Just Water?
Social Disparities and Drinking Water Quality in California's San Joaquin Valley

By

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ABSTRACT

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California's San Joaquin Valley is one of the world's richest agricultural regions yet it is also home to some of the greatest environmental problems, including drinking water contamination. After decades of intensive agriculture in the San Joaquin Valley (SJV), the region's aquifers and rivers are some of the most contaminated in the nation. This creates a notoriously difficult environmental problem to regulate, and related public health and environmental justice issues. Ninety-five percent of the SJV population relies on this contaminated groundwater for drinking thus creating an exposure risk. Contaminant exposures are further compounded by the fact that with high costs of treatment, few water systems are able to afford mitigation, especially under-resourced communities. Yet most of our understanding of water in the San Joaquin Valley concerns agricultural water use, or environmental water quality of rivers, streams and aquifers. Very little focuses directly on drinking water quality, and much less on the health and regulatory implications of this contamination.

My dissertation combines the fields of environmental health science and environmental justice to examine the relationship between exposures to contaminants and the socioeconomic characteristics of drinking water systems. Combining both fields allows me to explore which individuals and communities are most vulnerable to drinking water contamination, whether these groups are equipped to mitigate exposure at household, community or regional levels, and what underlying processes impact exposure. In doing so, this dissertation contributes to a growing field of research that addresses the impacts of contaminated drinking water supplies and inadequate service provision in the U.S., but still has considerable gaps. While the environmental justice literature focuses on the extent and causes of disproportionate environmental burdens, it has largely failed to examine drinking water issues. While the environmental health arena has contributed a plethora of studies on drinking water exposures and health outcomes, it has mainly focused on issues in the developing world, and has not always addressed social disparities in the U.S. with regards to water.

To fill these gaps, my dissertation addresses three sets of questions: 1) Are there social disparities in exposure to drinking water contaminants in California's San Joaquin Valley? 2) Are there social disparities in the ability of water systems to comply with drinking

water standards? 3) What are the social, political and environmental processes that explain the origins and persistence of observed disparities and their associated health and regulatory implications? Underlying these questions is a hypothesis that scale-alone (i.e., small system size) does not fully explain disparities in drinking water contamination and compliance abilities, and that a focus on demographic composition of water systems may further elucidate which communities are most vulnerable.

Using mixed methods, I answer these questions by focusing on community water systems throughout the Valley, and exploring the relationship between nitrate and arsenic contamination and community demographics. To answer the first two questions, I combine two main sets of historical datasets of drinking water quality maintained by the California Department of Public Health. With this data I estimate distribution water quality and contaminant exposure, and compliance with federal standards at the water system level. I then use statistical modeling techniques to examine the relationship between race, class and exposure to nitrate and arsenic in water systems. To answer the third question, I rely on primary ethnographic data that includes semi-structured interviews and participant observation with county and state regulators, drinking water advocates and community residents. I complement this primary data with media and document reviews relating to drinking water contamination in the San Joaquin Valley.

My results show that among smaller water systems, those serving larger fractions of Latinos have higher nitrate levels in their drinking water. This provides evidence of an environmental inequity. I also find that systems with lower rates of home ownership have higher arsenic concentrations in their drinking water. In addition, these systems have higher odds of receiving an arsenic maximum contaminant level violation. For arsenic, these results indicate that communities with fewer economic resources face a dual burden—they are not only exposed to higher arsenic levels, but are also served by non-compliant systems. I conclude by developing a new social epidemiology framework that captures the multiple challenges created by natural, built and social environmental factors. I use the framework to argue that these multi-level driving factors impact both coping abilities and exposure at the community and household level. In sum, my dissertation highlights the distributional and procedural inequities that exist with regards to drinking water contamination and compliance with drinking water standards. In doing so, this research challenges the notion that drinking water problems are only a matter of system size and elucidates the drinking water disparities that low-income communities and communities of color face.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF ACRONYMS AND ABBREVIATIONS

As—Arsenic
CDPH—California Department of Public Health
CWS—Community water system
DBCP—Dibromochloropropane
EJ—Environmental Justice
GIS—Geographic Information System
IQR—Interquartile Range
MCL—Maximum Contaminant Level
MCLG—Maximum Contaminant Level Goal
mg/L—milligrams per liter
NGO—non-governmental organization
NO_3 –Nitrate ion
PEP—Population potentially exposed
PICME—Permits, Inspections, Compliance, Monitoring and Evaluation
PPB—Parts per billion
PUC—Public Utility Commission
PWS—Public Water System
RPHL—Recommended Public Health Level
SJV—San Joaquin Valley
SDWA—Safe Drinking Water Act
SES—Socioeconomic status
TMF—Technical, Managerial and Financial
U.N.—United Nations
U.S. EPA—United States Environmental Protection Agency
WQM—Water Quality Monitoring database
μg/L—micrograms per liter

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INTRODUCTION

In the most prosperous state of the richest nation in the globe, there are towns with Third World problems.

-Benito Ortiz, Porterville Recorder, December 18th, 2004

Beginnings

At the end of my first semester of graduate school, this newspaper quote caught my eye. While I had spent the semester reading and discussing issues of access to clean drinking water in the Global South, four hours from the booming Bay Area were towns that lacked access to a constant supply of drinking water and frequently violated federal drinking water standards.

That spring, seeped in the theory of water economics, water and development, and river hydrology, I packed my bags and hopped on the last Amtrak train to Wasco, California, in Kern County. For a week I drove with colleagues from a local non-governmental organization (NGO) through endless stretches of fallow agricultural lands and bursts of orange groves. Under the blazing sun, farm workers wrapped their heads in bandanas and worked hunched over in the fields. Once off the main highways and thoroughfares, I could go forty-five minutes before passing any semblance of a town. And when I did, the small enclaves I passed were often dusty, desolate and run-down; though they did not lack passionate residents, local pride and a welcoming atmosphere.

On one of these stretches of highway, after passing a state penitentiary, and then miles of desolate industrial agriculture, I arrived in the town of Alpaugh. "Welcome to Alpaugh. Bienvenidos a Alpaugh" a friendly but faded mural read on a bridge that crossed over an agricultural canal full of water. Pulling up to the town's one café, I met Luisa[1], a Native woman who grew up on a nearby reservation and dreamed of moving to the "big town" of Alpaugh. She smiled, remembering her hopes. In recent years, Luisa had founded Community for a Better Alpaugh, and led the town on its quest for clean drinking water.

Over the course of the afternoon, Luisa gave me a tour of the town in her white pick-up truck, which overflowed with meeting notes and folders. This, she said, was the town's main well that had failed for several months, intermittently leaving the local school and residents without a piped water supply. Over here was the town's backup well that had also recently failed. She noted that both wells violated the federal drinking water standard for arsenic, one of the most carcinogenic water contaminants regulated by the Safe Drinking Water Act. Across the way, she pointed to the community's storage tank: it's supposed to get chlorinated to protect the town water supply. But, she noted, the system's operators don't always chlorinate because it's too expensive. Right next to each other were the two water agencies that governed water provision for nearly 70 years, for this town of roughly 700 residents. The state had recently forced them to join forces in order to be eligible for state funding.

Alpaugh was not alone, Luisa explained. Across the Valley, small, predominantly farm worker, unincorporated communities were plagued with contaminated drinking water. Households were forced to spend their limited income on purchasing bottled water *and* paying for their water utility bill. Sometimes local officials denied the requests of residents

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1 Though this interaction occurred before the start of my research, I have changed the respondent's name.

to clean dirty water, until communities organized and brought in state regulators and lawyers. All this, she said, and people still think this is just a local problem.

I returned to Berkeley with all of this churning in my mind. One year later, after a master's project focused on equity in watershed management in Brazil, the Valley was still calling to me. The fact that, as Ortiz notes, there were Third World problems in my own backyard made me rethink my personal mission to work in my ancestral Latin American homeland, and explore its reaches in the North. All that remained was to develop my analytical frame, hypotheses and research directions.

As that path unfolded, people often asked me one of five questions. Are there really drinking water problems in the U.S.? Isn't the issue of contaminated drinking water and demographics of water systems *just* an issue of economies of scale, where small systems simply face the biggest problems? In talking about social disparities are you implying that someone is deliberately polluting people's water? If there is no statistical correlation between race, class, and water quality does that mean there is no injustice? If people are drinking contaminated water, why don't they just move to a new community?

I have found it useful to continuously engage with these questions, not because I agree with their limited perspectives, but because they have required me to unearth the assumptions embedded in them, clarify why focusing on drinking water in the U.S. matters, and understand how best to frame my analysis and develop solutions that address the basic problems experienced by communities like Alpaugh. Thus these questions have guided my thinking, methods and analytical framings, and motivate the core goal of my dissertation—to understand the distribution and impact of drinking water contamination in California's San Joaquin Valley and develop a framework for addressing the multi-faceted problems. This introduction provides an overview of my research questions and the theories and studies upon which I build.

Overview

As emphatically noted in the above newspaper article, an array of drinking water-related problems still exists in the U.S, despite a history of investment in sophisticated water infrastructure and the existence of federal laws such as the Clean Water Act and Safe Drinking Water Act (SDWA) that regulate surface water contamination and protect the public's health (CWA 1972; SDWA 1974). Increasingly, evidence from the academic literature and grassroots groups has highlighted different aspects of these problems, from degrading infrastructure to unsafe levels of contaminants.

In many cases, small water systems face some of the greatest challenges. Of the 54,000 community water systems in the U.S., 56% serve fewer than 500 people (Roberson 2011). Lacking economies of scale, these small water systems have some of the most degraded infrastructure and are more likely than larger systems to violate drinking water regulations (Committee on Small Water Systems 1997; Cromwell 1997). Whether because investors often see small systems as a riskier investment, or because small systems that lack incorporation status are unable to levy bond monies, small systems are less likely to obtain loans for infrastructure improvements. As a result these systems often lack adequate finances, rendering them unable to hire capable staff or to develop long-term and sustainable alternatives to contaminated drinking water supplies (Shanaghan and Bielanski 2003). These economic and infrastructure problems can ultimately create an increased public health burden in these communities that are forced to endure ongoing drinking water contamination (Cromwell 1997).

Understandably, the dominant narrative is that these problems stem from a lack of economies of scale: smaller systems are worse off because of their size, and the costs that residents face are simply higher per capita costs. This dissertation uses empirical methods from environmental health and social epidemiology as well as theoretical foundations in environmental justice to explore the extent to which this assertion is true, and to propose alternate, albeit potentially complementary, explanations. For example, an increasing number of studies and anecdotal evidence have suggested that beyond scale, geography and socioeconomic status (SES) and the racial/ethnic composition of a community can compound scale-related problems. Many small rural water systems and those serving low-SES residents are unincorporated. Without a municipal government, unincorporated communities lack additional tax revenue and related fiscal support; these systems are even less able to supply safe water. And in low-income communities, residents are less able to afford rate increases, leaving them less able to mitigate contamination (Beecher 2003). Some studies suggest that in some places, communities of color, whether Latinos living in colonias in Texas and New Mexico, or African American enclaves in the rural South face greater challenges in accessing clean water, not only because of inadequate infrastructure, but due to historical discrimination, and inadequate incentives for outside service providers to render service (Marsh et al. 2010; Olmstead 2004; Pilley et al. 2009).

My dissertation explores challenges in access to safe drinking water in the U.S., the associated risks, and the populations most vulnerable to these problems. Focusing on the San Joaquin Valley, I examine social disparities in exposure to key drinking water contaminants and compliance with federal drinking water standards, and trace both the processes leading to exposure to these contaminants and the potential consequences of this exposure. I aim to understand distributional and procedural inequities related to drinking water contamination, develop a conceptual framework to disentangle the multiple factors driving drinking water burdens, and ultimately highlight critical points of intervention that may be useful at the regional, community and household levels.

In doing so this research challenges the popular notion that there are no drinking water problems in the U.S. While the burden of disease of drinking water contamination in the U.S. may be less than that in other areas of the world, and service provision near universal, the need to address existing challenges is still necessary. For just as a low-income family in the U.S. may appear much richer than a low-income family living in Latin America, this does not mean that the lived experience of poverty is any less harsh or impactful, nor that impacts of poverty or solutions to it should go unaddressed.

Theories and methods from environmental justice, social epidemiology and environmental health allow me to explore: who is most vulnerable; how environmental harms are distributed; and what procedures and processes may contribute to and exacerbate drinking water contamination. In this sense, an environmental justice frame allows us to look at health-related issues, as well as broader concepts of community vulnerability. An environmental health and social epidemiology perspective provides further motivation for analyzing how and why disparities in environmental contamination are important to consider in relation to social vulnerabilities and deficiencies in the built environment. Finally, an access to water and human rights perspective calls into question whether access to water in marginalized communities in the U.S. meets the World Health Organization's definition of access to improved water supplies[2], and underscores the importance of addressing challenges faced by some of the most marginalized groups in the U.S.

Research Questions, Key Arguments and Chapter Outline

Using a mixed-methods approach that combines statistical analyses and ethnographic methods, my dissertation seeks to answer the following questions:

1. Do community water systems serving higher fractions of ethnic/racial minorities and/or lower SES residents have worse water quality? In other words, are there social disparities[3] in exposure to drinking water contaminants in California's San Joaquin Valley?
2. Are there social disparities in the ability of water systems to comply with drinking water standards? In particular, do community water systems that serve higher fractions of ethnic or racial minorities and/or lower SES-residents have higher chances of being in non-compliance with federal drinking water standards[4]?
3. What are the social, political and environmental factors that create potential disparities and their associated health and regulatory implications?

The contribution of my research is threefold. First, it advances our understanding of environmental justice aspects related to drinking water contamination in the U.S., a topic that has been largely understudied. Second, I conduct what is, to my knowledge, the first in-depth academic study of social disparities in exposure to drinking water contaminants in California's San Joaquin Valley, and the first in the country that focuses on nitrate—one of the most common agricultural contaminants in the region (and country's) groundwater supplies. Thirdly, by integrating environmental health and social epidemiology frameworks my research advances theoretical understandings of the processes and patterns that determine community and household vulnerability to persistent contaminant exposure, and the ability to cope with contamination.

My dissertation addresses these questions in the following chapters. In *Chapter 1*, I present the details of my analytical framing. I describe U.S. drinking water challenges in relation to the international arena, and use an environmental justice framing to set-up a central focus of my research—one that seeks to explore and test the hypothesis that drinking water problems are only a problem of size, or scale. Using theories from environmental justice, environmental health and a series of historical analyses I lay the foundation for the rest of my dissertation that considers the vulnerability of water systems and residents through a social disparities angle (i.e., by race and socioeconomic status).

In *Chapter 2*, I present an overview of my methodological approach that consists of both quantitative and qualitative methods. While this chapter focuses primarily on the specific methods used, I argue that this dual approach is meant to provide a broader picture

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2 According to the World Health Organization, improved drinking-water sources include: piped water to the house or yard, public taps or standpipes, boreholes, protected dug wells, protected springs and rainwater collection (WHO 2009).

3 By social disparities, I employ a notion of distributive justice, in which one group may have more or less of a "good" or "harm". Environmental justice rests partially on part on this broad philosophical branch of justice, focusing on which communities bear a disproportionate burden of the harm, and thus face an "injustice". Throughout the dissertation, I use "social disparity" to focus on distributional injustices that vary by socioeconomic status and/or race and ethnicity.

4 This second question engages with a second notion of justice—procedural justice. This notion is largely Rawlsian (Rawls 1971) and is concerned not only with how a good or harm is distributed, but whether the process for achieving justice is fair.

of drinking water quality burdens than each method could on its own. Subsequent chapters outline specific methods in more detail, as relevant to the specific chapter at hand.

In *Chapter 3,* I analyze social disparities related to nitrate contamination in the San Joaquin Valley. Because the Valley is the site of some of the most intensive agriculture in the entire U.S., with some of the highest nitrate contamination in the country, and wide variation in nitrate levels, I explore the relationship between community demographics and nitrate levels. This chapter offers the first opportunity to challenge the "scale hypothesis"—that bad water quality is defined simply by the size of a water system. Using a statistical approach, I find that water systems serving higher percentages of Latinos have higher levels of nitrate in their drinking water, and that these effects are strongest in smaller systems. The main contribution of this chapter is that it presents the first study (to my knowledge) to focus on social disparities in nitrate exposure in the U.S., and it challenges the notion that scale alone defines water quality challenges.

In *Chapter 4* I build on the methodological approach of *Chapter 3*, but this time I focus on exposure to arsenic and the ability of communities to comply with the U.S. Environmental Protection Agency's (U.S. EPA) Revised Arsenic Rule. By focusing on what I call a "joint burden analysis", I argue for an expansion of the statistical analysis I used in *Chapter 3*. I start with the premise that the ultimate protection of the public's health in relation to drinking water rests not only on decreasing exposure to contaminants, but in enhancing the ability of water systems to comply with standards, a step which in and of itself can help ensure exposure is reduced. Thus I assess the relationship of two components—what is the public exposed to, and whether drinking water systems are complying—in relation to social disparities.

Arsenic offers an interesting case for several reasons. First, the SDWA's Revised Arsenic Rule offers an opportunity to examine both exposure and compliance with the new standard. Second, in the Valley, arsenic is predominantly naturally occurring. Its presence cannot be linked as readily to broader anthropogenic forces and patterns of development, as with nitrate. Thus arsenic provides a somewhat counter-case to nitrate, in that because its main source is natural, in theory one would not expect to see distributional inequities. Nevertheless, I find that those water systems serving higher percentages of low-SES residents have higher levels of arsenic in the drinking water, and that these systems also have higher chances of exceeding the federal standards. The main contribution of this chapter, therefore, is to simultaneously focus on both the exposure and compliance disparities associated with drinking water contamination.

While I find evidence of exposure and compliance disparities in *Chapters 3-4,* in *Chapter 5* I argue that the story does not end here. Engaging with social epidemiology approaches that support environmental justice analyses in going beyond statistical assessments, *Chapter 5* posits that a set of interacting and multi-level natural, built and social environmental factors drive exposure, and a community's ability to cope. This composite burden explains the origins and persistence of social disparities in exposure, and defines what I term "drinking water vulnerability". More specifically, the chapter uncovers how in conjunction with a baseline of contaminated source water, a series of historical planning policies have jointly shaped access to water supplies as well as to physical and financial resources for communities. These forces, alongside regulatory failures, a lack of community resources to successfully address contamination, and political disenfranchisement of local residents help explain the origins of social inequities in drinking water quality. That these same forces also influence coping capacities and lead to partial protection, at best, serves to further exacerbate the impacts of drinking water contamination and lead to its persistence.

In *Chapter 6,* I conclude by considering the broader policy implications of my findings and proposing next steps in research. Finally, the appendices, cited throughout the dissertation, provide additional supplemental information that the detail-oriented reader may find useful.

The implications of my research are relevant for the Valley as a whole and also extend to other parts of California and the U.S. While the San Joaquin Valley is worthy of study if only to better understand the region, the approach and findings in this dissertation may be relevant to other similar areas, such as California's Salinas Valley. Even more broadly, as I explore in *Chapter 1,* as a small, but growing body of literature increasingly addresses "Third World problems" in U.S. drinking water provision, these studies, including my dissertation increasingly highlight burdens faced in rural areas, within and across systems of different sizes.

CHAPTER 1
Background and Analytical Framing

The access of almost all 270 million U.S. residents to reliable safe drinking water distinguishes the United States in the twentieth century from that of the nineteenth century, and the United States from much of the rest of the world even for this century.
-Levin et al (2002)

Overview

This chapter provides contextual background that motivates my research questions. First, I place the U.S. in an international perspective and discuss key national drinking water challenges. Using theories and approaches from environmental justice research and activism, I then argue for an expansion of dominant conceptions of the root causes of drinking water challenges (i.e. low economies of scale). I bolster this argument with theories and approaches from environmental health sciences and social epidemiology. I end by motivating my use of the San Joaquin Valley as my study region.

U.S. Drinking Water in an International and Historical Context

We typically consider lack of access to safe and affordable water in a global context: worldwide, 1.2 billion people lack access to safe drinking water (World Health Organization 2009). This lack of access, alongside inadequate sanitation and hygiene, is a leading cause of disease. Annually, unsafe water and poor sanitation and hygiene cause an estimated 1.6 million deaths and are the fourth-most common set of risks impacting disability-adjusted life years (World Health Organization 2009). This burden of disease is felt most heavily in the lowest-income countries. For example, of the 88% of diarrheal deaths caused by unsafe water, sanitation or hygiene, more than 99% are in developing countries and regions with high mortality patterns, such as in parts of Africa and South-East Asia (World Health Organization 2009).

Given these statistics, it is no surprise that lack of access to improved drinking water sources is of particular concern in the global South. Simply put, piped water, public taps or standpipes, boreholes, protected dug wells, protected springs and rainwater collection are the norm in higher and middle-income countries (WHO 2009). In the U.S., for example, approximately 54,000 community water systems provide piped water to approximately 268 million residents on a year-round basis (U.S. Environmental Protection Agency 2009), and the US EPA proudly notes that the nearly 160,000 public water systems provide water to nearly all Americans at some point during their life (U.S. Environmental Protection Agency 2011d).

It follows that the U.S. is not considered a "problem area" in discussions on unsafe water and how to improve this dire situation. Indeed, the U.S. is often touted as a symbol of success for its provision of drinking water to the vast majority of its residents. The provision of centralized water and sanitation services in the 19th and early 20th centuries is often credited as one of the greatest public health victories in the country. In the U.S., an eightfold increase of filtration techniques reduced typhoid death rates from water supply by 55% from 1900 to 1913 (Ellms 1928). Though competing theories exist—including economic innovations and improved nutritional status—Cutler and Miller (2005) note that large-scale public infrastructure in the form of publicly supplied water, disinfection technologies and sanitation and refuse management were the major sources of health improvement, accounting for nearly half the mortality reductions in major cities (Cutler and Miller 2005).

Then, in the early 20th century, the Department of Health Services began to regulate drinking water quality. In 1976, the SDWA formally committed the U.S. government and the states to protecting the public from unsafe levels of contaminants. While widespread adoption of filtration and disinfection in the early 1900s had drastically reduced many waterborne illnesses, the act charged the U.S. EPA and drinking water systems to address the remaining water risks. It did so by establishing a cooperative program among local, state and federal agencies. Specifically, the SDWA required the development of primary drinking water standards, establishing maximum contaminant levels (MCLs) for both chemical and microbial contaminants. Interim regulations were adopted in 1975 on the basis of 1962 U.S. Public Health Services standards, and were amended several times through the early 1980s. The 1986 SDWA Amendments drastically updated the initial Act, proposing more realistic timelines for new standard setting, establishing wellhead protection programs, and including maximum contaminant level goals to go alongside the MCLs (FW Pontius 2003). The 1996 SDWA addressed further limitations by recognizing the need for source water protection, operator training, funding for water system improvements, and better public information (FW Pontius 2003; U.S. EPA 2009). Today, the SDWA regulates over 90 chemicals, and all public water systems are mandated to monitor and report water quality sampling results to their consumers (American Water Works Association 2003).

But despite a history of investment in sophisticated water infrastructure and the existence of federal laws such as the Clean Water Act and Safe Drinking Water Act (SDWA) that regulate source contamination and protect the public's health (CWA 1972; SDWA 1974), an array of drinking water-related problems still exists. Estimates indicate an annual funding gap of $11 billion for drinking water infrastructure (ASCE 2009). Compliance with federal regulations is not universally achieved (U.S. General Accounting Office 1990, 1999). Small systems face great challenges securing adequate funding to comply with federal drinking water standards and provide adequate service to local residents (Committee on Small Water Systems 1997; Roberson 2011). In the rural south, along the U.S.-Mexico border, and in California's agricultural regions, rural unincorporated areas are among the hardest hit, often lacking adequate infrastructure, service provision, and safe water (Heaney et al. 2011; Olmstead 2004; Pilley et al. 2009; Wilson et al. 2008a). In addition, despite wide coverage provided by community water systems, approximately 15% of the nation's population (~46 million) relies on water from unregulated private drinking water supplies[5] (e.g., private, household wells) (U.S. Environmental Protection Agency 2011c). In many ways, the situation for these residents is the most dire and least well understood, since the SDWA does not require private individuals to meet or monitor for compliance with federal drinking water standards.

Drinking Water Challenges in the U.S.

Certainly much is already known about key drinking water challenges in the U.S. By

5 In many ways, the situation for these residents is the most dire and least well understood, since the Safe Drinking Water Act does not require private individuals to meet or monitor for compliance with federal drinking water standards. While not the topic of this dissertation, it is still important to consider these individuals.

and large, these problems fall into one of three categories: water provision (infrastructure, pricing/financing, service provision); water safety (sources of contamination, waterborne disease, emerging diseases); and regulatory challenges.

Water Provision: Infrastructure and Financial Woes

Across the U.S., degraded infrastructure poses financial and health risks. Water lost to leaky infrastructure can represent significant costs to systems (Levin et al. 2002). Systems lacking adequate pressure and/or with leaky pipes may be at risk of cross-contamination. Back-siphonage of contaminants in these situations has caused waterborne disease outbreaks (Craun and Calderon 2006; Moore et al. 1993). The U.S. EPA estimates that the nation's water utilities must increase investments at least $151 billion over the next two decades to maintain public water infrastructure and to ensure safe and healthy community water supplies (U.S. Environmental Protection Agency 2001). Of this total, about $38 billion is for water treatment, $83 billion to repair and/or replace components of the distribution system, and $28 billion to protect watersheds and maintain storage reservoirs (Levin et al. 2002).

Even if limitless federal subsidies could fill these funding gaps, this would not address the real causes of inadequate system maintenance and ensure sustainability (Levin et al. 2002). As numerous authors have indicated, current institutional arrangements (e.g., fragmentation) that govern local public water providers and inadequate pricing and governance have led to low technical, managerial and financial capacity of water systems (Beecher 2003; Castillo et al. 1997; Ottem et al. 2003; Raucher et al. 2004). These challenges can lead to cyclical challenges in rural areas, where financial resources are more limited (Committee on Small Water Systems 1997). These areas tend to have higher unemployment and larger proportions of elderly populations, two factors that can further impair the ability of residents to afford system upgrades. While water rates in many water supply systems have been insufficient to cover long-run costs, residents in many of these communities are not be able to afford the increases. This creates a difficult conundrum—how to price adequately and equitably.

Water Safety: Sources of Contamination, Emerging Contaminants

Key sources of surface water contaminants include siltation, nutrients, pathogens, oxygen-depleting substances, metals, habitat alteration, pesticides, and organic toxic chemicals (Dubrovsky et al. 2010). These pollutants derive primarily from runoff related to human activities (Levin et al. 2002). Key sources of groundwater contamination include naturally occurring contaminants, such as arsenic (Welch et al. 2000), radioactive materials and other trace metals, as well as human-derived contaminants leaching into the groundwater (Storm 1994). Recharge of contaminated water into groundwater systems can also be a source of chemical and microbial contaminants (Levin et al. 2002).

Nationwide, agriculture is the most extensive source of water pollution, affecting 70% of impaired rivers and streams and 49% of impaired lake acres. Agriculture also impairs groundwater sources. Largely as a result of intensive agricultural practices, in California, nitrate is one of the leading groundwater contaminants. Nitrate is associated with numerous adverse health effects, from "blue baby syndrome" to reproductive effects in women (DeRoos et al. 2003; Fan and Steinberg 1996; Fan et al. 1987; Ward et al. 2005; Ward et al. 2010).

As a result of these contaminating activities, waterborne illness can still threaten the health of U.S. residents. Outbreaks of microbiological contaminants still make the headlines (i.e., *cryptosporidium* outbreaks in the 1990s in Milwaukee). While surface water supplies are the major risk for waterborne infectious disease, increasing evidence also shows that wells, especially relatively shallow wells, are also vulnerable to microbial contamination. The U.S. Centers for Disease Control and Prevention estimates that about half the documented waterborne disease outbreaks have a groundwater source (Barwick et al. 2000; Moore et al. 1993). Interestingly, because of poor data, the full extent of waterborne infectious diseases in the United States presently is not known and is thought to under-represent actual incidence (Barwick et al. 2000; Moore et al. 1993). For example, Levin (2002) notes that current incidence estimates are three to four orders of magnitude higher than the CDC data and that 6–40% of gastrointestinal illness in the United States may be water related.

In addition to microbial contamination, a number of health risks also occur from acute and chronic exposure to chemicals. For example, infants under six months are at risk of methemogloinemia ("blue baby syndrome") when ingesting water with high nitrate levels (i.e. > 45 mg NO3/L). At these levels, pregnant women are at risk of miscarriage and adverse reproductive effects, such as neural tube defects (Fan and Steinberg 1996; Ward et al. 2005). And newer evidence links chronic exposure to nitrate with various forms of cancer, most notably thyroid cancer (DeRoos et al. 2003; Ward et al. 2010). Arsenic, though largely naturally occurring in groundwater, is one of the most carcinogenic contaminants regulated by the Safe Drinking Water Act (Smith et al. 2002).

Emerging contaminants are also garnering increased attention. Among some of the most commonly cited are disinfection by-products, resulting from the chlorination of drinking water. The carcinogenic nature of disinfection by-products (Morris et al. 1992) has led some of these byproducts, such as trihalomethanes, to receive greater scrutiny and to be included for regulation under the SDWA. The presence and effects of pharmaceuticals in drinking water supplies have also garnered increased attention.

Regulatory Challenges of the Safe Drinking Water Act

Despite the passage of the SDWA, and the intention of its major amendments to address key concerns, a number of regulatory challenges still exist. To begin, the SDWA does not assure full provision of clean drinking water. To date, only 91 microbiological and chemical contaminants are regulated, though over two hundred more are present in drinking water (Environmental Working Group 2009). Even MCLs are not fully health protective since they are determined by considering technical and economic feasibility alongside health risks. The MCL for arsenic, for example is 10 parts per billion (ppb). But given that it is a known human carcinogen, the recommended public health level (RPHL) is zero, due to observed effects at lower levels, and the assumed dose-response model (National Research Council 2001; U.S. EPA 2010a). This RPHL, also referred to as the Public Health Goal or Maximum Contaminant Level Goals are non-enforceable, health-based goals that are set at a level at which no known or anticipated adverse human health effect occurs and allows for an adequate margin of safety, without regard to cost(American Water Works Association 1999)[6].

The contaminant-by-contaminant mode of regulation has additional limitations.

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[6] Whereas the MCL is based on a risk management approach, RPHLs or Maximum Contaminant Level Goals are based on a risk assessment approach.

Because each new standard is developed in isolation, adoption of new standards has been extremely slow (Roberson 2011). Furthermore, cumulative impacts—due to potentially additive or synergistic effects of multiple contaminants present in a drinking water source—have not been adequately considered. Further complicating this landscape is the fact that new chemicals continually enter the marketplace. In this process, whole classes of contaminants, including pesticides and pharmaceuticals, are largely unregulated and will likely remain so under the current regulatory paradigm. Partly as a result of these factors, in 2010 the U.S. EPA released a new approach to protecting drinking water and public health that included going beyond the traditional regulatory framework of addressing contaminants one at a time and look at regulating groups of chemicals (U.S. Environmental Protection Agency 2010a).

Even if existing standards were inclusive of all drinking water contaminants and reflective of the most precautionary levels, enforcement and implementation of the regulations by regulators and public water systems is far from perfect. In 2009, a series of articles in the New York Times exposed problems with monitoring, enforcement and general water quality (Duhigg 2009a, b, c, d, e). Past and current General Accounting Office (GAO) reports have highlighted similar problems (U.S. General Accounting Office 1990, 1999, 2011). In theory, water systems are required to provide water below the MCLs, or alert their customers when they exceed the limit (California Code of Regulations 2008e). But many water systems either fail to monitor water quality or to notify their customers of water quality levels (U.S. General Accounting Office 1990, 1999). Some studies explain this limited implementation of the SDWA as due to time and funding constraints (Scheberle 1997). But whatever the cause, the effect has been that large numbers of water systems fail to monitor, to report violations, or to notify their customers of unsafe water levels.

But an even broader challenge exists as well—weak institutions to adequately protect contamination of source water. For drinking water, this has largely to do with the fact that groundwater pollution is weakly regulated in the U.S. While the 1974 Clean Water Act was a landmark success in requiring protection of the nation's waters, the Act focused primarily on protecting the nation's surface waters. This may be due, in part, to the fact that the Act developed on the heels of the Cuyahoga River fires. In this case, the public and policy makers could certainly see surface water pollution more visibly. What's more, at the time, groundwater was still largely misunderstood, or simply not understood. Nash notes how little awareness there was of the connection between what was applied in the fields and what percolated into the ground (Nash 2006). The end result is that federal policies have been largely silent on the topic.

In the San Joaquin Valley, land subsidence due to excessive groundwater pumping may have been one of the first signals that something was going on underground (Galloway and Riley 2006), but even many regions in the U.S. lack comprehensive groundwater pumping monitoring plans. And, while the 1986 Amendments to the SDWA added a Wellhead Protection Plan to help protect groundwater supplies (F. Pontius 2003), these additions lack the regulatory teeth of the Clean Water Act. Similarly, while State and Regional Water Quality Control Boards are tasked with ensuring protection of groundwater quality, little enforceable regulations exist to ensure source water protection of the primary drinking water source for most Americans. This larger backdrop has important implications for drinking water quality. Rather than have the polluter pay for and protect against anthropogenic sources of drinking water contamination, individual communities must pay for cleaning contaminated water and ensuring protection of community health.

Water Supply and Other Challenges

Finally, water supply at both a regional and system level can create reliability problems. In many regions of the U.S., especially the arid West, water supply can be sporadic. Due to regional or local variation in the availability and amount of surface water supplies due to climate change, groundwater pumping by agriculture is expected to increase. This will create increased competition for groundwater supplies that are often used for drinking water purposes (Levin et al. 2002).

Drinking Water and Vulnerability: Water Policy Perspectives

Given the aforementioned challenges, the water policy literature generally discusses "vulnerability" to inadequate drinking water provision or unsafe water in two main ways. First, it focuses on how small water systems have the hardest time meeting and complying with water standards. At its most fundamental level, low economies of scale can compound these problems, because the per household cost of water supply can be significantly higher (Committee on Small Water Systems 1997). Even if loans for infrastructure improvements are obtained, for example, residents served by small water systems may pay more than three times the costs compared to those living in larger systems to finance construction of upgraded infrastructure or treatment plants. While some of these small water systems serve wealthy areas, such as unincorporated sub-urban subdivisions, many are in poorer regions, such as rural fringes. These systems have more difficulty raising the capital for infrastructure upgrades, or developing sustainable rates that result in adequate operating revenue. Lenders are often unwilling to give loans to these communities, if they perceive a small profit margin on the loan, or greater risk. With little revenue, there are less financial resources for water treatment and regulatory compliance (Committee on Small Water Systems 1997). More than their larger counterparts, these smaller systems lack technical, managerial and financial (TMF) capacity (Shanaghan and Bielanski 2003).

Service reliability can vary considerably in small systems, especially those in rural areas. By definition, these systems generally have fewer water uptake sites (e.g. wells or surface water intakes). In these systems, if one well breaks (as described by Luisa in Alpaugh) residents can lack of water. Seasonal variations can further impact water reliability (Committee on Small Water Systems 1997). And in systems with poor management, improper maintenance may occur. In the San Joaquin Valley, for example, some systems have failed to provide clean water simply because system operators failed to flush the system periodically (Community Water Center 2011). In small systems where treatment is unlikely and compliance difficult, these systems are vulnerable to outbreaks of waterborne illnesses and to serving harmful levels of contaminants. When these small systems are located in areas with contaminated sources (e.g. the San Joaquin Valley), the challenges are only multiplied, as the possibility of obtaining a clean alternative water source is more difficult.

Second, the water policy arena also considers vulnerability in relation to the vulnerability of subpopulations. This focus derives primarily from the 1996 amendments to the SDWA that noted that any proposed national primary drinking water standards should consider the effects of the contaminant on both the general population and sensitive subpopulations. Here, a sensitive population is defined as one that is at increased risk of an adverse health event or outcome after exposure (Griffiths 2003). Sensitive populations include babies, infants, pregnant women and the elderly. But in addition to these traditionally recognized groups, people with certain diseases, such as AIDS, cancer, or diabetes, may experience exacerbated effects from contaminant exposure.

Whether at the system or population-level, with certain caveats, additional aspects of vulnerability are rarely discussed. Socioeconomic characteristics of residents (e.g., race, poverty, etc) are only indirectly considered, when discussing the impact of resident household income on the ability of community water systems to obtain loans, or the ability of residents to afford rate increases (Beecher 2003; Committee on Small Water Systems 1997). And little mention is given to racial or ethnic disparities and how these may relate to system-level vulnerabilities or potential health risks. While the water policy arena's scale-oriented focus of the problem is important, it leaves much wanting. Are other sub-groups vulnerable? How vulnerable? Why? This dissertation aims to expand our notion of key vulnerabilities to consider with regard to drinking water provision.

Beyond Economies of Scale: Environmental Justice and Additional Dimensions of Vulnerability

The environmental justice (EJ) literature offers additional dimensions of vulnerability (e.g. race, class, distribution of harm and procedures), and additional approaches with which to measure other aspects of the drinking water burden, such as disparities and additional health risks and disparities.

As defined by the U.S. EPA, environmental justice is:

> *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. Fair treatment means that no group of people, including a racial, ethnic, or socioeconomic group, should bear a disproportionate share of the negative environmental consequences resulting from industrial, municipal, and commercial operations or the execution of federal, state, local, and tribal programs and policies* (U.S. EPA 2003).

While sometimes critiqued by activists for focusing on equity (i.e. distribution of the harm) rather than justice (i.e. no pollution should exist, broader focus on social justice) (Cole and Foster 2001), EPA's definition includes basic concepts of both distributional equity and of "fair treatment", or procedural justice. Distributional equity focuses on how an environmental good or harm is distributed. Procedural justice focuses on whether programs and policies are implemented fairly for different groups. In the context of drinking water, these concepts can help shape our understanding of geographic patterns of how contamination is distributed, what procedures may be driving these patterns, and disparities by race/class.

EJ activists have traditionally called for a more direct engagement with notions of environmental racism, and environmental justice per se. These, too, are relevant for drinking water considerations. The former concept refers to: "racial discrimination in environmental policy-making and enforcement of regulations and laws [and] the deliberate targeting of communities of color for toxic waste facilities…" (Chavis Jr 1994), thus emphasizing intentionality of harm. The latter speaks to EJ activists considering distributional equity as a crucial, but insufficient "entry way." In essence, both concepts emphasize the need to address the "social structure and institutional context in which environmental decisions are made" (Cole and Foster 2001) in order to attain a more holistic social justice agenda. For drinking water, these concepts are relevant as they beg a focus on the processes leading to contamination, not only on the distributional patterns.

But in general, there has been a paucity of EJ-focused drinking water studies. In 1993, Calderon et al conducted a seminal review examining the relationship between race, class and drinking water quality. Among other things, the study concluded that research endeavors should analyze: 1) existing datasets to assess exposure of populations to water contaminants, 2) the demographics of the populations exposed, 3) characteristics that affect exposure etc.

Since this study, researchers have slowly begun to focus on distributional and procedural aspects of drinking water. For the most part, distributional studies have focused on the association between community demographics and potential exposure to drinking water contaminants. For example, studies have examined the relationship between drinking water MCL violations and poverty and racial/ethnic minority status in San Joaquin County, California (Byrne 2003). Others have explored the extent of bacteriological and chemical contamination in unregulated drinking water sources in the Navajo Nation (Murphy et al. 2009), and the relationship between arsenic levels and community characteristics in Oregon (Stone et al. 2007).

A smaller body of studies has focused on procedural aspects of drinking water. Cory and Rahman (2009) assess the relationship between communities with high arsenic levels and demographics, but with a focus on whether there are implications for selective enforcement (i.e. unequal enforcement of laws). Though focused on the Clean Water Act, Imperial (1999) examines whether there is unequal access to Clean Water Act grants and funding mechanisms across counties. And, taking a more historical perspective, Troesken (2002) analyzes patterns of unequal drinking water provision in the rural south.

