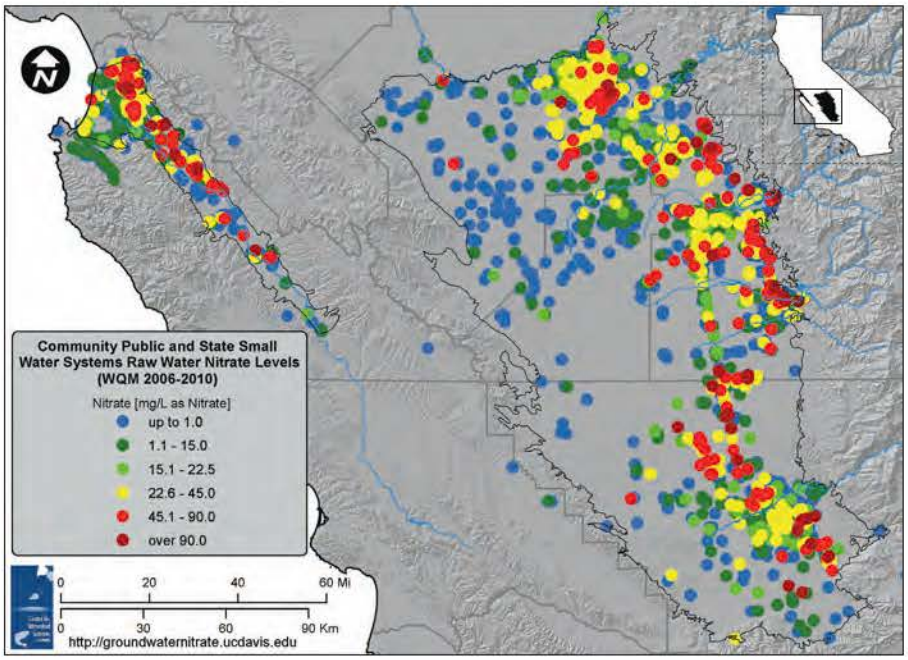
Nitrate in California's Groundwater

In 2008, Senate Bill SBX2 1 (Perata) required that the State Water Resources Control Board (State Water Board) prepare a Report to the Legislature to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the State … to clean up or treat groundwater, and ensure the provision of safe drinking water." The University of California prepared a set of reports under contract with the State Water Board for this purpose. This summary focuses on some major findings and promising actions, with details in the Main Report *Addressing Nitrate in California's Drinking Water* and accompanying Technical Reports.



Maximum reported raw-level nitrate concentration in community public water systems and state-documented state small water systems, 2006–2010. *Source:* CDPH PICME WQM Database.

Nitrate is one of the state's most widespread groundwater contaminants. It is principally a byproduct of nitrogen in fertilizer used to grow crops. Nitrate concentrations in public drinking water that exceed the **California Department of Public Health (CDPH) maximum contaminant level (MCL) for nitrate in drinking water of 45 milligrams per liter (as nitrate)** require often-costly water system actions to provide safe drinking water. This study focuses on the four-county **Tulare Lake Basin and the Monterey County portion of the Salinas Valley.** About 2.6 million people in these regions rely on groundwater for drinking water, including some of the poorest communities in California. The study area includes four of the nation's five counties with the largest agricultural production, representing about 40% of California's irrigated cropland and over half of the state's confined animal farming industry.

Nitrate in groundwater poses public health concerns for about 254,000 people in California's Tulare Lake Basin and Salinas Valley who are currently at risk for nitrate contamination of their drinking water. Of these, 220,000 are connected to community public (more than 14 connections) or state small water systems (5 to 14 connections), and 34,000 are served by private domestic wells or other systems that are smaller than the threshold for State regulation and which are largely unmonitored.

Summary of Key Findings

1. **Nitrate problems will likely worsen for several decades. For more than half a century, nitrate from fertilizer and animal waste has infiltrated into Tulare Lake Basin and Salinas Valley aquifers. Most nitrate in drinking water wells today was applied to the surface decades ago.**
2. **Agricultural fertilizers and animal wastes applied to cropland are by far the largest regional sources of nitrate in groundwater. Other sources can be locally relevant.**
3. **Nitrate loading reductions are possible, some at modest cost. Large reductions of nitrate loads to groundwater can have substantial economic cost.**
4. **Direct remediation to remove nitrate from large groundwater basins is extremely costly and not technically feasible. Instead, "pump-and-fertilize" and improved groundwater recharge management are less costly long-term alternatives.**
5. **Drinking water supply actions such as blending, treatment, and alternative water supplies are most cost-effective. Blending will become less available in many cases as nitrate pollution continues to spread.**
6. **Many small communities cannot afford safe drinking water treatment and supply actions. High fixed costs affect small systems disproportionately.**
7. **The most promising revenue source is a fee on nitrogen fertilizer use in these basins. A nitrogen fertilizer use fee could compensate affected small communities for mitigation expenses and effects of nitrate pollution.**
8. **Inconsistency and inaccessibility of data prevent effective and continuous assessment. A statewide effort is needed to integrate diverse water-related data collection activities by many state and local agencies.**

Adminsitrative Record
Page 3414

Table. Likely performance of promising state and agency actions for nitrate groundwater contamination.

Action	Safe Drinking Water	Groundwater Degradation	Economic Cost
No Legislation Required			
Safe Drinking Water Actions			
D1: Point-of-Use Treatment Option for Small Systems +	◆◆		low
D2: Small Water Systems Task Force +	◆		low
D3: Regionalization and Consolidation of Small Systems +	◆◆		low
Source Reduction Actions			
S1: Nitrogen/Nitrate Education and Research +		◆◆◆	low–moderate
S2: Nitrogen Accounting Task Force +		◆◆	low
Monitoring and Assessment			
M1: Regional Boards Define Areas at Risk +	◆◆◆	◆◆◆	low
M2: CDPH Monitors At-Risk Population +	◆	◆	low
M3: Consider Nitrogen Use Reporting +		◆◆	low
M4: Groundwater Data Task Force +	◆	◆	low
M5: Groundwater Task Force +	◆	◆	low
Funding			
F1: Nitrogen Fertilizer Mill Fee		◆◆◆	low
F2: Local Compensation Agreements for At-Risk Areas +	◆◆	◆	moderate
New Legislation Required			
D4: Domestic Well Testing in At-Risk Areas *	◆◆		low
D5: Stable Small System Funds	◆		moderate
Non-tax legislation could also strengthen and augment existing authority.			
Fiscal Legislation Required			
Source Reduction			
S3: Fertilizer Excise Fee	◆◆	◆	moderate
S4: Higher Fertilizer Fee in Areas at Risk	◆	◆	moderate
Funding Options			
F3: Fertilizer Excise Fee	◆◆	◆◆	moderate
F4: Water Use Fee	◆◆	◆◆	moderate

◆ Helpful

◆◆ Effective

◆◆◆ Essential

+ Legislation would strengthen.

* County health departments may have authority; CDPH requires legislation.

Adminsitrative Record
Page 34142

Addressing Nitrate in California's Drinking Water

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

EXECUTIVE SUMMARY

This Report and its associated eight Technical Reports were prepared by

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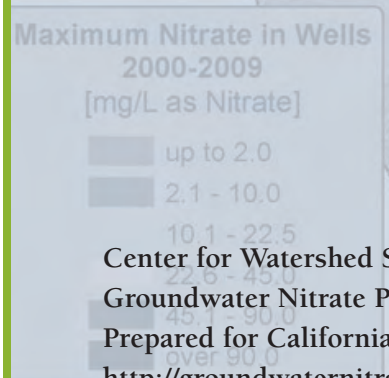


Center for Watershed Sciences • University of California, Davis
Groundwater Nitrate Project, Implementation of Senate Bill X2 1
Prepared for California State Water Resources Control Board • January 2012
http://groundwaternitrate.ucdavis.edu

Executive Summary

In 2008, Senate Bill SBX2 1 (Perata) was signed into law (Water Code Section 83002.5), requiring the State Water Resources Control Board (State Water Board), in consultation with other agencies, to prepare a Report to the Legislature to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the State…to clean up or treat groundwater, and ensure the provision of safe drinking water to all communities." The University of California prepared this Report under contract with the State Water Board as it prepares its Report to the Legislature.

This executive summary focuses on major findings and promising actions. Details can be found in the Main Report and eight accompanying Technical Reports.

Key Issues

Groundwater is essential to California, and nitrate is one of the state's most widespread groundwater contaminants. Nitrate in groundwater is principally a by-product of nitrogen use, a key input to agricultural production. However, too much intake of nitrate through drinking water can harm human health.

California's governments, communities, and agricultural industry have struggled over nitrate contamination for decades. **The California Department of Public Health (CDPH) has set the maximum contaminant level (MCL)** for nitrate in drinking water at 45 milligrams per liter (as nitrate). Nitrate concentrations in public drinking water supplies exceeding the MCL require water system actions to provide safe drinking water.

For this study, the four-county **Tulare Lake Basin and the Monterey County portion of the Salinas Valley are examined**. About 2.6 million people in these regions rely on groundwater for drinking water. The study area includes four of the nation's five counties with the largest agricultural production. It represents about 40% of California's irrigated cropland (including 80 different crops) and over half of California's dairy herd. Many communities in the area are among the poorest in California and have limited economic means or technical capacity to maintain safe drinking water given threats from nitrate and other contaminants.

Summary of Key Findings

1. **Nitrate problems will likely worsen for several decades. For more than half a century, nitrate from fertilizer and animal waste have infiltrated into Tulare Lake Basin and Salinas Valley aquifers. Most nitrate in drinking water wells today was applied to the surface decades ago.**
2. **Agricultural fertilizers and animal wastes applied to cropland are by far the largest regional sources of nitrate in groundwater. Other sources can be locally relevant.**
3. **Nitrate loading reductions are possible, some at modest cost. Large reductions of nitrate loads to groundwater can have substantial economic cost.**
4. **Direct remediation to remove nitrate from large groundwater basins is extremely costly and not technically feasible. Instead, "pump-and-fertilize" and improved groundwater recharge management are less costly long-term alternatives.**
5. **Drinking water supply actions such as blending, treatment, and alternative water supplies are most cost-effective. Blending will become less available in many cases as nitrate pollution continues to spread.**
6. **Many small communities cannot afford safe drinking water treatment and supply actions. High fixed costs affect small systems disproportionately.**
7. **The most promising revenue source is a fee on nitrogen fertilizer use in these basins. A nitrogen fertilizer use fee could compensate affected small communities for mitigation expenses and effects of nitrate pollution.**
8. **Inconsistency and inaccessibility of data prevent effective and continuous assessment. A statewide effort is needed to integrate diverse water-related data collection activities by many state and local agencies.**

Adminsitrative Record
Page 34144

Nitrate in groundwater poses two major problems and risks:

- **Public health concerns** for those exposed to nitrate contamination in drinking water; in California's Tulare Lake Basin and Salinas Valley, roughly 254,000 people are currently at risk for nitrate contamination of their drinking water. Of these, 220,000 are connected to community public (>14 connections) or state small water systems (5–14 connections), and 34,000 are served by private domestic wells or other systems smaller than the threshold for state or county regulation and which are largely unmonitored.
- **Financial costs of nitrate contamination** include additional drinking water treatment, new wells, monitoring, or other safe drinking water actions; over 1.3 million people are financially susceptible because nitrate in raw source water exceeds the MCL, requiring actions by drinking water systems. Nitrate contamination of drinking water sources will continue to increase as nitrogen from fertilizer, manure, and other sources applied in the last half century continues to percolate downward and flow toward drinking water wells.

Findings: Sources of Nitrate Pollution

Within the study area, human-generated nitrate sources to groundwater include (Figure ES-1):

- cropland (96% of total), where nitrogen applied to crops, but not removed by harvest, air emission, or runoff, is leached from the root zone to groundwater. Nitrogen intentionally or incidentally applied to cropland includes synthetic fertilizer (54%), animal manure (33%), irrigation source water (8%), atmospheric deposition (3%), and wastewater treatment and food processing facility effluent and associated solids (2%) (Figure ES-2);
- percolation of wastewater treatment plant (WWTP) and food processing (FP) wastes (1.5% of total);
- leachate from septic system drainfields (1% of total);
- urban parks, lawns, golf courses, and leaky sewer systems (less than 1% of total); and
- recharge from animal corrals and manure storage lagoons (less than 1% of total);
- downward migration of nitrate-contaminated water via wells (less than 1% of total).

Findings: Reducing Nitrate Pollution

Options for reducing nitrate pollution were identified for all sources. For cropland, where less than 40% of applied nitrogen is removed by crop harvest, 10 management measures (and 50 practices and technologies to achieve these management objectives) were reviewed that can reduce—but not eliminate—nitrate leaching to groundwater. These fall into four categories:

1. Design and operate irrigation and drainage systems to reduce deep percolation.
2. Manage crop plants to capture more nitrogen and decrease deep percolation.
3. Manage nitrogen fertilizer and manure to increase crop nitrogen use efficiency.
4. Improve storage and handling of fertilizers and manure to decrease off-target discharge.

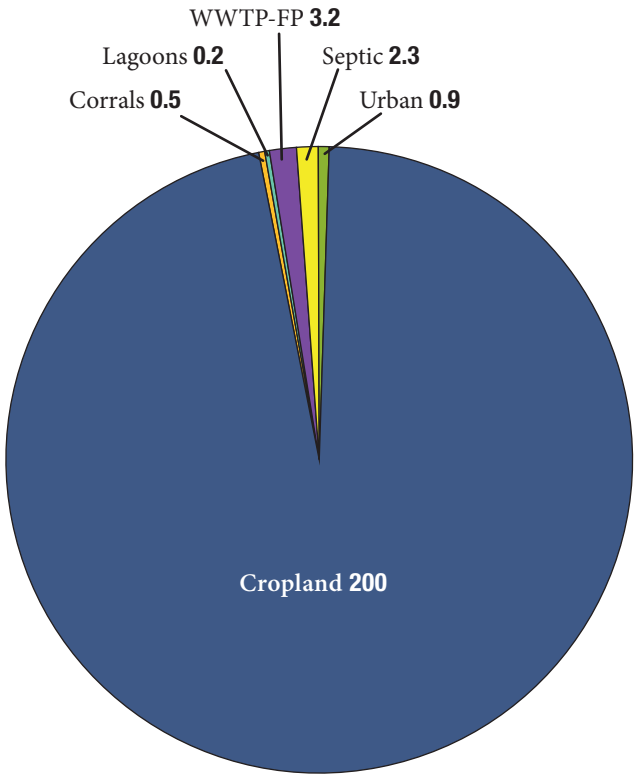
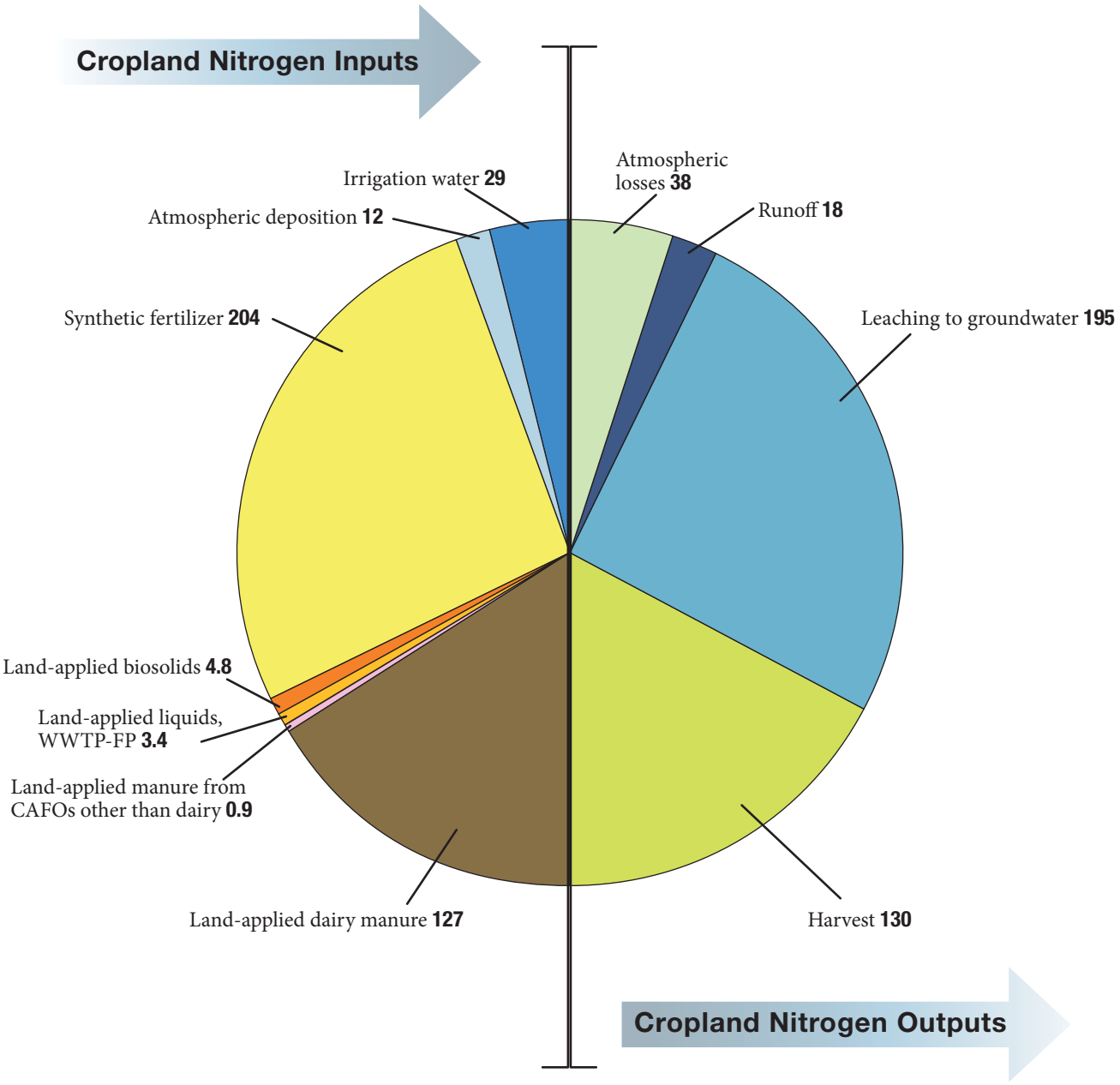


Figure ES-1. Estimated groundwater nitrate loading from major sources within the Tulare Lake Basin and Salinas Valley, in Gg nitrogen per year (1 Gg = 1,100 t).

Adminsitrative Record
Page 34145



Note: No mass balance was performed on 0.17 million ha (0.4 million ac) of nitrogen-fixing alfalfa, which is estimated to contribute an additional 5 Gg N/yr to groundwater. Groundwater nitrate loading from all non-cropland sources is about 8 Gg N/yr.

Figure ES-2. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. The left half of the pie chart represents total nitrogen inputs to 1.27 million ha (3.12 million ac) of cropland, not including alfalfa. The right half of the pie chart represents total nitrogen outputs with leaching to groundwater estimated by difference between the known inputs and the known outputs. Source: Viers et al. 2012.

Adminsitrative Record
Page 34146

Some of the needed improvements in nitrogen use efficiency by crops will require increased operating costs, capital improvements, and education. For some cropland, the high economic costs of nitrate source reduction sufficient to prevent groundwater degradation will likely hinder strict compliance with the state's current anti-degradation policy for groundwater (State Water Board Resolution 68-16).

Findings: Groundwater Nitrate Pollution

Groundwater nitrate data were assembled from nearly two dozen agencies and other sources (100,000 samples from nearly 20,000 wells). Of the 20,000 wells, 2,500 are frequently sampled public water supply wells (over 60,000 samples). In these public supply wells, about 1 in 10 raw water samples exceed the nitrate MCL. Apart from the recently established Central Valley dairy regulatory program in the Tulare Lake Basin, there are no existing regular well sampling programs for domestic and other private wells.

The largest percentages of groundwater nitrate MCL exceedances are in the eastern Tulare Lake Basin and in the northern, eastern, and central Salinas Valley, where about one-third of tested domestic and irrigation wells exceed the MCL. These same areas have seen a significant increase in nitrate concentrations over the past half century, although local conditions and short-term trends vary widely.

Travel times of nitrate from source to wells range from a few years to decades in domestic wells, and from years to many decades and even centuries in deeper production wells. This means that nitrate source reduction actions made today may not affect sources of drinking water for years to many decades.

Findings: Groundwater Remediation

Groundwater remediation is the cleanup of contaminated groundwater to within regulatory limits. Traditional pump-and-treat and in-place approaches to remediation, common for localized industrial contamination plumes, would cost billions of dollars over many decades to remove nitrate from groundwater in the Tulare Lake Basin and Salinas Valley. Timely cleanup of basin-scale nitrate contamination is not technically feasible.

Instead, long-term remediation by "pump-and-fertilize" would use existing agricultural wells to gradually remove nitrate-contaminated groundwater and treat the water by ensuring nitrate uptake by crops through appropriate nutrient and irrigation water management. Improved groundwater recharge management would provide clean groundwater recharge to mix with irrigation water recharge and partially mitigate nitrate levels in groundwater regionally.

Removal or reduction of contamination sources must accompany any successful remediation effort. Combining "pump-and-fertilize" with improved groundwater recharge management is more technically feasible and cost-effective.

Findings: Safe Drinking Water Supply

Nitrate contamination is widespread and increasing. Groundwater data show that 57% of the current population in the study area use a community public water system with recorded raw (untreated) nitrate concentrations that have exceeded the MCL at least once between 2006 and 2010. Continued basin-wide trends in nitrate groundwater concentration may raise the affected population to nearly 80% by 2050. Most of this population is protected by water system treatment, or alternative wells, at additional cost. But about 10% of the current population is at risk of nitrate contamination in their delivered drinking water, primarily in small systems and self-supplied households.

No single solution will fit every community affected by nitrate in groundwater. Each affected water system requires individual engineering and financial analyses.

Communities served by small systems vulnerable to nitrate contamination can (a) consolidate with a larger system that can provide safe drinking water to more customers; (b) consolidate with nearby small systems into a new single larger system that has a larger ratepayer base and economies of scale; (c) treat the contaminated water source; (d) switch to surface water; (e) use interim bottled water or point-of-use treatment until an approved long-term solution can be implemented; (f) drill a new well; or (g) blend contaminated wells with cleaner sources, at least temporarily.

There is significant engineering and economic potential for consolidating some systems. Consolidation can often permanently address nitrate problems, as well as many other problems faced by small water systems.

Solutions for self-supplied households (domestic well) or local small water systems (2–4 connections) affected by nitrate contamination are point-of-use (POU) or point-of-entry (POE) treatment and drilling a new or deeper well, albeit with no guarantee for safe drinking water.

Additional costs for safe drinking water solutions to nitrate contamination in the Tulare Lake Basin and Salinas Valley are roughly $20 and $36 million per year for the short- and long-term solutions, respectively. About $17 to $34 million per year will be needed to provide safe drinking water for 85 identified community public and state small water systems in the study area that exceed the nitrate drinking water MCL (serving an estimated 220,000 people). The annualized cost of providing nitrate-compliant drinking water to an estimated 10,000 affected rural households (34,000 people) using private

Adminsitrative Record
Page 34147

domestic wells or local small water systems is estimated to be at least $2.5 million for point-of-use treatment for drinking use only. The total cost for alternative solutions translates to $80 to $142 per affected person per year, $5 to $9 per irrigated acre per year, or $100 to $180 per ton of fertilizer nitrogen applied in these groundwater basins.

Findings: Regulatory, Funding, and Policy Options

To date, regulatory actions have been insufficient to control nitrate contamination of groundwater. Many options exist to regulate nitrate loading to groundwater, with no ideal solution. Nitrate source reductions will improve drinking water quality only after years to decades. Fertilizer regulations have lower monitoring and enforcement costs and information requirements than do nitrate leachate regulations, but they achieve nitrate reduction targets less directly. Costs to farmers can be lower with fertilizer fees or market-based regulations than with technology mandates or prescriptive standards. Market-based approaches may also encourage the development and adoption of new technologies to reduce fertilizer use.

Current funding programs cannot ensure safe drinking water in the Salinas Valley and Tulare Lake Basin. Small water system costs are high, and some of these systems already face chronic financial problems. Most current state funding for nitrate contamination problems is short term. Little funding is provided for regionalization and consolidation of drinking water systems. Policy options exist for long-term funding of safe drinking water, but all existing and potential options will require someone to bear the costs.

Promising Actions

Addressing groundwater nitrate contamination requires actions in four areas: (a) safe drinking water actions for affected areas, (b) reducing sources of nitrate contamination to groundwater, (c) monitoring and assessment of groundwater and drinking water, and (d) revenues to help fund solutions. Promising actions for legislative and state agency consideration in these areas appear below (see also Table ES-1). Starred (*) actions do not appear to require legislative action, but might benefit from it.

Safe Drinking Water Actions (D)

Safe drinking water actions are the most effective and economical short- and long-term approach to address nitrate contamination problems in the Tulare Lake Basin and Salinas Valley. These actions apply especially to small and self-supplied household water systems, which face the greatest financial and public health problems from nitrate groundwater contamination.

D1: Point-of-Use (POU) Treatment Option. CDPH reports on how to make economical household and point-of-use treatment for nitrate contamination an available and permanent solution for small water systems.*

D2: Small Water System Task Force. CalEPA and CDPH convene an independently led Task Force on Small Water Systems that would report on problems and solutions of small water and wastewater systems statewide as well as the efficacy of various state, county, and federal programs to aid small water and wastewater systems. Many nitrate contamination problems are symptomatic of the broad problems of small water and wastewater systems.*

D3: Regional Consolidation. CDPH and counties provide more legal, technical, and funding support for preparing consolidation of small water systems with nearby larger systems and creating new, regional safe drinking water solutions for groups of small water systems, where cost-effective.*

D4: Domestic Well Testing. In areas identified as being at risk for nitrate contamination by the California Water Boards, as a public health requirement, CDPH (a) mandates periodic nitrate testing for private domestic wells and local and state small systems and (b) requires disclosure of recent well tests for nitrate contamination on sales of residential property. County health departments also might impose such requirements.

D5: Stable Small System Funds. CDPH receives more stable funding to help support capital and operation and maintenance costs for new, cost-effective and sustainable safe drinking water solutions, particularly for disadvantaged communities (DACs).

Source Reduction Actions (S)

Reducing nitrate loading to groundwater is possible, sometimes at a modest expense. But nitrate source reduction works slowly and cannot effectively restore all affected aquifers to drinking water quality. Within the framework of Porter-Cologne, unless groundwater were to be de-designated as a drinking water source, reduction of nitrate loading to groundwater is required to improve long-term water quality. The following options seem most promising to reduce nitrate loading.

S1: Education and Research. California Department of Food and Agriculture (CDFA), in cooperation with the University of California and other organizations, develops and delivers a comprehensive educational and technical program to help farmers improve efficiency in nitrogen use (including manure) and reduce nitrate loading to groundwater. This could include a groundwater nitrate–focused element for the existing CDFA Fertilizer Research and Education Program, including "pump-and-fertilize" remediation and improved recharge options for groundwater cleanup.*

Adminsitrative Record
Page 34148

S2: Nitrogen Mass Accounting Task Force. CalEPA establishes a Task Force, including CDFA, to explore nitrogen mass balance accounting methods for regulating agricultural land uses in areas at risk for nitrate contamination, and to compare three long-term nitrogen source control approaches: (a) a cap and trade system; (b) farm-level nutrient management plans, standards, and penalties; and (c) nitrogen fertilizer fees.*

S3: Fertilizer Excise Fee. Significantly raising the cost of commercial fertilizer through a fee or excise tax would fund safe drinking water actions and monitoring and give further incentive to farmers for reducing nitrate contamination. An equivalent fee or excise tax could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

S4: Higher Fertilizer Fee in Areas at Risk. Areas declared to be at risk for nitrate contamination might be authorized to maintain a higher set of excise fees on nitrogen fertilizer applications (including synthetic fertilizer, manure, waste effluent, biosolids, and organic amendments), perhaps as part of a local safe drinking water compensation agreement.

Monitoring and Assessment (M)

Monitoring and assessment is needed to better assess the evolving nitrate pollution problem and the effectiveness of safe drinking water and nitrate source loading reduction actions. Such activities should be integrated with other state agricultural, environmental, and land use management; groundwater data; and assessment programs (source loading reduction actions)—along with other drinking water, treatment, and wastewater management programs (safe drinking water actions).

M1: Define Areas at Risk. Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.*

M2: Monitor at-Risk Population. CDPH and the State Water Board, in coordination with DWR and CDFA, issue a report every 5 years to identify populations at risk of contaminated drinking water and to monitor long-term trends of the state's success in providing safe drinking water as a supplement to the California Water Plan Update.*

M3: Learn from Department of Pesticide Regulation Programs. CalEPA and CDFA examine successful DPR data collection, analysis, education, and enforcement programs for lessons in managing nitrogen and other agricultural contaminants, and consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.*

M4: Groundwater Data Task Force. CalEPA , in coordination with CalNRA and CDPH, convenes an independently led State Groundwater Data Task Force to examine the efficacy of current state and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.

M5: Groundwater Task Force. CalEPA, CalNRA, and CDPH maintain a joint, permanent, and independently led State Groundwater Task Force to periodically assess and coordinate state technical and regulatory groundwater programs in terms of effectiveness at addressing California's groundwater quality and quantity problems. These reports would be incorporated into each California Water Plan Update.*

Funding (F)

Little effective action can occur without funding. Four funding options seem most promising, individually or in combination. State funding from fees on nitrogen or water use, which directly affect nitrate groundwater contamination, seem particularly promising and appropriate.

F1: Mill Fee. Increase the mill assessment rate on nitrogen fertilizer to the full authorized amount (CAL. FAC Code Section 14611). This would raise roughly $1 million/year statewide and is authorized for fertilizer use research and education.*

F2: Local Compensation Agreements. Regional Water Boards can require and arrange for local compensation of affected drinking water users under Porter-Cologne Act Water Code Section 13304. Strengthening existing authority, the Legislature could require that a Regional Water Board finding that an area is at risk of groundwater nitrate contamination for drinking water be accompanied by a cleanup and abatement order requiring overlying, current sources of nitrate to financially support safe drinking water actions acceptable to the local County Health Department. This might take the form of a local "liability district."*

F3: Fertilizer Excise Fee. Introduce a substantial fee on nitrogen fertilizer sales or use, statewide or regionally, to fund safe drinking water actions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs.

F4: Water Use Fee. A more comprehensive statewide fee on water use could support many beneficial activities. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities.

Adminsitrative Record
Page 34149

Table ES-1. Likely performance of promising state and agency actions for nitrate groundwater contamination.

Action	Safe Drinking Water	Groundwater Degradation	Economic Cost
No Legislation Required			
Safe Drinking Water Actions			
D1: Point-of-Use Treatment Option for Small Systems +	◆◆		low
D2: Small Water Systems Task Force +	◆		low
D3: Regionalization and Consolidation of Small Systems +	◆◆		low
Source Reduction Actions			
S1: Nitrogen/Nitrate Education and Research +		◆◆◆	low–moderate
S2: Nitrogen Accounting Task Force +		◆◆	low
Monitoring and Assessment			
M1: Regional Boards Define Areas at Risk +	◆◆◆	◆◆◆	low
M2: CDPH Monitors At-Risk Population +	◆	◆	low
M3: Implement Nitrogen Use Reporting +		◆◆	low
M4: Groundwater Data Task Force +	◆	◆	low
M5: Groundwater Task Force +	◆	◆	low
Funding			
F1: Nitrogen Fertilizer Mill Fee		◆◆◆	low
F2: Local Compensation Agreements for Water +	◆◆	◆	moderate
New Legislation Required			
D4: Domestic Well Testing *	◆◆		low
D5: Stable Small System Funds	◆		moderate
Non-tax legislation could also strengthen and augment existing authority.			
Fiscal Legislation Required			
Source Reduction			
S3: Fertilizer Excise Fee	◆◆	◆	moderate
S4: Higher Fertilizer Fee in Areas at Risk	◆	◆	moderate
Funding Options			
F3: Fertilizer Excise Fee	◆◆	◆◆	moderate
F4: Water Use Fee	◆◆	◆◆	moderate

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+ Legislation would strengthen.

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Adminsitrative Record
Page 34150

Addressing Nitrate in California's Drinking Water

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
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Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 34151

The health of our waters
is the principal measure
of how we live on the land.

—Luna Leopold

Adminsitrative Record
Page 34152

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Groundwater Nitrate Project, Implementation of Senate Bill X2 1

Prepared for California State Water Resources Control Board • January 2012

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Adminsitrative Record
Page 34153

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Contents

List of Tables v
List of Figures vi
Acknowledgments vii
Acronyms and Abbreviations viii

Executive Summary 1
Key Issues 2
Summary of Key Findings 2
Findings: Sources of Nitrate Pollution 3
Findings: Reducing Nitrate Pollution 4
Findings: Groundwater Nitrate Pollution 4
Findings: Groundwater Remediation 4
Findings: Safe Drinking Water Supply 4
Findings: Regulatory, Funding, and Policy Options 5
Promising Actions 5
Safe Drinking Water Actions 5
Source Reduction Actions 6
Monitoring and Assessment 6
Funding 7

1 Introduction 9

2 Sources of Groundwater Nitrate 15
2.1 Nitrogen Cycle: Basic Concepts 15
2.2 Sources of Nitrate Discharge to Groundwater 15
Natural Nitrate Sources 15
Human Nitrate Sources 16
Groundwater Nitrate Contributions by Source Category 17
Agricultural Sources 17
Urban and Domestic Sources 24
General Sources 29
2.3 Reducing Nitrate Source Emissions to Groundwater 29
Reducing Nitrate Loading from Irrigated Cropland and Livestock Operations 29
Reducing Nitrate Leaching from Municipal Wastewater Treatment and Food Processing Plants 34
Reducing Nitrate Contributions from Leaking Sewer Pipes and Septic Systems 34
Reducing Nitrate Leaching from Turfgrass in Urban Areas 34
Reducing Nitrate Transfer and Loading from Wells 34

Adminsitrative Record
Page 34155

3 Impact: Groundwater Nitrate Occurrence 35
3.1 Current Groundwater Quality Status 35
3.2 Cleanup of Groundwater: Groundwater Remediation 36
3.3 Existing Regulatory and Funding Programs for Nitrate Groundwater Contamination 39
The Dutch Experience 46
4 Impact: Drinking Water Contamination 47
4.1 Susceptible Populations 47
4.2 Alternative Water Supply and Treatment 51
Community Public Water System Options 52
Self-Supplied Households and Local Small Water System Options 54
Treatment to Remove Nitrate 55
Consolidation and Regionalization 57
4.3 Comparison and Discussion 58
Options for Small Community Public Water Systems 58
Options for Self-Supplied Households and Local Small Water Systems 60
4.4 Cost of Providing Safe Drinking Water 62
5 Policy Options for Nitrate Source Reduction and Funding 63
5.1 Nitrate Source Reduction Policy Options 63
Major Findings: Future Source Reduction Options 64
5.2 Funding Options 64
Major Findings: Future Funding Options 65
Payment for Ecosystem Services in New York City 66
6 Promising Solutions 67
6.1 Areas of Promising Action 67
Safe Drinking Water Actions 67
Source Reduction Actions 67
Monitoring and Assessment 69
Funding 69
6.2 Developing an Effective Solution Strategy 70
Options without Tax Legislation 70
Options Requiring Tax or Fee Legislation 71
6.3 Getting Organized 71
Informational Actions 71
Task Forces 71
6.4 Dilemmas for State Action 72
7 Conclusions 73
Literature Cited 75

Adminsitrative Record
Page 34156

Tables

Table ES-1. Performance of state and agency actions for nitrate groundwater contamination 8
Table 1. Major sources of groundwater nitrate. 18
Table 2. Major nitrogen fluxes to and from cropland 26
Table 3. Total nitrogen discharge to land and average total nitrogen concentration in discharge 27
Table 4. Measures for improving nitrogen use efficiency and decreasing nitrate leaching 30
Table 5. Effect of nitrate load reduction on applied water, revenues, crop area, and nitrogen application 32
Table 6. Total number of wells and samples, location, nitrate concentration, and MCL exceedance 36
Table 7. Total groundwater and remediation volume by sub-basin 41
Table 8a. Programs and statutes for reducing nitrate contamination in groundwater 42
Table 8b. Current planning and regulatory programs regarding nitrate in groundwater 43
Table 9. Existing funding sources for safe drinking water 44
Table 10. Number of years until community public water supply sources exceed nitrate MCL. 52
Table 11. Options for community public water systems 53
Table 12. Options for self-supplied households and local small water systems 55
Table 13. Influence of nitrate concentration on treatment selection 58
Table 14. Safe drinking water option costs for household and small water systems 60
Table 15. Least cost of short-term alternative water supply for community public water systems. 61
Table 16. Least cost of long-term alternative water supply for community public water systems. 62
Table 17. Regulatory options to reduce nitrate contamination to groundwater 63
Table 18. Future state funding options 65
Table 19. Performance of state and agency actions for nitrate groundwater contamination 68

Adminsitrative Record
Page 34157

Figures

Figure ES-1. Estimated groundwater nitrate loading from major sources. 3
Figure ES-2. Overview of cropland input and output 4
Figure 1. Maximum of raw water nitrate levels in public water systems, 2006–2010 10
Figure 2. Tulare Lake Basin and Salinas Valley 13
Figure 3. Overview of cropland input and output 19
Figure 4. Current typical fertilization rate for irrigated agricultural cropland 20
Figure 5. Estimated current average nitrogen uptake in harvested crop materials 22
Figure 6. Estimated historical agricultural development 23
Figure 7. Nitrogen applied, nitrogen harvested, and suggested nitrogen application reduction 25
Figure 8. Septic-derived nitrate leaching rates 28
Figure 9. Overall hazard index 31
Figure 10. Decrease in net farm income as a function of nitrogen efficiency improvements 33
Figure 11. Mean of the time-average nitrate concentration in wells, 2000–2009 38
Figure 12. Maximum nitrate concentrations, 2000–2009 39
Figure 13. Five-year moving average of wells exceeding the MCL in any given year 40
Figure 14. Public water system size distribution 48
Figure 15. Estimated locations of unregulated and unmonitored small water systems 48
Figure 16. Susceptible populations based on estimated vulnerability and water quality 49
Figure 17. Small water systems located near wells with nitrate greater than the MCL 50
Figure 18. Relationship between disadvantaged communities and delivered water quality 51
Figure 19. California drinking water systems treating or blending for nitrate 56
Figure 20. Study area systems treating or blending for nitrate 57
Figure 21. Minimum distance from a small water system to a larger system 59
Figure 22. Lowest-cost alternative supply for community public small water systems 61

Adminsitrative Record
Page 34158

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Adminsitrative Record
Page 34159

Acronyms and Abbreviations

AB	Assembly Bill
ac	Acre (about 0.4 hectares)
AF	Acre-foot (about 1,233 cubic meters)
AMBAG	Association of Monterey Bay Area Governments
AQUA	Association of People United for Water
ARRA	American Recovery and Reinvestment Act
AWP	Agricultural Waiver Program
BD	Biological Denitrification
BMP	Best Management Practices
CAA	Cleanup and Abatement Account
CalEPA	California Environmental Protection Agency
CAL FAC	California Food and Agriculture Code
CalNRA	California Natural Resources Agency
CCR	California Code of Regulations
CCR	Consumer Confidence Report
CDBG	Community Development Block Grant
CDFA	California Department of Food and Agriculture
CDPH	California Department of Public Health
CoBank	Cooperative Bank
CPWS	Community Public Water System
CRWA	California Rural Water Association
CV-SALTS	Central Valley Salinity Alternative for Long-Term Sustainability
CVSC	Central Valley Salinity Coalition
CWA	Clean Water Act
CWC	Community Water Center
CWSRF	Clean Water State Revolving Fund
DAC	Disadvantaged Communities
DPEIR	Draft Program Environmental Impact Report (of the Central Valley ILRP)
DPR	California Department of Pesticide Regulation
DWR	California Department of Water Resources
DWSAP	Drinking Water Source Assessment and Protection
DWSRF	Drinking Water State Revolving Fund
EDA	U.S. Economic Development Administration

Adminsitrative Record
Page 34160

EDR	Electrodialysis Reversal
ERG	Expense Reimbursement Grant Program
ERP-ETT	Enforcement Response Policy and Enforcement Targeting Tool
FFLDERS	Feed, Fertilizer, Livestock, Drugs, and Egg Regulatory Services
FMIP	Fertilizing Materials Inspection Program
FP	Food Processors
FREP	Fertilizer Research and Education Program
GAMA	Groundwater Ambient Monitoring and Assessment
Gg	Gigagram (1 million kilograms, about 1,100 tons)
ha	Hectare (about 2.5 acres)
HAC	Housing Assistance Council
HSNC	Historical Significant Non-Compliers
HUD	U.S. Department of Housing and Urban Development
I-Bank	California Infrastructure and Economic Development Bank
ILRP	Irrigated Lands Regulatory Program
IRWM	Integrated Regional Water Management
ISRF	Infrastructure State Revolving Fund
IX	Ion Exchange
KCWA	Kern County Water Agency
kg	Kilogram (about 2.2 pounds)
L	Liter (about 1.06 liquid quarts)
lb	Pound (about 0.45 kilogram)
LLNL	Lawrence Livermore National Lab
MCL	Maximum Contaminant Level
MCWRA	Monterey County Water Resources Agency
mg	Milligram (about 0.00003 ounce)
MHI	Median Household Income
MUN	Municipal or domestic water supply (beneficial use)
NDWC	National Drinking Water Clearinghouse
NMP	Nutrient Management Plan
NPDES	National Pollutant Discharge Elimination System
NRWA	National Rural Water Association
NUE	Nitrogen Use Efficiency
NWG	Nitrate Working Group

Adminsitrative Record
Page 34161

O&M	Operations and Maintenance
OW	EPA's Office of Water
PES	Payment for Ecosystem Services
PHG	Public Health Goal
PNB	Partial Nutrient Balance
POE	Point-of-Entry (for household water treatment)
Porter-Cologne	Porter-Cologne Water Quality Control Act (California Water Code § 13000 et seq.)
POU	Point-of-Use (for household water treatment)
PPL	Project Priority List
PWS	Public Water System
RCAC	Rural Community Assistance Corporation
RCAP	Rural Community Assistance and Partnership
RO	Reverse Osmosis
RUS	Rural Utilities Service
SB	Senate Bill
SDAC	Severely Disadvantaged Communities
SDWA	Safe Drinking Water Act
SDWSRF	Safe Drinking Water State Revolving Fund
SEP	Supplement Environmental Program
SHE	Self-Help Enterprises
SRF	State Revolving Fund
SSWS	State Small Water System
SV	Salinas Valley
t	Ton (U.S. short ton, about 907 kilograms)
TLB	Tulare Lake Basin
U.S. EPA	United States Environmental Protection Agency
U.S.C.	United States Code
USDA	United States Department of Agriculture
USGS	U.S. Geological Survey
WARMF	Watershed Analysis Risk Management Framework
WDR	Waste Discharge Requirements
WEP	Water Environmental Program
WMP	Waste Management Plan
WWTP	Wastewater Treatment Plant

Adminsitrative Record
Page 34162

Executive Summary

Adminsitrative Record
Page 34163

Executive Summary

In 2008, Senate Bill SBX2 1 (Perata) was signed into law (Water Code Section 83002.5), requiring the State Water Resources Control Board (State Water Board), in consultation with other agencies, to prepare a Report to the Legislature to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the State… to clean up or treat groundwater, and ensure the provision of safe drinking water to all communities." The University of California prepared this Report under contract with the State Water Board as it prepares its Report to the Legislature.

This executive summary focuses on major findings and promising actions. Details can be found in the Main Report and eight accompanying Technical Reports.

Key Issues

Groundwater is essential to California, and nitrate is one of the state's most widespread groundwater contaminants. Nitrate in groundwater is principally a by-product of nitrogen use, a key input to agricultural production. However, too much intake of nitrate through drinking water can harm human health.

California's governments, communities, and agricultural industry have struggled over nitrate contamination for decades. **The California Department of Public Health (CDPH) has set the maximum contaminant level (MCL)** for nitrate in drinking water at 45 milligrams per liter (as nitrate). Nitrate concentrations in public drinking water supplies exceeding the MCL require water system actions to provide safe drinking water.

For this study, the four-county **Tulare Lake Basin and the Monterey County portion of the Salinas Valley are examined**. About 2.6 million people in these regions rely on groundwater for drinking water. The study area includes four of the nation's five counties with the largest agricultural production. It represents about 40% of California's irrigated cropland (including 80 different crops) and over half of California's dairy herd. Many communities in the area are among the poorest in California and have limited economic means or technical capacity to maintain safe drinking water given threats from nitrate and other contaminants.

Summary of Key Findings

1. **Nitrate problems will likely worsen for several decades. For more than half a century, nitrate from fertilizer and animal waste have infiltrated into Tulare Lake Basin and Salinas Valley aquifers. Most nitrate in drinking water wells today was applied to the surface decades ago.**
2. **Agricultural fertilizers and animal wastes applied to cropland are by far the largest regional sources of nitrate in groundwater. Other sources can be locally relevant.**
3. **Nitrate loading reductions are possible, some at modest cost. Large reductions of nitrate loads to groundwater can have substantial economic cost.**
4. **Direct remediation to remove nitrate from large groundwater basins is extremely costly and not technically feasible. Instead, "pump-and-fertilize" and improved groundwater recharge management are less costly long-term alternatives.**
5. **Drinking water supply actions such as blending, treatment, and alternative water supplies are most cost-effective. Blending will become less available in many cases as nitrate pollution continues to spread.**
6. **Many small communities cannot afford safe drinking water treatment and supply actions. High fixed costs affect small systems disproportionately.**
7. **The most promising revenue source is a fee on nitrogen fertilizer use in these basins. A nitrogen fertilizer use fee could compensate affected small communities for mitigation expenses and effects of nitrate pollution.**
8. **Inconsistency and inaccessibility of data prevent effective and continuous assessment. A statewide effort is needed to integrate diverse water-related data collection activities by many state and local agencies.**

Adminsitrative Record
Page 34164

Nitrate in groundwater poses two major problems and risks:

- **Public health concerns** for those exposed to nitrate contamination in drinking water; in California's Tulare Lake Basin and Salinas Valley, roughly 254,000 people are currently at risk for nitrate contamination of their drinking water. Of these, 220,000 are connected to community public (>14 connections) or state small water systems (5–14 connections), and 34,000 are served by private domestic wells or other systems smaller than the threshold for state or county regulation and which are largely unmonitored.
- **Financial costs of nitrate contamination** include additional drinking water treatment, new wells, monitoring, or other safe drinking water actions; over 1.3 million people are financially susceptible because nitrate in raw source water exceeds the MCL, requiring actions by drinking water systems. Nitrate contamination of drinking water sources will continue to increase as nitrogen from fertilizer, manure, and other sources applied in the last half century continues to percolate downward and flow toward drinking water wells.

Findings: Sources of Nitrate Pollution

Within the study area, human-generated nitrate sources to groundwater include (Figure ES-1):

- cropland (96% of total), where nitrogen applied to crops, but not removed by harvest, air emission, or runoff, is leached from the root zone to groundwater. Nitrogen intentionally or incidentally applied to cropland includes synthetic fertilizer (54%), animal manure (33%), irrigation source water (8%), atmospheric deposition (3%), and wastewater treatment and food processing facility effluent and associated solids (2%) (Figure ES-2);
- percolation of wastewater treatment plant (WWTP) and food processing (FP) wastes (1.5% of total);
- leachate from septic system drainfields (1% of total);
- urban parks, lawns, golf courses, and leaky sewer systems (less than 1% of total); and
- recharge from animal corrals and manure storage lagoons (less than 1% of total);
- downward migration of nitrate-contaminated water via wells (less than 1% of total).

Findings: Reducing Nitrate Pollution

Options for reducing nitrate pollution were identified for all sources. For cropland, where less than 40% of applied nitrogen is removed by crop harvest, 10 management measures (and 50 practices and technologies to achieve these management objectives) were reviewed that can reduce—but not eliminate—nitrate leaching to groundwater. These fall into four categories:

1. Design and operate irrigation and drainage systems to reduce deep percolation.
2. Manage crop plants to capture more nitrogen and decrease deep percolation.
3. Manage nitrogen fertilizer and manure to increase crop nitrogen use efficiency.
4. Improve storage and handling of fertilizers and manure to decrease off-target discharge.

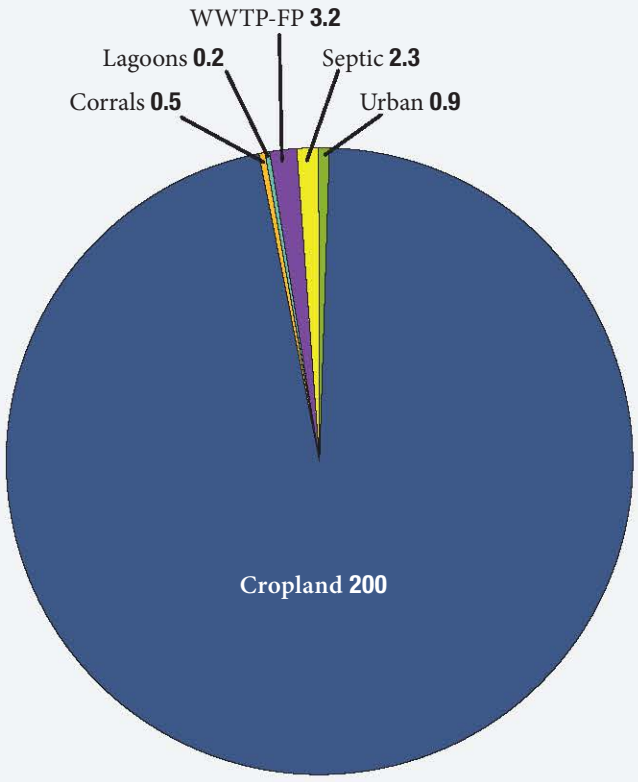
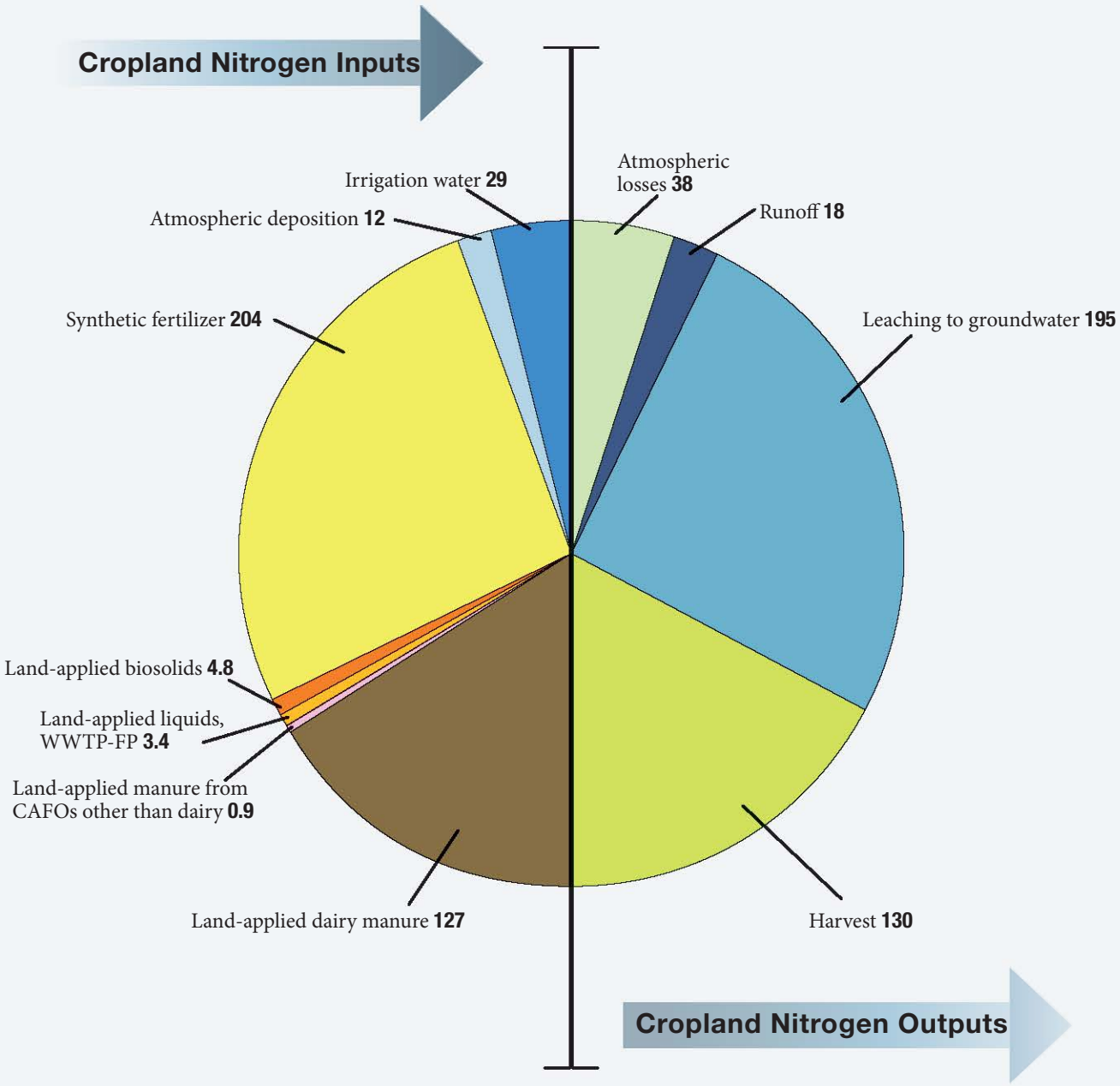


Figure ES-1. Estimated groundwater nitrate loading from major sources within the Tulare Lake Basin and Salinas Valley, in Gg nitrogen per year (1 Gg = 1,100 t).

Adminsitrative Record
Page 34165



Note: No mass balance was performed on 0.17 million ha (0.4 million ac) of nitrogen-fixing alfalfa, which is estimated to contribute an additional 5 Gg N/yr to groundwater. Groundwater nitrate loading from all non-cropland sources is about 8 Gg N/yr.

Figure ES-2. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. The left half of the pie chart represents total nitrogen inputs to 1.27 million ha (3.12 million ac) of cropland, not including alfalfa. The right half of the pie chart represents total nitrogen outputs with leaching to groundwater estimated by difference between the known inputs and the known outputs. Source: Viers et al. 2012.

Adminsitrative Record
Page 34166

Some of the needed improvements in nitrogen use efficiency by crops will require increased operating costs, capital improvements, and education. For some cropland, the high economic costs of nitrate source reduction sufficient to prevent groundwater degradation will likely hinder strict compliance with the state's current anti-degradation policy for groundwater (State Water Board Resolution 68-16).

Findings: Groundwater Nitrate Pollution

Groundwater nitrate data were assembled from nearly two dozen agencies and other sources (100,000 samples from nearly 20,000 wells). Of the 20,000 wells, 2,500 are frequently sampled public water supply wells (over 60,000 samples). In these public supply wells, about 1 in 10 raw water samples exceed the nitrate MCL. Apart from the recently established Central Valley dairy regulatory program in the Tulare Lake Basin, there are no existing regular well sampling programs for domestic and other private wells.

The largest percentages of groundwater nitrate MCL exceedances are in the eastern Tulare Lake Basin and in the northern, eastern, and central Salinas Valley, where about one-third of tested domestic and irrigation wells exceed the MCL. These same areas have seen a significant increase in nitrate concentrations over the past half century, although local conditions and short-term trends vary widely.

Travel times of nitrate from source to wells range from a few years to decades in domestic wells, and from years to many decades and even centuries in deeper production wells. This means that nitrate source reduction actions made today may not affect sources of drinking water for years to many decades.

Findings: Groundwater Remediation

Groundwater remediation is the cleanup of contaminated groundwater to within regulatory limits. Traditional pump-and-treat and in-place approaches to remediation, common for localized industrial contamination plumes, would cost billions of dollars over many decades to remove nitrate from groundwater in the Tulare Lake Basin and Salinas Valley. Timely cleanup of basin-scale nitrate contamination is not technically feasible.

Instead, long-term remediation by "pump-and-fertilize" would use existing agricultural wells to gradually remove nitrate-contaminated groundwater and treat the water by ensuring nitrate uptake by crops through appropriate nutrient and irrigation water management. Improved groundwater recharge management would provide clean groundwater recharge to mix with irrigation water recharge and partially mitigate nitrate levels in groundwater regionally.

Removal or reduction of contamination sources must accompany any successful remediation effort. Combining "pump-and-fertilize" with improved groundwater recharge management is more technically feasible and cost-effective.

Findings: Safe Drinking Water Supply

Nitrate contamination is widespread and increasing. Groundwater data show that 57% of the current population in the study area use a community public water system with recorded raw (untreated) nitrate concentrations that have exceeded the MCL at least once between 2006 and 2010. Continued basin-wide trends in nitrate groundwater concentration may raise the affected population to nearly 80% by 2050. Most of this population is protected by water system treatment, or alternative wells, at additional cost. But about 10% of the current population is at risk of nitrate contamination in their delivered drinking water, primarily in small systems and self-supplied households.

No single solution will fit every community affected by nitrate in groundwater. Each affected water system requires individual engineering and financial analyses.

Communities served by small systems vulnerable to nitrate contamination can (a) consolidate with a larger system that can provide safe drinking water to more customers; (b) consolidate with nearby small systems into a new single larger system that has a larger ratepayer base and economies of scale; (c) treat the contaminated water source; (d) switch to surface water; (e) use interim bottled water or point-of-use treatment until an approved long-term solution can be implemented; (f) drill a new well; or (g) blend contaminated wells with cleaner sources, at least temporarily.

There is significant engineering and economic potential for consolidating some systems. Consolidation can often permanently address nitrate problems, as well as many other problems faced by small water systems.

Solutions for self-supplied households (domestic well) or local small water systems (2–4 connections) affected by nitrate contamination are point-of-use (POU) or point-of-entry (POE) treatment and drilling a new or deeper well, albeit with no guarantee for safe drinking water.

Additional costs for safe drinking water solutions to nitrate contamination in the Tulare Lake Basin and Salinas Valley are roughly $20 and $36 million per year for the short- and long-term solutions, respectively. About $17 to $34 million per year will be needed to provide safe drinking water for 85 identified community public and state small water systems in the study area that exceed the nitrate drinking water MCL (serving an estimated 220,000 people). The annualized cost of providing nitrate-compliant drinking water

Adminsitrative Record
Page 34167

to an estimated 10,000 affected rural households (34,000 people) using private domestic wells or local small water systems is estimated to be at least $2.5 million for point-of-use treatment for drinking use only. The total cost for alternative solutions translates to $80 to $142 per affected person per year, $5 to $9 per irrigated acre per year, or $100 to $180 per ton of fertilizer nitrogen applied in these groundwater basins.

Findings: Regulatory, Funding, and Policy Options

To date, regulatory actions have been insufficient to control nitrate contamination of groundwater. Many options exist to regulate nitrate loading to groundwater, with no ideal solution. Nitrate source reductions will improve drinking water quality only after years to decades. Fertilizer regulations have lower monitoring and enforcement costs and information requirements than do nitrate leachate regulations, but they achieve nitrate reduction targets less directly. Costs to farmers can be lower with fertilizer fees or market-based regulations than with technology mandates or prescriptive standards. Market-based approaches may also encourage the development and adoption of new technologies to reduce fertilizer use.

Current funding programs cannot ensure safe drinking water in the Salinas Valley and Tulare Lake Basin. Small water system costs are high, and some of these systems already face chronic financial problems. Most current state funding for nitrate contamination problems is short term. Little funding is provided for regionalization and consolidation of drinking water systems. Policy options exist for long-term funding of safe drinking water, but all existing and potential options will require someone to bear the costs.

Promising Actions

Addressing groundwater nitrate contamination requires actions in four areas: (a) safe drinking water actions for affected areas, (b) reducing sources of nitrate contamination to groundwater, (c) monitoring and assessment of groundwater and drinking water, and (d) revenues to help fund solutions. Promising actions for legislative and state agency consideration in these areas appear below (see also Table ES-1). Starred (*) actions do not appear to require legislative action, but might benefit from it.

Safe Drinking Water Actions (D)

Safe drinking water actions are the most effective and economical short- and long-term approach to address nitrate contamination problems in the Tulare Lake Basin and Salinas Valley. These actions apply especially to small and self-supplied household water systems, which face the greatest financial and public health problems from nitrate groundwater contamination.

D1: Point-of-Use (POU) Treatment Option. CDPH reports on how to make economical household and point-of-use treatment for nitrate contamination an available and permanent solution for small water systems.*

D2: Small Water System Task Force. CalEPA and CDPH convene an independently led Task Force on Small Water Systems that would report on problems and solutions of small water and wastewater systems statewide as well as the efficacy of various state, county, and federal programs to aid small water and wastewater systems. Many nitrate contamination problems are symptomatic of the broad problems of small water and wastewater systems.*

D3: Regional Consolidation. CDPH and counties provide more legal, technical, and funding support for preparing consolidation of small water systems with nearby larger systems and creating new, regional safe drinking water solutions for groups of small water systems, where cost-effective.*

D4: Domestic Well Testing. In areas identified as being at risk for nitrate contamination by the California Water Boards, as a public health requirement, CDPH (a) mandates periodic nitrate testing for private domestic wells and local and state small systems and (b) requires disclosure of recent well tests for nitrate contamination on sales of residential property. County health departments also might impose such requirements.

D5: Stable Small System Funds. CDPH receives more stable funding to help support capital and operation and maintenance costs for new, cost-effective and sustainable safe drinking water solutions, particularly for disadvantaged communities (DACs).

Source Reduction Actions (S)

Reducing nitrate loading to groundwater is possible, sometimes at a modest expense. But nitrate source reduction works slowly and cannot effectively restore all affected aquifers to drinking water quality. Within the framework of Porter-Cologne, unless groundwater were to be de-designated as a drinking water source, reduction of nitrate loading to groundwater is required to improve long-term water quality. The following options seem most promising to reduce nitrate loading.

S1: Education and Research. California Department of Food and Agriculture (CDFA), in cooperation with the University of California and other organizations, develops and delivers a comprehensive educational and technical program to help farmers improve efficiency in nitrogen use (including manure) and reduce nitrate loading to groundwater. This

Adminsitrative Record
Page 34168

could include a groundwater nitrate–focused element for the existing CDFA Fertilizer Research and Education Program, including "pump-and-fertilize" remediation and improved recharge options for groundwater cleanup.*

S2: Nitrogen Mass Accounting Task Force. CalEPA establishes a Task Force, including CDFA, to explore nitrogen mass balance accounting methods for regulating agricultural land uses in areas at risk for nitrate contamination, and to compare three long-term nitrogen source control approaches: (a) a cap and trade system; (b) farm-level nutrient management plans, standards, and penalties; and (c) nitrogen fertilizer fees.*

S3: Fertilizer Excise Fee. Significantly raising the cost of commercial fertilizer through a fee or excise tax would fund safe drinking water actions and monitoring and give further incentive to farmers for reducing nitrate contamination. An equivalent fee or excise tax could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

S4: Higher Fertilizer Fee in Areas at Risk. Areas declared to be at risk for nitrate contamination might be authorized to maintain a higher set of excise fees on nitrogen fertilizer applications (including synthetic fertilizer, manure, waste effluent, biosolids, and organic amendments), perhaps as part of a local safe drinking water compensation agreement.

Monitoring and Assessment (M)

Monitoring and assessment is needed to better assess the evolving nitrate pollution problem and the effectiveness of safe drinking water and nitrate source loading reduction actions. Such activities should be integrated with other state agricultural, environmental, and land use management; groundwater data; and assessment programs (source loading reduction actions)—along with other drinking water, treatment, and wastewater management programs (safe drinking water actions).

M1: Define Areas at Risk. Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.*

M2: Monitor at-Risk Population. CDPH and the State Water Board, in coordination with DWR and CDFA, issue a report every 5 years to identify populations at risk of contaminated drinking water and to monitor long-term trends of the state's success in providing safe drinking water as a supplement to the California Water Plan Update.*

M3: Learn from Department of Pesticide Regulation Programs. CalEPA and CDFA examine successful DPR data collection, analysis, education, and enforcement programs for lessons in managing nitrogen and other agricultural contaminants, and consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.*

M4: Groundwater Data Task Force. CalEPA , in coordination with CalNRA and CDPH, convenes an independently led State Groundwater Data Task Force to examine the efficacy of current state and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.

M5: Groundwater Task Force. CalEPA, CalNRA, and CDPH maintain a joint, permanent, and independently led State Groundwater Task Force to periodically assess and coordinate state technical and regulatory groundwater programs in terms of effectiveness at addressing California's groundwater quality and quantity problems. These reports would be incorporated into each California Water Plan Update.*

Funding (F)

Little effective action can occur without funding. Four funding options seem most promising, individually or in combination. State funding from fees on nitrogen or water use, which directly affect nitrate groundwater contamination, seem particularly promising and appropriate.

F1: Mill Fee. Increase the mill assessment rate on nitrogen fertilizer to the full authorized amount (CAL. FAC Code Section 14611). This would raise roughly $1 million/year statewide and is authorized for fertilizer use research and education.*

F2: Local Compensation Agreements. Regional Water Boards can require and arrange for local compensation of affected drinking water users under Porter-Cologne Act Water Code Section 13304. Strengthening existing authority, the Legislature could require that a Regional Water Board finding that an area is at risk of groundwater nitrate contamination for drinking water be accompanied by a cleanup and abatement order requiring overlying, current sources of nitrate to financially support safe drinking water actions acceptable to the local County Health Department. This might take the form of a local "liability district."*

F3: Fertilizer Excise Fee. Introduce a substantial fee on nitrogen fertilizer sales or use, statewide or regionally, to fund safe drinking water actions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs.

F4: Water Use Fee. A more comprehensive statewide fee on water use could support many beneficial activities. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities.

Adminsitrative Record
Page 34169

Table ES-1. Likely performance of promising state and agency actions for nitrate groundwater contamination.

Action	Safe Drinking Water	Groundwater Degradation	Economic Cost
No Legislation Required			
Safe Drinking Water Actions			
D1: Point-of-Use Treatment Option for Small Systems +	◆◆		low
D2: Small Water Systems Task Force +	◆		low
D3: Regionalization and Consolidation of Small Systems +	◆◆		low
Source Reduction Actions			
S1: Nitrogen/Nitrate Education and Research +		◆◆◆	low–moderate
S2: Nitrogen Accounting Task Force +		◆◆	low
Monitoring and Assessment			
M1: Regional Boards Define Areas at Risk +	◆◆◆	◆◆◆	low
M2: CDPH Monitors At-Risk Population +	◆	◆	low
M3: Implement Nitrogen Use Reporting +		◆◆	low
M4: Groundwater Data Task Force +	◆	◆	low
M5: Groundwater Task Force +	◆	◆	low
Funding			
F1: Nitrogen Fertilizer Mill Fee		◆◆◆	low
F2: Local Compensation Agreements for Water +	◆◆	◆	moderate
New Legislation Required			
D4: Domestic Well Testing *	◆◆		low
D5: Stable Small System Funds	◆		moderate
Non-tax legislation could also strengthen and augment existing authority.			
Fiscal Legislation Required			
Source Reduction			
S3: Fertilizer Excise Fee	◆◆	◆	moderate
S4: Higher Fertilizer Fee in Areas at Risk	◆	◆	moderate
Funding Options			
F3: Fertilizer Excise Fee	◆◆	◆◆	moderate
F4: Water Use Fee	◆◆	◆◆	moderate

◆ Helpful

◆◆ Effective

◆◆◆ Essential

+ Legislation would strengthen.

* County health departments may have authority; CDPH requires legislation.

Adminsitrative Record
Page 34170

1 Introduction

The development of California's tremendous economy has not been without environmental costs. Since early in the twentieth century, nitrate from agricultural and urban activities has slowly infiltrated into groundwater. Nitrate has accumulated and spread and will continue to make its way into drinking water supplies. The time lag between the application of nitrogen to the landscape and its withdrawal at household and community public water supply wells, after percolating through soils and groundwater, commonly extends over decades.

This Report is an overview of groundwater contamination by nitrate in the Tulare Lake Basin and Salinas Valley. We examine the extent, causes, consequences, and costs of this contamination, as well as how it will likely develop over time. We also examine management and policy actions available for this problem, including possible nitrate source reduction, provisions for safe drinking water, monitoring and assessment, and aquifer remediation actions. The costs and institutional complexities of these options, and how they might be funded, also are addressed.

Addressing nitrate contamination problems in the Tulare Lake Basin and Salinas Valley will require decades to resolve, driven by the pace of groundwater flow and the response times of humans and institutions on the surface. Nitrate in drinking water today is a legacy contaminant, but years and decades from now the nitrate in drinking water will be from today's discharges. Assistance and management to improve drinking water supplies in response to nitrate contamination is a central and urgent policy issue for the State of California. Another major policy issue is the inevitability of widespread groundwater degradation for decades to come, despite even heroic (and ultimately expensive) efforts to reduce nitrate loading into aquifers. This introduction attempts to put the issue in a larger context.

Groundwater is essential to California. Groundwater is vital for California's agricultural, industrial, urban, and drinking water uses. Depending on drought conditions, groundwater provides between one-third and nearly one-half of the state's water supplies. As a source of drinking water, groundwater serves people from highly dispersed rural communities to densely populated cities. More than 85% of community public water systems in California (serving 30 million residents) rely on groundwater for at least part of their drinking water supply. In addition, approximately 2 million residents rely on groundwater from either a private domestic well or a smaller water system not regulated by the state (State Water Board 2011). Intensive agricultural production, population growth, and—indirectly—partial restoration of environmental instream flows have led to groundwater overdraft (Hanak et al. 2011). More protective health-based water quality standards for naturally occurring water quality constituents and groundwater contamination from urban and agricultural activities pose serious challenges to managing the state's drinking water supply.

Nitrate is one of California's most widespread groundwater contaminants. Nitrate is among the most frequently detected contaminants in groundwater systems around the world, including the extensively tapped aquifers in California's Central Valley and Salinas Valley (Figure 1) (Spalding and Exner 1993; Burow et al. 2010; Dubrovsky et al. 2010; MCWRA 2010; Sutton et al. 2011). Nitrate contamination poses an environmental health risk because many rural areas obtain drinking water from wells that are often shallow and vulnerable to contamination (Guillette and Edwards 2005; Fan and Steinberg 1996).

High levels of nitrate affect human health. Infants who drink water (often mixed with baby formula) containing nitrate in excess of the maximum contaminant level (MCL) for drinking water may quickly become seriously ill and, if untreated, may die because high nitrate levels can decrease the capacity of an infant's blood to carry oxygen (methemoglobinemia, or "blue baby syndrome"). High nitrate levels may also affect pregnant women and adults with hereditary cytochrome b5 reductase deficiency. In addition, nitrate and nitrite ingestion in humans has been linked to goitrogenic (anti-thyroid) actions on the thyroid gland (similar to perchlorate), fatigue and reduced cognitive functioning due to chronic hypoxia, maternal reproductive complications including spontaneous abortion, and a variety of carcinogenic outcomes deriving from N-nitrosamines formed via gastric nitrate conversion in the presence of amines (Ward et al. 2005).

Adminsitrative Record
Page 34171

Nitrate is part of the natural nitrogen cycle in the environment. Groundwater nitrate is part of the global nitrogen cycle. Like other key elements essential for life, nitrogen flows through the environment in a dynamic cycle that supports organisms ranging from microbes to plants to animals. Plants require nitrogen for growth, and scarcity of fixed soil nitrogen often limits plant growth. Specialized microorganisms can fix atmospheric elemental nitrogen and make it available for plants to use for photosynthesis and growth. The natural nitrogen cycle is a dynamic balance between elemental nitrogen in the atmosphere and reactive forms of nitrogen moving through the soil-plant-animal-water-atmosphere cycle of ecosystems globally. Production of synthetic nitrogen fertilizer has disrupted this balance.

Nitrogen is key to global food production. Modern agricultural practices, using synthetically produced nitrogen fertilizer, have supplied the nitrogen uses of plants to increase food, fiber, feed, and fuel production for consumption by humans and livestock. Agricultural production is driven by continued global growth in population and wealth, which increases demand for agricultural products, particularly high-value agricultural products such as those produced in California. Global food, feed, and fiber demands are anticipated to increase by over 70% over the next 40 years (Tilman et al. 2002; De Fraiture et al. 2010).

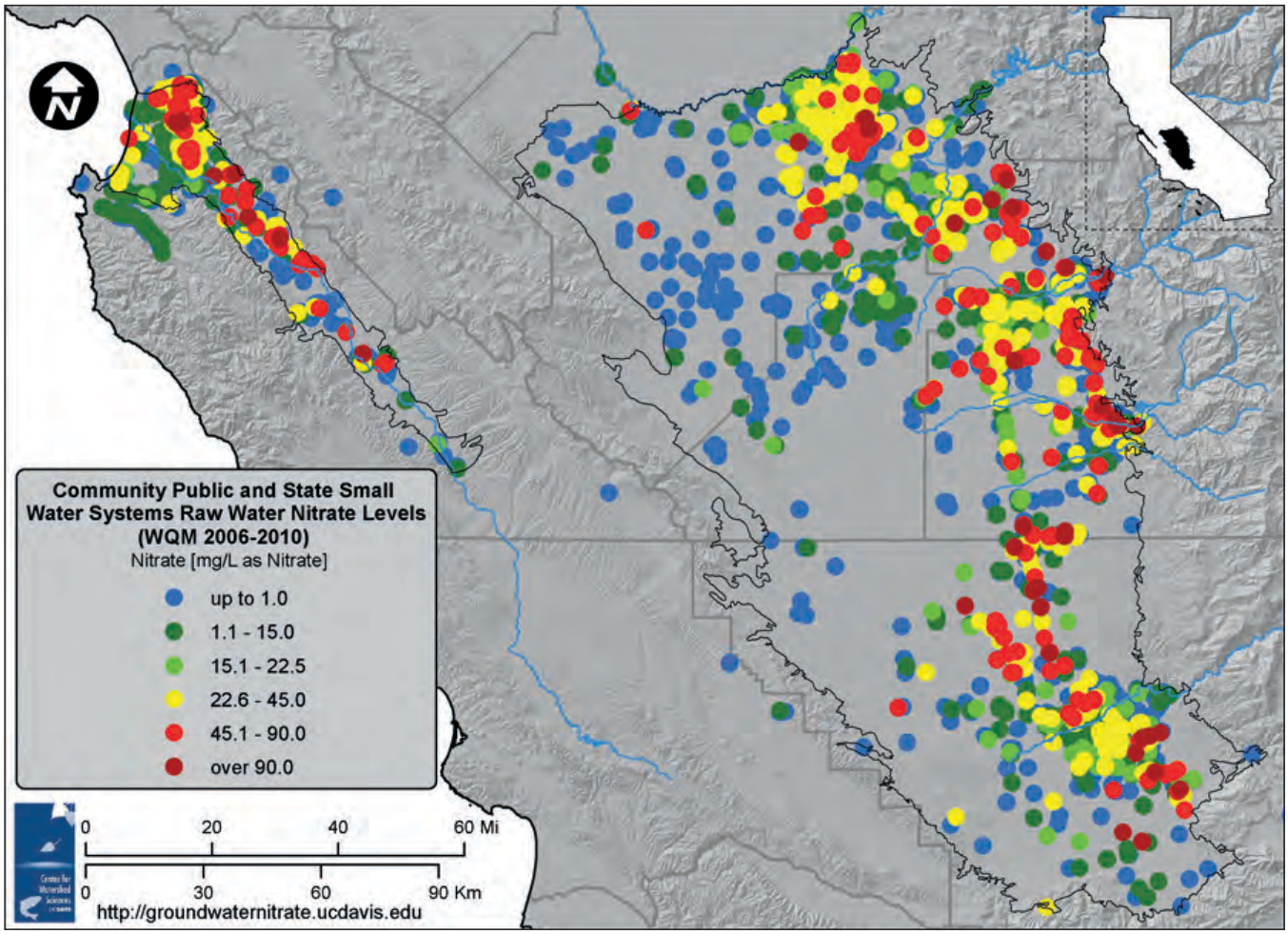


Figure 1. Maximum reported raw-level nitrate concentration in community public water systems and state-documented state small water systems, 2006–2010. Source: CDPH PICME WQM Database (see Honeycutt et al. 2011).

Adminsitrative Record
Page 34172

Intensive agriculture and human activities have increased nitrate concentrations in the environment. Greater use of nitrogen-based fertilizers, soil amendments such as manure, and nitrogen-fixing cover crops add nitrogen to deficient soils and dramatically raise crop yields. Technological advances in agriculture, manufacturing, and urban practices have increased levels of reactive forms of nitrogen, including nitrate, released into the atmosphere, into surface water, and into groundwater. The nearly 10-fold increase of reactive nitrogen creation related to human activities over the past 100 years (Galloway and Cowling 2002) has caused a wide range of adverse ecological and environmental impacts (Davidson et al. 2012).

The most remarkable impacts globally include the leaching of nitrate to groundwater; the eutrophication of surface waters and resultant marine "dead zones"; atmospheric deposition that acidifies ecosystems; and the emission of nitrogen oxides (NOx) that deplete stratospheric ozone (Keeney and Hatfield 2007; Beever et al. 2007; Foley et al. 2005). These widespread environmental changes also can threaten human health (Galloway et al. 2008; Guillette and Edwards 2005; Galloway et al. 2004; Townsend et al. 2003; Vitousek et al. 1997; Fan and Steinberg 1996; Jordan and Weller 1996).

California has decentralized regulatory responsibility for groundwater nitrate contamination. Nitrate contamination of groundwater affects two state agencies most directly. Sources of groundwater nitrate are regulated under California's Porter-Cologne Water Quality Control Act (Porter-Cologne) administered through the State Water Resources Control Board (State Water Board) and the Regional Water Quality Control Boards (Regional Water Boards). State Water Board Resolution 88-63 designates drinking water as a beneficial use in nearly all of California's major aquifers. Under the Porter-Cologne Act, dischargers to groundwater are responsible, first, for preventing adverse effects on groundwater as a source of drinking water, and second, for cleaning up groundwater when it becomes contaminated.

Drinking water in public water systems (systems with at least 15 connections or serving at least 25 people for 60 or more days per year) is regulated by CDPH under the federal Safe Drinking Water Act of 1972 (SWDA). CDPH has set the nitrate MCL in drinking water at 45 mg/L (10 mg/L as nitrate-N). If nitrate levels in public drinking water supplies exceed the MCL standard, mitigation measures must be employed by water purveyors to provide a safe supply of drinking water to the population at risk.

The California Department of Food and Agriculture (CDFA) and the Department of Water Resources (DWR) also have roles in nitrate management. The DWR is charged with statewide planning and funding efforts for water supply and water quality protection, including the funding of Integrated Regional Water Management Plans and DWR's management of urban and agricultural water use efficiency. CDFA collects data, funds research, and promotes education regarding the use of nitrogen fertilizers and other nutrients in agriculture.

SBX2 1 Nitrate in Groundwater Report to Legislature. In 2008, the California legislature enacted Senate Bill SBX2 1 (Perata), which created California Water Code Section 83002.5. The bill requires the State Water Board to prepare a Report to the Legislature (within 2 years of receiving funding) to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the state for the purposes of this section to clean up or treat groundwater, and ensure the provision of safe drinking water to all communities." Specifically, the bill directs the State Water Board to

> identify sources, by category of discharger, of groundwater contamination due to nitrate in the pilot project basins; to estimate proportionate contributions to groundwater contamination by source and category of discharger; to identify and analyze options within the board's current authority to reduce current nitrate levels and prevent continuing nitrate contamination of these basins and estimate the costs associated with exercising existing authority; to identify methods and costs associated with the treatment of nitrate contaminated groundwater for use as drinking water; to identify methods and costs to provide an alternative water supply to groundwater reliant communities in each pilot project basin; to identify all potential funding sources to provide resources for the cleanup of nitrate, groundwater treatment for nitrate, and the provision of alternative drinking water supply, including, but not limited to, State bond funding, federal funds, water rates, and fees or fines on polluters; and to develop recommendations for developing a groundwater cleanup program for the Central Valley Water Quality Control Region and the Central Coast Water Quality Control Region based upon pilot project results.

Adminsitrative Record
Page 34173

The bill designates the groundwater basins of the Tulare Lake Basin region and the Monterey County portion of the Salinas Valley as the selected pilot project areas. In June 2010, the State Water Board contracted with the University of California, Davis, to prepare this Report for the Board as background for its Report to the Legislature.

Project area is relevant to all of California. The project area encompasses all DWR Bulletin 118 designated groundwater sub-basins of the Salinas River watershed that are fully contained within Monterey County, and the Pleasant Valley, Westside, Tulare Lake Bed, Kern, Tule River, Kaweah River, and Kings River groundwater sub-basins of the Tulare Lake Basin. The study area—2.3 million ha (5.7 million ac) in size—is home to approximately 2.65 million people, almost all of whom rely on groundwater as a source of drinking water. The study area includes four of the nation's five counties with the largest agricultural production; 1.5 million ha (3.7 million ac) of irrigated cropland, representing about 40% of California's irrigated cropland; and more than half of California's dairy herd. More than 80 different crops are grown in the study area (Figure 2). This is also one of California's poorest regions: many census blocks with significant population belong to the category of severely disadvantaged communities (less than 60% of the state's median household income), and many of the remaining populated areas are disadvantaged communities (less than 80% of the state's median household income). These communities have little economic means and technical capacity to maintain safe public drinking water systems given contamination from nitrate and other contaminants in their drinking water sources.

Report excludes assessment of public health standards for nitrate. Public health and appropriateness of the drinking water limits are prescribed by CDPH and by U.S. EPA under SDWA. The scope of SBX2 1 precluded a review of the public health aspects or a review of the appropriateness of the nitrate MCL, although this is recognized as an important and complex aspect of the nitrate contamination issue (Ward et al. 2005).

"Report for the State Water Resources Control Board Report to the Legislature" and supporting Technical Reports. This Report for the State Water Board Report to the Legislature ("Report") has been provided in fulfillment of the University of California, Davis, contract with the State Water Board. This Report provides an overview of the goals of the research, methods, and key findings of our work, and is supported by eight related Technical Reports (Harter et al. 2012; Viers et al. 2012; Dzurella et al. 2012; Boyle et al. 2012; King et al. 2012; Jensen et al. 2012; Honeycutt et al. 2012; and Canada et al. 2012). The Technical Reports provide detailed information on research methods, research results, data summaries, and accompanying research analyses that are important for evaluating our results and findings and for applying our approach and results to other groundwater basins.

The Report takes a broad yet quantitative view of the groundwater nitrate problem and solutions for this area and reflects collaboration among a diverse, interdisciplinary team of experts. In its assessment, the Report spans institutional and governmental boundaries. The Report quantifies the diverse range of sources of groundwater nitrate. It reviews the current groundwater quality status in the project area by compiling and analyzing all available data from a variety of institutions. It then identifies source reduction, groundwater remediation, drinking water treatment, and alternative drinking water supply alternatives, along with the costs of these options. Descriptions and summaries are also included of current and potential future funding options and regulatory measures to control source loading and provide safe drinking water, along with their advantages, disadvantages, and potential effectiveness.

This set of Reports is the latest in a series of reports on nitrate contamination in groundwater beginning in the 1970s (Schmidt 1972; Report to Legislature 1988; Dubrovsky et al. 2010; U.S. EPA 2011). This Report has some of the same conclusions as previous reports but takes a much broader perspective, contains more analysis, and perhaps provides a wider range of promising actions.

Adminsitrative Record
Page 34174

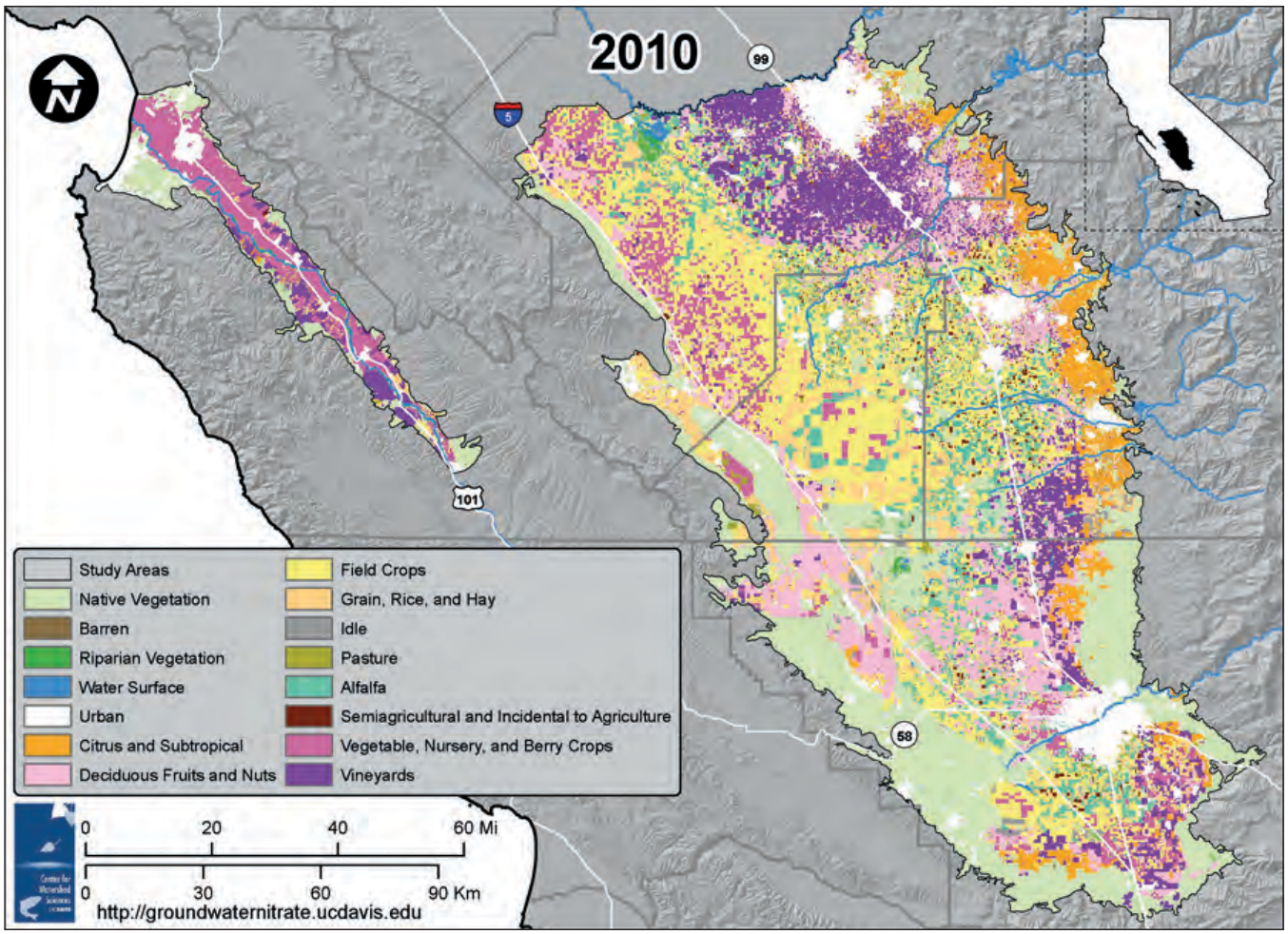
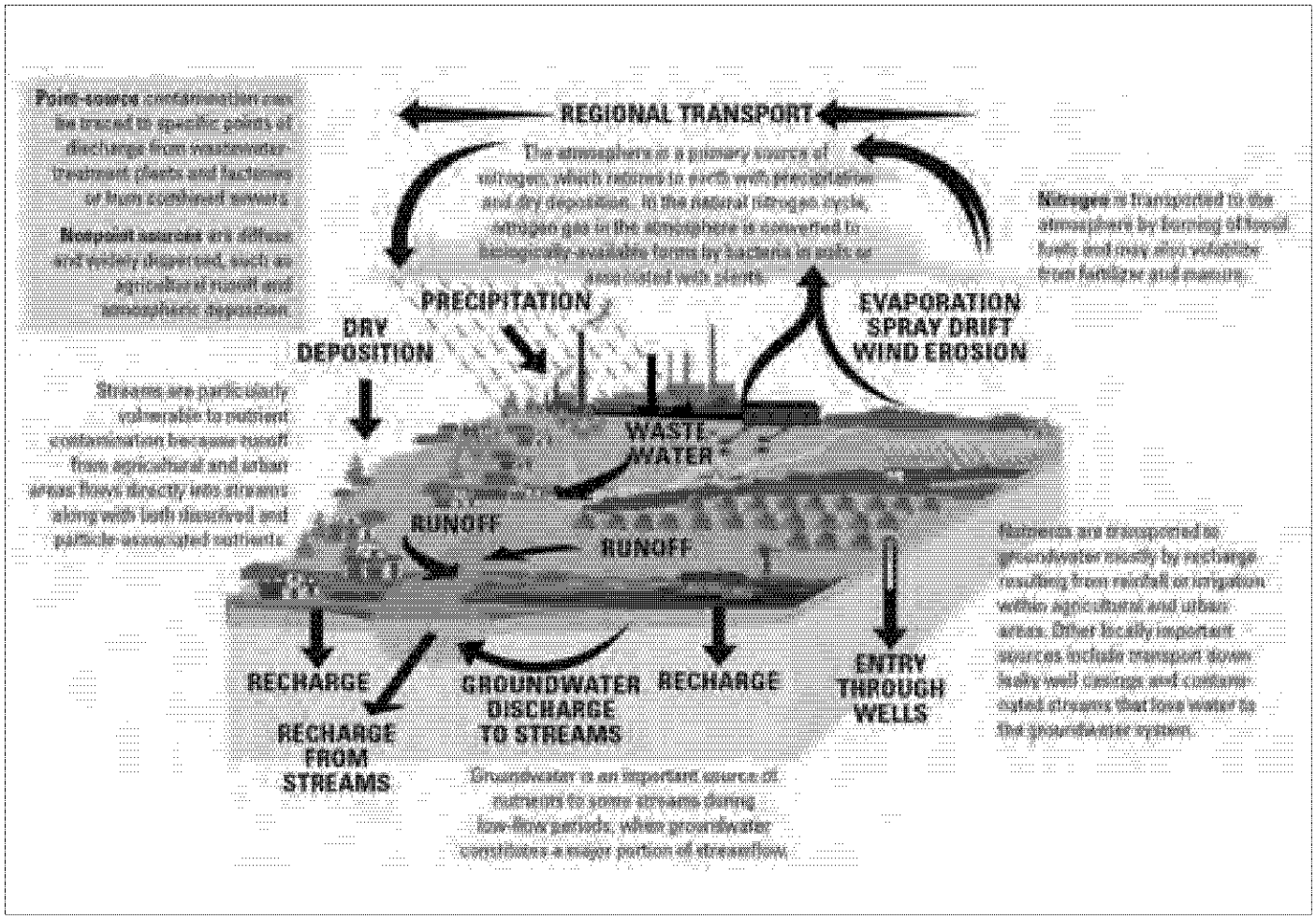


Figure 2. The Tulare Lake Basin (TLB) and Salinas Valley (SV) are the focus of this study. The study area represents 40% of California's diverse irrigated agriculture and more than half of its confined animal farming industry. It is home to 2.6 million people, with a significant rural population in economically disadvantaged communities. Source: Viers et al. 2012.

Adminsitrative Record
Page 34175



Source: Dubrovsky et al. 2010.

Adminsitrative Record
Page 34176

2 Sources of Groundwater Nitrate

2.1 Nitrogen Cycle: Basic Concepts

Nitrogen is an essential element for all living organisms. Nitrogen cycles through the atmosphere, hydrosphere, and biosphere. The dominant gas (78%) in the atmosphere is highly stable (inert) N_2 gas. Biological nitrogen fixation transforms N_2 gas into ammonia (NH_3), which is rapidly converted to the forms of nitrogen needed for plant growth. Nitrogen fixation is performed only by specialized soil and aquatic microbes. Other living organisms cannot use inert atmospheric N_2 directly but rely on accumulated soil organic matter, plants, animals, and microbial communities for nitrogen.

Soil nitrogen is most abundant in the organic form (N_{org}). Mineralization is a suite of processes performed by soil microbes that converts organic nitrogen to inorganic forms of nitrogen. The rates of mineralization depend on the environmental conditions such as temperature, moisture, pH, and oxygen content, as well as the type of organic matter available. The first product of mineralization is ammonium (NH_4^+), but under aerobic conditions, microbes can convert ammonium (NH_4^+) first to nitrite (NO_2^-) and then to nitrate (NO_3^-). Most plants use nitrate or ammonium as their preferred source of nitrogen (White 2006). Immobilization is the reverse of mineralization in that soil ammonium and nitrate are taken up by soil organisms and plants and converted into N_{org}.

The ultimate fate of "reactive" nitrogen (organic nitrogen, ammonium, nitrate, ammonia, nitrous oxide, etc.) is to return back to the atmosphere as N_2. For nitrate, this is a microbially mediated process ("denitrification") that requires an anoxic (i.e., oxygen-free) environment.

Groundwater is becoming a growing component of the global nitrogen cycle because of the increased nitrogen inflows and because of long groundwater residence times. Nitrate does not significantly adhere to or react with sediments or other geologic materials, and it moves with groundwater flow. Other forms of reactive nitrogen in groundwater are less significant and much less mobile: ammonia occurs under some groundwater conditions, but it is subject to sorption and rapidly converts to nitrate under oxidizing conditions. Dissolved organic nitrogen (DON) concentrations are generally much less than those of nitrate, except near wastewater sources, due to the high adsorption of DON to aquifer materials.

Groundwater nitrate inputs may come from natural, urban, industrial, and agricultural sources. Groundwater nitrate outputs occur through wells or via discharge to springs, streams, and wetlands. Discharge to surface water sometimes involves denitrification or reduction of nitrate to ammonium when oxygen-depleted conditions exist beneath wetlands and in the soils immediately below streams.

2.2 Sources of Nitrate Discharge to Groundwater

Nitrogen enters groundwater at varying concentrations and in varying forms (organic nitrogen, ammonium, nitrate) with practically all sources of recharge: diffuse recharge from precipitation and irrigation; focused recharge from streams, rivers, and lakes; focused recharge from recharge basins and storage lagoons; and focused recharge from septic system drainfields. Across major groundwater basins in California, diffuse recharge from irrigation, stream recharge, and intentional recharge are the major contributors to groundwater. Since groundwater is an important reservoir for long-term water storage, recharge is extremely important and desirable in many areas. Controlling nitrate in recharge and managing recharge are therefore key to nitrate source control.

Current groundwater nitrate, its spatial distribution, and its changes over time are the result of recent as well as historical nitrate loading. To understand current and future groundwater conditions requires knowledge of historical, current, and anticipated changes in land use patterns, recharge rates, and nitrate loading rates (Viers et al. 2012).

Natural Nitrate Sources

Nitrate occurs naturally in many groundwaters but at levels far below the MCL for drinking water (Mueller and Helsel 1996). The main potential sources of naturally occurring nitrate are bedrock nitrogen and nitrogen leached from natural soils. Surface water nitrate concentrations can be elevated in areas with significant bedrock nitrogen (Holloway et al. 1998), but they are not high enough to be a drinking water concern. During the early twentieth century, conversion of the study area's semiarid and arid natural landscape to irrigated agriculture may have mobilized two additional, naturally occurring sources of nitrate. First, nitrate was released from drained

Adminsitrative Record
Page 34177

wetlands at the time of land conversion due to increased microbial activity in agricultural soils; stable organic forms of nitrogen that had accumulated in soils over millennia were converted to mobile nitrate. Second, nitrate salts that had accumulated over thousands of years in the unsaturated zone below the grassland and desert soil root zone due to lack of significant natural recharge were mobilized by irrigation (Dyer 1965; Stadler et al. 2008; Walvoord et al. 2003). However, the magnitude of these sources (Scanlon 2008) is considered to have negligible effects on regional groundwater nitrate given the magnitude of human sources.

Human Nitrate Sources

Anthropogenic groundwater nitrate sources in the study area include agricultural cropland, animal corrals, animal manure storage lagoons, wastewater percolation basins at municipal wastewater treatment plants (WWTPs) and food processors (FPs), septic system drainfields (onsite sewage systems), leaky urban sewer lines, lawns, parks, golf courses, and dry wells or percolation basins that collect and recharge stormwater runoff. Incidental leakage of nitrate may also occur directly via poorly constructed wells. Croplands receive nitrogen from multiple inputs: synthetic fertilizer, animal manure, WWTP and FP effluent, WWTP biosolids, atmospheric deposition, and nitrate in irrigation water sources.

Source categories. For this Report, we estimated the groundwater nitrate contributions for 58 individual agricultural cropland categories, for animal corrals, for manure lagoons, for each individual WWTP and FP within the study area, for dairies and other animal farming operations, for septic system drainfields, and for urban sources. Contributions from dry wells and incidental leakage through existing wells were estimated at the basin scale. Groundwater nitrate contributions were estimated for five time periods, each consisting of 5 years: 1943–1947 ("1945"), 1958–1962 ("1960"), 1973–1977 ("1975"), 1988–1992 ("1990"), and 2003–2007 ("2005"); the latter is considered to be current. Future year 2050 loading was estimated based on anticipated land use changes (primarily urbanization). These categorical or individual estimates of nitrate leaching lead to maps that show nitrate discharge at a resolution of 0.25 ha (less than 1 ac) for the entire study area and its changes over a period of 105 years (1945–2050) (Viers et al. 2012; Boyle et al. 2012).

Separately, we also aggregated nitrate loads to groundwater

- by crop categories (e.g., olives, persimmons, lettuce, strawberries) and crop groups (e.g., "subtropicals," "vegetables and berries") averaged or summed over the entire study area;
- by county, totaled across all cropland, all WWTPs and FPs, all dairies, all septic drains, and all municipal areas; and
- summed or averaged for the study area.

Higher levels of aggregation provide more accurate estimates but are less descriptive of actual conditions at any given location. Aggregated totals are most useful for policy and planning.

We report nitrate loading to groundwater in two ways:

- Total annual nitrate leached to groundwater, measured in gigagrams of nitrate-nitrogen per year (Gg N/yr).[1] As a practical measure, 1 gigagram is roughly equivalent to $1 million of nitrogen fertilizer at 2011 prices.
- Intensity of the nitrate leaching to groundwater, measured in kilograms of nitrate-nitrogen per ha of use per year (kg N/ha/yr) [lb per acre per year, lb/ac/yr], which represents the intensity of the source at its location (field, pond, corral, census block, city) and its potential for local groundwater pollution.

How much nitrate loading to groundwater is acceptable? To provide a broad reference point of what the source loading numbers mean with respect to potential groundwater pollution, it is useful to introduce an operational benchmark that indicates whether nitrate leached in recharge to groundwater exceeds the nitrate drinking water standard. This operational benchmark considers that nearly all relevant anthropogenic nitrate sources provide significant groundwater recharge and therefore remain essentially undiluted when

[1] One gigagram is equal to 1 million kilograms (kg), 1,000 metric tons, 2.2 million pounds (lb), or 1,100 tons (t). In this report, nitrogen application to land refers to total nitrogen (organic nitrogen, ammonium-nitrogen, and nitrate-nitrogen). For consistency and comparison, total nitrate loading and the intensity of nitrate loading from the root zone to groundwater are also provided in units of nitrogen, not as nitrate. However, concentrations of nitrate in groundwater or leachate are always stated as nitrate (MCL: 45 mg/L) unless noted otherwise.

Adminsitrative Record
Page 34178

reaching groundwater. Our benchmark for "low" intensity versus "high" intensity of nitrate leaching is 35 kg N/ha/yr (31 lb N/ac/yr).[2] Aggregated across the 1.5 million ha (3.7 million ac) of cropland, the benchmark for total annual nitrate loading in the study area is 50 Gg N/yr (55,000 t N/yr). Total nitrate loading to groundwater above this benchmark indicates a high potential for regional groundwater degradation.

Estimating nitrate loading by source category. We used two methods to assess nitrate loading:

- a mass balance approach was used to estimate nitrate loading from all categories of cropland except alfalfa;
- alfalfa cropland and nitrate sources other than cropland were assessed by reviewing permit records, literature sources, and by conducting surveys to estimate groundwater nitrate loading (Viers et al. 2012).

Groundwater Nitrate Contributions by Source Category

Cropland is by far the largest nitrate source, contributing an estimated 96% of all nitrate leached to groundwater (Table 1). The total nitrate leached to groundwater (200 Gg N/yr [220,000 t N/yr]) is four times the benchmark amount, which suggests large and widespread degradation of groundwater quality. Wastewater treatment plants and food processor waste percolation basins are also substantial, high-intensity sources.[3] Septic systems, manure storage lagoons, and corrals are relatively small sources basin-wide, but since their discharge intensity significantly exceeds the operational benchmark of 35 kg N/ha/yr (31 lb N/ac/yr), these source categories can be locally important. The magnitude and intensity of urban sources (other than septic systems) does not suggest widespread impact to groundwater (Viers et al. 2012). The following sections provide further detail on these sources.

Agricultural Sources

Cropland sources: Overview. The five counties in the study area include 1.5 million ha (3.7 million ac) of cropland, about 40% of California's irrigated cropland. Agricultural production involves many crops and significant year-to-year changes in crops grown and crop yields. The dominant crop groups in the project area include subtropical crops (citrus and olives), tree fruits and nuts, field crops including corn and cotton, grain crops, alfalfa, vegetables and strawberries, and grapes (see Figure 2). The study area also supports 1 million dairy cows. These produce one-tenth of the nation's milk supply as well as large amounts of manure.

Cropland sources: Alfalfa. The mass balance approach is not applied to alfalfa because it does not receive significant amounts of fertilizer, yet alfalfa fixes large amounts of nitrogen from the atmosphere. Little is known about nitrate leaching from alfalfa; we used a reported value of 30 kg N/ha/yr (27 lb N/ac/yr) (Viers et al. 2012). In total, 170,000 ha (420,000 ac) of alfalfa fields are estimated to contribute about 5 Gg N/yr (5,500 t N/yr) in the study area. Alfalfa harvest exceeds 400 kg N/ha/yr (360 lb N/ac/yr), or 74 Gg N/yr (82,000 t N/yr), in the study area.

Cropland sources other than alfalfa. Unlike other groundwater nitrate source categories, cropland has many sources of nitrogen application, all of which can contribute to nitrate leaching. Principally, crops are managed for optimal harvest. Synthetic nitrogen is the fertilizer of choice to achieve this goal, except in alfalfa. Other sources of nitrogen are also applied to cropland, providing additional fertilizer, serving as soil amendments, or providing a means of waste disposal. These additional nitrogen sources include animal manure and effluent and biosolids from WWTPs, FPs, and other urban sources. Often do they replace synthetic fertilizer as the main source of nitrogen for a crop. Atmospheric deposition of nitrogen and nitrate in irrigation water are mostly incidental but ubiquitous.

For the mass balance analysis, external nitrogen inputs to cropland are considered to be balanced over the long run (5 years and more) by nitrogen leaving the field in crop harvest, atmospheric losses (volatilization, denitrification), runoff to streams, or groundwater leaching. Hence, cropland nitrate leaching to groundwater is estimated by summing nitrogen inputs to a field (fertilizer, effluent, biosolids,

[2] A typical groundwater recharge rate in the study area is roughly 300 mm/yr (1 AF/ac/yr). If that recharge contains nitrate at the MCL, the annual nitrate loading rate is 30 kg N/ha/yr (27 lb N/ac/yr). We allow an additional 5 kg N/ha/yr (4.5 lb N/ac/yr) to account for potential denitrification in the deep vadose zone or in shallow groundwater.

[3] The benchmark of 35 kg N/ha (31 lb N/ac) is not adequate for percolation basins, as their recharge rate is much more than 1 AF/ac. Instead, we consider actual average concentration (by county) of nitrogen in FP and WWTP discharges to percolation basins, which range from 2 to 10 times the MCL and 1 to 2 times the MCL, respectively (Viers et al. 2012).

Adminsitrative Record
Page 34179

manure, atmospheric deposition, irrigation water) and then subtracting the three other nitrogen outputs (harvest, atmospheric losses, and runoff).

In total, the 1.27 million ha (3.1 million ac) of cropland, not including 0.17 million ha (0.4 million ac) of alfalfa, receive 380 Gg N/yr (419,000 t N/yr) from all sources. Synthetic fertilizer, at 204 Gg N/yr (225,000 t N/yr), is more than half of these inputs (Figure 3). Manure applied on dairy forages or exported for cropland applications off-dairy (but not leaving the study area) is one-third of all nitrogen inputs. Atmospheric deposition and nitrate-nitrogen in groundwater used as irrigation water are approximately one-tenth of all nitrogen input. Urban effluent and biosolids application are small portions of the overall nitrogen input in the study area, but they are locally significant.

Table 1. Major sources of groundwater nitrate, their estimated total contribution in the study area, their percent of total contribution, and their estimated average local intensity, which indicates local pollution potential (actual total nitrate loading from these source categories is very likely within the range provided in parentheses)

	Total Nitrate Loading to Groundwater Gg N/yr* (range) [1,000 t N/yr (range)]	Percent Contribution to Total Nitrate Leaching in the Study Area	Average Intensity of Nitrate Loading to Groundwater kg N/ha/yr [lb N/ac/yr]
Cropland	195 (135–255) [215 (150–280)]	93.7%	154 [137]
Alfalfa cropland	5 (<1–10) [5 (<1–10)]	2.4%	30 [27]
Animal corrals	1.5 (0.5–8) [1.7 (0.5–9)]	0.7%	183 [163]
Manure storage lagoons	0.23 (0.2–2) [0.25 (0.2–2)]	0.1%	183 [163]
WWTP and FP† percolation basins	3.2 (2–4) [3.5 (2–4)]	1.5%	1,200‡ [1,070]
Septic systems	2.3 (1–4) [2.5 (1–4)]	1.1%	<10 – >50 [<8.8 – >45]
Urban (leaky sewers, lawns, parks, golf courses)	0.88 (0.1–2) [0.97 (0.1–2)]	0.5%	10 [8.8]
Surface leakage to wells	<0.4 [<0.4]	—	§

Source: Viers et al. 2012.

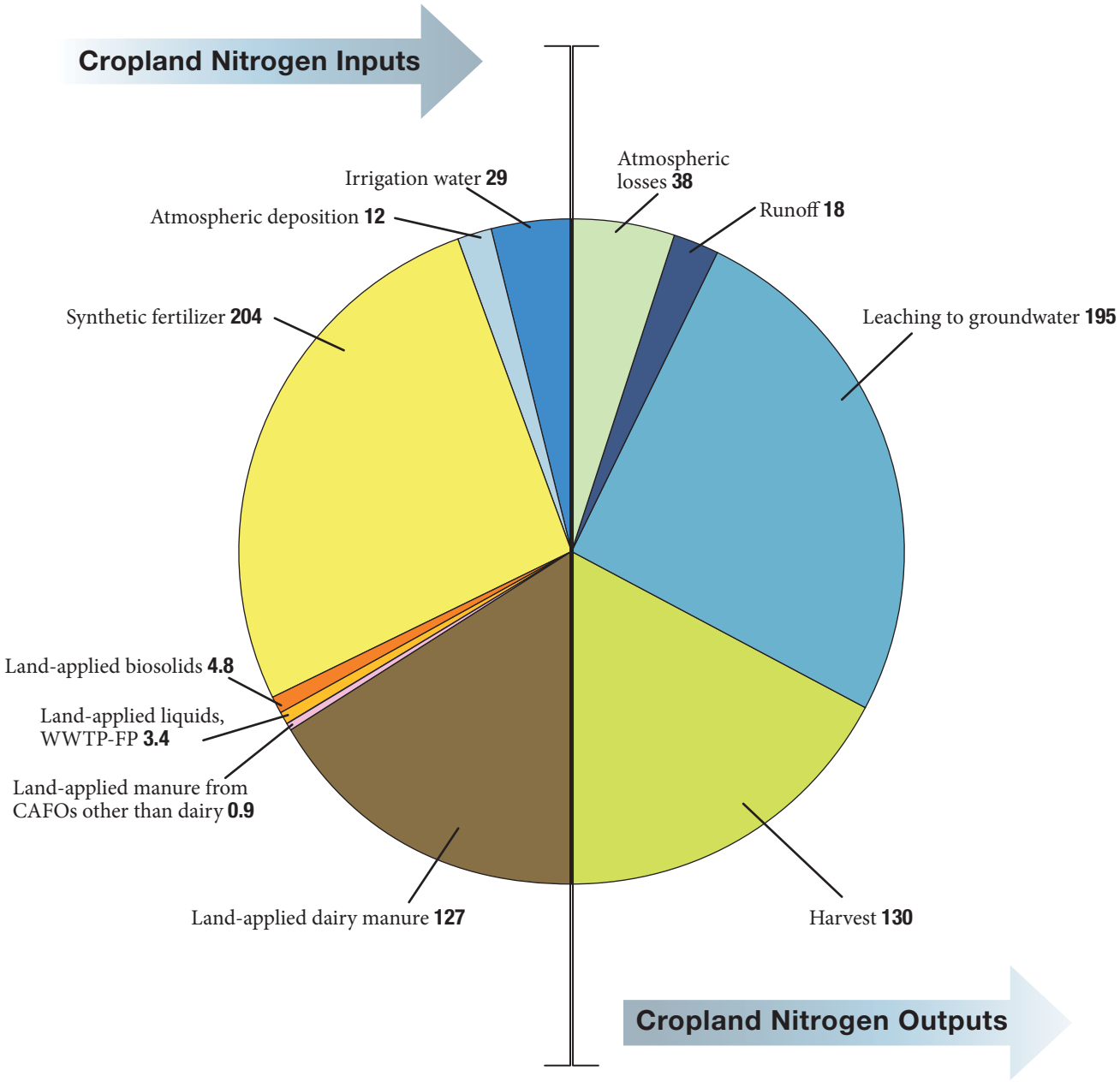
*At 2011 prices, 1 Gg N (1,100 t N) is roughly equivalent to $1 million in fertilizer nitrogen.

†WWTP = wastewater treatment plant; FP = food processor.

‡The benchmark of 35 kg N/ha/yr does not apply to WWTP and FP percolation basins, which may recharge significantly more water than other sources. Their nitrate loading may be high even if nitrate concentrations are below the MCL (Viers et al. 2012).

§Surface leakage through improperly constructed wells is based on hypothetical estimates and represents an upper limit.

Adminsitrative Record
Page 34180



Note: No mass balance was performed on 0.17 million ha (0.4 million ac) of nitrogen-fixing alfalfa, which is estimated to contribute an additional 5 Gg N/yr to groundwater. Groundwater nitrate loading from all non-cropland sources is about 8 Gg N/yr.

Figure 3. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. The left half of the pie chart represents total nitrogen inputs to 1.27 million ha (3.12 million ac) of cropland, not including alfalfa. The right half of the pie chart represents total nitrogen outputs with leaching to groundwater estimated by difference between the known inputs and the known outputs. Source: Viers et al. 2012.

Adminsitrative Record
Page 34181

On the output side, the total nitrate leaching to groundwater from cropland, not including alfalfa, comprises 195 Gg N/yr (215,000 t N/yr) and is by far the largest nitrogen flux from cropland, much larger than the harvested nitrogen at 130 Gg N/yr (143,000 t N/yr). The nitrogen leached to groundwater nearly matches the amount of synthetic fertilizer applied to the same cropland, suggesting large system surpluses of nitrogen use on cropland. Other outputs are small: atmospheric losses are assumed to be one-tenth of the inputs (Viers et al. 2012), and runoff is assumed to be 14 kg N/ha/yr (12.5 lb N/ac/yr) (Beaulac and Reckhow 1982).

Applying the benchmark of 50 Gg N/yr (55,000 t N/yr), groundwater leaching losses would need to be reduced by 150 Gg N/year (165,000 t N/yr) or more area-wide to avoid further large-scale groundwater degradation. Figure 3 suggests three major options to reduce nitrate loading to groundwater from cropland: develop techniques to make manure a useful and widely used fertilizer and reduce synthetic fertilizer application in the study area by as much as 75%; drastically reduce the use of manure in the study area; or significantly increase the agricultural output (harvest) without increasing the nitrogen input. Nitrate source reduction efforts will involve a combination of these options (see Section 2.3).

The following sections further discuss individual inputs and outputs that control agricultural cropland nitrate leaching.

Cropland inputs: Synthetic fertilizer (204 Gg N/yr [225,000 t N/yr]). Synthetic fertilizer application rates are estimated by first establishing a typical nitrogen application rate for each crop, derived from the literature, United States Department of Agriculture (USDA) Chemical Usage Reports,

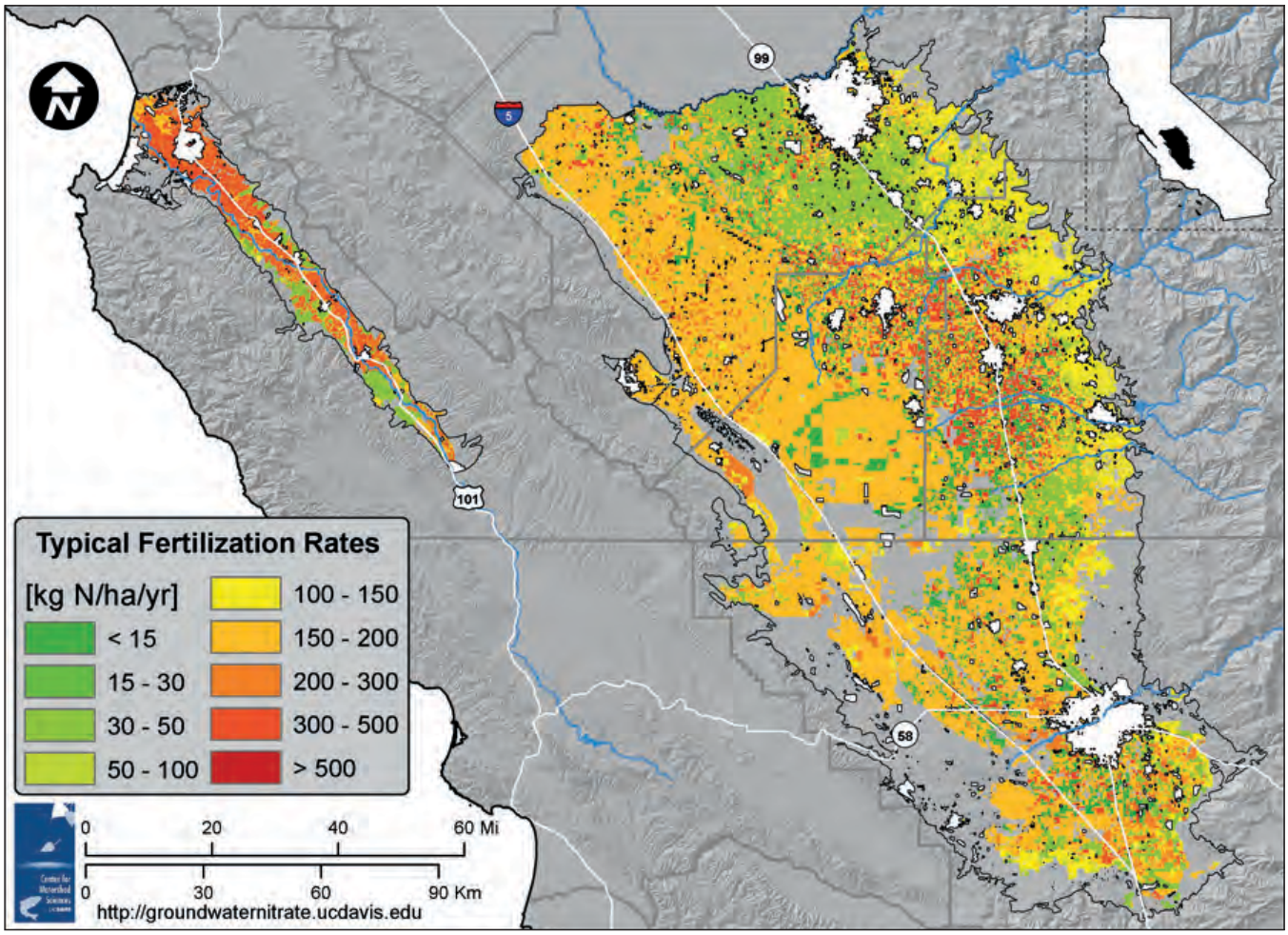


Figure 4. Current typical annual fertilization rates (1 kg/ha/yr = 1.1 lb/ac/yr) in irrigated agricultural cropland of the study area derived from the literature, USDA Chemical Usage Reports, and agricultural cost and return studies for each of 58 crop categories (does not include excess manure applications). Rates account for multi-cropping in some vegetable crops and double-cropping of corn and winter grain. Source: Viers et al. 2012.

Adminsitrative Record
Page 34182

and UC Davis ARE agricultural cost and return studies for each of 58 crop categories within 10 crop groups (Figure 4). In a second step, we assess whether some of the typical nitrogen application rate is met by other sources such as effluent, biosolids, and manure. The procedure varies with crop type, location, and aggregation level. Fertilizer needs not met by effluent, biosolids, or manure (see below) are assumed to be met by synthetic fertilizer, providing an estimate of synthetic fertilizer use at local (Figure 4), crop (see Figure 7), county (see Table 2), and study area (see Figure 3) levels. The magnitude of total estimated synthetic fertilizer use (204 Gg N/yr [225,000 t N/yr]) in the study area, on about 40% of California's irrigated land, is consistent with statewide average recorded sales of synthetic fertilizer used on cropland of 466 Gg N/yr (514,000 t N/yr) (D. Liptzin, pers. comm., 2012).

Cropland inputs: Animal manure (land-applied: 128 Gg N/yr [141,000 t N/yr]; corral and lagoon loading directly to groundwater: 1.7 Gg N/yr [1,900 t N/yr]). The Tulare Lake Basin houses 1 million adult dairy cows and their support stock (more than half of California's dairy herd), 10,000 hogs and pigs, and 15 million poultry animals. Dairy cattle are by far the largest source of land-applied manure nitrogen in the area (127 Gg N/yr [140,000 t N/yr]; see Figure 3). Manure is collected in dry and liquid forms, recycled within the animal housing area for bedding (dry manure) and as flushwater (freestall dairies), and ultimately applied to the land. Manure is applied in solid and liquid forms, typically on forage crops (e.g., summer corn, winter grain) managed by the dairy farm, or is exported to nearby farms (mostly as manure solids) and used as soil amendment. The amount of land-applied manure nitrogen is estimated based on: recently published studies of dairy cow, swine, and poultry excretion rates; animal numbers reported by the Regional Water Board and the USDA Agricultural Census; and an estimated 38% atmospheric nitrogen loss in dairy facilities before land application of the manure. Manure not exported from dairy farms is applied to portions of 130,000 ha (320,000 ac) of dairy cropland. Exported manure nitrogen is largely applied within the study area, mostly within the county of origin, on cropland nearby dairies.

Direct leaching to groundwater from animal corrals and manure lagoons is about 1.5 Gg N/yr (1,700 t N/yr) and 0.2 Gg N/yr (220 t N/yr), respectively (see Table 1).

Cropland inputs: Irrigation water (29 Gg N/yr [32,000 t N/yr]). Irrigation water is also a source of nitrogen applied to crops. Surface irrigation water is generally very low in nitrate. Nitrate in groundwater used as irrigation water is a significant source of nitrogen but varies widely with location and time. We used average nitrate concentrations measured in wells and basin-wide estimates of agricultural groundwater pumping (Faunt 2009) to estimate the total nitrogen application to agricultural lands from irrigation water, in the range of 20 Gg N/yr (22,000 t N/yr) to 33.4 Gg N/yr (36,800 t N/yr).

Cropland and general landscape inputs: Aerial deposition (12 Gg N/yr [13,000 t N/yr]). Nitrogen emissions to the atmosphere as NOx from fossil fuel combustion and ammonia from manure at confined animal feeding operations undergo transformations in the atmosphere before being redeposited, often far from the source of emissions. Nitrogen deposition estimates at broader spatial scales are typically based on modeled data. Nitrogen deposition in urban and natural areas was assumed to be retained with the ecosystem (Vitousek and Howarth 1991). In cropland, nitrogen deposition was included in the nitrogen mass balance. For the Salinas Valley, average aerial deposition is 5.6 kg N/ha/yr (0.6 Gg N/yr) (5.0 lb N/ac [660 t N/yr]). The Tulare Lake Basin receives among the highest levels in the state, averaging 9.8 kg N/ha/yr (11.3 Gg N/yr) (8.7 lb N/ac/yr [12,500 t N/yr]).

Cropland output: Harvested nitrogen (130 Gg N/yr [143,000 t N/yr]). The nitrogen harvested is the largest independently estimated nitrogen output flow from cropland. Historical and current annual County Agricultural Commissioner reports provide annual harvested acreage and yields for major crops. From the reported harvest, we estimate the nitrogen removed. For each of 58 crop categories, the study area total harvest nitrogen and total acreage used to estimate the rate of nitrogen harvested (Figure 5). All crops combined (not including alfalfa) contain a total of 130 Gg N/yr (143,000 t N/yr), with cotton (21 Gg N/yr [23,000 t N/yr]), field crops (28 Gg N/yr [31,000 t N/yr]), grain and hay crops (30 Gg N/yr [33,000 t N/yr]), and vegetable crops (30 Gg N/yr [30,000 t N/yr]) making up 85% of harvested nitrogen. Tree fruits, nuts, grapes, and subtropical crops constitute the remainder of the nitrogen export from cropland.

Adminsitrative Record
Page 34183

Historical Development of Fertilizer Use, Manure Production, Harvested Nitrogen, and Estimated Nitrate Leaching to Groundwater. Current and near-future groundwater nitrate conditions are mostly the result of past agricultural practices. So the historical development of nitrogen fluxes to and from cropland provides significant insight in the relationship between past agricultural practices, their estimated groundwater impacts, and current as well as anticipated groundwater quality. Two major inventions effectively doubled the farmland in production from the 1940s to the 1960s: the introduction of the turbine pump in the 1930s, allowing access to groundwater for irrigation in a region with very limited surface water supplies, and the invention and commercialization of the Haber-Bosch process, which made synthetic fertilizer widely and cheaply available by the 1940s.

The amount of cropland (not including alfalfa) in the study area nearly doubled in less than 20 years, from 0.6 million ha (1.5 million ac) in the mid-1940s to nearly 1.0 million ha (2.5 million ac) in 1960 (not including alfalfa) (Figure 6). Further increases occurred until the 1970s, to 1.3 million ha (3.2 million ac), but the extent of farmland has been relatively stable for the past 30 years.

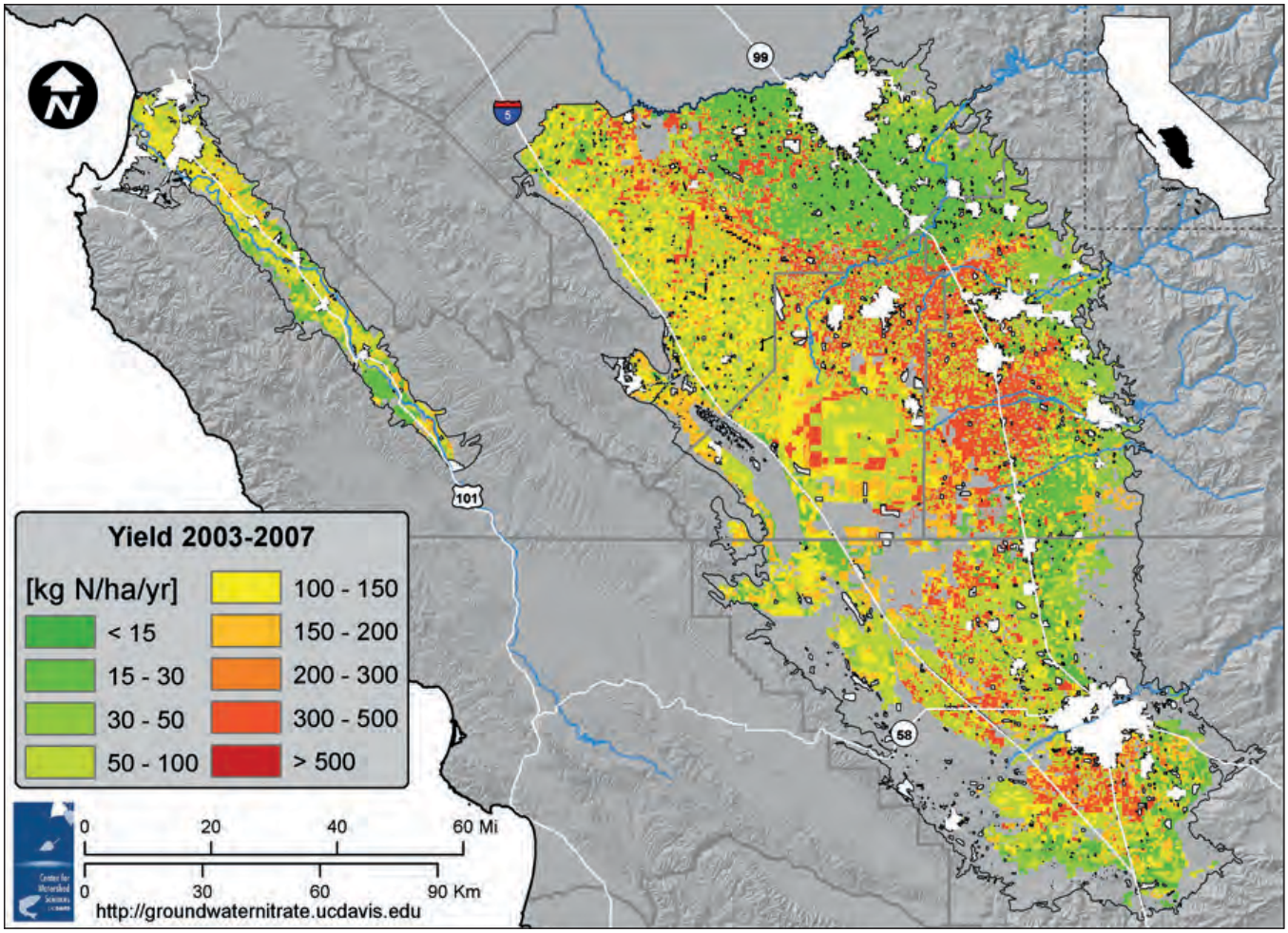


Figure 5. Current annual nitrogen removal rate in harvested materials (1 kg/ha/yr = 1.1 lb/ac/yr) derived from county reports of harvested area and harvested tonnage for each of 58 crop categories. Rates account for multi-cropping in some vegetable crops and double-cropping of corn and winter grain. Source: Viers et al. 2012.

Adminsitrative Record
Page 34184

In contrast, the harvested nitrogen has consistently increased throughout the past 60 years (see Figure 6). From 1945 to 1975, total harvested nitrogen increased twice as fast as farmland expansion, quadrupling from 20 Gg N/yr (22,000 t N/yr) to 80 Gg N/yr (88,000 t N/yr). Without further increases in farmland, harvests and harvested nitrogen increased by more than 60% in the second 30-year period, from the mid-1970s to the mid-2000s.

Synthetic fertilizer inputs also increased from the 1940s to the 1980s but have since leveled off. Between 1990 and 2005, the gap between synthetic nitrogen fertilizer applied and harvested nitrogen has significantly decreased.[4]

In contrast, dairy manure applied to land has increased exponentially, effectively doubling every 15 years (see Figure 6), from 8 Gg N/yr (9,000 t N/yr) in 1945 to 16 Gg N/yr (18,000 t N/yr) in 1960, 32 Gg N/yr (35,000 t N/yr) in 1975, 56 Gg N/yr (62,000 t N/yr) in 1990, and 127 Gg N/yr (140,000 t N/yr) in 2005, an overall 16-fold increase in manure nitrogen output. The increase in manure nitrogen is a result of increasing herd size (7-fold) and increasing milk production per cow (3-fold) and is slowed only by the increased nitrogen-use efficiency of milk production.

Until the 1960s, most dairy animals in the region were only partly confined, often grazing on irrigated pasture with

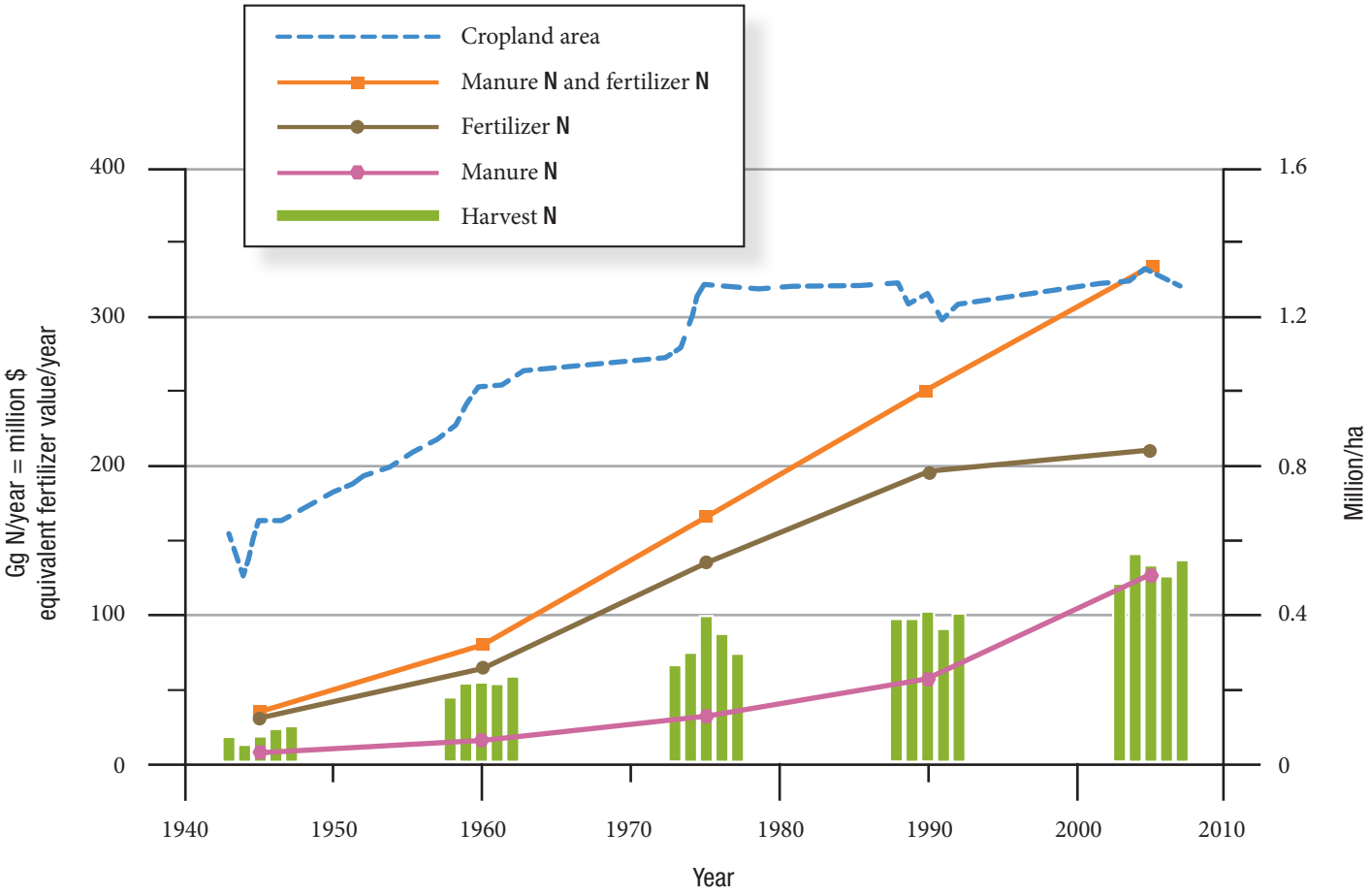


Figure 6. Estimated historical agricultural development in the study area (not including alfalfa): total harvested area, total harvested nitrogen in fertilized crops, fertilizer applied to cropland (5-year average), manure applied to cropland (5-year average), and sum of manure and fertilizer applied to cropland (5-year average). Not shown: In the study area, harvested alfalfa area grew from 0.12 million ha (0.3 million ac) in the 1940s to 0.2 million ha (0.5 million ac) around 1960, then leveled off to current levels of 0.17 million ha (0.42 million ac). Since the 1960s, nitrogen removal in alfalfa harvest has varied from 50 to 80 Gg N/yr. Note: 0.4 million ha = 1 million ac. Source: Viers et al. 2012.

[4] Fertilizer application rates and statewide fertilizer sales have grown little since the late 1980s.

Adminsitrative Record
Page 34185

limited feed imports. Manure from dairy livestock generally matched the nitrogen needs of dairy pastures. Since the 1970s, dairies in the Tulare Lake Basin have operated mostly as confined animal facilities, growing alfalfa, corn, and grain feed on-site, importing additional feed, and housing the animals in corrals and freestalls. The growth in the dairy industry has created a nitrogen excess pool that remains unabsorbed by crops (see Figure 6). Much of the nitrogen excess is a recent phenomenon (see Figure 6). With groundwater quality impacts delayed by decades in many production wells (see Section 3), the recent increase in land applied manure nitrogen is only now beginning to affect water quality in wells of the Tulare Lake Basin, with much of the impact yet to come.

Groundwater loading from irrigated agriculture, by crop group and by county. Significant differences exist in groundwater loading intensity between crop groups.[5] The intensity of groundwater loading is least in vineyards (less than 35 kg N/ha/yr [31 lb N/ac/yr]), followed by rice and subtropical tree crops (about 60 kg N/ha/yr [54 lb N/ac/yr]), tree fruits, nuts, and cotton (90–100 kg N/ha/yr [80–90 lb N/ac/yr]), vegetables and berry crops (over 150 kg N/ha/yr [130 lb N/ac/yr]), which includes some vegetables being cropped twice per year), field crops (about 480 kg N/ha/yr [430 lb N/ac/yr]), and grain and hay crops (about 200 kg N/ha/yr [180 lb N/ac/yr]). Manure applications constitute the source of nearly all of the nitrate leaching from these latter two crops. Without manure, field crops leach less than 35 kg N/ha/yr (31 lb N/ac/yr), and grain and hay crops leach 50 kg N/ha/yr (45 lb N/ac/yr). Figure 7 shows the rate of reduction (in kg N/ha/crop) that would be needed, on average across each crop group, to reduce groundwater nitrate leaching to benchmark levels.

At the county level, we aggregate cropland area, fertilizer applications (by crop category), manure output from individual dairies, effluent and biosolid land applications from individual facilities, and crop category–specific harvest. Differences in cropping patterns between counties and the absence or presence of dairy facilities within counties drive county-by-county differences in total groundwater loading and in the average intensity of groundwater loading (Table 2). Fresno County, which has fewer mature dairy cows (133,000) than Kings (180,000), Tulare (546,000), or Kern (164,000) Counties and also has large areas of vineyards (see Figure 2), has the lowest average groundwater loading intensity (103 kg N/ha/yr [103 lb N/ac/yr]). Monterey County is dominated by vegetable and berry crops (high intensity) and grape vineyards (low intensity).

Urban and Domestic Sources

Urban and domestic sources: Overview. Urban nitrate loading to groundwater is divided into four categories: nitrate leaching from turf, nitrate from leaky sewer systems, groundwater nitrate contributions from WWTPs and FPs, and groundwater nitrate from septic systems. For all these systems, groundwater nitrate loading is estimated based on either actual data or reported data of typical nitrate leaching.

Urban and domestic sources: Wastewater treatment plants and food processors (11.4 Gg N/yr [12,600 t/yr]: 3.2 Gg N/yr [3,500 t/yr] to percolation ponds, 3.4 Gg N/yr [3,800 t/yr] in effluent applications to cropland, and 4.8 Gg N/yr [5,300 t/yr] in WWTP biosolids applications to cropland). The study area has roughly 2 million people on sewer systems that collect and treat raw sewage in WWTPs. In addition, many of the 132 food processors within the study area generate organic waste that is rich in nitrogen (Table 3). Potential sources of groundwater nitrate contamination from these facilities include effluent that is land applied on cropland or recharged directly to groundwater via percolation basins, along with waste solids and biosolids that are land applied. Typically, WWTP influent contains from 20 mg N/L to 100 mg N/L total dissolved nitrogen (organic N, ammonium N, nitrate-N), of which little is removed in standard treatment (some WWTPs add treatment beyond

[5] Aggregated estimates were obtained from study area-wide totals for harvested area (by crop group), for typical nitrogen application, and for harvested nitrogen. The following averages were assumed: irrigation water nitrogen (24 kg N/ha/yr [21 lb N/ac/yr]), atmospheric nitrogen losses (10% of all N inputs), and runoff (14 kg N/ha/yr [12.5 lb N/ac/yr]). Most manure is likely land-applied to field crops, particularly corn, and to grain and hay crops. Little is known about the actual distribution prior to 2007 and the amount of synthetic fertilizer applied on fields receiving manure. As an illustrative scenario, we assume that two-thirds of dairy manure is applied to field crops and one-third of dairy manure is applied to grain and hay crops. In field crops, 50% of crop nitrogen requirements are assumed to be met with synthetic fertilizer, and in grain and hay crops 90% of their crop nitrogen requirements are assumed to be met by synthetic fertilizer. These are simplifying assumptions that neglect the nonuniform distribution of manure on field and grain crops between on-dairy, near-dairy, and away-from-dairy regions. However, corn constitutes most (106,000 ha [262,000 ac]) of the 130,000 ha (321,000 ac) in field crops, with at least 40,000 ha (99,000 ac) grown directly on dairies. Grain crops are harvested from 220,000 ha (544,000 ac). For further detail, see Viers et al. 2012.

Adminsitrative Record
Page 34186

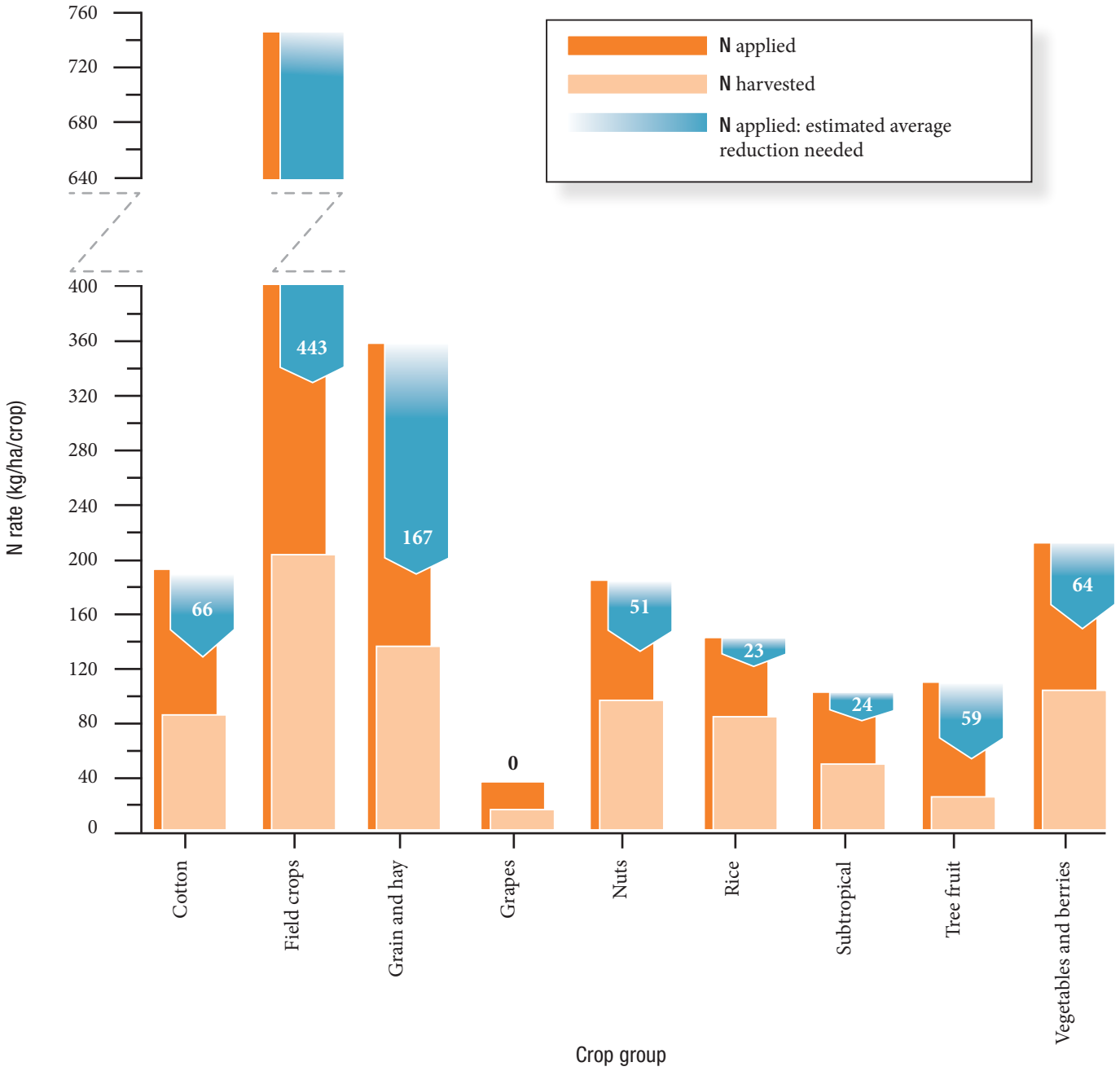


Figure 7. Nitrogen application reduction needed to reduce groundwater nitrate loading to less than 35 kg N/ha/crop, compared with average nitrogen applied (synthetic fertilizer and manure) and nitrogen harvested (all units in kg N/ha/crop). Rates are given per crop, and the required reduction does not account for double-cropping. Some vegetables and some field crops are harvested more than once per year. In that case, additional reductions in fertilizer applications would be necessary to reduce nitrate loading to less than 35 kg N/ha. Large reductions needed in field crops and grain and hay crops are due to the operational assumption that manure generated in the study area is applied to only these crop groups. Typical amounts of synthetic fertilizer applied ("N applied") to these crops, without excess manure, are 220 kg N/ha/crop for field crops and 190 kg N/ha/crop for grain and hay crops. Thus, without excess manure, average field crops and grain and hay crops may require relatively small reductions in nitrogen application. Source: Viers et al. 2012.

Adminsitrative Record
Page 34187

Table 2. Major nitrogen fluxes to and from cropland in the study area, by county (not including alfalfa)

	Synthetic Fertilizer Application Gg N/yr [1,000 t N/yr]	Manure Application Gg N/yr [1,000 t N/yr]	Land Applied Effluent and Biosolids, Gg N/yr [1,000 t N/yr]	Harvest Gg N/yr [1,000 t N/yr]	PNB* %	PNB_0 † %	Groundwater Loading Gg N/yr [1,000 t N/yr]	Groundwater Loading Intensity kg N/ha/yr [lb N/ac/yr]
By County								
Fresno	62.1 [68.3]	16.6 [18.3]	0.8 [0.88]	35.5 [39.1]	44.7	54.4	42.4 [46.7]	103 [92]
Kern	50.3 [55.4]	20.4 [22.5]	4.6 [5.0]	29.6 [32.6]	39.3	56.4	42.8 [47.2]	141 [123]
Kings	27.5 [30.3]	22.0 [24.3]	1.9 [2.1]	19.6 [21.6]	38.1	62.7	29.2 [32.2]	179 [160]
Tulare	36.0 [39.7]	67.3 [74.2]	0.7 [0.77]	32.7 [36.0]	31.4	72.5	65.1 [71.8]	236 [210]
Monterey	28.1 [30.9]	1.4 [1.54]	0.1 [0.11]	12.4 [13.6]	41.9	43.5	15.6 [17.2]	138 [123]
By Basin								
TLB	176 [194]	127 [140]	8.1 [8.9]	118 [130]	37.8	60.5	179 [197]	155 [138]
SV	28 [30.8]	1 [1.1]	0.1 [0.11]	12 [13]	41.9	43.5	16 [18]	138 [123]
Overall	**204 [225]**	**128 [141]**	**8.2 [9]**	**130 [143]**	**38.2**	**58.3**	**195 [215]**	**154 [137]**

Source: Viers et al. 2012.

Manure applications include non-dairy manure nitrogen (0.9 Gg N/yr [(990 t N/yr)] for the entire study area). Groundwater loading accounts for atmospheric deposition (9.8 and 5.6 kg N/ha/yr [(8.7 and 5 t N/yr)] in TLB and SV, respectively), atmospheric losses (10% of all inputs), irrigation water quality (22.8 kg N/ha/yr [20 lb N/ac/yr]), and runoff (14 kg N/ha/yr [12.5 lb N/ac/yr]) to and from agricultural cropland, in addition to fertilizer and manure application, and harvested nitrogen. Synthetic fertilizer application on field crops is assumed to meet 50% of typical application rates; on grain and hay crops, 90% of typical applications, with the remainder met by manure.

* PNB = partial nutrient balance, here defined as Harvest N divided by (Synthetic + Manure + Effluent + Biosolids Fertilizer N).

† PNB_0 = hypothetical PNB, if no manure/effluent/biosolids overage was applied above typical fertilizer rates.

Adminsitrative Record
Page 34188

conventional processes to remove nutrients including nitrate and other forms of nitrogen). Across the study area, WWTP effluent nitrogen levels average 16 mg N/L. Within the study area, 40 WWTPs treat 90% of the urban sewage. FP effluent nitrogen levels to percolation basins and irrigated agriculture average 42 mg N/L and 69 mg N/L, respectively.

Urban and domestic sources: Septic systems (2.3 Gg N/yr [2,500 t N/yr]). Crites and Tchobanoglous (1998) estimated that the daily nitrogen excretion per adult is 13.3 g. Approximately 15% of that nitrogen is assumed to either stay in the septic tank, volatilize from the tank, or volatilize from the septic leachfield (Siegrist et al. 2000). Based on census data, the number of people on septic systems in the study areas is about 509,000 for the Tulare Lake Basin and 48,300 for Salinas Valley. Total nitrate loading from septic leaching is 2.1 Gg N/yr (2,300 t N/yr) in the Tulare Lake Basin and 0.2 Gg N/yr (220 t N/yr) in the Salinas Valley. The distribution of septic systems varies greatly. The highest density of septic systems is

Table 3. Total nitrogen discharge to land application and average total nitrogen concentration (as nitrate-N, MCL: 10 mg N/L) in discharge to percolation basins from WWTPs and FPs, based on our surveys of WWTPs and the FP survey of Rubin et al. (2007)

	Biosolids Gg N/yr [1,000 t N/yr]	WWTP Land Application Gg N/yr [1,000 t N/yr]	WWTP Percolation Concentration mg N/L	FP Land Application Gg N/yr [1,000 t N/yr]	FP Percolation Concentration mg N/L
By County					
Fresno	0.006 [0.006]	0.40 [0.40]	18.5	0.42 [0.46]	56.2
Kern	3.1 [3.4]	0.92 [0.92]	17.7	0.56 [0.62]	43.9
Kings	1.6 [1.7]	0.09 [0.09]	11.2	0.26 [0.29]	2.1
Tulare	0.038 [0.044]	0.50 [0.50]	14.9	0.13 [0.14]	34.2
Monterey	0 [0]	0.09 [0.09]	13.9	0.05 [0.05]	22.1
By Basin					
Tulare Lake Basin	4.8 [5.3]	1.9 [2.1]	16.3	1.37 [1.51]	43.3
Salinas Valley	0 [0]	0.09 [0.09]	13.9	0.05 [0.05]	22.1
Overall	**4.8 [5.3]**	**2.0 [2.2]**	**16**	**1.4 [1.5]**	**42**

Adminsitrative Record
Page 34189

in peri-urban (rural sub-urban) areas near cities but outside the service areas of the wastewater systems that serve those cities (Figure 8). In the Tulare Lake Basin and Salinas Valley, 7.9% and 12.6%, respectively, of the land area exceeds the EPA-recommended threshold of 40 septic systems per square mile (0.154 systems per ha). Nearly 1.5% of the study area has a septic system density of over 256 systems per square mile (1 system/ha, or 1 system/2.5 ac). In those areas, groundwater leaching can significantly exceed our operational benchmark rate of 35 kg N/ha/yr (31 lb N/ac/yr).

Urban and domestic sources: Fertilizer and leaky sewer lines (0.88 Gg N/yr [970 t N/yr]). Fertilizer is used in urban areas for lawns, parks, and recreational facilities such as sports fields and golf courses. These land uses differ in their recommended fertilizer use, and there is almost no evidence of actual fertilization rates. Based on the most comprehensive survey of turfgrass leaching, only about 2% of applied nitrogen fertilizer was found to leach below the rooting zone (Petrovic 1990). For our nitrogen flow calculations, we assume a net groundwater loss of 10 kg N/ha/yr (8.9 lb N/ac/yr) from lawns and golf courses in urban areas (0.35 Gg N/yr [380 t N/yr]).

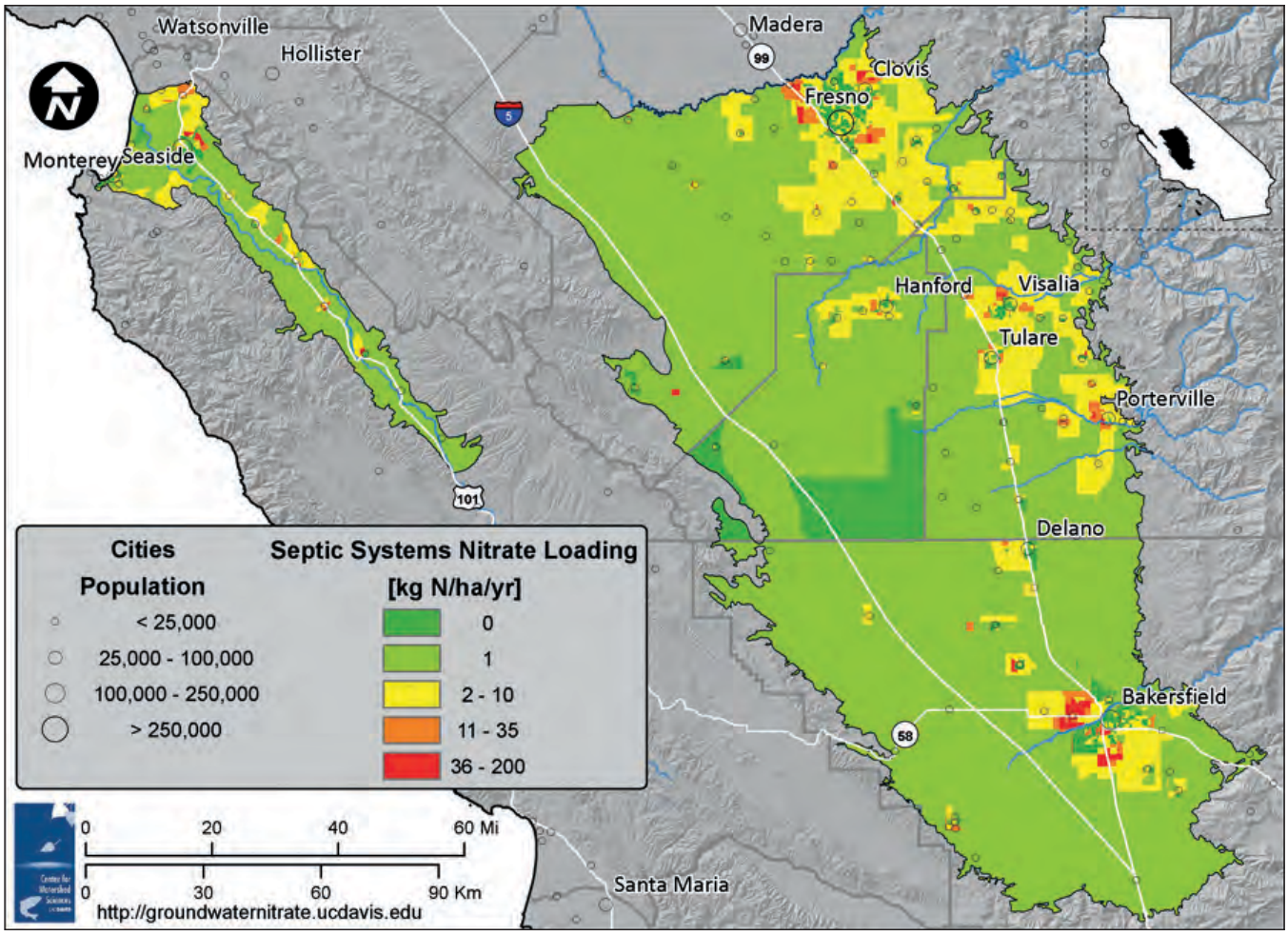


Figure 8. Septic-derived nitrate leaching rates within the study area. Source: Viers et al. 2012.

Adminsitrative Record
Page 34190

Sewer systems in urban areas can be a locally significant source of nitrogen. We use both reported sewer nitrogen flows and per capita nitrogen excretion rates to obtain total nitrogen losses via leaky sewer lines in urban areas. Nationally, estimated municipal sewer system leakage rates range from 1% to 25% of the total sewage generated. Given that much of the urban area within the study region is relatively young, we consider that the leakage rate is low, roughly 5% or less (0.53 Gg N/yr).

General Sources

General sources: Wells, dry wells, and abandoned wells (<0.4 Gg N/yr [<440 t N/yr]). Wells contribute to groundwater nitrate pollution through several potential pathways. Lack of or poor construction of the seal between the well casing and the borehole wall can lead to rapid transport of nitrate-laden irrigation water from the surface into the aquifer. In an inactive or abandoned production well, long well screens (several hundred feet) extending from relatively shallow depth to greater depth, traversing multiple aquifers, may cause water from nitrate-contaminated shallow aquifer layers to pollute deeper aquifer layers, at least in the vicinity of wells. Dry wells, which are large-diameter gravel-filled open wells, were historically designed to capture stormwater runoff or irrigation tailwater for rapid recharge to groundwater. Abandoned wells also allow surface water leakage to groundwater (spills) and cross-aquifer contamination. Lack of backflow prevention devices can lead to direct introduction of fertilizer chemicals into the aquifer via a supply well. Few data are available on these types of nitrate transfer in the Tulare Lake Basin or Salinas Valley. In a worst-case situation, as much as 0.4 Gg N/yr (440 t N/yr) may leak from the surface to groundwater via improperly constructed, abandoned, or dry wells, and as much as 6.7 Gg N/yr (7,400 t N/yr) are transferred within wells from shallow to deeper aquifers. Actual leakage rates are likely much lower than these worst-case estimates.

Groundwater Nitrate Loading: Uncertainty. The analyses above provide specific numbers for the average amount and intensity of nitrate loading from various categories of sources. However, discharges of nitrate to groundwater may vary widely between individual fields, farms, or facilities of the same category due to differences in operations, management practices, and environmental conditions. Also, average annual nitrate loading estimates for specific categories are based on many assumptions and are based on (limited) data with varying degrees of accuracy; the numbers given represent a best, albeit rough, approximation of the actual nitrate loading from specific sources. These estimates have inherent uncertainty. Very likely, though, the actual groundwater nitrate loading from source categories falls within the ranges shown in Table 1.

2.3 Reducing Nitrate Source Emissions to Groundwater

Although reduction of anthropogenic loading of nitrate to groundwater aquifers will not reduce well contamination in the short term (due to long travel times), reduction efforts are essential for any long-term improvement of drinking water sources. Technologies for reducing nitrate contributions to groundwater involve (a) reducing nitrogen quantity discharged or applied to the land and (b) controlling the quantity of water applied to land, which carries nitrate to groundwater (Dzurella et al. 2012).

Many source control methods require changes in land management practices and upgrading of infrastructure. Costs for mitigation or abatement vary widely and can be difficult to estimate. In particular, the quantity of nitrate leached from irrigated fields (the largest source) is determined by a complex interaction of nitrogen cycle processes, soil properties, and farm management decisions. Only broad estimates of the cost of mitigation per unit of decrease in the nitrate load are possible.

Reducing Nitrate Loading from Irrigated Cropland and Livestock Operations

Reduction of nitrate leaching from cropland, livestock, and poultry operations can come from changes in farm management that improve crop nitrogen use efficiency and proper storage and handling of manure and fertilizer. A common measure of cropland nitrogen use efficiency is the partial nitrogen balance (PNB), which is the ratio of harvested nitrogen to applied (synthetic, manure, or other organic) fertilizer nitrogen (Table 2).

We reviewed technical and scientific literature to compile a list of practices known or theorized to improve crop nitrogen use efficiency. Crop-specific expert panels

Adminsitrative Record
Page 34191

reviewed and revised this list of practices. Input from these panel members also helped to estimate the current extent of use of each practice in the study area and to identify barriers to expanded adoption.

PNB can be increased by optimizing the timing and application rates of fertilizer nitrogen, animal manure, and irrigation water to better match crop needs, and to a lesser extent by modifying crop rotation. Improving the storage and handling of manure, livestock facility wastewater, and fertilizer also helps reduce nitrate leaching. A suite of improved management practices is generally required to reduce nitrate leachate most effectively, and these must be chosen locally for each unique field situation. No single set of management practices will be effective in protecting groundwater quality everywhere. The best approach depends on the crop grown, soil characteristics of the field, and other specific factors. As summarized in Table 4, ten key farm management measures for increasing crop nitrogen use efficiency (and PNB) are identified and reviewed (Dzurella et al. 2012).

Although PNBs as low as 33% have been reported, a recent EPA report estimated that with the adoption of best management practices, PNB could increase by up to 25% of current average values (U.S. EPA 2011). Improvements in PNB are possible, but a practical upper limit is about 80% crop recovery of applied nitrogen (U.S. EPA 2011; Raun and Schepers 2008). This limit is due to the unpredictability of rainfall, the difficulty in predicting the rate of mineralization of organic nitrogen in the soil, spatial variability and nonuniformity in soil properties, and the need to leach salts from the soil.

Table 4. Management measures for improving nitrogen use efficiency and decreasing nitrate leaching from agriculture (local conditions determine which specific practices will be most effective and appropriate)

Basic Principle	Management Measure	Number of Recommended Practices
Design and operate irrigation and drainage systems to decrease deep percolation.	MM 1. Perform irrigation system evaluation and monitoring.	3
	MM 2. Improve irrigation scheduling.	4
	MM 3. Improve surface gravity system design and operation.	6
	MM 4. Improve sprinkler system design and operation.	5
	MM 5. Improve microirrigation system design and operation.	2
	MM 6. Make other irrigation infrastructure improvements.	2
Manage crop plants to capture more N and decrease deep percolation.	MM 7. Modify crop rotation.	4
Manage N fertilizer and manure to increase crop N use efficiency.	MM 8. Improve rate, timing, placement of N fertilizers.	9
	MM 9. Improve rate, timing, placement of animal manure applications.	6
Improve storage and handling of fertilizer materials and manure to decrease off-target discharges.	MM 10. Avoid fertilizer material and manure spills during transport, storage, and application.	9
		Total: 50

Source: Dzurella et al. 2012.

Adminsitrative Record
Page 34192

Based on expert panel commentary, several farm management practices that reduce nitrate leaching have been widely adopted in recent years in the study area, representing a positive change from past practices that have contributed to current groundwater nitrate concentrations. High PNB can sometimes increase yields and decrease costs to the producer (by decreasing costs for fertilizer and water). Alas, field data that document improvements in nitrate leaching from these actions are largely unavailable.

Significant barriers to increased adoption of improved practices exist. These include higher operating or capital costs, risks to crop quality or yield, conflicting farm logistics, and constraints from land tenure. Lack of access to adequate education, extension, and outreach activities is another primary barrier, especially for the adoption of many of the currently underused practices, highlighting the importance of efforts such as those offered by the University of California Cooperative Extension. The future success of leaching reductions through improved crop and livestock facility management will require a significant investment in crop-specific research that links specific management practices with groundwater nitrate contamination. Additional investments in farmer (and farm labor) education and extension opportunities are needed, as well as increased support for farm infrastructure improvements. Monitoring and assessment programs need to be developed to evaluate management practices being implemented and their relative efficacy.

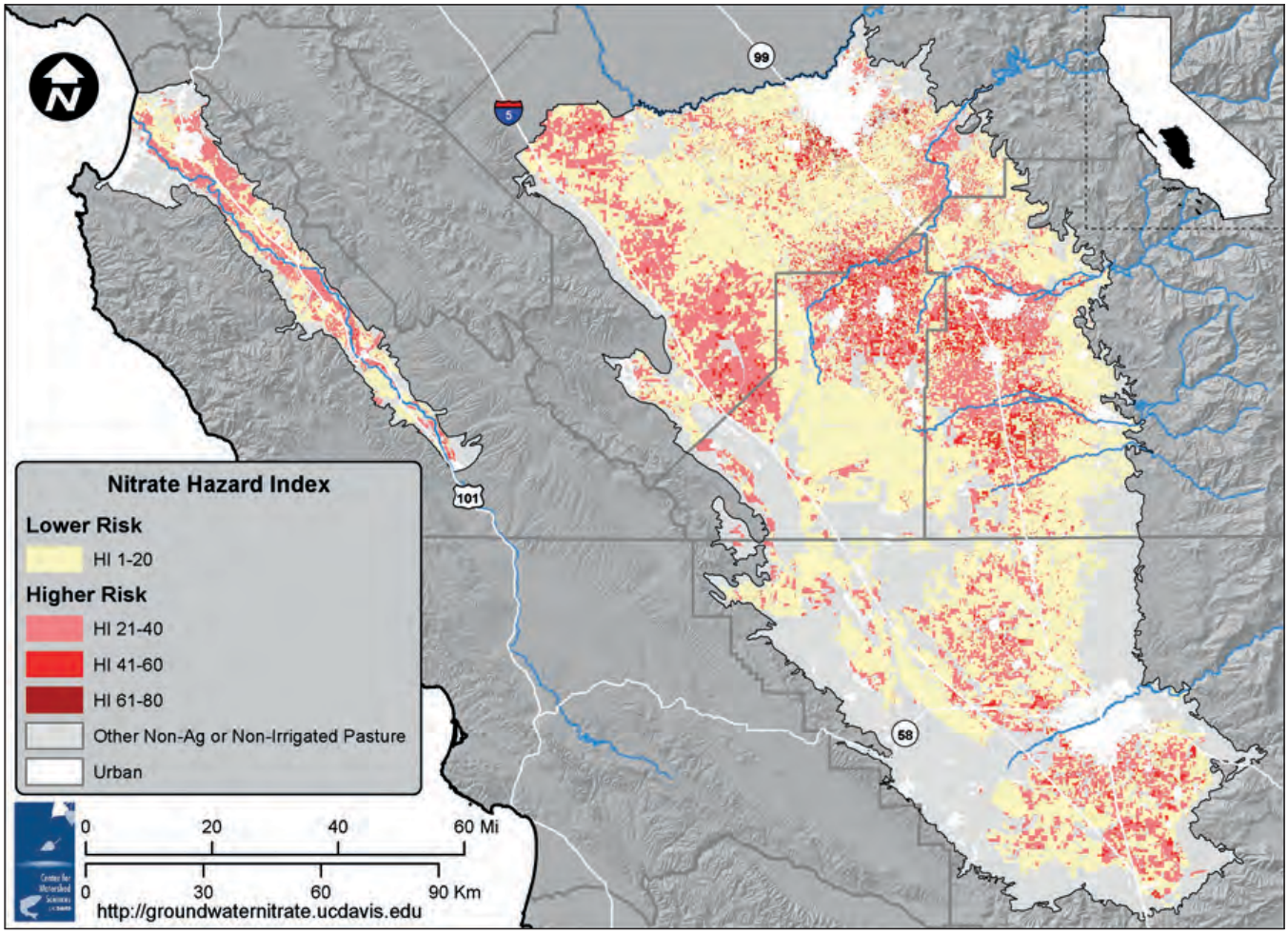


Figure 9. Overall nitrate hazard index calculated for the study area fields. Index values over 20 indicate increased potential for nitrate leaching from the crop root zone, benefiting most from implementation of improved management practices. Comparison between values in the higher-risk categories is not necessarily an indication of further risk differentiation, but it may indicate that multiple variables are involved in risk. Less-vulnerable areas still require vigilance in exercising good farm management practices. Source: Dzurella et al. 2012.

Adminsitrative Record
Page 34193

To establish the areas that would benefit most from improved management practices, we conducted a vulnerability assessment. Management-specific vulnerability was mapped using the UC Nitrate Hazard Index (Wu et al. 2005), which calculates the potential of nitrate leaching as a function of the crop grown, the irrigation system type in use, and the soil characteristics of each individual field. Based on this information, approximately 52% of irrigated cropland in the Salinas Valley and 35% of such land in the Tulare Lake Basin would most benefit from broad implementation of improved management practices (Figure 9).

A maximum net benefit modeling approach was developed to estimate relative costs of policies to improve PNB while maintaining constant crop yields for selected crop groups in the study area. Net revenue losses from limiting nitrate load to groundwater increase at an increasing rate (Table 5 and Figure 10). Our modeling results, although preliminary due to the lack of data on the cost of improving nitrogen use efficiency, suggest that reductions of 25% in total nitrate load to groundwater from crops will slightly increase production costs but are unlikely to affect total irrigated crop area, as summarized in Table 5. Smaller reductions (<10%) can be achieved at low costs, assuming adequate farmer education is in place (see Figure 10).

Greater reductions in total nitrate loading (>50%) are much more costly to implement, as capital and management investments in efficient use of nitrogen are required. Achieving such high load reductions may ultimately shift cropping toward more profitable and nitrogen-efficient crops or fallowing, as lower-value field crops and low-PNB crops lose

Table 5. Summary of how two groundwater nitrate load reduction scenarios may affect total applied water, annual net revenues, total crop area, and nitrogen applications, according to our estimative models for each basin*

Region	Scenario	Applied Water km^3/yr [million AF/yr]	Net Revenues $M/yr (2008)	Irrigated Land 1,000 ha [ac]	Applied Nitrogen Gg N/yr (%) [1,000 t/yr]
Tulare Lake Basin	base load	10.5 [8.5]	4,415 (0%)	1,293 [3,194]	200 (0%) [221]
	25% load reduction	10.0 [8.1]	4,259 (–3.5%)	1,240 [3,064]	181 (–9%) [199]
	50% load reduction	7.9 [6.4]	3,783 (–14%)	952 [2,352]	135 (–32%) [149]
Salinas Valley	base load	0.37 [0.30]	309 (0%)	92 [227]	18 (0%) [19]
	25% load reduction	0.33 [0.27]	285 (–7.5%)	83 [205]	15 (–16%) [16]
	50% load reduction	0.25 [0.20]	239 (–22%)	62 [153]	10 (–46%) [11]

Source: Dzurella et al. 2012.

* Irrigated land area and applied nitrogen in base load vary slightly from those reported in Section 2.2 due to land use data being based on Figure 2 (derived from DWR data) instead of County Agricultural Commissioner Reports (Figure 6).

Adminsitrative Record
Page 34194

favor economically. The average net revenue loss of reducing nitrate loading to groundwater is estimated to be $16 per kilogram of nitrogen at this 50% reduction level. Modeling a 7.5% sales fee on nitrogen fertilizer indicated an estimated reduction in total applied nitrogen by roughly 1.6%, with a 0.6% loss in net farm revenues.

Agricultural source reduction: Promising actions. Expanded efforts to promote nitrogen-efficient practices are needed. Educational and outreach activities could assist farmers in applying best management practices (BMPs) and nutrient management. Research should focus on demonstrating the value of practices on PNB and on adapting practices to local conditions for crop rotations and soils with the greatest risk of nitrate leaching. This especially includes row crops receiving high rates of nitrogen and/or manure that are surface- or sprinkler-irrigated. Research on the costs of increasing nitrogen use efficiency in crops would greatly benefit the capacity to estimate the economic costs of reductions in agricultural nitrate loading to groundwater. Research and education programs are needed to promote conversion of solid and liquid dairy manure into forms that meet food safety and production requirements for a wider range of crops.

We suggest that a working group develop crop-specific technical standards on nitrogen mass balance metrics for regulatory and assessment purposes. This nitrogen-driven metric would reduce the need for more expensive direct measurement of nitrate leaching to groundwater. Such

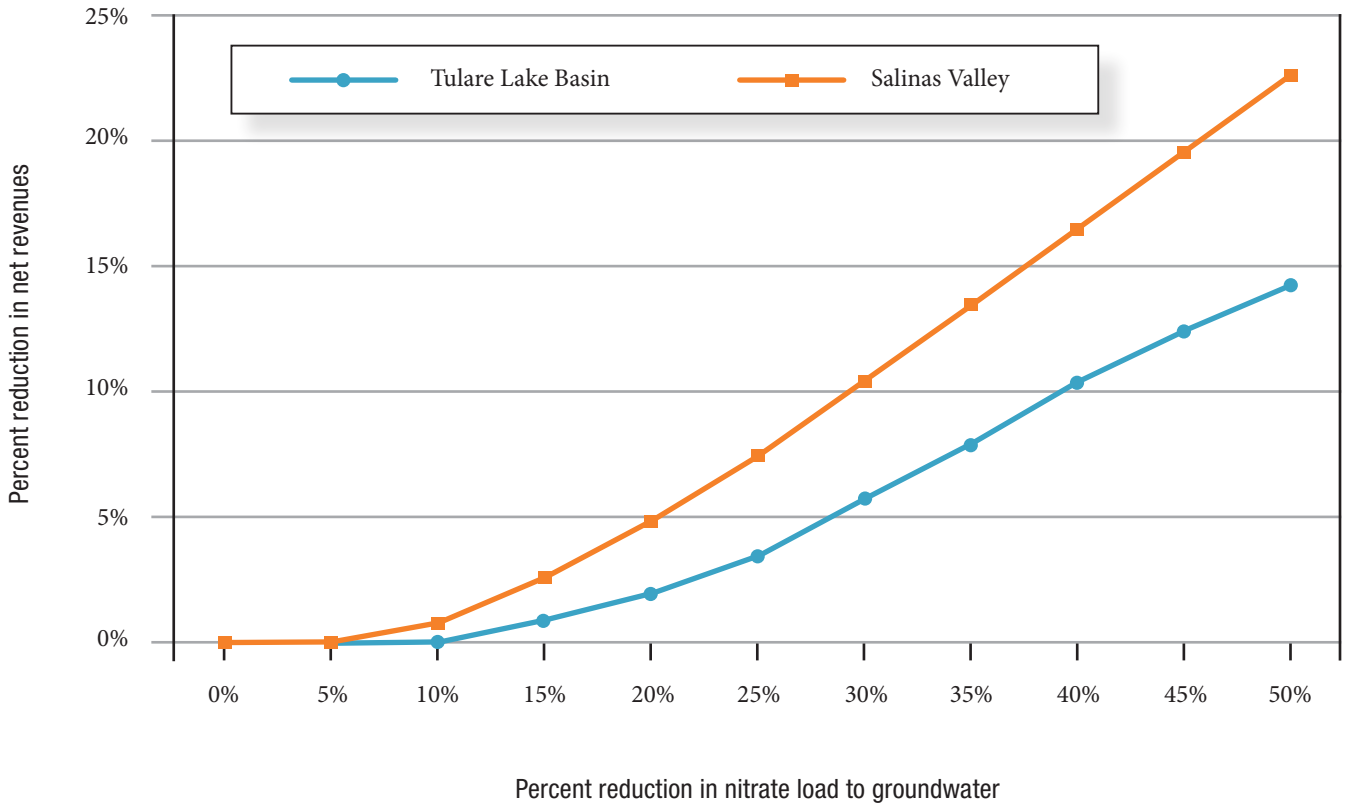


Figure 10. Percent reduction in net revenues estimated from different levels of reduction in nitrate loading to groundwater. Source: Dzurella et al. 2012.

Adminsitrative Record
Page 34195

metrics would also serve as a starting point to assist farmers in assessing their crop nitrogen use efficiency and be useful for nitrogen management. Finally, we recommend that a task force review and further develop methods to identify croplands most in need of improved management practices. Such a method should include consideration of soil characteristics (as in the UC Nitrate Hazard Index), as well as possible monitoring requirements.

Reducing Nitrate Leaching from Municipal Wastewater Treatment and Food Processing Plants

Implementation of nitrogen control options for WWTP and FP sources is feasible and useful. Nitrogen removal from wastewater can be accomplished using a variety of technologies and configurations; both biological and physical or chemical processes are effective. The selection of the most appropriate treatment option depends on many factors.

Estimated capital costs for nutrient removal from all wastewater (FPs and WWTPs) for facilities categorized as "at-risk" range from $70 to $266 million. Cropland application of wastewater treatment and food processing effluents can reduce direct groundwater contamination and total fertilizer application requirements of such fields, as the water and nutrients are effectively treated and recycled. These wastes should be managed in an agronomic manner rather than applied to land for disposal or land treatment purposes so that the nutrients are included in the overall nitrogen management plan for the receiving crops.

Optimizing wastewater treatment plant and food processing plant operations is another way to reduce nitrogen and total discharge volume. Facility process modifications may be sufficient in some cases. Groundwater monitoring is required for many facilities, but the data are largely unavailable since they are not in a digital format. To improve monitoring, enforcement, and abatement efforts related to these facilities, groundwater data need to be more centrally managed and organized digitally.

Reducing Nitrate Contributions from Leaking Sewer Pipes and Septic Systems

Retrofitting of septic system components and sewer pipes is the main way to diminish loading from these sources. Replacing aging sewer system infrastructure and ensuring proper maintenance are required to reduce risks to human health; such infrastructure upgrades also reduce nitrate leaching.

Loading from septic systems, significant locally, can be reduced significantly by two approaches where connection to a sewer system is not possible. Source separation technology can reduce nitrate loading to wastewater treatment systems by about 50%. Costs include separating toilets ($300–$1,100), dual plumbing systems ($2,000–$15,000), storage tank costs, and maintenance, pumping, heating, and transport costs (where applicable). Post-septic tank biological nitrification and denitrification treatment reduces nitrate concentrations below levels achieved via source separation technology but does not result in a reusable resource. Wood chip bioreactors have reduced influent nitrate by 74% to 91%, with costs ranging from $10,000 to $20,000 to retrofit existing septic systems.

Reducing Nitrate Leaching from Turfgrass in Urban Areas

Nitrate leaching from urban turfgrass, including golf courses, is often negligible due to the dense plant canopy and perennial growth habit of turf, which results in continuous plant nitrogen uptake over a large portion of the year. However, poor management can lead to a discontinuous canopy and weed presence, wherein nitrate leaching risk increases, especially if the turf is grown on permeable soils, is overirrigated, or is fertilized at high rates during dormant periods. The UCCE and UC IPM publish guidelines on proper fertilizer use in turfgrass. The knowledge and willingness of homeowners and groundskeepers to apply guidelines depend on funding for outreach efforts.

Reducing Nitrate Transfer and Loading from Wells

Backflow prevention devices should be required on agricultural and other wells used to mix fertilizer with water. Furthermore, local or state programs and associated funding to identify and properly destroy abandoned and dry wells are needed to prevent them from becoming nitrate transfer conduits. However, many well owners may not be able to afford the high costs of retrofitting long-screened wells to seal contaminated groundwater layers. As such, enforcement of proper well construction standards for future wells may be more feasible. Expenditures on retrofitting of existing dry and abandoned wells should be based on the contamination risks of individual wells. The nitrate contamination potential of wells needs to be identified as a basis for developing and enforcing improved, appropriate well construction standards that avoid the large-scale transfer of nitrate to deep groundwater in all newly constructed wells.

Adminsitrative Record
Page 34196

3 Impact: Groundwater Nitrate Occurrence

3.1 Current Groundwater Quality Status

We assembled groundwater quality data from nearly two dozen local, state, and federal agencies and other sources into a dataset, here referred to as the (Central) California Spatio-Temporal Information on Nitrate in Groundwater (CASTING) dataset (see Table 6 for information about data sources, Boyle et al. 2012). The dataset combines nitrate concentrations from 16,709 individual samples taken at 1,890 wells in the Salinas Valley and from 83,375 individual samples taken at 17,205 wells in the Tulare Lake Basin collected from the 1940s to 2011, a total of 100,084 samples from 19,095 wells. Almost 70% of these samples were collected from 2000 to 2010; only 15% of the samples were collected prior to 1990. Half of all wells sampled had no recorded samples prior to 2000 (Boyle et al. 2012).

Of the nearly 20,000 wells, 2,500 are frequently sampled public water supply wells (over 60,000 samples). Apart from the recently established Central Valley dairy regulatory program, which now monitors about 4,000 domestic and irrigation wells in the Tulare Lake Basin, there are no existing regular well sampling programs for domestic and other private wells.

From 2000 to 2011, the median nitrate concentration in the Tulare Lake Basin and Salinas Valley public water supply well samples was 23 mg/L and 21 mg/L,[6] respectively, and in all reported non-public well samples, 23 mg/L and 20 mg/L, respectively. In public supply wells, about one in ten raw water samples exceeds the nitrate MCL. Nitrate concentrations in wells vary widely with location and well depth. More domestic wells and unregulated small system wells have high nitrate concentrations due to their shallow depth (Table 6). Highest nitrate concentrations are found in wells of the alluvial fans in the eastern Tulare Lake Basin and in wells of unconfined to semi-confined aquifers in the northern, eastern, and central Salinas Valley (Figure 11). In the Kings, Kaweah, and Tule River groundwater sub-basins of Fresno and Kings County, and in the Eastside and Forebay sub-basins of Monterey County, one-third of domestic or irrigation wells exceed the nitrate MCL. Consistent with these findings, the maximum nitrate level, measured in any given land section (1 square mile) for which nitrate data exist between 2000 and 2009, exceeds the MCL across wide portions of these areas (Figure 12). Low nitrate concentrations tend to occur in the deeper, confined aquifer in the western and central Tulare Lake Basin (Boyle et al. 2012).

Nitrate levels have not always been this high. While no significant trend is observed in some areas with low nitrate (e.g., areas of the western TLB), USGS research indicates significant long-term increases in the higher-nitrate areas of the Tulare Lake Basin (Burow et al. 2008), which is consistent with the CASTING dataset. Average nitrate concentrations in public supply wells of the Tulare Lake Basin and Salinas Valley have increased by 2.5 mg/L (±0.9 mg/L) per decade over the past three decades. Average trends of similar magnitude are observed in private wells. As a result, the number of wells with nitrate above background levels (>9 mg/L) has steadily increased over the past half century from one-third of wells in the 1950s to nearly two-thirds of wells in the 2000s (Figure 13). Due to the large increase in the number of wells tested across agencies and programs, the overall fraction of sampled wells exceeding the MCL grew significantly in the 2000s (Boyle et al. 2012).

The increase in groundwater nitrate concentration measured in domestic wells, irrigation wells, and public supply wells lags significantly behind the actual time of nitrate discharge from the land surface. The lag is due, first, to travel time between the land surface or bottom of the root zone and the water table, which ranges from less than 1 year in areas with shallow water table (<3 m [10 ft]) to several years or even decades where the water table is deep (>20 m [70 ft]). High water recharge rates shorten travel time to a deep water table, but in irrigated areas with high irrigation efficiency and low recharge rates, the transfer to a deep water table may take many decades.

[6] Unless noted otherwise, nitrate concentration is given in mg/L as nitrate (MCL = 45 mg/L).

Adminsitrative Record
Page 34197

Once nitrate is recharged to groundwater, additional travel times to shallow domestic wells are from a few years to several decades and one to several decades and even centuries for deeper production wells.

3.2 Cleanup of Groundwater: Groundwater Remediation

Groundwater remediation is the cleanup of contaminated groundwater to levels that comply with regulatory limits. In the pump-and-treat (PAT) approach, groundwater is extracted from wells, treated on the surface, and returned to the aquifer by injection wells or surface spreading basins. In-situ treatment approaches create subsurface conditions that aid degradation of contaminants underground. In-situ remediation is not appropriate for contaminants spread over large regions or resistant to degradation. Both remediation methods typically also require removal or reduction of contamination sources and long-term groundwater monitoring.

Table 6. 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*

Data Source†	Total # of Wells	Total # of Samples	Location of Wells	Type of Wells	Years 2000–2010					
					# of Wells TLB	# of Wells SV	TLB Median mg/L nitrate	SV Median mg/L nitrate	TLB % > MCL	SV % > MCL
CDPH	2,421	62,153	throughout study area	public supply wells	1,769	327	12	8	6%	5%
CVRWB DAIRY	6,459	11,300	dairies in TLB	domestic, irrigation, and monitoring wells	6,459	—	22	—	31%	—
DPR	71	814	eastern Fresno and Tulare Counties	domestic wells	71	—	40	—	45%	—
DWR	26	44	Westlands Water District	irrigation wells	28	—	1	—	0%	—
DWR Bulletin 130	685	2,862	throughout study area	irrigation, domestic, and public supply wells	—	—	—	—	—	—
ENVMON	537	2,601	throughout study area	monitoring wells	357	180	—	27	52%	44%
EPA	2,860	4,946	throughout study area	—	—	—	—	—	—	—
Fresno County	368	369	Fresno County	domestic wells	349	—	18	—	15%	—
GAMA	141	141	Tulare County	domestic wells	141	—	38	—	43%	—
Kern County	2,893	3,825	Kern County	Irrigation, domestic wells	361	—	5	—	7%	—

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Adminsitrative Record
Page 34198

Groundwater remediation is difficult and expensive (NRC 1994, 2000). Groundwater remediation is done only very locally (less than 1 km^2 [<0.5 mi^2] to often less than 2 ha [<5 ac]). Cleanup of contaminants over a wide region is not feasible, and would require many decades and considerable expense. The success rate for cleanup of widespread groundwater contaminants is very disappointing (NRC 1994, 2000).

Because of the difficulty and poor success rates of plume remediation, an approach known as monitored natural attenuation (MNA) has become popular. MNA involves letting natural biochemical transformations and dispersion reduce and dilute contamination below cleanup goals, while monitoring to confirm whether MNA is adequately protecting groundwater quality. However, this approach is effective only for contaminants that transform to relatively harmless byproducts. The combination of circumstances that would favor denitrification of nitrate is generally lacking in California's alluvial aquifer systems (Fogg et al. 1998; Boyle et al. 2012), so MNA does not seem to be an effective way of remediating nitrate-contaminated groundwater in the study area.

The total estimated volume of groundwater exceeding the nitrate MCL in the Tulare Lake Basin and Salinas Valley is 39.7 km^3 (32.2 million acre-feet, AF) and 4.2 km^3 (3.4 million AF), respectively, more than the total groundwater

Table 6. *Continued*

Data Source†	Total # of Wells	Total # of Samples	Location of Wells	Type of Wells	Years 2000–2010					
					# of Wells TLB	# of Wells SV	TLB Median mg/L nitrate	SV Median mg/L nitrate	TLB % > MCL	SV % > MCL
Monterey County, Reports	239	1,018	Monterey County	monitoring, irrigation wells	—	98	—	14	—	36%
Monterey County, Geospatial	388	1,574	Monterey County	local small systems wells	—	431	—	18	—	15%
Monterey County, Scanned	452	5,674	Monterey County	local small systems wells	—	427	—	17	—	14%
NWIS	1,028	2,151	—	miscellaneous	76	4	35	0	36%	0%
Tulare County	444	444	Tulare County	domestic wells	438	—	22	—	27%	—
Westlands Water District	48	77	Westlands Water District	irrigation wells	31	—	4	—	0%	—

Source: Boyle et al. 2012.

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

† Data sources: CDPH: public supply well database; CVRWB Dairy: Central Valley RWB Dairy General Order; DWR Bulletin 130: data reports from the 1960–1970s, 1985; ENVMON: SWRCB 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: SWRCB 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.

Adminsitrative Record
Page 34199

pumped from the project area aquifers between 2005 and 2010 (Table 7). This is a basin-scale groundwater cleanup problem. Annual costs of traditional remediation would be on the order of $13 to $30 billion (Dzurella et al. 2012; King et al. 2012). This explains why no attempt at remediation of a contaminated groundwater basin on the scale of the Tulare Lake Basin or Salinas Valley has ever been undertaken. Except for cleanup of hot-spot sites, traditional remediation for nitrate is not a promising option.

A more promising remediation approach is what we refer to as "pump-and-fertilize" (PAF) (Dzurella et al. 2012; King et al. 2012). This approach uses existing agricultural wells to remove nitrate-contaminated groundwater and "treat" the water by ensuring nitrate uptake into crops through proper nutrient management. A disadvantage of PAF is that many irrigation wells are drilled deep to maximize the pumping rate, but most high levels of nitrate contamination are seen at shallower depths. Shallower nitrate-contaminated groundwater is en route toward the deep intake screens of many of the irrigation wells (Viers et al. 2012). One option is to drill intermediate-depth irrigation wells to intercept contaminated groundwater before it penetrates farther into the deeper subsurface. The cost, energy, and management requirements of this approach would need to be carefully evaluated, as it requires the drilling and operation of many shallower wells with smaller capture zones and smaller pumping rates at each well. At a regional or sub-regional scale, it may be an innovative alternative, although decades of PAF operations would be needed together with large reductions in nitrate leachate from the surface.

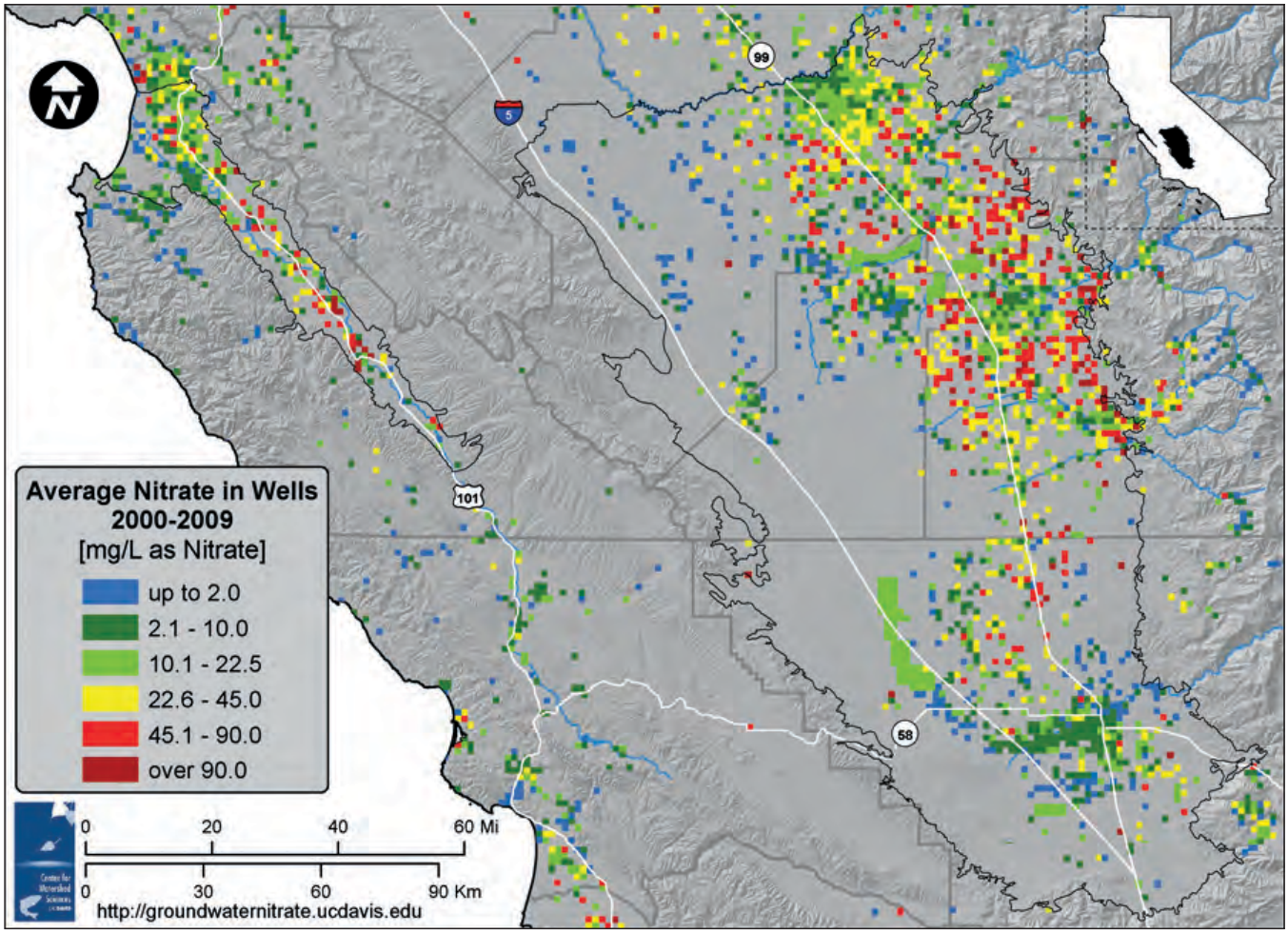

Figure 11. Mean of the time-average nitrate concentration (mg/L) in each well belonging within a square mile land section, 2000–2009. Some areas in the TLB are larger than 1 square mile. Source: Boyle et al. 2012.

Adminsitrative Record
Page 34200

Groundwater recharge operations could be managed to improve groundwater quality if the recharged water is of good quality and relatively low in nitrate (remediation by dilution). By introducing as much clean recharge water as possible, the long-term effects of contaminated agricultural recharge can be partially mitigated. But the large water volumes already affected would require decades of management.

Pump-and-fertilize along with improved groundwater recharge management are technically feasible, less costly alternatives than pump-and-treat and could help place regional groundwater quality on a more sustainable path. These alternatives should be accompanied by remediation of local nitrate contamination hot spots and long-term groundwater quality monitoring to track benefits of the strategy (for details, see King et al. 2012).

3.3 Existing Regulatory and Funding Programs for Nitrate Groundwater Contamination

Many regulatory and planning programs in the study area provide regulatory structure or technical and managerial support to water systems, communities, farmers, dairies, and others who deal with nitrate contamination in groundwater. Statutes also provide a regulatory framework for nitrate contamination of groundwater and drinking water. In the study area, there are several federal programs/statutes (Table 8a and Table 8b, blue), State programs/statutes (purple), and nongovernmental programs/agencies (orange) relevant to nitrate contamination and its effects on drinking water. Current regulatory/planning programs and statutes that have the ability to reduce groundwater nitrate contamination

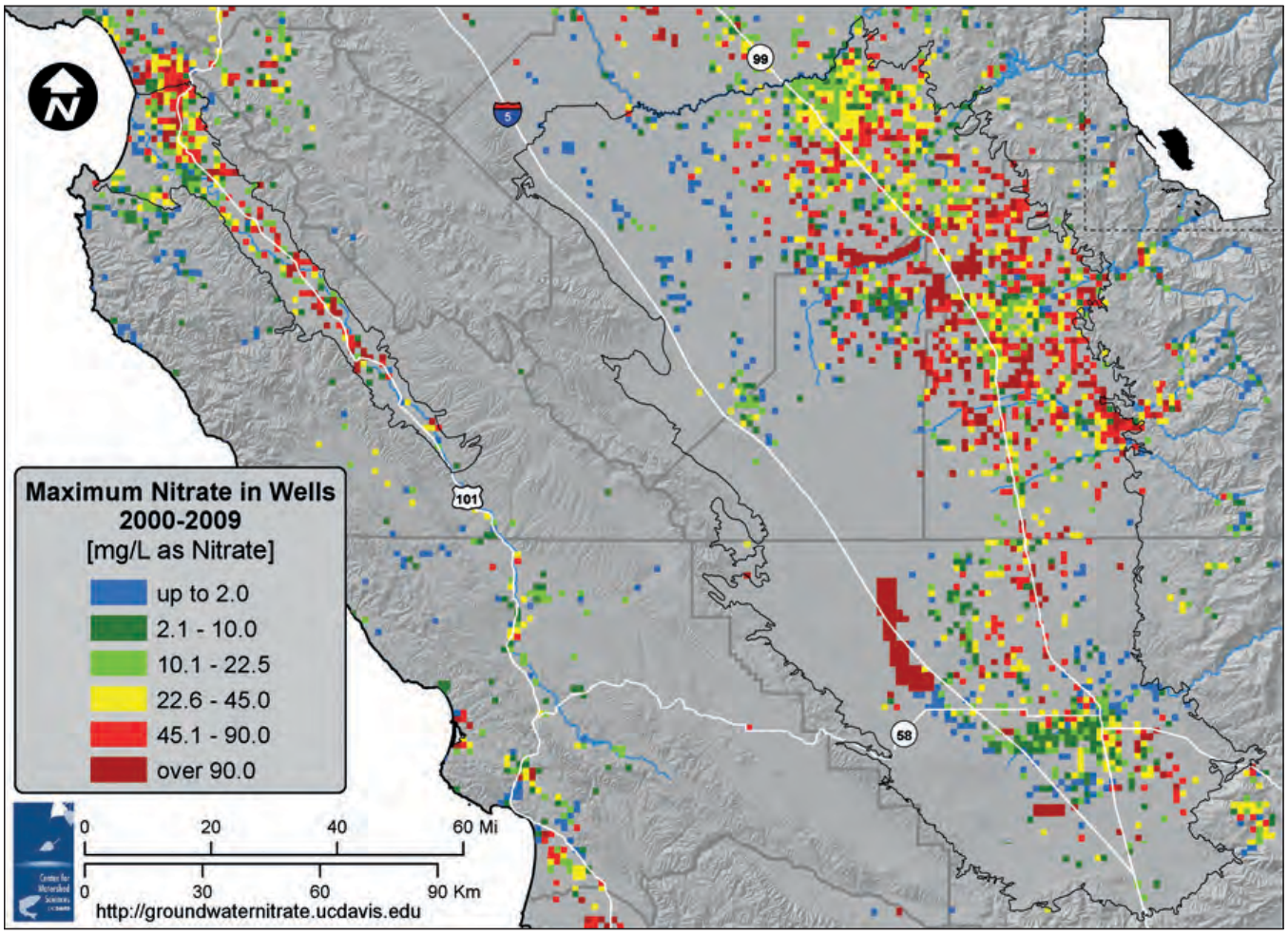

Figure 12. Maximum nitrate concentration (mg/L) measured at any time during 2000–2009 within a 1-square-mile land section. Some areas in the TLB are larger than 1 square mile. Source: Boyle et al. 2012.

Adminsitrative Record
Page 34201

are summarized in Table 8a. These programs/statutes have components that target nitrate source reduction or groundwater remediation. While providing a framework to address the groundwater nitrate issue, these programs have not been effective at preventing substantial nitrate contamination of groundwater used in drinking water supplies. Table 8b is a summary of current programs and statutes related to groundwater nitrate and drinking water. These provide for data collection, information, and education on nitrate sources and groundwater nitrate. Some of these programs regulate nitrate in drinking water.

In addition, several state, federal, and local agencies, as well as nongovernmental organizations, have established funding programs related to nitrate contamination in California's groundwater. A summary of existing funding sources to address problems related to nitrate in drinking water is shown in Table 9. In general, these programs are structured to provide assistance for activities related to alternative water supplies and nitrate load reduction. The State of California has eighteen relevant funding programs, administered by four agencies (Table 9, purple); the federal government manages an additional three funding programs (blue). Three large nongovernmental drinking water funding programs in the study area are highlighted in orange in Table 9. For a more detailed review, see Canada et al. (2012).

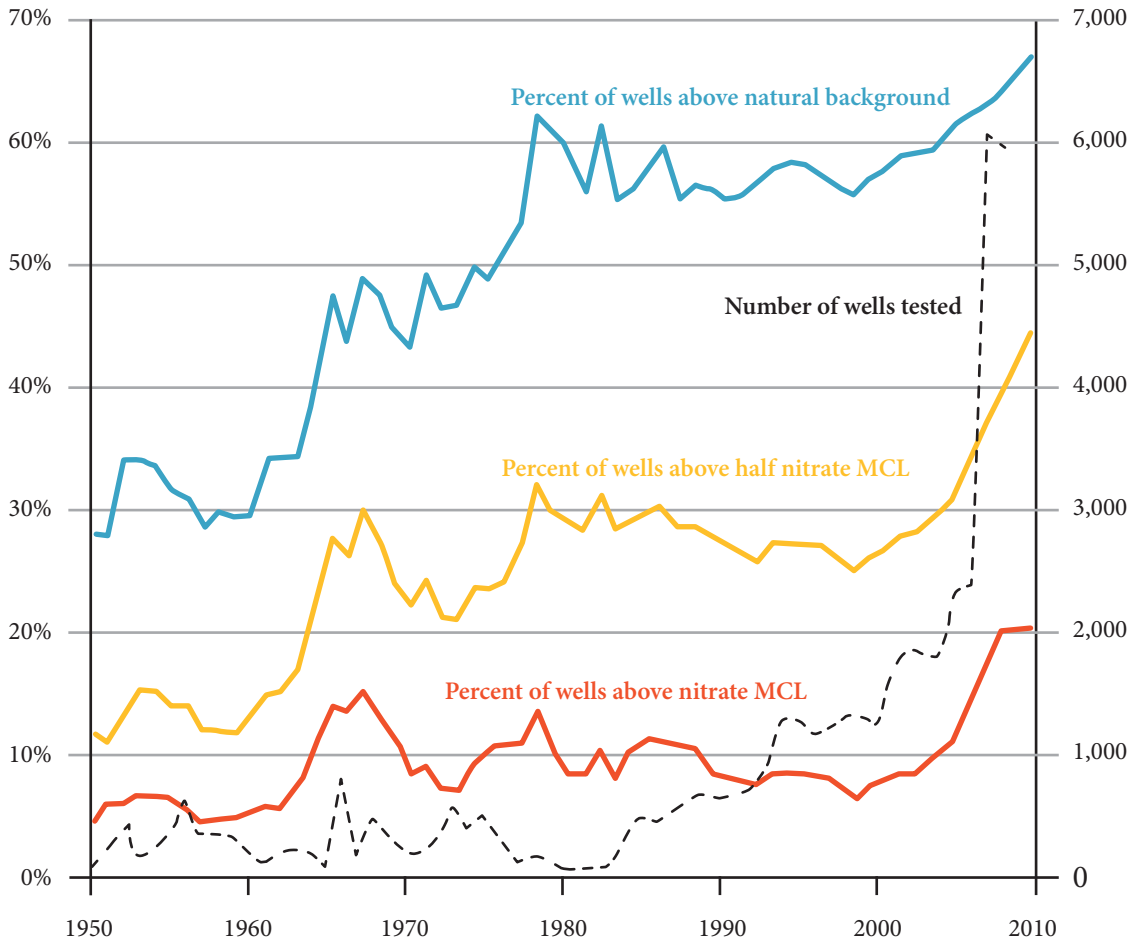

Figure 13. Five-year moving average of the percentage of wells for which the average annual measured concentration exceeded 9 mg/L (background), 22.5 mg/L (half of the MCL), and 45 mg/L (MCL) in any given year. Since the 1990s, an increasing number of wells other than public supply wells have been tested. In 2007, Central Valley dairies began testing their domestic and irrigation wells on an annual basis. Source: Boyle et al. 2012.

Adminsitrative Record
Page 34202

Table 7. Total groundwater volume* and estimated remediation volume by sub-basin

Sub-Basin	Total Groundwater Volume in Study Area km^3 [million AF]	Remediation Volume > MCL km^3 (% of total)	Remediation Volume > MCL million AF (% of total)
Tulare Lake Basin			
5-22.06–Madera	1.48 [1.2]	0.15 (10%)	0.12 (10%)
5-22.07–Delta-Mendota	3.21 [2.6]	0.16 (5%)	0.13 (5%)
5-22.08–Kings	115 [93]	12.75 (11%)	10.34 (11%)
5-22.09–Westside	64 [52]	1.67 (3%)	1.35 (3%)
5-22.10–Pleasant Valley	4.9 [4.0]	1.11 (23%)	0.90 (23%)
5-22.11–Kaweah	42 [34]	9.12 (21%)	7.39 (21%)
5-22.12–Tulare Lake	46 [37]	4.65 (10%)	3.77 (10%)
5-22.13–Tule	41 [33]	4.29 (11%)	3.48 (11%)
5-22.14–Kern	49 [40]	5.81 (12%)	4.71 (12%)
TLB TOTAL	366 [297]	39.7 (11%)	32.2 (11%)
Salinas Valley			
3-4.01–180/400 Foot Aquifer	8.46 [6.86]	0.91 (11%)	0.74 (11%)
3-4.02–Eastside	3.16 [2.56]	1.23 (39%)	1.00 (39%)
3-4.04–Forebay	5.59 [4.53]	1.37 (25%)	1.11 (25%)
3-4.05–Upper Valley	3.03 [2.46]	0.56 (19%)	0.45 (19%)
3-4.08–Seaside	0.78 [0.63]	0.07 (10%)	0.06 (10%)
3-4.09–Langley	0.44 [0.36†]	0.04 (9%)	0.03 (9%)
3-4.10–Corral de Tierra	0.60 [0.49‡]	0.002 (0.5%)	0.002 (0.5%)
SV TOTAL	22.1 [17.9]	4.19 (19%)	3.4 (19%)
Study Area Total	**315 [255]**	**43.9 (11%)**	**35.6 (11%)**

Source: King et al. 2012.

*Source: DWR 2010.

†Storage; actual groundwater volume not listed.

‡Source: Montgomery Watson Americas 1997, not listed in DWR Bulletin 118.

Adminsitrative Record
Page 34203

Table 8a. Summary of programs and statutes for reducing nitrate contamination in groundwater

Agency	Program/Statute (year created/passed)	Goal/Purpose
U.S. Environmental Protection Agency (U.S. EPA)	Supplemental Environmental Programs (SEP) (1998)	Environmentally beneficial project that a violator of environmental laws may choose to perform (under an enforcement settlement) in addition to the actions required by law to correct the violation.
State Water Resources Control Board (State Water Board)	Porter-Cologne Water Quality Control Act (1969)	Grants the State Water Board authority over state water quality policy and aims to regulate activities in California to achieve the highest reasonable water quality.
	Recycled Water Policy (2009)	Resolution No. 2009-0011: Calls for development of salt and nutrient management plans and promotes recharge of clean storm water.
Regional Water Quality Control Boards	Cleanup and Abatement Order (CAO)	CA Water Code § 13304: Allows the Regional Water Board to issue a directive to a polluter to require clean up of waste discharged into waters of the state.
Central Coast Regional Water Quality Control Board	Irrigated Lands Regulatory Program (ILRP) (2004, draft in 2011)	*General Conditional Waiver of Waste Discharge Requirements, 3-Tiered Agricultural Regulatory Program (2004):* Groundwater quality monitoring required to different degrees based on discharger's tier. Draft (2001) requires Tier 3 dischargers with high nitrate loading to meet specified Nitrogen Mass Balance Ratios or implement a solution that leads to an equivalent nitrate load reduction.
Central Valley Regional Water Quality Control Board	Irrigated Lands Regulatory Program (ILRP) (2003, draft in 2011)	*Conditional Wavier of Waste Discharge Requirements of Discharges from Irrigated Lands:* Interim program to regulate irrigated lands. Does not address groundwater. *Recommended ILRP Framework (2011):* Development of new monitoring and regulatory requirements (includes groundwater).
	CV-SALTS (2006)	Planning effort to develop and implement a basin plan amendment for comprehensive salinity and nitrate management.
	Dairy Program (2007)	*Waste Discharge Requirements General Order for Existing Milk Cow Dairies:* Confined animal facilities must comply with set statewide water quality regulations, and existing milk cow dairies must conduct nutrient and groundwater monitoring plans.
California Department of Food and Agriculture (CDFA)	Feed, Fertilizer, Livestock, Drugs, Egg Quality Control Regulatory Services (FFLDERS)	Manages licenses, registration and inspection fees, and a mill fee levied on fertilizer sales, to fund research and educational projects that improve fertilizer practices and decrease environmental impacts from fertilizer use.

Adminsitrative Record
Page 34204

Table 8b. Summary of programs and statutes related to groundwater nitrate and drinking water (data collection, information, education, or regulation of drinking water)

Agency	Program/Statute (year created/passed)	Goal/Purpose
U.S. Environmental Protection Agency (U.S. EPA)	Safe Drinking Water Act (SDWA) (1974, 1986, 1996)	Mandates EPA to set the drinking water standards and to work with states, localities, and water systems to ensure that standards are met.
	Phase II Rule (1992)	Established federal maximum contaminant level (MCL) for nitrate in public water systems.
	Enforcement Response Policy—Enforcement Targeting Tool	Focuses on high-priority systems with health-based violations or with monitoring or reporting violations that can mask acute health-based violations.
U.S. Department of Agriculture (USDA)	Rural Utilities Service: National Drinking Water Clearinghouse (1977)	Provides technical assistance and educational materials to small and rural drinking water systems.
California Department of Public Health (CDPH)	22 CCR § 64431	Established state maximum contaminant level (MCL) for nitrate in public water systems.
	Drinking Water Source Assessment and Protection (DWSAP)	Evaluation of possible contaminating activities surrounding groundwater and surface water sources for drinking water.
	Expense Reimbursement Grant Program (EPG)	Education, training, and certification for small water system (serving < 3,301 people) operators.
	Groundwater Ambient Monitoring and Assessment (GAMA)	Improves statewide groundwater monitoring and increases availability of groundwater quality information. Funded by Prop 50 and special fund fees.
Assembly Bill 3030	(1993)	Permits local agencies to adopt programs to manage groundwater and requires all water suppliers overlying useable groundwater basins to develop groundwater management plans that include technical means for monitoring and improving groundwater quality.
Kern County Water Agency (KCWA)	(1961)	Collects, interprets, and distributes groundwater quality data in Kern County.
Monterey County Health Department		Implements a tiered, regular nitrate sampling program based on increasing nitrate concentration for local small water systems and for state small water systems.
Southern San Joaquin Valley Water Quality Coalition	(2002)	Protects and preserves water quality in the Tulare Lake Basin through surface water quality monitoring and dissemination of collected data. Particular focus is on agricultural discharge areas. Does not currently focus on groundwater.
Tulare County Water Commission	(2007)	Discusses water issues impacting Tulare County and advises the Tulare County Board of Supervisors. Special focus on nitrate in groundwater and improving drinking water in small communities.
Monterey County Water Resources Agency (MCWRA)	(1947)	Provides water quality management and protection through groundwater quality monitoring (including nitrate levels) and research and outreach efforts to growers to improve fertilizer management and reduce nitrate leaching.
The Waterkeeper Alliance	Monterey Coastkeeper (2007)	Collaborates with the State Water Board to ensure effective monitoring requirements for agricultural runoff and more stringent waste discharge requirements for other nitrate sources.
Rural Community Assistance Partnership (RCAP)	(1979)	Uses publications, training, conferences, and technical assistance to help communities of less than 10,000 people access safe drinking water, treat and dispose of wastewater, finance infrastructure projects, understand regulations, and manage water facilities.
National Rural Water Association (NRWA)	(1976)	Offers drinking water system technical advice (operation, management, finance, and governance) and advocates for small/rural systems to ensure regulations are appropriate.
California Rural Water Association	(1990)	Provides online classes, onsite training, low-cost educational publications, and other forms of technical advice for rural water and wastewater systems.
Self-Help Enterprises (SHE)	Community Development Program (1965)	Provides technical advice and some seed money to small/rural/poor communities for the planning studies and funding applications associated with drinking water system projects.
Community Water Center	Association of People United for Water (AGUA) (2006)	Advocates for regional solutions to chronic local water problems in the San Joaquin Valley. Focused on securing safe drinking water, particularly from nitrate-impacted sources.

Adminsitrative Record
Page 34205

Table 9. Summary of existing funding sources for water quality investigations and safe drinking water

Agency	Program (year passed or created)	Funding Provided (in millions of dollars)
California Department of Public Health (CDPH)	Safe Drinking Water State Revolving Fund (SDWSRF) (1996) (grants and loans)	**Generally $100–$150:** Low-interest loans and some grants to support water systems with technical, managerial, and financial development and infrastructure improvements.
	Proposition 84 (2006) (grants) **(fully allocated)**	**$180:** Small community improvements. **$60:** Protection and reduction of contamination of groundwater sources. **$10:** Emergency and urgent projects.
	Proposition 50 (2002) (grants) **(fully allocated)**	**$50:** Water security for drinking water systems. **$69:** Community treatment facilities and monitoring programs. **$105:** Matching funds for federal grants for public water system infrastructure improvements.
State Water Resources Control Board (State Water Board)	Clean Water State Revolving Fund (CWSRF) (1987) (loans)	**$200–$300 per year:** Water quality protection projects, wastewater treatment, nonpoint source contamination control, and watershed management.
	Small Community Wastewater Grants (2004, amended 2007) (grants)	**$86 (fees on the CWSRF):** Loan forgiveness to small disadvantaged communities and grants to nonprofits that provide technical assistance and training to these communities in wastewater management and preparation of project applications.
	Proposition 50 (2002) (grants) **(fully allocated)**	**$100:** Drinking water source protection, water contamination prevention, and water quality blending and exchange projects.
	Agricultural Drainage Program (1986) (loans) **(fully allocated)**	**$30:** Addressing treatment, storage, conveyance or disposal of agricultural drainage.
	Dairy Water Quality Grant Program (2005) (grants) **(fully allocated)**	**$5 (Prop 50):** Regional and on-farm dairy projects to address dairy water quality impacts.
	Nonpoint Source Implementation Program (2005) (grants)	**$5.5 per year:** Projects that reduce or prevent nonpoint source contamination to ground and surface waters.
	Cleanup and Abatement Account (2009)	**$9 in 2010:** Clean up or abate a condition of contamination affecting water quality.
	Integrated Regional Water Management (IRWM) (2002) (grants) **(fully allocated)**	**$380 (Prop 50):** Planning ($15) and implementation ($365) projects related to protecting and improving water quality, and other projects to ensure sustainable water use.

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Adminsitrative Record
Page 34206

Table 9. *Continued*

Agency	Program (year passed or created)	Funding Provided (in millions of dollars)
California Department of Water Resources (DWR)	Integrated Regional Water Management (IRWM) (2002) (grants)	**$500 remaining (Prop 84):** Regional water planning and implementation.
	Local Groundwater Assistance Grant (2008) (grants)	**$4.7 anticipated for 2011–2012 (Prop 84):** Groundwater studies, monitoring and management activities.
	Proposition 82 (1988) (loans)	**$22:** New local water supply feasibility and construction loans.
	Water Use Efficiency Grant Program (2001) (grants)	**$15 in 2011 (Prop 50):** Water use efficiency projects for agriculture, such as: wellhead rehabilitation, water and wastewater treatment, conjunctive use, water storage tanks.
	Agricultural Water Conservation Loan Program (2003) (loans)	**$28 (Prop 13):** Agricultural water conservation projects, such as: lining ditches, tailwater or spill recovery systems, and water use measurement.
	Infrastructure Rehabilitation Construction Grants (2001) (grants) **(fully allocated)**	**$57 (Prop 13):** Drinking water infrastructure rehabilitation and construction projects in poor communities.
California Infrastructure and Economic Development Bank (I-Bank)	Infrastructure State Revolving Fund (ISRF) (1994) (loans)	**$0.25 to $10 per project:** Construction or repair of publicly owned water supply, treatment, and distribution systems.
U.S. Department of Agriculture (USDA)	Rural Utilities Service—Water and Environmental Programs (RUS WEPs) (loans and grants)	**$15.5:** Development and rehabilitation of community public water systems (less than 10,000 people), including: emergency community water assistance grants, predevelopment planning grants, technical assistance, guaranteed loans, and a household well water program.
U.S. Department of Housing and Development (HUD)	Community Development Block Grant (CDBG) (grants)	**$500 in 2010 for CA:** Community development projects: feasibility studies, final plans and specs, site acquisition and construction, and grant administration.
U.S. Department of Commerce	Economic Development Administration (EDA) (grants)	**Grants up to 50% of project costs:** supports economic development, planning, and technical assistance for public works projects.
Rural Community Assistance Corporation (RCAC)	Drinking Water Technical Assistance and Training Services Project (loans)	**$1.2 per year:** Administers funds from the US EPA Office of Groundwater & Drinking Water for infrastructure projects, including water.
The Housing Assistance Council (HAC)	Small Water/Wastewater Fund (loans)	**Up to $0.25 per project:** Loans for land acquisition, site development, and construction.
Cooperative Bank (CoBank)	Water and Wastewater Loan (loans)	**$1 per project:** Water and wastewater infrastructure, system improvements, water right purchases, and system acquisitions. **$0.05–$0.5 per project:** Construction costs.

Source: Canada et al. 2012.

Adminsitrative Record
Page 34207

The Dutch Experience

In response to increasingly intensive animal production and a growing awareness of its effects on nitrate concentrations in surface water and groundwater, the European Council Nitrate Directive (ND) (Council Directive 91/67/EEC) was established in 1991 as part of the European Union (EU) Water Framework. The ND imposes a performance standard of 50 mg/L nitrate on effluent, groundwater, and surface water quality levels within all EU countries. Furthermore, each country is required to establish nitrate contamination reduction plans, monitor program effectiveness, and regularly report their findings to the European Council (EC) (EU Publications Office). Compliance with the ND is costly in terms of time, expertise, and money; however, countries that do not meet ND standards face large fines from the EC. While the ND does very little in the way of explicitly specifying how countries should act in efforts to comply with these requirements, plans that do not propose to regulate manure application at ND standards (i.e., land application rates in the range of 170–210 kg N/ha) have been historically rejected.

As an agricultural hotspot, The Netherlands has struggled to meet the ND requisites. To fulfill the obligatory ND requirements (Ondersteijn 2002), the Dutch government first created the Mineral Accounting System (MINAS) in 1998 (Henkens and Van Keulen 2001). MINAS was a farm-gate policy created to ensure the balance of nitrogen and phosphorus inputs (fertilizer and feed) and outputs (products and manure) on individual farms via balance sheets (Oenema et al. 2005). MINAS resembled a farm-gate performance standard that was enforced by a penalty tax for excess nitrogen and phosphorus inputs: farms consuming more nitrogen or phosphorus than could be accounted for via harvest outputs would be fined per kilogram of nitrogen or phosphorus lost to the environment. As of 2003, fines of € 2.27/kg N ($1.40/lb N) were enforced, more than seven times the cost of nitrogen fertilizer at the time. MINAS was popular for its simplicity, and was well supported by government aid. RIVM (Netherland's National Institute for Public Health and the Environment), which monitors nitrogen and phosphorus soil and water concentrations nationally, reports that nitrogen surpluses in agricultural areas fell substantially beginning in 1998 as a result of its implementation. Nevertheless, the EU declared the Dutch MINAS policy noncompliant with ND requirements, stating that the policy did not directly regulate water nitrate concentrations (Henkens and Van Keulen 2001).

In response to the EU's rejection of MINAS, the Netherlands implemented an additional policy in 2002: the Mineral Transfer Agreement System (MTAS). MTAS was a cap-and-trade system that prescribed manure (not inorganic fertilizer) application rates (as per ND objectives) and allowed farmers to purchase surplus application rights from those farmers applying manure to their land below legal limits. Rather than repealing MINAS, however, the Dutch increased enforceable fines under MINAS to serve as a safety net under the newly implemented MTAS (Ondersteijn 2002). Following the enactment of MTAS, water nitrate levels continued to fall at pre-MTAS rates (Henkens and Van Keulen 2001; Ondersteijn 2002; Berentsen and Tiessink 2003; Helming and Reinhard 2009), suggesting that the implementation of MTAS in addition to MINAS had little or no additional effect.

Given the apparent futility of MTAS, and following the repeated rejection of MINAS by the European court of justice in 2003, both MTAS and MINAS were abandoned by the Dutch government by 2006. The two competing regulations were replaced by a composite policy that enforces nitrogen as well as phosphorus application standards for both manure and inorganic fertilizer, thereby satisfying both ND standards and the unique challenges encountered in Dutch territory, while minimizing administrative and economic costs. The composite policy remains in effect to date.

Adminsitrative Record
Page 34208

4 Impact: Drinking Water Contamination

About 2.6 million people in the Tulare Lake Basin and Salinas Valley rely on groundwater for drinking water. This section estimates the population susceptible to nitrate contamination of groundwater, identifies safe drinking water actions available and the most promising options to address nitrate groundwater contamination, and estimates the total cost of nitrate contamination to communities and households in these areas. This discussion summarizes more detailed examinations by Jensen et al. (2012) and Honeycutt et al. (2012).

4.1 Susceptible Populations

Groundwater nitrate contamination brings two forms of susceptibility: public health risks and the economic costs of avoiding such risks through treatment, source reduction, remediation, or alternative water supplies. California's Tulare Lake Basin and Salinas Valley are particularly susceptible to public health and financial risks from nitrate contamination for the following reasons (Honeycutt et al. 2012).

- Communities in this region are unusually dependent on groundwater. Less than 3% of the area's population is served by surface water alone.
- These areas have more and larger nitrate contamination sources than most other parts of California (Viers et al. 2012).
- Of the region's 402 community public and state-documented state small water systems, 275 are very small (15–500 connections) and 58 are small (501–3,300 connections) (Figure 14). Small and very small systems are about 81% of Tulare Lake Basin water systems (serving 89,125 people, 4% of the population) and about 89% of the Salinas Valley water systems (serving 23,215 people, 6% of the population).
- Many of these small systems rely on a single well, without emergency alternatives when contamination is detected. These small water systems are inherently less reliable and face higher per capita expenses to address nitrate contamination of groundwater.
- Roughly 10.5% and 2.6% of the populations of Tulare Lake Basin and Salinas Valley, respectively, use unregulated, unmonitored domestic wells, serving 245,000 people from 74,000 wells (Figure 15).
- The area has many poor communities that cannot afford drinking water treatment or capital-intensive alternative water supplies. Over 17% of the Tulare Lake Basin and 10% of the Monterey County population lives in poverty.

We estimated the population of these basins that is susceptible to significant financial cost and public health concerns from nitrate contamination in groundwater (Honeycutt et al. 2012). The drinking water source (groundwater well or surface water), history of nitrate contamination, size, and potential for contamination were considered for each water system and self-supplied rural household well location in this region. "Vulnerability" describes the intrinsic potential for a system to deliver drinking water to users with high nitrate levels based on the type of system and based on the number of water sources within the system. Vulnerability is scored as follows:

- Lower vulnerability is assigned to community public water systems (water systems with >15 connections) having more than one water source (i.e., more than one well), regardless of whether they treat their water to remove nitrate.
- Higher vulnerability is assigned to all other water systems: community public water systems with a single source (one well) and state small (5–14 connections), local small (2–4 connections), and household self-supplied water systems (domestic well).
- No vulnerability to nitrate groundwater contamination is assigned to water systems solely supplied by surface water.

Susceptible water users could be harmed by consuming drinking water containing contaminants or by the costs for avoiding such contamination. We define "susceptible population" as those

- served by a water system with multiple sources (wells) that has reported at least one delivered water nitrate MCL exceedance in the past 5 years, or
- served by a water system with a single source (well) that has reported at least one raw water nitrate MCL exceedance in the past 5 years, or

Adminsitrative Record
Page 34209

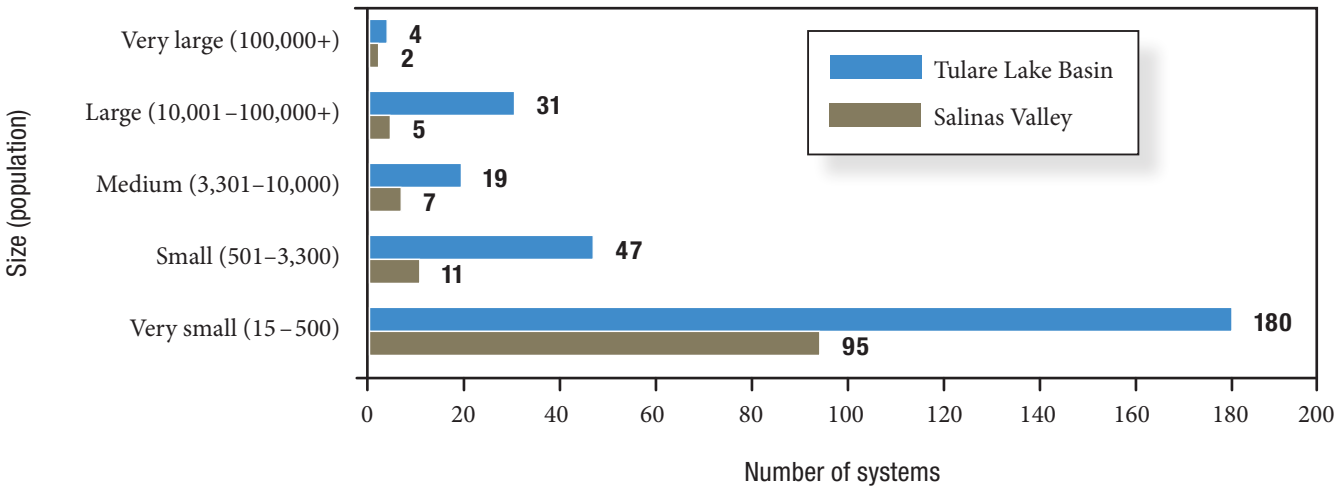

Figure 14. Community public and state-documented state small water systems of the Tulare Lake Basin and Salinas Valley. Source: CDPH 2010.

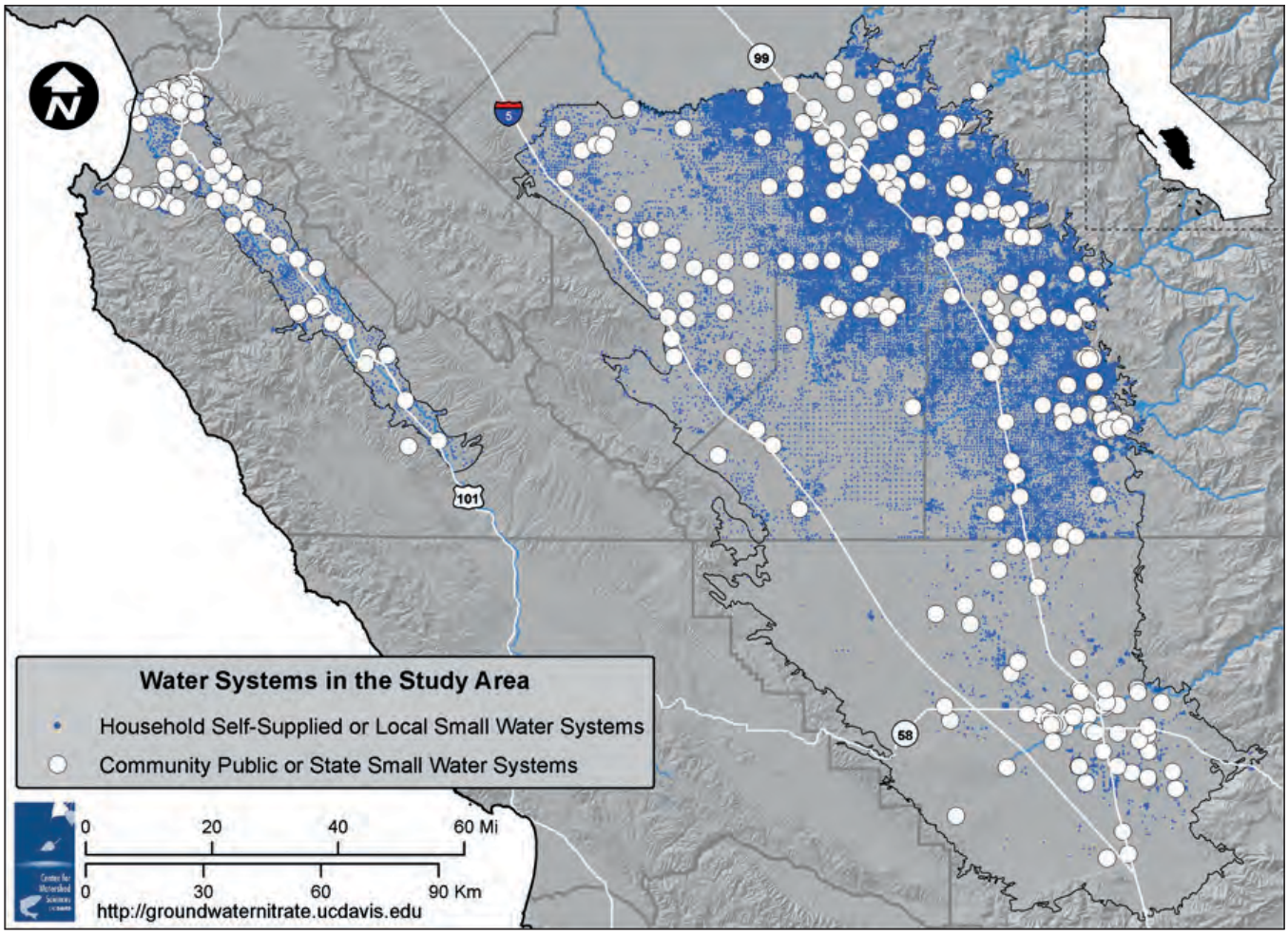

Figure 15. Estimated locations of the area's roughly 400 regulated community public and state-documented state small water systems and of 74,000 unregulated self-supplied water systems. Source: Honeycutt et al. 2012; CDPH PICME 2010.

Adminsitrative Record
Page 34210

- relying on domestic wells or local small water systems (fewer than 5 connections) in an area where shallow groundwater (<300 feet) has exceeded the nitrate MCL in the past (1989–2010), based on data from the UC Davis CASTING dataset (Boyle et al. 2012) or
- served by a water system lacking nitrate water quality data.

Figure 16 shows how these categorizations were used to classify populations and water systems. Of the 2.6 million people in the Tulare Lake Basin and Salinas Valley, 254,000 people have drinking water supplies susceptible to significant nitrate contamination. Of these, about 220,000 are connected to 85 community public or state small water systems with high or unknown susceptibility. For the majority of these systems, treatment will be expensive due to their small size (lack of economies of scale).

About 34,000 people are served by about 10,000 self-supplied household wells or local small water system wells at high risk for nitrate contamination given the known raw water quality exceedances in nearby wells (Figure 17). These systems are currently not regulated by the state or counties, and little public monitoring data exist for them.

Nine of 105 single-source small water systems in the study area exceeded the nitrate MCL at least once since 2006 and are not currently treating their water (CDPH 2010). Currently, 13 groundwater-supplied

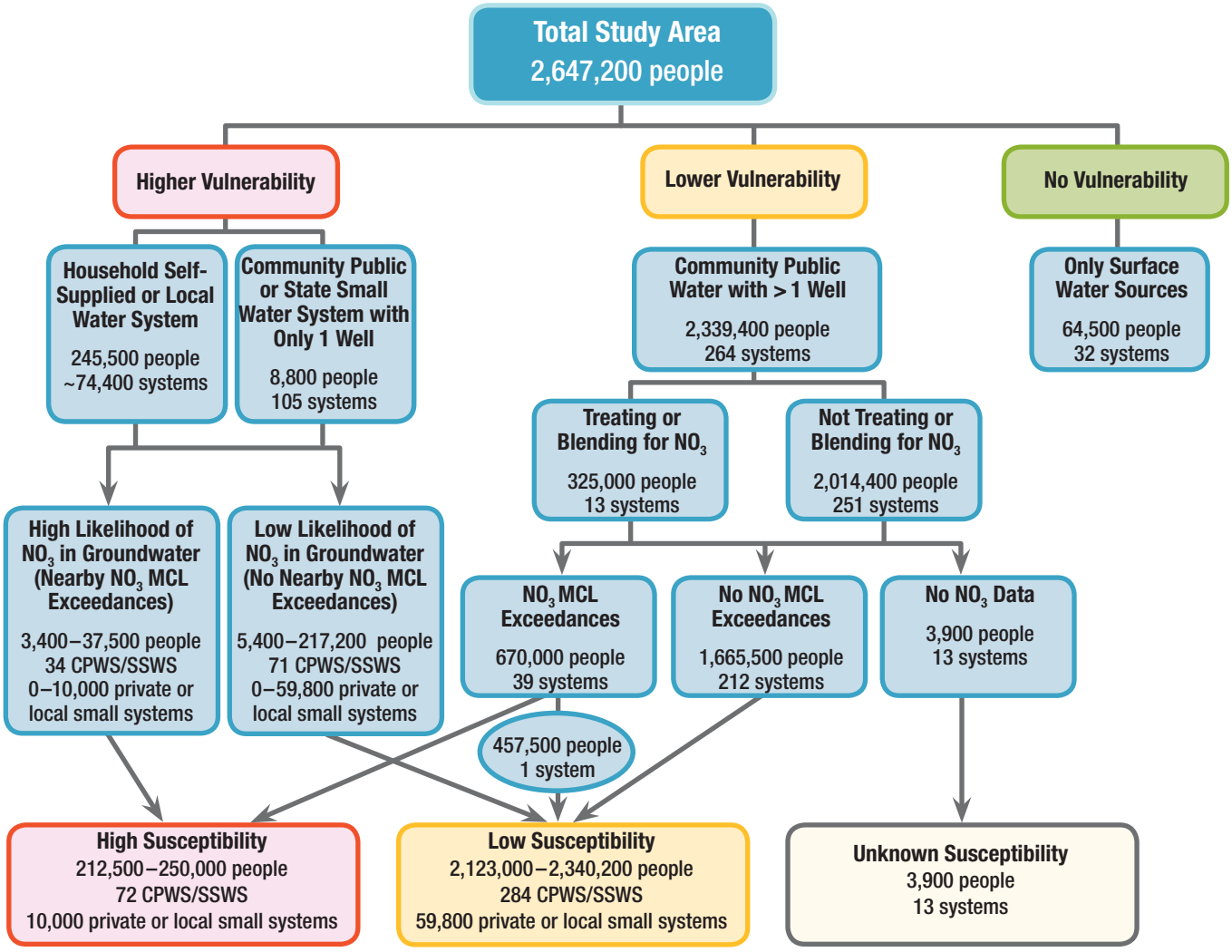

Figure 16. Classification of susceptible populations based on estimated vulnerability and water quality data for the study area. Due to different sources of data, the summation of the top row does not equal the total study area population. All population and connection information is approximate. CPWS: community public water system; SSWS: state small water system. Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34211

community public water systems and state small water systems treat for nitrate: 8 treat by blending and 5 by treatment processes (4 by ion exchange [IX] and 1 by reverse osmosis [RO]).

About 45% of the multiple-source systems that have delivered water exceeding the nitrate MCL serve severely disadvantaged and disadvantaged communities (SDACs and DACs) (Figure 18). DACs that are unincorporated, known as DUCs, often lack central water and sewer services. These DUCs are highly susceptible to nitrate contamination because they may lack a safe water source and are less financially able to resort to alternatives if their water source becomes contaminated. Since these areas have a large concentration of families with low incomes, community solutions to nitrate treatment or alternative water supply also might be difficult.

Over 2 million people in the study area are not classified as susceptible to a public health risk for nitrate contamination today. However, more than half of the study area population is considered to be at financial risk from nitrate contamination, having to potentially pay higher costs for treatment and monitoring because of regional groundwater contamination: A total of 1.3 million people (57%) in the area are served by community public water systems or state small water systems in which raw water sources have exceeded the nitrate MCL at least once between 2006 and 2010 (Figure 1 and Table 10). This includes over 457,000 people in the City of Fresno, which has nitrate exceedances in some wells but is taking measures to avoid this contamination, including significant expansion of surface water use.

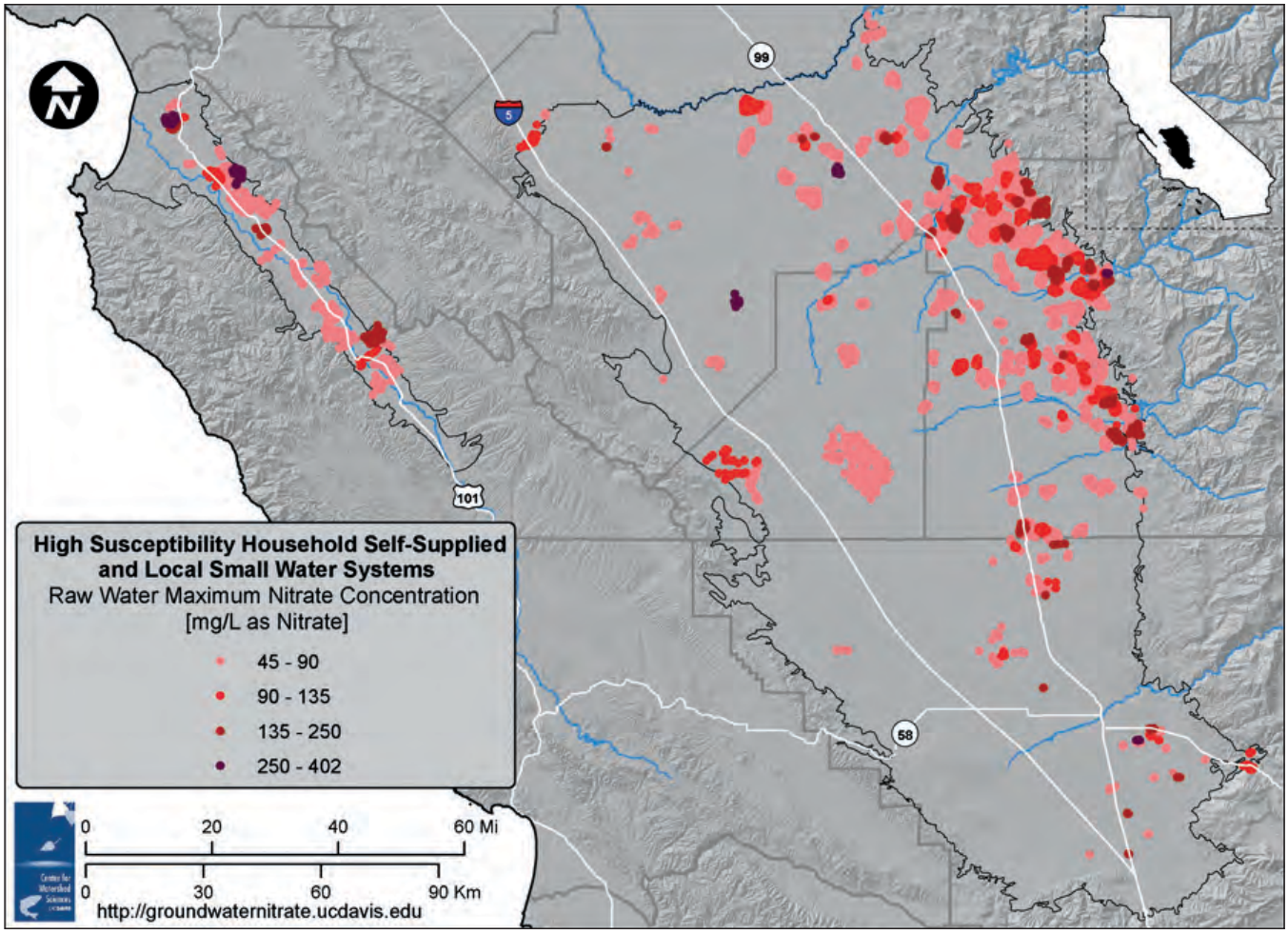

Figure 17. Household self-supplied and local small water systems located near wells having a maximum nitrate concentration value greater than the MCL. Source: 1989–2010 CASTING Database: GAMA, DWR, SWB, CDPH-CADWSAP, USGS, County Officials, Land Use Parcel Codes and DWR Land Use (see Honeycutt et al. 2012).

Adminsitrative Record
Page 34212

Severely disadvantaged communities (SDACs) are particularly vulnerable to financial costs. Of 51 community public water systems (serving about 714,000 people) in the study area with a raw source exceeding the nitrate MCL, most systems (40, serving about 379,000 people) are in a DAC. Thirteen of the 40 exceeding systems are in unincorporated areas (serving about 167,000 people), and 27 are in incorporated communities (serving about 212,000 people). They often cannot afford or organize and maintain capital-intensive solutions.

As past and current nitrogen applications migrate downward and through aquifers in the Tulare Lake Basin and Salinas Valley, populations susceptible to the costs and public health risks of nitrate contamination are likely to increase. Assuming unchanging and unabated basin-wide trends in CPWS raw nitrate groundwater levels since 1970, the financially susceptible population is estimated to increase from 57% currently to almost 80% or 1.9 million people by 2050 (not accounting for population growth, Table 10).

4.2 Alternative Water Supply and Treatment

Source reduction and aquifer remediation are insufficient to address drinking water nitrate contamination in the short- or near-term. In these cases, local water system authorities and users must select from a variety of treatment and alternative supply options. These options are summarized for community public water systems in Table 11 and for self-supplied

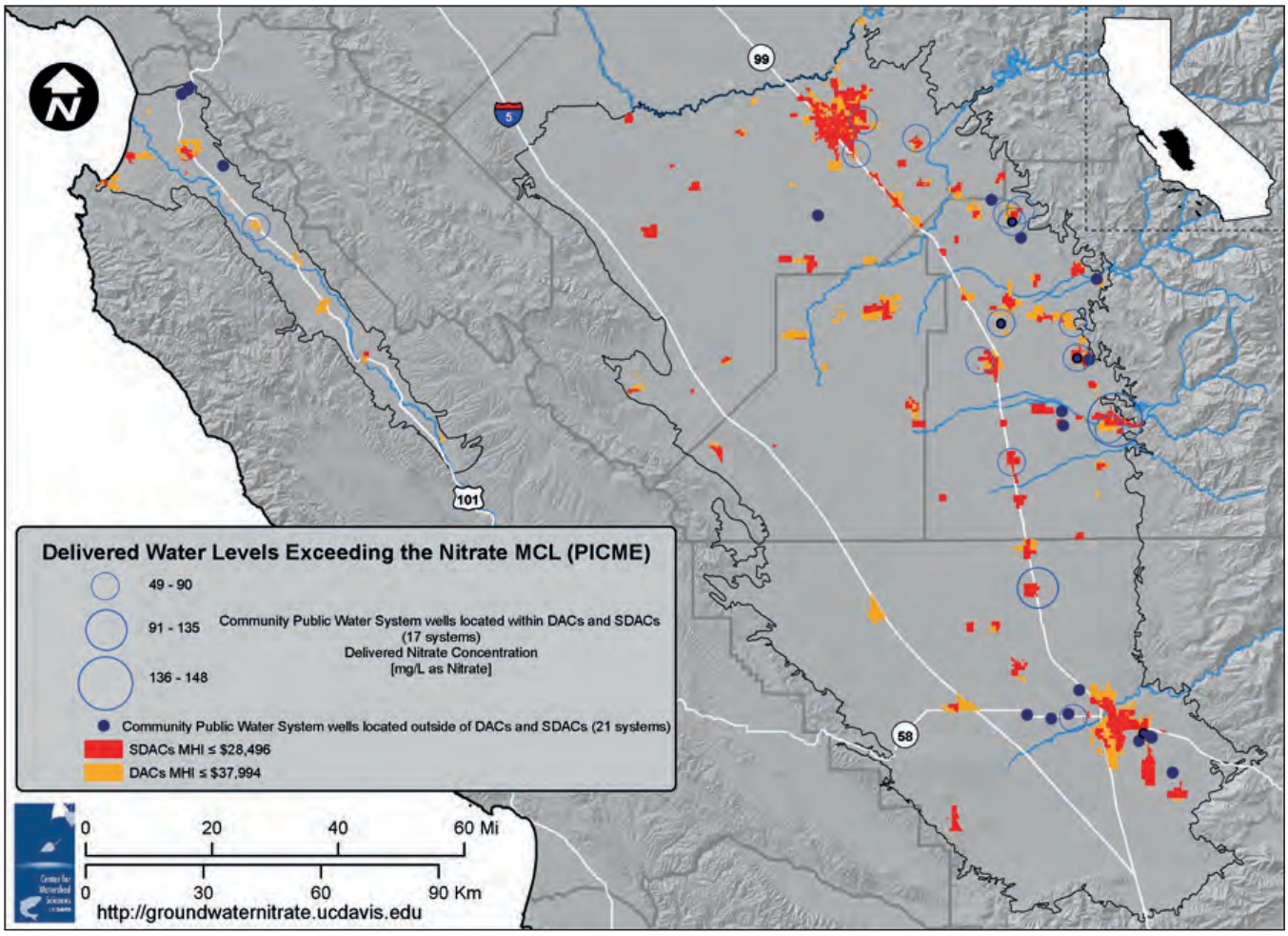

Figure 18. DACs, SDACs, and delivered water quality in multiple-source community public water systems. Source: CDPH PICME WQM 2006–2010; U.S. Census Bureau 2000, 2001 (see Honeycutt et al. 2012).

Adminsitrative Record
Page 34213

households and local small water systems in Table 12. This section further outlines these options (for details, see Honeycutt et al. 2012, and Jensen et al. 2012).

Community Public Water System Options

Each water system is unique, despite having many common problems and characteristics. No single solution will fit every community affected by nitrate in groundwater; each water system requires individual engineering and financial analysis.

The uniqueness of individual water systems is multiplied by the large number of small water systems in the Tulare Lake Basin and Salinas Valley. Small water systems have fewer and more expensive options per capita than do larger systems. They lack economies of scale and have fewer staff resources. Small water and wastewater systems also typically have disproportionately greater water quality and reliability problems and higher costs per capita (NRC 1997).

The options available for community public water systems faced with problems from nitrate contamination are summarized in Table 11. Blending is the most common approach to nitrate contamination for larger community public water systems with more than one water source. Water from the contaminated well is reduced, eliminated, or mixed with water from a safer water source. Eight community public water systems in the Tulare Lake Basin and Salinas Valley currently blend sources to comply with the nitrate MCL.[7]

Drilling a deeper or a new well is another common response to nitrate groundwater contamination. This approach can be cost-effective, but it is often only a temporary solution when nitrate contamination continues to spread locally and to deeper aquifers.

Treatment of community public water supplies is often explored and sometimes employed. A variety of treatment options are available (Jensen et al. 2012). Ion exchange and reverse osmosis are used for community public water system treatment in the basins. Additional treatment options, such as biological denitrification, may become economical and accepted in time (Jensen et al. 2012). However, treatment is expensive, especially for small systems. Under some circumstances, only a portion of extracted water is treated for nitrate because regulations can be met by blending treated water with water not treated for nitrate.

Management of waste concentrate or brine, by-products of ion exchange and reverse osmosis treatments, can also be costly. Options include discharge to a sewer or septic system, waste volume reduction using drying beds, trucking or piping for off-site disposal, deep well injection, and advanced treatment (Jensen et al. 2012).

Connecting to a larger system with reliable good-quality water can often solve many problems of small water systems, including nitrate contamination. This provides economies of scale in costs and greater access to expertise for resolving water system problems. However, connecting a small, often

Table 10. Estimated number of years until community public water supply (CPWS) sources exceed the nitrate MCL, and total affected population (not accounting for population growth)

Time for Maximum Recorded Raw Nitrate Level to Reach the MCL	Total Number of Affected CPWSs*	Total Affected Population*	Percent of Total CPWSs Population (study area)
0 years (2010)	77	1,363,700	57%
25 years (2035)	114	1,836,700	76%
40 years (2050)	127	1,903,300	79%

Source: Honeycutt et al. 2012.

* Based on raw water quality, not delivered quality susceptibility.

[7] Jensen et al. (2012) found a total of 23 water systems, including all types of water systems, in the study area that treat or blend to address the nitrate problem (10 blending systems, 10 IX systems, and 3 RO systems).

Adminsitrative Record
Page 34214

Table 11. Options for community public water systems

Option	Advantages	Disadvantages
Blending	• Simple nontreatment alternative. • Cost-effective, given suitable wells.	• Capital investment for accessing an alternative source. • Relies on availability and consistency of low-nitrate source. • Monitoring requirements. • Rising nitrate levels may preclude ability to blend.
Drilling a deeper or new well	• Potentially more reliable water supply. • Cheaper than bottled water for households using more than 8 gal/day.	• Potential decrease in source capacity. • Capital and operational costs increase with depth. • Potentially only a temporary quick fix; longevity depends on local hydrogeologic conditions and land use. • Risk of encountering other water quality concerns at greater depths (i.e., arsenic, manganese). • Pipeline costs if source area is far from original source.
Community treatment (IX, RO and EDR)	• Multiple contaminant removal. • Feasible, safe supply.	• Disposal of waste residuals (i.e., brine waste). • High maintenance and/or energy demands. • Resin or membrane susceptibility.
Piped connection to an existing system	• Safe, reliable water supply.	• Capital cost of pipe installation. • Connection fee. • Water rights purchase (surface water).
Piped connection to a new system	• Safe, reliable water supply.	• Capital cost of pipe installation. • High treatment system capital and O&M costs. • Water rights purchase (surface water).
Regionalization and consolidation	• Often lower costs.	• High capital and O&M costs.
Trucked water	• Community-wide distribution. • No start-up capital cost.	• Temporary “emergency” solution. • Not approved for new water systems.
Relocate households	• Safe, reliable water supply.	• Socially and politically difficult, extreme option. • Loss of property value and jobs. • Social, familial dislocation.
Well water quality testing (already in place)	• Water quality awareness. • Beneficial to blending.	
Dual system	• Hybrid of options. • Treating only potable.	• Possible consumption of contaminated source. • Cost of contaminated supply plus cost for POU system or trucked/bottled water, or capital dual plumbing costs.

Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34215

substandard system to a larger system often involves substantial initial capital costs to make the connection and to upgrade the smaller distribution system. Establishing connections also can pose institutional challenges (such as water rights and governance) and financial risks to the larger system.

Connecting several smaller systems into a new larger water system has many of the same advantages and costs of connecting small systems to an existing larger system. Establishing a new system also requires additional start-up costs for infrastructure and institutional development.

Institutional consolidation of several small systems avoids the costs of hydraulically connecting small systems, and it can provide a higher level of staff expertise and administrative economies of scale. This is attractive when systems are too small to merit full-time, trained staff and too scattered to economically connect their distribution systems and sources.

Trucking uncontaminated water to supply small communities allows the servicing of small scattered water systems, usually at a high cost. Trucking in water is generally seen as a temporary or emergency solution while a more permanent high-quality drinking water source is being developed.

Relocating households to a different area with better-quality water is an extreme approach that might be suitable if a small community is unviable for a variety of reasons and can not attract additional customer investments. Relocating households is likely to be accompanied by a loss of property values and local jobs, as well as social dislocation.

Two ancillary options that can supplement some of the above options are well water quality testing and the development of dual plumbing systems. Well water testing programs provide better and more timely information for awareness of nitrate contamination and can also provide useful information for blending. Dual plumbing systems separate potable from nonpotable water distribution systems, allowing a smaller quantity of contaminated water to be treated or conveyed from a higher-quality source for potable water uses.

The least expensive option is usually to stop using a nitrate-contaminated well and switch to another existing well, if a safer well is available. Similarly, many systems with more than one well blend water from a low-nitrate source or well with more contaminated supplies.

Self-Supplied Households and Local Small Water System Options

There are approximately 74,000 self-supplied households and local small water systems in the Tulare Lake Basin and Salinas Valley. Their nitrate contamination response options are summarized in Table 12 and discussed below.

Water supply options for self-supplied households and local small water systems are are similar to the options available to community public water systems, but are are similar to the options available to community public water systems, but are applied at a much smaller scale.

Drilling a deeper or new well can provide a reliable supply where better water quality exists. This option is costly, deeper wells can be accompanied by additional forms of contamination (such as arsenic), and new wells might provide only temporary relief if the nitrate plume is spreading deeper into the aquifer.

Treatment of household water supplies for nitrate is typically by reverse osmosis (RO). RO has advantages including the ability to remove multiple contaminants (where nitrate is not the only concern). However, household treatment does require some costs as well as additional burdens for maintenance, inspection, and operation of equipment. Treatment can be either point-of-entry (treating all household water use) or point-of-use (treating only potable water at household taps, usually the kitchen). As with centralized nitrate treatment, RO units create a concentrate or brine waste that requires disposal. Dilute waste streams, characteristic of RO, can sometimes be used for irrigation.

Connection to a larger system with more reliable water quality is a promising solution where a larger system is nearby. Such a connection often has a high cost, but it may provide a net economic benefit from lower long-term costs and delegation of many water quality concerns to qualified entities.

Trucking in water to the household or local small water system can be convenient and requires little start-up cost, but it is often expensive and is commonly considered to be a temporary solution. Bottled water use is similar to trucking in water, but it often entails a greater cost.

Households or local small water systems can relocate to avoid water quality problems, but this typically would involve some loss of property value. If the household or business is prosperous, relocation is unlikely. Poorer households are likely to feel any resultant loss of jobs or social dislocation more acutely.

Well water testing can better inform self-supplied users of their risks from nitrate contamination. These tests are not expensive. Dual plumbing systems can help reduce the amount

Adminsitrative Record
Page 34216

of water that is trucked in or treated, but it imposes additional costs and some risk of cross-connection of contaminated and safe water supplies.

Treatment to Remove Nitrate

Contaminated groundwater can be treated at a community treatment plant for all users, at the point-of entry-to residential or commercial buildings, or at the point of potable drinking water use (such as the kitchen sink). A variety of treatment options are available (Jensen et al. 2012). Ion exchange and reverse osmosis are used for community public water system treatment (Figures 19 and 20). RO is often used for point-of-use treatment in households and businesses. Additional treatment options, such as biological denitrification, may become economical and accepted (see Jensen et al. 2012). The effectiveness of treatment technologies across nitrate concentrations is summarized in Table 13.

Table 12. Options for self-supplied households and local small water systems

Option	Advantages	Disadvantages
Drilling a deeper or new well	• Potentially more reliable water supply. • Cheaper than bottled water for households using more than 8 gal/day.	• Potential decrease in source capacity. • Capital and operational costs increase with depth. • Potentially only a temporary quick fix; the nitrate plume follows groundwater movement. • Risk of encountering other water quality concerns at greater depths (i.e., arsenic, manganese). • Pipeline costs required if source area is far from original source.
Household treatment (RO)	• Multiple contaminant removal. • Low-nitrate water supply.	• Unless instructed, risk of improper handling or maintenance of equipment.
Regionalization and consolidation	• Cheaper treatment costs on a customer basis.	• High capital and O&M costs.
Trucked water	• Community-wide distribution. • No start-up capital cost.	• Temporary "emergency" solution. • Extra potable water storage required if a small community.
Bottled water	• Nitrate-free water supply. • No start-up cost.	• Inconvenience, monthly expenditure. • Temporary solution.
Relocate households	• Safe, reliable water supply.	• Unpleasant, extreme option. • Loss of property value and jobs. • Social, familial dislocation.
Well water quality testing	• Water quality awareness. • Beneficial to blending.	
Dual system	• Hybrid of options. • Treating only potable.	• Possible consumption of contaminated source. • Cost of contaminated supply plus cost for community treatment of potable supply and dual plumbing costs.

Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34217

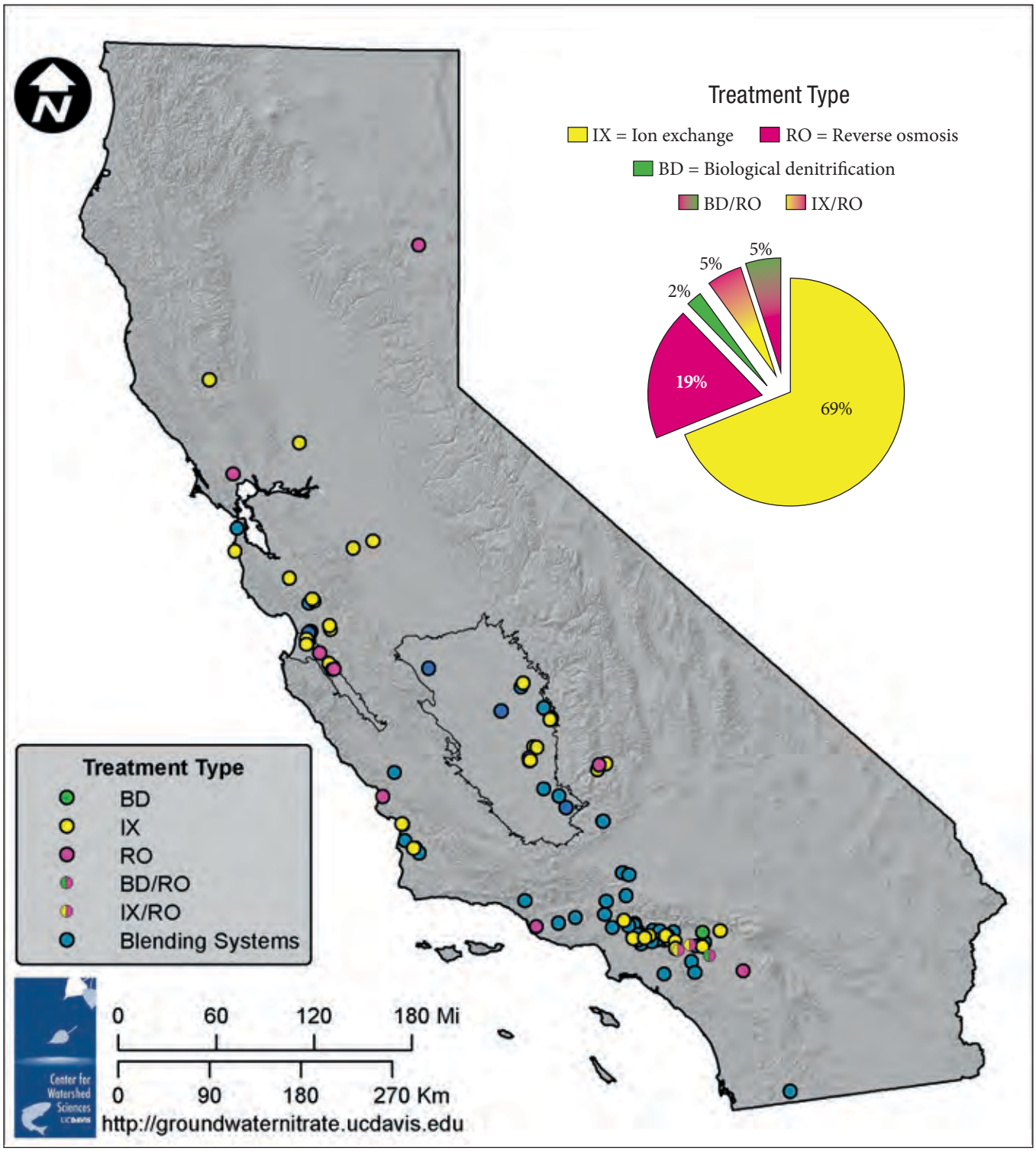

Figure 19. California drinking water systems treating or blending for nitrate, 2010. Source: Jensen et al. 2012.

Adminsitrative Record
Page 34218

However, treatment is expensive, especially for small systems. The development of treatment alternatives requires local engineering and development to accommodate local conditions. Nitrate contamination can be accompanied by other forms of groundwater contamination, including arsenic, magnesium, or pesticides, and treatment must accommodate the spectrum of water quality concerns as well as local water chemistry and distribution system conditions. Statewide, over 50% of nitrate treating systems utilize blending. Approximately 70% are using IX, and about 20% are using RO (Figure 19). In the Tulare Lake Basin and the Salinas Valley (Figure 20), 23 systems (of all types) were found to be treating and/or blending to address the nitrate problem (10 blending systems, 10 IX systems, and 3 RO systems).

Consolidation and Regionalization

Consolidation or regionalization of small systems is often suggested for addressing nitrate contamination and many other problems of small water systems. Although small systems are theoretically accountable and responsive to local customers, they often have diminished financial and technical resources that limit their ability to respond effectively or economically. Where a small system is near a larger system with superior water quality, connecting and consolidating these systems can provide a long-term remedy for the smaller system. Figure 21 shows the proximity of small systems (<10,000 people) in the Tulare Lake Basin and Salinas Valley to larger systems. Many small systems are reasonably close to potential long-term solutions.

However, the larger system may be concerned with financial and administrative burdens that may arise from upgrading the smaller system. Commonly, a smaller system must pay for the costs of connecting to a larger system as well as any distribution system upgrades needed to make the two systems compatible. This system upgrade burden on the financially weaker partner can require external financial assistance.

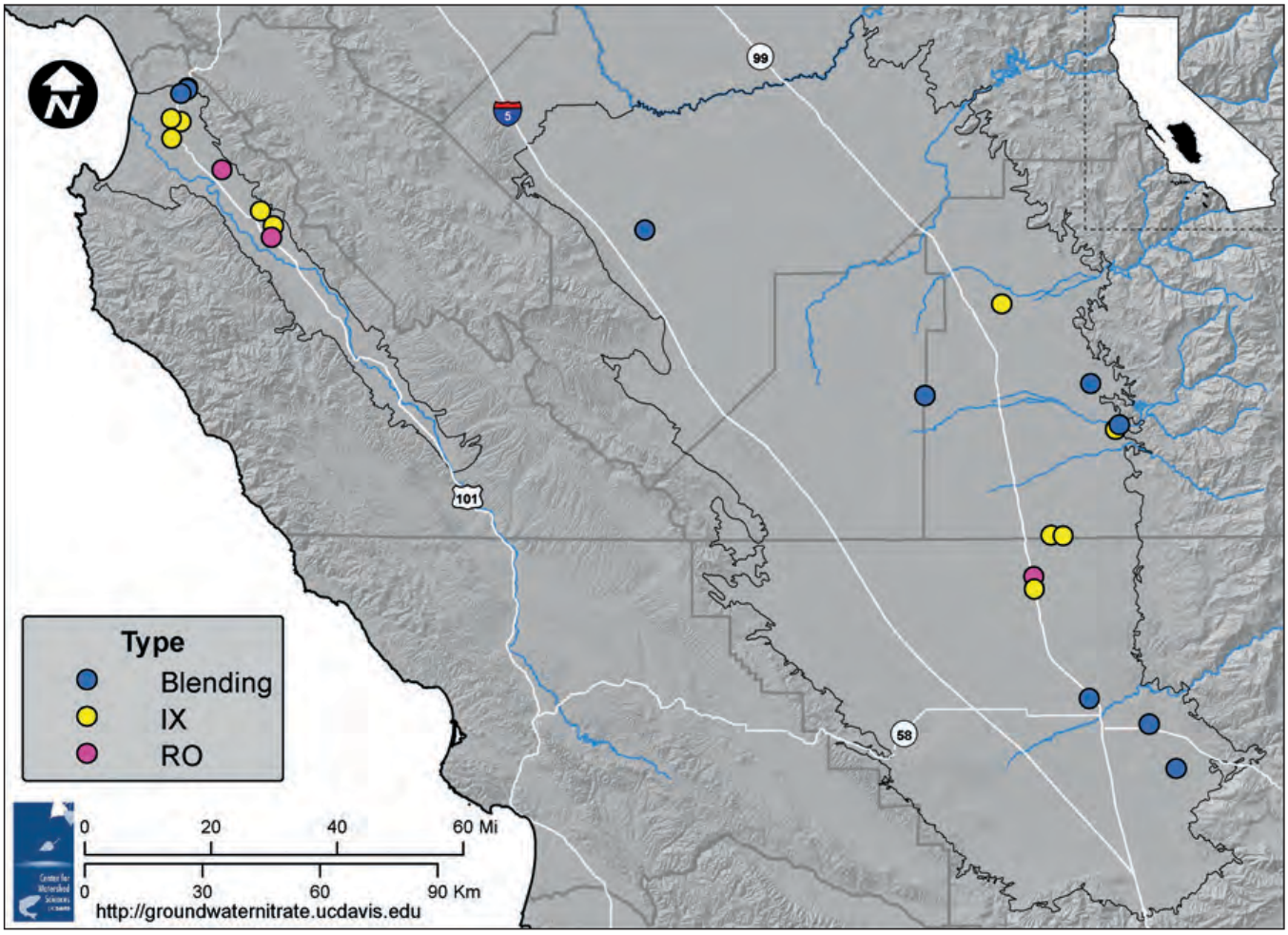

Figure 20. Utilities treating or blending for nitrate in the Salinas Valley and Tulare Lake Basin, 2010. Source: Jensen et al. 2012.

Adminsitrative Record
Page 34219

Many small systems are far from a larger system. For these cases, physical connection with a larger system is less financially attractive. However, even where systems remain hydraulically separated, consolidated operations, maintenance, and administration can sometimes have sufficient advantages to overcome financial barriers.

4.3 Comparison and Discussion

Economically promising and appropriate treatment and alternative water supply options have been identified (Honeycutt et al. 2012). These promising options give indications for state policy, and their costs are used to help estimate the overall cost of nitrate groundwater contamination in the Tulare Lake Basin and Salinas Valley.

Options for Small Community Public Water Systems

Estimated costs of options for community public water systems are compared in Table 14. Promising options for communities at risk of nitrate groundwater contamination are:

- **Consolidation to a larger system that can provide safe drinking water to more customers.** Although this option is viable for only a moderate number of systems, consolidation or regionalization of water systems can benefit a larger proportion of the vulnerable population and can help resolve many other long-term problems of small systems.
- **Consolidation of nearby small systems into a larger system** with a larger rate payer base and economies of scale. Even where small systems cannot economically connect to a large system, some opportunities exist to connect some small systems or to jointly manage several small systems to improve their overall financial condition.
- **Ion exchange treatment,** which is usually the most economical community treatment for groundwater contaminated by nitrate.
- **Interim point-of-use treatment or use of bottled water** until a more long-term and sustainable solution can be evaluated and implemented.
- **Blending of contaminated wells,** albeit temporarily if local nitrate contamination is expanding.

Table 13. Influence of nitrate concentration on treatment selection

Practical Nitrate Range	Option	Considerations
10–30% above MCL	blend	Depends on capacity and nitrate level of blending sources.
Up to 2× MCL	ion exchange	Depends on regeneration efficiency and costs of disposal and salt usage. Brine treatment, reuse, and recycling can improve feasibility at higher nitrate levels.
Up to many × MCL	reverse osmosis	Depends on availability of waste discharge options, energy use for pumping, and number of stages. May be more cost-effective than IX for addressing very high nitrate levels.
Up to many × MCL	biological denitrification	Depends on the supply of electron donor and optimal conditions for denitrifiers. Ability to operate in a start-stop mode has not yet been demonstrated in full-scale application; difficult to implement for single well systems. May be more cost-effective than IX for addressing high nitrate levels.

Source: Contact with vendors and environmental engineering consultants; Jensen et al. 2012.

Adminsitrative Record
Page 34220

A preliminary analysis was conducted to identify the short-term lowest-cost option for susceptible water systems in the project area to respond to nitrate contamination (Honeycutt et al. 2012). Results from this preliminary analysis, with and without point-of-use treatment for state small water systems, are summarized in Table 15 and Figure 22 (excluding POU). Due to public health and reliability concerns, point-of-use treatment is currently only allowed by CDPH as an interim action for very small water systems (serving <200 connections) facing nitrate pollution. In either case, drilling a new well appears to be the most economical solution for larger systems serving most of the susceptible population. In the long term, expanding nitrate contamination might reduce the viability of this option. If permanently allowed, point-of-use treatment for individual households would be economically preferred for most very small systems. Regionalization by connecting to a nearby larger system is attractive for a substantial minority of systems and about 10% of the susceptible population. The expense of groundwater treatment makes it relatively rare, but it remains important when other options are unavailable. Connection to surface water facilities was generally not found to be economical due to the high cost of surface water treatment facilities.

If expanding nitrate contamination precludes sustainable use of new wells, costs increase greatly for community public water systems to respond to nitrate contamination (Table 16). In this most constrained case, connecting to nearby larger systems (regionalization) is more common,

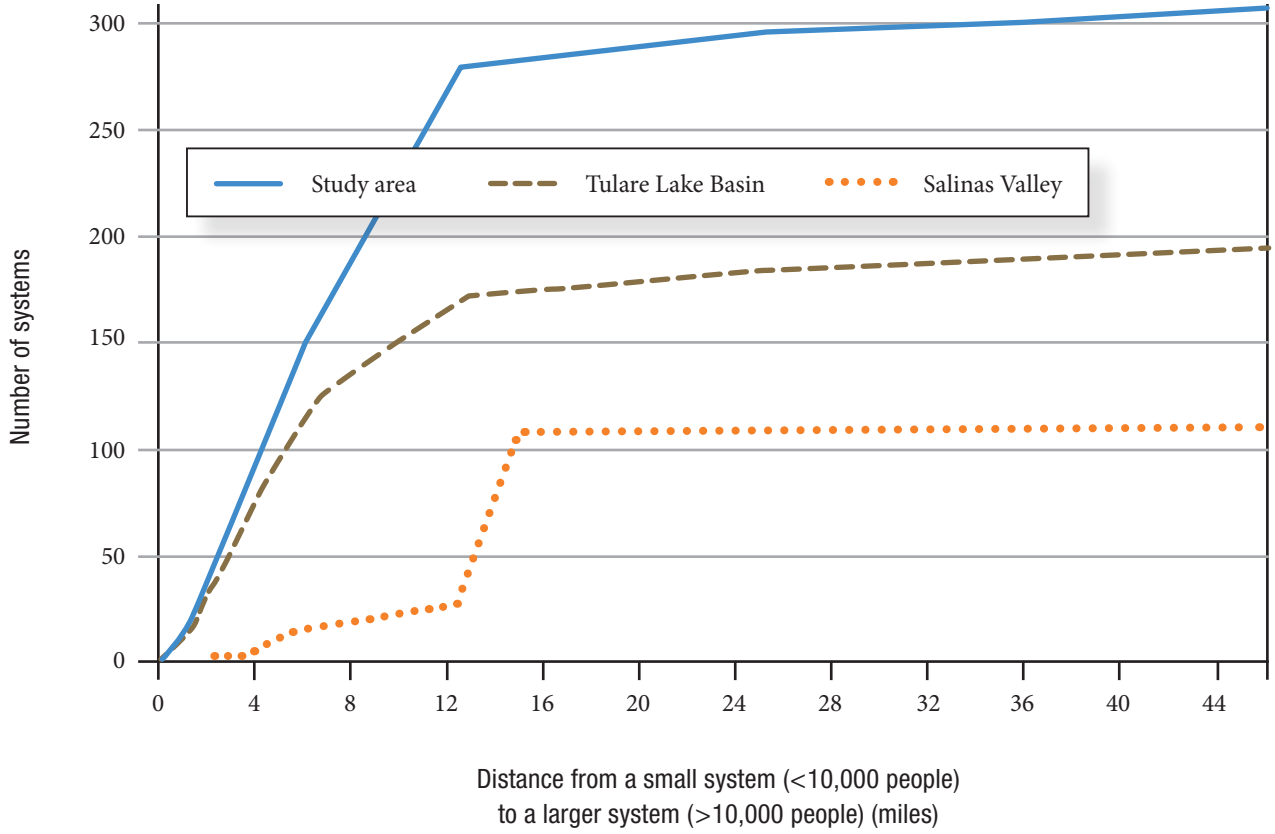

Figure 21. Cumulative distribution of the minimum distance from a small system (<10,000 people) to a larger system (>10,000 people) for the study area. Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34221

groundwater community treatment is common for small systems, and several of the largest systems (serving most of the susceptible population) switch to surface water treatment. The total estimated cost of alternative water supplies for susceptible community water systems more than doubles under this sustainable long-term scenario.

Options for Self-Supplied Households and Local Small Water Systems

Self-supplied and local small water systems have a smaller range of options (see Table 14). Point-of-use treatment is often the least-expensive option. Drilling a new well is sometimes more economical, where water use is greater and future nitrate contamination is less problematic.

Table 14. Safe drinking water option costs for self-supplied household and small community public water systems

Option	Estimated Annual Cost Range ($/year)	
	Self-Supplied Household	Small Water System (1,000 households)
Improve Existing Water Source		
Blending	N/A	$85,000–$150,000
Drill deeper well	$860–$3,300	$80,000–$100,000
Drill a new well	$2,100–$3,100	$40,000–$290,000
Community supply treatment	N/A	$135,000–$1,090,000
Household supply treatment	$250–$360	$223,000
Alternative Supplies		
Piped connection to an existing system	$52,400–$185,500	$59,700–$192,800
Trucked water	$950	$350,000
Bottled water	$1,339	$1.34 M
Relocate Households	$15,090	$15.1 M
Ancillary Activities		
Well water quality testing	$15–$50	N/A
Dual distribution system	$575–$1,580	$260,000–$900,000

Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34222

Table 15. Estimated cost of the lowest-cost short-term alternative water supply option for susceptible community public water systems and state small water systems based on system size and proximity to a larger system

Option	Number of Susceptible Water Systems		Population		Total Cost ($/year)	
	Including POU	Excluding POU	Including POU	Excluding POU	Including POU	Excluding POU
Drill new well	10	63	184,100	191,700	$10,144,000	$14,500,000
POU device for potable use	70	——	10,500	——	$1,320,000	——
Pipeline to a nearby large system (10,000+ system)	5	13	25,300	27,300	$865,000	$1,463,000
Groundwater treatment facility	0	9	0	900	$0	$450,000
Surface water treatment	0	0	0	0	$0	$0
Total	**85**	**85**	**219,900**	**219,900**	**$12,329,000**	**$16,413,000**

Source: Honeycutt et al. 2012.

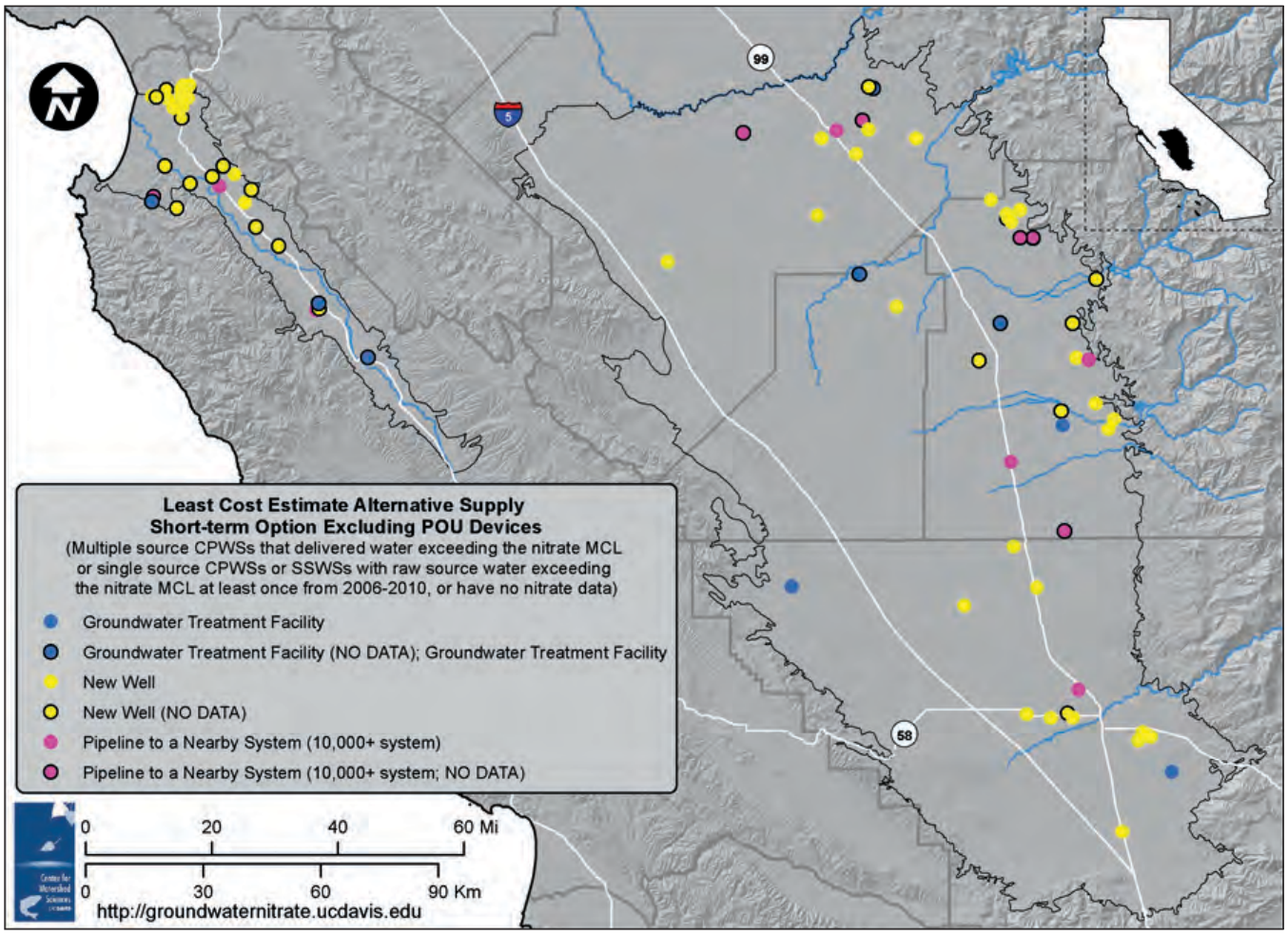

Figure 22. Lowest-cost alternative supply option (excluding POU systems) based on a high estimate of option costs for susceptible community public water systems and state small water systems (multiple source CPWS or SSWSs exceeding the nitrate MCL; or single-source CPWS or SSWSs exceeding the nitrate MCL at least once from 2006–2010; or those having no data). Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34223

4.4 Cost of Providing Safe Drinking Water

Roughly $12 to $17 million per year in additional costs in the near term will be needed to provide safe drinking water for people on community systems in the Tulare Lake Basin and Salinas Valley affected by nitrate contamination of groundwater (see Table 15). These costs are for 85 susceptible systems currently serving roughly 220,000 people. To provide safe drinking water for long-term solutions for these 85 systems will cost roughly $34 million per year if new wells are no longer sufficient. As additional systems become affected by nitrate contamination, these costs could increase.

The annualized additional cost of providing nitrate-compliant drinking water to the estimated 34,000 people (10,000 rural households) using domestic wells or local small water systems that are highly susceptible to current or future nitrate contamination is at least $2.5 million per year for point-of-use treatment for drinking purposes only. These costs could be lower if a manufacturing discount for bulk purchase of POU/POE systems were available. The lowest-cost POU option is used for all domestic well and local small water systems in the study area, estimated for both the short and long term. This does not include the cost of monitoring, public awareness, or regulatory programs to identify and reach out to this currently unregulated and unmonitored population.

The short-term cost to fund alternative water supplies for the highly susceptible nitrate-affected population amounts to $60 to $80 per susceptible person per year, $4 to $5 per irrigated acre per year for the 4 million acres of agriculture in these basins, or $75 to $100 per ton of fertilizer nitrogen (assuming about 200,000 tons of fertilizer nitrogen is applied in the study area). Allowing for only long-term, more viable, and sustainable alternative drinking water solutions for the affected population, the total cost amounts to $142 per susceptible person per year, $9 per irrigated acre per year, or $180 per ton of fertilizer in the long term.

Table 16. Estimated cost of the lowest-cost long-term alternative water supply options for susceptible community public water systems and state small water systems based on system size and proximity to a larger system

Option	Number of Susceptible CPWSs/SSWSs	Population	Total Cost ($/year)
Pipeline to a nearby system (10,000+ system)	29	36,600	$5,592,000
Groundwater treatment facility	51	8,000	$6,344,000
Surface water treatment facility	5	175,300	$21,532,000
Total	**85**	**219,900**	**$33,468,000**

Source: Honeycutt et al. 2012.

Adminsitrative Record
Page 34224

5 Policy Options for Nitrate Source Reduction and Funding

This section summarizes a range of policy options for reducing nitrate sources of contamination to groundwater and funding for resolving the problems of nitrate contamination. These options are drawn from the more detailed and extensive examination in Canada et al. (2012). Promising actions on future nitrate source reduction and funding options are discussed in Section 6.

5.1 Nitrate Source Reduction Policy Options

A wide range of policy options are available to reduce nitrate contamination to groundwater over time. We use four criteria for evaluating broad classes of regulatory options: the costs incurred by dischargers to reduce nitrate loading to achieve a nitrate standard (abatement costs), the costs of monitoring and enforcement, the information requirements, and the potential for raising revenues (for funding drinking water actions and other purposes related to nitrate contamination). These results are summarized in Table 17 and further described by Canada et al. (2012).

Specific technology mandates on farmers and agriculture will result in high per-unit costs for reducing nitrate contamination. Farming practices vary tremendously, even within these basins, so specific technology standards would be unlikely to be broadly effective or economical. Less-specific performance standards would provide more flexibility but still do not account for the variation in costs across farms. Nitrate or nitrogen fees or cap-and-trade approaches give farmers more flexibility to respond to required reductions in nitrate loading, thereby reducing the costs of nitrate abatement. If these actions are monitored and enforced based on nitrate leaching rates, much more costly and extensive on-site monitoring would be needed, whereas enforcement and accounting of fertilizer application requirements would be much less burdensome. Reducing nitrate leachate by imposing fees on nitrate or nitrogen has an added advantage of raising funds that may be used to compensate affected drinking water users. A cap-and-trade approach can also raise funds if nitrogen use permits are auctioned.

Hybrid options are also available to regulate nitrate. For nearly 15 years, the Netherlands has used a hybrid approach to manage nitrate (Kruitwagen et al. 2009; Ondersteijn et al. 2002). Under this system, agricultural sources are regulated using a performance standard combined with a fertilizer fee. (see "The Dutch Experience," p. 46). Hybrid regulations might be practical for managing nitrate leachate.

Information disclosure would have dischargers of nitrate or users of nitrogen make such information public. Water systems could also face more stringent water quality consumer reporting rules. Such disclosures should provide some motivation to reduce nitrate discharges.

Table 17. Summary of regulatory options to reduce nitrate contamination to groundwater

Regulatory Option	Abatement Costs	Monitoring and Enforcement Costs	Information Requirements	Revenue Raising
Technology mandate	high	Fertilizer application: low Nitrate leachate: high		no (unless fines)
Performance standard	medium			no (unless fines)
Fee	low			yes
Cap and trade	low			yes (if permits auctioned)
Information disclosure	medium	low	low	no (unless fines)
Liability rules	—	high	high	yes
Payment for water quality	low	low (if payment made to farmers) high (if payment made to state)	high	yes (if payment made to state)
De-designation of beneficial use	low	high	medium	no

Source: Canada et al. 2012.

Adminsitrative Record
Page 34225

Liability rules would make nitrate dischargers liable to users of drinking water and other groundwater users for the costs imposed by their discharges. If liability is established in courts, the costs could be quite high and may not necessarily result in much discharge reduction. Porter-Cologne Act Water Code Section 13304 might provide a useful framework.

Having water users or the state pay nitrate dischargers to reduce their dischargers ("payment for water quality") also has high transaction costs, without immediate effect to drinking water quality. But nitrate dischargers might find this an attractive long-term or preventive solution.

De-designating groundwater for drinking water use would shift all drinking water burdens to local water users. This would be administratively and politically awkward, acknowledging a permanent degradation to groundwater quality without compensating drinking water users.

Major Findings: Future Source Reduction Options

1. **Many options exist to regulate nitrate in groundwater, but there is no ideal solution.** The costs of regulatory options vary greatly, and while no option is perfect, some seem preferable to others.

2. **Regulating fertilizer application has lower monitoring and enforcement costs and information requirements than does regulating nitrate leachate, but it may be less effective in achieving nitrate reduction targets**. While the regulation of fertilizer application is easier to implement and enforce than the regulation of nitrate leachate, fertilizer regulation does not guarantee that water quality standards will be met. Due to nonuniform mixing, transport, and dispersion of nitrate in groundwater, it is difficult to quantify the impact of a unit of fertilizer on nitrate contamination of drinking water over time.

3. **Costs to farmers for reducing nitrate contamination can be lower with market-based regulations (fertilizer fees or cap-and-trade programs) than with technology mandates or prescriptive standards because of the additional flexibility farmers have in complying with market-based regulations.** Market-based instruments also encourage the development and adoption of new technologies to reduce fertilizer use, but they may lead to the formation of contamination hot spots.

4. **Well-defined and enforceable regulatory requirements are needed for liability rules to work.** In California, all groundwater is considered to be suitable, or potentially suitable, for municipal or domestic water supply and should be so designated by the Porter-Cologne Section 13304 which gives the California Water Boards authority to force polluters to pay for alternative water supplies for affected users of public water systems and private wells. Legislation might be useful to solidify Regional Board authority to apply this provision broadly.

5.2 Funding Options

Existing funding to address the costs of drinking water actions for communities and systems affected by nitrate contamination appears to be inadequate for many systems and largely requires drinking water users to bear the costs of groundwater contamination by others. The cost of nitrate contamination is felt disproportionately for small water systems (Honeycutt et al. 2012; Canada et al. 2012). Funding is also sparse for monitoring and for broad understanding of groundwater nitrate.

Many state, federal, and local programs exist to help fund local communities responding to nitrate contamination of their groundwater supplies, as discussed in Section 3 and Canada et al. (2012) and summarized in Table 9. Although current programs provide useful resources, they have been insufficient in addressing problems of nitrate groundwater contamination, particularly for smaller and poorer communities, who have less technical, managerial, and financial capacity for safe drinking water infrastructure and who are often ill-equipped for formal funding program applications.

A wide range of options is available to improve funding for drinking water supplies in areas affected by groundwater nitrate contamination, in addition to funding for nitrate source reduction and groundwater remediation activities. These options include state funding options summarized in Table 18 as well as traditional local water utility and tax options for funding water systems. These funding alternatives are addressed in greater depth by Canada et al. (2012). That examination and analysis led to the following findings for state funding and the promising options that are stated in Section 6.1(F).

Adminsitrative Record
Page 34226

Major Findings: Future Funding Options

1. **Many options exist to raise funds for safe drinking water and nitrate source reduction actions, but but all require that someone bear the cost, and many are awkward or insufficient.** Water use fees, groundwater pumping fees, bottled water fees, crop fees, and fertilizer fees are a few of the many potential sources for funding safe drinking water and source reduction actions.
2. **Some funding options give polluters a useful price signal.** Fertilizer (or nitrate leachate) fees and auctioned permits induce emitters to reduce fertilizer or nitrate use. Farmers do not pay sales tax on fertilizer in California.

Table 18. Summary of future state funding options

Option	Incentive to Reduce Nitrate	Who Pays	Example
Crop tax	no	producers and consumers of food	State Sales Tax Rate for Soft Drinks: The State of Maryland charges a 6% sales tax for soft drinks.
Fixed fee on drinking water agricultural water	no no	drinking water users agricultural users	Federal Communications Commission Universal Service Fee: A fixed fee placed on monthly phone bill to assure universal access to telecommunications for low-income and high-cost rural populations.
Volumetric fee on drinking water **agricultural water**	no low	drinking water users agricultural users	Gas Public Purpose Program Surcharge: A volumetric fee on gas bills in California to fund assistance programs for low-income gas customers, energy efficiency programs, and public-interest research.
Groundwater pumping fee	medium	agricultural groundwater users	Pajaro Valley Groundwater Pumping Fee: A per-acre-foot charge to secure financing for debt stabilization and to address groundwater overdraft.
Fee on bottled water	no	consumers of bottled water	California Redemption Value: A refundable fee placed on recyclable bottles at the point of sale.
Agricultural property tax	no	agricultural property owners	CA State Property Tax: A statewide ad valorem tax equal to a percentage of the purchase price is collected from all properties in the state, with some exceptions.
Fertilizer tax	high	consumers of fertilizer	Mill Assessment Program: The state imposes a fee of 2.1 cents per dollar on pesticide sales at the point of first sale into the state.
Nitrate leachate tax	highest	nitrate emitters	Duty on Wastewater: In the Netherlands, a tax of approximately $3.60 is imposed on each kilogram of nitrate in wastewater.
Cap and trade with auctioned permits	high/ highest	consumers of fertilizer and nitrate emitters	Title IV of the Clean Air Act Amendments: Established a tradable permit approach to control sulfur dioxide emissions. A small portion of permits sold in an auction.

Source: Canada et al. 2012.

Adminsitrative Record
Page 34227

Payment for Ecosystem Services in New York City

Currently, New York City participates in a payment for ecosystem services program for watershed protection. Under the U.S. Safe Drinking Water Act (SDWA), the city was required to meet the state water quality standards by either constructing a water filtration plant at an estimated cost of $6 billion in capital and $300 million in annual operating costs (Postel and Thompson 2005) or implementing a much less expensive watershed protection program. New York successfully requested a waiver from the SDWA filtration requirement and negotiated an agreement with upstream landowners and communities within the Catskill-Delaware watershed to establish a watershed protection plan. In 1997, a memorandum of agreement (MOA) was signed by state and federal officials, environmental organizations, and 70 watershed towns and villages to invest $1.5 billion over ten years to restore and protect the watershed (Postel and Thompson 2005). Program financing comes from bonds issued by the city and increases in residential water bills.

The program's fundamental activities include land acquisition; a program to manage and reduce agricultural runoff; a program for better forestry management; a program for enhanced stream management to reduce erosion and habitat degradation; improvements for wastewater infrastructure in the watershed; construction of an ultraviolet disinfection plant; and new regulation and enforcement of mechanisms to ensure continued water quality protection within the watershed (Postel and Thompson 2004). As of 2004, New York City has put $1 billion into the watershed protection program (Ward 2004). The negotiated partnership creates a watershed that provides high-quality drinking water, provides landowners with additional income, and improves recreational usage for nearby communities.

In this instance, negotiation or payment for ecosystem services led to the provision of safe drinking water at a lower cost than the default water filtration plant. By linking the ecosystem service providers with the beneficiaries, New York City successfully executed a comprehensive watershed protection program that delivers safe drinking water at a relatively low cost. New York City's watershed protection program is an example of a payment for ecosystem services program that guarantees the supply of high-quality drinking water and is financed via residential water bills and city bonds.

Adminsitrative Record
Page 34228

6 Promising Solutions

Many options are available to address the problems of drinking water quality, aquifer degradation, and economic costs from nitrate contamination of groundwater and its regulation. Of the many options available, some are more promising than others. But even among these promising options, major policy choices must be made.

6.1 Areas of Promising Action

Addressing groundwater nitrate contamination requires actions in four areas: (a) safe drinking water actions for affected areas, (b) reducing sources of nitrate contamination to groundwater, (c) monitoring and assessment of groundwater and drinking water, and (d) revenues to help fund solutions. Promising actions for legislative and state agency consideration in these areas appear below. Starred (*) actions do not appear to require legislative action, but might benefit from it. All actions are compared in Table 19.

Safe Drinking Water Actions (D)

Safe drinking water actions are the most effective and economical short- and long-term approach to address nitrate contamination problems in the Tulare Lake Basin and Salinas Valley. These actions apply especially to small and self-supplied household water systems, which face the greatest financial and public health problems from nitrate groundwater contamination.

D1: Point-of-Use (POU) Treatment. CDPH reports on how to make economical household and point-of-use treatment for nitrate contamination an available and permanent solution for small water systems.*

D2: Small Water System Task Force. CalEPA and CDPH convene an independently led Task Force on Small Water Systems that would report on problems and solutions of small water and wastewater systems statewide as well as the efficacy of various state, county, and federal programs to aid small water and wastewater systems. Many nitrate contamination problems are symptomatic of the broad problems of small water and wastewater systems.*

D3: Regional Consolidation. CDPH and counties provide more legal, technical, and funding support for preparing consolidation of small water systems with nearby larger systems and creating new, regional safe drinking water solutions for groups of small water systems, where cost-effective.*

D4: Domestic Well Testing. In areas identified as being at risk for nitrate contamination by the California Water Boards, as a public health requirement, CDPH (a) mandates periodic nitrate testing for private domestic wells and local and state small systems and (b) requires disclosure of recent well tests for nitrate contamination on sales of residential property. County health departments also might impose such requirements.

D5: Stable Small System Funds. CDPH receives more stable funding to help support capital and operation and maintenance costs for new, cost-effective, and sustainable safe drinking water solutions, particularly for disadvantaged communities.

Source Reduction Actions (S)

Reducing nitrate loading to groundwater is possible, sometimes at a modest expense. But nitrate source reduction works slowly and cannot effectively restore all affected aquifers to drinking water quality. Within the framework of Porter-Cologne, unless groundwater were to be de-designated as a drinking water source, reduction of nitrate loading to groundwater is required to improve long-term water quality. The following options seem most promising to reduce nitrate loading.

S1: Education and Research. California Department of Food and Agriculture (CDFA), in cooperation with the University of California and other organizations, develops and delivers a comprehensive educational and technical program to help farmers improve efficiency in nitrogen use (including manure) and reduce nitrate loading to groundwater. This could include a groundwater nitrate–focused element for the existing CDFA Fertilizer Research and Education Program (FREP), including "pump-and-fertilize" remediation and improved recharge options for groundwater cleanup.*

Adminsitrative Record
Page 34229

Table 19. Likely performance of promising state and agency actions for nitrate groundwater contamination

Action	Safe Drinking Water	Groundwater Degradation	Economic Cost
No Legislation Required			
Safe Drinking Water Actions			
D1: Point-of-Use Treatment Option for Small Systems +	◆◆		low
D2: Small Water Systems Task Force +	◆		low
D3: Regionalization and Consolidation of Small Systems +	◆◆		low
Source Reduction Actions			
S1: Nitrogen/Nitrate Education and Research +		◆◆◆	low–moderate
S2: Nitrogen Accounting Task Force +		◆◆	low
Monitoring and Assessment			
M1: Regional Boards Define Areas at Risk +	◆◆◆	◆◆◆	low
M2: CDPH Monitors At-Risk Population +	◆	◆	low
M3: Implement Nitrogen Use Reporting +		◆◆	low
M4: Groundwater Data Task Force +	◆	◆	low
M5: Groundwater Task Force +	◆	◆	low
Funding			
F1: Nitrogen Fertilizer Mill Fee		◆◆◆	low
F2: Local Compensation Agreements for Water +	◆◆	◆	moderate
New Legislation Required			
D4: Domestic Well Testing *	◆◆		low
D5: Stable Small System Funds	◆		moderate
Non-tax legislation could also strengthen and augment existing authority.			
Fiscal Legislation Required			
Source Reduction			
S3: Fertilizer Excise Fee	◆◆	◆	low
S4: Higher Fertilizer Fee in Areas at Risk	◆	◆	moderate
Funding Options			
F3: Fertilizer Excise Fee	◆◆	◆◆	moderate
F4: Water Use Fee	◆◆	◆◆	moderate

◆ Helpful

◆◆ Effective

◆◆◆ Essential

+ Legislation would strengthen.

* County health departments may have authority; CDPH requires legislation.

Adminsitrative Record
Page 34230

S2: Nitrogen Mass Accounting Task Force. CalEPA establishes a Task Force, including CDFA, to explore nitrogen mass balance accounting methods for regulating agricultural land uses in areas at risk for nitrate contamination, and to compare three long-term nitrogen source control approaches: (a) a cap-and-trade system; (b) farm-level nutrient management plans, standards, and penalties; and (c) nitrogen fertilizer fees.*

S3: Fertilizer Excise Fee. Significantly raising the cost of commercial fertilizer through a fee or excise tax would fund safe drinking water actions and monitoring and give further incentive to farmers for reducing nitrate contamination. An equivalent fee or excise tax could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

S4: Higher Fertilizer Fee in Areas at Risk. Areas declared to be at risk for nitrate contamination might be authorized to maintain a higher set of excise fees on nitrogen fertilizer applications (including synthetic fertilizer, manure, waste effluent, biosolids, and organic amendments), perhaps as part of a local safe drinking water compensation agreement.

Monitoring and Assessment (M)

Monitoring and assessment is needed to better assess the evolving nitrate pollution problem and the effectiveness of safe drinking water and nitrate source loading reduction actions. Such activities should be integrated with other state agricultural, environmental, and land use management, groundwater data, and assessment programs (source loading reduction actions), along with other drinking water, treatment, and wastewater management programs (safe drinking water actions).

M1: Define Areas at Risk. Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.*

M2: Monitor at-Risk Population. CDPH and the State Water Board, in coordination with DWR and CDFA, issue a report every 5 years to identify populations at risk of contaminated drinking water and to monitor long-term trends of the state's success in providing safe drinking water as a supplement to the California Water Plan Update.*

M3: Learn from Department of Pesticide Regulation Programs. CalEPA and CDFA examine successful DPR data collection, analysis, education, and enforcement programs for lessons in managing nitrogen and other agricultural contaminants, and consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.*

M4: Groundwater Data Task Force. CalEPA, in coordination with CalNRA and CDPH, convenes an independently led State Groundwater Data Task Force to examine the efficacy of current state and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.*

M5: Groundwater Task Force. CalEPA, CalNRA, and CDPH maintain a joint, permanent, and independently led State Groundwater Task Force to periodically assess and coordinate state technical and regulatory groundwater programs in terms of effectiveness at addressing California's groundwater quality and quantity problems. These reports would be incorporated into each California Water Plan Update.*

Funding (F)

Little effective action can occur without funding. Four funding options seem most promising, individually or in combination. State funding from fees on nitrogen or water use, which directly affect nitrate groundwater contamination, seem particularly promising and appropriate.

F1: Mill Fee. Increase the mill assessment rate on nitrogen fertilizer to the full authorized amount (CAL. FAC Code Section 14611). This would raise about $1 million/year statewide and is authorized for fertilizer use research and education.*

F2: Local Compensation Agreements. Regional Water Boards can require and arrange for local compensation of affected drinking water users under Porter-Cologne Section 13304. Strengthening existing authority, the Legislature could require that a Regional Water Board finding that an area is at risk of groundwater nitrate contamination for drinking water be accompanied by a cleanup and abatement order requiring overlying, current sources of nitrate to financially support safe drinking water actions acceptable to the local County Health Department. This might take the form of a local "liability district."*

F3: Fertilizer Excise Fee. Introduce a substantial fee on nitrogen fertilizer sales or use, statewide or regionally, to fund safe drinking water actions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs.

Adminsitrative Record
Page 34231

F4: Water Use Fee. A more comprehensive statewide fee on water use could support many beneficial activities. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities.

6.2 Developing an Effective Solution Strategy

Table 19 summarizes the required implementation levels and likely performance of promising actions identified above. Much can be done under existing authority and by existing agencies, although additional legislation could strengthen, augment, and further support these capabilities. While these actions include many helpful and effective solutions, none alone are sufficient to address the problems of groundwater nitrate contamination and the resulting drinking water problems. The most effective results will arise through a synergistic combination of major policy direction, legislation, and appropriate blends of of these identified actions.

Options without Fiscal Legislation

Without fiscal (tax, fee) legislation, there are several options to address drinking water or groundwater degradation, though each has a separate suite of choices. The most essential is having the Water Boards formally declare areas at risk for nitrate contamination. Such a declaration (M1) might entail a series of complementary actions, such as requiring domestic well testing in at-risk areas (D3), monitoring of at-risk populations (M2), and formation of a local compensation agreement or liability district for at-risk areas under Water Code Section 13304 (F2). Perhaps greater education and outreach to farmers in at-risk areas would also occur, along with discharger fees to fund safe drinking water actions to reduce nitrate discharges.

Porter-Cologne Act, Water Code Section 13304, states that "a cleanup and abatement order issued by the State Water Board or a regional Water Board may require the provision of, or payment for, uninterrupted replacement water service, which may include wellhead treatment, to each affected public water supplier or private well owner." This provides authority for the California Water Boards to require landowners contributing to nitrate in groundwater drinking water supplies to fund drinking water actions for affected public water supplies and private wells.

Using this authority, when a Regional Water Board establishes that an area is at risk for nitrate contamination of groundwater, it could simultaneously issue a cleanup and abatement order initiating a process for overlying landowners and contributors of nitrate to groundwater in that area to respond with an area drinking water compensation plan.

This process might involve requiring overlying landowners to support drinking water actions that comply with public health requirements established by the local County Health Department, including:

- an initial date by which groups of overlying landowners would submit a proposed area drinking water compensation plan for actions, implementation, and funding to the County Health Department;
- an intermediate date by which the appropriate Regional Water Board and County Health Department would approve such a plan, or one of their own, for overlying landowners to support drinking water actions; and
- a date by which any overlying landowner not complying with the area drinking water compensation plan would be required to cease and desist applications of nitrogen to overlying land exceeding a standard established by the Regional Water Board to protect drinking water users from nitrate pollution. This condition would apply to all overlying landowners if no alternative local compensation agreement drinking water action plan had been approved.

CDPH could issue suitable guidance to County Health Departments on establishing public health requirements.

County Health Departments would need to be empowered to collect fees from landowners pursuant to a drinking water action plan under a cleanup and abatement order. These fees would include the cost to the County Health Department of overseeing the drinking water action plan. Fees could be collected as part of annual county property tax assessments. This approach would provide a relatively organized and efficient means for landowners contributing nitrate to a contaminated aquifer to help decrease the additional costs incurred by drinking water users from nitrate contamination.

To protect public health, requiring testing of domestic wells in areas declared to be at risk of nitrate contamination seems prudent and in the public interest. Legislation seems needed to require such testing (perhaps periodically or on property sale), although perhaps this can be done by county

Adminsitrative Record
Page 34232

ordinance or administratively as a requirement to receive compensation under Water Code Section 13304.

Options Requiring Fiscal Legislation

Raising additional revenue to address nitrate issues seems to likely require legislation. The only exception is raising the small mill fee on fertilizer to its full authorized limit, which is approved for funding nitrogen use education and research activities.

Among these funding options, perhaps the most promising is to establish a statewide fee on the sale of nitrogen fertilizers, or a more administratively awkward fee on nitrogen use only in designated drinking water contamination risk areas. Such fees would act as both funding sources for safe drinking water actions and as an incentive to reduce nitrogen use, thereby somewhat reducing nitrate loading to groundwater. Partial rebates on these fees could be arranged for farmers who are involved in local area drinking water compensation plans or who have agreed to enforceable reductions in nitrate loads to groundwater.

6.3 Getting Organized

Many promising options are organizational. The management of nitrate groundwater contamination and its drinking water consequences is currently divided among several state agencies, each with historically derived authorities, purposes, and funding, as summarized in Section 3. In particular, the State and Regional Water Boards have the greatest authority under California's Porter-Cologne Act for groundwater quality. The California Department of Public Health and County Health Departments have authority over drinking water quality and public health. The California Department of Food and Agriculture has the greatest authority over fertilizer management and agricultural activities. The Department of Pesticide Regulation has no authority or direct interest in nitrate problems, but it has a successful, modern, integrated program for pesticide management, which may serve as a model for other forms of contamination, including nitrate. California's Department of Water Resources has overall water planning responsibility for the state, including oversight and funding authority for Integrated Regional Water Management Plans, and the State Water Board regulates water rights. The nitrate issues of the Tulare Lake Basin and Salinas Valley overlap several agencies. As environmental problems evolve beyond the origins of these agencies, there is often a need to evolve and coordinate the actions of different state and local agencies.

Nitrate contamination of groundwater is just one example of groundwater quality (and quantity) issues that many state agencies have in common. Each of the above agencies has its own groundwater monitoring, data, management, and often funding programs for groundwater overall or for individual groundwater quality or quantity concerns. Each of these agencies is facing, or will soon face, a range of similar and related groundwater problems regarding nitrate, pesticides, salts, and groundwater recharge and overdraft quantities.

Informational Actions

To help prepare the state to better address these problems, we propose several informational actions. Many informational actions could be triggered by requiring each of the California Water Boards to declare areas at risk of drinking water contamination from nitrate in groundwater (promising action M1). This finding is purely technical and seems well within the means of the Regional Water Boards, perhaps with some coordination from the State Water Board. A declaration of an area being at risk for nitrate groundwater contamination could also trigger several other informational actions. To protect public health, households and other very small water systems would be required to test drinking water wells for nitrate concentration upon sale and periodically thereafter (D4). Populations depending on groundwater in at-risk areas would also be reported to DWR for inclusion in state water planning efforts (M2). The "area at risk" designation could also serve to prioritize or trigger other funding, fee, education, monitoring, or regulatory actions.

Task Forces

We also propose four independently led task forces consisting of a core of agencies with overlapping interests. Having independent leadership would provide some assurance that each task force views the subject problem from more than just a collection of pre-existing agency perspectives.

- A task force on small water systems would seek to develop a common state policy for the problems of small water and wastewater systems in California. Small systems have inherent problems with higher costs, more precarious finance, and fewer technical and managerial resources, as they lack economies of scale. CDPH has long recognized these problems on the water supply side,

Adminsitrative Record
Page 34233

but there are likely to be benefits from addressing these local water and wastewater utility problems together.

- A task force on nitrogen mass accounting would explore the technical, economic, and institutional issues of having farms account for nitrogen and nitrate fluxes as a basis for regulation or fees. Currently, such detailed accounting is done for pesticides, air emissions, and dairy nitrogen, and it is being contemplated for salts and irrigation water. Having widespread and relatively detailed accounting for nitrogen would allow for some forms of economic management, such as cap and trade, and could also potentially support various educational and regulatory means of reducing nitrate loads to groundwater. This leads to a larger strategic question of whether the range of environmental emissions from agriculture should be accounted for separately by different agencies, gathered together in a single agency, or coordinated among separate agencies. Having a fragmented accounting system seems likely to increase costs and the regulatory burden, while reducing overall insight and understanding of environmental and agricultural problems. Accounting systems can be costly and time consuming for agencies and nitrogen users to administer.
- Two groundwater task forces are proposed. The first is in regard to groundwater data. A major difficulty in preparing this Report has been the fragmentation of groundwater data within and between agencies, as well as the lack of general access to groundwater data. Groundwater has become such an important issue that most agencies have their own groundwater activities. It is now critical that the state has a coherent and more forward-looking policy and technical capability for the collection and management of groundwater data. This issue seems sufficiently complex to call for a separate groundwater data task force.
- The many state interests and agencies involved with groundwater issues also seem to call for a periodic assessment of how effective these distributed programs are in practically addressing California's groundwater problems. This second independent groundwater task force would periodically review and report on the effectiveness of state groundwater activities to each California Water Plan.

6.4 Dilemmas for State Action

Groundwater nitrate contamination poses several overarching dilemmas and challenges for state policy, which will likely require broader discussions.

Local, statewide, or no compensation for pollution. In practice, the costs of pollution of drinking water sources are often borne by drinking water users. Some aspects of state policy (Water Code Section 13304) allow for fairly direct compensation for such costs. And general state support for water treatment also helps cover such costs. State general funds seem unlikely to be able to provide substantial support in the future, and many local communities, particularly small systems, are unlikely to have financial resources to cover such costs. Can the state establish a reasonable, relatively low-cost means to assess non-point source polluters for the drinking water (and perhaps other) costs entailed?

Degradation of groundwater. Current state law and policy does not allow degradation of groundwater quality to levels above water quality objectives defined in the applicable Basin Plan. However, no technological and institutional strategy has been found to economically reduce all nitrate discharges to levels that prevent further groundwater degradation. More modest approaches to reducing nitrate loads are likely to be economical. However, these more moderate reductions in nitrate loads would typically reduce the rate of groundwater degradation, but they would not always prevent degradation, particularly in the short term. If degradation is practically inevitable for some sources, how should state policy best oversee and regulate degradation?

Policy and policy implementation for environmental effects of land use. Both agriculture and urban land uses now face a host of environmental issues overseen by separate agencies and programs. The environmental causes and effects of nitrate contamination alone, for example, involve a diverse array of state agencies and programs. However, these same land uses also imply environmental impacts via pesticides, salinity, water use, air pollution, surface runoff, and endangered species. Many of these regulated (or potentially regulated) aspects interact environmentally, or their solutions have interactive effects and costs for land management. Is there a more effective and efficient policy approach to managing the environmental effects of land uses than mostly independent agencies and programs for each impact?

Adminsitrative Record
Page 34234

7 Conclusions

1. **Nitrate problems will likely worsen for decades.** For more than half a century, nitrate from fertilizer and animal waste have infiltrated into Tulare Lake Basin and Salinas Valley aquifers. Nitrate will spread and increase nitrate concentrations in many areas for decades to come, even if the amount of nitrate loading is significantly reduced. Most nitrate in drinking water wells today was applied to the surface decades ago.

2. **Agricultural fertilizers and animal waste applied to cropland are the two largest regional sources of nitrate in groundwater.** Although discharges from wastewater treatment plants, food processors, and septic tanks also contribute nitrate to groundwater and can be locally important, almost all of the regional groundwater nitrate contamination in the Tulare Lake Basin and Salinas Valley is from agricultural fertilizers and confined animal waste.

3. **Nitrate loading reductions are possible, some at modest cost. Large reductions of nitrate loads to groundwater can come at substantial economic cost.** Farm management is improving, but further improvements are necessary. While some are immediately achievable at modest cost, significant barriers exist, including logistical constraints and inadequate education. The cost of reducing nitrate loads to groundwater can be considerable for large reductions, especially on crops that require a substantial (much greater than 25%) decrease in nitrogen application from today's agronomically accepted, typical rates. Such dramatic reductions in fertilization rates without crop yield improvements can decrease net revenues by possibly several hundred million dollars per year within the study area.

4. **Direct remediation to remove nitrate from large groundwater basins is extremely costly and not technically feasible.** The volume of nitrate-contaminated groundwater is far larger than for urban contamination plumes. Standard pump-and-treat remediation to treat the groundwater underlying the Salinas Valley and Tulare Lake Basin would cost tens of billions of dollars. Instead, "pump-and-fertilize" and improved groundwater recharge management are less-costly long-term alternatives.

5. **Drinking water supply actions, such as blending, treatment, and alternative water supplies, are most cost-effective. Blending will become less available in many cases as nitrate pollution continues to spread.** Regardless of actions taken to reduce long-term nitrate loading to groundwater, many local communities in the Tulare Lake Basin and Salinas Valley will need to blend contaminated groundwater with cleaner water sources, treat contaminated well sources, or develop and employ safe alternative water supplies. Blending will become less available as an option in many cases as nitrate pollution continues to spread. The cost of alternative supplies and treatment for these basins is estimated at roughly $20 million to $36 million per year for the next 20 years or more.

6. **Many small communities cannot afford safe drinking water treatment and supply actions. High fixed costs affect small systems disproportionately.** Many small rural water systems and rural households affected by groundwater nitrate pollution are at or below the poverty level. Treatment and alternative supplies for small systems are more costly, as they lack economies of scale. Adherence to nitrate drinking water safety standards without substantial external funding or access to much less expensive treatment technology will potentially bankrupt many of these small systems and households.

7. **The most promising revenue source is a fee on nitrogen fertilizer use in these basins. A nitrogen fertilizer use fee could compensate affected small communities for mitigation expenses and effects of nitrate pollution. Under Water Code Section 13304, California Water Boards could also mandate that nitrate dischargers pay for alternative safe drinking water supplies.** Either mechanism would provide funds for small communities affected by nitrate pollution, allowing them to develop treatment or alternative water supplies that reduce the cost and effect of nitrate pollution over time.

Adminsitrative Record
Page 34235

8. **Inconsistency and inaccessibility of data from multiple sources prevent effective and continuous assessment. A statewide effort is needed to integrate diverse water-related data collection activities by various state and local agencies.** Throughout this study, we often faced insurmountable difficulties in gaining access to data already collected on groundwater and groundwater contamination by numerous local, state, and federal agencies. Inconsistencies in record keeping, labeling, and naming of well records make it difficult to combine information on the same well that exist in different databases or that were collected by different agencies. A statewide effort is needed to integrate diverse water-related data collection activities of various state and local agencies with a wide range of jurisdictions. Comprehensive integration, facilitation of data entry, and creation of clear protocols for providing confidentiality as needed are key characteristics of such an integrated database structure.

Adminsitrative Record
Page 34236

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Adminsitrative Record
Page 34237

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Adminsitrative Record
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Adminsitrative Record
Page 34240

The Tulare Lake Basin and the Salinas Valley, with 2.6 million inhabitants and home to nearly half of California's agricultural production, are the focus of this report. Nearly one in ten people in these two regions are currently at risk for nitrate contamination of their drinking water. Water systems providing water for half of these regions' population have encountered excessive nitrate levels in production wells at least once over the last five years.

An independent team of scientists at The University of California, Davis, was contracted by the State Water Resources Control Board to examine this problem. Working in consultation with an Interagency Task Force representing many state and local agencies, the authors undertake a uniquely broad and comprehensive assessment of the wide spectrum of technical, scientific, management, economic, planning, policy, and regulatory issues related to addressing nitrate in groundwater and drinking water for the Tulare Lake Basin and Salinas Valley.

This report identifies, describes, and quantifies past and current sources of nitrate, details the extent of groundwater nitrate contamination, and provides a comprehensive, up-to-date guide to the many options available to address the problems of drinking water quality, aquifer degradation, and economic costs from nitrate contamination of groundwater and its regulation. The report concludes by outlining promising actions in four key areas: safe drinking water actions for affected areas; reducing sources of nitrate contamination to groundwater; monitoring and assessment of groundwater and drinking water; and revenues to help fund solutions. Even among these promising options, major policy choices must be made. The research compiled in this report provides a foundation for informed discussion among the many stakeholders and the public about these policy choices.

The Center for Watershed Sciences at the University of California, Davis, brings a wide range of experts together to examine California's major water issues and problems. Its activities range from scientific and analytical modeling studies to major works on urgent problems. More about the Center can be found at watershed.ucdavis.edu.

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Adminsitrative Record
Page 34242

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 1:

Project and Technical Report Outline

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
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Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 34243

Project and Technical Report Outline

Technical Report 1

Addressing Nitrate in California's Drinking Water

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

Prepared by:

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California Nitrate Project, Implementation of Senate Bill X2 1

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Adminsitrative Record
Page 34244

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Contents

1 Significance and Scope of Work 1

2 Outline of Technical Reports 6

3 References 7

Adminsitrative Record
Page 34247

1 Significance and Scope of Work

The development of California's tremendous economy has not been without environmental costs. Since early in the twentieth century, nitrate from agricultural and urban activities has slowly infiltrated into groundwater. Nitrate has accumulated and spread and will continue to make its way into drinking water supplies. The time lag between the application of nitrogen to the landscape and its withdrawal at household and community public water supply wells, after percolating through soils and groundwater, commonly extends over decades.

This Report is an overview of groundwater contamination by nitrate in the Tulare Lake Basin and Salinas Valley. We examine the extent, causes, consequences, and costs of this contamination, as well as how it will likely develop over time. We also examine management and policy actions available for this problem, including possible nitrate source reduction, provisions for safe drinking water, monitoring and assessment, and aquifer remediation actions. The costs and institutional complexities of these options, and how they might be funded, also are addressed.

Addressing nitrate contamination problems in the Tulare Lake Basin and Salinas Valley will require decades to resolve, driven by the pace of groundwater flow and the response times of humans and institutions on the surface. Nitrate in drinking water today is a legacy contaminant, but years and decades from now the nitrate in drinking water will be from today's discharges. Assistance and management to improve drinking water supplies in response to nitrate contamination is a central and urgent policy issue for the State of California. Another major policy issue is the inevitability of widespread groundwater degradation for decades to come, despite even heroic (and ultimately expensive) efforts to reduce nitrate loading into aquifers. This introduction attempts to put the issue in a larger context.

Groundwater is essential to California. Groundwater is vital for California's agricultural, industrial, urban, and drinking water uses. Depending on drought conditions, groundwater provides between one-third and nearly one-half of the state's water supplies. As a source of drinking water, groundwater serves people from highly dispersed rural communities to densely populated cities. More than 85% of community public water systems in California (serving 30 million residents) rely on groundwater for at least part of their drinking water supply. In addition, approximately 2 million residents rely on groundwater from either a private domestic well or a smaller water system not regulated by the state (State Water Board 2011). Intensive agricultural production, population growth, and—indirectly—partial restoration of environmental instream flows have led to groundwater overdraft (Hanak et al. 2011). More protective health-based water quality standards for naturally occurring water quality constituents and groundwater contamination from urban and agricultural activities pose serious challenges to managing the state's drinking water supply.

Nitrate is one of California's most widespread groundwater contaminants. Nitrate is among the most frequently detected contaminants in groundwater systems around the world, including the extensively tapped aquifers in California's Central Valley and Salinas Valley (Figure 1) (Spalding and Exner 1993; Burow et al. 2010; Dubrovsky et al. 2010; MCWRA 2010; Sutton et al. 2011). Nitrate contamination

Adminsitrative Record
Page 34248

poses an environmental health risk because many rural areas obtain drinking water from wells that are often shallow and vulnerable to contamination (Guillette and Edwards 2005; Fan and Steinberg 1996).

High levels of nitrate affect human health. Infants who drink water (often mixed with baby formula) containing nitrate in excess of the maximum contaminant level (MCL) for drinking water may quickly become seriously ill and, if untreated, may die because high nitrate levels can decrease the capacity of an infant's blood to carry oxygen (methemoglobinemia, or "blue baby syndrome"). High nitrate levels may also affect pregnant women and adults with hereditary cytochrome b5 reductase deficiency. In addition, nitrate and nitrite ingestion in humans has been linked to goitrogenic (anti-thyroid) actions on the thyroid gland (similar to perchlorate), fatigue and reduced cognitive functioning due to chronic hypoxia, maternal reproductive complications including spontaneous abortion, and a variety of carcinogenic outcomes deriving from N-nitrosamines formed via gastric nitrate conversion in the presence of amines (Ward et al. 2005).

Nitrate is part of the natural nitrogen cycle in the environment. Groundwater nitrate is part of the global nitrogen cycle. Like other key elements essential for life, nitrogen flows through the environment in a dynamic cycle that supports organisms ranging from microbes to plants to animals. Plants require nitrogen for growth, and scarcity of fixed soil nitrogen often limits plant growth. Specialized micro-organisms can fix atmospheric elemental nitrogen and make it available for plants to use for photosynthesis and growth. The natural nitrogen cycle is a dynamic balance between elemental nitrogen in the atmosphere and reactive forms of nitrogen moving through the soil-plant-animal-water-atmosphere cycle of ecosystems globally. Production of synthetic nitrogen fertilizer has disrupted this balance.

Nitrogen is key to global food production. Modern agricultural practices, using synthetically produced nitrogen fertilizer, have supplied the nitrogen uses of plants to increase food, fiber, feed, and fuel production for consumption by humans and livestock. Agricultural production is driven by continued global growth in population and wealth, which increases demand for agricultural products, particularly high-value agricultural products such as those produced in California. Global food, feed, and fiber demands are anticipated to increase by over 70% over the next 40 years (Tilman et al. 2002; De Fraiture et al. 2010).

Intensive agriculture and human activities have increased nitrate concentrations in the environment. Greater use of nitrogen-based fertilizers, soil amendments such as manure, and nitrogen-fixing cover crops add nitrogen to deficient soils and dramatically raise crop yields. Technological advances in agriculture, manufacturing, and urban practices have increased levels of reactive forms of nitrogen, including nitrate, released into the atmosphere, into surface water, and into groundwater. The nearly 10-fold increase of reactive nitrogen creation related to human activities over the past 100 years (Galloway and Cowling 2002) has caused a wide range of adverse ecological and environmental impacts (Davidson et al. 2012).

The most remarkable impacts globally include the leaching of nitrate to groundwater; the eutrophication of surface waters and resultant marine "dead zones"; atmospheric deposition that

Adminsitrative Record
Page 34249

acidifies ecosystems; and the emission of nitrogen oxides (NOx) that deplete stratospheric ozone (Keeney and Hatfield 2007; Beever et al. 2007; Foley et al. 2005). These widespread environmental changes also can threaten human health (Galloway et al. 2008; Guillette and Edwards 2005; Galloway et al. 2004; Townsend et al. 2003; Vitousek et al. 1997; Fan and Steinberg 1996; Jordan and Weller 1996).

California has decentralized regulatory responsibility for groundwater nitrate contamination. Nitrate contamination of groundwater affects two state agencies most directly. Sources of groundwater nitrate are regulated under California's Porter-Cologne Water Quality Control Act (Porter-Cologne) administered through the State Water Resources Control Board (State Water Board) and the Regional Water Quality Control Boards (Regional Water Boards). State Water Board Resolution 88-63 designates drinking water as a beneficial use in nearly all of California's major aquifers. Under the Porter-Cologne Act, dischargers to groundwater are responsible, first, for preventing adverse effects on groundwater as a source of drinking water, and second, for cleaning up groundwater when it becomes contaminated.

Drinking water in public water systems (systems with at least 15 connections or serving at least 25 people for 60 or more days per year) is regulated by CDPH under the federal Safe Drinking Water Act of 1972 (SWDA). CDPH has set the nitrate MCL in drinking water at 45 mg/L (10mg/L as nitrate-N). If nitrate levels in public drinking water supplies exceed the MCL standard, mitigation measures must be employed by water purveyors to provide a safe supply of drinking water to the population at risk.

The California Department of Food and Agriculture (CDFA) and the Department of Water Resources (DWR) also have roles in nitrate management. The DWR is charged with statewide planning and funding efforts for water supply and water quality protection, including the funding of Integrated Regional Water Management Plans and DWR's management of urban and agricultural water use efficiency. CDFA collects data, funds research, and promotes education regarding the use of nitrogen fertilizers and other nutrients in agriculture.

SBX2 1 Nitrate in Groundwater Report to Legislature. In 2008, the California legislature enacted Senate Bill SBX2 1 (Perata), which created California Water Code Section 83002.5. The bill requires the State Water Board to prepare a Report to the Legislature (within 2 years of receiving funding) to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the state for the purposes of this section to clean up or treat groundwater, and ensure the provision of safe drinking water to all communities."Specifically, the bill directs the State Water Board to

> identify sources, by category of discharger, of groundwater contamination due to nitrate in the pilot project basins; to estimate proportionate contributions to groundwater contamination by source and category of discharger; to identify and analyze options within the board's current authority to reduce current nitrate levels and prevent continuing nitrate contamination of these basins and estimate the costs associated with exercising existing authority; to identify methods and costs associated with the treatment of nitrate contaminated groundwater for use as drinking water; to identify methods and costs to provide an alternative water supply to groundwater reliant communities in each pilot project basin; to identify all potential funding sources to provide resources for the cleanup of nitrate, groundwater treatment for nitrate, and the provision of alternative drinking water supply, including, but not limited to, State bond funding, federal funds, water rates, and fees or fines on polluters; and to

Adminsitrative Record
Page 34250

> develop recommendations for developing a groundwater cleanup program for the Central Valley Water Quality Control Region and the Central Coast Water Quality Control Region based upon pilot project results.

The bill designates the groundwater basins of the Tulare Lake Basin region and the Monterey County portion of the Salinas Valley as the selected pilot project areas. In June 2010, the State Water Board contracted with the University of California, Davis, to prepare this Report for the Board as background for its Report to the Legislature.

Project area is relevant to all of California. The project area encompasses all DWR Bulletin 118 designated groundwater sub-basins of the Salinas River watershed that are fully contained within Monterey County, and the Pleasant Valley, Westside, Tulare Lake Bed, Kern, Tule River, Kaweah River, and Kings River groundwater sub-basins of the Tulare Lake Basin. The study area—2.3 million ha (5.7 million ac) in size—is home to approximately 2.65 million people, almost all of whom rely on groundwater as a source of drinking water. The study area includes four of the nation's five counties with the largest agricultural production; 1.5 million ha (3.7 million ac) of irrigated cropland, representing about 40% of California's irrigated cropland; and more than half of California's dairy herd. More than 80 different crops are grown in the study area (Figure 2). This is also one of California's poorest regions: many census blocks with significant population belong to the category of severely disadvantaged communities (less than 60% of the state's median household income), and many of the remaining populated areas are disadvantaged communities (less than 80% of the state's median household income). These communities have little economic means and technical capacity to maintain safe public drinking water systems given contamination from nitrate and other contaminants in their drinking water sources.

Report excludes assessment of public health standards for nitrate. Public health and appropriateness of the drinking water limits are prescribed by CDPH and by U.S. EPA under SDWA. The scope of SBX2 1 precluded a review of the public health aspects or a review of the appropriateness of the nitrate MCL, although this is recognized as an important and complex aspect of the nitrate contamination issue (Ward et al. 2005).

"Report for the State Water Resources Control Board Report to the Legislature" and supporting Technical Reports. This Report for the State Water Board Report to the Legislature ("Report") has been provided in fulfillment of the University of California, Davis, contract with the State Water Board. This Report provides an overview of the goals of the research, methods, and key findings of our work, and is supported by eight related Technical Reports (Harter et al. 2012; Viers et al. 2012; Dzurella et al. 2012; Boyle et al. 2012; King et al. 2012; Jensen et al. 2012; Honeycutt et al. 2012; and Canada et al. 2012). The Technical Reports provide detailed information on research methods, research results, data summaries, and accompanying research analyses that are important for evaluating our results and findings and for applying our approach and results to other groundwater basins.

The Report takes a broad yet quantitative view of the groundwater nitrate problem and solutions for this area and reflects collaboration among a diverse, interdisciplinary team of experts. In its assessment, the Report spans institutional and governmental boundaries. The Report quantifies the diverse range of sources of groundwater nitrate. It reviews the current groundwater quality status in the project area by

Adminsitrative Record
Page 34251

compiling and analyzing all available data from a variety of institutions. It then identifies source reduction, groundwater remediation, drinking water treatment, and alternative drinking water supply alternatives, along with the costs of these options. Descriptions and summaries are also included of current and potential future funding options and regulatory measures to control source loading and provide safe drinking water, along with their advantages, disadvantages, and potential effectiveness.

This set of Reports is the latest in a series of reports on nitrate contamination in groundwater beginning in the 1970s (Schmidt 1972; Report to Legislature 1988; Dubrovsky et al. 2010; U.S. EPA 2011). This Report has some of the same conclusions as previous reports but takes a much broader perspective, contains more analysis, and perhaps provides a wider range of promising actions.

Adminsitrative Record
Page 34252

2 Outline of Technical Reports

The reports are organized to address the specific questions posed by SBX2 1.

Technical Report 2 – Sources of Groundwater Nitrate (Viers et al., 2012) describes the sources of nitrate in groundwater, reviews literature data on groundwater nitrate leaching from these sources, and assembles a wide range of data from numerous sources to quantify groundwater nitrate loading from individual categories of dischargers.

Technical Report 3 – Nitrate Source Reduction to Protect Groundwater Quality (Dzurella et al., 2012) provides an extensive review of both literature information and expert opinion collected specifically for this project on options and costs for reducing nitrate loading to groundwater.

Technical Report 4 – Groundwater Nitrate Occurrence (Boyle et al., 2012) reviews the hydrogeology and groundwater quality in the study area with an emphasis on nitrate. It also documents the assembly of a large database of groundwater nitrate data from the study area, which are then analyzed for historic trends and spatial distribution of nitrate. To provide an assessment of future groundwater quality, a groundwater model was developed that links the source loading developed in Technical Report 2 (Viers et al., 2012) to a groundwater transport model and predict both, historic and future groundwater quality.

Technical Report 5 – Groundwater Remediation (King et al., 2012) explains options for groundwater remediation. Traditional and alternative remediation options are explored. To the degree possible, it also details potential costs of these options. The chapter outlines groundwater quality management practices.

Technical Report 6 – Drinking Water Treatment for Nitrate (Jensen et al., 2012) is a comprehensive guide on treatment options for potable water supplies. Ion exchange, reverse osmosis, electrodialysis, and biological and chemical denitrification processes are explained, their advantages and disadvantages are described, as well as costs. The chapter also provides a guidance for selecting mitigation strategies to address nitrate in drinking water.

Technical Report 7 – Alternative Water Supply Options for Nitrate Contamination (Honeycutt et al., 2012) defines and identifies the population susceptible to nitrate in drinking water, reviews alternative water supply options for water systems and private well owners, and reviews strategies and costs for the implementation of alternative water supply options.

Technical Report 8 – Regulatory and Funding Options for Nitrate Groundwater Contamination (Canada et al., 2012) provides a summary of current regulatory programs and outlines potential future regulatory options to manage nitrate. It reviews current and potential funding options to address safe drinking water issues in the study area.

Adminsitrative Record
Page 34253

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Page 34255

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Adminsitrative Record
Page 34256

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Adminsitrative Record
Page 34257

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 2:

Nitrogen Sources and Loading to Groundwater

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
http://groundwaternitrate.ucdavis.edu

Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 34258

Nitrogen Sources and Loading to Groundwater

Technical Report 2

Assessing Nitrate in California's Drinking Water

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Adminsitrative Record
Page 34261

Contents

Tables iv

Figures iv

Acronyms and Abbreviations xii

Units xvi

Summary 1

1 Nitrogen Source Loading - Synthesis 3

1.1 Overview 3

1.2 Nitrogen Cycle: Basic Concepts 5

1.3 Nitrate Discharge to Groundwater 6

1.4 Natural Nitrate Sources 6

1.5 Human Nitrate Sources 6

1.6 Groundwater Nitrate Loading by Major Source Category 9

1.7 Comparative Analysis of Cropland Loading 26

1.8 Simulation of Groundwater Nitrate Sources and Loading 30

1.9 Concluding Remarks 49

2 Nitrogen Cycling and Mass Balance 53

2.1 The Biological Importance of Nitrogen 53

2.2 The Nitrogen Cycle 53

2.3 Natural ("Biogenic") Processes of the Nitrogen Cycle 55

2.4 Anthropogenic Sources of Nitrogen and Environmental Consequences 57

2.5 Environmental Effects of Excess Nitrogen 61

2.6 Nitrogen Fluxes in Croplands: A Mass Balance Approach to Groundwater Nitrate Loading 65

3 Cropland Nitrogen Loading 77

3.1 Introduction 77

Adminsitrative Record
Page 34262

3.2 Landuse, Fertilizer Nitrogen Application, and Harvest: Methods 85

3.3 Landuse, Nitrogen Application, and Nitrogen Harvest: Results 107

3.4 Sources of Uncertainty and Information Needs 131

4 Nitrogen Loading from Dairies, Feedlots, and other Animal Farming Operations 135

4.1 Introduction to N Loading from Dairies 135

4.2 Review of N Loading Rates from Dairy Corrals 138

4.3 Groundwater N Loading from Corrals: Methods and Results 141

4.4 Review of N Loading Rates from Dairy Lagoons 145

4.5 Groundwater N Loading from Liquid Manure Storage Lagoons: Methods and Results 147

4.6 Review of Nitrate Loading Rates from Irrigated Crop Fields with Manure Applications 149

4.7 Dairy Manure N Applications to Cropland 150

4.8 Review of N Loading from Non-Dairy Animal Farming Operations 162

4.9 Summary: Animal Farming as a Source of Groundwater Nitrate 163

5 Urban Landscape Nitrate Loading 165

5.1 Introduction 165

5.2 Methods 166

5.3 Results and Discussion 167

6 Domestic and Urban Wastewater Sources of Nitrogen Loading 169

6.1 Introduction to Domestic and Urban Wastewater Sources 169

6.2 Wastewater Treatment and Food Processing Facilities 170

6.3 Sewage Systems and their Contribution to Groundwater Nitrogen Loading in the Tulare Lake Basin and Salinas Valley 206

6.4 Septic Systems 215

7 Atmospheric Nitrogen Deposition 225

7.1 Introduction to Atmospheric Deposition 225

7.2 Atmospheric Deposition Methods 225

Adminsitrative Record
Page 34263

7.3 Atmospheric Deposition Results and Discussion 225
8 Natural Sources of Nitrogen 228
9 Wells as Sources of Nitrogen 230
9.1 Introduction to Wells as Sources of Nitrogen 230
9.2 Active Well Characteristics, Number of Wells, Well Seals, and Backflow Prevention 234
9.3 Dry Wells 242
9.4 Inactive and Abandoned Wells 244
9.5 Estimating N Loading and Transfer from Active, Abandoned, Improperly Destroyed, or Dry Wells 247
9.6 Summary and Conclusions 254
Literature Cited 257
1 Introduction 257
2 N Cycling 258
3 Cropland Nitrogen Loading 261
6 Domestic and Urban Wastewater Sources of Nitrogen Loading 268
7 Atmospheric N Deposition 274
8 Natural Sources of N 274
9 Wells as Sources of N 275
Appendix of Tables and Figures 278

Adminsitrative Record
Page 34264

Tables

Table 1. Major sources of groundwater nitrate, their estimated total contribution in the study area, their percent of total contribution, and their estimated average local intensity, which indicates local pollution potential.. 11

Table 2. Major nitrogen fluxes to and from cropland in the study area, by county. 19

Table 3. Total nitrogen discharge to land application and average total nitrogen concentration (as nitrate-N, MCL: 10 mg N/L) in discharge to percolation basins from WWTPs and FPs. 22

Table 4. Reporting years, by county, used to compare harvested land area in each county by three different data sources.. 27

Table 5. Comparison of harvested land area in each county 28

Table 6 . Comparison of harvested land area for each major crop group 29

Table 7. Study area summary of simulated CAML-based, spatially distributed nitrogen fluxes that account for mapped areas of cropland 1945 – 2050 (not including alfalfa), spatial distribution of WWTP, FP, and dairy facilities, spatial distribution of urban areas, and spatially variable atmospheric nitrogen deposition 36

Table 8. Statewide N emissions and ratio of past and future N emissions to current atmospheric N emissions 69

Table 9. Median groundwater nitrate in community public supply wells for 2000-2009, by DWR groundwater sub-basin (Boyle et al., 2012), groundwater use, and calculated basin-wide average nitrate-nitrogen application rate from pumped irrigation water. 70

Table 10. Data and estimation procedure to derive values for the multicropping rate of seven vegetable crops (celery, lettuce, spinach, broccoli, cabbage, cauliflower, and Brussels sprouts). 71

Table 11. Relationships between nitrate leaching and N inputs and/or drainage volume in free drained California sites) 79

Table 12. Farmland Mapping and Monitoring Program categories. 90

Table 13. UPlan Land Use Descriptions (adapted from UPlan Model Output Guide, Information Center for the Environment). 98

Table 14. UPLAN Business as Usual modeling scenario results showing change in land use over time. 103

Table 15. UPLAN Smart Growth modeling scenario results showing change in land use over time. 103

Table 16. The percentage change of N application rates for the given time periods 105

Adminsitrative Record
Page 34265

Table 17. Top 10 Agricultural Commodities in California, 2004-2005. 109

Table 18. BAU: Acreages for predicted UPlan urban expansion. 121

Table 19. Smart Growth acreages for predicted UPlan urban expansion. 121

Table 20. Number of different crop types or crop categories within each crop group. 122

Table 21. Historical changes in the total harvested area [ha] in the study area counties, by crop group (from ACR data) 125

Table 22. Historical changes in the total nitrogen typically applied [Gg N/yr] in the study area counties, by crop group, based on typical rates for each crop. 126

Table 23. Historical changes in average typical nitrogen application rate [kg N/ha], by crop group, obtained by dividing total typical nitrogen application rate by the total area 126

Table 24. Historical changes in total nitrogen harvested [Gg N/yr] in the study area counties, by crop group, based on ACR data of annual yields for each crop. 127

Table 25. Historical changes in average harvested nitrogen rate [kg N/ha], by crop group, obtained by dividing total harvested nitrogen (Table 23) by the total harvested area 128

Table 26. County crop areas, typically applied nitrogen, and harvested nitrogen in 2005, based on the median area of each crop between 2003 and 2007. 130

Table 27. Historic number of milk cows, not including dry cows 135

Table 28. County by county summary of corral area and of the lower and upper limits of estimated N loading to groundwater 142

Table 29. County by county summary of lagoon area and estimated largest possible N loading to groundwater from storage lagoons 147

Table 30. Number of milking cows, dry cows, and support stock (calves, heifers, etc.) in the Tulare Lake Basin study area, and the cropland acreage associated with dairies, total and by county 151

Table 31. Total amount of manure nitrogen excreted by dairy cattle in each TLB county, atmospheric nitrogen losses from manure, manure nitrogen sold off dairy, and manure nitrogen land applied within dairies 152

Table 32. Estimate of historical manure nitrogen excretion rates in the Tulare Lake Basin. 153

Table 33. Operational model on the fate of excreted nitrogen, by county. 157

Table 34. Matching of RB5 2010 Dairy database adult dairy animal numbers for 2007-2009 with the RB5 APN database of 2007 reported land area of dairy facilities and cropland for land application via dairy name. The table provides the number of adult cows and the total associated acreage of the match. .. 159

Adminsitrative Record
Page 34266

Table 35. Cropland and facilities acreage of assessor parcel numbers (APNs) reported by dairies to the RB5. Also shown are the acreages reported in the RB5 2010 dairy database as 2007 conditions. 160

Table 36. Total land area of cropland and other land uses within land parcels managed by a dairy 161

Table 37. Manure nitrogen from swine and poultry used for land application on cropland 163

Table 38. Turfgrass acreages in the study area..... 167

Table 39. Site considerations, design features, and characteristic effluent water quality for land treatment processes. 173

Table 40. Typical composition of domestic wastewater..... 174

Table 41. Fate of California biosolids in 2009. 177

Table 42. California counties receiving the greatest amount of biosolids in 2009..... 178

Table 43. Metric tons (Mg) of N applied annually in facility discharge (2010). 186

Table 44. Estimated metric tons (Mg) of N applied historically in facility discharge based on population change by county (WWTPs and FPs) and on change in specialty crop N (only FPs) between 1945 and 2010. 187

Table 45. Annual average N (kg N/ha/yr) discharged to irrigated land and percolation basins from WWTPs and FPs (averaged across all sites in each region) and corresponding total hectarage..... 188

Table 46. Average N concentration (mg/L) in discharge to irrigated land and percolation basins from WWTPs and FPs. 188

Table 47. Estimated metric tons (Mg) of biosolids produced or prepared annually. 196

Table 48. Nitrogen and flow characteristics by food processor type and basin for facilities with information available 200

Table 49. Sewer nitrogen loss by city population. 212

Table 50. Sewer nitrogen loss by wastewater treatment plant flow in MGD 213

Table 51. Calculated nitrogen loads (kg/ha/yr). 214

Table 52. Calculated sewer recharge rates (mm/yr)..... 214

Table 53. Summary of all estimates. 214

Table 54. Number of persons on septic in each county, for the regions of those counties that are within our study area boundaries. 219

Adminsitrative Record
Page 34267

Table 55. Land area with septic system densities below the threshold of 40 system per square mile, up to twice the threshold, one septic system per 2.5 acres, 2 systems per 2.5 acres, and above..... 222

Table 56. Yearly well construction totals for Kern, Kings, and Monterey Counties. 236

Table 57. Total numbers of Fresno County well construction permits since 1976 236

Table 58. Relative rates of infiltration found in lab tests of annular seal materials..... 239

Table 59. Performance rankings for eleven grout materials 240

Table 60. Estimated number of well destructions in five counties; 1977-2009 245

Table 61. Estimates of number of abandoned agricultural wells constructed between 1977 and 2009 in five counties using abandonment rate of 10% - 20% 246

Table 62. Number of abandoned public wells in five counties of the SBX2 1 study area..... 247

Table 63. Summary of the total amount of wells constructed, inactive or abandoned, and destroyed. 250

Table 64. Results of approximate worst case scenario for aquifer nitrate loading due to well leakage. 253

Appendix Table 1. Select estimates of NO3 leaching in California 277

Appendix Table 2. Hectares of land cover classes in the 2010 CAML Map 283

Appendix Table 3. 1990 Comparison of mapped vs. reported hectare totals..... 287

Appendix Table 4. 1977 Comparison of mapped vs. reported hectare totals..... 291

Appendix Table 5. 1960 Comparison of mapped vs.reported hectare totals..... 295

Appendix Table 6. 1946 Comparison of mapped vs. reported hectare totals..... 299

Appendix Table 7. Typical nitrogen applied, nitrogen yield, and partial nitrogen balance for each crop in each of five time periods 303

Appendix Table 8. Summary wastewater treatment and food processing facility data as reported or modeled 312

Appendix Table 9. Well completion records from the DWR South Central Region.....320

Adminsitrative Record
Page 34268

Figures

Figure 1. Estimated groundwater nitrate loading from major sources within the Tulare Lake Basin and Salinas Valley. 4

Figure 2. The Tulare Lake Basin (TLB) and Salinas Valley (SV) are the focus of this study. 10

Figure 3. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. 12

Figure 4. Current typical annual fertilization rates (1 kg/ha/yr = 1.1 lbs/ac/yr) in irrigated agricultural cropland of the study area. 14

Figure 5. Current annual nitrogen removal rate in harvested materials. 15

Figure 6. Nitrogen application reduction needed to reduce groundwater nitrate loading to less than 35 kg N/ha/crop, compared with average nitrogen applied (synthetic fertilizer and manure) and nitrogen harvested. 18

Figure 7. Estimated historical agricultural development in the study area (not including alfalfa) 20

Figure 8. Septic-derived nitrate leaching rates within the study area. 24

Figure 9. Simulated synthetic fertilizer application on cropland, including alfalfa, after accounting for the application of manure as fertilizer on dairy cropland. 37

Figure 10. Simulated land applied N from dairy manure (land applied on dairy cropland), and from WWTP/FP effluent and biosolids.. 38

Figure 11. Simulated land applied N from dairy manure (showing only land applied N on dairy cropland, but not exported dairy manure N applications), and from WWTP/FP effluent and biosolids. 39

Figure 12. Simulated land application of dairy manure N exported from dairies and land applied within the county of origin at rates proportional and in addition to typical N fertilizer needs in each crop. 40

Figure 13. Simulated land application of dairy manure N exported from dairies and land applied across the study area (not restricted to the county of origin) at rates proportional and in addition to typical N fertilizer needs in each crop. 41

Figure 14. Simulated nitrogen harvested in 2005 from all cropland including alfalfa. 42

Figure 15. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical “Scenario D”: All land applied manure N (62% of animal N excreted) is applied to corn, grain, and other field crops (not including alfalfa) under the direct management of dairies (no manure N exports). 43

Figure 16. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical “Scenario C (by county)”: hypothetically exported manure N from dairies (38% of animal N excretion) is land applied as soil amendment within the county of origin. 44

Adminsitrative Record
Page 34269

Figure 17. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario C (study area)": hypothetically exported manure N from dairies (38% of animal N excretion) is land applied as soil amendment across the study area (not restricted to the county of origin). 45

Figure 18. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario B (by county)": half of the hypothetically exported manure N from dairies (19.5% of animal N excretion) is land applied as soil amendment within the county of origin. 46

Figure 19. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario B (study area)": half of the hypothetically exported manure N from dairies (19.5% of animal N excretion) is land applied as soil amendment across the study area (not restricted to the county of origin). 47

Figure 20. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario A.". 48

Figure 21. Groundwater loading from non-cropland sources including leaky sewer lines, turf areas, golf courses, WWTP/FP percolation basins, dairy manure lagoons, and animal corrals, in 2005. 49

Figure 22. Dominant pathways of the nitrogen cycle 54

Figure 23. Recent Climate from TLB and SV Study areas (2001-2010) 78

Figure 24. Current evidence of the relationship between N leached and N inputs 81

Figure 25. Comparison of nitrate leaching estimates measured by soil sampling and suction lysimeters on a fine sandy loam in California 83

Figure 26. Flowchart of inputs to the CAML raster layer. 88

Figure 27. Input layers for the final CAML 2010 raster layer. 91

Figure 28. BAU modeling scenario 99

Figure 29. Smart Growth modeling scenario 100

Figure 30. BAU – Agriculture removal modeling scenario 101

Figure 31. Smart Growth – Agriculture removal modeling scenario 102

Figure 32. CAML 2010 Land Cover Map 108

Figure 33. Fresno County top crops in hectares 110

Figure 34. Tulare County top crops in hectares 111

Figure 35. Kern County top crops in hectares 112

Figure 36. Kings County top crops in hectares 113

Figure 37. Monterey County top crops in hectares. 114

Adminsitrative Record
Page 34270

Figure 38. 1946 Land Use from DWR Mapping. 117

Figure 39. Backcasted 1960 Land Cover Map 118

Figure 40. Backcasted 1977 Land Cover Map. 119

Figure 41. 1990 Land Use from DWR Mapping. 120

Figure 42. Location of included wastewater treatment plants and food processing facilities 183

Figure 43. Wastewater treatment plants: Total applied nitrogen and population served 190

Figure 44. Tulare Lake Basin wastewater treatment plants 191

Figure 45. Salinas Valley wastewater treatment plants 192

Figure 46. Tulare Lake Basin wastewater treatment plants 193

Figure 47. Salinas Valley wastewater treatment plants 194

Figure 48. Annual metric tons of biosolids produced by surveyed WWTPs 195

Figure 49. Large land application operations and WWTPs reporting direct land application of biosolids: Total annual mass of nitrogen [color] and hectarage in ha [diameter] 197

Figure 50. Large land application operations and WWTPs reporting direct land application of biosolids: kg N/ha/yr [color] and hectarage in ha [diameter] 198

Figure 51. Food processor type distribution 199

Figure 52. Total metric tons of nitrogen applied annually from food processor discharge 201

Figure 53. Tulare Lake Basin food processors: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) and total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] 202

Figure 54. Salinas Valley food processors: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) and total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] 203

Figure 55. Tulare Lake Basin food processors: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] 204

Figure 56. Salinas Valley food processors: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] 205

Figure 57. Sanitary sewer overflows in the Tulare Lake Basin and Salinas Valley reported in 2011. 207

Figure 58. Percent of households on septic systems, by blockgroup, from 1990 census data, in the study area. 220

Figure 59. Septic systems per hectare, in the study area 221

Adminsitrative Record
Page 34271

Figure 60. Septic-derived nitrogen leaching rates within the study area. 223

Figure 61. Atmospheric N deposition, spatially distributed across the study area, based on the Community Multiscale Air Quality model developed by U.S. EPA. 227

Figure 62. Schematic outline of a well. 231

Figure 63. Example of an abandoned well that poses a risk of direct surface runoff spillage into the well casing and into the aquifer 233

Figure 64. Possible backflow prevention device schematic. 241

Appendix Figure 1. Food processor locations 321

Appendix Figure 2. Wastewater treatment plant locations 322

Appendix Figures 3 through Appendix Figure 120 are available in a separate PDF file at

http://groundwaternitrate.ucdavis.edu

Adminsitrative Record
Page 34272

Acronyms and Abbreviations

ACR	Agricultural Commissioner Reports
AF	acre foot
AFOs	animal farming operations
APNs	assessor parcel numbers
BAU	"Business as Usual"
BNF	biological nitrogen fixation
C	carbon
C:N	carbon to nitrogen ratio
CAML	California Augmented Multisource Landcover
CDFA	California Department of Food and Agriculture
CDPH	California Department of Public Health
CH_4	methane
CIWQS	California Integrated Water Quality System
CMAQ	Community Multiscale Air Quality
CNA	California Nitrogen Assessment
CO	carbon monoxide
CO_2	carbon dioxide
CSU Chico	California State University, Chico
CVHM	Central Valley Hydrologic Model
CVRWB	Central Valley Regional Water Quality Control Board
CV-SALTS	Central Valley Salinity Alternatives for Long-Term Sustainability
CWA	Clean Water Act
CWC	California Water Code
Dairy General Order	Waste Discharge Requirements General Order No. R5-2007-0035
dairy-ΣN_{fertil}	actual total applied fertilzer N on a dairy
dairy-ΣN_{manure}	total available land applied manure N for an individual dairy
dairy-ΣN_{norm}	sum of N_{norm} across all dairy cropland receiving manure
DOC	dissolved organic carbon
DPR	California Department of Pesticide Regulation
DWR	Department of Water Resources
EPA	United States Environmental Protection Agency
EQ	Exceptional Quality
ET	evapotranspiration
FMMP	Farmland Mapping and Monitoring Program
FP	Food Processing
FPs	Food Processors
FRS	Facilities Registry System
Gg	Gigagram

Adminsitrative Record
Page 34273

Gg N/yr	Gigagrams of nitrate-nitrogen per year
GIS	Geographic Information System
GNLM	Groundwater Nitrate Loading Model
GRASS	Geographic Resources Analysis Support System
GRP	Glass Reinforced Pipe
HDPE	High Density Polyethylene
HNO_3	nitric acid
ICE UC Davis	the Information Center for the Environment at UC Davis
IPCC	Intergovernmental Panel on Climate Change
LULC	Land Use Land Cover
MCL	maximum contaminant level
Mg	Megagrams (metric tons)
mgd	million gallons per day
MRPs	Monitoring and Reporting Programs
MSLC	Multi-Source Land Cover
N	nitrogen
N_2	non-reactive nitrogen gas
N_2O	nitrous oxide
$N_{AreaManureExport}$	total amount of manure N exported in the county or study area
NASS	National Agricultural Statistics Survey
$N_{deposit}$	N from atmospheric deposition
N_{excess}	annual rate of N applied in excess of recommended rates
N_{fertil}	N from synthetic fertilizer
N_{GW}	N loading to groundwater
$(NH_2)_2CO$	urea
NH_3	amonia
NH_4^+	ammonium
$N_{harvest}$	N in the harvest
N_{irrig}	N contained in the source irrigation water (well, stream)
N_{loss}	N losses to the atmosphere via volatilization or denitrification
N_{manure}	N from manure, where applied
N_{norm}	typical amount of fertilization
NO_2^-	nitrite
NO_3^-	nitrate
NOI	Notice of Intent
Norg	organic nitrogen
NO_x	nitrogen oxides

Adminsitrative Record
Page 34274

NPDES	National Pollutant Discharge Elimination System
N_r	reactive nitrogen
N_{runoff}	N in surface runoff
$N_{TotalNorm}$	totalized recommended fertilizer application
$N_{WWTP-FP}$	N from WWTP/FP effluent or biosolids, where applied
O_3	ozone
OF	Overland Flow
PICME	Permits, Inspections, Compliance, Monitoring and Enforcement
PNB	partial nutrient balance
PNB_0	hypothetical partial nutrient balance
POTW	Publicly Owned Treatment Works
PUR	Pesticide Use Reports
PVC	Polyvinyl Chloride
RB5	Central Valley (Region 5) Regional Water Quality Control Board
Regional Water Board	Regional Water Quality Control Board
RI	Rapid Infiltration
RWDs	Reports of Waste Discharge
SAT	Soil Aquifer Treatment
SEP	Supplemental Environmental Project
SMR	Self Monitoring Report
SR	Slow Rate
SSO	Sanitary Sewer Overflow
State Water Board	State Water Resources Control Board
SV	Salinas Valley
TLB	Tulare Lake Basin
TN	Total Nitrogen
TP	Total Phosphorus
U.S. EPA	United States Environmental Protection Agency
UC	University of California
UC Davis	University of California, Davis
UC Davis ARE	Agricultural and Resource Economics at UC Davis
UIC	Underground Injection Control
Uplan	urban growth models; San Joaquin Valley Blueprint Planning Process
USDA	United States Department of Agriculture
USDWs	underground sources of drinking water
USGS	United States Geological Survey
VCP	Vitrified Clay Pipes
VOCs	volatile organic compounds
WARMF	Watershed Analysis Risk Management Framework

Adminsitrative Record
Page 34275

WCR	Well Completion Report
WDR	Waste Discharge Requirements
WWTP	Wastewater Treatment Plant

Adminsitrative Record
Page 34276

Units

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.5 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-).

To convert from:

Nitrate-N (NO_3-N)	→	Nitrate (NO_3^-)	multiply by 4.43
Nitrate (NO_3^-)	→	Nitrate-N (NO_3-N)	multiply by 0.226

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Adminsitrative Record
Page 34277

Summary

Nitrate loading to groundwater in the Tulare Lake Basin and Salinas Valley is widespread and chronic, and is overwhelmingly the result of crop and animal agricultural activities. Urban wastewater, septic systems, and other sources may have significant localized impact. Due to long transit times, the impact on groundwater resources is a legacy for years and decades to come.

The application of synthetic fertilizer and manure to agricultural crops, primarily under irrigated conditions, has resulted in high crop yields and the large-scale production of affordable food for the world's growing population. It has also promoted California – and the Tulare Lake Basin and Salinas Valley in particular – to the top of global crop production. The Tulare Lake Basin has also benefited from animal agriculture, where dairy commodities are the top economic producer. These agricultural operations, however, have not been without costs to the environment. A significant fraction of nitrogen applied in food production worldwide is in excess of crop needs, resulting in nitrate leaching to groundwater, eutrophication of aquatic ecosystems via surface run off, and air pollution from toxic emissions of ammonia and ozone-depleting greenhouse gases. Each of these negative environmental outcomes has the potential to impart significant impact on biogeochemical processes, ecosystem services, and human health. Current human activities cannot be sustained without commensurate and perhaps permanent degradation of vital natural resources, most specifically drinking water from groundwater aquifers. There are cumulative and long-term societal, environmental, and economic costs to our excess utilization of industrially-fixed nitrogen now used in cropland agriculture. Understanding these consequences requires a better scientific understanding of nitrate sources.

Using a mass balance approach, this technical report documents the extent and magnitude of nitrogen loading from anthropogenic and natural sources to groundwater in the Tulare Lake Basin and Salinas Valley. Our approach considers crop demand, fertilization, harvest, and volatilization in cropland agriculture, in addition to accounting for animal agriculture and localized sources, for the period of study (~1940–present). Cropland agriculture is the primary vehicle for nitrate loading to groundwater. Already widespread when agrichemicals first arrived in large quantities (ca. 1940), cropland areas further expanded into the late 1960s and crops diversified greatly over the past 60 years, with specialty crops that have higher nitrogen demands becoming of increasing importance. Animal agriculture, and particularly dairy production, is a dominant and widespread source of nitrogen in the environment. With a sustained, exponential increase in nitrogen output over the past sixty years, dairies currently supply about one-third of all nitrogen applied to cropland. Today's nitrogen loading will not materialize as contaminated groundwater for years to decades to come, and the current average loading rate is three to five times greater than the recognized maximum contaminant levels for drinking water in California. As the sources and fates of nitrogen are transient over space and time, it will require concerted action across many agroeconomic sectors to minimize the long-term degradation of groundwater aquifers.

While cropland and animal agriculture are the principal sources of nitrogen loading in the study area, other sources also require attention. Foremost is the role that wastewater treatment plants and food processing facilities play in distributing excess solids and effluent. We have shown that their

Adminsitrative Record
Page 34278

contributions on a localized basis can be quite considerable. Therefore, any reduction measures for this source will likely be directed towards protecting local drinking water supplies rather than regional loading reduction. Further, we have documented that there are important seasonal differences in discharge and land application management. This variation also exists in agricultural settings, with dominant phases of application and irrigation. Thus, it is important to recognize that localized drinking water supplies may be affected on a seasonal basis.

We have documented that on a regional scale, groundwater nitrogen loading from sewers is negligible in comparison to loading from fertilizers. However, at the local level, sewer leakage can be a significant source of nitrate contamination. Localized sources of raw sewage near domestic or public wells have the potential to detrimentally affect public health. This localized threat exists regardless of the negligible regional contribution of sewers to groundwater nitrogen. Similarly, we have investigated the local influence of septic systems on groundwater nitrogen and have found that contamination of domestic, unregulated drinking water wells may be a significant problem in peri-urban areas surrounding cities, or in areas of relatively high rural household density. While septic system contributions to regional nitrogen loading are minimal, it is still of local importance as a driver of nitrate contamination in drinking water. Other locally problematic sources of nitrogen include urban sources such as overfertilization of lawns and other ornamental landscapes.

Most nitrate is transported from sources to groundwater via soil percolation and recharge. But dry wells, abandonded wells, or improperly destroyed wells may act as rapid local conduits of nitrate contaminated surface runoff directly into groundwater. In addition, many deep wells may inadvertently act as conduits for deep aquifer contamination from shallow, nitrate-contaminated groundwater. We therefore consider these here as separate sources. Locally, significant nitrate contamination may result from these conduits.

Our mass balance approach to understanding the spatial and temporal dimensions of nitrogen loading is informed by observation and based on physical principles. However, it is made with inherent uncertainty. There are considerable information and data gaps in all phases of our analyses. There are few empirical studies specific to conditions found in the study area, especially with respect to agronomic practice, that document the source, transition, and fate of nitrogen in agroecosystems. Further, there are few if any data that provide for long-term composition in a manner that can explicitly quantify what was occurring where, and when. While a synoptic assessment such as ours is difficult, and is made with varying degrees of uncertainty, it does not invalidate our results. Rather, it emphasizes that despite limited information for given aspects of our study, our results, made with conservative assumptions, indicate that the magnitude of the problem far exceeds those degrees of uncertainty. Improvements to subsequent studies should focus on expanding the breadth and resolution of information necessary to reduce uncertainties, rather than in the methods themselves. These findings also suggest the need to develop and implement a programmatic monitoring and evaluation mechanism to capture, collect, and analyze information critical to understanding the source, extent, and magnitude of nitrate loading in California. California's long term prospects for social, environmental, and economic sustainability may depend upon it.

Adminsitrative Record
Page 34279

1 Nitrogen Source Loading - Synthesis

1.1 Overview

Groundwater is a vital natural resource extracted from subterranean aquifers for a broad array of purposes including agricultural irrigation, industrial production, and human drinking water supply. In California, access to and utilization of groundwater is indispensable. However, increases in human population density and concomitant urban uses, overdraft from expanding urban and agricultural demand, and worsening contaminant loading threaten the suitability and sustainability of groundwater as a hydrological resource in California. In select regions of California, nitrate leaching to groundwater impairs its beneficial use as drinking water and the source of contamination is poorly quantified.

We begin this Technical Report by describing the results of our extensive analysis. Section 1 contains a detailed description of the nitrate loading to groundwater from various sources, at the study area level, the county level, the source category level, and at the land parcel level. In Section 1, methods are explained only briefly and with a focus on the conceptual framework. Supporting methodological details not documented in Section 1 are found in the remaining sections of this Technical Report: a review of nitrogen cycling in the environment, which also provides the conceptual background for a detailed description of the technical and mathematical methods employed to perform the mass balance approach (Section 2); a description of the land use in the study area and agricultural crop categories considered, their spatial distribution, historic development, fertilization needs, harvest, and a review of known groundwater loading rates from croplands (Section 3), animal agriculture as both, a source of nitrate loading directly to groundwater and as a source of nitrogen applied to cropland (Section 4), and other sources of nitrate loading to groundwater and of nitrate application to croplands within in the study area, including: urban landscape (Section 5), food processors, wastewater treatment plants, sewer and septic systems (Section 6), atmospheric deposition (Section 7) and natural sources (Section 8). Wells as rapid conduits of nitrate from sources into groundwater and from contaminated shallow groundwater to deep groundwater are considered in Section 9. A comprehensive list of literature citations is provided in Section 10. Each section represents a separate–and in some cases technically complex–analysis. In its entirety, this Technical Report presents a large body of evidence documenting the current state of known nitrate source loading to groundwater, its extent, its magnitude, and the uncertainty about its magnitude in the SBX2 1 study area. Moreover, while geographically focused on the Tulare Lake Basin and Salinas Valley, much of the methodology and many of the underlying data developed for this report are applicable, with modifications in some cases, to other areas of California or similar semi-arid, irrigated agricultural regions around the world.

This technical report identifies relevant sources and quantifies relative amounts of nitrate loading to groundwater in the Tulare Lake Basin and Salinas Valley. As will be shown in this Technical Report, human-generated nitrate sources to groundwater in the study area include (Figure 1):

Adminsitrative Record
Page 34280

- cropland (96% of total), where nitrogen applied to crops, but not removed by harvest, air emissions, or runoff is leached from the root zone to groundwater. Nitrogen intentionally or incidentally applied to cropland includes
    - synthetic fertilizer (54%),
    - animal manure (33%),
    - irrigation source water (8%),
    - atmospheric deposition (3%), and
    - municipal effluent and biosolids (2%);
- percolation of wastewater treatment plant (WWTP) and food processing (FP) wastes (1.5% of total);
- recharge from animal corrals and manure storage lagoons (1% of total);
- leachate from septic system drainfields (1% of total);
- urban parks, lawns, golf courses, and leaky sewer systems (less than 1% of total); and
- downward migration of nitrate-contaminated water via wells (less than 1% of total).

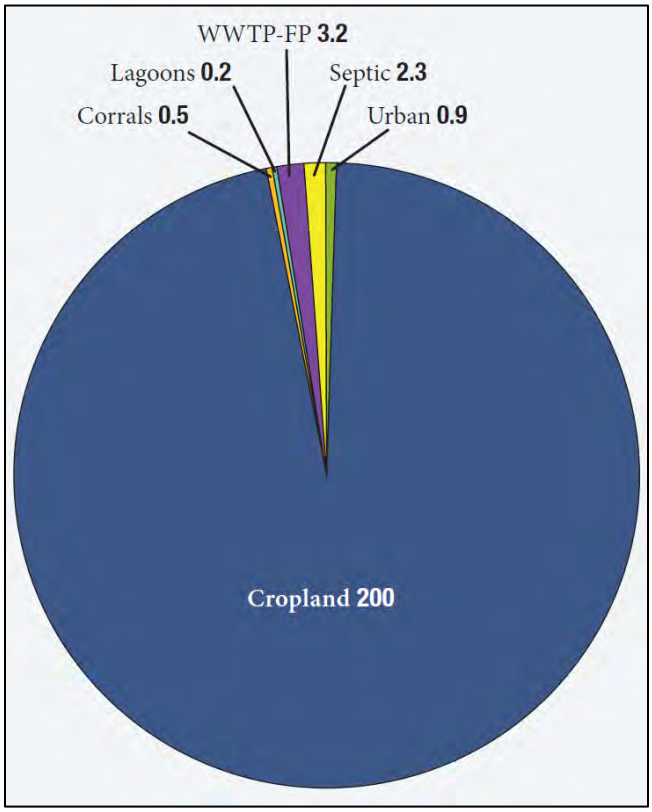

Figure 1. Estimated groundwater nitrate loading from major sources within the Tulare Lake Basin and Salinas Valley, in Gg nitrogen per year (1 Gg = 1,100 t).

Depending on the type of source, two principal methods are employed to assess nitrate loading:

- a mass balance approach was used to estimate nitrate loading from all categories of cropland except alfalfa;

Adminsitrative Record
Page 34281

- alfalfa cropland and nitrate sources other than cropland were assessed by reviewing permit records, literature sources, and by conducting surveys to estimate groundwater nitrate loading.

1.2 Nitrogen Cycle: Basic Concepts

Nitrogen is an essential element for all living organisms. Nitrogen cycles through the atmosphere, hydrosphere, and biosphere. The dominant gas (78%) in the atmosphere is highly stable (inert) N_2 gas. Biological nitrogen fixation transforms N_2 gas into ammonia (NH_3), which is rapidly converted to the forms of nitrogen needed for plant growth. Nitrogen fixation is performed only by specialized soil and aquatic microbes. Other living organisms cannot use inert atmospheric N_2 directly, but rely on accumulated soil organic matter, plants, animals, and microbial communities for nitrogen.

Soil nitrogen is most abundant in the organic form (N_{org}). Mineralization is a suite of processes performed by soil microbes that converts organic nitrogen to inorganic forms of nitrogen. The rates of mineralization depend on the environmental conditions such as temperature, moisture, pH, and oxygen content, as well as the type of organic matter available. The first product of mineralization is ammonium (NH_4^+), but under aerobic conditions, microbes can convert ammonium (NH_4^+) first to nitrite (NO_2^-) and then to nitrate (NO_3^-). Most plants use nitrate or ammonium as their preferred source of nitrogen (White 2006). Immobilization is the reverse of mineralization in that soil ammonium and nitrate are taken up by soil organisms and plants and converted into N_{org}.

The ultimate fate of "reactive" nitrogen (organic nitrogen, ammonium, nitrate, ammonia, nitrous oxide, etc.) is to return back to the atmosphere as N_2. For nitrate, this is a microbially mediated process ("denitrification") that requires an anoxic (i.e., oxygen-free) environment (see Section 2 for an expanded discussion).

Groundwater is becoming a growing component of the global nitrogen cycle because of the increased nitrogen inflows and because of long groundwater residence times. Nitrate does not significantly adhere to or react with sediments or other geologic materials, and it moves with groundwater flow. Other forms of reactive nitrogen in groundwater are less significant and much less mobile: ammonia occurs under some groundwater conditions, but it is subject to sorption and rapidly converts to nitrate under oxidizing conditions. Dissolved organic nitrogen concentrations are generally much less than those of nitrate, except near wastewater sources, due to the high adsorption of dissolved organic nitrogen to aquifer materials.

Groundwater nitrate inputs may come from natural, urban, industrial, and agricultural sources. Groundwater nitrate outputs occur through wells or via discharge to springs, streams, and wetlands. Discharge to surface water sometimes involves denitrification or reduction of nitrate to ammonium when oxygen-depleted conditions exist beneath wetlands and in the soils immediately below streams.

Adminsitrative Record
Page 34282

1.3 Nitrate Discharge to Groundwater

Nitrogen enters groundwater at varying concentrations and in varying forms (organic nitrogen, ammonium, nitrate) with practically all sources of recharge: diffuse recharge from precipitation and irrigation; focused recharge from streams, rivers, and lakes; focused recharge from recharge basins and storage lagoons; and focused recharge from septic system drainfields. Across major groundwater basins in California, diffuse recharge from irrigation, stream recharge, and intentional recharge are the major contributors to groundwater. Since groundwater is an important reservoir for long-term water storage, recharge is extremely important and desirable in many areas. Controlling nitrate in recharge and managing recharge is therefore a primary key to nitrate source control.

