homeowners who noted that their water had an odor (OR = 0.62), a sulfur odor (OR = 0.49), a sulfur taste (OR = 0.42), identified white/chalk staining on plumbing fixtures (OR = 0.56) and/or noticed white flakes in the water (OR = 0.40) were 1.6 to 2.5 times less likely to have elevated lead concentrations compared to homeowners who did not identify these characteristics. Median lead concentrations for homeowners who identified these characteristics were also significantly lower (Wilcoxon signed-rank tests, $p<0.05$), except for white/chalk staining ($p=0.06$). As median lead concentrations were not significantly different for white/chalk staining on plumbing fixtures, this characteristic may not be a preferred indicator. These characteristics are most likely associated with the formation of passivating films or scales, which could inhibit corrosion (Oliphant 1983; Sandvig *et al.* 2008).

Given that previous studies have linked perception of water quality issues with increased stewardship, these observations may be useful in designing public health messaging (Imgrund *et al.* 2011); although it is important to note that an individual's perception of drinking water quality is highly variable and can be influenced by perceived risk, familiarity, chemical composition, and demographics (Cuppett *et al.* 2006; Doria *et al.* 2009; Doria 2010; Mirlohi *et al.* 2011).

Limitations

As previously discussed, this work represents a collaborative effort with an ongoing extension effort. Homeowners volunteered to participate and therefore, had both the knowledge and resources to procure water quality testing, which may have introduced sample bias. It is unknown if the rates of contamination were elevated as concerned homeowners sought testing or reduced as a specific demographic was primarily targeted. The sampling protocol was designed to quantify contamination commonly found in private drinking water systems, with bacteria as a primary interest. Homeowners were instructed to collect samples from a non-swivel faucet with the aerator removed as these characteristics are believed to increase the probability of bacterial contamination. Without an aerator attached, homeowners were also instructed to collect samples at "low pencil-thin flow" to minimize the amount of water missed during collection; collecting at "low flow" may have resulted in an underestimation in particulate lead. The origin of the sample collected after five minutes of flushing was unknown and depended greatly on the characteristics of the system (e.g. presence of treatment, length of piping). With lead-bearing components upstream of the faucet and within the well, lead concentrations are highly dependent on location. Questionnaire information was voluntarily provided by the homeowner, and has not been verified. When quantifying particulate lead, samples were preselected based on physical characteristics, which may have introduced bias and resulted in the worst-case estimation of particulate lead, as discussed previously.

Conclusions

Almost 60% of samples submitted during this study exceeded at least one USEPA health-based standard for municipal systems. In keeping with past studies targeting private systems, bacterial contamination was primarily responsible for these violations, although metal leaching due to corrosion was not uncommon. Samples provided by 19% of households had lead concentrations above the USEPA action level of 15 μg/L. Average particulate lead composition was 47% for the randomly selected subsample set, and 75% for the preselected first draws. Particulate lead, on average, composed 62% of the total lead in corresponding flushed samples. The occurrence of particulate lead was highly variable, and future studies are highly encouraged to quantify all forms of lead in sampling and analytical procedures in order to ensure an accurate measure of potential human exposure.

Perhaps not surprisingly given the 1986 Lead Ban, households constructed before 1988 had higher lead concentrations; however, it is important to note that elevated lead concentrations were still observed in residences constructed after 1988. Lead appeared most likely to be leaching from the brass within faucets and/or fittings as lead concentrations were highly correlated with copper and zinc. There was no significant difference in lead concentrations based on identified plumbing materials (i.e. copper and plastic), which further supports that lead is leaching from brass fittings and fixtures. However, with the Reduction of Lead in Drinking Water Act, lead release from brass components should be reduced. Although elevated concentrations of lead were observed in samples from all types of systems, samples from dug/bored wells had significantly higher lead concentrations. This finding is likely due to the acidity of groundwater associated with shallow depths as pH values were also significantly lower in these wells. The influence of geology was not assessed during this study, but may be an important variable in understanding lead release as a large percentage of private systems do not use treatment that adjusts the acidity of the source water. Lead cannot be identified by smell or taste in drinking water, but increased concentrations were associated with obvious signs of corrosion such as blue-green staining and metallic taste, which may be useful in communicating the potential risks of corrosion and promoting private system maintenance. The demographic data highlighted the primary population sampled, which did not encompass a large portion of young adults (<50 years old). As children are at greatest risk of lead poisoning, future efforts to prevent waterborne lead exposure should focus on engaging this population.

For most of the private systems sampled in this study, flushing for five minutes reduced lead concentrations below 15 μg/L, although 2% of households experienced an increase in lead concentrations with flushing. This finding suggests that there may be other components within the plumbing network that release lead and/or particulate lead may have been mobilized. In order to develop effective remediation efforts and sampling protocols, an increased understanding of the mechanisms of lead release is essential.

Acknowledgements

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CHAPTER 3. SIMULTANEOUS INFLUENCE OF GEOLOGY AND SYSTEM DESIGN ON DRINKING WATER QUALITY IN PRIVATE SYSTEMS

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Abstract

Between 2012 and 2014, almost 3000 point of use water samples from private water systems (e.g. wells, springs) in Virginia were analyzed for common contaminants of human health and aesthetic concern. In addition, each sample was accompanied by a brief questionnaire detailing system characteristics. Approximately 55% of samples exceeded at least one health-based drinking water standard. This study evaluated the interactions between local geology and private system type to understand variations in water quality, which is critical when evaluating and prioritizing efforts to protect public health. In the context of lead, sodium, and total coliform, this study illustrated the importance of considering: (1) local geology as it dictates groundwater flow; (2) private system type as it determines the source aquifer and raw groundwater quality; and (3) household treatment devices as potential sources of additional water quality constituents.

Introduction

Private water systems (e.g. wells, springs) are the primary source of drinking water for 22% of households in Virginia, serving approximately 1.7 million residents (Kenny et al., 2009). As private systems are not regulated by the federal Environmental Protection Agency (USEPA), they are not subject to the standards outlined in the Safe Drinking Water Act (SDWA) (U.S. Environmental Protection Agency, 2013). Although federal and state agencies encourage private system homeowners to comply with the SDWA standards, ultimately homeowners are wholly responsible for ensuring the safety and quality of their household drinking water. Past studies report that 23-58% of private systems exceeded at least one SDWA health-based standard, with variations in quality attributed to local geology and/or type of private system constructed (DeSimone, 2009; Knobeloch, Gorski, Christenson, & Anderson, 2013; Pieper, Krometis, Gallagher, Benham, & Edwards, 2015; Swistock, Clemens, Sharpe, & Rummel, 2013).

The most common type of private system in the United States is the drilled well (Pieper, et al., 2015; Swistock, et al., 2013). These wells can be constructed in any geologic region using drilling methods and construction practices specific to the aquifer material (e.g. presence of a well screen; Figure 3-1a) (Gibbs, 1973; Virginia Water Resources Research Center, 1995; Waller, 1994). While drilled wells can vary greatly in depth (6-305 m), all typically have continuous casing of 0.1-0.15 m diameter that extends at least 0.2 m above the ground surface to prevent surface and shallow groundwater contamination. In contrast, in low yielding aquifers, such as overburdens or unconfined aquifers, dug and bored wells are often more advantageous than drilled wells. These shallow wells (9-30 m) have non-continuous casings of larger diameter (0.6-0.9 m), which provide additional water storage in the well (Figure 3-1b) (Gibbs, 1973;

Virginia Water Resources Research Center, 1995; Waller, 1994). Springs are an alternative household source water and occur naturally in areas where the land surface intersects flowing groundwater, as in low-lying regions and at the base of slopes (Figure 3-1c) (Virginia Water Resources Research Center, 1999; Waller, 1994). Homeowners may construct a spring box to divert water into the household or collect and transport water from a roadside spring.

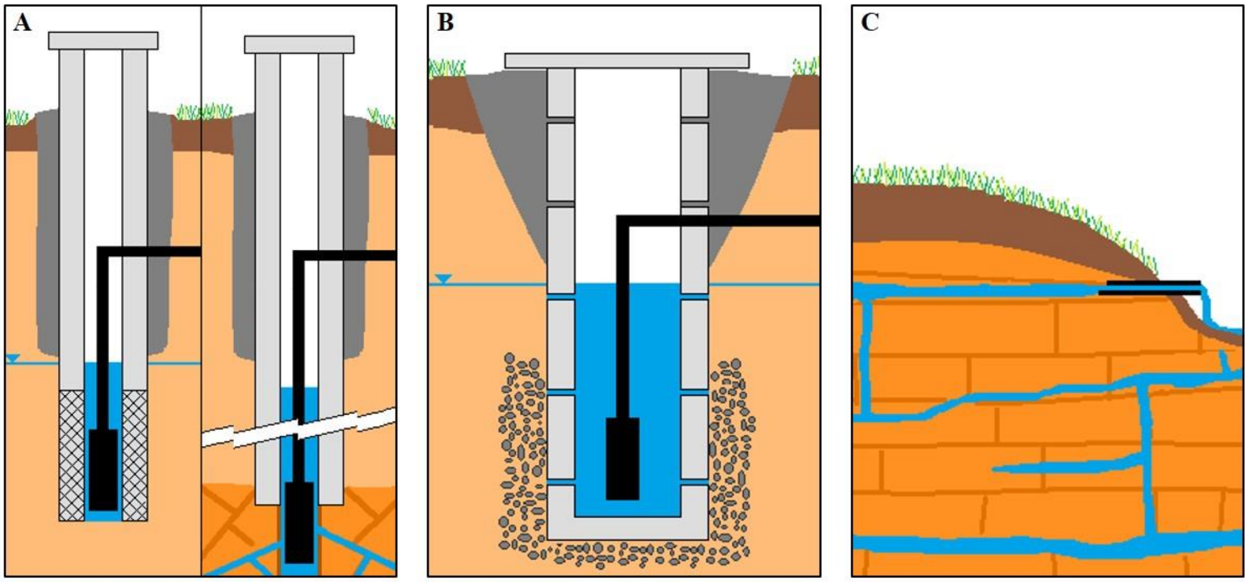

Figure 3-1. Types of private systems: (a, left): drilled well in an unconsolidated aquifer that is fully cased with a well screen; (a, right): drilled well in a consolidated aquifer that is cased only several feet into the bedrock; (b) dug/bored well in a shallow unconsolidated aquifer that has non-continuous casing; and (c) naturally occurring spring.

There are four primary geologic provinces in Virginia: Appalachian Plateau, Valley & Ridge, Blue Ridge-Piedmont, and Coastal Plain (Trapp & Horn, 1997). Differences in groundwater quality, as a result of underlying aquifer material, have been heavily documented (Heller, 2008; Nelms, Harlow, Plummer, & Busenberg, 2003; Trapp & Horn, 1997). To summarize this literature: in the Coastal Plain region’s unconsolidated and semi-consolidated aquifers, groundwater flows through pore spaces between sediment grains. Water quality varies considerably in this region and can be further complicated by saltwater intrusion. Though topographically different, the Blue Ridge and Piedmont regions both have fractured crystalline bedrock aquifers, with varying overburden thicknesses. Groundwater in this region flows through cracks and fractures and generally is acidic with lower specific conductivity, as the rocks are generally less reactive. In contrast, carbonate aquifers in the Valley & Ridge region are more prone to dissolution by groundwater, resulting in increased pH and networks of interconnected openings known as solution channels. Sedimentary rock underlying the Appalachian Plateau contains sandstone, shale, and coal, and groundwater flows through joint, fault, and bedding plane fractures.

Although previous studies have individually evaluated the influence of geology and private system type on water quality, studies have not considered water quality with respect to both variables simultaneously. Understanding the interactions between system location and construction is critical to evaluating and prioritizing the potential for health risks to people using private systems at both the regional and household level. The goal of this effort was to identify statistical trends linking private system type, location, and water quality at the point of use using a unique dataset of almost 3,000 samples. Information from this effort will be of direct use to environmental health practitioners, managers, and engineers attempting to reduce potential community health risks by improving private system maintenance and awareness.

Methods

Point of use water samples were collected from private systems by homeowners and analyzed through a long-standing Cooperative Extension program based at Virginia Tech described previously (Pieper, et al., 2015). In brief, participants were instructed to collect a first draw sample after a minimum stagnation of six hours and three additional samples after five minutes of flushing. Samples were analyzed for pH, conductivity (proxy for total dissolved solids), nitrate-N, sulfate, fluoride, and metals per standard methods 4500-H^+, 2510, 4110C, and 3125 B (American Public Health Association, American Water Works Association, & Water Environment Federation, 1998). Bacteria were quantified using the IDEXX Colilert 2000 method (www.idexx.com, Westbrook, MN, USA). In addition to collecting water samples, homeowners completed and submitted a brief questionnaire detailing system characteristics. In addition to collecting water samples, homeowners submitted answers to a questionnaire detailing system characteristics. The specific queries of relevance to this study are provided in Table 3-1, and the questionnaire is discussed in its entirety elsewhere (Allevi et al., 2013).

Table 3-1. Summary of questionnaire detailing system characteristics

Sample identification	- County of residence
Water source	- What household water supply source was drawn for sample (well, spring, cistern)? - If “well”, is it a dug or bored well, drilled well, or don’t know? - What is the well’s depth, if known? - What year was well constructed, if known? - What water treatment devices are currently installed?

Categorizing counties by underlying geology

Samples were collected and reported by county in keeping with Virginia Tech Institutional Review Board requirements. However, county boundaries do not necessarily conform to geologic formations, i.e. several counties lie within two geologic provinces. For the purposes of this study, if more than 75% of a given county’s area was within a geologic province, the county was considered to be solely within that geologic province. The six counties that did not meet this criteria were considered “mixed” and were not included in subsequent analyses (n = 213; Figure 3-2).

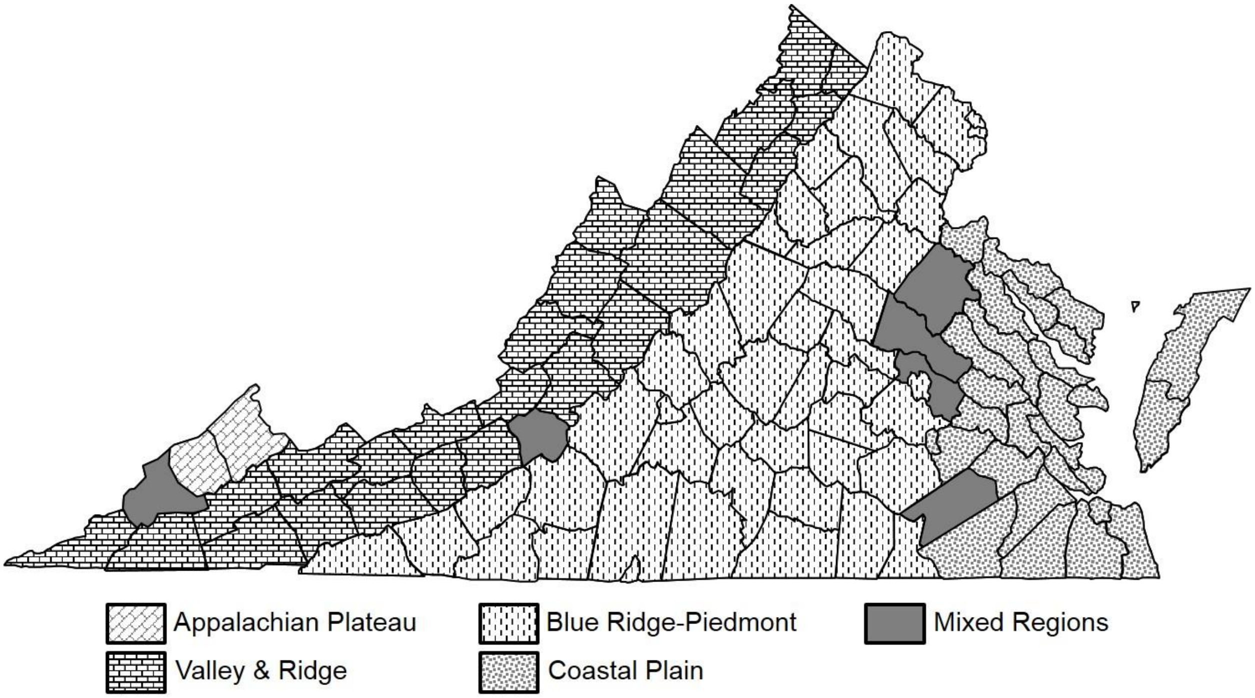

Figure 3-2. Counties in Virginia categorized by underlying geology

Statistical Approach

All statistical analyses were conducted in R version 3.0.2 (R Development Core Team, 2012) assuming an alpha of 0.05 as an indication of significance, unless otherwise noted. As the data was not normally distributed (Shapiro-Wilk; $p<0.05$), the nonparametric Kruskal-Wallis test was used to compare distributions of lead and sodium concentrations with respect to geology and/or system type. Due to the high percentage of non-detects, the test of equal proportions was used to compare rates of total coliform bacteria. The Bonferroni correction was selected as a post-hoc analysis for both analyses. To evaluate the interaction between geology, private system type, and presence of a specific contaminant, three dimensional contingency tables were used (Zeileis, Meyer, & Hornik, 2007). This strategy measured associations between the three variables using the Chi-Squared Test of Independence and mosaic plots were used to visually display the independence models.

