Chapter 5: Ecosystem services and human well-being

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Section 5.2: Clean drinking water

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Main Messages

The concentration of nitrate in California's surface water bodies seldom exceeds the federal maximum contaminate level (10 mg nitrate-N L^{-1}). As such, the use of surface water sources for drinking is generally considered low risk.

Nitrate levels in groundwater have increased over the past several decades, and in some parts of the state now exceed federal drinking water standards. This trend is likely to continue due to the time lag between the loss of nitrogen (N) to the environment and its accumulation in aquifers.

People in agricultural areas, particularly those with domestic wells, are more likely to be exposed to high levels of nitrate in their drinking water than those in urban and suburban areas. Groundwater from wells in the Tulare Lake Basin and Salinas Valley regularly exceed the federal MCL and an estimated 8.0-9.4% of residents (212,500 – 250,000 people) in these areas are "highly susceptible" to exposure to water in excess of 10 mg nitrate-N L^{-1}.

For most adults, the amount of nitrate and nitrite consumed via foods is much greater than the amount consumed through drinking water. Infants given water or foods high in nitrate can develop "blue-baby syndrome", a potentially fatal condition where their blood cannot transport oxygen.

The International Agency for Research in Cancer concluded that nitrate and nitrite are "probably carcinogenic to humans". Nitrate and nitrite can form nitrosamines, which are suspected to cause cancer. Consumption of nitrate and nitrite from all drinking water and food sources such as preserved meats, are associated with stomach cancer in some studies.

Nitrate and nitrite can have positive effects on the body. In some patients they are used to treat high blood pressure and reduce the risk of stroke.

Costs of treating nitrate contaminated drinking water can pose a significant financial burden on low-income households and the public and community water systems that serve disadvantaged communities. While state-wide estimates of the cost to address nitrate in public and community water systems are needed, recent studies suggest that an increase in public and private funding on the order of $17 – 34 million per year over many decades will be needed to implement required nitrate mitigation projects for water systems in the Tulare Lake Basin and Salinas Valley.

5.2 Clean drinking water

5.2.1. Trends in indicators of water quality

5.2.1.1 Maximum contaminant levels (MCL) for nitrate and nitrite in drinking water

This section describes the chemical and physical processes that affect N in California's drinking water, discusses the spatial and temporal patterns of N in surface and groundwater resources, as well as the human health and economic impacts. Drinking water in California is supplied by both surface and groundwater, with approximately 40% of the population in part relying on groundwater as a source for their drinking water (CDWR 2003). Drinking water is protected by regulating both the water sources and the water suppliers. California treats surface water and groundwater separately although they are physically linked (Figure 5.2.1). In general, the United States Environmental Protection Agency (US EPA) regulates surface water under the Clean Water Act, while the State Water Resources Control Boards implement federal regulations. In contrast, groundwater quality is regulated at the state and local level. Regardless of the source, the US EPA under the authority of the Safe Drinking Water Act has set maximum contaminant levels (MCL) of 10 mg nitrate-N L^{-1} and 1 mg nitrite-N L^{-1} for public drinking water supplies (US EPA 2009). "Self-supplied water systems" (domestic wells serving 1-2 households), "local small water systems" (systems serving 2-4 households) and "state small water systems" (systems serving 5-14 households) are not subject to this water quality regulation (see Glossary). Note that the units used by the US EPA are based on the mass of N in the nitrate or nitrite (e.g., mg nitrate-N L^{-1}), whereas European standards and guidelines are based on the mass of the nitrate molecule (e.g., mg $NO_3^- L^{-1}$). As such, 10 mg nitrate-N L^{-1} is equivalent to 45 mg $NO_3^- L^{-1}$ and 1 mg nitrite-N L^{-1} is equivalent to 3.3 mg $NO_2^- L^{-1}$.

[Figure 5.2.1]

5.2.1.2 The chemical and physical basis of nitrogen in drinking water

Sources of N in surface and groundwater include weathering of bedrock, mineralization of organic N in soil, atmospheric deposition of N, N fertilizers, livestock waste, septic systems and wastewater treatment plants (see mass balance in Chapter 4 for relative magnitudes). Ammonium (NH_4^+) and nitrate (NO_3^-) are the most abundant forms of reactive N that impact the quality of surface and groundwater resources in California. Since ammonium is positively charged it tends to adsorb to negatively charged soil particles and is thus not easily leached from the soil. However, under aerobic conditions ammonium is rapidly oxidized by microbes first to nitrite (NO_2^-) and then to nitrate through the process of nitrification. Nitrate is stable under aerobic conditions and highly mobile due to its negative charge and solubility in water. Hence, nitrate is generally the dominant form of N in both surface and groundwater. Although nitrite is chemically unstable and prone to oxidation to nitrate, it can also accumulate in surface and ground water to a limited extent. Denitrification, which converts nitrate to gaseous nitrogen (N_2) under anaerobic conditions, is the main pathway that removes N from surface and groundwater. When the rate of groundwater denitrification is low due to groundwater being high in oxygen and/or low in carbon, it takes many decades to attenuate the high groundwater nitrate loads (Green et al. 2008). In many of California's shallow, unconfined aquifers which experience intermittent recharge, the conditions are favorable for high dissolved oxygen and low organic carbon. Nitrous oxide (N_2O) is an intermediate product of both nitrification and denitrification. Since nitrous oxide is a potent greenhouse gas, it is an important driver of climate change. The amount of nitrous oxide released to the atmosphere will depend on environmental conditions and estimates of its fluxes have a high degree of uncertainty.

Since the US EPA has established maximum contaminant levels for nitrate and nitrite in drinking water, in this section we will focus primarily on these two forms of N. The US EPA has also issued criteria standards for both acute and chronic toxicity of certain juvenile fish species to elevated levels of dissolved ammonia (ammonia is the gaseous form of ammonium which dissolves in water) (US EPA 1999). However, dissolved ammonia is not considered a direct human health concern in drinking water because it rarely occurs at excessively high concentrations (WHO 1996). The presence of ammonia in excess of 0.165 mg ammonia-N L^{-1} (0.2 mg NH_3 L^{-1}) does have the potential to significantly reduce the efficacy of chlorine disinfection of drinking water supplies (WHO 1996). Elevated levels of dissolved ammonia are sometimes found downstream of wastewater treatment sites, septic systems, and agricultural sites receiving fertilizer (Parker et al. 2012; Lehman et al. 2004). Organic N carried in surface water that drains wetlands and to a lesser extent agricultural soils, also contributes to N contamination of drinking water (Diaz et al. 2008; Hedin et al. 1995; van Kessel et al. 2009). However, organic N is not a regulated contaminant in drinking water and is rarely monitored.

5.2.1.3 Nitrogen in surface water: Spatial and temporal trends

Surface water originates either as runoff which drains the land surface, or as groundwater that has come back to the surface. Thus, the degree of contamination depends on the activities that occur on land adjacent to lakes and rivers, as well as the N content of the groundwater that feeds surface water bodies. The groundwater that discharges into surface water bodies contains N from both natural and anthropogenic sources, and in some cases has traveled many kilometers over thousands of years (Figure 5.2.2). Runoff containing N from agricultural fields and other non-point sources, in addition to wastewater discharged from point sources, are important contributors to impaired surface water quality. As such, different land cover types and land management practices can strongly affect how much N is transported directly to surface waters. Studies suggest that rates of N transport to surface water bodies range from less than 1 kg N ha^{-1} yr^{-1} on some natural lands to approximately 20 kg N ha^{-1} yr^{-1} on agricultural lands (Beaulac and Reckow 1982; Jordan et al. 1997a, 1997b).

[Figure 5.2.2]

In California, it is well established that drinking water drawn from the vast majority of surface water sources has relatively low concentrations of nitrate (Figure 5.2.3). A large system of reservoirs, canals, and other water conveyances has been developed to move surface water from the Sierra Nevada mountains, the Sacramento/San Joaquin Delta (the Delta), and the Colorado River to the main urban areas along the coast. For example, the water supply for San Francisco and many East Bay cities is piped directly from the Hetch Hetchy Reservoir System on the upper Tuolumne River where nitrate concentrations are negligible (BAWSCA 2012). Virtually all of the surface water bodies in the Central Valley have median nitrate concentrations well below the EPA drinking water MCL (US EPA 2006; Figure

5.2.4). The concentration of nitrate in the surface water pumped from the Sacramento-San Joaquin Delta to Southern California for public drinking water supplies is less than 1 mg nitrate-N L^{-1} (4.45 mg $NO_3^- L^{-1}$)(US EPA 2006; CEPA 2010). Nitrate concentrations in California's Lower Colorado River are also well below the MCL and appear to be improving as nutrient levels in the Upper Colorado River (e.g., Lake Powell and Lake Mead) have declined since the 1960's (Paulson and Baker 1980). While less common, several studies also show that nitrate concentrations in some of the state's smaller rivers and sloughs (e.g., Pajaro River, Mud Slough) are sometimes above the regulatory limit (Ruehl et al. 2007; US EPA 2006). There is considerable evidence that e.g. the San Joaquin River is affected by nitrate from anthropogenic sources (Pellerin et al., 2008, Kendall et al., 2008), as is the Salinas River (Moran et al., 2011). In addition to the degradation of drinking water supplies, high nitrate concentrations in surface waters are linked to eutrophication and other ecological problems (see Section 5.5).

[Figure 5.2.3]

[Figure 5.2.4]

5.2.1.4 Nitrogen in groundwater: Spatial trends

In contrast to surface water supplies, studies indicate that nitrate contamination of groundwater is becoming a widespread problem in various parts of California (Harter 2009; Harter et al. 2012; Figure 5.2.5). While this problem is well-established[1] and broadly observed (Figure 5.2.3), the occurrence of nitrate in groundwater can vary considerably in three dimensional space and is influenced by a region's hydrologic features, soil type, and land-use patterns. Nitrate enters groundwater primarily via leaching which transports excess N from the soil surface through soil pore spaces in the vadose zone until it reaches the water table. Major sources of nitrate entering groundwater are fertilizers and livestock

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[1] Throughout the assessment, "reserve wording" was used to quantify areas of uncertainty in the available data and level of scientific agreement. See Supplemental Data Tables or Appendix 5.2.2 for further details.

manures which are applied in excess of a crop requirements. For some sources of N (e.g., dairy lagoons and septic systems), there is little opportunity for plant uptake because the nitrate never interacts with the rooting zone of crops or other vegetation. The importance of agriculture as a major source of N is demonstrated by the fact that nitrate concentrations in monitoring wells located in agricultural areas are often well-above background nitrate levels (2.0 mg nitrate-N L^{-1}; 9 mg NO_3 L^{-1}) (Boyle et al. 2012).

[Figure 5.2.5]

Several studies commissioned by the California State Water Resources Control Board have examined the current spatial patterns of groundwater nitrate in various parts of the state. Much of the recent work has focused on important agricultural regions in the Tulare Lake Basin (TLB) and Salinas Valley (SV) (Figure 5.2.6; Table 5.2.1). This study area accounts for approximately 40% of the state's cropland, 50% of the state's livestock, and 7% of the human population (Boyle et al. 2012). Between 2000 and 2009, public supply wells in the TLB and SV regions had median nitrate concentrations of 5.2 and 4.7 mg nitrate-N L^{-1} respectively (equivalent to 23 and 21 mg NO_3^- L^{-1}), with approximately 10% of samples exceeding the maximum contaminate level (10 mg nitrate-N L^{-1}; 44.5 mg NO_3^- L^{-1}). In several groundwater sub-basins of Fresno and Tulare Counties datasets consisting exclusively of domestic wells had exceedance rates of 30% to 45% (Boyle et al. 2012). Wells used for domestic and irrigation purposes often have higher concentrations than public supply wells due to their shallow depth and their proximity to agricultural land uses. In contrast, the deeper confined aquifers in the western and central Tulare Lake Basin and the northern sub-basin of the Salinas Valley also tend to have relatively low nitrate concentrations.

[Figure 5.2.6]

[Table 5.2.1]

While the Boyle et al. (2012) study was confined to the TLB and SV regions, other studies indicate that wells exceeding the drinking water MCL are also found in other parts of the state. Data from the State's Groundwater Ambient Monitoring and Assessment (GAMA) program (Belitz et al. 2003; GAMA 2015), which monitors thousands of wells throughout the state indicate that the drinking water standard is often exceeded in parts of the San Joaquin, Sacramento, and Santa Ana basins (Harter 2009; Figure 5.2.5). Recent efforts by Dubrovsky et al. (2010) to predict groundwater nitrate levels by projecting observed temporal trends provide estimates that are largely consistent with the work of Harter et al. (2012), Boyle et al. (2012) and Anning et al. (2012), and suggest further that shallow groundwater resources in the Imperial Valley are also above the MCL.

5.2.1.5 Nitrogen in groundwater: Historic trends and future projections

There is agreement across several recent studies conducted in California that groundwater nitrate levels have been increasing over the past several decades (Figure 5.2.3), particularly in major agricultural regions (Burow et al. 2008; Honeycutt et al. 2012; Boyle et al. 2012). For example, Honeycutt et al. (2012) report that since the 1970s average nitrate concentrations in public supply wells in the Tulare Lake Basin and Salinas Valley have increased by approximately 0.061 and 0.120 mg nitrate-N L^{-1} (0.27 and 0.53 mg $NO_3^- L^{-1}$) per year respectively. Likewise, nitrate levels in the eastern San Joaquin Valley have more than doubled between 1950 and the 2000, with some sites in shallow groundwater near Fresno and Modesto estimated to have concentrations approaching twice the federal MCL (Burow et al. 2007; Burow et al. 2008a; Burow et al. 2008a; Burow et al. 2012; Figure 5.2.7).

[Figure 5.2.7]

Boyle et al. (2012) state that concentrations of nitrate have also increased by a similar magnitude for domestic and irrigation wells located in the TLB and SV. Using the CASTING database which includes data from thousands of public supply, monitoring, domestic and irrigation wells located throughout the study area, they estimated an average increase in nitrate concentration of 0.08 mg nitrate-N $L^{-1} yr^{-1}$ (0.34 mg $NO_3^- L^{-1} yr^{-1}$) between 1950 and 2010 across all well types. For these regions, the proportion of wells testing higher than background levels of nitrate and above the MCL also increased over the past 6 decades (Figure 5.2.8). Despite reporting significant increases in groundwater nitrate over time, Boyle et al. (2012) also note that it is difficult to establish accurate historic trends because the data used in their analysis contained a relatively small number of samples prior to 1990 comprised mostly of public supply wells, and there was a large increase in the number of samples from domestic and irrigation wells at dairies beginning in 2007. Since public supply wells tend to be deeper and have somewhat lower nitrate concentrations than the shallow wells used for domestic and agricultural purposes, this change in data sources would tend to exaggerate the increasing trend particularly after 2007 (Figure 5.2.8). De-activation and abandonment of public supply wells with water quality problems also makes assessing temporal trends in GW nitrate a challenge. Acknowledging these caveats, the overall trend was still an increase in nitrate concentrations throughout the study area prior to 2007, albeit at a more gradual slope.

