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LEAD IN TAP WATER OF PUBLIC SCHOOLS NEAR DAYTON, OHIO

A thesis submitted in partial fulfillment
of the requirements for the degree
Master of Science

By,

BAYLEE STARK
B.S., Wright State University, 2017

2019
Wright State University

WRIGHT STATE UNIVERSITY

GRADUATE SCHOOL

December 13, 2019

I HEREBY RECOMMEND THAT THE THESIS PREPARED UNDER MY SUPERVISION BY Baylee Stark ENTITLED Lead in Tap Water of Public Schools Near Dayton, Ohio BE ACCEPTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF Master of Science.

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ABSTRACT

Stark, Baylee. M.S. Department of Earth and Environmental Sciences, Wright State University, 2019. Lead in Tap Water of Public Schools Near Dayton, Ohio.

Lead (Pb) is a human-health concern, especially with regard to exposures of children. Lead contaminated drinking water is a primary route of exposure for children; however, water sampling for Pb is voluntary in schools with a public water supply. This study examined Pb in tap water from public schools around Dayton, OH. Schools were selected to span a range of ages (construction year) and community socioeconomic status. Of the 28 schools contacted, seven responded “affirmatively” to sampling, two responded “negatively”, and 19 did not respond. None of the schools that were sampled had Pb concentrations exceeding the U.S. EPA guidelines for supplemental action, which is ≥ 10% of plumbing fixtures exceeding 15 μg/L. Only four of 100 fixtures sampled had Pb exceeding 20 μg/L, the concentration recommended for fixture removal in schools. As expected, increased Pb levels were associated with warmer water temperatures. Water from sink faucets had greater Pb levels than water from drinking fountains, and Pb concentrations were greater in initial water sample draws versus samples collected after a 5-minute flush. To combat the leaching of Pb into school tap water, older lead and brass containing fixtures should be replaced, and changes in physicochemical parameters should be monitored to identify risks of Pb exposure.

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ACKNOWLEDGEMENTS

First and foremost, I thank my family for being so supportive throughout my education. I thank my boyfriend, Aaron, who was kind enough to come to the schools to help me sample and encouraged me to write. Thank you to Mike Bottomley of the Statistical Consulting Center at WSU for sharing his statistical wisdom. I'd like to thank all of the mentors I've had over my years at WSU: Dr. Stacey Hundley for believing in me and taking me under her wing as an undergraduate student; Dr. Chad Hammerschmidt for being my advisor, and for being supportive even when times were tough at the University; Dr. Silvia Newell, for reaching out to me and providing support; and Dr. Mark McCarthy, for not being afraid to send my drafts back in red. To my fellow lab mates, past and present, thank you for showing me the ropes, and always being supportive. I thank the superintendents and staff of the following school districts for being the subject of inquiry and, to some, for allowing water sampling at their schools: Beavercreek City, Bellbrook-Sugarcreek Local, Centerville City, Dayton Public, Fairborn City, Greenon Local, Huber Heights City, Kettering City, Miamisburg City, Tipp City, Trotwood-Madison City, Vandalia Butler City, West Carrolton City, and Xenia Community. Lastly, thank you to every other faculty member and office staff person who has helped me along the way. As they say, it takes a village.

I. INTRODUCTION

Lead (Pb) is a well-studied environmental and human health concern and toxin (U.S. EPA, 2019; Gidlow, 2015; Goyer, 1993). Widespread human use of Pb on an industrial scale has occurred for over 5,000 years (Gidlow, 2015; Centers for Disease Control and Prevention, 2012). Due to its low melting point and ductility, Pb has been used in various products, such as pipes, paints, gasoline, ceramics, and plastics (Flora et al., 2012). While some countries, such as the United States, have ceased using Pb in these products and others, many other nations are still utilizing Pb in manufacturing today (Gidlow, 2015; Centers for Disease Control and Prevention, 2012).

Lead is primarily introduced to the human body through either: inhalation, via outdoor and indoor air, or ingestion, including drinking and eating (Boskabady et al., 2018; Prüss-Ustün et al., 2011). Lead exposure of adult humans has been linked to multiple adverse effects, including, for example, cardiovascular disease, impaired bone formation and brittleness, decreased kidney function, cancer, hypertension, reproductive problems, and even death in extreme doses (Lanphear et al., 2018; Centers for Disease Control and Prevention, 2012; Prüss-Ustün et al., 2011; Goyer, 1993). Lead levels in human blood have been declining in the United States over the past 40 years (Centers for Disease Control and Prevention, 2019; Gilbert and Weiss, 2006), which has been attributed to increased environmental regulations on Pb-containing materials (e.g., gasoline, paint, and solder) and reduced Pb concentrations in tap water (U.S. EPA, 2018; Centers for Disease Control and Prevention, 2012; Gilbert and Weiss, 2006). However, according to one population-based study, over 400,000 deaths annually in the United States are still attributed to health conditions caused by Pb exposure (Lanphear et al., 2018).

1.1 *Children and Pb exposure*

Children exposed to Pb, even at low levels, can develop cognitive, behavioral, and developmental deficits. Between 2000 and 2004, Pb exposure was estimated to contribute to 600,000 new cases of children with intellectual deficiencies each year (Prüss-Ustün et al., 2011) Adolescents experiencing long-term exposure to Pb have also displayed a disadvantage in reaching academic learning goals (McCrindle et al., 2017). School-aged children with elevated Pb concentrations in blood have repeatedly scored lower on both in comparison to peers (McCrindle et al., 2017; Miranda et. al, 2007; Chiodo et al., 2004; Bellinger et al., 1997). In one study of children having blood Pb levels less than 10 μg/dL, a 1.4-point decrease in IQ was observed for every 1 μg/dL increase in blood Pb (Canfield et al., 2003). In related studies with children, greater dose responses in the lowering of IQ were observed when blood Pb levels were less than 10 μg/dL, in comparison to blood Pb levels greater than 10 μg/dL (Tellez-Rojo et al., 2006; Canfield et al., 2003). These studies illustrate the importance of mitigating childhood Pb exposures, including at low levels, for which there is observed sensitivity to small changes in circulating Pb.

Due to the potential cognitive, behavioral, and developmental deficits that can occur when children are exposed to elevated levels of Pb, a circulating blood reference level of 5 μg/dL has been established as a threshold for initiation of public health action (Centers for Disease Control and Prevention, 2019). This threshold is thought to be conservative based on previous research studying dose responses, but other Pb studies, such as those mentioned previously, have revealed adverse effects on IQ and neurobehavioral development at blood Pb levels lower than 5 μg/dL (Tellez-Rojo et al., 2006; Chiodo et al., 2004; Canfield et al., 2003). Due to cognitive defects, such as inattention and decreased academic performance occurring at such low levels, it is difficult to determine a threshold for response (Safruk et al., 2017; Prüss-Ustün et al., 2011; Lanphear et al., 2005; Canfield et al., 2003). According to the U.S EPA, there is no safe level of Pb exposure, which is concerning because over 500,000 U.S. children aged 1–5 years are estimated to have blood Pb levels greater than the 5 μg/dL reference concentration (U.S. Centers for Disease Control and Prevention, 2019).