A growing number of geography and legal studies, often not labeled "environmental justice" studies, offer important insight into some of the processes underlying poor water quality and service provision. For example, some studies have addressed how U.S. municipalities have used their police powers (i.e. municipal legal authority) to provide or deny service to disadvantaged neighborhoods. Others have focused on historical patterns of unequal water service provision (Troesken 2002). More recently, researchers have explored how water service provision can be dictated by current discretionary planning (Marsh et al. 2010), historical exclusion of and deliberate growth around low-income and minority pockets (Anderson 2008; Lichter et al. 2007; Troesken 2002), and biased infrastructure decision-making at the county level (Anderson 2010) and city level (Heaney et al. 2011; Wilson et al. 2008a; Wilson et al. 2010). Anderson (2010) argues that part of the problem is that many unincorporated are "mapped out" of city boundaries; in appearing invisible, policy makers and regulators may have little information on conditions in these communities, and their needs go unmet.

Alongside grassroots efforts, these studies have helped highlight pockets of "hotspots", where drinking water provision is particularly poor. For example, along the U.S.-Mexico border, in states such as Texas and Arizona, exist dozens, if not hundreds, of settlements of predominantly Latino residents (Ward 1999). These colonias lack basic services, including drinking water (Olmstead 2004; Ramshaw 2011). In New Mexico, arsenic levels in similar communities exceed the MCL (Pilley et al. 2009). Areas of the rural south also lack basic water service provision, and have high rates of microbial contamination (Heaney et al. 2011). Similar problems exist in California's San Joaquin Valley, where PolicyLink, a California-based policy think tank estimates that several hundred unincorporated areas exist, and often lack paved streets, sewer provision or clean water (PolicyLink 2011).

Environmental Justice Meets Environmental Health and Social Epidemiology

Despite the strength of environmental justice studies in highlighting social disparities, it has been critiqued for not always connecting to health outcomes. Some authors ask—what are the implications of disproportionate hazards siting? Does proximity to siting confer increased vulnerability? At the root of these questions is a challenge to show the *impact* of procedural and distributional inequalities. Two public health fields—environmental health and social epidemiology—are particularly useful for addressing this challenge.

Environmental health helps assess health impacts more directly, mainly because of its focus on the relationship between environmental contamination and individual and community-level characteristics. Social epidemiology bolsters environmental health approaches by explicitly addressing the role of social vulnerabilities, and not just biological (e.g., age and genetic) susceptibility. Drawing on social-ecological models, the field recognizes the role of the social context in mediating and impacting health outcomes (deFur et al. 2007; Gee and Payne-Sturges 2004; Krieger 2001). In doing so, social epidemiology focuses not only on the role of individual-level characteristics, but community-level characteristics (e.g. neighborhood factors) in mediating community health. Thus, the field has focused increasingly on how multi-level factors may help explain observed disparities in health. Borrowing from deFur et al (2007), we can understand vulnerability as "how individuals or groups…respond to and recover from stressors inadequately or not as well as the average." De Fur et al argue that individuals or groups may be vulnerable as a result of their capacities, resources, coping mechanisms, supports and group characteristics (e.g. size and complexity). This dissertation, and *Chapter 5*, in particular, uses this concept of vulnerability to move beyond a solely scale-oriented explanation of drinking water challenges.

A final related focus related to social epidemiology is the field's growing focus on the role of the built environment in mediating exposure and disease. For example, Wilson et al (2008) argue that through the denial of basic amenities, such as sewer and water services, rural African American communities in North Carolina experience inequities in health risks. These authors subsequently argue for the importance of looking at differences in health-promoting (e.g. good water provision, parks) infrastructure, versus unhealthy infrastructure (e.g. degraded or non-existent water systems) (Wilson 2009; Wilson et al. 2008b). Thus *Chapter 5* engages with the multi-level nature of drinking water provision, and the need to address the built environment. In doing so, it begs a broader consideration of what types of interventions are necessary (beyond having a community pick up and move, for example).

The San Joaquin Valley: Motivation for the Study Area

This chapter ends by situating the aforementioned contexts and theories in relation to my study region, California's San Joaquin Valley. Stretching nearly seventy-five miles wide and four hundred and fifty miles long, the Central Valley spans nineteen counties, from Shasta County in the north to Kern County in the south. This Central Valley is often divided into two regions: the Sacramento Valley to the North, and the San Joaquin Valley to the South. These two valleys, defined by their major rivers, meet in the Bay-Delta, where the Sacramento and San Joaquin rivers converge.

The San Joaquin Valley (SJV) accounts for nearly two thirds of the entire Central valley and one fifth of the state of California. It is composed of eight counties: San Joaquin, Stanislaus, Merced, Madera, Fresno, Kings, Tulare and Kern counties (Figure 1.1). The 3.8 million inhabitants in the SJV (U.S. Census Bureau 2007b) account for roughly half the population of the entire Central Valley (~6.56 million inhabitants).

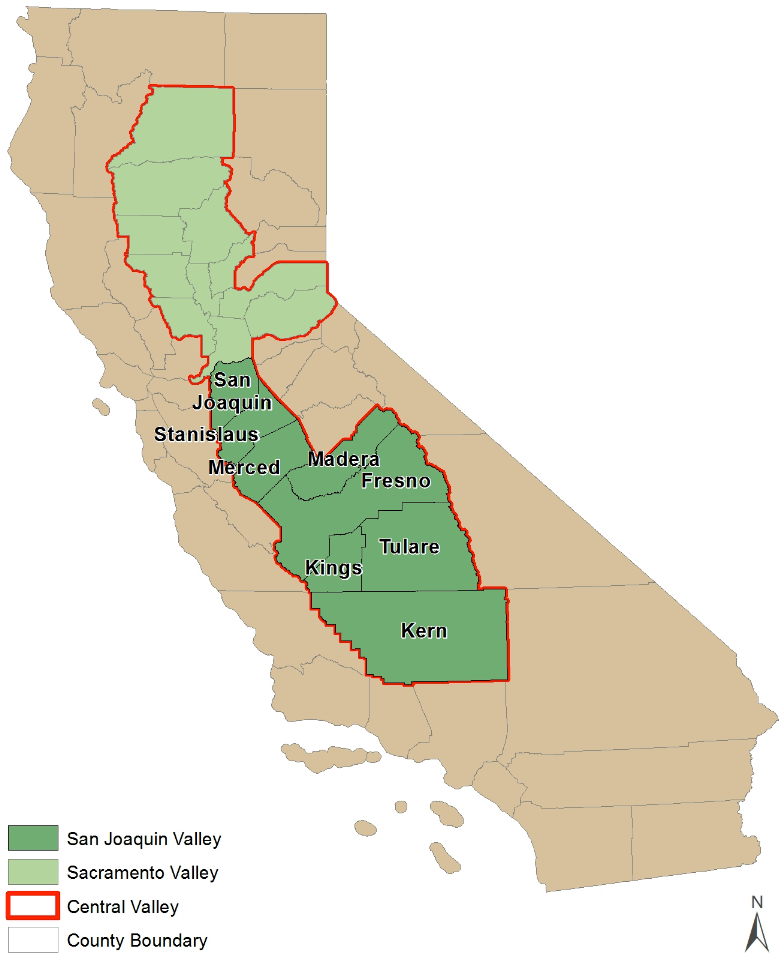

Figure 1.1. Map of the Central Valley. The Valley is comprised of the Sacramento and San Joaquin Valleys. Data source: California Department of Forestry and Fire Protection for County Boundaries.

To most, the Valley is a land of contrasts. On one hand it accounts for a vast majority of California's agricultural output; six of the eight counties rank among the top ten agricultural producers in the state (Table 1.1). As Walker (2004) notes, "In popular imagery, California's abundance is attributed directly to the munificence of nature: sunny Mediterranean summers, deep alluvial soils, and the Sierra snowpack." But related to this agricultural productivity are a host of acute environmental problems. For example, the SJV has some of the worst air quality in the US (Cowan 2005; Meng et al. 2010)). Not un-related, half of the counties in the SJV have asthma rates above the state mean (Millet et al. 2007). In 2000, eight of the eleven counties with the largest pesticide use in the state were in the Valley (Appendix A, Table A.7).

Water quality is equally impaired. The Valley has some of the most contaminated aquifers in the nation (Dubrovsky et al. 2010; Dubrovsky et al. 1998). And yet, as discussed previously, groundwater regulation has been nearly non-existent from a regulatory perspective. The Clean Water Act, for example, focuses primarily on protecting the nation's surface waters. The 1996 Amendments to the Safe Drinking Water Act acknowledged the need for source water protection (FW Pontius 2003) but is not mandated to protect groundwater, per se. Instead, the responsibility of providing clean water falls on the water system, not on outside polluters. In 2007, the Valley comprised 75%, 53% and 43% of all violations of the Safe Drinking Water Act, for nitrate, arsenic and total coliform, respectively, three of the top contaminants in the state and the region (Permits Inspections Compliance Monitoring and Enforcement (PICME) 2008).

But studies on the extent to which humans are impacted by water contamination have been somewhat limited. Dubrovsky et al (1998) note, for example, that the dire conditions of aquifers in the Valley have implications for human and environmental health. But not much more is said. A Department of Water Resources report addressed populations vulnerable to drinking water contamination, but it received little attention (Wilber 2003). Environmental health studies have primarily looked at exposure and specific health outcomes, such as between arsenic and cancer (Bates et al. 1995; Steinmaus et al. 2003), but have failed to look at regional trends.

My contribution in examining drinking water quality in the SJV starts here. Over 90% of the valley's population relies on groundwater deriving from these contaminated aquifers, but little research has focused on the health implications of this contamination. What are levels of key contaminants in drinking water? Which populations are most exposed? How may the built environment mediate exposure? All of these are questions relevant from an environmental health perspective that have been generally under-examined.

But equally relevant are environmental justice concerns. There is a general recognition that racial and ethnic minorities in the Valley face some of the worst living and health conditions, and are among the poorest (Table 1.2). For example, the Valley has one of the largest Latino populations in the state (40% in 2000, as per U.S. Census), though other racial or ethnic minorities include Hmong, African Americans, Assyrians, Thai, Vietnamese, Pakistanis, Laotian and Filipinos (see Table 1.3). High rates of Latino immigration present several issues, as these immigrants tend to: "be younger than the state average, have lower high school graduation rates, lack fluency in English, be disproportionately low-skilled, have higher birth rates and related family sizes, and higher rates of family poverty (Cowan 2005). In the context of disproportionate health risks and exposure, the aforementioned characteristics are of concern, not only because the low-income or communities of color are generally socially vulnerable, they may lack adequate health care with which to address or detect associated health effects.

The second reason an environmental justice perspective is relevant is given the condition of basic service provision in the Valley, and rural-urban divides. While the vast majority of the population lives in urban centers of Stockton, Fresno, Bakersfield and Visalia, the Valley is predominantly rural. Policy groups estimate that several hundred thousand people live in several hundred unincorporated areas, lacking basic municipal services such as sewers, paved roads and lighting, and relying on county governments (PolicyLink 2011; PolicyLink and California Rural Legal Assistance 2008). In many ways, these unincorporated areas go un-counted—a small fraction of these communities are counted as Census Designated Places, for example. Thus relatively little has been documented about the conditions in these places, including the quality of drinking water. And despite these relevant points, relatively few academic studies have addressed the intersection of drinking water quality, environmental health and environmental justice in the Valley, thus reflecting a general paucity of drinking water-environmental justice studies[7].

Finally, a focus on drinking water is relevant given increasing emphases on cumulative impacts of environmental contamination (Morello-Frosch et al. 2011; Sadd et al. 2011; Su et al. 2009). Numerous scholars working at the intersection of environmental health and environmental justice have begun to highlight the "triple jeopardy" that low-income and minority communities face. These studies argue that environmental health disparities may be, at least partially, understood as the interaction of disparities in exposure (i.e., distributional injustices), social vulnerability (e.g., poverty), and biological susceptibility (e.g., age, genetics) that act additively or synergistically (Gee and Payne-Sturges 2004; Morello-Frosch and Lopez 2006). In many instances drinking water contaminants are but one set of contaminants that residents may be exposed to (Morello-Frosch et al. 2011; Sadd et al. 2011; Su et al. 2009).

But while a drinking water-environmental justice focus is less present in the academic literature, social movements in the Valley have not ignored this link. Going back to the 1960s and 1970s, the labor and farm-workers movement aimed to improve living and working conditions for farm workers (Cole and Foster 2001). Drinking water quality can be seen as part of this struggle. In one community I attended, for example, a former labor organizer noted that Cesar Chavez would often tell people to not forget the water, as it carried many contaminants in it. Since the 2000s a thriving water justice movement has taken on this challenge more directly. In 2005, for example, the Environmental Justice Coalition for Water published a report that, among other things, examined the relationship between drinking water violations and county-level poverty rates(Environmental Justice Coalition for Water 2005). These grassroots groups have been responsible for having the United Nations (U.N.) Special Rapporteur on the Human Right to Water and Sanitation visit the Valley to document drinking water problems. These groups have also been at the helm of drafting human-right-water legislation (i.e., AB 685) for California.

[7] For a field of study that has developed alongside, and in relation to grassroots social movements, this is somewhat ironic, given that drinking water was at the core of the often-cited "birthplace" of the modern-day environmental justice movement. For example, when residents in Warren County, North Carolina protested the siting of a PCB-contaminated soil in the early 1980s, they were motivated by an underlying fear that this toxic would contaminate their drinking water sources. Perhaps because of the confluence of this nascent EJ movement with the anti-toxics movement, subsequent movement actions and research endeavors subsequently focused increasingly on the disproportionate siting of environmental hazards (United Church of Christ 1987). From here, academic studies began to focus mainly on air pollution (Axelrad et al 1999; Morello-Frosch and Lopez 2006; Morello-Frosch et al. 2011).

Table 1.1. Land-use and dairy statistics in San Joaquin Valley's Counties.						
	Approximate land area in acres (2002)	**Total cropland (2002), %**	**County Rank: Gross Value of Agricultural Production (Without Timber) (2006)*, in $1000**	**Number of Cows* (2006)**	**Number of Dairies (2006)***	**Average Number of Cows per Dairy (2006)***
Fresno	3,816,844	1,229,545 (32%)	1: 4,843,392	108,945	125	872
Kern	5,210,217	998,297 (19%)	4: 3,476,801	153,546	55	2,792
Kings	890,236	499,919 (56%)	11: 1,289,186	165,316	160	1,033
Madera	1,366,951	362,065 (26%)	13: 1,032,500	67,900	55	1,235
Merced	1,234,364	593,347 (48%)	5: 2,284,457	243,762	305	799
San Joaquin	895,540	574,552 (64%)	7: 1,684,871	103,480	136	761
Stanislaus	956,026	408,248 (42%)	6: 2,148,152	181,189	287	631
Tulare	3,087,340	770,484 (25%)	2: 3,870,843	466,592	341	1,368
Data source: *= (California Department of Food and Agriculture 2007); without *= (U.S. Department of Agriculture 2002)						

Table 1.2. Income and poverty statistics, San Joaquin Valley, 2007.			
	Total Population (2007) *	**Median Household Income (2007) in dollars+**	**% Below Poverty Level (2007)+**
California	36,553,215	$59,928	12.4
San Joaquin Valley	3,834,766	$46,470	17.8
Fresno	899,348	$46,547	20.0
Kern	790,710	$46,639	18.1
Kings	148,875	$45,087	17.2
Madera	146,513	$44,259	16.9
Merced	245,514	$46,544	19.3
San Joaquin	670,990	$51,874	14.2
Stanislaus	511,263	$50,367	13.6
Tulare	421,553	$40,444	23.2
Data source: * = (U.S. Census Bureau 2007b); += (U.S. Census Bureau 2007c)			

Table 1.3. Demographics by race and ethnicity, San Joaquin Valley, 2007.								
	Hispanic or Latino	*Not Hispanic or Latino*						
	% Hispanic or Latino (of any race)	% Non-Hispanic White	% Black or African American alone	% American Indian and Alaska Native Alone	% Asian alone	% Native Hawaiian and Other Pacific Islander alone	% Some other race alone	% Two or more races
California	35.7	43	6.1	0.5	12	0.3	0.4	2
San Joaquin Valley	46.4	40.5	4.55	.7	5.7	.21	.24	1.7
Fresno	47.6	36.4	4.9	0.6	8.5	0.1	0.2	1.6
Kern	45.1	42.8	5.6	0.6	3.7	0.1	0.3	1.8
Kings	47.4	38.8	7.5	0.9	3.4	0.1	0.1	1.7
Madera	49.3	41.8	3.8	1.1	2	0.1	0.2	1.8
Merced	51.7	35.5	3.5	0.5	6.5	0.3	0.5	1.4
San Joaquin	35.7	40.2	7	0.6	13.3	0.4	0.2	2.5
Stanislaus	38.2	50.9	2.7	0.6	5	0.5	0.3	1.9
Tulare	55.9	37.3	1.4	0.7	3.2	0.1	0.1	1.2
Data source: (U.S. Census Bureau 2007a)								

CHAPTER 2
Methods

Overview

This chapter provides an overview of the main methods used in this dissertation. While *Chapters 3-4* describe specific methods used, this chapter serves as a general map and defines key terms and approaches that are then explored in more depth in subsequent chapters. I used two main methodological approaches. The first was a quantitative analysis of water quality in the San Joaquin Valley. The second was a qualitative, predominantly ethnographic, approach to understanding broader burdens of drinking water.

Approach and Methods

Quantitative Analyses

Units of Analysis

The primary units of observation in this dissertation are community water systems (CWSs). CWSs are one of several categories of public water systems (PWSs) regulated under the SDWA (Figure 2.1). CWSs serve water year-round to at least 15 units or 25 people and include municipal systems, apartment complexes and mobile home parks. In contrast, *non-transient non-community* water systems serve at least 25 people who do not live at the location but who use the water for more than 6 months each year. These systems include factories, schools and office complexes. *Transient non-community* public water systems have at least 25 people per day and use water for short periods of time. These systems include restaurants and hotels (U.S. EPA 2010b). Because I seek to characterize the reality of drinking water faced by local communities throughout the Valley, this dissertation focuses on CWSs as the primary unit of analysis.

While the state retains primary legal responsibility, or "primacy", over all PWSs, most counties in California have opted to have primacy over smaller water systems, defined as those with less than 200 service connections[8]. In the San Joaquin Valley, only Kern and Fresno counties relinquished their primacy back to the state Department of Public Health. Kern did so starting in 1993, and Fresno starting in 2007 (as discussed in *Chapter 5*).
The majority of CWSs in the Valley (72%) serve fewer than 500 people, though the majority of the population is served by a handful of large systems, such as the cities of Fresno, Stockton and Bakersfield (CDPH 2008a). Three main ownership models exist—privately owned and regulated by the Public Utility Commission (6%), privately owned and not regulated by the Public Utility Commission (60%), and publicly owned (32%). A small fraction of systems are mixed private-public systems. For additional information on the Valley's water systems, see Appendix A.

8 Whether a system has fewer than 200 service connections serves as one threshold for defining small systems. Other commonly used thresholds consider CWSs serving fewer than 3,330 people as small; others consider those serving fewer than 500 people as small.

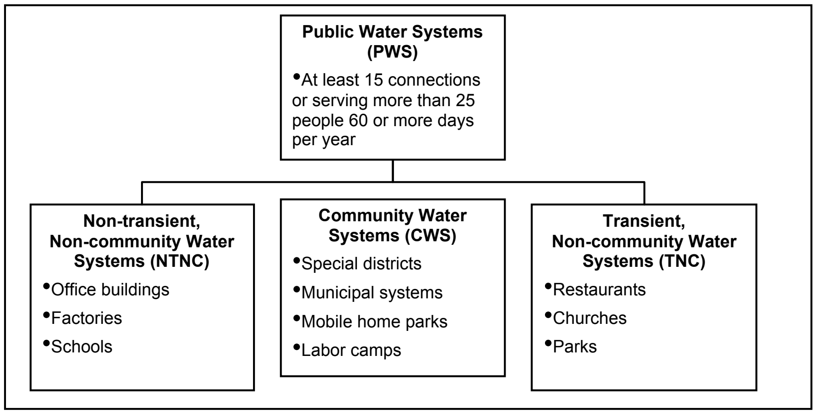

Figure 2.1. Breakdown of Public Water Systems. Adapted from the American Water Works Association (2003).

Primary Datasets and Sampling Frame

Two primary datasets served as the basis for most water-quality related estimates. The first is the California Department of Public Health's (CDPH) Water Quality Monitoring (WQM) database. This database houses historical water sampling data for all public water systems in the state. The second is CDPH's Permits, Inspections, Compliance, Monitoring and Enforcement (PICME) database. PICME contains ongoing records of all public water system's service characteristics (e.g., number of connections, population served) and violation histories. While regulators track adherence with the SDWA, regulators use PICME to record violations and compliance with them. Included in PICME are a range of violations, their dates and follow up actions requested.

I requested the WQM and PICME datasets from CDPH twice—once in 2007 and once in 2008 (to further update the data). To use and obtain the geographic coordinates for water systems, I was required to file a confidentiality agreement in order to use and map geographic locations of all public water systems in the state.

A list of all CWSs appearing in PICME served as the primary sampling frame from which to select CWSs in my quantitative analyses. Depending on the time period of interest (1999-2001 for nitrate, 2005-2007 for nitrate), from this full list, I selected CWSs that were actively in operation during each specified time period (See Appendix A, Table A.1 for a comparison of inactivated systems). Using Stata v10 (College Station, Texas), a statistical and data management software system, I selected only CWSs and cleaned the data for this attribute. For example, in some cases a hotel or restaurant was included as a CWS, when in fact this was a misclassification. After this cleaning, I further dropped two main types of CWSs—prisons and year-round youth facilities. While both of these are technically included as CWSs, for this research I wanted to focus on the non-institutionalized, residential population living in CWSs. Depending on the specific study focus (i.e., nitrate or arsenic), I then selected CWSs that had sampling or violation data for the contaminant of interest. These later steps are described in *Chapters 3* and *4*.

Contaminant Selection and Sampling Points

I focused primarily on arsenic and nitrate (as milligrams per liter of nitrate ion, NO_3) contamination, and to a lesser degree on dibromochloropropane (DBCP) and total coliform. Contaminants were selected on the basis of: 1) regulator and community input, 2) whether they are recorded in WQM and PICME, and 3) most-common contaminants. To begin, I met informally with regional and district engineers in the San Joaquin Valley to ask them what the key drinking water contaminants were. I also asked this of my key community partner (i.e. the Community Water Center). After generating this list, I then checked whether WQM and PICME had data on these contaminants.

Each contaminant was also selected based on its prevalence and toxicity. Nitrate, for example, is among the top contaminants in the state. Deriving largely from agricultural inputs (Dubrovsky et al. 2010; Dubrovsky et al. 1998), the Central Valley as a whole has some of this highest nitrate levels in the state. Arsenic, though naturally occurring is one of the most carcinogenic contaminants regulated by the SDWA (Smith et al. 2002), and is found throughout the Valley. Arsenic is also particularly interesting to track since after the revision of the Arsenic Rule in 2001, many more water systems were in violation due to the stricter standard (see *Chapter 4*). Additional contaminants of concern listed by regulators and drinking water experts, but not included in the study were: trihalomethanes, perchlorate and radon. These were not included because there were not enough data points on them in the 2008 WQM database.

Estimating Exposure and Compliance

To estimate exposure to drinking water contaminants, I used the WQM database. While all PWS are supposed to send in their water quality sampling results electronically (at least since 2007), many counties retain hard copies or electronic copies, and do not send in the sampling results due to database incompatibilities. This is a key source of measurement bias since data from water systems regulated by counties could appear less frequently in WQM. However, my statistical methods attempt to account for this bias, as described in *Chapters 3-4*.

Sampling points from point-of-entry sources for each CWS in WQM were used to estimate exposure. The details of this approach are described in *Chapters 3-4*. This focus on point-of-entry sources follows from the literature. For example, in a study of DBCP and nitrate in Fresno County, Whorton et al (1988) use water quality samples at point of entry to distribution system in their analysis of contaminant levels. Similarly, Cory and Rahman (2009) also use point-of-entry based samples. I further corroborated the use of these published approaches by discussing my approach with CDPH regulatory officials and maintainers of this database. These experts also agreed that the best sampling point that would represent the distribution system is at "point of entry" to the system, after the source has been treated (if there was treatment). However, two additional problems remained. First, regulators noted there would be inaccurate estimates of distribution water quality since larger systems can turn their sources on and off depending on water quality and supply needs. Secondly, I could not weight by production/flow of the source because this data on production was not available. Thus, I was forced to assume sources contribute equal volumes to the distribution system. Both limitations are addressed in *Chapter 3*. In a best-case scenario, this approach is somewhat more accurate for smaller water systems that only have a few wells. Steps for the selection and coding of point-of-entry sources are described in Appendix A.2.

Compliance data was derived from the PICME database. Data quality is again of concern with PICME. Major differences have existed between how the state and county-level regulators report violations. In some cases, for example, only one violation is noted for nitrate in a year, even if the water system was in violations in all four quarters. This is a topic discussed in greater depth in *Chapter 5*, and dealt with methodologically in *Chapter 4*. Table 2.1 provides a summary of the key variables or outcomes used in various parts of the research.

Estimating Community Demographics

While *Chapters 3-4* describe in detail how community-level demographics were estimated, this section gives a general overview. In essence, I followed four main steps of data collection and analysis. In Step 1, I digitized maps of water system service areas for Tulare and Fresno counties from hard copies, or by compiling existing Geographic Information System (GIS) data. To obtain these maps, I collected (with help of research assistants) all existing digitized boundaries from county planning and health departments. When these maps were not available, we called all remaining utilities and attempted to obtain hard copies. When this method did not work, we went to the county health department to obtain copies of water system files. In most cases, these files ended up having almost all water system boundaries. Having learned this latter point for Tulare County, for Fresno County I began with these water system files, and later sough to attain digitized GIS boundaries.

In Step 2, I used GIS to overlay these boundaries on top of U.S. Census blocks and block groups to estimate key demographic variables (e.g. income, race/ethnicity, etc). I used an aerial weighting technique to estimate population counts. This data was then used to compute the percent of each variable of interest. In Step 3, I used a separate layer of well and surface intakes to also estimate water system demographics. In Step 4, I compared the estimates from the two different approaches, using a regression analysis. Results of the aerial weighting and comparison of the two approaches are presented in Appendix B.1. Ultimately, the comparison of approaches indicated that I could proceed with the well-based estimation procedure.

Quantitative Analyses

In order to test the hypothesis that systems with lower rates of homeownership, and/or higher fractions of minority residents had higher levels of contaminants or violated the MCL more often, I used two main statistical regression models. The final model used to examine the relationship between demographics and concentration of contaminants is a linear regression model that reported robust standard errors clustered at the CWS-level (see *Chapters 3-4* for more details). Prior to selecting this as a final model, I had used a multi-level model to account for the fact that water system samples were clustered within point-of-entry sources, which were clustered within water systems. Results of this model are included in Appendix B.5. Ultimately, because I wanted consistent inference (unbiased standard errors) without model assumptions, my final models used the sandwich estimator of the standard error, allowing for correlation at the system level. Thus the final models presented in

Chapters 3-4 cluster at the system level, and provide robust inference under a more general model.

For analyzing compliance, I initially aimed to use a logistic regression model to analyze whether race/ethnicity and SES were associated with whether or not a system had a violation. But because of the small number of outcomes, the data would not support the estimation of a multivariate model, so I only examined bivariate associations, with p-values provided by nonparametric exact tests (Fisher's Exact Tests). All statistical analyses were conducted in Stata, and then integrated into GIS using ArcGIS software. Additional details are further described in *Chapters 3-4*.

Qualitative Methods

I used a number of qualitative methods to collect data on regulator, community and advocacy perspectives on drinking water problems in the Valley. These methods served as a means of understanding and validating my quantitative results. But more importantly, they allowed me to develop a broader analysis of water quality burdens in the Valley. Borrowing from the work of Pulido (1996), I worked from the premise that statistical analyses alone might provide an important, but incomplete picture of water quality burdens. Thus my qualitative methods were meant to gather additional information on patterns and problems driving potential drinking water contamination in the Valley.

Employing a loose community based participatory research model, I partnered with the Community Water Center for much of this research. Early on, the Community Water Center had come to me with questions about what the county-level trends looked like in terms of drinking water quality. Based on these initial conversations, we established a participatory model of research in which I periodically met with them to review the study questions and preliminary results, and they offered input and feedback.

This partnership became the vehicle through which I experienced the Valley and learned about impacted communities. Over the course of five years, I attended dozens of community meetings in places like Alpaugh, Seville, Tooleville and Plainview. At times I was asked to run analyses on water pricing or provide data on community characteristics (from data I had gathered). I went to press gatherings, and legislative hearings, and also attended the Tulare County Water Commission meetings. These meetings and participant observation served as an ethnographic base from which to observe the problems at hand, and understand the various viewpoints and stakeholders involved. Perhaps most importantly, while staying in the Valley, I participated in, or observed, Community Water Center staff and community meetings, gaining a deeper appreciation of their approach and viewpoints. This ethnographic process formed the core of my qualitative data collection experience.

In addition to this primary ethnographic approach, I also conducted informal and formal interviews with ten Valley-based non-profits, and approximately ten county and state regulators of the SDWA. After obtaining Institutional Review Board approval, I conducted interviews with non-profit staff from the Center on Race, Poverty and the Environment, the California Rural Legal Assistance Foundation, Self Help Enterprises, PolicyLink and the Clean Water Fund. These non-profit organizations were selected because of their involvement in community organizing, advocacy, policy or legislative involvement in drinking water issues in the Valley. Regulator interviewees included CDPH Regional and District Engineers that oversee implementation and compliance with the SDWA in the San

Table 2.1. Data and variable descriptions used in dissertation.

Key Variables	**Type of variable**	**Data Processing/Aggregation**	**Key Assumptions**	**Original Data Source**
Population	Count	Derived directly from datasets.	When PICME population not available, I used WQM data. Regulators continually update these numbers. Therefore, it is difficult to know the year for which population count is truly valid.	PICME & WQM databases
Connections	Binary (<200 connections, >199 Connections)	Turned continuous variable into binary variable	Same as above.	PICME & WQM
Source of Water	Categorical (Groundwater, surface water, or a combination of both)	The source of water for each *source* with a sample (i.e. *prim_sta_c*) was defined. Then, each water system was assessed for whether it had any GW source, or any SW source. The final code reflects whether the system as a whole had any of the two types of water sources.	Water system characterized on the basis of the water type for all its different sources.	WQM
Year inactivated	Year	Derived directly from datasets.		PICME
Nitrate concentration	Continuous	Sampling points for point-of-entry sources used to represent distribution quality	Assumes *flowpath* variables allow accurate depiction of point-of-entry sources. See *Chapters 3-4* for more details.	WQM
Arsenic concentration	Continuous	Same as above	See *Chapter 4* for more details.	WQM
Arsenic violations	Binary	Used two methods: 1) PICME violations, 2) whether source or system average exceeded MCL. See *Chapter* 3 for details.	See *Chapter 4* for more details.	PICME & WQM
Ownership type	Categorical (Public, private and PUC regulated, private non-PUC regulated, unknown)	Used WQM data in conjunction with list of CA PUC regulated systems, from PUC website.		WQM
Type of Water System	Categorical	Used system name and ownership type to categorize systems.	System name serves as key identifier. Not all labor camps are CWSs, so this category is likely an undercount	WQM
Regional location: Valley, Foothills, Mountains	Categorical	Used GIS contour lines to differentiate Valley, from Mountains and Valley floor.	Does not capture key hydrogeologic differences within each region.	GIS
% Home ownership	Percent	See *Chapters 3-4* for details on how this was estimated for each CWS.	Assumes 2000 data relevant for study period (see Chapters for more details).	U.S. Census 2000; block group level
% People of Color	Percent	Same as above	Same as above	U.S. Census 2000

Joaquin Valley, as well as county health regulators in charge of smaller water systems in their respective counties. Interview guides are included in Appendix A.3. Finally, I also conducted media analyses for select communities, as discussed in *Chapter 5*. I used a combination of field notes, excel analysis, and content analysis to examine qualitative data.

A Pictorial Tour of Select Communities

This final section provides some images of communities in the Valley where I worked, spent time, or focus on in subsequent chapters. The key communities include: Lanare, Alpaugh, Tooleville and Seville. Figures 2.2-2.3 depict some of the homes in the unincorporated community of Lanare, located in Fresno County. Drawing its name from the land speculator L.A. Nares, Lanare was established in the early 1900s[9]. From 1912 to 1925 the town had a post office and a railroad stop. Residents drew water from the Kings River, until its water was diverted largely for agriculture (David Bacon 2011). In 1970, Lanare Water Services was established and the first water well was approved (Nolen 2006). In the 1950s the town's population dwindled, but over the last few decades it began to increase again as working families have found it cheaper to live in than nearby Fresno. In 2010, Lanare had approximately 589 individuals (and approximately 140 households), of which nearly 88% were Latino. In 2009, the median household income was $36,806. (U.S. Census Bureau 2007a) In 2000, the median household income was $26,375 (U.S. Census Bureau 2000c). The low-income status of community residents and the management of the town's water utility is a topic of discussion in *Chapters 4* and *5*.

Figure 2.2. Image of house in Lanare, CA.
Photo credit: David Bacon (D. Bacon 2011)

Figure 2.3. Image of house in Lanare, CA.
Photo credit: David Bacon (D. Bacon 2011)

Figures 2.4-2.7 provide a glimpse of water-related aspects of Alpaugh, an unincorporated community in Tulare County. Figure 2.4 depicts a sign that welcomes passersby to Alpaugh, a town that in 2000 had roughly 761 residents, a median household income of $23,688 and approximately 66% owner-occupied housing units (U.S. Census Bureau 2000a). In the background an irrigation canal supplies water to nearby farms, but local residents rely on groundwater for their drinking water supply. That farmers have access to surface water that comes from Sierra snowmelt and/or the California Delta has mainly to do with the historical allocation of surface water rights and intense lobbying

[9] Though by other accounts, this occurred in the 1930s.

efforts to use these waters for irrigation purposes[10]. As a result, nearly 95% of residents in the Valley rely on groundwater for drinking water supplies, though approximately 15% of drinking water systems purchase and/or treat surface water from irrigation districts (Permits Inspections Compliance Monitoring and Enforcement (PICME) 2008). As discussed in *Chapters* 4-5, this water exceeds the arsenic MCL. In the 2000s, when attempting to fix a defunct well and obtain a cleaner water source (with lower arsenic levels) Alpaugh's two drinking water authorities—the Alpaugh Irrigation District and Tulare County Water Works District 1 were mandated to form a joint powers authority, Alpaugh Joint Powers Authority in order to receive state funding for drinking water improvements. Prior to this the two entities and the customers (residents and agriculture for AID) were often at odds with each other over drinking water supply and water rates. Ironically, the buildings housing the two former water districts two buildings, depicted in Figures 2.5-2.6, are located just a few dozen meters apart from each other, but functioned as separate entities until the Alpaugh Joint Powers Authority was formed. When unsafe arsenic levels and a broken main well left the community without a constant water supply, community residents and local leaders participating in the Committee for a Better Alpaugh (Figure 2.7) helped to attract statewide attention and ultimately garner state funding and interim drinking water supplies.

Figure 2.4. Entrance to Alpaugh. Photo credit: Carolina Balazs

Figure 2.5. The Alpaugh Irrigation District (AID) office, in Alpaugh, CA. Photo credit: Carolina Balazs.


Figure 2.6. Former Tulare County Water Works District 1 office. Today this building houses the Alpaugh Joint Powers Authority. Photo credit: Carolina Balazs.


Figure 2.7. Community meeting. A few of Alpaugh's residents involved in Committee for a Better Alpaugh and leader from the Center on Race, Poverty and the Environment. Photo credit: Carolina Balazs.

10 For a more detailed account of these historical factors see Reisner's Cadillac Desert and Arax and Wartzman's The King of California.

Figure 2.8. Drinking water well in Alpaugh.
Photo credit: Carolina Balazs

On Tulare County's east side, lies another unincorporated community with drinking water contamination, Tooleville. This community of approximately 300 residents is composed primarily of Latino families. As discussed in *Chapter 5,* due in large part to a shallower water table and nitrate-intensive citrus farming that occurs on the east side, Tooleville's drinking water has persistently exceeded the nitrate MCL. At local meetings, residents often wonder why they can't use the Friant-Kern Canal (Figure 2.9) as their primary drinking water source. This canal runs right behind their houses (Figure 2.10), but is used primarily for irrigation.

Figure 2.9. The Friant-Kern canal. To the left are orange groves. A few miles north is the community of Tooleville. Surface water goes primarily to farmers, and is used for drinking water in only a few select communities. Photo credit: Carolina Balazs

Figure 2.10. View from one of Tooleville's streets. At the end of the road, behind the levee lies the Friant-Kern canal.Photo credit: David Bacon (D. Bacon 2011)

A final piece of the pictorial tour provides glimpses of how local Valley residents have organized to voice their concerns over contaminated drinking water. In the community of Seville, for example, community leaders have toured (Figure 2.11), among others, the U.N. Special Rapporteur for the Human Right to Safe Drinking Water and Sanitation and even received honors from the U.S. Environmental Protection Agency (U.S. Environmental Protection Agency 2010b). Members of A.G.U.A. (the Association of People United for Water) drive up to Sacramento to provide community-perspectives and place community demands on addressing drinking water provision and contamination (Figure 2.12).

Figure 2.11. Seville community leader (left) giving community tour. Photo credit: Carolina Balazs

Figure 2.12. Valley residents attend a Regional Water Quality Control Board hearing in Sacramento. Photo credit: Carolina Balazs

CHAPTER 3
Social Disparities in Nitrate Contaminated Drinking Water in California's San Joaquin Valley

Overview

As discussed in *Chapters 1-2*, research on drinking water in the U.S. has rarely examined disproportionate exposures to contaminants faced by low income and minority communities. This chapter analyzes the relationship between nitrate concentrations in community water systems (CWSs) and the racial/ethnic and socioeconomic characteristics of customers. We hypothesized that CWS in California's San Joaquin Valley that serve a higher proportion of minority and/or lower socioeconomic status (SES) residents have higher nitrate levels, and that these disparities are greater among smaller drinking water systems. We used water quality monitoring datasets (1999-2001) to estimate nitrate levels in CWSs, and source location and Census block group data to estimate customer demographics. Our linear regression model included 327 CWSs and reported robust standard errors clustered at the CWS-level. Our adjusted model controlled for demographics and water system characteristics, and stratified by CWS size. Percent Latino was associated with a .04 mg NO_3/L increase in a CWS's estimated nitrate ion concentration (95% Confidence Interval (CI), -.08, .16) and rate of home ownership was associated with a .16 mg NO_3/L decrease (95% CI, -.32, .002). Among smaller systems, percent Latino and percent homeownership were associated with an estimated increase of .44 mg NO_3/L (95% CI, .03, .84) and a decrease of .15 mg NO_3/L (95% CI, -.64, .33), respectively. Our findings suggest that in smaller water systems, CWSs serving larger fractions of Latinos and renters receive drinking water with higher nitrate levels. This suggests an environmental inequity in drinking water quality.

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Introduction

An array of drinking water-related problems still exists in the U.S, despite a history of investment in sophisticated water infrastructure and the existence of federal laws such as the Clean Water Act and Safe Drinking Water Act (SDWA) that regulate source contamination and protect the public's health (CWA 1972; SDWA 1974). These problems range from increasing source contamination (Dubrovsky et al. 2010), exposure to chemical and microbial contaminants, poor implementation of water laws (Burke 2009; Duhigg 2009c) and degrading infrastructure (Levin et al. 2002). Rural areas often face the largest burden, as aquifers are contaminated from intensive agriculture and livestock production (Dubrovsky et al. 1998). Some rural unincorporated areas, such as some communities along the U.S.-Mexico border, lack access to adequate infrastructure, service provision, and clean water (Olmstead 2004; Pilley et al. 2009).

Despite these problems, there is a paucity of studies that examine social disparities in exposure to unsafe water. A literature review in the 1990s (Calderon et al. 1993) recommended that more quantitative analyses examine whether vulnerable populations, including people of color and the poor, are disproportionately impacted by drinking water contamination. Since then, a handful of studies have addressed different aspects of this issue. In San Joaquin County, California, one study found a weak but significant relationship between areas with higher poverty and greater proportions of minorities and poor drinking water quality (Byrne 2003). Research in the Navajo Nation found bacteriological and chemical contamination in unregulated drinking water sources (Murphy et al. 2009). In Arizona, researchers examined whether public water systems serving higher fractions of minority and low socioeconomic status (SES) residents were more likely to exceed the arsenic maximum contaminant level (MCL) as compared to those in less minority and higher income areas. They found a positive association between percent Latino and likelihood of an exceedance. However, they concluded that there was not an environmental justice issue because there was no difference between the percentage of Latinos in water systems with and without violations (Cory and Rahman 2009). In New Mexico, preliminary research documented high arsenic levels in drinking water sources serving predominantly Latino border communities known as colonias (Pilley et al. 2009).