[Figure 5.2.8]

Because of the time lag required for applied N to reach ground water, nitrate concentrations are likely to increase in the coming decades even if robust measures to minimize contamination are implemented (Harter 2009). While groundwater nitrate concentrations are likely to continue their upward trend, very few studies in California have been conducted to project of how fast nitrate levels may increase under various future land-use and groundwater protection scenarios (Figure 5.2.3). Building on their extensive dataset of wells in the Tulare Lake Basin, Boyle et al. (2012) have sought to address this knowledge gap by developing a process-based transport simulation model for non-point sources of nitrate across six hydrologic sub-basins (Kings, Westside, Tule, Kaweah, Tulare Lake, Kern sub-basins) within the TLB. The nitrate transport model projections exhibit significant spatial and temporal uncertainty due to inherent variability in N loading (i.e., N losses to the environment) across different land-use and source types (e.g., agricultural crops, septic systems, manure lagoons). But while the model may not forecast future groundwater nitrate levels with a high degree of accuracy it remains a useful tool for evaluating trends and the impact of alternative land-use management scenarios.

Using output from a N loading algorithm developed by Viers et al. (2012), Boyle et al. (2012) modeled four N loading scenarios that consider how changes in land use and N management from 1945 until 2050 may impact groundwater nitrate concentrations. Scenarios A and D assume that shifts in land use and improved N management will decrease N loading after 1990, while scenarios B and C assume that land use patterns and N management result in progressively higher N loading rates in the future (Figure 5.2.9). Under each of the scenarios the transport model projected increasing nitrate concentrations for all the sub-basins in all the groundwater sub-basins in the TLB region. For the Westside, Kaweah and Tule sub-basins, mean nitrate concentrations are all projected to exceed the drinking water MCL between 2005 and 2030 for both the A and C scenarios. By contrast, mean nitrate concentrations in the Kings, Tulare Lake and Kern sub-basins are not projected to reach the MCL threshold prior to 2050 (Figure 5.2.10). These results suggest that even with focused efforts to reduce N loading from non-point sources, nitrate contamination of groundwater resources is likely to become an increasingly intractable problem in certain regions. While the TLB and SV groundwater basins are likely to be among the most impaired in California, more studies are needed to evaluate and monitor the many other groundwater basins throughout the state.

[Figure 5.2.9]

[Figure 5.2.10]

5.2.2. Human exposure

5.2.2.1 Consumption of nitrate/nitrite in drinking water and food

As discussed above, nitrate and nitrite are the primary N species present in drinking water used for human consumption. In food, amino acids and proteins are the main N form, though it is also well established that nitrate and nitrite are often present in significant quantities (Figure 5.2.3). Nitrate levels can be very high in leafy green vegetables, carrots and silver beets, sometimes exceeding 677.4 mg nitrate-N kg^{-1} of vegetable (3,000 mg NO_3^- kg^{-1}) (Matallana 2010; Tamme 2010; Jaworska 2005; Santamaria 2006; Correia 2010; EFSA 2008). The amount of nitrate depends primarily on the type of crop, but is also influenced by the amount of fertilizer applied, environmental conditions, type of processing and storage time (Anjana 2007; Leszczyńska 2009; Prassad 2008; Chung 2004). Nitrates are used in processing and preserving meats, and can be found in concentrations greater than 22 mg nitrate-N kg^{-1} of food (100 mg NO_3^- kg^{-1}). Sodium nitrite is also commonly used to cure meats and meat products such as ham, bacon and sausage. In these foods nitrite is present at much higher levels than in drinking water, with some cured meats having average levels of 1.5 to 3.0 mg nitrite-N kg^{-1} food (5 to 10 mg NO_2^- kg^{-1})(EFSA 2008).

In drinking water, nitrate and nitrite are more of a problem when groundwater is the main source of drinking water rather than surface water. While consumption of foods, such as vegetables and processed meats are typically the main source of nitrate for most adults, in areas with high groundwater nitrate levels, drinking water can also be a significant means of exposure. Studies suggest that for those who consume drinking water well below the regulatory limit, only 7 to 11 % of total nitrate intake comes from drinking water (IARC 2010). However, when water sources have nitrate levels close to the regulatory limit (10 mg nitrate-N L^{-1}; 44.5 mg NO_3^- L^{-1}) as much as approximately 50 – 70 % of total nitrate intake may come from drinking water (IARC 2010; Correia 2010; Griesnebeck et al. 2010; EFSA 2010).

Nitrate in drinking water is a much more important exposure route for young infants if they are fed tap water or foods made with tap water. This is particularly important since infants under the age of six months are most susceptible to the harmful effects of nitrate. However, there is little information about nitrate exposure levels among young infants (EFSA 2010). In Romania, nitrate in drinking water, given in the form of tea, was the major source of nitrate exposure (Zeman 2002). A study by VanDerslice (2009) in rural Washington State found that less than 2% of the infants less than six months of age consumed any vegetables containing significant amounts of nitrate, leaving drinking water as the main source of dietary nitrate. In this same study approximately 10% of self-supplied households using private wells and 4.7% of households served by small water systems had nitrate levels over the federal MCL. Still for over half of the sample, total intake of nitrate was quite small at less than 0.5 mg nitrate-N kg^{-1} body weight (2.2 mg NO_3^- kg^{-1}) (VanDerslice 2009).

5.2.2.2 Exposure patterns in California

The US Geological Survey estimates that 7% of California residents rely on self-supplied water systems or small water systems serving fewer than 15 households, with the remaining 93% being supplied by public and community water systems (CWSs) (Kenny et al. 2005; see Glossary). Under the Safe Drinking Water Act, public and community water systems are required to regularly test for a wide range of contaminants. Based on data from the US EPA, 21 of the 3,049 active public and community water systems in California violated the federal MCL for nitrate during 2010 (US EPA 2011). These systems served an estimated 99,162 people, of which 92,158 were from one city water system that had a single nitrate MCL violation. Of the 21 systems that violated the nitrate MCL, 14 were located in Tulare County, with a combined service population of 5,458 residents. Overall, 0.3% of those served by public and community water systems in California are potentially exposed to nitrate levels greater than the MCL.

It is more difficult to assess the potential exposure facing self-supplied water systems that have individual wells or small water systems (see Glossary). Self-supplied and small water systems are required to test only after the well is drilled or at the time the property is sold. These results are not compiled centrally. Using data compiled from the California State Water Resources Control Board's GAMA Geotracker system, 9.8% of over 16,000 self-supplied wells tested had at least one value greater than the drinking water MCL, and 5.8% had an average level greater than this value (Figure 5.2.11; see Supplemental Data Tables and/or Appendix 5.2.2). Almost 30% of the wells had maximum nitrate levels greater than 3 mg nitrate-N L^{-1} (13.3 mg NO_3^- L^{-1}) indicating human impacts on the level of nitrate. These proportions varied across the state (Figures 5.2.12 and 5.2.13). These results should be thought of as general indicators, as the wells in this database included many types of wells, some of which were drilled specifically to characterize areas thought to have high nitrate levels.

[Figure 5.2.11]

[Figure 5.2.12]

[Figure 5.2.13]

While a statewide evaluation of the number of people exposed to elevated levels of nitrate in drinking water is still needed, a study by Honeycutt et al. (2012) has examined potential exposure in the Tulare Lake Basin and Salinas Valleys. This was done by developing a scheme for classifying the susceptibility of various private and public water systems (and the 2.65 million people they serve) based on both a qualitative definition of a water system's vulnerability (e.g., size of system, regulatory oversight, etc.), and any available data on groundwater nitrate concentrations between 2006 and 2010 (Figure 5.2.14). Using this approach they estimated that approximately 8.0 - 9.4% of the population in the study area (or 212,500-250,000 people) had 'high susceptibility' to nitrate exposure through drinking water that exceeded the federal MCL. A similar approach applied at the state level would likely be useful in evaluating the extent of exposure in other parts of the state. Service maps of California drinking water systems have now been completed for 90% of the California population who have public drinking water systems by the California Environmental Health Tracking Program (Wong et al. 2015).

[Figure 5.2.14]

5.2.2.3 *Disparities in exposure to nitrate/nitrite in California*

There are several studies that have looked at whether minority or low-income residents in California receive poorer quality drinking water than the rest of the state. However without knowing precisely which areas, and thus which people, each system serves, assessing social disparities in water quality is very difficult. The Environmental Justice Coalition for Water conducted a county-level analysis and found that counties with the highest number of drinking water violations had a higher proportion of people of Latino ethnicity than counties with the lowest number of violations (42% vs. 16%) (EJCW 2009). There were smaller disparities related to income; 17% of those living in counties with the highest number of violations were living below the poverty line as compared to 12% of those in counties with the fewest violations. Another study linked the wells used by 327 community water systems to the demographic characteristics of the people that lived around them (Balazs 2009). The proportion of residents who were Latino and the proportion who rented were significantly associated with wells that had higher levels of nitrate. They concluded that there was evidence of disparity in water quality levels based on ethnicity and poverty status.

There are also reports of predominantly low-income Latino communities in the San Joaquin Valley that are served by community water systems with elevated levels of nitrate (Firestone 2006). Of the 44 CWSs in California that violated the nitrate MCL in 2007, 74% (n=29) were located in this region (Pacific Institute 2011). Data available through the US EPA indicate that the number of systems with nitrate MCL violations dropped from 39 to 21 from 2007 to 2010, but that 76% of the systems (n=16) were still in the San Joaquin Valley (US EPA 2011).

Honeycutt et al. (2012) also examined the extent to which water systems serving disadvantaged communities in the Tulare Lake Basin and Salinas Valley exceeded the federal nitrate MCL between 2006 and 2010. In this study, “disadvantaged” and “severely disadvantaged” communities were defined as those having a median household income in 2000 below $37,994 and $28,496 respectively, which is equivalent to 80% and 60% of the statewide median household income of $47,493. Results of the study found that 51 of the 328 community water systems located in the study area exceeded the nitrate MCL. A total of 40 of the systems in violation were located in severely disadvantaged or disadvantaged communities that served approximately 379,000 people. While the studies provided above suggest that minorities and low income populations may face higher exposures to nitrate in drinking water, more detailed studies that link individuals to their specific water systems, or which actually test the levels of nitrate in their water, are needed to gain a better understanding of the disparities in water quality throughout California.

5.2.3 Human health effects of nitrate/nitrite

5.2.3.1 Adverse and beneficial effects

The consumption of nitrate and nitrite can have both adverse and beneficial effects on human health. Foods and drinking water containing high levels of nitrate and nitrite are thought to be related to three types of health problems: methemoglobinemia, adverse birth outcomes, and cancer. Studies examining these health risks are reviewed below. In addition, a small number of studies tentatively agree that increasing levels of nitrate concentration in drinking water is associated with increasing symptoms of subclinical thyroid disorders, such as hypothyroidism (Aschebrook-Kilfoy et al. 2012), although one of the studies found no such association (Ward et al. 2010). However nitrate and nitrite and other nitrogen-containing compounds are also used as therapeutic agents to lower blood pressure, and to reduce aggregation of platelets, and nitric oxide is an important signaling molecule to regulate cellular functions. (Figure 5.2.3).

5.2.3.2 Interpreting epidemiological evidence

Epidemiologic studies considered here are of two types: case-control and cohort. In case-control studies people with a disease are identified, and are compared in terms of exposure to similar people who do not have the disease. Optimally these people are randomly selected from the same population of the cases. In cohort studies, a group of initially disease-free people are observed over time, and are categorized by their level of exposure. The proportions of people that develop the disease are compared across exposure groups.

As with many environmental epidemiologic studies, there are conflicting results, with some studies showing associations while others do not. Results are influenced by the study design. Studies that are small are less apt to find a relationship when there is one. How exposure to nitrates and/or nitrites in drinking water and foods is measured can affect study results. Poor estimates of these exposures will often lead to results that show a weak or no effect, even if an association is present. In the studies reviewed, exposure was estimated in several ways. In many studies, the type of water system serving the person's house (public vs. private, groundwater vs. surface water) or the nitrate level (high nitrate water vs. low nitrate water) defined exposure, regardless of whether the person drank tap water or how much they drank. This can mask an effect of exposure. Only a handful of studies asked about, or actually observed, water consumption patterns. Exposure is particularly difficult to determine in studies of cancer, where the important exposures occurred several years before the cancer developed. Whether the exposure is due to nitrate or nitrite in food or water is also important as the risk may be different due to chemical reactions of nitrite with certain molecules in some foods (see Section 5.2.3.5).

It is quite possible that there are waterborne contaminants other than nitrate/nitrite that tend to be high when nitrate levels are high. For example, in intensely-cultivated areas both pesticide and nitrate application rates may be high. As such, effects seen with high nitrate/nitrite levels may actually be due to a different contaminant. These limitations need to be kept in mind when assessing the evidence from epidemiologic studies as we do below.

5.2.3.3 Methemoglobinemia (blue-baby syndrome)

Methemoglobinemia, or 'blue-baby syndrome', is a condition where infants become hypoxic and turn 'bluish', due to a lack of oxygen to the tissues. It is generally accepted that methemoglobinemia is caused when nitrite in the blood converts normal hemoglobin (which carries oxygen) to methemoglobin, a form which does not carry oxygen (National Research Council 1981; Wright et al. 1999). High levels of methemoglobin can lead to symptoms such as lethargy, dizziness, coma and even death. Nitrate in drinking water or food is converted to nitrite by bacteria in the stomach. The resulting nitrite, as well as nitrite consumed in foods, is absorbed into the blood stream, where it can change hemoglobin to the methemoglobin form (Figure 5.2.3).

Normal adults rarely demonstrate methemoglobinemia, as specific enzymes rapidly convert methemoglobin back to normal hemoglobin, so that less than 2% of hemoglobin is in the methemoglobin form at any time (Jaffe 1993). However, infants under six months of age have low levels of these enzymes, so with sufficient exposure, methemoglobin levels can build up to the point where the amount of oxygen delivered is substantially reduced. In addition, some of the blood of young infants is in a form (i.e. fetal hemoglobin) that is more easily affected by nitrites (National Research Council 1981; WHO 2003). Furthermore, infants have a higher gastric pH, leading to a greater conversion of nitrate to nitrite in the stomach.

The direct effect of nitrate ingested in drinking water and foods is difficult to determine due to the complex physiological processes involving nitrate and nitrite. For example, infants with diarrhea or other intestinal inflammation have had methemoglobinemia, even without any exposure to nitrates (Hegesh and Shiloah 1982; Hanukoglu and Danon 1996; Pollack and Pollack 1994; Lebby et al. 1993).

Subsequent studies have demonstrated that bacterial infections and inflammation of the bowel leads to production of nitric oxide (NO), which can also produce methemoglobin (Tannenbaum et al. 1978; Gupta et al. 1998; Witthoft et al. 1999; Levine et al. 1998; Wagner et al. 1984). Furthermore, when nitrite or nitric oxide reacts with hemoglobin to form methemoglobin, nitrate is produced (Lundberg et al. 2004). Some of the nitrate in the blood is gathered by the salivary glands and excreted into the mouth, where normal bacteria convert a part of it to nitrite. This nitrite and the remaining nitrate are then swallowed, creating a complex cycle.