1.2 *Lead in drinking water*

One of the major human exposure pathways to Pb is through drinking water (Chowdhurry et al., 2018; Deshommes et al., 2013). Lead can leach from water service lines, indoor plumbing, and plumbing fixtures into tap water in either particulate or soluble forms (Chowdhurry et al., 2018; Deshommes et al., 2010; Patch et al., 1998). Common sources of Pb leaching into tap water include Pb service lines, plumbing, solder, fixtures, and faucets (Lewis et al., 2017; Masters and Edwards, 2104; Deshommes et al., 2010). Recent research has also revealed the dangers of Pb leaching from galvanized steel piping (Clark et al., 2015). During the galvanization process, steel is coated with a zinc mixture, that often contains Pb, to prevent corrosion. In one study spanning multiple states, the Pb content on the surface of galvanized steel water pipes was found to range from undetectable to 1.8% by weight, and in some locations, Pb concentrations in the water from a dump-and-fill study were greater than 100 μg/L (Clark et al., 2015).

Multiple underlying factors contribute to leaching of Pb from plumbing, with two major contributors being the age of water lines and their degree of corrosion. Lead service lines that transmit water from the utility water line into buildings, and Pb plumbing fixtures are fewer in residential areas where houses were built after 1986 and implementation of the first amendment of the U.S. EPA's Safe Drinking Water Act (Cartier et al., 2011; Kimbrough, 2007; SDWA U.S. EPA, 1986). Therefore, it could be misconstrued that tap water from buildings constructed after 1986 is Pb-free, but highly leaded brass fixtures were not phased out of use in the U.S. until 1999, and many Pb-soldered fittings, Pb pipes, and galvanized pipes remain and are still contributing Pb to tap water (U.S. EPA, 2019; Chowdhurry et al., 2018; Deshommes et al., 2010).

The degree of leaching of Pb from service lines is influenced by water residence time, pH, temperature, maintenance repair, and abrupt changes in water chemistry, most commonly from either water source or treatment changes (Clark et al., 2015; Masters and Edwards, 2014; Cartier et al., 2011; Kim et al., 2011; Kimbrough, 2007; Edwards and Dudi, 2004). Corrosion and Pb leaching from pipes and fixtures are less when water is cooler and more alkaline compared to warmer and more acidic waters (Masters and Edwards, 2014; Kim et al., 2011; Schock, 1989). Concentrations of soluble Pb in tap water were positively correlated with water temperature (Kim et al., 2011). In the same study, corrosion of the protective scale lining in the pipe system was associated with lower pH. Stagnation times can also affect Pb leaching, as longer stagnation times are also associated with increased Pb concentrations in tap water (Dudi, 2004; Edwards et al., 2004; Edwards et al., 2002).

Current research is exploring the contribution of an area's socioeconomic status on the probability of Pb exposure through tap water (Balazs et al., 2012; Campbell et al., 2016; Switzer, 2017; McDonald and Jones, 2018). The U.S. EPA describes environmental justice as "the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income, with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies" (2019). The lack of environmental justice, environmental injustice, was one of the major factors of the Flint, Michigan Pb water contamination crisis (Campbell et al., 2016). Several studies in California have also found that populations composed of minorities and members with lower socioeconomic status were more affected by unsafe drinking water than other populations (Balazs et al., 2012; Switzer, 2017; McDonald and Jones, 2018).

The U.S. EPA regulates materials used to manufacture pipes, solder, and flux (a material applied to metal to help bonding while soldering) for water systems under the SDWA (U.S. EPA, 1986). Under the 2011 amendment of this act, these materials needed to be "lead-free"; that is, containing a weighted average of no more than 0.25% in the wetted surface material (U.S. EPA, 2011). In (1991), the U.S. EPA established the Lead and Copper Rule (LCR), a regulation establishing action level concentrations for Pb and copper in drinking water and corrosion control techniques for remediation (Goovaerts et al. 2017). With this rule, action is required to mitigate Pb leaching if concentrations in 10% of collected water samples exceed the action level of 15 µg/L (U.S. EPA, 1991), and in schools, it is recommended that a fixture be removed from service if Pb in water exceeds 20 µg/L. The LCR requires monitoring of community water systems, such as a municipal drinking water treatment plants, and non-community non-transient public water systems, such as a church or hospital that has its own water system. However, the rule does not require routine testing of Pb concentrations in publicly sourced tap water at schools and childcare facilities (U.S. EPA, 1991). Schools and childcare facilities are regulated under the SDWA only if they produce and distribute potable water from their own source. This regulatory loophole has resulted in 98,000 public schools and 500,000 childcare facilities in the U.S. that are not regulated under the SDWA (U.S. EPA, 2019; U.S. EPA, 1991). Due to this, almost all assessment of Pb in tap water of schools and childcare facilities is voluntary. In the context of childhood exposure to neurotoxic Pb, there is a need for greater information about Pb concentrations in school tap water.

I examined Pb in tap water from elementary and primary schools around the city of Dayton in southwest Ohio. Public schools targeted for this study included those listed in the Acknowledgments. The objective of this study was to assess relationships between Pb concentration in tap water and various common Pb leaching factors in elementary and primary schools (Chowdhurry et al., 2018; Clark et al., 2015; Masters and Edwards, 2014; Cartier et al., 2011; Deshommes et al., 2010; Kimbrough, 2007; Edwards and Dudi, 2004; Patch et al., 1998). I hypothesized that school buildings either built or renovated after the 1986 amendment of the SDWA was enacted (i.e., phase-out of Pb pipes and solder) would have lower Pb levels in tap water than older buildings. I also hypothesized that Pb concentrations would be related to water temperature and pH, with cooler water having lower Pb concentrations than warmer water, and more alkaline water having lower Pb concentrations than less alkaline water. With regard to median household income (MHI), I hypothesized that more affluent school districts would have lower Pb levels compared to less affluent school districts.

II. METHODS

2.1 Background

Dayton is the urban center of the Miami Valley and the fourth largest metropolitan region in Ohio (U.S. Census, 2010). Dayton was founded in the early 1800s, and as of 2010, the metropolitan region had a population of 1.1 million people (U.S. Census, 2010). Dayton is surrounded by many suburban cities and towns, having a variety of socioeconomic conditions and ages of infrastructure. The school districts in this study were chosen, in part, to reflect that diversity.