Results and discussion

Participation in water sampling

Between 2012 and 2014, 2899 homeowners from across the state submitted point of use private water system samples to the Virginia Household Water Quality Program, which does not include the 213 “mixed” samples. As only 10 homeowners submitted samples from the Appalachian Plateau, this region was not included in statistical analyses. Participation was highest in the Blue Ridge-Piedmont (n=1428), with a substantial amount of participation in the Valley & Ridge (n=927) and Coastal Plain (n=534).

Drilled wells were the most prevalent system type represented in this dataset, though system characteristics (e.g. depth) varied based on geologic region (Table 3-2). Drilled wells accounted for 82% of systems in the Valley & Ridge as compared to the 63% in the Coastal Plain. The mountainous topography of the Valley & Ridge is not ideal for shallow well construction, as reflected by the limited number of dug/bored wells documented, although it is uniquely suited for spring development (5% of systems). Dug/bored wells were primarily constructed in the surficial aquifer in the Coastal Plain and the overburden in the Blue Ridge-Piedmont. Although dug/bored wells are often considered a "dated" technology, it is worth noting that these wells are still being constructed, as evidenced by submitted construction dates as recent as 2011.

Table 3-2. Private system types and characteristics by geologic region

	Coastal Plain	**Blue Ridge-Piedmont**	**Valley & Ridge**
Total participants	534	1428	927
Drilled well	63% n=338	75% n=1065	82% n=756
Depth (m, median; 5th-95th percentile)	91.4 30.0-223.4	76.2 24.4-152.4	76.2 27.0-198.1
Year constructed (median; min-max)	2000 1945-2014	1995 1874-2014	1989 1900-2014
Dug/bored well	16% n=86	12% n=171	3% n=24
Depth (m, median; 5th-95th percentile)	11.0 4.8-57.9	16.6 9.1-61.0	30.5 3.7-100.6
Year constructed (median; min-max)	1979 1945-2011	1979 1876-2011	1973 1850-1990
Spring	<0.5% n=2	4% n=53	5% n=50
Unknown well type	17% n=91	8% n=120	9% n=79
Other or unknown system type	3% n=17	1% n=19	2% n=18

Overall private system water quality

Consistent with past literature assessing water quality at the point of use (Knobeloch, et al., 2013; Swistock, et al., 2013), approximately 55% of systems sampled (n=1586) exceeded at least one SDWA health-based standard and 21% (n=620) exceeded at least two SDWA health-based standards (Table 3-3). In keeping with observations from studies nationally, the most prevalent contaminant observed was total coliform bacteria (42% positive). Lead was the second most common source of health standard exceedance, with 18% of sample concentrations above the EPA action level of 15 µg/L. Although there are additional private system surveys addressing water quality within the well and source aquifers, this work focuses on point of use exposure, which includes potential waterborne contamination from components within the distribution system (e.g., plumbing, water treatment). To illustrate, while DeSimone (2009) reported that approximately 34% of wells tested positive for total coliform, no wells sampled had lead in water concentrations above 15 µg/L, which was most likely due to the collection of samples at the well head (i.e., point of entry). Therefore, to demonstrate the influences and interactions of geology and private system construction characteristics on water quality observations at the point of use, the incidence of these two most common contaminants of concern was explored further, along with sodium, which is frequently identified by health practitioners as of emerging concern in the coastal region.

Table 3-3. Water quality parameters reported by geologic region and private system type

Target water quality constituent	**SDWA Standard**	***All samples***	***Geologic region***			***Type of private system***		
			Coastal	**Blue Ridge-**	**Valley &**	**Drilled**	**Dug/bored**	**Springs**

				Plain	Piedmont	Ridge	wells	wells	
Exceeded at least one MCL Exceeded at least two MCLs			55% 21% n =2889	38% 12% n =534	60% 25% n =1428	58% 21% n=927	51% 17% n =2159	79% 45% n=281	90% 55% n =105
[†]Cadmium, in mg/L	MCL	0.005 mg/L	0.6% n =2886	0.2% n =534	0.7% n =1428	0.8% n=924	0.5% n =2158	1.1% n=281	0.0% n =103
[†]Chromium, in mg/L		0.1 mg/L	0.0% n =2886	0.0% n =534	0.1% n =1428	0.0% n=924	0.0% n =2158	0.0% n=281	0.0% n =103
[‡]Fluoride, in mg/L		4.0 mg/L	0.5% n =2889	2% n =534	0.0% n =1428	0.3% n=927	0.6% n =2159	0.4% n=281	0.0% n =105
[‡]Nitrate, mg/L		10 mg/L	2% n =2889	2% n =534	2% n =1428	2% n=927	2% n =2159	4% n=281	1% n =105
[‡]Total coliform, in cfu		ABSENT	42% n =2885	31% n =533	40% n =1426	50% n=926	37% n =2156	69% n=281	85% n =105
[‡]*E. coli*, in cfu		ABSENT	9% n =2885	4% n =533	7% n =1426	14% n=926	6% n =2156	15% n=281	40% n =105
[†]Copper, in mg/L	Action Level	1.3 mg/L	12% n =2886	6% n =534	19% n =1428	4% n=924	10% n =2158	27% n=281	17% n =103
[†]Lead, in mg/L		0.015 mg/L	18% n =2886	9% n =534	25% n =1428	13% n=924	17% n =2158	28% n=281	16% n =103
[‡]Fluoride, in mg/L	SMCL	2.0 mg/L	3% n =2889	12% n =534	0.5% n =1428	1.6% n=927	3% n =2159	1.1% n=281	2% n =105
[‡]Iron, in mg/L		0.3 mg/L	9% n =2889	11% n =534	9% n =1428	9% n=927	9% n =2159	8% n=281	11% n =105
[‡]Manganese, in mg/L		0.05 mg/L	10% n =2889	12% n =534	10% n =1428	9% n=927	10% n =2159	11% n=281	7% n =105
[‡]pH, in standard units		6.5-8.5	31% (27%[§]) n =2889	30% (14%[§]) n =534	45% (44%[§]) n =1428	8% (7%[§]) n=927	21% (19%[§]) n =2159	66% (64%[§]) n=281	47% (46%[§]) n =105
[‡]Sulfate, in mg/L		250 mg/L	1.5% n =2889	0.2% n =534	1.3% n =1428	3% n=927	2% n =2159	0.0% n=281	0.0% n =105
[‡]Total Dissolved Solids (TDS), in mg/L		500 mg/L	9% n =2889	12% n =534	1.8% n =1428	17% n=92	10% n =2159	4% n=281	2% n =105
[‡]Sodium, in mg/L	DWEL	20 mg/L	31% n =2889	62% n =534	13% n =1428	40% n=927	34% n =2159	12% n=281	5% n =105

MCL: Maximum Contaminant Levels – associated with risk to human health
SMCL: Secondary Maximum Contaminant Levels – associated with aesthetic considerations
DWEL: Drinking Water Equivalency Level (or guidance level)
cfu: colony forming units
[†]Measured in the first draw
[‡]Measured in the flushed sample
[§]Percent below pH of 6.5

Presence of total coliform bacteria (TC)

The presence of total coliform bacteria (TC) has generally been associated with shallow systems (e.g. dug wells, springs), given the high probability of surface-groundwater interaction (Allevi et al., 2013; Gonzales, 2008). Appropriate well construction and maintenance characteristics (e.g. presence of grout, sanitary well caps) designed to minimize surface water contamination are noted as effective in reducing contamination risk (Gonzales, 2008); however, contamination cannot be prevented solely through construction practices when bacteria are ubiquitous in the aquifer (Swistock & Sharpe, 2005). This study observed that the overall presence of TC was significantly higher (test of proportion, $p<0.05$) in springs (85%, n=105) compared to dug/bored wells (69%, n=281), which was significantly higher than drilled wells (37%, n=2156). Presence of TC also varied significantly between geologies; the Valley & Ridge had a higher percentage of exceedance (50%, n=926) as compared to the Blue Ridge-Piedmont (40%, n=1426) and Coastal Plain (31%, n=533).

Interactions between geology, private system type, and TC were evaluated with a three-dimensional contingency table, with results illustrated via a mosaic plot (Figure 3-3). In a mosaic plot, the width and height of each cell are proportional to the number of observations reported, and the color describes the association between the variables. For example, 18% of households during this study were from the Coastal Plain region (n=533; height of "CP" cells), 63% of these 533 observations were from drilled wells (n=337; width of "Dr" cell), and 20% of the 337 observations tested positive for TC (n= 68; height of bottom "CP" cell). The gray color of this cell indicates that there were fewer observations reported than expected under the assumption of independence, i.e. if these three variables were truly independent, there would have been a larger number of TC positive observations in drilled wells in the Coastal Plain. Red cells, as opposite to gray, imply that there were more observations reported than expected under the assumption of independence, and white cells show that the variables are independent. Lastly, darker colors note a stronger deviation from independence (i.e. larger magnitude of more/less observations reported than expected).

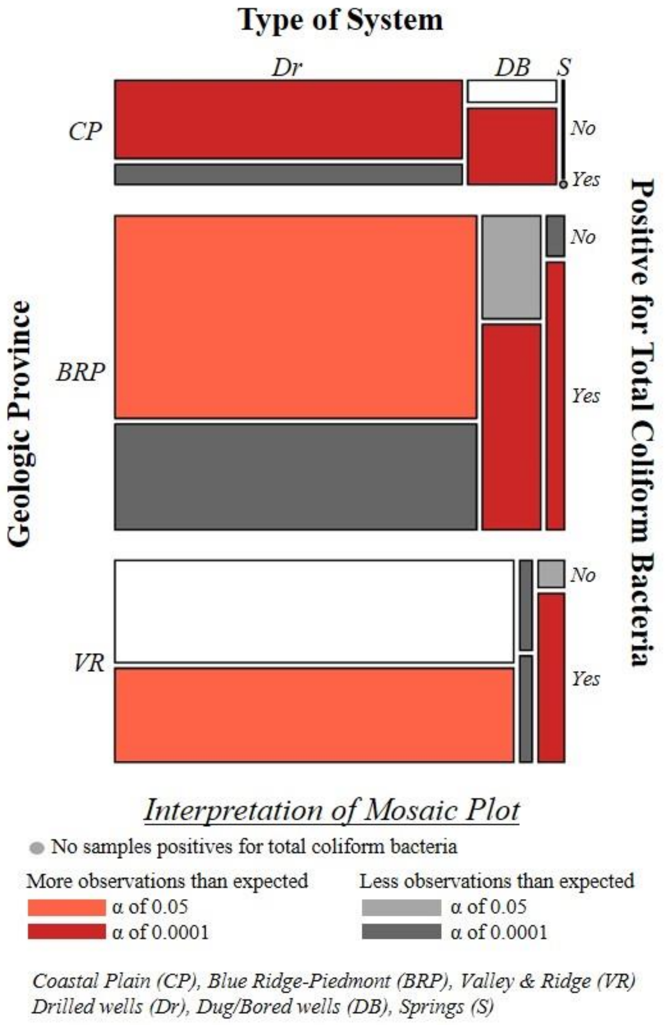

Figure 3-3. Presence of total coliform bacteria by geology and system type

Evaluating the interactions, trends in the presence of TC by private system type generally held true within regions (Figure 3-3). In the Coastal Plain, there were fewer TC positive samples in drilled wells than expected (20%, n=337), but more TC positive observations than expected in dug/bored wells (78%, n=86). This was also visible in the Blue Ridge-Piedmont, as springs (87%, n=53) and dug/bored wells (67%, n=171) had higher TC positive rates than expected and drilled wells had fewer positives (34%, n=1064). These results confirm previous reports that the presence of total coliform bacteria is more often associated with shallow systems, and highlights the importance of surface-groundwater interactions and shorter travel times associated with shallow depths. However, in the Valley & Ridge, springs (86%, n=50) and drilled wells (48%, n=755) both had more TC positive samples than expected. Solution channels within the carbonate geology in this region can readily transmit surface water to deeper depths, which allows TC to be observed in deeper wells. In addition, drilled wells in the Valley & Ridge had a significantly higher percentage of households with bacteria (test of proportion, $p<0.05$) compared to drilled wells in Blue Ridge-Piedmont, and the Blue Ridge-Piedmont had a significantly higher percentage than the Coastal Plain, which further emphasizes the importance of the groundwater flow paths (e.g. solution channels, fracture). There were fewer TC negative and positive samples than expected for dug/bored wells in the Valley & Ridge, which was attributed to fewer observations of this well type than expected assuming independence (n=24).

Lead concentrations

Higher lead concentrations observed in shallow systems (i.e. dug/bored wells and springs) have been attributed to the acidic nature of shallow groundwater, which is generally more corrosive (Pieper, et al., 2015; Swistock, Sharpe, & Robillard, 1993). Given previous knowledge of differences in groundwater acidity, it is not surprising that observed median lead concentrations were significantly different based on geology (Kruskal-Wallis, $p<0.05$). Lead concentrations in the Blue Ridge-Piedmont (5.6 µg/L, n=1428) were significantly higher than the Valley & Ridge (2.8 µg/L, n=925), which was significantly higher than the Coastal Plain (1.1 µg/L, n=534). Median concentrations were also significantly higher in dug/bored wells (7.6 µg/L, n=281) compared to drilled wells (3.6 µg/L, n=2158). Note that as lead in drinking water is primarily attributed to the corrosion of plumbing components (Triantafyllidou & Edwards, 2012), springs were not included in this analysis as the type of spring (i.e. roadside collection vs. spring box) was not documented.

Trends in observed lead concentrations differed greatly by region (Figure 3-4). In the Coastal Plain, there were fewer drilled wells exceeding the action level than expected (median <1 µg/L, n=338), but rates were higher than expected in dug/bored wells (median 9.0 µg/L, n=86). This can be directly linked to more aggressive water observed in dug/bored wells (median pH of 6.0 as compared to 8.2 for drilled wells), noting the importance of surface-groundwater interactions. In the Blue-Ridge Piedmont, there were more observations exceeding the action level than expected for both dug/bored wells (median 7.4 µg/L, n=171) and drilled wells (median 5.5 µg/L, n=1065). Acidic groundwater was observed in the shallow dug/bored wells (median pH of 6.2) and the deeper drilled wells (median pH of 6.7) in this region, which was attributed to the lack of buffering capability and fractured groundwater flow in the crystalline bedrock. Understanding differences in lead observations in the Valley & Ridge proved most complex: the carbonate bedrock can buffer the acidic groundwater, but solution channels can also readily transmit acidic groundwater to deeper depths. There were fewer observations of drilled wells in the Valley &

Ridge exceeding 15 µg/L than expected (2.9 µg/L, n=755). In addition, drilled wells in this region had a statistically lower median concentration (Kruskal-Wallis, $p<0.05$) than drilled wells in the Blue Ridge-Piedmont, which is consistent with knowledge of the buffering capabilities of these geologies (median pH of 7.3 in the Valley & Ridge). However, drilled wells in the Valley & Ridge had a statistically higher median concentration (Kruskal-Wallis, $p<0.05$) than drilled wells in the Coastal Plain, which highlights the importance of the groundwater flow paths, i.e. solutions channels in this region. Again, the relatively low number of dug/bored wells documented in the Valley & Ridge (n=24) limited analysis, but the median pH and lead concentration were 7.1 and 4.5 µg/L, respectively.

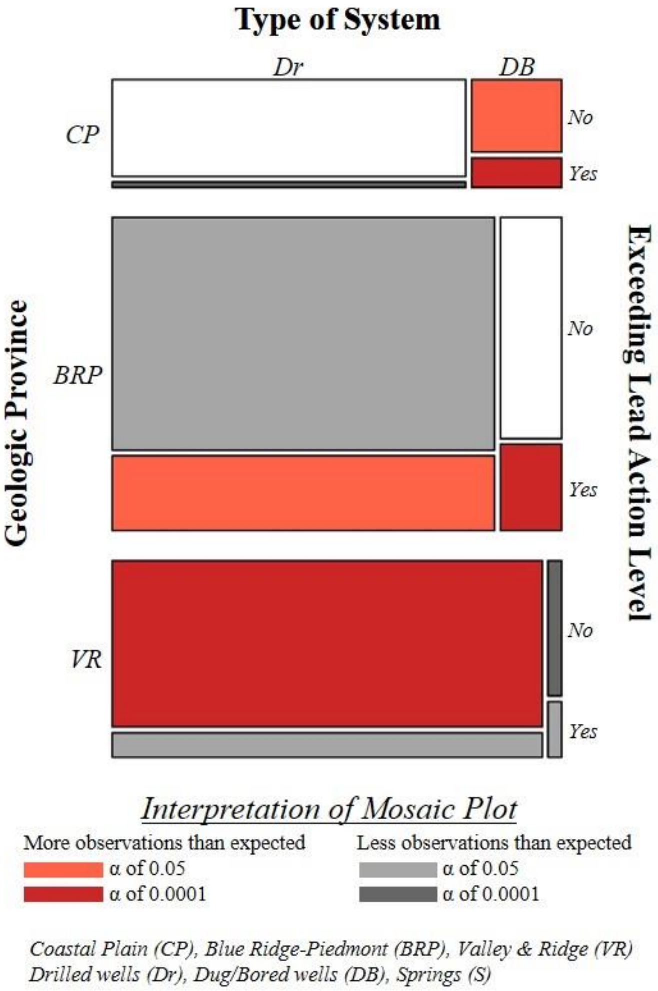

Figure 3-4. Lead concentrations by geology and system type

Sodium concentrations

Sodium is not associated with a maximum contaminant level (MCL), although the USEPA does provide a guidance level of 20 mg/L (U.S. Environmental Protection Agency, 2003). Elevated sodium concentrations in private systems have been associated with the use of water softeners, presence of sewage, application of road salt, and saltwater intrusion (Schmalzried & Keil, 2008; University of Rhode Island Water Quality Program, 2013). Median sodium concentrations in this study varied significantly by geologic region (Kruskal-Wallis, $p<0.05$); the Coastal Plain had a significantly higher median concentration (40.6 mg/L, n=534) compared to the Blue Ridge-Piedmont (6.7 mg/L, n=1428) and the Valley & Ridge (9.2 mg/L, n=927). Sodium also varied significantly across all system types (Kruskal-Wallis, $p<0.05$). Drilled wells had a significantly higher median concentration (9.0 mg/L, n=2159), while dug/bored wells (6.3 mg/L, n=281) and springs (2.6 mg/L, n=104) had a significantly lower concentration.

