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Municipal Drinking Water Quality and the Lead and Copper Rule (LCR): Conceptual Frameworks, Implementation Biases and Resultant Disparities in Community Health

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We must learn to think critically, be creative, and have the courage to become agents of social change. (Duntley-Matos et al. 2017)

Abstract

This paper evaluates the history and societal impacts of regulation of lead (Pb) in drinking water in the United States from the Safe Drinking Water Act (1974) to currently proposed revisions to the Lead and Copper Rule (LCR) (2021). This evaluation intends to link demonstrations of bias in implementation of drinking water policy to demonstrations of inequity in resultant drinking water quality to demonstration of disparity in health access within impacted communities. The central narrative presented herein with respect to Pb in drinking water is that decision frameworks that have underlain the development of drinking water policy have resulted in implementation biases that disparately impact the quality of municipal drinking water service provision in the U.S. That implementation biases create disparities in the quality of service provision that mirror fault lines of race and class in the United States is examined as demonstration of deep entrenchment of structured and reinforced bias by race × socio-economic status (SES) in how we think about 'deservingness' of health and resource access in this country.

Following evaluation of the history of drinking water regulation and service provision, this paper summarizes public health data related to impacts of elevated blood lead levels (BLL) and historical and contemporary trends in the relationship between the level of Pb in drinking water and BLL. Subsequently, this paper focuses specifically on the drinking water crisis that began in Flint, MI in 2014 following an economically-motivated switch in the city drinking water supply. This crisis resulted from a myriad of negatively reinforcing political, economic, legal and regulatory failures and has resulted in the exposure of the city's citizenry, including 10,000+ children, to elevated concentrations of Pb in residential drinking water. In evaluation of dynamics occurring in Flint during and following the acute phase of the drinking water crisis, this paper presents and evaluates frameworks for contextualizing community response within the crisis and provides strategies for external professional engagement – including public health engagement – in support of resilience for and within socially, economically and politically marginalized communities.

Introduction | Regulatory History

In 1974, the United States Congress promulgated the Safe Drinking Water Act (SDWA) to focus on protection of public health through regulation of municipal drinking water supplies. Under the SDWA, regulation entailed establishment and enforcement of criteria addressing contaminants that impact human health (primary regulations) and parameters such as appearance, taste and smell that impact drinking water aesthetics (secondary standards). Primary regulations are defined in terms of a maximum contaminant level (MCL) – a value associated with impacts on human health; or by a value or value range defined with respect to requirements of specified water system treatment techniques. Primary regulations, codified in 1975 as National Primary Drinking Water Regulations (NPDWR)[1], are updated by the US EPA through the setting of Maximum Contaminant Level Goals (MCLGs). MCLGs are health-based and non-enforceable, and provide contextual framework for adjusting MCLs based on advancements in understanding of health impacts from contaminant exposures. MCLs, in contrast, are enforceable and deviate from MCLGs based on *feasibility* of attainment. Factors that may make achievement of a MCLG infeasible include treatment costs and associated practical constraints on choice of water distribution or treatment technologies.

In 1985, US EPA proposed NPDWRs for 39 organic and inorganic chemicals and microbiological parameters, a list that included lead (Pb).[2] The 1985 MCL for Pb was set at 50 micrograms per liter of water (ug/L); the 1985 MCLG for Pb was set at 20 ug/L. Under NPDWR, monitoring occurs at the entry point of distribution, meaning at the discharge from the water treatment plant. Because the source of Pb in drinking water originates in the materials used in water distribution – including lead-containing service pipes, brass and bronze fittings, water meters, valves, faucets and leaded solder – a 1986 amendment to the SDWA banned the use of Pb-containing materials in drinking water pipes and plumbing, including in residential lines connected to municipal drinking water supplies. Chemically, lead (Pb) leaches, is solubilized, or scales into drinking water as the result of corrosion of service lines and pipe fittings, with

[1] https://www.govinfo.gov/content/pkg/CFR-2019-title40-vol25/xml/CFR-2019-title40-vol25-part141.xml

[2] There are currently (2021) 80+ contaminants on the NPDWR list; in addition to organic and inorganic chemicals and microbiological parameters, the list also includes radionuclides, disinfectants and disinfection by-products.

corrosion occurring either/both as the result of reactions between service line materials and the water's ambient chemistry and/or as the result of additives – such as chlorine – used to reduce the potential for microbiological growth in service lines and/or the transport of microbiological agents that cause illness. Water typically leaves a municipal treatment plant with a non-detectable concentration of Pb, and corrosion of pipes, fittings and solder within the service line network – including residential[3] services lines and fixtures – increases Pb levels by the point water reaches residence spigots/taps. In terms of geographic impacts, while the use of lead service lines and/or lead-lined steel or iron lines was somewhat restricted regionally in the U.S. (Cornwall, 2018), lead solder was used nationwide.

Temporally, installation of Pb service lines occurred in the U.S. from the late 19th century through the late 20th century (1986), with the majority of Pb service line emplacements having occurred prior to the 1940s (WWII). It has been estimated that there are currently 6 million Pb service lines in the U.S. and that 15-22 million residences are served by municipal water supplies that contain either partial or full lead service lines (Cornwell et al, 2016). Documented knowledge of the negative health impacts of use of Pb in drinking water service lines dates back to the 19th century in the U.S. and Europe, with the outlawing of Pb in drinking water pipes occurring in Europe as early as the 1870s.[4] Knowledge regarding the health implication of Pb exposure in drinking water, notwithstanding, Pb piping has long been used in water service networks because Pb is easily cut and jointed and the malleability of leaded materials allows for flexibility and displacement resistance in residential service lines connections (colloquially known as "goosenecks"). Use of Pb pipes and leaded solder (which contained up to 50% Pb by weight) in water main connections was common practice in the U.S. until the late 1950s and was not discontinued until the 1986 Federal ban (Cornwell et al., 2016).[5]

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[3] 'Residential', 'residence' and 'resident' are used throughout this paper to refer to individuals, where they live and where/how service lines connecting the municipal supply piping to the point of use for individuals or families are constructed; residential, residence and resident do not imply homeownership nor that drinking water consumers live in single family structures. Biases in assumptions regarding exposure scenarios are discussed later in this paper.
[4] Arguably, knowledge – or at least evidence – of the detrimental health impacts of consuming Pb in water or wine dates back to the Romans. https://arstechnica.com/science/2021/07/did-lead-poisoning-cause-downfall-of-roman-empire-the-jury-is-still-out/ | Accessed 09.16.21
[5] Solder containing 40-50% Pb can still be commercially purchased although is illegal to use in connection with drinking water plumbing Contemporary 'Pb-free' solder must contain no more than 0.2% Pb. Contemporary 'Pb-free' brass and bronze fittings used until very recently in drinking water distribution could legally contain up to 8%

In 1991, with the stated goal of halving elevated Pb exposures from drinking water by the year 2000, the USEPA promulgated the Lead and Copper Rule (LCR)[6]. The LCR superseded the previous MCL of 50 ug/L with an Action Level (AL) of 15 ug/L and defined the MCLG as zero. Recognizing that the principal driver of Pb in drinking water is corrosion *within* the pipe distribution network downstream of the water treatment plant, the LCR focused on achieving its goal by setting treatment requirements for controlling corrosion in the distribution network. Distinct from other chemicals on the NPDWR list, the LCR therefore mandated that Pb and Cu be monitored not just at the water treatment plant, but also at residential taps with taps providing a means for monitoring the effectiveness of corrosion control by sampling at the point of use. As stated in the LCR, the AL for triggering review of the effectiveness of the extant corrosion control treatment technology is determined via the concentrations of Pb and/or copper (Cu) at the 90th percentile level of a sampled water taps.[7] Specifically, with respect to the AL, municipal sampling of drinking water for assessment of Pb and Cu should entail residential collection of 1 liter (L) of water as 'first draw' (meaning, without flushing the tap or the sample container prior to sample collection) after water has been standing in residential pipes for at least 6 hours.

With respect to Pb, action levels are conceptually equivalent to MCLs in that both the MCL and the AL are based on technical or feasibility constraints versus health-based thresholds for safe exposure. A distinction between an AL and an MCL, however, and a principal reason that the LCR is an advancement over the prior NPDWRs is that exceedance of an AL triggers requirements for municipal response. Municipal response is mandated in terms of enhanced monitoring and/or treatment of source water, implementation or improvement of corrosion control, public education regarding the AL exceedance and replacement of Pb-containing service lines. As promulgated under the LCR, in the absence of AL exceedances, water utilities serving

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Pb; new regulations for 'Pb-free' brass and bronze limit Pb content to 0.25% by weight. As example: https://www.copper.org/applications/rodbar/pdf/a7036_brass_drinking_water.pdf | Accessed 09.18.21

[6] 40 Code of Federal Regulations (CFR) Part 141 Subpart 1 - https://www.epa.gov/dwreginfo/lead-and-copper-rule | USEPA (US Environmental Protection Agency), 1991. Drinking Water Regulations. 40 CFR Parts 141 and 142. *Federal Register*, 56:110:26460. October 1991.