Current groundwater nitrate, its spatial distribution, and its changes through time result from recent, as well as historical, nitrate loading. To understand current and future groundwater conditions requires knowledge of historical, current, and anticipated changes in land use patterns, recharge rates, and nitrate loading rates. Providing a comprehensive review of land use and nitrate loading rate information for the study area is a key objective of this technical report. Groundwater recharge is reviewed in Technical Report 4 (Boyle et al., 2012).

1.4 Natural Nitrate Sources

Nitrate occurs naturally in many groundwater basins but at levels far below the regulatory maximum contaminant level (MCL) for drinking water (Mueller and Helsel 1996). The main potential sources of naturally occurring nitrate are bedrock nitrogen and nitrogen leached from natural soils. Surface water nitrate concentrations can be elevated in areas with significant bedrock nitrogen (Holloway et al. 1998), but they are not high enough to be a drinking water concern. During the early twentieth century, conversion of the study area's semiarid and arid natural landscape to irrigated agriculture may have mobilized two additional, naturally occurring sources of nitrate. First, nitrate was released from drained wetlands at the time of land conversion due to increased microbial activity in agricultural soils; that is, stable organic forms of nitrogen that had accumulated in soils over millennia were converted to mobile nitrate. Second, nitrate salts that had accumulated over thousands of years in the unsaturated zone below the grassland and desert soil root zone due to lack of significant natural recharge were mobilized by irrigation (Dyer 1965; Stadler et al. 2008; Walvoord et al. 2003). However, the magnitude of these sources (Scanlon 2010) is considered to have negligible effects on regional groundwater nitrate given the magnitude of human sources.

1.5 Human Nitrate Sources

Human Nitrate Sources. Anthropogenic groundwater nitrate sources in the study area include agricultural cropland, animal corrals, animal manure storage lagoons, wastewater percolation basins at municipal wastewater treatment plants (WWTPs) and food processors (FPs), septic system drainfields (onsite sewage systems), leaky urban sewer lines, lawns, parks, golf courses, and dry wells or percolation

Adminsitrative Record
Page 34283

basins that collect and recharge stormwater runoff. Incidental leakage of nitrate may also occur directly via poorly constructed wells. Croplands receive nitrogen from multiple inputs: synthetic fertilizer, animal manure, WWTP and FP effluent, WWTP biosolids, atmospheric deposition, and nitrate in irrigation water sources.

Categories of Sources and Timeline. We estimated the groundwater nitrate contributions for 58 agricultural cropland categories, for animal corrals, for manure lagoons, for each individual WWTP and FP within the study area, for dairies and other animal farming operations, for septic system drainfields, and for urban sources. Contributions from dry wells and incidental leakage through existing wells were estimated at the basin scale. Groundwater nitrate contributions were estimated for five time periods, each consisting of 5 years: 1943–1947 ("1945"), 1958–1962 ("1960"), 1973–1977 ("1975"), 1988–1992 ("1990"), and 2003–2007 ("2005"); the latter is considered to be current. Future year 2020, 2035, and 2050 loading was estimated based on anticipated land use changes (increased urbanization).

Data on the Spatial Extent (Area) of Cropland. The actual spatial extent or area of cropland acreage cannot be precisely reconstructed. Except for perennial crops, the specific crops grown in a field (if any) change seasonally and yearly. Even perennial cropping patterns change significantly over time. Three major sources of information are available that provide estimates of the spatial area or extent (acreage) of cropland sources (see also Section 3 for expanded discussion):

California Augmented Multisource Landcover (CAML): Aerial photography and detailed field mapping conducted at nearly decadal time intervals by the Department of Water Resources and other agencies leads to a detailed spatial map of crop categories with field-by-field resolution, albeit it can only be a snapshot in time. Maps of crop categories (and the total land area of each category) are available for one year in the late 1990s or early 2000s, and for one year in the early to mid-1990s, depending on county. Older maps are simulated based on county Agricultural Commissioner reports.

Agricultural Commissioner Reports (ACR): County Agricultural Commissioner offices annually survey and report the total amount of land harvested and the total amount of harvest. Data are available for each year over the entire period of interest (1943 – 2007). Data are reported as county totals, by crop category, and are not mapped. Often referred to as Ag Commissioner data or reports herein.

National Agricultural Statistics Survey (NASS) and agricultural census: NASS compiles county agricultural commissioner data and also infrequently conducts an agricultural census of harvested area and crop yields, independent of the county agricultural commissioner. Agricultural census data are available for 1950, 1992, 1997, 2002, and 2007. Data are reported as county totals, by crop category, and are not mapped.

All three sources of information were used separately (and comparatively) to derive estimates of groundwater nitrate loading from cropland: County agricultural commissioner reported crop acreages were averaged over five-year periods representing five historical time periods (1945, 1960, 1975, 1990, and 2005) to derive estimates of nitrate groundwater loading by crop category, by crop group, by county, and for the study area. Data reported in Section 1.6 are based on the cropping area data provided by the ACR. We compare NASS agricultural census data for the year closest to the most recent

Adminsitrative Record
Page 34284

CAML mapping dates, by county and crop group, against the CAML and against the ACR derived nitrate loading (Section 1.7). The CAML information and the historic land use simulations generated from recent CAML maps were used to derive maps of groundwater nitrate loading with a resolution of 0.25 ha (less than 1 acre) for 1945, 1960, 1975, 1990, 2005, and 2050. Information in these maps was then aggregated to the crop category, crop group, county, and study area level (Section 1.8).

Spatial Granularity of Nitrate Source Loading Estimates. The groundwater nitrate loading estimates are computed and reported at four different levels of granularity or spatial resolution, depending on the source of information used and the amount of processing and aggregation:

- by land use parcel: individual categories of nitrate discharges to groundwater are mapped at a resolution of 0.25 ha (less than 1 ac) for the entire study area, in 15 year intervals between 1945 and 2050 (CAML based estimates);
- by crop categories (e.g., olives, persimmons, lettuce, strawberries) and crop groups (e.g., 'subtropicals', 'vegetables and berries'), averaged or summed over the entire study area, 1945 - 2005; (CAML, Ag Commissioner reports, NASS based estimates)
- by county, totaled across all cropland, all WWTPs and FPs, all dairies, all septic drains, and all municipal areas, 1945 - 2005 (CAML, Ag Commissioner reports, NASS based estimates); and
- summed or averaged for the study area, 1945 - 2005 (CAML, Ag Commissioner based estimates).

The higher levels of aggregation (coarser granularity, lower spatial resolution) provide more accurate estimates of nitrate loading for the spatial unit considered (crop category, crop group, county, study area) but are less descriptive of the actual loading in any given land parcel within each category. Aggregated totals are most useful for policy and planning.

We report nitrate loading to groundwater in two ways:

- Total annual nitrate leached to groundwater, measured in gigagrams of nitrate-nitrogen per year (Gg N/yr).[2] As a practical measure, 1 gigagram is roughly equivalent to $1 million of nitrogen fertilizer at 2011 prices.
- Intensity of the nitrate leaching to groundwater, measured in kilograms of nitrate-nitrogen per hectare of use per year (kg N/ha/yr) [lbs per acre per year, lbs/ac/yr], which represents the intensity of the source at its location (field, pond, corral, census block, city) and its potential for local groundwater pollution.

[1]. One gigagram is equal to 1 million kilograms (kg), 1,000 metric tons, 2.2 million pounds (lb), or 1,100 tons (t). In this report, nitrogen application to land refers to total nitrogen (organic nitrogen, ammonium-nitrogen, and nitrate-nitrogen). For consistency and comparison, total nitrate loading and the intensity of nitrate loading from the root zone to groundwater are also provided in units of nitrogen, not as nitrate. However, concentrations of nitrate in groundwater or leachate are always stated as nitrate (MCL: 45 mg/L) unless noted otherwise.

2. A typical groundwater recharge rate in the study area is roughly 300 mm/yr (1 AF/ac/yr). If that recharge contains nitrate at the MCL, the annual nitrate loading rate is 30 kg N/ha/yr (27 lb N/ac/yr). We allow an additional 5 kg N/ha/yr (4.5 lb N/ac/yr) to account for potential denitrification in the deep vadose zone or in shallow groundwater.

Adminsitrative Record
Page 34285

To provide a broad reference point of what the source loading numbers mean with respect to potential groundwater pollution, it is useful to introduce an operational benchmark that indicates whether nitrate leached in recharge to groundwater exceeds the nitrate drinking water standard. This operational benchmark considers that nearly all relevant anthropogenic nitrate sources provide significant groundwater recharge and therefore remain essentially undiluted when reaching groundwater. Our benchmark for "low" intensity versus "high" intensity of nitrate leaching is 35 kg N/ha/yr (31 lb N/ac/yr).[3] Aggregated across the 1.5 million ha (3.7 million ac) of cropland, the benchmark for total annual nitrate loading in the study area is 50 Gg N/yr (55,000 t N/yr). Total nitrate loading to groundwater above this benchmark indicates a high potential for regional groundwater degradation.

Estimating nitrate loading by source category. We used two methods to assess nitrate loading:

- a mass balance approach was used to estimate nitrate loading from all categories of cropland except alfalfa;
- alfalfa cropland and nitrate sources other than cropland were assessed by reviewing permit records, literature sources, and by conducting surveys to estimate groundwater nitrate loading (Viers et al. 2012).

1.6 Groundwater Nitrate Loading by Major Source Category

Cropland is by far the largest nitrate source, contributing an estimated 96% of all nitrate leached to groundwater (Table 1). The estimated total nitrate leached to groundwater (200 Gg N/yr [220,000 t N/yr]) ±30% is about three to five times the benchmark amount, which suggests large and widespread degradation of groundwater quality. Wastewater treatment plants and food processor waste percolation basins are also substantial, high-intensity sources.[4] Septic systems, manure storage lagoons, and corrals are relatively small sources basin-wide, but since their discharge intensity significantly exceeds the operational benchmark of 35 kg N/ha/yr (31 lb N/ac/yr), these source categories can be locally important. The magnitude and intensity of urban sources (other than septic systems) does not suggest widespread impact to groundwater (see Sections 5 and 6). The following provides further, more detailed discussion on these sources.

1.6.1 Agricultural Sources

Cropland sources: Overview. The five counties in the study area include 1.5 million ha (3.7 million ac) of cropland, about 40% of California's total irrigated cropland. Agricultural production includes many individual crops and significant year-to-year changes in crops grown and crop yields. The dominant crop groups in the project area include subtropical tree fruits (citrus and olives), deciduous tree fruits and nuts, field crops (including corn and cotton), grain crops, alfalfa, vegetables and strawberries, and grapes

[4] The benchmark of 35 kg N/ha (31 lb N/ac) is not adequate for percolation basins, as their recharge rate is much more than 1 AF/ac. Acual average concentration (by county) of nitrogen in FP and WWTP discharges to percolation basins range from 2 to 10 times the MCL and 1 to 2 times the MCL, respectively (see Section 6).

Adminsitrative Record
Page 34286

(see Figure 2). The study area also supports 1 million dairy cows. These produce one-tenth of the nation's milk supply as well as large amounts of manure.

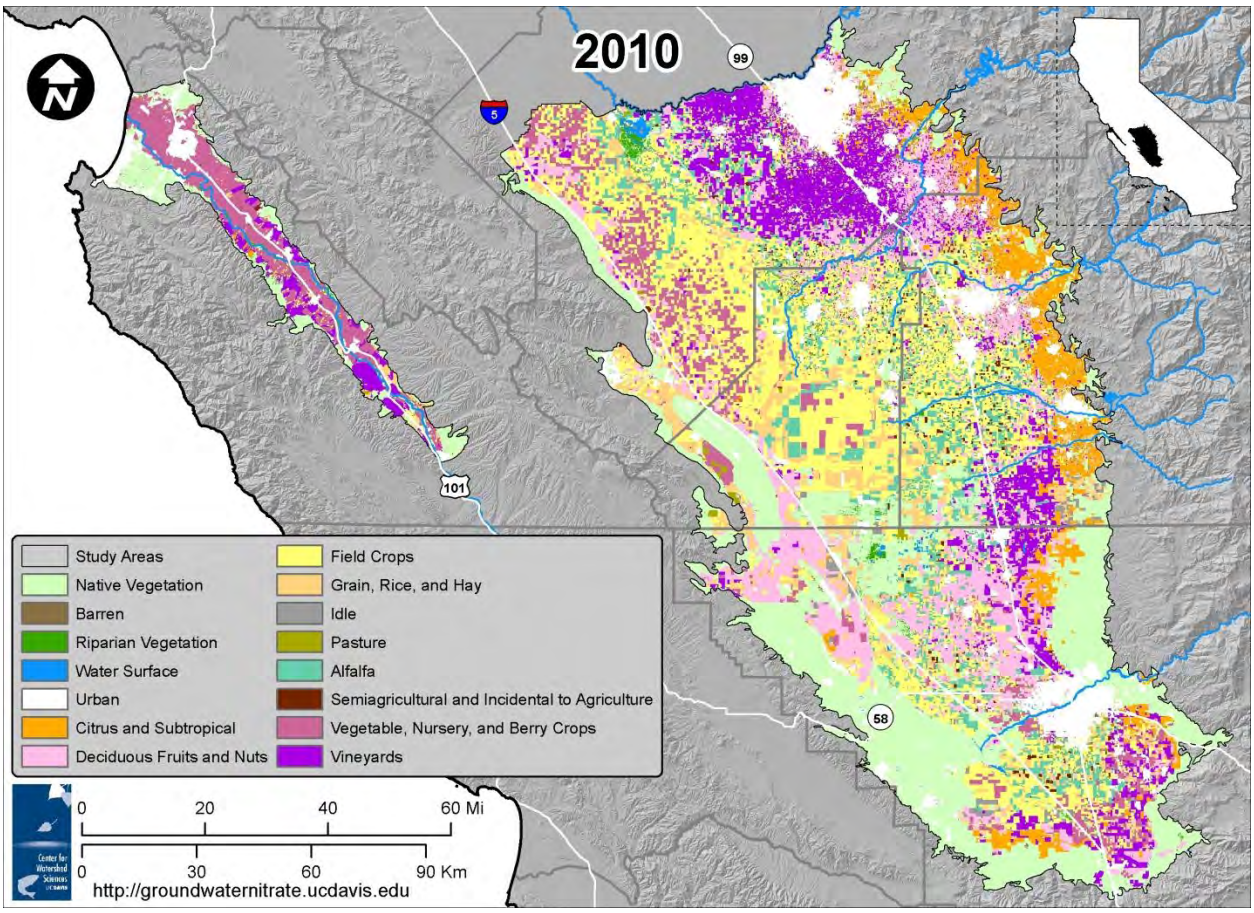

Figure 2. The Tulare Lake Basin (TLB) and Salinas Valley (SV) are the focus of this study. The study area represents 40% of California's diverse irrigated agriculture and more than half of its confined animal farming industry. It is home to 2.6 million people, with a significant rural population in economically disadvantaged communities. Spatial distribution of crop and other land use categories based on CAML data (see Section 3).

Cropland sources: Alfalfa. The mass balance approach is not applied to alfalfa because it does not receive significant amounts of fertilizer, while fixing large amounts of nitrogen from the atmosphere. Little is known about nitrate leaching from alfalfa; we used a reported value of 30 kg N/ha/yr (27 lb N/ac/yr) (Letey et al., 1979; Robbins et al., 1980, see Appendix Table 1). In total, 170,000 ha (420,000 ac) of alfalfa fields are estimated to contribute about 5 Gg N/yr (5,000 t N/yr) in the study area. Alfalfa harvest exceeds 400 kg N/ha/yr (360 lb N/ac/yr), or 74 Gg N/yr (82,000 t N/yr), in the study area.

Cropland sources other than alfalfa. Unlike other groundwater nitrate source categories, cropland has many sources of nitrogen application, all of which can contribute to nitrate leaching. Principally, crops are managed for optimal harvest. Synthetic nitrogen is the fertilizer of choice to achieve this goal, except in alfalfa and a few other leguminous crops (e.g., beans). Other sources of nitrogen are also applied to cropland, providing additional fertilizer, serving as soil amendments, or providing a means of waste disposal. These additional nitrogen sources include animal manure and effluent and biosolids from

Adminsitrative Record
Page 34287

WWTPs, FPs, and other urban sources. Often do they replace synthetic fertilizer as the main source of nitrogen for a crop. Atmospheric deposition of nitrogen and nitrate in irrigation water are mostly incidental but ubiquitous.

For the mass balance analysis, external nitrogen inputs to cropland are considered to be balanced over the long run (5 years and more) by nitrogen leaving the field in crop harvest, atmospheric losses (volatilization, denitrification), runoff to streams, or groundwater leaching. Hence, cropland nitrate leaching to groundwater is estimated by summing nitrogen inputs to a field (fertilizer, effluent, biosolids, manure, atmospheric deposition, irrigation water) and then subtracting the three other nitrogen outputs (harvest, atmospheric losses, and runoff).

Table 1. Major sources of groundwater nitrate, their estimated total contribution in the study area, their percent of total contribution, and their estimated average local intensity, which indicates local pollution potential. Actual total nitrate loading from these source categories is very likely within the range provided in parentheses.

	Total Nitrate Loading to Groundwater, Gg N/yr[1] (range) [1,000 t N/yr] (range)	Percent Contribution to Total Nitrate Leaching in the Study Area	Average Intensity of Nitrate Loading to Groundwater kg N/ha/yr [lbs N/ac/yr]
Cropland	195 (135 – 255) 215 (150 – 280)	93.7%	154 137
Alfalfa cropland	5 (<1 –10) 5 (<1--11)	2.4%	30 27
Animal corrals	1.5 (0.5 – 8) 1.7 (0.5 – 9)	0.7%	183 163
Manure storage lagoons	0.23 (0.2 – 2) 0.25 (0.2 – 2)	0.1%	183 163
WWTP and FP[2] percolation basins	3.2 (2 - 4) 3.5 (2 – 4)	1.5%	1,200[3] 1070
Septic systems	2.3 (1 – 4) 2.5 (1 – 4)	1.1%	<10->50 <8.8->45
Urban (leaky sewers, lawns, parks, golf courses)	0.88 (0.1–2) 0.97 (0.1 – 2)	0.5%	10 8.8
Surface leakage to wells	<0.4	—	—[4]

1. At 2011 prices, 1 Gg N (1,000 metric tons N or 1,100 t N) is roughly equivalent to $1 million in fertilizer nitrogen.
2 WWTP = wastewater treatment plant; FP = food processor.
3. The benchmark of 35 kg N/ha/yr does not apply to WWTP and FP percolation basins, which may recharge significantly more water than other sources. The nitrate loading may be high even if concentrations are below the MCL.
4. Surface leakage through improperly constructed wells is based on hypothetical estimates and represents an upper limit.

In total, the 1.27 million ha (3.1 million ac) of cropland, not including 0.17 million ha (0.4 million ac) of alfalfa, receive 380 Gg N/yr (419,000 t N/yr) from all sources. Synthetic fertilizer, at 204 Gg N/yr (225,000 t N/yr), is more than half of these inputs (Figure 3). Manure applied on dairy forages or exported for cropland applications off-dairy (but not leaving the study area) is one-third of all nitrogen inputs. Atmospheric deposition and nitrate-nitrogen in groundwater used as irrigation water are

Adminsitrative Record
Page 34288

approximately one-tenth of all nitrogen input. Urban effluent and biosolids application are small portions of the overall nitrogen input in the study area, but they are locally significant.

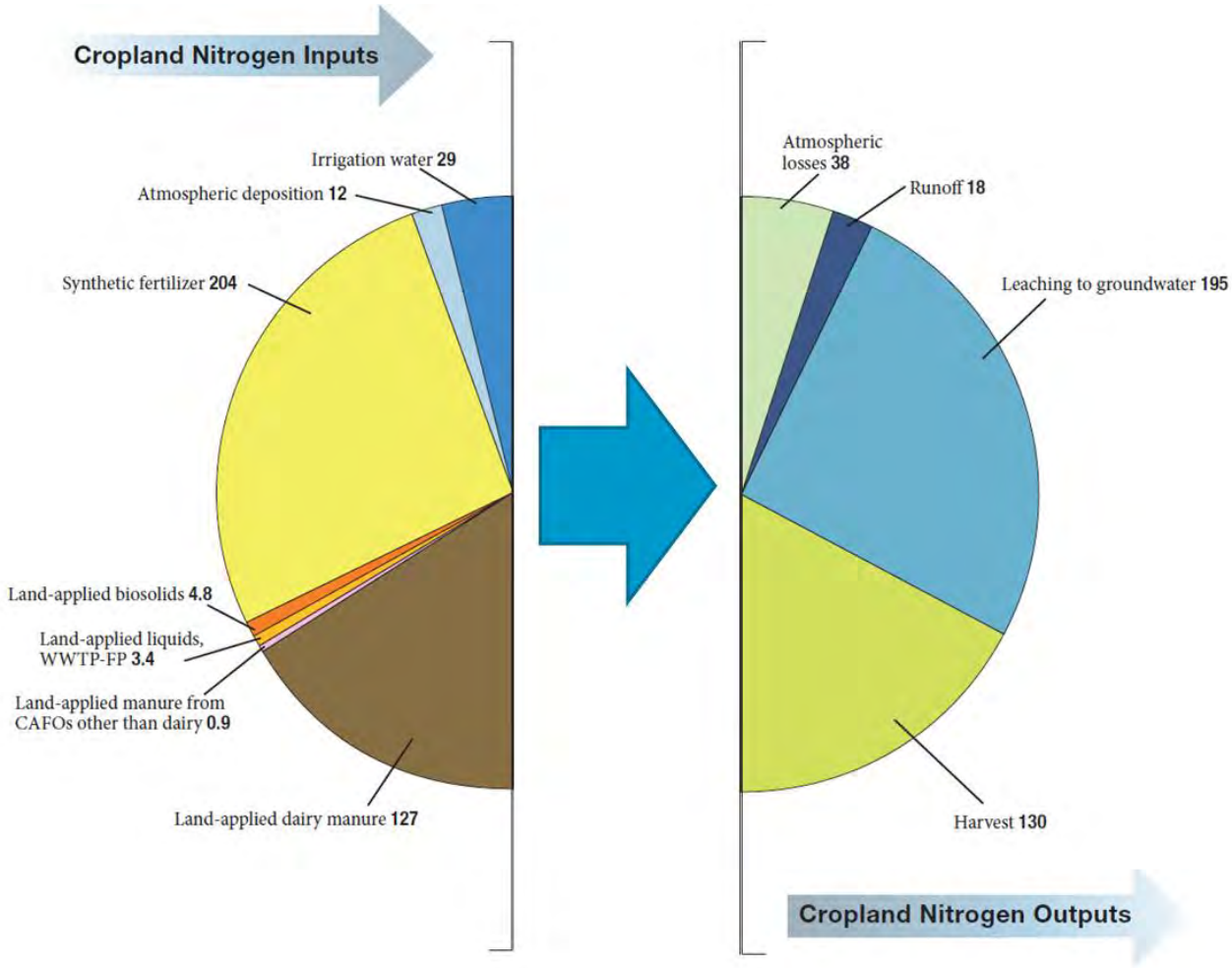

Figure 3. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. The left half of the pie chart represents total nitrogen inputs to 1.27 million ha (3.12 million ac) of cropland, not including alfalfa. The right half of the pie chart represents total nitrogen outputs with leaching to groundwater estimated by difference between the known inputs and the known outputs. No mass balance was performed on 0.17 million ha (0.4 million ac) of nitrogen-fixing alfalfa, which is estimated to contribute an additional 5 Gg N/yr to groundwater. Groundwater nitrate loading from all non-cropland sources is about 8 Gg N/yr

On the output side, the total nitrate leaching to groundwater from cropland (not including alfalfa) comprises 195 Gg N/yr (215,000 t N/yr) and is by far the largest nitrogen flux from cropland, much larger than the harvested nitrogen at 130 Gg N/yr (143,000 t N/yr). The nitrogen leached to groundwater nearly matches the amount of synthetic fertilizer applied to the same cropland, suggesting large system surpluses of nitrogen use on cropland. Other outputs are small: atmospheric losses are assumed to be one-tenth of the inputs (see Section 7), and runoff is assumed to be 14 kg N/ha/yr (12.5 lb N/ac/yr) (Beaulac and Reckhow 1982).

Adminsitrative Record
Page 34289

Applying the benchmark of 50 Gg N/yr (55,000 t N/yr), groundwater leaching losses would need to be reduced by 150 Gg N/year (165,000 t N/yr) or more area-wide to avoid further large-scale groundwater degradation. Figure 3 suggests three major options to reduce nitrate loading to groundwater from cropland: develop techniques to make manure a useful and widely used fertilizer and reduce synthetic fertilizer application in the study area by as much as 75%; drastically reduce the use of manure in the study area; or significantly increase the agricultural output (harvest) without increasing the nitrogen input. Nitrate source reduction efforts will involve a combination of these options (Dzurella et al., 2012).

The following sections further discuss individual inputs and outputs that control agricultural cropland nitrate leaching.

Cropland inputs: Synthetic fertilizer (204 Gg N/yr [225,000 t N/yr]). Synthetic fertilizer application rates are estimated by first establishing a typical nitrogen application rate for each crop, derived from the literature, United States Department of Agriculture (USDA) Chemical Usage Reports, and UC Davis ARE agricultural cost and return studies for each of 58 crop categories within 10 crop groups (Figure 4). In a second step, we assess whether some of the typical nitrogen application rate is met by other sources such as effluent, biosolids, and manure. The procedure varies with crop type, location, and aggregation level. Fertilizer needs not met by effluent, biosolids, or manure (see below) are assumed to be met by synthetic fertilizer, providing an estimate of synthetic fertilizer use at local (Figure 4), crop (see Figure 6), county (see Table 2), and study area (see Figure 3) levels. The magnitude of total estimated synthetic fertilizer use (204 Gg N/yr [225,000 t N/yr]) in the study area, on about 40% of California's irrigated land, is consistent with statewide average recorded sales of synthetic fertilizer used on cropland of 466 Gg N/yr (514,000 t N/yr) (D. Liptzin, pers. comm., 2012).

Adminsitrative Record
Page 34290

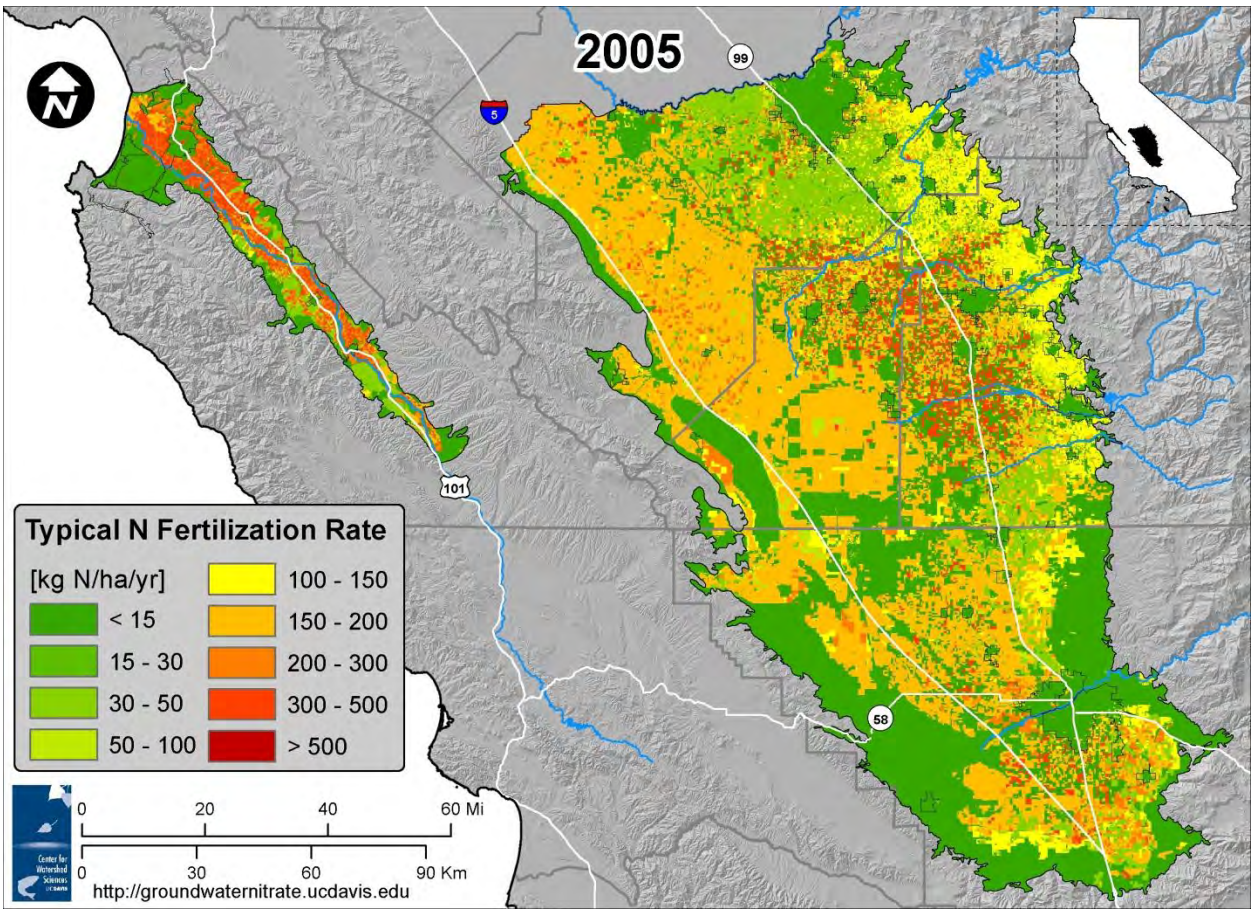

Figure 4. Current typical annual fertilization rates (1 kg/ha/yr = 1.1 lbs/ac/yr) in irrigated agricultural cropland of the study area derived from the literature, United States Department of Agriculture (USDA) Chemical Usage Reports, and agricultural cost and return studies for each of 58 crop categories (does not include manure applications). Rates account for multi-cropping in some vegetable crops and double-cropping of corn and winter grain. Spatial distribution of crop categories based on CAML data (see Section 3).

Cropland inputs: Animal manure (land-applied: 128 Gg N/yr [141,000 t N/yr]; corral and lagoon loading directly to groundwater: 1.7 Gg N/yr [1,900 t N/yr]). The Tulare Lake Basin houses 1 million adult dairy cows and their support stock (more than half of California's dairy herd), 10,000 hogs and pigs, and 15 million poultry animals. Dairy cattle are by far the largest source of land-applied manure nitrogen in the area (127 Gg N/yr [140,000 t N/yr]; see Figure 3). Manure is collected in dry and liquid forms, recycled within the animal housing area for bedding (dry manure) and as flushwater (freestall dairies), and ultimately applied to the land. Manure is applied in solid and liquid forms, typically on forage crops (e.g., summer corn, winter grain) managed by the dairy farm, or is exported to nearby farms (mostly as manure solids) and used as soil amendment. The amount of land-applied manure nitrogen is estimated based on: recently published studies of dairy cow, swine, and poultry excretion rates; animal numbers reported by the Regional Water Board and the USDA Agricultural Census; and an estimated 38% atmospheric nitrogen loss in dairy facilities before land application of the manure. Manure not exported from dairy farms is applied to portions of 130,000 ha (320,000 ac) of dairy cropland. Exported manure

Adminsitrative Record
Page 34291

nitrogen is largely applied within the study area, mostly within the county of origin, on cropland nearby dairies.

Direct leaching to groundwater from animal corrals and manure lagoons is about 1.5 Gg N/yr (1,700 t N/yr) and 0.2 Gg N/yr (220 t N/yr), respectively (see Table 1).

Cropland inputs: Irrigation water (29 Gg N/yr (32,000 t N/yr)). Irrigation water is also a source of nitrogen applied to crops. Surface irrigation water is generally very low in nitrate. Nitrate in groundwater used as irrigation water is a significant source of nitrogen but varies widely with location and time. We used average nitrate concentrations measured in wells and basin-wide estimates of agricultural groundwater pumping (Faunt 2009) to estimate the total nitrogen application to agricultural lands from irrigation water, in the range of 20 Gg N/yr (22,000 t N/yr) to 33.4 Gg N/yr (36,800 t N/yr).

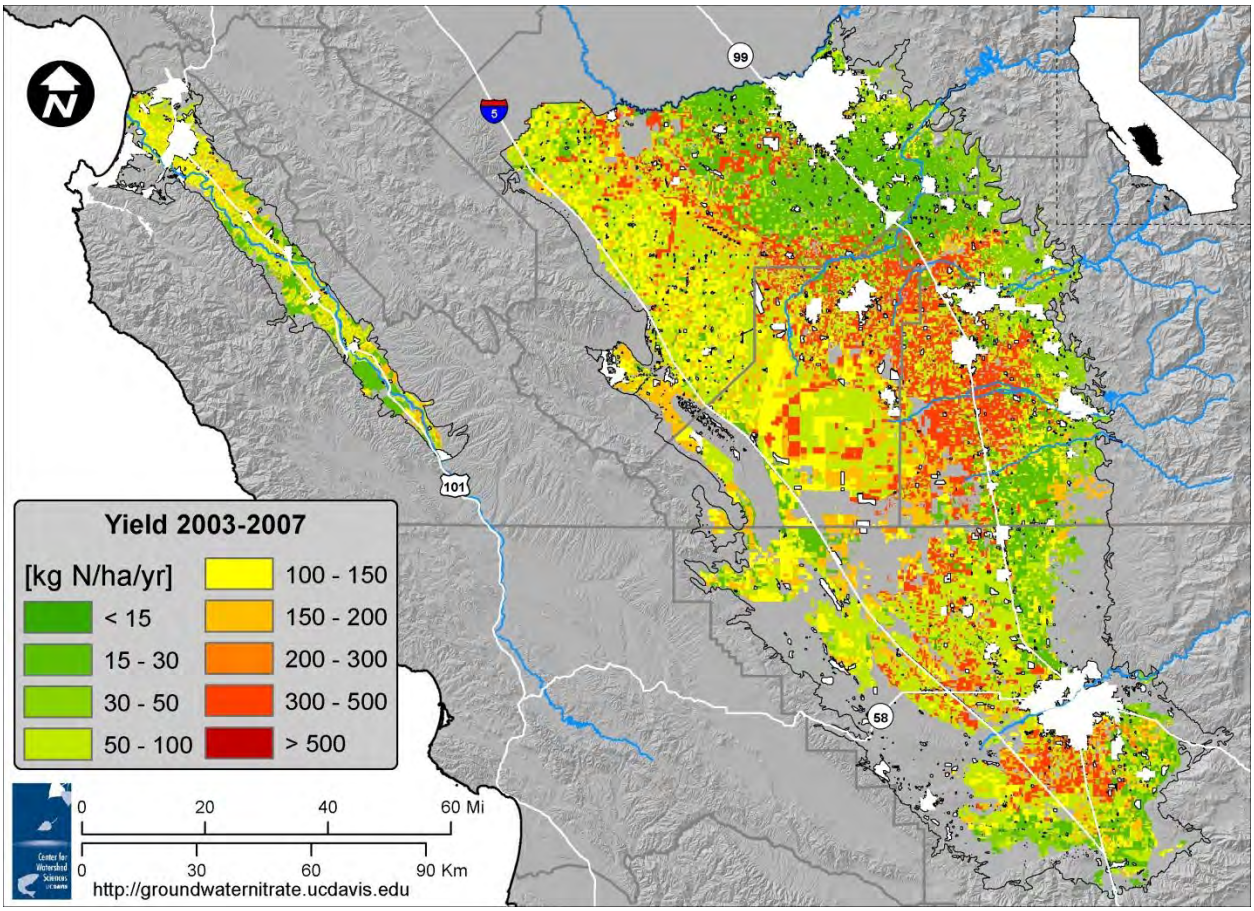

Figure 5. Current annual nitrogen removal rate in harvested materials (1 kg/ha/yr = 1.1 lbs/ac/yr) derived from county reports of harvested area and harvested tonnage for each of 58 crop categories. Rates account for multi-cropping in some vegetable crops and double-cropping of corn and winter grain. Spatial distribution of crop categories based on CAML data (see Section 3).

Cropland and general landscape inputs: Aerial deposition (12 Gg N/yr [13,000 t N/yr]). Nitrogen emissions to the atmosphere as NOx from fossil fuel combustion and ammonia from manure at confined animal feeding operations undergo transformations in the atmosphere before being redeposited, often

Adminsitrative Record
Page 34292

far from the source of emissions. Nitrogen deposition estimates at broader spatial scales are typically based on modeled data. Nitrogen deposition in urban and natural areas was assumed to be retained with the ecosystem (Vitousek and Howarth 1991). In cropland, nitrogen deposition was included in the nitrogen mass balance. For the Salinas Valley, average aerial deposition is 5.6 kg N/ha/yr (0.6 Gg N/yr) (5.0 lb N/ac [660 t N/yr]). The Tulare Lake Basin receives among the highest levels in the state, averaging 9.8 kg N/ha/yr (11.3 Gg N/yr) (8.7 lb N/ac/yr [12,500 t N/yr]).

Cropland output: Harvested nitrogen (130 Gg N/yr [143,000 t N/yr]). The nitrogen harvested is the largest independently estimated nitrogen output flow from cropland. Historical and current annual ACR data provide annual harvested acreage and yields for major crops. From the reported harvest, we estimate the nitrogen removed. For each of 58 crop categories, the study area total harvest nitrogen and total acreage used to estimate the rate of nitrogen harvested (Figure 5). All crops combined (not including alfalfa) contain a total of 130 Gg N/yr (143,000 t N/yr), with cotton (21 Gg N/yr [23,000 t N/yr]), field crops (28 Gg N/yr [31,000 t N/yr]), grain and hay crops (30 Gg N/yr [33,000 t N/yr]), and vegetable crops (30 Gg N/yr [30,000 t N/yr]) making up 85% of harvested nitrogen. Tree fruits, nuts, grapes, and subtropical crops constitute the remainder of the nitrogen export from cropland.

Groundwater loading from irrigated agriculture, by crop group and by county. Significant differences exist in groundwater loading intensity between crop groups.[5] The intensity of groundwater loading is least in vineyards (less than 35 kg N/ha/yr [31 lb N/ac/yr]), followed by rice and subtropical tree crops (about 60 kg N/ha/yr [54 lb N/ac/yr]), tree fruits, nuts, and cotton (90–100 kg N/ha/yr [80–90 lb N/ac/yr]), vegetables and berry crops (over 150 kg N/ha/yr [130 lb N/ac/yr]), which includes some vegetables being cropped twice per year), field crops (about 480 kg N/ha/yr [430 lb N/ac/yr]), and grain and hay crops (about 200 kg N/ha/yr [180 lb N/ac/yr]). Manure applications constitute the source of nearly all of the nitrate leaching from these latter two crop groups. Without manure, field crops leach less than 35 kg N/ha/yr (31 lb N/ac/yr), and grain and hay crops leach 50 kg N/ha/yr (45 lb N/ac/yr). Figure 6 shows the rate of reduction (in kg N/ha/crop) that would be needed, on average across each crop group, to reduce groundwater nitrate leaching to benchmark levels.

At the county level, we aggregate cropland area, fertilizer applications (by crop category), manure output from individual dairies, effluent and biosolid land applications from individual facilities, and crop category–specific harvest. Differences in cropping patterns between counties and the absence or presence of dairy facilities within counties are the main reason for county-by-county differences in total

[5] Aggregated estimates were obtained from study area-wide totals for harvested area (by crop group), for typical nitrogen application, and for harvested nitrogen. The following averages were assumed: irrigation water nitrogen (24 kg N/ha/yr (21 lbs N/ac/yr)), atmospheric nitrogen losses (10% of all N inputs), and runoff (14 kg N/ha/yr (12.5 lbs N/ac/yr)). Most manure is likely land applied to field crops, particularly corn, and to grain and hay crops. Little is known about the actual manure distribution prior to 2007 and the amount of synthetic fertilizer applied on fields receiving manure. As an illustrative scenario, we here assume that two-thirds of dairy manure is applied to field crops and one-third of dairy manure is applied to grain and hay crops. In field crops, 50% of crop nitrogen requirements are assumed to be met with synthetic fertilizer, in grain and hay crops 90% of their crop nitrogen requirements are assumed to be met by synthetic fertilizer. These are simplifying assumptions that neglect the non-uniform distribution of manure on field and grain crops between on-dairy, near-dairy, and away-from-dairy regions. However, corn constitutes most (106,000 ha (262,000 ac)) of the 130,000 ha (321,000 ac) in field crops, with at least 40,000 ha (99,000 ac) grown directly on dairies. Grain crops are harvested from 220,000 ha (544,000 ac). (For further detail, see Sections 3 and 4 of this Technical Report).

Adminsitrative Record
Page 34293

groundwater loading and in the average intensity of groundwater loading (Table 2). Fresno County, which has fewer mature dairy cows (133,000) than Kings (180,000), Tulare (546,000), or Kern (164,000) Counties and also has large areas of vineyards (see Figure 2), has the lowest average groundwater loading intensity (103 kg N/ha/yr [103 lb N/ac/yr]). Monterey County is dominated by vegetable and berry crops (high intensity) and grape vineyards (low intensity). The partial nutrient balance (PNB), which is the ratio of harvested N to cropland N inputs, varies from less than 35% in Tulare County to nearly 45% in Fresno County. If manure or other organic materials were applied only to within estimated typical fertilizer application rates, throughout the study area (Figure 4), then the resulting hypothetical partial nutrient balance (PNB_0) would range from nearly 45% in Monterey County to about 55% in Fresno County and Kern County, and to over 70% in Tulare County (Table 2). The difference between PNB and PNB_0 indicates the importance of accounting for all sources of nitrogen to cropland and the importance of properly managing organic nitrogen sources, especially manure.

Adminsitrative Record
Page 34294

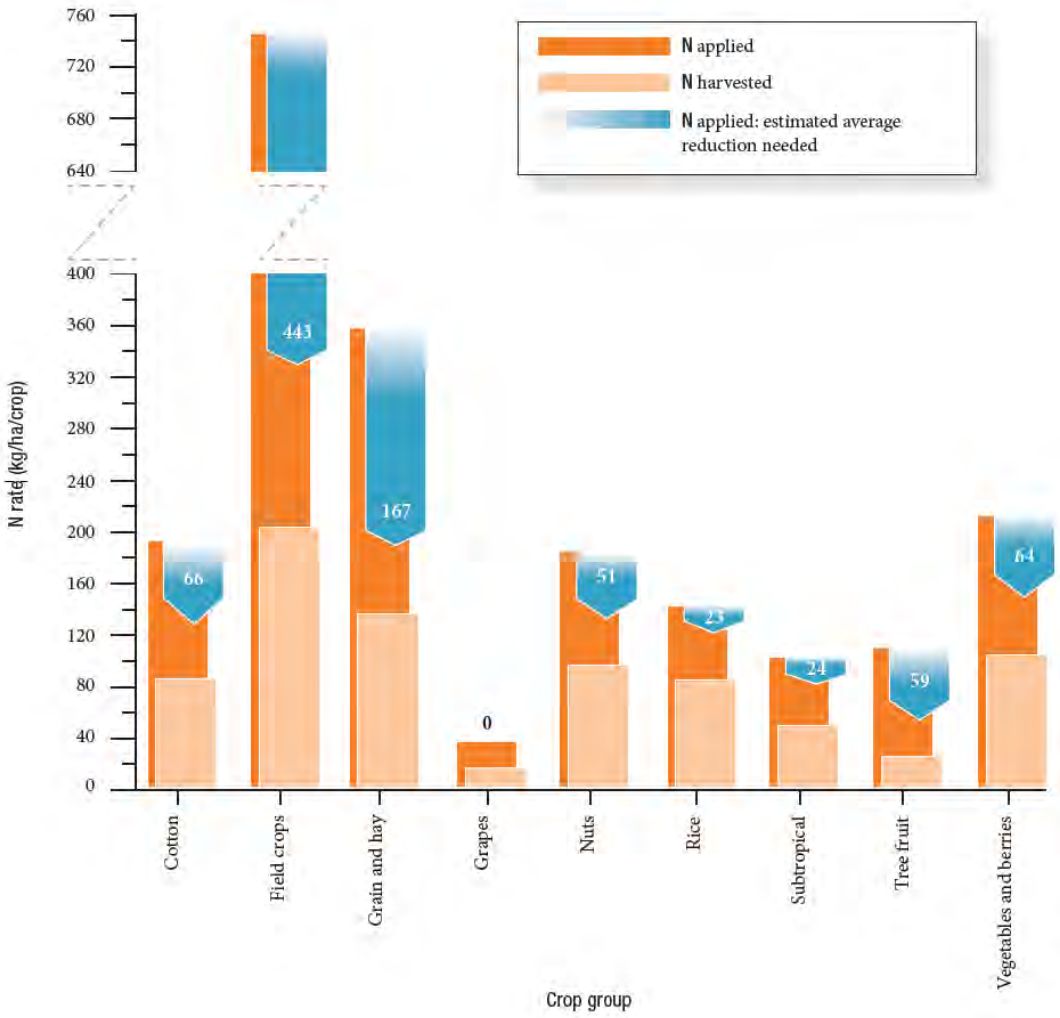

Figure 6. Nitrogen application reduction needed to reduce groundwater nitrate loading to less than 35 kg N/ha/crop, compared with average nitrogen applied (synthetic fertilizer and manure) and nitrogen harvested (all units in kg N/ha/crop). Rates are given per crop, and the required reduction does not account for double-cropping. Some vegetables and some field crops are harvested more than once per year. In that case, additional reductions in fertilizer applications would be necessary to reduce nitrate loading to less than 35 kg N/ha. Large reductions needed in field crops and grain and hay crops are due to the operational assumption that manure generated in the study area is applied to only these crop groups. Typical amounts of synthetic fertilizer applied ("N applied") to these crops, without excess manure, are 220 kg N/ha/crop for field crops and 190 kg N/ha/crop for grain and hay crops. Thus, without excess manure, average field crops and grain and hay crops may require relatively small reductions in nitrogen application.

Adminsitrative Record
Page 34295

Table 2. Major nitrogen fluxes to and from cropland in the study area, by county (not including alfalfa).

Table 2(a). Metric units.

	Synthetic Fertilizer Application Gg N/yr	Manure Application Gg N/yr	Land Applied Effluent and Biosolids Gg N/yr	Harvest Gg N/yr	PNB[1] %	PNB_0 [2] %	Ground-water Loading Gg N/yr	Ground-water Loading Intensity kg N/ha/yr
By County								
Fresno	62.1	16.6	0.8	35.5	44.7	54.4	42.4	103
Kern	50.3	20.4	4.6	29.6	39.3	56.4	42.8	141
Kings	27.5	22.0	1.9	19.6	38.1	62.7	29.2	179
Tulare	36.0	67.3	0.7	32.7	31.4	72.5	65.1	236
Monterey	28.1	1.4	0.1	12.4	41.9	43.5	15.6	138
By Basin								
TLB	176	127	8.1	118	37.8	60.5	179	155
SV	28	1	0.1	12	41.9	43.5	16	138
Overall	**204**	**128**	**8.2**	**130**	**38.2**	**58.3**	**195**	**154**

1. PNB = partial nutrient balance, here defined as Harvest N ÷ (Synthetic + Manure + Effluent +Biosolids Fertilizer N).
2. PNB_0 = hypothetical PNB, if no manure/effluent/biosolids overage is applied above typical fertilizer rates.
Note: Manure applications include non-dairy manure nitrogen (0.9 Gg N/yr (990 t N/yr) for the entire study area). Groundwater loading accounts for atmospheric deposition (9.8 and 5.6 kg N/ha/yr (8.7 and 5.0 lbs N/ac/yr) in TLB and SV, respectively), atmospheric losses (10% of all inputs), irrigation water quality (22.8 kg N/ha/yr (20 lbs N/ac/yr)), and runoff (14 kg N/ha/yr (12.5 lbs N/ac/yr)) to and from agricultural cropland, in addition to fertilizer and manure application, and harvested nitrogen. Synthetic fertilizer application on field crops is assumed to meet 50% of typical application rates; on grain and hay crops, 90% of typical applications, with the remainder met by manure.

Table 2(b). US Standard units

	Synthetic Fertilizer Application 1,000 t N/yr	Manure Appli-cation 1,000 t N/yr	Land Applied Effluent and Biosolids 1,000 t N/yr	Harvest 1,000 t N/yr	PNB[1] %	PNB_0 [2] %	Ground-water Loading 1,000 t N/yr	Ground-water Loading Intensity lb N/ac/yr
By County								
Fresno	68.3	18.3	0.88	39.1	44.7	54.4	46.7	92
Kern	55.4	22.5	5.0	32.6	39.3	56.4	47.2	123
Kings	30.3	24.3	2.1	21.6	38.1	62.7	32.2	160
Tulare	39.7	74.2	0.77	36.0	31.4	72.5	71.8	210
Monterey	30.9	1.54	0.11	13.6	41.9	43.5	17.2	123
By Basin								
TLB	194	140	8.9	130	37.8	60.5	197	138
SV	30.8	1.1	0.11	13	41.9	43.5	18	123
Overall	**225**	**141**	**9**	**143**	**38.2**	**58.3**	**215**	**137**

1 & 2. See notes in metric unit table (2a) above

Adminsitrative Record
Page 34296

1.6.2 Historical Development of Fertilizer Use, Manure Production, Harvested Nitrogen, and Estimated Nitrate Leaching to Groundwater.

Current and near-future groundwater nitrate conditions are mostly the result of past agricultural practices. So the historical development of nitrogen fluxes to and from cropland provides significant insight in the relationship between past agricultural practices, their estimated groundwater impacts, and current as well as anticipated groundwater quality. Two major inventions effectively doubled the farmland in production from the 1940s to the 1960s: the introduction of the turbine pump in the 1930s, allowing access to groundwater for irrigation in a region with very limited surface water supplies, and the invention and commercialization of the Haber-Bosch process, which made synthetic fertilizer widely and cheaply available by the 1940s.

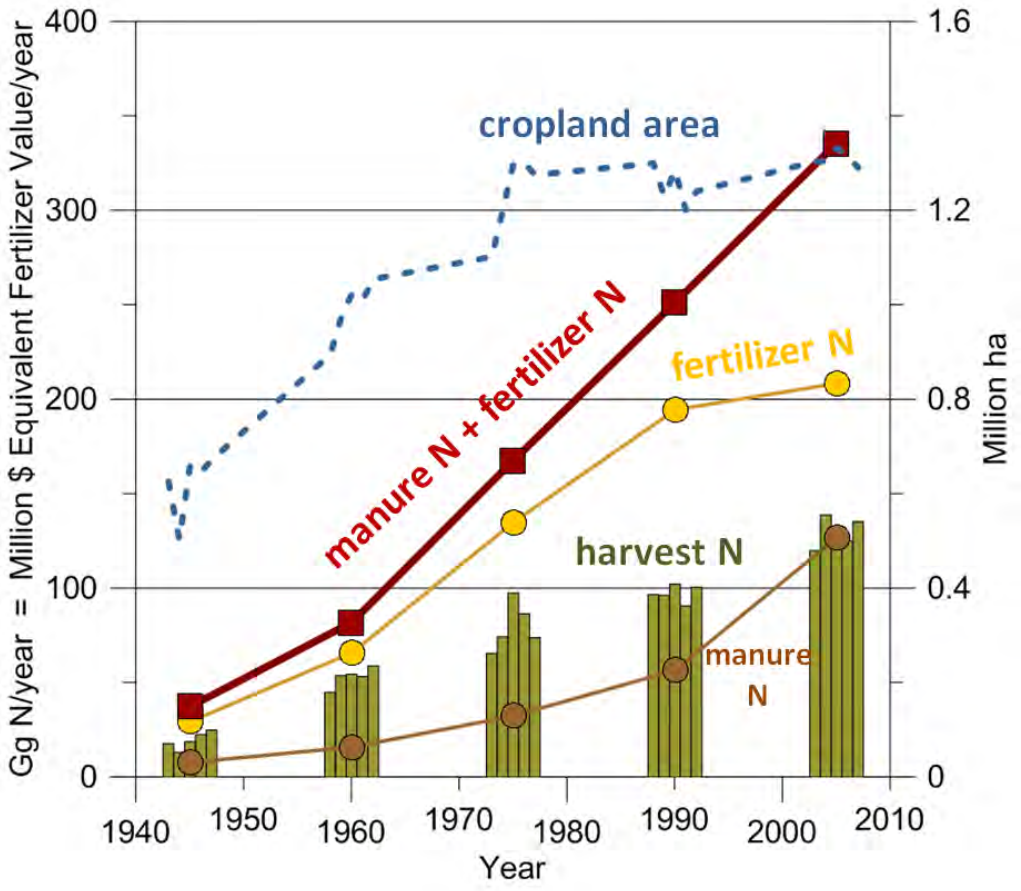

Figure 7. Estimated historical agricultural development in the study area (not including alfalfa): total harvested area, total harvested nitrogen in fertilized crops, fertilizer applied to cropland (5 year average), manure applied to cropland (5 year average), and sum of manure and fertilizer applied to cropland (5 year average).[6] Note: 0.4 million ha = 1 million ac.

[6] Not shown: In the study area, harvested alfalfa area grew from 0.12 million ha (0.3 million ac) in the 1940s to 0.2 million ha (0.5 million ac) around 1960, then leveled off to current levels of 0.17 million ha (0.42 million ac). Since the 1960s, nitrogen removal in alfalfa harvest has varied from 50 to 80 Gg N/yr.

Adminsitrative Record
Page 34297

The amount of cropland (not including alfalfa) in the study area nearly doubled in less than 20 years, from 0.6 million ha (1.5 million ac) in the mid-1940s to nearly 1.0 million ha (2.5 million ac) in 1960 (Figure 7). Further increases occurred until the 1970s, to 1.3 million ha (3.2 million ac), but the extent of farmland has been relatively stable for the past 30 years.

In contrast, the harvested nitrogen has consistently increased throughout the past 60 years (Figure 7). From 1945 to 1975, total harvested nitrogen increased twice as fast as farmland expansion, quadrupling from 20 Gg N/ yr (22,000 t N/yr) to 80 Gg N/yr (88,000 t N/yr). Without further increases in farmland, harvests and harvested nitrogen increased by more than 60% in the second 30-year period, from the mid-1970s to the mid-2000s.

Synthetic fertilizer inputs also increased from the 1940s to the 1980s but have since leveled off. Between 1990 and 2005, the gap between synthetic nitrogen fertilizer applied and harvested nitrogen has significantly decreased.[7]

In contrast, dairy manure applied to land has increased exponentially, effectively doubling every 15 years (see Figure 7), from 8 Gg N/yr (9,000 t N/yr) in 1945 to 16 Gg N/yr (18,000 t N/yr) in 1960, 32 Gg N/yr (35,000 t N/yr) in 1975, 56 Gg N/yr (62,000 t N/yr) in 1990, and 127 Gg N/yr (140,000 t N/yr) in 2005, an overall 16-fold increase in manure nitrogen output. The increase in manure nitrogen is a result of increasing herd size (7-fold) and increasing milk production per cow (3-fold) and is slowed only by the increased nitrogen-use efficiency of milk production.

Until the 1960s, most dairy animals in the region were only partly confined, often grazing on irrigated pasture with limited feed imports. Manure from dairy livestock generally matched the nitrogen needs of dairy pastures. Since the 1970s, dairies in the Tulare Lake Basin have operated mostly as confined animal facilities, growing alfalfa, corn, and grain feed on-site, importing additional feed, and housing the animals in corrals and freestalls. The growth in the dairy industry has created a nitrogen excess pool that remains unabsorbed by crops (see Figure 7). Much of the nitrogen excess is a recent phenomenon (see Figure 7). With groundwater quality impacts delayed by decades in many production wells (Boyle et al., 2012), the recent increase in land applied manure nitrogen is only now beginning to affect water quality in wells of the Tulare Lake Basin, with much of the impact yet to come.

1.6.3 Urban and Domestic Sources

Urban and domestic sources: Overview. Urban nitrate loading to groundwater is divided into four categories: nitrate leaching from turf, nitrate from leaky sewer systems, groundwater nitrate contributions from WWTPs and FPs, and groundwater nitrate from septic systems. For all these systems, groundwater nitrate loading is estimated based on either actual data or reported data of typical nitrate leaching.

Urban and domestic sources: Wastewater treatment plants and food processors (11.4 Gg N/yr [12,600 t/yr]: 3.2 Gg N/yr [3,500 t/yr] to percolation ponds, 3.4 Gg N/yr [3,800 t/yr] in effluent applications to

[7] Fertilizer application rates and statewide fertilizer sales have grown little since the late 1980

Adminsitrative Record
Page 34298

cropland, and 4.8 Gg N/yr [5,300 t/yr] in WWTP biosolids applications to cropland). The study area has roughly 2 million people on sewer systems that collect and treat raw sewage in WWTPs. In addition, many of the 132 food processors within the study area generate organic waste that is rich in nitrogen (Table 3). Potential sources of groundwater nitrate contamination from these facilities include effluent that is land applied on cropland or recharged directly to groundwater via percolation basins, along with waste solids and biosolids that are land applied. Typically, WWTP influent contains from 20 mg N/L to 100 mg N/L total dissolved nitrogen (organic N, ammonium N, nitrate-N), of which little is removed in standard treatment (some WWTPs add treatment beyond conventional processes to remove nutrients including nitrate and other forms of nitrogen). Across the study area, WWTP effluent nitrogen levels average 16 mg N/L. Within the study area, 40 WWTPs treat 90% of the urban sewage. FP effluent nitrogen levels to percolation basins and irrigated agriculture average 42 mg N/L and 69 mg N/L, respectively.

Table 3. Total nitrogen discharge to land application and average total nitrogen concentration (as nitrate-N, MCL: 10 mg N/L) in discharge to percolation basins from WWTPs and FPs, based on our surveys of WWTPs and the FP survey of Rubin et al. (2007).

Table 3(a). Metric units.

	Biosolids	WWTP Land Application	WWTP Percolation Concentration	FP Land Application	FP Percolation Concentration
By County	Gg N/yr	Gg N/yr	mg N/L	Gg N/yr	mg N/L
Fresno	0.006	0.40	18.5	0.42	56.2
Kern	3.1	0.92	17.7	0.56	43.9
Kings	1.6	0.09	11.2	0.26	2.1
Tulare	0.038	0.50	14.9	0.13	34.2
Monterey	0	0.09	13.9	0.05	22.1
By Basin					
Tulare Lake Basin	4.8	1.9	16.3	1.37	43.3
Salinas Valley Basin	0	0.09	13.9	0.05	22.1
Overall Average	**4.8**	**2.0**	**16**	**1.4**	**42**

Adminsitrative Record
Page 34299

Table 3(b). US standard units.

	Biosolids	WWTP Land Application	WWTP Percolation Concentration	FP Land Application	FP Percolation Concentration
By County	1,000 t N/yr	1,000 t N/yr	mg N/L	1,000 t N/yr	mg N/L
Fresno	0.006	0.40	18.5	0.46	56.2
Kern	3.4	0.92	17.7	0.62	43.9
Kings	1.7	0.09	11.2	0.29	2.1
Tulare	0.044	0.50	14.9	0.14	34.2
Monterey	0	0.09	13.9	0.05	22.1
By Basin					
Tulare Lake Basin	5.3	2.1	16.3	1.51	43.3
Salinas Valley Basin	0	0.09	13.9	0.05	22.1
Overall Average	**5.3**	**2.2**	**16**	**1.5**	**42**

Urban and domestic sources: Septic systems (2.3 Gg N/yr [2,500 t N/yr]). Crites and Tchobanoglous (1998) estimated that the daily nitrogen excretion per adult is 13.3 g. Approximately 15% of that nitrogen is assumed to either stay in the septic tank, volatilize from the tank, or volatilize from the septic leachfield (Siegrist et al. 2000). Based on census data, the number of people on septic systems in the study areas is about 509,000 for the Tulare Lake Basin and 48,300 for Salinas Valley. Total nitrate loading from septic leaching is 2.1 Gg N/yr (2,300 t N/yr) in the Tulare Lake Basin and 0.2 Gg N/yr (220 t N/yr) in the Salinas Valley. The distribution of septic systems varies greatly. The highest density of septic systems is in peri-urban (rural sub-urban) areas near cities but outside the service areas of the wastewater systems that serve those cities (Figure 8). In the Tulare Lake Basin and Salinas Valley, 7.9% and 12.6%, respectively, of the land area exceeds the EPA-recommended threshold of 40 septic systems per square mile (0.154 systems per ha). Nearly 1.5% of the study area has a septic system density of over 256 systems per square mile (1 system/ha, or 1 system/2.5 ac). In those areas, groundwater leaching can significantly exceed our operational benchmark rate of 35 kg N/ha/yr (31 lb N/ac/yr).

Adminsitrative Record
Page 34300

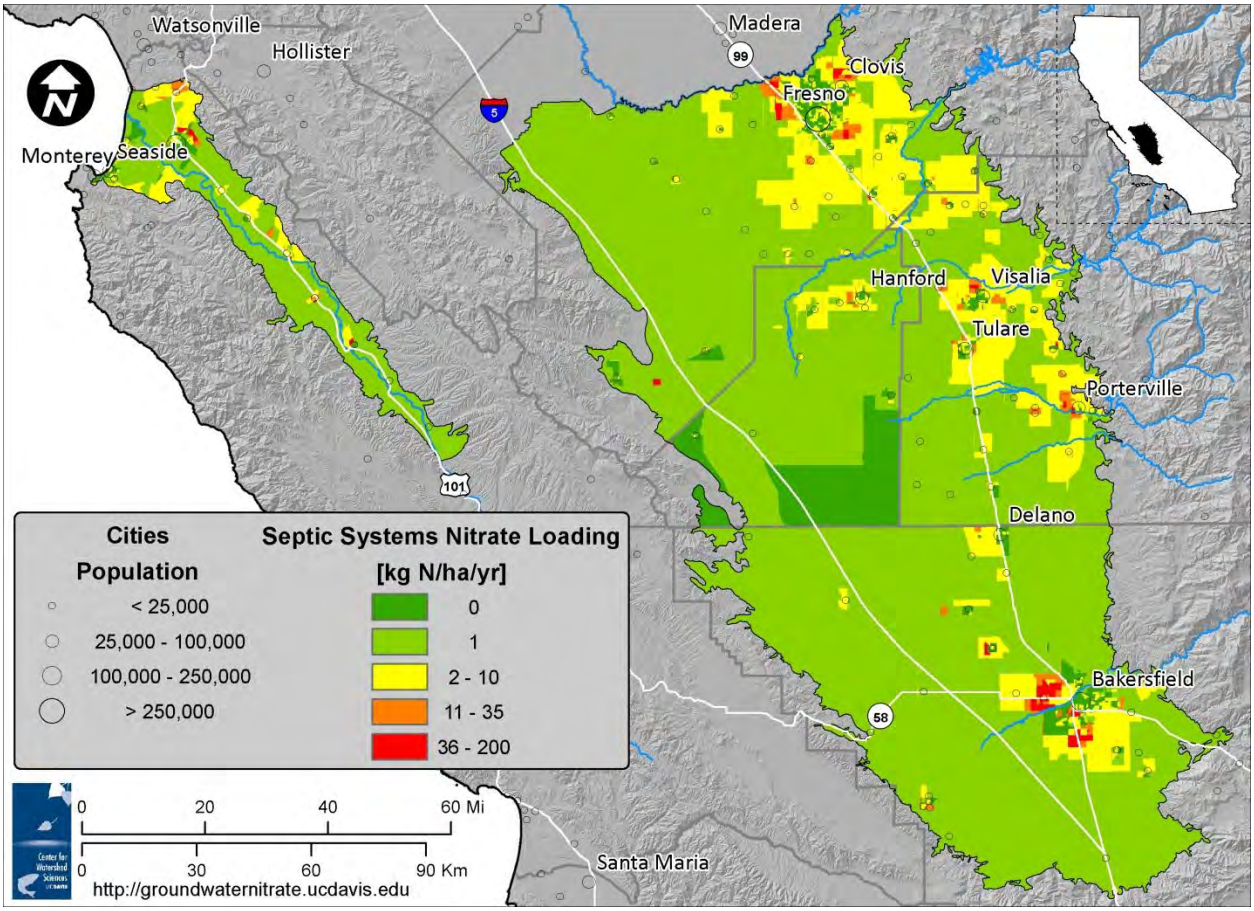

Figure 8. Septic-derived nitrate leaching rates within the study area.

Urban and domestic sources: Fertilizer and leaky sewer lines (0.88 Gg N/yr [970 t N/yr]). Fertilizer is used in urban areas for lawns, parks, and recreational facilities such as sports fields and golf courses. These land uses differ in their recommended fertilizer use, and there is almost no evidence of actual fertilization rates. Based on the most comprehensive survey of turfgrass leaching, only about 2% of applied nitrogen fertilizer was found to leach below the rooting zone (Petrovic 1990). For our nitrogen flow calculations, we assume a net groundwater loss of 10 kg N/ha/yr (8.9 lb N/ac/yr) from lawns and golf courses in urban areas (0.35 Gg N/yr [380 t N/yr]).

Sewer systems in urban areas can be a locally significant source of nitrogen. We use both reported sewer nitrogen flows and per capita nitrogen excretion rates to obtain total nitrogen losses via leaky sewer lines in urban areas. Nationally, estimated municipal sewer system leakage rates range from 1% to 25% of the total sewage generated. Given that much of the urban area within the study region is relatively young, we consider that the leakage rate is low, roughly 5% or less (0.53 Gg N/yr).

Adminsitrative Record
Page 34301

1.6.4 Wells, Dry Wells, and Abandoned Wells as Sources

Wells, dry wells, and abandoned wells (<0.4 Gg N/yr [<440 t N/yr]). Wells contribute to groundwater nitrate pollution through several potential pathways. Lack of or poor construction of the seal between the well casing and the borehole wall can lead to rapid transport of nitrate-laden runoff or irrigation water from the surface into the aquifer. In an inactive or abandoned production well, long well screens (several hundred feet) extending from relatively shallow depth to greater depth, traversing multiple aquifers, may cause water from nitrate-contaminated shallow aquifer layers to pollute deeper aquifer layers, at least in the vicinity of wells. Dry wells, which are large-diameter gravel-filled open wells, were historically designed to capture stormwater runoff or irrigation tailwater for rapid recharge to groundwater. Abandoned wells also allow surface water leakage to groundwater (spills) and cross-aquifer contamination. Lack of backflow prevention devices can lead to direct introduction of fertilizer chemicals into the aquifer via a supply well. Few data are available on these types of nitrate transfer in the Tulare Lake Basin or Salinas Valley. In a worst-case situation, as much as 0.4 Gg N/yr (440 t N/yr) may leak from the surface to groundwater via improperly constructed, abandoned, or dry wells, and as much as 6.7 Gg N/yr (7,400 t N/yr) are transferred within wells from shallow to deeper aquifers. Actual leakage rates are likely much lower than these worst-case estimates.

1.6.5 Groundwater Nitrate Loading: Sources of Uncertainty

The analyses above provide specific numbers for the average amount and intensity of nitrate loading from various categories of sources. However, discharges of nitrate to groundwater may vary widely between individual fields, farms, or facilities of the same category due to differences in operations, management practices, and environmental conditions. Also, average annual nitrate loading estimates for specific categories are based on many assumptions and are based on (limited) data with varying degrees of accuracy; the numbers given represent a best, albeit rough, approximation of the actual nitrate loading from specific sources. These estimates have inherent uncertainty. Very likely, though, the actual groundwater nitrate loading from source categories falls within the ranges shown in Table 1.

The range estimates about the loading rates to groundwater, given in Table 1, are explained in more detail in Section 3 (alfalfa), Section 4 (land applied manure N, animal corrals and manure storage lagoons), Sections 5 and 6 (urban sources including land applied N), and Section 9 (surface leakage to wells). For groundwater nitrate loading from cropland, the range estimate was based on an error analysis of the mass balance shown in Figure 3. The error analysis was performed using Monte Carlo simulation.[8] The analysis indicates that, with a 95% likelihood, groundwater N loading from cropland

[8] The error analysis of the study area wide mass balance was implemented using Monte Carlo simulation: Ten thousand random trials of the mass balance terms shown in Figure 3 were performed by computer simulation. For each random trial in the simulation, individual mass balance terms, except "leaching to groundwater", were randomly drawn from normal distributions with means equal to the individual N flux rates shown in Figure 3 and listed in the subsections above (Sections 1.6.1 and 1.6.3). The standard deviations of the normal distributions from which these random N flux rates were drawn were set based on an estimated accuracy of the overall study area N fluxes shown in Figure 3: Study area wide estimates for total synthetic fertilizer N application, total manure N land application, and total harvested N are assumed, at the 95% confidence level, to be within ±20% of the true value (standard deviation: 10%). Study area wide total WWTP/FP nitrogen land application, total atmospheric N deposition, total irrigation water N, total runoff N, and total atmospheric N losses are assumed, at the 95% confidence level,

Adminsitrative Record
Page 34302

(not including alfalfa) is in the range of 135 Gg N/yr to 255 Gg N/yr (about three to five times the operational benchmark of 50 Gg N/yr). The uncertainty about total N loading from cropland is dominated by the uncertainty about the largest terms in the mass balance, the total fertilizer N application, the total manure N application, and the total N harvest.

1.6.6 Validation of Groundwater Nitrate Loading Estimates with Field Data

The California Nitrogen Assessment[9] performed an analysis of field research on nitrate leaching to groundwater from various crops (see Section 3 for a summary). From a review of numerous field studies, a median groundwater nitrate loading rate was obtained. Multiplying the statewide acreage of cropland with the average field experiment-derived loading rates, the estimated statewide groundwater nitrate leaching from cropland is estimated to be 333 Gg N/yr or about 40% of all nitrogen inputs to cropland (Liptzin, personal communication, 2012). It is likely, that field experiments do not include a significant number of experiments with crops fertilized with large amounts of manure. In the study area, at least until the late 2000s, extremely high manure application rates occurred on less than 10% of the study area (the area under management by dairy facilities). The CNA estimate of 333 Gg N/yr groundwater leaching would not account for additional losses due to application of excess manure nitrogen. If we assume that statewide land application of manure amounts to be at least 200 Gg N/yr, and if we further assume that half of that N is applied in excess of typical fertilizer rates and therefore leached to groundwater, statewide groundwater nitrate leaching from cropland is on the order of 430 Gg N/yr. The study area represents over 40% of the statewide irrigated cropland area and more than 50% of its dairy herd, hence the study area fraction of the 430 Gg N/yr leaching loss would be over 180 Gg N/yr. This estimate is, roughly, based on leaching study estimates using the stated assumptions about manure N losses to groundwater. This value is within 10% of the total groundwater nitrate loading estimated from the mass balance analysis, and well within the confidence interval of the mass balance derived groundwater loading estimate for the study area, derived in the previous section.

1.7 Comparative Analysis of Cropland Loading

In this section, we compare the land areas designated for cropland and estimated from three different data sources. The data are not directly comparable as there is neither one particular year nor one specific area for which these data could be compared. Hence, the comparison is not of the highest quantitative accuracy. Spatially, the ACR data and the NASS Agricultural Census represent the entire counties, while the CAML data represent only the study area portion of these five counties. While

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to be within ±40% of the true value (standard deviation: 20%). The individual mass balance terms were assumed to be independent from each other. For each random trial, groundwater nitrate loading ("leaching to groundwater" in Figure 3) was obtained from mass balance, that is, as the difference between total inputs (each drawn randomly) and the (randomly drawn) outputs to atmospheric losses, runoff, and harvest. Thus, we obtained 10,000 randomized estimates of study area wide groundwater nitrate loading, reflecting the uncertainty in the various N flux terms of the mass balance. The groundwater nitrate loading values are normally distributed with a mean of 195 Gg N/yr and a standard deviation of 30 Gg N/yr. The 95% confidence interval for groundwater nitrate loading, which is the reported range, is determined by subtracting twice the standard deviations from the mean value to obtain the lower bound and by adding twice the standard deviation to the mean to obtain the upper bound).

[9] http://nitrogen.ucdavis.edu

Adminsitrative Record
Page 34303

almost all cropland of these five counties is in the study area, Kern County data reported by NASS and the ACR include significant crop acreage outside of the study area (in the high desert region of Kern County). Furthermore, the ACR data and the NASS Agricultural Census data represent the harvested land area, not the land area on the ground. "Harvested land area" represents the product of land area and the number of times that land area was harvested. The harvested land area on a triple cropped field is three times the size of the field itself. Multi-cropping is dominant almost exclusively among some vegetable crops and, to a lesser degree, on corn crops rotated with winter grains (i.e., double-cropping).

We selected datsets from the nearest years for comparison (Table 4). The most recent DWR landuse survey year set the year of interest. The closest NASS agricultural census year to that DWR survey and the digitized ACR year closest to the NASS Agricultural Census year were chosen for comparison to the CAML data. Up to seven year time difference occurred between the datasets chosen for comparison, which explains some, but not all of the discrepancies in the cropped land area.

Table 4. Reporting years, by county, used to compare harvested land area in each county by three different data sources. The CAML year is the year during which the Department of Water Resources last recorded and mapped land use distribution. The NASS Agricultural Census data were taken from the year closest to the CAML reference year. The ACR data were taken from the year closest to the CAML reference year (we only digitized and processed selected ACR data for this study, including 2003-2007 and 1987-1992).

County	ACR (Agricultural Commissioner Reports)	NASS Agricultural Census	CAML
Fresno	2003	2002	2000
Kern	2007	2007	2006
Kings	2003	2002	2003
Monterey	1992	1997	1997
Tulare	1992	1997	1999

Generally, the CAML data are in very good agreement with the ACR data – slightly lower than those reported in Fresno and Tulare County and about 5% larger in Kern County, despite the fact that Kern County has some land area outside of the study area. In Kings County, CAML maps report a significantly larger land area in crop production than the ACR data. In Monterey County, CAML shows only about three-quarters of the cropland production that the ACR data show. In Monterey County, where lettuce and other vegetables are frequently double-cropped, the large discrepancy reflects the difference between "harvested cropland" and actual "on-the-ground" cropland area (Table 5).

Adminsitrative Record
Page 34304

Table 5. Comparison of harvested land area in each county, as reported by three different data sources. Data are aggregated from crop-category specific data, by county. The county data used are from the calendar years indicated for the corresponding county – data source in Table 4.
Table 5(a). Metric units.

County	ACR (Agricultural Commissioner Reports) [ha]	*NASS Agricultural Census [ha]*	CAML [ha]
Fresno	479,021	417,437	470,250
Kern	377,980	231,511	397,480
Kings	207,750	152,569	244,243
Monterey	121,476	143,493	93,899
Tulare	313,382	216,840	309,096
TOTAL	**1,499,610**	**1,161,851**	**1,514,968**

Table 5(b). US standard units.

County	ACR (Agricultural Commissioner Reports) [ac]	NASS Agricultural Census [ac]	CAML [ac]
Fresno	1,183,687	1,031,509	1,162,013
Kern	934,009	572,076	982,194
Kings	513,361	377,006	603,538
Monterey	300,174	354,579	232,029
Tulare	774,384	535,823	763,793
TOTAL	**3,705,617**	**2,870,996**	**3,743,567**

The NASS Agricultural Census data indicate a significantly lower amount of land in production, when compared to the other two sources: only 1.2 million ha as opposed to 1.5 million ha reported by the other two sources. It is unclear, why there is such a significant discrepancy in total harvested area between the ACR and CAML data on one hand and the NASS data on the other hand.

Adminsitrative Record
Page 34305

Table 6 . Comparison of harvested land area for each major crop group, as reported by three different data sources. The data were aggregated from crop category-specific data in each county, with county data taken from the year indicated by the corresponding county – data source in Table 4. Since crop-groups are integrated across counties, these land areas do not represent a specific year and are computed here for a "best-possible" comparison between the three data sources only.

Table 6(a). Metric units.

Crop Group	ACR (Agricultural Commissioner Reports) [ha]	NASS Agricultural Census [ha]	CAML [ha]
Subtropical	88,696	78,917	101,697
Treefruit	70,979	55,212	88,102
Nuts	126,879	132,119	186,088
Cotton	259,284	222,140	244,624
Field Crops	144,078	29,042	200,913
Haylage	237,429	133,127	156,031
Alfalfa	145,869	124,197	149,076
Rice	2,098	2,449	5
Vegetables	262,596	210,359	178,583
Grapes	161,701	174,287	209,849
TOTAL	**1,499,610**	**1,161,851**	**1,514,968**

Table 6(b). US standard units.

Crop Group	ACR (Agricultural Commissioner Reports) [ac]	NASS Agricultural Census [ac]	CAML [ac]
Subtropical	219,173	195,008	251,299
Treefruit	175,393	136,432	217,705
Nuts	313,525	326,473	459,833
Cotton	640,705	548,920	604,479
Field Crops	356,024	71,764	496,467
Haylage	586,700	328,964	385,561
Alfalfa	360,450	306,897	368,375
Rice	5,184	6,052	12
Vegetables	648,889	519,808	441,288
Grapes	399,572	430,673	518,548
TOTAL	**3,705,617**	**2,870,996**	**3,743,567**

When we compare the land area (acreage) that is in production by crop group rather than county (Table 6), additional discrepancies between the three data sources become more apparent. In this report, we did not attempt to reconcile these data. Rather, we compute nitrate loading to groundwater using both the CAML data and the ACR data as the basis for the extent of cropland in the study area. Results are

Adminsitrative Record
Page 34306

reported separately: Section 1.6 above summarizes the N loading derived based on land area reported by county ACRs, historically and currently. In Section 1.8 below, we provide the N loading estimates based on CAML land area maps.

1.8 Simulation of Groundwater Nitrate Sources and Loading

The previous analysis does not provide a farm scale or field scale differentiation of nitrate source loading. In this section we provide a spatially more detailed analysis of groundwater nitrate loading that takes into account the specific N applications to cropland from individual WWTPs and FPs, and that takes into consideration the amount of manure generated on each of over 600 dairies and the availability of cropland on each of these dairies for land application of manure. We also take advantage of having available detailed maps of septic systems N loading to groundwater, maps of urban areas, golf courses, individual dairy corrals and of individual dairy lagoons, which provide a more detailed spatial context for groundwater nitrate loading.

A spatially detailed and historically dynamic set of nitrate loading maps is also needed to properly assess current and future groundwater nitrate contamination with groundwater models (see Technical Report 4, Dylan et al., 2012). In this section, we describe the results of our CAML-derived spatio-temporal simulation of groundwater nitrate sources and loading, which is more detailed than the crop and county-based analysis presented in Section 1.6.

The various source of groundwater nitrate are spatially distributed across the study area. The CAML maps distinguish individual fields at very high resolution. We also discretized some local sources, such as dairy lagoons and dairy corrals at a high resolution. To model groundwater nitrate loading across the study area, we divide the study area into 0.25 ha (0.6 ac) pixels, each of which has at least one assigned land use. The pixel size is sufficiently small to map individual fields, ponds, lagoons, and other sources with sufficient accuracy. We developed the so-called Groundwater Nitrate Loading Model (GNLM) to simulate direct nitrate loading to groundwater from non-cropland sources and from alfalfa and to simulate the various cropland nitrogen fluxes, including the on-farm and off-farm manure nitrogen distribution needed to compute cropland nitrate loading to groundwater by mass balance (see Sections 2 – 8 for details).

The mass balance modeling in GNLM is based on the same crop-category specific rates of typical N applications and N harvest as those used in the analysis in Section 1.6 above. But here, GNLM applies these rates to the actually mapped land area of each crop category provided by CAML (Figure 1), which yields the mapped distribution of typical N application rates (Figure 4) and N harvest (Figure 5).

In the CAML-based analysis with GNLM, we assume that seven vegetable crops are harvested multiple times per year: celery, lettuce, spinach, broccoli, cabbage, cauliflower, and Brussels sprouts. An analysis of NASS Agricultural Census data indicated that approximately 1.6 crops were planted and harvested per year in the 1990 period and 1.7 crops were planted and harvested per year in the 2005 period (see Sections 2 and 3 for details). We also assume that corn is always double-cropped with winter grain. Both assumptions are over-simplifying the variability in the agricultural systems of the TLB and SV, but

Adminsitrative Record
Page 34307

provide a best average approximation of management practices in the study area. For land areas with multiple crops per year, the typical N application rate and the harvested N rate were adjusted accordingly.

For effluent and biosolids applications, we identified specific cropland areas in the vicinity of individual WWTPs and FPs (Section 6). The GNLM model distributed known total nitrogen loading to these croplands each year. Similarly, about 600 dairies have provided information to the Central Valley Regional Water Quality Control Board (CVRWB) identifying cropland parcels owned and operated by the dairies. We used the parcel numbers to determine their location and then identified the land use by geospatial analysis with CAML. The GNLM uses this information to distribute the manure nitrogen generated on each dairy (Section 4) to those cropland parcels that are under a dairy's operation. Of those parcels, GNML selects on those for manure application where CAML indicates that field crops are grown that typically receive manure (e.g., corn, winter grain, and others, see Section 4).

After 1980, dairies began to export significant amounts of manure N (see Section 4) to neighboring farms, typically as soil amendment. The actual amount of manure N exported is not well known prior to the Central Valley Regional Water Board dairy general order. Since 2007, dairies report the amount of manure N exported, which can be used to constrain the amount of manure N exported in the past. For the spatially distributed modeling analysis, these data were not available, and we therefore developed six hypothetical scenarios with respect to the amount and fate of exported manure N. These scenarios broadly bracket the actual amount of manure exported by dairies. These scenarios also bracket the potential distribution of exported manure within each of the five counties, between the counties in the study area, and the hypothetical export of manure N to outside the study.

In 2005, after accounting for 38% atmospheric losses from excreted manure prior to land application, land application of manure accounts for 127 Gg N/yr. Manure exported by dairies is mostly solid manure or composted manure. Of the 6 hypothetical manure export scenarios (described below in more detail), scenarios A, B, and C assume that 77 Gg N/yr, a total of 38% of dairy manure excreted, nearly two-third of all land-applied manure, is moved off dairy (an additional 1.5 Gg N/yr land applied to cropland originates from other confined animal operations). This number reflects an approximate upper bound for the amount of manure N that can currently or in the near future be exported from dairies. Scenarios A, B, and C assume that 48 Gg N/yr are land applied on cropland within dairies. This reflects the order of magnitude of manure N that dairies can land apply on their own land within the foreseeable future under the CVRWB dairy general order: The order requires that total nitrogen application to cropland cannot exceed 140% - 165% of harvested nitrogen. Dairies in the study area currently manage approximately 120,000 ha (300,000 ac) of cropland. If much of the land under dairy management were converted to grow summer corn and winter grain at high yields, and if the required 140%-165% ratio of total N applied to total N harvested could be achieved while applying manure N at a rate of 400 kg N/yr (approximately two-third of the total N application), the total N applied on dairies, within the constraints of the dairy general order, would be about 48 Gg N/yr.

On cropland not managed by a dairy ("non-dairy cropland"), exported manure N is assumed to be applied in addition to synthetic fertilizer that meets 100% of the typical N application needs. The total N

Adminsitrative Record
Page 34308

application on non-dairy cropland[10] is therefore always more than 100% of the typical N application needs (see Sections 2.6.2 and 4.8.4 for details). On the other hand, Scenario D represents the (hypothetical) case that all manure is land applied on corn, grain, and other field crops on land that is under the direct management of dairies. In Scenario D, no manure is exported:

- "Scenario A": Manure exported by dairies does not affect the typical N fertilization rates (Figure 4) on non-dairy cropland within the study area, after accounting for the combined synthetic and organic sources of nitrogen fertilizer applied to non-dairy cropland. This is a hypothetical (future) scenario representing the possibility that manure exported from dairies
    - is applied to non-dairy cropland as part of the typical N fertilization rates,
    - is transported to areas completely outside the study area, possibly after some processing,
    - is intentionally processed and lost to the atmosphere,
    - or any combination thereof.
- "Scenario B (by county)": Half of the manure exported by dairies is applied as soil amendment on non-dairy cropland within the county of origin. The other half of the exported manure has the same fate as listed under "Scenario A". The manure exported by dairies for soil amendment within each county is distributed in direct proportion and in addition to the typical N fertilization needs of crops within that county (manure applied as soil amendment does not leave the county). This scenario represents the mid-point between "Scenario A" and "Scenario C (by county)".
- "Scenario B (study area)": Half of the manure exported by dairies is applied as soil amendment on non-dairy cropland within the study area (not restricted to the county of origin). The other half of the exported manure has the same fate as listed under "Scenario A". The manure exported by dairies for soil amendment in the study area is distributed across all non-dairy cropland in the study area in direct proportion and in addition to their typical N application needs. This scenario represents the mid-point between "Scenario A" and "Scenario C (by study area)".
- "Scenario C (by county)": All manure exported by dairies is applied as soil amendment on non-dairy cropland within the same county. The total manure exported by dairies within each county is distributed in direct proportion and in addition to the typical N application rates of crops within that county (manure does not leave the county).
- "Scenario C (study area)": All manure exported by dairies is applied as soil amendment within the study area (not restricted to the county of origin), and the total manure exported by all

[10] For purposes of simulating "exported manure N" to cropland other than dairy cropland that typically receives significant amounts of manure N (corn, grains, other field crops), the category "non-dairy cropland" is assumed to include vineyards, nut and tree crops, subtropical fruit, vegetables, and other non-forage crops managed by dairies, since they typically do not receive large amounts of manure application except as amendment.

Adminsitrative Record
Page 34309

dairies in the study area is distributed across all non-dairy cropland in the study area in direct proportion and in addition to their typical N fertilization rates.

- "Scenario D": No manure is exported by dairies. All manure is land applied on applicable forage crops within the dairy. Note that, groundwater nitrate loading on non-dairy cropland is therefore identical to that simulated in Scenario A. Groundwater nitrate loading on dairy cropland receiving manure is significantly higher under this Scenario than under the export scenarios.

The six manure export scenarios are coded into GNLM. Historically, for simulation purposes, manure N exports are assumed to be negligible (under all scenarios) prior to 1980, increase linearly from 0% to 38% between 1980 and 2005, and then stay constant at that rate through 2050.

Using the loading methods and mass balance considerations further described in Sections 2.6.2, and in Sections 3 through 8, the GNLM generates maps of spatially distributed groundwater nitrate loading in 1945, 1960, 1975, 1990, 2005, and 2050. These groundwater nitrate loading maps are used as input to the groundwater transport model of the Tulare Lake Basin, which simulates current and future nitrate concentration in wells throughout the Tulare Lake Basin (see Technical Report 4, Boyle et al., 2012).

Table 7 shows the study area totals of various nitrogen fluxes and of the groundwater nitrate loading derived from the CAML-based simulation with GNLM. Comparing the 2005 data from the CAML-based GNLM simulations with the estimates of cropland nitrogen fluxes and groundwater nitrate loading derived based on land areas reported by county ACRs, the magnitude of the nitrogen fluxes is similar and for some sources identical due to fixed source size (compare 2005 results in Table 7 with Figure 3).

The CAML-based GNLM results illustrate the spatial and temporal (historic) distribution of nitrogen fluxes and groundwater nitrate loading across the study area. Fertilizer applications are highest in vegetables in Monterey County and on double-cropped corn (and winter grain) land use areas in the central-eastern Tulare Lake Basin (Figure 9). Vineyards, located mostly around central Fresno County and southern Tulare County have among the lowest synthetic fertilizer application rates.

Manure, effluent, and biosolids applications under Scenario D (no manure exports from dairy) are focused on land areas designated for effluent and biosolids application and on field, grain, and hay crops within land parcels owned or operated by dairies (Figure 10). Application rates typically far exceed 500 kg N/ha/yr. Most of these occur in western Tulare County and northeast Kings County, but also along the central axis of the valley in Fresno County and Kern County.

Significantly lower rates of manure are applied to dairy cropland under Scenario A-C (an average of 38% of excreted manure nitrogen is exported from dairy facilities). Yet, on many dairy fields, the application rates still exceed 500 kg N/ha/yr (the largest category shown on the map). Biosolids and effluent applications are unaffected by the Scenario simulations (Figure 11). In Scenario C, the exported manure is distributed proportionally to the typical crop nitrogen needs, either within the county of origin (Figure 12), or within the study area (Figure 13). When manure is assumed to remain within the county of origin, the entire cropped area within Tulare County is subject to large amounts of manure nitrogen

Adminsitrative Record
Page 34310

being applied, not only on dairies (Figure 10), but also on all non-dairy cropland (Figure 12). The spatial distribution of manure nitrogen applied outside of dairies is due to the fact that exported manure is always applied proportional to the applied nitrogen needs of a crop (and it is in addition to "typical" fertilizer needs already being met by application of synthetic fertilizer). Vegetable crops are therefore receiving higher amounts of manure than, e.g., vineyards. When manure is simulated as being distributed across the study area, Monterey County receives large amounts of manure compost or amendment – in some cases in excess of 100 kg N/ha/yr, due to the high fertilization rates in vegetables. While, in reality, some compost is applied to vegetable crops in Monterey County, simulated rates for "Scenarios B and C (study area)" likely overestimate the amount of soil amendment applied in the Salinas Valley. Under these simulations, large amounts of manure are also applied to double-cropped corn – winter grain fields in Tulare County (high fertilization rats), which partially reflects current practices of applying manure primarily to these crops (Figure 13).

Harvested nitrogen is largest in the vegetable crops of the Salinas Valley, in double-cropped corn and grain fields in the central portion of the Tulare Lake Basin, and in alfalfa fields (Figure 14). Intermediate harvest rates are achieved in many other field crops and in nut crops. Largest nitrogen removal rates from fields therefore occur in the Salinas Valley, and on the Westside and in the central portion of the Tulare Lake Basin.

Simulated current (2005) groundwater nitrate loading, including direct percolation from urban areas, septic systems, percolation basin etc., as well as from cropland, is shown in Figures 15 – 20 for the six different manure export scenarios. Highest localized groundwater loading occurs on dairies in Scenario D, where all land applied manure N is applied on land within dairies. Groundwater nitrate loading on dairy cropland is significantly reduced by the hypothetical export of about two-third of the land applied manure nitrogen (Scenarios A-C). This represents, in very approximate terms, the minimum amount of manure that needs to be exported from dairies in the coming years under the 2007 CVRWB general order, such that dairy cropland can meet the required rate of total nitrogen application (140% to 165% of N harvested).

But manure export poses threats to groundwater quality impacts from cropland outside dairies (Scenarios B and C, Figures 16-19), unless manure is applied as part and within the constraints of a typical fertilization regime (Figure 20). That threat is also a function of how much of the exported nitrogen remains within the county of origin, how much is exported to other counties within the study area, and how much will potentially be exported outside the study area or processed for atmospheric loss (Scenarios A-C). Assuming that exported dairy manure remains within the county of origin and is applied as soil amendment to crops outside dairies, large groundwater nitrate loading would be expected particularly in Tulare County, which has the highest density of dairy animals (Figure 16). Even with approximately two-third of the manure exported off-farm and either managed to meet typical fertilization needs or removed from non-dairy study area cropland (Scenario A, Figure 20), the loading rates from dairy cropland remain relatively high. In the future, it may be possible to manage nutrients in corn, grain crops and other forage crops such that nitrogen needs are met mostly (rather than only to 50%, as simulated here) with manure nitrogen rather than fertilizer nitrogen, but without exceeding total nitrogen application limits imposed by the CVRWB dairy general order.

Adminsitrative Record
Page 34311

Non-cropland sources of groundwater nitrate loading other than septic systems are illustrated in Figure 21. The most intense sources are corrals and lagoons located on dairies and some WWTP/FP percolation basins. Other sources are not generally exceeding the operational benchmark leaching rate of 35 kg N/ha/yr (31 lbs N/ac/yr) (leaky sewer lines, turf areas, golf courses).

Maps of GNLM simulated nitrogen fluxes for 1945 – 2050 (including the 2005 maps shown in Figures 9 – 21),which are summarized in Table 7, are shown in Appendix Figures 3 through 120, available as a separate file at http://groundwaternitrate.ucdavis.edu.

The simulated groundwater nitrate loading demonstrates the overall spatial variability encountered across the study area due to the type of source, due to the management specifically of dairy manure, due to differences in nutrient management and harvested nitrogen between 58 different crop categories, due to the spatial distribution of percolation basins and liquid dairy manure storage lagoons, due to the location of urban and peri-urban areas, due to the density of septic systems density, etc. However, the data, particularly the cropland groundwater nitrate loading data, only represent averaged values for each crop category and for each source type. The simulations cannot take into account differences in groundwater nitrate loading due to different management practices by different landowners/source managers, due to differences in the physical characteristics between fields/orchards/vineyards of the same crop category, or differences in the specific design of individual septic systems, sewer systems, etc.

Since our estimates do not account for such differences, we caution that actual local groundwater nitrate loading at any location within the study area is likely to vary from those projected by the simulations shown in Figures 15-20. Nonetheless, the range of potential outcomes, across the landscape, across crop categories and source types, and across the listed range of manure management scenarios provides significant insight into both, the large variability of loading and the overall magnitude of groundwater nitrate loading associated with each source.

Future implementations of GNLM may be used to account for heterogeneous field-to-field or farm-to-farm variability in nitrogen fertilizer management. GNLM can also be modified to account for various future management scenarios for specific nitrate loading sources.

Adminsitrative Record
Page 34312

Table 7. Study area summary of simulated CAML-based, spatially distributed nitrogen fluxes that account for mapped areas of cropland 1945 – 2050 (not including alfalfa), spatial distribution of WWTP, FP, and dairy facilities, spatial distribution of urban areas, and spatially variable atmospheric nitrogen deposition. All fluxes shown in Gg N/yr. One Gg N/yr = 1,100 tons N/yr. Data are simulated using GNLM (See Section 2.6)

	1945	*1960*	*1975*	*1990*	*2005*	*2020*	*2035*	*2050*
Typical N application to cropland	40.2	85.2	144.6	220.1	240.5	240.5	240.5	240.5
Typical N application to alfalfa and pasture	5.5	13.3	19.4	3.1	3.5	3.5	3.5	3.5
Actual synthetic fertilizer N applied on cropland, Scenario A-C	39.7	84.2	139.0	209.1	228.4	228.2	228.0	227.9
Actual synthetic fertilizer N applied on cropland, Scenario D	39.7	84.2	139.0	209.1	228.2	228.0	227.9	227.8
Land applied manure (on-dairy), biosolids, effluent N, Scenario A-C	2.0	3.1	34.4	45.1	53.8	62.6	72.9	80.5
Land applied manure N to non-dairy cropland, Scenario A-C	0.0	0.0	0.0	14.0	78.0	88.0	99.8	106.1
Land applied manure (on-dairy), biosolids, and effluent N, Scenario D	2.0	3.1	34.4	58.0	131.7	150.6	172.7	186.5
Harvested N	19.4	54.6	70.7	112.8	141.1	141.1	141.1	141.1
Surface runoff N	9.5	14.4	17.7	17.9	19.2	19.2	19.2	19.2
Atmospheric N deposition on all land	13.4	20.0	24.5	25.5	20.1	17.0	12.2	7.4
GW nitrate loading (N) from cropland, Scenario A	23.0	28.0	88.6	133.1	124.9	130.0	136.8	141.4
GW nitrate loading (N) from cropland, Scenario B	23.0	28.0	88.6	139.2	158.9	168.4	180.3	187.6
GW nitrate loading (N) from cropland, Scenario C	23.0	28.0	88.6	145.3	193.3	207.3	224.5	234.6
GW nitrate loading (N) from cropland, Scenario D	23.0	28.0	88.6	145.7	195.3	209.6	227.1	237.3
Septic systems nitrate loading (N)	0.6	1.0	1.2	1.8	2.3	3.2	4.2	5.4
Total non-cropland nitrate loading (N) (not including alfalfa)	1.2	1.9	5.2	7.0	8.2	9.3	10.5	11.9
Alfalfa nitrate loading (N)	3.7	6.5	4.4	3.8	4.5	4.5	4.5	4.5

Note: "Cropland" in the above table always refers to "cropland, not including alfalfa". Typical nitrogen application represents the amount of N thought to be typically applied to a specific crop category (Figure 4). Where manure is applied on-dairy or where effluent or biosolids are applied, up to 50% of this amount is supplied by manure N (applies only after 1970). All manure N in excess of this 50% typically applied N, and all manure N applied outside of dairy-owned cropland is applied in addition to synthetic fertilizer applications that meet the typical nitrogen application needs. The harvested nitrogen reflects crop category specific harvest rates derived from acreage and yield data provided in county agricultural commissioner reports (Figure 5) and applied to the CAML crop category distribution maps for 2005 (Figure 1) or under historic and future conditions (see Section 3). Scenarios A, B, and C for cropland nitrate loading represent nitrogen mass balance modeling solutions assuming that no (A), half (B), or all exported manure N is applied as soil amendment at rates proportional and in addition to typical crop fertilizer rates (exports occur only after 1980). Scenario D assumes that no manure is exported from dairies, even after 1980. Total non-cropland nitrate loading other than from septic systems includes nitrate loading from urban lawns and leaky sewer systems, golf course, and from WWTP and FP percolation basins. After 1968, total non-cropland nitrate loading also includes loading from dairy corrals and dairy lagoons. For 2005, these N fluxes are similar to those obtained at the county and study area level using non-spatial county ACR data (Figure 3).

Adminsitrative Record
Page 34313

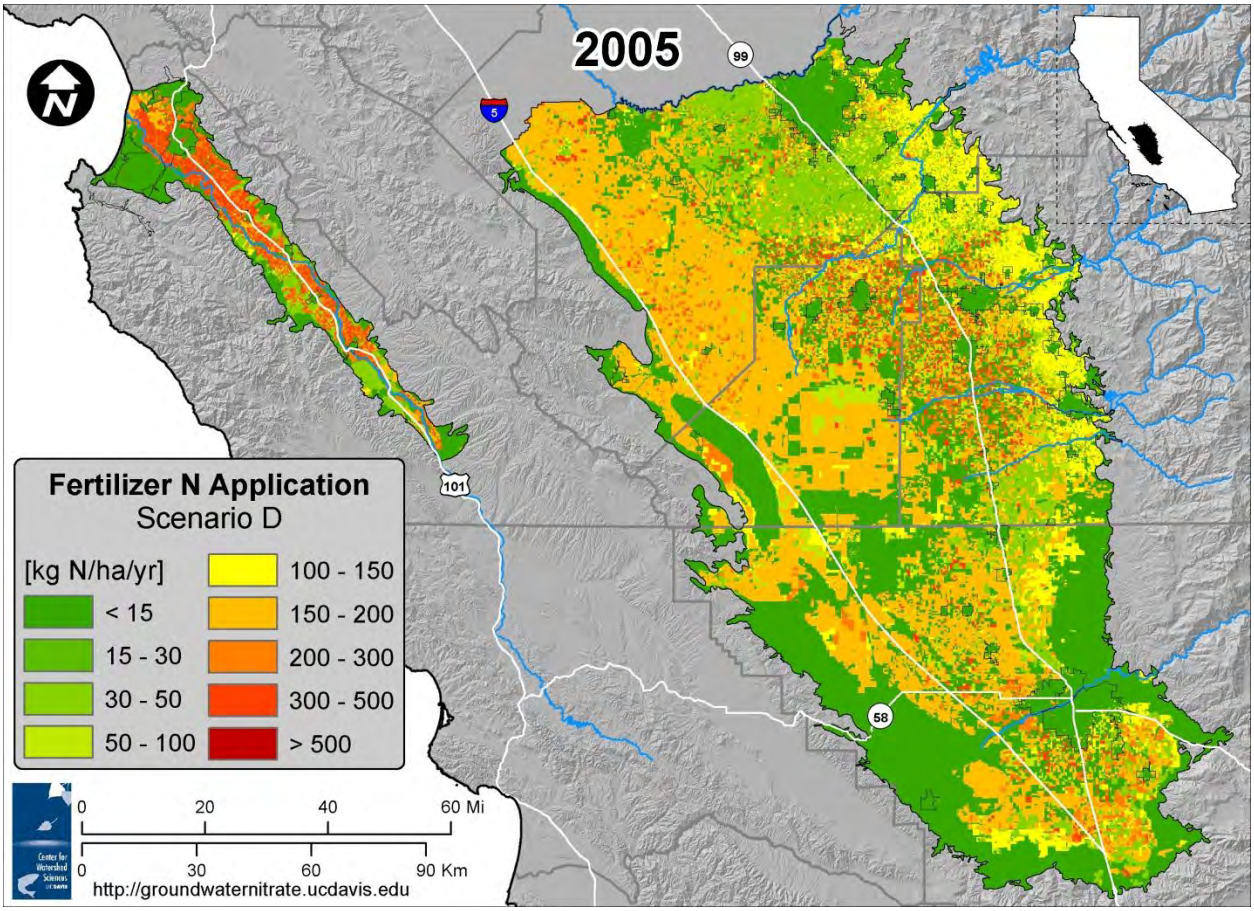


Figure 9. Simulated synthetic fertilizer application on cropland, including alfalfa, after accounting for the application of manure as fertilizer on dairy cropland (manure may make up to 50% of the applied nitrogen need). This map represents results for "Scenario D" (no manure exports from dairies). Differences to Scenarios A-C are very small (see Table 7). Simulated synthetic fertilizer applications account for 1.7x cropping in seven vegetable crops (broccoli, Brussels sprouts, cabbage, cauliflower, celery, lettuce, spinach) and double-cropping of all corn with winter grain.

Adminsitrative Record
Page 34314

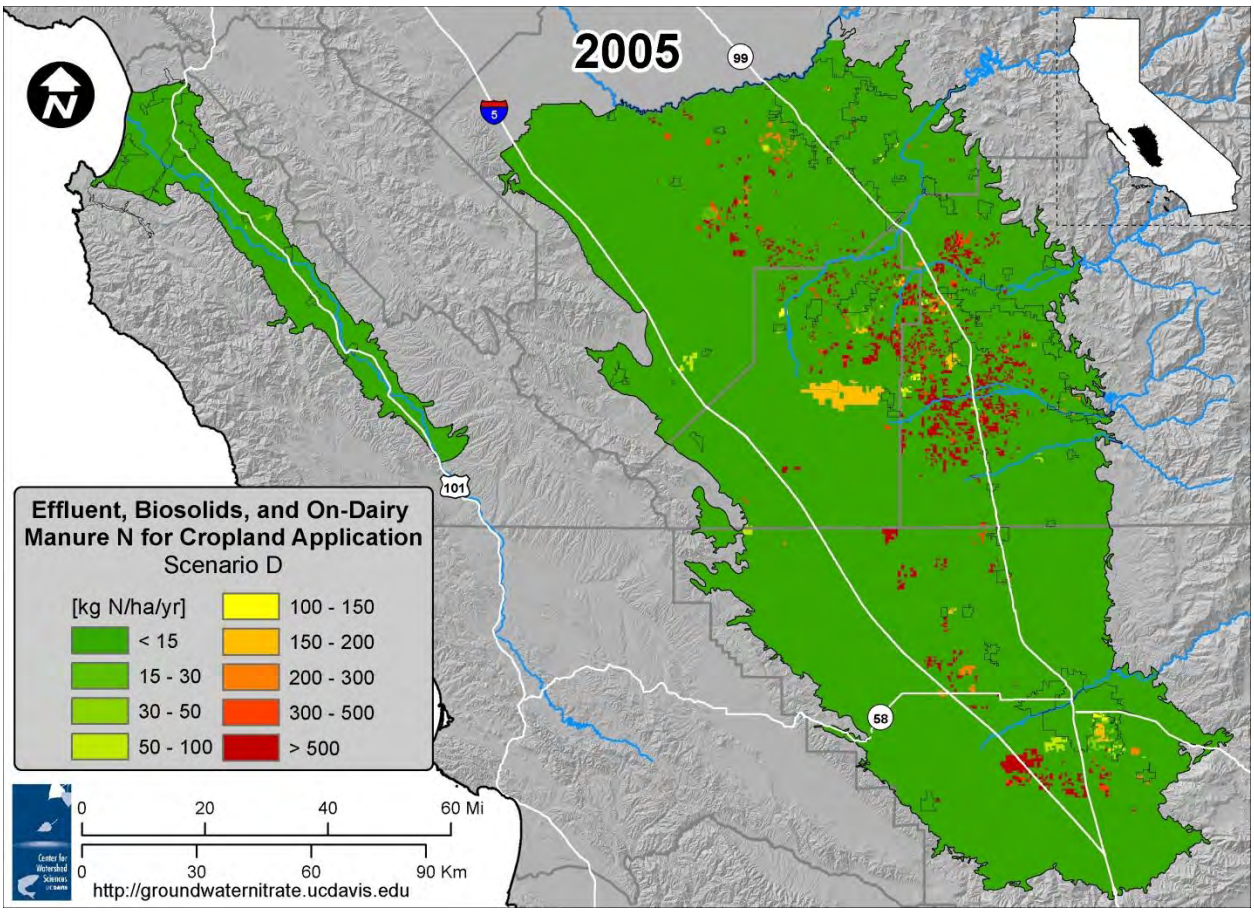


Figure 10. Simulated land applied N from dairy manure (land applied on dairy cropland), and from WWTP/FP effluent and biosolids. This map represents results for the hypothetical "Scenario D" (all manure is land applied on dairies, representing 62% of animal N excretion, no manure is exported from dairies).

Adminsitrative Record
Page 34315

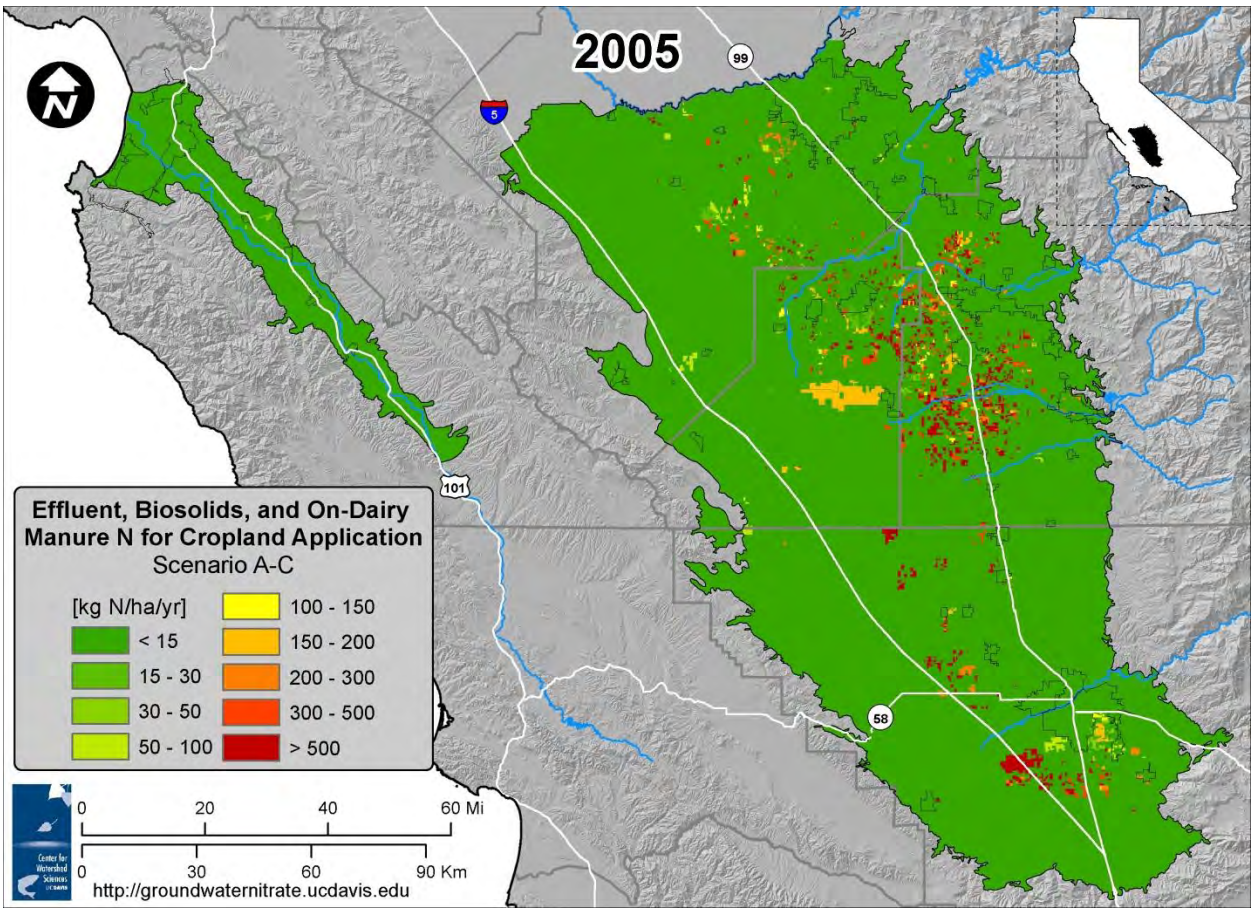


Figure 11. Simulated land applied N from dairy manure (showing only land applied N on dairy cropland, but not exported dairy manure N applications), and from WWTP/FP effluent and biosolids. This map represents results for Scenario A-C (a total of 24% of animal N excreted is land applied on dairies).

Adminsitrative Record
Page 34316

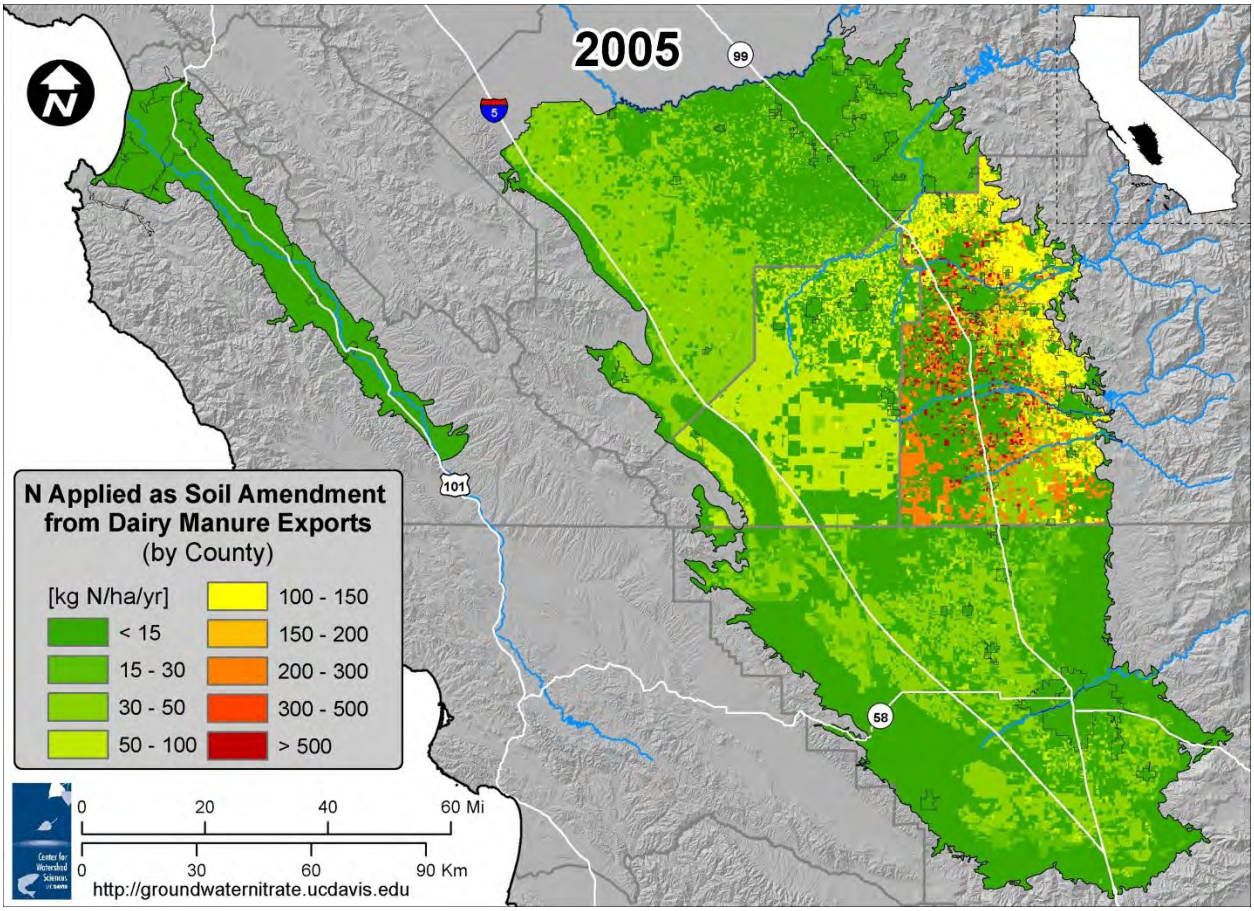


Figure 12. Simulated land application of dairy manure N exported from dairies and land applied within the county of origin at rates proportional and in addition to typical N fertilizer needs in each crop (see Figure 4). Total manure exports represent 38% of animal N excreted ("Scenario C (by County)").

Adminsitrative Record
Page 34317

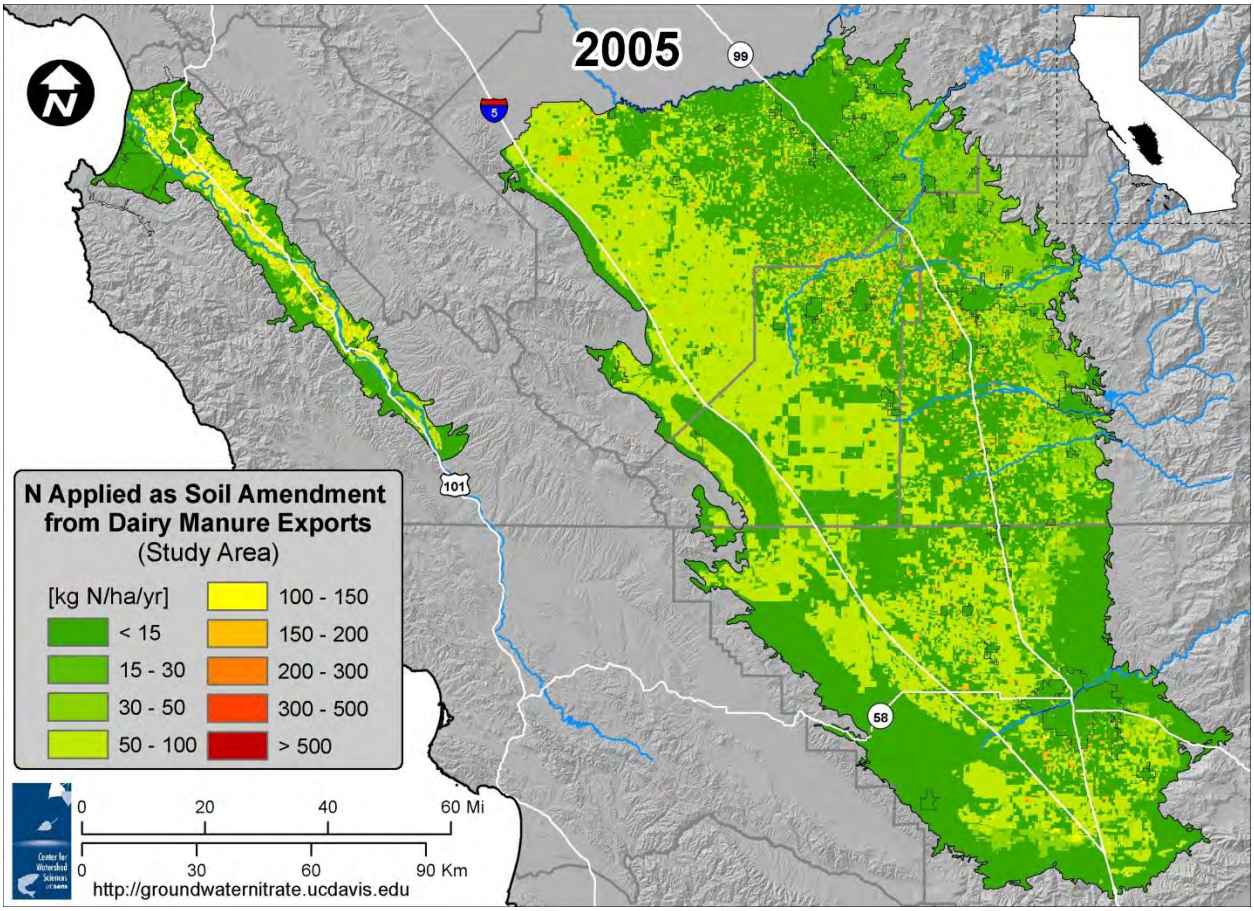


Figure 13. Simulated land application of dairy manure N exported from dairies and land applied across the study area (not restricted to the county of origin) at rates proportional and in addition to typical N fertilizer needs in each crop (see Figure 4). Total manure exports represent 38% of animal N excreted ("Scenario C (study area)").

Adminsitrative Record
Page 34318

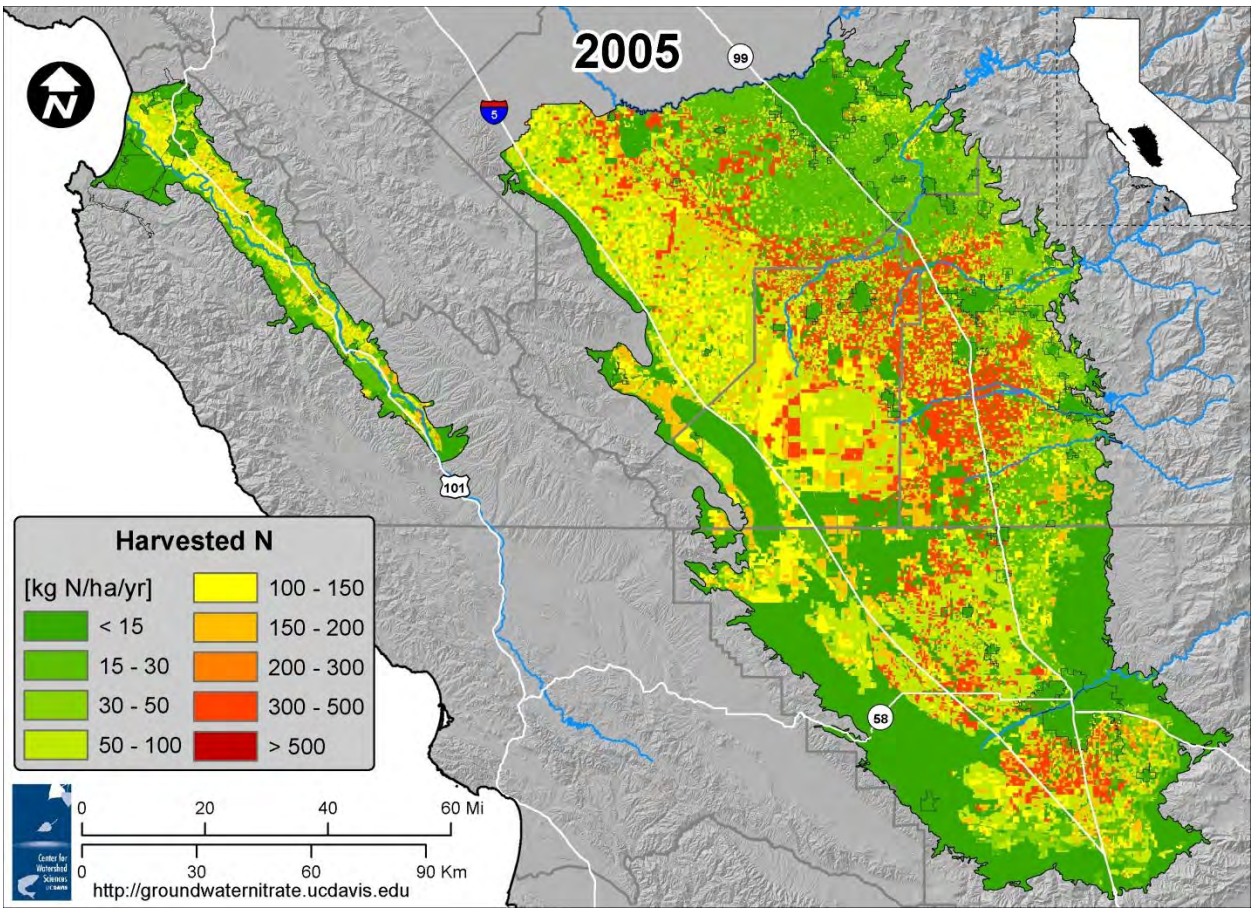


Figure 14. Simulated nitrogen harvested in 2005 from all cropland including alfalfa. The simulation assumes 1.7 annual crops in seven vegetable crops (broccoli, brussels sprouts, cabbage, cauliflower, celery, lettuce, spinach) and double-cropping of all corn acreage with winter grain.

Adminsitrative Record
Page 34319

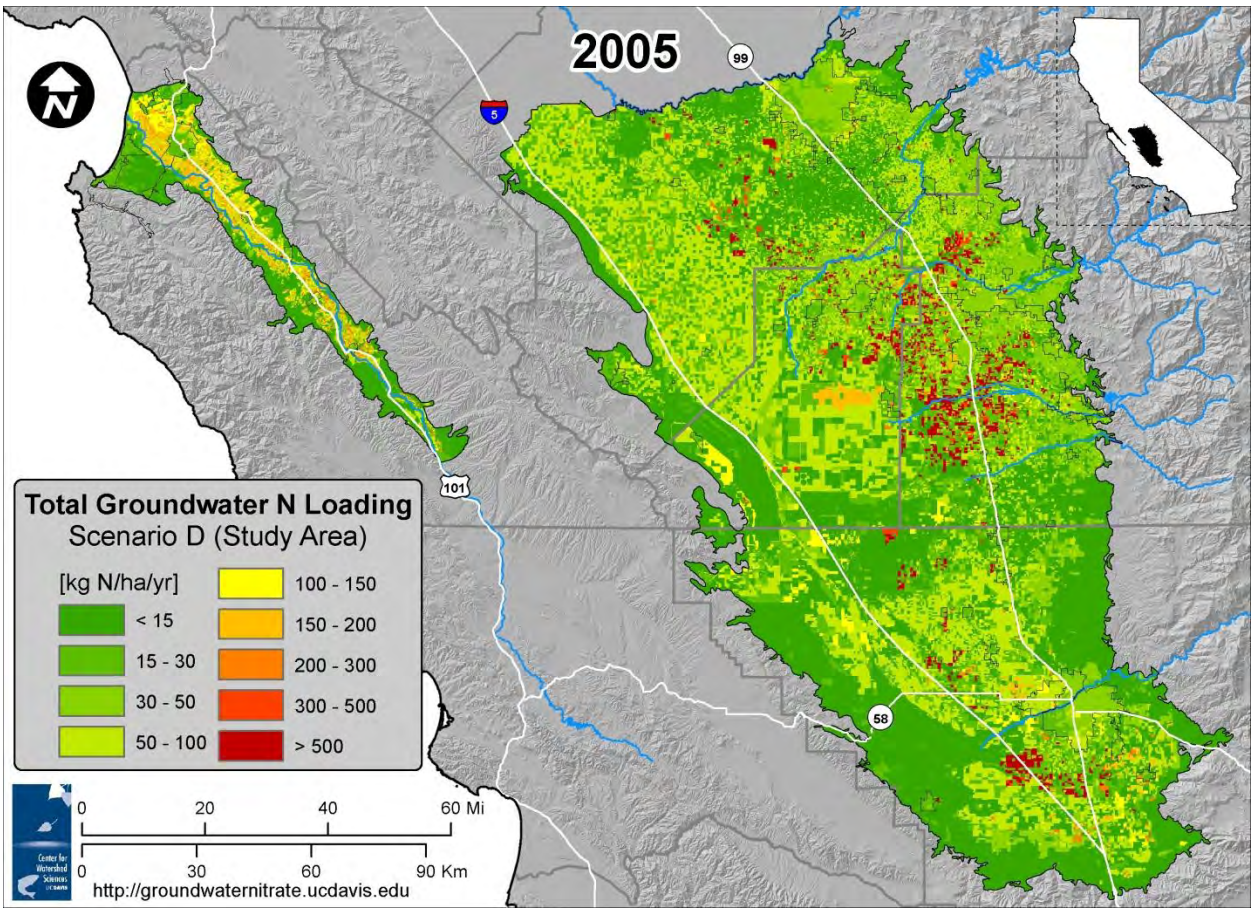


Figure 15. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario D": All land applied manure N (62% of animal N excreted) is applied to corn, grain, and other field crops (not including alfalfa) under the direct management of dairies (no manure N exports).

Adminsitrative Record
Page 34320

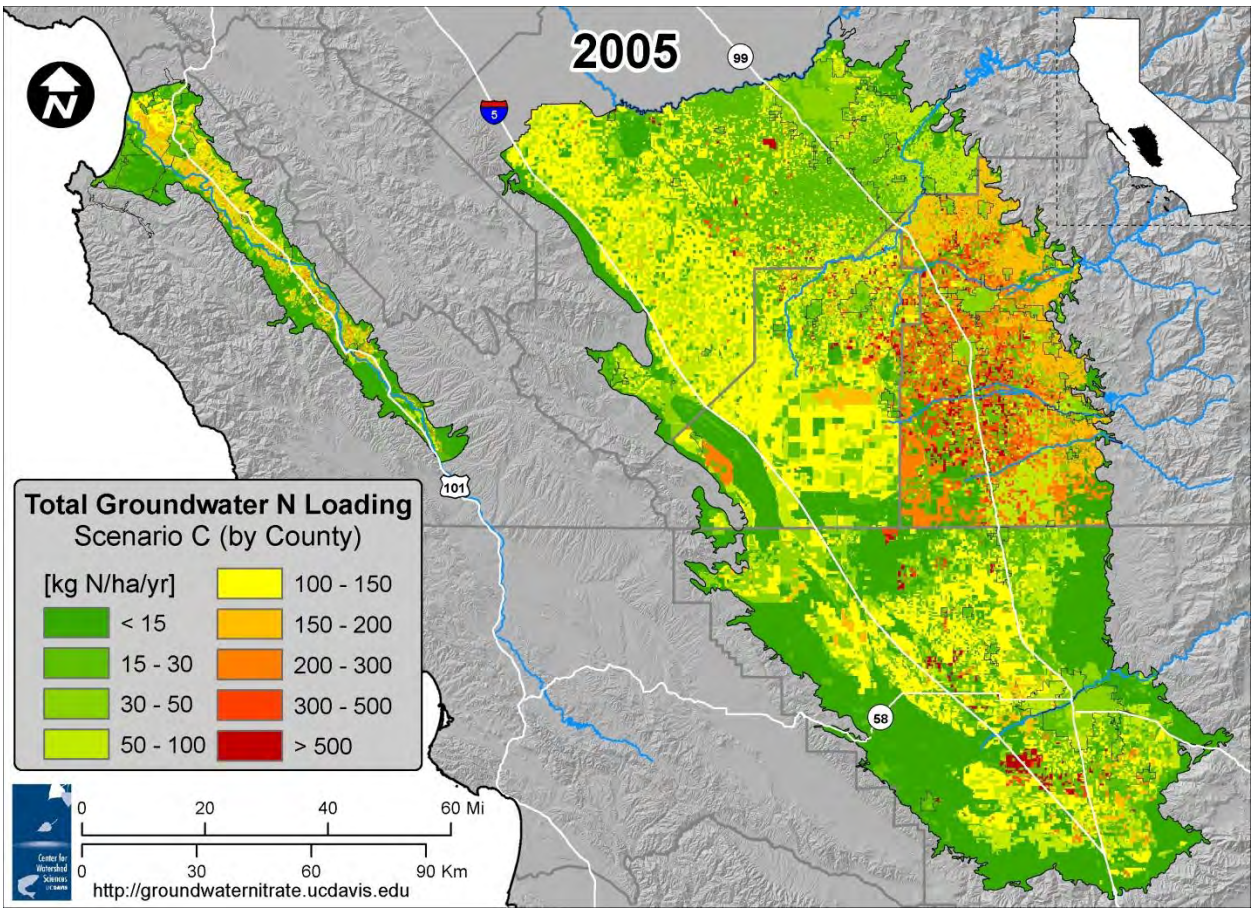


Figure 16. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario C (by county)": hypothetically exported manure N from dairies (38% of animal N excretion) is land applied as soil amendment within the county of origin. Exported manure N applications are proportional and in addition to synthetic fertilizer applications to meet typical fertilizer application needs (Figure 4).

Adminsitrative Record
Page 34321

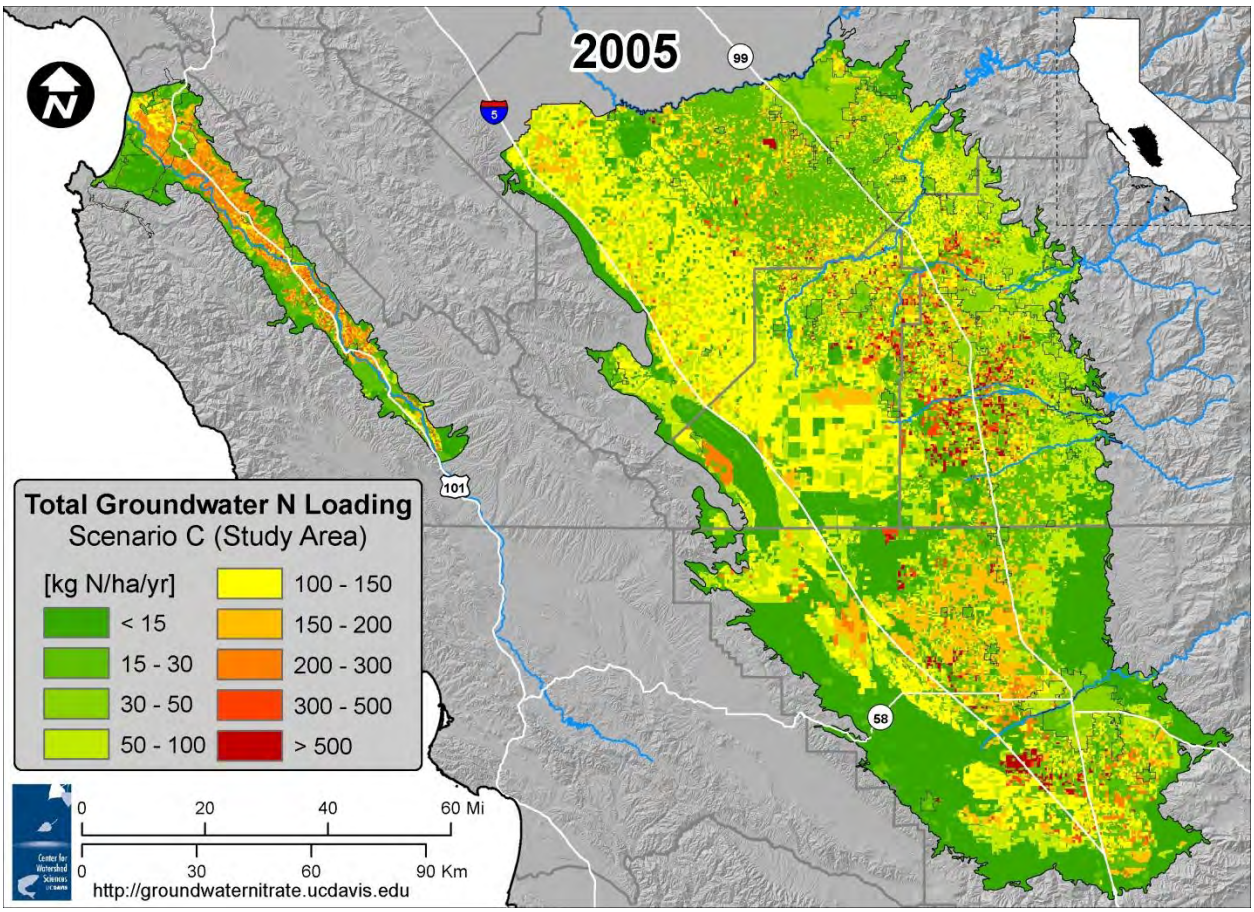


Figure 17. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario C (study area)": hypothetically exported manure N from dairies (38% of animal N excretion) is land applied as soil amendment across the study area (not restricted to the county of origin). Exported manure N applications are proportional and in addition to synthetic fertilizer applications to meet typical fertilizer application needs (Figure 4).

Adminsitrative Record
Page 34322

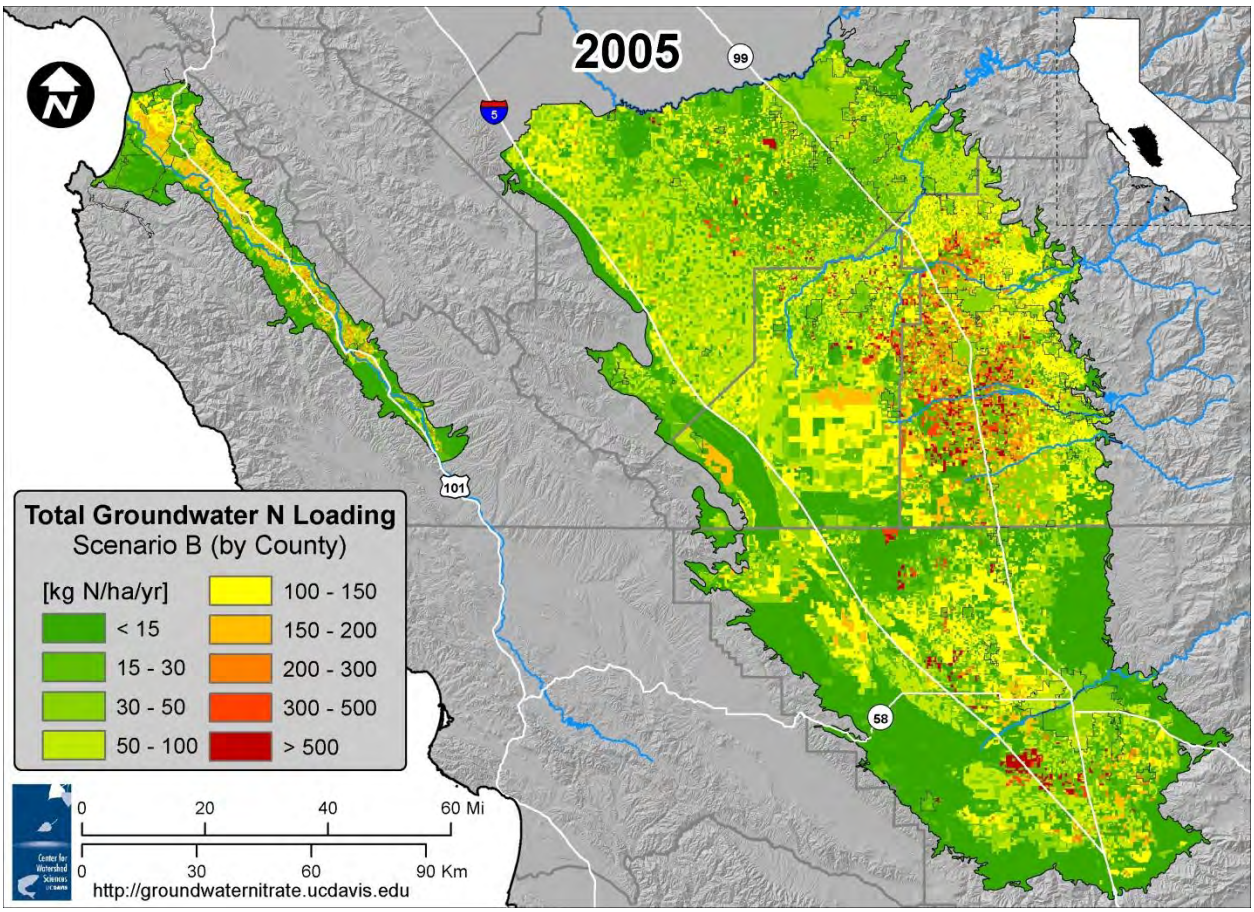


Figure 18. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario B (by county)": half of the hypothetically exported manure N from dairies (19.5% of animal N excretion) is land applied as soil amendment within the county of origin. These manure N applications are proportional and in addition to synthetic fertilizer applications to meet typical fertilizer application needs (Figure 4). The remaining half of the hypothetically exported manure N (19.5% of animal N excretion) is subject to the pathways explained under "Scenario A" (Figure 20).

Adminsitrative Record
Page 34323

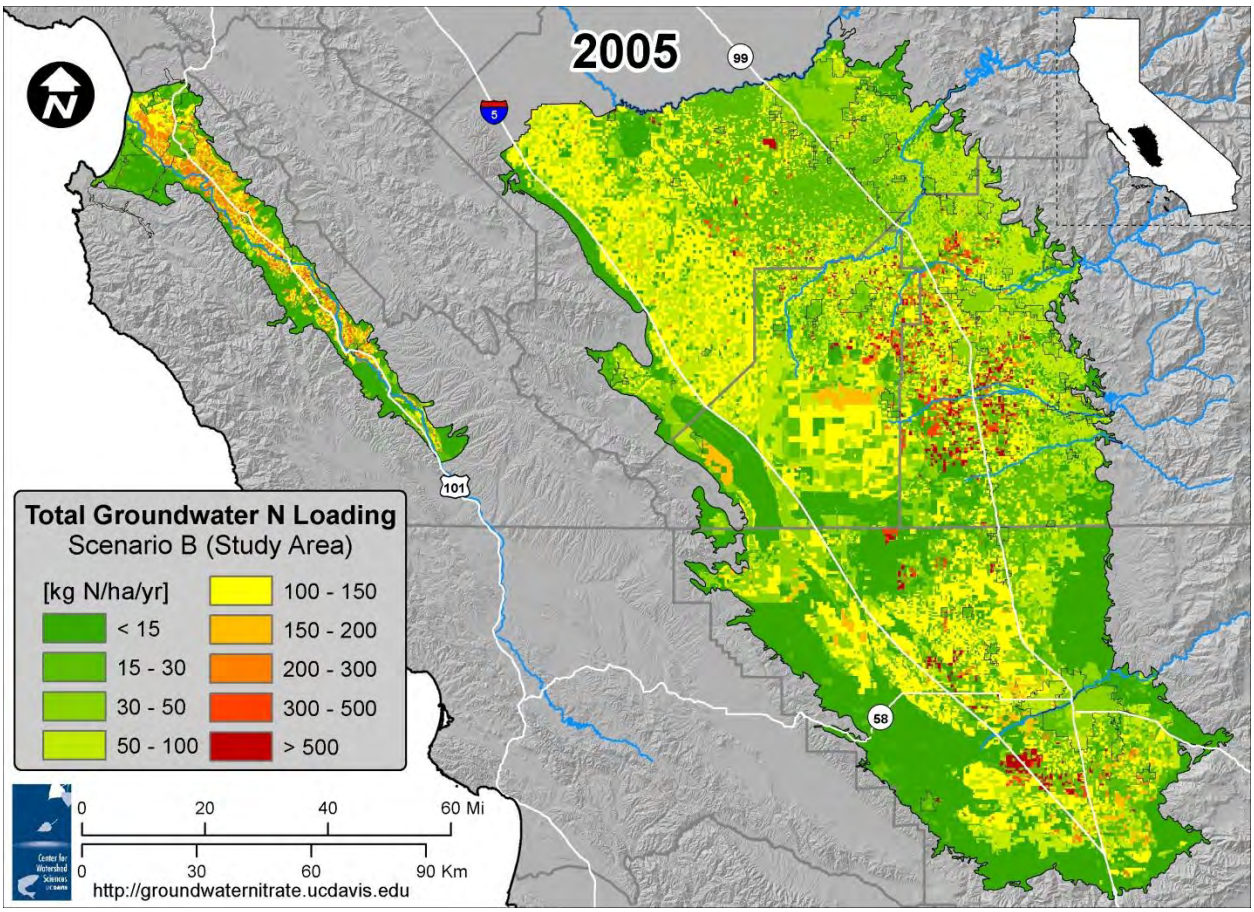


Figure 19. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario B (study area)": half of the hypothetically exported manure N from dairies (19.5% of animal N excretion) is land applied as soil amendment across the study area (not restricted to the county of origin). These manure N applications are proportional and in addition to synthetic fertilizer applications to meet typical fertilizer application needs (Figure 4). The remaining half of the hypothetically exported manure N (19.5% of animal N excretion) is subject to the pathways explained under "Scenario A" (Figure 20).

Adminsitrative Record
Page 34324

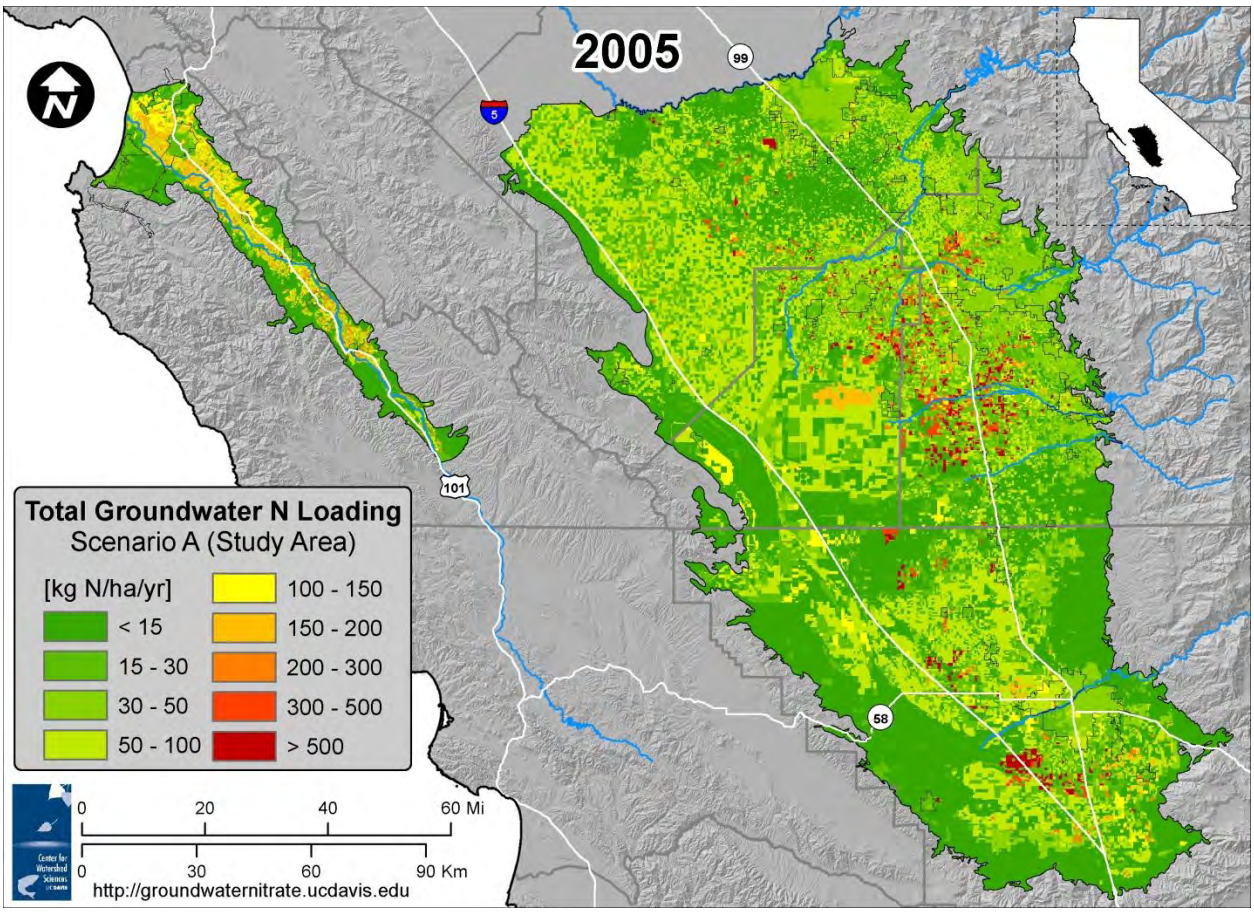


Figure 20. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario A": All of the hypothetically exported manure N from dairies (38% of animal N excretion) is subject to one of the following conceptual pathways: a) applied to non-dairy cropland, with synthetic N plus manure N not exceeding the typical N fertilization rates shown in Figure 4, effectively replacing up to 78 Gg N/yr (86,000 t N/yr) of synthetic fertilizer N with manure N; b) transported to areas completely outside the study area, possibly after some processing; or c) intentionally processed and lost to the atmosphere.

Adminsitrative Record
Page 34325

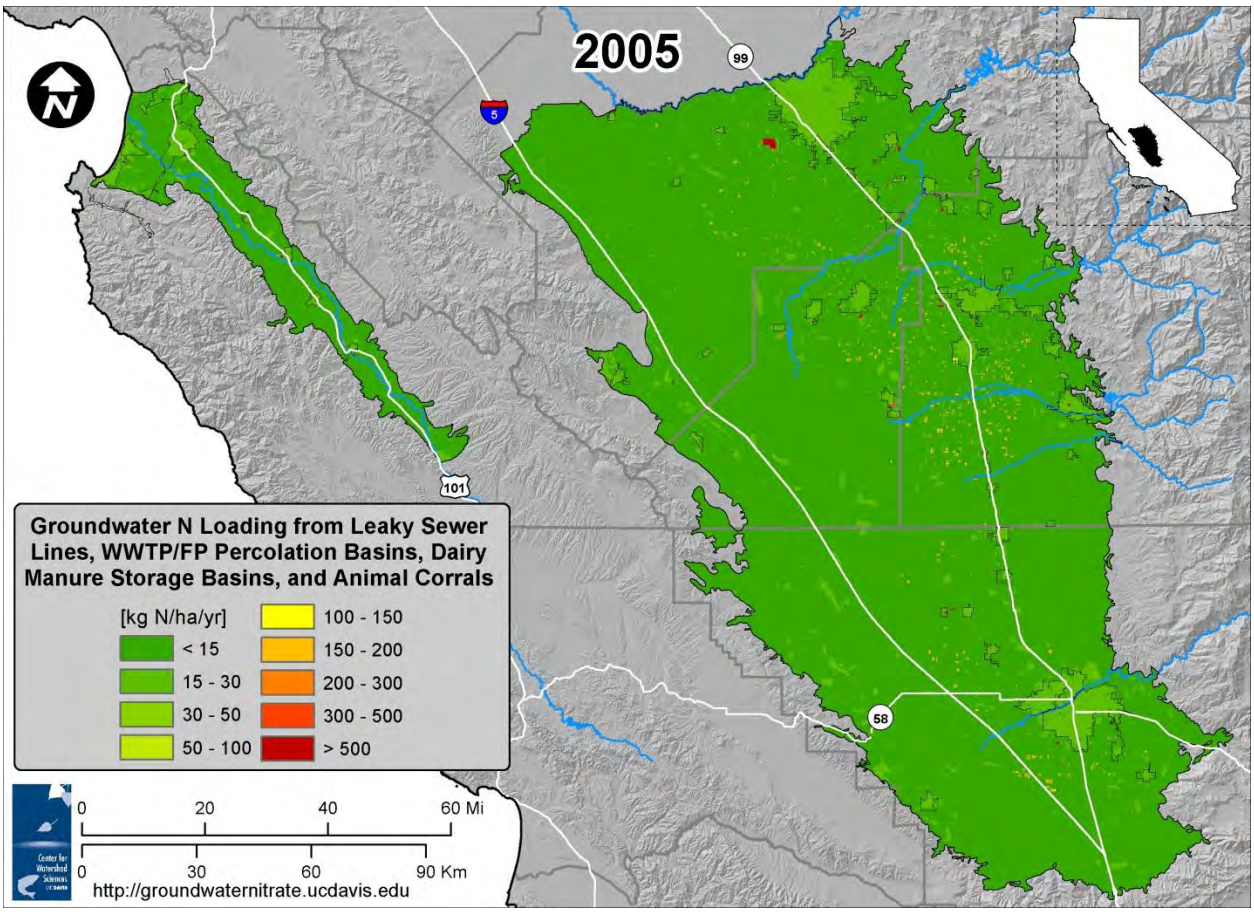


Figure 21. Groundwater loading from non-cropland sources including leaky sewer lines, turf areas, golf courses, WWTP/FP percolation basins, dairy manure lagoons, and animal corrals, in 2005. Septic systems loading (Figure 8) is not included here.

1.9 Concluding Remarks

In this section, we summarize and quantify the contributions of a wide range of nitrate sources to groundwater. The results, for the first time in one document, provide a comprehensive quantitative assessment of all groundwater nitrate sources and both, their overall regional contribution (Section 1.6) and the distribution of their local intensity (Section 1.8). Moreover, for the first time, sources are assessed continuously over a historically relevant period and into the future. The spatial source assessment covers 60 years of historic applications and projects 45 years into the future, at current management practices, and at anticipated urbanization rates.

Overall, cropland (not including alfalfa) is found to be the dominant source of nitrate in groundwater, contributing over 90% of all nitrate leached to groundwater. Other sources are locally important – often near urban areas - and may also lead to contamination of drinking water wells. These sources include septic systems in areas with relatively high density of unsewered homes, percolation ponds associated with municipal wastewater treatment plants and food processors, and manure storage lagoons on

Adminsitrative Record
Page 34326

dairies. Other sources, such as urban lawns, golf courses, leaky sewer lines may be locally important sources of nitrate, but are not considered a significant regional problem.

Cropland as the main source of groundwater nitrate in the study area is far from a single source. The study area features a globally unique diversity of dozens of major crops. It also experiences a wide range of soil and climate conditions and, more importantly, is managed by tens of thousands of individual farmers. Nitrate leaching undoubtedly varies within individual fields, between fields of the same crop, between farms, between counties, and between geographic sub-regions of the study area. With such inherent spatial (and temporal) variability, and given the additional complexity of the groundwater system itself (Boyle et al., 2012), it is tempting to lose sight of the overarching impact of agriculture on groundwater quality.

The large scale quantitative analysis of groundwater nitrate loading from cropland via the mass balance approach, aggregated from detailed data, allows for a clear identification of the major driving factors for groundwater nitrate loading – and of the constraints to addressing groundwater nitrate loading - independent of the large variability between crops, fields, and landowners in the study area.

The total amount of nitrogen intentionally or incidentally applied to study area cropland from various sources each year is about three times larger than the amount of nitrogen removed in the harvest,. This suggests significant system-wide inefficiencies in fertilizer use.

Synthetic nitrogen fertilizer makes up slightly more than half of the total nitrogen applied to cropland suggesting limited flexibility in reducing overall nitrogen application to cropland. In the Salinas Valley, synthetic nitrogen fertilizer is over 80% of all nitrogen applied to cropland.

Land applied dairy manure now constitutes more than one-third of the total nitrogen land applied, increasing from 2% of total N applied prior to the late 1960s and about 15% of total N applied to cropland in the mid-1970s (Table 7). Dairies as sources of groundwater nitrate are therefore a relatively recent phenomenon compared to synthetic fertilizer. In addition, nitrate in irrigation water pumped from groundwater and nitrogen from atmospheric deposition have also become a significant, if only incidental, source of nitrogen applied to cropland (about one-tenth of all nitrogen applied).

Approximately half of the nitrogen incidentally or intentionally applied to cropland is leached to groundwater, whereas the relative groundwater loss was only about one-quarter of all N applied to cropland in 1960. The estimated amount of nitrate losses to groundwater represent a net fertilizer value, at 2011 prices, of about $200 million per year. The amount of groundwater nitrate loading is of such magnitude that, no matter the uncertainty about the exact amount of groundwater loading, the overarching finding is that cropland recharge has and continues to significantly degrade groundwater quality in the study area.

The estimated amount of groundwater nitrate loading in 2005 (195 Gg N/yr or 215,000 t N/yr) is more than double the estimated amount of groundwater nitrate loading from cropland in the mid-1970s (82 Gg N/yr or 90,000 t N/yr), at similar recharge rates. This indicates that concentrations of nitrate in recharge have more than doubled over the past 30 years.

Adminsitrative Record
Page 34327

Importantly, the analysis also outlines significant constraints to reducing agricultural groundwater nitrate loading. If all cropland in the study area were under the CVRWB restrictions imposed on Central Valley dairy cropland, a restriction that would broadly reduce groundwater loading, the total allowable N application to cropland, at today's crop harvest output, would be on the order of 1.5 x 130 Gg N/yr (195 Gg N/yr or 215,000 t N/yr). This is about half of the current total N application to cropland (380 Gg N/yr or 420,000 t N/yr).

Significant reductions of cropland nitrogen applications cannot come from either atmospheric sources or irrigation water sources of N as these are incidental to the land. To the degree that changes in the economy of the study area are not desirable, continued application of urban and animal sources of N on cropland (effluent, biosolids, manure, and other organic materials) are also unavoidable. Large scale nitrogen-removal treatment of these sources would otherwise be needed. Together, these cropland N sources already provide about 90% (178 Gg N/yr or 196,000 t N/yr, Figure 3) of a 150% limit on the ratio of total cropland N applications to harvested N.

At the large-scale agricultural systems level, this suggests that significantly reducing groundwater nitrate loading in the intermediate to long-term is a two-pronged challenge:

First, significant reductions in synthetic fertilizer use would be needed, partly (or sometimes fully) replaced by nitrogen from organic sources, while crop yields are maintained or even improved. The necessary reduction in synthetic fertilizer use would largely be dictated by the ability to export organic sources of nitrogen out-of-state.

Second, nitrogen fertilizer from organic sources (largely dairy manure, but also biosolids and effluent) would be processed such that these can be economically distributed within the study area and – more importantly – such that these nutrients would effectively and efficiently replace synthetic fertilizer at a large scale across the study area. Alternatively, nitrogen from organic sources would be exported out-of-state or, perhaps, recycled in leguminous crops (alfalfa) currently not receiving significant amounts of fertilizer.

Technical Report 3 (Dzurella et al., 2012) reviews current practices to (at least partially) address the challenge in improving cropland nutrient management. The conversion of manure and other organic N sources into a synthetic fertilizer-like product, remains unaddressed in this report, but needs to be considered. The utilization of manure and organic wastes as an energy source, in bio-digesters or as biochar, does not remove significant amounts of nitrogen from the waste stream. But it may provide a manure or waste processing framework that is also amenable to separate nitrogen and salts[11] into a shippable, nutrient-accountable, and marketable product. Much more research and development is needed here.

The dual challenge of reducing overall N inputs to cropland by reducing synthetic fertilizer use while converting manure N to a product that can effectively mimic synthetic fertilizer is not unique to the

[11] Salt is another significant water quality concern associated with land application of manure. In the long-term, salinization of groundwater resources, unlike nitrate contamination, is detrimental not only to drinking water but also to irrigation water quality.

Adminsitrative Record
Page 34328

study area. To the degree that the study area represents over 40% of California's irrigated agriculture and half of its dairy herd, the challenge for California agriculture as a whole remains nearly identical. Beyond California, this is also a global challenge, driven by population growth, economic improvements in threshold countries, and the likely doubling of the demand for food, especially milk and meat products, fiber, and biofuel production over the next four decades, while expansion of global cropland area is expected to be very limited. Already, irrigated agriculture produces 40% of global food and fiber supplies on 20% of all cropland. Without significant shifts in national and global consumer food choices, global markets, in the long-term, will continue to provide incentives for further intensification of irrigated crop and animal production systems, in California, in the United States, and in agricultural regions around the world.

The following sections provide further detail on how the results presented in this first section were obtained. They also provide detailed description of individual source categories contributing to groundwater nitrate, how we conceptualized sources, the nitrogen fluxes associated with the operation of individual source categories, and a review of the literature on the nitrate contribution from these sources. Section 2 introduces the nitrogen cycle and provides the mathematical basis for the mass balance approach used on cropland, which receives nitrogen from many agricultural and urban sources. Section 3 describes current, historic, and future land uses with a focus on cropland as sources of groundwater nitrate, describes the methodology behind estimating cropland fertilizer use and harvest removal, and reviews groundwater loading estimates described in the literature. Section 4 focuses on animal sources, especially dairies, which constitute the overwhelming source of animal manure in the study area. Section 5 reviews nitrate loading from the urban landscape, specifically from turfgrass. Section 6 reviews other urban sources of groundwater nitrate including wastewater treatment plants and food processors, leaky sewer systems, and septic system leach fields. Sections 7 and 8 describe atmospheric and natural sources of groundwater nitrate. Section 9 reviews the role of active, abandoned, and dry wells as sources and conduits of nitrate to shallow and deep groundwater.

Adminsitrative Record
Page 34329

2 Nitrogen Cycling and Mass Balance

2.1 The Biological Importance of Nitrogen

The importance of nitrogen for life on Earth is evident in its ubiquitous presence in biological molecules such as amino acids, proteins and nucleic acids. Like other key elements essential for life, nitrogen flows through environmental systems in a dynamic biogeochemical cycle in which microorganisms and plants are an integral part. Plants require greater amounts of fixed nitrogen for growth than other essential nutrients. When soils are deficient in nitrogen and the requirement of plants is not adequately met, plant growth and health are depressed. In a unique biological relationship, specialized microorganisms that inhabit terrestrial and aquatic environments have evolved the ability to fix nitrogen and make it available for plants to utilize for photosynthesis and growth.

Modern agricultural management practices have leveraged the nitrogen requirement of plants to increase food production and to provide an adequate supply of food for consumption by humans and livestock. The application of nitrogen-based fertilizers, soil amendments, and the co-cultivation of leguminous cover crops provide nitrogen to deficient soils and dramatically augments crop yield. The provision of nitrogen subsidies (i.e., fertilization) to food crops is one of the most important contemporary agronomic advancements in meeting increasing global demand for food, fuel, and fiber. This advance has not been without consequence, however. Technological advances in agriculture, as well as in industrial manufacturing and urban practices, have disrupted the biogeochemical nitrogen cycle, principally by generating "fixed" (i.e., chemically reactive) nitrogen in excess of the assimilative capacity of ecosystems (Vitousek *et al.* 1997). In simple terms, Earth's biogeochemical budget is out of balance due to human activities. Emerging from this modern disruption of balanced nitrogen cycling is a wide array of adverse environmental effects and ecological impacts. The most remarkable impacts—which are ever increasing in magnitude on a global scale—include the leaching of nitrate that contaminates groundwater reserves, the eutrophication of surface waters and resultant "dead zones," atmospheric deposition that acidifies terrestrial, freshwater and coastal ecosystems, and the emission of the greenhouse gas, nitrous oxide (N_2O), that is also the dominant stratospheric ozone substance (Ravishankara et al. 2009). Moreover, these environmental changes are widespread and of high severity, and are increasingly associated with deleterious human health effects. The direct and indirect human health effects of human alteration to the global nitrogen cycling include acute poisoning, chronic exposure to newly emerged infectious diseases, and malnutrition facilitated by increased pestilence of food crops (see Fan and Steinberg 1996, Galloway *et al.* 2008, Galloway *et al.* 2004, Guillette and Edwards 2005, Johnson *et al.* 2010, Jordan and Weller 1996, Lavelle *et al.* 2005, Townsend *et al.* 2003, Vitousek *et al.* 1997 for extensive review).

2.2 The Nitrogen Cycle

The flows and fluxes of the major chemical elements on Earth (i.e., carbon, nitrogen, oxygen, hydrogen, phosphorous, sulfur) are regulated by an efficient system of biogeochemical cycling. Over geological

Adminsitrative Record
Page 34330

and evolutionary time (and until the advent of human agriculture), these biogeochemical cycles have provided the necessary nutrients in the proper amounts to support life on Earth. Nitrogen, as one of the major essential elements, participates in a global biogeochemical nutrient cycle in which the element exists in a continuous state of transformation and translocation within and among the atmosphere, hydrosphere, and biosphere. At any given time and space in the environment, nitrogen may be incorporated, released, or chemically converted by plants, animals, and microorganisms in air, water, and soils. This cyclical flow and flux of nitrogen in the Earth's natural systems is illustrated by the nitrogen cycle (Figure 22).

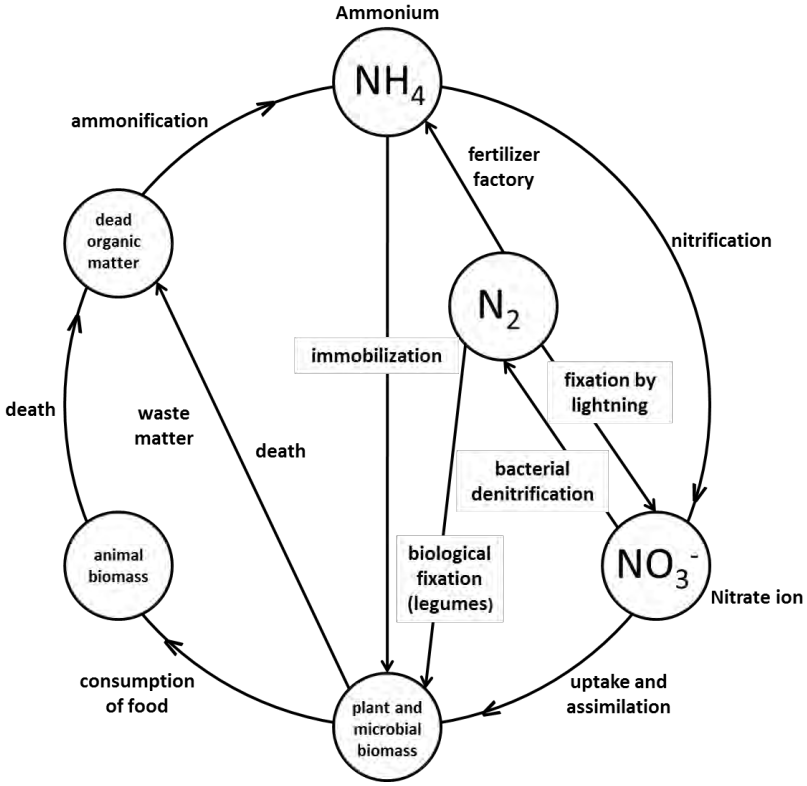


Figure 22. Dominant pathways of the nitrogen cycle (adapted from Stonecypher 2010).

Most of the elemental nitrogen on Earth exists in its gaseous, diatomic form (N_2), taking up 78% by volume of the Earth's atmosphere. At standard conditions, diatomic nitrogen is a colorless, odorless, inert gas. The chemical stability of elemental nitrogen can be attributed to the strength of the interatomic triple bond of the dinitrogen molecule (N_2). Excluding certain specialized microbial species, plants and most other living organisms are unable to use inert atmospheric N_2 directly for photosynthesis, cellular metabolism, and incorporation into structural membranes, cells, and tissues. Yet, as primary producers, plants serve in a critical role as the main entry point for the flow of nitrogen (and other nutrients) into higher levels of the trophic food web. Therefore, a pivotal stage in the cycling of nitrogen is its transformation from inert N_2 to a biologically useful—or reactive—form for plants. Once incorporated into plant tissue, nitrogen is accessible to the diets of higher-level trophic organisms such as humans and other animals.

Adminsitrative Record
Page 34331

2.3 Natural ("Biogenic") Processes of the Nitrogen Cycle

In the environment, nitrogen may be present in organic or inorganic form, and may be converted between the two forms by biological activity depending on local environmental conditions. In soils, most nitrogen is typically in organic form. This includes the living biomass in plant roots, animals, and microbial communities, but predominantly occurs as slowly decaying organic matter and stable humus. Although nitrogen is most abundant in soils in the organic form, it is inorganic nitrogen that plants can most easily utilize. Plants take up inorganic nitrogen in soils as nitrate or ammonium (NH_4^+) and convert it to organic form for use internally in metabolic reactions and growth. Later, the assimilated nitrogen cycles back to the soil as organic matter when plants and other organisms die and are decomposed by microorganisms.

Most of the transformations of N in the environment are biological, and largely microbial. The rates of these processes are dependent on the physical and chemical environment and the community of organisms present. In addition, human activity can alter the N cycle by changing the environment and by altering the amount of reactive N in the environment. In the following section, the fundamental processes of the nitrogen cycle are described with emphasis on cycling in soils.

2.3.1 Biological Nitrogen Fixation: N_2 to Ammonium

Biological nitrogen fixation,(BNF) is the enzymatic conversion of atmospheric N_2 to ammonia (NH_3), which is rapidly converted to less toxic forms. This process is performed by specialized free-living and symbiotic microorganisms (Franco and Muns 1982). The nitrogen fixing microorganisms responsible for biological N_2 fixation include species of *free-living* Cyanobacteria and Proteobacteria, and numerous species from the domain Archaea that inhabit soils and use N for their own needs. Additionally, *symbiotic* nodulating Rhizobial bacteria that infect and establish communities within the roots of leguminous plants (e.g., alfalfa and clover) and non-Rhizobial bacterial symbionts (e.g., Frankia) associated with a few non-leguminous actinorhizal plants (e.g., alder or *Alnus* spp.) fix nitrogen as well. In this case, the fixed N is exchanged with the plant in return for photosynthetically fixed carbon. This N acquired by N fixing organisms increases the overall N supply in ecosystems as it becomes generally available either after the death or consumption of the N fixing organisms. Prior to the industrial revolution, BNF was the most important process for N to become available in reactive forms in ecosystems.

2.3.2 Ammonification: Organic Nitrogen to Ammonium

Nitrogen mineralization is the microbial process which results in the conversion of organic forms of N in the soil to inorganic forms. Organic matter is biologically derived, and includes decaying plants and animals, microbial biomass, crop residues, humus, leaf litter, and scat. As a result of mineralization, organic nitrogen that was previously immobilized in living organisms is gradually rendered available to plants for uptake and use as inorganic ("mineralized") nitrogen. The rates of microbial activity associated with mineralization are relatively slow and dependent on the local physical and chemical factors

Adminsitrative Record
Page 34332

including temperature, moisture, pH, and oxygen content, as well as the type of organic material available. The first step in N mineralization, ammonification involves the release of ammonium during the decomposition of organic matter. The specialized microbes that can create the appropriate enyzmes derive energy from the chemical breakdown of the organic matter, but do not take up all N released by the breakdown of the more complex organic molecules.

2.3.3 Nitrification: Ammonium to Nitrate

Nitrification is the process by which reduced inorganic nitrogen in the form of ammonium is oxidized to nitrate. Under aerobic conditions, nitrifying microorganisms oxidize ammonium (NH_4^+) first to nitrite (NO_2^-) and ultimately to nitrate (NO_3^-). The gram-negative chemoautotrophic bacteria *Nitrosomonas* and *Nitrobacter* are largely responsible for the coupled nitrification processes. These organisms release the nitrate to the soil because they are using these nitrogenous compounds as a source of energy and not as a source of nitrogen for growth.

The nitrification reactions are coupled so that very little toxic NO_2^- accumulates in the soil, and nitrate production is favored. During the various enzymatic reactions of nitrification, several gaseous intermediate products can be emitted from soils prior to the formation of nitrate. Nitrogen may be emitted as nitric oxide (NO), which contributes to smog formation, or nitrous oxide (N_2O) which is a potent greenhouse gas and deplete stratospheric ozone. Once nitrate has been formed it can be immobilized by microbes or plants. However, any nitrate remaining in the soil solution can be easily leached from soils by rainfall or irrigation events because it is negatively charged and does not stick to soil particles.

2.3.4 Immobilization: Inorganic to Organic Nitrogen

Nitrogen immobilization represents the uptake of inorganic nitrogen from the soil by microbes, plants, fungi, and algae for assimilation and use in cellular metabolic reactions and the assembly of biomolecules. Immobilization is considered a loss of available nitrogen from soils, albeit temporarily until the new biomass is mineralized once again. An important factor influencing whether mineralization or immobilization by microbes is more likely in soils is the carbon to nitrogen ratio (C:N) of the organic matter (Ambus and Zechmeister-Boltenstern 2007). Most decomposers use carbon as a source of energy while simultaneously assimilating nitrogen for incorporation into structural cellular compounds. However, the uptake of carbon and nitrogen is dependent upon a certain critical C:N ratio, the equilibrium threshold of which hovers around 20:1. Soils that possess a high C:N (>30:1) will favor microbial N immobilization since adequate carbon is present for uptake by soil microbes. In cultivated soils, the C:N ratio increases when organic amendments such as crop residues or animal manure are applied as fertilizer. Conversely, soils with a lower C:N ratio (<20:1) will favor mineralization since inadequate carbon is present for microbial uptake and metabolism. When microbial immobilization is favored at high C:N ratios, there is more competition among microbes and plants for the inorganic N supply in the soil.

Adminsitrative Record
Page 34333

2.3.5 Denitrification: Nitrate to Dinitrogen Gas

The nitrogen cycle is closed by microbial denitrification, a critical pathway that returns reactive nitrogen from back to the atmosphere as N_2. The denitrification process is carried out by a variety of species of facultative anaerobic bacteria that are present in wide-ranging conditions, and in vastly different ecosystems. Important denitrifying bacteria include species of *Thiobacillus*, *Micrococcus*, and multiple species of *Psuedomonas*. These bacteria exhibit heightened metabolic activity in environments that are rich in nitrogen and organic matter and low in oxygen, including saturated soils and wetlands, heavily fertilized cropland, manure lagoons and animal lots, septic waste systems, wastewater discharges, and land-applied sludge and biosolids (Sprent 1987; Smith 1999). Because nitrate is typically produced in environments with oxygen present, denitrification will only proceed when nitrate is transported from where it is produced to environments with minimal oxygen.

Similar to nitrification, these organisms use nitrogen for purposes other than the production of organic molecules, like proteins. In the absence of oxygen, microbes can use nitrate as the terminal electron acceptor in a series of redox reactions involving electron transfers. The electron donor may be either a reduced inorganic compounds, or dissolved organic carbon (DOC). The chemical nature of the electron donor is important as it dictates the metabolic capacity, and thus, the type of bacteria involved in denitrification. For example, heterotrophic bacteria utilize DOC as electron donors, and are ubiquitous in most environments. In contrast, autotrophic bacteria utilize inorganic compounds, and are less abundant than their heterotrophic counterparts (Beller *et al.* 2002). This is particularly relevant in light of nitrate loading to anaerobic aquifers, where DOC may be limited, but where sulfide minerals such as pyrite may serve as the critical electron donor for denitrifying bacteria in unsaturated aquifer sediments (Schwientek *et al.* 2008; Spalding and Parrott 1992).

The prevalent pathway for returning nitrogen from the biosphere to the atmosphere is the reduction of NO_3^- to N_2, where NO_3^- is used as a terminal electron acceptor in the anaerobic respiration of denitrifying bacteria. The typical sequence of exothermic reductions proceeds as follows (Beller *et al.* 2002; Sprent 1987; Zumft 1997):

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$$

Nitrate → nitrite → nitric oxide → nitrous oxide → dinitrogen gas

Denitrification occurs in the opposite direction as nitrification. As with nitrification, NO and N_2O are intermediate byproducts that can be emitted during incomplete denitrification, and contribute to air pollution and greenhouse gas emissions.

2.4 Anthropogenic Sources of Nitrogen and Environmental Consequences

Contemporary explorations in biogeochemical research, such as ice core extraction and analysis, suggest that the earliest global nitrogen cycle was in a balanced state devoid of artificial nitrogen sources or

Adminsitrative Record
Page 34334

sinks (Canfield et al. 2010). The net result of the natural cyclical processes of biogenic fixation and denitrification effectively counteracted one other, placing the system in dynamic equilibrium (Canfield et al. 2010, Galloway et al. 1995). However, clear and compelling evidence demonstrates that the once steady-state conditions of the global nitrogen cycle have been significantly disrupted by the activities of modern humans (Galloway et al. 2004).

2.4.1 A New Paradigm: The "Nitrogen Cascade"

A modern approach to understanding the impact of excess nitrogen on ecological systems depicts the element as existing in two general pools: non-reactive nitrogen (N_2) and reactive nitrogen (N_r) (Galloway et al. 2003). N_2 refers to nitrogen in its inert diatomic gaseous form, the most abundant form of nitrogen on Earth. Conversely, reactive N (N_r) includes all other forms of biologically and chemically available nitrogen, regardless of source, or whether fixed via biological or anthropogenic processes. Whereas N_2 is inert and harmless, N_r is rapidly transformed. Thus, whatever the original form of N_r there is significant potential for it to be transformed to forms of N that can adversely affect the environment. Chemically, N_r comprises all inorganic and organic nitrogen-containing compounds at any given point in global, local, or compartmentalized nitrogen cycles. To illustrate, reduced inorganic species of nitrogen may take the form of ammonia (NH_3) and ammonium (NH_4^+). Oxidized inorganic nitrogen may include nitrous oxide (N_2O), nitric acid (HNO_3), nitrite (NO_2^-) and nitrate (NO_3^-), nitrogen oxides (NO_x) and a variety of other forms in water and the air (Sprent 1987). Finally, organic N consists of carbon-containing compounds such as amino acids, proteins, and nucleic acids (e.g., DNA and RNA) (Beever et al. 2007, Galloway et al. 2003).

The contemporary nitrogen cycle is characterized by anthropogenic modification of the rates and magnitude of transformations of N: the fixation of N_2 to N_r, reactions of the various N_r compounds, and the production of N_2 by denitrification. The multiplicity of consequences, both positive and negative, associated with N transformations have been summarized in the *nitrogen cascade* conceptual model (Galloway et al. 2003). A single molecule of nitrogen can have multiple effects between the time it is fixed and denitrified. For example, N can be industrially fixed to create fertilizer, applied to a crop field, immobilized into a plant that is harvested and eaten by humans, which would be a positive consequence. However, the N in food that we eat can cause nutrient enrichment in streams receiving treated wastewater followed by the biological conversion to the greenhouse gas and stratospheric ozone depleting compound nitrous oxide prior to being denitrified. Wherever and in whatever manner the N is fixed, there can be global consequences. Elevated levels of N_r have been linked directly to expanding anthropogenic activities, especially the creation and application of nitrogen-rich fertilizers used extensively in agricultural activities and human dependence on fossil fuel combustion for energy over the past century (Townsend et al. 2003).

2.4.2 Anthropogenic N_r

The rate of anthropogenic creation of N_r is rapidly exceeding the rate of natural biological fixation, and perhaps more seriously, the rate of denitrification of N_r to N_2 by microbial communities (Galloway et al.

Adminsitrative Record
Page 34335

2003). As a consequence, N_r is accumulating in aquatic, terrestrial, and atmospheric sinks (Galloway and Cowling 2002). At the current rate of anthropogenic nitrogen fixation, it is estimated that human contributed N_r will rise by 30% over the next three decades (Lavelle et al. 2005).

In the past 40 years, global nitrogen inputs have approximately doubled, while biological nitrogen fixation has decreased due to conversions of natural land and overall changes in land use (Galloway et al. 2004, Lavelle et al. 2005). The major sources of anthropogenic nitrogen flows to the environment are from agricultural, industrial, and urban practices, especially those associated with the production and use of food and energy. The magnitude of environmental loading from these anthropogenic sources is variable across time and space, but is geographically widespread and has continuously increased for the past six decades. Some loading sources may be more localized or concentrated in a geographic area, as exemplified by agricultural lands, dairies and concentrated animal feeding operations, and septic tank clusters ubiquitous in rural areas. These sources often compound, such as the use of grain, silage, and fodder as animal feed, which in turn concentrates nitrogen as excrement, which in turn is then re-applied to agricultural lands on a highly localized basis. Other loading sources are comparatively more diffuse, as illustrated by wastewater treatment plants and large-scale food processors that discharge biosolids to land, and as is the case with the profligate combustion of fossil fuels associated with aerial and automobile transportation. A notable commonality among anthropogenic nitrogen sources is that they are nonpoint sources, making them exceedingly difficult, if not impossible to contain. Consequently, the environmental impacts induced by such wide-ranging sources of N_r are cumulative and long-lasting, and include marked degradation of groundwater, surface waters, and air quality.

Agricultural, industrial, and urban practices all contribute to newly reactive nitrogen, but agricultural fertilization – overwhelmingly from the manufacture and application of synthetic fertilizers derived from petroleum – more than any other source, has substantially augmented anthropogenic nitrogen fixation and loading to the environment (Galloway and Cowling 2002). The principal consequences associated with excess nitrogen loading from agroecosystems are described in sub-sections below. However, to understand the *nitrogen cascade* it is important to recognize that the global driver for increased food (and fiber and biofuel) production comes from increasing global human population and its increasing wealth. Maintaining current production levels already requires substantial input of anthropogenically generated, synthetic nitrogen. Future increases in crop production at a similar or even reduced applied nitrogen levels requires significant changes in agricultural practices. The nitrogen cascade is a result of a complex interplay between agronomic practices and environmental variables, particularly soil and water. Given the potential negative consequences of limiting fertilizer or irrigation, practice has been to apply water and fertilizer in sufficient quantities to overcome any limitation. Increases in one factor often result in necessary increases in the other. For example, to achieve desired leaching levels for salinity control, additional water application has meant additional nitrogen application to assure plant uptake objectives. To date, the potential economic consequences of limiting factors have outweighed potential environmental costs, such as excess anthropogenic nitrogen in the environment. Presently we have technology and knowledge to reverse many of the contributing factors leading to the *nitrogen cascade*, but there are also scientific and economic limits to its minimization.

Adminsitrative Record
Page 34336

2.4.3 Reactive Nitrogen from Agroecosystems

The successful execution of the Haber-Bosch reaction (1909) was a foundational scientific accomplishment that soon after gave rise to the extensive use of synthetic nitrogen-based fertilizes in agricultural food production. The Haber-Bosch process is characterized by the high-pressure, high-temperature chemical conversion of N_2 to NH_3. Today, the industrial fixation of N_2 to produce NH_3 is widely used in the manufacturing of synthetic fertilizers, explosives, plastics, resins, nylon, and other raw materials. The industrial production of synthetic fertilizers, via the Haber-Bosch process, is seen as the overriding contributor of N_r in the global environment. Synthetic fertilization of agricultural crops has bestowed high crop yields, financial profitability, and the large-scale production of affordable food for the world's growing population. However, a significant portion of nitrogen applied in food production exceeds the necessary amount for desired crop yields. Indeed, much of the nitrogen applied is in excess of crop uptake. This excess nitrogen can leach into groundwater, cause eutrophication of aquatic ecosystems after surface runoff, and contribute to various forms of air pollution. Each of these negative environmental impacts has the potential to impart significant consequences to biodiversity and human health at local and regional levels. Cumulatively, the long-term societal, environmental, and economic costs of industrial fixation of nitrogen, its over-utilization, and resulting severely altered global nitrogen budget, are potentially dire.

2.4.4 Agricultural Application of Nitrogenous Fertilizers

When new plant growth occurs, nitrogen, carbon, and other nutrients are incorporated into the plant as organic biomass. Upon the natural death of plants or plant parts, some of this biomass is decomposed, and inorganic N is rendered available to other plants and soil microbes by the gradual process of mineralization. However, with the repeated harvesting of crops from cultivated soils, plant biomass is permanently, or at least disproportionately, removed from the soil-plant system as a potential source of nitrogen and other nutrients. Therefore, in cultivated soils, the harvesting of crops rapidly depletes soil fertility and establishes nutrient demands greater than natural cycles of mineralization and immobilization can satisfy. Moreover, soil fertility may be further diminished by poor irrigation practices that increase salinization and erosion. This occurs from the application of chemicals (e.g., pesticides that unintentionally suppress the diversity and total amount of beneficial soil biota), and by soil acidification resulting from atmospheric deposition of previously volatilized air pollutants (e.g., ammonia).

To support successive crop production within a harvested field, artificial or supplemental sources of reactive nitrogen in excess of naturally-occurring sources are required. These include minerals, synthetic inorganic fertilizers and nitrogen-rich irrigation water, as well as organic sources such as animal manure, biosolids, compost, and leguminous cover crops and residues. Because agricultural fertilization practices tend to result in surface runoff, leaching of nitrogen as nitrate, and increased gaseous nitrogen losses as NH_3 and NO, the objective of nitrogen supplementation on farmland should be to maximize uptake of nitrogen by crops while simultaneously minimizing losses of valuable nitrogen to soils and other environmental sinks. Therefore, sustainable nitrogen management in the context of

Adminsitrative Record
Page 34337

agroecosystems hinges on a pendulous balance between efficient nitrogen inputs and costly nitrogen losses that increasingly lead to collateral impacts on environmental systems.

2.5 Environmental Effects of Excess Nitrogen

2.5.1 Volatilization of Ammonia

Volatilization is the loss of nitrogen to the atmosphere as gaseous ammonia (NH_3). In soils, a dynamic equilibrium exists between ionized ammonium (NH_4^-) and un-ionized ammonia in solution. The extent to which one is formed over the other depends principally on three factors: pH, temperature, and moisture. In wet, cool, low pH soils, ammonia combines with water to form ammonium. Because soils have a net negative charge and ammonium is a positively charged ion, ammonium remains adsorbed on soil exchange surfaces and experiences relatively little mobility. The low mobility of ammonium in soils makes it readily available for absorption across the roots of plants. However, under dry, warm, and high pH conditions, ammonium is quickly converted back to gaseous ammonia, increasing the likelihood that it is lost by volatilization to the atmosphere.

$$H_3O^+ + NH_3 \rightleftharpoons H_2O + NH_4^+$$

Agricultural fertilizers and animal waste associated with concentrated animal operations are two major sources of ammonia volatilization to the atmosphere. Between 55 and 95% of annual anthropogenic discharges of NH_3 have been attributed to agriculture, particularly livestock operations and the land applications of fertilizers (Schepers and Raun 2008). Synthetic fertilizers are typically ammonia- or urea-based, though urea-based fertilizer is the most common type of fertilizer applied to agricultural soils, as it is the safer of the two types to handle. Urea [$(NH_2)_2CO$] is also a major component of nitrogenous waste from mammals, and is released to the environment in large amounts from the urine and feces on concentrated animal lots and manure application to fields. The urease enzyme is ubiquitous in soils and quickly hydrolyzes urea from these sources to ammonia, promoting gaseous loss of nitrogen from soils to the atmosphere.

Ammonia is highly reactive with oxide air pollutants in the atmosphere (including ubiquitous nitrogen oxides, NO_x), and its emission contributes to the formation of ambient particulate matter along with other major sources such as the combustion of fossil fuels and burning of biomass. Ambient particulate matter causes and exacerbates respiratory illness and lung disease, such as asthma and lung cancer, as well as cardiovascular disease (Hristov 2011, WHO Report 2003). Furthermore, volatilized nitrogen returns to the Earth's surface via atmospheric deposition of air pollutants. Atmospheric deposition is a major contributor to acidification and eutrophication of terrestrial and aquatic environments globally. Therefore, the impacts to air quality and water quality are intensified when excess nitrogen is applied as fertilizer or discharged as waste to the environment.

Adminsitrative Record
Page 34338

2.5.2 Reactive Nitrogen and Air Quality

2.5.2.1 Nitrous Oxide is a Greenhouse Gas and Depletes Stratospheric Ozone

Like other forms of reactive nitrogen, nitrous oxide (N_2O) has been increasing in the atmosphere for the past two centuries, with the greatest rate of increase occurring over the past few decades (Davidson 2009, Galloway 2004). N_2O is emitted as an intermediate byproduct of the biogenic processes of microbial nitrification and denitrification as well as a unintended product of fossil fuel combustion. Anthropogenic activities that have increased nitrogen loading to the environment have concomitantly augmented N_2O emissions by spurring the rates of microbial nitrification and denitrification in terrestrial and marine ecosystems. N_2O enhances the "greenhouse effect" by trapping heat from solar radiation onto the Earth's surface, raising regional and global temperatures and driving climate change. Although carbon dioxide (CO_2), methane (CH_4), and halogenated gases are also important greenhouse gases, N_2O is remarkable in its potential contribution as a greenhouse gas and global climate change due to its extensive atmospheric life – approximately 120 years (Howarth et al. 2005). It is the third most important greenhouse gas in terms of radiative forcing, and has been tracking increasing CO_2 concentrations in the past two decades with a 11% increase in concentration since 1998 (IPCC AR4).

In addition to its contribution to the greenhouse effect, and subsequently regional and global climate change, N_2O can undergo conversion to nitric oxide (NO) by reacting with oxygen in the atmosphere. The resulting NO reacts easily with stratospheric ozone (O_3), depleting the protective layer that shields the Earth from exposure to harmful levels of ultraviolet radiation (Beever et al. 2007).

2.5.2.2 Nitrogen Oxides Contribute to Ozone Formation and Urban Air Pollution

Nitrogen oxides (NO_x) are emitted by mobile and stationary sources, such as automobiles and smoke stacks, during the production and combustion of fossil fuels. When volatile organic compounds (VOCs), carbon monoxide (CO), and other air pollutants undergo photochemical oxidation in the presence of NO_x, highly reactive O_3 is produced. Whereas stratospheric O_3 is protective and a vital component of the Earth's upper atmosphere, tropospheric (ground level) O_3 is a dangerous constituent of urban smog. Owing to its highly oxidative and unstable chemical properties, ground level O_3 induces physiological and cellular damage in all living organisms, and exerts adverse impacts on human health, wildlife, natural vegetation, and the viability of food crops. Furthermore, NO_x can react with moisture in the atmosphere to form nitric acid (HNO_3). HNO_3 returns to Earth via atmospheric deposition of acid rain, acting as yet another exogenous source of reactive nitrogen to the biosphere, and raising the acidity of the receiving ecosystems.

2.5.3 Atmospheric Deposition

Atmospheric deposition is an increasingly important anthropogenic source of nitrogen to the biosphere and hydrosphere. Atmospheric deposition occurs when substances in the air are transported to the surface of the Earth. Deposition of a substance from the atmosphere may be termed "wet", referring to

Adminsitrative Record
Page 34339

the transport of substances via condensation and liquid precipitation (e.g., rain, snow, or fog), or "dry", as when substances are transported as aerosols, gases, or particulate matter.

Worldwide, the levels of trace nitrogen gases in the atmosphere have risen as a function of increased anthropogenic nitrogen fixation (get from EML). Fertilization of agricultural cropland, the combustion of fossil fuels, and the burning of biomass in the form of forest and grassland vegetation releases gaseous nitrogen as NH_3, N_2O, and NO_x. These highly reactive gases are emitted to the atmosphere, carried varying distances from the original source, and eventually deposited via precipitation in terrestrial and aquatic ecosystems. Therefore, atmospheric deposition of previously volatilized gaseous nitrogen constitutes an additional source of reactive nitrogen in the biosphere, and contributes to increased primary productivity, eutrophication, and acidification of surface waters.

Atmospheric deposition of nitrogen represents a significant environmental problem because in most temperate and boreal ecosystems, nitrogen is the rate limiting nutrient in primary productivity and biomass accumulation (Vitousek et al. 1997). The chronic deposition of a limiting nutrient such as nitrogen can dramatically disturb normal ecosystem equilibria, and alter the presence of intrinsic biota and the distribution of species, in addition to localized biogeochemical cycling and ecosystem functioning. For example, eutrophication in lakes, estuaries and coastal zones is increasing due to atmospheric deposition of nitrogen, as well as other nutrients commonly found in surface runoff (get from EML). The negative consequences of eutrophication are many, and include harmful algal blooms, the depletion of dissolved oxygen, and the release of toxic chemicals from decomposing organisms that endanger aquatic wildlife and lead to fish kills (Carpenter et al. 1998).

2.5.4 Eutrophication of Water Bodies

Eutrophication is the loading of surface water bodies with exogenous and excessive nutrients, especially nitrogen and phosphorus, which leads to significant increases in biomass in the form of algae and other phytoplankton. The deterioration in water quality associated with eutrophication is the most common environmental modification of freshwater ecosystems in the United States (Carpenter et al. 1998). The elimination of phosphorous in common detergents over the past several decades has now made nitrogen the most significant nutrient input and contributor to eutrophication of surface waters.

Excess nutrient loading derives from a variety of point sources (e.g., wastewater outflows and effluent) and non-point sources (e.g., surface runoff, groundwater seepage and discharge, and atmospheric inputs) (Smith et al. 1999). Streams and rivers, lakes and reservoirs, and estuarine and marine water bodies are the ultimate recipients of these excess nutrients, placing the waters at significant risk of eutrophication and threatening the aquatic wildlife. Eutrophication typically involves a distinct sequence of environmental impacts. Augmented levels of otherwise limited nutrients, including P and N, lead to harmful blooms of algae and other phytoplankton that result in the release of toxic compounds from some species. Aquatic hypoxia and anoxia also result when the phytoplankton die and oxygen-consuming decomposition occurs (Beever et al. 2007). Ecologically-important consumers such as fish, mollusks, and insects are vulnerable to the effects of the toxic compounds and reduced oxygen levels, and ultimately, shifts in species composition and widespread fish kills may occur. The harmful

Adminsitrative Record
Page 34340

algal blooms and the resulting mortality of aquatic organisms are responsible for the notorious malodor and degraded quality associated with green, eutrophic waters. Notably, episodic or continuous eutrophication can also be devastating for local fishing economies when toxic algal blooms cause fish and shellfish to be inedible and thus unmarketable (Smith et al. 1999).

2.5.5 Nitrate Leaching to Groundwater

Nitrate is the most ubiquitous contaminant of groundwater resources. Nitrate in groundwater may derive from a number of natural and anthropogenic sources including intrinsic geologic origins, application of synthetic fertilizers and animal manure, leakage from defective septic tanks, discharges of wastewater and biosolids from wastewater treatment plants and industrial food processors, atmospheric deposition of nitrogen pollution, and the over-cultivation of nitrogen-fixing crops. However, despite the multitude of potential sources, it is the extensive (i.e. widespread and chronic) use of agricultural fertilizers in intensive food production that has incomparably led to nitrate contamination of groundwater (Foley et al. 2005, Jordan and Weller 1996).

Nitrogen fertilizers are often applied to cultivated soils in exceedance of crop requirements because plant nitrogen use efficiency is often less than 100%. Therefore to overcome potential losses to the atmosphere, immobilization, denitrification, among many fates, application rates often exceed actual plant uptake. When coupled with excess water in a sufficiently permeable soil root zone, the excess nitrogen escapes via leaching through the soil profile. As a result of its charge, nitrate is not absorbed by most mineral soils, which can result in more rapid transport in leaching water relative to cations (Follet and Delgado 2002). When water is available to mobilize nitrate, it readily moves through interpore spaces and past the upper soil layers, through the intermediate vadose zone, and into subterranean aquifers (Follet and Delgado 2002). Nitrification and denitrification only occur where: a) appropriate bacterial population species are present, b) nitrate is present, and c) conditions favor metabolism of nitrogen. Deeper layers are one of many environments that can meet these conditions. Consequently, soluble nitrate may leach through the soil to groundwater aquifers and concentrate to levels that create a public health risk to populations that procure drinking water from groundwater wells (Jordan and Weller 1996, Power and Schepers 1989).

Notably, it is only within the medium of water that nitrate is transported to aquifers. In other words, even when unusually high levels of reactive nitrogen exist within the soil profile, nitrate leaching will not occur unless the rates of water infiltration (i.e. deep percolation) exceed the rates of evapotranspiration (Smith and Cassel 1991). For this reason, reducing deep percolation to groundwater from agricultural soil (by curbing inefficient or poorly practiced irrigation methods) is equally important as reducing excess levels of N fertilizer applied to cultivated lands. To put this in perspective, consider the two primary uses of water application in croplands: one, to maintain plant turgor for high-crop yield; and two, as a transporting medium to carry agrichemicals through the root zone. Without water, agrichemical transport through the root zone to plants and beyond to deep percolation would be impossible. Thus irrigation management is equally as important as nitrogen management in reducing groundwater contamination of agrichemicals.

Adminsitrative Record
Page 34341

Water in the root zone is not always from irrigation, as seasonal precipitation will affect soil saturation and transport capacity under saturated and unsaturated conditions. Thus, it is also important to acknowledge that changing hydroclimatic conditions include the increasing likelihood of extreme precipitation events that could potentially overwhelm well-designed irrigation strategies intended to reduce nitrate leaching. This reality reinforces the necessity of having strategies that account for rates of uptake in plants, fate in soils, and timing of application and removal of nitrogen in cropland agricultural practices.

2.6 Nitrogen Fluxes in Croplands: A Mass Balance Approach to Groundwater Nitrate Loading

In our study, we used a nitrogen mass balance approach to estimate nitrate loading from all cropland, by crop type (crop category), except for alfalfa cropland. Mass balance is the practice of analyzing physical systems by accounting for the amount of material entering and leaving a system of interest, and relies on the conservation of mass (i.e., mass can neither be created nor destroyed). This approach allows us to approximate flows of material, such as nitrate, that might otherwise have been unknown or difficult to measure (e.g., leaching to groundwater). This approach is often employed, including for nitrogen mass balance in surface waters of the Central Valley (Sobota et al. 2009). Water quality data from monitoring wells installed downgradient of fields receiving manure applications indicate that the nitrate concentration in recharge from these fields is closely related to the nitrogen losses estimated from a field-scale nitrogen mass balance (e.g., Burow et al., 2008; VanderSchans et al. 2009).

We do not include alfalfa into the cropland mass balance analysis, because most of the N taken up by alfalfa and removed as harvested N was obtained directly from atmospheric N via nitrogen-fixing bacteria in the root system of alfalfa plants. Only small amounts of fertilizer are typically applied to alfalfa. Manure is typically not applied to fields growing alfalfa except an unknown amount of solids that is sometimes applied prior to planting or after the last cutting in the fall. Little is known about nitrate leaching from alfalfa, which is most often grown in rotation with other field crops (corn, winter grain), particularly near dairies. Given the large amount of N fixation in alfalfa, which is directly related to its harvested N, the mass balance approach could not be performed for this crop. Instead, we use a groundwater leaching rate obtained from a field study in the 1970s (Letey et al. 1979). More research is needed to better understand the potential, if any, of alfalfa leaching to groundwater under various management practices.

The following subsections relate components of the N cycle, briefly describe known sources for N cycle components in the study area, and formulate our mass balance methodology. Methodological details for each component of the mass balance, and intermediate results, are presented in subsequent Sections (Sections 3 to 8). Our mass balance analysis does not take into account direct leakage of N into groundwater via wells, which is described separately in Section 9. A final comprehensive presentation of the results of the mass balance analysis is presented in Section 1.

Adminsitrative Record
Page 34342

2.6.1 Basic Concepts

Deriving current and historical estimates of nitrate loading to groundwater for a particular cropped field (cropland) requires, at minimum, two pieces of information: 1) the amount of N inputs to a field, N_{input}, including fertilizer, organic amendments (manure, effluent, biosolids, etc.), atmospheric deposition, and irrigation source water nitrate and 2) the amount of known N outputs from a field, $N_{output,}$ including harvested N, atmospheric losses, and runoff (see sections 2.6.2 and 2.6.3):

$$N_{GW} = N_{input} - N_{output} \qquad \text{(Eqn. 1)}$$

where N_{GW} is the mass of total nitrogen leached to groundwater (kg N/ha), mostly in form of nitrate-nitrogen (or a nitrate percursor).

Field-level N mass balances make one important assumption, in that they assume long-term (decadal or multi-decadal) steady state dynamics of soil N. That is, the amount of N mineralized from soil organic matter is equal to that immobilized by microbes. Hence, long-term N storage changes in soil structure are assumed to be negligible. The applicability of this assumption for California croplands, systems, and soils is unclear. It has been shown that the N in cultivated California soils has increased somewhat over the past 50 years, but the effect was only marginal (approximately 0.20%) (Singer 2001). The N accumulation is likely greatest soon after cultivation begins and decreases over time. The only study that directly tested the steady state assumption showed mixed results. Lund et al. (1982) examined long-term cropping on a variety of soils at four sites, mostly in the Santa Maria Valley. The results demonstrate that steady state assumption was valid for two of the four sites.

Despite its limitations, the mass balance approach presents clear advantages for estimating historical leaching rates. To begin with, using a mass balance approach allows one to calculate a field or soil N balance as the difference between the amount of N harvested and removed from the field in products and the amount of N fertilizer (organic or inorganic) applied. Calculating the rate of N applied in excess of plant uptake, referred to as "surplus", is important because it is nearly all released into the environment, with the majority transiting to groundwater. Further, isotopic N research has shown that less than 10% of the applied N is taken up in subsequent seasons (Ladha et al. 2005). It is possible that N immobilized into the soil may be released at time frames longer than 1-3 years following application, but N release at these timescales is not well constrained (Gardner & Drinkwater 2009). For this reason, we compute the nitrogen mass balance over an extended time period.

How large is the potential error due to the steady state assumption? If the total soil N increase was 0.2% over 50 years (Singer 2001), the total nitrogen flux into permanent soil storage would be 400 kg N/ha (360 lb N/ac). This amounts to an annual nitrogen flux into fixed soil storage of 8 kg N/ha/yr (7 lb/ac/yr), a fraction of the annual average nitrogen fertilizer and other N fluxes in agricultural lands (Section 1). Hence, the steady state-based mass balance approach is well suited for a *post hoc* analysis of long-term, decadal to multi-decadal, average nitrogen fluxes into and out of the root zone of agricultural lands.

Adminsitrative Record
Page 34343

2.6.2 Field Nitrogen Mass Balance in Cropland: Conceptual and Mathematical Model

The mass balance analysis is performed separately based on two different sets of data describing the amount of land area occupied by specific crops: one mass balance is computed for the land area of each crop reported in the county agricultural commissioner reports (ACR, see Sections 1.6 and 3), and another mass balance is computed for the land area using CAML mapped areas (see Sections 1.8 and 3).

The ACR land areas are reported in tables and can be used to obtain county-wide or crop-specific estimates of groundwater leaching. Unlike the tabularized ACR dataset, the digital CAML map allows for simulating the spatial distribution of biosolids, effluent, and manure N separately for each individual facility (including animal facilities), to the specific land these facilities own, and to the specific crops that these amendments are typically applied to.

In either case, groundwater nitrate loading from agricultural fields is computed based on a mass balance of the known or estimated inputs and outputs to an individual field in the CAML land use map or to an individual crop category (also considered a field) of the ACR tabularized data. The mass balance on agricultural cropland is performed regardless of the source of the nitrogen and applies equally to fields receiving commercial fertilizer, dairy manure directly on a dairy, dairy manure exported from dairies, effluent or biosolids from wastewater treatment plants (WWTPs), effluent or biosolids from food processors (FPs), or a combination thereof.

Because current and future groundwater nitrate concentrations are the results of a long history of nitrate loading, the annual mass balance is performed in 15 year intervals from 1945 to 2005 to 2050 (representing 8 time periods in 105 years). For groundwater modeling purposes (see Technical Report 4 by Boyle et al., 2012), annual groundwater loading at each field is linearly interpolated from those nine period years for which nitrate loading estimates were computed. For example, a field's groundwater nitrate loading in 2001 is equal to the sum {4/15th of the computed 1990 loading estimate + 11/15th of the computed 2005 loading estimate}.

The nitrogen mass balance is performed on the root zone of each field and considers only annualized fluxes into and out of the root zone. On the input side, each field root zone receives nitrogen from the following sources:

- N from atmospheric deposition, $N_{deposit}$
- N contained in the source irrigation water (well, stream), N_{irrig}
- N from synthetic fertilizer, N_{fertil}
- N from manure, where applied, N_{manure}
- N from WWTP/FP effluent or biosolids, where applied, $N_{WWTP-FP}$

Adminsitrative Record
Page 34344

On the output side, the following pathways are considered:

- N in the harvest, $N_{harvest}$
- N losses to the atmosphere via volatilization or denitrification, N_{loss}
- N loading to groundwater, N_{GW}
- N in surface runoff, N_{runoff}

We derive estimates of all of the above terms independent of the mass balance computation, except N_{GW}, which is estimated as closure to the basic mass balance equation:

$$N_{GW} = N_{deposit} + N_{irrig} + N_{fertil} + N_{manure} + N_{WWTP\text{-}FP} - N_{harvest} - N_{loss} - N_{runoff}$$

The terms on the right-hand side of this mass balance equation are defined as follows:

$N_{deposit}$: The amount of current and historic atmospheric N deposition, $N_{deposit}$, is described in section 7. Current atmospheric N deposition is spatially variable across the study area. For TLB land in agricultural production, excluding alfalfa, $N_{deposit}$ totals 10.696 Gg N/yr, at an average of 9.8 kg N/ha/yr (11,790 t N/yr at 8.7 lb N/ac/yr), while $N_{deposit}$ totals 0.848 Gg N/yr, averaging 5.6 kg N/ha/yr (940 t N/yr at 5.0 lb N/ac/yr), for the cropping area of the Salinas Valley. We used the current and historic statewide emissions data from the California Air Resources Board to estimate historic and future N deposition. Historic and future NO_x deposition was based on NO_x emissions reported by the California Air Resources Board (ARB).[12] As the ARB estimates begin in 1975, we assumed a linear decrease to zero NO_x emissions going backward to 1900. If the current decreasing trend in NO_x continues, then by 2050, there will again be zero NO_x emissions. The past and future of NH_3 emissions is poorly delineated because NH_3 is not a criteria pollutant. Similar to past NO_x emissions, we assumed a value of zero NH3 emissions for 1900. However, we assumed a linear increase to the current day based on the continued growth of livestock populations. Because of the uncertainty in NH_3 regulations, we considered three possible scenarios for 2050: 50% lower emissions, constant emissions, and doubled emissions. Calculations of the ratio of historic and future N deposition to current N deposition are assumed to be proportional to total statewide N emissions, shown in Table 8, below. For the simulation results presented in Section 1.8 and in the Appendix to this report, the intermediate scenario was used for the atmospheric N input.

[12] http://www.arb.ca.gov/app/emsinv/fcemssumcat2009.php

Adminsitrative Record
Page 34345

Table 8. Statewide N emissions and ratio of past and future N emissions to current atmospheric N emissions. The 2050 scenario assumes that NO_x emissions from automobiles become negligible. For 2050, three scenarios are considered for NH_3 emissions: half, current, and twice of current.

Year	Statewide NO_x emissions (Gg N)	Statewide NH_3 emissions (Gg N)	Total statewide N emissions (Gg N)	Ratio to current statewide emissions
1945	292	78	371	0.69
1960	393	105	497	0.92
1975	493	131	624	1.16
1990	499	157	656	1.22
2005	355	183	538	1
2050	0	92/183/366	92/183/366	0.68/0.34/0.17

N_{irrig}: The amount of nitrogen in irrigation water, N_{irrig}, can vary locally. For our analysis, N_{irrig} is approximated by assuming a study area average of 450 mm/year (1.5 AF/ac/yr) of irrigation water originating from groundwater in the TLB and 600 mm/yr (2 AF/ac/yr) of irrigation water originating from groundwater in the SV. The nitrate concentration in the irrigation water is assumed to be equal to the median nitrate in public water supply systems within each groundwater sub-basin (as defined by DWR) for the period between 2000 and current (Table 9, also see Technical Report 4 by Boyle et al., 2012). Surface water as irrigation water is assumed to contain only negligible amounts of nitrate nitrogen (also see Section 8 in this report). Prior to 2005, N_{irrig} is assumed to have increased linearly from zero in 1945 to the value shown in Table 9 for 2005. For future years, we assume that nitrate concentrations in irrigation water continue to increase at the same linear rate through 2050.

Adminsitrative Record
Page 34346

Table 9. Median groundwater nitrate in community public supply wells for 2000-2009, by DWR groundwater sub-basin (Boyle et al., 2012), groundwater use, and calculated basin-wide average nitrate-nitrogen application rate from pumped irrigation water.

Groundwater Subbasin	Sub-basin ID (DWR)	Median Nitrate [mg/L]	Groundwater Use [mm/yr]	N_{irrig} [kg/ha/yr]	N_{irrig} [lb/ac/yr]
Pressure Aquifer	3401	23	600	30.7	27.4
East Side	3402	29	600	38.7	34.5
Forebay	3404	17	600	22.7	20.3
Upper Valley	3405	4	600	5.3	4.7
Seaside	3408	10.6	450	10.6	9.5
Langley	3409	11	450	11.0	9.8
Corral de Tierra	3410	4	450	4.0	3.6
Madera	52206	3	450	3.0	2.7
Delta-Mendota	52207	1	450	1.0	0.9
Kings	52208	24	450	24.0	21.4
Westside	52209	4.8	450	4.8	4.3
Pleasant Valley	52210	0	450	0.0	-
Kaweah	52211	23.3	450	23.3	20.8
Tulare Lake	52212	1	450	1.0	0.9
Tule	52213	23	450	23.0	20.5
Kern	52214	16	450	16.0	14.3

For the county ACR based analysis presented in Section 1.6, we used ACR reported cropland area and an average N_{irrig} = 22.8 kg/ha/yr (20.3 lb/ac/yr) on irrigated cropland. Hence, the total annual nitrogen contribution from irrigation water in the study area is 29.0 Gg N/yr (26.4 and 2.6 Gg N/yr in the Tulare Lake Basin and Salinas Valley, respectively) [32,000 t N/yr total; 29,100 t N/yr and 2,900 t N/yr in the Tulare Lake Basin and Salinas Valley, respectively] (see Figure 3 in Section 1).

$N_{WWTP-FP}$, N_{manure}, N_{fertil}: The rate of effluent and biosolids nitrogen applied on fields belonging to WWTPs and FPs has been determined independently from permit records and other sources describing these facilities (see Section 6). The amount of manure N and the amount of synthetic fertilizer applied depend on the location of a field: If the field is *not* part of the cropland receiving direct application of (primarily liquid) manure within a dairy (see Section 4.8.5) or of effluent and biosolids within a WWTP-FP operation, then the amount of synthetic fertilizer, N_{fertil}, that is applied is equal to the typical amount of fertilization, N_{norm}, for the particular crop grown in the field for which the mass balance is performed (for tabulation of N_{norm}, see Section 3 and Appendix Tables). The soil amendment nitrogen is simulated as excess applied nitrogen.

Adminsitrative Record
Page 34347

In essence, this approach a) sums all manure generation in the TLB, b) distributes that manure proportionally within the TLB, based on crop type, while c) accounting for manure used on dairy property (see Section 4). Thus, a vineyard receives much lower soil amendment rates than, e.g., a lettuce field. The proportionality factor is equal to the total amount of manure N exported in the county or study area (depending on the scenario, see Section 1.8), $N_{AreaManureExport}$, divided by the totalized recommended fertilizer application rate for all croplands outside dairies and WWTP-FPs, $N_{TotalNorm}$. The amount of soil amendment nitrogen (from exported manure), N_{manure}, on a given field outside the application area of a dairy is computed by multiplying N_{norm} with the ratio $N_{AreaManureExport}$ / $N_{TotalNorm}$. This approach ensures that the amount of manure applied as soil amendment is exactly the amount of manure exported.

Double-Cropping: Typical fertilizer application rates, N_{norm}, which are discussed in Section 3, are for individual crops. For the N mass balance, we consider typical annual fertilization rates and annual harvest rates (see Section 3). The two types of crops identified in CAML that are typically subject to double cropping are corn and several vegetables (celery, lettuce, spinach, broccoli, cabbage, cauliflower, and Brussels sprouts). For our field-by-field N mass balance, we assume that all fields identified as "corn" are double-cropped with grain and we adjust both, the N_{norm} and the $N_{harvest}$ for these crops by adding the N_{norm} and $N_{harvest}$ values (see Section 3). This is an operational assumption. In practice, an unknown, but presumably small fraction of corn acreage is single cropped.

An analysis of the USDA agricultural census data for Monterey County was used to estimate the amount of multi-cropping in the seven vegetables listed above (Table 10). For the 1990 period, we assumed a multi-cropping factor of 1.6 for these seven vegetable crops. For the 2005 period, we assumed a multi-cropping factor of 1.7 for these seven vegetable crops. In simulating the mass balance, the values of N_{norm} and $N_{harvest}$ for these crops were multiplied with 1.6 and 1.7, respectively.

Table 10. Data and estimation procedure to derive values for the multicropping rate of seven vegetable crops (celery, lettuce, spinach, broccoli, cabbage, cauliflower, and Brussels sprouts). Rows A, B, and D are obtained from NASS agricultural census data for Monterey County.

	1987	1992	2002	2007
A: Harvested land area, seven multi-cropped vegetables [ha]	57,110	80,236	97,037	88,365
B: Harvested land area, all vegetables [ha]	70,812	94,376	110,464	102,750
C: Harvested land area, single-cropped vegetables (B minus A) [ha]	13,702	14,140	13,427	14,385
D: On the ground land area, all vegetables [ha]	50,888	65,269	73,194	66,111
E: On the ground land area, seven multi-cropped vegetables (D minus C) [ha]	37,186	51,129	59,767	51, 725
F: Number of crops harvested per year, seven vegetables (A divided by E)	1.53	1.57	1.62	1.71

On-Dairy N Use and WWTP/FP Land Application: Within a dairy, on fields receiving manure N (much of it from liquid manure), the amount of synthetic fertilizer, N_{fertil}, and the amount of manure N applied, N_{manure}, is computed differently from other cropland. Within dairies, the amount of fertilizer and

Adminsitrative Record
Page 34348

manure N applied, N_{fertil} and N_{manure}, is a function of the crop dependent agronomic annual fertilization rate, N_{norm}, and a function of the amount of manure available, which is determined by the number of animals in a dairy and across a county (see Section 4). For fields belonging to a WWTP or FP and receiving land application nitrogen from effluent or biosolids or both, $N_{WWTP-FP}$, the amount of synthetic fertilizer used is computed in an equivalent manner to fields on dairies.

To determine the amount of N_{fertil} and N_{manure} for each field, we make the following important assumptions:

1. The agronomic, "typical" annual fertilization rate, N_{norm}, is a crop-dependent value, which is listed in Section 3. That section also describes the historic variation, by crop, of the agronomic rate, N_{norm}.
2. A farmer will apply, at a minimum, the typical agronomic rate, N_{norm}, by using either fertilizer N, N_{fertil}, or manure N, N_{manure}, or a combination of both. As a result, the sum of fertilizer and manure applied is either equal to or in excess of typical rates, N_{norm}:

   $$N_{fertil} + N_{manure} \geq N_{norm}$$

   $$N_{fertil} + N_{manure} = N_{norm} + N_{excess}$$

   where N_{excess} is the annual rate of N applied in excess of recommended rates.

3. The following rules are assumed for non-dairy cropland including dairy cropland not used for application of liquid manure (e.g., tree crops, vineyards):
   a. On non-dairy cropland, all of N_{norm} is satisfied by applying commercial, non-manure fertilizer. Hence, for non-dairy cropland, $N_{fertil} = N_{norm}$
   b. On non-dairy cropland, any application of manure N, N_{manure}, obtained by export from a dairy (manure solids, composted manure solids), as described in Section 4, is thought to be applied as a soil amendment, but not to meet fertilization needs. In other words, the manure N applied is in excess of recommended fertilization rates, $N_{manure} = N_{excess}$
4. The amount of manure N exported from dairies is a fixed, county-specific fraction of the manure N excreted on each dairy, which in turn is a function of the number of adult animals on each dairy. Several potential scenarios for the amount and fate of the exported manure are simulated, as discussed in Section 1.8 and described in more detail in Section 4.
5. For manured dairy cropland (land use classes grain and cotton, field crops and corn, pasture), we assume the following:
   a. An individual dairy's total available land applied manure N, dairy-ΣN_{manure}, is distributed to manured dairy croplands of the individual dairy in relative proportion to a field's agronomically recommended rate, N_{norm}:
   b. For each dairy, we compute the sum of N_{norm} across all dairy cropland receiving manure, dairy-ΣN_{norm}.

Adminsitrative Record
Page 34349

c. All of a dairy's land applied manure N is distributed to manured cropland in relative proportion to each crop's agronomic needs, N_{norm}:

$$N_{manure} = N_{norm} \bullet \text{dairy-}\Sigma N_{manure} / \text{dairy-}\Sigma N_{norm}$$

d. The amount of manure applied relative to a field's agronomic N need, N_{manure} / N_{norm}, is constant across all manured dairy cropland of an individual dairy. It equals the ratio of a dairy's total land applied manure, dairy-ΣN_{manure}, to a dairy's total agronomic N need on its manured crops, dairy-ΣN_{norm}. From dairy to dairy, this ratio changes.

e. If sufficient manure N is not available, we assume that the difference between a dairy's N available from land-applied manure, dairy-ΣN_{manure}, and a dairy's agronomic N needs, dairy-ΣN_{norm}, is made up by synthetic fertilizer, N_{fertil}.

f. In addition, based on the agronomic practices that we observe among dairies in the Central Valley, we assume that a dairy satisfies at least half (50%) of its total agronomic N needs from synthetic fertilizer, regardless of the amount of land applied manure (at least until 2007, when the new Dairy General Order went into effect for the TLB). Hence the actual total applied fertilizer N on a dairy, dairy-ΣN_{fertil}, is the larger value of these two:

$$\text{dairy-}\Sigma N_{fertil} = \text{Max}(\text{dairy-}\Sigma N_{norm} - \text{dairy-}\Sigma N_{manure}, 0.5\ \text{dairy-}\Sigma N_{norm})$$

g. Like manure, synthetic fertilizer is distributed to individual manured fields in a dairy in relative proportion to the field's agronomic N needs:

$$N_{fertil} = N_{norm} \bullet \text{dairy-}\Sigma N_{fertil} / \text{dairy-}\Sigma N_{norm}$$

h. In this modeling approach, the total amount of excess N, N_{excess}, varies from dairy to dairy and – within a dairy – from manured crop category to manured crop category.

i. Dairies without excess N are those that generate a total amount of land-applied manure, dairy-ΣN_{manure}, that is less than half of a dairy's agronomic needs, dairy-ΣN_{norm}. Groundwater nitrate loading from crops on those latter dairies are comparable to those on non-dairy farms.

j. Fertilizer applications, N_{fertil}, in land application areas of WWTPs and FPs are simulated in the same fashion as N_{fertil} in land application areas of a dairy.

The above set of assumptions oversimplifies the actual complexity of dairy nutrient management, but it defines a rule-set that can be used to consistently simulate, for each individual agricultural field in the study area, the amount of fertilizer N applied each year, N_{fertil}, and the amount of manure N applied each year, $N_{manure,}$ if any, based on best available data.

$N_{harvest}$: The rate of nitrogen annually removed from a field with harvest, $N_{harvest}$, was obtained from county ACR data and is described in Section 3.

Adminsitrative Record
Page 34350

N_{runoff}: Surface runoff losses to streams are assumed to reach 14 kg N/ha/yr (13 lb N/ac/yr, Beaulac and Reckhow, 1982). This study focused mostly on watersheds in the Midwest, but the source is still widely cited. Calculations were also performed based on USGS data[13] for the Central Valley. We used the solver function in MS Excel™ for the 18 subwatersheds to find the best fit values of % cropland, urban land, and natural land in each watershed to predict the export coefficients for each land use. In both cases the value is 14-15 kg N/ha (13 lb/ac/yr). We note that this is a higher amount of N runoff than computed for the CV-SALTS Tule River pilot project with the WARMF watershed model (2-5 kg N/ha/yr or 2-5 lb N/ac/yr).[14]

N_{loss}: This is the annual rate of gaseous losses due to ammonium volatilization and denitrification in the root zone and at the land surface of cropland after application of commercial fertilizer or manure. The rate of nitrogen gases emitted from agricultural fields (N_2, N_2O, NH_3, and NO_x) in California is not well constrained. We use a default emissions factor of 10% of applied nitrogen to account for total gaseous emissions. The emission factor is derived from available data and reported as percentages of nitrogen applied:

- N_20: 1% The default emissions factor of direct field emissions used by the IPCC (De Klein et al. 2006).
- N_2: 1.8% This emissions factor is based on the average N_2:N_2O ratio reported in agricultural sites (Schlesinger 2009).
- NH_3: 3.6% Average emissions measured from 10 California fields (C. Krauter et al. 2009).
- NO_x: 2.1% Average emissions across 8 crops and 20 sites (Matson et al. 1997).

Based on these four fluxes, a total of 8.5% of applied nitrogen is emitted to the atmosphere as gas. Thus, the assumption to 10% is reasonable, if not conservative (also see the literature review in the Committee of Consultants report, Harter et al., 2007). N_{loss} is estimated to be 10% of all input N, not only synthetic fertilizer or manure N:

$$N_{loss} = 0.1\ N_{fertil} + 0.1\ N_{manure} + 0.1\ N_{WWTP\text{-}FP} + 0.1\ N_{deposit} + 0.1\ N_{irrig}$$

Inserting this equation into the mass balance equation above, we obtain the following equation, which defines annual groundwater nitrate-N loading rate [kg/ha/year] for each individual agricultural field in the study area:

$$N_{GW} = 0.9 * (N_{deposit} + N_{irrig} + N_{fertil} + N_{manure} + N_{WWTP\text{-}FP}) - N_{harvest} - N_{runoff}$$

The total amount of nitrate-N loading to groundwater from each field is computed by multiplying N_{GW} with the area of an individual field. This mass of N can then be summed across regions, counties, groundwater sub-basins, or the entire study area. The concentration of nitrate [mg N/L] in the recharge

[13] http://pubs.usgs.gov/sir/2010/5228/
[14] http://intpln.com/Docs/Final_SNSPIS_Report_Submittal_02.22.10_rs.pdf

Adminsitrative Record
Page 34351

from an individual field is computed by dividing the field-specific nitrate-N loading rate [kg/ha/yr] with the field-specific recharge rate [thousand m^3/ha/yr] (see Technical Report 4, Boyle et al., 2012).

Historic loading is simulated according to the following assumptions about historic changes in the above variables:

1. Typical nitrogen application rates, N_{norm}, vary linearly between estimates obtained for 1945 and those obtained for 1975. N_{norm} for current time (2005-2010) is assumed to have been constant since 1990.
2. Harvested N, $N_{harvest}$, is defined based on ACR data (see Section 3)
3. Manure excreted from dairies is adjusted according to the number of animals in each county and according to the average milk production for the above time periods. In 1945 and 1960, we assume that the amount of manure application outside of pasture is negligibly small and that most cows were grazed on irrigated pasture. Hence, no manure land application was simulated for this period (see Section 4).
4. Prior to 1981, manure N export from dairies is assumed to be negligible.Between 1980 and 2005, exports are assumed to increase linearly from 0% to the full export fraction defined by the individual export scenarios (see Section 1.8 and Section 4.8.4).
5. For each dairy, the same land parcels are considered for manure applications, going back to 1970, but the application rate accounts for the historic changes in land use on these parcels over time (see Sections 3 for landuse changes and Section 4.8.4 for crops receiving manure on dairies).

N_{GW} on Alfalfa: Alfalfa and clover are the two key crops identified in CAML that are leguminous, that is, they are able to capture inert atmospheric nitrogen and incorporate it as organic nitrogen into plant material. For the county mass balance, we did not consider alfalfa, since it does not receive significant amounts of fertilizer.

But for the spatially mapped N mass balance, the amount of nitrogen fixation in alfalfa is not taken into account in the mass balance equation above. We did not estimate the amount of N fixation, which depends on crop growth, soil status, and nutrient applications including atmospheric deposition, soil amendments including manure, and fertilizer. For the CAML analysis, we do consider alfalfa as part of the crop area that receives manure and atmospheric deposition as described above. We also consider the harvest N (per Section 3), atmospheric loss, and runoff losses as defined above.

However, without taking into account N fixation, the mass balance for alfalfa fields would be negative. To properly account for groundwater leaching from alfalfa, we use a fixed groundwater leaching rate in alfalfa of 30 kg N/ha/yr (27 lb N/ac/yr), which corresponds to values reported in a recent NSF study (Letey et al. 1979). Alfalfa land area in the 2010 CAML map encompasses 161,000 ha (400,000 acres). It is therefore important to properly account for N fixation by appropriately fixing the groundwater losses in alfalfa.

Adminsitrative Record
Page 34352

2.6.3 Nitrogen Mass Balance Computation: The "Groundwater Nitrate Loading Model" GNLM

The crop- and county-specific nitrogen mass balance based on ACR land area data was computed with MS Excel spreadsheets. Assumptions about manure distribution among crops are explained in the footnotes to the tables and figures in Section 1.6.

The nitrogen mass balance computed based on the CAML land area distribution was performed on 5 million, squared sub-field areas, of which each is 0.25 ha (0.6 ac) in size. The mass balance algorithm described above was coded into a Matlab® program that we call the "Groundwater Nitrate Loading Model" (GNLM). GNLM automatically performs the analysis for all eight time periods on each of the 5 million sub-field areas, including the various scenarios for exported dairy manure outlined in Sections 1.8 and 4. GNLM also includes the actual or simulated spatial distribution all non-cropland sources of groundwater nitrate: urban areas, golf courses, septic systems, dairy corrals, dairy lagoons, and percolation basins of WWTPs and FPs (see Sections 4–6). GNLM does not, however, include the effects of nitrate loading through dry wells, abandoned wells, or ill-constructed active wells (Section 9).

2.6.4 Potential Sources of Uncertainty in Mass balance Calculations

While we cannot quantify the exact amount of nitrogen loading that has taken place given the uncertainties expressed herein, our estimates do clearly demonstrate the magnitude of the issue and their relative sources. There is considerable uncertainty in the mass balance calculations employed herein, and variation is inherent in each parameter in the equation. Rates of emission are simply too variable in cropping systems at the scale of our analysis and, thus, it is impossible to determine the amount of nitrogen emissions for any given year, crop, or management very precisely. We therefore adopted an inclusive approach to provide a range of plausible nitrogen loading rates. The challenge in characterizing nitrogen loading should not be understated, as it should consist of both spatiotemporal accuracy and precise quantification. Our method provides a transparent and robust estimate of the potential loading rates over the past 60 years and into the future through 2050.

For direct sources of nitrate to groundwater, the research described in Sections 3 through 9 attempts to provide a reasonable range for the likely nitrate loading to groundwater (see Table 1 in Section 1 for a summary). An error analysis based on Monte Carlo simulation was performed on the study area total groundwater nitrate loading rate from cropland, as described in Section 1.6.5. A validation of the data against California-wide estimates of nitrate loading to groundwater obtained from a review of field studies (Section 3) is discussed in Section 1.6.6.

Adminsitrative Record
Page 34353

3 Cropland Nitrogen Loading

3.1 Introduction

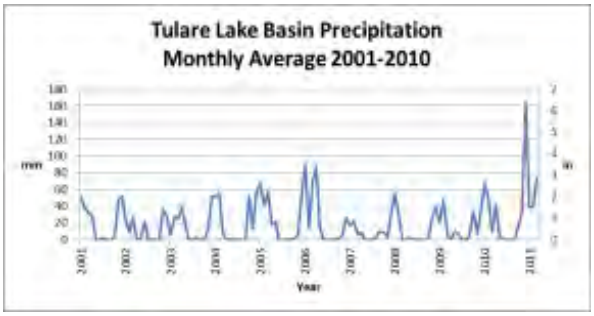
Nitrogen fertilizer use in crop production has long been recognized as a potential water pollution concern. Studies published as early as 1963 discussed N misuse in agriculture and the threat it posed to California groundwater resources (Harding et al. 1963). Scientific evidence continues to mount that fertilizer use contributes to nitrate percolating below the rootzone and accumulation in aquifers. Data derived from studies using radioactive isotopes, soil cores, soil water collection, irrigation and domestic well water collection, and mass balance all point to one conclusion: common fertilization, irrigation, and soil management practices place California groundwater resources at risk of nitrate contamination beyond established legal limits (Francis E Broadbent & Rauschkolb, 1977; K. R. Burow, Dubrovsky, & Shelton, 2007; Karen R Burow, Shelton, & Dubrovsky, 1996; Gardenas, Hopmans, Hanson, & Simunek, 2005; Jackson, Stivers, Warden, & Tanji, 1994; Mangiafico et al., 2009; Miller & Smith, 1976; Pang, Letey, & Wu, 1997; Pratt, 1979).

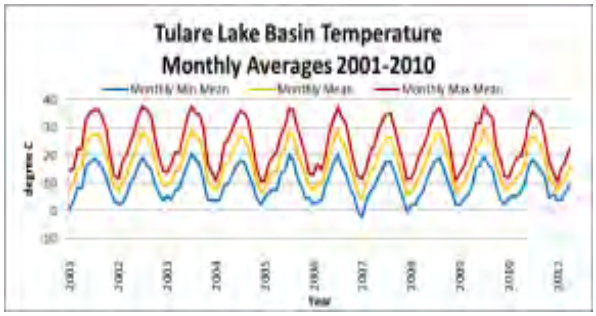
Nitrate leaching from cropland is a well-recognized and well-studied issue globally (Sutton et al. 2011). However, nitrate leaching in California's cropping systems is unique by comparison to other temperate agricultural areas. California's semi-arid climate creates two distinct management periods. During the summer growing season (approximately 15 April–15 October), conditions in the Salinas Valley (SV) and Tulare Lake Basin (TLB) are characterized by hot daytime air temperatures (> 35 °C, Figure 23) and negligible precipitation. The lack of summer precipitation, and the resulting dry soils, generally present low leaching potentials under non-irrigated management conditions. However, irrigation is regularly applied, sharply increasing the leaching potential. This is in contrast to most rain fed agricultural systems and the winter cropping season in the SV and TLB, which are characterized by cooler daytime air temperatures and measurable precipitation in the form of episodic rain (Figure 23). Episodic rain events in this part of California are highly variable on inter- and intra-annual bases (Neiman et al. 2008), but can produce periods of intense rainfall, saturated soil conditions, and localized flooding. These rain events can create periods of acute and sporadic nitrate leaching and runoff losses (Jackson 2000).

Deep percolation of nitrate laden water does not benefit crop production. Movement of nitrate beyond the root zone represents a financial loss for the farmer and is an environmental concern. However, leaching is sometimes a consequence of the need to control excess salinity. Generally, semi-arid agriculture systems characteristic of California and the TLB in particular, tend to accumulate salts in the root zone (Schoups et al. 2005). Elevated salts are toxic to plants and can reduce yields. Increasing the leaching fraction (amount of water moving beyond the root zone) is a primary way of removing salts from the root zone. It has long been thought that reducing irrigation to match evapotranspiration (ET) rates cannot be considered realistic for California croplands. Letey et al. (2011) reviewed the evidence for the impacts of salinity on crop growth, and concluded that often the negative effect on productivity was less than expected, and remark that the findings suggest the need for a refinement in conventional practice. However, the practice of leaching to control soil salt balance combined with the intrinsic

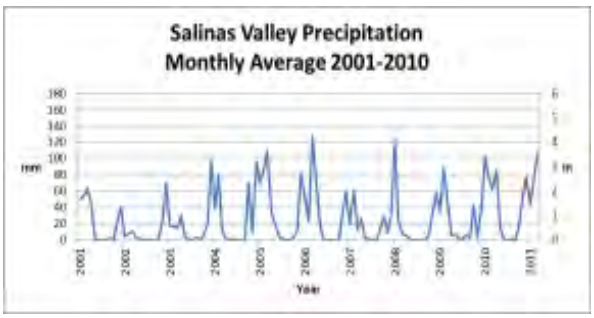
Adminsitrative Record
Page 34354

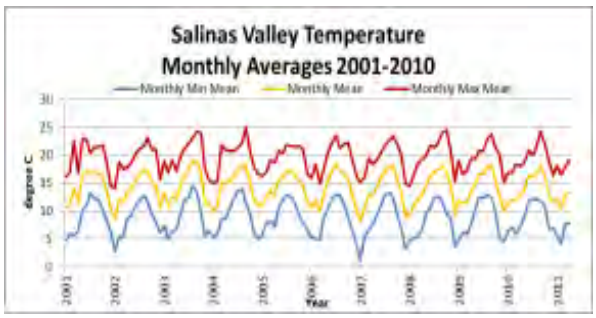
dynamics of the N cascade mean that some nitrate leaching from irrigated croplands is virtually inevitable.





Hanford 1 S, 36°19'N, 119°38'W, 72.2m AMSL





Salinas Municipal Airport, 36°40'N, 121°36'W, 22.6 m AMSL

Figure 23. Recent Climate from TLB and SV Study areas (2001-2010)

3.1.1 Management factors controlling nitrate leaching[15]

Nitrate leaching is a function of water movement and N dissolved in soil water solution. Thus, nitrate leaching can be represented by a formula with only these two factors: nitrate leaching loss (mass per area per year) = volume of water moving beyond the rootzone [volume per area per year] x nitrate concentration [mass per volume]). Field measurements of freely drained and tile drained sites growing a diverse set of annual and perennial crops throughout the major California agricultural regions established that this general equation is relevant for irrigated production of California (Table 11). From these positive relationships, it becomes apparent that management practices that either increase the amount of water percolating beyond the root zone or increase the amount of N in solution will fundamentally increase nitrate leaching losses.

[15] Readers are referred to Section 2.2 for discussion on the abiotic and biotic controls of nitrate leaching in soils.

Adminsitrative Record
Page 34355

Under irrigated conditions, nitrate leaching losses are correlated with irrigation system performance.[16] Poorly performing irrigation systems that distribute water non-uniformly, inefficiently, or both increase leaching potential. Models suggest that leaching increases exponentially when uniformity drops below a threshold of between 75% and 90% (Allaire-leung et al. 2001, Pang et al.1997). The heterogeneous spatial distribution of water applications and of soil properties causes differential soil infiltration within an irrigated field. Because irrigators often apply water to ensure that crop water needs are met everywhere in a field (to avoid plant water deficit), some areas of the field receive much greater amounts of water than others, where irrigation distribution uniformity is low.

The consequence of this management approach is increased infiltration and downward nitrate movement through parts of the soil profile. Over application of water, even under relatively uniform irrigation conditions, also contributes to low irrigation efficiency and deep percolation of nitrate. Stark et al. (1982) tested combinations of three N sources and three different amounts of irrigation (ranging from 1.0–2.0 x ET) in a trial with celery to determine nitrate movement in the soil profile. As subsequent studies in California have confirmed (e.g., Meyer & Marcum, 1998), greater amount of nitrate was found at depth when excess irrigation is applied. The importance of water management in general and high uniformity and efficiency in irrigation system management in particular cannot be understated as a primary means of minimizing nitrate leaching (see Technical Report 3, Dzurella et al., 2012).

Table 11. Relationships between nitrate leaching and N inputs and/or drainage volume in free drained California sites. (Source: Pratt 1984.)

Soil drainage	Relationship among leaching, N fertilizer, and water	Correlation (r)
Free	M = 11.7 + 3.05 W	0.77
Free	M = 13.0 + 0.469 N	0.68
Free	M = 54.5 + 0.0067 NW	0.79
Tile	M = -4.52 + 2.66 W	0.83
Tile	M = -48.9 + 3.82 N	0.72
Tile	M = 16.4 + 0.0042 NW	0.92

M = mass emissions (kg/ha), N = nitrogen inputs (kg/ha), W = drainage or effluent volume for free and tile drain systems, respectively (cm/ha).

[16] Irrigation system performance is evaluated based on two interrelated metrics of "uniformity" and "efficiency". Uniformity describes the spatial distribution of water applied or infiltrated across the field's extent. For example, one might imagine a field using furrow irrigation. Areas near the source of irrigation water (the head of the furrow) often receive substantially more water than the far end (the tail) due to the length of time it takes for water to move down the furrow and the need to minimize runoff. Efficiency is a ratio of the water consumed for beneficial purposes to the total water applied.

Adminsitrative Record
Page 34356

The role human decisions play in irrigation system performance and water management should not be overlooked. In SV and TLB, growers and their irrigators decide when, where, and how much water to apply. The operator manages soil water and, by extension, deep percolation. While pressurized irrigation systems, sprinklers and microirrigation, can precisely control water flow and thus have a greater technical potential for field uniformity and delivery efficiency, using a high-efficiency technology (e.g., drip) will only increase irrigation performance if managed properly. It is the management of those systems that results in optimal or non-optimal performance. Likewise, performance of surface irrigation systems are significantly influenced by operators and can achieve reasonable efficiency levels, though their absolute technical potential is far less than pressurized systems. As a point of reference, Hanson (1995) reported that efficiencies among irrigation types were similar in practice across nearly 1000 irrigation systems monitored in California. Drip and microsprinkler systems did not show appreciably higher performance (*ibid*.). Observed irrigation efficiencies ranged between 70 and 85% for both microirrigation and furrow irrigation. It is worth noting that actual efficiencies may be below or above this range, and that changes in management practice may have improved to capture the technical advantage of pressurized systems in the 16 years since this study was published. At least one study suggests that variance in efficiency may not have increased despite the recent use of more sophisticated equipment. When irrigation performance was measured on nine drip irrigated celery fields in the Salinas Valley, performance was low. Water application rates ranged between 85% and 414% of ET, indicating under- and over-irrigation were common despite advanced capabilities (Breschini & Hartz 2002). Celery may not be representative of other cropping systems less sensitive to water stress; however, the results illustrate the potential for current irrigation system mismanagement even with advanced technology. Though the ability to apply the desired amount of water with each application is limited by the configuration of the irrigation system and hence uniformity and efficiency are somewhat predetermined, there are many practices growers can use to optimize water delivery systems (Dzurella et al. 2012).

Although the drainage volume is the most significant predictor of nitrate leaching, the volume of leachate is only half of the equation. Also important is N concentration in the leachate itself. Generally, nitrate leaching is positively correlated with N inputs. In other words, as N is applied in increasing quantities, the potential for leaching loss also increases (Figure 24). This can simply be explained by the fact that leaching represents the greatest fraction of N loss from croplands, and thus increases with fertilizer use. The recognized objective of N fertilizer management is therefore straightforward: match the supply of nitrogen as closely as possible to the amount demanded by the cropping system (Cassman et al. 2002). Synchronization of soil-N supply with plant-N demand results in low levels of residual inorganic N, high efficiency, and low potential for pollution.

In practice, nitrogen fluxes in agricultural systems are a function of a multitude of biological and chemical processes whose rates vary across space (fields, farms, and landscapes) and time (days, months, years), and are subject to a series of constraints ranging from climate to cultivars to soil type to cultural practices. Thus, a grower is faced with balancing complex and variable relationships within and

Adminsitrative Record
Page 34357

between biology and technology. The challenge of managing these relationships – fundamentally a human endeavor – underlies the efficiency and inefficiency of N use (inorganic and organic) in croplands, as well as nitrate leaching and N's long term fate. While difficult, the main principles of improving system performance relative to N leaching have been successfully demonstrated in monitoring systems operating under various water quality permits, and can therefore be emulated.

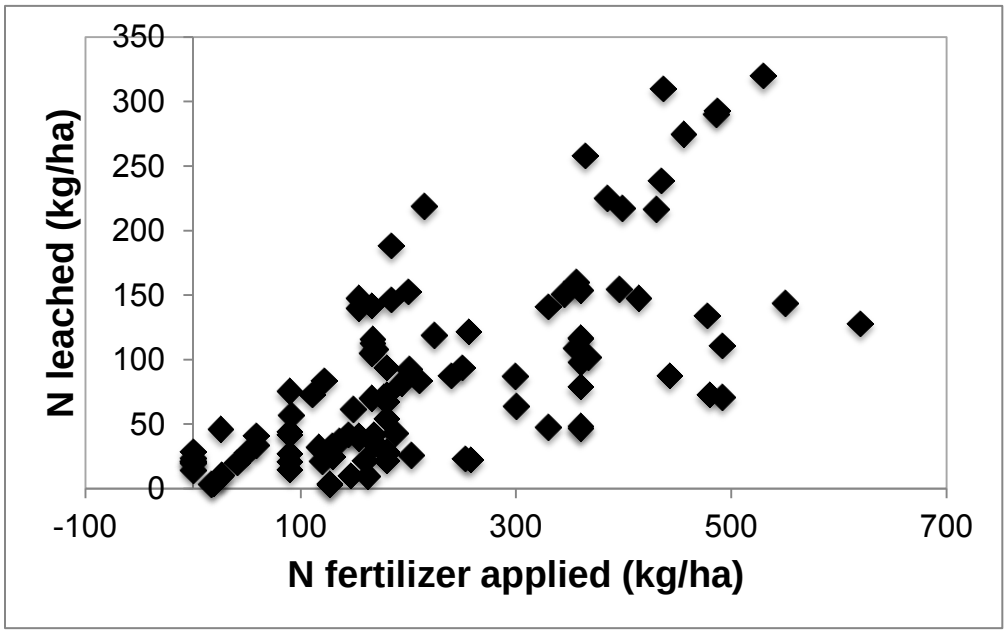


Figure 24. Current evidence of the relationship between N leached and N inputs. Based on a compilation of measurements taken in California, 1970 – 2010. Note: four outliers of high N inputs (> 1000 kg/ha) and high N leached (> 700 kg/ha) were omitted. Source: CNA (In Preparation) & Appendix I.

Nitrate leaching losses appear to be low if N fertilizer use does not exceed crop demand; whereas, once N uptake is exceeded, leaching potential increases exponentially (Broadbent & Rauschkolb 1977). Although N uptake is not the only determining factor in appropriate fertilization, this finding suggests that minimizing the amount of surplus N application is critical to controlling leaching loss. Rosenstock et al. (in review) estimate that crops in California assimilate an average amount equal to 54% of the N applied. Fruits and vegetables, many of which receive the most N per unit area, often recovered the least amount of N fertilizer. While the actual amount assimilated by a given crop will be a function of specific site and cropping system peculiarities, the exponential increase in leaching losses beyond crop demand thresholds presents growers and water quality managers an important point of reference from which to minimize nitrate leaching. It is important to note that perfectly matching N supply and demand is technically and biologically impossible. Under nearly all circumstances, even with best management practices, the amount of N assimilated by a crop will always be somewhat smaller than the amount of N (from all sources) applied to the field due to the constraints of farming (economics, infrastructure, labor, etc), the variability of soil and climate, and the complexity of N dynamics in the root zone, which cannot be perfectly predicted.

Adminsitrative Record
Page 34358

3.1.2 Field measurements of leaching in California

Despite awareness of the threat of N fertilizer to groundwater, data directly measuring nitrate flows beneath California cropland remain sparse. This is in part because of the difficulty in estimating leaching losses and partially because of the economic drivers of agricultural research; development of nutrient management practices to reduce nitrate leaching has focused on productivity and N use efficiency and not on directly quantifying leaching loss or remediation. Correlations between soil N surplus and leaching loss make the indirect approach informative. But the result is that only a small set of literature is available directly measuring leaching losses under California conditions. Much of the research was performed in the 1970s and 1980s with few measurements having been made since. Data collected during early studies represent N loading rates without changes in cropping, irrigation, and fertility practices and therefore remain uncertain.

As suggested by the preceding section, measuring nitrate leaching requires estimates of two factors: the volume of water moving beyond the root zone during a given period of time and the concentration of nitrate in that water. Gaining reliable measurement of either factor is not a trivial task as they occur well below soil surface and are highly variable within any given field, due to the intrinsic heterogeneity of soils and sediments. A number of methods to estimate or directly measure each factor have been developed over the last 30 years. The appropriate choice of monitoring tools depends on the goals of the research or the monitoring program. Many of the available methods have been applied in California. Seminal studies on leaching largely used soil cores to depths of 15 m and estimated leaching rates over more than 5 years. More recently, there has been a shift to suction lysimeters – also known as porous cups (Mangiafico et al. 2009) – and micro-lysimeters (Cabrera et al. 1993, Jackson 2000). Descriptions of techniques used to estimate nitrate leaching and their advantages and disadvantages can be found elsewhere (Webster et al. 1993, Weihermüller et al.2007).

Estimates of nitrate leaching will partially be a function of the method used to measure it. In general, the accuracy of a given method in predicting nitrate loss is inversely related to its cost and complexity. Broadbent and Carlton (1980) compared the results of soil coring with *in situ* extraction of soil solution using porous ceramic cups in corn fields on a Yolo loam soil in California with N fertilization rates of 90, 180, and 360 kg per ha, but both methods displayed considerable variability (F.E. Broadbent & Carlton, 1980). Webster et al. (1993) tested three measurement methods (soil cores, porous cups, and lysimeters) in arable cropland in England, and found good agreement in results from suction lysimeters and drainage lysimeters. Conversely, estimates derived from soil cores were generally lower and demonstrate significantly different seasonal patterns than the other methods calling into question their accuracy. Such results are concerning given many of the early estimates of leaching in California used deep soil cores to estimate nitrate leaching rates and transit to aquifers (Adriano, Pratt, et al. 1972; Adriano, Takatori, et al. 1972; Devitt et al. 1976). Because of the variability among methods, and variability of soil-nitrogen-water systems, estimates of nitrate leaching derived from relatively few measurements along a crop, soil, and management continuum must be interpreted with caution.

Adminsitrative Record
Page 34359

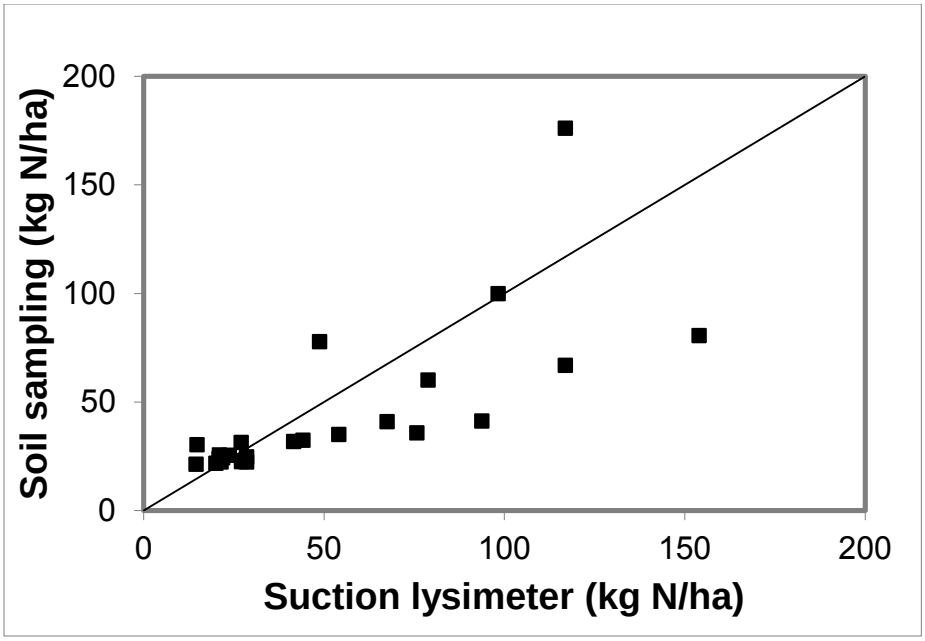


Figure 25. Comparison of nitrate leaching estimates measured by soil sampling and suction lysimeters on a fine sandy loam in California. Line is 1:1. (Source: Broadbent and Carlton 1980.)

The California Nitrogen Assessment compiled the available research measuring leaching losses from California croplands (Figure 25). By tabulating estimates reported in text or tables (but not including figures) within these published studies, a median of 78 kg ha^{-1} yr^{-1} was determined to be leached each year, which is equal to 30.2% of the applied N. Surprisingly, the median value was in near perfect agreement with the IPCC (2007) default emission factor for nitrate leaching based on global estimates of 30% (De Klein et al., 2006). The similarity between these leaching rates was unexpected because of the intense irrigated cropping systems in California, many of which utilize drip and micro-irrigation. It is likely that the median value reflects a bias towards measurements being made in the 1970s and 1980s, prior to widespread adoption of improved irrigation technology.

This analysis also determined that reported nitrate leaching losses in California irrigated cropland varied significantly, even when the same amount of N was applied. Estimated nitrate leaching losses depend on crop investigated, irrigation technology used, and length of measurements taken. Differential management creates various leaching potentials by altering the mineral N applied and potentially influencing irrigation technology and management. Cropping patterns reflect the relationship between specific crops, and inherent nutrient demands, and fidelity to specific soil types; thus, each combination of crop and soil may have inherently different nitrate leaching potentials (see Technical Report 3, Dzurella et al., 2012). As discussed previously, irrigation management can have a significant impact on nitrate leaching loss by decreasing the residence time of nitrate in areas of greatest root activity and movement of nitrate downward through the soil profile.

In the SV, measurements of nitrate leaching have been made in lettuce and cole crops. Nitrate leaching in lettuce fields has been estimated to range from 3 to 79% of N applied (Cahn unpublished, Jackson et al. 1994). The difference in the measurements seems to result from the period of observations. Cahn

Adminsitrative Record
Page 34360

et al. (unpublished) measured leaching within one season and found that leaching losses can be low utilizing an integrated nutrient and water management approach (<5 kg ha^{-1} $year^{-1}$). The results demonstrate that with low N inputs (< 125 kg N per ha) and strict water management (< 1.2 ET) nitrate leaching loss can be minimized in the systems. These results are in agreement with previous measurements by Jackson et al. (1994). However, when complete cropping considerations are taken into account (double cropping and overwinter fallow periods) nitrate leaching losses increased considerably. Jackson et al. (1994) calibrated and then applied the EPIC model, a biogeochemical model that estimates N cycling. Simulating the common double-cropping practice (two crops grown in the same field within one year) in the SV (lettuce-lettuce), nitrate leaching increased sharply to more than 146 kg N per ha. Increased nitrate leaching resulted from the mineralization of soil N from soil organic matter and crop residues between cropping events, and over the winter months when precipitation likely contributed to uncontrolled soil moisture percolation to below the root zone. LeStrange et al. (unpublished) estimated leaching losses from broccoli crops for N inputs of 134 kg/ha (120 lb/ac) and 269 kg/ha (240 lb/ac). At the higher application rate, the amount of nitrate leached increased 3 fold and was equal to double the relative percent (18% vs. 36%) of N applied (LeStrange, Mitchell, & Jackson, unpublished).

Earlier studies also estimated leaching in SV. In the mid- to late 1970s, a team of researchers from UC Riverside estimated leaching by taking samples from tile drainage effluent (Letey in Pratt, 1979). Their measurements largely taken from fields of vegetable crops suggest an average groundwater leaching rate of 34% of applied N from these systems. Collection of effluent allowed Letey and others to calculate concentrations of nitrate-N. On average concentration from tile drains in the SV were 187 mg/L nitrate (about four times the California drinking water standards). It is worth mentioning that measuring nitrate in tile drains as a proxy for leaching can distort leaching estimates, because tile drains change the matric potential of soils and may alter the observed estimates by increasing downward movement of water towards the drain. Therefore the accuracy of applying these estimates to non-tile drained fields remains uncertain.

Only a few estimates of nitrate leaching losses have been made in the TLB. Estimates made for corn and almonds (Pratt 1979) and nectarines (Onsoy et al.2005) suggest that nitrate losses were greater than 45% of the N applied. This may be partially explained by the coarse soil textures found at the study sites. Letey et al. (1977) measured nitrate in tile effluent on field sites in Tulare and Fresno counties. Tulare County sites showed low mass nitrate leaching losses (< 7% of N applied) while Fresno County sites appeared to leach more nitrate than N applied in three of the four sites studied. The latter findings of a net negative balance might have resulted from mineralization of N from soil organic matter or the mobilization of geologic nitrate (see Section 2.8).

Information gleaned from historical leaching studies must be interpreted with caution when extrapolating to estimate current and future leaching losses. A primary concern is that the cropping systems have changed over time. Yields, soil, water, and irrigation management as well as cropping practices are dynamic. That is obviously the case in the fairly widespread shift to orchard crops, but it is also true that even annual cropping systems (for which most of the early studies on leaching were conducted) have changed. Nitrogen and irrigation management methods have improved in response to

Adminsitrative Record
Page 34361

research by the University of California, and others. Examples of such technologies that have improved N efficiency include irrigation scheduling, splitting of N applications, and drip irrigation (see Technical Report 3, Dzurella et al., 2012). Only a few studies have measured nitrate leaching losses under drip irrigation. With few exceptions, much of past leaching measurement has centered on annual cropping systems. Many early studies also do not delineate which crops were being grown during a particular season (Adriano, Pratt, et al. 1972). The narrow breadth of crop diversity in these studies is problematic when one considers the large diversity of crops in California, and the diversity in irrigation, soil, and fertility management across fields and farms, between crops and even among fields of the same crops.

3.2 Landuse, Fertilizer Nitrogen Application, and Harvest: Methods

The previous section demonstrates the general lack of measured nitrate leaching rates in California, and more acutely the SBX2 1 Study Area encompassing the SV and TLB. Hence, the N mass balance approach (see Section 2.6) provides a significantly more rigorouse, consistent approach to estimating not only current, but also historical nitrate loading from cropland to groundwater. The mass balance approach quantifies the relative magnitude of N flows through the study system, and is akin to balancing a checkbook. Nitrogen mass balances have been applied at a variety of scales in California from the field or ranch to watershed to the entire state (Adriano, Pratt, et al. 1972), and fundamentally accounts for major inputs and outputs for a given production system. Performing the mass balance approach for cropland requires three pieces of information that are specific to the particular crop grown on a field: (1) knowledge of the location at which each specific crop is grown, (2) the typical amount of N fertilizer applied, by crop type, and (3) the amount of N harvested (removed) from a field, by crop type. Methods of deriving historical, current, and future estimates of these elements of the cropland mass balance are described in this section.

3.2.1 Land cover mapping

Global land cover change, in the form of natural habitat conversion to agricultural and urban uses, has long lasting and well-understood impacts on ecosystem processes. Recent studies suggest that the alteration of biogeochemical cycles – nitrogen and phosphorous cycles in particular – due to accelerated and wide-spread application of synthetic fertilizers is fundamentally changing the state and quality of ecosystems and their services (Vitousek et al. 1997), such as drinking water. Understanding the role of land use change through time – and potential surficial nitrate loading that could diminish water quality in groundwater aquifers – requires that historical, contemporary, and future land uses are not only quantified, but geographically determined. In effect, it requires a robust spatiotemporal framework of analysis.

For modeling of nitrate concentrations at drinking water wells in the SBX2 1 Study Area, it was necessary to understand the pattern of nitrogen loading on the ground surface over time. Nitrate in water supply wells of the study area have been lost from the root zone of a field, or from other sources, years and decades ago, when crop patterns and farm practices were considerably different. Similarly, nitrate loading from cropland today will affect groundwater concentrations in the foreseeable future. The

Adminsitrative Record
Page 34362

pattern of nitrogen loading across the study area is inherently a spatial issue, as different land uses will result in varying concentrations of surficial nitrogen at different locations from varying sources.

Recent advances in geospatial mapping techniques through the use of Geographical Information Systems (GIS) provide for a spatially enabled framework that combines mapping with analytical capabilities. In other words, mapping different land cover types enables modeling of nitrate loading, which, when integrated over space and time in a groundwater flow and transport model, can be used to compute well nitrate concentrations (Boyle et al., 2012). Spatial components to nitrogen loading include the locations of different crop types with varying fertilization regimes, dairy sites, the locations of septic systems and wastewater treatment plants, as well as fertilized lawns and turfgrass in urban areas, as described in other sections.

Because nitrogen loading to groundwater is cumulative over time, we developed land cover maps for several periods at approximately 15 year intervals over the past 60 years, as well as a current (circa 2005) land use map and two future land use projections. To develop the historical land use maps, we assembled statistics on crop areas at the county scale from agricultural commissioner reports (ACR) submitted by counties to the California Department of Food and Agriculture. We also provide a projected land cover map for the 2050 time frame based upon combining the current land use map with the results from an urban growth model. The GIS layer for current land use covers the entire state of California, whereas the land cover maps for the earlier time periods cover just the five counties (Monterey, Kern, Fresno, Kings, and Tulare) in the study region. The GIS layer for future land use is based on a statewide urban growth model, though our analyses are restricted to the study region.

3.2.1.1 Current Land Use

A map of current land use was developed to provide a statewide view of land cover using the most recent data sources as of June 2010. In the context of this project, the statewide view was necessary because it served as an input for a parallel project developing a nitrogen budget for the entire state of California (i.e., the California Nitrogen Assessment[17]). This map was based upon the earlier California Augmented Multisource Landcover (CAML) raster layer (Hollander 2007) developed at the Information Center for the Environment (ICE UC Davis) in 2007. This 2007 map augmented the earlier 2002 Multi-Source Land Cover (MSLC) map from the California Department of Forestry and Fire Protection by dividing its single agricultural class into the 8 agricultural classes used in the California Wildlife Habitat Relationships classification system (California Department of Fish and Game 1999), the primary focus of the MSLC map being on natural vegetation. The differences of the current map (henceforth CAML 2010) from the 2007 map include the following: 1) the data sources are up-to-date (the most recent being 2008); 2) given the agricultural focus of this project, the number of agricultural classes has been expanded, to a fairly large subset of the agricultural classes used in the DWR mapping (about 120 classes) and 3) the pixel resolution has been increased from 100 m to 50 m. A raster representation was chosen for later ease of analysis and processing: for instance, the spatial backcasting algorithm for

[17] http://nitrogen.ucdavis.edu

Adminsitrative Record
Page 34363

reconstructing historical land use that is described later on depends upon a grid cell-based contiguity and spread function.

Because different mapping efforts in the state emphasize different land cover themes, it was necessary to draw from four different data sources to compile the CAML 2010 map. These different data sources all have varying spatial resolutions, are in both raster and vector formats, and have varying levels of detail in the characteristics of land uses that they map. Figure 26 presents an overview of these datasets. First, the Department of Water Resources (DWR) Land Use Survey layers (California Department of Water Resources 2011) are a set of vector-formatted maps that emphasize agricultural land cover classes with 15 m accuracy for the linework. These have been compiled on a county-by-county basis with a return interval of about seven years. The dates of the surveys for the counties in our study region range from 1997 to 2006. The second data source was the Pesticide Use Reports (PUR) compiled by the California Department of Pesticide Regulation (California Department of Pesticide Regulation 2000). These provided supplemental information on crop types and were used in counties where no DWR surveys were performed. The PUR data are in tabular format and are spatially referenced to the nearest square-mile section (260 ha). The PUR data used in the CAML 2010 map date are from 2008. The third data source is the Farmland Mapping and Monitoring Program (FMMP) maps developed by the California Department of Conservation (California Department of Conservation 2011). These identify different types of farmlands (prime farmlands, grazing lands, etc.) and serve to track conversion of farmlands to urban lands over time. This is a vector data source with a minimum mapping unit of 10 acres. The FMMP data used in the CAML 2010 map serve as a source for urban boundaries, and dates from 2008. The final data source is the 2002 Multi-Source Land Cover (MSLC) map from the California Department of Forestry and Fire Protection (California Department of Forestry and Fire Protection 2002). This is a raster map with 100 m resolution that is used in the CAML 2010 map as a source of information on natural vegetation.

Figure 26 also outlines the workflow used in constructing the CAML 2010 map. The starting point for the CAML 2010 map was the MSLC layer from 2002. This layer combines the best regional vegetation maps into a single statewide raster map at a 100 m resolution. The land cover classes use the California Wildlife Habitat Relationships system, which is organized around differentiating habitat types for wildlife. As the MSLC layer collapses all irrigated agriculture types into a single land cover class it was used solely for the natural vegetation component of CAML 2010. The sole processing for the MSLC layer was simply to quarter the 100 m pixels into 50 m ones.

Adminsitrative Record
Page 34364

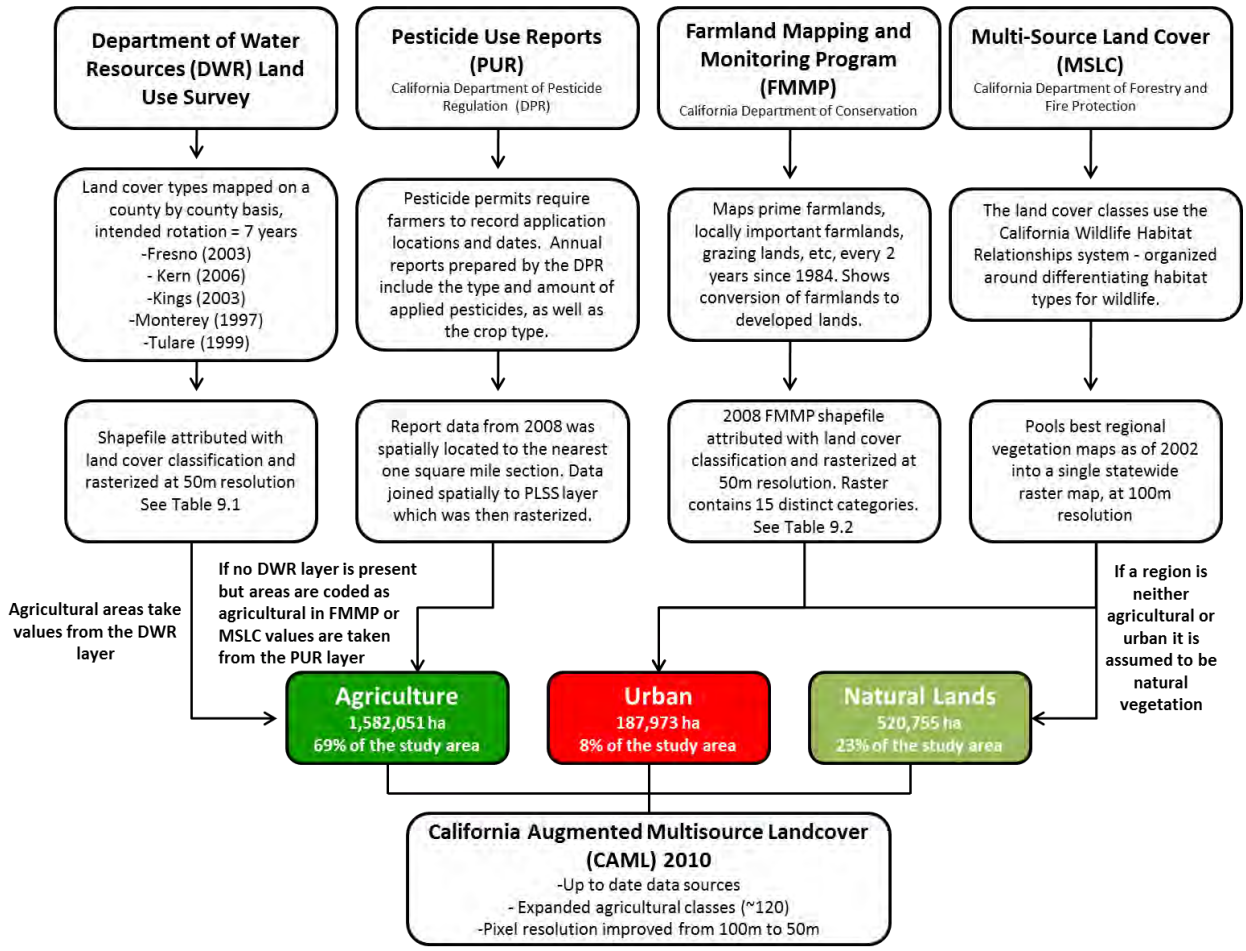


Figure 26. Flowchart of inputs to the CAML raster layer.

To fill in the agricultural regions of the CAML 2010 map, we started with the DWR land use maps. We used the most recent maps for each county, specifically 1997 for Monterey, 1999 for Tulare, 2000 for Fresno, 2003 for Kings, and 2006 for Kern County. DWR land use maps distinguish between 12 major land use classes (coded as "*class1*" in the DWR map attribute tables) including 8 major agricultural classes (grain and hay crops, rice, field crops, pasture, "truck" (i.e. vegetables and berries) crops, deciduous fruits and nuts, citrus, and vineyards. These agricultural classes are further subdivided into 89 subclasses (coded as "subclass1" in the attribute tables) that are mostly individual crop types (e.g., “cotton") but also include some lumped categories (e.g., “miscellaneous field crops"). Based on comparative corresponding areas in the DWR maps and the ACR data, most of the “miscellaneous” classes (e.g. miscellaneous subtropicals) represent minor crops. However, for the grain and hay class, the “miscellaneous” category accounts for a large fraction of the acreage because subclasses were not assigned. The GIS workflow was to load the vector shapefiles for each county into a single combined table in the spatial database PostGIS (Refractions Research 2008). Because the information on agricultural types was contained in two columns in this database table (both the "*class1*" column and the "*subclass1*" column), it was necessary to perform a relational database join across these two columns to convert them to a single integer coded table of land cover types. We then exported this

Adminsitrative Record
Page 34365

table in the spatial database to another shapefile which was then rasterized at 50 m resolution with integer-formatted values for the different land cover classifications (see Appendix Table 2) for the coding of the different land cover type).

Although this was not a concern for the study region, an issue that needed to be resolved for statewide mapping was that not all agricultural areas of the state have been mapped by DWR at any point in time even once, for example southern Santa Clara County. Yet these areas show up as agricultural regions in the MSLC map or the FMMP mapping. These areas with agricultural land classes needed an alternative source for their attribution, which was provided by PUR (California Department of Pesticide Regulation 2000). As a requirement of pesticide permits, farmers record application locations and dates with their county agricultural commissioner, who in turn reports these data to the Department of Pesticide Regulation. The PUR data include amounts and types of pesticides applied spatially located to the nearest one square mile section (260 ha), and include the crop type of application, listing about 207 different crop types. We converted the list of crop types in the PUR database to the lookup table used with the DWR maps and summed up the crop types by area for each square mile section, the rule being to assign each section the crop with the greatest total by area. The table was referenced spatially to a public land survey system layer for the state. The township-range-section map was then rasterized with the values for each pixel being the crop code for the majority crop type by area according to the PUR data within each section.

The final input dataset to the CAML 2010 map was the Farmland Mapping and Monitoring Program (FMMP) map data produced by the California Department of Conservation (California Department of Conservation 2011). These identify a number of different categories of lands, such as prime farmlands, locally important farmlands, grazing lands and so on for most counties in the state. FMMP has been mapping these in two-year intervals since 1984. Most importantly, FMMP has mapped conversion of farmlands to urban lands. We use the FMMP layer from 2008 as a source for urban boundaries. Like with the DWR vector dataset, we added all of the FMMP maps to a single table in PostGIS and then exported that to a shapefile, which was subsequently rasterized at 50 m resolution. The different FMMP categories are listed in Table 12).

Adminsitrative Record
Page 34366

Table 12. Farmland Mapping and Monitoring Program categories.

List of Categories in Farmland Mapping and Monitoring Program
Confined Animal Agriculture (CI)
Urban and Built-up Land (D)
Grazing Land (G)
Farmland of Local Importance (L)
Farmland of Local Potential (LP)
Natural Vegetation (nv)
Prime Farmland (P)
Rural Residential Land (R)
Farmland of Statewide Importance (S)
Semi-Agricultural and Rural Commercial Land (sAC)
Unique Farmland (U)
Vacant or Disturbed Land (V)
Water (W)
Other Land (X)
Unmapped Area (Z)

These four inputs to CAML 2010 were then combined (as illustrated in Figure 26), with Figure 27 showing the end map product. The urban regions are a combination of the urban areas from the MSLC and FMMP maps. The agricultural areas took values from the DWR layer where that was present. If no DWR layer was present, but the area was coded as agricultural in MSLC or FMMP, we took the values from the nearest PUR square-mile section, using a raster-based region growing algorithm to determine the crop type of the nearest section. If the region was neither urban nor agricultural, we assumed it was natural vegetation, and assigned values taken from the MSLC layer.

For the purposes of nitrate accounting, it is also necessary to keep track of double-cropping. The DWR land use maps provide information on multicropping in a field in the layer's attribute table. The *class2* and *subclass2* fields in this table give the second crop type if the polygon is double-cropped. We created a separate raster layer from this information, which presented the crop type if pixel was double-cropped. For the GNLM simulations, it was assumed that all areas classified as “corn” are double-cropped with winter grain, for the 1990s and the 2005 periods. For Monterey County, a different strategy was needed, since the DWR maps do not provide double-cropping information. This was handled by computing multiplicative factors based on the ratio of harvested acreage to land acreage in the NASS Agricultural Census for the annuals that are double cropped (see Section 2).

Adminsitrative Record
Page 34367

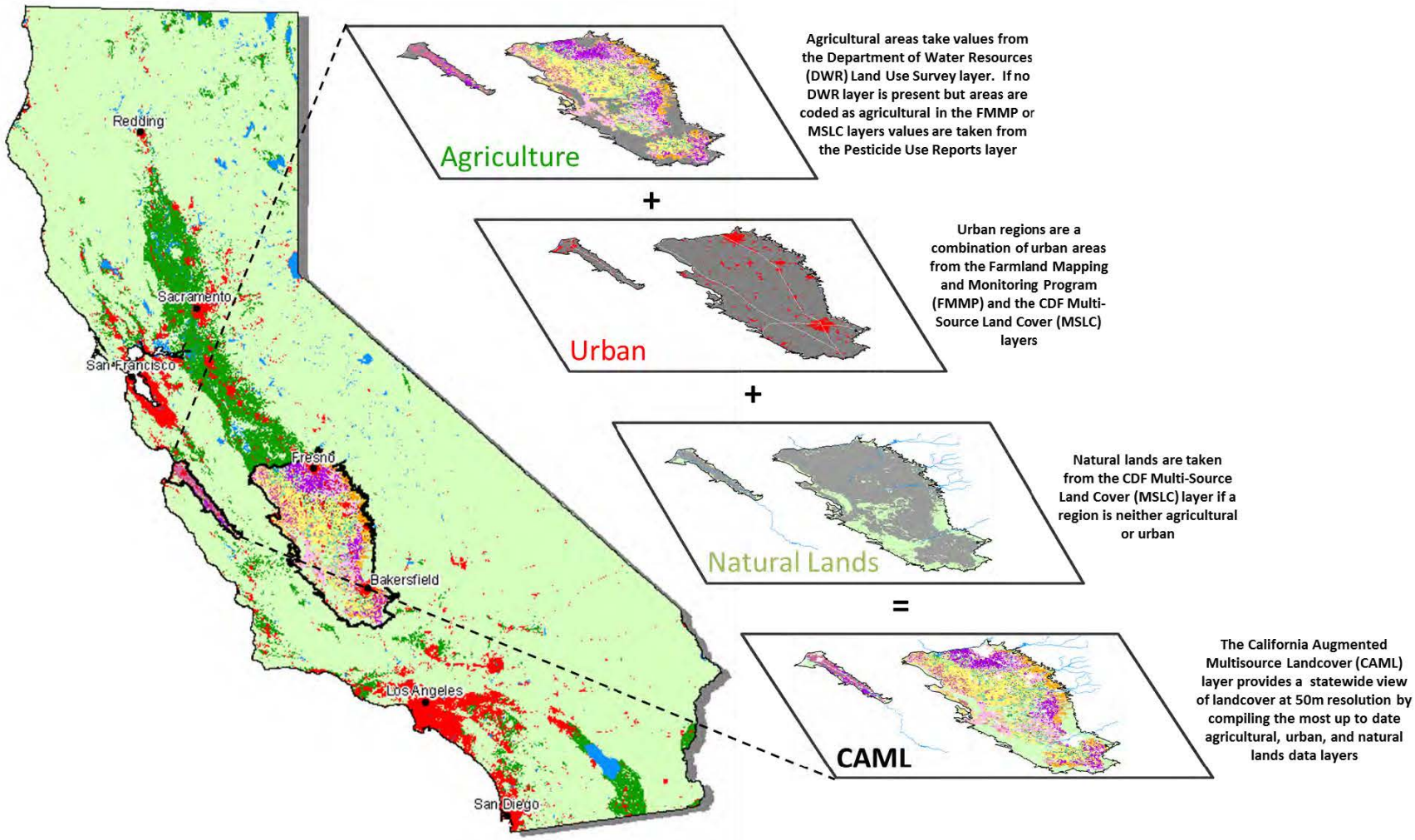


Figure 27. Input layers for the final CAML 2010 raster layer.

Adminsitrative Record
Page 34368

3.2.1.2 Historical Land Cover

We developed land cover maps for four broad time periods, designated as 1945, 1960, 1975, and 1990. Each of the time periods is centered on the corresponding year, but spans five years. In other words, each time period includes the year designated in addition to two years prior and two years post. In some cases, analysis was conducted on a single year (e.g., the median year), but is labeled as the corresponding time period. These time periods span three eras in mapping land cover. Source maps in 1945 and 1960 were all created on paper, whereas the 1975 era saw the first digital mapping products for land cover. By the 1990 period, detailed digital land cover maps were being created by a number of entities. These three eras called for different procedures in developing the GIS land cover maps.

1945 Land Cover Map

The 1945 time period corresponds roughly to beginning of the widespread application of synthetic fertilizers, and hence marks a significant point in the history of nitrogen use. For this time period, there are no map sources that provide crop type information at a field-by-field scale, so our aim in the mapping is to separate natural vegetation from agriculture and from urban areas. Differentiation of crops is handled using a simulation approach described below based on a statistical analysis of the cropping data collected from the county agricultural commissioner's reports (see below).

For the Tulare Lake Basin region, the initial map comes from the Central Valley Historic Vegetation Project from California State University, Chico (Geographical Information Center 2003). Using a wide variety of sources, this project developed a set of historic natural vegetation maps for the Central Valley for four periods: pre-1900, 1945, 1960, and 1995. Since the objective of the mapping effort was natural vegetation loss, particularly of riparian and wetland vegetation types, the CSU Chico maps do not distinguish urban areas from agricultural areas, necessitating additional data sources to differentiate those two land cover types. For this, we used 7.5 minute United States Geological Survey (USGS) topographic maps, taking advantage of the long revision cycle of these products. For instance in the pilot region of southwestern Tulare County, many of the quadrangles were last photo-revised in 1969, using a base that was originally published in 1951 from aerial photography taken in 1946. This 1946 date corresponds well to the 1945 time period of interest, and details from the 1951 base are often preserved in the current digital raster versions of the maps that are readily available online (e.g., http://www.atlas.ca.gov/quads/). To simplify digitizing urban boundaries, we started with the urban boundaries in the 1970s era digital USGS Land Use Land Cover (LULC) map (U.S. Geological Survey 1986) Visually overlaying these boundaries on georegistered images of USGS 7.5 minute quadrangle maps from the 1950 era, we then edited these boundaries to match the smaller urban extents in the 1950 era maps.

For the Salinas Valley region, two sources were used to distinguish between urban areas, natural areas, and agricultural areas. First, the USGS 7.5 minute quadrangles were used as in TLB to identify historic urban boundaries in locations where the time period of a revision of the map lined up well with the 1945 time period of interest. Second, the Wieslander Vegetation Type Maps (Kelly, Allen-Diaz, & Kobzina 2005) provide maps of vegetation cover in the 1930s for many parts of California, including the

Adminsitrative Record
Page 34369

Salinas Valley. Georegistered scans of these maps are available for the Salinas Valley, and many of the vegetation polygons on these maps have been digitized. Though the Wieslander mapping project emphasized natural community types, the maps do indicate crops and urban areas. We used a combination of the USGS 7.5 minute quadrangles and the Wieslander maps to adjust the boundaries of the USGS LULC digital map to reflect urban, agricultural, and natural vegetation conditions in 1945. This adjustment was performed in a similar manner to the procedure in the Tulare Lake Basin: the urban boundaries in the LULC map were used as a base for vector editing, with the vector boundaries being moved to correspond to the smaller urban boundaries in the Wieslander maps.

For both SV and TLB, crop statistics were derived from annual reports published by each county's agricultural commissioner's office (Fresno County Department of Agriculture, Kern County Department of Agriculture and Measurement Standards, Kings County Department of Agriculture Measurement Standards, Monterey County Office of the Agricultural Commissioner, Tulare County Department of Agriculture – Office of the Agricultural Commissioner/Sealer Agricultural Crop Reports, 1943 – 2007). These reports provide information pertaining to commodities produced in the county like crop type, harvested acreage, production and crop value. The purpose for using these data were twofold: 1) the production numbers for each commodity were used to help calculate the amount N removed the landscape during harvest (see section 2.3.2.3); and 2) the harvested acreage numbers were used in the backcasting model (see below) to help spatially reconstruct historic cropping patterns and land use in the study area.

In order to gain a more complete understanding of what the typical agricultural land use was within each county for each of the time periods represented by the specific target years, two years both preceding and succeeding the target year were included in the analysis. For example, the average agricultural land use for target year 1945 also includes crop data from years 1943, 1944, 1946 and 1947.[18] Where available, crop report data for each county within the target years were downloaded from each county's Agricultural Commissioner's webpage, where available. For counties whose ACR data were not available online, paper copies were obtained through Shields Library at UC Davis, and electronically scanned and saved in Adobe portable document format (.pdf). We created an MS Excel data form to compile crop data in a standardized format and included the following categories:

- year
- crop name
- DWR land use code
- NASS commodity code
- total ground acreage
- total harvested acreage
- total non-harvested acreage
- production unit
- production per acre

[18]Exceptions include: Kings and Monterey Counties (1942 was used in lieu of 1944 for Monterey County since no data were available for 1944 for either county. Kings also lacks 1943 data.).

Adminsitrative Record
Page 34370

The above data (except DWR land use code and NASS commodity code) were entered directly from the crop reports using both manual and electronic (Optical Character Recognition) methods. A visual comparison between the crop report spreadsheet and the .pdf version was performed at this time and any identified errors were corrected.

Once standardized, each of the spreadsheets was aggregated into one multi-year spreadsheet representing each county. The data were sorted by agricultural crop, and commodities were first combined by assigning the appropriate commodity code used by the National Agricultural Statistics Service (NASS). We further narrowed the number of commodities by matching each NASS commodity code to a DWR land cover code (via a lookup table). These DWR codes were derived from the value of the *class1* and *subclass1* columns in the DWR database. For each county for each year, the acreage and production were calculated for each DWR land cover representing a crop. These cropland DWR land cover codes are referred to as DWR crops from here onwards.

We assigned crops to individual field locations for N loading estimates. We used the following algorithm to simulate those crop locations. The algorithm worked as follows: from the earliest period for which we have a digital map of crop locations (1990) we compared the total area for each crop in that year and in our historic target period (for example 1945). There are two resultant possibilities, where either the area in the historic period is less than or equal to the area in the 1990 period, or it is greater than the area in 1990. In the first step of the algorithm we considered all crops where the historic area is less than or equal to the 1990 period. Proceeding crop-by-crop, we deallocated crop pixels so as to reduce the 1990 total area to the reported total for the earlier year. Within each crop, we chose pixels for de-allocation based upon the distance from centers of distribution of each crop considered. This distance was calculated by running a circular kernel summary filter over a binary presence-absence map of the particular crop, a procedure that results in the highest values at the center of distribution, with the sums diminishing as the distance increases from the center. Crop pixels are then deallocated in descending order by distance, so as to reduce the area of the crop to the area in the historic period. A small random value was added to each pixel in the distance map to allow for tie-breaking in the distance determination if needed. The rationale for this approach, rather than simply adjusting area by randomly de-allocating pixels, was that locations in which neighbors grow the same crop probably attract further increases in area, due to some combination of attractive growing conditions, access to water, processing, or transport, or perhaps simply social facilitation through experience and personal influences.

In the second step of the algorithm, we considered the crops where the area is greater in the historic period than the 1990 period. Proceeding in crop-by-crop order from most to least area in the historic period, we reallocated “deallocated” pixels so as to unify the area total for the historic period for that crop. This reallocation proceeds outwards in distance from pixels of each crop in 1990. That is, pixels adjacent to the 1990 fields were allocated first, then the next closest pixels are allocated, and so on, until the allocated acreage matches the historic acreage. This method of reallocating pixels was intended to preserve spatial patterning of crop types, and should be more realistic than random reallocation. This step of the algorithm was processed on a pixel basis rather than using the field boundaries provided by the DWR land use maps. Although a per-field basis crop allocation might better

Adminsitrative Record
Page 34371

reflect actual crop patterning for the simulated time period as compared to the employed per-pixel basis, doing so would have complicated the algorithm enormously. We developed and executed the algorithm as a Python module for the GIS GRASS (GRASS Development Team 2010) in a 100 m pixel resolution processing environment.

Two special cases were accounted for in the algorithm to more finely tune the resulting spatial pattern for a couple of crop types. In the first case, we disallowed reallocation of cotton east of Highways 99, 198, and 65. In the second case, de-allocation of citrus crops were biased to proceed west-to-east, since citrus was first planted at the line demarcating the eastern foothills, and spread east and west from there. These rules were adopted out of a concern that the allocation algorithm might result in unrealistic geographic distributions for these two crop types.

In all three time periods (1945, 1960, and 1975), the backcasting algorithm was executed directly from the 1990 digital land cover map to the seeded reference period. In other words, the 1975 backcasted map was not taken as the starting point for the 1960 backcasting, nor was the 1960 map taken as a base for the 1945 backcasting. This eliminated a possible source of correlated error that might occur if the maps were constructed sequentially.

1960 Land Cover Map

Developing this land cover map was similar to 1945, the one difference being that the California Department of Water Resources (DWR) had started mapping land use patterns, especially crop locations, as part of their program to evaluate water allocations (California Department of Water Resources 2011). This mapping was performed on paper, usually using 7.5 minute topographic quadrangles as a base. The early DWR land use maps have not been digitized, which makes them difficult to use in a GIS workflow without extensive preparation. From the DWR San Joaquin District we obtained scans of these land use maps for 1958 and 1968. These maps were not georegistered, and were instead used for reference on the side rather than in a GIS overlay. For the Tulare Lake Basin counties, the 1960 Central Valley vegetation map from Chico State was used to separate urban & agriculture regions from natural vegetation. As in 1945, we took the 1970s era USGS LULC base and clipped back urban polygons to provide urban boundary extent from the 1960 period. We referred to the DWR scans to identify urban boundaries from 1958, and referenced these boundaries to georegistered topographic quadrangles and the local road network. For the Salinas Valley we again started with 1970s LULC mapping, and referred to the USGS topographic quadrangle that was nearest to the 1960 time period (e.g.,1955 for the Soledad quadrangle) for reworking urban boundaries to the older zones. Crop placement was handled using the simulation technique discussed above for the 1945 period.

1975 Land Cover Map

By this time period the USGS LULC mapping was available. These products were mapped in the period from 1970 to 1985 from aerial photography at 1:250,000 scale and are classified to the second level of the Anderson land cover classification system (Anderson et al.1976), which retains greater detail than was needed to distinguish urban from agricultural from natural land cover. We obtained the portions of

Adminsitrative Record
Page 34372

the LULC map that covered the five counties of the study area. To evaluate the suitability of the LULC map for our use, we compared it with Landsat satellite imagery, specifically imagery from 1975 (Landsat 1 and 2) for the Salinas Valley, and imagery from Landsat 5 in 1984 for the Tulare Lake Basin. This was a qualitative check against an independent source of historical imagery to ensure that the boundaries of the LULC map corresponded reasonably well to actual land cover as visually interpreted. This check revealed no problems with the LULC map in terms of its classification of urban, agricultural, and natural land cover. We assigned crop types using the simulation technique described above for the 1945 land cover map.

1990 Land Cover Map

Beginning with this time period, digital versions of the DWR land use maps were available and we used these maps directly to assign crop cover and urban land use. We constructed a 1990-era land use map for the five counties of interest using the techniques described in more detail above for construction of the current land use map. The DWR maps we used were a 1989/1991 map digitized by the Monterey County Water Resources Agency for the Salinas Valley, a 1985 map of Tulare County digitized by Minghua Zhang at UCDavis, and the 1990 map for Kern County, the 1994 map for Fresno County, and the 1991 map for Kings County, all available from DWR.[19] These maps were merged with the California Department of Forestry and Fire Protection Multi-Source Land Cover Data layer (California Department of Forestry and Fire Protection 2002) for natural vegetation.

3.2.1.3 Future Land Use

We used the urban growth models developed for the San Joaquin Valley Blueprint Planning Process to project the extent of urban cover in the year 2050, referred to herein as UPlan(Johnston et al.2008). We used two UPlan scenarios, "Business as Usual" and "Smart Growth", to determine the total acreages by crop and other land cover types from CAML displaced by urban expansion by 2050. These data are intended to help determine the amount of N-loading removed from the landscape based on land use type. The "business as usual" (BAU) scenario predicts growth based on current growth patterns in California, with more people living in lower-density residential classes. The "smart growth" scenario predicts more compact growth with more people living in high-density living space concentrated around existing towns and cities (Bjorkmanet al. 2010).

UPlan is a GIS application developed by UC Davis and the California Department of Transportation. Developed in ArcGIS, UPlan projects future land use patterns in a spatial or mapped context, enabling users to utilize data outputs for environmental analyses. The UPlan runs used here were run at a raster resolution of 50 m. General assumptions of UPlan include: (1) population growth can be converted into demand for land use by applying conversion factors to employment households; (2) new urban expansion will conform to city and county general plans; (3) cell locations attract development at different rates, reflecting accessibility to transportation and infrastructure; and (4) some cell locations (e.g., lakes and streams) will not be developed, while other cell locations (e.g., sensitive habitats and floodplains) discourage development(Johnston et al. 2008).

[19] http://www.water.ca.gov/landwateruse/lusrvymain.cfm

Adminsitrative Record
Page 34373

Each UPlan model scenario was parameterized by the net predicted population growth for an area. Based on demographic and land use characterization inputs, the model determined the amount of land required for future housing, industry, and commerce. An attraction variable was created based on the assumption that development occurs near existing transportation infrastructure and urban areas (e.g., spatial layer attractors include highways, major and minor roads, city boundaries, ramps, and blocks with growth). Similarly, a discouragement variable was created from species of concern location found in the California Natural Diversity Database, in addition to the presence of floodplains, vernal pools, wetlands, protected areas, and existing urban areas to serve as detractors. A final suitability gradient was created by overlaying the attraction and detraction raster surface grids. Land use types were then allocated to areas in the suitability grid. The model allocated a certain land use type to a cell based on the cell value; the highest valued cells are filled first followed by incrementally lower valued cells until all the predicted acreage for a certain land use type was allocated. Developed land use types, often defined by local zoning categories, are broken into three major types: Industrial, commercial, and residential. UPlan allocated appropriate cells first to industrial uses, as they tend to be the most valuable and have the longest planning horizons. Remaining cells were then allocated to commercial uses, with residential taking up the most attractive remaining space. Similarly, there are several density categories within each type (Table 13), and, within each, the higher density uses were allocated first (Bjorkman et al. 2010).

Adminsitrative Record
Page 34374

Table 13. UPlan Land Use Descriptions (adapted from UPlan Model Output Guide, Information Center for the Environment).

UPlan Land Use	Abbr.	Density	Description	Scenario
Industrial	Ind	613 sq. ft./employee .23 FAR (Floor-Area Ratio)		Base Case Smart Growth
High-density Commercial	CH	498 sq. ft./employee .35 FAR		Base Case Smart Growth
Low-density Commercial	CL	856 sq. ft./employee .15 FAR		Base Case Smart Growth
Residential 50	R50	50 units/acre High Density	50+ unit apartments	Smart Growth
Residential 20	R20	20 units/acre Medium Density	2 unit, 3-4 unit, 5-9 unit, 10-19 unit and 20-49 unit apartments	Base Case Smart Growth
Residential 10	R10	10 units/acre Medium Density	Category created for future scenario; 20% of the ratio for R5 from the Base Case Scenario was shifted to R10 for the Smart Growth Scenario	Smart Growth
Residential 5	R5	5 units/acre Medium Density	Single family detached homes, <1 acre; single family attached homes, <1 acre	Smart Growth
Residential 1	R1	1 unit/acre Low Density	Single family detached homes, 1-9.9 acres; single family attached homes, 1-9.9 acres; and mobile homes, 1-9.9 acres	Base Case Smart Growth
Residential .5	R.5	2 acre lots Low Density	Category created for future scenario; 20% of the ratio for R.1 from the Base Case Scenario was shifted to R.5 for the Smart Growth Scenario	Smart Growth
Residential .1	R.1	10 acre lots Low Density	Single family detached homes 10+ acres; single family attached homes, 10+ acres; Mobile homes, 10+ acres	Base Case Smart Growth

We obtained statewide UPlan coverage from the Information Center for the Environment (ICE-UC Davis)[20] for both scenarios and extracted the model coverage to our study areas (Figures 28 and 29). Human population numbers were run by ICE for the specific study areas considered here. Total acreage removed was calculated by overlaying each UPlan scenario raster over the CAML raster layer in ArcGIS and using raster addition and subtraction based on value codes corresponding to residential and land use types (Table 14 and Table 15). The resulting layers (Figures 30 and 31) contained the areas of predicted urban expansion while retaining current land use attributes, allowing us to see which land uses would be urbanized in the future.

[20] http://ice.ucdavis.edu

Adminsitrative Record
Page 34375

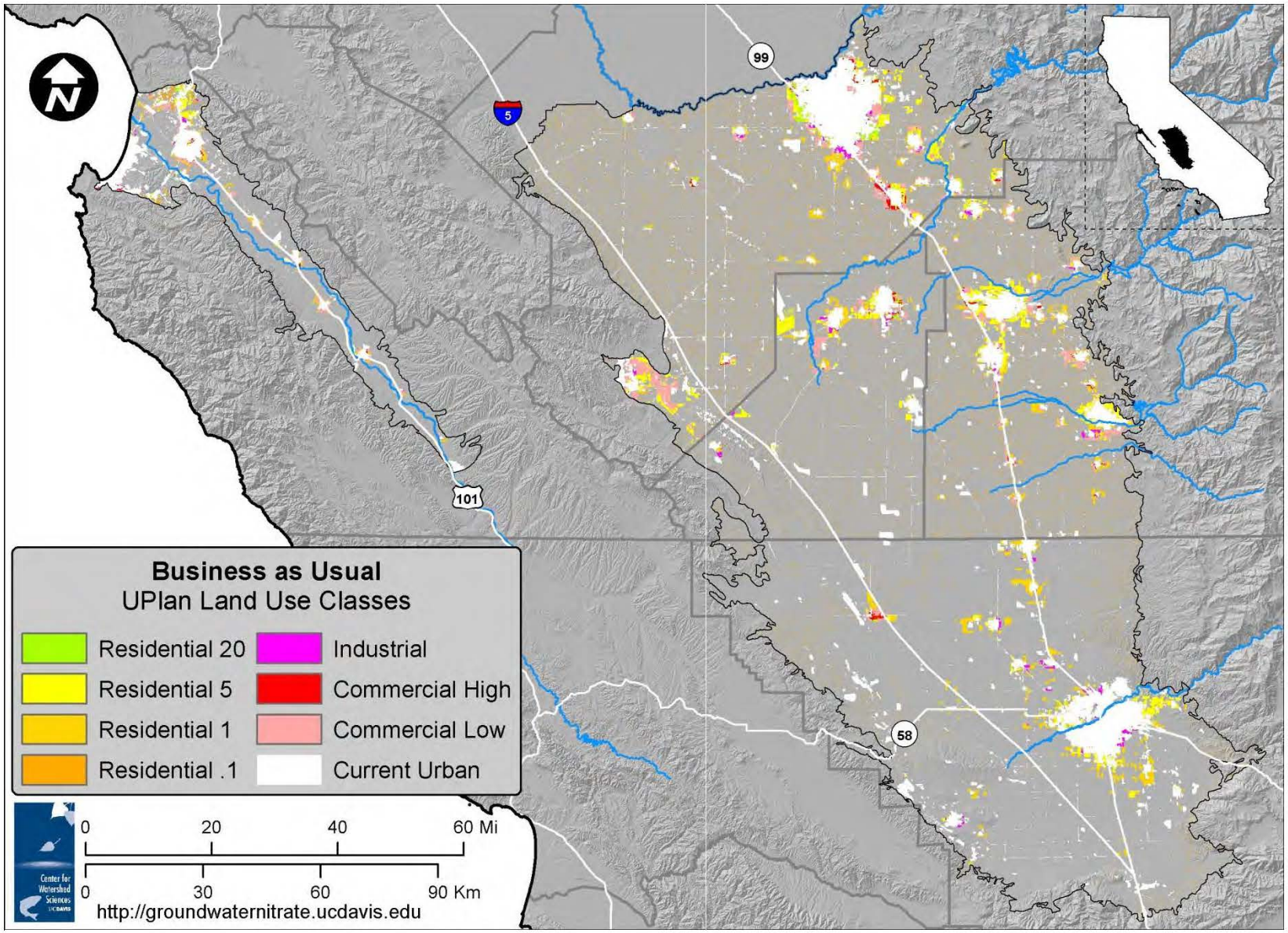


Figure 28. BAU modeling scenario (Information Center for the Environment)

Adminsitrative Record
Page 34376

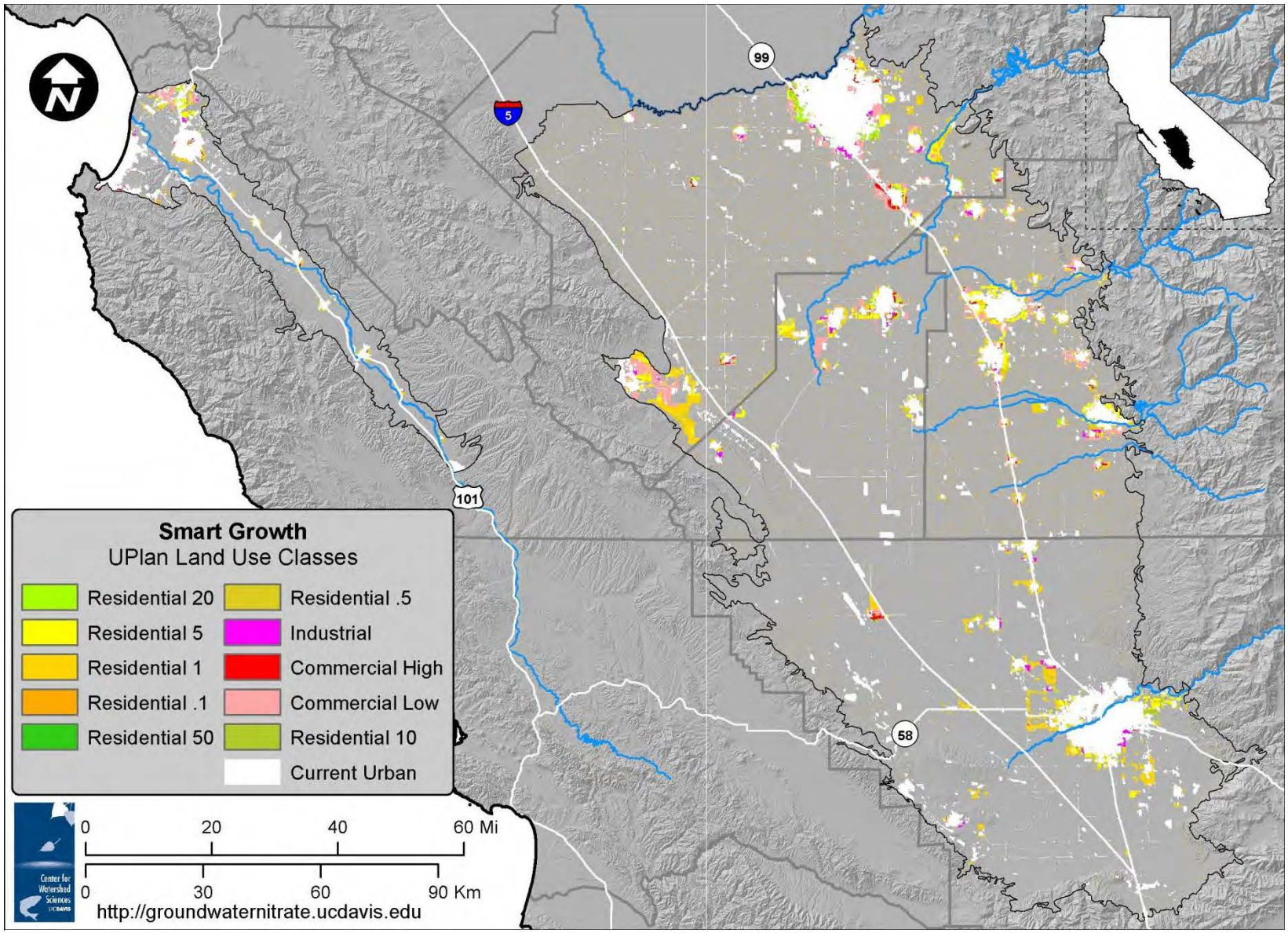


Figure 29. Smart Growth modeling scenario (Information Center for the Environment)

Adminsitrative Record
Page 34377

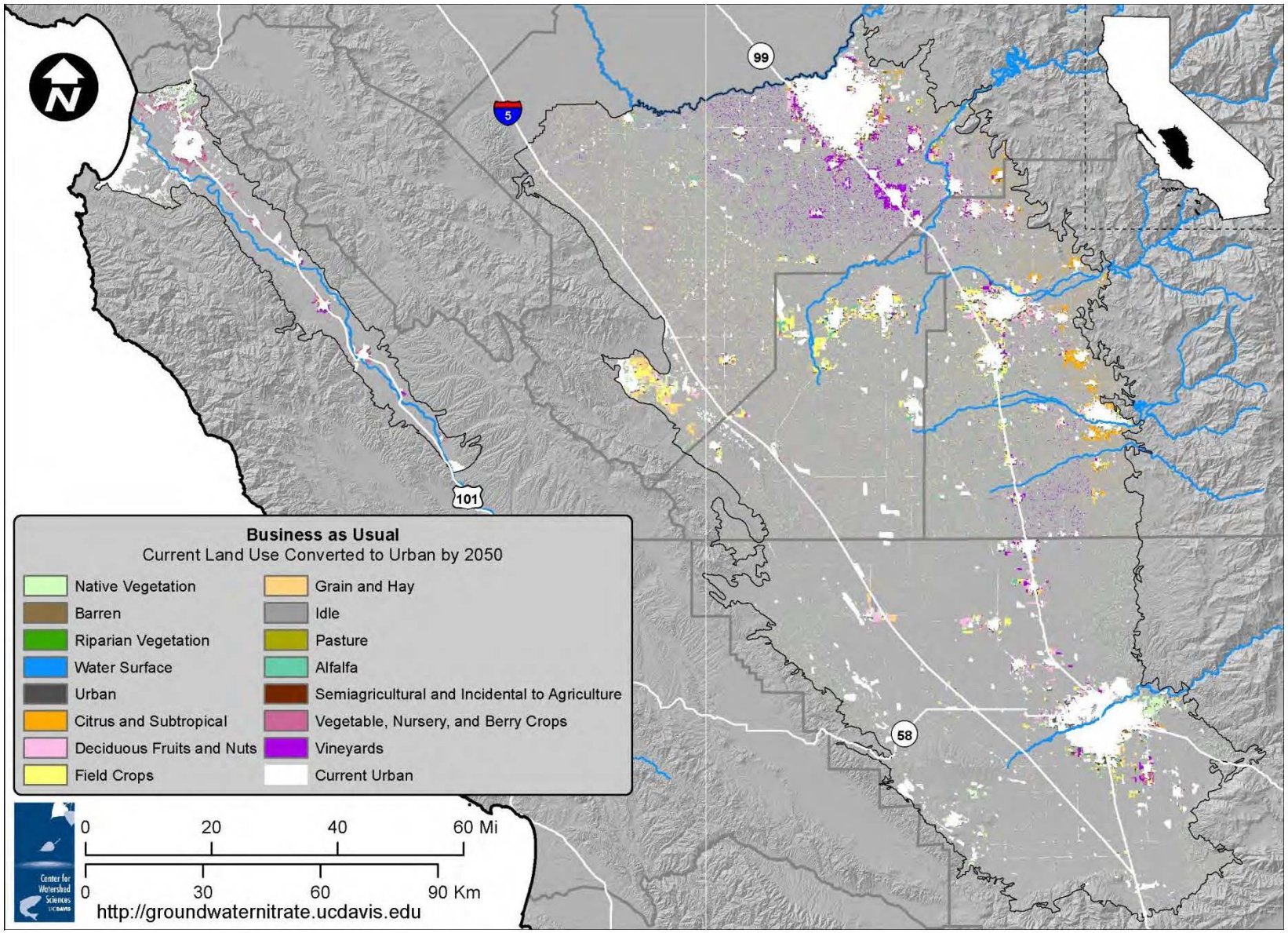


Figure 30. BAU – Agriculture removal modeling scenario (Information Center for the Environment)

Adminsitrative Record
Page 34378

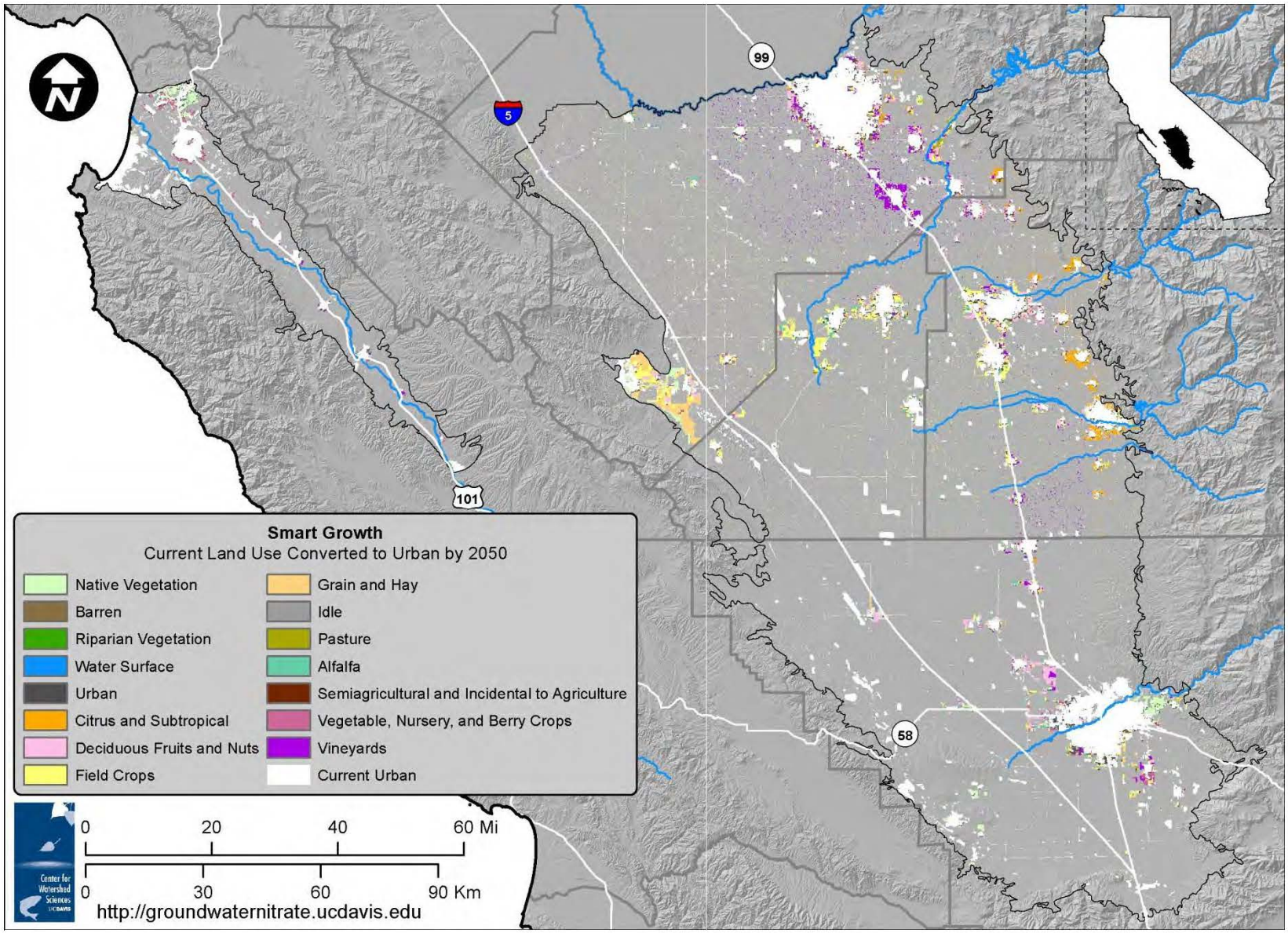


Figure 31. Smart Growth – Agriculture removal modeling scenario (Information Center for the Environment)

Adminsitrative Record
Page 34379

Table 14. UPLAN Business as Usual modeling scenario results showing change in land use over time.

BAU		
Land Use	**Hectares**	**Load Removed (Mg)**
Alfalfa	11630	0
Barren	127	0
Citrus and Subtropical	11997	621
Deciduous Fruits & Nuts	25511	1665
Field Crops	38734	3011
Grain and Hay	12818	927
Idle	1512	0
Native Vegetation	24885	0
Pasture	1905	17
Riparian Vegetation	261	0
Semiagricultural and Incidental to Agriculture	2879	0
Truck, Nursery, and Berry Crops	13759	1826
Urban	11621	0
Vineyards	21084	352
Water Surface	170	0
Total	**178892**	**8419**

Table 15. UPLAN Smart Growth modeling scenario results showing change in land use over time.

SMART		
Land Use	**Hectares**	**Load Removed (Mg)**
Alfalfa	8661	0
Barren	125	0
Citrus and Subtropical	9524	495
Deciduous Fruits & Nuts	21943	1436
Field Crops	30565	2480
Grain and Hay	12599	919
Idle	1223	0
Native Vegetation	20758	0
Pasture	1701	15
Riparian Vegetation	219	0
Semiagricultural and Incidental to Agriculture	2003	0
Truck, Nursery, and Berry Crops	9342	1235
Urban	11054	0
Vineyards	14362	240
Water Surface	137	0
Total	**144215**	**6819**

Adminsitrative Record
Page 34380

3.2.2 Estimating typical fertilizer use in crops

Few data are available to estimate current or historic N fertilizer rates. Fertilizer application rates are not widely reported, currently or historically (Rosenstock et al. In review). In order to develop a historical record of fertilizer application corresponding to the five time points (1945, 1960, 1975, 1990, and 2005), we compiled available data and extrapolated from known values based on trends of crop type. Rosenstock et al. (In Review) estimated fertilizer rates for the major food crops in California by taking the average of grower and expert surveys for 2005. A fertilizer application rate for the minor crops was calculated in a similar manner as part of the California Nitrogen Assessment (Liptzin & Dahlgren 2011). For all crops, the current fertilization rate was based on the average of the expert opinion in the UC Davis Cost Studies and the USDA chemical use surveys of growers in California. Estimated typical N application rates for each crop at each of the five time points of concern are compiled in Appendix Table 7 under the column "Napplied". The remainder of this section further describes how the "Napplied" values were determined.

For the cost studies, we compiled all available studies that reported N fertilization across all management regimes and regions of California from 2000-2009 for each CAML land cover type. We used the available USDA chemical use surveys available from 1999-2009. Depending on the crop, from 0 to 2 USDA surveys were averaged and from 0 to 5 cost studies were averaged. For each crop, we then averaged the two numbers obtained (one average value based on the USDA surveys and one average value based on the cost studies), giving each the same weight.

For CAML crop cover classes with multiple unique crops (e.g. peaches and nectarines), we calculated an area weighted average fertilization rate. For the lumped "miscellaneous" category of each CAML crop cover class, we assigned a fertilization rate of the most common crop within the DWR class (i.e., field crops = cotton, grain and hay = wheat, deciduous = prunes, subtropical = oranges, truck crops = lettuce).

Based on USDA surveys, it appeared there was little change in application rates between 1990 and 2005. Hence, 2005 N application rates were used to represent both time periods (1990 and 2005). This assumption is supported by the fact that N fertilizer sales have largely remained stable since approximately 1980.

For fertilization rates prior to 1990, the most comprehensive source of information was the Survey of Fertilizer Use 1973 (Rauschkolb & Mikkelsen 1978). The extensive statewide survey of UC staff asked more than 100 experts their opinions on fertilizer rates. The survey was completed in 1973 and published in 1978 and was chosen to represent the 1975 time point of our historic analysis.

The survey also reports average fertilizer use for 1960 and 1950 by major crop type (e.g., agronomic versus fruits and nuts). Estimates of fertilizer use in 1960 and 1945 were based on the relative changes reported in the Rauschkolb and Mikklesen (1978) (Table 16). The crops were scaled by the percent change between 1960 and 1975 values by crop type and likewise for the change between 1950 and 1960. The 1950 values were used in lieu of 1945 because they are the only known source of information. The relative rates of change for the major crop group that were used to scale historical N

Adminsitrative Record
Page 34381

application rates are reported in Rosenstock et al. (In Review). Of particular note, the estimated N application rate for fruits and nuts declined between 1945 and 1960.

Table 16. The percentage change of N application rates for the given time periods. Derived from Rauschkolb and Mikklesen (1978).

	Average N rate			% Δ		
Crop type	**1945**	**1960**	**1975[$]**	**1945 – 1960**	**1960 – 1975**	**1975 – 1990[%]**
Agronomic[^]	59	86	112	46	30	32
Fruits and nuts	131	110	123	-16	12	-12
Vegetables	101	154	198	52	29	38

[^] For 1960 and 1975, agronomic equals an area weighted average for forage and field crops.

[$] Equal to the reported 1973 data.

[%] Calculated as the average percentage from known 1975 values. We estimated N application rates in 1975 for crops not included in Rauchkolb and Mikklesen (1978) with this method, for 1 agronomic crop, 4 fruits and nuts, and 4 vegetables.

3.2.3 Estimating typical nitrogen removal in harvest

While agricultural production is quantified by several state and federal agencies on multiple spatial scales, the N in harvested products is not regularly reported. In order to calculate N yield, we combined crop production data with a database of crop N and moisture content. We used a four step process to convert the production data listed by commodity in the ACR data to harvested N by unique crop type. We then assigned each of these crop types to a specific crop cover type in the CAML map, which allowed us to calculate N yield by crop cover type.

First, we combined more than 200 crop commodities listed in the ACR data into 121 unique crops. The most common practice was to sum the production of crops reported with different end uses. For example, the four categories of broccoli (food service, fresh market, processing, and unspecified) were combined into one crop – broccoli. In some cases (peaches, lettuce, grapes), the production of different varieties of the same crop were summed, usually because a large fraction of the reported production was in the "unspecified" category. These production numbers represent the amount of material harvested from the land. Historically, in many cases, these numbers were reported in whichever unit the commodity was packaged in, i.e. bushels, sacks, or crates. Since the late 1950s, however, these numbers have been reported in pounds or tons. Using conversion rates provided by NASS where needed (Krug 2011), the total production (i.e. lugs, crates, cartons) reported for each commodity was converted to metric tons (1 metric ton = 1 megagram or Mg).

Secondly, we used the crop N and moisture contents from the USDA Crop Nutrient Tool[21] to convert harvested products to harvested N by crop. This database is by far the most comprehensive source of information on crop (not food) N and moisture content. However, most crops are represented by only a

[21] http://plants.usda.gov/npk/main

Adminsitrative Record
Page 34382

few publications. Further, half of all data sources used for the database were published prior to 1982. We matched the crops as closely as possible to the crops included in the database. In some cases we used nutrient contents for similar crops (e.g. tangerines for tangelos). In cases where we summed multiple varieties of the same crop (notably oranges and lettuce), we used the nutrient content for the most common reported variety (e.g. navel oranges for all oranges). As part of the California Nitrogen Assessment, the major commodity boards were invited to submit their own data on nutrient content. The only data from this additional source was for almonds from the Almond Board of California.

Finally, we further recombined the production data for the 121 crops into the 58 crop types associated with the CAML map described in section 2.3.2.1. For several important crops (such as corn, grapes, peaches, nectarines, melons, squash, and peppers), there were multiple unique crops with their own harvest rates, but only one CAML crop type. For example, the harvested N was calculated using separate production and nutrient contents for grain and silage corn, but the final production for the CAML crop type "grain and silage corn" in a specific county was the sum of harvested N of these two crops and the production came from the sum of the total harvested area of these two crops.

Many minor crops were lumped into a generic CAML crop type either because there was no unique CAML crop type or because there was no available fertilization rate for minor crops. For example, while grapefruit, lemons, oranges, dates, avocados, olives, and kiwis have a specific CAML crop subclass, the production of the commodities jojoba, tangerines & mandarins, limes, tangelos, kumquats, pomelo, citrus unspecified, pomegranates, quince, cherimoyas, guavas, feijoa, and prickly pears was summed into an area-weighted average as the miscellaneous subtropicals crop type.

The procedure provided a protocol for assembling the relevant data from the records available in the county ACRs. For each of the five counties, we canvassed the ACRs for five years centered around the period year of interest:

- 1943 – 1947 for the period year "1945"
- 1958 – 1962 for the period year "1960"
- 1973 – 1977 for the period year "1975"
- 1988 – 1992 for the period year "1990"
- 2003 – 2007 for the period year "2005"

By following the above protocol, a table was generated that shows, for each of the 25 years listed above, separately for each of the 58 CAML crop types, and separately for each of the five counties of interest the following two numbers: the total area harvested [ha] and the total amount of harvested N [kg/yr].

Due to difference in the amount harvested per acre between counties, and due to differences between counties with respect to the specific crops that were lumped into some of the 58 CAML crop types, the ratio of harvested N to harvested area varies both, between counties and between years within a specific period. For our purposes, however, we needed a single value of the harvested N rate (kg N/ha/yr) that was specific to crop type and period, but representative for all five counties and for all five years of a specific period. This was necessary because the typical N application rates were developed

Adminsitrative Record
Page 34383

only to the period and crop type, but not specific to county or year. No data are currently available to be that specific for fertilization rates.

This crop type and period specific harvested N rate was obtained by:

1. computing an area weighted average of the county-specific harvested N rates of each crop type, in each year, which is the sum of the total harvested N divided by the sum of the total harvested area in that year across all five counties;
2. then computing the period median harvested N rate of each crop type, in each period, from the five yearly values obtained in the previous step.

The period median was chosen over the period mean to avoid bias due to outliers among the five values from which each period harvested N rate was computed. The resulting harvested N rates ("Nharvest") [kg/ha/yr] for each crop type and each period are listed in Appendix Table 7 and provide an important parameter in the mass balance analyses described in Sections 1 and 2.6.

3.3 Landuse, Nitrogen Application, and Nitrogen Harvest: Results

3.3.1 Status and Trends in Land Use

3.3.1.1 Current Land Use

Figure 32 presents the CAML 2010 land cover map classified by major agricultural cover types and with the study area boundary superimposed upon it. This figure illustrates the main spatial patterns of agricultural cropping within the study area. In the Salinas Valley, vegetables and berries (truck crops) predominate with vineyards in the upper valley as well as along the slopes on the sides of the valley. In the Tulare Lake Basin, citrus orchards fall along the eastern edge of the basin. West from the citrus orchards, vineyards predominate in the eastern portion of Fresno County and the southeastern corner of Tulare County's valley floor. The western portions of Fresno and Tulare Counties as well as Kings County are dominated by a mix of field and vegetable crops, with deciduous orchards becoming an important part of the mix in Kern County.

Adminsitrative Record
Page 34384

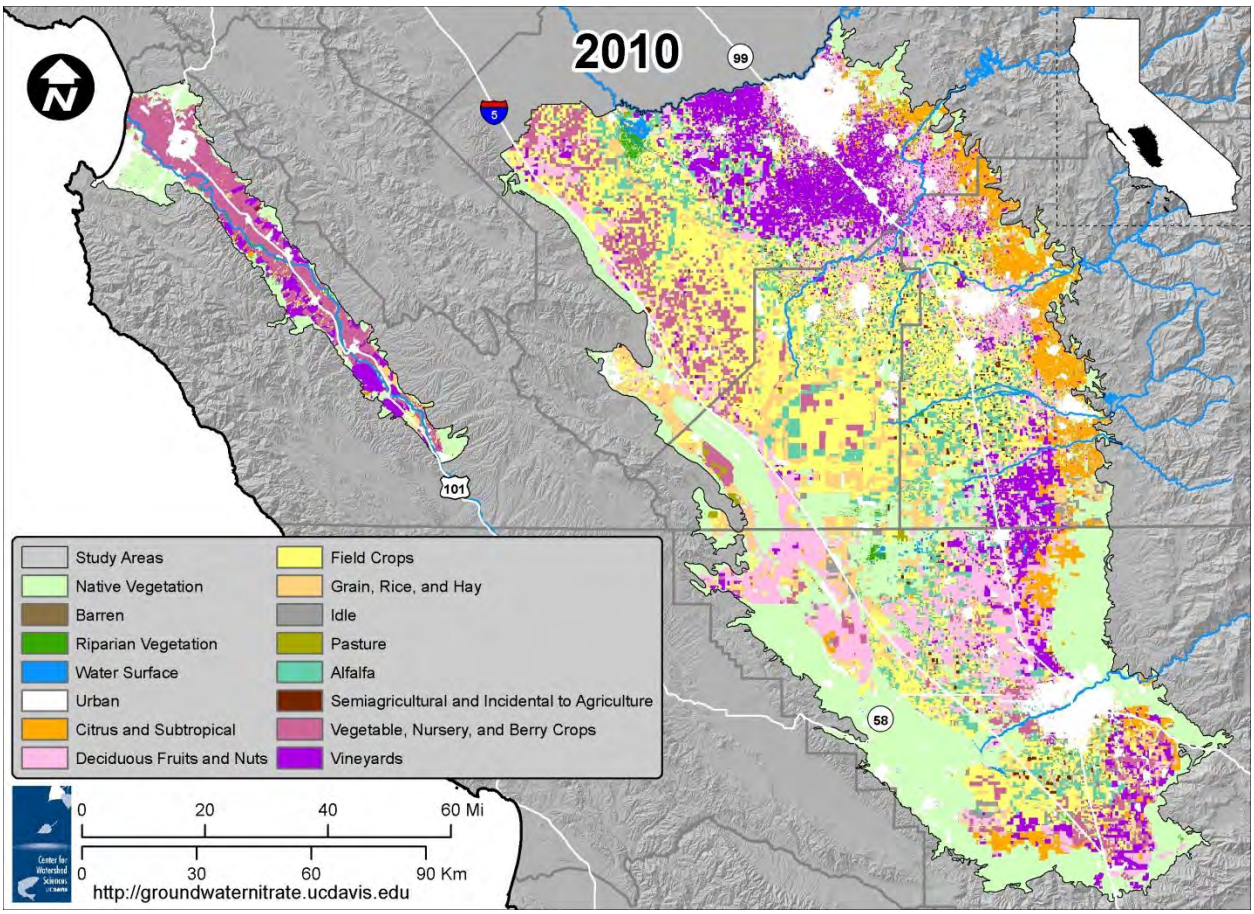


Figure 32. CAML 2010 Land Cover Map (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment)

Appendix Table 2 gives the area of land cover types in the CAML 2010 map within the study boundary apportioned by county. This table highlights specifics of the current cropping pattern. For instance, in Monterey County, lettuce is the predominant crop with 19,512 hectares (48,214 acres; 20.0% of total Monterey County crops), followed by vineyards at 19,234 hectares (47,527 acres; 19.7% of total Monterey County crops) and miscellaneous vegetable crops at 17,165 hectares (42,415 acres; 17.6% of total Monterey County crops). In Fresno County, grapes are the predominant crop at 105,801 hectares (261,434 acres; 21.7% of total Fresno County crops), followed next by cotton at 99,048 hectares (244,748 acres; 20.3% of total Fresno County crops). In Tulare County, oranges are the largest crop in area at 46,353 hectares (114,538 acres; 14.1% of total Tulare County crops), followed by alfalfa (which in the map is scattered throughout the western portion of the county) at 42,691 hectares (105,489 acres; 13.0% of total Tulare County crops). In Kings County cotton is the crop grown over the largest area at 71,412 hectares (176,459 acres; 28.1% of total Kings County crops). In Kern County, almonds are the crop with the largest area at 77,358 hectares (191,152 acres; 18.6% of total Kern County crops).

Adminsitrative Record
Page 34385

The compiled crop report data show that within our five county study area, the total number of hectares in agricultural production varied between 1943 and 2007. In some cases, counties experienced over a 100% increase in total hectares in agricultural production over this time period. Cropping patterns also emerged from the data allowing us to ascertain shifting agricultural trends within each county, and to track how they changed over time. When used in conjunction with the statistics showing the amount of N harvested off of the landscape as plant biomass, a clearer picture begins to emerge in terms of N not leaching into the ground but instead taken up by the plant. When we consider that all five of the counties within our study region are part of the top ten agricultural producing counties in California (Table 17), it becomes evident that being able to track how much N is removed with harvest is an important variable to consider for calculating potential nitrate leaching loss to groundwater.

Table 17. Top 10 Agricultural Commodities in California, 2004-2005. California Department of Food and Agriculture (CDFA) California Agricultural Resource Directory 2006

Rank	County	$ Value*	Main commodities
1	**Fresno**	**4,640,166**	**grapes, almonds, milk, tomatoes, cattle and calves**
2	**Tulare**	**4,360,854**	**milk, oranges, cattle and calves, grapes, alfalfa hay and silage**
3	**Kern**	**3,546,925**	**almonds and byproducts, grapes, milk, citrus, pistachios**
4	**Monterey**	**3,273,000**	**lettuce, strawberries, wine grapes, spinach, broccoli**
5	Merced	2,388,058	milk, chickens, almond meats, cattle and calves, sweet potatoes
6	Stanislaus	1,977,596	milk, almonds, cattle and calves, chickens, walnuts
7	San Joaquin	1,743,294	milk, grapes, almond meats, tomatoes, English walnuts
8	San Diego	1,531,307	foliage plants, woody ornamentals, avocados, bedding plants, cut flowers
9	**Kings**	**1,407,091**	**milk, cotton, cattle and calves, pistachios, alfalfa**
10	Imperial	1,286,066	cattle, alfalfa, leaf and head lettuce, carrots, livestock

*in thousands

County names in bold are within our study area

California Agricultural Resource Directory, 2006. CDFA

Adminsitrative Record
Page 34386

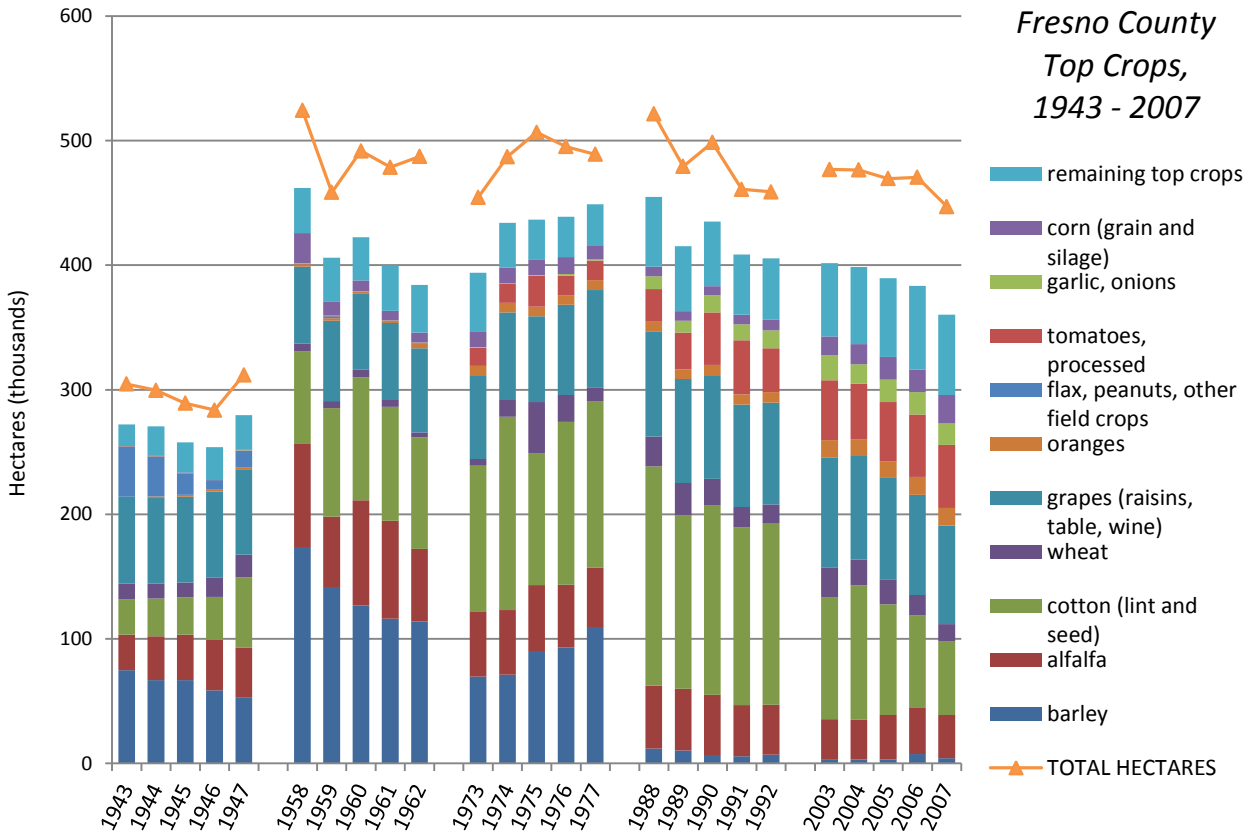


Figure 33. Fresno County top crops in hectares. Source: Fresno County Department of Agriculture Annual Crop and Livestock Reports

Fresno County, ranked number one in California and in the United States in terms of value of agricultural production (Table 17), experienced a 46% increase in agricultural hectares between 1943 and 2007 (Figure 33) The ACR for 2005 reveals that cotton and grapes are leading commodities in Fresno County, and make up 19% and 17%, respectively, of the hectares in production (in 2005) followed by tomatoes (including processing) (10%), alfalfa (7%) and wheat (5%) in the top 5 commodities. The California Department of Food and Agriculture (CDFA), however, lists grapes as the most valuable agricultural crop, followed by almonds, tomatoes, cotton and peaches in the top five Fresno County crops (CDFA, 2006).

Adminsitrative Record
Page 34387

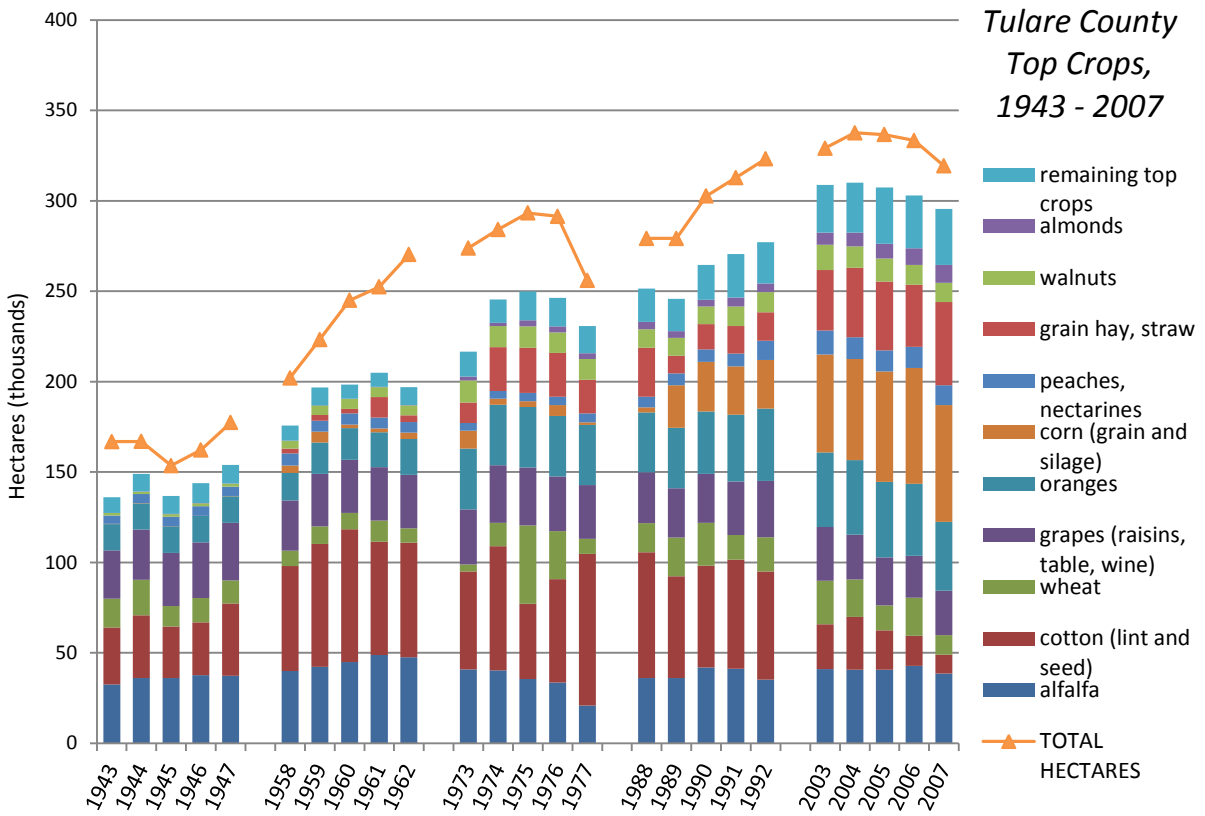


Figure 34. Tulare County top crops in hectares. Source: Tulare County Department of Agriculture and Office of the Agricultural Commissioner/Sealer Annual Crop and Livestock Reports

According to statistics compiled from the ACR data, Tulare County experienced a 91% increase in crop hectares between 1943 and 2007 (Figure 34) Although historically dominated by alfalfa, cotton, and grapes, contemporary Tulare County agricultural commodities have diversified to include corn (grain and silage), grain hay and straw, walnuts, oranges, and peaches. Tulare County was ranked the second largest agricultural producing county in the United States in 2005, with corn (for grain and silage) accounting for the largest number of crop hectares at 18%, oranges (12%), alfalfa (12%), grain hay (11%), and grapes (8%) comprising the top five crops in terms of hectares in production. The most valuable agricultural commodity by far in Tulare County is milk, worth almost $1.5 billion dollars in 2005. This makes sense considering that Tulare County, as the leading milk producing county in the U.S., has almost twice the number of cows as the second largest milk producing county, Merced County. Other sections of this report contain more information about the role of dairies in nitrate loading to groundwater (Section 2.4).

Adminsitrative Record
Page 34388

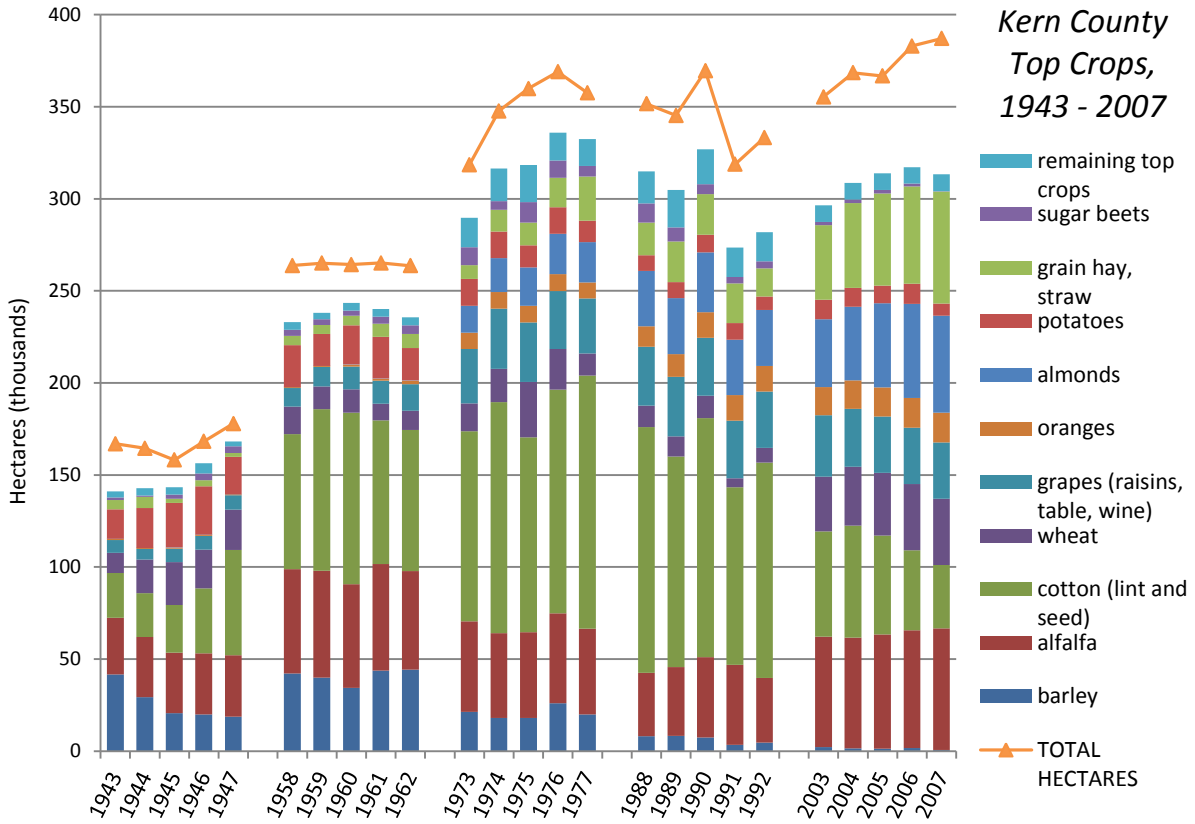


Figure 35. Kern County top crops in hectares. Source: Kern County Department of Agriculture and Measurement Standards Annual Crop Reports.

Kern County falls just below Tulare County in terms of one of the top ten agricultural producing counties, but experienced the highest increase of crop hectares in the Tulare Lake Basin (131%) between 1943 and 2007 (Figure 35). Alfalfa and cotton, 17% and 16% respectively, make up the largest percentage of crop hectares, followed by grain hay and straw (11%), almonds (10%), and grapes (9%). For Kern, the most valuable commodity in terms of production is also milk, with cotton, alfalfa, tomatoes, corn silage, and peaches as the most valuable crop commodities. Interestingly, pistachios are listed as one of the valuable crops for Kern County, which is not necessarily reflected in the crop acreages for 2005 (Figure 35).

Adminsitrative Record
Page 34389

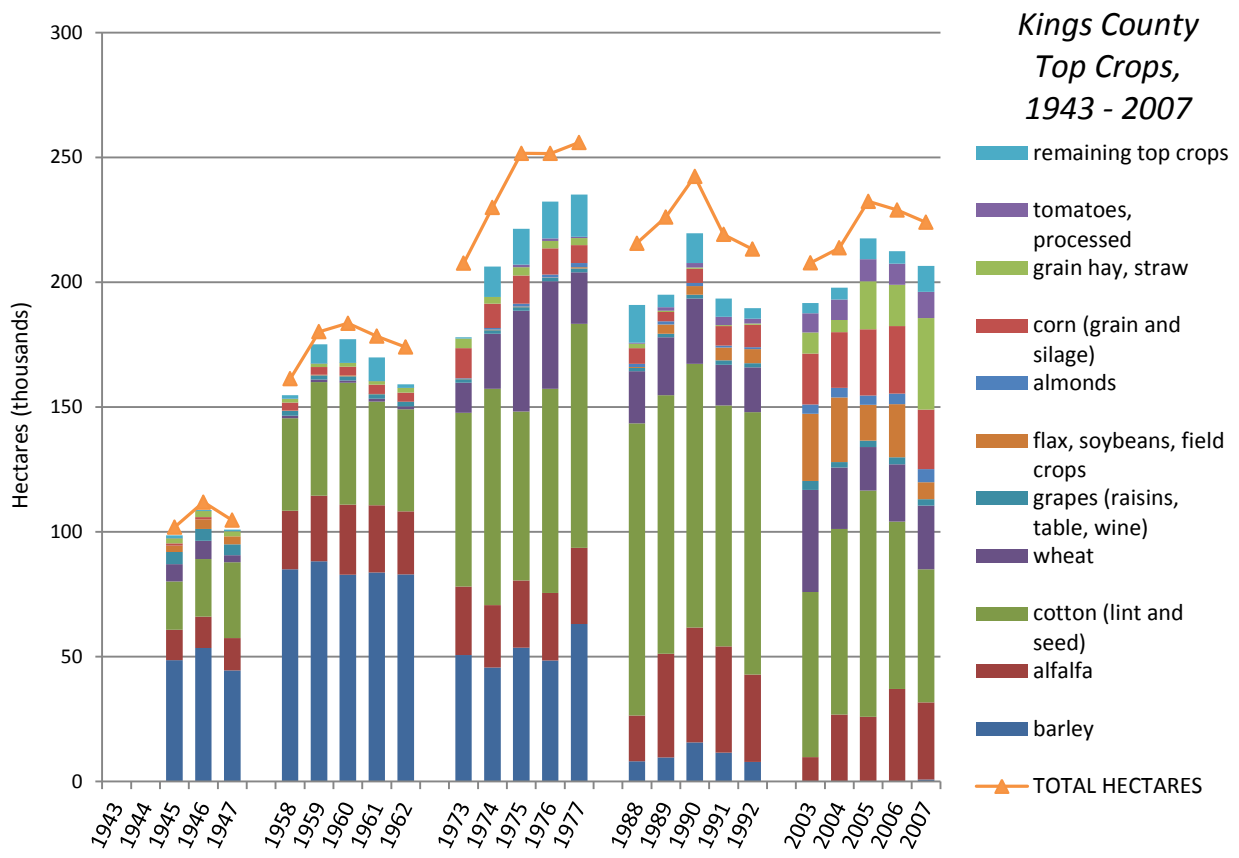


Figure 36. Kings County top crops in hectares. Source: Kings County Department of Agricultural Measurement Standards Annual Crop and Livestock Reports.

Kings County is ninth on the list of the top ten agricultural producing counties in the state, and has seen a 120% increase in crop hectares between 1943 and 2007 (Figure 36). Cotton has been, and continues to be, the largest crop in terms of crop hectares for Kings County, accounting for almost 40% of the crop hectares in 2005, followed by corn (grain and silage) (11%), alfalfa (11%), and grain hay and straw (8%). The most valuable commodities for Kings County, almonds and grapes, account for less than 3% of hectares in agricultural production. Kings County has been able to increase crop variability as well. Agricultural commissioner report data reflect that there are fewer hectares in cotton than there were in the two years preceding and succeeding 1990, but more variability in crop type.

Adminsitrative Record
Page 34390

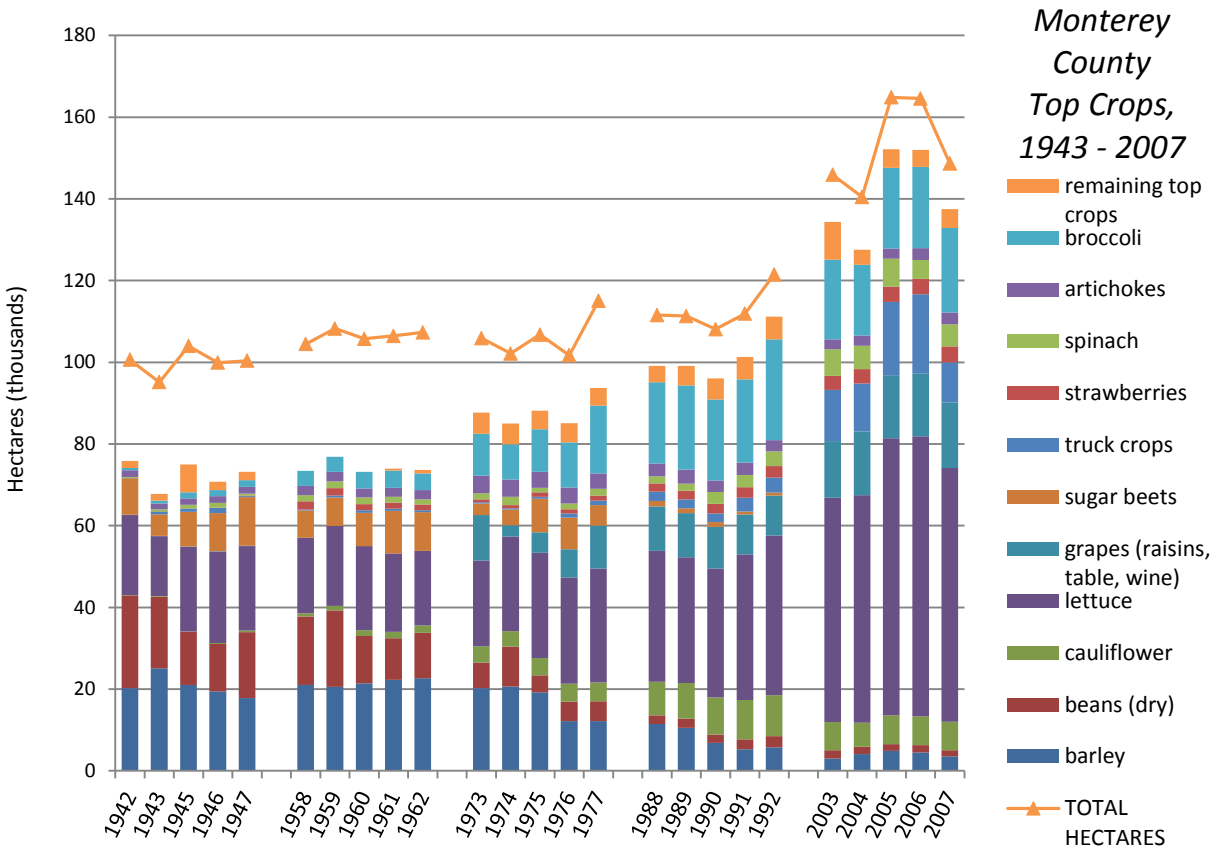


Figure 37. Monterey County top crops in hectares. Source: Monterey County Office of the Agricultural Commissioner Annual Crop Reports.

In the Salinas Valley, vegetable crops comprise more of the agricultural production than field crops. Ranking fourth on the list of the top ten agricultural producing counties in the state, Monterey County has experienced a 48% growth in crop hectares, some of which may be the effect of double cropping. With 41% of their crop hectares producing lettuce, Monterey County is the leading lettuce producer in California (and California grows three times as much lettuce as Arizona, the next largest producer in the United States). Broccoli and grapes comprise 12% and 9%, respectively, with other vegetable crops, like herbs, accounting for 8% of the crop hectares. Ranking second in terms of valuable crops for Monterey, strawberries represent only 2% of the crop hectares in the county (Figure 37)

3.3.1.2 Historical Land Cover

The spatial base for the backcasting of the crop patterns in the three earliest time periods is the 1990 land cover map, so it is useful to see how closely this land cover map, derived from DWR field surveys, matches the crop totals from the county crop reports. Appendix Table 3 presents a comparison of crop hectares for these two sources. Looking within pairs by county, some categories show good agreement between the areas in the land cover map and in the reports, yet some categories show substantial disagreement between DWR field surveys and county based crop reports. For instance, examples of crops that show good agreement overall on a percent basis are vineyards (code 2200) and cotton (code

Adminsitrative Record
Page 34391

1601), but even in those cases, the totals for individual counties can be off by tens of thousands of hectares (e.g., Fresno vineyards on the 1990 map having a total of 96,548 hectares (238,575 acres) versus a total of 83,223 hectares (205,648 acres) from the crop reports. By contrast miscellaneous grain and hay (code 700) shows 30,547 hectares (75,484 acres) in Fresno in the 1990 map, but only 9,267 hectares (22,900 acres) in the 1990 crop report. That particular case is perhaps explainable by the miscellaneous grain classification in the map lumping together totals for barley and wheat, categories which are broken apart in the crop reports, but if that logic is applied to the grain categories in Kern County, there is much more grain acreage in total in the crop reports if one sums in the acreages for barley and wheat. Another example of very poor agreement is pasture (code 1601), but as will be discussed below, this category is subject to a great deal of misclassification.

The choice of the three years to backcast spatial crop patterns was determined from the total overall cropped acres in the crop reports. Since two years of crop report data were compiled on either side of the target years 1945, 1960, and 1975, we could select data from any one of five years in each of the three time intervals. The selection was based upon which year had the median overall cropped acreage, summing across both Salinas Valley and Tulare Lake Basin counties in each five-year interval. The decision to choose the year with the median acreage was made in an attempt to use years that were most representative of land cover condition over each time interval, recognizing that the amount of land in production fluctuates with both weather and economic conditions. The years chosen were 1946, 1960, and 1977. In retrospect, 1977, and to a lesser extent 1960, mark drought years, and the fact that these years ended up as median years may reflect a falloff from a condition of peak production, which for the purposes of calculating nitrate loading to the environment can be considered conservative.

Another consideration in determining crop totals to use in the backcasting algorithm is double cropping, which is especially prominent in Monterey County. In Monterey County, the crop reports only indicate harvested acres rather than acres on the ground; in other words if lettuce is cropped twice in a year, the acres in the crop report would be twice the acres planted on the ground, the latter being what we want to map. We accounted for this by attempting to determine a “double cropping factor” for the six crops that are regularly double-cropped in the Salinas Valley: celery, lettuce, spinach, broccoli, cabbage, and Brussels sprouts. The factor was calculated by averaging the ratios of harvested acres to mapped acres for these six crops for the 1990 data. This factor was calculated as 2.19, which corresponds well to an expert opinion assessment of this factor of 2.2 (Timothy Hartz, pers.comm.). This level of double cropping is likely to be higher now than in earlier time periods, but no data were found to provide an estimate of the factor for the time periods to which the backcasting algorithm was applied. The double cropping factor was used to estimate acres on the ground for these six crops by dividing it into the values for the harvested acres.

When the backcasting algorithm was initially run, it was discovered in a couple of cases (Kern 1946 and Kern 1977) that there was more total crop acreage given in the reports than was available for allocation in the agricultural region of the base map. Since the base map was divided into three different land cover categories – agricultural land, natural vegetation, and urban land, the solution to this problem was to adjust the base map by changing natural vegetation to agricultural land. This conversion was done by expanding the agricultural land pixels outwards to remove natural vegetation. The number of pixels

Adminsitrative Record
Page 34392

changed from natural vegetation to agricultural lands corresponded to the difference in area between the total crop acreage and the agricultural lands in the original base map plus a factor of 5%. This factor of 5% was to account for idle agricultural lands, and was based on the fraction of idle lands versus cropped lands in Kern County in 1990.

Appendix Table 4 through Appendix Table 6 presents the patterns of agreement between the areas of crops in the backcasted maps and in the historical crop reports. This series of tables provides an internal validation check on the functioning of the algorithm. By design of the algorithm the county-by-county crop comparisons should be virtually identical, within several acres (or one one-hectare pixel) of each other. The crop comparisons show extremely close agreement on a percent basis, but nevertheless many have differences that are much greater than a single hectare. This is evidently due to slight problems in the coding of the algorithm that have not been identified to date. But since the overall agreement is extremely good – most crops showing errors of less than 0.1 % – the resulting maps should be suitable for use in subsequent modeling.

Figures 38–41 show historical land cover maps in chronological order. To summarize the overall pattern, all regions show more area in agriculture and less in natural vegetation forward in time. In the Salinas Valley, the overall crop mix shifts from a mix of grains and field crops (in particular, substantial acreages of barley, dry beans, and sugar beets, in 1946 and 1960, to domination by vegetable crops and vineyards, by 1990). The Tulare Lake Basin counties likewise show an increase in the amount of agricultural lands over time. Fresno County has shifted from large expanses of alfalfa and grain crops (primarily barley), the latter found especially at the eastern portion of the basin, in 1960 and 1977 to more field crops and vineyards by the current time period. In Tulare County, the citrus orchards increase in area from their extent in 1946 and 1960. In Kings County the crop mix becomes more diverse over in time, being less heavily dominated by cotton and barley. Kern County shows a large increase in the area of citrus orchards and vineyards going forward in time.

Adminsitrative Record
Page 34393

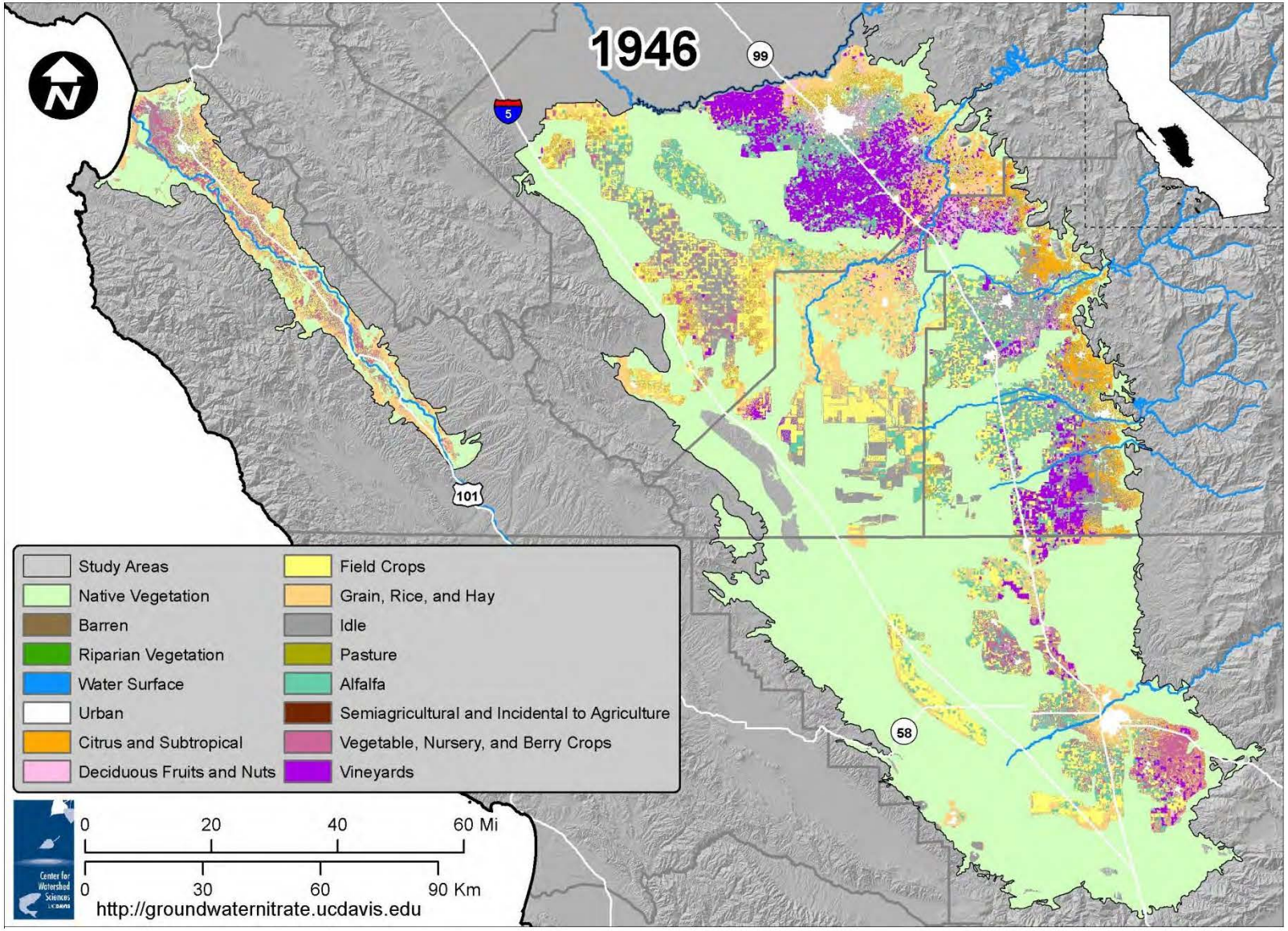


Figure 38. 1946 Land Use from DWR Mapping (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment).

Adminsitrative Record
Page 34394

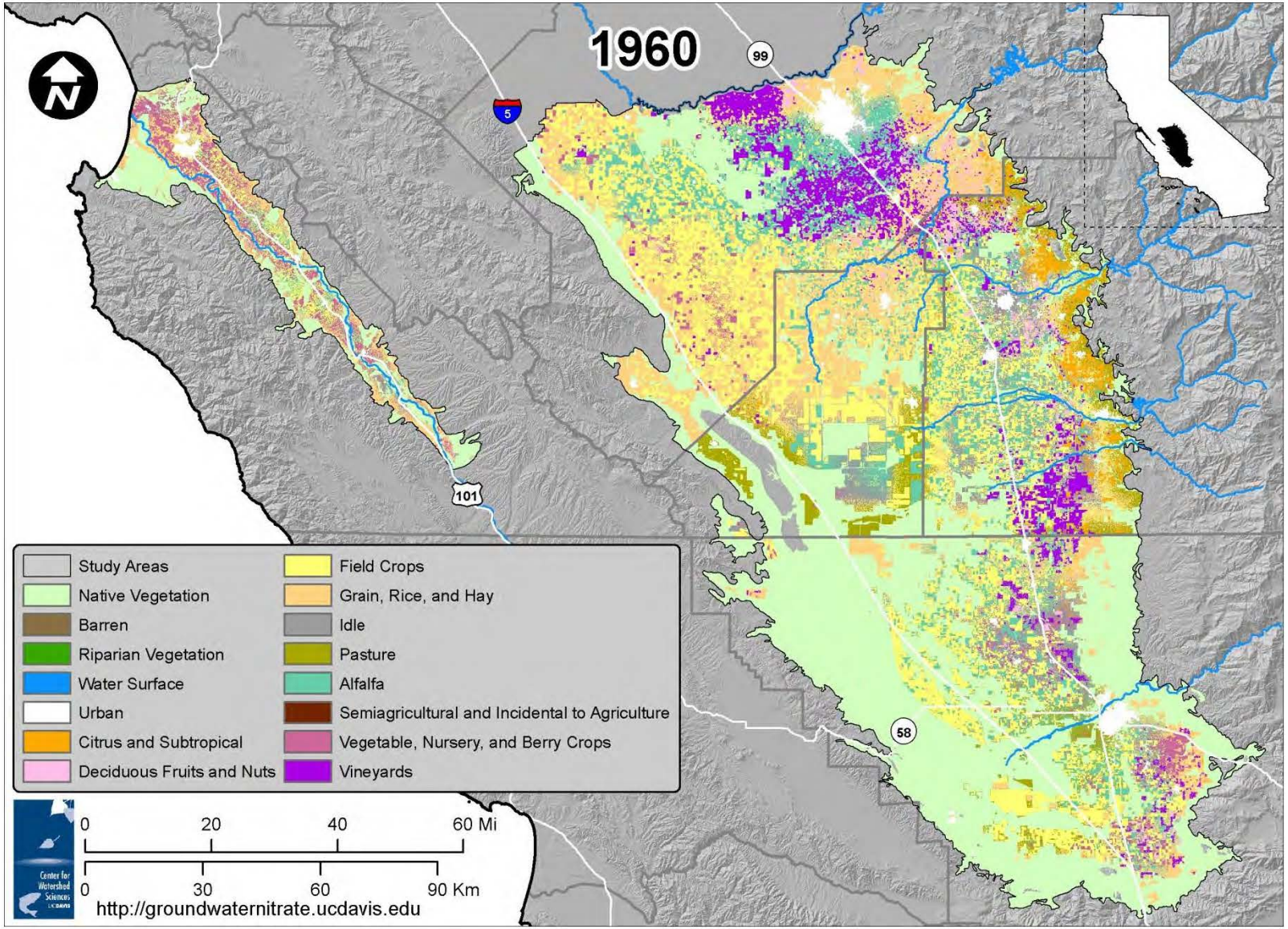


Figure 39. Backcasted 1960 Land Cover Map (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment)

Adminsitrative Record
Page 34395

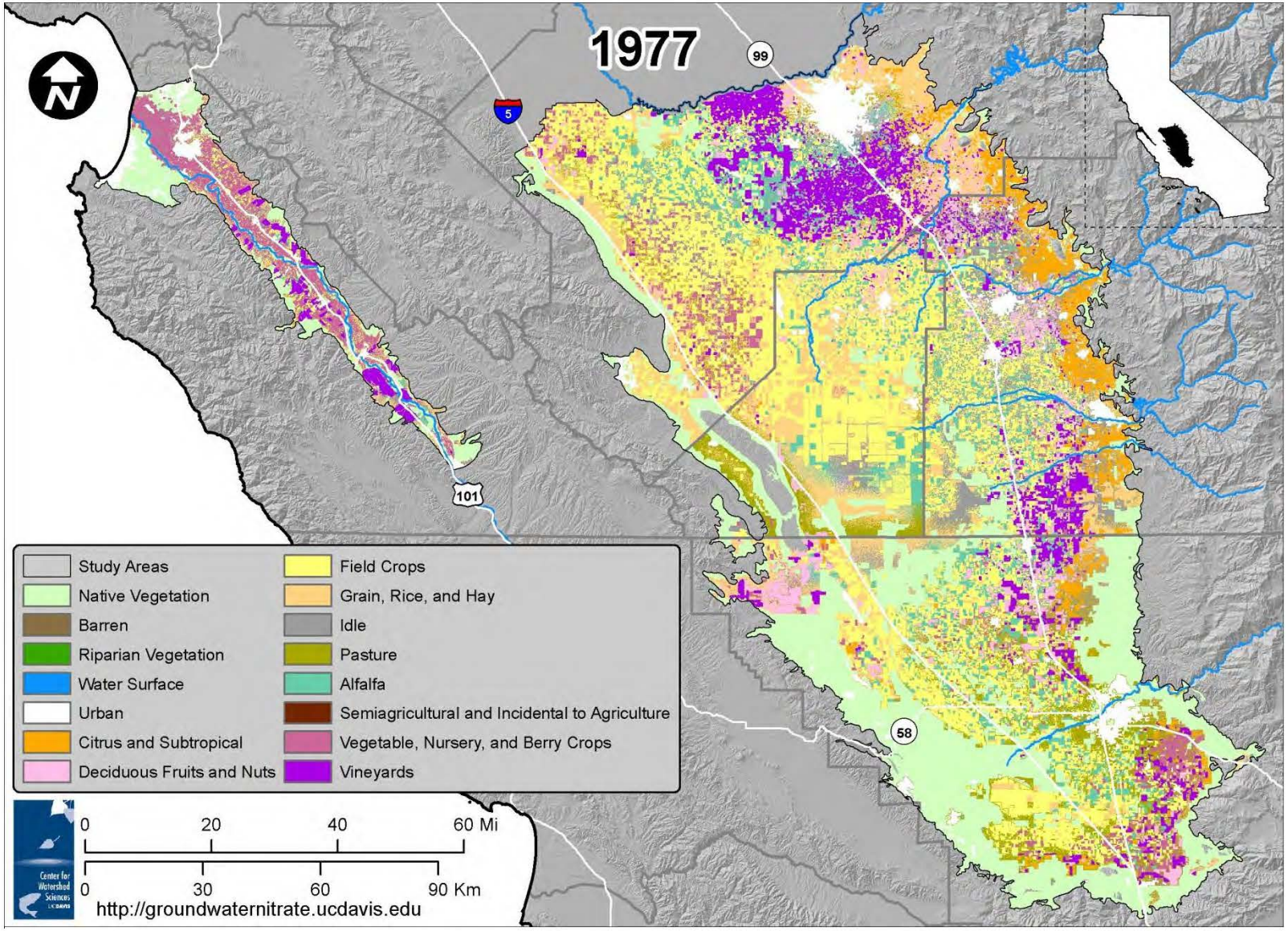


Figure 40. Backcasted 1977 Land Cover Map (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment).

Adminsitrative Record
Page 34396

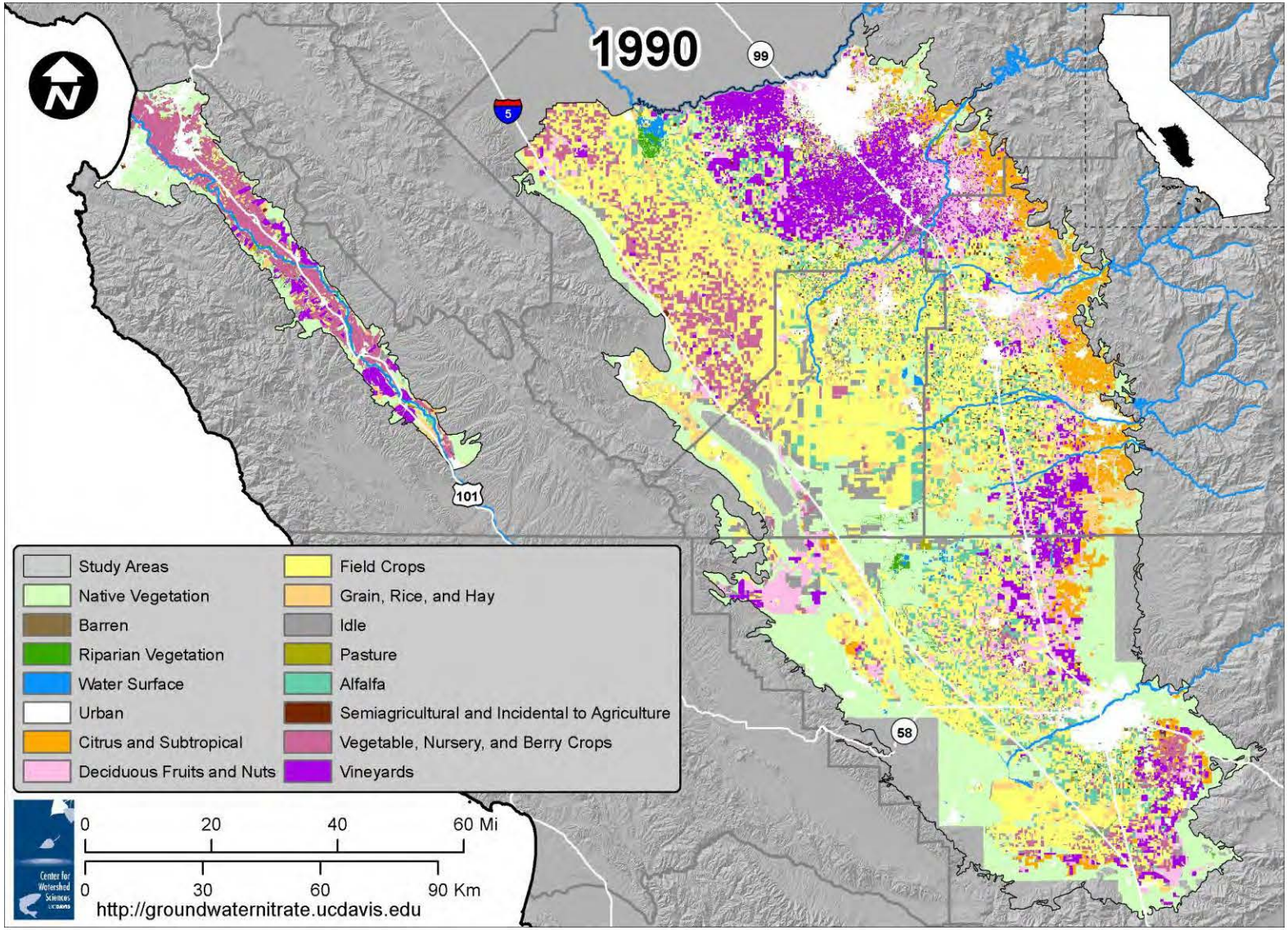


Figure 41. 1990 Land Use from DWR Mapping (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment).

Adminsitrative Record
Page 34397

3.3.1.3 Future Land Use

There are currently 203,844 hectares of urban area within our study area. The UPlan Smart Growth scenario predicts that urban areas will increase by a total of 71%, representing a 28% increase in the Salinas Valley and a 77% increase in the Tulare Lake Basin. For the BAU scenario, urban areas are predicted to increase by a total of 88% – a 40% increase in the SV and 94% increase in the TLB. The Salinas Valley contains 12% of the total urban area within our study area; the Tulare Lake Basin contains 88%.

Based on the BAU scenario a total of 178,892 hectares will be converted to residential, commercial, and industrial uses (Table 18). Agricultural land makes up 78% of this total acreage. The agricultural acreage removed based on the BAU scenario makes up 9% of the total agricultural acreage in the study area.

Table 18. BAU: Acreages for predicted UPlan urban expansion. "Type" indicates the zoning type. For example, "Residential 20" indicates 20 residential units per acre (per 0.4 ha).

Business as Usual (BAU)			
Type	**Hectares (SV)**	**Hectares (TLB)**	**Hectares (Total)**
Residential 20	262	3652	3914
Residential 5	1589	31007	32596
Residential 1	2782	29324	32106
Residential .1	1302	70549	71851
Industrial	296	4641	4937
Commercial High	365	2711	3076
Commercial Low	3468	26945	30413
Total	**10065**	**168827**	**178892**

Based on the Smart Growth scenario a total of 144,215 hectares will be converted with agricultural lands accounting for 76% of the total acreage in Table 19. The agricultural acreage removed based on the Smart Growth scenario makes up 7% of the total agricultural acreage in the study area.

Table 19. Smart Growth acreages for predicted UPlan urban expansion. "Type" indicates the zoning type. For example, "Residential 20" indicates 20 residential units per acre (per 0.4 ha).

Smart Growth			
Type	**Hectares (SV)**	**Hectares (TLB)**	**Hectares (Total)**
Residential 20	310	4422	4731
Residential 5	1147	19338	20484
Residential 1	2447	28111	30558
Residential .1	455	41916	42371
Residential 50	34	302	336
Residential .5	280	4718	4998
Industrial	292	4514	4807
Commercial High	377	2704	3081
Commercial Low	1621	26764	28384
Residential 10	119	4347	4465
Total	**7080**	**137134**	**144215**

Adminsitrative Record
Page 34398

3.3.2 Status and Trends in Typical Nitrogen Fertilizer Use and Harvested N

We applied the rates for typical nitrogen fertilizer use and harvested N to each crop type (or crop category) to two datasets describing cropland area of individual crop types: first, using the county ACR data for harvested land area; and second, using the areas mapped in CAML when simulating field-by-field N loading with GNLM. Section 1.6 summarizes the results of the ACR-based analysis. Section 1.8 summarizes the data for the GNLM simulations. The appendix also shows an extensive set of historic and current maps simulated with CAML-based GNLM. These include maps of actual synthetic fertilizer applications and harvested N. Synthetic fertilizer applications, in contrast to typical fertilizer applications, are computed with GNLM and account for the estimated amount of manure N that is applied as part of the typical fertilizer N rate (see Section 4).