Our research addresses several methodological limitations of previous studies, particularly with regards to appropriate unit of analysis, characterization of exposure and scale. For example, Byrne (2003) estimated average trichloroethylene levels and MCL exceedances in drinking water systems, and characterized exposure as a continuous measure across San Joaquin County; the community level, however, is more appropriate when considering community-level exposure. Cory and Rahman (2009) characterized the association between percent minority and a binary measure of arsenic exceedances, rather than of arsenic levels as a continuous variable. They also did not explore this association among smaller systems, where they noted that the majority of arsenic violations occurred.

Our study used the community as its unit of analysis to examine the relationship between nitrate concentration in community water systems (CWS) and social factors. CWS are public water systems that serve water year-round to at least 25 people or have more than 15 service connections (U.S. EPA 2010b). We characterized potential exposure to nitrate because it is one of the most common contaminants found in groundwater (Harter 2009; Spalding and Exner 1993), yet has received little attention with regard to social disparities in exposure.

Nitrate in drinking water is associated with methemoglobinemia (i.e. "blue baby syndrome") in infants (Fan and Steinberg 1996; U.S. EPA 2010a), though other risk factors include enteric infections (Charamandari et al. 2001; Fan and Steinberg 1996; Fan et al. 1987; Hanukoglu and Danon 1996; Sanchez-Echaniz et al. 2001; Ward et al. 2005; Ward et al. 2010) and foods high in nitrates (Sanchez-Echaniz et al. 2001). Epidemiologic data also suggest an association between nitrate levels in drinking water, reproductive toxicity, developmental effects, and various cancers (Ward et al. 2005; Ward et al. 2010), though the consistency of these associations varies. To protect against methemoglobinemia, the SDWA has established a maximum contaminant level (MCL) of 45 milligrams per liter (mg/L) as nitrate ion (NO3), or 10 mg/L as nitrate-nitrogen in drinking water (U.S. EPA 2010b).

California's San Joaquin Valley ("the Valley") is an important site for examining potential disparities in exposure to nitrate. With its intensive irrigated agriculture, the Valley has two of the most contaminated aquifers in the nation and some of the highest nitrate levels in the country (Dubrovsky et al. 2010; Dubrovsky et al. 1998). Because nearly 95 percent of the Valley's population relies on groundwater for drinking (Permits Inspections Compliance Monitoring and Enforcement (PICME) 2008) groundwater contamination is a particular health risk. This risk is compounded by the fact that with high costs of mitigation, few systems actually treat for nitrate. The Valley also has some of the highest rates of poverty and minority populations—particularly Latinos—in the state (U.S. Census Bureau 2007d). These communities are economically and socially disadvantaged, making it harder for them to afford mitigation, or address related health consequences of nitrate contamination. The continued use of nitrogen-based fertilizers (Dubrovsky et al. 2010; Ruddy et al. 2006) and the increasing demand for groundwater (Glover 2010) further highlight the importance of this contaminant, as exposure may become increasingly widespread.

Given this context, we used water quality monitoring data from the California Department of Public Health (CDPH) to analyze the association between racial/ethnic and socioeconomic status (SES) characteristics of people served by CWS and nitrate levels of these systems in the San Joaquin Valley. With few exceptions (Byrne 2003; Wilber 2003), there has been limited use of CDPH monitoring data to examine whether certain groups are disproportionately affected by exposure to drinking water contaminants. Similarly, despite an acknowledgment of the burden faced by small systems (Committee on Small Water Systems 1997), few studies have explored associated social disparities.

We hypothesized that CWS serving a higher proportion of minority or lower SES residents have higher nitrate levels and that these disparities are likely to be greater among smaller drinking water systems. Disparities in nitrate exposures, if they exist, could signal a potential environmental injustice. This analysis expands the emerging literature on drinking water quality and social disparities in the United States, and informs national and state level policy on the needs of under-resourced water systems.

Materials and Methods

Our units of observation were CWS in the Valley. We used three measures to test our study's hypotheses: 1) estimated average nitrate concentrations for each CWS to describe average water quality served to customers, 2) population potentially exposed (PEP) to three nitrate levels to estimate the population impacted by nitrate contamination, and 3) nitrate concentrations at points-of-entry into each CWS' distribution system to assess the relationship between demographic characteristics of customers and CWS nitrate levels. The first two measures were used in a series of descriptive statistical analyses. The third measure was used as the outcome variable in linear regression models that estimated the relationship between race/ethnicity, SES, and a system's nitrate concentration.

Sample Selection and Time Period

We included CWS that were active in the San Joaquin Valley between 1999 and 2001, had at least one point-of-entry source with a nitrate sample reported for this period, and had any source (i.e. point-of-entry or not) with geographic coordinate data available to estimate CWS demographics. *Point-of-entry sources* can be defined as sources of supply (e.g. well with no treatment or effluent from a well/surface water plant) that directly enter into the distribution system (Figure 3.1). We used nitrate-sampling data from 1999 to 2001 and demographic data from the 2000 Census. The sampling period represents one full compliance period under the SDWA (California Code of Regulations 2008c). Of the 873

CWS that were active during 1999 to 2001, 711 had sources with geographic coordinates. Of these, 327 (37%) had water quality sampling data and were included in our final sample.

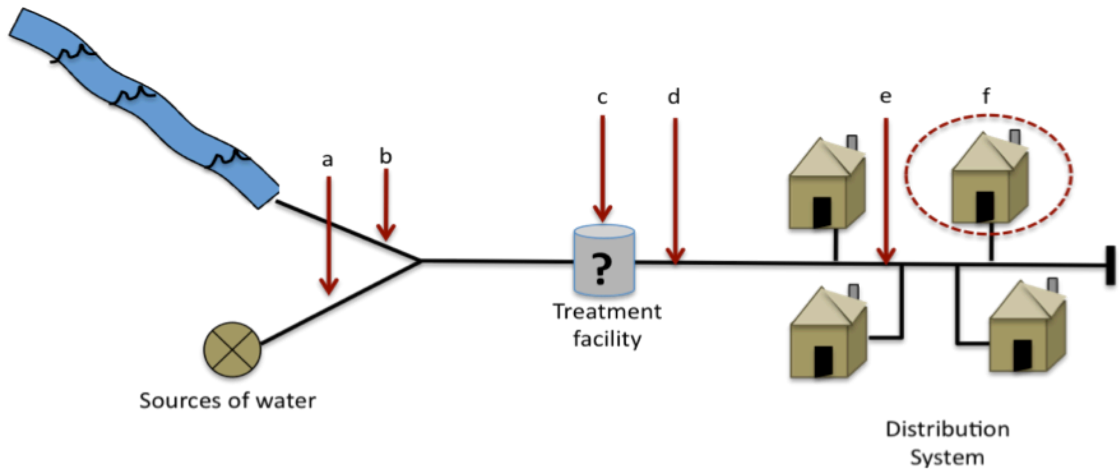

Figure 3.1. Schematic of a community water system (CWS) indicating: that water from a groundwater well or stream may be treated before entering into the distribution system, location of point-of-entry sources and use of proxy for tap water quality.

a=Water entering the distribution may flow from a groundwater well.
b=Water entering the distribution system may flow from a surface water source (i.e. stream)
c=Water may then be treated (different treatment techniques depending on the contaminant of interest, and original source).
d="Point-of-entry" into the distribution system. In this example, nitrate sample points would be used from point d, if points a and b flowed into the same point-of entry. Or, nitrate sample points would be averaged for points a and b each is a separate point-of-entry. In both cases, constant and equal flows are assumed.
e=Average nitrate level at point d is used to represent average water quality in the distribution system.
f=Nitrate levels in distribution system are a proxy for tap water quality.

Average Nitrate Concentration for CWS

To estimate nitrate concentration for each CWS, we selected two types of point-of-entry sources for inclusion (Figure 3.1): 1) sources in active use that had no treatment, or active sources that treated for contaminants other than nitrate, such as well fields or surface water plants, and 2) treatment plants in active use that potentially treated for nitrate. In both cases, we only included sources that were last in line to enter the distribution system (i.e. did not flow to another source before entering) as this could have resulted in double counting of nitrate levels. We used the CDPH's Permits, Inspections, Compliance, Monitoring and Enforcement (CDPH 2008a) database to identify source types, their location in relation to the distribution system, and their possible treatment techniques. If a plant had a treatment technique commonly used for nitrate (e.g. reverse osmosis), we assumed it treated for nitrate.

We then used nitrate-sampling data for these sources from CDPH's Water Quality Monitoring database (CDPH 2008B) to determine nitrate concentration at points-of-entry. This served as a proxy for water quality in each CWS' distribution system, and of tap water quality. Nitrate levels are unlikely to change from these entry points to the tap (unless systems chloraminate, which those in the Valley do not) (Haberman, R, personal communication). CWS using groundwater are required to sample each source for nitrate annually (unless a single sample or average of two samples exceeds the MCL, in which case the system must sample quarterly); CWS using surface water must sample quarterly

(California Code of Regulations 2008b). In practice, however, systems often fail to sample regularly (Haberman, R, personal communication). If a nitrate sample was below the detection limit of 2 mg NO3/L (California Code of Regulations 2008a), we took the square root of the value as a proxy for that sample's nitrate level (Lubin et al. 2004). We did not have flow measurements for the individual sources that contributed water to each CWS' distribution system. Therefore, we could not determine a flow-weighted measure of distribution water quality for each CWS based on the nitrate level measured in samples from each contributing source. Instead, we assumed that each point-of-entry source contributed independently and equally to a CWS' distribution system, and that each source contributed a constant amount to the system, regardless of season.

Next, we determined the average nitrate level for each point-of-entry source, and averaged the resulting values across all sources to estimate a system-wide nitrate level. The system-wide average was then used to categorize each CWS as: 1) low, defined as less than half the MCL (< 22.5 mg NO3/L); 2) medium (22.5 mg NO3/L to 44.9 mg NO3/L); and 3) high (≥ 45 mg NO3/L, the MCL for nitrate). These categories correspond to those used to assess source-level nitrate concentrations for regulatory purposes (California Code of Regulations 2008b). Besides the high category, the medium category is important to consider as research suggests there can be adverse health effects at half the MCL among susceptible subpopulations (DeRoos et al. 2003). In addition to calculating average nitrate levels, we used nitrate MCL violation data from PICME to verify whether systems with high nitrates did in fact receive violations, and to run a sensitivity analysis on the potentially exposed population.

Potentially Exposed Population

Using a method by Storm (1994), we computed the potentially exposed population (PEP) by apportioning the total population served by each CWS into three exposure categories based on the proportion of sources for that CWS with average nitrate levels that were low, medium, or high, as defined above. The population in each category was then summed across all CWS to estimate the total population potentially exposed to the three nitrate levels. The approach to calculate the PEP for the high-nitrate category is summarized by the following equation:

$$PEP_h = \sum_{i=1}^{327} (X_i \times s_{ih} / S_{it}) \qquad [1]$$

where X_i is the total population served in CWS *i*; s_{ih} is the number of sources for CWS *i* with average nitrate concentrations classified as high (*h*); and S_{it} is the total number of point-of-entry sources for CWS *i*. To calculate the PEP for the low (*l*) or medium (*m*) nitrate categories, we replaced s_{ih} with s_{il} or s_{im}, respectively. We used PICME 2008 data on the number of people served by each CWS to calculate the population size in each exposure category during 1999 to 2001. If the number of customers served by a CWS was not available from the PICME database, we used information from the CDPH Water Quality Monitoring database. To estimate population counts of potentially exposed individuals according to demographic characteristics (e.g. race/ethnicity) we multiplied the potentially exposed population in each nitrate category for each CWS by the estimated proportion of customers in each demographic subgroup for the CWS, and then summed these values across all CWS for each nitrate category. Because home ownership is based on housing units rather than population count, we did not derive a count of housing units.

Statistical Analysis of Nitrate Levels and CWS Characteristics

We used a linear regression model to analyze the relationship between CWS demographics and nitrate levels. We fit an *a priori*-selected model that controlled for known or hypothesized potential system-level confounders. We originally used a mixed model approach to account for clustering (Laird and Ware 1982). However, diagnostics of the mixed model indicated a very non-normal distribution of residuals (see Appendix B.5 for model comparison and residual assessment). Therefore, we used an approach that provided inference that was robust under laxer modeling assumptions. To derive the inference (i.e. standard errors), we clustered outcomes at the water system level (i.e. point-of-entry nitrate concentrations measured on a given day for a given source). Thus our final model reported sandwich-type robust standard errors (Huber 1967) that allow for arbitrary correlation, including correlation within or across sources in a CWS.

Our outcome variable, $Y_{ijk,}$ is nitrate concentration for the i^{th} water system, the jth source in system i, on day k (since January 1st, 1999). While nitrate samples from individual sources are our outcome measurements, the CWS is the primary unit of analysis, consistent with average nitrate level calculations discussed above. Our final model did not re-weight CWS with more samples (as the mixed model might have depending on the implied estimated correlation structure), as we wanted CWS to contribute based on a proxy of the number of people served. Thus systems with more measurements contributed more to the estimates. However, we addressed this assumption by stratifying by system size, to see if smaller CWS (with fewer samples) had a different effect than larger CWS. Because there was little difference between the estimates of the mixed model and the linear model, this provided evidence that the non-weighted approach of our final model was reasonable.

Key independent variables were the percent of Latino and non-Latino people of color served by CWS (referent category non-Latino whites) and percent home ownership in a CWS. Latinos were separately analyzed because they are the largest ethnic group in the Valley (40%, U.S. Census 2007). SES was represented by home ownership rate, which is a proxy for wealth and political representation (Krieger et al. 1997; Morello-Frosch et al. 2001; Oliver and Shapiro 1997). Because of our focus on CWS-level exposures, these variables were measured at the CWS-level. We assumed these remained constant for all three years.

Race/ethnicity and home ownership data were derived from the 2000 U.S. Census (U.S. Census Bureau 2000b). Since CWS service areas do not follow Census boundaries we used two spatial approaches in Geographic Information Systems (GIS) to estimate demographic variables for each CWS. We first compared an aerially-weighted approach using digitized CWS boundaries in two pilot counties (Tulare and Fresno) with a second approach joining spatial coordinates from CDPH data for all sources (well fields, surface water intakes and treatment plants) to Census block groups. Based on spatial and goodness-of-fit comparisons, we concluded it was reasonable to use the latter approach (see Appendix B.1 for details on the aerially-weighted approach and the comparison between the two methods). In brief, for each CWS, we estimated a population-based average of each variable across all block groups that included sources for the CWS. For example, if a CWS had two sources in two Census block groups, we determined the population-weighted average of the variable across both Census block groups and used that value to derive a percent estimate of demographic groups (e.g. 50% Latino) served by each CWS.

We controlled for other water system characteristics that could be potential confounders, including: source of water (ground water or groundwater and surface water versus surface water alone); whether the system served a city (i.e. incorporated) or an unincorporated area; ownership structure of the system (publicly versus privately owned and not regulated by the Public Utility Commission (PUC), with privately owned PUC-regulated as the referent category); system location (agricultural Valley floor or not); season (summer/fall or winter/spring); year of sampling (2000 or 2001, with 1999 as referent category); and number of service connections (< 200 or ≥200 connections). CWS with fewer than 200 connections are generally considered "small" (California Code of Regulations 2008a). We determined ownership structure by combining data in PICME with data from the PUC's list of regulated systems. We obtained all other characteristics from PICME. With the exception of year and season that were measured at the source-level, all covariates were measured at the water system level.

In addition to models including all CWS, we stratified by system size to assess if demographic effects on water quality might be stronger among smaller systems, and to test the hypothesis that scale alone explains water quality. We also used our final model to estimate the amount of nitrate contamination attributable to the proportion of the population that is Latino. We did so by using the final model to predict expected values for each observation if percent Latino equaled zero as described in Greenland and Drescher (1993). All statistical analyses were conducted using Stata v10 (College Station, Texas). We used Stata's cluster command (clustering at the CWS level) to derive robust standard errors.

Results

Descriptive Statistics

The 327 systems in our sample served approximately 2.95 million people, or 96% of the population served by CWS (Table 3.1). The distribution of average system-level nitrate concentrations is right-skewed, and ranges from 0 to 150 mg NO_3/L. This distribution and range is similar for average source-level nitrate concentrations as well as individual sampling points (see Appendix, Figures B.2-B.4). The mean proportion of Latinos served across these CWS was 32% with an inter-quartile range (IQR) of 10 to 50%. The mean proportion of homeownership was 70% with an IQR of 60 to 82%. Compared to all the CWS in the Valley active from 1999 to 2001, our study sample under-represented small CWS that have fewer than 200 connections (49% versus 73%, Table 3.1). The number of samples per source in systems with fewer than 200 connections ranged from 1 to 110 (mean=3.2), as compared to a range of 1 to 133 (mean=4.5) for systems with at least 200 connections. Six percent of samples had concentrations below the detection limit.

Overall, three percent (n=10) of all CWS in our sample had average nitrate concentrations above the MCL for at least some part of the study period, 10% (n=33) had average concentrations from half the MCL to the MCL, and 87% (n=284) had average concentrations below half the MCL (Figure 3.2). Of the ten systems with an average nitrate concentration over the MCL, nine had fewer than 200 connections and eight had only one or two sources (Appendix B.6, Table B.9). All but one of these ten systems received at least one MCL violation during the study period, and 14 CWS in our sample (serving ~92,268 people) received at least one MCL violation (CDPH 2008a).

CWS that served higher fractions of Latinos and lower fractions of homeowners (i.e. more renters) had higher average nitrate levels. Figure 3.3 shows that in the two highest Latino quartiles there were proportionately more systems with average nitrate concentration greater than the MCL (i.e. 5% and 7% in the two higher quartiles compared to 0% in both of the lower quartiles). These two quartiles also had the largest fractions of CWS in the medium nitrate category. The two quartiles with the lowest rates of home ownership had the largest proportions of systems in the "medium" and "high" nitrate categories (15% and 22%, respectively), compared to the two quartiles with the highest rates of home ownership (which had 7% and 8%, respectively).

Table 3.1. Community water systems (CWSs) included in study sample compared to all active CWS, San Joaquin Valley, CA 1999-2001.

Variable of Interest	Active CWS with Source Location N=711	CWS in Study N=327	CWS < 200 Con.: N=160	CWS ≥ 200 Con.: N=167
Total population	**3,047,822[a]**	**2,948,346**	**27,165**	**2,921,181**
Latino population (%)	34	39	29	40
White population (%)	58	47	64	47
Population above poverty level [b] (%)	57	57	59	57
Population served (mean/median)	4206 / 150	9016 / 565	170 / 100	17492 / 430
Incorporated [c] (%)	9	18	2	34
< 200 Connections	73	49	100	0
Only groundwater (GW)[d] (%)	89	90	97	84
GW and SW[d] (%)	5	8	3	13

Con. = service connections

[a] Approximately 71,418 people were served by CWS whose sources did not have geographic coordinates, and 80 systems had no population estimates available; this would bring the estimate of "true" population served by active CWS to at least 3,119,240.

[b] Above 200% the poverty level.

[c] A water system that serves a city that is a legally recognized municipal corporation with a charter from the state and governing officials that is incorporated, as opposed to a water system that serves an unincorporated area.

[d] Reference group=surface water only.

Of the population served in our sample, approximately 84.6% (~2,494,442 people) was potentially exposed to nitrate levels less than half the MCL, 15.2% (~448,729 people) to nitrate levels between half the MCL and the MCL, and 0.2% (~5,176 people) to nitrate levels at or above the MCL (Table 2). Of the 5,176 people served water with nitrates above the MCL, 56% were people of color (50% Latinos and 6% non-Latino), compared to 52% in the low and medium nitrate categories (Table 3.2). The percentage of Latinos served by high nitrate CWS was higher than the percentage of Latinos in our entire study sample (39%, Table 3.1). This percentage was also higher than the percentage of Latinos served by CWS in the other two nitrate categories (39 and 40% for low and medium nitrate, respectively).

Model Results

Multivariate modeling results are shown in Table 3.3. Unadjusted models indicate that percent Latino was positively and significantly (ß=.14; 95% Confidence Interval (CI), .04, .24) correlated with the average nitrate concentration in the distribution system. Conversely, home ownership was negatively correlated with average nitrate concentration, but marginally significant (ß=-.15; 95% CI, -.30, .003).

Our adjusted model suggests that, on average, a 1% increase in Latinos served by a CWS was associated with an increase of .04 mg NO_3/L (95% CI, -.08, .16). For home ownership, each percent increase was associated with a decrease of .16 mg NO_3/L (95% CI, -.33, .002). For systems with fewer than 200 connections, the association between percent Latino and home ownership and nitrate concentration was consistent with both the unadjusted model and adjusted model for all CWS, but the strength of the association for percent Latino increased. Specifically, on average, each percent increase in Latino was associated with a .44 mg NO_3/L increase (95% CI, .03, .84) in the smaller systems. A 1% increase in home ownership was associated with a .15 mg NO_3/L decrease (95% CI, -.64, .33), although the association was not statistically significant. In systems with at least 200 connections, neither race/ethnicity nor home ownership was associated with nitrate concentrations. Using the final model to predict expected values, we estimated that among small systems nitrate levels for CWS with 0% Latino would be on average 6 mg NO_3/L lower compared to CWS at the mean.

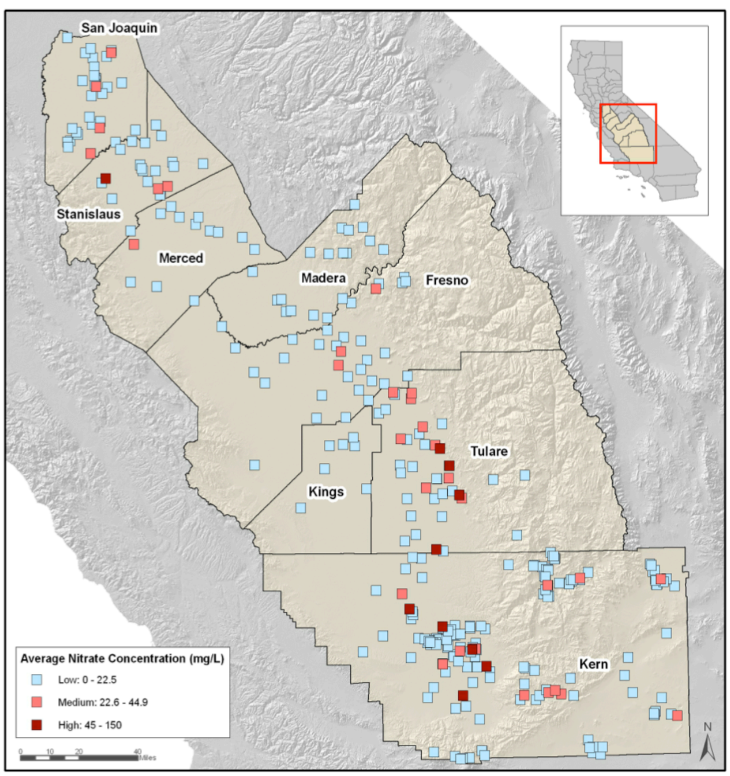

Figure 3.2. Average nitrate concentration[a] of Community Water Systems' (CWS [b,c]) in California's San Joaquin Valley, 1999-2001 (n=327)
[a] Estimate based on average of each point-of-entry source's average concentration.
[b] Sources of data: CDPH Water Quality Monitoring (2008) and CDPH (2008a)
[c] Approximate location of CWS are depicted, but not true boundaries. Due to close proximity of some CWS, map partially covers some CWSs.

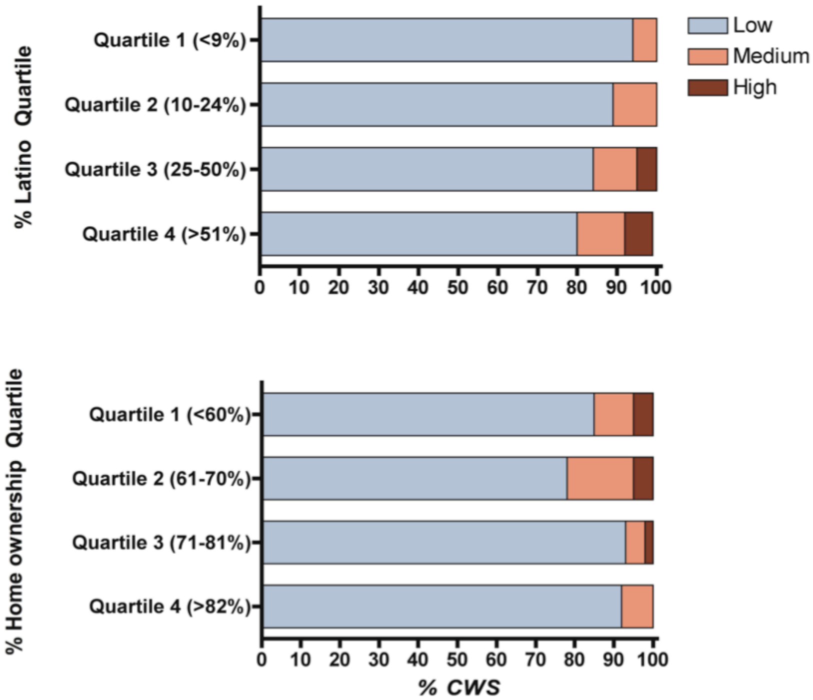

Figure 3.3. Percent of community water systems (CWS) with average nitrate concentration[a] that was low, medium and high[b], by quartiles of percent Latino and Home Ownership.

[a] Average system concentration is derived from the average of each source's average concentration at point of entry.
[b] Low=less than one half the maximum contaminant level (MCL) of 45 mg NO3/L; medium=one half the MCL up to the MCL, high=greater than or equal to the MCL.

Table 3.2. Demographic profile of total potentially exposed population (PEP[a]) in study sample by average level of nitrate concentration.

	Nitrate Level[b]		
Variable of Interest	**Low**	**Medium**	**High**
% Total Population (N=2,948,346)	84.6	15.2	0.2
% Latino (N=1,164,714)	39	40	50
% Non-Latino People of Color (N=389,336)	13	12	6
% White (N=1,394,296)	47	48	44

[a] Per water system, PEP= population of demographic variable of interest x (# of point-of-entry sources in one of three nitrate levels/total # of point-of-entry sources). PEP displayed in table is equal to sum across all water systems. This value can also be interpreted as the estimated number of people served water at this level.
[b] Low= less than one half the MCL of 45 mg NO_3/L; medium=one half the MCL up to the MCL (22.5 mg NO_3/L to the MCL (22.5 mg/L to 44.9 mg/L NO_3), high= greater than or equal to the MCL.

Table 3.3. Regression[†] for factors associated with nitrate concentration (mg NO_3/L) in community water systems (CWS), with beta coefficients, level of significance and 95% CI indicated.

Variable	**Model A[a]**	**Model B[a]**	**Model C[b]**	**Model D (<200 Con.)**	**Model E (≥ 200 Con.)**
Constant	14.2*** (9.1, 19.4)	27.1* (16.3, 38.0)	6.3 (-11.4, 24.0)	10.8 (-32.3, 53.9)	3.2 (-15.5, 21.9)
% Latino	.14*** (.04, .24)		.04 (-.08, .16)	.44** (.03, .84)	-.01 (-.12, .10)
% Non-Latino people of color	-.18 (-.62, .25)		-.15 (-.47, .18)	-.44 (-1.1, .27)	-.13 (-.45, .18)
% Home ownership		-.15* (-.30, .003)	-.16* (-.33, .002)	-.15 (-.64, .33)	-.10 (-.27, .07)
Incorporated			-4.4 (-9.3, .56)	-2.9 (-31.8, 25.9)	-4.1 (-9.3, 1.1)
Groundwater or combined[c]			9.7*** (4.3, 15.2)	na	11.7*** (7.9,15.4)
Private non-PUC regulated[d]			2.7 (-5.4, 10.9)	5.5 (-2.7, 13.7)	-.23 (-4.5, 4.1)
Public[d]			7.2*** (2.8, 11.5)	10.3 (-17.4, 38.0)	7.3 (3.6, 11.1)
< 200 service connections			9.1 (-2.5, 20.7)	na	na
Valley floor			7.9** (1.6, 14.2)	1.7 (-12.0, 15.4)	7.4 (1.0, 13.9)
2000[e]			1.3 (-.44, 3.1)	5.0*** (1.2, 8.8)	.71 (-1.1, 2.6)
2001[e]			1.4* (-.26, 3.1)	5.5*** (1.9, 9.1)	.67 (-.98, 2.3)
Summer/fall			1.3 (-.30, 2.9)	3.2** (.31, 6.3)	1.1 (-.72, 3.0)

[†]Regression with robust standard errors, clustered by community water systems. Coefficients represent the estimated difference in mean concentration at the system-level associated with a unit change in the covariate (95% CI). R^2=.13
[a]Unadjusted models, all CWS included
[b]Adjusted model, all CWS included
[c] Surface water is referent category
[d]Privately owned PUC-regulated CWS are referent category
[e] 1999 is referent year
Con.=connections
na=not applicable, as no CWS in this model run contains this factor, or all CWS have this factor
* p<.10, ** p<.05, *** p<.01

Discussion

To our knowledge, this is the first study to examine the relationship between nitrate levels in community water systems and social disparities in the U.S. After stratifying by system size, we found that among systems with fewer than 200 connections, those serving higher percentages of Latinos had higher nitrate levels. We found an inverse but not statistically significant association between home ownership and nitrate levels for smaller systems. For large systems, we did not find significant associations between race/ethnicity or home ownership and nitrate levels. Our findings corroborate previous drinking water studies (Byrne 2003) that find a positive relationship between percent minority and poor water quality, but are specific to nitrate contamination at the community level. That water quality varied by percent Latino or home ownership matters not only on account of environmental equity, but also because elevated nitrate levels could pose a greater hazard to sub-populations that may have less access to health care.

The association of race/ethnicity and SES with nitrate levels could be due to several factors. Race/ethnicity could have been related to proximity to agriculture, as well as the ability of residents to participate in the governance of their CWS. For example, in systems with higher fractions of Latinos, language abilities, citizenship status, or lack of political clout could inhibit residents from speaking out and demanding improvements in water quality (Michelson 2000). Home ownership could have been negatively associated with nitrate levels because renter-based communities may have had a lower capacity to pay for improvements in water infrastructure, or to hold a CWS accountable, assuming they received notices of violation as required (CCR 2008d). Or, it might indicate that a lack of economic resources may influence whether CWS can hire capable water managers, or comply with regulations.

That over 5,000 people in our study sample were potentially exposed to drinking water with nitrate concentrations above the MCL raises health concerns. As noted, acute and chronic health effects have been found for vulnerable populations (e.g. infants and pregnant women) exposed to nitrate over the MCL (Ward et al. 2005). Furthermore that many of the water systems in this category had only one or two sources can lead to increased chances of high exposure for residents if these are high-nitrate sources, as there is no immediate alternative source to draw from or blend with. These small systems often go years with high nitrate levels, until a new water source can be developed (Spath, D, personal communication). CDPH (2008a) data corroborates this observation: of the ten systems whose average was over the MCL, six had recurring MCL violations over an eight-year timeframe. In these systems, customers may be continually exposed (though exposure would be lower if people frequently use alternative sources). Additionally, as customers in the San Joaquin Valley often cope by purchasing bottled water, they pay two water bills—one for tap water and the other for bottled water (Moore et al 2011).

Our results also highlight a less frequently discussed issue—approximately 448,700 residents and 33 CWS in our sample were served water with nitrate levels less than, but exceeding half the MCL. Nitrate levels in these systems could be approaching the MCL in some cases. For example, one third of CWS in this category had average concentrations over 30 mg NO_3/L. Residents in these systems could be at risk of adverse health outcomes, and/or could experience additional economic costs. For example, in one study, the risk of colon cancer increased for certain susceptible subgroups (e.g. those with low Vitamin C intake, or high meat intake) whose water had nitrate levels greater than half the nitrate MCL for at least ten years (DeRoos et al 2003). More generally, while several authors have suggested that the current nitrate MCL could be increased to a less stringent level of regulation (L'hirondel and L'hirondel 2002), others have argued that the standard should not be changed (Ward et al 2005) because of uncertainties in the exposure assessment data and health effect estimates of the epidemiologic study upon which the current MCL is based (Walton 1951). Some authors also argue that the MCL includes no safety factor, and documented cases of infant methemoglobinemia have occurred at levels below the MCL (Fan and Steinberg 1996; Johnson and Kross 1990). Thus exposure to nitrates in the middle category is important to consider.

Monitoring of water quality by these CWS is also an important consideration. CWS with a source that has nitrate levels exceeding 22.5 mg NO3/L are required to increase their monitoring frequency from annual to quarterly sampling (California Code of Regulations 2008a). The cost of increased monitoring must be passed along to consumers. However, funding and staffing constraints can limit the capacity of small CWS systems to monitor, so these CWS may have nitrate levels approaching the MCL but neither the system operators nor customer base would know (Haberman R, personal communication). Such a scenario would undermine the aim of the SDWA, which is supposed to protect the public from harmful exposures, and requires systems to notify their customers so precautionary measures can be taken to reduce exposures (California Code of Regulations 2008e; Fan and Steinberg 1996).

This study used an appropriate unit of analysis (i.e. CWS) for estimating system-level nitrate exposure. The methods we used could be applied to other contaminants, and regions of the U.S. However, sources of error exist in our demographic estimate because: 1) surface intakes/well fields could fall in Census block groups not served by the CWS, 2) not all Census block groups served by a CWS have an intake/field located within them, and 3) Latinos in Census data could be undercounted due to legal status. Despite these potential errors, for the majority of CWS, sources fell within the same Census block groups that overlapped with CWS' service area boundaries. And, for nine of the ten systems in the high nitrate category, all sources were in the same Census block groups as those included in each CWS service area (see Supplemental Material). Additional sources of error include possible misclassification of points-of-entry to the distribution system due to errors in PICME. Furthermore, as the relative flow of different sources contributing to each was not known, our method may have over- or under-estimated average nitrate levels. However, at least among CWS with average concentrations over the MCL, the estimated concentrations were similar to the measured concentrations for which that CWS received one or more MCL violations. Our measure of exposure was limited by data availability—and thus for systems with fewer samples tested for nitrates, our estimate may be less accurate. While the population potentially exposed to nitrate over the MCL is small, it is likely to be an underestimate of the actual population impacted in the San Joaquin Valley. This is partly because our study under-represented smaller CWS, and also because we used average rather than maximum nitrate levels, or other measures of nitrate (e.g. MCL violations). Thus our estimate is likely to be a conservative measure of potential exposure. The estimate of the population potentially exposed may also contain some error, as there may be some differences among utilities in how population estimates are calculated. Finally, while our results are based on data that are ten years old, we believe that, at a minimum, they capture current trends in the Valley. Nitrate concentrations generally change slowly in deeper public supply wells, and have been increasing in most locations due to increasing fertilizer use (Dubrovsky et al. 2010).

Conclusion

Our study is one of the first to analyze the relationship between drinking water contamination, race/ethnicity and SES in the United States, and the first that focuses on nitrate contamination. Our results indicate that Latinos in the San Joaquin Valley may be disproportionately exposed to higher levels of nitrates, and that this exposure is particularly prevalent in smaller water systems. With the increasing use of nitrogen-based fertilizers and growing demand for groundwater, these trends are likely to worsen in future years. Regulatory and policy strategies to address scale-related vulnerabilities in drinking water quality have generally ignored the environmental justice implications for CWS. Given US EPA's renewed focus on environmental justice (U.S. Environmental Protection Agency 2011b) and the paucity of environmental justice studies on drinking water, this study highlights the importance of targeting funding for mitigation and source water protection efforts for under-served communities and those with nitrate levels over the MCL. Furthermore, there is a need for resources for both monitoring water quality and precautionary mitigation for communities whose nitrate levels are above one-half the MCL.

CHAPTER 4
Environmental Justice and Arsenic Contamination: Exposure and Compliance in Community Drinking Water Systems

Overview

Few studies of environmental justice examine inequities in drinking water contamination. Those studies that have, usually analyze either disparities in exposure/harm *or* procedural inequities. The 2001 Revised Arsenic Standard offers an opportunity to jointly address these burdens and broaden the scope of this field. In 2002, US EPA strengthened the drinking water standard for arsenic from 50 to 10 μg As/L. Policy-makers recognized that compliance costs would be higher for smaller systems, but gave limited attention to social disparities in exposure or capacity of systems to comply. We hypothesized that Community Water Systems (CWSs) serving a higher proportion of minority or lower SES residents have higher odds of non-compliance with the revised standard and higher arsenic levels. Using water quality sampling data for arsenic and maximum contaminant level (MCL) violation data for 464 CWSs in California's San Joaquin Valley (2005-2007) we ran bivariate tests and linear regression models. We found that MCL violations and higher arsenic levels were most common in CWSs serving predominantly socio-economically disadvantaged communities. Our findings suggest that environmental justice concerns related to arsenic are distributional *and* procedural: communities with low-socioeconomic status residents not only face disproportionate exposures, but unequal challenges of mitigating contamination.

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Introduction

Arsenic in drinking water is linked to skin, lung, bladder and kidney cancers (Chen et al. 1985; Fereccio et al. 2000; Tsai et al. 1999) and has one of the highest cancer risks among the contaminants regulated by the Safe Drinking Water Act (SDWA) (Smith et al. 2002). The most common exposure pathway is consumption of groundwater containing arsenic, which is generally naturally occurring but can also derive from anthropogenic sources (Prüss-Ustün et al. 2011). Many epidemiological studies examining health effects of arsenic in drinking water have been conducted in areas with extremely high levels (i.e., >100 μg As/L)—such as Argentina, Bangladesh and Taiwan. But high concentrations (i.e., 50-100 μg As/L) also occur in the U.S, especially in western regions such as Utah, Nevada, Arizona and California's San Joaquin Valley (Bates et al. 1995; Cory and Rahman 2009; Lewis et al. 1999; Steinmaus et al. 2003).

In 2001, the U.S. EPA approved tightening the arsenic drinking water standard from 50 to 10 μg As/L, on the basis of epidemiologic evidence and cost-benefit considerations (National Research Council 2001). Effective in 2002, the Revised Arsenic Rule required all public water systems to comply with the new standard by January 23, 2006 (U.S. Environmental Protection Agency 2011a). Public water systems supply piped water for human consumption to at least 25 people or at least 15 service connections for at least 60 days per year (U.S. EPA 2010b). Among critiques of the revised maximum contaminant level (MCL), some noted that smaller water systems would face higher costs of compliance per household (Jones and Joy 2006; Oates 2002), creating social inequities. But besides scale-related cost, little attention was given to potential social disparities in exposure to arsenic, or in the ability of smaller water systems to comply with the revised standard.

These critiques came on the heels of a call for increased attention on environmental justice implications of environmental contamination and related regulatory and mitigation policies and programs. Spurred partly by Executive Order 12898, the environmental justice order signed by President Clinton, studies have generally focused on two separate components of environmental justice—distributional inequities or procedural inequities. Distributional inequity addresses disparities in exposure or health outcomes. Procedural inequity is concerned with the inequitable implementation of policies and programs, including access to federal funds or impacts of historical planning efforts. Attention to each of these sub-components of environmental justice is, of course, warranted. We argue, however, that a joint focus -- on distributional and procedural inequities -- is most helpful for understanding the health and social implications of water policies, including the Revised Arsenic Rule.