[7] As example, if 100 water samples are collected and the data are arranged from the lowest concentration of Pb (numbered sample #1) to the highest concentration of lead (numbered sample #100), the 90th percentile level would be the Pb concentration of sample #90. If 50 samples were collected and arranged in the same way, the 90th percentile level would be the Pb concentration of sample #45 ($50 \times 0.9 = 45$).

fewer than 50,000 residents are not required to maintain a standing corrosion control program, while water utilities serving greater than 50,000 residents are required to maintain and optimize corrosion control as standard practice. While ALs are therefore an improvement over prior regulatory management strategies for Pb in drinking water, a significant concern with this approach as codified in the LCR is that AL-oriented sampling is focused primarily on confirmation of water treatment effectiveness versus explicit protection of health for residential consumers. With respect the Pb exposure, the distinction is important in that in implementation of the LCR, the EPA defined the AL at 15 ug/L as a balance between instrument/analytical capability and cost, projecting that up to 25% of municipal systems were (in the early 1990s) likely to register exceedances at this concentration at that time (Maas et al., 2005). Importantly with respect to public health, the LCR does not identify a Pb threshold above which fines would be levied (against municipalities or water utilities) for violating mandates to provide safe drinking water and protect human health.

Implementation Bias and Implications for Community Health

With respect to the regulatory background summarized above, a question to examine in evaluating the justification for selection of 15 ug/L as the AL for Pb in water under the LCR, is whether the selection of this value explicitly addressed the health impacts of this decision. Embedded within this question is, in turn, an additional significant question regarding the distribution by race and/or socio-economic status (SES) of these health impacts and whether that distribution reveals bias (implicit or explicit) on the part of regulators defining safety in municipal drinking water. Most importantly with respect to evaluating the context for selection of an AL > 0 for Pb in drinking water is the scientific consensus that for infants and children there is no physiologically safe exposure concentration for lead. Exposure to lead can result in neurological, developmental, cardiovascular, musculoskeletal and immunological damage, the results of which cause acute and long term impacts on individual, family and community health. Exposure to lead is also dangerous for pregnant women as the ability of Pb to cross the placental barrier (as well as the blood-brain barrier) creates significant potential for fetal exposure *in utero*.

Overall, the concentration of Pb one is exposed to, the total exposure amount, the form (e.g., in water, as dust, in soil, in food, in air) of exposure, the developmental timing, frequency and duration of exposure, and one's overall underlying health and nutritional status[8] all contribute to the potential severity of health impacts. Lead in the human body is typically measured in blood as a blood lead level (BLL) and is reported in units of micrograms of lead per deciliter (ug/dL) of blood. As summarized and contextualized by Brown and Margolis (2012), BLLs for Native Americans prior to European arrival have been estimated at 0.016 μg/dL; the estimated (measured) average BLL for children < 5 years old (y.o.) in the United States in 2000 was 1.9 μg/dL, a 100× increase resulting from ambient environmental exposures. Currently (2021), the United States Center for Disease Control and Prevention (CDC) applies a threshold BLL screening concentration of 5 µg/dL to identify children with elevated levels of Pb in blood. This screening concentration is a statistical threshold defined as the concentration threshold for the highest 2.5% BLLs for children less than 5 years old in the U.S.

As noted above, many factors including the timing, frequency and duration of exposure impact the magnitude of physiological effects, and relationships between exposure concentration, BLL and either short-term or long-term health consequences can vary considerably. Importantly, there has been no BLL measurable below which physiological, neurological or developmental impacts have not been documented. Currently, children < 6 y.o. represent more than 75% of the Pb-attributed disease impact in the U.S. and it is estimated that as many as 500,000 children have BLLs greater than or equal to (≥) the CDC 5 μg/dL threshold (Hauptman et al. 2017). While the majority of childhood exposures to lead are currently via contaminated dust or paint chips, it is estimated that up to 20% of early childhood Pb exposure results from municipal drinking water (Maas et al. 2005). Ngueta et al. (2016) have evaluated potential health risks for children < 5 y.o. who are exposed to Pb and have estimated that cumulative exposure in drinking water at concentrations significantly below the AL (including as low as 1 ug/L) can measurably increase BLL over exposure intervals on the scale of months or less.

Exposures via drinking water are also a significant concern for infants who are formula-fed, as contaminated drinking water may represent more than 85% of their Pb exposure (Katner

[8] It is important to state here that factors including the calcium and iron content of food, total calories consumed and the frequency of meals all influence the extent to which Pb in solid foods or liquids is absorbed into the body (https://www.canr.msu.edu/resources/fight_lead_with_nutrition). These data highlight the additional risks that Pb exposure poses for children and families who do not have enough to eat, including nutritionally dense foods.

et al., 2016) and infants (as well as children) intestinally absorb a significant percentage of ingested Pb (Rosen et al. 2020). With respect to the relationship between Pb concentrations in water and BLL, U.S. EPA has proposed a linear association with different – and steeper - absorption rates for children than for adults (Rosen et al., 2020). In confirmation of this differential absorption potential, Edwards (2009), in studying the drinking water crisis that occurred in Washington D.C. from 2000 – 2007, observed a significant relationship between the frequency of elevated BLLs and the 10% exceedance values under the LCR for drinking water in different neighborhoods within the city. The strength of this relationship was highest for children < 1.3 y.o. (versus older children or adults) and included correlation between elevated BLLs and elevated drinking water Pb concentrations, as well as between elevated BLL and differences in frequency of lead pipes and service lines between different affected neighborhoods in the city. The overall correlation between elevated drinking water Pb concentrations and elevated BLL in children has also been demonstrated in other studies including in Montreal, Canada, in which for every 1 ug/L increase in Pb in drinking water, there was a documented corresponding 35% increase in BLL for evaluated children (Ngueta et al., 2015) and in Flint, MI, in which significant geospatial overlap by neighborhood has been demonstrated for increasing BLL and increasing social vulnerability (as defined by a cumulative index assessing factors including nutritional status, housing age, socio-economic status and employment status (Hanna-Attisha, 2016).

Over the period of the 1970s – 2000s, the overall number of children with BLL ≥ 10 ug/dL declined from 13.5 million to 250,000, representing a decline from greater than 70% of children to fewer than 1% of children (Brown and Margolis, 2012; Rosen et al., 2020). This concentration of 10 ug/dL, 2× the CDC threshold for elevated BLL, represents the concentration of concern for state and local health agencies, above which case management visits are triggered for home visits and health education. As summarized by Rosen (2020), it is important to note that tap water sampling appears not to be routinely conducted as part of residence-based follow-ups for elevated BLL in children and when it does occur there is no consistent state-by-state protocol for how sampling should be conducted or whether water sampling should occur regardless of documented evidence (or not) of Pb-based paint chips or dust in the residence. Demographically, in the early decades (1970s – 1980s) of this recent era of Pb monitoring and abatement efforts, while only an estimated 2% of White children had significantly elevated BLL

(≥ 30 ug/dL), approximately 12% of Black children tested presented with BLL above this threshold for *significantly elevated* levels of Pb in blood. During this period, statistics presented in terms of SES were similarly skewed, with approximately 10% of children in low-earning households versus 1.2% of children living in higher-earning households presenting with BLL ≥ 30 ug/dL. By 2000, along with the overall significant decrease in children with BLL ≥ 10 ug/dL, statistically significant differences in BLL by race or SES in children at either the ≥ 30 ug/dL or ≥ 10 ug/dL threshold concentrations had disappeared (Brown and Margolis 2012).

Importantly, however, although statistically significant differences in BLL relative to elevated threshold values have disappeared, disparities in risks for potential exposure to Pb remain. Data from 2007-2008 demonstrate, as example, that average BLLs remain elevated for non-Hispanic Black children, with national pediatric screening results averaging 36% higher than BLL results for White children (1.9 μg/dL versus 1.4 ug/dL) (Brown and Margolis 2012). While these averaged values are below the current CDC threshold for elevated BLL, data continue to demonstrate that children exposed to even very low concentrations of Pb can suffer from attention deficit, behavioral challenges, growth stunting, hyperactivity, impaired coordination and impacts on IQ, reading ability and academic performance (NTP 2012; Rosen 2020).