The tap water for the school districts in this study is supplied from the Great Miami River Basin, including the Little Miami River Buried Valley Aquifer, Mad River Buried Valley Aquifer, and the Great Miami Buried Valley Aquifer. These aquifers are comprised of glacial sand and gravel deposits (Dumouchelle, 1998). The aquifer system is lined with shale and limestone beds, which contribute to the high mineral and calcium content of the ground water in this area (Dumouchelle, 1998). The ground water from the aquifers representing 6 of the 7 sampled schools have an average hardness greater than 10 grains/gallon, which is considered "very hard" water (USGS 2019).

I selected 14 school districts with public drinking water sources in Montgomery, Greene, and Clark Counties for sampling and analysis of Pb in tap water. These districts would not be subject to the LCR and would only be monitoring water voluntarily, if at all. These districts are located within a 20-mile radius of downtown Dayton and were chosen to span a range of school construction dates and average community socioeconomic status, as indicated by the MHI of district residents (Table 1).

2.2. Sampling inquiry

School district superintendents were contacted by email and asked whether they would facilitate water sampling from multiple tap water sources in particular school buildings at a prearranged date and time. Letters to superintendents indicated the identity of their schools would be kept anonymous to the public and identified numerically for the purpose of this study. Affirmative, negative, and no responses were recorded within 21 days from inquiries to sample water. School districts that were contacted for potential water sampling are listed in the Acknowledgments.

2.3 Water sampling bottles

Sampling bottles were made of either polytetrafluoroethylene (PTFE) or low-density polyethylene (LDPE), that were rigorously cleaned (Hammerschmidt et al., 2011). In contrast to the 1-L bottles described in the LCR, 500 mL bottles were utilized for water collection. Aside from the bottle size, water was sampled following LCR techniques (U.S. EPA, 1991). Sample bottles were acid cleaned by rinsing five times with ultrapure Milli-Q water, followed by a minimum six-day storage in 10% hydrochloric acid (HCl), and finished with a second 5x rinse with ultrapure Milli-Q water reagent grade water. Acid cleaned bottles were stored in doubled plastic zip bags prior to sampling events to prevent potential contamination.

Table 1. Physical and socioeconomic characteristics of school buildings examined in this study, including building construction year, median household income (MHI) of school district zip code, and date of water sampling.

School ID	Year built	MHI ($)	Date sampled
1	1963	88,939	3/4/2018
2	1960	88,939	5/18/2018
3	1932	107,705	5/31/2018
4	1958	59,706	5/31/2018
5	1961	39,137	7/18/2018
6	1956	70,070	7/18/2018
7	1997	59,108	1/2/2019

2.4 Water sampling

Tap water was sampled between May 2018 and January 2019 following LCR guidelines (U.S. EPA, 1991) (Table 1). Sampling that occurred during the summer break of the school district was conducted during daytime hours (08:00–15:00). Sampling during the academic year occurred either before school hours (prior to 08:00) or during weekends and academic holidays. These sampling periods were chosen to best satisfy the LCR requirement of a minimum 6-hour water stagnation period prior to sampling. Tap water sources were assumed to meet the ≥ 6-hour stagnation period due to students and staff not being present during the time prior to sampling.

In each school, an initial 500 mL water draw sample was collected from all drinking water fountains and also faucets in the kitchen, nurse's office, and teacher's lounge, if access was available. For the delayed draw sample, water was allowed to run from the source for five minutes before a 500 mL aliquot was collected. Delayed draw water samples were collected at one faucet per school to investigate Pb contributions from water service lines to the school. Delayed draw samples were taken from sink fixtures at each location due to ease of flushing. Temperature, specific conductance, and pH were measured immediately after sampling initial draws by collecting water in a graduated cylinder and measuring with a YSI Professional Plus sonde. The sonde was calibrated for pH and specific conductance using traceable standards from the U.S. National Institute of Standards and Technology (NIST). Field blanks were used to evaluate potential contamination during sampling. Field blanks consisted of reagent-grade water in acid cleaned sampling bottles that were transported with tap-water sampling bottles during sampling. Double-bagged water samples were stored in a laboratory cooler during transport back to Wright State University.

2.5 Laboratory analysis

Water samples and field blanks were returned to the laboratory and acidified to 2% with high-purity 16 M HNO_3 (J.T. Baker Instra-Analyzed). Acidified (>24 hours) water samples and field blanks were homogenized by inverting the sample bottle three times and then transferred from sample bottles into acid-cleaned, 15-mL polyethylene tubes for analysis. Aqueous calibration standards (0–25 µg/L Pb) were prepared by diluting a U.S. NIST traceable solution with 2% Instra-Analyzed 16 M HNO_3. A U.S. NIST certified solution, CLMS-2A, was also diluted using Milli-Q water to concentrations of 0.0, 0.5, 1.0, 5.0, 10.0, and 25.0 µg/L Pb, and acidified. All samples and standards were analyzed for Pb by inductively coupled plasma mass spectrometry (ICPMS). Samples were analyzed following U.S. EPA method 6020A. The method detection limit for Pb was 0.002 µg/L and was determined from precision of replicate analysis of a low sample (APHA, 1995). Recovery of known standard additions for Pb averaged 98.4 % ± 2.9 % (n = 14). After every 10 samples, a randomly selected sample was analyzed in triplicate for quality control. The average standard deviation of triplicate analysis for Pb was 0.8 ± 0.8 % relative standard deviation (n = 8 triplicate sets).

2.6 Statistical analysis

Analytical and physicochemical data was analyzed using SAS University Edition software, and Microsoft Excel. Relationships and comparisons explored included: Pb and temperature, Pb and pH, Pb from buildings built pre-LCR and post-LCR, Pb and MHI,

MHI and school district response to sampling, Pb and fixture type, and Pb in initial water draw versus Pb in water after a 5-minute flush. Lead concentrations followed a non-normal distribution and were thus transformed logarithmically to satisfy model assumptions of normality and constant variance. Linear regression and parametric comparisons were used to examine influences of the following parameters on Pb in tap water: water temperature and specific conductivity, building age, fixture type, MHI, and MHI and response to sampling. Lead versus temperature, Pb versus pH, Pb vs specific conductance, and Pb vs MHI were analyzed using regression models to model relationships between the two variables. Lead and fixture type, and school response to sampling compared to MHI, were analyzed using a t-test. Lead in the initial water draw sample compared to Pb in the 5-minute flush sample was assessed using a paired t-test. An analysis of Pb in the tap water of buildings built pre-LCR versus post-LCR, could not be completed due to the small sample size. An x-value of 0.05 was used to judge the significance of statistical tests, p-values and R^2 values are based on transformed Pb data.