Drilled wells in the Coastal Plain appeared to be influenced by potential saltwater intrusion, as there were more observations of elevated sodium concentrations in drilled wells than expected (61.8 mg/L, n=338) and fewer observations than expected in dug/bored wells (5.6 mg/L, n=86; Figure 3-5). The median sodium concentration in drilled wells was also significantly higher (Kruskal-Wallis, $p<0.05$) than dug/bored wells. However, it is worth emphasizing that high rates of elevated sodium concentrations were also observed in wells in the Valley & Ridge, mostly likely due to the common use of water softeners in this region. When the acidic groundwater dissolves carbonate bedrock, calcium and/or magnesium become soluble, resulting in increased water hardness and accompanying aesthetic concerns; consequently, 47% of households with drilled wells in the Valley & Ridge reported using a water softener. As water softeners rely upon the addition of sodium to precipitate out unwanted ions, the median sodium concentrations in drilled wells was 10.7 mg/L (n=756) and not surprising, there were more drilled wells exceeding the guidance level than expected under the assumption of independence. In contrast, springs had lower rates of exceedance than expected and a median concentrations of 1.2 mg/L (n=50), which further supports the use of water softeners as a primary cause of exceedance of the EPA guidance level in the Valley & Ridge. Systems in the Blue Ridge-Piedmont had fewer observations of elevated sodium concentrations than expected; however, 14% of systems exceeded 20 mg/L, which again, may be due to the use of water softeners that target iron removal.

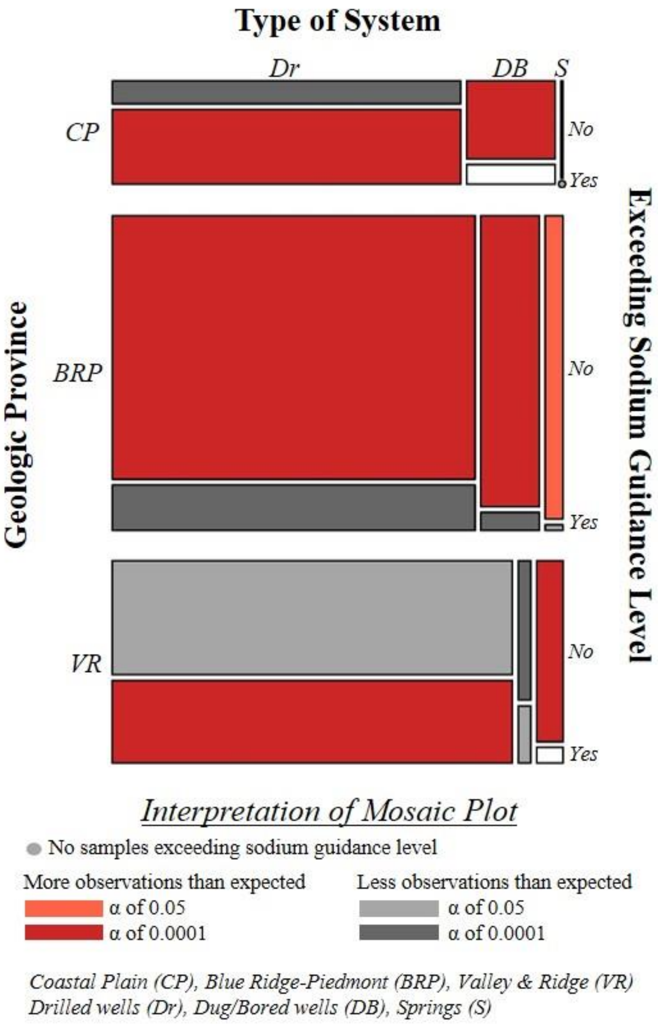

Figure 3-5. Sodium concentrations by geology and system type

Conclusion

This study aims to provide information and insights that can be communicated to private system homeowners in order to improve their understanding of private systems and potentially increase groundwater stewardship. Using lead, sodium, and total coliforms, the most common concerns in Virginia's private systems, this study demonstrates that: (1) geology in large measure dictates groundwater flow (e.g., solution channels, fractures), which influences the probability of surface-groundwater interactions; (2) the type of system constructed determines the source aquifer, which governs the raw groundwater quality being supplied to the household; and (3) the maintenance and/or monitoring of treatment devices can introduce water quality constituents that are not naturally present in raw, untreated groundwater. It is important to understand factors associated with different types of contaminants (e.g., bacteria, metals) in order to design interventions and educational materials to ensure appropriate actions at the household level.

The long-term maintenance and monitoring of private systems is solely the prerogative of individual homeowners, but is fundamental in maintaining water quality and protecting community health. To engage and empower homeowners, updated literature and resources highlighting the importance of testing and outlining risk factors associated with private water quality are essential. However, it is worth noting that approximately 8-17% of homeowners surveyed in this study did not indicate their well type, which is troublesome given the importance of well characteristics in remediation strategies. Therefore, educational resources addressing characteristics of private system and resulting water quality are also needed. This may continue to be presented through well-established cooperative extension efforts and state agencies, but also via the medical community as several of the waterborne contaminants are of health concern. It is imperative in all these efforts that materials be presented at appropriate reading and comprehension levels (Roy et al., 2015).

Additionally, more effort to characterize water quality in private water systems at the national level would provide context to this and similar recent studies, and serve to educate environmental health practitioners and regulators. The most comprehensive national study of private systems was conducted by the EPA over thirty years ago (Francis et al., 1982); recent studies suggest issues identified in the 1980s still persist and water quality in private systems is increasingly recognized as an area of environmental health concern (Backer & Tosta, 2011; Craun et al., 2010). Extension and state organizations must continue providing low-cost water testing surveys, but coordination of these efforts at a national level may provide a useful first step in understanding private water and improving recommendations communicated to private system homeowners.

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CHAPTER 4. PROFILING PRIVATE WATER SYSTEMS TO IDENTIFY PATTERNS OF WATERBORNE LEAD EXPOSURE

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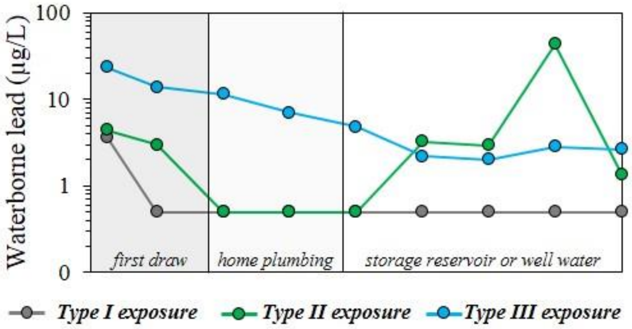
Status: Under committee review

Abstract



Although extensive literature documents corrosion in municipal water systems, only minimal data is available describing corrosion and metals release in private water systems (e.g., wells) that serve as a primary source of drinking water for approximately 47 million Americans. This study developed a profiling sampling strategy specifically tailored to evaluate lead release in these systems. When applied in a field study of 15 private systems, three patterns of lead release were documented: no elevated lead or lead elevated in the first draw only (Type I), erratic spikes of particulate lead (Type II), and sustained detectable lead concentrations (Type III). While flushing protocols as short as 15-30 seconds may be sufficient to reduce lead concentrations below 15 µg/L for Type I and III exposure, flushing may not an appropriate remediation strategy for Type II exposure. In addition, the sustained detectable lead concentrations observed with Type III exposure likely result from corrosion of components within the well, and therefore cannot be reduced with increased flushing. As profiling techniques are labor and sample intensive, recommendations for simpler sampling schemes for initial private system surveys aimed at quantifying lead and protecting public health are discussed.

Introduction

Contaminants of human health concern in private water systems (e.g., wells, springs) are increasingly recognized as a source of potential elevated health risk.[1-3] The proportion of waterborne disease outbreaks in private systems since 1971 continues to increase relative to public systems.[2] This is not necessarily surprising as private water systems (i.e., systems with less than 15 service connections and serve less than 25 individuals for at least 60 days per year), do not fall under the regulatory authority of the U.S. Environmental Protection Agency (USEPA) and are therefore not subject to the protections offered by the Safe Drinking Water Act (SDWA).[4] Approximately 15% of U.S. households serving more than 47 million residents, are currently reliant on these unregulated private water systems,[4] which are generally constructed in rural communities where municipal water service is not available.[5,6]

Preliminary efforts to survey water quality in private systems report that 23-58% of systems exceed at least one SDWA health-based standard.[5-7] However, waterborne metals in private systems related to the corrosion of plumbing components is only rarely assessed because there are no enforced regulations, sampling protocols have been designed for public utilities, and the financial burden falls completely on private system homeowners. The limited research available indicates that 12-19% of private systems sampled exceed the USEPA lead action level of 15 µg/L.[6-8] Observations of waterborne lead concentrations measured in "first draws" (e.g., samples collected following six hours' stagnation) as high as 2,800 µg/L in Pennsylvania and 24,740 µg/L in Virginia have been recorded. For perspective, a lead concentration of 5,000 µg/L or higher qualifies landfill leachate as hazardous waste.[9]

Under the Lead and Copper Rule (LCR), the USEPA monitors and regulates lead concentrations in municipal systems by collecting first draw samples in consumers' homes after a minimum of six hours of stagnation.[10] If more than 90% of the samples collected exceed the lead action level of 15 µg/L, the municipality must implement corrosion control techniques such as addition of corrosion inhibitors and/or adjustments of water quality parameters (e.g., pH, alkalinity) to reduce lead leaching to water from plumbing. Corrosion control strategies are not common in private systems, where the decision to install treatment is solely the prerogative of the homeowner. A recent survey of over 2,000 private systems in Virginia indicated that only 8% of systems had treatment devices that either adjusted water chemistry to control lead release or removed lead through filtration,[7] despite the fact that typical source water (e.g., groundwater) can be extremely aggressive (i.e., corrosive). Previous studies in Virginia, Pennsylvania, and New Jersey report that 18-44% of private systems' water samples are below the USEPA recommended minimum pH of 6.5,[6,7,11] which is an indicator of potential lead leaching from plumbing.[12,13]

The USEPA encourages homeowners with private wells to run the faucet for a few seconds to as long as two minutes to flush any contaminated stagnant water within the plumbing network to prevent waterborne lead exposure; however, this recommendation is only suggested as necessary for four weeks following installation of a new pump or brass faucet.[14] Although flushing may be a low-cost remediation strategy for many private systems, as elevated lead is often only observed in the first draw,[7] the recommended flushing protocols vary greatly with little confirmation of their effectiveness in reducing lead exposure. The CDC recommends a 1-2 minute flush, while Virginia Cooperative Extension suggests a minimum of 5 minutes.[15,16] Recent work targeting municipal systems suggest that effective flushing protocols need to be tailored for specific plumbing configurations and may never be entirely adequate in mitigating health risk if particulate lead is mobilized, which is associated with semi-random concentration spikes.[17-22] Particulate lead has been observed in private systems at levels as high as 99% of the total lead concentration, and, as in municipal systems, its occurrence appears to be sporadic and difficult to predict.[7,8]

Collection of a first draw grab sample only indicates if a given outlet releases an unacceptable lead concentration, and it provides little insight to water conditions in the remainder of the plumbing system.[23] To assess lead concentrations throughout plumbing systems, recent studies have used profiling techniques (i.e., analysis of sequential samples) to provide an in-depth understanding of the sources of lead to potable water.[17,18,22]

This research advances knowledge regarding lead contamination in private systems and develops management and sampling strategies to protect the health of consumers by: 1) developing a profiling sampling technique specifically tailored to private water systems; 2) investigating variations in waterborne lead release using this sampling protocol in 15 homes in Virginia; and 3) quantifying the potential magnitude and patterns of waterborne lead release in these private systems.

Methods

Study participants

Participants were recruited through the Virginia Household Water Quality Program (www.wellwater.bse.vt.edu), a Virginia Cooperative Extension program that offers water quality testing to private system homeowners in Virginia. During 2012 and 2013, 31% of 250 mL first draw samples submitted from participating homes in Albemarle County (37 of 119) contained lead concentrations above the USEPA action level of 15 µg/L. In late 2013 and early 2014, these homeowners were asked to participate in a complementary water quality follow-up study to investigate potential sources of waterborne lead. Participation was wholly voluntary, and 15 of the 35 homeowners contacted participated. Two of the 37 homeowners had incomplete contact information.

Profiling sampling protocol

Profiling the home plumbing system

Site visits were scheduled in advance to ensure that water remained stagnant for a minimum of six hours prior to sampling per LCR protocol. At each household, the volume of water between the kitchen faucet and pressure tank was estimated based on inspection of the plumbing and is referred to as volume ‘Z’ (1 to 3 L). To prevent the well pump from engaging every time a faucet is opened, private systems typically use a pressure tank that maintains water pressure within the home plumbing. While pressure tanks extend the life of well pumps, their presence interrupts traditional profiling sampling techniques, as these tanks store large volumes of water. Therefore, this profiling method controls water flow from the pressure tank to detect lead released from within the home plumbing system (e.g., brass, solder), as well as lead potentially derived from well components (e.g., galvanized iron, brass).

With electricity to the well turned off and flow driven solely by the pressure tank, a 250 mL and 750 mL first draw sample (Figure 4-1a, C1 and C2) were collected at full flow from the kitchen cold-water faucet to compare first draw results to the previous 2012/2013 extension effort and the LCR. Immediately after, consecutive 1 L samples were collected until ‘Z’ liters were collected. Two additional 1 L samples were then collected to evaluate water quality within the pressure tank and allow for error in volume estimates. These samples (C1-P2) quantified lead release within the home plumbing system. Subsequently, the kitchen cold-water faucet was flushed until the pressure tank read below 20 psi, which is below the common pump cut-in pressure of 30-40 psi.

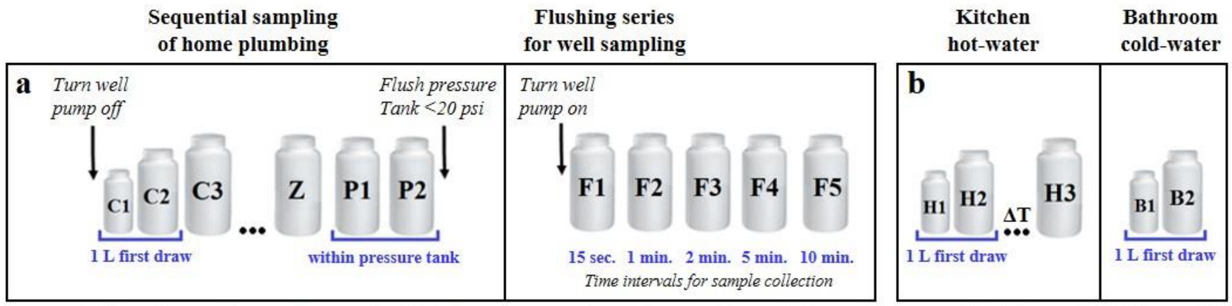


Figure 4-1. Illustration of the profiling protocol developed for private systems: (a) sequential samples were collected from the kitchen cold-water faucet, with the number of samples determined from the volume of water within the home plumbing (=Z liters). A series of flushed sampled were subsequently collected to quantify lead release from the well; and (b) samples collected from the kitchen hot-water faucet and bathroom cold-water faucet after completion of the kitchen cold-water faucet profile.

Flushing series to sample water within the well

With the electricity restored and the pump turned on, a series of samples were collected to detect lead in water within the well. It was assumed there would be relatively little contamination from the home plumbing as the supply to the house would be primarily fresh source water. Samples were collected after flushing the kitchen faucet for 15 s, 1 min, 2 min, 5 min, and 10 min (Figure 4-1a, F1-F5). As the average pressure tank fills within minutes, these flushed samples cannot be considered representative of plug flow. After the flushed samples were collected, two first draw samples (250 and 750 mL) were collected from the bathroom cold-water faucet (Figure 4-1b, B1 and B2) and kitchen hot-water faucet (H1 and H2) at full-flow to assess the influence of faucet ratings (i.e., drinking versus non-drinking water faucets) and temperature on lead release. The hot-water faucet was flushed until the temperature increased and an additional 1 L sample was collected (H3) to assess water quality within the water heater.