Here we describe in more detail the results of the county ACR based analysis to show historic changes in land area, fertilizer application rates, and harvest (Tables 20 - 25). Results are given for two aggregated levels: aggregated to the crop group (historically and current, and aggregated to the county level (2005 period only).

The crop groups include the following individual CAML designated crop types reported in historic ACRs:

- Alfalfa – alfalfa and (in the 1960 period only) pasture
- Field Crops – miscellaneous field crops, safflower, sugar beets, corn (grain and sileage), sorghum, sudan, beans (dry), sunflower
- Grain and Hay – miscellaneous grain and hay, barley, wheat, and oats
- Nuts – almonds, walnuts, and pistachios
- Subtropical – miscellaneous citrus and pomegranates, grapefruit, lemons, oranges, avocadoes, olives, kiwi
- Tree Fruit – miscellaneous tree fruit, apples, apricots, cherries, peaches and nectarines, pears, plums, prunes, figs
- Vegetables and berries – miscellaneous truck crops, artichokes, asparagus, beans (green), carrots, celery, lettuce, melons and squash, garlic and onions, green peas, potatoes, sweet potatoes, spinach, processed tomatoes, berries, strawberries, peppers, broccoli, cabbage, cauliflower, Brussels sprouts
- Grapes – raisin grapes, table grapes, and wine grapes are together considered one crop type

Table 20 indicates the number of crop types for which data are available within each crop group, for the five historic five-year periods. Generally, all crop types appear in all or almost all years.

Table 20. Number of different crop types or crop categories within each crop group.

CROP GROUP	1945	1960	1975	1990	2005
Alfalfa	1	2	1	1	1
Cotton	1	1	1	1	1
Field Crops	8	8	7	6	7

Adminsitrative Record
Page 34399

Grain and Hay	4	4	4	4	4
Grapes	1	1	1	1	1
Nuts	2	3	3	3	3
Rice	1	1	1	1	1
Subtropical	5	6	7	7	7
Tree Fruit	9	9	9	8	8
Vegetables and Berries	21	21	21	20	19
All Crops Except Alfalfa	52	54	54	51	51

Total harvested area, which experienced a large expansion - more than two-fold growth - between the 1940s and the 1970s, has increased by less than 10% since the 1970s. Alfalfa and small grain and hay crops expanded after World War II, but have remained stable or even decreased in land area over the past forty years. Field crops also have not seen much expansion since the 1960s. Cotton expanded rapidly until 1990 and his since seen drastic declines in harvested area. Over the past forty years, growth has been predominantly in land area used for specialty crops: grapes, nuts, tree fruit, subtropicals, and vegetables and berries (

Adminsitrative Record
Page 34400

Table 21). In many important vegetable crops, multiple crops are harvested each year (see Section 2 for details).

In total, the Tulare Lake Basin now accounts for over 90% (1,500,000 ha or 3,700,000 ac) of the study area's agricultural land, while the Salinas Valley contains 8-9% (about 100,000 ha or 250,000 ac) of the study area's agricultural land (see Section1 for a comparison of land areas between different data sources).

Today, cotton, grain and hay, and field crops make up 40% of the harvested cropland in the study area (about 600,000 ha or 1.5 million ac). Slightly over 10% of all cropland in the study area grows alfalfa. Vegetables account for slightly less than 20% of the harvested area, grapes and nuts each cover about 10% of the harvested area, and tree fruit and subtropicals each cover about 5% of the harvested area.

Adminsitrative Record
Page 34401

Table 21. Historical changes in the total harvested area [ha] in the study area counties, by crop group (from ACR data). Values shown are the median of five years of annual data for each period (year indicates the central year of the period). Areas harvested more than once are counted more than once (primarily applies to vegetables). Actual, on the ground land area is smaller. One hectare is equal to about 2.5 acres.

CROP GROUP	1945	1960	1975	1990	2005
Alfalfa	118,260	210,165	165,247	155,788	169,373
Cotton	104,796	275,464	393,782	429,732	246,810
Field Crops	54,179	99,175	129,095	105,304	131,538
Grain and Hay	210,651	353,793	304,459	161,263	223,468
Grapes	107,967	99,743	131,150	152,613	155,385
Nuts	3,412	5,879	32,463	71,170	136,717
Rice	3,148	10,197	7,790	2,686	2,098
Subtropical	21,100	20,691	58,475	68,055	88,420
Tree Fruit	25,155	25,535	27,978	45,657	61,719
Vegetables and Berries	75,809	90,490	132,626	209,524	280,433
All Crops Except Alfalfa	**606,217**	**980,967**	**1,217,818**	**1,246,004**	**1,326,588**

The total typical nitrogen applied as fertilizer (from synthetic fertilizer or manure fertilizer sources) is 225 Gg N/yr (238,000 tons N/yr). This is an increase of over 50% over the past 30 years. During the preceding 30 years, from the 1940s to the 1970s, fertilizer application had quadrupled. Vegetables and berries now account for more than one-quarter of typical fertilizer N applied, nuts for about 10% of total typical fertilizer applied. Cotton, field crops, and grain and hay crops together use more than half of all typical fertilizer applied (about 120 Gg N/yr or 123,000 tons N/yr, Table 22). Note that these typical application rates do not include other sources of nitrogen that may be applied to a field, such as manure amendments, other green wastes, municipal effluent, biosolids, irrigation water, or atmospheric deposition.

Adminsitrative Record
Page 34402

Table 22. Historical changes in the total nitrogen typically applied [Gg N/yr] in the study area counties, by crop group, based on typical rates for each crop (Appendix Table 7) and the area shown in Table x1. Values do not include excess manure or effluent (see Section 4) and other incidental nitrogen (atmospheric, irrigation water) applied to cropland. Shown is the median of five years of annual data for each period (year indicates the central year of the period). One Gg N = 1,100 tons N.

CROP GROUP	1945	1960	1975	1990	2005
Alfalfa	1.42	3.57	3.64	1.87	2.03
Cotton	6.60	25.34	47.25	82.08	47.14
Field Crops	3.34	8.90	16.53	18.43	29.07
Grain and Hay	10.98	26.01	30.77	25.23	42.06
Grapes	1.19	1.70	2.89	5.65	5.75
Nuts	0.72	1.08	6.35	13.15	25.32
Rice	0.16	0.74	0.74	0.38	0.30
Subtropical	3.84	3.09	10.29	7.10	9.21
Tree Fruit	3.63	3.03	3.72	5.19	7.09
Vegetables and Berries	6.50	11.68	22.06	44.72	59.21
All Crops Except Alfalfa	36.97	81.57	140.61	201.92	225.14

The application rates (the amount of fertilizer N typically applied per area) varies significantly among crops. Alfalfa needs very small amounts of fertilizer. Grapes also are relatively low fertilizer use intensive. Tree fruit and subtropicals are intermediate in their use of fertilizer N – slightly around 110 kg N/ha (100 lb N/ac), although these have been high N users historically (e.g., subtropicals using 176 kg N/ha, 160 lb N/ac, in the 1970s). Application rates in nuts has also been high historically, but remained relatively steady at about 190 kg N/ha (170 lb N/ac). Cotton, field crops, grain and hay crops, and vegetables and berries take the most intensive rates of N fertilizer, at about 190 - 220 kg N/ha (170 - 200 lb N/ac). All of these latter crops have seen continuous increases in fertilizer N rates over the past four decades, with cotton and vegetables remaining constant since about 1990 (Table 23).

The nitrogen removed by harvest is highest in alfalfa, which fixes most of its nitrogen content by direct fixation from the atmosphere. More than one-third (74 Gg N/yr, 81,000 tons N/yr) of the total harvested N in the study area comes from alfalfa and is used as animal feed. Field crops and grain and hay crops, which are also primarily used as animal feed, remove nearly another third of the total harvested N (about 60 Gg N/yr, 66,000 tons N/yr). All are highly valued sources of animal feed protein and represent more than half of the estimated animal protein consumed in the study area.[22]

Table 23. Historical changes in average typical nitrogen application rate [kg N/ha], by crop group, obtained by dividing total typical nitrogen application rate (Table 22) by the total area (

[22] As shown in section 4, total N excretion from dairy cows is about 204 Gg N/yr (224,000 N/yr). In modern dairies, excreted N represents about three-quarter of the protein intake of an adult milk cow, the remainder becomes milk protein. Hence feed protein intake in the study area is about 255 Gg N/yr (280,000 tons N/yr).

Adminsitrative Record
Page 34403

Table 21). One kg/ha = 0.9 lb/ac.

CROP GROUP	1945	1960	1975	1990	2005
Alfalfa	12	17	22	12	12
Cotton	63	92	120	191	191
Field Crops	62	90	128	175	221
Grain and Hay	52	74	101	156	188
Grapes	11	17	22	37	37
Nuts	211	183	196	185	185
Rice	50	73	95	143	143
Subtropical	182	150	176	104	104
Tree Fruit	144	119	133	114	115
Vegetables and Berries	86	129	166	213	211
All Crops Except Alfalfa	61	83	115	162	170

Fifteen percent of the nitrogen harvested is in vegetable crops and less than 10% is in nuts. Grapes, subtropicals, and tree fruit remove relatively smaller amounts of nitrogen (Table 24).

Table 24. Historical changes in total nitrogen harvested [Gg N/yr] in the study area counties, by crop group, based on ACR data of annual yields for each crop. Shown is the median of five years of annual data for each period (year indicates the central year of the period). One Gg N = 1,100 tons N.

CROP GROUP	1945	1960	1975	1990	2005
Alfalfa	34.13	72.50	59.57	59.84	73.84
Cotton	4.31	17.59	24.82	34.32	21.12
Field Crops	2.92	8.43	12.02	15.38	28.25
Grain and Hay	5.71	17.51	23.85	17.94	30.31
Grapes	1.00	1.36	1.89	2.23	2.64
Nuts	0.10	0.23	1.80	4.97	13.07
Rice	0.15	0.63	0.52	0.23	0.18
Subtropical	0.79	0.97	2.10	3.46	4.40
Tree Fruit	0.32	0.37	0.61	1.15	1.47
Vegetables and Berries	3.14	5.83	9.58	18.48	29.78
All Crops Except Alfalfa	18.44	52.91	77.19	98.15	131.21

Nitrogen removal rates are highest (over 400 kg/ha, about 400 lb/ac) in the leguminous alfalfa crop, a main reason for its use as a high protein animal feed. Field crops (predominantly corn) are another crop with relatively high nitrogen uptake, at over 200 kg/ha (about 200 lb/ac). Often, corn (here classified as field crop) are double-cropped with winter grains (oats, wheat, etc.). Annual fertilizer application rate and harvest removal rate are thought to be proportionally higher. Vegetables remove on the order of 100 kg N/ha or 90 lb N/ha per crop. Where double-cropped in a field, the annual rate of applied

Adminsitrative Record
Page 34404

fertilizer N and harvested N for these vegetables would double. Double- or multi-cropped forage (corn and grain) and vegetables are therefore the most intensive N fertilizer users as well as most intensive nitrogen harvests.

Cotton and nuts each remove just under 100 kg N/ha (90 lb N/ac). Even less nitrogen is removed by subtropicals (about 50 kg N/ha, 45 lb N/ac), tree fruit (about 25 kg N/ha, 22 lb N/ac), and grapes (about 17 kg N/ha, 15 lb N/ac, Table 25).

Over the past sixty years, all crops have seen dramatic increases in the rate of harvest removal. While typical nitrogen application rates have remained largely constant since the late 1980s, the ACR data suggest yield rate increases and, hence, increased nitrogen removal rates in all crops except subtropicals and tree fruit.

Table 25. Historical changes in average harvested nitrogen rate [kg N/ha], by crop group, obtained by dividing total harvested nitrogen (Table 23) by the total harvested area for each crop group (Table 21). One kg/h = 0.9 lb/ac.

CROP GROUP	1945	1960	1975	1990	2005
Alfalfa	289	345	360	384	436
Cotton	41	64	63	80	86
Field Crops	54	85	93	146	215
Grain and Hay	27	49	78	111	136
Grapes	9	14	14	15	17
Nuts	31	39	56	70	96
Rice	47	61	67	87	86
Subtropical	37	47	36	51	50
Tree Fruit	13	14	22	25	24
Vegetables and Berries	41	64	72	88	106
All Crops Except Alfalfa	30	54	63	79	99

Across all crops, not including alfalfa, the current average nitrogen harvested is about 100 kg N/ha (90 lb N/ac), compared to an average typical application rate of 170 kg N/ha (about 150 lb N/ac), not including incidental sources of N or soil amendments. Between 1975 and 2005, the average rate of typical fertilizer N application (weighted by the cropped area of each crop) has increased by 47% and the rate of N harvest has increased by about 56%. Most of the fertilizer application increase, however, occurred before the 1990s. Between 1990 and 2005, typical fertilizer application increase was less than 5%, while harvested N increased by 25%, with the most significant increase occurring in field crops and vegetables.

For field crops, it is conceivable, that the lower rate of typical fertilizer N increase, when compared to the increase in harvested N, is due to more extensive use of land applied manure that is not considered as part of the typical fertilization rates reported. This would be consistent with the large increase in manure N for land applications that occurred between 1990 and 2005 (see Section 4).

Adminsitrative Record
Page 34405

The county-by-county tally of harvested area, total typical nitrogen fertilizer applied, and total harvested nitrogen, and their rates, is shown in Table 26. Fresno County has the largest area of cropland, followed by Kern, Tulare, Kings, and Monterey County. Nearly 200 Gg N/yr (220,000 tons N/yr) are typically applied in the Tulare Lake Basin, on 94% of the land area, while almost one-fifth of the total N, 46 Gg N/yr (50,000 tons N/yr) is applied in Monterey County, on less than 10% of the cropland. The higher intensity of typical fertilizer N applications is due to the focus on vegetable and berry commodities in Monterey County. Table 26 also shows land area, typical fertilizer N application, and harvested N in alfalfa, cotton, field crops, and grain and hay crops, which are used in Section 1 to determine the amount of synthetic fertilizer versus manure N fertilizer that constitutes the total typical N fertilizer application in field crops (other than cotton) and in grain and hay crops.

Adminsitrative Record
Page 34406

Table 26. County crop areas, typically applied nitrogen, and harvested nitrogen in 2005, based on the median area of each crop between 2003 and 2007. Several crops are shown separately: alfalfa is shown, because it is not used for the cropland mass balance. Cotton is shown separately, although it is part of the "field crop" crop-group in the CAML classification. "Field crops" and "grain and hay crops" are used to compute manure distribution at the county and study area level based on ACR areas (see Section 1). One ha = 2.5 ac. One Gg N = 1,100 tons N.

	Fresno	Kern	Kings	Monterey / Salinas Valley	Tulare	Tulare Lake Basin	Study Area
Total cropland area harvested [ha]	454,424	366,957	193,754	151,391	317,762	1,332,896	1,484,287
Alfalfa area harvested [ha]	34,648	62,775	31,010	393	41,270	169,703	170,096
Area harvested with seven multipcropped vegetables [ha]	15,674	389	-	101,093	500	16,562	117,655
Total cropland area on the ground [ha]	448,547	366,811	193,754	113,481	317,574	1,326,686	1,440,166
Total cropland area on the ground, except alfafa	413,899	304,036	162,744	113,088	276,304	1,156,983	1,270,070
Typically applied N, including alfalfa [Gg]	65.67	53.22	31.62	28.42	45.68	196.19	224.60
Harvested N, including alfalfa [Gg]	50.60	56.96	33.10	12.55	50.73	191.40	203.95
Typically applied N in alfalfa [Gg]	0.42	0.75	0.37	0.00	0.50	2.04	2.04
Harvested N in alfalfa [Gg]	15.11	27.37	13.52	0.17	17.99	73.99	74.16
Typically applied N in cotton [Gg]	16.98	10.41	12.80	-	4.21	44.41	44.41
Harvested N in cotton [Gg]	7.61	4.66	5.74	-	1.89	19.89	19.89
Typically applied N in field crops [Gg]	5.16	1.06	5.84	0.52	16.04	28.10	28.62
Harvested N in field crops [Gg]	5.03	0.86	5.53	0.39	15.95	27.37	27.77
Typically applied N in small grain and hay [Gg]	5.29	16.60	8.02	0.63	11.58	41.48	42.11
Harvested N in grain and hay [Gg]	3.52	12.11	5.56	0.49	8.56	29.76	30.24

We assumed that the agricultural matrix (i.e. composition of crops) would remain static through 2050. While it is clear that changes in relative composition, and total acreages, change through time, it is also clear that most agricultural land is now in production and further that the range and types of crops cultivated in the study region are near saturation. Therefore, to determine total loading, and in the absence of any information detailing future conditions radically different than contemporary conditions, we only examined the impact of urbanization on nitrate loading. In other words, it is well documented

Adminsitrative Record
Page 34407

that farm land is readily converted to urban uses and further many studies have been conducted to determine the nature of future urbanization (i.e. the spatial extent and composition). The converse is rarely true in that urban lands are rarely if ever converted to agriculture; and the same pattern holds for either agricultural or urban lands being restored back to natural lands, though there are limited examples. Thus, we found that the total load reduction for the BAU scenario only removes less than 5% of the current loading. The Smart Growth scenario only removes about 3% of the total loading. In both cases the projected urban expansion only chips away at the total loading from agriculture and currently does not factor in the increased loading that would result from further development, including runoff from fertilized lawns and golf courses (Section 2.5), and changes in the number septic systems or number of persons served by sewers (Section 2.6).

3.4 Sources of Uncertainty and Information Needs

3.4.1 Current Land Cover Map: Uncertainty

Although the current land cover map was assembled from the best available sources, it does not represent a single point-in-time snapshot of land cover. The DWR land use surveys for the five counties in the study region range in time from 1997 to 2006. Maps and assessments of N loading derived from the current land cover map therefore reflect a patchwork ground condition that is from 5-14 years in duration. As long as the pattern of crop types change slowly over this time period, the temporal heterogeneity does not present a statistical problem for the N loading modeling. It should be remembered, though, that both the land cover mapping and much other spatial data used in this project represent general conditions over an extended period of about a decade. For land cover, the data sources that are available to cover the study region at a single point in time are either too coarse in spatial scale (for instance the crop information from the Department of Pesticide Resources Pesticide Use Reports) or inaccurate (for instance the USDA nationwide NASS mapping (U.S. Department of Agriculture, National Agricultural Statistics Service 2010), a satellite-image-based data source that works better for Midwestern agricultural landscapes than it does for California).

3.4.2 Historic Land Cover Map: Uncertainty

No attempt was made to validate the backcasted land cover maps against external map sources, and hence we do not know how accurately they portray the spatial patterns of cropping in the periods they represent. To a large degree, the data do exist to test the validity of this approach, but these data are mostly in the form of hardcopy blueprint maps stored in the archives of the various divisions of DWR, and it would be a massive and costly effort to digitize these maps to perform the analysis. In fact, the crop acreages match the values in the county reports well, but that is by design, and it is not clear how well the spatial proximity rule used to generate the backcasted maps actually reflects the reality of historical changes in cropping patterns.

Looking more closely at the backcasted maps, it is clear they are coarse approximations, with many dissimilar crops adjacent to each other at the one-hectare pixel scale. There is some tuning of the

Adminsitrative Record
Page 34408

algorithm that can be done to improve these data, such as decreasing the resolution of the maps while running the algorithm, but that approach may introduce more disagreement with the crop report totals through rounding errors. However, the fine-resolution details of the map may not impact modeling outcomes because of the lower resolution of the groundwater modeling. In the end, the groundwater modeling relies upon the N leaching rate and not the crop type.

Some of the categories used in the crop classification are problematic, particularly in terms of lumping crop types. For instance grain crops are sometimes reported in terms of their constituent crops (barley, wheat, etc.) and are lumped together at other times in a single category, depending upon which map or crop report is the source. This confusion leads to difficulties in interpreting changes in crop types over time. The most problematic category is pasture, a term which seems to elude rigorous definition: sometimes in crop reports pasture is used to characterize annual grasslands that are used as rangeland for grazing, a very different land cover type than an irrigated grassy lowland field, also termed pasture.

As mentioned above, the intensity of double-cropping in the past is not well-quantified, and the true "double-cropping factor" used to derive on-the-ground acres in the backcasted maps for six crops in Monterey County is probably well under the value of 2.19 used here. But since subsequent N loading modeling uses the harvested acreage for its calculations, rather than the on-the-ground acreage, the factor cancels itself out in the calculation of total N loading on a county-wide basis, so its actual value is less than critical. Changes in the value of the factor do however change the specific locations of particular crops, which in turn affect the fine-scale modeling of N reaching groundwater.

3.4.3 Crop Reports & Harvested N: Uncertainty

There are several sources of uncertainty in our approach to calculating harvested N rates:

1) Production statistics: While the NASS Census of Agriculture is generally thought of as the gold standard for agricultural statistics, it focuses on acreage and does not include the tonnage of production for most California crops. NASS does conduct annual surveys of acreage, tonnage, and price for most crops, but only at the state, not the county, level. To get a more precise accounting of production tonnage at the county level, and for specific crop types, the county ACR data were used. Although a comparison cannot be done for tonnage because of the lack of data, at the statewide level, the harvested acreages reported for most unique crops based on the ACR data are highly correlated with the NASS Census of Agriculture – a regression analysis comparing the NASS and ACR harvested acreage data shows a coefficient of determination value of 0.84. Grapes, for example, is one crop whose harvested acreage numbers are highly correlated between NASS and the ACR data, while reported numbers for cotton may differ depending on what each agency includes as part of that crop's acreage number (i.e. cotton lint, cotton seed). While there is no independent way to verify these data, the annual variability is relatively low.
2) Nutrient content: The USDA Crop Nutrient Tool is the most comprehensive database of its kind, but it is somewhat dated and less focused on the fruit and vegetable crops prevalent in California. There is some concern that the nutrient content has changed over time because of

Adminsitrative Record
Page 34409

differences in the varieties planted and the management practices, but there are few data to verify this in California specifically. For crops harvested as hay, the N content varies considerably with stage of development as well as the numbers of cuts for alfalfa. Error in the moisture content for most grains is insignificant for the calculations. For most fruits and vegetables, the moisture content is a bigger source of error than the N content. For example, the moisture content of fresh tomatoes is 94%; if the moisture content were really 92%, we would be underestimating the harvested N by 25%.

3) Combining of crops: For most widely planted cover types this is a minor source of error as the commodities are easily combined into unique crops and then cover types. While there is some error in the spatial distribution of N mass balance by combining different varieties of the same crop (e.g. wine grapes, raisins, and table grapes), the total N loading for the cover class should be representative.

Although the ACR data are a key source of information used to show how cropping patterns and agricultural production may have shifted over time, it is difficult to amalgamate what should be similar information across various counties. Often with these historical data, we have determined that a county might only list the total acres in production for a particular commodity, with no corresponding note as to how many of these acres are actually harvested. For example, Tulare County crop reports include numbers detailing both non-bearing and bearing acres, and the total production from the latter, whereas Monterey County acreage totals listed in the commissioner's report reflect the total number of acres harvested (which includes double and triple cropping). In this case the acreage totals listed might be twice or three times the actual ground acres. The challenge here is determining which of the acres reported contribute to production totals and which acres were out of production for that year.

Calculating production totals also proved challenging prior to 1958 for the counties in our project area, as agricultural production was often reported in whichever unit was used for a particular commodity (i.e. lugs, crates, sacks, cartons, etc.). In these historic crop reports, there is typically no mention of how units equate. Using current unit conversion rates to convert historic data are not always correct, as unit measurements have changed over time. In counties where no unit conversion rates could be determined, NASS provided an appropriate rate to use.

The relatively small amount of nitrogen contained in byproducts (e.g., almond hulls or cottonseed) is included in the calculation when data were available to estimate nitrogen content. Nitrogen content of products may deviate from the average values, especially for crops that are heavy consumers. The nitrogen content of such crops will vary depending on the mineral nitrogen available in the soil. Applying average values in this analysis is reasonable, however, because of the large scale of the analysis.

3.4.4 Information Needs

Combining the wide array of data developed and integrated into this analysis highlights a number of information needs. Chief among these needs is recognizing the importance of historical datasets and the need to have long, consistent time series for a large variety of data (e.g., cropping by type, yield,

Adminsitrative Record
Page 34410

irrigation, etc.). For the land cover mapping, the interval between DWR field visits is long (typically seven or more years.) Although DWR has a large archive of maps from their land cover mapping work from the 1950s through the 1980s, those information sources are available only in paper format. These maps provide a unique historical record of land cover change in California, but to properly use them for quantitative analyses, it is necessary for them to be digitized and brought into a proper digital framework (i.e. a GIS). Moreover, the lack of field level accounting for crop type, rotation type, and yield magnitude severely limit any prospective analyses at the scales necessary to hone nitrate leaching minimization strategies. Recent advances in high spatial and temporal resolution spectral remote sensing may provide the necessary information going forward; however, governmental entities will need to provide leadership and oversight in their collection and analysis.

Similar information needs pertain to county level cropping data. During the process of digitizing the ACRs, it became increasingly clear that the task of accurately comparing and measuring crop acres, both on the ground and harvested acres, and production in the Tulare Lake Basin and the Salinas Valley would benefit greatly from a concentrated effort to standardize historic, and in some cases contemporary, crop reports across counties. This might include a more comprehensive literature review of historic documents detailing specifics on agricultural commodity weights and measures from the 1940s through today. Although the agricultural commissioners' offices were extremely helpful in providing the available information, they noted that the historic information was difficult to obtain, as employees who created such data have long since retired and only base level reporting was in effect at the time. The National Agricultural Statistics Service's California field office and their Caudill Library were valuable resources for estimating historical unit conversion rates, using information from non-digital legacy documents.

Adminsitrative Record
Page 34411

4 Nitrogen Loading from Dairies, Feedlots, and other Animal Farming Operations

4.1 Introduction to N Loading from Dairies

The Tulare Lake Basin (TLB) portion of our study area is home to over half of California's dairy herd, housed predominantly in family-owned and operated confined animal farming operations. The TLB also houses one large beef cattle facility (Harris Ranch). This chapter (Section 4) focuses on dairy operations as the major source of animal manure nitrogen in the project area. Beef lots are also considered. Poultry and swine manure production is discussed in Section 4.9.

Table 27. Historic number of milk cows, not including dry cows (*from:* National Agricultural Statistics Service). The number of adult dairy cows (lactating and dry) is approximately 20% larger than the number of milk cows.

	1950	1992	2002	2007
Fresno Co.	34,695	72,350	90,550	114,768
Kings Co.	24,012	86,235	138,292	163,600
Tulare Co.	38,981	215,480	412,462	474,497
Kern Co.	9,962	34,566	74,708	124,756
TLB	**107,650**	**408,631**	**716,012**	**877,621**
Monterey Co. (SV)	**9,953**	**4,323**	**1,606**	**2,143**

The TLB is home to approximately 640 dairies with 1 million milking cows (lactating and dry) and over one million support cattle (calves, heifers). Over 10% of the national milk production occurs in the study area. In 2007, the Central Valley (Region 5) Regional Water Quality Control Board (RB5) adopted the Waste Discharge Requirements General Order No. R5-2007-0035[23] ("Dairy General Order") (Central Valley Regional Water Quality Control Board 2007), which regulates waste discharges from dairy operations. The Dairy General Order requires annual reporting as part of each operation's waste and nutrient management planning. For 2007, dairy operators reported 1,020,000 mature dairy cows in the four TLB counties. For the same four counties, the National Agricultural Statistics Service (NASS) reported 877,621 milk cows (about 1, 050,000 adult cows, assuming a ten month lactation period and a two month dry period) in its 2007 census and 716,612 milk cows in its 2002 census. The number of milk cows in 2007 was more than twice the herd size of 408,631 milk cows reported in the 1990 agricultural census (NASS 1990). It is an eight-fold increase in herd-size since 1950, at which time the agricultural census reported 107,650 milk cows for the four TLB counties.

[23] http://www.swrcb.ca.gov/rwqcb5/board_decisions/adopted_orders/general_orders/r5-2007-0035.pdf

Adminsitrative Record
Page 34412

In contrast, Monterey County (of which the Salinas Valley is a part), housed as many as 23,000 dairy cows in 1945 according to ACR data, but as few as 6,000 adult dairy cows in 1992 and less than 1,500 adult dairy cows in 2007. NASS reports 9,953 milk cows in 1950, 2,143 milk cows in 2007, and 24,686 beef cows in 2007. Today, the Salinas Valley itself houses only one significant cattle facility at the valley margin near Gonzales (Gallo Cattle Farm). The remainder of this chapter will therefore focus on the Tulare Lake Basin.

A typical dairy in the Central Valley consists of many different operational units, all of which can potentially become sources of groundwater nitrate:

- owner and worker housing,
- septic leach field(s),
- shops and equipment storage area,
- animal housing and exercise areas (freestalls, corrals) with
    - a central milking barn,
    - heifer corrals,
    - calf housing area,
    - feed storage area,
- solids manure storage areas,
- one or several liquid manure storage lagoon and settling basins,
- forage and other crop fields.

By far the largest land area of Central Valley dairies are their irrigated crop fields (mostly forage crops), which typically receive liquid manure and solid manure applications from the dairy as part of their nutrient management program. Dairy General Order reports provided by dairy owners to RB5 in 2007 totaled 130,000 ha (315,000 acres) of land application area (based on Existing Conditions Reports for 2007). We also mapped assessor's parcel numbers provided under the Dairy Order to RB5 (Fresno Office). Spatial analysis of these maps yield similar, albeit slightly smaller total areas. More importantly, they provide an impressive comparison of the size of cropland receiving manure applications versus the total acreage of the facilities themselves: dairies reported parcels with 109,500 ha (270,000 acres) of cropland and 11,900 ha (29,500 acres) of facility areas (corrals, milking barns, storage areas, and lagoons).

In the TLB, dairies operate either as a freestall operation, as a drylot operation, or as a combination of both. In a freestall dairy operation, adult animals are housed in covered freestalls that have access to exercise yards, which are here referred to as corrals. Freestalls are long rows of individual stalls bordered on the front side by a feed bunk and on the back side by a concrete-paved flush- and travel-lane used for both, manure collection and as access pathway for the animals to their stalls. The stalls themselves are unpaved and generally bedded with dry manure solids or other dry materials (e.g., almond hulls) that are refreshed frequently to keep the freestalls clean and comfortable for the animals.

Adminsitrative Record
Page 34413

Feed rations are distributed into feed bunks along the front of the freestalls. Two or three times daily, milking cows walk to a centrally located milking barn. The entire complex of freestall, flush-lane, and feed bunk is roof-covered to protect from sun and rain.

Animal manure (from liquid and solid excretions) accumulates primarily in the flush-lane that passes behind an individual animal's bedded stall. Flush-lanes are flushed two- to five times daily with recycled water from the liquid manure storage lagoon. Flush lanes are also used to traffic animals to and from the milking barn. Flush water is collected, passes through a mechanical solid separation system, and the liquid portion (with suspended solids) is stored in a manure storage lagoon. A number of different collection and solid separation systems are available and in use. Systems differ in their effectiveness of separating coarse solids and fine solids from the liquid fraction. Separated solids are generally stored in stockpiles or windrows for drying and storage. Dried, separated solids are reused for bedding in freestalls and corrals, as soil amendment in crop fields, or hauled off-property as soil amendment. Liquid manure is stored in manure storage lagoons ("lagoons") and recycled for flushing. All liquid manure is ultimately blended with irrigation water and used as fertilizer in crop fields associated with the dairy.

Drylots are earthen-surface exercise yards without flooring or plant cover, and usually without any roofing. So-called drylot dairies mostly lack flushlanes for the collection of manure, except in the milking barn area and its associated travel lanes. Animal excrements collect in the corral area, which is regularly scraped. Scraped solids are dried, sometimes (partially) composted and then either reused as bedding in the freestalls and corrals, used as soil amendment in fields, or sold off-dairy as soil amendment. The total roofed area (which may affect the amount of runoff diverted to a lagoon) in a drylot dairy tends to be less than in a freestall dairy.

Dairies also collect surface runoff from animal housing areas. Stormwater runoff from roof tops is often collected separately and diverted to stormwater drains. Any runoff that has come in contact with animal waste must be collected in the liquid manure storage lagoon.

Of the various management units within a dairy, the three major areas for potential groundwater nitrate loading are the corrals (uncovered animal holding areas), the liquid manure storage lagoons ("lagoons"), and the crop fields receiving either liquid or solid manure applications or both (manured cropland). Septic leach fields as a source of groundwater nitrate are reviewed in Section 6.4. Beef cattle feedlots, as a source of groundwater nitrate, are considered here to function similarly to dairy corrals, although the animal stocking rate may be significantly higher.

The following sections first provide a review of literature and field data of nitrate loading to groundwater, then describe the specific methods applied in this report to estimate groundwater nitrate loading from animal farming operations in the TLB. Data are provided and results presented and discussed. Separate methods were applied for corrals, lagoons, and manure irrigated croplands, as described below. Briefly, groundwater nitrate loading from corrals and lagoons is based on recharge rates and nitrate concentrations found in previous field studies, and based on the actual size of a corral or lagoon. Groundwater nitrate loading on manured cropland, as on other cropland, is estimated by

Adminsitrative Record
Page 34414

considering all nitrogen fluxes to and from an individual field, which are crop type dependent and include fertilizer and manure nitrogen applications, and harvest removal of nitrogen, among others. Groundwater nitrate loading on cropland is then estimated as the difference between nitrogen inputs to and outputs from an agricultural field (mass balance approach) rather than based on literature values.

A note on measurement units: Unless noted otherwise, this section (and others in this report) reports nitrate concentrations in water in mg nitrate (nitrate) per liter [mg/l], a unit for which the maximum contaminant level (MCL) in drinking water is 45 mg/L. However, in the agricultural context, fertilizer and manure nitrogen is applied in various forms including organic nitrogen (N-org), ammonium nitrogen (NH_4-N), and nitrate nitrogen (nitrate-N). For agronomic calculations (application rates, harvest rates, etc.), nitrogen mass flux of any of these forms is typically reported in mass of nitrogen (N), rather than in the mass of the specific nitrogen-form (organic N, ammonium, or nitrate), to allow for direct comparisons of these fluxes. When convenient, we therefore will sometimes be reporting nitrate concentrations in mg nitrate-N per liter, denoted by (mg N/L). In that case, the MCL in drinking water is 10 mg N/L (i.e. as nitrate-N), which is equivalent to 45 mg/L (as nitrate).

A note on unit conversions: Original measurements and estimates are all made in scientific units using the metric system and at least five significant digits. The scientific units are here reported to one, two, three, or more significant digits depending on the approximate accuracy of the estimate or measurement. Conversions to American units are sometimes made from the original number (with a large number of significant digits) and sometimes reflect a direct conversion of the number reported here (with limited number of significant digits). Regardless, we always report numbers of the American unit system with the same number of significant digits as the numbers reported in the scientific units.

4.2 Review of N Loading Rates from Dairy Corrals

The largest number of animal feedlots and corrals in the Tulare Lake Basin is associated with dairy facilities. In addition, there are several mostly small feedlots throughout the TLB and one large feedlot (Harris Ranch), and only a single cattle farm in the Salinas Valley. We mapped the total area of open dairy corrals in the Tulare Lake Basin in 2010 using a 2007 list of dairy addresses provided by the Regional Water Quality Control Board, Department of Water Resources year 2000, 2003, 1999, and 2006 land use surveys[24] for Fresno, Kings, Tulare, and Kern County, respectively, and 2009 aerial photography provided by the Department of Conservation Farmland Mapping and Monitoring Program[25] (FMMP) as the basis for digitization of the actual open corral area in dairies.

Feedlots and corrals are characteristically an un-vegetated, bare soil area where cattle spend all (dry-lot dairy) or part (freestall dairy) of their time. Animal stocking densities vary. Within the Tulare Lake Basin, our digitized maps of open dairy corrals on approximately 640 dairies show that these corrals encompass 8,316 ha (20,548 acres). Approximately half of the corral acreage in the TLB is concentrated in the Tulare County portion of the study area. The average stocking rate is on the order of 123 adult

[24] http://www.water.ca.gov/landwateruse/lusrvymain.cfm
[25] http://www.conservation.ca.gov/dlrp/fmmp/Pages/Index.aspx

Adminsitrative Record
Page 34415

animals per ha (50 adult animals per acre) (81 m^2 per adult animal). The same space is shared with an additional 1.4 support stock animals (calves and heifers) per adult cow (according to EPA data, see below). In addition, two relatively large beef cattle feedlots stand out, with exceptionally higher than average stocking rates. Harris Ranch, located on the Westside of the TLB, houses approximately 100,000 head of cattle on over 320 ha [800 acres]. The Joseph Gallo Cattle Company Feedlot, located at the edge of Salinas Valley east of Gonzales, houses up to 30,000 head of cattle on approximately 40 ha (100 acres, stocking rate of at least 13 m^2/head or 310 hd/ac). Together, these two feedlots average 300 to 800 head of cattle per hectare (120–320 head of cattle per acre, stocking rate of 12.5–33 m^2/head):

California regulations require that corrals have sufficient slope for rapid drainage during rainstorms. Ponding (storage of water from corrals) beyond 72 hours after the last rainfall is illegal. According to the 2007 Dairy General Order, B.6, p.16:

> "The milk parlor, animal confinement area (including corrals), and manure and feed storage areas shall be designed and maintained to convey all water that has contacted animal wastes or feed to the wastewater retention system and to minimize standing water as of 72 hours after the last rainfall and the infiltration of water into the underlying soils."

Typically, the corral surface soil consists of three distinct layers: a manure pack, a compacted black interface layer, and the underlying original soil (Mielke et al. 1974, Miller et al. 2008). The manure pack predominantly consists of fresh and aged manure, sometimes mixed with bedding material (in many cases dried, aged manure reapplied to the corral surface). The hoof action and weight of the animals lead to mixing of manure with the underlying original soil, and to the subsequent compaction of this mixing layer. The black compaction layer is typically from 5 to 15 cm thick (2-6 in). The hydraulic conductivity of this layer is much lower than that of the natural, underlying soil. This is due to:

1. compaction and mixture of manure with native soil materials; and
2. high microbial content of this interface layer, which forms biofilms that further impede water flow (Mielke et al. 1974).

For example, Miller et al. (2008) found in a study conducted in southern Alberta, Canada, that the hydraulic conductivity of corral floors is similar for medium-fine textured soils (33%–39% clay content) and for medium-coarse textured soils (12% clay content). This comes despite the fact that these are two hydraulically very different soil parent materials. Saturated hydraulic conductivities, measured in field infiltrometer experiments, ranged from 4 to 93 x 10^{-7} m/s (0.1 – 2.6 ft/d) (ibid.). Analysis of the chloride profile below three feedlot pen surfaces (aged 4, 5, and 53 years) revealed elevated chloride (4,000 ppm) only to 0.7 m (2.3 ft) depth (200 ppm below that depth). Average annual rainfall at this study site is 378 mm (15 in). A similar study at four older beef feedlots in central and northeastern Kansas (21 to 50 years of continuous operation) also used chloride to determine the leaching depth below the corral surfaces (Vaillant et al. 2009). Long-term average annual rainfall at these sites ranges from 630 mm to 880 mm (25 – 35 in), three times higher than in the Tulare Lake Basin (about 250 mm or 10 in). Underlying soil textures range from silty clay loam to loamy fine sand. Cattle stocking densities varied from 17 to 29 m^2/head (140-240 heads/ac). Nitrogen deposition rates in the pen surfaces were

Adminsitrative Record
Page 34416

estimated to range from 23,000 to 42,000 kg N/ha/year (21,000 – 37,000 lbs/ac/yr). The annual water equivalent in urine and manure at these sites was estimated to range from 1,100 mm to 1,460 mm (43-57 in). Following the results of Kissinger et al. (2007), 13% of this nitrogen was estimated to be available for leaching. However, the total amount of nitrogen found in the soil profile was only one-fifth of the estimated leachable nitrogen during the quarter to half century of feedlot operation (1,000 kg N/ha/yr [900 lbs/ac/yr] or about 3% of the excreted N). Ammonium and chloride concentrations, while highly elevated near the surface, reached background levels at depths of 1 m to 2 m (3–7 ft) below the corral surface. Nitrate concentrations in the soil profiles were also below background levels at depths of 2 m (7 ft) and lower with indication of anoxic conditions at some sites. The results suggested no significant leaching of nitrate to below 3 m [10 ft] from these four sites. Other researchers come to varied conclusions on the leaching potential of feedlot pens and corrals – some studies indicate leaching while others indicate no leaching (reviewed in Miller et al. 2008, Vaillant et al. 2009).

Harter et al. (2002) reported data from a monitoring well network across five dairies in the dairy region west of Modesto and Turlock (Stanislaus and Merced County), where groundwater is shallow (depth to water table less than 4.5 m (15 ft)) and where soils are well-drained and relatively coarse-textured. In that study, nitrate concentrations in monitoring wells downgradient of corrals averaged 293 mg/L (as nitrate) with a coefficient of variation of 0.45. While the nitrate concentration downgradient of corrals was often similar to those upgradient of the corrals, a significant increase in the groundwater salinity between upgradient and downgradient corral monitoring wells indicated that the downgradient nitrate originated from the corral area/production facility. In a few of the cases in the Merced/Stanislaus county study, upgradient nitrate concentrations were significantly less than downgradient of the corrals.

We would expect similar groundwater concentrations in dairies in the Tulare Lake Basin, where soils are similarly well-drained and where the water table is less than 15 m (50 ft). Few other studies have focused on leaching from corral areas in dairies. In a study of three Georgia dairies, impacted groundwater nitrate ranged from 212 to 608 mg/L (Drommerhausen et al. 1995).

In a mass balance and groundwater modeling study of a dairy landscape, VanderSchans et al. (2009) found that groundwater models are insensitive to leaching from corrals, but estimated that urine and manure adds approximately 500 mm/yr (20 in/yr) of equivalent water to the corral surface, much of which evaporates. Total leaching rates (from manure equivalent water and precipitation) were estimated to vary from 290 mm/yr (11 in/yr) to 580 mm/yr (23 in/yr) for a sloped and unsloped corral, respectively. Annual nitrogen loading to groundwater from corrals was estimated to be 872 kg N/ha (778 lbs/acre), obtained by calibrating the loading rate against measured monitoring well observation data. This value is consistent with the annual accumulation rate of 1,000 kg N/ha/yr (900 lbs N/ac/yr) found in the soil profile by Vaillant et al. (2009, see above).

Corrals as a source of groundwater nitrate were also reviewed in a report to the Central Valley Regional Water Quality Control Board (Brown et al. 2003). Facility-average nitrate concentrations in monitoring wells downgradient from corrals were reported to be ranging from 1 mg/L to as high as 110 mg/L (as nitrate). The total average across ten facilities was 58 mg/L. Five of those facilities with corral monitoring wells are located in the Tulare Lake Basin (Tulare/Kings/Fresno Counties). Average reported

Adminsitrative Record
Page 34417

nitrate concentrations at each of these dairies were 1, 18, 46, 95, and 110 mg/L. These concentrations are significantly lower than those found by Harter et al. (2002) in the Modesto area.

Harter et al. (unpublished data) recently completed an extensive groundwater sampling program in a monitoring well network spanning five dairies in Tulare and Kings County. The campaign included a total of seven corral monitoring well sites, with water table depths ranging from 15 m to 30 m (50 ft to over 100 ft). Over a 2.5 year period between 2007 and 2009, they measured nitrate at eight sampling events per year. Average nitrate concentration in corral monitoring wells was 55 mg/L (193 samples) with a coefficient of variation of 0.67. The *relative* variability is similar to the earlier study in the northern San Joaquin Valley, but the concentrations are significantly lower and comparable to Brown, Vence & Associates, 2004. Concentrations typically vary from below the MCL to as high as three times the MCL.

In contrast, soil cores taken at the monitoring well sites (Harter et al. unpublished data) during the well construction revealed elevated nitrate concentrations in the upper unsaturated zone, with concentrations typically above 200 mg/kg (dry soil) near the surface and gradually decreasing to 20-50 mg/kg at 10 m to 15 m (35 to 50 ft) depth. Elevated nitrate concentrations were associated with two older dairies that have been in operation for well over 50 years. The thickness of the affected unsaturated zone soil layer is significantly larger than in the Kansas feedlot study (Vaillant et al. 2009). If we interpret the 10 – 15 m (35–50 ft) penetration depth of elevated nitrate (and salinity) in these profiles as an expression of the downward water and solute movement rate underneath corrals, then the effective downward transport velocity in the unsaturated zone is 0.2 – 0.3 m/yr (0.7–1 ft/yr). Given an average moisture content of approximately 20% in deep alluvial vadose zones (Onsoy et al. 2005; Scanlon et al. 2010), the effective recharge rate under these corrals can then be estimated to be in the range of 40–60 mm/yr (about 2 in/yr). Consequently, the nitrogen loading rate, given an approximate average nitrate concentration of at least 90 mg/kg (20 mg N/kg) in the upper unsaturated zone profile, can be readily computed to be 60 – 90 kg N/ha/yr (50 –80 lbs/ha/yr). This is an order of magnitude less nitrogen loading from corrals than estimated by Vaillant et al. (2009) and by VanderSchans et al. (2009). Possible factors for this discrepancy are lower stocking rates, a significantly drier climate (250 mm [10 in] total precipitation) and higher annual ET than in either the northern San Joaquin Valley or in Kansas, and therefore both, lower recharge rates and higher atmospheric losses of nitrogen. Lower recharge rates and these lower nitrate loading rates are consistent with the uppermost groundwater nitrate concentrations found in Harter et al. (unpublished data) and in Brown, Vence & Associates (2004) (see above).

4.3 Groundwater N Loading from Corrals: Methods and Results

County and Study Area Nitrate Loading from Corrals: Review of literature data and field data from Tulare and Kings County provide a wide range of potential nitrogen loading and recharge rates in corral areas (see section 4.3). The only direct measurements of nitrogen in the deeper unsaturated zone below corrals in the Tulare Lake Basin indicate an annual loading rate that is on the order of at least 75 kg N/ha/year (70 lbs N/yr) with recharge rates around 50 mm/yr (2 in/yr) and corresponding soil moisture nitrate concentrations on the order of 675 mg/L (Harter et al. unpublished data, see Section

Adminsitrative Record
Page 34418

4.3). These data may provide a lower bound estimate of corral nitrogen leaching. However, these data do not include measurements from the first 2 m (7 ft) below the corral surface, where potentially most of the nitrogen is stored, but not transported to groundwater. Other studies vary in their estimation of recharge in corrals and exercise yards under similar climate conditions from less than 40 mm/yr to 300 mm/yr (2-12 in/yr) (Vaillant et al. 2009, VanderSchans et al. 2009), but at estimated nitrogen loading rates that are approximately one order of magnitude larger. Based on these latter results, an upper limit for the loading rate from corrals in the TLB is 1,000 kg N/ha/year (900 lbs/ac/yr).

Table 28. County by county summary of corral area and of the lower and upper limits of estimated N loading to groundwater. The numbers of dairies reflect 2007-2009 conditions.

	Number of Dairies	Corral Area [ha]	Corral Area [acres]	N leached below corral – Lower Limit [Mg/yr] (tons/yr)	N leached below corral – Upper Limit [Mg/yr] (tons/yr)
Fresno Co.	108	1,105	2,731	84 (93)	1,100 (1,200)
Kings Co.	162	1,574	3,889	120 (130)	1,600 (1,700)
Tulare Co.	315	4,168	10,300	320 (350)	4,200 (4,600)
Kern Co.	54	1,468	3,628	110 (120)	1,500 (1,600)
Total TLB	**639**	**8,316**	**20,548**	**630 (700)**	**8,300 (9,200)**

The corral area was obtained by digitizing corrals into a geospatial database using 2009 aerial imagery. The lower limit was obtained by assuming a loading rate of 75 kg N/ha/yr (70 lbs/ac/yr) (Harter et al. unpublished data). The upper limit was obtained by assuming a loading rate of 1,000 kg N/ha/yr (900 lbs N/ac/yr) (Vaillant et al. 2009). The numbers of dairies were provided courtesy of RB5. Numbers in parenthesis represent tons (1 ton = 2,000 lbs).

For the 8,316 ha (20,548 acres) of dairy corrals in the Tulare Lake Basin, potential groundwater nitrogen loading to groundwater is estimated to be in the range of 630 – 8,300 Mg N/yr (700 – 9,200 tons/yr). Approximately half of this load occurs in Tulare County (Table 28). Beef lot corrals may contribute an additional 5% to 10% of the total shown in Table 28, from 32 to 830 Mg N/yr (35 – 920 ton/yr).

For the TLB, given its relatively low precipitation, the large depth to groundwater underneath many dairies, and given the limited set of field data, the upper bound is likely a conservatively high value and likely to significantly exceed actual loading rates, perhaps by as much as one order of magnitude.

Although unknown sources of nitrogen loading to groundwater in the corral area include leaking underground pipelines for manure recycling within the production facility area, these here are not considered to contribute substantially to the above stated range of total loading rates from corrals.

Adminsitrative Record
Page 34419

Current Storage of N in Corrals: A potentially significant but unknown portion of the nitrogen excreted onto corrals is stored in the uppermost unsaturated zone at depths of 0-2 m (0–7 ft), possibly for years or even decades (Miller et al., 2008; Vaillant et al., 2009). Here we provide four independent estimates of the potential magnitude of soil nitrogen storage in the immediate subsurface below the dairy corrals in the Tulare Lake Basin.

The first estimate is based on a single soil core that we obtained from three boreholes drilled in corrals at two Kings County dairy sites with more than 30 m (100 ft) depth to groundwater and a corral age of at least 40 years. The highest measured total nitrogen concentration was 500 mg N/kg (0.05%) in a core extending from 0 – 0.6 m (0 – 2 ft). In cores below 0.6 m, concentrations were generally below the detection limit of 200 mg N/kg (<0.02% total nitrogen). Assuming a total storage of 500 mg N/kg (0.05%) and a bulk density of 1.5 g/cm^3, the total nitrogen storage within the upper 2 m (7 ft) is 15,000 kg N/ha (13,000 lbs/ac). At this level for the entire Tulare Lake Basin, the nitrogen storage would be 125,000 Mg N (138,000 tons) in 8,316 ha corrals.

A second estimate can be made based on the nitrogen excretion rate of dairy cows and the known stocking density. We make the following assumptions:

- the annual excretion rate is 198 kg N (437 lbs) per adult dairy cow, including the excretion from an additional 1.4 support stock per adult cow (see Section 4.7.1);
- the annual stocking rate is 81 m^2/adult cow (50 adult cows/acre)
- half of the annual excretion occurs onto corrals rather than into flush-lanes;
- 3% of the N excreted in corral areas remains within the soil profile (Vaillant et al., 2009; see above);
- The relative total excretion rate, compared to 2005 levels, is 6.0% in 1945, 12.5% in 1960, 25.5% in 1975, and 44.6% in 1990 (see Section 4.8.2); this accounts for historic changes in both, the number of adult cows in TLB and the excretion rate per cow;

The resulting total nitrogen accumulation within Tulare Lake Basin corrals is 3,000 Mg N/yr (3,400 tons) for 2005 or 62,000 Mg N (68,000 tons) since 1945. This is equivalent to 200 mg total N/kg (0.02%) in the upper 2.0 m (6.7 ft) soil profile across all corrals and consistent with the above field measurements.

A third estimate can be obtained using the carbon storage measured in feedlot soils of southern Alberta (Miller et al., 2008): about 300 g C/kg in the upper 4 cm (1.7 in) of corral soil (manure layer), about 100 g C/kg of carbon in the next 10 cm of soil (black layer), and as much as 30 g C/kg in the immediate subsoil. If we assume a carbon (C) to nitrogen (N) ratio of 14 (Vaillant et al., 2009), and further assume that nitrogen concentrations from 60 cm (2 ft) to 2.0 m (6.7 ft) are 500 mg/kg (see above), the total nitrogen storage in the upper soil profile amounts to 24,000 kg N/ha (22,000 lbs/ac) or 202,000 Mg N (222,000 tons) for the Tulare Lake Basin.

A fourth estimate to consider is the average soil nitrogen storage of 37,000 kg N/ha (33,000 lbs/ac) measured in 20 – 50 year old feedlot corrals in Kansas (Vaillant et al., 2009, see above). This represents

Adminsitrative Record
Page 34420

the upper 2.0 m (6.7 ft) immediately below the corral. For the Tulare Lake Basin, the equivalent total corral soil nitrogen storage would be 304,000 Mg N (336,000 tons).

Applied to the corral areas of the Tulare Lake Basin, these estimates would suggest a range of less than 62,000 Mg N (68,000 tons) to as much as 304,000 Mg N (336,000 tons). This range represents less than 1.5% to 7.5% of the total amount of N excreted by dairy cattle in the Tulare Lake Basin between 1945 and 2005. For several reasons, feed lot soil nitrogen storage reported for Kansas and Southern Alberta sites are thought to be higher than in the TLB: the climate is wetter (higher precipitation) at significantly lower temperatures; in addition, feed lots are generally managed to keep the lot moist and minimize dust, while dairy corrals in the TLB are managed to stay dry. Higher temperatures and lower moisture could result in more rapid mineralization of nitrogen. Also, feed lots typically remove solids once per year or less, while many dairies in the TLB remove solids twice per year and more. Overall, a reasonable estimate of the total nitrogen storage in TLB corrals is from less than 1.5% to as much as 3% of the total amount of N excreted by dairy cattle.

Importantly, as Vaillant et al. (2009) pointed out, the conversion of corral areas to cropland has a high potential of mineralizing and mobilizing this locally very large amount of nitrogen that is currently immobilized in the immediate subsurface of corrals. Such conversion and subsequent groundwater contamination can be avoided by removing the top layer of corral areas scheduled for conversion and by distributing the soil removed as a soil amendment on cropland, within the framework of a nutrient management plan.

Spatially Distributed Nitrogen Loading Model for Groundwater Model Input: For groundwater modeling (Technical Report 4, Dylan et al., 2012), two input datasets are needed: the recharge rate and the associated nitrate concentration. We use the recharge rate estimated by VanderSchans et al. (2009) for modeling nitrate loading from corrals, and one-fifth of the recharge concentration that was estimated in that study to account for the approximately 5 times lower average nitrate found in Tulare Lake Basin corral monitoring wells, when compared to the dairies studied by VanderSchans et al. (2009): recharge = 305 mm/yr (12 in/yr), recharge nitrate concentration = 270 mg/L. This estimate equals a nitrogen loading rate of 183 kg N/ha/year (163 lbs/ac/yr) and produces shallow groundwater concentrations consistent with those reported by Brown, Vence & Associates (2004) and to those that we have found in groundwater separated from corrals by thick unsaturated zones. Using this approach, the total current corral N loading to groundwater in the TLB is 1,500 Mg/year (1,700 tons/yr), near the lower end of the range indicated in Table 27. Groundwater nitrate loading rates are assigned directly to individually mapped corrals.

For the simulation of historic nitrate loading from corrals, we used a simplified conceptual scenario of the historic development of corral loading: nitrate loading in corrals is assumed to have been constant since 1975. Prior to 1960, contributions from (much smaller) corral areas are assumed to have been negligibly small with the dairy herd mostly on pasture. Between 1960 and 1975, we assumed a linear increase in corral nitrate loading from zero to 1975 rates.

Adminsitrative Record
Page 34421

4.4 Review of N Loading Rates from Dairy Lagoons

Like corrals, most liquid manure lagoons in the Tulare Lake Basin are associated with dairy facilities. In the Salinas Valley, the Gallo feedlot near Gonzales is the only major confined animal facility and it maintains storage lagoons to collect corral runoff. The total area of dairy lagoons in the Tulare Lake Basin was mapped in the same manner as the open corral area: using a 2007 database of dairy addresses provided by the Regional Water Quality Control Board the latest Department of Water Resources land use surveys for Fresno, Kings, Tulare, and Kern County to locate all dairies, and 2009 aerial photos provided by the Department of Conservation Farmland Mapping and Monitoring Program (FMMP) as the basis for digitization of the lagoons (see previous section).

Based on the digitized map, we find that, within the Tulare Lake Basin, there are nearly 2,300 dairy lagoons that encompass 1,265 ha (3,126 acres). Nearly all or all of these lagoons were built prior to the issuance of the Dairy General Order in 2007. Prior to 2007, regulatory requirements for the construction of liquid manure lagoons were governed under California Water Code Title 27, which required that lagoons are lined with soil containing at least 10% clay (for a review of the guidelines, see Brown et al. 2003). The soil liners typically develop a thin, but highly effective sludge layer that controls the seepage rate from the lagoon (Ham 2002).

Liquid manure stored in lagoons varies widely in composition and contains nitrogen in the form of dissolved organic nitrogen, dissolved ammonium, organic nitrogen bound to suspended solids, and ammonium nitrogen bound to suspended solids. Pettygrove et al. (2010) report two studies showing the range of total nitrogen in liquid manure to vary from less than 50 mg N/L to over 2,000 mg N/L (typically as ammonium nitrogen and organic nitrogen) depending on the various sources contributing to lagoon manure including the amount of rainfall collected and irrigation water added to the lagoon. One study of nine dairy lagoons over two years reported median lagoon nitrogen concentrations ranging from 164 mg N/L to 645 mg N/L, averaging 360 mg N/L (ibid., see their Table 5), another reported average TKN (total Kjeldahl nitrogen, a measure of the sum of organic and ammonium nitrogen) in eight dairy lagoons over two years ranging from 410 to 1,010 mg/L, an average of 670 mg N/L (ibid., see their Table 4).

Manure lagoons in Kansas are constructed similarly to those in California, and have been extensively tested for percolation rates. In the Kansas study, Ham (2002) used a highly sensitive water balance approach to estimate net (average) water lost from manure lagoons to groundwater. Twenty lagoons were tested (14 swine sites, 5 cattle feedlots, and 1 dairy). Seepage rates varied within a relatively narrow range, given the wide variety of underlying soils, from 0.07 to 0.88 m/yr (0.23 – 2.9 ft/yr), and averaged 0.4 m/yr (1.3 ft/yr). The effective hydraulic conductivity of the sealing layer that develops at the bottom of lagoons was estimated to be 1.8×10^{-7} cm/s (2.2 in/yr). Total estimated nitrogen loading rates to the unsaturated zone varied from site to site, ranging from 400 kg/ha/yr to 5,000 kg/ha/yr (360 to 4,500 lbs/ac/yr).

Harter et al. (2002) provided an extensive review of existing literature on lagoon leaching and presented field data from five dairies in Stanislaus and Merced County. Their data were also applied in a

Adminsitrative Record
Page 34422

groundwater modeling study that suggests a recharge rate of at least 0.8 m/yr (2.7 ft/yr) with nitrate concentrations on the order of 450 mg/L and a loading rate of 807 kg/ha/yr (720 lbs/ac/yr) (VanderSchans et al. 2009)—values that are confirmed by similar findings in Ham's Kansas study (see above). In the Merced/Stanislaus County study (Harter et al. 2002), groundwater conditions are considered highly vulnerable and lagoons are vertically separated from groundwater by less than 3 m (10 ft) and often less than 1 m (3 ft). Groundwater immediately downgradient of lagoons was frequently found to contain more ammonium than nitrate. Average total nitrogen concentrations in lagoon monitoring wells (including nitrate as nitrogen) were similar to those found for corrals: 55 mg N/L (equivalent to 248 mg nitrate/L). Concentration varied significantly, ranging from less than 10 mg N/L to over 100 mg N/L (45 mg/L–450 mg/L nitrate equivalent), with a coefficient of variation of 0.44. Dissolved ammonium-N will typically be converted to nitrate-N (at a one-to-one ratio in terms of nitrogen mass) as ammonium-laden groundwater moves into more oxic zones. These levels are several times higher than the regulatory limit for drinking water.

In a more recent study of five Tulare Lake Basin dairies, where depth to groundwater is more than 15 m (50 ft) and in most cases exceeded 25 m (80 ft), we found significantly lower total nitrogen concentrations in monitoring wells specifically drilled to monitor first encountered groundwater downgradient of lagoons: average nitrate concentrations were 42 mg/L with a standard deviation of 49 mg/L (162 samples from seven well sites next to six lagoons). At individual sites, average nitrate concentrations over the 2.5 year monitoring period varied from less than 5 mg/L to 122 mg/L. Four of the seven sites averaged nitrate concentrations below the MCL of 45 mg/L. Two lagoon monitoring wells on new dairy sites, built less than 10 years ago, had average nitrate concentrations of 35 mg/L and 22 mg/L. Two monitoring wells next to a lagoon constructed over 40 years ago averaged 4 mg/L and 8 mg/L (as nitrate). No significant ammonium was detected, except during well construction at one site with an old lagoon, in a thin, perched groundwater layer approximately 7 m (20 ft) below ground surface. Subsequent sampling from this perched layer did not yield sufficient water for sample analysis.

Brown, Vence & Associates (2004) summarized information from ten dairies equipped with monitoring wells as part of a regulatory enforcement action. Six dairies with lagoon monitoring wells are located in the Tulare Lake Basin. Their average nitrate concentrations were 15, 22, 22, <40, 87, and 205 mg/L, a range similar to that observed in our Tulare Lake Basin groundwater monitoring study.

Much of the nitrogen leached from the lagoon is – at least temporarily – stored in the unsaturated zone. Ham (2002) showed that significant amounts of nitrogen are stored in the vadose zone within 1 to 2 m (3–7 ft) below the bottom of the lagoon at sites that have operated for approximately one decade. Typical ammonium-nitrogen concentrations in this upper layer were found to be on the order of 500 mg/kg. In the Tulare and Kings County dairy study (Harter et al. unpublished data), we found similarly high concentrations of ammonium-nitrogen, but also of nitrate-nitrogen (in the 200–500 mg/kg range) in the near-surface soil immediately adjacent to lagoons and to depths of 10 m (35 ft). With further depth, nitrate-nitrogen concentrations decreased to levels ranging from 20-50 mg/kg and ammonium-nitrogen concentrations decreased to below 1 mg/kg.

Adminsitrative Record
Page 34423

4.5 Groundwater N Loading from Liquid Manure Storage Lagoons: Methods and Results

Based on the work by VanderSchans et al. (2009), nitrate loading to groundwater under generally vulnerable conditions (shallow water table, sandy aquifer) is estimated to be on the order of 800 kg N/ha/yr (720 lbs/ac/yr). Applying this leaching rate to all current dairy storage lagoons in the Tulare Lake Basin, which occupy a total of 1,265 ha, the total contribution of nitrogen to groundwater would be on the order of 1,000 Mg N/yr (1,100 tons/yr) (Table 29).

The upper limit was obtained by assuming a recharge rate of 365 mm/year (1.2 acre-feet/acre/year) and a combined ammonium-N and nitrate-N concentration of 500 mg N/L (1,825 kg N/ha/yr = 1,628 lbs/ac/yr). An alternative upper limit is obtained by assuming a loading rate of 800 kg N/ha/yr (714 lb/ac/yr), obtained for an older lagoon overlying an aquifer less than 3 m below ground surface and considered highly vulnerable (VanderSchans et al. 2009).

Table 29. County by county summary of lagoon area and estimated largest possible N loading to groundwater from storage lagoons based on leaching rates and lagoon N concentration as well as an alternative largest possible N loading to groundwater based on a loading rate of 800 kg N/ha/yr (714 lb/ac/yr).

	Number of Dairies	Lagoon Area [ha]	Lagoon Area [acres]	N leached below lagoon – Upper Limit [Mg/yr] (tons/yr)	N leached below lagoon – Alternative Upper Limit [Mg/yr] (tons/yr)
Fresno Co.	108	131	325	480 (530)	110 (120)
Kings Co.	162	221	547	810 (890)	180 (200)
Tulare Co.	315	704	1,740	2600 (2,800)	560 (620)
Kern Co.	54	208	514	760 (840)	170 (180)
Total TLB	**639**	**1,265**	**3,126**	**4,600 (5,100)**	**1,000 (1,100)**

On the other hand, Ham's (2002) work suggests that leaching rates from manure lagoons can be as high as 0.88 m/yr (3 ft/yr). Pettygrove et al. (2010) reported typical California liquid manure nitrogen concentrations as high as 1,000 mg N/L. These worst-case numbers would yield an upper limit for N loading on the order of 8,800 kg N/ha/year (7,900 lbs/ac/yr) – an order of magnitude higher than the estimate by VanderSchans et al. (2009). Such high leaching rates are likely to be sporadic only and are not considered to occur at every facility in the Tulare Lake Basin. For a reasonably conservative (high) upper limit of lagoon loading, we assume a leaching rate of 0.73 m/yr (2.4 AF/ac/yr), twice the average leaching rate of 0.37 m/yr (1.2 ft/yr) from Ham (2002) and similar to the leaching rate of VanderSchans et al. (2009), and a lagoon nitrogen concentration of 500 mg N/L, corresponding to an intermediate value of the two studies reported in Pettygrove et al. (2010). These numbers would suggest an upper limit for the lagoon loading rate of 3,650 kg N/ha/yr (3,260 lbs/ac/yr) or 4,600 Mg N/yr (5,100 tons/yr) across all lagoons in the Tulare Lake Basin (Table 29).

Adminsitrative Record
Page 34424

The low nitrate (and ammonium) concentrations found in monitoring wells recently constructed in the TLB adjacent to relatively old manure storage lagoons (Harter et al. unpublished data) suggests that, under conditions of deep water table (> 20 m below ground surface), either significant denitrification occurs or lateral movement across perching layers distributes the nitrogen across a larger recharge area.

Overall, an estimated range of 200 – 2,000 Mg N/year (220 – 2,200 tons/yr) total groundwater loading from lagoons appears most reasonable under current conditions.

Current Storage of N in Lagoons: Lagoons, like corrals, may store significant amounts of nitrogen either in a sludge layer at the bottom of the lagoon or in the subsurface below the lagoon. The organic nitrogen stored in the sludge layer or the lagoon is potentially stored there for long periods of time (years to decades) while the lagoon is operating. The magnitude of total amount of nitrogen stored in and below lagoons for the long-term can be estimated from measured sludge concentrations and from measured total nitrogen concentrations immediately below lagoons (see above). We offer the following estimate. We assume the following values to obtain an approximate upper limit of N stored in the sludge layer: average sludge layer thickness of 1 m (3.3 ft) across 1,265 ha (3,126 ac) of lagoons, a sludge dry matter solids content of 10%, a solids density of 2 g/cm^3, a nitrogen concentration of 1,500 mg/kg (0.15%) sorbed to solids, and 500 mg/L dissolved in the liquid. This totals 8,000 Mg N (8,800 tons) that is semi-permanently in storage at the bottom of lagoons. For the immobile organic nitrogen storage immediately below the lagoon, we assume that most of that nitrogen is found in the first 2.0 m (6.7 ft) at concentrations of 500 mg/kg (0.05%, see above) with a soil density of 1.5 g/cm^3. The nitrogen storage below the lagoon then amounts to 15,000 kg N/ha (13,400 lbs/ac). The total nitrogen stored below the subsurface in TLB dairy lagoons amounts to 19,000 Mg N (21,000 tons). In total, we estimate that the semi-permanent storage of organic nitrogen below and within the bottom of dairy lagoons is on the order of 27,000 Mg N (30,000 tons), about 3/4 of one percent of the total estimated N excreted by dairy cattle since the late 1960s, when lagoons began to be constructed.

For the overall mass balance analysis of nitrogen fluxes in TLB dairies, removal of nitrogen into semi-permanent storage within or below lagoons is therefore considered negligible. However, as for corrals, the conversion of lagoons to irrigated land (agriculture or urban) bears the risk of mineralization and subsequent mobilization of this locally very intensive nitrogen pool, leading to subsequent groundwater contamination. Removal of the nitrogen-rich sludge and subsoil layers prior to land conversion is an important preventive step.

Spatially Distributed Nitrogen Loading Model for Groundwater Model Input: To simulate nitrogen loading to groundwater, we assume an average recharge rate of 365 mm/year and an average nitrate concentration of 225 mg/L (182.5 kg N/ha/year). The loading rate is identical to that used for corrals and it is at the lower end of the suggested range for lagoons above. The total N loading from lagoons, using these values, is 230 Mg N/y (250 tons/yr).

For computer simulations of historic loading to groundwater and subsequent fate of groundwater nitrate, we assume that lagoon loading to groundwater was constant in time since 1970, despite the increasing cattle numbers. Prior to 1970, we assume that no lagoons existed in the Tulare Lake Basin.

Adminsitrative Record
Page 34425

Prior to 1970 and the passing of the Porter-Cologne Act in 1968, few lagoons existed, and many of the animals grazed on pasture for significant portions of the year.

4.6 Review of Nitrate Loading Rates from Irrigated Crop Fields with Manure Applications

Dairies in the Tulare Lake Basin, whether they are drylot or freestall operations, no longer maintain significant acreages of irrigated pasture land for cattle grazing (a practice common prior to the 1970s). Instead, animals are confined to corrals and freestalls, while agricultural land surrounding the animal production facility is used for the production of forage crops other than pasture. The most common forages in the Tulare Lake Basin are alfalfa (*Medicago sativa*), corn (*Zea mays*), sudan grass (*Sorghum bicolor* subsp. *drummondii*), and winter grains including triticale (*Triticale hexaploide*), oats (*Avena sativa*), wheat (*Triticum aetivum*), and barley (*Hordeum vulgare*). Dairies also manage vineyards, cotton, and other crops, which may be used for some (limited) manure application.

Harter et al. (2002, their Table 1) provided a review of existing data on nitrate leaching from manure cropland application areas. Nitrate concentration in leachate below the root zone and in domestic wells nearby such land application areas varied widely, from below detection limits to as much as five to eight times above the drinking water limit. In the same publication, data from monitoring wells on five dairy facilities, specifically downgradient of manure-treated forage fields were reported. The facilities were all located in the north-central San Joaquin Valley (Merced and Stanislaus Counties), on coarse-textured soils (sandy loams) with a shallow groundwater table (less than 5 m below ground surface). The average monitoring well nitrate concentration was six times above the drinking water limit (279 mg/L), with individual measurements varying widely (coefficient of variation of over 50%). Based on these concentrations and estimated recharge rates, nitrogen losses from manured fields to groundwater were estimated to be on the order of 280 kg N/ha/yr. VanderSchans et al. (2009), using a modeling approach that linked field recharge nitrogen fluxes to measured groundwater nitrate monitoring data on two of these dairies, estimated that nitrogen losses from manured fields ranged from 211 kg N/ha/yr (188 lbs/ac/yr) to over 700 kg N/ha/yr (630 lbs/ac/yr) with an average of 486 kg N/ha/yr (434 lbs/ac/yr). Values near the lower end of the above range were generally achieved under relatively strict nutrient management practices (see Technical Report 3, Dzurella et al. 2012) whereas the average and higher values for nitrate-nitrogen losses to groundwater represent traditional manure management practices.

Significantly lower nitrate concentrations were measured in an ongoing research project (Harter et al. unpublished data) in the Tulare Lake Basin: monitoring wells were installed to measure groundwater quality in the first encountered groundwater (not including aquitards) on five dairies in Kings and Tulare County, with water table depths of approximately 15 m (50 ft) at one dairy, and approximately 30 m (100 ft) at the other four dairies. Nitrate concentrations were measured eight times per year over a 2.5 year period in eight monitoring wells located downgradient of long-term manured cropland typically planted with corn and winter grain (often in a multi-year rotation with alfalfa), a similar land use to those dairies investigated by Harter et al. (2002). Average nitrate concentration was approximately 130 mg/L, three times the level of the MCL, and approximately half of the average nitrate concentration

Adminsitrative Record
Page 34426

reported for the northern San Joaquin Valley dairy study by Harter et al. (2002). Between monitoring wells, long-term average well nitrate concentrations typically ranged from 70 mg/L to 170 mg/L. One well, not included in the above average, consistently showed nitrate levels exceeding 300 mg/L, but the source of that water was not clear (ibid.).

Measured concentrations reported by Harter et al. (2002) and those modeled by VanderSchans et al. (2009) were found to be consistent with field mass balance estimates of nitrate leaching below the root zone. Groundwater nitrate leaching rates estimated from groundwater models that were calibrated to measured monitoring well nitrate concentrations compared favorably to nitrate leaching estimates obtained by closure of the field scale mass balance. In other words, the groundwater nitrate-nitrogen loading estimated from monitoring wells and groundwater flow dynamics was consistent with groundwater nitrate-nitrogen loading rates estimated from the difference between annual nitrogen application rates (inorganic fertilizer, manure nitrogen, atmospheric deposition, irrigation water) and the sum of crop nitrogen removal and atmospheric losses (Harter et al. 2002, VanderSchans et al. 2009).

This previous work showed that such a mass balance approach, while not exact, provides a valuable approximation of groundwater nitrate losses from manure applications. Over the past decade, this has led to the introduction of manure management practices that directly account for the nitrogen-fertilizer value of manure by measuring the amount and nitrogen-content of manure applied to fields, by timing the manure applications, and by including manure into the overall field fertilization schedule. The 2007 Dairy General Order issued by the Central Valley Regional Water Quality Control Board requires dairies to fully account for the nitrogen content of land applied manure and other nitrogen sources, while meeting a nitrogen application ratio (ratio of total nitrogen applied to total nitrogen removed in the harvest) of 140%–165%.

Historically—prior to the 2007 Dairy General Order—manure (liquid or solid) was typically applied during the spring and during the fall fallow seasons between harvest of summer/winter crops and planting of winter/summer crops on fields with corn and winter grains. Alfalfa, a leguminous crop capable of fixing nitrogen directly from atmospheric sources, may receive some solid manure prior to planting or after the last cutting in the fall, but generally receives little or no manure water application and only small amounts of fertilizer application. It is an important forage and widely grown on dairy farms. Farms also apply manure (mostly manure solids, but also manure liquids) to cotton fields, orchards, and vineyards, albeit in relatively moderate amounts.

4.7 Dairy Manure N Applications to Cropland

4.7.1 Total Amount of N Excreted at Each Dairy.

We had two data sources available to estimate the total amount of N excreted, some of which is then land applied. We initially used a table obtained from the U.S. EPA Region 9 ("Central Valley Dairies.dbf," courtesy of Don Hodge, U.S. EPA Region 9),[26] which contains data reported by dairy owners and

[26] http://www.epa.gov/region9/ag/dairy/locations.html

Adminsitrative Record
Page 34427

collected by RB5 during 2005. This database is referred to here as the "EPA 2005 dairy database." The EPA 2005 dairy database lists 621 individual dairies in Fresno, Kings, Tulare, and Kern County and – according to Central Valley RWQCB staff – represents 2004-2006 conditions. For each dairy, the list also provides the number of milking cows, the number of dry cows, and an estimated number of support cattle. The number of support cattle was set equal to 117% of the number of adult cows. We later obtained a similar table from RB5, which listed 639 dairies in the study area with 2007, 2008, and 2009 animal numbers and 2007 cropland acreage for each dairy. We refer to this table as the "RB5 2010 dairy database." Ultimately, we chose the latter database to estimate nitrogen excretion in manure on individual dairies, within each county, and study area wide.

The total amount of N excreted from cattle on each dairy identified in the RB5 2010 dairy database (Table 30) was estimated by assuming that the daily N excretion from lactating cows and dry cows is 462 g N d^{-1} and 195 g N d^{-1}, respectively (UC Committee of Consultants – Harter, 2007). This amounts to 153 kg N/yr (336 lbs/yr) excreted per adult cow, consistent with Pettygrove et al. (2010). To estimate the N excretion from support stock, we used the ratios in Table 1 of Pettygrove et al. (2010), which suggest that 25 kg N/yr (56 lbs/yr) are excreted by support stock for every adult cow, which – according to their Table 1 – excretes 148 kg N/yr (326 lbs/yr). Their computation was based on the assumption that, on average, each dairy has 0.17 calves (0-6 months) and 0.5 heifers (6 months to 24 months) per adult cow. We adopted the EPA estimate of 1.4 support stock per milk cow (lactating cows[27]) or 1.17 support stock per adult dairy cow, and scaled the Pettygrove et al. (2010) support stock excretion rate to 45 kg N/yr (101 lbs/yr) for the 1.17 support stock per adult dairy cow. Per adult cow, and including support stock, the total excretion rate is therefore 198 kg N/yr (437 lbs/yr).

In total, 202 Gg N/yr (223,000 tons/yr) are excreted by dairy cattle in the TLB (Table 31). More than half of the excreted manure is generated in Tulare County. The fate of this nitrogen, and how we estimate the breakdown between the three pathways (i.e., atmospheric losses, exported (sold) manure, and land application of manure within a dairy), is explained further below.

Table 30. Number of milking cows, dry cows, and support stock (calves, heifers, etc.) in the Tulare Lake Basin study area, and the cropland acreage associated with dairies, total and by county (based on data obtained from RB5, representing the most recent number of mature cows reported between 2007-2009, and in the text referred to as "RB5 2010 dairy database").

	Lactating Cows	Dry Cows	Support Stock	Cropland Acreage 2007 [ha (acres)]
Fresno County	110,793	21,795	155,110	25,067 (61,943)
Kings County	148,486	29,210	207,880	22,621 (55,897)
Tulare County	455,987	89,702	638,381	60,760 (150,140)
Kern County	137,147	26,980	192,006	19,059 (47,097)
Tulare Lake Basin	**852,412**	**167,688**	**1,193,377**	**127,507 (315,077)**

[27] In the EPA database, lactating cows are referred to as "milking cows" to which "dry cows" are added to obtain the total number of "adult cows"

Adminsitrative Record
Page 34428

Table 31. Total amount of manure nitrogen excreted by dairy cattle in each TLB county, atmospheric nitrogen losses from manure, manure nitrogen sold off dairy, and manure nitrogen land applied within dairies. The average dairy manure N loading rate is the arithmetic average across individual dairy's ratio of direct applied manure N [kg/yr] to cropland area [ha]. The countywide dairy manure loading rate is the county total direct applied manure N [kg/yr] divided by the county total dairy cropland [ha]. See text for further explanation.

	Manure N Excreted [Mg/yr]	Atmospheric Losses of N [Mg/yr]	Maximum Limit, Manure N Export [Mg/yr]	Minimum Limit, Direct Applied Manure N within Dairies [Mg/yr]	Minimum Limit, Average Dairy Manure N Loading Rate [kg/ha/yr]	Minimum Limit, Countywide Dairy Manure N Loading Rate [kg/ha/yr]
Fresno Co.	26,303	9,995	12,707	3,601	371	144
Kings Co.	35,252	13,396	12,913	8,943	521	395
Tulare Co.	108,256	41,137	41,960	25,159	546	414
Kern Co.	32,560	12,373	9,867	10,321	944	541
Tulare L.B.	**202,371**	**76,901**	**77,446**	**48,024**	**596**	**377**

4.7.2 Historic Dairy Cattle N Excretion Rates

Historically, the total nitrogen excretion in the TLB has been much less than the 2005 levels of N excretion. The total number of dairy cows has steadily increased over the past 60 years from nearly 110,000 milk cows in 1950 to nearly 880,000 milk cows in 2007. Also, the amount of milk produced per milk cow has tripled over the past 60 years, from a state-average of 7,150 lbs/yr in 1945[28] to 22,440 lbs/yr in 2007.[29] Over the same time period, the relative nitrogen content of milk, compared to the cow's feed intake has risen from approximately 21% in 1945 to 25% in 2005. Thus, the manure output per milk cow has increased somewhat less than three times between 1945 and 2005.

For purposes of estimating historic dairy N excretion rates and the amount of manure nitrogen used for cropland application, we estimate excretion rates from USDA agricultural census data for California. The ratios of historic excretion rates to the 2005 excretion rate estimated from these census data is then used to scale the excretion rate developed in Section 4.8.1 back to 1945, 1960, 1975, and 1990 (Table 32).

[28] http://usda01.library.cornell.edu/usda/nass/MilkProdDa//1940s/1946/MilkProdDa-02-15-1946.pdf
[29] http://usda01.library.cornell.edu/usda/nass/SB988/sb1022.pdf

Adminsitrative Record
Page 34429

Table 32. Estimate of historical manure nitrogen excretion rates in the Tulare Lake Basin based on USDA NASS California census data on milk production per head of cattle (hd) and total number of milk cows in the five study area counties.

Year	Milk Production [kg/hd/yr] (lbs/hd/yr)	Milk Nitrogen [kg N/hd/yr] (lbs/hd/yr)	Milk : Feed Intake Nitrogen Ratio	Excretion Rate [g N/milk cow/d] (lbs/milk cow/day	Number of Adult Dairy Cows in the TLB + SV	Total Excretion Ratio, relative to 2005	Total N Excretion in the TLB & SV [Gg N/yr] (tons N/yr)
1945	3,243 (7,150)	17 (37)	21%	173 (0.38)	141,124	0.060	12 (13,000)
1960	4,432 (9,770)	23 (51)	22%	223 (0.49)	225,510	0.124	24 (26,000)
1975	6,154 (13,566)	32 (70)	23%	292 (0.64)	352,089	0.255	49 (54,000)
1990	8,372 (18,456)	43 (95)	24%	376 (0.83)	478,668	0.446	86 (95,000)
2005	9,709 (21,404)	50 (111)	25%	413 (0.91)	977,887	1	194 (214,000)
2020	11,263 (24,831)	58 (128)	26%	432 (0.95)	977,887	1.129	219 (241,000)
2035	13,431 (29,612)	69 (153)	27%	489 (1.08)	977,887	1.280	248 (273,000)
2050	14,986 (33,039)	78 (171)	28%	520 (1.15)	977,887	1.360	264 (291,000)

Notes: The increase in milk N to feed N intake ratios is estimated to fit 1973 Committee of Consultant N excretion rate for California and approximate historic conditions. The number of cows in 1945 was assumed to be identical to the 1950 census data. The number of cows in 1960, 1975, and 1990 were estimated by linear interpolation of the 1950 and 1992 national agricultural census data. Similarly, the 2005 number of cows was estimated by linear interpolation of the 2002 and 2007 national agricultural census data. The historical total excretion rates for the TLB are based on the 2005 estimated N excretion and the N excretion ratio.

Since 1945, the total nitrogen excretion from dairy animals in the Tulare Lake Basin (with a very small fraction in the Salinas Valley, see Table 31), has risen exponentially, doubling every 15 years. Until the 1960s, much of the nitrogen excretion in the study area is assumed to have occurred on irrigated pasture where plant uptake rates were absorbed most of the manure nitrogen entering the root zone. However, since the early 1970s, liquid and solid manure is collected and land applied on crops. Since then, the amount of nitrogen that needs to be land applied – in direct proportion to the amount of nitrogen excreted - has increased five-fold.

Adminsitrative Record
Page 34430

4.7.3 Atmospheric nitrogen losses from manure prior to land application

Atmospheric losses of nitrogen from the total mass of nitrogen excreted are assumed to be 38%, which is based on a 2003 EPA draft report on ammonia emissions from manure (EPA 2003). This estimate is near the upper end of the range of atmospheric losses provided by the University of California Committee of Consultants (Harter 2007), which suggested that these losses may range from 20% to 40% of excreted N. We use the higher number to account for the fact that a significant number of dairies in the Tulare Lake Basin are drylot dairies, where atmospheric N losses tend to be higher than on freestall dairies.

Across the study area, 77 Gg N/year (85,000 short-tons/year) are lost to the atmosphere. The 38% loss rate is assumed constant across all dairies. Hence, more than half of all atmospheric losses occur in Tulare County, which houses over half the dairy animals in the TLB (Table 31).

4.7.4 Distribution of cropland applied manure nitrogen

For the years prior to the 2007 Dairy General Order, little is known about the actual distribution of cropland applied manure nitrogen including:

- The distribution across crops (crop categories)
- The distribution between on-dairy cropland and off-dairy cropland
- The distribution within county of origin and outside of the county of origin
- The distribution of synthetic fertilizer and manure nitrogen to meet applied fertilizer needs (discussed in Section 3 of this Technical Report)

Most manure is land applied to field crops, particularly corn, which – on dairies - is often double-cropped with winter grain. Manure is also likely being applied to grain and hay crops. Dried or composted manure solids may be applied as soil amendment to other crops including perennial crops. Limited amounts of manure are applied to alfalfa, typically before seeding, and occasionally at the end of the season.

Farmer's in the SV apply approximately 10 Mg/ha (~4 tons/acre) of compost (not necessarily dairy manure) once every other year. At 60% dry matter content and 2% nitrogen content, this is equivalent to approximately 60 kg N/ha/yr (50 lbs/ac/year). Furthermore, a composter in the TLB shared that he typically delivers compost over distances of a few to several tens of kilometers (few to tens of miles).

The overall exportation of manure from dairies to cropland outside dairy operated cropland can be a significant proportion of the nitrogen generated on the dairy, but typically is much smaller than the amount of manure nitrogen retained on dairies. Most of the manure exported, due to transportation cost, does not leave the county of origin and even less manure nitrogen leaves the study area.

Until recently (including the 2005 period), manure has been applied effectively as a soil amendment, in addition to synthetic fertilizer. Under the 2007 Dairy General Order, dairies are required to account for

Adminsitrative Record
Page 34431

both synthetic and manure nitrogen as well as other sources of nitrogen (e.g., irrigation water) in their nutrient management planning.

While future research of the dairy nutrient management data collected by RB5 will likely provide more detail on the distribution of manure, at least within dairy cropland, here we employed simplified manure distribution scenarios. These scenarios are designed to reflect the overall, very qualitative nature of what is known about the distribution of manure. The objective in designing these scenarios is to provide several scenarios for the likely quantitative distribution of manure in cropland application that can illustrate the potential range in groundwater nitrate loading and that can be used as more quantitative information on the distribution of manure becomes available.

Scenario for Crop-Group and County Analysis: For the mass balance analysis of crop- and county level groundwater nitrate loading based on land areas reported by the county agricultural commissioners (see Section 1 of this Technical Report), we make no distinction between manure land applied on dairies and manure land applied outside of dairies. We assume that all manure generated within a county is land applied within the county. Two-thirds of dairy manure is assumed to be applied to field crops and one-third of dairy manure is applied to grain and hay crops. In corn and other field crops (CAML classes 600, 602 to 612, but not including 601-cotton, see Appendix Table 2), 50% of crop nitrogen requirements are assumed to be met with synthetic fertilizer, in small grain and hay crops 90% of their crop nitrogen requirements are assumed to be met by synthetic fertilizer. For the mass balance analysis and to derive groundwater nitrate loading, the manure nitrogen available for cropland application is added to these synthetic fertilizer nitrogen applications for these two crop groups only.

CAML-based Analysis with the Groundwater Nitrate Loading Model (GNLM) – Scenarios A-D: For the CAML-based analysis, where the mass balance is computed on a field-by-field basis according to the CAML landuse maps (see Sections 1, 2, and 3 of this Technical Report), the amount of manure exported from dairies must be specified, and the specific crops receiving land applied manure must be specified as input to GNLM.

Within dairies, GNLM operationally assumes that manure nitrogen is applied, primarily as liquid manure, to the following CAML land use categories: field crops (600), cotton (601), sugar beets (605), corn (606), grain sorghum (607), sudan (9608), sunflowers (612), grain and hay (700), barley (701), wheat (702), oats (703), pasture (1600), and mixed pasture (1603). The numbers in parentheses refer to the CAML land use categories (see Section 3 of this Technical Report and Appendix Table 2).

Outside of dairies, and on dairy cropland other than the previously listed crop categories, exported manure (assumed to be dry manure or composted manure only) is distributed across all crop categories identified in CAML. For the amount of manure that is distributed off-dairies, we developed six hypothetical scenarios with the objectives

1. to broadly bracket the potential export (past, current, and future) of manure nitrogen from dairies (scenario D versus other scenarios) and
2. to broadly bracket the potential distribution of exported manure nitrogen between counties, study area, and areas outside of the study area (scenarios A-C).

Adminsitrative Record
Page 34432

The scenarios are:

- "Scenario A": Manure exported by dairies does not affect the typical N fertilization rates (Figure 4) on non-dairy cropland within the study area, after accounting for the combined synthetic and organic sources of nitrogen fertilizer applied to non-dairy cropland. This is a hypothetical (future) scenario representing the possibility that manure exported from dairies
    - is applied to non-dairy cropland as part of the typical N fertilization rates,
    - is transported to areas completely outside the study area, possibly after some processing,
    - is intentionally processed and lost to the atmosphere,
    - or any combination thereof.
- "Scenario B (by county)": Half of the manure exported by dairies is applied as soil amendment on non-dairy cropland within the county of origin. The other half of the exported manure has the same fate as listed under "Scenario A". The manure exported by dairies for soil amendment within each county is distributed in direct proportion and in addition to the typical N fertilization needs of crops within that county (manure applied as soil amendment does not leave the county). This scenario represents the mid-point between "Scenario A" and "Scenario C (by county)".
- "Scenario B (study area)": Half of the manure exported by dairies is applied as soil amendment on non-dairy cropland within the study area (not restricted to the county of origin). The other half of the exported manure has the same fate as listed under "Scenario A". The manure exported by dairies for soil amendment in the study area is distributed across all non-dairy cropland in the study area in direct proportion and in addition to their typical N application needs. This scenario represents the mid-point between "Scenario A" and "Scenario C (by study area)".
- "Scenario C (by county)": All manure exported by dairies is applied as soil amendment on non-dairy cropland within the same county. The total manure exported by dairies within each county is distributed in direct proportion and in addition to the typical N application rates of crops within that county (manure does not leave the county).
- "Scenario C (study area)": All manure exported by dairies is applied as soil amendment within the study area (not restricted to the county of origin), and the total manure exported by all dairies in the study area is distributed across all non-dairy cropland in the study area in direct proportion and in addition to their typical N fertilization rates.
- "Scenario D": No manure is exported by dairies. All manure is land applied on applicable forage crops within the dairy. Note that, groundwater nitrate loading on non-dairy cropland is therefore identical to that simulated in Scenario A. Groundwater nitrate loading on dairy

Adminsitrative Record
Page 34433

cropland receiving manure is significantly higher under this Scenario than under the export scenarios.

Historically, for simulation purposes, manure N exports are assumed to be negligible (under all scenarios) prior to 1980, increase linearly from 0% to 38% between 1980 and 2005, and stay constant at 38% after 2005 (scenarios "A", "B", and "C").

In the current version of GNLM (Section 2.6), the fraction of manure nitrogen exported is an arbitrary percentage set to 38% basin-wide, but varying from county to county in proportion to the ACR category "manure sold". While Scenario D brackets manure export at the lowest end (zero), a 38% export ratio brackets actual export ratios at the very high end (although a few individual dairies may export more). County- and study area specific ratios of hypothetical fractions of exported N are shown in Table 33

Table 33. Operational model on the fate of excreted nitrogen, by county.

Region	% N Excreted	% Atmospheric Losses before Land Application	% N Land Applied on Dairy Cropland	% N Land Applied Offsite
Fresno Co.	100	38.0	13.7	48.3
Kings Co.	100	38.0	25.4	36.6
Tulare Co.	100	38.0	23.2	38.8
Kern Co.	100	38.0	31.7	30.3
TLB	100	38.0	24.0	38.0

With 202 Gg N/yr (223,000 tons/yr) excreted, atmospheric losses prior to land application amount to 77 Gg N/yr (85,000 tons/yr). With this amount of atmospheric N losses, the N exports in 2005 are no more than 77 Gg N/yr (85,000 tons/yr) and the amount of manure nitrogen applied to cropland within dairies is at least 48 Gg N/yr (53,000 tons/yr, Scenarios "A"-"C"), but not exceeding 125 Gg N/yr (138,000 tons/yr, Scenario "D") (see Table 31).

In GNLM, all manure applications, within and outside of dairies, are distributed proportional to the nitrogen application needs of the particular crop grown on a specific field (see Section 2 of this Technical Report). Briefly, for the distribution of manure N within a dairy on the specific crops listed above (field crops, corn, etc.), we use the RB5 2010 dairy database to estimate the amount of manure N excreted on an individual dairy, and compute the scenario-specific amount of manure N applied to cropland within that individual dairy. The cropland associated with an individual dairy is obtained from the reported assessor parcel numbers and the CAML landuse map (see next section). At least 50%, but no more than 100% of the applied nitrogen need is met by synthetic fertilizer N, regardless of crop type. The remaining applied nitrogen needs are assumed to come from manure N. On many dairies, the total

Adminsitrative Record
Page 34434

amount of manure N applied exceeds 50% of the applied nitrogen needs and is therefore in excess of the applied nitrogen need.

The above estimation of the amount of manure N available for a) land application within the dairy, b) for application on cropland across the study area, and c) volatilized to the atmosphere is associated with significant uncertainties. Estimates of manure exports from dairies could be further improved by compiling the dairy data collected by RB5 under the 2007 Dairy General Order. Atmospheric N losses, assumed to be 38%, also are a significant source of uncertainty in estimating the amount of manure N land applied on dairies or exported.

Historic Simulation of Manure Nitrogen Application to Cropland: For the historic simulations of spatially distributed nitrogen applications to cropland, we assume that until the late 1960s, manure nitrogen is not land applied but excreted on irrigated pasture. Hence, for modeling purposes, dairy manure from any dairy application source or location (cropland, lagoon, or corral) is assumed to not contribute to groundwater nitrate loading prior to the 1970s. In the 1970s, land application of manure is assumed to be limited to cropland belonging to a dairy. No manure is exported from dairy-owned land prior to 1980. After 1980, exports of manure (Scenarios A-C) are assumed to gradually increase. GNML assumes that the full amount of export in Scenarios A-C is only reached in 2005. Between 1980 and 2005, the fraction of manure exported from dairies increases linearly from zero to the amount specified for 2005. In Scenario D, manure never leaves the dairy. All Scenarios are simulated through 2050.

4.7.5 Manure N cropland application on dairies: Identifying dairy cropland

For the CAML-based field-by-field analysis in GNML, a link between individual dairies and their associated fields must be created in a database to approximate the manure distribution within a dairy facility according to the number of cows in the dairy. One possible approach, taken previously by a pilot study for CV-SALTS, is to use a geographic information system (GIS) analysis that distributes manure nitrogen to cropland at agronomic rates, and selects a sufficiently large area of cropland. This approach assumes *a priori* that manure is distributed at agronomic rates. Another approach for identifying fields receiving manure is to consider the total acreage of dairy land identified, by dairy, in the RB5 2010 dairy database and identify the equivalent amount of cropland in the land use database described in Section 3. A minor shortcoming of this method is that the cropland areas identified in this way may include areas that are in fact facility and other non-crop acreage.

Here, we choose a third approach, based on the assessor parcel numbers (APNs) identified by dairies in their facility assessment of 2007. As part of the RB5 Dairy General Order, each dairy operator was required to submit a list of APNs that were either part of the facility or cropland potentially receiving manure. From RB5, we obtained a database that listed dairy name, and – for each dairy – the APNs of all parcels considered to be "facility" and of all parcels considered to be "cropland". We refer to this database, henceforth, as the "RB5 APN database". The RB5 APN database did not list address, or any other georeferences associated with the dairy name, only the county location. The dairy names in the RB5 APN database did not all match the dairy names in the RB5 2010 dairy database: Matches were found for 495 of 639 dairies. Within each county, all parcels in the RB5 APN database with unmatched

Adminsitrative Record
Page 34435

dairy names were combined into a single large virtual dairy, which was associated with the combined number of animals (and their manure) of those dairies in the RB5 2010 dairy database that were not matched with the RB5 APN database. Thus, we account for the total number of animals in the RB5 dairy database as well as the total dairy land area identified by APN numbers in the RB5 APN database.

In total, approximately one in eight cows (13%) is assigned to facilities for which APN parcels are unknown and approximately 5% of the APN parcels area are assigned to a dairy for which the animal numbers are unknown. For simulation purposes, this means that manure N application on the unassigned land areas is 2.5 times higher than the average on land with assigned animals (Table 34).

Table 34. Matching of RB5 2010 Dairy database adult dairy animal numbers for 2007-2009 with the RB5 APN database of 2007 reported land area of dairy facilities and cropland for land application via dairy name. The table provides the number of adult cows and the total associated acreage of the match.

	Number of cows with assigned APN land	Number of cows with unassigned APN land	APN land area with cows assigned [ha (acres)]		APN land area with no cows assigned [ha (acres)]	
Fresno	118,964	13,624	19,808	(48,946)	398	(984)
Kings	150,452	27,244	17,304	(42,759)	821	(2,030)
Tulare	482,289	63,400	61,095	(150,967)	3,944	(9,745)
Kern	137,834	26,293	19,736	(48,768)	1,456	(3,598)
Total TLB	889,539	130,561	117,943	(291,439)	6,620	(16,357)

Table 35 compares the total county-wide land area identified by the APN database and compares it against the total county wide land area reported in the RB5 dairy database (which does not identify, whether the reported land area is facility or cropland acreage). For the entire Tulare Lake Basin, the dairy land area identified by the RB5 APN database is 98% of the total land area listed (as total acreage per dairy) in the RB5 2010 dairy database.

We use the RB5 APN database to identify dairy land parcels on a digital map using the counties' APN GIS data layers. We thus create a digital map corresponding to the RB5 APN database (dairy APN GIS layer). Using GIS-based spatial analysis, we can overlay the dairy APN GIS layer with the CAML land use GIS layer described in Section 3. The spatial analysis within GIS allows us to identify the crop mix within the land area identified by dairy APNs; and it allows us to simulate the proper crops to which to apply manure within the area identified by a dairy as potentially receiving manure applications.

Adminsitrative Record
Page 34436

Table 35. Cropland and facilities acreage of assessor parcel numbers (APNs) reported by dairies to the RB5 (data provided courtesy of RB5, 2011). Also shown are the acreages reported in the RB5 2010 dairy database as 2007 conditions, for comparison. The last column is the ratio of the land area reported in the RB5 APN database and the land area reported in the RB 2010 dairy database.

	APN cropland (acres)	APN facilities (acres)	APN facilities/ cropland (acres)	APN total (acres)	RB5 2010 dairy database (acres)	APN total/ RB5 dairy (%)
Fresno County	36,771	13,159	-	49,930	61,943	81%
Kings County	41,329	3,460	-	44,789	55,897	80%
Tulare County	151,113	9,599	-	160,712	150,140	107%
Kern County	41,256	3,229	7,881	52,366	47,097	111%
Tulare Lake Basin	**270,469**	**29,447**	**7,881**	**307,796**	**315,077**	**98%**

For each county we summed the CAML land use areas within the areas identified as dairy APNs and computed the distribution of crops and other land uses within dairies: Approximately one-quarter (26%) of the acreage identified with the APNs is in alfalfa land use in CAML, another 57% of the APNs identified correspond to field crops, grain and corn crops, or pasture crops. Alfalfa is generally rotated with field crops, grain crops, and corn. The ratio of alfalfa acreage to field, corn, and grain crop acreage is not unreasonable. Manure is typically not applied to fields while they grow alfalfa except an unknown amount of solids sometimes applied prior to planting or after the last cutting in the fall. Two land uses that are unlikely to receive significant amounts of manure include farm structures (6%) and vineyards (3%), which make up most of the remaining land use identified by the APNs (Table 36).

For the field-by-field nitrogen mass balance computations in GNLM, we assume that non-exported manure is applied only to land within dairies ("direct manure applications"), as identified by the RB5 APN database, and within that area only to the following CAML land use categories: field crops (600), cotton (601), sugar beets (605), corn (606), grain sorghum (607), sudan (9608), sunflowers (612), grain and hay (700), barley (701), wheat (702), oats (703), pasture (1600), and mixed pasture (1603). The numbers in parentheses refer to the CAML land use categories (see Section 3).

The simulation process described here spatially allocates cropland specifically used for manure applications and associates that land with a dairy that has a known number of animals (see above). For the historic simulation of nitrogen budgets, we lack a similar knowledge base, but would like to use the same simulation approach. For simplicity, we assume that the land identified as currently belonging to a dairy, using the RB5 APN database, remained unchanged since 1975 (the first period for which land application of manure was considered to be significant).

Adminsitrative Record
Page 34437

To be consistent with the overall historic nitrogen fluxes, the number of animals associated with each facility was scaled according to the total number of animal excretion reported for the TLB historically (Table 27). Hence, in 1950, each facility is assumed to have less than one-tenth of the number of animal excretion than it has today (Table 27). The land use and crop mix within each dairy (within its associated parcels) change over time according to the historic land use simulations described in Section 3. The list of specific crops, to which on-dairy, direct manure applications were assigned, remains constant in time. But the simulated (back-casted) land use will vary over time (Section 3). In any given period, the actual parcels receiving manure directly on the dairy are reassigned according to that period's landuse distribution among the APN parcels of a specific dairy.

Table 36. Total land area of cropland and other land uses within land parcels managed by a dairy. The land area was computed by an overlay of the land area self-identified by dairies as APNs of land receiving manure, and reported to RB5, with GIS processed data on the crop type and landuse distribution in CAML (see Section 3). Results are obtained from a GIS spatial analysis of an overlay of APN identified dairy "cropland" parcels with the CAML land use map (see Section 3).

CAML Land Use within Land Parcels Managed by Dairies	Area Across All Dairy Parcels in the TLB [ha (acres)]	% of Total Dairy Land Area
Alfalfa	27,315 (67,498)	25.6
Farm structures	6,019 (14,873)	5.64
Field crops (including cotton)	16,211 (40,059)	15.19
Grains and corn	43,740 (108,085)	41
Idle cropland	292 (722)	0.27
Natural vegetation	4,035 (9,970)	3.78
Other crops	31 (77)	0.03
Pasture	626 (1,547)	0.59
Tree crops	3,624 (8,954)	3.4
Urban	1,047 (2,587)	0.98
Vegetable crops	817 (2,044)	0.78
Vineyards	2,927 (7,234)	2.74
TOTAL	**106,684 (263,650)**	**100**
Field + grain + pasture (typical crops used for manure applications)	**60,577 (149,691)**	**57**

A final note of caution: The data used as input for this land allocation simulation, on a field by field basis, are subject to potential errors. For example, it is likely that both, those parcels receiving manure and the APN identification of these parcels, were occasionally misidentified or that data were reported incorrectly. Parcels receiving manure may also change from year to year. It is unclear, whether the data

Adminsitrative Record
Page 34438

provided by an individual dairy facility represent the acreage used in 2007 only or the complete acreage of all crops typically used for manure applications, even if only on a rotating basis. Furthermore, the CAML land use cover described in Section 3 and used for the spatially distributed, field-by-field nitrogen loading mass balance analysis, represents only a snapshot of cropping conditions that are often transient from year to year and may not be the actual cropping conditions of 2007.

Hence, the simulation process described in this section can only be a much simplified conceptual approximation of complex processes in space and time involving people and land. The complexity of these processes is difficult to capture for current conditions, let alone under historic conditions, for which data cannot be collected retroactively. We emphasize that our approach is not designed to predict historic and current loading rates with high accuracy for each field or even for each individual dairy. Instead, our approach is designed to recreate the approximate conditions across all dairies in the study area, while preserving the variety of crops grown, and the variability in management practices between dairies, as expressed by animal numbers and land base. The simulation algorithm provides overall consistency in the conceptual approach, given the lack of historic landuse and land ownership data for more detailed modeling input.

4.8 Review of N Loading from Non-Dairy Animal Farming Operations

Besides dairies and beef lots, the study area is or has been used to raise poultry (i.e., chickens, turkeys) and swine. We estimated the amount of manure nitrogen used for land application based on number of animals reported in the same four national agricultural census reports from NASS from which the number of milk cows were tabulated in Table 27. We also included the 1945 agricultural census data. The NASS reports identify, by county, chickens, broilers, turkeys, and total hogs and pigs. We used the following annual total nitrogen excretion rates for these animals (D. Liptzin, personal communication, 2011; U.S. EPA, 2004): chicken (layers, inventory) - 0.55 kg N/yr/head, chicken (broiler sales) – 0.07 kg N/head, turkeys (sales) – 0.4 kg N/head, and hogs (inventory) – 5.9 kg N/yr/head. Atmospheric losses due to ammonia volatilization were estimated based on a 51% atmospheric loss rate for poultry and a 63% loss rate for swine (U.S. EPA, 2004, their Table E-2). Manure nitrogen not lost to the atmosphere is assumed to be applied to cropland across the study area as soil amendment, in addition to typical fertilization rates (see Section 3).

The agricultural census years do not all coincide with the five historic and current periods used in this study and centered on 1945, 1960, 1975, 1990, and 2005. For 1960 and 1975, land applied manure N was estimated by linear interpolation of the 1950 and 1992 data. For 1990, we assumed the same values as in the 1992 census. Linear interpolation of the 2002 and 2007 census data provided an estimate of 2005 land applied manure N from poultry and hogs. The data presented in Table 37 summarizes the total land applied swine and poultry manure nitrogen across all five counties. For 2005, the total in TLB and SV is somewhat lower than in 1990, when production peaked at 1 Gg N/year (less than 1% of the estimated 2005 dairy manure N land applied).

Adminsitrative Record
Page 34439

Table 37. Manure nitrogen from swine and poultry used for land application on cropland within the study area.

	Hogs and Pigs [Mg N/yr]	Chicken [Mg N/yr]	Turkey [Mg N/yr]	Total [Mg N/yr]
1945	170	53	0	223
1960	16	155	141	313
1975	19	308	311	638
1990	22	482	503	1,007
2005	0	456	406	862

4.9 Summary: Animal Farming as a Source of Groundwater Nitrate

Dairies represent the major animal farming industry in the Tulare Lake Basin with one million adult milking cows. Other animal farming operations (AFOs) within the study area include beef cattle feedlots (one in the Salinas Valley and one of significant size in the Tulare Lake Basin), and a small number of poultry operations and hog farms with approximately 10,000 hogs, 14 million broilers, and 2 million turkeys. Given the dominant size of the dairy herd in the study area, compared to other confined animal facilities, this chapter focuses on N loading to groundwater from dairies and feedlots.

Animal farming is a significant source of nitrogen due to the organic and ammonium nitrogen contained in the manure excreted by animals. In dairies, manure is collected in dry and liquid forms, recycled within the animal housing area for bedding (dry manure) and as flushwater (freestall dairies), stored in lagoons (liquid manure), and ultimately applied to the land. Manure is land applied in solid or liquid form, typically on forage crops (e.g., summer corn, winter grain) that are managed by the dairy farm or it is exported to nearby farms (mostly as manure solids) and used as a soil amendment. Nitrogen contained in manure applied to cropland or leached from corrals or lagoons can be a significant source of nitrate leaching to groundwater.

We consider three separate sources of nitrate within a dairy farm: open corrals and feedlots, manure storage lagoons, and manured cropland. Each of these sources contributes to groundwater nitrate via distinctly different mechanisms. Groundwater nitrate loading is estimated by different methods for each of these land use categories. Groundwater nitrate loading from corrals and lagoons is based on recharge rates and nitrate concentrations found in previous field studies, and based on the actual size of a corral or lagoon. Groundwater nitrate loading on manured cropland, as on other cropland, is estimated by considering all nitrogen fluxes to and from an individual field, which are crop type dependent and include fertilizer and manure nitrogen applications, and harvest removal of nitrogen, among others. At the county and study area level, land applied manure is added to the study area and county cropland mass balance. For the CAML-based spatially distributed simulation of groundwater nitrate loading, individual dairies and the cropland under their managemend is considered. For this simulation, we also consider six different hypothetical scenarios that bracket actual conditions for the

Adminsitrative Record
Page 34440

land application of manure to on-dairycropland versus off-dairy cropland. Groundwater nitrate loading on cropland is estimated as the difference between nitrogen inputs to and outputs from an agricultural field (mass balance approach) rather than based on literature values (see Sections 1 and 2).

Using recently published studies on dairy cow excretion and on atmospheric nitrogen losses in dairy facilities, along with county data on manure sales, and applying recent data collected by the Central Valley Regional Water Board, we estimate the nitrogen produced by the dairy herd, of which 38% is lost to the atmosphere as ammonia before land application of the manure. The amount of land-applied dairy manure nitrogen in the area is about 127 Gg N/yr [140,000 t N/yr] applied either directly to portions of 130,000 ha (320,000 ac) of dairy cropland or exported to nearby cropland. Due to transportation costs, manure nitrogen exports are limited to cropland within the study area, often nearby dairies. Land applied manure nitrogen becomes part of the cropland nitrogen mass balance, which includes other input terms. Groundwater leaching is determined based on the overall cropland mass balance.

Direct leaching of manure N to groundwater from animal corrals and manure lagoons is about 1.5 Gg N/yr (1,700 t N/yr) and 0.2 Gg N/yr (220 t N/yr), respectively. There is significant uncertainty about the overall magnitude of corrals and lagoons as groundwater nitrate sources. Actual loading may range somewhere between 0.5 to 8 Gg N/yr (about 500 to 9,000 t N/yr) for corrals and between 0.2 – 2 Gg N/yr (about 200 – 2,200 t N/yr) for lagoons. Other CAFOs in the study area generate a total of about 0.9 Gg N/yr (1,000 t N/yr) that is land applied as manure or compost.

Over the past 60 years, dairy manure applied to land has increased exponentially, effectively doubling every 15 years, from 8 Gg N/yr (9,000 t N/yr) in 1945 to 16 Gg N/yr (18,000 t N/yr) in 1960, 32 Gg N/yr (35,000 t N/yr) in 1975, 56 Gg N/yr (62,000 t N/yr) in 1990, and 127 Gg N/yr (140,000 t N/yr) in 2005, an overall 16-fold increase in manure nitrogen output. The increase in manure nitrogen is a result of increasing herd size (7-fold between the late 1940s and 2005) and increasing milk production per cow (3-fold), and is slowed only by the increased nitrogen-use efficiency of milk production.

Until the 1960s, most dairy animals in the region were only partly confined, often grazing on irrigated pasture with limited feed imports. Manure from dairy livestock generally matched the nitrogen needs of dairy pastures. Since the 1970s, dairies in the Tulare Lake Basin have operated mostly as confined animal facilities, growing alfalfa, corn, and other grain feed on-site, importing additional feed, and housing the animals in corrals and freestalls. The growth in the dairy industry has created a nitrogen excess pool that remains unabsorbed by crops. Much of the nitrogen excess is a recent phenomenon. With groundwater quality impacts delayed by decades in many production wells, the recent increase in land applied manure nitrogen is only now beginning to affect water quality in wells of the Tulare Lake Basin, with much of the impact yet to come.

Adminsitrative Record
Page 34441

5 Urban Landscape Nitrate Loading

5.1 Introduction

Urban sources of nitrogen tend to be intermediate in magnitude compared to natural areas and agricultural land uses, which are lower and higher respectively. There are three main pathways of N in urban areas: fertilizer use and application, human food consumption, and the household use of non-food N containing compounds. The per capita rates of food consumption and the ultimate fate of that food (wastewater treatment vs. disposal in landfills) are relatively well characterized in many areas. However, there are two other uses of N that are more difficult to quantify. The first is the household use of N containing products. One class of compounds is synthetically produced from the same ammonia feedstock as fertilizers. Examples of these synthetic compounds include nylon, polyurethane, and acrylonitrile butadiene styrene plastic. In addition many household products like shampoo and detergents contain synthetic N as well. Natural sources of non-food N to urban areas are predominantly derived from wood (lumber, paper, cardboard, etc.) while cotton and other fiber products are an insignificant source of N. Finally, pet waste from dogs and cats is a part of urban N dynamics. Though pet waste can pose a detriment to quality of surface waters, often for pathogenic reasons, its role in nitrate leaching to groundwater is comparatively minor, as this material is either disposed of in the landfill or is largely deposited on turfgrass where it is unlikely to leach to groundwater because of the high N retention in turfgrass soils.

For the purposes of the N balance in the present study, all of these urban N sources are ignored. However, there has been some suggestion that household products contribute N to wastewater (e.g., Baker et al. 2001), but in terms of mass, they are likely insignificant.

5.1.1 Landfills

Accumulation of nitrates in landfills is one potential source of loading as there are approximately 277 solid waste facilities, in various states of operation, throughout the study area (California Department of Resources Recycling and Recovery, 2011). Loading of nitrate to groundwater could be significant for landfills with active composting facilities depending on their management practices (i.e. if they store and compost nitrogen rich material over unlined areas) (M. Keeling, pers. comm.), which would be mobilized during precipitation events. Additional potential sources in landfills include biosolids, which are often degraded sufficiently prior to incorporation, and other organic material. While the quantities of these materials are largely unknown, the anaerobic state typically found within and below landfill environments would promote denitritification and biodegradation, and thus total leaching loss of nitrate to groundwater would be minimal (though releases of other forms of N to the environment, such as NO_x, may be significant).

Further, most landfills have sophisticated liners to minimize leaching, most are monitored for such leaching, and all facilities are regulated by local enforcement agencies. Therefore, with this information, and in conjunction with finding from previous studies (e.g., Hater et al. 2003, Wakida & Lerner 2006), we

Adminsitrative Record
Page 34442

have determined that any nitrate leachate contamination from landfills in the study area is comparatively negligible. For the purposes of nitrate loading, we assumed that both long-lived N containing compounds (natural and synthetic) as well as point source loading from landfills do not contribute to nitrate leaching in the study area.

5.2 Methods

For the purposes of N calculations, urban N use is described in two different sections. Urban fertilizer use is described in this chapter while wastewater is described in Section 6 of this report.

5.2.1 Fertilization rates

Fertilizer is used in urban areas for homeowner lawns, parks, and recreational facilities, such as sports fields and golf courses. These land uses vary in their recommended fertilizer use, but there is almost no data on actual fertilization rates. At the national scale, estimates by the Scotts Company suggest approximately 3,000 Gg N/yr (3.3 million tons N/yr) are applied as fertilizer on all turfgrass equally divided between homeowner application, commercial application, and recreational facilities. Based on the estimate of turfgrass acreage in California reported by Milesi et al. (2005), and scaling down the national estimate of turfgrass fertilizer use based on the population of California, the preliminary turfgrass fertilization rate calculated by the California Nitrogen Assessment is 50 kg N/ha (45 lb N/ac). The spatial location of turfgrass was based on the urban pixels in the 2010 CAML map. The amount of turfgrass in each pixel was based on the relationship described in Milesi et al. (2005) between impervious surface area and turfgrass acreage.

$$\text{percent turfgrass} = 79.53 - 0.83 \times (\text{percent impervious surface})$$

The impervious surface data was extracted from the impervious surface layer available in the 2001 National Land Cover Database.[30] Areas with less than 10% impervious surface were excluded as they tend to occur on the fringe of developed areas.

5.2.2 Nitrate Leaching

Based on the land cover in CAML we calculated a total of 31,741 ha (78,434 acres) of turfgrass within the study area (Table 38). The acreage varied between counties and represented between 12% and 23% of the urban land area depending on the county.

[30] http://www.mrlc.gov/nlcd2001.php

Adminsitrative Record
Page 34443

Table 38. Turfgrass acreages in the study area were based on empirical relationship between impervious surface area and the percent cover of turfgrass in urban land.

County	Turfgrass area (ha) [acres]	Golf course area (ha) [acres]	Turfgrass as a Percent of Urban Land (%)
Fresno	11,178 [27,621]	738 [1,824]	19
Kern	9,010 [22,264]	873 [2,157]	15
Kings	2,153 [5320]	212 [524]	12
Monterey	4,990 [12,331]	386 [954]	23
Tulare	4,410 [10,897]	369 [912]	15
Total	**31,741 [78,434]**	**2,578 [6,370]**	

Based on one of the most comprehensive surveys of turfgrass leaching, only about 2% of applied N fertilizer was found to leach below the rooting zone (Petrovic 1990). Nitrate leaching from turfgrass fertilization is thought to be negligible. Leaching of nitrate from turfgrass, when it occurs, is most likely at high rates of fertilization such as on golf courses and athletic fields. To account for this potential nitrate leaching we also assign a value of 10 kg N/ha/yr leached to groundwater from golf courses.

For the county and study area N leaching reported in Section 1.6, we assumed a worst case scenario of 10 kg N/ha/yr leached to groundwater from both turfgrass areas and golf course areas, as listed in the table above.

5.3 Results and Discussion

There are large uncertainties related to urban N use in terms of the spatial location of N use, the amount of N use, and the fate of this N use. Because many areas of turfgrass are small, it has been difficult to use traditional mapping and remote sensing techniques to identify the spatial location of turfgrass. One promising high resolution approach is the High Ecological Resolution Classification for Urban Landscapes and Environmental Systems (Cadenasso et al. 2007). This land classification system appears to be better at predicting N yields in streams than previous methods, but it has not been tested for predicting nitrate loading to groundwater.

Comprehensive survey data on fertilization rates are rare nationally and nonexistent for California. While we use a top down approach to estimate total fertilizer use and total turfgrass acreage, survey data can provide more spatially explicit patterns in turfgrass use. Our estimate of a N fertilizer application rate of 50 kg N/ha is lower than some survey data suggest. Both Flipse et al. (1984) for Long Island, and Law et al. (2004) for Baltimore, report on survey data for home lawns. In both locations

Adminsitrative Record
Page 34444

fertilization rates were approximately 100 kg N/ha. One reason the California estimate may be lower is that both studies report a strong relationship between fertilization rates and socioeconomic status. Since these surveyed areas may not be representative of the socioeconomic status of all areas with turfgrass and may be further biased because they include only responses from people who responded to the surveys, the data may not be representative of all households with turfgrass.

Nitrate leaching from turfgrass soils has been better studied in research plots and highly managed areas than in home lawns. We based our assumption, that nitrate does not leach from turfgrass, on the data compilation by Petrovic et al. (1990). It appears that the ability of turfgrass to sequester applied N fertilizer can last for decades (Raciti et al. 2008). Using isotopically labeled N, turfgrass fertilized at low application rates (49 kg N/ha/ application) resulted in leaching of less than 1% of applied fertilizer after a decade of constant fertilization (Frank et al. 2006). Turfgrass, however, is not completely immune to leaching. When leaching does occur, it is most likely on coarse textured soils with high fertilization rates (Sharma et al. 1996). There are relatively few peer reviewed studies of nitrate leaching from turfgrass in California. Wu et al. (2010) report that even at a rate of 195 kg N/ha, soil N concentrations are relatively low. One way to decrease the amount of fertilizer needed is to "grasscycle," i.e. to leave mulched grass clippings to the lawn (Harivandi et al. 1999).

Golf courses receive among the highest rates of fertilization of any turfgrass use. For example, fairways and greens likely receive in excess of 400 kg N/ha/yr (Wu et al. 2007). However, golf courses represent less than 10% of the turfgrass acreage and not the entire acreage is fertilized at rates this high. Even assuming that the entire acreage of golf courses receives 400 kg N/ha/yr and leaches half of this N, this amounts to a total of only 0.5 Gg N/yr. While this amount of N could cause localized contamination of waterbodies, golf courses represent only a minor fraction of the total N load in the study area.

For the final groundwater nitrate loading analysis (Section 1), we used the area identified as lawns and golf courses in urban areas, listed in the above table, and multiplied the area for turf and golf courses in urban areas with an overall relatively high upper rate of 10 kg N/ha/yr (8.9 lb N/ac/yr). This yields an estimated groundwater loading from urban turf and golf course areas of 0.35 Gg N/yr [380 t N/yr]). In the spatially distributed N loading analysis with the GNLM code, we specified that all urban areas, not designated otherwise (cropland, percolation basin), leach 10 kg N/h/yr (8.9 lb N/ac/yr, see Figure 21 in Section 1).

Adminsitrative Record
Page 34445

6 Domestic and Urban Wastewater Sources of Nitrogen Loading

6.1 Introduction to Domestic and Urban Wastewater Sources

Domestic and urban wastewater sources of nitrogen loading include wastewater treatment and food processing facilities, leakage from sewer systems, and discharge from septic systems. These sources were examined to include their overall contributions to groundwater N loading, to assess potential regional and local impacts of associated nitrogen discharged to groundwater, to explore nitrogen control measures, and to present N loading reduction strategies.

Effluent from wastewater and food processing facilities is discharged to groundwater through application to irrigated agriculture and percolation from recharge basins. Associated nitrogen loading varies with several factors including facility type, location, and application rate. Land application of facility effluent can be an effective way to reuse water and nutrients; however, with inappropriate land application practices, groundwater can be degraded. Detailed discharge information was collected and modeled to estimate the contribution of wastewater treatment and food processing facilities on groundwater nitrogen loading.

Aging infrastructure and insufficient maintenance of sewer systems can result in leakage from sewer pipes, leading to infiltration of raw sewage into the surrounding soil and ultimately into underlying groundwater. Poorly fitted pipes, aging collection systems, sanitary sewer overflows, and unsuitable piping materials all contribute to the leakage of raw sewage. Based on information in the literature and interviews with industry representatives, nitrogen loading from sewer leakage was estimated across the region of interest.

Septic systems, designed to treat domestic wastewater and for the prevention of human exposure to pathogens, also discharge nitrogen to the subsurface. The relative contribution of septic systems, regionally and locally was examined to assess their potential impact on groundwater nitrate levels. This was accomplished through literature review and modeling of the spatial distribution of septic systems.

While potentially significant locally, the regional impact of these sources on groundwater nitrate contamination is significantly lower than other sources of nitrogen in the area of interest. However, it is important to address associated nitrogen loading on a local scale, to protect drinking water sources; nitrogen reduction measures are discussed in Technical Report 3, Section 5 (Dzurella et al. 2012).

Adminsitrative Record
Page 34446

6.2 Wastewater Treatment and Food Processing Facilities

6.2.1 Background and Introduction

As potential nitrogen sources in the Tulare Lake Basin and Salinas Valley, wastewater treatment plants (WWTPs) and food processing facilities (FPs) were examined to:

- Assess their contribution to groundwater N loading,
- Determine the regional and local impacts of nitrogen in discharge,
- Examine nitrogen control measures, and
- Propose solutions for N loading reduction.

It is important to understand the dual nature of this discussion; wastewater treatment and food processing facilities can be sources of nitrogen and they can also be part of the solution. Potential sources of nitrate contamination from these facilities are:

- Effluent from WWTPs and FPs discharged for irrigation and/or groundwater recharge and
- Wasted solids from these facilities that are applied to land as a soil amendment.

Land application of effluent from these facilities can be an effective way to reuse water and nutrients, using natural processes in the soil and irrigated crops as a final stage of treatment. However, with inappropriate land application groundwater can be degraded. When discharges run the risk of negatively impacting groundwater, existing land application processes can be modified or facilities can be improved and potentially expanded to optimize operations and/or treat wastewater to a higher quality. The reduction of N loading from these facilities is discussed separately in Technical Report 3, Section 5.2 (Dzurella et al. 2012).

6.2.1.1 Permitting, Monitoring, and Waste Discharge Requirements

The California State Water Resources Control Board (State Water Board) was established in 1967 for the protection of water resources, with regional oversight by nine Regional Water Quality Control Boards (Regional Water Boards) across the state (State Water Resources Control Board 2011a). The Central Valley Regional Water Quality Control Board (Central Valley Regional Water Board) (Region 5) and the Central Coast Regional Water Quality Control Board (Central Coast Regional Water Board) (Region 3) are responsible for the permitting, monitoring and enforcement of regulations relevant to dischargers in the Tulare Lake Basin and the Salinas Valley, respectively (State Water Resources Control Board 2011a). The Federal Clean Water Act (CWA) requires a permit for discharge to surface waters administered through the National Pollutant Discharge Elimination System (NPDES) (United States Environmental Protection Agency 2011a). Extending the CWA to the protection of groundwater, the California Porter-Cologne Water Quality Control Act of 1968 mandates all dischargers to file a report of waste discharge with the

Adminsitrative Record
Page 34447

appropriate Regional Water Board. Unless a waiver[31] is granted, subsequent waste discharge requirements (WDR), issued by the Board, provide the guidelines that must be followed to protect beneficial water uses and maintain or improve water quality in accordance with the Regional Basin Plan (Brown and Caldwell and Kennedy/Jenks Consultants 2007). Non-compliance or violation of WDRs can result in the Regional Water Board mandating measures for remediation. Monitoring and Reporting Programs (MRPs) are delineated in WDRs to facilitate ongoing protection of water resources; monthly and annual monitoring reports are submitted to the Regional Water Board to ensure continued compliance with WDRs. Requirements for the disposal of approved solid wastes, including biosolids from WWTPs, are also dictated by WDRs.

6.2.1.2 Nitrogen Speciation

As discussed above in Section 2, the nitrogen cycle consists of transformation between various nitrogen species (Figure 22). Specific transformations in the nitrogen cycle that are pertinent to this discussion include (described in Section 2.3):

- Nitrogen Fixation – nitrogen gas is incorporated in organic matter
- Mineralization – organic nitrogen is converted to ammonia
- Nitrification – ammonia is converted to nitrite and nitrate
- Denitrification – nitrate is converted to nitrogen gas
- Immobilization – nitrate nitrogen and ammonium nitrogen are used by plants and/or microbes and incorporated in organic matter.

The same transformation processes that occur naturally in the environment are relevant to the treatment of nitrogen rich wastewaters. Fundamental to the reduction of nitrogen levels in discharges from wastewater facilities, nitrogen transformations in wastewater treatment are discussed in further detail in Technical Report 3, Section 5.2 (Dzurella et al. 2012).

6.2.1.3 Land Application of Discharge from Wastewater Treatment Plants and Food Processors

When appropriately permitted, effluent from WWTPs and FPs can be discharged to surface water, percolation basins, and/or agricultural fields and approved solid waste can be used as a soil amendment. While the primary focus of this study is nitrogen loading to groundwater, discharges from WWTPs and FPs to surface water were also taken into account for receiving surface waters identified as being sources of irrigation water. Land application of discharge from wastewater treatment and food processing facilities is a common method of waste stream disposal, enabling reuse of water and

[31] In accordance with California Water Code Section 13269 state and regional boards can waive WDRs for individual dischargers under the under the following conditions (CWC Section 13269):
1) "The state board or regional board determines, after any necessary state board or regional board meeting, that the waiver is consistent with any applicable state or regional water quality control plan and is in the public interest."
2) "A waiver may not exceed five years in duration, but may be renewed…"
3) "The waiver shall be conditional and may be terminated at any time by the state board or a regional board."
4) "Monitoring requirements shall be designed to support … the waiver's conditions;" however, "the state board or a regional board may waive the monitoring requirements … for dischargers that it determines do not pose a significant threat to water quality."

Adminsitrative Record
Page 34448

nutrients remaining in the effluent following treatment. Reuse of discharge water for irrigation[32] offers the benefit of minimizing the use of chemical fertilizer and conserving higher quality water sources for other beneficial uses (e.g., drinking water) rather than depleting them for irrigation purposes (Crites, Reed, & Bastian 2000).

One disposal option for FP waste is to discharge to Publicly Owned Treatment Works (POTW) (e.g., an existing municipal or industrial wastewater treatment plant) where appropriate treatment is already in place. Facilities accepting FP effluent are governed by NPDES permits and WDRs. As an alternative to disposal at Publicly Owned Treatment Works (POTW), land application of food processing waste can be a less costly method of disposal. However, to avoid degradation of groundwater, it is vital *"that wastes are applied to fields at reasonable rates, such that organic matter is broken down, [and] nutrients are taken up by crops or consumed by soil microorganisms…"* (Central Valley Regional Water Quality Control Board 2005, p. 4). Discharge to percolation basins enables direct groundwater recharge; however, the waste stream must be of a high enough quality to avoid degradation of underlying groundwater.

Land treatment methods can be categorized into three main types: Slow Rate (SR), Overland Flow (OF), and Soil Aquifer Treatment (SAT)/Rapid Infiltration (RI) (Crites et al. 2000; United States Environmental Protection Agency 2006). SR and SAT/RI are most pertinent to our analysis. SR land treatment refers to "*the application of wastewater to a vegetated soil surface*" whereby wastewater is treated through interaction with the root zone and soil (United States Environmental Protection Agency 2006, p. 1-2). The SAT/RI method refers to "*controlled application of wastewater to earthen basins in permeable soils at a rate typically measured in terms of meters of liquid per week…Treatment … is accomplished by biological, chemical and physical interactions in the soil matrix*" (United States Environmental Protection Agency 2006, p. 1-4). Table 39 summarizes site considerations, design features, and the resulting characteristic water quality reaching groundwater with proper implementation of land treatment processes (Crites et al. 2000).

[32] It is important to note that not all discharges from WWTPs and FPs are appropriate for reuse as irrigation water; discharge water must have suitable water quality characteristics to be used on crops (e.g., it would be inappropriate to irrigate a strawberry field with effluent from a WWTP and high salinity effluent would be damaging to certain crops.)

Adminsitrative Record
Page 34449

Table 39. Site considerations, design features, and characteristic effluent water quality for land treatment processes. (Source: Crites et al. 2000.)

Parameter	Slow Rate (SR)	Rapid Infiltration (RI)
Site Considerations		
Grade	20%, cultivated site 40%, uncultivated	Not critical
Soil Permeability	Moderate	High
Groundwater Depth	2 – 10 ft	3 ft during application 5 – 10 ft during drying
Climate	Winter storage in cold climates	Not critical
Design Considerations		
Application Method	Sprinkler or surface	Usually surface
Annual Loading, ft	2 – 20	20 – 400
Treatment area for 1 mgd, acres	60 – 700	7 – 60
Weekly Application, in	0.5 – 4	4 – 96
Minimum Pretreatment	Primary	Primary
Need for Vegetation	Required	Grass (sometimes)
Characteristic Water Quality After Land Treatment (mg/L, unless otherwise indicated)		
BOD_5	<2	5
TSS	<1	2
NH_3/NH_4^+ (as N)	<0.5	0.5
Total N	3	10
Total P	<0.1	1
Fecal coliform (#/100 mL)	0	10

6.2.1.4 Wastewater Treatment Plants

For regions with public sewers, wastewater from toilets, sinks, laundry, showers, dishwashers, and sometimes storm water, is conveyed to a central facility for treatment. Influent nitrogen levels typical of domestic WWTPs (raw sewage) are listed in Table 40. Although influent nitrogen levels vary with community water use, the annual mass loading of an individual treatment facility is directly related to the population served. Nitrogen loading from human waste can range from 2 – 15 g/capita/day (Henze, Loosdrecht, & Ekama 2008); according to (Crites & Tchobanoglous 1998b), the typical amount of excreted nitrogen is 13.3 g/capita/day. WWTPs serving larger populations generally discharge the greatest amount of total nitrogen. However, flow increases with population served as well, therefore, plants discharging the greatest total nitrogen annually are not necessarily discharging higher concentrations of nitrogen.

Adminsitrative Record
Page 34450

Table 40. Typical composition of domestic wastewater. (Source: Metcalf & Eddy 2003; Wisconsin Department of Natural Resources 2006; Henze et al. 2008.)

	Low	Medium	High
	mg/L as N		
Ammonia – N	12 – 20	25 – 45	50 – 75
Organic – N	8	15	35
Total – N	20 – 30	40 – 60	85 – 100

Effluent nitrogen levels are dependent on the level of treatment. For example, with only nitrification (ammonia to nitrate), the nitrate concentration in discharged water can be in the range of 20 – 30 mg/L nitrate-N, assuming complete nitrification (Wisconsin Department of Natural Resources 2006). Treatment consisting of both nitrification and denitrification can decrease effluent nitrogen concentrations below 10 mg/L N and advanced tertiary treatment can bring effluent nitrogen levels below 2 mg/L N (Metcalf & Eddy 2003). It is important to account for total nitrogen in discharged effluent (including ammonia, nitrate, nitrite, and organic nitrogen), rather than only nitrate, because other forms of nitrogen in discharged effluent can be transformed to nitrate after being discharged.

To assess N loading from WWTPs (and options for reducing N loading as discussed in Technical Report 3, Section 5.2, Dzurella et al. 2012), it is important to understand the distinction between conventional wastewater treatment and specialized treatment for nutrient removal.

6.2.1.5 Conventional Wastewater Treatment

WWTPs are generally designed to remove solids and organic matter through several standard unit processes. Nutrient removal is an additional process, beyond conventional wastewater treatment, used to decrease effluent levels of nitrate and/or phosphate. Preliminary treatment and primary treatment are designed to remove large and/or heavy objects capable of damaging downstream equipment as well as settleable and suspended solids. Unit processes can include screens, grinders, grit chambers, and primary clarifiers. The primary clarifier can remove up to 95% of settleable solids and up to 60% of total suspended solids, including a portion of influent organic matter (Metcalf & Eddy 2003). The small fraction of nitrogen removed in primary treatment is concentrated in primary sludge (Metcalf & Eddy 2003). Secondary treatment generally refers to the removal of organic matter and suspended solids via biological treatment (activated sludge) and a secondary clarifier, respectively. In secondary treatment, with a long enough hydraulic detention time, ammonia can be oxidized to nitrate through aeration and the activity of nitrifying bacteria. Conventional wastewater treatment historically did not extend beyond secondary treatment, with filtration in tertiary treatment as an optional step. Tertiary and advanced treatment can consist of a variety of additional unit processes to improve effluent water quality including nutrient removal (discussed below), filtration for additional solids removal, granular activated carbon to address organic chemicals, and, when extremely high quality water is necessary for reuse and recycling applications, reverse osmosis for the removal of numerous additional constituents. Disinfection is typically the final step in the treatment train.

Adminsitrative Record
Page 34451

6.2.1.6 Nutrient Removal

Nutrient removal in wastewater treatment has become increasingly prevalent over the past 30 years (United States Environmental Protection Agency 2008b). Nitrogen removal from wastewater can be accomplished using a variety of technologies and configurations; both biological and physical/chemical processes are effective. Treatment options for nutrient removal from wastewater are thoroughly described in the literature, with an abundance of material in engineering textbooks and state and federal guidance manuals/publications (Metcalf & Eddy 2003; United States Environmental Protection Agency 2008; Water Environment Federation 2010). The U.S. EPA guidance manual (2008) is a comprehensive resource describing available relevant technologies, their reliability, feasibility, and costs, based on case studies of full scale WWTPs.

With many potential configurations to achieve nitrification or combined nitrification and denitrification, biological nutrient removal is typically categorized as tertiary or advanced treatment and can be incorporated into the biological processes of secondary treatment (Metcalf & Eddy 2003). Biological nutrient removal (BNR) is accomplished through the provision of optimal conditions for the activity of various species of bacteria. Through biologically mediated transformation processes, influent organic nitrogen and ammonia are converted to nitrate and then to nitrogen gas. Additional methods used for nitrogen removal include chemical oxidation, air stripping, and ion exchange (Metcalf & Eddy 2003). With nutrient removal, effluent nitrogen levels can be decreased to less than 5 mg/L N (Metcalf & Eddy 2003). Treatment options for nutrient removal from wastewater are discussed in greater detail in Technical Report 3, Section 5.2 (Dzurella et al. 2012).

6.2.1.7 Recycling of Biosolids from Wastewater Treatment Plants

"Biosolids are primarily organic materials produced during wastewater treatment which may be put to beneficial use" (United States Environmental Protection Agency 2000a, p. 1). Additional options for biosolids disposal include incineration, landfilling, and composting. Through land application of biosolids, nutrients and organic matter are recycled, promoting plant growth and diminishing the need for inorganic fertilizer application. Biosolids are organic and less soluble than inorganic fertilizers; due to the slow release of nutrients, the risk of runoff and leaching is diminished (United States Environmental Protection Agency 2000a). In the liquid form (94 – 97% water), biosolids can be applied through injection or spraying. Through injection of biosolids into the top tilled layer of soil, nuisance conditions like odor and vector attraction can be minimized due to incorporation into the soil (National Biosolids Partnership 2005). (This is not to be confused with deep well injection for biosolids disposal, a completely different process for disposal of biosolids rather than land application of biosolids as a soil amendment.) After dewatering, in the solid form, biosolids can be applied using standard farming methods typical of manure application (United States Environmental Protection Agency 2000a). In accordance with the U.S. EPA's Code of Federal Regulations Title 40, Part 503, biosolids processing, or stabilization, is required to limit odors, kill pathogens, and sufficiently avoid attracting vectors (e.g., rodents, mosquitoes, etc.) (United States Environmental Protection Agency 2000a). Stabilization is accomplished through *"adjustment of pH, or alkaline stabilization, digestion, composting, and/or heat drying"* (United States Environmental Protection Agency 2000a, p. 2). Class A biosolids are treated to

Adminsitrative Record
Page 34452

the level of "exceptional quality" and can be applied without limitation. Class B biosolids have application restrictions to avoid hazardous exposure to pathogens. Costs and processing duration of class A and class B biosolids vary with facility size, sewage sludge characteristics, and treatment type.[33,34] Additionally, metal concentrations must not exceed federal limits as described in the U.S. EPA 40 CFR Part 503 (United States Environmental Protection Agency 2000a).

Through the State Water Board's General Order, Water Quality Order No. 2004-12-DWQ, state regulations ensure compliance with federal requirements and the California Water Code, by detailing waste discharge requirements for the use of biosolids as a soil amendment (State Water Resources Control Board 2011b). For compliance under the General Order, a Notice of Intent (NOI) must be submitted to the local Regional Water Board; land application of biosolids under the General Order is only permitted following receipt of a Notice of Applicability (State Water Resources Control Board 2011b). In addition to national and regional guidelines, there are county level ordinances governing local land application of biosolids. Regulations (as of 2008) for the counties of interest are as follows (Lauren Fondahl, Biosolids Coordinator, CWA Compliance Office, U.S. EPA Region 9 2011):

- Fresno
    - No Class B application on unincorporated lands has been allowed since 2001.
    - Class B application by small POTWs on city-owned land is allowed.
    - The majority of local biosolids are sent out of the county for composting (to Kern and Merced Counties).
- Kern
    - Only Exceptional Quality[35] (EQ) composted biosolids may be applied to unincorporated lands since 2003.
    - Class B application on city-owned lands is allowed.
    - The county has a long history of court action to control/limit biosolids land application.
- Kings
    - Only EQ composted biosolids may be applied throughout the county since 2006.

[33] "One study estimated costs for Class A alkaline stabilization ranging from $139 to $312 per dry ton of wastewater solids processed by facilities designed to serve wastewater treatment plants ranging in capacity from 10 to 60 million gallons per day. This estimated range demonstrates the economy of scale associated with larger systems. The capital costs cited in this same study ranged from $1.5 to $4.0 million and annual costs were estimated to range from $1 million and $4 million. This study concluded that alkaline stabilization was less expensive than composting or thermal drying (Sullivan, 1996)" (United States Environmental Protection Agency 2000b p. 6).

[34] "1. Aerobic digestion—Sewage sludge is agitated with air or oxygen to maintain aerobic conditions for a specific mean cell residence time at a specific temperature. Values for the mean cell residence time and temperature shall be between 40 days at 20 degrees Celsius and 60 days at 15 degrees Celsius. 2. Air drying—Sewage sludge is dried on sand beds or on paved or unpaved basins. The sewage sludge dries for a minimum of three months. During two of the three months, the ambient average daily temperature is above zero degrees Celsius. 3. Anaerobic digestion—Sewage sludge is treated in the absence of air for a specific mean cell residence time at a specific temperature. Values for the mean cell residence time and temperature shall be between 15 days at 35 to 55 degrees Celsius and 60 days at 20 degrees Celsius. 4. Composting—Using either the within-vessel, static aerated pile, or windrow composting methods, the temperature of the sewage sludge is raised to 40 degrees Celsius or higher and remains at 40 degrees Celsius or higher for five days. For four hours during the five days, the temperature in the compost pile exceeds 55 degrees Celsius. 5. Lime stabilization—Sufficient lime is added to the sewage sludge to raise the pH of the sewage sludge to 12 after two hours of contact" (CFR - Code of Federal Regulations).

[35] "The term *Exceptional Quality* is often used to describe a biosolids product which meets Class A pathogen reduction requirements, the most stringent metals limits (Pollutant Concentrations), and vector attraction reduction standards specified in the Part 503 Rule" (United States Environmental Protection Agency 2000a).

Adminsitrative Record
Page 34453

    - The majority of EQ compost goes to unincorporated lands and is from Kern County composting operations.
- Tulare
    - Only Class A/Class A equivalent biosolids may be applied, except
    - Class B application by small and medium POTWs on city-owned land is allowed.
- Monterey
    - No land application of biosolids is allowed.
    - A county landfill operates a biosolids composting pilot.
    - It is likely that some soil amendments imported into the county contain some biosolids.

Approximately half of national total biosolids are reused in land application; included land accounts for <1% of agricultural acreage (United States Environmental Protection Agency 2009). In U.S. EPA's Region 9 (including California), *"most biosolids…are used for growing agricultural non-food crops, for landscaping, as alternative daily land cover or final cover at landfills, or are landfilled. A very small amount is incinerated. There are several new or proposed projects for heat drying and use as fuel"* (United States Environmental Protection Agency 2011b). According to U.S. EPA Region 9, in 2009, 615,000 dry metric tons (dry weight) were produced in California. The fate of California biosolids in 2009 is listed in Table 41.

Table 41. Fate of California biosolids in 2009. (Source: Lauren Fondahl, Biosolids Coordinator, CWA Compliance Office, U.S. EPA Region 9 2011.)

Use	Percent of Total	Dry Metric Tons
Land Application	61	402,000
Class A[1]	*41*	*272,000*
Class B	*20*	*130,000*
Landfill	30	200,000
Surface Disposal	3.3	22,000
Incineration	2.8	19,000
Fuel for Kilns	2.1	14,000
Deep Well Injection	0.5	3,000
Other	0.2	1,000
Total Produced[2]	*93*	*615,000*
From Storage[2]	*7*	*45,000*
Total	**100**	**661,000**

[1] Class A biosolids include: 26% compost, 10% thermophilic digestion, 3% alkali treatment, 1% heat drying, and 1% air drying.
[2] As reported by U.S. EPA Region 9.

Locations receiving the greatest amount of biosolids are listed in Table 42; Kern County receives the greatest portion of California biosolids, some of which is composted and exported out of the county.

Adminsitrative Record
Page 34454

Table 42. California counties receiving the greatest amount of biosolids in 2009. (Source: Lauren Fondahl, Biosolids Coordinator, CWA Compliance Office, U.S. EPA Region 9 2011.)

County		Percent	Dry Metric Tons
Kern	**Total**	**27**	**180,000**
	Composters[1]		*97,000*
	Class A Land Applied		*81,000*
	Class B Land Applied		*2,000*
Yuma	**Total**	**12.4**	**82,000**
	Class B Land Applied		*71,000*
	Landfilled		*11,000*
Sacramento	**Total**	**6.6**	**44,000**
	Surface Disposal		*20,000*
	Class A Land Applied		*6,000*
	Class B Land Applied		*18,000*
San Bernardino	**Total**	**6.6**	**43,500**
	Composters		*30,000*
	Heat Drying/Fuel		*10,000*
	Class A Land Applied		*3,500*
Los Angeles	**Total**	**5.7**	**38,300**
	Composters		*13,000*
	Landfilled		*18,000*

[1] Includes compost exported and land applied outside of the county.

The nitrogen content of biosolids varies by source, wastewater treatment type and biosolids conditioning processes. The State Water Board's General Order indicates that biosolids nitrogen content can range from 2 – 10% (dry weight) (State Water Resources Control Board 2011b). According to the U.S. EPA Region 9 Biosolids Coordinator, Lauren Fondahl, composting, heat or air drying to prepare Class A biosolids decreases the nitrogen content from 5 – 6% to 1 – 2%; however, using other processes to prepare Class A biosolids can maintain higher nitrogen content. For the purposes of this study, the nitrogen content of biosolids is assumed to be 3.3% as listed in (Metcalf & Eddy 2003).

6.2.1.8 Food Processing Facilities

Reuse of food processing discharge through land application is a common disposal option for many types of food processing wastes and is well documented (Crites et al. 2000; Central Valley Regional Water Quality Control Board 2005; United States Environmental Protection Agency 2006; Brown and Caldwell and Kennedy/Jenks Consultants 2007). Land application of wastewater is common for a wide range of FP categories including brewery, vegetable and fruit canning and frozen foods, dairy, meat processing, and winery wastewaters (Crites et al. 2000).

Wastewater from FPs is characterized by the specific processing operations of the facility and by the food type; as such, waste volume and nitrogen content can vary widely between facilities. Steps in food

Adminsitrative Record
Page 34455

processing can include peeling, trimming, washing, mechanical operations, cooling, heating, canning, pureeing, juicing, blanching, cooking, drying/dehydrating, and cleaning of machinery and the facility (Liu 2007).

In-plant treatment of food processing waste prior to discharge is also dependent on food processor type and wastewater characteristics. For low strength wastewater, screening of the waste stream may be sufficient prior to land discharge. For high-strength wastewater, a combination of in-plant treatment processes may be implemented prior to land discharge, including biological treatment (activated sludge), aeration lagoons, trickling filters, settling basins, ion exchange and/or membrane processes (Wang et al. 2005; Liu 2007). Depending on the disposal method, different waste streams within the plant can be handled separately or they can be combined to meet disposal requirements. For example, non-contact cooling water may be appropriate for discharge to land without treatment, but high-strength wastewater may require extensive treatment onsite or at POTW. High-strength wastewater may be blended for dilution to meet effluent requirements for land application or to reduce disposal costs at POTWs. Some facilities discharge to onsite septic systems as well. A comprehensive guidance manual for waste management in the food processing industry was developed for the California League of Food Processors (CLFP) by (Brown and Caldwell and Kennedy/Jenks Consultants 2007).

It is important to note the seasonal differences in waste management from FPs. In highly agricultural areas, like the Tulare Lake Basin and the Salinas Valley, discharge may be handled differently during the growing season. Land application to irrigated agriculture may be the primary disposal method during the growing season, with alternative disposal methods the rest of the year. This seasonal variation must be taken into consideration; in close proximity to discharges, groundwater drinking water sources may be unaffected during one part of the year, but impacted by nitrate another part of the year. The potential for temporal variation can result in the need to address impacted drinking water supplies seasonally.

6.2.1.9 Recycling of Solid Wastes from Food Processing Facilities

Solid wastes from food processing operations are often reused as animal feed; however, certain solids can be composted and land applied as a soil amendment, a practice similar to leaving plant residual on a field after harvest.
According to the Central Valley Regional Water Board (Central Valley Regional Water Quality Control Board, Daniel Benas, Environmental Scientist, Compliance and Enforcement Unit 2011):

- Most FPs screen wastewater for solids before effluent is discharged.
- Solid wastes from food processing are often sold as animal feed.
- A small number of FPs dry solid wastes and apply to land as a soil amendment.

6.2.2 Nitrogen Loading from Wastewater Treatment and Food Processing Facilities

To address the nitrate problem, it is important to characterize the relative impact of nitrogen laden discharge from wastewater treatment and food processing facilities on groundwater in the Tulare Lake Basin and Salinas Valley. On a regional scale, the total mass loading from WWTPs and FPs is examined

Adminsitrative Record
Page 34456

to assess the relative contribution of these facilities to the nitrate problem. Locally, it is important to consider the risk to public drinking water supply wells and private domestic wells based on proximity to discharge locations and groundwater flow.

Excessive nitrogen loading to groundwater, due to application of food processing wastewater, has been reported at locations across the country. The Central Coast Regional Water Board estimates N loading to groundwater of 687 tons/year from municipal and industrial wastewater, accounting for 5.4% of total N loading to groundwater (Central Coast Regional Water Quality Control Board 2011). Historically, in the Central Valley, little or no groundwater monitoring was required to assess the impact on groundwater of land applied FP discharge; appropriate application practices were the primary goal of the Regional Board (Central Valley Regional Water Quality Control Board 2005). Additionally, in the past, facilities granted waivers of WDRs were generally not monitored and many waivers had no set expiration date. Changes in the California Water Code (CA Codes (wat:13260-13275)) resulted in the expiration of waivers by 2003 and the need to renew every five years thereafter (Central Valley Regional Water Quality Control Board 2005). As of 2005, nearly 50% of facilities discharging to land were monitoring groundwater.

Related groundwater monitoring data are available in paper files but not in digital format at the Region 5 office. Extraction of the monitoring data from paper reports was beyond the scope of this study. The related discussion in the 2005 Central Valley Water Board report was used as an alternative. This highlights a significant shortcoming of the current state of storage and management of data related to this study. Generally, a vast amount of data pertinent to this study exists; however, finding and accessing that information in a timely fashion is not always possible due to the lack of digital information.

According to the Hilmar SEP Project, food processors discharging the highest nitrogen loading in the Tulare Lake Basin are fruit and vegetable canning facilities (Rubin et al. 2007; Sunding & Berkman 2007; Sunding et al. 2007). In 2005, the Central Valley Regional Water Board estimated that land application practices at approximately 75% of food processing facilities discharging to land were degrading groundwater to some extent, though not specifically related to nitrogen loading (Central Valley Regional Water Quality Control Board 2005). In 2005, groundwater monitoring data from 13 facilities confirmed degradation of groundwater due to nitrate; an additional 25 facilities were listed as suspected of groundwater degradation due to nitrate (Central Valley Regional Water Quality Control Board 2005). The majority of these facilities are fruit and vegetable processors.

An update to the Central Valley Regional Water Board's Staff Report on FP discharges summarizes food processing dischargers in the Central Valley as follows (Central Valley Regional Water Quality Control Board 2006, p. 1):

- "119 processors discharge directly to Publicly Owned Treatment Works (POTWs), that are regulated by federal NPDES permits or by individual waste discharge requirements (WDRs);
- 212 processors discharge to land, and are regulated under individual WDRs issued pursuant to the California Water Code (CWC);

Adminsitrative Record
Page 34457

- 62 processors discharge to land and are enrolled under Order No. R5-2003-0106, the *Waiver of Waste Discharge Requirements for Small Food Processors*; and
- Approximately 250 wineries plus an unknown number of other food processors discharge to land, but have not submitted Reports of Waste Discharge (RWDs), as required by the CWC."

As discussed above, groundwater monitoring programs are used to assess the impact to groundwater in the vicinity of these facilities. The "Facilities-at-a-Glance" resource available via the California Integrated Water Quality System (CIWQS), through the State Water Board website, provides information on known violations for WWTP and FP dischargers. In the past 5 years only 6 of 132 FPs and 2 of 40 WWTPs have nitrogen violations listed in this database. This is in contrast to the data from the Central Valley Regional Water Quality Control Board (2005) report which states groundwater monitoring data from 13 FPs indicated degradation of groundwater due to nitrate with the potential for an additional 25 suspected FPs.

6.2.2.1 Nitrogen Loading – Methodology

N loading from land application of WWTP effluent, WWTP biosolids and FP effluent was assessed by first characterizing the nitrogen contribution of each of these sources. Land applied liquid discharges from WWTPs and FPs were examined, accounting for discharges to both irrigated agriculture and percolation basins. Biosolids production was detailed and data on land application of biosolids were collected. The total mass of nitrogen, total nitrogen concentration in discharges, and application rates (kg/ha/yr) were estimated based on collected data. To assess the distribution of N loading from these sources, information on discharge location and land area was collected and the corresponding spatial distribution of N loading from these sources was mapped.

The list of facilities in the region of interest was primarily developed from a master list from the State Water Board and the California Integrated Water Quality System Project (CIWQS) online database, with facilities extracted by county. Supplemental information was extracted from the U.S. EPA's Facilities Registry System (FRS). Facilities were geo-located and mapped; facilities outside of the project boundaries were excluded. For both WWTPs and FPs, any Waste Discharge Requirement (WDR) Reports available online were collected.

The final list of WWTPs was restricted to include facilities accounting for 90% of flow (based on design flow) in each basin of interest. The design flows for all WWTPs in each basin area were collected and summed. Starting with facilities having the largest design flow, WWTPs were added to the final list until 90% of the total design flow was included (see Appendix Table 8 for flow rate by facility). WDRs unavailable online were collected directly from the Regional Water Boards. Monthly and annual water quality monitoring reports (SMRs) were provided by the Central Coast Regional Water Board for all required facilities. SMRs for Central Valley facilities were reviewed at the Central Valley Regional Water Board office in Fresno and nitrogen levels in discharge were extracted from these reports on site. To ensure current information and to fill data gaps, WWTPs were surveyed via email and telephone. Available biosolids information was collected through communications with individual facilities and through contact with Lauren Fondahl from U.S. EPA Region 9.

Adminsitrative Record
Page 34458

For FPs in the Central Valley, information was extracted from a database developed as part of the Hilmar Supplemental Environmental Project (Hilmar SEP) by Hydrogeophysics, Inc. (Rubin et al. 2007; Sunding & Berkman 2007; Sunding et al. 2007). The Hilmar database is based on WDRs and monitoring reports filed with the Central Valley Regional Water Board from 2003 to 2005. WDRs and monitoring data were provided by the Central Coast Regional Water Board, as available, for FPs in the Salinas Valley.

Collected information includes: population served (WWTPs); design flow and actual flow; relative flow to recharge basins, surface water and irrigated agriculture; seasonal variation in flow and nitrogen levels; acreage of irrigated agriculture and/or percolation basins; nitrogen concentration in discharge (ammonia, organic nitrogen, nitrate, TKN, and total nitrogen, as available); fate and volume of biosolids; and treatment for nutrient removal (if any). Forty WWTPs and 132 FPs were included in the analysis (Figure 42). The information collected for these facilities was used to approximate N loading for 100% of WWTPs and FPs by calculating the percent of facilities for which information was collected and scaling up total N loading to account for 100% of facilities. Discharge to surface water was excluded except when specifically listed as a direct irrigation source. See Technical Report 4 (Boyle et al. 2012) for information on the relationship between surface water and groundwater in the study area.

Not all of the above information was available for all facilities; to fill data gaps, missing information was modeled based on the reported results of other facilities as follows:

- Unknown N concentration of discharge
    - FP: Correlation between N concentration in discharge and total flow by type of FP
    - WWTP: Correlation between N concentration in discharge and total flow of WWTP
- Unknown relative flow to recharge basins and irrigated agriculture
    - 50 – 50 split of flow to recharge basins and irrigated agriculture
- Unknown acreage of recharge basins and irrigated agriculture
    - Correlation between flow and acreage for recharge basins (WWTPs and FPs considered separately)
- Unknown total flow
    - Facilities were excluded from modeling and included only in total N loading estimates.

To assess historical N loading from WWTPs and FPs, applied nitrogen was scaled based on the ratio of county population in historical years (1945, 1960, 1975, 1990, 2005) and 2010 for each county (Department of Finance 2011; United States Census Bureau 2011; United States Census Bureau, as compiled and edited by Richard L. Forstall, Population Division, US Bureau of the Census, Washington DC 2011). These historical estimates were used in the Groundwater Nitrate Loading Model (GNLM) discussed in Section 3 of this report. Historical estimates of N loading from FPs were also assessed based on the historical change in annual N of specialty crops, also discussed in Section 3.

Loading from land application of biosolids was assessed using the assumption that the nitrogen content of biosolids is 3.3% as listed in (Metcalf & Eddy 2003). As a conservative estimate, biosolids reported as "wet" were assumed to be approximately 30% dry solids (United States Environmental Protection Agency 2000a); however, the solids content of dewatered (not dry) biosolids can vary considerably.

Adminsitrative Record
Page 34459

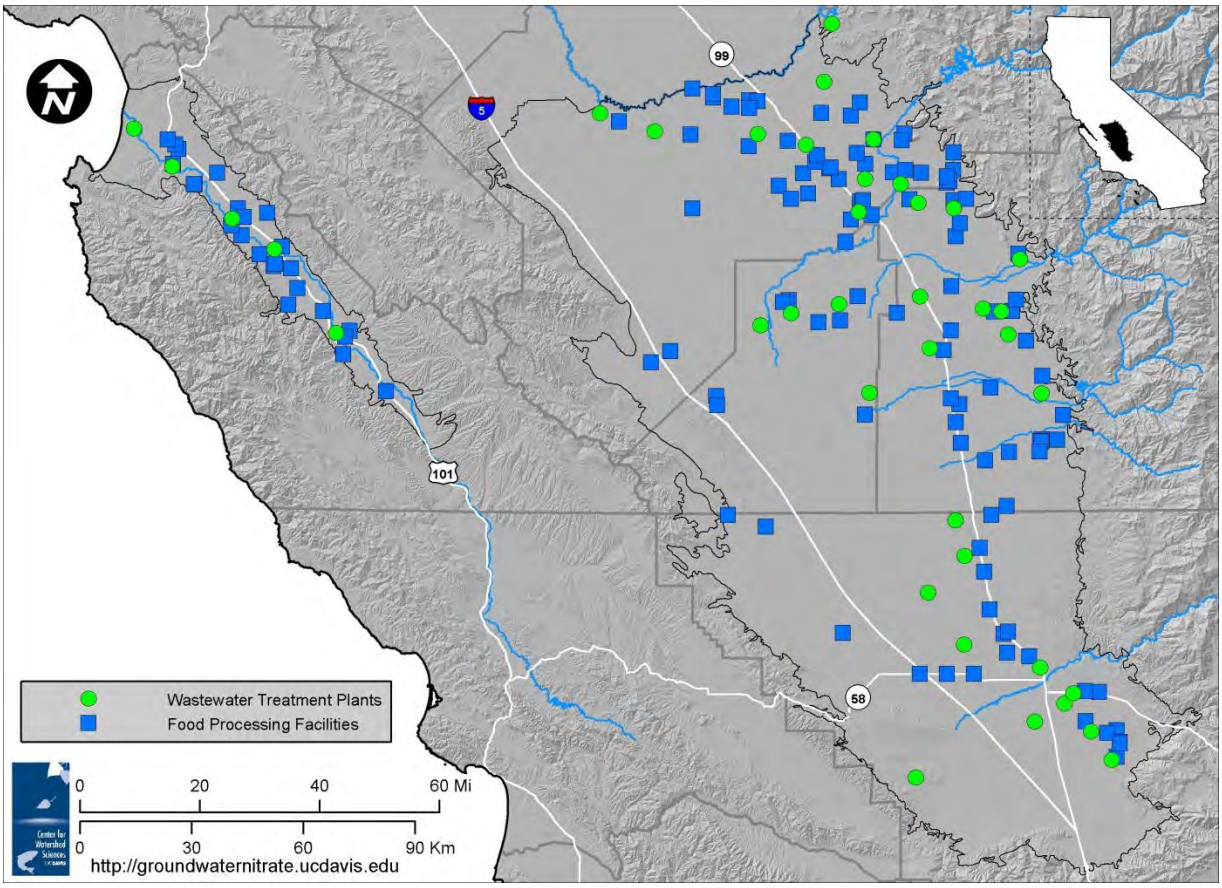


Figure 42. Location of included wastewater treatment plants and food processing facilities in the Tulare Lake Basin and Salinas Valley (FPs with an active discharge permit and the largest WWTPs comprising 90% of design flow in each basin were included in this analysis, see Appendix Table 8 **for facility specific information). (Source: California Water Boards.)**

6.2.2.2 Modeling of Nitrogen Leaching to Groundwater

Applied nitrogen data were collected for the above facilities. Given crop type, acreage, volume and nitrogen concentration, the leaching fraction can be modeled. With application of discharged nitrogen at rates less than or equal to plant uptake rates, replacing all, or a portion of chemical fertilizer application, reuse of discharge waters may have no detrimental impact on groundwater supplies (with respect to nitrogen). For the modeling of nitrogen leaching from WWTP and FP discharges, nitrogen load estimates were assigned to parcels in the vicinity of the facilities based on appropriate crop type (generally fiber, animal feed and fodder crops). Suitable parcels were selected until the approximate total land area for discharge was reached for each facility. N loading to groundwater via percolation basins was also spatially assigned with the selection of parcels or a portion of a parcel for each recharge area. Lastly, the land area of biosolids application was assigned an appropriate N loading rate by attributing estimated loading rates to parcels either specifically identified as application sites or estimated to be approximate application sites based on proximity to the facility and crop type. Additional information on the methodology for the estimation of nitrogen leaching to groundwater is included in Section 2.6. The following results and discussion refer to the data collected and estimated *applied* nitrogen from these sources.

Adminsitrative Record
Page 34460

6.2.2.3 Nitrogen Loading – Results and Discussion

Summary information for overall N loading from waste discharging facilities is presented and discussed below, followed by separate sections for WWTPs and FPs with a more detailed characterization of associated N loading. There are, however, sources of uncertainty in this analysis to be considered. These include:

- Effluent nitrogen monitoring data were not available for all facilities.
- The service population of WWTPs was not always available resulting in an estimation of population served from various sources, some of which may be outdated.
- When information was unavailable from the most reliable source, information from alternative sources was used to fill data gaps. For some facilities available information was limited or completely unavailable. The reliability and accuracy of data varied with source (from most certain to least certain):
    - From recent monitoring reports and direct contact with facilities
    - From recent monitoring reports and recent WDRs
    - From recent WDRs
    - From old WDRs
    - Modeling to fill data gaps (see Section 6.2.2.1)
    - No data available
- Small WWTPs (the WWTPs representing the final 10% of flow) were excluded from data collection to focus data collection efforts and to account for the largest nitrogen sources.
- Data for facilities operating with old permits may be outdated and data were unavailable for some facilities with pending permits.
- Effluent nitrogen levels were the focus of this analysis to determine the relative contribution of facilities to N loading; however, there is uncertainty in the estimation of leached nitrogen levels from applied nitrogen levels.
- In the surveying of WWTPs, some facilities indicated that additional fertilizer may be applied to supplement the nitrogen in land applied discharges. The extent of such practices and the impact to groundwater are unknown.
- Regarding the estimation of N loading from the land application of biosolids, the nitrogen content of biosolids varies (2 – 10%). Unless reported otherwise, the nitrogen content of biosolids was assumed to be approximately 3.3%, in accordance with Metcalf & Eddy (2003).
- The impact of evaporation and surface water recharge to ground water were excluded.
- N loading was assessed based on annual averages (of flow and N concentration). Seasonal variation may be a significant factor in the N loading from WWTP and FP facilities due to changes in applied water characteristics as well as irrigation and fertilization practices.

6.2.2.4 Summary of Results – Wastewater Treatment Plants and Food Processors

A total of 40 WWTPs accounts for 90% of WWTP flow within the study area. There is a total of 132 FPs within the study area; however, only 83 FPs are included in the detailed N loading analysis. A portion of the FPs is not actually required to report N information due to the expectation that nitrogen levels in

Adminsitrative Record
Page 34461

discharge will not lead to degradation of groundwater, with respect to nitrate. Some facilities, for which nitrogen monitoring data are not available, are granted a waiver of waste discharge requirements (WDRs).[31] Nitrogen data for 36 WWTPs and 63 FPs were collected. Modeling of the nitrogen content of discharge was necessary for 10% of WWTPs (4 out of 40 facilities) and 24% of FPs (20 out of 83 facilities). Thirty-seven percent of the total number of FPs (49 out of 132 facilities) could not be modeled due to insufficient information.

It is important to note that all of the current WWTP and FP nitrogen information is reported as applied levels rather than leached levels. Nitrogen reaching groundwater must be modeled based on land application method and crop type (for application to irrigated agriculture). Theoretically, if all discharged nitrogen from WWTPs and FPs were applied to land at rates less than or equal to plant uptake rates, then there would be no impact to groundwater from these facilities (with respect to nitrogen). Flow, nitrogen, and discharge details are listed by facility for all included WWTPs and FPs in Appendix Table 8; to match facilities with locations, WWTPs and FPs are numbered in Appendix Table 1 and Appendix Table 2 respectively.

Nitrogen application data can be viewed in several ways, each important for different reasons:

- The total mass of nitrogen applied (Table 43) is examined for comparison with fertilizer application and total N loading from other sources county- and basin-wide. This is important for a regional overview of N loading. A greater mass of applied nitrogen does not necessarily indicate a greater risk of contamination. For example, application of 2,500 metric tons of nitrogen over 50,000 acres with a total nitrogen concentration of 2 mg/L would not pose a threat to groundwater; however, application of 2 metric tons of nitrogen over 0.25 acres with a concentration of 500 mg/L could pose a significant threat on a local scale.

- The average application rate of nitrogen (kg/ha/yr) (Table 45) is examined for comparison with fertilizer application rates and total N loading from other sources, as well. This enables an assessment of the over-application of nitrogen; for high demand crops, a rough estimate of required nitrogen is 250 kg/ha/yr (~225 lbs/acre/yr), or 500 kg/ha/yr (~450 lbs/acre/yr) for double cropping.[36] Facilities exceeding this application rate risk contributing to nitrate contamination of groundwater. This is important both regionally and locally to pinpoint hot-spots and locate facilities that may require additional treatment or altered land application practices.

- The concentration of nitrogen in land applied discharge (Table 46) is examined to assess the potential for nitrate contamination, especially for discharge to percolation basins. In a worst case scenario, assuming direct recharge of groundwater from percolation basins with all nitrogen converting to nitrate and no denitrification, discharged nitrogen levels would be leached nitrate levels. Locally, this can be a great concern prior to migration and dilution in the aquifer.

[36] This is a rough estimate for high demand crops and is based on crop nitrogen demand for single and double cropping as discussed in Section 3.

Adminsitrative Record
Page 34462

Summary information, regarding applied nitrogen, is listed below (Table 43) by basin, county and across the entire area of interest. Total N applied for WWTPs was scaled up from the total for facilities representing 90% of flow (based on design flow) to estimate total N applied for all WWTPs in the study area. N data were collected or modeled for approximately 63% of FPs; totals are listed below for facilities reporting and separately scaled up to estimate total N applied for all FPs. These scaled up values were determined by incrementing the total N applied to reach 100% of facilities in each county. (Note: the latter is a maximum estimate and is likely an overestimation. Some facilities missing N information are not required to report because they are not considered a risk.) Total N applied from WWTPs is greatest in Fresno County, while total N applied from FPs is greatest in Kern County. Across the study area, nitrogen applied from WWTP effluent exceeds that from FPs by a factor of 3.2 (based on estimated totals). However, as previously mentioned, a greater mass of applied nitrogen does not necessarily indicate a greater risk of contamination; the land area over which WWTP effluent is applied far exceeds that of FP discharges and FP discharges are generally more concentrated (discussed below). Kings County deviates from the overall study area with a ratio of ~0.65 (total nitrogen applied from WWTP effluent to that from FP effluent); this is primarily due to the limited number of WWTPs in Kings County.

Table 43. Metric tons (Mg) of N applied annually in facility discharge (2010). [1 Mg = 1 metric ton = 1.1 tons.]

	WWTP (90% of flow)	WWTP (est. 100%)	FP (63% of facilities)	FP (est. 100%)
By County	Mg N/yr	Mg N/yr	Mg N/yr	Mg N/yr
Fresno	2,423	2,693	348	470
Kern	920	1,022	455	640
Kings	158	176	167	261
Tulare	764	849	100	149
Monterey	279	310	15	71
By Basin				
Tulare Lake Basin	*4,265*	*4,740*	*1,070*	*1,520*
Salinas Valley	*279*	*310*	*15*	*71*
Total	***4,544***	***5,050***	***1,085***	***1,591***

Note: Solids not included. Biosolids are discussed separately below. Due to insufficient data, application of FP solids is excluded from this analysis.

Historical application of nitrogen from WWTPs and FPs was estimated based on population change; estimated nitrogen application from WWTP and FP discharges in 1945, 1960, 1975, 1990, and 2005 is listed in Table 44. Back-casting of applied nitrogen using population as the scaling factor follows the same distribution pattern as above for the current time frame, scaled by the percent of current population for each year listed. These historical estimates were used in the Groundwater Nitrate Loading Model (GNLM) discussed in Section 3 of this report. Historical estimates of N loading from FPs were also assessed based on the historical change in annual N of specialty crops, also discussed in Section 3, resulting in a similar trend, with lower estimates for 1945 and a steeper increase in the past

Adminsitrative Record
Page 34463

20 years. Estimated historical applied nitrogen is provided only as reference and is based solely on the change in population between 2010 and previous years. Actual historical application may vary significantly from the estimates listed here as population is not the only factor affecting land applied nitrogen levels. Management of discharge from WWTPs and FPs varied significantly throughout the 1900's, based on numerous factors.

Table 44. Estimated metric tons (Mg) of N applied historically in facility discharge based on population change by county (WWTPs and FPs) and on change in specialty crop N (only FPs) between 1945 and 2010. [1 Mg = 1 metric ton = 1.1 tons.]

	Wastewater Treatment Plants				
	1945 (est. 100%)	***1960 (est. 100%)***	***1975 (est. 100%)***	***1990 (est. 100%)***	***2005 (est. 100%)***
By County	Mg N/yr	Mg N/yr	Mg N/yr	Mg N/yr	Mg N/yr
Fresno	737	1,059	1,292	1,932	2,511
Kern	249	355	433	662	910
Kings	51	57	79	117	165
Tulare	281	323	410	599	775
Monterey	88	148	200	266	303
By Basin					
Tulare Lake Basin	1,318	1,795	2,214	3,309	4,360
Salinas Valley	*88*	*148*	*200*	*266*	*303*
Total	***1,406***	***1,944***	***2,414***	***3,575***	***4,663***
Note: Solids not included. Biosolids are discussed separately below.					

	Food Processors				
	1945 (est. 100%)	***1960 (est. 100%)***	***1975 (est. 100%)***	***1990 (est. 100%)***	***2005 (est. 100%)***
By County	**Mg N/yr Population Basis (Mg N/yr Specialty Crop Basis)**				
Fresno	129 (43)	185 (71)	225 (129)	337 (244)	438 (413)
Kern	156 (58)	223 (96)	271 (175)	414 (332)	570 (563)
Kings	76 (24)	85 (39)	117 (71)	173 (135)	244 (230)
Tulare	49 (14)	57 (22)	72 (41)	105 (77)	136 (131)
Monterey	20 (6)	34 (11)	46 (19)	61 (37)	69 (62)
By Basin					
Tulare Lake Basin	*409 (139)*	*549 (228)*	*686 (416)*	*1030 (788)*	*1388 (1337)*
Salinas Valley	*20 (6)*	*34 (11)*	*46 (19)*	*61 (37)*	*69 (62)*
Total	***429 (145)***	***583 (239)***	***732 (435)***	***1090 (825)***	***1,457 (1,399)***
Note: Solids not included. Due to insufficient data, application of FP solids is excluded from this analysis.					

The current annual average kg N applied to irrigated agricultural crops and to percolation basins for WWTPs and FPs is listed in Table 45 by county and basin. The significantly higher values for percolation

Adminsitrative Record
Page 34464

basins are a product of small land area and are provided only for reference; applied concentration is a more important indicator of risk for recharge of groundwater (Table 46). Based on the required nitrogen estimate of 250 kg/ha/yr for high demand crops (assuming no double cropping), Tulare and Kern County averages indicate potential application of N above agronomic rates from WWTP discharge.

Table 45. Annual average N (kg N/ha/yr) discharged to irrigated land and percolation basins from WWTPs and FPs (averaged across all sites in each region) and corresponding total hectarage. [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

	WWTP Irrigation		***WWTP Percolation***		***FP Irrigation***		***FP Percolation***	
	kg N/ha/yr	Ha	kg N/ha/yr	Ha	kg N/ha/yr	Ha	kg N/ha/yr	Ha
By County	AVERAGE	TOTAL	AVERAGE	TOTAL	AVERAGE	TOTAL	AVERAGE	TOTAL
Fresno	166	1,673	1,932	1,028	155	3,048	306	61
Kern	255	8,259	754	130	200	2,389	797	93
Kings	27	7,183	330	240	121	1,308	42	8
Tulare	314	2,113	1,189	389	164	599	2,424	99
Monterey	177	4,917	1,163	176	24	258	151	23
Basin								
TLB	225	19,183	1331	1,788	163	7,344	1224	260
SV	177	4,917	1163	176	24	258	151	23
Overall	**220**	**24,100**	**1308**	**1,964**	**158**	**7,602**	**1137**	**283**

Note: Solids not included. Biosolids are discussed separately below. Due to insufficient data, application of FP solids is excluded from this analysis.

Table 46. Average N concentration (mg/L) in discharge to irrigated land and percolation basins from WWTPs and FPs.

	WWTP Irrigation	***WWTP Percolation***	***FP Irrigation***	***FP Percolation***
By County	mg/L N	mg/L N	mg/L N	mg/L N
Fresno	16.3	18.5	101.5	56.2
Kern	20.3	17.7	36.7	43.9
Kings	9.5	11.2	63.4	2.1
Tulare	15.3	14.9	35.1	34.2
Monterey	9.7	13.9	24.9	22.1
By Basin				
Tulare Lake Basin	*17.3*	*16.3*	*70.7*	*43.3*
Salinas Valley	*9.7*	*13.9*	*24.9*	*22.1*
Overall Average	***16***	***16***	***69***	***42***

Note: Solids not included. Biosolids are discussed separately below. Due to insufficient data, application of FP solids is excluded from this analysis.

Adminsitrative Record
Page 34465

Average concentrations in discharge to irrigated land and percolation basins from WWTPs and FPs are listed in Table 46. It is assumed that agricultural crops utilize the nitrogen in discharges used for irrigation. Concentrations of FP discharge applied as irrigation are significantly higher than those of WWTPs. The concentration of effluent discharged as direct groundwater recharge (percolation) can be a concern above 10 mg/L N. The significantly higher nitrogen concentrations of FP effluent discharged to percolation basins throughout most of the study area is of the greatest concern. In comparison with other N loading sources, the contribution of WWTPs and FPs is less significant on a basin-wide scale (refer to a table comparing all N sources); however, to avoid impacting groundwater nitrogen levels, the discharges must be properly managed.

6.2.2.5 Wastewater Treatment Plants – Results and Discussion

The following information is based on WWTP data collected and modeled for the top 90% of flow. Total annual effluent nitrogen relative to population served is illustrated in Figure 43; there is a direct correlation between population and nitrogen load with some variability for facilities treating combined domestic and industrial wastes. The facilities in red are those discharging the greatest number of metric tons of nitrogen per year. Population served by each WWTP is indicated by the diameter of each marker. WWTPs serving the largest population generally discharge the greatest amount of nitrogen. However, flow increases with population served as well, therefore, plants discharging the greatest total nitrogen annually are not necessarily discharging higher concentrations of nitrogen.

Forty percent of the reporting WWTPs discharge to both percolation basins and irrigated agriculture; 32.5% of wastewater facilities discharge only to percolation basins and 27.5% of wastewater facilities discharge only to irrigated agriculture. The relative land area and nitrogen applied to percolation basins versus irrigated agriculture are compared in Figure 44 (TLB) and Figure 45 (SV). Applied nitrogen is listed as concentration in mg/L as N for percolation basins to account for the possibility of direct recharge, while total annual metric tons of N applied is listed for irrigated agriculture to account for plant uptake. Regarding discharge to percolation basins, yellow, orange, and red markers indicate total nitrogen concentrations above the nitrate MCL. Regarding discharge to irrigated agriculture, yellow, orange, and red markers indicate more significant contributors to total mass loading. Acres of percolation basins and irrigated agriculture are indicated by marker diameter. Note the different land area scale; the total area of land application to percolation basins and irrigated agriculture is ~1,960 ha (~4,850 acres) and 24,100 ha (~59,550 acres), respectively. Highly concentrated discharge to percolation basins over many acres (larger, yellow to red markers on the left) indicates an increased likelihood of contributing to nitrate contamination. Greater total N applied to few acres of irrigated agriculture (smaller, yellow to red markers on the right) indicates an increased likelihood of contributing to nitrate contamination.

Adminsitrative Record
Page 34466

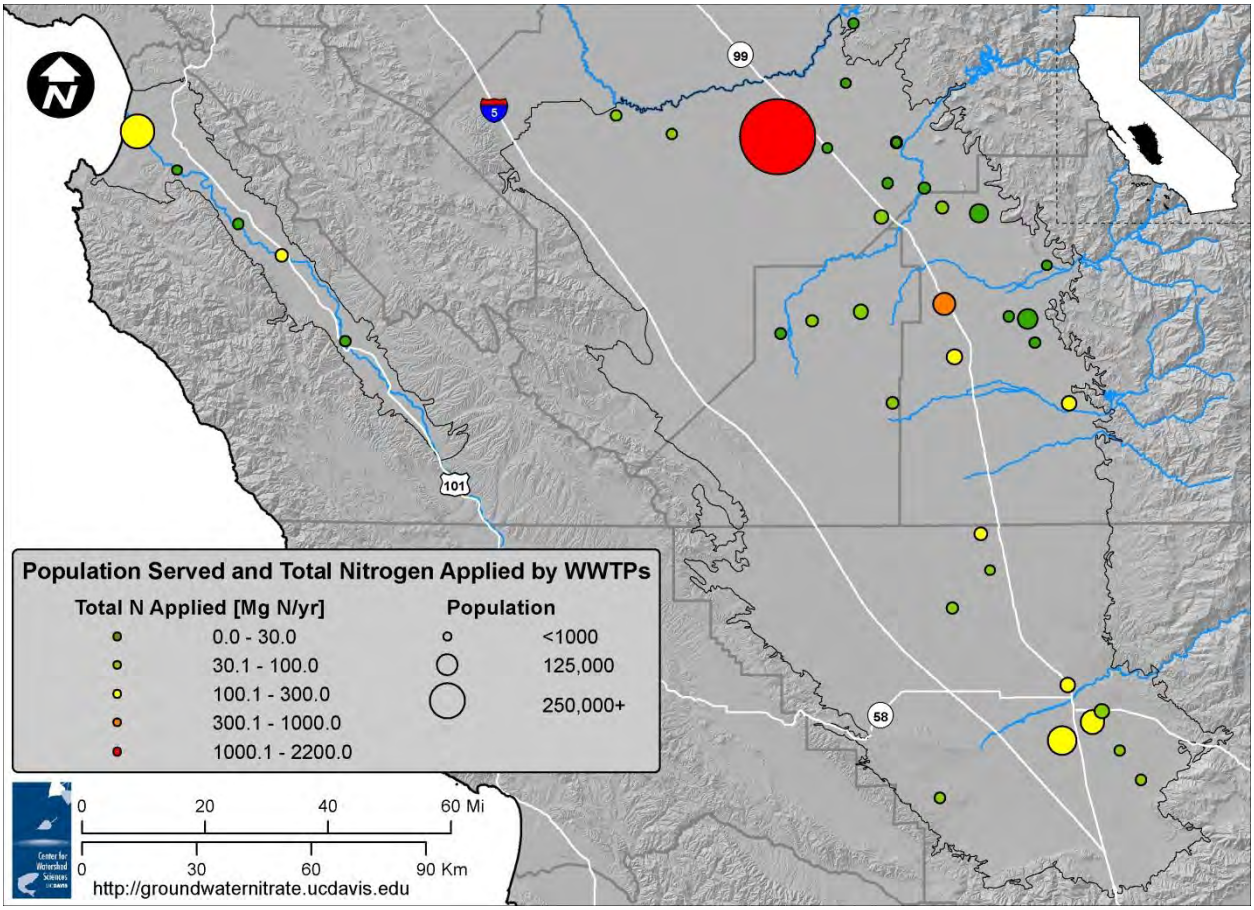


Figure 43. Wastewater treatment plants: Total applied nitrogen (metric tons N/Yr) [by color] and population served [by symbol diameter] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 Mg = 1 metric ton = 1.1 tons.]

Adminsitrative Record
Page 34467

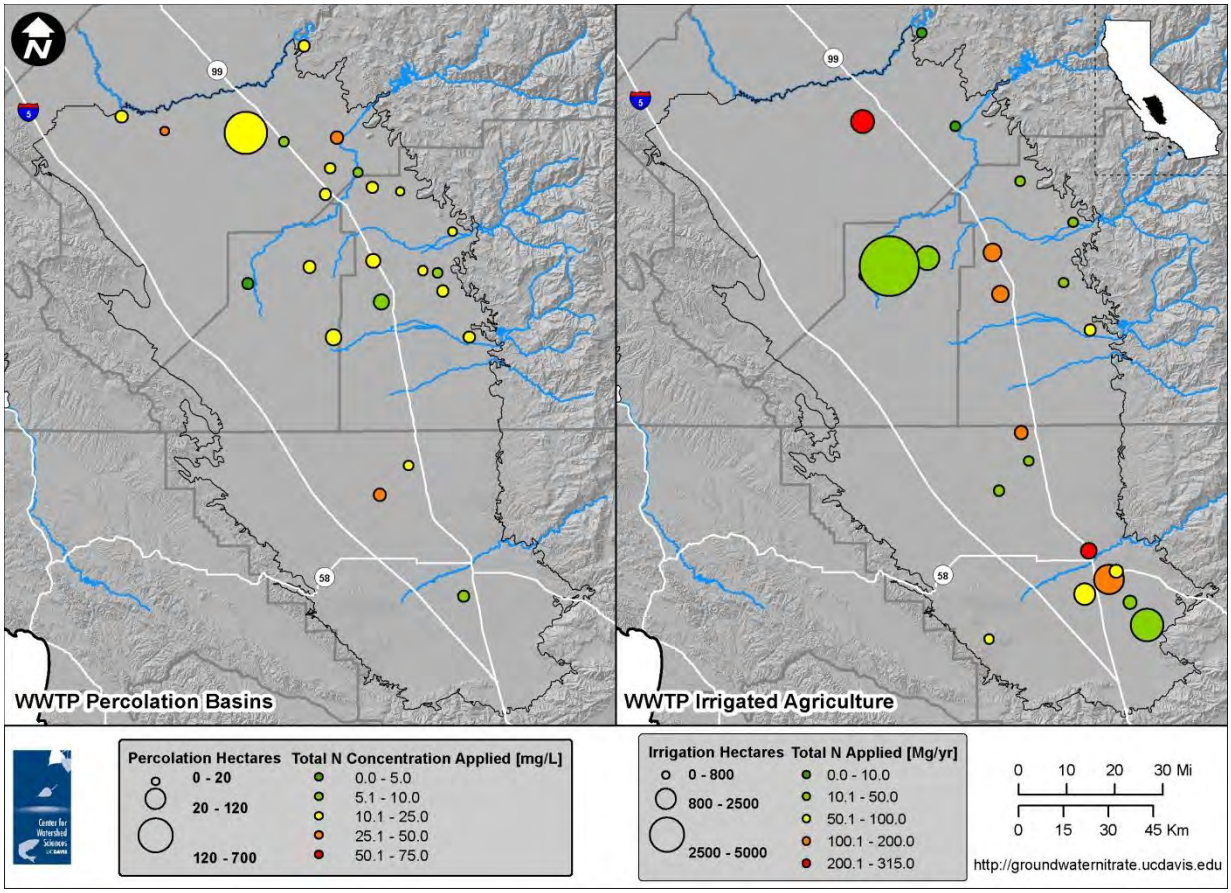


Figure 44. Tulare Lake Basin wastewater treatment plants: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) of total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 Mg = 1 metric ton = 1.1 tons, 1 hectare = 2.47 acres.]

Adminsitrative Record
Page 34468

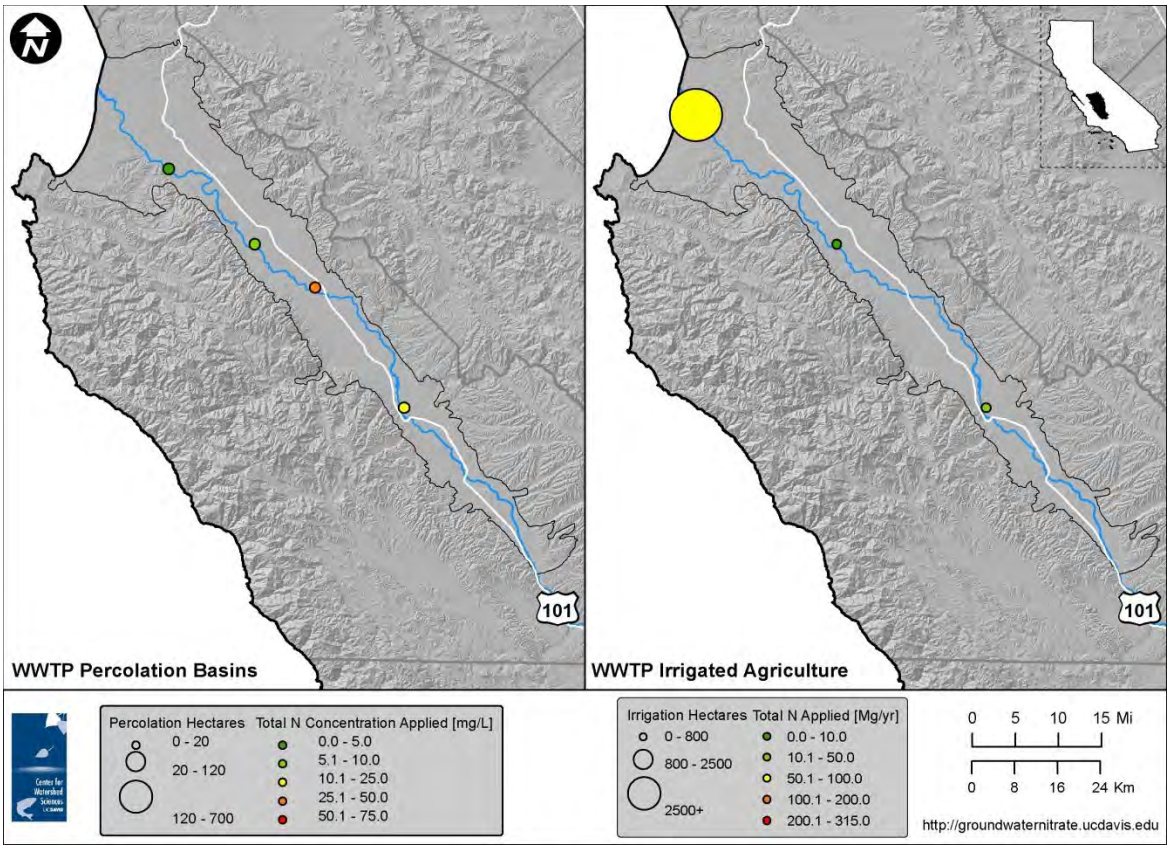


Figure 45. Salinas Valley wastewater treatment plants: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) of total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 Mg = 1 metric ton = 1.1 tons, 1 hectare = 2.47 acres.]

Figure 46 and Figure 47 illustrate the average kg/ha/yr of applied N from WWTP for comparison with fertilizer application rates and total N loading from other sources for the Tulare Lake Basin and Salinas Valley, respectively. This enables an assessment of the over-application of nitrogen; for high demand crops, a rough estimate of required nitrogen is 250 kg/ha/yr (~225 lbs/acre/yr), or 500 kg/ha/yr (~450 lbs/acre/yr) for double cropping.[37] Facilities exceeding this application rate (marked in orange and red) risk contributing to nitrate contamination of groundwater. This is important both regionally and locally to pinpoint hot-spots and locate facilities that may require additional treatment or altered land application practices.

[37] This is a rough estimate for high demand crops and is based on crop nitrogen demand for single and double cropping as discussed in Section 3.

Adminsitrative Record
Page 34469

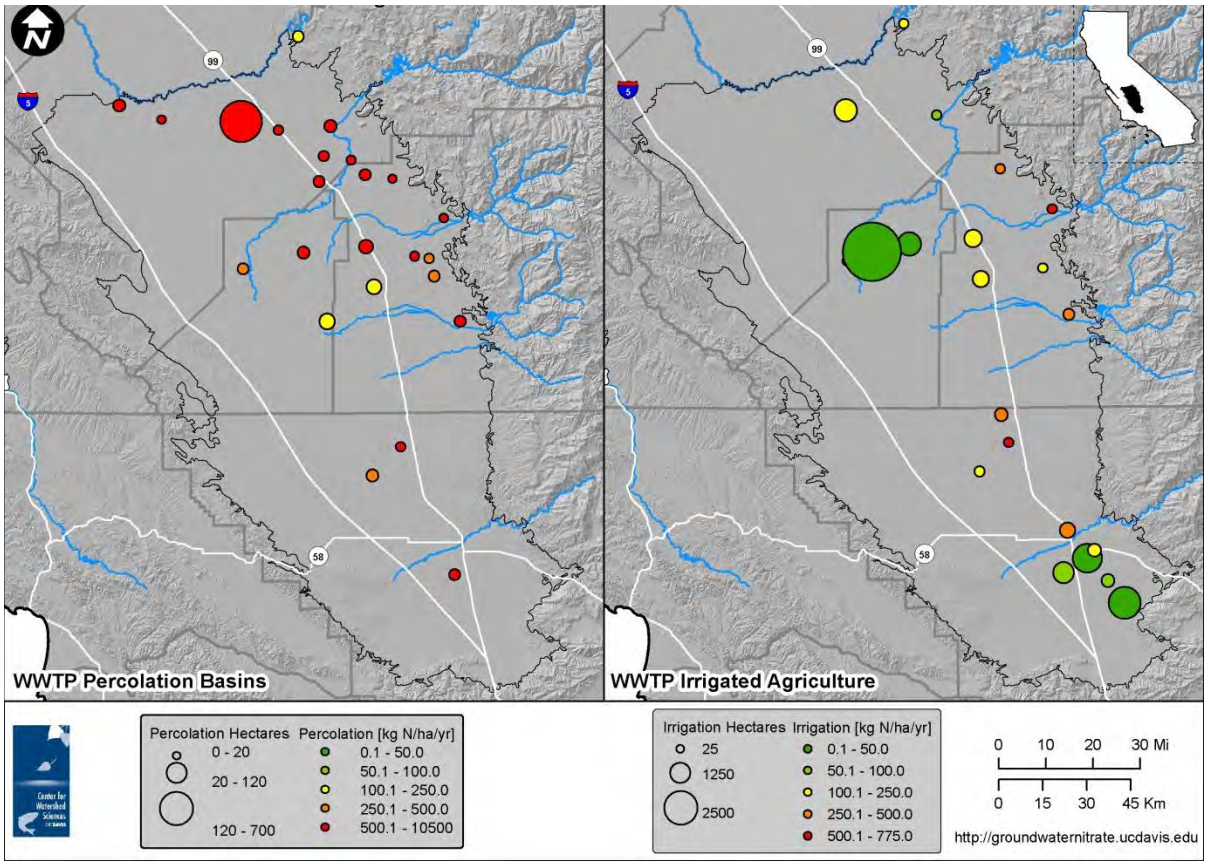


Figure 46. Tulare Lake Basin wastewater treatment plants: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

Adminsitrative Record
Page 34470

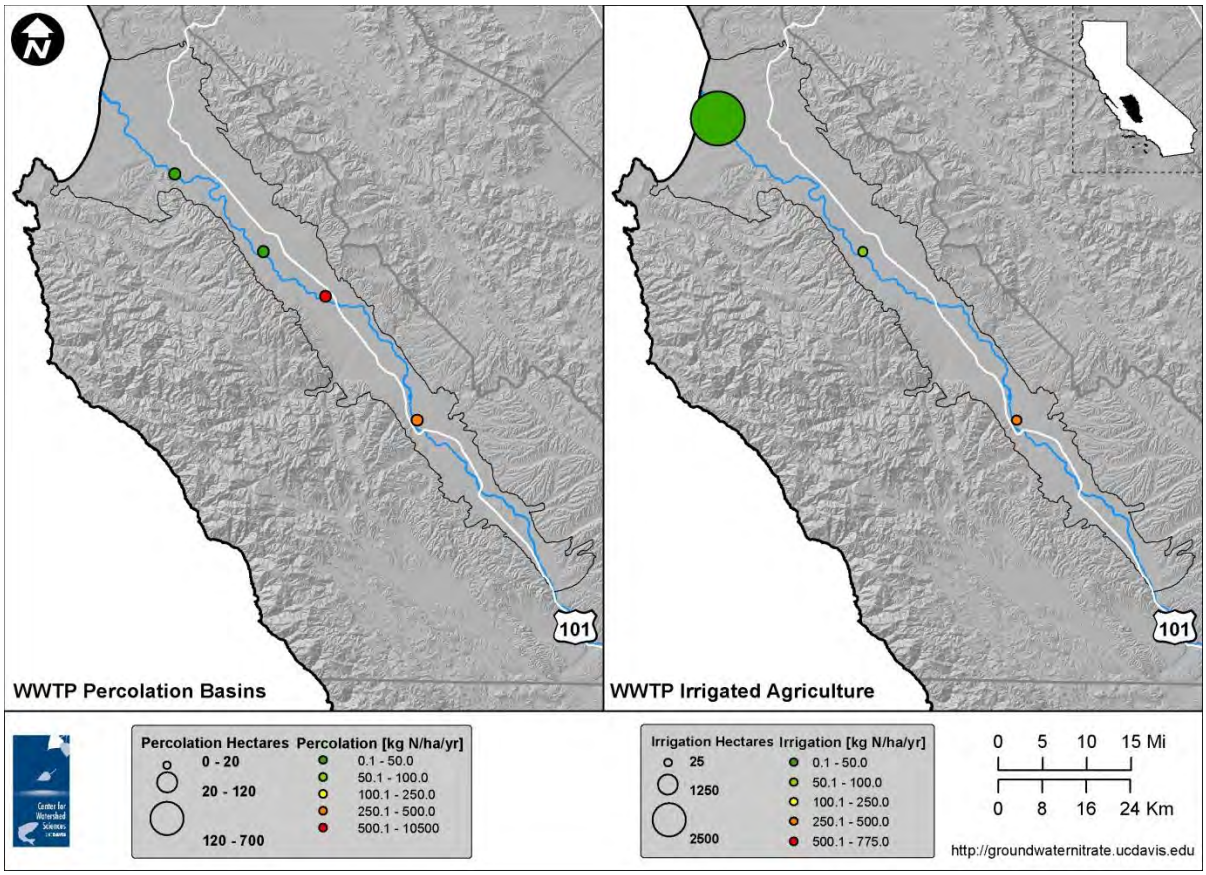


Figure 47. Salinas Valley wastewater treatment plants: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

Adminsitrative Record
Page 34471

6.2.2.6 Biosolids – Results and Discussion

Reported annual tons of produced biosolids from surveyed WWTPs are listed in Figure 48. Larger facilities process more wastewater and generally produce a greater amount of biosolids. The red markers represent the largest facilities (Fresno, Monterey, Visalia, Tulare, and Bakersfield plants).

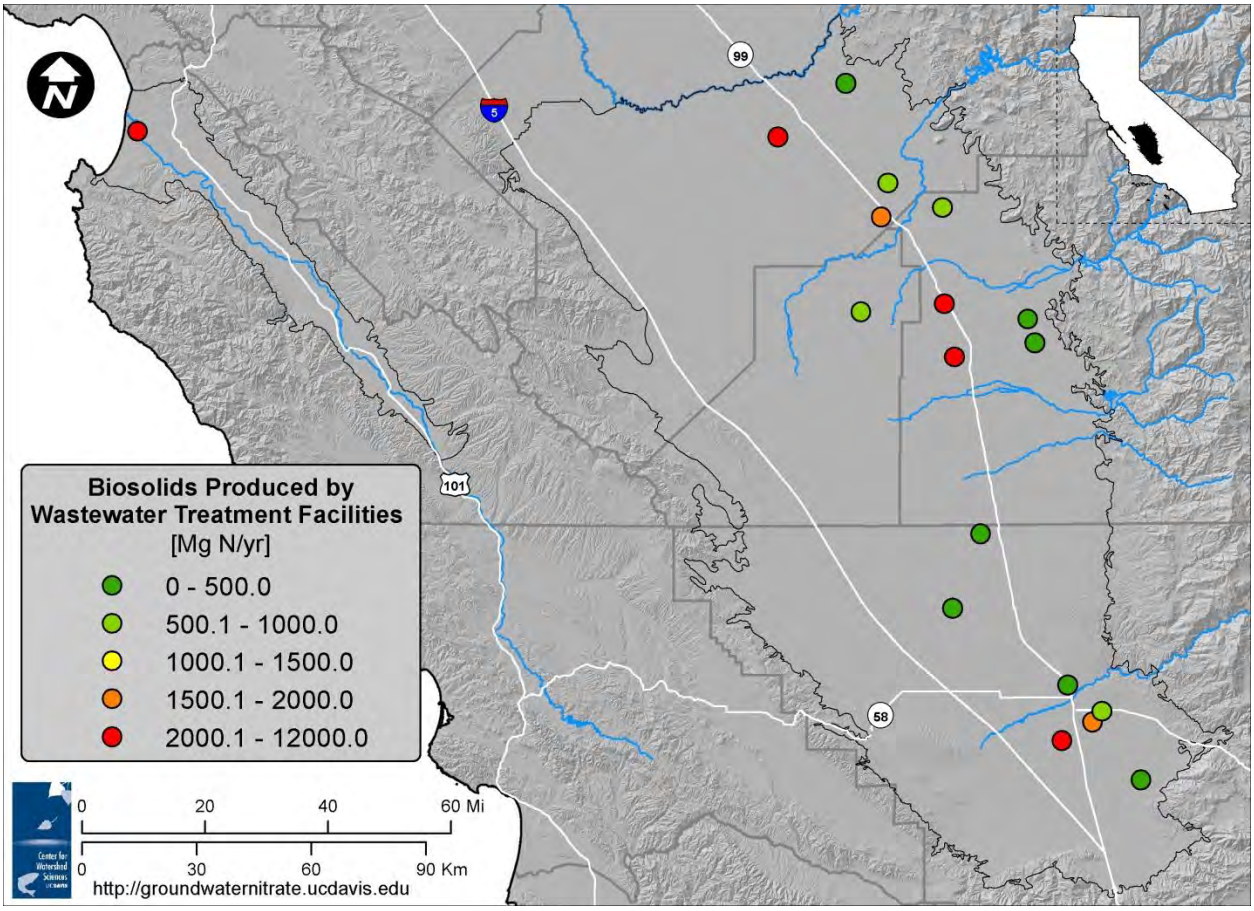


Figure 48. Annual metric tons of biosolids produced by surveyed WWTPs (~50% of facilities reporting, see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs.) [1 metric ton = 1.1 tons.]

Based on data reported to the U.S. EPA (United States Environmental Protection Agency 2008a) and collected through this analysis, Table 47 lists estimated total biosolids produced by county. The U.S. EPA estimates (column 1) include biosolids prepared by WWTPs and composting facilities; there are several large composting facilities within the study area which import biosolids from other counties, increasing the total. Small WWTPs (< 1 mgd flow) are not required to report biosolids information to the U.S. EPA and are therefore excluded from the totals listed in column 1. Column 2 lists total reported biosolids produced by WWTPs included in the analysis herein. Column 3 lists the estimated total biosolids produced by WWTPs in the study area; values of column 2 have been scaled up based on flow to estimate the total for 100% of wastewater flow in the study area.

Adminsitrative Record
Page 34472

Table 47. Estimated metric tons (Mg) of biosolids produced or prepared annually. [1 Mg = 1 metric ton = 1.1 tons.]

	[1] Biosolids Mg/yr (U.S. EPA, 2008)		*[2] Biosolids Mg/yr (Reported)*		*[3] Biosolids Mg/yr (Estimated Total)*	
By County	Solids*	Nitrogen	Solids*	Nitrogen	Solids*	Nitrogen
Fresno	17,732	585	14,438	477	17,318	572
Kern	140,948**	4,651	77,825	2,568	96,910	3,198
Kings	1,200	40	998	33	1,680	55
Tulare	3,815	126	6,829	225	9,435	311
Monterey	5,210	172	4,808	159	6,803	225
Basin						
Tulare Lake Basin	163,695	5,402	100,090	3,303	125,343	4,136
Salinas Valley	5,210	172	4,808	159	6,803	225
Total	168,905	5,574	104,898	3,462	132,146	4,361

*By dry weight.
**Includes 3 large composting operations which import biosolids from outside the study area (United States Environmental Protection Agency 2008a).

With the above listed restrictions on the land application of biosolids (Section 6.2.1.7), a significant portion of biosolids is composted and not directly tracked. Significant amounts of biosolids are imported into the region for composting and/or land application and some composted biosolids are exported from the counties of interest (mainly Kern County). Facilities reporting direct land application of biosolids and large land application operations (including composted biosolids), are mapped in Figure 49. The total reported land applied biosolids nitrogen in the Tulare Lake Basin is 4,768 Mg N/yr with application in Kern County and Kings County accounting for 99% of the total (3,135 Mg N/yr and 1,588 Mg N/yr, respectively). Monterey County does not permit application of biosolids ; however, it is likely that some soil amendments imported into the county contain some biosolids. As with the application of liquid effluent from WWTPs, the land application of biosolids nitrogen at rates less than or equal to plant uptake rates is important to avoid impacting groundwater nitrate levels. Measures are enforced to ensure appropriate application rates and to avoid contamination in storage, processing and transport operations. The largest contributors to total N application are in red. Land area is indicated by marker diameter. Small red markers would be of greatest concern, indicating the highest category of metric tons over a smaller land area. The largest marker corresponds with a large biosolids application farm; however, the total metric tons of N applied does not necessarily indicate degradation of groundwater. In this instance the large amount of nitrogen is applied over a large area of land of ~9,000 ha (22,000 acres).

Adminsitrative Record
Page 34473

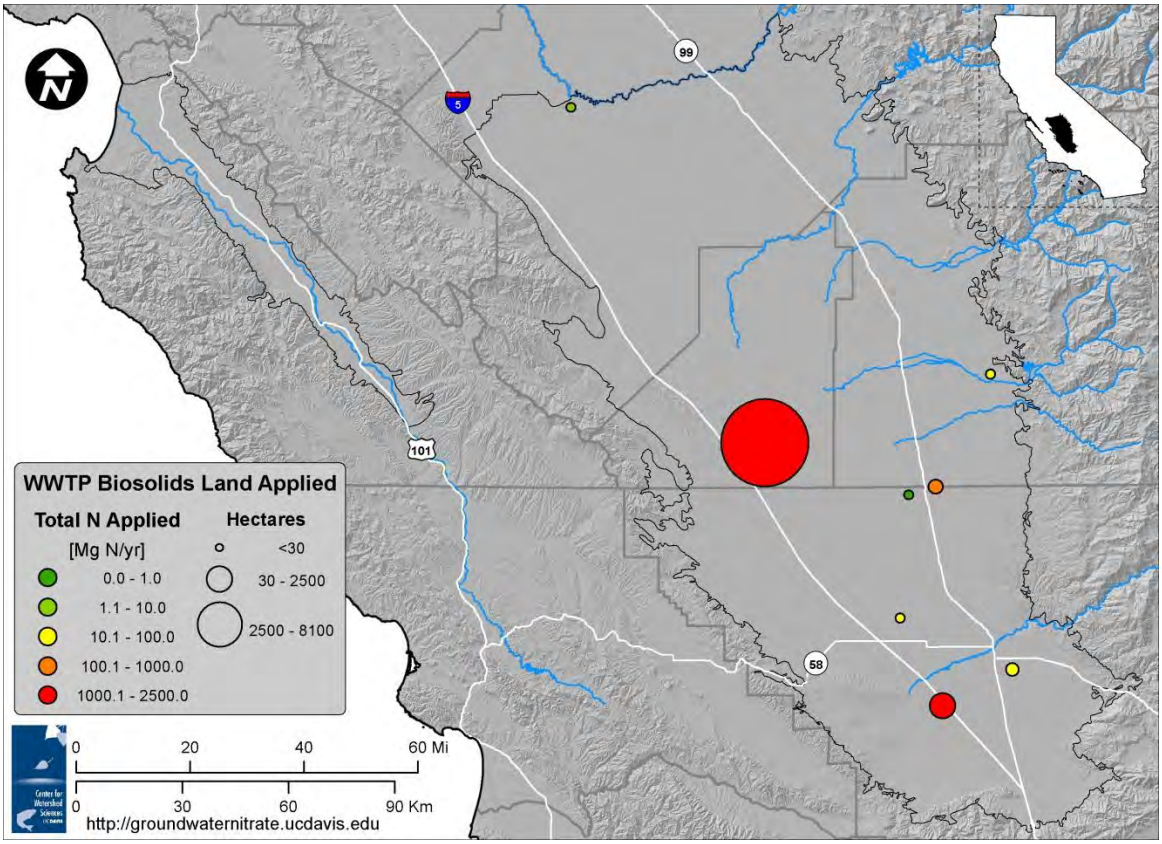


Figure 49. Large land application operations and WWTPs reporting direct land application of biosolids: Total annual mass of nitrogen [color] and hectarage in ha [diameter] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, U.S. EPA Region 9, Contact with Facilities, WDRs.) [1 metric ton = 1.1 tons, 1 hectare = 2.47 acres.]

Adminsitrative Record
Page 34474

Figure 50 illustrates the kg N/ha/yr of biosolids application for comparison with fertilizer application rates and total N loading from other sources. This enables an assessment of the over-application of nitrogen; for high demand crops, a rough estimate of required nitrogen is 250 kg/ha/yr (~225 lbs/acre/yr), or 500 kg/ha/yr (~450 lbs/acre/yr) for double cropping.[38] Facilities exceeding this application rate (marked in orange and red) risk contributing to nitrate contamination of groundwater. This is important both regionally and locally to pinpoint hot-spots and locate facilities that may require additional treatment or altered land application practices.

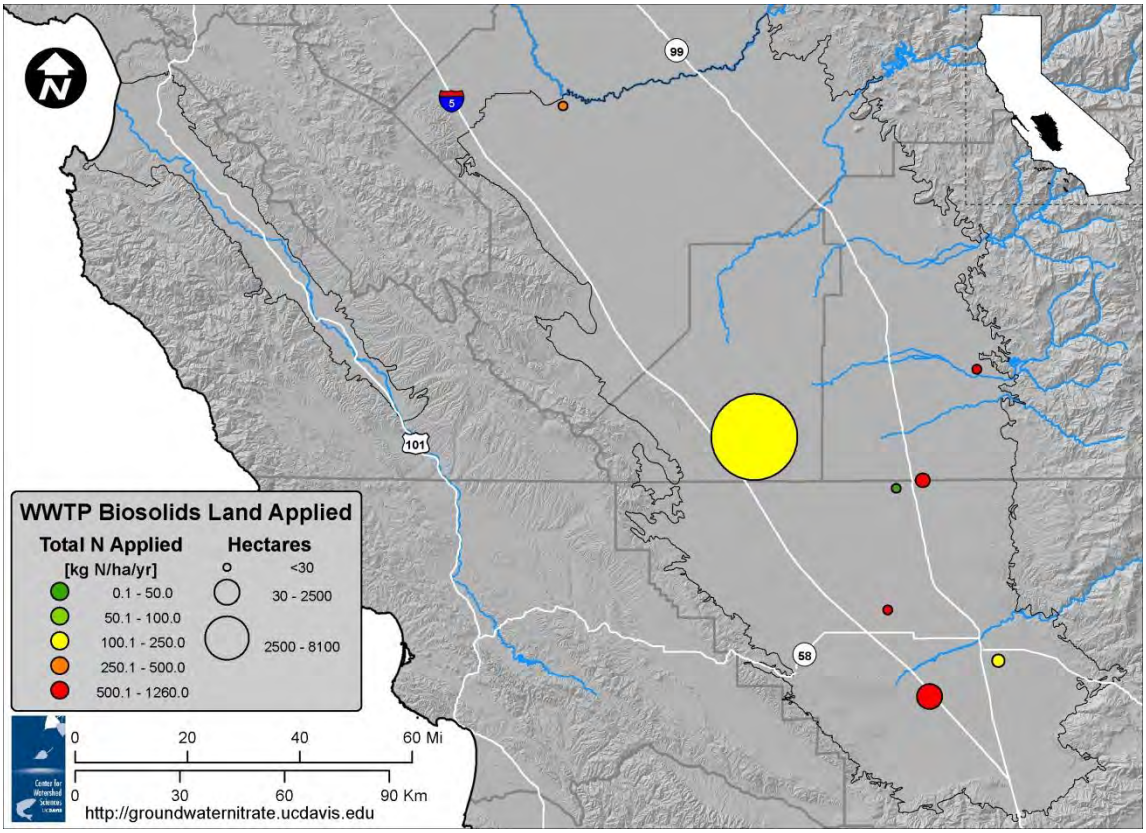


Figure 50. Large land application operations and WWTPs reporting direct land application of biosolids: kg N/ha/yr [color] and hectarage in ha [diameter] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, U.S. EPA Region 9, Contact with Facilities, WDRs.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

6.2.2.7 Food Processors – Results and Discussion

The following information is based on FP data collected and modeled, representing 63% of all food processing facilities in the study area. The type distribution of FPs is illustrated in Figure 51; two industrial WWTPs are included as well, because they receive a substantial amount of food processor discharges. Fruit and nut, winery, and vegetable operations account for 36%, 30%, and 10% of facilities, respectively.

[38] This is a rough estimate for high demand crops and is based on crop nitrogen demand for single and double cropping as discussed in Section 3.

Adminsitrative Record
Page 34475

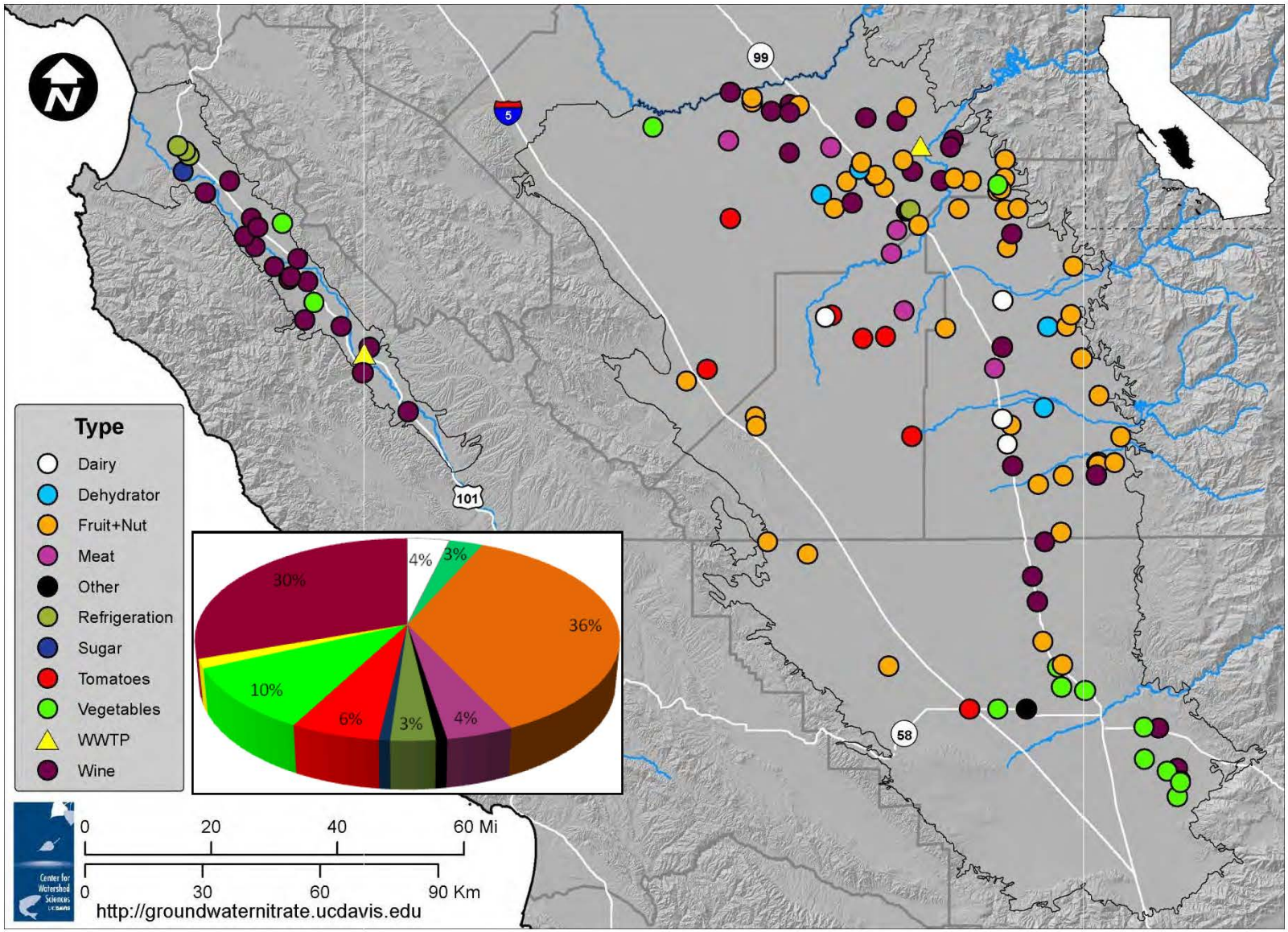


Figure 51. Food processor type distribution. The pie-chart shows the distribution of food processor type as a percentage of the total number of surveyed food processors (Source: California Water Boards, Geolocating by Address, WDRs.)

Adminsitrative Record
Page 34476

Discharge information by food processor type is listed in Table 48. Although the highest concentration of discharge reported is from a meat processing plant, this type of FP accounts for a very small portion of total flow across all facilities. However, the high concentration of nitrogen in discharge from this single meat processing plant is a concern. Despite lower effluent nitrogen concentrations, the largest contributors to total N applied from FPs are vegetable and tomato processing facilities, which contribute 22% and 29% of total N applied (Mg/yr), respectively.

Table 48. Nitrogen and flow characteristics by food processor type and basin for facilities with information available. [1 Mg = 1 metric ton = 0.001 Gg = 1.1 tons.]

ENTIRE STUDY AREA					
Type (# Facilities)	**Sum Total N (Mg/yr)**	**Average N (Mg/yr)**	**Sum Total Flow (mgd)**	**Average Flow (mgd)**	**Average [Total N] (mg/L)**
Fruit + Nut (43)	213.7	5.0	3.5	0.08	47.8
Wine (19)	157.1	8.3	1.8	0.09	64.0
Vegetables (8)	236.6	29.6	8.7	1.09	14.7
Tomatoes (6)	314.7	52.5	6.0	1.01	44.8
Meat (2)	71.0	35.5	0.1	0.04	520.0
Dairy (1)	7.6	7.6	0.3	0.25	22.0
Other (4)	83.7	20.9	1.3	0.32	21.0
TULARE LAKE BASIN					
Type (# Facilities)	**Sum Total N (Mg/yr)**	**Average N (Mg/yr)**	**Sum Total Flow (mgd)**	**Average Flow (mgd)**	**Average [Total N] (mg/L)**
Fruit + Nut (43)	213.7	4.97	3.5	0.08	47.8
Wine (18)	157.0	8.72	1.8	0.10	65.5
Vegetables (6)	224.0	37.3	8.0	1.33	15.8
Tomatoes (6)	314.7	52.5	6.0	1.01	44.8
Meat (2)	71.0	35.5	0.1	0.04	520.0
Dairy (1)	7.6	7.6	0.3	0.25	22.0
Other (2)	81.4	40.7	1.2	0.59	28.3
SALINAS VALLEY BASIN					
Type (# Facilities)	**Sum Total N (Mg/yr)**	**Average N (Mg/yr)**	**Sum Total Flow (mgd)**	**Average Flow (mgd)**	**Average [Total N] (mg/L)**
Wine (1)	0.050	0.050	0.001	0.001	36.4
Vegetables (2)	12.6	6.3	0.72	0.36	11.5
Other (2)	2.22	1.11	0.105	0.052	13.6

Nitrogen information was collected or modeled for approximately 63% of FPs; the corresponding total nitrogen applied annually across both basins is 1,085 metric tons (Figure 52).

Adminsitrative Record
Page 34477

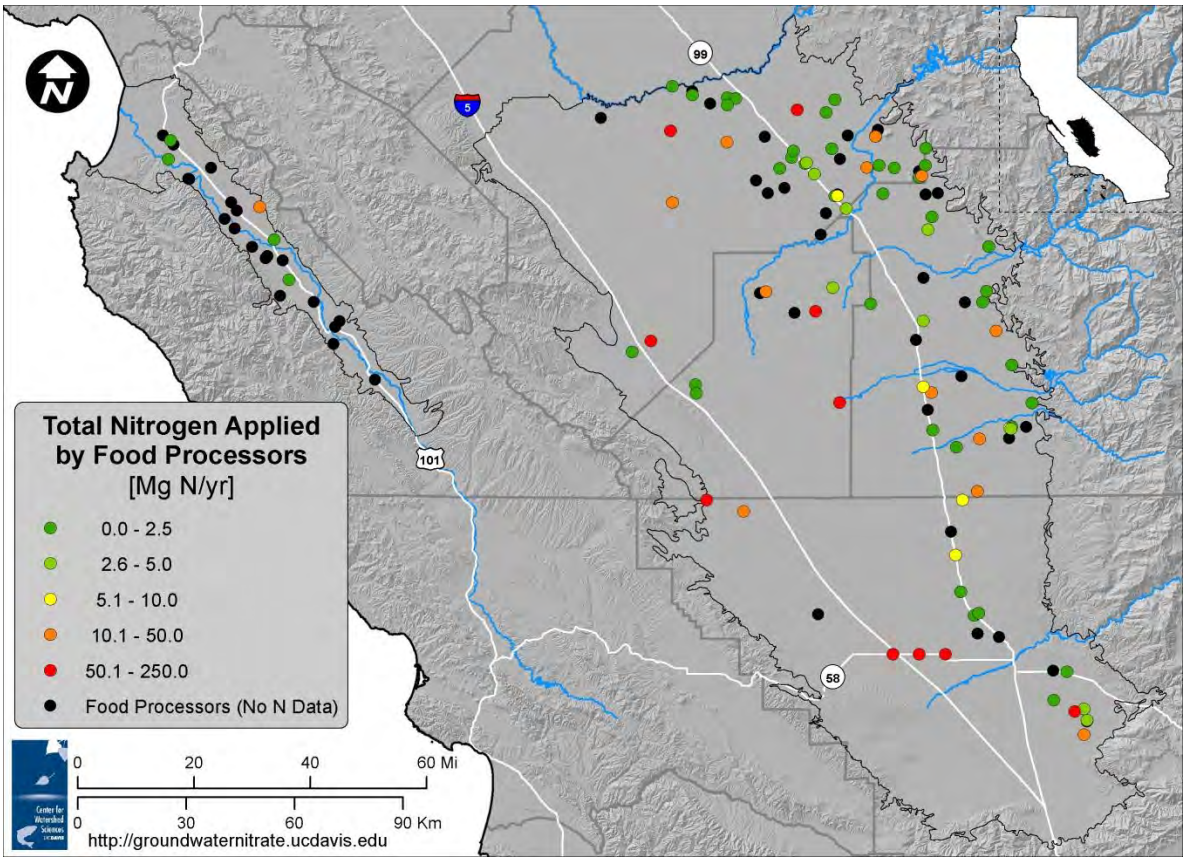


Figure 52. Total metric tons of nitrogen applied annually from food processor discharge (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [1 Mg = 1 metric ton = 0.001 Gg = 1.1 tons.]

Approximately 12.4% of FPs (reported or modeled) discharge to both percolation basins and irrigated agriculture; 33.3% of FPs discharge only to percolation basins and 54.3% of FPs discharge only to irrigated agriculture. The relative land area and nitrogen applied to percolation basins versus irrigated agriculture is compared in Figure 53 (TLB) and Figure 54 (SV). Applied nitrogen is listed as concentration in mg/L as N for percolation basins to account for the possibility of direct recharge while total annual metric tons of N applied is listed for irrigated agriculture to account for plant uptake. Regarding discharge to percolation basins, yellow, orange, and red markers indicate nitrogen concentrations above the nitrate MCL. Regarding discharge to irrigated agriculture, yellow, orange, and red markers indicate more significant contributors to total mass loading. Hectares of percolation basins and irrigated agriculture are indicated by marker diameter. Note the different land area scale; the total reported area of land application to percolation basins and irrigated agriculture is ~280 ha (~700 acres) and ~7,600 ha (~18,800 acres), respectively. Highly concentrated discharge to percolation basins over many acres (larger, yellow to red markers on the left) indicates an increased likelihood of contributing to nitrate contamination. Greater total N applied to few acres of irrigated agriculture (smaller, yellow to red markers on the right) indicates an increased likelihood of contributing to nitrate contamination.

Adminsitrative Record
Page 34478

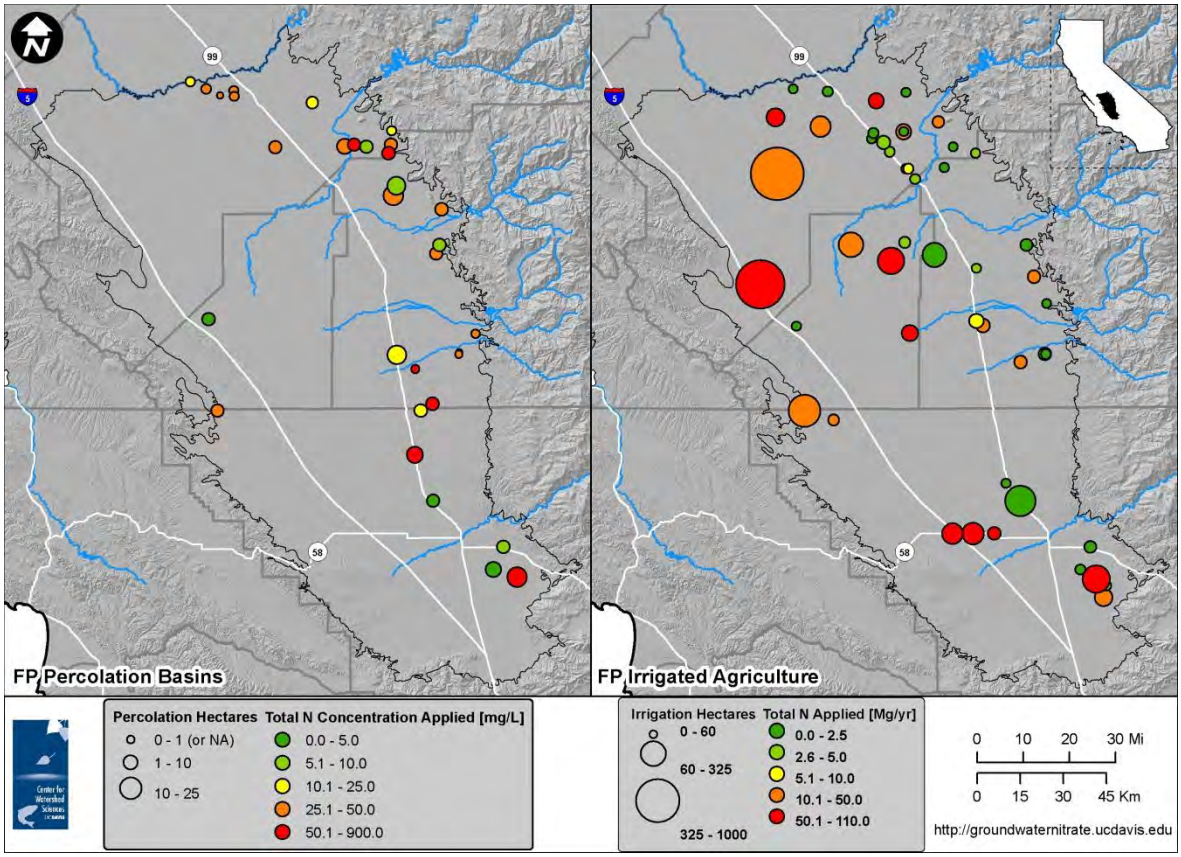


Figure 53. Tulare Lake Basin food processors: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) and total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [1 Mg = 1 metric ton = 0.001 Gg = 1.1 tons, 1 hectare = 2.47 acres.]

Adminsitrative Record
Page 34479

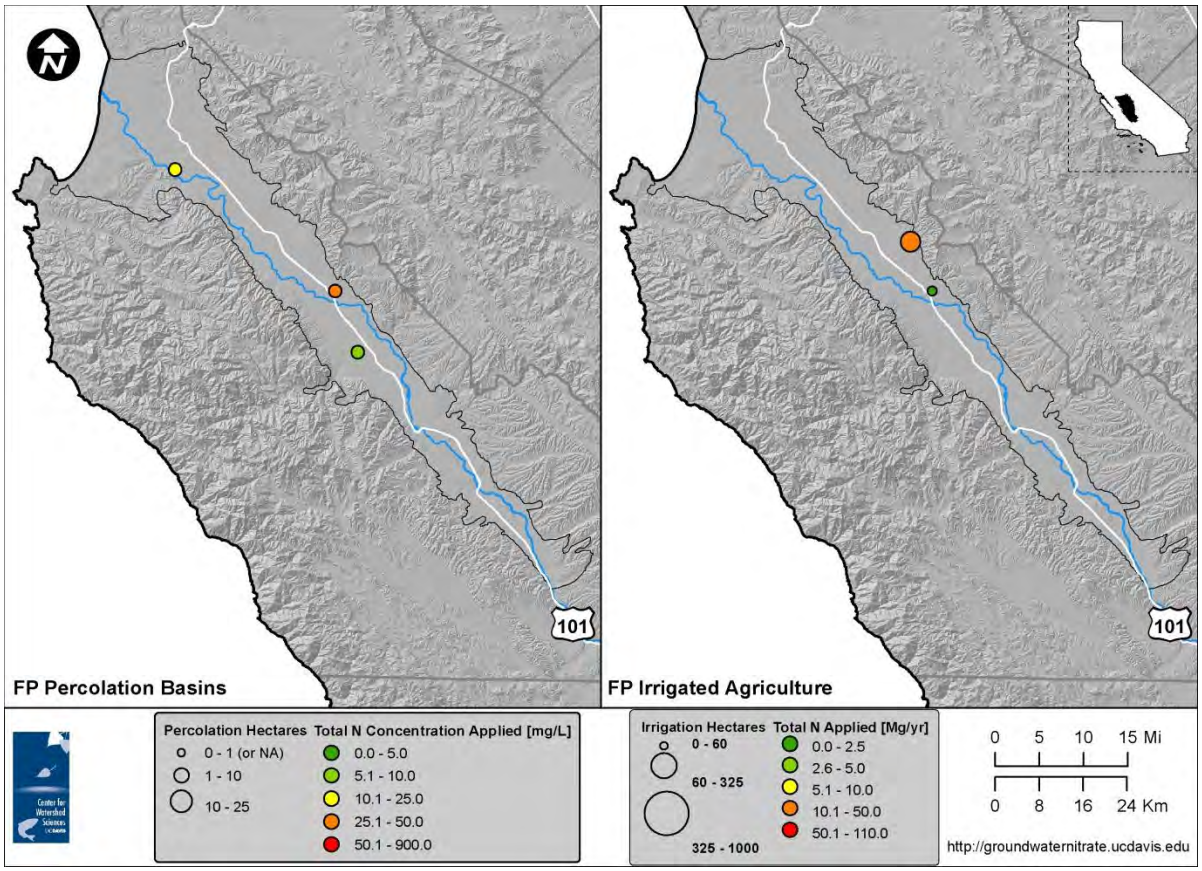


Figure 54. Salinas Valley food processors: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) and total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [1 Mg = 1 metric ton = 0.001 Gg = 1.1 tons, 1 hectare = 2.47 acres.]

Adminsitrative Record
Page 34480

Figure 55 and Figure 56 illustrate the average kg/ha/yr of applied N from FPs for comparison with fertilizer application rates and total N loading from other sources, in the TLB and SV, respectively. This enables an assessment of the over-application of nitrogen; for high demand crops, a rough estimate of required nitrogen is 250 kg/ha/yr (~225 lbs/acre/yr), or 500 kg/ha/yr (~450 lbs/acre/yr) for double cropping.[39] Facilities exceeding this application rate (marked in orange and red) risk contributing to nitrate contamination of groundwater. This is important both regionally and locally to pinpoint hot-spots and locate facilities that may require additional treatment or altered land application practices.

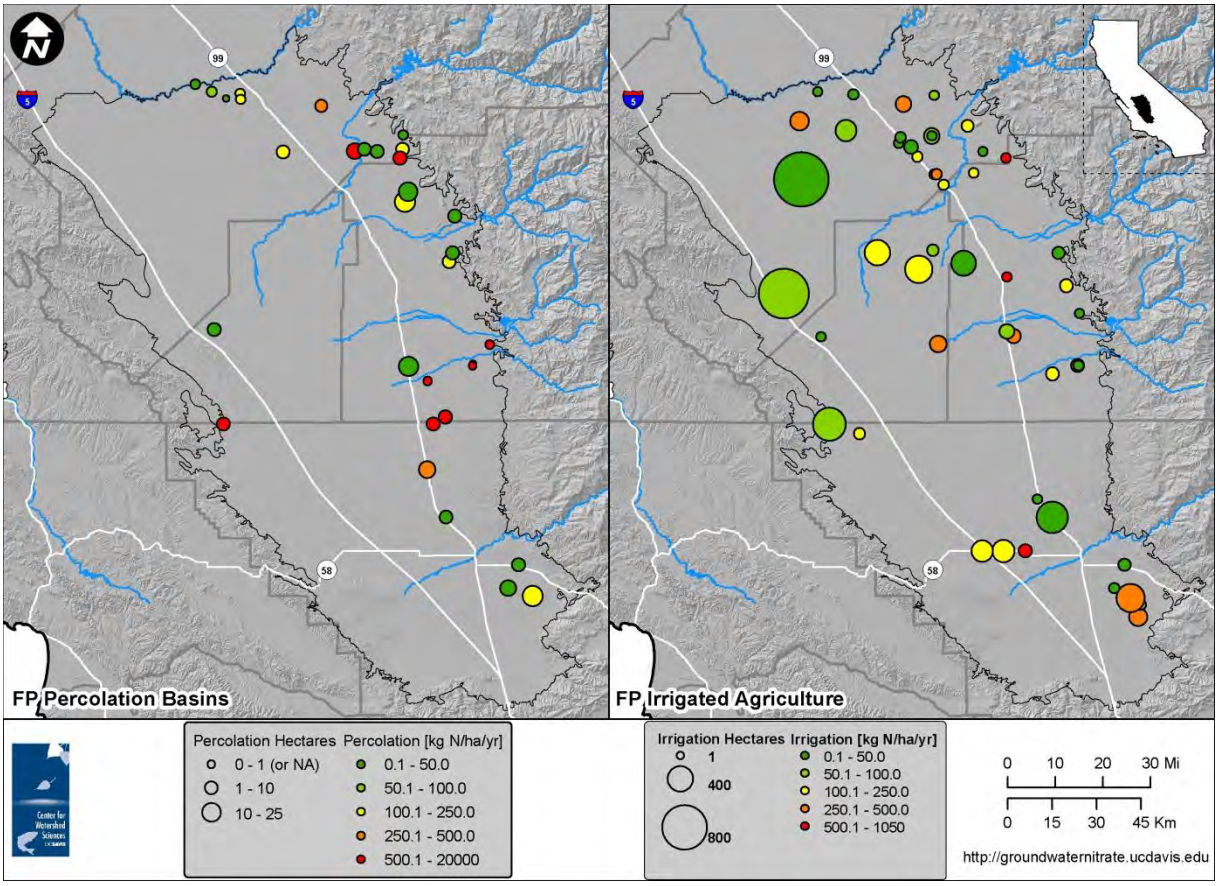


Figure 55. Tulare Lake Basin food processors: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

[39] This is a rough estimate for high demand crops and is based on crop nitrogen demand for single and double cropping as discussed in Section 3.

Adminsitrative Record
Page 34481

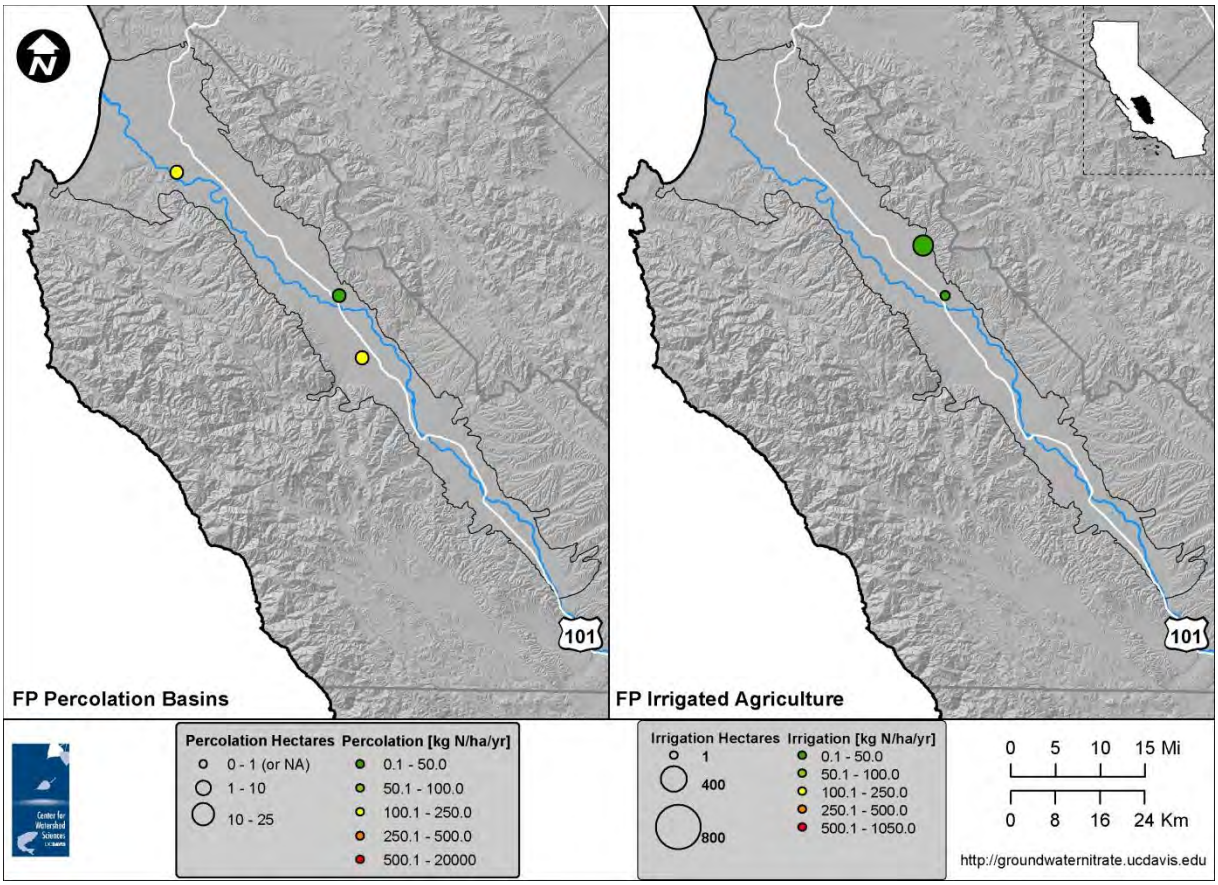


Figure 56. Salinas Valley food processors: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

6.2.3 Conclusions

With the potential to impact local drinking water supplies, N loading from WWTPs and FPs is of a greater concern locally than on a regional scale. This is evident in the GNLM simulation results, which spatially allocated the distribution of nitrogen from WWTP and FPs described in this section. Figures 10 and 11 in Section 1 include the amount of N in land application from biosolids and wastewater effluent. Figure 21 in Section 1 includes the amount of direct nitrate percolation from WWTP and FP percolation ponds. Additional analysis is necessary to assess the risk of these N sources to specific drinking water supplies. Groundwater monitoring is required for many of these facilities; however, the data are largely unavailable since they are not in a digital format. Compilation of the groundwater monitoring data from these facilities into a centrally-managed, digital format would prove highly beneficial to a more accurate assessment of the impact of their discharges to groundwater quality. While the contribution of these sources to regional N loading is less significant than that of agricultural sources, reduction measures can

Adminsitrative Record
Page 34482

be important for the protection of local drinking water supplies (see Technical Report 3, Section 5.2, Dzurella et al. 2012).

6.3 Sewage Systems and their Contribution to Groundwater Nitrogen Loading in the Tulare Lake Basin and Salinas Valley

6.3.1 Introduction to Sewer Systems

In this section we determine the contribution of raw sewage to groundwater nitrogen in the Tulare Lake Basin and the Salinas Valley. The piping that transports raw sewage to wastewater treatment plants must be carefully maintained. Leakage can cause wastewater to infiltrate the surrounding soil and reach the water table below. Poorly fitted pipes, aging collection systems, sanitary sewer overflows (SSOs), and unsuitable piping materials all contribute to the leakage of raw sewage. Sewage exfiltration, or leakage out of sewers, is also difficult to recognize, as it tends to occur underground and is not confined to any specific region. Figure 57 shows reported Sanitary Sewer Overflows (SSOs) over a four-month period in the Tulare Lake Basin and Salinas Valley counties. SSOs include any leakage, spill, overflow, or other discharge of sewage from sanitary sewer systems. Category 1 SSOs are those that spill at least 1000 gallons, or result in a discharge to surface water or a storm drain that does not return to the sanitary sewer system. Category 2 SSOs are all other overflows (State Water Resources Control Board 2011c).

The most common causes of SSOs are blockage or damage from tree roots, blockage by grease deposition, and blockage by debris. By volume, the primary causes are other (encompassing unknown cause, multiple causes, vandalism, operator error, maintenance, improper installation, valve failure, failure from diversion during construction, siphon failure, inappropriate discharge, and non-sanitary sewer system related), flow capacity exceedance, and pipe structural failure (State Water Resources Control Board 2011b).

Adminsitrative Record
Page 34483

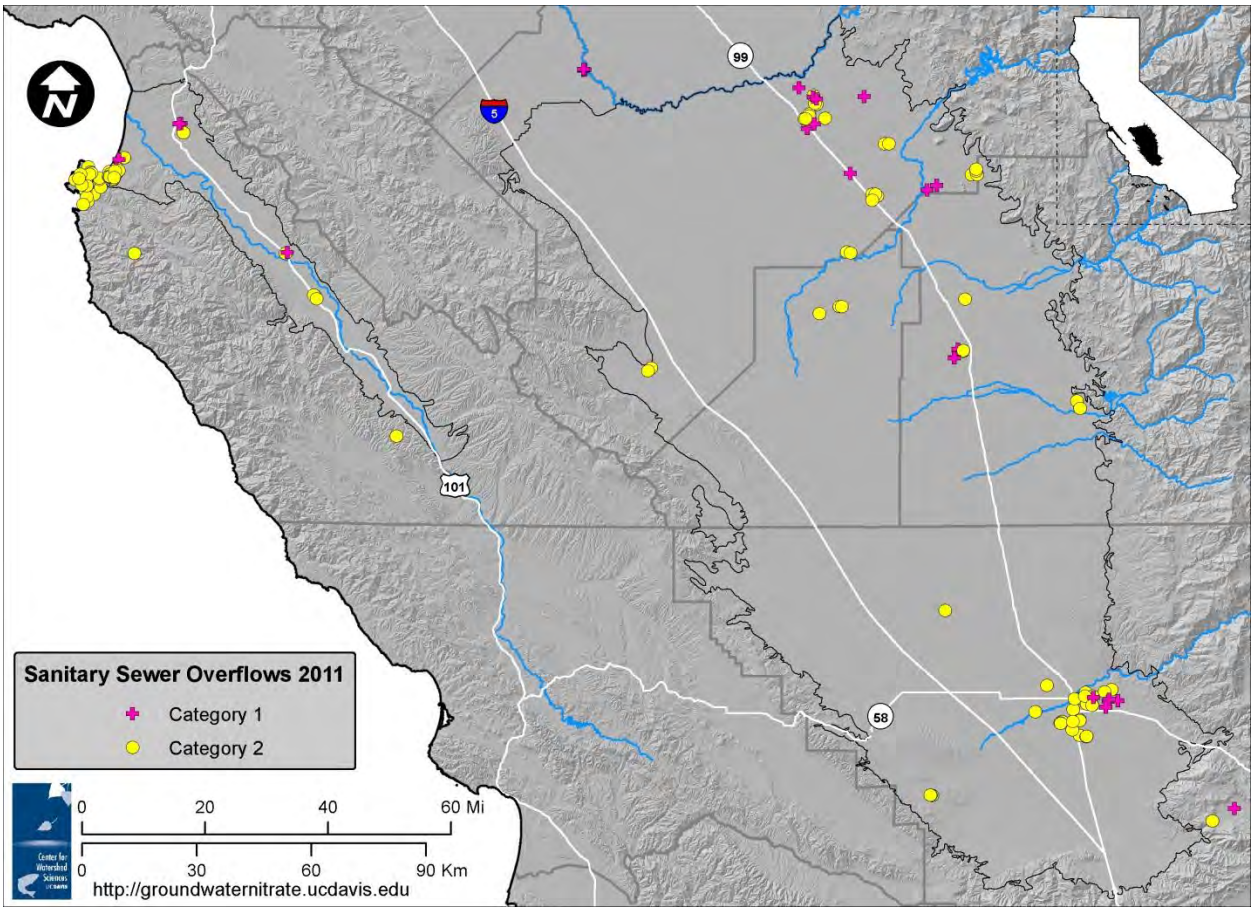


Figure 57. Sanitary sewer overflows in the Tulare Lake Basin and Salinas Valley reported in 2011. (Source: State Water Resources Control Board 2011c.)

6.3.2 Collection System Pipe Materials

This section is a brief review of collection system technology and relevant literature regarding nitrogen contribution to groundwater from sewer system leakage. Sewer pipes come in many forms, with a variety of currently and historically approved piping materials in use. As knowledge and technology of pipe materials have improved over the last century, sewer system pipes have greatly improved in efficiency and longevity.

Metals, especially cast iron and ductile iron, have historically been used as a piping material for sewers. These materials can withstand high pressure, and are effective in pressurized pumping systems. Cast iron pipes have been used as a piping material for the past several hundred years in Europe, and the past 150 years in North America. Recently they have largely been replaced with ductile iron pipes in new construction projects. Ductile iron pipes were introduced in 1955, and have been used more extensively since the mid-1960s. Ductile iron is stronger and more fracture-resistant than cast iron. Both cast iron and ductile iron pipes are susceptible to corrosion from sewer gases, such as hydrogen

Adminsitrative Record
Page 34484

sulfide gas. Consequently, both types of iron pipes are commonly coated with a cement mortar lining to prevent corrosion (Cutter 2009a).

Vitrified clay pipes (VCP) have been used for over 100 years, and are suitable for gravity pipe systems. VCPs are the most inert of all sewer pipes; they are corrosion-resistant to domestic sewage, hydrogen sulfide gas, and most industrial solvents (Cutter 2009b). They are strong, dense, perform well under many environmental conditions, and have a typical lifespan of at least 100 years (Ohlinger 2002). However, vitrified clay tends to fracture under pressure due to its rigidity. VCPs must therefore be carefully installed to ensure proper support and to avoid damage.

Plastic, in the form of polyvinyl chloride (PVC), high density polyethylene (HDPE), truss pipes, and glass-reinforced pipes (GRP), is another common pipe material. Plastic pipes fall into two main categories: thermoplastics and thermosets. PVC, HDPE, and truss pipes are all thermoplastics. They can be heated, formed, and reshaped repeatedly without changes in the material's physical properties. GRP is a thermoset pipe, composed of plastic that cannot be reshaped once it forms. All plastic pipes can be manufactured for use in either gravity sewer systems or pressure sewer systems, and all perform well underground. Plastic pipes are not as rigid as other materials, such as VCP. Proper installation is important to prevent the pipe from improperly bending (Ohlinger 2002). However, the flexibility of many plastics may prevent pipes from fracturing as readily as clay pipes. PVC is cheap, durable, and easy to assemble; it is the most widely used material in sewer systems in the United States and Canada (Rahman 2004). PVC pipes are also resistant to corrosion from many substances, including acids. PVC has been used for less than 40 years and has an uncertain longevity, but scientists estimate its lifespan as greater than 50 years (Ohlinger 2002).

Concrete pipes have been used for sewer and storm water systems since the 1800s. Concrete pipes are suitable for gravity pipes, and their rigidity makes them easier to install than many other types of pipes. Concrete pipes have been known to leak at the joints, but technology has greatly improved this issue (Cutter 2009b). However, concrete is still known to be problematic in some environments because it has the potential to corrode. Thus, many concrete pipes are lined with more inert materials, such as PVC, for protection. The US Army Corps of Engineers recommends a design life of 70 – 100 years for concrete pipes (American Concrete Pipe Association 2011).

Orangeburg pipe materials were used extensively in the past but are now believed to be inadequate for sewage transport. Orangeburg pipes, a brand name for a pipe composed of wood pulp and pitch, were used as early as the late 1800s, but especially during the 1950s – 1970s. Many cities turned to Orangeburg as a cheap alternative to other materials, such as cast iron. However, these brittle pipes deform under pressure, absorb moisture, and are prone to invasion from tree roots and deterioration from solvents (City of Ann Arbor 2006). Orangeburg pipes have a typical lifespan of 50 years, but are still present in many cities where they have surpassed 50 years in age. This material is no longer an acceptable piping material under most building codes, and has been largely replaced by PVC in new construction projects.

Adminsitrative Record
Page 34485

6.3.3 Methods of Determining Nitrogen Loading from Sewage Systems

To estimate the regional nitrogen input to groundwater from sewer system leakage, the sewer nitrogen outputs for the four largest cities within the study area–Fresno, Bakersfield, Salinas, and Visalia–were estimated individually. These cities account for approximately 60% of the total study area population on sewer systems, based on the 2010 population as estimated from the 2000 Census (US Census Bureau 2011), and Census 1990 estimates of the numbers of households on sewer systems at the block group summary level.

The contribution of leaky sewers to groundwater nitrogen was calculated using two techniques, one which relies on the known city population and one which relies on the known flow at the wastewater treatment facility. The estimates are most likely credible if they are in reasonable agreement (although similar estimates do not necessarily guarantee accuracy). Several common assumptions are made, including established values and generalizations about flow. In both estimates, the sewer leakage rate is assumed to be in the range of 1 – 25% of the total sewer flow (these values are determined by reviewing best available literature, which is later described in detail). Both estimates ignore sewage infiltration, and both calculations estimate total sewer exfiltration, rather than net exfiltration. It is also assumed that all nitrogen released from sewers will reach the groundwater as nitrate, and that the sewage composition is uniform.

Another important assumption is that nitrogen load from industrial waste is negligible. Industrial sources are not reliant on local populations, and the nitrogen content of industrial wastewater is industry-dependent. The industrial activities that do give off significant amounts of nitrate are largely related to agriculture, with few other industrial activities providing a negligible amount of nitrate (Metcalf & Eddy 2003). The overall nitrogen content from industrial sources is therefore difficult to assess. However, the wastewater treatment plants within the four study cities serve mainly domestic users, so this has little bearing on our overall calculations.

The first estimate for nitrogen output from domestic sewage is based on the population using a given sewer system. The average rate of human nitrogen excretion was assumed to be 4.85×10^{-3} Mg N/capita/year. Because individuals with septic systems do not utilize the sewage system, it was necessary to isolate the portion of each city population utilizing the sewage system. The population of the entire study area was partitioned by city and by household sanitation type. These values were derived from 2009 Census Estimates and the known population utilizing the two types of systems. This estimate may contain small population errors as collection system limits differ from city to city.

A second estimate for the total groundwater nitrogen from faulty sewage piping was calculated from the known average daily flows for each of the treatment plants and from the estimated average daily flow for the entire study area. The average daily flows for each of the four study cities was obtained from the local wastewater treatment plants, while the total flow in the study area was scaled up from the estimate for the actual total flow of 210 MGD for facilities accounting for 90% of WWTP flow in the region (see Section 6.2). The known average sewage flows for each treatment plant and for the entire region were used to estimate the sewage flow prior to losses. The difference between these two values

Adminsitrative Record
Page 34486

provided an estimate for the flow loss, and thus the nitrogen released to the subsurface. This calculation required estimating the total nitrogen concentration of untreated domestic sewage, which is available in the literature (Metcalf & Eddy 2003 p. 186). Average modern wastewater flows in the study area tend to fall in the range of 75 – 100 gallons/(capita*day), which is most consistent with a medium strength flow. The total nitrogen concentration of a medium strength flow is listed as 40 mg/L in this table, or 1.51×10^{-7} Mg N/yr. A total nitrogen concentration of 40 mg/L is consistent with data from the wastewater treatment plants (see also Section 6.2). Influent sewage reaching Bakersfield Waste Water Treatment Plant #3 has a total nitrogen concentration in the range of 38 – 42 mg/L. Fresno's untreated sewage is monitored for ammonia concentration, which averages 27 mg/L.

Little information is readily available regarding the leakage rates and other complications for particular sewage systems in the study area, as much of this information is not reported to the public. Interviews were conducted with city and wastewater treatment plant employees and with the Central Valley and Central Coast Regional Water Boards to obtain further information about these systems.

6.3.4 Results of Nitrogen Loading from Sewage Systems

Collection System Information: Bakersfield, Fresno, Salinas, Visalia

Fresno is the largest city within the study area, and contains the most extensive collection systems among the surveyed cities. The Fresno collection system includes 1,502 miles of piping, varying in size from under 8 inches to over 36 inches in diameter. Seventy-three percent of those pipes are 8 inches or less in diameter, with only 10% over 19 inches in diameter. This is a generally young system, with 73.5% of piping less than 50 years old. Only 6.9% of the piping surpasses 75 years in age. There are a variety of piping materials used in Fresno, but the vast majority are VCPs and PVC pipes (56.5% and 30%, respectively). The remaining 12.5% of Fresno collection piping consists of cast iron, concrete, and other types of plastic pipes. Fresno operates one large and one small treatment plant, with a combined average flow of approximately 66 million gallons per day (MGD), or ~3 $m^3 s^{-1}$.

Bakersfield has a total of 1,060 miles of collection system piping. Pipe sizes vary greatly throughout Bakersfield from 6 inch diameter pipes in small neighborhoods to large collection pipes, up to 60 inches in diameter. Bakersfield collection pipes were historically composed of VCP, and many older neighborhoods still contain these. Roughly 30 years ago, PVC became the preferred piping material, and most new pipes are constructed with PVC. The major collection pipes are 42, 48, or 60 inches in diameter, and are composed of concrete lined with PVC. The City of Bakersfield has a total of three treatment plants serving the community: Waste Water Treatment Plant #2, Waste Water Treatment Plant #3, and Kern County Sanitation Authority Waste Water Treatment Plant. These plants treat domestic and industrial wastewater. Flows at these facilities average 14 MGD, 17.5 MGD, and 1.25 MGD, respectively. The sum of these values represents the estimated total daily sewage output of Bakersfield; 32.75 MGD.

Salinas has a less extensive piping system, with 286 miles of collection system piping in total. Pipes vary in size from 6 to 60 inches. VCP is the most common piping material for smaller pipes, while the major

Adminsitrative Record
Page 34487

pipelines are mostly composed of HDPE. Until roughly 30 years ago, these pipelines were typically composed of concrete, until it was later detected that many sections of the concrete piping had badly deteriorated through a reaction with internal hydrogen sulfide gas. PVC pipes may be used to replace old pipes which are smaller in size, but PVC is not used for long portions of piping. The wastewater treatment plant for Salinas treats domestic wastewater only, and serves a larger area, including Salinas, Pacific Grove, Seaside, Monterey, Fort Ord Community, Marina, Castroville, and Moss Landing. The total flow from the Salinas wastewater treatment plant averaged 11.44 MGD this past year.

Visalia's collection system consists of 500 miles of piping. These pipes range in diameter from 6 inches in neighborhoods to 45 inches for major collection pipes. A wide range of materials are used, including VCP, concrete, and various PVC pipes. The City of Visalia has one main wastewater treatment plant, which averages a flow of 13 MGD.

Typical Sewage System Leakage Rates

There are three main methods by which scientists attempt to quantify sewer system leakage: field measurements, water balance calculations, and modeling exercises. These methods yield widely varying results for sewer leakage rates, as presented in numerous case studies. A City of Albuquerque study, using a water balance calculation, estimated Albuquerque sewage loss at 11% (Camp, Dresser & McKee Inc. 1998). This result is consistent with leakage rates calculated for several German wastewater systems (Amick & Burgess 2000), where leakage rates have been determined to range from 5 – 20% for sewers above the water table (Ellis et al. 2004). Other studies have broadened the range of potential leakage rates. One Nottingham study found a loss of sewage of only 1 – 2% through annual exfiltration (Ellis et al. 2004), while another study estimated worldwide sewer leakage rates as ranging from 8%, in high quality collection systems, to 20 – 25%, in poor quality systems (Amick & Burgess 2000). Several other estimates for exfiltration rates under normal conditions fall in the range of 1 – 25% (Amick & Burgess 2000; Ellis et al. 2004; Rutsch, Rieckermann, & Krebs 2005). This wide range in calculated leakage rates provides no indication as to the most appropriate value for use in our study area. Thus, providing both high and low end estimates of 1% and 25% is the most appropriate method of estimation due to regional leakage uncertainties.

Leakage rates for Salinas Valley and the Tulare Lake Basin are unknown, although information provided through the abovementioned interviews indicates that they may fall at the low end of the 1 – 25% leakage spectrum. Surface water infiltration to the sewage system tends to be a greater concern than sewage exfiltration in Salinas Valley. The lower elevations of Salinas Valley have a relatively high water table, and the interior pipe pressure can be lower than the pressure outside of the pipes. This promotes the movement of water into the sewer pipes from the surrounding soil.

The rate of human nitrogen excretion has been estimated to be 13.3 g N/capita/day, or 4.85×10^{-3} Mg N/capita/year (Crites & Tchobanoglous 1998a). Other estimates for the rate of human nitrogen excretion are within an acceptable error range of 13.3 g N/capita/day, such as one German water balance model, which used the value 13.7 g N/capita/day (Wolf et al. 2007). Another study estimated the human nitrogen input to sewage as ranging from 2 – 15 g N/capita/day (Henze et al. 2008). For our

Adminsitrative Record
Page 34488

calculations, we used the daily nitrogen output per person of 13.3 g N/capita/day (Crites & Tchobanoglous 1998a).

6.3.5 Methodological Calculations for Nitrogen from Sewage Systems

Estimate #1: Calculating Nitrogen Loss by City Population

Table 49 denotes the estimated populations utilizing sewage systems, yearly sewage nitrogen production (N_{tot}), and the yearly nitrogen losses for the exfiltration rates of 1% and 25% ($N_{L,low}$ and $N_{L,high}$). The total yearly nitrogen production (N_{tot}) was calculated as the product of the population on the sewer system and the yearly nitrogen production rate of 4.85×10^{-3} Mg N/capita/year. The yearly nitrogen loss was the product of the yearly nitrogen production and the sewage exfiltration values of 1% and 25%, or 0.01 and 0.25. $N_{L,low}$ and $N_{L,high}$ are the low and high end estimates for the expected range of sewer nitrogen exfiltration values.

Table 49. Sewer nitrogen loss by city population. [1 Mg = 1 metric ton = 1.1 tons.]

City	Pop. on sewer system	Total Sewer N (Mg/yr)	N lost (Mg/yr) 1% leakage	N lost (Mg/yr) 25% leakage
Bakersfield	268,691	1,304.4	13.0	326.1
Fresno	525,922	2,553.1	25.5	638.3
Salinas	136,929	664.7	6.7	166.2
Visalia	92,800	450.5	4.5	112.6
All study area	1,720,000	8,349.7	83.5	2,087.4

Next, these values were summed accordingly for comparison with the known fertilizer nitrogen values for Tulare Lake Basin and Salinas Valley. Salinas is the only city listed which is located within Salinas Valley, thus the estimate is 6.65 – 166 Mg N/yr. The values for Bakersfield, Fresno, and Visalia were summed to determine a value for Tulare Lake Basin of 43.1 – 1080 Mg N/yr. Because the population utilizing the sewer system was available for the entire study area, the amount of nitrogen released from exfiltrated sewage was also estimated, and found to be on the range of 83.5 – 2090 Mg N/yr.

Adminsitrative Record
Page 34489

Estimate #2: Calculating Nitrogen Loss by Sewage Flow

Table 50 illustrates the series of calculations used to determine the low and high end estimates for the amount of nitrogen lost (Mg N/yr) in each of the four cities of interest, and for the total study area. F_T is the sewage flow at the wastewater treatment plant (that is, the sewage flow after losses). Each F_T value was obtained directly from the wastewater treatment plant operators in the four study cities, and was estimated for the entire study area with the knowledge that the total actual flow for facilities accounting for 90% of WWTP flow in the region was 210 MGD (see Section 6.2). F_0 is the flow prior to any losses, and was calculated for the low and high end estimates as $F_0=F_T/0.99$ and $F_0=F_T/0.75$, respectively. F_L is the flow lost during sewage transport: $F_L=F_0-F_T$. The amount of nitrogen lost is the product of F_L and the total concentration of nitrogen in the wastewater, assumed to be 40 mg/L.

Table 50. Sewer nitrogen loss by wastewater treatment plant flow in MGD (106 gallons/day). [1 Mg = 1 metric ton = 1.1 tons.]

City	F_T (MGD)	F_0 (MGD)	F_L (MGD)	N lost (Mg/yr), 1% leakage
Bakersfield	32.75	33.08	0.33	18.3
Fresno	66.00	66.67	0.67	36.8
Salinas	11.44	11.56	0.12	6.4
Visalia	13.00	13.13	0.13	7.3
All Study Area	233.33	235.69	2.36	130.3

City	F_T (MGD)	F_0 (MGD)	F_L (MGD)	N lost (Mg/yr), 25% leakage
Bakersfield	32.75	43.67	10.92	603.3
Fresno	66.00	88.00	22.00	1215.9
Salinas	11.44	15.25	3.81	210.8
Visalia	13.00	17.33	4.33	239.5
All Study Area	233.33	311.11	77.78	4298.6

As before, these individual values were then used to estimate the sewage nitrogen input to groundwater for Salinas Valley and Tulare Lake Basin. The Salinas Valley estimate becomes 6.39 – 211 Mg N/yr. Bakersfield, Fresno, and Visalia values are summed to obtain 62.4 – 2060 Mg N/yr. The sewage nitrogen input for the entire study area is estimated to range from 130 – 4300 Mg N/yr.

Total Nitrogen Load Rate and Sewer Leakage Recharge for the Study Cities

Total nitrogen load (kg/ha/yr) for the four study cities was next computed based on the above estimates for the sewage nitrogen inputs to groundwater. The exact sewer system boundaries for each study city are uncertain, as sewer system operators for the cities were unable to provide this information. The area in hectares for each of the sewer systems are estimated as follows: Fresno at 35,969 ha, Bakersfield at 18,311 ha, Salinas at 11,634 ha, and Visalia at 10,338 ha (US Census Bureau 2011). The total nitrogen

Adminsitrative Record
Page 34490

load was then determined as the quotient of the yearly nitrogen load (Mg/yr) and the city area (ha). Results are displayed in Table 51.

Table 51. Calculated nitrogen loads (kg/ha/yr). [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

City	Area (ha)	N load (kg/ha/yr), Est #1	N load (kg/ha/yr), Est #2
Bakersfield	18,311	0.71–17.81	1.00–32.95
Fresno	35,969	0.71–17.75	1.02–33.80
Salinas	11,634	0.57–14.28	0.55–18.12
Visalia	10,338	0.44–10.89	0.70–23.17

Sewer leakage recharge was then estimated in mm/yr using the sewage flow losses (F_L) calculated using the second estimation method, and are displayed in Table 52. These values are presented as a range, as the values calculated for 1% and 25% leakage represent the low and high end estimates. The first method of estimation did not determine a total yearly leakage volume, and therefore cannot be directly used to determine sewer leakage recharge. The recharge rate was calculated as the sewage flow rate divided by the sewer system area, with appropriate unit conversions to mm/yr. The sewage recharge is likely to contain up to 40 mg/L of nitrate-nitrogen.

Table 52. Calculated sewer recharge rates (mm/yr). [1 hectare = 2.47 acres, 1 mm = 0.04 in.]

City	Area (hectares)	F_L range (MGD)	Recharge rate (mm/yr)
Bakersfield	18,311	0.331 – 10.9	2.50 – 82.37
Fresno	35,969	0.667 – 22.0	2.56 – 84.51
Salinas	11,634	0.116 – 3.81	1.37 – 45.29
Visalia	10,338	0.131 – 4.33	1.76 – 57.92

6.3.6 Discussion and Conclusion

The four cities surveyed account for only 60% of the population utilizing sewer systems. We scaled the total output to 100% of the population on sewer systems by linear scaling with population size, and obtained the final values for the potential range of leakage from urban sewer systems (shown in Table 53).

Table 53. Summary of all estimates. [1 Mg = 1 metric ton = 0.001 Gg = 1.1 tons.]

Region	Sewer N (Mg/yr), Est #1	Sewer N (Mg/yr), Est #2
Tulare Lake Basin	43.1 – 1080	62.4 – 2060
Salinas Valley	6.65 – 166	6.39 – 211
Entire Study Area	83.5 – 2090	130 – 4300

Adminsitrative Record
Page 34491

The two estimates compare reasonably well and provide a similar range of potential groundwater loading. The range across all estimates, though, is wide: 10^0–10^3 kg N/day. The largest source of uncertainty is the lack of an accurate value for sewer leakage rate in Tulare Lake Basin and Salinas Valley. Other sources of uncertainty contributing to this wide range are the population serviced by sewer systems, unavailability of sewer system boundaries, and lack of information on industrial contributions.

For the synthesis in Section 1, we assumed that 5% of the urban area population's human N excretion leaks to groundwater. We used the 2010 study area census population (2.78 million), subtracted the 2010 population on septic systems (0.56 million, see next section) and assumed the standard N excretion rate of 13.3 g/capita/d. In total, this yields 0.53 Gg N/yr (530 Mg N/yr, 580 tons N/yr) of groundwater nitrate contribution, consistent with the results above, which provide an overall range for this estimate. For the spatially distributed simulation of N loading to groundwater in the GNLM simulations, we assumed that 10 kg N/ha/yr are leaked to groundwater, uniformly throughout areas identified as urban in the CAML maps (Section 3).

Although these results, on a regional scale, and even at the mapped out local scale (Figure 21), indicate that groundwater nitrogen from sewers is negligible in comparison to groundwater nitrogen from fertilizers, sewer leakage can locally be a significant source of nitrate. A point-source of sewer leakage near a domestic or public well has the potential to detrimentally affect public health through contamination, regardless of the negligible regional contribution to groundwater nitrogen. Proper sewer maintenance is thus very important.

6.4 Septic Systems

6.4.1 Objectives in Septic Systems Analysis

This analysis has two primary objectives: 1) to estimate the spatially-distributed contamination of domestic (private) wells with septic-derived nitrate; and 2) to estimate the nitrate loading to groundwater from septic systems in the study areas. These two goals are similar, but require methodological resolution at different scales.

6.4.2 Nitrogen in Septic Systems

Septic systems are designed to prevent pathogens from reaching the soil surface where they may become a risk for human exposure. The reduction in pathogens is accomplished primarily by exposing the effluent of the system to soil microbes. Although a review of relevant literature by Siegrist et al. (2000) found 10-20% nitrogen removal in conventional septic systems, nitrogen removal is not a primary objective of these systems. Nitrogen removal in septic systems is an incidental combination of retention of solids in the septic tank, volatilization of NH_3, and denitrification either to N_2 (complete) or N_2O (incomplete).

Adminsitrative Record
Page 34492

Septic systems typically consist of a buried tank for settling and anaerobic decomposition with overflow to a leach field buried in a soil layer. Baffles in the tank prevent passage of solids to the leach field under proper operating conditions. The leach field consists of buried perforated pipe or open-bottomed chambers; in either case, contact with soil occurs at approximately 1 meter below the soil surface. A layer of slime is created at the soil contact surface.

Anaerobic conditions in the tank promote ammonification but not nitrification, therefore nitrogen in septic tank effluent to the leach field is dominated by ammonium (70-90%), with the remainder in organic form (Lance 1972; Nilsson 1990; Gold and Sims 2000, reviewed in Siegrist et al. 2000; Bunnell et al. 1999, reviewed in Eliasson 2002; Nizeyimana et al. 1996, Hantzsche and Finnemore 1992). Tank effluent ammonium ranges from 20 – 200 mg/L as nitrogen. Nitrification occurs in the aerobic layer of the soil below the leach field, typically the first 15 to 30 cm (Kaplan 1991; Siegrist et al. 2000; Bunnell et al. 1999; Robertson and Cherry 1992; Whelan 1988; Harman et al. 1996; Wilhelm et al. 1994), with less nitrification in silt/clay (soils with decreased hydraulic conductivity) compared to sandier soils (Cochet et al. 1990). Decreased hydraulic conductivity results in saturation and anaerobic conditions, thus reducing the oxygenation of nitrogen from NH_x forms to NO_x forms.

Ammonium can be transported without transformation to groundwater under several conditions: leach field soil is anaerobic due to saturation by irrigation or rainfall; soil particle adsorption capacity is reduced due to high loading rates of ammonium; and/or a high water table reduces the distance from slime layer to water table (Whelan and Barrow 1984). In addition, nitrate leaving the aerobic zone of the leach field can be denitrified below the nitrification zone. Cuyk et al. (2001) found 7-15% removal in the leachfield, at 60 - 90cm of infiltration; however, Brown (1984) found rates as low as 0.45% removal in the leachfield. Nonetheless, as a conservative estimate, we assume that all nitrogen leaving a properly functioning septic tank via a septic leachfield will eventually reach groundwater as nitrate (Whelan 1988).

6.4.3 Septic System Densities

The U.S. EPA in a 1977 report to Congress (USEPA 1977) referred to septic systems densities of 40 septic systems per square mile or more as “relatively high”. Although conversations with EPA scientists indicate that this was never intended to be used as a regulatory threshold or even a recommendation, several authors have used it as such (Yates 1985; Cantor and Knox 1986; New Alchemy Institute 1987; Borchard et al 2003; Horn 2010). To remain consistent with other authors who have investigated septic system densities, we will use that same value (40 systems per square mile, or 1 system per 16 acres, or 0.154 systems per hectare) as an arbitrary reference density above which septic system density is considered high.

Although studies have shown that septic systems can be sources of contamination in wells at much lower densities (Horn 2010; Yates 1985), these studies were concerned with pathogenic and other low-concentration toxins rather than nitrate specifically,therefore, these studies only needed to show that any interception of septic leachate by the well capture zone was likely. For nitrate, a higher threshold

Adminsitrative Record
Page 34493

would be required, and the non-uniformity of hydraulic conductivity, groundwater flow direction, and parcel sizes become more important than for a simple presence/absence assessment.

Short-circuiting of the groundwater system along well casings or naturally occuring pathways of high hydraulic conductivity between the surface and the well screen can result in contamination of any well placed close to a septic leachfield. This report does not address this problem of direct contamination due to poor placement of wells and septic systems. A much more detailed, case-by-case study would be required to estimate the rate of occurrence of this problem.

In the study area, each county has its own minimum required parcel size for septic systems. Fresno, Kings, Tulare, and Monterey Counties all require a minimum of 1 acre parcel size for septic system permits, and Kern requires 2.5 acres minimum. This information was obtained from county Environmental Health and Planning Agencies.

To qualitatively evaluate the incidence of nitrate contamination of drinking water wells with septic-derived nitrate, without expensive and time-consuming surveys and sampling, we examined septic system densities at several threshold levels.

6.4.4 Methods: Total Septic System Nitrogen-Loading to Groundwater

The 1990 Census asked two questions relevant to our investigation: "what is your water source?", and "how is your sewage disposed of?" These questions were tabulated at the census blockgroup level. The blockgroups in our study areas range in size from about 1 acre to over 600,000 acres. Blockgroups are delineated by the census such that housing density is relatively constant within in blockgroup. The relevant census data include the numbers of households in each blockgroup, the number of households on septic systems, and the number of households on private wells. These data are only available for the 1990 Census; subsequent Census questionnaires did not ask these questions, but did include the number of households in each blockgroup.

To calculate 2010 septic system density, we assumed that the fraction of households on septic systems was unchanged between 1990 and 2010. Using geographic information system (GIS) software (ESRI ArcGIS), the following procedure was used to create a one square kilometer raster grid indicating, for each raster cell, the septic system density (number of systems per square kilometer):

Adminsitrative Record
Page 34494

1. The 1990 spatial distribution of blockgroups was used to create a one-hectare raster grid of the 1990 variable "fraction of households on septic systems." The raster value (fraction of 1990 households on septic systems) is a spatial integration across blockgroups overlying an individual raster cell. The raster value accounts for the fractional overlying area of each blockgroup.
2. The 2010 spatial distribution of blockgroups was used to create a one-hectare raster grid of the 2010 variable "number of households." The raster value (number of households per hectare) is a spatial integration across blockgroups overlying an individual raster cell. The raster value accounts for the overlying area of each blockgroup and its household density. The two one-hectare raster-grids created in this step and in the previous step are spatially collocated (grid cells are in the exact same location);
3. Multiplication of the two raster values (fraction of households on septic systems, number of households per hectare) to produce a new raster value on the same one-hectare raster grid, "2010 number of households on septic systems per hectare;"
4. Equivalent to step 2, a one-hectare raster-grid of the value of the 2010 census variable "persons per household" was created. Multiplied with the raster-grid value "2010 number of households on septic systems per hectare," this yields a raster-grid value "2010 number of persons on septic systems per hectare."
5. Tchobanoglous et al. (2003) estimate that the daily nitrogen excretion per adult is 13.3 grams. Approximately 15% of that nitrogen is assumed to either stay in the septic tank or volatilizes from the tank or from the septic leachfield (see above, Siegrist et al. 2000). Thus, long-term average groundwater nitrate loading via septic systems is conservatively estimated to be 11.3 grams of nitrate-nitrogen per person per day (4.125 kg N per person per year). We multiply this value by the "2010 number of persons on septic systems per hectare" to obtain spatially distributed raster grid values "nitrate-N loading in kg per hectare."
6. Raster-grid values are summarized to obtain county and study area total nitrate-N loading.

6.4.5 Methods: Local Contamination

In order to quantitatively estimate the risk of contamination of drinking water wells by direct input from septic systems, we would need to know the likelihood that the capture zone of any given well includes a septic leachfield and the quantity of nitrate from this leachfield that would become entrained in the well. Therefore, the spatial distribution of leachfields and wells is a critical parameter for this analysis. Such information is both confidential and generally unrecorded. Although it may be possible to obtain data that describes the density of household wells in some counties, septic systems are unrecorded except as individual permits. A process to obtain these records would involve collecting and collating tens of thousands of permit documents from each county and extracting the relevant data from that collection. That effort is beyond the scope of the current project. Therefore, we conducted a qualitative assessment of direct contamination of wells by septic effluent instead.

We characterized the extent of direct contamination by comparison with several density thresholds: 40 septic fields per square mile (0.154 systems per hectare, or equivalent to 16-acre parcels), 80 septics per square mile (equalivalent to 8 acre parcels), the most stringent local agency threshold (that of Kern County) of 256 septics per square mile (0.988 systems per hectare, or 2.5-acre parcels), and twice that, 512 septics per square mile. The best model of septic system density that was available for this

Adminsitrative Record
Page 34495

assessment was the census blockgroup data. We used the raster-grid variable "2010 number of households on septic systems per hectare" (described above) to represent the density of septic systems.

Each polygon is assumed to have uniform septic system density, although density is most often distributed non-uniformly throughout each polygon. Therefore, this method will inevitably underestimate the maximum density of systems in any given census polygon, while over-estimating the fraction of that polygon that contains septics at the stated density.

6.4.6 Results: Total Septic System Nitrogen-Loading to Groundwater

Based on the 1990 and 2010 census data, we found that the total number of people on septic systems in the study areas was 509,000 for the Tulare Lake Basin and 48,300 for Salinas Valley (Table 54).

Table 54. Number of persons on septic in each county, for the regions of those counties that are within our study area boundaries.

Region	Persons on Septic Systems	Study Area
Fresno	182,516	Tulare Lake Basin Total: 509,015
Tulare	125,988	
Kings	18,867	
Kern	181,644	
Monterey	48,296	Salinas Valley

The annual nitrate load from septic systems to groundwater, based on the 1990 and 2010 census data, in the Tulare Lake Basin is 509,015 people x 11.3 g/day/person x 365 days/yr = 2.099 Gg N /yr. For Salinas Valley, that load is 0.199 Gg N /yr.

6.4.7 Results: Septic System Density and Regional Septic Nitrogen Leaching to Groundwater

Although the highest rate of septic use is in the most rural areas – areas furthest away from urban areas (Figure 58), the lower population densities in these areas result in low total densities of septics. We found that the highest densities of septic systems occurred in peri-urban (rural sub-urban) areas near cities, but outside the service areas of the wastewater systems that served those cities (Figure 58).

Adminsitrative Record
Page 34496

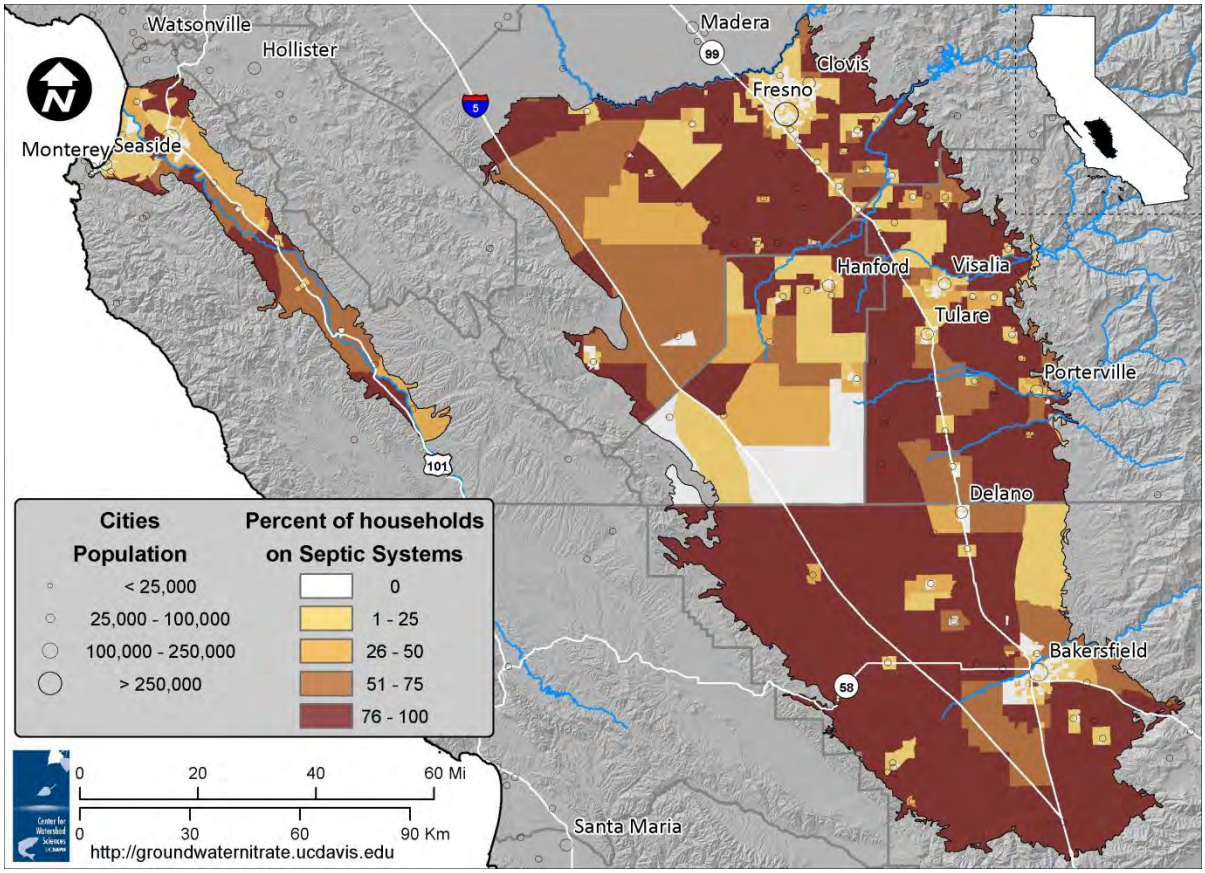


Figure 58. Percent of households on septic systems, by blockgroup, from 1990 census data, in the study area.

The system densities range from zero systems per hectare in city centers and in the southeast portion of Kings County to 5 or more systems per hectare in the peri-urban areas (Figure 59). Although the algorithm used to develop these values resulted in some small areas with more than 10 systems per hectare, these anomalous values are likely to be artifacts of the combination of the two census blockgroup polygon datasets.

Adminsitrative Record
Page 34497

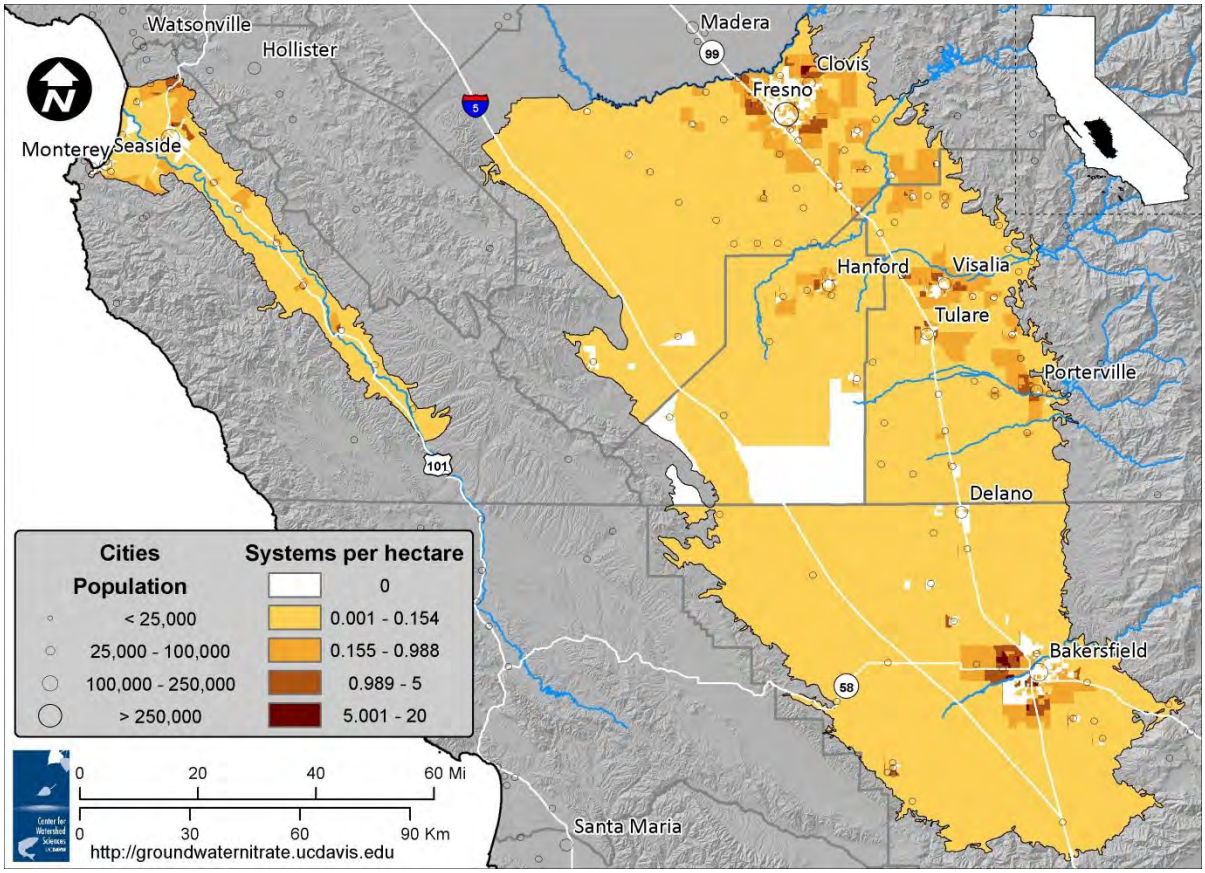


Figure 59. Septic systems per hectare, in the study area. Two thresholds are shown: 0.154 systems per ha (40 systems per square mile) and 0.99 systems per ha (1 system per acre), the maximum density allowed in some counties.

In the Tulare Lake Basin, 7.9% of the land area is over the arbitrary threshold of 40 septic systems per square mile. In the Salinas Valley, about 12.6% of the land area is over the arbitrary threshold of 40 septic systems per square mile (Table 55).

Adminsitrative Record
Page 34498

Table 55. Land area with septic system densities below the threshold of 40 system per square mile (0.154 systems per ha), up to twice the threshold (0.308 systems per hectare), one septic system per 2.5 acres (0.988 septic systems per hectare, 256 systems per square mile), 2 systems per 2.5 acres, and above.

	Salinas		Tulare		Study Area	
System Density	**Acres**	**% of region**	**Acres**	**% of region**	**Acres**	**% of region**
Under 40/sq.mi.	346,365	87.4%	4,846,931	92.2%	5,193,296	91.8%
40-80/sq.mi.	26,153	6.6%	213,205	4.1%	239,358	4.2%
80-256/sq.mi.	16,657	4.2%	133,056	2.5%	149,713	2.6%
256-512/sq.mi.	4,206	1.1%	25,970	0.5%	30,176	0.5%
Over 512/sq.mi.	2,780	0.7%	39,961	0.8%	42,741	0.8%

Due to variable local housing demographics, some regions tend to have more people living in a household than others. As a result, the actual loading of nitrogen to the groundwater may vary significantly at comparable density of septic systems (Figure 60). Based on the 2010 census data, household size in the study areas ranged from a blockgroup average of around 1 person per household to as many as 8 persons per household. In areas with high numbers of persons per household, the loading from septic systems is higher than predicted by our model, which uses the average number of person per household. Figure 60 shows the spatial distribution of nitrogen loading from septic systems in the study areas. Our estimate is that 0.47% of the Tulare Lake Basin study area, and approximately 0.68% of the Salinas Valley study area are subjected to nitrogen loads from septic systems that exceed the operational benchmark for our study of 35 kg/ha/yr.

Adminsitrative Record
Page 34499

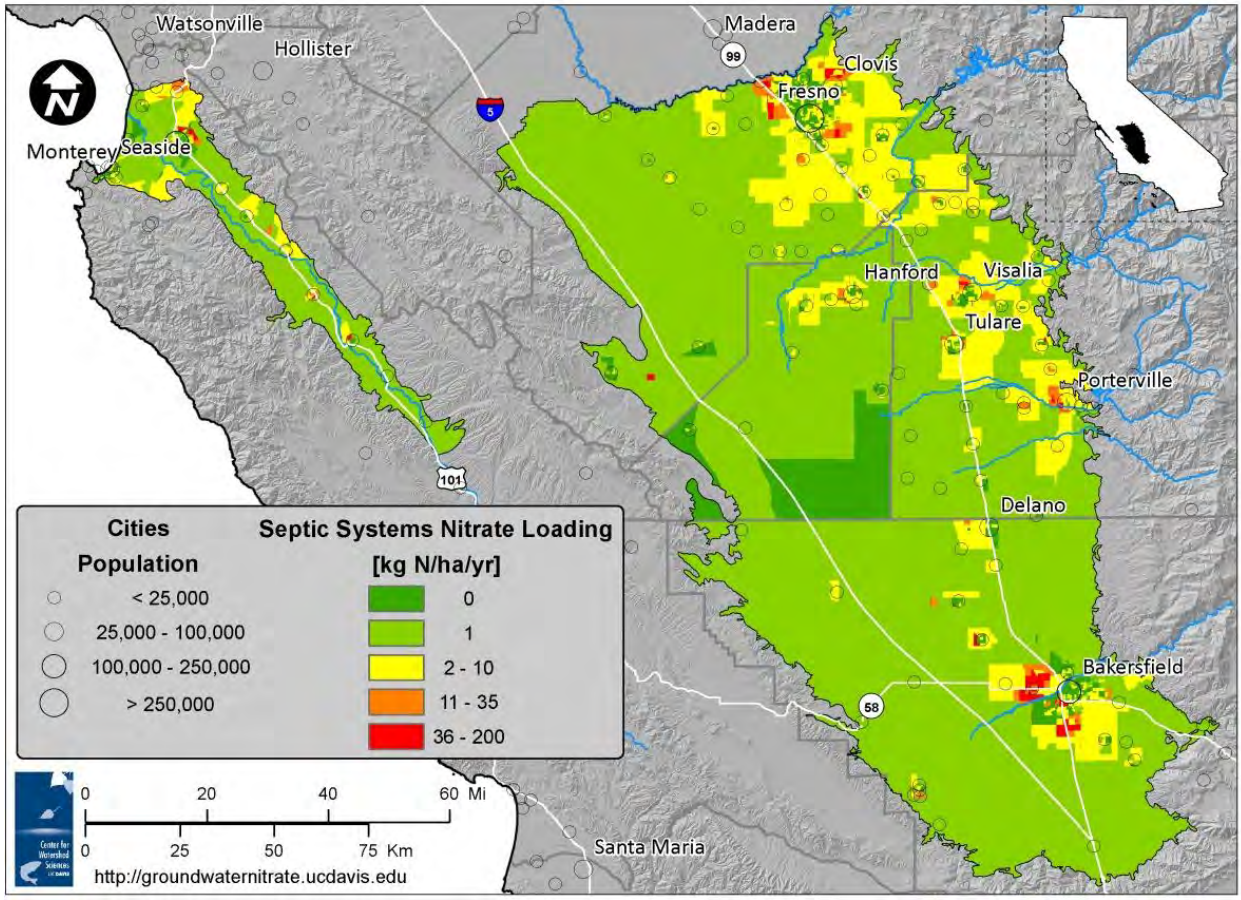


Figure 60. Septic-derived nitrogen leaching rates within the study area.

6.4.8 Discussion of Septic Systems Analysis

The method used to delineate septic system densities is based on the 1990 fraction of households on septic, and the 2010 number of households, for any given area. This likely overestimates the number of septic systems in some peri-urban areas. If the service area of a city sewer system has expanded to include an area that was unsewered in 1990, the now more densely populated area will appear to have very high septic system density. Further examination of the extent of sewer system service areas is required before we can properly evaluate this effect. A possible next step would be to analyze billing data from city sewer systems to delineate sewer system service areas.

Our analysis is providing an upper (highest possible) estimate of septic sewer leakage due to the assumption that all daily human waste is collected by the domestic septic system. In reality, residents spend some of their time outside the home, in urban areas, and some of the human waste will be collected in municipal sewer systems. The fraction of waste thus collected is not known.

6.4.9 Conclusions from Septic Systems Analysis

Total septic system contribution to nitrate loading to groundwater in the study area (2.298 Gg/yr) is about 2.4% of agricultural sources in the study area (see Section 1), and about 20% percent of the loading from Wastewater Treatment Plants and Food Processors (see Section 6.2 of this report). Taken as a contribution to the total nitrogen load to groundwater, septic systems are a minor problem. However, locally, septic systems are likely to contribute significantly to well water nitrate, particularly in areas of high septic systems density, surrounding cities, with highest densities observed around the cities of Fresno and Bakersfield, where nitrate loading to groundwater may be in the range of 10 - 50 kg N/ha (9 – 45 lb N/ac). The benchmark loading rate of 35 kg/ha/yr is exceeded by septic system nitrogen leaching in 0.49% of the study area (0.47% in the TLB, 0.68% in the SV). These areas include areas around the City of Salinas, and the smaller communities in the Salinas Valley, and the rural and peri-urban areas of Fresno, Hanford, Visalia, Tulare, Porterville, and Bakersfield in the eastern section of the Tulare Lake Basin.

Adminsitrative Record
Page 34501

7 Atmospheric Nitrogen Deposition

7.1 Introduction to Atmospheric Deposition

The two largest sources of N emissions to the atmosphere are NO_x from fossil fuel combustion, and ammonia volatilized from manure at concentrated animal feeding operations. This N undergoes various transformations in the atmosphere before being redeposited at distances ranging from meters to thousands of kilometers from the source of emissions. Atmospheric deposition occurs when airborne particles, gases and dissolved compounds are deposited to the land surface, either in precipitation (wet deposition) or as a result of one of several complex atmospheric processes (e.g., settling, impaction, and adsorption), which constitute dry deposition. Deposition can also be partitioned by the form of N deposited into oxidized N (largely derived from NO_x emissions) and reduced N (largely derived from ammonia emissions).

Many parts of California, especially in sparsely populated coastal areas, receive relatively low (<3 kg N/ha/yr) levels of atmospheric N deposition. However, the San Joaquin Valley receives among the highest levels in the state, typically greater than 10 kg N/ha/yr. Some of this N has been transported downwind from the San Francisco Bay Area while some is emitted in the study area, especially from urban areas and dairies (see Section 4). However, the geographic setting also plays a role: the dominant winds from the west tend to trap the pollutants in the valley against the foothills of the Sierra Nevada. Thus eastern cities, such as Fresno, tend to have much higher atmospheric concentrations and deposition.

7.2 Atmospheric Deposition Methods

Atmospheric deposition is rarely measured continuously and wet deposition is monitored much more frequently than dry deposition. The most comprehensive network of sample sites is run by the National Acid Deposition Program. There are approximately 10 sites in California included in this wet deposition monitoring program, but individual researchers have expanded the spatial distribution of measurements. Because N deposition varies spatially, especially dry deposition, N deposition estimates at broader spatial scales are typically based on modeled data. The most widely used model, the Community Multiscale Air Quality (CMAQ) model, was developed by the U.S. EPA. This model combines N emission inventories with meteorological data to estimate N deposition. The highest resolution version of CMAQ for California is a 4 km grid (Tonnensen et al. 2007). This estimate was updated by Fenn et al. (2011) to take into account the fact that measured rates of deposition exceeded the rates predicted by the model (Figure 61).

7.3 Atmospheric Deposition Results and Discussion

Over the entire study area, N deposition amounted to 20 Gg N/yr (see Section 1.8 for historic, current, and future results). N deposition in urban and natural areas was assumed to be retained within the

Adminsitrative Record
Page 34502

ecosystem. For urban areas, we assumed that N deposition is retained in urban soils due to the ability of turfgrass to sequester N, and the fate of N deposition to impervious surface would likely be to surface waters. Because most natural ecosystems are N limited (Vitousek & Howarth 1991), small amounts of N deposition typically act as fertilizer. However, at higher rates or chronic low rates of N deposition, changes in ecosystem function such as N leaching can start to occur (Aber et al. 1998). Many California ecosystems are receiving N deposition in excess of the critical load to maintain ecosystem function (Fenn et al. 2011) and in some cases nitrate loading to surface water has been documented. However, there is limited evidence that current rates of atmospheric deposition are resulting in loading to groundwater. Therefore, we have assumed that nitrate leaching of atmospherically deposited N is negligible in the natural lands contained in the study area.

In cropland, however, N deposition (from adjusted CMAQ estimates) was included as a loading input to the N mass balance (Sections 1 and 2) on a spatially disaggregated basis, as it can be mobilized to groundwater via irrigation. The average deposition rate is 9.8 kg N/ha (9 lb N/ac) in the Tulare Lake Basin and 5.6 kg N/ha (5 lb N/ac) in the Salinas Valley. Using the total on-the-ground ACR based cropland area (not including alfalfa) of 1,157,000 ha in the Tulare Lake Basin and 113,088 ha in the Salinas Valley (Section 3), we obtain an estimate of the total atmospheric N deposition. A value of 12.0 Gg N/yr is added to the overall N mass balance of croplands (except alfalfa) in the study area. Within the Tulare Lake Basin, N deposition on cropland (not including alfalfa) is 4.1, 3.0, 1.6, and 2.7 Gg N/yr for Fresno, Kern, Kings, and Tulare County, respectively.

Adminsitrative Record
Page 34503

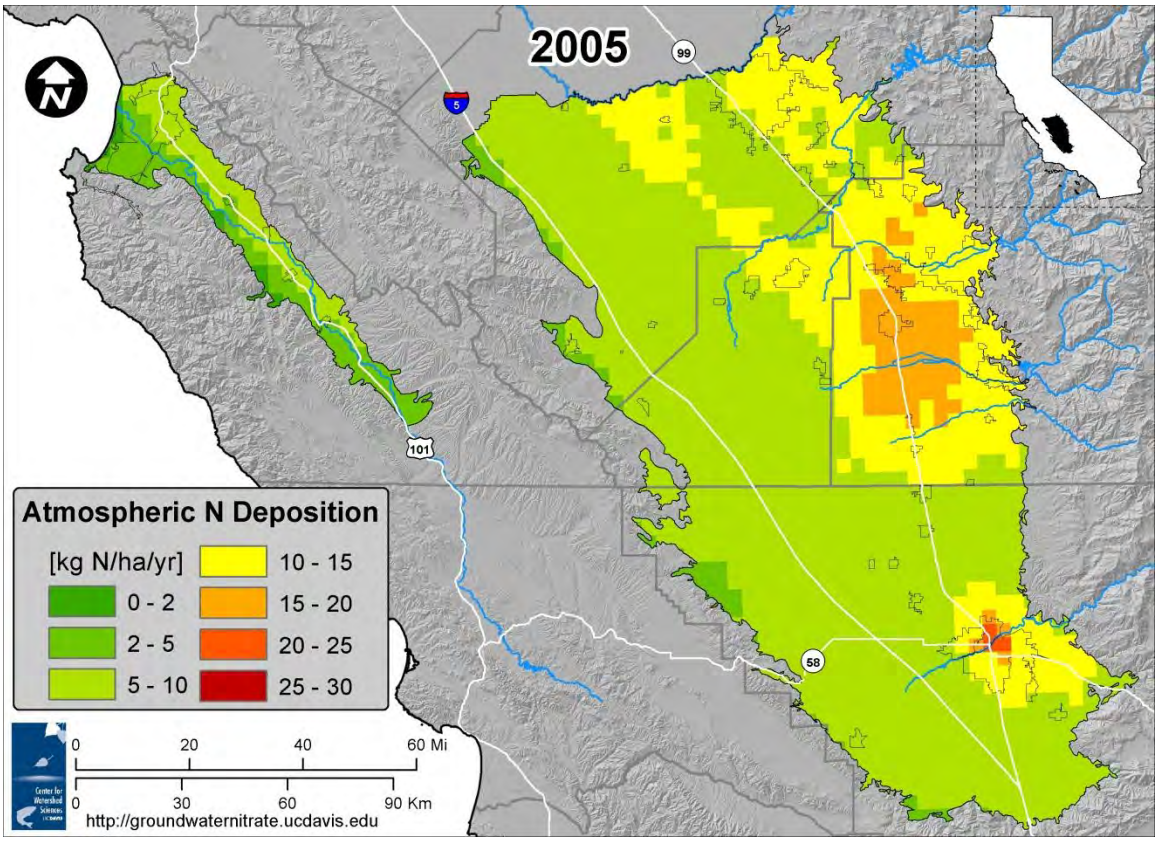


Figure 61. Atmospheric N deposition, spatially distributed across the study area, based on the Community Multiscale Air Quality model developed by U.S. EPA.

Adminsitrative Record
Page 34504

8 Natural Sources of Nitrogen

Groundwater nitrate concentrations in the absence of human activity are typically well below 3 mg NO_3^--N/L (Spalding & Exner 1993). However, researchers have documented the occurrence of relatively higher groundwater and surface water nitrate concentrations for diverse regions and under a variety of climatic conditions though the concentrations are well below the maximum contaminant limit. Geologic nitrogen (i.e. nitrogen in bedrock) has been suggested as a source for high nitrate concentrations for many areas in the United States, such as Cedar Valley in southwestern Utah (Lowe and Wallace 2001), central and southwestern Nebraska (Boyce et al. 1976), and in certain rock formations in Wisconsin, Utah, Colorado, and Wyoming (Sullivan et al. 1979). Geologic nitrogen has also been documented in several locations within California, including the Mokelumne River watershed (Holloway et al. 1998), parts of the San Joaquin Valley (Strathouse et al. 1980) and in the Klamath mountains (Morford et al. 2011). In general, it is marine rocks with high organic matter content that release N as they weather over time.

Several studies have suggested that geologic nitrogen may contribute to elevated nitrate concentrations in localized areas of the western San Joaquin Valley. However, the majority of the research has documented high nitrate concentrations in soils and the vadose zone, not groundwater, related to bedrock N. Extremely high soil NO_3^--N concentrations ranging from 1400-2000 mg/L have been documented in areas on the west side of the San Joaquin Valley, and were thought to be from indigenous sources of nitrate (Dyer 1965; Strathouse et al. 1980). This potentially represents over 1000 kg N/ha in the subsurface of areas with geologic N. Several other studies have documented that these high nitrate concentration are likely due to the high N content in several marine sedimentary rocks along the eastern flank of the Diablo Range to the west of the Valley and in the alluvial fans where sediments have been transported to the valley floor (Strathouse et al. 1980; Sullivan et al. 1979). Because there are relatively few unmanaged areas with native vegetation and soils in the San Joaquin Valley, it is difficult of find locations to sample for background nitrate concentrations related to geologic N. Nor is it possible to use isotopic analysis to distinguish N weathered from bedrock from fertilizer N (Fogg et al. 1998).

There are two other potential sources of “natural” N that can pollute groundwater because of human activity. First, semi-arid regions often contain large accumulations of nitrate salts in the subsurface. Small amounts of N from atmospheric N deposition or otherwise in excess of biological demand can leach below the root zone creating high concentrations of subsurface N regardless of any sources of bedrock N. However, because of the small amount of rainfall, there is not enough water transport to move the nitrate to groundwater (Stadler et al. 2008). This phenomenon has been documented in the Mojave Desert where nitrate accumulations can range up to 10,000 kg N/ha (Walvoord et al. 2003). While the climatic regime is similar in parts of the Tulare Lake Basin, there is limited evidence that similar nitrate accumulations occurred there. This nitrate would cause minimal groundwater contamination under the ambient climate regime (i.e., limited rainfall). Further, if such accumulations occurred in TLB wetlands, such as when actual evapotranspiration was greater than precipitation, and prior to the building flood control structures and widespread dyking (late 1800s) (Moyle 1995), it is likely

Adminsitrative Record
Page 34505

that leaching from episodic flooding would have negated any excess accumulation. However, if the land was converted to agricultural purposes and irrigated, the nitrate salts, along with other accumulated salts, combined with nitrate from decomposing organic matter, could have leached to groundwater due to aqueous mobilization and movement beyond the vadose zone (Dyer et al. 1965; Fogg et al. 1998).

A second source of nitrate released by human activity is also related to the conversion of natural lands to cropland. The physical disturbance of the soil can stimulate microbial activity resulting in the transformation of stable organic forms of N to more mobile forms like nitrate. This phenomenon has been well documented in the Great Plains (e.g., Scanlon et al. 2008), but there are relatively little data for California, in part because of the lack of uncultivated sites in the major agricultural regions (i.e., lack of experimental control). One metric to estimate the potential for N release associated with land conversion is the amount of soil organic matter. Based on data compiled by Post and Mann (1990), agricultural conversion results in N accumulation in surface soils when the C and N content are below 1% and 0.1% respectively. While many soils in former wetlands would be expected to exceed these concentrations, large areas of the Central Valley would have been below these concentrations because of low net primary productivity associated with the dry climate. Therefore, for this study we assumed that organic matter turnover is a comparatively unimportant source of nitrate loading to groundwater at the regional scale. A similar phenomenon has been described for urban areas with construction activities stimulating microbial activity in soils, and resulting in a localized pulse of N losses (Wakida and Lerner, 2006).

In summary, there are too few groundwater samples prior to human disturbance to definitively attribute any elevated nitrate concentrations to natural sources of N. While there are well documented examples of high N accumulation in the vadose zone, there is limited evidence that this nitrate would have resulted in significantly elevated groundwater N concentrations, and if it did, it would have happened in the very early decades of cultivation and irrigation in a pulse type event. At present, the high natural sources of N from bedrock represent a small fraction of the study area, as they are restricted to alluvial fans in the extreme western TLB. Therefore, for this study we have assumed that natural nitrate is a comparatively unimportant source of groundwater N.

Adminsitrative Record
Page 34506

9 Wells as Sources of Nitrogen

9.1 Introduction to Wells as Sources of Nitrogen

This section provides a review of wells as potential conduit for rapid nitrate migration into groundwater or as a source of nitrate transfer from shallower groundwater to deeper groundwater. The main purpose of this section is to give a rationale for and describe the methods used to determine the potential magnitude of nitrate contamination occurring via this pathway. A detailed technical review of proper well construction methods, backflow prevention techniques, and proper well destruction, however, is beyond the scope of this section.

The introduction provides a brief conceptual overview of the general mechanisms by which wells may become a conduit or source for nitrate contamination in shallow or deeper groundwater. Section 2.9.2 provides some technical background on the potential for active wells to incidentally leak contaminated water to groundwater and we estimate the total number of active wells in the study area. Section 2.9.3 is a brief review of dry wells. It discusses the potential extent these may contribute to groundwater nitrate pollution. In Section 2.9.4, we estimate the approximate number of abandoned and inactive wells. Finally, in Section 2.9.5, we provide a simplified and approximate estimate of the potential nitrate leakage due to surface discharges into wells and due to intra-well transfer from shallower aquifer sediments to deeper aquifer sediments. The estimates are based on the estimated number of active, dry, inactive, abandoned, or improperly destroyed wells provided in Sections 2.9.2 through 2.9.4.

Water supply wells are constructed by drilling an open borehole, slightly larger in diameter than the well itself, to the desired depth (Figure 62). The well – a long steel or plastic pipe with screened (slotted) sections – is inserted into the open borehole and positioned using centering devices. The annulus, which is the open space between the well pipe and borehole wall and which extends from the land surface to the bottom of the well, is back-filled with sand or gravel filter materials (“gravel pack”). Near the land surface, the annulus must be sealed with low permeability materials made from bentonite, cement, cement-bentonite mixtures, or similar materials (“surface seal”). The annulus ranges in width from 25 mm (1 inch) to 100 mm (4 inches). The State of California requires a surface seal within the annulus that is at least 15 m (50 ft), measured from the land surface down, for public and industrial wells, and 6 m (20 ft) for agricultural and private domestic well types. Some counties, such as Monterey County have more stringent requirements.

Adminsitrative Record
Page 34507

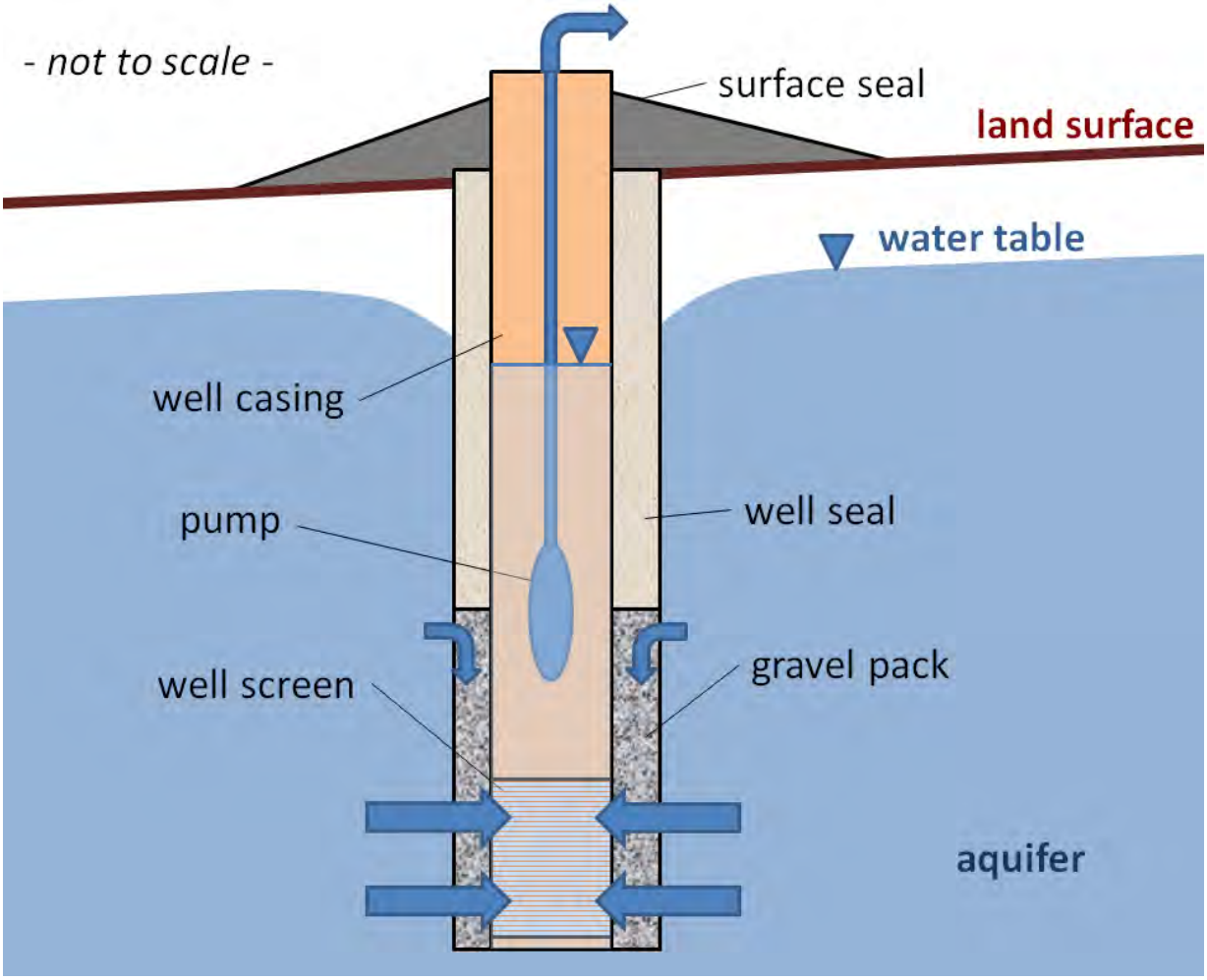


Figure 62. Schematic outline of a well. Well casing and the well screen are surrounded by the borehole annulus, which is packed with gravel materials around the screen and with well seal materials above the gravel pack. Water enters the well from the aquifer through the gravel pack and the well screen.

The purpose of the surface seal is twofold: a) to prevent the leakage of contaminants from the surface to the water table in the gravel pack along the outside of the well-pipe; and b) to prevent the leakage of often highly contaminated shallow groundwater along the gravel pack into deeper production aquifers. At the surface, a concrete pad prevents flow of contaminated water or fluids into the borehole.

Within unconsolidated sedimentary aquifer systems such as the alluvial basins of the Tulare Lake Basin and Salinas Valley, active wells, dry wells, inactive or abandoned wells, and improperly destroyed (decommissioned) wells are potential pathways for rapid transmission of contaminants from the land surface to groundwater or between aquifer units that are otherwise separated by low conducting, fine-grained silty and clayey materials. They therefore constitute a potential localized source of nitrate in groundwater or in deep groundwater. To the degree that the leakage from the land surface to groundwater would not have occurred, these wells are true groundwater nitrate sources from the landscape. They are a source of deep groundwater contamination, not from the land surface but from

Adminsitrative Record
Page 34508

already existing shallow groundwater contamination in the case, where water leaks through elongated well screens, within the well. As will be discussed in this section, the latter is a potentially significant source of internal nitrate contamination transfer within the aquifer.

In the following discussion, we consider common pathways associated with four potential sources: (seasonally) active wells, permanently inactive wells and abandoned wells, improperly destroyed wells, and dry wells.

Active wells are wells in current use. Many active wells are in use only seasonally for irrigation or supplemental municipal supply. An inactive public supply well may be operated under emergency situations for up to 15 days per year. Abandoned wells are wells that are no longer used and the associated pumping equipment has either fallen into disrepair or has been removed. A properly destroyed well is filled with grout or other impermeable material so water cannot flow through the well or the annulus. Dry wells are a special form of drain wells defined in more detail below.

Active Wells: Three potential pathways exist for groundwater contamination via active wells:

1. **Backflow**. Backflow occurs in wells when the head inside the well is less than the head outside the well, for example when the well pump is turned off, and no backflow prevention device is present. Normally, this merely injects water that was previously removed from the aquifer back into the well, however, in certain fertigation or chemigation applications, chemical fertilizer or pesticides can be injected into the aquifer. While regulations exist to prevent backflow of pesticides, there are currently no requirements for backflow prevention devices for wells used for fertigation with nitrogen fertilizer and no estimates of the number of backflow accidents or the amount of nitrate (or other contaminants) reaching groundwater via this pathway.

2. **Improperly installed or failing annular surface seal and/or well head completion.** Improperly constructed or failing annular surface seals allow transport of contaminants from shallow groundwater to deeper groundwater along the outside of the well casing. An absent, improperly constructed, or cracked pump base will allow contaminants from ground surface into the well borehole. The possibility of significant nitrate contamination of groundwater is particularly high where irrigation water containing high concentrations of nitrogenous compounds (nitrate, urea, ammonium) runs directly over the top of a well (e.g., in flood irrigated areas). Well construction guidelines by the State of California are 30 years old (California Department of Water Resources 1981, 1991) and are enforced by each county separately.

3. **Long or multi-aquifer screens or lacking/improperly installed aquitard seals.** Alluvial aquifer basins (such as the TLB and SV) characteristically consist of thick aquifer systems with multiple overlying water-bearing zones separated by interfingered layers of fine-grained silty or clayey sediment material. Especially in aquifer systems with thick aquitards, significant pressure differences may be present between overlying water-bearing zones (e.g., water level maps for confined and unconfined aquifers prepared by the Monterey County Water Resources Agency[40], or DWR[41]). In the TLB and SV, it is not

[40] http://www.mcwra.co.monterey.ca.us/
[41] http://www.water.ca.gov/iwris/

Adminsitrative Record
Page 34509

uncommon to find that most pumping occurs in intermediate and deeper water-bearing zones, while less pumping occurs in the shallower water-bearing zones, thus creating a downward pressure gradient. A well that is screened across multiple water-bearing zones or a well with a continuous gravel pack across multiple water-bearing zones becomes a conduit for groundwater flow between water-bearing zones during periods of low capacity pumping or no pumping. In California, these low or zero production periods occur during the non-irrigated winter season in both urban and agricultural regions. Nitrate-contaminated, shallower, younger groundwater can thus be introduced directly into deeper water-bearing zones that contain otherwise older and less nitrate-contaminated groundwater.

Abandoned Wells. Abandoned wells that are not properly decommissioned pose all of the same risks of an active well that is not pumping, including leakage through the seal, cross-aquifer contamination via head gradients and intra-well leakage between water-bearing zones (Figure 63). Problems tend to be worse in old, poorly constructed wells or where the well pipe or well seal has deteriorated over time. Abandoned wells further pose a risk when the surface protection has been removed or compromised, or when the surface opening of the well is visually concealed or camouflaged and left without effective protection from surface inflows. In irrigated areas with flood or furrow irrigation, abandoned wells may receive return flow.

Figure 63. Example of an abandoned well that poses a risk of direct surface runoff spillage into the well casing and into the aquifer (photo courtesy of David Von Aspern and Derek Jacks, Sacramento County).

Little work currently exists to estimate the number of abandoned wells in California. Legally, wells that are out of service must be destroyed properly (California Department of Water Resources 1981, 1991). In practice, many wells are improperly destroyed, particularly private domestic and irrigation wells. Not every abandoned well is a conduit of large amounts of nitrate (or other groundwater contaminants),

Adminsitrative Record
Page 34510

however, the possibility of direct and open connections to groundwater pose a high contamination risk to aquifers. Public supply wells, due to regulatory oversight, are usually professionally destroyed once no longer in use.

Dry Wells. Dry wells are used in urban and in agricultural areas for drainage of stormwater runoff and – in irrigated areas – of irrigation return water. They are commonly constructed in large diameter boreholes into which concrete culverts or large diameter steel pipes have been lowered and back-filled with gravel-sized materials that are highly permeable and inert. Dry wells are typically completed only to shallow depth and pose a high contamination risk mostly to the uppermost water-bearing zone.

9.2 Active Well Characteristics, Number of Wells, Well Seals, and Backflow Prevention

9.2.1 Introduction to Active Wells

Residents of the San Joaquin Valley rely on groundwater for domestic consumption. Information collected on typical water well characteristics and backflow prevention devices in the region provides a basis for estimating potential nitrate loading to aquifers via wells through the mechanisms described above.

Well numbers, requirements, and construction techniques vary county by county within this region, adding complexity to the issue of regional groundwater monitoring and regulation. In addition to California DWR well standards, many counties have local well requirements. Well seal requirements, typical well depths, typical well construction techniques, and well screen characteristics are subject to varying regulation.

9.2.2 Methods of Active Well Quantification

Information on wells, well construction requirements, well destruction, and dry wells (see below) was collected for all five counties within the study area: Fresno, Kern, Kings, Tulare, and Monterey counties. Information was sought out and collected at the county level by contacting county governments, at the state level through the Department of Water Resources, and by review of literature.

Each county was contacted for an estimate of the number of well construction permits issued over the past ten years. The departments responsible for well construction differed from county to county, and individuals from the following departments were able to provide information on this topic: Tulare County GIS, Kern County Department of Environmental Health, Kings County Community Development Agency, Fresno County Department of Environmental Health, and Monterey County Health Department Environmental Health Division. The Department of Water Resources South Central Region Office was also contacted for estimates of the number of well completion records on file over this time period, as this office is responsible for regulating a larger region which includes the counties of interest.

Adminsitrative Record
Page 34511

Well seal, back-flow prevention, well screen, well setback information was obtained from DWR, DPR, and individual county regulatory agencies. The effectiveness of various types of well seals was characterized through literature review.

9.2.3 Quantifying Well Construction Rates in the San Joaquin Valley

Availability and accuracy of the estimates of well construction over the past ten or more years varied considerably between county governments. Kern, Kings, and Monterey counties were able to provide yearly breakdowns in the number of permits issued for about the past 10 years. Well construction counts for 2011 only include those permits issued on or before June 30, 2011. Kern County maintains computerized records of well construction permits for the past decade, but many records dating earlier than 2006 may be missing (Kern County Department of Environmental Health, personal communication, 6 July 2011). There are a total of 1,581 domestic well records and 674 agricultural well records in the Kern county database, some of which are not included in the table below because they lacked construction dates or were dated earlier than 2001. Additionally, most of the older records have yet to be added to this database. For Kings County, an accurate record of all well permits issued since 2001 exists, including yearly totals, but breakdown between agricultural and domestic wells was not available (Kings County Community Development Agency, personal communication, 14 July 2011). In total, 2,012 well permits have been issued in Kings County since 2001, with older records yet to be added to the database. Monterey County has issued 2,370 well construction permits since 2000, with thousands of older records yet to be added to the county database (Monterey County Health Department, personal communication, 18 July 2011).The available yearly data for Kern, Kings, and Monterey counties are displayed in Table 56.

Fresno and Tulare Counties were unable to provide yearly breakdown of the number of permits issued. Accurate estimates for well construction permits issued in Tulare County over the past 10 or more years are not accessible, as permits are available but have not yet been digitally organized. The California Department of Water Resources holds well logs for over 20,000 Tulare County wells (Tulare County GIS Department, personal communication, 6-7 July 2011). The county estimates that a total of 30,000-40,000 wells have been constructed since 1930, with about 2,500 well completions since 2005 (*ibid.*). Tulare County has provided UC Davis with all available well construction permits, but they have not been counted.

Adminsitrative Record
Page 34512

Table 56. Yearly well construction totals for Kern, Kings, and Monterey Counties.

Year	Kern: Permits, Agriculture	Kern: Permits, Domestic	Kings County Permits	Monterey County Permits
2000	23	33		449
2001	23	31	169	380
2002	27	52	189	290
2003	15	87	214	290
2004	26	125	196	333
2005	49	255	199	291
2006	57	326	157	253
2007	87	270	202	229
2008	160	130	214	169
2009	106	106	246	135
2010	70	58	156	151
2011	22	40	70	61

According to Fresno County Department of Public Health, Fresno County holds an accurate record of well construction permits issued since the database establishment in 1976 (Fresno County Department of Public Health, Environmental Health Division, personal communication, 11 July 2011), with values displayed below in Table 57.

Because of the inconsistencies in available data between these five counties, the Department of Water Resources was also contacted and requested to provide the same type of information. The Department of Water Resources does not hold a record of well construction permits, but a record of well completion reports. The Department of Water Resources South Central Region Office, was able to provide yearly well completion counts by county and type from 1977-2009 for all five counties. This record appears more complete than any of the records from the individual counties and spans a longer time period, although some individuals may have failed to file a well completion report. DWR was also able to provide yearly well completion totals for domestic, agricultural, and municipal wells (DWR, personal communication, 20-21 July 2011). Well completion data are shown in Appendix Table 9. We note that these data are for the entire county areas and are not limited to the Tulare Lake Basin and Salinas Valley study area (groundwater basins). In particular, a significant amount of domestic wells would be constructed outside of DWR groundwater basins in fractured rocks of mountainous regions. Also, a large portion of Kern County and its groundwater basins are outside of the Tulare Lake Basin study area.

Table 57. Total numbers of Fresno County well construction permits since 1976

Domestic and Agriculture Well Permits	Public Well Permits	Well Destruction Permits
24,132	334	2,248

Adminsitrative Record
Page 34513

9.2.4 Typical Characteristics in Well Depths and Well Screens

As a result of local geology and differences in historical reliance on groundwater, typical well depths varied considerably through the San Joaquin Valley. Fresno County wells vary in depth throughout the county, with typical well depths around and exceeding 300 m (1,000 ft) on the west side of the valley and as little as 60 m (180 ft) on the eastside. Most Kern County agricultural and municipal wells extend 300 m (1,000 ft) in depth, while domestic wells are around 90 m (300 ft) in depth. Tulare, Kings, and Monterey counties were unable to provide values for typical well depths, due to local variability in well depth.

Well screen characteristics tend to vary by county as well. Fresno County well screens, in wells on the valley floor, are typically slotted and around 18 – 30 m (60-100) feet long. Hard rock wells, generally only found in the foothills, are typically open-bottomed and do not have a well screen. Most agricultural and municipal wells in Kern County penetrate the Corcoran Clay layer and are fully screened below the clay, but domestic wells do not tend to penetrate this layer, and contain much shorter screens. Throughout the area, most wells use mill slotted screens with slot size depending on aquifer materials and driller's choice.

9.2.5 County-Specific Wells and Well Seal Requirements

Well seal depth requirements for the counties of interest are available in each county's Code of Ordinances. Fresno, Kings, and Tulare Counties follow the standards enforced through the California Department of Water Resources: 15 m (50 ft) for public and industrial wells, and 6 m (20 ft) for other well types, including agricultural wells and domestic wells. Kern County wells exhibit more complex seal depth requirements (Kern County Environmental Health, personal communication, 8-11 July 2011). In locations where the Corcoran Clay layer is thought to be present, a geophysical log called an "E-log" or electrical log is obtained to determine the depth and extent of the clay, and the well is sealed from the bottom of the clay to the ground surface. Wells at dairies and in areas of high perched groundwater require a seal depth of 30 m (100 ft), while the seal depth for other agricultural, domestic, and industrial wells is 15 m (50 ft).

Monterey County also experiences high levels of nitrate contamination in some areas (see Boyle et al., 2012) and special requirements for wells exist not only at the county, but also at the local level. All Monterey County wells are required to be sealed to 15 m [50 ft] depth, although the Health Officer may require special well seal depth requirements in areas where groundwater quality problems are known (Monterey County, California 2011). Some areas have stricter requirements for all wells. The Salinas Valley, for example, currently uses approximately 3,000 agricultural wells (Monterey Department of Health- Division of Environmental Health, personal communication, 18 July 2011), but many domestic wells have been shut down in the Salinas Valley due to high groundwater nitrate content (Bryant 2002). Well construction is not permitted in the Fort Ord area, as the area contains contamination plumes. The Monterey Peninsula Water Management District also has greater regulation of groundwater use than

Adminsitrative Record
Page 34514

the rest of the county and enforces some additional requirements for wells: the installation of water meters and sounding tubes for water level measurement, and additional mandatory permitting for some drilling projects (Monterey County, California 2011).

Some counties also exhibit specific requirements as far as well seal materials. In Fresno County, well seals may consist of neat cement, cement grout, bentonite clay, or concrete (Fresno County, California 2011). Kern County wells may be sealed with neat cement, cement grout, cement, bentonite mixtures (powdered, granulated, pelletized, or chipped/crushed sodium montmorillonite clay), and low-permeability native soils (Kern County, California 2011). Monterey County wells must have a seal of cement, sand cement grout, neat cement/pozzolan/polymer mixture, bentonite clay, or a similar compound (Monterey County, California 2011). Tulare County well seals may consist of neat cement grout, sand cement grout, concrete, bentonite-cement grout, or bentonite clay (Tulare County, California 2011). Kings County did not list specific well seal materials in the county code of ordinances, but noted that the state standards should be followed for well construction unless otherwise noted (Kings County, California 2011).

Most counties within the study area did not have specific requirements for sealing the annulus if the well is screened through multiple aquifers. Kings, Monterey, Fresno, and Tulare County wells that are screened through multiple aquifers are not required to have the annulus between them sealed, except in areas where groundwater quality is already known to be a problem. Kern County wells follow a different guideline. Kern County does not allow wells to be screened through multiple aquifers (Kern County Environmental Health, personal communication, 8 July 2011).

Importantly, the standards described here are current standards. For example, in the past, a minimum 2 inch thickness was required for the annulus to place the gravel pack and seal. This minimum thickness is now 3 inches. Also, enforcement of these standards varies between counties and has improved our time. But with the long life-time of wells, a large number of older wells exist, dating from before the 1980s and 1990s, that completely lack a well seal or have a seal that does not meet modern minimum depth requirements and cannot be considered to provide effective protection against leakage.

9.2.6 Effectiveness of Various Sealing Materials

An important consideration during the well construction process is the effectiveness of the annular surface seal in the context of local environmental conditions. Recent literature suggests that these considerations may be crucial in preventing groundwater contamination, particularly in locations where wells penetrate multiple aquifers. One study tested the effectiveness and properties of sealing characteristics using a large-scale laboratory model, and investigated both infiltration though the seal and the seal's ability to withstand fracturing (Edil *et al.* 1992). Several bentonite drilling muds of varying viscosity and sand content were tested, along with neat cement, bentonite-cement, powder bentonite (Volclay), and granular bentonite (Benseal).

Researchers arrived at several conclusions for this experiment. Most importantly, it was confirmed that sealant success depends on both structural resistance to fracturing and on sealant hydraulic

Adminsitrative Record
Page 34515

conductivity. Researchers also identified which seals tended to be most effective overall (Table 58). Benseal-bentonite slurry grout provides an excellent seal, while neat cement and bentonite-cement grouts provide good seals. Volclay and the bentonite drilling muds form poorer seals and do not adhere to the well casing (Edil et al. 1992).

Table 58. Relative rates of infiltration found in lab tests of annular seal materials. Four different bentonite drilling muds, distinguished by sealant viscosity and sand content, are compared with four other typical annular seal materials (Adapted from Edil et al. 1992).

Seal Material [a: sealant viscosity (sec.qt0); b: sand content (%)]	Lowest Measured Infiltration Rate During First 10 Weeks [cm/s]
Bentonite drill mud (a:50; b: 10%)	4.60E-06
Bentonite drill mud (a:50; b:20%)	3.90E-06
Bentonite drill mud (a:70; b:10%)	3.80E-06
Bentonite drill mud (a:70; b:20%)	2.40E-06
Benseal	2.20E-07
Neat cement	4.50E-07
Bentonite-cement	8.10E-07
Volclay	2.27E-05

Another study sought to determine which type of sealant is most effective in actual wells (Christman, Benson, & Edil 2002). This study used an ultrasonic geophysical probe, which sends out a signal and receives return energy from the water-casing interface and the casing-seal interface that is characteristic of the material in the annulus. The ultrasonic geophysical tool was used at many locations along the length of the well, with a total of 35 wells tested.

The Christman study largely agreed with the results of the laboratory tests in the previous study. Differences in construction methods, site geology, and sealant types were all shown to influence the effectiveness of the seal. The results also indicated which seals and conditions were most unfavorable. The poorest seals were found when mud-rotary methods penetrated deep sand or gravel, as these coarse sediments collapsed into the annulus before the sealant was properly in place. Other seals which showed questionable results included those consisting of cement-bentonite grout or un-hydrated bentonite chips, as these materials remained too dry and promoted infiltration (Christman *et al.* 2002).

A recent study, the Nebraska Grout Study, has important implications for future well construction. This has been an ongoing study for over a decade, beginning in 1999 with the construction of a well with transparent casing. Sixteen months later, the well was revisited and found to contain large cracks in the grout column; the slurry grout shrank and cracked under drying conditions and never rehydrated, leaving cracks as well as space between the grout and the casing. This was an important finding, because the cracks provided a pathway by which surface waters could contaminate groundwater. This finding prompted the construction of other wells with transparent casing, spanning a variety of geological environments in Nebraska. Another important finding was that cement grouts do not bond with plastic well casing, maintaining a direct pathway for waters to mix in the ground. As of November

Adminsitrative Record
Page 34516

2010, this study is still ongoing, but it is believed that these findings will prompt future changes in well construction techniques and the creation of new grout materials (Ross 2010). Table 59 lists the performance rankings of eleven grout materials, with cement-sand the best overall sealant and bentonite slurry with <20% solids as the poorest.

Table 59. Performance rankings for eleven grout materials (adapted from Ross, 2010).

Grout Type	Performance Ranking	Visual Ranking
Cement-sand *	1	3.5
Bentonite chip	2	1
Neat cement – 7 gallons H2) *	3	5.5
Concrete *	4	8
Neat cement – 6 gallons H20 *	5	2
Cement-bentonite *	6	3.5
Bentonite slurry > 20%	7	5.5
Geothermal-sand ~60% **	8	10
Bentonite slurr = 20%	9	7
Geothermal ~20% **	10	11
Bentonite slurry <20%	11	9

* Based on maximum depth of dye in one- and 24-hour videos.
** Water level estimated from water table well.

9.2.7 Regulations for Backflow Prevention Devices

Chemigation is the process of applying chemical fertilizers or other chemicals through irrigation water. Due to concerns regarding possible contamination of the water source through chemigation, the U.S. EPA has put in place a series of regulations to prevent source water contamination through the use of backflow prevention devices. Under regulations from the California Department of Pesticide Regulation, all well users in California must follow a series of state regulations for backflow prevention devices during chemigation processes with pesticides. In California, backflow prevention devices are not required in the context of fertilizer preparation and use. The current design standards for backflow prevention devices can be found in the U.S. EPA's PR notice 87-1, written in 1987. This document lists six main requirements for sprinkler, furrow, and drip irrigation backflow prevention devices:

1. The system must contain a functional check valve, vacuum relief valve, and low pressure drain appropriately located on the irrigation pipeline to prevent water source contamination from back flow.
2. The pesticide injection pipeline must contain a functional, automatic, quick-closing check valve to prevent the flow of fluid back toward the injection pump.
3. The pesticide injection pipeline must also contain a functional, normally dosed, solenoid-operated valve located on the intake side of the injection pump and connected to the system interlock to prevent fluid from being withdrawn from the supply tank when the irrigation system is either automatically or manually shut down.

Adminsitrative Record
Page 34517

4. The system must contain functional interlocking controls to automatically shut off the pesticide injection pump when the water pump motor stops.
5. The irrigation line or water pump must include a functional pressure switch which will stop the water pump motor when the water pressure decreases to the point where pesticide distribution is adversely affected.
6. Systems must use a metering pump, such as a positive displacement injection pump (e.g., diaphragm pump) effectively designed and constructed of materials that are compatible with pesticides and capable of being fitted with a system interlock.

One possible design is shown schematically in Figure 64 (Zoldoske et al. 2004).

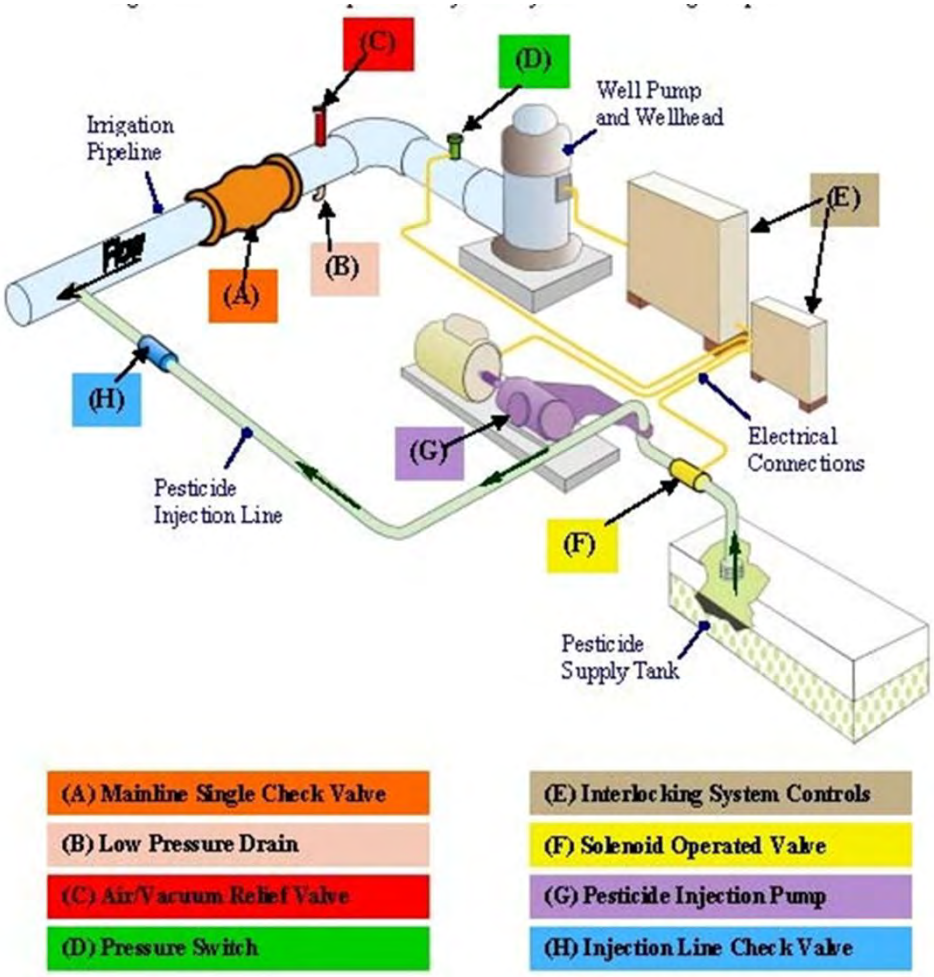


Figure 64. Possible backflow prevention device schematic (Zoldoske et al. 2004).

There are a variety of other backflow prevention devices. The air gap method is another acceptable backflow prevention device, which the California Department of Pesticide Regulation concludes is the most reliable form of backflow prevention. This method provides a physical separation between the source water and the pesticide treated water, with the source water pipe required to be at least two pipe diameters above the level of the pesticide-laden water below. A variety of other alternatives exist

Adminsitrative Record
Page 34518

as well. Either a chemigation valve or a gooseneck pipe loop can be used to meet the first listed requirement. A hydraulically operated check valve may be used in place of the solenoid controlled valve on the pesticide tank. A venturi can replace the positive displacement pesticide injection pump to draw pesticides into the irrigated water. The solenoid controlled valve and the quick-closing check valve can be replaced by a spring loaded check valve. The full list of acceptable alternatives to the required backflow prevention device in pesticide handling areas can be found in the CDPR document *Chemigation Safety Devices: Pesticide Label Requirements and Allowable Alternative Equipment* (Department of Pesticide Regulation 2001).

9.3 Dry Wells

9.3.1 Introduction to Dry Wells

Dry wells are structures that capture surface water runoff and redirect it into the ground. These vertical drains promote infiltration to groundwater because they are designed to be more permeable than the surrounding soils, and are usually filled with coarse grained sediments such as gravel. They are also an example of a direct-entry pathway to groundwater, a pathway by which runoff can bypass relatively less permeable soil layers and more quickly reach the water table. Dry wells are variable in depth, and may be just several meters [feet] in depth or 18-30 m [60-100 ft] in depth, depending on local soil characteristics and infiltration needs. Dry wells are widely used in many regions of the United States, as they are an effective method by which to dispose of excess surface water. They are most commonly used in agricultural settings, but dry wells are also an important drainage method for some urban locations, such as the City of Modesto.

Because a dry well provides a rapid pathway by which surface waters may reach deep groundwater sources, groundwater contamination through dry wells is an important concern known since at least the 1980s. Many questions persist regarding groundwater contamination via dry wells. A limited amount of information has been collected regarding the detrimental consequences to groundwater due to dry well use, because many farmers and other dry well users have been extremely reluctant to participate in past dry well studies. Here, we summarize the most common contaminants to enter and pass through dry wells, to compare agricultural and urban dry well systems, and to determine the feasibility of quantifying dry wells in the San Joaquin Valley of California.

9.3.2 Dry Wells and Groundwater Contamination

Several 1980s studies identified the presence of pesticides in groundwater in agricultural lands, but did not provide a direct link between dry wells as a potential source of contamination and pesticides in groundwater (Troiano & Segawa 1987; Pickett *et al.* 1990). A later study located simazine contamination in groundwater of the citrus growing region of Tulare County. Because Tulare County has a relatively impermeable layer about 6 -9 m (20-30 feet) below the ground surface and because simazine was found much lower in the ground, researchers concluded that direct-entry pathways such as dry wells must contribute toward groundwater contamination (DeMartinis & Royce 1998).

Adminsitrative Record
Page 34519

The potential for groundwater contamination exists even if a dry well is some distance away from a chemical application site. In one study, the California Department of Transportation surveyed rain water runoff near roads in San Joaquin County. Samples were collected near the road and at locations of suspected dry wells along the path of the runoff, and the samples were analyzed for several pesticides. Scientists found average concentrations of simazine, diuron, and bromacil to be 367.3 ppb, 219.8 ppb, and 8.5 ppb, respectively (Braun & Hawkins 1991). This study demonstrated that surface water runoff has the potential to incorporate detectable levels of pesticides which may then reach nearby dry wells, resulting in potential groundwater contamination.

9.3.3 Dry Well Regulations

The United States Environmental Protection Agency is the federal regulatory body for dry well usage (United States Environmental Protection Agency 2003). Dry wells are categorized as Class V injection wells, a broad characterization for storm drainage wells and other miscellaneous wells. Class V wells must follow the regulations of the federal Underground Injection Control (UIC) program because they have the potential to adversely impact underground sources of drinking water (USDWs). The UIC program protects USDWs by requiring that fluids entering Class V wells cannot pose a threat to the public water system (United States Environmental Protection Agency 2003). This program may require permits for drainage wells, but storm drainage wells don't require a permit if no USDWs are endangered and if the drainage well complies with federal UIC program requirements (United States Environmental Protection Agency 2003).

9.3.4 Urban Dry Wells

Dry well use is not restricted to agricultural land; dry wells are also present in some urban areas. The City of Modesto is one urban center that utilizes an extensive network of dry wells. Positive storm drains account for about 20% of Modesto's storm drainage system (City of Modesto), collecting runoff, conveying runoff through pipes, and discharging runoff directly to one of three surface waters: the Tuolumne River, Dry Creek, or Modesto Irrigation Canals. The remaining 80% of the storm drainage system consists of dry wells, which the City of Modesto staff refers to as rock-wells. These structures receive runoff from a catch basin in the street gutter, routing runoff for percolation to the subsurface below. Modesto dry wells are generally 7.5 – 15 m (25-50 feet) in depth, and are typically filled with gravel to promote rapid infiltration. Dry wells are the main drainage method in the older portion of the city's drainage system, and the system includes around 11,000 dry wells (City of Modesto). The city also maintains several large detention/retention basins, which are recreational areas during dry seasons and runoff collection locations during wet seasons, eventually draining through dry wells. A map of the entire drainage system is available on the City of Modesto website (City of Modesto 2008).

While Modesto dry wells may contain similar herbicides and pesticides as those found in agricultural dry wells, other industrial and household compounds are also typically found. Most notably, detergents and cleaners, petroleum products, and metals may be present in Modesto dry wells. The City of Modesto monitors the water quality of its storm drain system, and regularly collects and analyzes water samples.

Adminsitrative Record
Page 34520

Dry well samples are tested for many constituents, including nitrate. A full list of the regulatory limits of Modesto dry well contaminants is available through the City of Modesto (Creedon 2008). The City of Modesto website encourages citizens to limit pesticide use in the yard, noting that the pesticides diazinon and chlorpyrifos occur in storm water. These contaminants are the result of pesticide use to control household pests (City of Modesto 2011).

Although statistics on the nitrogen concentrations for storm water discharged to dry wells were not readily available, City of Modesto provided information that nitrate concentrations were 3.62 mg/L and 3.44 mg/L at two distinct sites on 17 February 2011. These same sites are reported to have total ammonium and organic nitrogen concentrations of 3.83 mg/L and 10.9 mg/L on 11 March 2011 (City of Modesto, personal communication, 12 September 2011). The latter indicates the potential for nitrate contamination above the MCL as organic and ammonium nitrogen eventually convert to nitrate.

9.3.5 Dry Well Prevalence

No dry well map exists for the San Joaquin Valley or the Tulare Lake Basin. (We did not conduct a survey for Salinas Valley). Unsuccessful past attempts to quantify dry wells indicate that any current regional dry well estimate would involve a great deal of uncertainty. One issue is that some land owners may be unaware of abandoned dry wells on their properties. A 1986 report estimated 5,000 abandoned drainage wells in the Central Valley, which may contribute toward groundwater contamination (Holden 1986). Over time, these abandoned dry wells could become forgotten and unreported in a future mapping project. However, a greater problem is farmers' wariness to report dry wells. One Tulare Lake Basin study noted that 7% of surveyed citrus farmers reported dry wells on their property (Pickett *et al.* 1990), but a later study detected pesticides throughout the regional groundwater system and concluded that dry wells were located on the lower end of almost all citrus groves in the area (DeMartinis & Royce 1998). DPR wanted to fund projects to quantify dry wells in the 1990s, but landowners strongly resisted participating in these studies. Accurate counts of dry wells are therefore unobtainable at this time.

9.4 Inactive and Abandoned Wells

9.4.1 Introduction to Abandoned Wells

Inactive, abandoned, and improperly destroyed wells are those that are no longer in use, seasonally not in use, or that have not been properly destroyed (Figure 65). They are of concern because they can be a direct pathway between surface water runoff and groundwater, and between shallow and deeper groundwater. These pathways can provide a rapid method by which surface water contamination can reach the groundwater. In contrast, well destruction is a process in which old wells are legally and safely destroyed. Well destruction processes require permits from the Department of Water Resources, and there are statewide well destruction requirements to prevent potential groundwater contamination. Well destruction is a costly processes, ranging from several thousand to several tens of thousands of dollars (see section 3.6). While well destruction may be very costly, well abandonment is not illegal under state law. Abandoned wells are a concern to the general public as they can be difficult to identify

Adminsitrative Record
Page 34521

and because they pose a clear threat to the quality of the local groundwater.In this section, we attempt to quantify the incidence of well destruction and well abandonment within Monterey, Fresno, Kern, Kings, and Tulare counties in California. Obtaining estimates of the incidence of well destruction was expected to be of greater certainty than obtaining estimates of the incidence of well abandonment. Because well destruction includes a permitting process, permits should be documented and stored by counties and the state. A public well which fails to meet water quality standards may become abandoned but will be documented. However, private abandoned wells are not formally reported and may be difficult to identify. Consequently, there is a great deal of uncertainty as to how many abandoned wells exist within this region.

9.4.2 Quantifying Permitted Well Destruction

Permitted well destruction values were estimated using data obtained from the Department of Water Resources South Central Region Office. The Department of Water Resources maintains well completion records for all California wells using a database that has been in place since 1977: domestic wells, irrigation wells, public wells, industrial wells, monitoring wells, well deepenings, and other permits. The "other" category includes several types of permits for various well modifications, such as those for well destruction, cathodic protection, well tests, vapor extraction, sparging, and direct push/injection. DWR estimates that 90-95% of permits in this category are well destruction permits. Using the yearly permit totals provided by DWR, the upper and lower bounds for the number of wells destroyed annually was calculated by multiplying the yearly totals by 0.90 and by 0.95. The yearly values for 1977-2009 were next added for each of the counties to obtain the total number of wells destroyed for each county. The entire dataset is summarized data below in Table 60.

Table 60. Estimated Number of Well Destructions in Five Counties; 1977-2009

County	Estimated # of Well Destruction	Estimated Destruction Rate (wells/yr)
Fresno	3,729-3,936	117-123
Kern	1,589-1,677	50-52
Kings	487-514	15-16
Monterey	1,883-1,987	59-62
Tulare	1,208-1,275	38-40
All Study Area	8,895-9,389	278-293

9.4.3 Quantifying Abandoned Wells

By definition, an abandoned well is one that has not been properly destroyed and documented; therefore, estimates obtained of the number of abandoned wells for the study area are likely to have a wide margin of error. The interviewed county employees provided rough estimates, at best, for the number of abandoned wells in each county. Kings County Community Development Agency could not offer an estimate for the number of abandoned wells in Kings County. Monterey County Environmental

Adminsitrative Record
Page 34522

Health was also unable to provide an estimate for the number of wells which have been abandoned in Monterey County, but noted that employees have encountered at least 85 potentially abandoned wells when conducting routine work. Tulare County estimated that thousands of abandoned wells may exist. Tulare County Environmental Health noted that 44 well destruction permits were on hold because well owners did not properly destroy their wells. The Kern County Environmental Health Division is aware of 180 abandoned wells and property owners have received notification of these violations, but no estimate exists of the total number of abandoned wells. Fresno County Environmental Health estimates that there are a total of 2,500-4,000 improperly abandoned wells in Fresno County.

Well abandonment rates for agricultural wells were estimated using information obtained from the Department of Water Resources. The Department of Water Resources South Central Region Office estimated that 10-20% of all agricultural wells are abandoned within the entire study area. The abandonment percentage is likely much lower for domestic wells because these wells are usually a home's main source of water. In cases where a home, formerly on a domestic well, is incorporated into a water supply system through extension of the water supply infrastructure, it is assumed that county involvement in the permitting process results in proper destruction of the well. Similarly, home owners who discontinue use of their well due to contamination are most likely to have discovered that contamination through testing conducted by their county environmental health department, and therefore will generally have been required to properly destroy the well. No attempt was made to quantify the number of domestic wells abandoned due to the abandonment of the homes themselves. For public wells, due to relatively consistent regulatory oversight, it was assumed that abandonment was well documented in PICME (discussed further below). Thus, agricultural wells are likely to be the main source of undocumented abandoned wells within the study area. Using the 10-20% rate from DWR, well abandonment estimates were calculated using yearly agricultural well completion record totals, which DWR staff provided. The agricultural well completion totals account only for agricultural wells completed during the years 1977-2009 (see Appendix Table 9), so older wells are excluded from the estimates. We used a range, 10% to 20%, to estimate the number of abandoned agricultural wells (Table 61).

Table 61. Estimates of number of abandoned agricultural wells constructed between 1977 and 2009 in five counties using abandonment rate of 10% - 20%.

County	Abandoned Agricultural Wells: 1977-2009
Fresno	496-991
Kern	147-294
Kings	150-300
Monterey	1444-288
Tulare	446-892
All Study Area Counties	**1,383-2,766**

Prior to 1977, the Department of Water Resources Well Completion Report (WCR) database estimates that as many as 70,467 wells had been completed. If we assume that the relative distribution of well types (agricultural, private domestic, public) was the same as from 1977 – 2009 (see Appendix Table 9,

Adminsitrative Record
Page 34523

Section 9.2.3), then about 21,000 agricultural wells had been constructed prior to 1977. At a 10-20% rate of abandonment, agricultural wells already abandoned by 1977 range from 2,100 to 4,200 wells. For the study area, this estimate suggests a total of 3,500 to 7,000 abandoned agricultural wells, a range that is consistent with the above county estimates.

The PICME database, maintained by the Information Center for the Environment (ICE) at UC Davis for the California Department of Public Health (CDPH) is a list of all California public water supply wells, and it provides information on these wells in tabular form, including well location and well status. The PICME database does not contain records for private domestic drinking water wells or irrigation wells. The PICME database was queried to gather information on the incidence of public well destruction and abandonment. This database lists the status of all public wells in California. The main categories in this database are active, inactive, abandoned, and destroyed wells. Inactive wells are those that are not in service for periods of one year or more, but that may be used in the future. Abandoned wells are those which are no longer in use, with no intention of future use. The PICME database was summarized by county for abandoned wells only. The results are displayed in Table 62. Comparing these values to the estimated number of abandoned agricultural wells, it is clear that abandoned agricultural wells greatly outnumber abandoned public wells. The PICME database provides the current status of its wells.

Table 62. Number of abandoned public wells in five counties of the SBX2 1 study area (Data Source: CDPH PICME database).

County	Abandoned Public Wells
Fresno	143
Kern	80
Kings	14
Tulare	106
Monterey	39
All Study Area	*382*

9.5 Estimating N Loading and Transfer from Active, Abandoned, Improperly Destroyed, or Dry Wells

9.5.1 Introduction to N Loading

Currently, no specific data exists that provides an accurate or even approximate estimation of the amount of nitrate (or other dissolved inorganic and organic nitrogen) introduced from the surface into groundwater, or transferred from contaminated water-bearing zones into uncontaminated water-bearing zones via active wells, seasonally or permanently inactive wells, abandoned wells, improperly destroyed wells, or dry wells.

Adminsitrative Record
Page 34524

Here we attempt an approximation based on estimated failure rates of active wells, the estimated number and size of abandoned wells described in the previous section, an estimate of dry wells in the project area described above, the estimated annual downward flow rate in any of these failing wells.

9.5.2 Methods for N Loading Quantification

How much water and nitrate leaks through a well? The amount of water and nitrate (or other nitrogenous compounds) leaked into or between aquifers depends on the diameter of the well, the filling material of the well in the case of dry wells, the well annulus material (filter pack material, sealing material), and the concentration of nitrate (and nitrogenous compounds) in the leaking water.

Here we consider two pathways: a leaky annulus and the inside of the well casing. We also consider two mechanisms of nitrate sources: surface discharge into a well due to a stormwater or irrigation event; and in-well leakage between overlying aquifers due to downward pressure gradients. We consider two types of wells, a domestic well with a 6 inch [174 mm] casing and a 2 inch [51 mm] annulus, and an irrigation or large municipal well with a 24 inch [704 mm] casing and 4 inch [102 mm] annulus.

To compute a flow rate through the annulus, we use the following formula:

flow rate [m^3/d] = gradient [m/m] x effective hydraulic conductivity [m/d] x cross-sectional area [m^2]

How many wells leak? Using the data presented in the previous 2 sections (2.9.3, 2.9.4), a summary of constructed, abandoned, and destroyed wells was prepared, including pre-1977 wells. The distribution of pre-1977 wells among the three well categories was assumed to be the same as that between 1977 and 2009. For the data presented in Table 63 we further used the total number of public supply wells abandoned, we assumed a small number (1%) of the total amount of domestic wells to be abandoned, and we used the upper limit of the range for the abandonment rate presented in previous section (20% of constructed wells). That provided an estimate of the total number of abandoned wells to date. We further used DWR's estimate of the total number of destroyed wells, and assumed that the distribution of destruction is similar to the distribution of abandoned wells among well types, which puts the majority of destroyed wells actually in the agricultural well sector (Table 63).

A fraction of the domestic wells listed in Table 63 are located outside the study area, in the foothills and mountains surrounding the study area, but within county boundaries. This fraction is likely proportional to the fraction of rural population living outside the study area relative to the total rural population in these counties. The larger agricultural and municipal wells are likely all located, with few exceptions, within the study area boundaries, which reflects the major production aquifer.

Besides the above DWR data on the number of well constructions, we have records for the number of active public supply wells in the study that serve public supply systems with more than 15 connections: 3,460 (from analysis of PICME data, see Section 9.4.3), a number that is consistent with the number of

Adminsitrative Record
Page 34525

active municipal wells recorded in Table 63 (3,320), although these two records reflect two different administrative systems.

Even more surprising is the strong agreement between the estimated number of domestic wells, based on DWR well construction permits (total active domestic wells: 75,421; see Table 64), and the estimated number of domestic wells for household self-supplied and local small water systems, based on 2010 census data, which yielded 245,490 people using 74,391 wells (see Table 5 in Technical Report 7 by Honeycutt et al., 2012). This would confirm a) that few domestic wells are abandoned; and b) that most of the domestic wells are within the study area.

Adminsitrative Record
Page 34526

Table 63. Summary of the total amount of wells constructed, inactive or abandoned, and destroyed per information provided in previous sections. The following data were used: 1977-2009 well completion records at DWR, total number of well completion records constructed pre-1977, PICME data on abandoned municipal wells, total number of well destruction records at DWR. The following assumptions were made: pre-1977 ratio of the number of wells between well-type is assumed equal to 1977-2009 distribution, negligible (1%) rate of domestic well abandonment, 20% rate (of total constructed) agricultural well abandonment, relative distribution well destruction numbers among well-types is equal to that for abandoned wells. We note that the above numbers are estimates.

	Total Number of Wells				Average Annual Rate			
	Domestic	Agricultural	Municipal	All Wells	Domestic	Agricultural	Municipal	All Wells
Fresno Co. 1977-2009	15,712	4,955	740	21,407	476	150	22	648
Kings Co. 1977-2009	1,501	1,500	84	3,085	45	45	3	93
Tulare Co. 1977-2009	5,722	4,462	439	10,623	173	135	13	321
Kern Co. 1977-2009	3,701	1,472	222	5,395	112	45	7	164
Tulare Lake Basin **Total**	26,636	12,389	1,485	40,510	806	375	45	1,226
Montery Co. 1977-2009	3,744	1,442	147	5,333	113	44	4	161
Total Constructed 1977-2009	30,380	13,831	1,632	45,843	919	419	49	1,387
Total Constructed Pre-1977	46,698	21,260	2,509	70,467	934	426	50	1,409
Total Constructed to Date	77,078	35,091	4,141	116,310	907	413	49	1,368
Total Abandoned to Date	771	7,018	382	8,171				
Total Destroyed to Date	886	8,064	439	9,389				
Total Active Wells	75,421	20,009	3,320	98,750				

The above estimate for active agricultural wells in the study area (20,000) is larger than the number of wells simulated based on the average groundwater pumpage estimated by Faunt et al. in their Central Valley Hydrologic Model (CVHM), a groundwater flow model of the Central Valley (Faunt et al. 2009). Total annual pumping rate in the TLB, averaged from 1962 to 2003, was 7.0 km^3/yr (5.7 million AF/yr) (*ibid.*,their Table B3). For the Salinas Valley, average total annual pumping is 0.63 km^3/yr (510,000 AF/yr) (see Technical Report 4, Chapter 3, Boyle et al. 2012). The study area total pumping rate is

Adminsitrative Record
Page 34527

approximately 7.5 km^3/yr (6.1 million AF/yr), of which approximately 90% are for agricultural usage. For the high resolution transport model of the TLB, we simulated wells based on a distribution of pumping rates (see Technical Report 4, Chapters 2 and 7, Boyle et al., 2012) and estimated that the total number of wells to meet the TLB pumping is 5,600 wells (not including domestic wells or other very small production wells)

Alternatively, estimates for the number of agricultural wells shown in Table 63, together with the known amount of agricultural pumping 6.8 km^3/yr (5.5 million AF/yr), can be used to derive an average pumping rate and check that number against known pumping rates: Assuming that practically all of that pumpage occurs during the summer 6 months, the average pumping rate on 20,009 wells is 1.3 m^3/min (343 gpm). The remaining 0.074 km^3/yr (60,000 AF/yr) pumped by municipal wells (3,320 wells) mostly within the same time period, yields an average pumping rate of 0.85 (225 gpm). These are average year pumping rates. Many agricultural wells only operate during dry years, when the total pumping in the TLB is as high as 11 km^3/yr (9 million AF/yr) or more, thus increasing the average pumping rate to approximately 1.9 m^3/min (500 gpm). These numbers are not unreasonable, but are at the lower end of the typical range of agricultural wells (many pumping 3.0 – 7.6 m^3/min or 800 - 2,000 gpm).

Given that the estimated pumping rate appears rather low, and if we assumed that the estimated number of active agricultural wells in Table 63 is correct, it would therefore need to be assumed that as many as 5,000 to 10,000 of these estimated 20,000 active agricultural wells are in fact inactive. Adding these to the inactive or abandoned wells already listed in Table 61, this calculation would indicate that at least 20,000 of the estimated 35,000 agricultural wells constructed to date within the study area are either inactive at any given time, or improperly destroyed and undocumented.

These data provide a basis from which to estimate the potential nitrate leaching into or within wells, whether they are active, inactive or abandoned, or improperly destroyed. To estimate a nitrate discharge into improperly sealed wells and nitrate leakage via wells between aquifers, we further make the following simplifying assumptions about the wells listed in Table 63. Other scenarios can be constructed for similar analysis. Actual data to support or rebut the assumptions below are not available. The example is used to illustrate the overall potential leakage potential.

- The largest one-third (33%) of all active 3,320 municipal and 20,009 irrigation wells span multiple aquifers and experience downward leakage in the annulus and/or inside of the well casing for half the year (180 days) –7,776 active wells with internal aquifer-to-aquifer leakage
- 1 in 100 large wells and 1 in 1,000 domestic wells have a faulty well-head and/or seal and are subject to irrigation or stormwater discharge that leaks directly into the well and the leaky annulus: 233 large active production wells, 78 large inactive production wells, and 75 small domestic wells.
- The deepest one-third (33%) of the inactive or abandoned 7,018 agricultural and 382 municipal wells span multiple aquifers and experience downward leakage in the annulus and/or inside of the well casing for the entire year (365 days) – 2,466 inactive or abandoned wells with year-round internal aquifer-to-aquifer leakage

Adminsitrative Record
Page 34528

- We assume that 4,000 of the estimated 5,000 Central Valley dry wells (Holden 1986) are within the study area and mostly within agricultural areas. At a rate of 1 in 4 wells, these are subject to significant nitrate leakage due to stormwater or irrigation run-on: 1,000 leaking dry wells.
- The effective nitrate (and other nitrogenous compounds) loading in water from irrigation or stormwater discharge into a well is 100 lbs N/acft (approximately 30 mg N/L). This represents a worst case scenario, so the average value may be significantly lower.
- The nitrogen loading in aquifer leakage (between overlying aquifers) is 50 lbs N/AF (approximately 15 mg N/L). This represents a worst case scenario. The actual average value is likely lower.
- We make further geotechnical assumptions appropriate for typical large production wells and typical domestic/small production wells. Values are listed in Table 64.

9.5.3 Results & Discussion of N Loading

Larger diameter production wells are capable of leaking a significantly larger amount of water than small diameter domestic wells. The annual surface discharge into an improperly sealed or leaking large well can amount to as much as the total amount of percolating recharge over an 8 ha (20 ac) irrigated parcel at a recharge rate of 300 mm/yr (1 AF/ac/yr). For small domestic wells, leakage is nearly an order of magnitude smaller than in large diameter wells.

Small domestic wells leak approximately an order of magnitude less nitrogen than large wells, and very few are thought to regularly be subject direct surface discharge into the well with high nitrate water. Hence, the total discharge into wells and nitrogen loading from small active wells in TLB and SV is nearly negligible.

The amount of nitrogen loading to groundwater from direct discharge into inactive or abandoned wells and dry wells is estimated to be approximately 200 Mg N/yr (220 tons/yr) total, two-thirds of which would be contributed by dry wells, which are designed specifically for field or stormwater runoff drainage. The mean nitrate value assumed here for dry well leakage assumes location in an agricultural field. Surface discharge into active wells with leaky seals also accounts for approximately 200 Mg N/year. These computed values are considered to be a worst case scenario. Actual surface discharge of nitrogen across the study area is likely smaller, as not every discharge event carries the high concentration of nitrate assumed in Table 64.

Adminsitrative Record
Page 34529

Table 64. Results of an approximate worst case scenario for aquifer nitrate loading due to well leakage in the study area (Tulare Lake Basin and Salinas Valley). Surface discharge refers to the incidental discharge of irrigation water or stormwater runoff into an improperly constructed well, a damaged well, or a dry well, either directly into the well casing or via the annulus. Intra-well leakage refers to the leakage of groundwater from shallow water-bearing units to deeper water-bearing units via the annulus between multiple aquifers.Input / Assumed Values (Worst Case Scenario).

	Large Production Wells	Domestic Wells
borehole diameter [inches]	32	10
casing diameter [inches]	24	6
downward gradient in well annulus, surface discharge [ft/ft]	1	1
downward gradient in well annulus for intra-well leakage [ft/ft]	0.5	0.5
hydraulic conductivity of the annulus [ft/d]	1,000	1,000
flow rate (gpm), surface discharge into well casing (assumed)	200	20
flow rate (gpm), intra-well leakage through well casing (assumed)	20	5
N concentration, surface discharge, lbs N/acft	100	100
N concentration, aquifer-to-aquifer leakage, lbs N/acft	50	50
days per year of surface discharge (from precipitation or irrigation)	20	20
days per year of aquifer leakage (no pumping)	180	180
number of active wells with frequent surface discharge	233	75
number of active wells with intra-well leakage	7,776	0
number of inactive wells with frequent surface discharge	78	
number of inactive wells with intra-well leakage	2,466	0
number of dry wells with frequent surface discharge	1,000	0
Output / Computed Values		
thickness of annulus [inches]	4	2
cross-sectional area of well [sq.ft.]	3.14	0.20
cross-sectional area of annulus [sq.ft.]	2.44	0.35
cross-sectional area of dry well [sq.ft.]	5.59	-
flow rate (gpm), surface discharge through annulus	12.69	1.81
flow rate (gpm), intra-well leakage through annulus	6.35	0.91
flow rate (gpm), dry well	28.93	-
flow rate (af/yr), surface discharge into well	18	2
flow rate (af/yr), surface discharge through annulus	1.13	0.16
flow rate (af/yr), intra-well leakage through well	16	4
flow rate (af/yr), intra-well leakage through annulus	5.08	0.73
flow rate (af/yr), dry well	2.58	-
total annual N load from surface discharge, into active wells [tons]	221	7
total annual N load from intra-well leakage, active wells [tons]	4,104	0
total annual N load from surface discharge, into inactive/dry wells [tons]	74	0
total annual N load from intra-well leakage, inactive wells [tons]	2,603	0
total annual N load from dry wells [tons]	129	-
Total Annual N Load [short-tons]	**7,130**	**7**

Adminsitrative Record
Page 34530

In contrast to the amount of direct surface discharge of N into groundwater via wells (nearly 400 Mg N/yr or 430 tons/yr), our analysis suggests that as much as 6,100 Mg N/yr (6,700 tons/yr) are leaked from shallow to deeper aquifers with lower head, thus potentially far outweighing the direct loading from surface spillage within the overall TLB and SV groundwater basin context. Given the large number of wells that may contribute to this rapid downward transfer of nitrate, the leakage of nitrate across multiple water-bearing zones must be considered a significant potential source of nitrate contamination in otherwise well-protected / longer-term protected, deeper water-bearing zones.

The total amount of aquifer-to-aquifer leakage resulting from the assumptions in Table 64 is 0.20 km^3/yr (164,000 AF/yr) in active wells and 0.13 km^3/yr (104,000 AF/yr) in inactive and abandoned wells, totaling 0.33 km^3/yr (268,000 AF/yr) of downward water transfer from the shallower aquifer. How realistic is such transfer of water between aquifers via long well screens? Faunt *et al.* (2009) developed a groundwater model for the Central Valley representing groundwater conditions from 1962 through 2003. Average total annual pumpage in the TLB within that period was estimated to be 7 km^3/yr (5.65 million AF/yr), 60.7% of the total pumping in the Central Valley. Their results also estimated intra-well aquifer-to-aquifer leakage of 0.49 km^3/yr (400,000 AF/yr) for the entire Central Valley. If the reported leakage is assumed to be distributed across the Central Valley in proportion to the groundwater pumping, then the TLB leakage rate is on the order of 0.30 km^3/yr (243,000 AF/yr), similar to the aquifer-to-aquifer leakage assumptions in Table 64.

The analysis depends on a number of broad assumptions listed above and in Table 9. The scenario analysis is strictly a mass balance analysis, and the results are directly proportional (and can therefore be easily scaled) to either the number of wells assumed to discharge / leak or the rate of discharge / leakage, both of which are highly uncertain numbers, and about which no specific further data exist in the TLB and SV groundwater basins. The above estimated nitrogen loading rates, from surface discharge to groundwater and from vertical cross-aquifer leakage, are reasonable estimates for an upper limit of loading. The actual downward nitrate transfer rate through all types of wells is thought to be on the order of 3,000 to 7,000 Mg N/yr (3,300 – 7,800 tons/yr).

We note that this transfer of nitrogen from shallow groundwater to deeper groundwater does not appear in the summary of nitrate sources (Section 1). This transfer does not constitute a new source of nitrate to groundwater. It does, however, constitute a source of nitrate for deep groundwater that would otherwise be unaffected.