There are relatively few studies of the relationship between nitrate exposure and the risk of methemoglobinemia (Fewtrell 2004). Working in Israel, Shuval et al. (1972) found no differences in the mean methemoglobin levels between infants with high nitrate tap water (11.3 – 20.3 mg nitrate-N L^{-1}; 50 – 90 mg NO_3 L^{-1}) as compared to those with low levels (< 1.13 mg nitrate-N L^{-1}; < 5 mg NO_3 L^{-1}). However, tap water accounted for only a small proportion of the infants' diet. A similar study in Illinois found only one statistically significant difference in methemoglobin levels (0.75% vs. 1.22%) comparing a dose of 1-19 mg NO_3 in the 2 hours prior to sampling to 20 – 50 mg NO_3. Neither level was above what is considered to be a normal level of methemoglobin (3%). (Craun et al. 1981). Knobeloch et al. (2000) report on two cases of clinical methemoglobinemia among infants both of whom had consumed formula prepared using well water. The concentrations of nitrate-nitrogen were 22.9 and 27.4 mg l^{-1}. One of the wells was negative for coliforms and the parents of the other child boiled the water before feeding. Diarrheal illness was not reported. In Washington State, a water quality study involving over 800 infants examined the effects of nitrate, total coliforms and *E. coli* on methemoglobinemia (VanDerslice 2009). The median nitrate-nitrogen concentration was 1.5 mg L^{-1} and 8.6% of the observations were above 10 mg L^{-1}. Approximately 25% of infants in the study also consumed water that tested positive for total coliforms. Results of the study reported small but statistically significant relationships between the amount of nitrate and total coliform ingested and elevated methemoglobin levels in infants (VanDerslice

2009). However, none of the infants in the study exhibited clear physical symptoms of methemoglobinemia. A case-control study in Romania found a significant relationship between nitrate intake and methemoglobinemia episodes, and a weaker association between diarrhea and methemoglobinemia episodes (Zeman et al. 2002). Nitrate exposure occurred through both diet and drinking water. Average nitrate consumption level was high among the cases whose water was > 10 ppm NO_3-N (> 22.6 mg nitrate-N kg^{-1} body weight day^{-1}; > 100 mg NO_3 kg^{-1} body weight day^{-1}). In a study conducted in Morocco Sadeq et al. measured methemoglobin levels in children up to eight years of age using wells and piped water (Sadeq et al. 2008). Information was solicited about consumption of common foods containing nitrate and nitrite, but diarrheal illness was not recorded. Children consuming well water with greater than 10 mg L^{-1} nitrate-N (> 44.5 mg NO_3-N L^{-1}) were 1.6 times more likely to have elevated methemoglobin levels (> 2% methemoglobin) than children drinking water from the piped supply (average = 2.99 mg NO_3-N L^{-1}). The proportion of children with elevated methemoglobin increased with age, peaking at six years of age. There was no evidence that food consumption was related to the increased prevalence of elevated methemoglobin.

The current MCL for nitrate is based primarily on two studies that found methemoglobin cases to occur only when the infant's water contained more than 10 mg nitrate-N L^{-1} (> 44.5 mg NO_3 L^{-1}) (Bosch et al. 1950; Walton 1951; US EPA 1990). With improvements in the understanding of nitrate reactions in the body, some researchers questioned the role of nitrate-contaminated water as a cause of methemoglobinemia (Avery 1999; L'Hirondehl et al. 2003). Specifically, these authors contend that high methemoglobin levels may be primarily due to nitric oxide created as a result of inflammation in the bowel, resulting from gastrointestinal infection. High nitrate wells are also more likely to be contaminated with enteric pathogens, so the apparent relationship between well water nitrate levels and methemoglobinemia may be due to microbiological contamination. As such, some scientists question whether the MCL should be set at a higher level (Avery 1999). It should be noted that the studies of VanDerslice (2009) and Zeeman (2002) observed effects of nitrate intake while controlling for diarrheal disease, and there have been carefully documented cases of methemoglobinemia where drinking water was above the nitrate MCL and there was no exposure to bacteriological pathogens in the drinking water nor evidence of diarrheal disease (Knobeloch et al. 2000).

This ambiguity has made it difficult for local and state public health officials to effectively respond to the problem of nitrate contamination of non-federally regulated water systems and private wells. Operators of some small systems express that there is no evidence of methemoglobinemia in their region despite a history of nitrate levels over 10 mg L^{-1}. The widespread nature of the problem, the high cost of treatment to remove nitrate or to develop alternative water sources, the lack of observed cases of methemoglobinemia and the knowledge that the condition can be fatal combine to make this a complicated issue for local and state environmental health practitioners.

5.2.3.4 Birth outcomes and birth defects

It has been suggested but remains unproven that exposure to nitrate may affect birth outcomes (Figure 5.2.3). The National Academy of Science (1981) suggested that a reduction in blood oxygen levels from the creation of methemoglobin might impact the development of the fetus, and there is some evidence of this in animal studies (Fan et al. 1987). In two studies, Tabacova et al. (1997, 1998) found that maternal methemoglobin levels were associated with the risk of pregnancy complications (pre-term birth, low birth weight, fetal distress, premature labor), and that the methemoglobin level in cord blood was strongly associated with the methemoglobin level in the mother’s blood. While these studies did not link methemoglobin levels to drinking water nitrate levels, Burkowski et al. (2001) found that women who lived in areas with median drinking water nitrate levels over 3 mg nitrate-N L^{-1} (< 13.3 mg NO_3 L^{-1}) were twice as likely to have a low birth weight baby as women exposed to very low nitrate levels (< 1.3 mg nitrate-N L^{-1}; < 5.8 mg NO_3 L^{-1}).

There have also been concerns that exposure to nitrates or nitrates during pregnancy can increase the risk of spontaneous abortion. One study found that women who had spontaneous abortions did not have higher or lower methemoglobin levels during pregnancy (Skrivan 1971). A comparison of women who consumed water with no nitrates to women who consumed low levels (0.1-5.5 mg L^{-1}) found no effect (Aschengrau et al. 1989), while a comparison of communities with insignificant levels of nitrate to communities with high levels (consistently over 40 mg L^{-1}) had no differences in fetal death rates (Gelperin et al. 1975). In 1996, the Center for Disease Control and Prevention (CDC) investigated a cluster of six spontaneous abortions in the same community in Indiana (CDC 1996). All women were consuming water from wells with nitrate-nitrogen levels ranging from 19.0 to 28.7 mg L^{-1}. A large hospital-based case-control study in Massachusetts found no association between nitrate levels in drinking water and stillbirth or congenital anomalies, but a weak association with neonatal deaths (Aschengrau et al. 1993).

There is some evidence that nitrate exposure is related to birth defects of the central nervous system (CNS). Women whose source of drinking water was groundwater with more than 15 mg nitrate-N L^{-1} (66.7 mg NO_3 L^{-1}) were more than three times as likely to deliver a baby with a defect of the CNS, as compared to women with low drinking water nitrate levels (Dorsch et al. 1984). Researchers in New Brunkswick conducted water sampling to ascertain exposure levels for a case-control study of CNS congenital malformations (Arbuckle et al. 1988). They observed an increased risk when nitrate levels were above 26 mg NO_3-N L^{-1} but only for those on well water. In Texas, Brender et al. (2004) found that women who drank water with more than 3.5 mg nitrate-N L^{-1} (15.6 mg NO_3 L^{-1}) and took medications that could produce nitrosamines were fourteen times more likely to have a baby with a neural tube defect (NTD). A study in California did not find a significant association between living in an area with nitrate above 10 mg nitrate-N L^{-1} and NTDs, but did find that there was four times greater risk of anencephaly, a specific type of NTD (Croen et al. 2001). The largest study, including more than 70,000 infants, found a modest increased risk for those mothers whose drinking water source was groundwater and thus more likely to be exposed to nitrate (Cedegren et al. 2002). A recent study of 60 congenital anomalies and 1635 controls assessed nitrate levels in both municipal water systems and estimated nitrate levels in private wells using geostatistical interpolation (Holtby et al. 2014). There was a statistically significant increase in the risk of any anomaly associated with average nitrate between 1 and 5.66 mg NO_3 L^{-1} (OR=2.44, CI = 1.05 – 5.66), and a similar effect, though not significant, for concentrations above 5.56 mg NO_3-N L^{-1}.

5.2.3.5 Cancer

Cancer refers to a large number of related diseases characterized by uncontrolled growth of cells. In some cancers this leads to the formation of a mass of cells that affect normal cells and organs. Nitrate and nitrite are not thought to cause cancer directly, but are precursors of N-nitroso compounds (NOCs) that are known animal carcinogens (IARC 2010). NOCs are a family of compounds formed through a reaction of nitrite with specific molecules that are found in amino acids (and other compounds). Amino acids are the building blocks of proteins. NOCs are formed in meat products that have been cured using nitrite, as well as from reactions that take place in the body. As such, the level of NOCs in the body depends on the amount of nitrate and nitrite consumed (as some nitrate is converted to nitrite in the mouth) and the amount of NOCs already present in foods (IARC 2010). Some drugs, including aspirin and antihistamines, can also react with nitrite to form NOCs (Brambilla 1985).

The relationship between the ingestion of nitrate, nitrite, foods containing NOCs and the level of NOCs in the body is very complex as many biochemical processes are involved. While increases in nitrate or nitrite exposure may increase the level of NOCs, this also depends on the availability of foods or drugs that contain amines. Vitamin C and other antioxidants can reduce the formation of NOCs (IARC 2010). A few studies have monitored the levels of specific NOCs in urine for individuals given specific diets, or with high and low levels of nitrate in drinking water. Subjects given higher nitrate intakes, and the chemicals which can form NOCs, excreted higher levels of the NOCs; taking vitamin C significantly reduced these levels (Vemeer et al. 1998; Vemeer et al. 1999; Mirvish et al. 1998).

Some NOCs are very reactive, and have been shown to damage and cause mutations in DNA. Studies using rodents have shown that the administration of NOCs induces tumors in the bladder, liver, nose, mouth, esophagus, kidney, pancreas, lymph, stomach, and in the nervous system (IARC 2010). Many of the studies observed an increased risk only when the animals was given both nitrate or nitrite and a source of amines (Pliss and Frolov 1991; Greenblatt et al., 1971; Greenblatt and Mirvish, 1973; Borzsonyi et al. 1976; Shank and Newberne 1976; Lijinsky 1984 and others, see IARC 2010 and Bryan et al. 2014)) Vitamin C and other antioxidants have been shown to reduce the formation of tumors for a given exposure to NOCs (IARC 2010). The evidence linking exposure to nitrite and nitrate with cancer in humans is not as clear. Studies relating nitrate, nitrite and NOCs to cancer in humans are presented below by cancer site. In addition, one study has found that ingestion of nitrate from dietary and drinking water sources was associated with an increased risk of thyroid cancer (Ward et al. 2010), but due to the small sample size and lack of other studies, no further discussion of thyroid cancer is included below.

Bladder and kidney cancer

There have been four studies that found the risk of bladder cancer not to be related to the total amount of nitrate from foods and water in their diet (Knekt et al. 1999; van Loon et al. 1998; Zeegers et al. 2006; Ward et al. 2003). Two other large studies, however, did find an effect. A study looking back at a cohort of over 20,000 women followed over time starting in 1986 found that the ones whose drinking water contained more than 2.5 mg nitrate-N L^{-1} (11.1 mg NO_3 L^{-1}) were nearly three times as likely to have been diagnosed with bladder cancer (Weyer et al. 2001). However, there was no relationship with total nitrate (food + water) intake. Nitrite, and nitrite plus nitrate in meats were associated with bladder cancer (29% increase) in a study including over 300,000 people in the US (Ferrucci et al. 2010). A smaller study, conducted in Hawaii, found that the chances of developing bladder cancer increased three-fold, but only among men of Japanese descent who consumed the highest levels of nitrite (Wilkens et al. 1996). A case-control study in Iowa also found an association between number of years of consuming water above 5 mg NO_3-N L^{-1} and renal cell carcinoma, but only among those with higher red meat consumption, or those with lower than average vitamin C consumption (Ward et al. 2007). Two recent large cohort studies that focused on nitrate, nitrite and NOCs in both found significant associations with renal cell carcinoma (Daniel et al. 2012; DellaValle 2013).

Cancer of the stomach and esophagus

Studies looking at cancers of the stomach and esophagus have examined the effects of nitrate separate from nitrite. Of 13 case-control studies that examined nitrate, only one (Boeing at al. 1991) of the three studies that assessed nitrate in drinking water found an association (Yang et al. 1998; Rademacher et al. 1992). The remaining 11 studies estimated total nitrate intake from foods and drinks; only two found a significant association (Ward et al. 2008; Rodgers et al. 1995; Jakszyn and González 2006; IARC 2010).

More than 20 studies have looked at the relationship between stomach and esophageal cancers and the amount of nitrite, or nitrite plus nitrate, in the diet (IARC 2010). Almost all of the studies that looked at smoked foods, preserved fish or preserved vegetables, and most of the studies of meats or processed meats, found significant associations with stomach cancer (IARC 2010). Studies which estimated total nitrite or NOC intake had variable results. However, most of the studies that considered both nitrite and antioxidant intake did find associations with stomach cancer.. When subjects consumed high levels of nitrite and low levels of antioxidants there was two to five times the risk of stomach cancer, as compared to subjects with low intake of nitrite and high intakes of antioxidants (Bruning-Fan and Kaneene 1993; Jakszyn and González 2006; IARC 2010). Ward et al. (2008) also observed both a non-significant association of nitrate from meats, and a significant association of nitrate from plant-derived foods, and stomach cancer. Two recent large cohort studies, however, did not observe associations with nitrite or nitrate and stomach cancer (Cross et al. 2011; Loh et al. 2011). In a recent review Bryan et al. observe the lack of evidence in recent cohort studies, but also conclude that the associations that have been seen have been primarily for nitrite exposure among people with low Vitamin C intake (Bryan et al. 2012).

Fewer studies have looked at cancer of the esophagus (Barretta et al. 2012). Navarro Silvera et al. (2011) observed an association with red meat consumption, while Cross et al. (2012) did not. A meta-analysis of studies examining the role of a ‘Western diet”, including higher red meat consumption, found no association, leaving us to conclude that an association between dietary levels of nitrate and cancers of the esophagus are suggested but unproven to date (Lui et al. 2014).

Breast and genital cancers

The only study that collected data over a period of time observed an increased risk of ovarian cancer with increasing drinking water nitrate levels, however there was association with breast cancer and an inverse relationship with uterine cancer (Weyer et al. 2001). Another study compared diets between over 300 women with breast cancer to a similar number without; they found no association with nitrite or nitrate intake, and a two-fold increase in risk for women with a higher intake of nitrate relative to their intake of folate (Yang et al. 2010). However in a follow-up study, nitrate intake from food or water was not associated with breast cancer overall, and was only associated among women with high folate intake, which was an unexpected finding (Inoue-Choi et al. 2012). Drinking water levels were not associated with breast cancer in the US (Brody et al. 2006). One study found no association of nitrate in diet and endometrial cancer (Barbone et al. 1993) while another found some association of drinking water nitrate levels during childhood and testicular cancer, but only for men currently living in urban areas, which casts doubt that the association is truly with nitrates (Møller 1997).

Brain cancers

There have been over 20 studies of the association of nitrate and/or nitrite in foods and drinking water and the risk of various brain cancers; almost all have been ecologic or case-control studies (IARC 2010). Of eleven case-control studies of brain cancer in adults, only one found any link of nitrate or nitrite exposure and the development of cancer; Ward et al. (2005) found an elevated risk for those with the highest intake of nitrate from food of plant origin. Nitrate intake from meats or drinking water were not associated with brain cancer. A large prospective study examined the relationship between dietary intake of NOCs and adult glioma and found no association, nor any protective effect of vitamin C (Dubrow et al. 2010).