Sampling in households with treatment devices with storage reservoirs

The profiling method was modified for households that had a treatment device with a storage reservoir (e.g., acid neutralizer, water softener). Due to the volume of stored water within the treatment unit, sampling water upstream of the pressure tank would not have been possible without extensive flushing. Therefore, in households with such treatment (n=7), after the sequential sampling of the home plumbing system, the flushing series began immediately, i.e., the pressure tank was not flushed below 20 psi.

Homeowner interviews and plumbing inspections

In addition to collecting drinking water samples, each homeowner was briefly interviewed to document behaviors since their initial water testing and describe previous participation in the extension effort (e.g., faucet sampled). To document plumbing materials (e.g., copper piping) and plumbing characteristics (e.g., size of pressure tank), the plumbing network was visually inspected at the kitchen and bath faucets, pressure tank, and well head.

General water quality analyses

Temperature and pH were recorded in the field using a Hanna Instruments HI 98129 probe per method 4500-H^+.[24] Alkalinity was measured within two weeks of collection per method 2330 B.[24] To quantify dissolved metals, a 10 mL aliquot was pipetted from each sample and filtered through a 0.45 μm filter on site. For metals analysis, samples (unfiltered and filtered) were acidified with 2% nitric acid and digested for a minimum of 16 hours prior to analysis using Inductively Coupled Plasma – Mass Spectrometry (ICP-MS) per methods 3030 D and 3125 B.[24] Blanks and/or spikes of known concentrations were processed every 10 samples for all analyses for data quality assurance and quality control (QA/QC) purposes.

Statistical analyses

Statistical analyses were conducted in R version 3.0.2 (R Development Core Team, 2012) assuming an alpha of 0.05 as an indication of significance. Due to the non-normal distribution (Shapiro-Wilk, $p<0.05$), the Wilcoxon signed-rank test was used for statistical analyses, including an unpaired test of 250 mL first draw lead concentrations (Figure 4-1a, C1), alkalinity, and pH measurements between households with and without an acid neutralizer installed; a paired test of lead concentrations in the 250 mL and 750 mL first draw samples (Figure 4-1a, C1 and C2) based on sampling locations (e.g., kitchen cold-water vs. hot-water); and a paired test to evaluate differences in lead concentrations in the 250 mL first draw between the extension effort and the follow-up study.

Results

First draw lead concentrations

During the original 2012/2013 extension clinics, 37 homeowners in Albemarle County, VA submitted first draw samples with lead concentrations that exceeded the USEPA action level. Of these, 15 participated in this follow-up study. The mean and median first draw concentrations for samples from these 15 households during the extension effort were 37.5 and 27.6 μg/L, respectively. Despite pre-selection of households with a history of "high" lead, only two had 250 mL first draw samples that contained lead above 15 μg/L during the follow-up (23.4 and 26.0 μg/L); the mean and median concentrations for all first draw samples were 7.6 and 4.2 μg/L. As a result, the 250 mL first draw samples collected during this follow-up study had significantly lower concentrations compared to the extension effort (Wilcoxon paired test, $p<0.05$). The median difference in lead concentrations between the two sampling efforts was -22.3 μg/L, which highlights the semi-random nature of lead release or the potential benefit of remedial measures implemented since the first sampling.

Only one of the two households with lead concentrations above the action level in the 250 mL first draw was also above the action level in the subsequent 750 mL sample (19.6 μg/L). However, a 750 mL sample from a third household exceeded 15 μg/L, although the initial 250 mL first draw sample did not (3.7 μg/L); the concentration in this sample was 54.9 μg/L, which was 87% particulate lead. Mathematically combining the initial 250 mL and 750 mL samples for comparison to the LCR's 1 L collection protocol for municipal systems, these three households would not be considered in compliance, as the equivalent 1 L first draw concentrations were 16.3, 21.2, and 42.1 μg/L.

Reduction in lead due to acid neutralizers

The lower than expected 250 mL first draw concentrations during this follow-up study could be attributed to the installation and/or servicing of treatment devices during the 1-2 year period between the extension effort and the follow-up study, i.e., upon obtaining their original results, homeowners took actions to improve their household water quality (Figure 4-2). During interviews (n=15), three homeowners indicated that they installed acid neutralizers based on their

2012/2013 extension water quality results. In addition, four homeowners serviced (e.g., changed the resin within) their acid neutralizers.

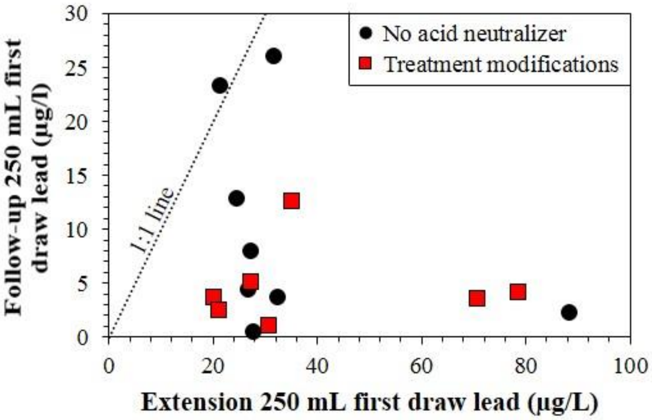


Figure 4-2. Paired 250 mL lead concentrations collected during the 2012/2013 extension effort and the follow-up study

Although there was no significant difference (p=0.35) in median 250 mL lead concentrations in households with an acid neutralizer (n=7; 3.7 µg /L) versus those without (n=8 6.2 µg /L), households with an acid neutralizer had significantly higher alkalinity (p=0.02; 96.9 compared to 21.3 mg/L as $CaCO_3$) and pH measurements (p=0.04; 7.3 compared to 6.3). As the relationship between corrosion and aggressive water is well documented,[12,13] the installation or servicing of acid neutralizers may be partly responsible for the lower observed lead concentrations during this effort relative to the 2012/2013 extension effort.

Reduction in lead due to sampling location

When homeowners (n=15) were asked to identify the faucet used during the 2012/2013 extension clinic, only four homeowners indicated that they collected samples from a drinking water (i.e., kitchen) faucet. Eight homeowners could not remember or reported both the bathroom and kitchen faucets, and three homeowners reported collecting from a non-drinking water faucet (e.g., bathroom faucet). One homeowner collected samples from a laundry sink as this faucet did not have an aerator and did not swivel, per the extension sampling instructions; however, this type of faucet is not approved for dispensing water for human consumption.[25] As faucet fixtures can contribute a large percentage of the total lead in the first draw,[26,27] the influence of the sampling location was further investigated.

Lead release from non-drinking water faucets

During the follow-up study, two first draw samples (250 and 750 mL; Figure 2-1b, B1 and B2) were collected from a bathroom faucet to quantify lead concentrations from a non-drinking water rated faucet. Seven of the 15 bathroom faucets had 250 mL first draw samples that measured lead above 15 µg/L, with mean and median concentrations for all samples of 19.6 µg/L and 13.4 µg/L, respectively. The 250 mL first draws collected from the bathroom faucet had significantly higher lead concentrations than kitchen 250 mL first draws (B1 vs. C1; p=0.03) with a median difference of 2.0 µg/L. As plumbing at the outlet is typically 9.5 mm (3/8 in.) diameter and supply lines are typically 12.7-25.4 mm (½-1 in.) diameter, the first 2.4 m (8 ft.) of plumbing would be approximately 250 mL of water. Therefore, a 250 mL first draw is likely primarily representative of lead release at the outlet only.

Concentrations in the subsequent 750 mL samples were generally much lower (median of 2.3 µg/L), with no significant difference in lead concentrations between the 750 mL bathroom and 750 mL kitchen first draw (B2 vs. C2; p=0.62). As the 750 mL sample was composed of water likely within the supply line rather than the faucet, this result is not surprising, though it does further emphasize the importance of sample location. Lead concentrations in the 750 mL first draw from the bathroom faucet were above the action level in two households (17.7 and 132.5 µg/L). In the household with a 750 mL bathroom sample concentration of 132.5 µg/L, 93% of the lead was in the particulate form.

Lead release from hot-water faucets

During the follow-up study, two first draw samples (250 and 750 mL; Figure 4-1b, H1 and H2) were collected from the kitchen hot-water faucet to evaluate the effect of temperature. Lead concentrations in the 250 mL first draw were above 15 µg/L in two households, with concentrations of 15.7 and 410.7 µg/L. However, the mean and median concentrations for all 250 mL first draw samples were 34.7 and 2.7 µg/L. There was no significant difference between the kitchen hot and cold-water faucets for the 250 mL first draw (H1 vs. C1; p=0.18) or the 750 mL first draws (H2 vs. C2; p=0.89). The subsequent 750 mL samples were generally much lower (median of 3.6 µg/L), but two 750 mL samples had elevated concentrations (15.9 and 122.6 µg/L). In the household with lead concentrations in the first draws of 122.6 and 410.7 µg/L, 98% of the lead was in the particulate form. High percentages of particulate lead was observed in all first draw samples collected from this household.

To assess water quality within the water heater, an additional sample (H3) was collected after flushing the hot-water faucet until the temperature increased (minimum increase of 15.3°C). Two households with recirculating systems were removed from this analysis. Only one water heater sample was above 15 µg/L, with a concentration of 137.2 µg/L, which was 99% particulate; this was not surprising given the corresponding 250 mL and 750 mL first draws were 410.7 and 122.6 µg/L. The overall mean and median concentrations for all hot water samples (n=13) were 13.5 and 1.6 µg/L. Lead concentrations in the 1 L water heater samples were statistically lower than the 250 mL first draws (H3 vs. H1; p = 0.02), but not statistically different from the 750 mL (H3 vs. H2; p=0.94).

Profiling

Past surveys of private systems have focused on quantifying lead in first draw samples as lead is primarily derived from the corrosion of plumbing components and rarely found in groundwater.[6-8,28] Studies have noted the potential for lead leaching from well components[29] although the USEPA suggests that exposure from well components would be minimized (<5 µg/L) after four weeks of use following installation of a well pump.[14]

Based on the 15 households profiled, three patterns of waterborne lead release were identified (Figure 4-3): no elevated lead or lead elevated in the first draw only (Type I), erratic spikes of particulate lead mobilized at semi-random times (Type II), and sustained detectable lead concentrations (≥1 µg/L) even with extensive flushing (Type III). Potential sources of lead release were evaluated by assessing the combinations of waterborne metals (e.g., copper, zinc).

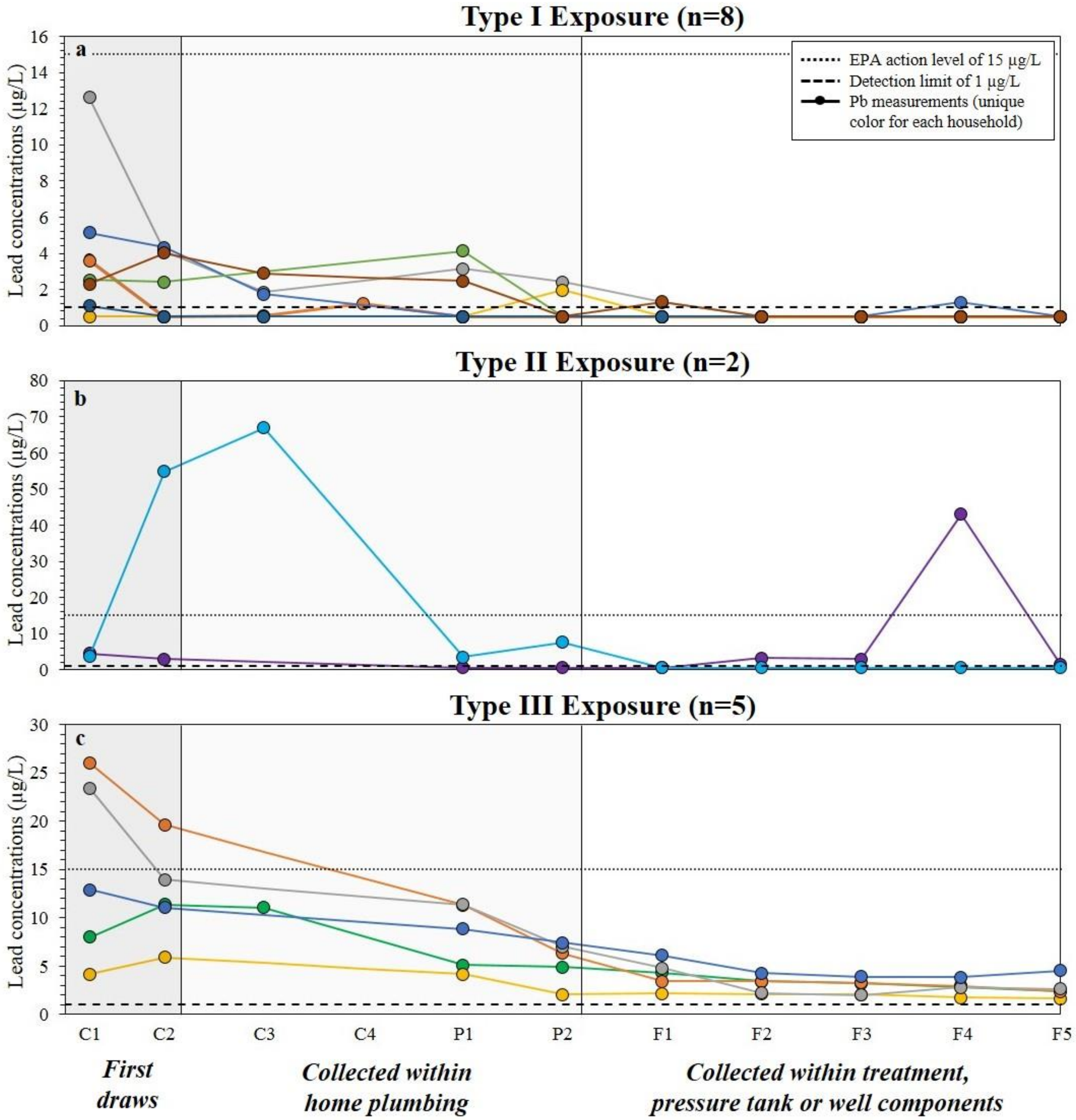


Figure 4-3. Patterns of waterborne lead release: (a) households with no elevated lead or lead elevated in the first draw only (Type I); (b) erratic spikes of particulate lead mobilized from plumbing during water use (Type II); and (c) sustained detectable lead concentrations (≥1 µg/L) even with extensive flushing (Type III). Note: samples C3 and C4 were not collected in all households due to shorter home plumbing lengths observed.

Type I

Conventional wisdom regarding lead exposure (Type I) is that water is safe or hazards are eliminated after a few seconds of flushing at a leaded brass faucet, and that this flushing is needed only for a few weeks after a new faucet is installed. Households identified as having Type I exposure (n=8) had non-detectable lead concentrations during the flushing series (<1 µg/L; Figure 4-3a). Detectable concentrations were observed in the home premise plumbing systems (C1-P2), with the highest concentrations observed in the first draw samples (C1 and C2) for six households. Brass components appeared to be the likely source of lead as water samples often contained elevated levels of copper and zinc (median C1 concentrations of 492 and 1108 µg/L, respectively), which are the three primary metals in brass alloys.[30] This is in keeping with other studies reporting the influence of brass faucet components.[7,27]

Type II

The occurrence of particulate lead (Type II) is highly variable and often associated with higher flow rates and hydraulic disturbances.[19,20,31,32] Samples from two households had lead concentrations above the USEPA action level with the lead completely in the particulate form. While lead concentrations in the home plumbing system of one household were low (C1-P2; max of 4.4 µg/L), there was a lead spike of 43 µg/L (99% particulate) in sample F4 after six minutes of flushing (Figure 4-3b, green). This sample also had higher concentrations of particulate copper and zinc (67.5 and 8.2 µg/L), which suggested sloughing of scale from a brass component(s) as a likely source. In the other household, particulate lead (87-97% composition) was mobilized during the profile, but the samples collected during the flushing series (F1-F5) had non-detectable concentrations (<1 µg/L; Figure 4-3b, purple). Samples C2 and C3 contained high levels of particulate tin and copper (11-14 and 141-170µg/L), which indicated that lead was likely released from solder.[33]

Type III

When components in the well are corroding (Type III), detectable background lead concentrations (≥1 µg/L) are present with continued water use, which constitutes a chronic potential source of exposure. The 250 mL first draw samples from two households in this category exceeded the USEPA action level; the remaining three households did not with concentrations between 4-13 µg/L. In all households, lead levels decreased in subsequent samples, but sustained detectable lead concentrations (2-6 µg/L) were observed throughout the flushing series (F1-F5; Figure 4-3c). These background concentrations were composed entirely of dissolved lead, which suggests collection of stagnant well water rather than mobilization of particulate lead. Concentrations of dissolved copper and zinc were elevated in samples with detectable lead concentrations (F1-F5; median concentrations of 125 and 50 µg/L), which suggests leaching of brass components in the well (e.g., pitless adapters, well pump).