In the context of drinking water quality and the protection of public health, the LCR requires that first-draw sampling be conducted at specific residences or, more generally, in neighborhoods with elevated risks of lead and/or copper contamination. Characteristics of residences or neighborhood that could be indicative of elevated Pb or Cu exposure risks via drinking water include older residences (built prior to 1950) and, more broadly, residences in neighborhoods in which previous sampling has documented elevated Pb and/or Cu concentration in first-draw sampling. For large municipal water systems (> 100,000 residents) first-draw sampling includes testing of taps at ~ 100 residences for determination of AL compliance. As per the requirement of LCR compliance sampling, the number of residences within the testing pool may be reduced by 50% if the 90% concentrations for Pb and Cu are below the respective ALs over two consecutive rounds of monitoring. For municipal water systems serving fewer than 100,000 residents, the maximum number of first-draw tests required for standard LCR compliance monitoring is 60; reduced frequency monitoring following demonstration of consistent compliance with LCR ALs is 30 tests or fewer.

Of legitimate concern with these low numbers of samples (as few as 0.1% of residents or, if it is assumed an average of 3 residents per domicile, $\leq 0.3\%$ of domicile taps) is whether acts of omission on the part of water managers could bias water testing results. Specifically, could choice of sampling locations bias the 90^{th} percentile values low relative to ALs and/or could bias exist in how AL exceedances are addressed in the event of failures in treatment technology. These questions are significant because if such omissions or bias occur in municipal water management and if these occurrences mirror social fault lines with respect to race and/or class, the results would create systemic inequalities in access to safe drinking water. Critically, as highlighted by VanDerslice (2011), while > 95% of the U.S. population is at least potentially served by municipal (and therefore, regulated) water systems, water utilities are not required to collect or analyze socio-demographic data for the users they serve. This lack of socio-spatial examination of service provision directly hampers robust assessment of potential disparity in safe drinking water provision.

What data do exist from the studies that have been undertaken, however, demonstrate both racial/ethnic and socio-economic bias in the quality of drinking water service provision. Switzer and Teodoro (2017) have paired SDWA compliance records from 2010-2013 with local-level demographic and economic data for water utilities serving > 1,000 people, and statistically evaluated direct and intersectional relationships between SDWA compliance, communities' racial/ethnic composition and SES. Results of their analysis demonstrate intersectional impacts of race/ethnicity and SES on drinking water quality, with the strength of the race/ethnicity impact on SDWA violations most pronounced in low-SES communities. Likewise, Butler et al. (2016) has compiled major instances of LCR AL exceedance for Pb in residential taps and/or public schools and has documented such bias in cities including Washington D.C. (2001 – 2004; impacted community is 61.5% BIPOC with 18.2% living below the poverty line); Columbia, SC (2005; impacted community is 48.3% BIPOC with 24.2% living below the poverty line); Durham NC (2006; impacted community is 53.6% BIPOC, with 18.1% living below the poverty line); Flint, MI (2014; impacted community is 62.6% BIPOC with 41.6% living below the poverty line); Jackson, MS (2015; impacted community is 81.6% BIPOC with 29.9% living below the poverty line); and Ithaca, NY (2016; impacted community is 43.7% BIPOC with 30.7% living below the poverty line).

Presented together, these data convincingly argue the reality of systemic inequalities as a function of race × SES in access to safe drinking water in this country. Importantly, in addition to water treatment requirements in the event of 10% exceedance values,[9] municipalities with AL exceedances are required to notify the occupants of the residence in a timely fashion (typically, within 30 days specifically to the resident(s) and within 60 days to the general public as a public service announcement) and to undertake a strategy of partial lead service line replacements (PLSLR). Under the PLSLR, a minimum of 7% of identified lead-containing service lines are required to be replaced annually until subsequent LCR sampling declines below the AL. With respect to PLSLR and SES, water utilities are required to offer to replace residents' leaded service line, but are not required to subsidize or cover the cost of replacement if residents are unable to afford the multi-thousand dollar replacement costs. With research demonstrating: (1) that PLSLR can result in both short-term (through physically dislodging Pb particles and scale in premise piping) and long-term (through the creation of a galvanic corrosion circuit between newly emplaced municipal copper pipes and remaining premise piping containing Pb) exposure concerns; and (2) that BLL in children can be higher in residences with PLSLR versus comparable residences in which municipalities have not undertaken PLSLR (Triantafyllidou and Edwards 2012), the inability to afford premise service line replacements can result in disproportionate Pb exposure risks as a function of SES. That is, the potential health impacts resulting from the means by which PLSLR is currently mandated and practiced represent a direct penalizing of the poor by local, state and federal governments.

Katner et al. (2016) have critically evaluated biases and failures in implementation of the LCR in Flint, MI during the prolonged drinking water crisis that surfaced in that city following

[9] If the 90% value exceeds the AL this is equivalent to saying that 10% of collected samples have Pb concentrations greater than 15 ug/L; the threshold approach of the LCR is sensitive to the number of values that exceed the AL, but not the actual Pb concentrations of those exceedances. In Flint, MI, as context, citizen monitoring of municipal drinking water in August 2015 – 16 months following the switch in city drinking water supply – the 90% value for Pb for first-draw samples (consistent with LCR sampling protocol) was 26.8 ug/L; 17% of samples tested exceeded the AL of 15 ug/L; the highest concentration measured was 158 ug/L or 10.5× the Action Level (Pieper et al., 2018).

an economically motivated switch in the city drinking water supply in 2014. The failure of state and federal regulators to protect the citizens of Flint, MI from elevated concentrations of Pb in city drinking water has resulted in thousands of children being exposed to dangerously high concentrations of Pb and, in addition to significant negotiated monetary settlements, replacement of city service lines and indictments against local and state officials, likely irrevocably damaged public trust in the quality and fair distribution of municipal water services in that city (as well as elsewhere). As detailed by Katner et al. (2016), failures both in implementation of the LCR and enforcement following AL exceedances occurred in Flint, MI, and included how samples were collected, where sampling was conducted, how samples were processed for laboratory analysis, how data were evaluated, how and when the public was notified, and how and when enforcement authority was (not) transitioned appropriately from the state to the federal government following reporting of LCR violations.

These failures are not unique to Flint, MI. Presenting summary data from a 2006 U.S. Government Accountability Office (GAO) national survey of LCR implementation and enforcement, Katner et al. (2016) note that as per the GAO, more than 30% of water utilities were not reporting water lead levels (WLL) to the EPA, more than 70% of water utilities were not sharing data with the EPA regarding implementation of LCR requirements following AL exceedances, and many utilities could not provide documentation that sampling locations were being chosen consistent with LCR protocols. As summarized by Katner et al. (2016), insufficient monitoring is likely resulting in a nationwide under-estimation of the extent and severity of Pb exposure risks in drinking water, a reality that may be undercutting public health efforts to eliminate childhood Pb poisoning in the U.S.

While, overall, monitoring data do suggest that the majority of large water utilities are in compliance with the LCR (e.g., Triantafyllidou and Edwards, 2012), the overall declining quality of U.S. drinking water infrastructure (awarded a C- in 2021 by the American Society of Civil Engineers in the annual ASCE infrastructure report card[10]), lack of funding for research in this domain (discussed further below), and well documented challenges with health-protective implementation of LCR sampling and data utilization, all raise legitimate concerns for our path

[10] https://infrastructurereportcard.org | Accessed 09.18.21

forward with drinking water quality in the United States. With specific reference to the LCR, it should be reiterated here that with the system-wide LCR focus on 90th percentile compliance for effective management of Pb in drinking water, there is no MCL for Pb in drinking water that would result in violation of Federal regulation. Moreover, while the LCR applies to residence supplied through municipal utilities, it specifically excludes the majority of schools (including daycare centers) that also rely on municipal water. The 90% of U.S. schools that rely on municipal water are covered under distinct regulation.[11] While not specifically the focus of this paper, Triantafyllidou and Edwards (2012) have compiled case studies of drinking water exceedances in public schools and have documented concerning incidents including extended time periods between determination of samples in exceedance of the LCCA threshold (20 ug/L) and public reporting (including 20+ years and information only released following FOIA filing) and drinking water Pb concentrations in significant exceedance of 5000 ug/L, a concentration equivalent to Federal designation of leachable hazardous Pb waste.[12]

In 2015, the National Drinking Water Advisory Council (NDWAC) provided input to the US EPA regarding potential long-term revisions to the LCR (NDWAC 2015). Previous updates and revisions to the LCR – focusing on monitoring frequency; treatment optimization requirements; service line replacements; and requirements for public education – had been undertaken in 2000[13] and again in 2007[14] following the drinking water crisis in Washington D.C. Recognizing that there is no safe level for Pb exposure and that the protection of water quality is both complicated and expensive, NDWAC concluded that water quality protection is a '*comprehensive shared responsibility between federal, state and local government, public and private utilities, and customers*' (NDWAC 2015). Critically, however, as noted by NDWAC,

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[11] Drinking water in schools and day care centers is covered under the 1988 Lead Contamination Control Act (LCCA); the LCCA, as with the LCR, does not have a regulatory structure for violations and is not enforceable, recommending only that sampled water does not exceed 20 ug/L Pb. Exceedance under the LCCA typically results in drinking fountains being removed which addresses one exposure concern although may not eliminate exposures entirely. Of specific additional concern could be exposures via drinking water in bathroom sinks and through cafeteria food preparation using Pb-contaminated water.