There is more evidence of an association with childhood brain tumors. Intake of nitrite in meats, and the combination of high nitrite intake and low prenatal vitamin intake during pregnancy were significantly related to brain tumors in their offspring (Preston-Martin 1996). The single study to look at nitrite levels in the drinking water (based on actual water samples) of women from their residence during pregnancy did find an association with the risk of brain cancer of their subsequent child (Mueller et al. 2004). A recent case-control study of childhood deaths due to brain tumors, using water quality information from piped water system found that having over 0.3 mg NO_3-N L^{-1} drinking water was associated with a significant increase in risk (Weng et al. 2011).

Rectal and colon cancer

A study in Iowa found that individuals exposed to drinking water nitrate above the MCL, and with low vitamin C diets, were twice as likely to develop colon cancer as those consuming water with nitrate levels below the MCL (DeRoos et al. 2003). They also observed a positive relationship between total nitrite intake and cancers of the colon and rectum (DeRoos et al. 2003). An earlier study in Iowa had found no association of nitrate levels in drinking water and colon cancer, and an inverse association with rectal cancer (Weyer 2001).A study in Wisconsin that included sampling of individual wells found that people who had been exposed to drinking water above the MCL were 2.9 times more likely to get colon cancer (McElroy et al. 2008). Deaths due to colon cancer were also studied in Taiwan; where the investigators reported that the effect of nitrate in drinking water varied depending on the level of magnesium (Chiu et al. 2010). In a large prospective study involving over 70,000 women, researchers found a large significant association of nitrate intake and colorectal, but only among women with lower vitamin C intake (DellaValle et al. 2014).

Leukemia and lymphoma

Leukemia refers to cancers that occur in the blood or bone marrow, while lymphomas are cancers of the lymph system. Of the four studies that examined the risks of nitrate or nitrite exposure, two studies (Ward et al. 1996, 2006) observed an association. In the first study, only nitrate in drinking water was associated with non-Hodgkin lymphoma, while dietary nitrate showed no significant association. The 2006 study, on the other hand, found that high dietary nitrite intake significantly associated with greater risk for non-Hodgkin lymphoma, while dietary nitrate intake was inversely associated with higher risk, and drinking water nitrate levels were not associated with risk of non-Hodgkin lymphoma. Other comparisons in these studies, and in two similar studies, did not demonstrate any association.

Pancreatic cancer

Of the four studies to assess nitrate exposure and cancer of the pancreas, none found any evidence of higher risks for higher levels of nitrate in water or higher dietary intake of nitrate (Howe et al. 1990;

Baghurst et al. 1991; Weyer et al. 2001; Coss et al. 2004). One study did see an increase in risk from increases in nitrite intake from meats (Coss et al. 2004).

Summary

There are mixed results on studies of nitrate and nitrite consumption and the risk of cancer. The strongest evidence relates to exposures from nitrites in foods, and a few studies have observed much greater risk among people with low vitamin C intake. The International Agency for Research on Cancer has concluded that "Ingested nitrate or nitrite under conditions that result in endogenous nitrosation [formation of nitrosamines in the body] *is "Probably carcinogenic to humans*" and has classified nitrate and nitrite in Group 2A, 'Probably carcinogenic to Humans' (IARC 2010; Figure 5.2.3). Neither the US EPA nor Health Canada have classified nitrate or nitrite in terms of carcinogenicity (US EPA 2011; Health Canada 2011).

5.2.3.6 Health benefits of ingested nitrate/nitrite

Nitrate and nitrite and other nitrogen-containing compounds are used as therapeutic agents (Butler, 2008). In clinical studies, nitrite (resulting from conversion of nitrate in the body) has been shown to lower blood pressure, and to reduce aggregation of platelets (Webb et al. 2008; McKnight et al. 1999; Lundburg et al. 2006; Gilchrist et al. 2011; Lundberg et al. 2011). Platelet aggregation can lead to the formation of clots in the circulatory system, a risk factor for stroke. Nitrate has been shown to increase oxygen delivery to oxygen-starved tissues, and help protect against injury to the heart resulting from a heart attack (Tang et al. 2011). There is evidence that nitrate and nitrite help the body's host defenses against bacterial pathogens (Lundberg et al. 2004). Several scientists have begun to question whether the beneficial aspects of nitrate and nitrite in foods outweigh the health risks and costs of addressing nitrate contamination in water supplies (Hord et al. 2009; Powlson et al. 2008; Katan 2009; Gilchrist et al. 2011; Kevil and Lefer 2011). The economic aspects are discussed in Section 5.2.4.

5.2.3.7 Research needs

Studies do show evidence that nitrate or nitrite in drinking water and/or foods is an important factor in the development of "blue-baby syndrome". Nitrate, nitrite and/or nitrosamines in foods are "Probably carcinogenic to humans" (IARC 2010), with consistent evidence that total exposure, and exposure via meats, is related to stomach cancer. Exposure to nitrate has been consistently associated with neural tube defects. However, given the complexities of these exposure disease relationships and the state of knowledge regarding these and other potential risks, it is very difficult to determine whether current regulatory limits are adequate to protect public health or possibly more stringent than is necessary (Ward et al. 2003; Ward et al. 2008; Powlson et al. 2008). There are several areas where more research is needed to examine the effects of nitrate and nitrite on potential health problems. Key areas of investigation should include birth defects, stillbirth and spontaneous abortion resulting from maternal exposure during pregnancy, and hypothesized impacts on thyroid disease and many types of cancer (Ward et al. 2010). Even for 'blue-baby syndrome', which has been linked to nitrate in drinking water for over 50 years, controversy remains about how much of a risk nitrate in drinking water poses, and whether factors other than nitrate are the actual cause of this potentially fatal disease.

The levels of exposure to nitrate are not well known, particularly differences in exposure related to income, race and/or ethnicity. Such research is limited by a lack of data describing which specific areas are served by which public water system (VanDerslice 2011). In addition, water quality data for private wells are not centrally collected, making it nearly impossible to create a comprehensive picture of exposure among those using private wells.

It is often very difficult to determine whether a disease is caused by an environmental exposure, and almost impossible to precisely know the relationship between the level of exposure and the risk of that disease for individuals of different ages, racial backgrounds, and states of health. Decisions about allowable levels of nitrate and nitrite, unfortunately, have to be based on the best information possible.

5.2.4 Economic costs of N in drinking water

5.2.4.1 Costs associated with human well-being

Studies evaluating the economic costs of nitrate and nitrite in drinking water are very limited in the peer-reviewed literature and only a few have focused on California in particular. The data that are available fall into four main economic categories; 1) the health costs associated with the human consumption of nitrate contaminated drinking water, 2) the household costs associated with strategies to reduce nitrate concentrations through “point of entry” and “point of use” treatment systems, 3) the household costs associated with avoiding consumption of contaminated water sources through purchasing clean water (e.g. bottled or trucked water) or drilling new wells, and 4) the costs associated with strategies to reduce nitrate concentrations through treatment, blending, and consolidation for public and community water systems of various sizes. Here we examine the economic costs associated with N in drinking water with a particular focus on understanding how each of these cost categories may influence human well-being.

5.2.4.2 Health costs

Accurate estimates of health costs are extremely scarce in the literature and are limited by the uncertainty present in the epidemiological studies that link drinking water nitrate/nitrite to the health outcomes discussed above (van Grinsven et al. 2010). Focusing on 11 countries in the European Union, van Grinsven et al. (2010) estimate that the health damages resulting from colon cancer related to nitrate in drinking water ranged between €0.1 – 2.4 per kilogram of N (i.e. \$0.14–3.38 kg^{-1} N) that is leached from agricultural land into groundwater supplies. Several other studies have speculated that health costs from the consumption of nitrate in drinking water are likely to be considerable (Hanley 1990; Innes and Cory 2001; Compton et al. 2010) but with the exception of van Grinsven et al. (2010), no other studies have attempted to estimate the actual costs associated with specific health outcomes (Figure 5.2.3). Despite the challenge of working with complex epidemiological phenomena, more efforts to estimate the possible health damages associated with methemoglobinemia, birth defects and other forms of cancer are needed.

5.2.4.3 Household costs

Cost studies dealing with the treatment and avoidance of nitrate/nitrite contaminated drinking water for single households with domestic wells are becoming increasingly common in the recent literature, and several have recently been conducted in California. Lewandowski et al. (2008) conducted a survey of households in Minnesota that estimated the cost of three treatment systems (reverse osmosis, distillation, anion exchange) and two avoidance strategies (bottled water, drilling new wells). In California, the Pacific Institute carried out a similar survey of households in the San Joaquin Valley that estimated the costs for reverse osmosis systems and bottled water (Pacific Institute 2011). Focusing on California's Tulare Lake Basin and Salinas Valley, Jensen et al. (2012) and Honeycutt et al. (2012) present the most comprehensive analysis of the household costs for various nitrate treatment and avoidance options to date. Across all four studies, single-household reverse osmosis systems required larger upfront costs than bottled water, but were generally the lowest cost alternative when initial capital and service costs were amortized over the lifetime of the system (Table 5.2.2). Since low income households are less able to afford the upfront costs of water treatment systems, these studies suggest that they may end up paying a large fraction of their household income to purchase relatively expensive bottled water and filtration systems or else continue consuming untreated water. For example, in one San Joaquin community (Beverly Grand) households spent 4.4% of their median income on vended and bottled water, filters, and tap water service; almost 3 times the 1.5% affordability threshold established by the US EPA (Pacific Institute 2011). These results, which are provisionally agreed upon by most, demonstrate how the cost of different household treatment options may have significant implications for both the health and economic wellbeing of low-income households (Figure 5.2.3).

[Table 5.2.2]

5.2.4.4 Costs to public and community water systems

Many public and community water systems have found that funds raised through local bond measures and/or fees levied on water users are often insufficient to pay for the water quality improvements required to meet federal drinking water standards (Pacific Institute 2011). In California, public and community water systems can also apply for public funds to address water quality through both the California Department of Public Health (CDPH) and US Department of Agriculture (USDA). In the San Joaquin Valley alone, approximately 100 projects to mitigate nitrate contamination were proposed by community water systems from 2005-2009, requesting $62 million for projects to address solely nitrate and a total of $150 million for projects that included nitrate among other water quality concerns (Pacific Institute 2011). Proposed projects included strategies such as drilling new wells, treating contaminated water, blending contaminated water with cleaner water from new sources, and consolidating available clean water sources (Table 5.2.3). The study also reported that the CDPH and USDA funded 16 of these proposed projects at an actual cost to the public of approximately $21 million over the 5 year study period (Table 5.2.4). While the authors also note that a detailed discussion of other public funding sources such as federal block grants for community development was beyond the scope of their analysis, the study does show that the need for projects addressing nitrate contamination in the San

Joaquin Valley (and likely elsewhere in the state) far surpasses the public resources which are currently available to do so.

[Table 5.2.3]

[Table 5.2.4]

Honeycutt et al. (2012) suggest that an additional $17 – 34 million per year over many decades may be required to ensure safe drinking water for the 85 public and community water systems in the Tulare Lake Basin and Salinas Valley that already exceed the federal MCL for nitrate. While their study examines two of the most vulnerable regions of the state, similar studies are still needed to estimate the costs required to address nitrate contamination in other regions (Figure 5.2.3). Also, since most of the water systems in violation were located in disadvantaged communities, challenging questions are being raised by water system managers and policy advocates regarding how to cover the costs of new projects aimed at reducing nitrate contamination and other water quality concerns. Given state and federal budget constraints, it is increasing likely that essential water quality projects serving vulnerable communities will be postponed, scaled back or abandoned (Pacific Institute 2011). As such, a growing number of policy studies have argued that allocation of public funds to address nitrate and other contaminants in private and community water systems should include a more robust assessment of the environmental and social justice concerns raised by vulnerable communities who are likely to pay a disproportionate share of the human health and remediation costs (Pacific Institute 2011; VanDerslice 2011; Firestone et al. 2006; Firestone et al. 2009).

Appendix 5.2.1: Glossary for Section 5.2

Maximum contaminant level (MCL) – Enforceable limits for nitrate and nitrite established to protect the public against consumption of drinking water that has concentrations of these contaminants high enough to present a risk to human health (e.g. 10 mg nitrate-N L^{-1} and 1 mg nitrite-N L^{-1}). (U.S. EPA)

Self-supplied water system - A water system that is not connected to a public water system, is assumed to be 1 to 2 households/dwelling units (or connections). These systems are not regulated under the Safe Drinking Water Act. (CDPH)

Local small water system - A water system that serves 2 to 4 households. These often draw on a single domestic well and depending on the county are sometimes s. These systems are not regulated under the Safe Drinking Water Act. (CDPH)

State small water system - A system for the provision of piped water to the public for human consumption that serves at least five, but no more than 14, service connections and does not regularly serve drinking water to more than an average of 25 individuals daily for more than 60 days out of the year. These systems are not regulated under the Safe Drinking Water Act. (CDPH)

Public water system - A system for the provision of water for human consumption through pipes or other constructed conveyances that has 15 or more service connections or regularly serves at least 25 individuals daily at least 60 days out of the year. These systems are regulated under the Safe Drinking Water Act. (CDPH)

Community public water system or community water system - A public water system that serves at least 15 service connections used by yearlong residents or that regularly serves at least 25 yearlong residents. These systems are regulated under the Safe Drinking Water Act. (CDPH)

Domestic well – Refers to the unregulated wells used for drinking water that serve self-supplied water systems and local small water systems.

Irrigation well - Refers to unregulated wells used for irrigation and other agricultural purposes, but not for drinking water.

Public supply well – Refers to the regulated wells used to supply drinking water to public and community public water systems.

Disadvantaged community - A census block group that has a Median Household Income (MHI) of less than 80% of the State of California's Median Household Income in 2000 (MHI = $47,493). (Honeycutt et al. 2012)

Severely disadvantaged community - A census block group that has a Median Household Income (MHI) of less than 60% of the State of California's Median Household Income in 2000 (MHI = $47,493). (Honeycutt et al. 2012)

Appendix 5.2.2: Reserve wording and data table for section 5.2[2]

Following the models of other assessments, "reserve wording" was used throughout the assessment to quantify areas of uncertainty (Box A5.2.1; modified from Ash et al. 2010), providing a more consistent analysis across chapters. This approach takes into account both the level of scientific agreement and amount of available evidence.

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[2] Full details are available in the Supplemental Data Tables.

[Box A5.2.1]

The following data table supplements and supports the statements and conclusions in the body of the assessment report. Data tables are not summaries of findings, but rather summaries of what is known and available to evaluate the causes, states, and consequences of N cycling in California. To this end, the data tables have two primary purposes: 1) summarize the sources of data and approaches used in the assessment and 2) systematically evaluate the quality of available data. Each table includes the indicators used in the assessment to measure flows of N, and for each indicator relevant sections of the assessment report are noted. The data tables are organized by broad categories of N flows. This framework was selected to maximize homogeneity of concepts and issues within each data table and explicitly show the linkages across chapters.