III. RESULTS AND DISCUSSION

3.1 Tap water

Lead concentrations from single fixtures across the schools ranged from 0.007–33.37 µg/L, and Pb averages of school buildings ranged from 0.86–7.42 µg/L (Table 2). Among all seven schools, only four of 100 total fixtures had Pb concentrations in water that exceeded the U.S. EPA school fixture removal recommendation of 20 µg/L. Although individual fixtures exceeded the LCR school fixture removal recommendation, none of the schools surpassed the LCR guidelines of 10% or more fixtures exceeding the 15 µg/L threshold. Thus, none of the sampled schools would fall under EPA guidelines for further corrective action under the LCR. pH of the sample set ranged from 7.53 to 9.07, temperatures ranged from 8.7 to 27.7 °C, and specific conductance ranged from 437 to 1640 µS/cm. There was no relationship between Pb and specific conductance in this data set ($p = 0.967$, ln Pb). Physicochemical readings were not collected at school 1. Of the 7 schools sampled, only one was built after 1990. The other six schools were built between 1930 and 1970.

3.2 MHI and Sampling Response

Only four of the 14 school districts contacted (including seven schools) agreed to facilitate water sampling. Three of the districts allowed sampling of multiple schools for a total of seven schools sampled in this study. In a comparison of response type to sampling inquiry ("yes" or "no"/no response), school districts that allowed sampling had

Table 2. Mean physicochemical characteristics, mean tap water Pb and standard deviation, and highest single fixture tap water Pb of school buildings examined in this study.

	Pb (µg/L)				Specific
School ID	All fixtures	Highest fixture	pH	Temperature (°C)	conductance (µS/cm)
1	2.78 ± 4.29	15.9	-	-	-
2	4.0 ± 2.5	7.38	8.1	19.6	1007
3	0.98 ± 1.78	5.77	8.1	19.9	886
4	3.12 ± 5.44	20.6	7.9	20.2	823
5	7.42 ± 11.5	33.3	8.5	21.7	489
6	2.13 ± 2.0	6.19	8.8	18.2	564
7	0.86 ± 1.21	2.98	7.9	12.0	862

a higher MHI than those who responded “no” or gave no response ($p = 0.006$, t-test, Figure 1). Previous studies have explored the influence of socioeconomic status on water contaminant exposure. In a study in California, groups with lower socioeconomic status and minority groups had increased potential for both initial and repeat violations of drinking water quality standards (McDonald and Jones, 2018). In a nation-wide review, SDWA violations are more common in low-income communities with higher populations of black and Hispanic individuals than high-income communities with lower populations of black and Hispanic individuals (e.g., Switzer, 2017).

A comparison of school district MHI to Pb levels had no statistical significance ($p = 0.634$, linear regression, Figure 2). This may have been influenced by small sample size and a repeating MHI for two school districts. Future analysis of how this variable is related to Pb levels may give better insight into potential environmental inequality of school districts in the Dayton area.

3.3 Influence of temperature

Lead concentration in tap water was weakly correlated to water temperature in initial water samples ($p < 0.001$, linear regression, Figure 3). Although Pb samples ranged almost four orders of magnitude among samples, water temperatures varied only by a factor of two in this study. This range is typical of water stagnating at ambient temperature or arriving slightly cooled in the case of water fountains. This relationship is in agreement with previous research on the influence of water temperature on measured

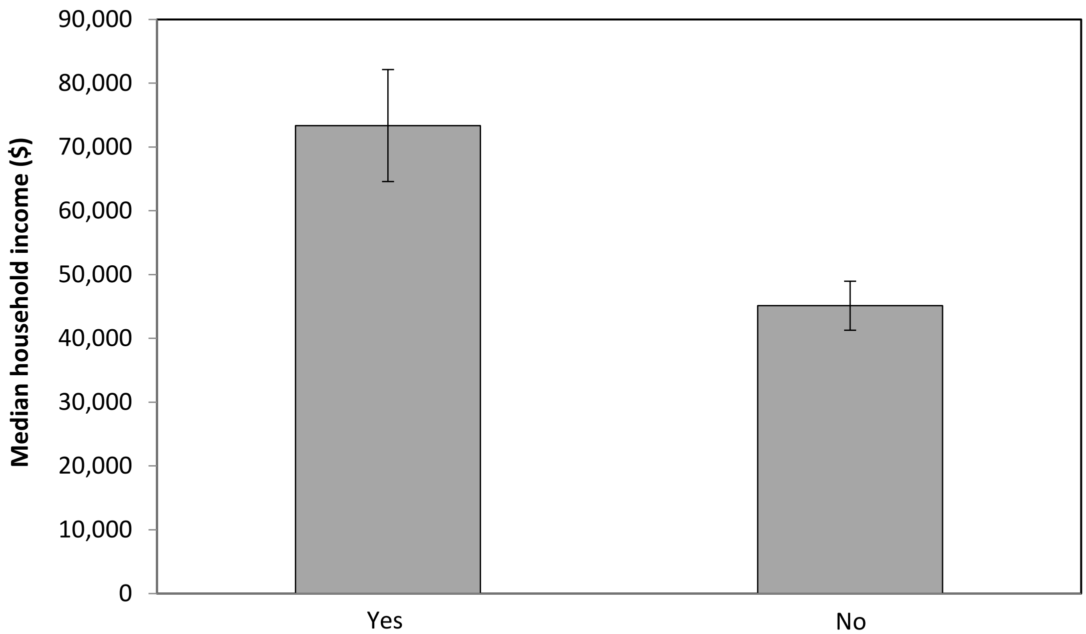

Figure 1. Median household income of school districts that responded “yes” and that responded “no” (or no response) (p = 0.006). Error bars are one standard error of the mean.