Compared to toxics siting and air pollution, inequities regarding drinking water contamination have received less attention in the EJ literature. But despite the near universal access to clean drinking water in the U.S (Levin et al. 2002) and successful passage of the Safe Drinking Water Act (Roberson 2011), there is evidence of regional inequities in water quality and service provision in places such as small border towns along the U.S.-Mexico border (Olmstead 2004; Pilley et al. 2009), African American communities in the rural South (Heaney et al. 2011; Wilson et al. 2008a) and unincorporated communities in California's San Joaquin Valley (Balazs et al. 2011). As with other environmental justice studies, research that has quantified inequities in drinking water has generally focused separately on exposure or on regulatory implementation. Specifically with respect to arsenic contamination, recent studies have explored potential inequities in exposure to arsenic (Pilley et al. 2009; Stone et al. 2007) and in enforcement of the arsenic standard (Cory and Rahman 2009). The Revised Arsenic Rule offers an opportunity to jointly address these burdens and thus broaden the scope of this field.

This study develops an integrated approach, which we term "joint burden analysis", to analyze environmental justice implications of arsenic contamination in the U.S. Our analysis focuses on both compliance and exposure burdens. Quantifying a water system's compliance with the arsenic MCL is important in order to know which systems are in violation, and to consider whether they are equipped to comply. Quantifying exposure levels and their distribution is equally important, given known health risks at levels even below the new standard (National Research Council 2001). Together, these analyses provide a picture of the joint burdens that water systems and residents may face.

We conducted our study in California's San Joaquin Valley, one of the poorest regions in the country with some of the most contaminated drinking water sources in California (Dubrovsky et al. 2010), including high nitrate and high arsenic levels (Bennett and Belitz 2010; http://pubs.usgs.gov/fs/2010/3079/). Social disparities in exposure to nitrate have already been documented in this region (Balazs et al. 2011). For this study, we hypothesized that CWSs that serve a higher proportion of minority or lower SES residents have higher odds of non-compliance with the revised arsenic standard and that these CWSs serve drinking water with higher levels of arsenic.

Materials and Methods

Sample Selection and Selection of Point-of-Entry Sources

We selected CWSs, located in California's San Joaquin Valley counties, that were in active use between 2005 and 2007, had any source with a geographic coordinate that could be used to estimate customer demographics, and had at least one active point-of-entry source with an arsenic sample reported during this period. Our time period represents one full compliance period under the SDWA, in which each CWS should have taken at least one arsenic sample (California Code of Regulations 2008c).

Point-of-entry sources are those that directly enter the distribution system. We selected two types of point-of-entry sources: 1) sources in active use that had no arsenic treatment, or that treated for contaminants other than arsenic, and 2) treatment plants in active use that potentially treated for arsenic (Figure 4.1). We used the California Department of Public Health's (CDPH) Permits, Inspections, Compliance, Monitoring and Enforcement (PICME) database (CDPH (California Department of Public Health) 2008a) to identify source types, their location in relation to the distribution system, and their possible treatment techniques. We confirmed the existence of arsenic treatment technologies with state and county regulators.

For the six CWSs with confirmed arsenic treatment plants that were in use during the study period, we used all point-of-entry arsenic sampling points prior to installation of treatment, and only sampling points from treatment plants after the installation date. For CWSs with no confirmed arsenic treatment, we selected systems where either all point-of-entry sources were labeled as untreated, or all point-of-entry sources were labeled as having treatment. In practice a CWS may both treated and untreated sources. But because the CDPH databases did not allow us to accurately ascertain whether untreated sources entered the distribution system if treated sources were also available, we selected CWSs in this manner as a conservative measure. We tested the sensitivity of this decision, by comparing regression results using our final sample to results using all CWSs. Our final sample included 464 of the 671 CWS active in the Valley from 2005 to 2007.

Outcome Measures and Independent Variables

For each CWS, we derived four main outcome measures: 1) officially recorded arsenic MCL violations, 2) average system and source-level arsenic concentrations, 3) population potentially exposed to arsenic, and 4) concentration of arsenic at point-of-entry to the distribution system. We used the first measure to analyze compliance. We used the second two measures to derive descriptive exposure statistics and run sensitivity analyses. We used the fourth measure as the outcome variable in a linear regression model.

Arsenic MCL Violations

The key outcome for our compliance analysis was officially recorded arsenic MCL violations derived from the PICME database. We created a binary variable indicating whether a system had received at least one MCL violation during the study period. This measure helped control for bias that could occur because CWSs with higher arsenic levels are required to sample more frequently (California Code of Regulations 2008a) thereby increasing the probability that they would receive more MCL violations. To consider the impact of under- or mis-reported violations, we ran sensitivity analyses using the number of CWSs whose average arsenic concentration at the source or system-level had exceeded the MCL.

Average System and Source-Level Arsenic Concentrations

We used arsenic water quality sampling data for the selected point-of-entry sources from CDPH's Water Quality Monitoring database (CDPH (California Department of Public Health) 2008b)to estimate arsenic concentrations in the distribution system (Figure 4.1). Using these data points, we derived the average arsenic concentration served by each CWS for the entire compliance period. We calculated this by averaging the average source concentration for each system during our time period. We assumed average system-level concentrations represent the water quality that residents received, as in previous studies (Cory and Rahman 2009; Stone et al. 2007). We also calculated each CWS's yearly average arsenic concentration to conduct sensitivity analyses.

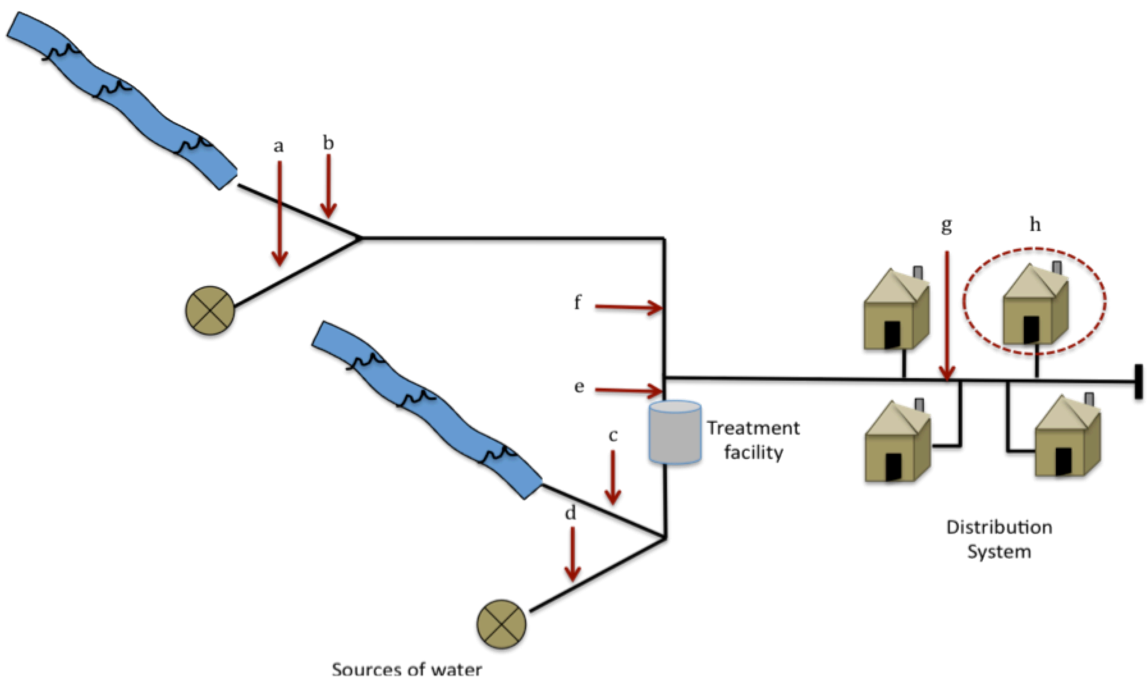

Figure 4.1. Schematic of a community water system (CWS) indicating: that water from a groundwater well or stream may be treated or untreated before entering into the distribution system, location of point-of-entry sources and use of proxy for tap water quality*.

* In a system with untreated sources, water entering the distribution may flow from a groundwater well (point a), or from a surface water source (point b). In a system with no treatment, if points a and b flow into the same point-of-entry (point f), the average arsenic levels of each source is averaged at point f. If points a and b do not flow into a common point-of-entry, each is, in essence, a point-of-entry. In a system with treatment, water from surface water (point c) or groundwater sources (point d) is treated for arsenic at a treatment facility, and point e is the common point of entry. The average concentration at all point-of-entry sources is used to represent average water quality in the distribution system (point g), which is a proxy for tap water quality (point h).

Because we did not have flow measurements for individual sources, we assumed that each point-of-entry source contributed independently, constantly and equally to a CWS's distribution system, regardless of season. For sampling points below the detection limit, we took the square root of the detection limit as a proxy for the arsenic concentration (Lubin et al. 2004). We categorized source and system-level averages into three concentration categories: 1) < 10 μg As/L ("low"), 2) 10-49.9 μg As/L ("medium"), and 3) ≥50 μg As/L ("high"). We used average source and system-level concentrations to create binary variables that we used in bivariate analyses. Here, average levels were coded as 1 (> 10 μg As/L), or 0 (< 10 μg As/L). Our use of average source-level concentrations, and the related binary variable, allows for a close approximation of whether a system exceeded the MCL since arsenic MCL violations are based on whether the average two consecutive samples for a source exceeds 10 μg As/L (California Code of Regulations 2008a).

Potentially Exposed Population

Using a method previously developed (Storm 1994), and described in Balazs et al (2011), we computed the population potentially exposed to the three aforementioned exposure categories. The approach to calculate the potentially exposed population (PEP) for the high-arsenic category is summarized by the following equation:

$$PEP_h = \sum_{i=1}^{464} (X_i \times s_{ih} / S_{it}) \qquad (1)$$

where X_i is the total population served in CWS i; s_{ih} is the number of sources for CWS i with average arsenic concentrations classified as high (h); and S_{it} is the total number of point-of-entry sources for CWS i. To calculate the PEP for the low (l) or medium (m) categories, we replaced s_{ih} with s_{il} and s_{im}, respectively. We used PICME data on the number of people served by each CWS to calculate the population size. If the number of customers served by a CWS was not available from the PICME database, we used information from the Water Quality Monitoring database. To estimate counts of potentially exposed individuals according to demographic characteristics (e.g. race/ethnicity) we multiplied the PEP in each arsenic category for each CWS by the estimated proportion of customers in each demographic subgroup for the CWS (e.g. 50% people of color), and then summed these counts across all CWSs for each arsenic category.

Concentration of Arsenic at Point-of-Entry

Arsenic sampling data for each point-of-entry source were used as the outcome variable in our regression model, as described under "Regression Model" below.

Analyses

Compliance Burden Analyses

We used our arsenic MCL variable to analyze whether CWSs with higher fractions of people of color or lower SES faced greater compliance burdens. Because only 34 CWS had at least one MCL violation we did not have enough outcomes to use multivariate regression techniques. Instead we ran difference of means tests (i.e., t-test) and Fisher's Exact tests for contingency tables, comparing the presence of at least one MCL violation by high or low levels of our variables of interest (i.e. race/ethnicity or homeownership). We used the median value of these variables to determine the threshold for high and low levels.

Exposure Analyses

To assess the relationship between demographics of customers served by CWSs and potential exposure, we first examined the demographic characteristics of the population potentially exposed to three different arsenic levels, and additional characteristics of the systems at those levels. To further analyze the relationship, we used our binary variable of average system-level arsenic concentrations to conduct difference of means tests and Fisher's Exact tests. These tests compared whether systems with high and low home ownership rates, or high and low percentages of people of color had a greater chance of having average arsenic levels in excess of the revised arsenic standard.

Regression Model

Finally, we examined the relationship between system-level demographics and arsenic levels using our continuous measure of arsenic concentrations. We used a linear regression model with robust standard errors to account for clustering. To derive the inference, we clustered outcomes at the CWS-level (i.e. point-of-entry arsenic concentrations measured on a given day for a given source). Our final model reported sandwich-type robust standard errors (Huber 1967) that allowed for arbitrary correlation, including correlation within CWS units. The a priori-selected model controlled for known or hypothesized potential system-level confounders.

The model's outcome variable, Yijk, was arsenic concentration for the ith water system, the jth source in system i, on day k (since January 1st, 2005). While arsenic samples from individual sources were our outcome measurements, the CWS was the primary unit of analysis, consistent with other calculations above. Our final model did not re-weight CWSs with more samples; thus systems with more measurements contributed more to the estimates. We addressed this issue by stratifying by system size to see if smaller CWSs (with fewer samples) had a different effect on water quality than larger CWSs.

Key independent variables were the percentage of people of color served by CWSs (referent category non-Hispanic whites) and percent home ownership in each CWS. SES was represented by home ownership rate, which is a proxy for wealth and political representation (Krieger et al. 1997). We also used the SES of residents as a measure of the economic resources available to a water system, which can affect the ability of a system to mitigate contamination (Committee on Small Water Systems 1997). Race/ethnicity and home ownership data were derived from the 2000 U.S. Census, measured at the CWS-level, and assumed to be constant for all three years (U.S. Census Bureau 2000b). Since CWS service areas do not follow Census boundaries we used a spatial approach in Geographic Information Systems (GIS) to estimate demographic variables for each CWS. In brief, for each CWS, we estimated a population-weighted average of each variable across all block groups that contained sources for the CWS. This value was used to derive a percent estimate of demographic characteristics (e.g. 50% homeownership) served by that CWS (Balazs et al. 2011).

We controlled for other potentially confounding water system characteristics including: source of water (ground water or groundwater and surface water versus surface water alone); system ownership (public, privately owned and not regulated by the Public Utility Commission (PUC), with private PUC-regulated as referent category); geographic location (Valley floor and foothills, with mountains as referent category); season (summer/fall or winter/spring); year of sampling (2006 and 2007, with 2005 as referent category); and number of service connections (<200 or ≥200 connections) . We determined ownership structure by combining data in PICME with data from the PUC's list of regulated systems. We obtained all other characteristics from PICME. With the exception of year and season, all covariates were measured at the water system level.

We stratified by system size to assess if demographic effects on water quality might be stronger among smaller systems, and to test the hypothesis that scale alone explains water quality. We used number of connections as a threshold for small versus large CWSs, where those with fewer than 200 connections are considered "small" (California Code of Regulations 2008a). We used our final model to estimate the amount of arsenic contamination attributable to the proportion of the population that are homeowners by predicting expected values for each observation if percent homeownership equaled 100%, as described in (Greenland and Drescher 1993). All statistical analyses were conducted using Stata v10 (College Station, Texas). We used Stata's cluster command to derive robust standard errors.

Results

Descriptive Statistics

The 464 CWSs in our study sample served 1.134 million people, representing 35% of the total population served by CWSs between 2005 and 2007, and 69% of all CWSs active through 2007 (Table 4.1). The mean percentage of people of color served by each CWS was 31% [inter-quartile range (IQR), 9-48%]. The mean percent of homeownership was 70% (IQR, 60-81%). The yearly average arsenic concentration in 2005, 2006 and 2007 was 7.0 μg As/L (median=3 μg As/L) 7.9 μg/L (median=2.5 μg As/L), and 6.8 μg As/L (median=3 μg As/L). Approximately 12% of samples were below the detection limit.

Nearly 16% of all CWSs in the sample had average arsenic concentrations between 10 and 50 μg As/L, and were therefore affected by the revised standard (Table 4.2). Among these, 66% had fewer than 200 connections, and 86% had three active wells or less. For each CWS with average arsenic in this range, the average percentage of a system's sources that exceeded the revised MCL was 87% (Table 4.2). Less than 1% of CWSs had average levels greater than 50 μg As/L. Among these, all had fewer than 200 connections. CWSs west of Highway 99 and in the central portion of the Valley have higher arsenic levels, as do some areas in the foothills and in southeastern Kern County (Figure 4.2).

Of the population served in our sample, approximately 14% was potentially exposed to arsenic levels over 10 μg As/L MCL (Table 4.3). Of the population potentially exposed to 10-50 μg As/L, 61% were people of color. This is higher than the corresponding percentage in the entire study sample (i.e., 55%, Table 4.1).

Table 4.1. Characteristics of community water systems (CWSs) in study sample including water quality and population served, and compared to all active CWS, San Joaquin Valley, CA (n=464), 2005-2007.

Variable of Interest	**Active CWS n=644**	**CWS in Study n=464**	**CWS < 200 Con.: n=324**	**CWS ≥ 200 Con.: n=140**
Total population (count)	3,037,785	1,134,017	49,340	1,084,677
Population Characteristics (%)				
Latino population	43	46	40	37
Non-Latino People of Color	13	9	8	9
White population	47	45	62	44
Population above poverty[a]	57	54	60	54
Water System Characteristics (%)				
Mean Latino	34	31	26	42
Mean Home Ownership	67	70	72	67
Population served (mean/median)	4,717 / 163	2,444 / 180	152 / 100	7,748 / 2537
Incorporated[b]	10	9	1	29
< 200 Connections	72	70	100	0
Groundwater Alone (GW)[c]	88	92	95	87
GW and surface water[c]	7	4	2	9
Publicly owned[d]	32	32	13	75
Privately owned non-PUC reg.[d]	60	61	80	16
Water Quality Characteristics				
Min-Max; mean (µg As/L)	NA	0-158; 6.0	0-158; 6.2	0-42; 5.7
IQR (µg As/L)	NA	1.4, 6.3	1.4, 6.2	1.4, 7.3
CWS with ≥1 As MCL Viol	44	34	15	19

Con. = service connections; NA=not applicable because not all active sources had arsenic samples; IQR= interquartile range.
[a] Above 200% the poverty level;
[b] A water system that serves a city that is a legally recognized municipal corporation with a charter from the state and governing officials that is incorporated, as opposed to a water system that serves an unincorporated area;.
[c] Reference group=surface water only;
[d] Reference group=privately owned and Public Utility Commission (PUC) regulated or unknown.

Table 4.2. Characteristics of community water systems (CWSs) at three average arsenic levels, 2005-2007, San Joaquin Valley, CA.			
	Average Arsenic Concentration		
CWS Characteristics	**<10 µg/L**	**10-49.9 µg/L**	**≥50 µg/L**
% CWS	84.4	15.5	.01
Mean Population Served (median)	2496 (180)	2277 (200)	127 (64)
% Privately owned, non-PUC Regulated	61	59	100
% < 200 Connections	70	66	100
Range of Mean Arsenic (µg As/L	0-9.9	10.1-41.7	59.5-158
Mean µg As/L (Median)	3 (2)	19 (16)	97 (85)
Mean % of Sources >MCL (IQR)	.7 (0,0)	87 (75, 100)	100 (100,100)
CWSs with arsenic treatment plant	2	4	0
IQR=Interquartile range; PUC=Public Utility Commission			

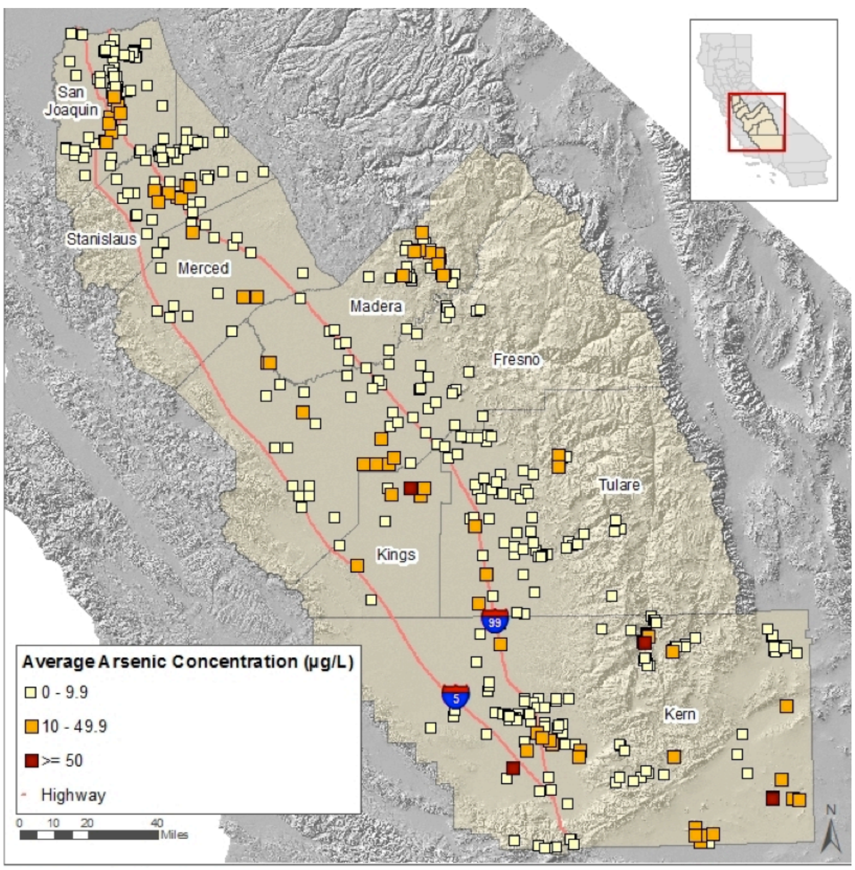

Figure 4.2. Average arsenic concentration of community water systems (CWS b,c) in study sample (n=464), 2005-2007, San Joaquin Valley, California.
[a] Estimate based on average of each point-of-entry source's average concentration.
[b] Data source: CDPH Water Quality Monitoring and PICME databases (CDPH 2008a,b)
[c] Approximate location of CWS are depicted, but not true boundaries. Due to close proximity of some CWS, map partially covers some CWS.

Table 4.3. Demographic profile of potentially exposed population (PEP [a]) at three average arsenic levels, 2005-2007, San Joaquin Valley, CA.			
	Average Arsenic Concentration		
Population Characteristics µg/L	**<10 µg As/L**	**10-49.9 µg As/L**	**≥50 µg As/L**
% Total Population (N=2,948,346)	86.1	13.7	0.2
% Latino (N=1,164,714)	45	49	16
% Non-Latino People of Color (N=389,336)	9	12	8
% Non-Latino White (N=1,394,296)	46	39	76

[a]Per water system, PEP= population count of demographic of interest x (# of sources in one of three arsenic level/total # of sources sampled). PEP displayed in table is equal to sum across all water systems. This value can also be interpreted as the estimated number of people served water at this level.

Statistical Analyses

Compliance Burdens and MCL Violations

Thirty-four CWSs, serving 151,391 people, received at least one arsenic MCL violation during the study period. CWSs serving higher percentages of homeowners had a 67% lower chance of having at least one MCL violation (Table 4.4). CWSs serving higher percentages of people of color had a 260% higher chance of having at least one MCL violation. Difference of means tests and sensitivity analyses were consistent (data not shown). Of the 34 CWSs with at least one arsenic MCL violation, 31 had average arsenic concentrations over 10 µg As/L and 3 had average concentrations of 8, 8.8 and 9.9 µg As/L (data not shown).

Table 4.4. Fisher's exact tests and related Odds Ratio (OR) comparing demographics in community water systems that received at least one MCL violation to those with zero violations, 2005-2007 (n=464).

Variable of Interest	≥ 1 MCL Violation	No MCL Violation	OR (95% CI)	p-value
High % Homeownership	12	269	.33 (.16, .67)	.003
Low % Homeownership	22	161		
High % People of Color	24	207	2.6 (1.2, 5.4)	.01
Low % People of Color	10	223		

[a]Fisher's Exact test compares high and low category for variable of interest, where threshold is determined by median value across all CWS.

Binary Measure of Exposure

CWSs serving higher percentages of homeowners had a 57% lower chance of having average arsenic levels exceed the arsenic MCL (Table 4.5). CWSs serving higher percentages of people of color had a 130% higher chance of having average arsenic levels exceed the MCL. Difference of means tests showed a similar pattern (data not shown). Sensitivity analyses indicated similar results when we considered whether a system had a source with average levels of at least 10 μg As/L, or yearly averages of at least 10 μg As/L (data not shown).

Table 4.5. Fisher's Exact Tests and related Odds Ratio (OR) comparing demographics for community water systems whose average arsenic was above or below the revised MCL, 2005-2007 (n=464).

Variable of Interest	≥ 10 μg As/L	<10 μg As/L	OR (95% CI)	p-value
High % Homeownership	28	233	.43 (.25, .72)	.002
Low % Homeownership	44	159		
High % People of Color	35	162	1.3 (.81, 2.2)	.3
Low % People of Color	37	230		

[a]Fisher's Exact test compares high and low category for variable of interest, where threshold is determined by median value across all CWS.

Absolute Measure of Arsenic Exposure

Results from the multivariate regression model examining the relationship between CWS demographics and absolute arsenic concentrations generally parallel descriptive findings. Unadjusted models had beta coefficients of -0.14 (95% Confidence Interval (CI), -0.34, 0.06) for homeownership, and -0.01 for percentage of people of color (95% CI, -.11, 0.08). Our adjusted model had a beta-coefficient of -0.27 (95% CI, -0.50, -0.05) for homeownership. This suggests that, on average, a 10% decrease in homeownership was associated with a 2.7 μg As/L increase, representing roughly one third the mean arsenic concentration across all CWSs (6.1 μg As/L, see Table 1). The beta coefficient for percentage people of color was -0.02 (95% CI, -0.13, 0.09) for percentage people of color. This suggests that a 10% increase in the percentage of people of color served by a CWS was associated with an increase of .2 μg As/L, though this association is not statistically significant.

Results from our stratified model (Table 4.6) suggest similar, but stronger, trends among smaller systems. Among systems with < 200 connections, the beta coefficient for homeownership was -0.43 (95% CI, -0.84, -0.03). This suggests that, on average, a 10% decrease in homeownership is associated with a 4.3 μg As/L increase, or nearly 70% of the mean arsenic concentration across all CWSs. The beta coefficient for percentage people of color was -0.17 μg As/L (95% CI, -0.36, 0.02), although this result was not significant. In systems with ≥200 connections, the coefficients on percent homeownership and people of color were -0.18 (95% CI, -0.40, 0.02) and 0.03 (95% CI, -0.09, 0.15), respectively; neither of these results are statistically significant. Using this final stratified model to predict expected values, we estimated that arsenic levels in CWSs with 100% home ownership would be on average 3.1 μg As/L lower, compared to CWSs at the mean.

Table 4.6. Regression[†] for factors associated with arsenic concentration (µg As/L) in community water systems (CWS), N=464.

Variable	Model A[a]	Model B[a]	Model C[b]	Model D (<200 Con.)	Model E (≥ 200 Con.)
Constant	20.0 (6.7, 33.3)	11.2 (6.1, 16.4)	9.7 (-11.8, 31.3)	18.2 (-11.9, 49.1)	8.7 (-11.7, 49.1)
% People of Color		-0.01 (-0.11, 0.08)	-0.02 (-.13, 0.09)	-0.17* (-0.36, 0.02)	.03* (-0.09, 0.15)
% Home ownership	-.14 (-0.34, 0.05)		-0.27** (-0.50, -0.05)	-0.43** (-0.84, -0.03)	-.19 (-0.40, 0.02)
Groundwater or combined[c]			11.4*** (7.5, 15.2)	11.5*** (6.1, 16.9)	8.4*** (4.2, 12.6)
Private non-PUC regulated[d]			5.6* (-1.0, 12.2)	8.5** (0.73, 16.3)	1.2 (-5.4, 7.9)
Public[d]			6.9** (0.61, 13.11)	7.5* (-0.76, 15.8)	6.4* (-0.99, 13.8)
< 200 service connections			2.6 (-1.2, 6.5)	na	na
2006[e]			2.8** (0.52, 5.1)	4.4** (0.27, 8.4)	1.8 (-.76, 4.3)
2007[e]			1.2 (-0.51, 2.9)	2.4* (-0.11, 4.9)	.52 (-1.8, 2.9)
Summer/fall			-.27 (-1.9, 1.4)	.43 (-3.1, 4.0)	-.27 (-2.1, 1.5)
Valley[f]			-1.4 (-6.5, 3.7)	6.4 (-2.3, 14.9)	-4.4 (-10.6, 1.8)
Foothills[f]			6.9* (0.32, 13.5)	12.1 (3.9, 20.4)	5.1 (-1.0, 11.3)

[†]Regression with robust standard errors, clustered by CWS. Coefficients represent the estimated difference in mean concentration at the system-level associated with a unit change in the covariate (95% CI); Con.=connections; na=not applicable, as no CWS in this model run contains this factor, or all CWS have this factor. R^2=.083
[a]Unadjusted models, all CWS included
[b]Adjusted model, all CWS included
[c]Surface water is referent category; combined refers to combination of groundwater and surface water sources
[d]Privately owned PUC-regulated CWS as referent category
[e]2005 is referent year
[f]Mountains is referent category * p<.10, ** p<.05, *** p<.01

Discussion

This study addressed distributional and procedural inequities associated with the revised arsenic standard in drinking water in the San Joaquin Valley. We found that communities with lower rates of home ownership and greater proportions of people of color had higher odds of having an MCL violation. We also found a negative association between homeownership rates and arsenic concentrations in drinking water, with a stronger effect among smaller CWSs. These results indicate that communities with fewer economic resources faced a dual burden—they were not only exposed to higher arsenic levels, but were also served by systems more likely to receive an MCL violation.

Nearly 14% of the population in the study sample was potentially exposed to average arsenic levels above the revised standard, highlighting the health risks faced by Valley residents. At the revised level, cancer risks are estimated to be 12 in 10,000 and 23 in 10,000 for bladder cancer among women and men, respectively, and 18 in 10,000 and 14 in 10,000 for lung cancer, among women and men (National Research Council 2001). While we did not find a significant association between race/ethnicity and arsenic levels, a disproportionate number of the population potentially exposed to levels ≥ 10 μg As/L were people of color, indicating that, as a whole, this group may still face disproportionate exposure.

Some limitations in our study are worth noting. There could be potential sources of measurement error in our dependent and independent variables. Under-reporting or under-issuing of violations could impact the count of MCL violations. Sensitivity analyses comparing MCL violations in our final sample to results including all CWSs yielded consistent results. Similarly, sensitivity analyses comparing results using the binary MCL variable to binary measures that used average source and system-level concentrations were similar. Because of this consistent negative relationship between SES and each of these measures, we expect minimal impact on our results. This does not, however, explain why 41 CWSs (out of 72) had average system-level concentrations above the MCL but had no violation recorded; this may be related to selective enforcement, and is worth further investigation.

There may also be some misclassification of points-of-entry into the distribution system. However, numerous sensitivity analyses, including and excluding CWSs with treated and untreated point-of-entry sources yielded consistent regression coefficients for home ownership. While results for estimated exposure and compliance burdens are nearly five years old, we believe that, at a minimum, they capture current trends because unless CWSs have installed treatment plants or are using water from new wells (which is unlikely for small systems), temporal variability of arsenic levels is likely small (Focazio et al. 1999).

There may be errors in our demographic estimates, as we had to use data from the U.S. Census 2000 to approximate demographics between 2005 and 2007. There could also be error in our demographic estimates from: 1) surface intakes/well fields falling in Census block groups not served by the CWS, 2) not all Census block groups served by a CWS having an intake/field located within them, and 3) Latinos in Census data being undercounted due to legal status. For the majority of CWS, however, sources fell within the same Census block groups that overlapped with CWS' service area boundaries (Balazs et al. 2011).

Our results are consistent with previous findings that CWSs with higher arsenic levels serve customers with lower income levels (Stone et al. 2007). Our results differ somewhat from previous research (Cory and Rahman 2009) that found that while percent Latino was positively associated with the likelihood of exceeding the arsenic MCL, so was high SES. This could be due to: differences in trends across states, our additional measurements of exposure and compliance, or our focus on CWSs rather than all public water systems.

Our results can be understood in the broader context of system-level capacity. Smaller water systems often lack the economies of scale and resource-base to ensure the technical, managerial and financial (TMF) capacity to reduce contaminant levels (Committee on Small Water Systems 1997; Shanaghan and Bielanski 2003). They may be less able to install treatment, apply for funding, or drill new wells. The socioeconomic status of residents directly influences TMF capacity, because it affects the ability of a water system to leverage internal (e.g., rate increases) or external (e.g., loans) resources (Committee on Small Water Systems 1997).

Thus, in our study, CWSs with lower SES may have been less able to support adequate TMF or to ensure compliance with the revised arsenic standard by 2007. That four of the six CWSs with treatment had more than 200 connections suggests that larger CWSs (i.e., with more resources and greater economies of scale) were able to comply more quickly with the revised standard, a result supported by previous research, as well (Shanaghan and Bielanski 2003). Furthermore, that the majority of CWSs with average arsenic concentrations over the revised standard were small and had a high fraction of their wells with high arsenic levels indicates that these systems had few alternative sources of clean drinking water to begin with, making short-term solutions unattainable.

Our results highlight the need to consider not only present exposure, but also the future mitigation potential of impacted water systems and the households they serve. Poorer CWSs faced the greatest exposure and compliance burdens, but these systems may be the least equipped to comply with EPA standards for at least three reasons. First, these CWSs are often less able to develop long-term plans to reduce contamination. For example, some low SES communities in the Valley have secured funding to upgrade their infrastructure, but their plans have failed to include steps to enter into compliance with the new standard (Boyles 2005). Second, low SES CWSs may be less able to apply for funding. By 2010, 13 of the 72 CWSs in our study with medium and high arsenic levels were not listed as having applied to the State Revolving Fund to help pay for mitigation options (CDPH (California Department of Public Health) 2011). These CWSs were mainly small (<200 connections) and had lower rates of home ownership (60% vs. 65%, $p<.10$) compared to CWSs that were listed. Current funding sources, such as State Revolving Fund, may further disadvantage small CWSs' efforts to mitigate arsenic exposures and comply with the standard because they require adequate TMF capacity. Finally, even with funding secured, low-SES water systems with low TMF capacity may be unable to maintain compliance. For example, some CWSs have installed arsenic treatment technologies, only to be forced to shut the plants down because they could not pay for ongoing treatment costs (Fresno County Grand Jury 2008).

This cycle of low SES, low TMF and low compliance ability of CWSs not only impacts mitigation potential and exposure levels, it can also result in significant economic burdens for poorer households. In general, CWSs that are able to mitigate arsenic contamination will incur costs that are passed along to customers. Low-income residents find it hard to pay these higher rates, or may oppose mitigation efforts because of the impact on household budgets (Beecher 2003). If a CWS cannot mitigate exposure, households may be forced to cope by buying bottled water, creating an additional economic burden. However, low-income residents may forgo such exposure-reduction measures, or only partially implement them (Moore et al. 2011). In these cases, if a CWS remains in continuous non-compliance, chronic arsenic exposure risks will be prolonged.

Overall, our findings suggest that environmental justice concerns related to arsenic are simultaneously distributional and procedural. Using a "joint burden" approach, we capture the extent of exposure and compliance burdens in the San Joaquin Valley, and foreshadow future challenges that residents and CWSs will likely face. Our work also highlights the need to better address how water systems serving low-SES residents can apply for and obtain resources, particularly if current funding criteria are tied to the technical, managerial and financial capacity of CWSs. Ultimately, regional solutions that consolidate smaller CWSs serving economically disadvantaged communities within larger CWSs may be the best approach to addressing these disparities. In the interim, however, small water systems serving low SES residents will need enhanced funding and technical support to reduce community-level arsenic exposures.

CHAPTER 5
The Drinking Water Vulnerability Framework

Isn't the issue of contaminated water *just* an issue of economies of scale,
where small systems simply face the biggest problems?

In talking about environmental injustices and contaminated drinking water
are you implying that someone is deliberately polluting people's water?

If there is no statistical correlation between race, class,
and water quality doesn't that mean there is no injustice?

Paraphrased questions commonly encountered throughout dissertation research

Overview

While *Chapters 3-4* provide a quantitative assessment of the relationship between drinking water quality and community-level demographics, this chapter traces the processes that shape disparities in drinking water quality at the regional, community and household level. Using data from participant observation and interviews, as well as secondary data on water quality, I develop a "drinking water vulnerability framework." This framework builds on social epidemiology and socio-ecological frameworks that address how multi-level environmental factors (i.e., natural, built and social) drive health disparities. Using the San Joaquin Valley as a case region, I trace the impact of the natural, built and social environments on both exposure and coping capacity. I argue that the interaction of these factors creates a "composite burden" that is composed of exposure and coping costs and shapes the differential ability of water systems and households to cope with drinking water contamination. More specifically, the framework uncovers how in conjunction with a baseline of contaminated source water, a series of historical planning policies have jointly shaped access to water supplies as well as to physical and financial resources for communities. These forces, alongside regulatory failures, a lack of community resources to successfully address contamination, and political disenfranchisement of local residents help explain the origins of social inequities in drinking water quality. That these same forces also influence coping capacities and lead to partial protection, at best, serves to further exacerbate the impacts of drinking water contamination and lead to its persistence. In sum, the composite burden explains the origins and persistence of social disparities in exposure, and defines what I term "drinking water vulnerability".

Background

The unincorporated community of Tooleville, California is located at the eastern edge of Tulare County's valley floor, at the foot of the rolling Sierra foothills that are covered with orange groves and dotted with small enclaves of residents. Historically a farm-worker community, Tooleville's demographic make-up has evolved with time. During the Dust Bowl era, several Oklahoma families first settled it. Today, the roughly 70 households living in Tooleville are predominantly Latino, farm-working families, with a median annual household income of $16,000[11]. Residents pride themselves on the beauty of their natural surroundings and their high rates of homeownership. Ms. Jimenez[12] still remembers the day her father, a farm-worker, purchased a home in Tooleville. "I was so proud that we owned a house." She still lives there and is passionate about staying in her community, despite the challenges Tooleville faces.

Like most small communities in the Valley, Tooleville residents rely on groundwater as their drinking water source. This reliance might be acceptable, except that since 1997, Tooleville's one well has exceeded the maximum contaminant level (MCL) for nitrate at least seven times. At these levels, infants are at risk of methemoglobinemia that can cause death, and women are at risk of a number of adverse reproductive effects. In some years, the total coliform rule has also been violated. Residents could boil the water to kill bacteria, but this would only further concentrate the nitrate.

These present day challenges represent but one layer of barriers to clean water. "Why can't we use the canal water instead of contaminated well water?" residents ask at community meetings. Despite the fact that just at the end of Tooleville's two roads runs the Friant-Kern Canal (one of the largest canals in the state of California, channeling Sierra snowmelt primarily to agriculture), Tooleville does not have legal access to this water source. By and large, farmers hold the surface water rights to the Canal, and even if Tooleville held legal rights the cost of treating surface water would be too expensive for this small community. Not only are water rights limited, historical planning processes have limited the financial and infrastructure resources available to the community. Until recently, Tulare County's General Plan listed Tooleville as one of the 15 communities for which public resources, including water infrastructure, should be potentially withheld.

Solutions have been hard to come by. Attempts to drill new wells have yielded poor results—the water all around the community is high in nitrates. This has left Tooleville with a persistent compliance and exposure burden, prolonging risks from exposure and household coping costs. Even coping mechanisms such as purchasing bottled water is only partially protective. Most residents have drunk the contaminated well water at some point, and still use it for cooking.

Regional solutions, such as connecting to the nearby city of Exeter have also been hard to achieve. For several years, Tooleville residents and county officials have hoped that Tooleville could consolidate with the City of Exeter that is less than two miles away and has more wells and cleaner water (Figure 5.1). Such consolidation would involve building a pipeline to supply Tooleville with some of Exeter's water. But the city has been slow to respond and has been more interested in expanding its spheres of influence in other directions. Residents believe this is intentional and discriminatory. As a low-income neighborhood, the community has little tax base to offer the city, they conjectured. The city cited prevailing wages as a barrier to consolidation, for which Exeter has since been exempted (see Senate Bill X29 and Senate Bill 110). But still, consolidation efforts stalled. In 2009, the California Department of Public Health stepped in, and has since been pressuring Exeter to connect to Tooleville. In the meantime, however, residents continue to rely on their one contaminated well, and pay twice for water—once for their utility bill, and once for bottled or vended water.

[11] By comparison, the annual median household income in California was $47,493 in 2000 (U.S. Census 2000b)
[12] Pseudonym used to protect privacy of this resident.