9.6 Summary and Conclusions

Wells can be an important and significant conduit for incidental nitrate discharge from the land surface into groundwater or for nitrate transfer from shallow, nitrate contaminated groundwater to deeper, clean groundwater. Wells become conduits for direct discharge of nitrate from the land surface to groundwater or from shallow to deeper groundwater due to:

Adminsitrative Record
Page 34531

- backflow from containers containing nitrogenous fertilizer compounds due to lack of backflow prevention devices
- lacking or leaky surface seals
- lacking, insufficiently deep, or incompetent well seals
- lack of seal between shallower, contaminated water-bearing units and deeper, uncontaminated water-bearing units of the study area aquifer system
- long screen intervals across multiple water-bearing units of variable water quality within the aquifer system

The above conditions may lead to nitrate contamination of groundwater in active wells, inactive wells, and abandoned wells. Improperly destroyed wells (only partially sealed, not properly filled with a competent seal) may also be conduits for nitrate leakage into groundwater. Dry wells are designed to rapidly infiltrate surface runoff directly into groundwater, by-passing the soil and deep unsaturated zone.

Over the last century, more than 75,000 domestic wells, over 35,000 agricultural wells, and over 4,000 public supply wells have been constructed in the study area.

Wells and well characteristics vary considerably between counties in the study area (Tulare Lake Basin and Salinas Valley), partly due to varying county requirements, partly due to varying groundwater conditions. Due to existing contaminant concerns in some areas, Monterey County well construction requirements differentiate between different groundwater sub-basins. Most of the counties do not require the annular space between two aquifers to be sealed, except in areas of known contamination concerns. However, Kern County does not allow wells to be screened through multiple aquifers.

Several projects have sought to determine the effectiveness of well seals, and determined that both well seal hydraulic conductivity and structural stability are important considerations during well construction. Two studies confirm that construction methods and site geology are as important to consider as the sealant material itself.

Regulation of backflow prevention devices is conducted at the state level, but only for pesticides (California Department of Pesticide Regulation). Regulations and enforcement to require similar backflow prevention in the mixing and application of fertilizer are lacking in California.

Dry wells provide a pathway for untreated surface waters to directly reach groundwater. Although no dry well count exists for the project area, there is ample evidence that dry wells are known to be abundant throughout the region, perhaps as many as a few thousand. At the basin scale, nitrate loading to groundwater via dry well is effectively part of total cropland nitrate loading. While likely an insignificant fraction compared to diffuse recharge through the unsaturated zone, it can be a locally significant nitrate conduit to groundwater. There are other types of structures that can act as a direct-entry pathway through the soil and lead to groundwater contamination, and more work is needed on this topic to determine the importance of dry wells toward groundwater contamination in comparison to other direct-entry pathways. Further research would be needed to create a dry well map for the study area, however, previous mapping attempts have been unsuccessful.

Adminsitrative Record
Page 34532

Most of the domestic wells are considered to be active or seasonally active. Approximately 3,300 public supply wells are known to be currently active. Between 10,000 to 20,000 agricultural wells are thought to be only seasonally active for irrigation, at least in dry years, and between 7,000 and 17,000 agricultural wells are estimated to be permanently inactive, or abandoned. Nearly 8,000 agricultural and nearly 2,000 domestic and public water supply wells have been destroyed under proper procedures.

A preliminary worst-case scenario analysis suggests that poor well construction, wells that are in disrepair, and dry wells may contribute as much as 0.4 Gg N/yr (430 tons/yr) to groundwater nitrate loading from various sources. In contrast, as much as 6.1 Gg N/yr (6,700 tons/yr) may be leaked from shallow to deeper aquifers with lower pressure potential.

Given the large number of wells that may contribute to this rapid downward transfer of nitrate, the leakage of nitrate across multiple water-bearing zones should be considered a significant potential groundwater-internal source of nitrate contamination in otherwise well-protected / longer-term protected, deeper water-bearing zones. The management of this nitrate transfer from shallow groundwater to deep groundwater is different from nitrate discharge sources. Nitrate is already in groundwater. Proper well construction can largely avoid this process.

Further research is needed to better understand and quantify down-well flow in seasonally active, inactive or abandoned wells screened across multiple aquifers to obtain better estimates of the regional deeper groundwater nitrate contamination via in-well downward flow from shallow aquifers.

Section 6 in Technical Report 3 (Dzurella et al., 2012) discusses remedies and costs for addressing these various conduits of groundwater nitrate contamination.

Adminsitrative Record
Page 34533

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Adminsitrative Record
Page 34534

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Appendix of Tables and Figures

Appendix Table 1. Select estimates of NO3 leaching in California. Methods follow: ie = anion exchange resin bags, sl = suction lysimeters, lys = lysimeter, ss = soil sample, H2O = water samples via tile drains. Robbins (1980) measurements from Idaho.

Crop	*Soil type*	*N input (kg ha^{-1} yr^{-1})*	*Water applied (cm ha^{-1} yr^{-1} or ET)*	*irrigation method*	*Depth of measurement (m)*	*N leached (kg/ha/yr)*	*% leached*	*Method*	*Source*
strawberry, barley	sandy loam	364	58	furrow	0 – 15	258	70.9	ss	Adriano 1972a
celery, sweet corn	clay loam	1271	76	furrow	0 – 15	725	57.0	ss	Adriano 1972a
cabbage, green onion, celery, romaine	loam	678	66	furrow	0 – 12	444	65.4	ss	Adriano 1972a
sugarbeets, grain	sandy loam	180	14	furrow	0 – 15	73	40.5	ss	Adriano 1972a
grain, sugarbeets	loamy sand	184	39	furrow	0 – 15	189	102.3	ss	Adriano 1972a
watermelon, carrots	fine loam	160	12	furrow	0 – 15	22	13.8	ss	Adriano 1972a
potatoes, cereal	sandy loam	330	13	furrow	0 – 15	48	14.4	ss	Adriano 1972a
potatoes, sweet corn	fine sandy loam	486	26	furrow	0 – 14	291	59.8	ss	Adriano 1972a
alfalfa, potatoes, barley	loamy fine sand	399	31	furrow	0 – 15	218	54.5	ss	Adriano 1972a
asparagus	sandy loam	112	45-53	furrow	0 – 15	32	28.9	ss	Adriano 1972b
asparagus	sandy loam	112	75-53	furrow	0 – 15	34	29.9	ss	Adriano 1972b
asparagus	sandy loam	560	45-53	furrow	0 – 15	217	38.7	ss	Adriano 1972b
asparagus	sandy loam	560	75-53	furrow	0 – 15	88	15.7	ss	Adriano 1972b
celery	sandy loam	135	180	furrow	0 – 15	87	64.5	ss	Adriano 1972b
celery (8), tomato, lettuce	sandy loam	405	180	furrow	0 – 15	194	47.8	ss	Adriano 1972b
carrot	loamy sand	280		sprinkler	1			ie	Allaire-Leung 2001
carrot	loamy sand	258		sprinkler	1	22.5	8.7	ie	Allaire-Leung 2001
citrus (watershed)	sandy loams + others		99	furrow, drip, and sprinkler		80.2	60.0	H_2O	Binghamton 1984
corn	fine sandy loam	0	.33 ET	furrow	0.24 – 0.3	28.6		sl	Broadbent and Carlton 1980
corn	fine sandy loam	90	.33 ET	furrow	0.24 – 0.3	44.2	49.1	sl	Broadbent and Carlton 1980
corn	fine sandy loam	180	.33 ET	furrow	0.24 – 0.3	67.6	37.6	sl	Broadbent and Carlton 1980
corn	fine sandy loam	360	.33 ET	furrow	0.24 – 0.3	98.3	27.3	sl	Broadbent and Carlton 1980
corn	fine sandy loam	0	.33 ET	furrow	0.24 – 0.3	23.8		sl	Broadbent and Carlton 1980
corn	fine sandy	90	.33 ET	furrow	0.24 – 0.3	75.8	84.2	sl	Broadbent and

Adminsitrative Record
Page 34555

	loam								Carlton 1980
corn	fine sandy loam	180	.33 ET	furrow	0.24 – 0.3	93.8	52.1	sl	Broadbent and Carlton 1980
corn	fine sandy loam	360	.33 ET	furrow	0.24 – 0.3	117	32.5	sl	Broadbent and Carlton 1980
corn	fine sandy loam	0	1.0 ET	furrow	0.24 – 0.3	21.1		sl	Broadbent and Carlton 1980
corn	fine sandy loam	90	1.0 ET	furrow	0.24 – 0.3	41.7	46.3	sl	Broadbent and Carlton 1980
corn	fine sandy loam	180	1.0 ET	furrow	0.24 – 0.3	54.2	30.1	sl	Broadbent and Carlton 1980
corn	fine sandy loam	360	1.0 ET	furrow	0.24 – 0.3	78.9	21.9	sl	Broadbent and Carlton 1980
corn	fine sandy loam	0	1.0 ET	furrow	0.24 – 0.3	20.1		sl	Broadbent and Carlton 1980
corn	fine sandy loam	90	1.0 ET	furrow	0.24 – 0.3	27.2	30.2	sl	Broadbent and Carlton 1980
corn	fine sandy loam	180	1.0 ET	furrow	0.24 – 0.3	22	12.2	sl	Broadbent and Carlton 1980
corn	fine sandy loam	360	1.0 ET	furrow	0.24 – 0.3	154	42.8	sl	Broadbent and Carlton 1980
corn	fine sandy loam	0	5/3 ET	furrow	0.24 – 0.3	21.5		sl	Broadbent and Carlton 1980
corn	fine sandy loam	90	5/3 ET	furrow	0.24 – 0.3	14.9	16.6	sl	Broadbent and Carlton 1980
corn	fine sandy loam	180	5/3 ET	furrow	0.24 – 0.3	27.1	15.1	sl	Broadbent and Carlton 1980
corn	fine sandy loam	360	5/3 ET	furrow	0.24 – 0.3	117	32.5	sl	Broadbent and Carlton 1980
corn	fine sandy loam	0	5/3 ET	furrow	0.24 – 0.3	14.6		sl	Broadbent and Carlton 1980
corn	fine sandy loam	90	5/3 ET	furrow	0.24 – 0.3	21	23.3	sl	Broadbent and Carlton 1980
corn	fine sandy loam	180	5/3 ET	furrow	0.24 – 0.3	28.6	15.9	sl	Broadbent and Carlton 1980
corn	fine sandy loam	360	5/3 ET	furrow	0.24 – 0.3	48.8	13.6	sl	Broadbent and Carlton 1980
greenhouse: roses		17.12	1.25 ET			3.65	21.3	lys	Cabrerra and Evans 1993
greenhouse: roses		26.7	1.25 ET			10.63	39.8	lys	Cabrerra and Evans 1993
greenhouse: roses		41.02	1.25 ET			20.25	49.4	lys	Cabrerra and Evans 1993
greenhouse: roses		19.19	1.10 ET			4.22	22.0	lys	Cabrerra and Evans 1993
greenhouse: roses		25.93	1.25 ET			9.82	37.9	lys	Cabrerra and Evans 1993
greenhouse: roses		50.27	1.50 ET			28.16	56.0	lys	Cabrerra and Evans 1993
lettuce		127	1.14 ET	furrow		3.5	2.8	sl	Cahn

Adminsitrative Record
Page 34556

									unpublished
lettuce		127	1.54 ET	furrow		4	3.1	sl	Cahn unpublished
lettuce		253	1.5 ET	furrow		23.5	9.3	sl	Cahn unpublished
corn, carrots	coarse-loamy	345		furrow	0.61 – 1.83	151	43.8	sl	Devitt 1976
corn, carrots	coarse-loamy	396		furrow	0.61 – 1.84	155	39.1	sl	Devitt 1976
lemons	sandy	26		furrow	0.61 – 1.85	46	176.9	sl	Devitt 1976
dates	coarse-silty	149		furrow	0.61 – 1.86	62	41.6	sl	Devitt 1976
cotton	sandy	492		furrow	0.61 – 1.87	71	14.4	sl	Devitt 1976
sorghum	fine	224		furrow	0.61 – 1.88	119	53.1	sl	Devitt 1976
cotton	sandy over clayey	203		furrow	0.61 – 1.89	26	12.8	sl	Devitt 1976
cotton	sandy over clayey	169		furrow	0.61 – 1.90	35	20.7	sl	Devitt 1976
lemons		134				35	27.2	ss	Embleton 1980
lemons		57				43	69.5	ss	Embleton 1980
lemons		165				67	42.2	ss	Embleton 1980
lemons		165				139	85.5	ss	Embleton 1980
lemons		57				31	57.6	ss	Embleton 1980
lemons		165				109	63.3	ss	Embleton 1980
lemons		486				239	60.2	ss	Embleton 1980
citrus	sandy loam			sprinkler			10.3	model	Gardenas 2005
grape	sandy loam			drip			25.9	model	Gardenas 2005
strawberry	sandy loam			surface tape			23.0	model	Gardenas 2005
processing tomatoes	sandy loam			subsurface tape			7.6	model	Gardenas 2005
citrus	loamy			sprinkler			5.9	model	Gardenas 2005
grape	loamy			drip			6.6	model	Gardenas 2005
strawberry	loamy			surface tape			12.2	model	Gardenas 2005
processing tomatoes	loamy			subsurface tape			0.2	model	Gardenas 2005
citrus	clay			sprinkler			1.0	model	Gardenas 2005
grape	clay			drip			1.0	model	Gardenas 2005
strawberry	clay			surface tape			8.5	model	Gardenas 2005
processing tomatoes	clay			subsurface tape			0.0	model	Gardenas 2005
citrus	silty clay			sprinkler			0.1	model	Gardenas 2005
grape	silty clay			drip			0.0	model	Gardenas 2005
strawberry	silty clay			surface tape			9.1	model	Gardenas 2005
processing tomatoes	silty clay			subsurface tape			0.0	model	Gardenas 2005
lettuce, lettuce	clay loam	184	16.9	furrow		146.4	79.6	model	Jackson 1994
lettuce, lettuce	clay loam	356	16.9	furrow		160.4	45.1	model	Jackson 1994

Adminsitrative Record
Page 34557

lettuce, lettuce	clay loam	356	11.9	furrow		108.9	30.6	model	Jackson 1994
broccoli	clay loam	134		furrow		25	18.4	ie	LeStrange FREP
broccoli	clay loam	269		furrow		99	36.7	ie	LeStrange FREP
potatoes, broccoli, beans	sandy loam, loam, loamy sand	250	0.94	furrow	0 – 6	94	37.6	ss	Lund 1982 + Lund (NSF)
artichokes	sandy loam, loam	300	0.64	furrow	0 – 6	64	21.3	ss	Lund 1982 + Lund (NSF)
cauliflower, broccoli	loam, sandy loam	620	1.28	furrow	0 – 3.6	128	20.6	ss	Lund 1982 + Lund (NSF)
lettuce, broccoli, celery	loam, sandy loam	550	1.44	furrow	0 – 4.2	144	26.2	ss	Lund 1982 + Lund (NSF)
almonds	sand	153	1.2		0.91 – 6.10	71	46.4	sl & model	Nolan 2010
corn silage	sandy loam	195	1.2		0.91 – 6.11	102	52.3	sl & model	Nolan 2010
peach		91	120 – 215	flood		57	62.6	mb	Onsy
peach		201	120 – 215	flood		93	46.3	mb	Onsy
peach		456	120 – 215	flood		275	60.3	mb	Onsy
citrus		154	70.66666667			39	25.3	ss	Pratt 1971
citrus		154	80			148	96.1	ss	Pratt 1971
citrus		167	87.09677419			116	69.5	ss	Pratt 1971
citrus		188	78.75			43	22.9	ss	Pratt 1971
citrus		122			6 – 15	84	68.9	ss	Pratt and Adriano 1973
citrus		111			6 – 15	73	65.8	ss	Pratt and Adriano 1973
citrus		256			6 – 15	122	47.7	ss	Pratt and Adriano 1973
citrus		414			6 – 15	148	35.7	ss	Pratt and Adriano 1973
citrus		330			6 – 15	141	42.7	ss	Pratt and Adriano 1973
citrus		194			6 – 15	82	42.3	ss	Pratt and Adriano 1973
citrus		154			6 – 15	41	26.6	ss	Pratt and Adriano 1973
citrus		154			6 – 15	140	90.9	ss	Pratt and Adriano 1973
citrus		167			6 – 15	113	67.7	ss	Pratt and Adriano 1973
citrus		168			6 – 15	42	25.0	ss	Pratt and Adriano 1973
asparagus		130			6 – 15	25	19.2	ss	Pratt and Adriano 1973
asparagus		144			6 – 15	42	29.2	ss	Pratt and Adriano 1973

Adminsitrative Record
Page 34558

asparagus		478			6 – 15	134	28.0	ss	Pratt and Adriano 1973
asparagus		492			6 – 15	111	22.6	ss	Pratt and Adriano 1973
celery		385			6 – 15	225	58.4	ss	Pratt and Adriano 1973
celery		1663			6 – 15	481	28.9	ss	Pratt and Adriano 1973
misc. row crops		437			6 – 15	310	70.9	ss	Pratt and Adriano 1973
misc. row crops		1525			6 – 15	912	59.8	ss	Pratt and Adriano 1973
misc. row crops		740			6 – 15	561	75.8	ss	Pratt and Adriano 1973
misc. row crops		210			6 – 15	84	40.0	ss	Pratt and Adriano 1973
misc. row crops		215			6 – 15	219	101.9	ss	Pratt and Adriano 1973
misc. row crops		480			6 – 15	73	15.2	ss	Pratt and Adriano 1973
misc. row crops		360			6 – 15	47	13.1	ss	Pratt and Adriano 1973
misc. row crops		530			6 – 15	320	60.4	ss	Pratt and Adriano 1973
misc. row crops		435			6 – 15	239	54.9	ss	Pratt and Adriano 1973
alfalfa	silt loam		68		0.25 – 2	44		ss	Robbins 1980
beans	silt loam		58		0.25 – 3	85		ss	Robbins 1980
beans	silt loam		1		0.25 – 4	87		ss	Robbins 1980
bens	silt loam		70		0.25 – 5	23		ss	Robbins 1980
peas	silt loam				0.25 – 6			ss	Robbins 1980
corn+n	silt loam	200	55		0.25 – 7	153	76.5	ss	Robbins 1980
corn	silt loam		48		0.25 – 8	60		ss	Robbins 1980
beans	silt loam		46		0.25 – 9	96		ss	Robbins 1980
wheat	silt loam		35		0.25 – 10	29		ss	Robbins 1980
beans	silt loam		15		0.25 – 11	17		ss	Robbins 1980
beans	silt loam		4		0.25 – 12	12		ss	Robbins 1980
alfalfa	silt loam		3		0.25 – 13	10		ss	Robbins 1980
corn+n	silt loam	170	45		0.25 – 14	108	63.5	ss	Robbins 1980
corn	silt loam		39		0.25 – 15	17		ss	Robbins 1980
turf	loamy sand, sandy loam	464	1 ET			27	5.9	lys	Wu 2007
turf	loamy sand, sandy loam	464	130 ET			27	5.8	lys	Wu 2007
turf	loamy sand, sandy loam	464	1 ET			11	2.4	lys	Wu 2007
turf	loamy sand, sandy loam	464	130 ET			10	2.2	lys	Wu 2007

Adminsitrative Record
Page 34559

turf	loamy sand, sandy loam	464	1 ET			9	2.0	lys	Wu 2007
turf	loamy sand, sandy loam	464	130 ET			8	1.8	lys	Wu 2007
vegetables	gilman	900			tile drain	152	16.9	H_2O	Letey 1979 NSF
vegetables	indio	350			tile drain	151	43.1	H_2O	Letey 1979 NSF
vegetables	indio	234			tile drain	120	51.3	H_2O	Letey 1979 NSF
grapefruit	coachella	0			tile drain	50		H_2O	Letey 1979 NSF
cotton	kettlemen	151			tile drain	350	231.8	H_2O	Letey 1979 NSF
melons	oxalis				tile drain	172		H_2O	Letey 1979 NSF
cotton	oxalis	150			tile drain	146	97.3	H_2O	Letey 1979 NSF
alfalfa, tomato	oxalis	118			tile drain	118	100.0	H_2O	Letey 1979 NSF
safflower, cotton	kettlemen				tile drain	100		H_2O	Letey 1979 NSF
cotton, sugar beets	oxalis	132			tile drain	46	34.8	H_2O	Letey 1979 NSF
barley, alfalfa	oxalis	63			tile drain	11	17.5	H_2O	Letey 1979 NSF
alfalfa	oxalis	0			tile drain	10		H_2O	Letey 1979 NSF
alfalfa, lettuce	coachella	410			tile drain	122	29.8	H_2O	Letey 1979 NSF
alfalfa	gilman	0			tile drain	30		H_2O	Letey 1979 NSF
wheat, alfalfa	imperial	292			tile drain	28	9.6	H_2O	Letey 1979 NSF
wheat	niland	203			tile drain	26	12.8	H_2O	Letey 1979 NSF
alfalfa, sugar beets	imperial	302			tile drain	18	6.0	H_2O	Letey 1979 NSF
cotton, wheat	imperial	654			tile drain	6	0.9	H_2O	Letey 1979 NSF
alfalfa	imperial	44			tile drain	4	9.1	H_2O	Letey 1979 NSF
none	panoche	0			tile drain	214		H_2O	Letey 1979 NSF
cotton	panoche	168			tile drain	90	53.6	H_2O	Letey 1979 NSF
alfalfa	panoche	0			tile drain	67		H_2O	Letey 1979 NSF
cotton	panoche	168			tile drain	46	27.4	H_2O	Letey 1979 NSF
cotton	panoche	168			tile drain	46	27.4	H_2O	Letey 1979 NSF
lettuce, celery	cropley-salinas	480			tile drain	930	193.8	H_2O	Letey 1979 NSF
lettuce, celery	pacheco	637			tile drain	383	60.1	H_2O	Letey 1979 NSF
celery	clear lake	728			tile drain	277	38.0	H_2O	Letey 1979 NSF
lettuce	clear lake	580			tile drain	138	23.8	H_2O	Letey 1979 NSF
lettuce	clear lake	580			tile drain	103	17.8	H_2O	Letey 1979 NSF
corn, sudan grass	columbia	717			tile drain	336	46.9	H_2O	Letey 1979 NSF
alfalfa	myers	0			tile drain	38		H_2O	Letey 1979 NSF
cotton	metz	280			tile drain	17	6.1	H_2O	Letey 1979 NSF
wheat	tulare	134			tile drain	7	5.2	H_2O	Letey 1979 NSF
cotton	pacheco	148			tile drain	4	2.7	H_2O	Letey 1979 NSF
cotton	pacheco	292			tile drain	3	1.0	H_2O	Letey 1979 NSF

Adminsitrative Record
Page 34560

Appendix Table 2. Hectares of land cover classes in the 2010 CAML Map.

Area of Land Cover Classes in CAML 2010 Map Within Study Boundary						
DWR Code number	***Land cover type***	***Fresno County hectares***	***Kern County hectares***	***Kings County hectares***	***Monterey County hectares***	***Tulare County hectares***
3	Annual Grassland	43,105	271,729	39,766	61,370	33,872
4	Alkali Desert Scrub	351	29,314	930	0	0
6	Barren	22	284	59	1,792	11
8	Blue Oak-Foothill Pine	21	45	0	19	15
9	Blue Oak Woodland	3,227	1,063	24	14,616	4,375
10	Coastal Oak Woodland	14	52	1	7,133	0
11	Closed-Cone Pine-Cypress	0	0	0	16	0
12	Chamise-Redshank Chaparral	0	0	0	90	0
13	Coastal Scrub	0	9	0	6,802	0
22	Freshwater Emergent Wetland	2,982	2,516	898	0	366
28	Lacustrine	64	41	102	0	3
32	Mixed Chaparral	3	0	0	368	0
35	Montane Hardwood-Conifer	0	0	0	27	0
36	Montane Hardwood	711	68	0	46	559
39	Perennial Grassland	0	490	0	0	0
43	Riverine	391	177	0	0	0
49	Saline Emergent Wetland	0	0	0	249	0
53	Urban	57,871	59,071	16,678	22,835	28,589
55	Valley Oak Woodland	0	106	0	118	11
56	Valley Foothill Riparian	263	147	384	4,177	332
57	Water	2,354	2,264	122	300	388
59	Wet Meadow	0	0	0	0	1
62	Undetermined Shrub Type	114	3,232	105	7	67
63	Undetermined Conifer Type	19	6	0	0	34

Adminsitrative Record
Page 34561

77	Eucalyptus	0	0	0	22	0
300	Citrus and Subtropical (Also Miscellaneous subtropical and jojoba)	343	706	5	60	274
301	Grapefruit	65	229	0	14	591
302	Lemons	469	2,154	0	499	1,589
303	Oranges	14,265	24,536	48	0	46,352
305	Avocados	2	0	0	0	77
306	Olives	332	307	136	6	8,040
308	Kiwis	170	92	105	1	611
310	Eucalyptus	156	50	128	161	69
400	Deciduous Fruits and Nuts	2,017	7,464	1,679	0	2,151
401	Apples	2,818	4,896	1,265	27	1,876
402	Apricots	524	353	352	12	188
403	Cherries	660	2,487	338	0	308
405	Peaches and Nectarines	16,599	1,735	4,285	0	12,659
406	Pears	123	0	24	0	520
407	Plums	8,290	849	1,424	0	8,601
408	Prunes	730	0	25	0	1,592
409	Figs	1,063	141	13	0	12
412	Almonds	35,115	77,356	5,135	0	7,386
413	Walnuts	2,285	866	4,999	109	14,257
414	Pistachios	4,957	25,418	4,114	0	4,091
600	Field Crops (includes Flax, Hops, Castor Beans, Miscellaneous Field, and Millet)	24,793	19,075	28,152	914	7,727
601	Cotton	99,046	45,861	71,410	0	28,307
602	Safflower	1,153	601	1,578	0	453
605	Sugar Beets	5,235	202	690	0	1,637
606	Corn (Field and Sweet)	12,348	9,973	20,197	57	42,275
607	Grain sorghum	79	1,747	287	0	1,101
608	Sudan	1,446	3,587	1,483	0	1,759

Adminsitrative Record
Page 34562

610	Beans (dry)	2,740	1,646	1,838	1,879	4,684
612	Sunflowers	9	0	0	0	9
700	Grain and Hay (includes miscellaneous)	27,309	46,120	35,064	6,234	27,853
701	Barley	218	380	329	83	8
702	Wheat	1,674	2,706	6,643	0	1,410
703	Oats	452	0	607	43	232
901	Idle – Cropped Past 3 Years	3,332	9,615	611	186	5,066
902	Idle – New Lands	272	3	0	413	31
1450	Native Vegetation	0	0	0	1	8
1455	Brush and Timber	240	0	0	0	0
1600	Pasture	295	0	0	51	68
1601	Alfalfa	33,159	38,985	33,736	506	42,690
1602	Clover	40	0	0	0	0
1603	Mixed pasture	3,753	1,958	3,256	416	2,298
1604	Native Pasture	1,025	147	8	73	427
1606	Miscellaneous grasses	147	0	0	0	38
1607	Turf farms	167	222	0	86	2
1800	Rice (includes rice & wild rice subclasses)	5	0	0	0	0
1901	Farmstead (with residence)	3,544	1,597	1,793	757	3,431
1902	Livestock feedlot operation	560	265	113	212	439
1903	Dairy farm	1,500	2,863	2,526	40	6,489
1904	Poultry farm	1,343	42	426	14	227
1905	Farmstead (without residence)	0	161	0	0	0
2000	Truck, Nursery, Berry Crops (includes cole mix, mixed, and misc. truck crops)	3,248	992	0	17,165	297
2001	Artichokes	2	0	0	3,550	5
2002	Asparagus	201	147	111	1,551	0
2003	Beans (green)	10	152	0	65	4
2006	Carrots	23	14,172	4,219	802	4
2007	Celery	0	1	0	1,917	0

Adminsitrative Record
Page 34563

2008	Lettuce	778	469	0	19,512	13
2009	Melons, squash, cucumbers	8,820	1,931	552	78	387
2010	Onions and garlic	10,775	4,693	1,822	1,679	198
2011	Peas	0	0	0	202	0
2012	Potatoes	11	2,308	0	275	0
2013	Sweet Potatoes	122	279	0	0	0
2014	Spinach	0	0	0	610	0
2015	Tomatoes (processing)	39,565	4,186	9,114	1,145	945
2016	Flowers, nursery, Christmas tree farms	444	1,514	0	728	875
2019	Bush berries	8	161	0	41	9
2020	Strawberries	58	74	1	3,428	9
2021	Peppers	651	2,053	0	2,258	29
2022	Broccoli	375	60	68	7,501	359
2023	Cabbage	0	20	0	575	24
2024	Cauliflower	50	4	60	2,120	169
2025	Brussels sprouts	0	0	0	146	0
2200	Vineyards (includes table grapes, wine grapes, and raisins)	105,799	45,738	3,509	19,234	35,569

Adminsitrative Record
Page 34564

Appendix Table 3. 1990 Comparison of mapped vs. reported hectare totals.

DWR Code	Crop Type	Fresno Map Hectares	Fresno Crop Report Hectares	Kern Map Hectares	Kern Crop Report Hectares	Kings Map Hectares	Kings Crop Report Hectares	Monterey Map Hectares	Monterey Crop Report Hectares	Tulare Map Hectares	Tulare Crop Report Hectares
300	Citrus and Subtropical (Misc.)	17	628	481	1,011	16	140	0	0	31	1,217
301	Grapefruit	2	0	164	739	0	0	13	0	204	204
302	Lemons	184	409	989	1,488	0	0	141	0	1,110	1,680
303	Oranges	11,064	8,155	16,172	13,905	27	0	14	0	41,767	34,949
304	Dates	0	0	0	0	0	0	0	0	0	0
305	Avocados	0	0	12	0	0	0	10	16	122	483
306	Olives	554	489	1,192	1,061	372	462	0	0	7,628	6,110
308	Kiwis	171	132	307	361	115	127	2	5	737	811
400	Deciduous Fruits and Nuts	191	267	0	1,322	0	156	0	0	0	912
401	Apples	3,383	0	2,386	1,979	507	102	15	141	3,074	457
402	Apricots	151	212	290	301	50	101	24	0	195	219
403	Cherries	139	0	29	26	5	0	0	0	2	55
405	Peaches and Nectarines	14,040	10,634	1,825	1,845	2,749	1,796	0	150	11,157	7,128
406	Pears	47	121	20	208	42	0	0	0	10	297
407	Plums	9,448	7,164	1,386	1,444	1,072	726	0	0	10,597	7,380
408	Prunes	454	488	97	48	0	0	0	0	864	2,255
409	Figs	1,538	1,214	309	193	22	0	0	0	24	18
412	Almonds	19,631	11,884	35,790	32,924	1,230	1,239	0	0	4,576	3,789
413	Walnuts	1,681	1,207	871	752	2,835	2,015	106	129	11,750	9,855
414	Pistachios	1,689	736	9,845	9,211	2,428	2,112	0	0	2,083	2,037

Adminsitrative Record
Page 34565

600	Field Crops (includes Flax, Hops, Castor Beans, Miscellaneous Field, and Millet)	27,310	0	26,946	0	17,537	3,546	1,791	587	5,151	8,466
601	Cotton	121,764	152,279	130,050	131,431	93,901	105,509	0	0	58,394	56,799
602	Safflower	7,450	2,671	1,888	2,227	19,294	14,461	0	0	1,446	0
605	Sugar Beets	4,508	8,094	4,309	5,348	2,336	267	1,162	1,109	839	1,700
606	Corn (Field and Sweet)	7,659	7,082	2,529	2,234	9,838	5,611	472	178	29,651	27,600
607	Grain sorghum	0	0	436	263	0	0	0	0	0	1,457
608	Sudan	977	0	647	0	347	0	84	0	817	0
610	Beans (dry)	608	5,261	2,387	4,503	165	736	550	2,023	1,527	5,747
612	Sunflowers	5	0	0	0	0	0	0	0	12	0
700	Grain and Hay (misc.)	30,547	9,267	23,639	22,460	15,460	520	9,586	530	40,072	14,083
701	Barley	0	6,758	0	7,580	13	15,618	0	6,863	0	9,348
702	Wheat	0	20,959	0	12,192	0	26,058	0	724	0	23,963
703	Oats	0	0	0	0	0	0	32	0	0	0
1600	Pasture	0	20,234	1	2,833	0	25,900	13	121	0	27,721
1601	Alfalfa	28,647	48,279	43,988	44,067	23,454	46,083	1,185	1,202	31,118	42,492
1602	Clover	21	0	0	0	0	0	0	0	0	0
1603	Mixed pasture	5,673	0	1,706	0	976	0	838	0	2,858	0
1604	Native Pasture	1,127	0	40	0	562	0	301	0	1,053	0
1605	Induced high water table native pasture	0	0	0	0	0	0	5	0	0	0
1606	Miscellaneous grasses	0	0	0	0	0	0	0	0	0	0
1607	Turf farms	29	0	99	0	107	0	0	0	0	0

Adminsitrative Record
Page 34566

1800	Rice (includes rice & wild rice subclasses)	0	2,509	216	236	0	0	0	0	0	0
2000	Truck, Nursery, Berry Crops (includes cole mix, mixed, and misc. truck crops)	2,603	3,885	1,168	4,968	242	1,398	13,842	2,107	1,259	3,613
2001	Artichokes	0	0	5	0	0	0	2,946	2,821	0	0
2002	Asparagus	189	0	281	348	381	0	1,871	1,955	0	0
2003	Beans (green)	0	0	500	0	136	0	408	0	337	0
2006	Carrots	0	591	3,621	11,959	0	0	373	1,287	0	0
2007	Celery	0	0	1	0	47	0	1,486	1,347	0	0
2008	Lettuce	5,488	6,924	1,345	3,188	85	113	12,303	14,408	71	0
2009	Melons, squash, cucumbers	9,991	17,122	2,085	4,072	1,009	601	124	161	567	0
2010	Onions and garlic	8,286	13,901	4,677	4,266	994	0	428	621	203	0
2011	Peas	12	0	7	0	87	0	0	0	0	0
2012	Potatoes	0	0	7,466	9,667	0	0	124	405	0	0
2013	Sweet Potatoes	17	554	324	0	40	0	0	0	0	0
2014	Spinach	0	0	0	0	0	0	1,207	1,349	0	0
2015	Tomatoes (processing)	37,970	42,087	1,995	2,104	4,423	1,821	841	3,144	274	0
2016	Flowers, nursery, Christmas tree farms	666	0	1,793	0	242	0	362	0	787	0
2019	Bush berries	5	0	12	0	0	0	93	320	3	0
2020	Strawberries	193	73	4	0	0	0	2,422	2,359	0	0
2021	Peppers	232	575	425	0	0	0	781	1,740	202	0

Adminsitrative Record
Page 34567

2022	Broccoli	0	2,015	0	0	0	223	11,643	9,028	8	0
2023	Cabbage	0	0	19	0	0	0	247	170	0	0
2024	Cauliflower	0	0	0	0	0	0	3,486	9,079	0	0
2025	Brussels sprouts	0	0	0	0	0	0	173	232	0	0
2027	Greenhouse	0	0	0	0	0	0	0	0	0	0
2200	Vineyards (includes table grapes, wine grapes, and raisins)	96,548	83,223	36,898	31,948	1,971	1,542	13,379	13,417	31,147	27,511

Adminsitrative Record
Page 34568

Appendix Table 4. 1977 Comparison of mapped vs. reported hectare totals.

DWR Code	Crop Type	Fresno Map Hectares	Fresno Crop Report Hectares	Kern Map Hectares	Kern Crop Report Hectares	Kings Map Hectares	Kings Crop Report Hectares	Monterey Map Hectares	Monterey Crop Report Hectares	Tulare Map Hectares	Tulare Crop Report Hectares
300	Citrus and Subtropical – Miscellaneous	523	523	499	499	0	0	0	0	1,332	1,333
301	Grapefruit	0	0	860	858	0	0	0	0	121	120
302	Lemons	337	337	1,758	1,757	0	0	0	0	2,116	2,131
303	Oranges	7,965	7,959	8,849	8,842	0	0	0	0	33,655	33,662
304	Dates	0	0	0	0	0	0	0	0	0	0
305	Avocados	0	0	48	48	0	0	0	0	564	569
306	Olives	643	642	2,666	2,665	546	546	0	0	6,077	6,069
308	Kiwis	0	0	45	44	0	0	0	0	2	0
400	Deciduous Fruits and Nuts	26	26	587	586	253	253	0	0	178	178
401	Apples	0	0	601	600	0	0	210	209	71	61
402	Apricots	171	171	93	93	70	71	87	87	66	65
403	Cherries	0	0	10	11	0	0	0	0	16	16
405	Peaches and Nectarines	6,623	6,617	2,023	2,021	1,022	1,021	0	0	5,008	4,959
406	Pears	0	0	115	115	0	0	0	0	104	105
407	Plums	3,697	3,694	1,035	1,034	411	410	0	0	5,063	5,037
408	Prunes	0	0	161	160	1	0	0	0	1,890	1,888
409	Figs	3,181	3,178	1,082	1,082	0	0	0	0	24	25
412	Almonds	6,829	6,824	22,153	22,136	2,021	2,019	0	0	3,352	3,341
413	Walnuts	1,652	1,652	394	395	2,128	2,127	80	73	11,693	11,685

Adminsitrative Record
Page 34569

414	Pistachios	0	0	5,464	5,459	1,418	1,416	0	0	378	378
600	Field Crops – Miscellaneous	0	0	0	0	578	512	90	119	16	0
601	Cotton	133,648	133,546	139,317	139,212	89,719	89,638	0	0	84,980	84,915
602	Safflower	3,632	3,630	647	647	12,384	12,150	122	121	110	110
605	Sugar Beets	4,414	4,411	5,751	5,747	976	976	5,012	5,020	1,791	1,789
606	Corn (Field and Sweet)	11,097	11,088	566	567	7,210	7,203	984	971	1,378	1,362
607	Grain sorghum	2,025	2,023	1,540	1,538	2,045	2,042	0	0	1,660	1,659
608	Sudan	0	0	0	0	0	0	0	0	1	0
610	Beans (dry)	2,211	2,210	2,026	2,023	0	0	4,898	4,891	2,571	2,568
612	Sunflowers	0	0	0	0	0	0	0	0	0	0
700	Grain and Hay (includes miscellaneous)	5,102	5,099	24,299	24,281	2,794	2,792	1,170	1,174	18,820	18,786
701	Barley	109,430	109,346	20,251	20,234	63,068	63,020	12,152	12,141	9,703	9,746
702	Wheat	11,016	11,007	12,151	12,141	20,658	20,642	405	405	8,349	8,362
703	Oats	0	0	0	0	0	0	606	607	102	101
1600	Pasture	16,201	16,187	75,329	75,272	22,268	22,252	607	607	5,286	5,293
1601	Alfalfa	47,789	47,753	47,052	47,016	30,604	30,581	3,720	3,713	21,068	21,044
1602	Clover	0	0	0	0	0	0	0	0	0	0
1603	Mixed pasture	0	0	0	0	0	0	0	0	2	0
1604	Native Pasture	0	0	0	0	0	0	0	0	3	0
1605	Induced high water table native pasture	0	0	0	0	0	0	0	0	0	0
1606	Miscellaneous grasses	0	0	0	0	0	0	0	0	0	0
1607	Turf farms	0	0	0	0	0	0	0	0	0	0

Adminsitrative Record
Page 34570

1800	Rice (includes rice & wild rice subclasses)	2,025	2,023	608	607	329	328	0	0	77	76
2000	Truck, Nursery, Berry Crops (includes cole mix, mixed, and misc. truck crops)	703	702	808	807	603	603	1,119	1,098	1,770	1,769
2001	Artichokes	0	0	0	0	0	0	3,770	3,764	0	0
2002	Asparagus	0	0	0	0	0	0	1,011	1,014	28	27
2003	Beans (green)	1,054	1,052	275	274	0	0	638	647	676	675
2006	Carrots	0	0	3,727	3,723	0	0	2,235	2,234	0	0
2007	Celery	0	0	0	0	0	0	1,253	1,244	1	0
2008	Lettuce	5,057	5,053	2,178	2,177	0	0	12,739	12,742	0	0
2009	Melons, squash, cucumbers	6,244	6,238	2,421	2,419	410	374	120	117	453	452
2010	Onions and garlic	1,157	1,155	4,630	4,626	0	0	2,846	2,849	1	0
2011	Peas	0	0	0	0	0	0	75	75	0	0
2012	Potatoes	0	0	11,746	11,736	0	0	1,506	1,505	0	0
2013	Sweet Potatoes	263	263	0	0	0	0	0	0	0	0
2014	Spinach	0	0	1	0	1	0	742	740	1	0
2015	Tomatoes (processing)	15,653	15,641	3,397	3,395	567	567	3,663	3,664	538	538

Adminsitrative Record
Page 34571

2016	Flowers, nursery, Christmas tree farms	0	0	0	0	0	0	0	0	0	0
2019	Bush berries	81	81	0	0	1	0	0	0	29	30
2020	Strawberries	65	65	0	0	0	0	1,208	1,222	0	0
2021	Peppers	234	235	0	0	0	0	1,896	1,908	184	185
2022	Broccoli	0	0	0	0	0	0	7,460	7,431	1	0
2023	Cabbage	0	0	0	0	0	0	350	345	0	0
2024	Cauliflower	0	0	0	0	1	0	4,581	4,593	1	0
2025	Brussels sprouts	1	0	1	0	1	0	264	262	1	0
2027	Greenhouse	0	0	0	0	0	0	0	0	0	0
2200	Vineyards (includes table grapes, wine grapes, and raisins)	78,200	78,141	30,195	30,172	1,510	1,509	13,669	13,620	30,243	30,204

Adminsitrative Record
Page 34572

Appendix Table 5. 1960 Comparison of mapped vs.reported hectare totals.

DWR Code	Crop Type	Fresno Map Hectares	Fresno Crop Report Hectares	Kern Map Hectares	Kern Crop Report Hectares	Kings Map Hectares	Kings Crop Report Hectares	Monterey Map Hectares	Monterey Crop Report Hectares	Tulare Map Hectares	Tulare Crop Report Hectares
300	Citrus and Subtropical – Miscellaneous	60	60	81	80	0	0	0	0	342	341
301	Grapefruit	7	7	4	3	0	0	0	0	64	63
302	Lemons	46	47	0	0	0	0	0	0	732	736
303	Oranges	1,457	1,456	990	989	0	0	0	0	17,806	17,734
304	Dates	0	0	0	0	0	0	0	0	0	0
305	Avocados	0	0	0	0	0	0	0	0	21	19
306	Olives	448	447	75	74	159	158	0	0	4,501	4,491
308	Kiwis	0	0	0	0	0	0	0	0	2	0
400	Deciduous Fruits and Nuts	39	38	168	168	1	1	0	0	83	83
401	Apples	48	47	83	82	0	0	265	264	115	106
402	Apricots	189	188	58	57	195	194	504	504	37	37
403	Cherries	0	0	0	0	0	0	40	40	11	12
405	Peaches and Nectarines	6,624	6,619	1,173	1,171	936	936	0	0	6,340	6,302
406	Pears	0	0	117	117	0	0	28	28	6	6
407	Plums	2,132	2,130	861	860	61	61	29	29	3,079	3,055
408	Prunes	41	41	3	3	1	0	0	0	448	447
409	Figs	5,501	5,497	0	0	0	0	0	0	109	110
412	Almonds	532	531	139	138	0	0	111	110	147	132
413	Walnuts	806	806	0	0	466	466	387	387	5,439	5,434

Adminsitrative Record
Page 34573

414	Pistachios	0	0	0	0	0	0	0	0	4	3
600	Field Crops – Miscellaneous	0	0	326	325	555	320	0	0	9	0
601	Cotton	98,841	98,766	94,242	94,170	48,885	48,833	0	0	74,374	74,317
602	Safflower	0	0	1,363	1,362	2,000	1,443	0	0	41	40
605	Sugar Beets	3,466	3,463	2,666	2,664	368	367	8,181	8,175	823	822
606	Corn (Field and Sweet)	8,756	8,749	1,253	1,251	3,532	3,528	1,317	1,315	1,934	1,924
607	Grain sorghum	12,960	12,950	9,941	9,934	2,136	2,134	0	0	14,174	14,164
608	Sudan	253	253	0	0	0	0	0	0	51	51
610	Beans (dry)	2,120	2,119	2,910	2,907	221	221	11,574	10,813	3,330	3,327
612	Sunflowers	61	61	0	0	85	85	0	0	163	162
700	Grain and Hay (includes miscellaneous)	20,552	20,535	5,447	5,442	1,512	1,510	4,455	5,042	2,569	2,530
701	Barley	126,769	126,674	34,727	34,701	82,799	82,736	21,466	21,448	18,222	18,211
702	Wheat	6,333	6,328	12,835	12,825	904	904	5,926	5,922	9,241	9,234
703	Oats	1,142	1,141	0	0	0	0	0	0	1,659	1,658
1600	Pasture	0	0	14,227	14,215	29,175	29,154	0	1,902	30,126	30,156
1601	Alfalfa	84,407	84,343	57,091	57,047	28,118	28,098	5,914	5,908	45,457	45,422
1602	Clover	0	0	93	93	0	0	0	0	0	0
1603	Mixed pasture	0	0	0	0	0	0	0	0	2	0
1604	Native Pasture	0	0	0	0	0	0	0	0	1	0
1605	Induced high water table native pasture	0	0	0	0	0	0	0	0	0	0
1606	Miscellaneous grasses	0	0	0	0	0	0	0	0	0	0
1607	Turf farms	0	0	0	0	0	0	0	0	0	0

Adminsitrative Record
Page 34574

1800	Rice (includes rice & wild rice subclasses)	8,167	8,161	1,653	1,651	16	16	1	0	738	737
2000	Truck, Nursery, Berry Crops (includes cole mix, mixed, and misc. truck crops)	324	325	0	0	33	32	674	672	61	60
2001	Artichokes	0	0	0	0	0	0	2,234	2,232	0	0
2002	Asparagus	0	0	0	0	0	0	0	0	565	564
2003	Beans (green)	44	45	0	0	1	0	1,101	1,101	32	29
2006	Carrots	69	68	413	412	0	0	2,634	3,191	0	0
2007	Celery	0	0	0	0	0	0	893	892	0	0
2008	Lettuce	491	490	97	97	0	0	9,391	1,202	40	40
2009	Melons, squash, cucumbers	11,479	11,469	2,660	2,658	763	763	181	180	1,054	1,053
2010	Onions and garlic	438	437	1,186	1,185	3	2	1,300	1,299	3	2
2011	Peas	0	0	798	797	0	0	1,043	608	49	49
2012	Potatoes	1,163	1,161	21,414	21,398	235	234	1,761	1,760	1,252	1,250
2013	Sweet Potatoes	272	271	76	76	6	6	0	0	3	3
2014	Spinach	49	49	0	0	0	0	1,641	748	0	0
2015	Tomatoes (processing)	162	162	0	0	9	8	2,674	2,671	378	380

Adminsitrative Record
Page 34575

2016	Flowers, nursery, Christmas tree farms	0	0	0	0	0	0	0	0	1	0
2019	Bush berries	111	110	0	0	0	0	48	49	21	21
2020	Strawberries	55	55	0	0	1	0	1,422	1,420	22	22
2021	Peppers	114	113	0	0	0	0	0	0	41	42
2022	Broccoli	0	0	0	0	1	0	1,866	1,822	1	0
2023	Cabbage	61	62	0	0	3	2	358	357	16	16
2024	Cauliflower	112	111	0	0	0	0	1,391	1,389	0	0
2025	Brussels sprouts	0	0	0	0	1	0	21	21	1	0
2027	Greenhouse	0	0	0	0	0	0	0	0	0	0
2200	Vineyards (includes table grapes, wine grapes, and raisins)	60,675	60,628	12,653	12,643	1,660	1,659	0	0	29,731	29,702

Adminsitrative Record
Page 34576

Appendix Table 6. 1946 Comparison of mapped vs. reported hectare totals.

DWR Code	Crop Type	Fresno Map Hectares	Fresno Crop Report Hectares	Kern Map Hectares	Kern Crop Report Hectares	Kings Map Hectares	Kings Crop Report Hectares	Monterey Map Hectares	Monterey Crop Report Hectares	Tulare Map Hectares	Tulare Crop Report Hectares
300	Citrus and Subtropical – Miscellaneous	242	241	45	44	0	0	0	0	454	455
301	Grapefruit	0	0	14	13	0	0	0	0	320	320
302	Lemons	0	0	4	3	0	0	0	0	514	516
303	Oranges	1,367	1,366	588	587	0	0	0	0	14,815	14,762
304	Dates	0	0	0	0	0	0	0	0	0	0
305	Avocados	0	0	0	0	0	0	0	0	3	3
306	Olives	486	486	168	168	119	118	0	0	2,994	2,985
308	Kiwis	0	0	0	0	0	0	0	0	2	0
400	Deciduous Fruits and Nuts	40	40	26	27	3	3	0	0	71	72
401	Apples	43	43	10	10	0	0	306	305	216	209
402	Apricots	835	835	173	173	1,063	1,062	741	739	365	365
403	Cherries	0	0	0	0	0	0	25	25	2	3
405	Peaches and Nectarines	3,877	3,875	261	260	1,293	1,292	0	0	5,443	5,414
406	Pears	9	8	32	32	0	0	90	91	17	17
407	Plums	764	764	750	750	59	60	0	0	1,728	1,706
408	Prunes	160	159	11	11	32	32	0	0	1,257	1,256
409	Figs	6,903	6,897	3	2	0	0	0	0	1,089	1,088
412	Almonds	90	89	69	68	0	0	1,499	1,499	172	158
413	Walnuts	260	259	4	5	242	241	227	218	1,548	1,546

Adminsitrative Record
Page 34577

414	Pistachios	0	0	0	0	0	0	0	0	1	0
600	Field Crops – Miscellaneous	7,785	7,778	0	0	3,863	3,859	0	0	8	0
601	Cotton	34,423	34,398	35,720	35,693	23,084	23,067	0	0	29,569	29,542
602	Safflower	0	0	0	0	0	0	0	0	1	0
605	Sugar Beets	1,012	1,012	3,574	3,571	291	291	9,343	9,344	641	640
606	Corn (Field and Sweet)	404	405	1,026	1,026	870	870	0	0	68	60
607	Grain sorghum	2,026	2,023	4,803	4,800	0	0	0	0	1,903	1,902
608	Sudan	0	0	1,013	1,012	0	0	0	0	244	243
610	Beans (dry)	130	129	419	418	2	2	11,652	11,640	1	0
612	Sunflowers	0	0	0	0	96	95	0	0	140	140
700	Grain and Hay (includes miscellaneous)	5,468	5,463	3,473	3,470	2,430	2,428	8,980	8,903	35	0
701	Barley	58,724	58,679	20,251	20,234	53,459	53,419	19,441	19,425	6,884	6,880
702	Wheat	15,796	15,783	21,181	21,165	7,291	7,284	8,910	8,903	13,764	13,759
703	Oats	2,430	2,428	1,701	1,700	1	0	0	0	283	283
1600	Pasture	18,226	18,211	405	405	0	0	0	0	12,143	12,141
1601	Alfalfa	40,499	40,469	33,621	33,596	12,554	12,545	0	0	38,070	38,040
1602	Clover	0	0	0	0	0	0	0	0	0	0
1603	Mixed pasture	0	0	0	0	0	0	0	0	2	0
1604	Native Pasture	0	0	0	0	0	0	0	0	0	0
1605	Induced high water table native pasture	0	0	0	0	0	0	0	0	0	0
1606	Miscellaneous grasses	0	0	0	0	0	0	0	0	0	0
1607	Turf farms	0	0	0	0	0	0	0	0	0	0

Adminsitrative Record
Page 34578

1800	Rice (includes rice & wild rice subclasses)	3,241	3,237	0	0	0	0	0	0	1	0
2000	Truck, Nursery, Berry Crops (includes cole mix, mixed, and misc. truck crops)	0	0	195	194	0	0	1,243	1,217	41	41
2001	Artichokes	0	0	0	0	0	0	1,617	1,619	0	0
2002	Asparagus	221	221	179	179	0	0	0	0	330	330
2003	Beans (green)	0	0	22	22	0	0	184	193	24	21
2006	Carrots	274	273	183	183	0	0	3,620	3,618	9	9
2007	Celery	21	20	0	0	0	0	306	295	537	536
2008	Lettuce	219	219	675	674	0	0	7,444	7,447	801	800
2009	Melons, squash, cucumbers	7,072	7,066	1,411	1,409	283	282	0	0	1,376	1,375
2010	Onions and garlic	98	97	1,037	1,036	0	0	844	850	21	20
2011	Peas	203	202	1,731	1,730	0	0	648	647	294	294
2012	Potatoes	851	850	26,488	26,468	33	33	142	142	3,522	3,519
2013	Sweet Potatoes	8	9	168	168	45	44	0	0	1	0
2014	Spinach	0	0	89	89	1	0	543	541	207	206
2015	Tomatoes (processing)	235	235	1,050	1,049	17	17	2,829	2,833	1,313	1,312

Adminsitrative Record
Page 34579

2016	Flowers, nursery, Christmas tree farms	0	0	0	0	0	0	0	0	1	0
2019	Bush berries	253	253	35	34	1	0	111	110	73	73
2020	Strawberries	0	0	0	0	0	0	113	103	34	33
2021	Peppers	0	0	153	153	0	0	31	40	34	34
2022	Broccoli	70	71	229	229	0	0	694	665	19	18
2023	Cabbage	73	73	16	16	1	0	152	148	0	0
2024	Cauliflower	0	0	62	62	0	0	208	202	71	71
2025	Brussels sprouts	0	0	0	0	0	0	74	74	0	0
2027	Greenhouse	0	0	0	0	0	0	0	0	0	0
2200	Vineyards (includes table grapes, wine grapes, and raisins)	68,818	68,765	7,655	7,649	4,797	4,794	121	41	31,186	31,164

Adminsitrative Record
Page 34580

Appendix Table 7. Typical nitrogen applied ("$N_{applied}$"), nitrogen yield ("$N_{harvest}$") [kg N/ha/yr], and partial nitrogen balance (PNB) for each crop in each of five time periods ("PERIOD"). $N_{applied}$ and $N_{harvest}$ represent the median of five years of data centered on the year listed in PERIOD. This analysis assumes that application rates from synthetic, manure, effluent, and biosolids fertilizer do not exceed the typical rate, $N_{applied}$.

DWR Code	CROP	CROP-GROUP	PERIOD	AREA [ha]	Napplied [kg/ha]	Nharvest [kg/ha]	PNB
300	citrus, pomegranates	Subtropical	1945	717	125	5.8	5%
300	citrus, pomegranates	Subtropical	1960	284	105	16.2	15%
300	citrus, pomegranates	Subtropical	1975	1,787	118	14.3	12%
300	citrus, pomegranates	Subtropical	1990	2,636	104	14.8	14%
300	citrus, pomegranates	Subtropical	2005	8,862	104	18.8	18%
301	grapefruit	Subtropical	1945	334	180	10.3	6%
301	grapefruit	Subtropical	1960	63	151	20.7	14%
301	grapefruit	Subtropical	1975	57	169	32.4	19%
301	grapefruit	Subtropical	1990	678	126	45.5	36%
301	grapefruit	Subtropical	2005	857	126	31.9	25%
302	lemons	Subtropical	1945	514	183	31.1	17%
302	lemons	Subtropical	1960	597	154	36.9	24%
302	lemons	Subtropical	1975	2,185	172	24.5	14%
302	lemons	Subtropical	1990	3,497	136	37.2	27%
302	lemons	Subtropical	2005	2,978	136	67.2	49%
303	oranges	Subtropical	1945	16,430	201	31.9	16%
303	oranges	Subtropical	1960	15,536	169	28.8	17%
303	oranges	Subtropical	1975	48,342	189	27.1	14%
303	oranges	Subtropical	1990	51,958	104	41.1	40%
303	oranges	Subtropical	2005	68,299	104	46.3	45%
305	avocadoes	Subtropical	1945		147		
305	avocadoes	Subtropical	1960	1	123	8.0	7%
305	avocadoes	Subtropical	1975	175	138	11.7	9%
305	avocadoes	Subtropical	1990	497	123	25.3	21%
305	avocadoes	Subtropical	2005	90	123	15.6	13%
306	olives	Subtropical	1945	3,105	96	77.0	80%
306	olives	Subtropical	1960	4,210	80	117.5	147%
306	olives	Subtropical	1975	5,914	90	119.1	132%
306	olives	Subtropical	1990	7,494	87	145.3	167%
306	olives	Subtropical	2005	6,374	87	128.2	147%
308	kiwi	Subtropical	1945		105		
308	kiwi	Subtropical	1960		88		
308	kiwi	Subtropical	1975	15	99	10.3	10%
308	kiwi	Subtropical	1990	1,295	111	19.0	17%

Adminsitrative Record
Page 34581

308	kiwi	Subtropical	2005	960	111	27.1	24%
400	persimmons, nuts (not walnuts or almonds)	Tree Fruit	1945	112	112	10.1	9%
400	persimmons, nuts (not walnuts or almonds)	Tree Fruit	1960	265	94	14.2	15%
400	persimmons, nuts (not walnuts or almonds)	Tree Fruit	1975	746	105	12.0	11%
400	persimmons, nuts (not walnuts or almonds)	Tree Fruit	1990	2,193	143	12.5	9%
400	persimmons, nuts (not walnuts or almonds)	Tree Fruit	2005	5,826	143	16.5	12%
401	apples	Tree Fruit	1945	495	94	12.3	13%
401	apples	Tree Fruit	1960	410	79	14.0	18%
401	apples	Tree Fruit	1975	724	88	14.9	17%
401	apples	Tree Fruit	1990	2,509	66	22.4	34%
401	apples	Tree Fruit	2005	1,752	66	19.0	29%
402	apricots	Tree Fruit	1945	2,942	103	13.2	13%
402	apricots	Tree Fruit	1960	965	87	25.0	29%
402	apricots	Tree Fruit	1975	382	97	21.2	22%
402	apricots	Tree Fruit	1990	757	104	29.3	28%
402	apricots	Tree Fruit	2005	1,382	104	29.1	28%
403	cherries	Tree Fruit	1945	29	113	9.0	8%
403	cherries	Tree Fruit	1960	41	95	6.6	7%
403	cherries	Tree Fruit	1975	14	106	4.7	4%
403	cherries	Tree Fruit	1990		75		
403	cherries	Tree Fruit	2005	2,610	75	10.9	14%
405	peaches, nectarines	Tree Fruit	1945	9,539	124	17.3	14%
405	peaches, nectarines	Tree Fruit	1960	12,708	104	20.3	20%
405	peaches, nectarines	Tree Fruit	1975	12,113	117	27.6	24%
405	peaches, nectarines	Tree Fruit	1990	20,116	114	29.3	26%
405	peaches, nectarines	Tree Fruit	2005	31,899	114	27.7	24%
406	pears	Tree Fruit	1945	148	137	10.5	8%
406	pears	Tree Fruit	1960	33	115	3.2	3%
406	pears	Tree Fruit	1975	207	129	5.8	4%
406	pears	Tree Fruit	1990	390	155	6.7	4%
406	pears	Tree Fruit	2005	602	155	19.8	13%
407	plums	Tree Fruit	1945	2,492	151	17.0	11%
407	plums	Tree Fruit	1960	5,234	127	9.0	7%
407	plums	Tree Fruit	1975	8,187	142	21.0	15%
407	plums	Tree Fruit	1990	15,841	114	20.6	18%
407	plums	Tree Fruit	2005	15,623	114	21.1	19%
408	prunes	Tree Fruit	1945	1,406	166	25.7	15%
408	prunes	Tree Fruit	1960	267	139	33.0	24%
408	prunes	Tree Fruit	1975	1,572	156	39.0	25%

Adminsitrative Record
Page 34582

408	prunes	Tree Fruit	1990	2,636	143	46.9	33%
408	prunes	Tree Fruit	2005	2,025	143	25.0	17%
409	figs	Tree Fruit	1945	7,992	182	3.7	2%
409	figs	Tree Fruit	1960	5,612	153	3.0	2%
409	figs	Tree Fruit	1975	4,033	171	2.0	1%
409	figs	Tree Fruit	1990	1,215	77	2.7	3%
409	figs	Tree Fruit	2005		77		
412	almonds	Nuts	1945	1,609	201	11.4	6%
412	almonds	Nuts	1960	734	169	31.1	18%
412	almonds	Nuts	1975	20,278	189	50.7	27%
412	almonds	Nuts	1990	46,223	197	61.1	31%
412	almonds	Nuts	2005	84,365	197	81.4	41%
413	walnuts	Nuts	1945	1,803	220	47.8	22%
413	walnuts	Nuts	1960	5,144	185	39.8	22%
413	walnuts	Nuts	1975	12,114	207	63.6	31%
413	walnuts	Nuts	1990	13,369	152	75.9	50%
413	walnuts	Nuts	2005	18,423	152	97.8	64%
414	pistachios	Nuts	1945		210		
414	pistachios	Nuts	1960	1	177	13.3	8%
414	pistachios	Nuts	1975	71	198	56.5	29%
414	pistachios	Nuts	1990	11,578	174	97.5	56%
414	pistachios	Nuts	2005	33,929	174	129.7	75%
600	field crops	Field Crops	1945	17,008	76	18.6	24%
600	field crops	Field Crops	1960	804	112	46.7	42%
600	field crops	Field Crops	1975	13,901	145	7.6	5%
600	field crops	Field Crops	1990	6,599	191	123.2	65%
600	field crops	Field Crops	2005	3,515	191	108.8	57%
601	cotton (lint and seed)	Cotton	1945	104,796	63	41.2	65%
601	cotton (lint and seed)	Cotton	1960	275,464	92	63.9	69%
601	cotton (lint and seed)	Cotton	1975	393,782	120	63.0	53%
601	cotton (lint and seed)	Cotton	1990	429,732	191	79.9	42%
601	cotton (lint and seed)	Cotton	2005	246,810	191	85.6	45%
602	safflower	Field Crops	1945	405	47	19.2	41%
602	safflower	Field Crops	1960	2,848	68	62.0	91%
602	safflower	Field Crops	1975	17,861	89	65.3	73%
602	safflower	Field Crops	1990	19,374	113	78.8	70%
602	safflower	Field Crops	2005	466	113	52.7	47%
605	sugar beets	Field Crops	1945	11,751	68	112.0	165%
605	sugar beets	Field Crops	1960	15,623	100	122.4	122%
605	sugar beets	Field Crops	1975	24,142	130	152.6	117%
605	sugar beets	Field Crops	1990	16,530	172	160.2	93%

Adminsitrative Record
Page 34583

605	sugar beets	Field Crops	2005	7,088	172	195.4	114%
606	corn (grain and silage)	Field Crops	1945	1,841	99	36.6	37%
606	corn (grain and silage)	Field Crops	1960	19,389	144	101.2	70%
606	corn (grain and silage)	Field Crops	1975	28,279	187	126.2	67%
606	corn (grain and silage)	Field Crops	1990	42,739	235	198.9	85%
606	corn (grain and silage)	Field Crops	2005	106,619	235	220.7	94%
607	sorghum	Field Crops	1945	8,377	62	42.2	68%
607	sorghum	Field Crops	1960	34,819	90	72.5	81%
607	sorghum	Field Crops	1975	32,400	117	76.7	66%
607	sorghum	Field Crops	1990	1,533	154	93.3	61%
607	sorghum	Field Crops	2005	2,527	154	100.3	65%
608	sudan	Field Crops	1945	1,256	96	35.6	37%
608	sudan	Field Crops	1960	202	141	7.8	6%
608	sudan	Field Crops	1975	59	183	9.9	5%
608	sudan	Field Crops	1990		242		
608	sudan	Field Crops	2005	3,843	242	477.8	197%
610	beans (dry)	Field Crops	1945	13,451	30	60.5	202%
610	beans (dry)	Field Crops	1960	25,182	43	71.6	167%
610	beans (dry)	Field Crops	1975	12,453	56	81.3	145%
610	beans (dry)	Field Crops	1990	18,529	100	94.2	94%
610	beans (dry)	Field Crops	2005	7,480	100	112.3	112%
612	sunflower	Field Crops	1945	90	35	24.0	69%
612	sunflower	Field Crops	1960	308	52	42.4	81%
612	sunflower	Field Crops	1975		67		
612	sunflower	Field Crops	1990		88		
612	sunflower	Field Crops	2005		88		
700	grain hay, straw	Grain and Hay	1945	19,268	77	37.3	48%
700	grain hay, straw	Grain and Hay	1960	34,495	113	51.8	46%
700	grain hay, straw	Grain and Hay	1975	40,429	147	141.0	96%
700	grain hay, straw	Grain and Hay	1990	41,931	194	124.7	64%
700	grain hay, straw	Grain and Hay	2005	116,187	194	156.4	81%
701	barley	Grain and Hay	1945	138,510	46	24.5	53%
701	barley	Grain and Hay	1960	283,991	67	49.3	74%
701	barley	Grain and Hay	1975	194,530	87	55.7	64%
701	barley	Grain and Hay	1990	45,737	62	63.1	102%
701	barley	Grain and Hay	2005	9,580	62	50.1	81%
702	wheat	Grain and Hay	1945	50,443	60	30.7	51%
702	wheat	Grain and Hay	1960	34,349	88	49.3	56%
702	wheat	Grain and Hay	1975	69,014	114	105.1	92%
702	wheat	Grain and Hay	1990	73,431	194	133.7	69%
702	wheat	Grain and Hay	2005	97,462	194	119.5	62%

Adminsitrative Record
Page 34584

703	oats	Grain and Hay	1945	2,430	42	24.5	58%
703	oats	Grain and Hay	1960	958	61	37.1	61%
703	oats	Grain and Hay	1975	486	79	138.2	175%
703	oats	Grain and Hay	1990	164	68	22.5	33%
703	oats	Grain and Hay	2005	239	68	28.7	42%
1600	pasture	Alfalfa and Pasture	1945				
1600	pasture	Alfalfa and Pasture	1960	122		11.3	
1600	pasture	Alfalfa and Pasture	1975				
1600	pasture	Alfalfa and Pasture	1990				
1600	pasture	Alfalfa and Pasture	2005				
1601	alfalfa	Alfalfa and Pasture	1945	118,260	12	288.6	2405 %
1601	alfalfa	Alfalfa and Pasture	1960	210,043	17	345.1	2030 %
1601	alfalfa	Alfalfa and Pasture	1975	165,247	22	360.5	1639 %
1601	alfalfa	Alfalfa and Pasture	1990	155,788	12	384.1	3201 %
1601	alfalfa	Alfalfa and Pasture	2005	169,373	12	436.0	3633 %
1602	pasture	Alfalfa and Pasture	1945				
1602	pasture	Alfalfa and Pasture	1960				
1602	pasture	Alfalfa and Pasture	1975				
1602	pasture	Alfalfa and Pasture	1990				
1602	pasture	Alfalfa and Pasture	2005				
1800	rice	Rice	1945	3,148	50	47.4	95%
1800	rice	Rice	1960	10,197	73	61.5	84%
1800	rice	Rice	1975	7,790	95	67.2	71%
1800	rice	Rice	1990	2,686	143	86.7	61%
1800	rice	Rice	2005	2,098	143	85.5	60%
2000	truck crops	Vegetables and Berries	1945	953	79	10.4	13%
2000	truck crops	Vegetables and Berries	1960	1,231	119	25.3	21%
2000	truck crops	Vegetables and Berries	1975	4,718	154	32.5	21%
2000	truck crops	Vegetables and Berries	1990	15,461	212	33.1	16%
2000	truck crops	Vegetables and Berries	2005	41,349	212	95.3	45%
2001	artichokes	Vegetables and Berries	1945	1,620	84	10.9	13%
2001	artichokes	Vegetables	1960	2,234	128	31.0	24%

Adminsitrative Record
Page 34585

		and Berries					
2001	artichokes	Vegetables and Berries	1975	3,929	165	45.1	27%
2001	artichokes	Vegetables and Berries	1990	3,056	193	58.7	30%
2001	artichokes	Vegetables and Berries	2005	2,504	193	63.0	33%
2002	asparagus	Vegetables and Berries	1945	509	80	21.4	27%
2002	asparagus	Vegetables and Berries	1960	582	121	16.9	14%
2002	asparagus	Vegetables and Berries	1975	1,501	156	19.2	12%
2002	asparagus	Vegetables and Berries	1990	2,305	155	29.6	19%
2002	asparagus	Vegetables and Berries	2005	2,125	155	34.5	22%
2003	beans (green)	Vegetables and Berries	1945	114	43	13.9	32%
2003	beans (green)	Vegetables and Berries	1960	1,176	66	40.8	62%
2003	beans (green)	Vegetables and Berries	1975	2,651	85	29.5	35%
2003	beans (green)	Vegetables and Berries	1990		135		
2003	beans (green)	Vegetables and Berries	2005	199	135	24.3	18%
2006	carrots	Vegetables and Berries	1945	4,030	67	19.1	29%
2006	carrots	Vegetables and Berries	1960	3,520	102	58.5	57%
2006	carrots	Vegetables and Berries	1975	6,423	132	71.7	54%
2006	carrots	Vegetables and Berries	1990	14,622	238	113.6	48%
2006	carrots	Vegetables and Berries	2005	1,354	238	58.5	25%
2007	celery	Vegetables and Berries	1945	1,205	105	49.5	47%
2007	celery	Vegetables and Berries	1960	1,956	160	86.7	54%
2007	celery	Vegetables and Berries	1975	2,394	206	85.6	42%
2007	celery	Vegetables and Berries	1990	2,806	284	98.1	35%
2007	celery	Vegetables and Berries	2005	3,992	284	108.2	38%
2008	lettuce	Vegetables and Berries	1945	22,224	89	18.3	21%
2008	lettuce	Vegetables and Berries	1960	19,589	136	38.1	28%
2008	lettuce	Vegetables and Berries	1975	31,848	175	61.6	35%
2008	lettuce	Vegetables and Berries	1990	42,196	212	80.4	38%

Adminsitrative Record
Page 34586

2008	lettuce	Vegetables and Berries	2005	74,015	212	86.1	41%
2009	melons, squash	Vegetables and Berries	1945	9,202	54	11.8	22%
2009	melons, squash	Vegetables and Berries	1960	16,137	81	21.2	26%
2009	melons, squash	Vegetables and Berries	1975	11,125	105	31.2	30%
2009	melons, squash	Vegetables and Berries	1990	19,493	162	34.8	22%
2009	melons, squash	Vegetables and Berries	2005	16,517	162	46.2	29%
2010	garlic, onions	Vegetables and Berries	1945	1,727	82	49.8	61%
2010	garlic, onions	Vegetables and Berries	1960	2,436	125	97.2	78%
2010	garlic, onions	Vegetables and Berries	1975	7,851	161	111.1	69%
2010	garlic, onions	Vegetables and Berries	1990	18,803	232	138.8	60%
2010	garlic, onions	Vegetables and Berries	2005	22,952	232	154.4	67%
2011	peas, green	Vegetables and Berries	1945	3,639	30	3.9	13%
2011	peas, green	Vegetables and Berries	1960	1,795	45	14.1	31%
2011	peas, green	Vegetables and Berries	1975	190	58	20.5	35%
2011	peas, green	Vegetables and Berries	1990	12	100	6.4	6%
2011	peas, green	Vegetables and Berries	2005	493	100	57.0	57%
2012	potatoes	Vegetables and Berries	1945	24,482	106	88.5	84%
2012	potatoes	Vegetables and Berries	1960	25,823	161	126.9	79%
2012	potatoes	Vegetables and Berries	1975	16,146	208	149.4	72%
2012	potatoes	Vegetables and Berries	1990	9,329	273	148.5	54%
2012	potatoes	Vegetables and Berries	2005	10,385	273	151.3	55%
2013	sweet potatoes	Vegetables and Berries	1945	212	61	29.3	48%
2013	sweet potatoes	Vegetables and Berries	1960	356	93	26.5	28%
2013	sweet potatoes	Vegetables and Berries	1975	251	120	50.0	42%
2013	sweet potatoes	Vegetables and Berries	1990	555	165	50.0	30%
2013	sweet potatoes	Vegetables and Berries	2005		165		
2014	spinach	Vegetables and Berries	1945	605	98	70.1	72%
2014	spinach	Vegetables and Berries	1960	1,508	150	87.1	58%

Adminsitrative Record
Page 34587

2014	spinach	Vegetables and Berries	1975	1,501	193	123.7	64%
2014	spinach	Vegetables and Berries	1990	2,359	154	109.7	71%
2014	spinach	Vegetables and Berries	2005	5,717	154	120.5	78%
2015	tomatoes, processed	Vegetables and Berries	1945	2,570	80	23.3	29%
2015	tomatoes, processed	Vegetables and Berries	1960	3,955	121	61.4	51%
2015	tomatoes, processed	Vegetables and Berries	1975	22,995	156	81.8	52%
2015	tomatoes, processed	Vegetables and Berries	1990	40,011	200	118.0	59%
2015	tomatoes, processed	Vegetables and Berries	2005	59,016	200	144.6	72%
2019	berries	Vegetables and Berries	1945	99	65	3.7	6%
2019	berries	Vegetables and Berries	1960	180	99	11.6	12%
2019	berries	Vegetables and Berries	1975	74	128	9.8	8%
2019	berries	Vegetables and Berries	1990	316	227	16.1	7%
2019	berries	Vegetables and Berries	2005	235	227	37.6	17%
2020	strawberries	Vegetables and Berries	1945	136	89	28.0	31%
2020	strawberries	Vegetables and Berries	1960	1,554	136	25.4	19%
2020	strawberries	Vegetables and Berries	1975	1,056	175	46.1	26%
2020	strawberries	Vegetables and Berries	1990	2,434	212	63.3	30%
2020	strawberries	Vegetables and Berries	2005	3,764	212	76.8	36%
2021	peppers (chili, bell)	Vegetables and Berries	1945	56	91	2.9	3%
2021	peppers (chili, bell)	Vegetables and Berries	1960	151	138	18.5	13%
2021	peppers (chili, bell)	Vegetables and Berries	1975	2,021	178	26.1	15%
2021	peppers (chili, bell)	Vegetables and Berries	1990	2,447	311	32.0	10%
2021	peppers (chili, bell)	Vegetables and Berries	2005	2,159	311	63.9	21%
2022	broccoli	Vegetables and Berries	1945	1,776	102	16.2	16%
2022	broccoli	Vegetables and Berries	1960	4,085	155	32.4	21%
2022	broccoli	Vegetables and Berries	1975	10,406	200	39.6	20%
2022	broccoli	Vegetables and Berries	1990	23,286	209	77.8	37%
2022	broccoli	Vegetables and Berries	2005	24,253	209	100.5	48%

Adminsitrative Record
Page 34588

2023	cabbage	Vegetables and Berries	1945	340	71	56.1	79%
2023	cabbage	Vegetables and Berries	1960	660	108	49.1	45%
2023	cabbage	Vegetables and Berries	1975	741	139	69.1	50%
2023	cabbage	Vegetables and Berries	1990	441	192	77.6	40%
2023	cabbage	Vegetables and Berries	2005	2,415	192	69.2	36%
2024	cauliflower	Vegetables and Berries	1945	168	97	60.3	62%
2024	cauliflower	Vegetables and Berries	1960	1,501	147	46.2	31%
2024	cauliflower	Vegetables and Berries	1975	4,232	190	43.7	23%
2024	cauliflower	Vegetables and Berries	1990	9,086	262	61.7	24%
2024	cauliflower	Vegetables and Berries	2005	6,991	262	78.4	30%
2025	brussel sprouts	Vegetables and Berries	1945	142	51	55.9	110%
2025	brussel sprouts	Vegetables and Berries	1960	60	78	74.6	96%
2025	brussel sprouts	Vegetables and Berries	1975	575	100	80.8	81%
2025	brussel sprouts	Vegetables and Berries	1990	508	138	124.0	90%
2025	brussel sprouts	Vegetables and Berries	2005		138		
2200	grapes (raisins, table, wine)	Grapes	1945	107,967	11	9.2	84%
2200	grapes (raisins, table, wine)	Grapes	1960	99,743	17	13.7	80%
2200	grapes (raisins, table, wine)	Grapes	1975	131,150	22	14.4	65%
2200	grapes (raisins, table, wine)	Grapes	1990	152,613	37	14.6	39%
2200	grapes (raisins, table, wine)	Grapes	2005	155,385	37	17.0	46%

Adminsitrative Record
Page 34589

Appendix Table 8. Summary wastewater treatment and food processing facility data as reported or modeled. (Source: California Water Boards, WDRs, SMRs, and Hilmar SEP Database.)

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
W-1	Fresno	FRESNO REGIONAL WWTF	65.20	23.20	2,089,970.82	9.78	1,485.20	313,495.62	55.42	708.20	1,776,475.20	
W-2	Fresno	SELMA-KINGSBURG-FOWLER CSD WWTF	2.90	13.00	52,089.00	-	-	-	2.90	42.49	52,089.00	
W-3	Fresno	REEDLEY WWTF	2.40	7.35	24,372.68	-	-	-	2.40	13.76	24,372.68	
W-4	Fresno	CLOVIS WWTF	2.30	6.30	20,020.41	2.30	-	20,020.41	-	-	-	I
W-5	Fresno	SANGER WWTF	1.67	28.02	64,653.08	-	-	-	1.67	64.75	64,653.08	
W-6	Fresno	KERMAN WWTF	1.20	37.00	61,346.19	-	-	-	1.20	5.87	61,346.19	
W-7	Fresno	Mendota WWTF	1.20	21.49	35,630.53	-	-	-	1.20	60.70	35,630.53	N
W-8	Fresno	MALAGA CWD WWTF	1.20	9.00	10,569.78	-	-	-	0.85	14.57	10,569.78	II
W-9	Fresno	PARLIER WWTF	1.10	10.62	16,140.68	-	-	-	1.10	28.33	16,140.68	
W-10	Fresno	FRESNO CO #41-SHAVER LAKE WWTF	1.00	19.74	27,274.19	0.50	65.26	13,637.09	0.50	44.84	13,637.09	N, D, A, III
W-11	Fresno	Millerton New Town WWTF and Recycling Operation	0.71	16.00	15,695.78	0.36	46.33	7,847.89	0.36	44.84	7,847.89	D, A
W-12	Fresno	SANGER INDUSTRIAL WWTF	0.25	16.30	5,630.31	0.25	76.08	5,630.31	-	-	-	

[1] "W - #" refers to Wastewater Treatment Plants, "F - #" refers to Food Processing Facilities. WWTPs representing 90% of municipal wastewater flow in each study area are included here, amounting to 40 WWTPs. Food Processors for which sufficient data were available (primarily from the Hilmar SEP database) or modeling was possible are included here, accounting for ~63% of FPs in the study area.

[2] "Total MGD" refers to the total flow leaving the facility. "Total mg N/L" refers to the effluent concentration of total nitrogen including nitrate, nitrite, ammonia and organic nitrogen. "Total kg N/yr" refers to the total mass of nitrogen discharged in liquid effluent to irrigated agriculture and percolation basins, combined.

[3] "Irrigation MGD" refers to the volume of flow land applied for irrigation. "Irrigation hectares" refers to the reported or modeled land area receiving irrigation discharges. "Irrigation kg N/yr" refers to the mass of nitrogen discharged in liquid effluent to irrigated agriculture.

[4] "Percolation MGD" refers to the volume of flow discharged to percolation basins for direct groundwater recharge. "Percolation hectares" refers to the reported or modeled land area receiving percolation discharges. "Percolation kg N/yr" refers to the mass of nitrogen discharged in liquid effluent to percolation basins.

[5] The "Notes" column indicates if modeling was used to estimate nitrogen, flow distribution and/or acreage and provides additional explanation for several specific facilities.

N: Modeled nitrogen. **D**: Modeled flow distribution. **A**: Modeled acreage.

I: Inconsistent discharge information. **II**: Remaining flow to surface water. **III**: This plant is located outside the TLB boundary to the northeast. **IV**: Small portion of flow to prison. **V**: Discharge to sewer. **VI**: Discharge to surface water only.

Adminsitrative Record
Page 34590

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
W-13	Kern	BAKERSFIELD WWTP #3	17.80	6.10	126,085.77	9.76	1,273.87	82,259.16	5.20	44.84	43,826.60	A
W-14	Kern	BAKERSFIELD WWTP #2	13.70	5.70	107,894.69	13.70	2,216.06	107,894.69	-	-	-	
W-15	Kern	NORTH OF RIVER WWTF	5.50	28.00	212,777.33	5.50	704.15	212,777.33	-	-	-	
W-16	Kern	DELANO WWTF	4.28	31.20	184,502.82	4.28	463.37	184,502.82	-	-	-	
W-17	Kern	KERN SANITATION AUTHORITY WWTF	3.90	9.89	53,292.43	3.90	445.15	53,292.43	-	-	-	
W-18	Kern	LAMONT WWTF	2.00	16.24	44,876.67	2.00	465.39	44,876.67	-	-	-	N
W-19	Kern	WASCO WWTF	1.80	26.00	64,662.20	0.90	157.83	32,331.10	0.90	64.75	32,331.10	D
W-20	Kern	TAFT WWTF	1.20	35.00	58,030.18	1.20	74.87	58,030.18	-	-	-	
W-21	Kern	ARVIN WWTF	1.10	23.60	35,868.18	1.10	2,428.11	35,868.18	-	-	-	
W-22	Kern	MCFARLAND WWTF	1.10	20.92	31,795.01	0.55	30.35	15,897.51	0.55	20.23	15,897.51	
W-23	Kings	HANFORD WWTF	4.90	10.70	72,441.01	2.45	1,618.74	36,220.51	2.45	58.27	36,220.51	D
W-24	Kings	LEMOORE WWTF	2.00	12.77	35,287.88	2.00	5,395.67	35,287.88	-	-	-	
W-25	Kings	LEMOORE NAS WWTF (naval services)	1.90	4.90	12,863.36	0.95	123.99	6,431.68	0.95	44.84	6,431.68	D, A
W-26	Kings	CORCORAN WWTF	1.30	18.12	30,919.31	-	-	-	1.24	136.78	30,919.31	N, IV
W-27	Monterey	MRWPCA REG TRTMT & OUTFALL SYS	21.00	3.51	67,895.31	14.00	4,856.23	67,895.31	-	-	-	II
W-28	Monterey	Soledad Sewage Treatment Plant	4.40	30.00	182,380.57	-	-	-	4.40	42.09	182,380.57	
W-29	Monterey	SALINAS INDUSTRIAL WWTP	2.10	0.09	261.14	-	-	-	2.10	44.52	261.14	
W-30	Monterey	KING CITY DOMESTIC WWTF	0.87	20.15	24,221.38	0.44	26.30	12,110.69	0.44	44.84	12,110.69	D, A
W-31	Monterey	GONZALES WW	0.53	5.40	3,954.34	0.27	34.59	1,977.17	0.27	44.84	1,977.17	D, A

Adminsitrative Record
Page 34591

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
W-32	Tulare	VISALIA WWTF	12.25	19.45	329,200.39	7.11	910.54	190,936.22	5.15	97.12	138,264.16	
W-33	Tulare	TULARE WWTF	12.00	10.00	165,800.52	10.80	809.37	149,220.47	1.20	121.41	16,580.05	
W-34	Tulare	PORTERVILLE WWTF	5.30	15.00	109,842.84	3.70	250.91	76,682.74	1.60	44.84	33,160.10	A
W-35	Tulare	DINUBA WWTF	2.25	16.85	52,382.60	-	-	-	2.25	40.47	52,382.60	
W-36	Tulare	LINDSAY WWTF	1.30	16.00	28,738.76	0.65	84.84	14,369.38	0.65	44.84	14,369.38	D, A
W-37	Tulare	CUTLER-OROSI WWTF	1.20	15.50	25,699.08	0.60	42.90	12,849.54	0.60	6.48	12,849.54	D
W-38	Tulare	FARMERSVILLE WWTF	0.92	20.00	25,422.75	-	-	-	0.92	14.24	25,422.75	
W-39	Tulare	WOODLAKE WWTF	0.92	16.00	20,338.20	0.46	14.16	10,169.10	0.46	3.89	10,169.10	D
W-40	Tulare	EXETER WWTF	0.90	5.18	6,441.35	-	-	-	0.90	16.19	6,441.35	
F-1	Fresno	LOS GATOS HURON PLANT	0.6789	78.49	73,619.45	0.6789	890.31	73,622.57	-	-	-	
F-2	Fresno	O'Neill Vintners Reedley Winery	0.5000	36.39	25,141.00	-	-	-	0.5000	14.89	25,141.00	N
F-3	Fresno	CONAGRA HELM TOMATO PROCESSING PLANT	0.4779	37.62	24,840.80	0.4779	969.63	24,842.43	-	-	-	
F-4	Fresno	GSV FRESNO WINERY	0.3002	38.00	15,759.82	0.3000	254.95	15,751.05	-	-	-	
F-5	Fresno	POM WONDERFUL FRUIT PROCESSING PLANT	0.2056	42.08	11,956.18	0.2056	146.09	11,953.79	-	-	-	N
F-6	Fresno	SUN-MAID KINGSBURG PLANT	0.1651	23.94	5,462.47	0.1650	18.21	5,458.12	-	-	-	
F-7	Fresno	THE WINE GROUP FRANZIA WINERY-SANGER	0.1518	65.05	13,643.94	0.1518	60.70	13,643.43	-	-	-	
F-8	Fresno	E & J GALLO WINERY FRESNO WINERY	0.1511	303.44	63,352.29	0.1511	141.64	63,348.27	-	-	-	
F-9	Fresno	DEL MONTE PLANT 25 (LAND APP)	0.0966	32.80	4,376.43	0.0966	31.57	4,377.80	-	-	-	

Adminsitrative Record
Page 34592

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
F-10	Fresno	E & J GALLO WINERY FRESNO WINERY	0.0965	62.31	8,308.93	0.0965	24.28	8,307.87	-	-	-	
F-11	Fresno	LION RAISINS SELMA PLANT	0.0888	29.26	3,589.77	0.0888	23.07	3,589.77	-	-	-	
F-12	Fresno	McCALL WINERY	0.0678	20.00	1,873.55	-	-	-	0.0678	6.68	1,873.55	
F-13	Fresno	BAKER COMMODITIES KERMAN DIVISION	0.0536	900.00	66,593.89	0.0536	202.34	66,589.63	-	-	-	
F-14	Fresno	NATIONAL RAISIN PLANT	0.0492	42.08	2,860.54	0.0492	99.15	2,860.54	-	-	-	N
F-15	Fresno	SUN-MAID ORANGE COVE PLANT	0.0313	285.64	12,334.22	0.0103	8.09	4,065.03	0.0210	7.65	8,287.92	A
F-16	Fresno	Paramont Farms El Dorado Facility	0.0267	42.08	1,552.36	0.0267	32.37	1,552.36	-	-	-	N
F-17	Fresno	FOWLER PACKING CEDAR AVENUE FACILITY	0.0231	42.08	1,342.82	-	-	-	0.0231	7.65	1,343.06	N, A
F-18	Fresno	BOGHOSIAN RAISIN PACKING PLANT	0.0218	4.26	128.49	0.0218	26.30	128.49	-	-	-	
F-19	Fresno	CHOOLJIAN BROS RAISIN DEHYDRATOR & PACKING PLANT	0.0217	10.70	320.12	0.0217	3.64	320.81	-	-	-	
F-20	Fresno	NORDMAN REEDLEY DISTILLERY	0.0189	302.71	7,888.23	0.0189	12.14	7,888.11	-	-	-	
F-21	Fresno	BALLANTINE REEDLEY PACKING FACILITY	0.0155	7.60	162.76	0.0078	15.91	81.38	0.0078	7.65	81.38	A
F-22	Fresno	VIE-DEL PLANT #2, KINGSBURG	0.0143	4.60	90.81	0.0143	14.16	90.81	-	-	-	
F-23	Fresno	FAMILY TREE REEDLEY PACKING HOUSE	0.0105	42.08	610.48	0.0105	2.83	610.48	-	-	-	N
F-24	Fresno	DEL REY PACKING	0.0091	45.11	565.93	0.0091	11.74	565.90	-	-	-	
F-25	Fresno	FIG GARDEN PACKING FACILITY	0.0086	42.08	499.12	0.0086	24.28	498.85	-	-	-	N

Adminsitrative Record
Page 34593

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
F-26	Fresno	SALWASSER SOUTH PLANT	0.0056	42.08	323.55	0.0028	3.72	161.63	0.0028	2.36	161.63	N
F-27	Fresno	SIX JEWELS DEHYDRATOR	0.0041	6.60	37.78	0.0041	4.86	37.75	-	-	-	
F-28	Fresno	LAMANUZZI & PANTALEO - FRESNO2	0.0036	42.08	210.47	-	-	-	0.0036	1.42	210.47	N
F-29	Fresno	LAMANUZZI & PANTALEO PLANT NO 1	0.0036	42.08	210.24	-	-	-	0.0036	2.02	210.24	N
F-30	Fresno	BOOTH RANCHES CITRUS PACKING FACILITY	0.0020	11.68	32.27	-	-	-	0.0020	1.46	32.27	
F-31	Fresno	SURABIAN PACKING CO, INC	0.0020	85.00	231.36	-	-	-	0.0020	7.65	231.36	A
F-32	Fresno	VITA-PAKT FRUIT PROCESSING & DEHYDRATING PLANT	0.0011	47.82	70.08	0.0011	26.30	70.10	-	-	-	
F-33	Fresno	BIANCHI VINEYARDS	0.0010	23.30	32.22	-	-	-	0.0010	1.21	32.19	
F-34	Fresno	NONINI WINERY	0.0001	36.39	3.57	-	-	-	0.0001	0.10	3.57	N
F-35	Kern	Grimmway Fresh Processing	3.6110	21.78	108,686.61	3.6110	411.57	108,685.87	-	-	-	
F-36	Kern	BOLTHOUSE BUTTONWILLOW PLANT	3.0134	14.96	62,284.73	3.0134	285.71	62,284.73	-	-	-	
F-37	Kern	J G BOSWELL TOMATO, KERN FACILITY	1.8740	21.00	54,374.28	1.8740	250.10	54,374.28	-	-	-	
F-38	Kern	FRITO-LAY CHIPS & PRETZELS MFG PLANT	1.1762	50.08	81,394.72	1.1760	77.70	81,377.66	-	-	-	
F-39	Kern	Grimmway Frozen Foods	1.0660	33.41	49,208.38	1.0660	195.87	49,209.52	-	-	-	
F-40	Kern	PARAMOUNT FARMS LOST HILLS FACILITY	0.9540	40.49	53,371.72	0.4770	503.02	26,684.49	0.4770	7.65	26,684.49	A
F-41	Kern	DELANO WINERY	0.2545	18.11	6,368.11	-	-	-	0.2545	8.09	6,368.11	
F-42	Kern	ARVIN PACKING SHED	0.1186	19.18	3,143.91	0.1186	32.37	3,142.82	-	-	-	

Adminsitrative Record
Page 34594

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
F-43	Kern	Grimmway Premier Packing	0.0978	3.80	513.58	-	-	-	-	-	-	V
F-44	Kern	SUN PACIFIC BAKERSFIELD PACKINGHOUSE	0.0715	0.66	65.01	0.0357	468.63	32.48	0.0357	7.65	32.48	A
F-45	Kern	MONARCH NUT COMPANY	0.0684	121.10	11,437.38	-	-	-	0.0684	7.65	11,438.02	A
F-46	Kern	HECK CELLARS	0.0562	52.81	4,103.82	-	-	-	0.0562	23.88	4,103.94	
F-47	Kern	Grimmway Mountain View Facility	0.0554	1.60	122.47	0.0277	29.14	61.24	0.0277	13.76	61.24	
F-48	Kern	PARAMOUNT FARMS KING FACILITY	0.0484	190.00	12,692.90	0.0484	52.61	12,705.85	-	-	-	
F-49	Kern	MCFARLAND WINERY	0.0434	111.34	6,676.21	-	-	-	0.0434	16.19	6,676.21	
F-50	Kern	EDISON WINERY	0.0284	5.13	201.49	0.0142	66.00	100.75	0.0142	7.65	100.75	D, A
F-51	Kern	SUN WORLD COMMODITY CENTER FACILITY	0.0011	42.08	61.29	0.0011	15.91	61.29	-	-	-	N
F-52	Kings	CORCORAN TOMATO PROCESSING FACILITY	1.4000	28.00	54,161.50	1.4000	161.87	54,161.50	-	-	-	
F-53	Kings	DEL MONTE FOODS PLANT #24	1.0697	41.81	61,796.26	1.0700	389.31	61,815.10	-	-	-	
F-54	Kings	OTP LEMOORE PLANT	0.5362	62.00	45,928.83	0.5361	364.22	45,925.97	-	-	-	
F-55	Kings	KEENAN FARMS PISTACHIO PLANT	0.1105	2.10	320.60	-	-	-	0.1105	7.65	320.62	A
F-56	Kings	NICHOLS PISTACHIO	0.1061	1.71	250.94	0.1060	327.80	250.81	-	-	-	
F-57	Kings	BAKER COMMODITIES HANFORD FACILITY	0.0230	140.00	4,448.98	0.0230	50.18	4,448.98	-	-	-	
F-58	Kings	CALIFORNIA PISTACHIO ORCHARDS PLANT	0.0018	107.00	269.07	0.0018	14.57	269.07	-	-	-	
F-59	Monterey	DOLE FRESH VEGETABLES, INC.	0.6000	13.33	11,046.46	0.6000	241.90	11,046.46	-	-	-	

Adminsitrative Record
Page 34595

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
F-60	Monterey	SENSIENT DEHYDRATED FLAVORS	0.1176	9.73	1,580.75	-	-	-	0.1176	7.65	1,580.75	A
F-61	Monterey	SPRECKELS SUGAR DIVISION	0.0662	20.23	1,851.48	-	-	-	0.0662	7.65	1,851.48	A
F-62	Monterey	UNI-KOOL ABBOTT ST	0.0388	6.90	369.42	-	-	-	-	-	-	VI
F-63	Monterey	ESTANCIA WINERY	0.0010	36.39	50.28	0.0005	15.91	25.14	0.0005	7.65	25.14	N, D, A
F-64	Tulare	SUNKIST GROWERS TIPTON PLANT	0.5463	48.33	36,475.90	0.5463	100.36	36,477.93	-	-	-	
F-65	Tulare	SWORLCO LAND APPLICATION SITE	0.3307	42.83	19,573.75	0.3307	87.41	19,571.56	-	-	-	
F-66	Tulare	Mozzarella Fresca Tipton Cheese Processing Plant	0.2500	22.00	7,599.19	0.2500	116.55	7,599.19	-	-	-	
F-67	Tulare	SETTON PISTACHIO PROCESSING PLANT NO 2	0.1370	57.50	10,884.11	0.1370	91.05	10,884.11	-	-	-	
F-68	Tulare	Setton Properties Terra Bella Facility	0.0909	57.50	7,223.28	0.0909	91.05	7,221.65	-	-	-	
F-69	Tulare	PORTERVILLE CITRUS PACKING HOUSE	0.0800	42.08	4,651.28	0.0080	25.90	465.13	0.0720	0.21	4,186.15	N
F-70	Tulare	THE WINE GROUP FRANZIA WINERY-TULARE	0.0760	29.70	3,118.71	0.0760	5.16	3,118.71	-	-	-	
F-71	Tulare	VENTURA COASTAL VISALIA DIVISION	0.0546	49.08	3,702.56	-	-	-	0.0546	24.28	3,702.56	
F-72	Tulare	SUN PACIFIC EXETER PACKINGHOUSE	0.0300	42.08	1,744.23	-	-	-	0.0300	7.65	1,744.23	N, A
F-73	Tulare	TREEHOUSE EARLIMART ALMOND PLANT	0.0250	57.00	1,968.88	-	-	-	0.0250	0.34	1,968.88	

Adminsitrative Record
Page 34596

ID [1]	County	Facility Name	Total [2]			To Irrigation [3]			To Percolation [4]			Notes [5]
			MGD	mg N/L	kg N/yr	MGD	Hectares	kg N/yr	MGD	Hectares	kg N/yr	
F-74	Tulare	LOBUE/EARLIBEST	0.0211	8.52	248.95	0.0106	66.00	124.46	0.0106	7.65	124.46	A
F-75	Tulare	PACKING HOUSE, ORANGE COVE	0.0144	42.08	835.14	-	-	-	0.0144	7.65	834.90	N, A
F-76	Tulare	GSV CUTLER WINERY	0.0111	8.51	130.38	-	-	-	0.0111	20.80	130.49	
F-77	Tulare	SEQUOIA ORANGE CO ACKINGHOUSE	0.0074	42.08	430.24	-	-	-	0.0074	0.14	430.24	N
F-78	Tulare	PORTERVILLE CITRUS PACKINGHOUSE	0.0060	42.08	348.85	-	-	-	0.0060	0.37	348.85	N
F-79	Tulare	SUN PACIFIC WOODLAKE PACKINGHOUSE	0.0056	42.08	325.59	-	-	-	0.0056	7.65	325.59	N, A
F-80	Tulare	CACCIATORE FINE WINES & OLIVE	0.0050	18.85	130.32	-	-	-	0.0050	22.26	130.22	
F-81	Tulare	EUCLID PACKING CITRUS PACKINGHOUSE	0.0040	7.88	43.53	0.0040	15.91	43.53	-	-	-	
F-82	Tulare	DINUBA PACKING PLANT	0.0033	8.00	36.25	-	-	-	0.0033	0.12	36.26	
F-83	Tulare	GOLDEN STATE CITRUS PACKING SHED	0.0022	42.08	127.39	-	-	-	0.0022	0.13	127.33	N

[1] "W - #" refers to Wastewater Treatment Plants, "F - #" refers to Food Processing Facilities. WWTPs representing 90% of municipal wastewater flow in each study area are included here, amounting to 40 WWTPs. Food Processors for which sufficient data were available (primarily from the Hilmar SEP database) or modeling was possible are included here, accounting for ~63% of FPs in the study area.
[2] "Total MGD" refers to the total flow leaving the facility. "Total mg N/L" refers to the effluent concentration of total nitrogen including nitrate, nitrite, ammonia and organic nitrogen. "Total kg N/yr" refers to the total mass of nitrogen discharged in liquid effluent to irrigated agriculture and percolation basins, combined.
[3] "Irrigation MGD" refers to the volume of flow land applied for irrigation. "Irrigation hectares" refers to the reported or modeled land area receiving irrigation discharges. "Irrigation kg N/yr" refers to the mass of nitrogen discharged in liquid effluent to irrigated agriculture.
[4] "Percolation MGD" refers to the volume of flow discharged to percolation basins for direct groundwater recharge. "Percolation hectares" refers to the reported or modeled land area receiving percolation discharges. "Percolation kg N/yr" refers to the mass of nitrogen discharged in liquid effluent to percolation basins.
[5] The "Notes" column indicates if modeling was used to estimate nitrogen, flow distribution and/or acreage and provides additional explanation for several specific facilities.
N: Modeled nitrogen. **D**: Modeled flow distribution. **A**: Modeled acreage.
I: Inconsistent discharge information. **II**: Remaining flow to surface water. **III**: This plant is located outside the TLB boundary to the northeast. **IV**: Small portion of flow to prison. **V**: Discharge to sewer. **VI**: Discharge to surface water only.

Adminsitrative Record
Page 34597

Appendix Table 9. Well completion records from the Department of Water Resources South Central Region.

Kern County

Year	Domestic	Agricultural	Municipal
1977	165	175	2
1978	221	139	3
1979	147	29	0
1980	111	58	0
1981	62	47	8
1982	55	16	4
1983	150	16	5
1984	109	13	1
1985	111	22	0
1986	129	4	3
1987	143	13	6
1988	158	22	7
1989	195	64	15
1990	169	48	12
1991	143	91	15
1992	136	23	5
1993	52	87	1
1994	112	59	15
1995	65	36	4
1996	37	26	5
1997	60	13	6
1998	49	6	2
1999	32	22	3
2000	70	29	4
2001	142	54	13
2002	85	46	7
2003	56	24	10
2004	185	46	13
2005	166	33	12
2006	158	31	17
2007	102	36	13
2008	90	64	3
2009	36	80	8
TOTAL	**3,701**	**1,472**	**222**
YRLY AVG	**112**	**45**	**7**

Monterey County

Year	Domestic	Agricultural	Municipal
1977	222	106	9
1978	169	49	7
1979	189	50	4
1980	163	30	1
1981	125	38	5
1982	83	45	3
1983	85	31	1
1984	83	33	0
1985	127	28	7
1986	103	29	3
1987	153	21	6
1988	168	37	3
1989	141	43	5
1990	172	44	10
1991	173	68	6
1992	132	77	3
1993	80	59	6
1994	67	44	4
1995	66	44	2
1996	57	46	3
1997	33	38	0
1998	43	41	2
1999	76	57	4
2000	97	37	12
2001	118	46	3
2002	103	31	3
2003	142	42	6
2004	138	47	6
2005	152	36	0
2006	113	27	5
2007	81	31	5
2008	59	37	6
2009	31	50	7
TOTAL	**3,744**	**1,442**	**147**
YRLY AVG	**113**	**44**	**4**

Tulare County

Year	Domestic	Agricultural	Municipal
1977	406	526	7
1978	263	340	5
1979	229	162	4
1980	167	100	3
1981	180	140	6
1982	70	55	4
1983	131	35	5
1984	191	72	5
1985	155	70	10
1986	194	46	8
1987	137	68	8
1988	215	96	10
1989	245	124	7
1990	281	168	21
1991	376	346	9
1992	367	295	13
1993	207	159	5
1994	191	145	8
1995	131	113	20
1996	100	75	32
1997	79	57	12
1998	66	58	10
1999	92	73	20
2000	84	75	24
2001	97	73	17
2002	89	96	15
2003	139	80	17
2004	124	107	30
2005	124	93	32
2006	154	45	23
2007	176	185	17
2008	129	203	19
2009	133	182	13
TOTAL	**5,722**	**4,462**	**439**
YRLY AVG	**173**	**135**	**13**

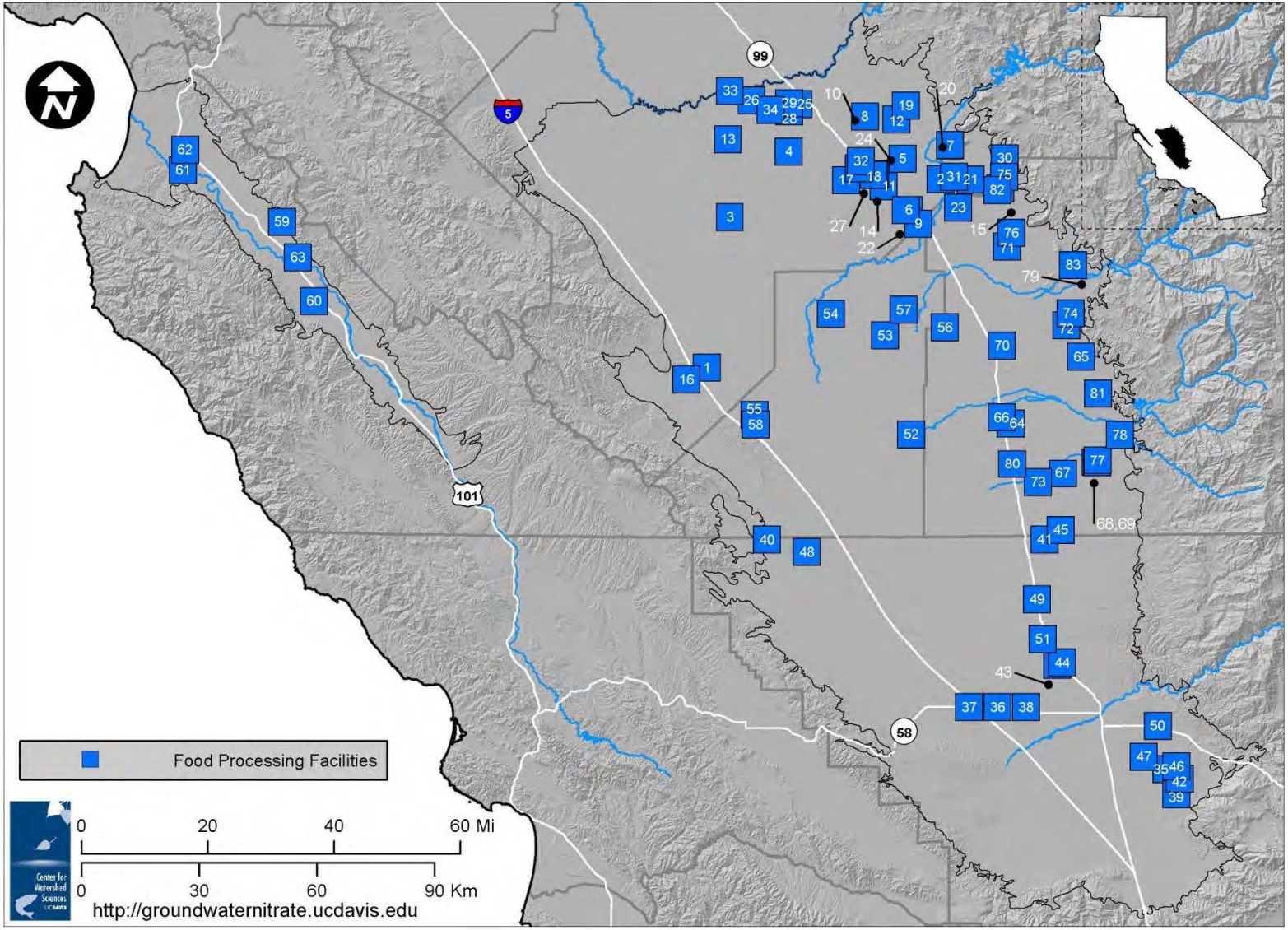
Fresno County

Year	Domestic	Agricultural	Municipal
1977	1,259	975	17
1978	993	609	22
1979	936	234	11
1980	736	188	15
1981	473	119	13
1982	324	96	8
1983	418	45	6
1984	408	55	7
1985	391	57	14
1986	419	34	7
1987	377	70	9
1988	470	99	12
1989	469	151	16
1990	611	200	29
1991	737	313	32
1992	769	223	56
1993	427	129	28
1994	451	143	28
1995	369	97	46
1996	327	71	13
1997	251	57	19
1998	265	54	11
1999	260	50	19
2000	268	84	31
2001	338	59	37
2002	416	87	21
2003	458	48	30
2004	580	80	55
2005	432	54	50
2006	430	51	40
2007	298	118	16
2008	173	150	16
2009	179	155	6
TOTAL	**15,712**	**4,955**	**740**
YRLY AVG	**476**	**150**	**22**

Kings County

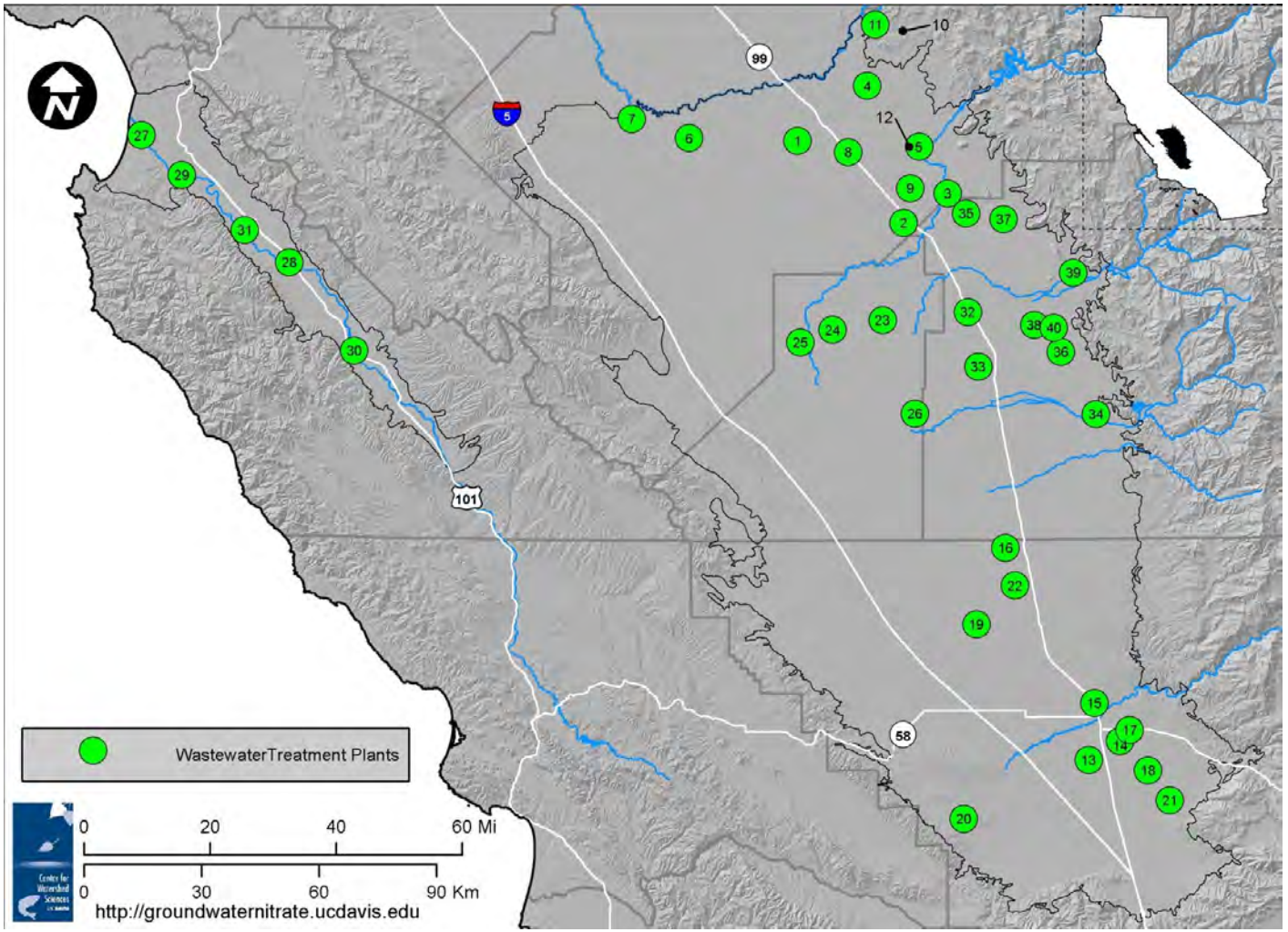
Year	Domestic	Agricultural	Municipal
1977	58	163	1
1978	61	103	3
1979	41	28	1
1980	31	40	0
1981	38	35	1
1982	18	18	1
1983	29	5	0
1984	32	13	2
1985	34	14	4
1986	21	10	0
1987	24	22	0
1988	26	34	0
1989	44	35	1
1990	37	79	5
1991	39	92	3
1992	100	94	0
1993	49	35	2
1994	55	61	4
1995	40	26	0
1996	34	18	4
1997	27	8	1
1998	34	11	2
1999	41	20	3
2000	36	36	2
2001	55	38	5
2002	45	63	5
2003	81	63	2
2004	41	56	8
2005	74	38	12
2006	70	32	3
2007	68	42	2
2008	56	89	2
2009	62	79	5
TOTAL	**1,501**	**1,500**	**84**
YRLY AVG	**45**	**45**	**3**

Adminsitrative Record
Page 34598



Appendix Figure 1. Food processor locations corresponding with facility numbering in Appendix Table 8. (Source: California Water Boards, Geolocating by Address, WDRs.)

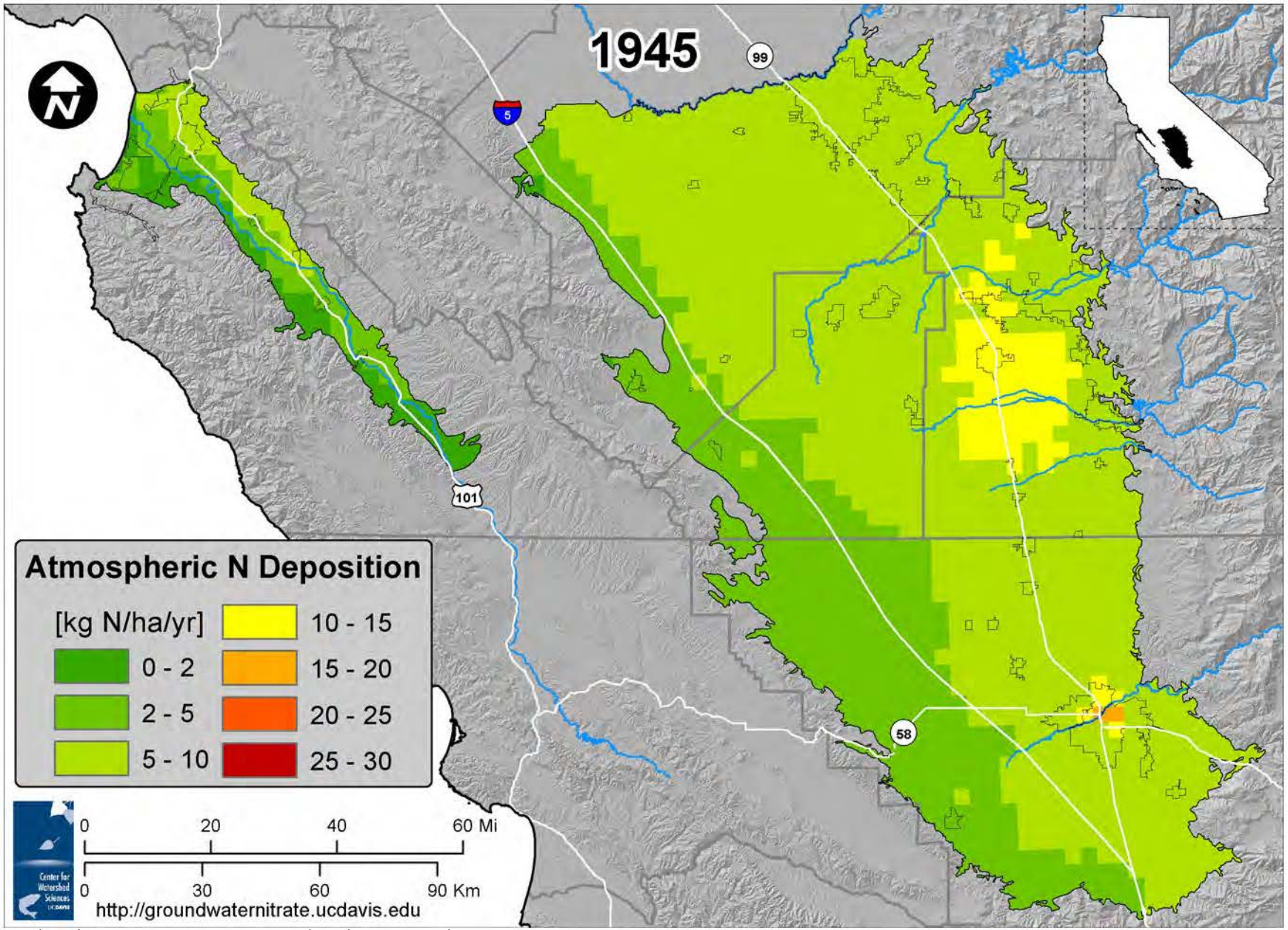
Adminsitrative Record
Page 34599



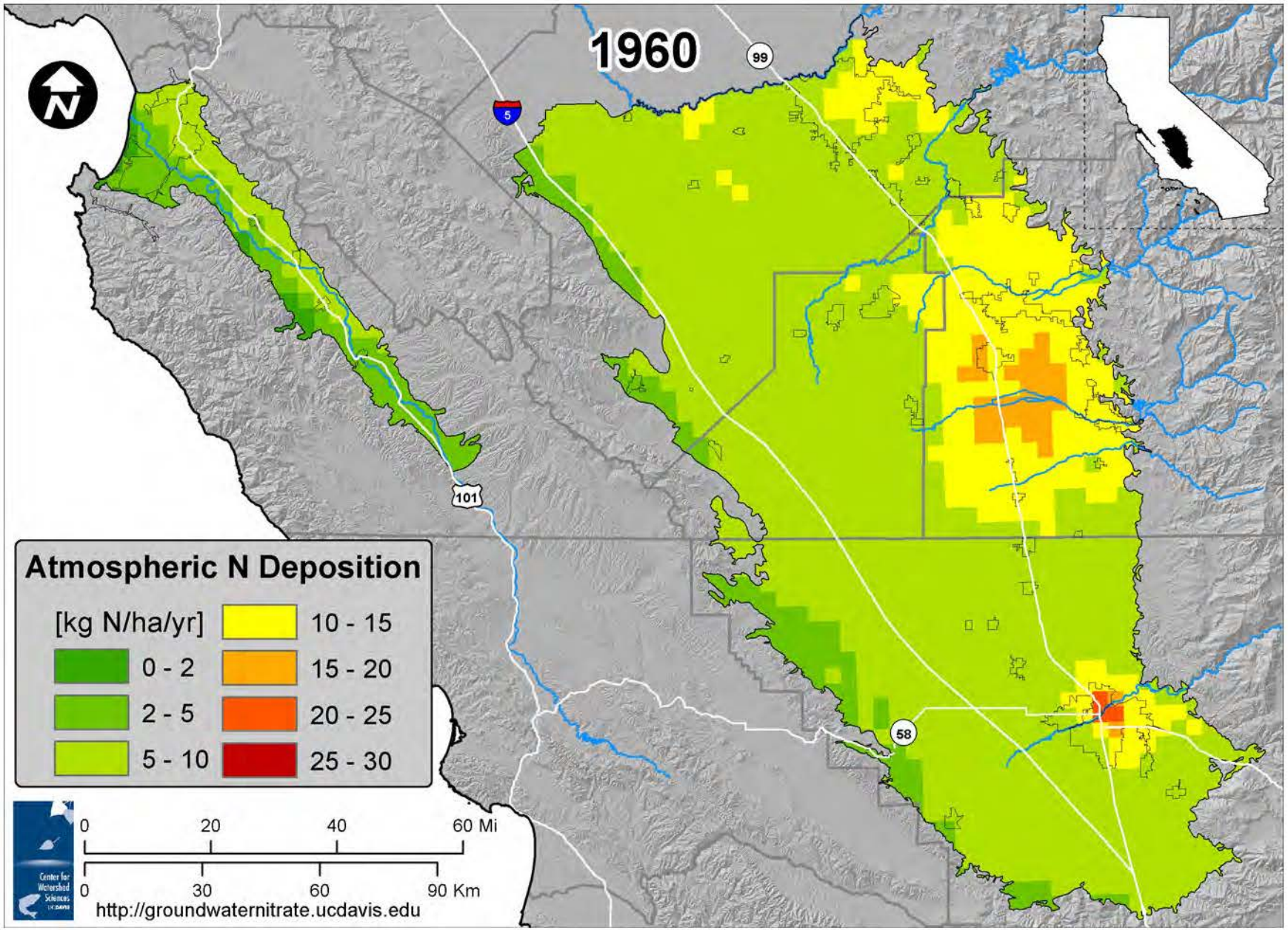
Appendix Figure 2. Wastewater treatment plant locations corresponding with facility numbering in Appendix Table 8. (Source: California Water Boards, Geolocating by Address, WDRs.)

NOTE: ADDITIONAL APPENDIX FIGURES 3 TO 120 ARE AVAILABLE IN A SEPARATE PDF FILE AT http://groundwaternitrate.ucdavis.edu

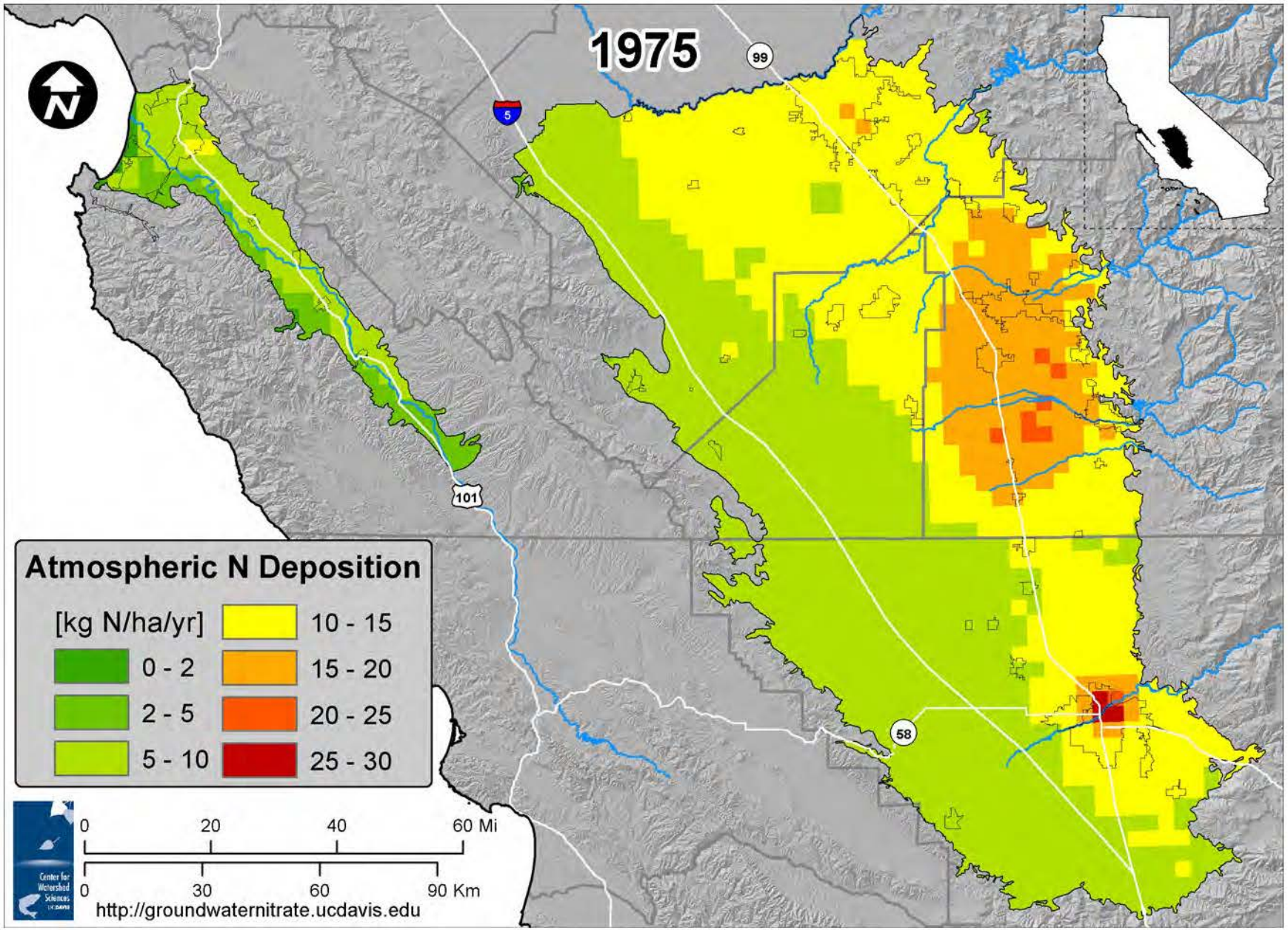
Appendix Figure 3. Estimated annual atmospheric nitrogen deposition in 1945.



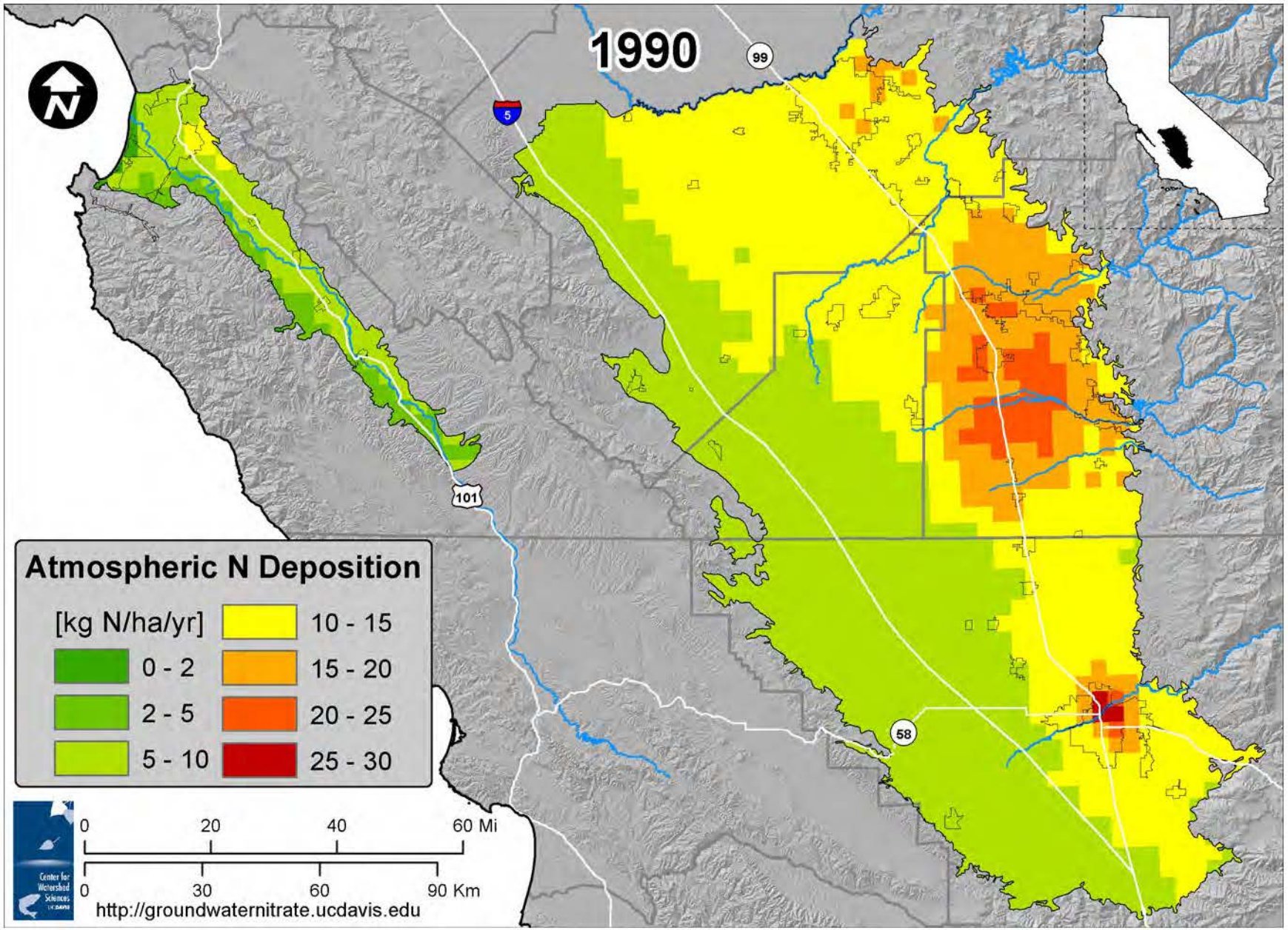
Appendix Figure 4. Estimated annual atmospheric nitrogen deposition in 1960.



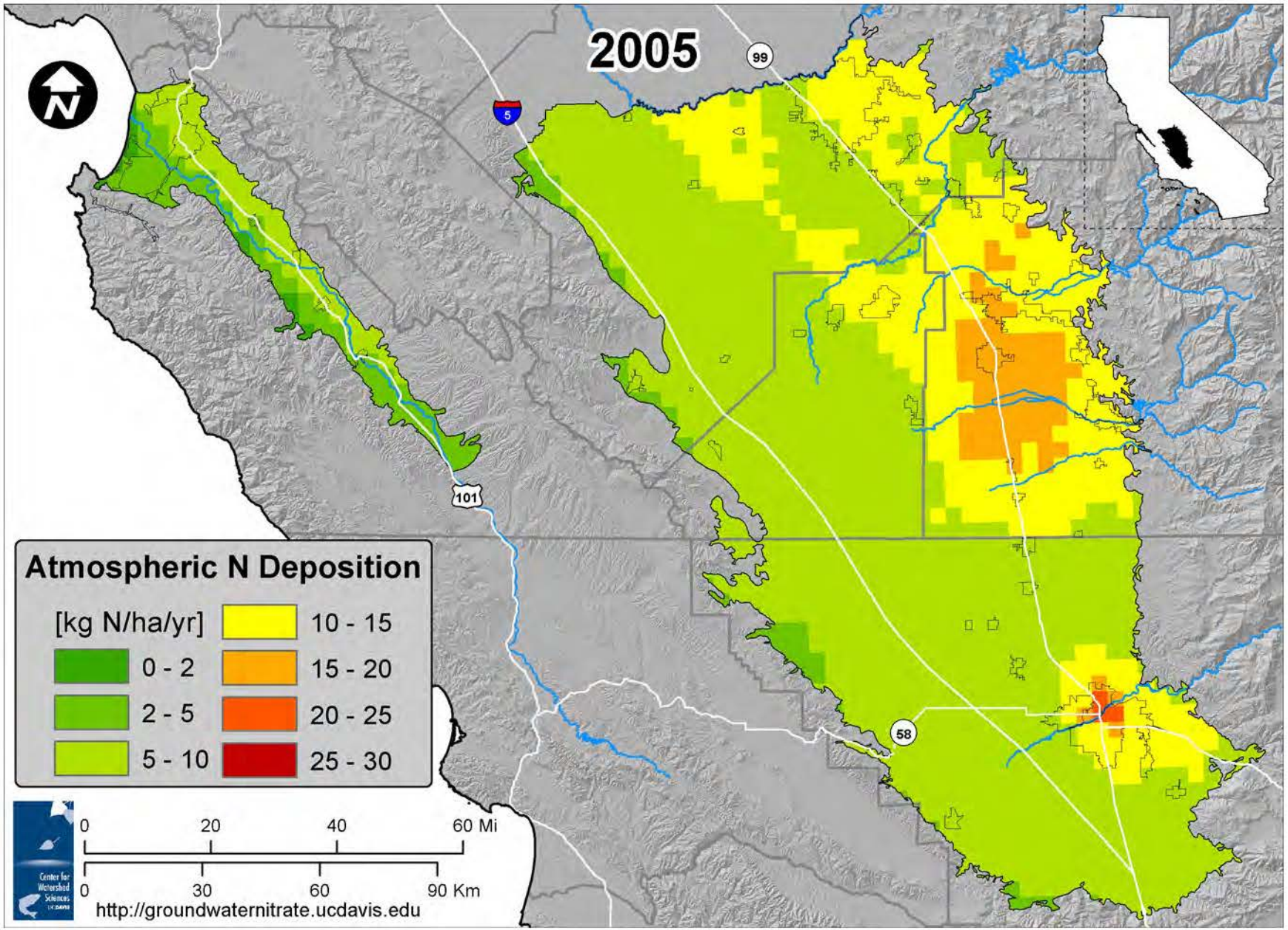
Appendix Figure 5. Estimated annual atmospheric nitrogen deposition in 1975.



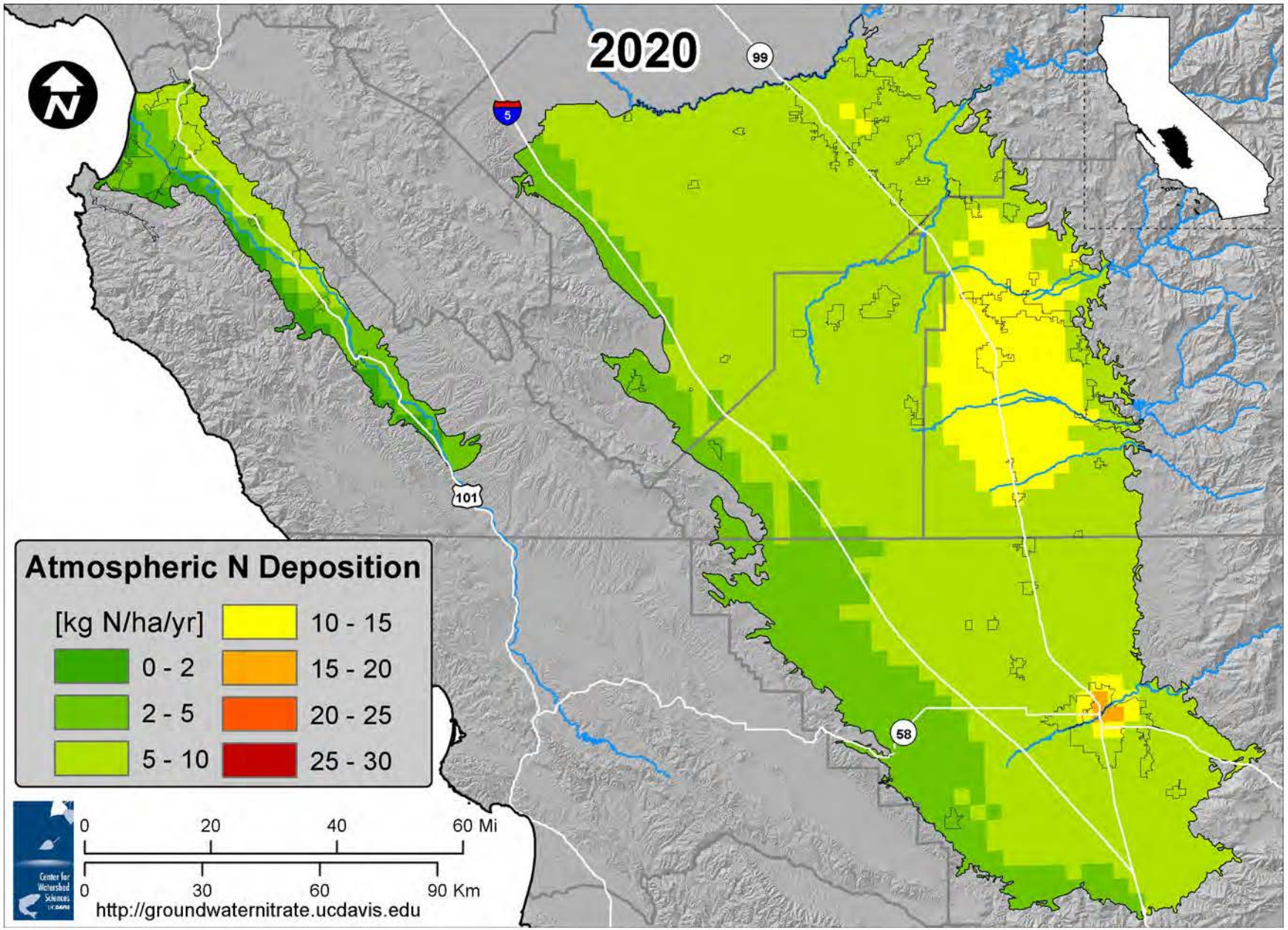
Appendix Figure 6. Estimated annual atmospheric nitrogen deposition in 1990.



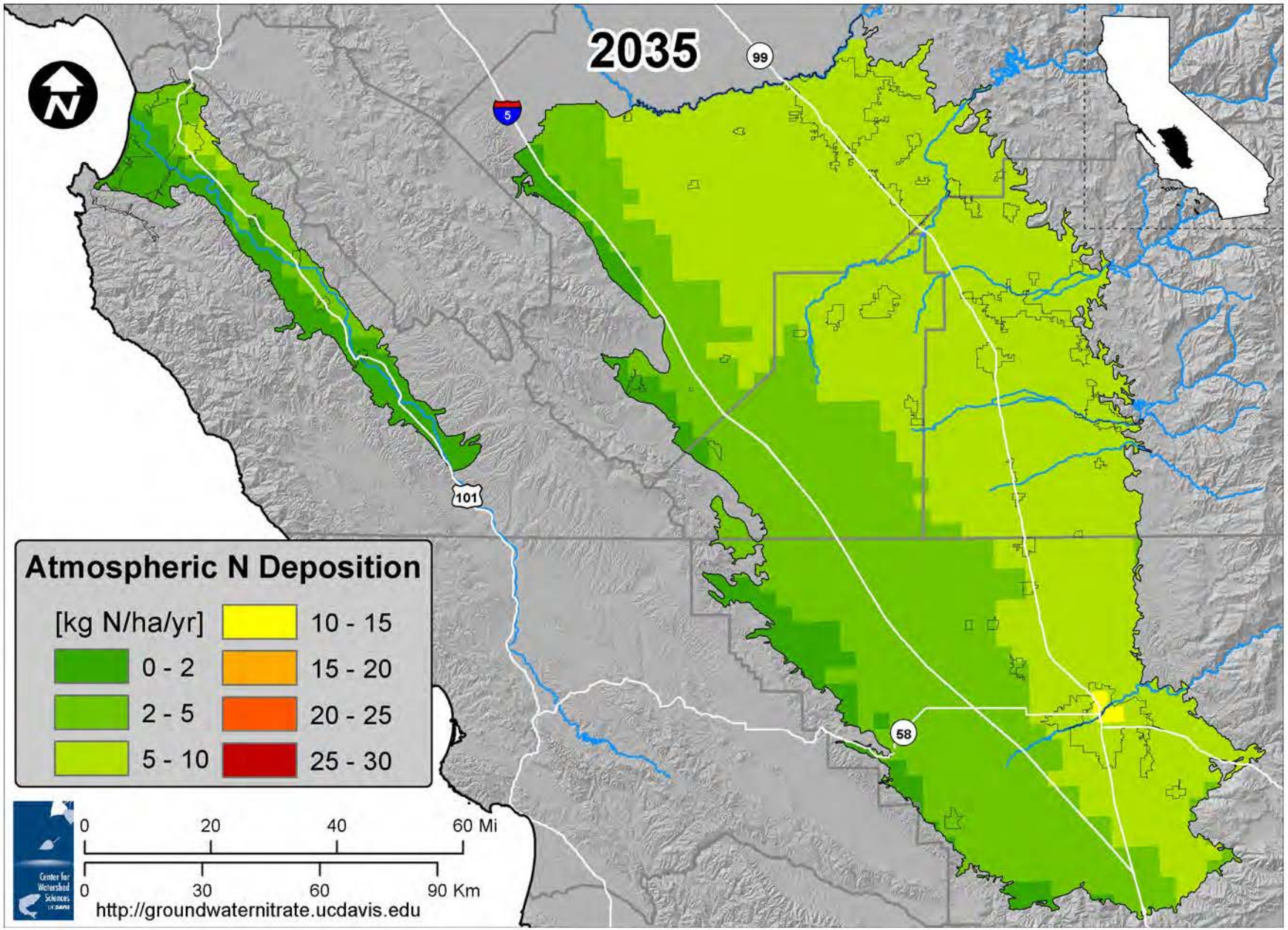
Appendix Figure 7. Estimated annual atmospheric nitrogen deposition in 2005.



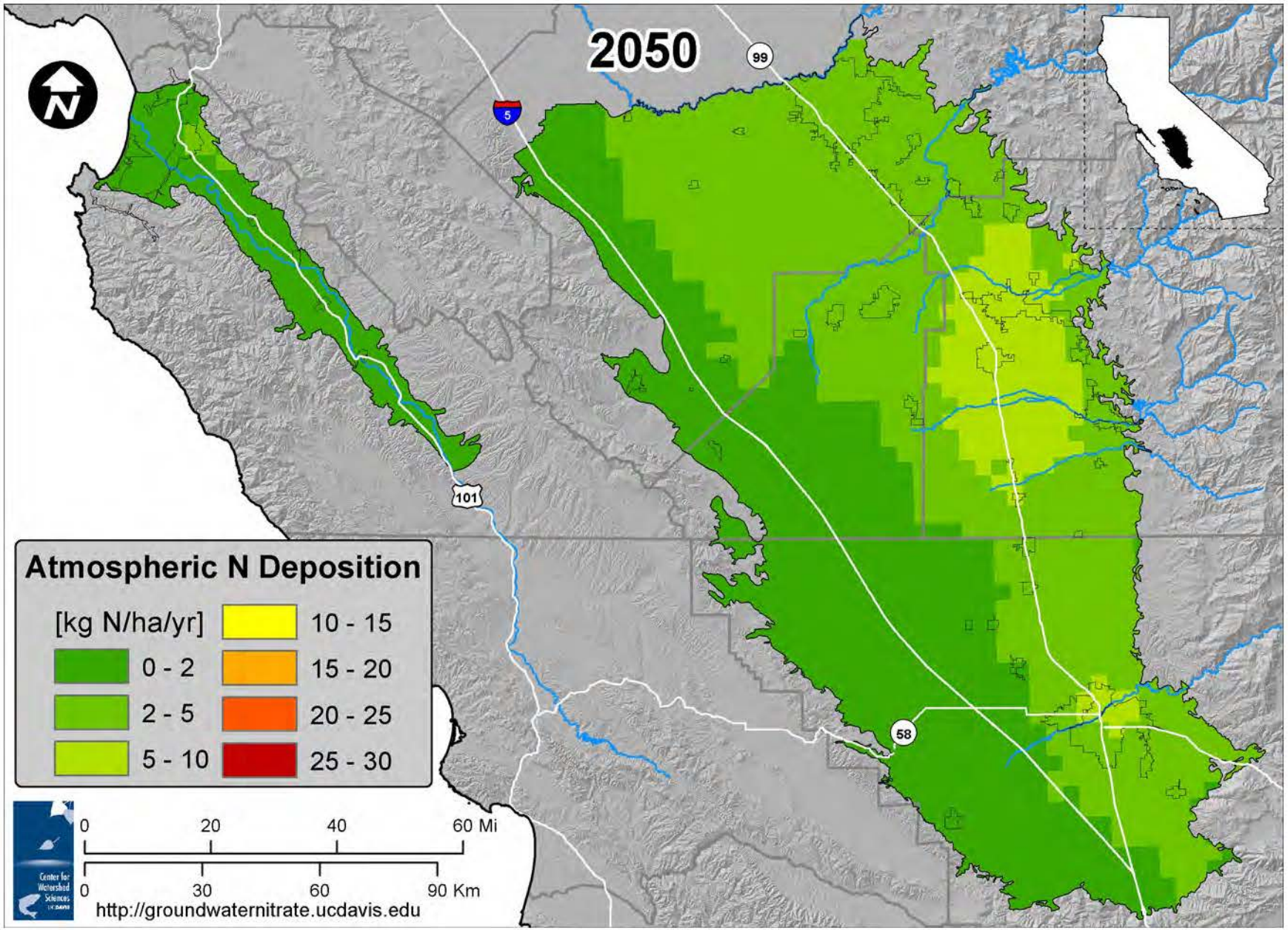
Appendix Figure 8. Estimated annual atmospheric nitrogen deposition in 2020.



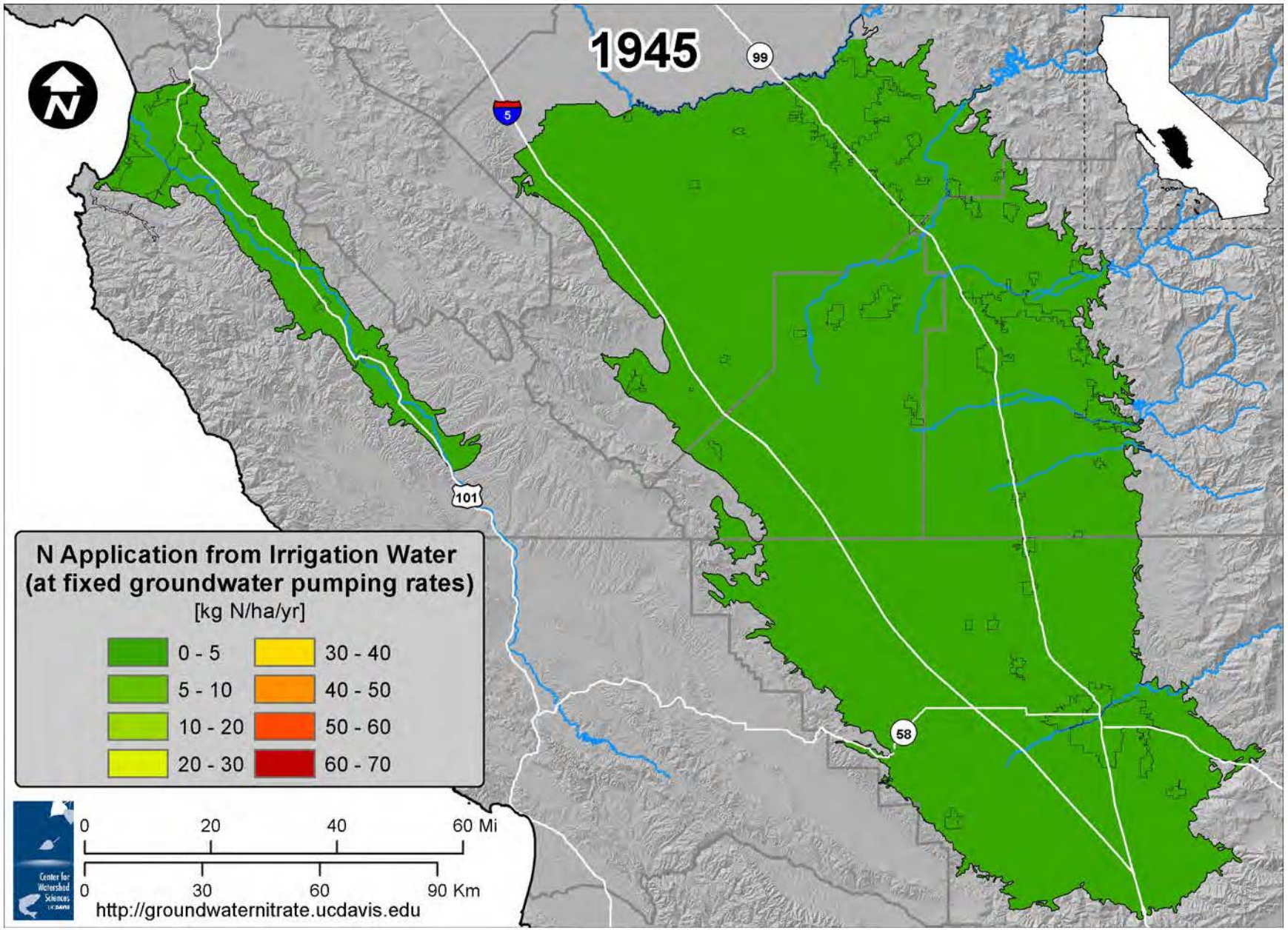
Appendix Figure 9. Estimated annual atmospheric nitrogen deposition in 2035.



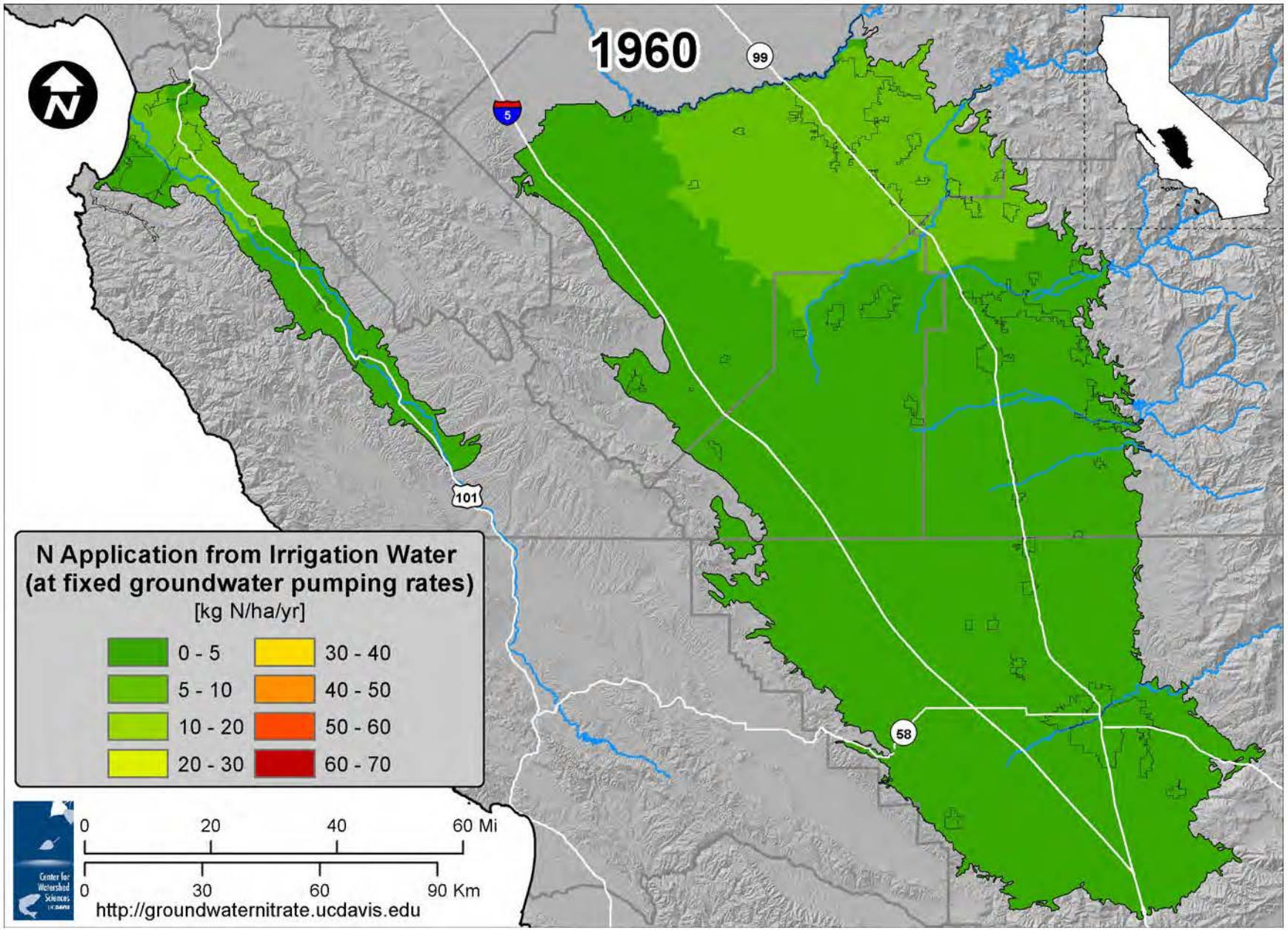
Appendix Figure 10. Estimated annual atmospheric nitrogen deposition in 2050.



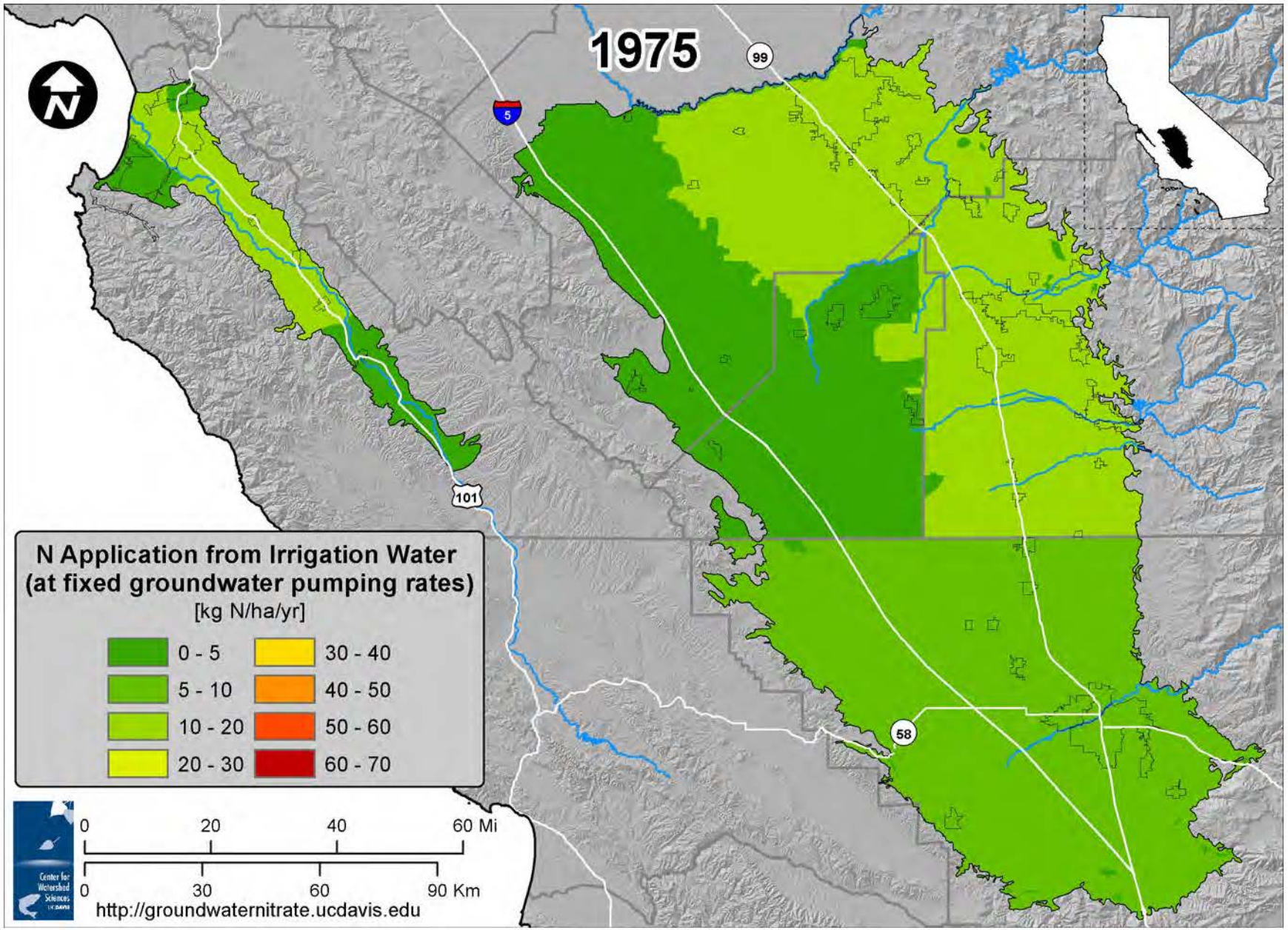
Appendix Figure 11. Estimated annual nitrogen in irrigation water applied to land in 1945.



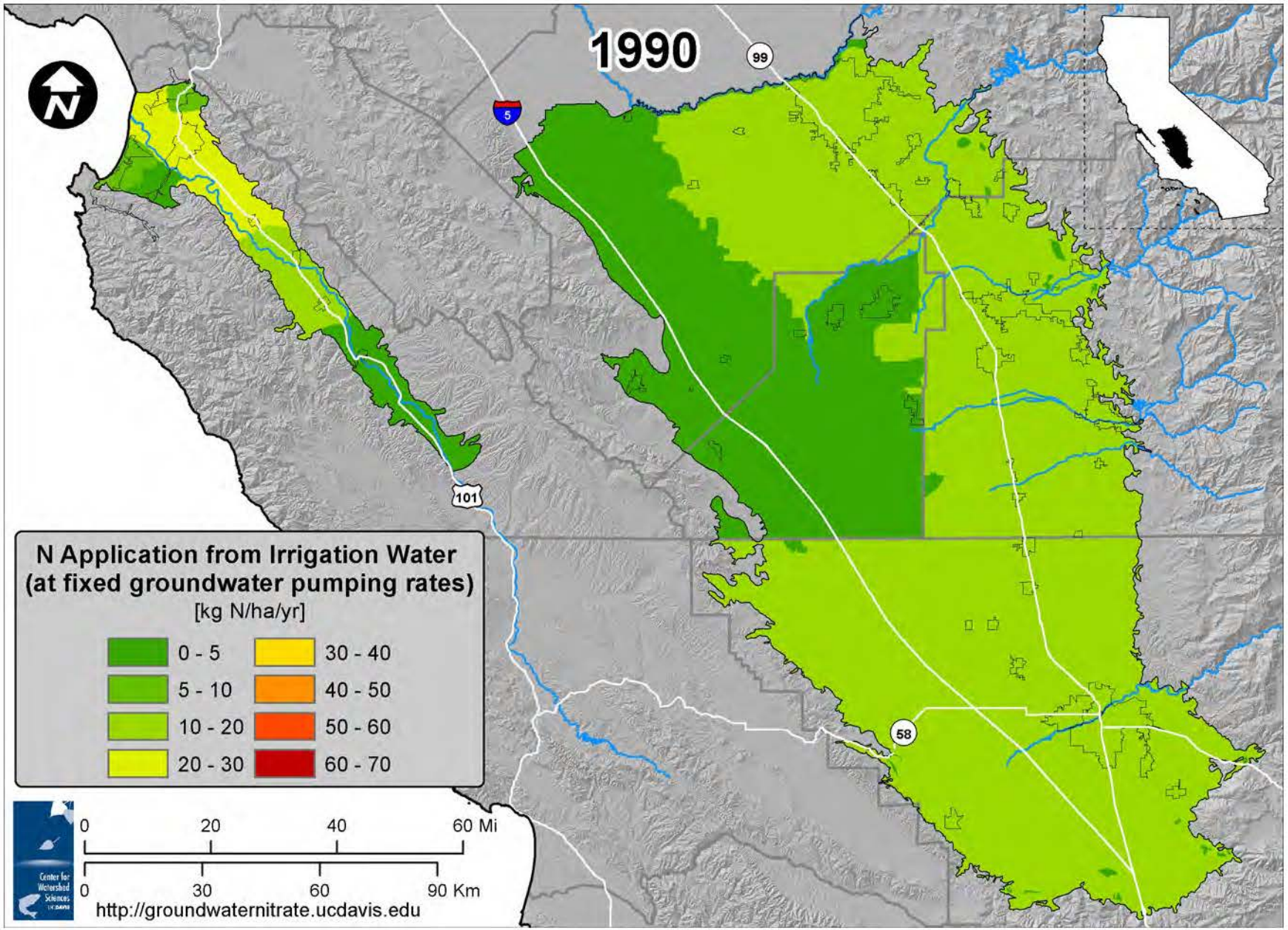
Appendix Figure 12. Estimated annual nitrogen in irrigation water applied to land in 1960.



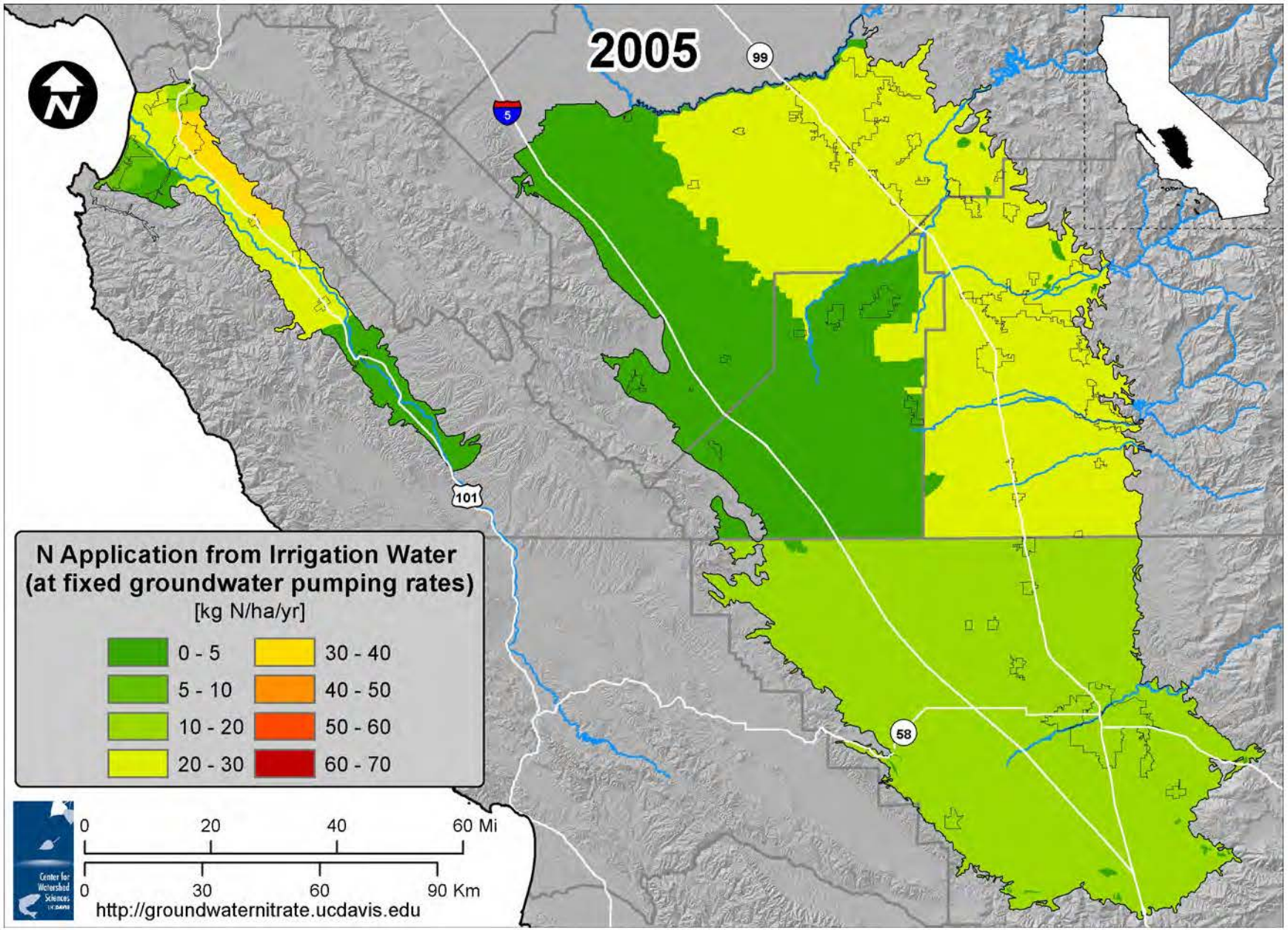
Appendix Figure 13. Estimated annual nitrogen in irrigation water applied to land in 1975.



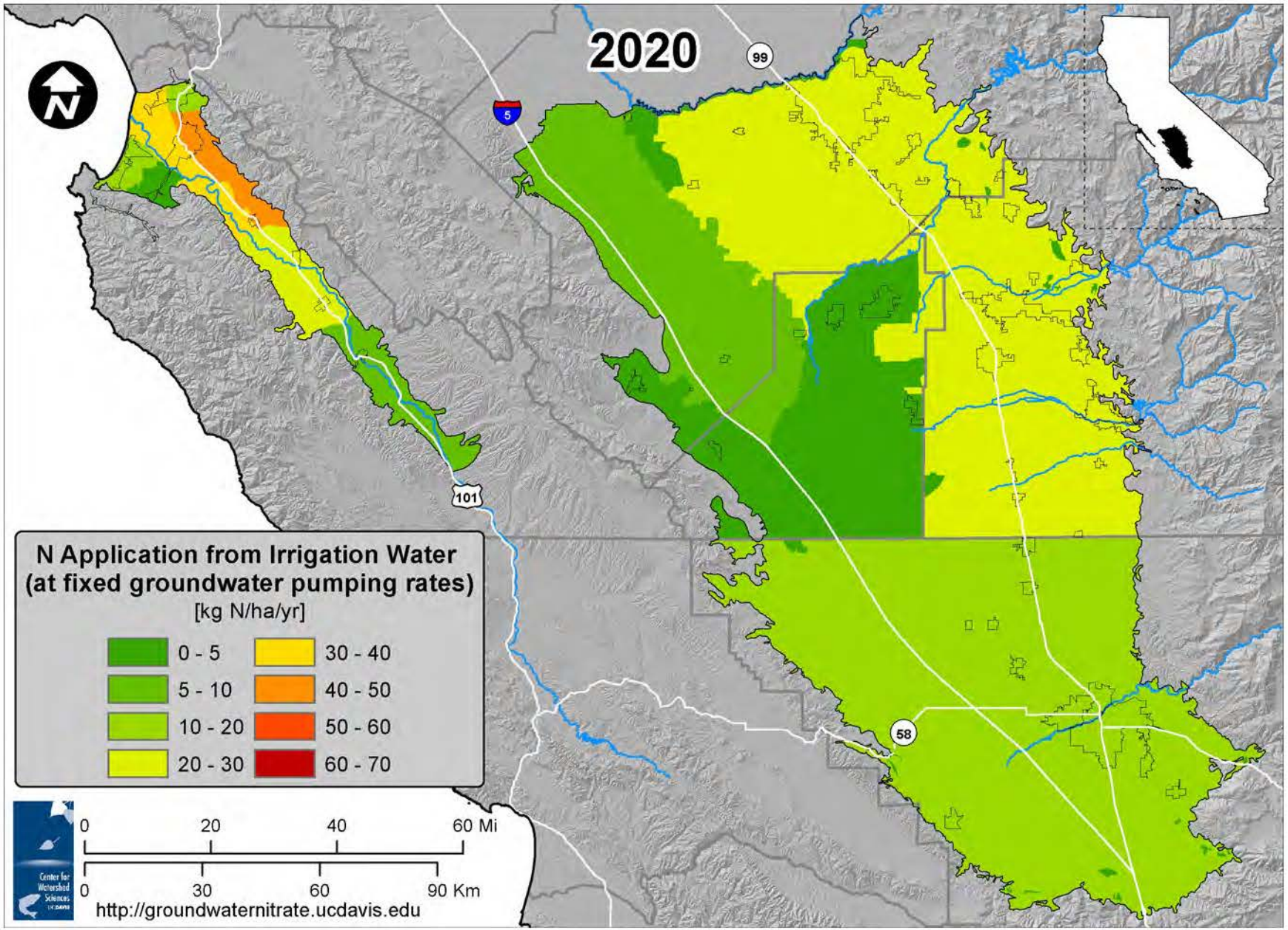
Appendix Figure 14. Estimated annual nitrogen in irrigation water applied to land in 1990.



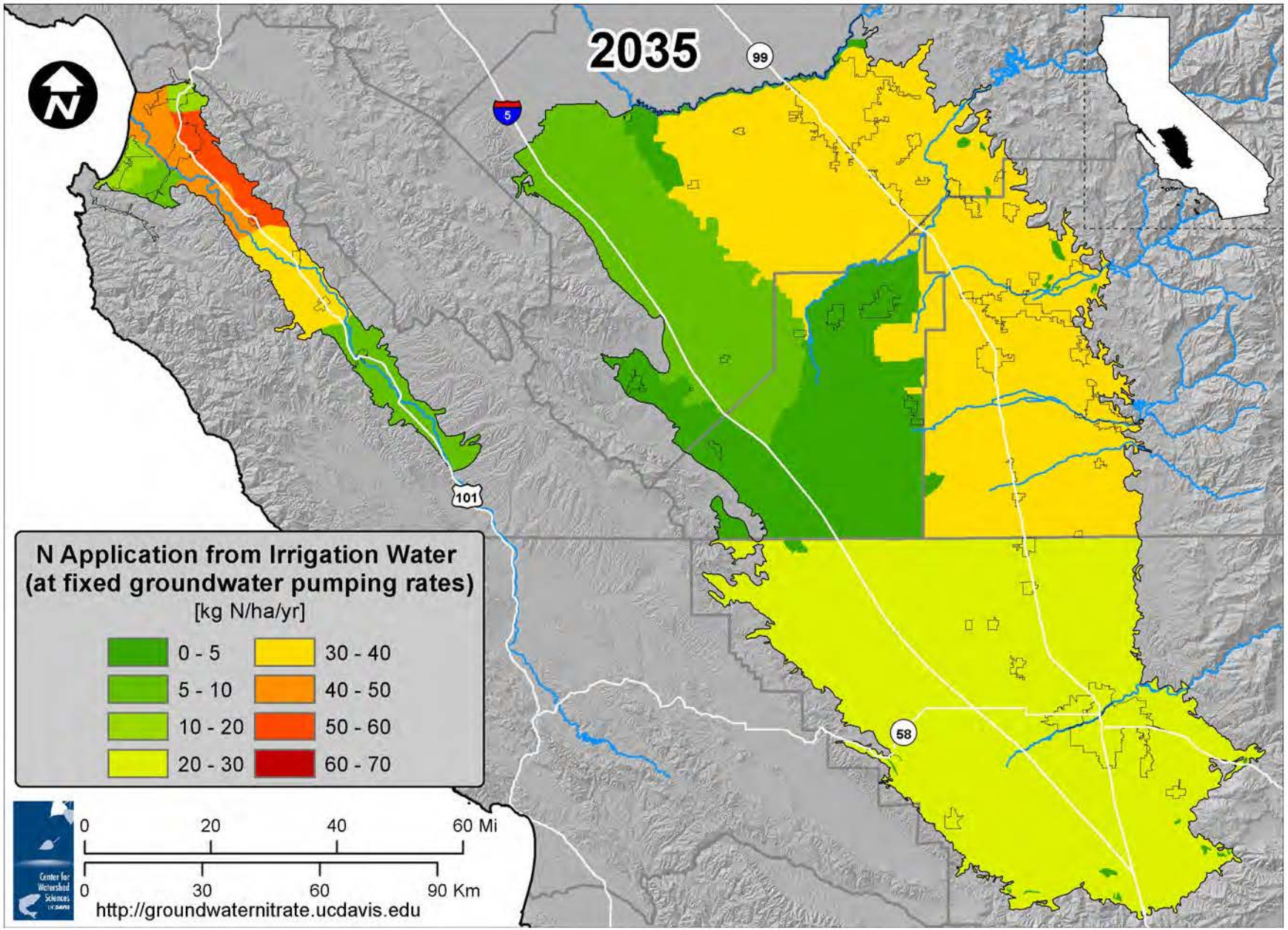
Appendix Figure 15. Estimated annual nitrogen in irrigation water applied to land in 2005.



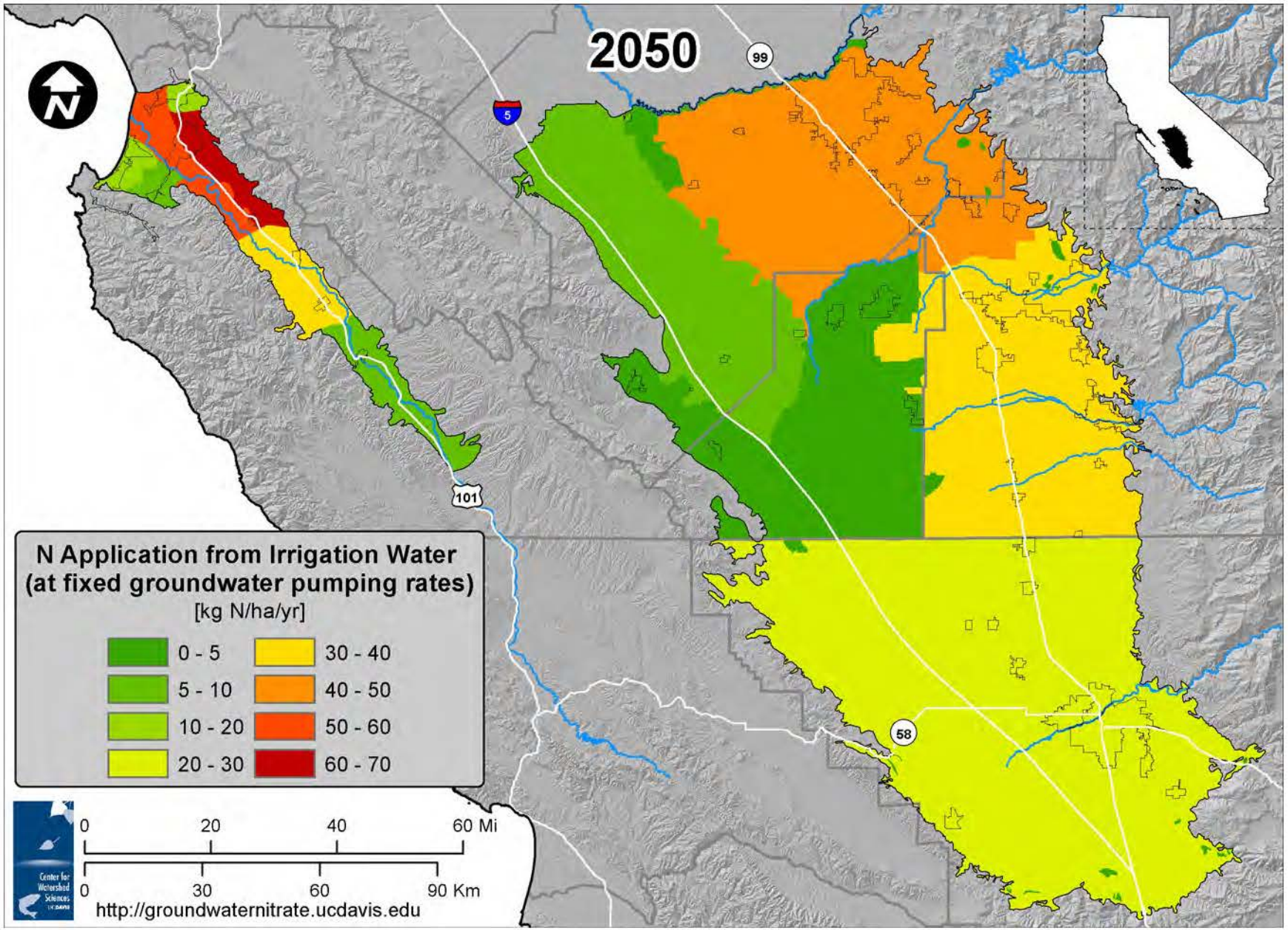
Appendix Figure 16. Estimated annual nitrogen in irrigation water applied to land in 2020.



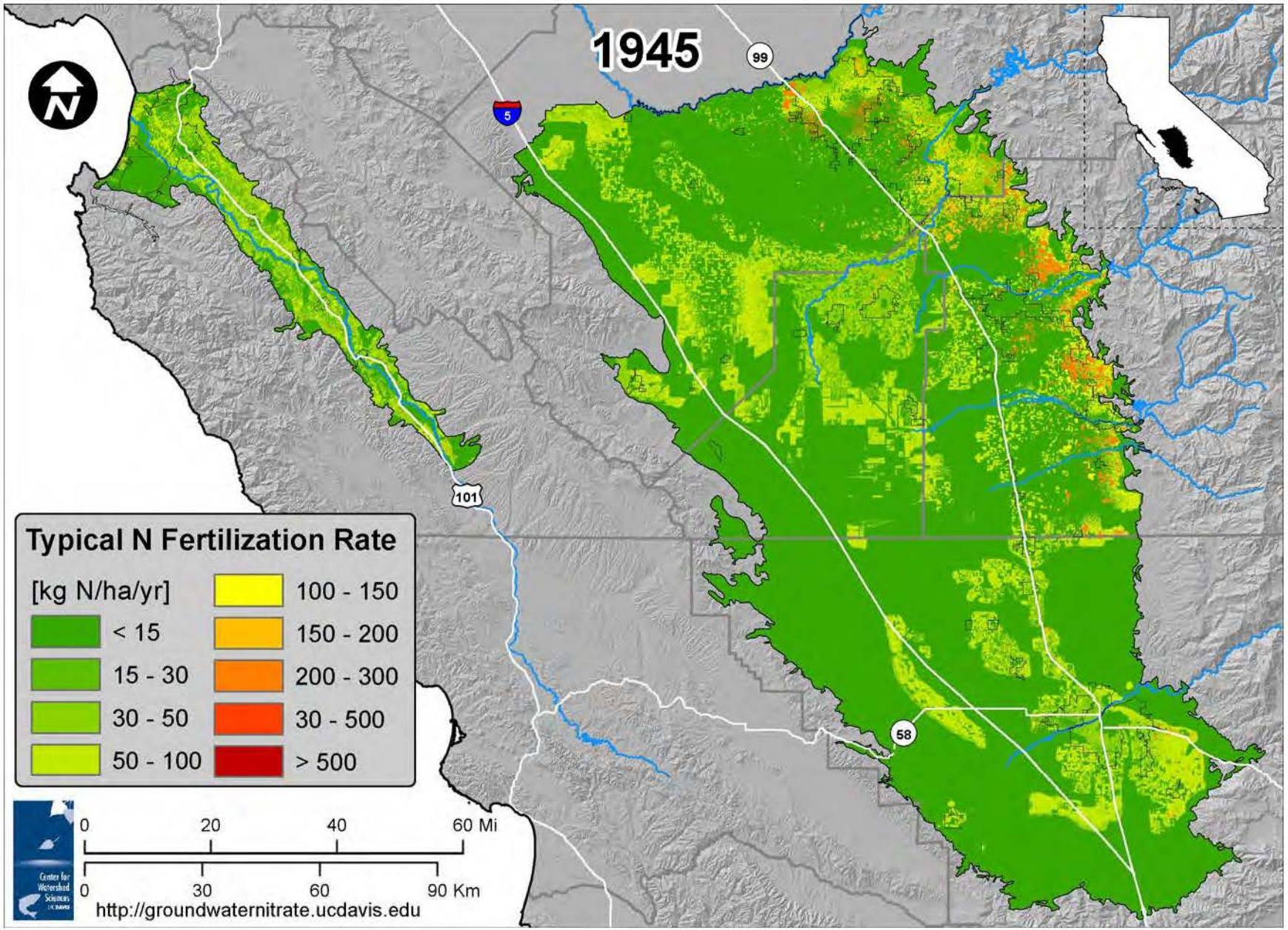
Appendix Figure 17. Estimated annual nitrogen in irrigation water applied to land in 2035.



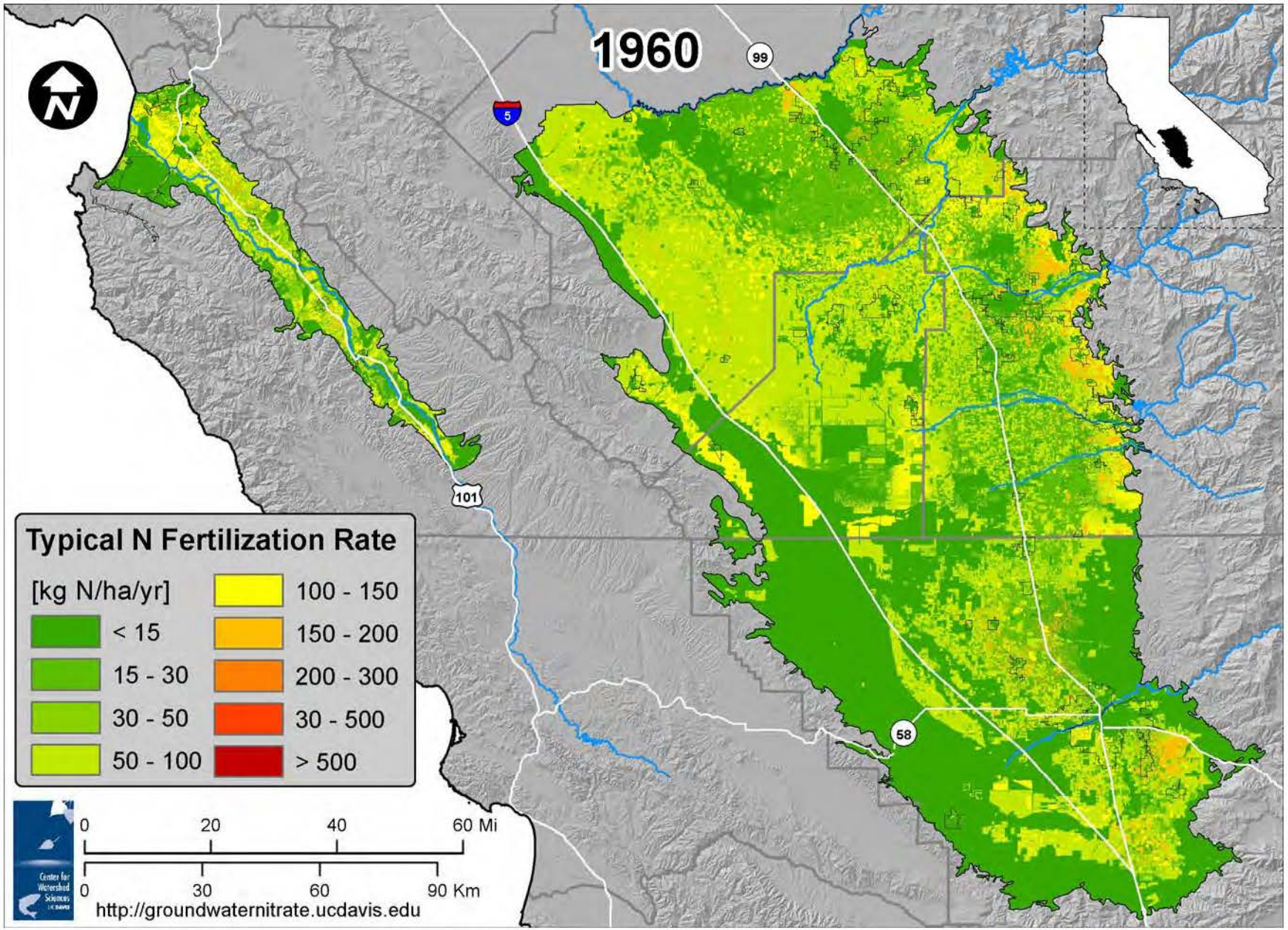
Appendix Figure 18. Estimated annual nitrogen in irrigation water applied to land in 2050.



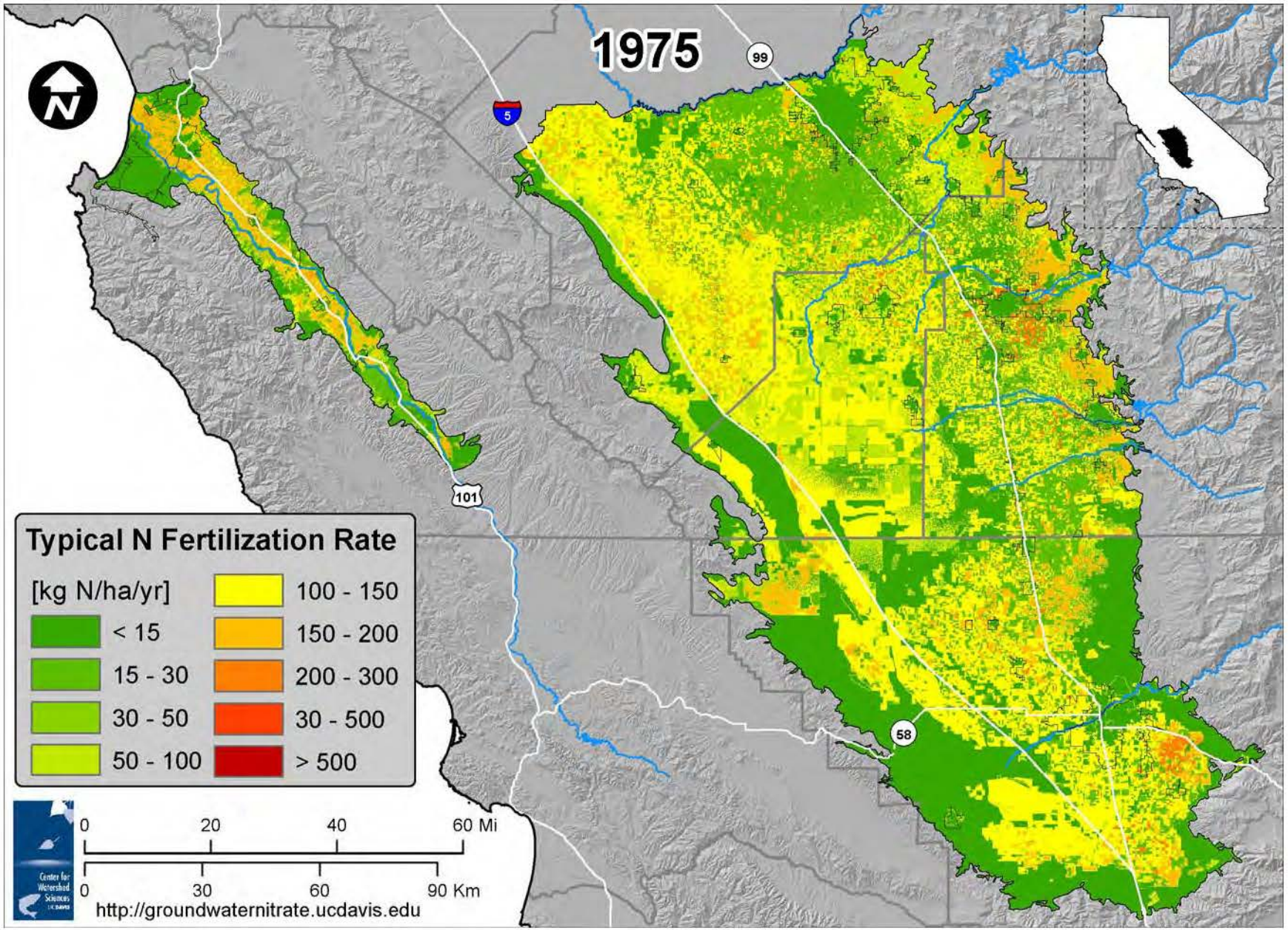
Appendix Figure 19. Typical nitrogen application rate in 1945.



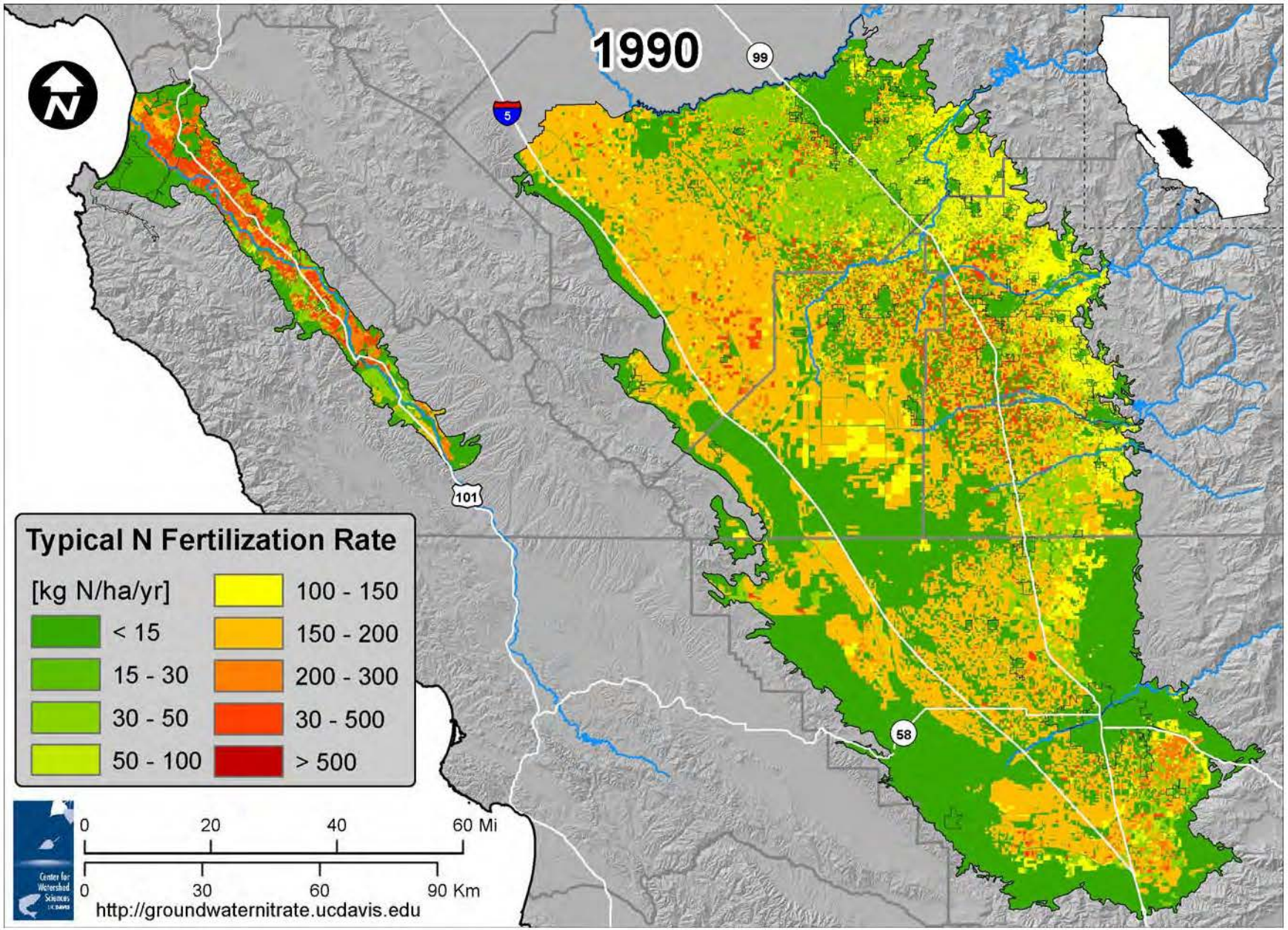
Appendix Figure 20. Typical nitrogen application rate in 1960.



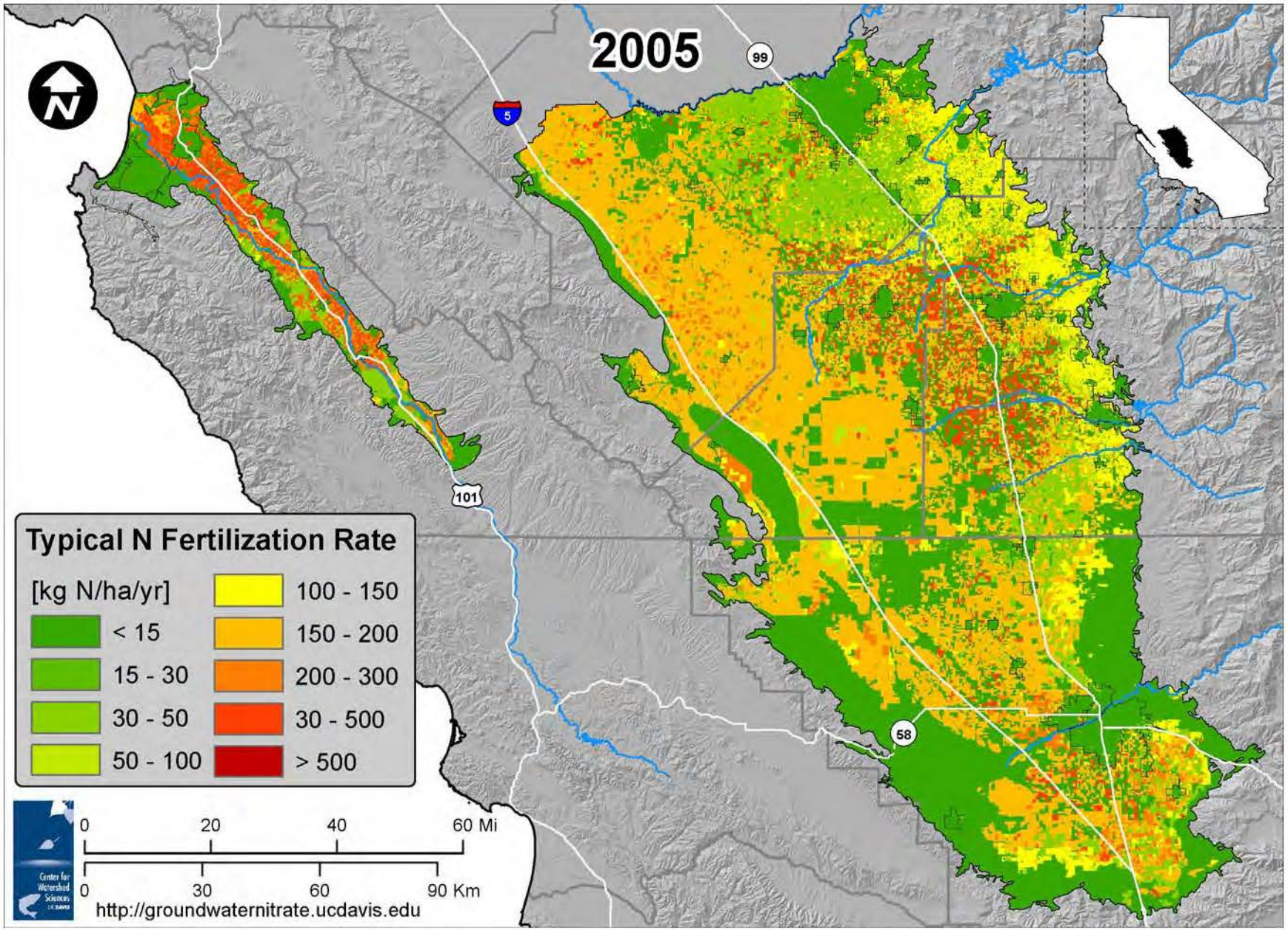
Appendix Figure 21. Typical nitrogen application rate in 1975.



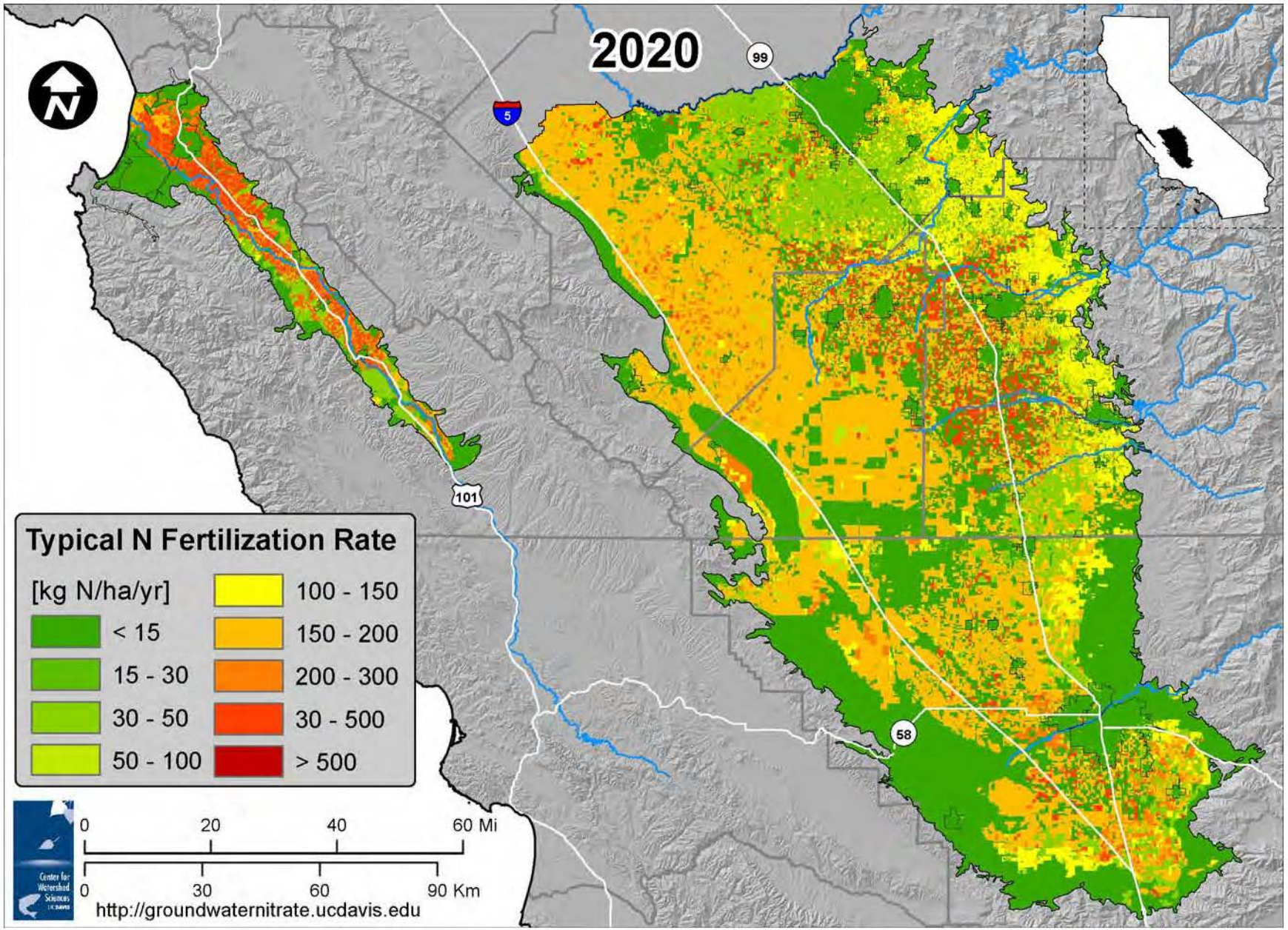
Appendix Figure 22. Typical nitrogen application rate in 1990.



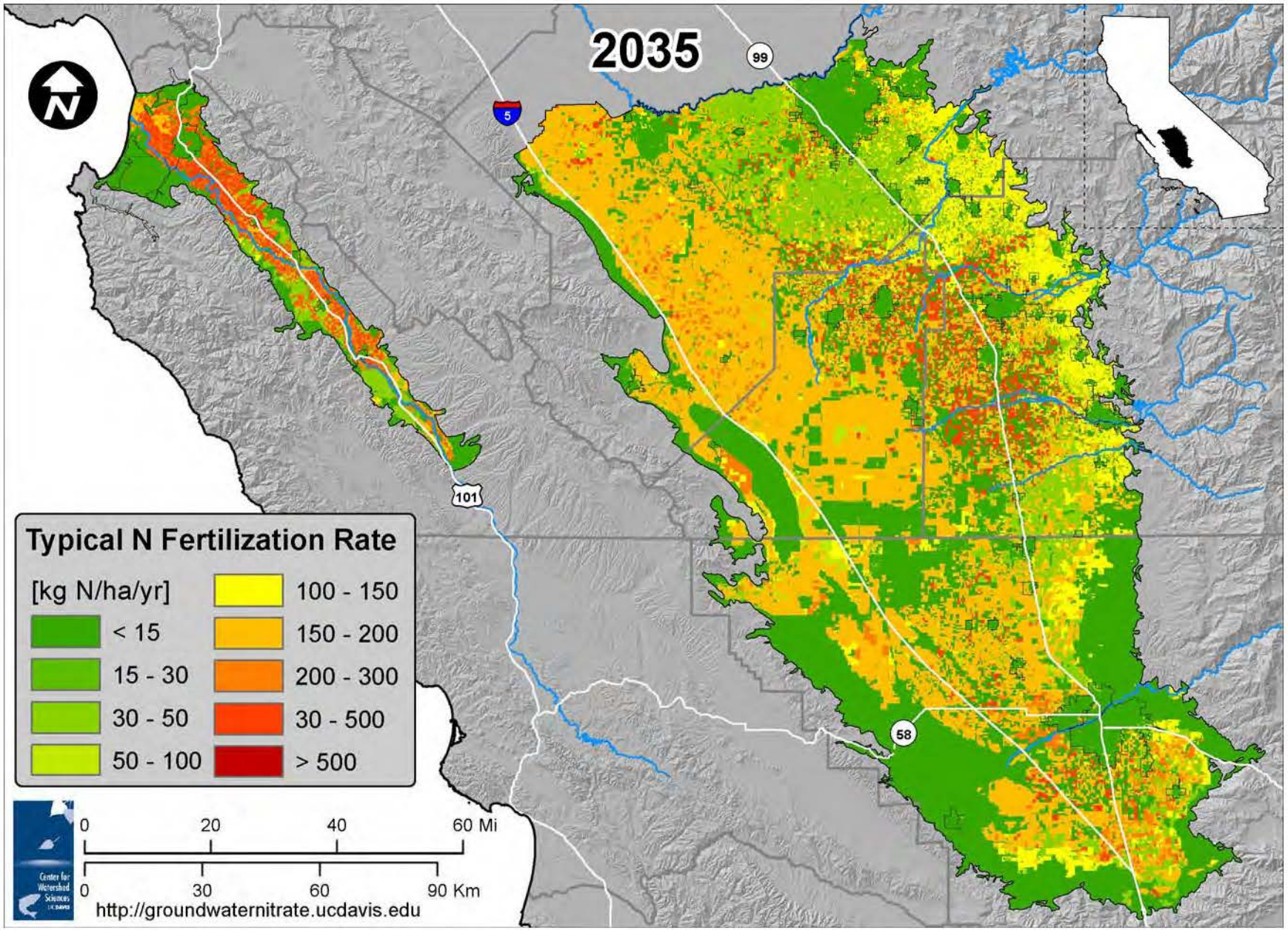
Appendix Figure 23. Typical nitrogen application rate in 2005.



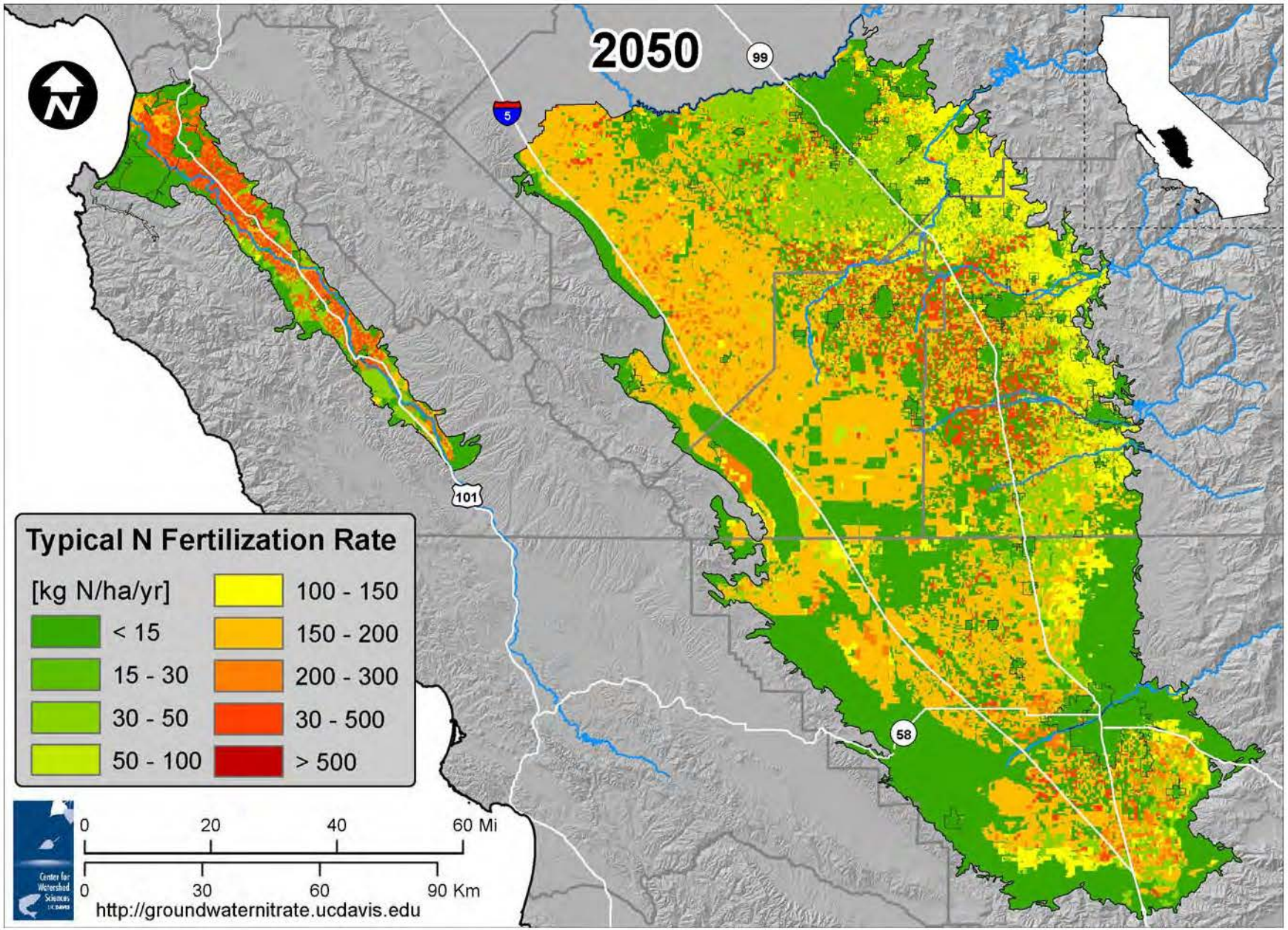
Appendix Figure 24. Typical nitrogen application rate in 2020.



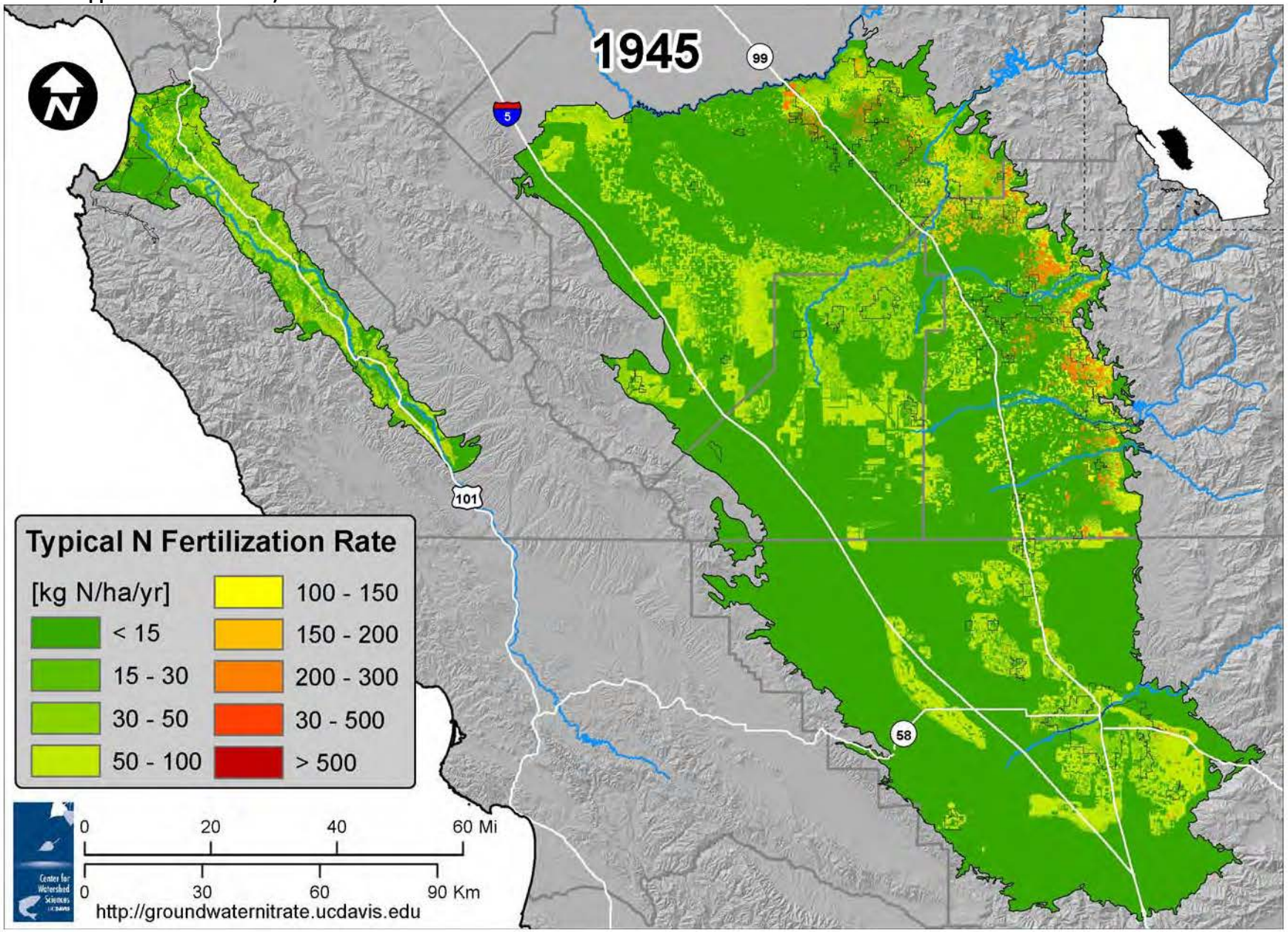
Appendix Figure 25. Typical nitrogen application rate in 2035.



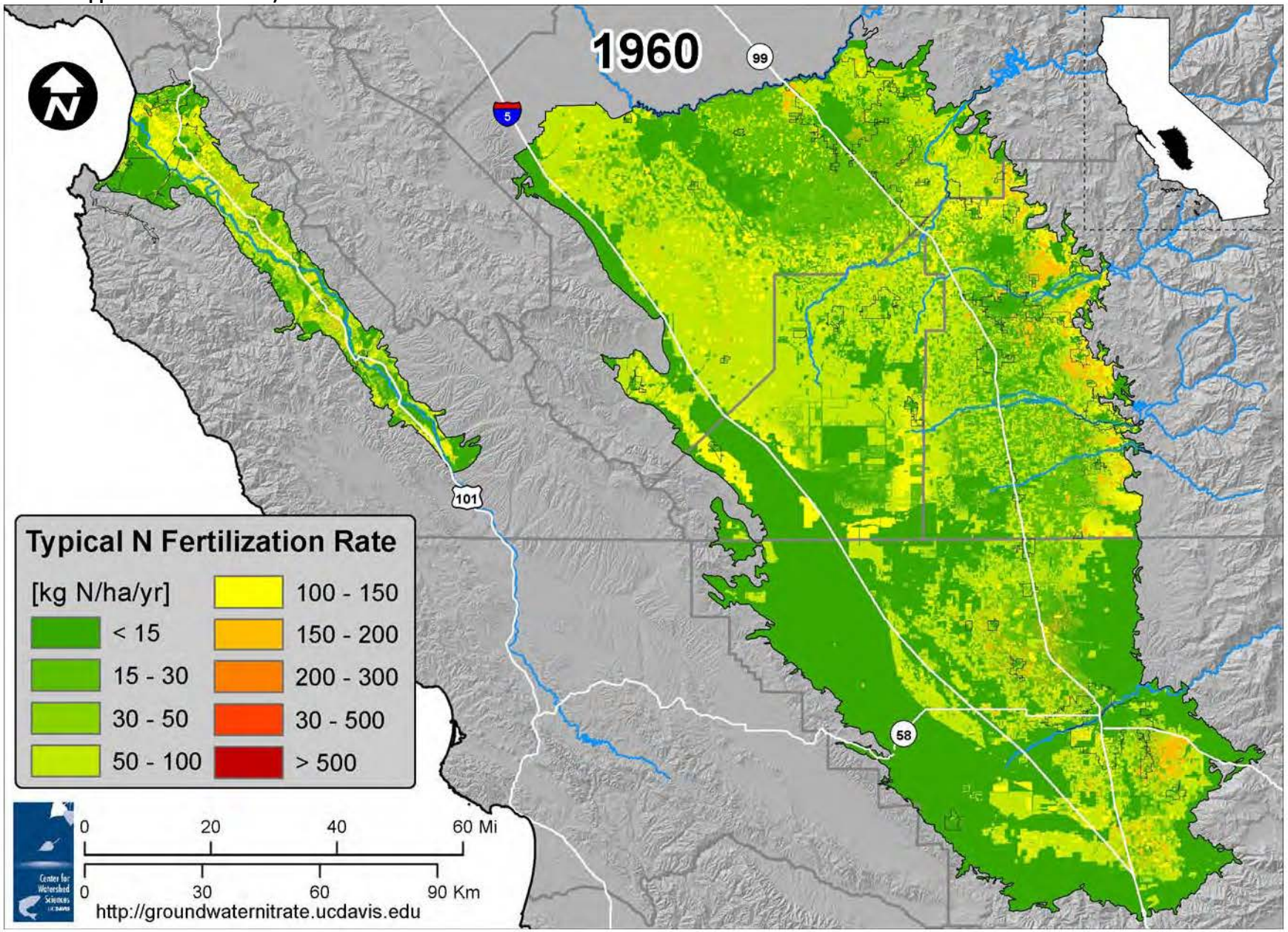
Appendix Figure 26. Typical nitrogen application rate in 2050.



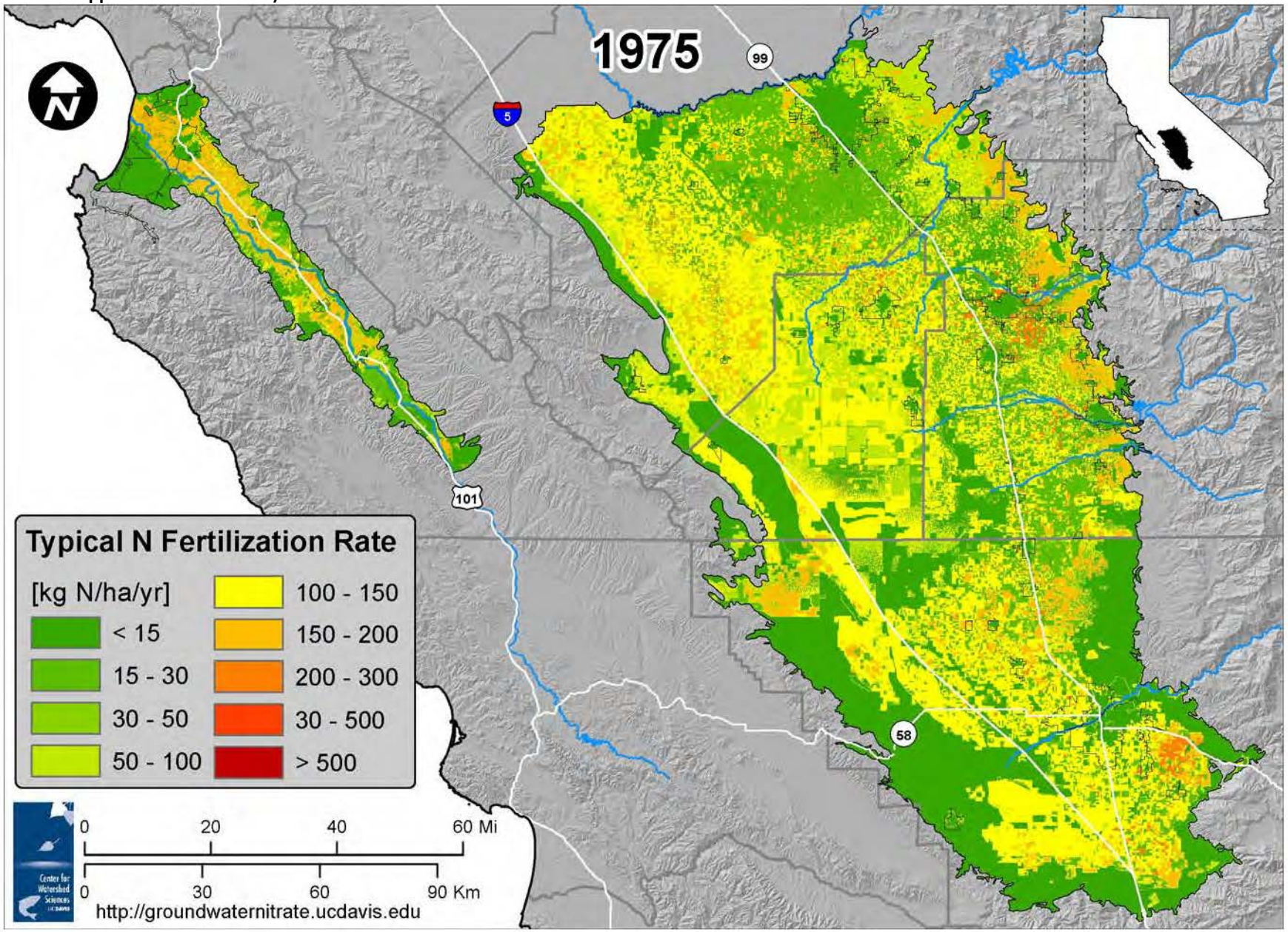
Appendix Figure 27. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 1945 (same as typical fertilizer application rate in 1945).



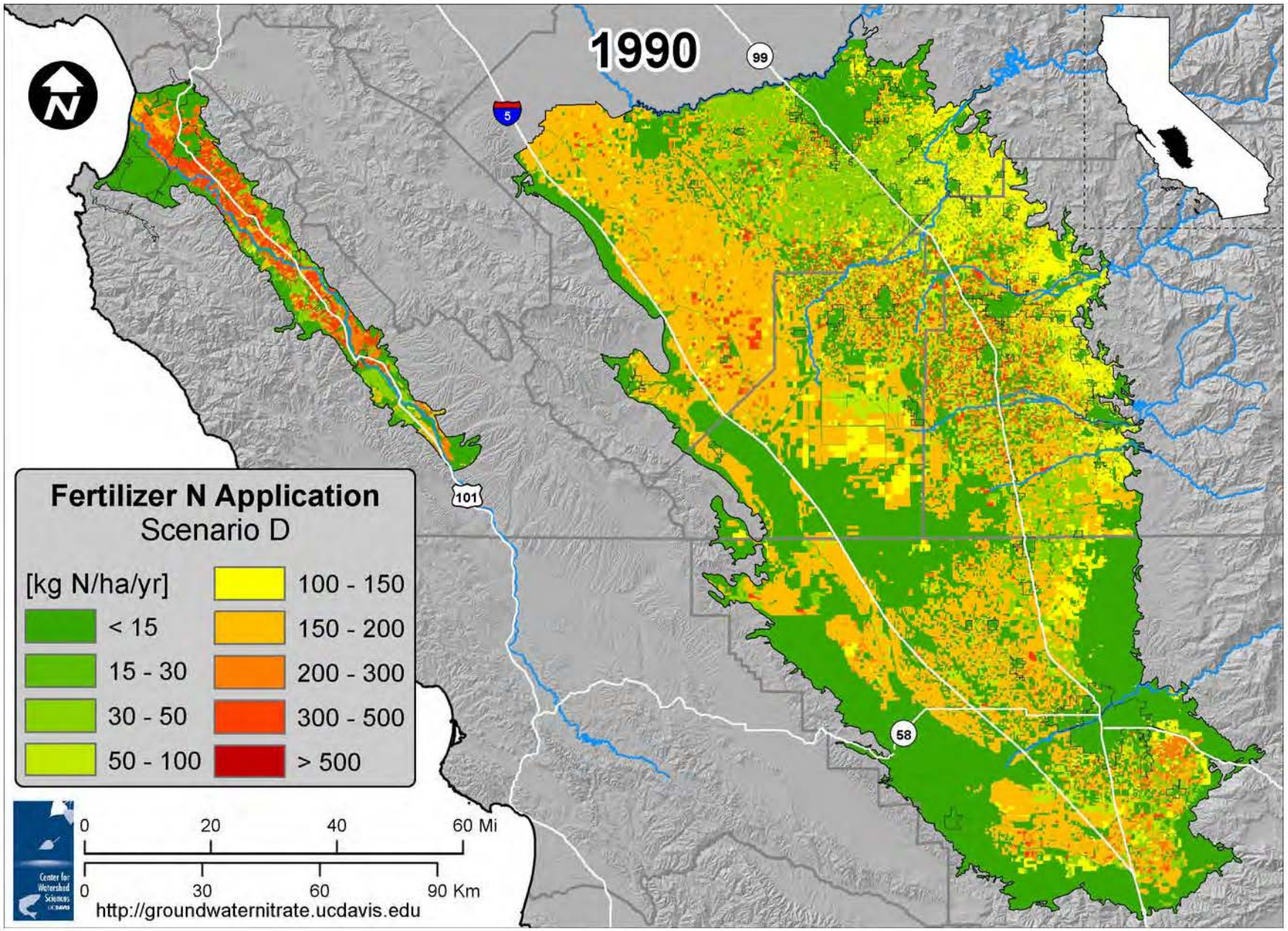
Appendix Figure 28. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 1960 (same as typical fertilizer application rate in 1960).



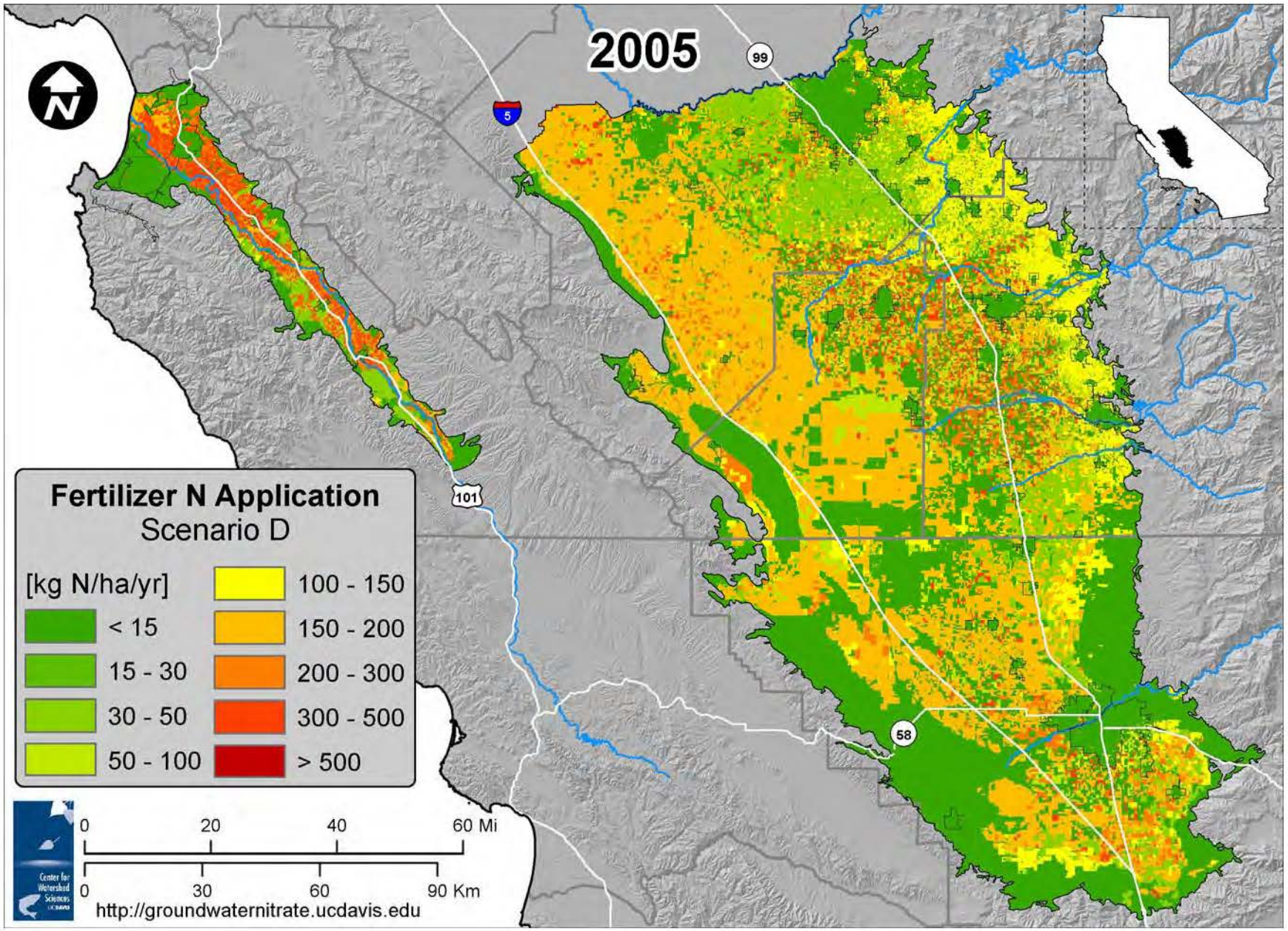
Appendix Figure 29. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 1975 (same as typical fertilizer application rate in 1975).



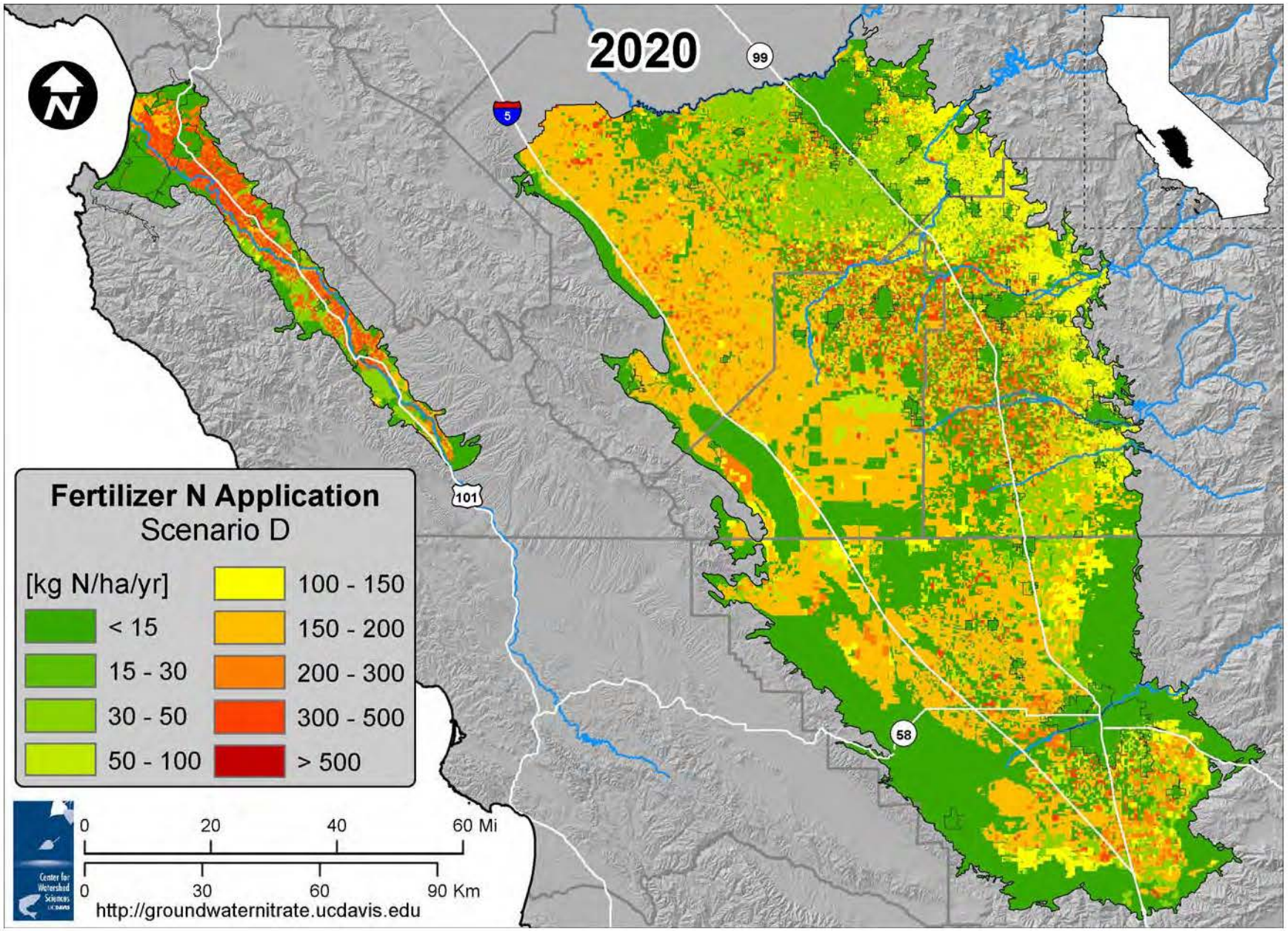
Appendix Figure 30. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 1990.



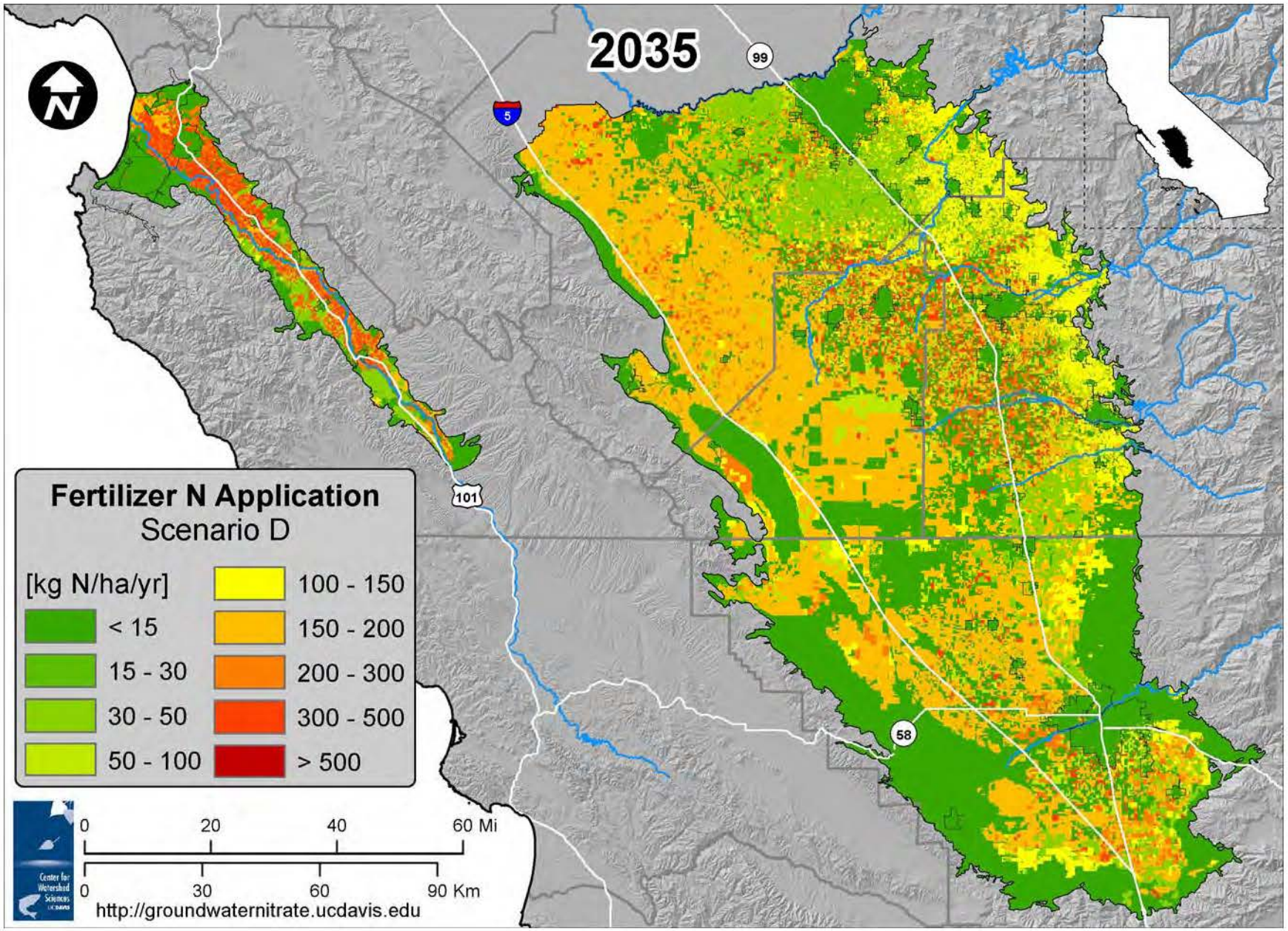
Appendix Figure 31. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 2005.



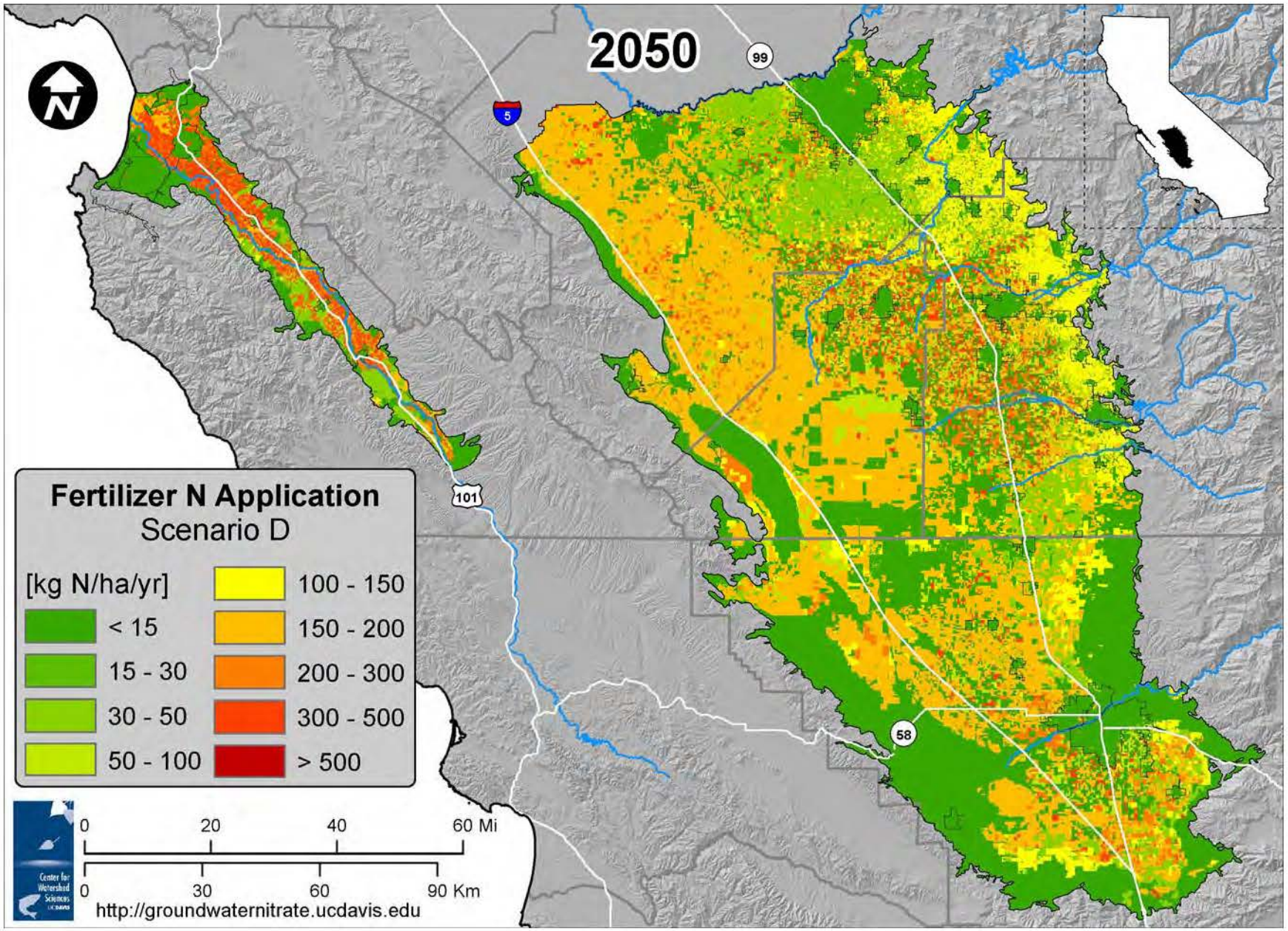
Appendix Figure 32. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 2020.



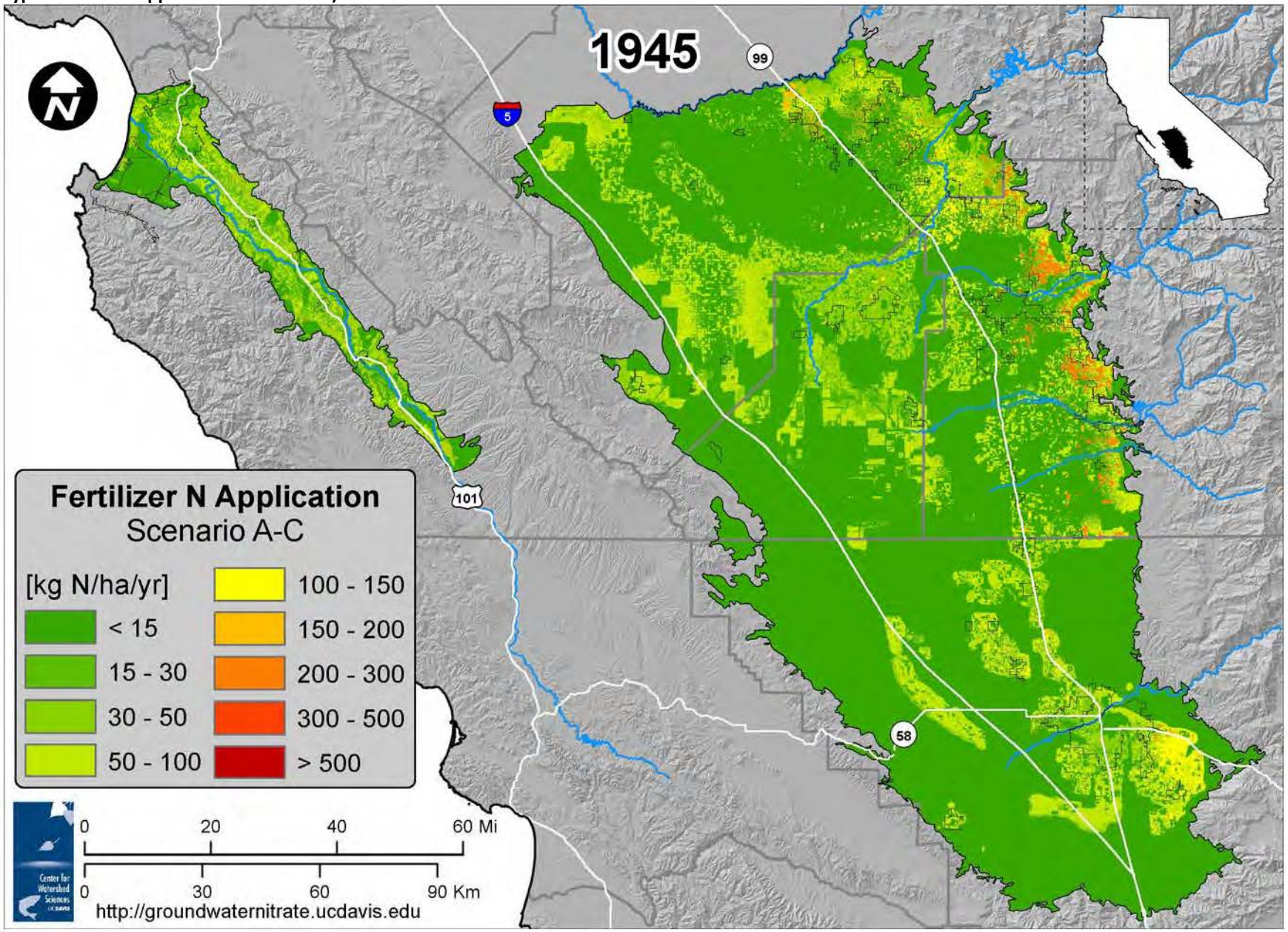
Appendix Figure 33. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 2035.



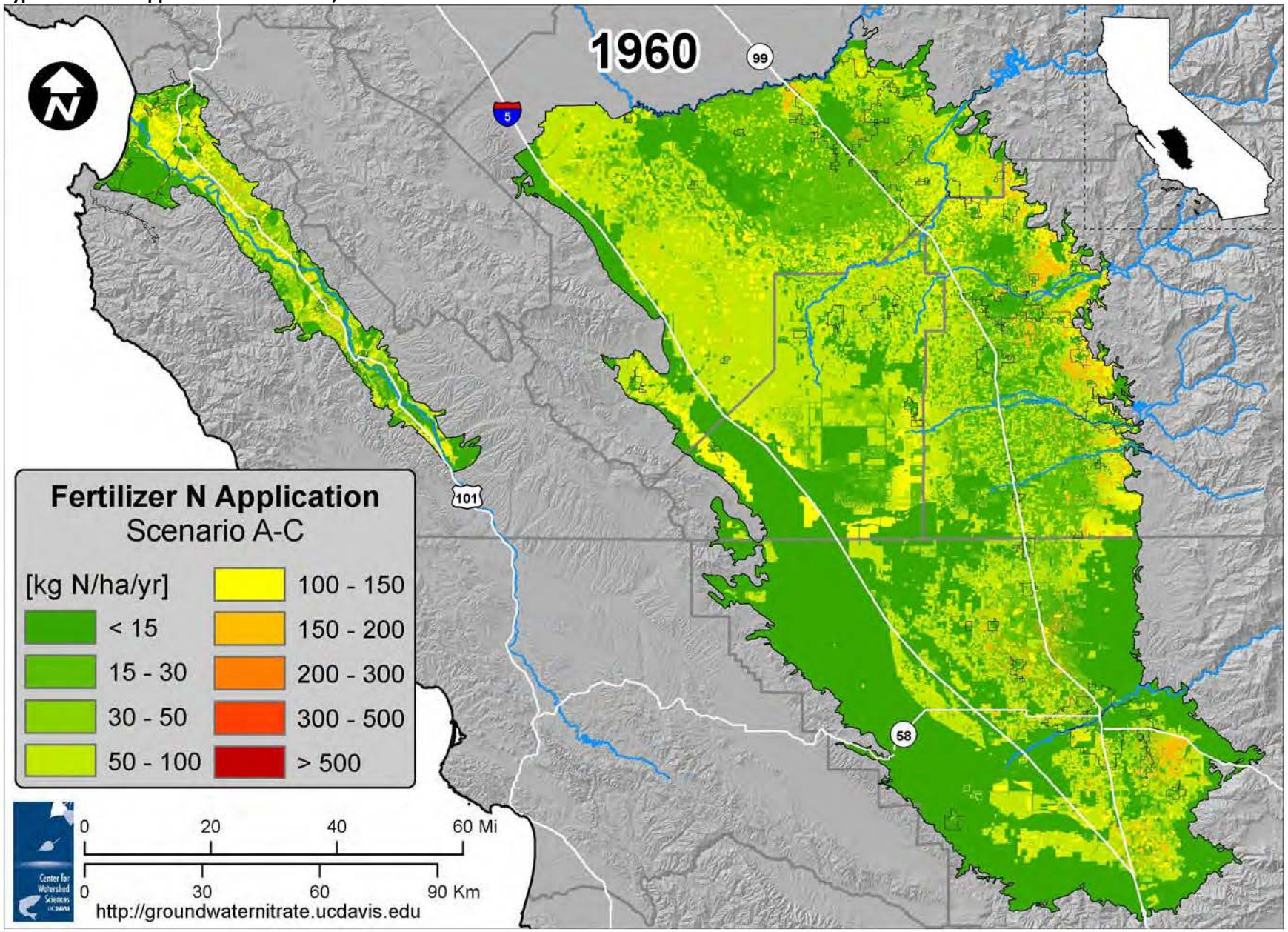
Appendix Figure 34. Actual synthetic fertilizer application rate, assuming that all manure stays within the dairy's cropland, in 2050.



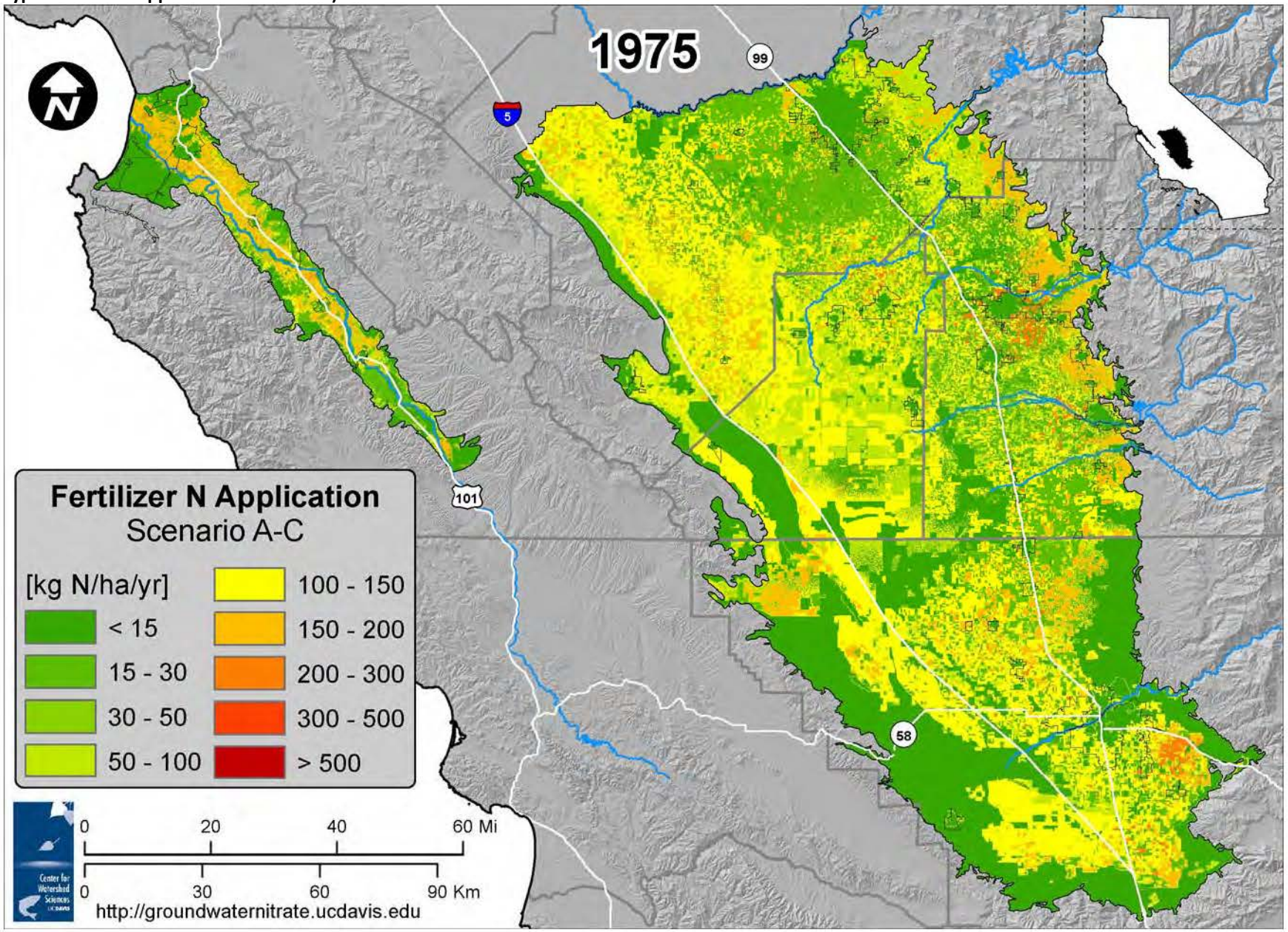
Appendix Figure 35. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 1945 (same as typical fertilizer application rate in 1945).



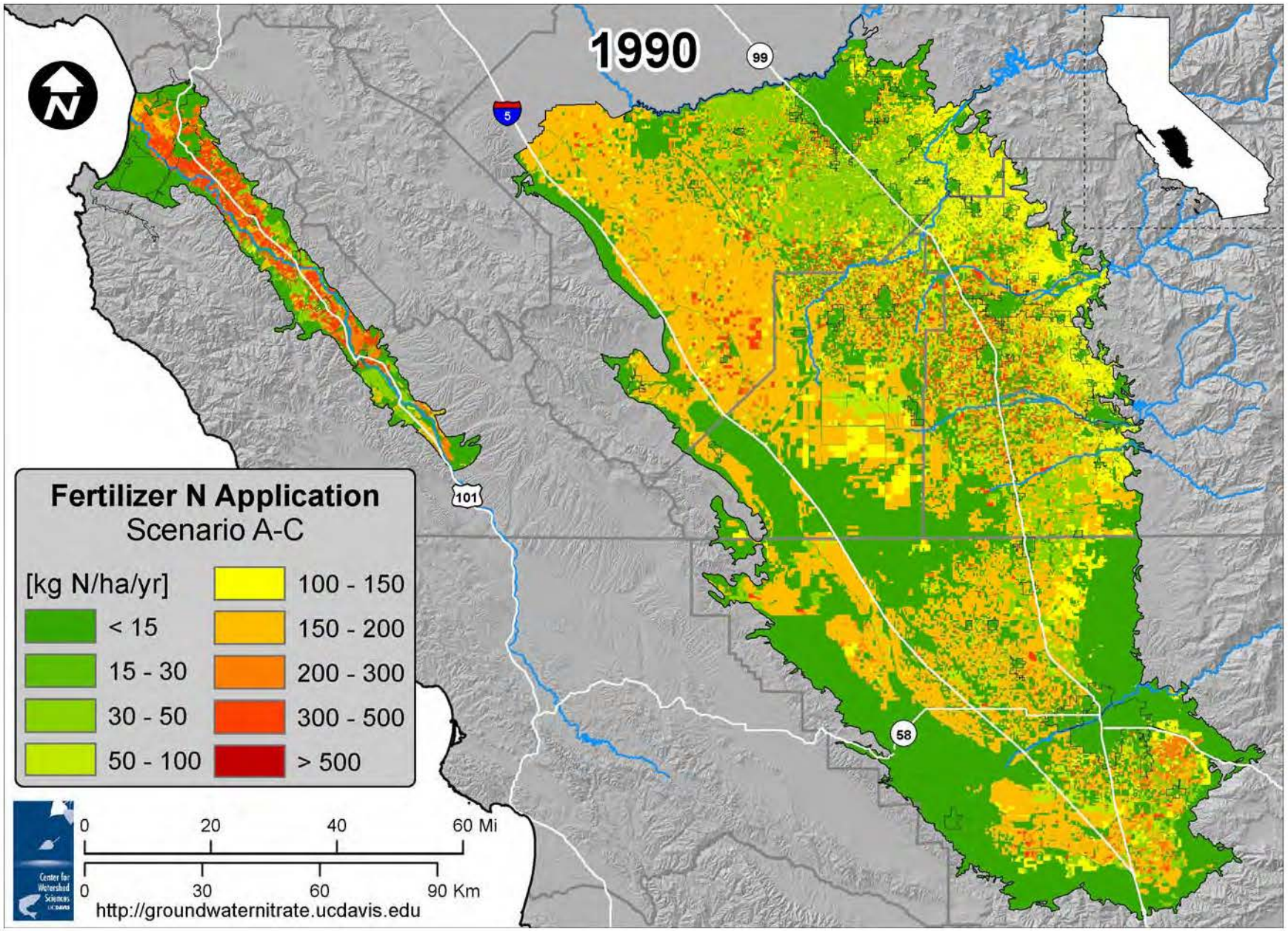
Appendix Figure 36. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 1960 (same as typical fertilizer application rate in 1960).



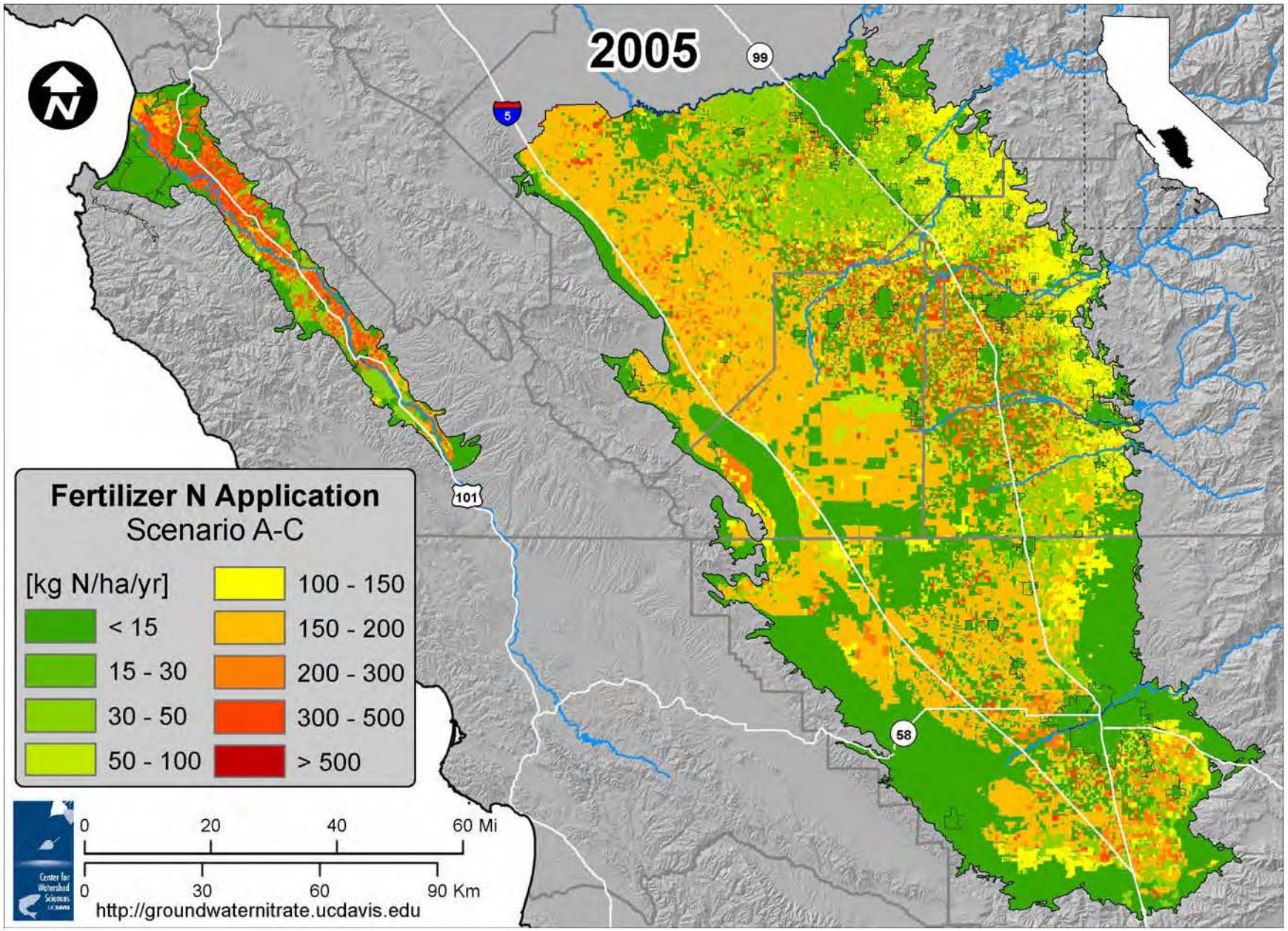
Appendix Figure 37. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 1975 (same as typical fertilizer application rate in 1975).



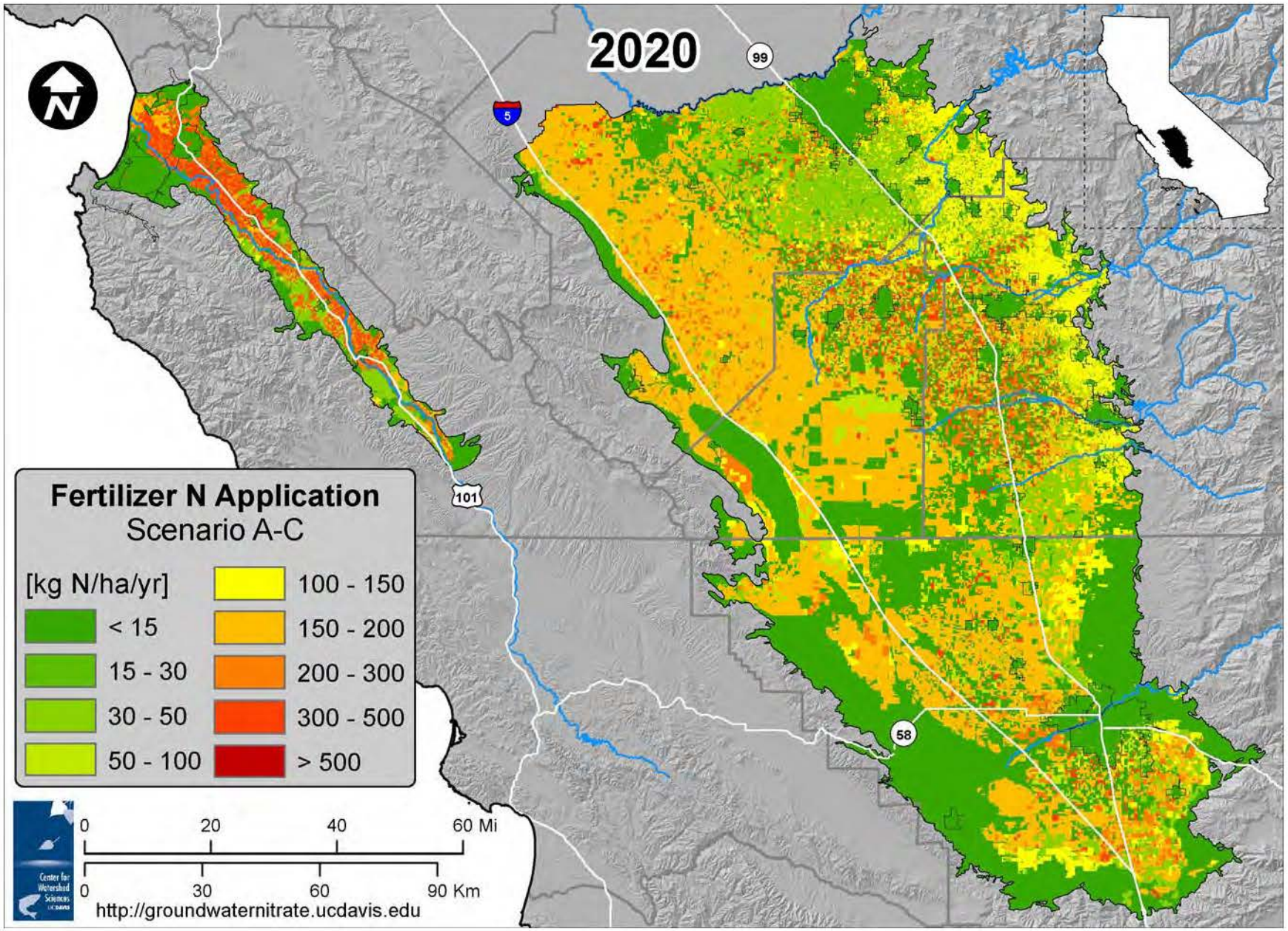
Appendix Figure 38. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 1990.



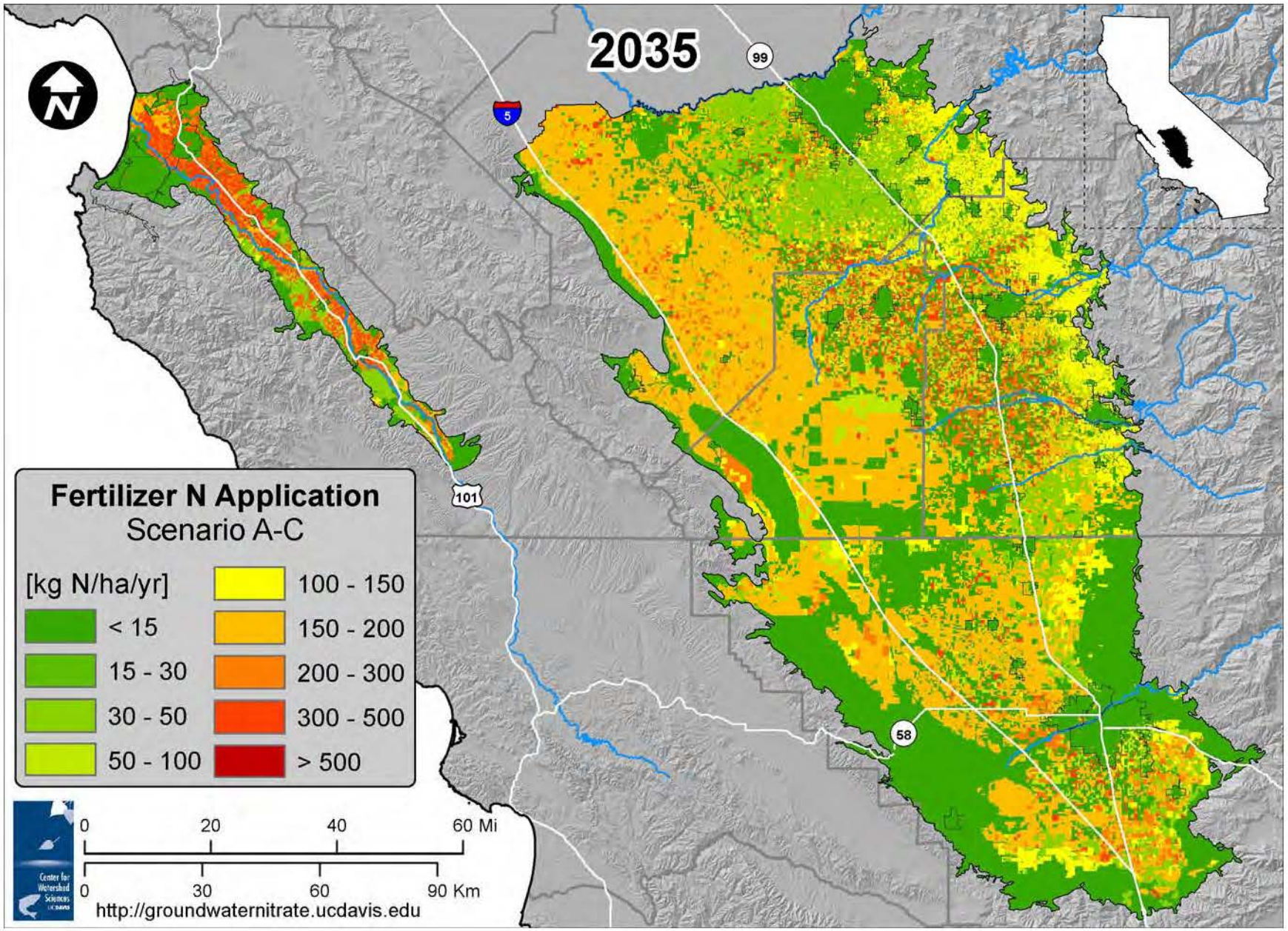
Appendix Figure 39. Actual synthetic fertilizer application rate, assuming that 38% of excreted of manure nitrogen is exported from dairies, in 2005.



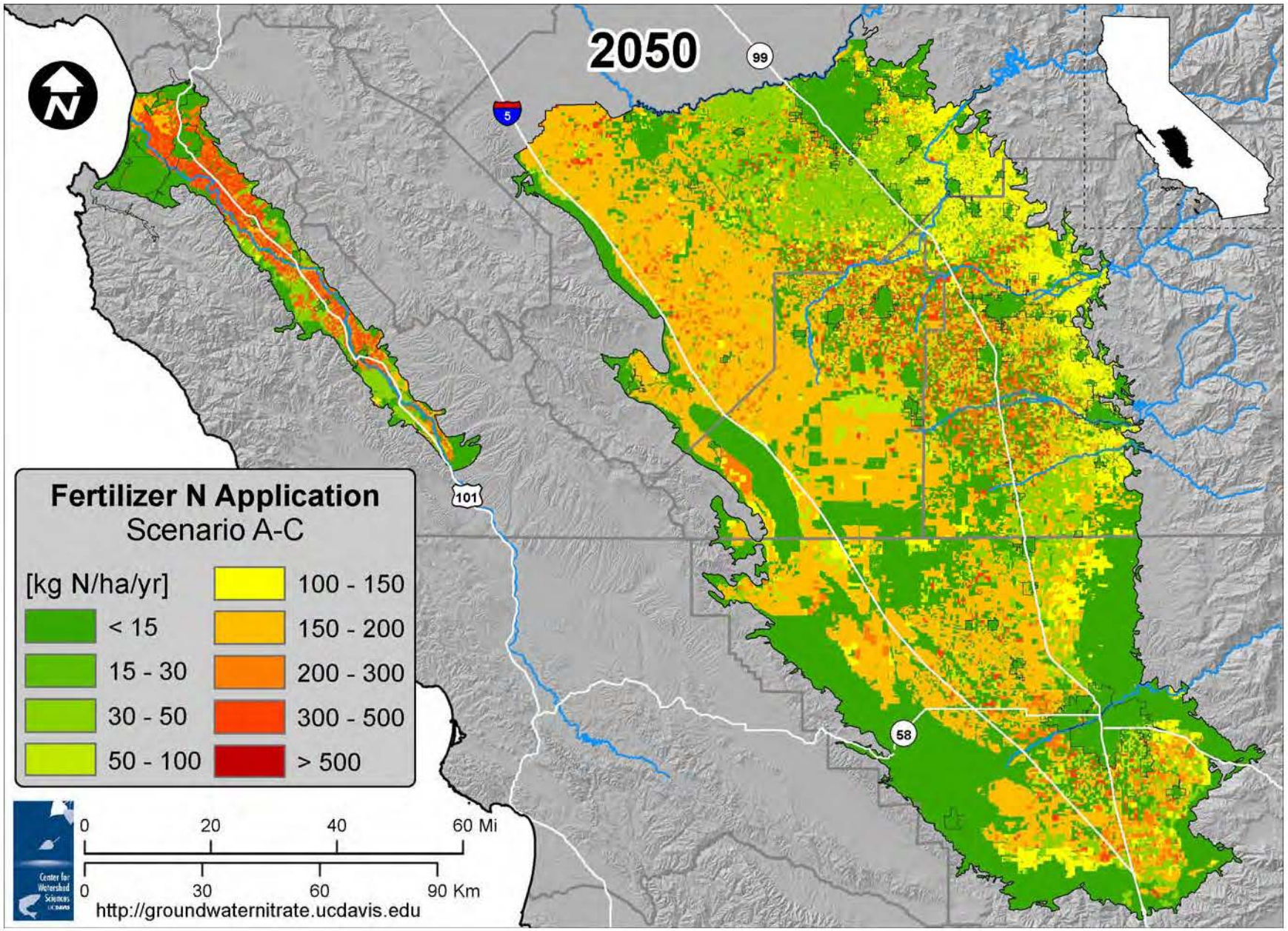
Appendix Figure 40. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 2020.



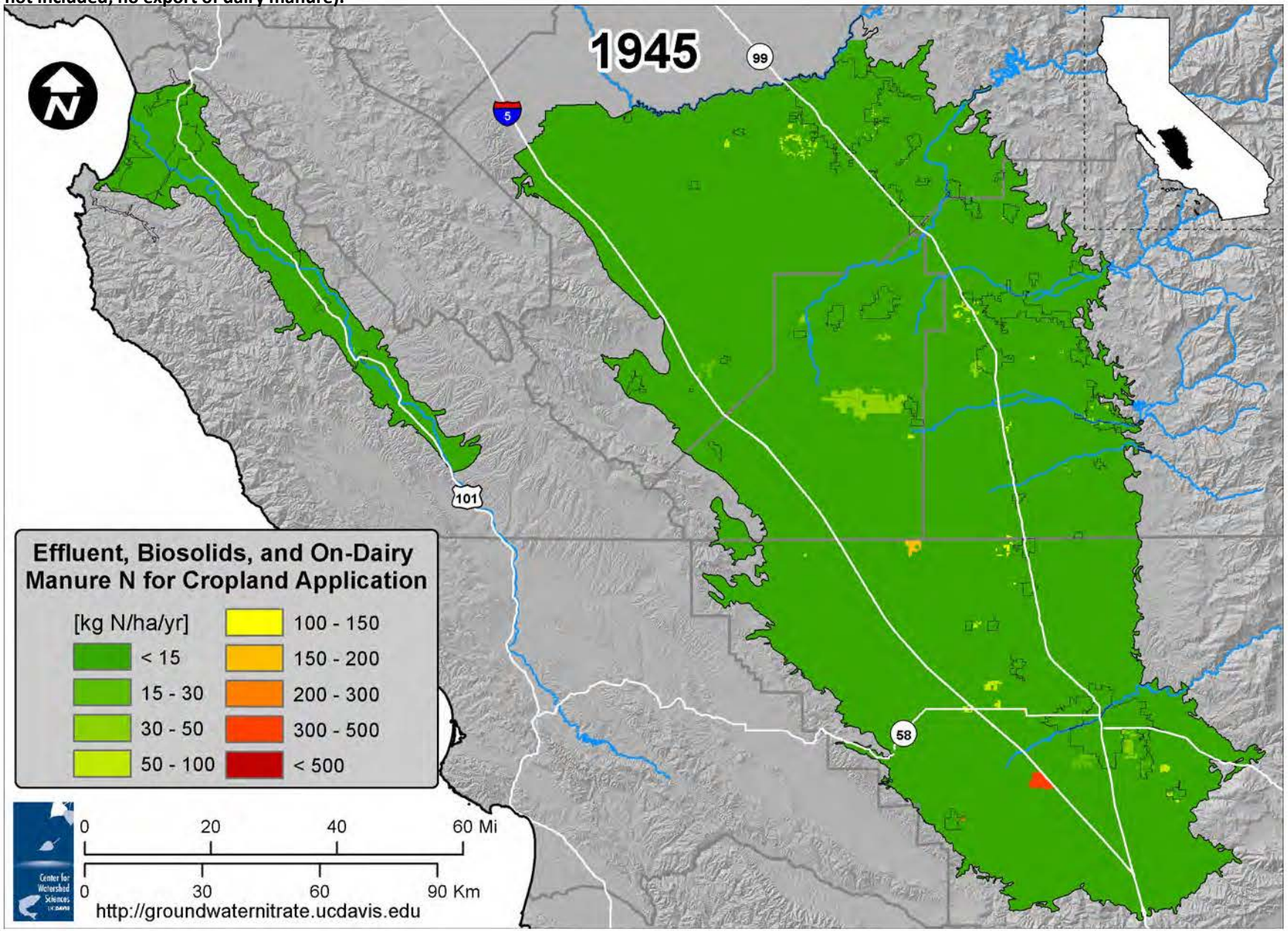
Appendix Figure 41. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 2035.



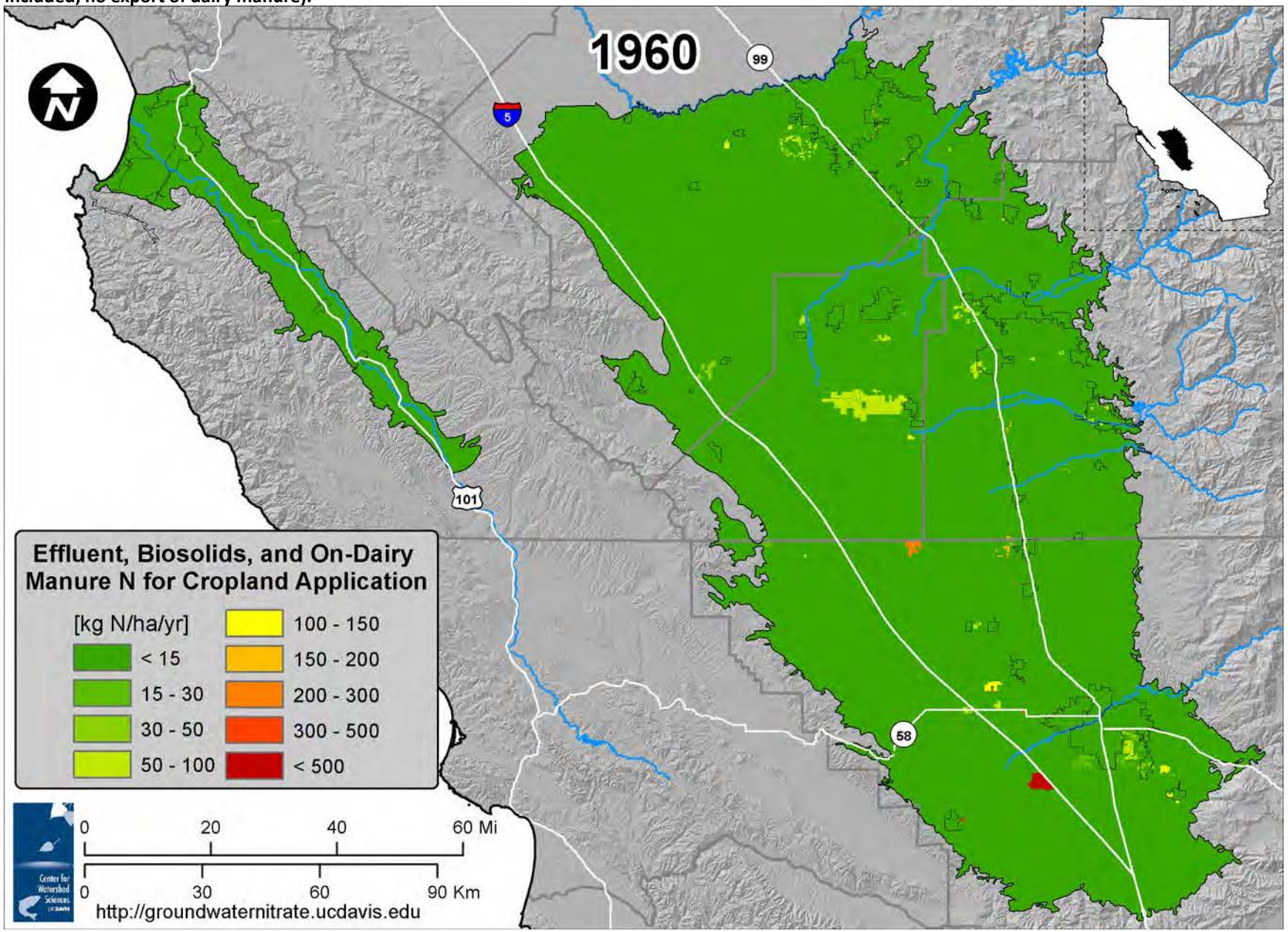
Appendix Figure 42. Actual synthetic fertilizer application rate, assuming that 38% of excreted manure nitrogen is exported from dairies, in 2050.



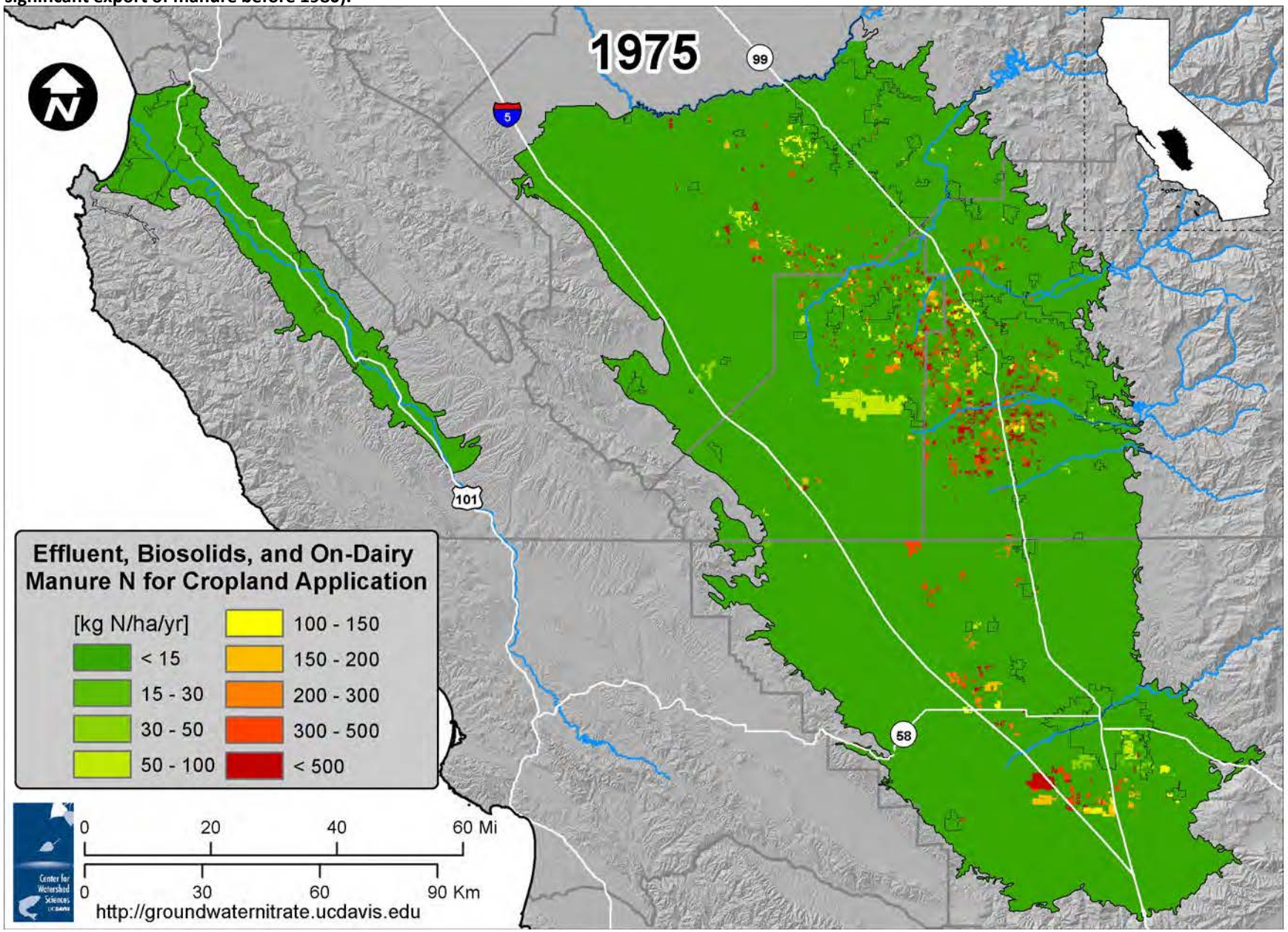
Appendix Figure 43. Land applied nitrogen from WWTP effluent, FP effluent, and biosolids, in 1945 (dairy animals assumed to be in pasture; manure N not included; no export of dairy manure).



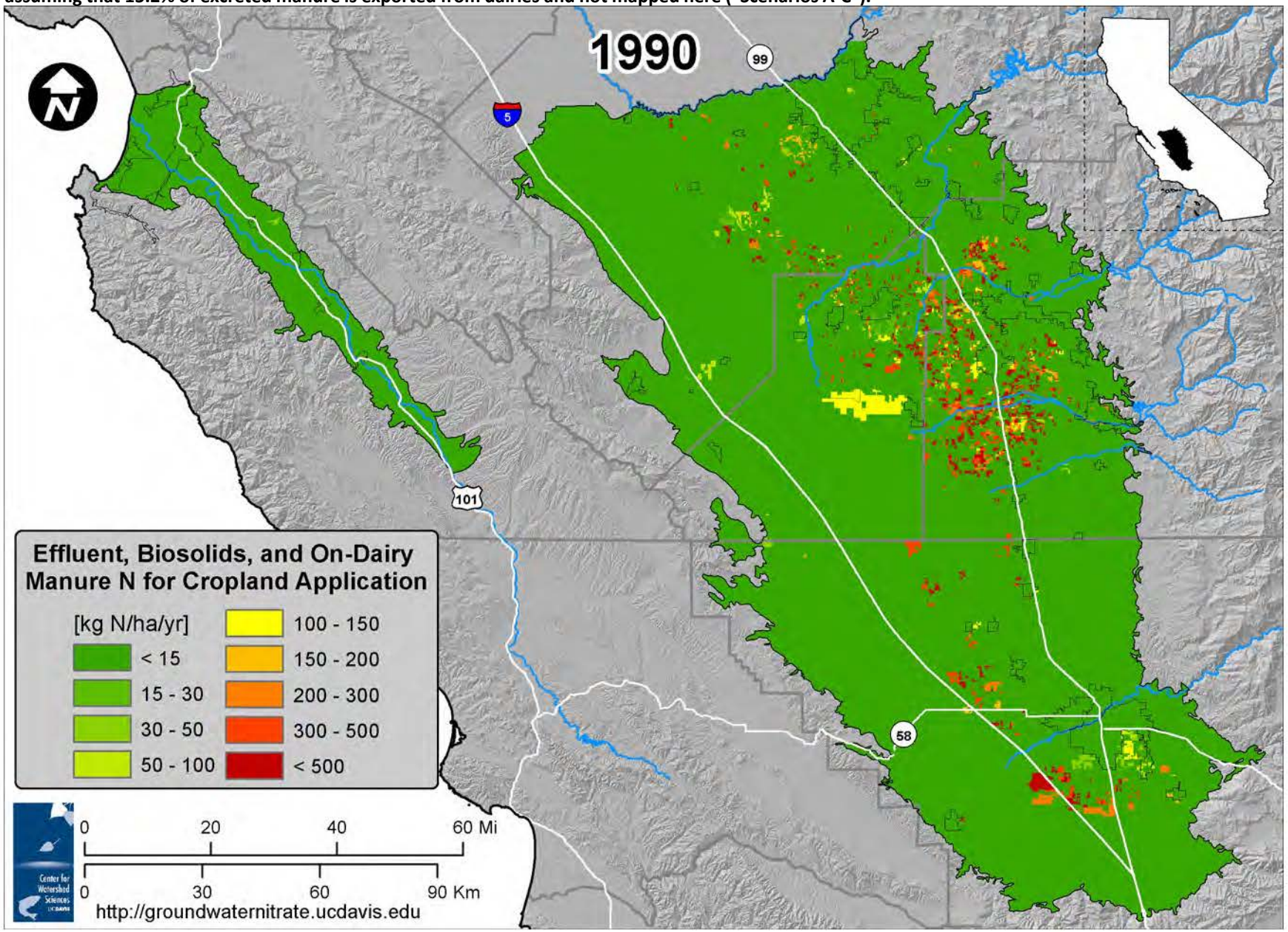
Appendix Figure 44. Land applied nitrogen from WWTP effluent, FP effluent, and biosolids, in 1960 (dairy animals assumed to be in pasture; manure N not included; no export of dairy manure).



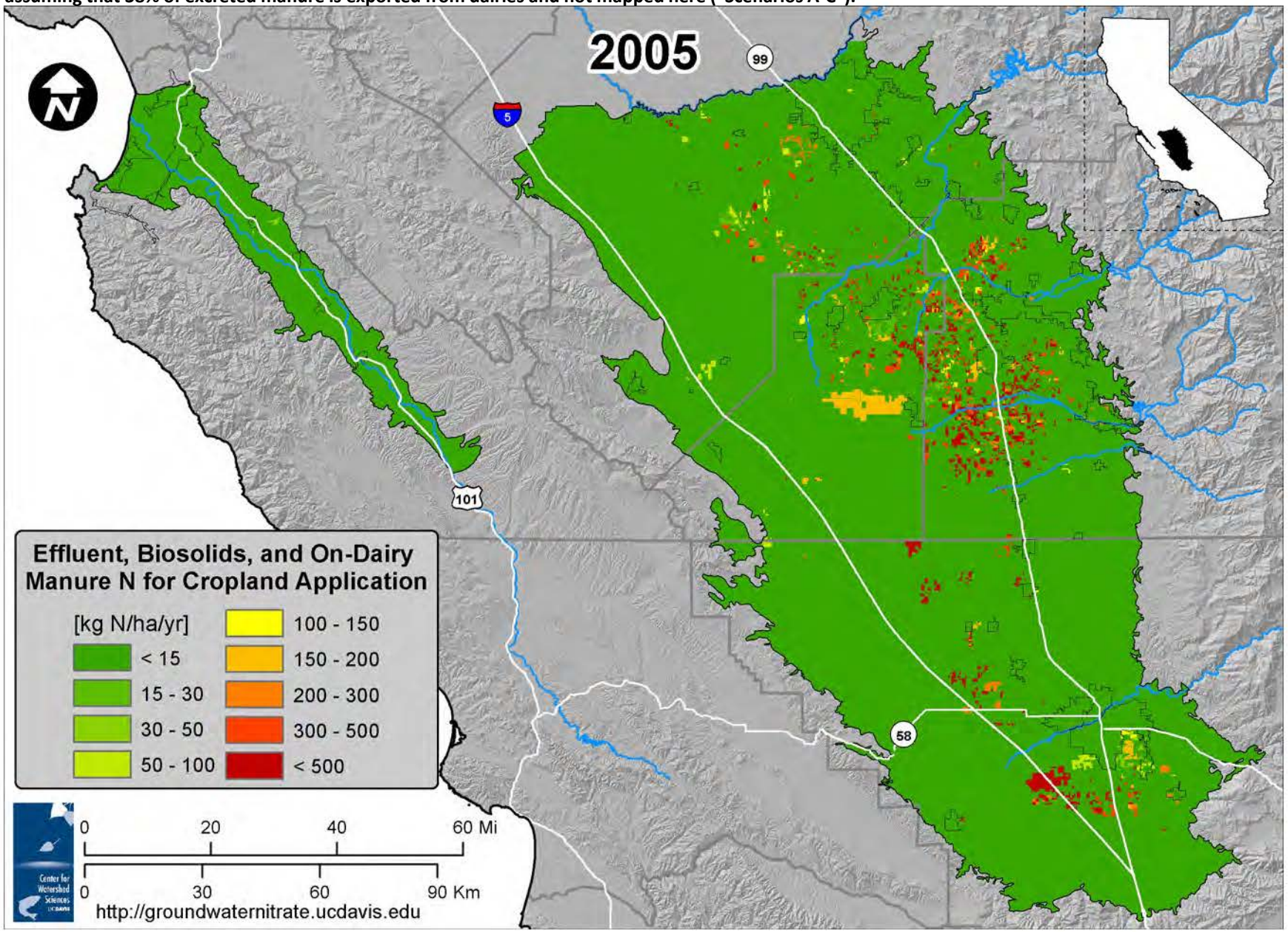
Appendix Figure 45. Land applied nitrogen from WWTP effluent, FP effluent, biosolids, and from dairy manure on dairy-controlled land areas, in 1975 (no significant export of manure before 1980).



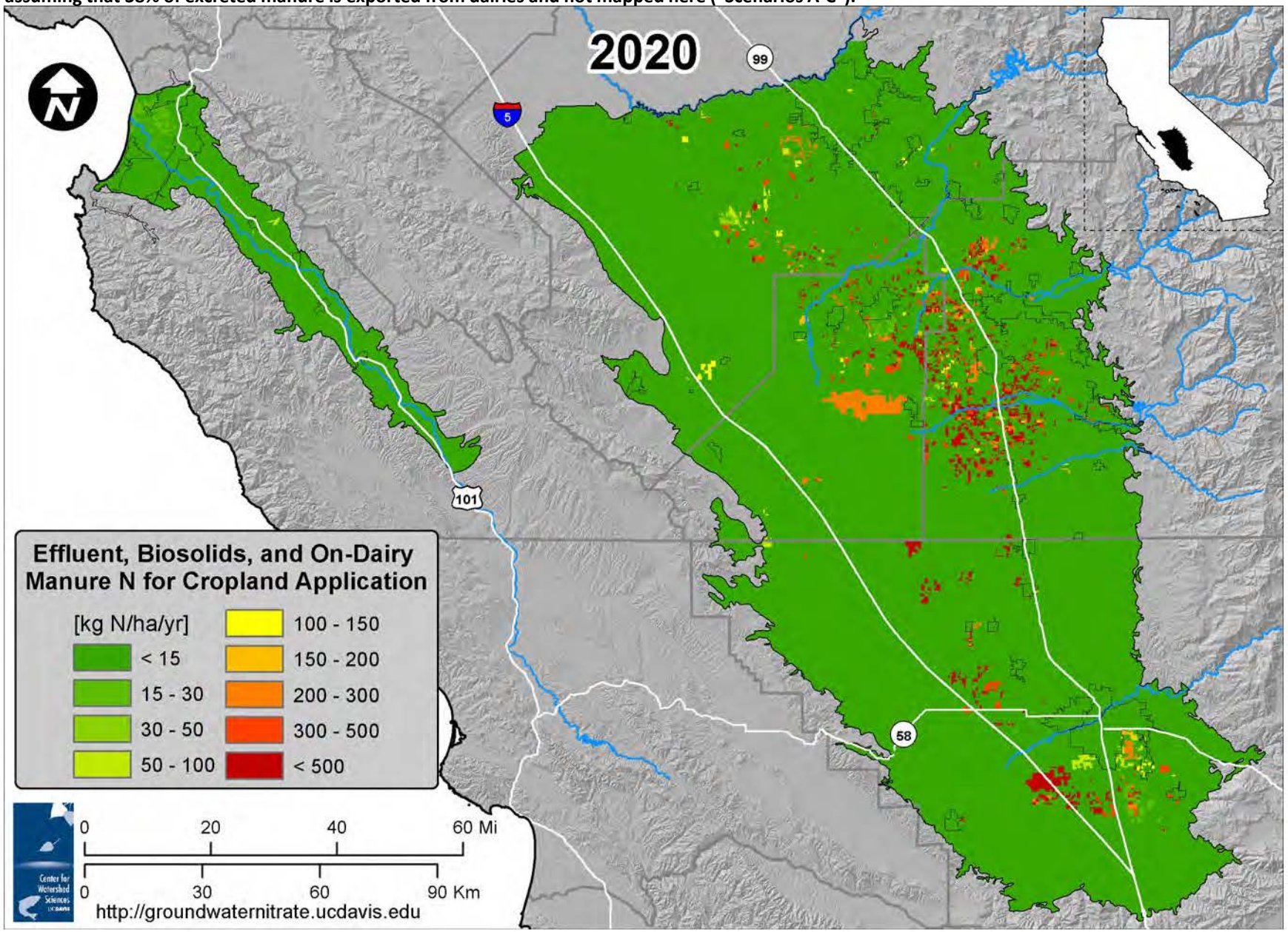
Appendix Figure 46. Land applied nitrogen from WWTP effluent, FP effluent, biosolids, and from dairy manure on dairy-controlled land areas, in 1990, assuming that 15.2% of excreted manure is exported from dairies and not mapped here ("Scenarios A-C").



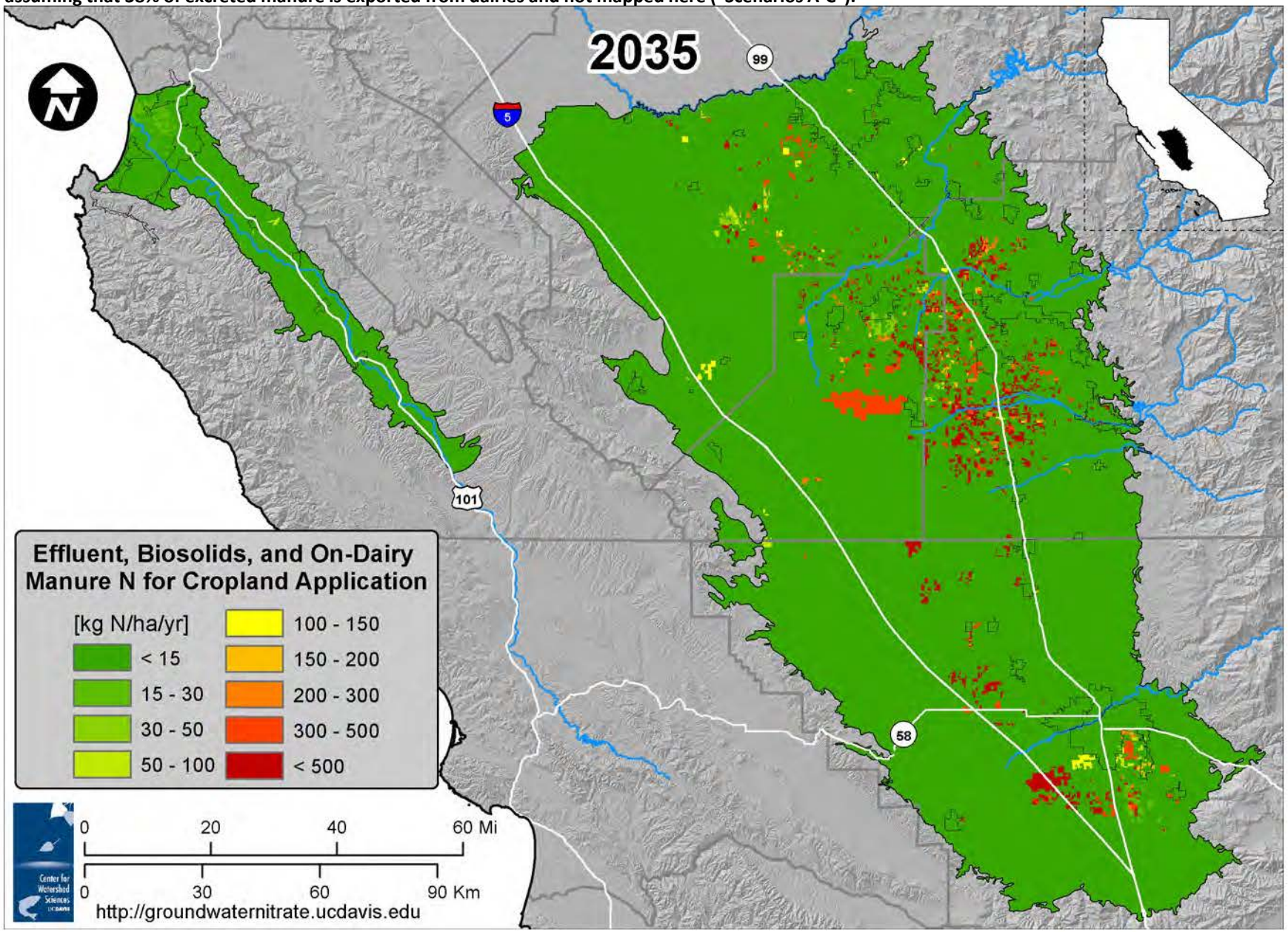
Appendix Figure 47. Land applied nitrogen from WWTP effluent, FP effluent, biosolids, and from dairy manure on dairy-controlled land areas, in 2005, assuming that 38% of excreted manure is exported from dairies and not mapped here ("Scenarios A-C").



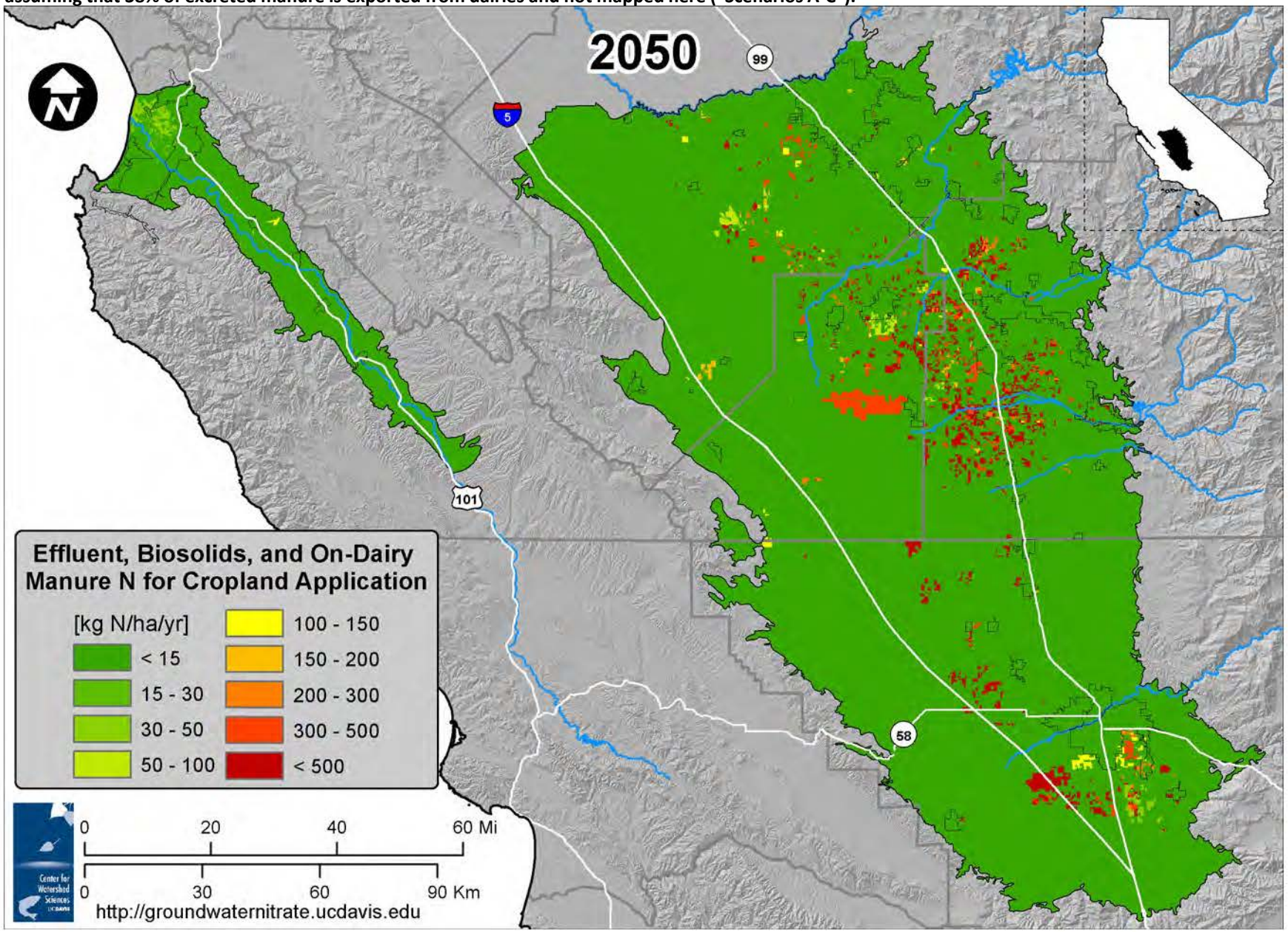
Appendix Figure 48. Land applied nitrogen from WWTP effluent, FP effluent, biosolids, and from dairy manure on dairy-controlled land areas, in 2020, assuming that 38% of excreted manure is exported from dairies and not mapped here ("Scenarios A-C").



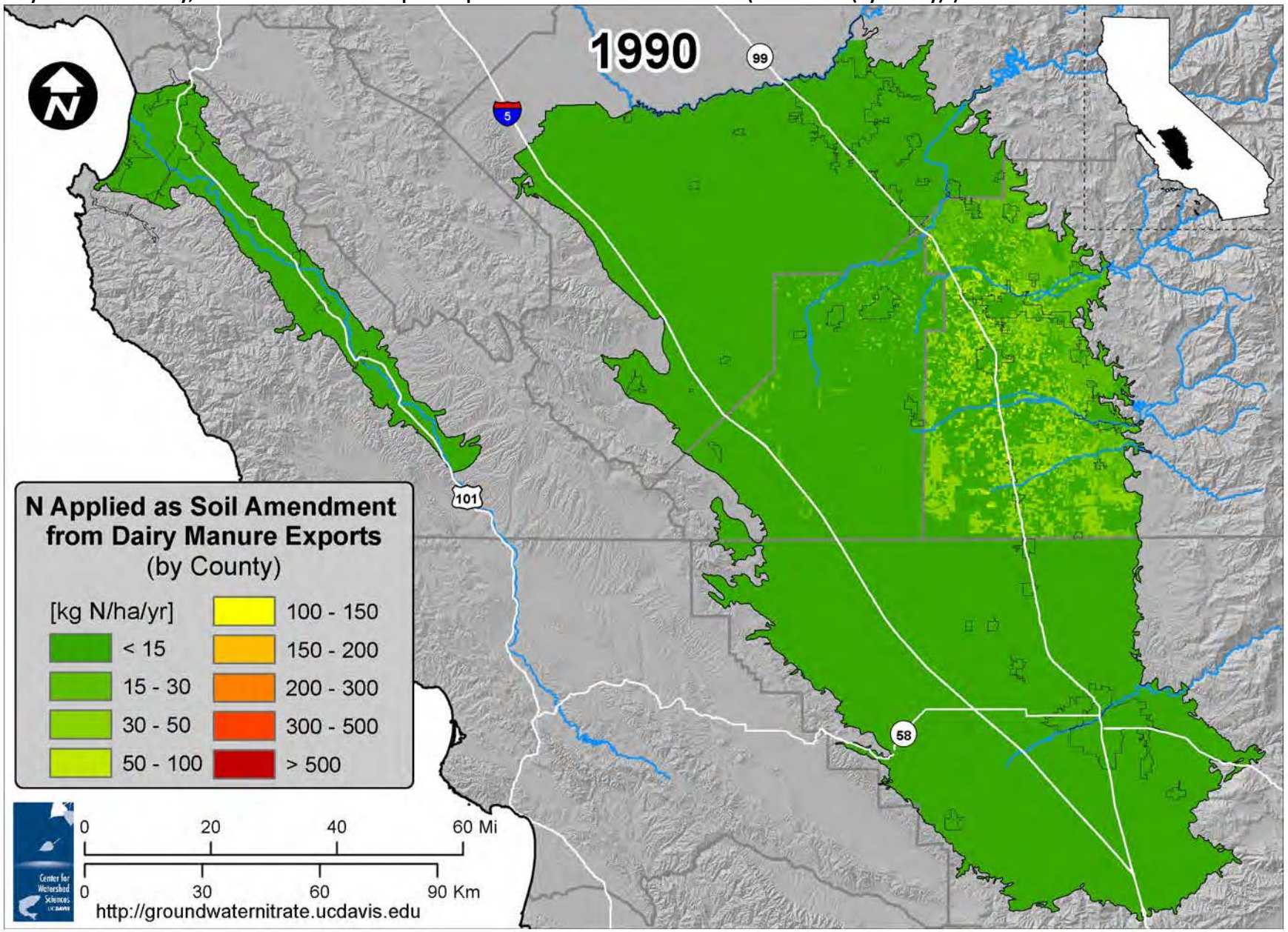
Appendix Figure 49. Land applied nitrogen from WWTP effluent, FP effluent, biosolids, and from dairy manure on dairy-controlled land areas, in 2035, assuming that 38% of excreted manure is exported from dairies and not mapped here ("Scenarios A-C").



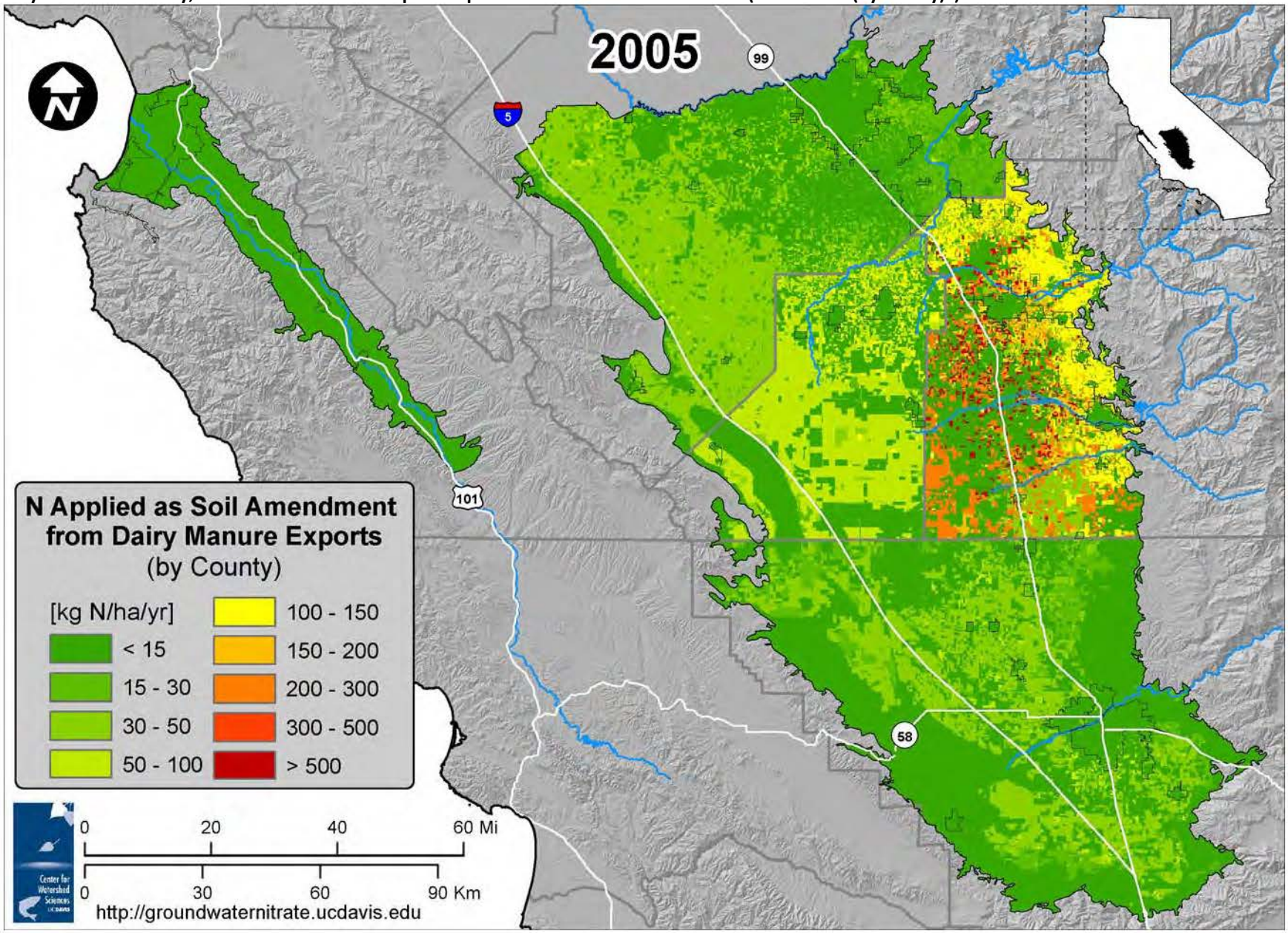
Appendix Figure 50. Land applied nitrogen from WWTP effluent, FP effluent, biosolids, and from dairy manure on dairy-controlled land areas, in 2050, assuming that 38% of excreted manure is exported from dairies and not mapped here ("Scenarios A-C").



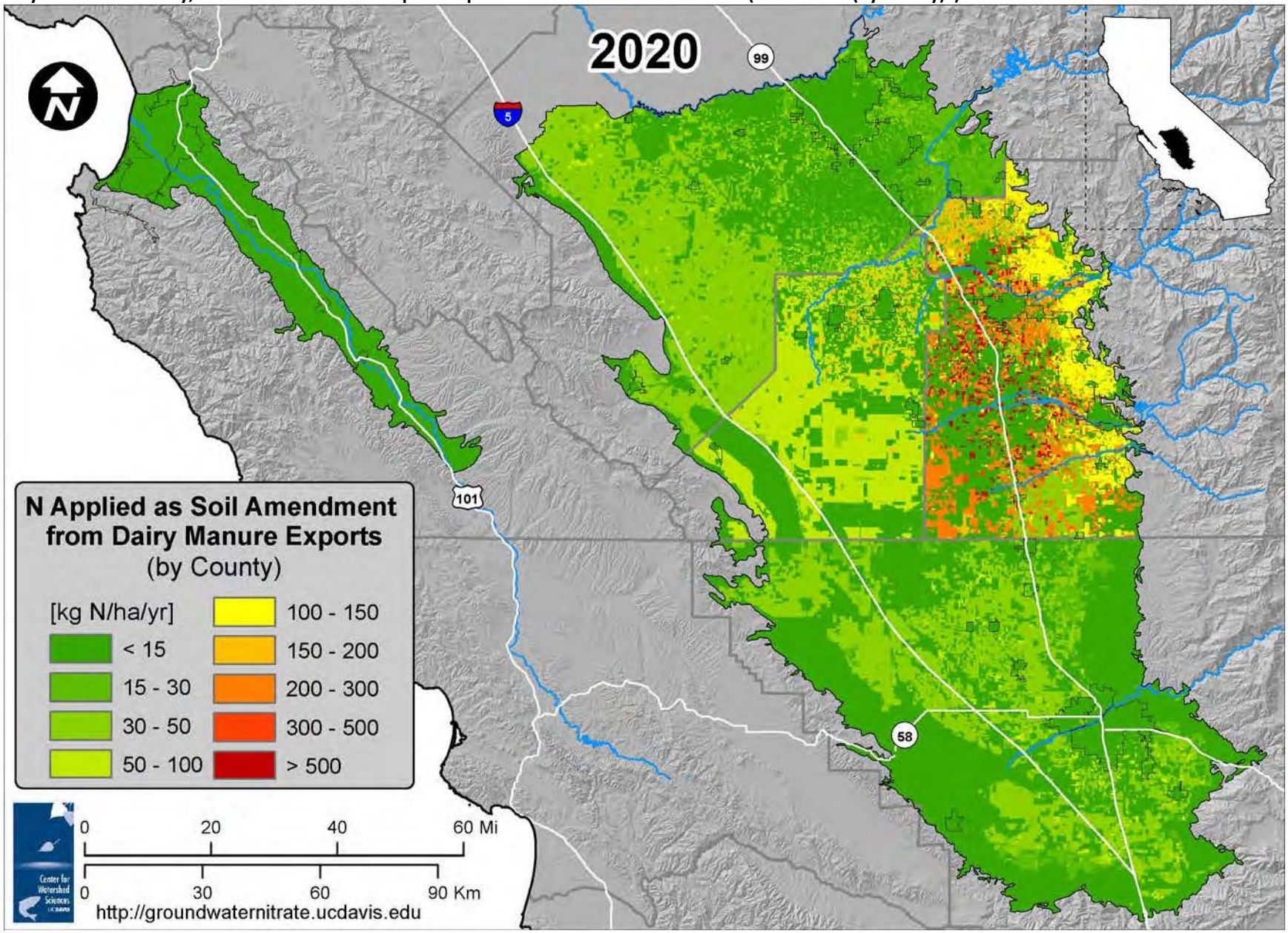
Appendix Figure 51. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the county, in 1990. Total manure exports represent 38% of animal N excreted ("Scenario C (by county)").



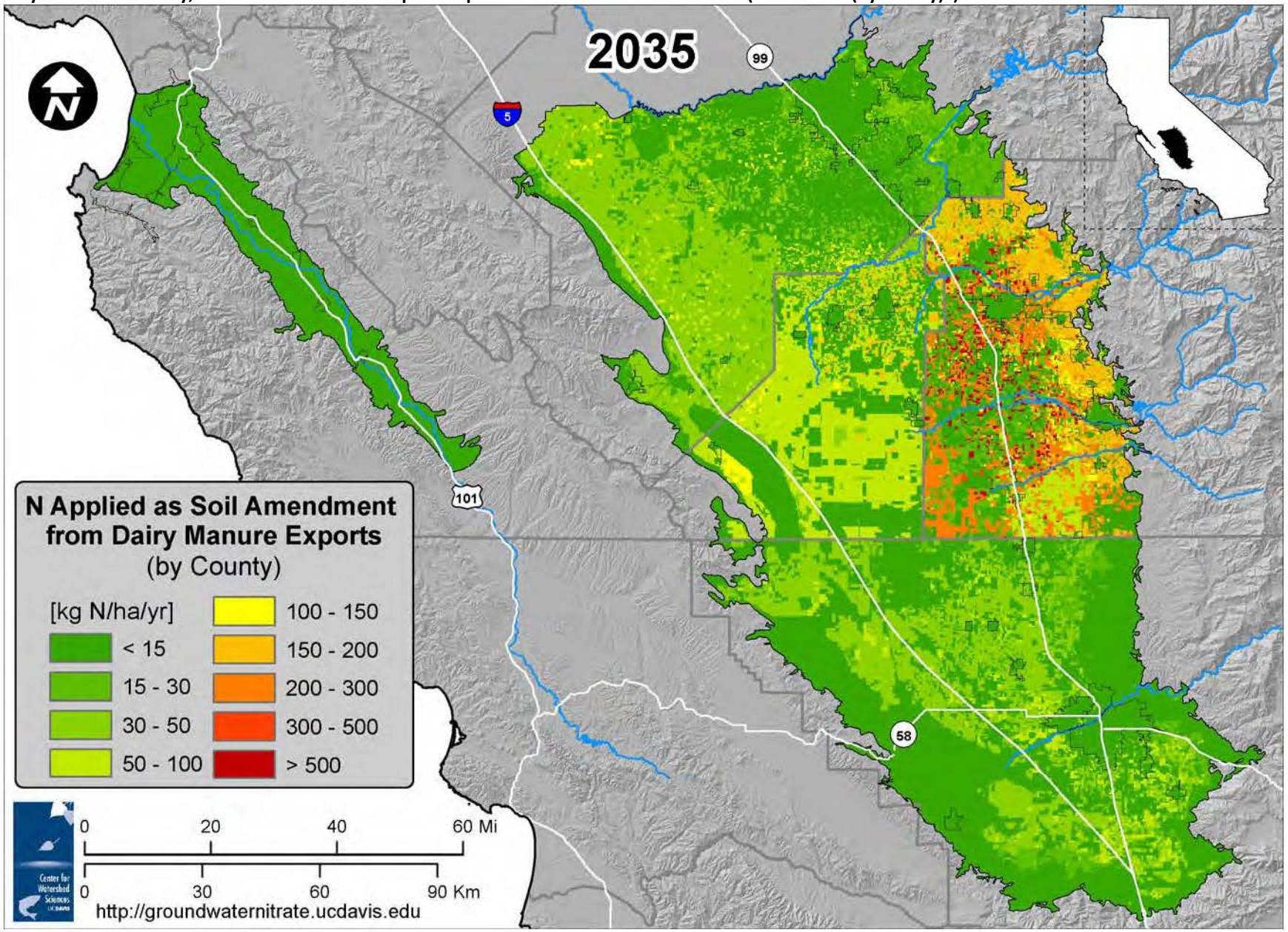
Appendix Figure 52. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the county, in 2005. Total manure exports represent 38% of animal N excreted ("Scenario C (by county)").



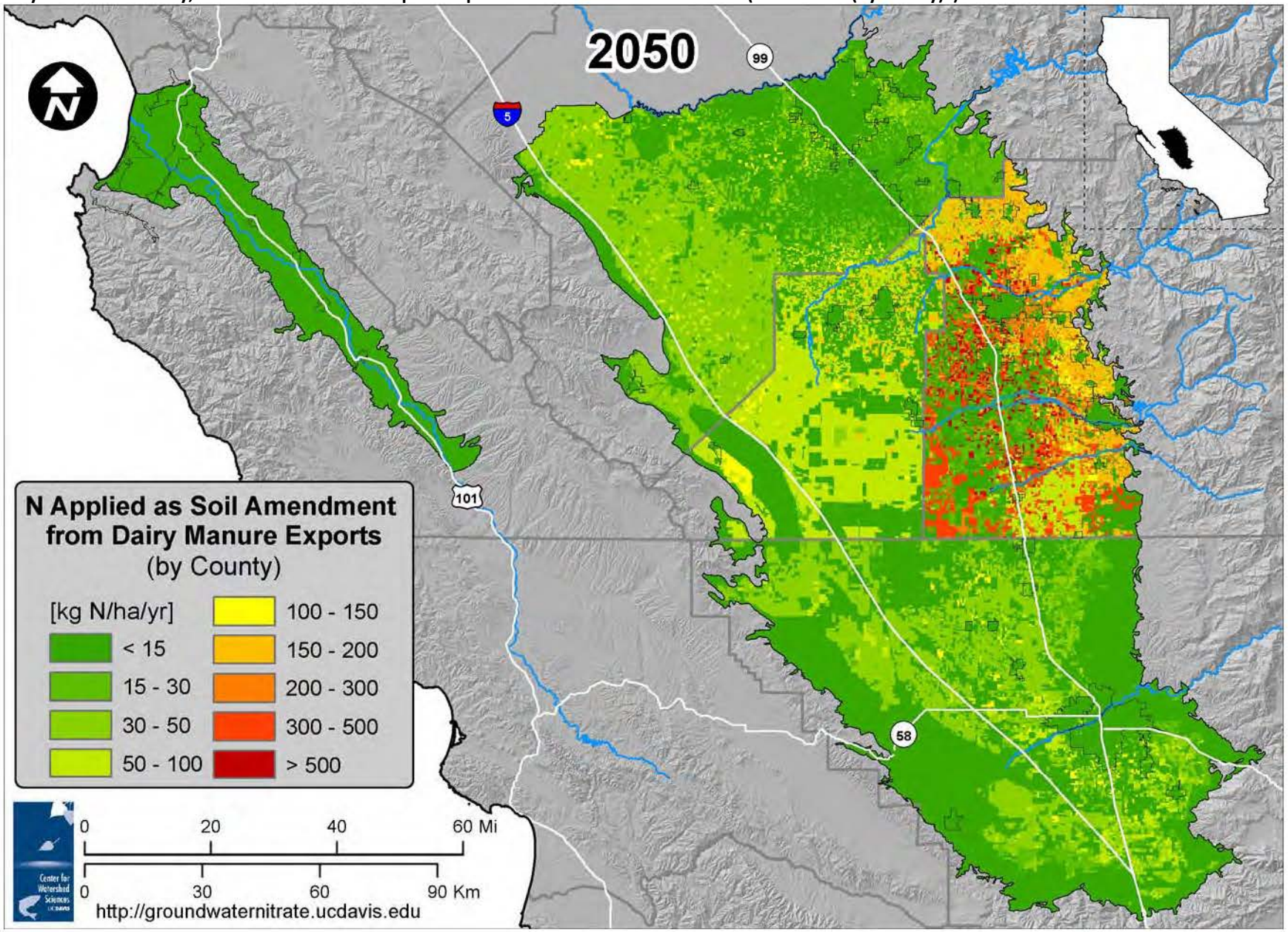
Appendix Figure 53. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the county, in 2020. Total manure exports represent 38% of animal N excreted ("Scenario C (by county)").



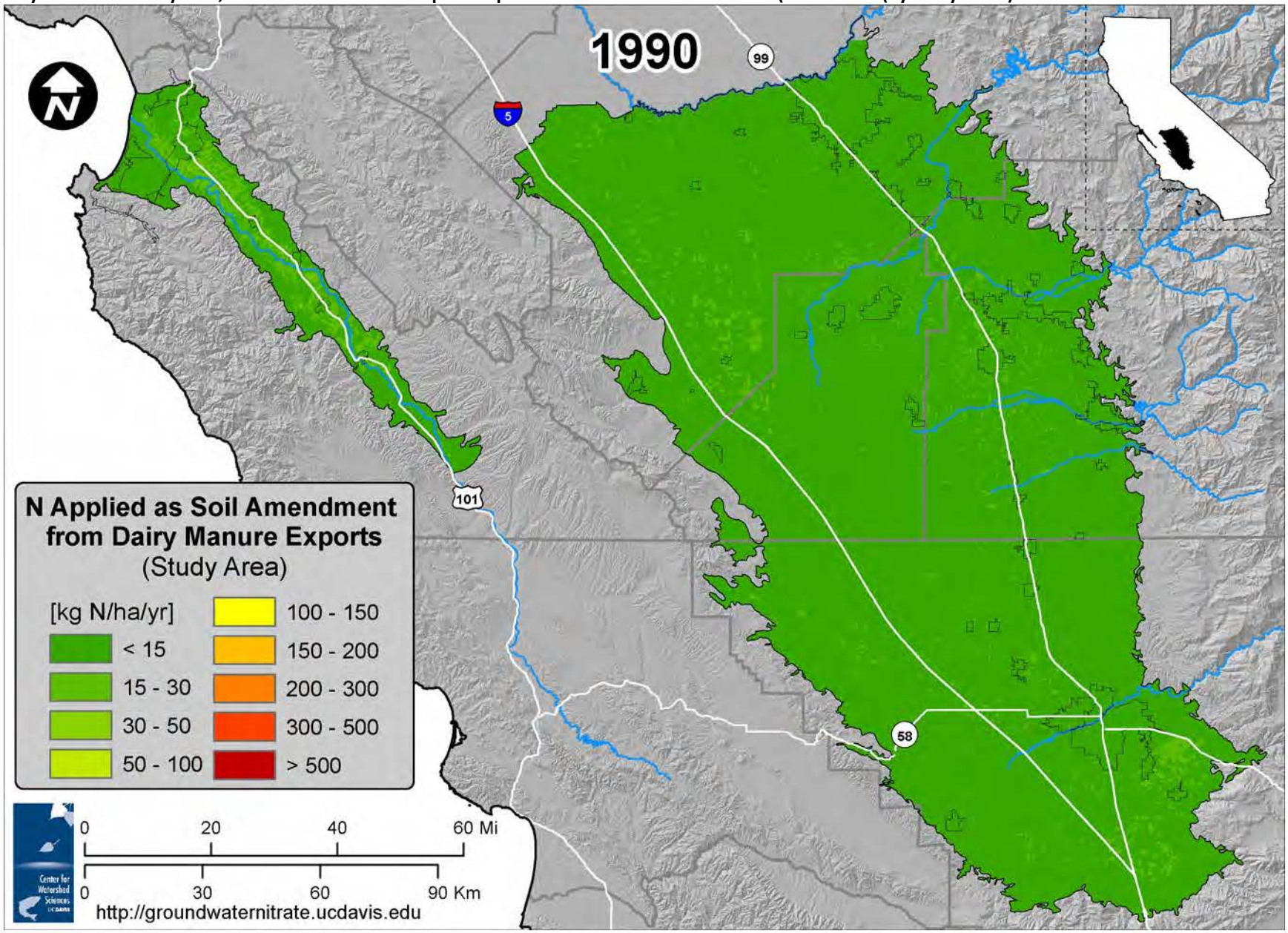
Appendix Figure 54. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the county, in 2035. Total manure exports represent 38% of animal N excreted ("Scenario C (by county)").



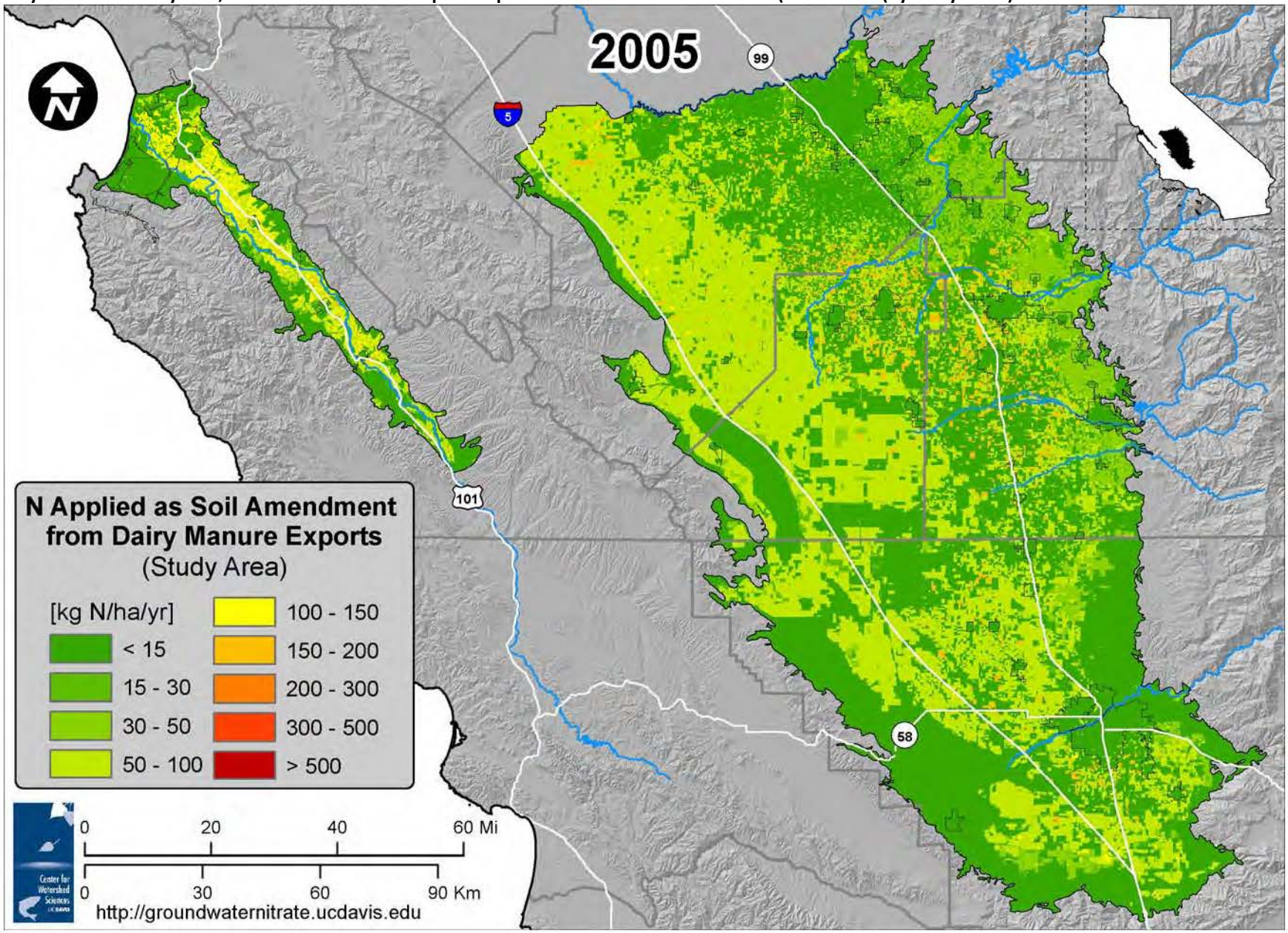
Appendix Figure 55. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the county, in 2050. Total manure exports represent 38% of animal N excreted ("Scenario C (by county)").



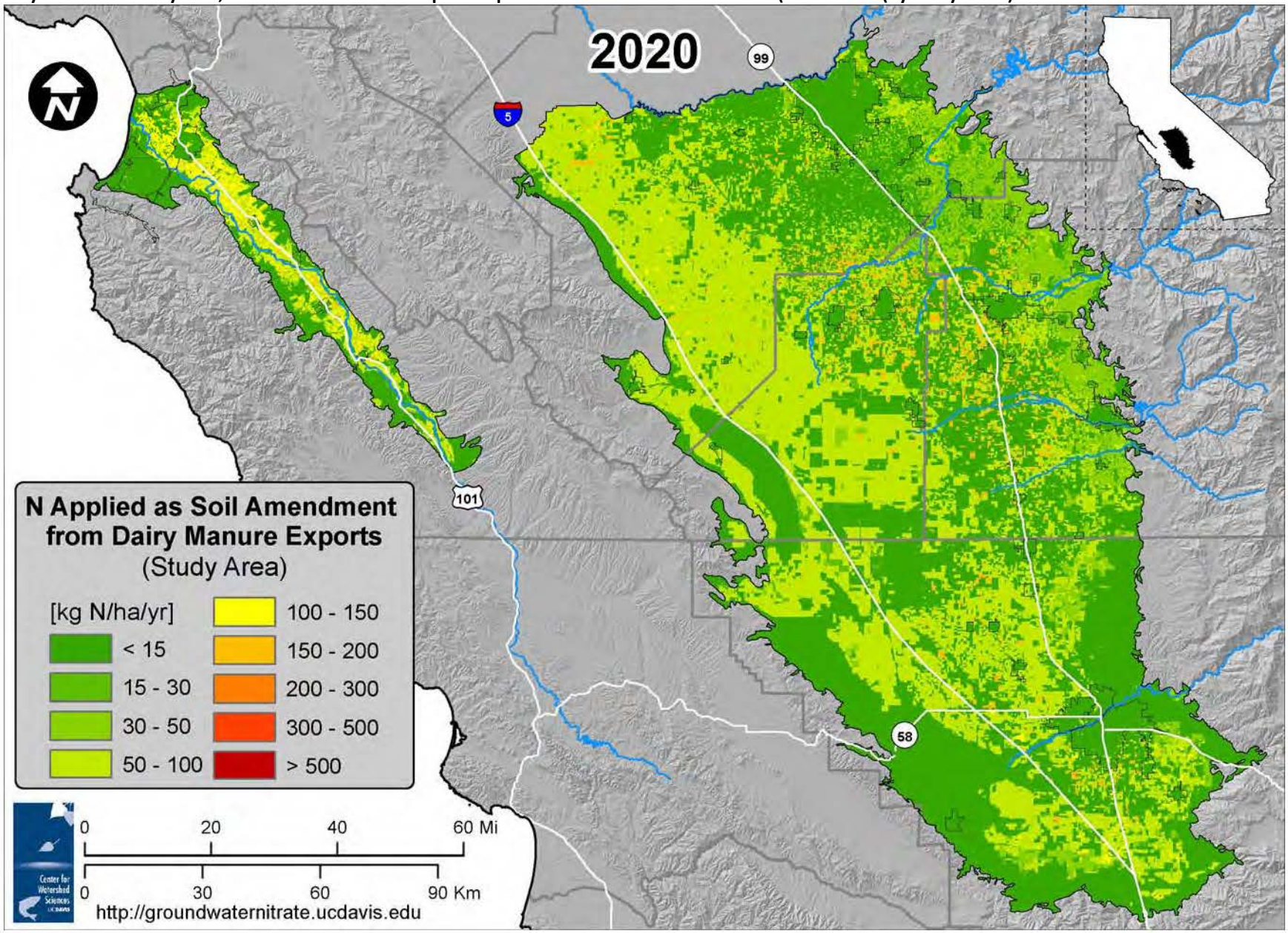
Appendix Figure 56. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the study area, in 1990. Total manure exports represent 38% of animal N excreted ("Scenario C (by study area").



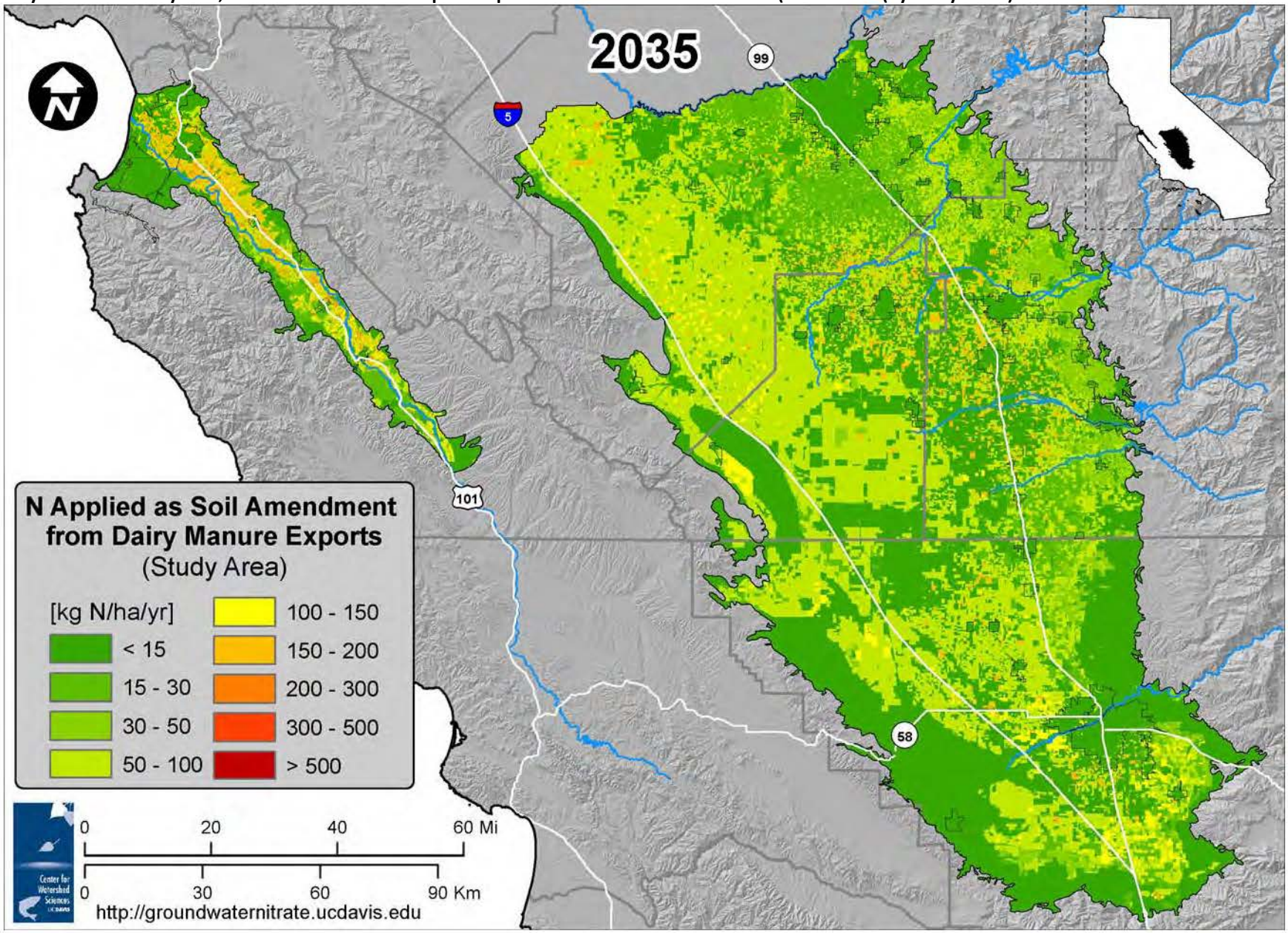
Appendix Figure 57. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the study area, in 2005. Total manure exports represent 38% of animal N excreted ("Scenario C (by study area").



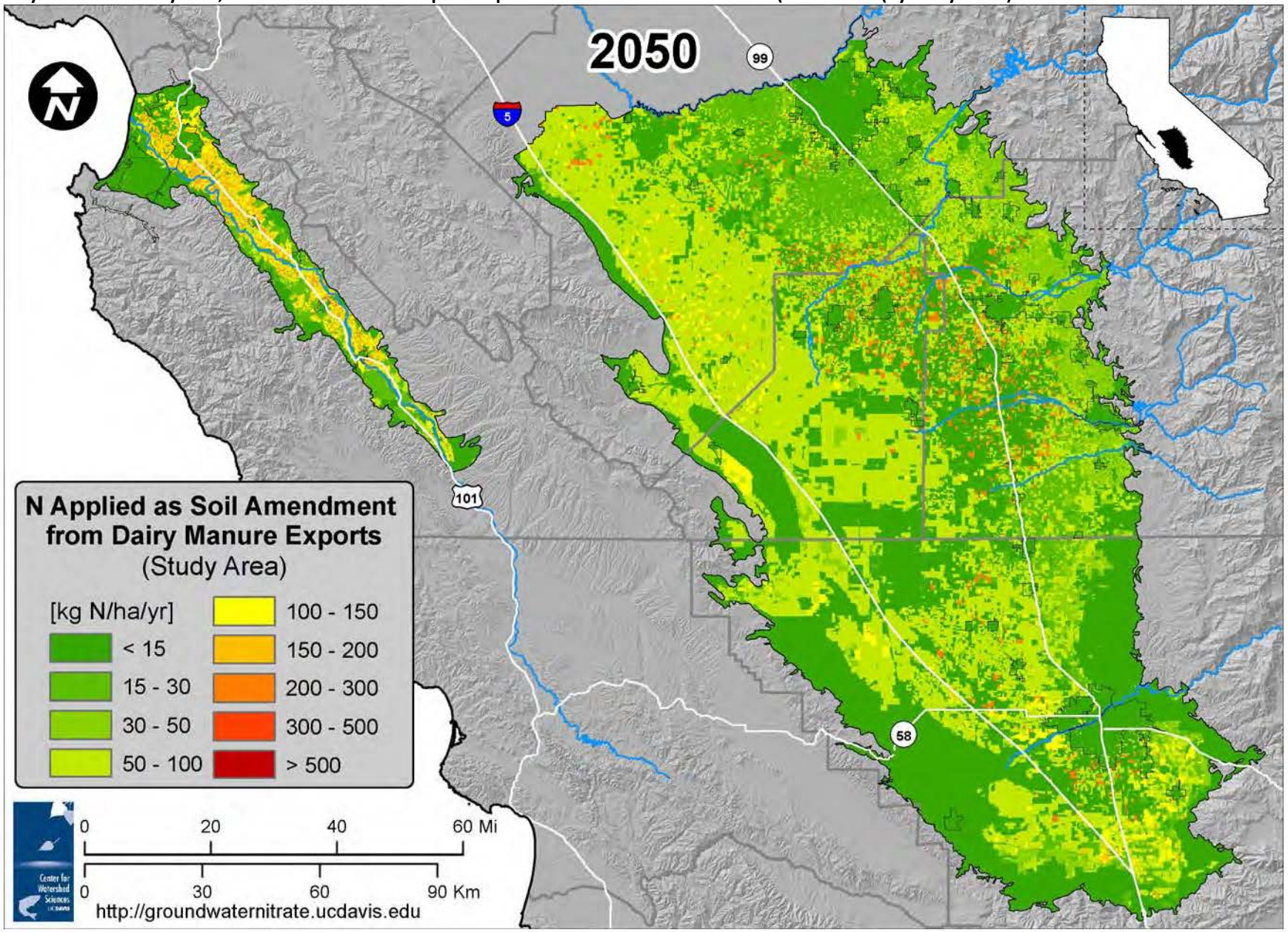
Appendix Figure 58. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the study area, in 2020. Total manure exports represent 38% of animal N excreted ("Scenario C (by study area").



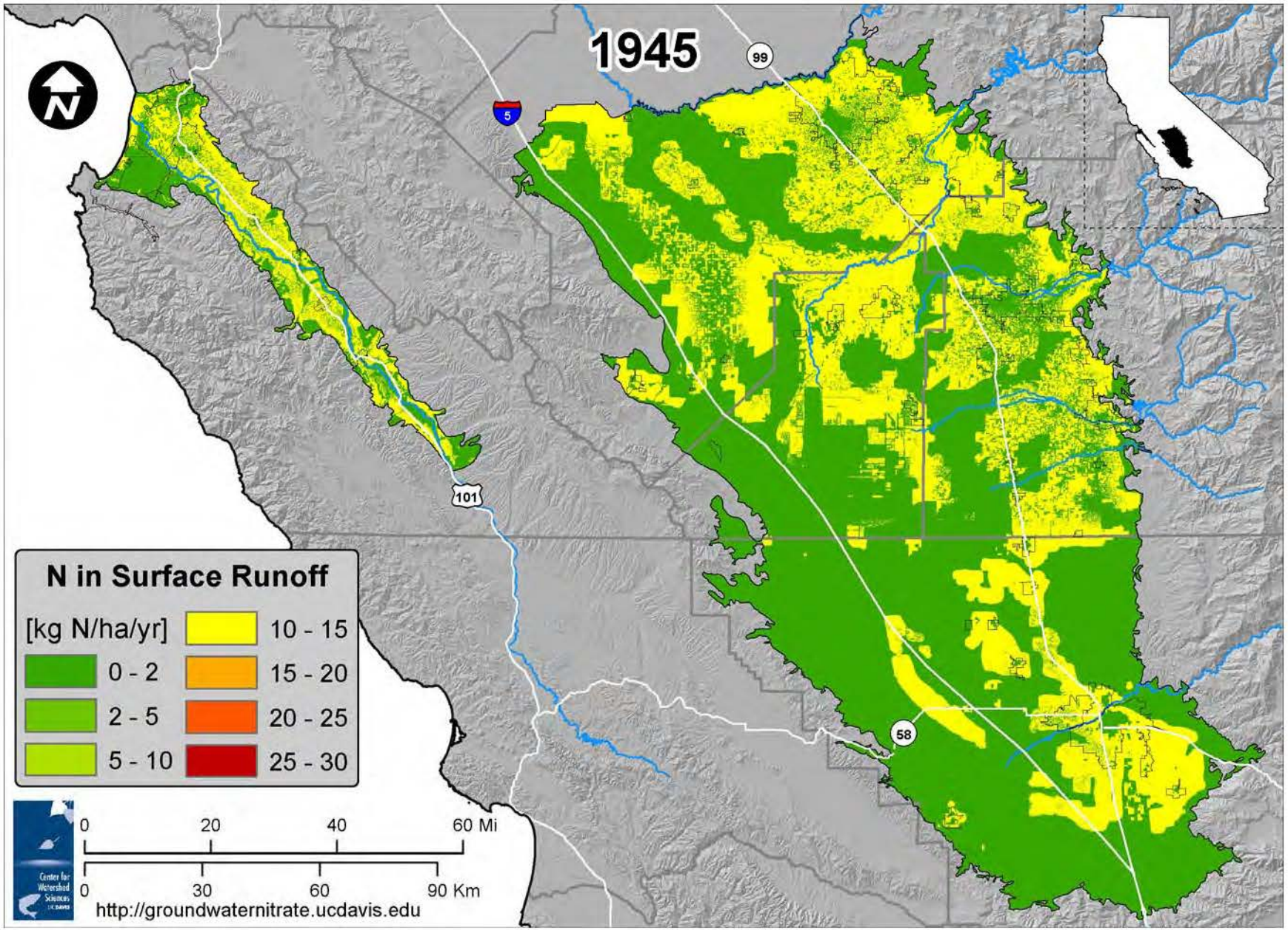
Appendix Figure 59. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the study area, in 2035. Total manure exports represent 38% of animal N excreted ("Scenario C (by study area").



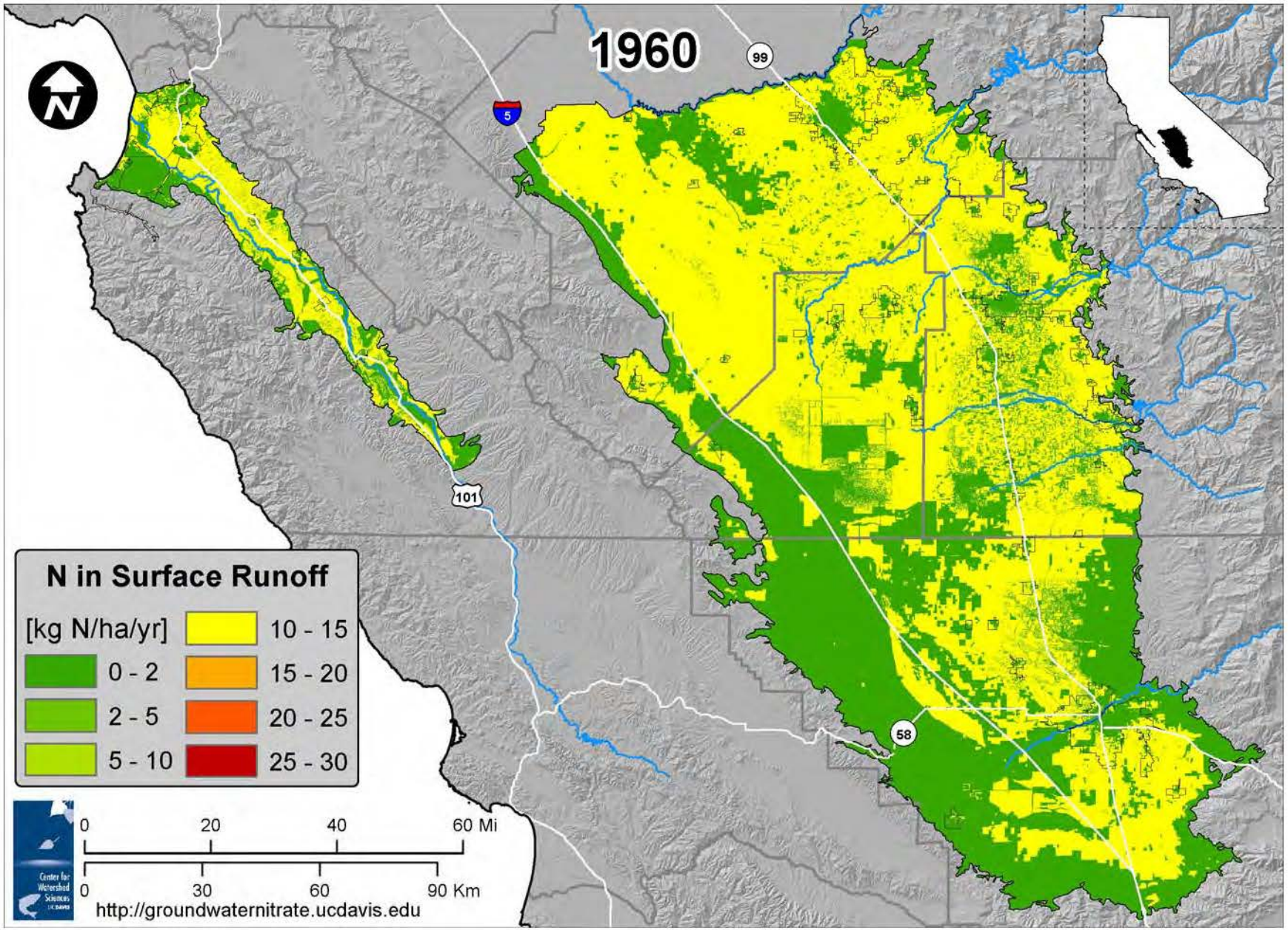
Appendix Figure 60. Land applied, exported dairy manure nitrogen used as soil amendment outside of dairy-controlled cropland, assuming that exports stay within the study area, in 2050. Total manure exports represent 38% of animal N excreted ("Scenario C (by study area").



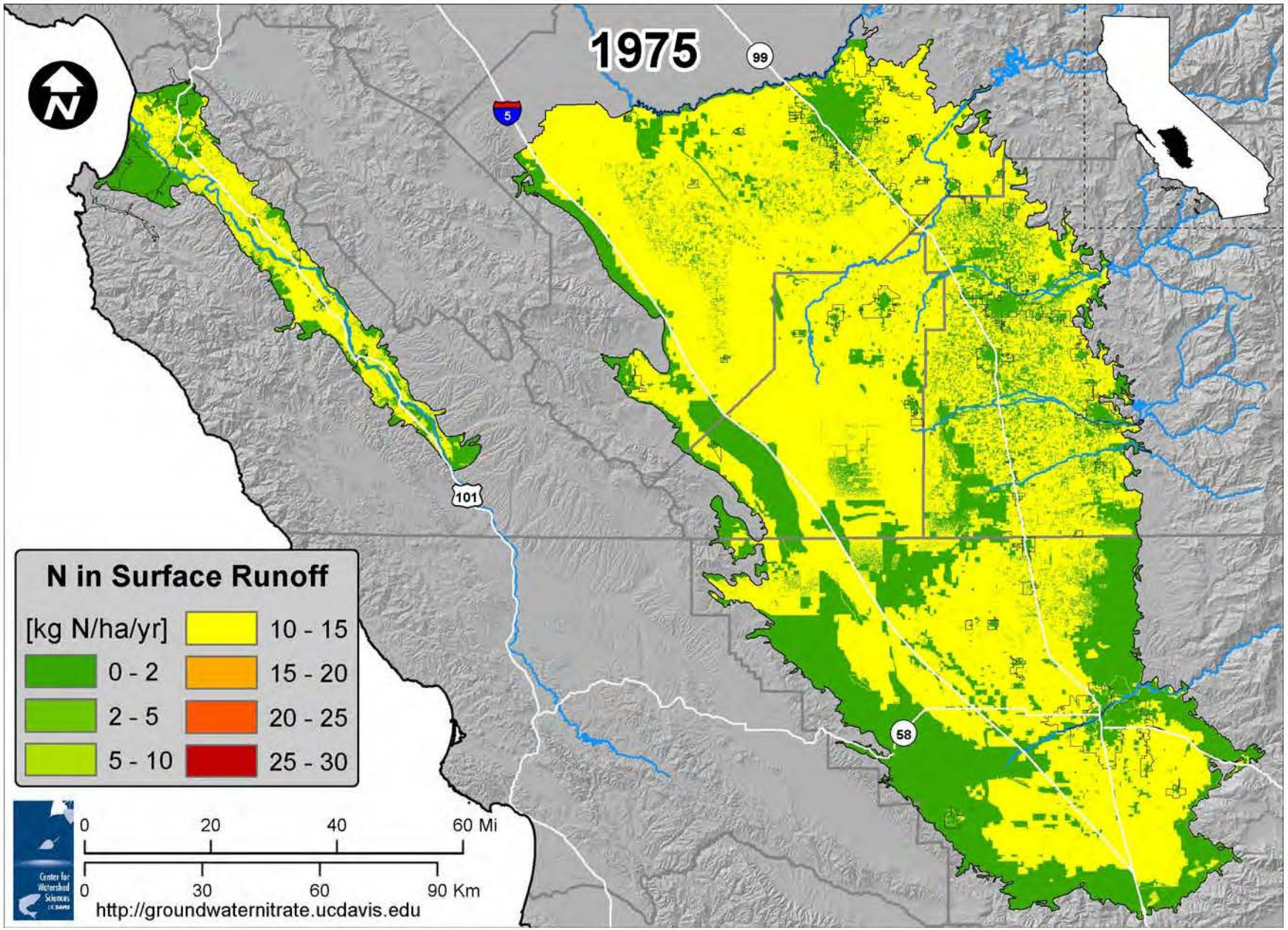
Appendix Figure 61. Nitrogen lost to surface runoff, in 1945.



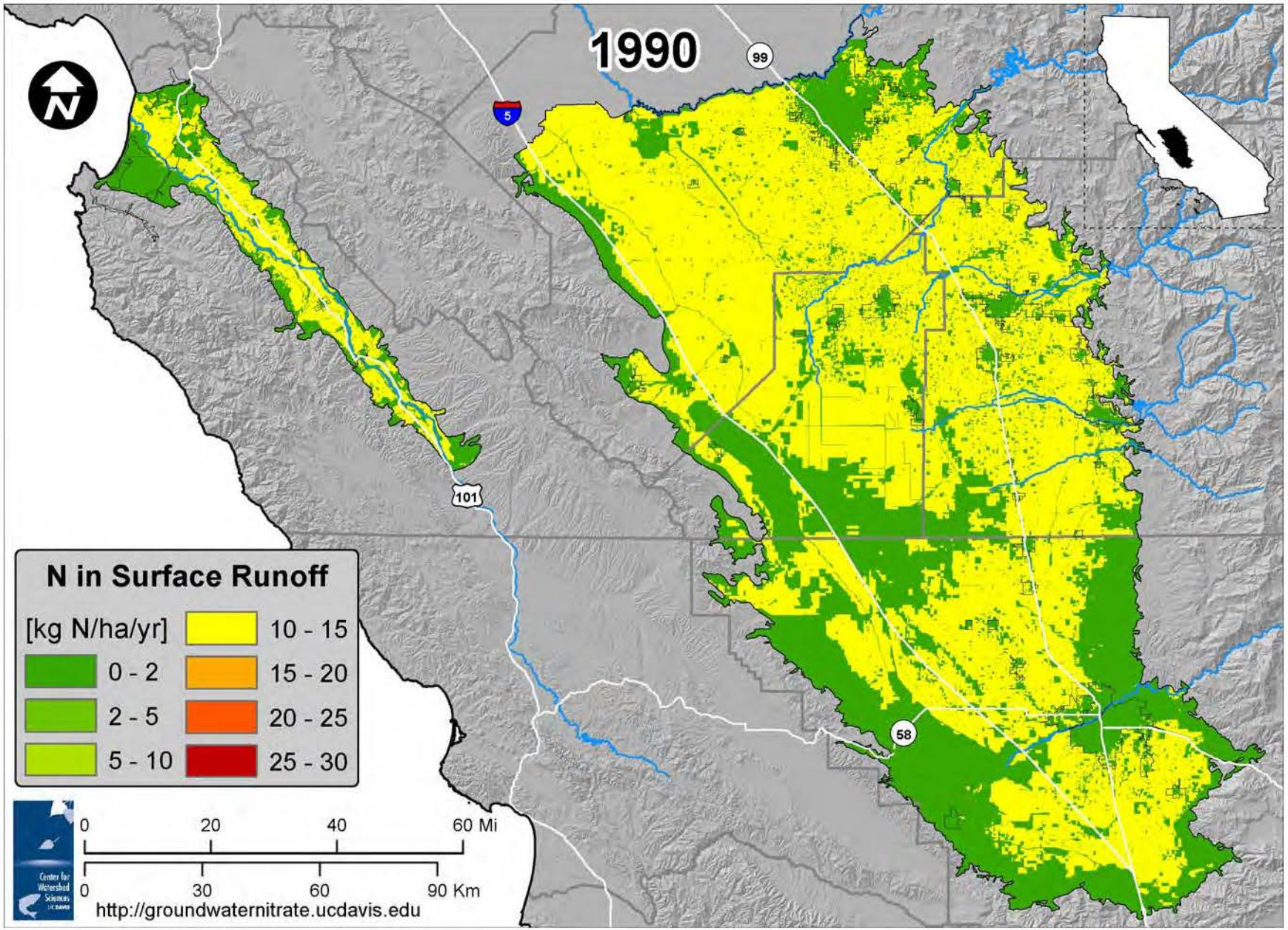
Appendix Figure 62. Nitrogen lost to surface runoff, in 1960.



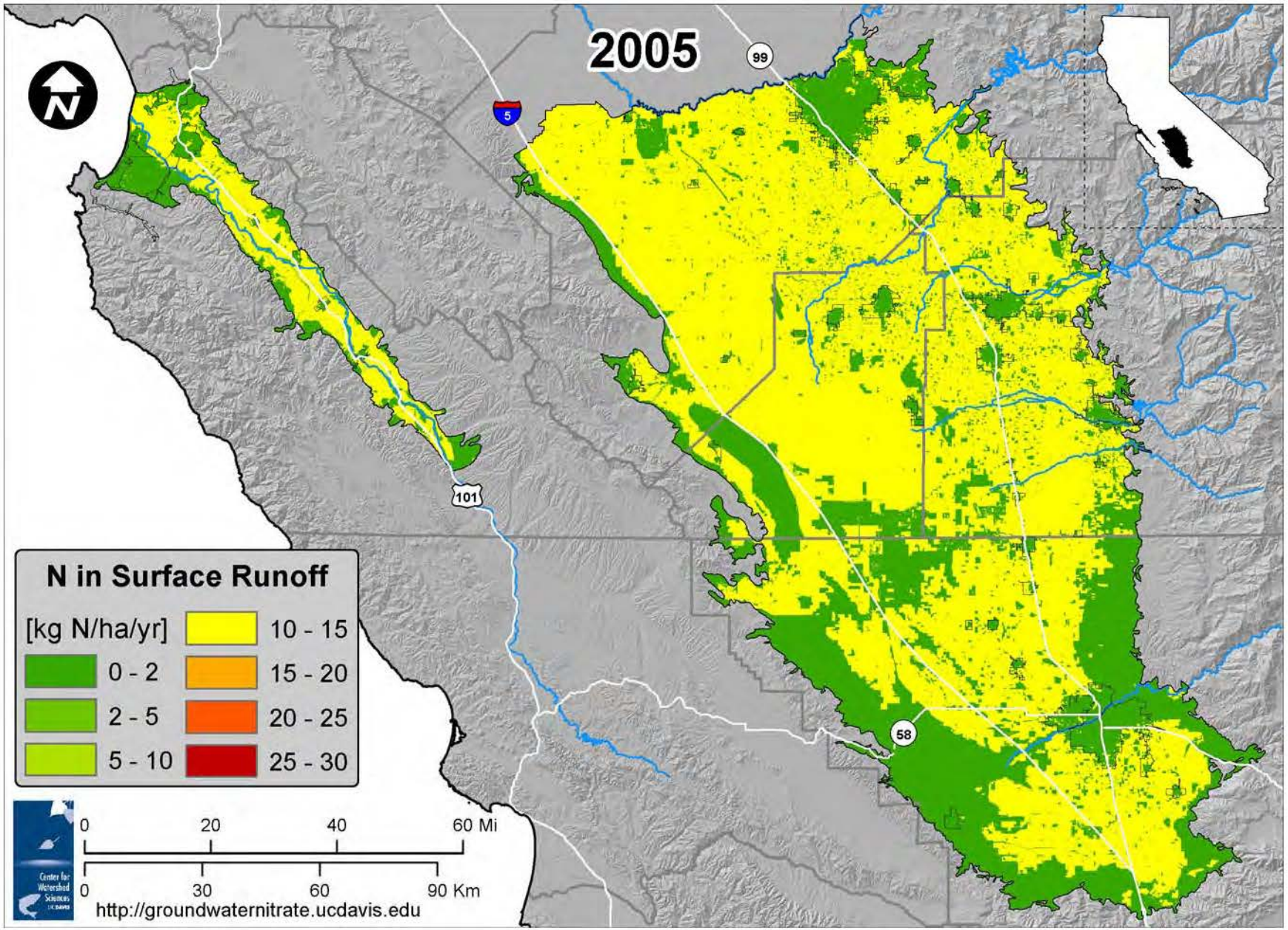
Appendix Figure 63. Nitrogen lost to surface runoff, in 1975.



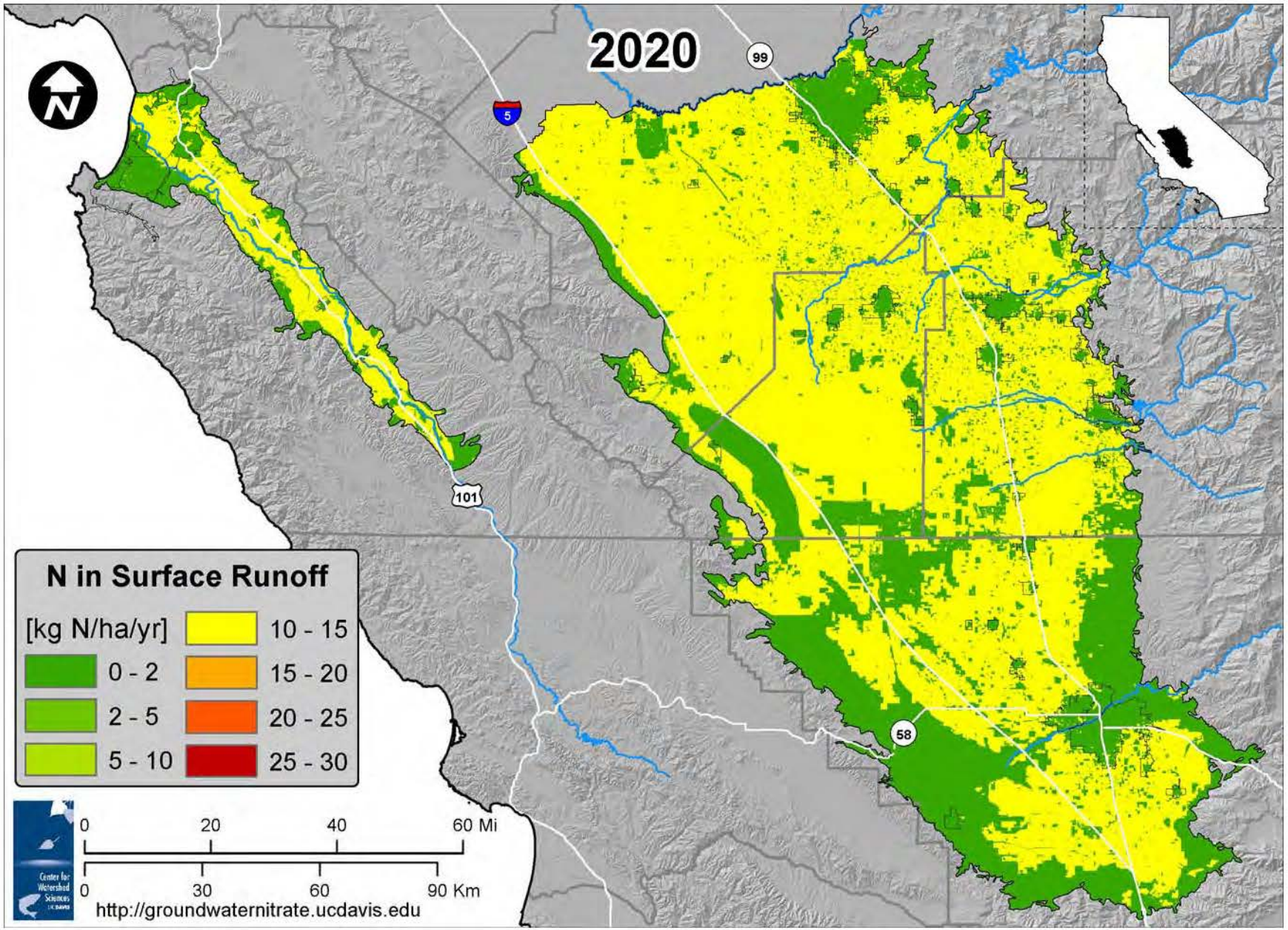
Appendix Figure 64. Nitrogen lost to surface runoff, in 1990.



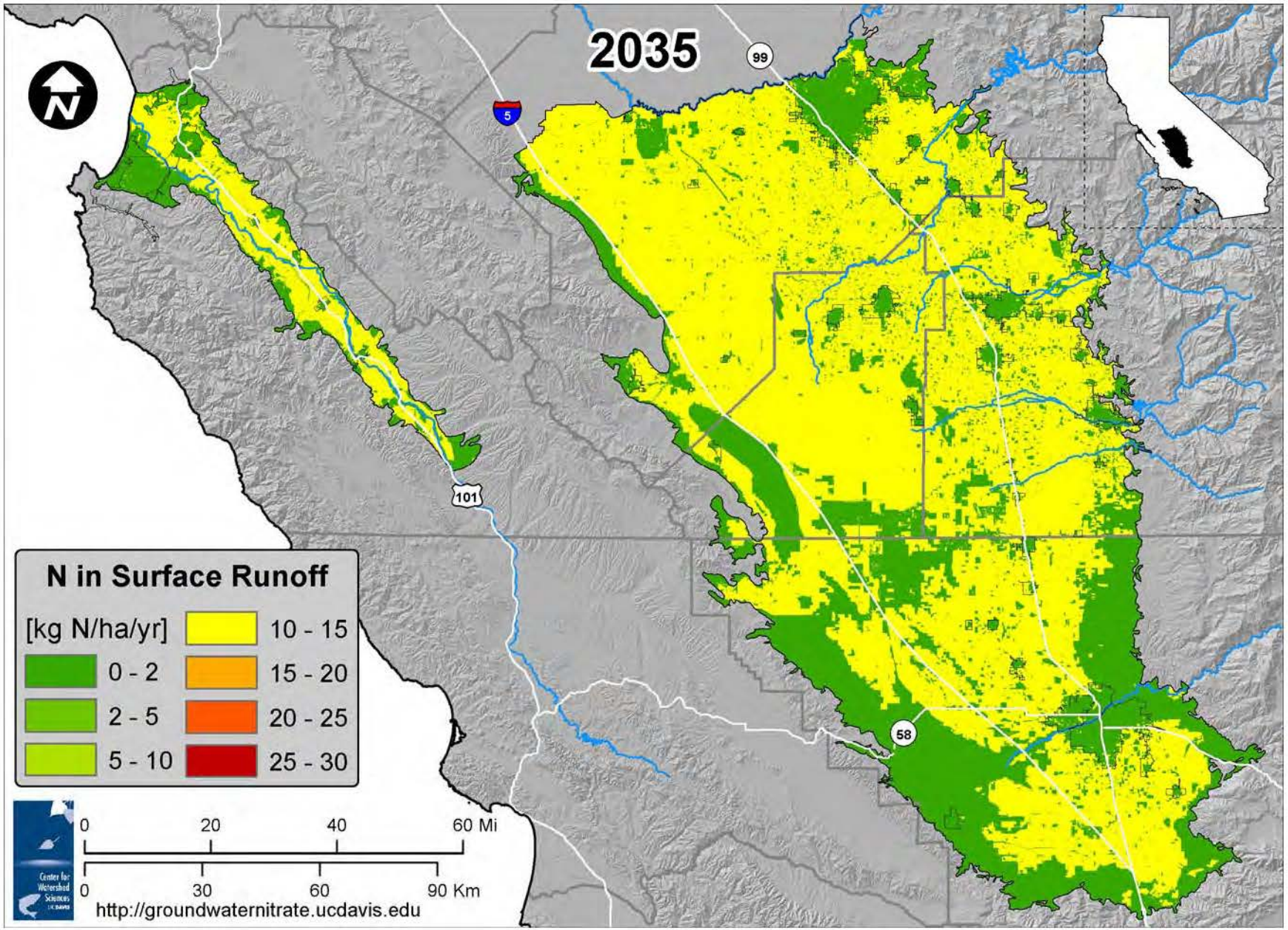
Appendix Figure 65. Nitrogen lost to surface runoff, in 2005.