[Table A5.2.1]

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Figure 5.2.1. Connectivity and utilization of surface water and groundwater resources. Source: Ponce (2007) adapted from California Department of Water Resources. [Return to text]

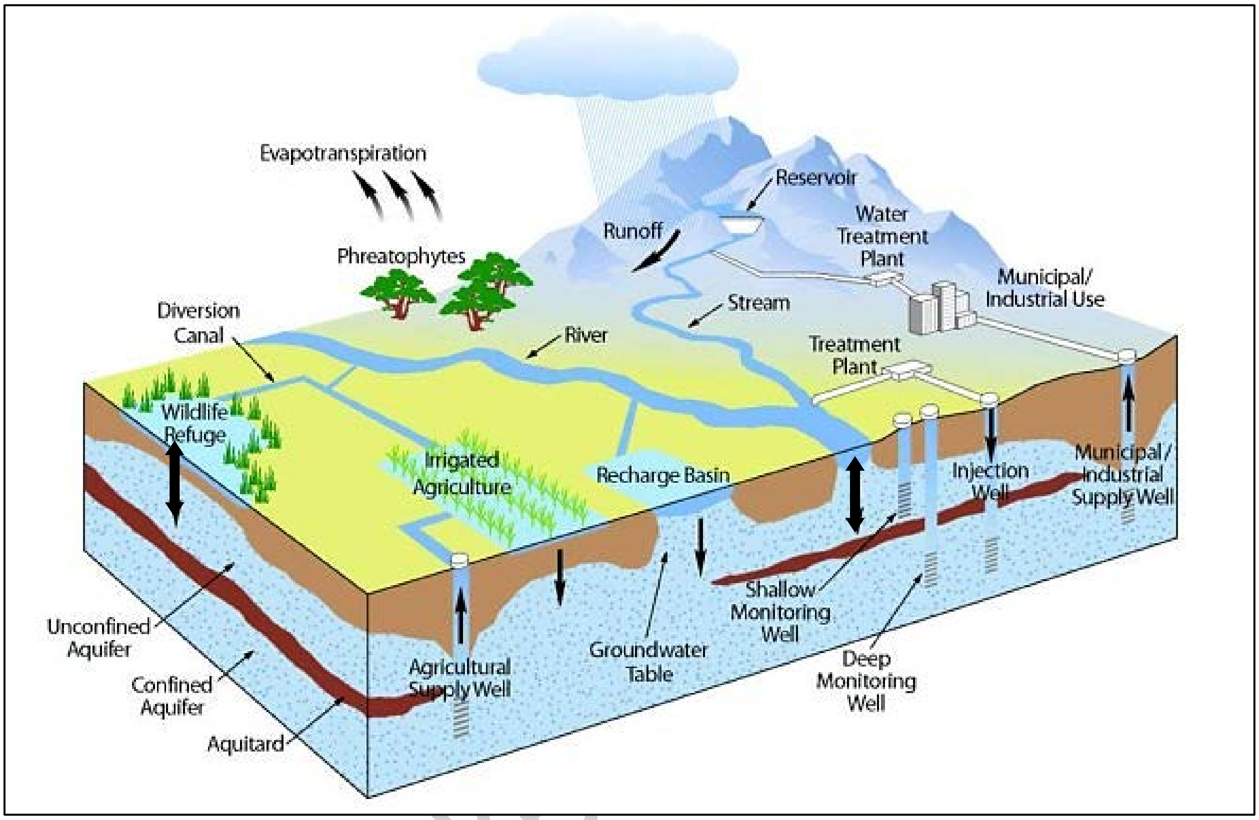
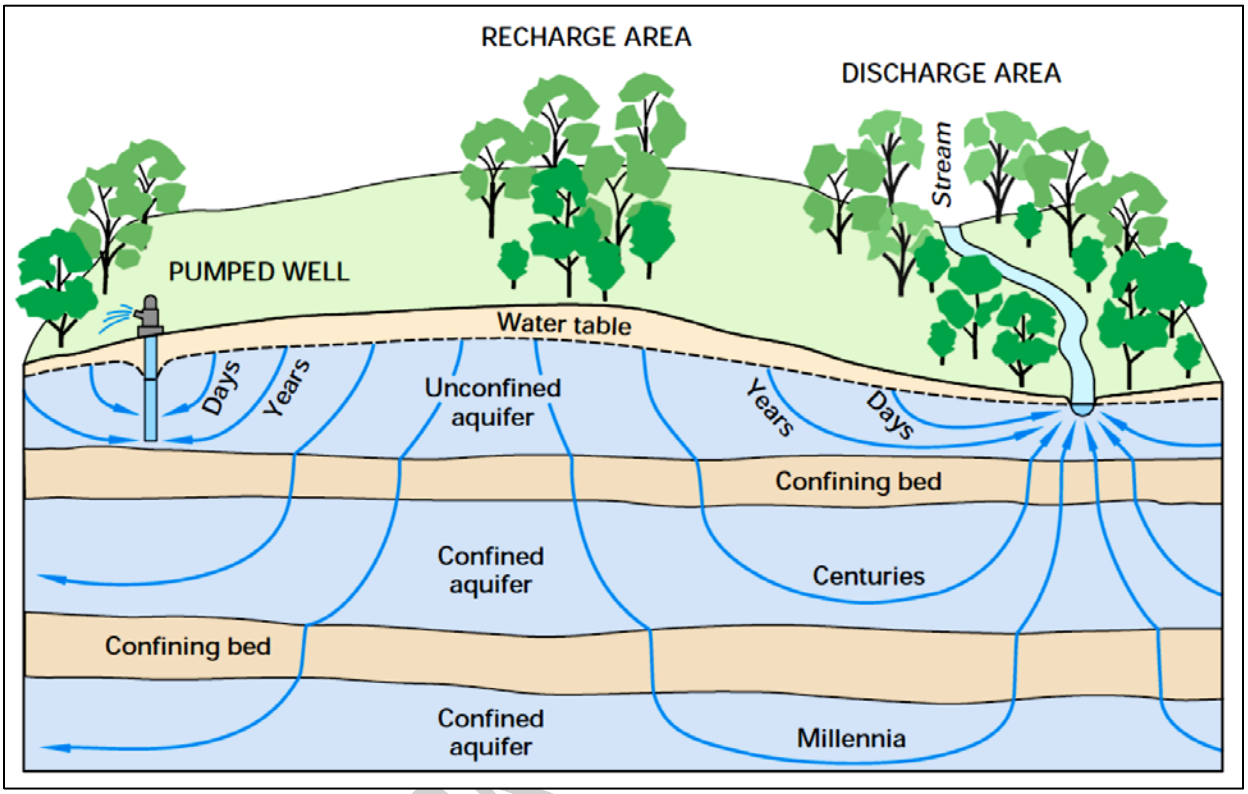
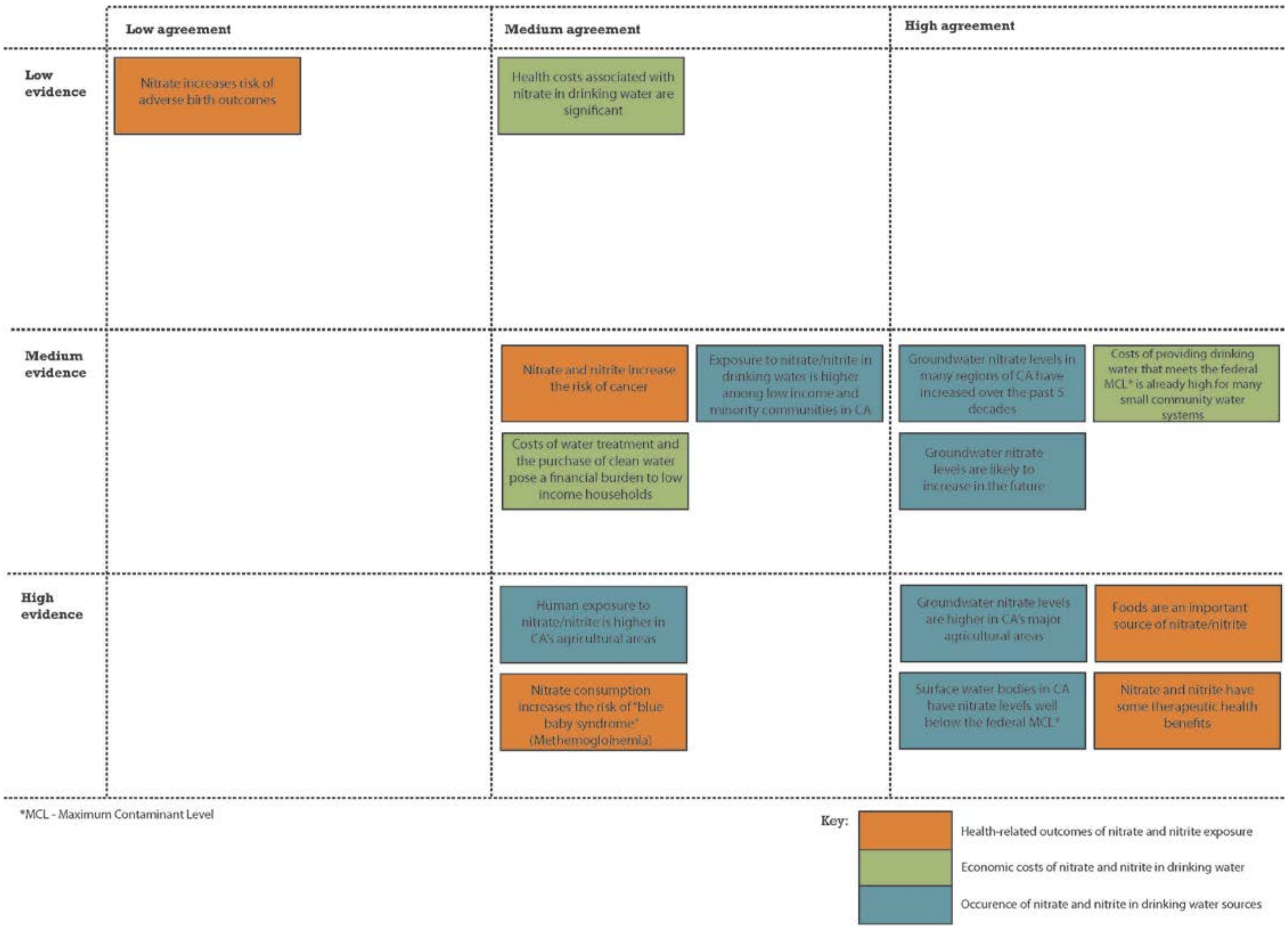


Figure 5.2.2. Spatial and temporal scales of groundwater movement. Ground-water flow paths vary greatly in length, depth, and travel-time from points of recharge to points of discharge in the groundwater system. Source: Winter et al. (1998). [Return to text]



1405 **Figure 5.2.3. Measuring uncertainty in nitrogen's impact on human well-being**. Amount of evidence and level of agreement on various aspects

1406 of nitrogen impacts on human well-being through contamination of drinking water resources. [Return to text]



	Low agreement	**Medium agreement**	**High agreement**
Low evidence	🟧 Nitrate increases risk of adverse birth outcomes	🟩 Health costs associated with nitrate in drinking water are significant	
Medium evidence		🟧 Nitrate and nitrite increase the risk of cancer 🟦 Exposure to nitrate/nitrite in drinking water is higher among low income and minority communities in CA 🟩 Costs of water treatment and the purchase of clean water pose a financial burden to low income households	🟦 Groundwater nitrate levels in many regions of CA have increased over the past 5 decades 🟩 Costs of providing drinking water that meets the federal MCL* is already high for many small community water systems 🟦 Groundwater nitrate levels are likely to increase in the future
High evidence		🟦 Human exposure to nitrate/nitrite is higher in CA's agricultural areas 🟧 Nitrate consumption increases the risk of "blue baby syndrome" (Methemogloinemia)	🟦 Groundwater nitrate levels are higher in CA's major agricultural areas 🟧 Foods are an important source of nitrate/nitrite 🟦 Surface water bodies in CA have nitrate levels well below the federal MCL* 🟧 Nitrate and nitrite have some therapeutic health benefits

*MCL - Maximum Contaminant Level

Key:
🟧 Health-related outcomes of nitrate and nitrite exposure
🟩 Economic costs of nitrate and nitrite in drinking water
🟦 Occurence of nitrate and nitrite in drinking water sources

1407

Figure 5.2.4. The range of $NO_3 + NO_2^-$ N concentrations expressed on both linear and log scales observed at different stations in the Central Valley and Delta. Box widths are proportional to the number of data points, shown next to the station name. Source: US EPA (2006). [Return to text]

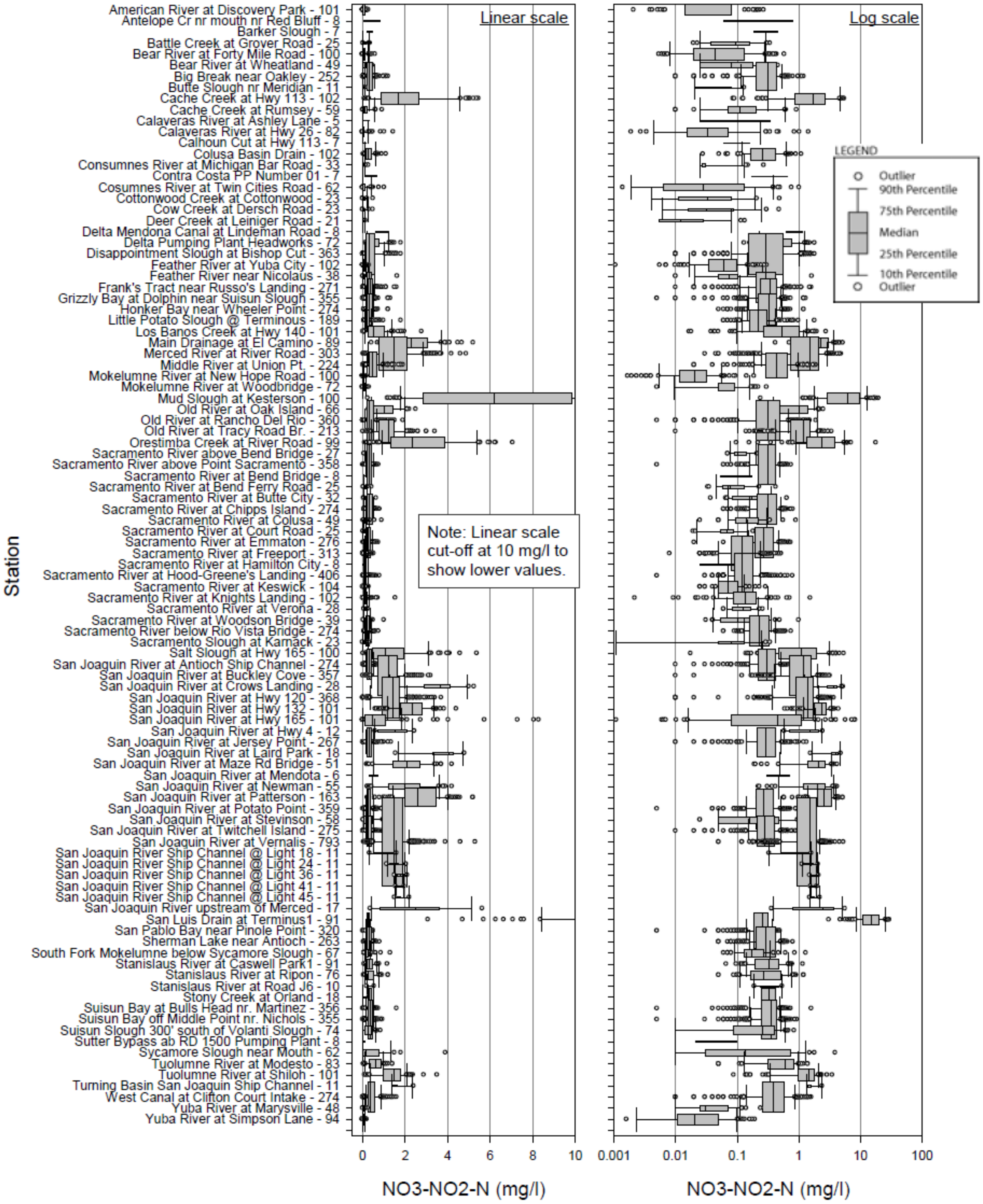


Figure 5.2.5. Groundwater nitrate concentrations measured in wells throughout California, 2009.