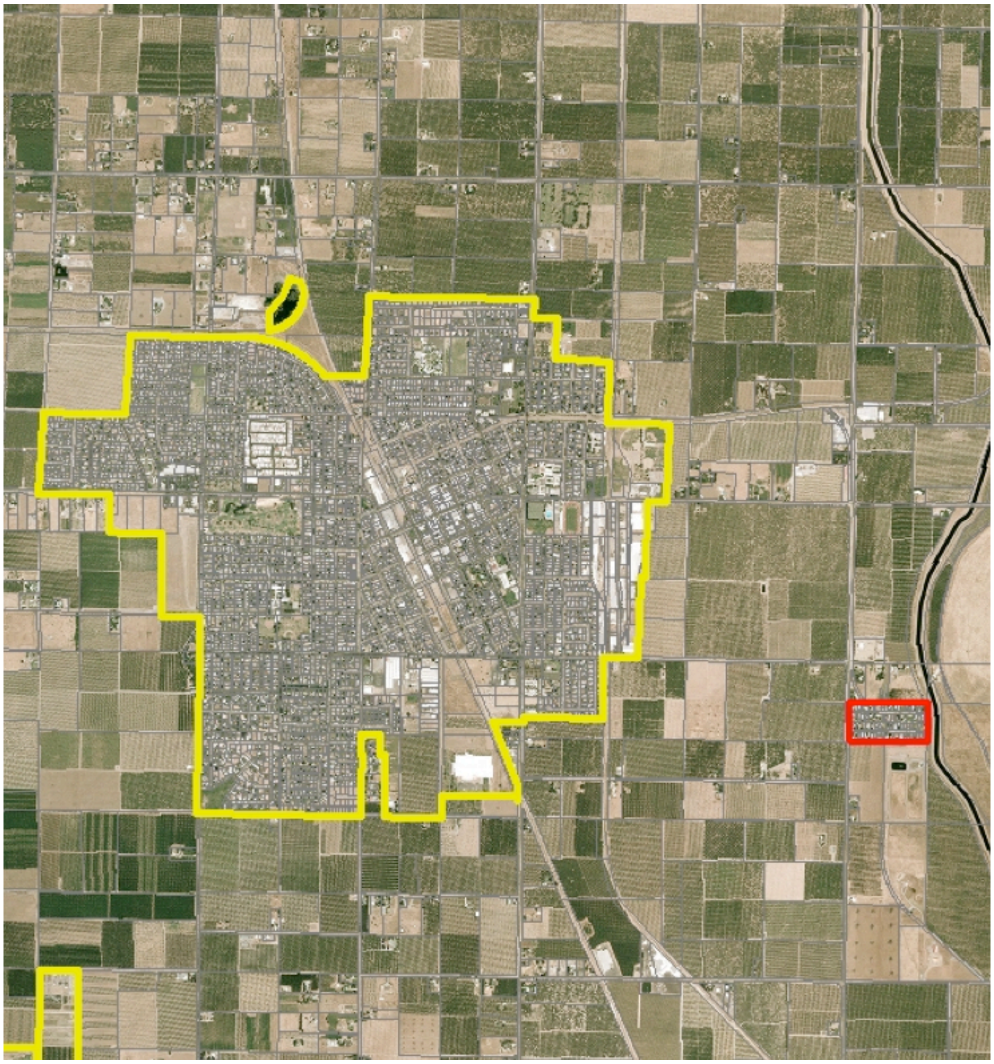

Figure 5.1. Aerial map of the City of Exeter (outlined in yellow) and Tooleville (outlined in red), less than two miles away. The Friant-Kern canal passes to the east of Tooleville.

The story of Tooleville is emblematic of what dozens of small, rural communities in California's San Joaquin Valley face—a multiplicity of social, built and natural factors that create a composite and persistent drinking water burden that threatens health and pocketbooks and may be exacerbated by race and socioeconomic class. More broadly, the story of Tooleville allows us to engage with the three questions noted at the beginning of this chapter. Tooleville highlights that that while small system size can make a system physically vulnerable (e.g., relying on only one well), a range of other factors also impact exposure and coping capacity. Second, Tooleville's story underscores that there is not just one actor whose actions can be credited with the single effect of "causing" drinking water pollution. In fact, multiple factors and actors are at play, and pollution itself is not the only factor to consider; coping capacity is equally of concern. Third, the composite burden—of exposure and coping costs—can create environmental injustices, regardless of whether statistical correlations prove a "significant" association between poor water quality and demographics.

Introduction

Drawing on this understanding of Tooleville, and with the three aforementioned questions as an echoing backdrop, this paper presents a drinking water vulnerability framework that examines the structural determinants of social disparities in drinking water quality. In doing so, this chapter builds on a growing body of literature that has sought to uncover how social processes intersect with environmental contamination and how race, class and related vulnerability affect health risks and outcomes (deFur et al. 2007; Gee and Payne-Sturges 2004; Sexton and Adgate 1999; Sexton et al. 1993). But while existing frameworks have unraveled why disparities in environmental exposure and health exist, this literature has largely left drinking water contamination unexplored. Given growing attention of disparities in access to clean water in marginalized U.S. communities, this chapter seeks to fill a critical understanding of how and why social disparities may exist in relation to drinking water, what the health and regulatory implications are, and how policies can better address related challenges. Thus the drinking water vulnerability framework expands existing social epidemiology frameworks, by presenting a unified discussion of the role of structural determinants in impacting exposure to drinking water.

This focus has also been largely lacking in the U.S. water policy literature. For example, the U.S. water policy literature has identified weaknesses in the regulatory system governing drinking water (i.e. Safe Drinking Water Act) and the vulnerabilities of small water systems (Committee on Small Water Systems 1997). But it has only loosely explored the role of structural factors such as community resources and external political linkages (Beecher 2003) in creating safe and affordable drinking water supplies.

Some environmental justice and geography studies offer more detailed insights regarding the structural problems associated with drinking water contamination and service provision. For example, researchers have examined the potential for environmental injustices due to selective enforcement (Cory and Rahman 2009), non-compliance with federal drinking water standards (Guerrero-Preston et al. 2008; Rahman et al. 2009) and inequities in access to Clean Water Act funding (Imperial 1999). Others have explored the extent to which rate regulation, cost of service extension, low ability to pay and weak political influence drive service provision (Olmstead 2004). Still other studies have traced the impact of inequitable municipal decision-making on provision of water services in under-served communities (Marsh et al. 2010; Troesken 2002; Wilson et al. 2008a; Wilson et al. 2010). Finally, government reports have further highlighted the extent of lack of monitoring and reporting by both water systems and regulators (U.S. General Accounting Office 1990, 1999, 2011). But while these studies provide greater understandings of the structural conditions shaping drinking water problems, overall they have focused on one aspect of the problem at a time and still do not present a comprehensive view of the multiple factors impacting exposure and mitigation capacity.

Building on these frameworks and gaps, the drinking water vulnerability framework: a) examines how multi-level social, natural and built environmental factors shape vulnerability to contaminant exposure, b) explores how these same factors also shape the capacity to cope with exposure, c) posits that community and household-level coping mechanisms present a feedback loop through which vulnerability is exacerbated, d) argues that the "composite burden" of exposure and inability to cope leads to the persistence of exposure, and e) explains why social disparities in exposure may exist and persist.

Ultimately, the framework uncovers how alongside a baseline of contaminated source water, a series of historical planning policies have jointly shaped access to water supplies as well as to and physical and financial resources for communities. These forces, in conjunction with regulatory failures, a lack of community resources to successfully address contamination, and political disenfranchisement of local residents help explain the origins of social inequities in drinking water quality. That these same forces also influence coping capacities and lead to partial protection, at best, serves to further exacerbate the impacts of drinking water contamination and lead to its persistence.

Development of the Framework: Empirical and Theoretical Bases

The drinking water vulnerability framework was developed using field data collected using a mixture of quantitative and qualitative methods. Participant observation and interviews are based on five years of interactions with residents, regulators and grassroots organizations in the southern San Joaquin Valley. During this period, I attended numerous community and water board meetings in unincorporated communities, many of them in Tulare County, as well as county and regional water meetings, water conferences, environmental justice alliance meetings, and testimonial hearings of community members to government and U.N. officials. I interviewed (multiple times when necessary) 12 California Department of Public Health regulators, and members of five community-based, non-governmental organizations (NGOs). This research occurred in collaboration with the Community Water Center, a community-based, drinking water NGO. As such, ongoing observations of the Center's organizing and advocacy efforts, and numerous informal discussions with the Center's staff and community base also informed the development of the framework.

This primary field data was complemented by quantitative analyses of drinking water quality and SDWA violations in community water systems across the entire San Joaquin Valley. These analyses used water quality monitoring datasets (i.e. Water Quality Monitoring database) collected by water systems and maintained by the CDPH, and monitoring and MCL violation data entered by state and county regulators into the Permits, Inspections, Compliance, Monitoring and Enforcement (PICME) database (CDPH 2008a).

I also conducted a media analysis of three key communities, Alpaugh, Lanare and Tooleville. The media review used three online news databases, Lexis Nexis Academic, NewsBank and Google News Archives. Search criteria included the name of each community and the words "water" or "drinking water" from 1999 to 2010.

Conceptually and theoretically, the framework draws on three different literatures: social epidemiology, environmental health and U.S. water policy. Insights from both social epidemiology and environmental health support a framework that emphasizes the interaction of multi-level factors and takes a socio-ecological approach. For example, the framework takes a "Multi-level dynamic" approach, where multiple scales and processes interact (sometimes over time) and create links between exposure and susceptibility (Krieger 2001). At the same time, the framework's underlying socio-ecological perspective builds on the premise that disparities in exposure derive not only from individual-level factors but also from multiple levels and multiple environments (e.g. social, natural and built) (Gee and Payne-Sturges 2004).

More specifically, the drinking water vulnerability framework expands existing socio-ecological frameworks in three ways. First, it explicitly focuses on drinking water. Second, it explicates the processes through which vulnerability intersects exposure and response (i.e., coping) pathways. Third, it posits that household and community-level coping mechanisms create a feedback loop through which vulnerability to exposure to drinking water contaminants is exacerbated.

These premises build on the theoretical core of three related environmental health and/or social epidemiology frameworks. Sexton et al (1993) expands the traditional exposure-disease paradigm (Lioy 1990) used in environmental health by positing that differential health risks may be associated with race and socioeconomic class due to exposure (e.g. proximity to source) and susceptibility-related (e.g. gender) attributes. Gee and Payne-Struges (2004) further refine Sexton et al's work by incorporating a multi-level perspective and exploring how vulnerability intersects the exposure-disease paradigm. deFur et al (2007) complement this approach by showing that vulnerability can impact exposure pathways between environmental factors and receptors (i.e. individual, community or population) and response pathways between receptors and outcomes. Fundamentally, these existing frameworks, and the one presented in this paper represent an evolution from the traditional exposure-disease paradigm (Lioy 1990; National Research Council 1991) that focuses solely on how sources of contamination determine concentrations of contaminants, which lead to exposure, ingested doses and finally to health impacts.

Using the National Environmental Justice Advisory Committee's view that vulnerability is composed of susceptibility, exposure, preparedness, and responsiveness (deFur et al. 2007; NEJAC 2004), the framework uncovers how multiple social, built and natural processes impact contaminant exposure and create susceptibility to it. In addition, the framework uses deFur et al's (2007) concept that differential responsiveness to exposure is shaped by the coping capacity and resources available to groups.

More generally, the framework's multi-level structure finds support in the recent literature on safe and sustainable water provision in the U.S. and methodological critiques of purely statistical analyses of environmental justice studies. Wilson et al (2008) for example, argue that the classic exposure-disease paradigm must address the role that infrastructure, laws (i.e., legal epidemiology) and regulation plays in producing adverse health outcomes. Tracing the on-the-ground impact of historical, structural and institutional processes, Marsh et al (2010) examine how municipalities provide or deny access to basic services, such as water provision, by exercising their police powers to determine which areas to annex or exclude from their city boundaries. By focusing on socio-political processes and the role of the built environment, these studies reflect the call (Pulido 1996; Pulido et al. 1996) for more historically-informed studies that trace the production of health inequities or environmental inequities.

The drinking water vulnerability framework also incorporates key factors affecting water system sustainability that are discussed in the U.S. water policy arena. For example, the 1996 SDWA Amendments have highlighted the importance of Technical, Managerial and Financial (TMF) capacity of water systems (Shanaghan and Bielanski 2003), and note how smaller systems that find it difficult to comply with the Act. Other studies recognize that water system sustainability depends on the larger socioeconomic and resource contexts of communities, including community income, population dynamics, availability of water resources, and regulatory institutions (Beecher 2003).

Drinking Water Vulnerability Framework

The drinking water vulnerability framework shows how multi-level driving factors affect coping and exposure. At the top of Figure 5.2, factors in the natural, built and social environments act within and across at least three distinct but mutually interacting levels: the regional, the community, and the household level. The *natural environment* includes ecological factors, such as soil types, hydrology and climate. The *built environment* represents human-modified spaces and infrastructures, such as agriculture and water system infrastructure. The *social environment* refers to institutions and characteristics of groups, including historical and present-day planning policies, governance institutions and community demographics.

The *region* could refer to a basin, the agricultural region of the San Joaquin Valley, or a county. For this framework, the *community* is defined by the physical service area of a community water system (CWS) that serves water year-round to at least 15 residential units or 25 people (e.g. municipal systems, mobile home parks) (U.S. EPA 2010b). Community-level factors can refer to characteristics of two key actors—the water system, composed of its physical infrastructure and governance institutions, and the aggregate make-up of community residents, the customers served and living within the water system service area. Thus community-level characteristics may be unique to the community (e.g. system-level infrastructure), or they may derive from the aggregation of household-level characteristics (e.g. percent homeownership). The framework includes characteristics of both types of actors since each influences coping mechanisms in unique ways (e.g., the water board can derive funding for a new well versus the household can install a point-of-use treatment device). The third level is the household. Because drinking water is often served at the household level, the framework includes factors describing the household, rather than individuals, though exposure ultimately occurs at the individual-level (as indicated by the addition of this level at the bottom of Figure 5.2). The framework does not expressly include private well owners, who are not connected to a public water supply.

Arrows interconnecting the three environments represent the cross-factor interactions that can occur. Dotted lines of each level indicate that multi-level interactions within each environment can also occur. Arrows pointing away from the driving factors show that these multi-level factors can influence source water quality, coping mechanisms and exposure. Table 5.1 lists specific examples of driving factors that I observed in the San Joaquin Valley or learned from the literature.

The framework then adds a new component to the classic exposure-disease paradigm (shown in un-highlighted boxes at the bottom): the role of coping mechanisms in influencing exposure. Exposure to drinking water contaminants in excess of SDWA standards necessitates mitigation and typically requires a CWS to implement a solution. However, when a CWS is incapable of doing so, the household must respond. Thus coping can occur at multiple levels. The result of coping mechanisms can range from zero or partial mitigation of contamination, to full mitigation that reduces exposure completely. The degree to which coping mechanisms are successful directly influences exposure. But the degree of exposure also dictates the need for coping. Thus a bi-directional arrow connects these two factors. Coping mechanisms result in related costs that may be incurred at one level (e.g. CWS), and passed down to another (e.g. household). These costs can also influence coping capacity, hence a second set of bi-directional arrows is included. These feedback cycles and resulting exposure and coping costs define a "composite drinking water burden". Whereas the driving factors include characteristics of actors (e.g. water board, household characteristics, etc) or describe contextual factors (e.g. policies or soil-type), the multiple levels highlighted under coping and exposure refer to the *actors* themselves, as these are the receptor sites that have the agency to cope.

Though a range of multi-level actors (e.g. region, community, household) can cope, this chapter focuses on the community and the household as the primary site of exposure impacts. The community level is relevant for policy purposes, since drinking water regulations are applied to and monitored in community water systems. In addition, while regional solutions are ultimately necessary, the impact of any regional efforts will be felt at the community level.

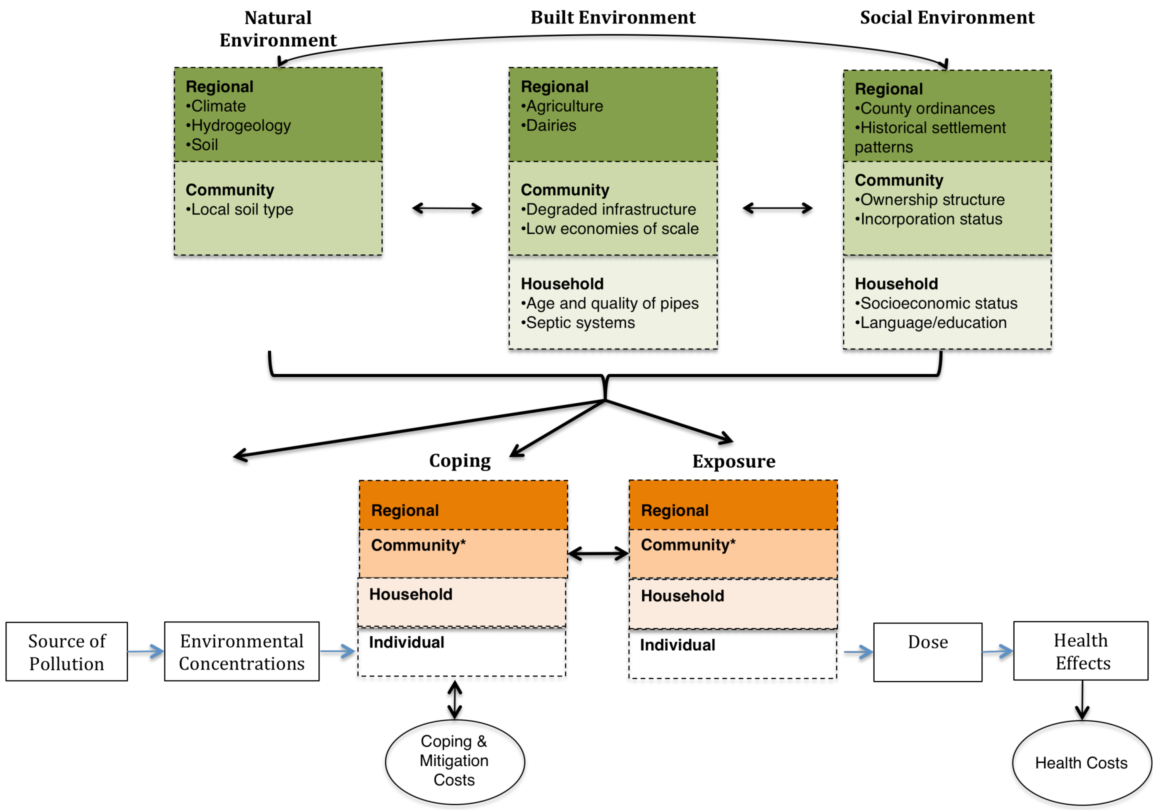

Figure 5.2. Drinking water vulnerability framework. The framework describes how multi-level factors impact exposure and coping capacity, emphasizes the feedback between coping capacity and exposure, and highlights the related costs that ensue. Dotted lines indicate that factors within three levels can interact. Colored boxes emphasize the focus of this framework, non-colored flow boxes capture traditional exposure-disease paradigm. Bullets provide specific examples of driving factors.

Multi-Level Driving Factors and Exposure

To explore how multi-level environmental factors shape vulnerability to and social disparities in exposure, this section first examines how natural environmental factors affect source water quality. Next, this section uncovers how these environmental factors interact with the built environment to create physical vulnerabilities to exposure at the community level. The framework highlights how in combination with these natural and built factors, social factors exacerbate vulnerability to exposure by defining access to water sources and determining the extent of physical, financial and regulatory resources that a community can attain.

To begin, factors from the natural and built environment, such as hydrogeology and land use practices shape source water quality, which in turn, partially defines baseline contaminant levels. For example, the climate and soil in Tulare County's eastern side creates favorable growing conditions for citrus trees that require high amounts of nitrate fertilizer application. Since the water table in this region is shallow (Figure 5.3), nitrate fertilizers can leach more rapidly into the water table and have a shorter travel time into well water (Nash 2006)[13]. As a result, communities, such as Tooleville, that are located along eastern edge of Tulare County's foothills have some of the highest nitrate levels in the Valley (Dubrovsky et al. 1998). In the western side of the Valley, the Corcoran clay layer plays a converse role. This impermeable layer requires that CWSs relying on groundwater drill deeper wells to obtain water (Galloway and Riley 2006). But at these deeper levels, the probability of drawing naturally occurring, arsenic-laden water increases[14], as in the case of Alpaugh.

Table 5.1. Driving factors that operate at regional, community and household levels, with examples listed in italics. Factors may act or occur uniquely within a level, or act similarly across levels.

Level	**Key Factors**		
	Natural Environment	**Built Environment**	**Social Environment**
Regional	• Climate • Hydrogeology: *soil type, depth-to-water*	• Land use practices: *cropping patterns, dairies*	• Planning policies: *County General Plans, ordinances, annexation procedures* • Regulatory policies: *implementation of SDWA* • Socioeconomic forces shaping development: *labor camps and unincorporated communities*
Community	• Local hydrogeology: *soil type, depth-to-water*	• Physical vulnerability of infrastructure vulnerability: *age and quality of distribution system* • Size of system: *# of wells or service connections* • Proximity to sources of contamination: *to dairies or agriculture* • Water type used: *groundwater, surface water, purchased water*	• Social vulnerability of community: *linguistic isolation, education levels, racial composition, community resources, SES* • Political clout of community • Technical, Managerial and Financial Capacity of system • Governance of water system: *ownership type, regulatory capacity, access to funding, allowance of civic engagement*
Household	N/A	• Physical vulnerability of home infrastructure vulnerability: *age & quality of pipes* • Proximity to sources of contamination near house: *septic*	• Social vulnerability of household: *SES, civic engagement capacity, economic capacity to mitigate exposure, capacity to mitigate individual exposure*

13 As indicated in Figure 5.2, parts of the water table are also shallow along the western edge of the Valley, but many of the communities in that area rely on surface water (e.g. Westlands Irrigation District).

14 Due to confidentiality measures associated with the relevant un-published document, I was unable to provide a citation of this source. Interested readers can email me for further information, and published versions of this chapter will cite the source appropriately.

But the impact of the aforementioned factors on source water quality, and eventually on exposure, cannot be understood without unraveling the interactive role of the social processes at play. These social factors have defined access to water resources, physical infrastructure of water systems and the level of resources and regulatory oversight available to communities. For example, the historical allocation of water rights and development of water resources in the Valley have played a direct role in determining drinking water quality. Historical government financing of large-scale water infrastructure projects enabled the storage and conveyance of vast quantities of Sierra snowmelt and Delta waters to farmlands. This enabled farmers to receive nearly unlimited surface water rights for their agriculture (see, for example (Reisner 1986)), but left 95% of the Valley's residents to rely on groundwater as the primary drinking water source (CDPH 2008a).

This might not matter, were it not for agriculture's contamination of groundwater due to chemical runoff from pesticides and fertilizers percolating into rivers and aquifers (Dubrovsky et al. 1998). But instead, due to the high rates of nitrogenous fertilizer use (Dubrovsky et al. 2010; Harter 2009), for example, the region contributes the largest share of California's nitrate MCL violations. In 2007, 75% of all of California's nitrate violations occurred in the San Joaquin Valley (CDPH 2008a).

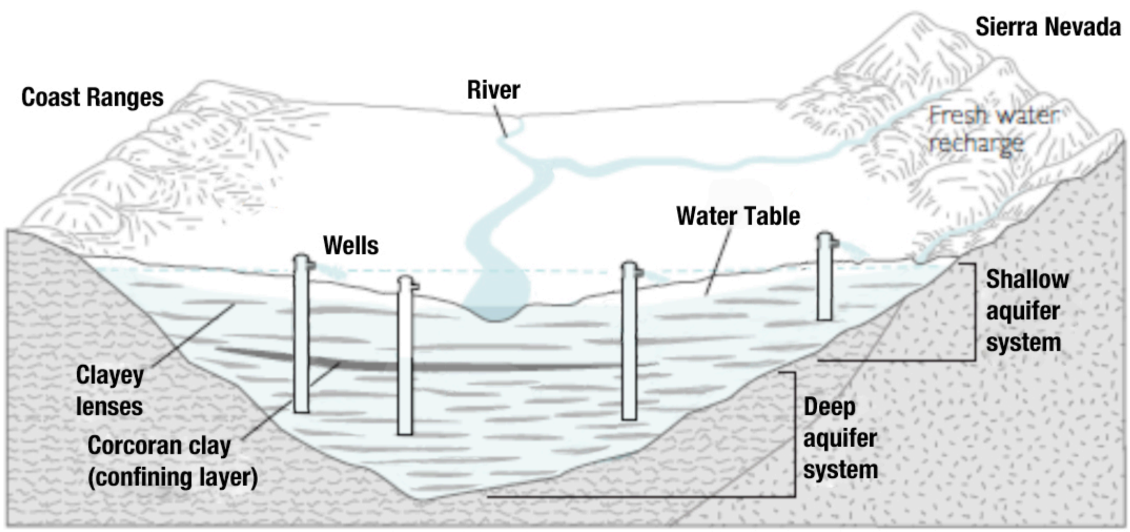

Figure 5.3. Cross-section of the Valley. Corcoran Clay layer on the left (west-side) and the shallower aquifers on the right (the eastside). Figure adapted from: (Galloway and Riley 2006).

Historical development patterns of rural communities have further increased vulnerability to exposure. In many unincorporated communities, the infrastructure started off poor. For example, in the unincorporated community of Plainview, engineers in the early 2000s found that oil pipes had been used in at least half of the water distribution network (Wilber 2003)(Figure 5.4). This reflects poor investment in these communities.

In other cases, regional planning policies further exacerbated vulnerability to exposure by explicitly depriving communities of adequate drinking water resources. For example, Section 2.D.3 of the 1971 Tulare County General Plan reads, "Public commitments to communities with little or no authentic future should be carefully examined before final action is initiated. These non-viable communities would, as a consequence of withholding major public facilities such as sewer and water systems, enter a process of long term, natural decline as residents depart for improved opportunities in nearby communities". Among the fifteen communities listed were Allensworth, Alpaugh, Lemon Cove, Plainview, Seville and Tooleville. Many of these communities were either historically labor camps, or are currently unincorporated communities, lacking their own tax-base and municipal representation. Such a policy exemplifies institutionalized instances of environmental/infrastructural 'redlining' where certain communities are intentionally deprived of adequate resources to ensure clean drinking water provision.

Figure 5.4. Old oil pipe found to be supplying water in the community of Plainview. Photo Credit: Erin Lubin.

These social processes suggest a direct relationship with the quality of the built environment in some communities, and can therefore not be extricated from the role that the built environment plays in increasing vulnerability to exposure. Dilapidated infrastructure can cause cross-contamination from leaky pipes, and requires costly upgrades. For many years, for example, the community of Seville (one of the fifteen communities listed in the General Plan) was unable to secure a grant to completely upgrade the system's pipe network. Thus community residents (rather than the water system operator) undertook partial upgrades to the distribution system's piping infrastructure (Figure 5.4). But because both the new and old infrastructure runs through open ditches, additional exposure pathways to contamination still exists, such as animal waste deriving from animals walking through the ditches (personal communication). Household-level infrastructure further creates exposure vulnerability. If homes are on septic systems, and the system has low water pressure, contamination from the septic system could leak into the system's distribution network, or pipes that run to the home.

Regulatory failures can interact with these factors, depriving communities of adequate regulatory protection, creating additional vulnerability to exposure. In interviews, county and state regulators noted that limited by funding and staff time they were forced to prioritize which contaminants to regulate, and which regulations to enforce. Some of this priortization derives from the Safe Drinking Water Act itself. Due to their designation as Tier 1[15] contaminants, for example, MCL violations of total coliform or nitrate violations, are explicitly prioritized over a system's failure to monitor contaminants. But in the process of prioritizing health risks due to short-term exposure (e.g., gastrointestinal illness in the case of nitrate, or "blue baby syndrome" in the case of nitrate), unforeseen exposure risks can occur.

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15 Tier 1 notification is required for: 1) total coliform MCL violation (including failure to take a repeat sample within 24 hours), 2) Violation of the MCL for nitrate/nitrite (including failure to take a repeat sample within 24 hours), 3) violation of a treatment technique that results in exceedance of maximum allowable turbidity level (unless the water system consults with CDPH within 24 hours and receives a waiver), 4) occurrence of a waterborne microbial disease outbreak or natural disaster that disrupts water supply, 5) other violation or occurrence of a contaminant that has potential for adverse effects due to short-term exposure, 6) violation of perchlorate MCL (California Code of Regulations 2008).

Figure 5.5. Pipe infrastructure in Seville. Before the community replaced the corroding pipes (on right) with a new PVC pipe (white pipe on left) distribution water quality was at risk of contamination. Even the upgraded pipes, however, are vulnerable to cross-contamination given that it runs in an open ditch. Photo credit: Carolina Balazs.

As an example, in 2007, Fresno County returned primacy of water systems with fewer than 200 connections to state-level regulators because county officials did not have the capacity to adequately implement the SDWA. Upon the take-over, state officials found that many of the CWSs had failed to monitor for several years, but had not been given monitoring violations by county regulators. What's more, state regulators estimated that many systems had been out of compliance for many years, though few MCL violations were recorded. The failures were thus multiple: lacking water quality monitoring data, county regulators had been unable to issue MCL violations, with no notices of MCL violations residents had lacked information on whether they faced exposure to harmful levels of contaminants.

Multi-level Driving Factors and Coping/Solutions

Factors Influencing Success of Community-Level Coping Mechanisms

If coping and mitigation strategies could adequately address drinking water contamination, vulnerability to exposure could be minimized. However, as the drinking water vulnerability framework highlights, multi-level factors also impede successful coping strategies at both the community and household level, thereby exacerbating vulnerability to exposure. In particular, physical vulnerability and inadequate financial and management resources at the community level, failures of the regulatory system to ensure timely and adequate information on near-term coping options, inadequate funding mechanisms at the regional level, and disenfranchisement of households combine to impact the success of coping mechanisms.

Similar to the baseline effects of the natural and built environment on source water quality, physical vulnerability of the water system—defined by the number of sources or service connections—can define initial baseline conditions that define near-term coping options. If a system has only one well, this means there are no immediate sources with which to blend water to reduce contaminant levels, nor any alternative water source to use to reduce contamination. Thus, in small systems, if one well exceeds a MCL this not only creates an immediate exposure impact, it also acts as a barrier to mitigating contamination, and prolongs potential exposure.

The community of Alpaugh is an example. For several years, its primary well exceeded the MCL for arsenic, and its back-up well was not consistently functioning. Lacking an immediate alternative water source, residents were forced to use water that exceeded the federal standard while the system sought to obtain funds for drilling a new well. Even today, Alpaugh's water supplies exceed the arsenic MCL, but residents still use the contaminated well water as they await a solution.

In conjunction with the physical vulnerability of small systems, the social vulnerability of residents can partly shape a system's TMF capacity, which ultimately shapes the ability of the water board to adequately plan for a sustainable and safe water supply (Shanaghan and Bielanski 2003). Social vulnerability can be defined in part by the racial/ethnic composition and SES of residents in a community. In systems with low-SES residents, for example, the board may be less qualified because it is difficult to hire and retain capable staff (Committee on Small Water Systems 1997). Poorly trained personnel may not know how to test the water contaminants, or be up-to-speed on funding and monitoring requirements. With few resources, the system has a hard time recruiting more experienced operators, and those that do well tend to move on to larger, more high-paying systems. In interviews, regulators noted that it is nearly impossible for these small communities to have the technical knowledge of how to run and operate a water system.

This can be seen in the community of Lanare, in Fresno County. In 2005, with arsenic levels exceeding the MCL, the Lanare Community Services District worked with an engineering firm to plan a solution to arsenic contamination. The firm recommended that Lanare install a treatment plant that would cost $1.3 million. Having secured grant money from a Community Block Grant, in July 2006 overjoyed residents celebrated the installation of a the treatment plant (Nolen 2007). Six months later, however, the plant was closed down. At first glance, the problem appeared to be financial—the water district was unable to collect sufficient revenue from residents to cover the operating expenses of the plant. But a Fresno County Grand Jury investigation found that the water system was bankrupt and suffered from poor management (Collins 2008). As the Grand Jury noted: "Because of mismanagement, unacceptable arsenic levels, and the absence of any other water source, the district is in crisis" (Fresno County Grand Jury 2008).

Funding mechanisms for developing new water sources or installing treatment facilities could help, but often further exacerbate the lack of immediate solutions and help explain social disparities in coping mechanisms. For example, recognizing the role of TMF capacity in determining compliance with the SDWA, Congress revised the 1996 SDWA Amendments to include the implementation of capacity development programs, especially for small water systems (Shanaghan and Bielanski 2003). But, at least in California, TMF capacity is still required for water systems to be eligible for some state funding of drinking water improvement projects (California Department of Public Health 2009a, b). Similarly, the American Recovery and Reinvestment Bill of 2009 set aside approximately $160 million to fund drinking water infrastructure through California's Safe Drinking Water State Revolving Fund. Among its provisions, however, it earmarked stimulus money for "high priority" drinking water projects if projects were "shovel ready" (California Environmental Protection Agency 2010).

These TMF and "shovel ready" requirements can be problematic for resource-poor communities. In both cases, the funding criteria define funding eligibility on the very bases of some of the fundamental weaknesses of resource-poor communities. A lack of financial resources at the community-level largely drives poor TMF in the first place. Hence, requiring economically disadvantaged communities to show adequate TMF can lead to an unbreakable chain of events. Without TMF, funding can be denied. But without funding, TMF may not be developed. Unless some support for developing TMF is provided, TMF may never be met[16]. Similarly, water systems that are "shovel ready" likely have a certain level of resources to begin. Making "shovel readiness" a requirement decreases the likelihood that some of the water systems with the most need would receive appropriate funding, since those systems likely lacked planning resources to develop plans in the first place.

A lack of community resources and the demographic composition of a community can further impede mitigation efforts when these factors are used as the basis for discriminatory use of municipal authority. Consolidation consists of a nearby community physically connecting its water infrastructure to a larger system, or sharing managerial functions (i.e. shared system operator) with one or more systems. Consolidation is a widely advocated solution to drinking water contamination in small communities because the per-user cost of improvements (e.g. new well) can be too expensive for a small system to shoulder, due to insufficient economies of scale. But, as seen with Tooleville, once physical (e.g. proximity) and engineering (e.g. infrastructure viability) constraints are considered, consolidation becomes a largely political process, and can be hindered by relations between richer and poorer systems or areas (Castillo et al. 1997; Ottem et al. 2003; Raucher et al. 2004).

When System-Level Coping Fails: Household-level Coping

When coping at the community water system-level fails, even partially, to mitigate contamination, households must cope. But social vulnerability of residents, impartial or inadequate coping strategies, and regulatory failures can impede successful mitigation at this level, creating yet another pathway of vulnerability to exposure. To begin, social vulnerability can disenfranchise residents, leaving them unable to obtain appropriate information or advocate for their needs. For example, local decision makers (i.e. water boards) can discriminate against residents on the basis of language, race/ethnicity or socioeconomic class, or homeownership. In the community of East Orosi, for example, residents providing testimony to the U.N. Special Rapporteur on the Human Right to Water and Sanitation noted that due to their ethnicity and accents in English they were continually turned away by water board administrators when they sought information about their water quality. In Alpaugh, and other mutual water companies, residents are required to be homeowners to vote on rate increases. This has been problematic because local residents largely opposed rate increases, and yet if they were renters they were not allowed to express their opinion.

A series of regulatory failures can further undermine household-level coping

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16 This is not to say that the *intention* of requiring TMF (to show long-term sustainability) is ill-founded, but rather that the practical implications can backfire for resource-poor communities.

mechanisms. For example, the SDWA has focused on a contaminant-by-contaminant mode of regulation, largely ignoring the need for public notices to address the effects of exposure to multiple contaminants. In 2007, 5% of the Valley's 677 active community water systems received an MCL violation for both nitrate and total coliform (CDPH 2008a). This is problematic for two reasons. First, because methemoglobinemia (the key health effect for which the nitrate MCL protects) is potentially associated with nitrate plus bacterial contamination of the water. This condition can favor the conversion of nitrate to nitrite and the occurrence of diarrhea, which increases the risk of methemoglobinemia in infants (Fan and Steinberg 1996). And second, because a violation of the coliform MCL triggers a boil-water orders as a temporary coping mechanism, and yet boiling water can increase concentrations of nitrate. And yet the SDWA has no clear stipulations on how residents can address multiple exposure risks.

Furthermore, by considering contaminants with health effects due to chronic exposure less important (at least with regards to Tier 1 notifications), the SDWA also fails to require water systems to provide adequate information to consumers on how to protect against long-term exposure. At a recent community meeting, a resident from Cutler, in Tulare county, noted that for years she has received Consumer Confidence Reports from her water system indicating that DBCP levels exceeded the legal limit, but that residents should not worry because health impacts were not based on immediate exposure, but rather lifetime exposure (see wording on "not an emergency" and "this is not an immediate risk" in Figure 5.5). She noted she had been living in her community for nearly 30 years—so was it true that she should not worry?

IMPORTANT INFORMATION ABOUT YOUR DRINKING WATER

Cutler Public Utility District Has levels of DBCP
Above Drinking Water Standards

Our water system recently failed a drinking water standard. Although this is not an emergency, as our customers, you have a right to know what you should do, what happened, and what we are doing to correct this situation.

We routinely monitor for the presence of drinking water contaminants. Water sample results collected Aug. 2008 thru Dec. 2009 showed dibromochloropropane (DBCP) levels of .25 ug/L. This is above the standard or maximum contaminant level (MCL) of 0.20 ug/L. This Well runs periodically.

What should I do?

- **You do not need to use an alternative (e.g., bottled) water supply.**
- This is not an immediate risk. If it had been, you would have been notified immediately. However, *some people who use water containing DBCP in excess of the MCL over many years may experience reproductive difficulties and may have an increased risk of getting cancer.*
- If you have other health issues concerning the consumption of this water, you may wish to consult your doctor.

Figure 5.6. Portion of a consumer confidence report for Cutler, CA.

What's more, the SDWA has inadequately addressed the need to provide information in multiple languages. Until the passage of the 2011 California Assembly Bill 938, for example, water systems were not required to provide notices in any language other than English when the system had a Tier 1 violation (California Code of Regulations 2008d). This leaves non-English speaking residents in many communities unable to understand public health warnings regarding violations in which short-term exposure could pose an acute health risk.

Even when households take actions to reduce exposure, coping mechanisms may not be effective. For example, households may purchase bottled water, but individuals may not consistently drink bottled water. In other cases, households may install treatments, such as a water filter. But this may still fail to reduce exposures if the treatment device does not treat for the proper contaminant, or if households fail to properly maintain the device, such as installing new filters on an ongoing basis (Moore et al. 2011).

In both cases coping mechanisms are only partially protective. And yet, households incur significant costs for these partially protective measures. For example, in some Valley communities, households can pay 4% to 10% of their monthly income on water expenditures (Moore et al. 2011). These expenditures include two costs, both the water utility bill and the coping cost of buying bottled or vended water. This expenditure comprises a significant fraction of a low-income household's income and is above the U.S. EPA's drinking water affordability criterion of 2.5% of median household income (U.S. Environmental Protection Agency 2003).

But more fundamentally, that households are forced to cope when a water system is unable is problematic for several reasons. First, impartial household coping mechanisms reflect the fact that households do not have the capacity to systematically address contaminant issues on their own. At the root of this is the issue that the so-called "right to know" (i.e., Consumer Confidence Reports required by the SDWA) do not readily translate in a household's "right to act" to effectively reduce exposures. This is especially the case for low-income households that lack financial resources to install and maintain good filtration systems or consistently buy and consume bottled water. Secondly, exposure to drinking water contaminants occurs at other points along the exposure pathway. For example, dermal absorption or inhalation of contaminants can occur when bathing and cooking, etc thus point-of-use devices do not adequately address all exposure pathways. Thus it is unrealistic that households may adequately improve water quality at all points-of-use. Third, in forcing coping strategies at the household level, costs are passed along directly to consumers, with no guarantee that the cost will lead to adequate protection or a long-term solution. Certainly when water systems develop a mitigation strategy, these costs are passed along to customers. But under these circumstances, there is some expectation that the end result is water quality of adequate standards (assuming mitigation was successful). These are all considerations that must be considered in the long-term development of policy solutions.

Discussion & Conclusion

The drinking water vulnerability framework unravels a cyclical and interactive relationship between the natural, built and social environments that impact water quality, contaminant exposure and mitigation capacity of water systems and households. The framework traces the development of a composite burden that includes the exposure and coping costs that water systems and households face. More specifically, the framework uncovers how a series of historical planning policies have jointly shaped access to water supplies as well as to and physical and financial resources that flow to small, rural communities. These forces, in conjunction with regulatory failures, a lack of community resources to successfully address contamination and political disenfranchisement of local residents, help explain the origins and persistence of social inequities in drinking water quality. That these same forces also influence coping capacities and lead to partial protection, at best, serves to further exacerbate the impacts of drinking water contamination.