Discussion and implications

Sampling protocols in private water systems

Although there is extensive literature evaluating sampling regimes for the detection of lead in water for municipal systems,[20] sampling protocols investigating private water quality vary considerably (e.g., stagnant vs. purged sampling, dissolved vs. total lead analysis, point-of-use vs. point-of-entry).[6-8,28] These variations render water quality assessments difficult to compare. Concurrent evaluation of a suite of parameters during this study provided a basis for recommending a uniform sampling protocol for application in private systems.

During this follow-up study, all homeowners (n=15) indicated using water at the kitchen faucet for drinking and cooking, but four homeowners also identified bathroom faucets as a possible drinking water source. Previous work has noted the difference in lead concentrations from non-drinking water faucets and potential underestimations when aerators are removed before sampling due to particulate lead in the aerator.[19,27] To accurately assess waterborne lead exposure in private water systems, sampling protocols should be modified to mimic normal consumption behaviors.

Without drinking water standards specific to private water systems, testing efforts often evaluate water quality using SDWA standards for public systems. To assess waterborne lead, the LCR evaluates 1 L first draw samples using the lead action level of 15 µg/L. However in unregulated school systems, the USEPA recommends collection of a 250 mL first draw sample with a threshold of 20 µg/L due to the smaller volume.[23] During this study, the median difference in lead concentrations between the 250 mL and calculated 1 L first draw collected from the kitchen cold-water faucet was 0.6 µg/L (C1 vs. combination of C1 and C2). Two households had 250 mL first draws above 20 µg/L, but three households had 1 L first draw samples exceeding 15 µg/L. First draw samples from bathroom cold-water faucets had a median difference between the 250 mL and calculated 1 L of 2.4 µg/L (B1 vs. combination of B1 and B2). While only two households had 1 L first draw samples that exceeded 15 µg/L, lead concentrations in the 250 mL first draw samples were above 20 µg/L in four households and 15 µg/L in seven households. Therefore, this research suggests that analysis of the 250 mL first draw sample compared to the lead action level of 15 µg/L is sufficiently protective for private water system surveys.

Waterborne lead exposure patterns

Remediation methods

Patterns of waterborne lead health risks at individual households were classified within three major categories: no elevated lead or lead elevated in the first draw only (Type I), erratic spikes of particulate lead (Type II), and sustained detectable lead concentrations (Type III). With the random detachment of particulate lead observed in Type II exposure, flushing may be ineffective in protecting health. For cases in which Type I and III lead release is confirmed, flushing protocols as short as 15-30 seconds are adequate to minimize concentrations below 15 µg/L, as this flushing duration would avoid human exposures to the first draw samples which were typically highest in lead concentration. However, Type III exposures pose a more significant health threat, as chronic lead exposure risks cannot be readily cleared from the line with reasonable flushing times.

Based on the 12 systems in this study where homeowners knew the approximate well depth, the average volume of water stored within the home plumbing system and plumbing connecting to the well pump was 72 L for households without treatment (n=6) and 100 L for households with treatment (n=6), which would require a minimum of 15 and 21 minutes to flush all stagnant water encountering well plumbing (average flow rate of 4.7 L/min). As a result, private systems homeowners would need to flush 2-3 times longer than the recommended flushing protocol for homeowners with a full or partial lead service line. According to the CDC, households connected to a municipal system via a leaded service line should flush a high-volume tap for at least 5 minutes before flushing the kitchen tap for 1-2 minutes.[34] As the average person in the U.S. uses approximately 375 liters of water per day,[35] flushing a private system to access fresh groundwater would account for 20-25% of a homeowner's total water use for the day, and seems impractical.

Acid neutralizers are recommended as a corrosion control option for private systems, however, these treatment devices only increase pH and alkalinity within the home plumbing system (i.e., do not remove lead). One household in this study installed an acid neutralizer after the 2012/2013 extension effort due to a major pipe failure. Despite having a pH of 7.4 and an alkalinity of 96.9 mg/L as $CaCO_3$ after installation, samples collected during the flushing series

(F1-F5) contained detectable lead (2 µg/L). While flushing protocols and acid neutralizers may be an effective remediation for some homeowners on private systems, the installation and maintenance of a certified point-of-use lead filter would provide improved protection for all systems.

Associated health risks

Assuming 15% of U.S. households are reliant on private systems and 19% of these systems have lead concentrations above 15 µg/L,[4,6,7] an estimated 9.1 million private system residents may be at risk of exposure to lead in water above the action level. As the 15 µg/L action level was determined when the CDC blood lead level of concern was 25 µg/dL (micrograms of lead per deciliter of blood), which is five times higher than the current threshold of 5 µg/dL,[36,37] achieving the action level itself might not be sufficiently protective. As cases of elevated blood lead levels (EBLLs; ≥5 µg/dL) in children have been associated with lead contamination from municipal water in the U.S., even in cities with modern corrosion control that meet the 15 µg/L action level,[38,39] characterizing waterborne exposure in private water systems is imperative to protect the public health.

To date, there has been no investigation of the contribution of lead in water to blood lead levels of children reliant on private systems in the U.S. Therefore, to assess the potential health risks associated with private well systems in Albemarle County, VA, observed waterborne lead concentrations were compared to concentrations documented in well systems in a recent study in Tamatave, Madagascar (Figure 4-4). This study in Tamatave documented lead release from well components and potential contribution to EBLLs using the USEPA Internal Exposure Uptake Biokinetic Model for Lead in Children (IEUBK model).[40]

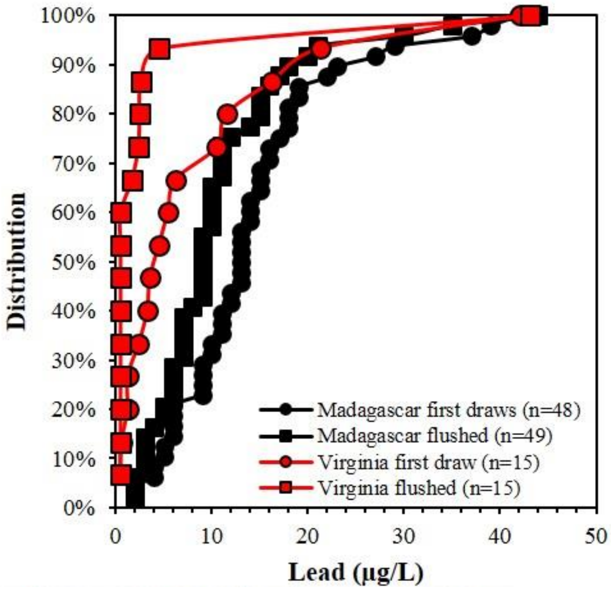


Figure 4-4. Cumulative distribution of waterborne lead concentrations in first draw and flushed samples collected in Albemarle County, VA and Tamatave, Madagascar

Although system construction and demographics are very different between Madagascar and Virginia, the upper 20% of these datasets had similar waterborne lead concentrations in the first draw samples – 80th percentiles were 18.0 µg /L in Tamatave and 15.3 µg/L in Albemarle County. Given that the Madagascar study estimated 15-70% of children in Tamatave were at risk of EBLLs due to lead in water concentrations between 4-54.5 µg/L, observations of EBLLs in children reliant on water from private systems in the United States with similar lead concentrations would not be unexpected. Further field research is encouraged to directly characterize health risks associated with metals exposure for families dependent on private systems in the U.S. to determine whether these estimates are reasonable.

It is also important to note that plumbing components in wells are submerged in groundwater, and as a result, the outer surfaces of these components can corrode and release lead. Based on the data and information provided, flushed samples collected in Tamatave appeared to represent stagnant water in contact with the outer surfaces of plumbing components. Despite flushing 2.5 times the volume of water within the plumbing in the well (average volume of 8.8 L using reported diameter of 26 mm), flushed samples still contained high levels of lead. This occurrence illustrates the potential for the outer surfaces of well components to release lead as well as highlighting the large volume of water associated with wells. Therefore, additional emphasis should be given to characterizing the unique flow patterns associated with private systems as this would aid in the development of remediation strategies.

Acknowledgements

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CHAPTER 5. EVALUATING NSF 61 COMPONENTS WHEN EXPOSED TO WATER REPRESENTATIVE OF PRIVATE SYSTEM

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Abstract

As private water systems generally have more aggressive water (e.g., lower pH and alkalinity), lead leaching potentials from NSF Standard 61 section 9 “lead-free” plumbing components were evaluated when exposed to more aggressive conditions. As expected, brass coupons exposed to aggressive water leached higher lead concentrations, but lead release was minimal from lead-free brass manufactured after 2014. However, the brass ferrules in faucet connectors released high levels of lead in the 30 mL volume. There is concern regarding multiple-material components as the new lead-free regulation allows leaded components as long as the relative wetted surface area is small. Lead release from lead-free galvanized steel was also evaluated as this material can be used for plumbing components in the well. Although observed lead concentrations were minimal, lead appeared to be adsorbing to the iron scale. With lead adsorbing to scale, there may be long-term consequences associated with the mobilization of particulate lead.

Background

Despite the well-known adverse health effects associated with human exposure, lead was a common additive to potable water plumbing components through at least January 2014 (Bowman & Hebenstreit, 2011; U.S. Environmental Protection Agency, 1989). Following the promulgation of the 1986 Lead Ban, the U.S. Environmental Protection Agency (USEPA) required the use of “lead-free” plumbing components in the installation and repair of drinking water infrastructure and on-site plumbing (U.S. Environmental Protection Agency, 1989). At the time, the USEPA defined “lead-free” as solder and flux that contained less than 0.2% lead, and piping and fittings that contained less than 8% lead, which banned the use of leaded solder (50% tin and 50% lead) and pure lead pipes. With the 2011 Reduction of Lead in Drinking Water Act, the allowable lead content in piping and fittings has been reduced to a weighted average of less than 0.25% lead with respect to wetted surfaces (NSF International, 2015).

As even these “lead-free” products still contained lead in small quantities, plumbing components also required certification by NSF International Standard (NSF) Standard 61 (NSF International, 2013) to ensure minimal leaching of health-based contaminants into water intended for human consumption. However, Dudi et al. (2005) demonstrated that a small pure lead pipe could pass NSF certification when lead results were normalized to a 1 L volume per the NSF protocol. Triantafyllidou and Edwards (2007) examined lead release from brass exposed to more aggressive water (i.e., pH = 7.4, alkalinity of 10 mg/L as $CaCO_3$) and observed that lead concentrations were 10 times higher than brass exposed to the test water outlined in the NSF protocol. In addition, as recent studies have identified “lead-free” plumbing components as the likely source of elevated waterborne lead in buildings connected to municipal systems (Boyd et al., 2008; Elfland et al., 2010), the protectiveness of NSF Standard 61 has been questioned.

Introduction

Approximately 15% of U.S. residents rely on private water systems (e.g., wells and springs) as a primary source of household water (U.S. Environmental Protection Agency, 2012). As private systems have less than 15 service connections and serve an average of less than 25 individuals for at least 60 days per year, these systems are not subjected to the standards outlined in the Safe Drinking Water Act, which includes the Lead Ban and Reduction of Lead in Drinking Water Act (U.S. Environmental Protection Agency, 2013). Although the USEPA believes that manufacturers are complying with lead-free regulations for private system well components, the USEPA has also reported that “submersible pumps are still leaching lead into water even though they have no more than 8.0 percent lead” (U.S. Environmental Protection Agency, 1994). While many public systems reduce the aggressiveness of their water through chemical additives such as orthophosphate, these interventions are uncommon in private water systems (Pieper et al., 2015; Swistock et al., 2013). Recent studies in Virginia and Pennsylvania report that 12-19% of samples from private systems exceed the USEPA lead action level of 15 µg/L (Pieper et al., 2015; Swistock et al., 2013). Perhaps not surprisingly, studies report that 18-44% of water from private systems had a pH below the EPA recommended 6.5, with measurements as low as 4.8 (New Jersey Department of Environmental Protection, 2008; Pieper et al., 2015; Swistock et al., 2013). While there is extensive literature on lead release in public water systems, public systems are not an appropriate surrogate for private systems due to the differences in water chemistry and treatment.

NSF certification. The USEPA regulates waterborne lead concentrations in households connected to a municipal system via the Lead and Copper Rule (LCR), which states that no more than 10% of households sampled can exceed the lead action level of 15 µg/L in the 1 L first draw sample (i.e., sample collected after 6+ hours of stagnation) (U.S. Environmental Protection Agency, 1991). As NSF Standard 61 Section 9 evaluates lead leaching from endpoint devices (i.e., components installed within the last 1 L segment of water), the “Q statistic” was developed to determine a component’s contribution to the 1 L first draw sample. When endpoint devices do not have a 1 L capacity and/or there are differences in surface area between the test apparatus and the product, results are normalized to comparable levels using the NSF normalization factor (Equation 5-1; NSF International, 2013).

$$Norm = Obs\ x \left(\frac{SA_F}{SA_L} x \frac{V_L}{V_F}\right) \qquad \text{Equation 5-1}$$

Where Norm are the normalized concentrations per NSF protocol, Obs are the observed concentrations during the NSF experiment, SA_F is the surface area exposed in the field, SA_L is the surface area exposed in the laboratory, V_F is the volume of water in contact with the product in the field, and V_L is the volume of water in contact with the product in the laboratory. Using the normalized lead concentrations, the Q statistic can be calculated (Equation 5-2), which determines acceptability of the product line.

$$Q = e^{\bar{Y}} e^{k_1 S} \qquad \text{Equation 5-2}$$

Where $\overline{Y}$ is the log-dosage mean of the normalized concentrations, S is the log-dosage standard deviation of the normalized concentrations, and k_1 is a NSF coefficient determined by the number of replicates. Previously, the Q statistic was set at a threshold of 11 µg/L, which allowed endpoint devices to release 11 µg lead when normalized to a 1 L first draw sample. As of July 2012, the Q statistic was lowered to a threshold of 3 µg/L for supply stop, flexible plumbing connectors and miscellaneous components, and 5 µg/L for all other endpoint devices, except commercial kitchen devices (NSF International, 2013).

Objective. The synthetic water specified in the NSF protocol for Section 9 end-point devices is non-aggressive with a pH of 8.0 and an alkalinity of 500 mg/L as $CaCO_3$, which is not representative of water quality in private systems. The goal of this study is to: (1) quantify lead release from lead-free brass coupons and faucet connectors; (2) assess potential lead leaching and recovery from lead-free galvanized steel; and 3) determine the associated acceptability (Q statistic) of these products.

Methods

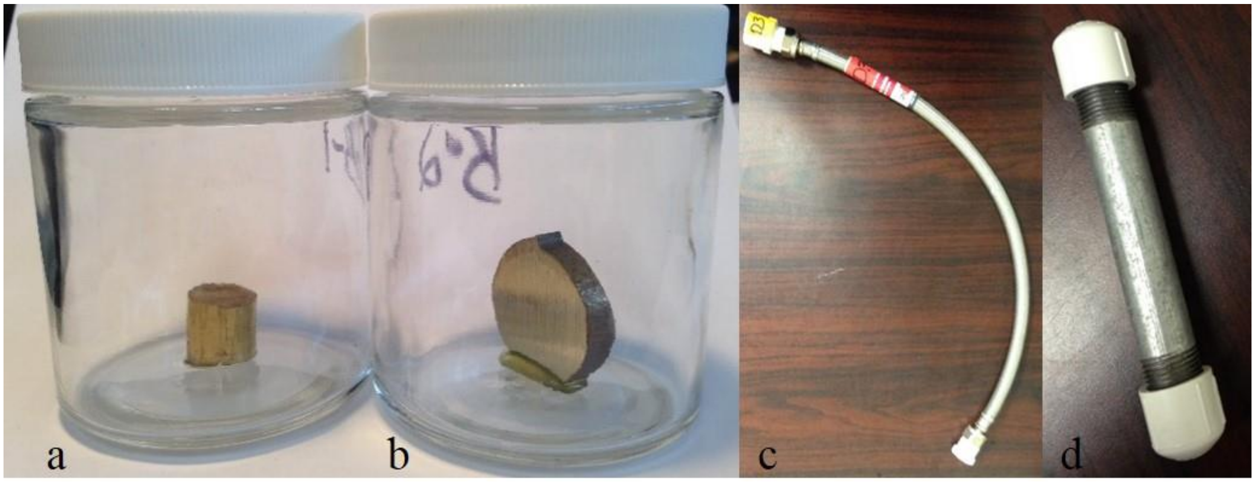
Water quality conditions. The Virginia Household Water Quality Program (www.wellwater.vt.edu) is a Virginia Cooperative Extension drinking water outreach program based at Virginia Tech. Over the past two and a half decades, this extension program has offered private system homeowners low-cost water quality testing for contaminants of health and nuisance concern, including pH and alkalinity, via Standard Methods. Of the 12,766 point-of-use water samples collected by the extension program between 1989 and 2009, 94% of private systems had water with a pH below 8.0 and 99.4% had water with an alkalinity below 500 mg/L as $CaCO_3$.