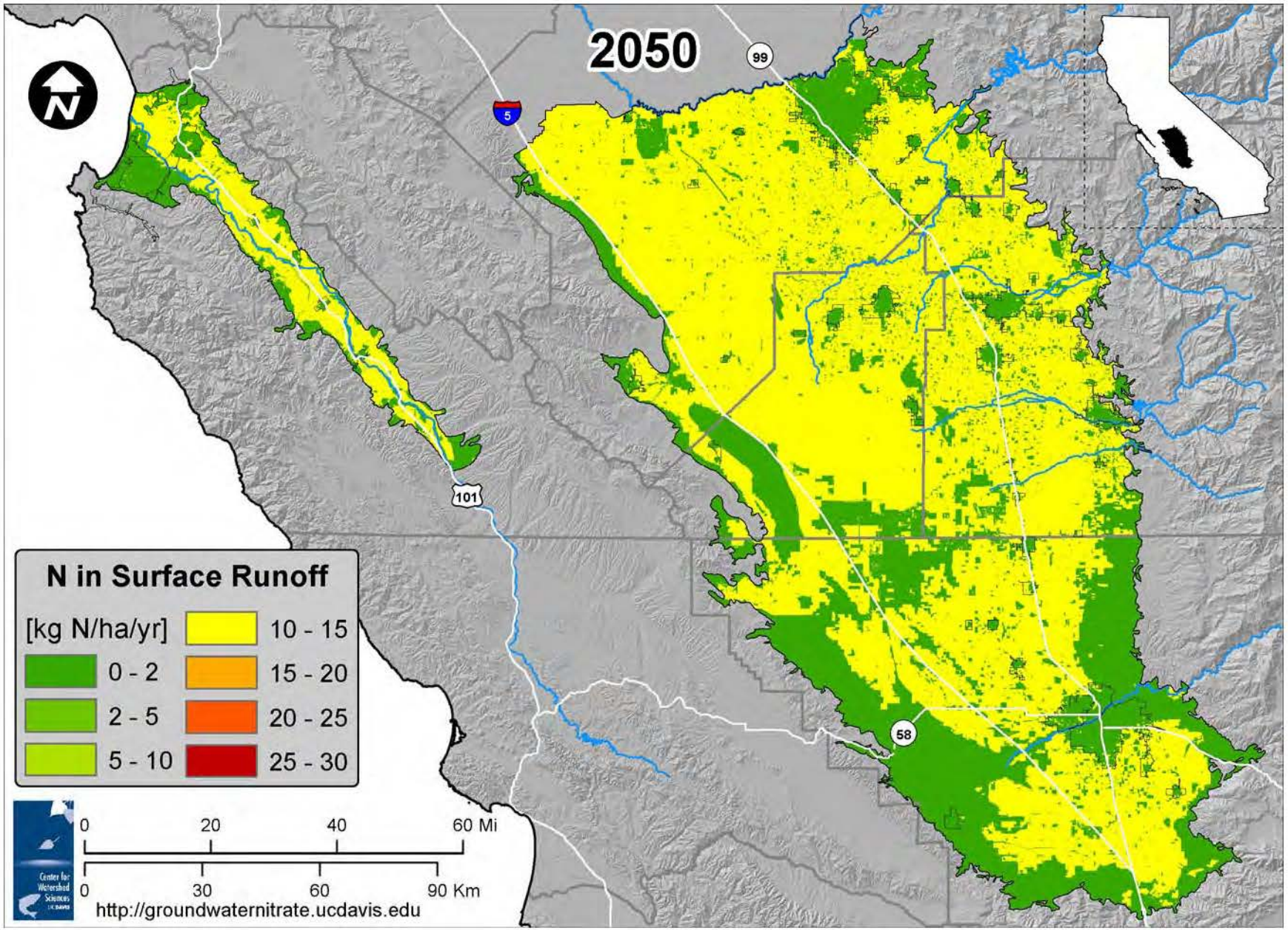
Appendix Figure 66. Nitrogen lost to surface runoff, in 2020.



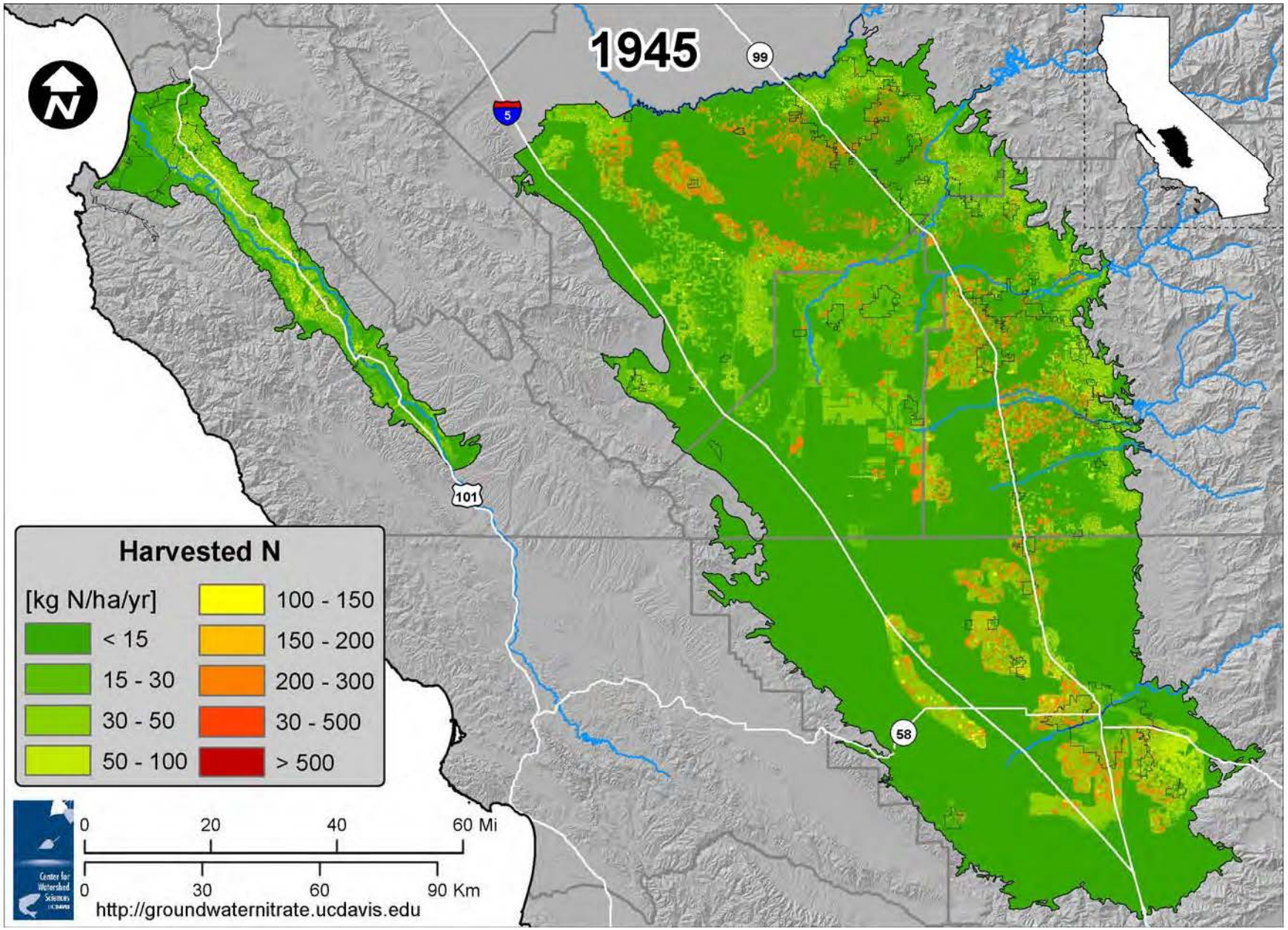
Appendix Figure 67. Nitrogen lost to surface runoff, in 2035.



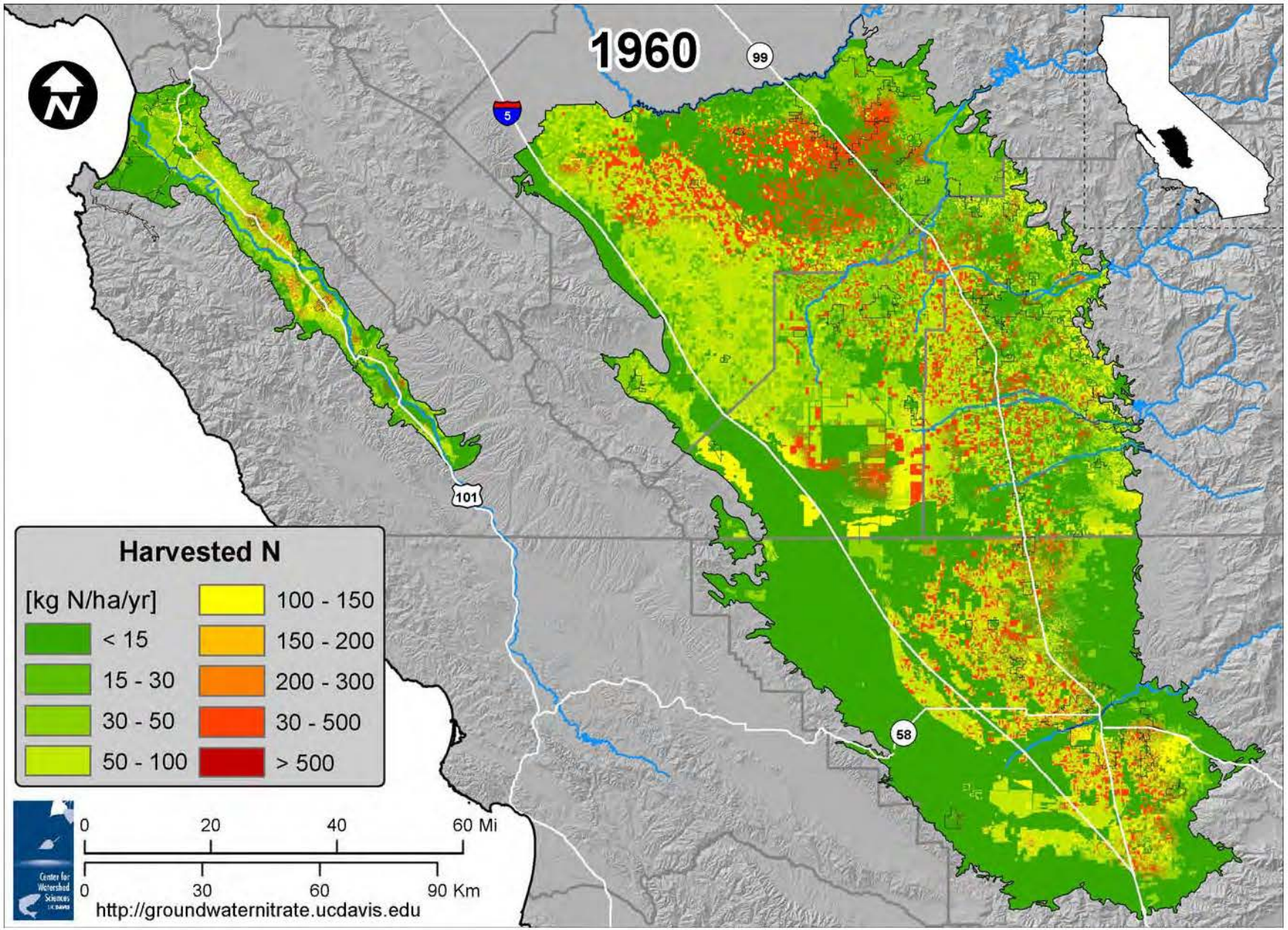
Appendix Figure 68. Nitrogen lost to surface runoff, in 2050.



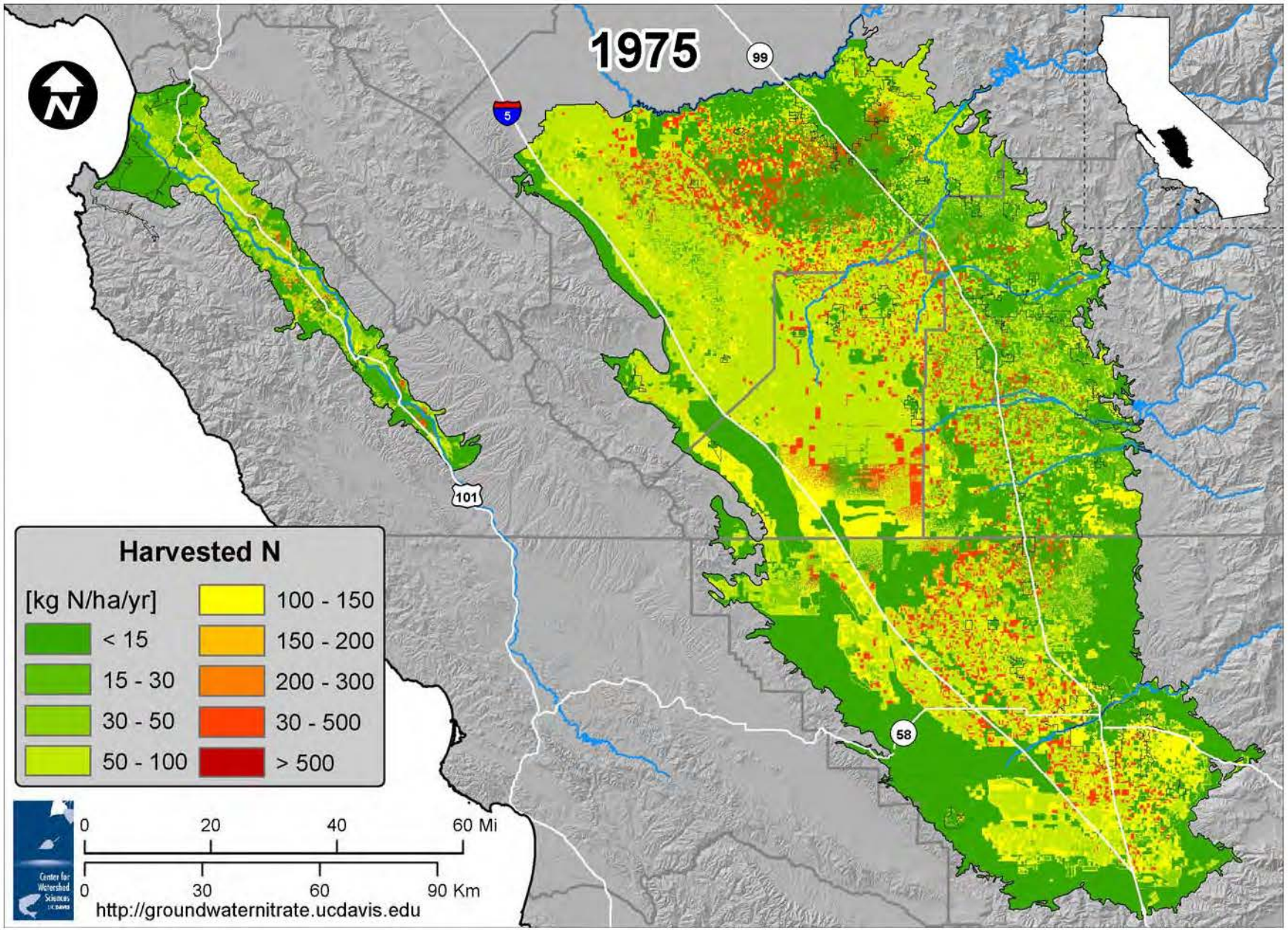
Appendix Figure 69. Nitrogen harvested, in 1945.



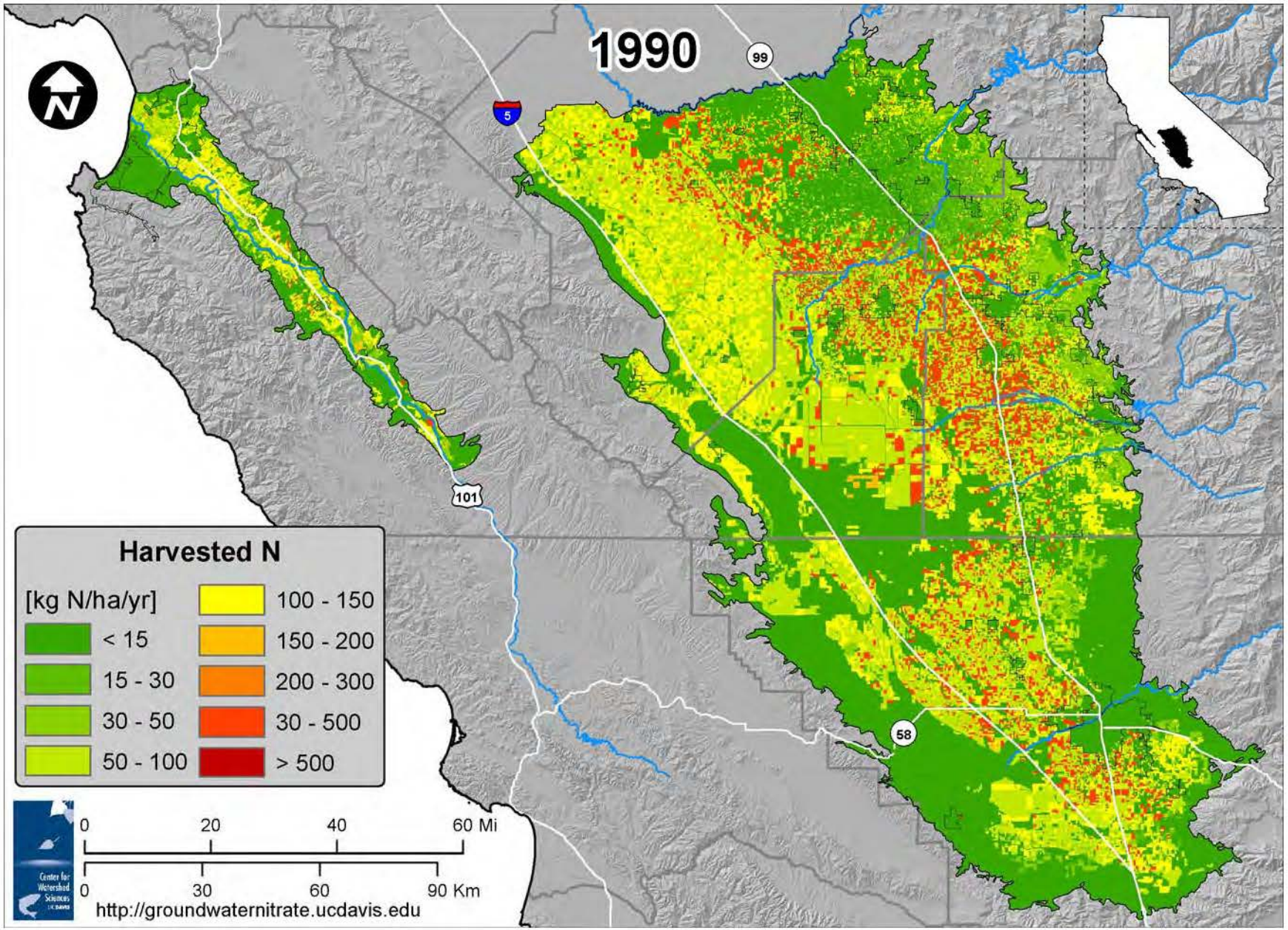
Appendix Figure 70. Nitrogen harvested, in 1960.



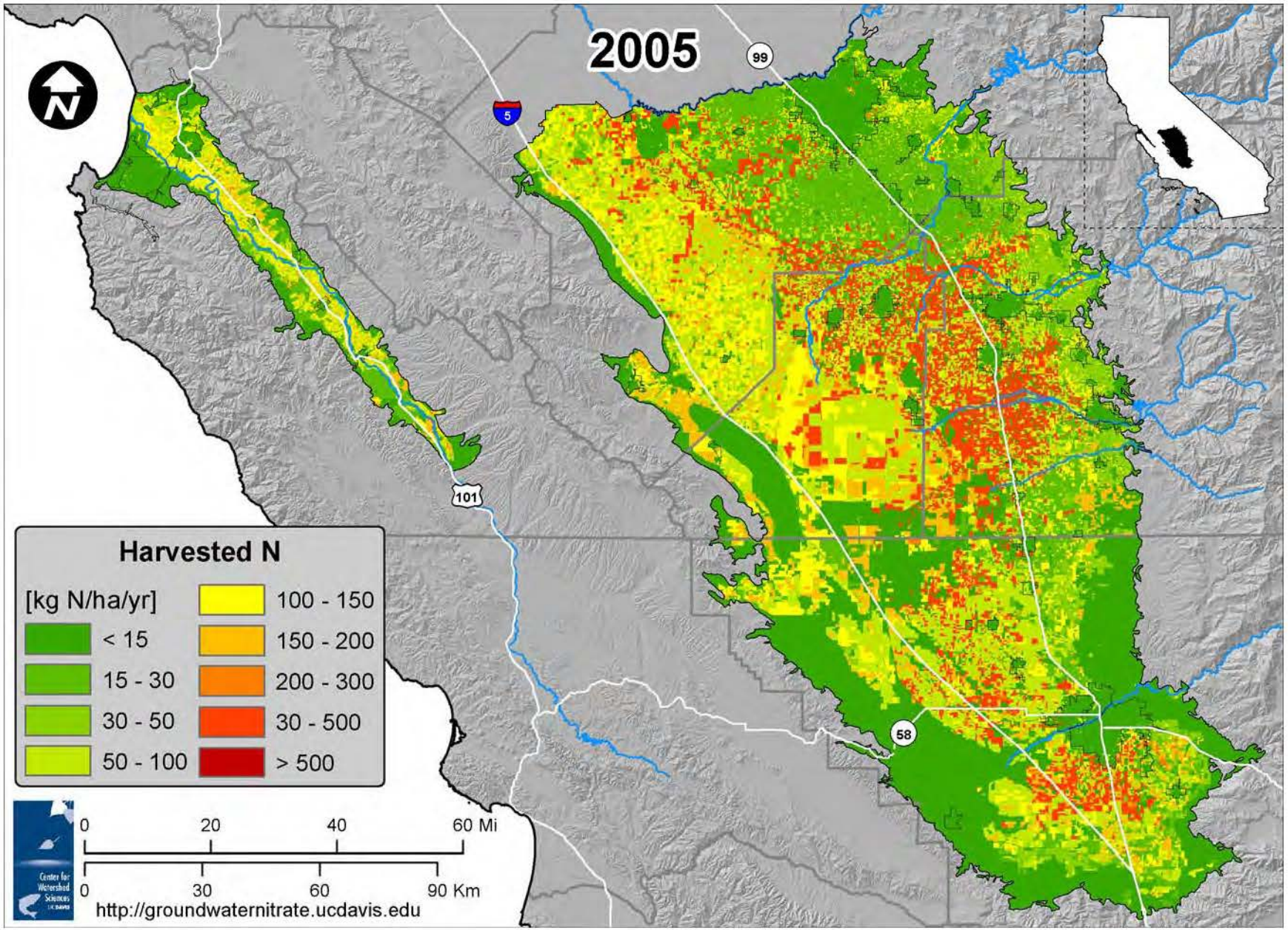
Appendix Figure 71. Nitrogen harvested, in 1975.



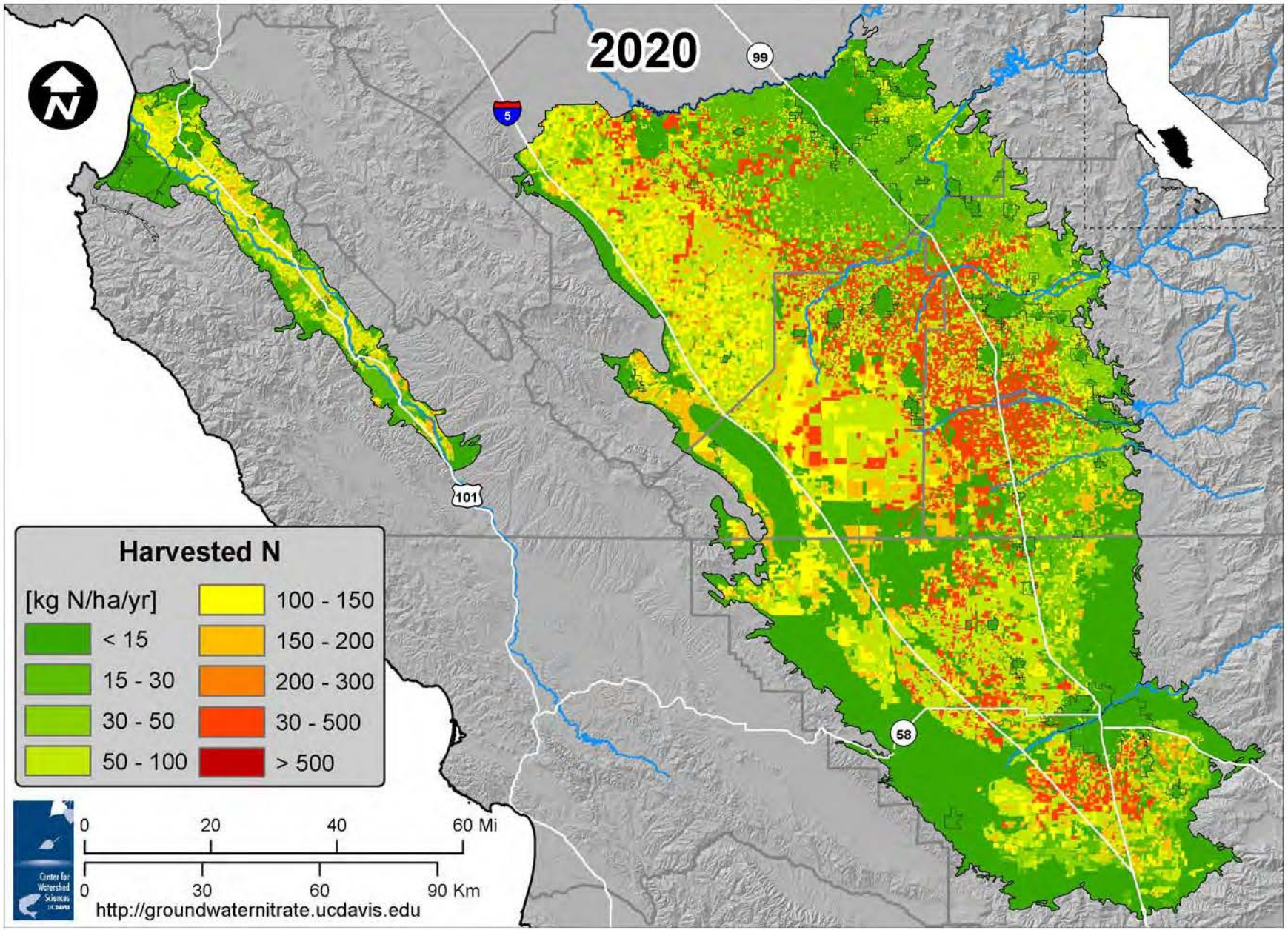
Appendix Figure 72. Nitrogen harvested, in 1990.



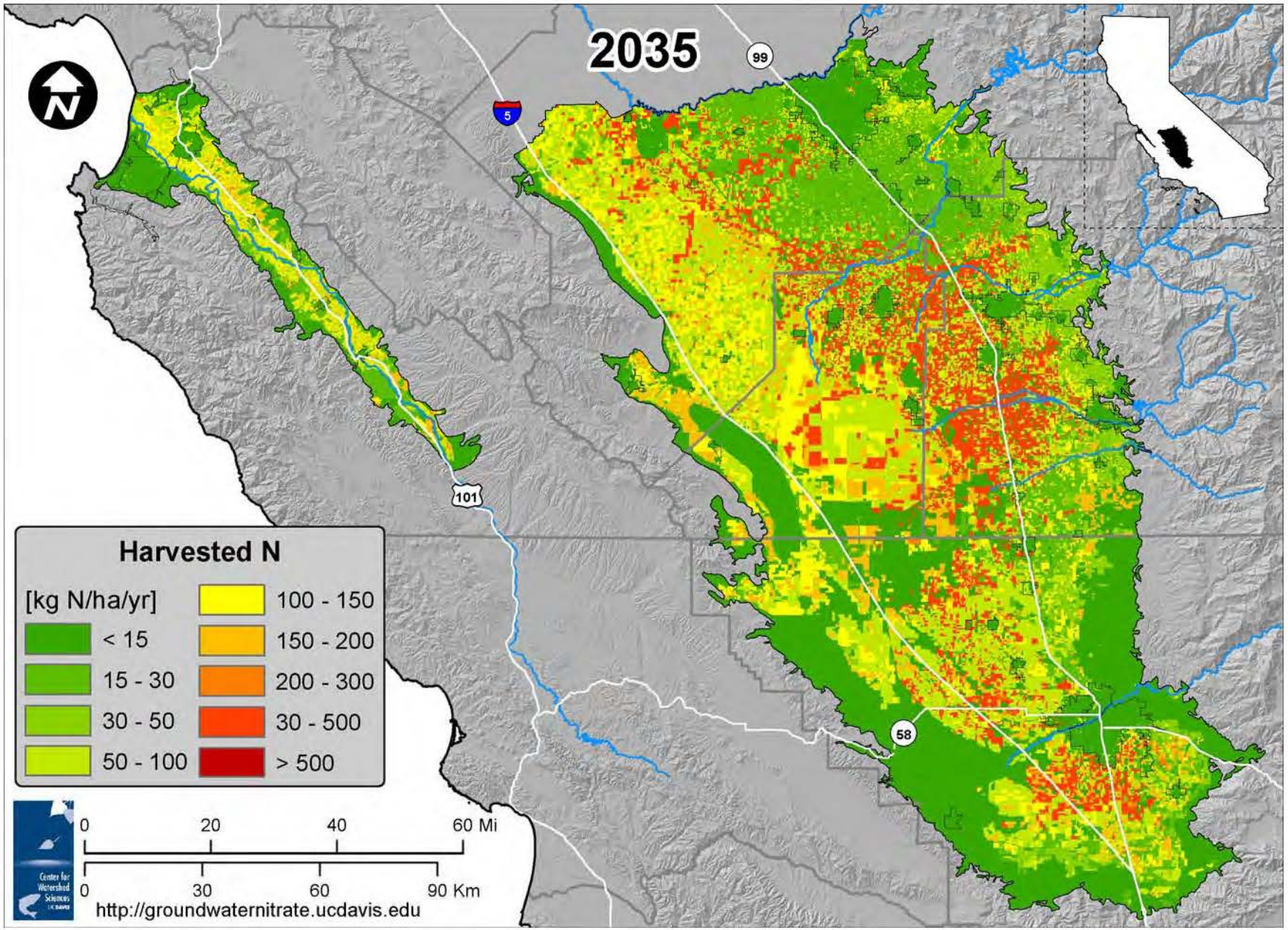
Appendix Figure 73. Nitrogen harvested, in 2005.



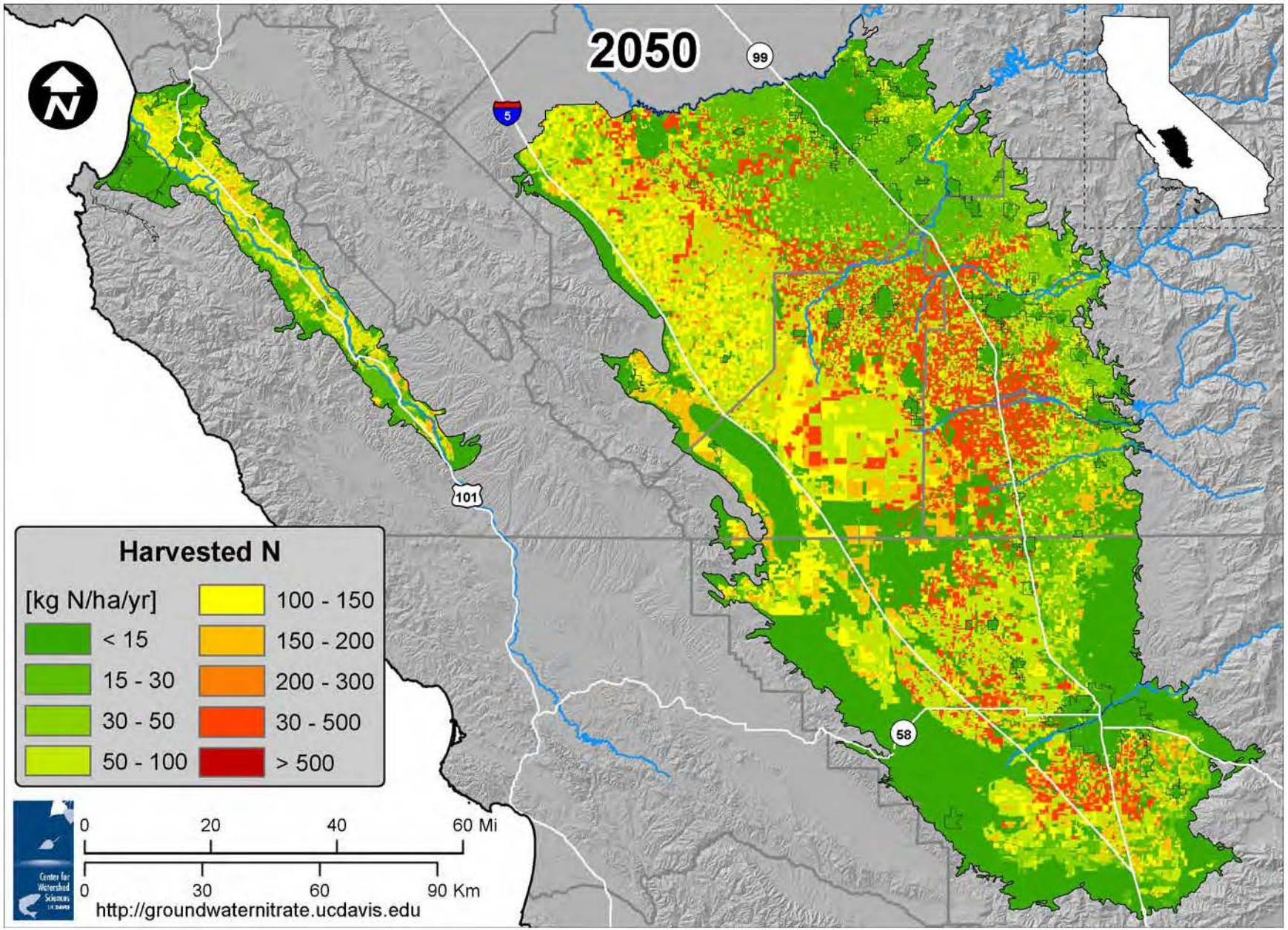
Appendix Figure 74. Nitrogen harvested, in 2020.



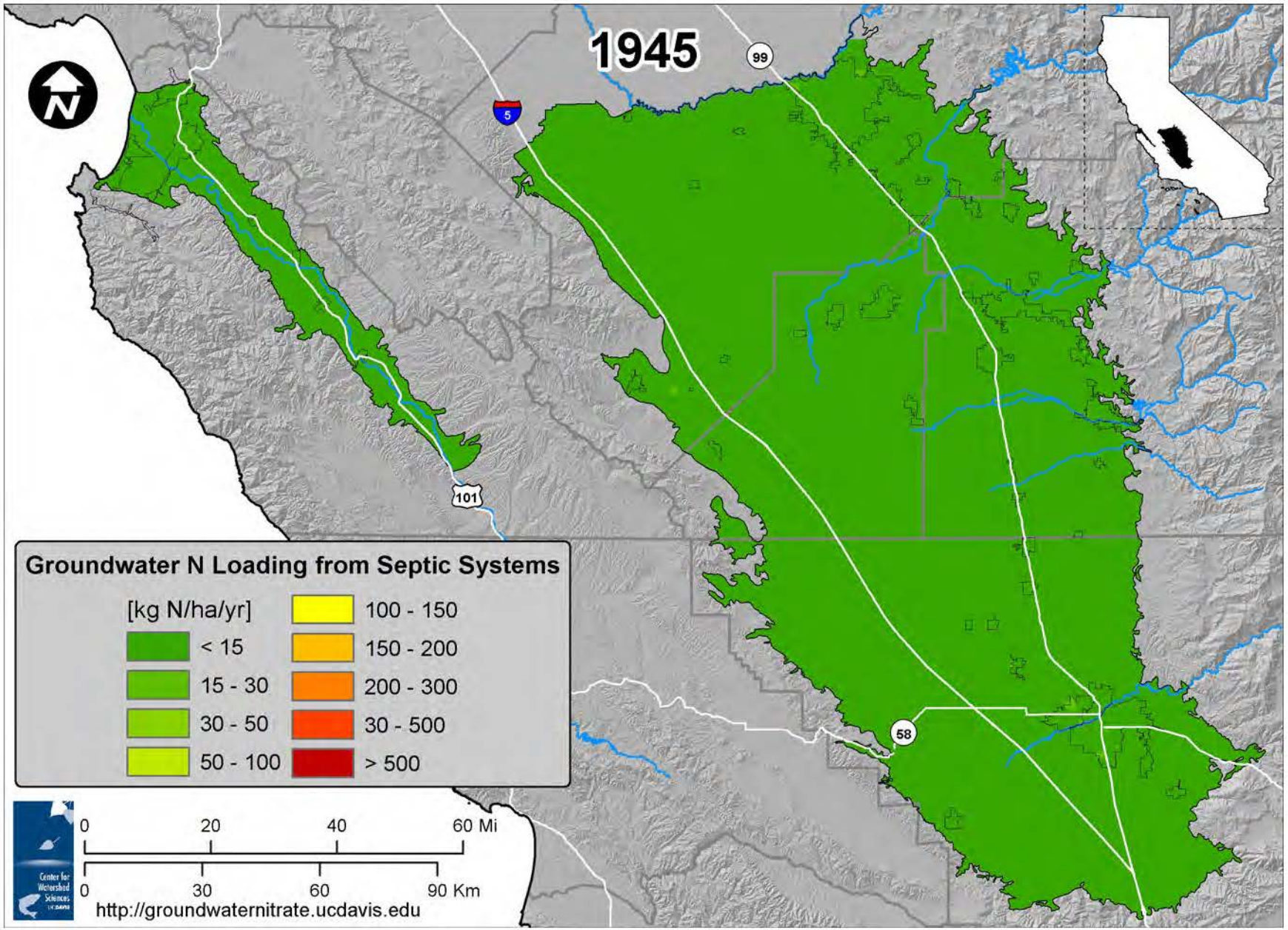
Appendix Figure 75. Nitrogen harvested, in 2035.



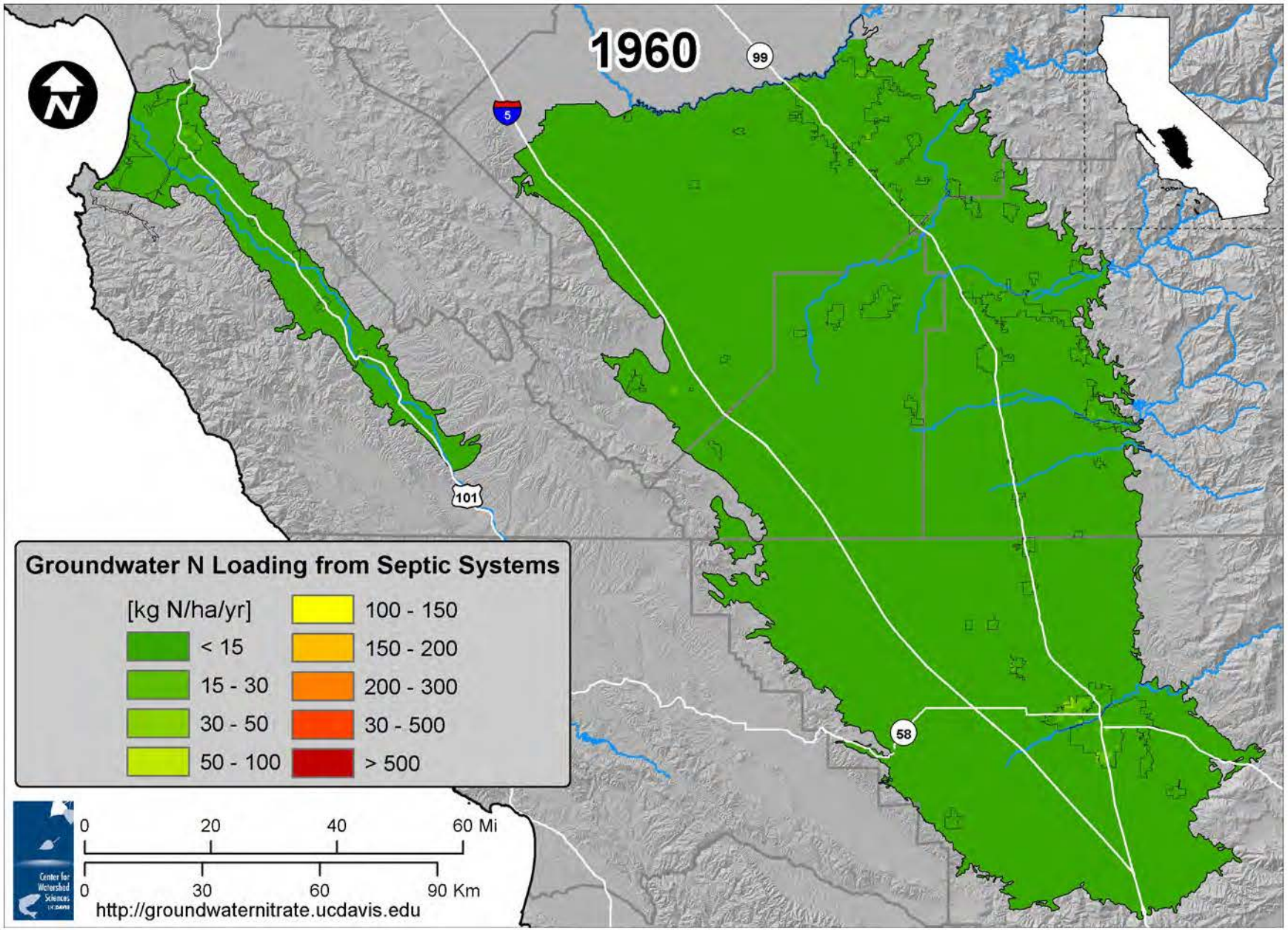
Appendix Figure 76. Nitrogen harvested, in 2050.



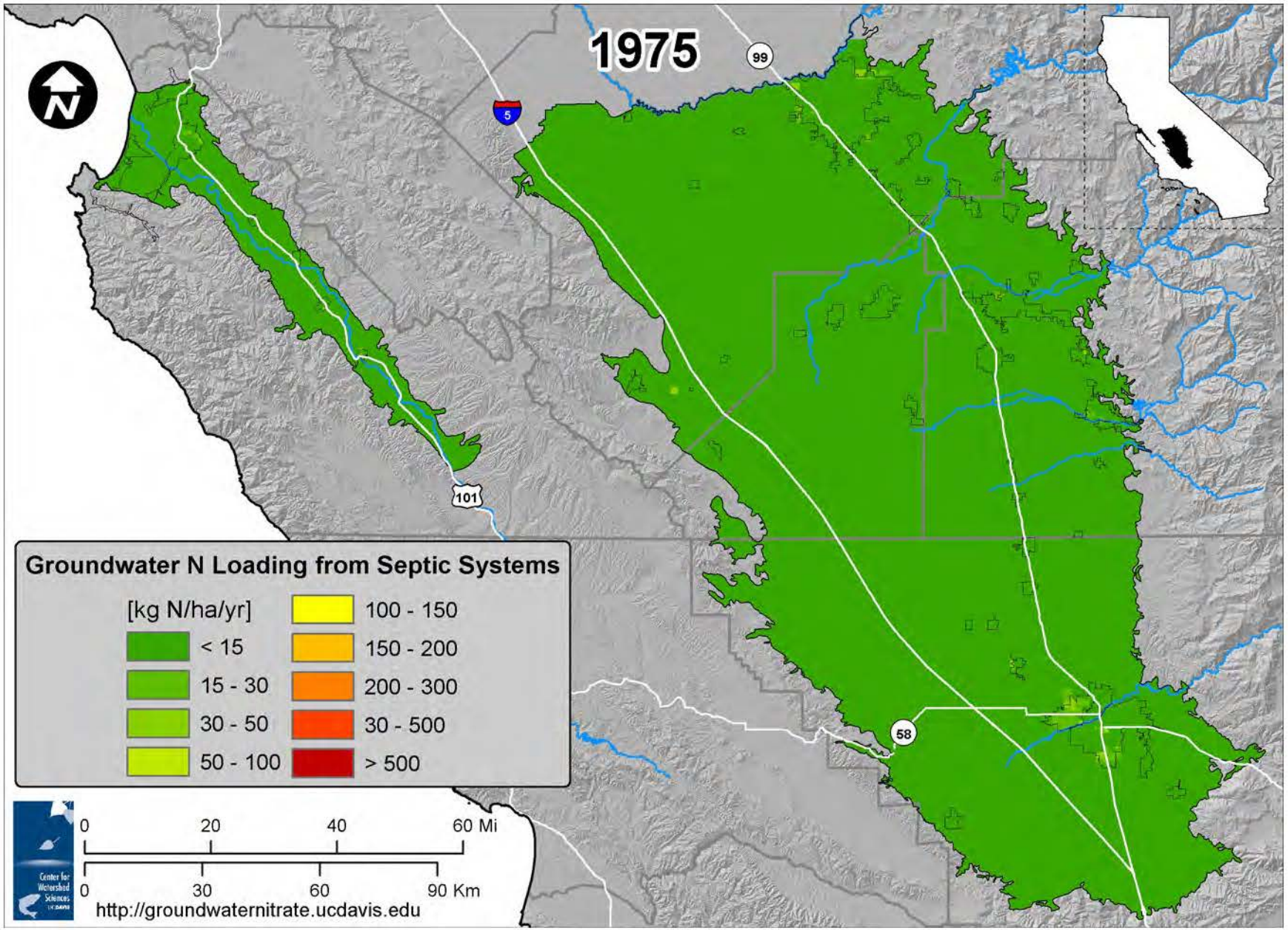
Appendix Figure 77. Nitrogen leached from septic systems, in 1945.



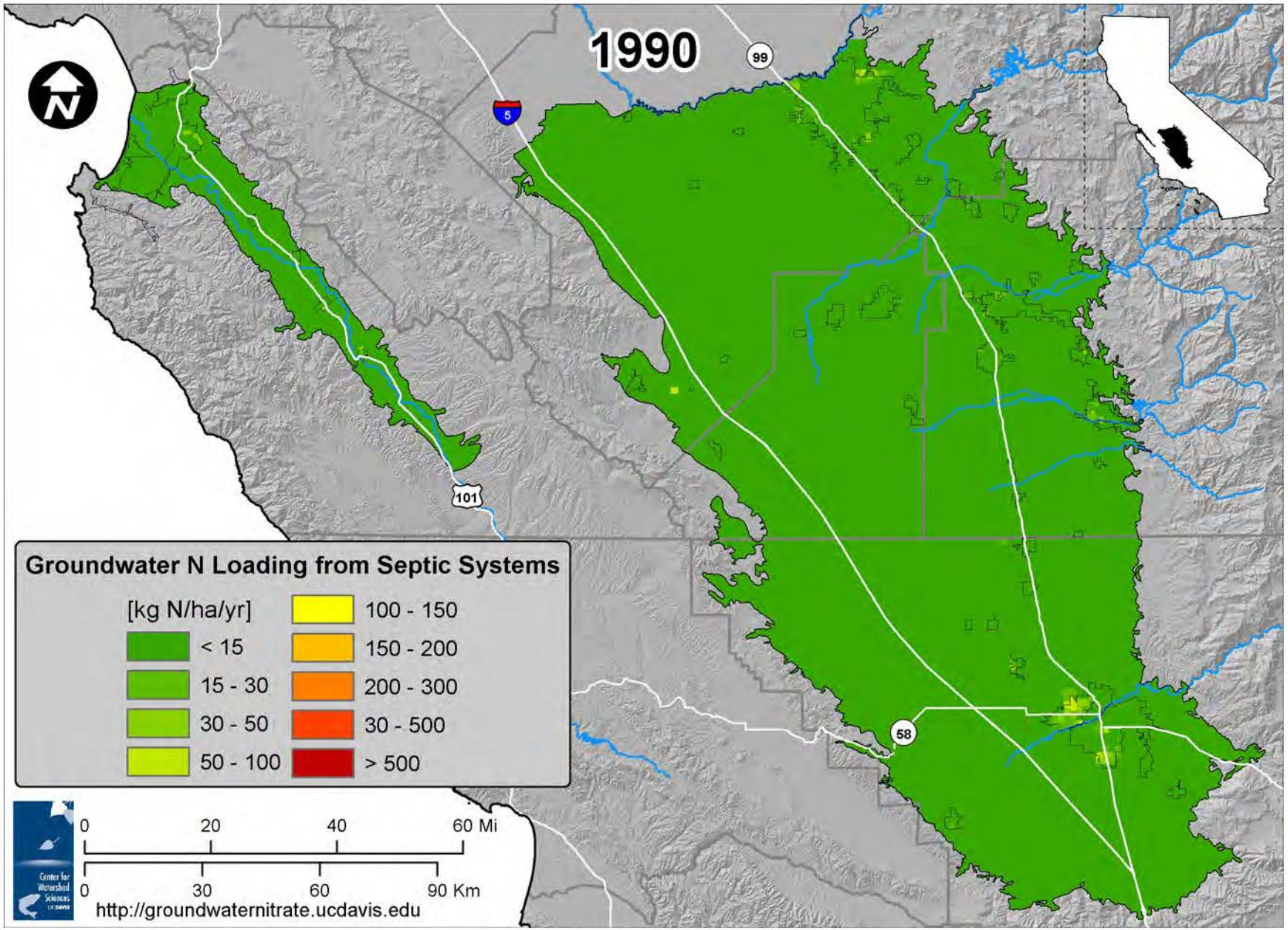
Appendix Figure 78. Nitrogen leached from septic systems, in 1960.



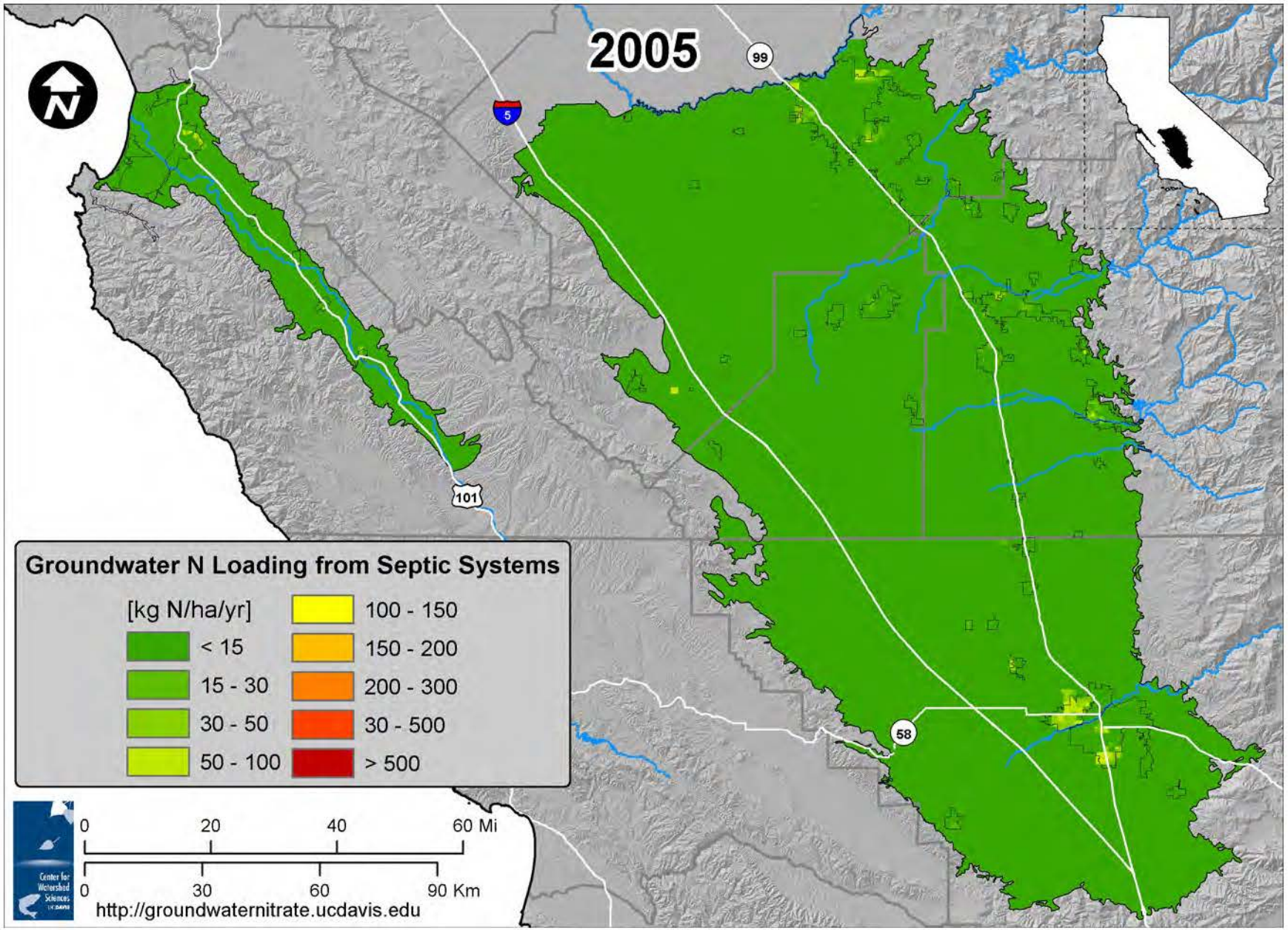
Appendix Figure 79. Nitrogen leached from septic systems, in 1975.



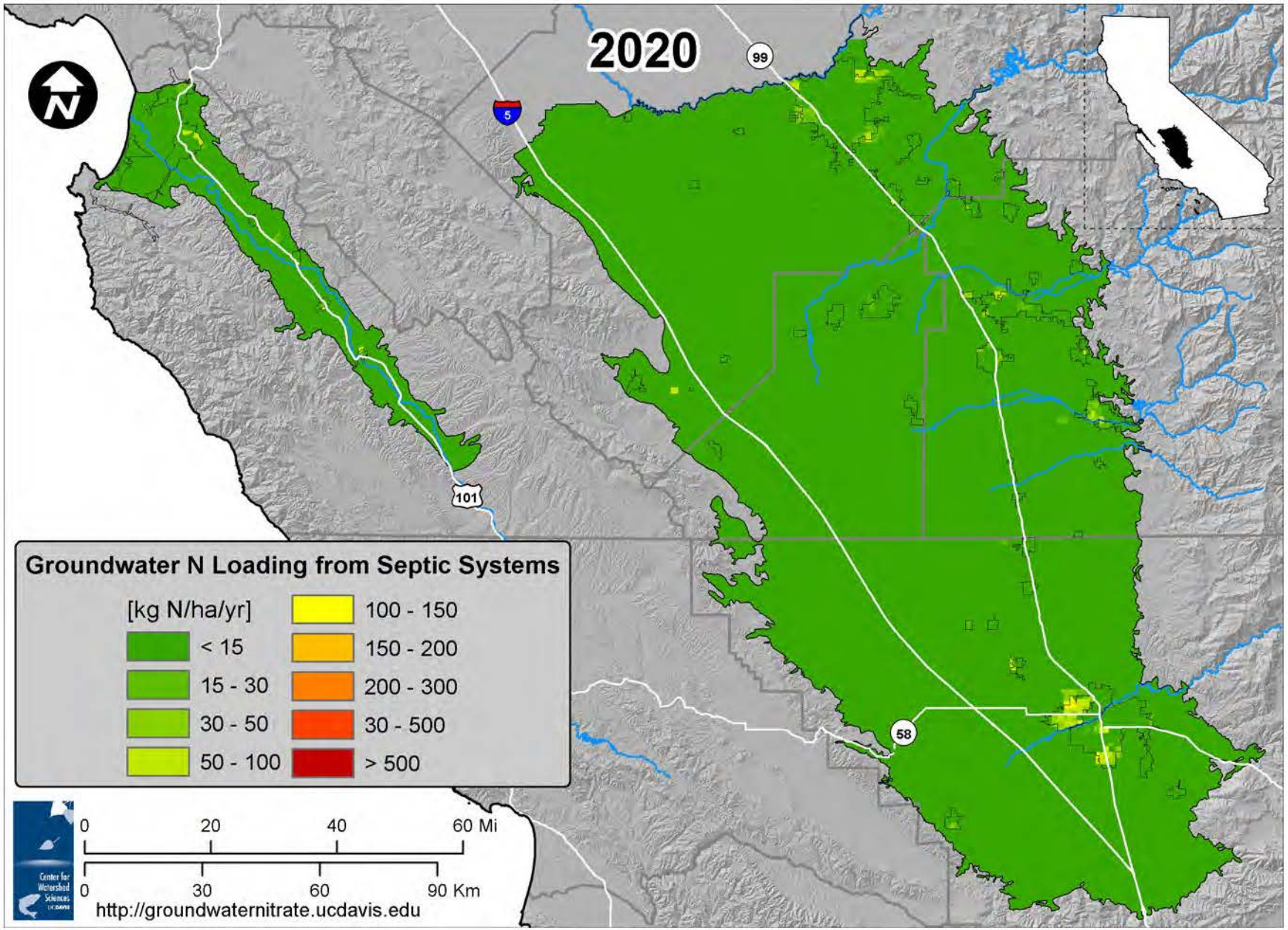
Appendix Figure 80. Nitrogen leached from septic systems, in 1990.



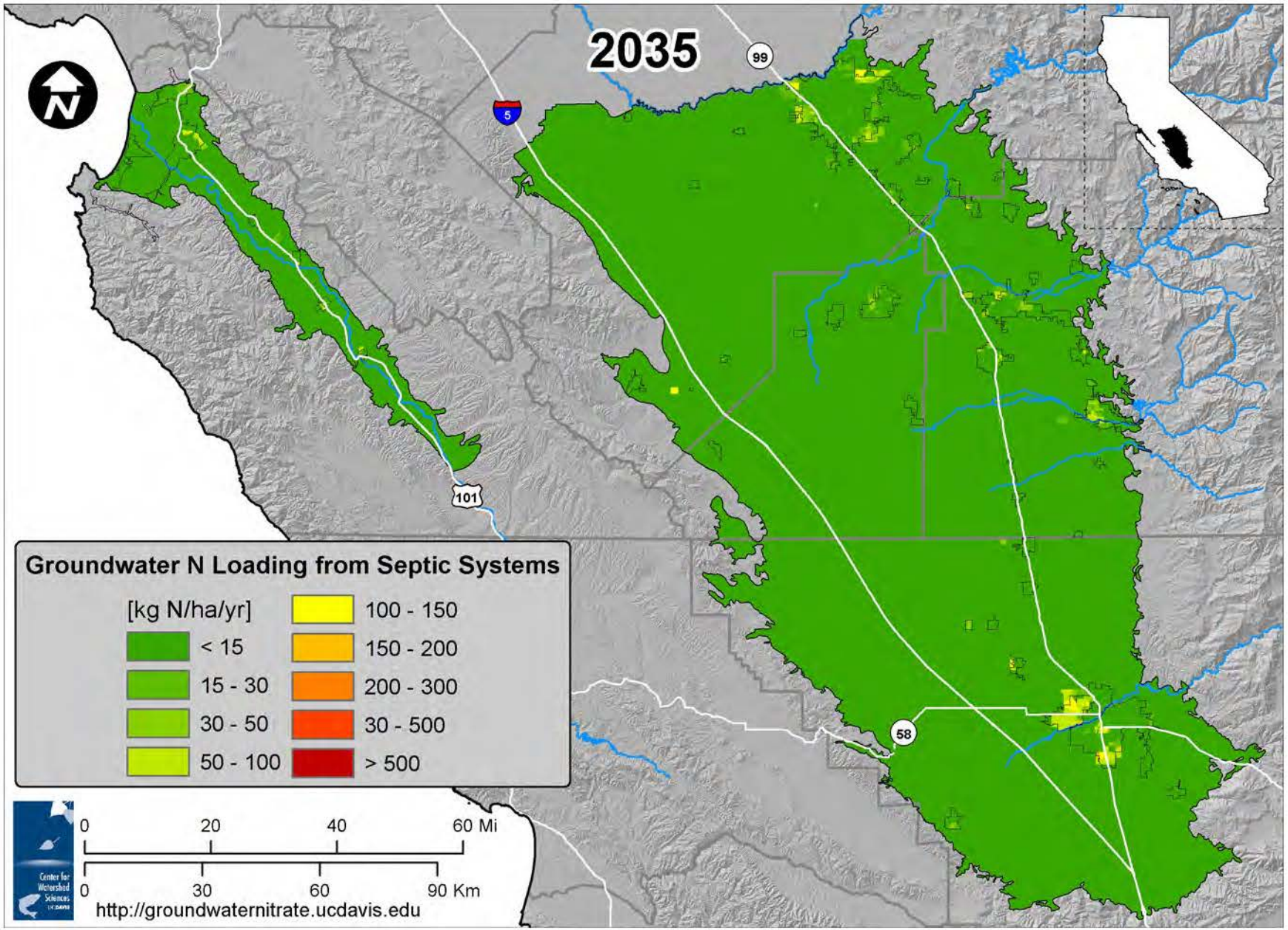
Appendix Figure 81. Nitrogen leached from septic systems, in 2005.



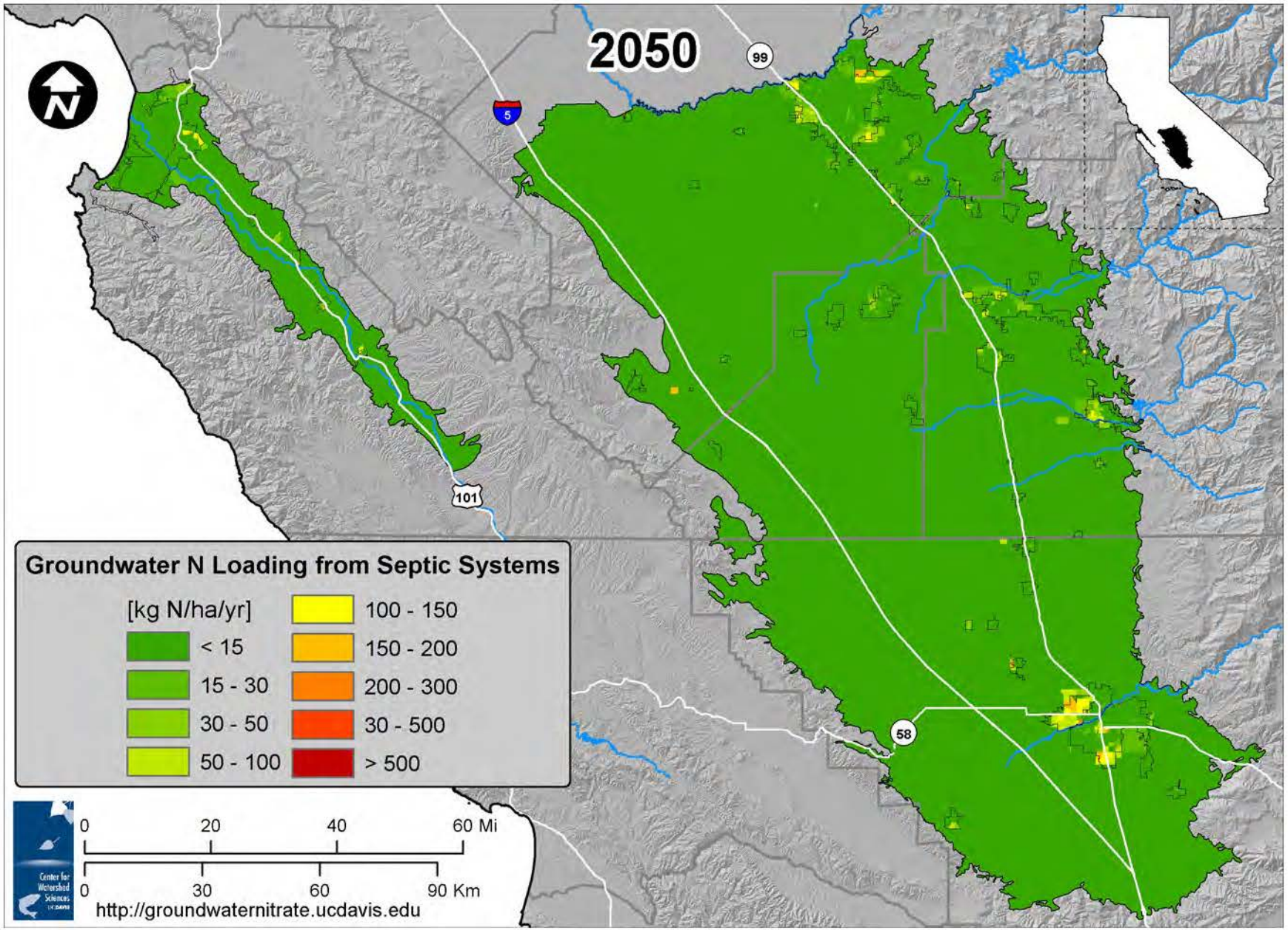
Appendix Figure 82. Nitrogen leached from septic systems, in 2020.



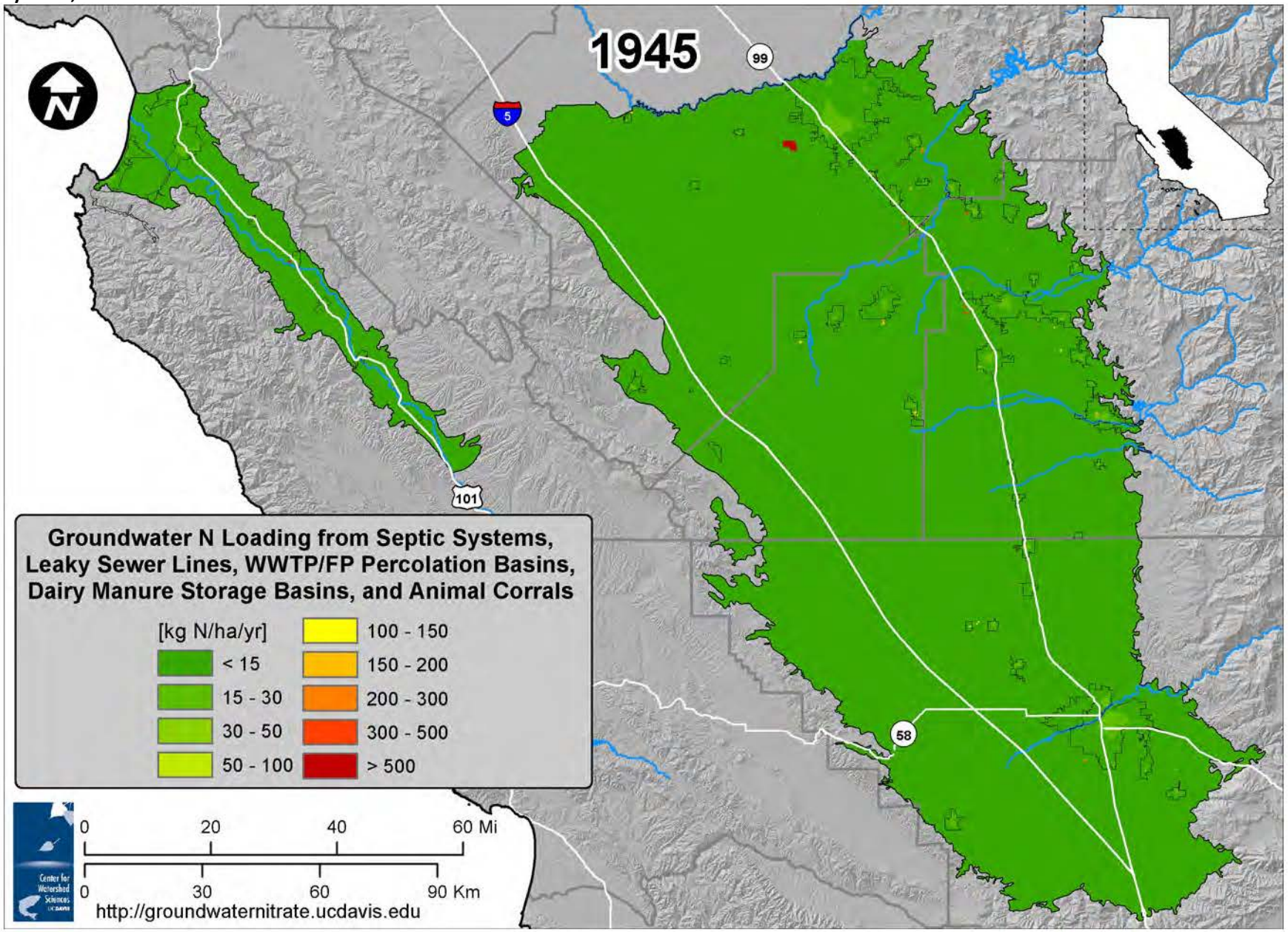
Appendix Figure 83. Nitrogen leached from septic systems, in 2035.



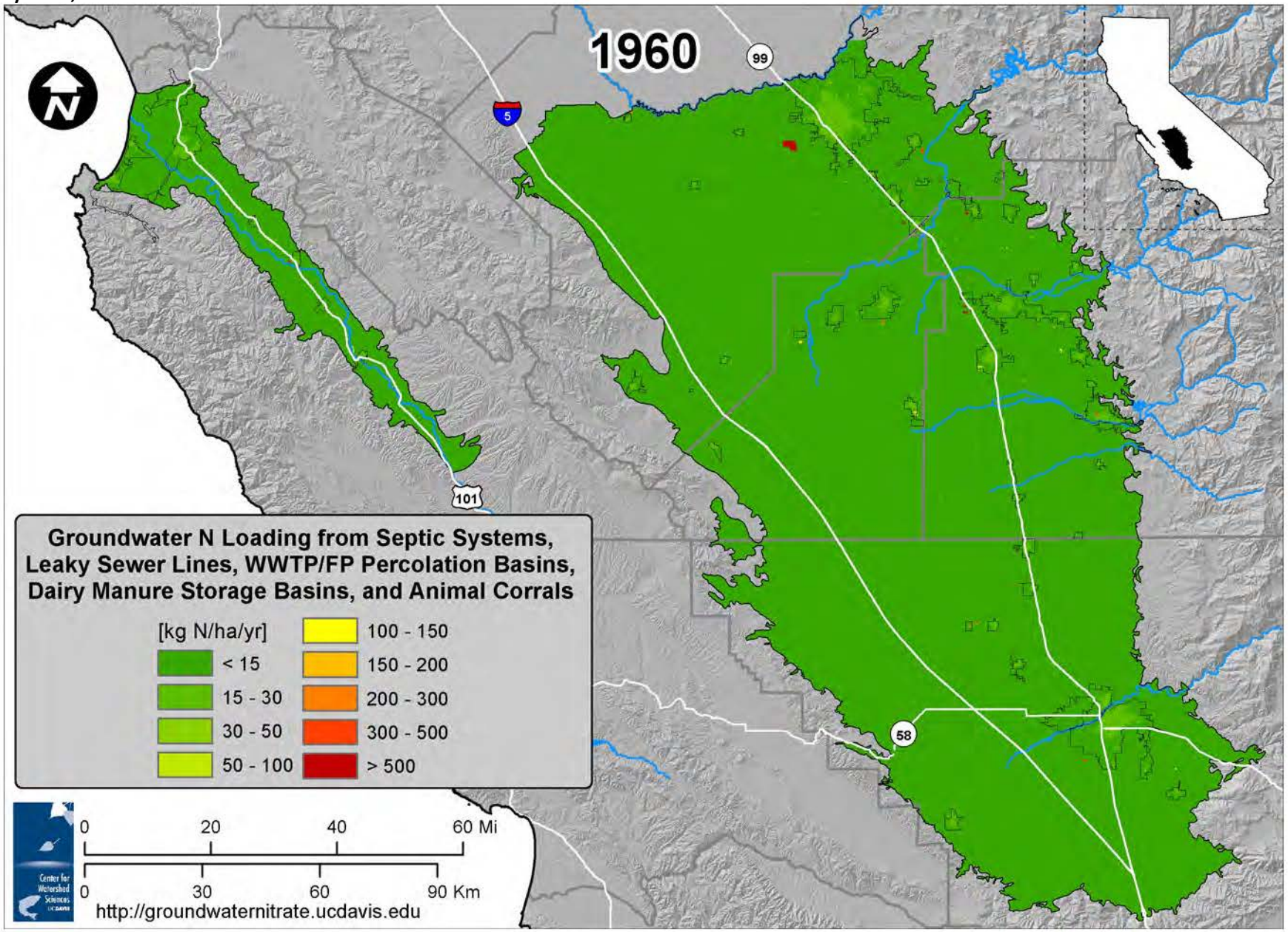
Appendix Figure 84. Nitrogen leached from septic systems, in 2050.



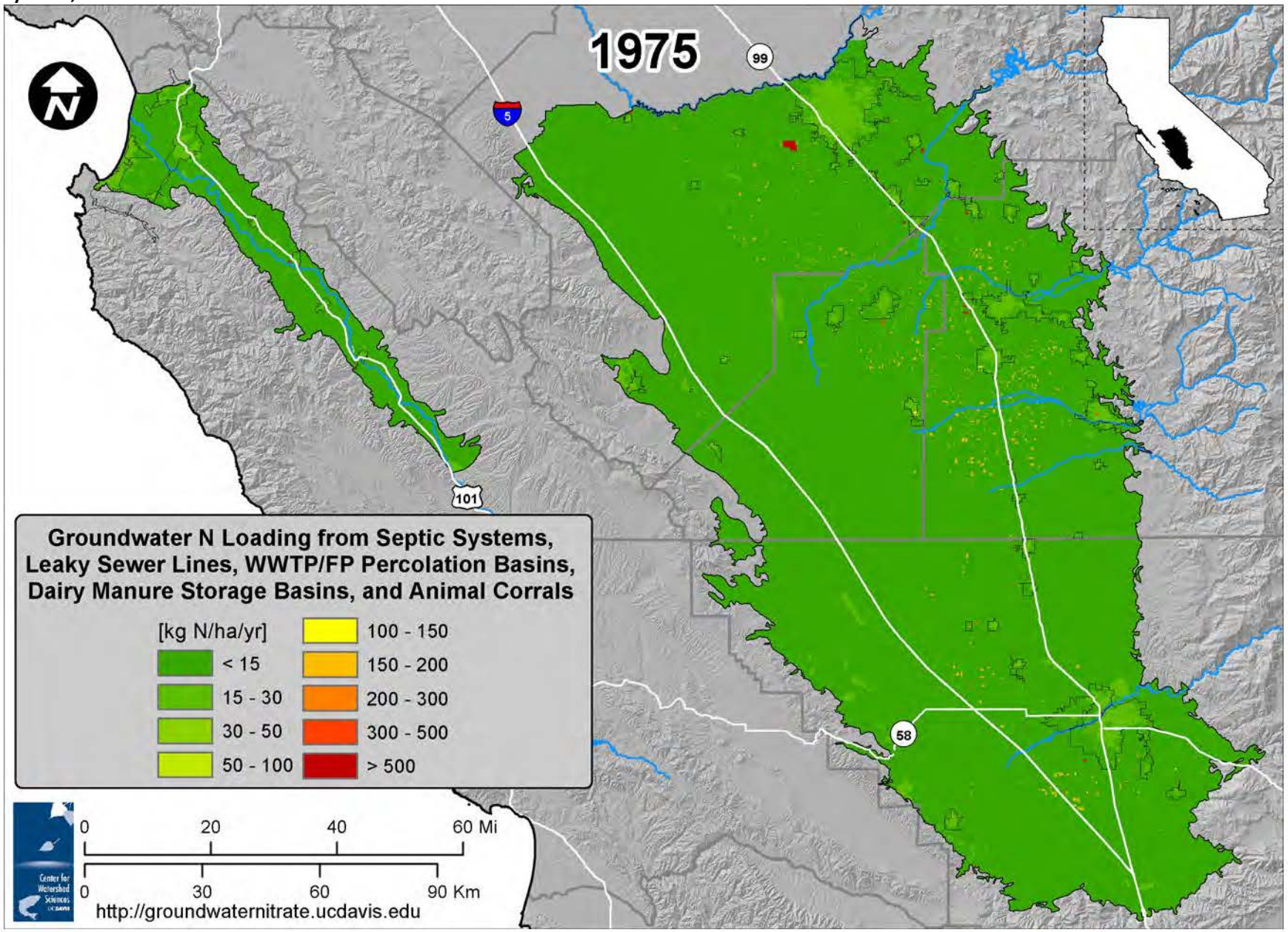
Appendix Figure 85. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 1945.



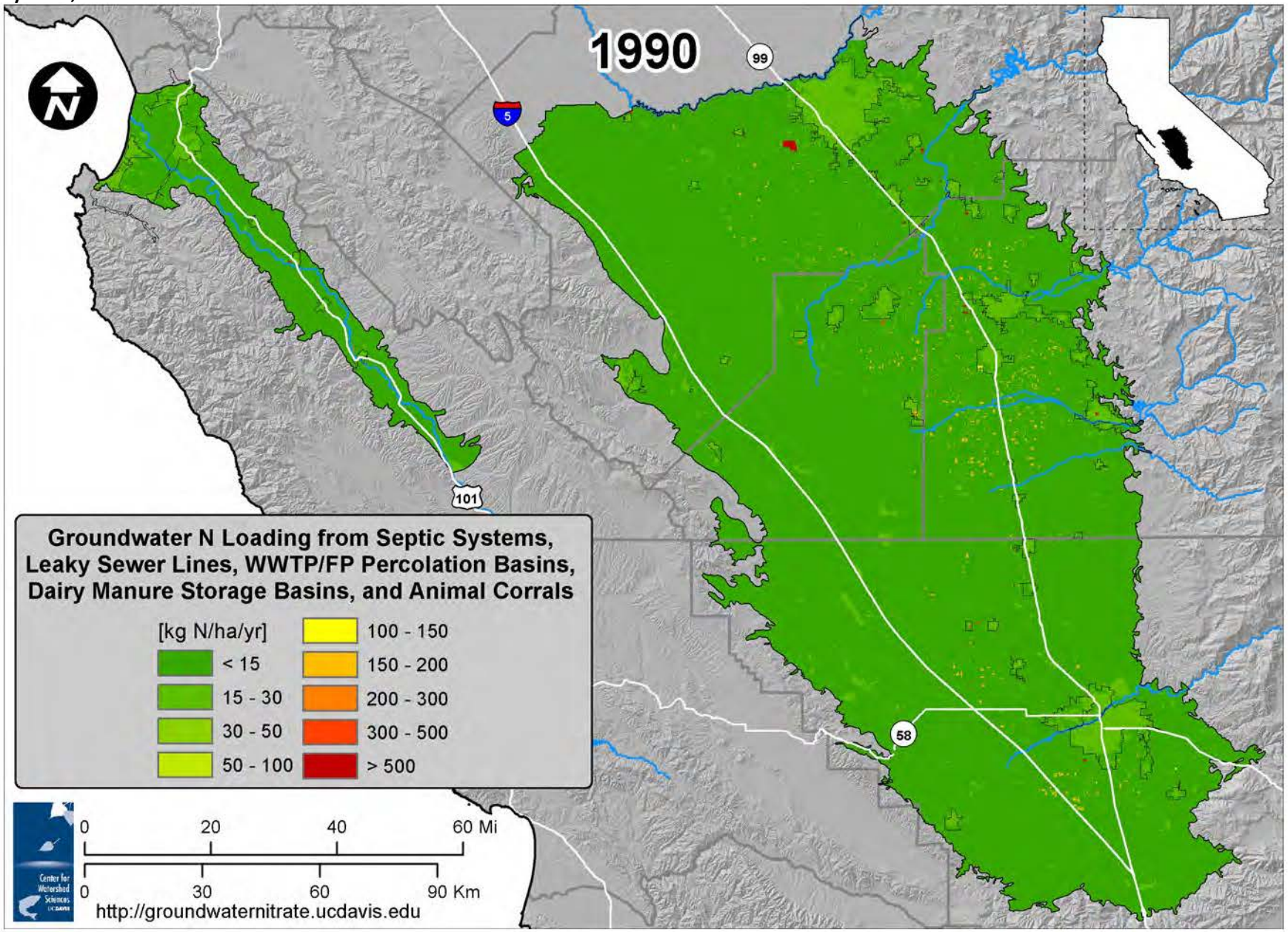
Appendix Figure 86. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 1960.



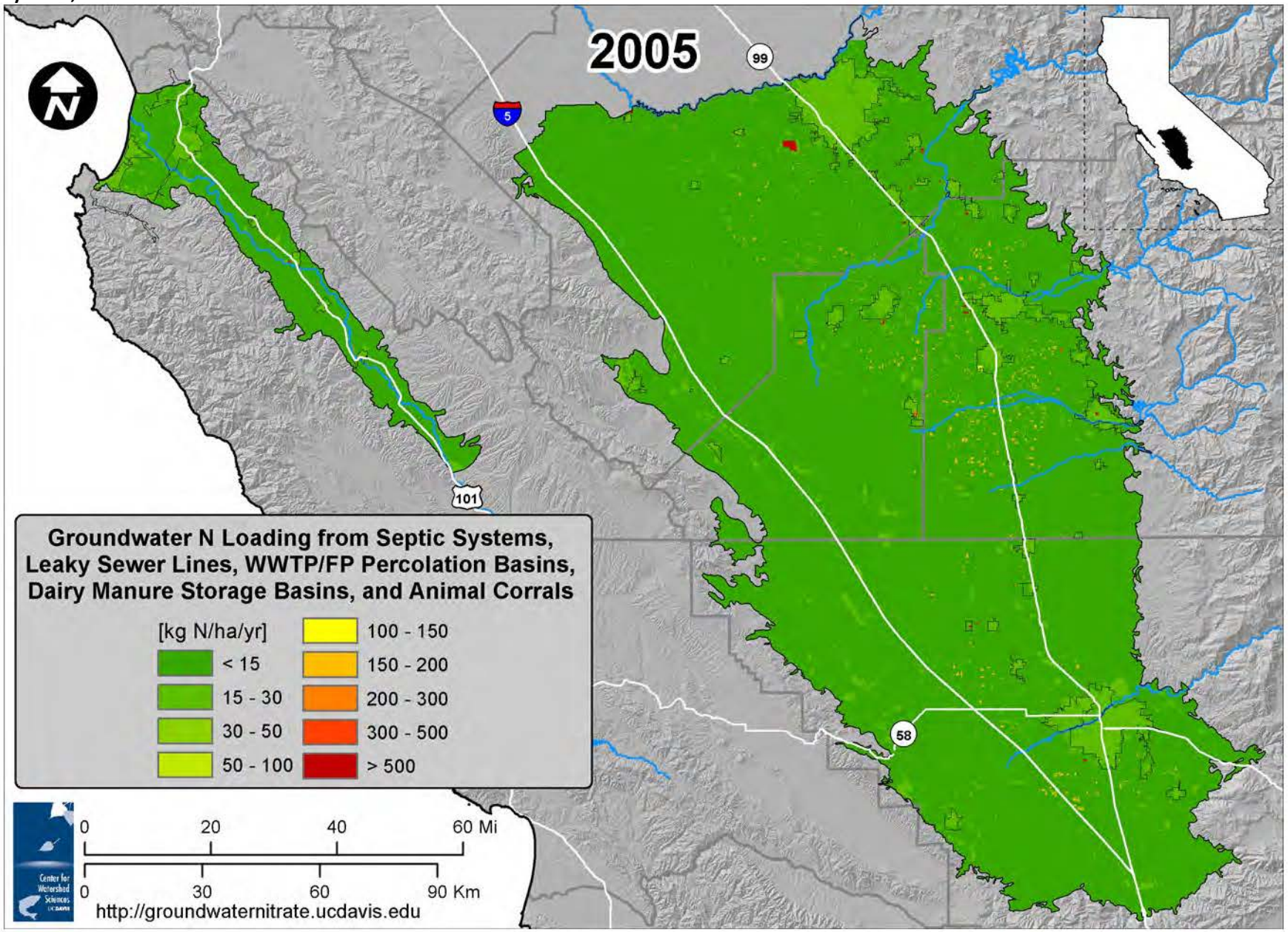
Appendix Figure 87. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 1975.



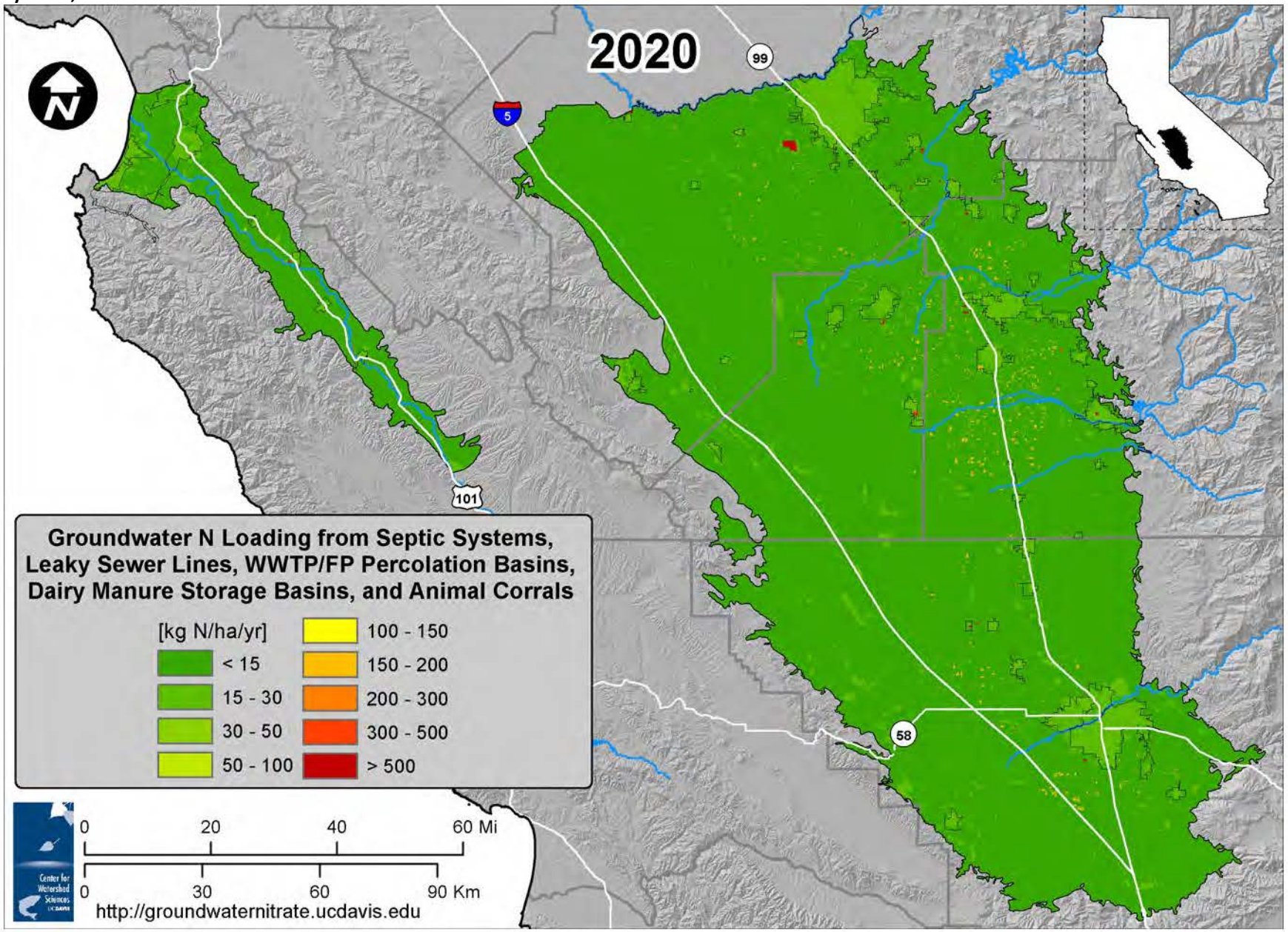
Appendix Figure 88. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 1990.



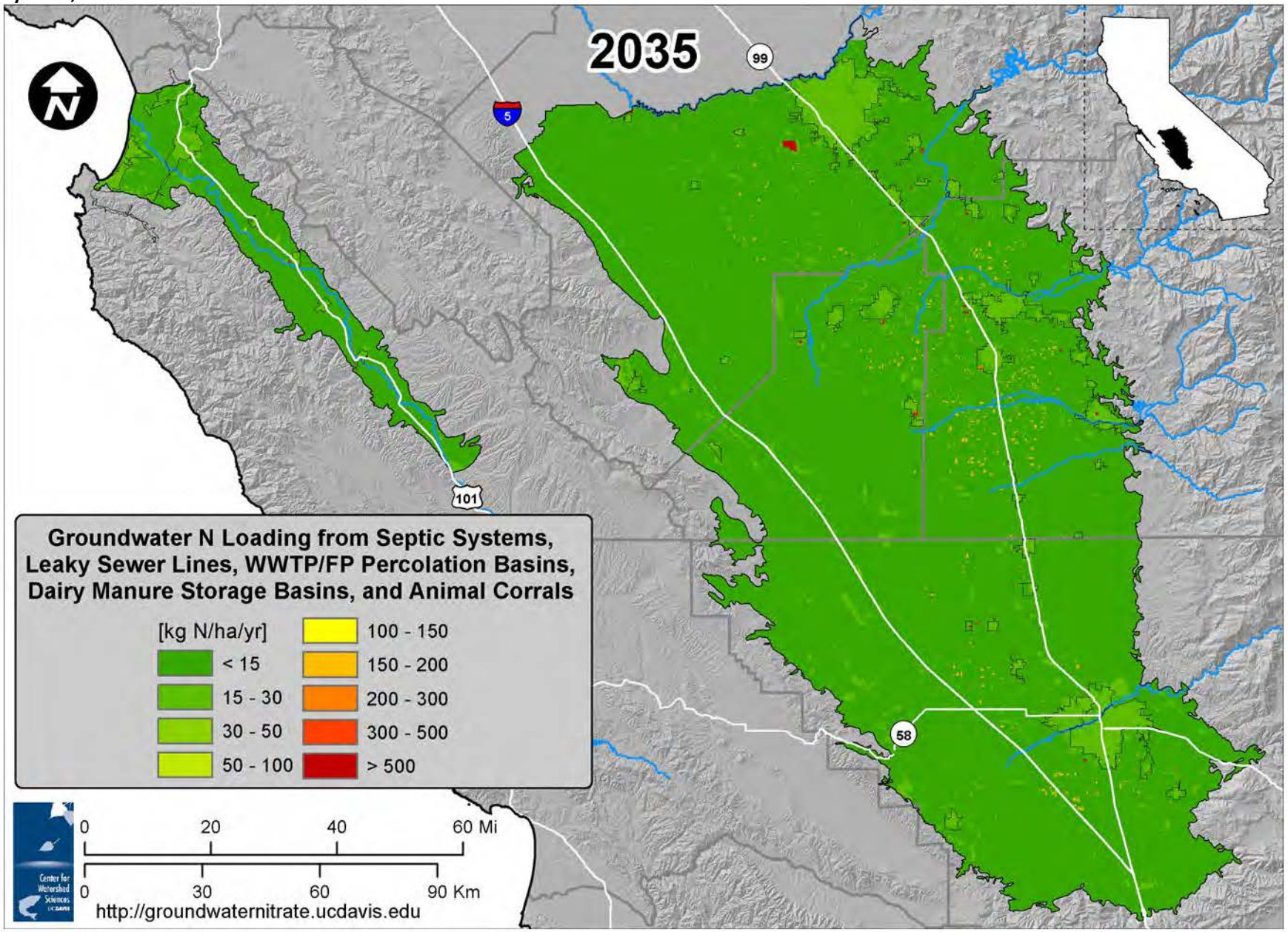
Appendix Figure 89. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 2005.



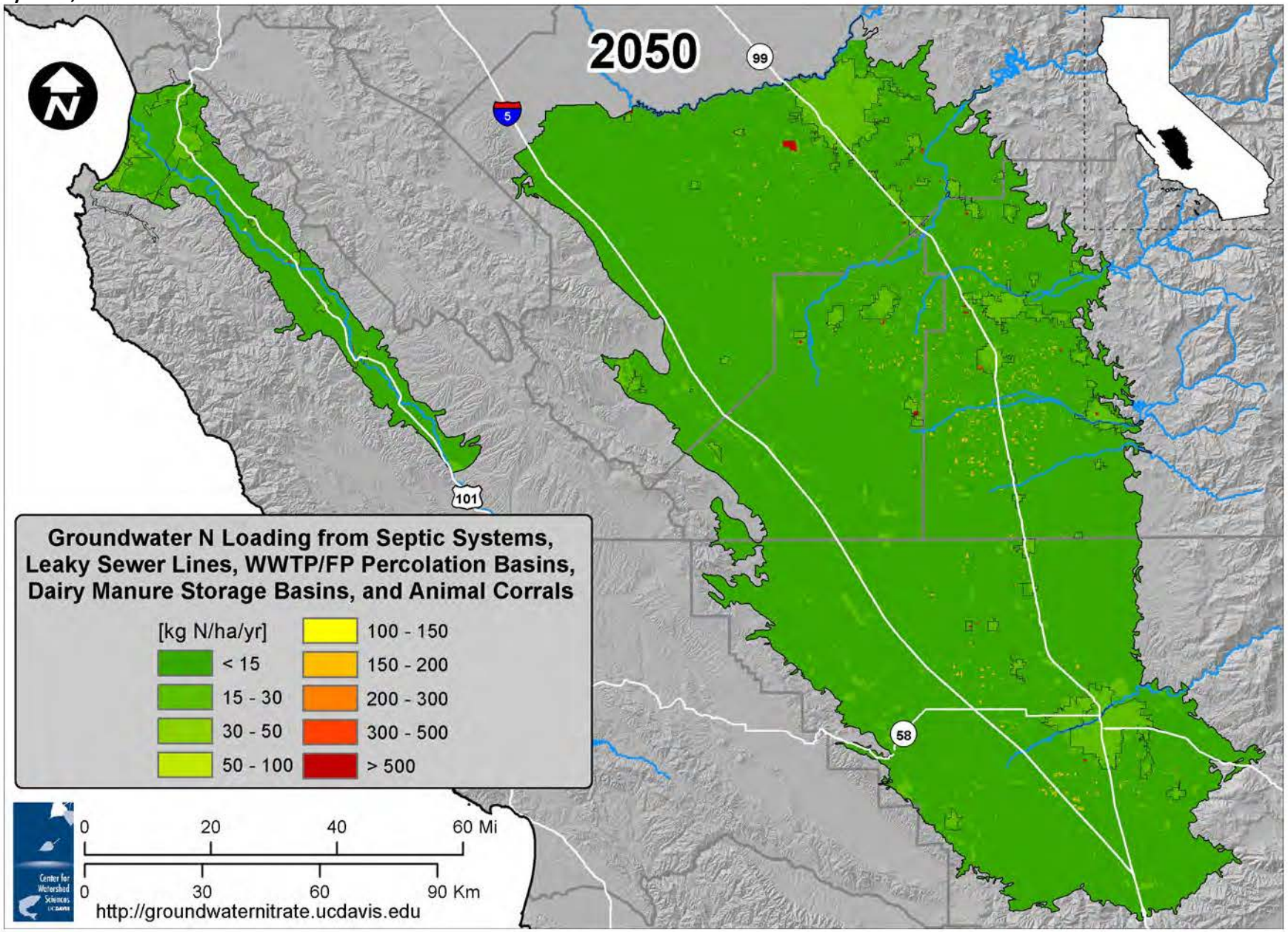
Appendix Figure 90. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 2020.



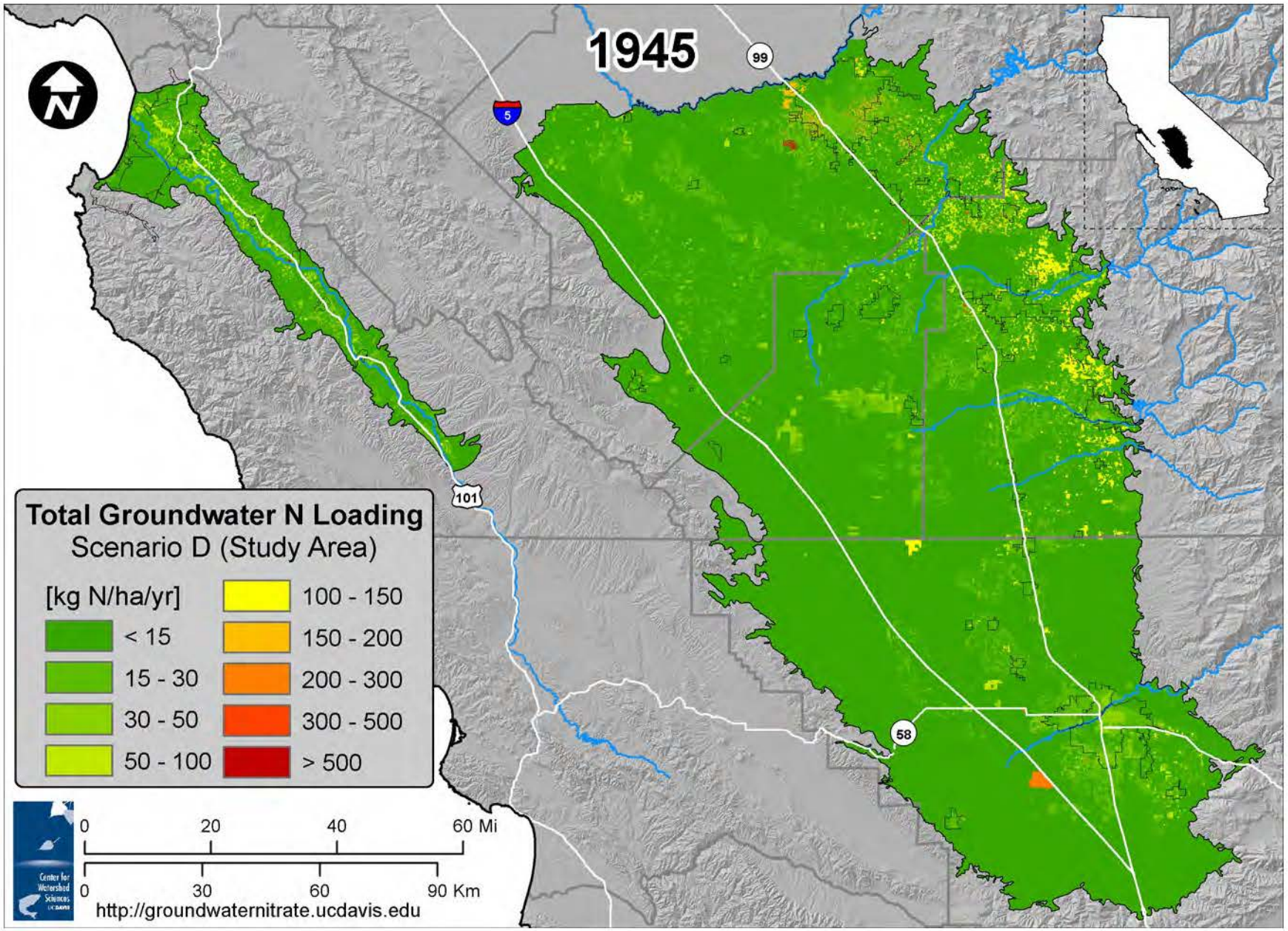
Appendix Figure 91. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 2035.



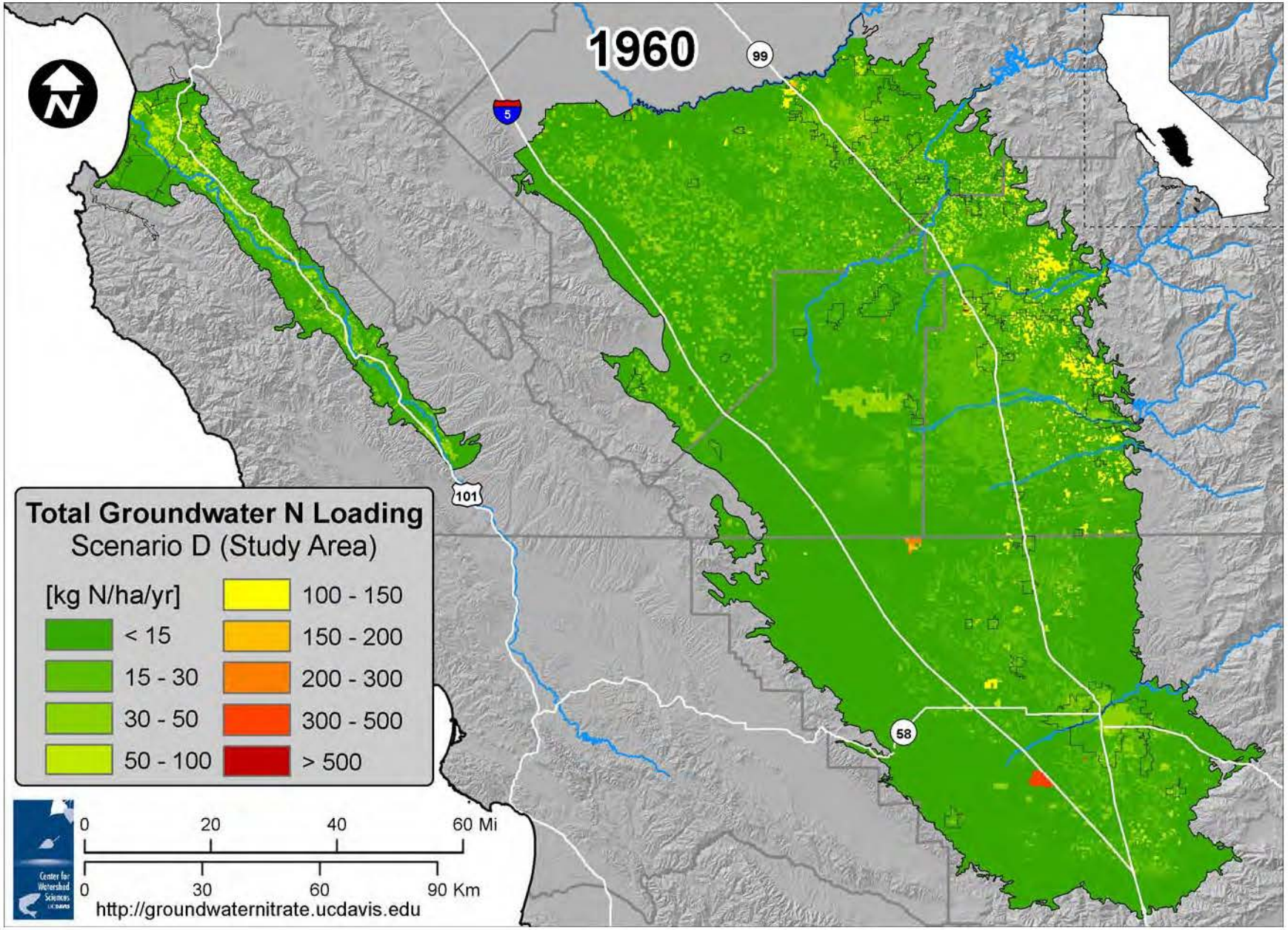
Appendix Figure 92. Direct percolation of nitrogen to groundwater from corrals, lagoons, WWTP & FP percolation ponds, golf courses, and leaky sewer systems, in 2050.



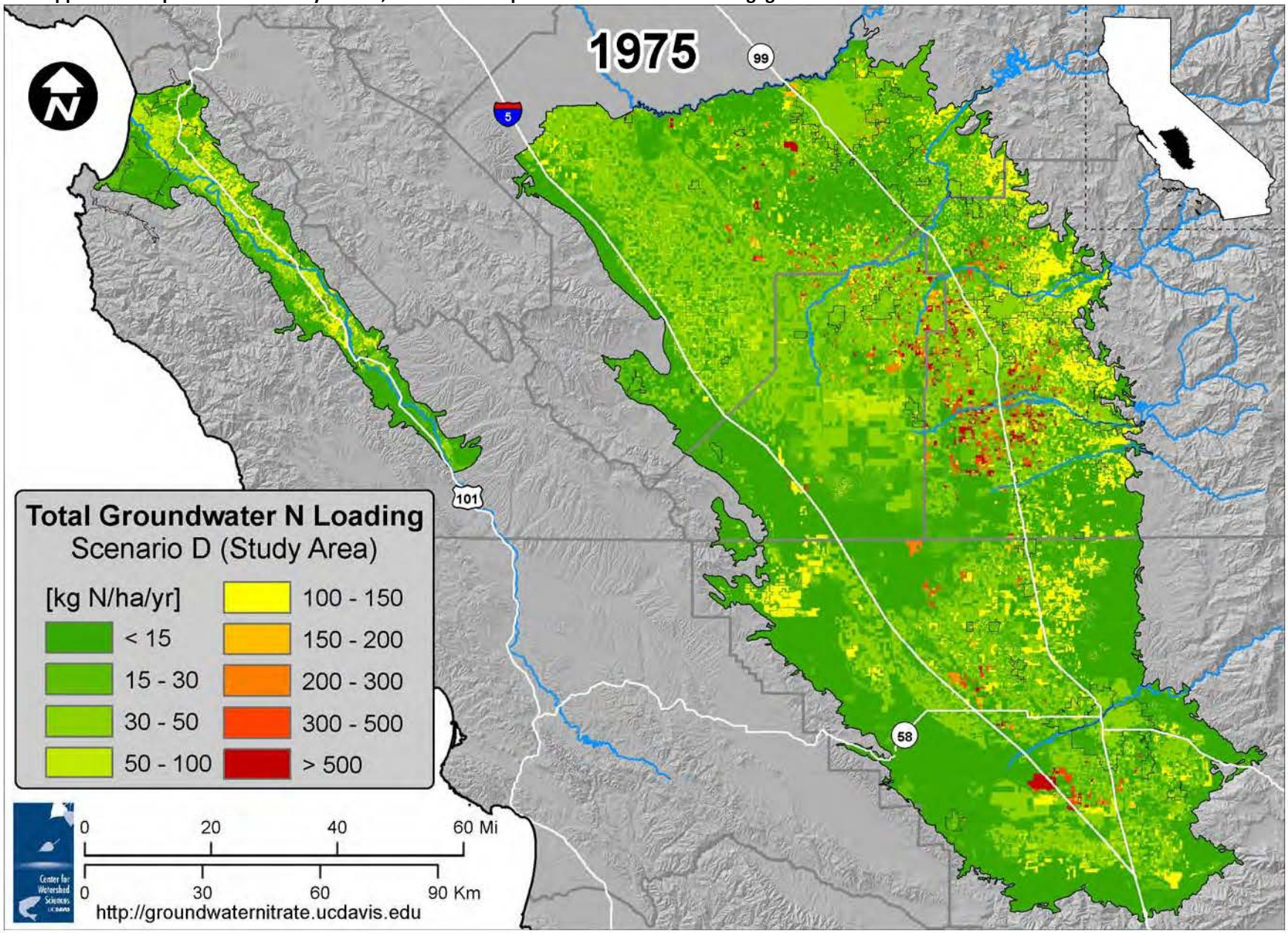
Appendix Figure 93. Total nitrogen loading to groundwater from all sources, in 1945. Dairy animals are in pasture and no manure is exported.



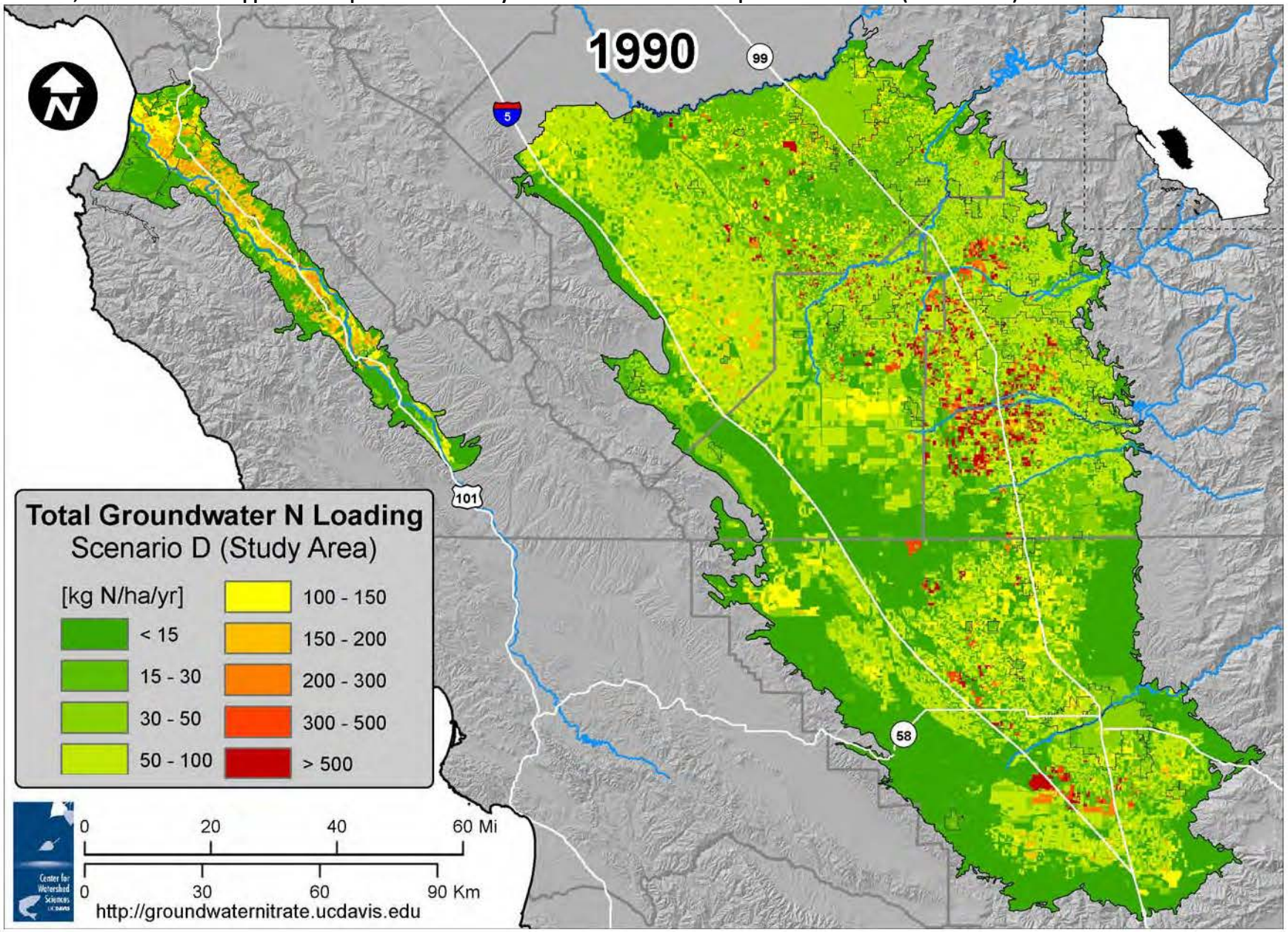
Appendix Figure 94. Total nitrogen loading to groundwater from all sources, in 1960. Dairy animals are in pasture and no manure is exported.



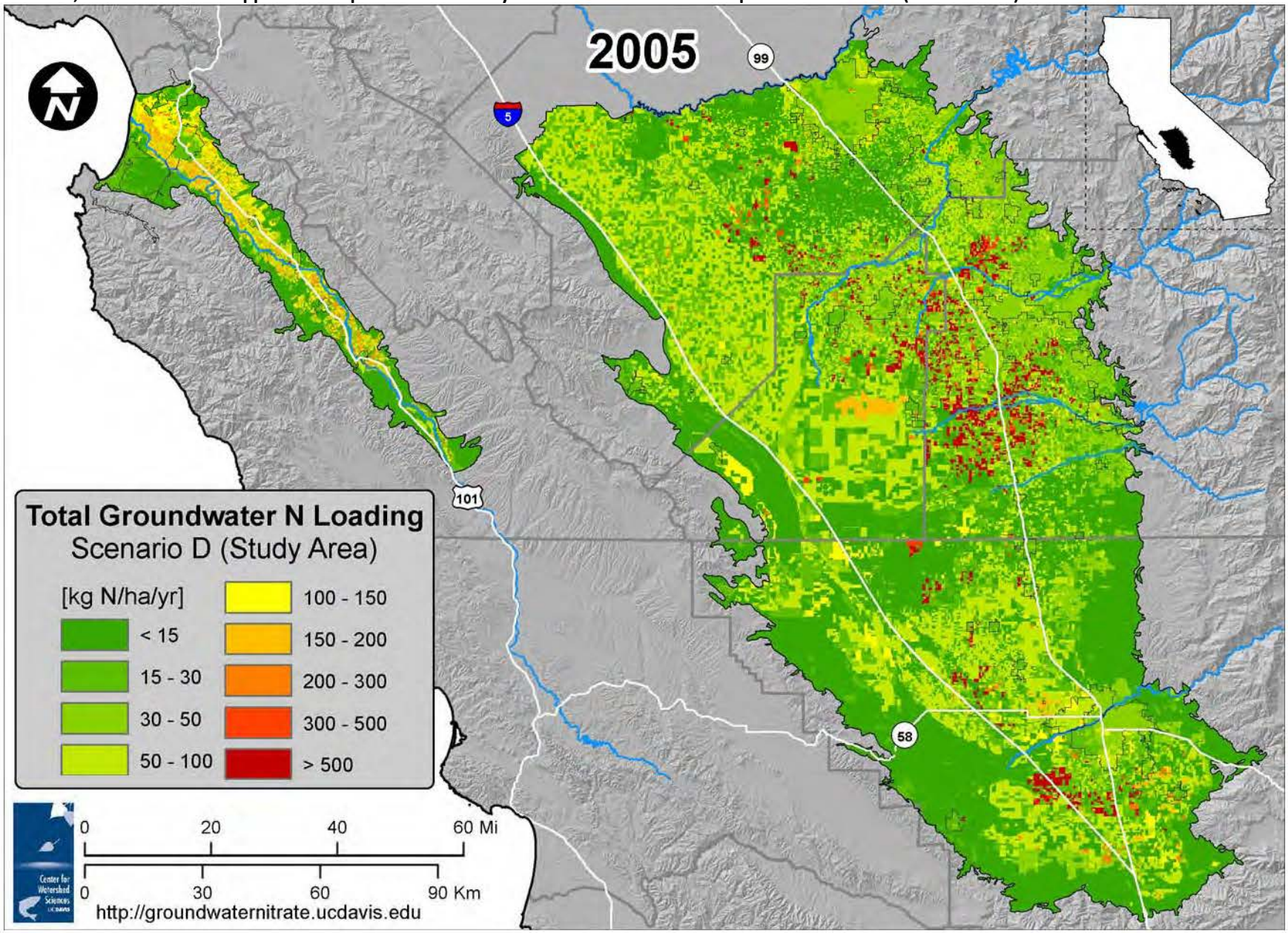
Appendix Figure 95. Total nitrogen loading to groundwater from all sources, in 1975. It is assumed that dairy animals are in corrals or freestalls, manure is land applied on cropland controlled by dairies, but manure exported from dairies is still negligible.



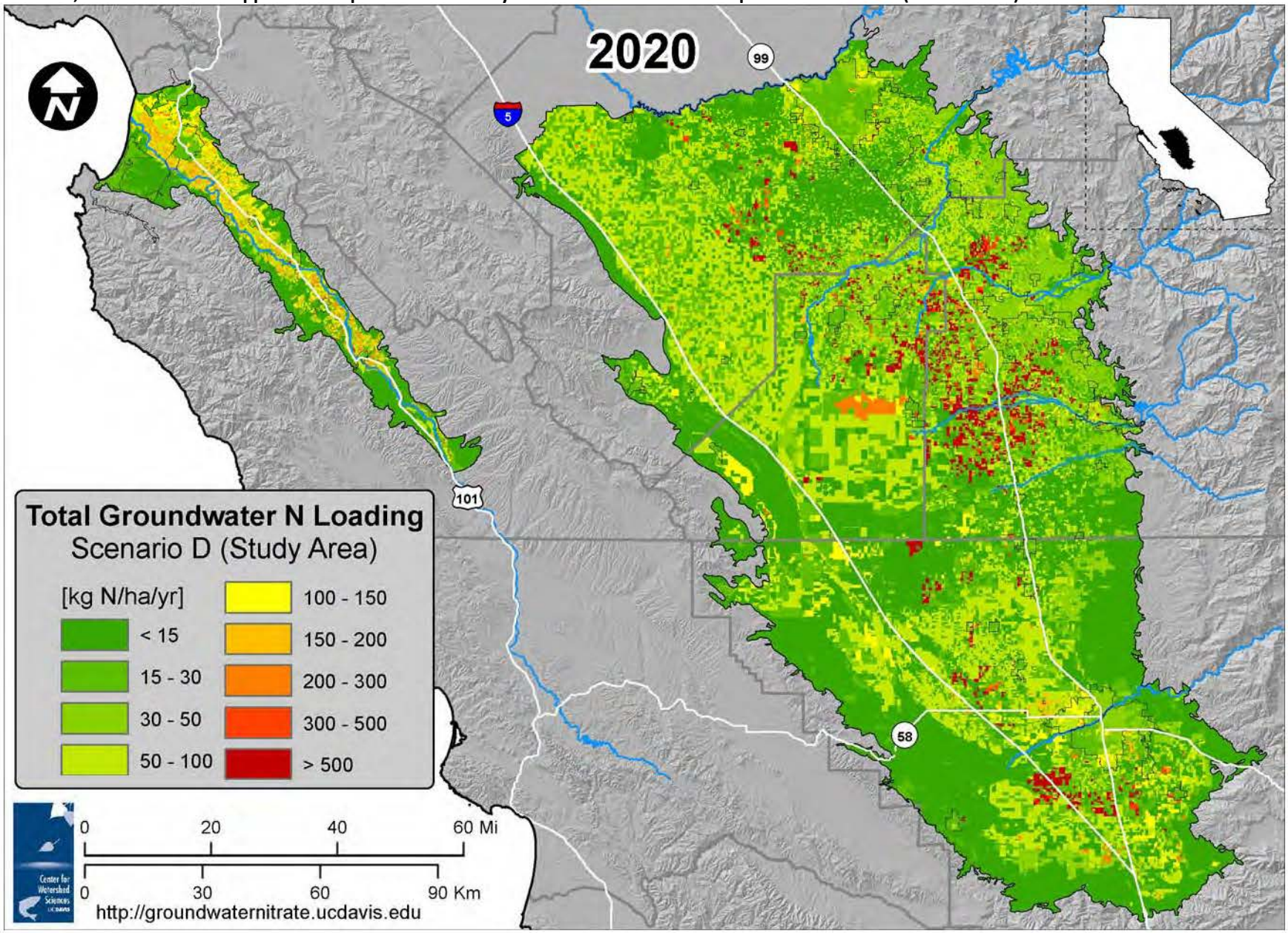
Appendix Figure 96. Total nitrogen loading to groundwater from all sources, in 1990. It is assumed that dairy animals are in corrals or freestalls. In this scenario, all manure is land applied on cropland controlled by dairies and no manure is exported from dairies ("Scenario D").



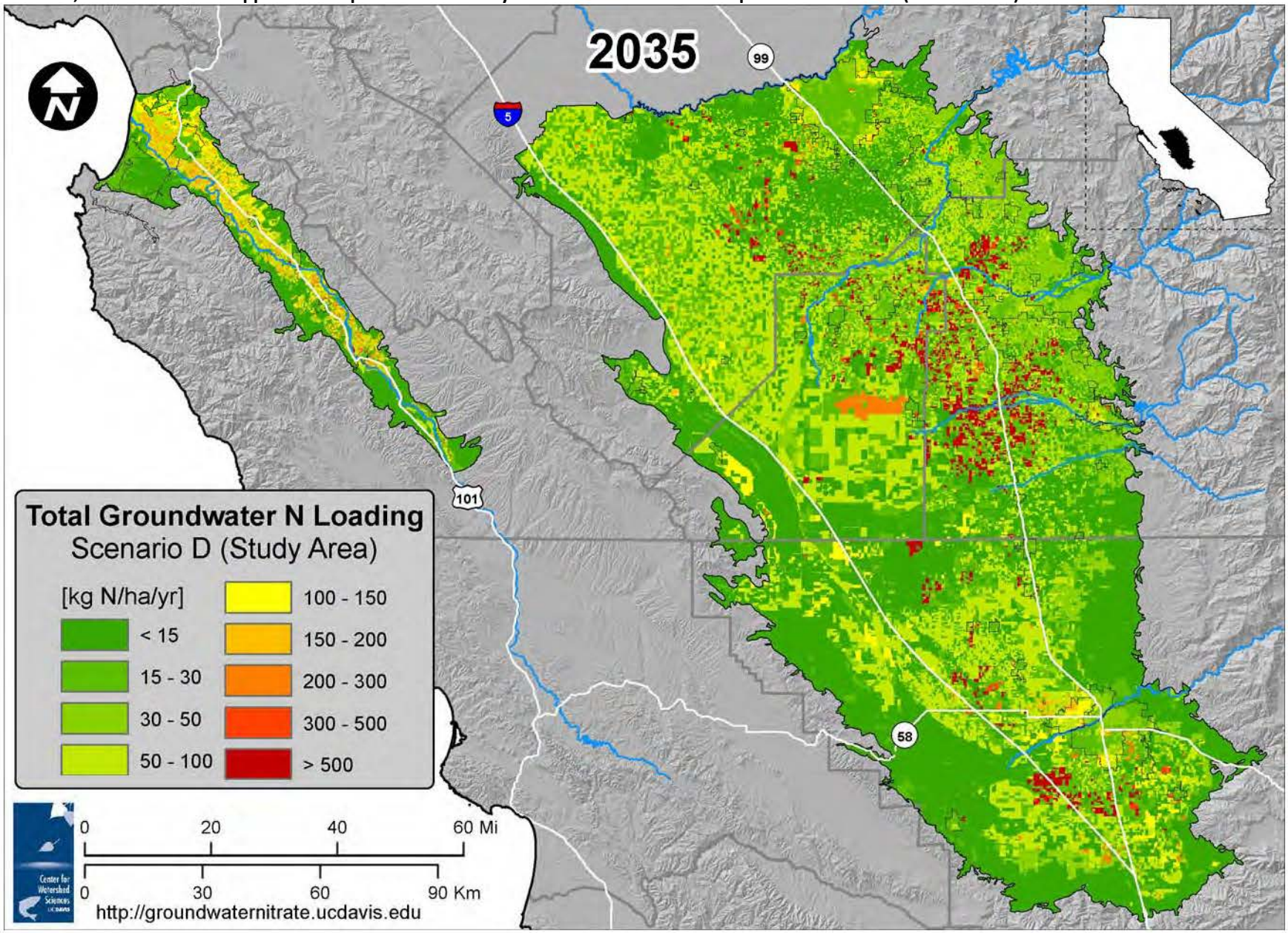
Appendix Figure 97. Total nitrogen loading to groundwater from all sources, in 2005. It is assumed that dairy animals are in corrals or freestalls. In this scenario, all manure is land applied on cropland controlled by dairies and no manure is exported from dairies ("Scenario D").



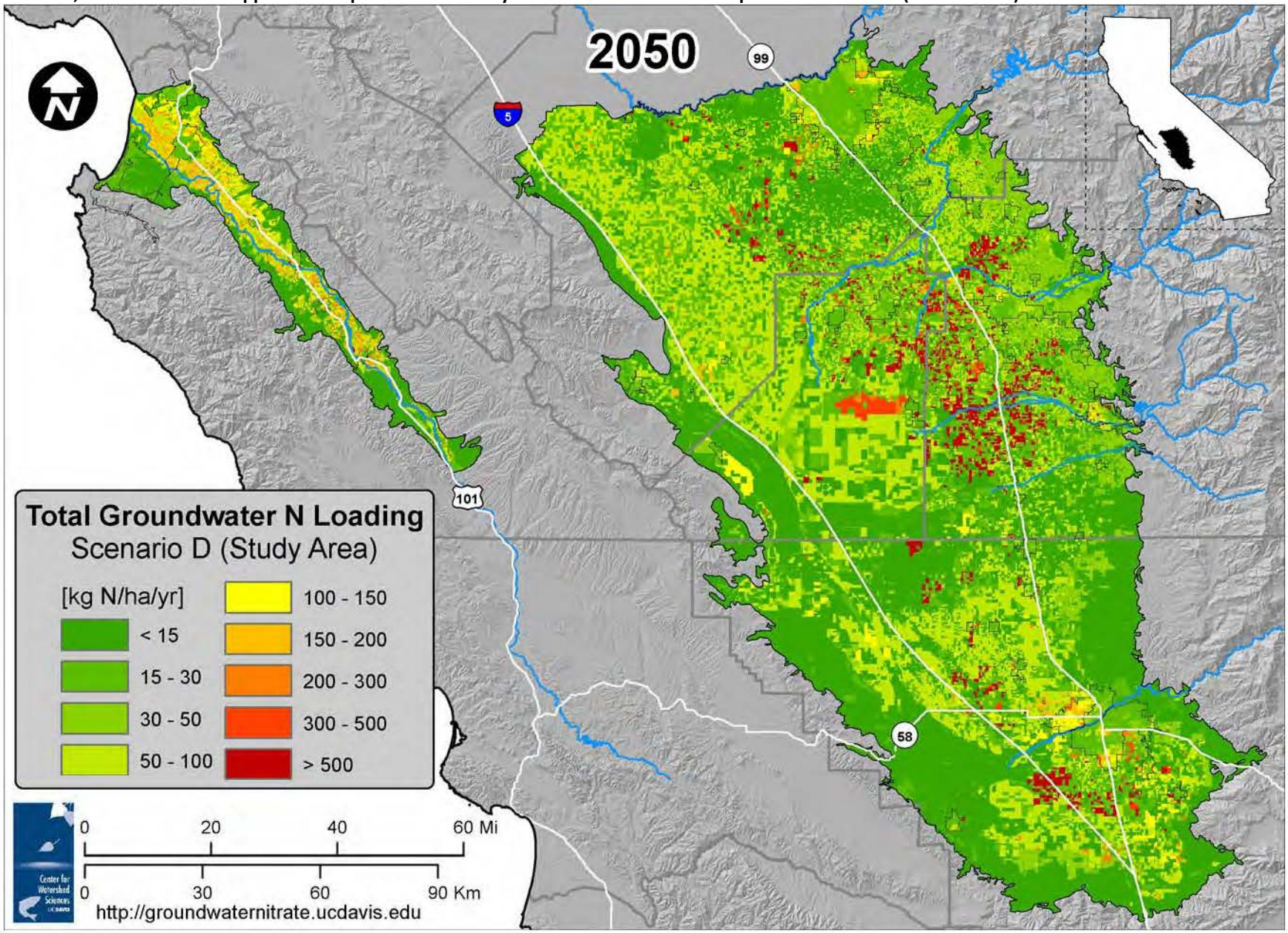
Appendix Figure 98. Total nitrogen loading to groundwater from all sources, in 2020. It is assumed that dairy animals are in corrals or freestalls. In this scenario, all manure is land applied on cropland controlled by dairies and no manure is exported from dairies ("Scenario D").



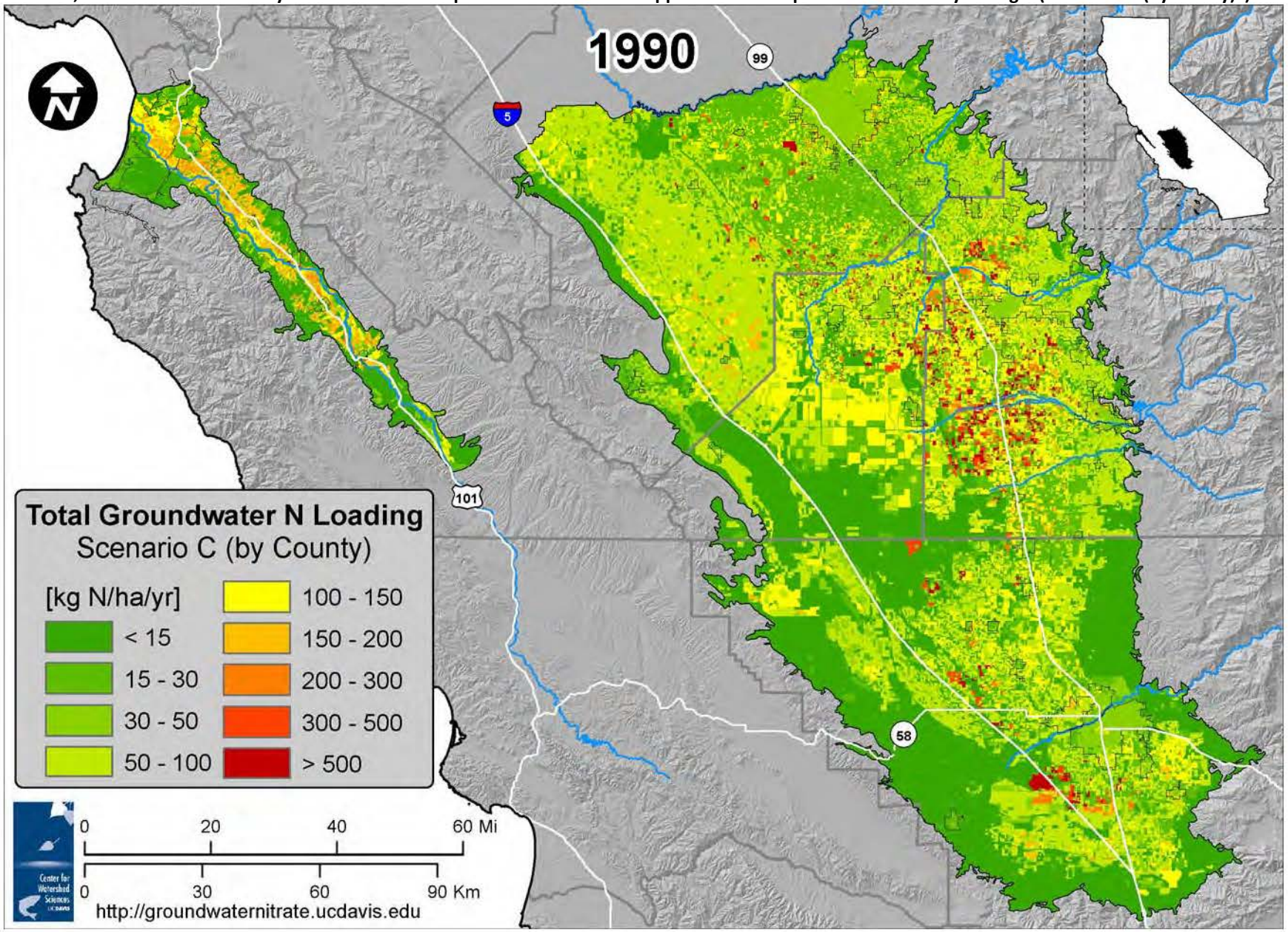
Appendix Figure 99. Total nitrogen loading to groundwater from all sources, in 2035. It is assumed that dairy animals are in corrals or freestalls. In this scenario, all manure is land applied on cropland controlled by dairies and no manure is exported from dairies ("Scenario D").



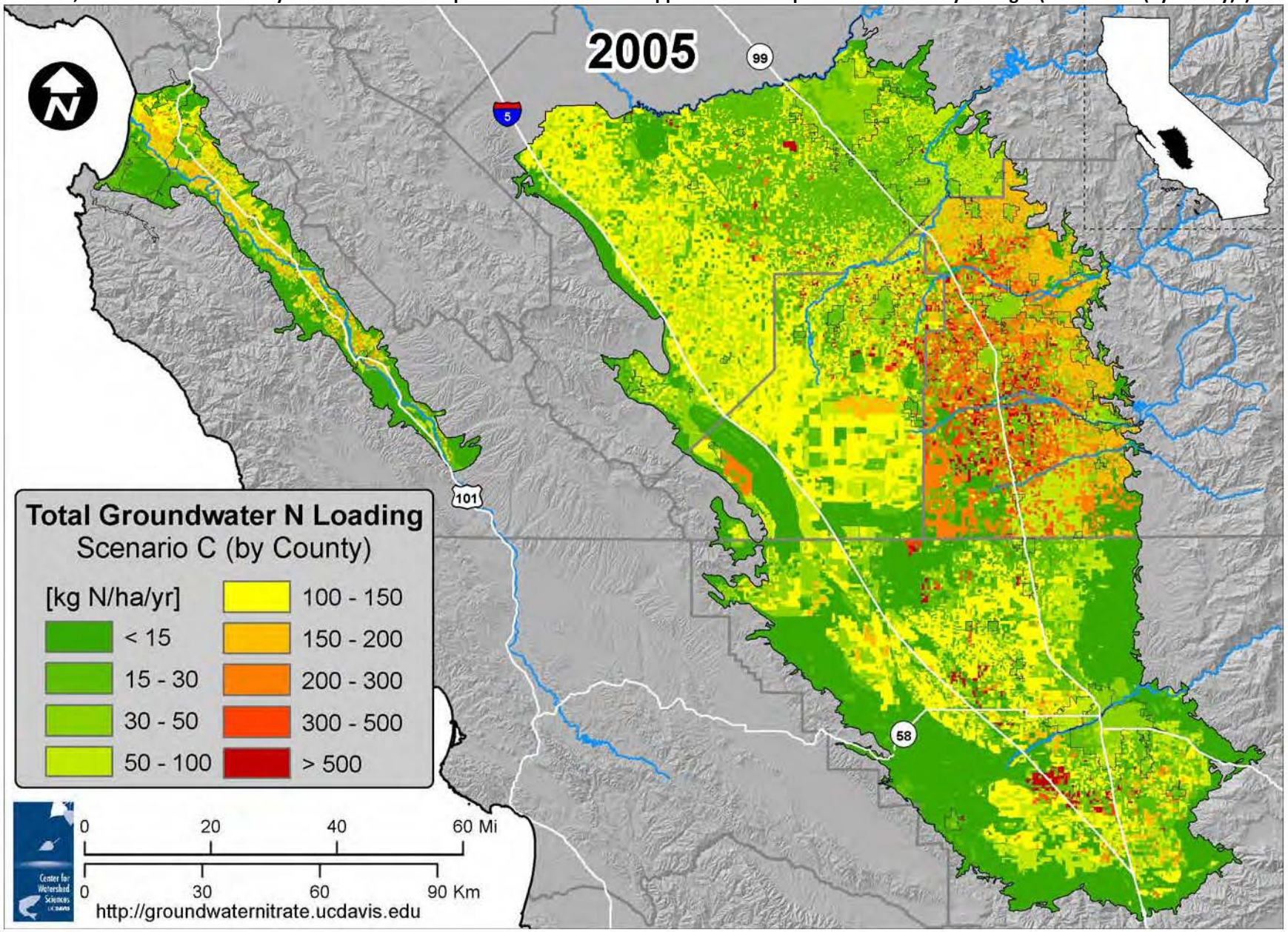
Appendix Figure 100. Total nitrogen loading to groundwater from all sources, in 2050. It is assumed that dairy animals are in corrals or freestalls. In this scenario, all manure is land applied on cropland controlled by dairies and no manure is exported from dairies ("Scenario D").



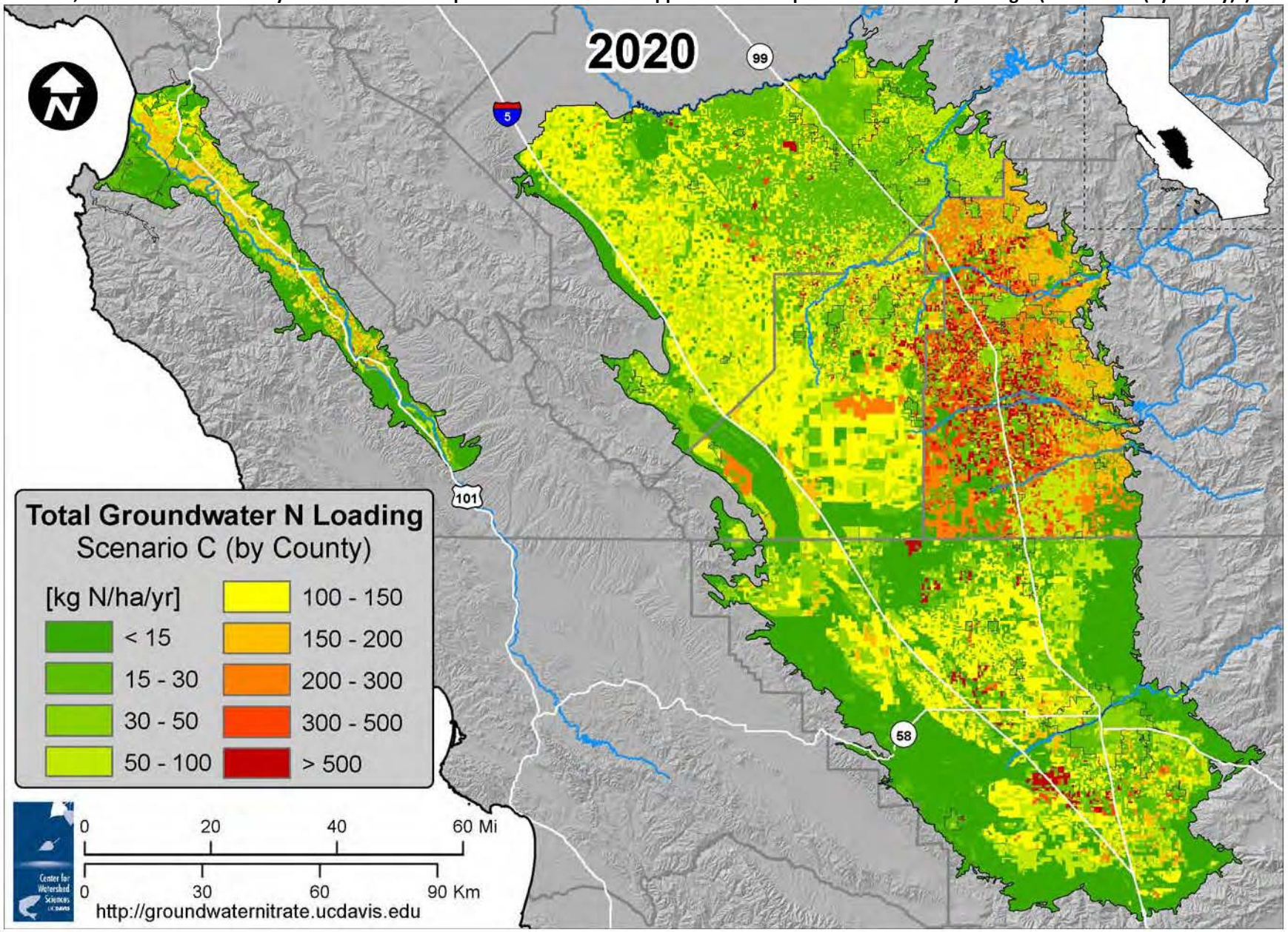
Appendix Figure 101. Total nitrogen loading to groundwater from all sources, in 1990. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the county of origin ("Scenario C (by county)").



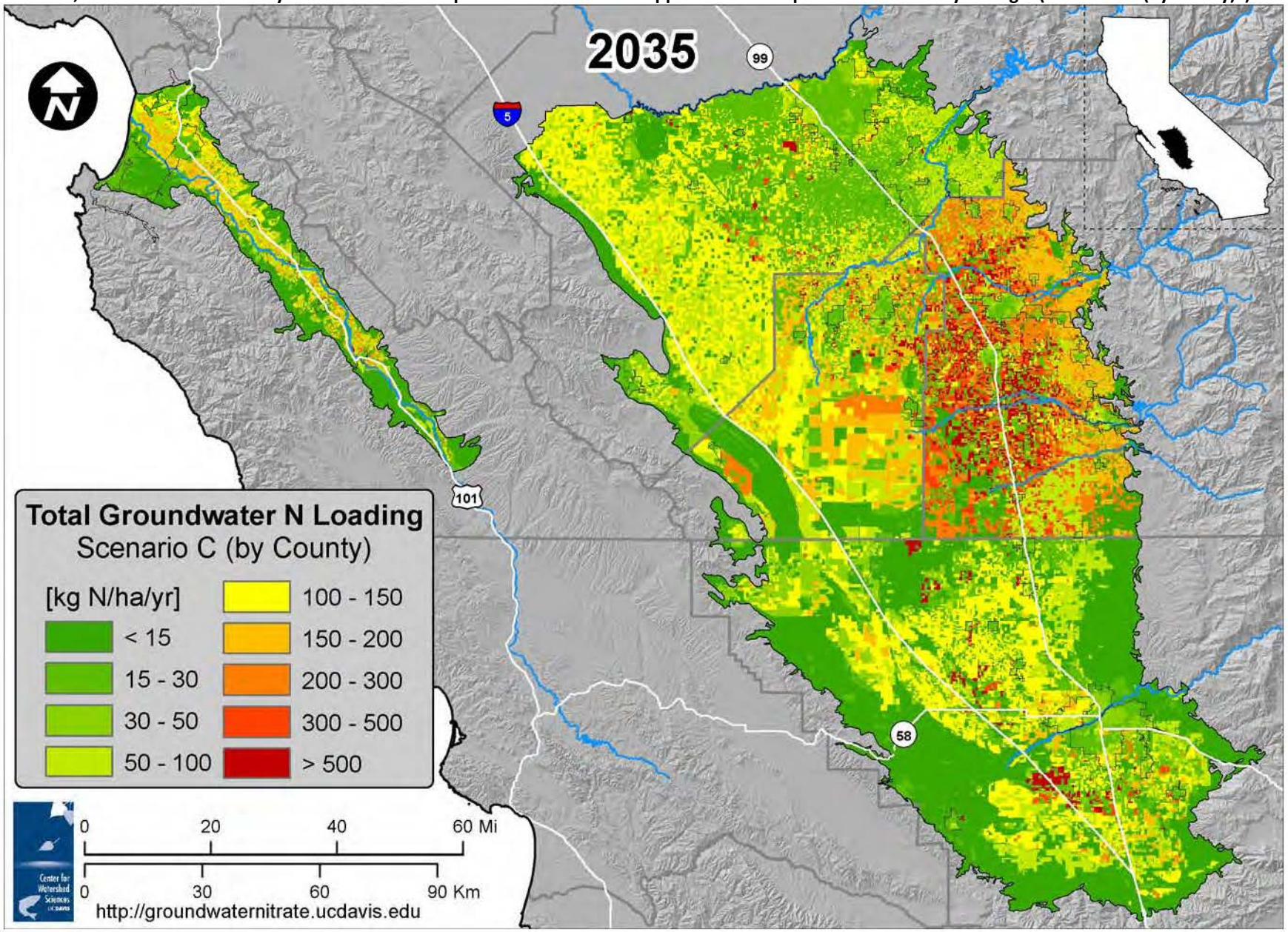
Appendix Figure 102. Total nitrogen loading to groundwater from all sources, in 2005. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the county of origin ("Scenario C (by county)").



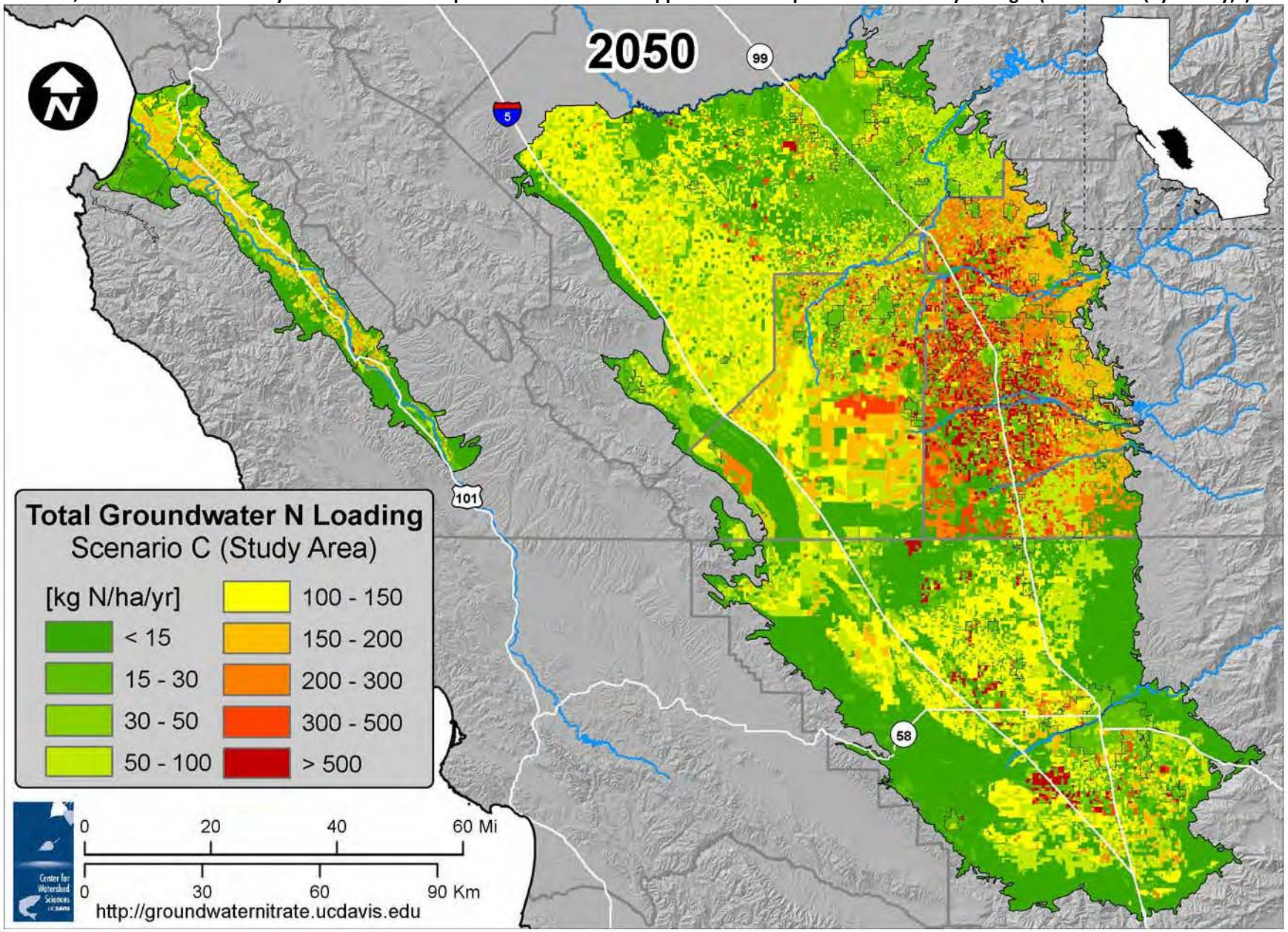
Appendix Figure 103. Total nitrogen loading to groundwater from all sources, in 2020. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the county of origin ("Scenario C (by county)").



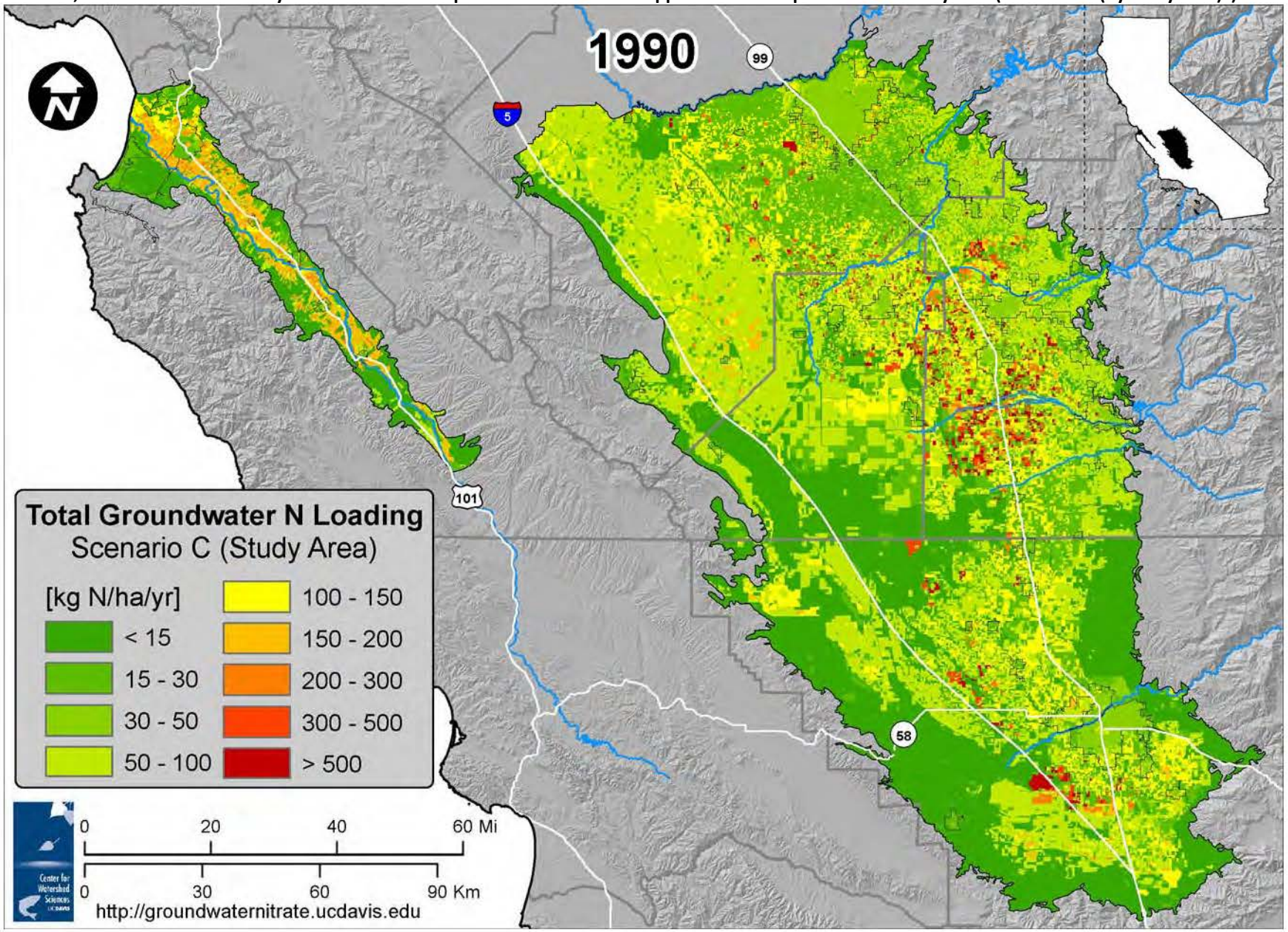
Appendix Figure 104. Total nitrogen loading to groundwater from all sources, in 2035. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the county of origin ("Scenario C (by county)").



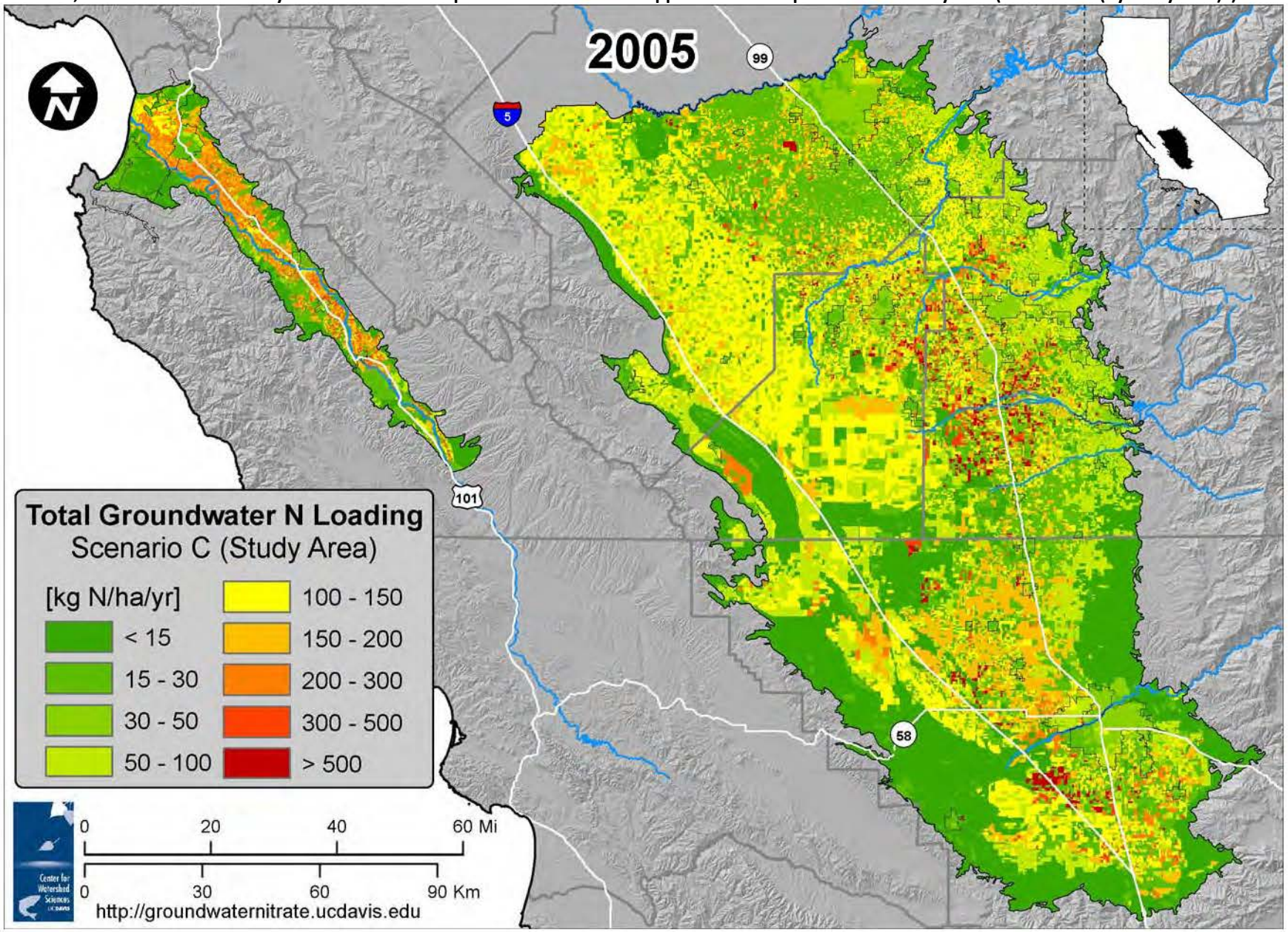
Appendix Figure 105. Total nitrogen loading to groundwater from all sources, in 2050. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the county of origin ("Scenario C (by county)").



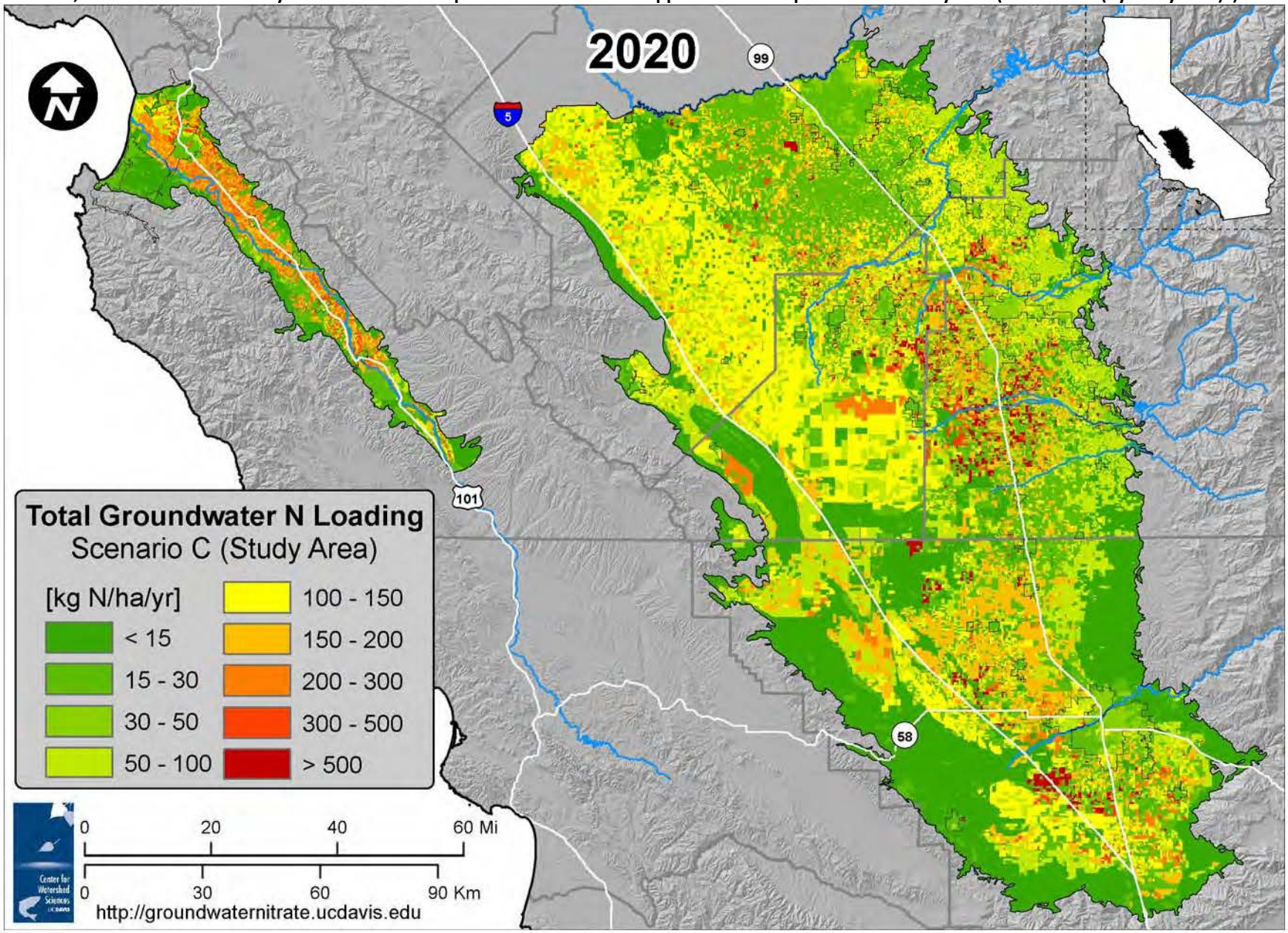
Appendix Figure 106. Total nitrogen loading to groundwater from all sources, in 1990. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the study area ("Scenario C (by study area)").



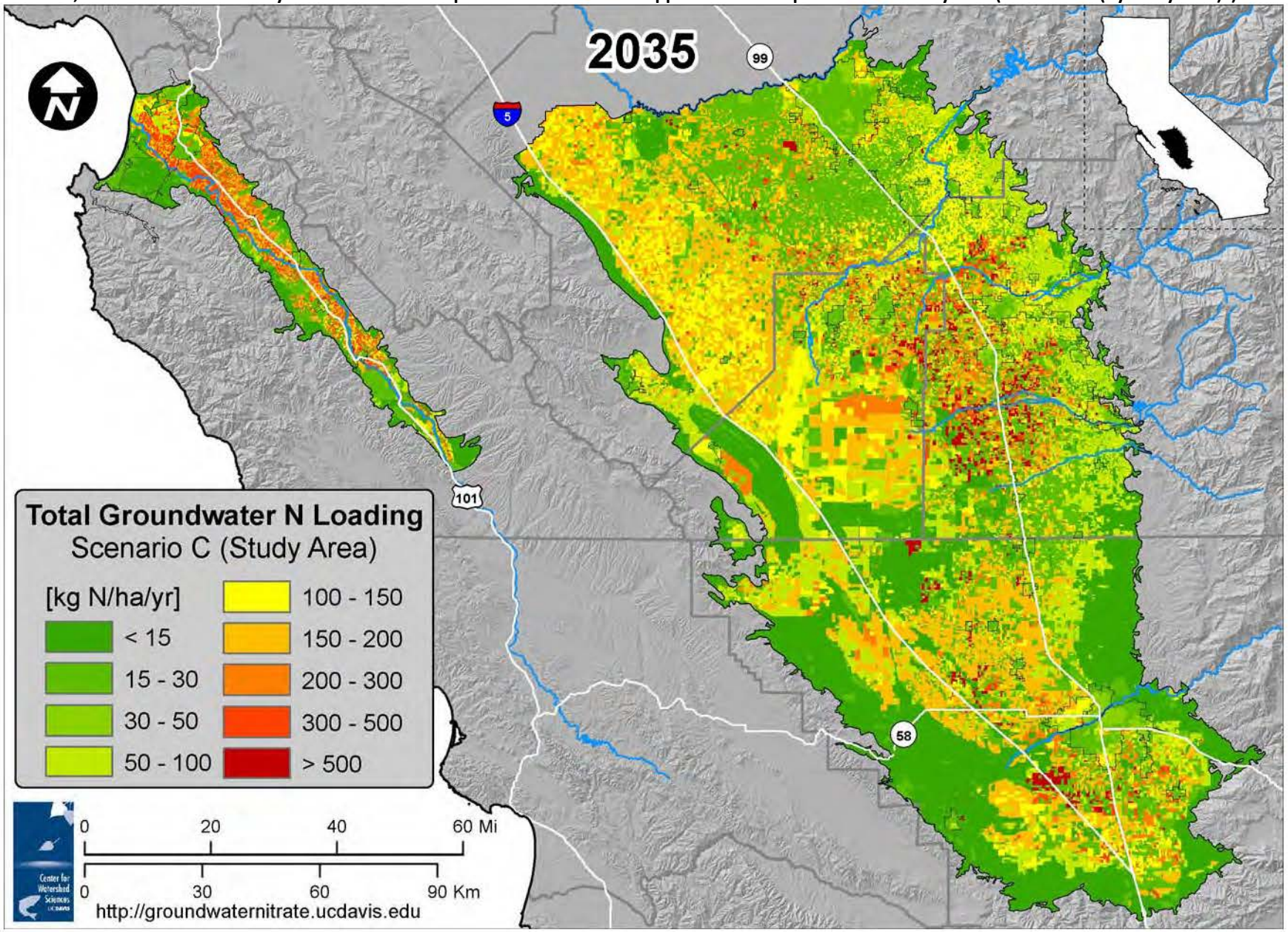
Appendix Figure 107. Total nitrogen loading to groundwater from all sources, in 2005. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the study area ("Scenario C (by study area)").



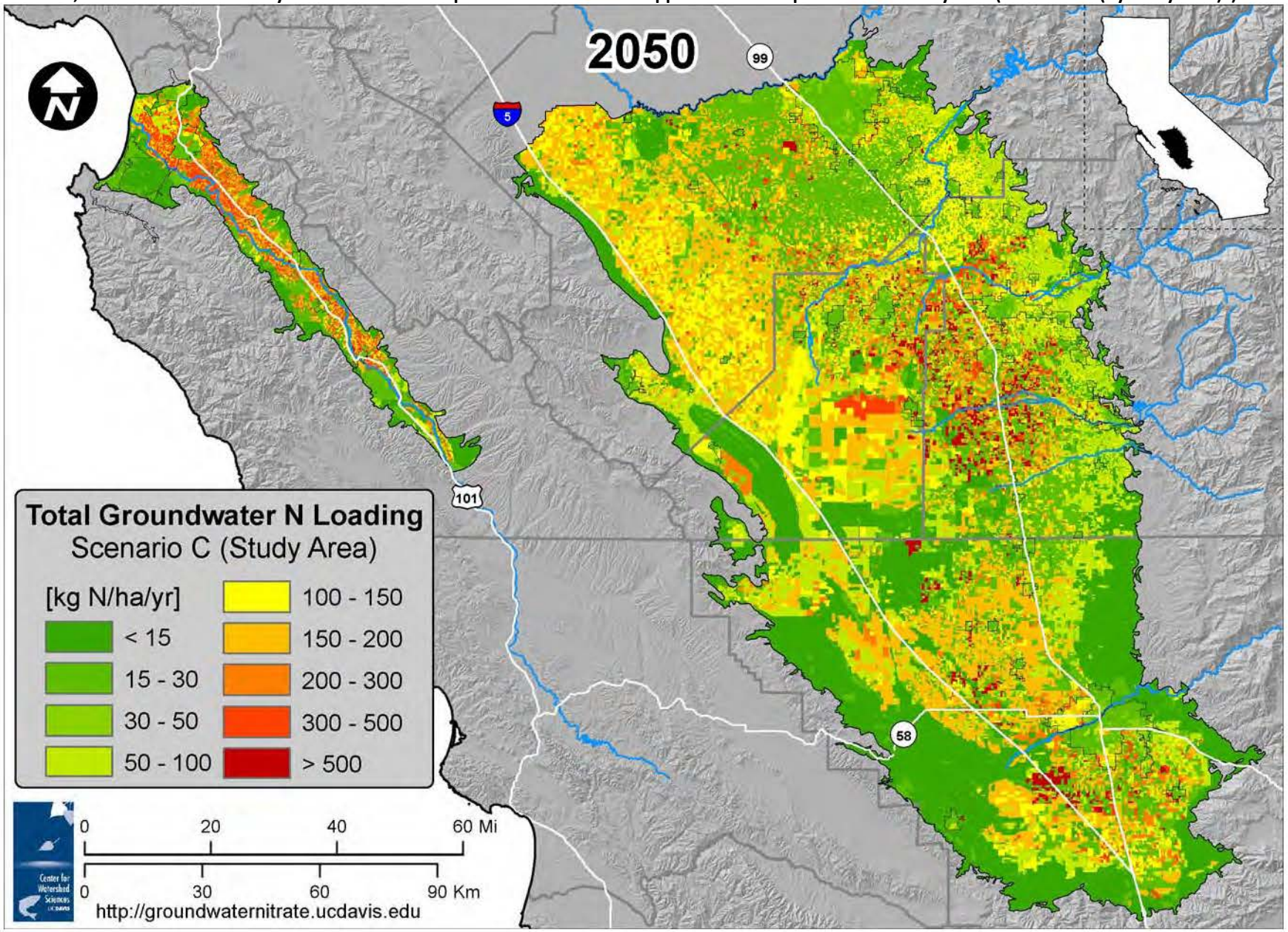
Appendix Figure 108. Total nitrogen loading to groundwater from all sources, in 2020. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the study area ("Scenario C (by study area)").



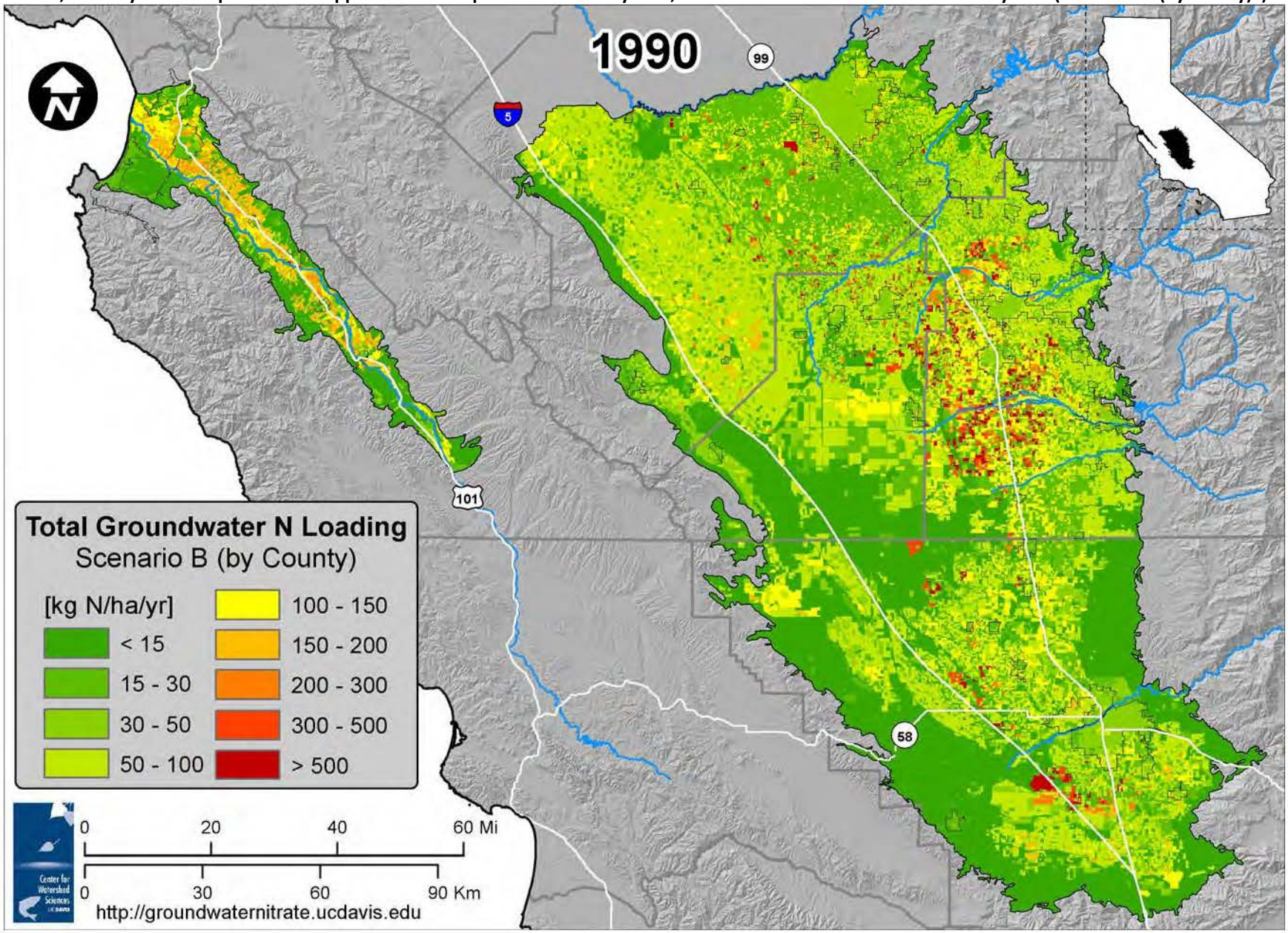
Appendix Figure 109. Total nitrogen loading to groundwater from all sources, in 2035. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the study area ("Scenario C (by study area)").



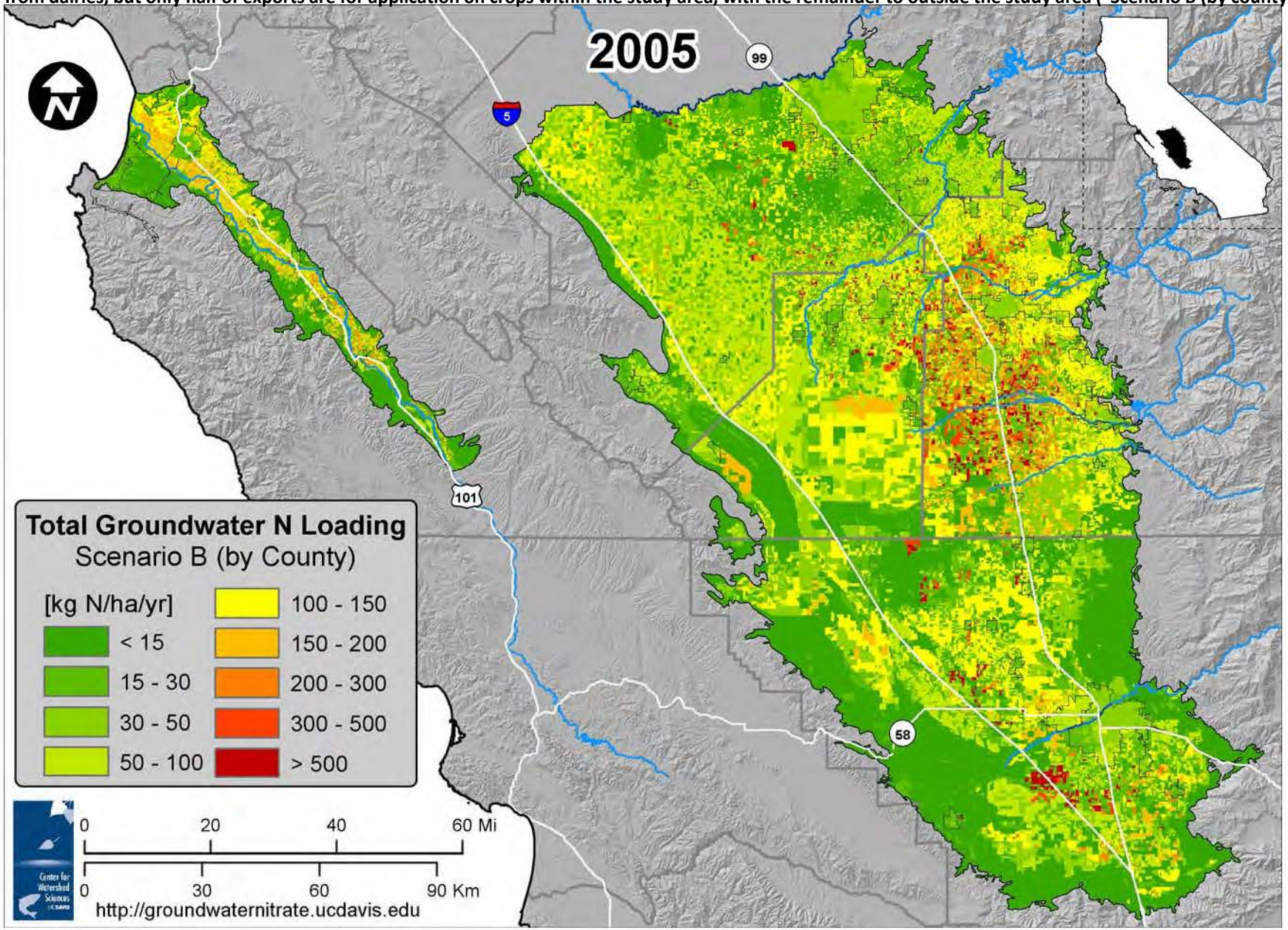
Appendix Figure 110. Total nitrogen loading to groundwater from all sources, in 2050. It is assumed that dairy animals are in corrals or freestalls. In this scenario, 38% of all excreted dairy animal manure is exported from dairies for application on crops within the study area ("Scenario C (by study area)").



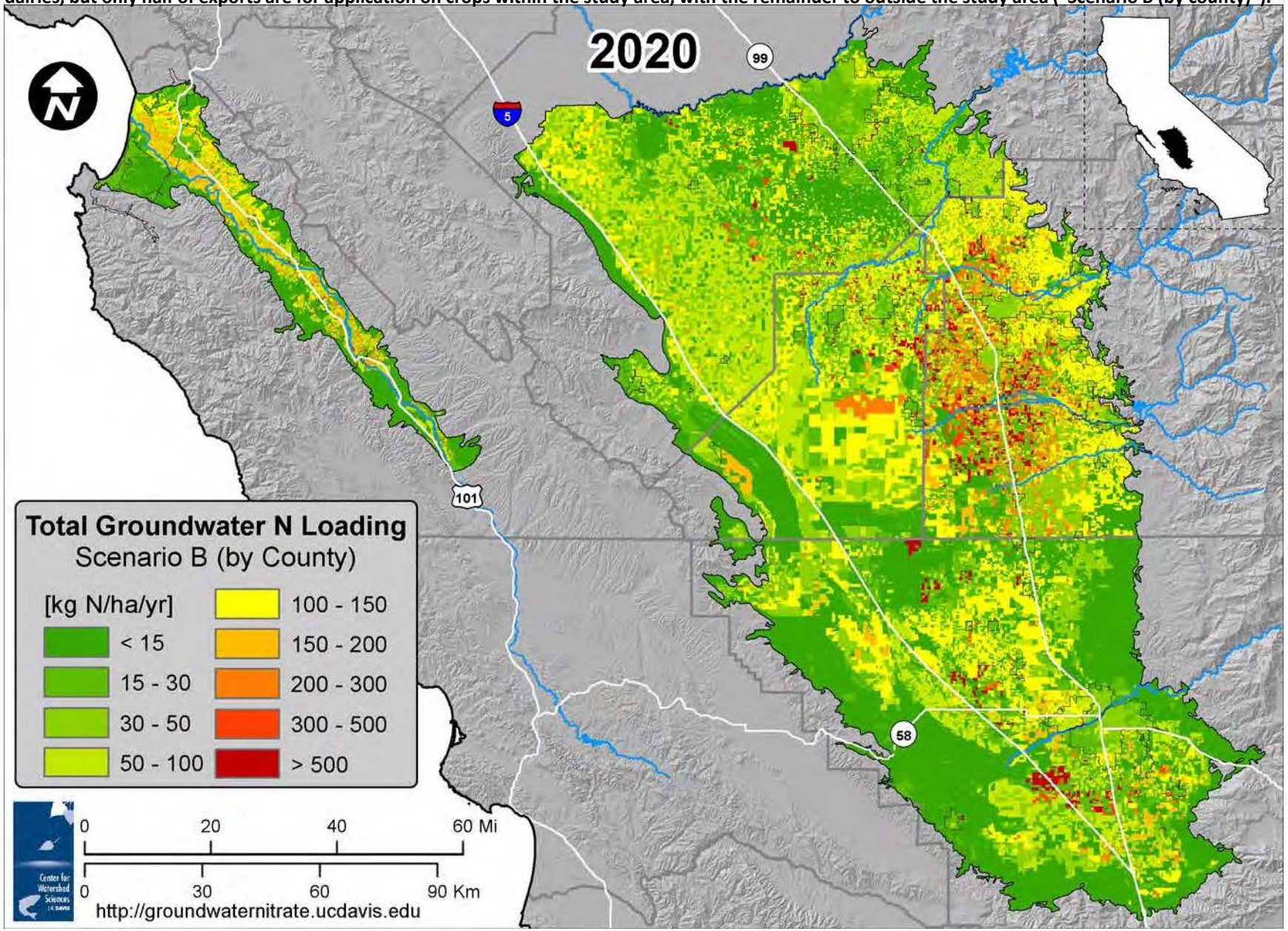
Appendix Figure 111. Total nitrogen loading to groundwater from all sources, in 1990. In this scenario, 38% of all excreted dairy animal manure is exported from dairies, but only half of exports are for application on crops within the study area, with the remainder to outside the study area ("Scenario B (by county)").



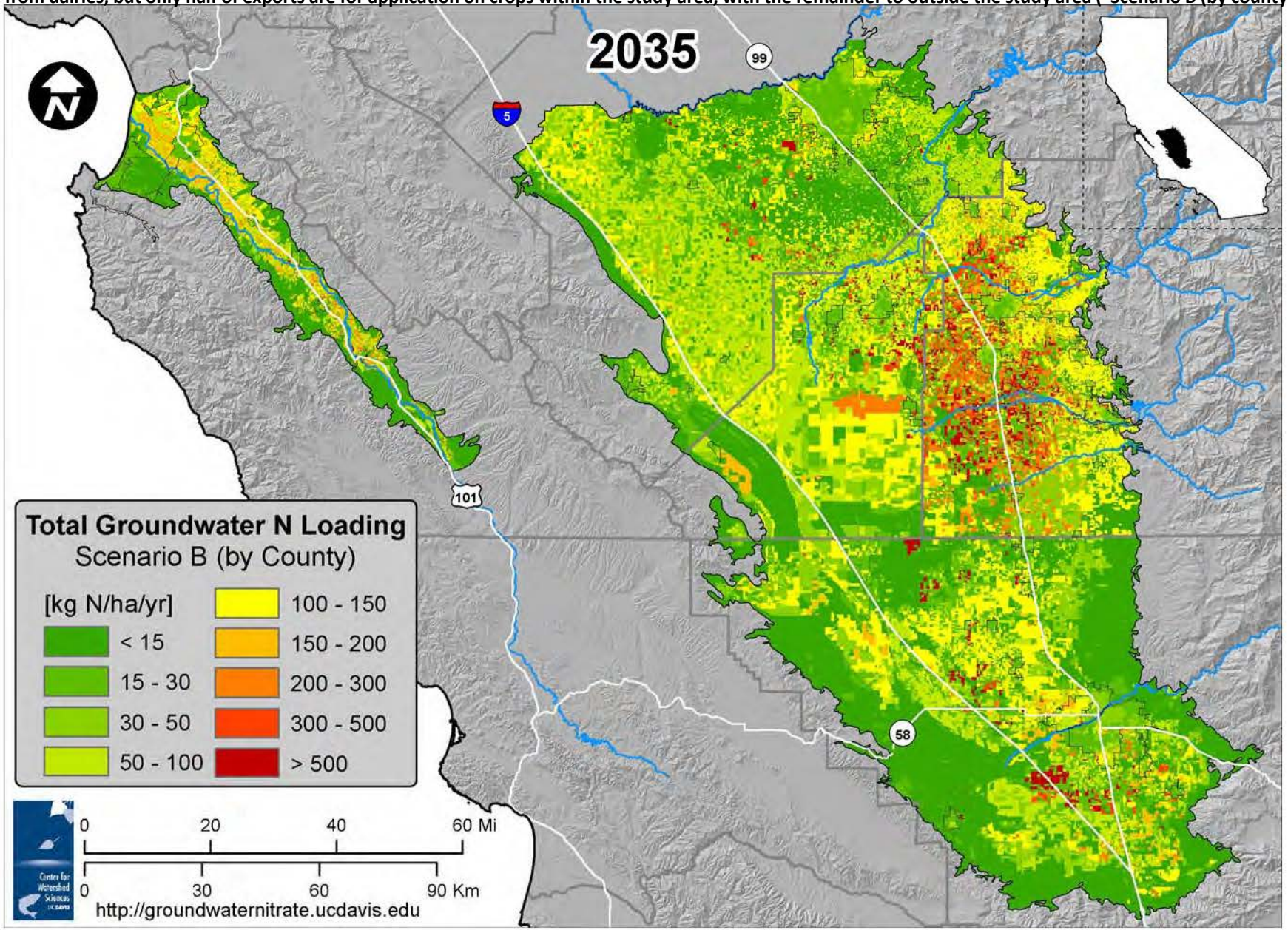
Appendix Figure 112. Total nitrogen loading to groundwater from all sources, in 2005. In this scenario, 38% of all excreted dairy animal manure is exported from dairies, but only half of exports are for application on crops within the study area, with the remainder to outside the study area ("Scenario B (by county)").



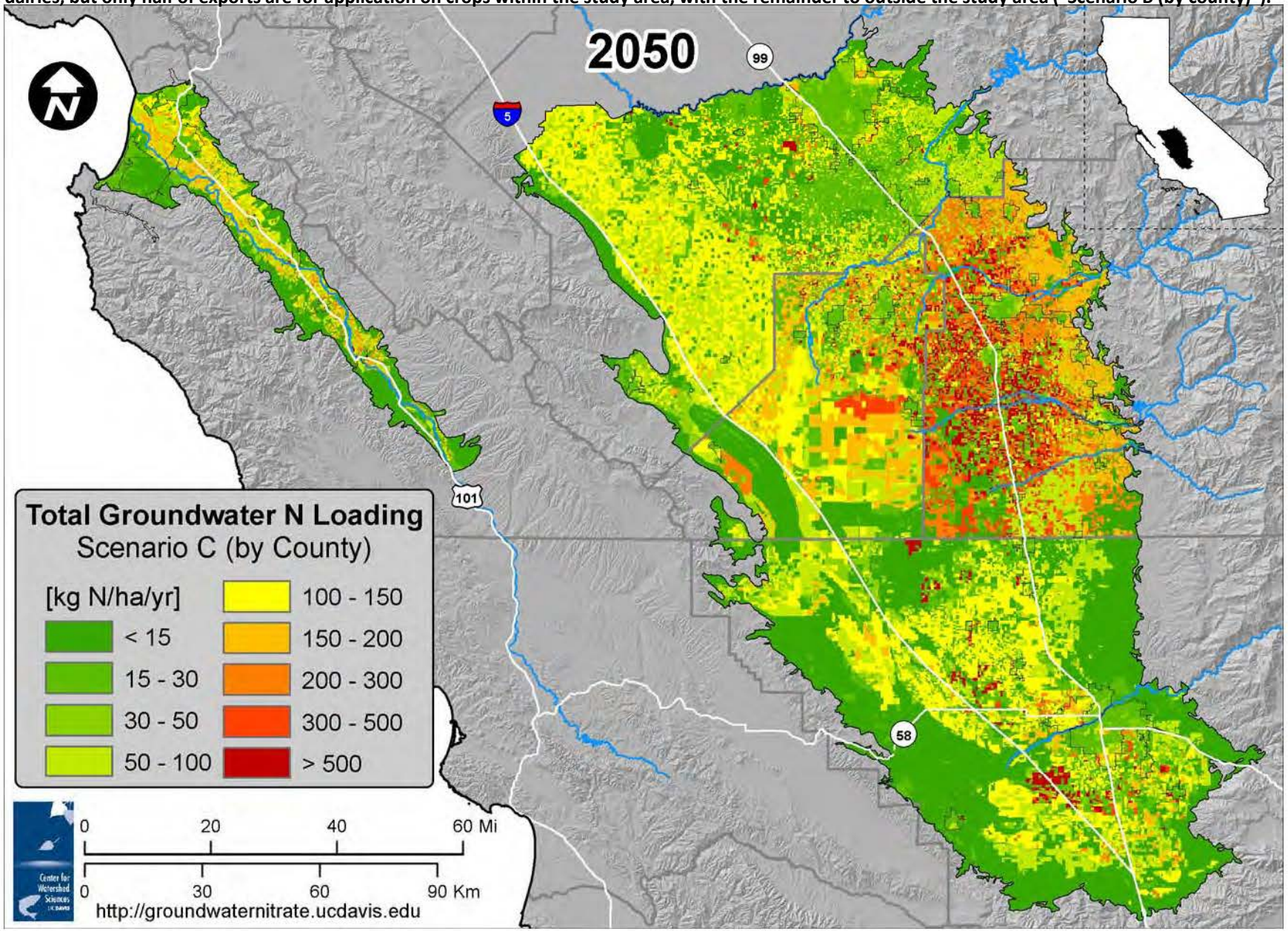
Appendix Figure 113. Total nitrogen loading to groundwater from all sources, in 2020. In this scenario, 38% of all excreted dairy animal manure is exported from dairies, but only half of exports are for application on crops within the study area, with the remainder to outside the study area ("Scenario B (by county)").



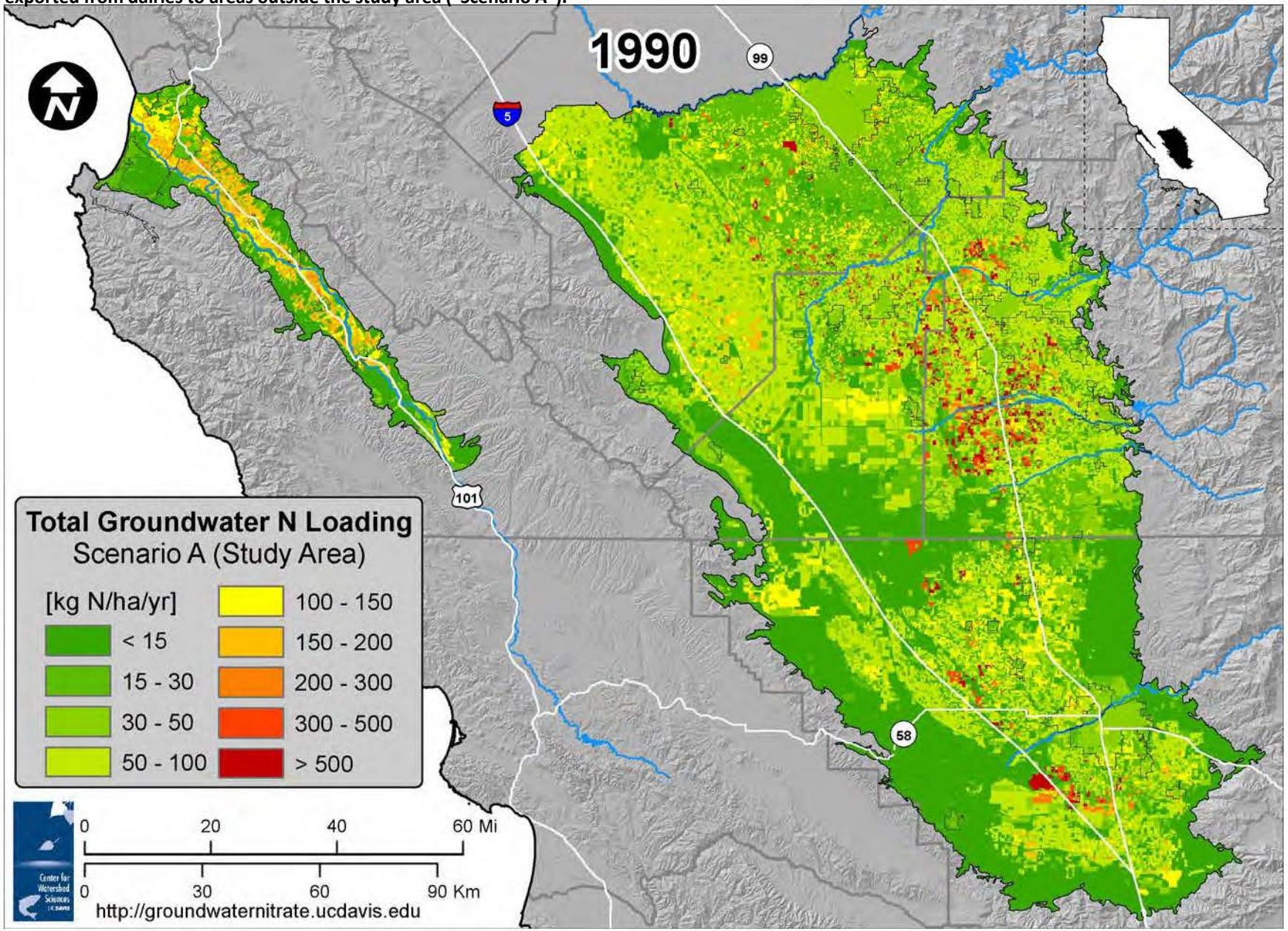
Appendix Figure 114. Total nitrogen loading to groundwater from all sources, in 2035. In this scenario, 38% of all excreted dairy animal manure is exported from dairies, but only half of exports are for application on crops within the study area, with the remainder to outside the study area ("Scenario B (by county)").



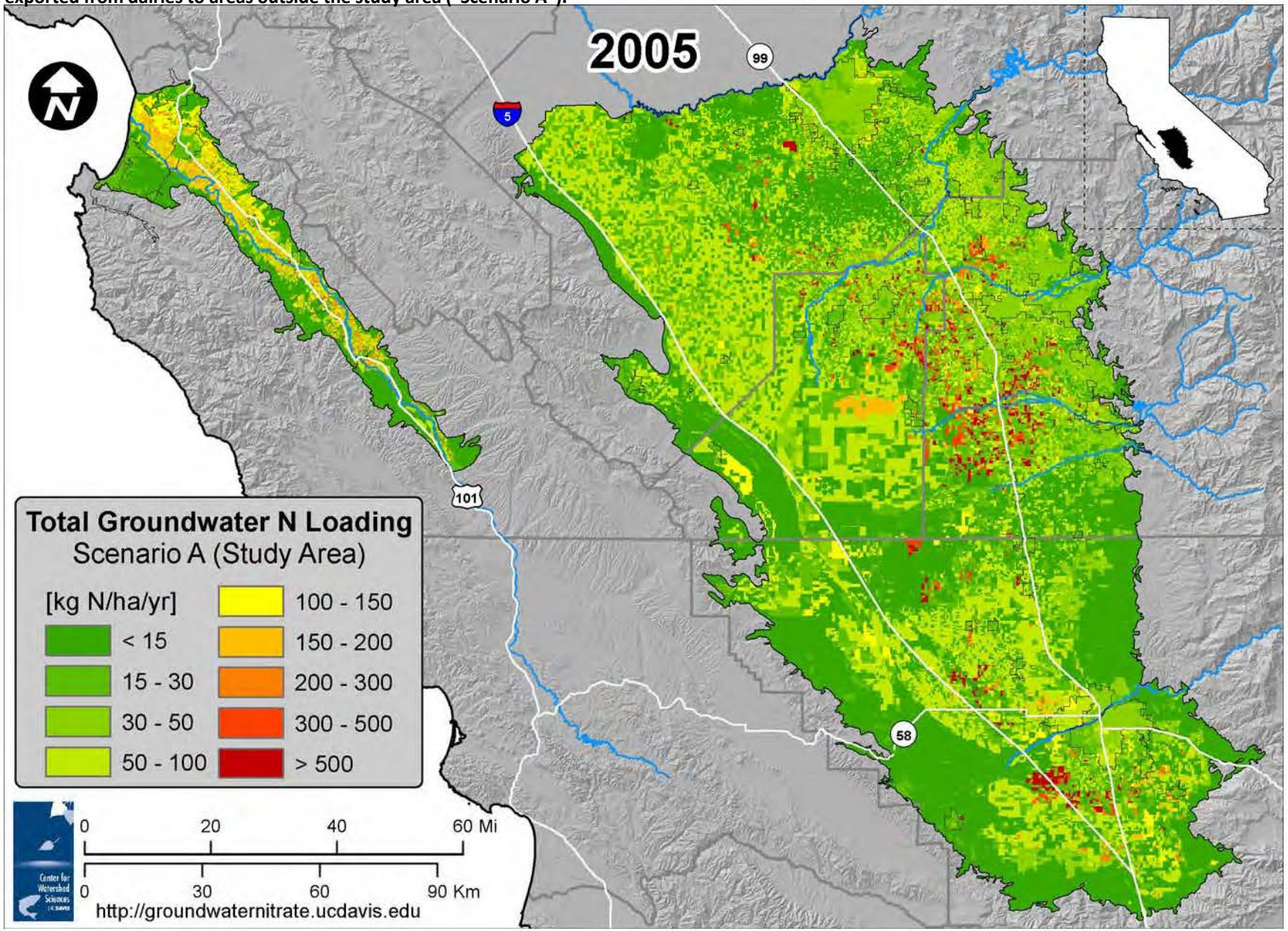
Appendix Figure 115. Total nitrogen loading to groundwater from all sources, in 2050. In this scenario, 38% of all excreted dairy animal manure is exported from dairies, but only half of exports are for application on crops within the study area, with the remainder to outside the study area ("Scenario B (by county)").



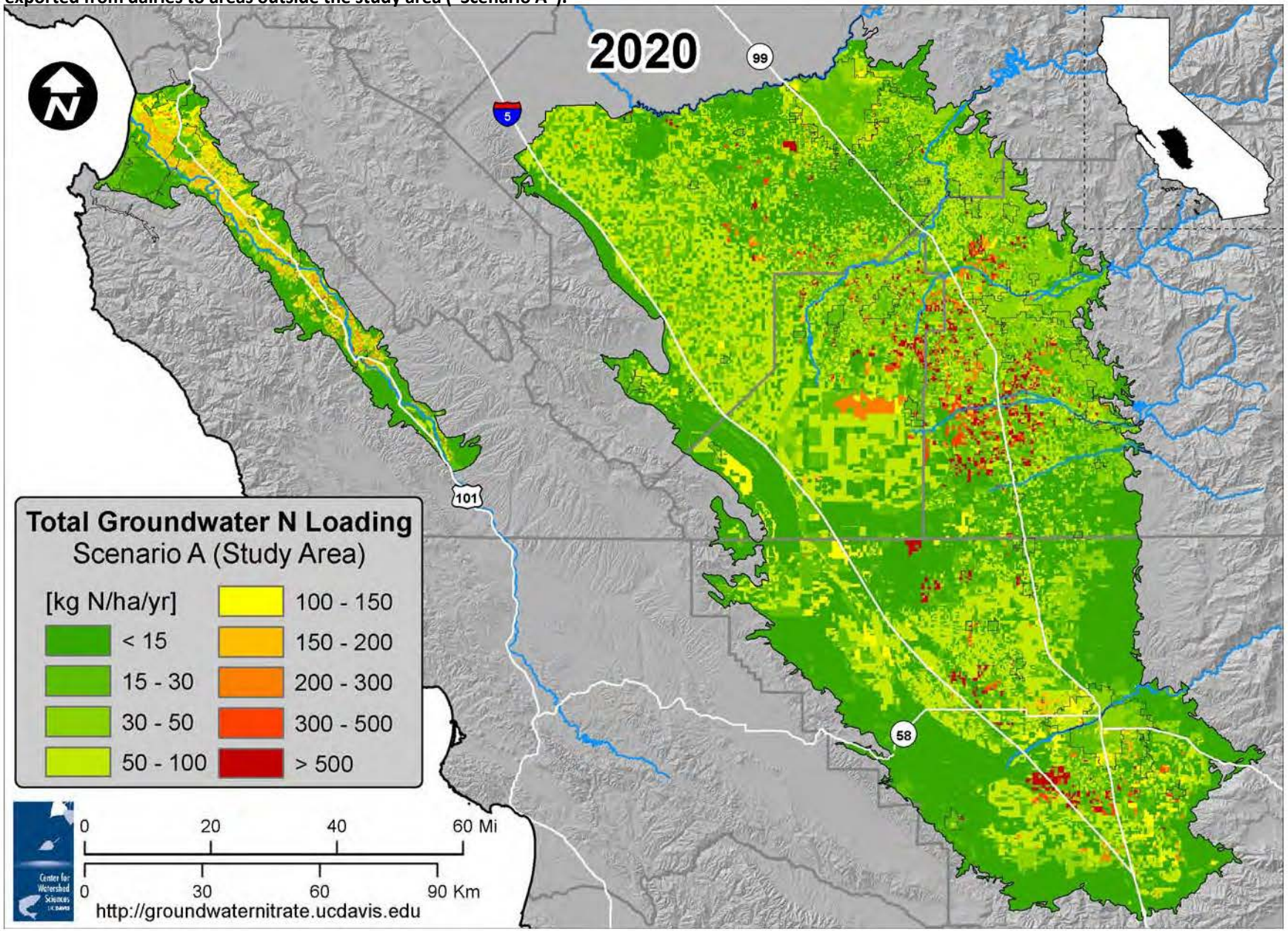
Appendix Figure 116. Total nitrogen loading to groundwater from all sources, in 1990. In this scenario, 15.2% of all excreted dairy animal manure is exported from dairies to areas outside the study area ("Scenario A").



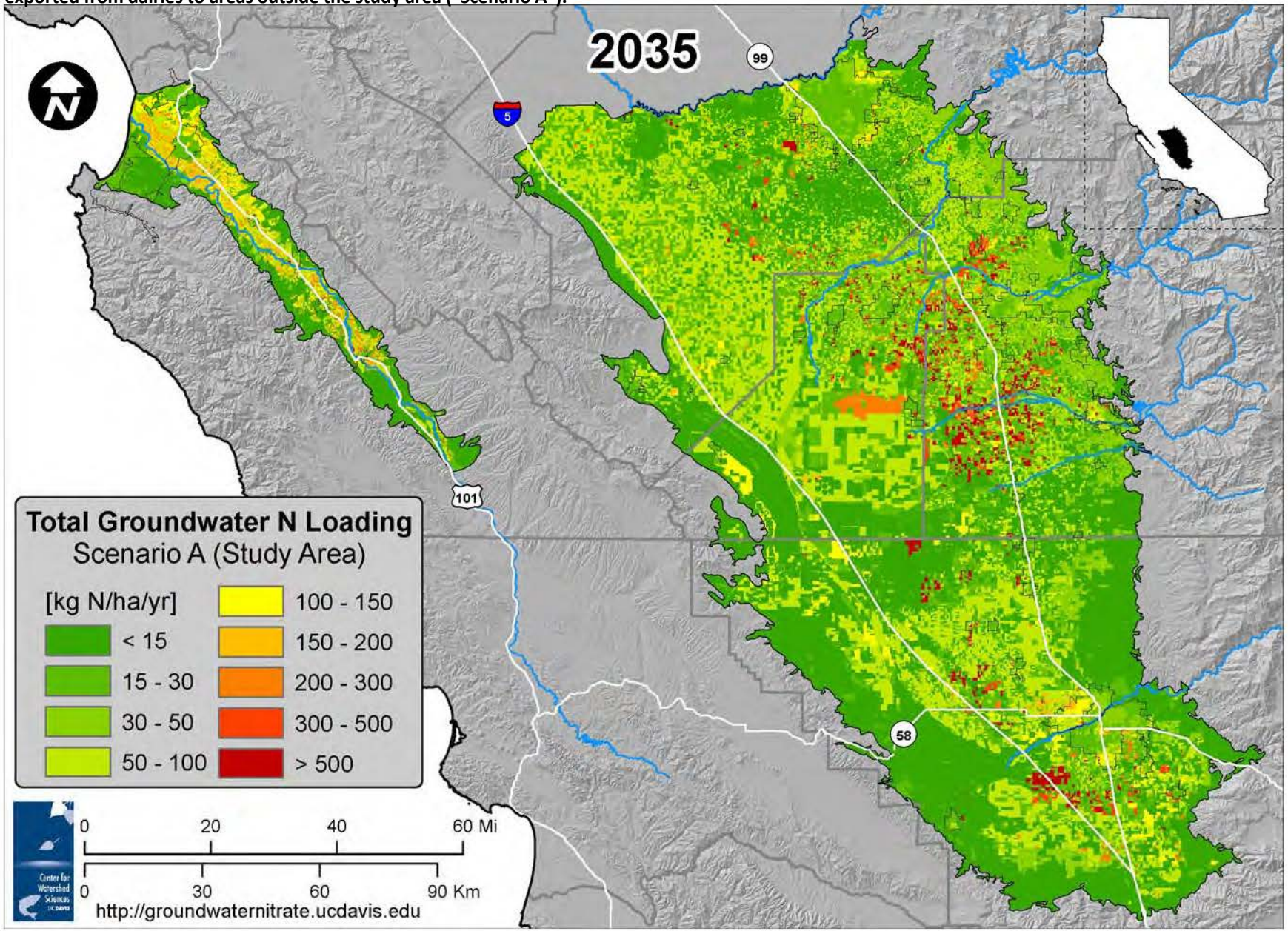
Appendix Figure 117. Total nitrogen loading to groundwater from all sources, in 2005. In this scenario, 38% of all excreted dairy animal manure is exported from dairies to areas outside the study area ("Scenario A").



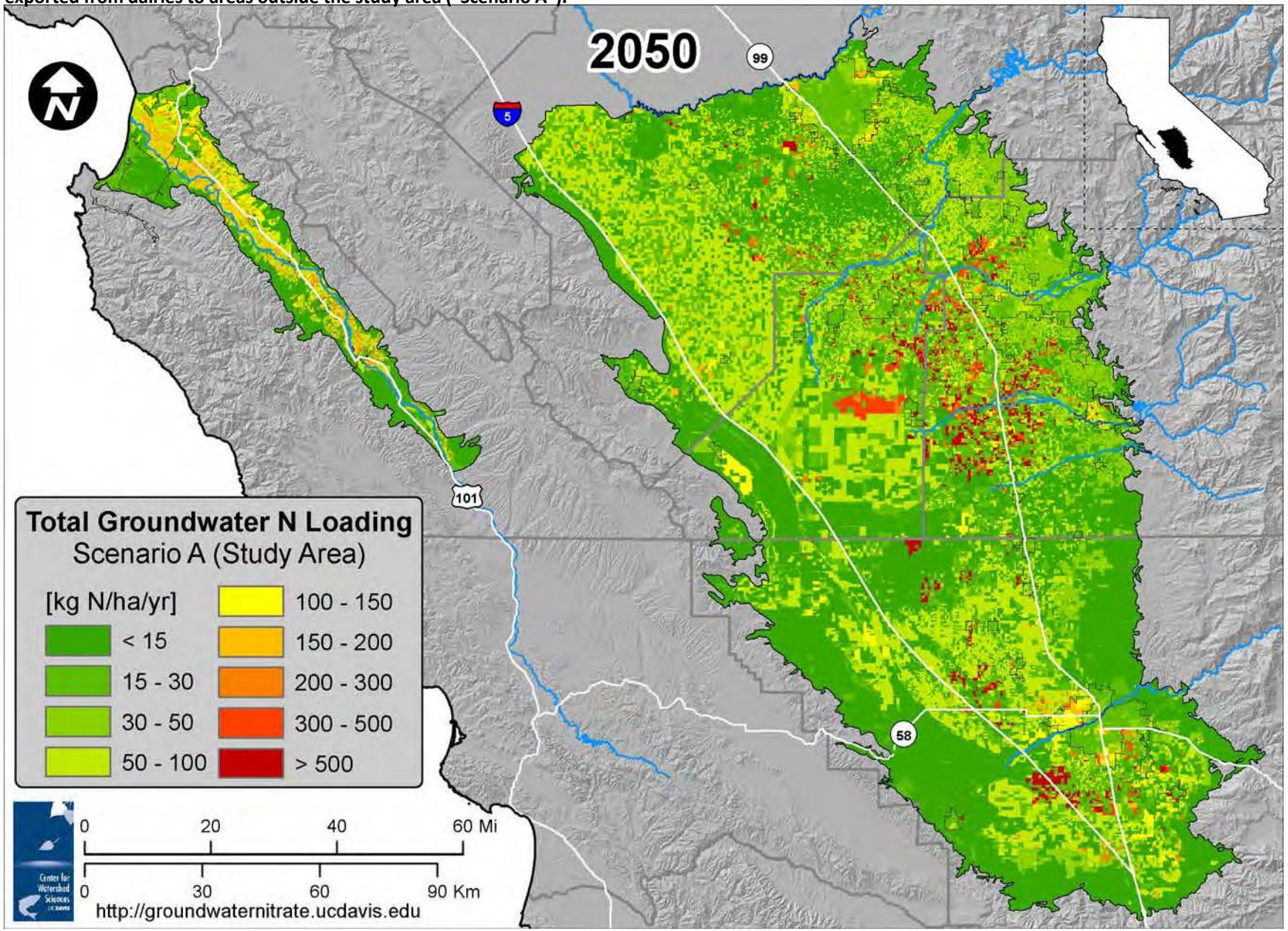
Appendix Figure 118. Total nitrogen loading to groundwater from all sources, in 2020. In this scenario, 38% of all excreted dairy animal manure is exported from dairies to areas outside the study area ("Scenario A").



Appendix Figure 119. Total nitrogen loading to groundwater from all sources, in 2035. In this scenario, 38% of all excreted dairy animal manure is exported from dairies to areas outside the study area ("Scenario A").



Appendix Figure 120. Total nitrogen loading to groundwater from all sources, in 2050. In this scenario, 38% of all excreted dairy animal manure is exported from dairies to areas outside the study area ("Scenario A").



Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 3:

Nitrogen Source Reduction to Protect Groundwater Quality

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
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Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 34719

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Technical Report 3

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Report for the State Water Resources Control Board Report to the Legislature

Prepared by:

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Tulare Lake Basin and Salinas Valley Pilot Studies

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Adminsitrative Record
Page 34720

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Contents

Tables v

Figures vi

Acronyms and Abbreviations viii

Unit Conversions ix

Summary 1

Reducing Nitrate Leaching Losses from Irrigated Cropland and Livestock Operations 1

Reducing Nitrate Leaching from Turfgrass in Urban Areas 4

Reducing Nitrate Leaching from Municipal Wastewater Treatment and Food Processing Plants 4

Reducing Nitrate Contributions from Leaking Sewer Pipes and Septic Systems 5

Reducing Nitrate Transfer and Loading from Wells 5

1 Introduction 6

2 Reducing Agricultural Nitrate Loading 8

2.1 Introduction 9

2.2 Methods 11

2.2.1 Approach to Identifying Potential Nitrate Reduction Strategies 11

2.2.2 Crop Management Practice Scenarios 12

2.2.3 Leaching Vulnerability Assessment 12

2.3 Management Measures (MM) and Recommended Practices (RP) for Reducing Movement of Nitrate to Groundwater from Crop Operations 18

2.3.1 Design and Operate Irrigation and Drainage Systems to Decrease Deep Percolation 20

2.3.2 Manage Crop Plants to Capture More Nitrate and Decrease Deep Percolation 38

2.3.3 Manage Nitrogen Fertilizer and Manure to Increase Crop Nitrogen Use Efficiency 42

2.3.4 Improve Storage and Handling of Fertilizer Materials and Manure 53

2.4 Management Measure Summary 57

Adminsitrative Record
Page 34722

2.4.1 Commonly utilized practices 61
2.4.2 Less Commonly Utilized Practices 63
2.5 Crop Management Practice Scenarios 69
2.5.1 Vegetable Crops 70
2.5.2 Tree Fruits, Nuts and Vines 73
2.5.3 Field Crops, Grain, and Hay 75
2.5.4 Forage and Silage Crops Receiving Manure 76
2.6 Vulnerability Assessment 77
2.6.1 Crops 78
2.6.2 Soils 79
2.6.3 Irrigation 80
2.6.4 Overall Hazard Index: Crop, Soil, and Irrigation 82
2.7 Conclusions 88
3 Agro-Economic Analysis of Nitrogen Source Reductions 91
3.1 Introduction 92
3.2 Crop Selection and Area of Study 94
3.3 Model 95
3.3.1 Conceptual Framework of Model: Nitrogen Use Efficiency, Nitrogen Surplus, and Irrigation Efficiency 95
3.3.2 Model Formulation 98
3.3.3 Water Capital Investments versus Irrigation Efficiency 100
3.3.4 Investments in and Costs of Increasing Nitrogen Use Efficiency 100
3.3.5 Nitrate Loading to Groundwater from Agricultural Production 102
3.3.6 Policy Simulations for Nitrogen Use Efficiency 103
3.4 Model Datasets 105
3.4.1 Irrigation and Cost Data 105

Adminsitrative Record
Page 34723

3.4.2 Nitrogen Use and Cost Data 105
3.5 Modeling Results 110
3.5.1 Policy Modeling 110
3.5.2 Modeling Results 110
3.5.3 A Tax on Nitrogen 117
3.5.4 A Penalty for Nitrogen Exceeding a Given Threshold 118
3.5.5 A Sensitivity Test 118
3.5.6 Limitations 119
3.6 Conclusions 120
4 Reducing Urban Landscape Nitrate Loading 121
5 Reducing Domestic and Urban Wastewater Nitrate Loading 122
5.1 Introduction 123
5.2 Nitrogen Reduction from Wastewater Treatment and Food Processing Facilities 124
5.2.1 Nitrogen Reduction – Methodology 124
5.2.2 Reducing Nitrogen Loading from Wastewater Treatment Plant Discharge – Nutrient Removal 125
5.2.3 Reducing Nitrate Loading from Food Processor Discharge 128
5.2.4 Reduction of Nitrate Loading from Wastewater Treatment Plants and Food Processors – Treatment Costs 129
5.2.5 Reduction of Nitrate Loading from Wastewater Treatment Plants and Food Processors in the Tulare Lake Basin and Salinas Valley 130
5.3 Methods and Cost of Sewer Leakage Prevention and Reduction 133
5.4 Reducing Nitrogen Leaching from Septic Systems 134
5.4.1 Source Separation Technology for Nitrogen Reduction 134
5.4.2 Post-septic Tank Nitrification/Denitrification 136
6 Reducing Nitrate Loading from Active Wells, Dry Wells, and Abandoned or Destroyed Wells 138
6.1 Introduction 139

Adminsitrative Record
Page 34724

6.2 Reducing Nitrate Loading from Direct Inflow into Active Wells 140

6.2.1 Implementation and Cost 140

6.2.2 Benefits 140

6.3 Reducing Nitrate Loading from Direct Inflow into Abandoned or Dry Wells and from Aquifer-to-Aquifer Leakage in Abandoned Wells 141

6.3.1 Implementation and Cost 142

6.3.2 Benefits 143

6.4 Reducing Nitrate Loading from Aquifer-to-Aquifer Leakage in Active Wells 144

6.4.1 Implementation and Cost 145

6.4.2 Benefits 146

6.5 Discussion and Promising Actions 147

7 Source Reduction: Overall Conclusions 150

Literature Cited 153

Section 2 Reducing Agricultural Nitrate Loading 153

Section 3 Agro-economic Analysis of Nitrogen Source Reductions 165

Section 4 Reducing Urban Landscape Nitrogen Loading 169

Section 5 Reducing Domestic and Urban Wastewater Nitrogen Loading 169

Section 6 Reducing Nitrogen Loading from Active, Dry, and Abandoned or Destroyed Wells 172

Adminsitrative Record
Page 34725

Tables

Table 1. Agricultural management measures available to decrease nitrate leaching to groundwater.....19

Table 2. Summary of management practice current extent of use and barriers to increased adoption. . 57

Table 3. Example management bundle: Lettuce. 71

Table 4. Example management bundle: Processing tomatoes. 72

Table 5. Example management bundle: Almonds. 73

Table 6. Example management bundle: Peaches. 74

Table 7. Example management bundle: Cotton. 75

Table 8. Example management bundle: Silage Corn. 76

Table 9. Crop groups and land areas for modeling economics of nitrogen source reductions 94

Table 10. Rules and justification to standardize data extraction from UC ARE Cost and Return Studies..108

Table 11. Costs of improving N fertilization practices..... 109

Table 12. Groundwater nitrogen load reduction scenarios and associated changes in total applied water, annual net revenues, irrigated land area, and applied nitrogen.. 112

Table 13. Treatment processes and associated bacterial species in wastewater treatment for nitrogen removal. 126

Table 14. Example of configurations and processes for biological nutrient removal in wastewater treatment. 127

Table 15. Physico-chemical technologies for nutrient removal from wastewater. 128

Table 16. Costs of BNR for example configurations with design capacity ranging from 1 to 10 mgd..... 130

Table 17. Number of facilities with > 250 kg N/ha/yr and > 500 kg N/ha/yr in discharges to irrigated agriculture and > 10 mg/L N in discharges to percolation basins. 131

Table 18. Estimated costs of BNR (U.S. EPA 2008) for at-risk facilities, based on wastewater flow. 132

Table 19. Summary of the estimated potential costs and benefits of N loading reduction measures associated with wells. 148

Adminsitrative Record
Page 34726

Figures

Figure 1. The UC Nitrate Hazard Index multiplicative matrix 15

Figure 2. Changes in water distribution uniformity from a single sprinkler nozzle when operated at different pressures. 32

Figure 3. Distribution uniformity of overlapped sprinklers for A) Low wind condition at regular pressure, B) High wind condition with lateral perpendicular to wind direction, C) High wind condition with lateral parallel to wind direction. 33

Figure 4. Alternating placement of lateral lines over successive irrigations. 34

Figure 5. Contrasting N uptake curves for corn and wheat. 42

Figure 6. Study area crop distribution categorized by the crop group's hazard index values 79

Figure 7. Soil hazard index value distribution for irrigated agricultural soil series 80

Figure 8. Irrigation hazard index value distribution. 82

Figure 9. Overall hazard index calculated for the study area fields 84

Figure 10. Distribution HI values by percent area for each component used in the determination of the overall hazard index values. 85

Figure 11. Composite hazard index values 86

Figure 12. Effective nitrogen versus applied nitrogen by management practice bundle 96

Figure 13. Tradeoff curve between Nitrogen Use Efficiency as Partial Nutrient Balance (PNB) and marginal costs per unit area to increase PNB 97

Figure 14. Sample tradeoffs between capital investments and irrigation efficiency for selected crops for a constant crop yield. 98

Figure 15. Simplified representation of crop production tradeoff surface between effective nitrogen use and land use. 101

Figure 16. Water use efficiency for the Tulare Lake Basin and the Salinas Valley at different levels of reduction in nitrate load to groundwater 111

Figure 17. Nitrogen use efficiency for the Tulare Lake Basin and the Salinas Valley at different levels of reduction in nitrate load to groundwater. 111

Figure 18. Relative applied water, net revenues, irrigated land, nitrogen applied, and nitrate load reduction as a function of reductions in the nitrate loading to groundwater by agriculture 113

Adminsitrative Record
Page 34727

Figure 19. Percentage reduction in net revenues estimated from different levels of reduction in nitrate loading to groundwater. 114

Figure 20. Average and marginal net revenue losses per kg of nitrate load reduction to groundwater. 115

Figure 21. Cumulative change in cropping patterns with respect to base conditions for selected crops in the Tulare Lake Basin. 116

Figure 22. Cumulative change in cropping patterns with respect to base conditions for selected crops in the Salinas Valley. 116

Figure 23. Average percent change, across all crops, in applied nitrogen and investment in nitrogen use efficiency per unit area with respect to current conditions. 117

Figure 24. Net revenue losses for both the TLB and the SV per tax rate level. 118

Adminsitrative Record
Page 34728

Acronyms and Abbreviations

BNR	Biological Nutrient Removal
CES	Constant Elasticity of Substitution
CIMIS	California Irrigation Management Information System
CPNB	Marginal cost of improving nitrogen use efficiency as Partial Nutrient Balance
DU	Distribution Uniformity
DWR	California Department of Water Resources
EDR	Electrodialysis Reversal
ET	Evapotranspiration
FP	Food Processor
GRP	Glass Reinforced Plastic pipe
HDPE	High Density Polyethylene pipe
HI	Hazard Index (Center for Watershed Sciences)
MM	Management Measure
NUE	Nutrient Use Efficiency
O & M	Operations and Maintenance
PMP	Positive Mathematical Programming
PNB	Partial Nutrient Balance
POTW	Publically Owned Treatment Works
PVC	Polyvinyl Chloride pipe
RO	Reverse Osmosis
RP	Recommended Practice
SBR	Sequencing Batch Reactor
SSURGO	Soil Survey Geographic Database (NRCS)
SV	Salinas Valley
SWAP	Statewide Agricultural Production Model
TLB	Tulare Lake Basin

Adminsitrative Record
Page 34729

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-).

To convert from:

Nitrate-N (NO_3-N)	→	Nitrate (NO_3^-)	multiply by 4.43
Nitrate (NO_3^-)	→	Nitrate-N (NO_3-N)	multiply by 0.226

Adminsitrative Record
Page 34730

Summary

Although reduction of anthropogenic loading of nitrate to groundwater aquifers will not reduce contamination in the short term (due to long travel times), reduction efforts are a critical component of any long term solution to the problem of high nitrate in drinking water. Technologies are available for reducing the transfer of nitrate to groundwater from surface sources. Such technologies involve (1) reducing the amount of nitrogen (N) discharged or applied to the land and (2) controlling the amount of water applied to land which serves as the carrier of nitrate. Many of these source control methods require changes in land management and upgrading of infrastructure.

In this report, we have considered the methods and associated costs for reduction of nitrate leaching losses from the major anthropogenic sources of nitrate loading to groundwater in the two study regions. The sources considered here are irrigated cropland, livestock operations, turfgrass and other urban landscaping, effluent from municipal wastewater treatment plants and food processing plants, sewers, septic systems, and abandoned, dry, and active wells.

Costs for mitigation or abatement vary widely and can sometimes be difficult to estimate. In particular, the amount of nitrate leached from irrigated crop fields (the largest source) is determined by a complex interaction of N cycle processes, soil properties, and farm management decisions; it is therefore virtually impossible to generalize mitigation costs per unit of nitrate load decrease, allowing for only broad estimations.

Reducing Nitrate Leaching Losses from Irrigated Cropland and Livestock Operations

Reduction of nitrate leaching from cropland and livestock and operations can be achieved through changes in farm management that result in improvements in crop nitrogen use efficiency (NUE), and by proper storage and handling of manures and fertilizers. NUE, sometimes referred to as partial nitrogen balance (PNB), refers to the percentage of N applied to cropland (from all sources) that is recovered by the crop and is therefore not lost to the atmosphere or to surface and groundwater.

To determine the actions needed to reduce nitrate leaching losses from crop and livestock operations, we reviewed technical and scientific literature to compile a list of practices that are known or theorized to improve crop NUE. We then convened crop-specific expert panels to review and revise this list of practices. We relied on input from panel members to estimate the current extent of use of each practice within the study area and to help identify the main barriers to expanded adoption.

To establish the proportion of acreage in the study areas that would benefit the most from increased adoption of improved management practices, we conducted a vulnerability assessment. Vulnerability was mapped by use of the UC Nitrate Hazard Index (Wu et al. 2005), which calculates a risk of nitrate leaching based on the crop grown, the irrigation system type in use, and soil characteristics of the field.

Adminsitrative Record
Page 34731

Cropland Nitrate Reduction, Findings

NUE can be increased by optimizing the timing and rates of applied fertilizer N, animal manures, and irrigation water to better match crop needs, and to a lesser extent by modification of crop rotations. Improving the storage and handling of manures, livestock facility wastewaters, and fertilizers also plays a role in nitrate leaching reduction. Although crop recovery of N inputs as low as 33% have been reported, a recent U.S. EPA report estimated that with the adoption of best management practices, NUE could increase by up to 25% of current average values (U.S. EPA Science Advisory Board 2011). While improvements in NUE are possible, there exists a practical upper limit of about 80% crop recovery of applied N (Ibid; Raun & Schepers 2008). This limit is due to the unpredictability of rainfall, the difficulty in predicting the rate of mineralization of organic N in the soil, spatial variability in soil properties, and the need to leach salts from the rootzone.

Note that, while it is certain that the mass of nitrate lost by leaching from the crop root zone can be reduced to well below the rate of loss that has resulted in the currently observed nitrate concentrations in affected aquifers, this does not necessarily mean that the concentration of nitrate can be reduced to the MCL, especially where the sole or main source of aquifer recharge is percolate from irrigated crop fields.

To most effectively reduce the mass of nitrate escaping the crop rootzone, a suite of improved practices is generally required, and these must be chosen according to the unique field situation. There is no one set of management practices that will be the most effective in protecting groundwater quality. The applicability and effectiveness of the practices vary according to field specific variables (crop and soil characteristics, as well as underlying hydrology). That being said, the basic principles of improved management are applicable to all operations, and again include optimizing application rates and timing of water, fertilizer, and manure applications to better match crop need, making adjustments to crop rotation strategies, and improving storage and handling of fertilizers and manure. Additionally, it is critical that manure-N be accounted for by reducing inorganic N applications accordingly.

Several of the practices known to reduce nitrate leaching have been adopted in recent years by farmers in the study area, representing a positive change from past practices that have contributed to current groundwater nitrate concentrations. While it is clear that improved management reduces the rate at which nitrate is leached, data are lacking that would allow an estimation of how the rate of leaching has changed as agricultural management has improved and to what degree additional management changes will affect loading rates. Management practices that are not widely used are generally associated with multiple barriers to adoption by farmers. These include higher operating or capital costs, perceived or real risks to crop quality or yield, conflicting farm logistics, and constraints associated with land tenure. Additional significant barriers include inadequate farmer education and insufficient research to adapt practices to local conditions.

Based on soil characteristics of the field, crop species grown, and irrigation system type in use, we estimate that approximately 52% of irrigated cropland in the Salinas Valley and 35% of such land in the Tulare Lake Basin is susceptible to significant nitrate leaching losses. This vulnerability estimate is not indicative of actual nitrate loading to groundwater and does not consider depth to groundwater, the

Adminsitrative Record
Page 34732

location of land relative to sensitive aquifers, and does not consider farmers' actual management practices. However, we can expect improved management in these areas to have the greatest positive impact on nitrate leaching.

A maximum net benefit modeling approach was used to estimate relative costs of policies aimed at improving NUE. One of the most uncertain input parameters employed in the economic modeling of agricultural nitrate source reductions is the actual cost of improving nitrogen use efficiency. Our modeling results suggest that modest reductions in N fertilizer application rates and increased adoption of related improved practices would increase production costs only slightly (assuming sufficient education in N management techniques). These modest reductions are thus deemed to be economically feasible without significant reductions in total irrigated area. As larger load reduction strategies are undertaken, the model predicts significant production costs increases, reducing net revenues considerably to the point of reducing net irrigated cropland within the study area. Lower value field crops and low NUE crops are especially vulnerable to area reductions, as higher economic returns will be required to cover the increased operating and capital costs associated with increased efficiency. A simulated nitrogen sales tax indicated such a tax could initiate additional grower motivation to adopt practices that contribute to NUE.

Cropland Nitrogen Reduction, Promising Actions

Expanded efforts to promote the adoption of nitrogen efficient practices are clearly needed. The educational barriers that are associated with many of the identified practices highlight the importance of funding research, education, and outreach activities to better assist farmers in applying best management strategies and nutrient management. The University of California Cooperative Extension (UCCE) and USDA Natural Resource Conservation Service (NRCS), for example, play integral roles in delivering quality educational information to both growers and crop advisors. Research should focus on demonstrating the impact of management practices on NUE, and on adaptation of practices to local or site-specific conditions in the crop rotations and soils that present the greatest risk of nitrate leaching. Such adaptive research should document the impact of improved practices on crop N use efficiency as well as on yield and profitability. Guidance in accounting for the nutrient value of manure-N is especially important. Additionally, research on the costs of increasing NUE would greatly benefit the capacity to better estimate the costs of reducing agricultural nitrate loading to groundwater.

Due to the waste discharge regulations imposed on Central Valley dairies in 2007,[2] transfer of significant amounts of manure from these dairies to other farms is increasing. It is not known which crop species and soils are receiving this manure or how the receiving farmers have integrated the manure into their N fertilization practices. We recommend development of adaptive research and education programs that will promote conversion of solid and liquid dairy manure into forms that meet the food safety and production requirements for a wider range of crop species. Providing guidance to non-dairy farms in co-managing conventional N fertilizers and manure-containing materials is necessary to ensure that the nutrient value of the manure is properly accounted for in fertilization activities.

[2] The General Order for Waste Discharge from Existing Dairies in the San Joaquin Valley, see http://www.waterboards.ca.gov/centralvalley/water_issues/dairies/dairy_program_regs_requirements/index.shtml

Adminsitrative Record
Page 34733

Supporting the development of crop-specific N-accounting methods that allow growers to evaluate their success in achieving high crop nitrogen use efficiency would be beneficial. An example of such a "nitrogen mass balance metric" is the ratio of the amount of N, from all sources, applied to a crop to the amount of N removed in the harvested crop. This serves in lieu of direct estimates of nitrate to the groundwater, which is extremely difficult to monitor at the individual farm or farm field scale.

Finally, additional promising actions include reviewing and further developing methods for identifying cropland areas that are at an increased risk or significantly vulnerable to nitrate leaching. Several methods have been used or are under consideration for doing this in California; some attention should be given to methods that can be used at the farm and county scale for on-farm applied research and technology transfer (outreach) activities. Such a method should include consideration of the spatial soil characteristics, as well as probable monitoring requirements.

Reducing Nitrate Leaching from Turfgrass in Urban Areas

Nitrate leaching from urban turfgrass including golf courses, is often negligible due to the dense plant canopy and perennial growth habit of turf, which results in continuous plant N uptake over a large portion of the year. Fertilizer N applied beyond plant need is often still taken up and utilized by the turf, increasing vegetative growth. However, poor management can lead to a discontinuous canopy and weed presence, wherein nitrate leaching risk increases, especially if growing on permeable soils, if over-irrigated, or if fertilized at high rates during dormant periods. The best strategy to reduce leaching from turfgrass is to simply follow recommended guidelines for the rate and timing of fertilizer application along with proper water management techniques. The University of California Cooperative Extension (UCCE) and UC Integrated Pest Management (UCIPM) program publish such guidelines. The practice of keeping fertilization rates and timing concordant with plant need has the added benefit to professional turfgrass managers of requiring less frequent mowing and a reduction of the volume of clippings requiring disposal. The knowledge and willingness of homeowners and groundskeepers to apply university and industry guidelines is primarily dependent upon funding for outreach efforts.

Reducing Nitrate Leaching from Municipal Wastewater Treatment and Food Processing Plants

Discharges from municipal wastewater treatment and food processing facilities, while not as regionally significant a source of groundwater nitrate as agriculture, do have locally important impacts to groundwater quality. Implementation of N control options for these sources is feasible and could be an important part of a multi-pronged approach.

Cropland application of wastewater treatment and food processing effluents can both reduce direct groundwater contamination as well as reduce the total fertilizer application requirements of such fields as the water and nutrients are effectively treated and recycled. Such wastes should be managed in an agronomic manner, such that the nutrients (especially nitrogen) present in the wastes, are included in the overall nitrogen management plan for the receiving crops. Appropriate agronomic application

practices should be used (especially with respect to application rate and timing). Biological and chemical treatment of these effluents can further reduce their impact on groundwater contamination. Which treatment option to employ depends upon the unique characteristics and limitations of the treatment plants. Costs of biological treatment options likewise vary widely; estimated capital costs for nutrient removal from all wastewater (FPs and WWTPs) for facilities categorized as "at-risk" range from $70 to $266 million depending on if the project is a retrofit or an expansion (associated operations and maintenance costs range from $3.2 - $20 million).

Optimizing wastewater treatment plant and food processing plant operations is another important consideration; limiting nitrogen and total discharge volume through in-plant process modifications may be sufficient for some facilities. Groundwater monitoring is required for many facilities, but the data are largely unavailable since they are not in a digital format. To improve monitoring, enforcement, and abatement efforts related to these facilities, groundwater data need to be more centrally managed and organized digitally.

Reducing Nitrate Contributions from Leaking Sewer Pipes and Septic Systems

Retrofitting of septic system components and sewer pipes represents the main avenue of diminishing loading from these sources. To reduce risks to human health, it is important to replace aging sewer system infrastructure and to ensure proper maintenance; necessary infrastructure upgrades will also reduce N leaching from leaking pipes. Our investigation of pipe-replacement costs indicate that the selection of pipe material is the cost determining factor, ranging from $7 - $55 per linear foot.

Loading from septic systems, a locally significant N source, can be reduced significantly by two approaches. Source separation technology, in which urine (representing approximately 80% of the total N in human waste) is removed from the waste stream and reused as a fertilizer, can be expected to reduce nitrate loading to onsite wastewater treatment systems by about 50%. Costs include separating toilets ($300 - $1100), dual plumbing systems ($2000 - $15,000), as well as storage tank costs, maintenance, pumping, heating, and transport costs (where applicable). The other option, post-septic tank biological nitrification and denitrification treatment, reduces N concentrations below levels achieved via source separation technology, but does not result in a reusable resource. Wood chip bioreactors have been shown to reduce influent nitrate by 74 – 91%, with costs ranging from $10,000 - $20,000 to retrofit existing septic systems.

Reducing Nitrate Transfer and Loading from Wells

Local or state programs and funding to identify and properly destroy abandoned and dry wells is needed to avoid their behaving as nitrate transfer conduits. However, it is also clear that many well owners may not be able to afford the high costs of retrofitting long-screened wells to seal contaminated groundwater horizons. As such we advocate that enforcement of proper well-construction standards for future wells may be more feasible and that expenditures on retrofitting of existing wells should be selected based on individual contamination risks.

Adminsitrative Record
Page 34735

1 Introduction

This report provides an assessment of the currently available technologies and management approaches for minimizing nitrate leaching from irrigated croplands and livestock facilities, turfgrass and other urban landscaping, wastewater treatment plants, food processing plants, sewers, septic systems, and wells. The assessment includes descriptions of the management measures, technologies and infrastructure upgrades that are available for reducing the load from these sources, and, where possible, their costs. Additionally, for cropland, barriers to expanded adoption by farmers of improved management practices are described.

Among the sources of nitrate analyzed in this report, irrigated cropland and livestock operations have the largest footprint and account for a much greater proportion of N in the environment than other sources (see Technical Report 2, Viers et al. 2012). Croplands represent a nonpoint N source, and in many instances there is a long delay – years to decades – between the initial escape of nitrate from the crop rootzone or other land surface, and the appearance of nitrate in drinking water wells. In Section 2 of this report, we provide a detailed overview (and a concise summary in Section 2.4) of management measures that are known to reduce the level of nitrate leaching to groundwater from crop fields. The report includes an estimate of the proportion of irrigated crop acreage within the study areas for which mitigation measures are likely to have the largest impact. This vulnerability estimate is based on soil properties, crop species, and irrigation system type, and does not consider underlying hydrology or whether farmers may have already adopted improved management measures and technologies.

Livestock, especially dairies, contribute significantly to the overall nitrate load in the Tulare Lake Basin (TLB). Because the animal waste is applied to croplands as a fertilizer and soil amendment, manure management is considered in tandem with inorganic N management in Section 2 of this report, along with proper storage and handling prior to cropland application. Regulations aimed at reducing nitrate discharges to groundwater were imposed on all dairies in the Central Valley in 2007 (Central Valley Regional Water Quality Control Board 2007). The regulations include animal housing, manure storage facilities, and dairy farm crop fields that receive manure. Croplands not associated with dairies (whether receiving manure or not) are not currently regulated for nitrate discharges to groundwater.

In Section 3, the cost of mitigation measures for irrigated crop production is evaluated by use of an agro-economic model. The model is based on the assumption that technologies are available to farmers that individually or in combination can be used to make modest improvements in crop nitrogen use efficiency. Very general assumptions are made about the likely management and equipment costs of such measures and the offsetting savings due to lower fertilizer use. Section 4 provides a brief overview of ornamental landscaping and turfgrass management options that minimize leaching losses from these areas.

Section 5 outlines nitrate loading from urban and domestic wastewater, sewers, and septic systems, all of which tend to have less regional impact but important local consequences. We discuss the infrastructure upgrades and improved technical standards necessary to reduce loading from these

Adminsitrative Record
Page 34736

sources. Costs and overall feasibility are included for each option, which tend to vary widely. In section 6, we examine how to reduce the contributions to aquifer nitrate contamination from wells, and the financial feasibility of the options available to address this source.

Finally, the Conclusions in Section 7 summarize the findings and promising actions concerning reduction of nitrate leaching from all of the above sources, as outlined in this report.

Adminsitrative Record
Page 34737

2 Reducing Agricultural Nitrate Loading

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Adminsitrative Record
Page 34738

2.1 Introduction

Crop production represents the largest source of groundwater nitrate in the study areas, while also playing an important role in recycling waste from urban areas, food processing, and most importantly, animal production. Livestock waste (especially dairy manure), a key source of the agricultural nitrate footprint, is primarily disposed of by application to croplands as a fertilizer and soil amendment. For this reason, manure is considered within the context of crop production, in addition to proper handling and storage prior to land application in this report.

A highly important industry, both economically and socially, crop production in California requires inputs of irrigation water and nitrogen. Nitrogen use efficiency (NUE) is dependent upon the management of both of these inputs. NUE, sometimes referred to as partial nitrogen balance (PNB), refers to the percentage of N applied to cropland (from all sources) that is recovered by the crop and therefore is not lost to the atmosphere or to surface and groundwater. As such, high NUE reduces losses to leaching while reducing unnecessary costs to the producer. Due to unavoidable N losses, complete crop recovery of all applied inputs is impossible to sustain. Our approach in this report is to identify and describe practices and technologies that are potentially available to growers for achieving high crop N use efficiencies, and for reducing, but not eliminating, nitrate leaching to the groundwater from cropped land. Although NUE as low as 33% crop recovery of N inputs has been reported, a recent U.S. EPA report estimated that with the adoption of best management practices, NUE could increase by up to 25% of current average values (U.S. EPA Science Advisory Board 2011).

In California's semi-arid climate, avoiding accumulation of salt in the crop rootzone requires some movement of water downward past the crop roots to leach the plant-toxic salts. Winter rains contribute to fulfilling this need but in low-rainfall years and in areas of reduced precipitation, such as the San Joaquin Valley, irrigation volumes applied must include water in excess of the evapotranspiration requirement (ET). With the leaching salt, any soluble nitrate present will likewise travel downward with the water, and once beyond the rootzone where it will no longer be available for plant uptake, the majority will make its way to groundwater. Efficient and uniformly distributed irrigation water is therefore a critical component of NUE, although ideal efficiency and uniformity are not possible due to spatial variability in soils (texture, water holding capacity, permeability, etc.) and the practical limits of the various irrigation systems available to growers.

The second component of maximizing NUE is to provide a plant-available form of N only at times and amounts required by the crop. Avoiding a large surplus of N in the soil at times of low crop demand or in excess of plant growth requirements is paramount to reduced leaching. While it is not possible to produce crops while maintaining zero soil nitrate and thus it is not possible to achieve zero leaching of nitrate, management measures are available that can contribute to optimization of NUE. Crop nitrogen and irrigation system design and management techniques that can help maximize crop N recovery (thereby limiting leaching losses) are well known. A selection of these practices is described in Section 2.3 of this report. Not all practices are appropriate or practically feasible for all farms and tandem use of a suite of practices is often required to significantly affect NUE.

Adminsitrative Record
Page 34739

Due to the typically long delay between the initial escape of the nitrate from the crop root zone and its appearance in groundwater, and the nature of croplands as a non-point source, it is not possible to provide quantitative estimates of the degree of reduction in loading that would result from adoption of any particular practice or combinations of practices. This is, in part, due to lack of survey data on the current use of many recommended practices and a lack of definitive information on how management practices have changed over time. Additionally, it is very difficult to relate increased adoption of a practice (such as nitrate soil testing) to a quantitative reduction in nitrate leaching, and the studies that do so are sparse and variable. Again, depending on the unique field situation (i.e., soil characteristics, crop type, and irrigation system, etc.), the effect a practice has on nitrate leaching will vary. It is clear that to significantly reduce leaching, adoption of a suite of improved management practices is the most effective and that the choice of which bundle of practices to adopt will always be field specific.

A detailed presentation of farm management practices follows the Methodology in Section 2.3. After introduction of an individual management measure, recommended practices to achieve the measure follow. Individual practices are briefly described, including an outline of how they contribute to crop NUE and estimation of the current use within the study area, and conclude with a discussion of the barriers to their expanded adoption.

A summary of the management measures and recommended practices follows in Section 2.4 emphasizing their current extent of use in the study areas while highlighting the barriers to increased adoption. In Section 2.5, examples of improved practice bundles that could be feasibly undertaken for specific crops, given specific barriers, are provided.

In Section 2.6, we illustrate the lack of a one-size-fits-all approach to farm management options, and how the selection of the appropriate suite of management practices will be highly variable and can only be attended to on a case by case basis. We provide a visualization of this variability within the study areas, and broadly identify the areas in which increased adoption of improved management practices may have the most effect. This was carried out by assessing the vulnerability of the study area fields to nitrate leaching using a Nitrate Hazard Index (Wu et al. 2005), based on each field's specific combination of crop, soil, and irrigation system choice. As such, the areas identified as especially vulnerable are the areas in which improved management practices will have the largest effect on groundwater aquifers.

Adminsitrative Record
Page 34740

2.2 Methods

2.2.1 Approach to Identifying Potential Nitrate Reduction Strategies

Crop production practices with the potential to maximize plant N recovery and reduce nitrate loading to groundwater were identified and refined via literature review and expert panel input and consensus. The 1998 UCANR publication "Nitrogen and Water Management for Cool-Season Vegetables" written in response to the Coastal Zone Management Act, provided a starting point for identifying these practices. The literature review[3] and expert panels (see below) provided the opportunity to significantly expand and revise the listed management practices. Case studies show unequivocally that nitrate leachate reductions are possible with the adoption of improved farm management strategies (Gheysari et al. 2009). However, such studies do not cover the range of crop, soil, and irrigation system combinations represented in California's diverse agricultural landscape. To address this greater diversity, five expert panels were convened to discuss and refine the draft list of practices and to provide insight into the current state of farm management practices specific to the two study areas.

2.2.1.1 Expert Panels

The following five expert panels met to review and refine the draft list of suggested practices, estimate their current extent of use and identify the barriers to increased adoption:

1. Salinas Valley cool-season vegetable and strawberries
2. Woody perennials (vineyards, tree fruits, and nuts)
3. Agronomic/field crops not receiving dairy manure
4. Forage/silage crops receiving dairy manure
5. Tulare Lake Basin vegetables and other truck crops.

Panels were set up as 6 – 10 person roundtable discussions, including the project leader and support staff, two UC Cooperative Extension farm advisors, two to three growers, and two to three crop management professionals or other industry personnel (most of the latter being Certified Crop Advisors). UCCE advisors were chosen based on their expertise and experience with the specified crop group's nitrogen and water management regimes within the study areas. The 10 UCCE farm advisors recommended growers and allied industry members with extensive knowledge of, and experience with, the crops and management practices in the study areas. To promote frank discussion, panel meetings were not open to the public and all members agreed to maintain confidentiality.

Each participant was provided with a project background statement and was asked to complete a detailed survey covering the draft list of practices. The survey asked panel members to rate each practice for mitigative potential and its current extent of use within the crop group and study area basin.

[3] Dr. Todd Rosenstock, of the California Nitrogen Assessment project, provided us a large repository of literature that we used in conjunction with our own.

Adminsitrative Record
Page 34741

A numerical summary of the results of these questionnaires was distributed at each panel meeting and was used to guide discussion and identify areas of consensus and disagreement. For practices identified as having high mitigative potential, panel members were asked to identify the main constraints or barriers to expanded adoption.

2.2.2 Crop Management Practice Scenarios

Managing nitrate leaching generally requires a tandem application of various mitigating practices (Power et al. 2001). Selection of suitable management practice suites will depend on the unique farm situation, requiring site-specific consideration of field variables such as crop and crop rotation practices, irrigation and management constraints, and soil and climate characteristics (Tilman et al. 2002; Fixen 2011).

Although some general farm management measures are important and generally applicable for all farm operations in terms of reducing the potential for nitrate leaching (e.g., improving irrigation efficiency), the specific practices used to implement these goals will change depending on the unique situation. For example, for grain production, converting to drip irrigation is not economically feasible; however, it may be possible to improve uniformity of existing surface gravity or sprinkler systems. In contrast, for production of many vegetables, irrigation with sprinklers beyond initial establishment may increase susceptibility to disease due to wetting of foliage and conversion to drip irrigation may be financially more feasible.

Examples of practice "bundles" that might be adopted to improve nitrogen use efficiency are presented as possible management change scenarios restrictive to a unique set of conditions. Six crops were selected to serve as these examples. These crops represent the major crop categories important to the two study areas (cool season vegetables, valley vegetables, field crops, silage crops, and permanent crops). For each selected crop or crop rotation, a bundle of feasible management practices was chosen based on literature reviews specific to that crop and expert panel input. For each scenario, potential barriers to adoption (e.g., high costs) are described.

2.2.3 Leaching Vulnerability Assessment

It is unlikely that any one set of farming practices would be effective in improving crop nitrogen use efficiency on all crop acres in the study area. It is also unlikely that mitigative measures are even needed on all cropped acres. For example, deep-rooted crop species grown on less permeable soils will be less likely to generate nitrate leaching when farmed with typical methods. In contrast, crops such as spinach, a shallow rooted species often grown on permeable loams, with a peak N uptake at the time of harvest, is more likely to generate nitrate leaching. In this situation more intense employment of effective management techniques will be justified.

We used a Nitrate Hazard Index (HI) developed in 2005 by the UC Water Resources Center to identify, at the scale of individual fields, the nitrate leaching risk based solely on soil, crop species, and irrigation

Adminsitrative Record
Page 34742

system type (Wu et al. 2005). Crop and irrigation types were identified using the most recent DWR land use maps for the five counties (Monterey 1997, Tulare 1999, Fresno 2000, Kings 2003, Kern 2006), and the Soil Survey Geographic (SSURGO) database (USDA NRCS). The HI was used to evaluate the area (at the time of the DWR surveys) where utilization of improved practices may have the most impact on nitrate leaching and where improved nitrogen and water use efficiency need special attention. The most important caveat to consider is that the DWR surveys represent only a "snapshot in time," that is, a one-time survey (generally in the summer) carried out as far back as 1997. As such, the overall pattern of risk is not reflective of current cropping systems, current irrigation system type, annual crop rotations, or associated irrigation system rotations. For example, drip irrigated vegetable fields may be rotated with similar vegetables in the winter months, or with furrow irrigated wheat; these rotational factors affect overall risk and cannot be spatially analyzed.

Spatially, agricultural nitrogen inputs do not correlate well with groundwater nitrate concentrations (U.S. Geological Survey 1999). Farm management practices heavily influence the fate of applied N, but equally important are the innate soil attributes, crop characteristics, and method of water delivery. Consideration of these three field-specific components can then provide guidance on the suitability of various management practices to the unique farm situation and indicate how much attention to reduced nitrate leaching strategies is warranted.

2.2.3.1 Nitrate Groundwater Pollution Hazard Index

In 1994, the California State Water Resources Control Board (State Water Board) appointed a Nutrient Technical Advisory Committee (TAC) to assess water quality problems associated with agriculture and make suggestions for addressing such issues (Irrigated Agriculture Technical Advisory Committee 1994; Wu et al. 2005). A hazard indexing methodology was conceptualized in which growers could identify field nitrate leaching vulnerabilities based on the soil characteristics, the crop grown, and the irrigation system utilized. An index or ranking method, such as this, allows for a way to quickly and easily determine risk severity and identify the major factors contributing to this risk, without requiring the large data set needed for more involved indexing methods (see for example Delgado et al. 2008) or modeling-based vulnerability assessments. The TAC suggested that growers with fields that scored high, in terms of nitrate leaching risk, should then be required to complete nutrient management plans.

The UC Center for Water Resources proceeded with this work, developing the matrix-based (overlay and index method) Nitrate Groundwater Pollution Hazard Index (HI) for irrigated agriculture (Wu et al. 2005) and making it available online to the public.[4] This online tool borrowed and built upon the conceptual framework of the TAC, assigning soil series, crop types, and irrigation systems individual leaching risk values through consideration of multiple factors by expert collaboration. Index values proposed for soils, crops, and irrigation systems were then subjected to external review by experts.

According to this HI system, soils are allocated a scale rating of 1 (low risk of nitrate leaching) to 5 (high risk) based on denitrification potential and water infiltration and transmission rates. NRCS soil series

[4] http://ucanr.org/sites/wrc/Programs/Water_Quality/Nitrate_Groundwater_Pollution_Hazard_Index/

Adminsitrative Record
Page 34743

descriptions were utilized by the developers of the HI to compile these values, with attention focused on the soil's drainage and permeability characteristics including typical pedon texture, restrictive layers, and evidence of mottles, (indicators of poor aeration). Additionally considered are color changes, depth to restrictive layers, degree of hardness, and organic matter content. For example, the Hanford series is a deep and well drained sandy loam with low organic matter content, moderately rapid permeability, and a lack of mottles or restrictive layer and is thus rated 5 (i.e., of the highest risk). The Cropley series on the other hand, is given a rating of 2; although moderately well drained, it has a clay to clay loam texture, light mottles with iron deposits, and a slow permeability. Note that the online HI allows users to include deep ripping as an option, which results in a decrease in the soil HI component. Deep ripping is a form of deep tillage that can improve drainage of soils having hardpan layers. It is commonly used to prepare land for planting of trees or vines. We could not assign this component in our analysis, although it is an available option in the online tool. In most situations, the addition of this factor does not move the overall hazard rating from one of low risk to high.

Soil series were extracted from a statewide layer produced by the local USDA-NRCS office that amalgamated the county-level SSURGO soils maps and assigned the dominant soil series by proportion to each map polygon. These polygons were then filtered by selecting only those that intersected irrigated agriculture. For any soil series of high area not found in the online HI tool, soil properties described in NRCS soil surveys were compared to the criterion described in Wu et al. (2005), and HI values assigned by a soil scientist. Fields of undefined soil series of very small area within the study area were excluded from the analysis.

The crops and irrigation components of the HI each range from 1 (low risk) to 4 (high risk). Crop rating is based on rooting depth, overall N requirements, how closely peak demand for N coincides with harvest, the ratios of N uptake and harvest removal to suggested N application rates, denitrification inherent to the crop, and, for permanent crops (trees and vines), whether leaf material is typically removed (i.e., if N stored in dropped leaves is available for subsequent crops). For example, lettuce, a shallow rooted annual that demands high N levels particularly just prior to harvest, is rated a 4 and considered high risk. Conversely, vineyards are considered a low risk crop and rated 1, due to their deep roots and generally low N requirements, among other factors.

In the case of irrigation systems, TAC suggestions were followed by the HI developers (with an updated scale from 1 to 4) wherein fertigation; environmental influence on water delivery; and the level of water application control and precision afforded the manager are considered. In all cases it was assumed that the system would be operating at maximum potential, in that application uniformity is not decreased by poorly maintained or faulty systems. Per the developers of the HI (Wu et al. 2005), fertigated drip and microirrigation are given a rating of 1 (2 without fertigation, a rare case), sprinklers without fertigation are rated a 3 (2 with fertigation, also rare), and all methods of surface irrigation are assigned a rating of 4.

In our irrigation analysis however, we deviated from these values slightly. An important caveat is that most drip irrigated annuals are germinated under sprinklers, a higher risk system due to the effects wind

Adminsitrative Record
Page 34744

and design flaws can have on uniformity. Fields are especially vulnerable to leaching during crop establishment due to the need to keep the seed bed (or transplant area) evenly moist at all times (to avoid risk of plant failure) via frequent irrigation events, coupled with the scarcity of plant roots to utilize any soil nitrate present at this time. The HI online tool does not differentiate mixed irrigation systems and designates fields pre-irrigated and germinated via sprinkler as the same risk as season-long micro- irrigation, as used for tree crops (HI 1). For our assessment however, we rated drip irrigated annual crops, which are germinated with sprinklers, to have an HI of 2. Based on feedback from experts, we considered an HI of 1 to be too low, ignoring the high risk of excessive pre-plant and germination sprinkler irrigation during times of low plant need. For example, Vázquez et al. (2005) showed that the greatest leaching losses occurred during crop establishment in drip irrigated tomatoes. For the spatial assessment it was necessary to assume that all drip irrigated annual fields were sprinkler established and designating the irrigation component value of 2. Similarly the spatial assessment required assumptions that all micro-irrigated crops are fertigated, while also assuming no fertigation for crops sprinkler irrigated through harvest.

The composite HI values, ranging from 1 to 80 (lowest to high risk) were produced by multiplying all three factor values (soil, crop, and irrigation) using an index matrix as shown in Figure 1. The authors of the HI proposed that fields identified as having an overall index below 20 are of low concern and that average management practices are usually adequate. As such, although continued vigilance is necessary for all fields, attention to optimizing NUE via good management practices is best focused in areas with greater risk for leaching. Any value over 20 is indicated as being cause for such concern, where adoption of more stringent management practices is paramount. The authors of the HI caution that comparing an HI value of 40 with 60, for example, is not especially useful, and that anything over 20 should be of equally high concern. More importantly, the HI can guide management improvement options by identifying the factors that contribute most to the high value. For example, if carrots (HI 2) are being grown on the high risk Hanford series (HI 5) and sprinkler irrigated (HI 3), it is clear that reducing risk on this field will need to be focused on tight irrigation management to optimize the system and match the rapid soil drainage characteristics. On the other hand, a high risk crop, such as broccoli, might be grown on a low risk soil with drip irrigation after being established with sprinklers. In this case, the high N levels required and the relatively high residuals left after harvest contribute the most to the risk value and incorporating a cover crop to scavenge residual N may represent a good choice for reducing risk in such a situation. For

	Soil					
Crop	1	2	3	4	5	Irrigation
1	1	2	3	4	5	1
1	2	4	6	8	10	2
1	3	6	9	12	15	3
1	4	8	12	16	20	4
2	2	4	6	8	10	1
2	4	8	12	16	20	2
2	6	12	18	24	30	3
2	8	16	24	32	40	4
3	3	6	9	12	15	1
3	6	12	18	24	30	2
3	9	18	27	36	45	3
3	12	24	36	48	60	4
4	4	8	12	16	20	1
4	8	16	24	32	40	2
4	12	24	36	48	60	3
4	16	32	48	64	80	4

Figure 1. The UC Nitrate Hazard Index multiplicative matrix with highly vulnerable situations highlighted in yellow (adapted from We et al. 2005)

Adminsitrative Record
Page 34745

the spatial analysis, the overall HI map was created by intersecting the soil HI polygon layer with the DWR polygon layer containing both crop and irrigation HI information.

There are important assumptions and caveats associated with the overall HI results. DWR surveys represent a single period in the year. Although information regarding double cropping is provided with most of these reports, it is only possible to spatially assign a single crop per field per layer, meaning that the resulting map is a "snapshot" in time. DWR surveys are repeated approximately every seven years, and the year of the most recent report for each county varied (Tulare 1999, Fresno 2000, Kings 2003, Kern 2006). Similarly, the time of year during which these surveys were carried out varied, but generally are confined to the summer months of June through September. Therefore, our evaluation of HI is skewed toward summer crop representation where, for example, cool season crops and winter fallow periods cannot be visualized simultaneously with the crops grown during the summer surveys. Likewise, irrigation system choice has evolved rapidly over the past 10-20 years, notably the increased adoption of drip and microsprinkler systems, reduction of surface gravity irrigated vegetable and permanent crops, and changes in impact sprinkler methods.

Similarly, crop production trends have changed since some of these surveys took place. For example, sugarbeets in the TLB are heavily overestimated in the DWR data (over 7,000 hectares reported, but near zero coverage currently remaining) due to the closure of one the last remaining processing facility in the state since the time of the surveys. These fields have likely been converted to higher risk species, given that sugarbeets are one of the few species with an HI rating of 1, and thus driving overall HI values up. Likewise, strawberry acreage in the Salinas Valley (SV) has increased substantially since the 1997 Monterey County survey date, due to strong financial incentives. In general, it is important to keep in mind when reviewing our results, that the agricultural landscape is very dynamic and has changed since these summer surveys were carried out between 1999 and 2006.

To simplify the crop HI map, some crops produced on very small acreages were grouped with crops of different HI values, but this was rare and restricted to very low acreage species. For example, olive trees, which are rated 1, were placed in the "orchard" category, where nearly all other tree crops are rated 2. In this case the olives represent only 2% of the total orchard area and thus, this and other similar crops were kept within the orchard group, rated HI 2 for the purposes of mapping (HI 1 figs and apricots, HI 3 grapefruit and kiwis, were also categorized with other HI 2 orchard species). Nearly all field, grain, and hay crops are rated HI 2, and the small area of dry beans and sudangrass (both rated HI 1) were kept in this category, while a separate category was created for the high acreage, HI 1 alfalfa, as well as HI 3 corn. Note that these groupings were only used to construct the *crop* HI analysis and map (Figure 6) and the actual (ungrouped) crop HI values were used to calculate the overall composite HI of each field, as displayed in the *overall* HI map (Figure 9). Vegetables were separated into those with HI values of 3 versus those of highest risk (HI 4). A small category was designated for lower risk vegetables consisting of carrots (HI 2) and the now extinct sugarbeet (HI 1) area.[5] Certain irrigated fields of very low

[5] Also included in this mixed HI 1-2 category were sweet potato, bush berry, and rice fields which collectively represent less than 0.01% of the study area acreage.

Adminsitrative Record
Page 34746

acreage that do not fit into any crop HI designation were not included in the HI analysis. These include turf farms, pasture, eucalyptus and Christmas tree plantations, and nursery operations.

For the four San Joaquin Valley counties, the DWR crop reports also include spatial irrigation system type for the same land parcels and years. The DWR surveys of individual fields for Monterey County did not include irrigation data. Although it is not realistic to assume that the same irrigation type is used for any one crop species, by choosing the method used for a majority of each crop, a spatial analysis could be performed for the Salinas Valley. To develop irrigation system type for individual fields, we relied on (a) expert panel estimates of the percent of crops irrigated by each method, and (b) the most recent county Ground Water Summary Report (MCWRA 2011), which summarizes irrigation methods used in portions of the Salinas Valley. Based on these sources of information, we assigned all field and grain crops to surface irrigation, and all tree crops (reported to be >70% citrus and >15% walnuts) and vines (nearly all winegrapes) to microsprinkler irrigation. Although drip irrigation is common in SV vegetables, the degree of uncertainty is wide. It is estimated that 50 – 70% of the vegetables (including strawberries) are currently drip irrigated, with the bulk of the rest furrow irrigated or sprinkler irrigated throughout the season. However, because there is no way of distributing such percent estimates for drip irrigation spatially, for purposes of our analysis, we assigned drip irrigation to all vegetable and strawberry acres in the Salinas Valley. This important caveat implies an underestimation of the area with high risk potential (HI>20) in the SV.

Note also that irrigation system choice has evolved rapidly over the past 10-20 years, notably the increased adoption of drip and microsprinkler systems, reduction of surface gravity irrigated vegetable and permanent crops, and changes in impact sprinkler methods. While the area of drip irrigated vegetables in the SV is overestimated in our analysis, there is some underestimation in the TLB, due to the much higher use of microirrigation in vegetables and permanent crops since the original surveys were carried out.

Adminsitrative Record
Page 34747

2.3 Management Measures (MM) and Recommended Practices (RP) for Reducing Movement of Nitrate to Groundwater from Crop Operations

While the complete elimination of agricultural nitrate loading to groundwater is not possible, adoption of improved farming management practices can help to mitigate this concern (Esser et al. 2002; Gheysari et al. 2009). The practices provided below hold some promise of contributing to the goal of reducing agricultural nitrate leaching losses and maximizing crop recovery of N inputs.

Management measures (MMs) described herein represent technologies or processes for reducing nitrate leaching to groundwater. Note that this report does not cover greenhouse and nursery production practices (refer instead to Newman 2008). After exploring the literature and conferring with members of the expert panels, a total of 10 of these measures with mitigative potential were identified. A total of 50 recommended practices (RPs) for achieving these management measure goals are reviewed in depth. All measures and practices fall into one of four categories:

1) Design and operate irrigation and drainage systems to reduce deep percolation
2) Manage crop plants to capture more N and decrease deep percolation
3) Manage N fertilizer and manure to increase crop N use efficiency
4) Improve storage and handling of fertilizers and manure to decrease off-target discharge[6]

For each category, a narrative description of the associated management measures and their accompanying practices follows. Practice descriptions refer to the literature in terms of mitigative potential and summarize panel findings, including an assessment of the extent of current use and, if promising in terms of mitigation, discussion of what barriers may be preventing more widespread adoption of the practice, follows. Although cost savings can often be realized with more efficient use of nitrogen and water, the barriers considered here include:

- Real or perceived risks to crop yield or quality
- Capital costs (infrastructure, equipment)
- Operating cost (labor, management, training needs, energy, physical inputs, etc.)
- Ranch logistics barriers
- Land tenure constraints
- Grower education, training, or demonstrations needed to speed technology transfer
- Insufficient adaptation of technology to local conditions

[6] Note that besides these categories, poor stand uniformity and N use efficiency can be affected by various factors including salinity, pests and diseases, non-N deficiencies, and soil chemical problems. Addressing these concerns is a common practice among growers but is clearly a critical component of maximizing crop N recovery.

Adminsitrative Record
Page 34748

The expanded narrative of the mitigative management measures follows the outline presented in Table 1 below. A suite of improved management practices, adopted in tandem, will usually be the most effective approach. Note that the mitigative potential of a practice will vary between individual fields depending on site-specific soil, irrigation and crop variables (Tilman et al. 2002; Fixen 2011). Thus, while the most effective suite of specific practices will be field specific, each of the four basic principles and management measures (depending on irrigation type) are universally applicable to irrigated fields. See Section 2.4 for a condensed review of the use and barriers of each practice specific to the study area.

Table 1. Agricultural management measures that can increase nitrogen use efficiency and decrease nitrate leaching to groundwater, including the number of described practices used to achieve each measure.

Basic Component	Management Measure	Number of Recommended Practices Described
Design and operate irrigation and drainage systems to decrease deep percolation	MM 1. Perform irrigation system evaluation and monitoring	3
	MM 2. Improve Irrigation scheduling	4
	MM 3. Improve surface gravity system design and operation	6
	MM 4. Improve sprinkler system design and operation	5
	MM 5. Improve micro-irrigation system design and operation	2
	MM 6. Make other irrigation infrastructure improvements	2
Manage crop plants to capture more N and decrease deep percolation	MM 7. Modify crop rotation	4
Manage N fertilizer and manure to increase crop N use efficiency	MM 8. Improve rate, timing, placement of N fertilizers	9
	MM 9. Improve rate, timing, placement of animal manure applications	6
Improve storage and handling of fertilizer materials and manure to decrease off-target discharges	MM 10. Avoid fertilizer material and manure spills during transport, storage and application	9
		Total: 50

Adminsitrative Record
Page 34749

2.3.1 Design and Operate Irrigation and Drainage Systems to Decrease Deep Percolation

Retention of soluble N within the root zone, where it is available for plant uptake, is achieved in part by good irrigation management. The amount of nitrate lost to leaching is related to the volume of water that percolates below the root zone, which in turn is related to the irrigation system performance (Letey et al. 1977; Allaire-Leung et al. 2001). Scheduling irrigation events such that the volume of applied water matches the crop water requirement (evapotranspiration or ET), and delivering water uniformly to the field, are both critical to increasing N use efficiency and reducing nitrate leaching. Non-uniform irrigation forces farmers to over-irrigate some parts of the field in order to ensure adequate delivery to the parts of the field receiving the least water.

For example, Pang et al. (1997) modeled expected nitrate leaching under various irrigation and fertilization regimes. Their models show that as irrigation uniformity decreases, leaching increases exponentially, even under lower and less optimal N application rates than would be encountered on functioning farms. Furthermore, N application rates that provided good yield under efficient irrigation were modeled to be too low under situations of inefficient irrigation uniformity, wherein excess water application in some areas of the field resulted in decreased yields as the water flushed the available N too quickly beyond the rootzone.

It is important to note that some extra water beyond ET must be applied for rootzone salinity control. This is particularly true in regions with very low rainfall (such as the southern San Joaquin Valley) and with very tightly managed irrigation systems, such as drip irrigation. In such situations, farmers must apply extra water specifically to move salt out of the root zone, inevitably leading to some nitrate leaching along with the salts. As discussed in Section 2.3.3, aiming for reduced soil N levels at times of low crop demand, when intentionally irrigating to leach salts, or during the winter rainy season can help to minimize the concern of concurrent nitrate leaching during salt management activities (Hanson & May 2011).

Maximizing distribution uniformity is complicated by the spatial variability of soil texture and permeability within a field or operation, as soil drainage characteristics affect retention and loss of available N within and beyond the rootzone (Letey et al. 1977; Goldhamer & Peterson 1984; Williams & Kissel 1991; Hanson et al.; 1998; Khosla et al. 2002). Such attributes are generally not easily alterable by the manager. In some situations however, it can be economically worthwhile to attempt manipulation of natural soil variability. Panel members brought up several examples of such manipulations. Draining of the marshy soils in Northwestern Monterey County increases aeration and, in turn, plant productivity; the installation of subsurface drains in such soils has proven to have beneficial economic consequences in terms of yield and diversification of potential crop palettes. Note that although the tile drains dramatically reduce levels of nitrate entering the groundwater, the drain effluent is then flushed into surface waters. Impacts of this diversion on surface water quality and the ecosystems associated with the Salinas River, Elkhorn Slough, and the Monterey Bay are cause of major environmental concern (see Section 2.3.1, MM 6 for further discussion of tile drains).

Adminsitrative Record
Page 34750

Another example of manipulation of soil variability is in row crops within the TLB. Many fields with highly variable alluvial sand deposits have had such sand spits removed and the fields regraded, allowing for more uniform drainage and water retention characteristics, and leading to increased water and nitrogen use efficiency. In both basins, it is not uncommon for field boundaries to have been reconfigured over the years to follow natural soil series boundaries more closely. Deep ripping, where underlying hardpans within the soil profile are broken up, is another example of a profitable practice aimed in part at increased water use efficiency, but especially at plant productivity, via changes to the soil rather than manipulations of water delivery systems, water application rates, or timing.

A variety of different irrigation system types (i.e., surface, sprinkler, and micro systems) and subtypes exist (e.g., basin, furrow, etc.), all with their own inherent maximum efficiencies and uniformities, assuming equal levels of management optimization. Determining which irrigation system is most suitable is guided by the crop type, soil type, water quality and availability, topography, and technological and economic considerations. Regarding water application, although perfect efficiency and perfect uniformity are not possible, their maximization can be attended to within the constraints of the existing system or by choosing to utilize systems that have, when operated correctly, higher efficiency thresholds.

Panel members estimated that the bulk of deciduous fresh market tree fruit as well as a very high proportion of field crops, such as grain and cotton, remain surface irrigated in the study area. Carrots, baby greens, and a still somewhat large proportion of onions and garlic were thought to be sprinkler irrigated throughout their lifecycle. The majority of the remaining direct seeded vegetables tend to be sprinkler irrigated during the early establishment period after which they are switched to drip or, to a lesser degree, surface irrigation for the remainder of their life cycles. Crop value and other economic concerns are the primary drivers behind the decision to install drip or microsprinkler irrigation systems. In recent years, a large proportion of acreage in processing tomatoes, leafy vegetables, strawberries, vineyards (especially winegrapes), citrus, and tree nuts have been converted from furrow or basin irrigation to drip and microsprinkler irrigation methods. In the SJV there is growing, yet still small, interest in mechanized sprinkler systems (center pivot and linear move) for some crops (see RP 3.6 for more information).

MM 1: Perform Irrigation System Evaluation and Monitoring

To meet the crop water requirement while minimizing deep percolation, the volume of water applied must be measured and application uniformity must be evaluated. Scheduling irrigation events to match the crop water requirement is difficult if the amount of water applied is not known or if water is applied to fields non-uniformly. Although slow or uneven water advance rate (in surface gravity irrigation systems), plugged lines or malfunctioning sprinkler heads provide visible cues to the need for maintenance, other symptoms of gradual efficiency decline can be slow to reveal themselves without periodic system evaluation. Over time, gradual loss of efficiency equates to potentially high levels of unnecessary nitrate leaching.

Adminsitrative Record
Page 34751

RP 1.1 Conduct irrigation system performance evaluation

Evaluation of system performance includes measurement of the volume of water applied to a field and the degree of uniformity of the application. Where water is inexpensive or where crop value is low, growers are less likely to carry out such evaluations. Where water is expensive, they are more likely to do so. This practice alone does not necessarily lead a land owner or grower to make system improvements or to reduce deep percolation losses, but it does make it possible to identify the magnitude of inefficiencies and the potential for improvement.

Irrigation system evaluation is standard practice on some farms; but for others it is rarely or never done Panel members indicated that the practice of performance evaluations varies widely, with some operations maintaining peak performance as part of their standard operational procedures and others more or less oblivious to their system's inefficiencies. Consensus was that there indeed exists room for improvement in this arena, although several barriers preclude enhanced adoption.

Barriers to expanded use of this practice include the following: In surface gravity and sprinkler systems, evaluation is usually time consuming and complicated and therefore more appropriately carried out by consultants. Growers who receive their water supply from district canals or who produce relatively low value field crops may view performance assessment data as having little value, because they cannot easily change the amount of water applied, or they cannot afford to make the changes that would improve performance. Performance evaluation of drip and microsprinkler systems is generally undertaken more often by growers, as the functioning of these systems is more dependent upon regular maintenance.

Often irrigation system complexity, e.g., multiple sources (several wells and surface waters) feeding single fields on varying days with varying levels of soil type uniformity, presents a barrier in that understanding all individual components and their interactions within the system requires more expertise than is present on-farm. Educational and consultant costs can be prohibitive. The time and labor involved with performing and analyzing results often lacks economic incentive unless an obvious problem forces action. In the past, DWR and the USDA-NRCS, in cooperation with UC Cooperative Extension and/or local water management agencies, operated so-called Mobile Laboratories, which provided irrigation system evaluation and advice, free of charge, to individual growers. Funding and expanding and marketing such programs certainly holds promise to improve wise-use of farm water resources.

RP 1.2 Install and use flow meters or other measuring devices to track water volume applied to each field at each irrigation

Knowledge of the volume of water applied is the crucial first step in minimizing excess water applications and reducing leaching risk (Mosley & Fleming 2009). Although nominal pump capacity multiplied by operating time can provide an estimate of how much water is applied to a specific field, the actual rate at which water moves through the irrigation piping system may not equal the rate at which the pump is rated due to, e.g., total pressure head, impeller obstructions, etc. Thus, totalizing

Adminsitrative Record
Page 34752

flow meters are considered the better method. Tracking water applications can identify potential problems associated with excess irrigation before they cause economic and environmental problems. Where water is expensive, flow meters are a financially sound investment. Knowledge of water flow and pressure are especially crucial in drip irrigation systems.

In the TLB, some crop consultants maintain records of irrigation water applications for their clients. Additionally, the water district and how they allot and price their water can influence use of flow meters. For those that do track their water flow with such devices, panel members indicated that it was primarily used more often as an accounting technique rather than one of long-term monitoring and irrigation management.

Barriers to the use of flow meters include equipment and installation costs. Also, there are technical limitations to flow metering. Inaccurate readings can result from pipes not being full at the time of reading, the influence of check valves and pipe turns, user errors in reading and use, or a faulty mechanism within the meter itself. Additionally, lack of understanding of water budget data and the complexity of evaluation (e.g., when fields are irrigated from multiple sources) can limit the usefulness of flow meters.

RP 1.3 Conduct pump performance tests

In non-pressurized (surface gravity) systems, deteriorating pump performance contributes to irrigation inefficiency. Pump performance tests are more important for these surface systems than for pressurized systems such as drip and sprinkler, where flow volume is controlled to some degree by emitter or nozzle characteristics. Although pump inefficiency will not affect water delivery rate in pressurized systems, it can lead to higher pumping cost. Salvaging the money lost to poor pump performance is thus an incentive for this practice.

Costs of these tests can sometimes present a barrier, although not in the case of those portions of the southern Central Valley serviced by Southern California Edison, which offers pump performance tests free of charge. The Center for Irrigation Technology at California State University Fresno, in collaboration with Pacific Gas & Electric, also offers reduced cost pump performance testing, education, and partial repair incentive rebates through the Advanced Pumping Efficiency Program (formerly the Agricultural Pumping Efficiency Program). When free or low cost, these tests are often preferred by growers over the more costly installation of flow meters. The availability of the Advanced Pumping Efficiency Program appears not to be well known, and the growers who do not know about this free and reduced cost testing are less likely to conduct pump performance tests.

MM 2: Improve Irrigation Scheduling

Scheduling irrigation timing and amounts to coincide with actual water needs, rather than by calendar date, is an important component of any good water management program. Synchronizing water and nutrient applications with crop needs reduces nitrate leaching (Meisinger & Delgado 2002; Gehl et al. 2005; Gheysari et al. 2009). To be effective, such scheduling does not only consist of evaluating current

Adminsitrative Record
Page 34753

soil and crop water status, but also must include anticipating and forecasting future water needs (Martin et. al. 1991). Frequency will depend on the crop needs and their rooting depth, soil water holding capacity and drainage characteristics, and the specific irrigation system. Weather- and plant-based scheduling and soil moisture guidance can be used in tandem or independently to guide choices of when and how much water to apply. All have advantages and disadvantages and tandem use will enhance the overall predictive capacity. Optimal frequency of water application will be dictated by weather, crop characteristics, soil water holding capacity, and availability of water to the farmer. Good irrigation scheduling is important for all crops and soils, although it is especially critical when growing crops on sandy soils and with shallow rooted species.

In spite of years of effort by public agencies and universities to educate growers on the concepts of water budgeting and crop water use, calendar scheduling is still common, particularly in field and forage crops. This is due, in part, to tradition or lack of understanding, but in many instances, ranch logistics, labor, and water availability constrain the farmer's ability to adjust irrigation schedules. Inexpensive water in some areas and a lack of penalty for over-irrigating also promote poor scheduling, while expensive water districts are home to some of the study area's most efficient and conscientious irrigators. Growers who do not track their water use are often simply unaware water is being wasted. Panel members stressed the necessity of continuing education and that farmer access to irrigation specialists is critical to seeing a reduction in the typical dominance of calendar scheduling. NRCS and UCCE outreach has increased grower awareness and knowledge of the importance of good scheduling and the tools available to guide timing decisions, but continued outreach and intensity will be required to see increased use of advanced scheduling techniques.

RP 2.1 Use weather-based irrigation scheduling

Weather-based scheduling is based on crop evapotranspiration (ET) estimates calculated from recent weather data and normal-weather forecasts. The "reference ET" is then adjusted with a crop species specific factor that accounts for plant geometry. In California, a network of weather-stations, known as the California Irrigation Management Information System (CIMIS), provides crop ET estimations in real time via the internet to help growers schedule their water applications. Use of CIMIS technology is proven to increase water use efficiency (Hartz et al. 1994). In a SV lettuce trial, use of CIMIS during the germination and establishment phase led to a reduction in total water applied and reduced nitrate leaching losses (Smith & Cahn 2011).

Note that where the groundwater is shallow, CIMIS is generally not appropriate, as the method assumes that soil water changes between irrigations is equal to crop ET. This is not the case in shallow groundwater areas due to crops extracting some of their water from the perched groundwater (Hanson & Ayars 2002). Otherwise, irrigating to crop ET should be sufficient to meet salinity leaching fraction requirements. The occasional exception comes up in drip irrigated fields when winter and spring rainfall is low, where sprinkler irrigation may be needed to reduce salts during stand establishment (Hanson & May 2011).

Adminsitrative Record
Page 34754

Panel member commentary indicated that the use of CIMIS and weather-based scheduling was limited according to crop, crop management needs, user education, irrigation system type, and water district delivery constraints. Vegetable and tree fruit and nut producers tend to utilize ET data fairly regularly, whereas calendar scheduling remains common in field crop production. Although real-time ET/CIMIS data provides the most precise guidance, historical ET and weather data is more often utilized by growers. Historical data are readily available and do not require active access to CIMIS and adaptation for the specific crop from the reference ET using established crop coefficients.

Specific crops react to irrigation events in ways that complicate scheduling, requiring managerial attention beyond expected ET. For example, tree fruit yield and quality will be compromised without sufficient irrigation during their ripening process, driving growers to want to overcompensate, whereas nut crops are generally more tolerant of deficit irrigation in the late season. Ranch logistics can also present as a barrier, for example, needing to access the field to spray or cultivate can trump an optimized irrigation schedule. Labor schedules can also complicate irrigation scheduling, especially in surface irrigated systems.

Calculating crop coefficients demands specific knowledge that follows a learning curve, thus educational requirements were also cited by panel members as a possible barrier to increased use. For some crops, more research is required to determine their specific coefficients and corresponding ET. Water district delivery schedules often complicate the ability of growers to optimize their irrigation schedule. A tendency is to over-irrigate in the spring when ditch water is abundant, with poorer supply during the dry summer leading to deficit irrigation in some crops. Forage, silage, grain, and other field crop producers and their advisors agreed that calendar based irrigation scheduling was the more common practice in these crops due to logistics of harvest and field operations and the fact that gravity based systems simply do not allow the manager the option of manipulating how much water is applied in any one run.

In spite of these barriers, increased grower education, especially in terms of water budgeting, could boost utilization of weather-based scheduling techniques, in particular the increased use of real-time ET data. The greatest potential for this happening is in areas where water is expensive, although it is precisely in these areas where growers are already paying very close attention to water use efficiency.

RP 2.2 Use plant-based irrigation scheduling

In general, where weather and soil based scheduling can tell a grower *how much* to irrigate at any given time, plant-based methods can be a superior technique for determining plant water status and therefore are more precise guides to irrigation *timing* (Fulton et al. 2008). At its simplest, plant-based irrigation scheduling consists of the farmer observing plant appearance or condition in the field and identifying the first signs of stress. Stress however, tends to affect productivity before observable physical symptoms appear. In recent years, several ground-based and remote sensing techniques and instruments have become available for use in crop production.

Adminsitrative Record
Page 34755

Infrared thermometers detect plant canopy temperature increases when plants reduce their stomatal conductance in response to water stress. This technology has proven to be a useful scheduling tool in some crops and is associated with enhanced crop water use efficiency, and, when irrigations are automated based on such measurements, significant labor reduction (Peters & Evett 2008). The most widely used tissue water status method is the use of pressure chambers, or so called pressure bombs, used to approximate leaf water potential via the inverse relationship this has to the pressure needed to draw sap from a cut leaf stem. Although labor intensive, this method is a relatively commonly employed plant-based scheduling tool in tree crops and some field crops (Hanson & Ayars 2002). Pressure bombs are generally more labor intensive than infrared readings. Note that the reliability of both pressure bomb and infrared temperature techniques are affected by environmental factors such as fog and clouds. For a full overview of all plant-based methods see Jones (2004).

Barriers to plant-based scheduling are similar to those cited for weather-based scheduling, including irrigation system type, water availability and district delivery schedules that are more compatible with calendar scheduling, conflicts with various field operation needs, and other ranch logistics and complexity considerations. Precise technical skill is required of many plant-based scheduling tools and atmospheric differences can cause variability (when using pressure bombs) requiring a large sample pool which increases labor requirements. These tools are used by some tree crop, field, and vegetable producers or their hired irrigation schedulers in the study areas, although of all plant-based techniques, simple visual observation of plant status is the most utilized.

RP 2.3 Use soil moisture content to guide irrigation timing and amount

Tracking soil moisture levels can alert the producer to crop water need sooner than observable wilting and can aid irrigation scheduling. Keeping soil moisture at or within 20% of field capacity is ideal for most crops (Pettygrove et al. 1998). Manually checking soil moisture content with a shovel or soil sampling tube is a traditional farmer method for determining irrigation timing. For annual cropping systems, this is still the most important technique. Various instrumented or automated methods for measuring soil water status exist, including tensiometers, psychrometers, gravimetric (weighing) devices, and neutron probes. Such devices may be used in tandem with ET and plant-based irrigation scheduling methods. These devices are in use by some study area growers.

Although most instruments are relatively straightforward to use, are generally precise, and lend themselves well to automation, a limitation of using soil moisture content to guide irrigation timing is that soil moisture may vary greatly within a field due to heterogeneity of soil texture, crop condition, and water application during the previous irrigation, thus requiring multiple sensors to achieve an acceptable level of accuracy (Jones 2004). Growers may be reluctant to install such devices in row crop fields where they could be damaged by tractors and interfere with harvesting practices. Again, irrigation system, water and pipe availability, and other field requirements will sometimes limit flexibility and stand in the way of scheduling irrigations according to plant need, whether determined via ET, plant status, or soil moisture levels.

Adminsitrative Record
Page 34756

RP 2.4 Avoid heavy pre-plant or fallow irrigations

Studies have shown that nitrate leaching is high following fallow period irrigation events (Jackson et al. 1994; Cavero et al. 1999; Jackson 2000; Smith & Cahn 2011). Without adequate plant roots to take up available soil N during these times, any soil N present is potentially lost to leaching. For example, Di & Cameron (2002) report two studies wherein up to 95% of residual N was lost during rainy fallow periods following harvest of the cash crop. However, pre-plant and fallow period irrigations are often required to prepare the seedbed for planting and address salinity concerns (Hanson & Ayars 2002; Hanson & May 2011). Besides germination and salt control requirements, pre-plant irrigation is also sometimes used to encourage deep rooting (e.g., in cotton). Vázquez et al. (2005, 2006) found that the highest leaching losses in tomato fields occurred during the plant establishment phase. Jackson et al. (1994) found that the highest leaching occurred during the germination of the second lettuce crop of the season in the SV, due to the rapid decomposition of the first crop's residues, raising soil N to high levels at a time of little plant uptake. Soil characteristics and weather will influence the ability of farmers to adjust pre-plant irrigation. Note that leaching will not be high, even with heavy irrigations or rain events, if soil N is low during this time. Thus, for crops, soil types, and weather conditions that demand heavier pre-plant or fallow irrigations, focusing on minimizing soil N content during these time periods is critical (see RP's in MM 7, MM 8, and MM 9).

Panel members agreed that annual crop growers are becoming more aware that early establishment water needs are lower than what tradition may have held, although there is a legitimate risk of reduced germination and/or establishment should pre-plant irrigations not be sufficient. This risk was cited as the largest barrier, while emphasis was also given to the role that soil characteristics play in fallow period irrigation needs. Most panel members suggested that there was still room for improvement in reducing fallow period and early establishment water applications, although farmers would likely require more evidence of what is possible before risking crop establishment and yield potential. As such, fueling research and education should help increase adoption of this important management practice.

MM 3: Improve Surface Gravity System Design and Operation

Gravity irrigation systems, in which water flows directly across the soil surface, are traditionally the most utilized crop water irrigation type both in the state and in the study area, although there has been a steady decline in use of these surface systems as growers of some crops convert to drip or sprinkler systems. The comparatively lower capital and operating costs of surface gravity methods of irrigation are attractive to growers, and some crops (especially field and forage crops) and soil types are best served by surface systems in general. In these systems, enough water must be applied to advance the water to the end of the field and this usually limits the capacity for efficiency. The upper end of the field is often subject to deep percolation as the water slowly advances to the opposite field end. Irrigation uniformity is often degraded by long field lengths, uneven slopes, uneven soil compaction or surface roughness in furrow bottoms, and natural variations in soil texture and the soil water intake characteristics (Power et al. 2001).

Adminsitrative Record
Page 34757

Surface gravity systems, when operated near peak performance, deliver water at 70-85% efficiency (Hanson 1995). Improving water application efficiency through the practices listed below can reduce nitrate leaching.

RP 3.1 Convert to surge irrigation

In surge irrigation, also known as "bumping", the water is turned on and off as it advances down the furrow. After halting the water flow, water is then run into an alternate set of furrows where it is again turned off before reaching the end of the field. While the new set is watered, the original set of furrows drain. This switching between sets continues until the water has reached the end of the field in both sets. Because the first part of the field has already been wetted prior to the rest of the run, the infiltration is lower in these areas and water advances faster towards the areas that have not yet been saturated. As such, uniformity is increased and the total amount of water required to thoroughly wet the end of the field is decreased by up to 30-80% compared to conventional furrows (those that have not be torpedoed—see RP 3.4) (Coupal & Wilson 1990; Schepers et al. 1995; Power et al. 2001; Rodríguez et al. 2004; Schwankl & Frate 2004). Fertigating while utilizing surge irrigation can also increase N uniformity (Boldt et al. 1994). Advance inflow rates can be further minimized in furrows that have been compacted, as described in RP 3.4 (Yonts et al. 1996).

Although Hanson (1989) argues that the cost of surge irrigation equipment is relatively low and easy to implement on existing systems, panel members felt differently, citing equipment costs, labor costs, and logistics as representative of large barriers that may be difficult to overcome. This technique requires gated pipe and, ideally, automatic valves, both of which are an additional capital cost for farms that are not already equipped as such. Without gated pipe and surge valves, significant additional labor time is required to manually open and close valves. Most dairy grain, silage, and forage production fields do not lend themselves well to utilizing this technique, due to the costs involved, incompatible valves or gates, and bad experiences with malfunctioning equipment (Schwankl & Frate 2004).

RP 3.2 Use high flow rates initially, then cut back to finish off the irrigation

By pushing the water over the surface more quickly during the initial irrigation phase and reducing the subsequent advance rate, otherwise known as cut-back, it is possible to increase uniformity and application efficiency (Evans 1977; Hanson 1989; Mohammed et al. 2006). A constant flow rate from start to finish is the norm and displays reduced uniformity compared to the employment of cut-back methods.

Amongst panel members, the practice was not perceived to have substantial mitigative potential and most agreed that it was not a highly utilized method of irrigating. This is partially due to the expense of the extra labor required to manually operate the valves or the capital cost of automated technologies that would reduce labor costs. Coupal & Wilson (1990) found that this practice may only be economically viable if gated pipe were already in place, although this may depend on the value of the crop and water costs. This practice is usually unsuitable in furrows and provides greater advantage in

Adminsitrative Record
Page 34758

border check irrigation systems which are commonly used for grain, silage corn, and other forages in the Tulare Lake Basin.

RP 3.3 Reduce irrigation run distances (i.e., field lengths) and decrease set times

Long field lengths are one of the primary reasons why higher efficiency cannot be met by surface irrigation systems. This is due to the minimum water application not being determined by crop needs, but rather by the amount needed to reach the end of the field. For example, if 8 inches of water are needed to reach the field end, but the crop has only used 5 inches of water since the last irrigation, some of the excess water applied (3 inches in our example) is sure to be lost below the rootzone. If the field is long, it could be cut in half where water set times should be reduced to match the reduced time required for the water to reach the of the field. This shorter field length and run-time would increase distribution uniformity and reduce overall water use, thereby increasing efficiency and reducing opportunity for leaching (Schwankl & Frate 2004). For example, Hanson (1989) showed that by splitting a 300 m field in half, subsurface drainage likewise decreased by 50 percent.

This practice removes significant land from production, reducing total farm yield, because it requires a supply ditch or pipe and tailwater ditch, as well as tractor turning areas for each of the shorter field sections. Additionally there is an expense for installation of new pipe. Members of the silage and manure-receiving crop panel agreed that even with the cost barrier, it still may be cheaper than, for example, switching to a mechanized center pivot system. However, overall, it is likely that, until there is a more substantial economic incentive, the practice will remain underutilized due to the loss of farming area and subsequently reduced productivity. Most fruit tree orchards in the TLB and SV vegetable fields are already short in length.

RP 3.4 Increase flow uniformity among furrows (e.g., by compacting furrows)

Improving the surface uniformity among furrows necessarily improves water distribution uniformity. Tractors can pull behind them heavy tubes (torpedoes), wheels, or balls (bolas) for the purpose of smoothing furrow bottoms to achieve a more rapid water advance rate. (Schwankl et al. 1992) showed that torpedoes increased water advance rates by 15 to 30% compared to non-torpedoed furrows. Such compaction practices have been shown to reduce water application and overall leaching risk (Musick et al. 1985; Schwankl & Frate 2004).

Panel members agreed that there is currently moderate use of such equipment amongst growers utilizing furrow irrigation. To see increased adoption, some members suggested that growers would need more convincing that the practice would be worth it in their particular situation in terms of crop, soil, and field length. The practice is not helpful, for example, in cracking clay soils, and is likely unnecessary in short fields. Equipment operating cost and education are the primary barriers, with a need for additional research specific to the study area conditions to bolster the conclusion that any water savings associated with such compacting devices are economically worthwhile to the producer.

Adminsitrative Record
Page 34759

RP 3.5 Grade fields as uniformly as possible

To advance water at a uniform rate and to achieve more uniform infiltration across the field, proper grading is essential, with a slope that remains as constant as possible. Poorly graded fields will result in poor distribution uniformity and addressing this issue will lessen leaching risk in surface irrigated fields (Schepers et al. 1995; Mosley & Fleming 2009).

Panel members agreed that with the advent of laser leveling, uniformly graded fields are well represented. Typically fields are graded via land planes for every crop and should then be laser leveled every three years. Growers also noted that further grading-related improvements have been employed by some growers, such as altering the grade slightly towards the end of the field in order to allow water to move faster at the top of the field (where infiltration would otherwise be higher) and slower at the bottom, allowing for a more uniform infiltration.

RP 3.6 Where high uniformity and efficiency are not possible, convert to drip, center pivot, or linear move systems

In some situations, improving surface irrigation systems using the practices listed above will yield only moderate increases in efficiency despite optimal management. This may be due to soil characteristics, incompatibility of the practices with the specific crop grown, or excessive costs. In such cases it may be more economically feasible to consider an alternative irrigation system that offers the possibility of higher system performance. Drip irrigation and mechanized sprinkler systems give the operator greater control over water applications and thus, when managed correctly, can increase irrigation efficiency (Spalding et al. 2001; Hanson & Ayars 2002; Tilman et al. 2002). It is important to keep in mind that if these systems are not properly maintained and managed (including scheduling irrigation timing and amounts to meet crop need) their performance can be less than that of a well running surface system. In other words, nitrate leaching is correlated to the distribution uniformity and efficiency regardless of the specific irrigation system. Nevertheless, the inherently more precise control offered by these systems can translate into significant reductions in deep percolation when managed correctly.

Although converting to any of these technologies will incur significant capital costs, it may work out that such a switch is still more economically worthwhile than staying with a system that performs poorly (Fereres et al. 1982). For example, Hanson et al. (1998) found that in a cracking clay soil, due to preferential flow of water down the soil cracks beyond the rootzone, it was difficult to improve the surface system performance and an alternative system may be needed to reduce deep percolation. Similarly, Goldhamer & Peterson (1984) showed that on a sandy loam soil, cotton irrigated with a linear move sprinkler system yielded slightly better with significantly less deep percolation than a furrow system, and even with higher costs, the net return was slightly higher with the mechanized system. Center pivot systems have been shown to reduce water application by as much as 60-72% as compared to conventional furrow systems (Schepers et al. 1995). Drip irrigation systems can also substantially reduce water application requirements and, in some crops such as tomatoes, consistently increase yields (Hanson et al. 1997, 2001; Hanson & May 2003). Vázquez et al. (2006) showed that water loss was reduced when water is applied with a high frequency, such as is usual in microirrigation systems. A

Adminsitrative Record
Page 34760

trend towards adopting such technologies in lieu of surface systems has been increasing steadily in the nation, state and study areas (Edinger-Marshall & Letey 1997; Dillon et al.1999; Zoldoske 2002; Orang et al. 2008; MCWRA 2009). Dillon et al. (1999) also found that farmers who adopted more efficient irrigation systems also integrated new, more efficient nutrient management techniques faster than those who had not upgraded their irrigation systems. Additionally, as discussed in MM8, RP 8.3, N fertilizer in microirrigation systems is usually delivered directly in the irrigation water (fertigation), which further enhances NUE by way of the small multiple N doses better matching crop need (see RP 8.3 for more information on split applications) (Vázquez et al. 2006).

The main barrier to switching from a surface system to a more efficient irrigation system is the economic cost-return ratio. For lower value crops, the cost may prove insurmountable unless simultaneously switching to a higher cash crop or adding one to their rotation (Hanson & Ayars 2002). Drip and continuous move systems are not adaptable to all crops (such as grain or hay crops) or may show little advantage. For example, fresh market deciduous tree fruits are grown in small fields where the short-run surface irrigation distribution uniformity can be quite high already and the cost of installing a pressurized system would make little economic sense. Center pivot and linear move systems are not widely used in California, although installation is increasing slowly. Center pivot systems are not favored by some growers due to lost productivity at the corners of the fields and the economic burden associated with the reduced yield that accompanies the reduced effective field size. Additionally, for air quality reasons, dairy farmers in the Central Valley have not been permitted to apply liquid manure through sprinklers. Salinas Valley panel members indicated that the small field and planting block sizes that are common in the region made linear move systems impractical.

Switching to drip irrigation continues to be a favored practice among producers of many high value crops, especially vegetables, nuts, citrus, and winegrapes. Educational barriers often prevent newly installed drip systems from being operated at their maximum efficiency, due to the learning curve involved in managing the much more complicated and labor intense irrigation system. An additional caveat to keep in mind with this improvement is that most annual crops still require sprinkler irrigation during the leaching-vulnerable phase of initial establishment, which can somewhat reduce the positive impact drip has on leaching reduction.

MM 4: Improve Sprinkler System Design and Operation

Optimizing sprinkler systems requires proper design, operation, and maintenance. Irrigation uniformity with sprinkler systems is primarily dependent upon spacing of nozzles along the lateral, lateral spacing along the mainline, proper nozzle maintenance, maintenance of correct pressure within the entire piping system, and wind speed (Hanson et al. 2008). In general, the continuous move center-pivot and linear-move systems offer greater uniformity than hand-move and wheel-line systems when properly maintained (Spalding et al. 2001; Gaudi et al. 2007; Hanson et al. 2008) and investing in such systems offers mitigative potential. Furthermore, better sprinkler distribution uniformity can equate to cost savings to the grower (Chen & Wallender 1984), depending on overall capital investment requirements and level of uniformity prior to change.

Adminsitrative Record
Page 34761

RP 4.1 Monitor flow and pressure variation throughout the system

As all sprinkler systems are pressurized, an important managerial requisite is monitoring for variation in pressure. As shown in Figure 2 below, pressure variation can cause water distribution uniformity issues related to inadequate spray breakup (Hanson et al. 2008). Design flaws and insufficient maintenance are the primary causes of pressure and flow problems within the system. Alternatives for addressing pressure problems include adjusting or unclogging contributing heads individually, investing in flow control nozzles, or redesign of the system (e.g., removing heads in excess of what pressure allows) or adjusting pipe gradients (Smajstrla et al. 1997).

The consensus among panel members was that better monitoring of pressure variation is a practical improvement opportunity. The labor costs and expertise required to properly diagnose problems and maintain optimal functioning of sprinkler systems were identified as the main impediments to increased grower attention to these matters. However, panel members agreed that such barriers should not be insurmountable, especially if growers were provided improved access to irrigation specialists, such as NRCS and UCCE staff, or through reinstatement of free or reduced fee irrigation evaluation services, such as the past DWR/NRCS Mobile Laboratories program.

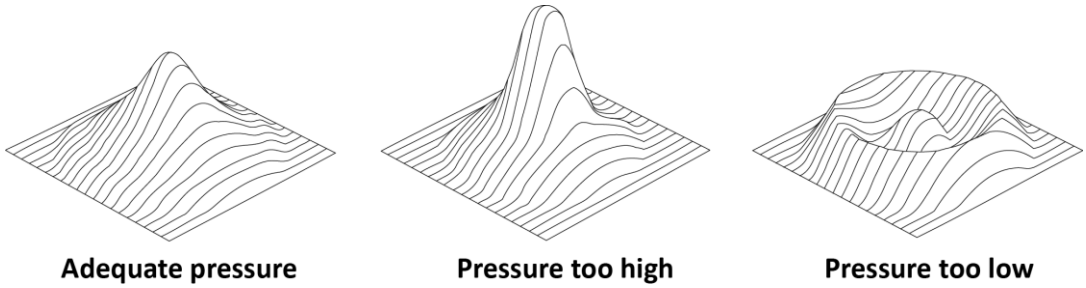

Figure 2. Changes in water distribution uniformity from a single sprinkler nozzle when operated at (from left to right) pressures meeting their design specifications, or when pressure is too high or too low (adapted from Trimmer & Hanson 1986).

RP 4.2 Repair leaks and malfunctioning sprinklers, follow manufacturer recommended replacement intervals

Leaks and worn or clogged nozzles contribute significantly to poor system efficiency. These are common occurrences and regular maintenance is required when employing these pressurized systems (Gaudi et al. 2007). Leaks simply waste water and result in high levels of deep percolation on location, as do poorly performing worn or clogged nozzles that can both over- and under-apply water. More often than not, these problems are not necessarily obvious or come on gradually, such that the issues are not identified unless regular system evaluations are made. Panel members agreed that nozzle maintenance is sometimes overlooked by growers and attention could be improved. It was suggested that continued vigilance may require regular reminders during field days and other educational events. Irrigation advisors indicated that a significant portion of operations in the TLB do not follow the suggested nozzle 'lifespans' (nozzles should be replaced every 4 years), keeping them in operation beyond optimal

Adminsitrative Record
Page 34762

performance capacity. Similarly, there is room for additional attention by growers to replacement of impact "spoons" and sprinkler head seals. Extension specialists indicated that growers sometimes avoid regular uniformity evaluations until major problems demand attention. Cost of upgrades and the labor and expertise required to monitor and maintain sprinkler systems were thought to be the main barriers to the needed increased adoption of this practice. Again, programs such as those that have been offered in the past (DWR/NRCS Mobile Laboratories), offering free or low cost irrigation evaluations would allow for earlier detections of gradual declines.

RP 4.3 Operate sprinklers during the least windy periods

Operating sprinklers when it is windy results in non-uniform distribution of water, as shown in Figure 3 below. Panel members agreed that most growers are aware of this problem, but in order to provide water to all fields, they typically must irrigate round the clock, even during windy periods. The Salinas Valley is subject to relatively high winds, making it difficult to optimize sprinkler system performance, although real-time tracking of wind forecasts could allow for some delay in irrigation. Minimizing the distance between laterals can help reduce wind interference of distribution uniformity and the effect of such minimal spacing can be accomplished in hand move systems by alternating lateral locations over successive irrigation sets (see RP 4.4) rather than investing in additional pipe and other necessary equipment.

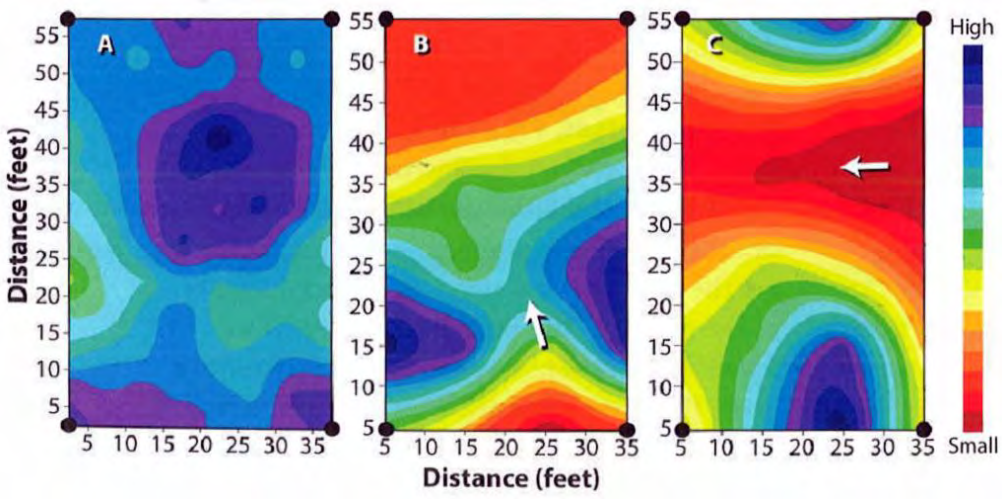

Figure 3. Distribution uniformity of overlapped sprinklers for A) Low wind condition at regular pressure, B) High wind condition with lateral perpendicular to wind direction, C) High wind condition with lateral parallel to wind direction. Black dots represent sprinkler locations and arrows the wind direction (from Hanson et al. 2008, reproduced with permission).

RP 4.4 Reduce distance between lateral lines or alternate lateral line location over successive irrigations

Improvements to water distribution uniformity may be achieved by reducing the distance between lateral lines in cases of high wind or poor pressure. While Sanden et al. (1999) found no statistical

Adminsitrative Record
Page 34763

difference in distribution uniformity between 33 foot and 48 foot spacing in a Kern County carrot field, they reported that 40 foot spacing demonstrated the most consistent water and nitrogen use efficiencies, especially in high wind situations. Kasapligil (1990) demonstrated that spacing laterals 25-30 feet apart allows for 80% uniformity with 3-8 mph winds. If the capital cost of decreased spacing becomes prohibitive, alternating the lateral locations over successive irrigations and placing them midway between the locations of the previous cycle, will effectively cut the spacing in half, as shown in **Error! Reference source not found.** (Kincaid 1982; Trimmer & Hanson 1986). This is especially useful in ituations of lower pressure and has been shown to be economically beneficial (Chen & Wallender 1984). Panel members indicated that offsetting laterals is practiced by some producers, but not all. Shallow-rooted crops that require frequent water applications are not adaptable to this practice, as it is difficult to keep up with the multiple irrigations required in short time periods. In such cases, simply reducing the distance between solid set lateral lines to 30-40 feet is the best option. Capital costs for additional hardware may be required. Labor and ranch logistical issues may present additional barriers to use of this practice.

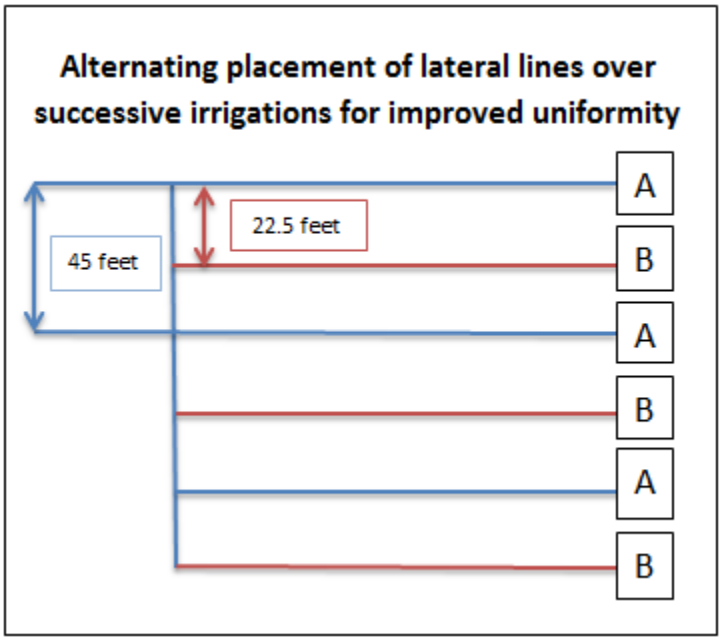

Figure 4. With lateral sprinkler lines set 45 feet apart on the mainline, uniformity can be improved by alternating the location of the laterals for each irrigation set A and B, effectively cutting in half the distance between them, resulting in improved water distribution uniformity.

RP 4.5 When pressure variation is excessive, use flow control or pressure regulating nozzles

As discussed in RP 4.1 and as Figure 2 indicates, pressure variation is a cause for poor distribution uniformity. If pressure variation is difficult to address via re-design and good maintenance and management, installing flow control or pressure regulating nozzles can effectively resolve the issue. When used, regular monitoring of the rubber in these nozzles for degradation and periodic replacement are required to keep the flow control utility functional. Panel members indicated that such equipment is

used by some producers, especially in hilly areas where pressure variation is a bigger problem. The main barrier is the equipment and installation costs, although the achieved improvements to uniformity may offset some of these costs via water savings, given good maintenance practices.

MM 5: Improve Drip and Microsprinkler System Design and Operation

Very high application efficiency and uniformity are possible with drip and microsprinkler systems, but poor system design and poor monitoring and maintenance have resulted in lower performance than even some furrow systems (Pettygrove et al. 1998). Burt (2004) found that most of the uniformity problems were due to either pressure differences or to alterations of flow rate among emitters caused by emitter clogging and wear. Pressure variation has been addressed to some extent with the improved pressure compensating emitters that are now more common in today's drip systems, although this does not apply to microsprinkler systems or in cases of poor design. Other farm management activities can also affect drip system performance. Ensuring tractor tires do not compact the soil above soft buried drip tape and being careful when cultivating are both essential to good performance. Panel members also stated that attention to animal populations is important. For example, crows removing orchard emitters, gophers destroying subsurface driplines, or fire ant populations drawn to the localized water source can slowly destroy the tape over time.

Maintenance of drip and microsprinkler systems is complicated by the fact that problems are not as readily visible as in surface gravity and sprinkler systems. Root penetration into buried drip can be slow to reveal itself and close spacing of emitters often allows clogging issues of both surface and buried lines to go unnoticed for some time without vigilance in monitoring. Problems with design such as inappropriate hose lengths, variable distance between emitters, and the installer's failure to follow manufacturer recommendations of emitter to pressure ratios may also not be detected for some time without regular monitoring of efficiency or uniformity. Members of the tree fruit and nut panel indicated that it was not uncommon for 1 gallon-per-hour emitters to be replaced with 2 gallon-per-hour emitters on the same line and system, resulting in inefficiencies. As with other irrigation system types, soil variations within a single field demand growers water according to the least common denominator (that is, the most water demanding soil type). This can result in excess water applications in the areas of higher water holding capacities, increasing potential for nitrate leaching. Design of the system around such variability can help mitigate this concern (see RP 5.3). Soil and crop characteristics also need to be considered when comparing drip emitters, surface drip tape, buried tape, and microsprinklers (Gardenas et al. 2005). In areas of rolling topography for example, pressure compensating emitters are a justified investment.

RP 5.1 Use appropriate lateral hose lengths to improve uniformity

Excessive hose length causes pressure drop and poor uniformity. Generally this is an uncommon problem in that engineering technology and design has improved over time. In cases where the systems have not been updated, reconfiguring the system will present some capital and labor costs, but replacement costs are generally expected with drip systems, and increased water savings usually help to

Adminsitrative Record
Page 34765

offset such costs. Long lateral hose lengths present less of a problem in steep topography if flowing downward.

RP 5.2 Check for clogging potential and prevent or correct clogging

Mineral-heavy or highly alkaline groundwater can precipitate solids independently or in reaction to fertilizer delivered through the drip system, clogging small emitters and reducing efficiency (Schwankl & Prichard 1990). Calcium carbonate or lime precipitates are common and can be resolved via a number of products on the market that lower pH or otherwise prevent or address the problem. Standard operating procedure should include checking for possible precipitate issues when installing a new system, when switching fertilizer products, or when utilizing a new water source. Biological growth and physical particles can also present clogging issues, especially when irrigating using surface waters prone to hosting microorganism populations. Proper filtration and filter maintenance can address these issues with additional biocide or pH reduction treatments as necessary to control algal or bacterial clogging. Regardless of the reason for clogging, regular flushing should be performed, allowing water to run through the system with open-ended laterals. After fertilization or the use of other chemical additives, lines should be flushed to clean emitters, with maintenance flushing recommended twice a month to keep the system in optimal condition.

Panel members agreed that filtration, chemical treatments, and flushing are common grower practices, although issues regarding proper methodology can still present a problem, as can a lack of proactive management (i.e., only checking the system when problems appear). Distribution uniformity and scheduling methodologies vary widely and optimizing system performance needs more attention by some growers. Panel members noted that treatment options (maintenance products and techniques) have generated "an industry in and of itself," but agreed there is some room for increased vigilance by growers on good system maintenance and proper scheduling, again indicating that educational reminders would benefit to this end. Labor and capital costs can also contribute to suboptimal efficiency of drip systems.

MM 6: Make Other Irrigation Infrastructure Improvements

Besides evaluation, scheduling, and irrigation system choices, groundwater protection should be prioritized at the site of water entry (i.e., the well) and exit from the field (i.e., drain and tail water), as discussed below.

RP 6.1 Installation of subsurface drains

Anaerobic conditions generally reduce plant productivity and thus in areas with shallow water tables (some areas on the west side of the southern SJV, some areas in the northern Salinas Valley), it can be economically advantageous (in terms of yields and diversification of potential crop palette) to install subsurface drains to keep the water below the rootzone and to help leach salts. A certain portion of the applied irrigation water and any dissolved nitrogen that percolates with it beyond the rootzone will then flow out of these drains rather than entering the aquifer. The fraction of percolating water that enters

Adminsitrative Record
Page 34766

these drains versus that which moves directly to the groundwater will be high if the spacing between drains is sufficiently small and the placement of drains is not excessively deep. Depending then on drain design, soil and hydrological characteristics, drains may keep a significant amount of applied nitrate from entering the groundwater (Drury et al. 2009; Schipper et al. 2010a; Woli et al. 2010).

However, the resulting high nitrate and often saline effluent from these drains however, usually acts as a surface water contaminant. The disposal or treatment of this waste then represents a major problem associated with this practice. For example, the drain effluent flushed into northwestern Monterey County surface waters has negative consequences on the ecosystems associated with the Salinas River, Elkhorn Slough, and the Monterey Bay. Technology for treating the effluent on-site via denitrification chambers is currently being researched and developed with some promising results (Schipper et al. 2010). These systems are not yet commercially available in California and it is yet unclear if deeper drain installations (in areas without high water tables that interfere with plant health) might represent a feasible or cost effective means of diverting nitrate from groundwater should treatment systems become commercially available.

The subsurface drains that have been installed in the study area are primarily for the purpose of addressing perched water tables. The drain effluent in Monterey County passes through local sloughs and ultimately into the ocean, creating surface water quality concerns. A few growers in both regions transfer the tile drain water to evaporation ponds, blend it with fresh water and reuse it, although this is not common, and the collection ponds can sometimes pose a risk to wildlife. Until treatment or disposal options are further developed and further research specific to California is conducted, subsurface drain installation will not represent a feasible option for groundwater nitrate leaching reduction.

RP 6.2 Backflow prevention

Farms that practice fertigation are required to protect against backflow of fertilizer into wells. Local, state, and industry guidelines should be heeded when selecting preventative measures. For the most reliable prevention of backflow, multiple devices often are used in some combination. Regularly checking that the device is properly maintained and functioning optimally is critical to ensure fertilizers do not enter the well. While backflow prevention devices are in high use amongst growers, proper maintenance may not be less than optimal. Producers need to keep in mind that, with all devices, there will be pressure loss, which tends to increase as the flow rate increases.

Check valves require a minimum backpressure to completely seal, tending to leak if the minimum pressure is not maintained. Installation of a low pressure drain upstream from the check valves will intercept any leakage that might occur. An inspection port should also be installed upstream from the valve to ensure maintenance of pressure and proper valve seating. Check valves should be installed between the injection point and the well, as well as in the injection pump discharge pipe. Vacuum-relief valves prevent vacuum formation caused by water flowing back down the well column after the pump stops. All of these items should be utilized simultaneously and regularly checked for functionality.

Adminsitrative Record
Page 34767

By maintaining an air gap between the water supply and the fertilizer vessel, an actual backflow prevention device may not be required. However, this method is unsuitable for pressurized systems. Alternatively, interlocking circuits can be used, where the irrigation pump and chemical injection pump are wired to shut off whenever the irrigation pump shuts off.

2.3.2 Manage Crop Plants to Capture More Nitrate and Decrease Deep Percolation

Besides managing nutrient and water applications in a way that increases crop N use efficiency, nitrate leaching potential is also affected by other crop and field management decisions, including crop rotation and tillage choices (Tilman et al. 2002). Tillage practices such as deep ripping, chiseling, and slip plowing will increase available rooting depth, allowing plants an increased opportunity to "catch" percolating nitrate. However, depending on soil structure and fertilization method, identical tillage practices can have opposite effects on leaching, reducing it in one situation while increasing it in others (Di & Cameron 2002). Similarly, risks of leaching, greenhouse gas volatilization, surface water contamination, and erosion are all affected differently according to choice of tillage practices (Smith & Cassel 1991). Due to the complex effects of tillage schemes, in this section focus is given to managing crop rotations.

MM 7: Modify Crop Rotation

Crop rotation decisions involve choice of crop species, crop sequence, length of fallow periods, and whether cover crops will be grown. Plant species and varieties may differ in rooting depth, rate of N uptake during the season, the amount and quality of residues left in the field following harvest, and other characteristics that influence total soil N and leaching risk. Careful selection of crop rotation schemes, based on knowledge of such traits, can reduce nitrate leaching hazard.

RP 7.1 Grow cover crops

Nitrate leaching potential is substantial on fallow fields during periods of rain, as well as during pre-plant and early establishment irrigations (Cavero et al. 1999; Jackson 2000). In processing tomatoes for example, Vázquez et al. (2006) observed that both water and N losses were highest during their establishment period. Di & Cameron (2002) note two studies reporting that, following harvest of the last crop, up to 95% of residual mineral N can be lost during rainy fallow periods. A cover crop is a non-cash crop[7] grown to improve soil structure and lessen risk of soil erosion or nutrient loss; add organic matter to the soil; and sometimes to help prevent or break a crop disease or insect cycle. By incorporating the prior cash crop's residual N into their biomass, the amount of N ultimately subject to deep percolation is reduced. The subsequent cash crop can then capitalize on cover crop N residuals.

It has been repeatedly shown that growing a cover crop during otherwise fallow periods, or between rows of permanent crops, reduces nitrate leaching and improves overall soil quality (Jackson et al. 1993; Wyland 1996; Brandi-Dohrn et al. 1997; Delgado 1998; Beaudoin et al. 2005; Feaga et al. 2010; Ramos et

[7] A broader definition may include crops with a marketable harvest, rotated with the main cash crop. See RP's 7.2, 7.3 and 7.4.

Adminsitrative Record
Page 34768

al. 2010). In their covercropping meta-analysis, Tonitto, David, & Drinkwater (2006) reported that use of non-leguminous cover crops reduced leaching by 70% on average and by 40% in legume-based rotations, with the latter simultaneously fixing additional N and in some cases increasing cash crop yields. Using leguminous species, otherwise known as green manures, can add 100-300 kg N per hectare via N-fixation depending on time of year and how quickly their residues decompose (Sheenan 1992). The additional N fixed by leguminous cover crops should be accounted for in the nitrogen budget (see RP 8.8). With any cover crop, one caveat is that if early rainfall is significant, much of the nitrate remaining after the harvest of the main crop can be moved beyond the reach of the cover crop roots depth, or will be in excess of the cover crop's nitrate uptake capacities (Feaga et al. 2010).

A wide variety of crops can be used for this purpose in both annual and perennial systems and much research has been performed regarding which species and species-mixes may provide the best benefit for specific goals or situations. Snapp et al. (2005) reviewed multiple cover cropping studies across specific climates (including California's Mediterranean zone), comparing costs and benefits and evaluating barriers to increased adoption. According to this review the additional benefits of cover cropping (beyond reduction in nitrate leaching) include reduced erosion due to increased water infiltration; improved weed control; increased soil organic matter (leading to improved soil structure, tilth and water holding capacity); improved pest and disease management; and possible yield increases.

Snapp et al. (2005) also outline potential disadvantages of covercropping: harboring of certain cash crop diseases and pests; increased weed problems; problems associated with the residues (temporary decreases in available nutrients while they are broken down); interruption of the planting schedule (delayed planting); along with additional costs for labor, seed, and water. The authors emphasized that the benefits of cover crops tend to be long-term and somewhat cumulative. In addition, making informed decisions regarding the appropriate cover-cropping options for the unique farm situation can require heavy information gathering and experience by trial and error. Specific to the Salinas Valley, Smith & Cahn (2010) additionally point out that the high land rent rates in this area further discourage covercropping, where growing a non-cash crop is economically risky.

Panel members agreed with these literature findings, adding that although growers in the study areas utilize the practice, it was not especially common due to disease hosting concerns, capital and labor costs, delayed planting of the main cash crop, conflicts with tree harvesting methods that require a clean floor, and most especially, the fact that cover crops increase irrigation requirements when water availability is already a concern for their main cash crops. The latter was cited as the largest deterrent to growers incorporating cover crops into their rotations for both study areas. The multiple benefits of cover-cropping were thought to be generally well-known by growers and, despite some of the barriers, most panel members considered it a practice with potential to be more widely utilized should additional local and crop-specific trials occur along with associated educational outreach.

RP 7.2 Include deep-rooted or "nitrogen scavenger" crop species in annual crop rotations

Crops differ in terms of their rooting depth, their total N requirements and uptake curves over the season, the level and nature of the residuals left after harvest, and other characteristics that influence N

Adminsitrative Record
Page 34769

uptake efficiency and leaching. So-called nitrogen scavenger crops have deep root systems that can capture and recycle N leftover from the previous crop that would otherwise be leached if the ground were left fallow or planted again to a shallower rooted species (Osterli & Meyer 1977; Delgado 1998; Bassil et al. 2002; Delgado et al. 2008). For example, Zhou et al. (2008) showed that when following corn with wheat, the nitrate that has leached below the root zone of the corn is taken up by the deeper-rooted winter wheat that follows it. Unlike the more traditional cover crops (such as vetch), deep rooted N-scavengers are not restricted to the winter months, and using them in a crop rotation scheme usually produces a harvestable crop. Although scavenger crops may require N fertilization, they may also obtain sufficient N from the residuals of the previous crop (Bassil et al. 2002). Examples of such crops include sugarbeets, sorghum, ryegrass and other winter cereals and forages, as well as oilseeds, such as safflower. Alfalfa and other perennials can also be excellent N scavengers (see RP 7.4).

Barriers to incorporating N scavengers into crop rotations are similar to those cited for cover crops, especially the poor economic incentives. Rotation options are also sometimes limited by buyer preferences and requirements. Similarly, growers often do not want to plant crops associated with an uncertain market. For example, in the Tulare Lake Basin, oilseed processing facilities contract with certain growers to meet market demands, leaving non-contracted growers with high economic risk should they grow a crop for which little additional market share exists. There is also very little market for forages in the Salinas Valley, and in the TLB, sugarbeets, although an ideal scavenger and once representing well over 7000 hectares (19,000 acres) of the study area, are no longer grown due to the closure of the last processing facility in the Central Valley. Land tenure, land rent rates, and current price of such crops also need to be considered when developing rotation schemes. Often, crops with high extractive capacity are also of lower economic value, and thus, the high establishment costs present a significant economic barrier to alternative rotation schemes.

RP 7.3 Grow more crops per year (e.g., dairy forage triple crop rotation)

Growing more cash crops per year in annual cropping systems reduces the length of the fallow periods during which there is greater risk of nitrate leaching loss. The third crop should allow for an updated fertilizer application timing regime and ideally would be an N scavenger (see RP 7.2) or one that requires relatively small amounts of additional fertilizer. The best example of this practice is the addition of a late summer sudangrass crop into the more traditional summer silage corn-oats dairy forage double crop. Planted immediately after harvest of the silage corn, the sudangrass can then capture remaining residual soil N, and its presence may allow for manure or dairy lagoon water applications that would otherwise contribute to an ever-increasing soil N content at risk of leaching. As such, the addition of the third forage reduces the risk of nitrate leaching loss later in the winter compared to the more typical double crop.

Some dairy forage crop producers in the TLB are triple cropping in this manner. The main barriers to expanded use of this practice are the cost of the extra field operations and lack of availability of irrigation water. The extra forage produced tends to also be relatively low in feeding value. Additionally, the insertion of sudangrass crop into the rotation in the fall may delay planting of the winter cool-season

Adminsitrative Record
Page 34770

grass forage (oats, wheat, etc.). With later planting dates, there is an increased risk of early rains preventing entry of planting equipment into the field.

RP 7.4 Include perennial crop rotation (e.g., alfalfa or perennial grasses)

Perennial grasses, clovers, and alfalfa are generally very effective in capturing nitrogen that might be lost in annual cropping systems. Perennial forages, if irrigated, actively take up nutrients and water from the soil over a greater portion of the year than annuals, even where two or three annual crops are grown in sequence during a 12-month period. Legumes in general, and alfalfa especially, have the ability to obtain all needed nitrogen from the air (so-called biological N fixation, BNF) and therefore do not require any N fertilizer applications. However, when nitrogen is available in the soil due to presence of residual manure or fertilizer N, the BNF process stops, and the plants switch to the soil N source. Alfalfa is the most important high protein livestock forage in California and is known as "Queen of the Forages." Alfalfa is a very deep rooted species and can capture N from the soil at greater depths than corn or cotton. In a corn lysimiter trial, Toth & Fox (1998) demonstrated that nitrate leaching was reduced in fields incorporating alfalfa into the crop rotation. Watts reported that, within a year, alfalfa can remove nearly all nitrate from the soil up to several meters deep and that the crops following alfalfa in the rotation require less fertilizer N to be applied (Power et al. 2001). Pettygrove & Putnam (2009) estimate that the average alfalfa credit is between 50 and 80 pounds of N per acre (56-80 kg N/ha). In their literature review, (Power et al. 2001) found that the residual effects of alfalfa can last up to 5 years, with the caveat that nitrate leaching may not be reduced if the fertilization scheme for the following crop does not adequately account for the legume N credit. Panel members agreed that although many growers were aware of the N credit, they were unsure of how to utilize values in an N budget without fear of yield reduction (see RP 9.8 for more information on N budgeting).

Deep rooted perennial grass is also effective in reducing soil N levels (Dear et al. 2001). Entz et al. (2001) found that perennial grasses extracted higher levels of nitrate than continuous alfalfa in the shallower soil profile (<120cm), with similar extraction levels below this depth. California native perennial grasses can be used between tree and vineyard rows, and because they are summer dormant, offer the added benefit of not requiring additional water application (Ingels et al. 1994). In Central Valley vineyards, perennial clover and native grasses used between grape rows reduces vine vigor (a beneficial effect) and can reduce N fertilizer application needs (King & Berry 2005), although more water use may result (Monteiro & Lopes 2007).

Barriers to utilizing perennial crops in rotations are primarily economic. Perennial forages generally produce much less income than, for example, vegetables or other field crops such as cotton. In some parts of the TLB where water is quite limited, farmers may have enough water to grow tomatoes or cotton, for example, but not enough to produce alfalfa. Dairy producers, who can utilize the harvest for feed, will tend to only grow alfalfa when they know they will be able to produce enough of their other forages. Additionally, alfalfa can be more sensitive to complications associated with manure applications than other silage crops, so it is not as useful for recycling their manure. Harvesting conflicts preclude the use of perennial cover crops in raisin production and some fruit and nut trees due to the need for a "clean" vineyard/orchard floor. In their review, Crews & Peoples (2005) suggest that

Adminsitrative Record
Page 34771

decreasing N losses from agroecosystems will require increased utilization of perennials, although more research and information dissemination are required.

2.3.3 Manage Nitrogen Fertilizer and Manure to Increase Crop Nitrogen Use Efficiency

Synchronization of fertilizer and manure application rates and timing to match plant requirements is critical to reducing groundwater nitrate contamination from agricultural crop production. Total crop N requirements vary widely among crops as do nutrient uptake patterns throughout the growing season (**Error! Reference source not found.**). The rate, timing, placement and source of fertilizer materials are nder the control of the grower and make up the critical components of matching N supply with demand. Environmental variables, outside of grower control also affect the balance of supply and demand and should be reflected upon, as should crop restrictions on irrigation systems (including fertigation) and requisite cultural practices. These interrelated aspects of N synchrony, although complex, need all be considered simultaneously when managing N applications to increase N use efficiency and better control N leaching.

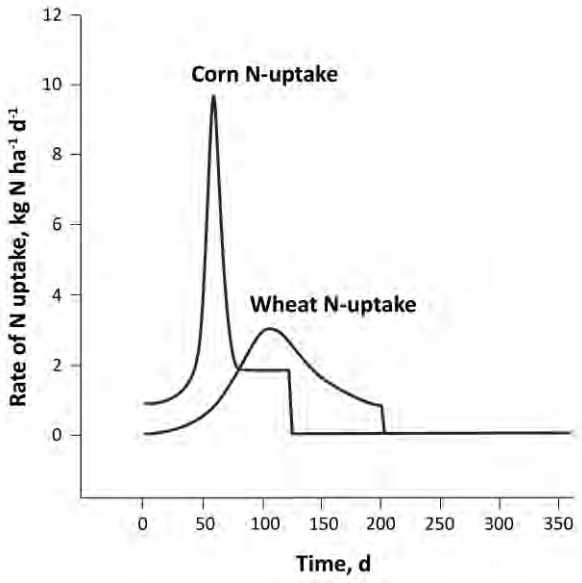

Figure 5. Contrasting N uptake curves for corn and wheat, exemplifying the crop-specific nature of optimizing timing of N applications. (adapted from Pang & Letey 2000, with permission).

MM 8: Improve Rate, Timing, and Placement of Nitrogen Fertilizers

As discussed in the following RPs, applying fertilizer at rates and times consistent with crop needs reduces leaching potential (Di & Cameron 2002; Meisinger & Delgado 2002; Meisinger et al. 2006; Li et al. 2007; Gheysari et al. 2009; Fessehazion et al. 2011). Besides proper rate and timing of fertilizer applications, nitrate leaching is affected by placement and the nitrogen source. Attention to variability in soil characteristics, water infiltration and areas of root proliferation can reduce ill-placed N

Adminsitrative Record
Page 34772

applications. Banding of fertilizer in row crops (as opposed to broadcast spreading) or with the irrigation water (fertigation) delivers fertilizer to areas of higher root densities, thereby decreasing N leaching risk (Kemper et al. 1975; Lehrsch et al. 2000). Urea-based products are better incorporated below the soil surface than broadcast onto the surface due to risks of volatilization losses. Attention to placement is important in early establishment periods when roots are yet poorly developed and irrigation tends to exceed ET. Nitrogen source (e.g. urea, ammonia, ammonium nitrates, manure, compost) plays a role in gaseous emissions (especially gaseous NH_3) and nitrate leaching, but effects are highly variable according to environmental and management factors and thus debatable. Discussion regarding N source is thus kept minimal in this document, primarily differentiating between organic and synthetic fertilizer sources.

RP 8.1 Adjust nitrogen fertilizer rates based on soil nitrate testing

Applying N rates above plant need increases the risk of nitrate leaching losses. Maintenance of profitable yield plateaus at certain fertilization rates and overfertilizing beyond this point is not uncommon in California (Hartz et al. 1994; Hartz 2006). Soil nitrate testing is a technique for estimating plant-available N in the soil and reducing risks of over-application. Overall crop N needs are very small until plants are well established and growing rapidly. The N concentrations in the small rooting zone needed for plant growth at this stage can be met by careful placement of small amounts of starter or preplant N fertilizers when needed. Often, prior to the rapid growth phase, residual N in the soil can meet early plant needs.

Soil testing at or just prior to planting can indicate if enough residual N is available for plant establishment, allowing fertilizer application to be delayed, to the benefit of groundwater (Di & Cameron 2002). Preplant soil nitrate testing identifies fields with adequate N for plant establishment and presidedress soil nitrate testing (PSNT) helps to determine in-season fertilization needs (both rates and timing). Applying this knowledge can provide substantial reductions in overall N application rates (Hartz et al. 2000; Krusekopf et al. 2002; Li et al. 2007; Malone et al. 2010). For example, Breschini & Hartz (2002) reported that PSNT use in California lettuce fields could reduce seasonal N applications by 40 percent. Split applications, based partially on soil testing, can also reduce total N applications (see RP 8.3). Soil nitrate testing at or just after harvest can indicate whether N fertilizer applications were excessive and may be useful in adjusting future rates. Both laboratory testing and on-farm quick tests are available. The on-farm quick tests offer a simple and relatively inexpensive way for growers to routinely monitor and gauge fertilization needs without having to wait for the return of laboratory results (Hartz, Smith, et al. 1994).

Barriers to soil nitrate testing and/or adequate use of soil testing include associated costs (primarily labor and management), complexity of collection regime, and spatial variability within a field. Spatial variability of soil nitrate is often very high, requiring that a large number of samples be taken to adequately characterize a field. High spatial variability is especially a problem in no-till fields and under drip irrigation. In drip-fertigated fields, plant roots and residual fertilizer N may be concentrated in a small zone of wetting, leading to some uncertainty in where to collect soil samples. Panel members agreed that a large portion of growers utilize soil testing, although they were unsure of relative

Adminsitrative Record
Page 34773

frequency and overall attention to the variability concerns noted above. On-farm quick-tests are not yet especially common, with the majority of growers contracting with commercial laboratories for testing. Growers of grain and other field crops (besides cotton) are exceptions to the trend of utilizing soil testing to guide application rates , although over-application of N in cereals is less common than in other field crops due to risk of lodging.[8]

RP 8.2 Adjust timing of nitrogen fertilization based on plant tissue analysis

Timing fertilizer applications to match crop need is critical to increasing nitrogen use efficiency and decreasing leaching potential (Adriano et al. 1972; Peacock et al. 1991; Christensen et al. 1994; Neetson et al. 1999; see also RP 9.3 below). Again, total N requirements are usually quite low during early plant establishment. For example, Pang et al. (1997) modeled leaching with various water application rates, fertilizer rates, and application timing. Higher N rates and earlier application dates produced the highest leaching fractions in the model, whereas delaying the second N application in grains reduced modeled leachate significantly (multiple dosing is reviewed in RP 8.3).

Plant tissue analysis can indicate if nutrient levels are adequate for meeting plant needs. During mid and late season, soil testing becomes a less accurate indicator of crop N needs. Using tissue testing in conjunction with continued soil testing can provide a superior gauge for timing applications (Pettygrove et al. 1998). These tests are primarily useful for determining nutrient deficiencies and not as useful for determining surplus N levels (Pang et al. 1997; Hartz et al. 2000; Breschini & Hartz 2002). The primary benefit of the tests then is to guide smaller applications over time by indicating when plant N levels are adequate. Note that if a defiency is detected via tissue analysis, it may be too late to fully compensate for the nutrient shortage. In-season and long-term monitoring of tissue testing result trends in relation to fertilizer application dates, and/or midseason soil testing results, can sometimes be indicative of opportunities to reduce or delay fertilizer applications (Pettygrove et al. 1998).

Soil nitrate testing in woody perennials is somewhat impractical due to their large rootzone and thus tissue analysis in these crops is generally a standard practice, most commonly using whole leaf tissue tests. Spatial variability within the tree can lead to uncertainty in comparing whole leaf tissue test results. Petiole testing is also common in some crops (e.g., cotton and grapes). The measured value can be affected by the time of day, temperature, presence of cloud cover, and plant water status, leading to uncertainty in interpretation. Similar to the literature cited above, some panel members felt that, with the test better at detecting deficiencies than surplus, its effect on leaching is questionable. Vegetable and row crop panel members disagreed over tissue testing's overall usefulness in annual crops, citing confounding factors of the results as well as the fact that crop damage or loss can occur before the lab can return the results.

[8] Lodging, most often associated with cereal crops, refers to the permanent loss of the plant's upright position, negatively affecting yields and increasing risk of disease, etc. Excess N applications can enhance lodging risk in cereals.

Adminsitrative Record
Page 34774

RP 8.3 Apply nitrogen fertilizer in small multiple doses rather than single large dose

To better match crop N uptake patterns (see Figure 5) fertilizer should be applied in multiple doses rather than a single application. In research on both tree and row crops, this practice has repeatedly been shown to increase N use efficiency, reduce total N fertilizer requirement, and reduce subsequent leaching below the rootzone (Chapman 1951; Russelle et al. 1981; Pang et al. 1997; Malusa et al. 2001; Breschini & Hartz 2002; Gehl et al. 2005; Quiñones et al. 2005; Vázquez et al. 2006; Gheysari et al. 2009; Lopus et al. 2010). For some crops, seasonal N uptake curves have been published and can be used by growers to estimate weekly or even daily N uptake quantities. These can be used together with soil and tissue testing to make N fertilizer timing and rate decisions. Split fertilizer application is especially straight forward and highly utilized in drip irrigation systems (fertigating), although sprinkler and furrow fertigation systems can also lend themselves to the practice, as can tractor applied sidedressings. Leaching is typically more highly reduced with fertigation, as it normally equates to higher application uniformity as well as higher numbers of split applications. For example, Alva et al. (2003) showed higher reductions in leaching from lemon orchards when N application was split between more frequent (15X) fertigations as compared to less frequent (4X) tractor applied sidedressings.

Barriers to split application practices primarily are cost and educational requirements. The practice is employed by growers in the study areas, largely independent of irrigation methodologies, although in field crops less so than in vegetable and permanent crop production. Panel members suggested that where splitting between one or two applications may be common in non-fertigated systems, it is often justified to split fertilization into three, four, or more applications, and thus there is room for improvement.

RP 8.4 Measure nitrate content of irrigation water and adjust fertilizer rates accordingly

If not regularly testing irrigation water, especially where the nitrate level in an irrigation water source is high (> 5 ppm N) and/or fluctuates during the season, N application above plant need is more likely. Any nitrate present in water from irrigation wells can be used to grow the crop, simultaneously reducing spending on N fertilizer. This is the central premise of the pump-and-fertilize groundwater remediation concept discussed in Technical Report 5 (King et al. 2012). Irrigation water samples should be taken after the pump has run for at least 15 minutes and kept cool or frozen en route to the laboratory for analysis. Several test kits are commercially available and can be used for on-farm testing.

To calculate the appropriate N fertilizer credit, farmers should use the volume of water actually used by the crop, which is usually less than the amount applied. Crop water use can be estimated with sufficient accuracy for this purpose with the CIMIS crop evapotranspiration online tool (see RP 2.1)

Panel members indicated that although some growers are measuring their irrigation water for nitrate and reducing fertilizer rates accordingly, many do not. Generally, more testing and accounting takes place in areas with relatively higher well water N, and dairy farmers are now required to test their water annually. Adjusting fertilizer rates based on nitrate levels in well water is complicated by the seasonal variation of N levels and by irrigation of individual fields using multiple water sources (e.g., additive N

Adminsitrative Record
Page 34775

levels of multiple wells and/or dilution by surface waters). Panel members also suggested some reluctance by growers that rent their farmland to test their landlord's water.

RP 8.5 Use low rates of foliar nitrogen instead of higher rates applied to soil

Foliar applications of urea based N fertilizer, used in conjunction with reduced rates of soil applied N, have shown some success in increasing nitrogen use efficiency. For example, Lovatt (1999) showed that, with specifically timed foliar applications of low-biuret urea, citrus and avocado fruit set and fruit size increased, improving overall yields. While maintaining yields, leaching risk from the soil is reduced when a portion of the N needs are applied to the leaves (Dong et al 2005; Gonzalez et al. 2010). Johnson et al. (2001) found that peach quality and yield were equivalent when applying 112 kg/ha (100 lb/ac) N to soil versus a treatment with 56 kg/ha foliar application and 56 kg/ha (50 lb/ac) soil application.

Although foliar application of some micronutrients is more common, especially in tree production, foliar application of nitrogen is not as common and is used only on a very restricted number of crops, mostly subtropical tree fruits. In general, it is not as practical in field and vegetable crops, due to their relatively high N requirements. Foliar applications of N fertilizers must be applied in small doses to prevent damage to plant tissue, and thus the cost of applying materials can be significantly higher than for ground application. Also, conversion of many tree fruit and nut orchards to drip or microsprinkler irrigation and use of fertigation has eliminated some of the advantage of foliar application of N (foliar micronutrient application is still proving to be useful in some tree crop production regardless of irrigation system). Panel members agreed that although the practice may hold some promise, more research is required to prove the value of benefits compared to costs in various crops.

RP 8.6 Vary nitrogen application rates within large fields according to expected need

Large fields often vary in their soil characteristics, which in turn affects water holding capacity, N availability, and plant uptake of nitrogen. By varying N application rates according to soil physical and chemical characteristics, rather than applying the same rate everywhere, increased nitrogen use efficiency can be achieved (Khosla et al. 2002; Mann et al. 2010).

In site specific management, or so-called precision agriculture, application rates of fertilizers, pesticides and irrigation water are varied within a field to better match the field variability, such as variable soil characteristics or pest and disease pressures. Satellite imagery, GIS, and other technologies are used to detect plant and soil characteristics within the field and zone the field according to needs. GPS-equipped tractors can then be automated to make applications by zone. Precision agriculture is increasingly advancing technologically and has been adopted commercially primarily in field crops and some tree crops outside of California, where increased nitrogen use efficiency is based on site-specific management zones (Khosla et al. 2008; Mann et al. 2010). An important caveat to this practice is that although it can enhance NUE and/or yields, if the reason for a zone's low soil N is due to, for example, lower water holding capacity (i.e. an area of sandier soil), then adding more N to this area is actually going to contribute to additional loss of nitrate below the rootzone, and thus acting as a counterproductive practice in terms of groundwater quality.

A survey one decade ago of agricultural scientists from California universities (Wagner Weick 2001), indicated that advances in the GIS and GPS technologies used in precision agriculture were expected to greatly impact the state's agricultural efficiency, and in time this may be the case. Highly automated precision agricultural systems are as of yet uncommon in California and will likely remain as such until the technologies are further developed to meet local conditions and needs. Large (>80 acres) fields of combine-harvested crops in rainfed systems may have high yield potential variation, as well as high soil variability and slope variability within the same large field. Californian irrigated fields often have more uniform conditions in that rain does not complicate water application uniformity, a large percentage of crops are grown in flat valley floors, and, in most rotations, fields here tend to be smaller, meaning less variable soil characteristics. Additionally, more research is required to develop some of the techniques for the myriad crops grown here, as again, the most developed techniques are specific to large fields of grains and other combine-harvested crops. Precision agriculture techniques that may be considered currently available to California growers relate more to irrigation than fertilizer applications, and with the former it generally only applies to micro-irrigation systems, and to a lesser extent, sprinkler systems. Additional irrigation infrastructure would also be required (such as valves or soil moisture sensors). Along with the capital costs for programs and infrastructure, additional barriers include the lack of interoperability between programs and the significant time required to manage and analyze data.

Panel members agreed that the few growers who do manage to site specific variables, utilize past harvest information and soil samples to guide their N application rates spatially. This is relatively common in fruit production within the TLB, facilitated by the fact that often different varieties are planted within the same field or block and it is already necessary to manage on a row-by-row or block-by-block basis. The practice is largely not applicable to Salinas Valley producers as fields are typically small and where field boundaries have been updated to stay within more homogenous soil zones.

RP 8.7 When fertigating in surface gravity systems, use delayed injection procedure

Injection of N fertilizers into surface-run irrigation water is a practical way to provide nutrients in mid- or late-season when the crop is too big to permit tractor applications, and is commonly used in corn, tomatoes, and other row crops. The delayed injection technique is a method for improving the spatial distribution of the applied nutrients by adjusting the timing of the fertilizer injection in the water. In this method, the water is allowed to advance 25-50% across the field before fertilizer valves are opened. The fertilizer rapidly "catches up" with the advancing water front and therefore reduces the time that nitrogen is subject to leaching at the upper end of the field (Pettygrove et al. 1998). The procedure enhances N distribution uniformity and may decrease the total amount of N fertilizer needed (Schwankl & Frate 2004; Pettygrove et al. 2007).

Performance of the technique requires some additional labor time for opening the fertilizer tank out of sync with starting the irrigation run. Also, where surface gravity irrigation is already adequately uniform, there will be little or no advantage to delayed injection. Panel members agreed that although the practice has some mitigative potential, the logistical concerns associated with the additional time and labor requirements have kept adoption of this method relatively low within the study area.

Adminsitrative Record
Page 34777

RP 8.8 Develop a nitrogen budget that includes crop nitrogen harvest removal, supply of nitrogen from soil, and other inputs

A nitrogen budget is a tool for planning N fertilizer applications and for evaluating over time the N use efficiency achieved in a field. Crop N harvest removal is defined as expected yield multiplied by the expected N concentration in the harvested product. A simple budget is made by considering this harvest removal amount in conjunction with N inputs, including fertilizer, manure, soil N levels, and irrigation water N levels. After harvest each year, inputs and outputs are updated with actual values and the budget utilized in planning for the next year.

In many situations, N budgeting is not very useful for annual planning of N fertilizer requirements. This is due to the uncertainty in the magnitude of N losses due to leaching, runoff, and gaseous losses. Additional uncertainty lies in the amount of N that is likely to be provided to the plant by mineralization, the conversion of organic N forms (from plant residues, soil organic matter, and any applied manure or compost) to plant available (inorganic) N forms during the growing season. These factors may vary from year to year and among fields, and general values from the literature are not case-specific enough to allow useful budgets to be constructed. However, with careful recordkeeping and with experience, growers may be able to use a budgeting approach to approximate, for example, adjustments to N fertilizer rates in response to expected changes in crop yield or in N content of irrigation water.

Nitrogen budgets may have some use in assessing crop N use efficiency achieved in the recent past. N budgets for individual fields are currently required of dairy producers in the Central Valley under the General Order Waste Discharge Requirements, implemented in 2007. The intended purpose of these budgets is to allow the producer and the regulatory agency to evaluate, after the fact, whether nitrate leaching losses were likely low, as would be suggested by high crop N use efficiencies. Compliance with the N budgeting provision requires producers to measure or estimate and record the following on each crop in each field: crop yield, N content of harvested crops, N content of all irrigation water and volume of water applied, weight and N content of all manures (liquid and solid) applied, and amount of N fertilizers applied. These data would allow comparison of total N inputs to crop N harvest removal, thus to infer from the difference whether leaching losses of N may be excessive. This is a relatively new regulation, and it is not known whether such N budgets will provide useful information, either to farmers or to the regulatory agency. It is reasonable to ask whether such a recordkeeping approach used for crops (other than on the currently regulated dairies in the Central Valley, where most of the crops are forage species) would provide the information needed to evaluate crop N use efficiency.

Nitrogen budgeting, that is, measuring the difference between N inputs and N removed in the harvested crop, and tracking this over a period of years, appears to have the most potential as an assessment tool in annual crop rotations (on a multi-year scale), especially vegetables, forages, and grain crops. It is more difficult to envision its utility for grapes, tree fruits, and nuts, because in those crops, the permanent part of the plant (trunk, branches, and roots) stores nitrogen over a period of years. Additionally, for some crops (e.g., watermelon, lint cotton, leaf lettuce), obtaining a representative sample of the harvested crop and preparing the sample for laboratory analysis would be difficult, and protocols for doing this would need to be developed.

Adminsitrative Record
Page 34778

Panel members stated that emphasis needs to be placed on the fact that crop fertilization requirements are considerably larger than harvest removal. Members agreed that keeping this fact in mind, harvest removal does serve as an important grower evaluation tool of their overall nitrogen use efficiency on a multi-year scale. Considerable additional management time and the level of expertise required to track and analyze an N budget are the main barriers to this practice, as well as record privacy concerns. Along with the need for significant educational outreach, additional research is required to evaluate average per-crop removal rates to accommodate increased use of this figure in nutrient budgeting calculations (Heckman et al. 2003).

RP 8.9 Use controlled release fertilizers, nitrification inhibitors, and urease inhibitors

In recent decades, a number of products have emerged that have the potential to increase overall N use efficiency. Slow release fertilizers, nitrification inhibitors, and urease inhibitors have been shown to benefit crop N use efficiency as discussed below. Note that these products are not applicable to most drip irrigated crops.

Controlled (or slow) release fertilizer (CRF) pellets are either compounds of low water solubility or are coated with a material that restricts the dissolution of soluble N over time. Some success in increasing N use efficiency has been documented with some of these products (Diez et al. 1995; Chen et al. 2008). For example, Zvomuya et al. (2003) showed that Nitrate leaching was more significantly reduced in potato fields fertilized with polyolefin-coated urea than in those receiving split applications of uncoated urea. However, some studies have pointed to more soluble N being left in the soil after harvest than with traditional fertilizer, which is then subject to leaching (Delgado & Mosier 1996). Furthermore, it has been suggested that these products are most suited to conditions outside of California where summer rains minimize grower control over water applications (and thus leaching) and where growers have less room to synchronize N applications with plant need than California growers do (Hartz & Smith 2009). Although California strawberry production makes heavy use of these relatively expensive fertilizers, the drip irrigation and fertigation (offering precise control over both water and N applications, respectively) already common to California vegetable fields, would likely show little benefit with the use of CRF products. Panel members agreed that outside of strawberries, controlled release products are not highly used due both to their relatively high costs, as well as uncertainty regarding how product N release rates match crop N demands.

Nitrification inhibitors reduce the oxidation of ammoniacal N to nitrate, where the former is plant available, but not subject to leaching unless transformed. Nitrification inhibitors have been shown to increase N use efficiency and decrease leaching losses, and in some cases increase yields (Serna et al. 1994; Shi & Norton 2000; Cameron & Di 2002; Di & Cameron 2004; Quiñones et al. 2009). In his meta-analysis of the nitrification inhibitor nitrapyrin, Wolt (2004) found that in 75% of the studies reviewed, use of the inhibitor increased soil N retention and yields while decreasing volatilization and nitrate leaching. Gaseous losses have also been shown to decrease substantially with the use of the nitrification inhibitors (Dittert et al. 2001; Cameron & Di 2002; Hatch et al. 2005; Chen et al. 2008; Carneiro et al. 2009). Panel members indicated that although the use of these products is more

Adminsitrative Record
Page 34779

common than use of slow release fertilizers, altogether they are not especially popular due to cost-return uncertainties.

Urease inhibitors slow the conversion of urea to ammonia, thereby reducing the potential volatilization loss of ammonia nitrogen when urea is left on the surface of the soil for more than a few hours. Immediate incorporation of urea fertilizers following application will mostly eliminate such volatilization losses. In the study areas, there are only a few situations (e.g., orchards or pastures receiving broadcast application) where urea is subject to this loss. By decreasing these gaseous losses, urease inhibitors have been shown to increase nitrogen use efficiency by reducing total N application requirements, thereby decreasing leaching risk (Watson et al. 1994; Gioacchini et al. 2002; Chen et al. 2008; Sanz-Cobena et al. 2008; Watson et al. 2008; Dawar et al. 2010a; Zaman et al. 2010; Dawar et al. 2010b; Sanz-Cobena et al. 2011). Panel members agreed that these products are utilized by some growers, but not the majority. New products are continuing to be developed and introduced into the marketplace and additional research is warranted as N use efficiency has already been shown to be significantly enhanced by both urease and nitrification inhibitors. Thus, with more research and educational outreach, the existing uncertainty and cost barriers may be overcome.

MM 9: Improve Rate, Timing, and Placement of Animal Manure Applications

Animal manures include both solid and liquid materials and wastewater from animal housing (e.g., dairy lagoon water). Such materials represent a significant source of crop nutrients in Central Valley.

Nearly all dairies in the Central Valley collect a portion of the cattle excreta by flushing concrete surfaces with water. The concrete surfaces include milking facility floors and lanes in barns and corrals. Additionally, during the rainy season, runoff from calf pens or other animal housing must be retained and usually is collected in the same system as the flushed manure water. On many dairies in the Tulare Lake Basin, more than 50% of all animal excreta is collected in this manner and stored in retention ponds as a dilute liquid, so-called “lagoon water”, prior to application to crops. This liquid manure cannot be economically transported for long distances and is usually applied to dairy forage crops and less commonly to non-forage crops on adjacent farms.

All dairies in the Central Valley are subject to regulations aimed at preventing manure nutrients from polluting surface and ground water. [9] A key feature of the regulation is the requirement that total N from all sources (manure, fertilizer, irrigation water) applied to a field not exceed 140 (or in special circumstances 165%) of removal of N by the harvested crop.

To comply with this N loading limit, while retaining current dairy stocking rates, many dairy producers in the Central Valley transfer manure solids (and to a far lesser extent, lagoon water) to other farms. A constraint to this transfer is that application of raw manure and even fully composted manures is restricted on some crops by processors and other buyers. Due to food safety concerns, use of manure products on crops destined for human consumption is decreasing.

[9] The General Order for Waste Discharge from Existing Dairies in the San Joaquin Valley, see: http://www.waterboards.ca.gov/centralvalley/water_issues/dairies/dairy_program_regs_requirements/index.shtml

Adminsitrative Record
Page 34780

Thus, there is a need to identify appropriate N management practices for two distinct situations: (1) application of manure and dairy lagoon water on dairy crop fields that fall under the General Order, and (2) application of manure on farms in the region that are not regulated under the General Order. In the second situation, farmers are not subject to a regulatory nitrate loading limit, and they are more likely to be applying manure (or compost made with this manure) without reducing N fertilizer application rates. Additional research is required to address the recent increase in off-farm transfer of dairy wastes and to develop proper manure recycling methods. This should include a comprehensive education and outreach effort aimed at growers not accustomed to accounting for organic N in their nutrient budgets.

Compared to commercially-manufactured inorganic fertilizers, manure, both solid and liquid forms, is far more heterogeneous in its physical properties and nutrient form and amount. It is more difficult to apply to fields uniformly, and its nutrient value to crops is less certain due to difficult-to-predict release rates of plant-available nutrient forms, fluctuating (and sometimes high) volatilization rates, influence of animal nutrition, and other factors.

Approaches for minimizing nitrate leaching losses from crop fields receiving dairy manure are listed here. For dairies, this includes installing or upgrading the infrastructure needed for collecting, storing, and applying lagoon water to fields. For non-dairy farms, this encompasses management of solid manure or composts made with manure. Recommended practices listed here are focused on careful characterization of manure nutrient content, nutrient budgeting, and more uniform application to fields. Proper collection and storage of manure prior to cropland application is covered in section 2.3.4.

RP 9.1 Apply moderate rates of manure and compost, and use materials with high available nitrogen content to meet the peak nitrogen demands

Where possible, growers should not rely solely on low-N manures, composted manures, or decomposed manures for the entire supply of N to the crop. Leaching losses have been found to be higher in manure fertilized fields than those strictly fertilized with inorganic N sources, due to poor synchronicity between plant demand and mineralization of residuals (Kirchmann & Bergström 2001). Only a small portion (5-30%) of the organic N in such materials will be converted to plant-available forms in the season following application. Therefore, to support the current seasonal crop N needs, a very high application rate would be required, resulting in a buildup of residual organic N that is released and subject to leaching when crop N demand is low. A more N-conserving strategy is to use moderate rates of low-N supplying manure and rely on inorganic N fertilizers or high N-supplying organic materials, such as dairy lagoon water, high-N poultry manure, blood meal, etc. to meet peak N needs.

The main limitation to this approach is the physical heterogeneity of manure and the uncertainty in behavior of manures and other organic fertilizers, which limits the farmer's ability to apply precisely timed and measured doses of N and other nutrients. Additionally, lowering manure N applications while retaining current dairy stocking rates translates into manure needing to be spread over significantly larger areas, often outside of the cropped property managed by dairy owners (as addressed in the overview of MM 9 above). The cost of this transportation is thus an additional barrier to adoption and

Adminsitrative Record
Page 34781

the logistics involved in off-farm application will often also involve additional management and regulatory cost.

RP 9.2 Incorporate solid manure immediately to decrease NH_3 volatilization loss

Fresh manures, non-composted manures, and dairy lagoon water contain high levels of ammonium N, and are subject to ammonia volatilization loss. This loss creates uncertainty in the N fertilizer value of these materials, which may then prompt a farmer to apply additional N to reduce the risk of N deficiency. Ammonia may be lost within hours of application, and even very shallow incorporation into the soil will prevent this loss (Rotz 2004). By incorporating manure quickly, the available N forms will be conserved, allowing for a more reliable reduction of inorganic N fertilizer rates. It is a relatively common practice to quickly incorporate surface applications of solid manures and composts, especially on fields associated with dairies. Although manure product use on crops destined for human consumption is decreasing due to food safety concerns, there is some room for improvement by these growers (not associated with dairies) to match timing of custom manure spreading with incorporations. Note that solid manure applied to alfalfa and irrigated pasture and dairy lagoon water applied with furrow irrigation water cannot be incorporated into the soil.

RP 9.3 When applying liquid manure in surface gravity irrigation systems, use the delayed injection procedure to improve application uniformity

As discussed in RP 8.7, the delayed injection procedure can improve the application uniformity of fertilizers and other materials during furrow irrigation, and this is applicable to liquid manure injections as well (Schwankl & Frate 2004). In this practice, the liquid manure is not injected into the surface gravity irrigation water until the water has advanced part way across the field. The same barriers as cited in RP 8.7 (increased labor and conflicts in ranch logistics) were noted by members of the manure panel, with the addition of field delivery logistics being complicated by the fact that the lagoon may be some distance from the field to which the lagoon water is being applied.

RP 9.4 Use quick test methods to monitor dairy lagoon water nitrogen content immediately before and during application, and adjust application rate accordingly

Dairy producers in the Central Valley are required by regulation to analyze the composition of dairy lagoon water applied to each field on a quarterly basis. For many large dairies, due to high variability of the lagoon water composition, sampling and analysis needs to be more frequent in order to provide a better basis for nutrient applications. A quick test method for both the ammonium and organic N fractions has been developed, which is being used by some dairy producers as an on-farm method to guide targeted lagoon water N applications. According to panel members, the quick tests are not highly utilized in the study area.

Limiting the use of this test are the extra labor required for collection and analysis of the sample, and the extra management and education needed for recordkeeping, calculations, and interpretation of the data. Also, the test developed by UC for this purpose generates a small amount of mercury-containing

Adminsitrative Record
Page 34782

waste, which must be disposed of in accordance with state and county public health law. Panel members added that, due to highly variable N concentrations and the need for testing to coincide as much as possible with planned application, development of a continuous monitoring device would be highly beneficial. At the very least, a quick test for lagoon water that does not have the mercury as a byproduct would be useful.

RP 9.5 Develop a nitrogen budget that includes crop nitrogen harvest removal, supply of nitrogen from manure, and other inputs

When using dry or liquid manure, it is important that adjustments to inorganic fertilizer application rates be made by determining the N content and N availability of the manure that will be applied. An N budget can then be created that, along with manure N-content, will consider N already present in the soil and irrigation water, as well as crop harvest removal, as discussed in RP 8.8. In setting N application rates, credit for residual N from composts and manures applied in previous years should be included. This can seem like a minor adjustment to N rates, but, as residual N builds up in the soil over a period of years with repeated applications of manure or compost, mineralization from this source will become significant. A commonly recommended strategy is to apply higher rates of compost/manure at first, then decrease over a period of years to a long-term sustainable rate based on the total (rather than immediately available) N content of the manure.

Besides the barriers described for RP 8.4, N budgets where manure is a source of nutrients are less certain than for crops receiving N only from inorganic commercial fertilizers, due to the uncertainty of mineralization rates of the organic N in manure. Variation in manure composition (for example, from one truck load to the next or during lagoon water pumping) can significantly increase costs of manure sampling and analysis (see RP 9.4).

RP 9.6 Calibrate solid manure and compost spreaders

Many growers applying manure or other organic amendments to their cropland hire custom application companies, and such companies usually weigh materials, in order to relate the weight to the volume capacity of the spreader. Knowing the water content and the weight of materials (and not merely the volume) applied is important, because the nutrient content of such materials is usually reported by analytical laboratories on a dry weight basis. Several techniques for calibrating application equipment exist. Barriers to improved management of this activity include the fact that calibration may be inconvenient, requiring extra time. Additionally, for dairy producers applying manure with their own equipment (rather than hiring a custom applicator), maintenance and calibration of spreaders is often not a high priority.

2.3.4 Improve Storage and Handling of Fertilizer Materials and Manure

Proper handling of fertilizers, manure, and other farm input materials is relatively straightforward with little to no extra time or money involved for producers. Simply ensuring farm workers are educated and

Adminsitrative Record
Page 34783

trained to remain vigilant in regards to the handling of farm inputs can minimize risks to water quality. The following practices are adapted from Pettygrove et al. (1998).

MM 10: Avoid Fertilizer Material Spills and Manure Discharges During All Phases of Transport, Storage, and Application

Government and industry standards for fertilizer storage facilities exist and should be met by producers. These practices are applicable to all operations and generally have minimal financial and time costs. Training staff and regularly reminding them of the importance of observing protocol regarding the transport, storage, and application of farm input materials need to be standard operating procedure. If hiring custom applicators, farmers should verify that applicator employees are trained as well, prior to entering the business arrangement. Written cleanup procedures posted in storage, mixing and loading areas enhances staff awareness of spill cleanup protocol. Again, most operations already observe these recommendations, but ensuring vigilance should be a priority given the risks to water quality when protocol is not followed or accidents take place. Supervision of inexperienced staff during mixing and transfer activities can reduce potential for spills. Products should be clearly and properly labeled to avoid handling mix-ups.

Permanent mixing, loading, and storage areas should be located downslope of, and an adequate distance away from wells and surface water. These areas should be equipped with an impermeable floor with a curb to collect spills, and storage areas should have a permanent roof. Regular checks for signs of cracks or seepage should be conducted. As a precaution, a temporary plastic lined berm can be used when mixing and loading are performed in the field.

Storing minimal amounts of product onsite reduces the potential for spills and contamination. Hiring custom applicators and purchasing products close to the time of need are two ways to reduce the need for on-farm storage. The bermed containment area in storage and mixing areas should have a capacity to contain a minimum of 110% of the volume of product in storage.

RP 10.1 Do not overfill trailers or tanks. Cap or cover loads

Overfilling equipment beyond what it can safely handle unnecessarily increases the risk of spillage and waste. Similarly, ensuring liquid or solid fertilizer loads are properly capped or covered will keep materials in place, decreasing the opportunity for wind-born losses. This practice applies to all operations, has virtually no cost involved in its adoption, and should pose no risk to yield or quality of harvests.

RP 10.2 When transferring fertilizer, take care not to allow materials to accumulate on the soil

Bulk materials should be unloaded over concrete or other impermeable surfaces to allow for quick clean up. If leaks are noticed during liquid transfers, the transfer should be halted and the leaks attended to prior to resuming transfer. Taking precautions when utilizing a belt loader to transfer dry fertilizer can

Adminsitrative Record
Page 34784

reduce the amount of fertilizer that accumulates below the belt. By opening the hopper door of the transport vehicle slowly and only after the belt has turned on, waste will be reduced.

RP 10.3 Maintain all fertilizer storage facilities and protect them from the weather

Storage facilities should be protected from the weather with a permanent roof, and should include an impermeable floor equipped with a berm to contain spills.

RP 10.4 Clean up fertilizer spills promptly

Should a spill occur, immediate attention should be given to contain the product. The goal is to reduce the amount of contamination resulting from the spill as much as possible. As such, care should be taken to ensure spilled material does not enter drains, waterways, sewers or wells. Dry spills should not be cleaned up with water. After cleanup, the spilled product should be applied to the field uniformly and at rates compatible with crop need.

RP 10.5 Shut off fertilizer applicators during turns and use check valves

A simple money- and environment-saving practice, turning off applicators when turning the tractor at non-cropped row ends saves otherwise wasted fertilizer and reduces contamination risks. Check-valves prevent leaks while the equipment is off and can be installed at all critical points besides just shank orifices.

RP 10.6 Maintain proper calibration of fertilizer application equipment

Applicator rates should be verified prior to fertilizer and manure application to the field and adjusted accordingly. Similarly, water and chemical flow metering devices should be calibrated for precise injection.

RP 10.7 Wellhead Protection

Wellhead protection via installation of wide concrete backstops circling the well is also mandated. Many wells were built prior to this legislation however, and often are only retrofitted once some other issue comes about, at which point the investigating well-technician mandates the upgrade. Poorly maintained protective devices, such as cracks in the casing also are a cause for concern. Panel members stated that on some farms fertilizer or pesticide tanks have been installed in close proximity to wells, posing a risk despite all other components being up to code. Generally however, these source protection practices are in high use.

RP 10.8 Distribute rinse water from fertilizer application equipment evenly throughout field

Rinse water should be treated just as fertilizer and applied evenly to the crop in order to avoid creating areas with nutrient concentrations beyond what is required by the crop. This can be easily accomplished by discharging the rinse water into the irrigation stream, for example.

Adminsitrative Record
Page 34785

RP 10.9 Avoid manure spills/discharges during transport, storage, and application

Manure should be transported, stored, and applied so as to prevent spills or discharges that would pollute water with nutrients and pathogens. Methods for safe handling and storage of manure are described by the Livestock and Poultry Environmental Learning Center, specifically the Animal Manure Management *Handling and Storage Options for Manure* section (2011).[10] The Central Valley Regional Water Quality Control Board adopted the General Order for Waste Discharge from Existing Dairies[11] in May of 2007. In addition to manure land application practices (see MM 9), the General Order regulates design and operation of liquid manure storage lagoons and corrals. Requirements for manure storage ponds include:

- An adequate storage volume, with the pond designed to create mixing (i.e., a uniform composition) of liquid to be delivered to crop fields
- A distribution system (pumps, pipelines) that enables delivery of small doses of lagoon water to fields, along with a flowmeter
- A conveniently located sampling valve

[10] http://www.extension.org/pages/8638/handling-and-storage-options-for-manure
[11] http://www.waterboards.ca.gov/centralvalley/water_issues/dairies/dairy_program_regs_requirements/index.shtml

Adminsitrative Record
Page 34786

2.4 Management Measure Summary

In this section we summarize all recommended practices in table form below (Table 2) and with short recapitulations of each practice's estimated current extent of use within the study areas and associated barriers to increased adoption.

Table 2. Summary of management practice current extent of use and barriers to increased adoption.