Green dots represent 0 - 10 mg NO_3^- L^{-1}. Yellow dots represent >10 - 45 mg NO_3^- L^{-1}. Red dots represent ≥ 45 mg NO_3^- L^{-1}. Source: Harter (2009). [Return to text]

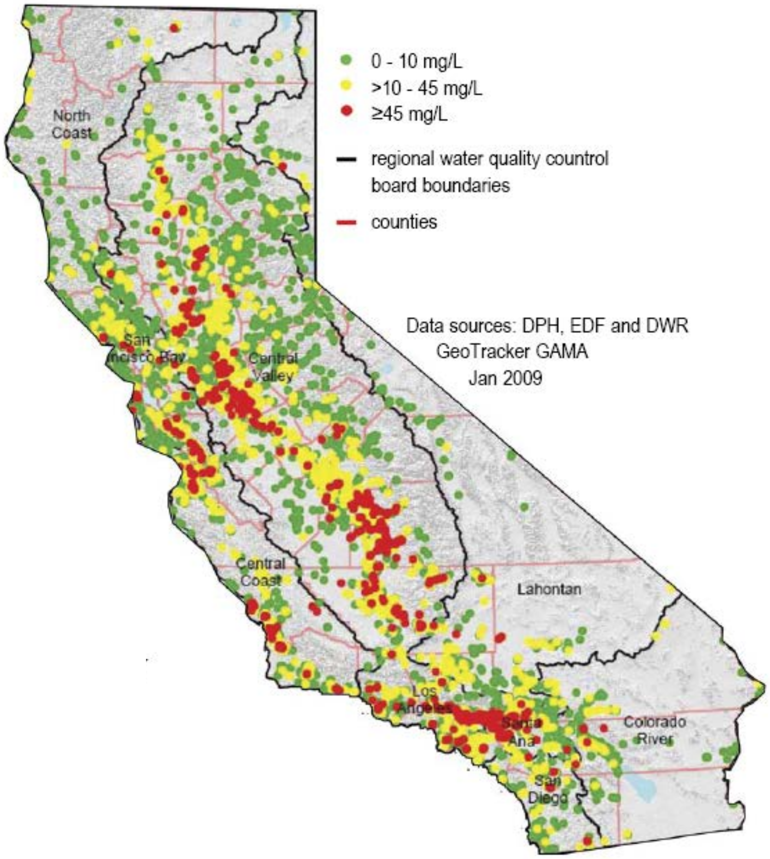


Figure 5.2.6. Average concentration of NO_3^- in wells during 2000-2009 in the Salinas Valley and Tulare Lake Basin. Red and deep red indicate wells over MCL. Yellow indicates wells above ½ MCL. Source: Boyle et al. (2012). [Return to text]

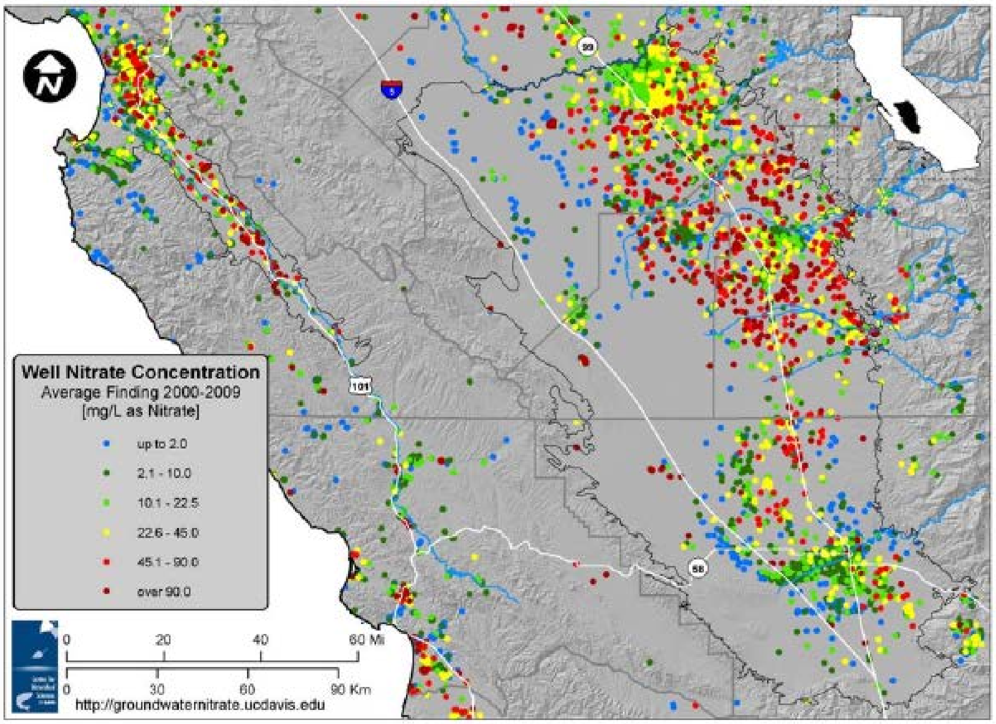


Figure 5.2.7. Estimated concentrations of nitrate in recharge and observed concentrations of groundwater nitrate in monitoring wells in Modesto, California 1950-2000. Observed concentrations of nitrate from ground water sampled in 2003 –2005 (Modesto) are plotted against corresponding interpreted ages from age-dating tracers. Estimated concentrations of nitrate from nitrogen fertilizer applications represent 50% of the nitrogen fertilizer applications divided by the area of fertilized land, dissolved in 0.4 m yr^{-1} of recharge in Modesto. MCL, maximum contaminant level. Source: Burow et al. (2008b). [Return to text]

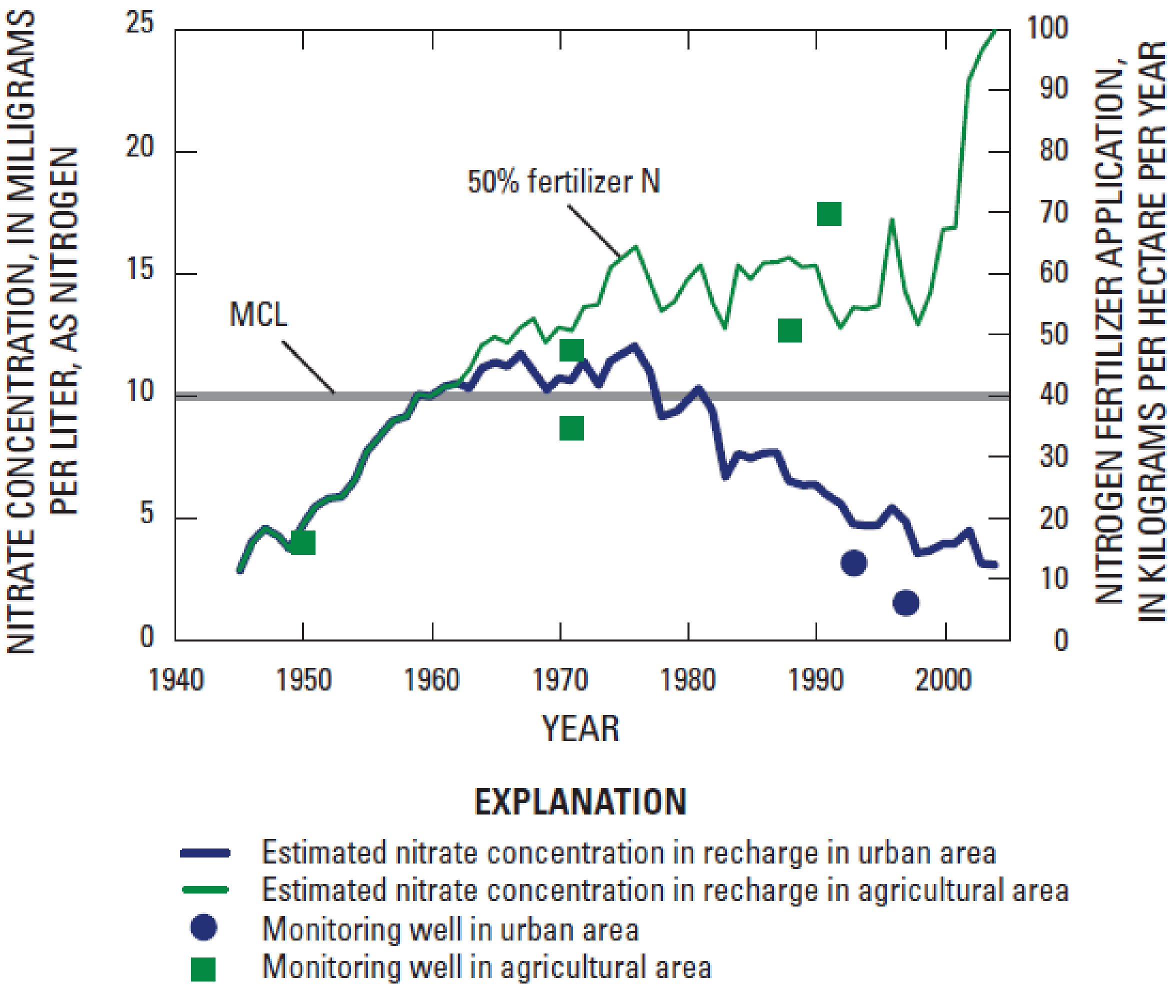


Figure 5.2.8. Five-year moving average of the percentage of Salinas Valley and Tulare Lake Basin wells with average annual NO_3 levels > 9 mg / L (background), 22.5 mg / L (1/2 MCL) and 45mg / L (MCL), 1950-2010. Prior to 1990 most wells sampled were public supply wells, There are only a small number of samples available for the 1950-1970 period thus trend during this period should be interpreted with care. In 2007, Central Valley dairies began testing domestic and irrigation wells which greatly increased the number of samples available for analysis. Source: Boyle et al. (2012). [Return to text]

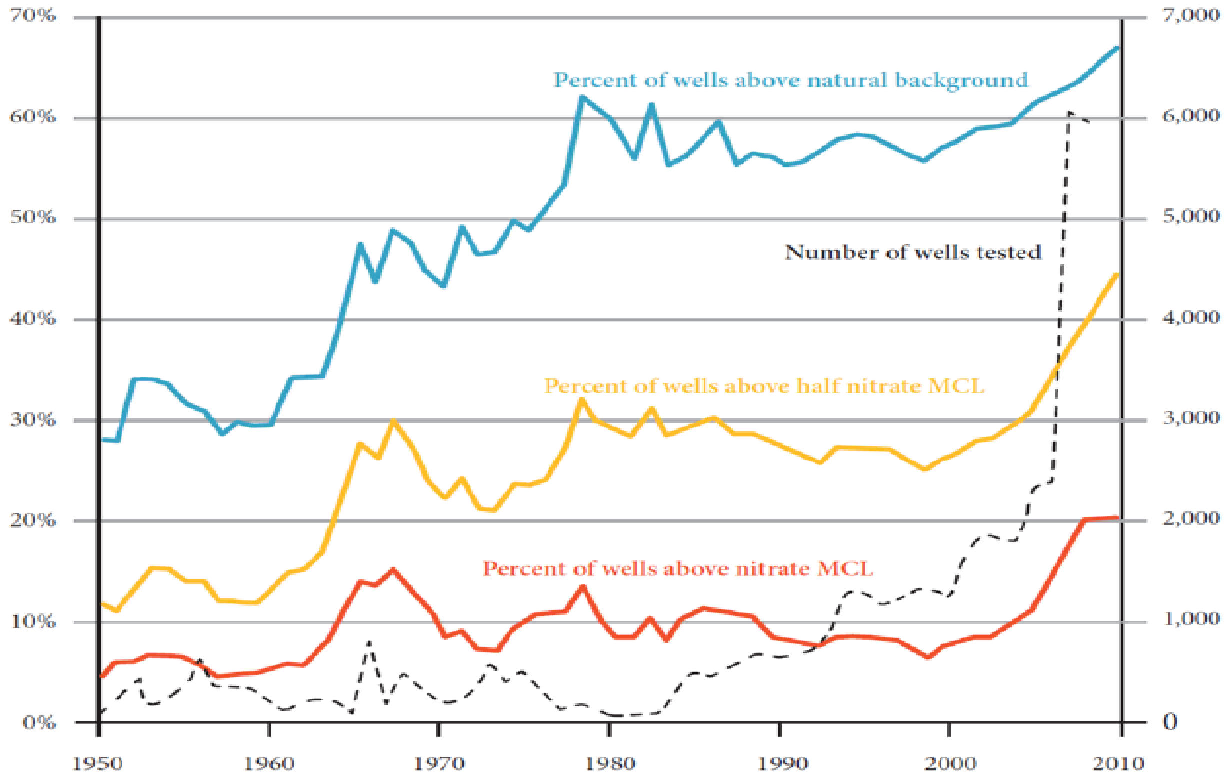


Figure 5.2.9. Cumulative distribution of N loading per year for model scenarios A, B, C, and D. The left panels correspond to the N loading output algorithm of Viers et al. (2012) expressed in kg N ha^{-1} and mg L^{-1} of groundwater as nitrate which are used as input to the Non-point Source Assessment Tool (NPSAT) simulation model. Scenarios A and D assume declines in N loading, while B and C assume increased N loading over time. Source: Boyle et al. (2012). [Return to text]