To evaluate lead release from NSF certified components under water conditions more representative of water quality in private systems, seven water conditions were developed based on the extension dataset (Table 5-1). Consistent with the NSF synthetic water, hardness was set at 100 mg/L for all seven modified water conditions. Although hardness levels varied in the extension dataset, maintaining a constant chloride-to-sulfate mass ratio (CSMR) was important, as CSMR influences corrosion (Edwards & Triantafyllidou, 2007). As private systems typically do not install continuous disinfection treatment devices, a chlorine residual was not included in any water conditions, including the NSF synthetic water.

Table 5-1. Water conditions developed based on extension dataset of 12,766 samples

Condition	Population percentile	pH	Alkalinity (mg/L as $CaCO_3$)	Hardness (mg/L)
A	94 (99.4*)	8.0	500	100
B	75	7.5	175	100
C	50	7.0	80	100
D	25	6.4	30	100
E	10	6.0	15	100
F	5	5.7	10	100
G	1	5.2	5	100
H	Min	4.0	0	100

*Percent of private system homeowners with an alkalinity below 500 mg/L as $CaCO_3$



NSF certified lead-free testing apparatuses: (a) 3% Pb C36000 brass coupon epoxied in a 125 mL glass vial, (b) <0.25% Pb C87850 brass coupon epoxied in a 125 mL glass vial, (c) <0.25% Pb faucet connector sealed at the bottom with a threaded polyvinyl chloride cap and a threaded polyvinyl chloride male adapter added to the top, and (d) <0.25% Pb galvanized steel nipple sealed at the both ends with threaded polyvinyl chloride caps.

Certified NSF lead-free plumbing components.

Lead Ban lead-free brass coupons. With the Reduction in Lead in Drinking Water Act in effect as of January 2014, C36000 brass is no longer permitted in new construction or repairs of drinking water infrastructure as this brass alloy contains 2.5-3.7% lead. However, as plumbing components composed of this brass may still be present in households constructed prior to January 2014, lead leaching from C36000 brass was evaluated. Brass coupons with 3% lead of 12.7 mm (0.50 in.) diameter and 12.2 mm (0.48 in.) height were epoxied to the bottom of 125 mL glass vials, which maintains the ratio of brass surface area to water observed in faucets (Triantafyllidou & Edwards, 2007). As C36000 brass was expected to have the highest lead release due to the higher lead content, brass coupons were exposed to all eight water conditions (A-H). Each condition was tested in triplicate.

Reduction in Lead in Drinking Water Act lead-free brass coupons. For single material components (e.g., brass check valve), the Reduction in Lead in Drinking Water Act requires a weighted average of <0.25% lead. To evaluate lead release from lead-free brass alloys manufactured after 2014, C87850 brass coupons containing 0.09% lead were epoxied to the bottom of 125 mL glass vials; these brass coupons had double the exposed surface area of the C36000 brass. Since C87850 brass contains low levels of lead, brass coupons were only exposed to the three most aggressive test water conditions, and tested in triplicate (F-H).

Reduction in Lead in Drinking Water Act lead-free faucet connectors. For multi-material components, the Reduction in Lead in Drinking Water Act requires a weighted average of less than 0.25% lead with respect to the wetted surfaces. While lead-free faucet connectors comply with this new regulation, these components can contain leaded brass ferrules. To evaluate lead release from brass ferrules in lead-free faucet connectors manufactured after 2014, 12.7 mm x 50.8 cm (0.5 in. x 20 in.) braided stainless steel faucet connectors were exposed to the three most aggressive test water conditions and tested in triplicate (F-H). Each faucet connector was sealed at the bottom with a threaded polyvinyl chloride cap, and a threaded polyvinyl chloride male adapter was added to the top to ensure a water tight seal. The male adapter was covered with parafilm for each period of stagnation.

Reduction in Lead in Drinking Water Act lead-free galvanized steel nipples. To quantify lead release from the zinc-lead galvanized coating of galvanized steel, 12.7 mm (0.50 in.) diameter schedule 40 galvanized steel nipples were evaluated. As these lead-free components were manufactured after 2014 and contained 0.1% lead, nipples were exposed to the three most aggressive test water conditions (F-H) as well as the NSF test water condition (A) to serve as a control, and tested in triplicate. Each nipple was sealed at both ends with a threaded polyvinyl chloride cap.

NSF Standard 61 Section 9 testing protocol. To evaluate lead leaching potentials, each test apparatus was exposed to water quality conditions using a static dump-and-fill method. According to the NSF protocol, each apparatus was subjected to varying periods of stagnation (2-64 h) for 19 days with lead quantified on days 3, 4, 5, 10, 11, 12, 17, 18, and 19. All test apparatuses were filled completely with test water to exclude air. For QA/QC purposes, to ensure lead release was not due to the glass vials and/or epoxy, an epoxy control was exposed to the NSF standard 61 test water (A). Lead samples were analyzed via Inductively Coupled Plasma – Mass Spectrometry (ICP-MS) per Standard Method 200.8 (American Public Health Association et al., 1998). Galvanized steel is not typically installed in premise plumbing, but continues to be used for well components such as drop pipes. To understand potential lead leaching from the galvanized coating, the 12 galvanized steel nipples exposed to their water quality according to the NSF section 9 protocol, but lead was only quantified on day 19.

Spike and recovery to examine sorption. Because released lead levels were less than expectations based on zinc release and the content of the galvanized coating, testing was conducted to determine if the galvanized coating could sorb lead released by corrosion and accumulate the lead in scale. Each test water put into the galvanized pipe was spiked with an initial concentration of 100 µg/L soluble lead, and then placed into the pipe. Any decrease in lead from this initial value would demonstrate uptake of lead in water by the plumbing material and accumulation in the scale. As before, each test nipple was exposed to its water quality condition using a static dump-and-fill method, and lead was quantified following the 16 hour overnight stagnation.

Results and discussion

Effect of pH and alkalinity on leaching. Based on private water quality observations provided by the Virginia Cooperative Extension, C36000 brass coupons were exposed to eight water conditions with pH and alkalinity ranging from 4.0-8.0 and 0-500 mg/L as $CaCO_3$, respectively. Consistent with the literature, lead leaching from brass coupons was higher when exposed to more aggressive water (Figure 5-1). Brass coupons exposed to water conditions E-H released lead concentrations exceeding the EPA action level of 15 µg/L in the 125 mL samples collected throughout the 19 day experiment, while lead concentrations in samples exposed to conditions A-D decreased below 15 µg/L during the last week of the experiment (days 17-19). On day 19, lead release was 4-16 times higher from brass exposed to conditions E-H compared to condition A (Figure 5-2).

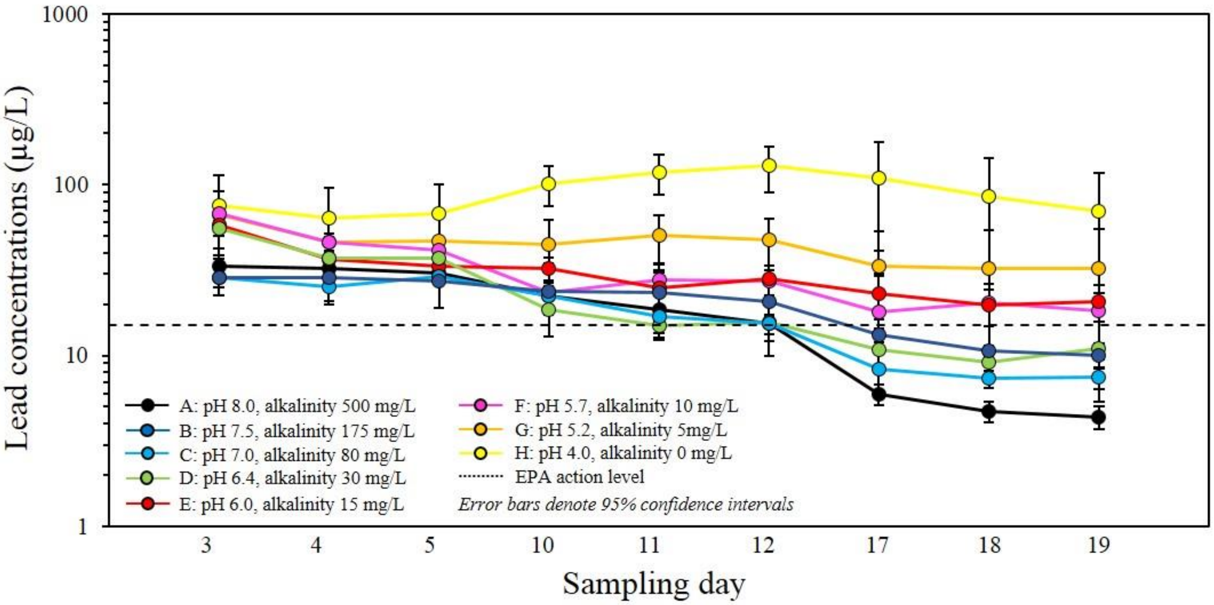


Figure 5-1. Mean lead released from C36000 brass coupons

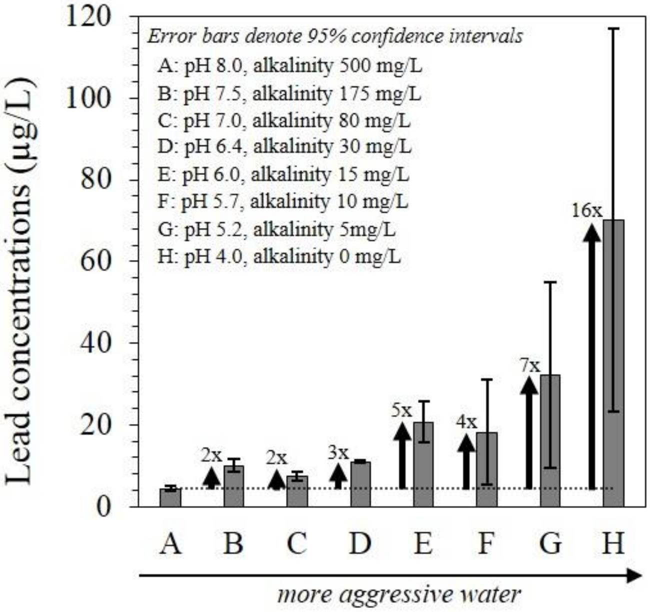


Figure 5-2. Mean lead released from C36000 brass coupons on day 19 under each water condition; increases in release relative to the standard NSF test water are provided to the left of each column.

When normalized to a 1 L first draw sample via Equation 5-1, all samples were below 15 µg/L during the last week of the experiment, with concentrations ranging from <1 to 13.6 µg/L. In order to be certified as lead-free prior to 2012, products were not permitted to leach more than 11 µg lead to the 1 L first draw sample. Based on the lead concentrations observed during the 19 day experiment and the variation between the triplicates, brass coupons exposed to conditions G and H had Q statistics (Equation 5-2) that were greater than 11 (Table 5-2). Therefore, private systems with a pH less than 5.2 and alkalinity less than 5 mg/L as $CaCO_3$ (i.e., approximately

1% of private systems in Virginia according to our historical data) are potentially at risk for elevated lead in water if C3600 brass is present in plumbing networks.

Table 5-2. Day 19 lead observed lead concentrations and calculated Q statistics for NSF certified lead-free testing apparatuses.

Test water	**3% Pb C36000 Brass Coupon**		**<0.25% Pb C87850 Brass Coupon**		**<0.25% Pb Faucet Connector**		**<0.25% Pb Galvanized Steel Nipple**	
	Day 19 Pb in 250 mL (µg/L)	Q	Day 19 Pb in 250 mL (µg/L)	Q	Day 19 Pb in 30 mL (µg/L)	Q	Day 19 Pb in 30 mL (µg/L)	Q
NSF	4.4	2.1	-	-	-	-	0.8	0.1
B	10.0	4.4	-	-	-	-	-	-
C	7.4	2.1	-	-	-	-	-	-
D	11.0	2.8	-	-	-	-	-	-
E	20.7	5.6	-	-	-	-	-	-
F	18.2	9.0	BDL	0.1	13.9	1.5	14.4	1.7
G	32.2	20.6	BDL	0.1	27.8	1.5	8.2	3.1
H	70.1	40.6	BDL	0.2	540.4	31.5	4.2	1.1

In 2012, the NSF reduced the Q statistic to 5 µg/L for most endpoint devices including brass fittings. With this lowered threshold, conditions E and F in addition to G and H had Q statistics greater than 5 (i.e., pH < 6.0; approximately 10% of past observations of water samples in Virginian private systems). Although the installation of C36000 brass components is no longer permitted in plumbing networks constructed or repaired after 2014, it is likely these components are widespread in plumbing networks constructed prior to 2014, rendering a larger number of homes vulnerable. These observations are not surprising given past field evidence; Dodrill and Edwards (1995) observed that no public drinking water system with a pH above 7.8 and an alkalinity above 174 mg/L as $CaCO_3$ exceeded the EPA's lead action level of 15 µg/L, but over 70% of systems with a pH below 7.4 and an alkalinity below 30 mg/L as $CaCO_3$ exceeded the action level. In addition, surveys of private systems report correlations between elevated lead concentrations and lower pH measurements (i.e., below 6.5) in samples taken at the point of use (Pieper et al., 2015; Swistock et al., 2013) .

Effect of reducing allowable lead content on single-material components. The Reduction of Lead in Drinking Water Act reduced the allowable lead content in plumbing components from no more than 8% to less than 0.25% lead with respect to wetted surfaces, which translates to no more than 0.25% lead for a component composed of a single material. To evaluate potential lead release from brass containing less than 0.25% lead, C87850 brass was exposed to the three most aggressive water conditions (F-H). Although these brass coupons leached lead, concentrations never exceeded 15 µg/L in the 125 mL samples (max of 2.5 µg/L) and non-detectable lead concentrations were reported for all water conditions on days 3-19. When normalized to a 1 L first draw sample, all lead measurements were below 1 µg/L throughout the 19 day experiment. Not surprisingly, conditions F-H had Q statistics that were less than 5 (Table 5-2).

Effect of reducing allowable lead content on multi-material components. Although the entire component must have less than 0.25% lead under the Reduction of Lead in Drinking Water Act, individual components can have higher lead content due to the relative wetted surface area. As an example, brass ferrules can contain up to 3% lead as these components only account for 8% of the total wetted surface. Therefore, faucet connectors with brass ferrules that contain 3% lead were exposed to the three most aggressive conditions (F-H). Lead released from the brass ferrules exposed to water condition F exceeded the EPA action level in the 30 mL samples, except for day 19 (Figure 5-3). The brass ferrules exposed conditions G and H leached lead above the action level throughout the experiment, with concentrations ranging from 28-673 µg/L. As this experiment only quantified lead release from the three most aggressive waters, further research is needed to evaluate additional less aggressive water conditions. However, these findings are consistent with the observed lead leaching results from the C36000 brass coupons, which highlights the higher potential for lead leaching under more aggressive water conditions.

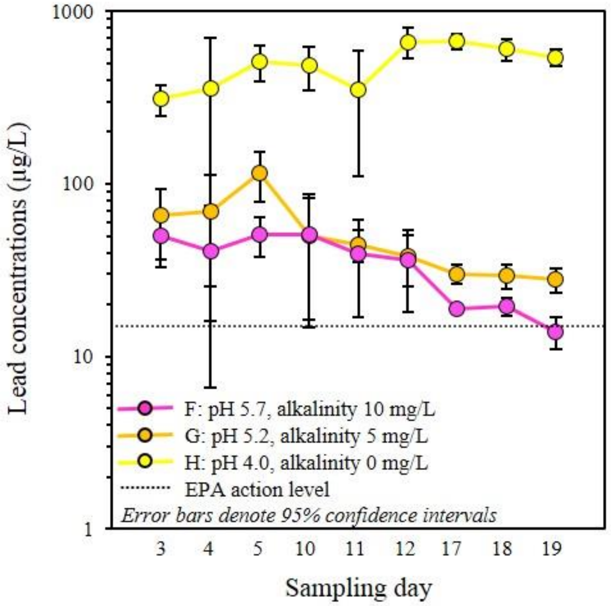


Figure 5-3. Mean lead released from lead-free faucet connectors

Although the brass ferrules leached elevated lead concentrations, when normalized to a 1 L first draw sample, concentrations were below 15 µg/L for conditions F and G (max normalized lead was 3.5 µg/L) and the Q statistics for both conditions were less than 3 µg/L (Table 5-2), which means that faucet connectors exposed to this aggressive water would not contribute more than 3 µg/L lead to the 1 L first draw sample. The Q statistic for condition H was 31.5 µg/L and the normalized lead results were still elevated, with concentrations as high as 20.2 µg/L. It is important to note that condition B was the most aggressive water documented in the Virginia Cooperative Extension dataset and represents the worst-case scenario.