[12] https://www.epa.gov/sites/default/files/2015-10/documents/chap7_0.pdf | Accessed 09.24.21

[13] 65 FR 1950, U.S. EPA 2000 | https://www.govinfo.gov/content/pkg/FR-2000-01-12/pdf/00-3.pdf#page=50 | Accessed 09.23.21

[14] https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100DP2P.txt | USEPA. 2007. 40 CFR Parts 141 and 142. Drinking Water Regulations. Part IV: Short- Term Regulatory Revisions and Clarifications. October 2007. Accessed 09.23.21

revisions to the LCR that focus principally on optimization strategies for corrosion control do a disservice to public health and raise questions of disparate impacts and environmental justice on issues of (for example) being unable to finance household service line replacements. Of specific proposed inclusion in the revised LCR are health-based requirements for supporting residential-requested tap water testing as well as the establishment of residence-based ALs that trigger direct reporting of results both to residents and to appropriate local and state agencies to ensure medical follow-up is provided as necessary (NDWAC 2015).

In addition, under evaluation with respect to long-term revisions to the LCR is the question of decreasing the AL for Pb to < 15 ug/L. As might be expected, this question represents an area of contention between overall agreement that Pb exposures should be reduced (and so warranting a lower AL) and awareness that ALs are often misapplied or misinterpreted as safe exposure levels, thereby complicating interpretation of what a *safe* concentration level at a home tap might be. Conceptually, while reducing the Pb exposure AL from 15 ug/L to 10 ug/L or lower may be achievable through optimization of corrosion control and replacement of lead service lines, a lower AL will create challenges with laboratory instrumentation (current analytical detection limits for Pb can be as high as 5 ug/L [Cornwell, 2018]), and still represents a Pb concentration 5-10× higher (at least) than concentrations considered medically safe for pregnant women, infants and children.[15] The crux of the issue here is then one of distinguishing between revisions to the LCR that will improve individual and community health versus revisions that will instead further complicate interpretation and implementation of an already complex treatment technology-focused rule.

From the vantage of drinking water consumers, support of residentially-requested professional testing at homes, apartments, and schools is a constructive revision to the LCR. Currently, although a variety of point of use (POU) water test kits for Pb do exist, study results suggest that there is no commercially available POU test that consistently and reliably detects Pb at concentrations at or near 15 ug/L (Kriss et al. 2021). Likewise, the potential regulatory approval of specific at-home POU water filters is a positive proposed revision to the LCR. For

[15] The American Academy of Pediatrics (https://www.aap.org), as example, considers 1 ug/L as the maximum recommended allowable concentration of Pb in drinking water, although there are practical limits on analytical capabilities for routine water sampling for Pb at this concentration (Masters et al, 2020).

POU water filters, in contrast to POU water test kits, effective, available and fairly inexpensive filters do exist for limiting Pb exposure in drinking water, especially for children (e.g., Deshommes et al. 2010). Ideally, free POU filters along with a comprehensive and consistent framework for filter exchange should be established to provide at-tap protection for residents who are concerned for their and their families health, but unable to shoulder the financial burdens of paying for this necessary peace of mind.

Nationally, as highlighted by Triantafyllidou and Edwards (2012) as well as by Rosen (2020), there is surprisingly little Federal attention, and therefore, directed funding, for strategies to minimize exposure to Pb through drinking water contamination. Rosen (2020) notes, as example, that over the years for which funding data were publicly available (2008 – 2016), less than \$1.5M in Federal research funding was allocated to studying Pb in drinking water. Relatedly, Triantafyllidou and Edwards (2012) have observed that with limited Federal attention directed toward public health, funding decisions around Pb exposure prevention have been forced into a hierarchy of perceived risks versus rewards in designing interventions for reducing exposures from soil, dust, paint chips and drinking water. Of specific concern here – beyond overall indignation that such prioritization is required in the context of roughly \$6.5B in Federal non-defense spending over this same interval (2008 – 2016) of which approximately 50% was allocated to health or health-adjacent spending with approximately 13% of that funding (\$46B) directed generally toward Pb-related health research (Rosen, 2020) – are questions of why so little attention continues to be paid to helping residents of this country manage infrastructure-related health exposures in the midst of our ongoing national infrastructure crisis and, most importantly, whether those individuals and communities most likely to bear the burdens of chemical exposures are present, represented and centered in framing and discussion of chemical monitoring, hazard mitigation and health promotion.

With respect to questions of bias in funding and directions in health-related research it is also critically important that we see and address omissions and assumptions that underlie cost-benefit or risk-versus-reward analyses. Such analyses often serve as foundation for creation and implementation of policies designed – at least on the surface and with unexamined disregard for underlying and embedded social biases – *in service to the greater good.*[16] As an example focused on the setting of the MCL and subsequent AL for Pb in water, in 1986 the U.S. EPA Office of Policy, Planning and Evaluation conducted a cost-benefit analysis focused on reducing the concentration of Pb in U.S. drinking water (Levin, 1987). This analysis was undertaken in conjunction with the inclusion of Pb in NPDWRs and in parallel with the drafting of the SDWA Lead Ban (enacted in 1986) that mandated the use of "lead-free" pipes and solder in installation or repair of plumbing that carries water for human consumption. The analysis included a summary of what was known at the time about the occurrence of Pb in drinking water; the health effects of Pb exposure on children and adults; an assessment of cost savings with respect to corrosion control and pipe material stability; and a summary of annual estimated costs and benefits associated with Pb control in drinking water.

Results of this analysis suggested that (based on 1985 dollars and considered as annual costs or savings), reduction of Pb concentrations in water from 50 ug/L (the MCL at the time) to 20 ug/L could result in roughly $900 million annual benefits including savings in avoided medical expenses for children ($27M) and adults ($292M), avoided compensatory education costs for children suffering from acute Pb exposure ($81M) and water treatment to control corrosivity and thereby protect utility and domicile plumbing ($525) (Levin, 1987) Annual costs associated with reducing corrosivity of water and improving water quality (and thereby limiting Pb dissolution and scaling and reducing Pb exposure at the tap for children and adults) were estimated at $240M, generating an overall estimated benefit-to-cost ratio ~ 4:1 for reducing Pb in drinking water from the (then) MCL of 50 ug/L to 20 ug/L.

Overall, this analysis, while strongly supportive of the social benefits of Pb exposure reduction, contains significant biases and omission in the generation of the values from which the costs versus benefits were assessed. While this analysis was conducted 35 years ago and we are arguably now in a different place socio-culturally in our collective ability to see and address race- and class-based biases in decision-making, there is value in highlighting the blind spots that

[16] See Washington and Foster (2016) for a conversation regarding Flint, MI and what Dr. Foster details as the lack of individual and community redress for violation of environmental laws "*geared towards the greatest pollution reductions for the greatest number of people*," versus being geared toward addressing the reality of inequality.

underlie (in this case) monetary or monetized assessments of resource use and protections for human health. As presented, the analysis begins with a rounded estimate of 42 million Americans currently (1987) exposed to Pb at a concentration > 20 ug/L in drinking water (Levin, 1987). This estimate was derived based on four factors: number of municipal water users, percentage of residential water samples > 20 ug/L, number of new housing starts,[17] and number of individuals per household per new housing start. In addition to overall uncertainties in the size of the population relying on municipal water in the U.S., additional biases and omissions in this calculation include: (1) a likely underestimation of the population exposed to significantly elevated concentrations of Pb in solder (reason: estimate only focuses on single family purchased dwellings and does not assess whether residents of newly constructed or repaired rental units or apartments are similarly exposed); and (2) a potentially significant underestimation of the percentage of domicile taps with Pb concentrations > 20 ug/L (reason: estimate is drawn from private company data[18] without confirmation that individuals able to hire a contractor for water testing experience similar Pb exposures to those without equivalent financial resources.