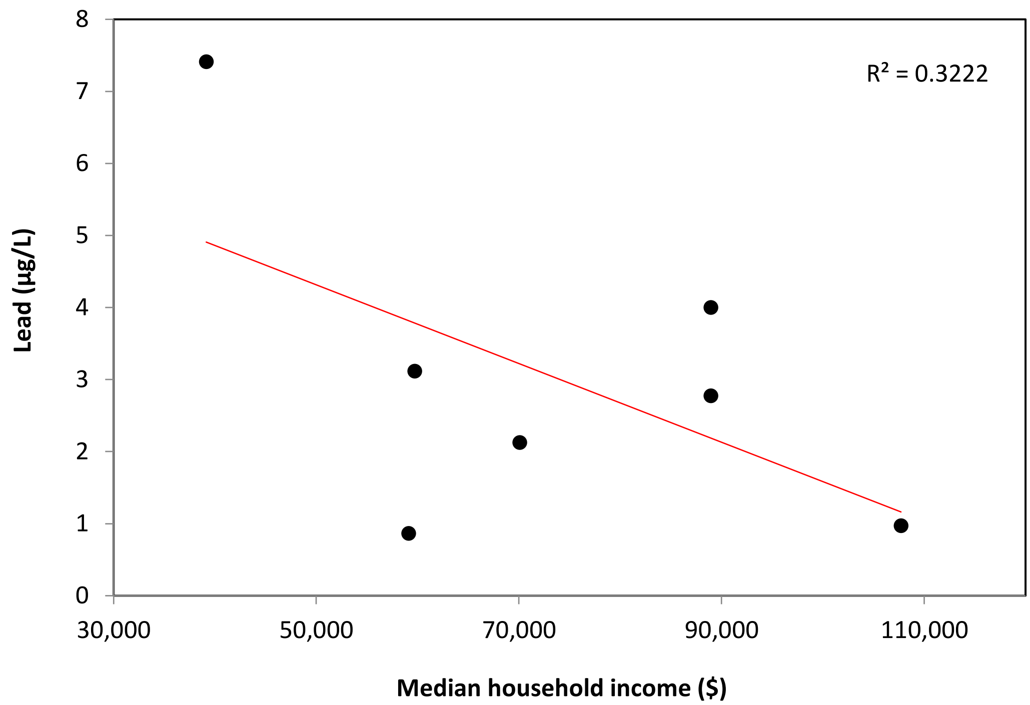

Figure 2. Correlation between mean Pb concentration (µg/L) and the median household income of each school district ($p = 0.634$, ln Pb).

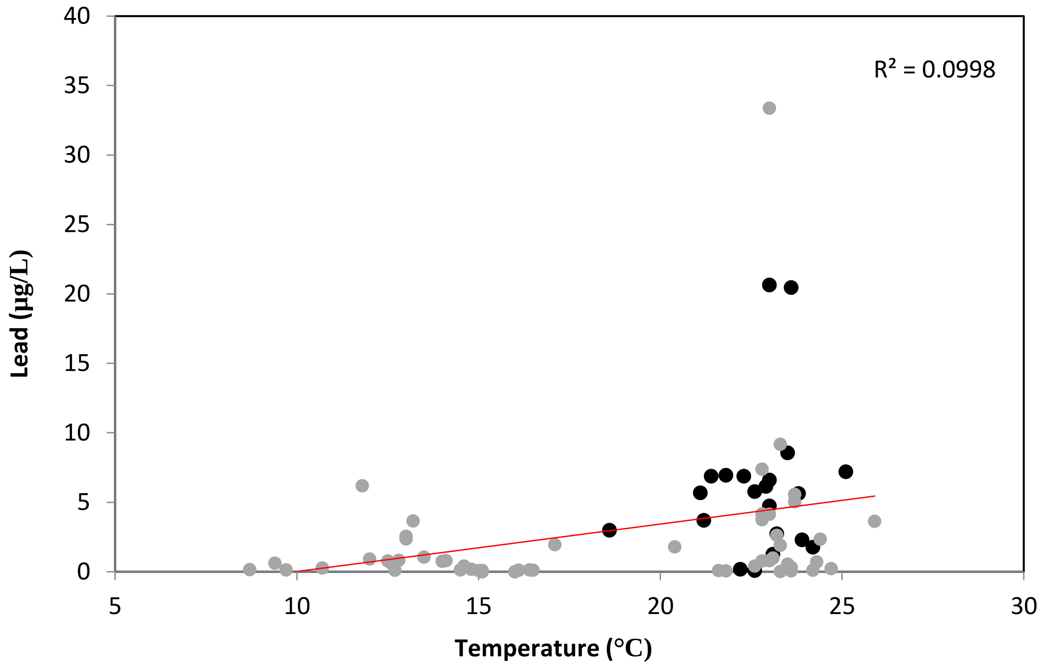

Figure 3. Correlation between Pb concentration (µg/L) and the temperature of tap water samples. Water from sink fixtures are denoted by black markers, water from water fountains are denoted by grey markers ($p < 0.001$, ln Pb).

Pb concentrations. Higher water temperatures can accelerate reaction and dissolution rates in plumbing and fixtures (Kim et al., 2011; Bryant, 2004). The temperature of water samples drawn from sink fixtures were warmer than 18° C (Black markers, Figure 3), while temperatures of water samples drawn from water fountains spanned the entire temperature range (Grey markers, Figure 3). This is due to the water from sink faucets sitting at ambient temperatures within the fixtures, and water from water fountains being either ambient or chilled. Due to water sitting stagnant in the plumbing, Pb concentrations can also fluctuate seasonally, with higher concentrations in warm summer months compared to winter (Kim et al., 2011). In this study, water was sampled during only one sampling event at each school. Seasonal changes within each water system could not be determined, but seasonal variation in Pb concentration may also be present within these schools.

3.4 Influence of pH

Tap water was alkaline (range = 7.53–9.07) and Pb concentrations in tap water were unrelated to pH ($p = 0.834$, linear regression, Figure 4). Formation and dissolution of protective carbonate films on pipes is dependent on pH (Schock, 1989). Water that is more alkaline is more efficient at forming protective films, and a pH of 8–10 can inhibit leaching (Schock, 1989). According to the U.S. EPA, a non-corrosive water system is defined as having a pH > 7.8 and alkalinity between 30 and 100 mg $CaCO_3$/L. The relatively high alkalinity of tap water samples in this study (range = 7.53–9.07) likely helped prevent Pb leaching from pipes and fixtures into the water systems (U.S. EPA, 2003). The water samples in this study were unfiltered, so particulate inorganic species of Pb may also be present. The samples that had higher Pb concentrations in this study may