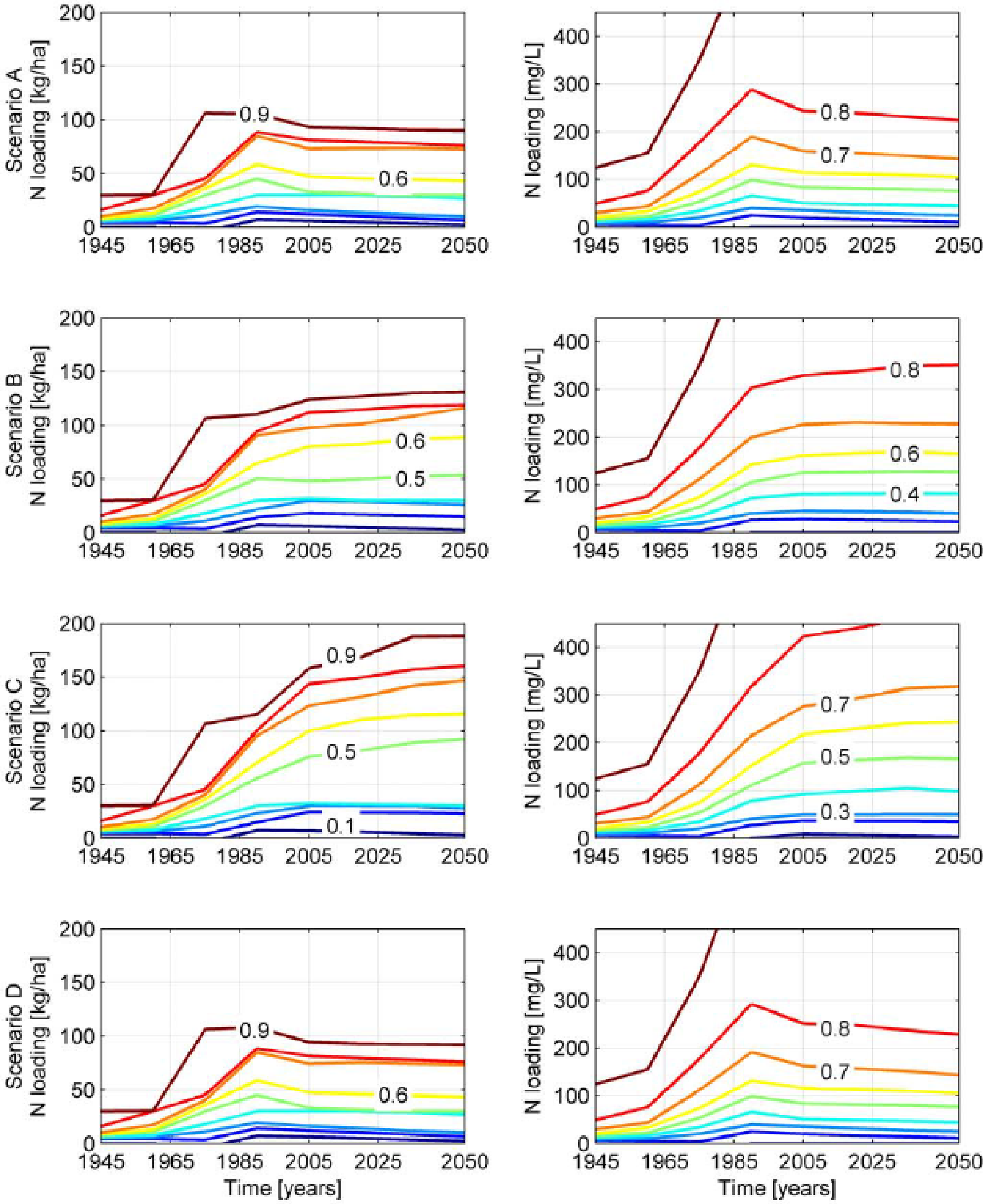


Figure 5.2.10. Historic and projected change in groundwater nitrate concentrations for six regions in the Tulare Lake Basin study area under model scenarios A and C. Scenario A assumes decreased N loading as nitrate over time, while scenario C assumes an increase in N loading. Source: Boyle et al. (2012). [Return to text]

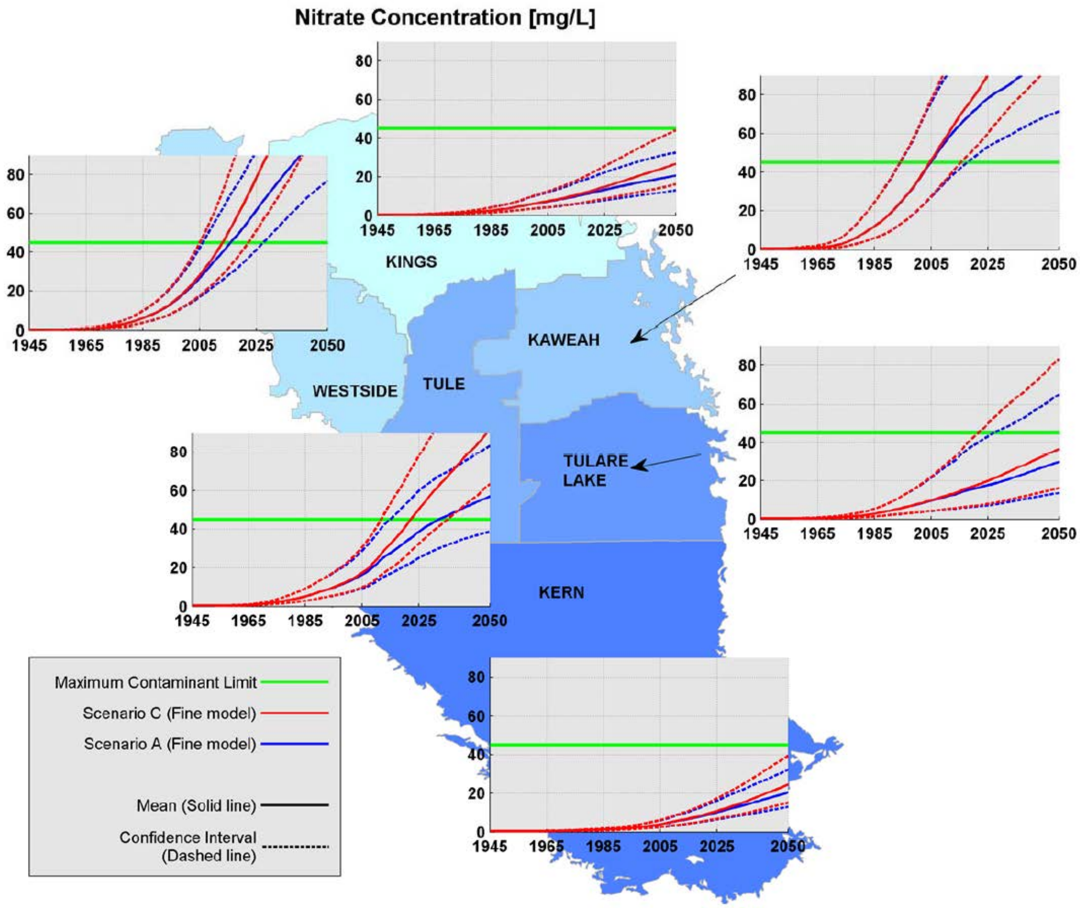


Figure 5.2.11. Average concentration of nitrate as N in California, 2008-2010. Wells with high levels have been displayed on top of the other wells to show areas where higher nitrate levels are more commonly observed. Data source: data compiled from the California State Water Resources Control Board. [Return to text]

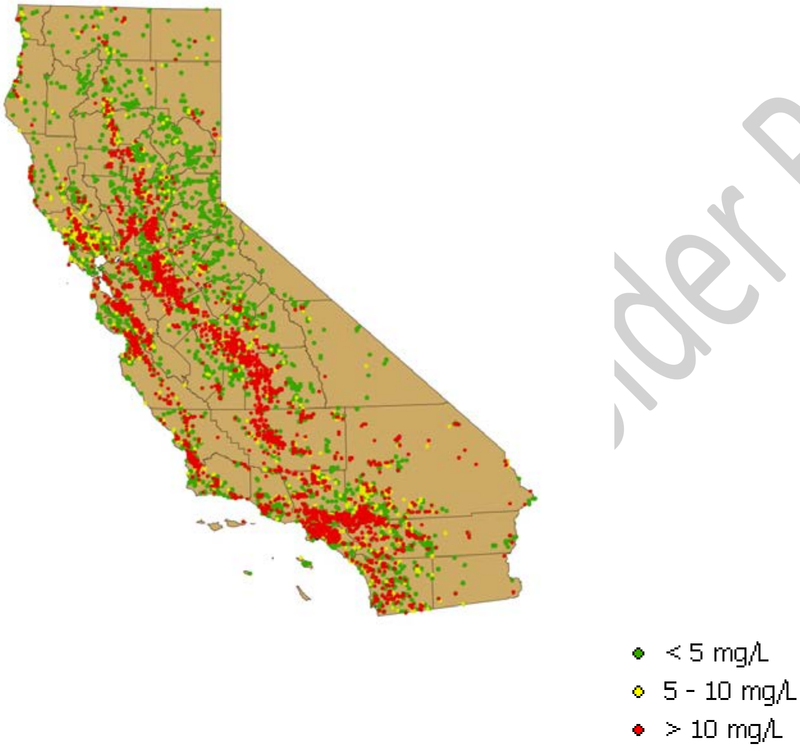


Figure 5.2.12. Proportion of wells in California with maximum nitrate concentrations greater than 10 mg nitrate L^{-1} (44.5 mg NO_3 L^{-1}) by county during 2008-1010. Data source: data compiled from the California State Water Resources Control Board. [Return to text]

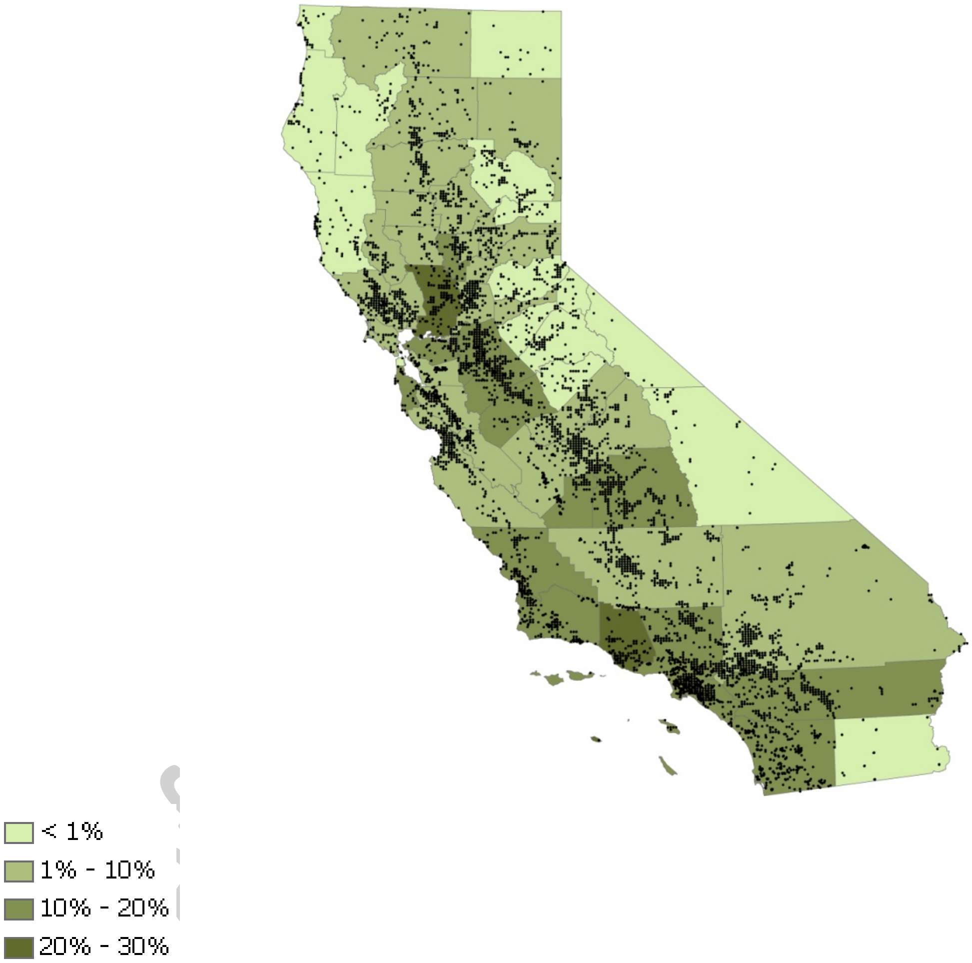


Figure 5.2.13. Proportion of wells in California with maximum nitrate concentrations greater than 3 mg nitrate L^{-1} (13.3 mg NO_3 L^{-1}) by county during 2008-1010. Data source: data compiled from the California State Water Resources Control Board. [Return to text]

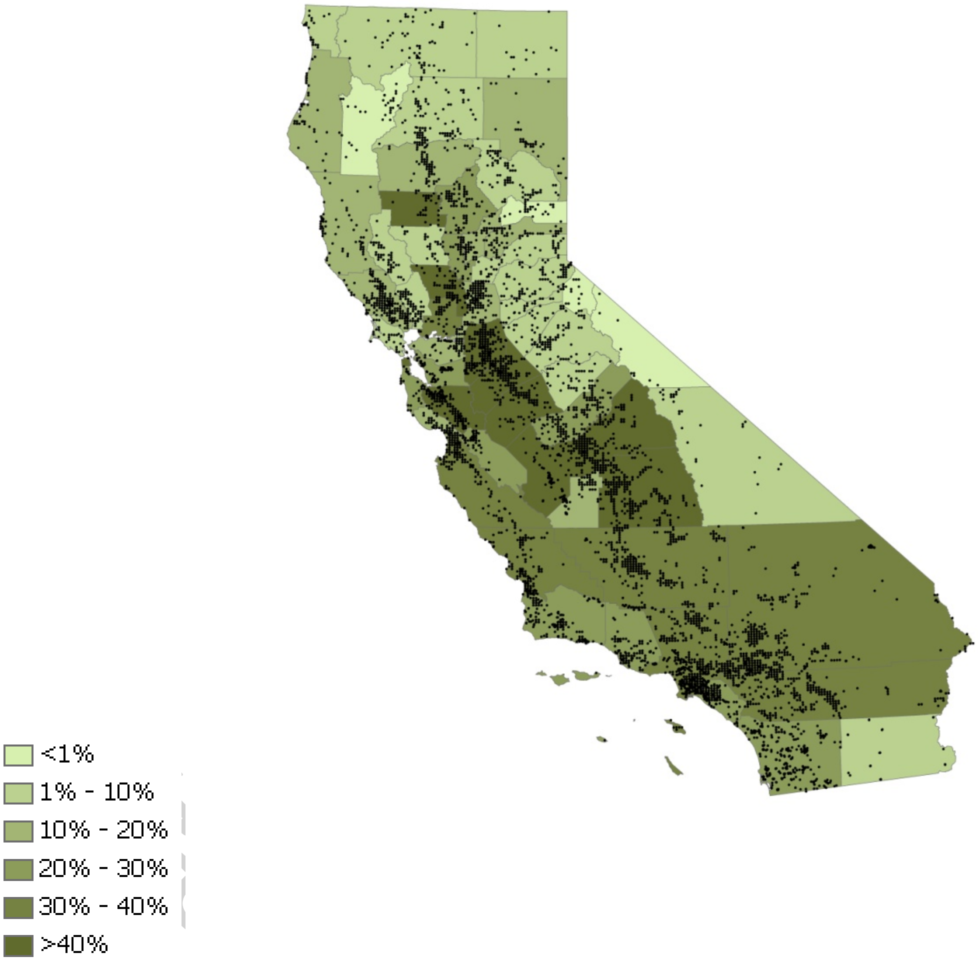


Figure 5.2.14. Classification of susceptible water systems and populations based on estimated vulnerability and water quality data in the Tulare Lake Basin and Salinas Valley study area. Source: Honeycutt et al. (2012). [Return to text]