The drinking water vulnerability framework and related analyses has important implications for the water policy arena. To be certain, numerous policies have attempted to address drinking water contamination and address challenges that small water systems face, including American Economic Recovery Act monies for improvement of drinking water infrastructure and State Drinking Water Revolving Funds to help finance drinking water projects. But unless future incarnations of these policies address some of the issues highlighted by the framework, these policies are unlikely to improve drinking water conditions in the most disadvantaged communities.

To this end, specific policy recommendations include helping to fund and develop TMF capacity in small or disadvantaged systems, such as training and educating local operators and water board members on technical and financial management of water systems. While these efforts are already underway[17], concerted focus on improving TMF capacity in disadvantaged communities is critical. In addition, policies and funding mechanisms should not necessarily use TMF capacity as a requirement for funding (at least in disadvantaged communities), but should find ways to support it when it is lacking. This would support the development of a locally competent workforce and it would enhance long-term sustainability of water systems by actually funding TMF. Similarly, rather than given priority to "shovel ready" projects, funds may need to be made available to systems that are "planning ready." These funds would help small and/or disadvantaged systems develop initial engineering and financial plans for contaminant mitigation and infrastructure needs. In the end, implementation of these recommendations may require separate funding mechanisms and cycles for different types of systems.

In a similar fashion, the general promotion of water system consolidation—be it physical connection of a small system to a larger one, or sharing of management capacities—should address the political and social barriers noted in this paper. Given the local politics that can compromise consolidation efforts, such efforts may be more successful if facilitated by a regional drinking water development program. For example, a regional planning body with decision-making weight and authority could help ensure that consolidation decisions are not left to isolated cities and communities. This may require abdication of some level of municipal authority over such decisions (something many cities are loath to surrender), but may be the only way for consolidation efforts to be implemented. As an example, regional consolidation efforts in New Mexico's have been particularly successful because they have brought together diverse stakeholders and decision makers and defining joint areas of need (Amador Surgeon 2010).

[17] For example, on November 17, 2011, the State Water Board approved a contract with California Rural Water Association (CRWA) to "provide up to $500,000 in wastewater-related technical assistance to small, disadvantaged communities (SDACs) statewide, defining a SDAC as a public body with fewer than 20,000 persons, and an annual median household income (MHI) of less than 80 percent of the statewide MHI." The types of technical assistance that will be offered included: "a) Preparation of financial assistance applications; b) Review of proposed project alternatives to assist in identifying low-cost, sustainable approaches; c) Assistance with planning and budgets, including capital improvement planning; and d) Assistance with community outreach, awareness, and education...." (Source: email of notification on file with author).

Finally, future amendments to the SDWA would be beneficial. For example, efforts to ensure that water systems are able to comply with monitoring and reporting violations should be given particular priority. Similarly, drinking water regulations should increasingly address the co-occurrence of contaminants, and how to adequately inform residents about the impacts and temporary protective measures to take.

Beyond these policy implications, the framework has implications for future research. First, the framework serves as a call for environmental justice and social epidemiology-oriented drinking water research to focus on a broader set of outcomes (e.g. coping costs, or multiple contaminants) in relation to drinking water contamination. Second, the framework begs for research to examine the structural determinants of contaminated drinking water in other regions. Results from these new efforts as well as from the application of existing studies (Anderson 2008; Marsh et al. 2010; Wilson 2009; Wilson et al. 2008a; Wilson et al. 2010; Wilson et al. 2008b) could help to further refine the existing framework.

Third, if consolidation efforts are indeed one key solution to addressing contamination, research efforts should further examine the underlying barriers to consolidation. For example, water policy experts often say that one of the main reasons why consolidation efforts are slow is because smaller systems fear losing local autonomy, and they therefore stall or block consolidation efforts. Research on consolidation could examine the hypothesis that barriers to consolidation are not only explained by a fear of "loss of local autonomy", but also by a deeper set of social, economic, political processes. While geographers and legal scholars have addressed these issues in the context of annexation an explicit drinking water focus would benefit drinking water-related solutions.

In sum, the drinking water vulnerability framework moves us beyond a single-contaminant mode of analysis, and helps to uncover the multi-layered processes that explain the origins and persistence of contaminated drinking water and related social disparities. And, it helps us see that solutions—whether for the household, community or region, must address, among other things, the vulnerability of residents, the role of political clout, and a need for regional interventions, not just community-by-community fixes.

CHAPTER 6
Conclusions: Research Contributions, Policy Recommendations and Future Research

Overview

Focusing on drinking water contamination in California's San Joaquin Valley, my dissertation examined: 1) whether communities with greater percentages of people of color, or lower homeownership rates had higher levels of drinking water contaminants and greater odds of non-compliance with drinking water standards, and 2) the role of structural determinants in explaining the origins and persistence of drinking water contamination and social disparities. Through quantitative analyses of potential exposure to nitrate and arsenic, and the development of a socio-ecological drinking water vulnerability framework I uncovered the multi-layered burdens related to contaminated water in the San Joaquin Valley.

I found that communities with higher percentages of Latinos had higher nitrate levels in their drinking water systems, and that those with lower rates of homeownership had higher arsenic levels and greater chances of exceeding federal safety standards. For both nitrate and arsenic, I found that water quality is worse in smaller communities. This is significant because it represents a dual burden—not only do small systems face difficulty in complying with drinking water standards, the people living in these systems are some of the most socially and economically vulnerable, and may be the least able to afford mitigation. While these contaminant-specific findings are informative, the drinking water vulnerability framework emphasized that social disparities in vulnerability to exposure can also be viewed beyond a contaminant-by-contaminant basis. Thus I showed how beyond any statistical associations between race, class and water quality, a series of structural factors have shaped the origins and persistence of disparities to contamination, and the degree of coping capacity. These findings represent important contributions and implications for the environmental health and environmental justice fields, and for the water policy arena. This concluding chapter outlines these implications, and ends by exploring next steps for a related research agenda.

Research Contribution

Do race and class matter in relation to risks from drinking water contamination? Calderon et al's seminal review article in the 1990s asked this very question (Calderon et al. 1993). Relying primarily on case studies, the authors recommended that future studies make use of existing quantitative datasets to explore this question more definitively. As I noted throughout the chapters in this dissertation, only a handful of studies have taken advantage of water quality monitoring datasets to explore these questions.

This, therefore, is the first contribution of this dissertation: I explored the under-examined relationship between drinking water quality and health disparities. While much research has examined the role that environmental conditions play in producing and maintaining health disparities (Gee and Payne-Sturges 2004; Lee 2002; Morello-Frosch et al. 2011; Sexton 2000), most studies have focused little on the role of drinking water quality. While we have information on pockets of highly contaminated water in regions of the U.S., few studies have examined the composition of vulnerable populations, and whether social disparities in exposure exist. By focusing on the degree to which race/ethnicity and socioeconomic class are associated with potential exposure to drinking water contaminants, this dissertation represents an important step in exploring the role that poor drinking water quality may play in contributing to health disparities (through its focus on exposure).

My dissertation also offers the first detailed exploration of the relationship between demographics and water quality in California's San Joaquin Valley, and is the first such study (to my knowledge) to focus on nitrate contamination and social disparities. These discussions have been largely absent in the water quality and environmental health literature focused on the Valley and the greater U.S. For example, while the Valley is one of the most well known agricultural regions in the country, and its water quality status is relatively well documented (Burow et al. 2008; Dubrovsky et al. 2010; Dubrovsky et al. 1998), few studies have attempted to systematically explore distributional and procedural inequities related to drinking water. After documenting water contamination in the region's rivers and aquifers, for example, Dubrvosky et al (1998) note the importance of understanding the impacts of this contamination on human health. A lay-oriented report had examined the relationship between drinking water violations and poverty and ethnicity at the county-level (Environmental Justice Coalition for Water 2005), but failed to address this topic at the community-scale, the appropriate unit of analysis, given that this is the cite of exposure to contaminants. To my knowledge, one study (Byrne 2003) specifically tackled environmental justice questions in San Joaquin County, but as I discuss in *Chapter 3*, this study focused on only one county.

By focusing on exposure and compliance burdens (as in *Chapter 4*), and developing a holistic Drinking Water Vulnerability Framework (in *Chapter 5*), this dissertation also bridged issues related to health, regulatory challenges and structural determinants of exposure disparities. I showed how it is not only important to consider whether socially vulnerable groups (e.g., Latinos or renters) are more exposed to harmful contaminants, but how burdens of non-compliance create dual challenges in low-income communities, leading to potential exposures and coping costs.

My dissertation also offers a number of methodological contributions. First, my work utilized two under-examined datasets on drinking water quality—the CDPH's WQM and PICME databases. During the course of my dissertation, regulators advised me of the pitfalls of these datasets for research endeavors. These datasets, they cautioned, were for regulators to examine compliance, but they had many inconsistencies that would make it difficult to use for research. For example, the frequency of water quality sampling varied widely across systems, county-level data was more likely to be missing because of database incompatibilities (at least in part, as I discuss in *Chapter 5*), and this data did not represent exposure to contaminants, per se. But in *Chapters 3-4* I accounted for these data limitations and still answer my basic research questions. Numerous conversations with CDPH colleagues and a few existing studies helped support my general approach of using point-of-entry samples to estimate average distribution water quality (Cory and Rahman 2009; Stone et al. 2007; Whorton et al. 1988).

In a similar way, my approach to estimate community-level demographics pushed a basic methodological boundary. While CDPH collects water quality and system-level information, it does not gather data on the demographics of water users. From a public health perspective, this would seem an important piece of information to assess, but as far as the SDWA is concerned, the purpose of maintaining these datasets is to simply ensure compliance with drinking water standards.

Beyond simply using existing data in a new ways, however, this dissertation contributes methodologically and conceptually to the drinking water-environmental justice arena. Viewed as a package, for example, *Chapters 3-4* establish the need to consider both exposure and compliance burdens. Furthermore, *Chapter 5* emphasizes the need to consider structural determinants that impact potential exposure. Such an approach reflects current approaches in social epidemiology (Gee and Payne-Sturges 2004; Wilson 2009), and critical environmental justice and geography arguments to move beyond purely statistical analyses of potential environmental injustices (Pulido 1996; Pulido et al. 1996). In addition, the drinking water vulnerability framework expands the focus beyond exposure to contaminants and emphasizes the need to consider additional outcomes, such as coping capacity of communities and households.

Policy Implications

My dissertation has several implications for the water policy and public health arenas, in both international and national contexts. Viewed from an international perspective, my findings beg us to consider placing the challenges faced by San Joaquin Valley communities in an international water policy context. When community residents are forced to buy bottled or vended water because a well has broken (as in the case of Alpaugh), or because water contamination exceeds federally-set standards, this calls into question whether all U.S. residents have access to "improved" drinking water sources, as defined by the World Health Organization (World Health Organization 2009).

Furthermore, by highlighting the challenges to obtaining clean drinking water in some of the most marginal communities in California, this dissertation also pushes us to consider issues of access in the context of the human right to water. As de Albuquerque, the U.N. Special Rapporteur on the Human Right to Water and Sanitation notes, "[by] its nature, a human rights analysis focuses on the situation of the most marginalized and excluded...While these groups comprise a small proportion of the population…they require priority attention" (United Nations General Assembly: UN Human Rights Council 2011). Applying these international concepts of access and rights requires a thorough consideration of just what the U.S. and California residents believe are the basic needs that should be provided with respect to drinking water provision.

At a state and regional level, my findings support specific funding and planning solutions for the Valley's populations. Certainly long-term solutions must address source water protection. In the interim, however, small water systems serving low-SES residents need enhanced funding and technical support to reduce community-level contaminant exposure. But funding mechanisms must be careful to not use the very drivers of vulnerability (e.g. poor community capacity) as eligibility criteria for obtaining funding. This may require a separate tier of funding opportunities for disadvantaged communities. For example, rather than require that systems be "shovel ready", funds may need to be made available to systems that are "planning ready." These funds would help small and/or disadvantaged systems develop initial engineering and financial plans for contaminant mitigation and infrastructure needs.

Ultimately, regional solutions that consolidate smaller water systems serving economically disadvantaged communities with larger ones may be the best approach to addressing coping and exposure disparities. But here again, support for such a regional solution must address the political and social barriers that may lead socially vulnerable or disadvantaged communities to be by-passed or excluded from consolidation efforts. *Chapter*

5 proposes the development of regional-level decision-making bodies that help facilitate consolidation efforts and not leave ultimate decisions to individual cities. This is but one idea, and thinking through other similar ideas will be necessary. For example, in 2011, Tulare County began a pilot program that brings together a diverse set of community, farming and policy stakeholders to help delineate sustainable drinking water solutions in disadvantaged communities.

Nationally, the SDWA would benefit by increasingly addressing regulation and mitigation of co-occurring contaminants. The U.S. EPA's current drinking water strategy has expressed support for a more cumulative impacts-oriented regulatory approach. For example, the Drinking Water Strategy notes that one key goal is to "address contaminants as groups rather than one at a time so that enhancement of drinking water protection can be achieved cost-effectively" (U.S. Environmental Protection Agency 2010a). But the benefit of such an approach should not only be framed in terms of cost-effectiveness. Such an approach can also help communities address exposure to multiple contaminants, and ensure that residents know how to take appropriate mitigation measures when this happens.

In the end, even these regional and national solutions are not enough, if the source of groundwater contamination is not also addressed. To date, agriculture has been allowed to externalize their contamination activities, and communities are ultimately forced to pay for mitigation measures. A truly innovative and just set of drinking water policies would ensure that source water is adequately protected, and that the industries impacting exposure levels are responsible for helping to pay for the cost of mitigation, and for reducing contaminant levels.

In essence, a multi-pronged and multi-level intervention strategy is needed. In the near-term, communities must receive adequate support to enter into compliance and provide safe drinking water. But in the longer-term regional solutions must be developed. These regional solutions, such as enhanced support for consolidation efforts and rigorous groundwater protection efforts, will help shape the long-term viability of a sustainable drinking water supply for all Valley residents (and beyond).

Next Steps in Research

This dissertation represents a first step in addressing social disparities in drinking water contamination in the San Joaquin Valley, but it also sets the foundation for future research. Because each dissertation chapter reads as a stand-alone paper, the limitations of the core analytical chapters (*Chapters 3-5)* are addressed specifically in each. Even so, key limitations are still worth summarizing as they help inform next steps in research. One main limitation pertains to exposure estimates. While *Chapters 3-4* estimate potential exposure, I did not estimate individual-level exposure. Such an effort would entail measuring contaminant levels at different point-of-use sites in the house (e.g. tap, shower, etc), and surveying residents on their daily water uses and intakes. As such, the associations I examine between water quality and community demographics do not speak to actually amounts of ingested contaminants, but only potential levels in drinking water. Giving the various coping mechanisms that residents do undertake (e.g., buying bottled water or installing filtration systems), future research would benefit from conducting a more detailed and specific exposure assessment. As a point of comparison, future work should compare distribution water quality estimates (as estimated in *Chapters* 3-4) with concentration of contaminants at the tap. Future research efforts should also focus on conducting more epidemiology-focused studies of health effects related to contaminants.

A final area for further consideration is in the cumulative impacts arena. Recognizing that multiple sources of contamination exist (e.g. from air, land and water), community health and advocacy groups have long pushed for an integrated framework to assess these co-existing environmental problems, often referred to as cumulative impacts (NEJAC 2004). In doing so, these groups have pushed policy makers and scientists to think about the science of cumulative impacts, and how decision-making can better incorporate a cumulative focus. A cumulative impacts drinking water research agenda would move beyond a contaminant-by-contaminant analysis, analyze multiple geographic sites of exposure, and integrate drinking water into broader cumulative impact assessments that include air and land.

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APPENDIX A
Appendix to Chapter 2

Appendix A.1 Snapshot of a Waterscape: Drinking Water Systems in the San Joaquin Valley

Ownership of Public Water Systems

The ownership status of a PWS is best understood by breaking it down into several layers. Recall that by definition, all PWS are regulated by the SDWA and serve the *public,* regardless of ownership type. Thus all PWS can be considered "public", though this implies nothing of the ownership structure of the PWS.

PWS ownership is classified as either public or privately owned. Private refers to being investor-owned, either by a private company, or a group of individuals (e.g. mutual water company or a homeowners association). Private ownership can be further broken down into two sub-categories: private PWS that are regulated by the California PUC and private PWS exempt from regulation by the PUC. The 4 criteria for being PUC regulated are that a PWS be: 1) investor owned (i.e. any corporation or person that owns, controls or manages provision of water), 2) serve the public, 3) receives payment for the services and 4) not exempt (Firestone 2009). Exempt systems generally include investor-owned systems such as homeowner associations, residential apartment complexes, mobile home parks and mutual water companies[18], or a "water as accommodation" system where a neighbor might share her well water with another neighbor.

Community Water Systems in the San Joaquin Valley

Active versus Inactive Systems

Between 1993 and 2007, 879 CWS were in active operation in the SJV, and supplied water to the public at some point within that time frame. However, as of 2007, 671 CWS were in active operation, indicating that during that time period approximately 208 systems were inactivated or destroyed. An inactive system does not supply water to the residents, either because the system closed down, or consolidated with a nearby system. The remainder of this appendix focuses on the 671 CWS that were active as of 2007.

Number of Community Water Systems and Population Served

The 671 actively operating CWS in the SJV, representing slightly over 20% the state's approximately 3100 active CWS. Comprising over one quarter of all systems in the Valley, Kern County has 187 active systems (Table A.1). Kern County is followed by Fresno, with 113 CWS, and Tulare with 100 CWS. These are also the three counties with the highest numbers of inactivated systems, from 1993 to 2007. From 1993 to 2007, the percent of CWS that were inactivated in each county (as a fraction of the total number of active systems

[18] A mutual water company (mutual) is a nonprofit mutual benefit corporation that is organized to sell, distribute, supply or deliver water for irrigation purposes or domestic use (Firestone, 2009).

in each county) were 34%, 24%, 23%, 5%, 37%, 12%, 22%, and 28% in Fresno, Kern, Kings, Madera, Merced, San Joaquin, Stanislaus, and Tulare, respectively.

Table A.1. Active and inactivated community water systems by county, 1993-2007.

	# of CWSs Active in 2007	% CWSs Active in 2007	# CWSs Active Between 1993 and 2007	# CWSs Inactivated Between 1993-2007	% CWSs Inactivated 1993-2007 (of Total)
Fresno	113	17%	170	57	34%
Kern	187	28%	245	58	24%
Kings	17	3%	22	5	23%
Madera	63	9%	66	3	5%
Merced	26	4%	41	15	37%
San Joaquin	96	14%	109	13	12%
Stanislaus	69	10%	88	19	22%
Tulare	100	15%	138	38	28%
Total	**671**	**100%**	**879**	**208**	**24%**

As of 2007, CWS in the SJV serve approximately 3 million people, or 80% of the SJV's 3.8 million inhabitants. As a fraction of the county's total population, CWS in Stanislaus County serve the most number of people (88% of the county's approximately 510,000 inhabitants). Stanislaus County is followed closely by Fresno County (CWS serve 85% of the 900,000 inhabitants). Interestingly, Madera serves the smallest fraction of its total population, at 66% (Figure A.1 and Table A.2).

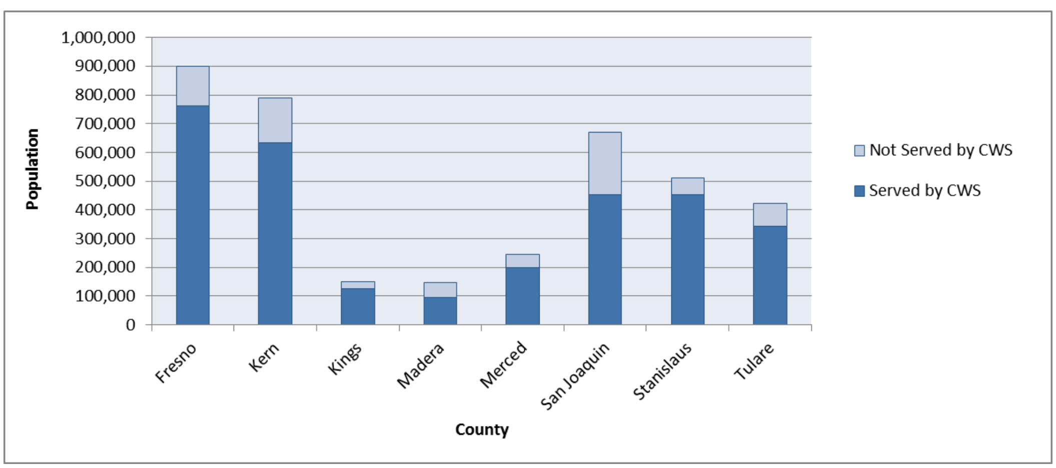

Figure A.1. Population served by community water systems in the eight SJV counties. Data source: Permits, Inspections, Compliance, Monitoring and Evaluation database (CDPH 2008a) and (U.S. Census Bureau 2007d).

Table A.2. Population and percent served by CWS.			
County	**Population Served by CWS**	**Total County Population**	**% of Population Served by CWS***
Fresno	762,461	899,348	85%
Kern	634,597	790,710	80%
Kings	124,268	148,875	83%
Madera	96,167	146,513	66%
Merced	199,706	245,514	81%
San Joaquin	452,683	670,990	67%
Stanislaus	451,960	511,263	88%
Tulare	343,094	421,553	81%
Total	**3,064,936**	**3,834,766**	**80%**
Data source: CDPH (2008a) and (U.S. Census Bureau 2007d). *Assumes difference between total county population and population served by CWSs are on private supply. This does not account for population count errors in the WQM and PICME databases.			

Beyond population served, one can consider the density of systems per population and land area. Here, the story is quite different. Madera County has the highest density of CWS per 100,000 people[19] (43 systems for every 100,000 people). Merced County has the lowest density of CWS per 100,000 people, indicating that, at least in part, a larger fraction of the population is served by larger systems. However, in terms of the density of systems per one hundred square miles[20], San Joaquin has the highest density (6.9 CWS) per 100 square miles), followed by Stanislaus County (4.6 CWS per 100 square miles). These numbers are not altogether surprising in that these are the two counties with the highest density of population per square mile. Kings County has the lowest density of systems, at 1.2 CWS per 100 square miles (Table A.3).

Table A.3. Population and CWS density statistics.				
County	**Total Area (square miles)**	**Population Density (People per square mile)**	**Density of systems per 100 square miles**	**Density of systems per 100,000 people[21]**
Fresno	5,963	150.8	1.9	12.6
Kern	8,141	97.1	2.3	23.6
Kings	1,391	107.0	1.2	11.4
Madera	2,136	68.6	2.9	43.0
Merced	1,929	127.3	1.3	10.6
San Joaquin	1,399	479.6	6.9	14.3
Stanislaus	1,494	342.2	4.6	13.5
Tulare	4,824	87.4	2.1	23.7
Total	**27,276**	**140.6**	**2.5**	**17.5**
Data source: U.S. Census Bureau (2007a)				

[19] Density based on total county population, not total population served by active CWS.

[20] Because some counties, such as Tulare County, have a significant portion of the county designated as state or national parks, a more accurate density calculation would exclude these park areas. Doing so would likely change some of these estimates.

[21] Density of CWS per 100,000 people uses total county population for 2007 as its denominator.

Size of Community Water Systems

CWS vary in size, both in terms of number of service connections and population served (Figure A.2). Overall, the vast majority of the 671 CWS serve communities with fewer than 500 residents. Specifically, 72% (n=483) of all CWS in the Valley serve less than 501 people, and 14% serve 501 to 3300 people. Only 1% (n= 5) of all CWS serve more than 100,000 people (Figure A.2). Examples of such systems include the City of Bakersfield (i.e. California Water Service-Bakersfield), in Kern County, the City of Modesto in Stanislaus County, the City of Fresno in Fresno County, and the City of Stockton (i.e. California Water Service-Stockton) in San Joaquin County. At a county level, the number of CWS that serve less than 501 people ranges from 53% to 79% (Table A.4). These breakdowns reflect commonly cited statistics that state that the majority of systems in the U.S. are very small systems (serving less than 501 people) (Committee on Small Water Systems 1997).

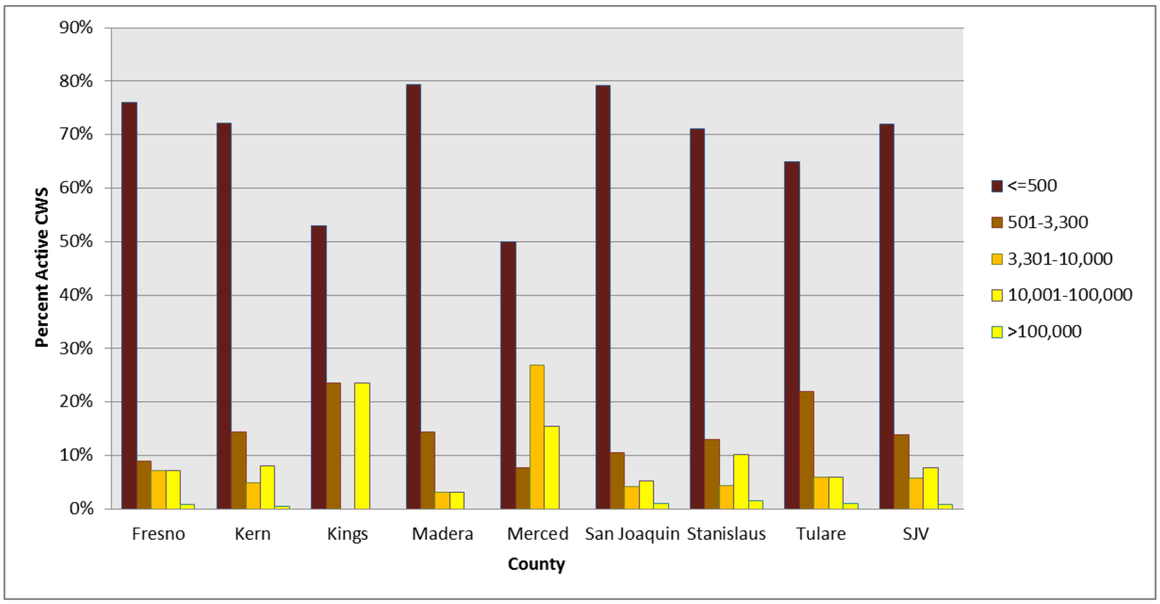

Figure A.2. Percent of active CWS in 2007 by population category. SJV label refers to the total across all San Joaquin Valley counties. Data source: CDPH (2008a).

While the majority of CWS are very small, they serve only 70,000 people (2% of population). Eighty-six percent (~2.6 million people) of the Valley's population served by CWS is served by CWS that serve over 10,000 people. The spatial distribution of these systems can be seen in Figure A.3.

Table A.4. Total number of people served by CWS within each population category* and the percentage of the county's population served by CWS that this represents.						
County	**<=500**	**501-3,300**	**3,301-10,000**	**10,001-100,000**	**>100,000**	**Total Population**
Fresno	14,237 (2%)	13,119 (2%)	56,034 (7%)	22,1560 (29%)	45,7511 (60%)	762,461
Kern	16,020 (3%)	30,887 (5%)	54,487 (9%)	301,996 (48%)	231,207 (36%)	634,597
Kings	1151 (1%)	7,322 (6%)	0	115,795 (93%)	0	124,268
Madera	7,234 (8%)	13,948 (15%)	9,395 (10%)	65,590 (68%)	0	96,167
Merced	2,230 (1%)	3,200 (2%)	37,898 (19%)	156,378 (78%)	0	199,706
San Joaquin	9,273 (2%)	12,370 (3%)	22,866 (5%)	236,397 (52%)	171,777 (38%)	452,683
Stanislaus	7,606 (2%)	13,372 (3%)	18,554 (4%)	200,428 (44%)	212,000 (47%)	451,960
Tulare	11,510 (3%)	30,971 (9%)	39,067 (11%)	154,546 (45%)	107,000 (31%)	343,094
Total	**69,261**	**125,189**	**238,301**	**1,452,690**	**1,179,495**	**3,064,936**

Data source: CDPH (2008a)

* Five population ranges are indicated: less than or equal to 500, 501-3,300 people, 3,301-10,000 people, 10,001-100,000 people and greater than 100,000 people. Under each range is the total population served by systems whose populations fall within this range. For example, in Fresno county 12,586 people are served by the smallest sized systems (i.e. those serving less than 501 people).

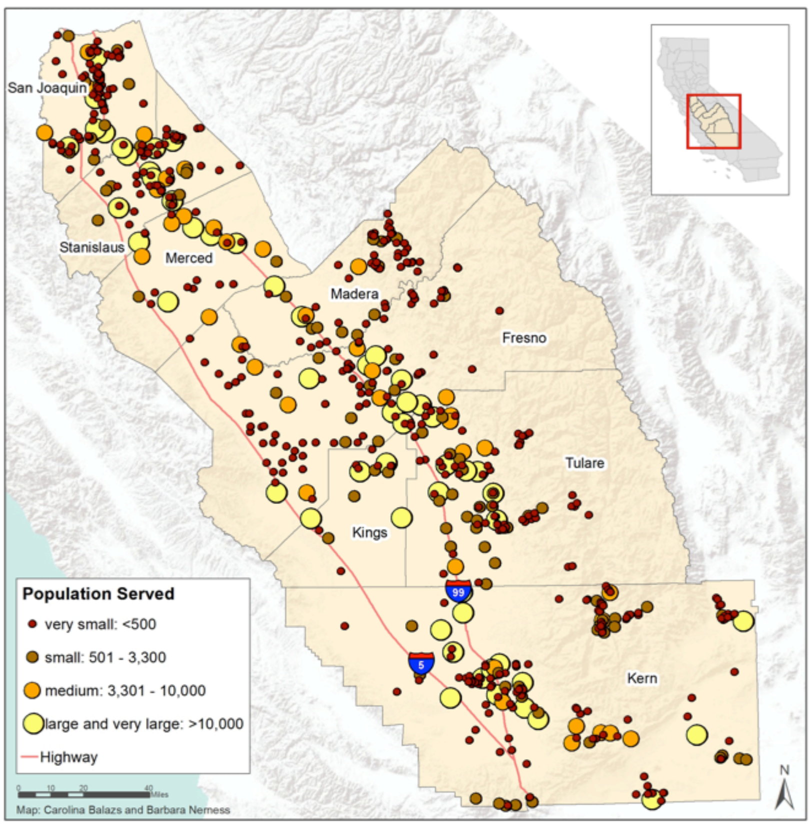

Figure A.3. Map of Population Served by Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. Data source: CDPH (2008a) for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary.

Source of Water

Of the 671 currently active systems, all but 15 (2%) systems have data on the source of water used. Of those 656, 577 (88%) rely solely on groundwater as their source of water. Forty-seven CWS (7%) rely solely on surface water (Figures A.4-A.5). The remaining 5% rely on a combination of surface and groundwater.

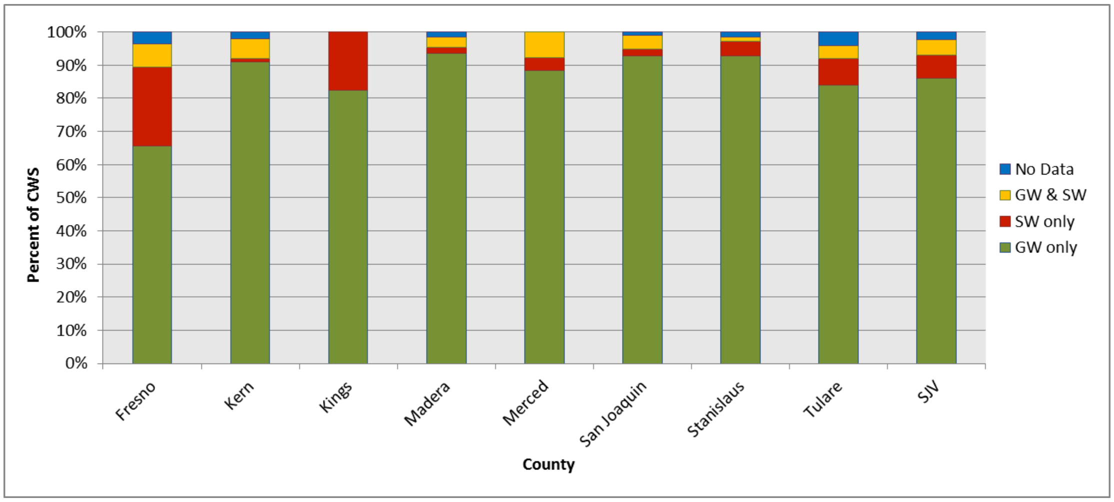

Figure A.4. Percent of CWSs relying upon one of three types of water sources. GW=groundwater only only, SW=surface water only, GW & SW=combination of both. Data source: CDPH (2008a).

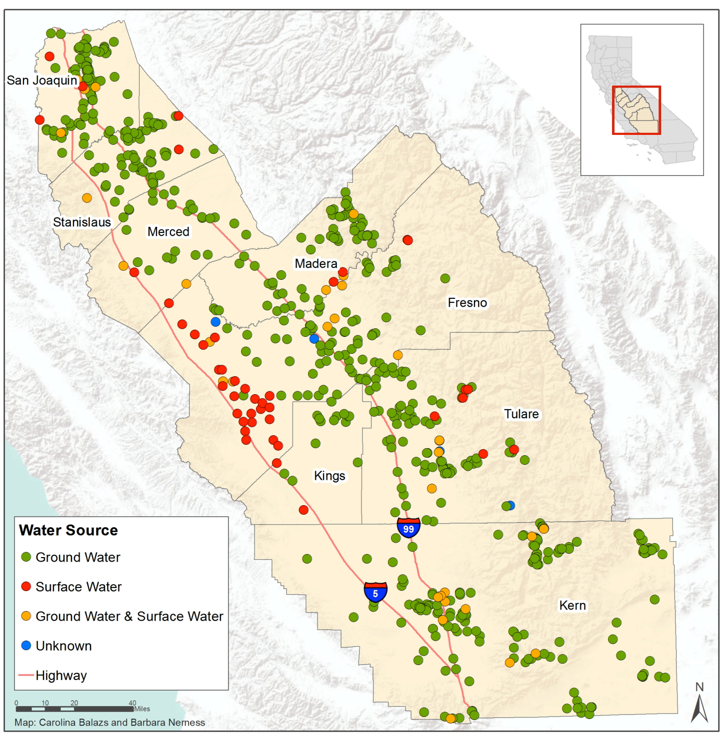

Figure A.5. Map of Water Source for Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. Data source CDPH (2008a) for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary.

While the majority of systems rely entirely on groundwater, the largest CWS rely more on a combination of groundwater and surface water. In particular, approximately 18% of medium-sized CWS (i.e. 3,300 to 10,000 people) rely on combined sources. Approximately 12% of all large CWS (i.e. 10,001-100,000) rely on a combination of groundwater and surface water sources. But 60% of all CWS (3 out of 5) of the very large systems (i.e. greater than 100,000 people) rely on combined water sources (Figure A.6 and Table A.5).

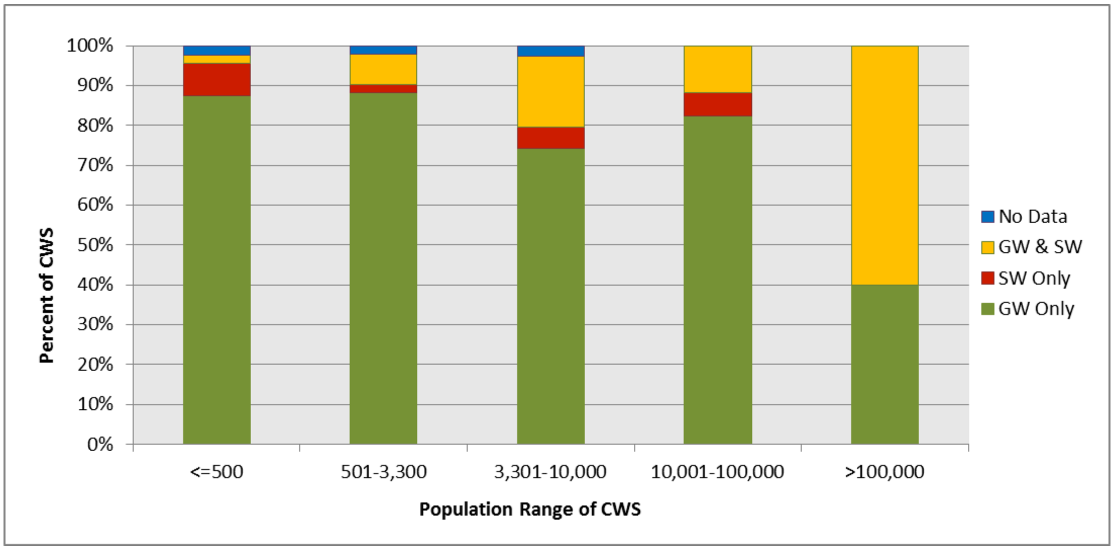

Figure A.6. Percent of CWSs relying on different water sources, by population category. GW=groundwater only only, SW=surface water only, GW & SW=combination of both. Data source: CDPH (2008a)

Table A.5. Number of CWS with one of three types of water sources.					
County	**GW only**	**SW only**	**GW & SW**	**No Data**	**Total**
Fresno	74 (65%)	27 (24%)	8 (7%)	4 (4%)	113
Kern	170 (91%)	2 (1%)	11 (6%)	4 (2%)	187
Kings	14 (82%)	3 (18%)	0	0	17
Madera	59 (94%)	1 (2%)	2 (3%)	1 (2%)	63
Merced	23 (88%)	1 (4%)	2 (8%)	0	26
San Joaquin	89 (93%)	2 (2%)	4 (4%)	1 (1%)	96
Stanislaus	64 (93%)	3 (4%)	1 (1%)	1 (1%)	69
Tulare	84 (84%)	8 (8%)	4 (4%)	4 (4%)	100
SJV	577 (86%)	47 (7%)	32 (5%0	15 (2%)	671
GW=groundwater only only, SW=surface water only, GW & SW=combination of both. Data source: CDPH (2008a)					

Ownership Type

Sixty-six percent (n=442) of the CWS in the Valley are privately owned[22]. Privately owned systems make up 66% (442) of the CWS, including both PUC (6%) and non-PUC regulated (60%). At the county level, private, non-PUC regulated CWS comprise 35% of all

[22] Estimates of ownership are based on the author's classification, using data from the Public Utility Commission's website, system0 name, corroboration with PICME's own classification and confirmation of some systems by the California Department of Public Health officials.

CWS in Merced County and up to 74% in San Joaquin County. Interestingly, in Merced County, there are more publicly owned systems than private non-PUC regulated systems (Figure A.7). Ownership type does not appear to follow a particular spatial pattern (Figure A.8).