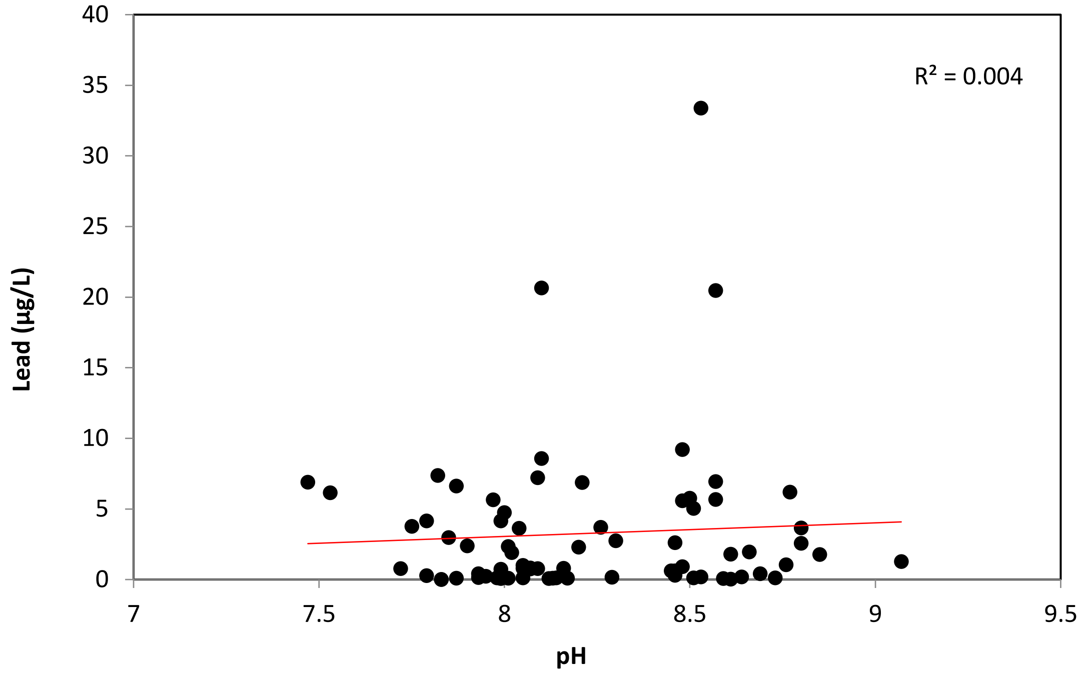

Figure 4. Correlation between Pb concentration (µg/L) and the pH of tap water samples (p = 0.834, ln Pb).

also contain lead-carbonate, which can leach from the protective scale that lines the plumbing. Pb leaching is often increased with greater acidity in a water distribution system, and most metal leaching from plumbing occurs when there are changes to pH (Clark et al., 2015; Cartier et al., 2011; Kimbrough, 2007; Schock, 1989).

3.5 Fixture type

Water samples were collected from water fountains and sink faucets. Lead concentrations were higher in initial draw water samples from sink faucets (n = 25) than initial draw water samples from drinking fountains (n = 75) (p = 0.007, t-test, Figure 5). Little research has specifically explored the influence of water fountains versus water fixtures on Pb concentration. While certain components of fixtures are known to contribute to Pb in water, such as brass fittings, Pb solder, galvanization in the manufacturing process, and the use of Pb containing alloys in fittings (Lewis et al., 2017; Masters and Edwards, 2104; Deshommes et al., 2010), research focusing specifically on the differences between water fountains and other types of faucets was not found during the literature review.

Based on the findings in this study, along with previous research linking water temperature to Pb levels (Kim et al., 2011; Bryant, 2004), concentrations may be lower in water fountains in part due to the water being chilled, as cooler water inhibits dissolution rates. The composition (e.g. pipe material, fittings, and solder type) of the water fountains may also play a role in the reduced Pb levels.

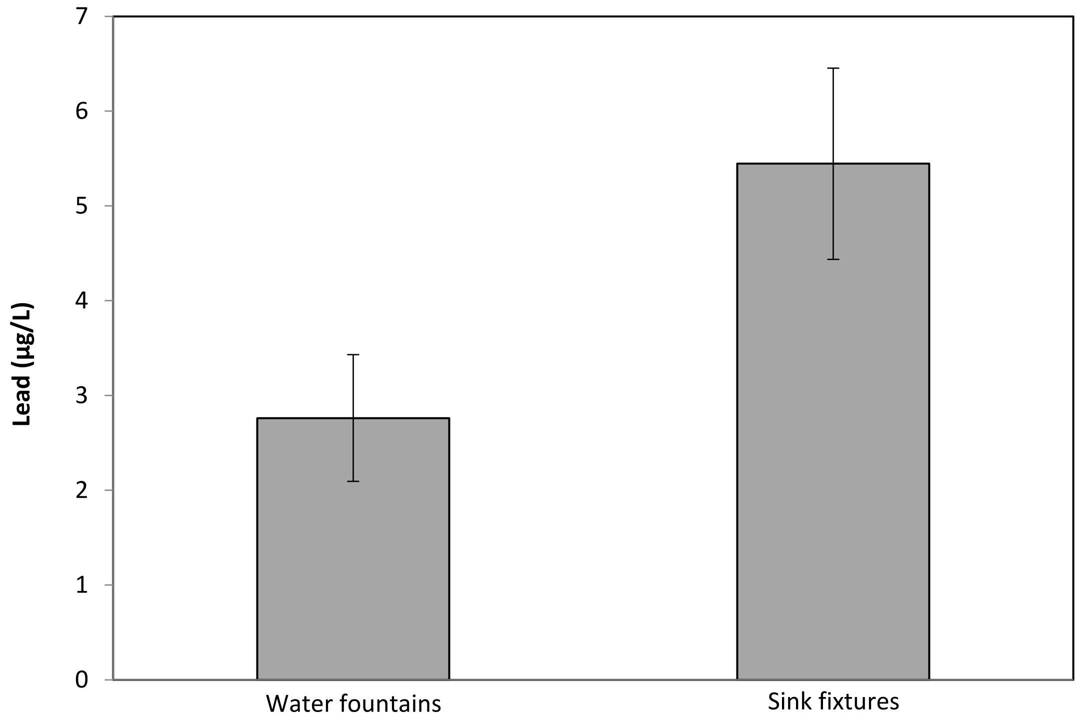

Figure 5. Lead concentration in tap water samples from sink fixtures (µg/L) (n = 25) and water fountains (µg/L) (n = 75) (p = 0.007). Error bars are one standard error of the mean.

3.6 School building construction year

The construction year of the school buildings had no influence on Pb concentration in tap water samples ($p = 0.892$, linear regression, Figure 6). After the first amendment of the SDWA was passed in 1986, restrictions were placed on lead containing plumbing materials for tap water (SDWA U.S. EPA, 1986). This sample set had only one school building built after the onset of those restrictions. The schools in this study may also have had renovations since initial construction, replacing pipes and fixtures over time. Due to the limited school age range, and potential updates to school building plumbing, school construction year was not a representative indicator of Pb concentration in this study. With a larger sample size, building construction year may be more significant.

3.7 Stagnation

Much of the pb in school tap water is derived from intra-school sources as opposed to the water service line. Water samples from the initial water draw had higher concentrations of Pb than those collected after a 5-minute flush ($p = 0.001$, paired t-test, Figure 7, Table 3). Therefore, the majority of Pb leaching into the tap was from the fixtures, as opposed to the school building service lines, which may or may not contain Pb. As stated previously, samples collected after a 5-minute flush are indicative of metals originating from either the source or service lines, as opposed to in-home plumbing and fixtures (Goovaerts et al. 2017; U.S. EPA, 2013). Due to the initial draw samples being elevated, the schools sampled in this study could likely observe a reduction in Pb concentrations at the tap by replacing their old fixtures, or by flushing their water before use. This solution is also more cost effective than replacing the school water service line.