With respect to costs associated with children's exposure to elevated Pb concentrations in drinking water, the approach employed to assess human capacity loss included calculation of: costs of (short-term) remedial/compensatory education; costs associated with decreased future earning potential as the result of IQ loss; and savings (if water lead levels declined to 20 ug/L) in medical attention sought from the effects of acute Pb poisoning (Levin 1987). Underlying this cost-benefit approach regarding children's health are omissions regarding: major physiological effects such as long term kidney or liver damage that occur with prolonged Pb exposure; assessment of whether the presumed 3 years of (part-time) compensatory education is actually sufficient to prevent lasting cognitive damage following prolonged Pb exposure; and evaluation of the merits of only costing that which could be priced at the time, namely the cost of acute medical intervention for BLL > 25 ug/dL (Levin 1986).

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[17] Annually; the most significant exposure window for Pb from solder used in residential service line construction occurs during the initial years of water consumption; thus 'number of new housing starts' provided the baseline 'unit' for elevated exposures following 'new' construction.
[18] Data generated by Culligan Water Softening Company (Levin 1986)

Of concern here, and the reason that this analysis has assessment value for this report although it is dated, is that while the answer that was generated – that there is a 4:1 benefit in societal savings relative to industrial costs for reducing Pb in drinking water from 50 ug/L (the MCL at the time) to 20 ug/L – has considerable value in the argument regarding the importance of reducing Pb exposure, the demographic under-estimations of exposure (most particularly with respect to race and SES), as well as the absence of discussion of long-term and integrated health consequences in the lives and the circumstances of individuals who are not well represented in these exposure scenarios, makes the actual numbers meaningless. That is, as example, if the BLL threshold used in calculation of potential medical savings is 5× higher than the current CCD-defined threshold for considering a BLL *elevated* and likely 10-25× higher than levels medically demonstrated to result in long-term health impacts for children (e.g., NTP 2012; Rosen 2020), what is the actual value in so unrealistically constraining exposure and response scenarios so as to create a monetized comparison of '*worthwhileness*' or '*deservingness*'?

There is a dissonance created in the linearizing, quantifying and simplifying of reality so as to permit 'rationalized' (in this case, monetary) analyses that results in both the obscuring of the actual *challenges* of accessing clean water (if you can't afford to call a private water service company) as well as the rendering invisible of marginalized people and communities whose health is most likely to suffer from its absence. That rationalized policies prioritize solutions and analyses that are purely technical and rationally neutral within socio-political systems that are anything but, creates dynamics in which decision-makers are willing to accept significant potential risks to residents' well-being as the r*easonable* consequence of their economically sound decisions.[19] That is, that the LCR is generally functioning as intended is not equivalent to saying that health-first legal and/or technical policies are in place to pro-actively identify community water systems that may be on the brink of failing or to ensure that protections are in place for communities in the event that it is their water system that fails next (e.g., Jacobson et al.

[19] As example, promulgation and implementation of the LCR as the strategy for monitoring the quality of municipal drinking water provides a threshold-based approach for action/response that makes it possible to simultaneously say that while annually, approximately 90% of community water systems (CWS) are in compliance with the LCR, failures in compliance for the remaining 10% of CWS leave > 20 million residents exposed to drinking water impairments (including impairments responsible for the > 16 million reported annual cases of acute gastroenteritis linked to municipal drinking water in the U.S.) (Allaire et al., 2018).

2020). This structural shortcoming to the implementation framework of the SDWA is a failure of public health protection on many levels.

As explored by Hughes (2020) as well as others (e.g., Cassano and Benz 2019; Jacobson et al. 2020), the cumulative impact on a community of failures in how policies are created, implemented and superseded is as damaging to the body of the community as the cumulative impact of chemical exposures is to the body of the individual.[20] As has been documented in examination of underlying factors that amplified the drinking water crisis in Flint, MI, the appointment of an Emergency Manager – a fiscal decision invoked in cities in response to significant economic distress – emplaces and empowers a politically-appointed official to make financially-motivated short-term decisions without explicit regard for public health or community protections (e.g., Hughes 2020). That slightly more than half (53%) of Michigan's Black population has found themselves forced to live under austerity-emplaced Emergency Managers over the past 8 years versus only 2% of MI's White residents[21] is sufficient demonstration of how the structured role of rationalized, economically-motivated decision-making creates conditions of health disparity for impacted communities.

Community Health – Conceptual Frameworks

In the context of frameworks, the first section of this paper evaluated the LCR and the regulatory framework through which water quality is monitored and exceedances of the AL for Pb in drinking water are addressed. As explored in this evaluation, the regulatory framework for limiting or preventing exposure to Pb in municipal drinking water is constructed on a foundation

[20] Jacobson et al. (2020) review the overlaps and omissions in how authority is shared (or not) across levels of government focused on drinking water quality. As they document in analysis of the drinking water crisis occurring in Flint, MI, the legal framework for implementation of the SDWA and the legal framework for protection of human health are not well integrated. As specific example, neither US EPA nor the Michigan state Department of Environmental Quality (MDEQ) – the two agencies with (sometimes competing) prevention authority for protection of drinking water (via implementation and enforcement of the LCR) – have agency expertise in public health nor do decision made regarding water quality by either agency require public health consultation or input. Likewise, Michigan Department of Health and Human Services (MDHHS), the agency with state-level authority regarding protection of human health, has no capacity for surveillance, detection, investigation or intervention with respect to exceedances of the LCR AL. The Emergency Manager framework, in turn– invoked and enacted in Flint (as well as elsewhere) in response to fiscal austerity triggers, supersedes ALL local authority for prevention, surveillance, detection, investigation or intervention to protect either human health or drinking water quality.

[21] https://www.michiganradio.org/politics-government/2017-12-06/lawsuit-states-emergency-manager-law-discriminates-against-black-communities || Accessed: 09.21.21.

of treatment technology optimization. Specifically, the AL for triggering review of the effectiveness of the extant corrosion control treatment technology is determined via the concentrations of Pb and/or copper (Cu) at the 90th percentile level of sampled water taps. Conceptually, this foundation is focused on the prevention of technological failure more specifically than it is focused on health goals for safe consumption of municipal drinking water.

This distinction matters significantly when viewed through the lens of crescive crises. Explored by Beamish (2002), crescive crises are those events to which decision-making culture doesn't respond until a threshold has been exceeded. With specific focus on the context evaluated in this paper – public health and drinking water quality – crescive crises are what occur when a system is calibrated toward a default that prioritizes operational efficiency over community health, the outcome of which is to undervalue the impacts of chronic exposures on marginalized individuals and communities. They are what occur when complexity is reduced to a binary over/under that allows a box to be checked confirming that there isn't a problem today. Crescive crises are the result of a mindset that defaults to making hierarchical decisions for and from a biased perspective on the common good. The threshold that ultimately triggers response can be as a significant change relative to a level previously and commonly considered as 'background' or, as has occurred in Flint, MI as well as elsewhere, the result of individuals and agencies skirting their mandates until they are caught red-handed. These categories of thresholds or precipitating events are, of course, often tightly linked because they describe a similar paternalistic worldview in decision-makers.

One specific outcome of this default calibration toward maintaining the status quo is the tendency to ascribe a sense of the unusual to an event that is, in fact, an early indicator of system instability. A challenge with applying these narratives of the unusual to these events is that what may be the overlapping/reinforcing aspects or phenomena that result in amplification of action or impact above 'background' may have as much or more to do with the media narrative as with the actual supposed uniqueness of the event. As example, the drinking water crisis in Flint, MI is the result of a series of disparate factors – cultural, economic, political, technical, environmental, historical – that have overlapped spatially. Importantly, however, this overlap of factors isn't unique, and as discussed earlier in this paper, is occurring – and disproportionately impacting –

marginalized communities in many places simultaneously. Also importantly, while the news media may move on from coverage in any specific location and allow the sense (distinct from the reality) of crisis to recede from public view, the issues themselves – including the impacts on public health resulting from decisions made by the wrongly centered few – don't go away.