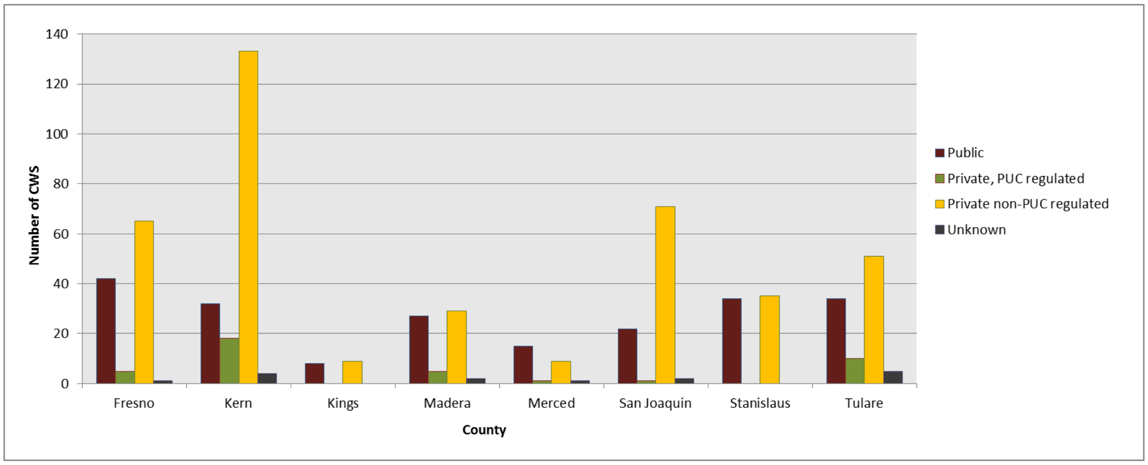

Figure A.7. Ownership of CWSs by county. Data source: CDPH (2008a) and CA PUC

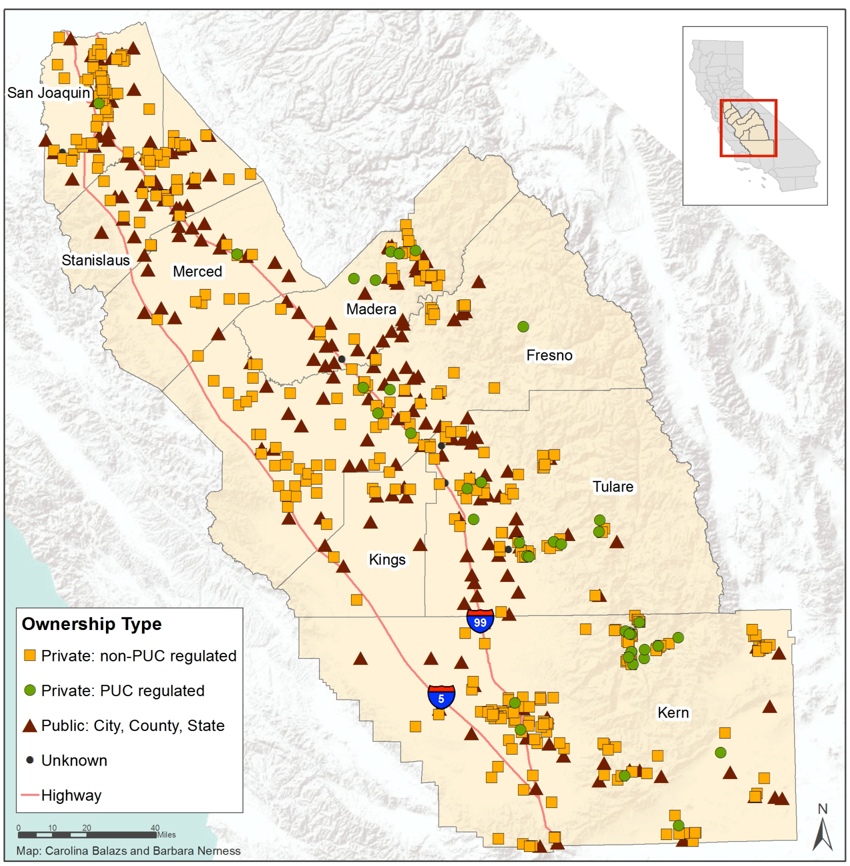

Figure A.8. Map of Ownership of Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. Data source: CDPH (2008a) for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary.

When one thinks of CWS, often what comes to mind are small neighborhoods or towns served by a particular water system provider. Included in this definition, however, are mobile home parks and stand-alone apartment buildings, as well as labor camps where farm laborers reside year-round. Figure A.9 indicates different types of CWS based on using the system name as a general identifier. The most numerous individual categories are mobile home parks or apartments (n=111, or 17%), mutual water companies (n=104, 15%) and city-owned (n=61, 9%). Forty-seven (7%) are farms such as ranches, dairies or labor camps. Of course, because these categories were "artificially" differentiated by the author, it does not mean that mobile homes are the dominant type of CWS. For example, if one sums the number of home owner associations, estates, mutuals, water systems, etc, the total far surpasses that of mobile home parks and apartments. Thus this distinction is helpful insofar as it helps tabulate the various categories, by general type. The number of people served by these different system categories is also important to keep in mind, as the 61 city systems serve 65% of the population served by CWS in the Valley.

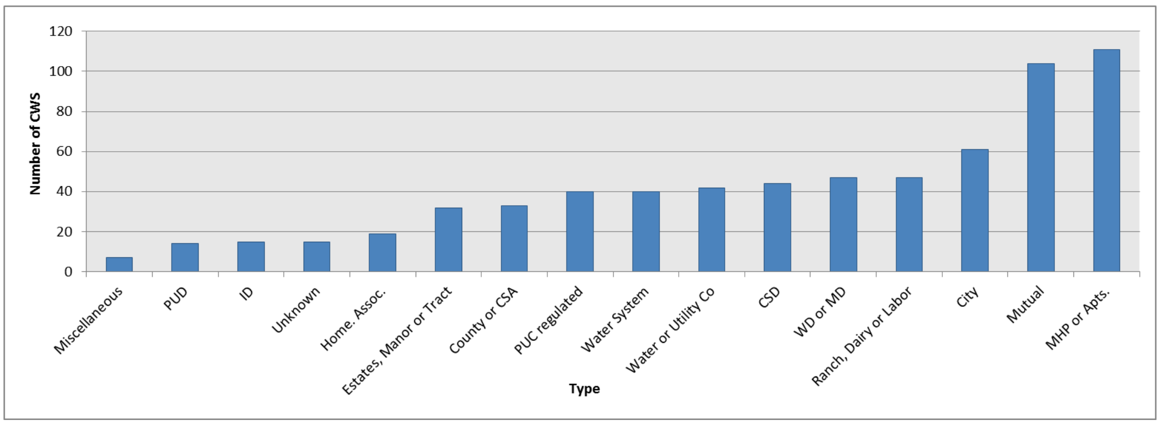

Figure A.9. Number of CWS by type of system.

Regulating Entities

The majority of CWS are county-regulated. In particular, anywhere from 50% (Merced County) to 76% (San Joaquin and Madera County) of all CWS are county-regulated. While Kern and Fresno counties have a considerable amount of systems under 200 connections, in 1993 and 2007, respectively, these counties gave up their LPA status, thus making the state the new regulating entity.

Demographics of CWS

The following figures highlight economic and racial-ethnic patterns across the CWS in the Valley. These statistics were generated using source locations and census block group data, as described in the *Chapter 3*. In general, one sees a consistent set of low-income and highly Latino customer base in western Fresno County and in northeastern Kern county. In Fresno, many of these systems are farm labor camps. Another common trend is that San Joaquin has some of the more affluent systems (relative to others in the Valley), and the eastern Sierra foothills have lower poverty levels and whiter customer bases (Figures A.10-A.13). Table A.5 highlights the interquartile range for percent Latino, home ownership and poverty at the water system level, across all systems that had source locations (which permitted a demographic estimate).

Table A.6. Estimated demographics at the community water system level.					
	Statistic				
Variable of Interest	**25th percentile**	**50th percentile**	**75th percentile**	**mean**	**N**
% Latino	9	26	51	33	644
% Home Ownership	57	71	80	68	644
% Above Poverty	43	61	73	58	644
Median Household Income	$28,919	$36,572	$45,885	$39,894	644

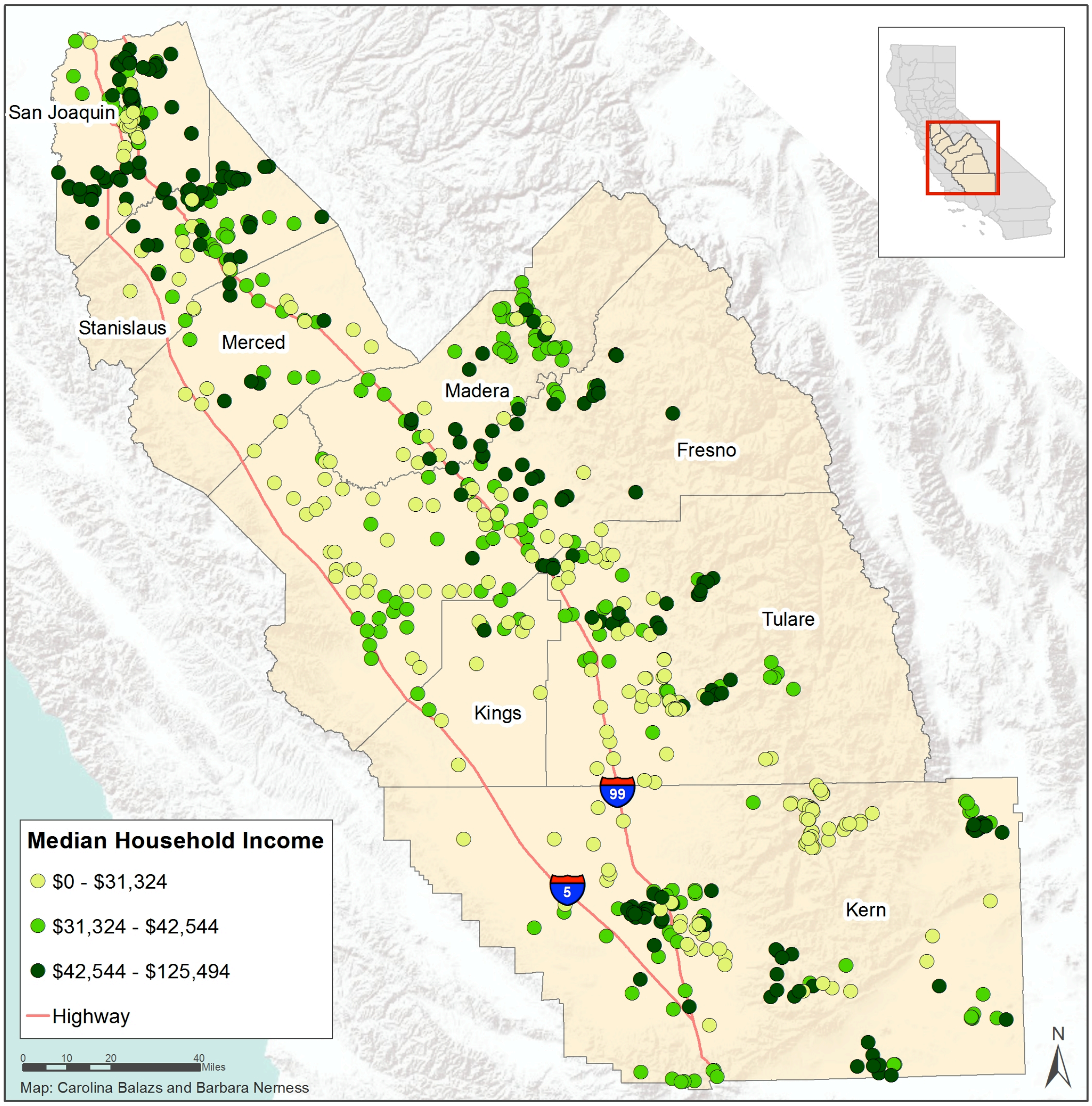

Figure A.10. Map of Median Household Income for Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. Data source: CDPH (2008a) for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary; US Census 2000 & Balazs et al (2011) for demographics.

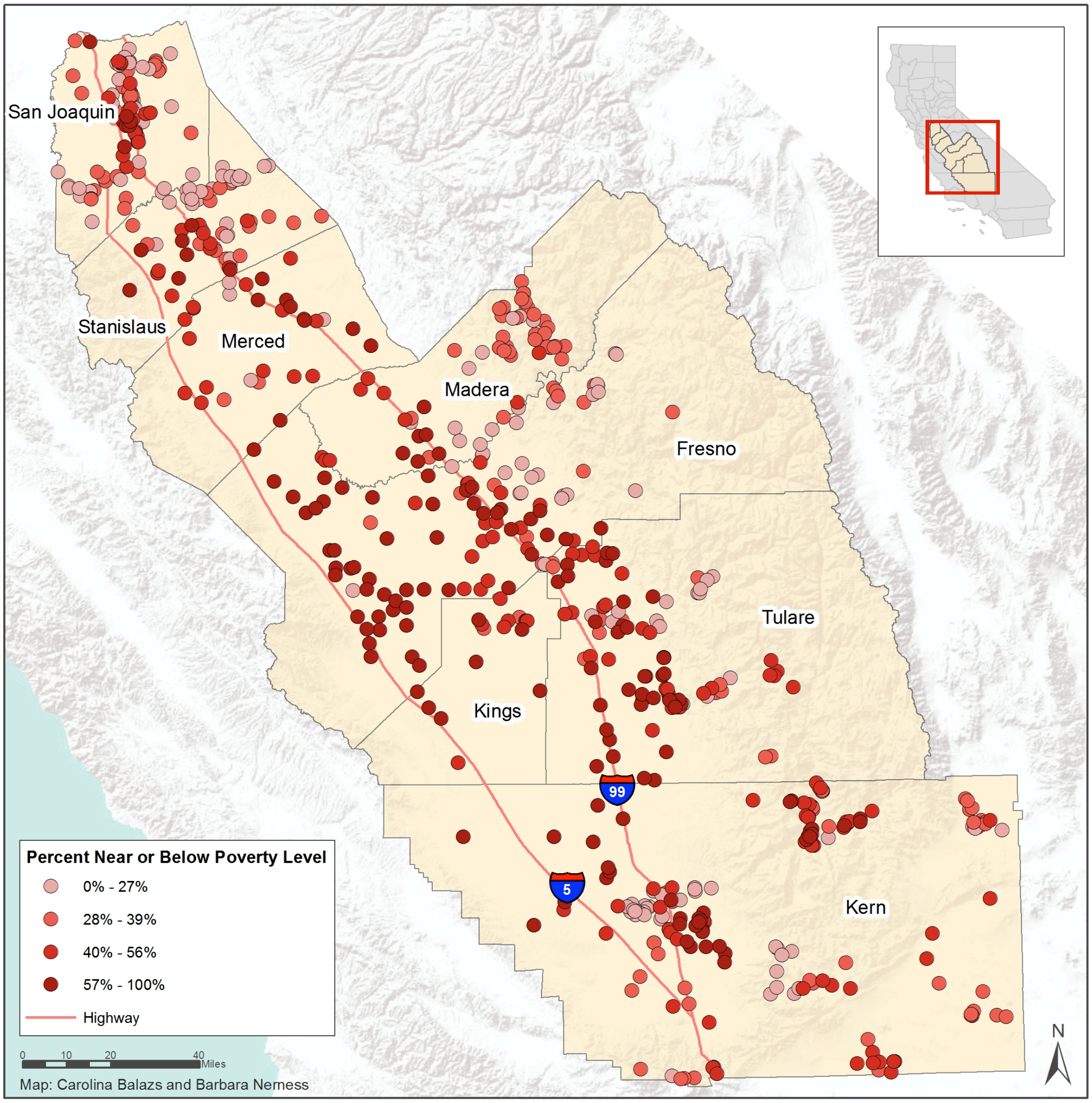

Figure A.11. Map of Percent Near or Below Poverty Level for Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. 'Below poverty' includes households with a ratio of .99 or less below the poverty level in 2000. 'Near poverty' includes households with a ratio of 1-1.99 (i.e. 100-199% above the poverty level). Data source: CDPH (2008a) for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary; US Census 2000 & Balazs et al (2011) for demographics.

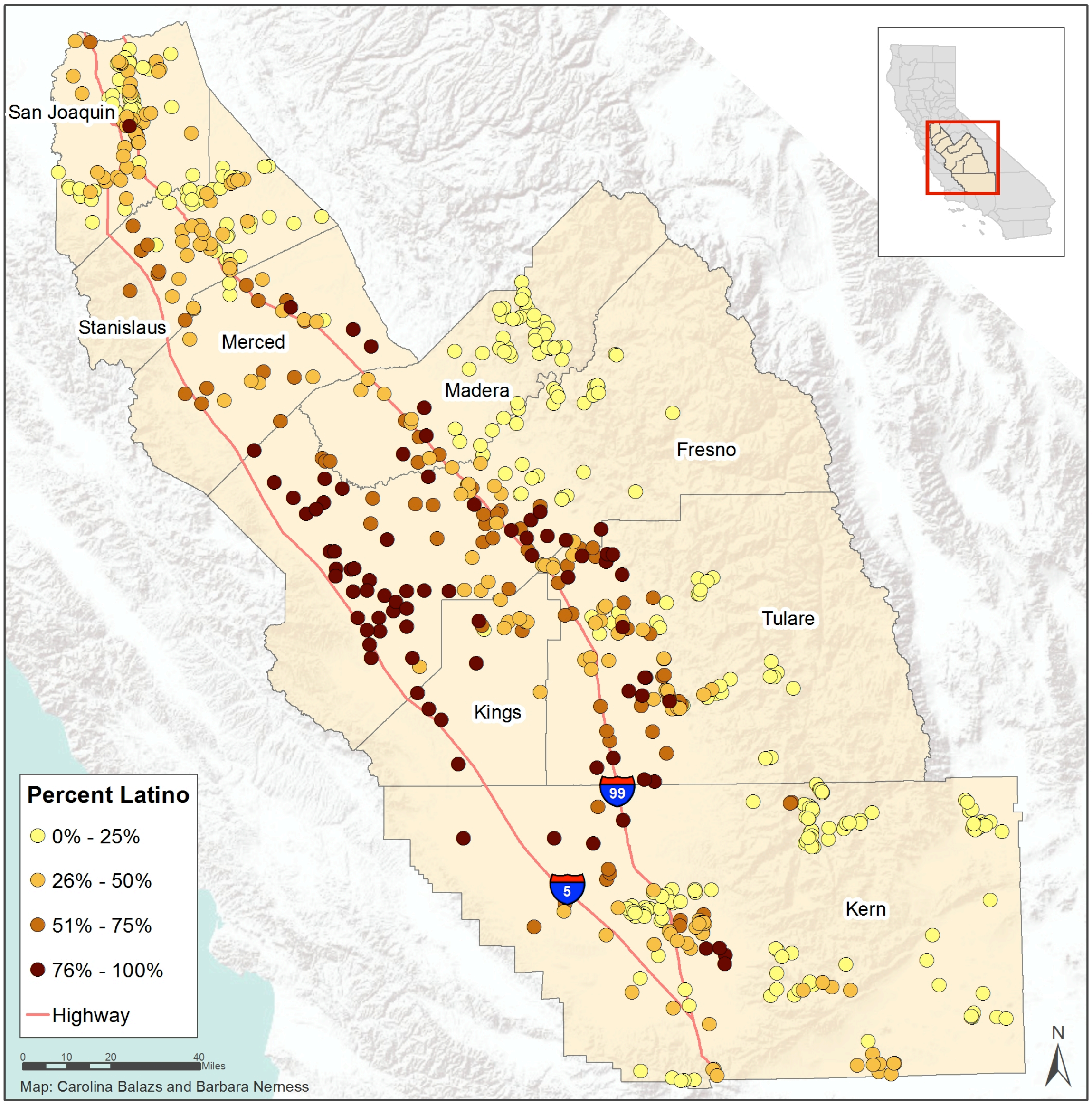

Figure A.12. Map of Percent Latino for Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. Data source: CDPH (2008a) for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary; US Census 2000 & Balazs et al (2011) for demographics.

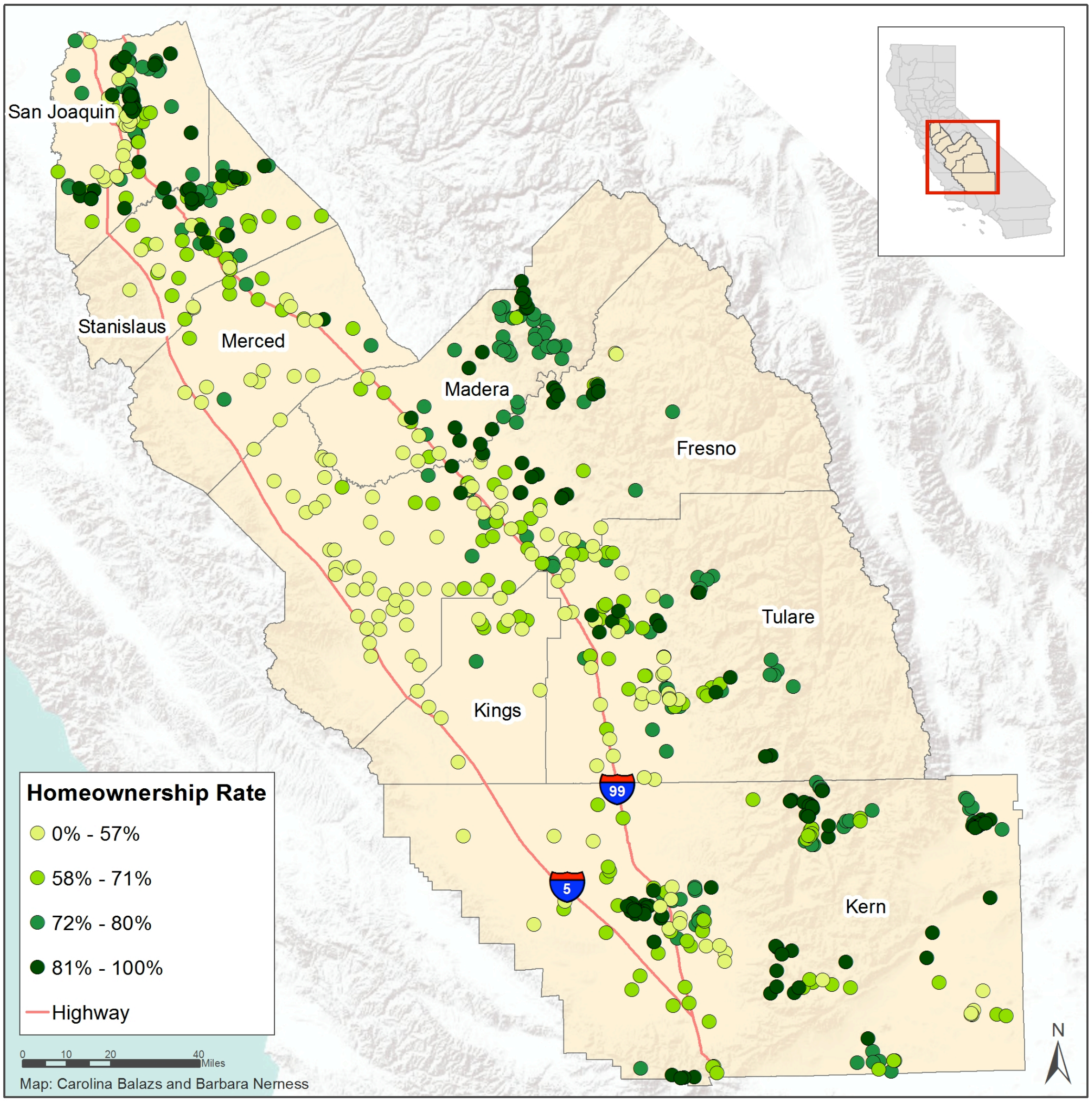

Figure A.13. Map of Homeownership Rate for Active CWS in 2007. Circles depict approximate location of CWS, but not boundaries. Data source: Permits, Inspections, Compliance, Monitoring and Enforcement (PICME) database for location & characteristics of CWS; California Department of Forestry and Fire Protection for County Boundary; US Census 2000 & Balazs et al (2011) for demographics.

Table A.7. Pesticide use in San Joaquin Valley counties with ranking of each county relative to all counties in California, 2000.	
County	Pesticide Use (in pounds applied)/ Rank
Fresno	34,797,885/ 1
Kern	22,570,893/ 2
Kings	5,229,958/ 10
Madera	9,549,731/ 5
Merced	7,621,119/ 7
San Joaquin	11,241,711/ 4
Stanislaus	4,680,374/ 11
Tulare	16,457,558/ 3
Data source: (California Department of Pesticide Regulation 2001)	

Appendix A.2 Steps to determine points-of-entry

The following steps describe the general process I took to determine what water quality samples to include in my analyses noted in *Chapters 3 and 4*, which ultimately define my "points of entry" (see *Chapters 3-4* for a discussion and definition of this term). Specific coding are on file with the author and can be requested, if necessary. This section gives general steps, for those readers interested.

1. Throughout this description, the variable "prim_sta_c" refers to the primary station code of any source. In this setting, a "source" can be an intake source, a well, a treatment plant, etc. Here, source does not only refer to the original source of water.
2. The variable entity_info was used to determine whether any source (i.e., prim_sta_c) was an active source supplying drinking water. Sources coded as: AR (active raw), AT (active treated), AU (active untreated), CM (combination blend mixed), CR (Combination blend raw), CT (Combination blend treated), CU (combination blend untreated), DR (Distribution raw), DT (Distribution treated), PR (purchased raw), PT (purchased treated) were kept. Sources coded as AB (abandoned), AG(Agricultural), DS (Destroyed), IR (Inactive raw), IT (Inactive treated), IU (Inactive untreated), PN (Pending), SR (Standby Raw), ST (Standby treated), SU (standby untreated), MW (Monitoring well), WW (Recycled water) were dropped.
3. Then, using the srec_type code (could be P for plant, S for source, E for point of entry), I selected only sources whose srec_type= "S" to identify whether that source flowed into another source (i.e. treatment plant, another source, etc). To determine this, I used the variable soupath.
4. Separately, I determined which sources were treatment plants, srec_type= "P" by and created a prim_sta_c_plant variable.

5. Then, using the treatment table, I used existing data to identify what kind of treatment the plant had, and whether that treatment might be relevant for arsenic or nitrate
6. Then, for water quality sampling results for each contaminant I identified whether the source (i.e. prim_sta_c) flowed into a plant or not. I coded each prim_sta_c with the following codes: 1= "Source doesn't flow into any other source"; 2= "Source Flows to Plant with relevant treatment"; 3="Source Flows to Plant with treatment, but not relevant for contaminant of interest"; 4= "Plant has relevant treatment but no information on what source flows into it"; 5= "Plant has no treatment information and no source information"; 6= "Plant has relevant treatment and source information"; 7= "Plant has no relevant treatment or source information". These codes thus included information on both the source and the plant.
7. In later modeling steps, I then used these codes to estimate "point-of-entry" sources that would describe water entering the distribution system. For nitrate (*Chapter 3*), for example, I kept the following codes: 1, 3, 4, 6. The choice of inclusion of these codes was meant to include sources that enter directly into the distribution system, or plants for which one could assume the sources flowed in to the system.

Given these steps, there are, of course possibilities of error, deriving from either the raw data itself and how accurate it is, to the assumptions of the steps. To remedy these steps I met with the database manager at DPH that oversees this particular data. And, to address impact of potentially errors, I originally ran sensitivity analyses including all samples to see how sensitive results were to my coding approach. As sensitivity analyses yielded relatively consistent results, I opted to use this coding approach, as I assumed it would best capture "point-of-entry" sources.

Appendix A.3 Interview guides

The following are the interview guides—for informal and formal interviews—used throughout the study. The interview guide for state and county regulators evolved into a stricter format, as noted below. Interviews addressed the following topics: (1) perception of drinking water quality problems and impacts, (2) impacts of historical factors on drinking water quality, (3) impacts of institutional and regulatory factors on drinking water quality, and (4) financial constraints relating to drinking water.

i. *SDWA Regulators: County and State*

Description of Job

1. Describe your job, length of time.
2. What is your role in working with water systems? With the public?
3. How do you work with DPH in Sacramento?
    a. Are there differences in how Sacramento vs Valley staff are involved in the Drinking Water Program (DWP)?
4. Regarding your job description, are there tasks you have to prioritize?
    a. What influences your decision to prioritize these things?

Drinking Water Quality & Impacts

1. Describe drinking water quality in your district. Does this vary from the other districts?
2. Which are the most impacted types of CWS? Why?
3. What would you define as the real impacts?
    a. For example: risk of not monitoring? Not knowing what's in your water?
    b. The exposure to the contaminants?
    c. Inability to fix infrastructure?
4. Is this a general view shared by all District Engineers and/or Sacramento DPH?
5. [later in order] Are there things that the water system can do? Residents can do to address these issues?
6. [later in order] Any reason why homeowners would be in better shape?

Regulatory System

1. Monitoring
    a. Can you walk me through how monitoring happens?
        i. How do you work with systems to meet monitoring requirements?
        ii. What happens if a system is failing to monitor? Do they automatically get a monitoring violation?
        iii. How much priority is given to whether a system monitors?
        iv. Can you describe the ½ the MCL level—how do systems deal with that
            1. Costs of monitoring
        v. State vs County differences in system monitoring

2. Data reporting
    a. Can you walk me through how data reporting happens?
        i. Challenges in this process?
        ii. State vs County

3. MCL Violations
    a. Can you walk me through what you do when a system exceeds an MCL level?
        i. How do you work with systems when they exceed an MCL?
        ii. Are there pieces that you have to prioritize?
            1. Contaminants
            2. Reporting of the MCL—how to decide if an MCL gets recorded
        iii. Difference b/t County & DPH in issuing MCLs?
        iv. In reporting MCLs?
4. How County LPAs work/what are constraints/what do those systems face that other systems may not?
    a. Impact on small systems?

Broad SDWA Questions

1. What is your philosophy in implementing the SDWA?
2. How would you describe the strengths of implementing the SDWA in the region?

3. How would you describe the constraints or challenges you face with regards to implementing the SDWA?
4. What differences do you see in state-regulated versus county-regulated systems/approaches.
5. Does anything need to be changed or improved to address SJV problems—regulations and in policy?
6. What are the drinking water problems that you would fix if you could?
7. Given current reality, what's the most feasible type of solution to the problems your systems face?

ii. Non-profit/For-profit organizations

(1) Drinking water problems and impacts: I will ask each respondent to describe the perceived drinking water problems and impacts for their community water system and systems throughout the region.
(2) Historical factors: I will ask each respondent to describe the historical factors that have influence water quality in their water system and region.
(3) Institutional and regulatory factors: I will ask each respondent to describe how planning, decision-making, agency and regulatory factors have had an impact on their water system and/or community.
(4) Financial constraints: I will ask each respondent to describe the financial reality of providing clean and affordable drinking water drinking water in their area of service.

iii. *Community member interview guide*

(1) Drinking water problems and impacts: I will ask each respondent to describe the perceived drinking water problems and impacts. Potential key areas will be health impacts, environmental impacts, service impacts, community well-being impacts, etc.
(2) Historical factors: I will ask each respondent to describe the historical factors that have influenced water quality in their community or in their general region.
(3) Institutional and regulatory factors: I will ask each respondent to describe how planning, decision-making, agency and regulatory factors have had an impact on their community or their individual life.
(4) Financial constraints: I will ask each respondent to describe the extent of the financial burden of drinking water quality on their individual life, or the life in their community.

APPENDIX B
Appendix to Chapter 3

Appendix B.1 Details on GIS-based estimates of demographics

Estimating Customer Demographics

Digitized Boundaries and Aerial-Weighting Approach

We employed two methods to estimate demographics, and selected one for use in this study. In this first method, we collected hard and digital copies of system boundaries and digitized these in GIS for two pilot counties—Fresno and Tulare. We then estimated water system demographics by using digitized water system boundaries for all CWS in Fresno and Tulare counties and spatially joining these boundaries to block groups in GIS. We used the resulting area of block groups falling within the service area to create an aerial-based weight for the demographics. While aerial weighting is widely used when estimating demographic statistics in GIS, it assumes that the population within the census block (or block group) is homogenously distributed. The formula this approach was:
[1]

$$Z_i = (\sum_{j=1}^{j=n}[(x_j / X_j) * p_j] / \sum_{j=1}^{j=n}[(x_j / X_j) * P_j]) \times 100$$

Where Z is the percent of the variable of interest (i.e. percent Latino) in system *i; j* identifies a particular census block group*;* p_j is the population count of the variable of interest (e.g. white, Latino, number of owner-occupied units, etc) in census block group j; x_j refers to the area of the census block group *j* overlapping with the water system boundaries; X_j refers to the total area of census block group; P_j refers to the total population in census block group j. The numerator is the aerially weighted sum of the variable of interest, whereas the denominator is the population weighted total. Thus, in a universe with one water system and two census block groups (j=1 and 2) overlapping partially with a water system, and where the area of block group 1 is 20 km^2 and the area of block ground 2 is 10 km^2, and the area overlapping with the water system is 1 km^2 and 2 km^2 in block group 1 and 2, respectively, and block 1 has 200 Latinos and block 2 has 100 Latinos, the estimated number of Latinos would be (the numerator):

$[(1km^2/20\ km^2) *200]+[(2km^2/10km^2)*100]=30$ people,
and the fraction of Latinos (if the total weighted population was 100) would be:
(30/100)*100=30% Latino.

Surface intake/well field-based approach

To determine whether we could use a faster estimation procedure we compared the aforementioned approach to a second one, which we ultimately used. This second approach is a population-weighted average that joins surface intake and well field locations (“intakes/fields”, which we also refer to as “sources”) to block groups, but does not weight aerially. Here, the formula is:

[2]

$$Z_i = \sum_{j=1}^{n}(p_j + ...p_n) / \sum_{j=1}^{n}[(P_j + ..P_n]$$

Here, Z_i refers to the percent of the variable of interest in system i, p_j refers to the population count of the variable of interest in census block group j (e.g. number of Latinos), in which a given well field/intake falls; and P_j refers to the total population in block group j.

Assumptions and Sources of Error

Because no demographic information exists for CWS in the Valley, state or nation, both approaches described are estimates of reality. Each contains several sources of error, making demographic estimates from either imperfect, though reasonable, given data limitations.

Sources of error in the boundary-digitized approach (approach 1) can derive from a few key factors. First, while nitrate concentration was modeled for 1999-2001, we collected boundary layers in 2007. For certain water systems—especially large cities (i.e. Fresno and Visalia), the 2007 boundaries would extend further than 2001 boundaries. Second, error can occur from the assumption of homogeneity. In rural areas, where block groups are large, it is possible that there is spatial variability in terms of the population distributed within a block group. Our aerial weighting method assumes homogeneity of distribution of the population. This could mean that systems in more rural areas have more error. In urban areas, a related error could occur. For example, there are water systems served adjacent to, or within, city boundaries (e.g. mobile home parks, or other unincorporated areas). The true demographics of these places could be different from that of the neighboring city, and yet the same block group information is used for both the city and the smaller water system demographics.

In the case of our second approach (using source locations joined to block groups), additional errors can enter. Well fields/intake locations may not fall within the CWS service area, or at least the block group served by the CWS. In addition, not all block groups served by a CWS may have a well field/intake location, causing demographic estimates to rely solely on block groups that have well field/intake location.

Conducting a detailed quantification of the error from the digitized boundary approach is beyond the scope of the paper. However, we quantified potential *sources of error* by seeing how often the aforementioned situations arose using the well field/intake location-based approach. We assessed potential error in two different ways. First, we used our digitized service area boundaries and assessed how close well fields or surface water intakes were to an associated service area. We did this by using two different datasets: 1) 249 digitized water system boundaries for Tulare and Fresno county, 2) all coordinates for intakes/fields for the 249 systems. We used all water systems, not just those in our study sample to have the largest sample. We conducted two different assessments: 1) what fraction of intakes/fields and systems were within the CWS boundary (i.e. service area), 2) what fraction of intakes/fields and systems were within 1000 feet (~.2 miles) of the boundary.

We found the following that 76% of intakes/fields were within the associated CWS service area (or "boundary"). An additional 13% were within 500 feet of the boundary, and an additional 4% (n=65) were within 1000 feet. Thus 93% of sources were within 1000 feet of the service area. In terms of systems, half (n=128) had all of their sources within the CWS boundary, and an additional 25% (n=64) had all of their sources within 500 feet, and 5% more had all their sources within 1000 feet. Thus 82% of CWS had all their source intakes within 1000 feet. This remainder presents some possibility for error. However, this shows that a majority of sources were within 1000 feet of their service area, a relatively short distance (~.20 miles).

Given that we used geographic coordinates of intakes/fields, what is perhaps more pertinent to assess is the fraction of sources that are within a block group that is served by at least part of the CWS. By "served" we mean that a block group boundary physically intersects with the CWS service area, and we thus assume that residents in that portion of the block are "served" by the water system. This can be exemplified in Figure 3.

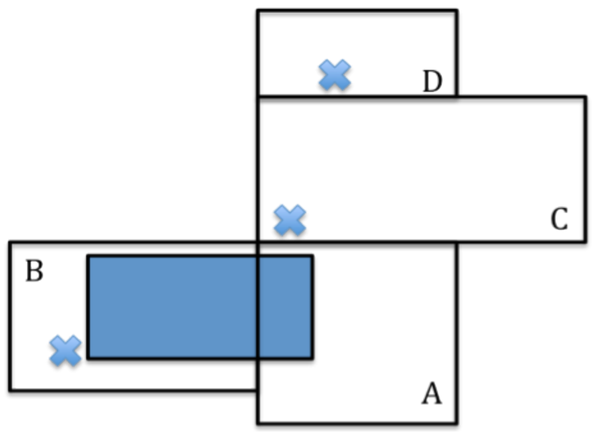

Figure B.1. Schematic of a theoretical example of a community water system (CWS). CWS (shown in blue) that serves portions of two different block groups (A & B), and has well fields or surface water intakes (shown as a blue "x") in three different block groups (B, C, & D), not all of which are served by the CWS.

Ninety-eight percent of sources were within a block group that is served by some portion of the CWS (e.g. block group B, Figure B.1). The remaining 2% of sources were in a block group not served by the CWS (e.g. block groups C and D, Figure 3). At the system level, 93% of systems had all of their intakes/fields within at least one block group that is served by some portion of the CWS (this may mean that not all block groups served by the CWS had a source in them). Among the 7% (n=17) of systems that had at least some of their intakes/fields in block groups not served by CWS (such as the situation shown in Figure B.1), 5 systems did not have any of their intakes/sources in a block group served by the CWS (this would be Figure B.1, but without a well in block group B), representing a key source of error, though the fraction of systems affected is relatively small. Among systems in our study sample, only 1 system had intakes/fields not in a block group served by the CWS, and for this system the wells were in the adjacent block group.

We also quantified the fraction of block groups that intersected with a portion of the CWS boundary (i.e. residents in this area would be "served" by the CWS) but did not have a source located within the block group (e.g. block group A, Figure B.1). We found that 491 block groups (among 106 CWS) do not have an intake/field located within them. This represents a significant fraction of the total systems assessed (42%), and likely explains some of the difference when comparing our aerial weighting method to our intake/field method (suggesting we would encounter similar error in our study sample). However, without knowing the true demographic of each system, it is difficult to say how much this impacts the demographic estimate.

For our study sample, for eight of the ten systems whose average was over the MCL, all sources were within the CWS service area and shared the same block groups as those served by the CWS. The ninth system had all sources within the same block group as that served by the CWS, and the sources were within 500 feet of the community. The tenth had two-thirds of its sources in block group not served by the CWS. Thus, while there may be some error due to our use of block group estimates, we expect minimal error for these systems.

In sum, we have two main sources of error (between the two different methods): 1) when a block group that is not served by CWS boundary is included in the point-based method, 2) when a block group that is served by the CWS does not have a source located within it and is excluded from the point-based method.

Goodness-of-fit Test

We compared our two approaches—digitized boundary with aerial weighting to point-based estimates by running a goodness-of-fit test regressing the point-based estimates against the aerial-weighted demographic, for percent Latino (Table B.1) and percent homeownership (Table B.2). This allowed us to assess how close both methods were to each other. By examining the R^2 values for our two key variables of interest (percent Latino and percent home ownership) we determined our source-based approach reasonably (i.e. $R^2>=.80$) resembled the digitized approach, especially for the percent Latino estimate. The R^2 is lower for home ownership ($R^2=.48$). But, since neither approach is the "gold standard", given spatial assessments and the fact that digitized boundaries were not available across the Valley, we concluded that using source locations was a reasonable approach.

Table B.1. Regression† of estimated percent Latino customers in Community Water Systems (CWS) using digitized CWS boundaries against estimated percent Latino estimated using source coordinates.