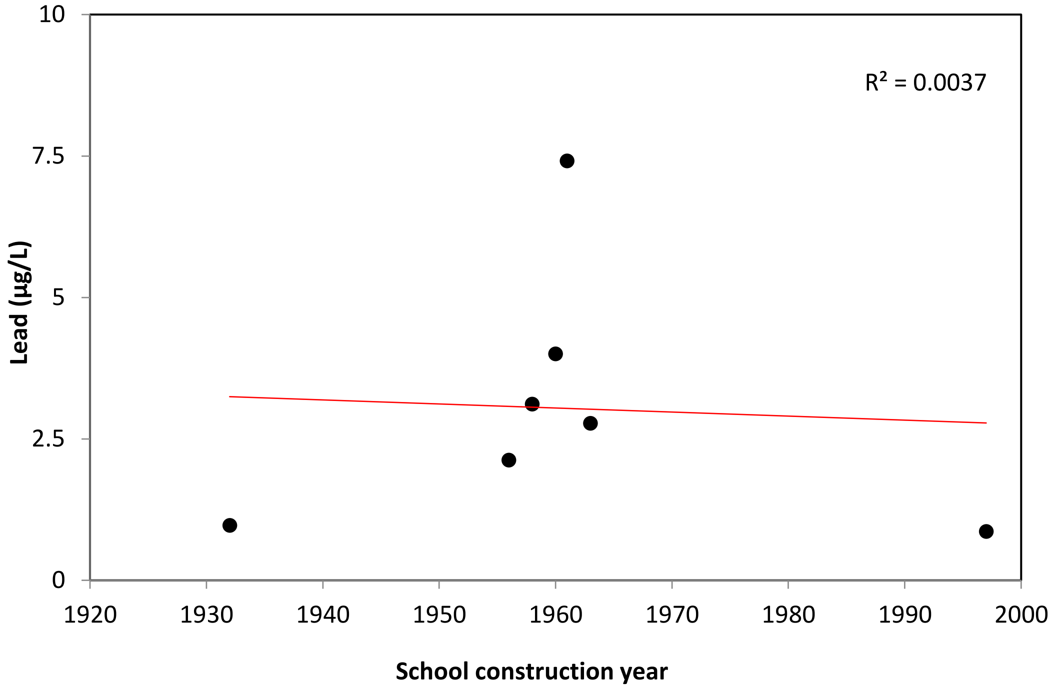

Figure 6. Correlation between mean Pb concentration and school construction year ($p = 0.892$, ln Pb).

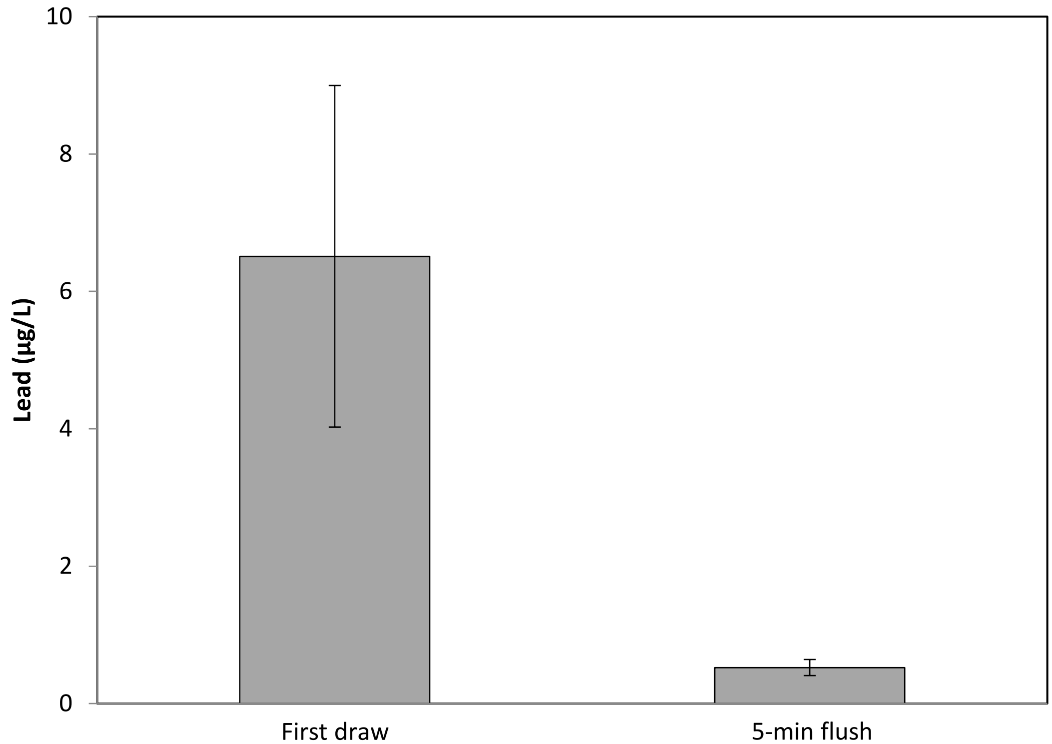

Figure 7. Lead concentration in first draw samples (µg/L) and 5-min flush samples (µg/L) ($p = 0.001$). Error bars are one standard error of the mean.

Table 3. Lead concentration in initial water draw and 5-minute flush by school.

School ID	Initial draw (µg/L)	5-min flush (µg/L)
1	2.3	0.2
2	3.7	0.7
3	5.8	1.1
4	8.6	0.7
5	20.5	0.2
6	1.8	0.4
7	3.0	0.4

IV. SUMMARY

Lead exposure of children and adolescents is a continuing problem across the United States and globally. Tap water is still one of the major sources of Pb exposure. The prevalence of Pb in school tap water is largely unknown, as water testing for Pb in public school facilities that do not have their own water supply is voluntary and not regulated under the LCR. In this study of seven community schools, four fixtures surpassed the LCR school fixture removal recommendation of 20 µg/L. Fixture type and water temperature were both significant factors in elevated Pb levels. In a comparison of first draw samples versus a 5-minute flush, indoor plumbing and fixtures were the major contributors of Pb, rather than the service line. In a comparison of school district MHI to response to sampling, school districts that responded yes to sampling had higher MHIs than those that responded no (or did not respond). In a future study, a comparison of the MHI of school districts to the Pb concentration in their tap water could reveal if environmental inequality is a contributing factor to Pb levels in the Dayton area. To assess potential exposure risks at school facilities, water testing should be carried out at all non-regulated schools and childcare facilities across the United States. A larger study encompassing more school districts would also improve strength of correlations between these drivers, as other factors may also be contributing.

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VI. APPENDIX

Dear XX,

I am a professor of aquatic chemistry at Wright State University (https://people.wright.edu/chad.hammerschmidt). I have studied metals in water for over 20 years, and I am proposing a unique and free service for XX elementary.