Much has been written about the myriad and negatively reinforcing political, economic, social, legal and regulatory failures that have created the crescive crisis in Flint, MI, and some of that context has been summarized herein. The failures at play in Flint, MI are structural (borne of history); spatial (city-wide, although with considerable access disparity with respect to information and resources); relational (alternating power hoarding and power defaulting between agencies as advantageousness); individual; and somatic (embodied). At the level of the individual, it is important to acknowledge that implementation of the LCR is not value-neutral and bias has been documented in where compliance-oriented tap water samples were (and continue to be) collected (or not); how tap water samples were collected; how the data were evaluated and what was done (or not) in response to the review of the compliance data (e.g., Goovaerts 2017; Henderson and Wells 2020; Pieper et al., 2018). As also highlighted in these studies, bias can (and did and does) also occur via absence of response, including inaction in the face of crescive data; regulatory indifference and both overt and subtle blaming of individuals for their personal lack of resources (e.g., Cassano and Benz 2019; Pulido 2016; Hammer 2019). As has been investigated and reported by the Michigan Civil Rights Commission:

> *The people of Flint have been subjected to unprecedented harm and hardship, much of it caused by structural and systemic discrimination and racism that have corroded your city, your institutions, and your water pipes, for generations. When the last of the civil lawsuits and the Attorney General's criminal investigations are completed, and relief dollars from state and federal sources are exhausted, what will remain is a city and its people who will continue to fight against built-in barriers but whose voices – as a matter of public right – must never be stifled or quelled again. The Michigan Civil Rights Commission and the Michigan Department of Civil Rights will do their part to proactively engage you, listen to you, advocate for you and defend you, to the fullest extent allowed by the civil rights laws of the state of Michigan.*[22]

[22] https://www.michigan.gov/documents/mdcr/VFlintCrisisRep-F-Edited3-13-17_554317_7.pdf |Accessed: 10.08.21

Dimensionally, as noted above, health-impacting scenarios or contexts can be conceptualized as structuralized, as well as spatialized. A spatialized context describes three dimensional space and maps the geographic extent of the now. Spatialized context describes how an individual moves through their worldscape and/or how a location - an intersection, a neighborhood, a city - exists within a larger organizational framework (in this case: a neighborhood, a city, a county or state). A spatialized worldscape is racialized if differences exist as a function of race in how an individual or a location (such as a neighborhood) experiences health, including safety. A structuralized context, in extension, expands upon this mapping to include time; it describes a four dimensional map of context. Structuralized worldscapes that are grounded in history, as with the water crisis in Flint, MI are racialized. A structuralized worldscape that looks to the future may be less so if spatialized disparities as a function of race and/or SES can be lessened in the now.

In addition to this conceptual temporal pivot, an additional pivot with relevance to the questions examined in this paper is in the viewpoint transition from a structural-spatial focus on prevention of failure – such as occurs with the LCR and AL triggers for Pb and/or Cu – to the community-individual-embodied focus on the creation of psychological space for health. One fulcrum on which these opposing perspectives (im)balance is in the extent to which the perpetuation of health inequity in the face of crescive crises is understood as an active or passive response to socio-cultural demands to address structural indifference and hostility. From an activist perspective, this statement is asking the questions: where are the levers for structural change and how can they be accessed in ways that unbalance existing cultural, regulatory, political, and economic frameworks of statis? Additionally, how can this work be done in ways that increase the capacity for health for communities who have been actively under-resourced by these same frameworks?

Wilson (2009) explores this question through an ecological | ecosystem framework built on the premise that living conditions and the capacity for health in urban landscapes are determined through higher order structures built on the foundation of the U.S. race-based development policies. Those policies include both legal (*de jure*) and administrative (*de facto*) segregation and restrictive covenants, suburbanization, urban renewal and gentrification. As summarized by Wilson (2009), within and undergirding the concept of (dis)functioning society, are therefore active processes of underdevelopment and destabilization that perpetuate imbalances in the capacity for quality of life; processes that operate on the macro- and meso-scale as functions of race × SES. Thus, with respect to health, while individual-scale factors may influence the specific choices that any person makes, it is larger social | structural factors constraining (or, for some, amplifying) opportunity that place both visible and unseen boundaries around the capacity for health. Capacity, in this context, includes both the ability to avoid situations, dynamics and exposures that are harmful or health-restricting and the ability to access resources (e.g., nutritious foods; medical support; physical mobility; safety) that are health-improving, and should be seen as actively shaped by the existence and totality (if present) of buffers and protections to the cumulative impact of socio-physical stressors that are pressed on to vulnerable communities and that function to undermine individual and community resilience. Within this framework, health must therefore represent more than simply the absence of acute dangers, limitations and exposures.

In discussion of the active imbalancing of environmental resource access that creates disparities in community capacity for health, Wilson (2009) defines a human ecological continuum between salutogenesis – those processes, factors and dynamics that support, enrich and enhance capacity for health – and pathogenesis – those processes, factors and dynamics that diminish, constrict or undermine this capacity. Examples of human ecological salutogens – that is, components of social and built environments that increase a community's capacity for health include sufficient and high quality housing stock, urban greenspaces, adequate medical treatment facilities, functioning water and sanitation infrastructure, transportation infrastructure that improves access without impeding physical mobility, available and reasonably priced nutritious foods and safe recreational opportunities for children (amongst others). Examples of human ecological pathogens, in contrast, include those factors that act as social, physical, economic, cultural and environmental stressors resulting from how and where infrastructure is placed (or not); how and where infrastructure is maintained (or not); and how and where individual and community opportunities are constrained (or not) by societal decisions regarding who is deserving of local environments generally free of visible drug use, violence, police interference, industrial facility siting and housing dereliction.

With a specific focus on drinking water, Balazs and Ray (2014) build on Wilson's (2009) assessment of active imbalancing through creation of a framework for foregrounding disparities in chemical exposures. As explored in their research, the meso- and macro-scale factors that create or constrain access to sufficient and safe drinking water cannot be equally mitigated or confronted by all municipal water users, thereby creating disparity in individual and community capacity for self-protection from socio-technical pathogenesis (as per Wilson 2009). As traced by Balazs and Ray (2014) for agricultural communities in the San Joaquin Valley (CA), persistent and composite exposures to multiple contaminants in drinking water (including nitrate and arsenic) distribute by race/ethnicity and SES as the result of inadequate water monitoring, unequally applied regulatory protection, sub-standard service provision (resulting from unjustly enforced rate structures and/or significantly delayed relaying or responding to water quality violations) and municipal decisions regarding annexation versus exclusion from provision of city services, resulting in the active withholding of resources from specific communities. These factors driving disparity are spatialized as a function of community location and structuralized by history, as they result from and continue to be exacerbated by past (and ongoing) constraints regarding what environmental, socio-political and infrastructural (e.g., built environment) conditions specific communities were (are) permitted to live within. As framed by Balazs and Ray (2014), the structure of pathogenesis for agricultural communities in the San Joaquin Valley (CA) is a negatively reinforcing feedback loop of multi-level constraints, within which a community's inability to self-protect creates exacerbated potential for chemical exposure.

Importantly, within the disparity framework delineated by Balazs and Ray (2014), the '*unleveling of the field*' is foregrounded as the on-going and active imbalancing of access that it is. That is, when an individual is rendered unable to reach toward health (salutogenesis) by socio-economic or cultural constraints on their ability to access the services that society claims to offer without disparity, they are being told to cope within a structure that withholds protective measures. As highlighted by Balazs and Ray (2014) in discussion of costs incurred by agricultural communities in CA to access safe drinking water,[23] and as echoed throughout this

[23] The US EPA affordability criterion for drinking water for small community water systems (<10,000 people) is defined as 2.5% of median U.S. household income (MHI); for an annual MHI of $67,000 (U.S. 2020), this equates to $1,6750/yr. https://www.awwa.org/Portals/0/AWWA/ETS/Resources/AffordabilityAssessmentTool.pdf

paper, regulatory frameworks for maintaining drinking water infrastructure routinely, and in myriad ways, fail lower-income and otherwise marginalized neighborhoods and communities.

What must also therefore be brought into relief within a disparity framework is a community's actual capacity to respond to (in this case) failures in drinking water protection and resultant chemical exposures. Conceptually, *capacity to respond* is, itself, commonly viewed within a disparity or deficit framework. As example, the Social Vulnerability Index (SVI)[24] created and maintained by the U.S. Center for Disease Control and Prevention (CDC) and the Agency for Toxic Substances and Disease Registry (ATSDR), aggregates census tract-level data on 15 parameters grouped under parameter headings of Socioeconomic Status;[25] Household Composition & Disability;[26] Minority Status & Language;[27] and Housing & Transportation.[28] While created initially as tool for county- and state-level officials to assess and respond to community needs in the preparation for and/or aftermath of natural disasters, the SVI is increasingly been used in socio-temporal contexts including in conjunction with historical insurance mapping to examine long-term cultural, economic and health impacts of *de jure* 20th Century segregation policies,[29] and as a public health assessment tool for identifying individuals and communities in need of non-acute (i.e., not in direct response to a natural disaster) support (e.g., Gay et al., 2016 in assessment of youth activity and physical fitness within the built environment). The underlying framework for these questions is significantly important for understanding constraints on capacity to access to health through a focus on the negative (pathogenic) impacts of place on health. What these models do not build, however, is an understanding of how to reduce vulnerability by increasing resilience and capacity. That is, how do we (in any way that *we* can be defined constructively for this purpose) strengthen community

(Accessed 10.10.21) As stated in Balazs and Ray (2014), low-income agricultural communities in the San Joaquin Valley (CA) can be required to spend up to 10% of monthly income on water service provision and still not achieve salutogenesis with respect to water quality. The crisis in water affordability is a growing problem in the United States and was one of the specific economic factors that underpinned the city of Flint, MI decision to seek a new source of drinking water, the decision that created the Pb exposure crisis still developing.