Variable	Coefficient
Constant	3.17* (-.51, 6.8)
% Latino	.92*** (.86, .98)

†Regression for 224 water systems in pilot study comparison of methods in Tulare and Fresno Counties. Systems with geographic coordinates and digitized boundaries (n=224) in Tulare and Fresno county pilot study included for comparison.
$R^2=.80$
* p<0.10, ** p<0.05, *** p<0.01

Table B.2. Regression† for estimated percent home ownership in Community Water Systems (CWS) using digitized CWS boundaries against percent home ownership estimated using source coordinates.	
Variable	Coefficient
Constant	34.48 (-3.0, 11.9)
% Home Ownership	.88*** (.76, .99)

†Regression for 224 water systems in pilot study comparison of methods in Tulare and Fresno Counties. Systems with geographic coordinates and digitized boundaries (n=224) in Tulare and Fresno county pilot study included for comparison.
R^2=.50
* p<0.10, ** p<0.05, *** p<0.01

Appendix B.2 Variability of nitrate levels across sources

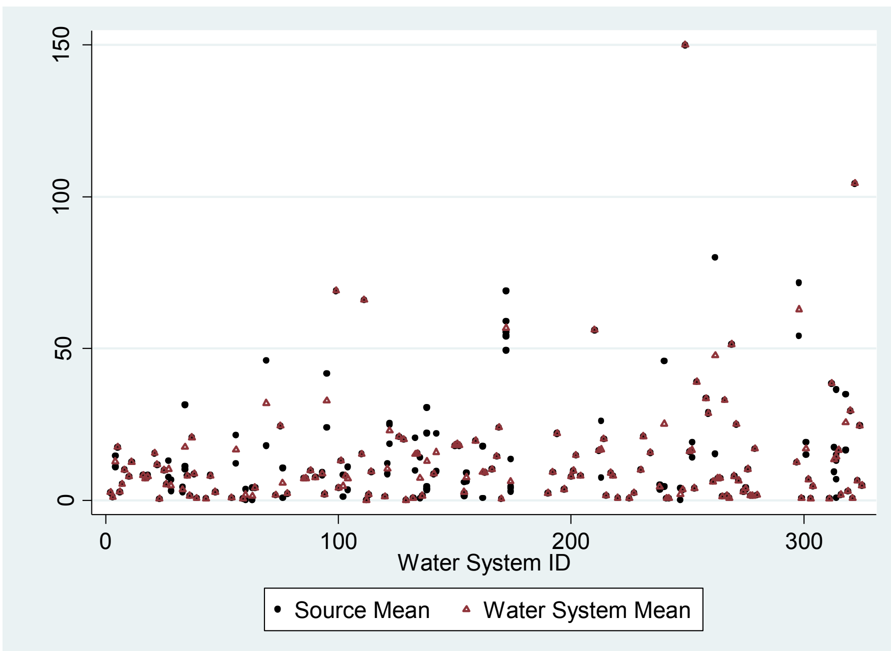

Figure B.2. Variability of mean nitrate concentration for point-of-entry sources within water systems for water systems† under 200 connections (n=160).
† Individual water systems are identified by water system ID on the x-axis.

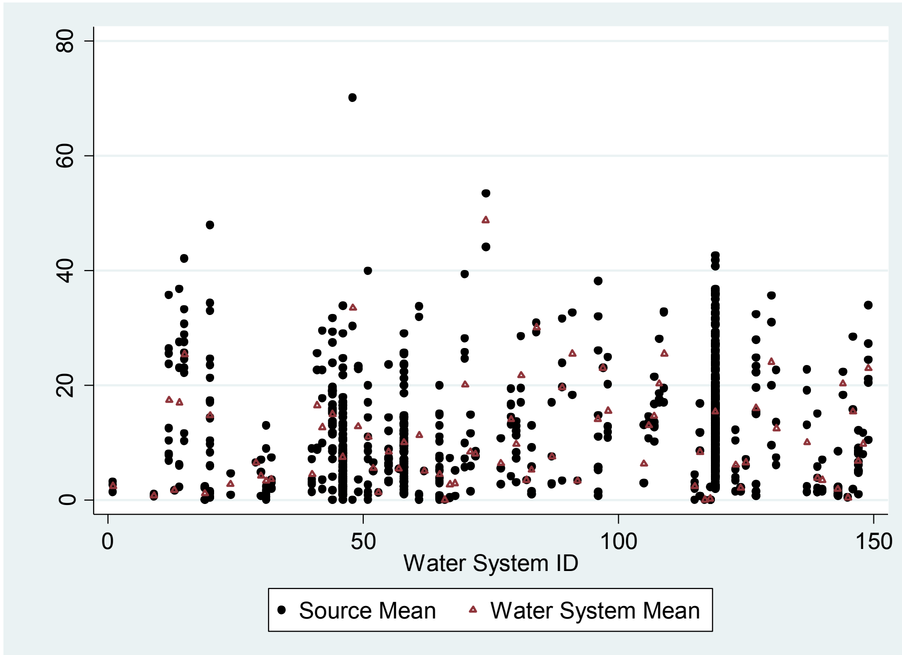

Figure B.3. Variability of mean nitrate concentration for point-of-entry sources within water systems for water systems[†] over 200 connections (n=167).
[†] Individual water systems are identified by water system ID on the x-axis. For ease of presentation, first half of systems included in this figure.

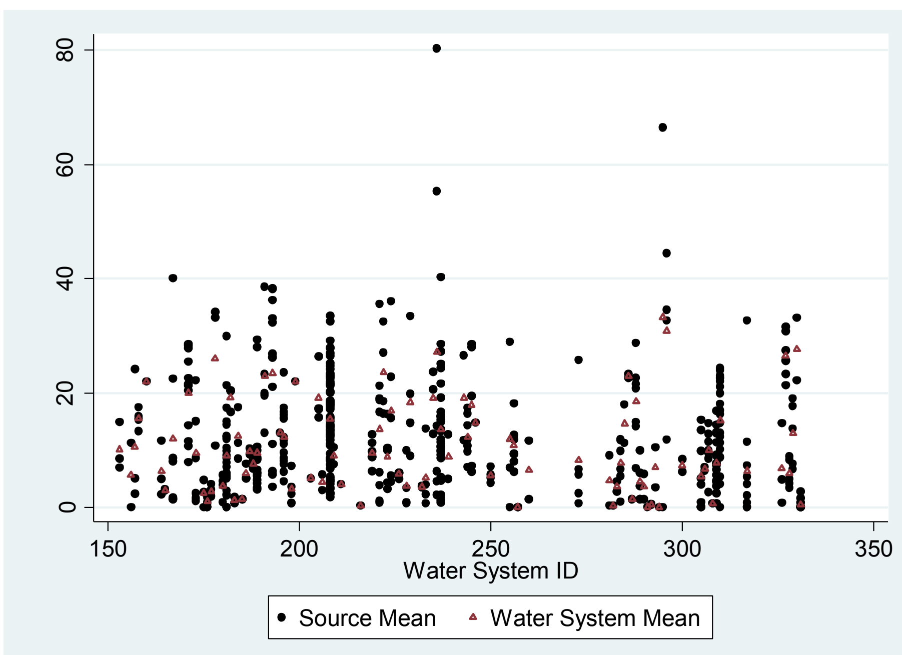

Figure B.4. Variability of mean nitrate concentration across sources within water systems for water systems[†] over 200 connections (n=167).
[†] Individual water systems are identified by water system ID on the x-axis. For ease of presentation, second half of systems included in this figure.

Appendix B.3 Statistics on Nitrate Samples per Source

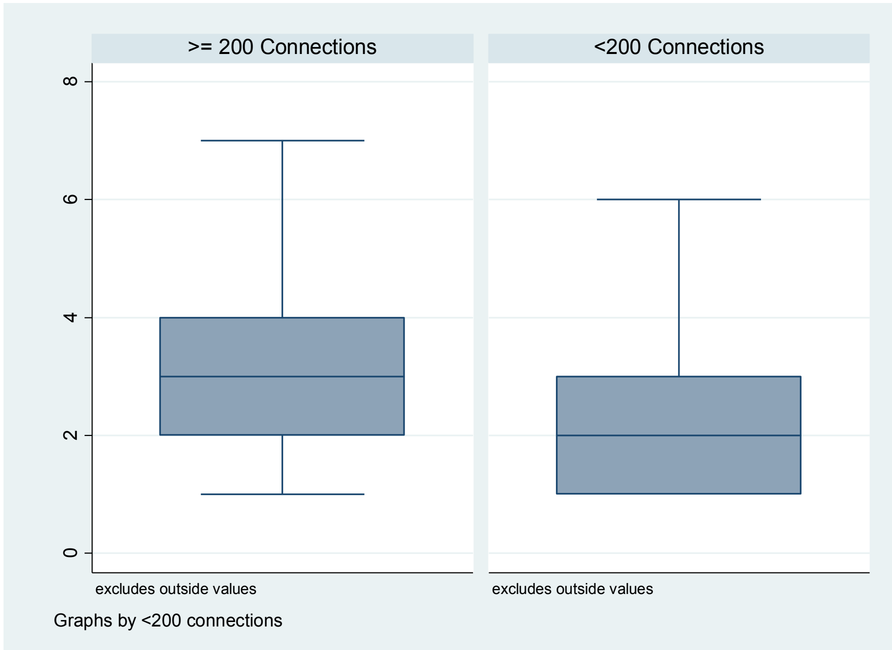

Figure B.5. Box plot of number of samples per source, f1999 to 2001.

Appendix B.4 Distribution of nitrate concentration at system, source and sample-level

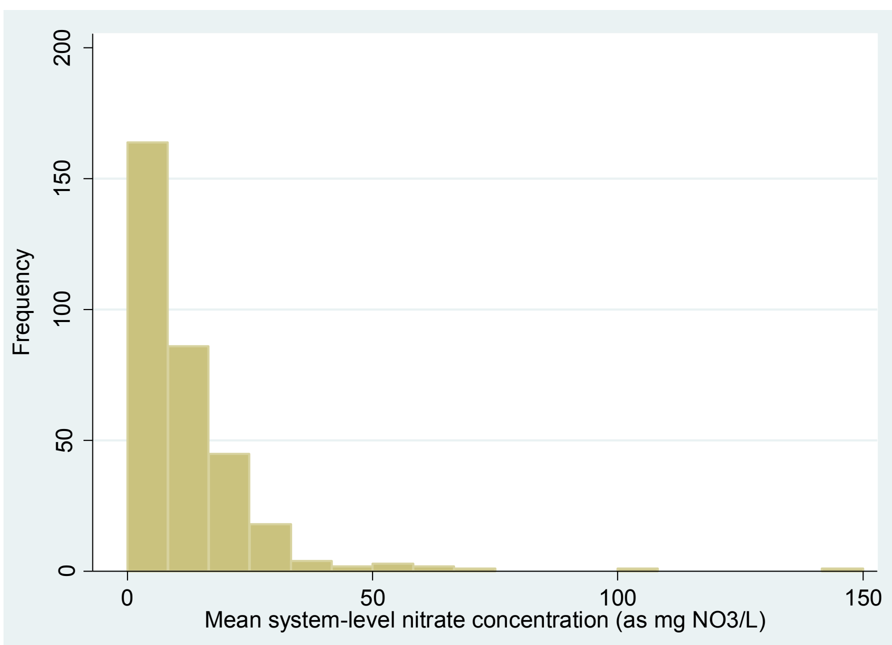

Figure B.6. Histogram of mean system-level nitrate concentrations, for all community water systems (CWSs) in nitrate study sample (n=327), 1999-2001.

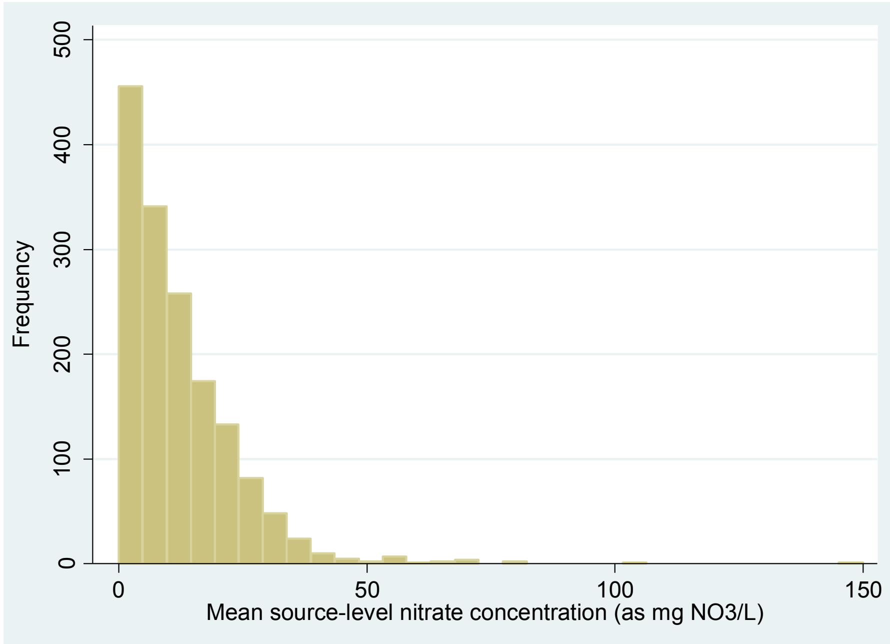

Figure B.7. Histogram of mean source-level nitrate concentrations (1,551 sources), across all community water systems (CWSs) in nitrate study sample (n=327), 1999-2001.

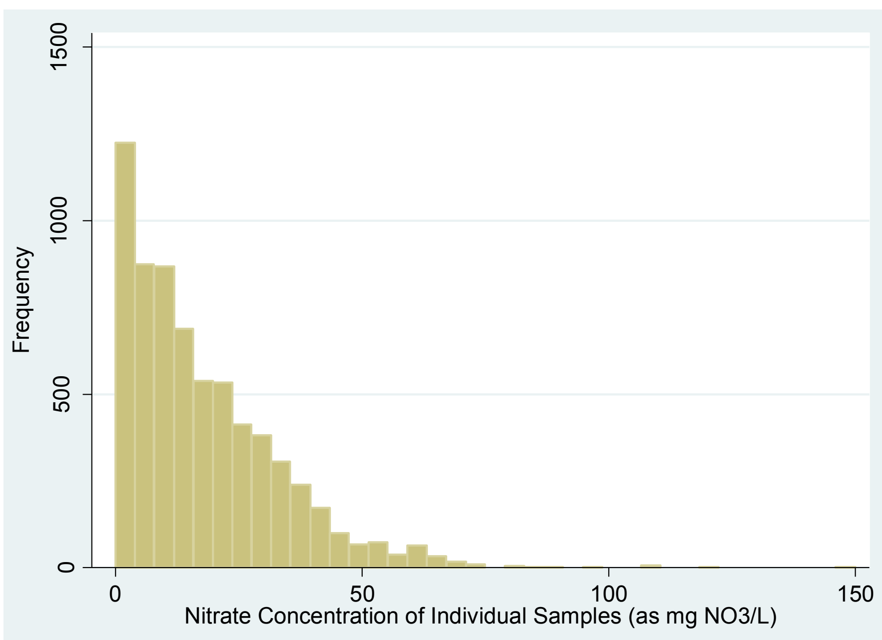

Figure B.8. Histogram of nitrate concentrations for individual samples (n=6,660), across all sources and systems, 1999-2001.

Appendix B.5 Results of original multi-level model

This appendix presents the original multi-level model first used before conducting residual assessments. The outputs highlight the sensitivity analyses and residual diagnostics we conducted to test for the appropriateness of this model. These requests were partially requested by reviewers of our journal article. After conducting a second residual diagnostic, we found that the assumptions of normality were being violated. This prompted us to explore the sensitivity of four other model options.

The following sections present the following model outputs and residual diagnostics: 1) Original xtmixed model, 2) Original xtmixed model, stratified by size, 3) Original model, excluding residuals whose absolute values are greater than 10, 4) Original model, that uses the square root of the findings (as the distribution of the residuals in the original model looked more like it was a negative binomial distribution, rather than using raw nitrate samples, 5) Clustered model, where robust standard errors are used, but outcomes are clustered at the system level. This model also drops the variable "sources" as we found that it was highly correlated with number of connections (i.e. both are measures of system size in some way). We ran the clustered model with sources as well, but as the one without sources is our "final" model, we show these results alone. The follow tables and figures explain the step-by-step process of selecting the "final" model presented in *Chapter 3*.

Table B.3. Multi-level regression for factors associated with nitrate concentration in community water system. Coefficients represent change in mean concentration for unit change in covariate (95% CI).

Variable	Model A[a]	Model B[a]	Model C[b]	Model D[c]	Model E[d]
Fixed Effects					
Constant	8.9 (5.8 ,12.0)	25.5 (-.29 , -.10)	-985 (-1405 , -564)	-1398 (-3483- 686)	-925 (-1334, 517)
% Latino	.11 (.06 ,.16)		.07 (-.005 , .15)	.19 (.02 , .36)	.009 (-.06 , .08)
% non-Latino people of color	-.06 (-.30 ,.18)		-.14 (-.36 ,.13)	-.10 (-.60 ,.39)	-.18 (-.40 , .05)
% Home ownership		-.19 (-.28 , -.10)	-.14 (-.26 , -.02)	-.25 (-.49 , -.02)	-.04 (-.15 , .07)
Incorporated			-.68 (-4.7 , 3.3)	6.2 (-17.4 , 29.9)	1.3 (-1.6 , 4.1)
No. of sources with samples			.12 (-.16 , .41)	3.4 (.87 , 6.0)	.01 (-.18 , .20)
Groundwater or combined			.3 (-6.0 , 18.0)	na	8.8 (-.99 ,18.6)
Private non-PUC regulated			2.1 (-2.8 , 7.1)	2.9 (-6.2 , 12.11)	
Public			-.19 (-5.1 , 4.8)	-4.5 (-20.5 ,11.4)	.11 (-3.8 , 4.0)
< 200 service connections			3.8 (-.32 , 7.1)	na	na
Valley floor			3.2 (-.47, 6.8)	4.9 (-1.4 , 11.2)	2.0 (-1.9 , 5.9)
Year			.50 (.29 , .71)	71 (-.33 , 1.75)	.46 (.26 ,.67)
Summer/fall			1.3 (.88 , 1.6)	3.2 (1.4 , 5.0)	1.1(.69 , 1.4)
Random Effects					
SD Ψ^{3e}	10.7 (.64)	10.8 (.64)	10.4 (.63)	13.2 (1.3)	6.5 (.55)
SD Ψ^{2f}	7.9 (.19)	7.9 (.19)	7.9 (.19)	8.1 (1.2)	7.8 (.18)
SD Ψ^{1g}	6.4 (.06)	6.4 (.06)	6.3 (.06)	9.8 (.31)	5.9 (.06)
N (CWS), level 3	327	327	327	157	170
N (sources), level 2	1551	1551	1551	217	1334
N (observations), level 1	6660	6660	6660	701	5959

[a] Unadjusted models [b] Adjusted model [c] Fewer than 200 service connections [d] Greater than or equal to 200 connections [e] Between-system standard deviation= square root of variance of random intercept for system. [f] Between-source standard deviation=square root of variance of random intercept for source. [g] Standard deviation of observations within systems and source.

As requested by journal reviewers, we then included Year dummies (Table 2), and both the histogram of residuals and the q-q plot that shows our residual diagnostics, and normality assumptions being violated. This led us to conclude that the xtmixed was giving very biased inference because this difference is not just due to the slight changes in the coefficients.

Table B.4. Mixed model with year dummies included

Mixed-effects REML regression				Number of obs	= 6660	
	Observations per Group					
Group Variable	No. of Groups	Minimum	Average	Maximum		
system_no	327	1	20.4	1219		
prim_sta_c	1551	1	4.3	133		
Log restricted-likelihood = -23394.836				Wald chi2(13)	= 102.45	
				Prob > chi2	= 0.0000	
finding	**Coef.**	**Std. Err.**	**z**	**P>|z|**	**[95% Conf. Interval]**	
% Latino	.069	.039	1.77	0.077	-.007	.146
% people of color	-.118	.127	-0.92	0.356	-.368	.1327
% home owner	-.14	.0612	-2.35	0.019	-.263	-.023
incorporated	-.542	2.045	-0.26	0.791	-4.55	3.46
# sources	.021	.0384	0.55	0.584	-.054	.096
Gw or gwsw	6.54	6.289	1.04	0.298	-5.78	18.86
Private non-puc	2.15	2.561	0.84	0.400	-2.86	7.17
public	-.288	2.549	-0.11	0.910	-5.28	4.70
<200 conn.	3.47	2.16	1.60	0.109	-.777	7.72
In valley	3.13	1.877	1.67	0.096	-.552	6.80
2000	.389	.211	1.84	0.066	-.025	.803
2001	.987	.214	4.60	0.000	.567	1.41
Summer/fall	1.258	.190	6.62	0.000	.885	1.63
_cons	8.736	8.432	1.04	0.300	-7.79	25.26

Random-effects Parameters	**Estimate**	**Std. Err.**	**[95% Conf. Interval]**	
system_no: Identity				
sd(_cons)	10.44	.634	9.27	11.76
prim_sta_c: Identity				
sd(_cons)	7.87	.190	7.51	8.25
sd(Residual)	6.35	.062	6.23	6.47

LR test vs. linear regression: chi2(2) = 7474.48 Prob > chi2 = 0.0000

Note: LR test is conservative and provided only for reference.

The histogram of residuals for the mixed model looked normal, with the exception of outliers that are almost not visible with this plot.

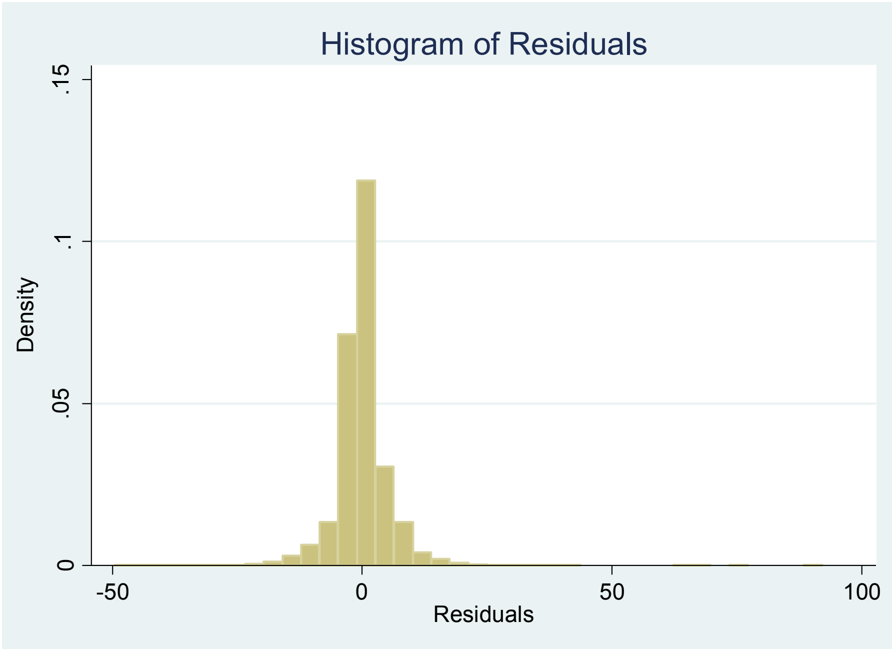

Figure B.9. Histogram of residuals for mixed model.

To further test for normality, I ran a q-q plot of residuals. Under normality conditions, the line residuals should plot linearly, but they did not.

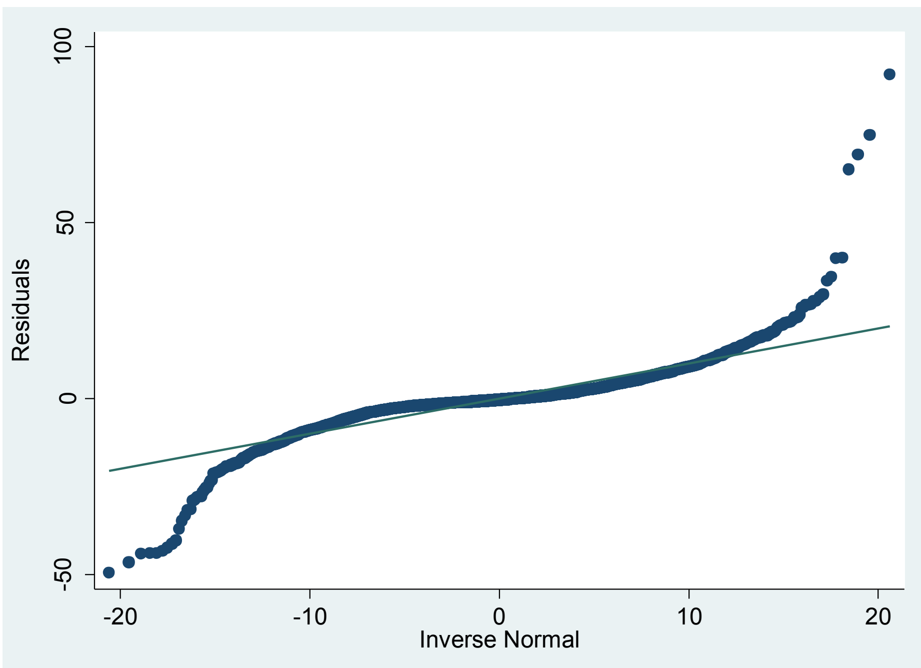

Figure B.10. Q-Norm plot of residuals, all systems included.

Table B.5a. Stratified results for CWS with >199 connections using original mixed model with year dummies.

Mixed-effects REML regression				Number of obs		= 5947
	Observations per Group					
Group Variable	No. of Groups	Minimum	Average	Maximum		
system_no	167	1	35.6	1219		
prim_sta_c	1329	1	4.5	133		
Log restricted-likelihood = -20383.357				Wald chi2(12)		= 102.45
				Prob > chi2		= 0.0000
finding	**Coef.**	**Std. Err.**	**z**	**P>|z|**	**[95% Conf. Interval]**	
% Latino	.009	.034	0.27	0.784	-.0584	.077
% POC	-.185	.117	-1.58	0.115	-.4171	.045
% home owner	-.0407	.0575	-0.71	0.478	-.1535	.071
incorporated	1.157	1.486	0.78	0.436	-1.755	4.06
# of sources	.0111	.0249	0.45	0.656	-.0378	.0600
Gw or gwsw	8.752	5.003	1.75	0.080	-1.052	18.55
Private non-puc	-1.246	2.602	-0.48	0.632	-6.347	3.855
public	.206	2.015	0.10	0.918	-3.743	4.155
In valley	2.048	2.027	1.01	0.312	-1.925	6.021
2000	.469	.2042	2.30	0.021	.0694	.8699
2001	.933	.2086	4.47	0.000	.5247	1.342
Summer/fall	1.060	.185	5.72	0.000	.6971	1.423
_cons	3.217	7.276	0.44	0.658	-11.04	17.47

Random-effects Parameters	**Estimate**	**Std. Err.**	**[95% Conf. Interval]**	
system_no: Identity				
sd(_cons)	6.57	.559	5.565	7.768
prim_sta_c: Identity				
sd(_cons)	7.81	.185	7.45	8.188
sd(Residual)	5.88	.060	5.762	6

LR test vs. linear regression: chi2(2) = 6377.85 Prob > chi2 = 0.0000

Note: LR test is conservative and provided only for reference.

Table B.5b. Stratified results for CWS with <200 connections
using original mixed model with year dummies.

Mixed-effects REML regression	Number of obs	= 713

	Observations per Group			
Group Variable	No. of Groups	Minimum	Average	Maximum
system_no	160	1	4.5	110
prim_sta_c	222	1	3.2	110

Log restricted-likelihood = -2808.333	Wald chi2(11)	= 52.51
	Prob > chi2	= 0.0000

finding	Coef.	Std. Err.	z	P>|z|	[95% Conf. Interval]	
% Latino	.197	.0884	2.24	0.025	.0245	.3710
% POC	-.115	.2512	-0.46	0.647	-.605	.3774
% home owner	-.245	.1215	-2.02	0.044	-.483	-.0071
incorporated	5.118	12.29	0.42	0.677	-18.98	29.22
# sources	2.431	1.478	1.64	0.100	-.467	5.329
Private non-puc	6.007	4.96	1.21	0.227	-3.73	15.74
Public	-1.895	8.166	-0.23	0.816	-17.901	14.11
In valley	4.229	3.237	1.01	1.31	-2.115	10.57
2000	-.2838	1.125	-0.25	0.801	-2.488	1.921
2001	1.226	1.074	1.14	0.253	-.8781	3.332
Summer/fall	2.992	.9145	3.27	0.001	1.199	4.784
_cons	14.746	11.42	1.29	0.197	-7.652	37.14

Random-effects Parameters	Estimate	Std. Err.	[95% Conf. Interval]	
system_no: Identity				
sd(_cons)	13.57	1.309	11.23	16.401
prim_sta_c: Identity				
sd(_cons)	7.912	1.191	5.891	10.62
sd(Residual)	9.706169	.3060511	9.124479	10.32494

LR test vs. linear regression: chi2(2) = 690.11 Prob > chi2 = 0.0000

Note: LR test is conservative and provided only for reference.

To test for alternative model types, I used the original model and attempted to drop outliers from the analysis, by excluding those residuals whose absolute value was greater than 10 (we also examined this for absolute values greater than 5). Upon checking coefficients and standard errors, I found that while the coefficient stays rather constant on percent Latino, the coefficient and standard error changes significantly on home ownership. Because of this, we determined that even after excluding outliers, the xtmixed model is giving biased inference. Had the exclusion of the outliers not led to significant differences in our estimates, inference and conclusions we could have concluded that they did not significantly contribute to either the estimates or inference. But this was not, in fact, the case.

Table B.6. Mixed models keeping systems only if absolute(residual)<10

Mixed-effects REML regression				Number of obs		= 6240
Group Variable	No. of Groups	Minimum	Average	Maximum		
		Observations per Group				
system_no	324	1	19.3	1199		
prim_sta_c	1540	1	4.1	98		
Log restricted-likelihood = -18757.617				Wald chi2(12)		= 84.94
				Prob > chi2		= 0.0000
finding	**Coef.**	**Std. Err.**	**z**	**P>|z|**	**[95% Conf. Interval]**	
% Latino	.0654427	.0351793	1.86	0.063	-.0035074	.1343929
% POC	-.0992326	.1138568	-0.87	0.383	-.3223879	.1239226
% home owner	-.0487532	.0555519	-0.88	0.380	-.1576329	.0601265
incorporated	-.0258445	1.821611	-0.01	0.989	-3.596137	3.544448
# sources	.1322636	.1305169	1.01	0.311	-.1235449	.3880721
Gw or gwsw	6.747713	5.661021	1.19	0.233	-4.347684	17.84311
Private non-puc	1.899885	2.261722	0.84	0.401	-2.533009	6.332779
public	.7787303	2.268881	0.34	0.731	-3.668194	5.225655
<200 Conn	3.263749	1.896535	1.72	0.085	-.4533915	6.98089
In valley	2.864651	1.877437	1.71	0.088	-.4222844	6.151587
year	.289863	1.677039	5.04	0.000	.1770485	.4026775
Summer/fall	.6143538	.1046236	5.87	0.000	.4092954	.8194122
_cons	-578.107	115.3824	-5.01	0.000	-804.2523	-351.9617

Random-effects Parameters	**Estimate**	**Std. Err.**	**[95% Conf. Interval]**	
system_no: Identity				
sd(_cons)	9.220376	.5585297	8.188165	10.38271
prim_sta_c: Identity				
sd(_cons)	7.995799	.1723083	7.665113	8.34075
sd(Residual)	3.270616	.0336307	3.205361	3.3372

LR test vs. linear regression: chi2(2) = 12678.59 Prob > chi2 = 0.0000

Note: LR test is conservative and provided only for reference.

Table B.7. Stratified model for systems with <200 connections, including systems with absolute(residuals)<10.

Mixed-effects REML regression				Number of obs		= 653

Group Variable	No. of Groups	Observations per Group Minimum	Average	Maximum
system_no	157	1	4.2	85
prim_sta_c	218	1	3.0	85

Log restricted-likelihood = -2010.6988	Wald chi2(11)	= 41.82
	Prob > chi2	= 0.0000

finding	Coef.	Std. Err.	z	P>|z|	[95% Conf. Interval]	
% Latino	.1805632	.0754895	2.39	0.017	.0326064	.32852
% POC	-.0446494	.2160003	-0.21	0.836	-.4680021	.3787034
% home owner	-.0529085	.1062135	-0.50	0.618	-.261083	.155266
incorporated	2.805334	10.51154	0.27	0.790	-17.7969	23.40757
Sources	2.675071	1.276382	2.10	0.036	.1734075	5.176734
Private non-puc	5.704552	4.30302	1.33	0.185	-2.729212	14.13832
public	.4610844	6.999208	0.07	0.947	-13.25711	14.17928
In valley	3.516442	2.76463	1.27	0.203	-1.902134	8.935017
2000	.7406234	.347254	2.13	0.033	.0600181	1.421229
2001	1.1516	.332477	3.46	0.001	.4999573	1.803243
Summer/fall	.4622626	.2987012	1.55	0.122	-.123181	1.047706
_cons	.4711525	9.914725	0.05	0.962	-18.96135	19.90366

Random-effects Parameters	Estimate	Std. Err.	[95% Conf. Interval]	
system_no: Identity				
sd(_cons)	11.92714	1.081089	9.985787	14.2459
prim_sta_c: Identity				
sd(_cons)	8.349043	.7944669	6.928499	10.06084
sd(Residual)	2.825555	.0958902	2.643728	3.019888

LR test vs. linear regression: chi2(2) = 1644.17 Prob > chi2 = 0.0000

Note: LR test is conservative and provided only for reference.

Because the q-norm plot looked somewhat like a negative binomial distribution, we also took the square root of the findings and used this as the outcome variable (Table B.6). This has the drawback of having less easy-to-interpret coefficients. But even after checking residuals, the assumption of normality is not met. So we decided this was still not a reasonable approach to take.

Table B.8. Mixed model using square root of findings.

Mixed-effects REML regression			Number of obs		=	6660

	Observations per Group			
Group Variable	No. of Groups	Minimum	Average	Maximum
system_no	327	1	20.4	1219
prim_sta_c	1551	1	4.3	133

Log restricted-likelihood = -9290.1534	Wald chi2(13)	= 105.12
	Prob > chi2	= 0.0000

finding	Coef.	Std. Err.	z	P>|z|	[95% Conf. Interval]	
% Latino	.0081323	.004763	1.71	0.088	-.0012029	.0174675
% POC	-.0118366	.0155282	-0.76	0.446	-.0422713	.0185981
% home owner	-.0097029	.0074615	-1.30	0.193	-.0243272	.0049215
incorporated	.179602	.2434086	0.74	0.461	-.2974702	.6566741
sources w	.021072	.0384337	0.55	0.584	-.0542567	.0964007
Gw or gwsw	1.789062	.7882053	2.27	0.023	.2442078	3.333916
Private non-puc	.4839952	.3091396	1.57	0.117	-.1219073	1.089898
public	.1197855	.3039967	0.39	0.694	-.4760371	.7156081
<200	.3452339	.2624288	1.32	0.188	-.1691171	.8595849
In valley	.4890554	.2288794	2.14	0.033	.04046	.9376507
2000	.0419373	.024412	1.72	0.086	-.0059093	.0897839
2001	.1099545	.0247756	4.44	0.000	.0613951	.1585138
Summer/fall	.1452352	.0220262	6.59	0.000	.1020646	.1884058
_cons	.7150814	1.043063	0.69	0.493	-1.329285	2.759448

Random-effects Parameters	Estimate	Std. Err.	[95% Conf. Interval]	
system_no: Identity				
sd(_cons)	1.197437	.0756389	1.057997	1.355254
prim_sta_c: Identity				
sd(_cons)	1.144232	.0261529	1.094105	1.196657
sd(Residual)	.7321485	.0072211	.7181314	.7464393

LR test vs. linear regression: chi2(2) = 7838.12 Prob > chi2 = 0.0000

Note: LR test is conservative and provided only for reference.

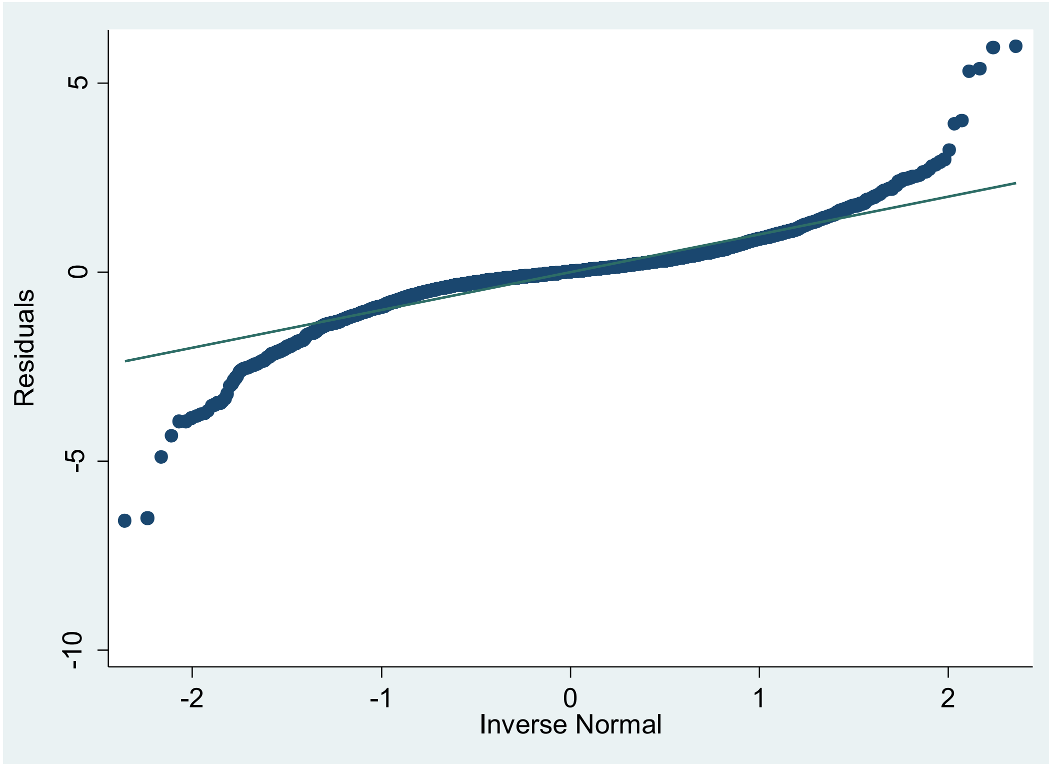

Figure B.11. Q-Norm plot where outcome variable equals log(nitrate sample).

Ultimately, we decided to use a clustered model with robust standard errors. This final model included year dummies (as requested by a reviewer) and dropped number of sources since it was highly correlated with connections. Because the inference is not robust to misspecification of the model, and the diagnostics above suggest model-based inference would have been biased, we report robust standard errors. The results of this final model are included in *Chapter* 3. This final model allowed for arbitrary correlation within the highest level of the hierarchy (i.e. water system), which is the only level I had to thus specify.

Appendix B.6 Characteristics of CWS With Average Nitrate Over the MCL

Table B.9. Description of 10 systems in study sample whose average nitrate concentration was over the MCL.

System	Ownership Type	# of Sources with samples (proxy for # of sources)	Estimated average nitrate concentration[a] (mg NO_3/L)	Years for which MCL violations issued and associated nitrate concentration (nitrate concentration) [b]	Difference[c] between estimated average nitrate concentration and reported concentration in year for which MCL violation was given (mg NO_3/L)
1	City Tract	2	48.7	No violation in time period[d]	0
2	Private, Mutual	1	69	2001 (69)	-44
3	Private, mutual	1	66	2000 (110)	-1.7 in 1999 -10.3 in 2000
4	Irrigation District	6	56.7	1999 (57) 2000 (67)	-16.9 in 2000 -21.9 in 2001
5	Private, Labor Center	2	56.1	2000 (73) 2001 (78)	-16.9 in 2000 -21.9 in 2001
6	Private, Labor Camp	1	150	2000 (150)	0
7	Private, Mutual	2	47.6	1999 (80)	32.4
8	Private, Mutual	1	51.3	2000 (48)	-3.3
9	Private, Mutual	3	62.8	2000 (47.9) 2001 (54)	14.9 in 2000 8.8 in 2001
10	Private, Labor Center	1	104.4	1999 (115) 2000 (106) 2001 (96.75)	-10.6 in 1999 -1.6 in 2000 7.65 in 2001

[a] Estimated average nitrate concentration derived from study.
[b] Data source for year of violation and concentration of violation (at the source-level) derived is the Permits, Inspections, Compliance, Monitoring and Evaluation (PICME) database. Nitrate concentration in mg NO3/L.
[c] Where a system had more than one year with a violation, the difference is noted for each year. A negative number denotes that system-level average was below the concentration for which the MCL was given.
[d] MCL violation in 1998 for 46.5 mg NO3/L, just one year before study period.