During the past two years, one of my M.S. students has been examining lead concentrations in Dayton tap water via door-to-door sampling. Lead in drinking water is a public health concern because of its neurotoxicity, especially to children. Of the 100+ Dayton homes that she sampled, we found that less than 5% had lead concentrations in tap water that were greater than the U.S. EPA action level of 15 parts per billion (ppb). This result was expected good news.

However, a news article published on Jan 22 in the *Dayton Daily News* suggested that lead concentrations in tap water of southwest Ohio schools might be a problem based on a contractor's findings (https://www.daytondailynews.com/news/local-schools-find-lead-water-more-now-plan-test/qoXNFZi0brfbhnVLTEax8M/). I was initially surprised and did not believe the contractor's results, so, with the superintendent's permission, I resampled the Bellbrook elementary school water that was tested previously. In contrast to the contractor's results, I found that none of the school water sources had lead concentrations above the U.S. EPA action level, and some of the contractor's results were 100-fold greater than our measurements. My hypothesis is that the contractor responsible for collecting and analyzing the water samples does shoddy work, to be frank. The problem here is two-fold: 1) school districts that can afford lead testing may be getting bad data and therefore making expensive and unneeded plumbing renovations (and causing undo concern of parents, like me), and 2) districts that cannot afford testing do not know the concentration of lead in their water. In both cases, I see this as an environmental injustice.

I have a first-year M.S. student, Baylee Stark, who is planning to investigate lead concentrations in drinking water from schools in southwest Ohio (Montgomery, Greene, Warren, and Clark Counties). She intends for her future career to be in environmental health. Her thesis research is seated in the question of risk factors (e.g., school age, district median income) for potentially elevated lead concentrations in public school water. Lead is especially detrimental to young children, so her research will be focused on elementary and intermediate schools. With this project she also will be providing a public health service. I am writing to ask for your permission and logistical help to sample tap water from XX Elementary, which Baylee and I have selected based on the school's location, age, and other school-district characteristics according to Baylee's hypotheses. We would prefer to sample water in the morning, and it will not take more than one hour. An itemized report of results will be sent to you directly and confidentially, from me. We will not leak school-specific results to the press. We will, however, and by nature of the research, report results in Baylee's thesis and a subsequent publication in the peer-reviewed literature, but none of the results will be directly traceable to specific schools. With your permission, XX will only be acknowledged as a participant, in the acknowledgments section of her thesis and with other schools, for participating in the study. All sampling, analysis, and data reporting will be of no cost to you; the cost is covered by me and Wright State University.

Please reply to me about your willingness to participate in this study and with either any questions or concerns you may have. We would like to schedule sampling times in June and July. Please reply by email and either suggest a time that we can talk or indicate who else in your district that I should contact.

Thank you for considering my request.

Kind Regards

Figure A1. Example of the email sent to school superintendents inquiring about school sampling access.

Table A1. School ID, Pb concentration, pH, temperature and fixture type (WF = water fountain, S= sink fixture) of each individual water sample.

School ID	Lead (µg/L)	pH	Temperature (°C)	Fixture type
1	0.424			WF
1	0.630			WF
1	1.492			S
1	1.819			WF
1	0.339			WF
1	1.442			WF
1	0.174			WF
1	1.726			S
1	2.118			WF
1	2.339			S
1	0.214			WF
1	0.495			WF
1	0.528			WF
1	0.254			WF
1	0.203			WF
1	9.186			WF
1	5.406			WF
1	14.34			WF
1	15.93			WF
1	3.411			S
1	5.271			WF
1	0.321			WF
1	0.447			WF
1	0.444			WF
1	0.449			WF
2	7.376	7.82	22.8	WF
2	3.757	7.75	22.8	WF
2	0.795	8.05	14.1	WF
2	2.379	7.90	13.0	WF
2	6.141	7.53	22.9	S
2	6.888	7.47	22.3	S
2	4.153	7.79	22.8	WF
2	6.881	8.21	21.4	S
2	1.958	8.66	17.1	WF
2	6.610	7.87	23.0	S

2	0.623	8.45	9.40	WF
2	0.755	8.07	14.0	WF
2	0.917	8.48	12.0	WF
2	3.696	8.26	21.2	S
2	3.633	8.04	25.9	WF
2	7.204	8.09	25.1	S
2	4.730	8.00	23.0	S
2	2.296	8.20	23.9	S
2	5.677	8.57	21.1	S
2	6.946	8.57	21.8	S
2	0.618	8.46	12.6	WF
3	5.766	8.50	22.6	S
3	0.085	8.01	22.6	S
3	0.098	8.17	15.1	WF
3	2.748	8.30	23.2	S
3	0.800	8.16	23.0	WF
3	0.081	8.12	21.6	WF
3	0.091	8.13	15.0	WF
3	0.721	7.99	24.3	WF
3	0.128	8.14	14.5	WF
3	0.073	7.99	21.8	WF
3	0.109	8.05	16.1	WF
4	0.419	7.93	14.6	WF
4	0.222	7.95	24.7	WF
4	20.64	8.10	23.0	S
4	8.570	8.10	23.5	S
4	2.340	8.01	24.4	WF
4	0.105	7.87	16.5	WF
4	0.128	7.98	16.4	WF
4	0.824	8.07	12.8	WF
4	4.146	7.99	23.0	WF
4	5.638	7.97	23.8	S
4	0.997	8.05	23.1	WF
4	0.008	7.83	16.0	WF
4	0.007	7.83	15.1	WF
4	0.774	8.09	22.8	WF
4	1.910	8.02	23.3	WF
5	0.174	8.64	14.8	WF
5	0.309	8.46	23.6	WF
5	9.194	8.48	23.3	WF
5	0.558	8.46	23.5	WF

5	0.111	8.51	12.7	WF
5	1.799	8.61	20.4	WF
5	0.063	8.59	23.6	WF
5	0.022	8.61	23.3	WF
5	31.73			WF
5	33.37	8.53	23.0	WF
5	20.46	8.57	23.6	S
5	2.617	8.46	23.2	WF
5	5.029	8.51	23.7	WF
5	0.190	8.53	22.2	S
5	5.568	8.48	23.7	WF
6	1.769	8.85	24.2	S
6	0.123	8.73	24.2	WF
6	1.264	9.07	23.1	S
6	1.051	8.76	13.5	WF
6	6.191	8.77	11.8	WF
6	0.401	8.69	22.6	WF
6	3.658	8.80	13.2	WF
6	2.560	8.80	13.0	WF
7	2.980	7.85	18.6	S
7	0.154	8.29	8.70	WF
7	0.133	7.93	9.70	WF
7	0.267	7.79	10.7	WF
7	0.782	7.72	12.5	WF