YQ (risk to yield or quality, poor market) CC (capital cost--infrastructure, equipment) OC (operational cost--labor, management, training) RL (ranch logistics) LT (land tenure) ED (education, training, demonstration needs) IT (insufficiently developed technology or more research needed)	**Current Extent of Use**		**Barriers to increased adoption**
	Common practice in some crops	Less common practice	
1. Perform irrigation system evaluation and monitoring			
1.1 Conduct irrigation system performance evaluation		x	OC, LT, ED
1.2 Use flow meters or other measuring devices to track water volume applied to each field at each irrigation	x		CC, OC, ED
1.3 Conduct pump performance tests	x		OC, ED
2. Improve Irrigation scheduling			
2.1 Use weather-based irrigation scheduling	x	field crops[12]	OC, RL, ED, IT
2.2 Use plant-based irrigation scheduling		x[12]	OC, RL, ED
2.3 Use soil moisture content to guide irrigation timing and amount	x	field crops[12]	OC, RL, ED
2.4 Avoid heavy pre-plant or fallow period irrigations		x	YQ, RL, ED
3. Improve surface gravity system design and operation			
3.1 Convert to surge irrigation if appropriate		x	CC, OC, RL, ED
3.2 Use high flow rates initially, then cut back to finish off the irrigation		x	OC, RL, ED
3.3 Reduce irrigation run distances (i.e., field lengths) and decrease set times		x[13]	YQ, CC, OC, LT, ED
3.4 Increase flow uniformity among furrows, e.g., by compacting furrows, use of "torpedos"		x	OC
3.5 Grade fields as uniformly as possible	x		OC, ED
3.6 Where high uniformity and efficiency are not possible, convert to drip, center pivot or linear move systems	x		CC, OC, LT, ED

[12] According to panel members, calendar scheduling remains common in much of the TLB's field, grain and hay production due to the barriers cited. Tree crop and vegetable producers tend to make more use of ET and CIMIS data, although historical ET is more commonly utilized over the more effective real-time CIMIS. Plant or soil moisture monitoring techniques are also frequently used in tandem to guide their water application rate and timing decisions.

[13] TLB fruit orchards and SV vegetable fields utilizing surface irrigation already have reduced run distances.

Adminsitrative Record
Page 34787

YQ (risk to yield or quality, poor market) CC (capital cost--infrastructure, equipment) OC (operational cost--labor, management, training) RL (ranch logistics) LT (land tenure) ED (education, training, demonstration needs) IT (insufficiently developed technology or more research needed)	**Current Extent of Use**		**Barriers to increased adoption**
	Common practice in some crops	Less common practice	
4. Improve sprinkler system design and operation			
4.1 Monitor flow and pressure variation throughout the system		x	OC
4.2 Repair leaks and malfunctioning sprinklers, follow manufacturer recommended replacement intervals	x	x (optimal frequency)	CC, OC, ED
4.3 Operate sprinklers during the least windy periods (when logistically possible)	x		RL
4.4 Use offset lateral moves		x	OC, RL, IT
4.5 When pressure variation is excessive, use flow control nozzles		x	CC, LT, ED
5. Improve drip and microsprinkler system design and operation			
5.1 Use appropriate lateral hose lengths to improve uniformity	x		ED, CC
5.2 Check for clogging potential and prevent or correct clogging	x	x (optimal frequency)	OC, CC, ED
6. Make other irrigation infrastructure improvements			
6.1 Installation of subsurface drains		x	CC, IT[14]
6.2. Backflow prevention	x		CC, ED
7. Modify crop rotation			
7.1. Grow cover crops		x	YQ, CC, OC, RL, ED, IT
7.2 In annual crop rotations, include deep-rooted or "N scavenger" crop species, such as oilseeds, and sugarbeet		x	YQ, CC, OC, RL
7.3 Grow more crops per year, e.g., dairy forage triple crop rotation		x	OC, RL
7.4 Include perennial crop in rotation, e.g., alfalfa or perennial grasses		x	CC, RL, LT

[14] Subsurface drains are already commercially developed; rather effluent disposal and the option of installing drains outside of areas of perched water tables need more research and development.

Adminsitrative Record
Page 34788

YQ (risk to yield or quality, poor market) CC (capital cost--infrastructure, equipment) OC (operational cost--labor, management, training) RL (ranch logistics) LT (land tenure) ED (education, training, demonstration needs) IT (insufficiently developed technology or more research needed)	**Current Extent of Use**		**Barriers to increased adoption**
	Common practice in some crops	Less common practice	
8. Improve rate, timing, placement of N fertilizers			
8.1 Adjust N fertilizer application rates and timing based on soil nitrate testing	x	x (frequency, quick tests)	YQ[15], OC, ED
8.2 Adjust timing of N fertilization based on plant tissue analysis	x (where appropriate)		YQ[1515], OC, ED
8.3 Apply fertilizer N in small multiple doses rather than single large doses	x	x (optimal frequency)	OC, ED
8.4 Measure N content of irrigation water and adjust fertilizer rates accordingly		x	OC, LT[16], ED
8.5 Use low rates of foliar N to replace a portion of soil applied N (when applicable)		x	OC, ED, IT
8.6 Vary N rates within large fields according to expected need, rather than applying the same rate everywhere		x	OC, CC, ED, IT
8.7 When N fertigating in surface gravity systems, use delayed injection procedure		x	OC, RL, ED
8.8 Develop N budget that includes crop N harvest removal, supply of N from soil, and other inputs		x	OC, ED, IT
8.9. Use controlled release fertilizers, nitrification inhibitors and urease inhibitors		x	YQ[15], CC, ED, IT
9. Improve rate, timing, placement of animal manure and organic amendment applications			
9.1 Apply moderate rates of manure and compost, and time the use of materials with high available N content to meet the peak N demands		x	YQ, OC, RL, ED, IT
9.2 Incorporate solid manure immediately to decrease NH_3 volatilization loss	x		OC, ED

[15] If soil or tissue tests or use of inhibitors indicate an opportunity to delay or reduce fertilizer applications, the grower may still have concern for yield or quality impacts should they not retain an over-application buffer to mediate this.

[16] Land tenure may present a challenge due to renting growers being hesitant to report contaminated water sources to landlords.

Adminsitrative Record
Page 34789

YQ (risk to yield or quality, poor market) CC (capital cost--infrastructure, equipment) OC (operational cost--labor, management, training) RL (ranch logistics) LT (land tenure) ED (education, training, demonstration needs) IT (insufficiently developed technology or more research needed)	**Current Extent of Use**		**Barriers to increased adoption**
	Common practice in some crops	Less common practice	
9.3 When applying liquid manure in surface gravity irrigation system, use delayed injection procedure to improve application uniformity[17]		x[17]	OC, RL, ED, IT
9.4 Use quick test methods to monitor dairy lagoon water N content immediately before and during application, and adjust application rate accordingly	x	x (optimal frequency)	OC, ED, IT[18]
9.5 Develop N budget that includes crop N harvest removal, supply of N from manure and other inputs	x (in dairies as regulated)	x	OC, ED, IT
9.6 Calibrate solid manure and compost spreaders	x		OC, RL, ED
10. Avoid fertilizer material and manure spills during transport, storage and application			
10.1 Do not overfill trailers or tanks. Cap or cover loads	x		ED
10.2 When transferring fertilizer, take care not to allow materials to accumulate on the soil	x		ED
10.3 Maintain all fertilizer storage facilities and protect them from the weather	x		CC, OC, ED
10.4 Clean up fertilizer spills promptly	x		ED
10.5 Shut off fertilizer applicators while turning at the end of the field and use check valves	x		OC, ED
10.6 Maintain proper calibration of fertilizer application equipment	x		ED
10.7 Wellhead protection	x		ED
10.8 Distribute rinse water from fertilizer application equipment evenly throughout field	x		OC, RL, ED
10.9 Avoid manure spills/discharges during transport, storage and application	x		ED

[17] Note dairy lagoon water is very rarely applied to crops other than forages.
[18] Although such tests are already commercially available, development of a quick test without a hazardous byproduct and/or a continuous monitoring system would significantly improve adoption.

Adminsitrative Record
Page 34790

2.4.1 Commonly utilized practices

California's high-value agricultural framework already demands from its practitioners a certain amount of "future-oriented and innovation-seeking attitudes" in order to survive economically (Brodt et al. 2006). This is reflected in the number of practices listed herein that have already been widely and voluntarily adopted by the farmers of our study areas. UC Cooperative Extension, the USDA NRCS, and other research, education, and outreach organizations have been instrumental in providing the technical advice and the proof that farmers demand when changes in management are under consideration (Pence & Grieshop 2001; Goodhue, Klonsky, & Mohapatra 2010).

The rate and period of time over which management practices have changed is largely unknown due to sparse survey data. Additionally, field and crop variables alter effects of various management schemes on leaching (Tilman et al. 2002; Fixen 2011). These facts, coupled with the usually long travel time of nitrate to groundwater, mean that it is impossible to say with certainty to what degree future groundwater nitrate concentrations will respond to the positive management changes already instituted and their increasing use.

Based on input from the five expert panels convened during April and May 2011, we conclude that the following practices that contribute to N use efficiency have been adopted by a significant portion of the growers in the two project study areas and represent a positive change from past practices (as summarized in Table 1).

MM 2: Improve Irrigation Scheduling, RP's 2.1, 2.2, 2.3, 2.4

The use of weather based irrigation scheduling tools (i.e., CIMIS), in conjunction with monitoring of soil moisture and/or plant water indices, has the potential to reduce excessive water applications, keeping nutrients within the rootzone longer. Tree crop producers using micro or drip irrigation, as well as many vegetable producers, commonly utilize ET data (usually historical) to guide their scheduling and more often monitor soil moisture or plant water status as well. Calendar scheduling still dominates field crop and forage production. Barriers to the use of these scheduling tools for all crops are described in section 2.3.1 and 2.4.2.

MM 3: Surface Irrigation Improvements, RP's 3.5 and 3.6

- Optimizing field grades by use of laser leveling technology is in widespread use.
- Converting to mechanized sprinkler or drip irrigation systems is employed by growers when economically feasible and crop appropriate. High value crops, such as most vegetables and many orchards and vineyards have already largely been converted to more efficient drip or microsprinkler systems.

Adminsitrative Record
Page 34791

MM 4: Sprinkler Irrigation Improvements, RP's 4.2 and 4.3

- Attention to irrigation system repair needs is generally adequate, with leaks and malfunctioning nozzles repaired or replaced as necessary. More regular monitoring and attention to gradual declines in performance would be beneficial however, particularly with better attention to manufacturer recommended nozzle and sprinkler component "life spans."

- Growers avoid operating sprinklers during periods of high wind as much as possible, although scheduling concerns can make this difficult.

MM 5: Drip and Microsprinkler System Improvements, RP's 5.1 and 5.2

- Engineering of drip and microspinkler systems has continually improved, as has the understanding of their optimal management. Appropriate lateral hose lengths are the norm in newly installed systems and the higher upkeep and replacement needs of these systems (as opposed to gravity systems, for example) allows for design improvement opportunities that are generally attended to by producers.

- Growers have greatly improved their understanding of the maintenance requirements of drip and microsprinkler systems, generally attending well to clogging from chemical and mechanical contaminants through both prevention and treatment. Commonly, new users of these systems experience a learning curve, and management may not be optimal during this phase and may remain as such without adequate education.

MM 6: Make Other Irrigation Infrastructure Improvements, RP 6.2

- Concrete backstops for wellhead are mandated by law on newly constructed wells and thus well attended to by growers. Exceptions include older uninspected wells, although growers are more often than not conscientious of wellhead protection measures such as not locating fertilizer tanks too close to wellheads.

MM 8: Rate, Timing, and Placement of Nitrogen Fertilizers, RP's 8.1, 8.2, 8.3, 8.6

- Soil testing to help guide fertilizer application rates is a common practice in some row crop production and nearly all large scale vegetable crop production. However, increased use of quick testing methods or more frequent laboratory testing has the potential to further refine rate decisions throughout the season.

- Plant tissue analysis to guide fertilizer timing decisions is also a relatively common practice in some operations but may be unhelpful in some crops where result turnaround time and rapid changes in growth make it difficult to reflect realtime changes in the field.

- Splitting fertilizer applications over the course of the growing season to better match plant uptake curves is relatively common and represents a large contribution to overall nitrate

Adminsitrative Record
Page 34792

leaching reduction and improvement over past practices. In non-fertigated crops, although split applications are common, further division of N applications is likely warranted to optimize timing with plant needs.

MM 9: Rate, Timing, and Placement of Animal Manure, RP's 9.2, 9.5

- Immediate incorporation of solid manure is a common practice that reduces gaseous N losses and thus increases N use efficiency.
- Estimation of annual harvest removal for N budgeting purposes is a newly mandated practice for California dairy operations in the Central Valley. However, these new limitations to on-farm recycling of manure have resulted in increased transfer of manure off-dairy. The consequence is an increased application of organic N to unregulated fields by producers that may not be properly accounting for the N content. The value of requiring the N budgeting methodology for crops that do not regularly receive dairy manure, as a means of reducing nitrate leaching, is uncertain at this time.

MM 10: Material Storage and Handling

- All recommended material storage and handling practices are generally assumed to be followed by the bulk of producers although continued vigilance in the form of regular educational reviews is important. The new dairy wastewater discharge regulations are applicable to management and construction of lagoons and corrals along with manure land application protocols.

2.4.2 Less Commonly Utilized Practices

Significant barriers limit the ability of producers to adopt some of the practices known or theorized to lessen leaching of nitrate to groundwater from agricultural land. The feasibility of overcoming some of these barriers varies, with some practices facing significant impediments to increased adoption and others with fewer obstructions to overcome their underutilization. As outlined in Table 1, these include economic barriers (operation and capital costs or yield risks) of varying degrees, educational requirements, farm operation logistical conflicts, land tenure conflicts, and need for additional research.

For example, growers who rent their land may be reluctant to invest in equipment that will remain with the land owner (land tenure conflict). Water district delivery schedules or pumping capacity may logistically constrain the farmer's ability to more precisely time irrigations to match crop need for water (logistical conflict). Additionally, some practices have good theoretical potential to reduce nitrate leaching from irrigated cropland, but require more research, development, and/or verification of cost effectiveness before becoming widespread commercially viable options (newer controlled release fertilizers, for example).

Adminsitrative Record
Page 34793

In many cases, increased adoption of improved practices is enhanced by access to effective educational programs. Panel members discussed the educational barriers associated with a number of management practices regarding water and nitrogen budgeting. Outreach and knowledge transfer efforts are crucial to increasing adoption of improved management practices. Delivery of this information will largely depend upon certified crop advisors, UCCE agents and NRCS personnel. According to some panel member commentary, attention to delivery methods needs to be considered when meeting the training and continued educational needs of growers. They emphasized that text heavy pamphlets and web pages are often ineffective and that field days, how-to videos, and picture-laden pamphlets are, in their experience, a more effective and efficient means of reaching the audience and increasing voluntary adoption of improved practices. Additionally, as many management tasks are carried out by farm laborers who communicate primarily in Spanish, attention to language is important when developing knowledge transfer activities and materials.

As summarized in Table 2, according to the literature and expert panel members, implementation of one, or a combination, of the following practices represents an opportunity to reduce agricultural nitrate leaching risk with increased adoption; however, overall feasibility will be determined by the noted barriers. Recall that not all practices are adaptable or appropriate to all crops, soils, or irrigation systems, and thus the extent of any increased use is also limited by farm-specific characteristics.

MM 1: Perform Irrigation System Evaluation and Monitoring, RP 1.1

- Irrigation system performance evaluations are the critical first step to addressing water use efficiency and uniformity of application. Although the mere act of evaluation will not mitigate nitrate leaching, acting upon the knowledge gained has the potential to reduce leaching from some fields. Past programs (i.e., DWR/NRCS Mobile Irrigation Laboratories) were successful in providing these evaluations free of charge, and with the decline of this service, the percentage of growers that keep up with regular performance evaluations has also declined, due primarily to cost and time constraints.

MM 2: Improve Irrigation Scheduling, RP's 2.1, 2.2, 2.3, 2.4

- The use of weather based irrigation scheduling tools (i.e., CIMIS), in conjunction with monitoring of soil moisture and/or plant water indices, has the potential to reduce excessive water applications, keeping nutrients within the rootzone longer. Vegetable and tree crop producers do commonly utilize ET data (mostly historical data) to guide their scheduling and often monitor soil moisture as well (especially in orchards), but calendar scheduling remains the most common method of scheduling in field and forage crops. Barriers for all crop groups include conflicting labor schedules, ranch logistical conflicts (e.g., timing of necessary pest control measures), water availability, field variability, and irrigation system constraints (generally more difficult in surface systems than drip or sprinkler systems). Better grower education and access to irrigation specialists can provide the training required to maintain and use these technologies and can help overcome one of the main barriers (besides logistical

Adminsitrative Record
Page 34794

issues) that keep better water budgeting (especially use of real-time ET data) from being more highly utilized.

- Growers are typically aware of the disadvantages of excessive pre-plant and fallow period irrigations and many try to apply the bulk of their N fertilizers after pre-irrigation and stand establishment. However, salt control concerns and the fear of poor crop germination and establishment (along with inexpensive water in some areas) tend to keep over-application of water during this vulnerable period relatively common. The reduced nutritional needs of crops during early establishment equates to high loss levels only if soil nitrate is significantly present during this phase. High water application will not present a problem when soil N levels are low, and, as such, strict N management can be equally critical to overall N use efficiency. Keeping in mind the importance of salinity control and successful plant establishment, the potentially highly mitigative practice of conservation of water during fallow and early growth periods deserves to be a continued focus of educational outreach programs.

MM 3: Improve Surface Gravity Design and Operation, RP's 3.1, 3.2, 3.3, 3.4

- Surge and cutback irrigation methods offer increased water use efficiency in surface irrigation systems, but the increased labor costs, ranch logistics and, in some cases, capital costs have led to the underutilization of these techniques. While automated valves can reduce the labor cost barrier, they would impose equipment costs. Note surge irrigation is only appropriate on some soils in furrow irrigated row crops.

- Reduced field length can improve irrigation efficiencies and reduce leaching risk, but the significant economic barriers (reduced yield through loss of land, and increased capital and operational costs) are difficult to overcome in most circumstances. Even so, the number of fields under a half mile has increased since times past. Most fruit orchards and the majority of Salinas Valley vegetable fields are already short in length.

- Optimizing flow uniformity via furrow compaction through the use of torpedoes, bolas, or other weighted objects is utilized by some furrow-irrigating growers, although not the majority. Research on cost-effectiveness may overcome economic barriers, while educational opportunities exist to increase adoption.

MM 4: Improve Sprinkler Design and Operation, RP's 4.1, 4.2 and 4.4

- Better monitoring of pressure and flow variations within sprinkler irrigation systems represents a feasible improvement option. The implementation challenges of labor costs and required expertise should be surmountable especially if programs, such as the DWR/NRCS Mobile Laboratories irrigation evaluation services offered in the past, were reinstated.

- Nearly all growers using sprinklers attempt to maintain the equipment. However, a more proactive maintenance schedule would encourage earlier detection of gradual declines. More

Adminsitrative Record
Page 34795

timely nozzle, head seal, and impact "spoon" replacement schedules could also be beneficial. Labor and capital costs are the main impediments to optimized sprinkler maintenance.

- Offsetting lateral moves generally improves irrigation uniformity and, although not utilized by a majority of growers that sprinkler irrigate, the practice is not rare. Additional labor costs present the main barrier, along with logistical issues, such as limitations to adjusting irrigation scheduling. Additionally, more research is warranted in comparing the mitigation potential between offsetting laterals or simply spacing the laterals closer together, and which may be the most cost-effective option.

MM 5: Improve Drip and Microsprinkler Design and Operation

- Although drip systems are capable of high distribution uniformity and offer theoretically more control over matching water application rates to specific plant needs, without adequate attention to system maintenance and scheduling, the overall efficiency can drop below that of a well-managed surface system. Distribution uniformity and scheduling practices are quite varied, and increased vigilance would be beneficial. Labor and capital costs, along with a need for adequate training, present barriers to drip systems meeting optimum efficiencies.

MM 6: Make Other Irrigation Infrastructure Improvements, RP 6.1

- Closely spaced subsurface drains can divert a significant amount of any leached nitrate from entering the groundwater in areas of perched water tables, thus representing a groundwater contamination reduction strategy where they are installed. However, the drainage water must be treated, and current methods for treatment are not yet fully developed. Should treatment technology improve, much more research will be required to determine whether installation of deeper drains in areas with deeper water tables might be an effective option for mitigating groundwater pollution in these areas.

MM 7: Cover Crops and Crop Rotations, RP's 7.1, 7.2, 7.3, 7.4

- Covercropping has been shown to significantly reduce leaching during fallow periods when excess N may remain in the upper soil horizons. Increased grower use is impeded by intertwining and sometimes unpredictable aspects, depending on both the cover crop and cash crop species. More educational outreach and research (crop and site specific), along with economic incentives would allow for some increased use of cover crops. Although logistical issues (water availability, planting schedule disruption, disease introduction) will continue to present barriers in some sectors, the evidence suggests this practice has high mitigative potential and some increased adoption is likely feasible, but the added cost of covercropping (especially the cost of the additional water needed to produce it) with little economic return still needs to be addressed to see more widespread adoption.

Adminsitrative Record
Page 34796

- The incorporation of deep rooted scavenger species or perennial species into crop rotation schemes faces barriers similar to covercropping, including capital and operating costs and planting schedule disruptions, but does provide some opportunity for income, unlike traditional cover crop species. Alfalfa, an excellent scavenger, is already grown extensively for the numerous dairies in the TLB, although its rotation diversity is somewhat limited (e.g., not used in vegetable crop rotations, etc.).

MM 8: Rate, Timing, and Placement of Nitrogen Fertilizers, RP's 8.4, 8.6, 8.7, 8.8, 8.9

- Delayed injection of fertilizers into surface irrigation systems can improve fertilizer application uniformity, although economic incentive to overcome the additional labor costs is currently lacking. Labor costs would be traded for capital costs, should there be development of automation devices for this practice.

- Automated precision agriculture technology has been shown to enhance N use efficiency, particularly in the rainfed mid-West. The economic benefit of these technologies in the diverse irrigated fields of California has been mixed. Further research and development are required before these technologies can significantly and consistently benefit the high crop diversity found in California.

- Nutrient budgeting that accounts for harvest removal, as well as nitrogen already present in irrigation water and soil organic matter pools, is a complex tool that, in some instances, can be used by farmers to identify opportunities to reduce N fertilizer or manure application rates and timing. Increased adoption is dependent upon more crop specific research and associated education and outreach, as well as addressing the economic burdens associated with the operating costs of increased recordkeeping and analysis.

- Research shows potential for increased N use efficiency using some of the recently commercialized nitrification and urease inhibitors. Additional research under more variable and specific field conditions is needed to show product cost-effectiveness and the ability to reduce crop nitrogen fertilizer requirements.

MM 9: Rate, Timing, and Placement of Animal Manures, RP's 9.1, 9.3, 9.5

- Reducing plant-available nitrogen release from organic matter at times of low crop need can be accomplished by only applying moderate amounts of solid manures and composts and supplementing with inorganic nitrogen fertilizers to meet crop need. Estimating release rates from organic N sources still represents a problem in calculating optimal application rates of both organic and inorganic fertilizers in these systems. The increasing levels of dairy manure that is transferred off-farm to unregulated fields (often to growers that do not properly account for organic N in their fertilization regimes) demands additional research, monitoring, and education efforts.

Adminsitrative Record
Page 34797

- Delaying injection of liquid manure into surface irrigation water can enhance uniformity of nutrient application, but logistical constraints and the current lack of automated methods for doing this burdens producers with labor costs, making the practice difficult to consistently implement.

- Monitoring dairy lagoon water N content via quick test methods is not highly utilized at present although the new dairy waste discharge regulations now require quarterly sampling and analysis. By testing just prior to every application, a better estimate of total N application is expected. Although the practice is meant to address the nutrient level fluctuation and varying dilution ratios, the variability of concentration values, inadequately educated farm personnel, and cost concerns stand as major barriers, along with additional research and development of a quick test that is not associated with a hazardous waste byproduct as is currently the case.

In summary, it is certain that with the increased adoption of improved management practices, the mass of nitrate leached from the crop rootzone can be reduced to well below the rate of loss that has resulted in the currently observed nitrate concentrations in affected aquifers. Furthermore, the rate is expected to have already decreased to some degree due to increasing use of improved management practices by growers in the study areas. The applicability and effectiveness of any one management practice, or group thereof, varies according to field specific variables (crop and soil characteristics), and the current level of NUE achieved. Overcoming significant barriers remains a hurdle, although moderate increases in use of improved practices should be feasible. Logistical, educational, and financial constraints are the most frequently cited impediments to increased adoption of better management practices.

Adminsitrative Record
Page 34798

2.5 Crop Management Practice Scenarios

In this section, hypothetical scenarios of combinations of practices are presented for example crops of four main crop groups: vegetables; tree crops and grapes; field crops, including grain and hay; and forages receiving dairy manure. The purpose of presenting these scenarios is not to prescribe particular management measures for these crops, but rather to illustrate the complexity in decision making for growers and the barriers that will need to be addressed.

Maximizing crop nitrogen use efficiency (NUE), and thereby minimizing nitrate leaching losses, often requires a combination or "bundle" of management measures, rather than just a single practice. For example, improvements in both irrigation and fertilizer application timing may be needed, and making only one of these improvements would likely have little impact.

Because of the wide variation in soil properties, crop characteristics, water supply availability, weather, and production economics, the particular combination of practices for achieving efficient N use will be farm- and even field-specific.

However, certain management measures and recommended practices are applicable to all farming operations, regardless of crop category, including:

- Irrigation system evaluation and monitoring
- Improving irrigation scheduling via use of weather-, plant-, or soil-based scheduling systems
- Adjusting N application rates and timing based on soil or tissue testing and irrigation water N content
- Recordkeeping for evaluation of success with any altered management scheme
- Wellhead protection, backflow protection, and proper storage and handling of fertilizer products

Other recommended practices are appropriate only for certain crop species, irrigation systems, soils, or climatic conditions. Selection of management practice bundles requires site-specific considerations, as field and crop variables alter effects of various management schemes (Tilman et al. 2002; Fixen 2011). Thus, many of the practices described in sections 2.3 may be very good choices for a particular farm, but may not be applicable to the entire, diverse crop group.

Crop-specific examples of improved management options that could be used in tandem to improve nitrogen use efficiency are presented for each crop group. Crops used in these examples occupy a large acreage within the study area and have been the focus of N management research and, in some cases, nitrate leaching.

The objective for all management changes is essentially the same for all scenarios: increase N and water use efficiency, reduce deep percolation of nitrate-laden water, and maintain crop yields and quality. Evaluation of the outcomes of adoption of new practices is a crucial component of any change in management and should be considered an implied management measure for all example scenarios.

Adminsitrative Record
Page 34799

Records of crop yield and quality, total water use, total N use, costs, and other notable outcomes must be evaluated and analyzed for effectiveness and cost-benefit ratio.

Again, because each unique farming situation demands individualized management plans, the scenarios presented here are provided to give a more realistic sense of the constraints or barriers a producer may face, and are not meant as required formulas for reduction of N transfer to groundwater for these crops. For example, covercropping in orchards may be feasible in some cases, while in other orchard crops harvest machinery (sweepers) requires that the orchard floor be smooth and firm, making covercropping an impossibility. Irrigation system and soil characteristics of individual fields will also influence strongly the impact of practices on crop N use efficiency and Nitrate leaching losses.

2.5.1 Vegetable Crops

Vegetable crops grown in the study areas are both highly diverse and profitable with a well-developed infrastructure surrounding their production and marketing. These crops, because of their (usually) shallow root systems are inherently less efficient in use of water and nitrogen; and some of the vegetable crops, for example leafy greens, require an ample supply of N up to the point of harvest. However, opportunities to increase these efficiencies via good management do exist, and many practices known to contribute to reductions in Nitrate leaching have already been adopted by a large portion of producers (as outlined in section 3). For example, the generally profitable nature of vegetables has led in recent years to widespread conversion from furrow and overhead sprinklers to drip irrigation systems, which can offer more precisely controlled water delivery under good management and maintenance. Besides the practices listed above that are applicable to all crop categories, practices generally recommended specifically to vegetable crops include weather-based irrigation scheduling; restricting heavy fallow period and pre-plant irrigation as much as possible; covercropping; soil N quick-tests and split fertilizer applications.

Following are examples of practice bundles that might be adopted for lettuce production, a very important crop in the Salinas Valley (Table 3); and processing tomatoes, a highly important TLB crop (Table 4).

Adminsitrative Record
Page 34800

Table 3. Example management bundle: Lettuce.[19]

Region: Salinas Valley **Crop: Lettuce** **Irrigation: Sprinkler to establish, then drip**	
Improved Practices	**Costs & other barriers**
Irrigation system evaluation and monitoring	
Conduct irrigation system performance evaluation and identify opportunities to improve uniformity and reduce water application during sprinkler phase	• Consultant services to measure sprinkler water distribution and application efficiency
Install and use flow meters to track water volume applied to each field at each irrigation	• Flow meters and installation • Software to manage large amount of data • Management time to review information
Irrigation scheduling	
Use weather-based irrigation scheduling and focus on reducing sprinkler irrigations used during germination and establishment	• Customized software • Training of managers • Irrigator work schedules may be affected.
Improve rate, timing, placement of N fertilizers	
Adjust N fertilizer rate based on more intensive soil nitrate testing	• Cost of collection and analysis of soil samples • Management time to interpret results
Measure N content of irrigation water and adjust fertilizer rates accordingly	• Cost of analysis. • Recordkeeping • Management time to interpret results.

[19] (Smith & Cahn 2011) Lettuce trial: CIMIS guided germination irrigation reduced water application by ~2.5" and led to reduced leaching per lysimeters. Use of presidedress soil tests allowed for average reduction of 61.6 kg N/ha (55 lb N/acre) with no change in yield. Low residue covercropping considered possible tool as well, but only in soils of low to moderate soil nitrate levels. Breschini & Hartz (2002) Lettuce trial: PSNT-guided fertilization plots averaged 8mg/kg lower residual soil NO_3-N than controls. Jackson et al. (1994) Lettuce field trial (EPIC model): Highest leaching coincided with sprinkler irrigation to germinate the second crop. Model without pre-plant irrigation yielded ~40% decrease in leaching. Jackson et al. (1993) Lettuce trial: Non-leguminous covercropping reduced winter leaching without interrupting planting schedule.

Adminsitrative Record
Page 34801

Table 4. Example management bundle: Processing tomatoes.[20]

Region: San Joaquin Valley **Crop: Processing tomato from transplants** **Irrigation: Sprinkler to establish, then drip**	
Improved Practices	**Costs & Other Barriers**
Irrigation system evaluation and monitoring	
Conduct irrigation system performance evaluation and identify opportunities to improve uniformity and reduce water application during sprinkler phase	• Consultant services to measure sprinkler water distribution and furrow distribution uniformity and application efficiency
Install and use flow meters or other measuring devices to track water volume applied to each field at each irrigation	• Flow meters, installation, software. • Management time to review information.
Irrigation scheduling	
Use weather-based irrigation scheduling	• Improved software. • Training of managers. • Irrigator work schedules may be affected
Improve rate, timing, placement of N fertilizers	
If very high N fertilizer rates are being used to provide for the increased yields under drip irrigation, experiment with lower N rates, e.g., similar to or lower than rates used for furrow-irrigated tomatoes.	• Potential risk to yield • Difficult for farmers to conduct rate comparisons on commercial fields
Adjust N fertilizer rates based on soil nitrate testing during season	• Cost of collection and analysis of soil samples • Some management time to interpret results
Use soil nitrate testing at or immediately after harvest and consider results in determining the next year's N fertilizer program	• End-of-season soil analyses can be difficult to interpret
Measure N content of irrigation water and adjust fertilizer rates accordingly	• Cost of analysis • Management time to interpret results

[20] Hartz & Bottoms (2009) Processing tomato trial: Switch to drip allowed for a seasonal N application reduction sufficient to maintain the increased yield that comes with drip irrigation for this crop. (Hartz, LeStrange, & May 1994) CIMIS ET data for drip irrigation scheduling reduces N fertigation requirements.

Adminsitrative Record
Page 34802

2.5.2 Tree Fruits, Nuts and Vines

Tree fruits and nuts represent a large industry of their own in the southern San Joaquin Valley TLB study area, representing over 35 % of the irrigated agricultural land of the basin at the time of the most recent DWR reports. Vineyards, including table, raisin and wine grape production are also an extremely important in both valleys, representing over 18% of the total TLB agricultural land and 20 % of the Salinas Valley agricultural production. According to the most recent DWR reports, permanent crops in the TLB amount to over 566,000 hectares (over 1,400,000 acres) or 58 % of the irrigated agricultural land. Vines, deciduous fruit, subtropical fruit and nut trees are all managed differently from each other, with contrasting N and irrigation needs. Tissue testing is generally advisable over substantial soil testing regimes in these crops.

Table 5. Example management bundle: Almonds.[21]

Region: San Joaquin Valley **Crop: Almonds** **Irrigation: Microsprinkler**	
Improved Practices	**Costs & Other Barriers**
Irrigation scheduling	
Use weather-based irrigation scheduling	• Custom software • Training of managers and irrigators • Irrigator work schedules may be affected
Improve rate, timing, placement of N fertilizers	
Collect leaf samples for nutrient analysis on routine basis, not just for trouble shooting	• Cost of collection and analysis of soil samples. • Management time to interpret results.
Consider published UC leaf sample N critical values in making N fertilizer use decisions	• Recordkeeping • Consultant costs • Lack of confidence in tissue sampling and recommended critical values; lack of UC multi-year research
Time N fertilizer applications according to UC recommendations	• Management time • Adjustment to labor schedule
Measure N content of irrigation water and adjust fertilizer rates accordingly	• Cost of analysis. Management time to interpret results.

[21] Brown (unpublished) Almond trial: Split applications and microirrigation.

Adminsitrative Record
Page 34803

Table 6. Example management bundle: Peaches.[22]

Region: San Joaquin Valley **Crop: Peaches** **Irrigation: Furrow or basin**	
Improved Practices	**Costs & Other Barriers**
Irrigation scheduling	
Check soil moisture to aid decision on timing and amount of water applied	• Water supply delivery schedules may limit grower flexibility in irrigation timing
Use weather-based irrigation scheduling	• Custom software • Training of managers and irrigators • Irrigator work schedules may be affected • Water supply delivery schedules may limit grower flexibility in irrigation timing
Improve rate, timing, placement of N fertilizers	
Collect leaf samples for nutrient analysis on routine basis	• Cost of collection and analysis of soil samples • Management time to interpret results
Consider published UC leaf sample N critical values in making N fertilizer use decisions	• Recordkeeping • Consultant costs • Lack of confidence in tissue sampling and recommended critical values; lack of UC multi-year research
Split N applications during season, and time applications according to UC recommendations	• Management time • Adjustment to labor schedule
Measure N content of irrigation water and adjust fertilizer rates accordingly	• Cost of analysis • Management time to interpret results

[22] Malusa, Buffa, & Ciesielska (2001) Peach trial: Split application of fertilizer retains yield. (Johnson et al. 2001) Peach trial: Applying 50% N requirements as foliar retains yield. Niederholzer et al. (2001) Peach trial: Spring N applications sufficient to maintain growth and yields, no need for fall N application.

Adminsitrative Record
Page 34804

2.5.3 Field Crops, Grain, and Hay

Field crop, grain, and hay production amounts to over 55% of the irrigated agricultural land in the TLB, and cotton alone accounted for nearly half of this production per most recent DWR surveys. In the Salinas Valley, grain and hay production accounted for about 10% of the production in 1997. Most of these crops are surface irrigated, with a smaller percentage being sprinkler irrigated. In general, drip is either totally non-applicable or economically unfeasible. Optimizing surface irrigation efficiency is thus of high importance in this group, which is more adaptable to using plant-based irrigation scheduling and incorporating perennials into crop rotation schemes than vegetables.

Table 7. Example management bundle: Cotton.[23]

Region: San Joaquin Valley **Crop: Cotton** **Irrigation: Furrow**	
Improved Practices	**Costs & Other Barriers**
Irrigation improvements to reduce leaching	
Grade fields so that slope is decreased in bottom ~one quarter of field	• Potential cost of additional earth moving during field grading
Use weather-based irrigation scheduling	• Custom software • Training of managers and irrigators • Irrigator work schedules may be affected • District water delivery schedules may limit grower flexibility in scheduling
Modify crop rotation to capture more N	
Include alfalfa in rotation and reduce N fertilizer application on following crop	• Water availability • Alfalfa hay price is sometimes unattractive compared to alternative crops in rotation • Lack of definitive UC “alfalfa N credit” methods and values
Improve rate, timing, placement of N fertilizers	
Adjust N fertilizer rates based on soil nitrate testing	• Cost of soil sample collection and analysis • Management time to interpret results
Measure N content of irrigation water and adjust fertilizer rates accordingly	• Cost of analysis and management time to interpret results • Data collection and interpretation can be complex (e.g., when irrigating fields from multiple sources (wells and surface waters)

[23] Hutmacher et al. (2005) Cotton trial: Accounting of N in irrigation water, soil testing, and tissue testing necessary to overcome yield reduction concerns and economic return (fertilizer cost savings versus increased management costs).

Adminsitrative Record
Page 34805

2.5.4 Forage and Silage Crops Receiving Manure

These typically surface-irrigated crops are most often grown as livestock feed; consequently, they receive a substantial portion of their nutritional needs from manure, and have less economic return than other crop groups of the study areas. New regulations, with an aim to decrease nitrate leaching concerns from dairy farms, include N testing and budgeting.

Table 8. Example management bundle: Silage Corn.[24]

Region: San Joaquin Valley **Crop: Silage corn** **Irrigation: Furrow with dairy lagoon water applied in some irrigations**	
Improved Practices	**Costs & Other Barriers**
Irrigation scheduling	
Use weather-based irrigation scheduling	• Custom software • Training of managers and irrigators • Irrigator work schedules may be affected
Improve rate, timing, placement of N fertilizers	
Collect corn stalk samples for nitrate analysis at harvest and evaluate manure/lagoon water/fertilizer program efficiency; make improvements in following year	• Cost of collection and analysis of plant • Lack of California research to show effectiveness of this practice
Measure N content of fresh irrigation water and adjust fertilizer and dairy lagoon water rates accordingly	• Cost of analysis • Management time to interpret results
Improve rate and timing of dairy lagoon water and manure application	
Measure NH_4 content of lagoon water frequently during applications and adjust application rate accordingly	• Additional cost for collection and analysis beyond quarterly sampling required by dairy waste discharge regulation • Dairy lagoon water infrastructure (pumps and piping) may not allow for fine-tuning of application rates or for very low flow rates
Adjust N fertilizer rates based on soil nitrate testing	• Cost of collection and analysis of soil samples • Management time to interpret results • Excessive spatial variability of soil nitrate may deter usefulness • Not effective in highly permeable soils

[24] Gehl et al. (2005) Silage trial: Weather-based irrigation scheduling and split N applications reduced leaching.

Adminsitrative Record
Page 34806

2.6 Vulnerability Assessment

Our approach in this report is to identify and describe practices and technologies that are potentially available to growers for achieving high crop N use efficiencies. We have also indicated which practices appear to be currently in common use by farmers and the barriers (such as high costs or perceived risk to yields) to increased adoption. In this section we provide an analysis of the results of the nitrate groundwater pollution hazard index, assessing the portion of cropland in each of the two study regions where the risk of Nitrate leaching loss is highest, and therefore where adoption of more N-efficient practices may have the greatest effect.

To make this assessment, we have used the Nitrate Hazard Index (HI) (Wu et al. 2005) as described in the methods section above (2.2.3). The HI assigns an index value from 1 to 80 (very low to very high risk of nitrate leaching loss) based on the soil series, crop species grown, and the type of irrigation system in use in any one field. An HI under 20 is considered to be of low risk to leaching and any value over 20 is considered potentially highly vulnerable to nitrate leaching. The HI does not take into account farmer use of specific management practices, depth to groundwater, or the underlying hydrogeology. Rather than an indicator of actual nitrate loss or pollution to groundwater, the HI is an indicator of how much focused effort a farmer needs to place on efficient N management, where attention to improved management may have the most effect on groundwater quality, and, by identifying which factors contribute most to risk, the HI can be used to help guide choice of specific management practices.

It is important to emphasize again that the HI values do not reflect the management decisions that contribute highly to overall leaching risk, as discussed in the bulk of this report. A slowly draining field cropped to an inherently lower risk species for example, may in fact pose a significant hazard to groundwater quality if poorly managed in regards to water and nitrogen use efficiency. The specific utility of the hazard indexing tool to the grower is rather then to help identify which inherent component or components (crop, soil, or irrigation system) are driving risk, and as such providing information useful for guiding the choices of which alternative management options may be the most appropriate for the given situation. The analysis provided here serves as a showcase of the variability that exists between single fields within the study area and the associated variability in appropriate farm management options. In areas identified as being of high risk, the actual loading will be highly influenced by how the grower manages their crop N use efficiency. In this way, the analysis provides an estimation of where adoption of mitigative management practices would likely have the most effect on nitrate leaching and the associated groundwater quality at the basin scale.

Individual component HI values for soil, crop, and irrigation system were assigned to each irrigated agricultural field in the study areas using the online HI tool database, and based on the most recent crop, irrigation, and soil surveys available to us. We present an analysis of each individual component and follow with the results of the overall vulnerability, computed by multiplying the component values together. A more detailed discussion of the methods, caveats, and assumptions involved in this analysis can be found in in the Methods Section 2.2.3.

Adminsitrative Record
Page 34807

Our analysis indicates that 32% of the irrigated cropland in the Tulare Lake Basin and 52% of the irrigated cropland in the Salinas Valley is especially vulnerable to low crop N use efficiency based on the combinations of crop, soil, and irrigation system characteristics assigned. That is, these percentages represent the area with HI values greater than 20, the critical point over which potential for risk is high as suggested by the authors of the HI (Wu et al. 2005). Note that if any one of the three components – soil, crop, or irrigation system – has a value of 1 (the lowest risk category), the overall HI will be less than 20, regardless of the values for the other two factors. For example, an intrinsically higher risk irrigation system utilized on a highly permeable soil will still not compute as a particularly vulnerable situation (that is, an HI>20) if the crop grown is classified as very low risk, such as alfalfa.

2.6.1 Crops

The majority of the Salinas Valley irrigated land is cropped to relatively high risk cool season vegetables. The high risk is a reflection of shallow rooting, high market sensitivity to N deficiency, and harvest of the plants at the peak of growth and N requirements. Based on the most recent DWR crop survey (1997), 70% of the crops in this basin are vegetables rating either HI 3 or HI 4; low risk vineyards (HI 1) cover another 20% of the area and the remaining ~10% of the area is in orchards and field crops with an HI of 2. We expect that, for the most part, current crop acreages are similar, with the exception of strawberries which have increased in production in the past decade.

In contrast, in the Tulare Lake Basin, over 50% of the irrigated cropland was devoted to lower risk permanent crops (vineyards and orchards) and alfalfa (at the time of the DWR surveys: Fresno 2000, Kern 2006, Kings 2003, and Tulare 1999). An additional 35% of the area is devoted to production of HI 2 rated field, grain, and hay crops (excluding corn, an HI 3 crop), with just under 15% of the irrigated agricultural land planted to the higher risk crops of vegetables and corn.

The spatial distribution of these crop groups according to their HI value is shown in Figure 6 below.

Adminsitrative Record
Page 34808

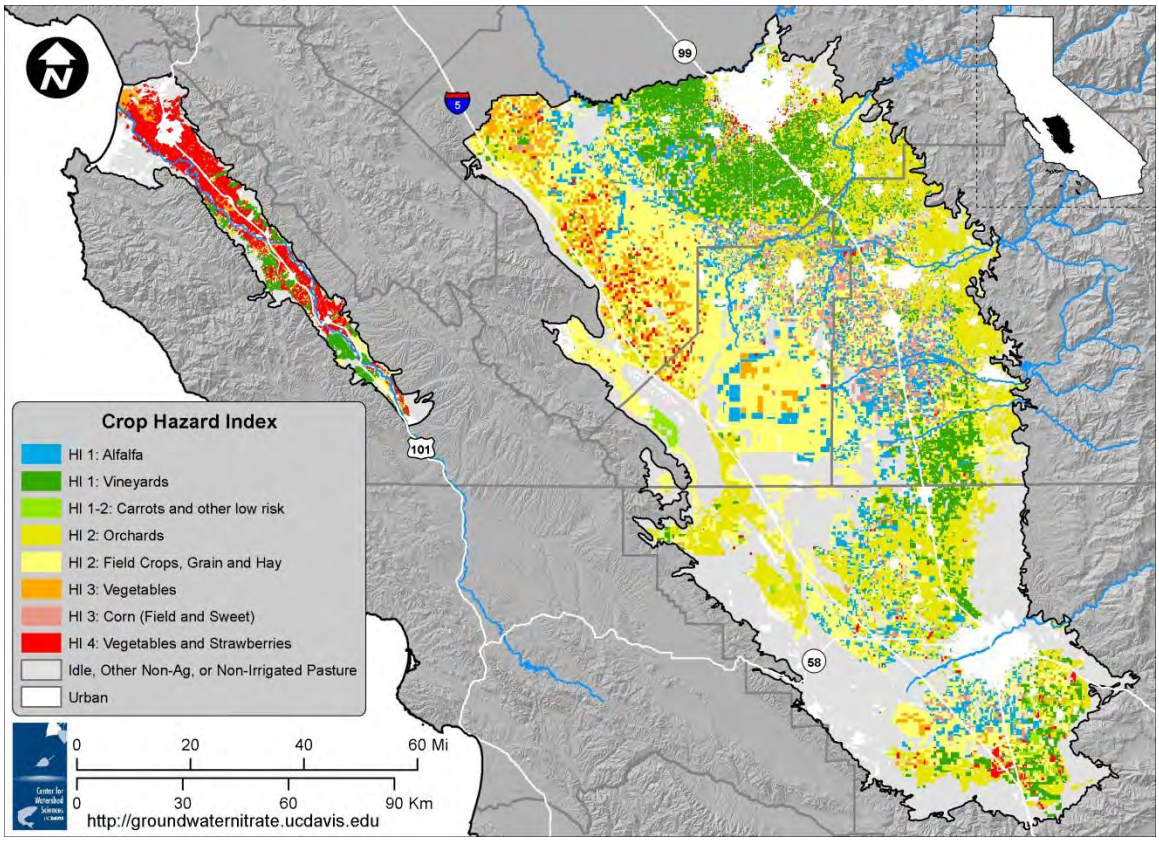

Figure 6. Study area crop distribution categorized by the crop group's hazard index values, based on the most recent DWR reports for each county (see text). Higher values denote crops with higher nitrate leaching risk.[25] (Source: DWR, UCANR WRC, internal designations, and Wu et al. 2005.)

2.6.2 Soils

The majority of both the TLB and SV soils are rated HI 3 or above, 62% of the TLB irrigated agricultural area and 72% of the SV area. The SV has only minimal areas rated 2 or below, primarily around Gonzales and to the east and southeast of King City. Higher risk soils predominate and are rated 4 and 5 all along the river, widening south of the confluence of the Arroyo Seco and Salinas Rivers to Greenfield and continuing south to the end of the valley. The TLB has larger areas of reduced permeability (HI 1-2) surrounding and north of the old Tulare Lake bed and along the eastern edge of Tulare County, as well as areas of Kern County. Higher risk soils surround Fresno, Hanford, and Visalia and extend south in a loose belt roughly following Highway 99 into McFarland, spreading south through the fields surrounding

[25] Note that the category "HI 1 or 2, Carrots and other low risk" is nearly 70% carrots (HI 2), with the bulk of the remaining proportion in sugarbeets (HI 1): sugarbeets are no longer produced in the basin. The category "HI 2 Orchards" includes 5 tree varieties not classified as HI 2: Olives, Apricots and Figs are rated HI 1 and Grapefruit and Kiwis are rated HI 3: all of these crops are of very low area comparatively and thus were included in the orchard category as a simplification strategy despite the different HI values. For the same reason, the "HI 2 Field Crop" category includes both dry beans and sudangrass, both rated HI 1, but where each represents a very small portion of the total category area. The category "HI 4 Vegetables" includes both onions and garlic, although garlic is rated as an HI 3 vegetable. The DWR surveys on which this map is based groups both crops together and does not differentiate the fields, thus the need to categorize garlic with the HI 4 vegetables.

Adminsitrative Record
Page 34809

Wasco and Shafter, and finally beyond Bakersfield to the southeast (Figure 7). The highest risk soils (HI 5) surrounding Fresno are planted mainly to vineyards (a low HI crop), although much of the remaining HI 5 area and a large portion of HI 4 soils are cropped with higher risk species or are irrigated with higher risk systems, driving up the overall HI rating (see Section 2.6.4).

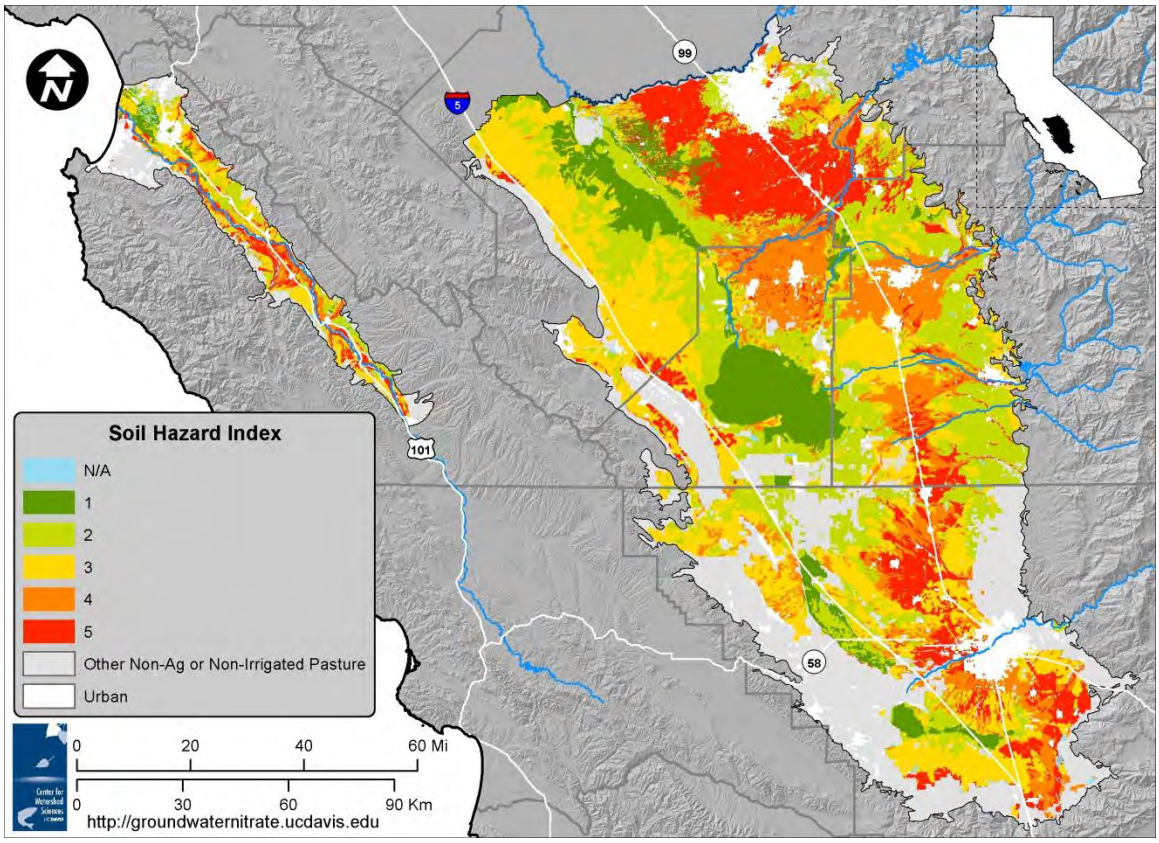

Figure 7. Soil hazard index value distribution for irrigated agricultural soil series. (Source: NRCS, SSURGO, UCANR WRC, internal designations, and Wu et al. 2005.)

2.6.3 Irrigation

As shown in Figure 8, at the time of the DWR surveys (Tulare 1999, Fresno 2000, Kings 2003, and Kern 2006), the majority of the TLB (over 68%) was recorded as being surface irrigated. According to the DWR reports for the study area, about 8% was sprinkler irrigated and nearly 25% of the area was already under drip or microsprinkler irrigation, especially in orchards and vineyards surrounding Fresno and those along the eastside of the basin (Figure 8). Permanent crops irrigated by these systems (60% in the DWR survey) will receive an overall HI of 20 or less regardless of the value for the soil component of the HI due to the HI value of 1 given the irrigation component.

The consensus of our tree and vine panel was that currently, much more than 60% of the permanent tree and vine crop area in the TLB is irrigated by drip or microsprinkler. Furthermore, according to members of our expert panel for TLB vegetables, nearly all of the processing tomato acres in the TLB are

now irrigated by drip, and very few acres are irrigated by furrow systems – a change from the 85% furrow irrigated tomatoes reported in the DWR survey. They also confirmed that a significant number of other vegetable crop growers in the TLB had likewise adopted drip irrigation, but at a lower rate than processing tomatoes. A simulation was performed to look at how the overall HI may change should drip and microsprinkler use increase among permanent and vegetable crop producers and is discussed in the next section (2.6.4).

It is safe to assume that unlike vegetables, nuts, and citrus, most field crops remain surface irrigated and that the relative area of these crops has remained similar, with only small areas converting to potentially more efficient automated sprinkler systems (center pivot, linear move, or LEPA). According to panel members, carrots, onions, and some garlic are often sprinkler irrigated throughout the cropping pattern. Thus, the expected reduction in leaching vulnerability due to changes in irrigation methodology presents itself more heavily in the movement towards higher proportions of drip irrigated vegetables, vineyards, and orchards (especially nuts and citrus).

For the Salinas Valley, there are no recent irrigation surveys that include spatial information. The 2010 Monterey County Groundwater Summary Report (MCWRA 2011) details present and past irrigation system usage for just over 71,000 of the nearly 95,000 hectares of irrigated agriculture in the Salinas basin, although not spatially. Of that area, nearly 55% was reported to be drip irrigated, and 20% each sprinkler and surface systems (expected for the 2011 season). This drip irrigated area includes the valley's vineyards, as well as vegetable and berry growers who reported only 46% of their crops to be on drip. Based on input from our Salinas Valley expert panel, we estimate that 50 to 70% of the total vegetable acreage is irrigated by drip. As we do not know which vegetable acres are irrigated by drip, to create the field-specific irrigation layer of the HI estimate, we made the simplifying assumption that all vegetables in the SV are drip irrigated (except carrots, spinach, broccoli, and onions designated as sprinkler irrigated) and that all field crops are surface irrigated. As drip irrigation contributes to a lower composite HI value than furrow or sprinkler, this simplification results in a larger area of low-HI vegetable fields than actually exists in the SV, and therefore underestimates the vulnerability to nitrate lost below the rootzone (see Section 2.2.3 for more information).

Adminsitrative Record
Page 34811

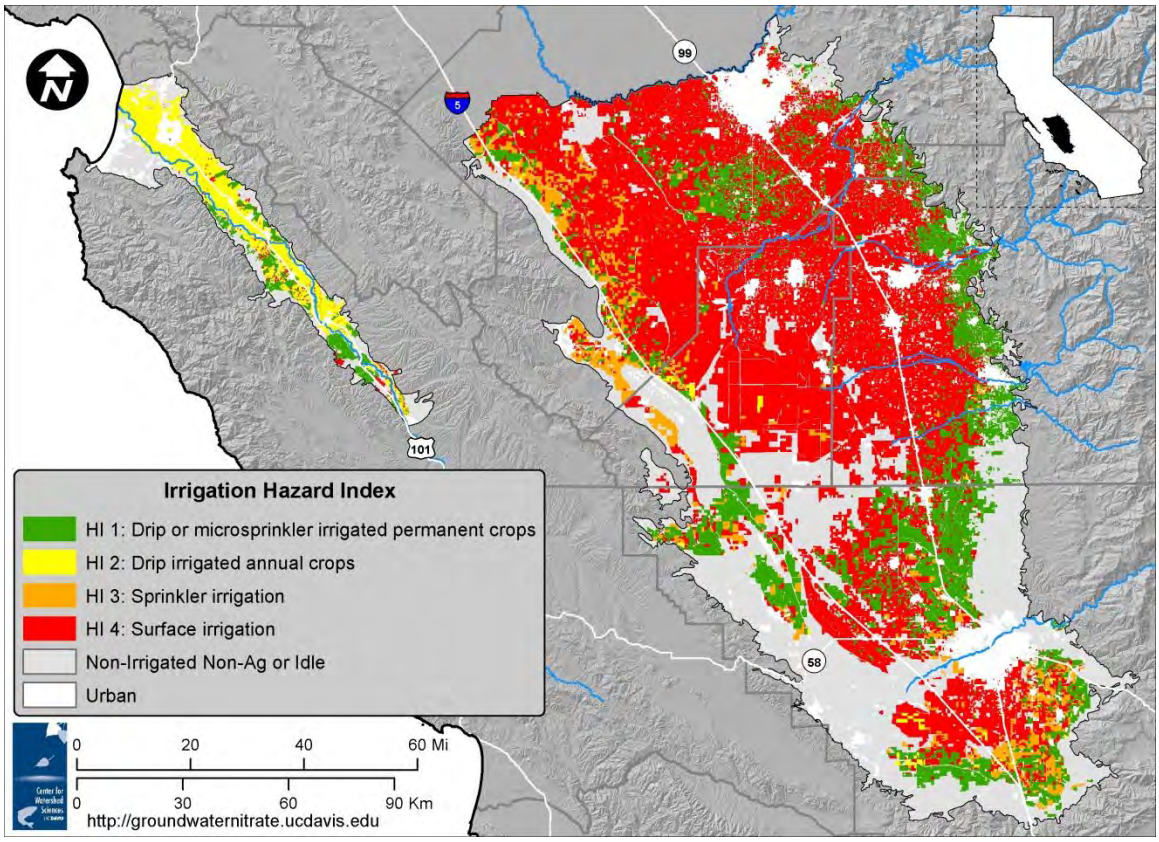

Figure 8. Irrigation hazard index value distribution. The most recent DWR reports for SV did not include irrigation data and thus for spatial representation it was necessary to designate 100% of each crop group to one specific irrigation system.[26] The prevalence of drip irrigated vegetables in the Salinas Valley is thus highly overestimated. In contrast, drip and microsprinkler irrigation is underestimated in the TLB due to the increased use of these systems in general (primarily in orchards, vineyards, and vegetables) since the time of the DWR irrigation survey data for these counties. (Source: DWR, UCANR WRC, internal designations, and Wu et al. 2005.)

2.6.4 Overall Hazard Index: Crop, Soil, and Irrigation

A large portion of the area of each basin falls within the low vulnerability definition (an overall HI of 20 or less) as shown in Figure 9. In areas in which HI<20, there is not as likely to be a significant decrease in N lost to leaching by adoption of improved practices. Either the soils are not as readily leached, or crop-irrigation combinations already make it easier to manage N conservatively. However, this does not mean that growers actually are managing optimally in these areas or that producers in such areas should not equally attempt to optimize their N use efficiency.

[26] In the SV, for the purpose of spatial analysis and calculation of an overall HI, field crops were designated as surface irrigated, tree crops (the majority of which are lemons in SV) and vineyards as micro-irrigated, and vegetables as drip irrigated (with the exception of carrot, spinach, onion, and broccoli fields designated as sprinkler irrigated)—see Section 2.2.3 for more information. Current actual estimates of drip irrigated vegetables in the SV are between 50% and 70% total.

Adminsitrative Record
Page 34812

In the Salinas Valley, 52% of the irrigated agriculture has an HI above 20 and is therefore considered vulnerable. The actual percent is even higher. As more recent irrigation type data were not available, we assumed all vegetables and strawberry fields to be irrigated by drip systems, and all vineyards and orchards to be drip or microsprinkler irrigated. In reality, it is estimated that only 50 – 70% of that acreage is irrigated by drip or microsprinkler, which would raise the percent of land with an HI>20. Highly permeable soils and shallow rooted leafy greens and other vegetable crops contribute to the large area of land that requires careful management to attain high crop NUE and low leaching losses (see Figure 10). However, with the large portions of the vegetables already converted to fertigated drip systems, increases in NUE have certainly occurred since times past, assuming proper maintenance and management of these systems is in place. Fallow period N management, especially during the rainy season and when pre-irrigating or germinating annual crops, continues to be critical, regardless of the irrigation/fertigation system used. Increased research, education and farmer outreach, and development of funding avenues will be required to increase adoption of other potentially high impact practices, such as fallow period covercropping.

In the Tulare Lake Basin, the areas of reduced concern are primarily those with the low risk soils, in areas of drip and microsprinkler irrigated trees and vineyards, and in areas in which alfalfa was recorded as growing at the time of the surveys. Although the majority of the area is cropped to lower risk crop species (Figure 6), the prevalence of higher risk surface irrigation and well-drained soils results in 1/3 of the basin having a calculated HI above 20, based on the available data. These results indicate that tight water and N management is required, especially in corn and vegetable production, as well as in the surface irrigated field crops grown on high risk soils. The TLB's high concentration of dairies contributes to overall leaching risk from both land application of manure and liquid manure storage lagoons and, to a much lesser extent, corrals (see Technical Report 2, Section 4, Viers et al. 2012). Most of the corn in the TLB is grown as silage for dairy cow consumption, and most of this silage corn receives manure and/or dairy lagoon water fertilizations applied by surface irrigation. Furthermore, these dairies are concentrated in areas of more permeable soils, driving risk upward. However, rotational effects may negate some of this risk, in that alfalfa, a highly effective perennial N scavenger (see MM 7, RP 7.2 and RP 7.4) is often included in the rotation scheme of forage crop producers.

Adminsitrative Record
Page 34813

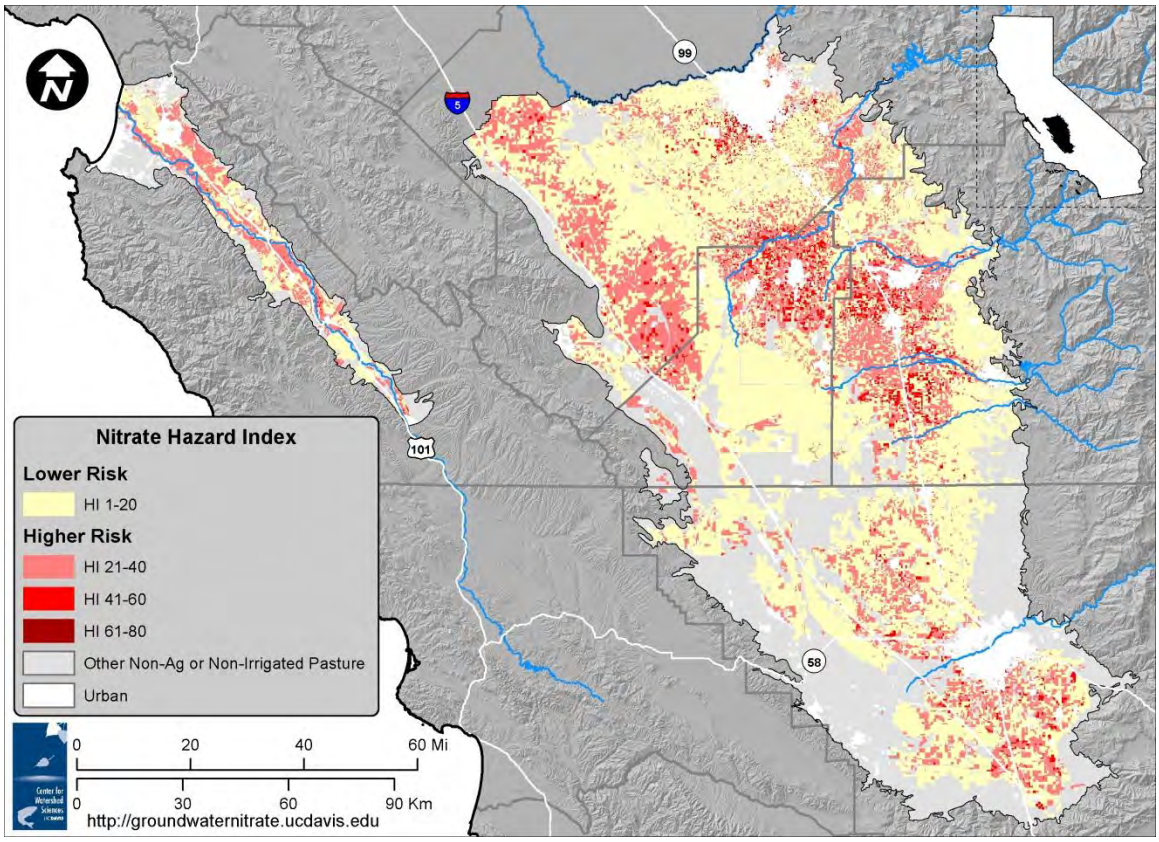

Figure 9. Overall hazard index calculated for the study area fields according to the most recent county DWR crop and irrigation data (Monterey 1997, Tulare 1999, Fresno 2000, Kings 2003, Kern 2006) and NRCS SSURGO soil series. All index values over 20 are considered vulnerable and indicate where improved management practices will likely have the greatest effect. Comparison between values in this category is not necessarily an indication of further risk differentiation, but may indicate that multiple variables are involved in risk. The less vulnerable areas still require vigilance in exercising good farm management practices. (Source: DWR, USDA, UCANR WRC, internal designations, and Wu et al. 2005.)

Again, the calculated final hazard index value is representative of risk of nitrate loss below the crop rootzone and not necessarily loading to groundwater, and is independent of management factors currently in use: vulnerability is based only on inherent characteristics of the soil, crop, and irrigation system in each polygon at the time of the DWR surveys (as described in Section 2.2.3). The proportion of the component HI values for each region is shown in Figure 10 below.

Changes in crop rotation will affect the overall HI distribution, especially if there is a continuing conversion on the more vulnerable (i.e., permeable) soils from lower-value, furrow irrigated crops to higher-value, drip or micro sprinkler-irrigated crops. Additionally, dairy forage crop producers routinely rotate corn, a high risk crop, with alfalfa, a nitrogen scavenging crop. Some of the most vulnerable fields shown in Figure 9 (primarily around Hanford, Visalia, and south beyond Tulare) are those on which field corn is grown (see Figure 6) and if rotated with deep rooted alfalfa (see RPs 7.2-7.4), a highly significant crop in the area, this may reduce risk slightly.

Adminsitrative Record
Page 34814

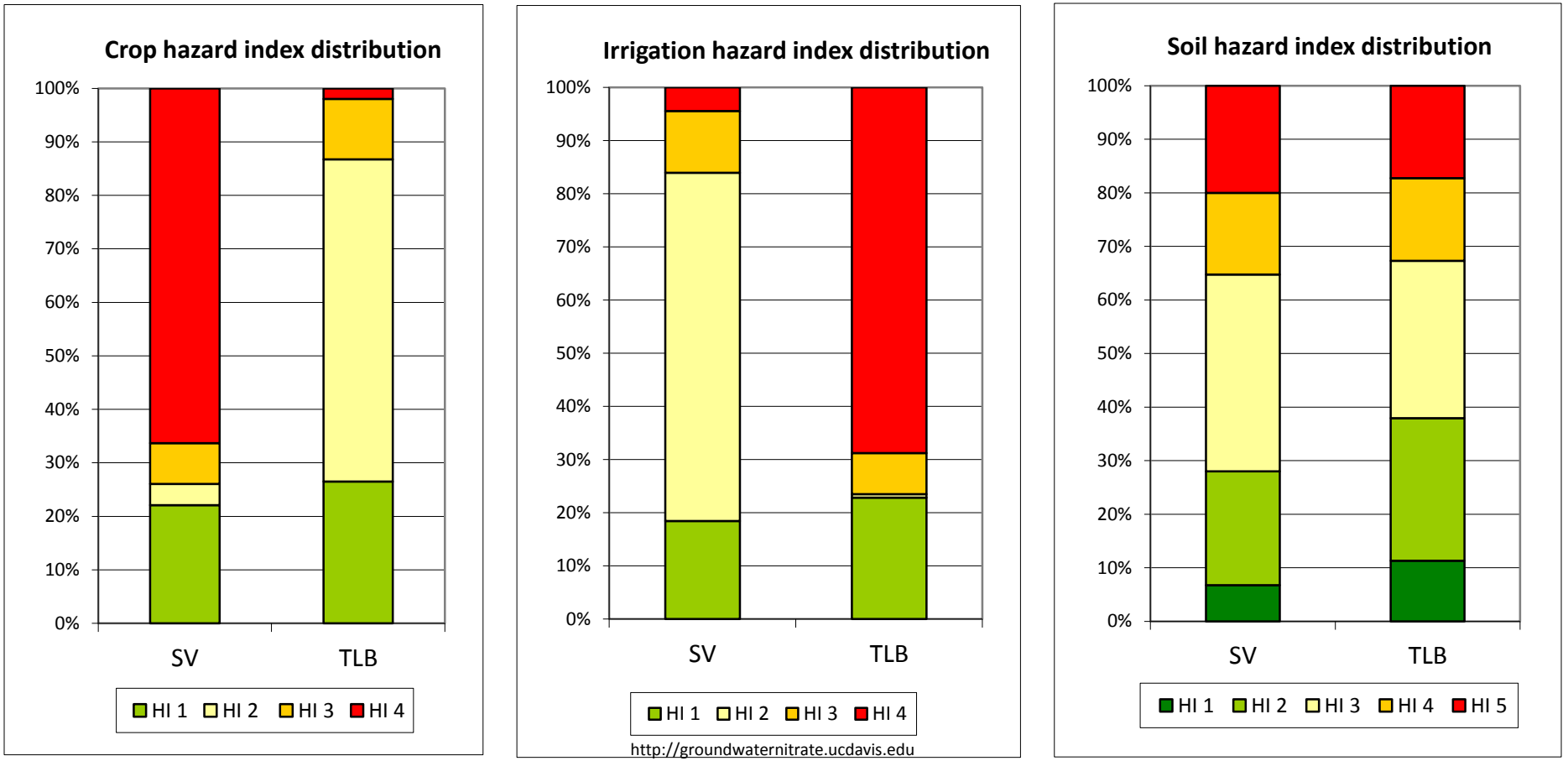

Figure 10. Distribution HI values by percent area for each component used in the determination of the overall hazard index values. SV = Salinas Valley. TLB = Tulare Lake Basin. Center figure: Note that all vegetables, strawberries, orchards, and vineyards in the SV had to be designated as drip or microsprinkler irrigated for spatial demonstration purposes (except spinach, broccoli, carrots, onions, and garlic; designated as sprinkler irrigated). This overestimates the percentage of lower risk irrigation in the SV, as it is estimated that only 50-70% of these crops are irrigated via low volume systems, rated HI 1-2 (see text for discussion).

Adminsitrative Record
Page 34815

A simulation was performed to illustrate how changes in irrigation system choice affect the overall vulnerability of the TLB. Over the past 20 years in the Tulare Lake Basin, many orchards, vineyards, and vegetable fields have been converted from surface gravity irrigation to low volume irrigation systems (drip and microsprinkler). Figure 11 shows both the overall HI value distribution in each region and illustrates the impact on the overall HI if the remaining area of these crops (orchards, vineyards, and vegetables) in the TLB were converted to such low volume systems. The rightmost bar in Figure 11 shows that the proportion of cropland with low vulnerability (HI<20) increases from 67% to about 78% due to such a conversion. It is likely that the conversion of orchards, vineyards, and vegetable fields to low volume irrigation will continue to increase in the coming years.

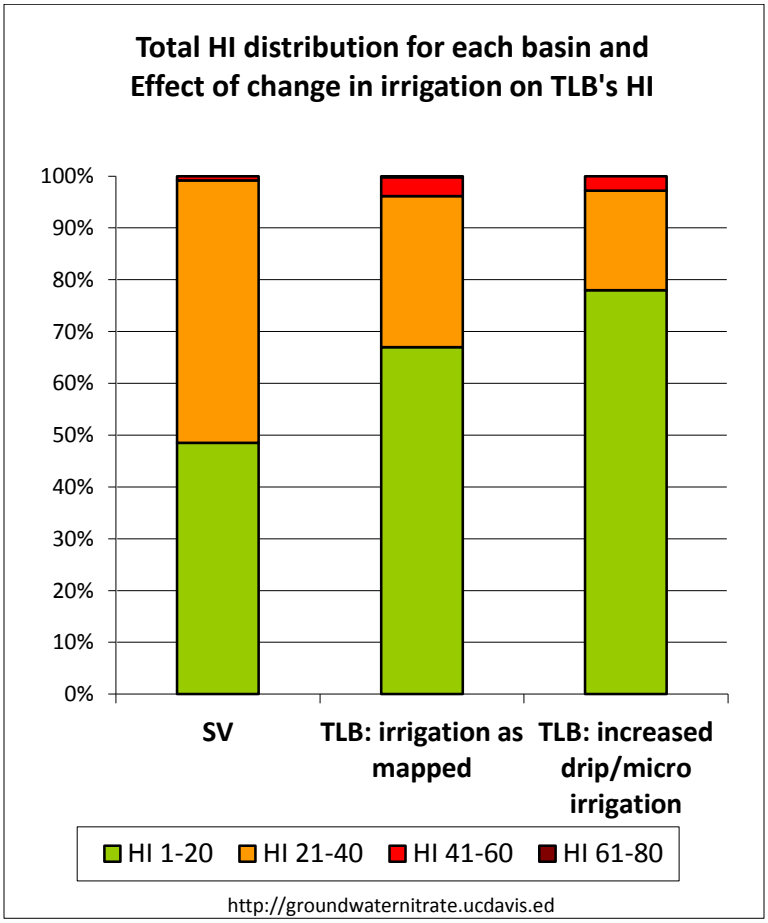

Figure 11. Composite hazard index values. Righthand bar shows a ~10% reduction in the area of high vulnerability in the TLB following an increase in the adoption of drip or microsprinkler irrigation systems over surface systems. That is, a change from the scenario as mapped (Figure 9, per DWR survey dates-- Tulare 1999, Fresno 2000, Kings 2003, and Kern 2006), to one where 100% of all vegetable, orchard, and vineyard area is drip or micro-irrigated. Note that while further conversion is expected in the area, a 100% adoption rate would be unrealistic, as it is inappropriate for some crops in those categories.

This exercise, again, ignores how these fields and irrigation systems are actually managed. As discussed in section 2.3.1 (see RP 3.6), optimally managed surface irrigation systems can outperform poorly managed and maintained drip systems in terms of overall efficiency and uniformity. However, due to

Adminsitrative Record
Page 34816

the increased control the operator has with low volume irrigation systems, a higher efficiency is expected when properly managed. These systems are not appropriate for some crops and remain economically unfeasible for others. Interest in center pivot and linear move sprinkler systems has increased in the San Joaquin Valley in recent years. These systems can deliver water with higher efficiencies than some of the traditional surface gravity systems; however, it is too soon to know whether these will be used on a significant fraction of cropland.

In summary, about 50% of irrigated cropland in the Salinas Valley and 25-35% of irrigated cropland in the Tulare Lake Basin has a nitrate hazard index over 20 and therefore is considered especially vulnerable to nitrate leaching losses according to the UC Nitrate Hazard Index tool. This evaluation is based on field soil series characteristics and the most recent spatial surveys of crop species grown and type of irrigation used on each field. It does not take into consideration management practices in actual use, crop rotational effects, or distance to groundwater. Thus while these results are not indicative of risks to groundwater (rather to risk of nitrate losses below the rootzone), the analysis illustrates the variability within the study areas and provides an estimate of the proportion of cropland (at a resolution similar to USDA county soil surveys) where use of mitigative practices would be most effective in decreasing overall nitrate leaching.

Adminsitrative Record
Page 34817

2.7 Conclusions

Nitrate leaching from the crop root zone is controlled by the interaction of farm management practices with inherent properties of the soil, weather, and crop species being grown. We have identified 10 farm management measures that contribute to efficient use of nitrogen by crops and detailed individual practices that can be used to achieve the management goals. Use of these practices, individually or in combination, can reduce nitrate leaching losses. Many of the identified practices are suited for use only with certain crop species or irrigation systems, and a few are suited for use only on certain soil types. The extent to which any one practice can contribute to NUE is site and crop specific. A suite of management practices appropriate to each field will have the most impact on NUE and leaching reduction. In general, NUE can be increased by optimizing the timing and rates of applied fertilizer N, animal manures, and irrigation water to better match crop needs, and to a lesser extent by modification of crop rotations and careful on-farm handling and storage of fertilizer and manure.

Based on the commentary from our expert panels, we found that a number of the identified practices have been adopted by study area farmers in recent years. An example is the conversion of vegetable fields from furrow and sprinkler to drip irrigation methods. While drip irrigation provides the key infrastructure for allowing reduced nitrate leaching losses, some farmers using drip have not yet optimized their systems to minimize leaching. For example, nitrogen fertilizer may still be applied in excessive amounts or at inappropriate times. Practices that are not already commonly in use by farmers in the study areas are generally associated with multiple barriers to an increased rate of adoption. High operating or capital costs and ranch logistical constraints represent significant and common barriers for a number of practices. Lack of access to adequate education and outreach activities is also one of the primary barriers for many of the less utilized practices.

Even where farmers are using “best management” or the recommended methods analyzed in this report, there are practical limits to the crop nitrogen use efficiency that can be achieved. The limits are due to spatial variability of soil hydraulic properties within fields, unpredictability of rainfall, the complexity of the N cycle and difficulty in predicting the rate of mineralization of organic N in the soil, and the need to leach salts from the rootzone. Where crop N use efficiency is low, significant improvements are possible with use of currently available technology, but due to the basic constraints mentioned here, NUE values above 80% are not likely achievable. However, understanding of the soil-crop system continues to improve, and future technologies may allow farmers to manage inputs even more precisely. It should be noted that nitrogen use efficiency should not be the only standard used to evaluate practices. In fact, the highest numeric NUE value may be achieved at yield levels that are not economically viable.

Not all croplands are especially vulnerable to nitrate leaching. Increasing the use of improved practices in areas that are the most vulnerable to leaching will have the most impact on groundwater nitrate concentrations. Based only on inherent characteristics of soil, crop, and type of irrigation system in use we identified those areas that are at least likely to present nitrate leaching, and those areas that are more vulnerable. We found that approximately 52% of irrigated cropland in the Salinas Valley and 35%

Adminsitrative Record
Page 34818

of such land in the Tulare Lake Basin is vulnerable to nitrate leaching, utilizing a Nitrate Hazard Index (Wu et al. 2005). While this assessment is not indicative of actual nitrate loading rates to groundwater, nor does it take into account actual management practices on individual fields or their location relative to sensitive aquifers, it does provide an estimate of the number of acres and general location of fields where attention to improving crop N use efficiency could have the most benefit.

While farm management has most certainly improved since times past, the rate of change has not been documented. Due to the generally long travel time of nitrate molecules to aquifers, current measurements of well water nitrate levels are most representative of past management regimes. The effect any one practice has on leaching is variable and depends on climate, soil characteristics, crop characteristics, crop rotation strategies, irrigation strategies, and other factors. For all of these reasons, it is impossible to assess the level of impact the improved management regimes employed by today's producers will have on future groundwater quality, and impossible to assess to what degree increased adoption of mitigative practices will have on this quality. However, it is certain the impact will be positive and that current average management, while an improvement over past practices, can still be considerably improved in terms of groundwater protection. Tandem implementation of improved management practices, chosen in relation to each unique farm situation, is the best approach to reducing nitrate leaching from agricultural fields.

Promising Actions: Based on all of the above findings, it is clear that expanded efforts to promote the adoption of nitrogen efficient practices are needed:

- The educational barriers that are associated with many of the identified practices highlight the importance of increased funding of research, education, and knowledge transfer (outreach) activities, to better assist farmers in applying best management strategies and nutrient management.
- More research is required that directly compares the effect of changes in management on nitrogen use efficiency or nitrate leaching under California conditions.
- Support development of crop-specific N-accounting methods that allow growers to evaluate their success in achieving high crop nitrogen use efficiency. An example of such is a "nitrogen mass balance metric" which is the ratio of the amount of N from all sources applied to a crop to the amount of N removed in the harvested crop. This would serve in lieu of direct estimates of nitrate to the groundwater, which is extremely difficult to monitor at the individual farm or farm field scale. Input:harvest ratios need to be averaged over multiple years to provide useful data.
- Implementation of a task force to review and fine tune methods for identifying cropland areas that are most vulnerable to nitrate loss below the rootzone. Several methods have been used or are under consideration for doing this in California; some attention should be given to methods that can be used at the farm and county scale for on-farm applied research and outreach activities. Such methods should include consideration of the spatial soil characteristics, as well as probable monitoring requirements.
- As the recently instituted Central Valley dairy regulations (CVRWQCB 2007) restrict on-farm cropland applications of manure, the volume of off-dairy transfer of animal waste to

Adminsitrative Record
Page 34819

unregulated fields has increased. It is not known which crop species and soils are receiving this manure or how the receiving farmers have integrated the manure into their N fertilization practices. Development of adaptive research and education programs that will promote conversion of solid and liquid dairy manure into forms that meet the food safety and production requirements for a wider range of crop species holds promise. Providing guidance to non-dairy farms in co-managing organic and conventional N sources is necessary to ensure that the nutrient value of the manure is properly accounted for in fertilization regimes.

Adminsitrative Record
Page 34820

3 Agro-Economic Analysis of Nitrate Source Reductions

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Acknowledgments

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Adminsitrative Record
Page 34821

3.1 Introduction

Improving nitrogen and water management on croplands is an important strategy to minimize future groundwater contamination. Nutrient, soil, and water management practices capable of reducing the impacts of croplands on groundwater quality are available (as reviewed in Section 2 of this report). However, implementation of new practices often requires increasing the intensity of management and hence changes costs and farming profitability. We develop a novel methodology of determining how policies aimed to reduce nitrate loading to groundwater would financially impact farmers in the Tulare Lake Basin and the Salinas Valley of California.

Widespread application of synthetic nitrogen fertilizers is a foundation for California's robust agricultural economy. However, excessive use has contaminated groundwater throughout California's agricultural regions (Stephany et al. 1998, Burrow et al. 1998; Zhang et al. 1998). Nitrate in groundwater is a public health concern. A majority of Californians rely on groundwater for their primary source of drinking water (DWR 2003) and ingesting nitrate has been linked to a series of negative health outcomes (Ward et al. 2005). Agriculture is both the largest contributor of nitrate to groundwater (see Technical Report 2, Viers et al. 2012) and is a primary driver of local economies in the Tulare Lake Basin and the Salinas Valley, as the five counties that comprise these regions are among the most agriculturally productive in the United States. Poorly planned nitrogen management policies may have significant impacts on the agricultural productivity and, consequently, economic health of these regions.

Various technologies and practices can help crop producers use nitrogen more effectively and reduce leaching losses, as outlined in Section 2 of this report. Conventional wisdom suggests that reducing nitrate loading will demand more intensive management and cost growers more money to produce, reducing net profit. The dual goals of maintaining profitability and reducing leaching potential may not always be at odds; N monitoring has been regarded as a possible low cost strategy (Knapp and Schwabe 2008) and can even lead to net profit increases (Hartz 1994). Identifying and implementing practices that produce similar win-win results has the potential to preserve the rural agricultural economy and groundwater for future generations.

In practice, farming operations often change several practices simultaneously (Technical Report 2, Viers et al. 2012). Suites of practices effectively increase nitrogen use efficiency and decrease pollution potential (Broadbent and Carlton 1978; Stark et al. 1983; Meyer and Marcum 1998; Letey et al. 1982). Combinations of production practices can be thought of as "bundles" of practices as they co-produce the desired benefits (see Section 2 of this report). Similar to individual practices that decrease leaching, bundles typically require capital costs for improving technology and additional operational costs by moving towards more intensive labor practices and education. Few studies quantify the costs of implementing technology bundles and their results on nitate loading. Knapp and Schwabe (2008) offer an example of a dynamic multi-year approach that accounts for water and nitrogen application as well as irrigation systems bundles. In our approach we account for water and nitrogen use efficiency, yet our focus is on the economics of nitrate leaching to groundwater as a result of both regulation and cost policies.

Adminsitrative Record
Page 34822

In this study, we develop a novel approach to estimate the cost of a menu of policies on farm profits. It is worth noting that we address only the production economics perspective. It is not the intention of this analysis to estimate the costs of the externalities to human health and other natural resources. Thus, results of our economic analysis of farm sector impacts need to be examined within a larger examination of societal trade-offs and options that considers alternatives outside the farm sector, including those for mitigation options such as drinking water treatment to address impacts of declining groundwater quality.

Modeling the interaction between nitrogen fertilizer, irrigation water use, crop mix, crop yield, and the costs and revenues of agricultural production is rather complex and involves a number of uncertainties. Previous research efforts have focused on the policy aspects of regulating nitrate, with less attention to economic effectiveness. Daberkow et al. (2008) offer a comprehensive literature review on economic modeling of public policies for changing nitrogen use practices in agriculture. In general, adverse impacts to farm income result from taxes on fertilizer or nitrogen effluent, or increasing limits on nitrogen application or effluent discharge. Effectiveness and costs vary across studies, but literature seems to concur that modest improvements in nitrogen use efficiency may come at little cost to farm net income (Knapp and Schwabe 2008).

Policy instruments for reducing groundwater nitrate loading may vary in effectiveness and ease of application. The policymaker needs to take into account heterogeneity in production to address equity issues from applying a given policy. Helfand and House (1995) found that, for nitrate in the Salinas Valley, implementing individual input taxes may lead to more socially optimal solutions, but such taxes are harder to apply. In contrast, second best policy instruments such as output taxes or uniform taxes or cutbacks may be close to the best performing policy and are often easier to apply. Nevertheless Helfand and House (1995) found that taxing a single production input such as N can be effective. Conversely, Knapp and Schwabe (2008) conclude that taxing water may be more cost effective than taxing N. In our approach, we simultaneously consider tradeoffs in costs and efficiency in nitrogen and irrigation, which accounts for different tax, cost, and nitrate load restriction schemes for water and nitrogen use efficiency. Thus we undertake an economic assessment of nitrogen limiting and nitrogen price policies. We complement our work with a sensitivity analysis that considers increases in the marginal costs of improving nitrogen use efficiency.

Adminsitrative Record
Page 34823

3.2 Crop Selection and Area of Study

A wide range of crops are grown in the Tulare Lake Basin and Salinas Valley, and they account for much of California's agricultural production. The final crop categories and their currently harvested areas are shown in Table 9. This crop mix covers more than 90 percent of the irrigated crop area of the two regions.

Table 9. Crop groups and land areas for modeling economics of nitrogen source reductions in crop farming.[27]

Crop group	Tulare Lake Basin (Acres)	Tulare Lake Basin (Ha)	Salinas Valley (Acres)	Salinas Valley (Ha)
Alfalfa	367,578	148,754	599	242
Almonds and Pistachios	407,007	164,710	--	--
Corn[27]	209,731	84,875	125	51
Cotton	605,154	244,897	--	--
Grain and Field[27]	667,910	270,293	18,618	7,535
Lettuce	3,048	1,234	48,209	19,510
Orchards[27]	212,056	85,816	342	138
Strawberries	364	148	8,492	3,436
Subtropical Tree Fruit[27]	305,691	123,709	1,423	576
Tomato[27]	132,804	53,744	2,858	1,157
Vegetables	175,085	70,854	100,470	40,659
Vineyards[27]	475,484	192,422	46,157	18,679
All other uses	1,699,329	687,694	169,622	68,643
Total	**5,261,242**	**2,129,149**	**396,916**	**160,626**

[27] Crop grouping follows DWR land and water use surveys available at http://www.water.ca.gov/landwateruse. Areas are from the California Augmented Multisource Landcover Map (CAML, http://cain.ice.ucdavis.edu/caml). Alfalfa is excluded from our analysis because it is often a nitrogen sink and our emphasis is on crops with higher NUE improvement potential. Field and Grain may include safflower, dry beans, sunflower, barley, wheat, oats, and excludes corn (all classes) and cotton. Likewise, vegetable crops are a composite of about twenty crops, excluding tomatoes (fresh and processing), strawberries and flowers, and others listed individually. Orchards exclude almonds and pistachios. Subtropical includes olives, avocado, citrus, and other. Small acreages are excluded. Corn includes all classes of corn and vine crops include grapes only. Kiwis and other vine fruit besides grapes are within the subtropical category.

Adminsitrative Record
Page 34824

3.3 Model

We developed a self-calibrated profit-maximizing model for agricultural production to assess the economic impact on farmers of policies that reduce nitrate loading from croplands.[28] Because nitrate loading to groundwater in irrigated cropping systems is largely a function of nutrient and water management, the model is based on economic and environmental consequences of changes in nutrient use and irrigation efficiency. It is assumed that better management will require additional monetary input (e.g., infrastructure cost, labor cost, costs for information and education, etc.), but will reduce total nitrate loading from croplands. The model allows for tradeoffs between monetary investments in production inputs (management practice bundles) and total nitrogen and water use. The optimization model maximizes profits from farming, while keeping yields constant in the constraint set definition.

3.3.1 Conceptual Framework of Model: Nitrogen Use Efficiency, Nitrogen Surplus, and Irrigation Efficiency

The mass of nitrate leaching to groundwater from irrigated croplands is a simple function of the amount of nitrogen applied times the quantity of water moving beyond the rootzone. Both of these critical pieces of information (and remedial strategies) must be considered. Our modeling approach presented here allows producers to adopt changes to both or either factors.

For the nitrogen part of the equation, our model is based on two interrelated metrics that, together, represent nitrate leaching potential: nitrogen use efficiency and nitrogen surplus. Measures of nitrogen use efficiency (NUE) are ratios of nitrogen taken up by the plant to nitrogen applied.[29] Given the total applied nitrogen N, a generally smaller *effective* amount of nitrogen, $\tilde{N}$, is used and taken up by the crop. The ratio $\tilde{N} / N$ is the "partial nutrient balance" (PNB), one measure of NUE.[30] Nitrogen surplus, or 'surplus,' equals the amount of N remaining in the field after harvest; in the model description section below we present, in equation form, these interrelationships. Hence, surplus is the difference between nitrogen applied and that taken up by the crop or surplus = amount of nitrogen applied * PNB. If the total applied nitrogen were equal to the effective nitrogen (100% NUE), $\tilde{N}$ as a function of N, $\tilde{N}(N)$, would yield a straight line with a 1:1 slope (Figure 12 below). We represent management practice bundles requiring specific capital and other investments in terms of their nitrogen use efficiency curves. For each practice bundle, the nitrogen use efficiency at low N application rates tends to be very high (albeit with low yields) and the value of $\tilde{N}(N)$ is close to the 1:1 line.[31] As the N application rate

[28] In this setting, the self-calibrated model (Howitt 1995) employs non-linear cost functions derived from the first order conditions of a farm profit maximization program, such that input use in the base policy model calibrates to those in the base dataset. These cost functions are referred to as Positive Mathematical Programming (PMP) cost functions after Howitt (1995).
[29] There are at least 18 ways to calculate nitrogen use efficiency (Ladha et al. 2005).
[30] It is worth noting that PNB is an indiscriminate proxy for NUE. That means that it cannot differentiate between nitrogen in plant material derived from mineralization or that derived from fertilizer. Thus, the inherent fertility of the soil can have a marked impact on PNB.
[31] In this way, our model aims to reflect a primary concern of using NUE as a measure of sustainable nitrogen management. The most nitrogen efficient producers are those that apply too little nitrogen. It is one of reasons that we chose to base our approach on a suite of indicators that included NUE, but also surplus and irrigation efficiency.

Adminsitrative Record
Page 34825

increases, the nitrogen uptake typically increases at a slower rate. Hence, the nitrogen efficiency decreases and the $\tilde{N}(N)$ curve levels off relative to the 1:1 line of $\tilde{N}(N)$ (see Figure 12 below). Plotting such curves for various (hypothetical) management practice bundles on a single graph, we can compare the nitrogen use efficiency of various practices. Bundles with lower slopes at a given nitrogen application rate are considered less desirable (e.g., Bundle 0), whereas bundles with higher slopes are preferred (Figure 12).

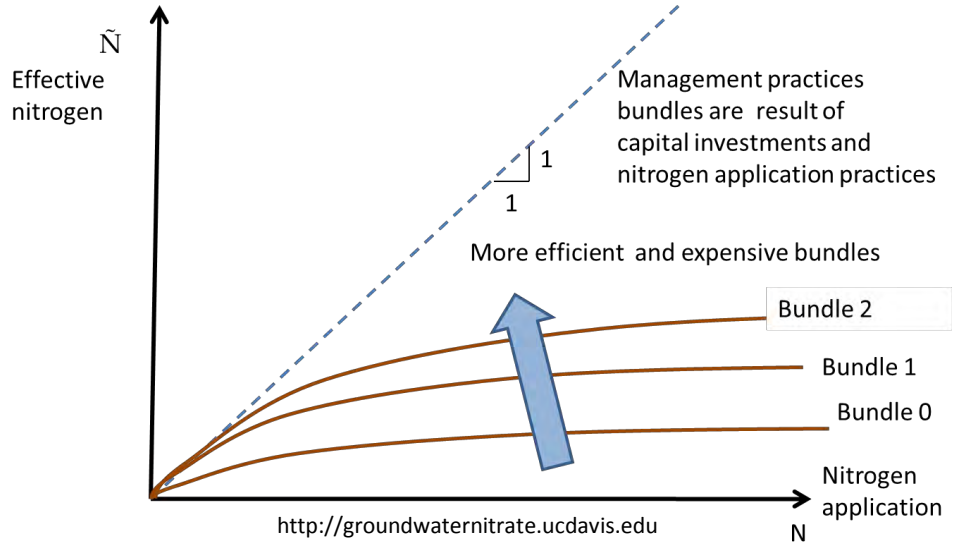

Figure 12. Effective nitrogen versus applied nitrogen by management practice bundle. Bundle 0 refers to practices before any improvements. Bundles 1 and 2 refer to more efficient and expensive bundles.

We use a substitution relationship between capital investments for nitrogen application practices and total nitrogen application calibrated to surveyed costs of application bundles. These tradeoff curves follow a Constant Elasticity of Substitution (CES) functional form and assume effective nitrogen use remains constant. One challenge for our approach is that bundles are discrete costs (i.e., they are either adopted or not adopted by the farmer) and therefore must be approximated to a non-linear function as shown in Figure 13. This figure shows an entropy-adjusted tradeoff curve of the cost of improving PNB versus the reciprocal of PNB for almonds and pistachios in Tulare Lake Basin. With respect to Figure 12 above, the lower right end of the tradeoff curve in Figure 13 corresponds to Bundle 0. As nitrogen use efficiency is increased by moving to the left along the tradeoff curve, the farmer shifts to more costly bundles (i.e., bundles 1, 2 or 3 in Figure 12). Since yields have to be maintained, the slope of *superior* bundles in Figure 12 increases at an increasing rate as do costs, as seen in the left portion of Figure 13. With the maximum entropy approach it is possible to estimate expected values for the parameters of a given curve with small or incomplete datasets. The theory behind the maximum entropy approach was developed by Shannon (1948) to quantify the expected value of a random variable.

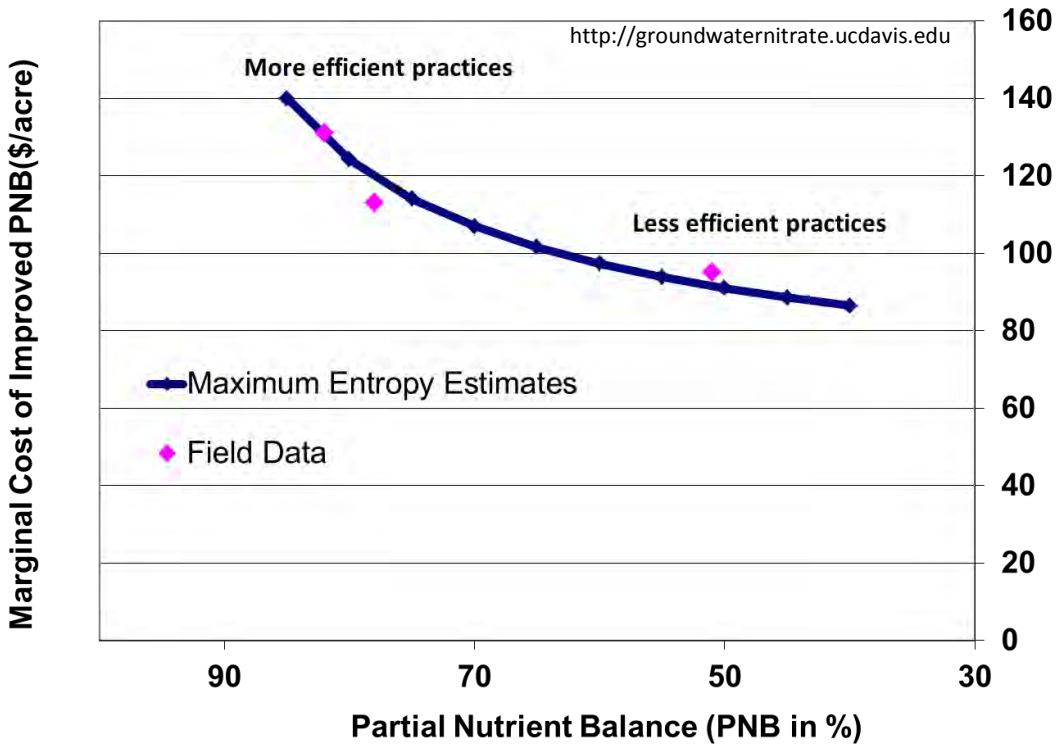

Figure 13. Tradeoff curve between Nitrogen Use Efficiency as Partial Nutrient Balance (PNB) and marginal costs per unit area to increase PNB. Within the tradeoff curve, yields are assumed constant. Higher nitrogen use efficiency (left in the horizontal axis) is achieved by increasing capital investments (labor, technology, and management) in nitrogen use efficiency (vertical axis).

Likewise, for irrigation efficiency, we assume that bundles with higher irrigation efficiency require capital investments to maintain crop yields. We measure irrigation efficiency as the inverse ratio between applied water and evapotranspiration of applied water. The United States Bureau of Reclamation (1997) conducted a study on the Central Valley to parameterize this tradeoff relationship (Figure 14).

Adminsitrative Record
Page 34827

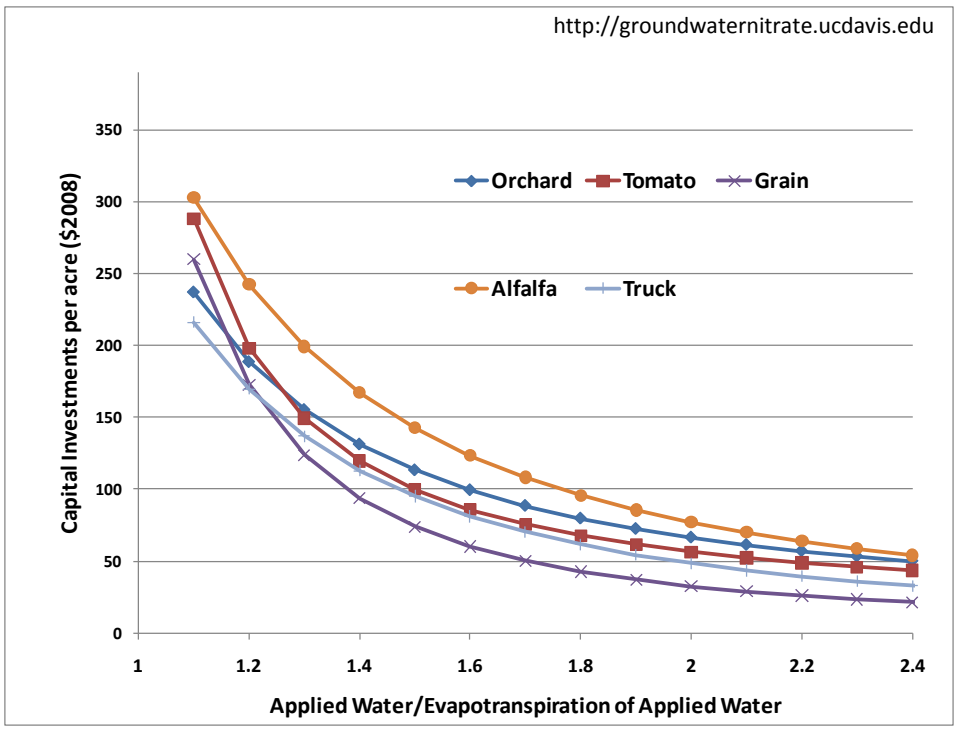

Figure 14. Sample tradeoffs between capital investments and irrigation efficiency for selected crops for a constant crop yield (adapted from USBR 1997).

Information on irrigation technology and an approximation of the tradeoffs between capital investment and efficiency exists from previous studies (USBR 1997). However, with the exception of Knapp and Schwabe (2008), very few analyses have been published comparing cost of improved N use practices, crop N use efficiency, and economics of nitrate leaching to groundwater. In the following subsection we present the model formulation and assumptions to derive such relationships for bundles of nitrogen management practices.

3.3.2 Model Formulation

Our model follows a multi-step calibration process using a CES function with two nests: efficient water use and effective nitrogen. In our first step we employ a Leontief technology which allows no substitution among inputs. The production function for the farmer for each crop includes six variable inputs:

- Land
- Water
- Supplies(Supp)
- Capital investments in water application (Wcap)
- Nitrogen (AppN)
- Investments in improving nitrogen use efficiency and management (CPNB)

Adminsitrative Record
Page 34828

Supplies refers to other variable inputs including labor and farming supplies, other than nitrogen and water, which have been lumped into an amalgam of variable production costs per acre. In our program, *capital investments* refers to expenditures on equipment, management and operation costs, which may include additional training of personnel, increased supervision, and use of crop consulting services. Two tradeoff curves exist in the model as mentioned above: one for water versus water capital investments and another for nitrogen versus nitrogen capital investments. Medellin-Azuara et al. (2012) present the full set of equations of a similarly nested model for irrigation efficiency only. In our present application we provide a simplified set of equations for the three steps. The footnote in this page describes the variables, parameters and subindexes employed in the equations to follow. In the first step the objective function (Eqn. 1) is given by:[32]

$$\max Z = \sum_g \sum_i \left(V_{gi} Y_{gi} XL_{gi} - \sum_j \left[a_{gij} XL_{gi} \omega_{gij} \right] \right) \quad \text{(Eqn. 1)}$$

Where *Z* stands for total net revenues; V_{gi} , Y_{gi} crop *i* (Table 12) in region *g* (TLB and SV) prices and yields, respectively; and $XL_{gi,}$ is the decision variable for input allocation of land. On the cost side, the parameters a_{gij} and ω_{gij} are the Leontief coefficient normalized to land and the unit cost coefficient, in region *g* and crop *i*.

The program is constrained in Eqn. 2 to a limiting amount of water and land:

$$\sum_i a_{gij} XL_{gi} \le b_{gj},\ j \in \{land, water\} \quad \text{(Eqn. 2)}$$

with supplies, effective water, and effective nitrogen use as in Eqn. 3, being and $\tilde{X}_{gij}$ the base *observed* value of the production input:

$$a_{gij} XL_{gi} = \tilde{X}_{gij} \forall g, i,\ j \in \{EffW,\ EffN,\ Suppl\} \quad \text{(Eqn. 3)}$$

In other words, the objective function (Eqn. 1) maximizes the net revenues for a limited amount of land, water, and for a given amount of supplies, water and nitrogen use efficiency. By comparing the optimized values for land, water cost, nitrogen cost, and crop allocation (including costs of increasing

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[32] Sub-indexes ***i***, ***j***, and ***g*** stand for crop group (see Table 12), input (land, effective water, effective nitrogen, investments in water use efficiency, investments innitrogen use efficiency, applied water and applied nitrogen) and region (TLB and SV). The two virtual inputs on *j* : effective water (EffW) and effective nitrogen (EffN) are used in the two lower level nests.
Z, NL2, net revenues in the objective functions respectively for the linear and last optimization steps.
$\tilde{X}_{gij}$ observed input use in region *g, for crop i.*
XL, **XN**, and **XNN:** Decision variables land in the linear program, nested CES, and main CES function..
$\mathbf{V_{gi}}$ and $\mathbf{Y_{gi}}$, $\mathbf{b_{gj}}$: Respectively, Price per ton, and yields in tons per unit land area of crop *i* in region *g*; and available land and water (limited resources) for region *g*, where$\mathbf{b_{gi}}$= $\tilde{X}_{gij}$ *and* $j \in \{land, water\}$
$\mathbf{a_{gij}}$ and $\boldsymbol{\omega_{gij}}$ Leontief coefficient (normalized to land) and linear cost of production input *j* for crop *i* in region *g.*
β1, β: Respectively share parameters for the nested (effective water and effective nitrogen) and main (production) CES functions.
$\boldsymbol{\tau_{1gi}}$, $\boldsymbol{\tau_{gi}}$ Scale parameter of the nested and the main (production) CES functions respectively.
$\boldsymbol{\alpha_{gij}}$, $\boldsymbol{\gamma_{gij}}$**:** parameters of the PMP quadratic cost function,

Adminsitrative Record
Page 34829

efficiency) at different water and nitrogen use efficiencies, we can compare the cost of improving nitrogen use efficiency, which in turn has the potential to decrease nitrate groundwater load. The model of nitrogen application efficiency versus capital investments in nitrogen use efficiency and the model of water capital investments versus irrigation efficiency are described below.

3.3.3 Water Capital Investments versus Irrigation Efficiency

Capital investments for improvements in irrigation efficiency versus total applied water can be likewise modeled following USBR (1997). The evapotranspiration of applied water for each crop is used as a proxy for irrigation efficiency. Formulation is as in Eqn. 4:

$$a_{gi,EffW} XL_{gi} = \tau 1_{gi}^{W} \left\{ \left[\beta 1_{gi}^{W} a_{gi,water} XL_{gi} \right]^{\rho_i} + \left[(1-\beta 1_{gi}^{W}) a_{gi,Wcap} XL_{gi} \right]^{\rho_i} \right\}^{1/\rho_i} \qquad \text{(Eqn. 4)}$$

Information to calibrate this component exists from previous work by the USBR (1997). The effective water amount, on the left hand side is given by the *observed* evapotranspiration of applied water. The parameters τ_{gi} and $\beta 1_{gi}$ are the scaling and the share factors in CES functional form. On the right hand side, $a_{gi,EffW} XL_{gi}$ for applied water and capital investments in applied water, represent the factors within the water efficiency nest that may substitute for each other. Finally, ρ_i is given by the elasticity of substitution[33] σ_i of crop *i,* such that $\rho_i = (\sigma_i - 1)/\sigma_i$.

3.3.4 Investments in and Costs of Increasing Nitrogen Use Efficiency

The second nested component[34] in the objective function (Eqn. 1) is used to represent tradeoffs between nitrogen application and costs associated with improving nitrogen use efficiency, assuming agricultural yields are not negatively affected by these improvements. We employ, again, a constant elasticity of substitution relationship between costs of reduction and nitrogen application (Eqn. 5), such that:

$$a_{gi,EffN} XL_{gi} = \tau 1_{gi}^{N} \left\{ \left[\beta 1_{gi}^{N} a_{gi,AppN} XL_{gi} \right]^{\rho_{Ni}} + \left[(1-\beta 1_{gi}^{N}) a_{gi,CPNB} XL_{gi} \right]^{\rho_{Ni}} \right\}^{1/\rho_{Ni}} \qquad \text{(Eqn. 5)}$$

Where, $a_{gi,EffN} XL_{gi}$ or effective nitrogen is proportional to the PNB and corresponds to the vertical axis in Figure 12. On the right hand side, applied N and capital investments in NUE conform the substitutable factors in this second nest. The rest of the parameters are as in the water efficiency nest (Eqn. 4). In Figure 13, the PNB is shown on the horizontal axis and the investments in NUE per unit area, given by $a_{gi,CPNB} XL_{gi}$, are shown on the vertical axis. $a_{gi,CPNB} XL_{gi}$ is the estimated unit cost of labor, supervision,

[33] The elasticity of substitution is a measure of how one production input can substituted for another. Low elasticities of substitution indicate low substitution possibilities (such as water for labor). Constant elasticity of substitution production functions constrain the ability to substitute among production inputs to remain constant at any level of production.

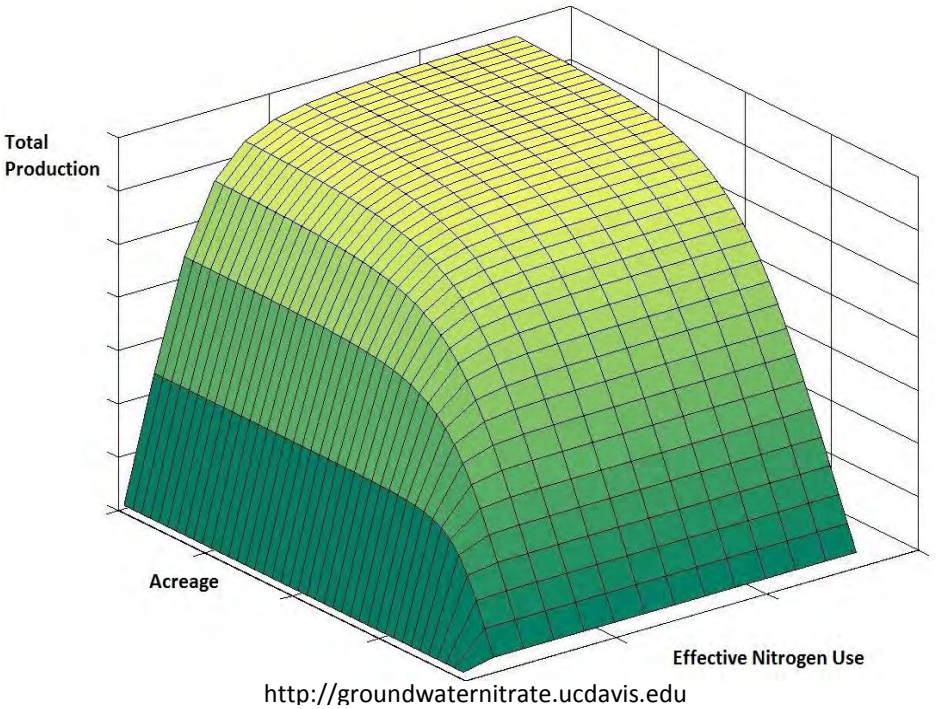
[34] The components are called nested because the left hand side variable in Eqn. 4 and 5 are nested in the constant elasticity of substitution production function in Eqn. 1.

Adminsitrative Record
Page 34830

and materials associated with the level of NUE on the horizontal axis, over a c*onstant yield* tradeoff curve.

In this case, the substitution parameter ρ_{Ni} was estimated empirically using a maximum entropy approach, as only a small dataset existed for PNB versus costs per unit area required to achieve that particular PNB. Maximum entropy theory (Jaynes 1957; Shannon 1948) makes maximum use of the existing information to estimate probability distribution of a particular parameter. Figure 13 exemplifies an estimation taken for this study.

A tradeoff between production and inputs is shown in the simplified graph below (Figure 15), where tradeoffs between nitrogen uptake, planted acres, and crop production are illustrated. This illustrates that total production is the result of combining inputs, which also share tradeoff relationships between them.



http://groundwaternitrate.ucdavis.edu

Figure 15. Simplified representation of crop production tradeoff surface between effective nitrogen use and land use.

In addition to Eqns. 1 – 5, a calibration constraint on XL_{gi} that restricts land to observed values $\tilde{X}_{gi,land}$ is employed following Howitt (1995). Once a solution to the linear program of these six equations is found, the second step in our model consists of taking the Lagrangian of the land use constraint to estimate a PMP quadratic cost function as in Medellin-Azuara et al. (2012). In the third and fourth steps (not shown here), the parameters for the CES water efficiency and nitrogen use efficiency are obtained. The resulting calibrated program in the fifth step is given by Eqns. 6 – 9, below.

Adminsitrative Record
Page 34831

$$\max NL2 = \sum_g \sum_i v_{gi} \left[\tau_{gi} \left(\sum_j \beta_{gij} XNN_{gij} \right)^{\rho_2} \right]^{1/\rho_2} - \sum_g \sum_i \sum_j \left(\alpha_{gij} XNN_{gij} + \gamma_{gij} XNN_{gij}^2 \right) \quad \text{(Eqn. 6)}$$

The first term in the right hand side of Eqn. 6 represents the main CES production function containing two nests of Eqns. 7 and 8. The last term in Eqn. 6 represents the calibration PMP cost function (Howitt 1995). To the objective function, two nested CES and a resource constraint follow:

$$XNN_{gi,EffW} = \tau_{gi} \left\{ \left[\beta 1_{gi}^{W} XNN_{gi,water} \right]^{\rho_i} + \left[(1 - \beta 1_{gi}^{W}) XNN_{gi,Wcap} \right]^{\rho_i} \right\}^{1/\rho_i} \quad \text{(Eqn. 7)}$$

$$XNN_{gi,EffN} = \tau_{gi} \left\{ \left[\beta 1_{gi}^{N} XNN_{gi,AppN} \right]^{\rho_{Ni}} + \left[(1 - \beta 1_{gi}^{N}) XNN_{gi,CPNB} \right]^{\rho_{Ni}} \right\}^{1/\rho_{Ni}} \quad \text{(Eqn. 8)}$$

$$\sum_i XNN_{gi} \le b_{gj} \ , \forall \ \text{g,j} \ \in \{land, water, suppl\} \quad \text{(Eqn. 9)}$$

The mass balance and policy constraints are described in the next subsections. Modifications to the mass balance constraints and costs of inputs (second term in Eqn. 6), allow us to model the cost of different policy options.

3.3.5 Nitrate Loading to Groundwater from Agricultural Production

In our modeling approach, to estimate nitrate load to groundwater in irrigated systems, a simplifying assumption is that nitrogen use efficiency cannot exceed the irrigation efficiency. In other words, farmers that employ efficient irrigation practices are more likely to adopt (if they have not done so already) more efficient nitrogen application practices. In addition, to compute groundwater nitrate loading, we assume that 10 percent of the applied nitrogen is lost to the atmosphere as ammonia, nitrogen oxides, or di-nitrogen gas. The remaining 90% of the (annually) applied N is either taken up by the crop or leached to groundwater. The groundwater nitrate load is the maximum between fraction of applied nitrogen that remains after atmospheric losses and nitrogen removed by harvest. The maximum potential fraction of nitrogen that can leach into groundwater is thus formally:

$$GW_{NO_3load,gi} = Max \ \{0, XNN_{gi,AppN}(0.9 - PNB_{gi})\} \quad \text{(Eqn. 10)}$$

Where $GW_{NO3load,gi}$ in Eqn. 10 is the estimated groundwater nitrate load and the rest of the terms, as defined earlier. Note that nitrate load to groundwater cannot be negative thus, there is a minimum of zero in the aforementioned computation. In Eqns. 11 and 12 below, we present Partial Nutrient Balance as related to surplus, harvested, and total applied nitrogen:

$$PNB_{gi} = 1 - \frac{SurN_{gi}}{AppN_{gi}} = 1 - \frac{AppN_{gi} - HarN_{gi}}{AppN_{gi}} = \frac{HarN_{gi}}{AppN_{gi}} \quad \text{(Eqn. 11)}$$

Adminsitrative Record
Page 34832

$$HarN_{gi} = PNB_{gi} AppN_{gi} \quad \text{(Eqn. 12)}$$

$SurN_{gi}$ is nitrogen surplus and $HarN_{gi}$ is the nitrogen removed by harvest. We also assume that irrigation efficiency is greater or equal to PNB, as some farming operations may, for example, have a well-managed drip irrigation system with a high water use efficiency, but still have very low PNB due to remaining inefficient nitrogen management practices. We assume that a high PNB cannot be achieved when irrigation efficiency is low.

One caveat of this approach is that there may be events or seasonal cases in which the irrigation efficiency is poor, yet nitrate leaching is also low. This may occur, for example, if soil nitrate concentration is low during pre-irrigation or during the winter (rainy season), where groundwater recharge is high. Likewise, reducing soil nitrogen during these times represents an improved practice. In our approach we consider total annual groundwater nitrate loading, and are particularly concerned with losses during the irrigation season.

3.3.6 Policy Simulations for Nitrogen Use Efficiency

Our policy simulations are aimed at estimating changes in agricultural revenues from changing cropping patterns (including increases in fallow land area), resulting from implementation of nitrate load reducing policies. We also estimate changes in revenue from efficiency improving management measures, taxes on nitrogen use and maximum load limits, and other policies. We are not concerned here with neither the specific aspects of such policies, nor with the feasibility of these policies. Instead we investigate expected shifts in cropping patterns and changes in farm revenues at different levels of restrictions to nitrate leaching.

To represent restrictions in nitrate load to groundwater, our modeling approach imposes a constraint on the amount of nitrogen such that, for region *g*, the maximum nitrate load is given by:

$$\sum_i GW_{NO_3load,gi} \leq Red_g \sum_i \tilde{X}_{gi,AppN}(0.9 - PNB_{gi}) \quad (13)$$

On the left hand side groundwater nitrate load for region *g* is as in Eqn. 10; Red_g is the policy determined factor to reduce loading to groundwater by some percentage for region *g*, and the summation over *i* is the *current* groundwater nitrate load from crop *i* in region *g,* assuming that, from the *observed* applied nitrogen ($\tilde{X}_{gi,AppN}$), 10 percent is lost to the atmosphere and the rest is removed by harvest. Thus, Red_g would equal one unit if we are modeling a base case with no reductions, and 0.75 if we are cutting down nitrate load to groundwater by 25%. Water use efficiency is constrained to be greater than NUE, such that the weighted PNB is less than the weighted water use efficiency in a region.

In summary, the process consists of five steps: 1) linear land constrained program (Eqns. 1 to 5); 2) estimation of a calibration PMP cost function; 3) parameterization of the irrigation efficiency nest; 4) parameterization of the nitrogen efficiency nest; and 5) base calibrated model (Eqns. 6 to 11). In this

Adminsitrative Record
Page 34833

fifth step, we maximize regional producers' surplus (Eqn. 6, above) accounting for tradeoffs between costs and efficiency in irrigation and nitrogen management (Eqns. 7 and 8), resource constraints (Eqn. 9), mass balance (Eqn. 10), and policy-based nitrate leaching limits (Eqn. 13).

Adminsitrative Record
Page 34834

3.4 Model Datasets

The model is calibrated based on publically available datasets. Because data are insufficient to estimate a baseline and improved irrigation and nitrogen set of practices for all crops in the two study regions, we opted to perform the analysis on crop groups. Data were thus aggregated into crop groups based on an area-weighted average. One shortcoming of using this crop group approach is the aggregation of the response: all crops within a group are assumed to respond equally to costs of improvement.

3.4.1 Irrigation and Cost Data

Production input usage for land, water, labor, and supplies (excluding nitrogen) are obtained from the Statewide Agricultural Production Model (SWAP) .[35] Irrigation efficiency, the ratio of evapotranspiration of applied water to applied water, was taken from the California DWR crop group as reported in the California Water Plan Update (DWR 2009).[36] The capital costs per unit area for irrigation efficiency were obtained from USBR (1997) and scaled up to 2008 dollars, as were the rest of the monetary costs on inputs. Production information from UC Davis agricultural cost and return studies was employed for crops that were not included in the original SWAP formulation including lettuce and strawberries.[37] The irrigation technology parameters employed for the CES trade-off curves in Figure 14 follow USBR (1997).

3.4.2 Nitrogen Use and Cost Data

Because data are generally unavailable to estimate nitrogen use or cost data for individual practices, let alone, bundles of practices, we developed datasets to estimate efficiency and costs for three scenarios of practices: a current baseline scenario (Bundle 1), an improved scenario (Bundle 2), and an idealized and most efficient scenario (Bundle 3). Bundle 1 represents the efficiency and cost of current practices. Bundle 2 represents the scientifically tested improvement in nitrogen management possible with currently available practices. Bundle 3 represents the presumed benefits for NUE, surplus, and nitrate loading and economic costs for practices that are under development or not yet practically feasible at scale.

3.4.2.1 PNB of Bundles

The first step in developing the dataset was to estimate the PNB for each of the three bundles. Bundle 1 was presumed to be the baseline or current practice. For Bundle 1, we calculated PNB from available statistics.[38] Calculating a PNB requires knowing yield, moisture, and nitrogen content of the crop, as well as nitrogen application rates. We derived these values from the following sources: yields (USDA 2011a), moisture and nitrogen content (USDA 2011b), and nitrogen application rates (Rosenstock et al. in

[35] http://swap.ucdavis.edu
[36] http://www.waterplan.water.ca.gov
[37] http://coststudies.ucdavis.edu/
[38] It is worth noting that statistics on nitrogen usage are notoriously difficult to come by and any statistics found can at best be an approximation due to the variation among fields, farms, and regions.

Adminsitrative Record
Page 34835

review). Because the PNB of Bundle 1 reflects statewide average reported values, it aggregates across all of the practices that are occurring currently. This includes both advanced nitrogen management practices in some cases, as well as relatively traditional nitrogen management practices in others. Yet we assume that the PNB derived from the statewide averages is equal to the PNB for the most common unimproved bundles. Implicitly this means that depending on the current extent of adoption of improved practice bundles, the baseline PNB may be an overestimate.

Bundle 2 is the so-called "improved," scientifically verified, collection of practices. For this bundle, PNB figures were compiled through a literature review. We surveyed published literature and collected unpublished data to find the most recent research on nitrogen management in California for 22 economically important crops (see footnote of Table 9 for references). These studies and data reflect recently developed and tested nitrogen and irrigation best management practices. We made every attempt to include research from field-scale nitrogen trials. Research conducted at research stations was excluded when other research existed because these trials generally outperform results observed in a grower's field. That is, PNB will be higher under research-station conditions than in the field. Data relevant to calculating improved PNB values using the above equation were obtained adhering to the following prescription:

- Only reported yield and nitrogen application rates that corresponded to realistic nitrogen application rates for a particular crop were used. We only included data from research scenarios with reasonable application because nitrogen rate trials historically also include a zero-N treatment (no nitrogen applied) and an excessive nitrogen treatment with large amounts of nitrogen applied. Including these two values in the calculations would have potentially biased the PNB values.
- Where research reported the amount of nitrogen in the harvested portion of crop, those values were used directly. Where research only reported yield, but not crop nitrogen content, the amount of nitrogen in the crop was calculated based on the USDA Crop Nutrient Tool (USDA 2011b).

Bundle 3 represents the highest potential gains plausible. Many practices that are currently used by growers or are under development, such as weather based irrigation scheduling in cool-season vegetables, and will potentially reduce nitrate loading further than the improved practices identified above. However, data quantifying PNB and nitrate loading are not currently available. We represented these most efficient practices by including a third hypothetical bundle in our tradeoff between nitrogen application improvement and the costs of those improvements, in which we assumed PNB to be 5% higher than in the improved practice bundle.

3.4.2.2 Costs of Bundles

Estimated costs of 'bundles' are unavailable, especially when considering the range of crops grown in the study regions. Because of the paucity of data, we developed an index to estimate costs and the differences in costs among the bundles (hereafter referred to as the 'cost ratio'). The cost ratio

Adminsitrative Record
Page 34836

estimates the relationship between the cost of applying fertilizing materials (labor, machine time, information, etc.) and the fertilizing materials themselves. The cost ratio is based on the assumption that improving PNB generally results from more active management, demanding greater resources. As nitrogen and water management improve, the relative cost of application compared to fertilizer itself increases.

Cost ratios for the baseline and improved scenarios for each crop group were derived from the UC Agricultural and Resource Economics Department Cost and Return Studies (CS, http://are.ucdavis.edu). Cost ratios are not costs directly reported in the CS studies, rather they are estimates calculated from the available data. Estimated costs of bundles were developed to be consistent with the agronomic practices used to calculate PNB (e.g., industry standard practices for Bundle 1 and the practices used in the nitrogen trials for Bundle 2). Cost studies are inconsistent in how they present data, so we adopted a few basic rules to standardize estimations (Table 10). Rules needed to be created to disaggregate co-mingled costs. For example, CS studies often include the labor costs of fertigation with irrigation. Yet, increasing the frequency and number of fertigation events is a practice to increase NUE and decrease pollution potential. A fraction of the irrigation labor therefore must be attributed to the fertigation costs to obtain a realistic estimate.

Often there were CS studies that approximately reflected baseline and improved practices (e.g., furrow versus drip irrigation). In these cases, costs were derived from studies created for each practice. However, when improved practices represent only slight modifications of existing systems or if only one recent CS was available, costs for both baseline and improved PNBs were estimated from the same study. Since we created ratios of costs and they were consistent within a study, the ratios are comparable across studies.

The PNB and the respective cost ratios can be found in Table 11. These points correspond to Figure 13 and are employed to estimate the depicted CES relationship. Improvements in NUE modeled in this study lay within the continuum of this entropy-estimated relationship and will not necessarily correspond to Table 11 data points.

Adminsitrative Record
Page 34837

Table 10. Rules and justification to standardize data extraction from UC ARE Cost and Return Studies. These data were used to produce cost ratios for baseline and improved PNB.

Practice	Issue	Rule	Calculations
Fertigation	Labor costs of fertigation often included with irrigation labor costs.	Cost of labor depends on the number of fertigation events.	< 2 events: 5% of irrigation labor
			2 – 5 events: 10% of irrigation labor
			> 5 events: 15% of irrigation labor
Mixed operations	Fertilizer is often spread in conjunction with other operations.	Cost of labor is proportional to the number of operations.	Divide labor by number of operations performed.
Fertilizer blends	Fertilizer costs often includes other elements.	The cost of fertilizer is proportionate to the amount of N in the fertilizer blend.	Divide % N by total % NPK and multiply by cost of material.
Pest control advisor/consultant	Consultants advise on more than fertility decisions	Fertility is largely an add-on service estimated at 20% of cost.	Multiply custom cost by 0.20. Consultant costs limited to improved PNBs.
Diagnostic tests	CS only include more than 1/season.	Scale cost by the number used to create PNB.	Cost of test x number of tests.
Fertilizer rates	CS fertilizer costs usually for different N fertilizer rates than used in either PNB.	Scale costs to match fertilizer rate used to create PNB.	Use cross products to scale costs.

Adminsitrative Record
Page 34838

Table 11. Costs of improving N fertilization practices. The cost ratio is the ratio of cost of labor to N fertilizer materials.[39]

	Current practice		Improved practice	
Crop	PNB[1] (%)	Cost ratio[2]	PNB[1] (%)	Cost ratio[2]
Almond and pistachio	51	0.04	78	0.07
Corn	89	0.17	90	0.22
Cotton	61	0.05	67	0.15
Grain	80	0.13	91	0.19
Lettuce	35	0.24	51	0.35
Strawberry	34	0.07	55	0.14
Other vegetable crops	42	0.21	46	0.34
Other tree crops	36	0.12	33	0.36
Processing tomatoes	66	0.18	89	0.21
Subtropical	44	0.23	47	0.47
Vineyards	51	0.22	63	0.31

[1] PNB = (N exported in harvest/N applied) * 100

[2] Cost ratio = Labor : Fertilizing Material

[3] Excludes alfalfa, pasture, rice, and sugarbeet

[39] Data sources used to calculate PNB of improved practices: Brown, P.H. (personal communication), Allaire-Leung et al. (2001), Hartz et al. (2000) Fritschi et al. (2005), Frate et al. (2008), Peacock et al. (1991), Christensen et al. (1994),Ali. & Lovatt. (1994), Saenz et al. (2001), Johnson et al. (2001), Hartz et al. (1993), Rosenstock et al. (2010), Southwick et al. (1995), Meyer and Marcum (1998), Bendixon (1997), Hartz et al. (1994), Hartz and Bottoms (2010) and Richardson and Meyer (1990).

Adminsitrative Record
Page 34839

3.5 Modeling Results

3.5.1 Policy Modeling

We modeled two different baseline crop mixes, one for the Tulare Lake Basin and one for the Salinas Valley, considering the so-called Nitrogen Hazard Index grouping as described in section 2.2.3 and 2.6 of this report. The hazard index is an indicator of nitrate leaching vulnerability based on soil characteristics, the crop grown, and the irrigation system utilized on a specific field. Similar approaches have been used to quantify vulnerability of groundwater in agricultural regions (Loague et al. 1996). We employed existing cost information in the SWAP model and information on the likely PNB and its cost before and after application of best management practices. The model calibrates for all selected crops and production factors to within 3% of the *observed* input values.

Four policies were modeled: 1) a 25 percent reduction in the total nitrate load to groundwater, 2) a 50 percent reduction in the total nitrate load to groundwater, 3) a tax on applied nitrogen of 7.5% and 4) a $2.00 surcharge per kg of nitrogen applied, for fields that have exceeded a maximum allowed leaching rate of 35 kg/ha/yr. Finally, we test the robustness of our approach by undertaking a sensitivity analysis of the marginal cost of improving nitrogen use efficiency. In previous work, tax or fee-based N reduction policies have been found to have the highest social costs (Helfand & House 1995); however, these uniform input taxes and regulations (same for all users) are close to the socially optimal solution when accurate pollution charges are hard to implement. We quantify the impact of uniform-type regulations for agricultural production in our study area.

3.5.2 Modeling Results

Our preliminary model results indicate a relatively mild economic adjustment response in terms of cultivated land at the 25% nitrate load reduction level. However, a 50% reduction from the current nitrate load across the entire region translates into higher production costs and some decreases in net revenues from farming, even assuming yields are maintained by improving nitrogen and irrigation efficiency. Figure 16 and Figure 17, below, show the percent change in measures of water and nitrogen use efficiency by scenario and region. Base water use efficiencies are on the order of 72% and 68% for all crops in the Tulare Lake Basin and Salinas Valley, respectively. Baseline nitrogen use efficiency for the crops analyzed is 51 for the TLB and 40 for the SV. Overall, the TLB has higher NUE than the SV. The Salinas Valley has a higher value crop mix; however, the TLB has a higher proportion of more nitrogen efficient crops such as corn, processing tomatoes, almonds, and pistachios.

Adminsitrative Record
Page 34840

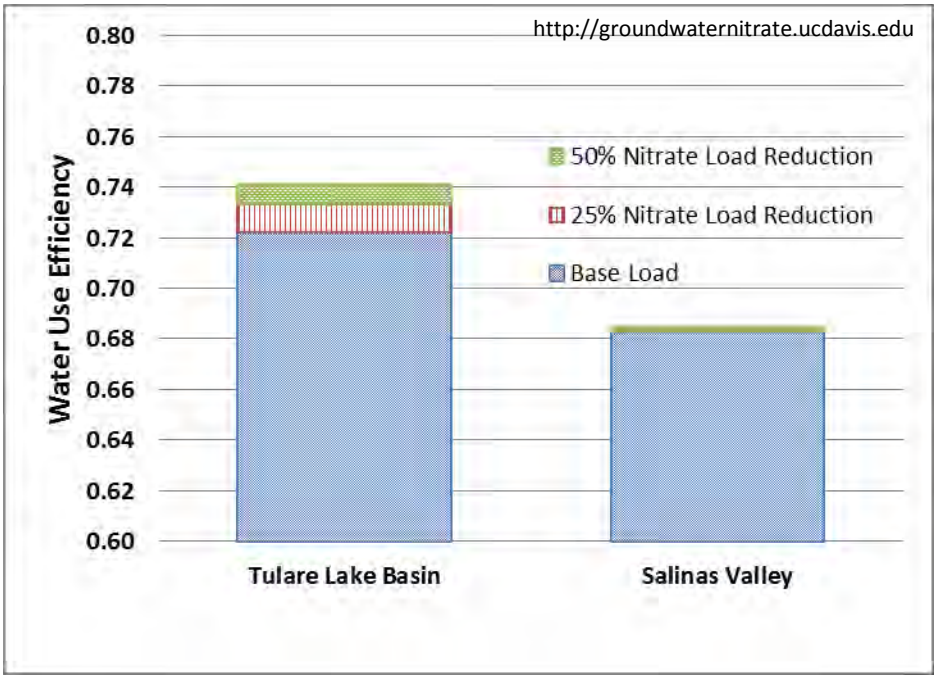


Figure 16. Water use efficiency for the Tulare Lake Basin and the Salinas Valley at different levels of reduction in nitrate load to groundwater.

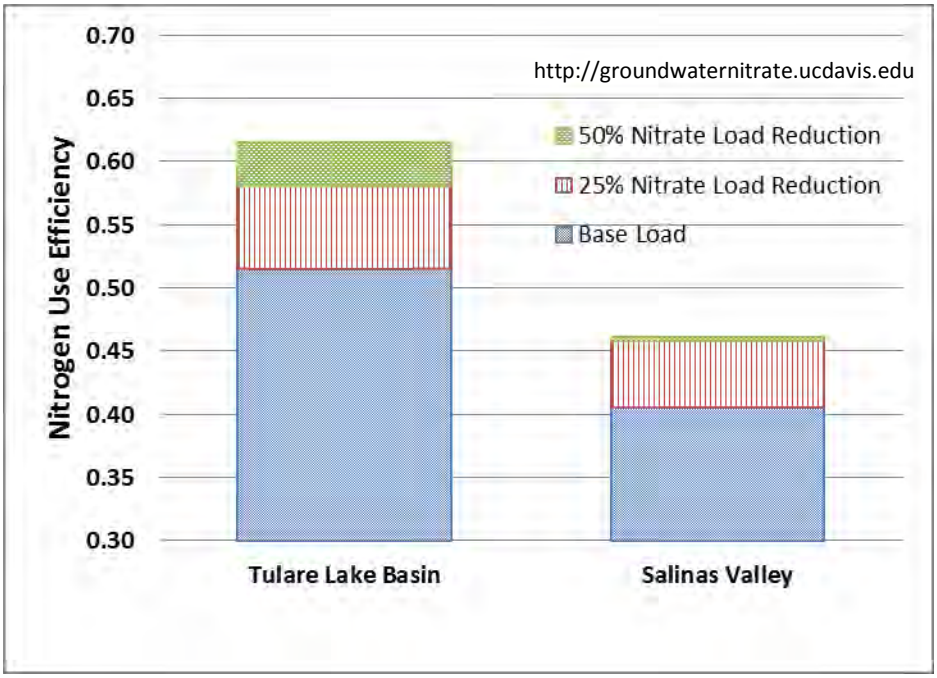


Figure 17. Nitrogen use efficiency for the Tulare Lake Basin and the Salinas Valley at different levels of reduction in nitrate load to groundwater.

Adminsitrative Record
Page 34841

Both water and nitrogen use efficiency increase with restrictions on total nitrate load to groundwater (Figure 16 and Figure 17, above). The marginal cost (not shown) of increasing irrigation efficiency is higher than the marginal cost for increasing nitrogen efficiency. Thus, the model allocates fewer resources to improve irrigation efficiency. To represent the interaction of water and nitrogen use efficiency on nitrate percolation to groundwater, we have constrained the model so that water efficiency is always greater or equal to nitrogen use efficiency.

Table 12 shows how applied water, applied nitrogen, land area, and net revenues would change with 25% and 50% nitrate load reductions. Reducing deep percolation loading by 25% can be achieved at relatively small costs in net farm revenue, assuming basic N management education is in place. However, net revenue losses increase at an increasing rate as greater reductions are sought. On average, in the TLB, a reduction of 3.6 metric tons of applied nitrogen for every 405 hectares (1000 acres) must be enacted to achieve a 25% decrease in nitrate load to groundwater.[40] For the Salinas Valley, this reduction is close to 5.6 metric tons per 405 hectares. For a 50% reduction in nitrate load to groundwater, the required reduction per 405 hectares increases to 5.2 metric tons for the TLB and 12.9 metric tons for the Salinas Valley. In the TLB, net revenue losses of 14% result from a 50% reduction in nitrate load to groundwater, which is four times the cost of a 25% reduction (3.5% loss in net revenue), with similar relationships in the SV. Figure 18, below, summarizes the relative changes in applied water, net revenues, irrigated land area, applied nitrogen, and nitrate load reductions for 0, 25 and 50 percent reductions in nitrate load.

Table 12. Groundwater nitrate load reduction scenarios and associated changes in total applied water, annual net revenues, irrigated land area, and applied nitrogen. The model keeps crop yields constant.

Region	Scenario	Applied Water km^3/yr [million AF/yr]	Net Revenues $2008 M/yr	Irrigated Land 1000 Ha [ac]	Applied Nitrogen Gg N/yr (%) [1000 t/yr]
Tulare Lake Basin	Base Load	10.5 [8.5]	4,415 (0%)	1,293 [3,194]	200 (0%) [221]
	25% load reduction	10.0 [8.1]	4,259 (-3.5%)	1,293 [3,064]	181 (-9%) [199]
	50% load reduction	7.9 [6.4]	3,783 (-14%)	952 [2352]	135 (-32%) [149]
Salinas Valley	Base Load	0.37 [0.30]	309 (0%)	92 [227]	18 (0%) [20]
	25% load reduction	0.33 [0.27]	285 (-7.5%)	83 [205]	15 (-16%) [17]
	50% load reduction	0.25 [0.20]	239 (-22%)	62 [153]	10 (-46%) [11]

[40] This is calculated from Table 12 as the difference in the ratio of total nitrogen applied (last column Table 12) to irrigated land area (fifth column) for each scenario.

Adminsitrative Record
Page 34842

At base load conditions average weighted PNBs of 0.51 and 0.40 are estimated for TLB and SV, respectively. If a 25% reduction in the nitrate load to groundwater is implemented, weighted average PNB increases to 0.58 and 0.44 for TLB and SV, respectively. Similarly, the ratio of applied nitrogen to effective nitrogen decreases under nitrate load to groundwater restricting policies. Conversely the ratio of investments in NUE to effective nitrogen, weighted by region, increase as the nitrate load to groundwater is restricted. Changes in these sets of ratios suggest crop farming adaptation to nitrate load reduction to groundwater policies by reducing applied nitrogen, increasing NUE via investments in NUE, reducing irrigated crop areas or switching to more N efficient and/or profitable crops.

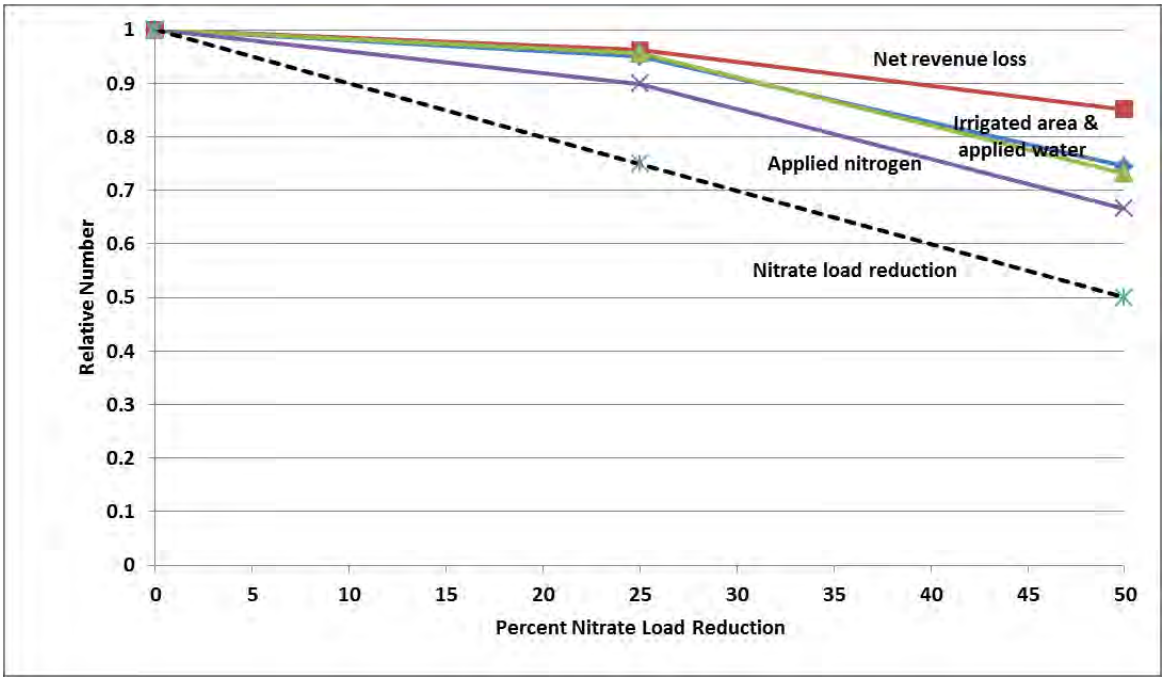


Figure 18. Relative applied water, net revenues, irrigated land, nitrogen applied, and nitrate load reduction as a function of reductions in the nitrate loading to groundwater by agriculture averaged over the Tulare Lake Basin and Salinas Valley.

Net percentage of revenue losses shown in Figure 19 below correspond to the net revenue in Table 12. Net revenue losses start increasing rapidly with larger reductions in total nitrate loading to groundwater. Reductions in total nitrate loading by 25% have an average cost of $8.1 per kilogram ($3.7/pound) of applied nitrogen in the TLB and $9.71 per kilogram ($4.4/pound) in the SV. When loading to groundwater is reduced by 50%, the average cost per kilogram of reduced nitrogen is approximately $9.7 per kilogram ($4.4/pound) for the TLB and $9.1/kg ($4.1/pound) for the SV.

Adminsitrative Record
Page 34843

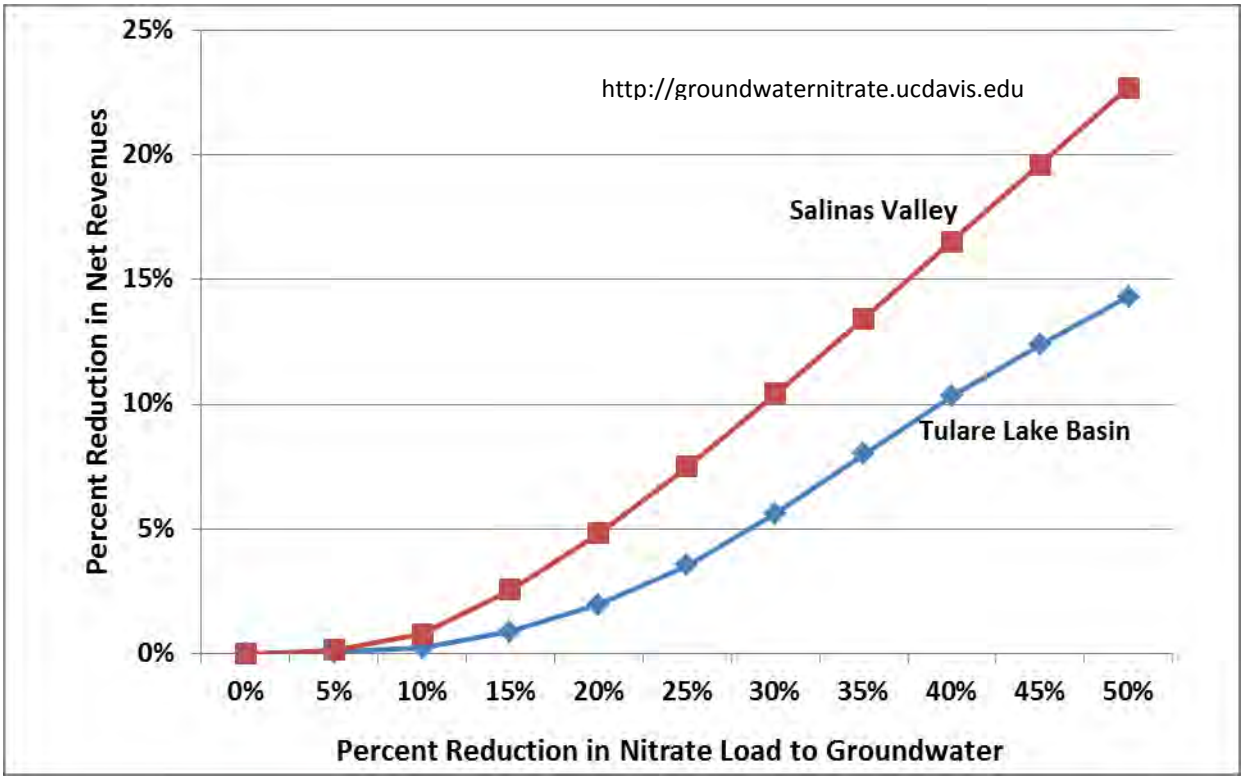


Figure 19. Percentage reduction in net revenues estimated from different levels of reduction in nitrate loading to groundwater.

Furthermore, Figure 20 below shows the marginal and average net revenue losses by kg of nitrate load reduction. As limits on nitrate leaching to groundwater increase, the average cost (as net revenue loss) per kg of nitrate load to groundwater increases. However, the marginal cost of nitrate load reductions to groundwater has two regions. In the first one, the marginal cost increases reaching a maximum, while in second one the marginal cost decreases and starts to flatten. This is analogous to stages of production in which a marginal product increases through stage 1, and is above average product but decreasing in stage 2 of production (Perloff 2004). This measure is useful for estimating costs of nitrate load limiting policies over the range analyzed in this study. For instance, the average net revenue loss per kg of nitrate load to groundwater is roughly $8/kg when the total nitrate load to groundwater is reduced by 25 percent. At this level, the marginal[41] net revenue loss per kg of nitrate load reduction is $18/kg, nearly twice as much as the average net revenue losses at 25% nitrate load reduction.

[41] The marginal net revenue loss per kg of nitrate load reduced to groundwater is the cost of an additional percent unit of reduction in the total nitrate load to groundwater.

Adminsitrative Record
Page 34844

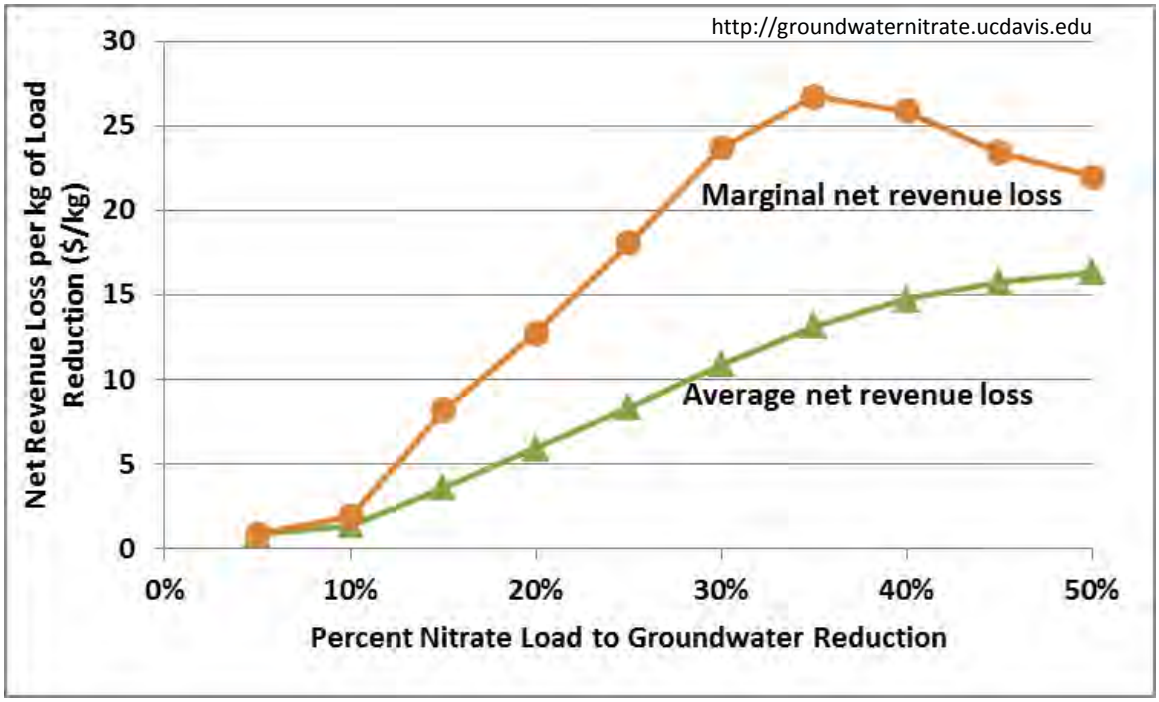


Figure 20. Average and marginal net revenue losses per kg of nitrate load reduction to groundwater.

We also present the resulting cropping pattern changes, from applying the two policies, in Figure 21 and Figure 22, below, for selected crops. At higher levels of reduction (e.g., 50%) cotton, corn, and other field and grain crops face the largest reductions in the Tulare Lake Basin. Likewise, the irrigated field and grain crops area is reduced in the SV, where higher value crops are grown. Irrigated area for high-value crops such as strawberries and lettuce remain about the same. However, vegetable crops as a group, due to their generally relatively low NUE, are subject to significant reductions in their irrigated crop area at highly restrictive nitrate loading policies. Similarly in the TLB, vegetable crops face significant reductions in their crop area when the total nitrate load to groundwater is required to be reduced by half.

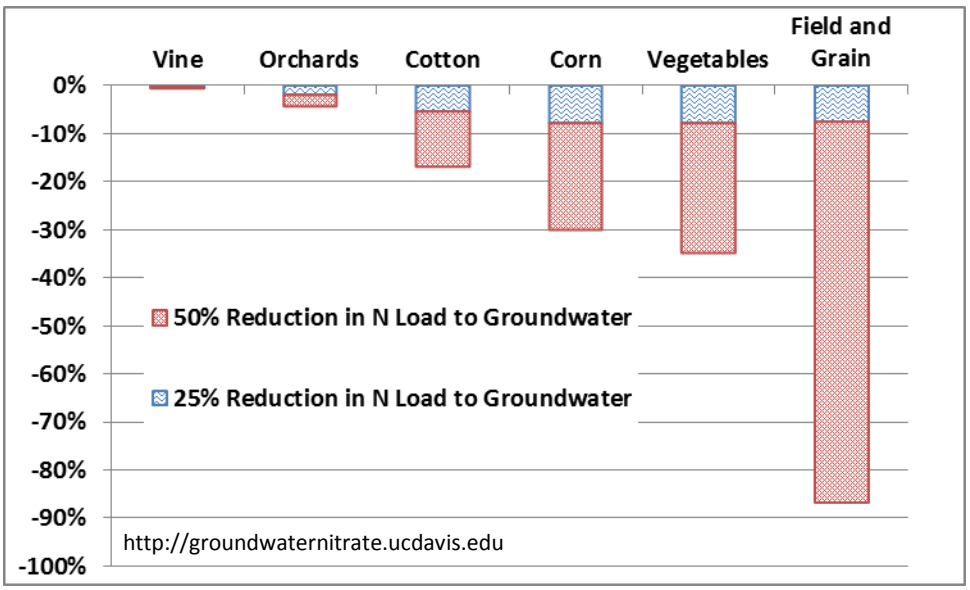


Figure 21. Cumulative change in cropping patterns with respect to base conditions for selected crops in the Tulare Lake Basin.

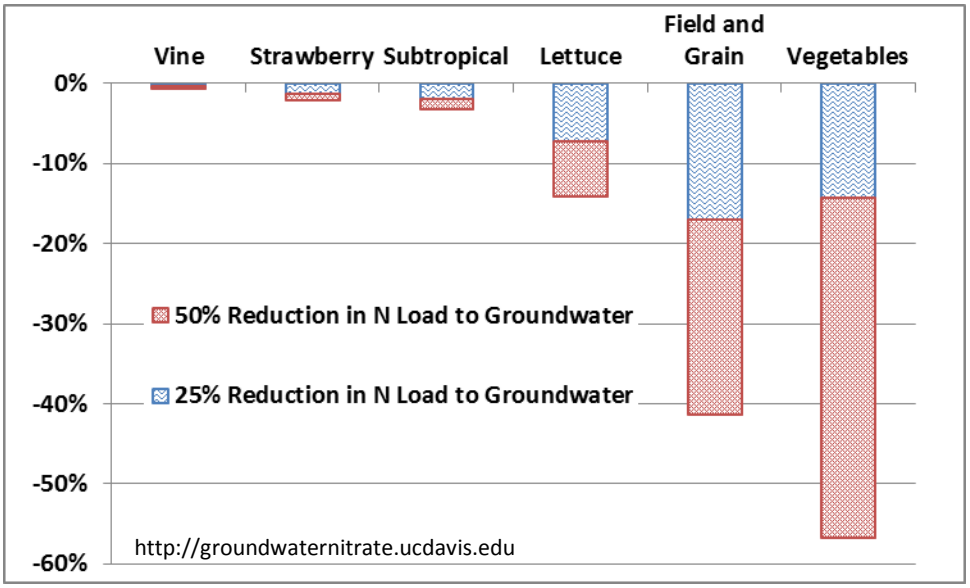


Figure 22. Cumulative change in cropping patterns with respect to base conditions for selected crops in the Salinas Valley.

Figure 23, below, illustrates the substitution between total applied nitrogen and investments in nitrogen use efficiency per unit area. The light shaded (blue) bars correspond to the percentage reduction of applied nitrogen per unit area, whereas the dark (red) shaded bars correspond to increases in the capital employed to produce the same crop mix per unit area. An interesting result is that the investment in nitrogen use efficiency improvements is maximized under the 25% reduction scenario, but is not increased with the 50% scenario for both study area regions.

Adminsitrative Record
Page 34846

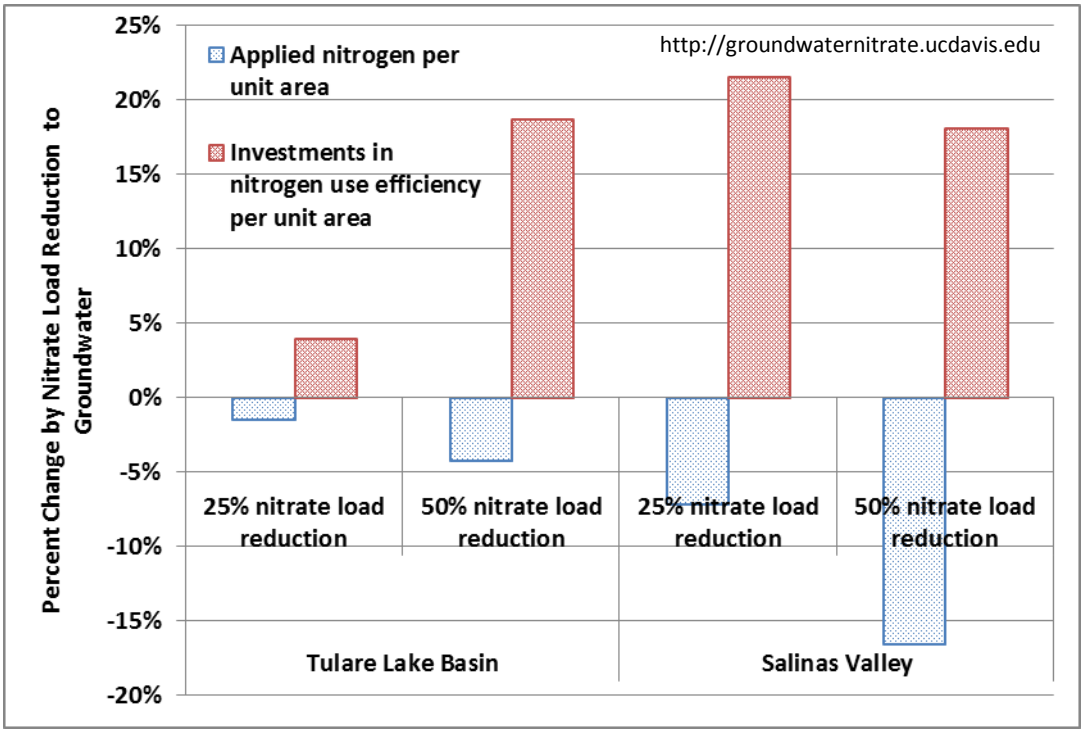


Figure 23. Average percent change, across all crops, in applied nitrogen and investment in nitrogen use efficiency per unit area with respect to current conditions.

3.5.3 A Tax on Nitrogen

Levying a tax on the use of nitrogen is one way to simultaneously reduce nitrogen use and to raise revenues to fund alternative water supplies. Currently the sale of nitrogen fertilizer is not subject to the California sales tax. The economic model is run under conditions that this tax relief is removed and the purchase of nitrogen is subject to the normal 7.5% California sales tax. Under this tax, the model predicts that farmers will respond in several ways to minimize the costs of the tax. There is a small difference in revenues and reductions in the levels of nitrogen applied in response to cost increase and are mostly offset by increases in investment in improving nitrogen use efficiency. Overall, nitrogen application is reduced by roughly 1.6% for both study area regions. Interestingly, the total irrigated acreage remains essentially unchanged. Figure 24 shows net revenue losses for both the TLB and the SV from zero to 50% tax on applied nitrogen. The trend is nearly linear for tax levels below 50% on applied nitrogen. Cropping patterns are maintained roughly constant with respect to base conditions. Net revenue losses for both the TLB and the SV from a sales tax policy of 7.5% are close to $29.4 million (0.6% of base net revenues).

Adminsitrative Record
Page 34847

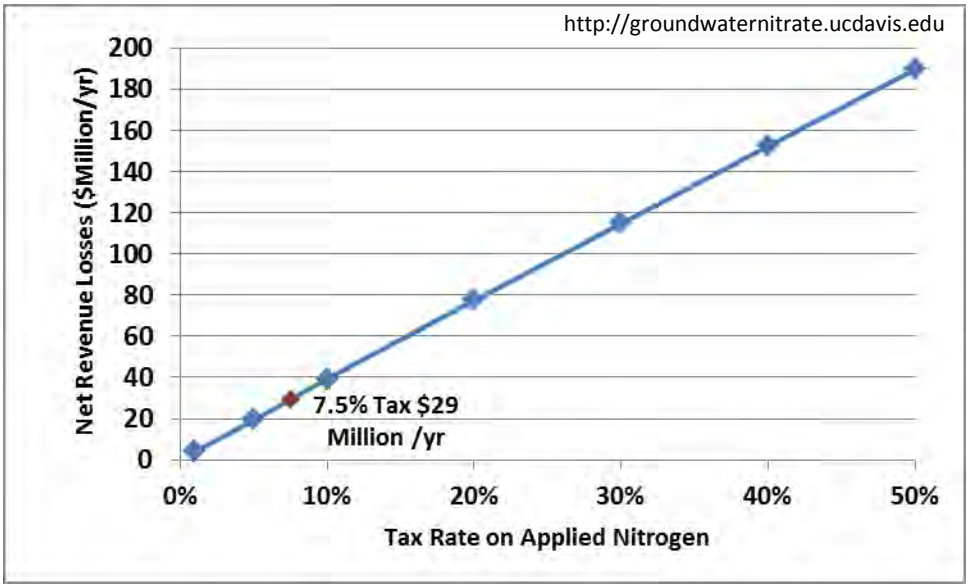


Figure 24. Net revenue losses for both the TLB and the SV per tax rate level.

3.5.4 A Penalty for Nitrogen Exceeding a Given Threshold

If a penalty of $2 per kg of N applied were set for any field exceeding a maximum leaching threshold of 35 kg/ha/yr (32 lb/acre/yr), irrigated crop area reductions are more evident in the model. For the TLB a 4.5% reduction in irrigated crop area may occur as a result of this threshold-based penalty. For the SV, this reduction can be as high as 5.6%. Total revenue losses can be 2.3% and 4.4% for the TLB and the SV, respectively, and are roughly in line with irrigated crop area reductions. Net revenue losses in this case exceed, on a percentage basis, land use reductions. Nearly 20% and 26% reductions in net revenues can be expected for the TLB and the SV, respectively. Thus, a high penalty on nitrogen application for crops with estimated loads to groundwater exceeding 35 kg/ha seems more effective in helping reduce total nitrogen application and load to groundwater. Reductions on the order of 3.5% in total revenues and 13% in the load to groundwater could be achieved; however, the net revenue loss can be high for this option.

3.5.5 A Sensitivity Test

One of the most questionable parameters is the marginal cost of improving nitrogen use efficiency. The sensitivity of the model results to this cost assumption is tested by doubling the cost in the initial model data. The model is then calibrated using the higher marginal cost and the same elasticities of substitution and supply as the base results model. When the 7.5% tax is coupled with the higher marginal cost for improving NUE, the difference in the results is interesting. The 7.5% tax results in higher reductions in total nitrogen applied of nearly 2.3% in the TLB, and 3.2% in the SV. Under these assumptions, the cost minimizing policy is to also reduce the total irrigated agricultural area by 2.3% in the TLB and 3.2% in the SV. The high cost of substituting a technology for improved nitrogen use efficiency makes it a cheaper alternative to reduce irrigated crop acreage of certain crops than to adopt

Adminsitrative Record
Page 34848

the efficiency enhancing practice. Crop area reductions are concentrated in field crops and corn, grown both in the TLB and in the SV, to less than 10% of the base cultivated land. Likewise, net revenues decrease by 10.4% in the TLB and 15.3% in the SV. For the nitrate load to groundwater restriction policies, at a 25% reduction, irrigated crop area decreases 10% for the TLB and 15% for the SV. This sensitivity analysis confirms our expectations: the cost at which substitution between capital for improving NUE and the resulting NUE is a critical parameter in determining both the cost and type of policy response of the model.

The range over which we can substitute best management practices for nitrogen use efficiency is critical to the costs of both policy scenarios. We had great difficulty in finding reliable measures of the ability to substitute in the agronomic literature. Additional research on this topic is required to more reliably model the cost of nitrogen reduction policies.

3.5.6 Limitations

Several limitations are worth mentioning in our modeling approach. First, aggregation of crops may bias crop farming response to nitrate load limiting policies in both directions. Second, the restriction that keeps yields constant will clearly over-estimate the cost of both limiting nitrate load and nitrogen cost policies, as higher NUE may result in higher yields and therefore increased gross revenues from farming (Hartz 1994). Third, carryover nitrogen (Knapp and Schwabe 2008) and crop rotation may influence multi-year cropping decisions, currently not captured in our modeling approach, which may result in cost over estimation of the policies modeled in our study. Finally, given California's market power for some specialty crops, irrigated crop area shifts may actually have an endogenous price effect which will influence production decisions that might also partially compensate the estimated losses to agriculture. A more comprehensive approach to capture price effects (Medellin-Azuara et al. 2012) could be used. With these limitations in mind, our approach proves to be useful in eliciting likely crop response and costs of nitrogen use efficiency management for California.

Adminsitrative Record
Page 34849

3.6 Conclusions

Consistent with the literature (Larson et al. 1996; Vickner et al. 1998; Knapp & Schwabe 2008), small reductions in nitrate leaching to groundwater from croplands can be made at relatively low costs. However, nitrogen-limiting policies that require larger reductions, on the order of half the current loads to groundwater, may have significant consequences on net farm revenues and cause significant reductions in total irrigated area. A sales tax on applied nitrogen may reduce total applied nitrogen and groundwater nitrate loading, with a very modest effect on net farm income. The response to policy measures is shown to be sensitive to both the cost of increasing nitrogen use efficiency and the range over which improved efficiency can substitute for applied nitrogen.

The cost of reducing nitrate loading to groundwater from irrigated crop farming appears to increase with reductions that are greater than 25%, but this will depend on the true costs of implementing efficiency improving management practices. Adjustments occur in three ways: 1) changes in nitrogen use efficiency, 2) changes in irrigation efficiency, and 3) changes in cropping patterns. In constructing the model, we found that the ability and cost of improving nitrogen use efficiency is exceptionally difficult to define quantitatively, given current agronomic studies and available data. As shown in our sensitivity analysis, the marginal cost of increasing nitrogen use efficiency is clearly the most critical parameter in terms of uncertainty, and should be the focus of additional empirical field studies such as those done for irrigation efficiency, before policies are based on results such as these.

Several conclusions arise from this work:

1) Modest increases in nitrogen use efficiency will increase production costs, but are unlikely to affect total irrigated crop area. Less than 4% of the total irrigated area and net revenues will be lost with such modest increases to NUE via improved management practices.
2) Larger reductions in excess nitrogen will be much more costly to implement and may ultimately lead to net reductions in irrigated crop areas, higher costs in terms of net revenue losses, and shifting cropping patterns towards more profitable and nitrogen efficient crops. In this case, more than 20% of the total irrigated grain and field crops area would be reduced.
3) A sales tax on applied nitrogen may slightly decrease total applied nitrogen with some loss in farm net revenues. A sales tax of 7.5% could help reduce applied nitrogen by nearly 2% under the modeling and cost assumptions developed here.
4) Larger than estimated marginal costs for increasing nitrogen use efficiency increases farming response to nitrogen limiting and tax policies. A two-fold increase in the marginal cost of improving nitrogen use efficiency results in net revenue losses of more than 14% in the TLB and 21% in the SV when total nitrate loading is limited by 25% of base values.

Despite the information uncertainty, due to poor understanding of costs for nutrient management and nitrogen use efficiency improvements, this analysis shows that it is fruitful and insightful to combine quantitative economic and agronomic data into a model that can reflect the cost differences in the level and location of reductions in groundwater nitrate load. This type of modeling will improve with better development of field data.

Adminsitrative Record
Page 34850

4 Reducing Urban Landscape Nitrate Loading

Prepared by: G. Stuart Pettygrove

Turfgrass is grown in parks, golf courses, athletic fields, and school grounds, and around residential and industrial buildings. In many of these locations, maintenance of a high visual quality (i.e., a green and healthy appearance) has significant economic value. High visual quality of turfgrass usually requires N fertilization; as the cost of N fertilizer materials and application is low compared to the economic value of the turf, there is potential for over application, resulting in leaching of N to groundwater.

Recent studies conducted by the University of California show that Nitrate leaching from the rootzone of well managed turfgrass is very low (Wu et al. 2007, 2010). In one study, leachate passing the roots of bermudagrass turf had a lower concentration of N than the water used for the irrigation, even though the turf received N fertilizer applications that are typical of resort golf courses (Wu et al. 2007). Low leaching N loss is due to the dense continuous plant canopy and root system of healthy turfgrass and its perennial growth habit, which allows for continuous N uptake over at least 8 months of the year depending on the species used. Furthermore, N applications at rates exceeding the minimum needed to maintain acceptable appearance simply results in "luxury uptake" by the turf and increased growth.

Poorly managed turf with a discontinuous canopy and presence of weeds, if it is growing on permeable soils and is overirrigated, presents a greater risk of Nitrate leaching loss, especially if fertilized at high rates during the dormant season of bermudagrass or similar tropical species of turfgrass. It is less likely that turfgrass in such condition in residential areas will be managed according to professional N fertilization guidelines; however, it also is less likely that in such situations, high rates of N fertilizer will be applied.

Therefore, an effective approach to limiting Nitrate leaching losses from urban turfgrass to very low levels is to follow standard N fertilization guidelines published by land grant universities and the golf course industry. These guidelines provide recommendations for N fertilizer rates in pounds N per 1000 square feet and for application timing. All such guidelines recommend somewhat smaller doses of N than often were recommended in the past. Also recommendations are for multiple small doses of N rather than single large doses. UC recommends more frequent, smaller applications to turf on soils having a low water holding capacity (i.e., on sandy soils). In some situations, the use of slow-release fertilizers will allow for reduced frequency of application, while maintaining turf quality; this type of product is in use on professionally managed turfgrass. An incentive for golf course managers and other landscape managers to use conservative N application rates is the less frequent mowing needed and smaller volume of clippings requiring disposal.

Turf N management guidelines suitable for both homeowners and professionally managed turf exist and can be accessed on the internet. Such guidelines are published by the University of California Integrated Pest Management Program (University of California Integrated Pest Management 2011) and by the Oregon Chapter of the Golf Course Superintendents Association of America (Oregon Golf Superintendents Association 2009).

Adminsitrative Record
Page 34851

5 Reducing Domestic and Urban Wastewater Nitrate Loading

Prepared by:

Vivian B. Jensen, Jeannie Darby, Nicole De La Mora, Thomas Harter, Aaron M. King

Acknowledgments

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Adminsitrative Record
Page 34852

5.1 Introduction

As potential nitrogen sources in the Tulare Lake Basin and Salinas Valley, wastewater treatment plants (WWTPs) and food processing facilities (FPs) were examined to:

- Assess their contribution to groundwater nitrate loading,
- Determine the regional and local impacts of nitrogen in discharge,
- Examine nitrogen control measures, and
- Propose solutions for nitrate loading reduction.

It is important to understand the dual nature of this discussion; wastewater treatment and food processing facilities can be sources of nitrogen and they can also be part of the solution. Potential sources of nitrate contamination from these facilities are:

- Sewer leakage due to aging/compromised infrastructure,
- Effluent from WWTPs and FPs discharged for irrigation and/or groundwater recharge, and
- Wasted solids from these facilities that are applied to land as a soil amendment.

A comprehensive assessment of potential nitrate loading from land applied discharge from WWTPs and FPs is provided in Technical Report 2, Section 6.2 (Viers et al. 2012). Also, Technical Report 2, Section 6.3 covers potential nitrate loading from sewer leakage. The below discussion is intended as an overview of options to prevent and reduce N loading from these sources.

Land application of effluent from these facilities can be an effective way to reuse water and nutrients, using natural processes in the soil and irrigated crops as a final stage of treatment. However, with inappropriate land application groundwater can be degraded. When discharges run the risk of negatively impacting groundwater, existing land application processes can be modified or facilities can be improved and potentially expanded to optimize operations and/or treat wastewater to a higher quality.

Septic systems in rural and peri-urban regions of the study areas are locally significant sources of nitrate leaching to groundwater (see Technical Report 2, Section 6.4, Viers et al. 2012). We discuss options for reduction of this source of nitrogen, as well.

Adminsitrative Record
Page 34853

5.2 Nitrogen Reduction from Wastewater Treatment and Food Processing Facilities

How can nitrate loading be reduced? How feasible are reduction options? What are the costs?

Land application of effluent from WWTPs and FPs can be an effective way to reuse water and nutrients, using natural processes in the soil and irrigated crops as a final stage of treatment. However, with inappropriate land application, groundwater can be degraded. When discharges run the risk of negatively impacting groundwater, existing land application processes can be modified or facilities can be improved and potentially expanded to optimize operations and/or treat wastewater to a higher quality.

Before considering the implementation of additional treatment processes to minimize nitrate loading to groundwater from land applied discharge, adjustment to land application practices should be considered. With discharge application at appropriate rates, proper onsite storage, and monitoring to avoid groundwater degradation, land application may be the best waste management option available for certain facilities.

As discussed previously (see Technical Report 2, Section 6.2, Viers et al. 2012), land treatment methods can be categorized into three main types: Slow Rate (SR), Overland Flow (OF), and Soil Aquifer Treatment (SAT)/Rapid Infiltration (RI) (Crites, Reed, & Bastian 2000; United States Environmental Protection Agency 2006). SR and SAT/RI are most pertinent to our analysis. For food processing facilities, with the selection of the most appropriate land treatment method based on site and water quality characteristics, the high cost of disposal at publicly-owned treatment works (POTW) can be avoided and the need for onsite treatment systems may be minimized. Low strength wastewaters may only require screening before land application; however, high strength waste streams must be treated prior to discharge. To ensure the protection of drinking water sources, the appropriate measures must be determined and implemented based on the needs and characteristics of individual facilities.

5.2.1 Nitrogen Reduction – Methodology

A brief literature review of available options to reduce nitrate loading to groundwater from WWTPs and FPs was conducted including both treatment and non-treatment options. Cost information was collected from published literature and case studies. Cost information can be used to assess the feasibility of options to reduce nitrate loading from WWTPs and FPs relative to the options available to address more significant sources (e.g., fertilizer). Treatment information was collected from WWTPs in the study area (top 90% of flow based on design flow, see Technical Report 2, Section 6.2 (Viers et al. 2012) for additional information), supplementing a survey of WWTPs performed by the State Water Board across CA (State Water Resources Control Board 2010).

A list of at-risk WWTPs and FPs (i.e., WWTPs which may need to adjust land application practices and/or include nutrient removal in the treatment process) was developed, based on the collected nitrate

Adminsitrative Record
Page 34854

loading data (Technical Report 2 Section 6.2, Viers et al. 2012). For high demand crops, a rough estimate of required nitrogen is 250 kg/ha/yr (~225 lb/acre/yr), or 500 kg/ha/yr (~450 lb/acre/yr) for double cropping.[42] Facilities exceeding this application rate risk contributing to nitrate contamination of groundwater. This is important both regionally and locally to pinpoint hot-spots and locate facilities that may require additional treatment or altered land application practices. Facilities annually discharging total N greater than 250 kg/ha/yr (~225 lb/acre/yr) to agriculture and/or discharging effluent having total N concentrations greater than the MCL to percolation basins are included in the list. Facilities annually discharging total N greater than 500 kg/ha/yr (~450 lb/acre/yr) were also investigated to account for the possibility of double cropping. Treatment costs for nutrient removal were gathered from U.S. EPA documentation (United States Environmental Protection Agency 2008) and utilized to estimate costs for nitrate loading reduction from at-risk WWTPs and FPs.

5.2.2 Reducing Nitrogen Loading from Wastewater Treatment Plant Discharge – Nutrient Removal

"Conventional biological treatment processes designed to meet secondary treatment effluent standards typically do not remove total nitrogen (TN) or total phosphorus (TP) to an extent sufficient to protect certain receiving waters" (United States Environmental Protection Agency 2008). However, nutrient removal in wastewater treatment has become increasingly prevalent over the past 30 years. Treatment options for nutrient removal from wastewater are thoroughly described in the literature, with an abundance of material in engineering textbooks and state and federal guidance manuals/publications (Metcalf & Eddy 2003; United States Environmental Protection Agency 2008; Water Environment Federation 2010). The U.S. EPA guidance manual (2008) is a comprehensive resource describing available relevant technologies, their reliability, feasibility, and costs, based on case studies of full scale WWTPs.

Nitrogen removal from wastewater can be accomplished using a variety of technologies and configurations; both biological and physical/chemical processes are effective. The selection of the most appropriate treatment option is dependent on many factors including:

- Influent wastewater quality characteristics
- Required capacity
- Effluent discharge limitations
- Financial constraints
- Operations and Maintenance (O & M) demands
- Site limitations (land area, temperature, etc.)
- Electron donor source
- Future need for expansion

With many potential configurations to achieve nitrification or both nitrification and denitrification, biological nutrient removal is typically categorized as tertiary or advanced treatment and can be incorporated into the biological processes of secondary treatment (Metcalf & Eddy 2003). Additional

[42] This is a rough estimate for high demand crops and is based on crop nitrogen demand for single and double cropping as discussed in Technical Report 2, Section 3 (Viers et al. 2012).

Adminsitrative Record
Page 34855

methods used for nitrogen removal include chemical oxidation, air stripping, and ion exchange (Metcalf & Eddy 2003).

Biological nutrient removal (BNR) is accomplished through the provision of optimal conditions for the activity of various species of bacteria (Table 13). Through biologically mediated transformation processes, influent organic nitrogen and ammonia are converted to nitrate and then to nitrogen gas. Optimal conditions are process dependent and different BNR configurations can be designed to facilitate each stage. For example, nitrification is an aerobic process, requiring sufficient oxygen, while denitrification requires anoxic conditions and an electron donor (wastewater is likely sufficient).

Table 13. Treatment processes and associated bacterial species in wastewater treatment for nitrogen removal (adapted from United States Environmental Protection Agency 2008).

Nitrogen Transformation	Process	Bacteria
Organic nitrogen → NH_4^+	Ammonification	Various
$NH_4^+ + 3/2O_2 + 2HCO_3^- \rightarrow NO_2^- + 2H_2CO_3 + H_2O$	Nitrification (1)	*Nitrosomonas*
$NO_2^- + ½\ O_2 \rightarrow NO_3^-$	Nitrification (2)	*Nitrobacter*
NO_3^- + organic carbon → $N_{2(g)} + CO_{2(g)} + H_2O + OH^-$	Denitrification	Denitrifiers

Denitrifying bacteria require an electron donor (substrate) for the reduction of nitrate to nitrogen gas. In conventional wastewater treatment, substrate addition is not typically needed, because the wastewater contains sufficient carbon for denitrification to occur. However, depending on source water, substrate addition may be required. Feed water composition may need to be further augmented with the addition of nutrients required for cell growth (phosphorus for example). Autotrophic bacteria utilize sulfur or hydrogen as an electron donor and inorganic carbon (typically carbon dioxide) as a carbon source for cell growth (Eqns. 14 and 15), while heterotrophic bacteria consume an organic carbon substrate, such as methanol, ethanol, or acetate (Eqn. 16) (Mateju et al. 1992; Kapoor & Viraraghavan 1997).

$$11S^0+0.5CO_2+10NO_3^-+2.54H_2O+1.71NH_4^+ \rightarrow 0.92C_5H_7O_2N+11SO_4^{2-}+5.4N_2+9.62H^+ \quad \text{(Eqn. 14)}$$

$$H_2 + 0.35\ NO_3^- + 0.35\ H^+ + 0.052\ CO_2 \rightarrow 0.010\ C_5H_7O_2N + 0.17\ N_2 + 1.1\ H_2O \quad \text{(Eqn. 15)}$$

$$1.08\ CH_3OH + NO_3^- + H^+ \rightarrow 0.065\ C_5H_7O_2N + 0.467\ N_2 + 0.76\ CO_2 + 2.44\ H_2O \quad \text{(Eqn. 16)}$$

Various species of bacteria are responsible for denitrification including *Thiobacillis denitrificans, Micrococcus denitrificans, Pseudomonas maltophilia* and *Pseudomonas putrefaciens* (Kapoor & Viraraghavan 1997).

BNR processes fall into three major categories: suspended growth, biofilm, and hybrid systems (Water Environment Federation 2010). In suspended growth systems (e.g., activated sludge), "bacteria are kept in suspension under appropriate conditions to allow them to grow and consume pollutants from the

Adminsitrative Record
Page 34856

water" (Water Environment Federation 2010 p. 65). In contrast, biofilm systems encourage surface growth of bacteria (e.g., trickling filters and moving bed bioreactors (MBBRs)). Sample configurations of BNR are listed in Table 14 (United States Environmental Protection Agency 2008; Water Environment Federation 2010).

Table 14. Sample configurations and processes for biological nutrient removal in wastewater treatment (United States Environmental Protection Agency 2008; Water Environment Federation 2010).

• Activated Sludge (see cyclical aeration, fixed film)	• MBR (In-tank vs. external tank)
• Moving Bed Bioreactor (MBBR)	• Alt MBBR
• Effluent Filter/Denitrifying Filter	• Fluidized Bed
• Ludzack-Ettinger	• Modified Ludzack-Ettinger
• Four-stage Bardenpho	• Sequencing Batch Reactor (SBR)
• ANAMMOX	• SHARON
• InNitri	• Biodenitro
• BABE	• Oxidation Ditch
• Trickling Filter	• Biologically Activated Filters
• Wuhrman	• Step Feed Activated Sludge
• Simultaneous NDN	• SHARON-ANAMMOX
• Schreiber countercurrent aeration process	• Constructed Wetlands

While there are many configurations and types of systems, all BNR processes for nitrogen removal require an aerobic nitrification zone and an anoxic denitrification zone. The Municipal Nutrient Removal Technologies Reference Document (United States Environmental Protection Agency 2008) includes a decision matrix to facilitate technology selection based on key factors including site, water quality, and operation considerations.

Physico-chemical treatment options (Table 15) include breakpoint chlorination, gas stripping, ion exchange, and membrane processes (reverse osmosis (RO) and electrodialysis reversal (EDR)). While technologically feasible, the high costs of physico-chemical processes generally limit application of these options for nitrogen removal from wastewater (United States Environmental Protection Agency 2008). Also, treatment processes resulting in a brine waste stream can be particularly problematic for inland regions like the Tulare Lake Basin, due to the expense of brine disposal.

Adminsitrative Record
Page 34857

Table 15. Physico-chemical technologies for nutrient removal from wastewater (adapted from Water Environment Federation 2010).

Breakpoint Chlorination	Ammonia Stripping
Ion Exchange	Chemical and Catalytic Denitrification
Membrane Processes (RO and EDR)	Various Media (Bauxite/Zeolite)

For details on the WWTPs in the study area, with respect to treatment, including a discussion of treatment costs for facilities with greater contributions to nitrate loading, see Section 5.2.5 *Reduction of Nitrogen Loading from Wastewater Treatment Plants and Food Processors in the Tulare Lake Basin and Salinas Valley*, below.

5.2.3 Reducing Nitrate Loading from Food Processor Discharge

There are several options for avoiding the installation of an on-site treatment system for nitrogen removal at FP facilities:

- Optimization of food processing practices to moderate nitrogen levels in discharge or minimize waste volumes requiring disposal,
- Blending of low and high strength waste streams to meet permit limits,
- Reuse of process water to decrease total waste flow,
- Consideration of alternative disposal options, including deep well injection and POTWs,
- Use of discharge for irrigation of agricultural crops rather than direct groundwater recharge through percolation ponds to reduce nitrate loading to groundwater,
- Implementation of appropriate measures to avoid leaching of groundwater from storage vessels prior to land application (lining of storage ponds),
- Modification of land application practices to limit nitrogen application rates to levels less than or equal to plant uptake rates by decreasing flow rate, decreasing nitrogen levels, or increasing acreage.

When alternative options have been eliminated, treatment options for nitrogen removal from FP wastewater are similar to those described above for WWTPs. Additional information is available in the literature; (Isosaari, Hermanowicz, & Rubin 2010) provide a comprehensive discussion of natural wastewater treatment options pertinent to waste management in food processing facilities. Additionally, the Hilmar SEP report includes a detailed discussion of treatment options for food processing wastewater; the primary focus of the Hilmar analysis is salinity, but nitrate is considered as well (Rubin et al. 2007; Sunding et al. 2007; Sunding & Berkman 2007).

Details on the FPs in the study area, with respect to treatment, including a discussion of treatment costs for facilities with greater contributions to nitrate loading, follow in the next section.

Adminsitrative Record
Page 34858

5.2.4 Reduction of Nitrate Loading from Wastewater Treatment Plants and Food Processors – Treatment Costs

When options for in-plant optimization and improvements to land application practices have been exhausted, N loading can be reduced through the implementation of additional treatment measures. The Municipal Nutrient Removal Technologies Reference Document (United States Environmental Protection Agency 2008) includes a comprehensive cost analysis of retrofitting and expansion for BNR. Based on case studies developed by the U.S. EPA, "the costs for modification ranged from a low of $0.20 to a high of $5.25 per gpd capacity" (United States Environmental Protection Agency 2008 pp. 4–34). Costs varied based on facility size, water quality parameters, treatment level, and process; overall costs per unit capacity are higher for small systems. Costs were modeled and cost curves were developed based on process and system capacity.

Expansion of facilities refers to "a parallel train and no increase in design flow" while retrofitting facilities includes the adaptation of existing facilities and is therefore less expensive (United States Environmental Protection Agency 2008). Table 16 lists BNR costs for multiple configurations; costs were averaged over three system sizes: 1, 5, and 10 mgd (United States Environmental Protection Agency 2008). Life cycle costs were calculated, "by first annualizing the capital cost at 20 years at 6 percent interest. The annualized capital cost was then added to the annual O&M cost to obtain total cost. This cost was then divided by the annual flow to get the life-cycle cost per million gallons (MG) treated" (United States Environmental Protection Agency 2008 p. ES–17).

Although the Hilmar SEP report includes a detailed cost analysis of treatment options for food processing facilities, the primary focus of that investigation is salinity. The costs and technologies for treating high salinity waters are different from those for nitrate treatment. Salt loading is a significant problem in parts of California and the results of the Hilmar SEP study provide an extensive volume of information. The costs listed for biological treatment are pertinent to this discussion. Costs associated with the discharge of food processor effluent to the Tulare WWTP, provided as an estimate of the costs of biological treatment, are $1,070/MG ($1.07/kgal) (Sunding et al. 2007 p. 571).

Adminsitrative Record
Page 34859

Table 16. Costs of BNR for example configurations with design capacity ranging from 1 to 10 mgd (United States Environmental Protection Agency 2008).

Process	Capital Cost ($/gpd capacity)	O&M Cost ($/MG treated)	Life Cycle Cost ($/MG treated)
Retrofit			
Denitrifying Filter[1]	1.00	303	543
Step Feed[2]	0.83	105	303
MLE[3]	0.88	92	275
Extra Basins for PID[4]	0.52	65	190
Expansion			
Denitrifying Filter[1]	1.00	303	543
Step Feed[2]	1.73	383	792
MLE[3]	1.97	408	883
Extra Basins for PID[4]	0.79	145	335
SBR[5]	2.40	398	970

[1] Denitrifying filter with a total N target of 3 mg/L.
[2] Step feed configuration with retrofit taken as 1/3 of step feed, with a total N target of 5 mg/L.
[3] Modified Ludzack-Ettinger process with retrofit taken as 1/3 of MLE and total N target of 5 mg/L.
[4] Extra basins for PID (phased isolation ditch) with a total N target of 5 mg/L.
[5] SBR (Sequencing Batch Reactor), with a total N target of 5 mg/L, provided for comparison with Tulare WWTP.

5.2.5 Reduction of Nitrate Loading from Wastewater Treatment Plants and Food Processors in the Tulare Lake Basin and Salinas Valley

Fifteen percent of the WWTPs included in this study reported the use of some form of nutrient removal to decrease effluent nitrogen levels. For reference, total annual nitrogen levels (including nitrate-N, nitrite-N, ammonia-N, and organic-N) discharged from facilities with treatment for nutrient removal average 12.54 mg/L N, while the average across all other facilities is 16.65 mg/L N. It is important to keep in mind, however, that WWTPs treating to remove nitrogen in effluent are doing so because of high nitrogen levels.

A list of at-risk WWTPs and FPs (i.e., facilities which may need to adjust land application practices and/or include nutrient removal in the treatment process) was developed, based on the collected nitrate loading data (Technical Report 2, Section 6.2, Viers et al. 2012). Facilities discharging an annual total N greater than 250 kg/ha/yr (223 lb/ac/yr) to agriculture and/or discharging effluent having nitrogen concentrations greater than the MCL to percolation basins are included in the list.[43] Facilities discharging an annual total N greater than 500 kg/ha/yr (446 lb/ac/yr) are also listed to account for the possibility of double cropping.[43] It is important to note that this analysis is not suggesting that all of these facilities are contributing to groundwater nitrate contamination; groundwater monitoring data would be necessary to determine the relative impact (if any) of discharges from these facilities on

[43] This is a rough estimate for high demand crops and is based on crop nitrogen demand for single cropping as discussed in Technical Report 2, Section 3 (Viers et al. 2012).

Adminsitrative Record
Page 34860

groundwater nitrate levels. However, based on the assumptions herein, the facilities included in Table 17 are the top contributors to applied nitrogen of all the facilities included in this investigation.

Table 17. Number of facilities with > 250 kg N/ha/yr and > 500 kg N/ha/yr in discharges to irrigated agriculture and > 10 mg/L N in discharges to percolation basins.

	WWTP Irrigation	WWTP Percolation	FP Irrigation	FP Percolation
By County	# > 250 kg N/ha/yr (# > 500 kg N/ha/yr)*	# > 10 mg/L N	# > 250 kg N/ha/yr (# > 500 kg N/ha/yr)*	# > 10 mg/L N
Fresno	0 (0)	8	6 (2)	11
Kern	4 (2)	2	3 (1)	5
Kings	0 (0)	2	1 (0)	0
Tulare	3 (1)	7	2 (1)	10
Monterey	1 (0)	2	0 (0)	2
By Basin				
TLB	*7 (3)*	*19*	*12 (4)*	*26*
SVB	*1 (0)*	*2*	*0 (0)*	*2*
Overall Count**	**8 (3)**	**21**	**12 (4)**	**28**

Note: Solids not included.
*To allow for double cropping.
**Total # Facilities: 24 WWTP, 39 FP. Some facilities discharge to both irrigated agriculture and percolation ponds.

Based on wastewater flow, capital and O&M costs associated with treatment for nutrient removal for the at-risk facilities listed in Table 17 can be estimated. As mentioned above, wastewater treatment costs vary with numerous factors; these numbers are intended as a rough guide and actual costs may vary significantly from those listed here. These costs have been estimated based on system flow and the above cost information reported for biological nutrient removal in wastewater treatment (Table 16). Available BNR costs (United States Environmental Protection Agency 2008) for 1, 5, and 10 mgd treatment systems are used here as an approximation of actual costs. Many of the food processing facilities have very low wastewater flows (<< 1 mgd); BNR capital costs are therefore expected to be significantly underestimated for these low flow systems lacking economies of scale. In practice, these facilities would likely consider modification of land application practices or in-plant optimization, rather than installation of a small scale BNR system. Similarly, facilities listed above as exceeding application levels for irrigated agriculture are likely to first consider addressing groundwater degradation risk through changes in land application practices. Costs were estimated only for those facilities with land application (irrigation or percolation) of discharges having nitrogen concentrations above 10 mg/L as N. This includes all at-risk facilities listed above (24 WWTPs and 39 FPs) as some facilities discharge to both irrigated agriculture and percolation ponds.

Total capital costs for BNR treatment of all wastewater (FPs and WWTPs) from the above listed at-risk facilities range from $70 – $135 million and $107 – $266 million, for retrofit and expansion projects,

Adminsitrative Record
Page 34861

respectively. Associated annual O&M costs range from $3.2 – $15 million and $7.2 – $20 million, for retrofit and expansion projects, respectively. Table 18 lists estimated cost information for BNR by basin for at-risk WWTPs and FPs.

Table 18. Estimated costs of BNR (U.S. EPA 2008) for at-risk facilities, based on wastewater flow.

# Facilities: 24 WWTP, 39 FP	WWTP Retrofit	WWTP Expansion	FP Retrofit	FP Expansion
Total by Basin				
TLB Capital Costs ($ x10^6)	62.0 – 119.2	94.2 – 234.8	5.5 – 10.6	8.4 – 20.9
TLB Annual O & M ($ x10^6)	2.8 – 13.2	6.3 – 17.8	0.252 – 1.2	0.561 – 1.6
SVB Capital Costs ($ x10^6)	2.7 – 5.3	4.2 – 10.4	0.035 – 0.067	0.053 – 0.132
SVB Annual O & M ($ x10^6)	0.125 – 0.582	0.279 – 0.785	0.002 – 0.007	0.004 – 0.010
Overall Total				
Capital Costs ($ x10^6)	64.7 – 124.5	98.3 – 245.2	5.6 – 10.7	8.4 – 21.0
Annual O & M ($ x10^6)	3.0 – 13.8	6.6 – 18.5	0.25 – 1.2	0.57 – 1.6

An additional, more general strategy for the reduction of nitrate loading from these sources is simply to improve management of water quality data. Groundwater monitoring is required for many of these facilities; however, the data are largely unavailable since they are not in a digital format. To improve monitoring, enforcement, and abatement efforts related to these facilities, groundwater data need to be centrally managed and organized digitally.

Adminsitrative Record
Page 34862

5.3 Methods and Cost of Sewer Leakage Prevention and Reduction

A comprehensive assessment of potential nitrogen leakage from sewer systems is provided in Technical Report 2, Section 6.3 (Viers et al. 2012). Here we briefly discuss potential prevention and reduction options.

The main causes of sewer exfiltration are outdated and poorly fitted pipes, according to the majority of interviewed county employees. Surveys of city and county personnel indicated that sewage system leakage is generally considered a minor problem within the study area, although some survey participants mentioned particular sources of pipe leakage: Concrete pipes were considered problematic due to corrosion from hydrogen sulfide gas. Old vitrified clay pipes (VCP) pipes are burdened with numerous cracks and leaks, and a U.S. EPA study confirms that the sewers most susceptible to exfiltration are old VCP pipes (Amick & Burgess 2000). Poorly fitted pipes were cited as a potentially greater cause of leakage than any particular piping material. While identifying problematic piping material is important, the age of the piping system is also crucial. The oldest neighborhoods (primarily in the downtown areas) are most likely to exhibit leakage problems. However, proper sewer maintenance is an important process for sewers of all ages in preventing exfiltration.

Replacing outdated or inadequate pipe materials will decrease sewer exfiltration. Although many piping materials are suitable for sewage transport and are adequate replacements for older pipes, the longevity of the chosen replacement piping material is an important consideration. Pipes vary greatly in cost depending on material. Typical costs of some gravity pipes are as follows: $10.30 per linear foot for PVC, $14.50 per linear foot for concrete, $50 – $55 per linear foot for VCP, and $6.85 per linear foot for HDPE (Cutter 2009). Pressure pipes tend to be more expensive, as they are often thicker to withstand greater pressures (Cutter 2009).

Further research is needed to adequately evaluate potential costs associated with addressing sewer leakage in the study area, a source of groundwater nitrate for which few data exist (see Technical Report 2, Section 6.3 in Viers et al. 2012).

Adminsitrative Record
Page 34863

5.4 Reducing Nitrogen Leaching from Septic Systems

Septic systems in the study area are shown to have a small but important impact on overall groundwater nitrate loading/leaching. In certain locales, generally near the outskirts of larger cities, these systems are leaching nitrogen at rates similar to that of irrigated agriculture. An important distinction between this source and the agricultural source, is that septic systems do not require high concentrations of dissolved nitrogen to function properly – they merely discharge nitrogen as a byproduct. Therefore, it is possible to take measures to reduce or eliminate this nitrogen source without concern for a negative impact on the local economy. Septic tanks are, in fact, a low-hanging fruit in some areas.

Two methods of nitrogen reduction are presented below: source-separation and post-septic tank biological denitrification.

5.4.1 Source Separation Technology for Nitrogen Reduction

Segregation of urine from wastewater can be accomplished using separating toilet technology or urinal fixtures. Urine-separating toilets divert urine to a storage tank for processing, removing it from the septic (or sewer) waste stream. Urine-separation toilets have been approved for use in California. The California Department of Transportation has recently installed a separate storage tank for diversion of waste from ultra-low flow urinals at a highway safety roadside rest area in Southern California. The urine diversion was implemented to reduce the size and improve the performance of the onsite wastewater treatment system (anoxic treatment wetland). Based on model estimates, the urine diversion is expected to reduce the nitrogen loading to the treatment system by about 50 percent.

Urine separation toilets have been investigated extensively in several European and African countries, both in laboratory and in field (household) settings (Johansson 2000; Vinneras 2001; Jönsson 2004; Muench 2009; Rossi et al. 2009). The urine separation is accomplished generally with a split bowl, with urine collection in the front and fecal and solid wastes collected in the back.

Performance

Around 80% of the nitrogen (as well as 50% of the phosphorus and 90% of the potassium) in human waste is contained in the urine (Vinneras 2001). Given a known loading rate and assuming a 75% recovery rate (Rossi et al. 2009), the leaching rate of a system can be calculated. If one considers, for example, a septic nitrate loading/leaching rate of 66 kg N/ha/yr (59 lb/ac/yr) as seen in some of the areas northeast of Fresno (see Technical Report 2, Viers et al. 2012), the same region with urine-separating toilets would have a nitrate loading/leaching rate of (66 x 0.2) + (0.25 x 0.8 x 66) = 26.4 kg N/ha/yr (23.5 lb/ac/yr).

Adminsitrative Record
Page 34864

Pathogens

While urine itself is not generally considered pathogenic (Björklund 1999; Vinneras 2001), cross-contamination with feces poses a risk (Vinneras 2001). However, simply storing urine at 20°C (room temperature) for 6 months is sufficient treatment to produce fertilizer safe for any crop (Höglund 2002). The storage conditions allow for near complete conversion of urea-nitrogen to ammonia, resulting in a product that is sterile. With average soil temperatures below 1-meter depth around 16-18°C in most of the study area, only a small amount of energy would be expended in maintaining the required temperature in these tanks.

Economics

These toilets cost between $150 and $1300 depending on the level of sophistication of the technology (based on review of prices from the following companies: Separett[44] and Ecovita[45]). Some urine-separating toilets are also composting toilets, designed to produce compost from the solids portion. Other urine-separation toilets are designed to send the solids to the sewer or septic waste system, some with reduced flush-water volumes. Other costs associated with the technology include dual plumbing systems downstream of the toilet, storage tank costs, maintenance, pumping, heating, and transport costs (where applicable).

Dual plumbing systems are required from the toilet to the storage tank. A typical cost for such a renovation would range from $2,000 for simple systems with bathrooms already on an outside wall to $15,000 for more complex systems involving multiple stories, multiple bathrooms, or longer transport to storage tanks through existing structure (walls). To properly treat the stored urine, two 6-month storage tanks would be required. Assuming a reasonable value for urine excretion for humans, 0.6 to 1.2 L/day (0.15 to 0.3 gallons per day) (Schouw et al. 2002), and a household size of 3.5 individuals, a 6-month storage tank should be on the order of 800 liters capacity. The cost of two such tanks ($400 each for HDPE) plus excavation for underground emplacement ($500) and a low-power warming system ($500) to maintain 20°C amounts to about $1,800. Assuming an additional $200 per year of maintenance cost (pumping and electrical heating), partially offset by sale of the fertilizer product, total cost for a system, including tank, dual plumbing, toilet, and maintenance, annualized at a 5% discount rate over 20 years is between $525 and $1,650 per year. The cost for new construction could be significantly less than for renovation.

Reuse

Urine that is collected by source separation systems is most often stored on-site for use as fertilizer in the immediate environs of the residence, but some installations utilize on-site temporary storage tanks that are periodically serviced by pumping (Berndtsson 2006). The urine from many households can thus be accumulated in a centralized location and used for larger scale application to crops.

As a replacement for conventional fertilizers, urine has several benefits. First, it should be cheaper than chemical fertilizers, being essentially a byproduct of normal household activity. Second, urine requires

[44] http://www.separett-usa.com
[45] http://www.ecovita.net

Adminsitrative Record
Page 34865

minimal processing prior to use as fertilizer and therefore is more energy efficient over its lifecycle than chemical fertilizers (Jönsson et al. 2004). Third, urine contains less heavy metals than either chemical or manure fertilizers (WHO 2006), and less pharmaceuticals and hormones than manure (Magid 2006; Hammer and Clemens 2007). Farmers surveyed in Sweden reported no negative impact on crop production (Berndtssen 2006).

Urine from a single individual will fertilize about 0.035 hectares (0.086 acres) at agricultural rates (Jönsson et al. 2004). If one assumes that residents of areas with more than one system per hectare will be encouraged to implement source separation, then 29,500 hectares (72,900 acres) of peri-urban and rural residential land, or approximately 77,800 households (roughly 250,000 persons) will be affected (see Technical Report 2, Viers et al. 2012). If all those households implemented urine separation, this amounts to enough urine-N to provide agricultural rates of fertilizer for around 8,750 hectares (21,600 acres) of farmland or landscaping.

Future Outlook

Recent experiences in San Diego, Los Angeles, and San Ramon (Guest et al. 2009) indicate that Californians may not be prepared to implement urine separation, or similar waste-recycling technologies on a wide scale. As further research is conducted and costs of the technology decrease, this option for reduction of nitrate loading/leaching to groundwater will become more attractive, particularly in areas with high septic system densities.

5.4.2 Post-septic Tank Nitrification/Denitrification

Septic systems present an opportunity to intercept and treat water with nitrogen concentrations many times the MCL before it reaches the aquifer. In certain locales of the study area, this can result in sizeable reductions in nitrate loading/leaching to groundwater. In contrast to urine separation systems, treatment does not result in a reusable resource; however, it can reduce the concentration of nitrogen in septic leachate to lower levels than urine separation. Current research on wood-chip bioreactors (WCBRs) indicates great potential for reduction of nitrate (Leverenz et al. 2010; Schipper et al. 2010; Moorman et al. 2010).

Performance

Robertson and Cherry (1995) used an in situ bioreactor to treat septic leachate from around 60 ppm nitrate-N to 2 – 25 ppm nitrate-N. Robertson et al. (2000), in a series of field trials lasting up to 7 years, found that Wood Chip Permeable Reactive Barriers (WCPRBs) treating high-nitrate septic leachate removed between 74 and 91 percent of influent nitrate. These WCPRBs were two horizontal denitrification walls installed below septic leach fields, and a vertical denitrification wall installed to intercept a septic leachate plume. A fourth bioreactor in this study was installed as a sub-surface containerized Wood Chip Bioreactor (WCBR) to treat low-concentration agricultural drainage. The low-concentration WCBR removed 58 percent of influent nitrate.

Adminsitrative Record
Page 34866

Economics

A typical septic leach field for a household of four is 400 square meters (0.1 acres), and infiltrates around 300 cubic meters (0.24 acre feet) of water per year. For a WCBR with a 10-day residence time, this would require less than 10 cubic meters of woodchips. A 10 cubic meter (13 cubic yards) denitrification bed under the entire 400 square meters would likely be too thin to produce appropriate hydraulics. In the denitrification bed study by Robertson et al. (2000), a slurry of silt and wood chips was used to increase hydraulic retention time, but in a household setting, a more economical approach would be to install a separate tank for the WCBR between the septic tank and the leach field. The woodchips in this tank could be replaced or serviced easily as needed, whereas a layer of wood chips below the leach field would be expensive and difficult to maintain.

Effluent from the septic tank contains nitrogen in the form of ammonium, requiring nitrification before it can be denitrified in a WCBR. Nitrification is accomplished by passing the septic effluent through an unsaturated media (typically sand) aerobic filter. Two types of systems are typically used, single pass systems that require no dosing tank, and multi-pass systems that require a dosing tank, but use a much smaller filter (Metcalf and Eddy 2003). Single pass systems, 30 m^2 (323 square feet) of surface area and 1 m (3.3 feet) deep, use a pump from the septic tank to spray effluent onto the large surface of the filter at a slow enough rate to maintain unsaturated conditions. Effluent then flows by gravity from the outlet of the filter to the WCBR tank and then to the leach field.

In a multi-pass system, the filter is only 2 m^2 by 1 m (22 square feet by 3.3 feet) deep, but a 500 gallon recirculating tank is required. Effluent flows from the septic tank to the recirculating tank by gravity, is pumped into the filter, and flows by gravity back to the recirculating tank, with a portion flowing to the WCBR and then to the leach field. Mixing of the partially nitrified effluent from the filter with the septic tank effluent accomplishes partial denitrification (up to around 60% removal). In a multi-pass system, the WCBR is a finishing component and could be smaller sized. Such a system would cost between $10,000 and $20,000 to install as a retrofit to an existing septic system (Crites and Tchobanoglous 1998, EPA 2002). Bio-filtration for nitrification is a common technology in wastewater treatment and has been extensively tested. Off the shelf components and systems are available. Monitoring of these systems is easily accomplished due to containerized treatment.

Future Outlook

Based on the analysis presented in Technical Report 2, (Viers et al. 2012), there are roughly 77,800 households using septic systems in areas with septic system densities above one per hectare. These households produce about 1,031 metric tonnes (1,136 short tons) per year of leached nitrogen through their septic systems – about half of the total nitrogen leached by septic systems by our estimation. The reduction of this nitrogen load with nitrification/denitrification systems as described above would be most limited by the implementation coverage, as near 100% removal of nitrate is not difficult to achieve with a properly functioning nitrification/denitrification system. In addition, effluent from these systems can be expected to have reduced presence of pathogens and other constituents due to biological treatment in the bioreactor.

Adminsitrative Record
Page 34867

6 Reducing Nitrate Loading from Active Wells, Dry Wells, and Abandoned or Destroyed Wells

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Adminsitrative Record
Page 34868

6.1 Introduction

Most nitrate is transported from sources to groundwater via soil percolation and recharge. However, dry wells, abandonded wells, or improperly destroyed wells may act as rapid local conduits of nitrate contaminated surface runoff directly into groundwater. In addition, many deep wells may inadvertently act as conduits for deep aquifer contamination from shallow, nitrate-contaminated groundwater (for details, see Section 9 of Technical Report 2, Viers et al., 2012). The role of these wells in aquifer contamination is not limited to nitrate. In urban and industrial areas, many other contaminants may rapidly be leaked into groundwater and across multiple overlying aquifer units into deeper groundwater. Similarly, in agricultural areas, farm chemicals such as pesticides may also be rapidly leaked into groundwater or transferred to deep groundwater along these conduits.

Preventing contamination from leaky or improperly constructed active, abandoned, or dry wells is therefore a significant concern, beyond nitrate impacts. Proper construction and destruction of wells can largely avoid this type of contamination, which affects groundwater quality at the local scale, even if it is not a large source of nitrate at the basin scale. This section discusses potential solutions to remedy these sources and to prevent future groundwater contamination from these direct pathway sources. Specifically, this section discusses options to reduce loading by backflow prevention, options to reduce loading from existing or abandoned wells by retrofitting and proper destruction, and the potential costs of such actions.

Much of the information presented was assembled by informal interviews with hydrogeological professionals to obtain an overall assessment of reduction options, but this section does not provide a formal, extensive review of the available technical literature.

Adminsitrative Record
Page 34869

6.2 Reducing Nitrate Loading from Direct Inflow into Active Wells

Nitrate loading to groundwater from this source is best avoided by rigorous backflow prevention programs and by proper well construction, including a functioning annular seal below the surface, proper surface completion, and protection of the well per recommendations from the California Department of Water Resources (DWR 1981; DWR 1990).

6.2.1 Implementation and Cost

Inspection of newly constructed wells by county personnel upon well completion, where not already done, is one possibility for enforcing existing regulations on proper well design (currently an unfunded mandate although funding could be provided through, for example, a well permit fee). Implementing a requirement of having the well construction and especially the seal placement of large municipal and agricultural supply wells supervised and certified by a registered civil engineer, geologist, or hydrogeologist would be a further, more rigorous option that is already exercised for the construction of many monitoring wells and large public water supply wells, but which is an uncommon practice for the construction of irrigation wells. The cost of county inspection (at least $500 - $1000 for basic inspection) or professional certification (at least $2,000, assuming about ten hours of billable consulting hours) would add less than 5% to the cost of intermediate to large production well construction. With an average of 468 large productions wells drilled annually in the TLB and SV basins (see Technical Report 2, Section 9, Viers et al. 2012), the total annual cost of this measure is on the order of at least $1.5 million. This estimate does not include enforcement for domestic wells. The cost of installation and equipment for backflow prevention varies depending on type. Prevention by air gap requires no additional device. Mechanical devices may cost a few to several hundred dollars per system. Assuming an average cost of $200 - $400 per well, annual costs for installing backflow prevention on new wells is on the order of $1-2 million. It is unknown how many existing agricultural wells would need backflow prevention to prevent fertilizer chemical backflow.

6.2.2 Benefits

Relative to other measures, this reduction option addresses future well construction, but does not address construction problems (other than backflow prevention) at existing wells.

Adminsitrative Record
Page 34870

6.3 Reducing Nitrate Loading from Direct Inflow into Abandoned or Dry Wells and from Aquifer-to-Aquifer Leakage in Abandoned Wells

The first step to reduce N loading to groundwater via direct inflow into abandoned and dry wells is a field reconnaissance program that identifies the location of abandoned and dry wells (many of which may have little or no obvious surface structure). The second step is to properly destroy wells and dry wells using developed protocols (California Department of Water Resources 1981, 1990). Proper well destruction requires improved guidelines and county implementation. Currently, many counties in the study area require the destruction of the near-surface portion of the well and the placement of some sort of seal to prevent inflow to wells. However, this does not seal the aquifer-to-aquifer leakage, especially from the most vulnerable shallow aquifer zone. Proper destruction of wells includes removal and pressure grouting of the existing well and well-bore, or, at a minimum, perforation (sometimes referred to as laceration) of the well-casing and pressure-grouting of the gravel pack and casing.

The "Sacramento County Abandoned Well Restoration Project" provides an excellent case study for a county-directed well destruction program (David Von Aspern, personal communication, October 9, 2011). In 2009 and 2010, the county's Environmental Management Department (EMD) received $1.4 million funding from a Special Environmental Project (SEP) to locate abandoned wells and provide support to well owners for proper destruction of abandoned wells under the county's water well ordinance. Funding was used for a year-and-a-half of field canvassing and program administration, for an extensive public outreach program, and for providing waivers of the "inactivation permit" fee (normally $426 per inactivated well) to owners of abandoned wells. Approximately 60 wells have been destroyed over a two year period. Roughly 460 illegally abandoned wells have been discovered and processed thus far. Destruction costs encountered (by well owners) in that program vary widely. The lowest price for an agricultural well destruction over the initial two year phase of the project was $4,000. The highest price was $13,000 (including County permit costs). Wells have ranged in depth from less than 200 feet to 300 feet. Costs are a function of (ibid.):

- well depth
- well construction and design
- geology
- availability of a well log, in the absence of which a down-hole camera is needed prior to designing the destruction protocol
- number(s) and sizes of voids and cavities created during well use
- obstruction(s) present in the borehole, such as a "stuck pump"
- whether the well was previously part of a "hand-dug" well or a "pit well." When this is the case, most often the pit is filled with junk and debris, which must be removed, sometimes using manual labor.

Adminsitrative Record
Page 34871

- type of perforation (laceration) used for destruction (blast[46] or mechanical)
- presence of secondary constraints, such as a) newer buildings having been constructed too close to the well, b) powerlines too close, thereby interfering with drill rig, c) newer wells built too close to the old one, d) if the groundwater had become contaminated at some point, necessitating containerization and legal disposal of water coming out of the well as it is grouted from the bottom up, e) proximity to residential subdivision.

Other estimates for proper well destruction that we have obtained typically range from $10,000 for shallower wells (up to 400 ft) to over $50,000 for deep wells, including an example of a consultant estimate of $60,000 for perforating and pressure grouting a deep, 240 m (800 ft) well.

6.3.1 Implementation and Cost

It may be possible to identify abandoned or dry wells with steel casing through remote sensing and on-the-ground validation of the type of well (inactive, abandoned, dry). Without remote sensing, local and state governments will largely rely on information voluntarily provided by landowners or other forms of identifying abandoned and dry wells. Incentives programs by the county and state may motivate private landowners to identify the location of abandoned wells, particularly, if the landowner can avoid some of the financial liability associated with the proper abandonment of wells or dry wells. Dry wells may be operated by public entities (cities) and therefore more forthcoming in the identification. Depending on the methods used, the (partial) identification of abandoned and dry wells (not including the cost of proper well abandonment itself) may vary widely (from $1 million to $10 million per county).

Proper well destruction (including that of dry wells) typically includes sealing the borehole simultaneously with pulling the existing piping out of the borehole. Often, the latter cannot be accomplished and requires the overdrilling of the existing borehole, an operation that is as expensive as the construction of a new well. Alternatively, the well casing is lacerated and then pressure-grouted. Conservatively, assuming that the average cost of proper well destruction is on the order of $20,000 and that approximately 1,233 wells (see Technical Report 2, Section 9, Viers et al. 2012) need proper destruction due to their potential exposure to large amounts of surface run-on with nitrate-laden water, the one-time cost for this measure is on the order of $25 million for the TLB and SV basins.

Further, assuming that the number of wells newly abandoned each year is 94 wells per year, 20% of the number of newly constructed agricultural and municipal wells (468 wells per year, see Technical Report 2, Section 9, Viers et al. 2012), and that the proper destruction of these newly abandoned wells (not including newly abandoned domestic wells) can be done at an average cost of $20,000/well, the total ongoing cost of proper well destruction is on the order of $2 million per year for the TLB and SV basins.

[46] This method has a significant potential to lead to ineffective abandonment in some cases.

Adminsitrative Record
Page 34872

6.3.2 Benefits

Relative to other measures, this reduction permanently prevents a significant amount of nitrate loading to groundwater, some from direct inflow of stormwater and irrigation water into wells. But more importantly, this measure would primarily prevent large amounts of aquifer to aquifer nitrate leakage (estimated to be as much as 2,600 Mg N/year). Perhaps more importantly, this measure prevents the leakage of other harmful contaminants into groundwater.

Adminsitrative Record
Page 34873

6.4 Reducing Nitrate Loading from Aquifer-to-Aquifer Leakage in Active Wells

In seasonally active, inactive, abandoned, or improperly destroyed wells with screens across multiple aquifer systems, the task of reducing N loading from a shallower to a deeper aquifer system requires measuring and/or estimating the amount of vertical leakage under pumping and under non-pumping conditions. A measurement program would be designed to also assess seasonal fluctuations. Vertical leakage estimates could include measurement of a vertical velocity profile under pumping conditions as well as non-pumping conditions inside the well screen. If downward flow exists, the nitrate concentration would need to be monitored, possibly at a regular interval (e.g., annually, every 3 years) to determine whether the downward flow is associated with a transfer of nitrate into deeper aquifer layers. Shallow wells would not be subject to these guidelines except for extending the surface seal to near the top of the screen.

In wells that are found likely to be transferring nitrate into deeper aquifer layers, the following options prevent/reduce the nitrate loading to the deeper layers:

- Well retro-fitting (lining sections of screen, cementing portions of well, blocking casing flow with packers), which is most appropriate in newer wells with large diameter existing casing.
- Destruction of the existing well and new construction. The cost of destruction and new construction may not be significantly higher than retrofitting, especially in smaller diameter wells; it is sometimes easier to identify and target high quality, high producing aquifer formations in a new borehole than in an already existing well. The decision between retrofitting and new construction needs to be done on a case by case basis.

In newly constructed wells, the following are recommended measures to help prevent the leakage of nitrate-laden groundwater into deeper layers or the occurrence of natural contaminants in well water. These measures require good test hole information on both aquifer yields and water quality, collected prior to actual well construction:

- Where possible (favorable water quality, aquifer productivity), construction of irrigation well screens in nitrate-contaminated aquifer horizons with no screen sections extending into deeper, cleaner aquifer horizons. This may limit the amount of water that can be pumped from an individual well and require construction of multiple smaller-production irrigation wells.
- Construction of well screens below contaminated zones with proper annular sanitary seal around a blank casing from the surface to near the top of the (first) well screen; use of high quality screen material (e.g., stainless steel) with large open area (e.g., continuous wire-wrap type screens), use of high quality inert filter pack materials (e.g., well-rounded quartz gravel), and proper well development, all of which provide the basis for a well yielding high quality water from limited thickness, high-yielding aquifer layer.
- Construction of a proper annular seal around blank casing between aquifer layers (across aquitards) and simultaneous installation of well-packers inside the well casing that prevent the

Adminsitrative Record
Page 34874

leakage of groundwater across aquifer horizons during or after groundwater pumping operations.

- Minimization of well screen length and number of screen segments by using screens with a high open area ratio (e.g., continuous wire-wrap screens) that are placed only against the most permeable aquifer horizons and that do not span contaminated aquifer horizons (would lead to cross-aquifer contamination within the well). This may limit the amount of well-production and require construction of a larger number of smaller-production wells.
- Regional programs (such as Kern County's) and guidelines that prevent construction of well screens across multiple aquifer systems.

6.4.1 Implementation and Cost

The implementation of these measures will require establishing new incentive programs or implementing new regulatory programs that control the design and construction of large production wells in nitrate-contaminated groundwater basins, or both. As a principal policy objective, agricultural pumping would largely occur from wells in shallower groundwater - up to 150 m (500 ft) depth – and be sealed off against deeper, uncontaminated groundwater, where water quality, hydrogeology, and other potential constraints are favorable. Domestic, small community, and large community groundwater supplies could be pumped from deeper groundwater, with design, assessment, and construction ensuring complete seals to the depth of the screen, and with screen placement at depth focusing on avoiding other natural contaminants or undesirable constituents (see above, also see Technical Report 5, King et al. 2012). These programs can be developed, for example, as part of a regional Groundwater Management Plan, a county ordinance, or as part of an Integrated Regional Water Management Plan.

Proper design and construction of new wells according to the guidelines above may increase the cost of large irrigation well construction. On the other hand, proper construction and well development may provide some long-term savings in pumping costs due to energy savings (reduced hydraulic well losses) and decreased well maintenance costs (absence of suspended sediments that cause wear and tear on pumping equipment). Over the life-time of the well (many decades), the above measures may actually decrease the overall cost of well installation and maintenance. Current irrigation well construction tends to be with long screens, low open screen areas (slotted well screens rather than continuous wire-wrap or louvered screens), and poor well development, leading to poor well efficiency. For large drinking water wells that are properly constructed and developed (proper well design with necessary pilot hole testing and assessment, deep sanitary seals, targeted short screen sections at desired depths, proper well development), the costs are on the order of at least $1 – 2 million per well (depth: 800 feet or more).

The cost of testing existing deep wells for the potential of cross-aquifer nitrate leakage is estimated to be on the order of a few thousand dollars per well, if a rigorous regional program were implemented that provides economies of scale. Assuming, for illustration purposes, that most of the estimated 20,000 existing irrigation wells in the TLB and SV are tested, the total one-time expense for the reconnaissance program would be at least $100 million.

Adminsitrative Record
Page 34875

Costs for retrofitting with packers and sealing off contaminated aquifer horizons, redrilling new wells and abandoning leaky wells, or other methods, may vary widely from well to well. Assuming an average cost of $30,000 for retrofitting one-third of the 20,000 irrigation wells, leads to a total one-time cost on the order of **$200 million**, which does not include the cost for drilling additional wells due to potentially reduced yields in retrofitted wells.

6.4.2 Benefits

Relative to other measures, this reduction option prevents a significant amount of nitrate loading to groundwater (estimated to be as much as 4,000 Mg N/year, currently, and significantly more in future nitrate pollution avoidance).

Adminsitrative Record
Page 34876

6.5 Discussion and Promising Actions

A long history of inadequate well design and well construction that was – and often continues to be – largely not focused on reducing the potential for contamination, has led to two major nitrate contamination related impacts:

1. Short-circuiting of contaminants directly from the surface into the well or short-circuiting of contaminants in the shallowmost aquifer (near the water table) into the well.
2. Cross-aquifer contamination between aquifers through borehole leakage, either in the filter pack or in the well via the well screen, or both.

The well construction issues that lead to either one of these two conditions can be summarized into the following categories:

1. improper or poor construction of the mandatory sanitary surface seal in the annulus around the well casing;
2. mandatory sanitary seal depths (15 m (50 ft), except in Monterey County: 30 m (100 ft)) are insufficient and do not prevent movement of contaminated shallow groundwater through the filter pack into a well screened in a deeper, higher quality aquifer;
3. long intake screens that span multiple aquifer and aquitard horizons with highly variable water quality;
4. improper pump location in the upper screened section of the well possibly leading to preferential pumping of predominantly shallow, often more nitrate-contaminated water;
5. poor well design due to a lack of proper assessment of the hydrogeology by a trained professional using the appropriate geophysical and geochemical probing tools;
6. poor choice of well construction material leading to a perceived need for long screens and to poor sanitary/annular seal construction;
7. lack of backflow prevention devices on wells used to mix water with agricultural fertilizer.

The above issues are particularly widespread in the construction of private domestic and irrigation wells. Well owners are often poorly informed or misinformed about trade-offs between up-front construction costs (which can be significantly higher in properly constructed wells), long-term energy, well and pump maintenance, and water treatment costs (which may quickly reach levels much higher than the additional construction investment for a properly constructed and developed well).

Policy measures to address these issues include:

- providing appropriate informative materials for (future) well owners
- improved well construction standards/guidelines and appropriate, regular enforcement, possibly funded through, for example, well construction fees or energy incentives programs
- proper continuing education and training programs for well drilling and consulting professionals

Adminsitrative Record
Page 34877

- improved licensing requirements/training for drilling and consulting professionals involved in the design of groundwater wells
- well and backflow prevention inspection programs.

Table 19, below, summarizes the N reduction measures described above, the associated costs, and the associated estimated reduction in annual N loading to groundwater. The least cost measure, and one that has a large potential for future increased N loading avoidance is the implementation of a backflow prevention program and of rigorous well construction standards for wells that penetrate more than the shallow-most aquifer layer.

Rigorous well construction standards can avoid both surface inflow into the well casing or annulus and – much more importantly from an aquifer management perspective – they may avoid the silent and unseen leakage of shallow contaminated groundwater into deeper, uncontaminated horizons. Relative to the cost of other measures, implementation of more rigorous well construction standards may in many cases be nearly cost-neutral given that proper well construction, while initially more expensive, may be offset by significant reduction in long-term energy and maintenance/well rehabilitation cost. This should therefore be a high priority measure. However, it does require more rigorous enforcement by the counties of their well construction permit regulations (additional funding may be possible through increased permit fees for certain well types (e.g., deep wells), or through incentive programs by energy providers).

Table 19. Summary of the estimated potential costs and benefits of N loading reduction measures associated with wells. The categorization of "benefits" as significant is not meant to be interpreted as a recommended measure. The benefit indicates merely whether the overall impact on groundwater nitrate in the study area is thought to be measureable and beneficial in the long run. Public policy debate must decide whether the avoided impact is worth the cost of these measures.

Reduction Measure	Cost for Entire Study Area (one-time or annually)	Benefit to Overall TLB & SV N Load Reduction
Backflow prevention program	$1-2M per year (on new wells)	Significant to avoid local hot spots
Enforcement of proper well construction	$1.5M per year (on new wells)	Significant in the future – several 1,000 Mg N/yr
Identification of abandoned/dry wells	$1M to $10M once	
Proper abandonment of abandoned/dry wells	$25M once + $2M per year	Significant - less than 1,000 Mg N/yr
Testing existing deep wells for leakage	$100M once	
Retrofitting existing deep wells for leakage avoidance	$200M once	Significant - 4,000 Mg N/yr

Adminsitrative Record
Page 34878

Retrofitting existing wells to avoid cross-aquifer leakage is a potentially expensive option, but may avoid significant deeper aquifer contamination that is out of sight and typically ignored. A least-cost alternative may be the sealing of existing screens in the uppermost, most nitrate contaminated aquifer horizons (or the opposite – sealing off all deeper, uncontaminated aquifer horizons).

Identifying and properly destroying inactive/abandoned or dry wells may be difficult and costly to implement. It does not necessarily have a significant impact on groundwater nitrate loading from surface inflow, but it could avoid significant N loading from cross-aquifer leakage in deep abandoned wells. For those, a potentially lower-cost alternative to casing removal or overdrilling is to perforate the screen and casing and to fill the casing and gravel pack by pressure-grouting. Sacramento County is an example of a recent program to address this issue.

We note that costs provided in this section are based on numerous assumptions, are order-of-magnitude estimates, and are intended for illustration purposes only. Costs do not reflect any actual survey of well drilling or retrofitting costs. Further assessment of the extent of well leakage and of technologies and costs to addressing well leakage is needed.

Adminsitrative Record
Page 34879

7 Overall Conclusions: Source Reduction

Technologies are available for reducing the rates of nitrate leaching to well below those that historically have contaminated groundwater. Generally, reduction of nitrate leaching involves changes in management and upgrading of infrastructure. Costs for doing this vary widely. We have considered the methods and associated costs for reduction of nitrate leaching losses from the major anthropogenic sources of N loading to groundwater in the two study regions. These include irrigated cropland, livestock operations, turfgrass and other urban landscaping, wastewater treatment plants and food processing plants, septic systems, and abandoned wells.

Reduction of agricultural nitrate leaching from cropland and livestock and poultry operations is primarily dependent upon changes in farm management and improving crop nitrogen use efficiency (NUE). NUE can be increased by the correct timing and rates of applied fertilizer N, animal manures, and irrigation water, and to a lesser extent by modification of crop rotations. Improving the storage and handling of manures, livestock facility wastewaters, and fertilizers also plays a role in nitrate leaching reduction. Tandem implementation of improved management practices, chosen in relation to each unique farm situation, is the best approach to reducing nitrate leaching from agricultural fields.

While improvements in N use efficiency are possible, a practical upper limit is set by unpredictable rainfall, difficulty in predicting the rate of mineralization of organic N, and most especially, soil spatial variability and the need to leach salt from the crop rootzone. It is estimated that by implementing recommended practices, crop N recovery can reach 60-80% of N inputs. While improved management will lead to a reduction in the mass of nitrate lost by leaching, it is unlikely that the concentration of nitrate can be reduced to the MCL, especially where the sole or main source of aquifer recharge is percolate from irrigated crop fields.

Based only on inherent characteristics of the soil, crop species grown, and irrigation systems in use, we conclude that approximately 52% of irrigated cropland in the Salinas Valley and 35% of such land in the Tulare Lake Basin is susceptible to significant nitrate leaching losses. Improved management in these more highly vulnerable areas is expected to have the most impact on nitrate leaching reduction. A number of the practices known to reduce cropland nitrate leaching have been adopted in recent years by farmers in the study area. Those management options that are less utilized are generally associated with multiple barriers to adoption by farmers. These include higher operating or capital costs, risks to crop quality or yield, and conflicting farm logistics. Additional significant barriers include inadequate education and inadequate adaptation of practices to local conditions and research to this end.

Expanded efforts to promote the adoption of nitrogen efficient practices are clearly needed. The educational barriers that are associated with many of the identified practices highlight the importance of expanded funding of research, education, and outreach activities to better assist farmers in applying best management strategies. For example, Cooperative Extension, the USDA Natural Resources Conservation Service, and independent crop advisors play important roles in delivering such educational information. More on-farm research is required to demonstrate the impact of practices on NUE and N

Adminsitrative Record
Page 34880

leaching for specific crops and soils. Development of crop-specific nitrogen application metrics that relate, for example, total N applied to total N harvested in the crop, needs to be further explored and supported. To allow for identification of the areas in which improved management is needed most, a task force charged with fine-tuning existing options for spatial nitrate leaching risk assessment methods is recommended. Such a method would need to include consideration of the spatial soil characteristics, as well as probable monitoring requirements.

Finally, as the recently instituted Central Valley dairy regulations[47] restrict on-farm cropland applications of manure, the volume of off-dairy transfer of animal waste to unregulated fields has increased. It is not known which crop species and soils are receiving this manure or how the receiving farmers have integrated the manure into their N fertilization practices. We recommend development of adaptive research and education programs that will promote conversion of solid and liquid dairy manure into forms that meet the food safety and production requirements for a wider range of crop species. Providing guidance to non-dairy farms in co-managing organic and conventional N sources is necessary to help ensure the nutrient value of manure is properly accounted for in fertilization regimes.

The cost of improving crop NUE is difficult to define quantitatively and there is wide uncertainty in the marginal costs of such improvements. Therefore it is not possible to assign a cost (incurred for on-farm improvements) to a defined or quantitative lowering of cropland nitrate leaching. Our model suggests that modest reductions in N fertilizer application rates and increased adoption of related improved practices will increase production costs slightly, assuming adequate levels of education. Thus, modest reductions are concluded to be economically feasible in the general sense. As larger load reduction strategies are undertaken, the model predicts production costs to increase enough to result in larger reductions in irrigated cropland within the study area, or to decrease the area of land used for lower revenue crops such as grain, hay, and other field crops. A simulated nitrogen sales tax revealed that it could initiate additional grower motivation to adopt practices that contribute to NUE.

Studies confirm that leaching from urban turfgrass areas, including golf courses, are often negligible due to the dense plant canopy and perennial root system that allows for continuous N uptake over a large portion of the year. Additionally, often any excess N above plant need will still be taken up and utilized by the turf, increasing vegetative growth. However, poor management can lead to a discontinuous canopy and weed presence, wherein nitrate leaching risk increases, especially if growing on permeable soils, if over-irrigated, or if fertilized at high rates during dormant periods. The best leaching reduction strategy from this source then is to simply follow recommended guidelines of fertilizer application rate and timing. The UCCE and UCIPM publish such guidelines. The practice of keeping fertilization rates and timing concordant with plant need has the added benefit to professional turfgrass managers of requiring less frequent mowing and reducing the volume of clippings requiring disposal.

Discharges from municipal wastewater and food processing treatment facilities, while not as regionally significant as agriculture, do have locally important impacts. Implementation of control options for these sources is feasible and an important part of a multi-pronged approach.

[47] The General Order for Waste Discharge from Existing Dairies in the San Joaquin Valley, see http://www.waterboards.ca.gov/centralvalley/water_issues/dairies/dairy_program_regs_requirements/index.shtml

Adminsitrative Record
Page 34881

For locations where discharges from WWTPs and FPs may be detrimental to groundwater quality, reduction strategies include transfer of food processing waste to publically owned treatment works and modification or addition of treatment (e.g., biological nutrient removal). The most cost effective solutions for management of N loading from WWTP and FP discharges may be an adjustment to the current practices to limit N application to levels that can be accommodated by irrigated crops and soil degradation. As such, the discharge should be managed in an agronomic manner such that the nutrients, especially nitrogen present in the wastes, are included in the overall nitrogen management plan for the receiving crops.

Another primary consideration is the optimization of operations; limiting nitrogen and total discharge volume through in-plant process modifications may be sufficient for some facilities. Lastly, the technology for nutrient removal is proven and can be implemented where other options are not sufficient to limit N loading to groundwater. Costs of implementing reduction strategies are widely variable and generally facility dependent. Evaluation of nitrate loading due to waste discharges and impacts of operational changes is difficult at some facilities due to recordkeeping shortcomings. Groundwater monitoring is required for many of these facilities; however, the data are largely unavailable since they are not in a digital format. To improve monitoring, enforcement, and abatement efforts related to these facilities, groundwater data need to be centrally managed and organized digitally.

The need to replace aging infrastructure is a concern across the US; while leaking sewer pipes can lead to nitrate contamination of groundwater, this poses a smaller threat to public health than pathogen contamination. Choice of pipe material determines the cost of sewer retrofits. Septic system contributions to groundwater N are another locally significant source. At a reasonable cost, these sources can be reduced or eliminated using existing technology, including local reuse of nitrogen through source-separation and post-septic tank biological denitrification. Urine separation technology, in particular, has the potential to reduce the use of chemical fertilizer, representing possible cost savings for farmers and reducing energy expenditure.

Contributions to aquifer and deep aquifer nitrate contamination from wells (permanently or seasonally inactive, abandoned, or dry wells) are best addressed through development and implementation of rigorous well construction standards for future wells. Reconnaissance of abandoned and improperly destroyed wells and their proper destruction is another step toward avoiding aquifer contamination from either surface inflow to wells or cross-aquifer flow within wells. Expenditures on retrofitting of existing wells are high and should be selected based on their individual contamination risks.

Adminsitrative Record
Page 34882

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Adminsitrative Record
Page 34883

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Adminsitrative Record
Page 34884

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Adminsitrative Record
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Adminsitrative Record
Page 34904

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 4:

Groundwater Nitrate Occurrence

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
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Adminsitrative Record
Page 34905

Groundwater Nitrate Occurrence

Technical Report 4

Addressing Nitrate in California's Drinking Water

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Report for the State Water Resources Control Board Report to the Legislature

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Contents

Tables vi

Figures viii

Acronyms and Abbreviations xiv

Unit Conversions xvii

Summary 1

1 Introduction 3

2 Tulare Lake Basin Hydrogeology and General Water Quality 5

2.1 Physical Setting 6

2.1.1 Location 6

2.1.2 Hydrologic Subbasins 7

2.2 Hydrology 9

2.2.1 Overview 9

2.2.2 Water Budget 9

2.2.3 Groundwater Levels 12

2.3 Hydrogeology 14

2.3.1 Geology 14

2.3.2 Groundwater Recharge 19

2.3.3 Flow Modeling 20

2.3.4 Transport Modeling 20

2.4 Water Quality 21

2.4.1 Natural Contaminants 21

2.4.2 Groundwater Chemistry 22

2.4.3 Anthropogenic Contaminants 26

3 Salinas Valley Hydrogeology and General Water Quality 31

3.1 Physical Setting 32

3.1.1 Location 32

3.1.2 Subareas 32

3.2 Hydrology 35

3.2.1 Overview 35

Adminsitrative Record
Page 34909

3.2.2 Water Budget 35
3.2.3 Groundwater Levels 39
3.2.4 Seawater Intrusion 40
3.3 Hydrogeology 44
3.3.1 Geology 44
3.3.2 Aquifer Heterogeneity 45
3.3.3 Flow Modeling 47
3.3.4 Transport Modeling 49
3.4 Water Quality 51
3.4.1 Nitrate Data Collection 51
3.4.2 Nitrate Analysis Reports 51
3.4.3 Nitrate in Groundwater 53
3.4.4 Additional Reports 56
3.4.5 Nitrate Transport - Aquifer Heterogeneity & Groundwater Age 59
3.4.6 Groundwater Chemistry 60
4 Development and Description of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database 63
4.1 Introduction 64
4.1.1 Objectives 64
4.1.2 Background 64
4.1.3 Data Sources 67
4.1.4 Well Identification Codes 76
4.1.5 Potential Future Improvements to a Nitrate Database 78
4.2 Methods 80
4.2.1 CASTING Database Structure 80
4.2.2 Developing the NO3GW and NO3GT Datasets for CASTING 80
4.2.3 Adding Data to CASTING 83
4.2.4 Attributing Wells with Depth Information 84
4.2.5 Mapping of Nitrate in Groundwater 85
4.3 Spatial Distribution of Wells Data in CASTING 87
4.3.1 Spatial Distribution of All Wells in the Database 87
4.3.2 Spatial Coverage by Data Source 87

Adminsitrative Record
Page 34910

4.3.3 Depth of Wells 92
4.4 Test Origins and Temporal Distribution 96
5 Analysis of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database 98
5.1 Introduction 99
5.1.1 Disparate Data Sources and Collection Methods 99
5.1.2 De-clustering of Data Sources 100
5.2 Current Spatial Distribution of Nitrate in Wells 103
5.2.1 Dataset Overview 103
5.2.2 Exceptionally High Values and Treatment of Outliers 103
5.2.3 Frequency Distribution 104
5.2.4 Spatial Distribution of Nitrate Concentration 106
5.3 Effect of Data Source, Well Type, and Well Depth on Groundwater Nitrate 113
5.3.1 Effect of Data Source 113
5.3.2 Depth-Dependency of Nitrate 117
5.4 Effect of Soil Type and Vadose Zone Thickness 121
5.5 Nitrate Above and Below the Corcoran Clay 124
5.6 Nitrate in Wells near Streams 135
5.7 Historic Trends 137
6 Groundwater Nitrate Forecasting: Assessment of Vadose Zone Nitrate Transport 148
6.1 Introduction 149
6.2 Modeling with HYDRUS 1D 150
6.2.1 Calculating Leachate Fluxes 150
6.2.2 Soils 152
6.2.3 Soil Column Configurations 153
6.2.4 Modeling Nitrate as a Conservative Solute 153
6.2.5 Climate Data 155
6.2.6 Modeling 156
6.2.7 Results 156
6.2.8 Modeled Irrigation 158
6.2.9 Verifying Downward Velocity Estimates 159

Adminsitrative Record
Page 34911

6.3 GIS Map Creation 161
6.3.1 Land Use 161
6.3.2 Field Water Budgets 161
6.3.3 Depth to Water Table 162
6.3.4 Results and Discussion 162
7 Groundwater Nitrate Forecasting/Modeling 167
7.1 Introduction and General Conceptual Overview of the Approach 168
7.1.1 Nonpoint Source Pollution a Global Groundwater Quality Threat 168
7.1.2 Key Differences Between Nonpoint Source Pollution and Point Source Pollution Dynamics . 169
7.1.3 Groundwater Nonpoint Source Assessment Tools 170
7.1.4 Proposed Modeling Framework 172
7.2 Nonpoint Source Assessment Toolbox 173
7.2.1 Conceptual Approach 173
7.2.2 High Resolution Groundwater Velocity Field Computation 174
7.2.3 Transport Simulation: Streamlines 177
7.2.4 Streamline Computation 178
7.2.5 Unit Response Function Approach 181
7.2.6 Transport Simulation along Streamlines 181
7.2.7 Unit Response Function Parameterization 182
7.2.8 Summary of Construction Phase 183
7.2.9 Implementation Phase: Computing Breakthrough Curves at CDSs 184
7.3 Approach to Generate Representative Groundwater Velocity Distribution 186
7.3.1 Steady-state Model for Tulare Lake Basin 186
7.3.2 Conceptual Background 190
7.3.3 Groundwater and Transport Simulation 198
7.4 NPSAT Implementation Based on Coarse Model 205
7.5 Results 209
7.5.1 Nitrogen Loading Scenarios 209
7.5.2 Model Validation 214
7.5.3 NPSAT Modeling Predictions 221
8 Groundwater Nitrate Occurrence: Overall Conclusions 228

Adminsitrative Record
Page 34912

9 Literature Cited 229

Section 1 Introduction 229

Section 2 Tulare Lake Basin Hydrogeology and General Water Quality 229

Section 3 Salinas Valley Hydrogeology and General Water Quality 232

Section 4 Development and Description of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database 237

Section 5 Analysis of the California Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database 238

Section 6 Groundwater Nitrate Forecasting: Assessment of Vadose Zone Nitrate Transport 239

Section 7 Groundwater Nitrate Forecasting/Modeling 240

Section 8 Groundwater Nitrate Occurrence: Overall Conclusions 248

Section 10 Appendix A - Review of Domestic Well Monitoring Programs for the U.S. 50 States 249

Section 11 Appendix B – Denitrification in Central California Soils and Aquifers 254

10 Appendix A - Review of Domestic Well Monitoring Programs for the U.S. 50 States 260

10.1 Introduction 260

10.2 Summary 260

11 Appendix B – Denitrification in Central California Soils and Aquifers 269

11.1 Denitrification Processes 269

11.2 Denitrification in Aquifers 270

11.3 Denitrification in Soils 272

11.4 Study Area Characterization 274

11.5 Denitrification Potential in Tulare Lake Basin and Salinas Valley 276

11.6 Conclusions 277

Adminsitrative Record
Page 34913

Tables

Table 1. Subbasins within the Tulare Lake Basin, California. 8

Table 2. Annual average groundwater budgets from hydrologic models in the Central Valley. 11

Table 3. Distinguishing characteristics of the subareas of the Salinas Valley, California. 33

Table 4. Average values of groundwater budget in acre-feet/year. 38

Table 5. Urban groundwater pumping by city or area. 39

Table 6. Groundwater models of the Salinas Valley. 48

Table 7. Groundwater sampling reports for the Salinas Valley. 53

Table 8. Number of wells sampled by subarea and year 56

Table 9. Nitrate/water quality studies for the Salinas Valley. 57

Table 10. Salinas Valley Basin studies. 58

Table 11. Geological studies of the Salinas Valley Basin. 59

Table 12. Water system classification by constituency, connections, and duration of service per year .. 65

Table 13. Tests and wells by data-source within the study area boundaries.. 76

Table 14. Well identification codes and agency users. 77

Table 15. Priority of screen top data sources. 84

Table 16. Number of nitrate tests in CASTING, within the study area, by county and time period. 96

Table 17. 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): number of wells measured, median nitrate concentration, and percentage of MCL exceedance for the TLB and the SV. 114

Table 18. Summary of database records for nitrate samples for the current period (after 1999) including the data source, the number of wells measured by each data source, the number of wells-years, the median nitrate value, and the percent of annual average well nitrate values that exceeded 9, 22.5, 45, and 90 mg/L nitrate. 116

Table 19. Median nitrate and exceedance probability as a function of well depth and location on nitrate concentration. 117

Table 20. Mean and 95% confidence interval, by groundwater basin and well depth category, of the mean of decadal well means of the log of nitrate. 120

Adminsitrative Record
Page 34914

Table 21. Median nitrate concentration and exceedance probability in wells, grouped by Department of Pesticide Regulation (DPR) groundwater protection zone. 121

Table 22. Difference in median values of nitrate concentration (mg/L as nitrate) in equal area sample sets between wells screened above and below the Corcoran Clay, by decade. 127

Table 23. Results of Paired-Sample Wilcoxon Signed Rank tests. 134

Table 24. Regional Kendall test results on the de-clustered equal area statistics for nitrate. 143

Table 25. Statistics of the least-square estimates of the regression slope (mg/L per year) of nitrate concentrations versus time for each well with at least two sample values, grouped by groundwater region and decade. 145

Table 26. Statistics of the least-square estimates of the regression slope (mg/L per year) of nitrate concentrations versus time for each well with at least two sample values 147

Table 27. Hydraulic parameters for the three soil types used. 153

Table 28. Soil column construction. 153

Table 29. Solute transport parameters. 154

Table 30. Monthly average precipitation and reference evapotranspiration for the city of Visalia. 155

Table 31. Advective travel time to the groundwater table in years for three soils and six depths. 157

Table 32. Travel times compared between simulated daily and weekly irrigation. 159

Table 33. Downward velocities using linear regression of modeled travel times and velocities based on the annual flux divided by the average water content. 160

Table 34. Average soil water content based on 70% and 90% irrigation efficiencies for cotton and alfalfa grown in sand and clay soils. 160

Table 35. Summary of domestic well regulations, testing programs, and educational resources by state. 262

Adminsitrative Record
Page 34915

Figures

Figure 1. Hydrologic subbasins of the Tulare Lake Basin, California. 7

Figure 2. Depth to groundwater for the Tulare Lake Basin and Salinas Valley. 13

Figure 3. Spatial distribution of sediments of the Corcoran Clay as modeled in the Central Valley Hydrologic Model (CVHM). 16

Figure 4. Fans of Northern and Central San Joaquin Valley. 17

Figure 5. Conceptual diagram of the San Joaquin aquifer system. 18

Figure 6. Based on the sediment origins, the TLB is divided into three groundwater regions: the Eastside Alluvial Fans, the Basin, and the Westside Alluvial Fans region. 19

Figure 7. Piper diagram of groundwater samples from Eastside Fans sediments. 24

Figure 8. Piper diagram of groundwater samples from Westside Fans sediments. 25

Figure 9. Piper diagram of groundwater samples from Basin sediments. 26

Figure 10. Nitrate levels in the uppermost 3 m of groundwater, simulated by statistical models based on general land use, groundwater, vadose zone thickness, and soils information. 30

Figure 11. Four major subareas of the Salinas Valley. 34

Figure 12. A significant increase in the use of micro irrigation has been observed in the Salinas Valley. 37

Figure 13. Groundwater levels in the subareas through time. 40

Figure 14. Map of seawater intrusion through time for the 180-ft aquifer. 42

Figure 15. Map of seawater intrusion through time for the 400-ft aquifer. 43

Figure 16. The two major groundwater regions used for the statistical analysis of groundwater nitrate data in Section 5. 45

Figure 17. Vertical cross section through a 3D heterogeneity model of subsurface soils in the Pressure subbasin of the Salinas Valley. 46

Figure 18. Horizontal cross section through a 3D heterogeneity model of subsurface soils in the Pressure and Eastside subbasins of the Salinas Valley 46

Figure 19. Wells showing percent young water less than 50 years overlain upon nitrate concentration contours. 50

Figure 20. Nitrate trends from MCWRA synoptic sampling program. 55

Adminsitrative Record
Page 34916

Figure 21. Piper diagrams of public supply water systems in the Salinas Valley. Concentrations are nitrate as nitrate. Data is from the CDPH STORET database. 62

Figure 22. Process diagram for adding new wells to CASTING database. 82

Figure 23. Regression of top of perforated interval from well depth. 85

Figure 24. All wells in the database as of December 17, 2011. 87

Figure 25. CADWSAP wells. 88

Figure 26. U.S. EPA STORET wells. 89

Figure 27. The State Water Board Environmental Monitoring wells and USGS Priority Basins wells. 89

Figure 28. County-sourced wells. 90

Figure 29. Regional Projects. 90

Figure 30. Dairy wells, data from the Central Valley Regional Water Board. 91

Figure 31. Population density and CASTING well distribution. 91

Figure 32. Wells with depth to top of perforation. 92

Figure 33. Distribution of depth to top of perforated interval for all wells with available data. 93

Figure 34. Distribution of depth to top of perforated interval for household domestic drinking water wells with available data. 93

Figure 35. Household domestic wells in CASTING with depth data available. 94

Figure 36. Comparison of mean depth to top of perforations between household wells and water system wells, by groundwater subbasin. 95

Figure 37. Number of samples collected per year, with maximum and average nitrate (as nitrate) mg/L (each year), in CASTING database 97

Figure 38. The two physiographic groundwater regions in the Salinas Valley and their equal area cells used for spatial de-clustering 101

Figure 39. The three physiographic groundwater regions of the Tulare Basin study area and their corresponding equal area cells, ... 102

Figure 40. Histogram of $\log_{10}(c)$ (x-axis), where c is the sample nitrate value, for all 100,029 non-zero individual samples, and for all 23,377 non-zero decadal means of $\log_{10}(c)$ at individual wells. 105

Figure 41. Average concentration of nitrate (mg NO3/L) in wells during the decade 2000 to 2010. 108

Adminsitrative Record
Page 34917

Figure 42. Maximum concentration of nitrate in wells during the decade 2000 to 2010. 109

Figure 43. Average nitrate by PLSS, apx. 2.59 km^2 1 mi^2) 110

Figure 44. Maximum nitrate by PLSS, apx. 2.59 km^2 (1 mi^2) 111

Figure 45. Back-transformed mean and confidence interval of decadal well means of log nitrate [y-axis: in mg/L nitrate] for the five major groundwater regions in the study area without equal area de-clustering 112

Figure 46. Back-transformed mean and confidence interval of the mean over all equal areas of the logarithm of the median in each equal area of annual well means, separately for the five major groundwater regions in the study area.. 112

Figure 47. Means and 95% confidence interval of means for DPR groundwater protection zones 122

Figure 48. Locations of wells underlain by the Corcoran Clay, and the position of their screens relative to the Clay. 125

Figure 49. Number of nitrate samples each decade in wells screened above and below the Corcoran Clay 126

Figure 50. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1950's.. 128

Figure 51. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1960's. 129

Figure 52. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1970's. 130

Figure 53. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1980's. 131

Figure 54. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1990's. 132

Figure 55. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 2000's. 133

Figure 56. Average nitrate concentration by PLSS section.. 136

Figure 57. Five-year moving average of the percentage of wells for which the average annual measured concentration exceeded 9 mg/L (background), 22.5 mg/L (half of the MCL), and 45 mg/L (MCL) 138

Figure 58. Back-transformed mean of the logarithm of equal area decadal medians that were computed from ten years of annual well means. 139

Adminsitrative Record
Page 34918

Figure 59. Mean of the equal area decadal exceedance probablility computed from ten years worth of annual well means in each equal area. 140

Figure 60. Back-transformed annual mean of the logarithm of equal area annual medians that were computed for one year of annual means of each measured well. 141

Figure 61. Annual crop coefficient curves. 150

Figure 62. Daily flux of leachate based on 70%irrigation efficiency for the 5 modeled crop cycles. 152

Figure 63. Travel times for a conservative solute in sandy soil. 158

Figure 64. Relationship between annual flux and downward velocity for three soil types. 158

Figure 65. Depth to the water table. 162

Figure 66. Travel time based on sand soil. 164

Figure 67. Travel time based on loam soil. 165

Figure 68. Travel time based on clay soil. 166

Figure 69. Typical spatial variability of a land use in an intensively managed semi-arid agricultural region with significant groundwater pumping for irrigation, Tule River Groundwater Subbasin 169

Figure 70. Schematic illustration of Simplistic Domain Decomposition Method. 176

Figure 71. Geometric definition of symbols involved in the weight calculation and Weight function. .. 177

Figure 72. Distribution of exit points around the well screen. 178

Figure 73. Prism element. 180

Figure 74. Calculation of Unit Response Function. 182

Figure 75. Construction phase of the NPSAT. 184

Figure 76. Simulation phase of the NPSAT. 185

Figure 77. Simulated cumulative annual changes in aquifer storage. 187

Figure 78. Water budget of CVHM. 188

Figure 79. Steady state head distribution for the water year 1996. 189

Figure 80. Comparison of averaged heads against the monthly heads used for averaging. 190

Figure 81. Empirical cumulative distribution function of maximum pumping rate. 191

Adminsitrative Record
Page 34919

Figure 82. Empirical cumulative distribution function of Depth [m] (distance between water table and bottom of the well screen). 191

Figure 83. Correlation between well depth and screen length. 192

Figure 84. Left) Empirical cumulative distribution function of the number of wells per square mile. Right) histogram of the number of simulated wells per square mile. 193

Figure 85. Locations and pumping rates of simulated wells in the TLB study area. 194

Figure 86. Empirical cumulative distribution function of simulated pumping rates. 195

Figure 87. Sampling range for screen length. 196

Figure 88. Empirical cumulative distribution function of simulated well depth. 197

Figure 89. Correlation between simulated well depths and screen lengths. 197

Figure 90. Discretization based on finite elements. 198

Figure 91. Histogram of degrees of freedom for the simulated subdomains. 199

Figure 92. Domain decomposition of study area into overlapping subdomains. 200

Figure 93. Illustration of weighting scheme between adjacent subdomains. 201

Figure 94. Computed streamlines based on the fine simulation model. 203

Figure 95. Unit Response Functions. 204

Figure 96. Streamlines based on coarse model and analytical particle tracking. 205

Figure 97. Comparison of statistics of streamline characteristics between the two simulation models 207

Figure 98. Comparison between streamlines computed with the detailed and coarse model. 208

Figure 99. Cumulative distribution of N loading per year. 211

Figure 100. Exceedance probabilities of N loading rates as nitrate 212

Figure 101. Cumulative distribution function of N loading based on the coarse model. 213

Figure 102. Exceedance probabilities of leaching rates as nitrate based on coarse model. 214

Figure 103. Decadal trends for the three regions based on measured data.. 215

Figure 104. Decadal trends of exceedance probabilities based on measured data.. 215

Figure 105. Comparison of concentration decadal trends between measured data and model predictions. 216

Adminsitrative Record
Page 34920

Figure 106. Comparison of decadal trends of exceedance probability of 9 mg/L, between measured data and model predictions. 218

Figure 107. Comparison of decadal trends of exceedance probability of 22.5 mg/L, between measured data and model predictions. 219

Figure 108. Comparison of decadal trends of exceedance probability of 45 mg/L, between measured data and model predictions. 220

Figure 109. Breakthrough curves based on model simulation for each well for the three alternative scenarios. 222

Figure 110. Comparison between coarse model and fine model based on scenario D. 223

Figure 111. Comparison between scenario A and C based on the fine model predictions. 224

Figure 112. Exceedance probabilities for 22.5, 45, and 90 mg/L based on the fine model. 225

Figure 113. Exceedance probabilities for 22.5, 45, and 90 mg/L based on the coarse model. 226

Adminsitrative Record
Page 34921

Acronyms and Abbreviations

ADE	Advection Dispersion Equation
AMBAG	Association of Monterey Bay Area Governments
ANOVA	Analysis of Variance
APN	Assessor's Parcel Numbers
AW	Applied Water
BMPs	Best Management Practices
BOR	Bureau of Reclamation
BTC	Breakthrough Curve
C2VSIM	California Central Valley Groundwater-Surface Water Simulation Model
Ca^{2+}	Calcium
CADWSAP	California Drinking Water Source Assessment Program
CAML	California Augmented Multisource Landcover
CASTING	California Ambient Spatio-Temporal Information on Nitrate in Groundwater
CDPH	California Department of Public Health
CDS	Compliance Discharge Surface
CFC	Chlorofluorocarbon
CIMIS	the California Irrigation Management Information System
CIP	Canal Integration Program
Cl^-	Chloride
CO_3^{2-}	Carbonate
COMSOL	Multiphysics Finite Element Analysis Simulation Software
CVHM	Central Valley Hydrologic Model
CVRWB	Central Valley Regional Water Board
CV-SALTS	Central Valley Salinity Alternatives for Long-Term Sustainability
CWS	Community Water System
DAU	Detailed Analysis Units
DBCP	Dibromochloropropane
DD	Domain Decomposition
DGW	Depth to Groundwater
DIP	Distribution Integration Program
DLPA	DWR Division of Local Planning and Assistance
DOC	Dissolved Organic Carbon
DPR	Department of Pesticide Regulations
DSR	Data Summary Reports
DWR	California Department of Water Resources
ecdfs	Empirical cumulative distribution functions
E-Clay	Corcoran Clay
EPA	(U.S.) Environmental Protection Agency
ET	Evapotranspiration

Adminsitrative Record
Page 34922

Eto	Evapotranspiration
FEFLOW	Simulation model for subsurface flow and transport processes
FEM	Finite Elements Method
FKC	Friant-Kern Canal
FREDSIM	FRiant Economics-Driven SIMulation model
GAMA	California State Water Board's Groundwater Ambient Monitoring and Assessment
GIS	Geographic Information system
GLEAMS	Ground Water Loading Effects of Agricultural Management Systems
GPS	Geographic Positioning System
GWQB	Ground Water Quality Bureau (New Mexico)
HCO_3^-	Bicarbonate
HR	Hydrologic Region
ICE	UC Davis Information Center for the Environment
IGSM	Integrated Groundwater and Surface-Water Model
IPCC	Intergovernmental Panel on Climate Change
IrrEff	Irrigation Efficiency
K^+	Potassium
KCWA	Kern County Water Agency
KDWCD	Kaweah Delta Water Conservation District
KRCD	Kings River Conservation District
K_C	standardized crop coefficient
L	liter
LPA	Local Primacy Agency
LR	Leaching Requirement
MAF	Million Acre-feet
MassDEP	Massachusetts Department of Environmental Protection
MCEM	MCWRA Agricultural Environmental Monitoring Program
MCL	Maximum Contaminant Level
MCWRA	Monterey County Water Resources Agency
MCDEH	Monterey County Department of Environmental Health
mg	milligram (one-thousandth of one gram)
µg	microgam (one-millionth of one gram)
Mg^{2+}	Magnesium
Na^+	Sodium
NASS	National Agricultural Statistics Service
NAWQA	National Water-Quality Assessment
ND	Non-detect
NMED	New Mexico Environment Department
NO_3^-	Nitrate
NOAA	National Oceanic and Atmospheric Administration
NPS	Nonpoint Source

Adminsitrative Record
Page 34923

NPSAT	Non-Point Source Assessment Tool
NTNCWS	Non-Transient Non-Community Water System
NWIS	National Water Information System (USGS)
PA	Principal Aquifers
Pdfs	Probability Distribution Functions
PICME	Permits, Inspections, Compliance, Monitoring, and Enforcement Database, CDPH
PLSS	Public Land Survey System
ppm	Parts per million
Precip	Precipitation
PS-Code	Primary Station Code
PWS	Public Water System
PWTA	Private Well Testing Act
RK	Regional Kendall
RWQCB	Regional Water Quality Control Board
SDWA	Safe Drinking Water Act
SID	State ID
SJV	San Joaquin Valley
SO_4^{2-}	Sulfate
SOM	Soil Organic Matter
STORET	"STOrage and RETrieval", EPA's database of water quality information
SV	Salinas Valley
SVB	Salinas Valley Basin
SVIGSM	Salinas Valley Integrated Ground Water and Surface Model
SWAP	Source Water Assessment Programs
SWN	State Well Number
SWP	State Water Project
SWRCB	State Water Resources Control Board
SWS	Small Water System
TDS	Total Dissolved Solids
TLB	Tulare Lake Basin
TNCWS	Transient Non-Community Water System
U.S. EPA	United State Environmental Protection Agency
URF	Unit Response Functions
USBR	United States Bureau of Reclamation
USDA	United States Department of Agriculture
USGS	United States Geological Survey
WGA	Well Generation Algorithm
WWD	Westlands Water District
YR	year

Adminsitrative Record
Page 34924

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-).

To convert from:

Nitrate-N (NO_3-N)	→	Nitrate (NO_3^-)	multiply by 4.43
Nitrate (NO_3^-)	→	Nitrate-N (NO_3-N)	multiply by 0.226

Adminsitrative Record
Page 34925

Summary

This report reviews the hydrogeology and groundwater quality in the Tulare Lake Basin (TLB) and Salinas Valley (SV). We also assembled groundwater quality data from nearly two dozen local, state, and federal agencies and other sources into a dataset, here referred to as the (Central) California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) dataset. The dataset combines nitrate concentrations from 16,709 individual samples taken at 1,890 wells in the Salinas Valley (SV) and from 83,375 individual samples taken at 17,205 wells in the Tulare Lake Basin (TLB) collected from the 1940s to 2011, accounting for a total of 100,084 samples from 19,095 wells. Almost 70% of these samples were collected from 2000 to 2010; only 15% of the samples were collected prior to 1990. Half of all wells sampled had no recorded samples prior to 2000.

Of the 19,000 wells, approximately 2,500 are frequently sampled public water supply wells (over 60,000 samples). Apart from the recently established Central Valley dairy regulatory program, which now monitors about 4,000 domestic and irrigation wells in the Tulare Lake Basin, there are no regular well sampling programs for domestic and other private wells. These latter are sampled sporadically by county agencies and through research programs.

From 2000 to 2011, the median nitrate concentration in the Tulare Lake Basin and Salinas Valley public water supply well samples was 23 milligrams per liter (mg/L) and 21 mg/L (as nitrate), respectively, and in all reported non-public well samples, 23 mg/L and 20 mg/L (as nitrate), respectively. In public supply wells, about one in ten raw water samples exceed the nitrate maximum contaminant level (MCL) of 45 mg/L (as nitrate). Nitrate concentrations in wells vary widely with location and well depth. More domestic wells and unregulated small system wells than public supply wells have high nitrate concentrations due to their shallow depth. The highest nitrate concentrations are found in wells of the alluvial fans in the eastern Tulare Lake Basin and in wells of unconfined to semi-confined aquifers in the northern, eastern, and central Salinas Valley. In the Kings, Kaweah, and Tule River groundwater subbasins of Fresno and Kings County, and in the Eastside and Forebay subbasins of Monterey County, one-third of domestic or irrigation wells exceed the nitrate MCL. Consistent with these findings, the maximum nitrate level, measured in any given land section (1 square mile) for which nitrate data exist between 2000 and 2009, exceeds the MCL across wide portions of these areas. Low nitrate concentrations tend to occur in the deeper, confined aquifer in the western and central Tulare Lake Basin.

Nitrate levels have not always been this high. While no significant trend is observed in some areas with low nitrate (e.g., areas of the western TLB), USGS research indicates significant long-term increases in the higher-nitrate areas of the Tulare Lake Basin, which is consistent with the CASTING dataset. Average nitrate concentrations in public supply wells of the Tulare Lake Basin and Salinas Valley have increased by 2.5 mg/L (±0.9 mg/L) per decade over the past three decades. Average trends of similar magnitude are observed in private wells. As a result, the number of wells with nitrate above background levels (> 9 mg/L) has steadily increased over the past half century from one-third of wells in the 1950s to nearly

Adminsitrative Record
Page 34926

two-thirds of wells in the 2000s. Due to the large increase in the number of wells tested across agencies and programs, the overall fraction of sampled wells exceeding the MCL grew significantly in the 2000s.

The increase in groundwater nitrate concentration measured in domestic wells, irrigation wells, and public supply wells lags significantly behind the actual time of nitrate discharge from the land surface. The lag is due, first, to travel time between the land surface or bottom of the root zone and the water table, which ranges from less than 1 year in areas with shallow water table (<3 m (10 ft)) to several years or even decades where the water table is deep (>20 m (70 ft)). High water recharge rates shorten travel time to a deep water table, but in irrigated areas with high irrigation efficiency and low recharge rates, the transfer to a deep water table may take many decades.

Once nitrate is recharged to groundwater, additional travel times to shallow domestic wells are from a few years to several decades, with travel times of one to several decades, and even centuries, for deeper production wells.

Denitrification (the natural attenuation of nitrate) is most likely to occur in fine-grained anoxic clay layers, the most prominent of which is the Corcoran Clay separating the upper semi-confined to unconfined aquifer from the lower confined aquifer along and next to both sides of the trough of the Tulare Lake Basin; the several thousand feet thick clay and silt units underlying the former bed of the Tulare Lake south of Hanford; and clay units confining the Pressure aquifer system in the northwestern Salinas Valley. These clay layers have limited effect on most groundwater production wells, which generally obtain most of their water through coarser sand and gravel aquifer sediments that are connecting recharge areas with wells. In very shallow groundwater discharge areas to surface water, denitrification may occur where significant sources of organic carbon are present, e.g., in riparian and marshland areas along the valley trough of the Salinas Valley and Tulare Lake Basin. Some removal of nitrate by denitrification may also occur in deeper reducing aquifer sediments occurring typically at more than 500 to over 1,000 feet depth throughout the Tulare Lake Basin. However, due to the large age of groundwater in deep anoxic zones, it is currently uncertain, to which degree anoxic conditions may slow down or prevent future nitrate pollution at that depth.

Adminsitrative Record
Page 34927

1 Introduction

Ninety-eight percent of the population in the Tulare Lake Basin and Salinas Valley rely on groundwater as a source of drinking water. Most of the study area's residents are provided drinking water through public water supply systems or small water systems that own and operate groundwater wells. About one in ten residents obtain their drinking water from private domestic wells. An assessment of current and future quality of groundwater used as drinking water is of critical interest to the public, and to local, state, and federal agencies charged with protecting water resources and providing safe drinking water.

Nitrate has long been known to be a widespread groundwater pollutant in California's groundwater basins. It is associated with fertilizer use, land application of manure and organic wastes, and disposal of urban and domestic sewage. The objective of Technical Report 4 is to review our current understanding of nitrate in groundwater in the Tulare Lake Basin and the Salinas Valley, and to provide a comprehensive assessment of past, current, and future distribution of nitrate.

This report begins with a review of the general characteristics of the Tulare Lake Basin and Salinas Valley physiography, hydrology, hydrogeology, and water quality (Sections 2 and 3). These sections describe and summarize the known occurrence and trends of nitrate in groundwater as described in research studies and government reports over the past half century.

In this study, we also made an effort to collect a comprehensive inventory of data on nitrate occurrence described in reports, housed in county and districts offices, available from state and federal databases, or through individual research groups. In Section 4, we describe the methods used to assemble the CASTING (California Ambient Spatio-Temporal Information on Nitrate in Groundwater) database. The database provides an efficient interim electronic dataset platform for the research team to perform spatial and trend analysis, to map nitrate occurrence, and - after further completion and quality control - for the transfer of the data to the State Water Resources Control Board's publicly accessible Geotracker GAMA groundwater quality database.

The database includes data from publicly accessible data sources as well as data that have never before been part of a thorough regional water quality evaluation, providing an opportunity to update previous studies and assessments, many of which have focused on local areas. With the CASTING dataset, we perform a statistical evaluation of historic and current nitrate occurrence across the study area (Section 5).

Field data are useful to understand current water quality and historic trends, where such data exist. But models are needed to assess groundwater quality where measurements have not been made, and to assess and predict future dynamics of groundwater nitrate, which result from past, current, and future nitrate loading to groundwater. Technical Report 2 (Viers et al. 2012) describes a century of nitrate loading, 1945 to 2050, distributed across the entire study area, reflecting the large spatial and temporal variability in nitrate loading. In this report, we use these data, in conjunction with a newly developed groundwater modeling tool to predict groundwater quality across the study area over the next 40 years. In Section 6 we develop a method to estimate nitrate travel times from the root zone (the point of

Adminsitrative Record
Page 34928

reference for the nitrate loading calculations described in Technical Report 2, ibid.) through the vadose zone and into groundwater. Section 7 describes the development of a new groundwater modeling tool, specifically to evaluate pollution from diffuse non-point sources in large number of wells distributed over a large aquifer system such as the Tulare Lake Basin. The model simulates the temporal dynamics of groundwater nitrate at thousands of irrigation and drinking water supply wells, from 1945 to 2050. Simulation results for the last 60 years are compared to data, where available to validate the modeling tool.

The appendices contain a general overview of the potential role of denitrification in naturally attenuating groundwater nitrate as well as information on domestic well monitoring programs across the United States. Denitrification is an important natural attenuation process. Further research is needed to determine the long-term (multi-decadal) influence of denitrification on groundwater nitrate distribution, particularly in the confined aquifer system of the central and western Tulare Lake Basin.

Adminsitrative Record
Page 34929

2 Tulare Lake Basin Hydrogeology and General Water Quality

Prepared by:

Dylan Boyle, Thomas Harter

Adminsitrative Record
Page 34930

2.1 Physical Setting

2.1.1 Location

The Central Valley of California ("Central Valley") covers a large and central portion of the state of California. The total area is approximately 58,000 km^2 (22,500 square miles), with a length of over 700 km (450 miles) and a width of 60 to 100 km (40 to 60 miles). It is characterized by minimal topographic expression, an essentially flat region with surface slopes less than 1% throughout most of its area. The Central Valley is further divided into its two major watersheds, the Sacramento Valley in the north and San Joaquin Valley in the south. The San Joaquin Valley is further divided into the northern externally-draining San Joaquin River Basin and the southern internally-draining Tulare Lake Basin (TLB). The San Joaquin Basin drains into the Sacramento – San Joaquin River Delta (the Delta), by way of the San Joaquin River. Rivers entering the endorheic[2] TLB terminate in Tulare Lake, which has been drained and reclaimed for irrigated agriculture since the mid-20th century. The TLB is the widest part of the Central Valley (Figure 1).

[2] Endorheic refers to a closed basin which has no outflow to other bodies of water.

Adminsitrative Record
Page 34931

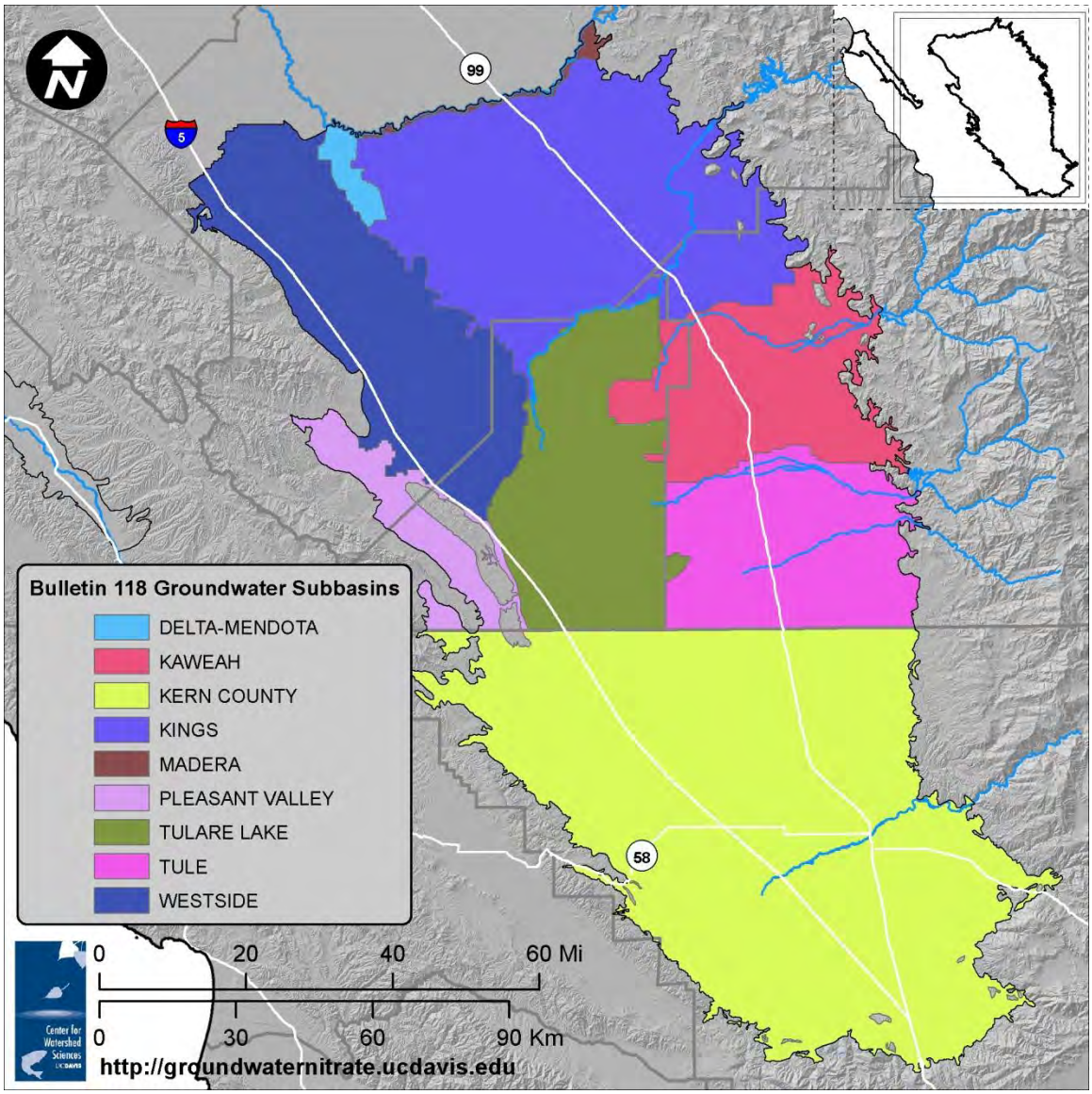

Figure 1. Hydrologic subbasins of the TLB, California. (Source: DWR 2003.)

2.1.2 Hydrologic Subbasins

The TLB comprises over 20,000 km^2 (8,000 square miles) of the valley portion of the Tulare Lake Hydrologic Region (HR) as defined in California Department of Water Resources (DWR) Bulletin 118 (California Department of Water Resources 2003). The northern boundary of the TLB is defined by the westward flowing San Joaquin River upstream and east of the city of Firebaugh and by a shallow watershed divide to the west of Firebaugh. The Kettleman Hills and the Temblor Range of the Coast Ranges of California form the western boundary of the TLB. To the south, the TLB is bordered by the Tehachapi Mountains. The Sierra Nevada foothills form the eastern boundary. The crystalline bedrock formations of the Sierra Nevada slope in a southwesterly direction beneath the sediments of the TLB, forming its lower boundary. The TLB is divided into subbasins based on geology, hydrologic barriers, and

institutional boundaries. The groundwater subbasins within the study area are listed in Table 1 and shown in Figure 1. Over the past century, all major rivers in the TLB have been regulated with dams and reservoirs located along the Sierra Nevada foothills. Since the mid-20th century, most stream runoff has been used for irrigation and lakes have been drained and used for irrigated agricultural production.

Table 1. Subbasins within the TLB, California. (Source: DWR 2003.)

Subbasin Name	Subbasin Number	Subbasin Area km^2 [mi^2]
Kings	5-22.08	3,950 [1,530]
Westside	5-22.09	2,590 [1,000]
Pleasant Valley	5-22.09	588 [227]
Kaweah	5-22.11	1,803 [696]
Tulare Lake	5-22.12	2,117 [818]
Tule	5-22.13	1,898 [733]
Kern County	5-22.14	7,874 [3,040]

Adminsitrative Record
Page 34933

2.2 Hydrology

2.2.1 Overview

The region is characterized by a semi-arid to arid Mediterranean climate with most precipitation occurring during the winter months and nearly no precipitation during the hot summer months. Much of the water supply in the TLB has its origins in the Sierra Nevada snowpack, and to a much lesser extent in precipitation that falls in the Coast Ranges and Tehachapi Mountains. Winter storms deposit large quantities of snow in the Sierra Nevada and spring melt water discharges into the Valley by way of the TLB's major rivers.

The major rivers of the TLB are the Kings River, the Kaweah River, and the Kern River. The three rivers originate from steep, mountainous watersheds of the snow-covered Sierra Nevada Mountains east of the TLB where the highest elevation (Mt. Whitney) is nearly 4,500 m (15,000 ft). In the past, the Kings, Kaweah, and Tule Rivers, under natural conditions, drained directly into Tulare Lake, while the Kern River drained into Kern and Buena Vista Lakes. All three lakes were historically terminal lakes that occasionally connected via surface sloughs and, at very high water stage, drained into the San Joaquin River via Fresno Slough. Now that the lakes have been drained, and development of groundwater has lowered the water table, the basin is largely considered to have no natural outflow of water from the basin (except for evapotranspiration).

2.2.2 Water Budget

Spring runoff is captured in large reservoirs along the Sierra Nevada foothills, and subsequently released into a network of natural stream channels and canals that serve irrigation and natural water needs and recharge groundwater. The largest canals in the TLB are the Friant-Kern Canal (FKC) along the entire eastern edge of the TLB, and the State Water Project (SWP) and San Luis Canal along the western and southern edge of the TLB. The canal networks associated with the FKC and SWP connect in the southern TLB. Together they are part of the backbone of California's modern water transfer network: The FKC transfers San Joaquin River (and some Kings River) runoff from their respective reservoirs to irrigation districts along the east- and southeast-side of the TLB. The SWP imports water from the much wetter Sacramento Valley via the Delta to irrigation districts in the western TLB and to other southern California regions.

Ideas to transfer "excess" water from the Sacramento Valley to the San Joaquin Valley were formulated as early as 1870s (California Department of Water Resources 1994). Over time, various private and government schemes have transformed the natural hydrology of the Central Valley. Surface water was used for irrigation needs as early as the 1700s (Bertoldi et al. 1991), but as agricultural efforts intensified, it was realized that this water had to be supplemented. The Miller and Lux agricultural enterprise formed in the mid 1800's and by 1900 several canals had been constructed for the purpose of delivering water to the southern San Joaquin Valley (Igler 2001). Groundwater resources started being developed in the 1880s (Bertoldi et al. 1991) and allowed cities to begin to flourish in the San Joaquin

Adminsitrative Record
Page 34934

Valley. As early as 1900, groundwater levels had fallen, requiring the development of larger extraction pumps to withdraw the deeper water. Around 1930, the improved deep well turbine pump was developed concurrently with the expansion of rural electrification (Galloway et al. 1999). As a consequence, water could be pumped from an even greater depth, and larger yields could be attained. Shortly thereafter, the invention of the Haber-Bosch process (which created nitrate from nitrogen gas present in the atmosphere) led to large-scale production of industrial nitrogen fertilizer. As a result, the mid-20th century began a period of quickly expanding and intensifying irrigated agricultural production. Groundwater pumping was excessive, particularly along the western TLB and in the southern TLB, due to the lack of significant surface water inflows to these areas. Importation of northern California water via the Sacramento River, the Delta, and the SWP began in the late 1940s which alleviated groundwater overdraft temporarily. In the 1990s and 2000s, restrictions on water transfers from the Sacramento Valley to the SWP, due to ecological concerns in the Delta combined with a series of droughts, led to a reduction in surface water imports and a resurgence of groundwater overdraft.

Today the Central Valley aquifer system, if treated as a single aquifer, is the second most highly pumped aquifer in the United States (Faunt 2009). The TLB accounts for approximately 35% of California's total annual groundwater withdrawal, roughly 5 cubic kilometers (4.34 million acre-feet (MAF)) (California Department of Water Resources 2003).

Several models have been created to understand the water budget for the region. Two established models for the Central Valley are the Central Valley Hydrologic Model (CVHM) and the California Central Valley Groundwater-Surface Water Simulation Model (C2VSIM). Each model also calculates water budgets for sub-regions within the Central Valley. Table 2 shows the groundwater budgets for the Central Valley as a whole, and for the TLB. The results from both models show that the TLB and the Central Valley as a whole have been losing groundwater (groundwater storage) for the past several decades. For the Central Valley, the results shown for CVHM indicate an average annual loss of 1.604 km^3 (1,300,000 acre-ft/year) over the years 1962-2003 (Faunt 2009), and the results shown for C2VSIM indicate an annual average loss of 2.491 km^3 (2,020,108 acre-ft) for the years 1962-2003 (Table 2) (Brush 2012). The final report for C2VSIM has not been published and values shown should be considered preliminary results.

For the TLB, the models calculate similar losses of storage. For the TLB, CVHM estimates an annual average loss in storage of -2.005 km^3 (-1,626,000 acre-ft) (Faunt 2009) and C2VSIM estimates an annual loss of -2.022 km^3 (-1,639,582 acre-ft) (Table 2) (Brush 2012). According to both models, the TLB has the greatest average storage loss out of all the subbasins for the years modeled. Of course, the reason the models can show a greater net loss in storage for the TLB, compared to the Central Valley as a whole, is due to other subbasins having net groundwater gains through time.

Four additional models have been recently developed for modeling water budgets of smaller regions within the TLB; the models are the Kings Integrated Groundwater and Surface Water Model (Kings IGSM), the Kaweah Delta Water Conservation District Model, the FRiant Economics-Driven SIMulation model (FREDSIM), and a model of the Kern Water Bank (Mellier et al. 2001; Ruud et al. 2003; Wrime, Inc. 2007).

Adminsitrative Record
Page 34935

Table 2 provides water budgets for the Kings IGSM, FREDSIM, and Kaweah Delta model. Model results show an average net loss of water for all three regions. The FREDSIM model has an economic framework, and the range of average storage loss shown is based on different economic scenarios. A water budget for the Kern County Water Bank model was not provided in the modeling report; however, Kern County Water Agency (KCWA) reports yearly water budget estimates for the water bank (PBS&J Inc. 2007). From 1995 to 2005, estimates show an annual average increase in storage of almost 100,000 acre-ft.

Table 2. Annual average groundwater budgets from hydrologic models in the Central Valley.

Model	Average Annual River Recharge km^3 [acre-ft]	Average Annual Recharge (except river percolation) km^3 [acre-ft]	Average Annual Groundwater Pumpage km^3 [acre-ft]	Average Annual Net Storage Change km^3 [acre-ft]
Central Valley Hydrologic Model (CVHM), USGS	0.3700 [300,000]	9.374 [7,600,000]	-11.471 [-9,300,000]	-1.604 [-1,300,000]
CVHM (Tulare), USGS	0.865 [701,000]	3.932 [3,188,000]	-6.968 [-5,649,000]	-2.005 [-1,626,000]
California Central Valley Groundwater-Surface Water Simulation Model (C2VSIM)**, DWR**	-0.173 [-140,095]	8.886 [7,203,690]	-11.157 [-9,045,364]	-2.491 [-2,020,108]
C2VSIM (Tulare Basin)**, DWR**	0.127 [102,606]	4.306 [3,491,054]	-6.459 [-5,236,202]	-2.022 [-1,639,582]
Kings Integrated Groundwater and Surface Water Model, KRCD	0.397 [321,700]	1.626 [1,318,500]	-2.223 [-1,802,000]	-0.200 [-161,900]
Kaweah Delta Water Conservation District Model, KDWCD	0.083 [67,000]	-0.104 [-84,000*]	-	-0.021 [-17,000]
FRiant Economics-Driven SIMulation model (FREDSIM], UC Davis	-	-	-	-0.155 to -0.355 [-126,000 to -288,000**]
Kern Water Bank, DWR	-	-	-	0.118 [95,525***]

*For the Kaweah model, the Average Annual Recharge value listed represents the net recharge which includes groundwater pumping. **The range of estimated groundwater storage change is based on different economic scenarios (e.g., pumping costs and surface water delivery costs). ***Storage change is based on KCWA estimates. ****Values shown for C2VSIM should be considered preliminary results.

Adminsitrative Record
Page 34936

2.2.3 Groundwater Levels

Groundwater levels generally fluctuate annually. This is largely due to groundwater pumping in the summer, causing a lowering of water levels, and subsequent recharge in the winter and spring, providing an increase in storage and raising groundwater levels. Levels also change through time on longer scales, due to periods of drought when groundwater is heavily pumped, and conversely, during periods of high precipitation when greater amounts of surface water are used and groundwater is not relied on as much. The Department of Pesticide Regulation, while investigating groundwater vulnerability to surface contamination, required an estimated depth to groundwater (DGW) coverage for the Central Valley (Spurlock 2000). For the purpose of their study, average DGW coverage was determined for California using spatial DGW measurements. Taken in the months of January through May, 260,000 later winter-spring DGW measurements were selected to represent water table levels. Kriging was used to interpolate between data points, and interpolation was restricted to distances less than 3.2 km (2 miles) from raw data boundaries. A map of the DGW coverage is shown in Figure 2. The primary source of the DGW data was the DWR Division of Local Planning and Assistance (DLPA). Other sources included the United States Geological Survey, San Benito County Water District, Santa Clara Valley Water District, and the Monterey County Water Resources Agency (Spurlock 2000). Although groundwater levels fluctuate through time, the map in Figure 2 shows areas where DGW is generally shallow, and other areas where it is considerably deeper. This has a direct impact on nitrate travel times to wells, as the hydraulic conductivity of unsaturated soils (above the water table) is much lower when compared to their saturated equivalent. The reader is referred to Section 6 *Nitrate Travel Time Through the Vadose Zone in the Tulare Lake Basin and Salinas Valley* for a more in-depth discussion on vadose zone transport.

Adminsitrative Record
Page 34937

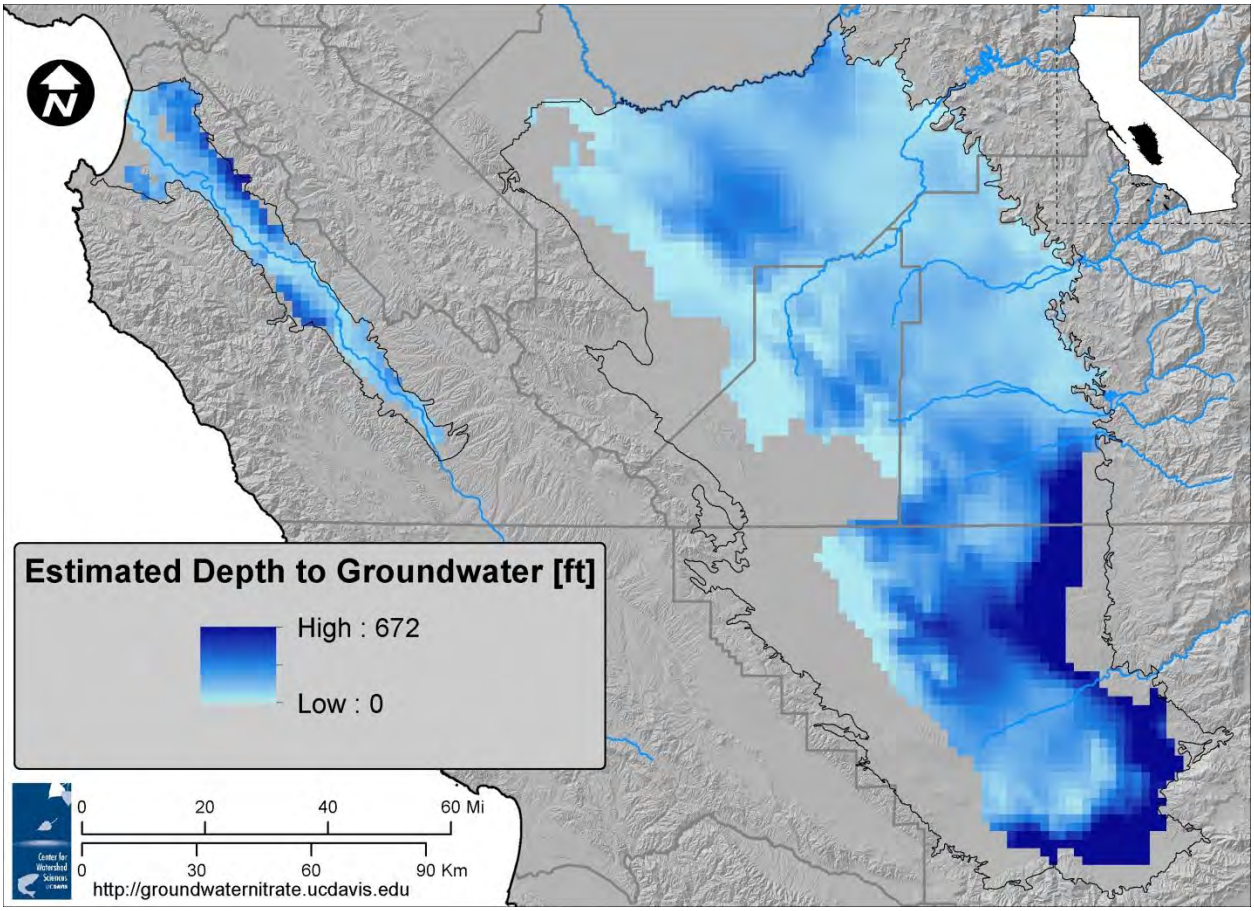

Figure 2. Depth to groundwater for the TLB and SV. (Source: Spurlock 2000.)

Adminsitrative Record
Page 34938

2.3 Hydrogeology

2.3.1 Geology

The Sierra Nevada Mountains to the east of the TLB are formed mainly of pre-Tertiary rocks of igneous and metamorphic origin. Sloping southwesterly from the foothills, the granitic bedrock lies beneath the fluvial and alluvial sediments which comprise the Central Valley aquifer systems. The Coast Ranges, to the west of the TLB, were formed by complex folding processes due to the convergent plate boundary to the west, and are both marine and volcanic in origin. The consolidated sediments of the Coast Ranges are of Jurassic, Cretaceous, and Tertiary age (Croft & Gordon 1968; Croft 1972; Planert & Williams 1995; Faunt 2009).

The Central Valley is a structural trough, with largely unconsolidated to semi-consolidated alluvial and fluvial sediment covering the basement complex (Faunt 2009). These sediments comprise the framework which stores the Valley's groundwater supplies. The aquifer systems of the TLB are generally composed of coarser sandy and gravelly sediments within a framework of finer-grained sediments (Weissmann et al. 2005). Through time, rivers and smaller streams emerging from the foothills of the Sierra Nevada and Coast Ranges have carried sediments from the mountains and deposited them on the valley floor, forming alluvial fans consisting of complexly arranged stream bed deposits (mostly sands, gravels); overbank deposits (sands and silts); and flood basin deposits (silts and clays), with inter-bedded lacustrine lakebed sediments (clays) (Weissmann et al. 2005). The coarse grained sediment bodies within the subsurface aquifers are the portions which conduct water the easiest and provide the majority of the water to pumping wells.

Throughout the San Joaquin Valley and the TLB, the shallower aquifer systems are generally composed of alluvial deposits derived from the Coast Ranges and Sierra Nevada Mountains, and flood-basin deposits near the valley trough (Lauden & Belitz 1991). Coast Range alluvium (derived from marine-origin sedimentary formations) skirts the western-most portion of the TLB, while the central and eastern portion of the TLB is dominated by the alluvial fans and plains formed by the streams discharging sediments from the granitic Sierra Nevada mountains (Figure 4) (Miller et al. 1971; Belitz & Heimes 1990; Weissmann et al. 2005; Faunt 2009). Coast Range alluvial deposits interfinger with Sierra Nevada alluvial deposits beneath the valley trough to form a heterogeneous organization of sediments. The Sierra Nevadan alluvial sediments are generally coarser in texture than the Coast Range alluvium, due to their granitic origin, as compared to the marine sediments derived from the Coast Range (Miller et al. 1971; Belitz & Heimes 1990).

Fine-grained lacustrine, paludal, and flood-basin deposits also underlie the valley trough. These finer-grained deposits are hydrologically important, clay and silty-clay deposits, and have been labeled as the A- through F-Clays. The E, C, and A-Clays comprise a significant portion of the TLB aquifer system framework (Croft & Gordon 1968; Croft 1972). The Corcoran Clay (E-Clay) is a large confining unit in the western portion of the valley and its spatial distribution is shown in Figure 3 as it is modeled in CVHM (Faunt 2009). It forms the upper confining boundary for many of the aquifers in the western Central

Adminsitrative Record
Page 34939

Valley. Historically, the Corcoran Clay was thought to be a continuous impermeable layer which separated the upper unconfined aquifer and the lower confined aquifer. Figure 3 shows that, although it is relatively continuous over a large area of the Central Valley, it has significant spatial variability within it. Areas containing larger fractions of coarser grained material likely provide an exchange of water between the upper and lower aquifers compared to areas with no coarse grained sediments. More importantly, studies suggest that the development of groundwater has likely increased the connection between the aquifers via well bore holes, creating unimpeded pathways through the confining layer (Williamson et al. 1989; Belitz & Phillips 1995).

Weissmann et al. (2005) performed a study of the role that alluvial and fluvial fans have played in the creation of San Joaquin Valley sediment fill. In common with the rest of the San Joaquin Valley, the eastern flanks of the TLB are characterized by many large fluvial fans that coalesce before terminating at the valley trough. In the TLB, the major fans are those that originate from the larger rivers of the Sierra Nevada Mountains.

Adminsitrative Record
Page 34940

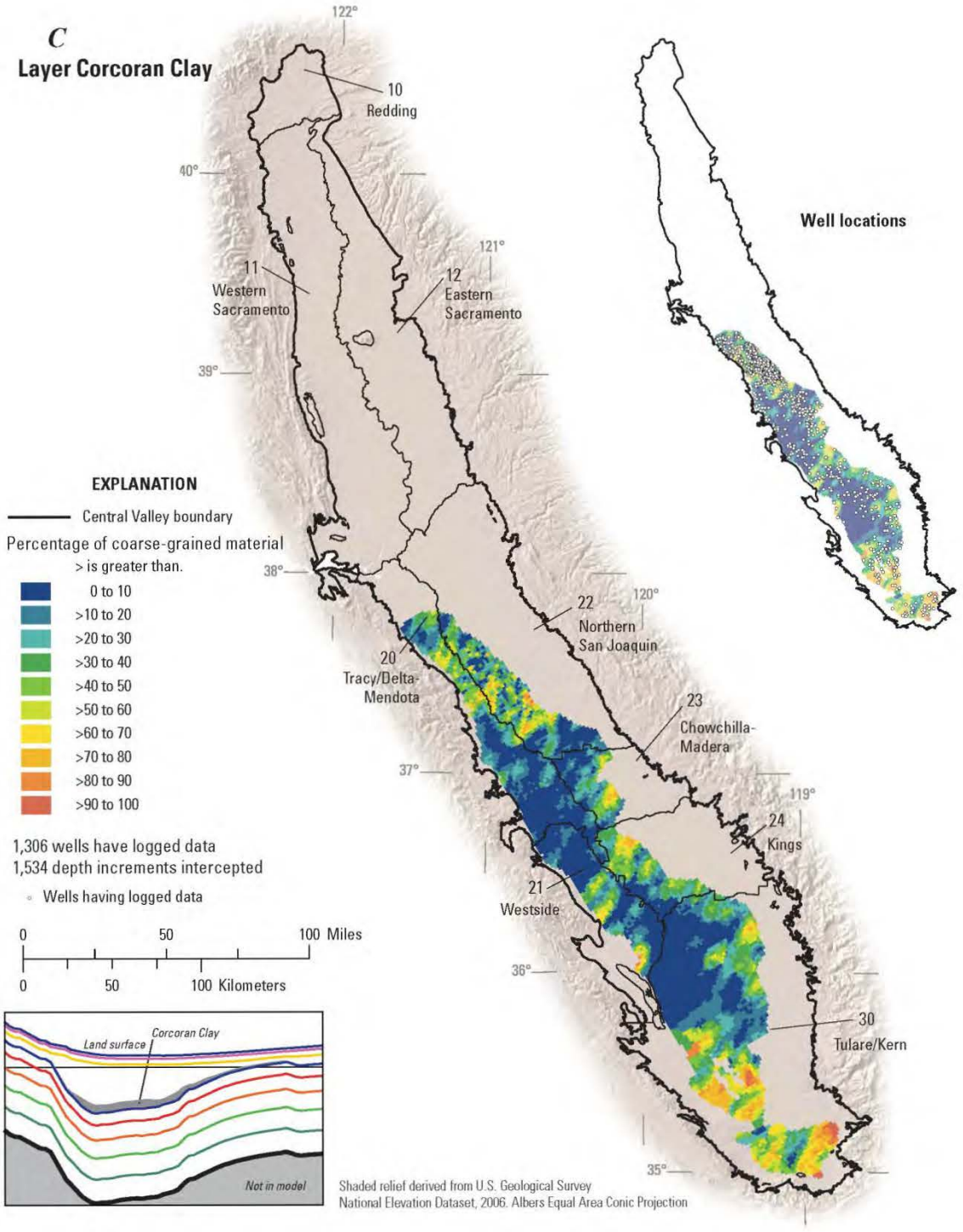

Figure 3. Spatial distribution of sediments of the Corcoran Clay as modeled in the Central Valley Hydrologic Model (CVHM). (Source: Faunt 2009.)

Adminsitrative Record
Page 34941

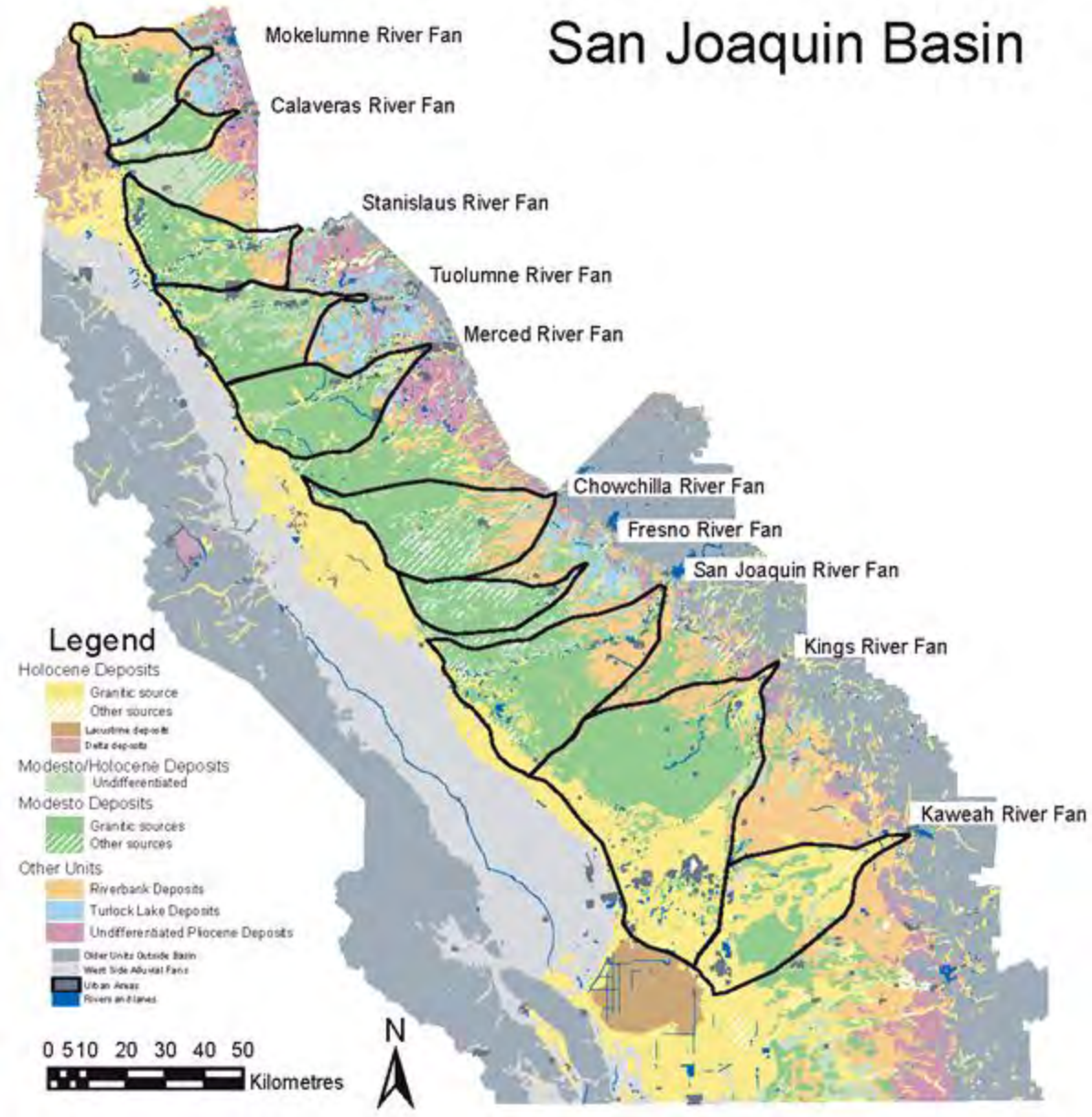

Figure 4. Fans of Northern and Central San Joaquin Valley from *Alluvial Fans: Geomorphology, Sedimentology, Dynamics*, Geological Society of London Special Publication 251, Geological Society, London. The Kings and Kaweah River Fans are in the TLB. (Source: Weissmann et al. 2005.)

The thickness of aquifers in the TLB varies throughout the basin. The aquifers that contain fresh groundwater are generally thickest in the southern portion of the TLB, in the vicinity of Bakersfield. The thickness of aquifer material bearing freshwater is on average 600m (2,000 ft), but can locally be greater than 4,000 ft (Planert & Williams 1995). Beneath the continental deposits are sediments of marine origin (Faunt 2009). Figure 5 from Faunt (2009) shows a conceptual diagram of the aquifer system.

Adminsitrative Record
Page 34942

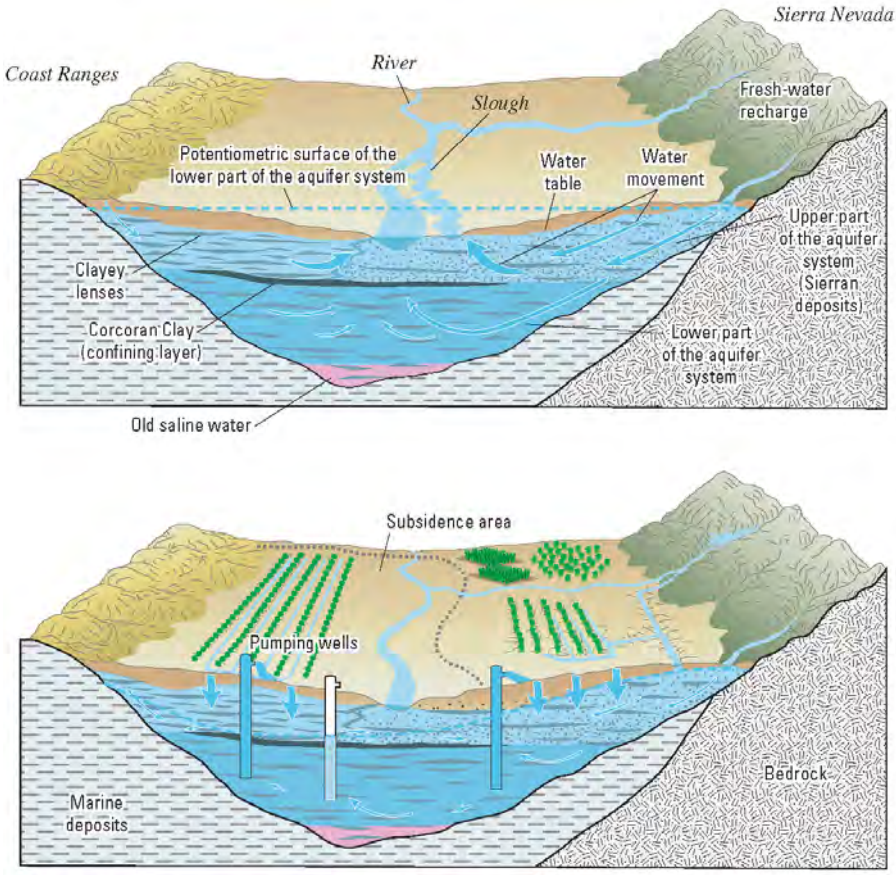

Figure 5. Conceptual diagram of the San Joaquin aquifer system. The first image (top) shows pre-development conditions where groundwater typically discharged to surface water bodies. The second image (bottom) shows that the mechanism for discharge from the aquifer is now dominated by groundwater pumping. (Figure from: Faunt 2009, as modified from Belitz & Heimes 1990; Galloway et al. 1999.)

The marine sediments, deposited in an ocean environment, often contain old saline water. In general, the freshwater portion of the aquifer system is comprised of much younger water than that of the deep marine aquifers. Although freshwater is generally found in continental deposits, in certain areas it can be hard to delineate between the marine and continental deposits (Bertoldi et al. 1991). In the western portion of the TLB, the regionally elevated-TDS groundwater found in the freshwater aquifer systems is the result of the water flowing through the Coast Range alluvial fans composed of marine sediments (Deverel & Gallenthine 1988).

For the regional nitrate analyses in Section 5, the TLB was divided into three separate regions based on sediment origins. The three groups, referred to as the Eastside Alluvial Fans, Basin, and Westside Alluvial Fans groundwater regions, were delineated by the USGS and refer to the origin of the alluvial sediments present in these regions (Burow et al. 1998). The role of denitrification (the natural attenuation of nitrate) in these regions is further explained in Appendix B of this report.

Adminsitrative Record
Page 34943

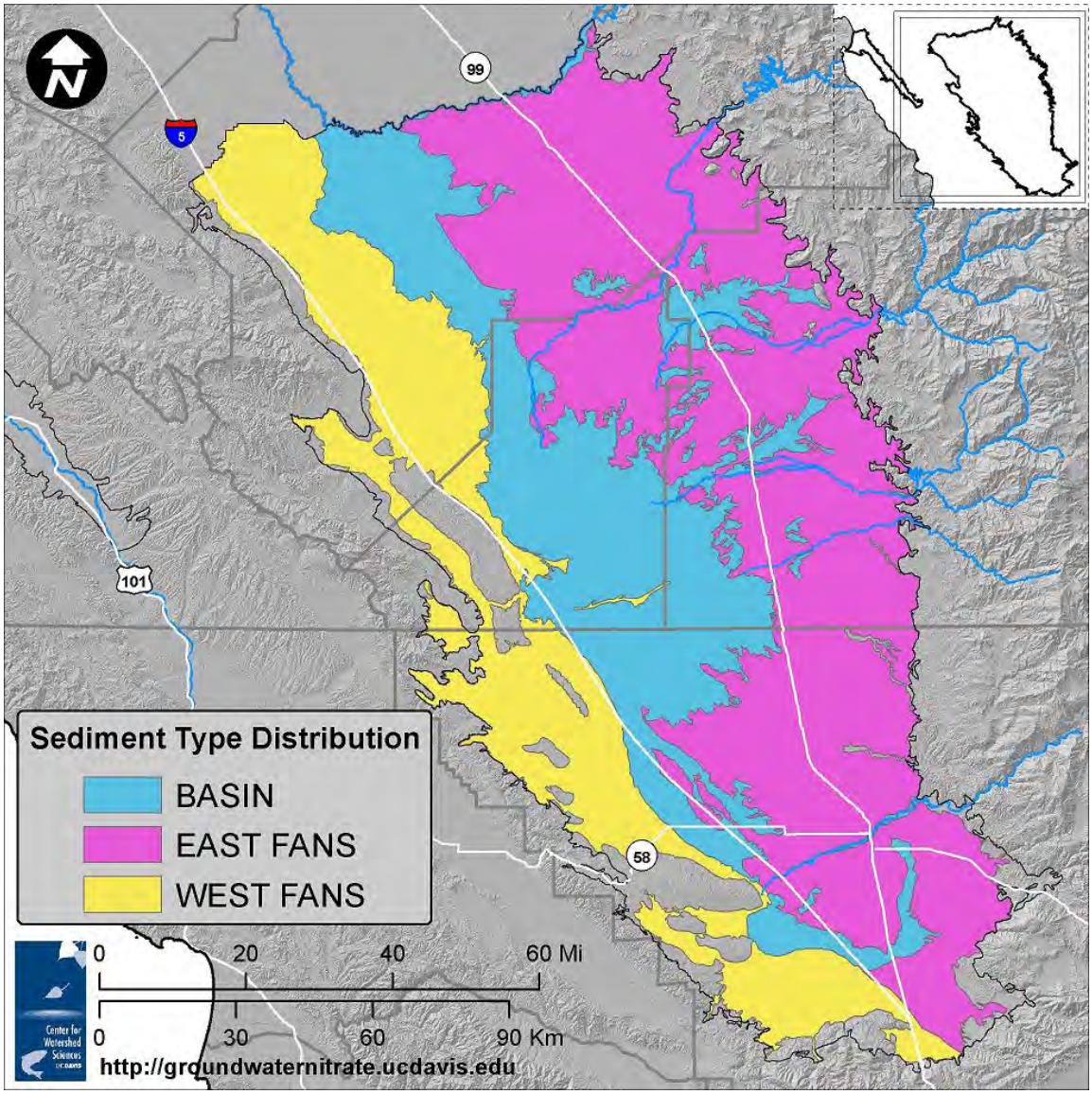

Figure 6. Based on the sediment origins, the TLB is divided into three groundwater regions: the Eastside Alluvial Fans, the Basin, and the Westside Alluvial Fans region.

2.3.2 Groundwater Recharge

Groundwater recharge in the study area comes mainly from irrigation return flow and surface sources such as streams and managed recharge projects. The majority of surface water reaches the groundwater by first being applied as crop irrigation. Groundwater pumping is the main outflow from the groundwater basin. Estimates of groundwater recharge and discharge for the TLB can be found in Table 2 of Section 2.2.2 *Water Budget*.

Some cities and water districts have attempted to counteract declining groundwater storage through artificial recharge programs. These are usually situated over areas within the TLB having comparatively

Adminsitrative Record
Page 34944

higher-permeability sediments in the shallow subsurface. Kern County, which has a highly valuable agricultural economy while also being very arid, uses surface water from sources such as local streams, the California Aqueduct, and the Friant-Kern Canal for "groundwater banking" (Mellier et al. 2001). When "excess" surface water is available during the wet season, it is transported to suitable areas and allowed to seep into the subsurface to be withdrawn later from the aquifer when it is needed. Major artificial recharge projects are currently operated by the Arvin-Edison, Kern, and Semitropic Water Storage Districts. These groundwater banking operations have a combined storage capacity of approximately 3 million acre-feet (more than 5 times greater than Millerton Lake[3]) (California Department of Water Resources 2003). The success of these projects has led to other districts proposing their own, such as the Madera Ranch Project to the north. The City of Visalia already has its own groundwater recharge program.

2.3.3 Flow Modeling

Section 2.2.2 *Water Budget* provides the water budget results from several groundwater models which encompass all or a portion of the TLB. As a part of this study, a steady state model was developed using input stresses (e.g., groundwater recharge and pumping) from the CVHM model for the purpose of modeling the transport of nitrate in the subsurface. The reader is referred to Section 7 *Nitrate Occurrence: Groundwater Transport Modeling* for more information on the development of the steady state model.

2.3.4 Transport Modeling

To date, there is no groundwater model that models nitrate transport for the TLB. A Non Point Source Assessment Tool (NPSAT) was developed as a part of this study to model nitrate transport at the basin scale and to investigate variable source loading scenarios in regards to nitrate. The reader is referred to Section 7 *Nitrate Occurrence: Groundwater Transport Modeling* for more information.

[3] Millerton Lake is an artificial reservoir constructed on the San Juaquin River near the town of Friant for surface water storage.

Adminsitrative Record
Page 34945

2.4 Water Quality

Generally, the groundwater found within the aquifer system as outlined above is suitable for the majority of municipal and agricultural uses, though local water quality impairments can be found. The primary groundwater quality impairments are commonly due to high TDS, nitrate, arsenic, selenium, boron, pesticides and herbicides, and organic compounds (Planert & Williams 1995; California Department of Water Resources 2003). The source of these constituents of concern varies; some are naturally occurring and others originate from anthropogenic activities.

2.4.1 Natural Contaminants

Fresh water, suitable for water supply needs, is defined as having a TDS measurement of less than 2,000 mg/L. Higher levels are defined as brackish and saline. Saline and connate[4] water can be found within the fresh water-bearing continental deposits; most saline and connate water is below the fresh water. This saline water comes from a variety of potential sources, including upward migration of old marine water (present during the deposition of the marine sediments) or through the process of evaporative concentration (Farrar & Bertoldi 1988).

The marine sediments that make up the Coastal Ranges naturally contain a variety of constituents that are of concern in surface and groundwater within the Central Valley. Through dissolution of the marine sediments, minerals and ions are released, and water flowing through such sediments increases in TDS. This has consequentially lead to elevated TDS levels in much of the west-side of the TLB (Deverel & Gallenthine 1988). This is in contrast to the east-side, where water originates in the Sierra Nevada Mountains. The granitic rocks of the Sierra Nevada Mountains dissolve at a much slower rate than the marine sediments and therefore contribute much less mineral content (TDS) over time (Planert & Williams 1995).

In the vertical direction, TDS readings generally start high at the water table, due to the infiltration of high-TDS agricultural return water from the surface, and decrease with depth, until relatively pristine pre-modern (<1900's) age water is reached. Below this, in the deeper portions of the subsurface, salinity again increases due to the presence of old water with high concentrations of dissolved minerals, as well as connate seawater present from when the marine sediments were deposited.

High TDS water can be a problem in deep wells, which are commonly found in the western and southern portions of the TLB. These deep wells, while trying to avoid pumping the shallow contaminated water, may penetrate deep enough to withdraw high-TDS groundwater or water that has been affected by high salinity water. In addition to the naturally accumulated TDS in waters found in the valley trough, evaporation close to the land surface concentrates the groundwater leading to further elevated TDS. In the western and central portions of the TLB, accumulation of solutes in shallow groundwater has had negative effects on agricultural production. The Central Valley Salinity Alternatives for Long-Term Sustainability (CV-SALTS) coalition, a non-profit organization, was formed in 2008 to address salinity

[4] Water present when the aquifer sediments were deposited.

Adminsitrative Record
Page 34946

issues in the Central Valley. For more information on salinity in California, visit the CV-SALTS website at http://www.cvsalinity.org/.

Another natural contaminant, selenium, is also found within the marine Coast Ranges sediments. Selenium can be concentrated by evapotranspiration in western and central portions of the TLB when this water is used for irrigation (Deverel & Gallenthine 1988). Selenium is toxic to many living creatures, and has the potential to bioaccumulate. A well-documented case of selenium contamination occurred at the Kesterson Wildlife Refuge further to the north, where selenium caused harm to waterfowl, fish, insects, plants, and algae (Ohlendorf et al. 1990).

Arsenic is an additional contaminant of recent concern. Arsenic concentrations are often the result of water with iron- or manganese-reducing conditions, which provide conditions that dissolve iron and manganese oxides present on sediments containing sorbed arsenic. Oxic alkaline (high pH) water also has the potential to desorb arsenic from oxides present on sediments (Belitz et al. 2003; Welch et al. 2006). The dry lake-beds within the TLB, namely Tulare, Kern, and Buena Vista are known to have elevated arsenic.

2.4.2 Groundwater Chemistry

As discussed before, the sediments through which groundwater travels can affect the chemistry of the water through dissolution of the minerals present. The following three piper diagrams plot groundwater samples from California Department of Public Health (CDPH) wells for the three geological regions of the TLB; the Eastside Fan, Westside Fan, and Basin sediments (Figure 6, Section 2.3.1). Although there is considerable scatter to the data, due to the large physiographic region of our study area and the natural heterogeneity of the aquifer, the samples generally reflect the aquifer sediments present in these regions.

The major ions present in water are calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), chloride (Cl^{-}), sulfate (SO_4^{2-}), carbonate (CO_3^{2-}), and bicarbonate (HCO_3^{-}). The relative amounts of each constituent, when plotted on a piper diagram (Figure 7-9), provide information on regional flow paths and origins of the water. For example, Ca^{2+} and Mg^{2+} concentrations are often elevated in shallow groundwater and decrease along flow paths due to exchange for Na^{+} present on clay-rich sediments, resulting in more Na^{+} and less Ca^{2+} and Mg^{2+} the longer water has spent traveling through the subsurface. Additionally, groundwater tends to continuously dissolve carbonate and other minerals found naturally in geological materials as it travels through the aquifer system. This results in greater amounts of HCO_3^{-} in older groundwater.

Samples from Eastside Alluvial Fans sediments, while covering the broad spectrum of water types, are concentrated in the left portion of the figure, meaning that the water contains little sulfate, and higher portions of calcium and magnesium (Figure 7). This is a reflection of the granitic sediments coming from the Sierra Nevada Mountains.

Adminsitrative Record
Page 34947

Samples from Westside Alluvial Fans sediments show the opposite. While the number of samples is significantly less than those from the Eastside Alluvial Fans sediments, the majority of the samples plot in the upper portion of the piper diagram. This signifies that the groundwater samples contain elevated levels of sulfate compared to the samples taken from the Eastside Alluvial Fans sediments. This reflects the origin of the Westside Alluvial Fans sediments, having primarily a marine origin.

The Basin sediments represent a mixture of both west and east fans sediments . The TLB as a whole, however, is dominated by Eastside Alluvial Fans sediments. The samples taken from the basin sediment region shows that the water chemistry is more similar to the water found in the east fans as compared to the water found in the west sediments. Groundwater from this region generally contains more calcium and magnesium and less sulfate.

In all three regions, nitrate is also plotted with the major ion chemistry. As mentioned before, as water travels through aquifer sediments, it generally exchanges calcium and magnesium for sodium. Water containing more sodium compared to calcium and magnesium can generally be assumed to have been present in groundwater longer than water containing little sodium, compared to calcium and magnesium. In the basin and Eastside Fans sediments the piper plots show that the “older” water generally contains lower nitrate than the “young” water. Section 3.4 provides more in depth discussion of groundwater chemistry, age, and nitrate contamination in the context of the Salinas Valley. Appendix B discusses the role of denitrification.

Adminsitrative Record
Page 34948

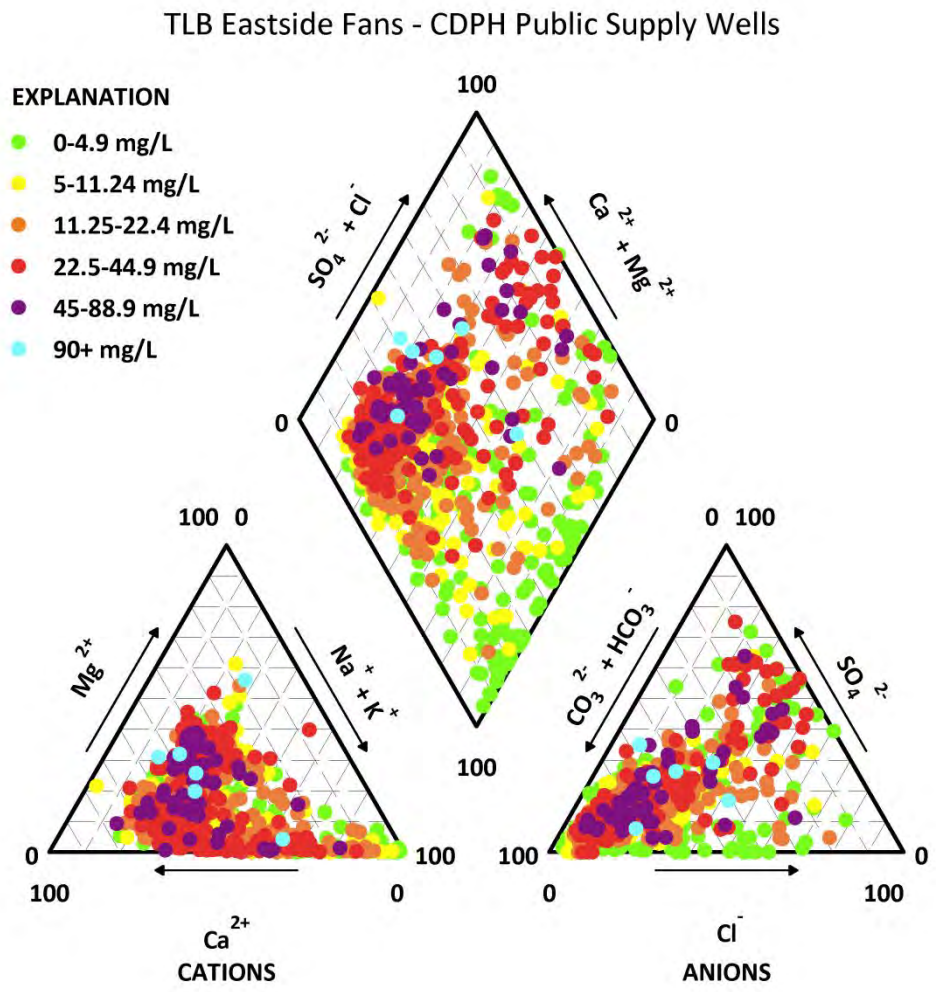

Figure 7. Piper diagram of groundwater samples from Eastside Fans sediments. Concentrations are of nitrate as nitrate.

Adminsitrative Record
Page 34949

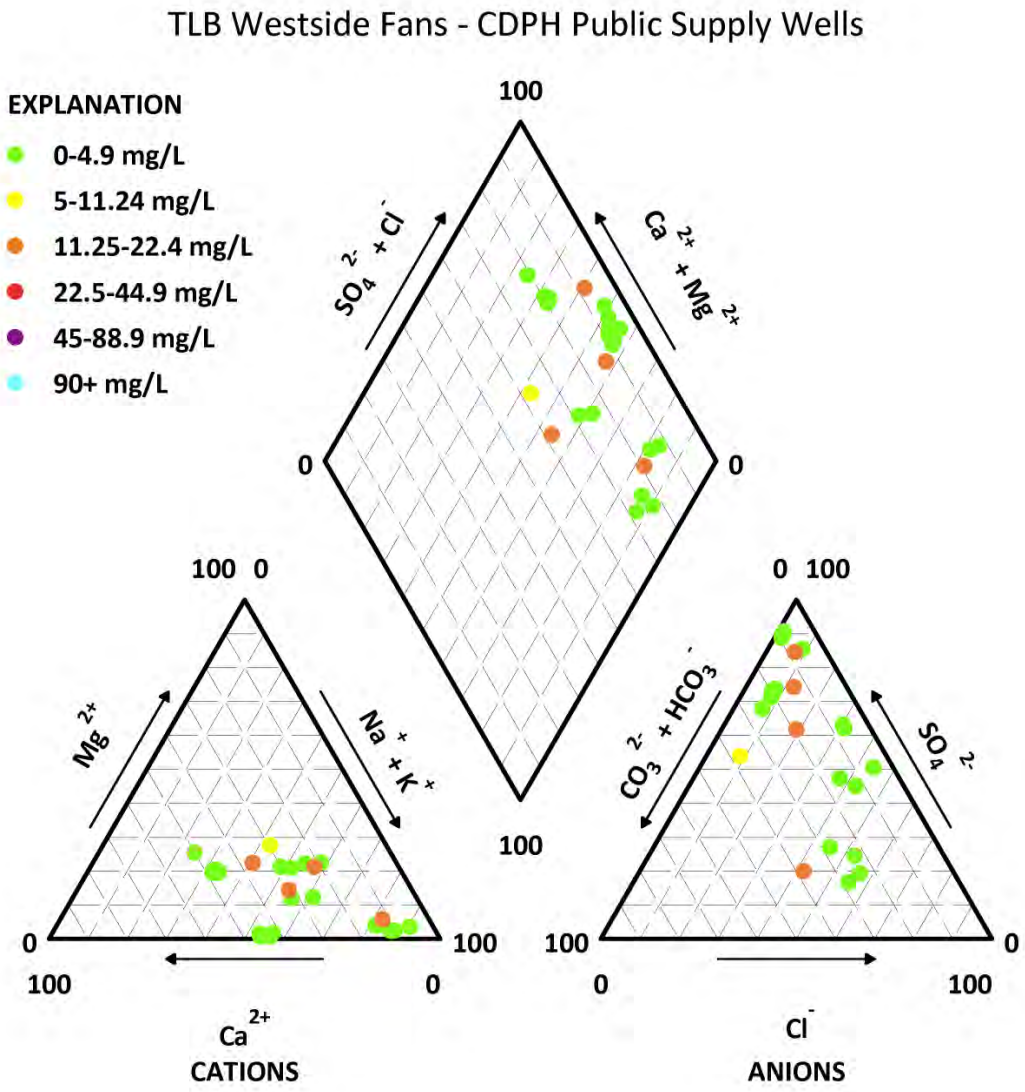

Figure 8. Piper diagram of groundwater samples from Westside Fans sediments. Concentrations are of nitrate as nitrate.

Adminsitrative Record
Page 34950

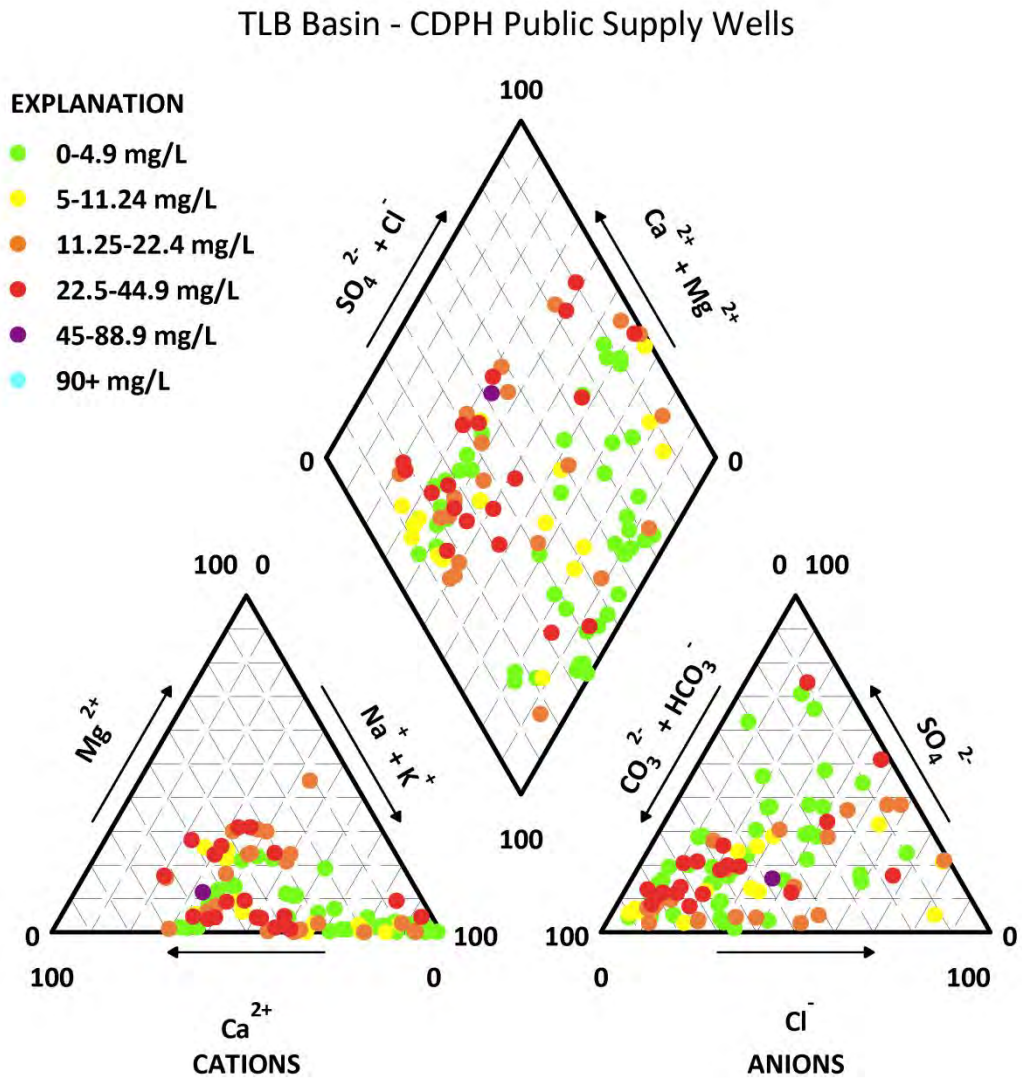

Figure 9. Piper diagram of groundwater samples from Basin sediments. Concentrations are of nitrate as nitrate.

2.4.3 Anthropogenic Contaminants

Nitrate is the most wide-spread pollutant in the TLB and it has been studied extensively. Anthropogenic sources include fertilizer applied to crops; animal operations, such as dairies and feedlots; and human sources, such as wastewater treatment plant effluent and septic tanks. Nitrate contamination is prevalent in every subbasin within the TLB. While nitrate is a naturally-occurring constituent of groundwater, concentrations of nitrate are being detected in the TLB that are well above what is considered natural or "background" concentrations. Levels of nitrate below 9 mg/L (as nitrate) are generally considered background (Mueller & Helsel 1996), and levels above 18 mg/L (as nitrate) are thought to reflect water that has been impacted, or contaminated from anthropogenic activities (Nolan

Adminsitrative Record
Page 34951

et al. 2002). The maximum contaminate level (MCL) for nitrate, as established by the U.S. EPA, is 45 mg/L (as nitrate).

Anton et al. (1988) presented the first regional-scale examination of nitrate in water wells in a report to the California State Legislature. The San Joaquin Valley (including the TLB) and the Central Coast (including the SV) geographic regions were examined as part of that study. The U.S. EPA STORET database was the primary source of data for the project, but most of the discussion centers on reportage of conclusions from previously unidentified studies. The Anton report characterizes the state of nitrate in groundwater for each of the Tulare Basin counties (Fresno, Kings, Tulare, and Kern). Fresno County is described as having elevated nitrate mostly in the eastern portions of the valley, with lower nitrate concentrations in the mid-valley, and slightly elevated concentrations in the western edge of the valley. Kings County is characterized as being relatively free of high nitrate problems in groundwater. The high nitrate region of eastern Fresno County is described as extending through the eastern valley portion of Tulare as well, and into Kern County (discontinuously from the city of Fresno to the city of Bakersfield). Anton states that 33 small and one large water system in that band have been in violation, but the timing of those violations is not mentioned. The report goes on to describe Kern County as having some of the highest nitrate concentrations in wells, with 34 small systems in violation. Anton also mentions a study from 1982 by Kern County Water Agency that found that the area affected by nitrate concentrations near or exceeding the MCL had expanded from 127 km^2 (49 mi^2) in 1958 to 963 km^2 (372 mi^2) in 1979.

A 1998 study by the USGS investigated nitrate and pesticide contamination of groundwater below several crop types grown in the TLB (Burow et al. 1998). Sixty monitoring wells were installed to sample nitrate and pesticide levels in groundwater in 1994-1995. The wells were placed along approximate groundwater flow paths beneath three land uses in the eastern San Joaquin Valley. The crop types included almond orchards, vineyards, and corn/alfalfa/vegetable crops. The three land use settings were thought to be representative of the range of crops grown in the San Joaquin Valley.

The high levels of fertilizer application, along with the rapid infiltration rate, led to many of the samples obtained from almond orchards to be very high in nitrate levels. About 40% of the samples contained nitrate levels above the MCL of 45 mg/L (as nitrate) (Burow et al. 1998). For vineyards, nitrate levels were much lower, representing approximately 6% of the Central Valley land area. About 15% of samples taken from a vineyard land use setting had nitrate levels greater than the MCL. For the alfalfa, corn, and vegetable land use, nitrate levels were closer to levels found in almond orchards. Alfalfa, corn, and vegetable cover approximately 12% of the Central Valley, and these crops tend to be planted in fine-grained sediments with low dissolved oxygen measurements and a slow infiltration rate. Nitrate levels were higher than the MCL in 35% of samples obtained from these crops. Two wells had concentrations below the detection threshold of 0.05 mg/L. Nitrate levels exceeded the background level (defined in the report as 13.2 mg/L (as nitrate) in 75% of the samples and exceeded the MCL in 30% of the well samples.

Overall, Burow et al. (1998) also found that 68% of the wells tested had detectable levels of pesticides. The main pesticides found were simazine, dibromochloropropane (DBCP), atrazine, desethyl atrazine,

Adminsitrative Record
Page 34952

and diuron found in 37, 30, 25, 25, and 15 percent of the samples, respectively. All of the pesticides except DBCP were below respective U.S. EPA MCL limits for drinking water (if a limit existed). DBCP exceeded its MCL of 0.2 micrograms per liter (µg/L) in 25% of the wells tested.

USGS Circular 1159 (Dubrovsky et al. 1998) details the findings from investigations conducted under the National Water-Quality Assessment (NAWQA) for the San Joaquin-Tulare Basins. From September 1992 to August 1995, data were collected from a total of 88 domestic wells. Sixty of the domestic wells were located (20 each) in 3 different landuse settings: almond orchards, vineyards, and corn/alfalfa/vegetable crops. These are the same 60 wells that were examined in Burow et al. (1998), referenced above. The remaining 28 wells, plus 2 of the 60, were chosen as representative of the regional aquifer, and had a median nitrate concentration of 20.3 mg/L (as nitrate). Five of the 30 (17%) exceeded the MCL. Taken all together, 25% of the 88 domestic wells sampled for the project exceeded the MCL, and 77% exceeded the accepted background level (defined as 2 mg/L nitrate as nitrate). Elevated nitrate in shallow groundwater (i.e., domestic wells) was associated with agricultural land uses and coarse-grained sediments.

Returning to the same area in 2003, Burow et al. (2007) resampled the 60 monitoring wells for nitrate and pesticide contamination. The study showed (through chlorofluorocarbon (CFC) age dating) that water less than 10 m (33ft) below the water table had mean ages generally less than 15 years old. Water samples from depths greater than 60 m were shown to have mean ages greater than 45 years old. Burow et al. compared nitrate concentrations measured in their localized sampling with regional monitoring networks and found comparable results at the regional scale. Concentrations were the highest and most variable in the more shallow waters with median values and variability decreasing with depth. Comparing the 2003 samples to previous samples taken in 1994-1995, nitrate was found in have increased from 35.3 mg/L to 101.4 mg/L (as nitrate) in samples taken from the shallow groundwater. Analysis also determined the regional waters to be generally oxic and therefore little natural attenuation was expected to be taking place. Given the increases in nitrate concentrations, the oxic conditions, and vertical gradients present in the aquifer system, Burow et al. theorized that the shallow contaminated water would eventually make its way to deeper depths where public wells generally withdraw water.

Burow et al. (2008) examined historical data in the San Joaquin and Tulare Basins using data from 1,437 National Water Information System (NWIS) wells, 3,216 U.S. EPA STOrage and RETrieval (STORET) wells, and 1,689 Permits, Inspection, Compliance, Monitoring, and Enforcement (PICME) wells. In addition, the 2008 study included data from resampling of most of the wells sampled in the 1998 studies (Burow et al. 1998; Dubrovsky et al. 1998). The 2008 study concluded that nitrate concentrations were increasing over time and decreased with depth, and that drinking water supplies were significantly degraded by nonpoint source (agricultural) nitrogen.

In 2008, as part of the California State Water Resources Control Boards' Groundwater Ambient Monitoring and Assessment (GAMA) Priority Basin Project, the USGS sampled 99 public supply wells and irrigation wells in Kings, Kaweah, Tule, and Tulare Lake regions (Burton & Belitz 2008). These public supply wells tend to be drilled to depths that do not have elevated nitrate, are located predominantly in urban and peri-urban areas, and have much deeper screens than the monitoring well network used by

Adminsitrative Record
Page 34953

Burow et al. (2007). To obtain spatially unbiased data, public supply wells were sampled using an equal area grid, with a single well selected for each equal area cell. The results showed that six wells (~6%) were found to contain nitrate above the MCL. More than half of the samples contained at least one detectable pesticide, and seven pesticides were present in more than 10% of the samples. However, almost all of the samples' concentrations of pesticides were well below drinking water standards for the constituent.

Under the same GAMA program, a report for Kern County was published in 2008 by the USGS (Shelton et al. 2008). Using the same methodologies for unbiased spatial sampling, 2 out of 17 samples analyzed for nitrate contained concentrations above the drinking water standard, and all but one sample had detectable (>0.26 mg/L as nitrate) concentrations of nitrate. At least one pesticide was detected in 29 out of the 47 wells analyzed, and 5 pesticides were found in more than 10% of the samples. All pesticide concentrations, however, were below drinking water thresholds, with the majority of the concentrations being less than one-thousandth of the thresholds.

In 2010, the USGS National Water Quality Assessment Program published a national assessment of nutrient impacts on groundwater (Dubrovsky et al. 2010). The assessment included a national statistical model to predict groundwater nitrate concentration in the shallow-most groundwater based on land use data, general hydrogeologic information, and soils information. For large portions of the TLB, but also for the SV, the national USGS statistical nitrate model simulates shallow groundwater nitrate values that exceed the drinking water limit (Figure 10).

It is important to remember that nitrate contamination found in the TLB is not from a single land use or process. Nitrate detected in a single groundwater sample is often from a combination of sources. LLNL, using isotopic data of groundwater samples from a domestic well survey program, concluded that although the most contaminated samples were associated with an organic source (i.e. manure, septic, etc.), the majority of the samples containing elevated levels of nitrate had a signature that indicated the contamination was due to multiple sources, meaning a combination of organic and inorganic sources (Singleton et al. 2011). High concentrations of nitrate were found in groundwater associated with all land use categories (Ibid.).

The GAMA reports discussed here and in section 3 are those which specifically pertained to our study area. For more information on SWRCB's GAMA program, and additional reports, including reports on nitrate occurrence and fate, the reader is referred to SWRCB's GAMA website http://www.waterboards.ca.gov/gama/.

Adminsitrative Record
Page 34954

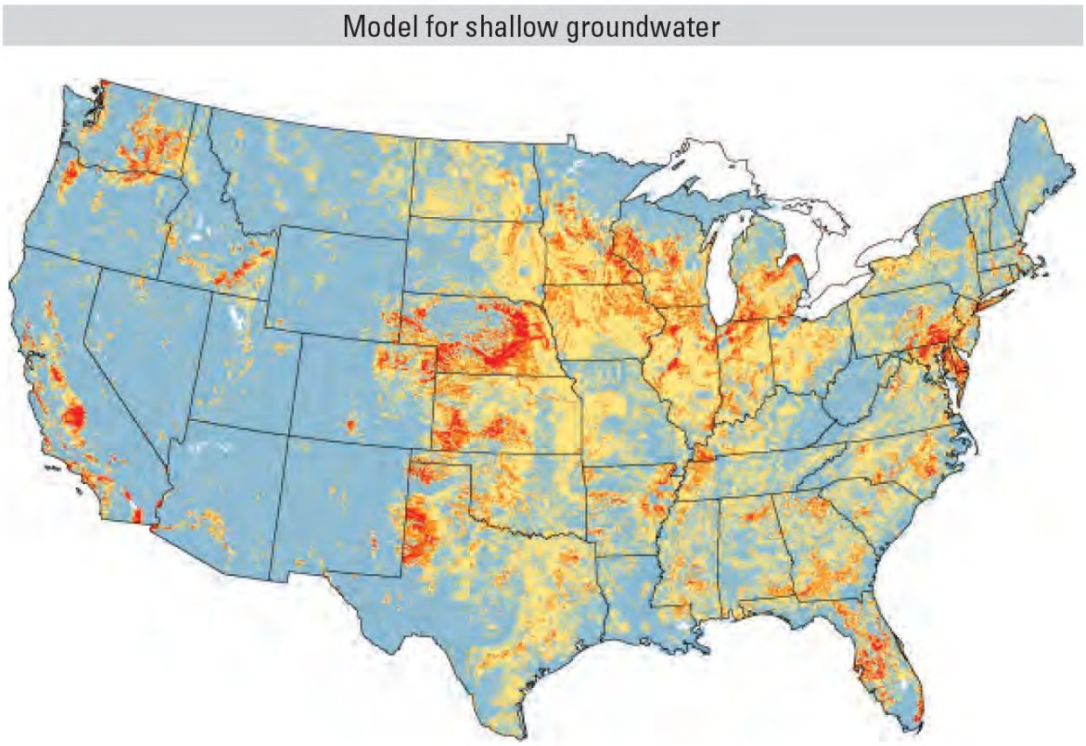

Figure 10. Nitrate levels in the uppermost 3 m of groundwater, simulated by statistical models based on general land use, groundwater, vadose zone thickness, and soils information. Red indicates concentrations above the drinking water MCL. (Source: Dubrovsky et al. 2010).

Adminsitrative Record
Page 34955

3 Salinas Valley Hydrogeology and General Water Quality

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Adminsitrative Record
Page 34956

3.1 Physical Setting

3.1.1 Location

The Salinas Valley (SV), located about 160km (100 miles) south of San Francisco, CA, is an intermontaine valley which contains an aquifer system composed of fluvial and alluvial fan sediments. It is bound to the northeast by the Gabilan and Diablo Ranges, to the southeast by the Sierra de Salinas and Santa Lucia Ranges, and to the northwest by Monterey Bay. The valley floor is approximately 85 miles long, ranging from 3 to 10 miles wide containing roughly 91,134 hectares of agricultural land and 22,835 hectares of urban land, according to California Augmented Multisource Landcover (CAML) in 2010 (Viers et al. 2012). The study area consists of the SV groundwater basin from Monterey Bay, southeast to the town of San Ardo. The basin extends beyond San Ardo and is hydrologically connected to the Paso Robles Basin; however, the study area for this report was concerned with the major subareas of the Salinas River watershed (Figure 11) that lie within Monterey County. Almost all of the land is used for agriculture: lettuce (19,512 hectares or 48,215 acres); vineyards (19,234 hectares or 47,528 acres); and truck, nursery, and berry crops (17,165 hectares or 42,415 acres) make up the majority of the crops grown. Viers et al. (2012) provides a more detailed discussion on current and historic land use in the SV.

3.1.2 Subareas

The SV has been divided into four main subareas by the DWR (Durbin et al. 1978). These areas are known as the Pressure, Eastside, Forebay, and Upper Valley subareas (Figure 11). The four subareas do not correspond to subbasins or watersheds, but were based on the differences in hydrogeologic properties and sources of groundwater recharge (GW Recharge) (Durbin et al. 1978; Salinas Valley Ground Water Basin Hydrology Conference 1995). Durbin et al. (1978) discuss three important characteristics that differentiate the areas; confining conditions, specific capacity of wells, and the source of groundwater recharge (Table 3).

Confining Conditions

The Pressure area is generally thought to be composed of 3 semi-confined to confined aquifers known as the 180-ft, 400-ft, and Deep aquifers. The names are based on the approximate depths to reach each aquifer (Durbin et al. 1978; Boyle Engineering Corporation 1986; Montgomery Watson Americas, Inc. 1997). The Eastside subarea is considered semi-confined, the Forebay semi-confined to unconfined, and the Upper Valley subarea largely unconfined.

Specific Capacities

Specific capacities of wells generally increase up-valley. The Eastside area has an average of 447 m^2/day (25 (gal/min)/ft), the Pressure area 1073 m^2/day (60 (gal/min)/ft), the Forebay 1788 m^2/day (100(gal/min)/ft), and the Upper Valley 2682 m^2/day (150 (gal/min)/ft) (Durbin et al. 1978).

Adminsitrative Record
Page 34957

Groundwater Recharge

Previous work indicates that the Pressure area is recharged largely by irrigation and stream recharge in roughly equal amounts. The Forebay and Upper Valley areas receive recharge from irrigation return water and infiltration from the Salinas River, with estimates indicating that the river provides approximately twice as much recharge as irrigation return. The Eastside area is the only subarea that the Salinas River does not flow through, and the majority of its recharge is from irrigation return water (Durbin et al. 1978; Boyle Engineering Corporation 1986; Montgomery Watson Americas, Inc. 1997).

Table 3. Distinguishing characteristics of the four largest subareas of the SV, California. (Source: Durbin et al. 1978, Montgomery Watson Americas, Inc., final report 1997.)

Subbasin Name	Subbasin Area (acres)	Average Specific Capacity (gal/min)/ft	Confining Conditions	Dominant Source(s) of Recharge
Eastside	74,000	25	Semi-confined and unconfined	Irrigation Return
Pressure	91,000	60	Three confined aquifers	Irrigation Return and Salinas River
Forebay	87,000	100	Semi-confined and unconfined	Salinas River and Irrigation Return
Upper Valley	92,000	150	Unconfined	Salinas River and Irrigation Return

Adminsitrative Record
Page 34958

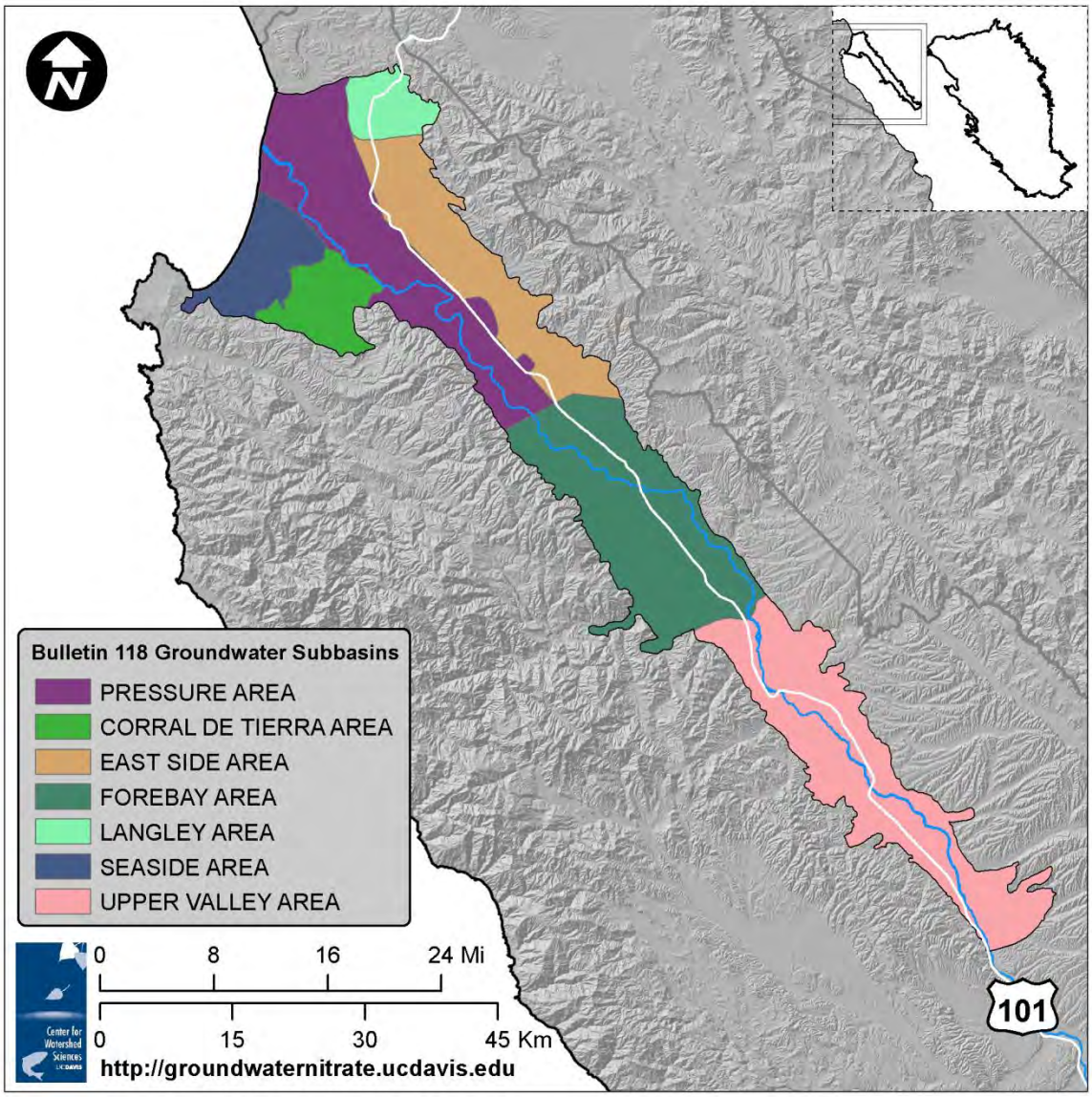

Figure 11. Seven subareas of the SV. The four largest areas considered are the Pressure Area (also referred to as the 180 ft/400 ft aquifers), the Eastside area, the Forebay Area, and the Upper Valley Area (Source: DWR 2003.)

Adminsitrative Record
Page 34959

3.2 Hydrology

3.2.1 Overview

The SV has a Mediterranean climate and receives precipitation almost exclusively in the winter months. The months from November to April account for 87% of the annual total, while the summers consist of moderate temperatures and very little rain (Planert & Williams 1995). Annual rainfall varies within the basin, averaging from 10 inches (25.4 cm) in the valley to over 30 (76.2 cm) inches locally at high altitudes (Durbin et al. 1978).

The Salinas River is the largest river in the valley, running the entire length of the SV. The river is also considered to be influent (loses water to the subsurface) for the majority of its length. Historically (before large scale development of groundwater) the river was likely a discharge zone for groundwater in the lower reaches, as is the case in most undeveloped basins. In 1846 the river was described as such: "as it approaches the ocean, is broad and fertile, and there are many fine ranchos upon it. But higher up, [the Salinas River] becomes dry in the summer" (Verardo & Verardo 1989 p.27). This implies that in the summer months, when there was almost no rainfall in the valley, the lower portions of the river still contained water flowing to the ocean, indicating that groundwater was discharging to the surface. In addition, the reclamation ditch (constructed in 1917) was built with the intention of draining the wetlands and lakes present in the lower basin (Casagrande & Watson 2006), again implying that groundwater was historically discharging to the surface. When groundwater pumping began in the valley in the early 1900s, the groundwater system of the valley was changed significantly. By the 1940s, the Salinas River was largely dry in the summer months. Due to its proximity to Monterey Bay, the groundwater basin also historically discharged to the ocean. This discharge mechanism has also been altered as seawater intrusion has been documented in the coastal regions since the 1930s (Montgomery Watson Americas, Inc. 1993).

3.2.2 Water Budget

Due to the seasonal variations in rainfall and river discharge, decreasing groundwater levels, and increasing seaward intrusion on the coast, the Nacimiento and San Antonio reservoirs were constructed (1957 and 1965, respectively) to regulate flow of the Salinas River to facilitate increased recharge to the groundwater basin. By sustaining a perennial flow in the Salinas River through controlled discharge from reservoirs, the groundwater basin has the potential to be recharged throughout year. Since construction of the reservoirs, multiple studies have concluded that stream recharge is the greatest contributor to groundwater recharge for the basin as a whole (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1993; Salinas Valley Ground Water Basin Hydrology Conference 1995). Stream recharge, however, is not distributed equally along the river, with some subareas receiving much more river recharge than others (Table 4). Shallow confining layers in the northern portion of the valley prevent significant infiltration, whereas in the southern valley the semi- and unconfined aquifers facilitate

Adminsitrative Record
Page 34960

greater amounts of groundwater recharge. Groundwater level responses in each subarea reflect this unequal recharge as seen in Section 3.2.3 *Groundwater Levels*.

Irrigation is the second most significant source of recharge to the valley as a whole (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1993; Salinas Valley Ground Water Basin Hydrology Conference 1995). According to a 2009 Monterey County Water Resources Agency (MCWRA) groundwater summary report, the amount of groundwater currently pumped for irrigation in the valley totals 416,421 acre-ft (Monterey County Water Resources Agency 2011). Groundwater modeling from the Salinas Valley Integrated Ground Water and Surface Model (SVIGSM) indicates that between 1970-1994, an annual average of 189,000 acre-ft was recharged to groundwater as deep percolation (excluding river recharge), the majority as agricultural return. Evapotranspiration by crops and direct evaporation from soils account for the difference, removing roughly two thirds of the total applied. As more irrigation in the SV has been converted to micro-irrigation, recharge from irrigation has potentially decreased (Figure 12) although a significant potential for increasing irrigation efficiency in micro-irrigation often remains to be captured (Tim Hartz, personal communication). Precipitation and boundary flow (from adjacent land outside of the DWR boundaries) contribute to the recharge of the groundwater basin; however, these are likely small compared to irrigation return water and stream recharge (Montgomery Watson Americas, Inc. 1993).

Adminsitrative Record
Page 34961

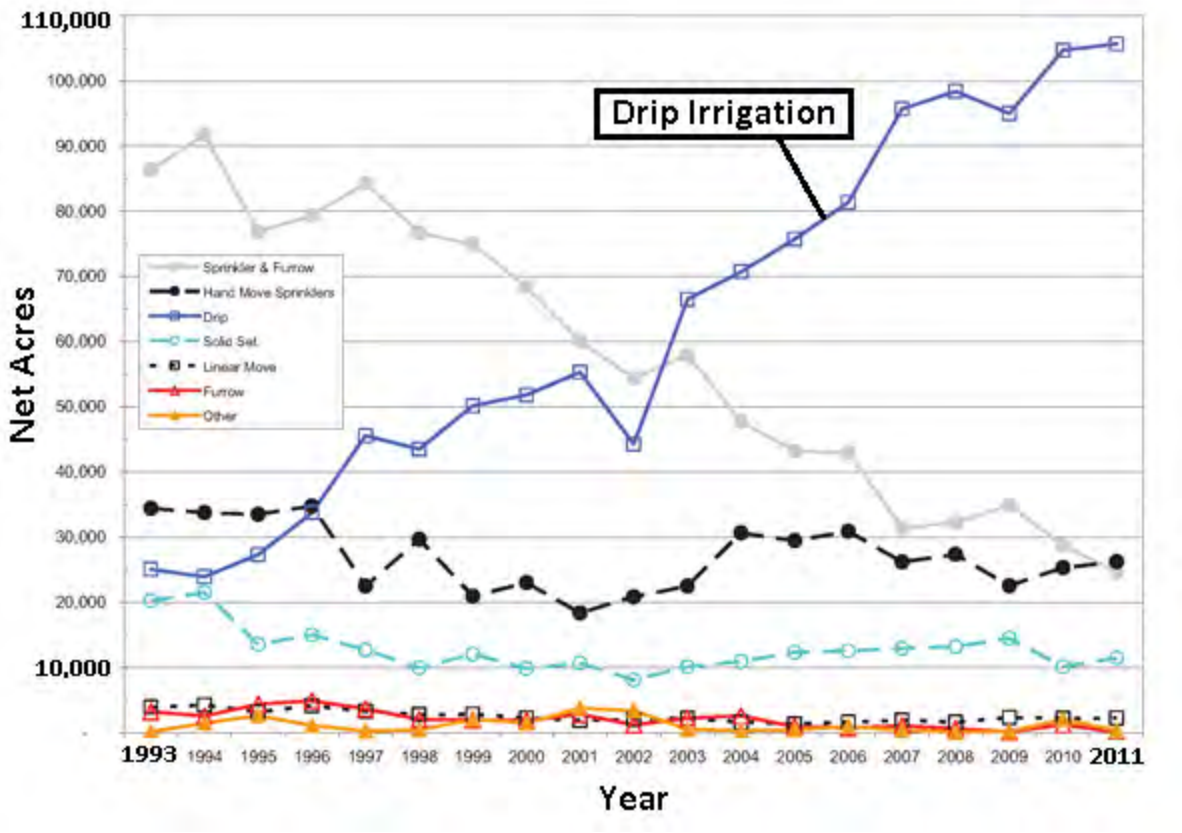

Figure 12. A significant increase in the use of micro irrigation has been observed in the SV. Reprinted with permission. (Source: MCWRA 2011.)

Discharge from the basin occurs mainly from groundwater pumping. According to a MCWRA 2010 Groundwater Summary Report, 460,443 acre-ft of water was pumped in 2010. Agricultural pumping accounted for 90% of all groundwater pumping, while urban pumping amounted to only 10% (Monterey County Water Resources Agency 2011). Portions of the Salinas River may still provide a discharge zone for groundwater in some locations; however, it is likely small compared to the discharge through well pumping.

A water budget for the SV (Table 4) comes from model outputs from the SVIGSM (Montgomery Watson Americas, Inc. 1997) and data from annual groundwater extraction reports published by MCWRA. The breakdown of agricultural pumping and urban pumping in Table 4 is based on MCWRA reported averages during 1995-2010, whereas the remaining values are averages from the SVIGSM (1970-1994). Model results show that the Eastside and Pressure subareas have experienced significant loss in storage during the 25 years modeled. The Forebay and Upper Valley have remained essentially constant, with the Upper Valley showing a slight increase in storage through time. Table 5 shows pumping broken down by city/area and comes from the latest groundwater extraction report provided by MCWRA (MCWRA 2011).

Adminsitrative Record
Page 34962

Table 4. Average values of groundwater budget in acre-feet/year.

In acre-feet/year	Pressure	Eastside	Forebay	Upper Valley
GW Recharge[1]	54,337	33,189	44,060	57,753
Stream Recharge[1]	59,958	1,830	102,634	98,956
Seawater Intrusion[1]	15,000	---	---	---
GW Pumping - Total[1]	133,176	84,833	157,585	143,826
GW Pumping–Agricultural[2]	101,203	86,013	140,919	133,389
GW Pumping–Urban[2]	22,227	12,402	7,076	4,255
Net Change in Fresh GW[1]	-11,000	-33,000	0	3,000

[1]Montgomery Watson SVIGSM model averages 1970-1994 (Montgomery Watson Americas, Inc. 1993).
[2]MCWRA extraction reports 1995-2010.

Adminsitrative Record
Page 34963

Table 5. Urban groundwater pumping by city or area. (Source: Monterey County Water Resources Agency 2011.)

City or Area	Urban Pumping-2010 (Acre-Feet/year)	Urban Pumping-Avg* (Acre-Feet/year)	Percentage of Total-2010	Percentage of Total-Avg*
Castroville	810	878	1.7%	1.95%
Chualar	121	132	0.3%	0.29%
Former Fort Ord	2,469	2,755	5.7%	5.97%
Gonzales	1,282	1,277	3.2%	2.75%
Greenfield	2,152	1,588	5.1%	3.49%
King City	3,089	3,415	6.3%	7.51%
Marina Coast WD	1,765	2,031	4.3%	4.47%
Other Areas	11,383	7,510	20.3%	16.38
Salinas	16,819	21,992	43.3%	47.79%
San Ardo	100	130	0.3%	0.29%
San Lucas	36	60	0.1%	0.12%
Soledad	2,293	2,329	5.3%	5.04%
Soledad Prisons	1,702	2,330	4.1%	4.93%
Total	44,022	45,960	100.0%	100%
*Averages calculated from 1995-2010 MCWRA extraction reports.				

3.2.3 Groundwater Levels

Development of groundwater in the SV basin initially caused a lowering of groundwater levels throughout the basin; however, the most lasting and significant drops have been observed in the Pressure and Eastside subareas. Since the construction of the Nacimiento and San Antonio reservoirs, water levels have risen to roughly 1944 levels and have become relatively stable in the Forebay and Upper Valley subareas. The Pressure and East Side subareas have shown continued decline, although the East Side has experienced the most significant decline. Figure 13 shows water levels from representative wells for the four subareas.

The differences in water levels over time for each subarea can be explained by the geology, groundwater pumping, and proximity to the Salinas River. According to the SVIGSM and through direct groundwater level measurements, the Eastside and Pressure subareas have experienced considerable overdraft during 1970-1994 (Montgomery Watson Americas, Inc. 1997). The Salinas River flows through every subarea except the Eastside, which is why increased recharge from the Salinas River has had little effect on water levels in this region. Almost all of the groundwater recharge in the Eastside subarea comes from irrigation return flow; however, this is far less than is pumped from the subsurface. River

Adminsitrative Record
Page 34964

recharge and irrigation return in the Pressure subarea is limited by laterally extensive confining layers that thin out and disappear in the adjacent subareas (Kennedy/Jenks Consultants 2004). This inhibits recharging of the confined and semi-confined aquifers below. Groundwater pumping has generally exceeded recharge in both the Pressure and Eastside subareas, which has resulted in the loss of groundwater storage over time. The Forebay and Upper Valley subareas are for the most part unconfined to semi-confined, and the Salinas River is able to provide significant recharge to the aquifer system. On average, the river provides greater than or equal amount of water to the subsurface in relation to irrigation return in these subareas (Table 4). The combined recharge has matched or exceeded the amount of water pumped from the subsurface, which has allowed groundwater levels to recover. Model results indicate that the Upper Valley has even seen an average increase in storage through time.

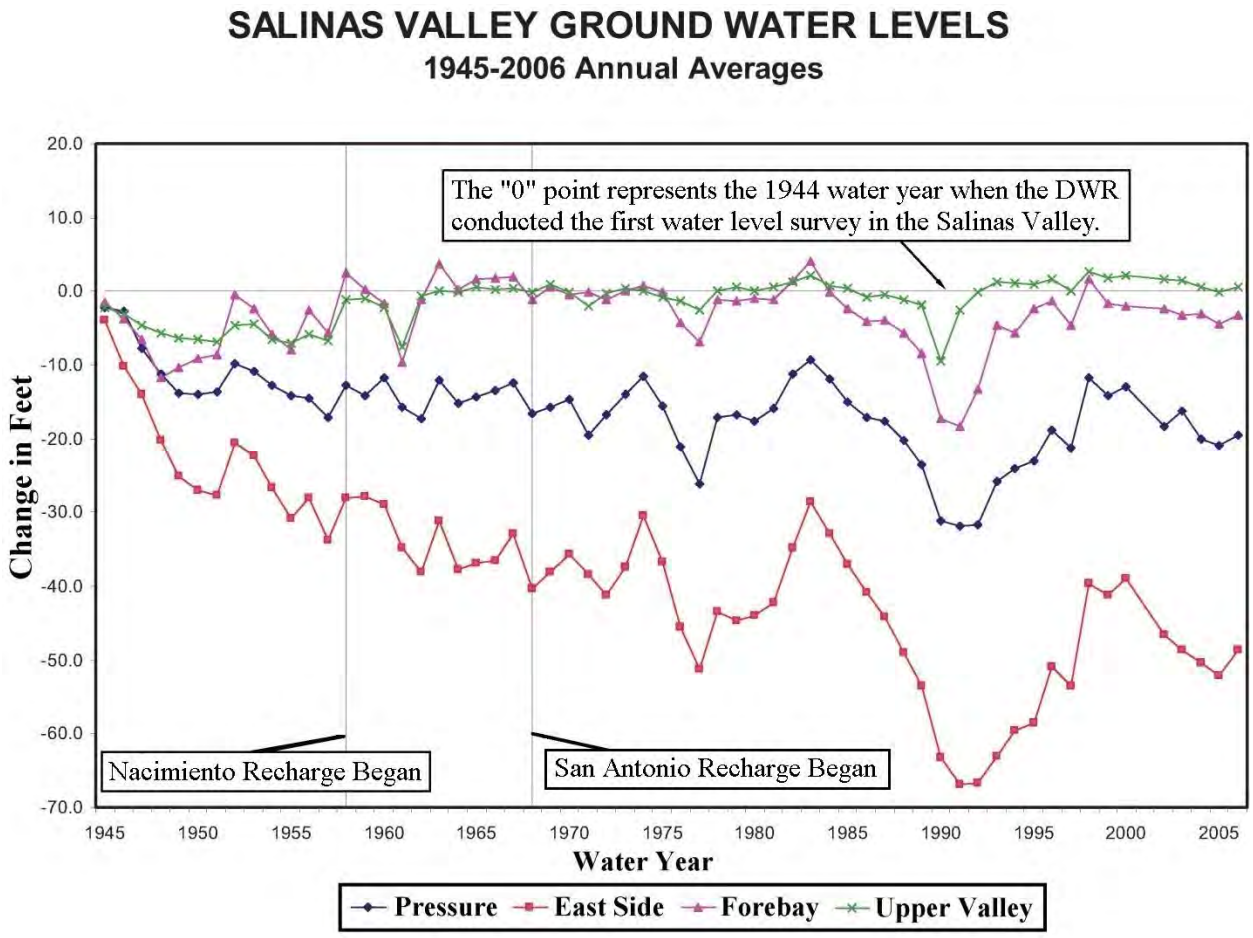

Figure 13. Groundwater levels in the subareas through time. (Modified from Johnson 2009.)

3.2.4 Seawater Intrusion

Seawater intrusion (Figure 14 and Figure 15) has been recognized in the SV since the 1930's (Casagrande & Watson 2006). The intrusion is the result of a reversal of groundwater flow near Monterey Bay.

Adminsitrative Record
Page 34965

Historically, fresh water likely discharged into the ocean below sea level where the aquifers outcrop into Monterey Bay; however, groundwater pumping has caused seawater to enter the groundwater basin. The reversal of flow can occur when the groundwater head in the aquifer is lower than that of an adjacent saltwater body (in this case, the Pacific Ocean). This creates a lateral hydraulic gradient, which results in lateral groundwater flow inland. Currently the saline/freshwater transition is operationally defined by TDS concentration of 500 ppm, and the time progression of this front for the 180-ft and 400-ft aquifers show that seawater has moved inland over time (Figure 14 and Figure 15, respectively).

A paper prepared for the MCWRA in 1999 offered a solution to the seawater intrusion problem which consisted of limiting pumping near the Monterey Bay area and delivering water to this area from upgradient (Salinas Valley Ground Water Basin Hydrology Conference 1995). The conclusion of the study was that seawater intrusion was not the result of a supply problem, but a problem of water distribution.

Seawater intrusion continues today; however, its rate of encroachment has been slowed due to the completion of the reservoirs and improved management strategies. In accordance with the recommendation of the Salinas Valley Ground Water Basin Hydrology Conference paper, a surface water diversion system is being built as part of the Salinas Valley Water Project to deliver surface water to the coastal areas near Castroville to replace groundwater pumping (Monterey County Water Resources Agency 2001). These and other basin supply studies can be found in Table 9 through Table 11 in Section 3.4.4.