[24] https://www.atsdr.cdc.gov/placeandhealth/svi/index.html | Accessed: 10.12.21

[25] Specific parameters: income relative to Federal poverty level; employment; income; and high school diploma.

[26] Specific parameters: aged 65 or older; aged 17 or younger; civilian with a disability; single parent household.

[27] Specific parameters: minority status; individuals in household age 5+ who speak English 'less than well'.

[28] Specific parameters: housing as multi-unit structures; mobile homes; crowding (metric not defined); absence of vehicle; group quarters

[29] Not Even Past: Social Vulnerability and the Legacy of Redlining - https://dsl.richmond.edu/socialvulnerability/ | Accessed: 10.12.21

capacity and improve resilience such that acute response in the face of crescive crises is not the default operational strategy?

Strengthening Community Capacity – Recommendations

An evaluation of context for strengthening community capacity and engagement requires explication of strategies across differing degrees of validation of existing power dynamics and politico-technical hierarchies. One visible form that engagement within existing politico-technical hierarchies can take is community involvement in the creation of scientific narratives. In Flint, as example, and with respect to the narrative of the acute water crisis, researchers from Virginia Polytechnic Institute and State University (Virginia Tech) actively collaborated with members of the Flint community to conduct multiple rounds of tap water testing over the interval 2015 – 2017 (e.g., Pieper et al. 2018). As conceived by Virginia Tech, the collaboration included the university researcher team – providing Federal grant funding, analytical support and a technical plan for water testing – and residents from Flint – providing local knowledge and connections to publicize and facilitate community-wide sampling. The community team from Flint was coordinated by a local homeowner who had been the index resident ("Resident Zero") initiating discussion with a US EPA Region 5 regulator who was raising concerns[30] about water quality in Flint. Ultimately, 5 municipal system-wide water sampling campaigns were conducted involving 268 residences participating in at least one of the 5 rounds of community data collection (Pieper et al., 2018). The results of this sampling campaign were instrumental in generating the data that forced local, state and Federal response to the emergent drinking water crisis.

Roy and Edwards (2019), in reviewing the research collaboration initiated during the initial years (2014 – 2015) of the water crisis, focus on what they define as a distinction between community-engaged science and citizen science. Specifically, Roy and Edwards (2019) draw attention to what they perceive as a feedback between the damage to public trust inflicted by municipal mishandling of the water crisis and the platform handed to individuals and citizen-science organizations with leverage out of proportion to their knowledge of science and/or public health practice. From their perspective, the most significant negative impact of this dynamic was

[30] Miguel A. Del Toral – June 2015 Memorandum - http://flintwaterstudy.org/wp-content/uploads/2015/11/Miguels-Memo.pdf | Accessed 10.17.21

the robust platform that community distrust provides for the dissemination of misinformation. As example, publicly recognizable activists' insistence that city water should not be used for washing or bathing (EB[31]) coupled with the promotion and distribution of an unvalidated proprietary water testing method (MR[2]; WaterBug™) combined to create a strong counter-current message that may have led directly to the summer 2016 outbreak of gastrointestinal Shigellosis in Flint Shigellosis is spread through poor hygiene, including specifically, lack of hand-washing and bathing. Importantly, with respect to the 2016 outbreak in Flint, MI, residential tap water samples collected during the outbreak were negative for the *Shigella* bacteria (indicating that the bacteria were not coming from the drinking water), suggesting that widely disseminated misinformation on the dangers of bathing may have exacerbated conditions facilitating an acute and extended community outbreak of this illness (Roy and Edwards, 2019).

As specific counterbalance to how community engagement is framed by the Virginia Tech initiative and amplified in turn by the extensive (and savior-oriented) media response to the Water Team's presence in Flint, Johnson and Key (2018) and Carrera et al. (2019) return the focus to the Flint community's centrality in their own narrative. In identifying the root cause of the water crisis in society's moral and technical divestiture from the lived experiences of African American communities, Johnson and Key (2018) and Ezell et al. (2021) highlight the reality that media narratives focused on external saviors create a storyline that frames success in terms that actively perpetuate that divestiture. Such framing – described by Carrera et al. (2019) as '*violation by experts in a crisis situation'* disregards the perspectives, voices and experiences of marginalized communities who most directly receive the brunt of health impacts from society's technological failures.

As counterpoint to this framing – and specifically with focus on who holds funding, how research questions are framed and resolved, and the impact of these factors on who the media reaches toward for narrative framing of crises – Johnson and Key (2018) identify what they believe is critical for successful community-centered (versus external savior-centered) public health initiatives; specifically, commitment to continual re-focusing on community priorities and perspectives; and equitability in attribution of credit, sharing of expertise and data, and allocation

[31] Activist Erin Brockovich (EB); and actor/activist Mark Ruffalo (MR)

of public and philanthropic funding for research and services. Moreover, with community presence and narrative centered in conceptualizing engagement, it should be argued that the most significant consequence of the damage to public trust inflicted by municipal mishandling of the water crisis is in the direct impact of that damage to community health (mental, physical, emotional, spiritual) and resilience.

As summarized by Carrera et al. (2019), if community resilience is the goal of engagement on issues of environmental and public health crises (both from within affected community and from outside of those communities), that goal can only be reached for, fought for and elevated if it is centered in how engagement is approached. While community participation is always and absolutely necessary for integrated public health solutions, how that participation is conceived, invited and embodied matters significantly for whether success gets defined in terms of resolving a specific technical problem (e.g., reducing the concentration of Pb in drinking water) or strengthening community capacity to self-advocate (e.g., re-allocating resources and attention to address biased social structures and dynamics). The distinction here is one between framing the crisis in Flint as one that began with the city switch to the Flint River water supply in early 2014 and ended with municipal re-connection to the Detroit water supply (i.e., Lake Huron) in late 2015 versus an understanding of the crisis as one of marginalization, disengagement and chronic neglect that only became visible and 'news-worthy' to non-residents (including the scientific community, external media and the political and regulatory bureaucracy) at the point that it became not possible to not respond and engage.

Within communities impacted by disengagement and chronic neglect, resilience can (and should and does and needs to) reside in the capacity for response to acute crises (in technical infrastructure, in the case of Flint) without damage to or loss of community social structure (e.g., Carrera et al. 2019). That so much of what gets defined as acute socio-technical crisis is, in reality, crescive as a result of underlying socio-economic hierarchies and structures of racism requires a distinction be made between resilience as 'capacity to effect progressive change in the re-balancing of underlying disparities' versus 'capacity to return to conditions that preceded the crisis'. This distinction is therefore one between resilience as an active embodying and experiencing of healing versus a suggestion that health lies is in the capacity to adjust to conditions of racial and socio-economic inequity. Critically, the distinction here focuses on the reality that responding to crises takes time, energy and community capacity that in the absence of that crisis could have been spent in engagement on other community priorities. This perspective is offered in direct contrast to media narratives that either ignore local community capacity to effectively organize or that center the role that crisis is (externally) perceived to have played in catalyzing or galvanizing community response.[32]

A significant strategy for interaction with politico-technical hierarchies that is also often misinterpreted or mis-framed by media narratives is self-support outside of hierarchies. Ezell (2021) document the entirely unsurprising and generally under-reported longer term physiological and psychological impacts of the Flint water crisis on adults within Flint communities. Significantly, results from community-based questionnaires distributed in Flint in 2019 demonstrated that more 25% of adult respondents met evaluation criteria for trauma (29% of respondents) and either/both depression and anxiety (26% of respondents) (Ezell, 2021). These under-reported longer term mental health consequences of crescive public health crises are currently significantly under-acknowledged and untreated and create what Miller and Wesley (2016) describe as the potential for escalating and enduring trauma that is reinforced through fear of uncertain consequences and health impacts. With sometimes limited medical and broader socio-cultural support for impacted communities following city-scale crises and with often great potential for media dissemination of false narratives of emerging resilience (and therefore allowance for non-response),[33] self-support is imperative capacity for survival.