Although faucet connecters exposed to conditions F and G satisfied the Q statistics, lead concentrations in the 30 mL samples ranged from 14-116 µg/L during the experiment. This highlights a potential issue with the NSF protocol requiring lead results to be normalized to a 1 L sample. While the LCR protocol outlines the collection of 1 L first draw samples, the USEPA recommends 250 mL first draw samples for unregulated school systems as this volume is more representative of a serving consumed by a child (U.S. Environmental Protection Agency, 1991, 2006). When evaluating the allowable lead in a 250 mL sample against the new Q statistic of 5 and 3 µg/L, a 250 mL can contain up to 20 and 12 µg/L lead, respectively (Figure 5-4). Therefore, as Dudi et al. (2005) illustrated with Section 8 components, normalizing smaller volumes to a 1 L volume may not be protective of public health as lead in water hazards can be diluted.

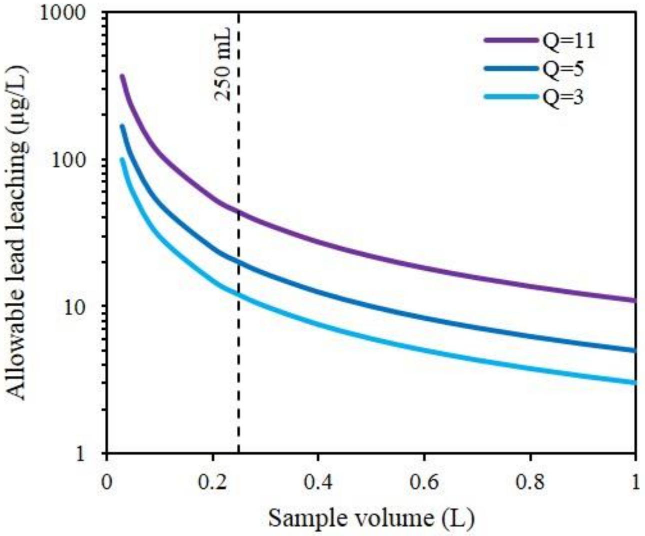


Figure 5-4. Comparison of lead release by sample volume

Adsorption of lead to iron. Galvanized steel nipples were exposed to the three most aggressive conditions (F-H), but lead concentrations were only quantified on day 19 of the NSF experiment. Despite having 0.1% lead in the galvanized coating, galvanized nipples exposed to conditions F-H released lead concentrations ranging from 4.2-14.4 µg/L in the 30 mL samples (Figure 5). Galvanized steel nipples exposed to the NSF test water (A) as a control released non-detectable lead concentrations (<1 µg/L). When normalizing the lead results and calculating the Q statistic, the galvanized nipples exposed to the four conditions had Q statistics less than 5 (Figure 5-5). Galvanized steel nipples exposed to the NSF test water (A) as a control released non-detectable lead concentrations (<1 µg/L). When normalizing the lead results and calculating the Q statistic, the galvanized nipples exposed to the four conditions had Q statistics less than 5 (Table 5-2). While brass coupons exposed to condition H released the highest lead concentrations, lead release was highest when galvanized nipples were exposed to condition F. It was hypothesized that this occurrence was due to the adsorption of lead to the iron pipe and/or scale, which has been documented in previous literature (De Rosa & Williams, 1992). Therefore, to assess this theory, the galvanized nipples were exposed to their corresponding water conditions for an additional day, but each conditions was spiked with a concentration of approximately 100 µg/L lead to quantify lead recovery and evaluate adsorption (Figure 5-6).

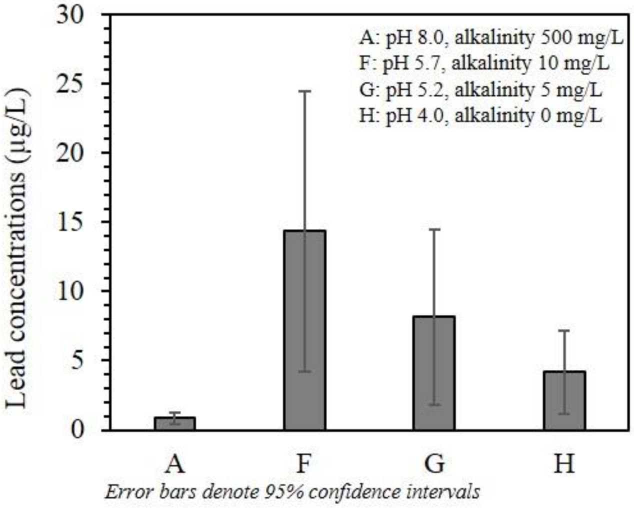


Figure 5-5. Mean lead released from galvanized steel nipples on day 19

As before, the highest lead concentrations were observed in condition F samples (40% recovery) and lowest concentrations in condition G samples (17% recovery). When exposed to the NSF test water (A), 6% of the lead was recovered, which is in keeping with the mechanics of iron corrosion (Masters & Edwards, 2015; Snoeyink & Wagner, 1996). Lead appeared to be retained by the galvanized component, most likely adsorbing to the iron scale, which highlights the potential limitations of a short-term 19 day NSF experiment in evaluating long-term health risks to consumers. If lead adsorbs to the iron scale, galvanized plumbing components essentially become lead “sponges” which could serve as later sources of high particulate lead release. While there is minimal official documentation of plumbing components used in private systems, wells greater than 180 m (600 ft.) in depth require a stronger drop pipe material due to the stresses induced by the pump and the weight of the drop pipe, and galvanized steel is often selected. Prior to 2014, galvanized coatings contained 0.5-1.4% lead (ASTM International, 2013). As a 305 m (1000 ft.) galvanized drop pipe with 1% lead in the coating would have the same lead in water contact effect as a 3.05 m (10 ft.) section of pure lead pipe, this is of concern. Despite the reduced allowable lead content reduced as of 2012, a 305 m galvanized drop pipe with 0.25% would be equivalent to a 0.76 m (2.5 ft.) pure lead pipe.

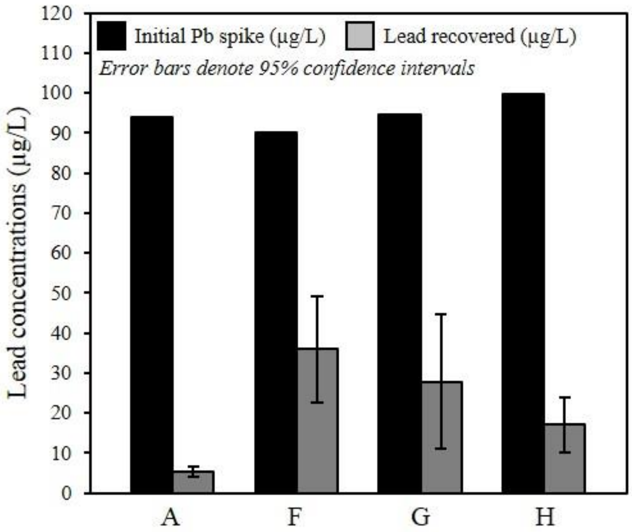


Figure 5-6. Lead recovery from galvanized steel nipples on day 20

Acknowledgements

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CHAPTER 6. FOLLOW-UP WATER QUALITY ANALYSIS IN ALBEMARLE COUNTY, VA

Kelsey J. Pieper, Leigh-Anne Krometis, Daniel Gallagher, Brian Benham, and Marc Edwards

Introduction

As municipal water services are not available in all communities in the United States approximately 15% of homeowners, mostly in rural areas, receive their household water from a private water system (e.g., wells, spring) (USEPA, 2012). Private water systems are not regulated by the Environmental Protection Agency (USEPA), which means homeowners are responsible for ensuring safe water quality through efforts such as regular testing and installation of treatment devices. Due to the voluntary nature of this maintenance and monitoring, studies typically report high rates of waterborne contaminants in these systems (DeSimone, 2009; Knobeloch *et al.*, 2013; Pieper *et al.*, 2015; Swistock *et al.*, 2013).

Despite the reported water quality in these systems, there is minimal effort to evaluate homeowners' behavioral actions and changes water quality after receiving water quality results. In addition, information regarding system and household characteristics provided by the homeowners via questionnaires are rarely validated and confirmed. Therefore, this study aims to measure changes in water quality and document behaviors actions within the 1-2 year following participation in an extension water quality survey.

Methods

Participation in Virginia Household Water Quality Program, 2012-2014

The Virginia Household Water Quality Program (VAHWQP; www.wellwater.bse.vt.edu), a Virginia Cooperative Extension program, provides private system homeowners the opportunity to participate in low-cost water quality testing. Homeowners participating in a water testing clinic are provided the necessary sampling materials and instructions for a small fee (~$50). On predetermined mornings, homeowners collect four point-of-use (i.e., consumers' tap) water samples: a 250 mL first draw sample for metals analysis, two 250 mL samples collected after five minutes of flushing for metals and physicochemical parameters analysis, and a 100 mL sample for bacterial analysis. In addition to collecting water samples, homeowners are asked to complete a questionnaire describing system characteristics (e.g., use of treatment).

Participation in complementary follow-up study, 2013-2014

Due to the high rate of participation in water quality testing, former VAHWQP participants from Albemarle County were selected as the sample population for follow-up study. During late 2013 and early 2014, the 108 homeowners from Albemarle County who participated in the 2012 and/or 2013 VAHWQP water quality clinic were offered an opportunity to participate in a complementary follow-up water quality analysis (Figure 6-1). Although 119 samples were submitted during the two year period, several homeowners participated both years. Of the 108 homeowners that were contacted, 44 homeowners volunteered to participate and 12 homeowners were not contacted as their contact information was incorrect or incomplete. The 15 homeowners from the high lead category participated in the profiling effort described in Chapter 3, and the remaining 29 homeowners from the low lead category participated in a simple resample study. All water quality data are pooled here for the purpose of analysis.

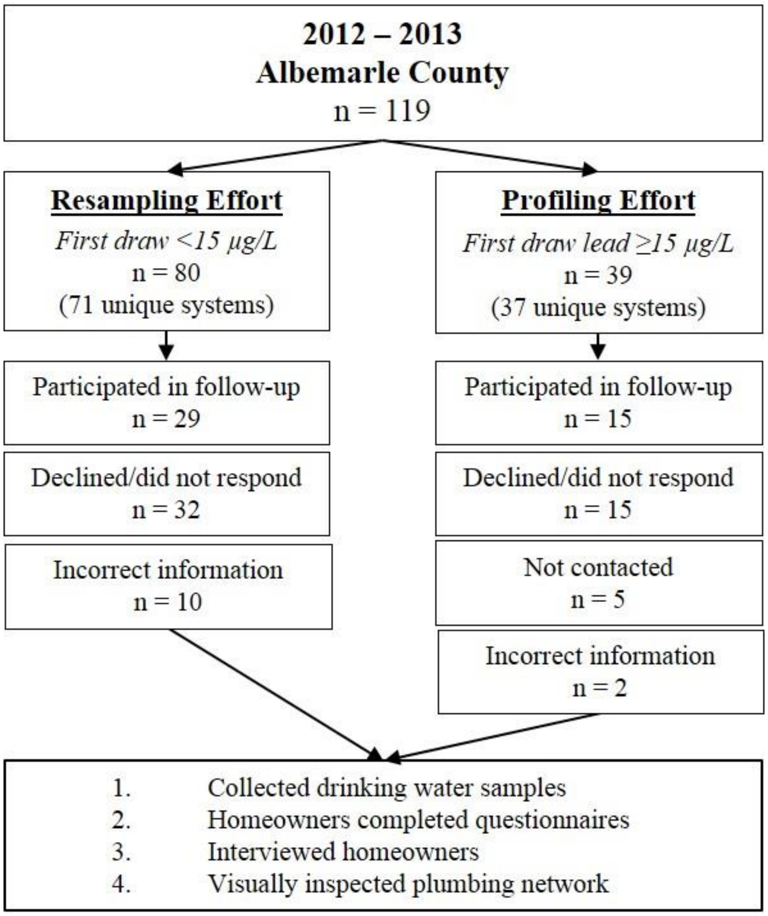

Figure 6-1. Flowchart illustrating participation in follow-up study in Albemarle County, VA

Resampling protocol for households with low lead

During the site visit, homeowners were provided with the necessary sampling materials, which included: sampling instructions, a timer, and sampling bottles. The sampling method was demonstrated to ensure proper collection of samples as well as to collect two additional samples, which were analyzed for pH and alkalinity. Homeowners were instructed to collect five samples from the kitchen cold-water tap (without removing the aerator) on a predetermined morning following a minimum six hours of stagnation. Although additional samples were collected, this study focuses on the 250 mL first draw sample for metals analysis and the 100 mL sample for bacterial analysis. While conducting site visits, the research team also interviewed homeowners to document behaviors since the initial VAHWQP testing, perception of water quality, and treatment maintenance.

General water quality analyses

All water quality targets were analyzed via Standard Methods: pH, conductivity, nitrate-N, sulfate, and fluoride per methods 4500-H^+, 2510, and 4110C (American Public Health Association et al., 1998). The defined substrate Colilert method with Quanti-tray 2000s was used to quantify total coliform and *E. coli* (www.idexx.com, Westbrook, MN, USA). Metals were quantified using Inductively Coupled Plasma – Mass Spectrometry (ICP-MS) per methods 3030 D and 3125 B (American Public Health Association et al., 1998). For data quality assurance and quality control (QA/QC), blanks and spikes of known concentrations were processed every 10 samples. Alkalinity was measured within two weeks of collection via titration per method 2330 B (American Public Health Association et al., 1998).

Statistical analysis

All statistical analyses were conducted in R version 3.0.2 (R Development Core Team, 2012) assuming an alpha of 0.05 as an indication of significance. The Wilcoxon signed-rank test compared alkalinity and pH measurements between homeowners who installed a chemical treatment device and homeowners who did not.

Results and discussion

Variations in water quality in Albemarle County

Observed water quality during VAHWQP clinics, 2012-2013

Between 2012 and 2013, 119 samples from Albemarle County were submitted to a VAHWQP clinic, and 49% exceeded at least one EPA health-based standard (Table 6-1), which is in keeping with water quality trends observed on the state level. However, only 21% of samples from Albemarle County tested positive for total coliform bacteria and less than 1% tested positive for *E. coli*, which is considerably lower than the reported rates of 46% and 10% for the state. However, samples from Albemarle County had higher rates of waterborne metals concentrations as 33% and 27% of samples had elevated levels of lead and copper. Not surprisingly, 45% of systems had a pH below 6.5, the lower end of the EPA recommended range of 6.5 to 8.5. The number of households with water treatment systems was comparatively high with 63% of homeowners identified having treatment installed (n=75). The most common type of treatment was a sediment filter (42%, n=50) followed by acid neutralizer (16%, n=19).

Table 6-1. Comparison of original Extension versus follow-up sampling results

Target water quality constituent	**SDWA Standard**		***Extension 2012-2013***	***Follow-up 2013-2014***
Exceeding at least one MCL			49% n=119	48% n=44
[‡]Total coliform, in cfu	MCL	ABSENT	21% n=119	30% n=44
[‡]*E. coli*, in cfu		ABSENT	0.8% n=119	0.0% n=44
[†]Copper, in mg/L	Action Level	1.3 mg/L	27% n=119	18% n=44
[†]Lead, in mg/L		0.015 mg/L	33% n=119	9% n=44
[‡]pH, in standard units	SMCL	6.5-8.5	47% (45%[§]) n=119	30% (27%[§]) n=44

MCL: Maximum Contaminant Levels – associated with risk to human health
SMCL: Secondary Maximum Contaminant Levels – associated with aesthetic considerations
cfu: colony forming units
[†] Measured in the first draw
[‡] Measured in the flushed sample
[§] Percent below pH of 6.5

Observed water quality during follow-up study, 2013-2014

The follow-up study occurred in 2013 and 2014 and 48% of the 44 systems sampled exceeded at least one health-based standard (Table 6-1). When comparing water quality results between the two sampling campaigns, samples submitted by 10 of the 44 homeowners during the VAHWQP survey tested positive for total coliform bacteria, and only five of these households had samples that tested positive again during the follow-up study. However, an additional 7 households provided samples that were positive for total coliform bacteria in the follow-up study, despite samples during the VAHWQP survey testing negative. While 15 of the 44 households previously had elevated lead in water (≥15 µg/L), only 2 of the households had samples that still contained elevated lead during the follow-up study. Although, two additional households had water samples with elevated lead concentrations in the follow-up, despite previously submitted samples with lower lead concentrations.

Variations in observed water quality between the two sampling campaigns could be attributed to several factors such as sampling location, sampling method, and the collection of a grab sample (Triantafyllidou et al., 2007). However, there may be behavioral actions and system modifications that influence observed water quality, which are rarely discussed in the literature (Knobeloch et al., 2013; Swistock et al., 2013).

Behavioral changes between initial testing and follow-up study

During site visits, homeowners were interviewed to discuss behavioral changes and/or system modifications that homeowners may have implemented between the VAHWQP testing and the follow-up study. Forty eight percent of homeowners indicated that they had not modified their behavior or system since the initial testing (n=21). However, 9% of homeowners (n=4) had changed their drinking and/or cooking water source, 11% had shock chlorinated their well (n=5), and 20% had installed or replaced a treatment device (n=9).