Adminsitrative Record
Page 34966

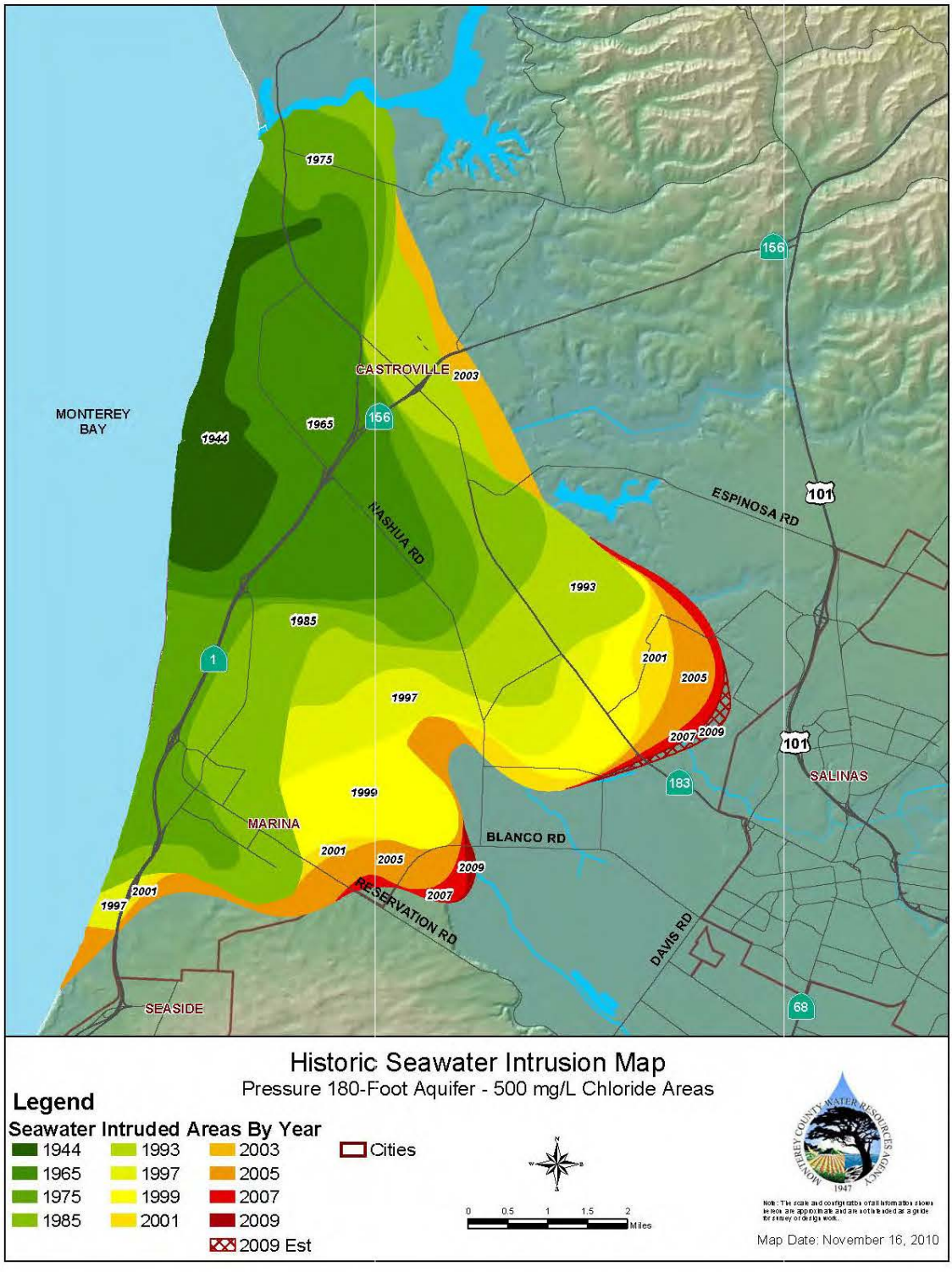

Figure 14. Map of seawater intrusion through time for the 180-ft aquifer. (Source: MCWRA 2010a.)

Adminsitrative Record
Page 34967

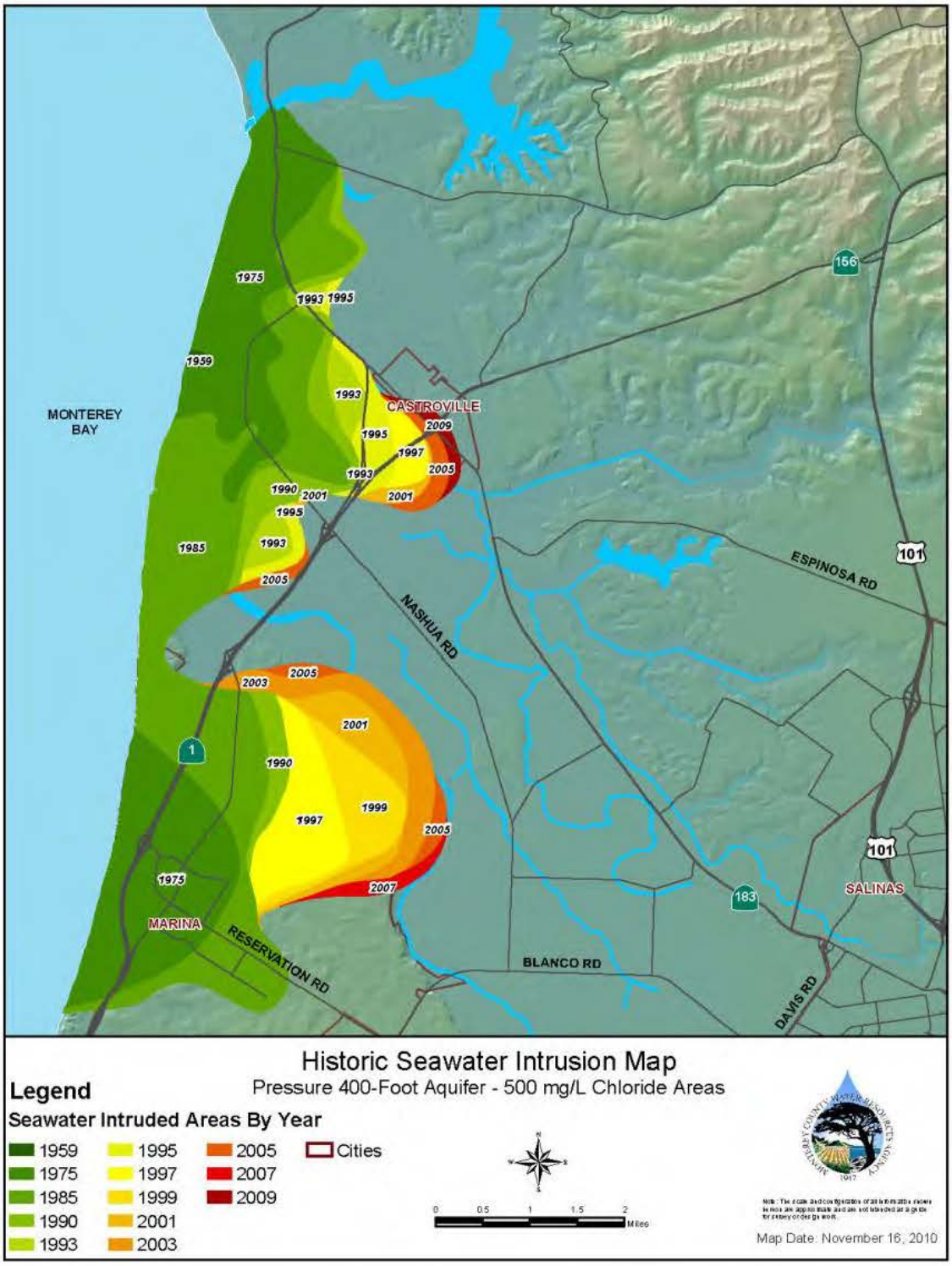

Figure 15. Map of seawater intrusion through time for the 400-ft aquifer. (Source: MCWRA 2010b.)

Adminsitrative Record
Page 34968

3.3 Hydrogeology

3.3.1 Geology

The aquifer system in the SV was deposited into a subsiding geologic basin, or trough, by fluvial processes (e.g., the Salinas River) and alluvial fans that mainly occur between the valley walls and the ancestral Salinas River. The bedrock underlying the sediments has an elevation, at its lowest point, of roughly 30m (100 ft) above sea level in the upper portion of the SV, but drops to over 600m (2000 ft) below sea level at its lowest point in the northern valley near Monterey Bay (Durbin et al. 1978). The geologic history of the basin involves a complex interaction with the adjacent sea as it has risen and fallen due to glacial periods. Recessions and transgressions of the ocean combined with alluvial and fluvial processes in the valley have resulted in a highly heterogeneous configuration of sediments. The aquifer system in the lower portion of the SV contains semi-continuous clay deposits thought to have been deposited by the marine transgressions. Such clay layers form the aquitards which create the confined and semi-confined conditions in the Pressure and adjacent subareas.

The general stratigraphic framework of the basin is composed of 3 hydrostratigraphic units, which sit upon a granitic basement and are overlain by recent (Quaternary) unconsolidated fluvial and alluvial sediments. The following geologic descriptions are largely from the 1978 USGS model report and 1997 MCWRA model report (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1997). The Monterey Formation, of Miocene age, is found above the basement granite and is composed of shale and large mudstone layers with thin sandstone beds towards the base. Lying conformably above the Monterey Formation and unconformably above the granitic basement (where the Monterey Formation is nonexistent) are the Santa Margarita/Purisima Formations (Pliocene). Stratigraphically similar, these two formations make up the bulk of the semiconsolidated rocks of marine origin. Both consist of a range of sedimentary rocks from sandstone to siltstone. Above the semiconsolidated marine deposits are unconsolidated materials which consist of the Paso Robles Formation (Pliocene/Pleistocene), Aromas Sand, and recent (Quaternary) alluvium. These unconsolidated sediments are of non-marine origin and make up the majority of the fresh water bearing aquifer material in the SV. The Paso Robles Formation is generally of fluvial origin; however, it contains sediment from alluvial, fluvial, and lacustrian environments. The recent alluvium consists of wind-blown sand, alluvial, and river sediments.

For the nitrate analyses in Section 5, the SV was divided into two regions as per a USGS GAMA 2005 study (Kulongoski & Belitz 2007). The two areas used were the Monterey Bay area, which encompasses the Pressure, Eastside, Langley, Seaside, and Corral de Tierra subareas, and the Salinas Valley area, which contains the Upper Valley and Forebay subareas (Figure 16). For the Monterey bay area, the subbasins were combined based on their similar Quaternary Deposits (Kulongoski & Belitz 2007). The Forebay and Upper Valley subareas were combined based on their similar geology (Kulongoski & Belitz 2007).

Adminsitrative Record
Page 34969

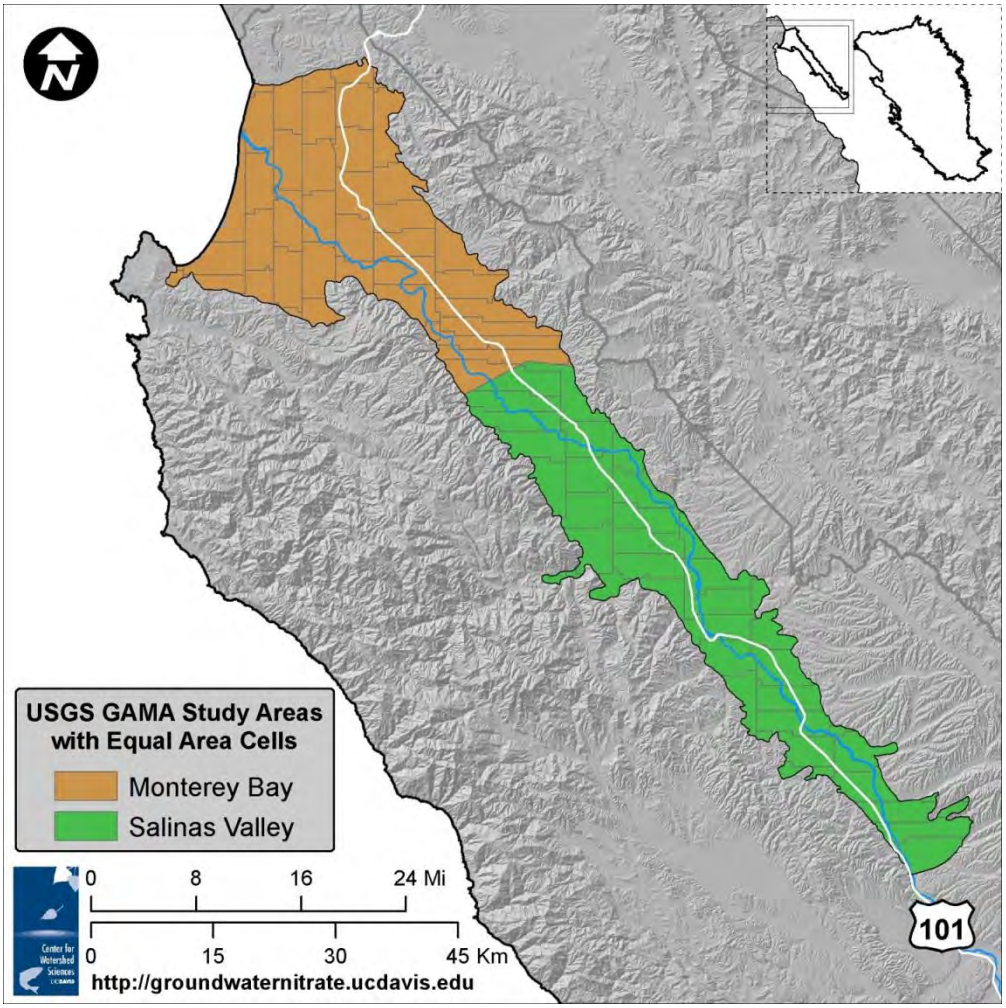

Figure 16. The two major groundwater regions used for the statistical analysis of groundwater nitrate data in Section 5. The equal area cells shown were used in spatial analyses to avoid spatial bias of the inherently clustered well test data (see Section 5) (Source: Kulongoski & Belitz 2007).

3.3.2 Aquifer Heterogeneity

It is well known that, within stratigraphic units that are composed of alluvial and fluvial sediments, the sediments are highly heterogeneous. Further, alluvial sediments that compose aquifers commonly consist of more aquitard sediments (silts, muds and clays) than aquifer sediments (sands and gravels), and the SV is no exception (Figure 17 and Figure 18).

To represent this heterogeneity and its effects on transport of nitrate, Fogg et al. (1999) created three-dimensional, geostatistical models of the dominant sediment textures in the Valley, and two examples are shown in Figure 17 and Figure 18. Geostatistical approaches produce more realistic models of spatially varying subsurface properties than simplified layered models. These models are particularly useful for representing the inevitable windows that breach aquitards (where commonly they are assumed and modeled as perfectly continuous). Fogg et al. (1999) demonstrated that the confining bed

Adminsitrative Record
Page 34970

above the 180-ft aquifer could be an effective confining bed with respect to the hydraulic conditions in the system while still having enough coarse-grained pathways through them to allow localized nitrate contamination from the surface. This is consistent with the occurrence of localized nitrate contamination in the confined aquifers of the Pressure subarea.

For more information on the geology of the SV, refer to the list of hydrogeological studies in Table 9 through Table 11.

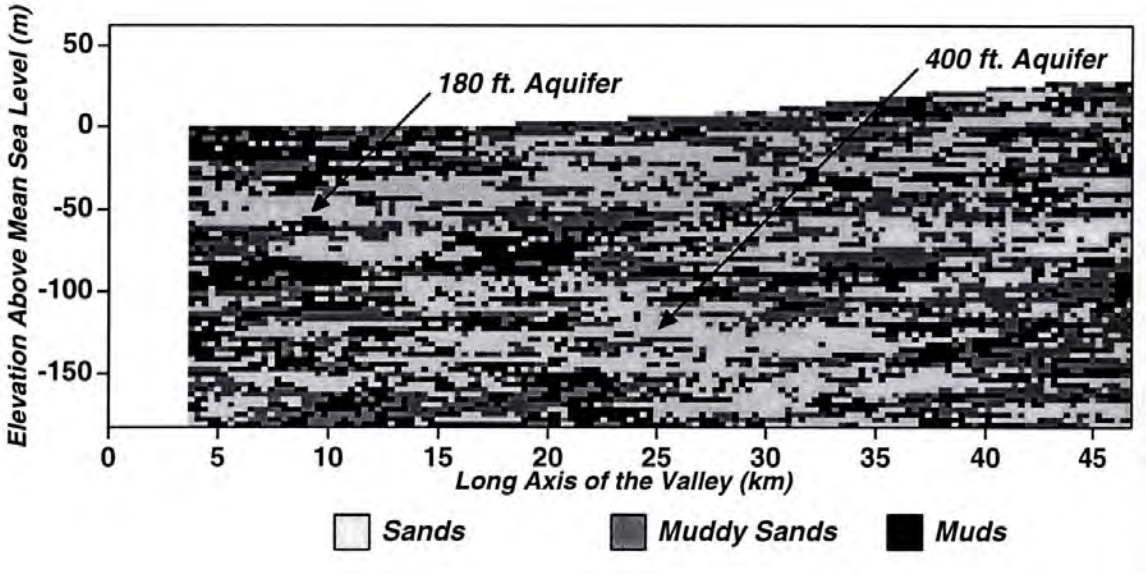

Figure 17. **Vertical cross section through a 3D heterogeneity model of subsurface soils in the Pressure subbasin of the SV. (Source: Fogg et al. 1999.)**

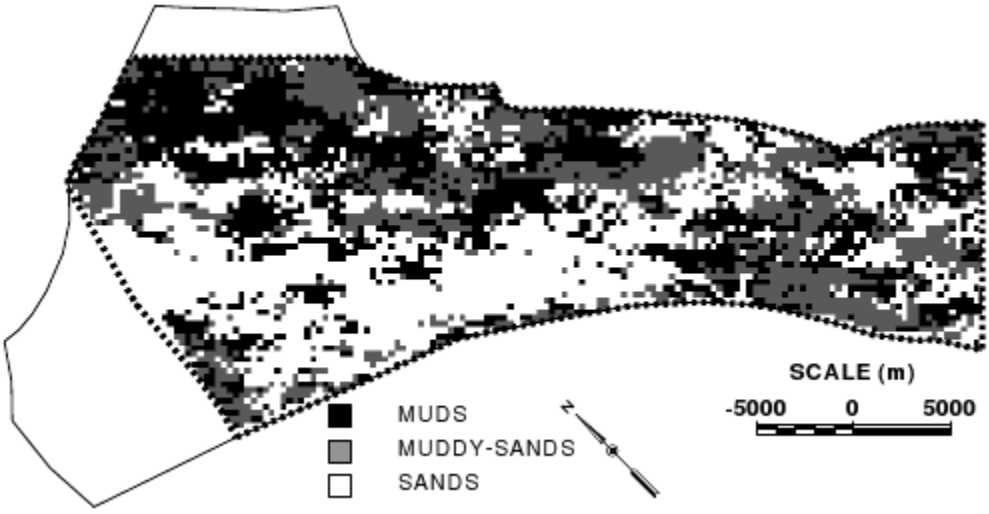

Figure 18. **Horizontal cross section through a 3D heterogeneity model of subsurface soils in the Pressure and Eastside subbasins of the SV. (Source: Fogg et al. 1999.)**

Adminsitrative Record
Page 34971

3.3.3 Flow Modeling

In the past, there have been several studies that have modeled all or portions of the SV aquifer system (Table 6). Initially the purpose of these models was to define a water budget for the basin. As models became more sophisticated, they were used to investigate the impacts of groundwater pumping on seawater intrusion. Of the many models, there have been three widely used computer models of the SV: Durbin et al. (1978), Boyle Engineering Corporation (1987), and Montgomery Watson Americas, Inc. (1997) models.

In 1978, the USGS, working with the United States Army Corps of Engineers, developed two transient models: a two dimensional model and a three dimensional model (Durbin et al. 1978). The goal of the model was to investigate the water budget of the aquifer system. The following year, a report was written explaining the results of various management scenarios ran with the model and also qualitatively discussing the results with respect to seawater intrusion (Kapple & Johnson 1979). Following the model's completion, there have been additional modifications to the original version. In 1985, Hydrocomp adapted the USGS model to investigate the effects of groundwater development of the Deep/900-ft aquifer in the Pressure subarea (Hydrocomp, Inc. 1985). In 1988, the model was modified to incorporate new water budget data to simulate basin management options (Yates 1988).

In 1986, Boyle Engineering developed a new model, FEGW-14 for the SV groundwater basin (Boyle Engineering Corporation 1986). The model was used to calculate groundwater budgets and quantify seawater intrusion. A 1987 report by Boyle Engineering provided an alternative analysis to the original report, investigating new water management scenarios. The model was updated again in 1990, incorporating new data and additional basin management scenarios (Boyle Engineering, Inc. 1990).

In the 1990's, Montgomery Watson Americas, Inc. developed an Integrated Ground Water and Surface Model code (IGSM) which was being adapted for several groundwater basins in California at the time. In 1993, the model was applied to the SV groundwater basin to investigate the groundwater budget and include quantitative simulations of seawater intrusion. A final report, an update from the initial 1993 draft report, was published in 1997 and explored various management strategies to increase infiltration and prevent further seawater intrusion (Montgomery Watson Americas, Inc. 1997).

Adminsitrative Record
Page 34972

Table 6. Groundwater models of the SV.

Title	Creator	Year	Discretization	SS/Trans.	Purpose
2-d and 3-d Digital Flow Models For The Salinas Valley Groundwater Basin, California	Durbin et al. (U.S. GEOLOGICAL SURVEY)	1978	Finite Element	Transient	Water Budget
Model Simulation of Various Water-Resource Management Alternatives in the Salinas Valley Ground-Water Basin, California	Kapple and Johnson (U.S. GEOLOGICAL SURVEY)	1979	Finite Element	Transient	Water Budget
A Steady - State Finite Difference Model Of The Groundwater Of Salinas Valley	Karen Nelsen	1985	Finite Difference	Steady State	Water Budget
Modeling of the Deep Zone In The Salinas Valley Groundwater Basin	Hydrocomp (Modified '78 USGS model)	1985	Finite Element	Transient	Seawater Intrusion (Deep/900-ft aquifer)
Salinas Valley Ground Water Model	Boyle Engineering	1986	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Ground Water Model Alternative Analysis	Boyle Engineering	1987	Finite Element	Transient	Water Budget and Seawater Intrusion
Simulated Effects Of Ground-Water Management Alternatives For The Salinas Valley, California	Eugene Yates (Modified '78 model)	1988	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Groundwater Model: Model Update And Future Hydrologic Analyses	Boyle Engineering	1990	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Integrated Ground Water And Surface Model	Montgomery Watson Americas, Inc.	1993	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Integrated Ground Water and Surface Model Update	Montgomery Watson Americas, Inc.	1997	Finite Element	Transient	Water Budget and Seawater Intrusion
Mapping Ground Water Susceptibility to Nitrate and Pesticide Contamination	Monterey County Water Resources Agency	1997	N/A	N/A	Ground Water Susceptibility
Groundwater Vulnerability Assessment: Hydrogeologic Perspective And Example From Salinas Valley, California	Graham Fogg, Eric Labolle, Gary Weissmann	1999	Finite Difference	Steady State	Nitrate Transport

Adminsitrative Record
Page 34973

3.3.4 Transport Modeling

Two regional scale groundwater vulnerability studies have been completed in the SV. The first was a Geographic Information System (GIS) based approach, which studied a portion of the SV from Chualar to Greenfield (Zidar 1997). The study used spatial maps of various parameters including agricultural use, soil texture, and depth to groundwater. These data were used with the Ground Water Loading Effects of Agricultural Management Systems (GLEAMS) model. The GLEAMS model is a root zone water quality model that provides an assessment of shallow groundwater vulnerability to contamination. Because the transit times of deeper groundwater from the surface to pumping wells are generally on the order of decades to centuries, the GLEAMS model does not account for effects of deeper hydrogeologic conditions on aquifer vulnerability to contamination. In addition to nitrate, other applied agricultural chemicals, such as pesticides, were investigated as part of the study.

A second vulnerability study (Fogg et al. 1999) used regionally detailed, 3D models of the subsurface heterogeneity to evaluate vulnerability of the deeper groundwater (which supply the irrigation and drinking water wells) to contamination from the surface. Backward-in-time, random-walk particle tracking was used to generate groundwater age distributions for groundwater produced by wells at a depth of 55 m (180-ft aquifer). Age distributions for individual wells were split into young (post WWII industrial production and wide spread use of artificial fertilizers) and old water (pre-industrial production of fertilizers). Wells with higher fractions of “young water” were assumed to represent wells which likely contained contaminated agricultural return water. Using a 50-year simulation, those wells which contained significant portions of young water were compared with current distributions of nitrate throughout the SV. Figure 19 shows that the simulated distribution of young water compared reasonably well with the nitrate spatial distribution in 1988 (concentrations shown are nitrate - as nitrate); wells that contained high fractions of young water were located in areas with known nitrate contamination. This demonstrated that, in addition to investigating surface processes, aquifer heterogeneity is a crucial component to determining the vulnerability of groundwater to contamination from the surface.

Adminsitrative Record
Page 34974

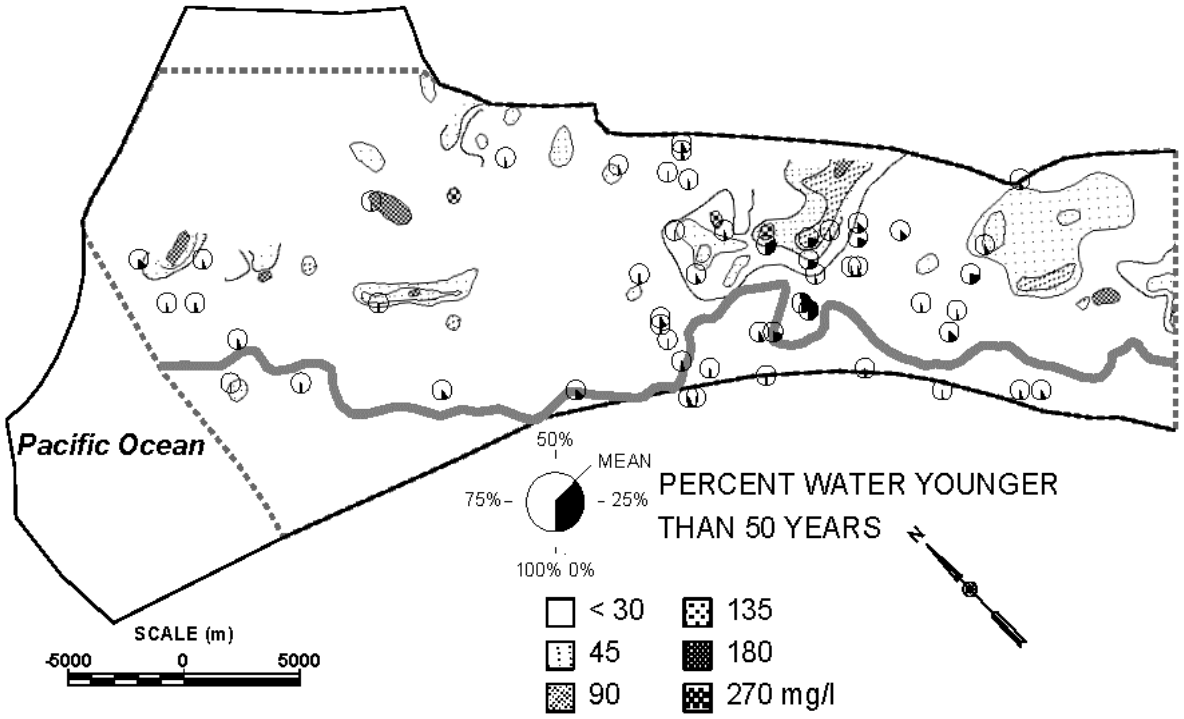

Figure 19. Wells showing (in black) the fraction of simulated young water less than 50 years overlain upon measured nitrate concentration contours. (Source: Fogg et al. 1999.)

Adminsitrative Record
Page 34975

3.4 Water Quality

In the SV basin, elevated levels of nitrate have been detected in groundwater since the 1950's (Zidar & Thomasberg 1995). Since the advent of synthetic fertilizers in the post WWII era, intensive agriculture has been the dominant form of land use in the SV. Nitrate travels at the same rate as groundwater and is often the first indication that a well may be compromised by contaminated groundwater recharge. In 1978, a study by the Association of Monterey Bay Area Governments (AMBAG 1978) determined that elevated nitrate levels in wells were primarily the result of agricultural practices at the surface, and recommended establishing a monitoring network for nitrate and other chemicals of interest (Snow et al. 1988). In response to this, MCWRA, working with the DWR developed an ambient monitoring program consisting of approximately 450 wells, the majority being irrigation wells. In addition to MCWRA's monitoring network, nitrate levels are regularly sampled from public wells to ensure that water meeting the U.S. EPA drinking water guidelines is being delivered. The Monterey County Department of Environmental Health (MCDEH) mandates that any water supply system with two connections or more must be tested annually. At the state level, systems with 15 or more connections (or serving more than 25 people for more than 60 days out of the year) are required to be tested annually.

3.4.1 Nitrate Data Collection

There are three programs that currently sample groundwater at the basin scale to assess the state of nitrate contamination in the SV. The first is the synoptic sampling of irrigation wells by MCWRA mentioned above. The second program, also run by MCWRA, samples a network of approximately 40 monitoring wells that are installed throughout the valley. The third is a program where public water systems are required to systematically test their well water, the results which are reported to MCDEH. In the past, a program by DWR (Bulletin 130 series) sampled irrigation, domestic, and public supply wells in the 1960's, the 1970's, and 1985 in 5 hydrologic regions of California (California Department of Water Resources 1963-1974, 1985). The data pertaining to the SV has since been incorporated into MCWRA's database. More recently, in 2005, a GAMA study by the USGS sampled public supply wells throughout the basin as part of a larger Priority Basins project. In 2011, Monterey County domestic wells were sampled for the GAMA Domestic Well program (http://www.swrcb.ca.gov/gama/domestic_well.shtml).[5]

3.4.2 Nitrate Analysis Reports

In the Central Coast region, Anton et al. (1988) describes the SV as severely impacted by nitrate. The report states that 113 of 1,207 wells of small water systems were in violation in 1986, and 6 of 180 wells of large systems were "closed" due to nitrate. In the unconfined aquifer regions of the SV, 48% of monitored wells exceeded the nitrate MCL, according to an unnamed study Anton mentions. The report also mentions that another study had predicted that all the unconfined aquifer regions of the SV would

[5] Data from this survey were not available until 2012 and are not included in the CASTING dataset described in Section 4.

Adminsitrative Record
Page 34976

have nitrate concentrations of 1.9 to 4.4 times the MCL by the year 2000. A study in the Prunedale area of Monterey County found that 27% of the 154 "private and public" wells in the area were over the MCL for nitrate, and that both shallow and deep wells were affected.

The two *Nitrates in Groundwater* studies (Snow et al. 1988; Zidar & Thomasberg 1995), and Technical Memorandum (2010) in Table 7 are from the synoptic well sampling program performed by the MCWRA. The analyses largely consist of irrigation well sampling; however, they also include the 40 dedicated monitoring wells operated by MCWRA. The results from the studies are briefly discussed in Section 3.4.3 *Nitrate in Groundwater,* where the mean concentrations from each report are plotted in Figure 20. The two *Water Resources Data Reports* listed in Table 7 (Monterey County Water Resources Agency 1996, 1997) are from the same sampling program; however, they provide a snapshot of a particular year, rather than analyzing the changes in nitrate concentration over a time period.

As part of the California State Water Resources Control Boards' GAMA Priority Basin Project, the USGS collected water quality, isotopic, and age tracer samples from public supply wells in 2005. Two reports have been published as a result of this sampling: a data summary report in 2007 (Kulongoski & Belitz 2007) and an interpretive report in 2011 (Kulongoski & Belitz 2011). Water supply systems with two or more connections require annual testing in Monterey County, and the results are reported to Monterey County Public Health. There are currently no reports regarding spatial and temporal trends in public supply wells of the SV; however, data from the reports found in Table 7 and public supply well data from MCDEH are included in the CASTING database. A description of the database and all of the data included in it can be found in Section 4. An analysis of the database is provided in Section 5.

Adminsitrative Record
Page 34977

Table 7. Groundwater sampling reports for the SV.

Title	Year(s) Published	Agency
Bulletin 130 Hydrologic Data: Central Coast Area	1963-1975, 1985	DWR
Nitrates in Groundwater 1978-1987	1988	MCWRA
Nitrates in Groundwater 1987-1993	1995	MCWRA
Water Resources Data Report, Water Year 1993-1994	1996	MCWRA
Water Resources Data Report, Water Year 1994-1995	1997	MCWRA
Ground-Water Quality Data in the Monterey Bay and Salinas Valley Basins, California, 2005—Results from the California GAMA Program*	2007	SWRCB
Technical Memorandum 1993-2007	2010	MCWRA
*An additional report (Kulongoski & Belitz 2011) provides an interpretation of the data in this report.		

3.4.3 Nitrate in Groundwater

While some well samples contain nitrate concentrations that are many times greater than the drinking water standard (MCL), the majority of the public supply wells in the SV have concentrations below the MCL (see Section 5 *Analysis of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater Database*). This is to be expected, given that any well with more than 2 connections is tested annually in Monterey County and reported to the MCDEH. If the nitrate concentration is found to be above 50% of the MCL, the well is required to be sampled every 3 months. Once the MCL of a particular contaminant is exceeded, wells are often destroyed, abandoned, or their use is discontinued. Once a well is found to be contaminated, further sampling is often not performed due to its destruction/abandonment. This effectively removes potentially high nitrate samples from the record, maintaining the statistic that the majority of wells sampled are below the MCL.

Of the 34 wells sampled for the GAMA program (31 public supply and 3 monitoring wells), nitrate plus nitrite (as nitrogen) was detected in 24 wells, with two (8%) of the samples being over the MCL (Kulongoski & Belitz 2011). This is consistent with our analysis of state and local public supply wells from the CASTING database, which show that generally one out of ten public supply wells are above the MCL. While an analysis of public supply wells provides information regarding the quality of water which is delivered for the purpose of drinking, a more insightful picture of the overall groundwater quality can be obtained through the synoptic well sampling program of MCWRA. This program samples irrigation wells on an annual basis, indiscriminate of the nitrate concentrations previously found. Some of the wells

Adminsitrative Record
Page 34978

have been sampled repeatedly for nearly 60 years and contain concentrations of nitrate many times over the MCL.

MCWRA's long-term irrigation well sampling database was unavailable to our research group for reasons of confidentiality, nevertheless three reports analyzing this data and apparent trends have been published by MCWRA (Snow et al. 1988; Zidar & Thomasberg 1995; Monterey County Water Resources Agency 2010c). The mean (average) concentrations are plotted in Figure 20. The data show a general increase in nitrate concentrations through time for the valley, although the latest data for the Upper Valley subarea suggest that, on the average, concentrations did not changed significantly from 1993 to 2007. The Pressure subarea confined aquifers (Pressure 180, Pressure 400, and Pressure Deep) have the lowest average concentrations of nitrate compared to the other subareas of the basin, due to semi-continuous clay layers (aquitards) between the aquifers and the surface, slowing the transport process. The regression lines in Figure 20 show different rates of increase for nitrate concentrations ranging from 0 mg/L/year (Pressure Deep) to 2.23 mg/L/year (Upper Valley). The dip in values for 1993 is possibly explained by the number of wells analyzed for that particular year, roughly twice the amount as other years (Table 8).

While the SV has been home to many different land uses in the past and present (animal grazing, diary farms, vegetable row crops, pasture, etc.) LLNL found that the largest source of nitrate contamination to groundwater was from irrigated agriculture. Geochemical and isotopic results from groundwater and surface water samples containing nitrate above an established background level indicated that the nitrate contamination had a signature consistent with inorganic fertilizers (Moran et al. 2011).

Adminsitrative Record
Page 34979

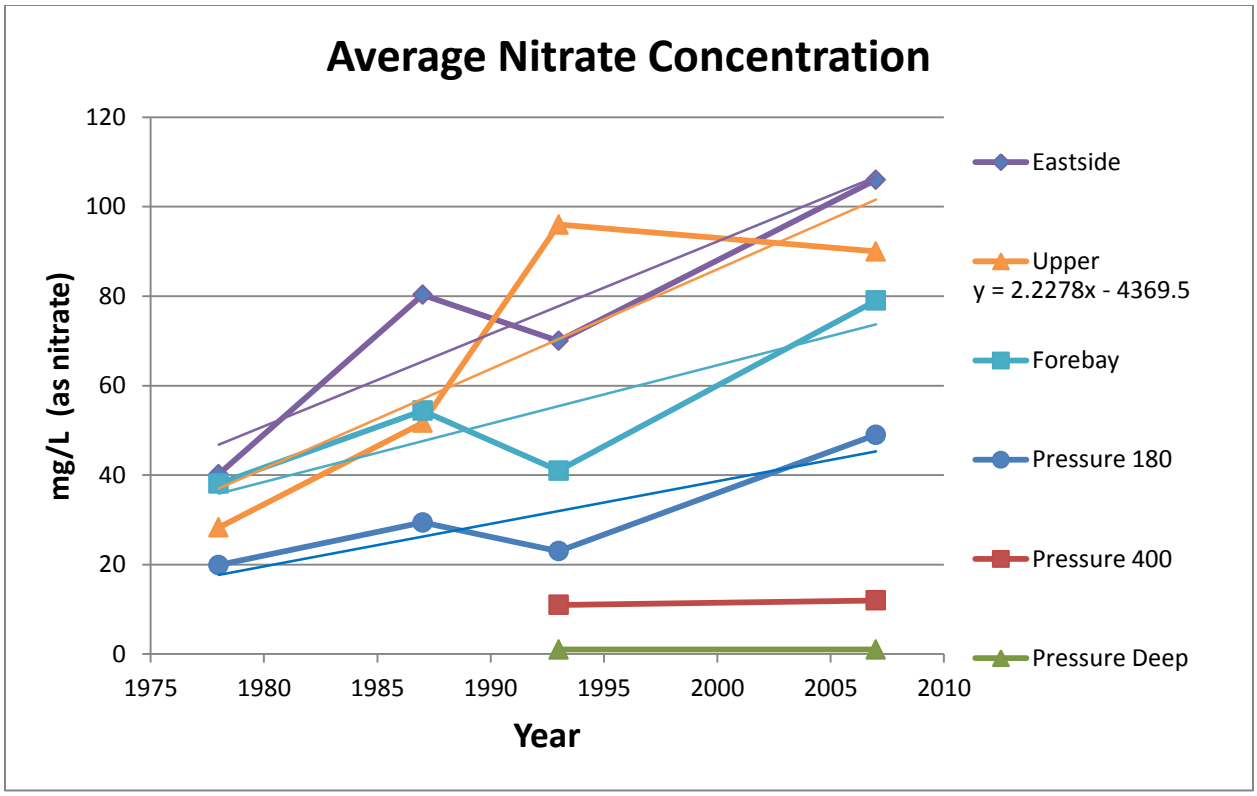

Figure 20. Nitrate trends from MCWRA synoptic sampling program. Values are average nitrate concentrations for each subbasin for 1978, 1987, 1993, and 2007. (Source: Snow et al. 1988; Zidar & Thomasberg 1995; Monterey County Water Resources Agency 2010c)

Adminsitrative Record
Page 34980

Table 8. Number of wells sampled by subarea and year as reported in MCWRA's *Nitrate in Groundwater* Reports.

SUBAREA	1978	1987	1993	2007
Pressure – 180 ft aquifer	61	61	95	28
Pressure – 400 ft aquifer	N/A	N/A	98	44
Pressure – Deep aquifer	N/A	N/A	6	5
Eastside	40	40	64	15
Forebay	34	34	73	41
Upper	16	16	34	19
TOTAL	**151**	**151**	**370**	**152**

3.4.4 Additional Reports

Table 9, Table 10, and Table 11 include a list of additional reports provided by MCWRA concerning the geology, water supply, and water quality in the SV. MCWRA has the following reports archived and is the organization to contact for obtaining those documents listed below that cannot be found through conventional sources.

The GAMA reports discussed here and in section 2 are those which specifically pertained to our study area. For more information on SWRCB's GAMA program, and additional reports, the reader is referred to SWRCB's GAMA website http://www.waterboards.ca.gov/gama/.

Adminsitrative Record
Page 34981

Table 9. Nitrate/water quality studies for the SV.

Title	Year	Author	Prepared For
Ground Water in California with Particular Reference to Salinas Valley	1949	T.R. Simpson	
CLEAN WATER: Water Quality Management Plan for the Monterey Bay Region	1978	Association of Monterey Bay Area Governments (AMBAG)	Monterey and Santa Cruz Counties
Nonpoint Sources Of Groundwater Pollution in Santa Cruz and Monterey Counties, California	1978	H. Esmaili and Associates, Inc.	
Report Of The Ad Hoc Salinas Valley Nitrate Advisory Committee	1990	Zidar et al. (1990) Monterey County Flood Control & Water Conservation District	
Demonstration Program to Reducing Nitrate Leaching Through Improvements to Irrigation Efficiency and Fertilizer/Cover Crop Management	1993	Varea-Hammond (1993), Monterey County Agricultural Extension Service University of California	MCWRA
Reducing Nitrate Leaching Through Improvements to Irrigation Efficiency And Fertilizer Management: 205 (J) Phase V Final Report to the State Water Resources Control Board	1993	Zidar et al. (1993), Monterey County Water Resources Agency	State Water Resources Control Board
Salinas Valley Ground Water Basin Seawater Intrusion Delineation/Monitoring Well Construction Program	1993	Staal, Gardner, and Dunne Inc.	
Wellhead Protection for Rural Communities Facing Threats from Nonpoint Source Nitrate Contamination, Case Study, King City, Salinas Valley, California	1995	Zidar (1995a), Monterey County Water Resources Agency	MCWRA
Irrigated Agriculture Technical Advisory Committee Report to the California State Water Resources Control Board	1994	Technical Advisory Committee	State Water Resources Control Board
Salinas River Basin Management Plan: BMP Task:2.06.1 Water Quality Assessment Report	1995	Win (1995)	MCWRA
Northern Salinas Valley Watershed Restoration Plan: Nonpoint Source Pollution in Coastal Harbors and Sloughs of the Monterey Bay Region: Problem Assessment and Best Management Practices	1997	Barron (1997), Moss Landing Marine Laboratories	Association of Monterey Bay Area Governments
2001 Nitrate Management Survey Results Report	2002	Monterey County Water Resources Agency	
Implementation of Public Outreach and Education Elements of the Salinas Valley Nitrate Management Plan	2003	Thomasberg (2003), Monterey County Water Resources Agency	Central Coast Regional Water Board State Water Resources Control Board
California GAMA Special Study: Nitrate Fate and Transport in the Salinas Valley	2011	Moran et al.	State Water Resources Control Board

Adminsitrative Record
Page 34982

Table 10. SV Basin studies.

Title	Year	Author
Salinas Basin Investigation	1946	Simpson (1946), Department of Public Works, Division of Water Resources
Data Report On Phase I - Salinas Valley Groundwater Basin Management Program	1974	Ott (1974a), Monterey County Flood Control and Water Conservation District
Summary Report On Phase I – Salinas Valley Groundwater Basin Management Program	1974	Ott (1974b), Monterey County Flood Control and Water Conservation District
Groundwater Management Measures for the Salinas Valley Corps of Engineers Urban Studies Report	1979	Army Corps Of Engineers
Optimal Conjunctive Use of Groundwater and Surface Water in the Salinas Valley	1985	Yates
Reconnaissance Report: Salinas Valley Water Transfer Project	1992	Laska
Groundwater Extraction Management Study Report	1992	Hurst (1992), MCWRA
Marina County Water District, Salinas Valley Groundwater Basin Overview	1992 (Draft)	Nolte and Associates
Water Resources Management in the Salinas River Basin. Volume 2: Review of Studies and Plans	1992	Price
Salinas River Basin Water Resources Management Plan study: Task 2.06.1 Report Water Quality Assessment	1992	Silas Snyder Larry O'Hanlon
Hydrogeology and Water Supply of Salinas Valley (White Paper)	1995	Salinas Valley Ground Water Basin Hydrology Conference
North Monterey County Hydrogeologic Study Volume I: Water Resources	1995	Zidar
North Monterey County Hydrogeologic Study Volume II: Critical Issues Report and Interim Management Plan	1996	Zidar
North Monterey County Water Issues Action Plan	1997	Barron
Monterey County Groundwater Management Plan	2006	MCWRA
Salinas Valley Water Project	2007	MCWRA

Adminsitrative Record
Page 34983

Table 11. Geological studies of the SV Basin.

Title	Year	Author	Prepared For
Geology of the Lower Portion, Salinas Valley Ground Water Basin	1969	Ford (1969), DWR	
Geology of Southern Monterey Bay and its Relationship to the Ground Water Basin and Salt Water Intrusion	1970	Green (USGS)	
Selected Geological Cross Sections in the Salinas Valley Using GEOBASE	1992	Hall	
Hydrogeologic Assessment - Salinas River Basin Management Plan Salinas Valley Geohydrologic Study	1994	Fugro West, Inc.	Montgomery Watson
Hydrogeologic Assessment. Salinas River Basin Management Plan Salinas Valley Geohydrologic Study: Construction Of Monitoring Well Clusters.	1995	Fugro West, Inc.	Montgomery Watson
Hydrogeologic Investigation of the Salinas Valley Basin in the Vicinity of Fort Ord and Marina Salinas Valley, California	2001	Harding ESE	

3.4.5 Nitrate Transport - Aquifer Heterogeneity & Groundwater Age

Rates of water movement are commonly slower in the unsaturated zone as compared to groundwater because the former have much lower hydraulic conductivity values than their saturated equivalents (Radclifee & Simunek 2010). A modeling study of vadose zone transport at the Wing Ranch in the SV showed a range of modeled travel times for bromide, a conservative tracer. Based on different heterogeneity realizations, travel times from the surface to the water table (35 m or 115 ft) were estimated to be from 4 to 14 years, depending on the framework of the subsurface sediments (Burow 1993). An approximate travel time of 10 years at that location was assumed reasonable. Nitrate is usually assumed to behave conservatively in the subsurface as well, and would be expected to have similar travel times. As a part of this study, advective travel times through the vadose zone were estimated on a regional scale, providing a spatial distribution of minimum and maximum travel times for the entire study area. The study involved 1D transport modeling with program HYDRUS 1D. Rather than explicitly defining the local heterogeneity of the unsaturated soils, the model was run with three homogeneous cases: sand, loam, and clay soil, thought to represent the fastest, intermediate, and longest probable travel times. Average travel times for the entire SV, based on the sand, loam, and clay soils are 5.57 years, 16.97 years, and 23.83 years, respectively. The maps and a description of the methods can be found in Section 6 *Groundwater Nitrate Forecasting: Assessment of Vadose Zone Nitrate Transport*.

After reaching the water table, water can take decades to centuries to migrate to deeper production wells (Fogg et al. 1999). Furthermore, wells draw in a mixture of waters that may have a broad distribution of ages, often ranging from years to centuries. Because of this, groundwater pumped by wells typically consists of both post-agricultural development groundwater that post-dates the

Adminsitrative Record
Page 34984

introduction of large-scale use of fertilizer and irrigation, and pre-agricultural development groundwater that pre-dates the agricultural contaminant sources. One can think of the pumped groundwater as a mixture of old, pre-1940s-1950s, uncontaminated water and young, post-1950s, potentially contaminated groundwater. As time continues, the fraction of the pumped groundwater that is "young" or post 1950s increases, and the potential for upward trending nitrate concentrations increases if the sources of nitrate contamination have not diminished appreciably. Water found at shallow depths commonly has larger fractions of young water, and thus higher concentrations of nitrate associated with the recent recharged water. Local geology can often have a significant influence in the distribution of ages found in water due to areas of high hydraulic conductivity and areas of low conductivity (Fogg et al. 1999). The interconnectedness of the high conductivity sediments can lead to faster travel times, while areas of low conductivity can impede groundwater and associated contaminants. This explains, to some degree, why some deeper wells can have greater concentrations of nitrate than shallower wells. As discussed previously in Section 3.3.4, the vulnerability map of Fogg et al. (1999) showed that the "percent of young water" at a depth of 55 m (180 feet) compared favorably when overlain on top of a map of actual nitrate occurrence. Wells containing higher fractions of young water appear to roughly match the occurrences of higher nitrate concentrations found from well sampling (Figure 19). Denitrification reduces nitrate concentration. In the Salinaas Valley, denitrification has been found to occur in the immediate vicinity of the Salinas River and its contact with groundwater, and is likely to also occur in the clay layers confining the aquifers of the Pressure area. The potential role of denitrification is further explained in Appendix B of this report.

3.4.6 Groundwater Chemistry

Hydrogeologic characterization of groundwater includes major ion chemistry and its evolution along groundwater flow paths. The major ions present in water are calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), chloride (Cl^{-}), sulfate (SO_4^{-}), carbonate (CO_3^{2-}), and bicarbonate (HCO_3^{-}). The relative amounts of each constituent, when plotted on a piper diagram (Figure 21), provide information on regional flow paths and origins of the water. For example, Ca^{2+} and Mg^{2+} concentrations are often elevated in shallow groundwater and decrease along flow paths due to exchange for Na^{+} present on clay-rich sediments, resulting in more Na^{+} and less Ca^{2+} and Mg^{2+} the longer water has spent traveling through the subsurface. Additionally, groundwater tends to continuously dissolve carbonate and other minerals found naturally in geological materials as it travels through the aquifer system. This results in greater amounts of HCO_3^{-} in older groundwater.

Both trends (exchange of Na^{+} and increasing HCO_3^{-}) can be seen in the water samples taken from municipal wells in the SV. Before the development of groundwater resources in an alluvial-fill basin like the SV, the predominant direction of groundwater flow is usually horizontal, with relatively small vertical flow. Water tables also usually follow the slope of the surface elevation. Therefore, we would expect to find younger waters in the shallow unconfined Upper Valley subarea, and progressively older (and geochemically evolved) water to be found further down the valley.

Adminsitrative Record
Page 34985

Piper plots were constructed from samples collected from public supply wells (Figure 21) and reflect the expected geochemical evolutionary trend. In the Upper Valley, all water samples fall in the upper left portion of the diamond graph, meaning higher proportions of Ca^{2+} and Mg^{2+} (relative to Na^{+}) as well as lower proportions of HCO_3^{-}. The diagram suggests that the water here is relatively "young." Moving into the Forebay subarea, the data display a bit more scatter to the plot (some samples plotting further down and to the right), signifying that this water is more geochemically evolved and has traveled in the subsurface for a longer period of time. However, it is not until the relatively deep Pressure and Eastside aquifers that the more geochemically evolved water appears. The samples in the Pressure and Eastside aquifers, in general, plot much further down and to the right with less Ca^{2+}/Mg^{2+}, more Na^{+}/K^{+}, and more HCO_3^{-}. These relative proportions suggest older water, which fits our conceptual model of the geochemical evolution of groundwater in the SV where we expect to find older, more evolved water in the lower portion of the valley. The degree of scatter in the data also tells us something of the heterogeneous nature of the groundwater system in the SV. In the semi and unconfined aquifers of the Forebay and Upper Valley subareas we see primarily younger water, with less variability in their chemical evolution, which suggest faster travel times to supply wells. The Pressure and Eastside subareas show more scatter to the data which implies that some wells in these subareas are withdrawing very old water, while others are withdrawing younger water. The presence of confining conditions in these areas tends to cause longer travel times, while the heterogeneous nature of aquifers create a wide range of travel times.

Nitrate concentrations are also shown in Figure 21. Concentrations above 22.5 mg/L nitrate-nitrate (red, purple, and blue), generally plot in the upper portion of the diagram. Concentrations below this (green and yellow) plot further down. Groundwater containing nitrate concentrations over 22.5 mg/L likely has a large fraction of young contaminated water, which is reflected in the location where it plots in Figure 21. The location of groundwater samples containing nitrate concentrations below 22.5 mg/L on the piper diagram suggests that this water contains a higher fraction of older, pre-industrial, recharge water that has not been contaminated by recent (post-1950's) agricultural activities.

Adminsitrative Record
Page 34986

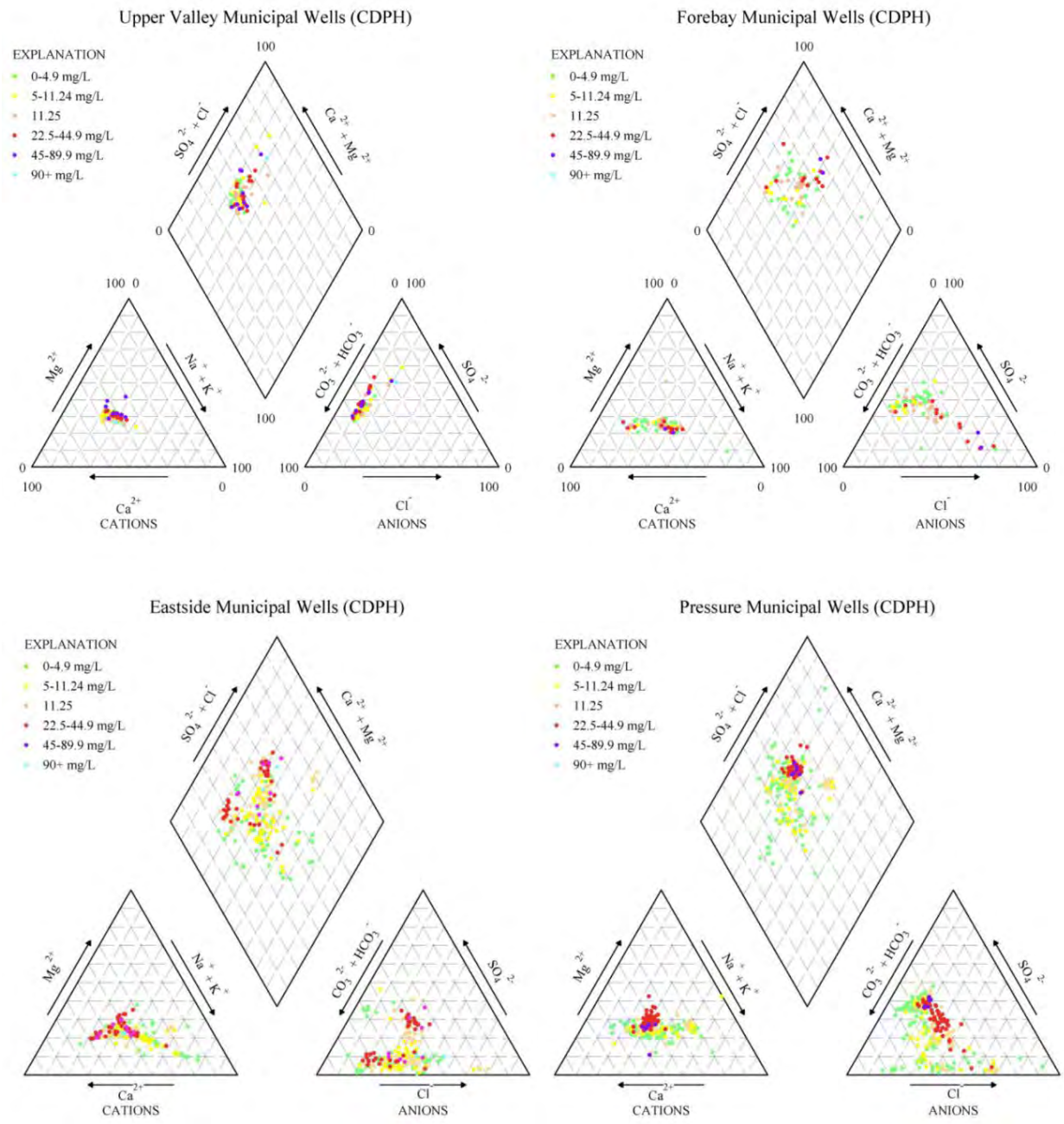

Figure 21. Piper diagrams of public supply water systems in the SV. Concentrations are nitrate as nitrate. Data is from the CDPH STORET database.

Adminsitrative Record
Page 34987

4 Development and Description of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database

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Adminsitrative Record
Page 34988

4.1 Introduction

4.1.1 Objectives

The objective of Section 4 is to describe the origins, development, organization, and scope of the database used by this study to examine current, historical, and rate of change of nitrate concentration in groundwater in the study area, the TLB and the SV. We acquired nitrate test data for groundwater in our study regions from federal, state, and many local agencies and reports; attempted to geo-locate every tested well, if at all possible, with an estimate of the spatial precision of the geo-location; used multiple, redundant methods to check for duplication within and between data sources; and created a geographic information system (GIS) database focused on sample x, y, depth, time, and nitrate concentration. In this section we describe the source datasets and document the protocols used to assemble the data into a single database, herein referred to as CASTING (California Ambient Spatio-Temporal Information on Nitrate in Groundwater). A discussion of the spatio-temporal distribution of available data on nitrate in groundwater is presented.

Note that well log information is confidential by current state law. Information from well completion reports and on the exact location of public or private wells will therefore not be made public.

4.1.2 Background

Nitrate test data are collected by federal, state, county, and local agencies for regulatory and scientific purposes. Agencies from which we obtained data for our study area include the United States Geological Survey (USGS), the United States Environmental Protection Agency (EPA), the United States Bureau of Reclamation (BOR), the California Department of Public Health (CDPH), the California Department of Water Resources (DWR), the California State Water Resources Control Board (State Water Board), the Central Coast (Region 3) and Central Valley (Region 5) Regional Water Quality Control Boards (Regional Water Boards), environmental health agencies of Fresno, Kern, Kings, Monterey, and Tulare Counties, Westlands Water District (WWD), the Kern County Water Agency (KCWA), and the Monterey County Water Resources Agency (MCWRA).

Under current drinking water regulations (U.S. Safe Drinking Water Act [SDWA] of 1974, amended 1986, 1996), the state and local regulatory agencies require public water supply system operators to regularly test their raw water and their finished (treated) water provided to households for biological, chemical, and radiological constituents, including a test of nitrate concentration. Water systems report their water quality data to the local agency or directly to CDPH, depending on their size. Nitrate concentration in drinking water is regulated at three levels in California. These regulatory levels are defined by the number of people and number of days served by a system, and by the number of connections served by each water system. Water systems with 15 or more connections or serving more than 25 people for more than 60 days a year are considered public water systems (PWS) under the federal Safe Drinking Water Act, and are regulated by CDPH.

Adminsitrative Record
Page 34989

The U.S. EPA distinguishes the following types of PWSs:

- Community Water System (CWS): A public water system that supplies water to the same population year-round.
- Non-Transient Non-Community Water System (NTNCWS): A public water system that regularly supplies water to at least 25 of the same people at least six months per year, but not year-round. Some examples are schools, factories, office buildings, and hospitals which have their own water systems.
- Transient Non-Community Water System (TNCWS): A public water system that provides water to 25 or more people, 60 days or more per year, in a place such as a gas station or campground where people do not remain for long periods of time.

CDPH uses the U.S. EPA definitions as well as further classifying water systems by number of persons and number of service connections (Table 12).

Table 12. Classification of water systems by constituency, connections, and duration of service per year.

<table>
<tr><td></td><td colspan="2">Connections:</td><td><5</td><td>5+</td><td><15</td><td>15+</td><td><200</td><td>200+</td></tr>
<tr><td>Duration of Service</td><td colspan="2">Persons served:</td><td colspan="3"><25</td><td colspan="3">25+</td></tr>
<tr><td rowspan="3">N/A</td><td rowspan="3">Small Water System (SWS)[1]</td><td rowspan="4">Classification defined by</td><td colspan="5" rowspan="3">connections</td><td rowspan="3"></td></tr>
<tr></tr>
<tr></tr>
<tr><td rowspan="3"><60 days/year</td><td rowspan="2">Local Small Water System (LSWS)</td><td rowspan="2">connections and (persons, duration)</td><td rowspan="2"></td><td colspan="3" rowspan="2"></td></tr>
<tr></tr>
<tr><td>State small Water System (SSWS)</td><td></td><td>connections and (persons, duration)</td><td colspan="3"></td></tr>
<tr><td>≥60 days/year</td><td>Community Public Water System (CPWS)[2]</td><td colspan="3"></td><td colspan="3">connections or (persons, duration)</td></tr>
</table>

[1] Classification as a SWS does not preclude classification as any of the other types. SWSs may be regulated by CDPH or by LPA.

[2] A CPWS is a system for the provision of water for human consumption that has 15 or more service connections OR regularly serves 25 individuals at least 60 days a year (CDPH 2010 b,c).

From Table 12, systems with fewer than 200 connections are considered small water systems (SWSs). Under the provisions of Section 116330 of the California Health and Safety Code, CDPH has delegated regulatory authority over SWSs to 35 counties (called local primacy agencies, or LPAs). In our project area, the county environmental health departments of Kings County, Monterey County, and Tulare County are LPAs, but neither Fresno County nor Kern County have local primacy (i.e., in these counties, CDPH regulates SWSs). Fresno County had LPA status until 2006, when they relinquished that responsibility to CDPH.

Adminsitrative Record
Page 34990

Drinking water systems serving between five and 14 connections are referred to as "state-smalls," and are regulated by the county, by CDPH, or not at all, depending on the county. Systems serving fewer than five connections are referred to as "local-smalls" and are regulated at the discretion of each county.

In general, household (domestic) wells are not regulated, although in the last decade many counties have begun to collect a single nitrate test from each newly drilled drinking water well at the time the well is drilled, including domestic wells (in our study area, this includes Fresno, Tulare, Monterey, and Kern counties).

CDPH maintains the largest portion of the regulatory test result data in the Permits, Inspections, Compliance, Monitoring, and Enforcement system information database (PICME). PICME includes data collected on all PWS sources of drinking water - surface water or groundwater, treated or untreated, active or inactive. Samples are collected by licensed water system operators and taken to certified laboratories that upload the resulting water quality data to PICME. The PWSs listed in PICME serve drinking water to approximately 95% of California's residents.

Besides the public health regulatory agency data, there are several other sources of nitrate test results for wells in our study area, which are summarized below. For each of these data sources we include information on the specific intention/purpose of the program, the scope of the program (where/how data were collected, the targeted group of wells, timeframe of monitoring), and the types of information included in the database. These data were compiled, reviewed, and assembled into a new database that is here referred to as the CASTING (California Ambient Spatio-Temporal Information on Nitrate in Groundwater) database. The CASTING database serves as an intermediate electronic storage format from which data will be transferred to Geotracker GAMA, the State Water Board's groundwater quality database. The following sections review the data sources for CASTING, the procedures used for assembling the various data into CASTING, and the quality assurance/quality control procedures that were applied.

4.1.2.2 Privacy Concerns

Currently, neither the locations, nor the physical characteristics of privately owned wells (whether part of public water systems or private domestic drinking water systems or household wells) are public information. Although DWR maintains copies of all well driller logs, that detail the engineering, geology, and location of all drilled wells in the state, these logs are themselves not public information. In the process of our analysis, we obtained permission to access many well driller logs, as well as precise locations and other private information from various agencies. The permission we obtained included license to use the data, but not to disseminate the data. Therefore, precise locations and physical characteristics of particular wells are not disclosed within this report.

Adminsitrative Record
Page 34991

4.1.3 Data Sources

CADWSAP (California Drinking Water Source Assessment Program)

Purpose – Under the federal Safe Drinking Water Act (SDWA), source water assessment programs (SWAP) for surface and groundwater were mandated to produce geo-locations (typically as geographic positioning system (GPS) coordinates) and vulnerability assessments for all active drinking water sources. CADWSAP is the California implementation of the federal SWAP program. The CADWSAP database is housed at the Information Center for the Environment (ICE) at UC Davis, and consists of a subset of PICME data that is intended to include all active source waters, with GPS coordinates for each water source location.

Scope – Within our study area, CADWSAP provided 3,616 wells with 60,792 individual nitrate samples dating from the early 1980's to 2010.

Data – In CADWSAP, nitrate data are reported as NO_3-N, NO_3-NO_3, or NO_3-N + NO_2-N combined, all in mg/L. These wells are all geo-located with GPS and are considered to have high spatial precision (better than 1 m accuracy for spatial location). Many of these well records also include the depth to the top of the well screen and the length of the well screen (perforated interval). CADWSAP wells are identified by a unique Primary Station Code (PS-Code) used only by CDPH and contributing counties. The format of the CADWSAP database provided the template for the CASTING database.

GAMA Domestic Well Project

Purpose – The GAMA program was established under the California Groundwater Quality Monitoring Act of 2001 (AB 599 2001) to provide a statewide comprehensive assessment of groundwater quality. The GAMA program is divided into three projects: the domestic well water quality (this section), priority basin water quality (next section), and special studies. No special studies projects were completed within our study area by the date at which we stopped adding datasets, however, several special studies were completed in 2011[6] that are not included in our database. The GAMA Domestic Well Project, sampled for nitrate (and other constituents) in household wells starting in 2002. The objective of the project is to develop a baseline for assessment of drinking water quality in household wells.

Scope – The domestic well project has collected data in six counties to date, of which Tulare County, (sampled in 2006) and Monterey County (sampled in 2011) are in our study area. As with the special studies, the Monterey County Domestic Well study was not completed before our final cutoff date for

[6] 1). California GAMA Special Study: Nitrate Fate and Transport in the Salinas Valley Jean E. Moran*, Bradley K. Esser, Darren Hillegonds, Marianne Holtz*, Sarah K. Roberts, Michael J. Singleton, and Ate Visser.
2). California GAMA Program: Impact of Dairy Operations on Groundwater Quality. Bradley K. Esser, Harry R. Beller, Steven F. Carle, G. Bryant Hudson, Staci R. Kane, Roald N. Leif, Tracy E. LeTain, Walt M. McNab and Jean E. Moran. Lawrence Livermore National Laboratory, P.O. Box 808, Livermore, CA 94550.
3). Denitrification in a Shallow Aquifer Underlying a Dairy Farm: New Approaches to Characterization and Modeling: UCRL-PRES-207404

Adminsitrative Record
Page 34992

adding data into CASTING, and is not included here. The Tulare County project collected one sample per well and collected samples from 181 wells out of the approximately 26,000 domestic wells in Tulare county. Of these, 141 wells are within the study area.

Data –Highly accurate GPS data exists for these wells; however, for confidentiality reasons, wells are randomized to within ½ mile from the actual well locations. The nitrate concentrations reported by the GAMA program are in units of mg/L as nitrate. Well depths are available for some wells as reported by the well owners. The accuracy of the well depth information is uncertain. In general, domestic wells are drilled to depths less than 100 m (300 ft). GAMA Domestic Well Project wells are identified by a GAMA ID assigned by the State Water Board.

GAMA Priority Basin Project

Purpose – The GAMA Priority Basin Project, conducted by the USGS in coordination with the State Water Board (see previous section), was designed to assess water quality conditions in key groundwater basins (priority basins) that provide over 90 percent of all groundwater use in the State. This project focuses on the quality of ambient groundwater in aquifers used for drinking water supply and to establish a baseline groundwater quality monitoring program.

Scope – The Priority Basin Project has sampled wells from 116 groundwater basins (as defined by DWR Bulletin 118), divided into 35 study units. The Southern San Joaquin Valley study unit that overlapped our study area was completed in 2006. The Monterey/Salinas study unit was completed in 2005. The wells for the Priority Basin Project were sampled one time each, and consist of water supply wells, groundwater monitoring wells, and irrigation wells. In these Priority basin study units, samples were collected from 141 wells, 83 of which are located within our study area. Groundwater quality samples were collected and analyzed, at low detection limits, for a wide variety of natural and anthropogenic chemical (including nitrate), bacteriological, and radiological constituents. The Western San Joaquin Valley study unit was completed in 2010 and the data are not available as of this writing.

Data – These wells were geo-located using field GPS by USGS field personnel and are assumed to be accurate to within 7 m (20 ft). Nitrate concentrations in this dataset are reported in mg/L as nitrate. Well construction information was available for these wells in most cases. The construction information was obtained by the USGS for the Priority Basin Project from DWR well-logs. CADSWAP depth data was used in cases where construction data were not available in the Priority Basins reports. Priority Basin wells are identified by a unique ID code assigned by USGS.

USGS - NWIS

Purpose – The National Water Information System (NWIS) is the general clearinghouse database of locations for the USGS. The purpose of the database is to maintain continuity of identification and attribution of data collections across the many studies conducted by USGS on wells and other locations.

Scope – Within our study area, NWIS included 1,735 wells with 3,850 test results. NWIS identifies wells by the state well number (SWN). About 450 of the NWIS well SWNs matched wells already in the

Adminsitrative Record
Page 34993

CASTING database. Since many agencies and programs, notably the CDPH database and the Central Valley Regional Water Board (CVRWB) dairy database, do not use the SWN, an unknown number of the NWIS wells may overlap with other wells in CASTING. The tests are unlikely to be duplicates, as the tests in NWIS are only those collected by the USGS.

Data – NWIS wells are geo-located by various methods but these methods are not recorded in the database. It is therefore not possible to determine the accuracy of the reported geo-locations. The nitrate data are reported in mg/L as nitrate or in mg/L as nitrate-nitrogen. Nearly all of these wells included the depth of the completed well, but the top of the screened interval was not available.

Westlands Water District Irrigation Wells Project

Purpose – In 2008 and 2009, certain groundwater wells within the District were tested for water quality as part of the Canal Integration Program (CIP) and the Distribution Integration Program (DIP). The CIP allowed wells of sufficient water quality to pump into the San Luis Canal for distribution to downstream laterals. The DIP provided the growers an opportunity to pump water into the District's lateral for use on downstream lands served by the lateral. DWR was involved in a one-time sampling program designed to assess the water quality in irrigation wells in Westlands Water District. We obtained data from both Westlands Water District and from DWR.

Scope – The DWR project included a set of 39 wells with one water quality sample each, taken in 2008. All wells are located in Westlands Water District, 36 in western Fresno County and six in western Kings County. These wells were primarily irrigation wells and none were duplicates of CADWSAP wells. The Westlands Water District dataset included 84 water quality records from 71 wells, including 31 of the DWR wells.

Data –Well locations of 42 wells reported by DWR were geo-located with GPS by DWR and are accurate to within 7 m (20 ft). The nitrate concentrations in this dataset are reported in mg/L as nitrate-N, and were converted for our purposes to mg/L as nitrate. The remainder of the wells in the Westlands Water District database were identified by the State Well Number (SWN), defined below. No depth data were collected for these wells.

State Water Board Environmental Monitoring Wells

Purpose – The State Water Board maintains a large number of wells for monitoring of water quality at environmental cleanup sites. These wells are drilled specifically to monitor contaminated groundwater sites, such as leaky underground storage tanks, and other spills.

Scope – There are 537 monitoring wells with nitrate test results in our study area. Many other wells are monitored that do not have nitrate test results and these other wells were not included in our dataset. The monitoring wells are typically placed in clusters around cleanup sites, and are often not representative of regional groundwater quality.

Adminsitrative Record
Page 34994

Data – Nitrate concentrations in these wells are often quite high, as nitrate is frequently associated with the cleanup sites. These well locations are geo-located and are accurate to within 7 m (20 ft). The nitrate concentrations in this dataset are reported in mg/L as nitrate-N, and are converted for our purpose to mg/L as nitrate. These wells are identified by a unique code that is only used at the State Board, although a few also referenced the State Well Number.

U.S. EPA STORET Database

Purpose – The U.S. EPA warehouses data from many agency sources in the STORET legacy files database. These data were provided to the STORET facility by EPA's Region 9 water agency, the US Bureau of Reclamation, and California's State Water Board. Data from many different projects are collected in the STORET dataset and the original purpose of the underlying data collection efforts are not retained as an information item in the legacy database that we accessed for this project.

Scope – The U.S. EPA provided us with 16,031 samples from 7,083 wells. These wells were located throughout the TLB study area. For the SV, the Public Lands Survey System (PLSS) coverage of quarter-quarter sections is not available, which prevented us from locating wells using the State Well Number (SWN) system used in that study basin.

Data – These wells were not provided with accurate location data. However, all but 12 of the records included a SWN, so it was possible to locate the wells to the so-called quarter-quarter section, an area of 16.2 hectares (40 acres) comprising one of 16 (4x4) squares within a one-square mile (640 acre) land section. Only those wells with SWNs that matched a quarter-quarter section located within the study area were considered for inclusion in our database.

Without precise locations or any way of cross-referencing to other identification systems, it was unclear how many of the STORET tests and wells are duplicates of data that were already obtained from other databases (e.g., from PICME). To avoid duplication, we included only those wells from STORET into the CASTING database that were located in quarter-quarter-sections for which CASTING did not have a record of nitrate data. The STORET data do not include construction information such as well depth or screened interval. The test results are reported to U.S. EPA in several different concentration units, including mg/L NO_3-NO_3, mg/L NO_3-N plus NO_2-N, and mg/L NO_3-N. For use in CASTING, we converted all values to mg/L as nitrate.

Central Valley Regional Water Board Dairy Wells Monitoring Data

Purpose – Since 2007, the Central Valley Regional Water Board (CVRWB) has collected nutrient and chemistry data from irrigation, domestic, barn, and monitoring wells sampled by dairy owners. This groundwater monitoring and reporting program is the Central Valley Dairy General Order.[7]

[7] http://www.swrcb.ca.gov/rwqcb5/board_decisions/adopted_orders/general_orders/r5-2007-0035.pdf

Adminsitrative Record
Page 34995

Scope – We obtained the 2007, 2008, and 2009 data reported by dairies to CVRWB from the CVRWB Fresno office. Results are provided to CVRWB in annual reports provided by each dairy. The dataset includes 11,300 samples from 6,459 presumably unique wells sampled in 2007, 2008, and/or 2009.

Data – Data included dairy name and address, owner-selected names of wells, nitrate data, and date of the sample collection. No information was provided on the well construction details or well use. Well names in some - but not all - cases were indicative of the well use (e.g., "irrigation well #3"). Sample collection is once annually, but not necessarily in yearly intervals. Since well names are provided by the owners or their consultants, naming inconsistencies between years result in uncertainties about the identity of wells between years. For our analysis, we assumed that nitrate records were from the same well only if names between years were identical. These data were provided in spreadsheets. While Regional Water Board 5 also has maps for many of the dairies that identify the well location, these maps were not digitally geo-located for the SBX2 1 project. Instead, wells are associated with the address location and assumed to be within approximately 1 mile of that address. The only spatial reference available is the street address. It is unknown how many of these addresses represent the well locations and how many represent the business office or home of the dairy operator. Unfortunately, some dairies from Tulare and Kings Counties lacked the names of the cities, and therefore were not geo-located to the street address provided. Instead, those wells were located at the county seat – Visalia in Tulare County and Hanford in Kings County. Data provided included results in mg/L NO_3-N, NO_3-NO_3, and total N, all of which were converted to mg/L nitrate prior to inclusion into CASTING. Some samples were described as surface water sources; these were ignored for the purpose of this study. Some samples were reported as “DRY,” and these were also ignored. There were 2,224 samples reported both as NO_3-N and as NO_3-NO_3. Most of these samples were related by the correct mass ratio of NO3:N = 4.4268:1, but 663 were not. Of the 663, 20 were more than 25% different and of those, 12 were more than 50% different. In all 663 cases, the average of the two values was used and included as mg/L nitrate.

DWR Bulletin 130

Purpose – DWR Bulletin 130 reports are produced by DWR in cooperation with USGS to describe hydrologic conditions in surface and groundwater in California. These reports are produced for five regions of the state. Volume 3 covers the Central Coast, including our SV study area. Volume 4 covers the San Joaquin Valley.

Scope – This dataset included 859 samples from 256 wells in the SV, and 2,339 samples from 1,717 wells sampled in the TLB. These wells were sampled during 1963 to 1975 and in 1985.

Data – The data consist of the State Well Number, chemistry test results, the sample date, and in some cases the well use type. These data were digitized from PDF reports. These wells were located to the centroid of quarter-quarter sections, based on their SWN, in the TLB. In the SV, the quarter-quarter section map required to locate these wells is only available through Monterey County Water Resources Agency, which was unable to provide that map to our project. Therefore, SV wells in this dataset (and all other datasets with only SWN for location) are located to the centroid of the section identified in the SWN.

Adminsitrative Record
Page 34996

Burow et al. 1998

Purpose – Burow et al. (1998) compiled a set of wells data as part of a study on the occurrence of nitrate and pesticides in the eastern San Joaquin Valley.

Scope – This dataset included 53 samples from 30 wells sampled in 1986 or 1987 and 1995. Exact dates of samples were not available.

Data – The data consist of depths to the top and bottom of the perforated interval, the State Well Number, and the well use (all were household domestic wells). These data were digitized from a report and located by their SWN to the centroid of quarter-quarter sections where possible. Some wells were only possible to locate to the centroid of sections.

MCWRA Environmental Monitoring Wells

Purpose – MCWRA monitors 40 deep wells in Monterey County for monitoring of sea water intrusion and groundwater quality.

Scope – This dataset included 407 samples from 39 wells sampled from 1993 to 2009.

Data – These data were provided with GPS coordinates with accuracy assumed to be on the order of 10 meters. Top and bottom of perforated interval, SWN, sample dates, and chemical parameters were included in the dataset.

MCWRA 93/94 and 94/95 Water Year Reports

Purpose – MCWRA (1994, 1995) produced two water year reports including data on groundwater quality.

S*cope* – This dataset included 769 samples from 105 wells sampled in 1994 and 1995.

Data –These data were digitized from printed reports. They were located using their State Well Numbers to identify section centers. MCWRA did not provide actual locations, or reference maps to identify quarter-quarter sections.

MCWRA 2002

Purpose – MCWRA (2002) was a report on the shallow-well sampling program started in 1998. This program was designed to assess nitrate contamination in drinking water wells in SV.

Scope – This dataset included 243 samples from 76 wells sampled in 1999 and 2001. The report from which these data were acquired does not include precise dates.

Data – These data were digitized from a report and located by their SWN to the centroid of PLSS sections. MCWRA did not provide actual locations, or reference maps to identify quarter-quarter sections.

Adminsitrative Record
Page 34997

MCWRA has developed a nitrate well monitoring program with private well owners, that is subject to confidentiality agreements, and that has been operating since the early 1990s. Due to confidentiality agreements, MCWRA did not provide these data to our project, with the exception of the 40 environmental monitoring wells described above. Throughout most of California, the State Well Number can be used to locate a well to its quarter-quarter section (¼ mile by ¼ mile), but in the SV, the PLSS has not been digitized except by MCWRA and Monterey County. Neither MCWRA nor Monterey County has been able to provide UC Davis or the State Water Board with that digitized PLSS. Therefore, all wells in the SV study area that are located by SWN are located only to the section (one square mile).

Department of Pesticide Regulations (DPR)

Purpose - The Department Pesticide Regulations (DPR) maintains a voluntary domestic well sampling program to verify the efficacy of its pesticide regulatory program, particularly in areas that are DPR designated vulnerable groundwater zones.

Scope – The program includes 96 domestic wells of which 71 wells are located within the study area in eastern Fresno County and in Tulare County. Wells have been sampled annually during spring since 2001 (data are available through 2010) and also in the fall of 2001 and 2002.

Data – The data shared between DPR and UCD are subject to a confidentiality agreement. Accuracy of reported well locations is reported as the centroid of the section in which the well resides, roughly within one half mile of the actual location. The dataset includes depth to groundwater, well depth, DPR vulnerability zone, sampling date, and nitrate water quality data. Wells fall into two groups: those located in so-called "runoff" zones and those located in so-called "leaching" zones.[8]

County Data

Each of the five counties in our study area has provided additional well nitrate data that were included in CASTING. All of these data were collected for the purpose of maintaining regulatory compliance either with drinking water regulation or well permit regulation. With the exception of Kings County, all of the counties in the study area conduct one-time nitrate tests at the time each new well is drilled.

Tulare County

Tulare County provided us with assessor's parcel numbers (APN) of the domestic and local-small wells in the county that have been tested for nitrate. These wells were tested once, at the time they were drilled. The APNs for these domestic wells were used to select and locate those parcels from the county parcel GIS layer. Although APNs can change, it is expected that this will cause only a small number of inaccuracies. For geo-location reporting in CASTING, the centroid of each selected parcel was chosen as the approximate location of the well. The maximum spatial error for these wells is estimated as [½ x (Parcel Area)$^{1/2}$].

[8] http://www.cdpr.ca.gov/docs/emon/grndwtr/

Adminsitrative Record
Page 34998

Tulare County also provided several dozen test results for PWS wells that were not duplicates of CADWSAP records. In CASTING, these test results were assigned to their respective CADWSAP wells based on their PS-Code (discussed below).

Kings County

Kings County was able to provide us with documentation of well locations and test results for 20 wells, which we integrated into CASTING. Several of these were duplicates of CADWSAP wells, but included non-duplicate test results. Most of the wells provided by Kings County had only a few or a single nitrate test result. The non-duplicate wells were geo-located to the nearest second of latitude/longitude, (approximately 33 meters).

Fresno County

Fresno County provided us an opportunity to manually digitize well log and initial well water quality test results for household domestic wells. Original data are on paper records located in the County Health Department. The digitization effort covered 437 household domestic drinking water wells in our study area, with one nitrate sample each. These wells are located primarily by geo-coded addresses, with those that did not have satisfactory addresses located by the APN. Although APNs can change, it is expected that this will cause only a small number of inaccuracies. The wells are assumed to be in the centroid of the parcel with the corresponding APN. The precision of each APN-located well's location is calculated as one-half the square root of the area of the parcel. Of these wells, 377 had legible perforated interval top and bottom depth construction information, and 329 had legible estimated yield information.

Kern County

Kern County delegates all relevant data management to the Kern County Water Agency. KCWA has provided us with electronic data for over three thousand wells in the county. Most of these wells are associated with a single test for nitrate. While the bulk of these are irrigation wells, there is also a mix of household domestic, small water system supply, and monitoring wells in the dataset. The Kern County wells are geo-located by State Well Number, with a few exceptions having more precise locations based on manual re-location using aerial photography within a GIS.

Monterey County –scans of paper records

Monterey County provided us with scanned paper copies of well-driller reports and nitrate test results for approximately 1,000 local small and domestic wells. These wells have been sampled since the early 1990's although most records are much more recent. From these records, nearly 8,000 nitrate test results from 634 wells were manually digitized by UC Davis. Of the digitized wells from this effort, 24 were located by GPS coordinates, 350 were able to be geo-coded by street address, 17 were located to the center of PLSS sections, 223 were located to the center of cities, 10 were matches of wells already in the database, and 10 wells fell outside the study area. Although State Well Numbers were available for many of these wells, the lack of any available reference dataset of PLSS quarter-quarter section maps from SV meant that wells were not geo-located as accurately as the SWN would allow. The driller logs include the depth to screen and screen length in almost all cases.

Adminsitrative Record
Page 34999

Monterey County – electronic spreadsheets

Monterey County also provided a dataset of well locations and a separate set of nitrate tests for state small and local small water systems and domestic wells. The well location set consisted of 431 unique geo-referenced points, and another 498 wells with addresses associated. We were able to geo-locate 389 of these addressed wells, for a total of 820 unique geo-referenced well locations. The nitrate test results are reported by system name or number and not by a specific well identification. Therefore, for systems with multiple wells, tests could not be associated with specific wells or well locations. For this reason, multi-well systems were ignored. The remaining 680 systems have unique locations (a single well) to which nitrate test results can be matched. Numerous inconsistencies were found in the system naming and ID codes, indicating that the locations and test results associated with certain locations are of uncertain reliability. This dataset is known to overlap the digitized dataset described above; however, no common identification code exists between the two datasets. It is unknown at this time how many of these wells overlap with the previous dataset, which was manually digitized from the scans provided by Monterey County.

Adminsitrative Record
Page 35000

Table 13. Tests and wells by data-source within the study area boundaries. Some data-sources provided tests that were duplicates of tests already in the CASTING database, resulting in cases such as the NWIS data-source with more wells than tests; duplicate test results were filtered out of the final database. The total number of wells in the study area in the CASTING version of December 17, 2011, is 20,286.

Source Dataset Code in CASTING	Number of Test Results	Number of Wells	Brief Description of Data-source
CADWSAP	60,792	3616	Subset of PICME dataset with high-quality locations and SWAP implemented
DAIRY	11,271	6459	Regional Board 5 dataset of wells on dairies
MOCODig	5,666	452	Digitized by UC Davis staff: records from MCWRA
EPA	4,871	2883	Legacy STORET data
KeCo	3,412	3007	Supplied by Kern County Water Agency
EnvMon	2,598	537	SWRCB Environmental Monitoring wells
DWRL	2,015	1978	Digitized by UC Davis staff from DWR Bulletin 130 reports
NWIS	1,684	1735	USGS database of tested wells
MoCo	1,572	462	Supplied by Monterey County Environmental Health
MCEM	1,018	229	Digitized by UC Davis staff from water quality reports in Monterey County
DPR	814	71	DPR domestic wells survey
TCEHS	444	444	Domestic wells data supplied by Tulare County Environmental Health
FRCO	368	369	Domestic wells data supplied by Fresno County Environmental Health
GAMA	141	141	SWRCB GAMA Domestic Wells Study
USGS	83	83	USGS/SWRCB GAMA Priority Basin study
KiCo	82	20	Supplied by Kings County Environmental Health
CIP	63	63	Canal Integration Project (Westlands Water District and DWR)
DWR	44	39	Data supplied by DWR from tests in western San Joaquin Valley in 2008
KRB98	36	19	Digitized from Burow, 1998
TuCo	17	579	Supplied by Tulare County GIS staff
DIP	14	14	Distribution Integration Project (Westlands Water District and DWR)

The 21 datasources incorporated into the CASTING database (Table 13) comprise the most comprehensive dataset of well nitrate test results in existence for the TLB and SV. CASTING is intended to be a prototype of a state-wide database for nitrate in groundwater. The varying levels of confidentiality of the datasets incorporated into CASTING require that the database be managed carefully for privacy concerns. All portions of the database that may legally be made public will be as soon as technically feasible.

4.1.4 Well Identification Codes

There are several different identification schemes used throughout the state (Table 14). The State Well Number (SWN) is theoretically applied to every drilled well and is filed with DWR in the well-driller log

Adminsitrative Record
Page 35001

for each well. The SWN is based on the Public Land Survey System (PLSS), which defines the meridian, township, range, section, and quarter-quarter section that contains each well, and includes a serial number to distinguish each well from other wells in the same quarter-quarter section. As such, it is possible to (approximately) locate a well by its SWN to within an eighth of a mile distance. For many private well owners, such accuracy is cause for privacy concerns. Lack of understanding of the cadastral survey has further led to some problems with the SWN. For example, California has three meridians upon which the township, range, and section numbers can be based (Humboldt, Mount Diablo, and San Bernardino meridians). When this meridian is not included in the SWN, it can be difficult to identify in which region (of the 3) the well belongs, particularly in the TLB. The SWN is the only ID code that is meant to be universal within California. Unfortunately, many agencies do not record the SWN of wells that are sampled in their projects.

The database of drill logs that DWR maintains is the only database that is intended to be comprehensive of all wells in the state, but this database is only for construction information such as depth, screen interval, estimated yield, and drilling date. The database is currently maintained in the form of paper files and scanned images of such files. It is not organized in a manner that allows for convenient access to large numbers of well data files. Although some well records in the database have a SWN recorded on the paper file, the electronic files are identified by a serial number that has no direct relationship to the SWN or any other identification code used by any agency. DWR does not maintain a relational database of this serial number as it relates to the SWN.

The only other commonly used, statewide-applicable code is the so-called State ID (SID). This code consists of the 13 digit latitude/longitude in degrees, minutes, and seconds for the well. This code is far less common than the SWN, and has higher precision, on the order of 30 m (100 ft), (though also more likelihood of error) than the SWN. It also produces confidentiality concerns. Other identification codes are used by specific agencies, projects, or regional organizations, and are generally not cross-referenced to each other, the SWN, or the SID.

Table 14. Well identification codes and agency users.

Code Name	Abbreviation Alternate Name	Example	Agencies/Dataset that use it
PS-Code	PICME Pkey	5400544002	CDPH, Counties
System Number	PWSID	5400544	CDPH, Counties
State Well Number	SWN	011S021E18F001M	SWRCB, DWR, USGS, EPA
GAMA ID		TULE-O7	SWRCB, USGS
State ID	SID	365803119432101	SWRCB, DWR, USGS, EPA
Electronic Data Format Name	EDF	W-5C	SWRCB, DWR, USGS

CDPH and many of the counties use a code called a Primary Source Code, or PS-Code. This code consists of a 7-digit serial number assigned to each water system that provides drinking water to at least two households, followed by a 3-digit serial number that designates the particular source (well, spring, stream, tap, tank, or other point at which water is tested).

Adminsitrative Record
Page 35002

The GAMA Domestic Well Project uses well IDs based on the county in which the well is located, followed by a serial number. These codes are unique, but generally not related to the SWN or other identification codes. As a result, although they are convenient for distinguishing between GAMA domestic wells, it is difficult if not impossible to compare distinguish these wells from other wells that have been sampled in the area. The GAMA Priority Basins project uses codes that appear to have been assigned irregularly, with some codes consisting of the county name followed by a serial number, and some others based on groundwater basins or other geographic features. These codes are based on the study area defined in the Priority Basins Studies. Many of these wells are also identified by their SID, thus allowing some cross-referencing with other well sampling projects.

Environmental monitoring wells with information maintained by the State Water Board are assigned names within each separate project by the sampling contractor, without consideration of wells at other monitoring sites that have already been assigned names. Some of these wells have the SWN as the name, but most use a code such as "MW-1" or similar. There are many examples of duplicated well names within that database. To avoid confusion, a project identification code is assigned by the State Board to distinguish between projects. Within a single project, well names are not duplicated. The well name and project ID together provide a unique identification to differentiate between wells with identical well names. A latitude-longitude based identification of reasonably high accuracy is also available for these wells. However, this information is not used as a site ID, because of the possibility of confusion with other, less-accurately defined, coordinate-based ID schema.

4.1.5 Potential Future Improvements to a Nitrate Database

The database represents what has been possible to collect over the 18 months of this project. There are potential future improvements to this database that were outside the scope of this project:

- Tulare County has additional domestic well nitrate data in digital format that we were unable to obtain. It is unclear how many records are not covered in CASTING.
- GPS locations of Tulare County domestic wells that are currently not in CASTING may be available from Tulare County.
- The dairy regulatory program currently does not maintain an electronic database of groundwater nitrate data obtained annually from 1,800 producers. Data are currently entered from paper records into a spreadsheet. However, important information is not collected consistently:
    - Well identification codes may change from year to year.
    - Well locations and well construction information (screen depth) are not known. This is a task that could be done by a single entity for all program participants at once, by working with DWR well log archives.
    - Records are not kept in a suitable database format; significant data manipulations and simulation are needed to bring data records into a format amenable for database archiving and (statistical) analysis.

Adminsitrative Record
Page 35003

    - We recommend that the CVRWB work with DWR and State Water Board to set up an internal work process to enter data provided by dairy producers into a high quality database and to follow up with producers on data gaps and data inconsistencies. In principle, this system could also be setup whereby certified consultants enter data collected under the dairy regulatory program directly into an electronic database, including groundwater data.
    - At a minimum, groundwater nitrate data should be associated with:
        - Well identification number
        - Well location
        - Well construction information
        - Sampling date.
- MCWRA has a confidential, private network of wells monitored for nitrate. Data are not included in CASTING, unless they have been published (typically data for which the collection was funded by public grants).
- Fresno County and presumably other counties have a significant number of paper records with water quality data from domestic and other wells that we have not been able to digitize as part of this project.
- Many wells in the CASTING dataset have no information or incomplete information on both well use and well construction. A major effort is required to match these wells with DWR well log information. While this information may not be unambiguously available for all wells, a substantially larger number of wells would have well depth and more precise location information than currently in CASTING.

Adminsitrative Record
Page 35004

4.2 Methods

4.2.1 CASTING Database Structure

The CASTING database of wells and test results consists of two large tables. The first table consists of a single record (row) for each well, with attributes (columns) for location information, any identifying codes of names, and any available physical characteristics of the well. This database table is called NO3GW and consists of geo-located datapoints (wells). Each well in this table is assigned an internal unique new ID called "NO3_ID."

The second database table is a tabular set of test results and is designated NO3GT. Each record refers to a single nitrate sample test. The record consists of attributes identifying names and codes, the sample date, the sample result, the agency from which the data were acquired, and the NO3_ID for the well.

The NO3GT dataset can be queried or related to the NO3GW dataset via the attribute NO3_ID. Many organizations maintain databases of NO3 data and of well information, often separately. In many cases, different agencies record separate samples for the same wells. To reduce duplication of data within CASTING that would cause incorrect results from statistical and spatial analyses, new data from outside sources must be compared with wells already in CASTING by the various agency ID codes, the well name, well location, and well test date and results.

4.2.2 Developing the NO3GW and NO3GT Datasets for CASTING

The CADWSAP database was used as the starting point for the development of the CASTING wells database. The dataset is much larger than most of the other well databases used; it is also the most reliable source of water quality testing data and it has the broadest set of recorded parameters. Each well in CADWSAP has a unique ID (PS-Code) from PICME. Any one well may have between one and several hundred test results. It is therefore not practical to store the test results in the same dataset as the well information. PICME, the parent dataset for CADWSAP, uses a database called WQM for storing the water quality information. Each test result has the PS-Code for its associated well, and this code is used to query the WQM database for results associated with wells of interest.

We extracted all raw or untreated water sample data from CADWSAP wells located within our study area. CADWSAP also includes records for all treated or blended sources of drinking water, and for non-well (surface water) sources of drinking water, but we did not include these sources in CASTING. Each well added from the CADWSAP database was then assigned a unique code that is called the "NO3ID" code. This code is simply a serial number preceded by the prefix "NO3_" and is only used for internal record-keeping. NO3ID has no spatial reference incorporated, nor any other identifying characteristics that could trigger confidentiality concerns. The resulting wells table (spreadsheet) is called NO3GW (for nitrate in groundwater - wells) within CASTING.

Adminsitrative Record
Page 35005

We used the PS-Codes of these wells to extract the WQM test results for all those wells. The NO3IDs we assigned to each well in the study area were then attached to the test results. The resulting test results database (a spreadsheet or electronic table) is called NO3GT (for nitrate in groundwater - tests) and is the second of two databases in CASTING.

Subsequently, each new well or set of wells obtained from an outside data source was checked against the existing database to find matches, either by codes or well owner name. As new wells were acquired, the ID codes and names associated with those wells were added into the database in order to maintain a record of the source. Additionally, a separate attribute field (column) is created for each data source in both tables (NO3GW and NO3GT), named generally by the data source. All wells from that data source were assigned a single-digit integer value in that attribute. The integer value is associated with the method of geo-location and is described in the metadata of the CASTING database. For example, all wells in the EPA STORET dataset are located by their SWN, so they all receive a "1" in the "EPA" attribute. Similarly, the wells in the Tulare County Domestic Wells dataset provided by Tulare County Environmental Health are located through one of 3 different methods (by Parcel Number, by address, or by GPS coordinates), so each well gets either a"1", "2", or "3" in the "TCEHS" attribute.

Each well might be present in several datasets, but each test record only comes from one dataset. Therefore, in NO3GW it is possible to find many records with values in multiple dataset columns, but in NO3GT each record has only one of these columns filled in. For example, a well that was added from CADWSAP would receive a non-zero value in the CADWSAP column of NO3GW, if it were then found to also exist in the STORET database, it would receive a non-zero value in the STORET column as well, but the new test result would be on a new row in NO3GT. Thus, overlap between data sources can be tracked (in NO3GW), while the source of each test result is retained (in NO3GT).

The process of integrating new data into the NO3GW database is shown schematically in Figure 22. This process adds the well and its associated parameters such as ID codes, data source, depths, screen intervals, or other physical or ownership characteristics to the spatial dataset (NO3GW) of wells in the study area. Once a well was identified and assigned a NO3ID, the test results associated with that well are attached to the NO3GT database with the associated NO3ID.

Adminsitrative Record
Page 35006

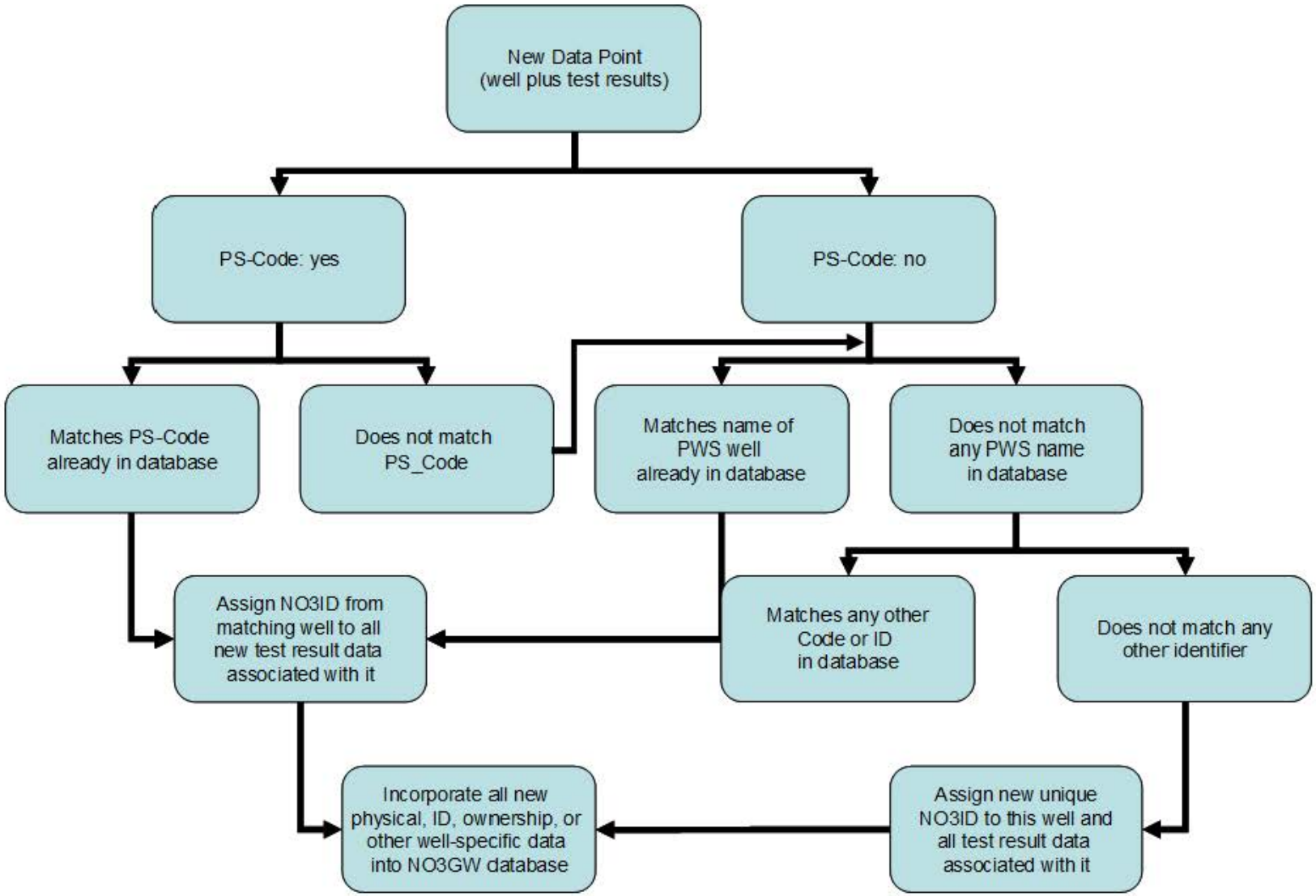

Figure 22. Process diagram for adding new wells to CASTING database.

Adminsitrative Record
Page 35007

4.2.3 Adding Data to CASTING

The input databases were developed by different agencies at different times, and thus very few of these databases were in similar formats. Each database was examined for formatting, unit, and precision prior to assimilation into CASTING. The locations of wells were provided in some cases, and these locations were assessed for horizontal precision. For datasets without location data, locations were generally acquired through address geocoding, or based on the SWN.

4.2.3.1 Non-Detect Data

For nitrate test results, all values were converted to nitrate as NO_3. This value is contained in the "FINDING" attribute (column) of the NO3GT database of CASTING. Tests for which "non-detect" (ND), "0" or "less than x" (where x is a detection limit) was reported were assigned the detection limit in the "FINDING" attribute and "<" in the "x_mod" attribute of the NO3GT database. Statistical analyses using these assigned values generally follow the method described in Helsel (2005) of using ½ the detection limit in cases where a non-detect is reported. In cases where the detection limit was not reported by the data provider, it was assumed to be 2 mg/L as NO_3.

4.2.3.2 Precision of Well Locations

In the NO3GW database of CASTING, the attribute "x_y_prcsn_m" gives the likely maximum error in the Northing and Easting location of the well, in meters (m). This value was developed based on the method of location used for each well. When the location of the well was provided by the input database, but the method of location was unclear, this attribute is generally left blank.

For wells located by Assessor's Parcel Number (APN), the SWN, (either to the quarter-quarter (QQ) section or the full section), the value of the precision attribute is equal to ½ the square root of the area of that parcel, QQ section, or section, in meters. When wells were located by their SWN in this manner, it is unavoidable that all wells identified in the same QQ section (or section) will appear in a single location at the centroid of the QQ section (or section). The precision of well locations derived from APNs is variable depending on the size of the parcel. Also, the APN can change when properties change ownership or are subdivided or split, and this will introduce unknown positional error. The precisions of wells located to the QQ section and section are approximately 200 m and 800 m, respectively.

In the SV, no section survey has been conducted, but a virtual PLSS dataset has been built by the Bureau of Land Management (BLM) to the section level. It is our understanding that a similar, though likely not identical, virtual PLSS has been developed by the Monterey County Water Resource Agency to the quarter-quarter section level. This product of the MCWRA was used to assign SWNs to wells that agency tested, and these SWNs were available in public documents recording the sampling data; however, the virtual quarter-quarter section dataset was not provided. Therefore, wells located by SWN in the SV are only located to the section centroid, based on BLM's virtual PLSS, and may have ½ mile or more of Northing and/or Easting error. Wells in several datasets are located by GPS coordinates and these are

Adminsitrative Record
Page 35008

assumed to be within 20 m of the correct location in most cases. Some wells are located through geocoded addresses, and the precision of these wells is assumed to be 50 m.

4.2.4 Attributing Wells with Depth Information

Some data sources provide well depth, some provide screen depth and/or screen interval length, and some provide no or inconsistent depth data. For the purpose of this study, screen top and bottom depths are acquired wherever possible.

Where well screen top and bottom depths are provided by the originator, we have incorporated those values into the database. When two data sources provide conflicting information about depth for a particular well, priority is given based on the hierarchy shown in Table 15.

Table 15. Priority of screen top data sources.

Priority	Top of Screened Interval Data Source
1	Driller Log
2	USGS data
3	CADWSAP or other agency-sourced data
4	Seal depth from agency source, taken as top of screened interval
5	CADWSAP depth to static water, taken as top of screened interval
6	Regression vs well depth from agency source
7	Regression vs owner reported well depth
8	Unassigned
9	No depth information provided

Where 2 or more agencies reported different depths, we checked for a driller log source and used that if possible. If no driller log source was available or identifiable, we looked for other data points that might invalidate any of the agency-reported depths. If we could not invalidate any data, we choose the deeper value (this only occurred in a very few cases). All values were retained in the NO3GW dataset in case later information needs arise that require us to retrieve these values.

In many cases, the information obtained included well depth, but not the depth to the top or bottom of the perforated interval. A regression was built to model these depths based on well depth using the Priority Basin Project wells. These data include more reliable and comprehensive construction information for the wells than that contained in the CDPH database, and are the only large dataset for which we have the depth to the top and bottom of perforated interval as well as the total well depth.

The Priority Basin Project wells were identified by either a GAMA ID or SID (Table 14). The State Water Board provided a cross-walk between these codes, the SWN and the CDPH PS-Code. We used the GAMA ID, STATEID, and SWN to join these wells with data extracted from the Data Summary Reports (DSR) published on the State Water Board GAMA website for the USGS Priority Basin Project. Based on the information thus joined, we derived an empirical relationship between completed well depth and

Adminsitrative Record
Page 35009

depth to top of perforated interval (Figure 23). The depth to the bottom of the perforated interval is nearly always the same as the depth of the completed well. The relationship between depth to top of perforations and well depth is nearly linear, but becomes less certain with increasing depth.

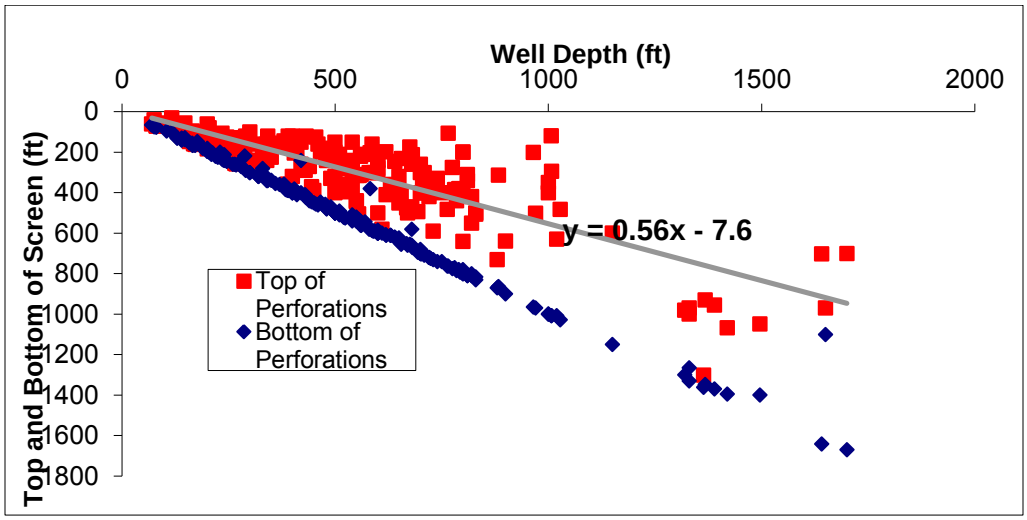

Figure 23. Regression of top of perforated interval from well depth. Based on the USGS GAMA Priority Basins Project dataset.

The linear regression of depth to top of perforation from well depth is:

Top of Perforations = [well depth (0.56) – 7.6] (Eqn. 1)

This is based on 185 wells that had top and bottom of perforation depths as well as well depth. These data were contained in the DSR datasets produced by the USGS Priority Basins project.

Of the 4,686 wells with Depth to Top of Perforation in the current version of CASTING, 2,144 are derived from Eqn. 1. The majority of these were wells with only well depth provided, based on Agency reporting.

Each method, or source of information about the depths to top and bottom of perforated interval, was retained in the CASTING database. This allows users to select data for analysis based on the level of confidence in the depth data.

4.2.5 Mapping of Nitrate in Groundwater

ESRI ArcGIS™ 10 was used to develop the CASTING database and to create maps showing the distribution of nitrate in wells in the study area, at different time periods, regional scales, and depths. Nitrate concentration is mapped in relation to rivers, the trough of the TLB, population, and overlying land use characteristics. We used this GIS to produce tables relating the concentration of nitrate to these other variables for statistical analysis. The California Teale Albers NAD83 projection was used for all datasets.

Adminsitrative Record
Page 35010

Gradients developed as described above were joined back to the spatially referenced wells and mapped in ArcGIS™. Nitrate concentrations and gradients of nitrate concentrations were compared across depths and regions. Wells were distinguished by several attributes: depth to the top of the screen, groundwater subbasin, nearness to major natural streams, and well-type.

Adminsitrative Record
Page 35011

4.3 Spatial Distribution of Wells Data in CASTING

4.3.1 Spatial Distribution of All Wells in the Database

In the December 17, 2011 version, CASTING consists of 25,214 wells, with 2,109 in the SV area and 18,177 wells in the TLB area. The remaining 4,928 wells are outside the study area, but are retained to provide boundary conditions for modeling purposes such as interpolations of groundwater nitrate and well density.

Well locations are unevenly distributed across the study area (Figure 24). The wells are of relatively low density in the southwestern region of the TLB and in the upper SV. The Tulare Lake bed region in Kings County is effectively devoid of any records of well nitrate, which coincides with a privately-owned, surface water irrigated farming area (Boswell Farms) that has almost no private residences. The highest density of wells in the database is found in the metropolitan areas of Fresno, Hanford, Visalia, Porterville, Delano, and Bakersfield. A high density of wells with nitrate data also exists in areas west and southeast of Bakersfield in the valley portion of Kern County (Figure 24).

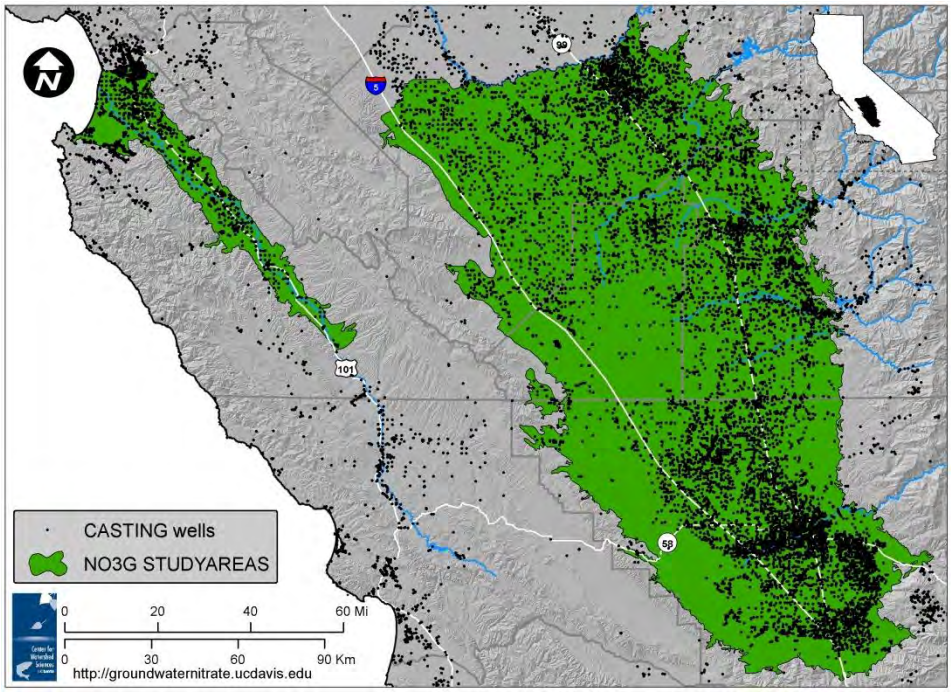

Figure 24. All wells in the database as of December 17, 2011.

4.3.2 Spatial Coverage by Data Source

The CADWSAP database (5,429 wells) had the widest coverage of the various source databases. Even so, CADWSAP lacked good coverage in some regions of the study area (Figure 25). The U.S. EPA STORET database (3,077 wells) is well-distributed in the TLB, but lacks any coverage in the SV portion of the

Adminsitrative Record
Page 35012

study area (Figure 26). All other datasets were restricted to either very low density coverage (the State Water Board Environmental Monitoring wells, USGS Priority Basins wells – Figure 27) or were very regional (county datasets – Figure 28; DPR, DWR, GAMA Domestic – Figure 29). The Central Valley Regional Board provided 6,459 wells in the study area, most in the eastern portion of the TLB (Figure 30). The following maps show these distributions.

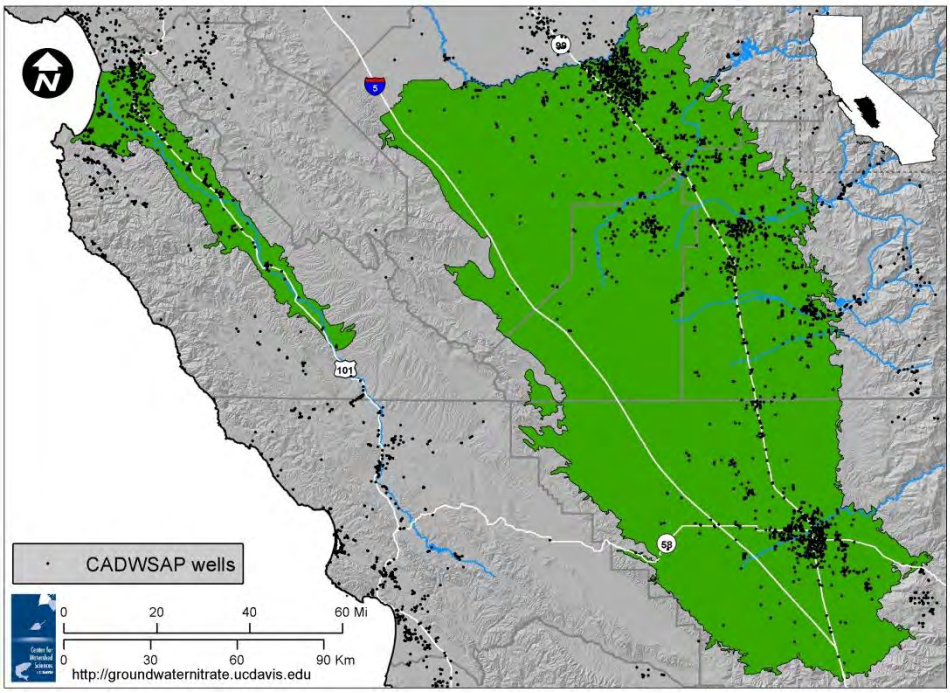

Figure 25. CADWSAP wells.

Adminsitrative Record
Page 35013

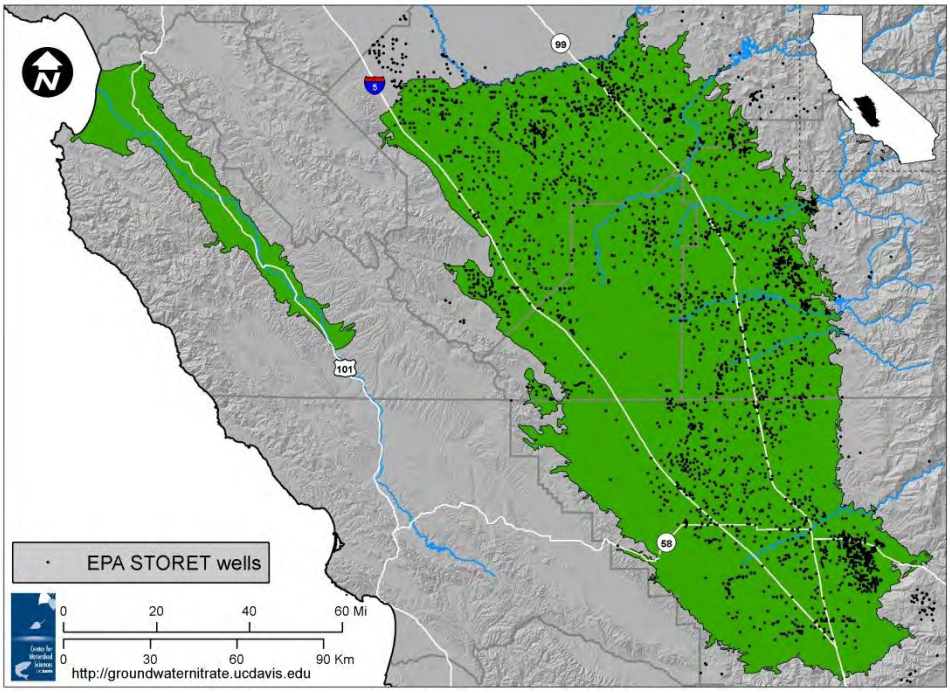

Figure 26. U.S. EPA STORET wells.

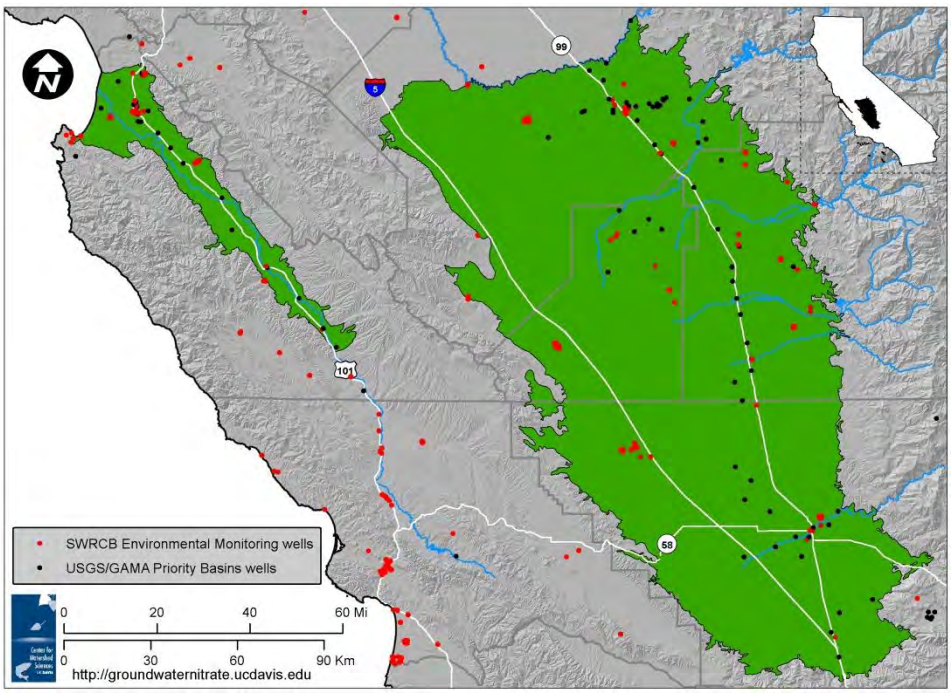

Figure 27. The State Water Board Environmental Monitoring wells and USGS Priority Basins wells.

Adminsitrative Record
Page 35014

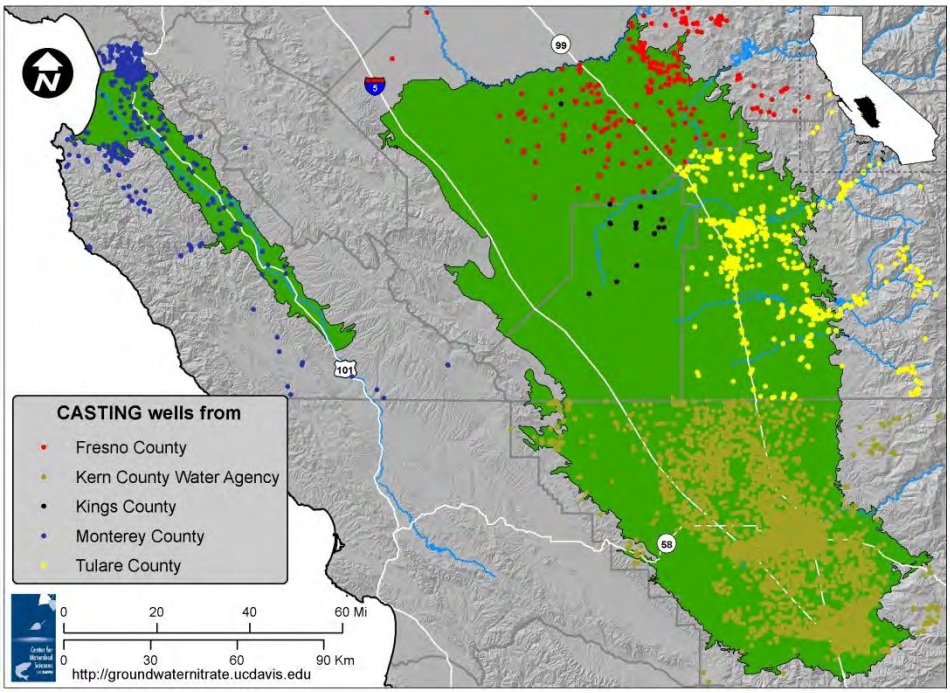

Figure 28. County-sourced wells.

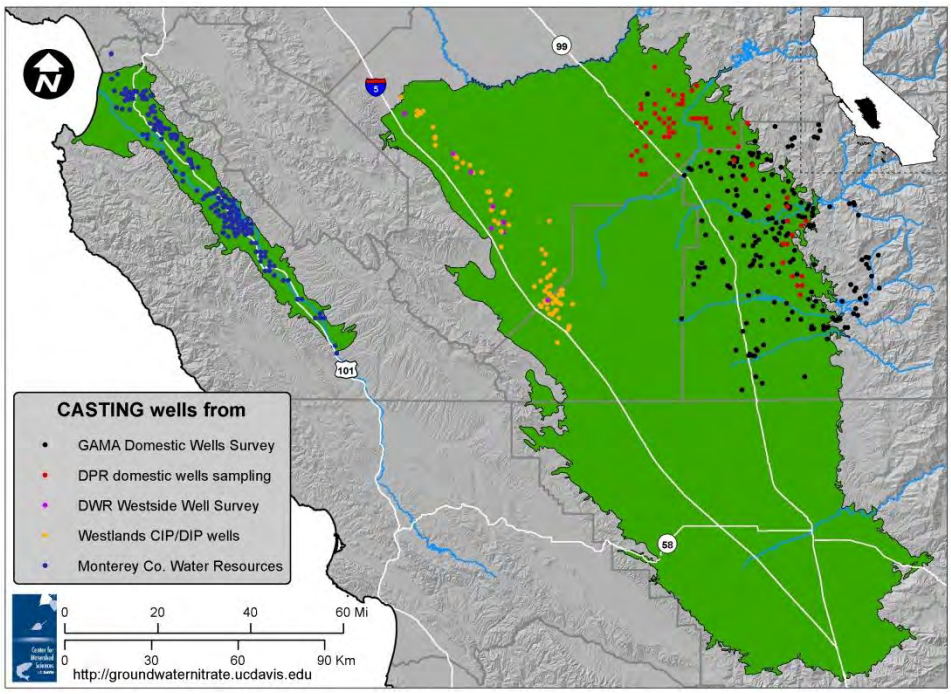

Figure 29. Regional Projects.

Adminsitrative Record
Page 35015

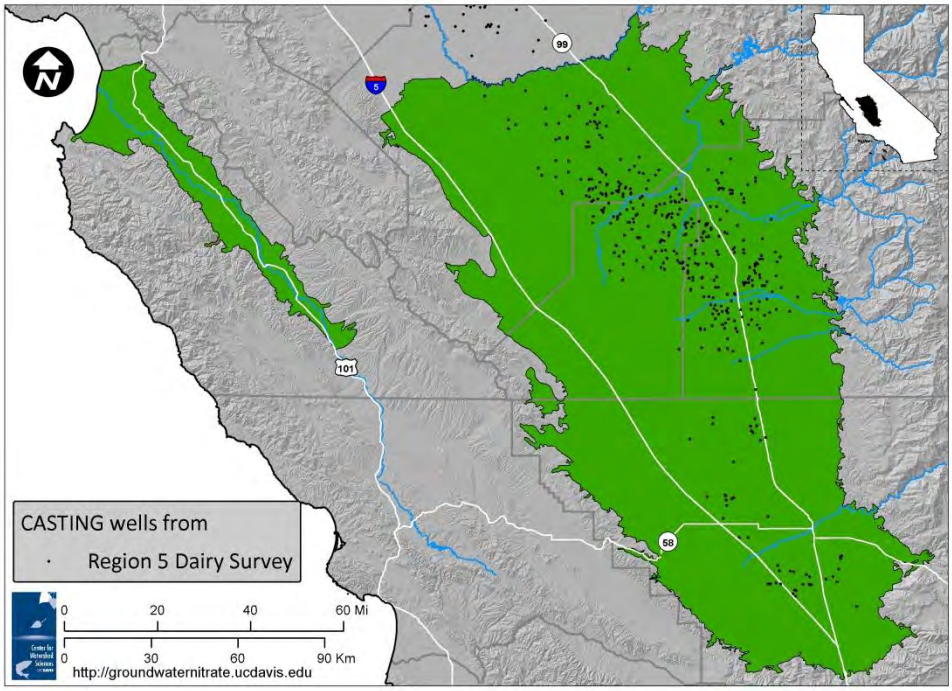

Figure 30. Dairy wells, data from the Central Valley Regional Water Board.

The wells in the CASTING database are distributed similarly to the population of the study area (Figure 31). Wells for which data has been collected tend to be clustered in areas with higher population density and along major roads.

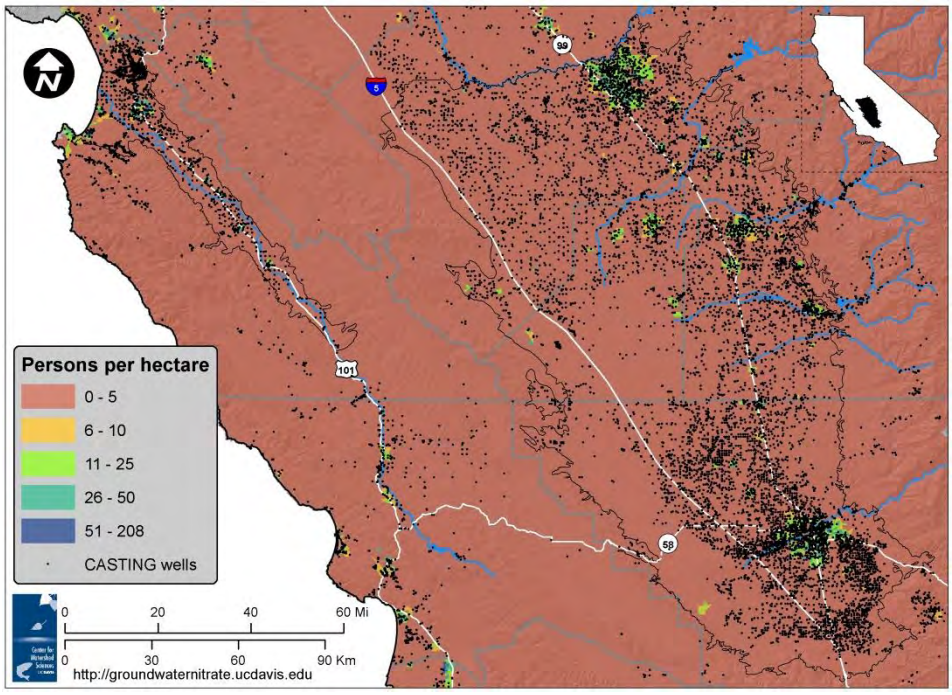

Figure 31. Population density and CASTING well distribution.

Adminsitrative Record
Page 35016

4.3.3 Depth of Wells

We were able to obtain or estimate depth to top of perforations for 4,686 of the wells in the database within the study area. Depths to the top of the perforated interval were found to average approximately 79 meters (259 feet) in our dataset, with the shallowest well at 1.8 meters (6 feet) depth, and the deepest well at 741 meters (2,431) feet below ground surface. The standard deviation from the mean of well depth to top of perforation was 72.2 meters (237 feet) for the entire set of wells with depths. The spatial distribution of the wells with depth information was roughly similar to that of the overall database (Figure 32), but with lower density throughout the study area. Depth classes were defined as "Shallow" (0 – 90 m or 0-300 ft below ground surface), "Medium" (90 – 180 m or 300-600 ft), and "Deep" (> 180 m or > 600 ft).

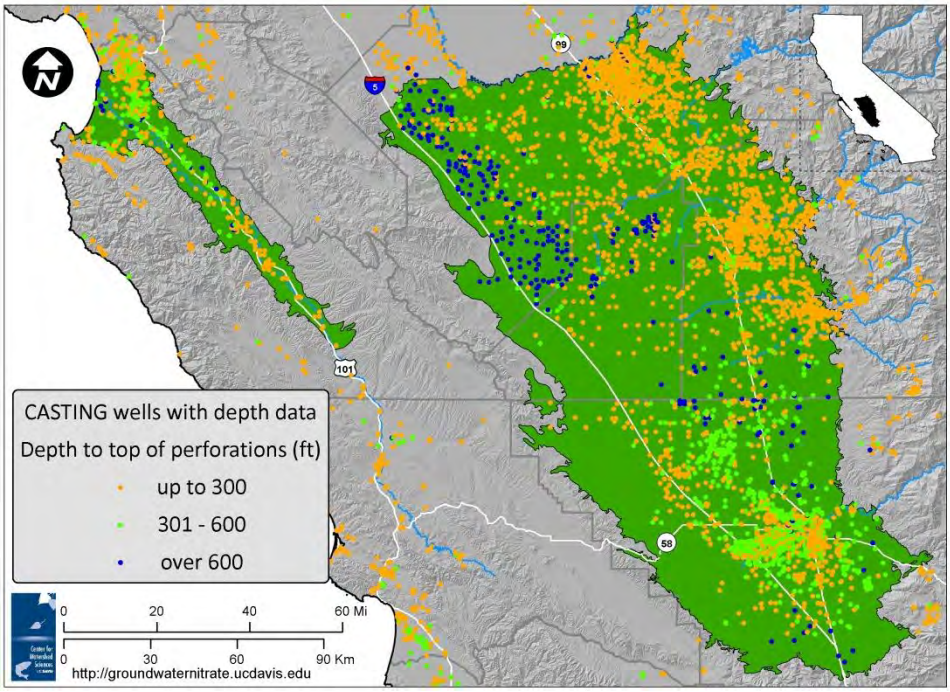

Figure 32. Wells with depth to top of perforation.

For all wells with depths, the distribution of depths is heavily skewed to the higher end (Figure 33). For domestic household wells only (data from Fresno County, GAMA Domestic Wells Survey, CVRWB Dairy Program, and DPR Domestic Wells Survey, Figure 34) this skew is less pronounced. This set of wells consists of wells that are known to be household wells; the mean, max, and standard deviation of domestic well depths to top of perforated interval were 258 ft, 1832 ft, and 97 ft, respectively.

Adminsitrative Record
Page 35017

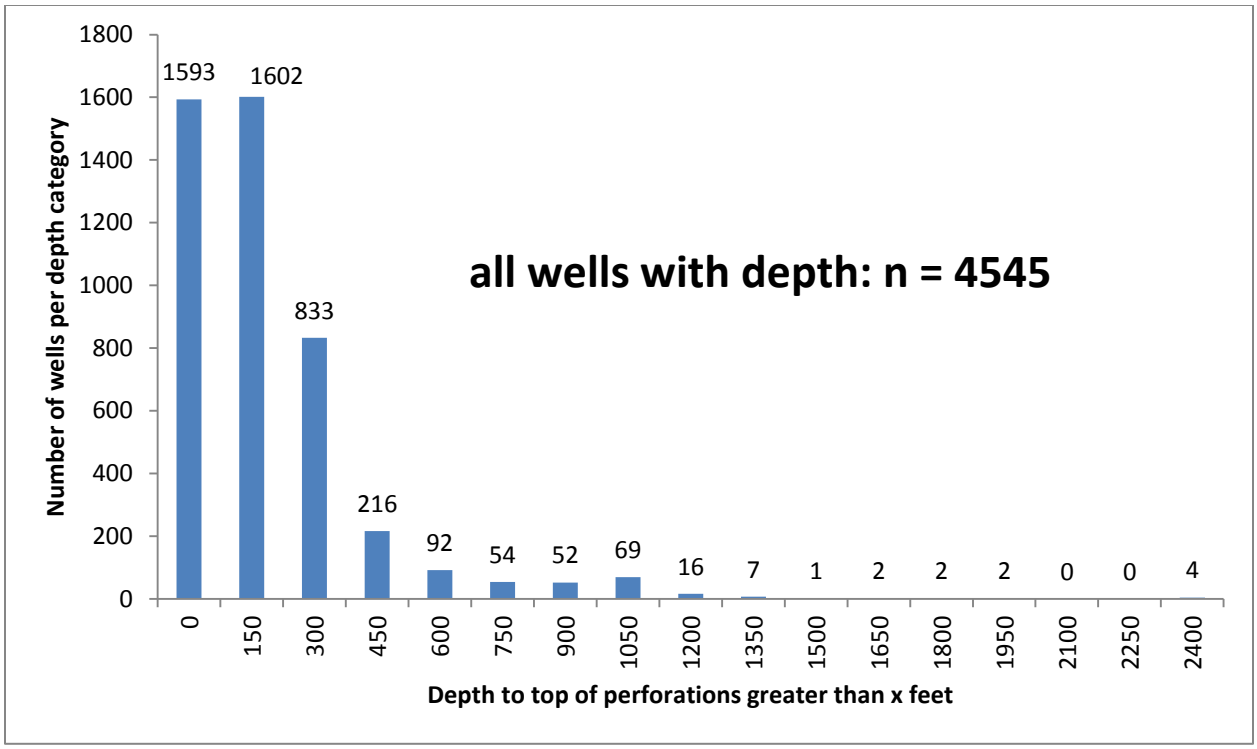

Figure 33. Distribution of depth to top of perforated interval for all wells with available data.

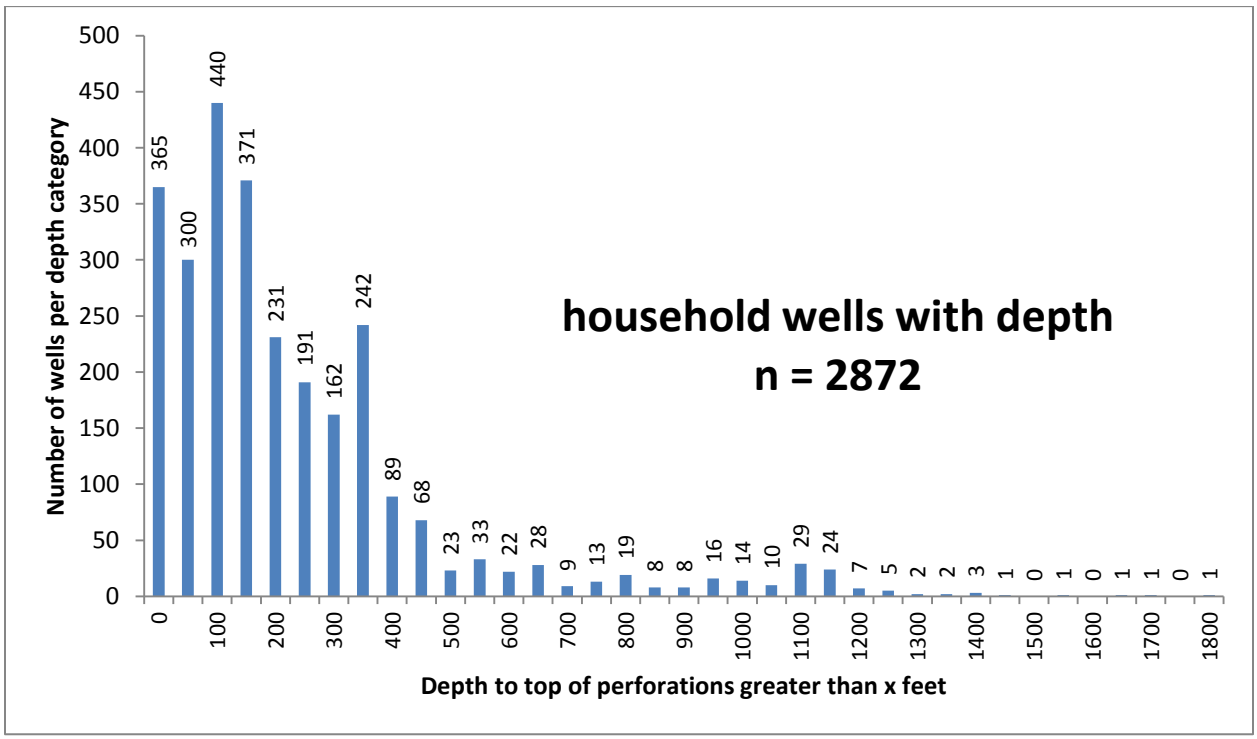

Figure 34. Distribution of depth to top of perforated interval for household domestic drinking water wells with available data.

Adminsitrative Record
Page 35018

Wells known to be non-household wells averaged 79.6 meters (261 feet) to top of perforations, with a standard deviation of 191 feet. Most of the household wells in CASTING are located in Tulare, Fresno and Monterey Counties (Figure 35), whereas other types of drinking water wells are more evenly distributed throughout the study area (Figure 25). This spatial bias may reduce the value of this direct comparison between the household and non-household wells.

Between groundwater subbasins, distinct differences in depth distribution arise, making a study-wide comparison of domestic well depths with public supply well depths difficult. Therefore, DWR Bulletin 118 groundwater subbasins were used to group wells for a spatially distributed comparison between household wells and water system wells. The number of wells of each type (household and water system), and their average and standard deviation of depth to top of perforations, paired by subbasin, are shown in Figure 36.

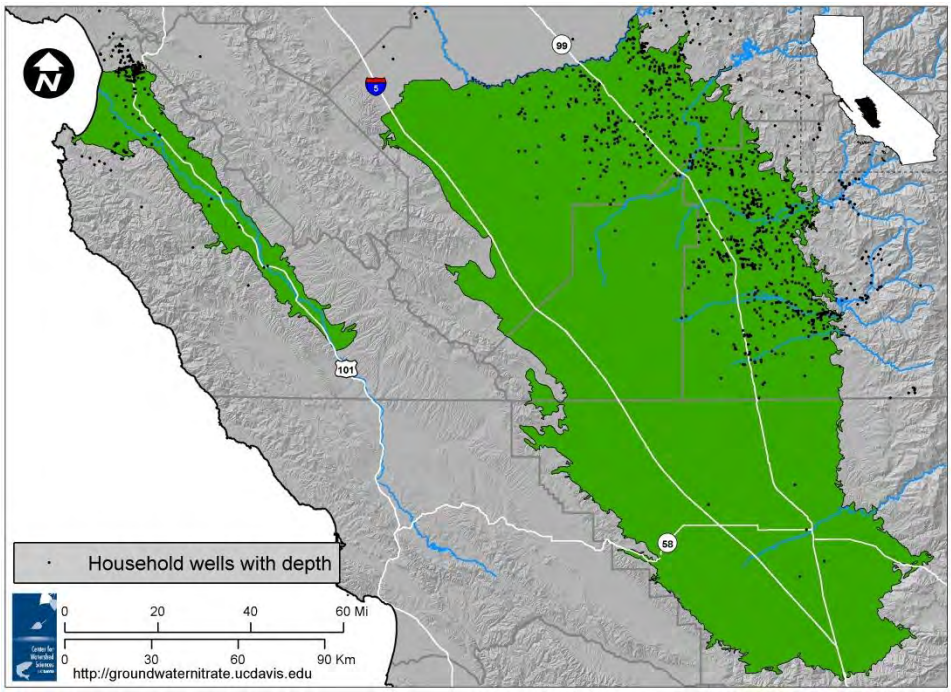

Figure 35. Household domestic wells in CASTING with depth data available.

Adminsitrative Record
Page 35019

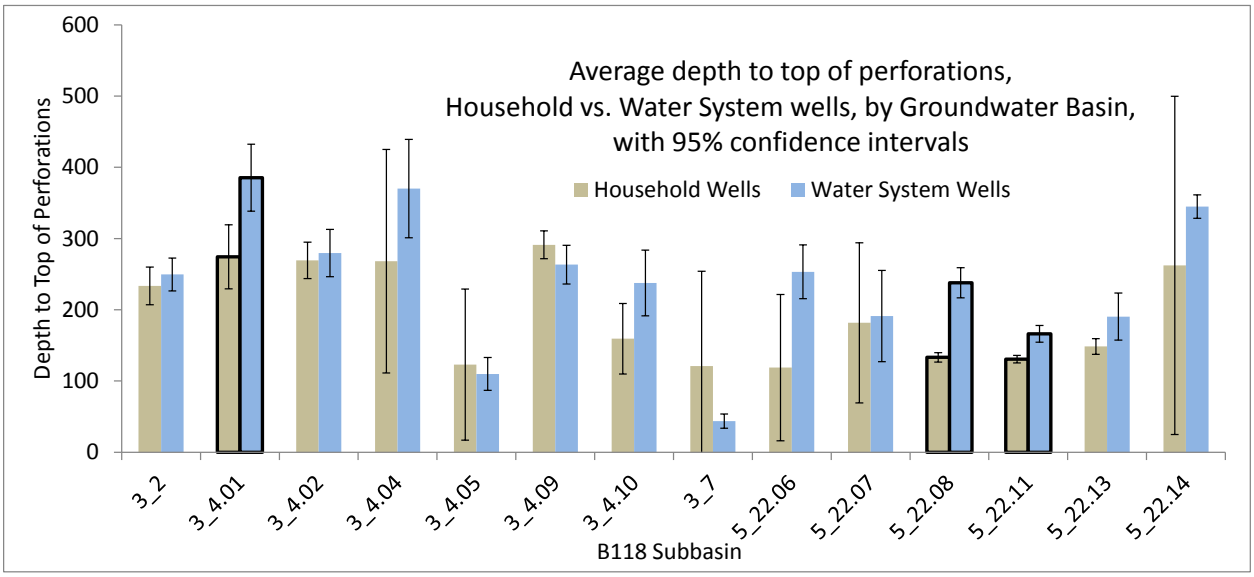

Figure 36. Comparison of mean depth to top of perforations between household wells and water system wells, by groundwater subbasin. Bold outlines indicate significantly (α=0.05) different values.

The three subbasins that produced significantly different values of mean depth to top of perforations all had deeper drinking water system wells than their household wells. However, several subbasins show household wells of essentially the same depth as drinking water system wells. This implies that in certain areas, the assumption that community water system wells are deeper than household wells does not hold.

Adminsitrative Record
Page 35020

4.4 Test Origins and Temporal Distribution

We have acquired records for 133,329 test results associated with the wells in and near the study area. Of these tests, 97,005 are on wells in the study area, with 16,663 in the SV and 80,342 in the TLB. We did not differentiate between tests taken by different projects – all tests are treated as equal value. Any test that was performed on any particular well is associated with that well, regardless of the source of the test data. The bulk of the tests in the test database are from the CDPH WQM database. Where test results from other sources were available for CADWSAP wells, these tests are simply included in the tally of tests for those wells. The database of well nitrate samples consists of test results spanning the years from 1900 to 2010 (Figure 37).

Well information is distributed unevenly in time (Table 16). Regular monitoring began in the 1950s with a few hundred records available per year in the study area through the 1980s (a total of approximately 13,000 nitrate test results prior to 1990). During the 1990s, nitrate sampling records collected each year rapidly increased (approximately 15,000 nitrate data are included in CASTING for the 1990s). Over 70 percent of the data in CASTING – approximately 68,000 nitrate records - were collected since the year 2000. The data from 2010 is incomplete.

Table 16. Number of nitrate tests in CASTING, within the study area, by county (study area) and time period. Total for both study area is 97,005 test results.

	Fresno	Kern	Kings	Tulare		(TLB)		Monterey (SV)
1900-1949	30	38	4	44		116		4
1950's	797	2,645	142	521		4,105		0
1960's	872	1,462	205	655		3,194		245
1970's	712	1,840	201	826		3,579		194
1980's	1,059	937	202	671		2,869		933
1990's	3,500	4,045	180	3,080		10,805		4,883
2000's	27,616	9,180	3,593	16,080		56,469		9,563
2010's	1,295	340	50	547		2,232		887
all years	35,881	20,487	4,577	22,424		83,369		16,709

Adminsitrative Record
Page 35021

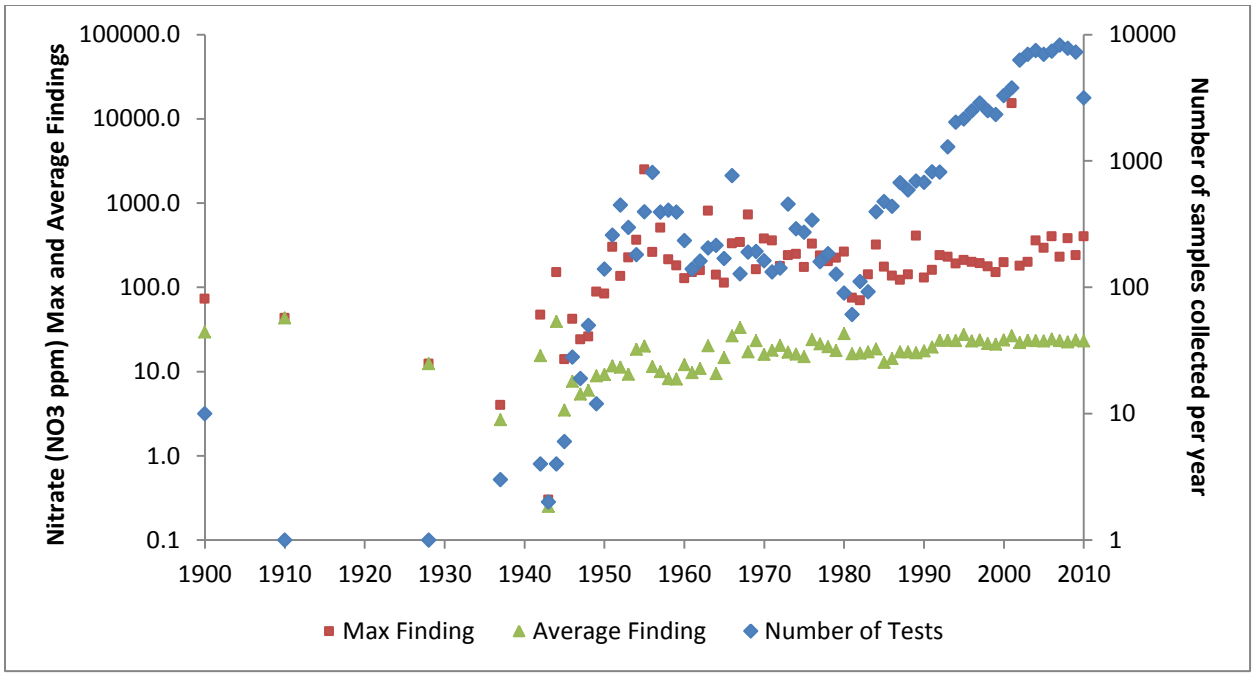

Figure 37. Number of samples collected per year, with maximum and average nitrate (as nitrate) mg/L (each year), in CASTING database.

Adminsitrative Record
Page 35022

5 Analysis of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database

Prepared by:

Thomas Harter, Aaron King, Dylan Boyle, Giorgos Kourakos

Adminsitrative Record
Page 35023

5.1 Introduction

In this section, maps and statistical analyses of the spatio-temporal distribution of nitrate concentration in wells in the study area are presented. Unless mentioned otherwise, all nitrate concentrations [mg/L] are expressed for nitrate as nitrate.

Nitrate concentrations in wells vary widely with location and well depth. More domestic wells and unregulated small system wells have high nitrate concentrations due to their shallow depth (Table 20). Highest nitrate concentrations are found in wells of the alluvial fans in the eastern TLB and in wells of unconfined to semi-confined aquifers in the northern, eastern, and central SV (Figure 41). In the Kings, Kaweah, and Tule River groundwater subbasins of Fresno and Kings County, and in the Eastside and Forebay subbasins of Monterey County, one-third of domestic or irrigation wells exceed the nitrate MCL. Consistent with these findings, the maximum nitrate level, measured in any given land section (1 square mile) for which nitrate data exist between 2000 and 2009, exceeds the MCL across wide portions of these areas. Low nitrate concentrations tend to occur in the deeper, confined aquifer in the western and central TLB. From 2000 to 2011, the median nitrate concentration (the concentration exceeded by half of all samples) in the TLB and SV public water supply well samples was 23 mg/L and 21 mg/L, respectively, and in all reported non-public well samples, 23 mg/L and 20 mg/L, respectively. In public supply wells, about one in ten raw water samples exceeds the nitrate MCL.

Nitrate levels have not always been this high. While no significant trend is observed in some areas with low nitrate (e.g., areas of the western TLB), USGS research indicates significant long-term increases in the higher-nitrate areas of the TLB (Burow et al. 2008), which is consistent with the CASTING dataset. Average nitrate concentrations in public supply wells of the TLB and SV have increased by 2.5 mg/L (±0.9 mg/L) per decade over the past three decades. Average trends of similar magnitude are observed in private wells. As a result, the number of wells with nitrate above background levels (> 9 mg/L) has steadily increased over the past half century from one-third of wells in the 1950s to nearly two-thirds of wells in the 2000s (Figure 57). Due to the large increase in the number of wells tested across agencies and programs, the overall fraction of sampled wells exceeding the MCL grew significantly in the 2000s.

5.1.1 Disparate Data Sources and Collection Methods

The data analyzed in this section were gathered from many different sources (see Section 4). As such, the spatial accuracy and precision of these data are variable, with some wells located to a high degree of certainty and others very poorly located. Nitrate concentration data, in contrast to spatial data, was generally collected with consistent protocols across the various datasets, though detection limits have improved over time. The difference between filtered and unfiltered nitrate concentrations is assumed to be insignificant.

Adminsitrative Record
Page 35024

5.1.2 De-clustering of Data Sources

Wells documented in the CASTING database are very unevenly distributed across the study area. In addition, specific data sources target only specific sub-areas within the study area (e.g., public supply wells are dominantly located in urban areas). For some of the statistical analyses described here, we performed a spatial de-clustering of the dataset to estimate nitrate concentrations by aquifer proportions (Belitz et al. 2010). The study area was first divided into five major physiographic groundwater regions based on sedimentological, hydrological, and geological characteristics.

For the SV, the study area was divided into two regions following the USGS GAMA 2005 study (Kulongoski & Belitz 2007). The two areas used were the "Monterey Bay" portion of the SV, which encompasses the Pressure, Eastside, Langley, Seaside, and Corral de Tierra groundwater subbasins, and the "Salinas Valley" area, which contains the Upper Valley and Forebay groundwater subbasins. For the Monterey Bay area, the subbasins were combined based on the similarity in Quaternary deposits (Kulongoski & Belitz 2007). The Forebay and Upper Valley subareas were combined based on their similar geology (Kulongoski & Belitz 2007).

The TLB study area was divided into three separate regions based on sediment origins. The three groups, referred to as the Eastside Alluvial Fan, Basin, and Westside Alluvial Fans regions, were delineated by the USGS and refer to the origin of the alluvial sediments present in these regions (Burow et al. 1998).

Each of the five regions in the study area is further divided into spatially distributed, randomized equal area cells (Figure 38 and Figure 39). The equal area cells for the Monterey Bay and Salinas Valley regions were those used in a 2005 USGS GAMA study of nitrate and pesticide contamination in the SV (Kulongoski & Belitz 2007). For the Monterey Bay region, the study area for this report did not include a northern region included in the 2005 USGS GAMA report, and therefore those cells located outside study area boundary were not included in the analysis provided here. The Monterey Bay region was divided into 48 equal area cells, the Salinas Valley into 31 equal area cells, each measuring 25 km^2 (9.7 sq. miles). The equal area cells within the Eastside Alluvial Fan, Westside Alluvial Fan, and Basin regions of the TLB were generated for this project using the same method by Scott (1990) that was also applied in the USGS GAMA studies. Cell size in the three TLB regions was 81 km^2 (31 sq. miles) : The Eastside Alluvial Fans was divided into 120 equal area cells, the Basin into 65 equal area cells, and the Westside Alluvial Fans into 56 equal area cells (Figure 39). The equal area cells are used to determine representative statistical values for each equal area cell (for example, the mean nitrate concentration in a cell). Further statistical analysis is then performed on the values representative of each cell, thus giving equal weight to each cell area. Details of the statistical analyses performed on equal area cells and the results of the analyses are described in the following sections. A variety of statistical tests are performed. The sections below specifically identify when the equal area cells are used for de-clustering the data (e.g, Table 20, Table 24, Figure 46, Figures 58-60).

Adminsitrative Record
Page 35025

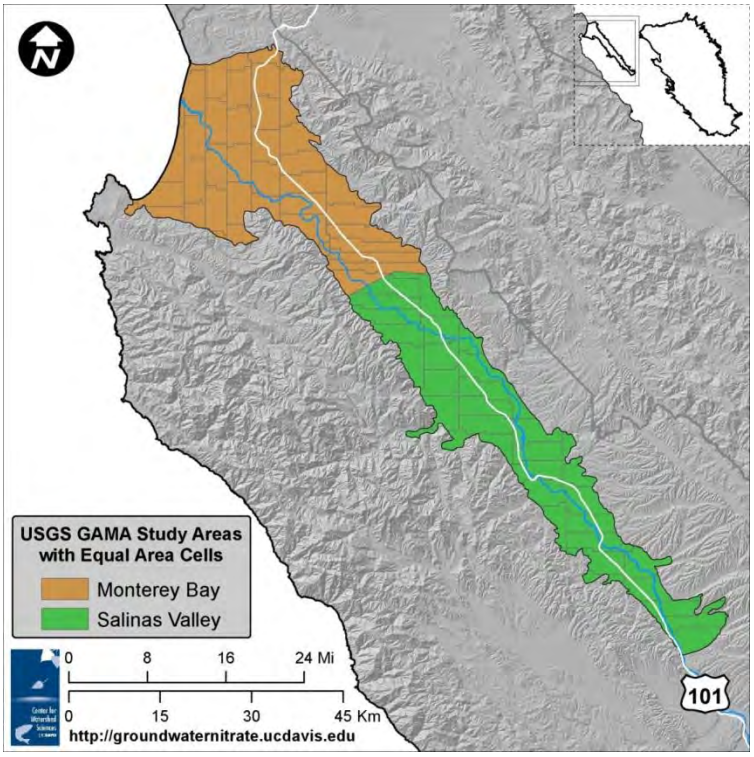

Figure 38. The two physiographic groundwater regions in the SV and their equal area cells used for spatial de-clustering. (Source: Kulongoski & Belitz 2007.)

Adminsitrative Record
Page 35026

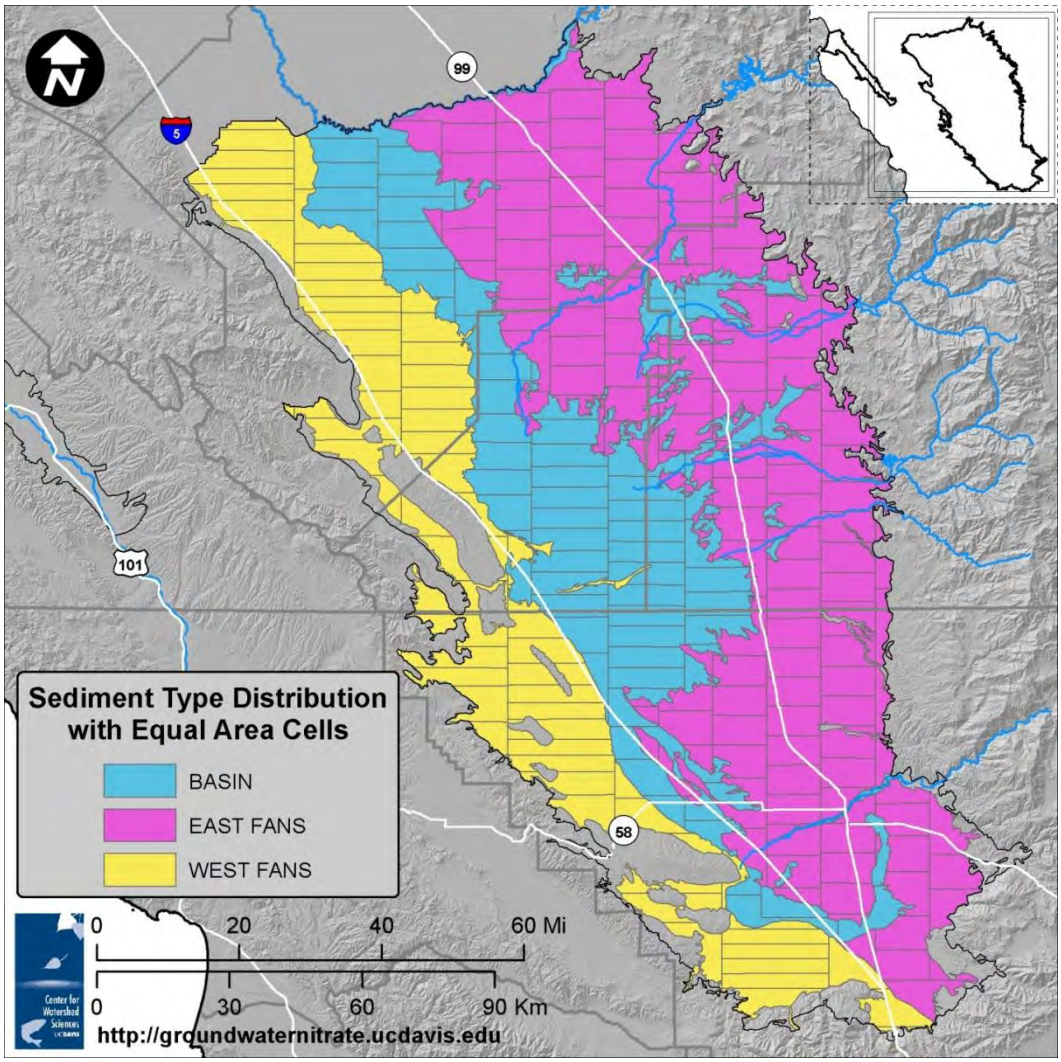

Figure 39. The three physiographic groundwater regions of the Tulare Basin study area and their corresponding equal area cells, which we generated for this report using the procedure of Scott (1990).

Adminsitrative Record
Page 35027

5.2 Current Spatial Distribution of Nitrate in Wells

5.2.1 Dataset Overview

The CASTING database version used for the analysis in this section is versioned 2012-01-29. The database contains a total of 100,084 samples from 19,095 wells within the study area and analyzed for nitrate. The dataset combines nitrate concentrations from 16,709 individual samples taken at 1,890 wells in the SV and from 83,375 individual samples taken at 17,205 wells in the TLB collected from the 1940s to 2011. Almost 70% of these samples were collected from 2000 to 2010; only 15% of the samples were collected prior to 1990. Half of all wells sampled had no recorded samples prior to 2000.

Of the 19,000 wells, approximately 2,500 are frequently sampled public water supply wells (over 60,000 samples) located often in urban or peri-urban areas. The Central Valley dairy regulatory program, established in 2007, annually monitors about 4,000 domestic and irrigation wells in the TLB, located mostly in agricultural and rural areas. Apart from these two programs, there are no existing other regular well sampling programs for domestic or other private wells. The exceptions are a ten year old monitoring program on 71 domestic wells in eastern Fresno and Tulare County (DPR), sampled once or twice annually; and Monterey County Water Resources Agency's private well monitoring program data, ongoing for two decades, but with most data confidential and not accessible to this study.

The distribution of nitrate values among the 100,084 samples is highly skewed with most values (86%) less than the drinking water limit of 45 mg/L. One quarter of all samples measure less than 6.8 mg/L, half of all samples measure 19 mg/L or less, and one quarter of all samples measure more than 32.3 mg/L. One in twenty samples exceeds 79 mg/L and one in a hundred samples exceeds 194 mg/L (more than four times the drinking water standard).

5.2.2 Exceptionally High Values and Treatment of Outliers

A very small fraction of samples, 202 records or two-tenths of one percent of all records, indicate nitrate concentrations of 500 mg/L or higher (more than one order of magnitude above the drinking water limit). Of these samples, most are associated with environmental monitoring wells (166 samples) with the highest recorded concentration being nearly 26,000 mg/L. All these environmental monitoring well samples were taken in the 2000s (after 1/1/2000). Another 26 samples are associated with USGS research projects (NWIS database). It is likely that these wells also sample shallow-most groundwater. Among the 36 wells with unknown well use or not designated as environmental monitoring wells (Well Use is *not* "Q"), one sample is 2,500 mg/L, 7 samples are between 1,000 and 1,700 mg/L, and 28 samples are between 500 and less than 1,000 mg/L. Only five of these 36 samples were taken since January 1, 2000 and among these five samples, the maximum concentration is 903 mg/L.

Unless explicitly mentioned, we did not remove any of these values as outliers. Instead, for many analyses, samples taken from wells designated as environmental monitoring wells (except the "monitoring wells" in the Monterey County Water Resources Agency agricultural environmental

Adminsitrative Record
Page 35028

monitoring program, “MCEM”) were considered separately or excluded. Among the 1,018 MCEM monitoring well samples (from 239 wells), only one sample exceeded 500 mg/L (value: 681 mg/L), and it was taken prior to 1/1/2000. MCEM wells that are designated as “monitoring wells” were not considered part of the “environmental monitoring well” set, since these are not installed to monitor existing site contamination. MCEM wells are included in the analyses below, where the analysis is designated as excluding environmental monitoring wells.

5.2.3 Frequency Distribution

The frequency distribution (histogram) of nitrate values is best fitted with a log-normal distribution. Statistical tests (Chi-Square test and Kolmogorov-Smirnov test, Haan, 1977) for goodness-of-fit with the log-normal distribution are significant at the 1% level. The same goodness-of-fit tests, at the 1% significance level, suggest that the following aggregated nitrate data are also log-normal distributed:

- The 19,095 mean nitrate values computed for each well, with the mean computed for the entire period of record.
- The 23,402 mean nitrate values computed for each well for each decade with well sample records (“decadal well means”).
- The 55,541 mean nitrate values computed for each well for each year with well sample records (“annual well means”).

The $\log_{10}(c)$ of all samples, where c is the sample nitrate value, and the mean of $\log_{10}(c)$ for each well for each decade statistically fit a normal distribution, although the histogram is slightly skewed (Figure 40). The sample mean of $\log_{10}(c)$ is 1.116 (13.1 mg/L of nitrate) and the standard deviation is 0.631, more than a half order of magnitude. The decadal well means of $\log_{10}(c)$ have a mean of 0.929 (8.3 mg/L of nitrate) and a standard deviation of 0.753, nearly one order of magnitude. Because the 2,500 public supply wells make up more than half of all samples, but only one-eighth of all wells, statistics on the total set of all 100,000 samples will be highly biased toward measurement in public supply wells. The distribution of means computed for each well is a better representation of the nitrate distribution in the study area than the distribution of the full sample set. An even less biased analysis is obtained by applying statistical measures to the equal area cells described in the previous section (de-clustered analysis).

Adminsitrative Record
Page 35029

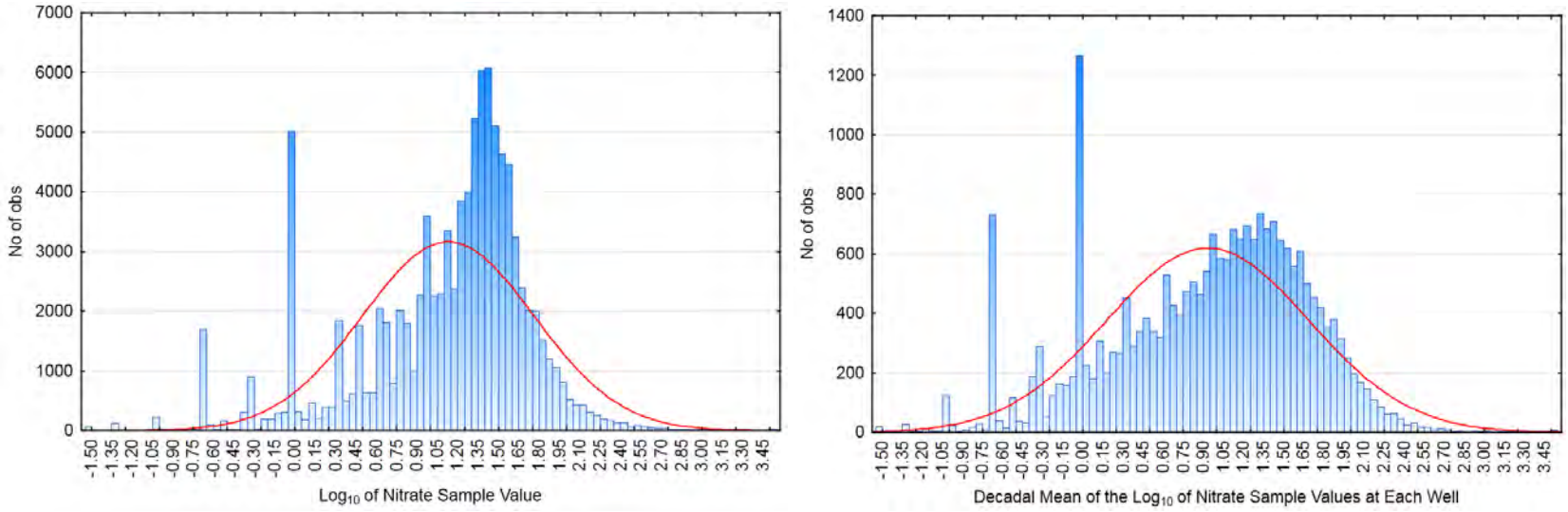

Figure 40. Histogram of $\log_{10}(c)$ (x-axis), where *c* is the sample nitrate value, for all 100,029 non-zero individual samples (left), and for all 23,377 non-zero decadal means of $\log_{10}(c)$ at individual wells (right). The distributions statistically fit a normal distribution, although they are slightly skewed. A large number of samples are recorded at half of often reported detection limits (detection limits were converted to actual concentrations at half of the detection limit). In logarithmic units, the MCL for nitrate is 1.65. X-axis values of -1, 0, 1, 2, and 3 correspond to nitrate values of 0.1, 1, 10, 100, and 1000 mg/L, respectively.

Adminsitrative Record
Page 35030

Due to the skewed distribution of nitrate, much of the remaining analysis focuses on statistical measures that are not dependent on distributional assumptions and that are robust against outliers or high values. In particular, we use medians (exceeded by exactly half of the sample populations) and exceedance probabilities relevant to nitrate. The exceedance probability is the probability that nitrate exceeds a certain value, that is, the fraction of wells in which nitrate exceeds a certain value. In particular, we consider the exceedance probabilities for the following four thresholds:

- 9 mg/L nitrate, which is the level that is often used to separate background nitrate concentrations from anthropogenically influenced nitrate concentrations
- 22.5 mg/L nitrate, which is half of the maximum contaminant level (MCL)
- 45 mg/L nitrate, the drinking water MCL
- 90 mg/L nitrate, twice the drinking water MCL

We focus our analysis on nitrate data collected since January 1, 2000. We consider the 2000s period representative for overall current conditions. Older data are considered for the time trend analysis (see below). Furthermore, we consider the following sample populations for which we compute statistical values:

- All raw sample values ("sample values")
- The annual median nitrate concentration at each well ("annual well median")
- The annual mean nitrate concentration at each well ("annual well mean")
- The decadal mean nitrate concentration at each well ("decadal well mean")
- The decadal mean of $\log_{10}(c)$ at each well, where c is the nitrate concentration ("decadal well mean log nitrate")

5.2.4 Spatial Distribution of Nitrate Concentration

There are 10,120 wells in the TLB study area sampled at least once during the 2000's decade. The back-transformed median concentration[9] of the decadal well means of log nitrate is 12 mg/L. The median of the annual well means is 15 mg/L. About one-quarter (24%) of all wells have a decadal well mean of log nitrate exceeding the MCL, while nearly 20% of annual well mean concentrations exceed the MCL. There are 1,768 wells in the TLB that have been sampled at least 4 times. Of these, 406 (23%) exceeded the MCL for nitrate at least once in that decade.

In the SV, there are 1,474 wells sampled at least once during the 2000's decade. The back-transformed median concentration of the decadal well means of log nitrate is 9.4 mg/L. The median of the annual well means is 14 mg/L. About 13% of all wells have a decadal well mean of log nitrate exceeding the MCL, and 15% of annual well mean concentrations exceed the MCL. In the SV, 729 wells have been

[9] The "backtransformed median" of a log-transformed variable is equal to 10 to the power of the median of the log10-tranformed variable. For example if the median of the log10-transformed nitrate concentrations is 1.3, then the backtransformed median is $10^{1.3}$ = 19.95.

Adminsitrative Record
Page 35031

sampled at least 4 times during the 2000s, of which 215 (29%) exceeded the MCL for nitrate at least once in that decade.

The above represent different statistical measures that illustrate the range of "typical" or median nitrate in groundwater and of the rate at which the MCL is exceeded across various statistical approaches. The decadal well means represent one mean for each well, regardless of the number of samples from that well or in how many years during the decade it would have been sampled. In the computation of the median of decadal well means, each well weighs equally. In contrast, for each well there may be one or several annual well means per decade. Therefore, wells with multiple years of sampling weigh proportionally more in the computation of the 'median of annual well means' than wells with a single sample during the decade. In the median of annual well means, for example, public wells weigh generally ten times more than private domestic wells, because most public wells are tested at least once each year (one annual well mean per year per well), while most domestic and other private wells in the database have only been sampled once (except those in the "dairy" dataset). In the TLB and the SV, the median of annual well means is higher than the back-transformed median of decadal well means of log nitrate, indicating that wells with multiple samples tend to have higher nitrate values. In contrast, in the TLB, the MCL exceedance rate among decadal well means of log nitrate is higher than among annual well means. To further investigate these differences, we will analyze subsets of these data (e.g, public supply wells, domestic wells) further in the following sections. Importantly, the diverse measures given here all demonstrate the significant percentage of wells that exceed the MCL and provide a measure of the magnitude of median nitrate concentration in groundwater.

In the TLB, wells in the eastern and central portions of the valley tend to have higher average concentrations of nitrate than those in the western part of the valley (Figure 41). The Westside Alluvial Fans region of the TLB has fewer wells, but the data that are available suggest lower average levels of nitrate in these wells compared to the eastside. In the SV, the higher average concentrations are observed in the northeastern, central, and southern portion of the SV. The map of maximum values observed in each well (Figure 42) appears visually very similar to average values (Figure 41) due to the large number of wells with a single or few measurements.

Due to the density of wells in some areas of the study area, it is difficult to visualize all wells directly. To better understand the distribution of nitrate concentration in the study area, the average concentration of nitrate in wells for each Public Lands Survey Section for the years 2000 through 2009 were calculated (Figure 43). In any given section, there may be anywhere from one well with a single sample to many wells with one or multiple samples per well. The average nitrate concentration in each section was computed from the average nitrate concentration at each well in that section in 2000 – 2009. Thus, for the average nitrate concentration in each section, each well was counted only once, regardless of the number of samples. A map showing the maximum nitrate concentration observed anywhere within each section between 2000-2000 was similarly prepared (Figure 44).

When the median of all annual well mean nitrate values are computed for each equal area (see Section 5.1 for maps of equal areas), the de-clustered median nitrate concentrations over all equal areas (not including environmental monitoring wells) are 16 mg/L (TLB Eastside Fans), 2.6 mg/L (TLB Basins), 1.5

Adminsitrative Record
Page 35032

mg/L (TLB Westside Fans), 13 mg/L (Monterey Bay region), and 12 mg/L (Salinas Valley region) (see Figure 45). For a de-clustered analysis of exceedance probabilities, we compute the MCL exceedance probability for each equal area separately, based on the number of annual well means exceeding the MCL within each equal area during the 2000s (not including environmental monitoring wells). The de-clustered average MCL exceedance rate (average of the exceedance rate in each equal area) in each of the five groundwater regions is 20% (TLB Eastside Fans), 11% (TLB Basins), 14% (TLB Westside Fans), 14% (Monterey Bay region), and 22% (Salinas Valley region) (see Figure 46).

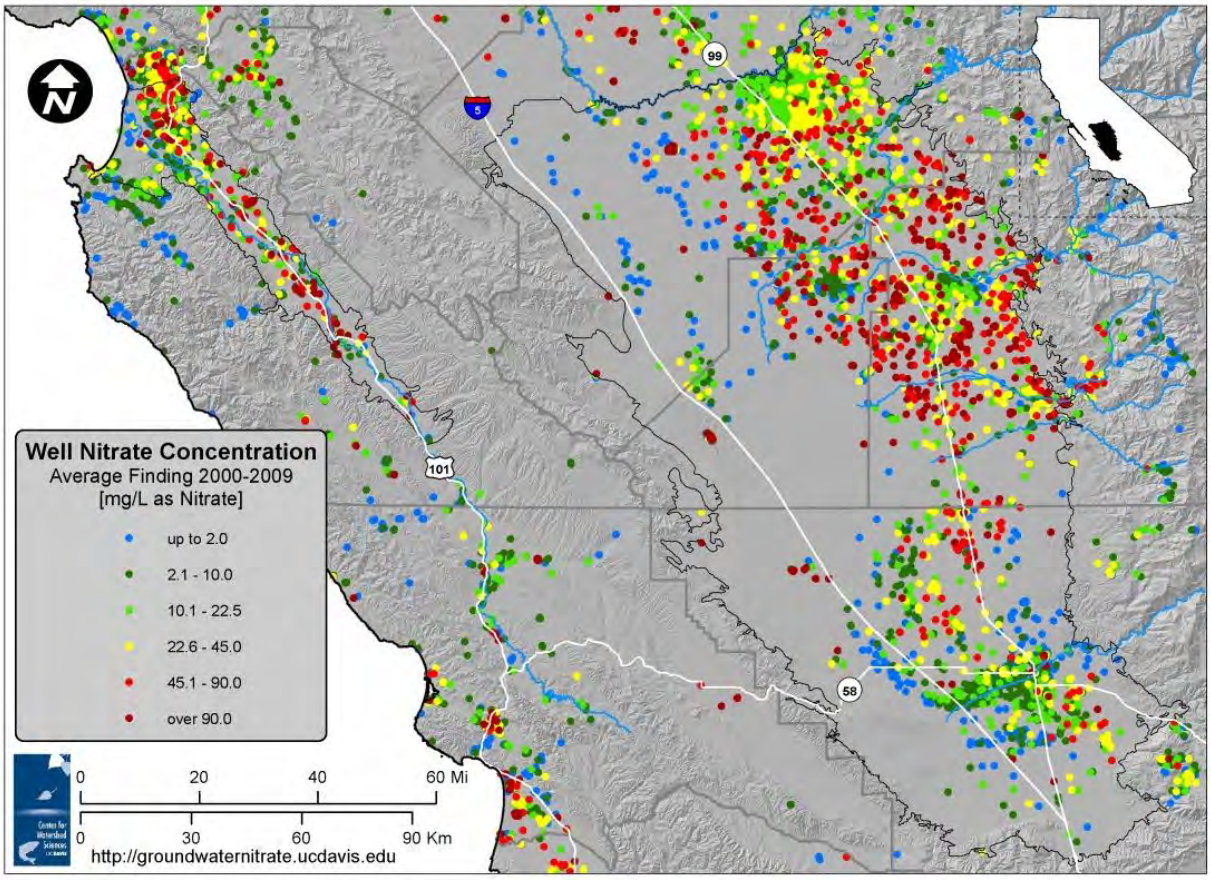

Figure 41. Average concentration of nitrate (mg NO3/L) in wells with at least one sample during the decade 2000 to 2010 (not including SWB Geotracker environmental monitoring wells). Red and dark red indicate wells with average concentration above the MCL. Yellow indicates wells that are above one half the MCL, but below the MCL.

Adminsitrative Record
Page 35033

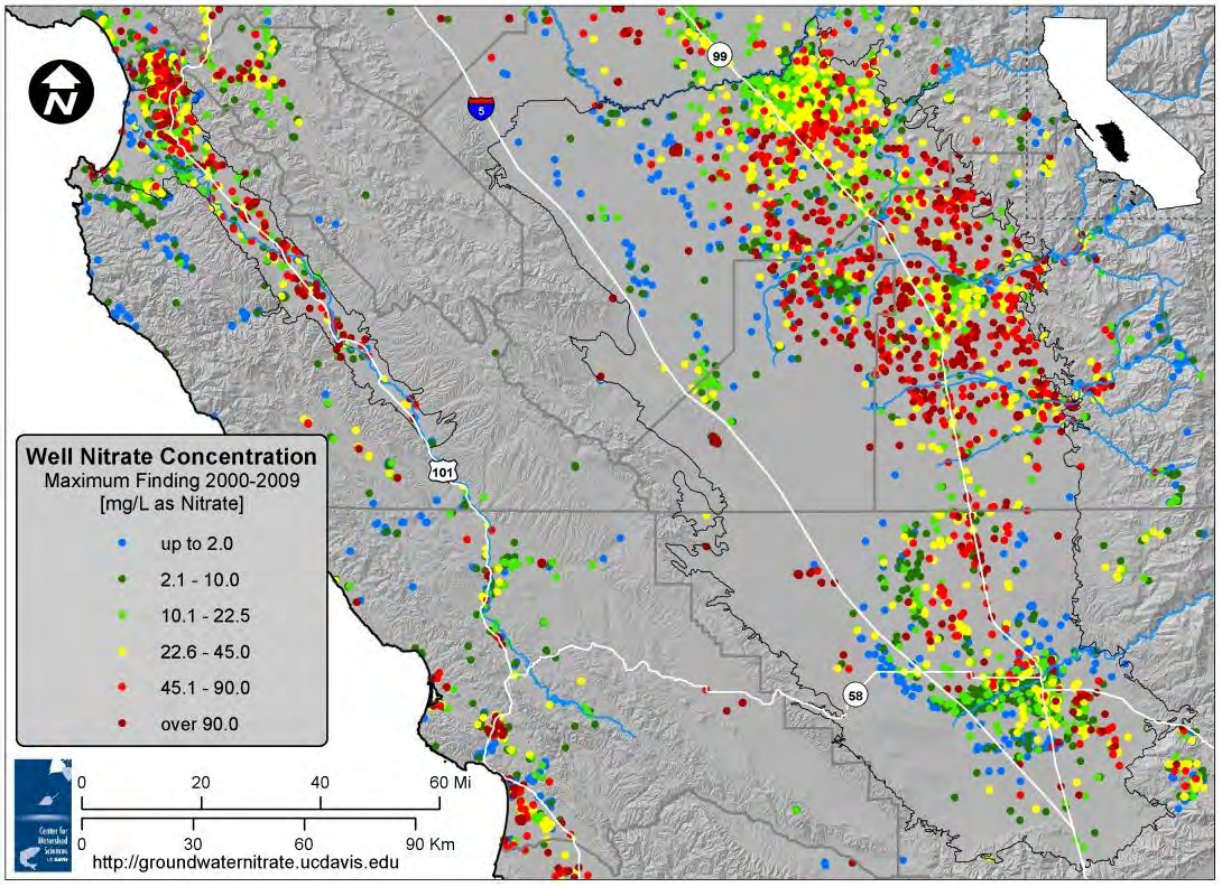

Figure 42. Maximum concentration of nitrate in wells with at least one sample during the decade 2000 to 2010 (not including SWB Geotracker environmental monitoring wells). Red and dark red indicate wells with a maximum measured concentration over the MCL. Yellow indicates wells with a maximum measured concentration that is above one half the MCL but below the MCL.

Adminsitrative Record
Page 35034

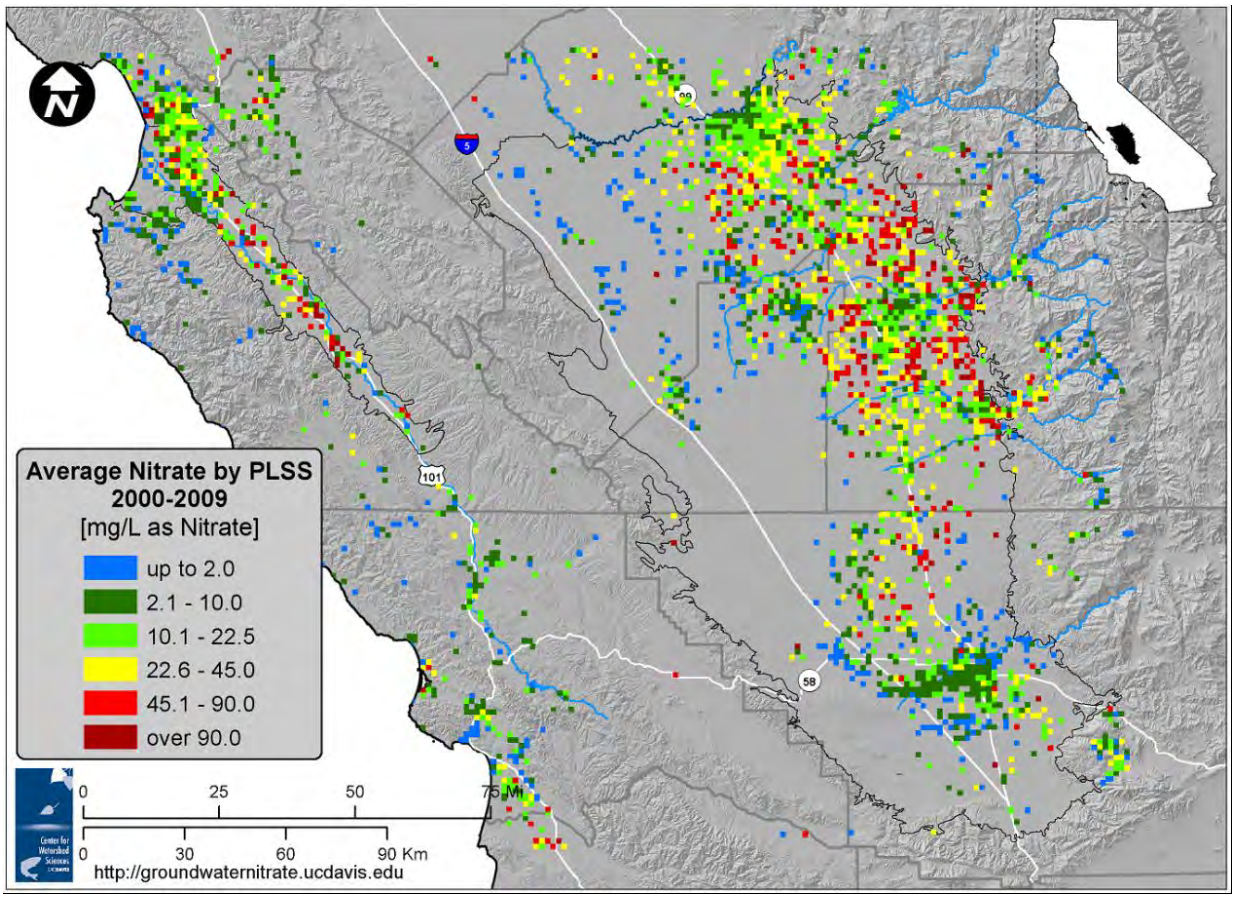

Figure 43. Average nitrate by PLSS section, approximately 2.59 km^2 (1 mi^2), computed from the average at each well of all samples obtained during the years 2000 to 2009 (not including samples from SWB Geotracker environmental monitoring wells). Thus, each well is only counted once toward the average nitrate in each section, regardless of the number of samples obtained for each well. Red and dark red sections are those where the average of all samples in a PLSS section is over the MCL. Yellow indicates PLSS sections where the average of all known samples in this time period is above one half of the MCL.

Adminsitrative Record
Page 35035

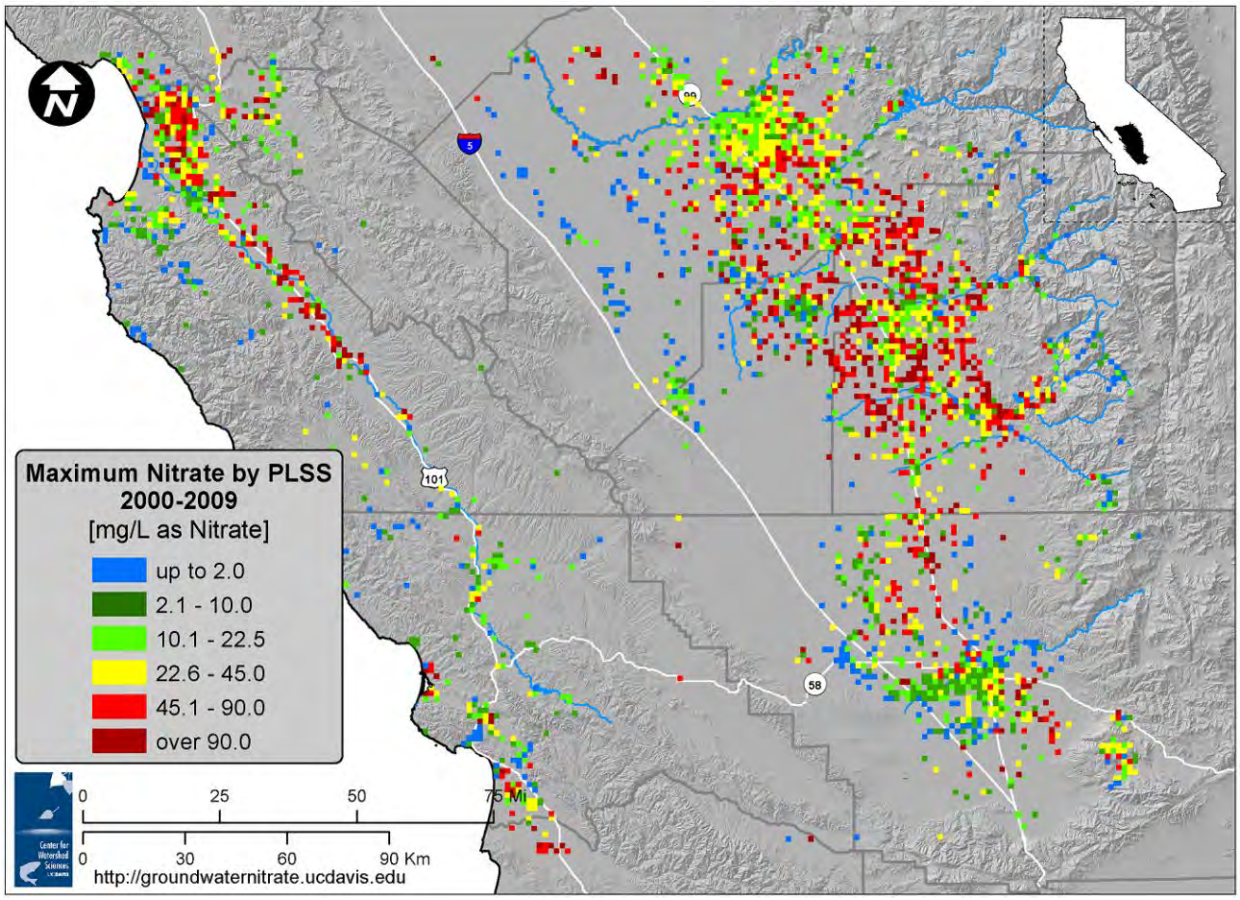

Figure 44. Maximum nitrate concentration measured between 2000 and 2009 in each PLSS section (not including samples from SWB Geotracker environmental monitoring wells), where each section is approximately 2.59 km^2 (1 mi^2).

We de-cluster the data by first taking the annual well means for the 2000s (except environmental monitoring wells), then computing the median of all annual well means in each equal area, and finally taking the mean of the logarithm of equal area medians across all equal areas within each groundwater region (Figure 46). The logarithm is taken because equal area medians are log-normally distributed across the equal areas of the study area. When the data are de-clustered by equal area, the TLB (Central) Basin area has a lower back-transformed mean nitrate (3.2 mg/L) than before de-clustering (7.8 mg/L). All other groundwater regions have similar mean nitrate with or without de-clustering (compare Figure 45 with Figure 46).

Adminsitrative Record
Page 35036

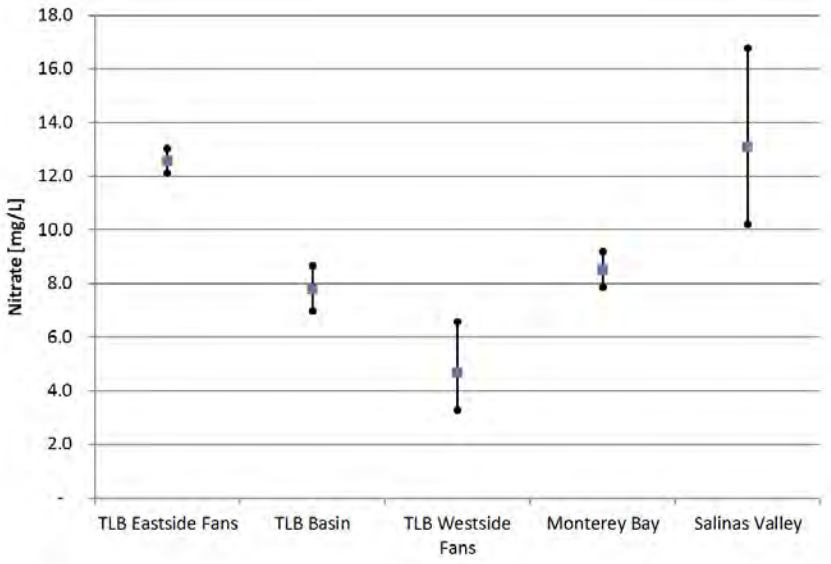

Figure 45. Back-transformed mean and confidence interval of decadal well means of log nitrate [y-axis: in mg/L nitrate] for the five major groundwater regions in the study area without equal area de-clustering. Includes all wells sampled in the 2000s, except environmental monitoring wells.

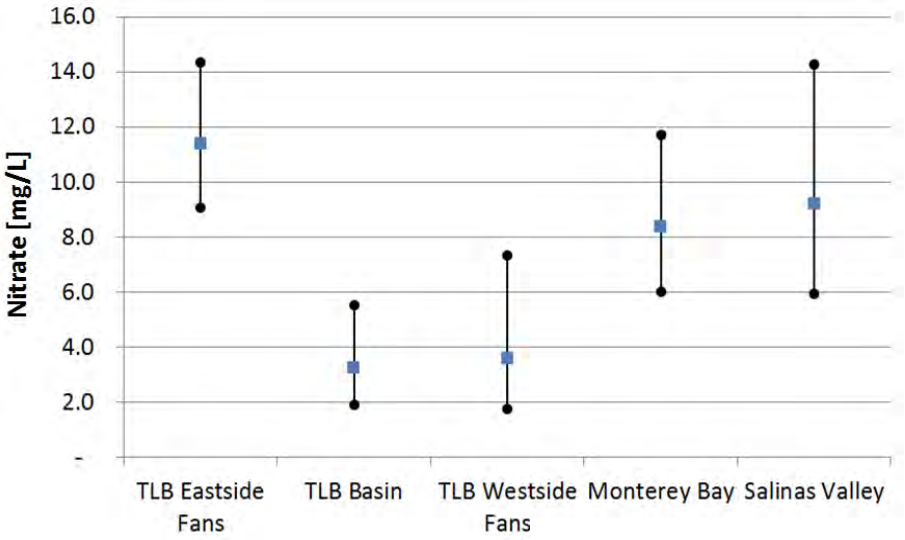

Figure 46. Back-transformed mean and confidence interval of the mean over all equal areas of the logarithm of the median in each equal area of annual well means [y-axis: in mg/L nitrate], separately for the five major groundwater regions in the study area. Note that the medians of equal area annual well means are log-normally distributed. Includes all wells sampled in the 2000s, except environmental monitoring wells. Confidence intervals are broader due to the smaller number of data points (number of equal area cells) when compared to the previous figure's number of data points (number of wells) used for the statistical analysis.

Adminsitrative Record
Page 35037

5.3 Effect of Data Source, Well Type, and Well Depth on Groundwater Nitrate

5.3.1 Effect of Data Source

Many of the individual datasets that comprise the CASTING database for the 2000s represent groundwater nitrate samples taken for specific purposes, from specific well types, from a specific region, or under a specific program. This includes the CADWSAP dataset (public water supply wells, mostly located in urban areas, sampled frequently) and the dairy dataset ("Dairy", representing irrigation, domestic, and monitoring wells on dairies and sampled annually since 2007, under the dairy general order). It also includes the various county datasets collected from individual wells once, typically after construction of new domestic wells (Tulare and Fresno Counties, some of the Kern County records), on local small community water systems (Monterey County), or for regional monitoring purposes (DPR, GAMA, MCEM). Two large datasets in CASTING cover only historic (pre-2000) water samples and did not continue into the 21st century and were not considered here: U.S. EPA STORET and DWR legacy data.

We consider the data separately by major data source (Table 17 and Table 18) to illustrate the variability between different groundwater sampling programs. Nitrate samples were aggregated by well and year: for each well *i*, we first obtained the average nitrate concentration, $\langle c \rangle_{ij}$, in each year *j* ("annual well mean"). We then performed an exceedance probability analysis on the ensemble of all annual well means, $\langle c \rangle_{ij}$, using all wells *i* (of a particular grouping) and all years *j* (2000 – 2011) within the dataset. Table 17 shows, by data source, the number of wells for which samples were available, and the number of samples, the median nitrate value [mg/L], and the exceedance probability for the key threshold levels. Due to the significantly different nitrate concentrations in the five major groundwater regions, separate analyses were also performed for the major groundwater regions of the study area. Data for 2010 were incomplete, but included. Only few data were available for 2011 (not included in all analyses). Table 18 breaks down the total number of annual average well nitrate values available between 1/1/2000 and 2011 (number of well-years), by data source and by region.

Adminsitrative Record
Page 35038

Table 17. 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 TLB and SV: number of wells measured, median nitrate concentration, and percentage of MCL exceedance for the TLB and the SV.[2]

Data Source[1]	Total				TLB (2000 – 2010)			SV (2000 – 2010)		
	# Wells	# Samples	Well Locations	Well Type[3]	# Wells	Median[2] [mg/L NO_3^-]	% > MCL[2]	# Wells	Median[2] [mg/L NO_3^-]	% > MCL[2]
CDPH	2,421	62,153	throughout study area	PS	1,769	12	6%	327	8	5%
CVRWB DAIRY	6,459	11,300	dairies in TLB	D, I, M	6,459	22	31%			
DPR	71	814	eastern Fresno and Tulare Counties	D	71	40	45%			
DWR	26	44	Westlands Water District	I	28	1	0%			
DWR Bulletin 130	685	2,862	throughout study area	D, I, PS						
ENVMON	537	2,601	throughout study area	M	357		52%	180	27	44%
EPA	2,860	4,946	throughout study area							
Fresno County	368	369	Fresno County	D	349	18	15%			
GAMA	141	141	Tulare County	D	141	38	43%			
Kern County	2,893	3,825	Kern County	D, I	361	5	7%			
Monterey County, Reports	239	1,018	Monterey County	I, M				98	14	36%
Monterey County, Geospatial	388	1,574	Monterey County	LS				431	18	15%
Monterey County, Scanned	452	5,674	Monterey County	LS				427	17	14%
NWIS	1,028	2,151		Misc.	76	35	36%	4	0	0%
Tulare County	444	444	Tulare County	D	438	22	27%			
Westlands Water District	48	77	Westlands Water District	I	31	4	0%			

[1]Data Source: CDPH: public supply well database; CVRWB Dairy: Central Valley RWB Dairy General Order; DWR Bulletin 130: 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] Median and percent MCL exceedance were computed based on the annual mean nitrate concentration at each well for which data were available (not spatially de-clustered). Depending on the number of years (between 2000 and 2010) for which data are available for a specific well, a well may be represented by one to eleven datapoints. See Table 18 for details.

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

Adminsitrative Record
Page 35039

For the various datasets, the public water supply wells (CADWSAP dataset) generally have among the lowest rates of MCL exceedances, and the by far lowest median of annual well means (less than 13 mg/L). Other datasets with less than 10% exceedance rates of the MCL are the Kern County wells (except the group of wells on the Westside Alluvial Fans), and the Westlands Water District irrigation wells dataset provided by WWD and DWR. Except for the public supply wells, these wells were mostly measured just once. The Westlands Water District wells are of unknown depths, but, based on their location, are likely deep wells pumping from the lower portions of the upper, semi-confined aquifer or from the confined aquifer below the Corcoran Clay. The Kern County dataset, which also has relatively low nitrate median, includes irrigation wells, many of which may tap deeper portions of the aquifer.

Many of the public supply wells (CADWSAP dataset) are located in urban or semi-urban areas and are impacted by urban recharge, particularly in the city of Fresno. The CADWSAP dataset includes many wells that are screened only at depths exceeding 100 m (330 ft). Wells with chronically elevated nitrate levels are often taken off-line by the water purveyor. Data representing water quality conditions after the well is taken off-line would not be sampled and therefore not be included in the dataset.

Monitoring wells (those in the State Water Board Environmental Monitoring Wells dataset and monitoring wells identified, by name, as monitoring wells in the Dairy dataset) have the highest exceedance rates for the MCL and also for the 90 mg/L threshold. More than half of these monitoring wells, deployed typically to track a known or suspected contamination problem, exceed the nitrate MCL and most of these latter wells measure more than 90 mg/L, twice the MCL. Monitoring wells are typically constructed to measure first encountered groundwater, with well screens at or immediately below the water table. They represent the water quality of recent recharge at the location of monitoring. Many of these are deployed specifically to locations with suspected or known contamination problems.

The datasets that consist exclusively of domestic wells (Fresno and Tulare County, DPR, GAMA Tulare project, and some of the Dairy dataset wells, where identified) show varying rates of nitrate MCL exceedance, ranging from about 30% to 45%. Wells monitored by the Monterey County Water Resources Agency and the Monterey County local system wells (serving less than five connections) show similar exceedance rates in the SV. Only the local systems wells in the northern SV (“Monterey Bay”, which includes the Pressure Aquifer and Eastside Basin sub-regions of SV) have lower MCL exceedance rates (less than 15%). Median values in domestic wells of the DPR and GAMA dataset, at about 40 mg/L, are highest in the Eastside Alluvial Fans region of the TLB. Median concentrations are lower in the central basin and Westside Alluvial Fans region of TLB

Adminsitrative Record
Page 35040

Table 18. Summary of database records for nitrate samples for the current period (after 1999) including the data source, the number of wells measured by each data source, the number of wells-years (the sum of all wells, each multiplied by the number of years during which it was sampled), the median nitrate value (exceed by 50% of all samples), and the percent of annual well means that exceeded 9, 22.5, 45, and 90 mg/L nitrate.[1]

Region	Data Source	# of Wells	# Well-Years	Median [mg/L]	> 9 mg/L	> 22.5 mg/L	> 45 mg/L	> 90 mg/L
TLB Eastside Fans	CADWSAP	1,597	11,867	12.5	62.0%	26.1%	5.7%	0.8%
	DAIRY	5,310	9,078	23.0	67.5%	50.3%	30.8%	11.1%
	DPR	70	666	40.5	89.2%	72.8%	45.9%	12.8%
	FRCO	325	325	19.3	73.8%	45.8%	15.4%	1.2%
	GAMA	124	124	41.0	83.9%	68.5%	44.4%	19.4%
	KECO	248	248	5.1	29.8%	12.5%	8.5%	2.8%
	KICO	9	14	10.0	50.0%	28.6%	14.3%	7.1%
	NWIS	75	128	34.9	75.8%	60.9%	35.9%	5.5%
	TCEHS	394	395	22.2	75.7%	49.9%	27.1%	5.1%
	USGS	6	14	32.9	85.7%	57.1%	42.9%	14.3%
	Mon. Wells	298	553	53.8	77.0%	66.0%	52.8%	37.6%
TLB Central Basin	CADWSAP	147	912	8.0	47.0%	17.1%	6.7%	1.5%
	DAIRY	995	1,776	17.7	60.1%	44.8%	26.5%	8.7%
	FRCO	23	24	1.0	25.0%	12.5%	12.5%	0.0%
	GAMA	15	15	32.4	86.7%	66.7%	33.3%	13.3%
	KECO	91	91	4.4	35.2%	13.2%	3.3%	0.0%
	TCEHS	34	34	23.2	82.4%	50.0%	26.5%	5.9%
	Mon. Wells	114	323	122.2	74.0%	66.6%	61.3%	56.7%
TLB Westside Fans	CADWSAP	20	108	1.5	26.9%	5.6%	0.0%	0.0%
	DWR	27	27	1.0	22.2%	3.7%	0.0%	0.0%
	KECO	11	11	6.4	45.5%	45.5%	27.3%	18.2%
	WWD	30	54	3.4	33.3%	9.3%	0.0%	0.0%
	Mon. Wells	29	61	62.0	62.3%	59.0%	52.5%	45.9%
SV-Pressure Aquifer, Eastside, and Monterey Bay[2]	CADWSAP	270	1,511	8.1	48.6%	21.6%	4.6%	0.3%
	MCEM	31	126	6.7	47.6%	37.3%	31.7%	18.3%
	MOCO	395	1,319	18.0	70.6%	40.9%	12.4%	1.7%
	MOCODig	412	1,839	17.0	65.6%	41.3%	13.8%	1.5%
	Mon. Wells	170	570	25.6	60.0%	51.2%	43.0%	27.9%
SV-Forebay and Upper Valley[3]	CADWSAP	57	296	7.0	42.9%	18.9%	4.4%	0.3%
	MCEM	67	72	38.3	66.7%	56.9%	44.4%	23.6%
	MOCO	36	123	24.0	69.9%	51.2%	39.8%	22.0%
	MOCODig	15	56	22.5	60.7%	50.0%	35.7%	23.2%
	Mon. Wells	10	22	77.2	81.8%	72.7%	59.1%	45.5%

[1] Higher nitrate concentrations are exceeded by fewer samples. Only datasets with at least 5 wells and at least 10 well-years are listed. Monitoring well ("Mon. Wells") records are analyzed separately and consist of those from the State Water Board's Geotracker and those in the "DAIRY" dataset.

[2] The "Monterey Bay" area, designated by the 2005 USGS GAMA basin study, encompasses the following Bulletin 118 subbasins: Pressure (180/400 foot aquifer), Eastside, Langley, Seaside, and Corral de Tierra areas.

[3] This area is called the "Salinas Valley" area in the 2005 USGS GAMA basin study.

Adminsitrative Record
Page 35041

5.3.2 Depth-Dependency of Nitrate

Given the diversity of well sampling programs, the lack of consistent information on well depth or screen length, and the wide range of the number of samples taken at individual wells, a quantitative regression to determine the depth-dependency of nitrate is not possible. Instead, we compared five groups of wells (not exclusive of each other, in other words, some wells may occur in multiple groups) to qualitatively determine, whether there is a potential effect with well-depth:

- Monitoring wells: State Water Board environmental monitoring wells and monitoring wells labeled as such in the CVRWB dairy dataset. Thought to be screened at depths of less than 30 m (100 ft) below the water table;
- Shallow private wells:
    - Domestic wells or wells of local water systems with less than five connections, designated as such in CASTING. Mostly (but not exclusively) screened at depths of less than 100 m (330 ft);
    - Private wells, including explicitly designated domestic wells, with known depth to the top-of-screen of 60 m (200 ft) or less;
- Shallow private and public wells: All wells, including public supply wells and those identified as domestic wells, with known depth to the top-of-screen of 60 m (200 ft) or less;
- All public supply wells (CADWSAP dataset), including those with unknown screen depth and those with deeper screens, and all wells from the Westlands Water District dataset (WWD and DWR).

Medians and exceedance probabilities of annual well means for these five groups were computed as described in the previous section (for Tables 17 and 18) and are shown in Table 19. As discussed above, the monitoring wells have the highest exceedance rate for both the MCL and the 90 mg/L (twice the MCL) threshold. Their median concentrations range from less than the MCL in the northern SV to over 120 mg/L in the central TLB.

Shallow private wells located in the eastern TLB and in the southern portion of the SV (Forebay and Upper Valley subbasins) have MCL exceedance rates of more than 30% and nearly 45%, respectively, in agreement with exceedance rates found in previous studies (i.e., Burow et al. 1998). Lower exceedance rates are observed in domestic wells elsewhere. Median concentrations are as high as 27 mg/L or as low as 16 mg/L.

Along the central basin of the TLB, the MCL exceedance rates in shallow private wells are more moderate, about one-quarter. Less than five such wells are located on the Westside alluvial fans of the TLB, not sufficient for a significant determination of exceedance rates in that groundwater region.

In the northern SV (Monterey Bay area, Pressure and Eastside subbasins), the exceedance rate is between 10% and 15% for domestic and shallow private and public supply wells, but only 5% for all public supply wells. In the northern SV, the Eastside subbasin is considered more vulnerable than the Pressure subbasin due to overlying agricultural activities and the lack of a confining layer. There,

Adminsitrative Record
Page 35042

domestic wells and local water supply wells (less than five connections), but also public supply wells with less than 60 m (200 ft) to the top-of-screen have a somewhat higher MCL exceedance rate of 17% (not shown in Table 19). In comparison, the MCL exceedance rate in the Pressure subbasin specifically is 12% for domestic and local supply wells and 0% for shallow public supply wells (not shown in Table 19). Shallow private wells known to have less than 60 m (200 ft) to the top-of-screen in either of these two subbasins have an exceedance rate of about 14%.

For further statistical analysis, we simplify the depth classification into two groups: we compare the group of wells known to either be a household or local small systems supply well or known to have less than 60 m (200 ft) to the top of the well screen ("shallow" wells), against the group of wells that either have more than 60 m (200 ft) to the top of the well screen or are not designated as monitoring, household, or local small systems wells ("non-shallow" wells). Wells that have neither any depth information nor any well use classification are not included in this comparison. The mean of the decadal well means of the log of nitrate are significantly higher in the "shallow" wells than in the "non-shallow" wells (Table 20). Even after spatial de-clustering, distinctly higher means of nitrate are obtained in the "shallow"equal area zones of each groundwater region than in the "non-shallow" equal area zones (Table 20).

Adminsitrative Record
Page 35043

Table 19. Median nitrate and exceedance probability of annual well means as a function of well depth and location on nitrate concentration.[1]

Region	Well Depth Category	# of Wells	# Well-Years	Median	> 9 mg/L	> 22.5 mg/L	> 45 mg/L	> 90 mg/L
TLB Eastside Fans	Monitoring	298	553	53.8	77.0%	66.0%	52.8%	37.6%
	Domestic	1,749	2,879	27.4	75.3%	55.9%	33.0%	9.6%
	<200', priv.	785	1,143	27.2	78.8%	56.6%	31.7%	7.5%
	<200', all	1,241	4,682	16.5	69.1%	38.1%	15.4%	2.8%
	public	1,597	11,867	12.5	62.0%	26.1%	5.7%	0.8%
TLB Central Basin	Monitoring	114	323	122.2	74.0%	66.6%	61.3%	56.7%
	Domestic	257	387	21.3	63.8%	47.8%	26.9%	8.0%
	<200', priv.	70	71	19.0	67.6%	42.3%	23.9%	4.2%
	<200', all	118	404	16.5	67.3%	34.2%	14.6%	3.7%
	public	148	913	8.0	47.0%	17.1%	6.7%	1.5%
TLB Westside Fans	Monitoring	29	61	62.0	62.3%	59.0%	52.5%	45.9%
	<200', priv.	2	2	24.1	50.0%	50.0%	0.0%	0.0%
	<200', all	3	9	1.5	11.1%	11.1%	0.0%	0.0%
	public	77	189	1.5	28.0%	6.3%	0.0%	0.0%
SV-Pressure Aquifer, Eastside, and Monterey Bay	Monitoring	170	570	25.6	60.0%	51.2%	43.0%	27.9%
	Domestic	530	1,970	16.0	65.0%	38.3%	14.0%	2.0%
	<200', priv.	108	458	16.0	65.3%	37.6%	13.1%	2.2%
	<200', all	146	678	14.6	63.6%	36.3%	10.0%	1.5%
	public	270	1,511	8.1	48.6%	21.6%	4.6%	0.3%
SV-Forebay and Upper Valley	Monitoring	10	22	77.2	81.8%	72.7%	59.1%	45.5%
	Domestic	33	105	22.0	67.6%	49.5%	40.0%	22.9%
	<200', priv.	62	84	36.3	61.9%	56.0%	42.9%	21.4%
	<200', all	79	193	10.0	52.8%	34.2%	22.3%	9.8%
	public	57	296	7.0	42.9%	18.9%	4.4%	0.3%

[1] Monitoring wells ("Monitoring") are typically measuring the shallow-most groundwater (upper 15 m – 30 m [50 ft – 100 ft]). Domestic wells ("Domestic") are typically screened in the upper 100 m [330 ft], but can be as deep as 150 m [500 ft], rarely more. Some wells have an actual or estimated record of screen depth and length. Here we selected wells with the top of the screen at less than 60 m [200 ft], not including public supply wells ("<200', priv."); those that include public supply wells, often located in urban areas ("< 200' "); and finally all public supply wells and a few deep irrigation wells in Westlands Water District ("public").

Adminsitrative Record
Page 35044

Table 20. Mean and 95% confidence interval, by groundwater basin and well depth category, of the mean of decadal well means of the log of nitrate.[1]

Groundwater Region	Depth	Number of Wells *(# equal areas)*	-95% CI	Mean	+95% CI
TLB Eastside Alluvial Fans	shallow	2,294 (*95*)	13.9 (*11.4*)	14.8 (*14.0*)	15.7 (*17.2*)
TLB Eastside Alluvial Fans	deep	5,804 (*119*)	11.4 (*7.7*)	11.9 (*9.5*)	12.4 (*11.8*)
TLB Central Basins	shallow	317 (*33*)	9.6 (*2.4*)	11.6 (*4.1*)	14.1 (*7.2*)
TLB Central Basins	deep	956 (*46*)	6.4 (*1.7*)	7.3 (*2.9*)	8.3 (*4.7*)
TLB Westside Alluvial Fans	shallow	3 (*3*)	0.1 (*0.1*)	6.2 (*6.2*)	741.9 (*741.9*)
TLB Westside Alluvial Fans	deep	86 (*29*)	3.0 (*2.2*)	4.2 (*4.1*)	5.9 (*7.9*)
Monterey Bay	shallow	462 (*31*)	8.6 (*7.8*)	9.7 (*10.8*)	10.8 (*14.9*)
Monterey Bay	deep	612 (*34*)	6.8 (*4.0*)	7.6 (*5.9*)	8.5 (*8.7*)
Salinas Valley	shallow	106 (*23*)	12.5 (*8.1*)	17.0 (*15.0*)	23.2 (*27.6*)
Salinas Valley	deep	67 (*21*)	5.6 (*5.5*)	8.4 (*7.8*)	12.6 (*11.0*)

[1] In parentheses, for comparison, are the de-clustered data, obtained from the mean of the log-transformed equal area medians, where the medians are obtained from all annual well means. Data shown here are back-transformed from their logarithmic values and shown in [mg/L] nitrate. Includes all wells sampled in the 2000s, except environmental monitoring wells.

Adminsitrative Record
Page 35045

5.4 Effect of Soil Type and Vadose Zone Thickness

To examine the effect of soil type and depth to groundwater, we categorize the location of wells by the groundwater protection zones that the Department of Pesticide Regulations (DPR) has designated[10] to protect against groundwater contamination. Groundwater protection zones occur only where depth to groundwater is less than 20 m (70 ft) and where soils are either highly permeable ("Leaching" zones) or where a shallow hardpan occurs ("Runoff" zones).

In the TLB, concentrations are significantly higher within groundwater protection zones than outside of groundwater protection zones (Table 21). Wells in "Runoff" zones and "Runoff or Leaching" zones have significantly higher nitrate than those in "Leaching" zones. In "Runoff" zones of the TLB central basin and the TLB Eastside alluvial fans, more than one third (33%) of wells (not including monitoring wells) exceeds the MCL, while only 12% – 16% of wells outside of DPR groundwater protection zones exceed the MCL in these areas of the TLB (Table 21).

In contrast, fewer than 40 out of over 1,200 wells in the SV are within groundwater protection zones and no significant difference is observed between nitrate in wells located within DPR groundwater protection zones and those outside of those protection zones.

Table 21. Median nitrate concentration and exceedance probability in wells, grouped by Department of Pesticide Regulation (DPR) groundwater protection zone. Includes all wells sampled in the 2000s, except environmental monitoring wells.

Region	DPR Groundwater Protection Zone	# of Wells	# Well-Years	Median	> 9 mg/L	> 22.5 mg/L	> 45 mg/L	> 90 mg/L
TLB Eastside Fans	(Outside)	6,661	17,770	13.3	61.4%	33.9%	15.7%	5.0%
	Leaching	647	2,330	16.6	69.4%	40.4%	14.0%	3.8%
	Runoff	814	2,626	32.3	86.3%	64.0%	35.3%	9.7%
	Runoff or Leaching	40	140	20.9	86.4%	45.7%	21.4%	10.0%
TLB Central Basin	(Outside)	903	2,013	6.0	41.6%	24.3%	11.5%	2.8%
	Leaching	7	23	17.0	82.6%	39.1%	39.1%	17.4%
	Runoff	390	800	33.1	88.0%	60.8%	37.5%	13.4%
	Runoff or Leaching	8	19	50.0	89.5%	57.9%	57.9%	26.3%
TLB Westside Fans	(Outside)	89	201	1.8	29.4%	9.0%	2.0%	1.5%
SV-	(Outside)	1091	4,716	14.0	61.2%	35.0%	11.0%	1.6%
Pressure Aquifer,	Leaching	21	73	12.0	61.6%	27.4%	9.6%	0.0%
Eastside, and Monterey Bay	Runoff	4	15	5.0	6.7%	0.0%	0.0%	0.0%
SV-Forebay	(Outside)	160	508	10.0	53.5%	34.1%	20.9%	10.6%
and Upper Valley	Leaching	15	39	12.0	59.0%	38.5%	20.5%	10.3%

[10] http://www.cdpr.ca.gov/docs/emon/grndwtr/index.htm

Adminsitrative Record
Page 35046

Given the small number of wells in DPR protection zones of the SV, statistical significance of DPR zones with respect to nitrate concentration in wells is tested only on the entire datase (wells measured in the 2000s, not including environmental monitoring wells) and not by groundwater region. We also consider wells in "Runoff or Leaching" zones as belonging into the "Runoff" category, since median nitrate values of that small group of wells resemble more closely those in the "Runoff" category than those in the "Leaching" category. Analysis of Variance (ANOVA) tests on the decadal well means of log nitrate show that the DPR zonation (outside, "Leaching," and "Runoff"), which represents the depth to groundwater and soil type classification, has a statistically highly significant influence on the fate of nitrate in groundwater. The back-transformed mean of decadal well means of log nitrate is 9.8 mg/L (9.4 – 10.1 mg/L) for wells outside of DPR protection zones; 12.1 mg/L (10.8-13.6 mg/L) in "Leaching" zones; and 29.4 mg/L (27.7 – 31.3 mg/L) in wells located in "Runoff" zones or "Runoff or Leaching" zones (parentheses indicate back-transformed 95% confidence intervals of means, Figure 47).

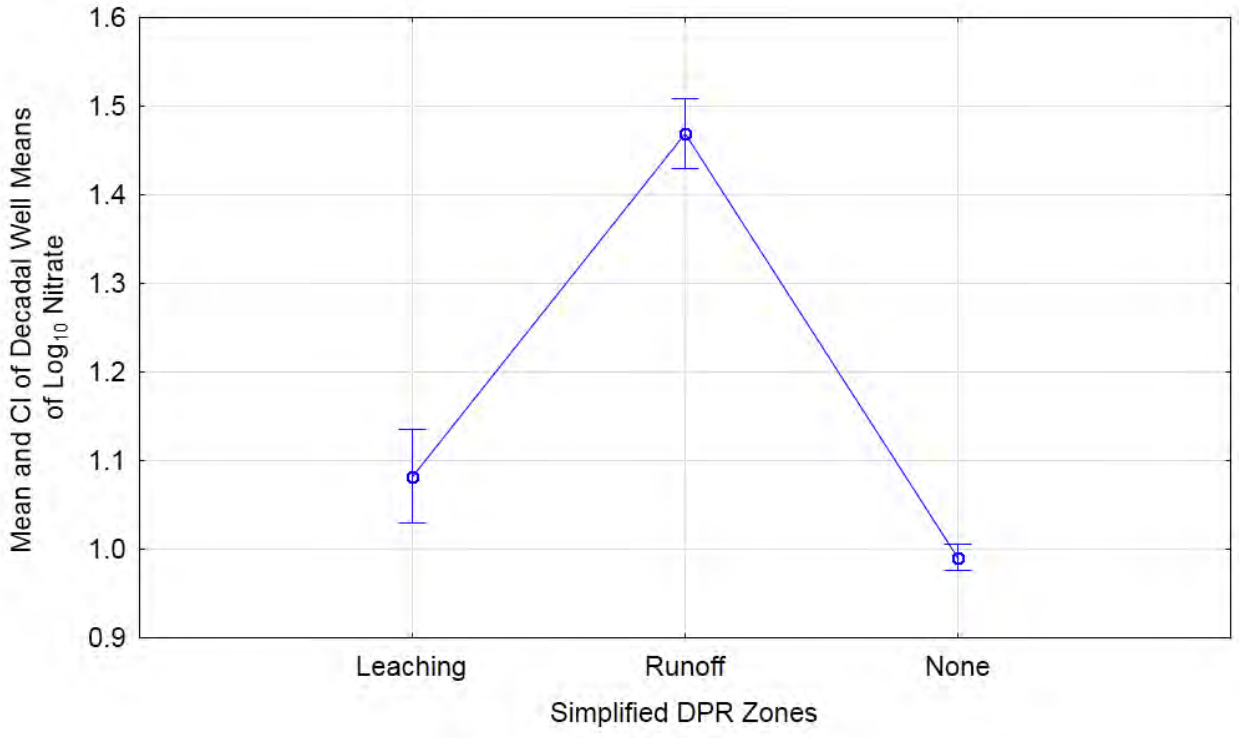

Figure 47. Means and 95% confidence interval of means for DPR groundwater protection zones. Here, "Runoff" refers to wells in either "Runoff" DPR zones or "Runoff or Leaching" DPR zones. Includes all wells sampled in the 2000s, except environmental monitoring wells.

Most "Leaching" zones are located in central Fresno County (south, southwest, and southeast of the city of Fresno) where the dominant crop is vineyards, some are located in the SV. Most "Runoff" zones are located in eastern Fresno County and Tulare County, which have a wide variety of crops. Kings and Kern County have few areas designated as protection zones. Vineyards (40%) and urban areas (13%) make up more than half of the "Leaching" zones, another 20% are deciduous fruit and nut. In contrast less than 20% of "Runoff" and "Runoff or Leaching zones" are vineyard (11%) or urban (8%). It is possible that the relatively lower nitrate concentration in "Leaching" zones, which are dominated by vineyard land use, is

Adminsitrative Record
Page 35047

due to relatively lower nitrogen use in vineyards, when compared to other crops (see Technical Report 2, Viers et al. 2012). Other potential causes for the difference between these three groups are: differences in temporal (time) delay in the transport of nitrate to groundwater through the vadose zone; different irrigation water use efficiencies due to presence/absence of hardpan or due to soil texture (light soil vs. heavy soils); and differences in denitrification rates between these soils.

Adminsitrative Record
Page 35048

5.5 Nitrate Above and Below the Corcoran Clay

The Corcoran Clay underlies the Central Valley from San Joaquin County to Kern County in the western two thirds of the Valley for most of that extent. In the study area, the depth to the Corcoran Clay varies from approximately 10 to 250 meters (32.8 to 820.2 feet) below ground surface. This aquitard unit is between 6 and 55 meters thick. Historically, the Corcoran Clay has been recognized as a regionally extensive barrier that prevents flow between an upper semi-confined aquifer system and a lower confined aquifer system. Recent (>1950's) development of groundwater in the region has likely altered the effect of the barrier due to the many large bore wells that have been drilled through the confining feature to access the lower aquifer (Panert 1995; Bertoldi 1991; Benito 2008). With strong pressure gradients present below and above such confining layers, abandoned and active wells likely function as conduits between groundwater systems which would otherwise be separated (see Viers et al. 2012, Section 9). Flow through the clay aquitard itself is subject to strong denitrification due to the anoxic conditions within the aquitard.

Although development of groundwater may have degraded the ability of the Corcoran Clay to prevent contamination of deeper groundwater from shallow groundwater, we test here at the regional scale if there is a significant difference between wells screened above and below the Corcoran Clay. Of the wells in the study area for which we were able to obtain screened interval values (both top and bottom of screened interval), 825 are screened below the Clay, and 642 are screened above it (Figure 48). CASTING contains 7,358 tests for these wells. Figure 49 shows the distribution of these samples, for wells screened above and below the Corcoran Clay, by decade.

Adminsitrative Record
Page 35049

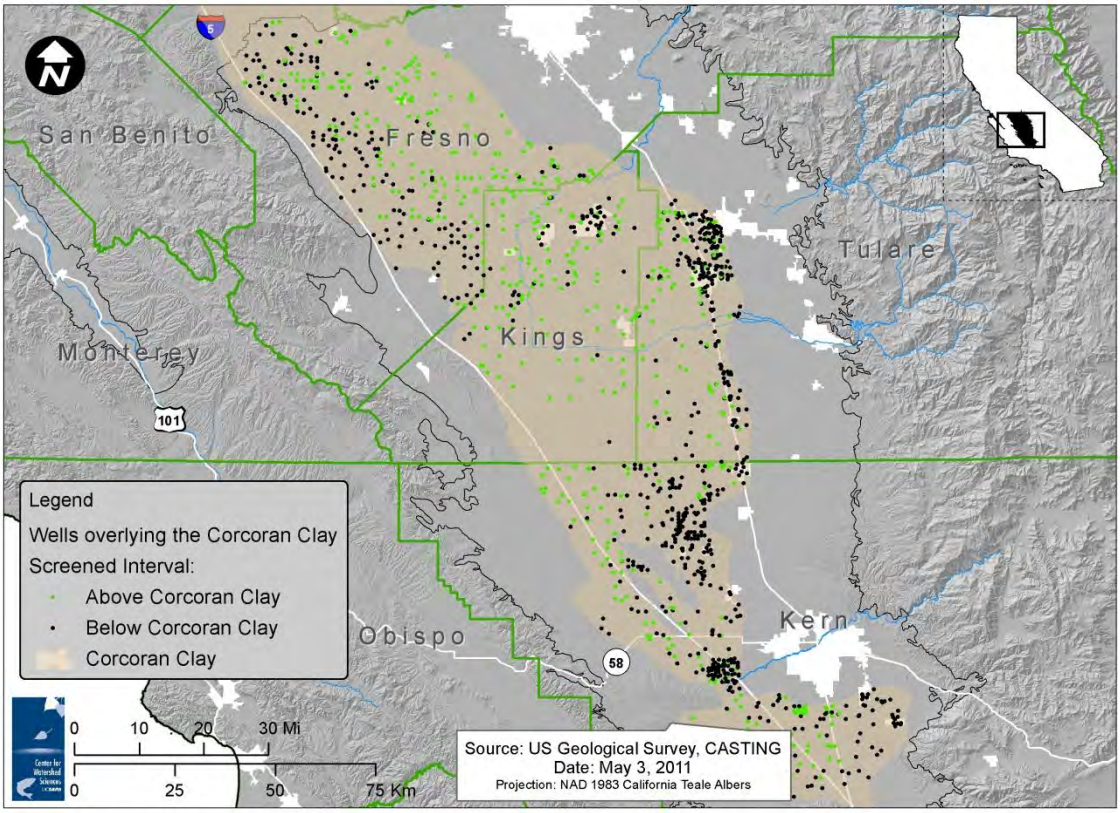

Figure 48. Locations of wells underlain by the Corcoran Clay, and the position of their screens relative to the Clay.

Adminsitrative Record
Page 35050

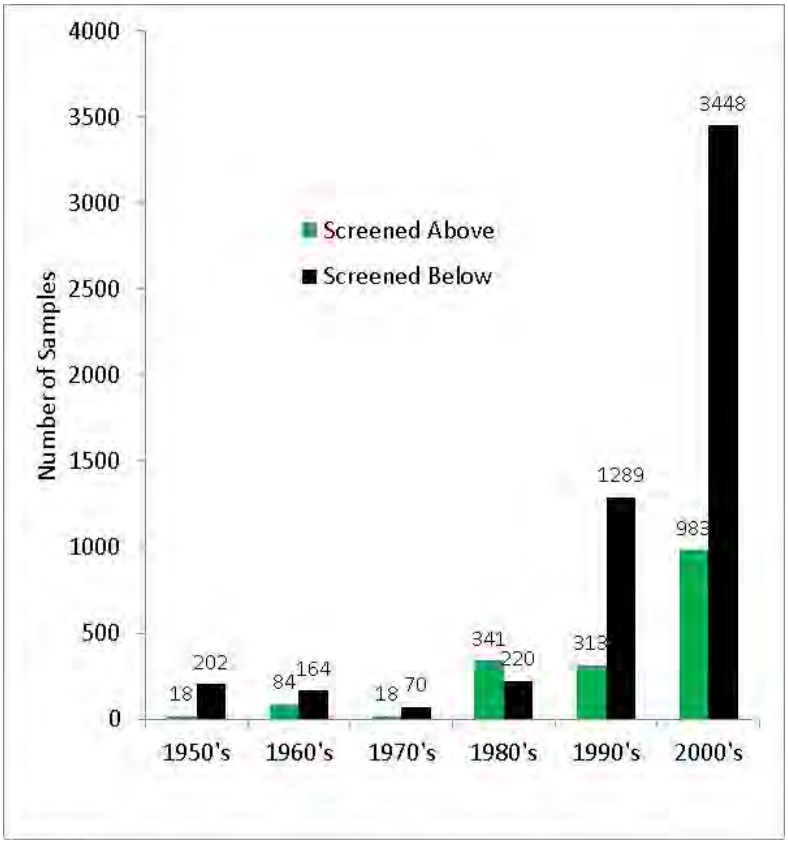

Figure 49. Number of nitrate samples each decade in wells screened above and below the Corcoran Clay.

The decadal median nitrate finding for each of the 1,467 wells with perforated interval information and within the horizontal boundaries of the Corcoran Clay was used for comparison between decades (1950's through 2000's) and between those wells screened above and those screened below the Clay. For each well, the median nitrate finding was found for each decade for which CASTING has nitrate data.

Each well is either screened above or below the Corcoran Clay, therefore direct comparison of the nitrate concentrations above and below the aquitard, in a single borehole, is not possible. To avoid introduction of bias, the area of the Corcoran Clay was divided into equal area hexagons (approximately 21,400 acres, or 8,660 hectares, each), such that the centroid of each is within the outline of the Corcoran Clay, as defined by Claudia Faunt in the Central Valley Hydraulic Model (Faunt 2009). Within each hexagon, for each decade, the median of the decadal median of nitrate concentration in wells above and below the Corcoran Clay was determined. The set of equal area medians above the Corcoran Clay was compared to the same set below, using the Mann-Whitney test for significance at $\alpha = 0.05$ (results in Table 22). This is a non-paired test, meaning that all of the above-clay findings were compared to all of the below-clay findings, including those hexagons that had only one or the other.

Adminsitrative Record
Page 35051

Table 22. Median values of the median of decadal well median nitrate concentration (mg/L as nitrate) in hexagonal equal area sample sets for wells screened above and below the Corcoran Clay, by decade. Non-paired comparison.

	Median NO_3 concentration by decade					
Screen Position	**2000's**	**1990's**	**1980's**	**1970's**	**1960's**	**1950's**
Above	7.1	1.4	15.0	6.7	1.2	1.1
Below	2.1	2.6	4.0	2.0	1.5	1.2
Significance at $\alpha = 0.05$	NO	NO	YES	NO	NO	NO

No significant difference between wells screened above and wells screened below the Corcoran Clay was found for any decade except the 1980's. Although the 1980's decade did produce a significant result, examination of the source data for that decade showed that the 94 highest nitrate concentration findings were all derived from the NWIS input dataset in the last 4 years of the decade. Each of these 94 findings was the result of a single test on a well, and none of these wells was ever tested before, or since. Unfortunately, NWIS does not provide any information regarding the project objectives for particular data points. As these nitrate concentrations were 1 to 2 orders of magnitude higher than all other nitrate values produced during that decade, excluding known environmental monitoring wells, it is likely that these were samples taken from wells specifically targeting high nitrate concentrations as part of a particular USGS study. Of the 442 samples collected in the 1980's above or below the Corcoran Clay, 301 were from the NWIS database.

From this non-paired analysis of the entire Corcoran Clay within the study area, we might infer that, at the regional scale, the Corcoran Clay does not separate aquifers of significantly different nitrate concentration. However, a paired analysis reveals more information.

Since some parts of the Corcoran Clay were sampled in some decades and not in others, the non-paired method presents an uncontrolled view in that each sample (hexagon) does not have a relevant comparison sample. Without pairing, the test compares values above the Corcoran Clay in one area to values below the Corcoran Clay in another area, and large-scale regional trends may bias those differences.

To find area-specific differences, those hexagons with values both, above and below the clay in a given decade were considered. Too few data were available from the 2010s to analyze that decade. The following series of figures shows the hexagonal equal area cells used in both the non-paired and paired analyses for the decades from 1950 to present (Figure 50 through Figure 55). Note that there are no cells for which data exist both above and below the Corcoran Clay for all decades.

Adminsitrative Record
Page 35052

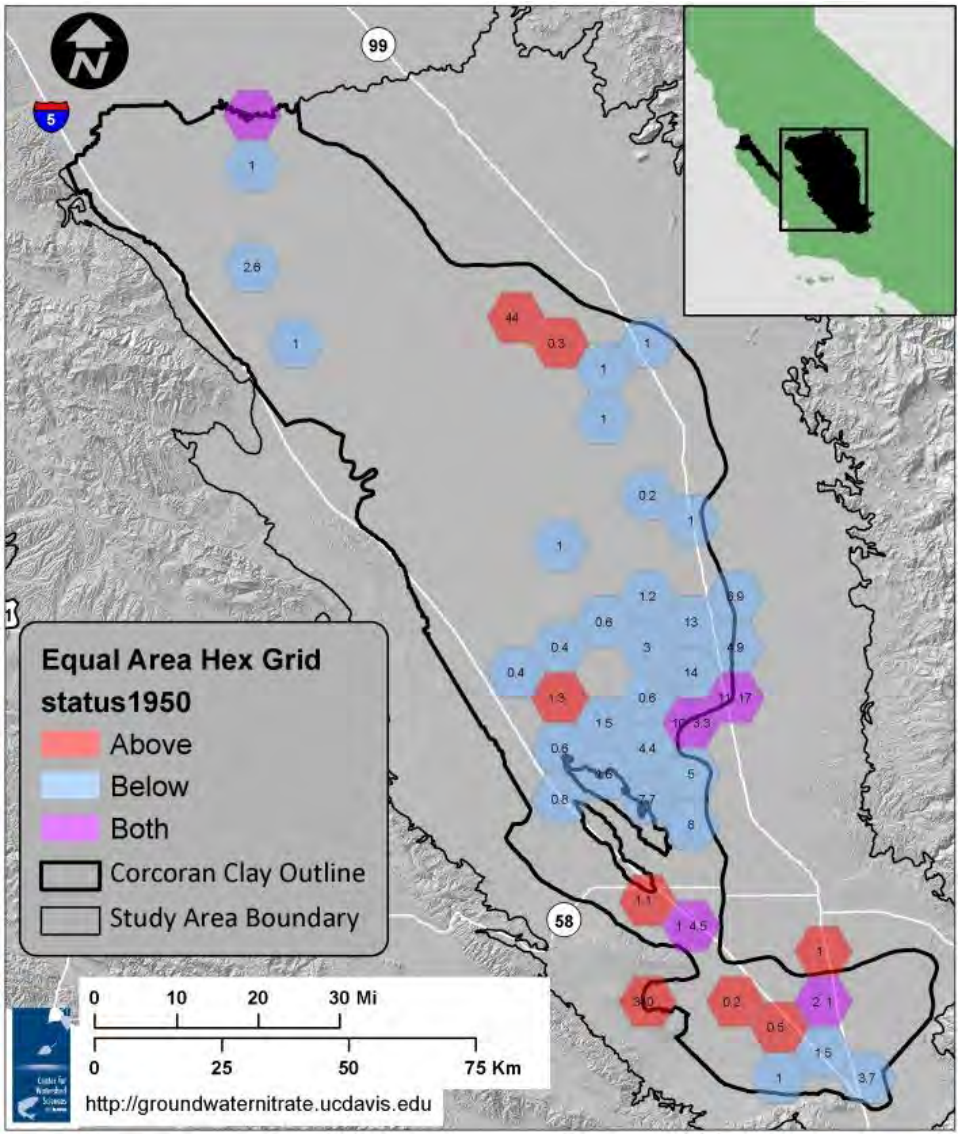

Figure 50. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1950's. Red hexes represent regions with data available from wells screened above the Clay only; blue represents areas with data from below the Clay; purple areas have data from above and below the Clay. The values printed on each hexagon indicate the median of all decadal median nitrate concentrations from the wells in that hexagon – in purple cells, the value on the left is from the above-clay wells and the value on the right is from the below-clay wells.

Adminsitrative Record
Page 35053

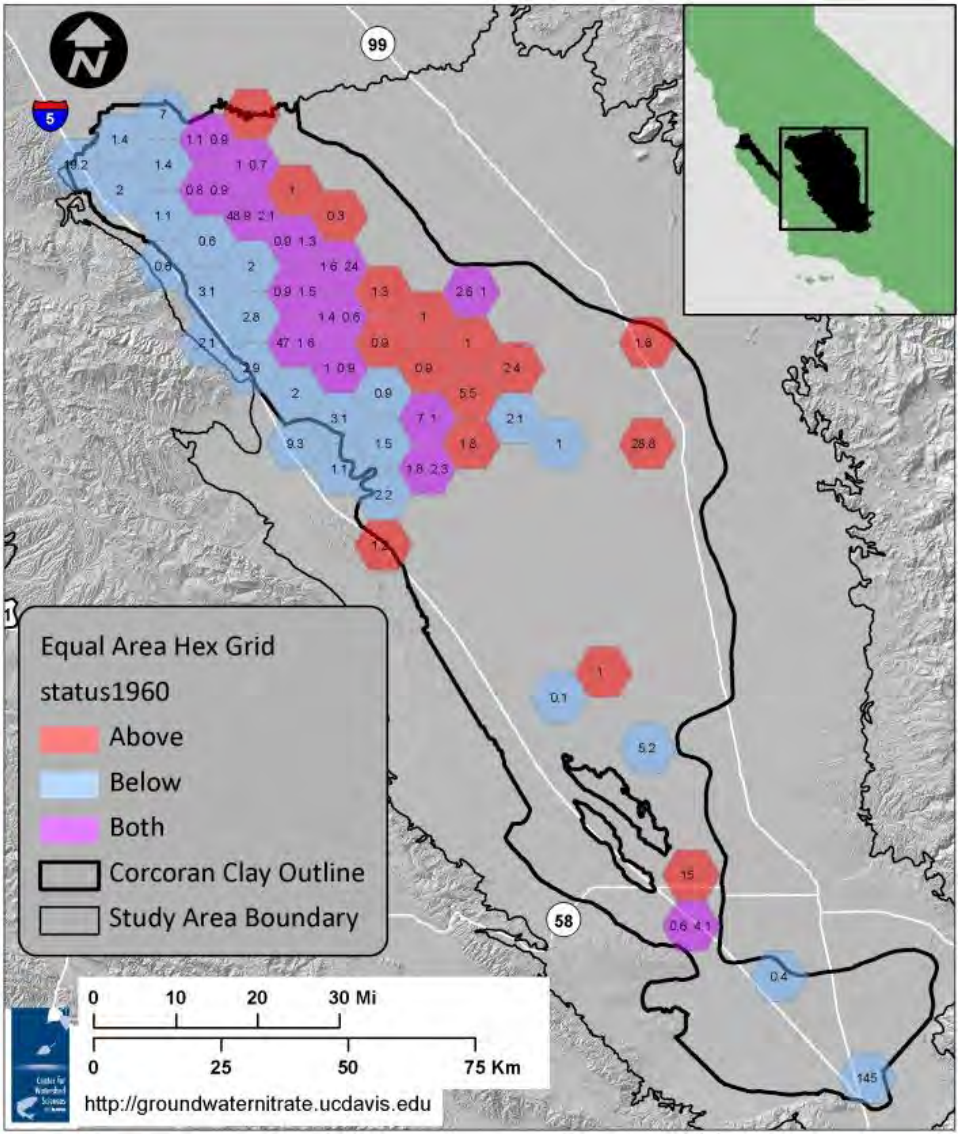

Figure 51. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1960's.

Adminsitrative Record
Page 35054

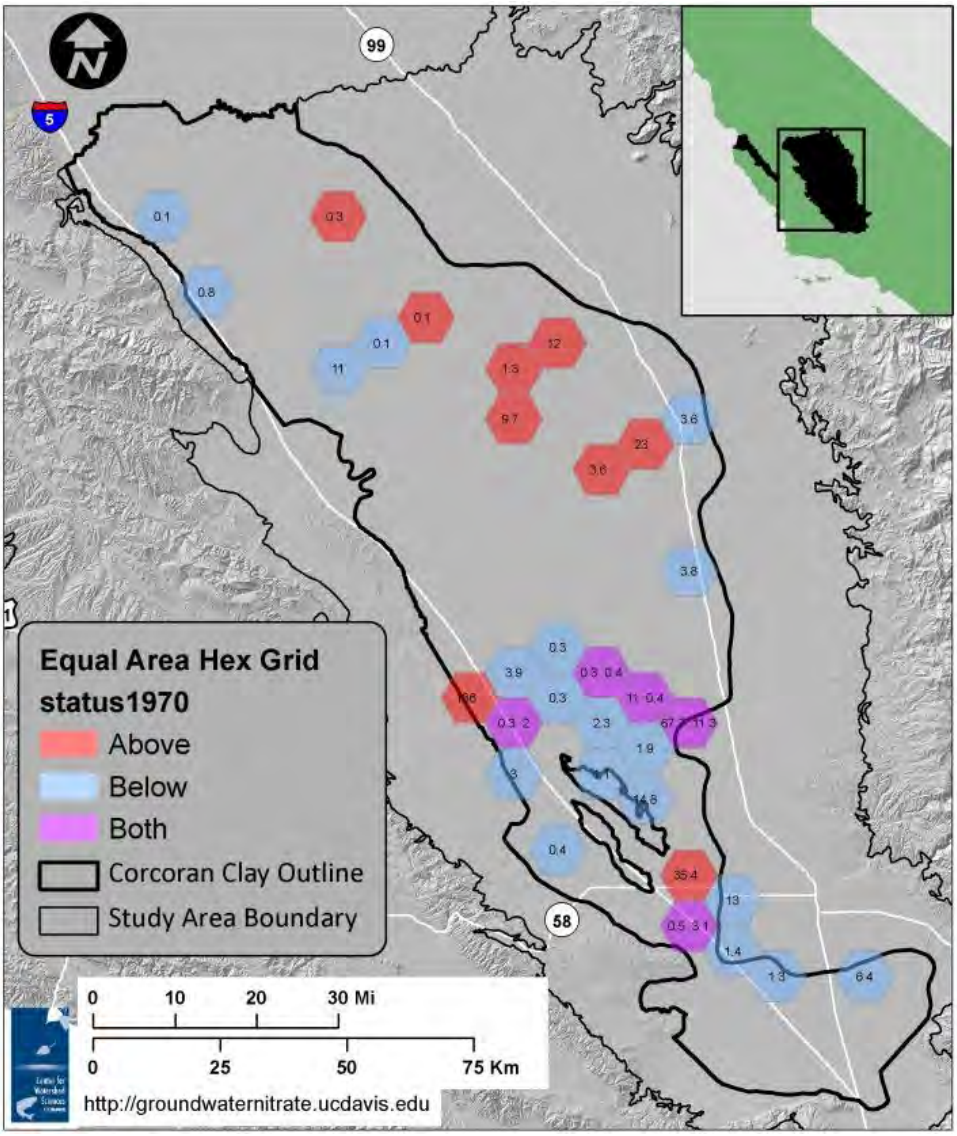

Figure 52. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1970's.

Adminsitrative Record
Page 35055

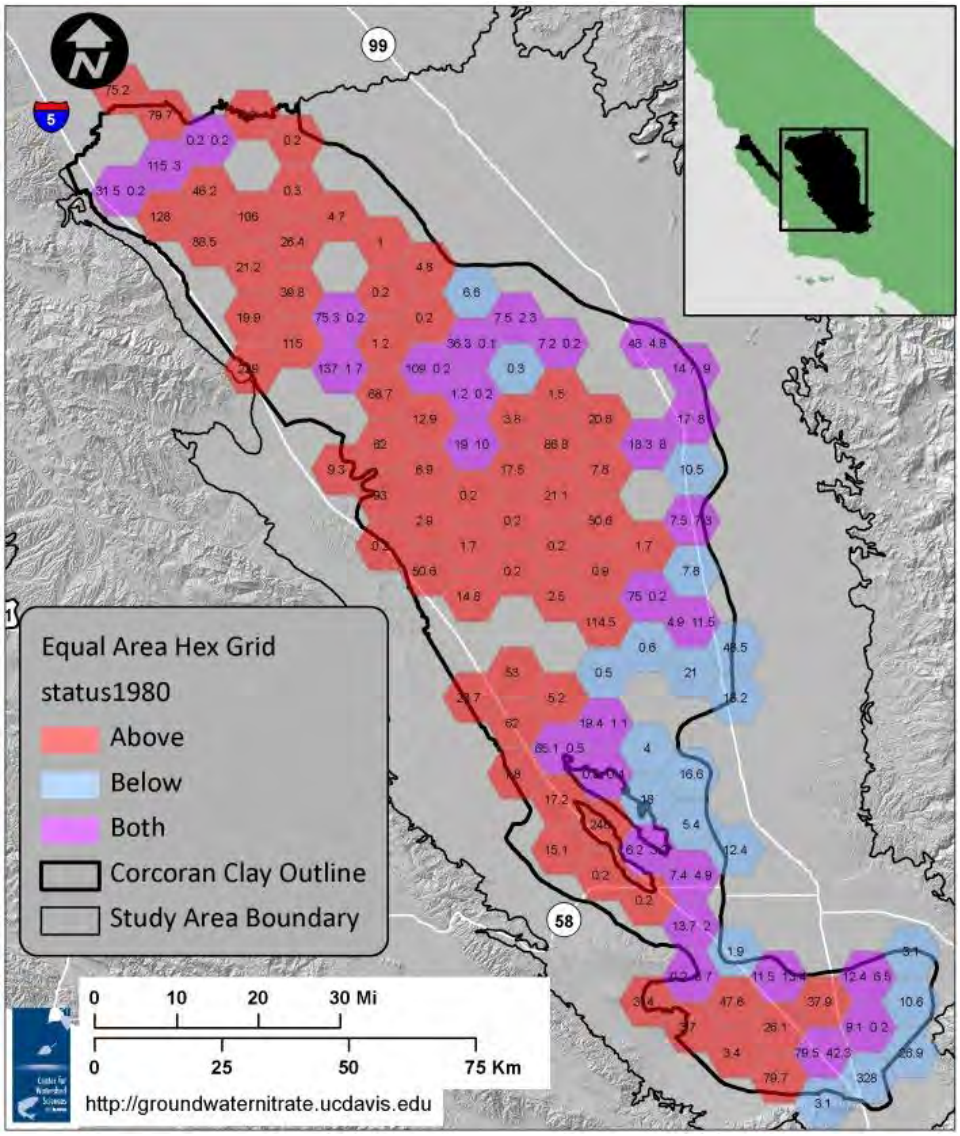

Figure 53. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1980's.

Adminsitrative Record
Page 35056

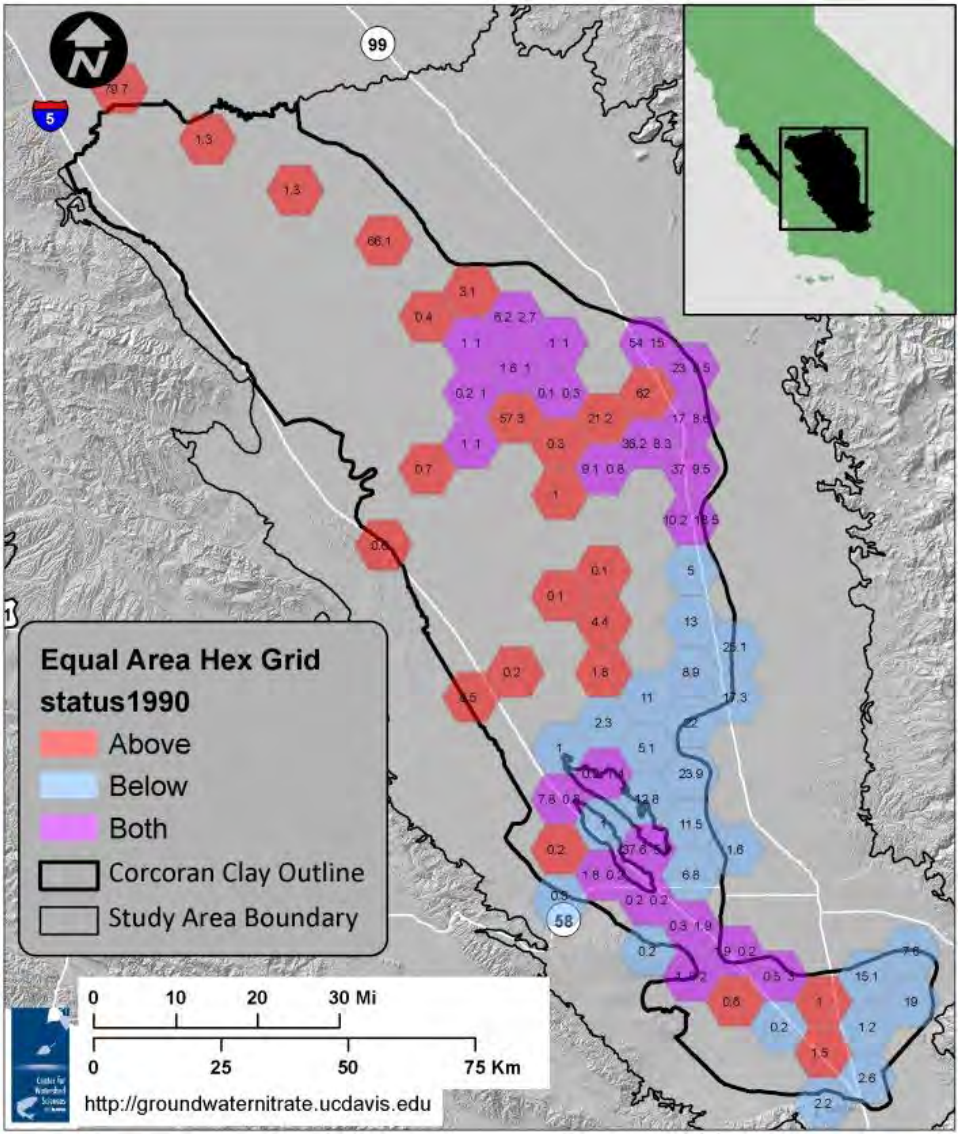

Figure 54. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1990's.

Adminsitrative Record
Page 35057

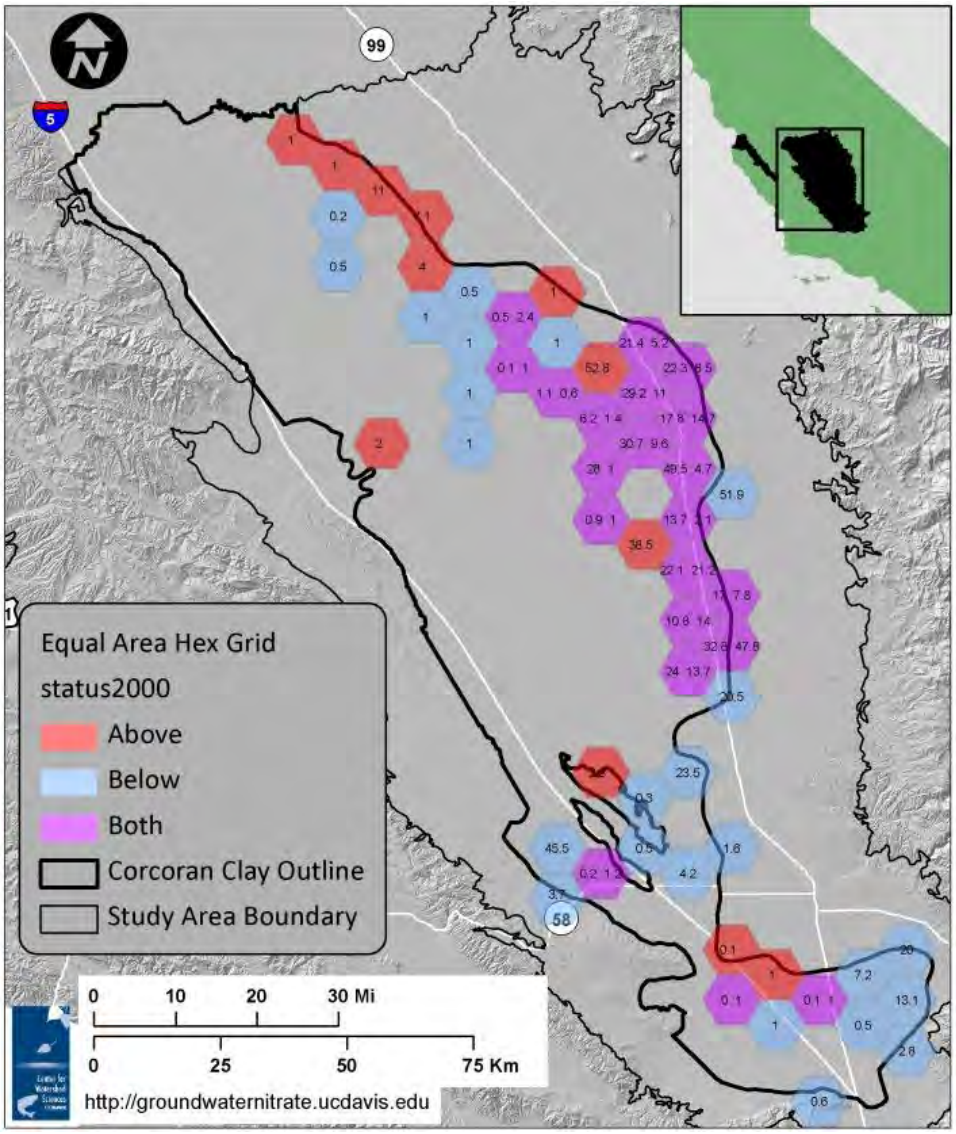

Figure 55. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 2000's.

For hexagon/decade combinations with values both above and below the Corcoran Clay, the below value was subtracted from the above value. A positive result implies that for that decade, in that hexagon, the nitrate concentration was higher above the Corcoran Clay than below it. Median values were used in this analysis as well.

The paired Wilcoxon rank sum test produced very different results from the non-paired analysis. In the 1950's, 1960's, and 1970's, no significant difference was found between the above and below datasets. However, in the 1980's, 1990's and 2000's, the above-clay concentrations were significantly higher than the below-clay data. Not counting the 1980's (due to the bias of the NWIS dataset), the difference in the mean nitrate above and below the Corcoran Clay in the recent two decades has been on the order

Adminsitrative Record
Page 35058

of 7 – 8 mg/L, nearly twice as much as the difference between “shallow” and “non-shallow” wells found across the study area (see previous section).

Table 23. Results of Paired-Sample Wilcoxon Signed Rank tests.

Decade	p-value	hex count	Above = Below ?	Mean of Above - Below
2000's	0.018	21	Reject	7.5
1990's	0.0076	20	Reject	7.1
1980's	0.000011	29	Reject	27.9
1970's	0.34	5	Do Not Reject	12.5
1960's	0.27	14	Do Not Reject	5.3
1950's	0.5	4	Do Not Reject	-0.4

Based on this analysis, the Corcoran Clay appears to be a significant water quality boundary. Lower nitrate concentration below the Corcoran Clay are the result of multiple factors: groundwater below the Corcoran Clay is significantly older than above the Corcoran Clay, significant portions of the confined aquifer below the Corcoran Clay are geochemically reducing and therefore favorable for in situ denitrification (Burow et al. 1998), and the aquitard represents a significant barrier to downward flow within the aquifer system. On a regional level, this analysis seems to confirm these notions.

On the other hand, a number of wells screened below the Corcoran Clay have nitrate concentrations above typical background levels (more than 9 – 18 mg/L). Also, we caution that a relatively small area of the Corcoran Clay is represented by this analysis because few equal areas, mostly near the eastern boundary of the Corcoran Clay, have sufficient data for a comparison of nitrate values above and below the Corcoran Clay (see Figures above). To the degree that we find nitrate concentrations above background levels in wells screened below the Corcoran Clay, even in the Basin and Westside Alluvial Fans, the dataset indicates that wells may be significant conduits of nitrate, from the upper aquifer system to the confined lower aquifer system below the Corcoran Clay.

Adminsitrative Record
Page 35059

5.6 Nitrate in Wells near Streams

In addition to the obvious groundwater quantity benefits, groundwater banking could become an important strategy for improving the regional groundwater quality. Figure 56 shows the average nitrate concentration found in pumping wells by Public Land Survey System (PLSS) section. The circles on the map highlight areas near rivers that have lower nitrate concentrations as compared to wells further away from the rivers. It is likely that these wells have relatively low nitrate due to the fact that the water they are withdrawing consists of a high fraction of uncontaminated river recharge, and a small amount of contaminated recharge via irrigation return water. Additionally, areas near rivers are often irrigated with low nitrate surface water. The resulting leachate from these crops is potentially less than that from crops grown further away from rivers as those fields are often irrigated with pumped groundwater containing elevated levels of nitrate.

Within the city of Fresno, south of the San Joaquin River, large amounts of surface water and storm water are intentionally recharged. Wells that are further from major rivers generally appear to have higher nitrate concentrations. The water withdrawn from these wells likely has a higher fraction of contaminated irrigation return water. At the basin scale, managing recharge by increasing the amount of uncontaminated recharge versus contaminated recharge through artificial recharge basins, as well as managed reservoir releases to maintain stream flow (and therefore stream recharge), has the potential to positively affect the long term trends in water quality within the TLB.

Adminsitrative Record
Page 35060

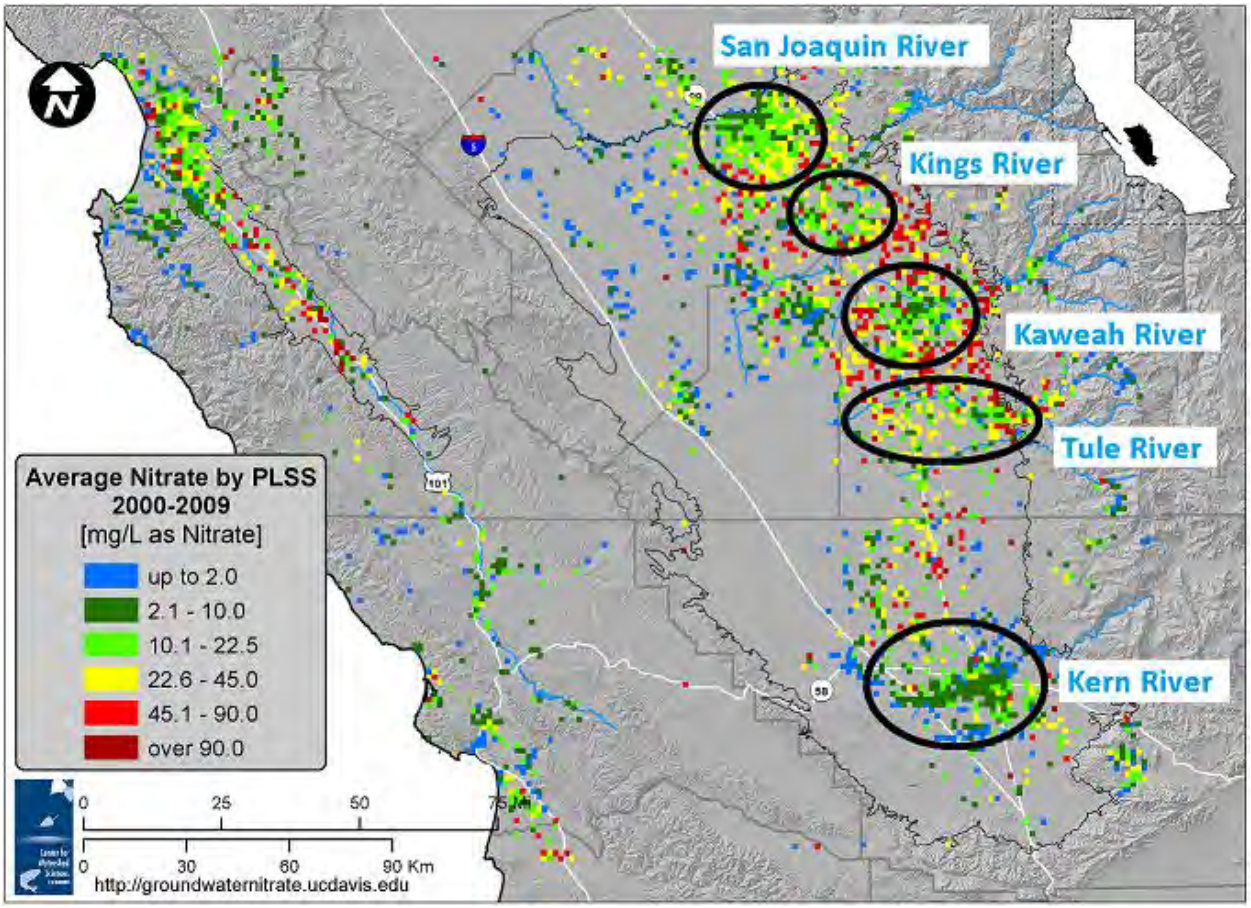

Figure 56. Average nitrate concentration by PLSS section. Circles highlight areas where major rivers enter the study area. These are also areas characterized by wells low in nitrate.

Adminsitrative Record
Page 35061

5.7 Historic Trends

The CASTING database represents a large dataset of primarily the current period (since 1/1/2000) with wide regional representation across the study area. CASTING also represents a view back in time. In the study area, only a few tens to a few hundred wells were tested each year for nitrate during much of the 1950s, 1960s, 1970s, and 1980s (Figure 57). Many of these wells, especially after the 1960s, represent testing in public water supply wells. In the late 1980s, the number of wells tested annually increased to over 500, exceeding 1,000 wells in the early 1990s, and exceeding 2,000 wells by the early 2000s. In 2007 – 2010, the dairy regulatory program in the TLB added about 4,000 wells, thereby more than doubling the number of wells tested annually (Figure 57). Over the next decade and beyond, the dairy dataset will provide an opportunity to obtain a much better understanding of groundwater nitrate and long-term trends in agricultural regions of the San Joaquin Valley and TLB, if implemented properly. Currently missing information includes consistent sampling protocols, collection of well construction information (particularly screen location), and consistent data management.

For the period from the 1950s to current, public water supply wells are the only set of wells with long-term records of groundwater nitrate, although those wells exceeding the MCL will often be abandoned or destroyed, which potentially introduces a statistical bias into a trend analysis. Apart from public supply wells (and the recent dairy well program), only the DPR dataset on 71 domestic wells in Fresno and Tulare County, operated for the last decade, has consistently sampled from the same set of wells. Many wells have been sampled once or few times, not allowing for long-term trend analysis. Monterey County Water Resouces Agency administers a confidential set of private well samples monitored for nitrate over the past two decades. Only selected published records, funded from public grants, have been available for this study.

First, we consider the long-term changes in exceedance probabilities across the entire set of wells sampled each year, whether these are sampled once or multiple times. Figure 57 shows that the fraction of well samples exceeding the background threshold level of 9 mg/L has steadily increased over the past sixty years, albeit with significant interannual variations. Exceedance rates have increased from about one-third of wells in the 1950s to more than two-thirds of wells exceeding the background level in the late 2000s. In the late 1970s and 1980s, the exceedance rate seemed to hold steady at about 55%. That was also during a period with relatively few samples taken (Figure 57).

The half-MCL (22.5 mg/L) threshold exceedance rate increased from near 10% in the 1950s to about 30% in the early 1980s, with a spike in the late 1960s. During the 1980s and 1990s, the exceedance rate held steady, but began to increase again in the late 1990s, with a large jump in the late 2000s, when the exceedance rate increased to 45%. On the other hand, the exceedance rate of the MCL has held relatively steady ranging from 5% to 15% over the second half of the 20th century, but increasing significantly since the late 1990s.

The uneven distribution of samples across space, time, and data sources (monitoring purposes), and the relatively low number of samples prior to 1990, make it very difficult to obtain a highly accurate estimate of long-term trends in the study area that also accounts for differences due to variation in land

Adminsitrative Record
Page 35062

use, soils, vadose zone thickness, well depth and aquifer stratigraphy, and other factors potentially influencing nitrate concentrations in wells. Furthermore, most of the samples taken prior to 2000 are focused on urban and peri-urban areas of the study area. Any analysis of long-term trends must therefore be carefully conducted and interpreted. Here we perform a number of additional tests to further elucidate and interpret apparent trends seen in Figure 57.

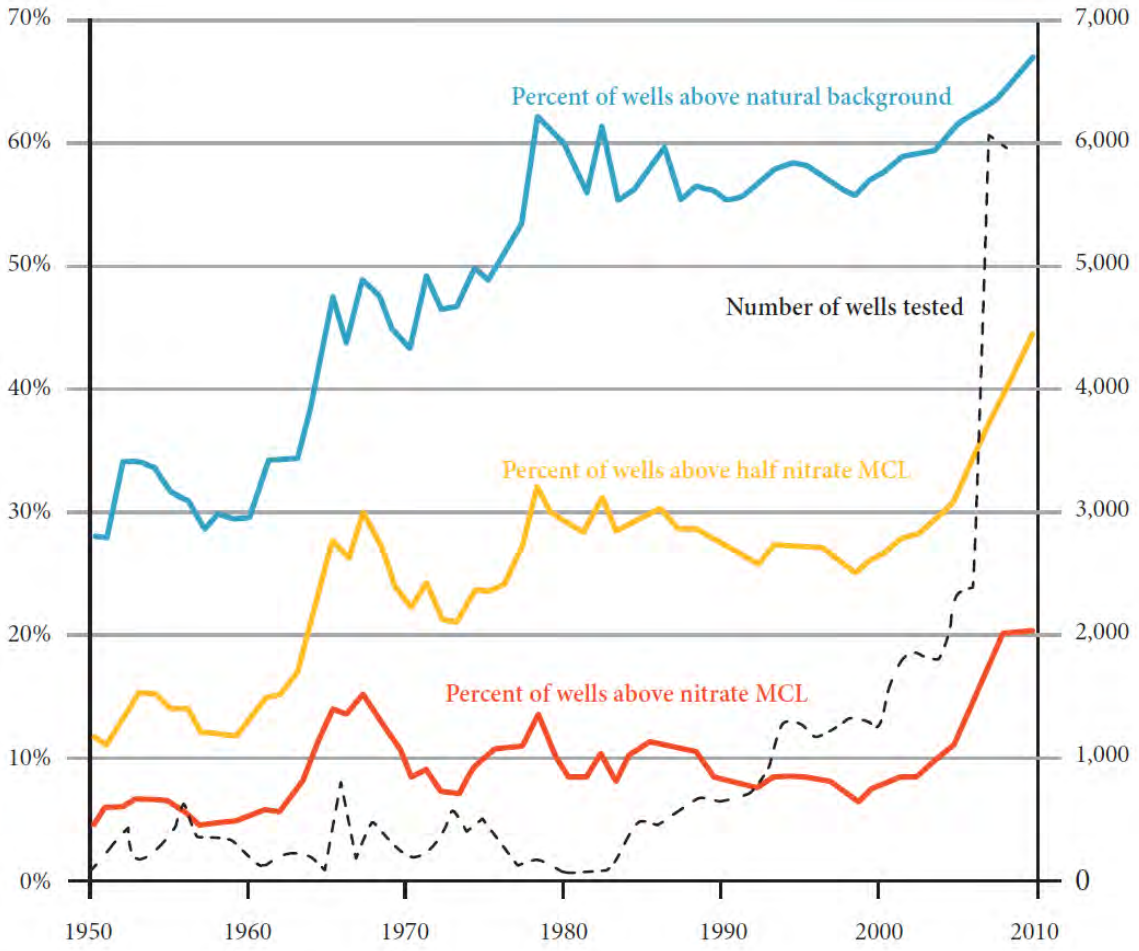

Figure 57. Five-year moving average of the percentage of wells for which the average annual measured concentration exceeded 9 mg/L (background), 22.5 mg/L (half of the MCL), and 45 mg/L (MCL) in any given year. Since the 1990s, an increasing number of wells other than public supply wells have been tested. In 2007, Central Valley dairies began testing their domestic and irrigation wells on an annual basis.

For an investigation of historical trends, we again consider the five physiographic groundwater regions of the study area described above and use de-clustered nitrate values to illustrate the development of nitrate concentration distribution over the past sixty years. We first consider the median and exceedance probabilities of annual well means in each equal area and each decade, not including environmental monitoring wells (see Section 5.1 for a description of equal areas). The medians are log-normally distributed across equal areas. We calculated the mean and confidence interval of the log of medians in each of the five groundwater regions, for each decade (Figure 58). We computed the mean

Adminsitrative Record
Page 35063

across all equal areas of each equal area's exceedance probability, based on annual well means (Figure 59) and also computed 95% confidence intervals of these means (Figures 58, 59).

Increases in mean nitrate concentrations are observed mostly in the Eastside Alluvial Fans region of the TLB and the Monterey Bay region of the Salinas Valley (Figure 58). Between the 1950s and the 2000s, with all areas equally weighted, mean nitrate levels have increased about 7 mg/L. The 1980s saw significant spikes in mean nitrate in the TLB Basins, TLB Westside Alluvial Fans, and the Salinas Valley regions. However, the number of samples taken during the 1980s was relatively low, and the confidence interval is wide. In the TLB, much of the high nitrate samples collected in the 1980s were associated with a sampling program archived in the USGS NWIS dataset.

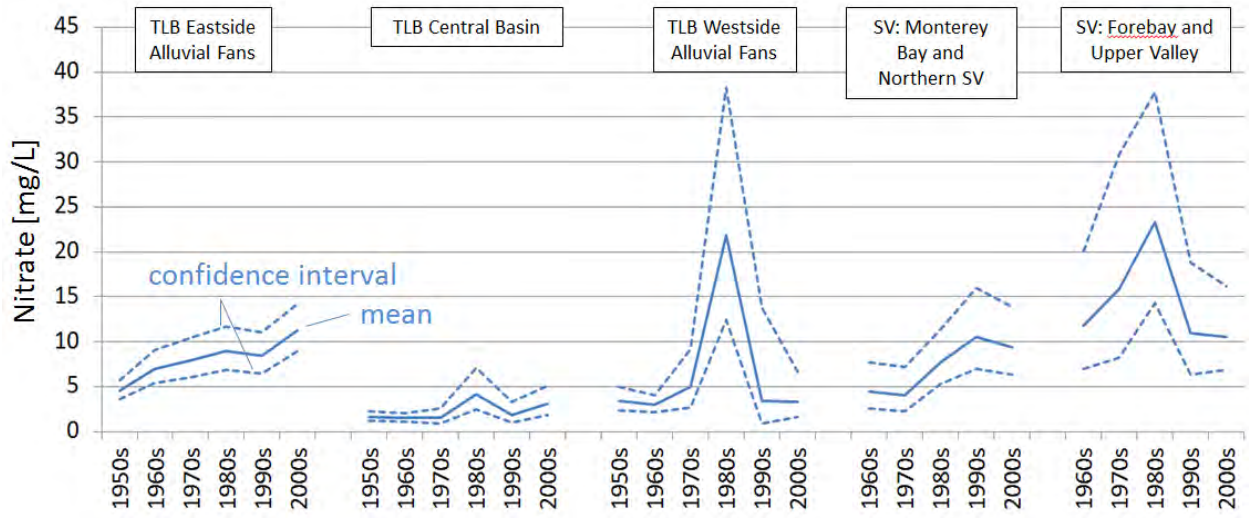

Figure 58. De-clustered, back-transformed mean of the logarithm of equal area decadal medians that were computed from ten years of annual well means. From left to right: TLB Eastside Alluvial Fans, TLB Central Basin, TLB Westside Alluvial Fans, Northern SV and Monterey Bay, SV Forebay and Upper Valley. Confidence intervals of the mean are at the 95% level (back-transformed).

Exceedance probabilities for all three thresholds considered (9 mg/L, 22.5 mg/L, and the MCL at 45 mg/L) show long-term increases in the TLB Eastside, the TLB Basin, and the Monterey Bay and Northern Salinas Valley groundwater regions. Again, the spike of high nitrate samples collected in the 1980s is apparent, if not as strong as in the medians (Figure 59). The decadal changes seem to indicate a decrease in exceedance probabilities between the 1990s and 2000s for both Monterey County regions (Monterey Bay and Salinas Valley, Figure 59). A more detailed look at annual (rather than decadal) medians in each equal area of those two regions (approximately log-normal distributed) reveals that the decrease is mostly limited to the mid-1990s, when sample size was relatively small (and confidence intervals are wide). During the 2000s, no significant decrease in median nitrate concentration was observed in the two Monterey County regions (Figure 60) with a potential upward trend in the past five years in both regions.

Adminsitrative Record
Page 35064

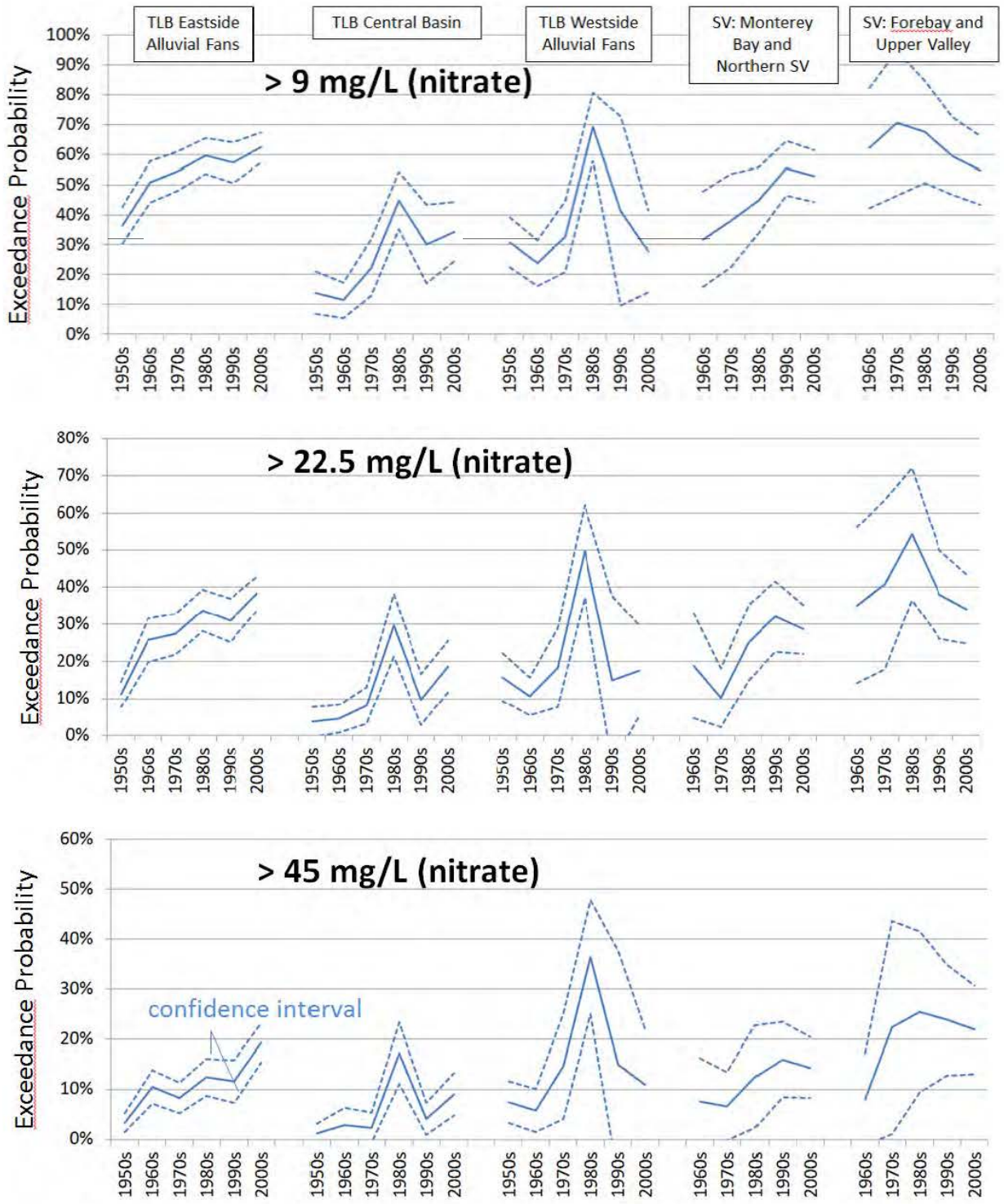

Figure 59. De-clustered mean of the equal area decadal exceedance probablility computed from ten years worth of annual well means in each equal area. From left to right: TLB Eastside Alluvial Fans, TLB Central Basin, TLB Westside Alluvial Fans, Northern SV and Monterey Bay, SV Forebay and Upper Valley. From top to bottom: exceedance probability of nitrate above 9 mg/L (top), nitrate above 22.5 mg/L (center), and nitrate above the MCL of 45 mg/L (bottom). Confidence intervals of the mean are at the 95% level.

Adminsitrative Record
Page 35065

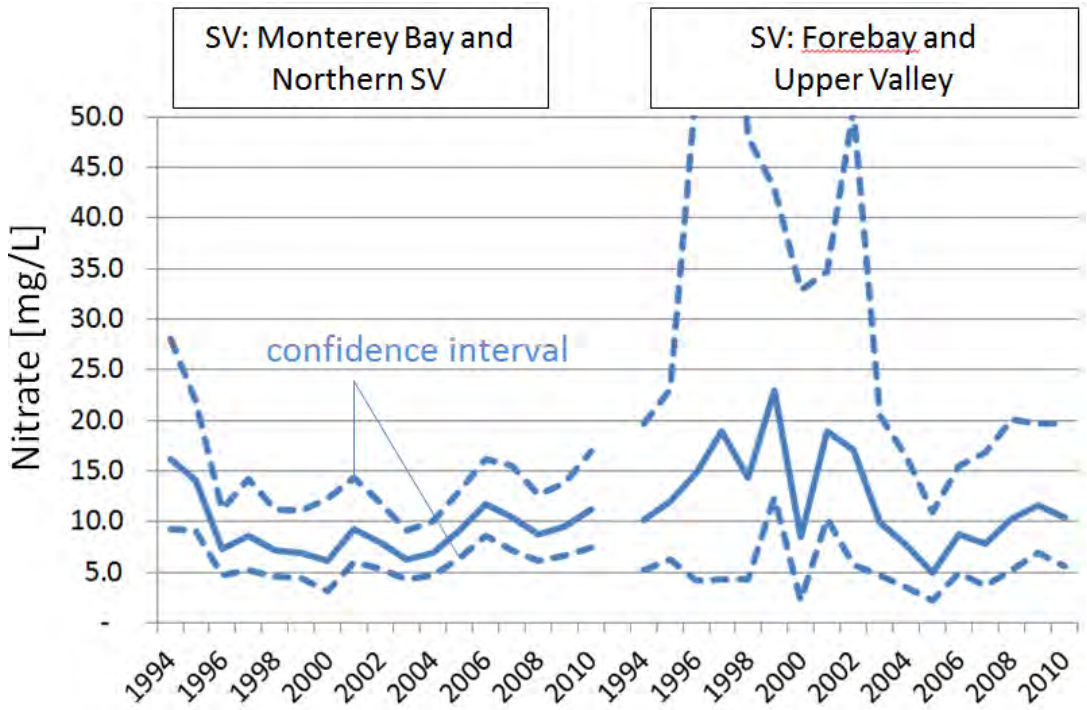

Figure 60. De-clustered back-transformed annual mean of the logarithm of equal area annual medians that were computed for one year of annual means of each measured well. Monterey Bay and Northern Salinas Valley (left), Salinas Valley Forebay and Upper Valley (right). Confidence intervals of the mean are at the 95% level (back-transformed).

To quantitatively assess the changes in nitrate levels through time, a regional statistical test called the "Regional Kendall" (RK) test (Helsel & Frans 2006) was used to test for long-term nitrate concentration trends in the study area. The RK tests whether a regional (or temporal) trend is present in a dataset, and whether that trend is significant or not. In the RK test, the tau value is a correlation coefficient that ranges from -1 to +1. A value of +1 or -1 would indicate a perfect positive or negative correlation. A zero would signify no correlation. The p-value represents the probability that the null hypothesis is true, where the null hypothesis that there is no significant trend in the data. Typically a p value of 0.01 or 0.05 (signifying 99% confidence and 95% confidence) are used as a threshold to conclude whether or not the trend is significant. For this trend analysis, a 99% confidence threshold was used to signify a significant trend ($p \leq 0.01$). The RK test operates on concentration differences between all pairs of nitrate samples taken over time from the same well (or equal area), for wells (or equal areas) with at least two measurements (e.g., each year, each decade, etc.). The test is based on the counts of sample pairs with increasing concentration trend and on the counts of pairs with decreasing concentration trend over time.

To test for trend, the Regional Kendall test was performed on the yearly medians of nitrate for the equal area cells (Helsel & Frans 2006). First, the median value of all tests within a single year for a given well was found. Next, the median value of all wells for each year, for each equal area cell was found. The

Adminsitrative Record
Page 35066

result is a time series of nitrate data for each equal area cell, where each cell has at most one value per year (if no data were available for a particular year within a cell it would have no value for that year). Given the uneven distribution of samples over time, the dataset was broken into two time periods: 1949–1999 and 2000–2011. The analysis on the first time period represents historical trends while the analysis on the second time period represents recent trends. The results of the RK test are shown in Table 24.

Three tests resulted in a p-value of greater than 0.01 (no significant trend): the TLB Westside Alluvial Fans area for the years 2000–2011, and for both periods of the "Salinas Valley" region of the SV. All other regions and time periods resulted in positive trends that were statistically significant. All significant trends show that nitrate is increasing, with some areas increasing faster than others. For the TLB, the Eastside Alluvial Fans region was found to be increasing the fastest. The granitic and generally coarse sediment coming from the Sierra Nevada Mountains provides for rapid transport of water applied at the surface to the groundwater below. The TLB Westside Alluvial Fans, and the TLB Basin sediments are characteristically much finer compared to the TLB Eastside Alluvial Fans sediments, and we see slower rates of nitrate increases in these regions.

The greatest trends for the study area were observed in the Monterey Bay and TLB Eastside Fans regions. From 2000–2011, the RK test shows that the median values of the equal area cells were increasing by 0.4 (mg/L)/year in the Monterey Bay region, and 0.19 (mg/L)/year in the TLB Eastside Fans region. The Salinas Valley sub-region (Forebay and Upper Valley subbasins) did not result in any significant trend. River recharge is greatest in this portion of the SV. Aquifer conditions also range from semi-confined to unconfined aquifer conditions. This setting likely creates a very heterogeneous system which might obfuscate a trend from showing in the analysis.

The values shown are also very small in magnitude. It is important to remember that the test was performed on the median values of a dataset consisting of mostly drinking water wells. The test is insensitive to outliers (e.g., contaminated wells) and is therefore more a measure of the general water quality of the region than any trend observed in individual wells. A positive trend documents that nitrate concentrations for the region as a whole are increasing, and that nitrate contamination of wells is more of a regional process rather than a localized phenomenon. The RK test results are in agreement with the more qualitative observations made on the long-term changes in nitrate concentration shown in Figure 58, Figure 59, and Figure 60. The magnitude of the upward trend is also consistent with the observed changes in mean nitrate, particularly in the TLB Eastside Alluvial Fans region and the Monterey Bay and Northern Salinas Valley region.

Adminsitrative Record
Page 35067

Table 24. Regional Kendall test results on the de-clustered equal area statistics for nitrate. Trends shown in blue are statistically not significant.

	Tau	P	Annual Change in NO_3^-
TLB Eastside Fans, 2000-2011	0.148	0.000	0.1900
TLB Eastside Fans, 1949-1999	0.136	0.0000	0.07561
TLB Basin, 2000-2010	0.141	0.0011	0.01214
TLB Basin, 1950-1999	0.117	0.0000	0.02146
TLB Westside Fans, 2000-2010	0.089	0.3496	0.000
TLB Westside Fans, 1949-1999	0.097	0.0029	0.03984
Monterey Bay, 2000-2011	0.175	0.0011	0.4000
Monterey Bay, 1962-1999	0.163	0.0001	0.1164
Salinas Valley, 2000-2011	-0.143	0.1472	-0.8750
Salinas Valley, 1962-1999	0.097	0.1392	0.1750

To further illustrate the long-term trend of nitrate values in wells, we pursue an investigation of the statistics of the average trend in nitrate values at each well that has at least two measurement data. We investigate the average trend at each well over the entire period of observation and, separately, for each decade of observation. To obtain an average trend of nitrate concentration at a well, we performed a simple linear regression between measured nitrate values and time for the entire period over which a well was sampled and, separately, for each decade at which a well was sampled at least twice. The regression was performed using a least-square estimation procedure to obtain the regression slope. The regression slope was used as a measure of the average trend in nitrate at a given well over the time period of interest. The regression slope expresses average nitrate changes in a well in units of concentration change per year [mg/L/yr]. Here, we refer to the regression slope at a well as the "well trend" or the "decadal well trend," if the regression slope was performed for a specific decade.

Well trends and decadal well trends are distributed nearly symmetrically around zero with a large number of samples showing very small or zero gradients. The number of non-zero well trends (positive or negative) decreases hyper-exponentially as the absolute value of the well trend (or decadal well trend) increases. We computed the mean and its 95% confidence interval, the median, and the lower and upper quartiles of the well trends across all wells within each groundwater region (Table 25). Data were analyzed separately for the major groundwater regions. Well trends and decadal well trends larger than 20 mg/L/yr or smaller than 20 mg/L/yr were considered outliers and not included in the analysis. Results of the statistical analysis are shown in Table 25.

The mean of well slopes over the entire period of measurement is positive in all five regions. The lower confidence interval for the mean slope in all five regions is also larger than zero indicating that the mean is significantly different from zero. This confirms the results of the RK test and would suggest that even the Salinas Valley (no significant trend according to the RK test), over the entire period of measurement, had an increasing trend in nitrate concentrations.

For the study area as a whole, the average well trend is 0.34 mg/L/yr. Half of all wells have a trend larger than 0.08 mg/L and 25% of all wells have a long-term trend of more than 1.11 mg/L/yr.

Adminsitrative Record
Page 35068

In the TLB, the Eastside Alluvial Fans region and the Basins region have the fastest increasing well trend. The mean in the Eastside Fans area is at 0.3 mg/L/yr; there, half of all wells increase at a rate higher than 0.08 mg/L/yr and one-quarter of all wells increase at a rate higher than 1.25 mg/L/yr. In the Basins region, the mean well trend is 0.45 mg/L/yr (median: 0.03 mg/L/yr, upper quartile: 1.22 mg/L/yr). Lower rates of increase are found for the Basins regions, where the mean well trend is 0.16 mg/L/yr (median: 0 mg/L/yr). A quarter of wells there exceed a rate of increase of 0.5 mg/L/yr.

Adminsitrative Record
Page 35069

Table 25. Statistics of the least-square estimates of the regression slope (mg/L per year) of nitrate concentrations versus time for each well with at least two sample values, grouped by groundwater region and decade.[1]

GW Region	Period	Number of Slopes	CI -95%	Mean Slope	CI +95%	Median Slope	Lower Quartile	Upper Quartile
Study Area	**total period**	7,694	0.23	0.34	0.44	0.08	-0.40	1.11
TLB Eastside Fans	**total period**	5,128	0.15	0.30	0.44	0.09	-0.48	1.25
	1950s	226	-0.74	-0.15	0.43	0.00	-0.49	0.78
	1960s	190	-0.72	0.05	0.83	0.01	-1.36	2.12
	1970s	366	0.01	0.33	0.66	0.13	-0.46	1.36
	1980s	521	-0.48	-0.14	0.20	0.00	-1.13	1.04
	1990s	1,142	-0.16	-0.01	0.14	0.00	-0.59	0.49
	2000s	3,756	0.14	0.32	0.51	0.09	-0.79	1.61
TLB Basin	**total period**	850	0.08	0.45	0.81	0.03	-0.29	1.22
	1950s	92	-0.50	0.08	0.65	0.00	-0.29	0.23
	1960s	63	-1.15	-0.50	0.15	0.00	-0.58	0.40
	1970s	33	-1.99	-0.14	1.71	0.00	-0.07	0.61
	1980s	31	-1.70	-0.40	0.90	0.08	-0.25	0.59
	1990s	81	-0.56	0.13	0.83	0.02	-0.19	0.50
	2000s	575	0.28	0.80	1.31	0.22	-0.73	2.74
TLB Westside Fans	**total period**	298	-0.33	0.16	0.65	0.00	-0.59	0.50
	1950s	140	-1.30	-0.46	0.37	0.00	-1.16	0.45
	1960s	57	-0.97	0.16	1.29	0.08	-0.40	0.75
	1970s	9	-0.67	-0.06	0.56	-0.39	-0.70	0.24
	1980s	11	-1.77	2.03	5.83	0.60	0.00	5.86
	1990s	8	-1.81	-0.31	1.18	0.03	-1.31	0.94
	2000s	42	-2.13	-0.14	1.84	-0.11	-2.23	1.21
Monterey Bay, Pressure Aquifer, and SV Eastside	**total period**	1,185	0.07	0.25	0.42	0.07	-0.22	0.72
	1960s	39	-0.41	0.44	1.29	0.19	0.02	0.63
	1980s	35	-1.46	0.58	2.62	0.00	-0.94	0.99
	1990s	220	-0.99	-0.32	0.35	0.00	-2.45	1.35
	2000s	556	0.16	0.44	0.71	0.13	-0.21	0.95
Salinas Valley, Forebay and Upper Valley	**total period**	1,024	0.13	0.29	0.45	0.04	-0.26	0.79
	1990s	233	0.85	1.50	2.14	0.54	-0.22	3.78
	2000s	20	-0.24	0.76	1.76	0.45	-0.07	1.56

[1] Regression slopes for each well are computed for the entire period of records ("total period") and, separately, for each decade with at least two measurements on one well. Environmental monitoring wells are excluded.

Adminsitrative Record
Page 35070

In the SV, the Monterey Bay region experiences long-term nitrate increases similar to those in the eastern TLB, as expected from the RK test and the change in average regional nitrate levels, discussed above. The mean well trend is 0.25 mg/L/yr with half of all wells exceeding 0.07 mg/L/yr and one quarter of all wells increasing at a rate of 0.72 mg/L/yr or more. In contrast, the Forebay and Upper Valley subbasins of the SV have a mean rate of nitrate increase of 1.5 mg/L/yr, half of all wells increase at a rate of 0.54 mg/L/yr or more, one quarter of all well trends are 3.78 mg/L/yr or more.

Table 25 also lists the mean, medians, and quartiles of the decadal well trends, within each region. When analyzed by groundwater region and decade, half of all wells or more have increasing decadal well trends (median is zero or greater than zero), except in few cases (TLB Westside in the 1970s and 2000s, and Salinas Valley region in the 1970s).

Specifically for the most recent decade (the 2000s), mean decadal well trends were 0.3, 0.8, -0.14, 0.29, and -0.36 mg/L/yr for the TLB Eastside, TLB Basin, TLB Westside, Monterey Bay, and Salinas Valley regions. The two negative trends are associated with the by far smallest sample sets (42 wells in the TLB Westside region and 116 wells in the Salinas Valley) when compared to the other three regions. The decadal well trends are consistent with the RK test and with the changes in average nitrate across wells observed in these regions (Figure 58 and Figure 59). In the five regions, for the last decade, half of all wells (median) had decadal well trends of 0.09, 0.22, -0.11, 0.04, and 0.00 mg/L/yr, or more, respectively. One quarter of all wells (upper quartile) in these five regions had nitrate concentration increases at a rate of 1.61, 2.74, 1.21, 0.79, and 0.83 mg/L/yr or more, respectively.

The study area has four major continuous nitrate monitoring programs: public supply wells (since the 1980s); a private and confidential monitoring program in the SV administered by the Monterey County Water Resources Agency (MCWRA) since the 1990s; the Department of Pesticide Regulations (DPR) domestic well monitoring program in groundwater protection zones of Fresno and Tulare Counties; and – most recently – the dairy monitoring program for which CASTING contains the 2007 through 2009 records.

For comparison, we performed the well trend and decadal well trend analysis separately for these datasets (Table 26). The number of public supply wells with multiple data steadily increased from less than 600 in the 1980s to 1,300 in the 1990s and nearly 2,000 in the last decade. The number of wells monitored in the other programs is relatively steady over time, although we only had access to some published MCWRA records. The long-term mean well trend in public water supply wells, in the MCWRA monitoring wells, and (albeit over only 3 years) in the Dairy Program wells was comparable at 0.27, 0.41, and 0.45 mg/L/yr, respectively. Within each monitoring program, half of all wells (median) had trends of 0.08, 0.07, and 0.11 mg/L/yr or more, respectively; and one-quarter of wells (upper quartile) increased at a rate of 0.54, 3.32, and 3.78 mg/L/yr or more. The mean upward trend is significantly different from zero for the public supply wells and for the dairy wells, but not for the MCWRA monitoring well program, which is only about one-tenth in size when compared to either the public water supply well or dairy well sampling program.

Adminsitrative Record
Page 35071

The 69 domestic wells in the DPR program, on average, have a decreasing well trend of -0.27 mg/L/yr. More than half of the wells (median) have a decreasing trend (the median is -0.32 mg/L/yr). One quarter of all wells (lower quartile) decrease at rates of -1.46 mg/L/yr or faster, while one quarter of wells (upper quartile) increase at rates of 1.08 mg/L/yr or faster. About half of these wells are located in "runoff" protection zones, while the other half is located in "leaching" protection zones. Well trends in both zones cover a similar range. Hence, the type of protection zone does not appear to affect the rate of change in nitrate concentration.

Table 26. Statistics of the least-square estimates of the regression slope (mg/L per year) of nitrate concentrations versus time for each well with at least two sample values, grouped by data collection programs and decade.[1]

Data Source / Collection Program	Period	Number of Slopes	CI -95%	Mean Slope	CI +95%	Median Slope	Lower Quartile	Upper Quartile
Public Supply Wells (CADWSAP)	**total period**	2,190	0.18	0.27	0.35	0.08	-0.14	0.54
	1980s	558	-0.20	0.11	0.41	0.02	-0.92	1.24
	1990s	1,311	-0.10	0.04	0.18	0.00	-0.56	0.50
	2000s	1,927	0.21	0.31	0.41	0.11	-0.18	0.66
MCWRA Monitoring Program	**total period**	206	-0.57	0.41	1.38	0.07	-2.67	3.32
	1990s	171	-0.06	1.12	2.30	0.55	-2.12	5.14
	2000s	59	-2.62	-0.64	1.35	0.00	-5.89	2.77
DPR Domestic Wells – all	2000s	69	-0.91	-0.27	0.38	-0.32	-1.46	1.08
DPR - Leaching zones only	2000s	30	-1.33	-0.20	0.92	-0.47	-1.17	1.08
DPR - Runoff zones only	2000s	39	-1.12	-0.32	0.48	-0.31	-1.49	1.12
Dairy General Order	2000s	2,600	0.17	0.45	0.73	0.11	-2.35	3.78

[1] Regression slopes for each well are computed for the entire period of records ("total period") and, separately, for each decade with at least two measurements on one well. Environmental monitoring wells are excluded.

In summary, on average across the study area, groundwater nitrate concentrations are increasing at rates of about 1 to 10 mg/L per decade, particularly in the eastern TLB and in the northern Salinas Valley. The RK test provides the statistically most robust measure of groundwater nitrate trends. The trends observed in the RK test are consistent with our statistical analysis of well trends in the five groundwater regions, and within the four existing continuous groundwater nitrate programs. Locally (in a well), nitrate concentrations may vary significantly over time. Some wells experience a downward trend in nitrate, while other wells experience an upward trend in nitrate. The variability in the dynamics of nitrate at wells is very large. Not all areas of the study area are seeing groundwater nitrate increases. At the TLB Westside and in the Forebay and Upper Valley of the SV, trends in groundwater nitrate are ambiguous.

Adminsitrative Record
Page 35072

6 Groundwater Nitrate Forecasting: Assessment of Vadose Zone Nitrate Transport

Prepared by:

Dylan Boyle, Thomas Harter

Adminsitrative Record
Page 35073

6.1 Introduction

Best Management Practices (BMPs) in agriculture are developed for a number of reasons, often with the intent of minimizing the release of chemical pollutants to surface and groundwater. A key challenge in evaluating BMPs, in the context of groundwater contamination, is the lag time from when a BMP is enacted, and when its effects can be seen in the water withdrawn from a well. The goal of Section 6 is to show spatially, areas within the TLB and SV where the effect of BMPs will be observed the soonest at the water table, and to identify which areas will experience significant lag. Although heterogeneity can cause preferential flow through the vadose zone (Harter et al. 2005), resolving detailed soil structure for 2D and 3D simulations is not feasible at the scale of thousands of square miles. In light of this, three maps were created based on three homogeneous soil types: sand, loam, and clay soil, representing the quickest, intermediate, and slowest probable travel times of nitrate to the water table, respectively. HYDRUS 1D was used to model travel times to the water table by specifying daily leachate fluxes, depth to the water table, and soil type. Fluxes of agricultural return water were determined by mass balance using the differences between calculated evapotranspiration (ET) from a field and the amount of water applied through natural precipitation and irrigation (including various irrigation technologies and their associated efficiencies). We show that travel time is essentially driven by the amount of water that infiltrates past the root zone of a crop, the depth of the water table, and the hydraulic properties of a soil. The hydraulic conductivity of unsaturated soils is a function of their water content, and therefore the rate in which water (and dissolved solutes) travels in the subsurface is influenced by the quantity of water infiltrating past the root zone of a crop. Factors such as irrigation efficiency, annual precipitation, and crop evapotranspiration contribute to this flux.

First, a small modeling exercise will show that annual water budget information is sufficient for estimating travel time. Six representative crops grown in the study area (alfalfa, citrus, cotton, almonds, corn, and grain) are used in a 1D soil column model to simulate solute travel time of a conservative (non-reactive and non-sorbing) solute to the water table. Daily water budgets for each crop are calculated to determine the amount of water leaching past the root zone. These fluxes are used to estimate travel time to the water table using the software HYDRUS 1D, a flow and transport model for unsaturated soil. This initial study shows that travel time to the water table can be approximated from annual water budget information for a particular field. The results from the modeling are then applied to agricultural fields in the SV and TLB to produce regional maps of nitrate travel time to the water table (figures 64-66). The modeling results provide a helpful tool for regional planners by distinguishing where adjusted BMPs can have a relatively quick impact to water quality at the water table, and areas where a significant lag will take place.

Adminsitrative Record
Page 35074

6.2 Modeling with HYDRUS 1D

6.2.1 Calculating Leachate Fluxes

The first step in calculating leachate fluxes is determining crop ET, the amount of water needed by a specific crop. Daily water needs for a crop are calculated by using a standardized crop coefficient (Kc) multiplied by reference evapotranspiration (ETo)(Snyder et al. 1989). In California, evapotranspiration from a standard/well maintained grass (reference crop) is monitored at hundreds of locations and data are provided through the California Irrigation Management Information System (CIMIS). Multiplying the ETo by a crop coefficient gives a daily estimate of the water used by a crop through evapotranspiration, storage in the tissue, and evaporation from the adjacent soil. Annual crop coefficient curves, which provide a daily Kc value for a crop, are shown in Figure 61 for the representative crop cycles chosen. Crop coefficient curves were generated from methods outlined in University of California's Cooperative Extension *Leaflet 21427* (Snyder et al. 1989) for a typical annual growing season. For portions of the year during which a particular crop is not being cultivated, an estimated "crop coefficient" for bare soil was used (Snyder et al. 1989). It is evident from the figure that each crop requires different amounts of water at different times of the year. The crop coefficient curves were combined with average monthly ETo and average monthly precipitation for the Visalia area, in the central-eastern TLB, to determine irrigation requirements for the crop cycles.

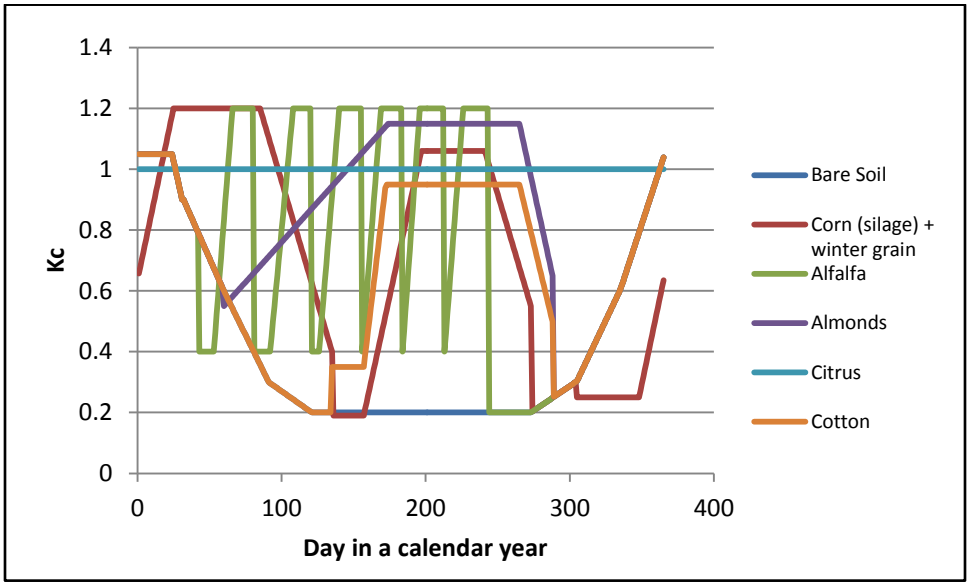

Figure 61. Annual crop coefficient curves.

Irrigated fields generally require more water than will be transpired by the plant. The two main reasons for this are the need to prevent salt buildup at the root zone, and to accommodate irrigation inefficiencies (non-uniform application of water). This results in water traveling past the root zone into the deeper vadose zone. Calculation of this flux is done by subtracting the amount of water transpired,

Adminsitrative Record
Page 35075

held in the plant tissue, and evaporated from adjacent soils from the amount of water applied via irrigation and natural precipitation.

For each day of the year, one the following calculations was made for each crop:

Precip - (ETo x Kc) + ((ETo x Kc x LR) - Precip) / IrrEff = Daily Leachate under Irrigation (Eqn. 2)

Precip - (ETo x Kc) = Daily Leachate without Irrigation (Eqn. 3)

ETo is the potential evapotranspiration, Kc is the crop coefficient, LR is the leaching requirement needed to control salt buildup in the root zone, IrrEff is the irrigation efficiency, and Precip is the daily average precipitation. Daily Leachate is the amount of water passing the root zone each day assuming a well maintained crop and daily irrigation applications. Eqn. 2 was used if precipitation was not sufficient to fulfill the daily water requirement for a particular crop and additional irrigation was necessary. Eqn. 3 was used if precipitation was sufficient to meet the daily water requirement including the salt leaching requirement.

In Eqn. 2, the daily ETo value (obtained by dividing the long-term monthly average by the number of days in the month) was multiplied by the daily crop coefficient to determine the amount of water required by the crop for a particular day. The water required was multiplied by a leaching requirement (1.1 or 10%), which was assumed for all crops for salt control. Average precipitation (based on monthly long-term averages divided by number of days of the month) was subtracted from the water required by the crop to determine the water deficit needed to be supplemented by irrigation. Irrigation, however, is never 100% efficient due to differences in drip line pressure, sprinkler distribution patterns, fields not being perfectly sloped, and spatial variability in soil permeability. Because of such factors, more water must be applied to a field to ensure that every location on a field receives the minimum requirement of water. Three irrigation efficiencies (90%, 80%, and 70%) were used to represent modern efficiencies of drip, sprinkler, and furrow/flood irrigations, respectively. After irrigation efficiency was included, a daily amount of applied irrigation water was obtained. Subtracted from this number was the amount of water required by the crop (note: a crop's Kc also accounts for surrounding soil evaporation) that was not being met by precipitation. The excess amount of water was specified as the daily flux of water passing the root zone, referred to here as the Daily Leachate.

In the case where precipitation was sufficient to meet the daily needs of the plants (including a leaching requirement), Eqn. 3 was used and the excessive precipitation was specified as the daily flux of water. There was assumed to be no runoff.

Daily leachate values were calculated for a 365 day calendar year and can be seen in Figure 62. Fluxes shown are based on 70% irrigation efficiency. Table 31 shows total annual leachate fluxes calculated for each crop and efficiency.

Adminsitrative Record
Page 35076

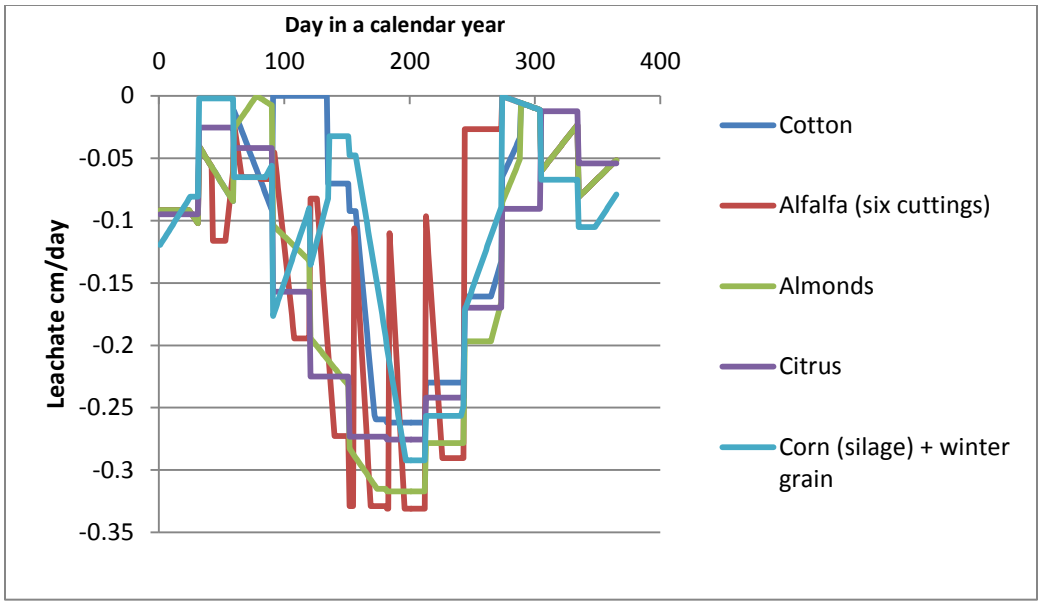

Figure 62. Daily flux of leachate based on 70%irrigation efficiency for the 5 modeled crop cycles. Negative values imply a downward flux in HYDRUS 1D.

6.2.2 Soils

Heterogeneity of subsurface soils usually results in preferential flow (Burow 1993; Harter et al. 2005). Every location in the study area has a unique soil profile, but defining field scale heterogeneity on the scale of thousands of square miles is not possible. Instead of detailed soil profiles, three homogeneous soil profiles were used: sand, loam, and clay soil. Hydraulic parameters for the soil types were obtained from the Rosetta Soil Catalog, included in the HYDRUS 1D software (Simunek et al. 2005). Sandy soil was used to represent the fastest probable travel time of a solute as it has the highest saturated hydraulic conductivity. Clay soil was assumed to represent the longest probable travel time as it has the lowest saturated hydraulic conductivity. Loam soil is thought of as the average soil type, as most soils fall between clay and sandy soil. Table 27 shows the respective parameters for each soil. The Van Genuchten – Mualem single porosity model was used to represent each soil type in HYDRUS 1D.

Adminsitrative Record
Page 35077

Table 27. Hydraulic parameters for the three soil types used.

Parameter	Sand	Loam	Clay
Residual soil water content	0.05	0.08	0.07
Saturated soil water content	0.43	0.43	0.38
Parameter *alpha* in soil water retention function [cm^{-1}]	0.15	0.04	0.01
Parameter *n* in the soil water retention function	2.68	1.56	1.09
Saturated hydraulic conductivity [cm/day]	712.8	24.96	4.8
Tortuosity parameter in the conductivity function	0.5	0.5	0.5

6.2.3 Soil Column Configurations

A total of 18 different soil columns were modeled using six different depths and three soil types. Nodal spacing was set at 4 cm for each soil column and initial solute concentrations were set to zero (Table 28). Initial pressure heads for each node were set to -100 cm except the last (bottom), which was specified with a head equal to zero in order to simulate the water table and resulting capillary fringe. The default temperature of 20° C was used.

Table 28. Soil column construction.

Column depth	Node Spacing	Number of Nodes	Initial Column Head [cm]	Bottom Node Head [cm]	Temperature (C)	Initial Concentration
2m	4cm	51	-100	0	20	0
5m	4cm	126	-100	0	20	0
10m	4cm	251	-100	0	20	0
20m	4cm	501	-100	0	20	0
30m	4cm	751	-100	0	20	0
40m	4cm	10001	-100	0	20	0

6.2.4 Modeling Nitrate as a Conservative Solute

Nitrate is generally assumed non-reactive when present in oxic unsaturated soils. For the purpose of this study, nitrate was assumed to behave conservatively and was modeled as a non-reactive solute.

Adminsitrative Record
Page 35078

Table 29 displays the soil/solute properties used to model nitrate transport in the unsaturated zone (U.S. Department of Agriculture 2011). Longitudinal dispersivity generally changes with the soil type and column depth, but was held constant at 10 cm in this study. Assuming constant longitudinal dispersivity has little to no effect on simulation results, because travel time is determined here as the time until the concentration at the water table is 50% of the input concentration. This travel time is primarily controlled by the advective velocity and not dispersion.

Table 29. Solute transport parameters.

Soil Type	Sand	Loam	Clay
Bulk Density [g/cm^3]	1.69	1.43	1.25
Longitudinal Dispersivity [cm]	10	10	10
Diffusion in water [cm^2/day]	1.55	1.55	1.55

Adminsitrative Record
Page 35079

6.2.5 Climate Data

Average monthly precipitation and ETo data were used when calculating daily leachate fluxes. Standard monthly ETo was obtained from the California Irrigation Management Information System (CIMIS) for the city of Visalia, and daily Eto was calculated by dividing the monthly average by the number of days in the month (California Department of Water Resources a). Similarly, average monthly precipitation was obtained from the National Oceanic and Atmospheric Administration (NOAA) for the city of Visalia, and daily precipitation was calculated by dividing the monthly average by the number of days in the month (Nation Oceanic and Atmospheric Administration). Table 30 below shows the monthly values used.

Table 30. Monthly average precipitation and reference evapotranspiration for the city of Visalia.

Precipitation														
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
City Visalia	**mm**	51.56	49.53	54.61	20.32	9.40	3.56	0.25	0.51	6.35	16.51	29.71	37.84	280.16
	inch	2.03	1.95	2.15	0.80	0.37	0.14	0.01	0.02	0.25	0.65	1.17	1.49	11.03
Reference Evapotranspiration (Eto)														
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Station Visalia-33	**mm**	22.09	42.41	84.83	130.30	172.21	194.82	199.64	175.51	125.22	82.04	38.35	21.08	1288.542
	inch	0.87	1.67	3.34	5.13	6.78	7.67	7.86	6.91	4.93	3.23	1.51	0.83	50.73

Adminsitrative Record
Page 35080

6.2.6 Modeling

HYDRUS 1D (Simunek et al. 2005) was used to model the transport of nitrate in the vadose zone. HYDRUS 1D solves Richards' Equation (Eqn. 4) for water flow in the vadose zone and Fickian based advection-dispersion equations for solute transport (Radclifee & Simunek 2010).

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}[K(\theta)\left(\frac{\partial \varphi}{\partial z} + 1\right)] \qquad \text{(Eqn. 4)}$$

Crops were simulated for a period of 27 years. Daily leachate values were modeled as a specified flux for the upper boundary condition. If, for example, 2 mm was leached in a particular day, a 2 mm/day flux would be assigned for the entire day. The lower boundary condition was specified as a constant head equal to zero. This allowed the water table to be specified, and for the creation of a capillary fringe.

Fluxes for the initial four years were simulated with a concentration of zero so that a normal wetting pattern could be established. After this, fluxes were assigned a concentration of 1. This was done to "cycle up" the soils and generate adequate initial soil moisture content before the solute was applied. Travel time to the groundwater table was defined as the time when the relative concentration at the water table began to exceed a value of 0.5, reflecting the advective velocity of the solute.

6.2.7 Results

Table 31 below shows the results of the modeling, with travel time values shown in years. Figure 63 shows that there is a linear relationship between the water table depth and travel time for a particular annual flux rate. Using this linear relationship, intermediate depths may be interpolated based on the regression line, which is the downward velocity. The downward velocities were then plotted against their respective annual fluxes to determine the relationship between annual leachate flux and the downward velocity of a solute. This was performed on the model results from the three soil types and is shown in Figure 64. In the next section, these relationships were used to estimate downward solute velocities in lieu of detailed modeling to estimate travel times to the water table.

Adminsitrative Record
Page 35081

Table 31. Advective travel time to the groundwater table in years for three soils and six depths. Spaces are left blank when solute transport to a particular depth was not achieved in 27 years.

Advective Travel Time to the Groundwater Table in Years																
		Alfalfa (six cuttings)			Almonds			Citrus			Cotton			Corn (silage) + winter grain		
Irrigation efficiency		70%	80%	90%	70%	80%	90%	70%	80%	90%	70%	80%	90%	70%	80%	90%
Total Leached (cm/yr)		46.6	31.0	18.8	52.8	34.2	19.7	50.8	31.8	17.1	37.1	25.6	16.6	39.8	26.5	16.2
Soil column depth (m)		Alfalfa (six cuttings)			Almonds			Citrus			Cotton			Corn (silage) + winter grain		
Sand	2	0.45	0.54	0.86	0.47	0.56	0.85	0.46	0.57	1.01	0.58	0.68	1.07	0.31	0.63	1.07
	5	0.78	1.48	2.22	0.76	1.35	2.15	0.79	1.51	2.39	1.24	1.73	2.39	1.17	1.5	2.54
	10	1.8	2.78	4.35	1.73	2.69	4.13	1.76	2.78	4.79	2.43	3.29	4.82	2.3	3.13	4.99
	20	3.87	5.44	8.55	3.46	5.16	8.33	3.6	5.4	9.41	4.79	6.53	9.66	4.44	6.27	9.81
	30	5.76	8.39	12.8	5.18	7.69	12.3	5.39	8.08	14.0	7.13	9.86	14.4	6.6	9.49	14.8
	40	7.75	11.1	17.1	7.04	10.2	16.5	7.25	10.9	18.8	9.5	13.2	19.2	8.87	12.7	19.5
Loam	2	1.32	1.68	2.78	1.08	1.63	2.73	1.15	1.68	3.14	1.61	2.15	3.19	1.31	2.08	3.33
	5	2.85	4.32	6.84	2.71	3.94	6.59	2.75	4.18	7.5	3.78	5.17	7.66	3.42	4.95	7.9
	10	5.96	8.58	13.5	5.25	7.95	13.0	5.46	8.39	14.8	7.32	10.3	15.1	6.8	9.92	15.5
	20	11.8	17.2		10.6	15.8		11	16.8		14.7	20.4		13.6	19.7	
	30	17.8			16.0			16.5			21.9			20.5		
	40				21.3			22.1								
Clay	2	1.49	2.35	3.64	1.45	2.08	3.57	1.46	2.34	4.08	1.79	2.69	4.15	1.69	2.6	4.32
	5	3.73	5.74	9.6	3.52	5.46	9.2	3.57	5.66	10.6	4.82	7.13	10.8	4.39	6.78	11.1
	10	7.77	11.7	19.4	6.94	10.7	18.6	7.43	11.6	21.4	9.88	14.4	21.9	9.25	13.8	22.5
	20	15.8			14.1	21.7		14.6			19.9			18.5		
	30				21.2			22.0								
	40															

Adminsitrative Record
Page 35082

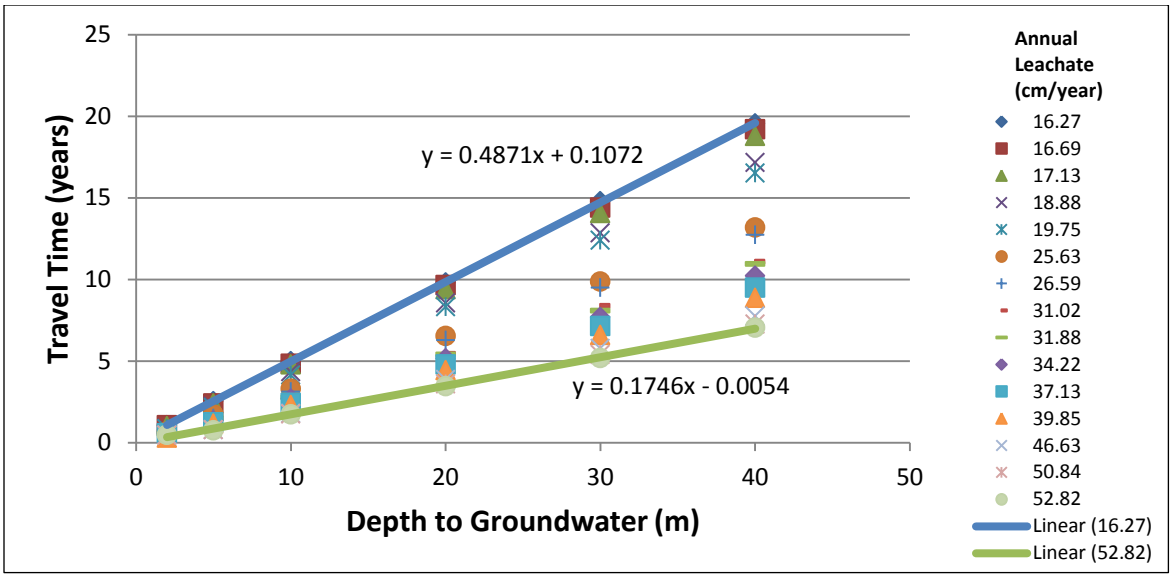

Figure 63. Travel times for a conservative solute in sandy soil.

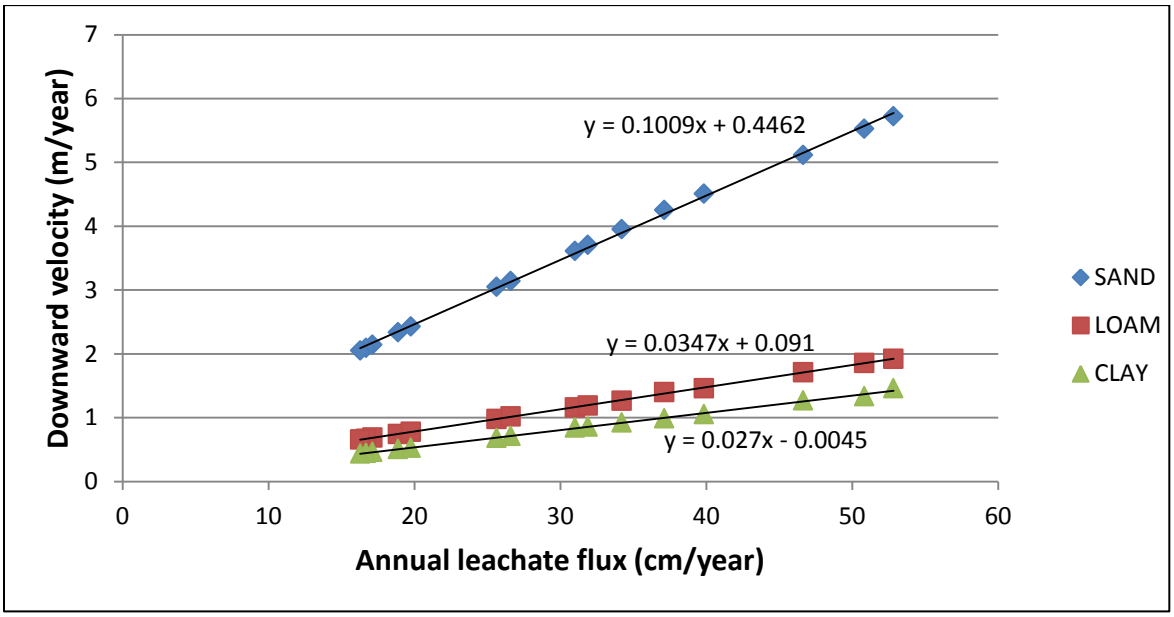

Figure 64. Relationship between annual flux and downward velocity for three soil types.

6.2.8 Modeled Irrigation

Daily fluxes, as previously modeled, are not representative when compared to actual irrigation scheduling. Farmers generally irrigate periodically, often several days to weeks apart. Does this make a difference in travel times? For Alfalfa (six cuttings), daily leaching fluxes were summed every seven days to simulate weekly irrigation to see if this changed the travel times significantly. Table 32 shows that travel times did not change significantly given the time scale under investigation (years to decades). As

Adminsitrative Record
Page 35083

can be seen in Table 32, the largest difference of 0.2 years was observed at 90% efficiency and 50 m depth to the water table, which is insignificant on the decadal time scale.

Table 32. Travel times compared between simulated daily and weekly irrigation. Values are in years.

	Alfalfa (Six Cuttings), Travel Time in years					
Depth	**Daily Irrigation**			**Weekly Irrigation**		
Efficiency	**90%**	**80%**	**70%**	**90%**	**80%**	**70%**
2m	0.75	0.55	0.48	0.7	0.5	0.43
5m	2.12	1.42	0.77	2	1.37	0.75
10m	4.22	2.77	1.81	4.07	2.72	1.76
20m	8.48	5.42	3.87	8.37	5.36	3.84
30m	12.79	8.33	5.75	12.66	8.17	5.68
40m	17.09	11.08	7.75	16.9	10.99	7.65
50m	21.33	13.72	9.71	21.13	13.62	9.62

6.2.9 Verifying Downward Velocity Estimates

In order to confirm the downward velocities calculated in Figure 64, annual fluxes were divided by average soil water content. This calculation computes an effective vertical velocity that should correspond to the distance traveled in a year according to the mass balance. Cotton and Alfalfa, 70% and 90% irrigation efficiencies, and sand and clay soils were used to verify the results. To establish average soil water content, an observation node was placed at a depth of 5 m in a 20 m soil column. Average soil water content was calculated by averaging the soil water content values recorded at every time step after the conservative solute had reached the observation point (when the concentration at the node exceeded 50%). This was done to ensure that values of water content before the bulk of the water had reached the node (which would reflect the arbitrary initial condition for water content used in the simulation) did not bias the calculated average water content. The results show that our calculated downward velocities using regression analysis (of the relationship between breakthrough time and depth to water table) compare very well with those calculated using annual flux rates and average soil water content. The results are shown in Table 33 and Table 34, below. The results demonstrate that the key variables controlling downward movement of nitrate are the annual water flux rate below the root zone and the effective moisture content of the unsaturated zone between the water table and the root zone.

Adminsitrative Record
Page 35084

Table 33. Downward velocities using linear regression of modeled travel times and velocities based on the annual flux divided by the average water content.

Efficiency	Annual Flux (m/yr)	Velocity in sand soil		Velocity in clay soil	
		Using regression (m/yr)	Annual flux/Avg. Water Content (m/yr)	Using regression (m/yr)	Annual flux/Avg. Water Content (m/yr)
Cotton - 70%	**0.3713**	4.193	4.240	0.998	0.990
Cotton - 90%	**0.1669**	2.130	2.080	0.446	0.450
Alfalfa - 70%	**0.4663**	5.151	5.208	1.255	1.238
Alfalfa - 90%	**0.1888**	2.351	2.365	0.505	0.508

Table 34. Average soil water content based on 70% and 90% irrigation efficiencies for cotton and alfalfa grown in sand and clay soils.

Efficiency	Annual Flux (m/yr)	Sand soil average water content	Loam soil average water content	Clay soil average water content
Cotton - 70%	**0.3713**	0.088	0.272	0.375
Cotton - 90%	**0.1669**	0.079	0.249	0.371
Alfalfa - 70%	**0.1888**	0.089	0.277	0.377
Alfalfa - 90%	**0.4663**	0.081	0.252	0.372

Adminsitrative Record
Page 35085

6.3 GIS Map Creation

In the previous section, it was verified that three parameters could be used to estimate travel time to the water table: 1) soil type 2) total annual amount of water leached and 3) depth to water table. This section takes the previous modeling and applies the results to the SV and TLB. Three maps (Figure 66, Figure 67, and Figure 68) were generated showing estimated travel time of a solute to reach the groundwater table based on a sand, loam, and clay soil.

6.3.1 Land Use

Detailed land use for the study area was constructed as part of the Technical Report 2 (Viers et al. 2012) and is a combination of Monterey, Kern, Tulare, Kings, and Fresno County surveys created by the DWR (see Technical Report 2, for methods used for land use map construction, Viers et al. 2012). Counties are not surveyed for land use each year, so the map used to define current land use is a collection of several surveys from 1997–2006, with most data coming from the year 2000. The land use surveys cover many land uses such as agricultural, urban, waterways, and natural vegetation. Only those fields classified as agricultural were used in this study.

6.3.2 Field Water Budgets

DWR divides the state into 400+ spatial units for the purpose of hydrological budgeting, referred to as Detailed Analysis Units (DAU) (California Department of Water Resources). The land use maps were spatially joined to DWR's Detailed Analysis Unit map, to assign each agricultural field the corresponding DAU in which it lies. Each DAU has water budget data estimated for 20 major crops, and is based on local climate and irrigation technologies. Two of the attributes provided by DWR's Annual Water Use spreadsheets are applied water (AW) and evapotranspiration (ET). Applied water is the total amount of water applied via irrigation taking into consideration local irrigation technologies and local climate (precipitation, ETo, etc.). AW includes only water that must be applied *in addition to* precipitation. ET provides an estimate to the amount of water that is taken up into the plant tissue, transpired, and evaporated from the soil surface. The DWR provides this detailed water budget information for the years 1998–2001. This study utilized the average AW and ET values for these four years. The average of the four years was used as it likely represents the long-term average for the study area, given that, in California, these years represent a transition from a wet year (1998) to a dry year (2001).

Precipitation was obtained from Oregon State University's PRISM Climate Group, and is based on average annual precipitation from 1971–2000 (PRISM Climate Group 2006). Precipitation was added to AW to obtain the net amount of water applied to a particular field. ET was subtracted from this to determine the net amount of water which remained. This was assumed to be the annual amount of leachate and was used in combination with depth to the water table and soil type to determine the travel times.

Adminsitrative Record
Page 35086

6.3.3 Depth to Water Table

Depth to the water table (Figure 65) was obtained as a point file, which defines depth to groundwater values at the center of PLSS sections in the study area (Spurlock 2000). Fields were assigned a depth to water table value based on the nearest point. If a field contained multiple points, the field was assigned the average value of the points. Although this file provides the best coverage of depth to the groundwater table for our study area, it does not cover it in its entirety. For this reason, if a field was more than 2.5 km (1.6 mi) from the extent of the data coverage, it was not included in the study.

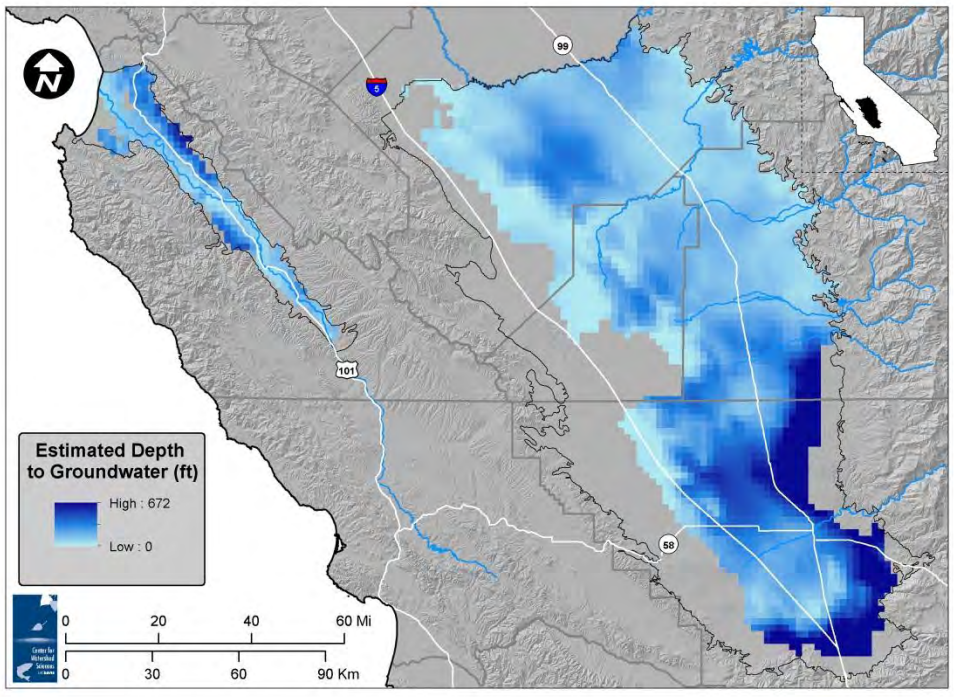

Figure 65. Depth to the water table. (Source: Spurlock 2000.)

6.3.4 Results and Discussion

Three maps are shown in Figure 66, Figure 67, and Figure 68 and represent the estimated travel time from the surface to the groundwater table for three soil types: sand, loam, and clay.

The travel time results shown in Table 31 (for solute transport in sandy soil) are similar to those obtained in a previous study where models of subsurface heterogeneity were employed. Burow,

Adminsitrative Record
Page 35087

studying nitrate transport in the vadose zone in the SV, found travel times between 4 and 14 years for a depth to groundwater of 35 m (Burow 1993). Burow used two dimensional statistical realizations of subsurface sediments to model the natural variability of hydraulic parameters often found in alluvial and fluvial deposits. The differences in travel times obtained by Burow were based on different realizations of subsurface heterogeneity. Looking at Table 31, similar travel times are seen for the sandy soil travel times to a depth of 30 m (98 ft)(5.18–14.8 years); however, the differences between travel times are a function of annual leachate fluxes.

It is well known that preferential flow exists in unsaturated sediments (Burow 1993; Harter et al. 2005). Harter et al. demonstrated that travel times through a heterogeneous configuration of sediments are much quicker than those through uniformly distributed sediments due to preferential flow through the coarser grained sediments (Harter et al. 2005). Therefore, it is expected that travel times to the water table are likely closer to the results based on sandy soil (Figure 66). The presence of laterally extensive low conductivity layers (e.g., clays and silts) may impede solute travel times; however, even these hydrologic features are known to have spatially variable hydraulic properties that can lead to faster than expected travel times via preferential flow.

Travel times established by groundwater modeling in the saturated zone may underestimate total travel time of a solute from the surface to a well screen due to the additional time needed for a solutes traveling through the unsaturated zone. The study presented here shows that the amount of additional travel time can be significant, depending on the soils present, crops being grown, and depth to the water table.

Adminsitrative Record
Page 35088

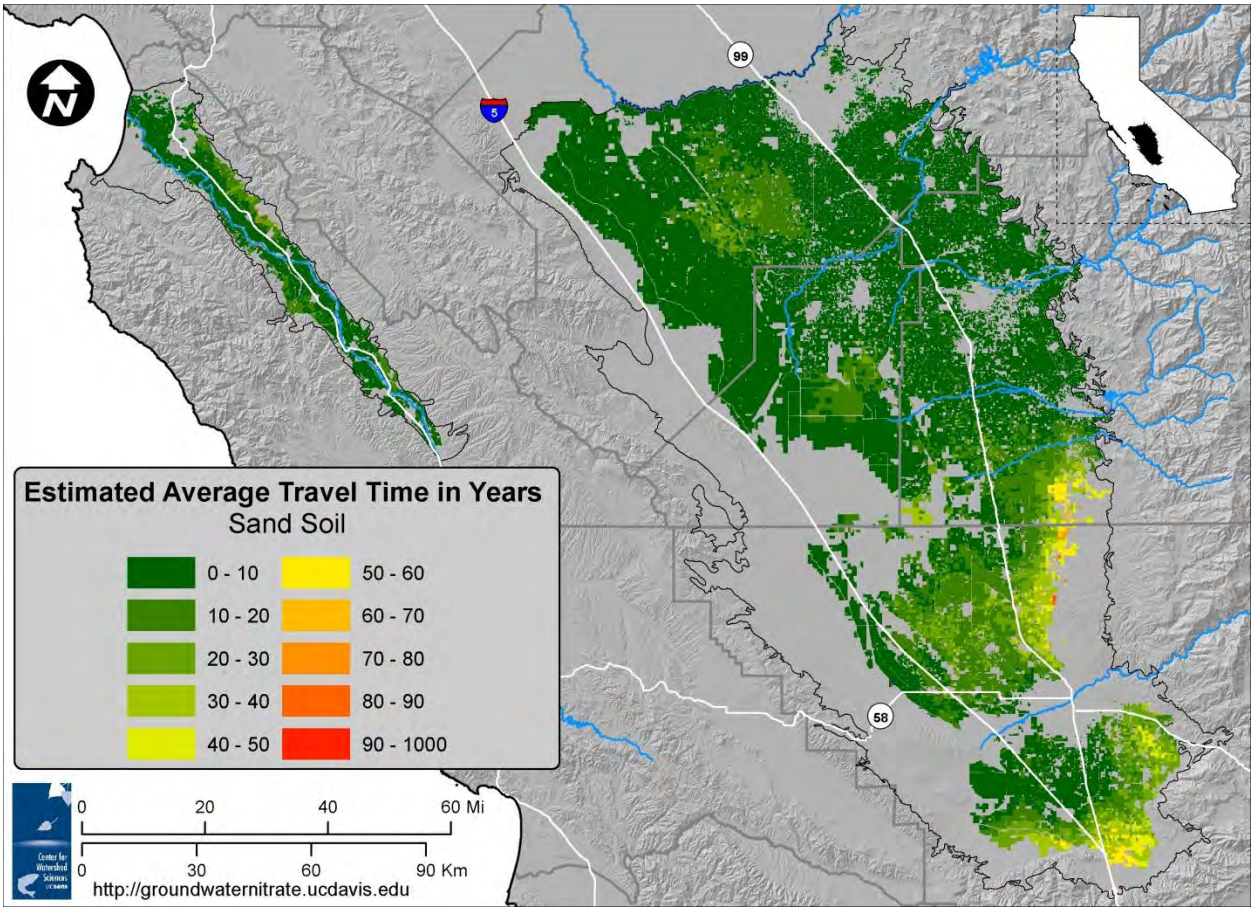

Figure 66. Travel time based on sand soil.

Adminsitrative Record
Page 35089

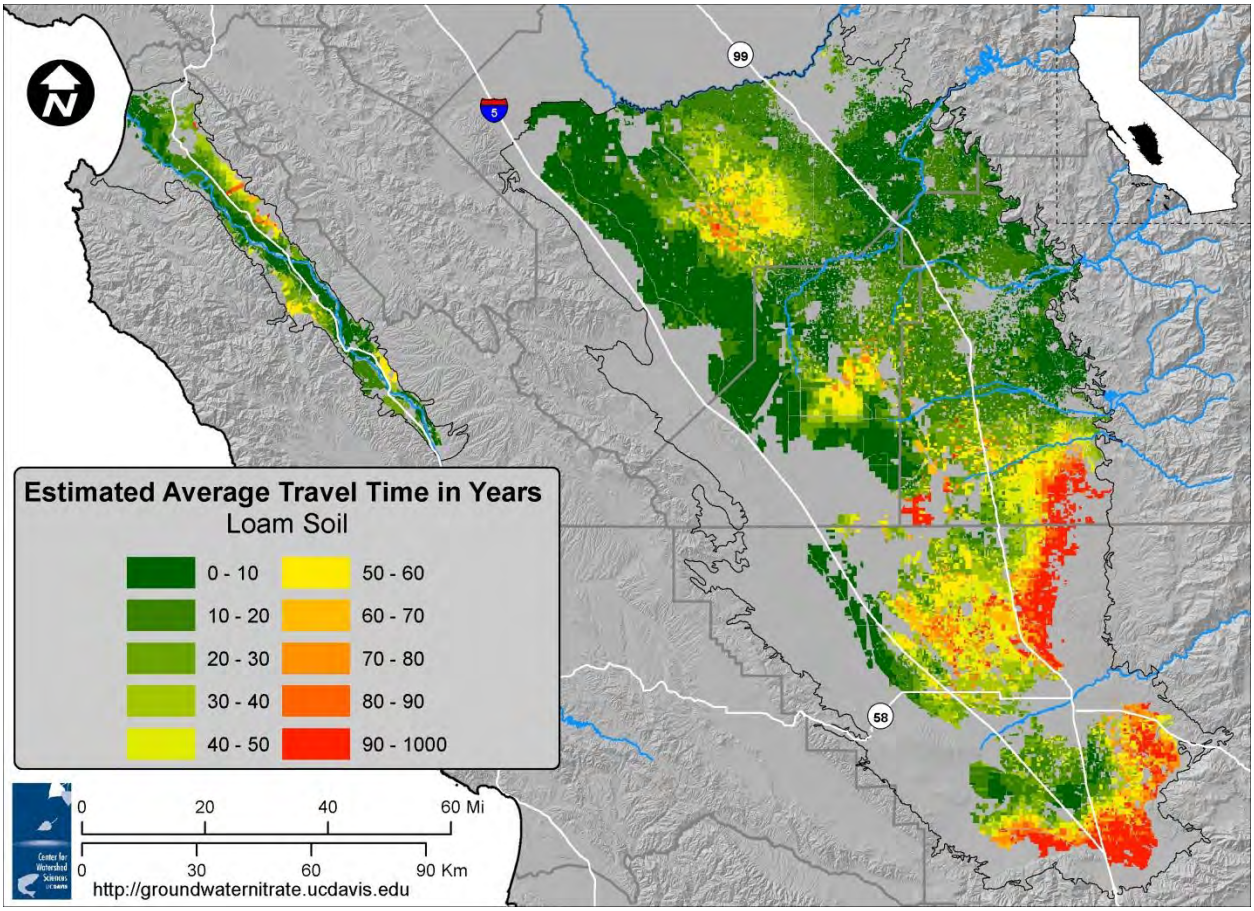

Figure 67. Travel time based on loam soil.

Adminsitrative Record
Page 35090

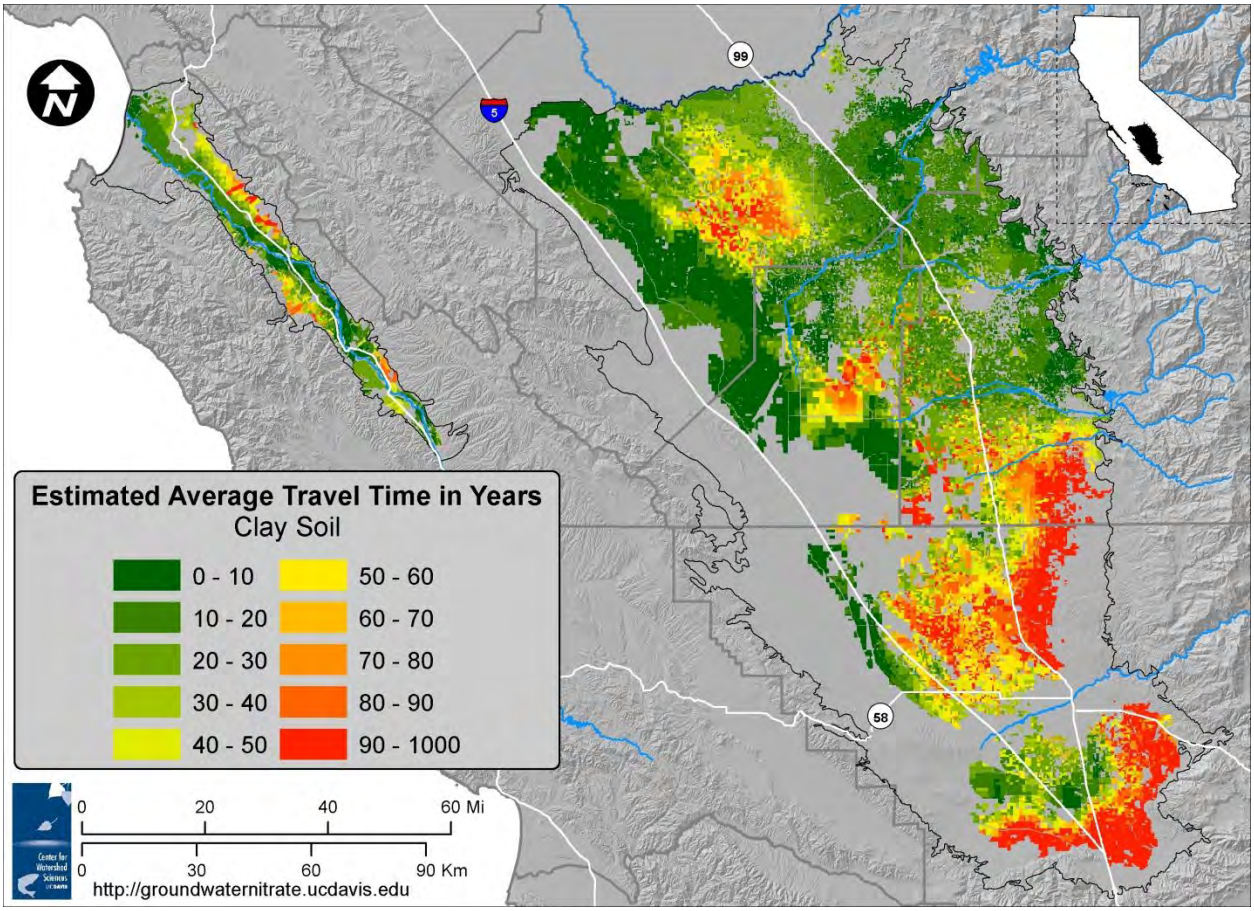

Figure 68. Travel time based on clay soil.

Adminsitrative Record
Page 35091

7 Groundwater Nitrate Forecasting/Modeling

Prepared by:

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Adminsitrative Record
Page 35092

7.1 Introduction and General Conceptual Overview of the Approach

7.1.1 Nonpoint Source Pollution a Global Groundwater Quality Threat

Nonpoint source (NPS) pollution of groundwater has been recognized as a key water quality problem worldwide (WWAP 2009). Agriculture is considered the most dominant NPS polluter of groundwater, primarily through emissions of nitrogen and salt, but also pesticides and other farm-chemicals (Humenik et al. 1987; Bower 2000; VanDrecht et al. 2003; Vitousek et al. 2010; Watanabe et al. 2010). Nitrate-nitrogen is considered by far the most common type of groundwater contamination associated with agricultural activities (e.g., Spalding and Exner 1993; Harter et al. 2002; Spruill et al. 2002; VanDrecht et al. 2003; Burow et al. 2010). Groundwater nitrate is largely derived from fertilizer nitrogen and animal nitrogen applied in agriculture, where nitrogen is a vital nutrient for plant growth (WWAP 2006, p. 117). The increasingly intensive use of nitrogen-based fertilizers in agriculture has allowed global food production to stay ahead of rapid population growth (Almasri and Kaluarachchi 2007; Stadler et al. 2008; Laftouhi et al. 2003), but at potentially significant cost to current and future water quality in production wells (Corwin and Wagenet 1996). With further growing world population and higher standards of living, food consumption is estimated to increase 70% over the next four decades, while global land and water resources have limited growth reserves. Intensification of agriculture will therefore continue (Molden 2007). Besides nitrate, long-term salinization of groundwater basins from nonpoint sources, particularly in semi-arid and arid irrigated agricultural regions is a second critical threat to groundwater quality around the globe (Burkhalter and Gates 2005; Martín-Queller et al. 2010).

The degradation of groundwater resources not only impacts ecosystems worldwide via return flow of groundwater to surface water (Bouwman et al. 2009), but it affects both irrigation water (salinity) and drinking water quality (nitrate, salinity, pesticides, pathogens). Approximately half of the global population depends on groundwater as a drinking water source (UN WWAP 2003; Giordano 2009). In contrast, most of the global population in intensively farmed agricultural regions such as the California Central Valley, the North-American High Plains and Floridan aquifers, Central Europe's unconsolidated aquifers, the Indo-Gangetic aquifer complex, and the North China plains, relies on – often shallow – groundwater (e.g., Power and Schepers 1989; Chakraborti et al. 2011). Globally, 43% of consumptive water used in irrigation is groundwater (Siebert et al. 2010). Furthermore, particularly in Europe and North America, the need to protect drinking water quality for large populations sectors has driven and continues to drive NPS policy (e.g., Sonneveld and Bouma 2003; Dowd et al. 2008).

Sound policy requires thorough scientific understanding of nonpoint sources and how they work, and of the linkage between nonpoint sources and groundwater discharges to users or affected ecosystems (domestic wells, irrigation wells, urban/municipal wells, springs, discharges to stream reaches). Significant scientific effort has been dedicated to understand, manage, and monitor potential sources, to understand the dynamics of NPS pollutants in the vadose zone and in groundwater, and to assess the environmental and public health consequences of NPS pollution of groundwater (Addiscott and

Adminsitrative Record
Page 35093

Wagenet 1985; Corwin et al. 1999; Pavlis et. al. 2010). The spatio-temporal and process complexity of NPS pollution of groundwater on one hand and the number and large diversity of affected stakeholders on the other hand (Figure 69) requires management of large datasets, the bridging of possibly huge datagaps, upscaling, the use of potentially complex models, and – most importantly – that science effectively communicates with policy and decision makers (King and Corwin 1999; National Research Council 1993).

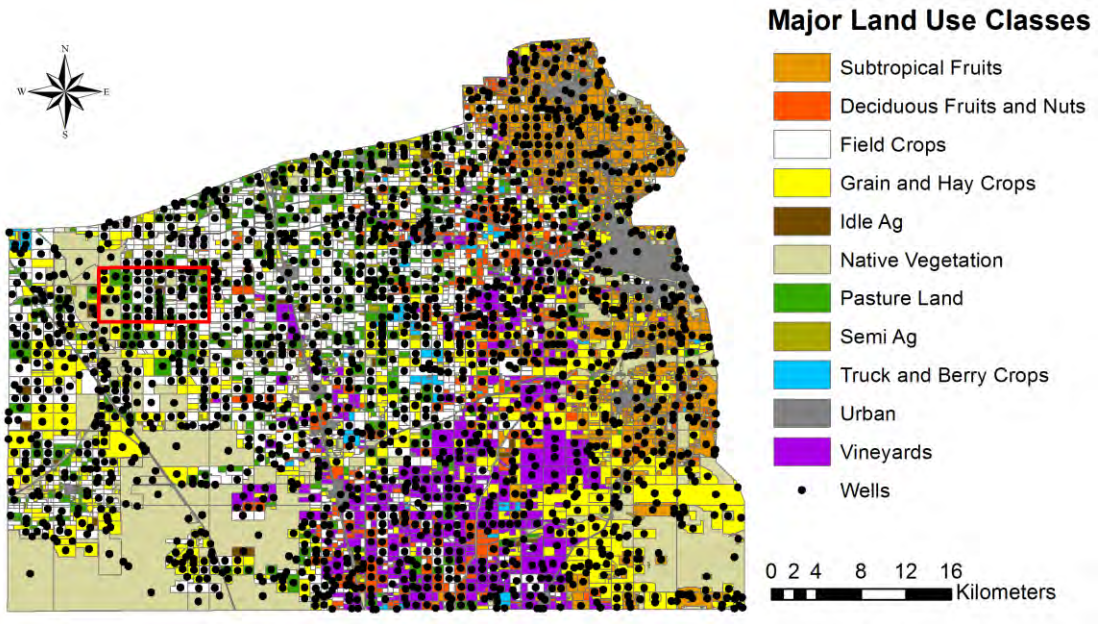

Figure 69. Typical spatial variability of a land use (and implicitly, associated diffuse pollution) in an intensively managed semi-arid agricultural region with significant groundwater pumping for irrigation (black dots), Tule River Groundwater Subbasin, Central Valley Aquifer System, California. (Source: modified from Ruud et al. 2004.)

7.1.2 Key Differences Between Nonpoint Source Pollution and Point Source Pollution Dynamics

Agricultural activities are the dominant nonpoint source of groundwater nitrate and salts. Other significant sources include urban wastewater discharge, septic systems, wastewater holding ponds, and atmospheric deposition. For assessment, planning, and regulatory purposes, nonpoint source pollution, particularly from nitrate and salt, has a distinctly different problem set than point sources:

1. Control measures for point sources have been in development now for four decades (e.g., U.S. Code Title 42 Chapter 103 U.S. Comprehensive Environmental Response, Compensation and Liability Act of 1980), while control and monitoring measures for nonpoint sources of groundwater have only begun to be developed over the last one to two decades (e.g., EU Nitrate Directive[11] and California Salt and Nutrient Basin Plan[12] development).

[11] http://ec.europa.eu/environment/water/water-nitrates/index_en.html

Adminsitrative Record
Page 35094

2. Point sources, most commonly, are accidental spills of limited duration contributing a negligible fraction of basin recharge. Nonpoint sources are commonly associated with natural or intentional, managed or unmanaged sources of recharge that provide a significant fraction or even the majority of a groundwater basin's recharge on a time-varying but continuous basis (GWSP 2008; UN/WWAP 2006; Burow et al. 2010).
3. Point sources tend to be of limited spatial and temporal extent. One-time spills may occur over areas from less than 0.1 ha (0.25 acres) to few ha in size (Freeze and Cherry 1979; Bower 2000; Domenico and Schwartz 2008). In contrast, nonpoint source pollution can occur repeatedly across the majority of the land surface area of entire groundwater basins, particularly in agricultural regions. Sources are spatially near-contiguous, while individual source facilities (liable parties) range from ten to several hundred hectares in size and are characterized by large spatial as well as temporal variability and, hence, uncertainty. For example the volume of irrigated water and its concentration in nitrogen vary significantly with crop type and season. In addition there is significant uncertainty related to the estimated amounts of excess nitrate that leach to groundwater (Loague and Corwin 1998). Similar uncertainty exists in rain-fed crops, where nitrate leaching depends on various factors such as rainfall intensity, air temperature (Xin-Qiang et al. In press), the form of nitrogen application (Pilbeam et al. 2004), N application rate, form, and timing (Basso et al. 2010), etc.
4. Point sources of pollution are often very intensive (i.e., the associated pollution level (concentration) at and near the source can be many orders of magnitude above the regulatory limit). Nonpoint source pollution is typically of low intensity (i.e., at concentration levels near the regulatory limit and up to one order of magnitude above the regulatory limit (e.g., nitrate and salts)) (Burow et al. 2010).

These differences between point source pollution and nonpoint source pollution of groundwater require that assessment methods, monitoring approaches, and regulatory frameworks for nonpoint source control do not simply copy the approaches taken in the point source arena, but that methods are developed specifically for nonpoint sources.

7.1.3 Groundwater Nonpoint Source Assessment Tools

Studies have developed various modeling tools for assessing and predicting aquifer pollution impacts or concentration levels in response to land use and management strategies. These models can be generally grouped into three categories:

1. Overlay and index methods, where different parameters of spatially distributed hydrologic, geographic, soils, and source information are combined to give an estimation of the vulnerability in the form of an index (National Research Council 1993; Pavlis et al. 2010) such as DRASTIC (Aller et al. 1987), SINTACS (Civita and De Maio 2004), etc.
2. Statistical methods that estimate the vulnerability by correlating spatial variables with actual occurrence of pollutants in the groundwater (Pavlis et al. 2010) such as regression (Nolan et al. 2002; Worrall et al. 2000), fuzzy logic (Uricchio et al., 2004), etc.

[12] http://www.swrcb.ca.gov/centralvalley/water_issues/salinity/index.shtml

Adminsitrative Record
Page 35095

3. Process based methods to simulate the contaminant transport using mathematical formulas (Fogg et al. 1999).

The majority of these latter models are limited to the simulation of pollutants in the vadose zone, while simple methods such as zero-order mixing models (Mercado 1976; Lee 2007) or vertical plug-flow models (Refsgaard et al. 1999; Hansen 1991; Cho and Mostaghimi 2009) are used for the estimation of the fate of contaminants in the saturated zone. These kinds of approaches are not able to properly capture the spatial and temporal variability of contaminant loading across large aquifer systems.

Detailed spatio-temporal nonpoint source impact assessment in an aquifer requires numerical flow and transport models in two and three dimensions. A few studies developed analytical solutions to governing flow and transport equations (Leij & Dane 1990; Fry et al. 1993; Perez-Guerrero et al. 2009); however, their applicability is limited to simple geometry cases under special boundary conditions. Coupled numerical solution schemes of groundwater flow and transport have been applied in groundwater remediation studies and nonpoint source prediction models at relatively small scale sites (Trowsdale and Lerner 2007) or large scales (Carle et al. 2006), but often using relatively coarse gridded solutions (Almasri and Kaluarachchi 2007; Jiang and Somers 2009; Zhang and Hiscock In Press).

The implementation of a fully three-dimensional flow and transport model for nonpoint source assessment is largely limited by computational resources; typically, current numerical flow and transport models are designed with 10^5 to 10^8 degrees of freedom (particle lines, finite difference cells, finite elements), allowing for $10^2 – 10^3$ discretization points per dimension. At typical (point source) contamination sites to which these are applied, the resulting spatial discretization is on the order of 10^{-1} to 10^2 meters (Carle et al. 2006). On the other hand, the simulation of entire groundwater basins affected by, for example, agricultural nonpoint source pollution, being tens to hundreds of kilometers across (e.g., Floridan aquifer system, High Plains aquifer system, Central Valley aquifer system, North China Plain aquifer system, Indo-Gangetic aquifer system), would require spatial grids that are four to six orders of magnitude larger ($10^9 – 10^{14}$ degrees of freedom), at effectively similar discretization. The latter is necessary to properly capture individual sources (e.g., crop fields, lagoons, septic leach fields) and the impacts to individual contaminant sinks or receptors (wells, stream reaches, springs) across a basin (Bloomfield et al. 2006). The application of classic numerical contaminant modeling approaches to model nonpoint sources and their impacts at multiple locations (sinks) across entire basins is generally beyond current computational capacities. This is particularly true if the focus is on the impact to individual production wells (domestic, municipal, irrigation), of which there may be tens of thousands across a single groundwater basin.

To render the computational burden tractable, alternative methods have been proposed. For example Lin et al. (2010) developed a simplified numerical model where the governing equations of 3D groundwater flow and contaminant transport are replaced by a 2D finite element approximation in x-y direction, with a 1D finite difference approach for the vertical direction. Almasri and Kaluarachchi (2007) used surrogate models such as Modular Neural Networks in order to predict nitrate contamination in the Sumas-Blaine aquifer in Washington State, but performance was found inferior to the classical fate and transport model.

Adminsitrative Record
Page 35096

A widely used alternative technique is the streamline simulation model, where a multi-dimensional simulation problem is decoupled into multiple one-dimensional problems (Martin and Wegner 1979). Streamline models have been used extensively in petroleum engineering (Blunt et al. 1996; Baker et al. 2002). Jang and Choe (2002) utilized the streamline model to simulate solute transport in fractures, and found that the Breakthrough Curves (BTCs) from simulations matched excellently with experimental data. Bandilla et al. (2009) combined an analytic element based solution of groundwater flow with the streamline method neglecting transverse dispersivity effects. Recently, Herrera et al. (2010) proposed an improved version of the method for simulating reactive solute transport in porous media. Streamline methods have also been used for model calibration (Jang 2007; Jang and Choe 2002, 2004), where the flow domain is decomposed into streamlines and the calibration parameters are adjusted along the streamlines. The efficiency of the streamline method stems from neglecting transverse numerical dispersion. Depending on the modeling objective, the method is computationally far less demanding than a full three-dimensional solution at equivalent high resolution.

Although the streamline method has been established as a reliable alternative solution to simulate transport in aquifers, simulation time for long simulation periods can be exceptionally large. For environmental managers interested in evaluating or optimizing multiple nonpoint source management scenarios, these simulation models are not practical. Tools for efficiently evaluating the long-term impacts of past, current, and (alternate) future nonpoint source loading scenarios at the groundwater basin scale are still lacking.

7.1.4 Proposed Modeling Framework

In this report we developed and use a very efficient, yet highly resolved transport simulation approach that accounts for and takes advantage of the distinct attributes of nonpoint source pollution (as opposed to point source pollution). Our objective is to design a Non Point Source Assessment Tool (NPSAT) applicable to large groundwater basins with a highly heterogeneous, but spatio-temporally continuous coverage of nonpoint sources, that gives stakeholders, decision makers, and environmental managers specific information on the time-dependent statistical distribution of nonpoint source pollutants in a finite-sized ensemble of discrete groundwater discharge surfaces (e.g., well screens and streambeds). The latter comprise a distributed set of discrete compliance surfaces, which can be further categorized (grouped) by significantly controlling factors such as depth, hydrogeologic sub-regions, landscape and land use regions, etc. The concentration history at a discharge surface is controlled by aquifer properties and their spatial distribution, by groundwater pumping and other discharges, and by the spatio-temporally variable, continuous nonpoint source pollution fluxes across the recharge surfaces of the aquifer. Specifically, our objective is to develop an efficient physically-based hydrogeological modeling framework to predict time-dependent pollutant concentration histograms and their probability distributions for compliance surfaces in a groundwater basin under spatio-temporally variable nonpoint source loading.

Adminsitrative Record
Page 35097

7.2 Nonpoint Source Assessment Toolbox

7.2.1 Conceptual Approach

Groundwater flow is governed by Darcy's law and the conservation of mass (Bear 1979):

$$\nabla \mathbf{q} + Q_s = \nabla(\mathbf{K}\nabla h) + Q_s = S\frac{\partial h}{\partial t} \qquad \text{(Eqn. 5)}$$

subject to appropriate initial and boundary conditions. Here,$\mathbf{q}$ is the Darcy flux, h is the hydraulic head, $\mathbf{K}$ is the hydraulic conductivity tensor, Q_s represents a vector of sources and/or sinks, S is the storage coefficient, and t represents time. The governing equation of contaminant transport in groundwater is (Bear 1979):

$$R\frac{\partial c}{\partial t} = \nabla \cdot (\mathbf{D}\nabla c) - \nabla(\mathbf{v}c) + G \qquad \text{(Eqn. 6)}$$

where c is the concentration of the contaminant,$\mathbf{v} = \mathbf{q}/\theta$ is the velocity field, θ is the porosity of the porous medium, $\mathbf{D}$ is the dispersion tensor, t represents time, G represents sources and sinks (e.g., via recharge, wells) and R is the retardation factor (Putti et al. 1990). Pollutant concentrations (Eqn. 6) in a groundwater basin are controlled by spatio-temporally variable, dynamic sources and sinks of water and associated (dissolved) pollutants (Figure 69) and by spatially distributed (heterogeneous) aquifer properties.

Of particular interest to groundwater quality protection and management is the pollutant concentration (historical, current, and future) in water discharged from a (finite) set of individual wells or gaining stream reaches (Compliance Discharge Surfaces-CDSs) within a groundwater basin. The pollutant concentration history of the well water or stream reach discharge is here referred to as the breakthrough curve (BTC). The BTC at the CDS is controlled by the pollutant loading history in the source area of the CDS and by the solute reactions and dispersion along the groundwater flow paths between source area and CDS. The CDS source area is defined as the recharge area associated with all groundwater flow discharging into the CDS. Generally, recharge and pollutant loading within the source area may be spatially and temporally variable and not all locations within the source area contribute to the associated CDS at all times, due to transient changes in groundwater flow direction.

To yield the solution of Eqns. 5 and 6 tractable for nonpoint source pollution at the basin scale, yet with sufficiently high resolution, we make three critical simplifications: first, we assume that groundwater flow is steady-state:

$$\nabla \cdot (\mathbf{K}\nabla h) + Q_s = 0 \qquad \text{(Eqn. 7)}$$

Second, we assume that transverse dispersion in Eqn. 6 is negligible (longitudinal dispersion only), and, third, we assume that pollutant reactions are limited to first order degradation, linear sorption, or a combination thereof.

Adminsitrative Record
Page 35098

We support the first assumption with the following heuristic consideration: a nonpoint source pollutant entering an aquifer at a specific location, but continuously over a period of time may discharge at multiple proximate CDSs at different times if groundwater flow is sufficiently transient. However, our focus here is on exceedance probabilities and hence on the ensemble set of BTCs across a group of CDSs (and, hence, a group of source areas), which are much less sensitive to transient changes in source area. Source areas of CDSs may partially overlap due to transient flow conditions. Hence, a steady-state flow approximation still allows for capturing both central tendencies (mean travel time) and the degree of variability of travel time within a CDS and between CDSs.

The second assumption is thought to introduce limited error, because the lateral extent of nonpoint sources is large relative to the length scale of transverse dispersivity or transverse macrodispersivity (Neuman 1990; Gelhar et al. 1992; Kim et al. 2004). The third assumption has been found to be applicable to a wide range of nonpoint source pollutants, including salinity, nitrate, and pesticides (Beltman et al. 1995; Lindenschmidt 2006; Almasri and Kaluarachchi 2007).

The steady-state flow problem Eqn. 7 is separable from the transport problem Eqn. 6 and here solved subject to the appropriate aquifer domain and boundary conditions using a finite element method (FEM). The grid resolution is chosen to capture the spatial pollutant loading variability as well as the flow dynamics around individual CDSs with sufficient detail. For example, the average size of individual sources in a typical agricultural region (California, Central High Plains, North China Plains, Central Europe) varies from 10^2 to 10^6 m^2 (10^3 – 10^7 ft^2); hence, the maximum size of a side of an element is in the range from 10 m (~33 ft) to 1000 m(~3300 ft). Near the CDSs, resolution is on the order of 10 m (~33 ft) (e.g., Figure 69 (Pilot study area)) to provide appropriate flow field resolution near the well.

7.2.2 High Resolution Groundwater Velocity Field Computation

Nonpoint contamination sources exhibit significant variability across source types (agricultural crops, septic leach field, ponding basins/lagoons) and among similar sources managed by different landowners and subject to variable land uses (e.g., varying crops). Nonpoint source loading is also highly variable in time. To assess nitrate and other NPS pollution in domestic and public supply wells, spatial variability must be resolved to the scale of individual wells and their source area or finer. For some NPS pollutants, such as nitrate and salinity, which typically vary over a relatively narrow range near regulatory limits (+/- one order of magnitude), temporal variations of source loading at the annual and inter-annual scale drive subsequent well pollutant levels, while shorter-term and very small-scale variations in the pollutant signal are absorbed by the mixing that occurs in typical production well screens. Proper resolution of physical transport processes in groundwater requires high resolution computer models and limits the application of process models to large groundwater basins (10^3 to 10^5 km^2 ($3.8x10^2$ -$3.8x10^4$ mi^2). For example, the average size of individual sources in a typical agricultural region (California, Central High Plains, North China Plains, Central Europe) varies from 10^2 to 10^6 m^2 (10^3 – 10^7 ft^2) (i.e., the maximum size of a side of an element or cell likely is in the range from 10 m (33 ft) to 1000 m (3280 ft)). Process-based model applications have therefore been limited to site-specific studies of pollution, primarily point source pollution, particularly if aquifer heterogeneity is also considered.

Adminsitrative Record
Page 35099

In this report the process-based regional modeling is achieved by introducing an automated two-step domain decomposition method where the domain is first simulated using a coarse resolution and then is divided into several overlapping sub-domains for either sequential or parallel high resolution simulation of flow and transport. Boundary conditions for the sub-domains are interpolated from the coarse grid solution, while a weighted average scheme is used to smooth the velocity field across sub-domain boundaries.

In general Domain Decomposition (DD) Methods are special techniques for solving linear or non-linear systems of equations arising from the discretization of partial differential equations (Smith et al. 1996). The majority of the DD methods require intervention at a computing level that existing simulation models either do not provide (COMSOL, FEFLOW) or that is very difficult to implement (MODFLOW, MT3DMS, HYDROGEOSPHERE). Here we use a simplistic domain decomposition method, which has the advantage that it can be combined seamlessly with the majority of the existing simulation models.

According to the simplistic DD, the aquifer is initially treated as a single domain Ω and simulated using a coarse discretization, resulting in a coarse hydraulic head field h^c. Secondly the domain Ω is divided into N sub-domains $\Omega_1, \Omega_2, \dots, \Omega_N$ of very fine resolution. In this report we choose to divide the domain into orthogonal sub-domains, but the same method can be extended to any arbitrary sub-domain shape. We define the boundaries $\partial\Omega$ of domain Ω as external, and the artificial boundaries Γ_n as internal. Note that $\partial\Omega_n$ (i.e., the boundary of sub-domain) consists either of part of $\partial\Omega$ and Γ_n or exclusively of internal boundaries Γ_n. To assign boundary conditions to internal boundaries Γ_n we use an interpolation method $I_{\Omega\to\Gamma_n}$ where the unknown boundary conditions Γ_n are interpolated from the coarse solution h^c and the fine head field h_n^f for each sub-domain n is calculated independently. At the end of this process, h^f is the union of all individual simulation results $h^f = h_1^f \cup h_1^f \cup \dots \cup h_N^f$.

However, based on our simulation results already obtained, it was found that the resultant fine head field h^f exhibits discontinuities across the internal boundaries Γ_n. To alleviate this problem, we propose to offset the artificial sub-domains boundaries at a specified distance l_{of}, thus producing an overlapping zone of width $2l_{of}$. As shown in Figure 70 the aquifer domain can be of any arbitrary shape, and the simulated area of each sub-domain is defined as the intersection of domain Ω and the extended sub-domain boundary. In Figure 70, the solid red lines correspond to the non-extended sub-domain and the green dashed lined to the extended sub-domain after the offset operation.