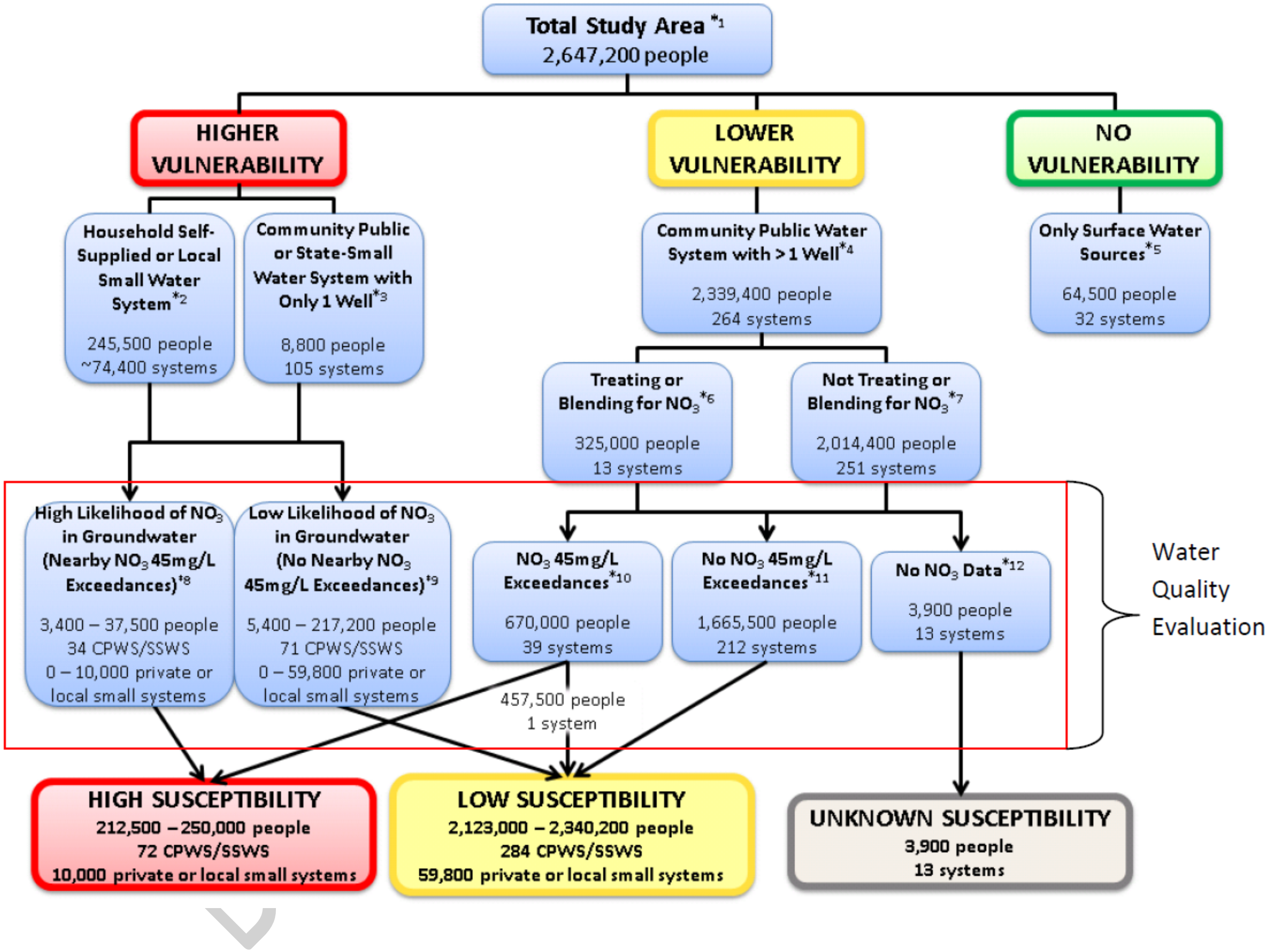


Table 5.0.1: Ecosystem services affected by increased N in the environment. Positive and negative impacts of N on various environmental and human health services are indicated using a plus or a minus. Source: Adapted from Compton et al (2011) and USEPA (2012). [Return to text]

Type	Ecosystem service	Impact on benefit	Mechanism of impact	N-related cause	Source
Provisioning	Production of food and materials	+	Increased production and nutritional quality of food crops	N fertilizer increases crop growth	Synthetic and organic N fertilizer
		+	Increased production of building materials and fiber for clothing or paper	N fertilizer increases crop growth	Synthetic and organic N fertilizer
		-	Soil acidification, nutrient imbalances and altered species composition	Acid deposition	Fossil fuel combustion, and agriculture
	Fuel Production	-/+	Increased N inputs required for some biofuel crops can affect other services	N fertilizer increases crop growth	Synthetic and organic N fertilizer
		+	Increased use of fossil fuels to improve human health and well-being across the globe[3]	Increase energy availability	Fossil fuel combustion
Supporting and Regulating	Drinking water	-	Increased nitrate concentrations lead to blue-baby syndrome, certain cancers	Nitrate into water	Agriculture
		-	Increased acidification and mobility of heavy metals and aluminum	Acid deposition	Fossil fuel combustion, and agriculture
	Clean Air	-	NO_x-driven increases in ozone and particulates exacerbate respiratory and cardiac conditions.	NO_x into air; $PM_{2.5}$, O_3 and related toxins	NO_y and NH_x from fossil fuel combustion, and agriculture
		-	Increased allergenic pollen production	Pollen production	Crops with airborne pollen
		-	Stimulation of ozone formation, which in turn can reduce agricultural and wood production	Ozone and acid deposition	Fossil fuel combustion
	Visibility	-	Increased NO_x in air stimulates formation of particulates, smog and regional haze	Fine particulate matter	NO_y and NH_x from fossil fuel combustion and agriculture
	Climate regulation	+/-	Variable and system-dependent impacts on net CO_2 exchange	N deposition	Fossil fuel combustion, agriculture

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[3] This impact is not addressed in Chapter 5. Please refer to Section 3.4 for a discussion of fuel combustion as a direct driver.

Type	Ecosystem service	Impact on benefit	Mechanism of impact	N-related cause	Source
		-	Stimulation of N_2O production, a powerful greenhouse gas	N_2O into air	Agriculture, animal manure management, sewage treatment, fossil fuel combustion
	UV Regulation	-	Increased N_2O release, which has strong-ozone-depleting potential	N_2O into air	Agriculture, animal manure management, sewage treatment, fossil fuel combustion
Cultural	Swimming	-	Stimulation of harmful algal blooms that release neurotoxins (interaction with phosphorus)	Excess nutrient loading, eutrophication, variable freshwater runoff	Fossil fuel combustion, agriculture
		-	Increased vector-borne diseases such as West Nile virus, malaria and cholera	Excess nutrient loading, eutrophication, variable freshwater runoff	Fossil fuel combustion, agriculture
	Fishing	+	Increased fish production and catch for some very N-limited coastal waters	Nutrient loading, N deposition	Fossil fuel combustion, agriculture
		-	Increased hypoxia and harmful algal blooms in coastal zones, closing fish and shellfish harvests	Excess nutrient loading, eutrophication, variable freshwater runoff	Fossil fuel combustion, agriculture
		-	Reduced number and species of recreational fisheries from acidification and eutrophication	Atmospheric deposition of HNO_3, NH_3 and ammonium compounds	Fossil fuel combustion, agriculture
	Hiking	-	Altered biodiversity, health and stability of natural ecosystems	N deposition	Fossil fuel combustion, agriculture
	Other	-	Altered biodiversity, food webs, habitat and species composition of natural ecosystems	N deposition	Fossil fuel combustion, agriculture
		-	Damage to buildings and structures from acids	Acid deposition	Fossil fuel combustion, agriculture
		+/-	Long range trans-boundary N transport and associated effects (both negative and positive)	N deposition	Fossil fuel combustion, agriculture

Table 5.2.1. Data sources with the total number of samples recorded, total number of sampled wells, location of wells, type of wells, and for the last decade (2000-2010) in the Tulare Lake Basin and Salinas Valley: number of wells measured, median nitrate concentration, and percentage of MCL exceedance for the Tulare Lake Basin and the Salinas Valley. Source: Adapted from Boyle et al. (2012). [Return to text]

			Tulare Lake Basin (2000-2010)[3]			Salinas Valley (2000-2010)[3]		
Data Source[1]	**Well Type[2]**	**Total # Samples**	**# of Wells**	**Median mg NO_3 L^{-1}**	**% > MCL**	**# of Wells**	**Median mg NO_3 L^{-1}**	**% > MCL**
CDPH	PS	62,153	1,769	12	6%	327	8	5%
CVRWB Dairy	D, I, M	11,300	6,459	22	31%			
DPR	D	814	71	40	45%			
DWR	I	44	28	1	0%			
ENVMON	M	2601	357		52%	180	27	44%
Fresno Co.	D	369	349	18	15%			
GAMA	D	141	141	38	43%			
Kern Co.	D, I	3825	361	5	7%			
Monterey Co. - Report	I, M	1018				98	14	36%
Monterey Co. - Geospatial	LS	1574				431	18	15%
Monterey Co. - Scanned	LS	5674				427	17	14%
NWIS	Misc.	2151	76	35	36%	4	0	0%
Tulare Co.	D	444	438	22	27%			
Westlands Water Distr.	I	77	31	4	0%			

[1] Data Source: CDPH: public supply well database; CVRWB Dairy: Central Valley RWB Dairy General Order; DWR data reports from the 1960-1970s, 1985; ENVMON: State Water Board Geotracker environmental monitoring wells with nitrate data (does not include data from the CVRWB dairy dataset); EPA: STORET dataset; Fresno County: Public Health Department; GAMA: State Water Board domestic well survey; Kern County: Water Agency; Monterey County, Reports: data published in reports by MCWRA; Monterey County, Geospatial: Health Department geospatial database; Monterey County, Scanned: Health Department scanned paper records; NWIS: USGS National Water Information System; Tulare County: Health and Human Services; Westlands Water District: district dataset. Some smaller datasets are not listed. Individual wells that are known to be monitored by multiple sources are here associated only with the data source reporting the first water quality record.

[2] D = domestic wells, I = irrigation wells, LS = local small system wells, M = monitoring wells, PS = public supply wells.

[3] Median and percent MCL exceedance were computed based on the annual mean nitrate concentration at each well for which data were available.

Table 5.2.2. Estimated capital, service, and annualized costs for single self-supplied households using alternative nitrate treatment and avoidance strategies. [Return to text]

Strategies	Capital Costs [1]	Service Costs [2]	Annualized Costs [3]	Location, Year	Reference
Treatment Strategies		---------------------------- \$ household^{-1} ----------------------------			
Reverse osmosis	\$855	\$87	\$130	Minnesota, 2006	Lewandowski et al. (2008)
	\$100-\$300	\$80-\$150	\$93-\$221	California, 2010	Pacific Institute (2011)
	\$406-\$1,200	\$190-\$200	\$250-\$360	California, 2010	Honeycutt et al. (2012)
	\$330-\$1,430	\$110-\$330	NR	Idaho, 2007	Jensen et al. (2012)
Distillation	\$961	NR	NR	Minnesota, 2006	Lewandowski et al. (2008)
	\$275-\$1,650	\$440-\$550	NR	Idaho, 2007	Jensen et al. (2012)
Ion exchange	\$1,600	NR	NR	Minnesota, 2006	Lewandowski et al. (2008)
	\$660-\$2,425	NR	NR	Idaho, 2007	Jensen et al. (2012)
Avoidance Strategies		---------------------------- \$ household^{-1} ----------------------------			
Bottled water	NA	\$190	\$190	Minnesota, 2006	Lewandowski et al. (2008)
	NA	\$380	\$380	California, 2010	Pacific Institute (2011)
	NA	\$1,260	\$1,260	California, 2010	Honeycutt et al. (2012)
Trucked Water	NR (storage)	\$950	\$950	California, 2010	Honeycutt et al., (2012)
Drill Deeper well[4]	\$50-\$200 ft^{-1}	\$60	\$860-3,300	California, 2010	Honeycutt et al., (2012)
Drill new well[5]	\$7,200	NR	\$144	Minnesota, 2006	Lewandowski et al. (2008)
	\$25,000-\$40,000	\$60	\$2,100-3,300	California, 2010	Honeycutt et al., (2012)

[1] NA indicates not applicable.

[2] NR indicates data not reported.

[3] See Lewandowski et al. (2008), Pacific Institute (2011), and Honeycutt et al. (2012) for assumptions, equations, discount rates and amortization periods used to calculate the annualized costs for various strategies.

[4] Honeycutt et al. (2012) assume that an existing well is deepened from 300 to 500 ft and has a pump efficiency of 0.60, and 0.15 kWh.

[5] Honeycutt et al. (2012) assume that a new 300 ft well with a pump efficiency of 0.60, and 0.15 kWh.

Table 5.2.3. Cost of community water supply projects proposed to California Dept. of Public Health and US Dept. of Agriculture during 2005-2009 where nitrate contamination was identified as the sole problem (Adapted from Pacific Institute, 2011). Data source: CDPH SRF Project Priority List (August, 2010) and CDPH Proposition 84 Draft Project Priority List (February, 2011). [Return to text]

Project type	# of Proposals	Ave. Project Cost	Min. Project Cost	Max. Project Cost	Total Cost of Proposals
		-- US $ --			
Feasibility study	6	$55,500	$25,000	$80,000	$333,000
Drill new well	17	$1,203,529	$100,000	$4,700,000	$20,460,000
Treatment	11	1,372,659	$150,000	$4,500,000	$15,099,250
Consolidation of sources	8	$1,169128	$250,000	$5,008,020	$9,353,020
Infrastructure to blend sources	1	$100,000	$100,000	$100,000	$100,000
Project type unclear	11	$774,718	$100,000	$2,000,000	$8,521,900
Drill new well and/or treatment	2	$581,500	$300,000	$863,000	$1,163,000
Drill new well and/or	2	$631,250	$262,500	$1,000,000	$1,262,500
Consolidation and/or treatment	4	$1,030,250	$621,000	$1,500,000	$4,121,000
Consolidation and/or blending	1	$1,500,000	$1,500,000	$1,500,000	$1,500,000
Total Proposed	**63**	**$982,757**	**$25,000**	**$5,008,020**	**$61,913,670**

Table 5.2.4 Cost of community water supply projects funded by California Dept. of Public Health and US Dept. of Agriculture during 2005-2009 where nitrate contamination was identified as one of several problems addressed (Adapted from Pacific Institute, 2011). Data source: CDPH SRF Project Priority List (August, 2010) and CDPH Proposition 84 Draft Project Priority List (February, 2011). [Return to text]

Project type	**# Funded by CDPH or USDA**	**Ave. Project Cost**	**Min. Project Cost**	**Max. Project Cost**	**Total Cost of f Funded Projects**
		-- US $ --			
Consolidation funded by CDPH	6	910,114	200,000	1,505,367	5,480,472
New well funded by CDPH	6	1,017,090 $	492,955	2,290,000	5,535,455
New well and consolidation funded by CDPH	2	4,306,225	1,150,000	7,462,450	8,612,450
New well funded by USDA	2	687,500	375,000	1,000,000	1,375,000
Total funded	**16**	**--**	**--**	**--**	**21,003,377**