Source of water for drinking and cooking

Homeowners were asked to describe the locations (i.e., outlets) used to draw water for drinking and cooking. As excepted, 98% of homeowners (n=43) consume water from their kitchen tap and 100% (n=44) use water from the kitchen tap for cooking. Although, several homeowners also use bottled water as a source of water for drinking (23%, n=10) and cooking (5%, n=2). The homeowner that does not drink the water from the kitchen faucet relies solely on bottled water for drinking, but cooks with water from the kitchen faucet. Homeowners also noted drinking water from the refrigerator tap (34%, n=15) and bathroom tap (41%, n=18).

When discussing behavioral changes in water use, 9% of homeowners (n=4) reported changing their water source(s) for drinking and/or cooking. Three homeowners are now flushing their kitchen tap prior to use, which is assumed to be due to the high waterborne metals concentration observed during VAHWQP testing; waterborne lead concentrations were 27-32 µg/L during the VAHWQP clinic. Along with flushing, one homeowner also started using bottled water. The fourth homeowner has started cooking with water from the kitchen tap rather than exclusively with bottled water due to the good water quality results.

Shock chlorination

As bacteria is the most prevalent contaminant in private systems (Allevi et al., 2013; DeSimone, 2009; Pieper et al., 2015; Swistock et al., 2013, shock chlorination is often recommended to homeowners. While there is no standard protocol, methods outline pouring household bleach (i.e., sodium hypochlorite) into the well, typically at a concentration of 200 mg/L and leaving it stagnant for 12 to 24 hours before flushing (Benham and Ling, 2011). Since participation in VAHWQP water testing, five homeowners shock chlorinated their well. Three of these systems had submitted water samples that tested positive for bacteria and one household had undergone major renovations. Water samples from these five households tested negative for total coliform bacteria during the follow-up study. One homeowner expressed interest in shock chlorination, but was unsure of the method. The lack of a standard recommended protocol and difficulty in estimating well volume may prove this treatment to be challenging for many homeowners.

Understanding household characteristics

The interview provided an opportunity to further evaluate homeowners' understanding of their private systems. Only 27% of homeowners knew the exact depth of the well; although 45% of homeowners could approximate, with well depths being described as "greater than" or "within" a specific range. Unfortunately, 27% were unable to provide any type of estimate. One important variable in private system maintenance is the volume of water stored within the system, which includes the volume of water stored within the well. When discussing the year the well was constructed, 61% of homeowners were able to identify the year of construction, which was often associated with the construction of the home. While 9% of homeowners could not identify a year, 30% could provide an approximate timeframe (e.g., 1980s, within last 10-15 years).

Use of treatment devices

Install treatment after VAHWQP water quality results

Based on previous water quality results, 9 homeowners installed a treatment device between the initial VAHWQP testing and the follow-up study. One homeowner installed a point-of-use device that removes sodium as the water softener increased his/her sodium concentration. Four homeowners installed a device that corrects water acidity (e.g., acid neutralizer), one installed an iron removal unit, and two installed filtration units. Lastly, one homeowner replaced an old water softener.

Overall treatment

Higher rates of treatment use were observed during the follow-up study compared to VAHWQP, as 33 of the 44 homeowners (75%) indicated having at least one device installed. Ten homeowners had installed a sediment filter, and 23 had chemical treatment (i.e., acid neutralizer, iron removal and/or water softener) with 9 of these systems also having an accompanying sediment filter. Of the 23 chemical devices installed, 14 were acid neutralizers, which is consistent with the lead in water observations reported on the county level (34% of submitted samples exceeded lead action level). When interviewing homeowners about their system characteristics, 3 homeowners stated that no treatment was installed, yet during the inspection of the plumbing network, sediment filters were found.

Servicing treatment units

Homeowners typically described a timeframe for servicing chemical treatment, which varied depending on water use (3-4 months to once per year) and type of treatment. Eight homeowners with a chemical unit indicated that a treatment company maintains and regularly services their devices while homeowners maintain and service their units in the remaining 15 households. The water quality from households with chemical treatment was typically between a pH of 7 and 8 and alkalinity of 75 to 175 mg/L as $CaCO_3$; only one household had a pH below the EPA recommended 6.5 (Figure 6-2). In addition, alkalinity tended to be higher in systems serviced by a treatment company. Although most homeowners appeared to have a timeframe for maintenance, one homeowner discussed changing the resin in his/her acid neutralizer based on the extension water quality results. The servicing of filters varied greatly, which is in keeping with the nature of these treatment devices. Most homeowners indicated changing and/or cleaning the filter when there was a problem (n=7). Despite all homeowners having aerators on their faucets, homeowners communicated that they do not clean their aerators (39%, n= 17) or only clean them when there is a problem (e.g., reduced flow; 55%, n=24).

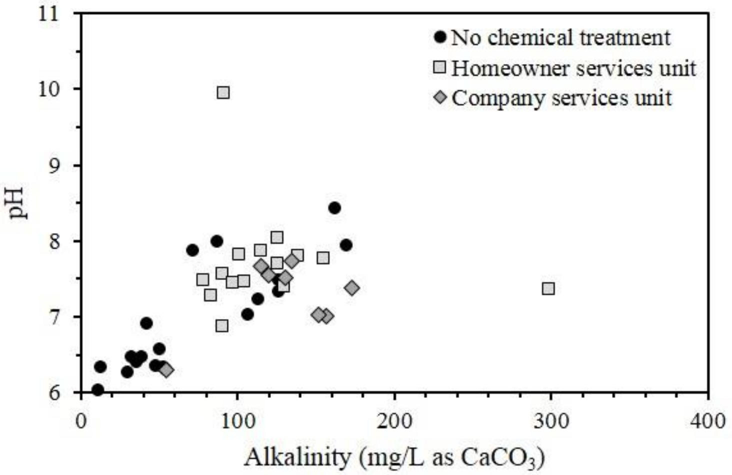

Figure 6-2. pH and alkalinity measurements in households without chemical treatment, chemical treatment devices serviced by homeowners and chemical treatment devices serviced by treatment companies.

To assess the potential influence of treatment on pH and alkalinity, households were categorized based on the use of chemical treatment. The 11 households that did not have treatment and the 10 households that only had a sediment filter were classified as 'no chemical treatment'. The remaining 23 households used at least one form of chemical treatment, which included water softener, acid neutralizer, and/or iron removal. For households that had a chemical treatment installed, pH and alkalinity were significantly higher (Wilcoxon test, $p<0.05$) compared to households without chemical treatment (Figure 6-2). The median pH and alkalinity for households with chemical treatment was 7.5 and 120 mg/L as $CaCO_3$ compared to 6.5 and 50 mg/L as $CaCO_3$ for households without chemical treatment. Of the 4 households that installed an acid neutralizer before the follow-up study, there was an average increase of 1.2 pH units.

Perception of water quality

The majority of the 44 homeowners (59%) identified an issue with staining. When asked to further characterize the staining, homeowners described it as blue-green (n=13), limescale (n=7), and rusty staining (n=10). In addition, 14 of these 27 homeowners noticed the staining predominately in the bathroom, in particular the toilet. Eight homeowners commented that aesthetic characteristics were noticeable before they installed treatment and several homeowners (n=4) noted aesthetic issues when treatment devices needed to be serviced. When asked why the treatment device was installed, homeowners communicated concern with acidic water (n=12), sediment/particles (n=8), iron (n=4), hard water (n=4) or that the previous owner had installed a device (n=4).

Well siting practices

In 1992, the Virginia Department of Health promulgated the Private Well Regulations, which established minimal construction standards and horizontal setback from known sources of contamination. Although these regulations do not address long-term maintenance or monitoring, the standards outline well integrity characteristics that minimize surface water contamination (Virginia Department of Health, 1992). As wells constructed prior to 1992 are not subject to these regulations, the VAHWQP developed ten tips for ensuring safe well water, which is in keeping with the Private Well Regulations (Virginia Household Water Quality Program, 2013). Based on these two documents, well heads were inspected and characteristics were documented during the follow-up study (Table 6-2). It is important to note that variances are permitted when setback distances are not feasible (e.g., siting well closer than 50 ft. to septic system), but regulations usually require additional barriers to contamination (e.g., extra grouting).

Table 6-2. Comparison of well practices to private well regulations and presence of total coliform bacteria

	All wells (n=44)		**TC+ wells (n=13)**	
	n	*% with category*	*n*	*% TC+ with category*
Well covering				
Sanitary cap	22	50%	6	27%
Well cap	18	41%	5	28%
Well cover	3	7%	2	67%
Well seal	1	2%	0	0%
Ground Slope				
Yes	6	14%	2	33%
No	27	61%	6	22%
Not visible	11	25%	5	45%
Casing height				
<12 in.	10	24%	3	30%
>12 in.	31	76%	9	29%
Cracks in casing				
Yes	1	2%	1	100%
No	42	98%	12	29%
Distance to septic				
<50 ft.	4	10%	0	0%
>50 ft.	35	90%	12	34%
Distance to road				

<100 ft.	22	50%	6	27%
>100 ft.	22	50%	7	32%
Distance to surface water				
<100 ft.	5	11%	1	20%
>100 ft.	39	89%	12	31%
Downslope of contamination				
Yes	9	21%	4	44%
No	34	79%	9	26%
Swampy or low-lying area				
Yes	4	9%	1	25%
No	39	91%	12	31%

Based on well head inspections, most wells were cited away from known sources of contamination and were not constructed in swampy or low-lying areas (Table 6-2). To prevent surface water contamination, VAHWQP recommends covering the well casing with a sanitary well cap or seal for a drilled well and a well cover for dug/bored wells. While all wells were protected with a cap or cover, 52% had a sanitary well cap or seal (n= 23) and 7% had a well cover (n=3). The traditional well cap may allow surface water and/or insects to enter through the unsealed sides, and 41% of systems had this covering. It is also recommended that well casing extend a minimum of 12 in. above the ground surface. Most wells extended at least 12 in. and only one well head had visible cracks in the casing. Lastly, the ground surface should be graded to slope away from the well casing to prevent water from pooling at the well head. This recommendation was difficult to assess as 25% of the time the well head was covered with grass or stone. However, most commonly, the ground adjacent to the casing was flat and not sloped away. As these regulations and recommendation address surface water contamination, the presence of total coliform bacteria was evaluated with respect to each category. Due to the limited number of samples collected, it is difficult to identify patterns associated with bacterial contamination. However, the link between bacterial contamination and well construction characteristics have been established (Gonzales, 2008). Therefore, sampling campaigns addressing bacterial contamination should collect samples at multiple locations. Understanding bacterial loading in the groundwater, well water, upstream and downstream treatment units would provide insight into bacterial contamination and the influence of proper construction as well as maintenance of treatment.

Conclusion

Although water quality surveys have been documenting the incidences of waterborne contaminants in private systems for decades, there has been minimal effort to understand the impacts and outcomes upon receiving these results. Evaluating 44 households in Albemarle County, VA, this study observed variations in water quality between the sampling campaigns and documented behavioral actions taken by the homeowners. While more than half the homeowners interviewed modified their consumption behaviors and/or implemented remediation strategies, there was confusion among others regarding proper protocols and routine maintenance. In addition, perception of water quality appears to be a primary driver for behavioral actions such as the installation and maintenance of treatment devices. Therefore, further research is paramount to understand the behaviors of homeowners and their link to observed water quality.

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CHAPTER 7. FINAL REMARKS AND SUGGESTED FUTURE RESEARCH

Characterizing water quality in private water systems is imperative as an estimated 47 million U.S. residents are reliant on these systems. This dissertation research documents and investigates the incidence of elevated waterborne lead concentrations in private systems as almost 1 in 5 households sampled in Virginia have lead in water above the EPA lead action level of 15 µg/L, with concentrations as high as 24,740 µg/L. As expected, based on the literature addressing corrosion in municipal water systems, elevated lead concentrations were associated with more aggressive water, which is common in private systems due to the limited use of chemical treatment.

Although there are numerous similarities between private and municipal water systems, private systems are considered individual distribution systems. This is important for well systems as source water (i.e., well water) may be contaminated as plumbing components are submerged in groundwater without options for corrosion control. Furthermore, the certification of "lead-free" plumbing components does not take into account conditions observed in private water systems (e.g., aggressive waters). Private system homeowners are at greater risk as brass components leached higher lead concentrations when exposed to more aggressive waters than outlined in the NSF protocol.

While this dissertation research advanced the knowledge of corrosion control and public health protection in private water systems, it also identified critical areas of research that need to be addressed. These research areas include:

1. Identifying associations between waterborne lead concentrations in private systems and human health outcomes;
2. Examining the performance of common low-cost lead remediation strategies using water representative of private system;
3. Documenting the performance and maintenance of homeowner installed treatment units; and
4. Evaluating the impact of commonly recommended chemical treatments on water quality and well-infrastructure longevity.

Identifying associations between waterborne lead concentrations in private systems and human health outcomes

While there is general consensus that deteriorating lead-based paint is the most widespread and significant source of lead exposure in urban areas (American Academy of Pediatrics Committee on Environmental Health, 2005; Levin et al., 2008), the USEPA has acknowledged that the percentage of exposure attributed to drinking water is rising. Even 25 years ago, drinking water was thought to contribute greater than 85% of exposure for infants consuming formula (U.S. Environmental Protection Agency, 1991). Cases of EBLLs in children from water have been associated with lead contamination even in cities with modern corrosion control meeting the 15 µg/L EPA action level (Brown et al., 2011; Edwards et al., 2009), which highlights the importance of characterizing waterborne exposure in unregulated private water systems that do not utilize corrosion control treatment devices. With the detection of lead poisoning at its early stages is generally revealed only by blood lead testing, efforts are needed to evaluate the contribution of lead in water to blood lead levels of children reliant on private systems.

Examining the performance of common low-cost lead remediation strategies using water representative of private system

In situations where flushing is not effective or adequate, in-home treatment units, such as reverse osmosis, particle filters, lead filters, and pH adjustments may be recommended. Unfortunately, the use of these treatment devices is not common in private systems despite the very aggressive water conditions documented (Pieper et al., 2015; Swistock et al., 2013). Although previous studies have examined lead removal rates from common POU filters (Deshommes et al., 2010b), the extreme conditions encountered in private water systems was not considered. Furthermore, the effectiveness of POU devices after the filter's published capacity was not evaluated, which is important as homeowners may not replace their filters as recommended. Therefore, laboratory studies evaluating the protectiveness and effectiveness of low-cost lead remediation strategies when exposed to the different patterns of waterborne lead release observed in private systems before and after capacity has been met need to be conducted.

Documenting the performance and maintenance of homeowner installed treatment units

Using data collected during selected VAHWQP drinking water clinics, 42% of homeowners submitted samples that tested positive for total coliform bacteria, but only 4% of homeowners (n=135) had a treatment device that targeted bacterial contamination (i.e., UV light or chlorinator). Furthermore, among the 4% of the population with bacterial treatment, 36% of systems with a UV light (36 of 101) and 47% with a chlorinator (16 of 34) tested positive for total coliform bacteria. Similar trends were observed with treatment options designed to filter waterborne lead or optimize corrosion. While homeowners were installing treatment devices that corrected specific water quality concerns, in some situations, water quality at the point-of-use did not meet drinking water guidelines for municipal systems. It is unclear whether this is due to extremely poor source water quality or ineffective treatment; however, these observations highlight a lack of understanding regarding treatment installation and maintenance. To evaluate the influence of installation and maintenance of common household treatment devices, water samples should be collected pre and post treatment. In addition, in-depth interviews with homeowners should outline maintenance schedules and remedial strategies.

Evaluating the impact of commonly recommended chemical treatments on water quality and well-infrastructure longevity

Although shock chlorination is commonly recommended to consumers to disinfect the well, there is no standard protocol. Researchers also note that shock chlorination may not be effective in reducing bacteria due to the presence of biofilms (Artiola et al., 2013), and short-term increases in metals release after shock chlorination have been observed (Seiler, 2006). Calcium carbonate, iron carbonate, and hydrated iron oxides can precipitate onto well components to form scale (Clarke, 1980). One well rehabilitation technique is to 'acidize' the well, which introduces a strong acid (hydrochloric, sulfamic or phosphoric acid) to dissolve and/or cleave the deposits. As with shock chlorination, there is no standard method, but some protocols outline adding a solution of 15% concentrated hydrochloric acid to the well and allowing up to 24 hours for the desired chemical reactions to occur (Clarke, 1980).

The effects of chemical treatments such as shock chlorination and acidization in private systems are not well understood as there is no standard protocol or prior research for these methods. Based on research in municipal systems, these chemical treatments could increase corrosion release due to the acidic conditions (Dodrill and Edwards, 1995; Schock, 1990; Triantafyllidou and Edwards, 2007), higher chloride (Edwards and Triantafyllidou, 2007; McNeill and Edwards, 2001), and long periods of stagnation (Deshommes et al., 2010a). However, shock chlorination is used to remediate sulfur issues (e.g. odor) (Seiler, 2006), and elevated sulfate concentrations have been reported to inhibit corrosion (Oliphant, 1983). Acidification aims to remove scale (Clarke, 1980), which may reduce water contamination as lead can accumulate in scale, but the low pH can mobilize very high levels of lead contained in the scale (Deshommes et al., 2010a). Therefore, the precise outcome of each remedial measure cannot be predicted based on any reasonable theory or prior data, making this research a high priority as these remediation may be beneficial or detrimental.

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