Self-support in this context is distinct from citizen participation in scientific research or other demonstrations of engagement within hierarchies of power. Specifically, self-support or *self-provisioning* (e.g., Sadler et al., 2021) centers the networks and organizations that hold people together rather than reinforcing the linearity of structures and dynamics that vertically link individuals to power (or not). With respect to Flint, MI, networks of self-support that pre-

[32] This particular form of pernicious media narrative needs to be seen as the self-soothing savior narrative that it is.
[33] It should be noted here (and repeatedly) that this discussion does not absolve anybody with resources from offering support and demanding change from any vantage that can help level disparity. That a community has capacity for self-provisioning is not equivalent to saying that assistance isn't needed. Michael Harriot – journalist, poet and Senior Writer for The Root – has recently noted: "*If [white people] knew about these disparities, they would have to conclude that they benefit from them, which might make them feel bad. But it also might make the next generation of [young people]work to eliminate this structural discrimination. A person can only fix a problem if they know the problem exists, and educating [young people] on America's history of systemic racism is the only way to fix it*." | https://www.theroot.com/maybe-white-people-should-feel-bad-1847918518 | Accessed 10.23.21

dated (and now post-date) the acute phase of the drinking water crisis include, as examples, community mutual aid networks – both within Flint and across a wider network of MI cities (e.g., Duntley-Matos et al., 2017; Howell et al., 2019) that draw support and resources through interfaith and interdenominational church networks, grassroots engagements to coordinate and distribute resources, including water and food,[34] and, to the extent that they remain mission-aligned, local Foundation and non-profit supports. Critically, and significantly, self-support outside of the politico-technical hierarchies of power is entirely interwoven with protest and rebellion *against* those politico-technical hierarchies. That is, within a framework of spatialized and structuralized racism and historical inattention, self-provisioning is a centrally activated and necessarily disruptive demonstration of individual and community resistance.

Conclusions

From the vantage of community response to the water crisis, the multiple and conflicting currents of information amplified by the megaphone of media, have generated engagement across a broad spectrum from pro-active (or reactive) coordination to reactive (or pro-active) disruption. It is not my place (nor anyone's observing from a distance the dynamics that occur in response to crescive crises) to critique or judge the ultimate value of disparate community responses in Flint, MI. To do so without clearly understanding how *successful* engagement is conceived, framed, and enacted within a community, is to willfully ignore how legitimate anger at histories of inaction and inattention distribute differently by race and class (as well as gender) within society. In their reflections on the Flint Water Study Team's participation in Flint, MI, Roy and Edwards (2019) conclude that "*social justice advocates are sometimes willing to openly embrace faulty methods or even misconduct to achieve a populist objective, thereby creating a high likelihood of confrontation in future high profile cases like the Flint Water Crisis.*" This statement, grounded in the premise that populist objectives are of lesser value than rigorous scientific objectives, suggests clearly that, from the researchers' perspective, it is the steadying

[34]Examples: Edible Flint: http://www.edibleflint.org | Community Foundation of Greater Flint: https://www.cfgf.org | MSU: https://www.canr.msu.edu/news/community_gardens_cultivate_healthier_neighbors_in_flint | Flint Mutual Aid and Assistance: https://www.facebook.com/groups/766085643915802/ | 2015 International Social Movements Gathering – Detroit & Flint – including the Detroit to Flint Water Justice Journey protest and community walk: http://peoplestribune.org/pt-news/2015/06/international-social-movements/ | Accessed 10.23.21

and guiding hand of science that protects impacted communities from the instabilities and disruptions that impede the resolution of crises.

While a statement like Dr. Roy and Dr. Edward's may well be true from some perspectives (e.g., consider the current health crisis in the U.S. resulting from opposition to COVID-19 vaccination mandates), it also negates the reality that different objectives can be achieved by different methods employing or embracing different visions of *success*. Specifically, the worldview centered by Dr.'s Roy and Edwards prioritizes order over action and conceives of confrontation as the messy result of poorly actualized science. What Roy and Edwards (2019) need to acknowledge directly – and do not – in this framework is that showing up professionally in opposition to unjust regulations, policies or laws is not precisely equivalent to showing up in service to the individuals who are impacted by the injustice; these objectives are, of course, linked, but they are also distinct. While it may be possible, professionally, to avoid confrontation while conducting science in support of rebalancing socio-economic and cultural marginalization, the ability to do – and/or to claim the right to frame that confrontation if it does occur as counterproductively disruptive – is a poor measure of achievement or success.[35] As Carrera et al. (2019) note, if a disconnect exists between how academic scientists views community partnerships (often, as an access point for samples and local labor for that sample collection) and how community members view interaction with academic scientists (as a potentially useful strategy and platform for elevating voice and concerns), disagreement or conflict shouldn't be surprising.

In describing challenges and limitations imposed on community members in Flint during invited collaboration in the 2012 - 2016 city Master Planning Process, Lederman (2019) describe a similar potential for conflict involving, in this case, technocrats in civic management and urban planning. As described by Lederman (2019), if community involvement is solicited within a

[35]As endnote to Roy and Edwards (2019), the authors state: "Dr. Roy's professional presentation on the case study presented in this paper at an academic conference (fPET 2018 on May 31 2018) was disrupted by Mr. Paul Schwartz. Schwartz also handed out copies of a letter, which Dr. Edwards alleged to be defamatory, to participants at this event. Dr. Lambrinidou and Resident A (of Flint, MI) are both discussed in this paper and also named defendants, along with Mr. Schwartz, in Dr. Edwards' lawsuit alleging personal defamation, which Dr. Roy is not a party to. However, the defendants' lawyer, Mr. Bill Moran, sent Dr. Roy Twitter messages opining "[Dr. Edwards] should quietly exit this case," and has repeatedly attacked him publicly on Twitter (e.g., "You are no victim, Sid. Grow the f*ck up").

narrowly conscripted set of choices and frameworks, the results often achieve the objectives of the decision-makers (e.g., the ability to point to public engagement within urban planning as demonstration of responsiveness to community needs) without ceding either the authority to determine the scope of engagement or the capacity to challenge it to the community for whom (but not, actually, with whom) decisions are being made. Such theater of participation is deeply damaging to public trust in municipal services and agencies.

As has been discussed throughout this paper, the damage to public trust that municipal, state and Federal neglect creates is both utterly justified and, rooted as it is in a long history of the same, very difficult to overcome. During the 2015-2016 interval of the water crisis in Flint, it is true both that residents within Flint believed accurately and angrily that the magnitude of the city's water contamination problems was driven directly by structural racism at the level of the municipality, its agencies and the state of MI (e.g., Hammer, 2019), and that state public health agencies and practitioners within that framework were working diligently and earnestly to build community collaborations to coordinate public health messaging; conduct Pb exposure assessments; and link affected community members with guidance and resources for self-protection and medical follow-up (e.g., Hanna-Attisha et al., 2016; Ruckart et al., 2019).

Likewise, it is true that although indictments have been handed down against local and state officials on charges including misconduct in office, willful neglect of duty and perjury in response to their individual actions in perpetuation of Pb exposures in the community of Flint,[36] these individuals are not simply aberrative in agencies that can otherwise point toward consistent demonstration and implementation of equity-oriented policies and programming. Indeed, that officials are individually corrupt does not absolve the system they work within, specifically because – as has been documented throughout this paper – the system itself in which these officials operate is racially and socially biased. From the perspective of those on the receiving end of environmental and social injustice, the challenges that such dynamics create have significant potential to further escalate distrust. To wit, during the 2014-2015 interval in which the magnitude of drinking water contamination in Flint was known to regulators but prior to any action taken to protect public health, the EPA Region 5 Water Division Branch Chief is on

[36] Nine Indicted on Criminal Charges in Flint Water Crisis Investigation https://www.michigan.gov/som/0,4669,7-192-47796-549541--,00.html | Accessed 10.16.21.

record as having sent the following in an e-mail: "*I'm not so sure Flint is the community we want to go out on a limb for*."[37] The only useful answer to this blandly incendiary comment is a question: *why not Flint*? to be followed by a second and equally urgent question: *Isn't this who the technological improvements and process optimizations required to provide clean and safe municipal water should be for*?

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[37] https://www.detroitnews.com/story/news/michigan/flint-water-crisis/2016/03/15/hearing-epa/81805068/ | Accessed 09.24.21

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