Let i be a discrete point, located within the j non-extended sub-domain and also within M extended sub-domains (eg., Figure 70, the point i is located within the 2nd non-extended sub-domain and in the 8th extended sub-domain eg., $M = 1$). Each sub-domain returns a slightly different solution $h_i^j \neq h_i^k \neq h_i^{k+1} \neq \cdots \neq h_i^M$ as a result of the different boundary conditions, where $k \in [1, M]$ are the subdomain IDs. In the case of nonconforming meshes, an interpolated value is used for the heads $h_i^k, h_i^{k+1}, \dots h_i^M$.

Adminsitrative Record
Page 35100

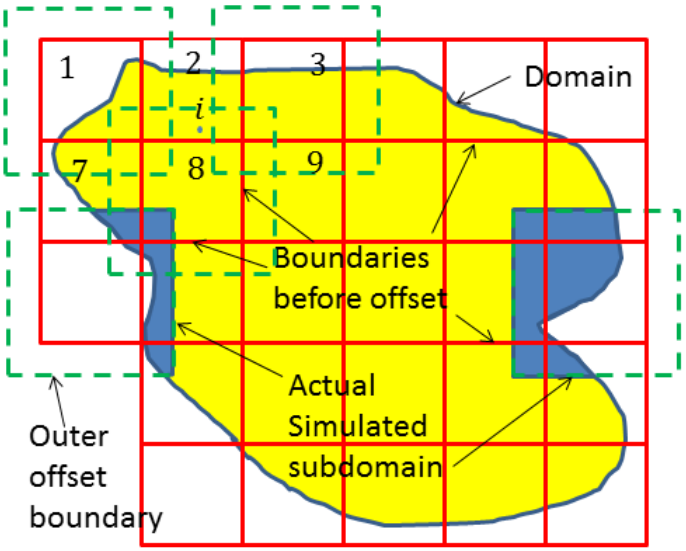

Figure 70. Schematic illustration of Simplistic Domain Decomposition Method.

Subsequently the $M+1$ head values can be averaged to obtain the head for point i. However, it has been observed that points closer to the boundaries have a larger head discrepancy from true head values. To counteract this problem, we propose a weighted average scheme, where the weights are taken proportional to the distance from the barycenter[13] of each sub-domain:

$$\bar{h}_i = \frac{1}{\sum_n^M w_i^n} \sum_{n=0}^{M} w_i^n h_i^n; \qquad n \in [j, k, k+1, \dots, M] \tag{Eqn. 8}$$

where h_i^n is the calculated head of point i based on the solution of sub-domain Ω_n and w_i^n is the weight of point i with respect to sub-domain n, calculated by the following empirical formula:

$$w_i^n = \min\left\{ W\left({}^{l_x}/_2, l_{of}, |x_i - x_c^n| \right), W\left({}^{l_x}/_2, l_{of}, |x_i - x_c^n| \right) \right\} \tag{Eqn. 9}$$

where x_i, y_i are the coordinates of the point i, l_x, l_y are the sub-domain lengths before the offset operator along the x and y directions, respectively (see Figure 71 left), x_c, y_c are the coordinates of the barycenter of the orthogonal and W is a geometric function defined as:

$$W(l, d, x) = \frac{x - (l + d)}{-2d} \tag{Eqn. 10}$$

It can be seen from Eqn. 10, that input arguments l and d of Eqn. 10 are constants for each sub-domain. Eqn. 9 returns zero weights for the points that lie on the outer offset edges and weights of ones at the inner offset edges, with weights that vary linearly within the overlapping zone between these two numbers (Figure 71, Right). The negative weights outside the extended orthogonal are not used and

[13] The barycenter of an object is the center of Mass

Adminsitrative Record
Page 35101

they are ignored. The result of the above formulation is a smooth velocity field across the artificial boundaries.

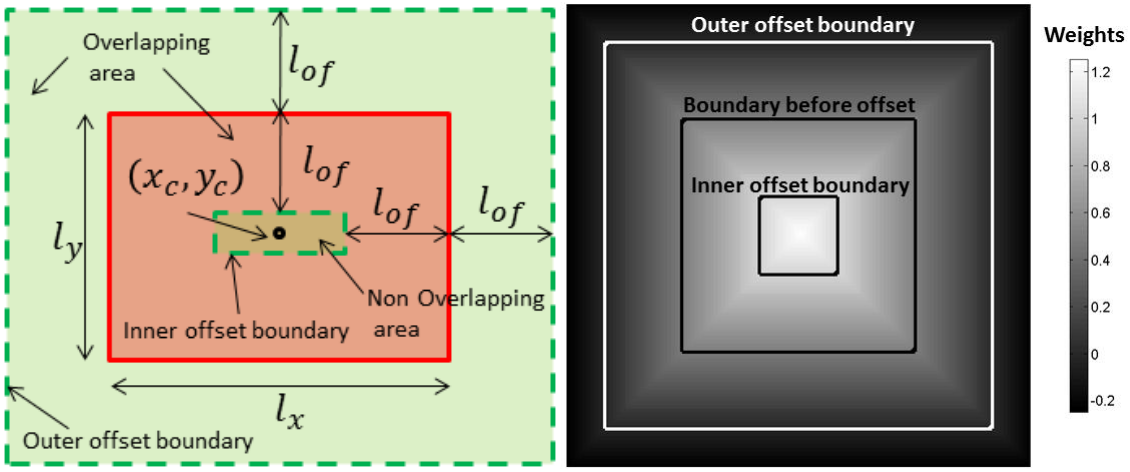

Figure 71. Left) Geometric definition of symbols involved in the weight calculation, Right) Weight function.

The velocity field is then used by transport simulation model of NPSAT for a highly efficient, streamline-based quasi-3D solution of an (arbitrary) solute transport equation yielding a pollutant unit response Functions (URF).

7.2.3 Transport Simulation: Streamlines

Neglecting transverse dispersion, the transport equation (Eqn. 6) is solved through an ensemble of one-dimensional streamline-based solutions focused on the CDSs rather than a fully three-dimensional solution. Obtaining a quasi-3D solution with the streamline approach specifically for CDS locations can be significantly more efficient than computing fully three-dimensional transient solutions over the entire groundwater nonpoint source contamination domain for time-horizons spanning decades to centuries (Thiele 2001).

Here, we define streamlines by their exit points on the CDS and use backtracking to the source area (Figure 72). Exit points of streamlines are distributed across each CDS such that each streamline represents a known fraction of flow into the CDS. Streamline exit points around a well screen CDS are organized in multiple horizontal layers, each with a finite number of exit points per layer: $N_s = N_l \cdot N_{ppl}$ where N_l is the number of layers, N_{ppl} is the number of exit points per layer and N_s is the total number of streamlines exiting on the well screen. Streamline contributions to the CDS BTC are weighted based on the exit velocity at the CDS.

Adminsitrative Record
Page 35102

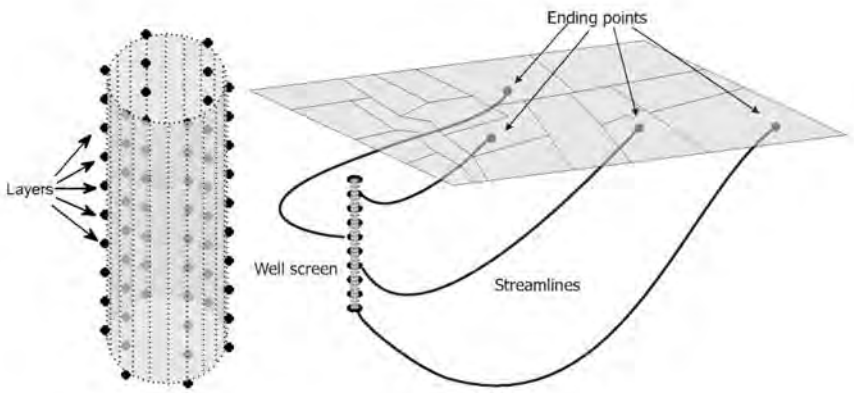

Figure 72. Distribution of exit points around the well screen.

The accuracy of the breakthrough curve simulation at the CDS is determined by the number of streamlines used relative to the (spatio-temporally variable) pollutant loading across the source area (source loading). For computational efficiency, a balance must be sought between accuracy and numerical efficiency. The choice of the number of streamlines used is application-specific and depends on the desired accuracy. Given typical uncertainties and inaccuracies associated with estimating groundwater flow parameters and source loading, a practical simulation goal is to obtain discharge (well) concentrations that have a numerical accuracy within 5% of the true mathematical solution or, alternatively, at the 5%–10% level of a problem-specific contaminant concentration level of interest (e.g., drinking water limits for nitrate, salinity), whichever is larger. Consider a nonpoint source concentration that varies between 10% (background) and 1,000% (intensive source) of the regulatory control level while recharge is relatively uniform. This is typical for nitrate and salinity pollution from agricultural landscapes (Harter et al. 2002; Burow et al. 2010). In this case, the scenario requiring the highest resolution occurs if 1% of the CDS source area has the maximum concentration of 1,000% of the regulatory control level, while the remainder of the source area recharges at a background concentration of only 10% of the regulatory control level. Hence, the high polluter adds ($1\% \cdot 1000\%$=) 10% of the regulatory concentration level to the background concentration at the CDS. The number of streamlines must be sufficiently large to ascertain that the procedure captures the 1% of the source area with high concentration. Generally, from 10^2 to 10^3 streamlines are therefore needed to properly simulate the BTC at such a CDS.

7.2.4 Streamline Computation

To define the streamline associated with each exit point on the CDS, a backward particle tracking is performed for each of the N_s exit points, and the positional and velocity vectors are computed for

Adminsitrative Record
Page 35103

$N_s \times N_{CDS}$ streamlines, where N_{CDS} is the number of CDSs included in the simulation area. Backward particle tracking for streamlines has two distinct advantages; transport is computed only for the part of the aquifer that is of interest to the simulation outcome (concentration hydrographs at contaminant sinks), and by using backward particle tracking to define streamlines, we avoid the so-called "weak-sink" problem in numerical solutions of (5) (Zheng and Wang 1999).

For each starting point $i_s \in [1, N_s]$ a backward particle tracking is performed until the particle intersects the water table, yielding a streamline $S_{i_s}^{i_{CDS}}$. In groundwater, the streamlines describe the time t for a particle to travel a certain distance s within the groundwater velocity field. Mathematically the streamlines are expressed by the following first-order, initial-value, ordinary differential equation:

$$\frac{d\vec{x}_p}{dt} = \vec{v}_p(\vec{x}_p, t) \qquad \text{(Eqn. 11)}$$

$$\vec{x}_p(t = \tau_p) = \vec{x}_{p_0}$$

where $\vec{x}_p$ is the position vector, $\vec{v}_p$ is the pore velocity vector, and $\vec{x}_p(t = \tau_p)$ is the starting point (e.g., exit point of streamline Figure 72). Eqn. 11 can be solved analytically (Pollock 1994) or numerically using any known numerical methods. In this report we used a hybrid numerical integration method. The method is a mix of the predictor-corrector scheme and the fourth order Runge-Kutta integration method. The method starts by calculating the velocity $\vec{v}_p$ at a given point $\vec{x}_p$. The velocity is computed through interpolation based on the head field. Next we estimate a new point $\vec{x}_{p+1}^{(1)}$ using an explicit Euler formula:

$$\vec{x}_{p+1}^{(1)} = \vec{x}_p + \vec{v}_p \cdot h \qquad \text{(Eqn. 12)}$$

Where h is a predefined step. According to the predictor-corrector scheme, the position $\vec{x}_{p+1}^{(1)}$ is improved iteratively using the corrector:

$$\vec{x}_{p+1}^{(i+1)} = \vec{x}_p + \frac{1}{2}(\vec{v}_p + \vec{v}_p^{(i)}) \cdot h; \quad i = 1,2,3 \qquad \text{(Eqn. 13)}$$

The iterations stop when the discrepancy between the positions of points i and $i + 1$ is smaller than a specified threshold.

Typically, based on our simulations, the predictor-corrector scheme converges after 2–5 iterations. However, it was observed that in some cases (i.e., very irregular geometry of mesh elements) the predictor-corrector scheme either does not converge fast or in very few cases does not converge at all. To counteract this, we employed a fourth order Runge-Kutta integration method when the iterations of the predictor-corrector integration exceed a certain number without converging (e.g., 10). In these cases the new position $\vec{x}_{p+1}^{(1)}$ is calculated using Eqn. 12 where $\vec{v}_p$ is the weighted average of four velocities along the direction of flow.

Adminsitrative Record
Page 35104

A critical step in the particle tracking method is the computation of velocity, based on a hydraulic head field. In general, velocity is defined as the negative gradient of head divided by the aquifer porosity (i.e., $v = -\boldsymbol{K}\nabla H/\theta$, where $\boldsymbol{K}$ is the hydraulic conductivity tensor, H is the hydraulic head, and θ is the porosity)**.** According to the finite element method a continuous hydraulic head $H(x, y, z)$ field is approximated as the weighted average of the heads H_i at specific discrete locations $i = [1,2, \dots, N]$ (i.e., the nodes of the finite element mesh) weighted by the shape functions $N_i(x, y, z)$, $H(x, y, z) = \sum_{i=1}^{N} N_i(x, y, z)H_i$. Similarly, the gradient of the hydraulic head is approximated as the weighted average of the heads H_i multiplied by the gradient of the shape functions $\nabla N_i(x, y, z)$. Since the hydraulic heads H_i are computed from the solution of the groundwater flow equation, the calculation of velocity is simplified to the calculation of the shape function derivatives. However in 3D models, shape function derivatives are very complex expressions of x, y, z. In this report we used exclusively prism elements (Figure 73) and isoparametric shape functions, which simplify the derivative calculations.

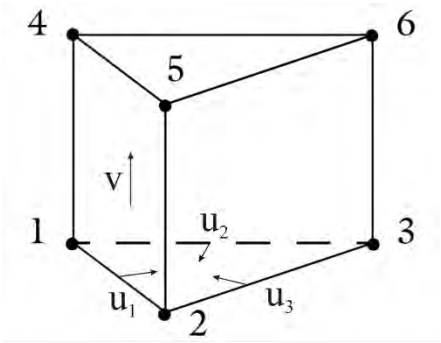

Figure 73. Prism element.

Isoparametric shape functions are defined on an element local coordinate system and in the case of prism elements are expressed as the following:

$$\begin{aligned} N_1 &= u_1\,(1-v)/2 \\ N_2 &= u_2\,(1-v)/2 \\ N_3 &= u_3\,(1-v)/2 \\ N_4 &= u_1\,(1+v)/2 \\ N_5 &= u_2\,(1+v)/2 \\ N_6 &= u_3\,(1+v)/2 \end{aligned} \qquad \text{(Eqn. 14)}$$

where $u_1 + u_2 = u_3$. Note that shape functions are defined in a local coordinate system $N(u_1u_2, v)$ and velocity computation requires the derivatives with respect to the physical coordinate system $\partial N(x, y, z)$. However the conversion is straightforward and is obtained by the following equation:

$$\partial\mathbf{N}(x, y, z) = \mathbf{J}^{-1}\mathbf{G}\cdot\mathbf{N} \qquad \text{(Eqn. 15)}$$

where $\mathbf{G}$ is the gradient operator in the local coordinate system $\mathbf{G} = [{}^{\partial}/_{\partial u_1}\ {}^{\partial}/_{\partial u_2}\ {}^{\partial}/_{\partial v}]^T$, $\mathbf{N} = [N_1, N_2, \dots, N_6]$ and $\mathbf{J} = \mathbf{G}\cdot\mathbf{N}\cdot[\mathbf{x}\ \mathbf{y}\ \mathbf{z}]$, where $\mathbf{x} = [x_1, x_2, \dots, x_6]^T$, $\mathbf{y} = [y_1, y_2, \dots, y_6]^T$, and $\mathbf{z} = [z_1, z_2, \dots, z_6]^T$.

Adminsitrative Record
Page 35105

In the above analysis we used linear shape functions. Higher order elements can be also used,at the expense of computational complexity and increased CPU runtime.

After the calculation of the streamlines, each one consists of a positional vector $\mathbf{x}_i$, and velocity vector $\mathbf{v}_i$. The positional vector $\mathbf{x}_i$ contains the distance along the streamline $S_{i_s}^{i_{CDS}}$ measured from the initial point i_s, and the velocity vector $\mathbf{v}_i$ contains the velocity magnitude that corresponds to positions of vector $\mathbf{x}_i$. Note that the last element of vector $\mathbf{x}_i$ is the key link of the streamline $S_{i_s}^{i_{CDS}}$ with the nonpoint source loading function L_j; $j \in [1, N_{land}]$, and eventually links each exit point on the CDS with a contamination source L_j.

7.2.5 Unit Response Function Approach

The linearization of the transport problem (Eqn. 6) allows for the application of the principle of superposition (Jury and Roth 1990); the concentration history at any streamline exit point on the CDS, due to a temporally variable source loading history at the associated source boundary, can be computed as a superposition of solutions of the so-called unit response functions (URF). URFs have been widely used for the simulation of rainfall-runoff processes (Saghafian 2006; Jukic and Denic-Jukic 2009), where the URF is known as unit hydrograph. Researchers also employed URFs as transfer functions to simulate solute transport in the unsaturated zone (Jury 1982; Jury et al. 1982, 1986; Jury & Roth 1990; Heng and White 1996; Stewart and Loague 2003; Jaladi and Rowell 2008; Mattern and Vanclooster 2010) and in watersheds (Botter et al. 2006). Here, the transfer function concept explored in Jury and Roth (1990) is interpreted as a transfer function across a finite-sized three-dimensional streamtube linking a fraction of the source area with a fraction of the CDS. Each streamline represents an infinite number of stream-filaments (particle paths) within the associated streamtube (Ginn 2002).

7.2.6 Transport Simulation along Streamlines

For each streamline, a one-dimensional transport model is applied to compute the URF. Generally, any transport model may be applied within the NPSAT framework to compute the URF provided that the superposition principle can be applied (e.g., Continuous Time Random Walk (Berkowitz et al. 2006), Fractional Advection Dispersion equation (Meerschaert et al. 1999), the tempered one-sided stable residence time density (Cvetkovic 2011), and others). Here, we use the one-dimensional Advection Dispersion Equation (ADE) (Jury and Roth 1990):

Adminsitrative Record
Page 35106

$$\frac{\partial c}{\partial t} = \frac{\partial}{\partial x}\left(D\frac{\partial c}{\partial x} - vx\right) \quad \text{(Eqn. 16)}$$

subject to:

$$\begin{aligned} &c(x,0) = 0 \\ &c(0,t) = 1\,;\, t > 0 \text{ (Heaviside step function input)} \\ &\left(\frac{\partial c}{\partial t}\right)_{x=x_{\max}} = 0 \end{aligned} \quad \text{(Eqn. 17)}$$

where D represents the effective macrodispersion, given by $D = \tilde{\alpha}_L v$. The macrodispersivity, $\tilde{\alpha}_L$ intrinsically accounts for the effects of aquifer heterogeneity within the streamtube represented by the 1D streamline. Consistent with field experiments (Gelhar et al. 1992) and numerical experiments (Green et al. 2010), the longitudinal macrodispersivity, $\tilde{\alpha}_L$ is scaled relative to the length of the streamline,

$$\tilde{\alpha}_L = f(L_s) \quad \text{(Eqn. 18)}$$

where L_s is the streamline length. The velocity varies along the streamline, and is calculated from the flow solution by the norm $v = \sqrt{v_x^2 + v_y^2 + v_z^2}$. The solution to Eqn. 16 is obtained numerically for each streamline. The streamline URF is computed from the resulting solution $c(t)$

$$URF = c(t) - c(t-1) \quad \text{(Eqn. 19)}$$

This ensures that the area of the URF is always equal to 1 (Figure 74).

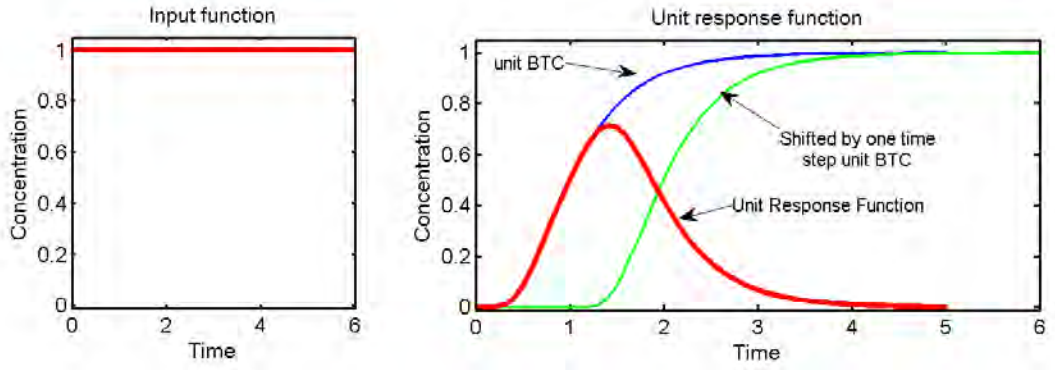

Figure 74. Calculation of Unit Response Function.

7.2.7 Unit Response Function Parameterization

Streamline URFs are archived for retrieval during the forecasting phase of the NPSAT. We found that the shape of the URFs is similar to that of common probability distribution functions (pdfs) found in the statistical literature and in statistical software. Both functions intrinsically integrate to a unit area. We

Adminsitrative Record
Page 35107

can therefore readily fit the empirically determined URFs to a library of pdf functions using existing software. Pdfs are typically defined by two to three parameters and the function itself. The fitting procedure used consists of a gradient based optimization method aiming at minimization of the error between the empirical and fitted URF. Rather than archiving on the order of $10^4 – 10^5$ bytes of data per CDS (10^2 or more datapoints per each of $10^2 – 10^3$ streamlines), this procedure reduces the archive to 3 to 4 numbers (2–3 parameters and one index to identify the function) per streamline or between 10^2 and 10^3 bytes of data per CDS. This allows for efficient data storage and forward modeling in applications to large groundwater regions, where the number of CDSs may range from 10^4–10^6 or even higher.

7.2.8 Summary of Construction Phase

Use of the URF approach computationally decouples the transport process from the nonpoint source loading process. URFs can be computed a priori without knowledge of the actual nonpoint source concentration history. We call this the NPSAT construction phase. The NPSAT construction phase requires the following steps (Figure 75):

- Geospatial mapping of the individual land use parcels and their (non-transient, average) recharge, of the CDSs (e.g., wells) and their (non-transient, average) discharge, and of other boundary conditions
- Computation of a detailed three-dimensional steady-state groundwater velocity field using a high-resolution numerical solution to Eqn. 7 with distributed recharge, groundwater pumping, and groundwater discharge to streams
- Geospatial mapping of the desired distribution of streamline exit points on the set of CDSs
- Backward computation of streamlines from their CDS exit points to the water table (source area)
- Computation of a URF as the one-dimensional solution of Eqn. 16 separately for each streamline, given a unit input step function, and fitting a parametric function to this empirically obtained URF at each streamline, a step that drastically reduces the data storage requirements.

The construction phase yields a geospatial database of the location of CDSs (wells, drains, springs, stream reaches), an identification of associated streamlines, the parameters and a code identifying the form-function of the URF for each streamline, and each streamline's recharge and discharge (beginning and end) location.

Adminsitrative Record
Page 35108

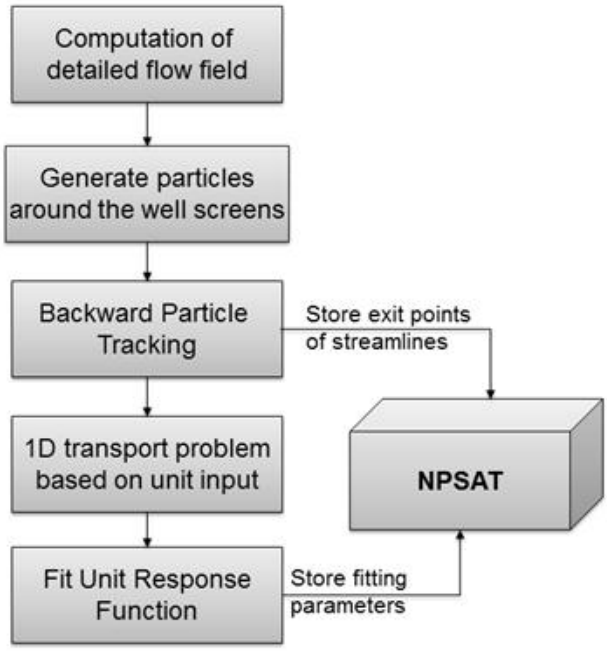

Figure 75. Construction phase of the NPSAT.

7.2.9 Implementation Phase: Computing Breakthrough Curves at CDSs

In the implementation phase of the NPSAT, the BTC of a CDS is computed by convoluting each streamline-specific unit response function with the actual, location-specific nonpoint source loading function, then performing a flux-weighted integration of streamline-output concentrations at time t over all streamlines exiting in a specific CDS (model prediction phase). Suppose that N_s streamlines and associated unit response functions $\mathbf{URF} = \{URF_1, URF_2, \ldots, URF_{N_s}\}$ were computed for the $i_{CDS}{}^{\text{th}}$ compliance discharge surface. The source loading functions associated with the $\mathbf{URF}$ are denoted as $L_i; i \in [1, N_s]$. For each streamline, the concentration history is obtained by convolution:

$$BTC_j(t) = \sum_{d=0}^{t} L_j(t-d) \cdot URF_j(d) \qquad \text{(Eqn. 20)}$$

where d increases in the summation at time step intervals and t is the total runtime of the transport model. Eqn. 20 is the numerical approximation of the general convolution operator between two functions f and g expressed as $f * g = \int_0^t f(\zeta)g(t-\zeta)d\zeta$. After the calculation of the concentration history at each streamline, the BTC of the CDS is computed from:

Adminsitrative Record
Page 35109

$$\overline{BTC}_{i_{CDS}}(t) = \frac{1}{w_1 + w_1 + \cdots + w_{N_s}} \sum_{j=1}^{N_s} w_j \cdot BTC_j(t) \qquad \text{(Eqn. 21)}$$

where w_j is the weight that represents the amount of flow that corresponds to each streamline.

Unlike in watershed NPS modeling, where the CDS output of individual stream reaches or tributaries is effectively integrated at the watershed outlet (Basso et al. 2010), the solute output at individual groundwater well CDSs does not further mix among CDSs (similarly in the case where a large number of low order stream reaches are considered separately). Instead, the BTCs provide the basis for constructing time-dependent pollutant exceedance probability distribution functions (pdfs) across user-specified specific population sets of CDSs (e.g., domestic wells, irrigation wells, drinking water wells, stream reaches) within the modeling domain. This stochastic analysis is the final step in the NPSAT process (Figure 76).

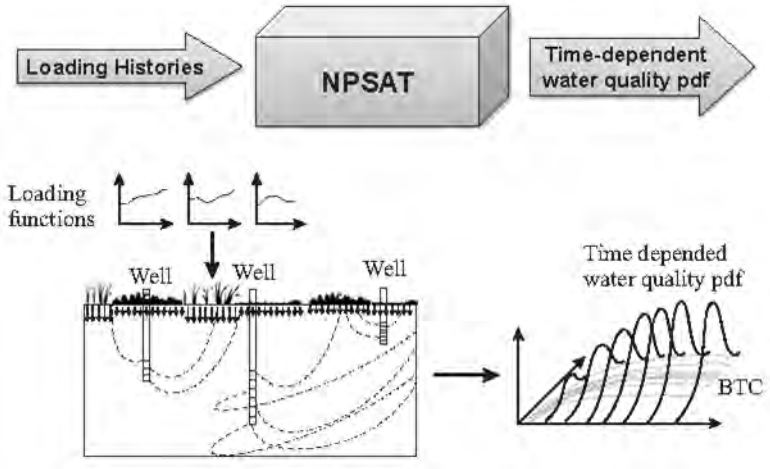

Figure 76. Simulation phase of the NPSAT.

Adminsitrative Record
Page 35110

7.3 Approach to Generate Representative Groundwater Velocity Distribution

One of the main assumptions of the proposed modeling approach is the aquifer flow field is steady state. While this assumption might not be valid when a monthly or even seasonal temporal scale for the analysis is used, it is justifiable with the rationale that the model will be used for prediction at the annual temporal scale (see justification in subsection 7.2.1 Conceptual approach).

To choose a representative steady state flow field for the study area we used two criteria: 1) the change of storage volume in the aquifer for the selected period should be as small as possible (i.e., the volume of water that entered the groundwater aquifer must be equal to the losses) and 2) the various stresses for the selected period have to correspond with the average stresses for the study area.

7.3.1 Steady-state Model for Tulare Lake Basin

For the TLB the analysis is based on the Central Valley Hydrologic Model (CVHM) (Faunt 2009). The CVHM is a transient state groundwater model, with a monthly time step, coupled with the FARM process (Schmid et al. 2006) which is used as a water budget tool to estimate the groundwater stresses due to agricultural practices (e.g., groundwater recharge and pumping, etc.). The CVHM simulates the groundwater flow for the entire Central Valley aquifer for the years 1962 – 2003. According to CVHM transient simulation, the storage in the aquifer decreases over the 41-year simulation period. Figure 77 illustrates the cumulative change in groundwater storage for four basins of the Central Valley. The violet line corresponds to the TLB. It can be seen that, despite the general declining trend, there is a two-year period (1995 – 1997) where the change in storage is rather negligible (see red oval Figure 77). The next requirement is that the stresses need to be representative of the study area. Figure 78 shows the simulated water budget from the CVHM. In particular it shows the agricultural pumping, the surface water deliveries, the water delivery requirements for irrigation (groundwater plus surface water deliveries), and the landscape recharge. Note also that three years, 1975, 1990, and 1998, have been designated as typical, dry, and wet, respectively. Interestingly, for the years 1996 and 1997 the simulated agricultural pumping, surface water deliveries, total delivery requirements, and the landscape recharge are very close to those of the typical year (compare the ends of the arrows which point to the amount of acre feet per year for the typical year 1975, and the year 1996 in Figure 78).

Adminsitrative Record
Page 35111

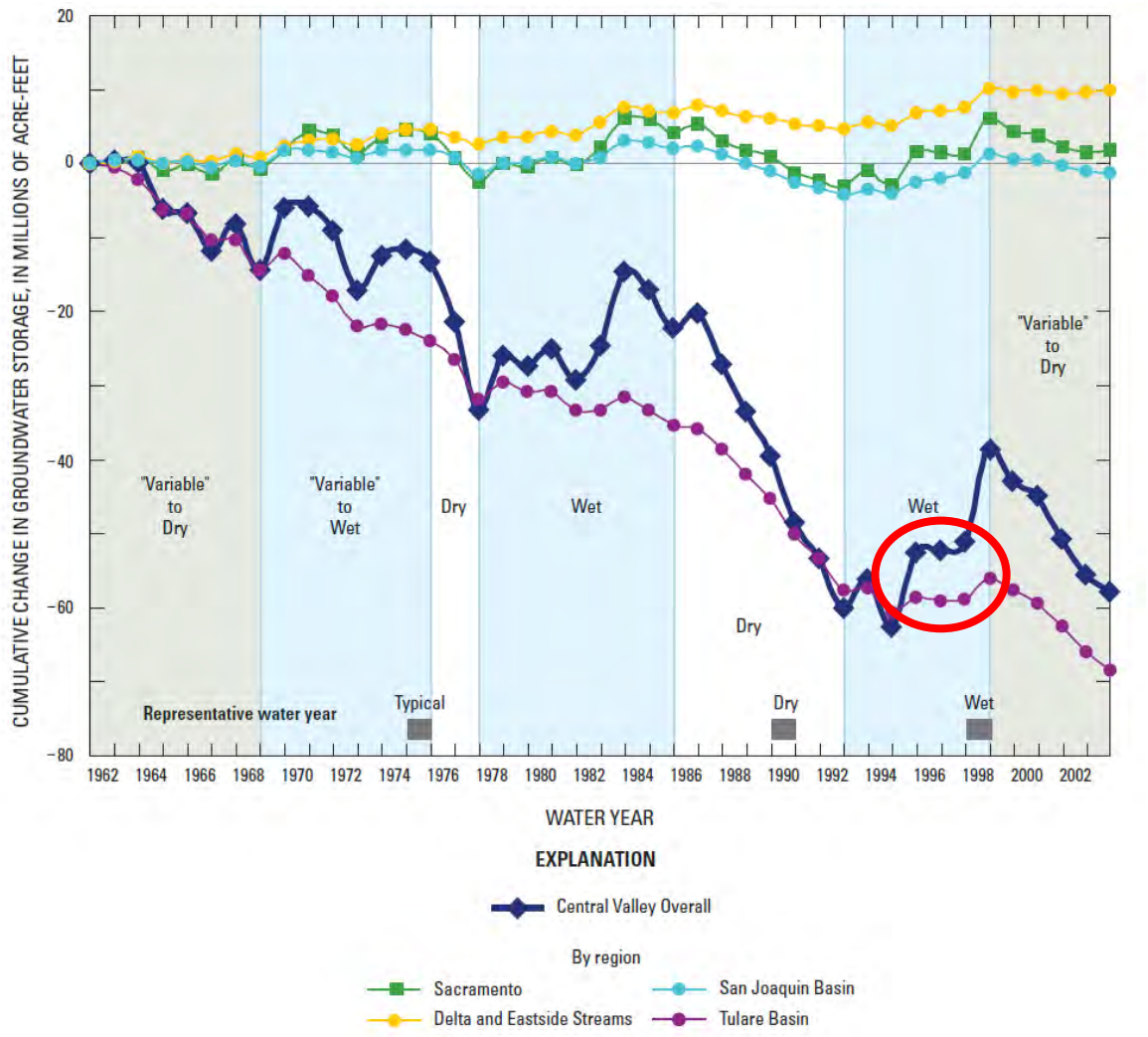

Figure 77. Simulated cumulative annual changes in aquifer storage (reprinted with permission from: Faunt 2009, p. 77, Figure B9.). Red circled area indicates the years chosen to represent steady-state stress conditions.

Adminsitrative Record
Page 35112

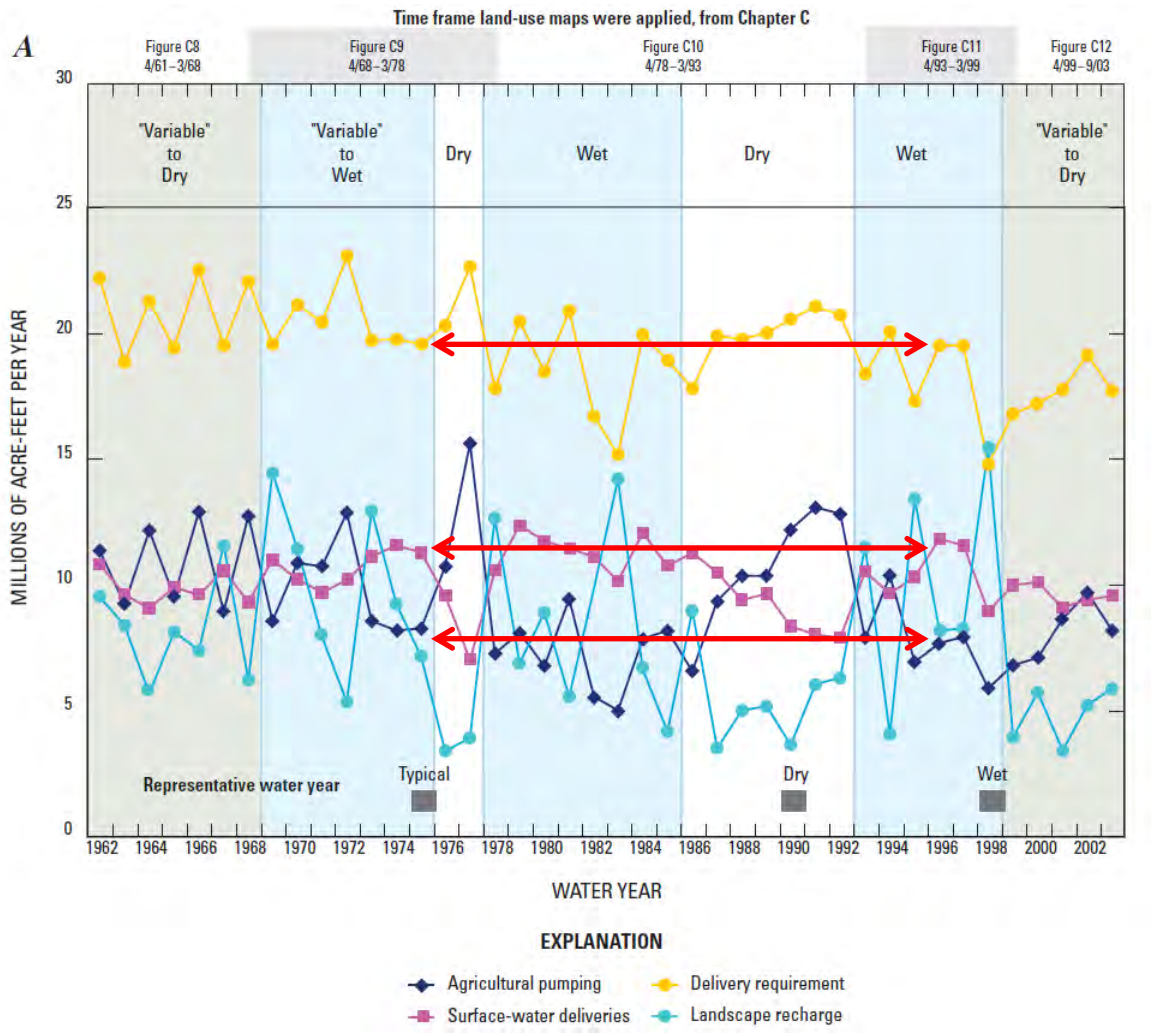

Figure 78. Water budget of CVHM, where "delivery requirement" is the irrigation water requirement from both groundwater and surface water sources. (Reprinted with permission from: Faunt 2009, p. 73, Figure B6.)

For the simulation we chose the water year spanning Oct. 1996 – Sep. 1997 as the stresses remain nearly constant compared to the previous year.

However, the CVHM model is a transient state model with a monthly step, where the head distribution, groundwater recharge, and groundwater pumping, vary over the year. To obtain a steady head field we average the monthly rates for the water year 1996. The mean hydraulic head relative to mean sea level is shown in Figure 79. The heads vary between -10 to 150 m (-33 to 492 ft). The head field is one of the main drivers of the velocity field and how the contaminants are transported through the aquifer. Therefore an accurate representation of head is very important. Figure 80 shows the cumulative distribution of the discrepancy between the averaged head field and each monthly head field of the year 1996. The discrepancy between the transient simulated head values and averaged head field is less than 1 m for 80% of the head values and less than 2 m for 90%.

Adminsitrative Record
Page 35113

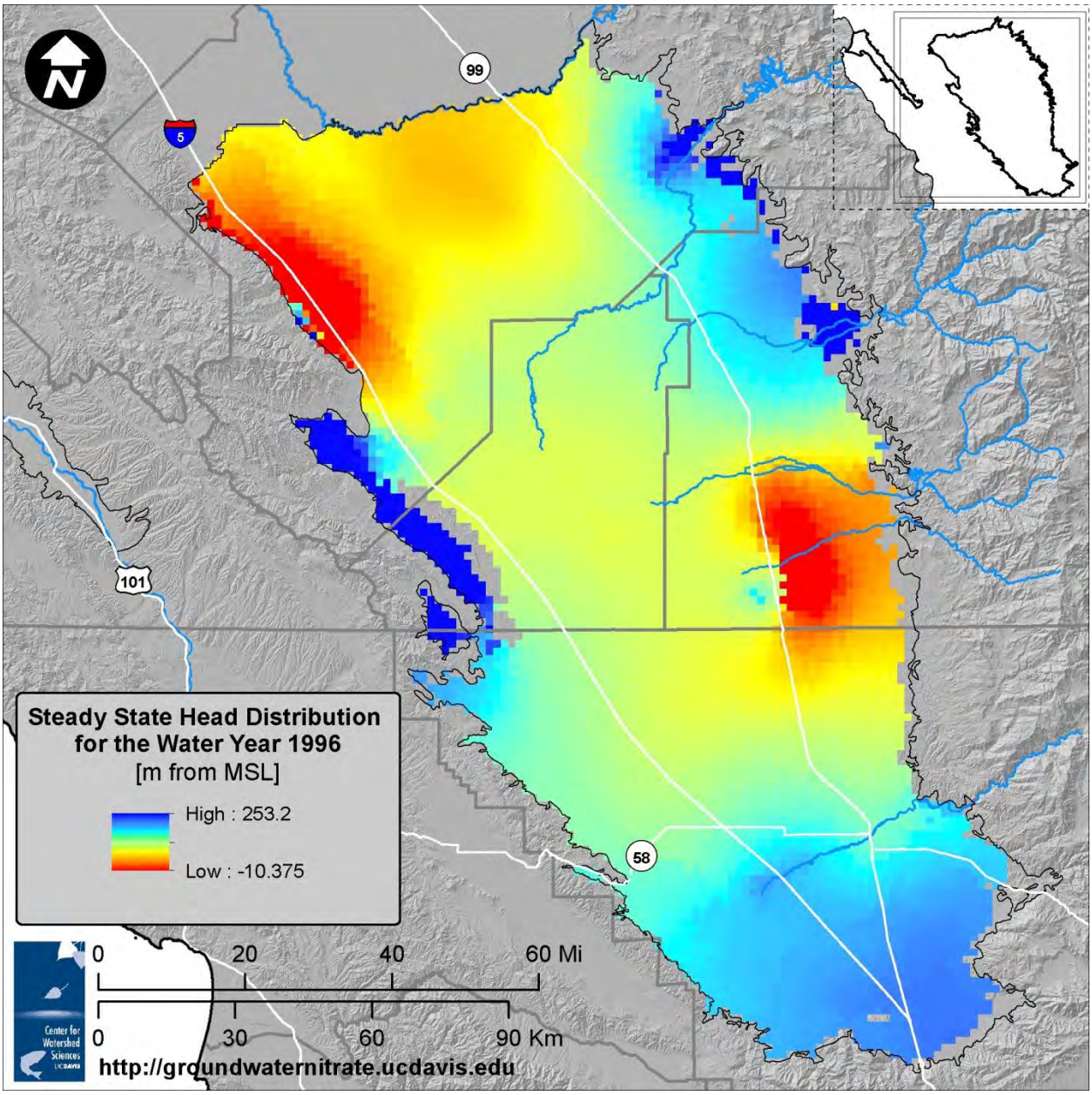

Figure 79. Steady state head distribution for the water year 1996.

Adminsitrative Record
Page 35114

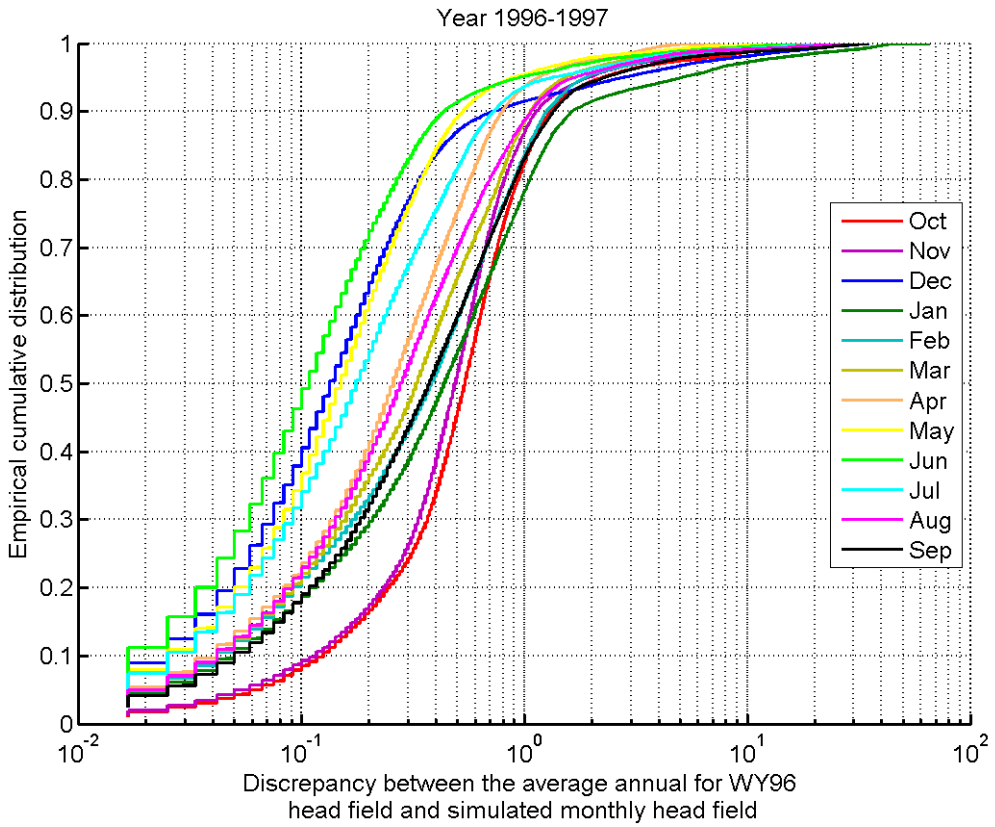

Figure 80. Comparison of averaged heads against the monthly heads used for averaging.

7.3.2 Conceptual Background

Well Generation Algorithm

The CVHM uses finite difference approximation with discretization of one square mile. Therefore various stresses such as groundwater recharge, groundwater pumping, and stream interactions, are assigned to the center of each cell, and the assigned values correspond to aggregation of all stresses located in each cell (e.g., the pumping rates of all the wells that are found in a square mile). To compute BTCs for wells with typical characteristics (i.e., pumping rates, well depths and screens that are closer to reality), in the refined model we developed an algorithm for generating wells with representative pumping rates, screen length, and depth. In fact the Well Generation Algorithm (WGA) is an automated way to distribute the well stresses back to individual wells. Since the location of the real wells is unknown, we use the WGA for generating random locations and random pumping rates, which satisfy the overall mass balance and honor the well characteristics, such as pumping rates, screen depth, and length typical of the study area. The input data for the WGA are the empirical distribution functions (ecdfs), which describe the well pumping rates, screen lengths and depths. The ecdfs used in this report are illustrated in Figure 81, Figure 82, and Figure 83.

Adminsitrative Record
Page 35115

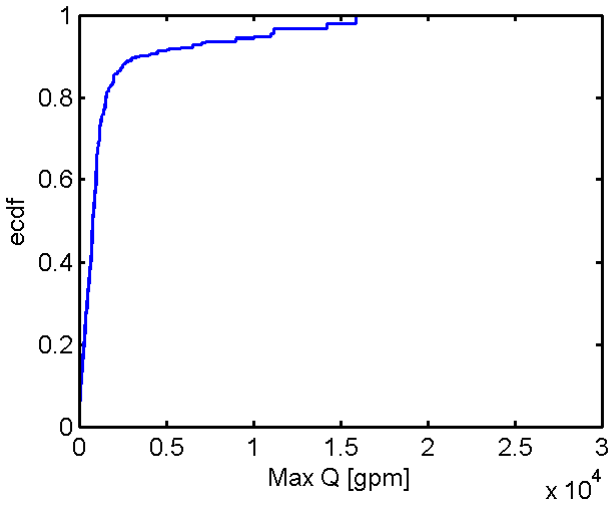

Figure 81. Empirical cumulative distribution function of maximum well pumping rate.

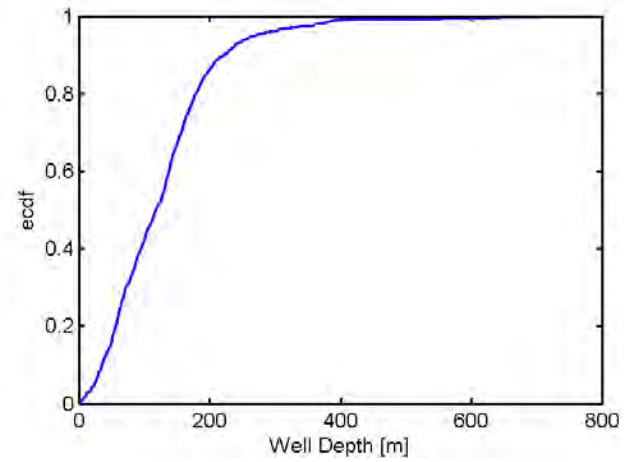

Figure 82. Empirical cumulative distribution function of well depth [m] (distance between water table and bottom of the well screen).

The ecdf of pumping rates is shown in Figure 81. Note that the data correspond to the maximum capacity of each well in the sample and not the actual pumping rate, which is unknown. In addition, note that most of the wells, if they operate at their maximum design capacity, they do so approximately 5 months per year (dry season) while during the remaining months, they either do not operate or operate at reduced rates. Based on expert opinion, it was suggested to sample from a subset of the full distribution, excluding the extremely large rates (>0.9) or negligible rates (<0.05). Note also that, in the simulation, we are interested in the yearly average rates, therefore the sampled rates were further reduced to correspond with yearly rates.

Adminsitrative Record
Page 35116

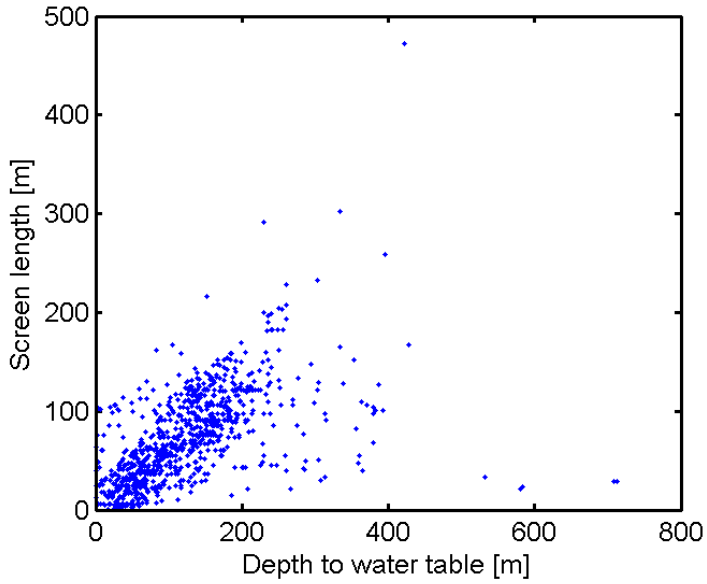

Figure 83. Correlation between well depth and screen length.

The steps of the WGA to generate locations and pumping rates are outlined as follows and explained in more detail below:

I. Choose a cell with assigned pumping rate
    1. Generate a well location
    2. Generate pumping rate and keep track of cumulative pumping
    3. If cumulative pumping is less than the cell assigned pumping rate, repeat steps 1 and 2
    4. Else, subtract the difference from the nearest cell, and proceed to the next cell

II. Generate random depth and screen length based on the distribution

III. If the depth is greater than 100 m assign the depth to a well with large production rate

IV. If the depth is less than 100 m assign the depth to any well

V. Check that there is 3 m (10 ft) of screen for every 100 gpm in coarse material. If not repeat steps II-V.

According to the pseudocode for each cell (square mile) of the CVHM discretization with an assigned pumping rate, we randomly generate wells within the cell using the formula $\mathbf{x} = \mathbf{x}^{min} + \mathbf{r}(\mathbf{x}^{max} - \mathbf{x}^{min})$ which generates uniformly distributed numbers within the specified limits (e.g., cell extent) where $\mathbf{x} = [x, y]$ are the coordinates of the generated well and $\mathbf{r}$ is a vector of two random numbers from the uniform distribution between 0 and 1. In addition, we specified a minimum threshold distance equal to 300 m, so that the wells are not in close proximity. The generation of the wells is a sequential process and each new well is accepted only if the distance with the closest existing well is greater than the threshold. To assign pumping rates, we sample randomly from a subset of the empirical cumulative distribution function. When the cumulative pumping rate of the generated wells per cell exceeds the

Adminsitrative Record
Page 35117

rate of the cell, which is specified by the CVHM, we proceed to the next cell. However, to maintain the water budget and at the same time to honor the pumping ecdf, the difference between the total cumulative pumping rate and the specified CVHM rate, is subtracted from the nearest cell to the last generated well in the current cell. Based on the above approach the number of wells per cell (i.e., wells per square mile) is dictated first by the cell pumping rate, (e.g., the higher the rate the more wells will be generated) and secondly by the shape of the ecdf.

In total, the algorithm generated 5,486 wells within the TLB study area. It can be seen (Figure 84) that the average number of wells per square mile is 4 while 80% of the simulated wells are found at a density less than or equal to 5 wells per square mile. This is also in agreement with reality, according to expert opinions. The simulated well locations are illustrated in Figure 85. The regional spatial distribution of the simulated wells is dictated by the parent model, here CVHM, and the CVHM pumping fluxes for the representative period chosen for the steady-state flow field (here: 1996). The WGA introduces an artificial variability at the scale of and within the CVHM grid cells. Due to the much finer discretization, the spatial local distribution of the wells obtained from WGA is therefore different and more detailed than with CVHM.

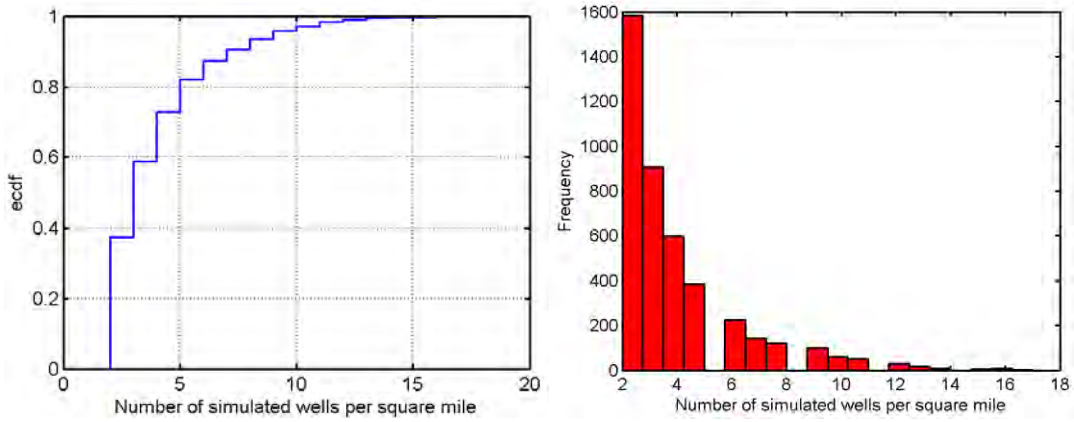

Figure 84. Left) Empirical cumulative distribution function of the number of wells per square mile. Right) histogram of the number of simulated wells per square mile.

Adminsitrative Record
Page 35118

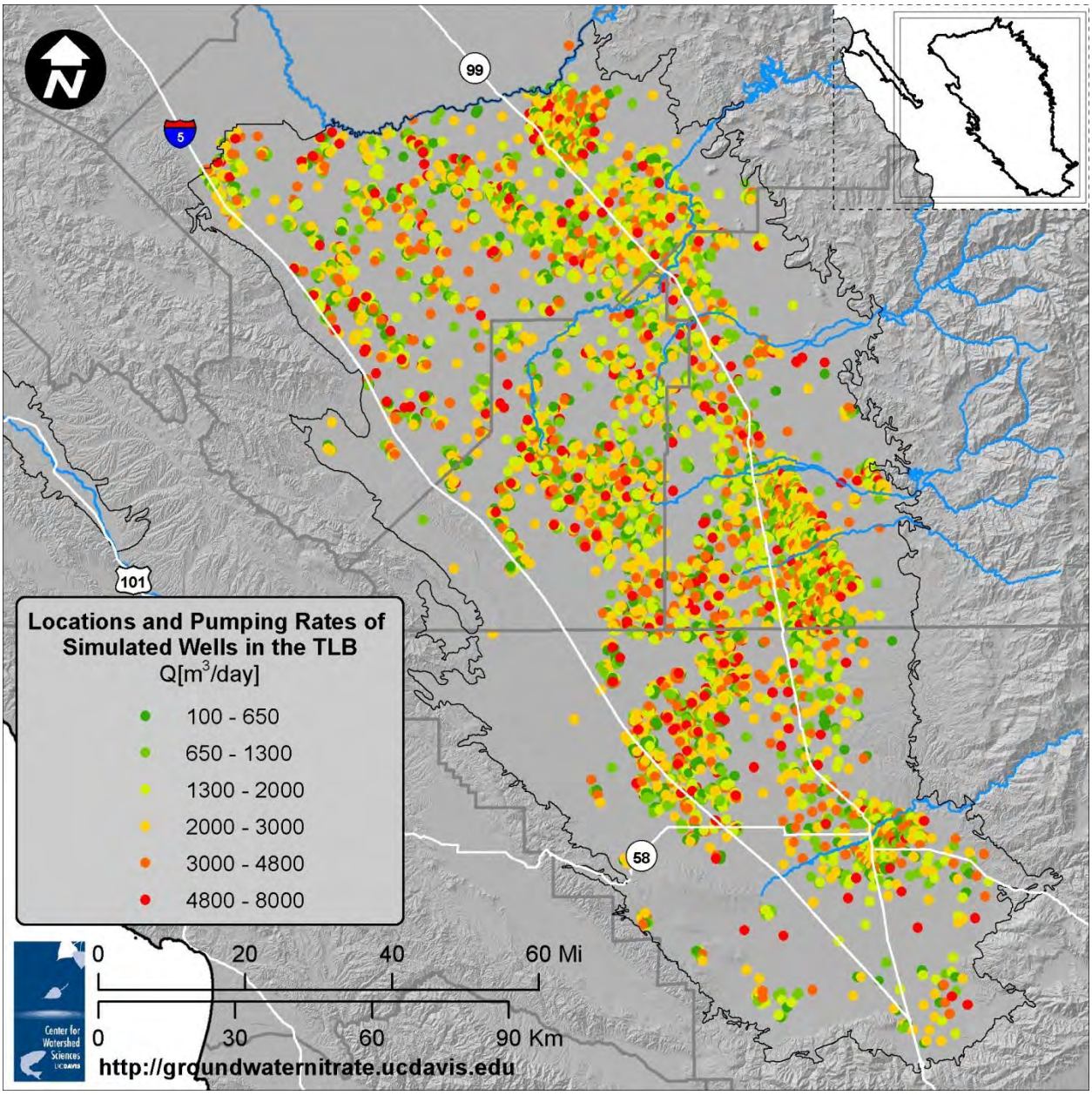

Figure 85. Locations and pumping rates of simulated wells in the TLB study area.

The distribution of pumping wells in Figure 85 reflects long-term average agricultural and urban water extractions, as represented by the year 1996 water budget (see discussion above and Figure 78). Under "normal year" conditions, surface water supplies along the easternmost portion of the TLB are sufficient to supply most irrigation water. Hence, most pumping wells in the easternmost part of the TLB are for urban uses (e.g., City of Fresno, City of Visalia, City of Porterville).

The distribution of pumping rates of the simulated wells is illustrated in Figure 86. Since we used a subset of the original distribution, the simulated distribution does not match the data, yet the median

value is approximately equal to 800 gpm, assuming a 5-month pumping period, which is a reasonable assumption for the particular study area.

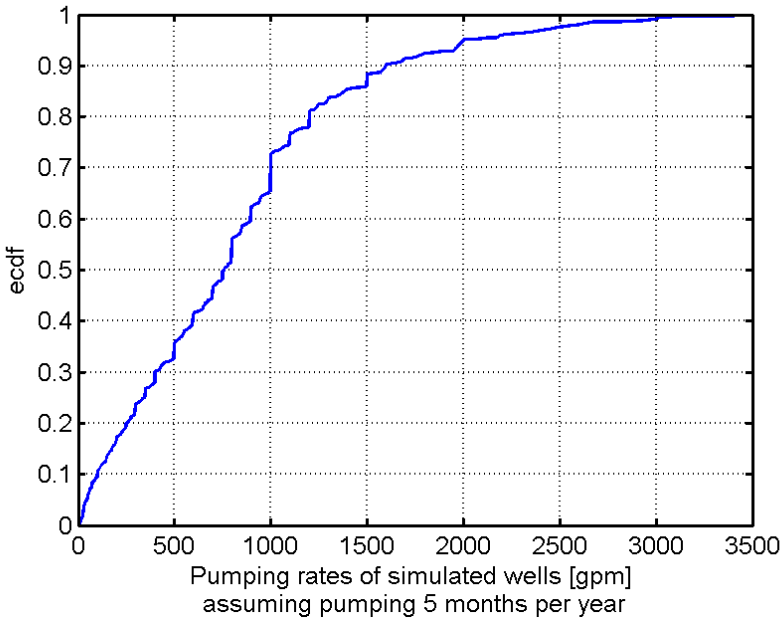

Figure 86. Empirical cumulative distribution function of simulated pumping rates.

After the generation of well locations and pumping rates, we assigned depth and screen lengths. Similarly the ecdf of well depths (Figure 82) and the correlation between screen lengths with depth (Figure 83) are used to generate random values. However special treatment is required to avoid assigning very large pumping rates to shallow wells. To do so we imposed a to sample from the upper half of the depth distribution when the pumping rate exceeds the median pumping rate of the pumping rate ecdf. However, when the rate is less than the median we use the full range of the distribution. In addition to pumping rate generation, special treatment is required for the screen length generation. Typically the shallow wells have shorter screen lengths than the deep wells. Figure 83 shows the depth of the well against the screen lengths. Ideally, an empirical bivariate probability density function is computed; however, this requires a large amount of data. Here we choose a two-step approach where, for each well, we first generate the depth and then the screen length. During the screen length generation, instead of sampling from the full distribution, we sample from a narrow range of the distribution. For example, suppose that d is the depth generated by sampling the depth distribution (Figure 82). To assign screen lengths we identify the samples with depth within the range $[r - d, r + d]$; hence, we use only a limited number of samples instead of the full distribution and construct a local ecdf, which describes the distribution of the well screen lengths in the vicinity of the selected depth d (Figure 87). The range r depends on the density of sample points (wells) around d and it is determined in an automated fashion. For each well, the algorithm starts with a very small value for the range r and increases the range until a sufficient number of sample points are found to construct a local ecdf.

Adminsitrative Record
Page 35120

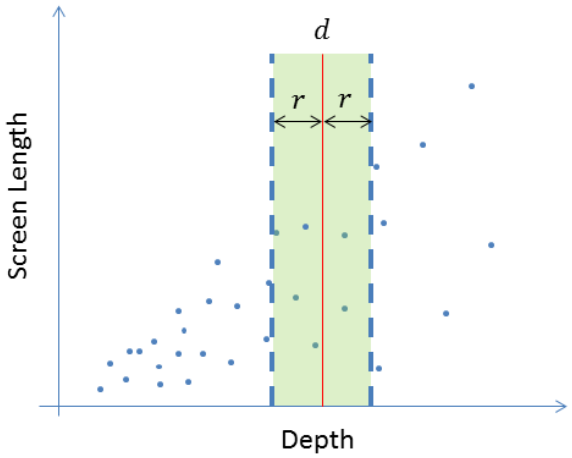

Figure 87. Sampling range for screen length.

Based on the depth and screen length distributions we assigned random well depths and screen lengths to the simulated wells that were generated in the previous step, according to their pumping rates. In order to generate an unbiased depth-screen length distribution, first we split the simulated wells into two groups. One group includes all the wells with simulated pumping rates greater than the median (i.e., 800 gpm), while the second group actually contains all the wells. Then we generate random depth d and random screen length based on the ecdfs. If the generated depth d is greater than 100 m (328 ft) then the well is designated as deep and the depth is assigned randomly to one of the wells that belongs to the first subgroup (of large production wells) and the well is removed from both sets. If the generated depth is less than 100 m then the pair of depth-screen length is assigned arbitrarily to one of the wells of the second group which contains all the wells. It can be seen that it is possible for wells with relatively small pumping rates to have large screen lengths, yet the suggested approach does not allow large wells to have small screen lengths. At the same time, since all the distributions are sampled independently, we avoid introducing bias. This becomes apparent by comparing Figure 82, which shows the ecdf of depths based on the real data and Figure 88 that shows the ecdf of the simulated well depths. The correlation between well depth and screen length is illustrated in Figure 89, which is very similar to the correlation of the real data (Figure 83). Last, we check if the material between the screen lengths can support the assigned pumping rate. We utilized an empirical formula where, for each 100 gpm we require at least 3 m (10 ft) of coarse material (e.g., K> 10 m/day (33 ft/day)). If the well does not meet this restriction, we repeat the depth and screen length generation.

Adminsitrative Record
Page 35121

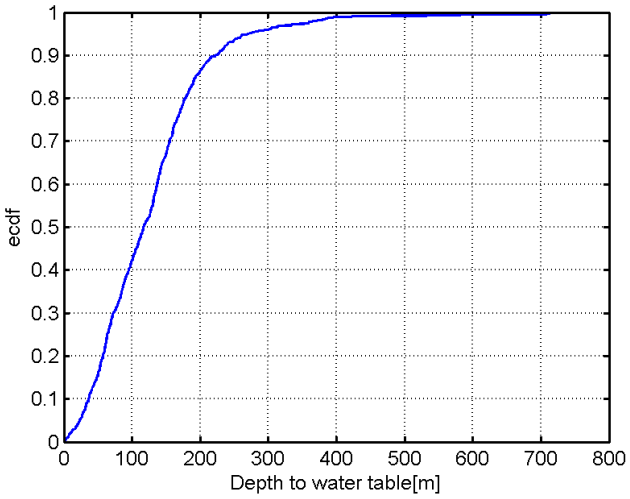

Figure 88. Empirical cumulative distribution function of Simulated well depth (distance between simulated water table and bottom of the well).

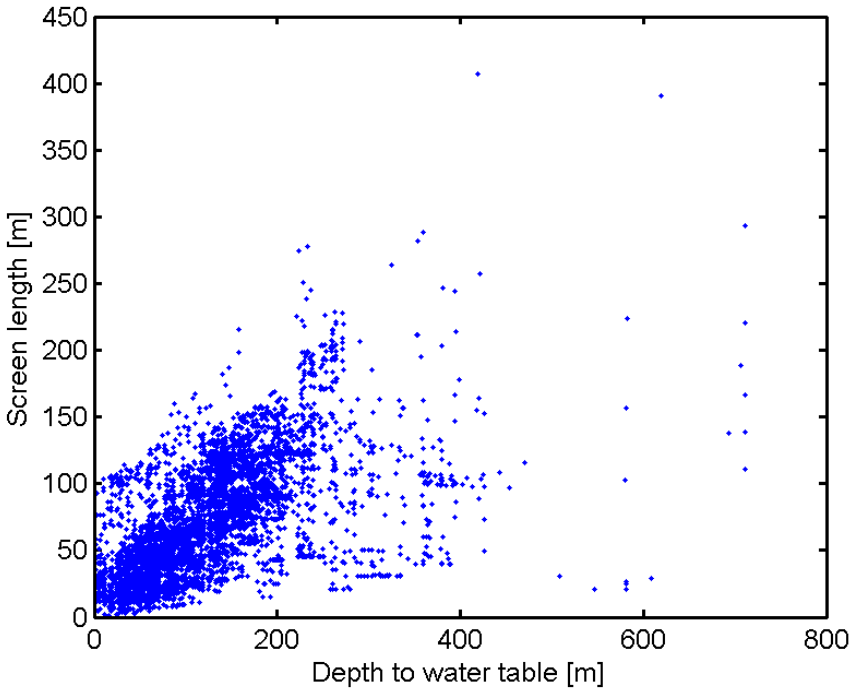

Figure 89. Correlation between simulated well depths and screen lengths.

Adminsitrative Record
Page 35122

7.3.3 Groundwater and Transport Simulation

After the generation of wells, all the additional stresses, which were defined as monthly rates, such as groundwater recharge stream leakage interactions, were averaged to obtain representative yearly rates. The hydraulic conductivity values and the ratios between horizontal and vertical conductivities were obtained from the input files of CVHM model.

To simulate the steady state flow field that corresponds to the water year 1996, we use finite element discretization, where the discretization varies from few meters ~10 m (33 ft) near wells to several hundred meters. Figure 90 illustrates an example of the fine discretization that was used in the steady state solution. The red cells correspond to the cells of the CVHM (i.e., each cell is 2.6 km (1 sq.mi)). For the vertical discretization we used 25 layers as opposed to the 10 layers used in CVHM.

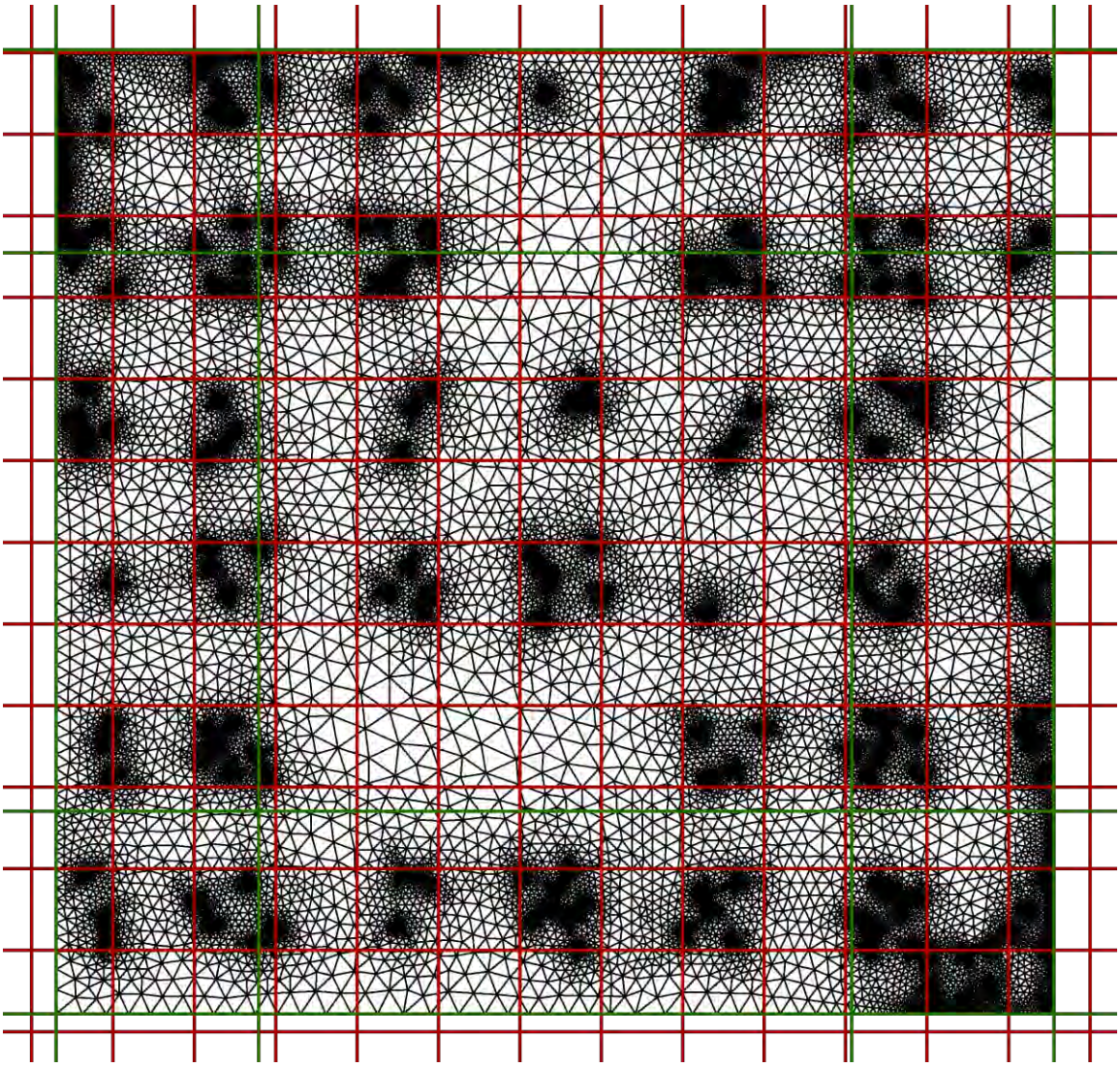

Figure 90. Discretization based on finite elements. The red lines correspond to the cells of the CVHM model. The dense black areas indicate the highly detailed discretization around the wells.

Adminsitrative Record
Page 35123

To obtain a steady state solution at this level of discretization, we employed the simplistic domain decomposition method presented above. According to the method, the domain was first divided into overlapping subdomains. Figure 92 shows the division of the study area into overlapping subdomains. In total, the subdomain was divided into 97 overlapping subdomains. The dimension of each subdomain was approximately 20 x 20 km., while the width of the overlapping zone was constant and equal to 2 km.

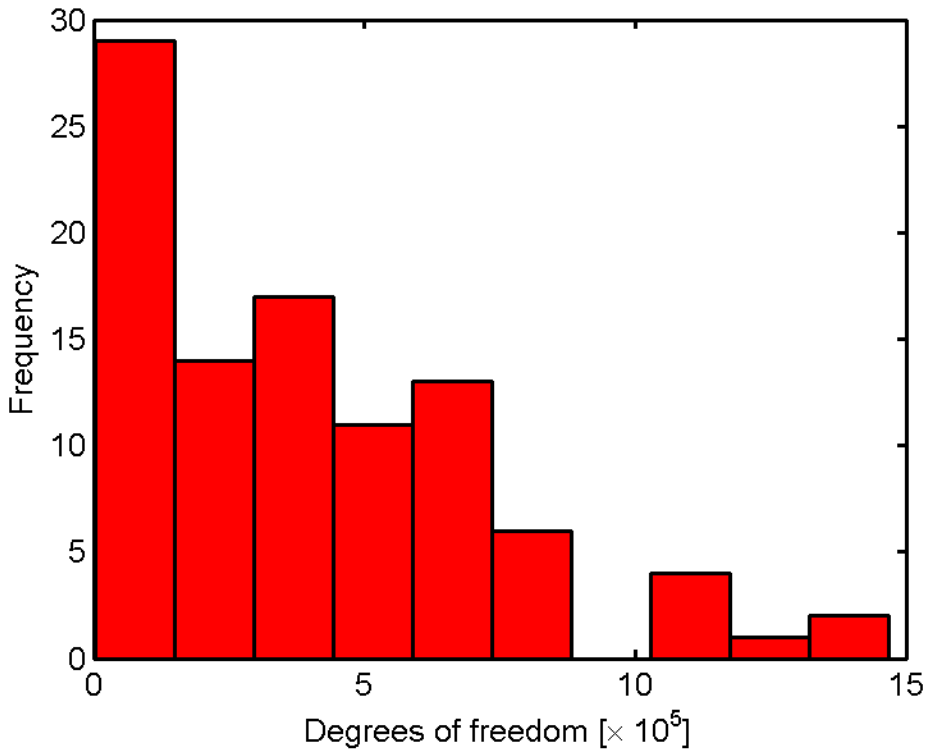

Figure 91. Histogram of degrees of freedom for the simulated subdomains.

The degrees of freedom for each subdomain vary from 3.1×10^3 to 14.8×10^5 (Figure 91). The total number degrees of freedom is 39×10^5. Note that in our approach the meshes of adjacent subdomains were not conforming (i.e., within the overlapping zone the nodes of two subdomains do not coincide). According to the simplistic domain decomposition method, each subdomain was solved independently. The boundary conditions for the inner boundaries of each subdomain were interpolated from the coarse solution of CVHM. To obtain a divergent free velocity field we used the proposed (section 7.2.2) weighted averaging scheme.

Adminsitrative Record
Page 35124

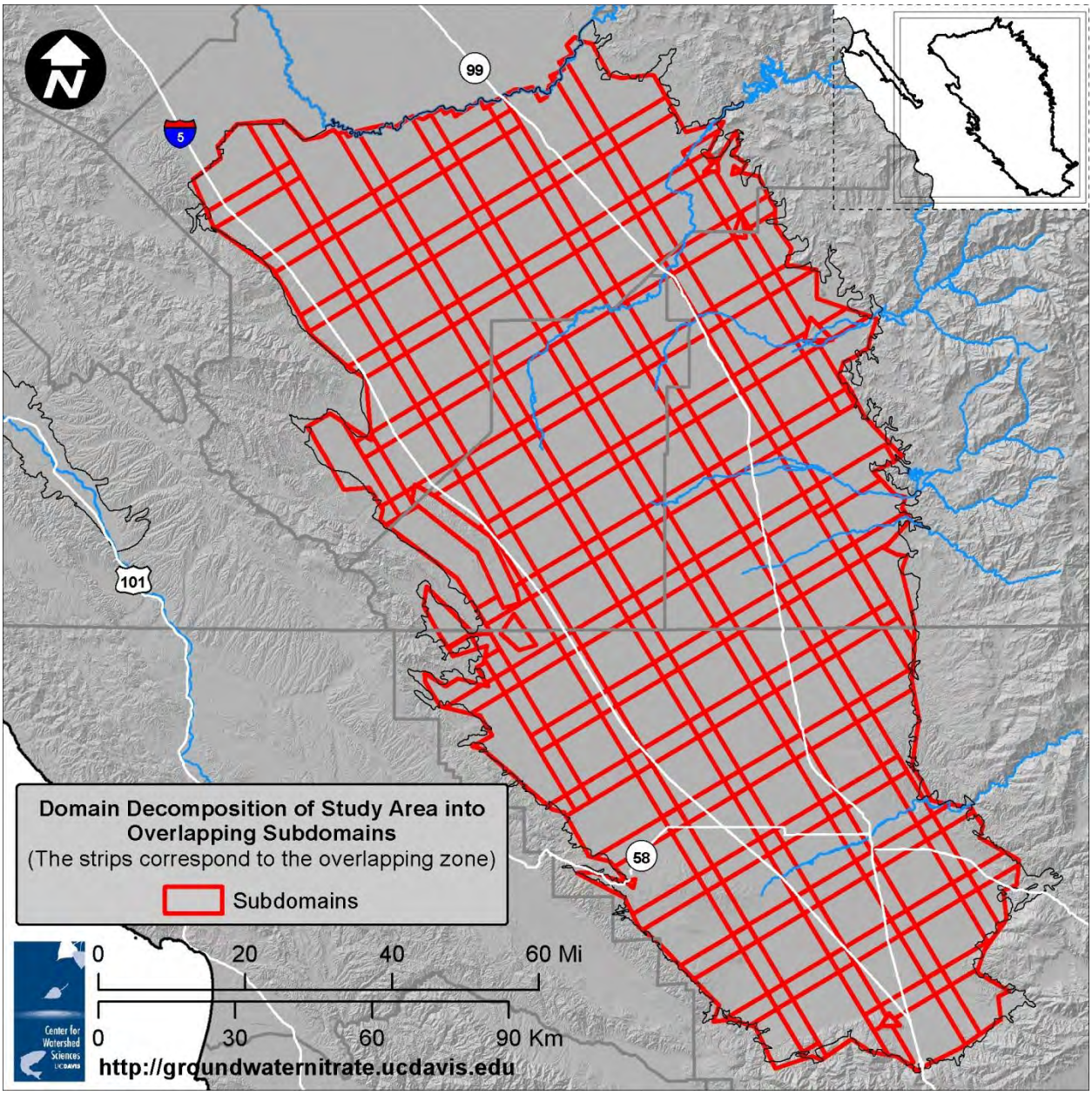

Figure 92. Domain decomposition of study area into overlapping subdomains. The strips correspond to the overlapping zone.

Adminsitrative Record
Page 35125

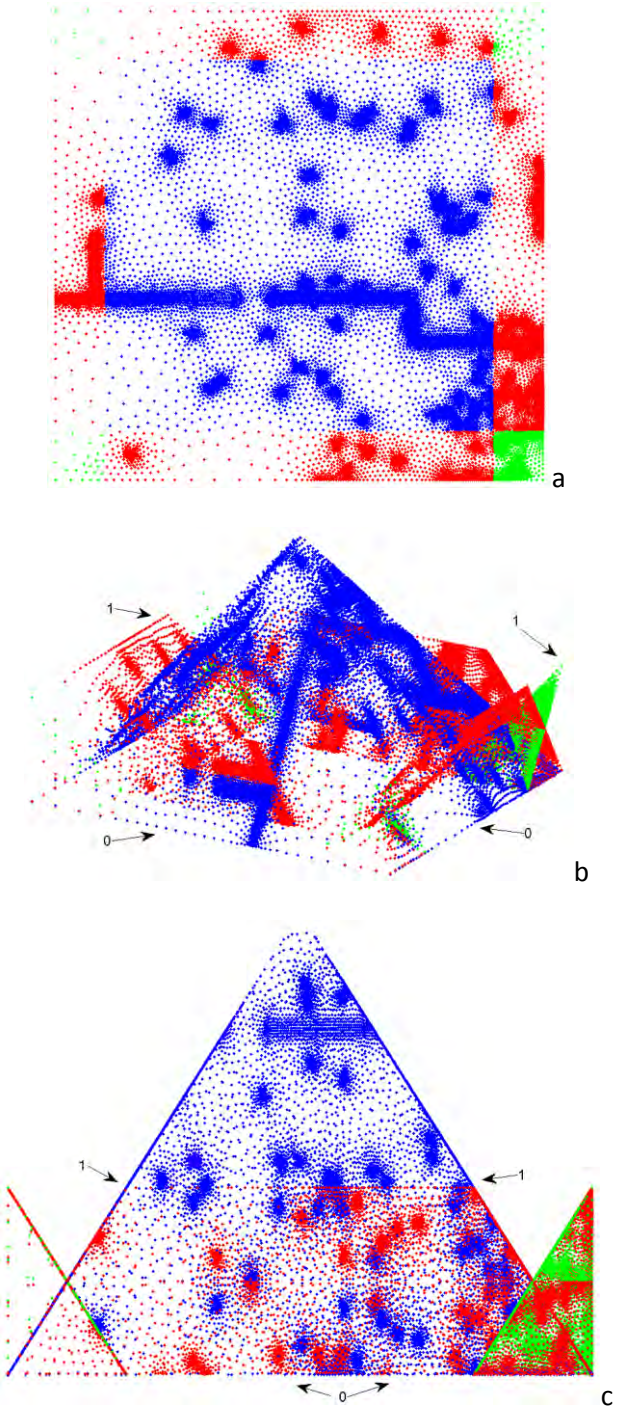

Figure 93. Illustration of weighting scheme between adjacent subdomains.

The hydraulic heads for the nodes within the non-overlapping area were obtained directly from the head solution of each subdomain. For the nodes in the overlapping zones, first, we computed the weights associated with each subdomain and then the node heads were calculated as the weighted average. Figure 93a illustrates the nodes of one of the subdomains. To keep the figure simple the triangularization has been removed from the figure. The blue dots in Figure 93a correspond to the

Adminsitrative Record
Page 35126

nodes within the non-overlapping area. The nodes with red correspond to the overlapping zone between 2 subdomains and the green to the overlapping -zone between 4 subdomains. In the other two panels, Figure 93b and Figure 93c, the height of each node corresponds to the weight associated with each subdomain. It can be seen that the weight of the main subdomain (blue dots) is zero at the boundaries and increases linearly. Similarly, the weights of the adjacent subdomains are zero at their boundaries and increase linearly as well (see red and green dots). The final head field is the weighted average of the subdomain head distribution. For example, the heads of the main subdomain (blue dots) of Figure 93 at the boundaries will be given zero weight, since the error is expected to be large due to the interpolated (from the coarse solution) boundary constraints. As we move towards to the center of the main subdomain (blue dots) the velocity obtained from the main subdomain will be given higher weight, while the weight from the adjacent subdomains decreases. Hence, the overlapping zone creates a smooth transition zone between subdomains

The head and subsequently the velocity field were used for particle tracking. According to the methodology, for each well we released 100 particles, uniformly distributed, around the well screen and tracked backwards until they exit the aquifer. Since the number of wells is 5,486, the simulated streamlines equal to 548,600 (Figure 94).

Adminsitrative Record
Page 35127

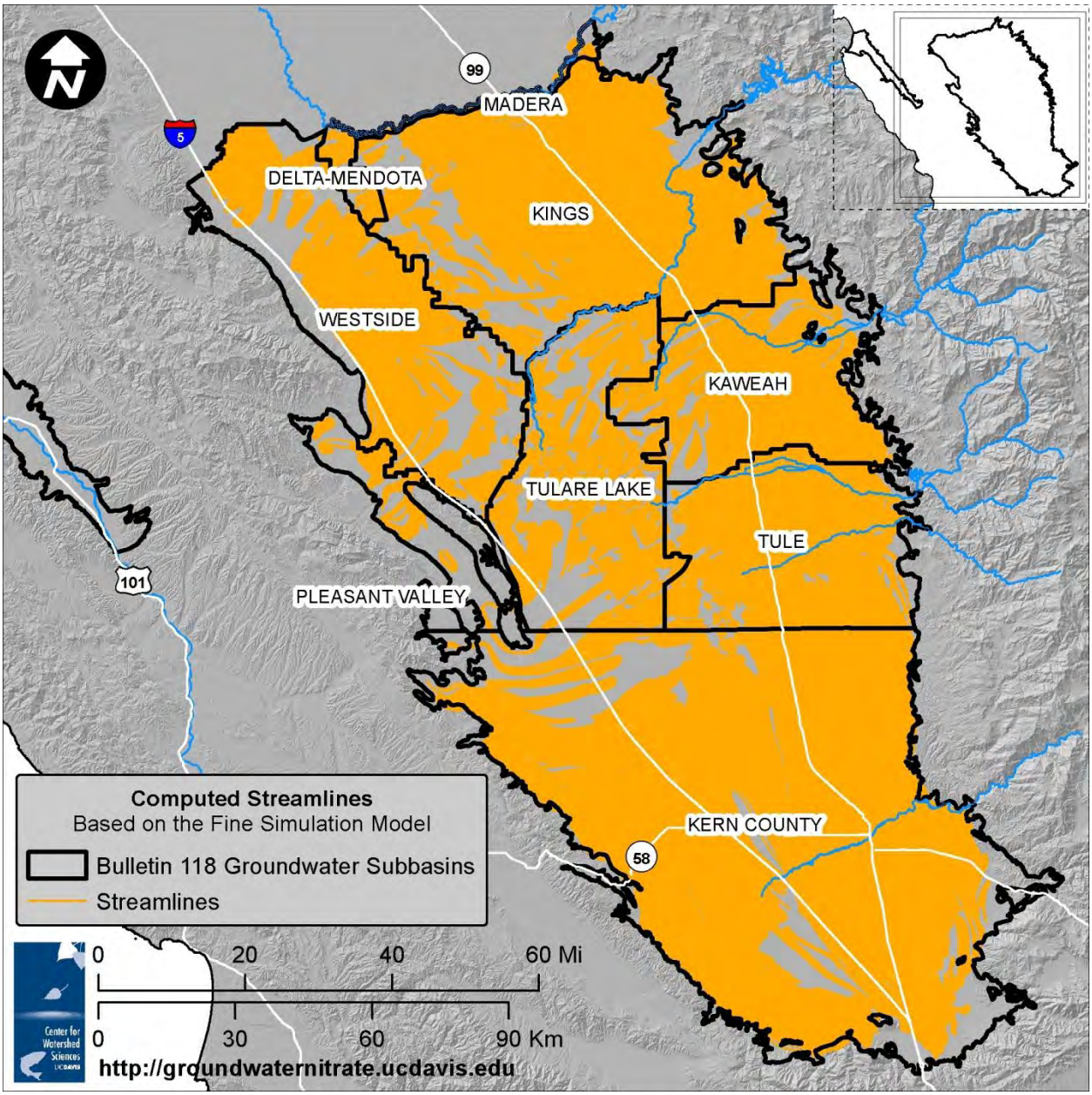

Figure 94. Computed streamlines based on the fine simulation model.

Then, for each streamline the 1D ADE transport problem was solved to obtain unit response functions. Here we used the analytical solution of 1D ADE where the longitudinal dispersivity was a function of streamline length and the velocity was set equal to the advective velocity. Note that, during the 1D transport problem, we assumed unit input loading to obtain a unit response function for each streamline, which was subsequently stored into a GIS database and was used for predictions based on different loading scenarios. Figure 95 illustrates the computed URFs for two of the simulated wells. We choose one shallow and one deep well to point out the different responses one should expect. It can be seen that the travel time distributions for shallow wells are shorter between few years and few decades, as opposed to deep wells where the range of travel time distribution spans from several decades to

centuries. In addition, the shape of the URF for shallow wells is spiky, indicating that shallow wells exhibit greater concentrations compared to deep wells.

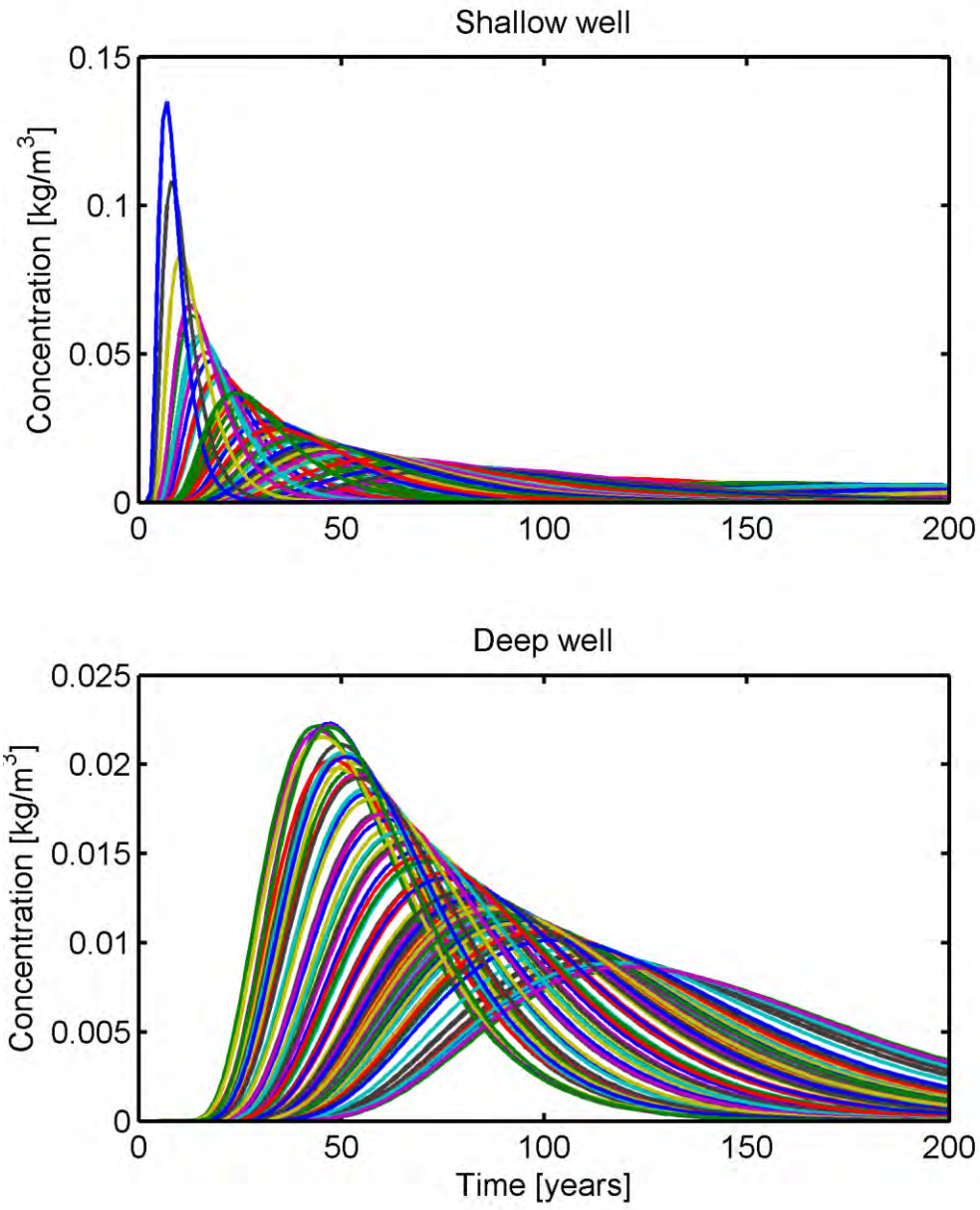

Figure 95. Unit Response Functions.

Adminsitrative Record
Page 35129

7.4 NPSAT Implementation Based on Coarse Model

In addition to the highly detailed model, we applied the NPSAT using the CVHM coarse approximation. Similar to the detailed simulation, to obtain an average steady state flow field for the water year 1996, we averaged the monthly flow fields which were obtained from the output of CVHM. Based on analytical particle tracking (Pollock 1994), we perform backward particle tracking until the particles exit from the water table. The particle starting points are identical to those used in the detailed model. Yet, due to coarse approximation, there is no cone of depression around each simulated well, resulting in an unrealistic shape of streamlines around the wells, which can be seen locally (Figure 98).

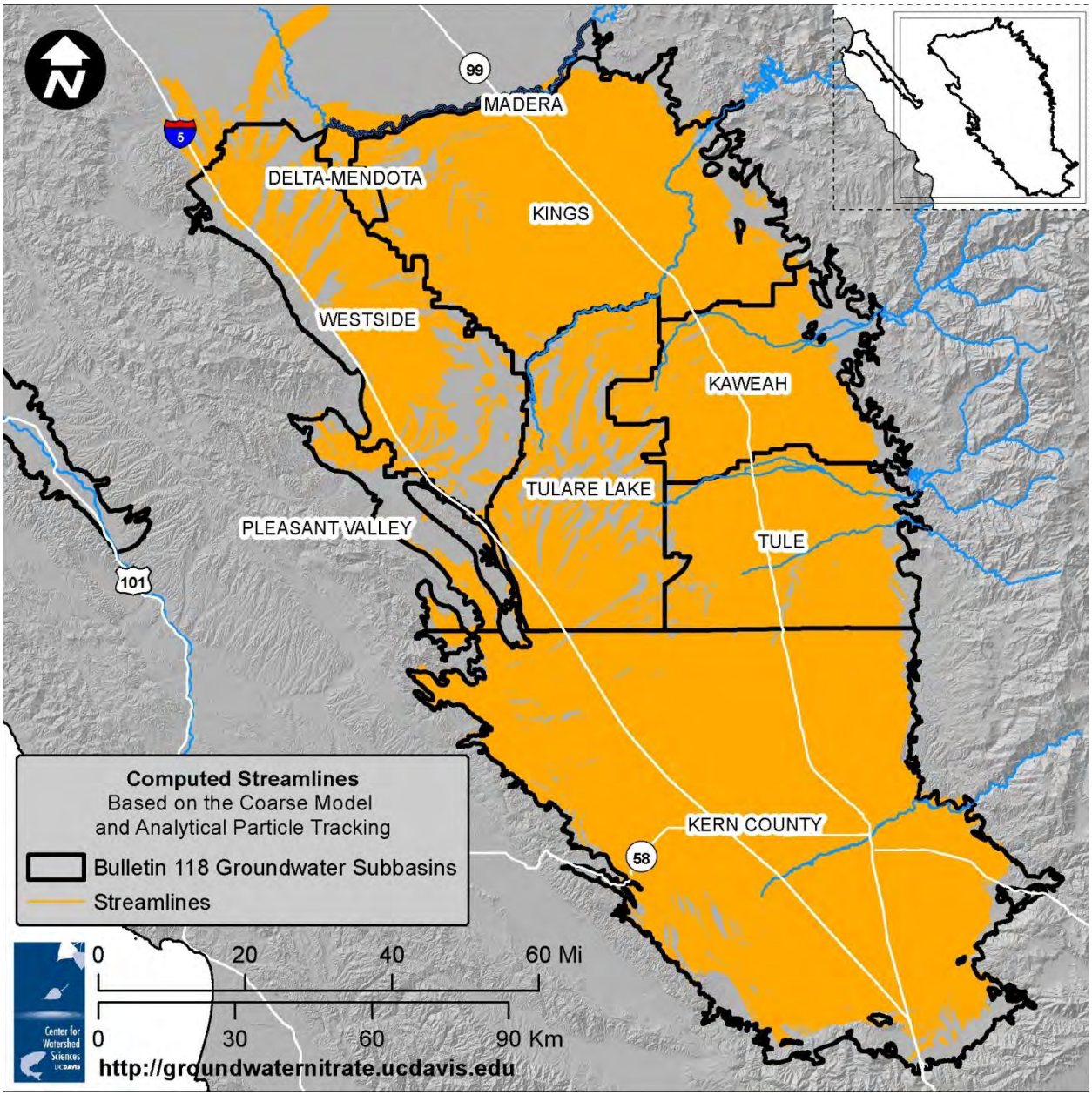

Figure 96. Streamlines based on coarse model and analytical particle tracking.

Adminsitrative Record
Page 35130

The streamlines based on the coarse approximation are shown in Figure 96. It can be seen that, in general, the streamlines follow the same pattern as in the highly detailed simulation. In both cases the streamlines cover the study area and in both cases there are fewer streamlines in the basins of Tulare Lake and Westside compared to the other basins. This can be attributed to the fact that there are fewer wells, but also that the particle travel distance is shorter in some cases.

Although the average shape of streamlines seems similar in both cases, the statistics of the streamlines reveal significant differences (Figure 97). One of the major differences is the distribution of the streamline length. In the coarse approximation the logarithm of streamline length appears normally distributed with a mean value of about 15,000 m (15 km), while for the highly detailed simulation, the distribution exhibits two peaks, one close to 6,000 m (6 km) and one close to a few hundred meters. The significant difference in streamline length is reflected also in the age distributions. While the logarithm of age appears normally distributed, the mean value for the detailed model is approximately 1.8 (~65 years), but it is 2.1 (120 years) for the coarse model. In addition there is a significant percentage of streamlines in the detailed model with travel time less than a year.

Adminsitrative Record
Page 35131

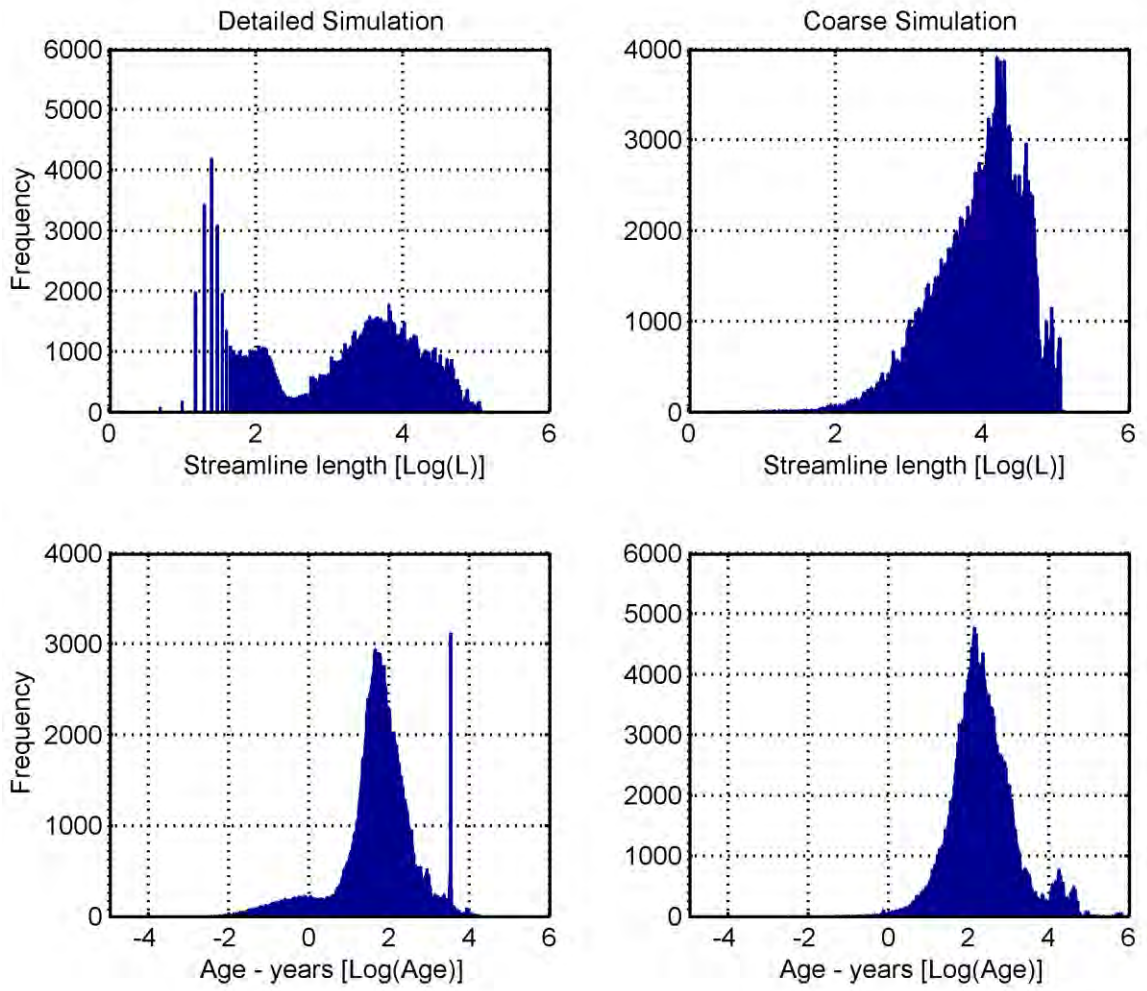

Figure 97. Comparison of the distribution of streamline length [m] and age [years] between the two simulation models. A log of 0, 1, 2, 3, and 4 corresponds to 1, 10, 100, 1000, and 10,000 units (m or years), respectively.

One of the main reasons for these differences is the fact that, in the highly detailed simulation, the hydraulic head field was depressed around the wells, due to fine discretization. Note that in coarse simulation the local cone of depression around the simulated wells was absent. Although in both models the particles originated from the exact same positions, in the fine model the particles are actually closer to the water table, due to the cone of depression, which was absent in the coarse mode. Therefore for the fine model the particle's travel time to exit the aquifer was shorter compared to the coarse model, as well as the length of their streamline. In addition, it was not unusual that part of the screen length was above the water table and the particles above the water table in the detailed model were ignored. To illustrate the difference we have plotted the streamlines for one of the wells (Figure 98 top panel) and zoomed around the well (bottom panels. The blue lines correspond to the coarse model and the red to the highly detailed. The streamlines shown here were selected as representative. Indeed, in most cases the streamlines based on the coarse model where longer and narrower compared to the detailed model, due mainly to the absence of a cone of depression around the well.

Adminsitrative Record
Page 35132

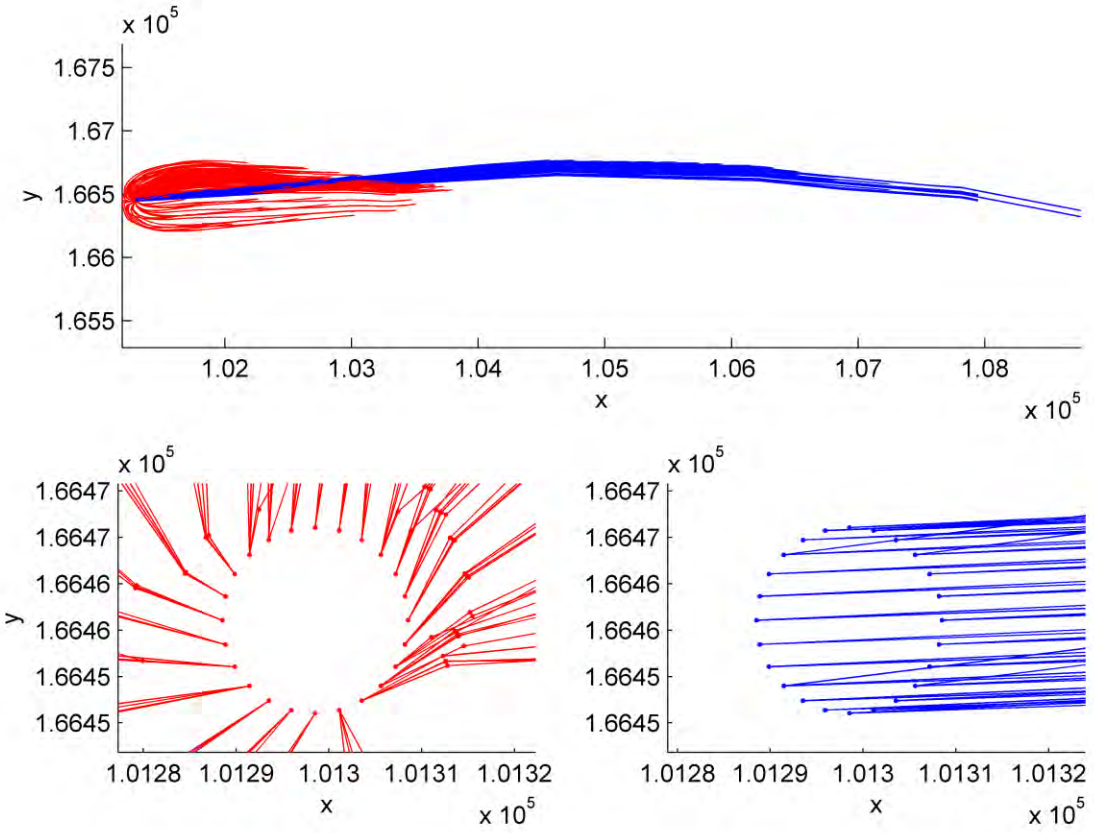

Figure 98. Comparison between streamlines computed with the detailed and coarse model. Red streamlines correspond to detailed model and blue to coarse model. Top panel shows the overall streamline shapes of the 100 streamlines for a particular well. The bottom panels show the orbits of particles around the wells. Due to the absence of a cone of depression on the coarse model (blue lines) the streamlines are actually straight lines.

Adminsitrative Record
Page 35133

7.5 Results

7.5.1 Nitrogen Loading Scenarios

The computed Unit Response Functions (URFs) can be rapidly convoluted with alternative loading scenarios to calculate real breakthrough curves for each well. In this report we examined the impact of four alternative nitrogen loading scenarios. A detailed description of the development of N loading scenarios is discussed in Section 1.8 of Technical Report 2 (Viers et al. 2012). Briefly, the four scenarios represent different management of excess manure N (nitrogen) exported from dairies after 1980: In Scenario A, exported manure N either leaves the study area or is incorporated into nutrient management on non-dairy crops such the typical amount of N applied to crops does not change. In scenario B (by study area) and C (by study area), half and all of exported manure, respectively, is applied to non-dairy cropland across the study area as an amendment in addition to typically applied fertilizer N resulting in higher groundwater nitrate loading. In scenario D, all manure N is applied on manured crops within dairies resulting in very high nitrate loading to groundwater on dairies, but elsewhere identical to Scenario A. The approach taken to modeling manure applications is also used to model food processing waste application, and application of biosolids and WWTP effluent nitrogen, whether it is to cropland or directly into recharge facilities.

The output of the N loading algorithm developed by Viers et al. (2012) is four alternative scenarios of the N leaching into into the groundwater table. The leaching rates are given in kg/ha for 8 different years (1945, 1960, 1975, 1990, 2005, 2020, 2035, and 2050). For the simulation purposes we assumed that the recharge is constant over the 106 years of simulation, while the nitrate leaching rates vary linearly between these yearly values. Groundwater water recharge is a very important parameter, not only due to its influence on the flow field, but also because it determines the final leaching concentration rates. The loading histories need to be converted in units kg/m^3 (i.e., concentration). Therefore, prior to convolution of loading functions with the URF, we divided the N rates with the recharge. Figure 99 shows the cumulative distribution of loading rates assigned to each streamline. The left panels shows the N loading leaching rates in kg/ha (e.g., as they were computed by the N loading algorithm), while the right panels show the actual concentration that is assigned as input to each streamline. Note that this plot takes into account the loading rates for each streamline. Therefore, when multiple streamlines originate from the same land parcel, the corresponding rate is counted multiple times. During the prediction phase this does not affect the outcome as the final breakthrough curve is the weighted average of the streamline BTC (eqn. 21)

Similarly, in Figure 100 we plotted the exceedance probabilities of half the MCL, the MCL, and twice the MCL, that is the probability that wells exceed half the MCL, the MCL, or twice the MCL. Note that these three concentration limits will serve as reference values in the prediction phase. It can be seen that, in year 1945, approximately 40% of streamlines (i.e., 40% of the whole well source area) were associated with land parcels that leach with rates of half of the MCL, while 20% and 13% exceed the MCL and twice the MCL, respectively. Having these conditions as starting points, loading increases somewhat linearly until the year 2005, where 83%, 68% and 55% of the streamlines are associated with land parcels that

Adminsitrative Record
Page 35134

exceed the 22.5 mg/L, 45mg/L and 90 mg/L, respectively. For future predictions, four different loading scenarios are considered (see details in Section 1.8 of Technical Report 2, Viers et al. 2012). Outside of dairies, there is no difference in N loading to groundwater between “Scenario A” and “Scenario D” as manure exports either don’t affect typical N application or there are no manure exports to areas outside dairies. Post-1990, these two scenarios represent significantly less loading than “scenario B (study area)” and “scenario C (study area)”, which apply half and all exported manure to cropland, respectively, in addition to typical fertilizer applications. No manure exports occur in 1975 and prior to 1975 in any of these scenarios. Hence the loading distribution across the landscape is identical between the scenarios for 1945 – 1975.

Adminsitrative Record
Page 35135

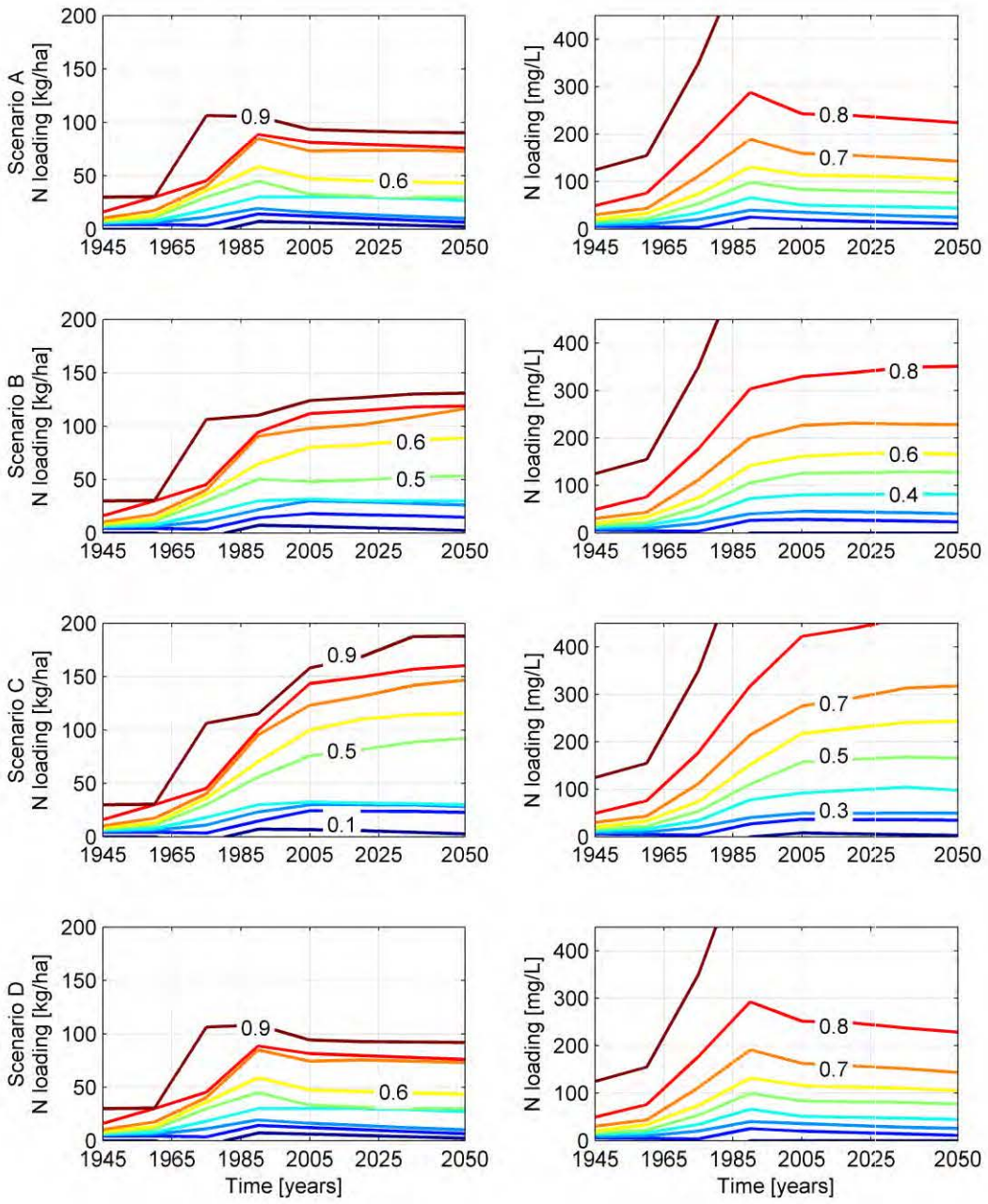

Figure 99. Cumulative probability distribution of N loading per year assigned to each streamline. The left panels correspond to N loading with units kg/ha -output of N loading algorithm Technical Report 2 (Viers et al. 2012), while the right panels correspond to the actual loading of N as nitrate (45 mg/L drinking water limit), which is used as input to NPSAT simulation model. All scenarios here are "by study area" and are explained in Section 1.8 of Technical Report 2 (Viers et al. ,2012).

Adminsitrative Record
Page 35136

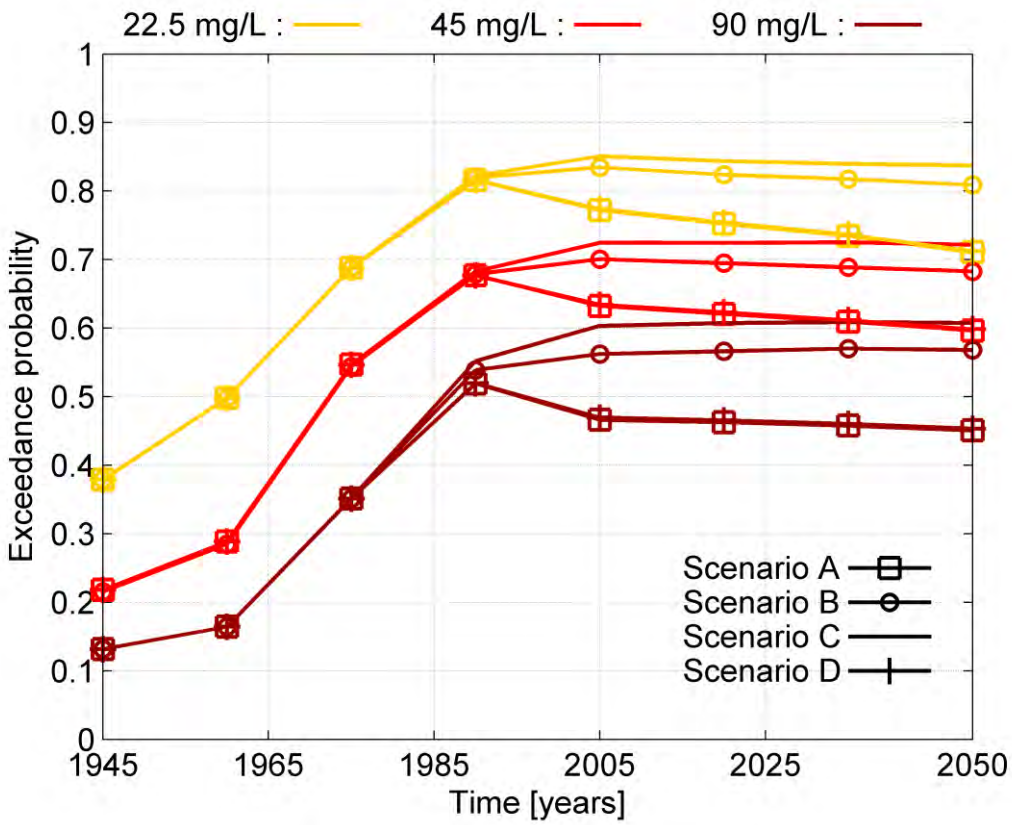

Figure 100. Exceedance probabilities of N loading rates as nitrate. All scenarios used here distribute excess nitrogen "by study area" and are explained in Section 1.8 of Technical Report 2 (Viers et al. ,2012).

The above loading functions were computed based on the detailed simulation model. Yet, when the coarse simulation model is used to convert the rates to concentrations, the figures change significantly. Figure 101 illustrates the cumulative distribution functions of N loading over time based on the coarse model. While the output of the N loading algorithm is identical in both cases (compare left panels in both figures) the loading distribution is primarily a function of the starting points of streamlines (i.e., the point where the particles exit the aquifer during backward particle tracking). The four scenarios exhibit similar behavior as described previously (i.e., in scenario A and D the leaching rates are reduced after 1990, in scenario B [by study area] leaching rates remain nearly constant, and in scenario C [by study area] there is an increase in loading), but the magnitude of the overall loading is less compared to the detailed scenario. The main reason is that, in coarse simulation, the particles were likely to connect wells to the higher recharge areas, therefore, during the conversion from mass kg/ha to mg/L the concentrations appear to be smaller. The main reason for that can be attributed to the nature of the particle tracking algorithm that was used in each case (e.g., analytical particle tracking for the coarse case and numerical particle tracking for the detailed simulation.) Furthermore, the streams for the coarse model are simulated by one square mile cells, while in detailed simulation the streams were assumed line sources with width approximately 50 m. It was found that, in the coarse model, a larger number of streamlines originate from cells associated with streams. Finally, the exceedance probabilities of the three reference concentration limits are shown in Figure 102. By comparing Figure 100 and Figure

Adminsitrative Record
Page 35137

102 we see that the N leaching into the groundwater concentration rates for the coarse model are 5% to 10% less than those of the fine model.

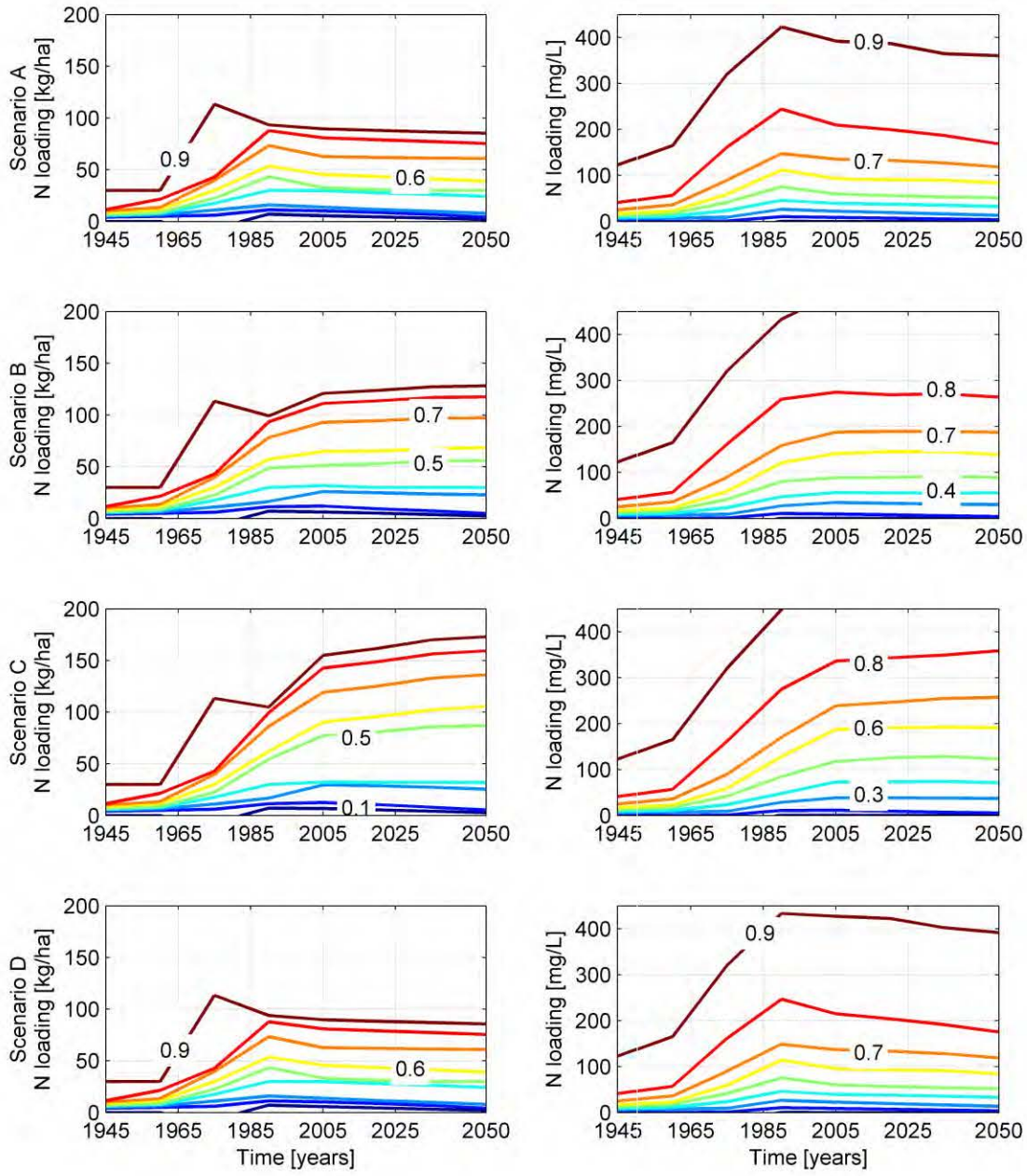

Figure 101. Cumulative distribution function of N loading based on the coarse model. All scenarios used here distribute excess nitrogen "by study area" (see Section 1.8 of Technical Report 2, Viers et al. ,2012).

Adminsitrative Record
Page 35138

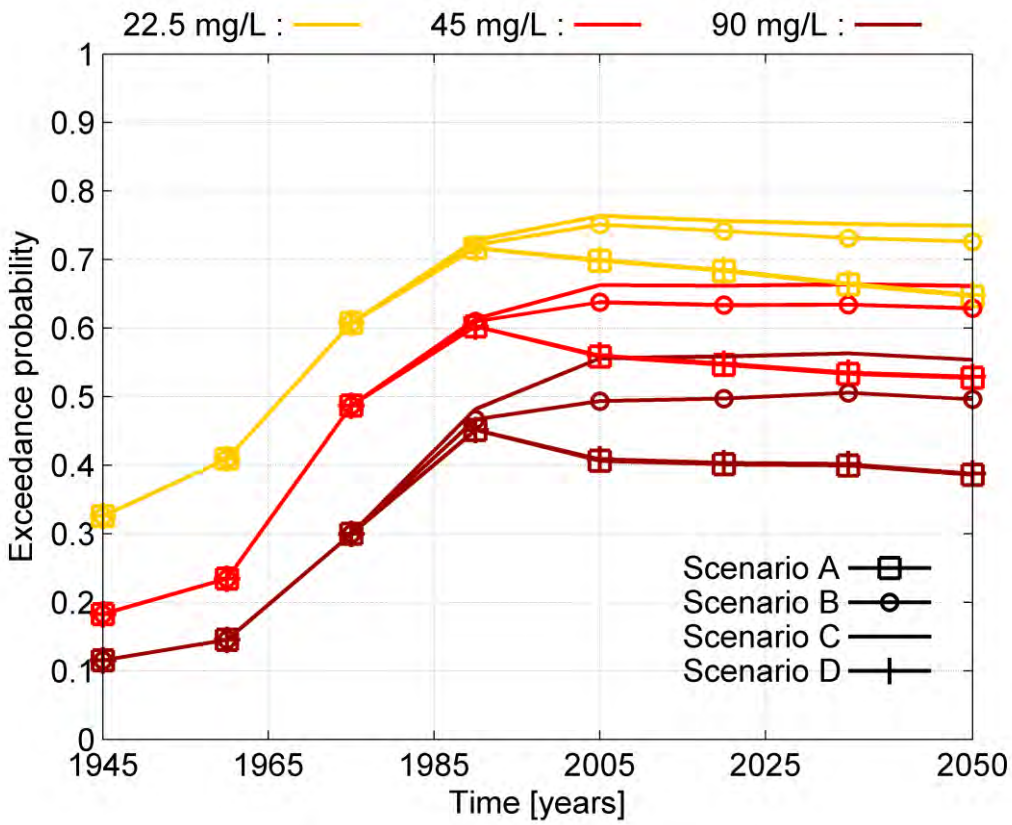

Figure 102. Exceedance probabilities of leaching rates as nitrate based on coarse model.. All scenarios used here distribute excess nitrogen "by study area" (see Section 1.8 of Technical Report 2, Viers et al. ,2012).

7.5.2 Model Validation

Prior to using the model for future predictions we perform a validation step, where the outcome of the model is compared against real data that have been gathered from various sources (section 4.4). The rationale for the data gathering and processing is explained in detail in section 4.4; however, for completeness of the section we summarize the main steps. In the well database there is a large number of samples; however, these samples are not uniformly distributed, neither spatially nor temporally. To alleviate both non uniformities the study area was divided into a number of equal area cells (Figure 39). Then, starting with 1950 and through 2010, for each cell, we compute the median of the annual well concentration means on a decadal basis. At the end of this process we obtain one representative concentration value for each cell for each decade. To calculate a de-clustered decadal average concentration for a particular region we compute the mean and the confidence interval of the mean of the decadal medians of the cells located in that region after they are log transformed. The back-transformed decadal means for the three regions of the TLB (Figure 39) are shown in Figure 103 and Figure 104, with the 95% confidence interval of the mean.

Adminsitrative Record
Page 35139

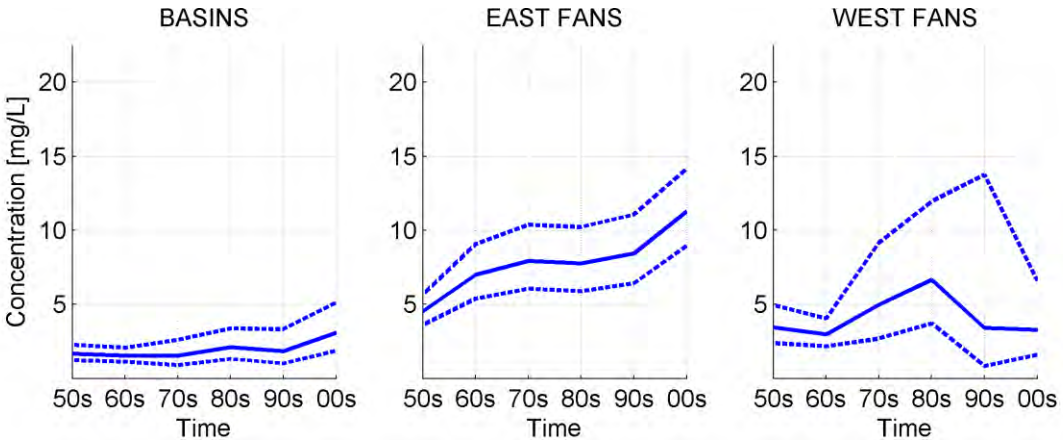

Figure 103. Decadal means of nitrate concentration for the three regions based on measured data. The solid line represents the mean and the dashed line the confidence interval.

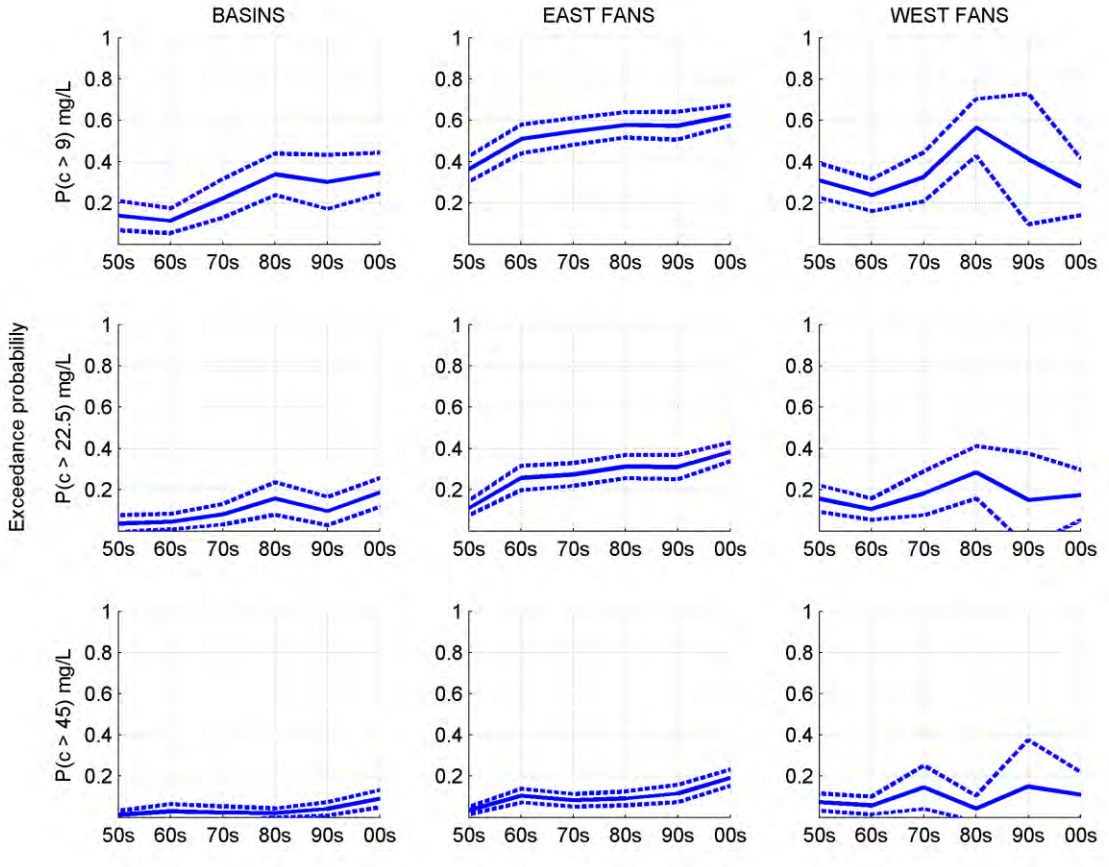

Figure 104. Decadal exceedance probabilities based on measured data. The solid line represents the de-clustered mean exceedance probability within each region and the dashed line the 95% confidence interval.

Adminsitrative Record
Page 35140

Next, the measured data are compared against the simulated trends, based on the two modeling approaches, the fine and coarse model. Similar to the measured data, the simulated results were grouped into the same equal area cells and the de-clustered decadal means and confidence intervals of the means were calculated as described previously.

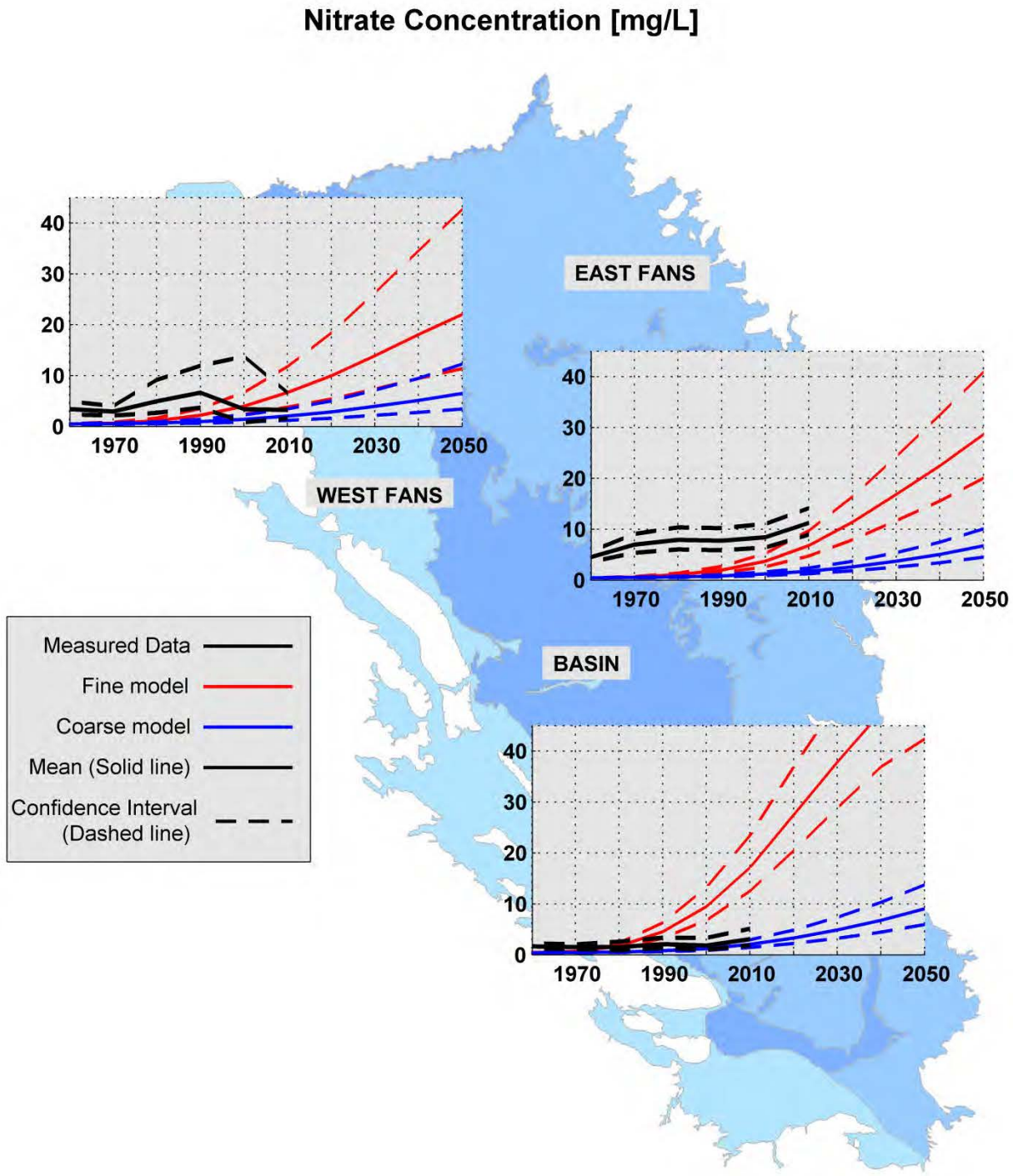

Figure 105. De-clustered, back-transformed mean of equal area log nitrate medians in each region of annual well means. Comparison of mean and 95% confidence interval between measured data and model predictions.

Adminsitrative Record
Page 35141

First we compare the de-clustered mean concentration data against the model responses (Figure 105). The solid lines represent the de-clustered, back-transformed mean of log transformed decadal medians of annual well means calculated per equal area cell and the dashed line represents the confidence intervals. In general, both models fail to follow the early decades 1950–1980, with an exception for the Basin region, yet this is due to the very low measured concentrations. Note that, in all regions, the responses of both models are practically identical until the 1970s and both calculate very low concentrations at the beginning of the simulation. This is due to the fact that the groundwater nitrate loading model does not account for nitrate loading from any sources prior to 1945 – groundwater is assumed to be "clean" in 1945. The model also neglects any background nitrate concentration, which is typically on the order of ~4 mg/L nitrate or less. A second reason is that the groundwater model does not include shallow domestic wells with very low pumping rates, which are often affected before the larger production wells with longer screens, simulated here, are affected by higher nitrate concentrations. For the Basin region, the coarse model fits better to the data compared to the fine model. In the Basin region, the fine model predicts a more rapid increase in nitrate concentrations than the measured data suggest. For the Eastside Fans region, early nitrate data are higher than predicted by either model. For the 1990s and 2000s, the fine model, predicting a more rapid increase than the coarse model, comes relatively close to measured de-clustered mean. The coarse model predicts, on average, very low concentrations, less than 10 mg/l, while the upper limit of fine model is approximately identical with the lower limit of the data for the last decade. Note also that the slope for the last decade is somewhat similar to the slope suggested by the fine model. For the Westside Fans region, there is similar discrepancy as in the Eastside Fans region. The fine model predicts a slightly upward trend for the 1980s, which becomes more apparent during the 1990s, then significantly overpredicts the observed average low nitrate concentration measured during last decade (2000-2010). The coarse model underpredicts even further the higher average nitrate concentrations observed during the 1980s in the Westside Fans region, but appears to better predict the more recently observed low average nitrate concentrations. We caution that for the Westside Fans, the measured decadal mean values are based on much fewer measured data than for the other regions, with the source of data and the associated types of wells measured varying significantly over decades (see discussion in Section 5). Overall, it appears that the response of the fine model is closer to what is measured for the Eastside Fans than the coarse model. In contrast, the coarse model, showing a slower rise in nitrate concentrations than the fine model, due to the above discussed inherent conceptual model differences, compared better to the measured average nitrate concentrations in the Basin and Westside Fans regions. Neither model takes into account denitrification, which may be a significant process within the Corcoran Clay separating the upper from the lower aquifer system in the Basin and Westside Fans region. Future modeling efforts may be performed to accommodate denitrification in the Corcoran Clay and other portions of the aquifer system, for which denitrification rates would have to be determined.

Adminsitrative Record
Page 35142

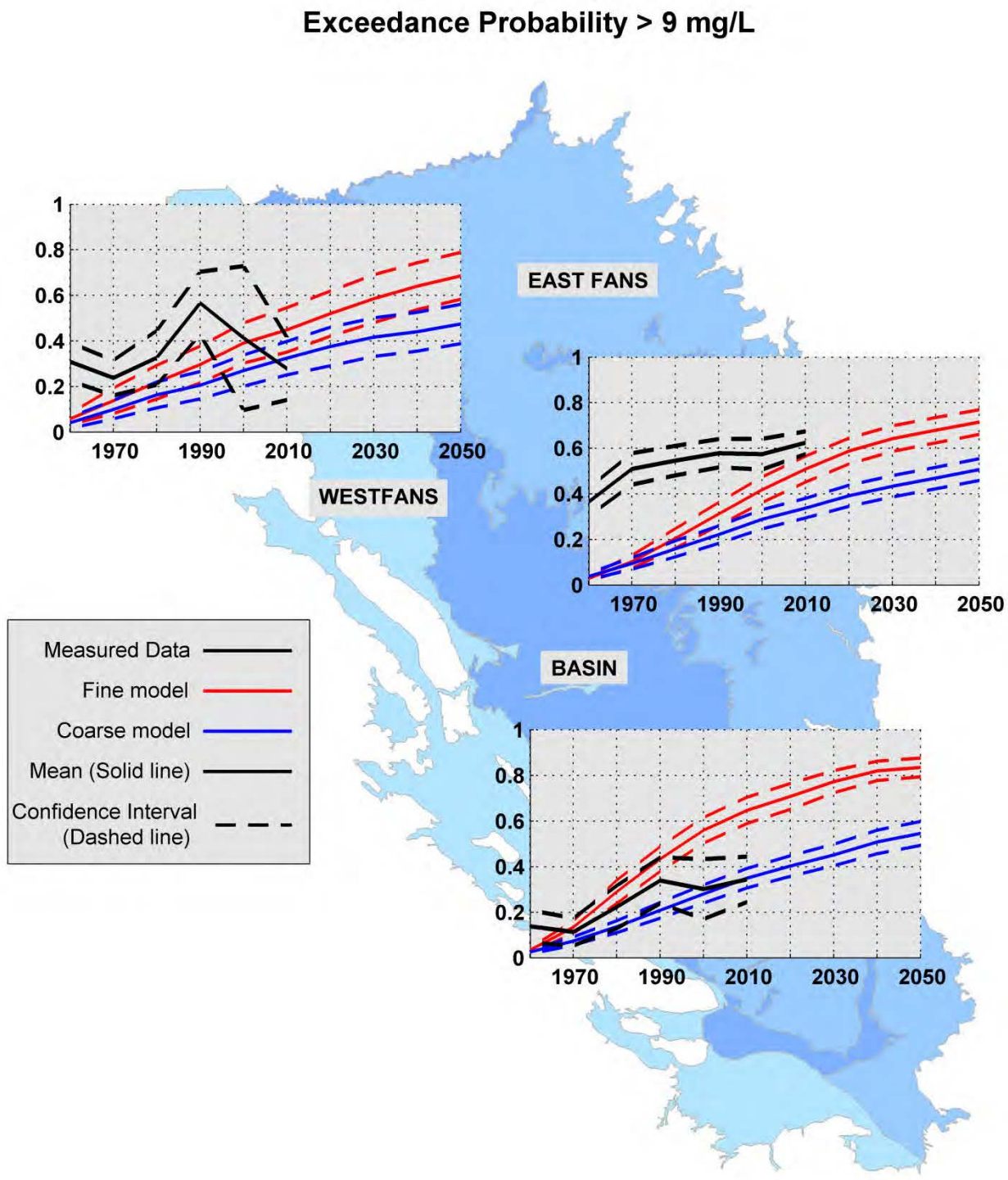

Figure 106. Comparison of decadal trends of exceedance probability of 9 mg/L, between measured data and model predictions.

Adminsitrative Record
Page 35143

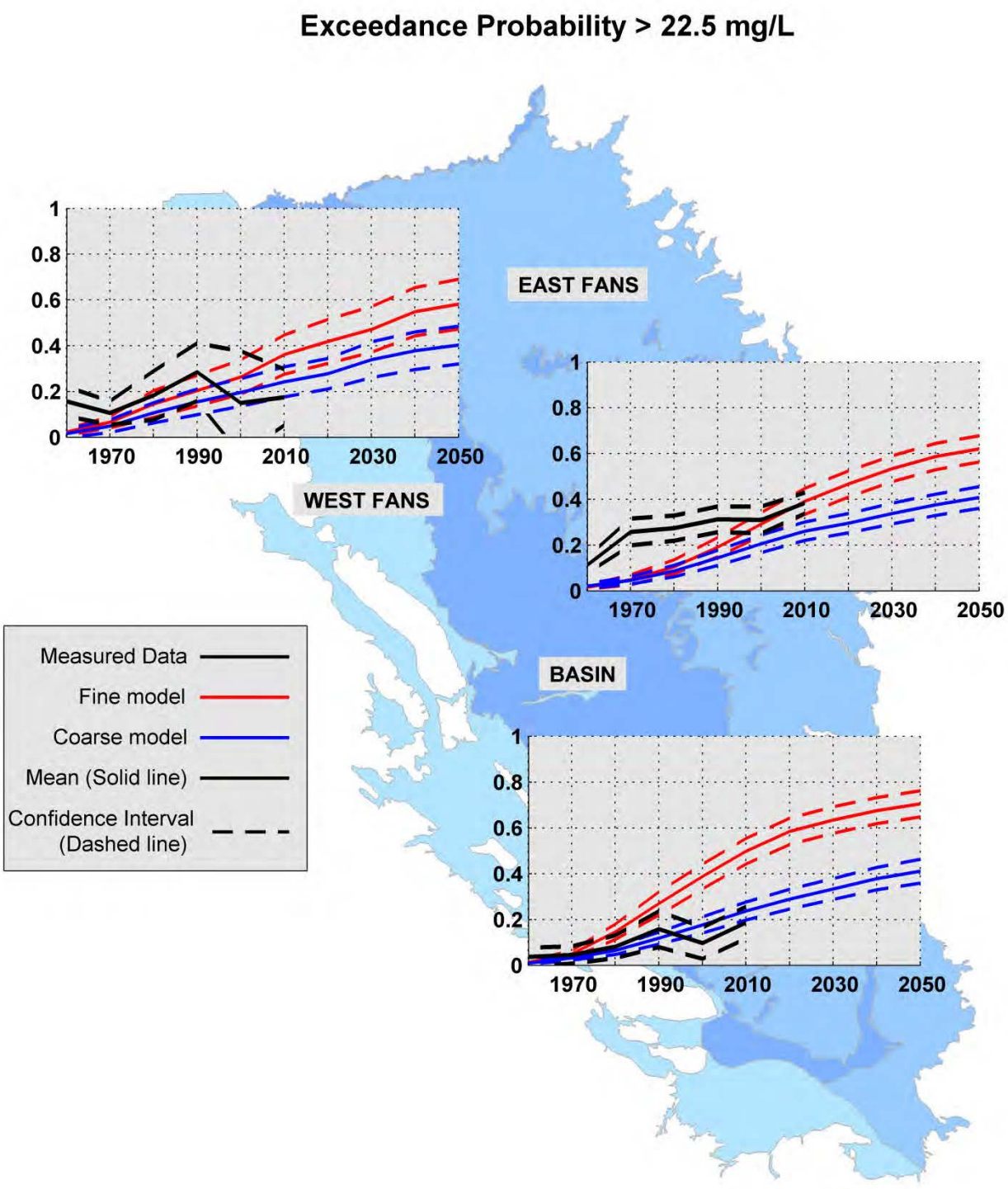

Figure 107. Comparison of decadal trends of exceedance probability of 22.5 mg/L, between measured data and model predictions.

Adminsitrative Record
Page 35144

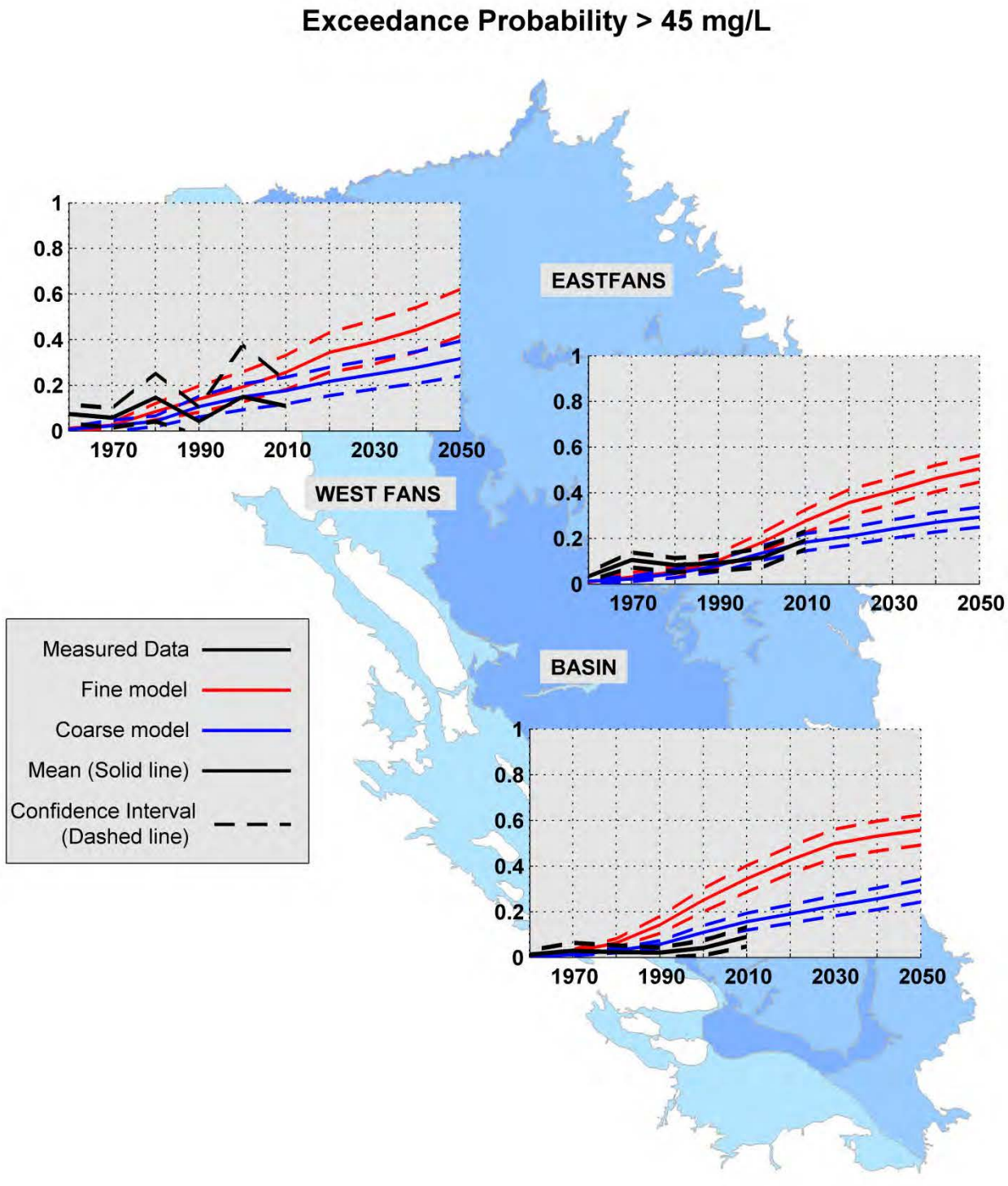

Figure 108. Comparison of decadal trends of exceedance probability of 45 mg/L, between measured data and model predictions.

Another way to study the decadal trends is to examine the exceedance probabilities for a given concentration. To this end we computed the exceedance probabilities for concentration levels 9, 22.5 and 45 mg/L. Note that the drinking water limit is 45 mg/L. Figure 106 illustrates the mean decadal

Adminsitrative Record
Page 35145

exceedance probabilities of 9 mg/L and the confidence intervals based on the measured data and the responses of the two models per region. Interestingly, for the Basin region the fine and the coarse models follow the upper and lower limit of confidence intervals of the real data for the 70s and 80s, but they fail to reproduce the abrupt change that occurs between 80s and 90s. For the Eastside Fans region both models underestimate their predictions during the first decades, while the response of the fine model are somewhat closer the measured data. Both models behave in a similar manner for the Westside Fans region predicting a continuous increase, yet based on the data there is a significant decreasing trend from the 80s onwards, which is not captured by either model, but maybe due to a sampling bias in the measured data. The response of the models for the exceedance probabilities of 22.5 mg/L (Figure 107) is very similar to the exceedance probabilities of 9 mg/L. Both models fail to capture the decreasing trends for the Westside Fans and Basins regions, while the predictions of the fine model are closer to real data for the Eastside Fans region.

Interestingly, the models are showing better fit for the exceedance probabilities of the MCL (45 mg/L) (Figure 108). In antithesis with the previous figures, the trends in all three regions show a general increasing trend. For the Westside Fans and Eastside Fans regions, both models respond similarly, although the coarse model seems to fit slightly better with the real data. For the Basin region, both models predict a significant upward trend after the 70s; however, based on the real data, the upward trend only occurs two decades later.

In conclusion, the measured nonpoint source pollution system behaves in certain cases very different than what would be expected based on the model. In every case, both models failed to reproduce any downward trend. The fine model predictions seem to fit to the measured data better for the last decade, especially for the Eastside Fans region. Comparison between the models confirms, in certain cases, our expectation that the coarse model tends to underestimate, while the fine model tends to overestimate the actual concentration levels, although there are certain cases where this general “rule” is violated. Accounting for denitrification, with independently measured denitrification rates would provide a tool to improve the average model predictions when compared to average measured data.

7.5.3 NPSAT Modeling Predictions

The N loading functions for each scenario were convoluted with the URFs, a process that involves only analytical calculations; hence, we were able to calculate each alternative scenario A, B (by study area), C (by study area), and D (for detailed description of these scenarios, see Section 1.8 of Technical Report 2; Viers et al., 2012). The BTC for each well are shown in Figure 109. The thin colored lines represent the well BTCs, the red lines show the drinking water limit, and the black thick lines depict the exceedance probabilities. It can be seen that there is a significant number of wells with very fast BTCs (i.e., shallow wells with relatively small screen lengths), and also a significant number of wells that exhibit very slow responses (e.g., deep wells).

Adminsitrative Record
Page 35146

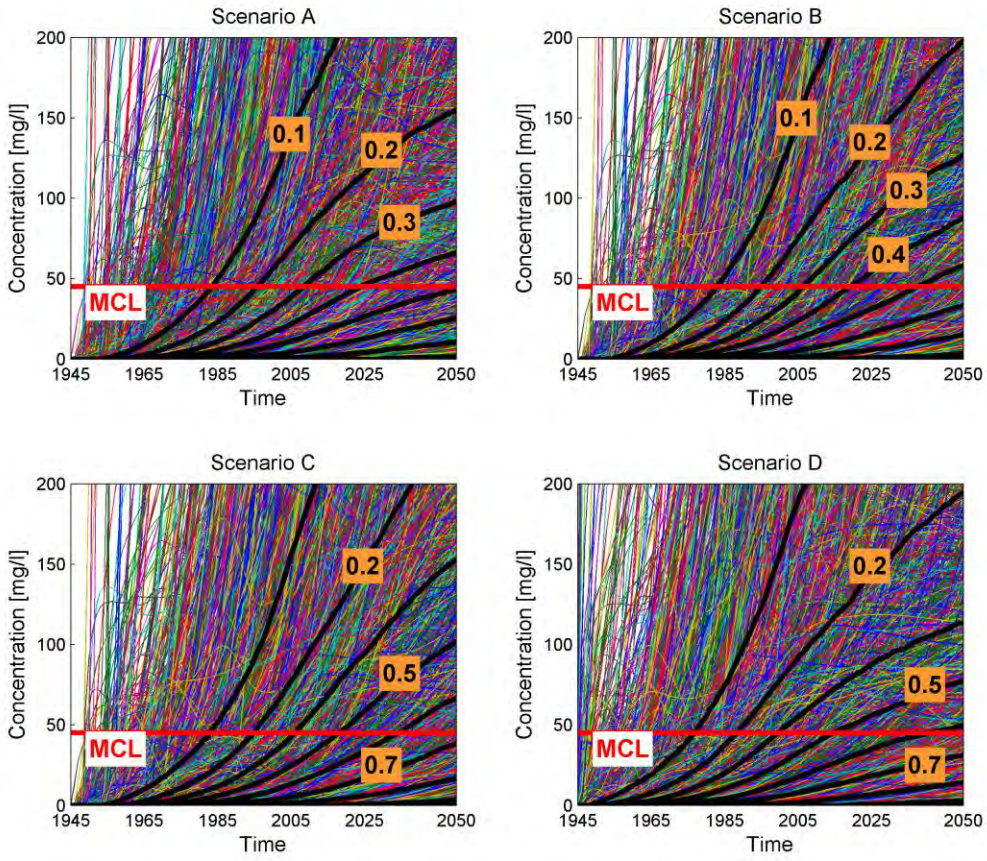

Figure 109. Breakthrough curves based on model simulation for each well for the four alternative (by study area) scenarios (thin colored lines). The black thick lines correspond to exceedance probabilities and the red lines show the drinking water limit (45 m/L). Scenario details are provided in Section 1.8 of Technical Report 2 (Viers et al., 2012).

Although the simulated BTCs are temporally uniformly distributed, their spatial distribution is non-uniform. Therefore we utilized a similar method as in the previous section (4.7.5.2) to process the results, where the TLB area was split into six basins (Kern, Kings, Tule, Tulare Lake, Kaweah, and Westside). Each basin was divided into equal area cells, and, according to the previous method, we computed the means and confidence intervals for each basin. Figure 110 shows the comparison between the fine model and the coarse model using the loading functions of scenario D. It can be seen that in all basins the fine model (red lines) predicts higher concentrations compared to coarse model (blue lines). Note also that the uncertainty is rather high in most basins (e.g., Kaweah, Tule), and in particular for the future predictions. In addition, the two models exhibit large discrepancies, yet a few safe conclusions can be drawn. For example, both models predict that for the Kings and Kern basins the concentrations will remain below the drinking water limit. On the other hand, both models predict high concentrations for the Kaweah basin. By the year 2050 the mean concentration, based on the fine model, is expected to be twice the MCL, and the coarse model estimates concentrations close to MCL with but significant variance in its estimation). This may be attributed to the fact that Kaweah basin

Adminsitrative Record
Page 35147

contains the majority of dairies. As far as the remaining three basins are concerned, the models exhibit large discrepancies. For example, the coarse model predicts that the concentration in Tule basin will not exceed 10 mg/L by 2050 (the upper confidence limit), while the fine model predicts that the concentration will not exceed, but will be very close to the MCL, yet, this figure is associated with large variance. Note the upper confidence limit is approximately twice the MCL. For the basins on the west side, the coarse model predicts a similar upward trend, where the mean concentration by the year 2050 is expected to be 20 mg/L, approximately. On the other hand, the fine model predicts significantly higher concentrations.

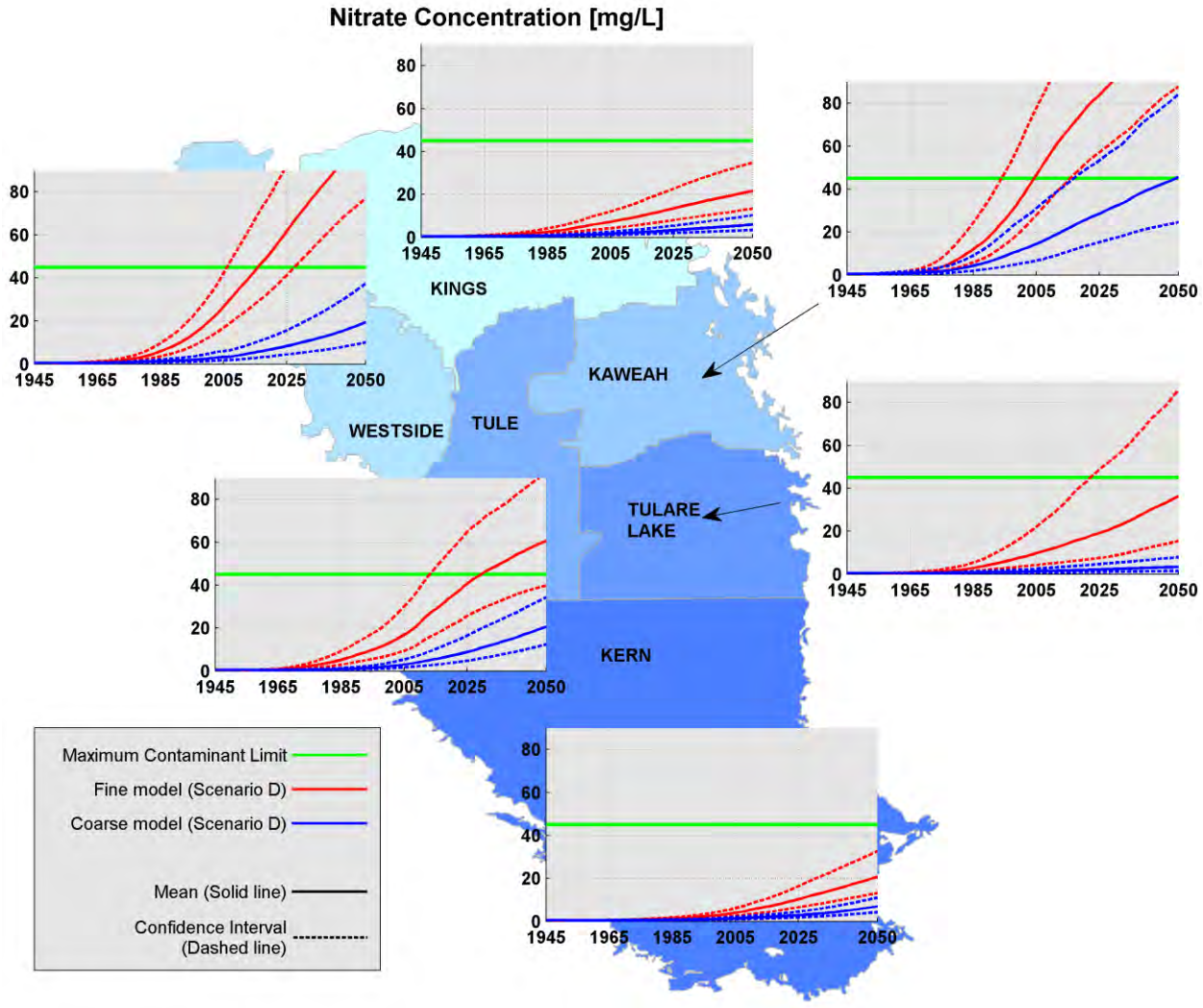

Figure 110. Comparison between coarse model and fine model based on scenario D.

Adminsitrative Record
Page 35148

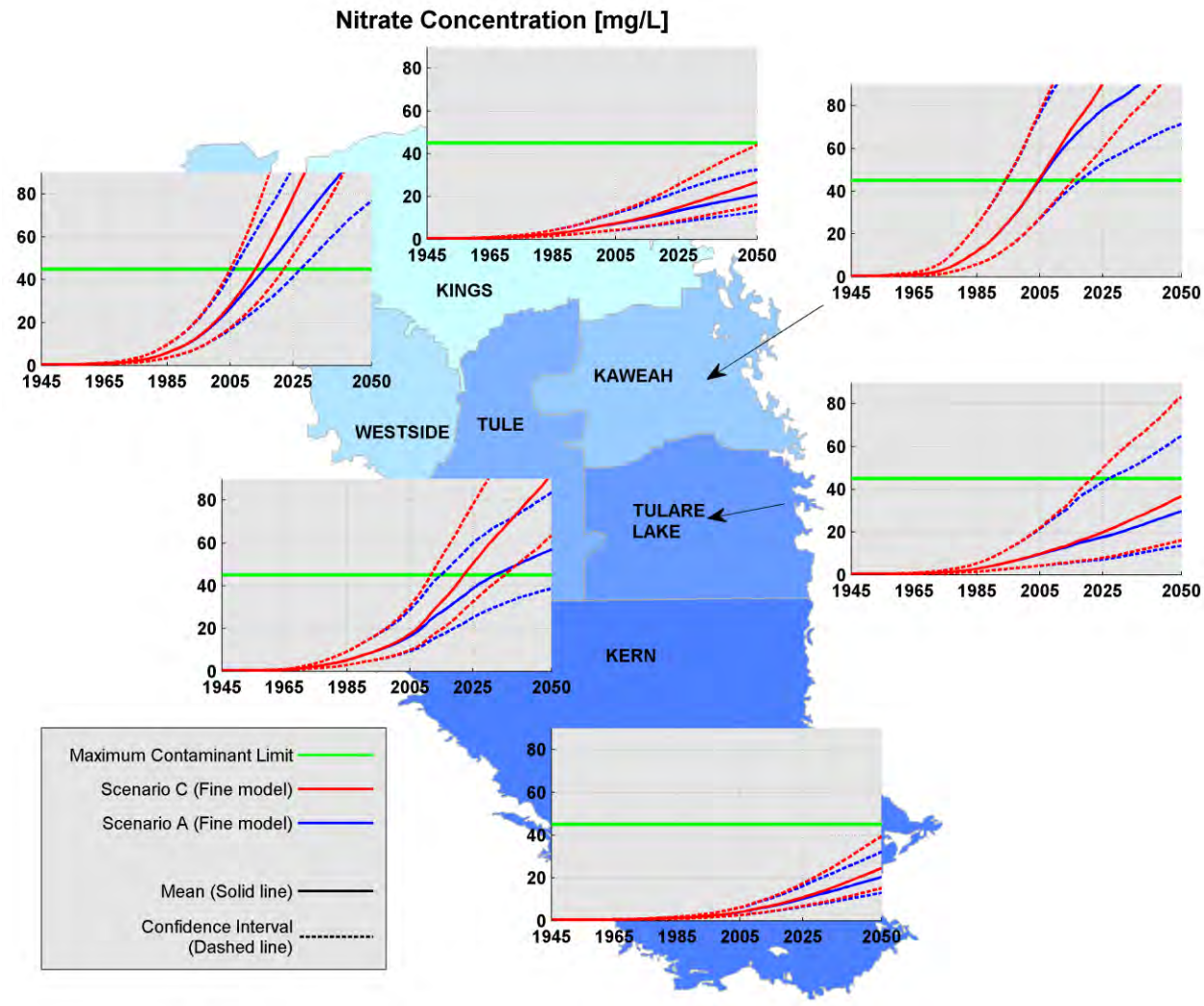

Figure 111. Comparison between scenario A and C based on the fine model predictions.

While the models may not accurately capture the actual responses they can still be useful tools to evaluate the impact of alternative scenarios. Figure 111 plots the predictions of the fine model based on the loading scenarios A and C. Scenario C is considered here the worst scenario. It can be seen that the impact of scenario C for the basins of Kern, Tulare Lake, and Kings is rather negligible. In antithesis, the basins of Kaweah, Tule, and West side are significantly affected.

Last, we compute the exceedance probabilities for each basin, based the fine and the coarse models for three given limits (e.g., 22.5 mg/L (half the MCL), 45 mg/L (MCL), and 90 mg/L (twice the MCL)). Figure 112 shows the exceedance probabilities based on the fine model using scenario D, while Figure 113 shows the exceedance probabilities based on the coarse model and the same scenario D. By comparing the two figures we observe similar discrepancy between the two models. The fine model tends to predict higher exceedance probabilities compared to the exceedance probabilities of the coarse model.

Adminsitrative Record
Page 35149

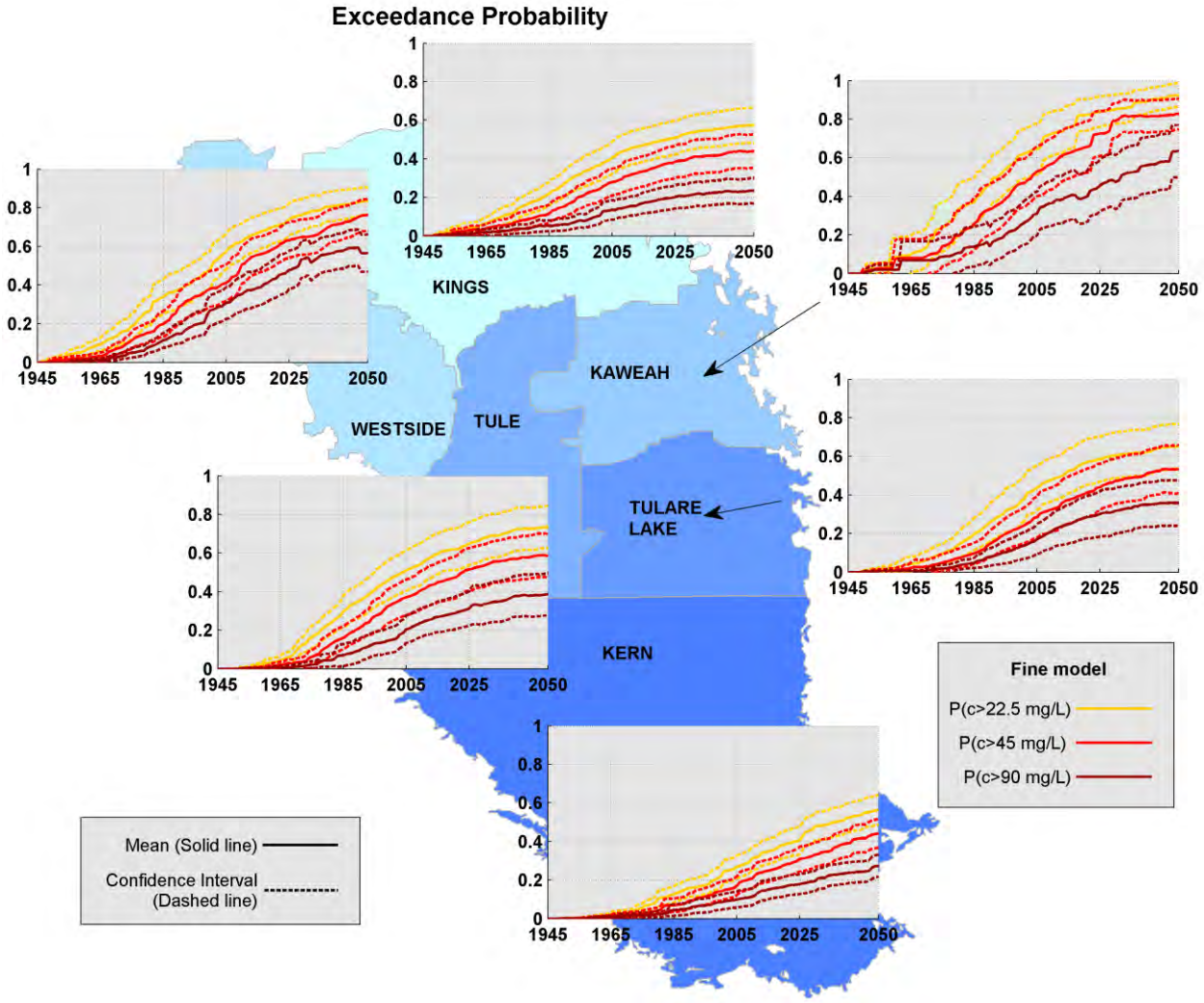

Figure 112. Exceedance probabilities for 22.5, 45, and 90 mg/L based on the fine model.

Adminsitrative Record
Page 35150

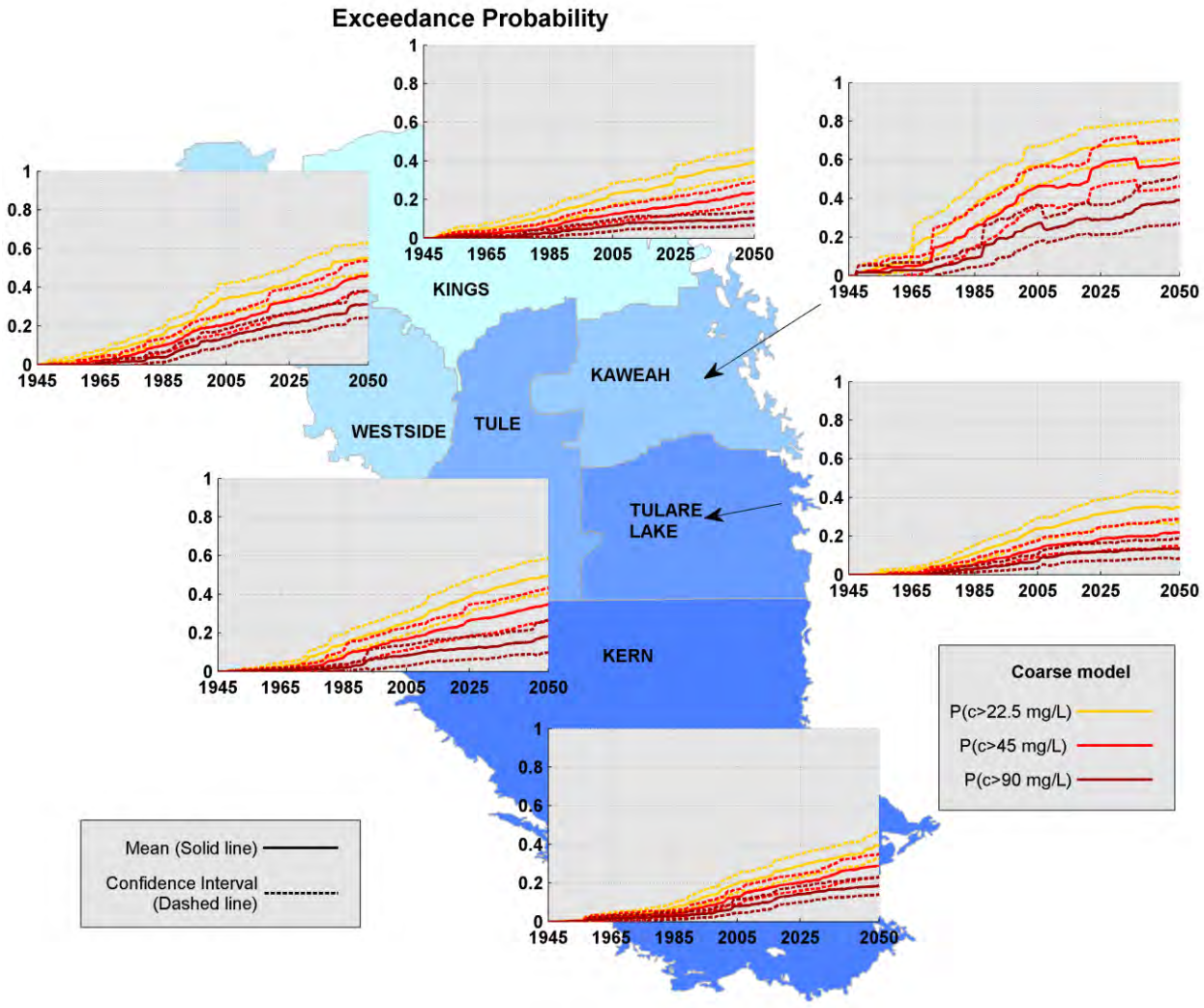

Figure 113. Exceedance probabilities for 22.5, 45, and 90 mg/L based on the coarse model.

For example, by the year 2050, the exceedance probability of the drinking water limit for the basin of Kern, based on the fine model, is approximately 30%, while based on the coarse model the same probability is ~42%. Similarly for the other regions, the two models exhibit a discrepancy on the order of 10% to 20%.

In conclusion, it is apparent that there is a strong disagreement between the model predictions, yet the two models are consistent as far as the impact of the alternative scenarios is concerned. In addition, they are consistent regarding the trends for each basin. The basin of Kaweah exhibits the highest concentration levels among the six basins (e.g., 80% and 60% of wells exceed the drinking water limit based on fine model and coarse model, respectively). For the Kern and Kings basins the two models converge that the concentration and exceedance probabilities will remain low, while a notable increase is expected for the west basins.

Adminsitrative Record
Page 35151

The difference between the two models indicates that two similar models (e.g., both are based on the CVHM model) may exhibit very different results. As shown in the previous paragraph, the streamline length distributions were quite different between the two models, with the coarse model having longer streamlines with older ages. Therefore, it is not surprising that the coarse model predicts lower concentrations. Although the fine model captures more precisely the local variability of the head field, one of the main drawbacks of the fine model is that it fails to maintain the general water balance due to the simplistic decomposition method. For example 85% of the groundwater recharge is diffuse recharge and the remaining corresponds to stream recharge. Yet the number of streamlines that originate from the streams, based the fine model, is approximately 6.5%. For the coarse model it appears that 20% of the streamlines originate from the streams. Therefore, these two models can be seen as the two extremes, where the fine model overestimates nitrate contamination and the coarse model tends to underestimate.

Overall the simulation of nonpoint source pollution for large groundwater basins is a subject that requires further research and deeper understanding of the transport mechanisms. The fine model approach, with adjustments to the simplified domain decomposition method and after accounting for denitrification, is the most promising approach. The coarse model approach, while apparently better in predicting average nitrate concentrations in some instances than the fine model, when comparing to actual average regional data, has been shown to have intrinsic weakness in the computation of travel paths that the fine model is able to address better. As a result, the coarse model appears to make better predictions especially in areas, where in fact denitrification is a significant controlling factor. Further improvements to the fine model approach are needed, and independently measured denitrification rates in the TLB region, to improve on the current modeling capability.

Adminsitrative Record
Page 35152

8 Overall Conclusions: Groundwater Nitrate Occurrence

- 100,000 nitrate records from over a half century of groundwater nitrate sampling on 19,000 wells were archived into a geospatial database.
- Availability of nitrate groundwater data is spatially highly non-uniform and most data represent only the past one to two decades. Much fewer data are available for previous decades.
- Statistical and geospatial analyses consistently show that nitrate concentrations have been increasing in much of the study area over the past 60 years with the exception of the western TLB, where wells are relatively deep and where denitrification in the Corcoran Clay, separating the upper from the lower aquifer system, may be a significant factor.
- The largest increases and highest nitrate concentrations are observed in the unconfined and semi-confined aquifer systems of the central-eastern TLB and in the northeastern and central SV.
- In many areas of the central and eastern TLB and SV, from 20% to over 30% of domestic wells and wells with relatively shallow screen exceed the nitrate MCL.
- The fraction of shallower wells that exceeds the nitrate MCL is likely to continue to increase for some time to come.
- Travel times in the unsaturated zone range from less than one year to decades, depending mostly on depth to groundwater, and on recharge rates under agricultural operations.
- We developed a new groundwater modeling tool that can be used in conjunction with estimated nitrate loading maps for past, current, and future conditions (Viers et al. 2012) to simulate the development of nitrate exceedance probabilities in sub-regional aquifer systems of interest. The model is consistent with measured nitrate concentration distributions and suggests significant increases in nitrate exceedance rates over the coming years.

Adminsitrative Record
Page 35153

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Adminsitrative Record
Page 35181

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Adminsitrative Record
Page 35182

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Adminsitrative Record
Page 35183

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Adminsitrative Record
Page 35184

10 Appendix A - Review of Domestic Well Monitoring Programs for the U.S. 50 States

Prepared by: Katherine Lockhart

10.1 Introduction

Nitrate contamination of groundwater in California's Central Valley may pose a health risk to residents who rely on well water for their drinking and cooking needs. Drinking water supplied by private domestic wells is not currently regulated in California. Well owners are personally responsible for ensuring their water meets drinking water standards and for treatment if water from their well is contaminated. However, many well owners do not regularly test their well water.

There are a variety of reasons why well owners often choose not to test their well water including: cost, lack of information on how/where/what to test for, lack of knowledge about groundwater contamination risks, or choice. Requirements for private domestic well water quality analysis or water quality testing programs and education could help to provide an accurate assessment of the quality of the groundwater for those who rely on domestic wells. California can look to other states that are currently requiring domestic well water quality analysis or that provide testing and educational programs (if any), as an example.

The purpose of this review was to analyze existing requirements for water quality analysis of domestic well water, state-by-state. For this review, links from the Environmental Protection Agency (EPA) website "State Private Drinking Water Wells Web Sites" (United States Environmental Protection Agency 2012) were followed and webpages were reviewed state-by-state. In some cases, the state Health Department was contacted for more information. Educational programs or information for well owners and programs or information provided for water quality testing were also noted (when information was available). These educational and water testing programs and resources are included as examples of the type of assistance available in many states by a multitude of agencies at the local and regional level.

10.2 Summary

This online survey of domestic well water quality regulations and programs demonstrates that some states have only minimal requirements for the testing of domestic well water, and most states have no requirements at all. Among states that do require domestic well water testing, basic analysis requirements vary and include testing upon well installation, repair, maintenance, upon property transfer, or every 5 years for rental properties. This survey found only one state that is currently requiring on-going analysis of private domestic well water: New Jersey requires the owners of rental properties have the properties well water tested once every 5 years and provide the results to the tenants.

Adminsitrative Record
Page 35185

Several states had extensive and helpful information online for homeowners concerned about the water quality from their domestic well (Rhode Island as an example) and many states provided basic information such as lists of certified laboratories or recommended analytes. However, many states may benefit from offering better online resources for homeowners on domestic well water quality.

In terms of sampling assistance or financial support for water testing, this survey found several examples. Financial assistance programs include waiving or reducing the testing fee for low income households, providing free analysis for nitrate if the household includes an infant under six months of age, or providing free testing to well owners in certain high-risk areas. For sampling assistance, in many cases, websites mentioned that local or county health departments would collect a water sample for the well owner for a small fee. Several local and county health department websites offered a detailed explanation of sample collection procedures if the well owner called the department and many departments provide test bottle kits for free or for a small fee. New Mexico's Water Fair and Water-Quality Outreach Program is exemplary in providing free testing while simultaneously delivering rural community education.

Information obtained for each state is summarized below (Table 35) alphabetical order by state. States that have an actual or potential requirement on domestic well water quality testing are highlighted in bold and underlined. This survey was completed in the summer of 2011 and more recent information may be available and/or information may have changed since then. The review does not include many educational programs provided by local, regional, or state agencies. These can be readily found through internet searches. Some examples are included in the table.

Adminsitrative Record
Page 35186

Table 35. Summary of examples for domestic well regulations, testing programs, and educational resources by state. States with regulatory requirements for domestic wells are in bold and underlined.

State	Summary	Department
Alabama	Resources: The Alabama Cooperative Extension System organizes the Home*A*Syst program that offers educational information for well owners (Alabama Cooperative Extension System 2005).	Alabama Cooperative Extension System
Alaska	Resources: Information is provided for well water testing and interpretation of the results (Alaska Department of Environmental Conservation Division of Environmental Health Drinking Water Program 2007).	Alaska Department of Environmental Conservation Division of Environmental Health Drinking Water Program
Arizona	Resources: The University of Arizona Well Owners Help Program's website offers information on wells, advice on well water testing, and a link to the Arizona Department of Health contact for a list of certified labs. The program is conducting a study of well water quality in Cochise County and has offered Saturday workshops for well owners (Arizona Board of Regents 2009). The University of Arizona's Arizona Wells Program offers a web service where interested persons can search for information on nearby wells. However, due to budget cuts, Arizona Wells is currently not operating (University of Arizona 2010).	Arizona Board of Regents and The University of Arizona
Arkansas	Regulatory: Mortgage companies often require safe drinking water results before closing on a home mortgage. Most mortgage companies require that testing be done in an EPA-certified laboratory. Arkansas has no EPA-certified private laboratories so the testing must be done at the Water Microbiology Laboratory in Little Rock or at one of the certified municipal laboratories that provides the service. Testing is probably for coliform organisms only (Arkansas Department of Health 2011).	Arkansas Department of Public Health
California	Regulatory: Testing of wells may be required upon installation. Resources: The Department of Water Resources webpage provides a list of laboratories, analyte[14] recommendations, test result interpretation information, and water treatment information (California Department of Water Resources 2011). The State Water Resources Control Board's Groundwater Ambient Monitoring and Assessment Program (GAMA) has sampled domestic wells in six county focus areas at no cost to the well owners. The State Water Resources Control Board has published many reports from GAMA program findings (State Water Resources Control Board 2012).	California Department of Water Resources
Colorado	Some monitoring of water use is required in specific basins, however it was not clear who performs the monitoring (Colorado Division of Water Resources 2011).	Colorado Division of Water Resources

[14] "analyte" is an individual chemical to be tested in a water sample.

Adminsitrative Record
Page 35187

Connecticut	Regulatory: Upon well installation or property transfer, well owners are required to test the well water for total coliform, nitrate, nitrite, sodium, chloride, iron, manganese, hardness, turbidity, pH, sulfate, apparent color and odor. If pesticides are suspected to have been used in the immediate area, the sample may be required to be analyzed for alachlor, atrazine, dicamba, ethylene dibromide (EDB), metolachlor, simazine and 2, 4-D (Connecticut Department of Public Health 2007). Resources: For additional well water testing, the University of Connecticut Cooperative Extension System offers recommendations on what to test for and provides a link to water testing laboratories University of Connecticut Cooperative Extension System 2012).	Connecticut Department of Public Health and the University of Connecticut Cooperative Extension System
Delaware	Resources: Test kits are available for a fee through the Division of Public Health Office of Drinking Water, with laboratory recommendations provided (State of Delaware, Delaware Department of Natural Resources and Environmental Control, Division of Water 2011).	State of Delaware
Florida	Resources: County health departments will provide instructions on testing and/or collect a sample for the homeowner for a small fee. County health departments provide lists of certified labs (Florida Department of Health, Division of Environmental Health, Bureau of Water Programs 2011).	Florida Department of Health
Georgia	Resources: University of Georgia Cooperative Extension offers Home*A*Syst resources online. Home*A*Syst offers extensive information on water quality risks and helps homeowners decide how to better protect their water quality and determine if their water could potentially be contaminated. County extension offices offer advice on how to collect a sample and where to send it. The Extension also prepares circulars on uranium and arsenic contamination for homeowners and has held workshops in the past covering general water quality, uranium, radon, and arsenic (University of Georgia Cooperative Extension Service, Housing and Environment 2003).	University of Georgia Cooperative Extension
Hawaii	Resources: University of Hawaii Cooperative Extension Service offers a fact-sheet on private well water quality and provides phone numbers for the local health department for various areas (University of Hawaii at Manoa, College of Tropical Agriculture and Human Resources, Cooperative Extension Service 2000).	University of Hawaii Cooperative Extension
Idaho	Resources: The Idaho department of Environmental Quality will provide a list of labs and regional health departments offer to help interpret lab results (Idaho Department of Environmental Quality 2011).	Idaho Department of Environmental Quality
Illinois	Resources: There is no state laboratory fee for the testing of private wells for coliform or nitrate if they meet the following conditions: 1) newly constructed wells, 2) wells that have been affected by flooding, 3) in support of studies related to water borne disease, and 4) wells serving infants under 6 months of age (Dalsin, G.J. 2011). In some cases the local health department will collect the sample for the homeowner. In other cases, the home-owner can request a sample kit that includes sample mailing instructions. Local health departments may charge a testing fee or nominal fee for sample collection (Illinois Department of Public Health, Environmental Health 2011).	Illinois Department of Public Health
Iowa	Regulatory: Well owner is responsible for collecting and testing a well water sample for coliform and nitrate when a well is installed or repaired (between 10-30 days after the well is put into service) (Iowa Department of Natural Resources 2012).	Iowa Department of Natural Resources

Adminsitrative Record
Page 35188

Indiana	Resources: "The Private Well Complaint Response Program receives complaints, investigates, and samples at-risk private water wells which are suspected of being contaminated by man-made contaminants." Homeowners can contact their local county health department to see if their well qualifies (Indiana Department of Environmental Management 2011).	Indiana Department of Environmental Management
Kansas	Resources: Some local health and environmental departments provide screening tests. Kansas State University has a brochure that recommends where to look up labs and where to get help with interpreting well water test results (Powell, G.M., Bradshaw, M.H., & Dallemand, B. 1999).	Kansas State University
Kentucky	Regulatory: KAR 5:037 requires domestic well owners to develop a groundwater protection plan that includes testing for fecal coliform and other contaminants of concern once a year. The plan can be the generic form found at the Energy and Environment Cabinet Department for Environmental Protection website (Kentucky Energy and Environment Cabinet, Department for Environmental Protection 2011). Resources: The KY-A-Syst for the Home program run by the University of Kentucky Cooperative Extension offers well educational information and advice on water testing (University of Kentucky College of Agriculture Cooperative Extension Service 2000).	Energy and Environment Cabinet Department for Environmental Protection and the University of Kentucky College of Agriculture Cooperative Extension Service
Louisiana	Resources: The department of health provides a link to analytical labs and can provide recommendations on what to test for (Louisiana Department of Health and Hospitals, Center for Environmental Health Services 2011).	State of Louisiana Department of Health and Hospitals
Maine	Resources: The department of environmental health has a link to labs and can provide recommendations on what to test for. Assistance with water treatment or well repairs may be available to low-income households (Maine Department of Health and Human Services, Center for Disease Control and Prevention, Division of Environmental Health 2012).	Maine Center for Disease Control and Prevention Division of Environmental Health
Maryland	Regulatory: Well testing may be required when a house is sold or refinanced. Local health departments may test well water (Miller, T.H. 2007).	University of Maryland Extension
Massachusetts	Regulatory: Wells are regulated at the local level and rules may vary. Resources: While there is no state requirement to have well water tested (although there may be from mortgage lenders or local Boards of Health), the Massachusetts Department of Environmental Protection (MassDEP) recommends that all homeowners with private wells do so, and use a state certified laboratory. The Department of Environmental Health website provides recommendations on tests and a link for certified labs (Massachusetts Department of Environmental Protection 2011).	Massachusetts Department of Environmental Protection
Michigan	Resources: Well drillers or mortgage lenders may test domestic well water and the homeowner can access the report. Local health departments may collect a sample as a part of the well inspection and permitting process (Michigan Department of Environmental Quality, Water Bureau 2008).	Michigan Department of Environmental Quality Water Bureau
Minnesota	Resources: County health agencies can perform nitrate and bacteria testing and some operate labs. There is a $30-$40 fee. The Minnesota department of health offers recommendations on what to test for and how often (Minnesota Department of Health 2011).	Minnesota Department of Health

Adminsitrative Record
Page 35189

Mississippi	Resources: The Mississippi State University Extension website has a checklist well owners can go through to determine whether or not their well is at a low, medium, or high risk for contamination. The extension also provides recommendations for what to test for, how often to test, and where testing samples can be analyzed (Bonner, J. 2010).	Mississippi State University Extension
Missouri	Resources: Missouri Department of Health provides free sample bottles and low cost analysis. St. Charles County Division of Public Health offers recommendations on what to test for and how often to test (St. Charles County Division of Public Health 2001).	St. Charles County Division of Public Health
Montana	Resources: Montana State University Extension Water Quality offers the Well Educated Program. "The Well Educated Program guides private well owners through the process of testing water quality, provides materials to help interpret test results, and offers insight on ways to help protect drinking water resources. The program is offered as a service for Montana well owners to provide affordable well testing services accompanied by test result interpretation, while simultaneously providing a useful water quality data source for managers" (Montana State University Extension Water Quality Program 2009 and 2010).	Montana State University Extension Water Quality Program
Nebraska	Resources: Nebraska Department of Health and Human Services provides information on what to test for and where to have sampled analyzed. The Public Health Environmental Laboratory can provide sampling for a fee. Water sample bottles are available from the Public Health Environmental Lab and many local health departments and county extension agents (Skipton, S.O., Woldt, W., Dvorak, B.I. & Pulte, R. 2008). The Nebraska Department of Environmental Quality helps organize "Test Your Well" events that offer free nitrate testing for private well owners (The Groundwater Foundation 2006).	Nebraska Department of Health and Human Services and Nebraska Department of Environmental Quality
Nevada	Resources: Nevada Division of Environmental Protection, Bureau of Safe Drinking Water provides information on laboratories (State of Nevada, Nevada Division of Environmental Protection, Bureau of Safe Drinking Water 2012).	Nevada Division of Environmental Protection, Bureau of Safe Drinking Water
New Hampshire	Regulatory: Mortgage lenders or specific towns may require testing. Resources: New Hampshire Department of Environmental Services website provides recommendations on when and what to test for and provides laboratory information (New Hampshire Department of Environmental Services 2008).	New Hampshire Department of Environmental Services
New Jersey	Regulatory: In 2001, the Private Well Testing Act (PWTA) passed into law, aiming at information disclosure about private drinking water wells. Under the PWTA, certain wells must be tested before a house can be sold. Buyers or sellers must have the water tested and review the results before the close of title. Landlords of certain properties must also test for certain drinking water parameters every five years and provide a written copy of the result to their tenants. Results must also be reported electronically to the New Jersey Department of Environmental Protection. Sample collection and analysis are conducted by laboratories certified by the NJDEPs Office of Quality Assurance. Individuals are responsible to pay for the tests which cost between $450-$650. Parameters include total coliform, VOCs, nitrate, lead, arsenic, mercury, 48-hour gross alpha particle radioactivity, pH, iron, and manganese. Not all tests are required in all areas of the state (Atherholt, T.B., Louis, J.B., Shevlin, J., Fell, K. & Krietzman, S. 2009).	New Jersey Department of Environmental Protection

Adminsitrative Record
Page 35190

New York	Resources: New York State Department of Environmental Health website offers recommendations on what to test for and how often. They provide a link to a list of state certified labs (New York State Department of Health, Bureau of Water Supply Protection 2006).	New York State Department of Health, Bureau of Water Supply Protection
New Mexico	Resources: The New Mexico Environment Department (NMED), Ground Water Quality Bureau (GWQB) conducts free testing of domestic wells throughout the rural areas of the state by holding "water fairs" ten times a year. At the water fairs, domestic well water is tested for free for electrical conductivity, fluoride, iron, nitrate, pH, and sulfate using portable analytical equipment. Domestic well owners are also educated about water quality issues and how they can help preserve or improve water quality in their communities. "This program has proven to be very popular with the general public and continues to provide NMED with valuable information on ground water quality in rural communities. The NMED continues to receive numerous requests for water fairs from community organizations, NMED Field Offices, other State, County and City agencies, and private citizens. The Water Fair and Water-Quality Outreach Program is an important tool for identifying possible non-point source water quality problems" (New Mexico Environment Department, Groundwater Quality Bureau 2012).	New Mexico Environment Department
North Carolina	Regulatory: Wells must be tested when constructed or repaired. The local health department is responsible for the testing and must report the results to the homeowner. Constituents include: arsenic, barium, cadmium, chromium, copper, fluoride, lead, iron, magnesium, manganese, mercury, nitrates, nitrites, selenium, silver, sodium, zinc, pH, and bacterial indicators. Resources: Local health departments can provide information if a homeowner wants to re-test the water (North Carolina Department of Environment and Natural Resources 2012).	North Carolina Department of Environment and Natural Resources
North Dakota	Resources: North Dakota Department of Health offers advice on what to test for and provides a list of labs on their brochure (North Dakota Board of Water Well Contractors 2007).	North Dakota Board of Water Well Contractors and North Dakota Department of Health
Ohio	Resources: The state is leading several nonpoint source pollution investigations that include sampling domestic wells (Ohio Department of Natural Resources, Division of Soil and Water Resources – Ground Water Mapping & Technical Services 2011).	Ohio Department of Natural Resources
Oklahoma	Regulatory: Most lending institutions require a water test before they will approve a loan for purchase or construction of a home. Resources: The Oklahoma Cooperative Extension Service offers educational information on wells and advice on water testing (Oklahoma Cooperative Extension Service 2012).	Oklahoma State University Cooperative Extension Service
Oregon	Regulatory: Homeowners are responsible for testing their water when there is a property transfer. Constituents required are nitrate, total coliform bacteria, and arsenic. Results must be reported to the Department of Human Services Drinking Water Program and to the buyer within 90 days. No financial assistance is available (Oregon Health Authority, Public Health Division 2011).	Oregon Health Authority

Pennsylvania	Regulatory: The state of Pennsylvania does not regulate private domestic wells. Resources: The Department of Environmental Protection offers educational information online concerning well water quality and testing (Pennsylvania Department of Environmental Protection 2011). The Penn State Extension Master Well Owner Network trains volunteers to provide homeowners with assistance for their private water systems (Penn State Extension 2012). The Center for Rural Pennsylvania has conducted at least one study throughout the state that sampled rural domestic wells for a variety of constituents (The Center for Rural Pennsylvania 2009).	Pennsylvania Department of Environmental Protection, Penn State Extension and the Center for Rural Pennsylvania
Rhode Island	Regulatory: State law currently requires testing of new private wells or for certification of occupancy requests (rental properties). State law will be requiring testing for real estate transfers in the near future. Constituents include coliform bacteria, nitrate, turbidity, and chloride. Testing must be done at a state certified lab. Resources: Rhode Island Department of Health provides a web-based interactive map that recommends constituents for well water analysis based on a wells location (Rhode Island Department of Health Private Well Testing Viewer 2011). Rhode Island Department of Health also provides sampling kits, instructions, and a laboratory testing order form. Sampling kits are based on initial/annual testing, 3-5 year testing, and 5-10 year testing, with different kits for each testing period (only the initial testing is required). The price of each test is listed on the form so well owners can pick and choose if they don't want to run the full suite of analytes (State of Rhode Island Department of Health 2011).	Rhode Island Department of Health
South Carolina	Regulatory: Fecal coliform test is required when a new well is installed ($20 extra charge on the well permit). Resources: Additional water quality tests for domestic wells are performed for a small fee and are sometimes waived or reduced depending on the homeowners income or if the well is part of a department groundwater contamination investigation. Constituents include: total or fecal coliform, metals and minerals, other inorganic parameters, volatile organic chemicals, herbicides, pesticides and other synthetic organic parameters. Funds come from funds appropriated for public drinking water oversight monitoring. Free sample kits are provided (South Carolina Department of Health and Environmental Control 2011).	South Carolina Department of Health and Environmental Control
South Dakota	Regulatory: The Centennial Environmental Protection Act of 1989 requires that all new domestic wells drilled in South Dakota are tested for bacteria and several selected chemicals. Resources: South Dakota Department of Environment and Natural Resources provides recommendations for additional well testing and an explanation of why constituents are important. South Dakota Department of Health provides sample bottles (South Dakota Department of Environment and Natural Resources 2011).	South Dakota Department of Environment and Natural Resources
Tennessee	Resources: The Tennessee Department of Environment and Conservation, Division of Water Resources published a Healthy Well Manual that recommends bacteriological testing of well water once every two years and provides advice on well maintenance (Tennessee Department of Environment and Conservation, Division of Water Resources 2012).	Tennessee Department of Environment and Conservation, Division of Water Resources

Adminsitrative Record
Page 35192

Texas	Regulatory: The Texas Commission on Environmental Quality requires that a well water sample be analyzed for coliform bacteria when a well pump is repaired or replaced (Texas Commission on Environmental Quality 2004) and they may sample domestic well water for a specific chemical or for coliform bacteria if they receive a request by a doctor (Texas Commission on Environmental Quality 2003). Resources: The Texas Water Resources institute, Texas Well Owner Network offers well sample screening and provides educational information for well owners (Texas Water Resources institute, Texas Well Owner Network 2012).	Texas Commission on Environmental Quality and the Texas Water Resources Institute.
Utah	Resources: The Utah State University Cooperative Extension provides information on how and what to test for (Utah State University Cooperative Extension 2012), and have a results interpretation tool online (Utah State University Cooperative Extension, Agricultural & Water Quality 2012).	Utah State University Cooperative Extension
Vermont	Resources: The Vermont Department of Health offers recommendations and provides test kits. There may be a charge for test kits and there is a charge for testing. Landlords are required to "provide safe drinking water" but the website did not provide details on what that entails (Vermont Department of Health, Agency of Human Services 2011).	Vermont Department of Health
Virginia	Resources: The Virginia Master Well Owner Network (VAMWON) consists of trained citizens who are available to educate citizens about well water quality and well construction. In addition to VAMWON, the Virginia Household Water Quality Program (VAHWQP) organizes drinking water clinics where well owners can have their water tested and get information on test results (Virginia Tech 2012).	Virginia Tech
Washington	Regulatory: "In most counties, when a home with a private well is bought or sold, the county health or planning department, or the lending institution involved, may require the seller to provide water-sampling results to show the water is safe to drink" (Washington State Department of Health, Division of Environmental Health, Office of Drinking Water 2010).	Washington State Department of Health
West Virginia	Resources: The local health departments may preform tests and provide guidance and recommendations, if requested (West Virginia Department of Health and Human Resources, Public Health Sanitation Division 2011).	West Virginia Department of Health and Human Resources
Wisconsin	Resources: The Department of Natural Resources, Bureau of Drinking Water and Groundwater provide recommendations for testing (University of Wisconsin, Stevens Point, College of Natural Resources, UW Extension, Central Wisconsin Groundwater Center 2010).	Wisconsin Department of Natural Resources, Bureau of Drinking Water and Groundwater and the Central Wisconsin Groundwater Center
Wyoming	Resources: Wyoming Department of Environmental Quality has recommendations for what to test for and how often (a tiered system) (Wyoming Department of Environmental Conservation Division of Environmental Health Drinking Water Program 2007).	Wyoming Department of Environmental Quality, Water Quality Division

Adminsitrative Record
Page 35193

11 Appendix B – Denitrification in Central California Soils and Aquifers

Prepared by: Megan Mayzelle and Thomas Harter

11.1 Denitrification Processes

Nitrate fate below the rootzone can be affected by four processes: soil retention, assimilatory reduction, dissimilatory reduction, and denitrification. Of these, only denitrification can act as a major N sink; the others only temporarily immobilize it (Korom 1992). Denitrification refers to the four-stage process (Bothe 2007) of converting nitrate (NO_3) to nitrogen (N_2) gas, a gas which composes 80% of the Earth's atmosphere (Beller et al. 2002). Soil and aquifer denitrification can help mitigate nitrate loading, thus reducing the environmental and health risks associated with groundwater nitrate contamination. Quantifying the denitrification capacity of a system and its spatial distribution is thus essential to understanding and predicting the ultimate impact of nitrate loading and to designing appropriate approaches to remediation (Ibid.; McMahon & Chapelle 2008).

Active denitrification is evidenced by a loss of nitrate from the system, an accumulation of N_2 gas, and quantitative evidence that the apparent loss of NO_3 is not entirely accounted for by dilution (Ibid.). Denitrification is traditionally considered a microbially-mediated process occurring under anoxic or anaerobic conditions where sufficient dissolved organic carbon (DOC) is present to serve as an electron donor (Korom 1992; Bothe 2007). Various studies, however, suggest that denitrification can occur under a variety of hydrological conditions.

The onset of anaerobic biological denitrification generally occurs when dissolved oxygen is below 0.25 mg O_2/L H_2O (McMahon & Chapelle 2008). Meanwhile, denitrification by facultative anaerobic bacteria has been shown to occur in soils at O_2 levels approaching that of air-saturated water)—8.6 mg O_2/L at 25°C up to more than 14 mg O_2/L at 0°C (Lloyd 1993; Burt et al. 1993; Water on the Web 2007). Research has demonstrated aerobic denitrification to be widespread in natural environments, and several researchers have suggested that it may even be predominant among denitrifying bacteria (Lloyd 1993). Unlike most anaerobic denitrifiers (Korom 1992), however, aerobic denitrifiers generally only partially mediate the denitrification process, thus producing harmful nitrogen oxide gases (Lloyd 1993) instead of harmless N_2. These gases include nitric oxide (NO), which, when exposed to oxygen, is rapidly converted to nitrogen dioxide, a major air pollutant. Nitrous oxide (N_20), a highly reflective greenhouse gas, can also form (Korom 1992). Of the 1,500 soil denitrifiers identified by Gamble et al. (1977), only 146 are capable of complete denitrification. While some methods, such as molecular markers and genetic fingerprinting, exist for identifying taxonomic groups of bacteria, the current understanding of active denitrification populations is still limited by identification procedures (Bothe 2007). Both groundwater and surface waters have been identified as important contributors of emissions of N_2O (Drecht et al. 2003; Bothe 2007); aerobic denitrification may be one explanation of this contribution.

Adminsitrative Record
Page 35194

The presence of heterotrophic versus autotrophic denitrifiers is another possible catalyst for the denitrification process. While many studies demonstrate that denitrification is limited by the availability of dissolved organic carbon (DOC) (Starr & Gillham 1993; DeSimone & Howes 1998), some suggest that denitrification by autotrophic denitrifiers can occur in the absence of DOC (Beller et al. 2002) by using reduced manganese, iron, and sulfides as electron donors (Moran et al. 2011; Korom 1992). There has also been evidence of abiotic denitrification (chemodenitrification) occurring at slightly basic pH under laboratory conditions (Van Hecke et al.1990) as well as in groundwater (Ibid.) and soils (White, R. 2006). Other research shows chemodenitrification producing NO_2 in isolated instances (Ibid.) and only after bacteria have mediated the initial steps of the denitrification process (Brons 1992; Korom 1992). While the tendency of current research is to assume the presence of microbial denitrifiers in anaerobic conditions with DOC as a limiting factor, aerobic denitrification and chemodenitrification may also affect soil and aquifer denitrification (Burt et al. 1993; Bothe 2007).

11.2 Denitrification in Aquifers

The challenges of assessing the occurrence of aquifer denitrification include accurate quantification of hydrogeochemical conditions and spatial tracking (Tesoriero et al. 2007; Rupert 2008). While the segregate and sequential nature of redox processes allow aquifer water quality data to be used to determine denitrification capacity at a given point in the aquifer (McMahon & Chapelle 2008), attributing changes in aquifer NO_3 concentration to denitrification is always accompanied by uncertainties with respect to hydrologic mixing, dispersal, and flow paths (Beller et al. 2002; Rupert 2008 Green et al. 2010), especially in areas with high-capacity pumping wells (McMahon & Chapelle 2008; McMahon et al. 2008). Factors such as recharge rates (Rupert 2008), the depth of the water table (Wright, Belitz, & Johnson 2004; McMahon & Chapelle 2008; Landon et al. 2010), spatial variability in surface-level fertilizer and manure application (Puckett & Cowdery 2002; Puckett et al. 2002; Sobota, Harrison, & Dahlgren 2009), location of the test point within the aquifer (Smith & Duff 1988), and measurement error (Beller et al. 2002) may also contribute to NO_3 concentration variability. Fluctuation in the relative abundance of stable isotopes can provide insight into to the hydrological processes—precipitation, evaporation, mixing, etc.—contributing to the recharge of the aquifer being tested (Moran et al. 2011; Wright et al. 2004), as well as any sources—rocks, manures, plants, etc--of background nitrate levels (Moran et al. 2011). The utilization of various robust indicators is ultimately key to corroborating the predicted denitrification capacity of any given system. While research has pointed toward highly heterogeneous forms of denitrification (described above), anaerobic heterotrophic microbial denitrification is currently the most well-documented (White, R. 2006; Bothe 2007; Rupert 2008) and widely accepted as the dominant form of the process. Thus, when attempting to characterize and predict the denitrification capacity of a system, indicators of environmental conditions permitting this particular denitrification pathway should be considered the most reliable (Bradley, Fernandez, & Chapelle 1992).

Given that oxygen depletion by aerobic microbial respiration occurs over time (Tesoriero et al. 2007), the age of water in a system can often serve as evidence of denitrification potential (Beller et al. 2002). However, this correlation has been shown to be invalid in some studies of the California San Joaquin

Adminsitrative Record
Page 35195

Valley and Salinas Valley aquifer systems (Tesoriero et al. 2007; Moran et al. 2011). The presence of almost exclusively anthropogenic compounds, such as tritium, a radioactive by-product of nuclear reactions (US NRC 2011), is used to age-date water, since most of these chemicals have been produced since 1950 (GAMA 2011). In a study of principle North American aquifers, McMahon & Chapelle (2008) further showed that NO_3 concentrations are significantly larger (indicating less denitrification activity) in water samples containing > 0.5 mg O_2/L and concomitant (Beller et al. 2002) Mn^{2+} and Fe^{3+} concentrations of < 0.05 mg/L and < 0.1 mg/L , respectively. Concordantly, Rupert (2008) observed through a national decadal study that wells with anoxic conditions had significantly smaller increases in nitrate concentrations as a consequence of anthropogenic NO_3 loading than mixed or oxidized wells. McMahon & Chapelle (2008) have thus identified the aforementioned concentrations as threshold values for denitrification to occur. Nevertheless, when using oxygen concentration as an indicator of denitrification capacity, the presence of oxygen-rich microenvironments as well as subsequent blending with oxygen-rich waters must be taken into consideration before a system can be characterized as anaerobic (Ibid.).

An aquifer's porous matrix can provide further important clues as to its denitrification potential. A blue, greenish, or gray color indicates anoxic conditions; these sediments generally represent subaqueous deposition and tend to be relatively higher in organic matter and calcium carbonate than oxidized sediments (Croft 1972). Additionally, since most denitrifiers apparently reside in the aquifer's porous sediment matrix, denitrification rates in sediment core samples are generally higher than rates in water samples (Korom 1992; Tesoriero et al. 2007). In fact, denitrification rates in one aquifer varied from 1.2 to 74.4 nmol/g H_2O/day, while denitrification in the saturated sediments surrounding the same aquifer ranged from 3.8 to 233.6 mmol/m^2/day (Bradley et al. 1992). Various samples may thus be necessary when attempting to characterize biological denitrification potential in order to accurately represent the aquifer's total denitrification capacity and biological diversity. A microbial presence can also be determined by amending an aquifer with an organic carbon source, such as sucrose (Hiscock et al. 1991) and observing sucrose degradation and/or CO_2 production. The addition of an organic carbon source can also help identify a situation in which DOC concentrations are a limiting factor of denitrification capacity (Bradley et al. 1992).

Stable isotope ratios of N_2 and O_2 (Böttcher et al. 1990) and N^2/H^2O concentrations (Beller et al. 2002) are also frequently used as indicators of denitrification. These methods are especially advantageous when investigating age-dated aquifers, since N_2 quantity can then be used to determine the rate of denitrification (Ibid.). Nevertheless, measurements have been constrained by the difficulty in determining the atmospheric N_2 component as opposed to that produced by denitrification (Ibid.). These methods of denitrification quantification have been recently improved by new technologies, such as membrane inlet mass spectrometry, which allows precise determination of the dissolved concentrations of N_2 and O_2 (Ibid.). An additional benefit of this field test is that it quantifies dissolved argon concentrations (Ibid.), which further corroborate indications of denitrification activity (McMahon & Chapelle 2008) by determining the amount of N_2 produced by denitrification as opposed to normally-occurring concentrations (Beller et al. 2002). McMahon & Chapelle (2008) establish a framework for assessing redox processes based on such easy-to-measure and relatively inexpensive parameters (Ibid.).

Adminsitrative Record
Page 35196

In this framework, favorable conditions for denitrification are identified as: $O_2 < 0.5$ mg/L, $NO_3 \geq 0.5$ mg/L, $Mn^{2+} < 0.05$mg/L, and $Fe^{2+} < 0.1$mg/L. In addition to these parameters, McMahon & Chapelle (2008) recommend that other redox indicators (including MH_4, H_2S, CH_4, and H_2) be measured as additional verification. While the application of this framework will not always result in a single-redox diagnosis, McMahon & Chapelle (2008) assert that even a mixed redox diagnosis will be useful in the context of assessing water quality issues.

Aquifer denitrification is likely to remain a topic of interest both within and beyond the scope of groundwater nitrate contamination. Potential exists for amending anaerobic environments with ethanol and acetate (Lorrain et al. 2004) or even sawdust (Israel et al. 2009) to increase denitrification. The possibility of adding nitrate to aquifers to serve as an electron acceptor in the degradation of hazardous organic compounds has also been explored (Alvarez & Vogel 1995; Pelz et al. 2001), although studies have shown that such organic compounds can coexist with high nitrate concentrations in aquifers without significant nitrate denitrification (Moran et al. 2004) . Site restriction for such systems is also very stringent to ensure that drinking water sources remain uncontaminated (Isosaari et al. 2010). Additionally, the role of denitrification of contaminants in the production of anthropogenic N_2O will remain highly relevant as governments face higher accountablity for their greenhouse gas production (Gon 2005; UNFCCC 2007).

11.3 Denitrification in Soils

Soil denitrification depends on the combined effects of hydrological conditions, soil properties, temperature, fertilizer type and application rate, and crop or plant characteristics (Burt et al. 1993; Drecht et al. 2003). Rates of soil denitrification are highly temporally and spatially variable, and can reach 4.5 – 9.0 kg NO_3/ha/day for short periods (White 2006). As in aquifers, denitrification in soils is best known to occur when denitrifying bacteria are present, DOC is available, water content is high, and oxygen levels are concomitantly low (air < 15% porosity at field capacity) (Burt et al. 1993).

NO_3 is highly mobile in soils, and is hence prone to leaching out of the root zone and into the subsoil (>50 cm) during periods of water and NO_3 application in excess of plant demand (Fageria & Baligar 2005). Leaching is the major source of NO_3 in subsoils (Bothe 2007) and in aquifers (Buczko & Kuchenbuch 2010). Additionally, leaching tendency is greater on arable land than grassland (Barraclough et al. 1983) and is further influenced by crop type (McLay et al. 2001), fertilizer type and application rate (Sobota et al. 2009), soil texture and structure (White 2006), and management techniques (Burt et al. 1993) iincluding irrigation (Burt et al. 1993; White 2006). Although most soil denitrification occurs in the top 10-20cm of soil (White 2006), where organic matter concentrations are high, subsoil denitrification has reportedly denitrified up to two-thirds of agricultural nitrate input to a shallow glacial outwash system (Puckett & Cowdery 2002; Puckett et al. 2002). Nevertheless, prevailing conditions can vary considerably from the root zone to the subsoil, meaning that rootzone denitrification capacity is not a determinant of subsoil denitrification capacity (White 2006).

Humid climates (Drecht et al. 2003) and large annual precipitation (White 2006) result in a high leaching rate and short residence time of nitrate in the soil (Drecht et al. 2003); high leaching and short residence

Adminsitrative Record
Page 35197

times can be logically correlated to heavy irrigation activities as well (Burt et al. 1993), especially when temporally near fertilizer applications (White 2006). In contrast, dry climates with restricted precipitation have lower leaching rates and a higher residence time of nitrate in the soil (Drecht et al. 2003). Soil properties also affect denitrification by influencing soil water capacity, oxygen status, and leaching tendencies. Fine-texture and low porosity (either due to massive structure or a large percentage of fine micropores) both tend to hold water more tightly than sandy soils (White 2006), consequently reaching anaerobic conditions more easily and maintaining them for longer periods (Drecht et al. 2003). Poorly drained soils also tend more toward being anaerobic (Burt et al. 1993); although denitrification can begin when soil air falls below 15% of soil pore space (White 2006), rates increase exponentially as water-filled pore space moves upward of 90% (Bothe 2007). Inundated fields, such as for wetland rice, are an extreme example of this condition (Drecht et al. 2003). On the other hand, sandy and high porosity soils tend toward leaching and low soil water saturation (high soil air fraction) (White 2006), which can carry residual soil nitrate into deeper soil layers toward groundwater aquifers. Regardless of the soil type, denitrification outside of periods of irrigation (Burt et al. 1993) or rainfall does not generally occur in shallow agricultural soil systems due to aeration by tilling (Bothe 2007). This combination of high fertilizer input and low denitrification rates results in significant levels of nitrate present in agricultural soils compared to other ecosystems (Burt et al. 1993). Short bursts of microbial growth during irrigation are being studied as a potentially important source of N_2O gas (Smart et al. in press).

While agricultural soils themselves do not normally experience extended periods of denitrification, fertilizers applied only to the surface of the soil have significant volatilization capacity (Burt et al. 1993). Animal manures are particularly vulnerable since they provide NO_3 as well as abundant DOC to denitrifiers (Ibid.); manures also have a particular tendency toward partial denitrification (White 2006), resulting in N_2O production (Korom 1992). Animal feed composition and the length and method of storage alter the composition of manure, which subsequently affects soil denitrification capacity and concomitant N_2O emissions following application (Bothe 2007). A strong knowledge of the plant-available-N content of fertilizers and the site-specific mineralization tendencies is additionally necessary to decrease N losses by adjusting application rates to best reflect crop uptake patterns (Ibid.). The Intergovernmental Panel on Climate Change (IPCC) has developed a model designed for large-scale utilization that combines fertilizer and manure application rates, soil type, and number of animals to determine N_2O emission factors (Gon 2005); this model estimates the default emission factor for N_2O fertilizers and manures at 1.25% of N applied to soil, with an additional 2.5% of all nitrate leached to groundwater eventually being released as N_2O gas as well (Bothe 2007). Such models may be useful in determining state-, region-, or even farm-wide N_2O emissions over time; they also indicate, at the regional level, very limited losses of NO_3 to denitrification being assigned in these models relative to the total amount of NO_3 fluxes to groundwater.

Temperature can serve as a general indicator of rate of microbial denitrification, since cool temperatures generally reduce microbial activity (Korom 1992). Processes are very slow below 10°C (White, R. 2006) but can be stimulated by abundant carbon amendments (Burt et al. 1993). The optimum temperature for denitrification is approximately 40°C. Temperature, along with pH, can

Adminsitrative Record
Page 35198

further affect the ratio of N_2O to N_2 produced by denitrification processes; in cool, acidic conditions, relatively more N_2O is produced, whereas if pH >6 and temperatures 25°C or greater, N_2 is favored (White 2006). This effect of pH and temperature changes on N_2O : N_2 ratios is immediate, and thus a direct effect on N_2O production, as opposed to being as a result of a change in microbial population composition (Burt et al. 1993).

The rate of denitrification over a short period of time can be measured in the field by injecting acetylene, which blocks the reduction of N_2O to N_2, into the soil and collecting N_2O emissions from a limited area (Drecht et al. 2003). This method has a number of disadvantages, including acetylene leaching from the soil, and is logistically inapplicable to subsoil layers since disturbance may aerate the soil (Bothe 2007). N-labeling is an alternative that allows long-term measurements without soil disturbances, but that requires specialized laboratory analyses and expensive materials (Ibid.). Field-scale mechanistic models can also be used in combination with crop production models to simulate denitrification and the events affecting it, such as crop N uptake, changes in soil organic matter (SOM) composition, and leaching (Drecht et al. 2003).

Interesting possibilities exist for exploiting natural soil denitrification processes to mitigate nitrate contamination including overland flow systems, constructed wetlands, and pond systems. Overland flow systems utilize vegetation, microbial communities, and a limited-percolation deep soil layer to facilitate denitrification across a sloped field (Isosaari et al. 2010). While total denitrification in such systems is positively correlated with field length (up to 60 meters) and DOC availability, site-specific evaluations are necessary to determine the sustainable denitrification capacity of a given system (Ibid.). Constructed wetlands, especially those including wetland plants and DOC amendments, offer an indefinite and socially appealing option for catalyzing denitrification. These systems remove a median 18-37% of nitrogen from wastewater; this percentage can be augmented by recycling effluent back into the system (Ibid.). Pond systems, while working on a similar principle, have an inconsistent denitrification rate of 20-80% and require a large land area (Ibid.).

11.4 Study Area Characterization

The Salinas Valley Basin (SVB) extends 120 miles southeast from the Monterey Bay down through the center of Monterey County on the central coast of California (Kulongoski & Belitz 2007). Monterey surpassed Kern County in 2009 to become the 3rd most agriculturally productive county in the nation; agricultural product value for Monterey County now totals more than $4 billion (USDA NASS 2010). Groundwater meets 95% of agricultural, municipal, and industrial water needs in the SVB (Saavedra 2007) and has been historically overdrawn as a result (MCWRA 2007). Most public supply wells and some irrigation wells source from deep groundwater (Moran et al. 2011; Fogg et al. 1999); recharge occurs principally from stream percolation and irrigation return water (California Department of Water Resources (2003).

The SVB has mean annual precipitation of 38 centimeters and a mean annual temperature of 14°C with generally little seasonal variation (Kulongoski & Belitz 2007). The valley is filled with about 800 meters of marine and terrestrial sediments, including as much as 610 meters of saturated alluvium. Water-

Adminsitrative Record
Page 35199

bearing units are unconsolidated to semi-consolidated, interbedded gravel, sand, and silt lying above a consolidated granitic basement; these include the Monterey Formation, Santa Margarita Formation, the Paso Robles Formation, and recent alluvium at the surface (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1997). The Salinas Valley groundwater basin includes the main aquifers at their respective depths: the 180-foot, the 400-foot, and the deep 900-foot aquifers (Saavedra 2007). While the northeast side and seaside portions of these aquifers are confined and semi-confined (Fogg et al. 1999) by clay aquitards (Moran et al. 2011), the 400 and 180 aquifers become unconfined in the midsection of the Valley (Forebay subbasin) and in the southern portion of the SVB (Upper Valley subbasin), respectively (Fogg et al. 1999; California Department of Water Resources 2003). While anoxic groundwater can be found in the northern portions of the valley (Moran et al. 2011), groundwater samples in the Forebay and Upper Valley subbasins area have an average O_2 concentrations of 3.9 mg/L and an average temperature of 19.3°C (Kulongoski & Belitz 2011). Their combined total aquifer storage capacity is approximately 8,820,000 acre feet (California Department of Water Resources 2003). Groundwater overdraft has reduced water levels by approximately 25 meters, resulting in 3-5 mile wide seawater intrusions in the 180 and 400 aquifers (Moran et al. 2011; California Department of Water Resources 2003; MCWRA 2007).

With the exception of the seaside area, groundwater NO_3 concentrations in the 180 and 400 aquifers have been steadily on the rise since the 1950s, primarily as a result of nonpoint agricultural contamination (Fogg et al. 1999; California Department of Water Resources 2003). Nitrate concentrations in the SV show high spatial and temporal variability, probably due to movement and mixing caused by the draw from well pumps, as well as the varying depths of the tested wells (Moran et al. 2011). The California DWR reports that 9.7% of public supply wells in the SV exceeded the NO_3 maximum contaminant load (MCL) from 1994-2000 (Ibid.). In a 1995 study of irrigation and monitoring wells, 23 of 35 wells (66%) tested in the Upper Valley region and 30 of 81 wells (37%) tested in the Forebay region exceeded the MCL for NO_3; the average NO_3 concentrations were 98 and 45 mg NO_3/L respectively (California Department of Water Resources 2003). In contrast, a 2005 study of the Forebay and Upper Valley Aquifers detected NO_3 in 6 of 19 wells (32%) tested, with an average well NO_3 concentration of only 3.15 mg/L as N (Kulongoski & Belitz 2007) . Most recently, Moran et al. (2011) have reported one deep drinking water well in the San Jerardo area as having NO_3 concentrations ranging from 69-130 mg/L throughout 2010 and sporadic concentrations ranging from 100-681 mg/L throughout the SV.

The Tulare Lake Basin (TLB), comprised of portions of Fresno, Kern, Kings, and Tulare counties, sits in the south of the San Joaquin Valley (SJV) of California. Five of the eight counties in the SJV rank among the top ten agriculturally most productive in the nation, with the market value of agricultural products sold from the SJV totaling $18.3 billion (USDA NASS 2007). Groundwater serves as the primary drinking water source for nearly 90% of SJV residents (Community Water Center's Health and Drinking Water Series 2011). While depth to groundwater in the Central Valley is generally shallow (California Department of Water Resources 2003), water levels in the TLB are comparatively much lower (Faunt 2009). TLB groundwater levels dropped nearly 17 feet from 1970 through 2000, with 7 feet of loss occurring between 1999 and 2000 alone (California Department of Water Resources 2003).

Adminsitrative Record
Page 35200

Groundwater in the Central Valley is generally well-oxygenated (Beller et al. 2002), and principally recharged by surface irrigation and stream recharge (Groundwater Management Technical Committee 1999; California's Groundwater Bulletin 2006).

The TLB receives an average of 18 cm annual precipitation and has an average annual temperature of 20°C, with large seasonal variations from 0°C to 39°C (National Weather Service). The TLB is filled with up to 9,700 meters of marine and continental sediments (California Department of Water Resources 2003). The younger alluvium sits largely near and above the water table (ibid.) underlain by the generally poorly sorted deposits of the older alluvium and continental deposits (Croft and Gordon 1968; Croft 1972). These are moderately to highly permeable water-bearing units (Croft 1972; California Department of Water Resources 2003) and form the principle aquifers for the region (Croft & Gordon 1968). The Corcoran Clay, the most prominent of various very low permeable clay layers spreading over the region divides the subsurface into an upper unconfined to semi-confined aquifer and a lower confined aquifer (Croft & Gordon 1968; California Department of Water Resources 2003). Anoxic conditions generally prevail in the Corcoran Clay and other, less prominent clay layers, often designated as the A-clay, B-clay, C-clay, D-clay, and F-clay (the Corcoran Clay is also referred to as the E-clay). Anoxic conditions also prevail in the hundreds to several thousand feet thick lacustrine and paludial clay deposits underneath the lakebed of (former) Tulare Lake west of the town of Corcoran, which yield very little water of poor quality (Croft & Gordon, 1968).

11.5 Denitrification Potential in Tulare Lake Basin and Salinas Valley

A comparison of NO_3 concentrations between various North American principle aquifers (PA) revealed that the Central Valley sand and gravel PAs, along with the western volcanic PAs, have the largest NO_3 concentrations on the continent (McMahon & Chapelle 2008). This finding is consistent with the high N loading from fertilizer and manure recorded in those areas. Electron transfers from Mn^{4+} and Fe^{3+} reduction is relatively small in these aquifers and redox conditions are the least heterogeneously distributed of all North American PAs (Ibid.). Movement of NO_3 into the aquifers would be expected to divert electrons from these compounds since NO_3 sits higher in the succession of terminal electron-accepting processes.

Subsurface conditions are often variable and change on regional and local scales (Green et al. 2008; Green et al. 2010; Landon et al. 2011). In general, a lack of correlation between anaerobic conditions and groundwater age has been observed in the study area (Moran et al. 2011; Tesoriero et al. 2007). On the whole, it is thus likely that denitrification does not affect nitrate concentrations significantly in the production aquifer system of the Central Valley (Ibid.; Tesoriero et al. 2007).

While not the dominant process affecting nitrate concentrations in the Central Valley as a whole, denitrification may be affecting concentrations in certain areas of the valley, in particular in shallow groundwater near the valley trough. Previous investigations in the San Joaquin Valley have characterized the groundwater as being mostly oxic. Anoxic conditions that may lead to denitrification have been found to be more prevalent in shallower parts of the aquifer system where groundwater has historically discharged into wetlands and streams near the central trough of the valley (Davis et al. 1959;

Adminsitrative Record
Page 35201

Croft & Gordon 1968; Croft 1972; Bertoldi et al. 1991; Dubrovsky et al. 1993; Burow et al. 1998; Singleton et al., 2007; Landon et al. 2011). There, a shallow groundwater table with historically upward groundwater flow and the presence of large amounts of organic matter provide shallow subsurface conditions with extremely limited oxygen availability but large amounts of dissolved organic carbon as electron donor for microbially mediated redox reactions that create anoxic conditions in the shallow-most aquifer sediments.

Anoxic conditions generally prevail below the oxic zone, in deeper sections of the older alluvium and continental deposits, typically below depths of 500 to more than 1,000 ft throughout the Tulare Lake Basin (Croft 1972). Due to the large age of groundwater in the anoxic zone, it is currently uncertain, to which degree such anoxic conditions may slow or prevent future nitrate pollution (Landon et al. 2011).

Soil denitrification in the TLB and SV plays a potentially significant role. Riparian zone, shallow aquifer, and stream/groundwater denitrification have been shown to remove up to 93 mg NO_3/L from the Merced River, a gaining river (one whose flow increases due to groundwater discharging to the surface) just north of the TLB in the SJV (Domagalski et al. 2008). Similar results have been found for the Parajo River, a losing river (one whose flow decreases due to surface water recharging groundwater) just north of the mouth of the SV at Monterey Bay. Ruehl et al. (2007) report that NO_3 concentrations in the Pajaro River decrease downstream across 11km while other ions concentrations remain unchanged, with denitrification rates peaking where seepage loss and surface-subsurface exchange are the greatest and stream discharge is the lowest (Ibid.). Moran et al. (2011) have further found NO_3 concentrations in monitoring wells near the Salinas River to be significantly lower than those located directly in the Salinas River (about 10 mg NO_3/L). Isotopic signatures of the water samples indicated that denitrification was taking place, and that this process may occur in other areas where similar conditions exist near the river. Denitrification was not, however, found to occur in areas away from the Salinas River where oxic groundwater (high levels of dissolved oxygen) was found.

11.6 Conclusions

The naturally occurring process of denitrification in soils and aquifers is increasingly being recognized for its relevance to current anthropogenic NO_3 and N_2O contamination. Scientific understanding of the various denitrification pathways has been restricted by effective assessment techniques, especially with respect to aquifer and subsoil denitrification potential. Oxygen concentrations, DOC concentrations, and microbial presence have been acknowledged as key conditions of system denitrification potential; other factors, including pH, temperature, fertilizer types and rate, vegetation type, and hydrologic processes such as irrigation and precipitation, have also been identified as important influences. Nonpoint agricultural emissions, including inorganic fertilizers and manures, are the principle sources of NO_3 contamination of groundwater in the SV and TLB of California. Although some initiatives have begun to analyze this issue, the problem of water insecurity in the SV and TLB deserves further attention. While the potential for aquifer denitrification is limited where most groundwater is pumped in these areas, managed surface-level denitrification systems merit additional exploration as a plausible form of NO_3 contamination mitigation.

Adminsitrative Record
Page 35202

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 5:

Groundwater Remediation and Management for Nitrate

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
http://groundwaternitrate.ucdavis.edu

Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 35203

Groundwater Remediation

Technical Report 5

Addressing Nitrate in California's Drinking Water

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

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Adminsitrative Record
Page 35204

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Contents

Figures iii

Tables iv

Acronyms and Abbreviations v

Unit Conversions vi

Summary 1

1 Introduction 2

2 Groundwater Remediation Options 4

Estimation of Volume for Basin-wide Remediation 4

Hypothetical Plume Size 6

2.1 Pump-and-Treat 7

2.1.1 Pump-and-Treat Using Drinking Water Treatment 8

2.1.2 Pump-and-Treat Using Remediation Basins – Wood Chip Bioreactors (WCBRs) 14

2.2 Phytoremediation of Nitrate in Groundwater 16

2.2.1 Background 16

2.2.2 Phytoremediation of Nitrate 17

2.2.3 Required Acreage for Complete Treatment 17

2.2.4 Phytoremediation Conclusions 18

2.3 In Situ Denitrification 19

2.3.1 In Situ Bioremediation/In Situ Redox Manipulation (With Injection of Carbon Source) 22

2.3.2 In situ Denitrification Using Permeable Reactive Barrier 28

2.4 Pump and Fertilize (PAF) at the Basin Scale 38

2.4.1 Overview of Pump and Fertilize 38

2.4.2 Understanding the Value of Irrigation Water Nitrogen 41

2.4.3 Current Use of Irrigation Water Nitrate in Fertilizer Calculations 41

Adminsitrative Record
Page 35207

2.5 Management of Groundwater Recharge 42

3 Summary and Conclusions 43

4 Literature Cited 44

Appendix 51

Adminsitrative Record
Page 35208

Figures

Figure 1. Pump-and-treat remediation - plume-scale application.....13

Figure 2. Hypothetical in situ bioremediation scenario.....23

Figure 3. Barrier configuration of in situ biological denitrification.....24

Figure 4. General schematic of a permeable reactive barrier.....28

Figure 5. Permeable reactive barrier configurations: Continuous, Reactive vessel, Funnel and gate.....29

Figure 6. Potential application of PRBs in the Tulare Lake Basin (right) and Salinas Valley (left).....34

Figure 7. WCPRB for treatment of infiltrate from a single half-section field or orchard.....35

Figure 8. Hypothetical placement of 120 miles of WCPRBs to intercept and treat infiltrate from a high nitrate-loading area with shallow groundwater.....36

Adminsitrative Record
Page 35209

Tables

Table 1. Total groundwater and remediation volume listed by subbasin (DWR 2003). 6

Table 2. Drinking water treatment cost estimation. 11

Table 3. Estimated basin-wide pump-and-treat water treatment costs using drinking water treatment technologies 12

Table 4. Acreages of land used for application of effluent from wastewater treatment facilities in the study areas, compared to estimates of the acreage needed for complete transpiration using phreatophyte trees 19

Adminsitrative Record
Page 35210

Acronyms and Abbreviations

AF	Acre Feet
BD	Biological Denitrification
CD	Chemical Denitrification
HRT	Hydraulic Retention Time/Hydraulic Residence Time
ISB	In Situ Bioremediation
ISRM	In Situ Redox Manipulation
MCL	Maximum Contaminant Level
MGD	Million Gallons per Day
PRB	Permeable Reactive Barrier
SV	Salinas Valley
TLB	Tulare Lake Basin
WCBR	Wood Chip Bioreactor
WCPRB	Wood Chip Permeable Reactive Barrier

Adminsitrative Record
Page 35211

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Adminsitrative Record
Page 35212

Summary

This report explores methods and costs of remediation of groundwater nitrate contamination in the Tulare Lake Basin and Salinas Valley. Groundwater cleanup, or remediation, is one of the most difficult actions in the environmental sciences, even when done on the scale of a small contaminant plume (1000s of cubic meters). Remediation of entire groundwater basins has never been attempted at the scale and depths of the Salinas Valley and Tulare Lake Basin, on the order of billions of cubic meters. This analysis shows that direct remediation to remove nitrate from large groundwater basins is extremely costly and not technically feasible. In situ remediation, though technologically infeasible as a regional remedy, is appropriate in certain areas of shallow groundwater with high contaminant levels. Traditional pump and treat (ex situ) methods are too slow to produce results on the regional scale in an acceptable time frame, prohibitively expensive, and impractical to implement.

A novel form of pump-and-treat remediation is possible in the study area, but not yet widely practiced. By monitoring the nitrogen content of pumped irrigation water, and taking that ambient nutrient source into account when calculating additive fertilizer amounts, farmers can reduce the amount of nitrogen input to the aquifer. Such an approach is not unlike phytoremediation and is herein called "pump-and-fertilize." Pump-and-fertilize is part of an effective nutrient management program. Due to the nature of irrigation with groundwater, it is as much a source reduction method as it is a form of groundwater remediation.

Hot spot and pump-and-fertilize remediation alone will not solve the problem of groundwater nitrate contamination. A new approach is needed that combines regional groundwater and nitrogen management strategies. This approach would include monitoring, source reduction, maximization of clean groundwater recharge across agricultural landscapes, pump-and-fertilize, and in situ treatment targeted at shallow, high-concentration plumes to place regional water quality on a trajectory toward improvement. Modeling and monitoring of regional groundwater quality are integral to an effective long-term management strategy.

Adminsitrative Record
Page 35213

1 Introduction

Groundwater remediation is the cleanup of contaminated groundwater to levels that are in compliance with regulatory limits. This involves either ex situ or in situ methods. In the ex situ approach, groundwater is extracted by wells, treated on the surface, and put back in the subsurface by injection wells, percolation basins, or similar means. This approach, referred to as pump-and-treat (PAT), uses water treatment technologies to remove or reduce contaminants in the pumped groundwater. The difficulty and cost associated with this approach largely stems from the often intractable problem of pumping not removing enough of the contamination from the aquifer in a reasonable time frame (e.g., less than a human lifetime). In situ remediation requires a detailed understanding of existing subsurface conditions in order for it to be effective. In the in situ approach, subsurface conditions are created that favor the degradation of the contaminants into less harmful products. The in situ approach is not appropriate for contaminants that are spread over large regions or are recalcitrant to degradation. Both ex situ and in situ methods are typically accompanied by removal or reduction of the sources of contamination.

Groundwater remediation is one of the most difficult tasks in environmental cleanup (NRC 1994; NRC 2000). Historically, groundwater remediation has only been done at the plume scale (< 1 km^2). Recalcitrant contaminants, such as nitrate, are difficult to remove, and can require decades of effort. Moreover, the success rate for cleanup of recalcitrant groundwater contaminants is poor (NRC 1994; NRC 2000). It is not unusual for plumes of recalcitrant contaminants to undergo remediation for several decades without reaching cleanup goals. Plume containment through flow barriers and/or manipulation of groundwater head gradients can be part of a remediation strategy, or it can be a goal in itself when remediation is deemed impossible.

The cost and difficulty of plume remediation rises dramatically with the age[2], size, and depth of the plume. Older plumes typically have a substantial fraction of their mass residing in the portions of the groundwater system with lower hydraulic conductivity, such as silts and clays in aquifer systems like the Tulare Lake Basin and Salina Valley. Groundwater contamination that entered the subsurface over a time period of months to years will take much longer to flush from the system, on the order of decades to centuries or more, whether active or passive remediation approaches are used. Under such cases (more the norm than the exception), speeding up the cleanup would require increasing the rate of molecular diffusion, an impossibility in all but relatively small aquifer volumes where thermal energy remediation techniques could be applied.

Because of the difficulty and poor success rates of plume remediation, an approach known as monitored natural attenuation (MNA) has become popular in recent years. In MNA, the natural attenuation processes of biochemical transformation and dispersion reduce and dilute contaminant mass to below cleanup goals. Simultaneous monitoring confirms whether MNA is adequately protecting groundwater quality. This approach is only effective for contaminants that can transform to relatively harmless byproducts via biological or chemical transformation.

[2] Plume age is defined as the period in which a plume has been migrating in the subsurface.

Adminsitrative Record
Page 35214

While it is possible for denitrification to fully degrade nitrate, the combination of circumstances that would favor denitrification is generally lacking in California's alluvial aquifer systems (Fogg et al. 1998). In natural aquifer systems, these circumstances include anaerobic conditions, a carbon source, and substantial microbial activity. Given the very low carbon content of most alluvial aquifer materials in California, and that microbial activity in soils is orders of magnitude greater than in the underlying alluvial deposits, the greatest potential for denitrification is in the biologically active soil zone, found in the upper 1-2 meters of the earth. Soils tend to be orders of magnitude more microbiologically active than the underlying, geologic parent material (Kazumi and Capone 1994). Because of California's semi-arid climate, however, the water table is typically at least 10 m deep, leaving the soil zone largely without the anaerobic conditions needed to drive denitrification. Well below the water table, where anaerobic conditions tend to occur, the natural geologic materials generally lack the carbon source and the microbial activity required for useful rates of denitrification. Given the ongoing spread of nitrate contamination in California's aquifer systems, it is reasonable and prudent to view most nitrate in California groundwater as recalcitrant to denitrification unless protected by thick overlying clay layers. Thus, MNA is unlikely to be a viable remediation strategy.

Favoring nitrate remediation, the regulatory limit is often only a factor of 2 to 10 times lower than the typical nitrate concentrations in contaminated groundwater in the current study areas (see Technical Report 4, Boyle et al. 2012, for discussion of current nitrate concentrations in wells in the study areas). However, this does not mitigate the problem of the scale of the contaminated basins. Plume-scale remediation may be worthwhile in certain parts of the basins, but, there are no examples of remediation of entire groundwater basins on the scale of the Tulare Lake Basin or Salinas Valley.

Adminsitrative Record
Page 35215

2 Groundwater Remediation Options

Remediation in the classical sense for the basin-scale groundwater nitrate contamination in the Tulare Lake Basin and Salinas Valley is not technically feasible in a reasonable period of time. Once aquifers are regionally contaminated, subsurface heterogeneity, together with the processes of groundwater flow and transport, combine to render the reversal of the contamination extraordinarily difficult. It is nevertheless useful to consider remediation options at multiple scales, to provide context and understanding of the processes involved for policy makers. For context, we examine a scenario in which the basin contamination problem is treated with the PAT method. Additionally at the basin scale, we suggest regional groundwater and nitrate management practices that can be used to improve groundwater quality over the long term. We emphasize that nitrate source reduction is key to any successful long-term solution to the nitrate problem in the study areas. As part of a regional strategy, we also examine the feasibility of removal of hot spots or highly concentrated sources of contamination, including options for local-scale (plume scale and somewhat larger) nitrate remediation.

Part of a regional groundwater and nitrate management strategy involves using existing wells and modified agricultural practices to simultaneously remove contaminated groundwater and reduce nitrate loading at the surface through the use of nitrogen uptake from irrigation water by crops, accompanied by optimized fertilizer application. We refer to this practical, basin-scale approach as pump-and-fertilize. This approach is further described in Section 2.4 of this report.

At the plume scale, several options for the remediation of nitrate impacted groundwater are available; guidance documents have been developed by various agencies to assist in the selection, design and management of remediation systems (see U.S. EPA 1990 and ITRC 2002). Plume-scale remediation options are discussed in Sections 2.1.1 and 2.3 of this report.

Unfortunately, the data needed to accurately estimate the basin-wide volume of groundwater requiring remediation by direct measurement do not exist. Most of the nitrate contamination is in the upper portions of the saturated groundwater systems where little well monitoring data exists, while most of the data on nitrate occurrence are from deeper portions of the groundwater systems where nitrate has either not yet arrived or is gradually increasing due to downward migration from above. We therefore estimated the volume of contaminated groundwater by calculating the fraction of the basin surface area in which well data indicated nitrate contamination and using this fraction to estimate that portion of the total groundwater volume that is contaminated.

Estimation of Volume for Basin-wide Remediation

The CASTING database (detailed in Technical Report 4, Boyle et al. 2012) was queried for wells with any nitrate data available between January 1, 2000 and December 31, 2009[3]. Thiessen polygons were created for each of these wells. A Thiessen polygon is defined by a point of interest, in this case, the

[3] We note that, for this computation, an (earlier) version of the CASTING database was used that did not include the Central Valley Regional Water Board dairy wells.

Adminsitrative Record
Page 35216

well. All the area that lies closer to the particular well than to any other well comprises the Thiessen polygon for that well. The average nitrate concentration in each well during the period from 2000 to 2009 was assigned to each respective Thiessen polygon.

To simplify calculations, and because we were more concerned with the total volume than with the distribution of heterogeneity, we chose to ignore non-aquifer sediments (silts and clays) entirely, although these sediments may store considerable volumes of contaminated groundwater. Therefore, rather than assuming a total porosity to calculate the volume of water requiring remediation, an effective porosity of 0.1 was used.

The area overlying each groundwater basin, taken from Department of Water Resources (DWR) Bulletin 118 (2003), was intersected with the Thiessen polygons for the wells. The areal fractions of each basin contained in its intersected Thiessen polygons were calculated and then multiplied by the total volume of the basin (DWR 2003) to produce a volume underlying each Thiessen polygon. The nitrate concentration (average from the 2000's) from each well was assigned to its corresponding volume. The volumes that contained nitrate levels above ½ the Maximum Contaminant Level (MCL) [scenario 1] (>5 mg/L as N or 22.5 mg/L as nitrate) and above the MCL [scenario 2] (> 10 mg/L as N or 45 mg/L as nitrate), for each basin, are summarized in Table 1.

For scenario 1, using the fraction of wells exceeding ½ the MCL, the estimated volumes of groundwater to be remediated in the Tulare Lake Basin and Salinas Valley are 94.2 km^3 (76.4 million acre feet, AF) and 8.9 km^3 (7.2 million AF), respectively. For scenario 2, using the fraction of wells exceeding the MCL, the estimated volumes to be remediated in the Tulare Lake Basin and Salinas Valley are 39.7 km^3 (32.2 million AF) and 4.2 km^3 (3.4 million AF), respectively.

Adminsitrative Record
Page 35217

Table 1. Total groundwater and remediation volume listed by subbasin (DWR 2003).

Sub-Basin	Total Groundwater Volume in Study Area (Million AF)	Remediation Volume > ½ MCL (Million AF) [% of Total]	Remediation Volume > MCL (Million AF) [% of Total]
Tulare Lake Basin			
5-22.06 – Madera	1.2	0.31[26%]	0.12 [10%]
5-22.07 – Delta-Mendota	2.6	0.13 [5%]	0.13 [5%]
5-22.08 – Kings	93	29.73 [32%]	10.34 [11%]
5-22.09 – Westside	52	3.10 [6%]	1.35 [3%]
5-22.10 – Pleasant Valley	4.0	2.52 [63%]	0.90 [23%]
5-22.11 – Kaweah	34	13.98 [41%]	7.39 [21%]
5-22.12 – Tulare Lake	37	11.77 [32%]	3.77 [10%]
5-22.13 – Tule	33	6.65 [20%]	3.48 [11%]
5-22.14 – Kern	40	8.16 [20%]	4.71 [12%]
TLB TOTAL	*297*	*76.4 [26%]*	*32.2 [11%]*
Salinas Valley Basin			
3-4.01 – 180/400 Foot Aquifer	6.86	2.21 [32%]	0.74 [11%]
3-4.02 – Eastside	2.56	1.56 [61%]	1.00 [39%]
3-4.04 – Forebay	4.53	2.10 [46%]	1.11 [25%]
3-4.05 – Upper Valley	2.46	0.99 [40%]	0.45 [19%]
3-4.08 – Seaside	0.63	0.23 [36%]	0.06 [10%]
3-4.09 – Langley	0.36 [1]	0.09 [24%]	0.03 [9%]
3-4.10 – Corral de Tierra	0.49 [2]	0.07 [15%]	0.002 [0.5%]
SV TOTAL	*17.9*	*7.2 [41%]*	*3.4 [19%]*
STUDY AREA TOTAL	***315***	***83.6 [27%]***	***35.6 [11%]***

[1] Storage, actual groundwater volume not listed.
[2] From Montgomery Watson (1997), not listed in Bulletin 118.

The estimated volumes of groundwater requiring remediation are based on available nitrate data for wells across the study area. Note that the density of wells having nitrate data is not consistent across the study area. As a result of this heterogeneity, uncertainty in the model is variable and increases inversely with the density of wells in any given region. These volumes are used to estimate rough costs for remediation options listed below.

Hypothetical Plume Size

Plume size, boundary, depth, and volume are site specific. For the purposes of this analysis a typical plume is defined as having a width of 500 m (1,640 ft), a depth from land surface to plume bottom of 75 m (246 ft), and a length of 2000 m (6,562 ft), spanning 100 ha (250 acres), with a total groundwater volume (porosity of 0.1) of 7.5 million cubic meters (6,080 AF). Typical plume characteristics are based on what would be expected for plumes beneath waste discharge areas, lagoons, and excess fertilizer or manure application to agricultural fields. A thorough analysis of site characteristics would be required for accurate plume delineation and cost estimates.

Adminsitrative Record
Page 35218

2.1 Pump-and-Treat

Pump-and-treat, a type of ex situ remediation, refers to the extraction of contaminated water from the subsurface followed by treatment and subsequent discharge of treated water to groundwater (i.e., through reinjection or percolation) or surface water. Most commonly known for the remediation of organic chemicals at superfund sites, PAT has been implemented in a variety of applications to address both inorganic and organic constituents of concern. Currently, the application of PAT is shifting toward containment efforts rather than complete plume remediation, protecting public health and limiting the migration of contaminant through the capture and treatment of the leading edge of contaminant plumes (U.S. EPA 2005; U.S. EPA 2007a; U.S. EPA 2007b; Faris 2011). PAT is generally considered when contaminant plumes can be clearly defined; however, for the purpose of this investigation, the extreme example of basin-scale treatment will also be considered.

It is important to be mindful of the long-term nature of nitrate movement into and within the aquifer. Over many years, nitrate is in contact with the heterogeneous porous media of the subsurface, with several key processes affecting their fate and transport, including advection, dispersion, and diffusion. Complete removal of the contaminant is virtually never accomplished in one round of PAT. Water that is pumped from the subsurface comes generally from the highly conductive materials, while water within areas of low conductivity is removed much more slowly, if at all. The concentration of nitrate in the low-conductivity regions is not necessarily less than that in the high-concentration regions. Thus, reinjected, clean water will come into contact with untreated water and diffusion of nitrate from the latter to the former will prolong remediation efforts. These migration processes of diffusion and slow advection are much slower than the groundwater movement in the coarse-grained material of the system. PAT remediation is therefore a lengthy process, with diminishing returns, even for plume-scale contamination.

Options for the treatment of extracted water include existing wastewater or drinking water treatment facilities, newly constructed treatment facilities dedicated to groundwater remediation, constructed wetlands, and remediation basins (ITRC 2000; ITRC 2002). These treatment options can be applied for remediation in a PAT scenario.

Options for the fate of treated water include reinjection wells, percolation basins, infiltration galleries, and discharge to surface water or to storage. For the purposes of this exercise, it is assumed that treated water will be returned to groundwater due to the need to replenish the irrigation and drinking water supplies. Infiltration galleries are not appropriate for any but small-scale recharge projects.

Percolation requires land area for the percolation ponds and allows little control over groundwater recharge rate and depth. This option may be most suitable for recharge from remediation basins due to the low volume rate of output of remediation basin treatment, and the typical placement of remediation basins in areas where space is not a limiting factor. Infiltration basins are also beneficial in regions where protecting the shallow groundwater is the primary goal, as opposed to protecting the deeper water. Hydraulic head in the region below the percolation basin will be increased, causing groundwater flow away from the basin laterally.

Adminsitrative Record
Page 35219

Reinjection requires reinjection wells and provides control over the depth and location of recharge. The extraction and injection of water can be used to control hydrologic flow by controlling head gradients. In some situations, decreasing or reversing gradients can be used to contain contaminants by preventing migration beyond the bounds of current contamination. By increasing the local head gradient, flow rate can be increased, thus flushing targeted subsurface regions (Isherwood et al. 1992).

Monitoring will be required of both the effluent from the treatment system to ensure recharge of clean water, and of the groundwater to assess remediation progress. Optimization of the system is fundamental to cost effective long-term operation; extraction from some wells may not be necessary for the same duration as others (U.S. EPA 2005).

2.1.1 Pump-and-Treat Using Drinking Water Treatment

Basin-scale application of PAT using drinking water and wastewater treatment technologies is presented as an extreme option with remediation of the entire aquifer. Local application of this remediation method to hot-spots/nitrate plumes is also considered.

Nitrate Treatment

Water treatment to address high nitrate levels consists of two major categories; reduction technologies and removal technologies. Reduction technologies include biological denitrification (BD) and chemical denitrification (CD), both of which transform nitrate through reduction to other nitrogen species, preferably to innocuous nitrogen gas. The removal technologies include anion exchange (IX), reverse osmosis (RO) and electrodialysis/electrodialysis reversal (ED/EDR), which remove nitrate and other constituents, to a concentrated waste stream requiring disposal. Disposal of this waste stream can be costly and disposal options are particularly limited for inland communities.[4] Wastewater treatment plants are generally designed for the removal of organic material but can also include biological nutrient removal (BNR) to remove nitrogen through nitrification and denitrification.[5] In the selection of the most appropriate treatment method, it is important that co-contaminants are taken into consideration.[6] Co-contaminants must also be considered when re-injecting the treated water. Discharge or injection permits often need to be obtained, and removal of other constituents may be required. Each of the above treatment technologies has the potential to remove other constituents of concern to varying degrees, in addition to nitrate. For example, biological treatment has been used for removal of both perchlorate and nitrate, while reverse osmosis can address many co-contaminants simultaneously including arsenic, salt, nitrate, and others.

Biological treatment was successfully deployed for treatment of nitrate contaminated groundwater in San Diego, CA, decreasing nitrate levels from ~18 mg/L to <4.5 mg/L as nitrate (4 mg/L to <1.0 mg/L as

[4] See Technical Report 6, Section 3 (Jensen et al. 2012), for more information on these technologies in the context of drinking water treatment.
[5] See Technical Report 3, Section 5 (Dzurella et al. 2012), for more information on wastewater treatment technologies for nutrient removal.
[6] See Technical Report 6, Section 3 (Jensen et al. 2012) for additional information on the impact of water quality parameters on the selection of treatment.

Adminsitrative Record
Page 35220

nitrate) with discharge to a nearby creek (Envirogen 2010). In an assessment of ex situ remediation options for perchlorate, biological treatment was found to be the most commonly considered alternative, accounting for 69% of case studies reported (NASA 2006). Treatment for perchlorate and nitrate can be similar; both anion exchange and biological treatment are employed for the removal of these constituents. Capital and operations and maintenance costs of biological treatment have been described as being lower than costs for anion exchange, especially when the cost of waste brine disposal for anion exchange is considered (Harding ESE 2001 from NASA 2006). "In ex situ denitrification, the University of Colorado has shown that a complete denitrification system costs approximately $60 per 3,780 L treated (Silverstein 1997)" (ITRC 2000, p. 40).

In Southern California, large desalters have been used to address salinity. Consisting predominantly of reverse osmosis treatment (sometimes supplemented with anion exchange treatment), these large treatment plants are also being used to reduce high nitrate levels. Chino Desalters I and II are drinking water treatment plants combining RO and IX technologies. Using the combined treatment technologies, nitrate levels are decreased from 70-300 mg/L as nitrate (16-68 mg/L as N) in the influent to 22-35 mg/L as nitrate (5-8 mg/L as N) in the effluent with influent TDS of 1100 mg/L (CDA 2010) (blending is also used, see Technical Report 6, Jensen et al. 2012, for more information).

For the purposes of this investigation, we assess only one round of extraction, treatment, and reinjection (per unit of water) to remove nitrate from the groundwater. Accordingly, these scenarios also assume no additional nitrogen inputs into the groundwater. However in practice, as discussed above, reinjected, clean water will come into contact with untreated water and mass transfer of nitrate from the latter to the former in the presence of subsurface heterogeneity will prolong remediation efforts.

Two treatment options will be considered: biological treatment and a combined treatment system using reverse osmosis and anion exchange (RO/IX). Biological denitrification was selected as an appropriate treatment option based on the prevalence of its use in remediation scenarios. Although biological denitrification is not commonly used in the U.S. for drinking water treatment, two full-scale biological denitrification plants are being implemented in California for drinking water treatment (Webster & Crowley 2010; Brown & Bernosky 2010).[7] In the context of PAT remediation, biological denitrification offers the advantage of low cost treatment, and, in contrast to RO/IX, biological denitrification reduces nitrate to nitrogen gas rather than concentrating nitrate in a waste stream. The combined RO/IX treatment system, while capable of addressing many constituents of concern (including nitrate), produces brine that is costly to dispose of.

Basin-Scale Pump-and-Treat Remediation with Drinking Water Treatment

For the remediation of an entire basin, the portion of groundwater impacted by nitrate would need to be extracted, treated and recharged. From above (Table 1), for scenario 1, using the fraction of wells exceeding ½ the MCL, the corresponding total volumes to be remediated in the Tulare Lake Basin and Salinas Valley are 94.2 cubic kilometers (76.4 million AF) and 8.9 cubic kilometers (7.2 million AF),

[7] See Technical Report 6 (Jensen et al. 2012) Section 3.4.5 for related case studies.

Adminsitrative Record
Page 35221

respectively. For scenario 2, using the fraction of wells exceeding the MCL, the corresponding total volumes to be remediated in the Tulare Lake Basin and Salinas Valley are 39.7 cubic kilometers (32.2 million AF) and 4.2 cubic kilometers (3.4 million AF), respectively. We do not consider this basin-scale PAT scenario to be either economical or feasible. This scenario is presented for context and to convey the scale of the problem.

For complete aquifer treatment, multiple large treatment plants would be required throughout the Tulare Lake Basin and Salinas Valley; the number, location, and distribution of plants would require an analysis of pipeline and pumping costs (for transport to and from the plant) relative to treatment plant costs. While it would be less costly to build and operate one very large treatment plant than multiple smaller plants, the cost of transporting water across the regions of interest would be insurmountable. In this simplified hypothetical example, the cost of treatment is assessed on the basis of a $/1000 gallons cost for a large plant (10 million gallons per day (mgd)). Pipeline and pumping costs for transport of water from remote locations to a large centralized facility are not included and would increase the total cost.

Factors affecting treatment cost include facility capacity (how much water), source water quality (including nitrate concentration), environmental factors (temperature and pH), and target effluent nitrate concentration. In this analysis, the focus is the removal of nitrate. Costs of nutrient removal in wastewater treatment include numerous processes other than denitrification and are thus not directly applicable. Instead, treatment costs for biological denitrification are used as a proxy for simulation of denitrification in BNR. Treatment costs are based on direct contact with facilities[8] and include capital and O&M costs adjusted to 2010 dollars. Costs reported are for complete plant operation and would therefore include monitoring and pumping costs. Total capital costs were converted to annualized capital costs per 1000 gallons ($/kgal) based on Eqn. 1.

$$\text{Annualized Capital Cost} = [\text{Capital Cost (\$)} * \text{Amortization Factor}] / [\text{Flow (MGD)} * 1000\ \text{kgal/mgal} * 365\ \text{days/year}] \quad \text{(Eqn. 1)}$$

An amortization value of 0.0802 was used which corresponds with an interest rate of 5% over 20 years (Eqn. 2).

$$\text{Amortization Factor} = (1+i)^N/((1+i)^N - 1)/i) \quad \text{(Eqn. 2)}$$

Where i = interest rate and N = number of years

Annual O&M costs were calculated based on Eqn. 3 to convert total annual O&M costs to $/kgal.

$$\text{O\&M Cost (\$/kgal)} = [\text{O\&M Cost (\$)}] / [\text{Flow (MGD)} * 1000\ \text{kgal/mgal} * 365\ \text{days/year}] \quad \text{(Eqn. 3)}$$

[8] Cost estimation is based on drinking water treatment costs reported by BIOLOGICAL DENITRIFICATION treatment systems and a combined RO/IX system (See Table 2).

Adminsitrative Record
Page 35222

Annualized Capital Cost ($/kgal) and O&M Cost ($/kgal) were summed to determine Total Annualized Cost ($/kgal). The corresponding costs of treatment using a) biological denitrification, and b) reverse osmosis combined with anion exchange, are listed in Table 2. For reference, the costs of treatment using anion exchange alone are also provided. Listed biological treatment costs are based on a published cost analysis for a 10 MGD drinking water treatment plant (Meyer et al. 2010). Costs for the RO/IX combined scenario are based on reported costs of a drinking water treatment plant using reverse osmosis and ion exchange with a brine line for disposal (CDA 2010). Costs of ion exchange treatment alone are based on a published costs analysis for a 10 MGD drinking water treatment plant using evaporation ponds for brine waste management (Meyer et al. 2010).

Table 2. Drinking water treatment cost estimation.

	Capital Cost ($/kgal)	O&M Cost ($/kgal)	Total Annualized Cost ($/kgal)
Biological Treatment[1]	0.43	0.75	1.18
Reverse Osmosis and Ion Exchange[2]	0.83	1.80	2.63
Ion Exchange[3]	0.36	0.87	1.23

[1] 10 MGD system, Meyer et al. (2010). Other costs available for smaller systems from Webster & Togna (2009), and Carollo Engineers (2008).
[2] CDA (2010).
[3] 10 MGD system, Meyer et al. (2010).

Based only on treatment costs and the calculated volume to be treated, the total annualized cost of remediation in the Tulare Lake Basin and Salinas Valley is listed in Table 3. The total annualized cost of remediation across the entire study area would be $32.1 billion [scenario 1] and $13.7 billion [scenario 2] using biological treatment and $71.6 billion [scenario 1] and $30.5 billion [scenario 2] using a combined RO/IX system. The duration of remediation would depend on the number of facilities and their design capacity; however, costs listed above are based on a 20 year amortization. To remediate the entire basin under scenario 2 (nitrate above the MCL, the lesser total volume) in a 20 year time frame, using multiple 10 mgd treatment plants, more than 140 plants would be required in the TLB and more than 15 plants would be required in the SV. This does not account for the proximity to high nitrate areas or the distribution of treatment plants that would be required. For remediation of the estimated volume of groundwater exceeding ½ the MCL, additional plants would be required. The significantly lower costs of biological treatment in this remediation scenario illustrate the importance of accounting for disposal costs. The use of the removal technologies (RO/IX), especially on such a large scale, would not be possible without complete optimization of waste recycling or an inexpensive means of disposal. Such an operation would be more feasible with coastal access to the ocean for disposal of waste brine; water treatment facilities with high brine/concentrate waste volumes (e.g., desalters and desalination plants) are typically located near an ocean. As mentioned above, the costs of the RO/IX combined treatment scenario are based on a treatment plant with access to a brine line. In the context of drinking water treatment, the management of waste brine is discussed is greater detail in Technical Report 6, Section 6.4 (Jensen et al. 2012).

Adminsitrative Record
Page 35223

Table 3. Estimated basin-wide pump-and-treat water treatment costs using drinking water treatment technologies.

	Total Annualized Remediation Cost (2010 $)			
	Biological Denitrification Treatment		Combined RO/IX Treatment	
	Scenario 1 (> ½ MCL)	Scenario 2 (> MCL)	Scenario 1 (> ½ MCL)	Scenario 2 (> MCL)
TLB	29.4 billion	12.4 billion	65.5 billion	27.6 billion
SV	2.8 billion	1.3 billion	6.2 billion	2.9 billion
TOTAL	*32.1* billion	*13.7* billion	*71.6* billion	*30.5* billion

We note that only a small portion of all groundwater in the two study areas is used for drinking water. The remaining remediated groundwater would be ultimately (re-) pumped and used for crop irrigation, a water use for which the nitrate regulation does not need to be met, and for which elevated nitrate levels can help meet crop nitrogen needs. Assuming that future nitrate loading to groundwater was to not exceed maximum allowable nitrate levels, this scenario would illustrate the cost of one-time complete aquifer remediation to address legacy contamination. However, as previously mentioned, significant low conductivity material present in the aquifers will provide diffuse sources of nitrate for decades, requiring more than one round of PAT remediation, thus, this estimate is likely to be a low estimate of cost and time requirements.

Basin-wide remediation, by whatever means, will be a long-term process. Short-term solutions would be needed to address nitrate levels in public and private drinking water supply wells during the period of time needed for complete aquifer remediation. These (short- and intermediate-term) costs would need to be added to the cost for remediation.[9] If additional treatment is required to avoid groundwater degradation due to constituents other than nitrate, costs would likewise increase. Measures to ensure a significant reduction of ongoing nitrogen loading to groundwater would also need to be implemented simultaneously, further adding to the cost of this scenario. Again, this scenario was presented for completeness only; we do not consider basin-wide pump-and-treat to be either economical or feasible.

Plume-Scale Pump-and-Treat Remediation with Drinking Water Treatment

PAT remediation at the local level can be implemented to address current and historical discharges of nitrogen that have created highly concentrated nitrate plumes (e.g., beneath an unlined waste discharge pond). Remediation of such known hot-spots in the vicinity of drinking water sources has the potential to avoid the need for drinking water treatment for nearby water systems and household wells. Through remediation of high-nitrate plumes, contamination can be mitigated before nitrate leaches deeper into the aquifer and disperses to impact a larger area. Remediation of a highly concentrated nitrate plume is more cost effective than remediation of a much larger volume of water with diluted contaminant levels, such as a basin-wide scenario as presented above. It is important to keep in mind that nitrate contamination is generally dispersed basin-wide and from numerous non-point and point sources. A

[9] See the analysis of alternative water supplies in Technical Report 7 (Honeycutt et al. 2012).

Adminsitrative Record
Page 35224

plume-scale remediation scenario is only applicable to point-source pollution. PAT remediation of a contaminant plume is illustrated in Figure 1.

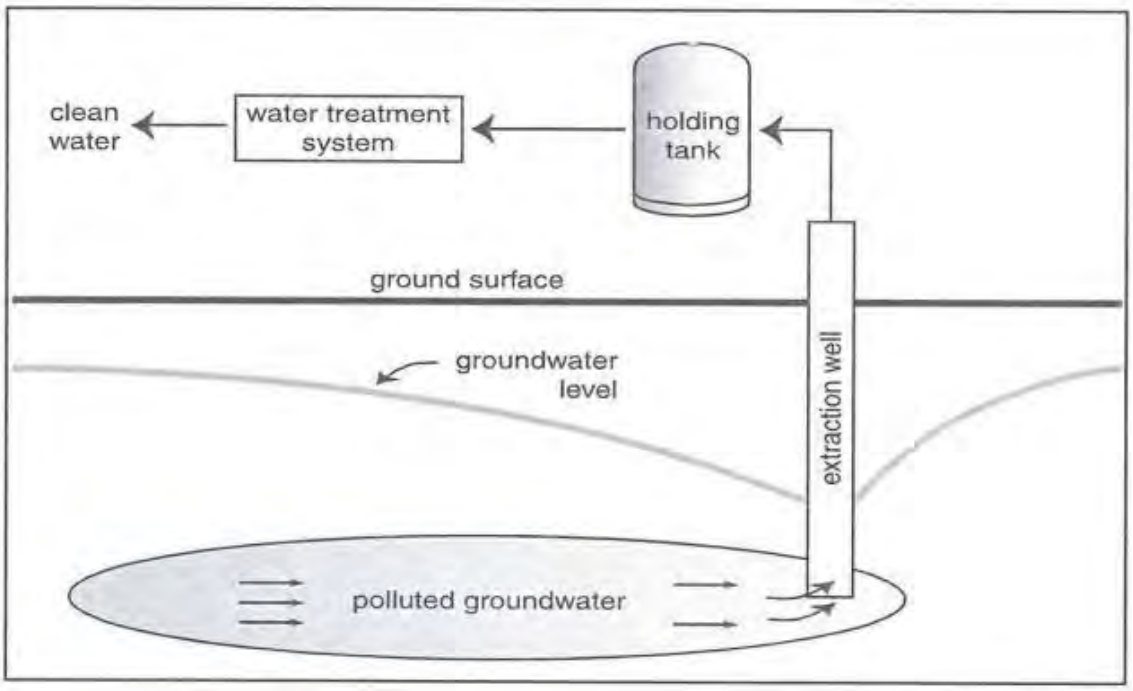

Figure 1. Pump-and-treat remediation - plume-scale application. (Source: U.S. EPA 2001b.)

Locations of hot spots and plume delineation would be required to locate the extent and boundary of the high-nitrate plume.[10] Optimization of placement of extraction and reinjection wells would be needed to ensure capture of the contaminant plume. For the purposes of this analysis, a typical plume is defined as discussed above in the Section *Hypothetical Plume Size*.

Drinking water treatment options and costs in this scenario are the same as those listed in Table 2 above ($/kgal), which include pumping, but not new well construction. Two treatment types are considered here; however, the selection of the appropriate treatment technology will be dependent upon the nitrate concentration, the presence and concentration of co-contaminants (e.g., arsenic, salt, perchlorate, etc.), availability of affordable disposal options, and additional site-specific characteristics. As in the consideration of basin-wide remediation, costs are for a large treatment plant (10 mgd). If a smaller treatment plant were deemed more appropriate, total capital and O&M costs would decrease; however, the cost per 1000 gallons generally increases as plant size decreases. The need for additional extraction, monitoring and reinjection wells will increase total costs. Preliminary costs associated with plume delineation are also excluded.

The calculated total annualized costs of plume-scale remediation in this scenario are $2.3 million and $5.2 million, for biological treatment and combined RO/IX treatment, respectively. Using a 10 mgd

[10] A nitrogen loading analysis of the study areas, along with a cumulative nitrogen loading map is presented in Technical Report 2 (Viers et al. 2012).

Adminsitrative Record
Page 35225

plant, the entire plume volume would be remediated over a period of 6 months; however total annualized costs are based on a 20 year amortization. In the actual application of drinking water treatment technology for remediation of a plume of this size, a smaller plant would be more appropriate, having a capacity of about 0.3 mgd for remediation over 20 years. The priorities for a given site would determine whether the more appropriate option would be the use of a large plant for faster remediation or the use of a smaller plant for remediation over a longer period of time with lower upfront capital costs, but long-term O&M costs. To address multiple plumes across the study area, a plausible option would be the use of a portable treatment system that could be used to sequentially address multiple plumes over a long duration. The costs and sizing of such a system would require additional research.

It is somewhat counter-intuitive to treat extracted groundwater to drinking water standards and then return the clean water to the aquifer, rather than using treated water directly for potable water supply. This scenario was presented for completeness, with the intent of showing how treating highly contaminated nitrate plumes can avoid dispersion of nitrate to a wider impacted area.

2.1.2 Pump-and-Treat Using Remediation Basins – Wood Chip Bioreactors (WCBRs)

Denitrification with solid carbon sources has been used in treatment of wastewater, groundwater, and agricultural runoff. The most common tested applications are denitrification walls for shallow groundwater, basins or beds for concentrated discharges, and horizontal layers for leachate. Typically, shredded or chipped wood is used as the carbon source. Wood chips provide biochemical oxygen demand that strips all available oxygen from the water, creating the habitat for denitrifying bacteria. These bacteria use the carbon of the wood chips as an electron donor in the process of biological denitrification.

Above-ground Wood Chip Bioreactors (WCBRs) for remediation of nitrate with wood chip biomass are simple to install and have been proven effective in agricultural runoff treatment (Blowes et al. 1994; Moorman et al. 2010; Schipper et al. 2010), and decentralized wastewater treatment (Leverenz et al. 2010). A typical installation consists of a basin 1 to 3 meters deep, lined with an impermeable layer that is filled with wood chips. Water to be treated is injected into one end of the basin through a manifold and removed from the basin at the other end.

Longevity of the treatment system becomes more important with increasing cost and difficulty of installation. Sub-surface (below-grade) installations are more difficult to maintain, while at-grade or above-grade containerized bioreactors can be maintained and monitored very easily, however, sub-surface systems allow use of the over-lying land. Moorman et al. (2010) found a half-life of 36.6 years for wood chips under anoxic conditions in a bioreactor treating agricultural drainage. When the wood chips were exposed to oxygenated water periodically, this dropped to 4.6 years. Robertson et al. (2000) concluded that wood chip bioreactors operated to maintain anoxic conditions could function without replacement of the chips for decades with a consumption rate of about 3% of original wood chips after 7 years of operation.

Adminsitrative Record
Page 35226

A variant of this technology also incorporates wetland plants such as the common cattail, *Typha latifolia* (Leverenz 2010). The plants themselves take up nitrate, increasing the efficiency of the system. The roots of the plants preferentially fill in voids in the wood chip media that could otherwise develop into preferential flow paths leading to short-circuiting of the bed. The plants are also visual indicators of the function of the WCBR. A well-functioning, planted WCBR should have healthy plants at the upstream end and unhealthy or no plants at the downstream end, where available nitrogen should be eliminated.

Two anoxic treatment wetland systems (WCBRs planted with wetland plants) have been installed at safety roadside rest areas by the California Department of Transportation to remove nitrogen from restroom wastewater (Leverenz 2011). The systems, which have been in operation for approximately six months, are located near Shandon, CA, and El Centro, CA, and treat approximately 15,140 and 37,850 liters per day (4,000 and 10,000 gal/d), respectively. Each system is composed of two horizontal plug flow reactors operated in parallel, with a total wood chip volume of 344 cubic meters (450 cubic yards) for each system. The hydraulic retention times/hydraulic residence times (HRT) for these systems are approximately 9 and 3.6 days, respectively. The influent nitrate concentrations for both systems range from 20 to 40 mg nitrate-N/L, while the effluent has no detectable nitrate.

A one year experiment conducted by Blowes et al. (1994) tested the performance of a pair of pilot-scale WCBRs in treating agricultural tile-drain runoff. These 200 liter (53 gallon), unplanted bioreactors treated up to 60 liters (16 gallons) per day from inflow concentrations up to 27 mg/L as nitrate (6 mg/L as N) to effluent concentrations below the detection limit (0.09 mg/L as nitrate, 0.02 mg/L as N).

Residence time is the key parameter for sizing bioreactors. Robertson and Cherry (1995) observed that groundwater flow rate was inversely related to the nitrate removal efficiency of sub-surface wood chip denitrification walls used for treatment of groundwater. This is due to the dependence of removal rates on hydraulic residence time in the reactor. Residence times in WCBRs vary with temperature, influent concentration and desired effluent concentration, but typical values range from 1.3 to 15 days (Blowes et al. 1994; Robertson et al. 2000; Greenan et al. 2009; Leverenz 2010; Schipper et al. 2010; Moorman et al. 2010).

The porosity of wood chips in a packed basin ranges from 0.3 to 0.5 (void volume over total volume) (Robertson et al. 2000; Hamel & Krumm 2008). Assuming the middle of that range, and a 10-day hydraulic residence time, a reactor designed to treat 38 liters (10 gallons) per minute (equivalent to 54,510 liters (14,400 gallons) per day) would require about 1,380 cubic meters (1,800 cubic yards) of wood chips.

Wood chip prices per cubic yard from commercial vendors are typically between $7 and $20, depending on location, season, and quality. Higher quality chips have less fines and inorganic material than low quality chips. Fines may reduce porosity in the wood chip bed, thus creating the potential for reduced flow through parts of the bed, and reduced denitrification efficiency. Inorganic materials such as sand, metal (from nails, etc.), or paint will have no significant effect on denitrification at typical levels found in low-quality chips, such as chips made from pallets or construction/demolition waste.

Adminsitrative Record
Page 35227

In a PAT scenario, at-grade[11] WCBRs could be used to enhance denitrification of pumped well water. The limiting factors on this treatment technology would be the cost and supply of wood chips, the depth of the groundwater level, and the cost of pumping. Additionally, water quality regulations present a possible complicating factor for use of the WCBR technology for PAT. WCBRs do not treat for all constituents, and thus the effluent may require further treatment prior to use as drinking water or reinjection to groundwater.

In the first several months of operation, WCBRs produce effluent with high dissolved organic carbon content. This problem is simple to mitigate through cascade aeration, but does add some cost as well as requiring space.

2.2 Phytoremediation of Nitrate in Groundwater

2.2.1 Background

Phytoremediation is defined in the 1999 U.S. EPA Phytoremediation Resource guide as "the direct use of living plants for in situ remediation of contaminated soil, sludges, sediments, and ground water through contaminant removal, degradation, or containment" (U.S. EPA 1999, p. vii). Some phytoremediation schemes rely on the ability of plants to take up contaminants into their tissues, which are then harvested or otherwise removed, while others use the plants to produce a soil environment for microbial degradation of contaminants. Phytoremediation can be conducted with terrestrial or wetland plant species. Phytoremediation is most useful as a method of interception of contaminants on their path to the aquifer, though treatment of aquifer contaminants in situ is possible for shallow aquifers under certain circumstances.

Terrestrial plant phytoremediation can be used to intercept nitrate from septic leach fields and other shallow subsurface applications, such as land application of pumped groundwater (a pump and treat alternative) or wastewater from municipal or industrial sources. Contaminated runoff from flood or furrow irrigated agricultural fields could also be treated via phytoremediation. The pump and fertilize concept described in Section 2.4 of this report is essentially a phytoremediation option, whereby the constituent nitrogen in pumped groundwater for irrigation is accounted for in the calculation of fertilizer input rates. Within the study area, another example of the use terrestrial plants to reduce groundwater nitrogen loading is the land application of effluent and solid wastes from food processing and wastewater treatment facilities.[12]

Application of phytoremediation with terrestrial plants is limited to the vadose zone and the top surface of the saturated zone. Roots of these plants do not grow deeply into the saturated zone even when that is very shallow.

[11] The top of the basin is level with the surrounding land surface.
[12] See Technical Report 2, Section 6.2 (Viers et al. 2012) and Technical Report 3, Section 5.2 (Dzurella et al. 2012) for additional information.

Adminsitrative Record
Page 35228

Dense plantings with large evapotranspiration rates can create a zone of depression in a shallow water table, causing flow towards the phytoremediation site, enabling the remediation of saturated-zone groundwater; however, this in situ application is unlikely to be feasible in most of the study area, due to lack of sufficiently shallow groundwater. Again, in general, nitrate phytoremediation projects have been more successful when implemented as part of a long-term strategy to control nitrogen flux to groundwater rather than as treatment for contaminated groundwater. Targeted applications, designed to treat contaminated flows at their source, have been tested and found effective (Schnoor 1995; U.S. EPA 1999; Perry 2009), and could be implemented in the study areas.

Typical terrestrial plants used for nitrate phytoremediation include phreatophyte trees (e.g., poplar, willow, cottonwood, aspen), grasses (e.g., rye, bermuda, sorghum, fescue), and legumes (e.g., clover, alfalfa, cowpeas) (Schnoor 1997). Phreatophyte trees transpire much more water than typical agricultural crops (Blaney 1958). Root depths of the listed tree species are essentially never over 3 – 4 meters (9.8 – 13.1 feet), and can be much shallower depending on soil conditions (Crow 2005). The mature root systems of rye and sorghum can extend to around 1.4 meters (4.6 feet) in ideal conditions, while alfalfa and clover taproots can extend to over 3 meters (9.8 feet), but are rarely over 4 meters (13.1 feet) (Weaver 1926). These rooting depths are sufficient for the uptake of nutrients (and other contaminants) in leachate of septic systems (typical leach field depths range from 1 to 2 meters (3.3 to 6.6 feet) below ground surface).

2.2.2 Phytoremediation of Nitrate

McKeon et al (1996), in an investigation of phytoremediation with 2 phreatophyte species, estimated 4.1 metric tonnes (4.5 short-tons) per year of nitrate removal on the 24 hectare (59.3 acres) site, assuming no grazing and non-manipulated canopy coverage rate (25% coverage was assumed). Maximum nitrate concentration in the plume was 1,200 milligrams per liter. Plume volume at time of the study was 2×10^6 m^3 (1,620 acre feet). They further estimated that pumping the groundwater to irrigate the trees would result in full remediation to acceptable levels (44 mg/L) of the entire plume in 20 years. This form of phytoremediation was entirely based in the accumulation and assimilation of nitrate in and by the plants.

Phytoremediation of contaminated flows in constructed wetlands is mediated primarily through enhanced denitrification, although accumulation and assimilation also occur. In a review of this technology, Horne (2000) found removal rates of established stands of wetland plants from 540 to 1220 mg/L per m^2 per day. This type of remediation requires that the water to be treated is either pumped from the aquifer, or intercepted before entering the aquifer.

2.2.3 Required Acreage for Complete Treatment

Complete phytoremediation requires transpiration of 100% of the flow to be treated, thereby removing all of the contaminant of interest. As an example of a small-scale phytoremediation application, we assume phreatophyte trees are used to treat the leachate from a septic system on a 4-person household. Phreatophyte trees typically transpire around 3000 liters (793 gallons) of water per tree per

Adminsitrative Record
Page 35229

year, and typical phytoremediation plantings are at a density of 3700 trees per hectare (1500 per acre) (Schnoor 1997). Assuming a typical flow rate of 800 liters (211 gallons) per day for a household of 4 persons (Tchobanoglous et al. 2003), a septic system phytoremediation site would need an area of roughly 16 by 16 meters (52 feet by 52 feet) of trees to transpire all of the wastewater flow from the septic system. This simple exercise ignores the problems inherent in planting trees over a leachfield, such as root growth into the leach field apparatus. Though the details of such an application would need to be dealt with, it is conceivable that phytoremediation of septic leachate could be used in rural residential areas where parcel sizes are large enough to accommodate plantings of this size, but small enough (high septic system density) that septic systems contribute a substantial amount of nitrogen to groundwater.

At a larger scale, the outflow of a wastewater treatment plant such as the Visalia municipal plant, with a flow of 12.25 MGD, would require about 3800 acres of trees for complete transpiration, or 150 ha (307 acres) of trees per MGD. The Visalia plant currently applies 7.1 MGD of its effluent to 910 ha (2250 acres) of silage and cotton crops, with the balance of the effluent wasted to a 97 ha (240 acre) percolation basin. At the 150 ha (307 acres) per MGD rate, 882 ha (2180 acres) of trees would be required for complete transpiration of 7.1 MGD. Table 4 summarizes the agricultural lands applied effluent flows in comparison to the acreage required for complete transpiration based on the 150 ha (307 acres) per MGD estimate.

2.2.4 Phytoremediation Conclusions

Phytoremediation is most useful for interception of contaminated flows rather than as an in situ treatment, except in areas of very shallow groundwater. Sufficient plantings for complete phytoremediation of nitrate require substantial areas (150 ha (307 acres) per million gallons per day with phreatophyte trees). Currently, many wastewater treatment facilities in the study areas apply effluent to adequate acreage; however, this can be misleading, as the rate of transpiration of agricultural crops is much lower than that of the optimal plant species.

Adminsitrative Record
Page 35230

Table 4. Acreages of land used for application of effluent from wastewater treatment facilities in the study areas, compared to estimates of the acreage needed for complete transpiration using phreatophyte trees.

Facility Name	Reported Ag-applied Flow (MGD)	Reported Ag Acreage	Est. Acreage for Complete Transpiration	Reported as Pct of Estimate
WOODLAKE WWTF	0.46	35	141	25%
MCFARLAND WWTF	0.55	75	169	44%
KING CITY DOMESTIC WWTF	0.435	65	134	49%
TAFT WWTF	1.2	185	368	50%
PORTERVILLE WWTF	3.7	620	1136	55%
CUTLER-OROSI WWTF	0.6	106	184	58%
TULARE WWTF	10.8	2000	3316	60%
DELANO WWTF	4.28	1145	1314	87%
KERN SANITATION AUTHORITY WWTF	3.9	1100	1197	92%
NORTH OF RIVER WWTF	5.5	1740	1689	103%
VISALIA WWTF	7.105	2250	2181	103%
BAKERSFIELD WWTP #3	9.76	3148	2996	105%
LEMOORE NAS WWTF (naval services)	0.95	306	292	105%
FRESNO CO #41-SHAVER LAKE WWTF	0.5	161	154	105%
GONZALES WW	0.265	85	81	105%
LINDSAY WWTF	0.65	210	200	105%
MILLERTON NEW TOWN WWTF AND RECYCLING	0.355	114	109	105%
FRESNO REGIONAL WWTF	9.78	3670	3002	122%
BAKERSFIELD WWTP #2	13.7	5476	4206	130%
WASCO WWTF	0.9	390	276	141%
LAMONT WWTF	2	1150	614	187%
SANGER INDUSTRIAL WWTF	0.25	188	77	245%
MRWPCA REG TRTMT & OUTFALL SYS	14	12000	4298	279%
HANFORD WWTF	2.45	4000	752	532%
ARVIN WWTF	1.1	6000	338	1777%
LEMOORE WWTF	2	13333	614	2171%

2.3 In Situ Denitrification

As an alternative to groundwater extraction and treatment, under appropriate conditions, nitrate impacted groundwater can be addressed in situ. In situ methods can be less costly than ex situ options and have the ability to directly target the groundwater contaminant plume while taking advantage of naturally occurring processes of denitrification. Two major categories of in situ denitrification are considered: Enhanced In Situ Biological Denitrification (EISBD)/In Situ Redox Manipulation (ISRM) and Permeable Reactive Barriers (PRBs).

Adminsitrative Record
Page 35231

The In Situ Bioremediation (ISB) Team of the Interstate Technology and Regulatory Council (ITRC) developed a detailed reference document as a guide for in situ remediation options (ITRC 2002). The ITRC (2000, 2002) specifically addresses options for the remediation of nitrate impacted groundwater and states, "All indications point to enhanced in situ biodenitrification as a reasonable remediation alternative for nitrate- (NO_3) contaminated groundwater" (ITRC 2002, p. iv). See Figure A-1 of the Appendix for a detailed decision-tree on the application of ISB for nitrate. Extensive information is also available in the literature on the use of PRBs for the remediation of various groundwater constituents (U.S. EPA 1998; ITRC 1999; U.S. EPA 2002; FRTR 2002).

Important considerations in the application of in situ denitrification are the mobility and mixing capability of water and contaminants in the subsurface, redox conditions, and the maximum depth of the contaminant plume. The key to successful in situ remediation is the exposure of the contaminant plume to the treatment zone; both ISB and PRBs can operate as a barrier through which contaminant migration is blocked as nitrate is destroyed within the plume. When injecting a carbon substrate, the substrate must be available across the plume, for remediation to occur. Unfortunately, subsurface heterogeneity of material properties, such as permeability, render any such injection procedure very inefficient because most of the injectate flows preferentially in relatively localized volumes of the subsurface, thereby bypassing most of the contaminant volume. PRBs have the advantage that the contaminated groundwater moves passively through the PRB for nitrate to be removed. If the PRB can be sufficiently deep and laterally extensive, cleanup can be very effective for the region down gradient of the PRB.

In situ remediation options rely on denitrification in the subsurface to reduce nitrate to other nitrogen species; denitrification requires an electron donor for the reaction to proceed. Nitrate can be reduced through biological denitrification or chemical denitrification. Generally in situ denitrification through the injection of a carbon source (EISBD and ISRM) reduces nitrate through biological denitrification, while PRBs can operate through biological and/or chemical denitrification, depending on the design of the system. PRB remediation and ISB can be combined and sometimes the barrier configuration of ISB is referred to as a PRB. These remediation options are examined separately below in greater detail.

Biological Denitrification

Denitrification occurs naturally in the environment as part of nitrogen cycling, but can be promoted in the subsurface by providing appropriate conditions. Control and monitoring of water quality characteristics, including temperature, pH, salinity, and oxidation reduction potential (ORP), can be fundamental to the stability and efficiency of biological denitrification processes (WA DOH 2005). For biological denitrification, near neutral pH is preferred (7-8) and temperatures below 5°C (41°F) can inhibit denitrification (WA DOH 2005). Biological denitrification uses denitrifying bacteria to reduce nitrate to innocuous nitrogen gas in the absence of oxygen (anoxic conditions). The reduction of nitrate proceeds stepwise in accordance with Eqn. 4.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad \text{(Eqn. 4)}$$

Adminsitrative Record
Page 35232

Denitrifying bacteria require an electron donor (substrate) for the reduction of nitrate to nitrogen gas. Autotrophic bacteria utilize sulfur or hydrogen as an electron donor and inorganic carbon (typically carbon dioxide) as a carbon source for cell growth (Eqns. 5 and 6), while heterotrophic bacteria consume an organic carbon substrate, like methanol, ethanol or acetate (Eqn. 7) (Mateju et al. 1992; Kapoor & Viraraghavan 1997).

$$11S^0+0.5CO_2+10NO_3^-+2.54H_2O+1.71NH_4^+ \rightarrow 0.92C_5H_7O_2N+11SO_4^{2-}+5.4N_2+9.62H^+ \quad \text{(Eqn. 5)}$$

$$H_2 + 0.35\ NO_3^- + 0.35\ H^+ + 0.052\ CO_2 \rightarrow 0.010\ C_5H_7O_2N + 0.17\ N_2 + 1.1\ H_2O \quad \text{(Eqn. 6)}$$

$$1.08\ CH_3OH + NO_3^- + H^+ \rightarrow 0.065\ C_5H_7O_2N + 0.467\ N_2 + 0.76\ CO_2 + 2.44\ H_2O \quad \text{(Eqn. 7)}$$

Eqns. 5 through 7 illustrate the overall denitrification reaction defining the stoichiometric relationship between electron donor, carbon source and nitrate in the production of cells and the conversion of nitrate to nitrogen gas. Not all nitrogen is converted to nitrogen gas. Some nitrogen is required for cell growth. The governing stoichiometric equation indicates the necessary dose and varies with the substrate used. For example, the stoichiometric factor for acetic acid is 0.82 moles of acetic acid per mole of nitrate (Dördelmann et al. 2006).

Various species of bacteria are responsible for denitrification, including *Thiobacillis denitrificans, Micrococcus denitrificans, Pseudomonas maltophilia* and *Pseudomonas putrefaciens* (Kapoor & Viraraghavan 1997). Denitrifiers are naturally present in the subsurface and bioaugmentation is not typically required (i.e., denitrifiers generally do not need to be added). Due to slower bacterial growth rates, autotrophic denitrification offers the advantage of minimizing biomass accumulation; however, autotrophic denitrification requires alkalinity to supply the inorganic carbon source for cell growth (Della Rocca et al. 2006).

Chemical Denitrification

The general mechanism of chemical denitrification involves the transfer of electrons from an electron donating metal to nitrate. As in biological denitrification, nitrate is reduced in accordance with Eqn. 4. However, in contrast with biological denitrification, chemical denitrification often reduces the nitrogen in nitrate to the least oxidized form, ammonium (Eqn. 4a) (Huang et al. 1998; Hao et al. 2005).

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \quad \text{(Eqn. 4)}$$

$$NO_3^- \rightarrow NO_2^- \rightarrow NH_4^+ \quad \text{(Eqn. 4a)}$$

Nitrate is exposed to an electron donating metal by passing the treatment stream through granular media. Particle size, surface area and surface chemistry are important media characteristics related to the efficiency of nitrate removal.

Due to the extensive research focused on the use of zero valent iron (ZVI) in chemical denitrification, ZVI will serve as a preliminary example. There is some variation in the use of ZVI. Forms of application include powdered iron, stabilized iron as nanoparticles, and iron filings. Relevant reactions are listed in Eqns. 8 through 13 (Huang et al. 1998; Hao et al. 2005; Xiong et al. 2009). Nitrate can be reduced to

Adminsitrative Record
Page 35233

nitrite (Eqn. 9), ammonia (Eqn. 10) or nitrogen gas (Eqn. 13) by ZVI. Following nitrate reduction to nitrite, nitrite can then be reduced to ammonia (Eqn. 12). Nitrate can also be reduced by the hydrogen gas that is produced from corrosion reactions (Eqn. 8) to ammonia (Eqn. 11).

$$Fe^{o} + 2H^{+} \rightarrow H_{2(g)} + Fe^{2+} \quad \text{(Eqn. 8)}$$

$$Fe^{o} + NO_3^{-} + 2H^{+} \rightarrow Fe^{2+} + NO_2^{-} + H_2O \quad \text{(Eqn. 9)}$$

$$4Fe^{o} + NO_3^{-} + 10H^{+} \rightarrow NH_4^{+} + 4Fe^{2+} + 3H_2O \quad \text{(Eqn. 10)}$$

$$NO_3^{-} + 4H_2 + 2H^{+} \rightarrow NH_4^{+} + 3H_2O \quad \text{(Eqn. 11)}$$

$$3Fe^{o} + NO_2^{-} + 8H^{+} \rightarrow 3Fe^{2+} + NH_4^{+} + 2H_2O \quad \text{(Eqn. 12)}$$

$$5Fe^{o} + 2NO_3^{-} + 6H_2O \rightarrow N_{2(g)} + 5Fe^{2+} + 12OH^{-} \quad \text{(Eqn. 13)}$$

The reduction of nitrate by iron is characterized by an increase in pH and consumption of hydrogen ions. pH is a significant controlling factor for this treatment method (Hao et al. 2005). The kinetics of nitrate reduction by ZVI have been thoroughly covered in the literature to determine the reaction rate under various conditions. For example, Alowitz & Scherer (2002) examined the nitrate reduction rates of three types of iron; findings indicate that reduction rate increases with decreasing pH.

2.3.1 In Situ Bioremediation/In Situ Redox Manipulation (With Injection of Carbon Source) – Local

In situ bioremediation (ISB) "requires simultaneous evaluation of subsurface hydrogeology, contaminant interactions, and biology/biochemistry. It necessitates the ability to scientifically understand, predict, and monitor the collocation of contaminants, substrates, nutrients, and microbial processes in situ to achieve bioremediation. It is a system designed to establish optimized subsurface conditions, utilizing injected substrates and nutrients to enhance natural biodegradation, the ultimate result of which is accelerated destruction of the target contaminant…" (ITRC 2002, p. 7).

In situ bioremediation/redox manipulation (ISRM) is accomplished by injecting an electron donor into the groundwater plume such that bacteria can utilize the electron donor in the denitrification process, reducing nitrate to nitrogen gas (Figure 2). The addition of injectate enables denitrification to occur much faster than it would naturally (i.e., natural attenuation) (ITRC 2000). ISB requires plume delineation, monitoring wells, and injection wells. "This technology has the potential of remediating sizeable nitrate plumes in groundwater systems" (ITRC 2000, p. iii). Nevertheless, as mentioned above, experience shows that geologic heterogeneity typically results in very poor contact between any injectate and the groundwater contamination due to preferential flow and bypass. Additional information specific to the application of ISRM is available in the literature (see DOE 2000).

Adminsitrative Record
Page 35234

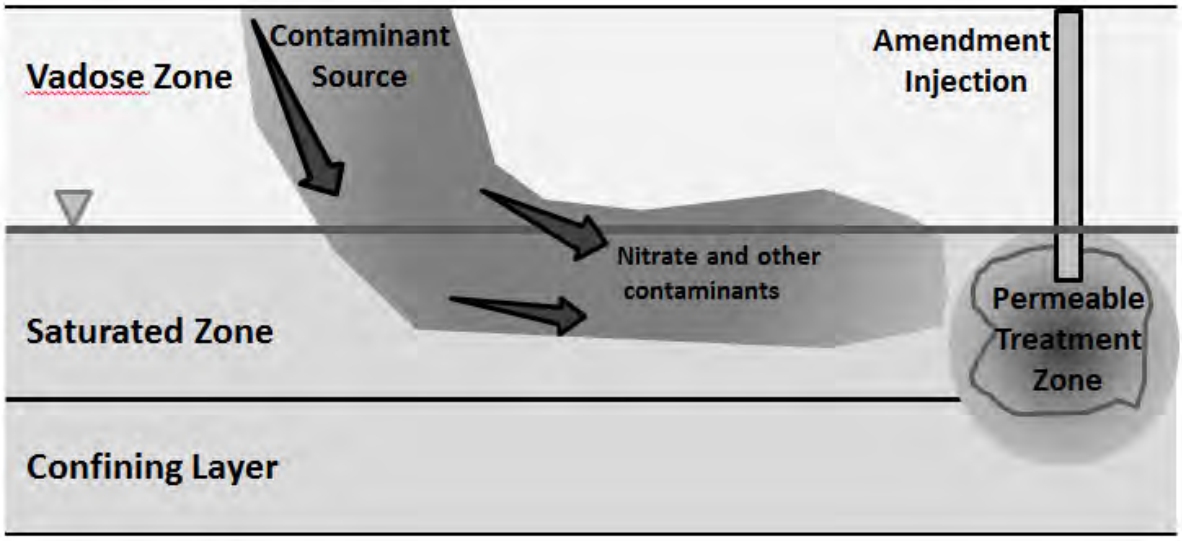

Figure 2. Hypothetical in situ bioremediation scenario. (Source: adapted from DOE 2000.)

For successful implementation and operation of an ISB system, a thorough characterization of the site and ISB system is essential, including the following (ITRC 2002; See also Figure A-1 of the Appendix):

- site history
- hydrologic parameters
- contaminant definition
- geochemical parameters
- potential risks
- analysis of contaminant transformations
- plume delineation and source control
- analysis of subsurface interactions (e.g., ORP, O_2, appropriate carbon source, limiting nutrients)
- regulatory and permitting requirements
- pilot testing
- monitoring

According to the ITRC, advantages of ISB include low-cost, rapid remediation, and the potential for "complete plume remediation," while disadvantages of this remediation option include "impact to geochemistry, regulatory concerns, and biomass buildup" (ITRC 2000, p. 15). While biomass management is fundamental, in part, to avoid uneven distribution of injectate, a problematic concern is the buildup of biomass and the potential for well and aquifer clogging, which can be detrimental to the remediation system. Management of biomass can be accomplished through selection of the optimal carbon source and through various operational practices. Acetate has been shown to limit biomass buildup, pulsed injection can minimize both biomass buildup and oxidizers, while acids and biocides have been utilized to control biofouling (ITRC 2002). Variability in the hydraulic conductivity across the

Adminsitrative Record
Page 35235

plume can also reduce the efficiency of an ISB project; low conductivity areas will have limited to no contact with the injectate, leaving the nitrate in these regions untreated.

ISB can be implemented in a number of configurations to remediate and contain nitrate contamination including general well placement designed for maximal plume remediation, a downstream barrier configuration (Figure 3), a daisy well configuration (Kahn & Spalding 2003), and in parallel with PRBs (discussed separately). Using injection wells, the remediation depth is limited primarily by the permeability of the subsurface and the depth of injection wells. The electron donor (carbon source) can be delivered through reinjection by mixing extracted nitrate contaminated water or by alternating injections of amendment and nitrate laden water (pulsed injections); the latter limits the risk of biomass accumulation in the vicinity of the injection well.

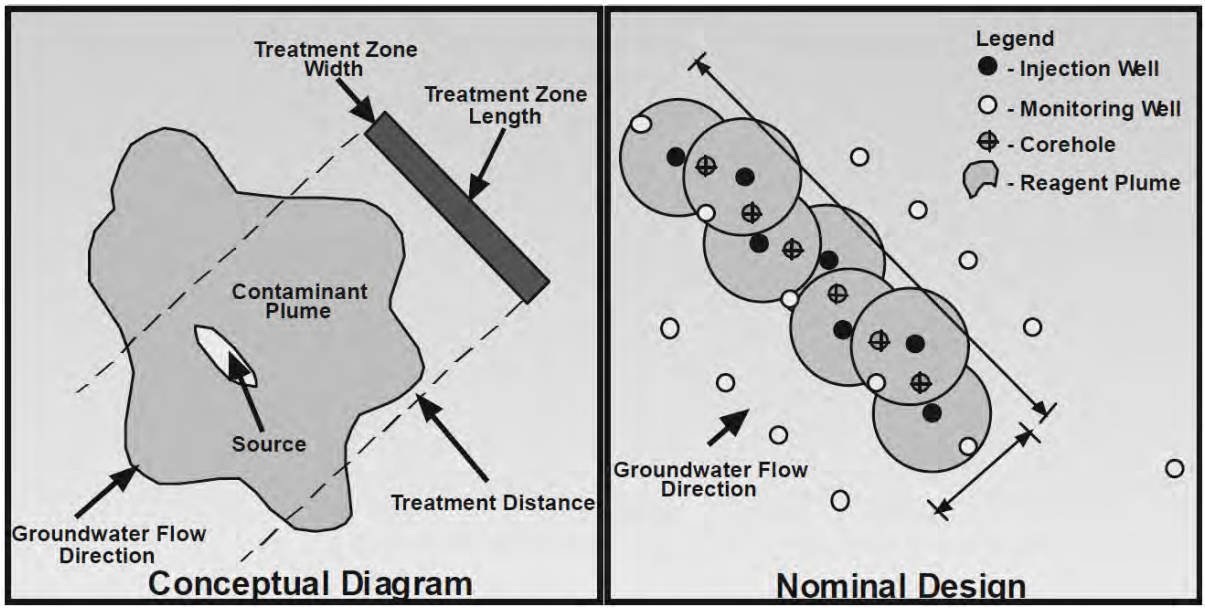

Figure 3. Barrier configuration of in situ biological denitrification. (Source: DOE 2000.)

Application of In Situ Bioremediation to Address Nitrate Impacted Groundwater

Case Study: Nebraska (Khan & Spalding, ITRC)

In Nebraska, research by the University of Nebraska-Lincoln (UNL) used a daisy well configuration to promote biological denitrification of groundwater surrounding an inactive municipal well, with acetate as the substrate (Khan & Spalding 2004). Reduction wells (15 cm diameter) were placed 18 m from the centrally located municipal well, in a circular configuration. Oxidation wells (5 cm diameter) were placed similarly, but at a distance of 9 m from the central municipal well. Oxidation wells were included for the injection of an oxidizer to decrease residual carbon and oxidize any nitrite to nitrate. Site characteristics include a shallow, unconfined aquifer of 22 m thickness. The heterogeneous aquifer is composed of predominantly sand and gravel. Modeling software (Modflow and Modpath) facilitated system design. The extraction flow rate of the system was 12.6 L/s (~0.3 mgd or ~200 gpm) resulting in an appropriate residence time of the carbon source for denitrification to occur (~ two days). The maximum screened depth of the wells was ~25 m.

Adminsitrative Record
Page 35236

The acetate dosage (pulse length) was varied and alternated with injection of extracted nitrate contaminated water. No clogging problems were detected over a three month period. Nitrate levels decreased by ~45%, from an initial nitrate concentration of 55.7 mg/L as nitrate (12.6 mg/L as N) to a final nitrate concentration of 29.3 mg/L as nitrate (6.6 mg/L as N). While water quality improved to meet regulations with respect to nitrate, "the total plate count [for coliform] exceeded the maximum permissible limit (500 cfu/mL)" (Khan & Spalding 2004, p. 3382). For long-term operation, the injection line, accessories, and injection wells should be cleaned regularly (with a hydrogen peroxide solution). Hydrogen peroxide can also be injected in the oxidizing wells to decrease dissolved organic carbon (DOC) levels in extracted water; however, this was unnecessary as extracted DOC levels in this project were close to background levels.

Reported costs associated with the remediation system are (Khan & Spalding 2004):

- Capital and Installation $75,000 (2004 dollars)
- Chemicals
  - Acetate $0.05/1000 L (0.19/1000 gal)
  - Cleaning $0.06/1000 L (0.23/1000 gal)
- O&M $0.16/1000 L (0.61/1000 gal)

Other UNL remediation projects include similar systems that are successfully addressing nitrate concentrations as high as 177 mg/L as nitrate (40 mg/L as N) (ITRC 2000). The daisy well configuration surrounding a municipal well is a specific scenario that would likely be operated continuously for the protection of a drinking water source, rather than an option to specifically remediate a plume.

Case Study: Mineral Processing Facility (Garret & Hudson)

Large scale in situ bioremediation was used at a 28 ha (70 acre), shut down mineral processing facility to address nitrate levels ranging from ~45 mg/L as nitrate (10 mg/L as N) to more than 10,000 mg/L as nitrate (2,258 mg/L as N), with the highest concentrations found beneath an evaporation pond (Garrett & Hudson 2005). Over the three-year study period, an average of 41% reduction in nitrate levels was achieved across the 19 ha (48 acre) nitrate plume using methanol as the amendment for denitrification. Initially, a total of 24 injection wells and 14 monitoring wells were used; injection wells were placed both upstream and downstream of the contaminant source. Thirty-five more injection wells and 18 more monitoring wells were placed within the pond area, while 22 injection wells and seven monitoring wells were placed in another contaminated area on site. In addition to methanol, a nutrient solution was injected. In regions treated for a longer duration (>2 years) of the project, nitrate levels were decreased by 69%. Costs of the remediation system were not discussed; however, the system was designed to be as simple as possible, in part to minimize capital and O&M costs.

Case Study: New Mexico (Nuttall & Dutta, Dutta et al., Mohr, Faris)

Lastly, a research project by the University of New Mexico (Nuttall & Dutta N.D.) implemented in situ bioremediation using a bio-barrier (aka. bio-curtain) design. The technology consists of a radially arranged set of injection wells that maintain appropriate electron donor level for denitrification in a ring

Adminsitrative Record
Page 35237

about the extraction well. The site, located in the South Valley of Albuquerque, NM, spanned 223 ha (550 acres) with a total plume volume of more than 6 billion L (1.6 billion gal) and nitrate levels approaching ~1,330 mg/L as nitrate (300 mg/L as N). The objective of the pilot scale study was to assess the feasibility of both plume remediation and containment with injection of molasses and nutrients. The pilot system was tested for more than a year with nitrate levels reduced to < 4.4 mg/L as nitrate (< 1mg/L as N) (Dutta et al. 2005). After four months, biomass buildup resulted in clogging and interrupted system operation; a bleach solution was subsequently used for biomass management. The use of an inexpensive amendment as electron donor and the ability to recharge the curtain repeatedly allow for a potentially cost-effective remediation system with minimized operation and maintenance demands (Dutta et al. 2005). According to researchers,

"The ability to direct groundwater flow using a biofilm barrier could be used to channel contaminated groundwater to an active treatment zone while also contributing to bioremediation of the water. In situations where groundwater flow is minimal, pumping strategies to draw the contaminated groundwater into an active treatment zone could be enhanced with biofilm barrier technology. This technology has commercial value for assisting agricultural businesses, such as feedlots, hog farms, and fertilizer suppliers, in reducing their environmental impact and ensuring the availability of safe drinking water" (Nuttall & Dutta N.D., p. 205).

Following subsequent research and consideration of various remediation options,[13] the New Mexico Office of Natural Resources Trustee (ONRT) opted to utilize ISB to address the nitrate plume in Mountain View, NM (Mohr 2009). Although the bio-curtain falls into the category of ISB, the terminology is not particularly clear for this unique system as it is also called a PRB. For reference, the following is excerpted from a published article (Mohr 2009, p. 417) discussing remediation at this location:

"It is anticipated that if a biodenitrification barrier could be constructed within and down the gradient of the Mountain View contamination site, natural groundwater gradient flow through the barrier would stimulate denitrification (Faris, 2007a). Barriers to successful in situ biodenitrification include the proper placement of food, need for additional nutrient injections, and the potential for clogging or biofouling (Faris, 2007a). Typical biobarriers require injection of the food every 10 feet; given the size of the Mountain View nitrate plume, 460 injection points would be required to build one biobarrier (Faris, 2007b). The estimated cost of building the biobarrier to remediate the Mountain View nitrate plume is approximately $1.5 million, with another $500,000 for soil and groundwater testing and assessment (Faris, 2007a)."

The New Mexico site is further discussed below in Section 2.3.2 on Permeable Reactive Barriers (PRBs).

[13] The following remediation options were considered for remediation of the nitrate plume in Mountain View, NM,: "taking no action, pumping the nitrate-contaminated water for agricultural or industrial use, pumping the contaminated water and treating it for reinsertion into the aquifer or for other beneficial use (ex situ biodenitrification), or treating the nitrate in place through manipulation of natural biodentrification processes..." (Mohr 2009, p. 415).

Adminsitrative Record
Page 35238

Estimation of Costs of ISB/ISRM

In addition to that reported in the above research, cost information for ISB/ISRM systems is available in the literature through various agencies (ITRC 2000; DOE 2000; U.S. EPA 2001a; ITRC 2002; ITRC 2008; www.frtr.gov); however, it is important to note that the costs for remediation projects are site specific, varying with location, site and water quality characteristics. Capital and O&M costs for ISB/ISRM are highly dependent on depth to the plume, as related to drilling, injection, and pumping costs.

Cost components of Enhanced In Situ Biological Denitrification (EISBD) consist of the following categories (ITRC 2000, p. 39): "Chemical Amendments, engineered Amendment Injection Systems, well Construction, system Maintenance, and Monitoring."

The majority of available cost information is for the use of ISB to address contaminants other than nitrate, thus, the costs associated with nitrate remediation of the Mountain View, NM, nitrate plume are likely the best example of nitrate plume remediation. General cost information for ISRM is available in the literature (see DOE 2000).

Application of ISB to the TLB and SV

For the purposes of this analysis, a typical plume is defined as discussed above in the Section *Hypothetical Plume Size*. The total plume volume of the typical plume is greater than that discussed above for the New Mexico site (due to greater depth); the additional depth makes it difficult to accurately extend the published cost estimates of the NM site to our typical plume. While the feasible depth of ISB for remediation is theoretically dependent on the depth of injection wells, remediation to greater depths will be more costly and less reliable, due, in part, to inconsistencies in subsurface geology. It is expected that the cost of ISB remediation for a typical plume in the study area would exceed that of the NM site (>$2 million), whereas for a more shallow plume, this may not be the case.

It is important to note that the costs presented here are just one example of nitrate plume remediation costs using ISB; there are numerous configurations, amendments, and site specific variables that affect remediation costs. Obstacles associated with the application of ISB in the TLB and SV might include:

- Plume depth, hydrologic and geologic factors (lack of strong confining layer at shallow depth)
- Depth to groundwater, often exceeding 50 feet to 100 feet
- Limited access for site characterization, injection, extraction and monitoring wells, depending on location (e.g., the middle of an operating dairy or an actively farmed field)
- Cooperation of stakeholders and public perspectives/education
- Project funding sources (especially for legacy contamination and the lack of a clearly definable responsible party)
- Regulatory and permitting requirements

However, the extensive research and experience thus far with ISB for nitrate impacted groundwater (in Mountain View, New Mexico) offers a precedent for the potential application of this technology as a plume-scale remediation option.

Adminsitrative Record
Page 35239

2.3.2 In situ Denitrification Using Permeable Reactive Barrier – Local

PRBs can be used to remove nitrate from groundwater in situ through biological denitrification or chemical denitrification (see above for more information). Barriers containing reactive media (e.g., ZVI, solid phase organic carbon, oil coated sand) can be installed in the path of groundwater flow (Figure 4), supplying the necessary components for denitrification.

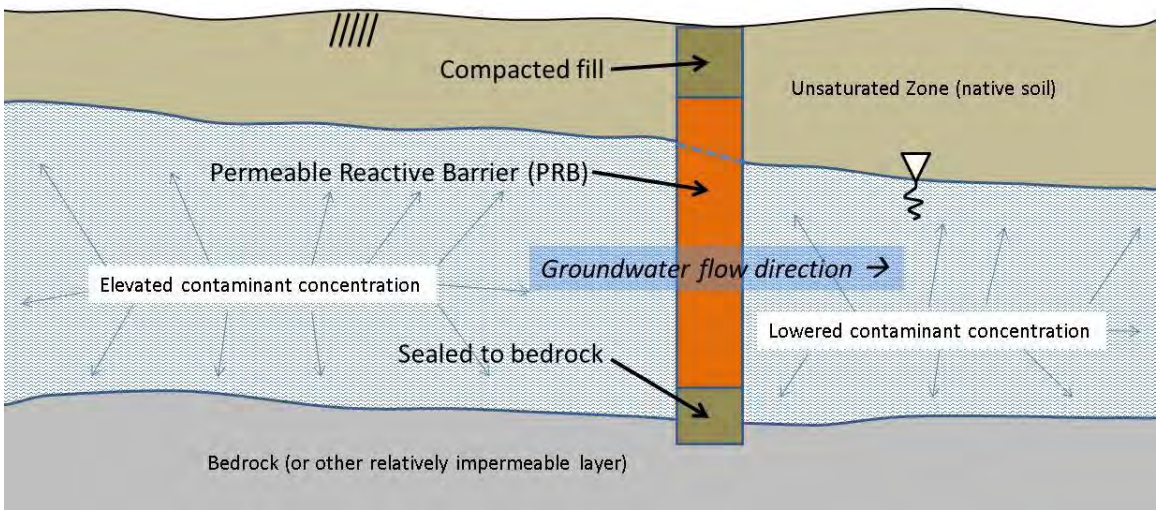

Figure 4. General schematic of a permeable reactive barrier.

The denitrification zone of PRBs can also be augmented by injection of amendments to provide optimal conditions. PRBs can be implemented in several configurations including cross-flow continuous barriers to treat diffuse contaminant plumes, funnel and gate installations that channel contaminated flow through a narrow reactive barrier, and reactive vessel designs for containment of point-source plumes before they have spread (Figure 5). According to the U.S. EPA (2001c, p. 2):

- "PRBs work best at sites with loose, sandy soil and a steady flow of groundwater.
- The pollution should be no deeper than 50 feet.
- Since there is no need to pump polluted groundwater above ground, PRBs can be cheaper and faster than other methods.
- There are no parts to break, and there is no equipment above ground so the property can be used while it is being cleaned up.
- There are no energy costs to operate a PRB because it works with the natural flow of groundwater."

PRB remediation systems can require significantly less maintenance than alternative remediation options; however, plume depth will be a significant determining factor affecting the feasibility of application. Trenching is a significant portion of the costs associated with the implementation of PRBs

Adminsitrative Record
Page 35240

and the deeper the required barrier, the more costly the project. For depths greater than 30 feet, specialized equipment may be necessary; PRBs can be installed as deep as 120 feet or more, but costs will increase with depth (Gavaskar et al. 2000; NFESC 2002 as cited in Perry 2008). Based on U.S. EPA recommendations (2001c) PRBs are generally more appropriate for contamination less than 15 m (50 ft) deep.

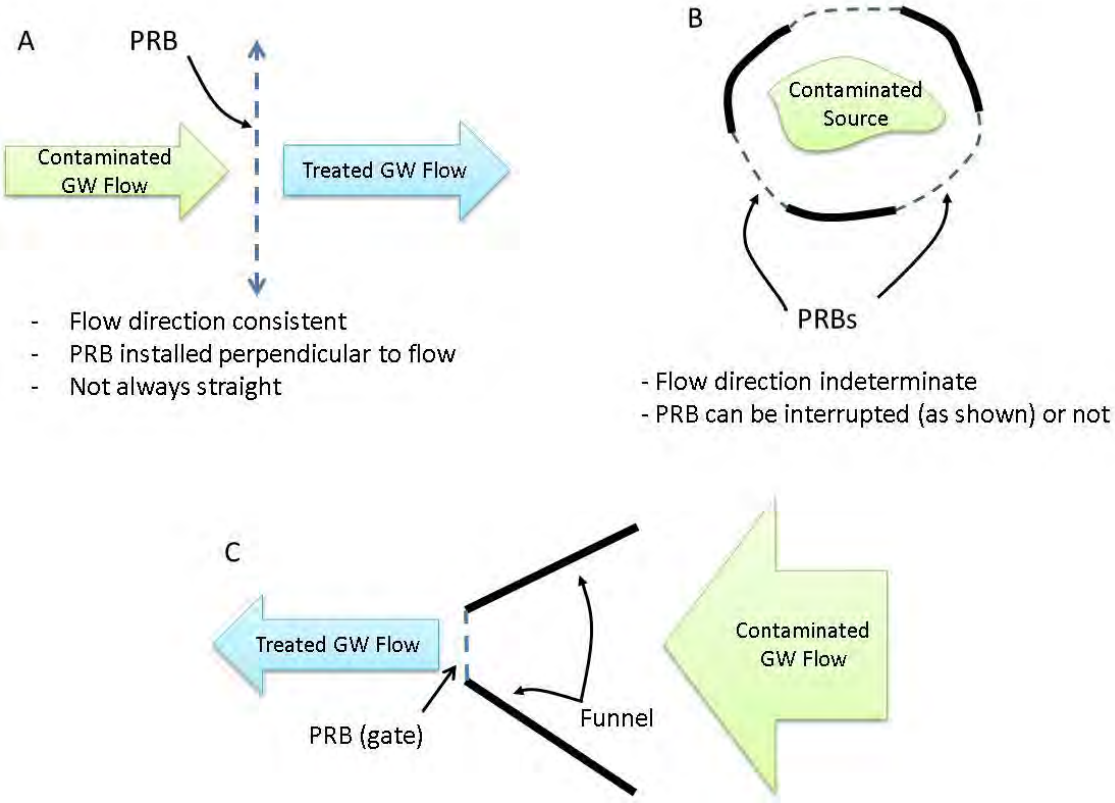

Figure 5. Permeable reactive barrier configurations: A) continuous, B) reactive vessel, C) funnel and gate.

One area of research focuses on the selection of the most appropriate amendment for biological denitrification using PRBs. For example, Hunter (2001) examined the use of vegetable oil as an electron donor in biological denitrification. The use of an insoluble substrate minimized biomass blockage, a problem common with the use of soluble substrates like ethanol, methanol, and acetate. The barrier was composed of soybean oil-coated sand and effectively decreased the nitrate levels from a starting concentration of ~89 mg/L as nitrate (20 mg/L as N) to below the MCL for a period of 15 weeks, with a flow rate 1100 L/week. After 15 weeks, insufficient oil remained for denitrification. High chemical oxygen demand, TSS, and turbidity in the effluent of the reactor indicate a longer sand bed was needed; however, the author suggests that in situ application of this type of biological reactor would decrease these factors naturally. With a withdrawal point far enough from the barrier, subsequent drinking water treatment requirements would be limited to disinfection. The most significant problem encountered in this study was the exhaustion of substrate; an effective means of substrate addition must be found (injection for example), but this was not explored. The estimated life of the PRB was 2.5-12.5 years

Adminsitrative Record
Page 35241

depending on several key factors including flow, nitrate concentration and dissolved oxygen concentration.

2.3.2.1 Application of Permeable Reactive Barriers (PRBs) to Address Nitrate Impacted Groundwater

Numerous examples of the application of PRBs for treatment of groundwater impacted by nitrate and other contaminants are available in the literature (U.S. EPA 1998; ITRC 2000; Blowes et al. 2000; U.S. EPA 2002; DOD 2002; FRTR 2002). PRBs can be implemented as a stand-alone denitrification barrier or can be augmented by the injection of amendments.

Case Study: Tennessee (FRTR)
An example of chemical denitrification, an iron-based PRB was installed in Tennessee for the remediation of nitrate and uranium; initial nitrate concentrations ranged from 20 to 150 mg/L (assumed to be nitrate as NO_3^-) (FRTR 2002; FRTR 2011). A total of 503,000 liters (133,000 gallons) of groundwater were treated using a PRB 67 m (220 ft) long and 7.6 m (25 ft) deep in a funnel and gate configuration. An iron and peat mixture was used to address nitrate; levels were reduced by 75%. Installation costs for this demonstration project were $943,000. Operation and maintenance costs were not provided.

Case Study: Canada (Robertson et al.)
Robertson et al. (2000) discuss several examples of the use of biological denitrifying PRBs to address nitrate contamination from septic tanks, and one example that addresses the runoff/drainage from an agricultural field. At these sites the barrier was composed of 15% – 100% cellulose and nitrate concentrations were reduced between 58% and 91%, from as high as 252 mg/L as nitrate (57 mg/L as N) (Robertson et al. 2000). These PRBs successfully removed nitrate from groundwater through passive treatment, with results indicating a lifespan of ten years or more before necessary restocking of the PRB.

Case Study: New Mexico Revisited (Faris, ONRT, Intera)
Current plans for remediation of the Mountain View, NM, nitrate plume discussed above (see the case study listed in Section 2.3.1 of this report), include the use of several PRBs for the most concentrated portions of the plume (Faris 2011; ONRT 2011). The remediation plans at this site consist of multiple PRBs using an ISB (in situ bioremediation) bio-curtain configuration. The remediation costs are budgeted for $4 million and are planned to "remove 450,000 pounds of nitrate from the groundwater plume hot-spots allowing the remainder of the plume to naturally attenuate to below State groundwater standards in less than 20 years" (ONRT 2011). An extensive remedial investigation of the Mountain View site began in mid-2009 resulting in the compilation of a detailed report, for the New Mexico Environment Department (NMED) and New Mexico Office of Natural Resources Trustee (ONRT), including monitoring data, plume delineation, characterization of the subsurface and hydrogeology, and an assessment of the feasibility of remediation (Intera 2010). The breadth of this report highlights an important part of any remediation project; the costs of preliminary planning, site characterization, monitoring, and feasibility studies must be considered, as they can be significant. The Mountain View, NM, nitrate plume has a long history, with numerous projects investigating and cataloging the site; the costs to finalize site characterization and feasibility (for the development of the Intera report) were

Adminsitrative Record
Page 35242

$250,000 (Faris 2011). This figure excludes pre-existing monitoring wells and additional past investigative efforts.

Additional unique variations of the application of PRBs might also be considered including circular barriers around wells and installation in drainage channels.

2.3.2.2 Costs of PRBs Reported in the Literature

At the Mountain View, NM, site, the latest total cost estimate budgets the project around $4 million for multiple PRBs to address hot-spots, removing 450,000 pounds of nitrate over a period of 4 years. Additional details of the remediation plans for this site are currently in development.

As indicated above, PRB remediation costs are largely dependent on the required depth of the barrier. As reported by Gavaskar et al. (2000), estimated costs for trenching to depths of 30 feet range from $2 – $10 per sq. ft., while excavation deeper than 80 feet can range from $2 – $55+ per sq. ft. PRB installation costs have been estimated to range from $50 – $300 per vertical foot using Caisson-based construction[14] (Gavaskar et al. 2000).

The following example capital costs are summarized for a PRB remediation system of a Chlorinated Volatile Organic Compounds (CVOC) plume at Dover Air Force Base (AFB).[15] Although the Dover AFB site is for CVOCs rather than nitrate, comprehensive costs are included here as a representative example of PRB costs for a PRB system of the same scale as the Dover AFB site. In this example, the PRB depth, width, and thickness for the funnel and gate project were 39 ft, 68 ft, and 4 ft, respectively, which captures 50 ft of the plume across 25 vertical ft (Gavaskar et al. 2000). The estimate includes preconstruction costs as well as the costs of materials and construction (Appendix B of Gavaskar et al. 2000). Preconstruction costs include "site assessment, site characterization, laboratory testing, PRB modeling and design, procurement of materials and construction contractors, and regulatory overview…and can constitute as much as 50% of the total capital investment in the PRB" (Gavaskar et al. 2000, p. 128). O&M costs refer to any ongoing costs over the life of the project. Estimates provided by Gavaskar et al. (2000) for a full-scale PRB at Dover AFB (operating at 10 gpm which would equate to ~5.3 million gal/year) are:

[14] Caissons are steel temporary retaining walls installed progressively during excavation to maintain integrity of the walls, and removed after fill has been placed.

[15] The exact dimensions of the plume were not reported; however, the operating capacity of the PRB system, estimated to be 10 gpm, would equate to ~5.3 million gal/year. Please refer to Gavaskar et al. (2000) for additional information.

Adminsitrative Record
Page 35243

- Capital Costs
    - Preconstruction: $365,000
    - PRB Construction (including materials): $587,000
    - Total Capital Cost: $947,000
- O&M Costs
    - Annual Operating Costs: $148,000
    - Additional Long-term Maintenance (every ten years): $421,000
    - Total Annual O&M Cost: $190,100

The most significant elements of the annual operating costs listed above include quarterly groundwater sampling of 40 wells ($80,000), quarterly CVOC analysis ($20,000) and data analysis, reporting and regulatory review ($40,000); these O&M costs highlight the importance of accounting for sampling, chemical testing, and data analysis. O&M costs can vary widely with not only the scale of contamination, but also monitoring and reporting requirements.

Capital costs for PRBs can be greater than those of PAT remediation, but the long-term ongoing operations and maintenance cost savings can make PRBs the more financially prudent option. Estimated costs for a PAT system comparable to the above PRB system at Dover AFB (capable of the same level remediation) indicate a break-even point of the PRB for this site after 8 years of operation assuming a 30 year project duration and a 20 year media life.

In McGregor, TX, PRB trenches were constructed over one mile long and 25 ft deep for the remediation of perchlorate contaminated groundwater; the initial perchlorate concentration was reported as 27,000 ppb (ug/L)(DOD 2002). Less than one month after start-up, perchlorate levels were reduced by a minimum of 90% (DOD 2002). Installation of the PRB totaled $833,000. "Capital cost avoidance has been estimated at more than $3 million compared to *ex situ* technologies. In addition, operation and maintenance costs are estimated at $5,000 per year versus $100,000 per year for the *ex situ* technologies" (DOD 2002, p. 2). In addition to treating for perchlorate rather than CVOCs, the significantly lower O&M costs for the McGregor PRB remediation, in comparison with the Dover PRB remediation, is assumed to be due to differences in site characteristics, sampling regime, and reporting requirements.

The use of PRBs for remediation at the McGregor, TX, and Dover, DE, sites are provided only as examples of the full-scale application of PRBs; it is important to keep in mind that these PRBs were used to address contaminants other than nitrate and costs are not only variable across sites, but also across contaminants. Additional costs of PRB remediation are listed in the literature on a case study basis (Gavaskar et al. 2000; U.S. EPA 2001d; U.S. EPA 2003).

2.3.2.3 Application of PRBs Within the Study Areas

Costs associated with the use of PRBs for remediation in the Tulare Lake Basin and Salins Valley are expected to be similar to those described above, depending on plume size and site considerations.

Adminsitrative Record
Page 35244

However, it is important to note that the costs presented here are just examples of plume remediation costs using PRBs; there are numerous configurations, amendments, and site specific variables that would affect remediation costs.

The application of PRBs for the remediation of nitrate contaminated groundwater in the Tulare Lake Basin and Salinas Valley would be limited to areas with shallow contamination which accounts for only a small portion (about 26%) of the area of interest. From Figure 6, it is apparent that the northeast region of the Tulare Lake Basin study area is the most likely part of the study areas to be appropriate for PRB treatment for two reasons: first, the shallow groundwater in that region makes PRB treatment possible, and second, the high concentrations of nitrate in that region affords better return on the cost to implement than installations in areas with lower concentrations of nitrate. Because the PRB systems are capable of treating water at high nitrate concentration as easily as at low concentration, the best benefit to cost is found in the former.

Adminsitrative Record
Page 35245

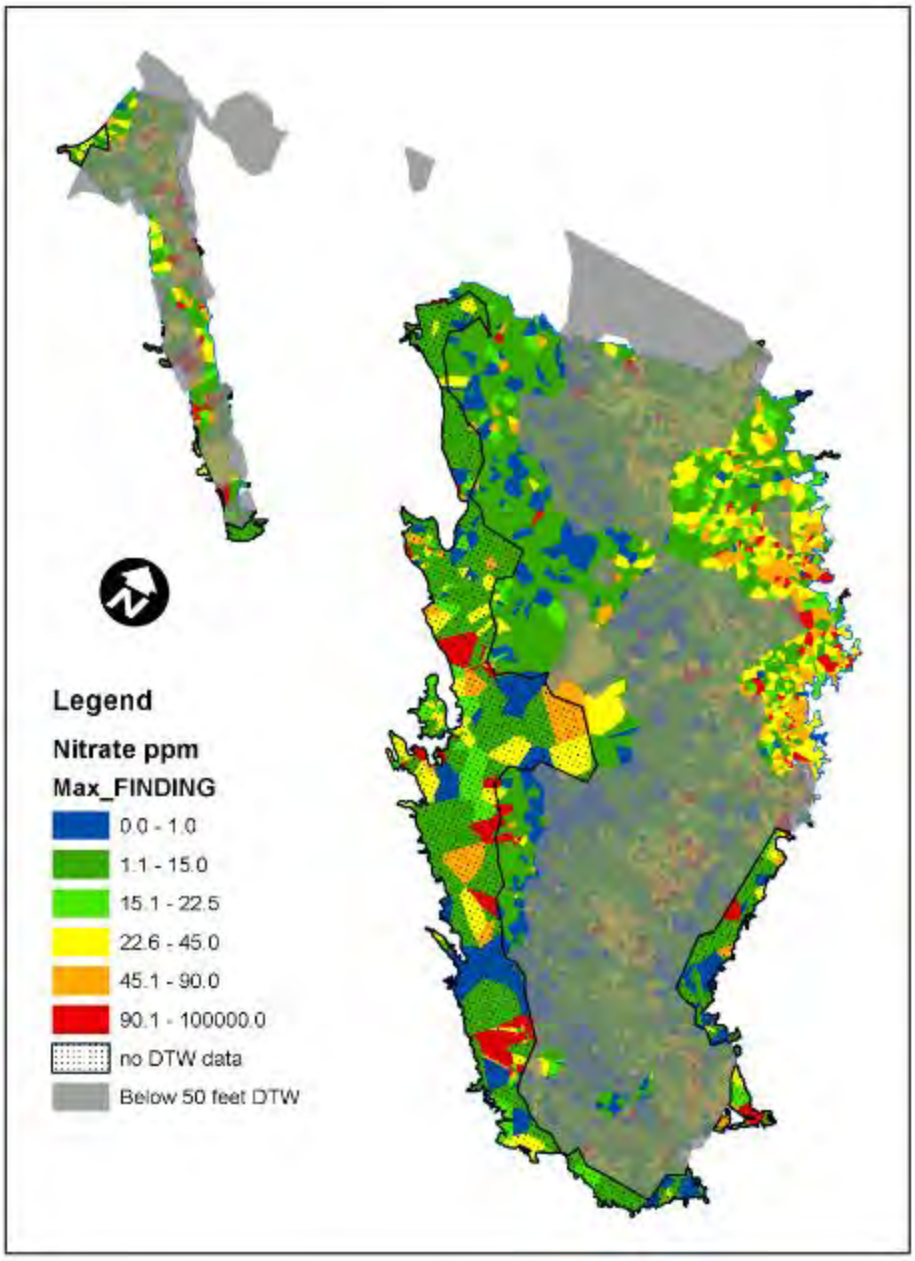

Figure 6. Potential application of PRBs in the Tulare Lake Basin (right) and Salinas Valley (left).

2.3.2.4 Biological Denitrification with Wood Chip Substrate

As a plausible alternative to chemical substrates, wood chip PRBs (WCPRBs) are examined as an example of PRB application in the Tulare Lake Basin. WCPRBs can be implemented as vertical denitrification walls to intercept flow of high-nitrate, shallow groundwater moving in predictable flow paths.

The eastern slopes of the northern Tulare Lake Basin are heavily planted with citrus crops that require large amounts of nitrogen fertilizer. Although a definite link has not been established between the applied fertilizer and the groundwater nitrate levels, wells in these areas are prone to elevated nitrate

Adminsitrative Record
Page 35246

levels. In the same area, the groundwater levels are frequently less than 50 feet below ground surface. The groundwater gradient in this area is to the southwest, toward one of the more heavily populated parts of the study area. This eastern portion of the Tulare Lake Basin represents the most favorable conditions for in situ treatment with WCPRBs in the study area. A brief discussion of the effort and estimated cost of employing this remediation option follows.

In the Tulare Lake Basin, typical infiltration rates of 1 foot per year, and typical down-gradient movement of 50 to 100 feet per year imply that a WCPRB placed 50 feet deep (starting from the groundwater surface) could intercept groundwater flow from nearly a mile up-gradient. Assuming a required residence time of 10 days and groundwater flow velocity of 100 feet per year, the denitrification wall would need to be 2.74 feet thick. Assuming a denitrification wall of 50 foot depth, starting at 30 feet below the ground surface, and running for a half mile to intercept the infiltrate from a half-section field (Figure 7), a WCPRB would require about 21,500 cubic yards of excavation, and about 13,400 cubic yards of wood chips. Assuming a low-estimate cost for excavation of $20 per cubic yard, $10 per cubic yard for backfilling, wood chips delivered at $20 per cubic yard, the cost to install such a WCPRB would be roughly $779,000, not including the costs of site inspection, plume delineation, permitting, and monitoring.

Figure 7. WCPRB for treatment of infiltrate from a single half-section field or orchard, assuming 100 feet per year lateral groundwater travel, 1 foot per year of infiltration, and 10-day residence time in the WCPRB. Over 13,400 cubic yards of wood chips would be required.

As a sub-regional approach to remediation, WCPRBs could be installed for miles roughly parallel to groundwater depth contours (Figure 8). Interception and treatment of nitrate contaminated groundwater from a strip of impacted agricultural land approximately one mile wide, with WCPRBs following depth to water contours from 20 miles north of Visalia to 20 miles south east of Visalia, would require placement of about 40 miles of WCPRB at a cost on the order of $62.5 million. Considering that the area of impacted groundwater along the eastern edge of the basin varies from 2.5 to 5 miles in width, multiple WCPRBs would need to be placed in sequence perpendicular to the groundwater flow gradient, totaling on the order of 120 miles of WCPRB. Implementation of this WCPRB example is estimated to cost roughly $180 million, and require over 1.6 million cubic yards of wood chips (with the same assumptions as the ½ section example above).

Adminsitrative Record
Page 35247

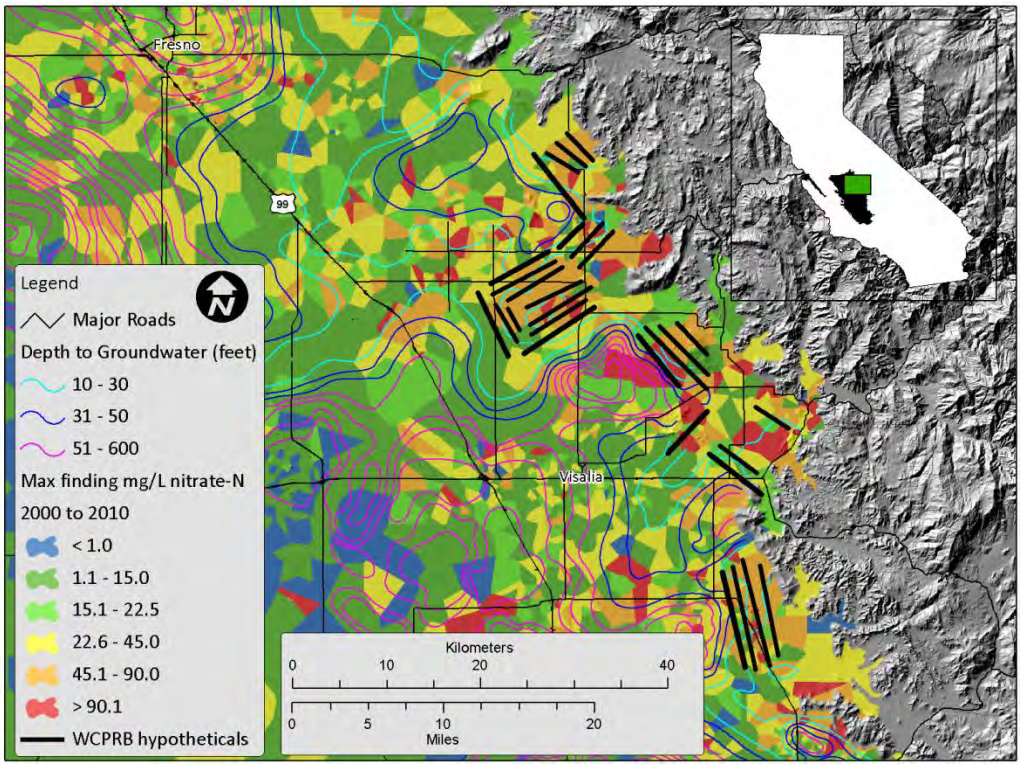

Figure 8. Hypothetical placement of 120 miles of WCPRBs (thick black lines) to intercept and treat infiltrate from a high nitrate-loading area with shallow groundwater.

At these scales, the supply of wood chips for bioreactors will likely become limiting. However, the hypothetical ½-mile application described above would consume the equivalent of about 15% of the yearly delivered volume of a single typical medium-sized wood chip supplier in the central valley (based on interviews with regional woodchip delivery companies). Therefore, it is feasible that the 120 miles of WCPRB could be installed over a period of several years.

2.3.2.4 Wood Chip Bioreactor (WCBR) Treatment of Tile-Drain Effluent

Another targeted application of wood chip biological denitrification technology incorporates the use of tile drains as collectors of infiltrate, with treatment of the collected irrigation infiltrate in a bioreactor. Robertson et al. (2000) demonstrate this application. In a similar approach, a 9-year field-scale experiment by Moorman et al. (2010) used a pair of shallow WCPRBs placed on either side of a tile drain to treat agricultural drain water prior to entry into the tile drain. This treatment resulted in a reduction from 97 mg/L to 39 mg/L nitrate as NO_3^- (22 mg/L to 8.8 mg/L nitrate as N).

Such smaller scale, on-farm technologies could be used to target high-nitrate crops before the contamination reaches the groundwater, and should reduce costs when compared with treatment options restricted to treating diluted or deep water. Similar to the WCPRB application above, this technology is applicable only in limited areas – though not the same areas. Tile drains are only used in a small portion of the agricultural areas in California, specifically, those areas with very shallow, perched groundwater (on the order of 1 to 3 meters depth to water). Tile drains are used in the lower Salinas

Adminsitrative Record
Page 35248

Valley and the north-western Tulare Basin (southern San Joaquin). Data on the distribution of tile-drains is not available in the public record, so a survey of agricultural practices, combined with investigation of tile-drain effluent nitrate concentrations, would be required before an evaluation of this application could be accomplished. No extensive on-farm testing of WCBR treatment of tile-drain effluent has been conducted to date in California. Based on the rate constants in Leverenz (2010), a typical installation designed to treat a 100 acre field would cost between $20k and $35k and would require approximately 1 acre of land.

Regional Water Quality Control Boards have issued waste discharge permits for in situ treatment of leaking sub surface tanks (petroleum) using injection of oxidizing agents, and PRBs of zero-valent iron for treatment of groundwater plumes contaminated with heavy metals. WCPRBs would require similar permits.

2.3.2.5 PRB Conclusions

Permeable reactive barriers are useful as in situ treatment for nitrate, however, in the Tulare Lake Basin and Salinas Valley, feasibility of PRBs is limited by depth to groundwater (a quarter of the study area has shallow enough groundwater). PRBs are most cost effective when used to treat relatively high concentrations, further restricting their regional value. As with any in situ treatment regime, drinking water supplies may need to be protected by other means until remediated groundwater reaches drinking water wells. Woodchips are a viable alternative to more expensive materials (e.g., iron powder) in the Tulare lake Basin, where chipped orchard trees supply ample material for large scale implementation of WCPRBs.

Obstacles associated with the application of PRBs in the TLB and SV include:

- Plume depth, hydrologic, and geologic factors
- The limited area within the SV and TLB with a shallow enough depth to groundwater allowing for PRBs. Of the 22,660 square kilometers of the study area, only 26% (6,000 square kilometers) has groundwater at 50 feet depth or less, the recommended (U.S. EPA 2001c) maximum depth to groundwater for application of PRBs.
- The time lag for remediated water to reach drinking water wells
- Limited access for site characterization, PRB installation, and monitoring wells (depending on location, e.g., the middle of an operating dairy or actively farmed field)
- Cooperation of stakeholders and public perspectives/education
- Project funding sources (especially for legacy contamination and the lack of a clearly definable responsible party)
- Regulatory and permitting requirements

Adminsitrative Record
Page 35249

2.4 Pump and Fertilize (PAF) at the Basin Scale

Full, basin-scale application of pump-and-treat (PAT) methods is not practical, due to the prohibitively high costs associated with the required construction and operation of a vast network of contaminant capture wells for decades, possibly centuries. Moreover, vast amounts of groundwater would have to be treated and reinjected. The construction and energy costs alone would be enormous. Pump-and-fertilize (PAF) refers to accounting for any nitrate already present in irrigation water when determining fertilizer needs. The PAF option could be done at a small fraction of the cost of pump-and-treat (PAT) but requires changes in land, fertilizer, and irrigation management. Many of these changes are technically feasible. Some farmers in the study area already employ improved farm management practices that include pump and fertilize.[16]

2.4.1 Overview of Pump and Fertilize

Traditional PAT involves construction of contaminant recovery wells to capture the contamination. In agricultural groundwater basins such as the TLB and SV, however, there already exist thousands of irrigation wells that provide most of the groundwater used in the region. These existing irrigation wells capture a significant fraction of the recharge, including any high nitrate water stemming from crop irrigation and fertilizer application. PAF would use the existing wells and the existing pumping schedule of those wells to capture nitrate contaminated groundwater. PAF could potentially include the drilling of new irrigation wells specifically to capture high nitrate, shallower groundwater. Furthermore, since this water is used to grow crops, PAF would use crops to remove nitrate-N from irrigation water by applying less commercial fertilizer, commensurate with the amount of N already in the pumped irrigation water. PAF is, in effect, a regional phytoremediation approach that makes use of existing irrigation wells and nitrogen uptake by crop production. PAF is therefore an intrinsic element of proper nutrient management (see Technical Report 3, Dzurella et al., 2012). It has to be operated as part of a farm's nutrient management efforts. Unless additional irrigation wells are installed specifically to capture high nitrate groundwater for irrrigation, PAF may not necessarily be called out as an active remediation scheme, but may simply be considered part of a farm's nutrient management.

The PAF approach requires more careful management of both, nitrate-N in irrigation water and the nitrogen applied as fertilizer. The amount of N applied to each field through irrigation water needs to be measured, and commercial (or organic) fertilizer applications must be adjusted downward to account for nitrogen applied with the irrigation water. Because nitrate concentrations in pumped groundwater can vary considerably in space and time, frequent monitoring and adaptive nutrient management is required.

Some long-term remediation at the basin-scale is possible by using nitrate contaminated groundwater to grow crops and by reducing the nitrate concentration of water percolating below the crop root zone. To ascertain how much reduction in nitrate loading this method could accomplish, pilot field projects

[16] See Technical Report 3 (Dzurella et al. 2012) for more information on extent of use of best management practices by growers in the study area.

Adminsitrative Record
Page 35250

would need to be conducted, quantifying the improvement in nitrate leaching or in groundwater quality beneath the fields. Regional groundwater contaminant transport modeling studies would also need to be conducted to assess the necessary time scale of this remedial action to achieve the stabilization and reduction of groundwater itrate concentrations to acceptable levels. Such models would provide the basis for regional groundwater quality management with respect to nitrate but also with respect to other contaminants such as arsenic and salts that also affect drinking and irrigation water quality. Moreover, groundwater quality management models would help estimate how much of the downward migrating nitrate would be captured by existing and additional wells, depending on pumping rates and schedules. Ultimately, the goal would be to manage groundwater quantity and quality jointly to improve the sustainability of both.

A disadvantage of the PAF approach is that many existing irrigation wells are designed to pump at large extraction rates, requiring that they are drilled to relatively large depth drawing water across multiple aquifer layers and allowing for sometimes large water level drawdowns near the well. Shallow-to-intermediate depth nitrate contaminated groundwater is therefore not efficiently intercepted by these wells (also see Technical Report 4, Boyle et al., 2012). An alternative option to better capture high nitrate shallow-to-intermediate depth groundwater is to drill intermediate-depth irrigation wells that would intercept contaminated groundwater before it further penetrates the deeper subsurface. This approach would require a significant and careful capital investment because shallower wells would have smaller capture zones, and therefore a much larger numbers of such wells would need to be drilled and operated. The above-mentioned groundwater quality management models would be an integral tool in the evaluation of the potential costs and benefits of constructing additional shallow to intermediate depth irrigation or capture wells.

A number of factors will determine the cost for an extensive pump and fertilize scheme, including the type of irrigation system, the number of wells on a farm, and the degree to which groundwater is used for irrigation purposes (exclusively or mixed with surface water). There is also cost associated with creating and setting up such a program on-farm, including costs for education, training, and planning, as well as for infrastructure changes. Of those factors, we here consider only the cost of testing water quality on a sufficiently regular basis to provide the farmer with confidence in the fertilizer nitrogen content of groundwater applied as irrigation water. Based on an informal survey of analytical costs for testing nitrate in water, the estimated cost is approximately $15/test for nitrate. Testing for other forms of nitrogen (ammonium or organic nitrogen) is more expensive, but generally not necessary for purposes of managing pump-and-fertilize: the total concentration of non-nitrate N in groundwater is typically less than 2 mg N/L. Additionally assuming that sample collection and shipping costs $15, the total cost is $30 per sample. If a nitrate sensor or nitrate test kit is used, the sample cost may be lower, albeit the cost for sample collection, in-field analysis, and instrument maintenance remains.

Within the study area, we estimate that there are between approximately 6,000 and 20,000 agricultural irrigation wells (see Section 9.5 in Technical Report 2, Viers et al., 2012). These wells pump, on average, 7.9 km^3 (6.1 million acre-feet) per year (see Technical Report 4, Boyle et al., 2012) and are active throughout much of the 3.8 million acres of irrigated cropland within the study area (see Technical Report 2, Viers et al., 2012).

Adminsitrative Record
Page 35251

To estimate cost, we assume each well is tested five times during each irrigation season to ensure that varying nitrate levels are adequately captured for the farmer to properly account for the nutrient value in the irrigation water. Per well, the sampling is then on the order of $150 per year. We note that on many wells, it will not be necessary to sample more than once or twice per irrigation season ($30 - $60 per year) once a sufficiently long record is established showing relatively constant nitrate values over the season. The analytical cost is only a small fraction of the amount of fertilizer value gained in this process: Assuming that a typical well pumps at least 200 acre-feet per irrigation season, and assuming that water contains an average 22.5 mg/L nitrate (5 mg N/L, half of the MCL), the amount of nitrogen "fertilizer" obtained from a well is 2,800 lb N per year, at a 2012 value of approximately $1,400, if used to replace commercial fertilizer.

For 6,000, 10,000, or 20,000 active wells, the total annual cost of regular nitrate data collection to estimate the irrigation water fertilizer value, is $0.9 million, $1.5 million, or $3 million for the entire study area, at least initially; and likely lower in the longer term.

The median nitrate concentration in public supply wells during the last decade varies by groundwater sub-basin (see Technical Report 4, Boyle et al. 2012). The wells in the Westside and Tulare Lake Central Basin sub-basins typically have median nitrate concentrations well below 10 mg/L as nitrate. There, pump-and-fertilize may be of limited use until nitrate levels rise in the future – a likely consequence of the long-term groundwater transport processes – unless networks of shallow production wells are installed to provide capture of the upper aquifer portions. In the Kern sub-basin, median nitrate concentration is 16 mg/L as nitrate, while the median concentration in the Salinas Valley main aquifer and in the Kings, Kaweah, and Tule subbasins range from 20 to 25 mg/L as nitrate (see Technical Report 4, Boyle et al. 2012). The overall median nitrate concentration is 21 mg/L as nitrate. In 7.85 km^3 (6.1 million acre-feet) of irrigation water, this concentration gives over 35,000 GgN/yr. At current nitrogen fertilizer costs exceeding $1 per kg N ($0.50-$0.75/lb N), the theoretical "fertilizer value" of irrigation water is over $35 million. If a pump-and-fertilize program can take advantage of at least one-third to half the groundwater's fertilizer value, the net savings in fertilization costs could still be in the range of $10 - $20 million, exceeding the necessary investment in monitoring nitrate concentration in the groundwater by one order of magnitude.

There are multiple potential on-farm challenges to adopting a pump-and-fertilize program. Perhaps the largest is information, education, and training. Farmers may be unaware of the fertilizer value of their irrigation water, and/or do not have the means to properly interpret groundwater quality data for their irrigation wells in terms of accounting for its fertilizer value. Some farms have complex and seasonally varying irrigation setups with varying input from one or more wells and surface waters, making proper accounting of the fertilizer value that much more difficult and prone to error. For some of the high fertilizer need crops, the amount of nitrogen applied with irrigation water from groundwater (on the order of 10 - 100 kg N/ha depending on nitrate concentration and groundwater use), may be thought to be too insignificant to be used.

Nonetheless, basic pump-and-fertilize management - proper accounting for the nitrogen content of irrigation water in nutrient management planning - is an essential part of modern nutrient management.

Adminsitrative Record
Page 35252

Additional pump-and-fertilize management components, such as requiring that a larger number of agricultural wells be screened only in the shallower (higher nitrate) portions of the aquifer, would need to be evaluated for their cost and feasibility.

2.4.2 Understanding the Value of Irrigation Water Nitrogen

Nitrogen in groundwater pumped for irrigation has been shown to be important both, for crop nitrogen uptake and in field effluent nitrogen loads. In a detailed mass-balance study of a corn field in Yolo County, California, King et al (2009) found that influent irrigation water nitrate was an important constituent in post-irrigation runoff nitrogen content. In the endorheic (hydrologically closed) Tulare Lake Basin, field runoff eventually percolates to groundwater unless it is artificially captured and stored. Although the Salinas Valley does drain to the ocean (via the Salinas River and Elkhorn Slough), it is also subject to percolation of irrigation runoff waters.

Martin et al (1982) conducted a field-calibrated modeling study of nitrogen uptake by corn in Nebraska, with results that showed that nitrogen uptake efficiency by corn from irrigation water was actually higher than that from synthetic fertilizer nitrogen. This finding suggests that, in some cases, it may be possible to replace commercial fertilizer nitrogen with irrigation water nitrate nitrogen at a replacement rate of less than 1:1 (irrigation water N to commercial fertilizer N), further supporting the above economic valuations.

2.4.3 Current Use of Irrigation Water Nitrate in Fertilizer Calculations

The expert panels conducted for the current study (see Technical Report 3, Dzurella et al, 2012) indicated that although irrigation water nitrate testing is somewhat common, properly reducing fertilizer applications based on irrigation water nitrate content is less of a common practice. In areas of high groundwater nitrate, growers tend to make more of an effort to account for it. The complexity of the irrigation water source, the irrigation scheduling, and the need for technical expertise are the most important barriers. Many growers operate irrigation regimes that incorporate both surface water as well as multiple wells (at varying levels of nitrate concentration), meaning the pumping and piping (which are not static) contributes significantly to the complexity of accounting for irrigation water nitrate. In the Tulare Lake Basin, farms are much larger than in the Salinas Valley, increasing the tendency for complex irrigation modes. Growers also indicated some reluctance to test water in wells on leased land, citing privacy concerns of landlords. In the end, while more and more growers in the study area are aware of the issue, the complexity and associated learning curve keeps proper accounting a less common practice.

A survey of growers in the Salinas Valley was conducted in 2001 (MCWRA 2002). This survey was voluntary. Of 314 growers who received the survey, 107 growers responded, which represented 49% of the irrigated agricultural acres in the Salinas Valley. No conclusions are drawn regarding the applicability of these results to non-surveyed growers. The survey found that 66% of the area was farmed by growers who stated they accounted for nitrate in some of their calculations of fertilizer application rates. Monterey County Water Resources Agency recommends to growers that they account for

Adminsitrative Record
Page 35253

irrigation water nitrogen content. In a fact sheet produced by MCWRA, (1999), a detailed description of how to take this nutrient source into account is presented, and it appears that farmers in that region use it (MCWRA 2002). No such technical assistance is widely available to growers in the Tulare Lake Basin.

In Nebraska, and in Australia, farmers are encouraged to account for irrigation water nitrogen (nitrate) when calculating the amount of fertilizer they will apply. The Nebraska Cooperative Extension office offers advice and training to this end, including tables of irrigation water nitrogen content by region (of Nebraska) that can be used by farmers in the absence of irrigation well water nitrogen monitoring data (Ferguson et al 1994). The Queensland (Australia) Department of Environment and Resource Management Reef Protection Package (2009) (addressed specifically to sugar-cane growers in coastal areas where nitrogen leaching to groundwater impacts coral reefs in the near-shore ocean environment) gives their recommended method for calculating the amount of reduction in fertilizer nitrogen necessary to account for nitrate in irrigation water. Both the Queensland and Nebraska methods recommend that a 1-for-1 reduction (nitrogen in irrigation water for nitrogen in fertilizer) be made.

2.5 Management of Groundwater Recharge

In the Tulare Lake Basin the dominant source of groundwater recharge is irrigation, while infiltration of surface water from streams is a secondary source. Direct precipitation also contributes some recharge. A basic premise of regional groundwater quality management is that if most of the recharge to a basin is contaminated with recalcitrant compounds like nitrate, the groundwater quality is more vulnerable and likely non-sustainable. If however, a less contaminated source of recharge, such as stream infiltration or recharge ponds can be augmented to decrease the ratio of contaminated-to-clean recharge, regional groundwater quality will improve and is more sustainable. A good example of the effects of relatively clean recharge from streambed infiltration can be seen in the vicinity of the Leaky Acres recharge facility in Fresno, where groundwater nitrate tend to be lower than groundwater that is receiving most of its recharge from irrigation (Technical Report 4, Boyle et al., 2012). There is also evidence suggesting that recharge from the Salinas River results in lower groundwater nitrate concentrations near the river.

As climate change results in less storage of surface water in California reservoirs owing to earlier snow melt and flood control requirements, subsurface storage of water will become increasingly necessary to mitigate the loss of snow water storage. Moreover, if some of this earlier winter runoff can be captured and diverted to streams and groundwater recharge operations, the volumes of 'clean' recharge will increase. The beneficial effects of clean recharge on the nitrate problem would need to be estimated through regional scale modeling of the groundwater quality under various pumping, irrigation, and recharge scenarios.

Adminsitrative Record
Page 35254

3 Summary and Conclusions

The basin-scale pump-and-treat (PAT) approach is not recommended for implementation in the study areas due to its economic and logistic impracticality. Hot-spot source reduction through local-scale remediation methods such as injection wells and PRBs for in-situ treatment in targeted areas, together with regional scale management of irrigation water nitrate and optimized fertilizer application (pump-and-fertilize, PAF) are the most promising actions and will likely improve groundwater quality over the long term. The benefits of these measures, and the time necessary for these changes to occur, can be estimated through agricultural-field scale monitoring at focus sites and through regional scale modeling of groundwater quality. Some farmers are applying PAF now, but technical support and incentives are needed to encourage broader implementation, especially in the Tulare Lake Basin, but also in Salinas Valley.

Implementation will require regional groundwater quality management models for determining the combinations of N source reductions, localized remediation, irrigation water and N management (PAF), streambed recharge, and groundwater pumping distributions that will bring about improvement in the groundwater quality on a time scale of decades to centuries. The long time frame required for such actions to succeed will present both policy and implementation challenges. The policy must still be developed based on current scientific knowledge, some of which is presented in this chapter, together with science that will come from the needed groundwater quality management models. Because the effects of any practices set into motion by policy will unfold slowly, it will be important to use an adaptive management approach, in which predictions of trends in groundwater quality are regularly checked against monitoring data that are then used to recalibrate models, assumptions, and policies.

Adminsitrative Record
Page 35255

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Adminsitrative Record
Page 35259

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Adminsitrative Record
Page 35261

Adminsitrative Record
Page 35262

Appendix

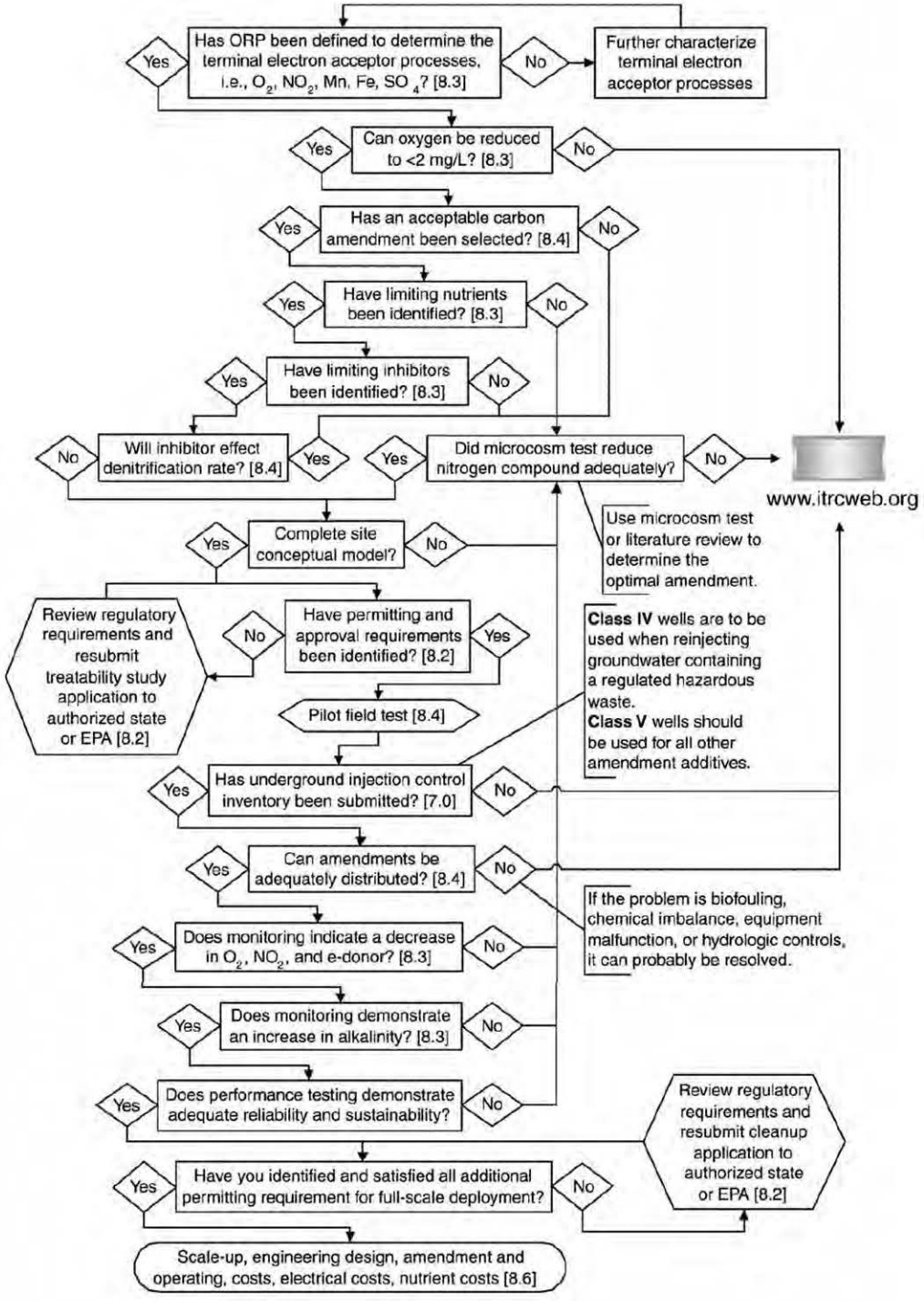

Figure A-1. Decision-tree for the application of in situ bioremediation for nitrate (reproduced with permission from ITRC 2002).

Adminsitrative Record
Page 35263

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 6:

Drinking Water Treatment for Nitrate

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
http://groundwaternitrate.ucdavis.edu

Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 35264

Drinking Water Treatment for Nitrate

Technical Report 6

Addressing Nitrate in California's Drinking Water

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

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Contents

Tables v

Figures vii

Acronyms and Abbreviations ix

Unit Conversions xi

Summary 1

Objective 1

Background 1

Approach 2

Findings 5

Non-Treatment Options 5

Treatment Options 5

Conclusions 9

1 Introduction 12

1.1 Management Options for Nitrate in Potable Water 12

2 Non-Treatment Options for Nitrate Contaminated Potable Water 14

2.1 Well Abandonment, Inactivation, and Destruction 14

2.2 Wellhead Protection and Land Use Management 14

2.3 Development of Alternative Sources and Source Modification 14

2.4 Blending 16

3 Treatment Options for Nitrate Contaminated Potable Water 18

3.1 Ion Exchange (IX) 21

3.1.1 Conventional Ion Exchange 21

3.1.2 Ion Exchange - Design Considerations 23

3.1.3 Ion Exchange - Cost Considerations 28

Adminsitrative Record
Page 35269

3.1.4 Ion Exchange - Selected Research 29
3.1.5 Ion Exchange - Summary of Advantages and Disadvantages 29
3.1.6 Modifications to Conventional Ion Exchange 29
3.1.7 Ion Exchange - Case Studies 35
3.2 Reverse Osmosis (RO) 48
3.2.1 Reverse Osmosis - Design Considerations 48
3.2.2 Reverse Osmosis - Cost Considerations 53
3.2.3 Reverse Osmosis - Selected Research 54
3.2.4 Reverse Osmosis - Summary of Advantages and Disadvantages 54
3.2.5 Reverse Osmosis - Improvements and Modifications 55
3.2.6 Reverse Osmosis - Case Studies 57
3.3 Electrodialysis (ED/EDR/SED) 73
3.3.1 Electrodialysis - Design Considerations 75
3.3.2 Electrodialysis - Cost Considerations 77
3.3.3 Electrodialysis - Selected Research 78
3.3.4 Electrodialysis - Summary of Advantages and Disadvantages 78
3.3.5 Modifications to Electrodialysis 79
3.3.6 Electrodialysis - Case Studies 80
3.4 Biological Denitrification (BD) 88
3.4.1 Biological Denitrification - Design Considerations 90
3.4.2 Biological Denitrification - Cost Considerations 95
3.4.3 Biological Denitrification - Selected Research 96
3.4.4 Biological Denitrification - Summary of Advantages and Disadvantages 96
3.4.5 Biological Denitrification - Case Studies 97
3.5 Chemical Denitrification (CD) 106

Adminsitrative Record
Page 35270

3.5.1 Zero Valent Iron (ZVI) 106
3.5.2 Catalytic Denitrification 108
3.5.3 Chemical Denitrification - Design Considerations 108
3.5.4 Chemical Denitrification - Emerging Technologies 110
3.5.5 Chemical Denitrification - Cost Considerations 113
3.5.6 Chemical Denitrification - Selected Research 114
3.5.7 Chemical Denitrification - Summary of Advantages and Disadvantages 114
3.6 Brine Treatment Alternatives and Hybrid Treatment Systems 114
3.6.1 Electrochemical Destruction of Nitrate in Waste Brine 115
3.6.2 Catalytic Treatment of Waste Brine 116
3.7 Residential Treatment (Point-of-Use, Point-of-Entry) 117
4 Tulare Lake Basin and Salinas Valley - Water Quality Analysis 120
4.1 Water Quality - Treatment Interference 121
4.2 Water Quality - Co-contaminants 121
4.3 Water Quality and Treatment Selection 128
5 Addressing Nitrate Impacted Potable Water Sources in California 130
5.1 Well Abandonment, Destruction and Inactivation 130
5.2 Survey of Blending and Treating Systems 133
6 Treatment Cost Analysis 139
6.1 Costs by Treatment Type 141
6.2 Costs by System Size 144
6.3 Costs by Water Quality Parameters 147
6.4 Disposal Costs 149
7 Guidance for Addressing Nitrate Impacted Drinking Water 152
7.1 Checklist for the Selection of Mitigation Strategy 152

Adminsitrative Record
Page 35271

7.2 Decision Trees 153

8 Summary and Conclusions 155

9 Literature Cited 157

10 Appendix 175

10.1 Tables of Selected Research 175

Adminsitrative Record
Page 35272

Tables

Table S.1. Utilities included in the case studies. 4

Table S.2. Potable water treatment options for nitrate management 6

Table S.3. Comparison of major treatment types. 11

Table 1. Utilities included in the case studies. 19

Table 2. Potable water treatment options for nitrate management. 20

Table 3. Selection of full-scale ion exchange installations for nitrate removal. 21

Table 4. Summary of design considerations for conventional IX. 24

Table 5. Selected published costs of ion exchange systems for nitrate removal. 29

Table 6. Summary of design considerations for reverse osmosis. 49

Table 7. Selected costs of reverse osmosis systems for nitrate removal. 54

Table 8. Summary of design considerations for electrodialysis/electrodialysis reversal. 75

Table 9. Sample EDR O&M costs. 78

Table 10. Full-scale biological denitrification systems for potable water treatment. 89

Table 11. Summary of design considerations for biological denitrification for nitrate removal. 90

Table 12. Membrane biological reactor configurations. 94

Table 13. Cost information for biological denitrification of potable water. 96

Table 14. Summary of design considerations for chemical denitrification. 109

Table 15. Selected research on brine treatment alternatives and hybrid systems 115

Table 16. Costs of POU treatment for nitrate removal. 119

Table 17. Summary of water quality data of high nitrate wells in Tulare Lake Basin and Salinas Valley. 120

Table 18. Comparison of major treatment types 129

Table 19. Influence of nitrate concentration on treatment selection. 129

Table 20. Incidence of abandonment, destruction, and inactivation of nitrate impacted wells. 131

Table 21. Population and nitrate levels of systems in the study area treating or blending for nitrate ... 137

Adminsitrative Record
Page 35273

Table 22. Nitrate level, well depth and well capacity for a Tulare County blending system. 138

Table 23. Cost estimation using U.S. EPA cost curves of IX for arsenic removal. 140

Table 24. Summary of anion exchange and reverse osmosis cost information by system size. 146

Table 25. An exercise in the estimation of treatment costs based on nitrate levels. 148

Table 26. Brine disposal costs. 150

Table A.1. Selected research on the use of ion exchange for nitrate removal. 175

Table A.2. Selected research on the use of reverse osmosis for nitrate removal. 176

Table A.3. Selected research on the use of electrodialysis for nitrate removal. 177

Table A.4. Selected research on the use of biological denitrification for nitrate removal. 178

Table A.5. Selected research on the use of chemical denitrification for nitrate removal. 181

Table A.6. Advantages and disadvantages of the five major treatment options for nitrate removal. 182

Adminsitrative Record
Page 35274

Figures

Figure S.1. Summary of nitrate management options. 2

Figure 1. Selective well screening using a packer/plug. 16

Figure 2. Conventional ion exchange schematic. 22

Figure 3. Process flow diagram for counter current MIEX® process. 30

Figure 4. Vessel rotation in Calgon Carbon countercurrent ISEP® system. 32

Figure 5. Example of flow through the ISEP® system. 32

Figure 6. Example of an Envirogen multiple bed proprietary anion exchange system. 33

Figure 7. Process schematic of weak base ion exchange for nitrate ("WIN" process). 35

Figure 8. Reverse osmosis schematic. 48

Figure 9. Flow chart of the HERO™ process. 56

Figure 10. Electrodialysis reversal schematic. 73

Figure 11. Illustration of electrodialysis membrane stack. 74

Figure 12. Biological denitrification schematic. 88

Figure 13. FBR configuration. 98

Figure 14. FBR treatment system schematic. 100

Figure 15. Surface chemistry of ZVI particles 107

Figure 16. Process schematic for denitrification using SMI-III®. 111

Figure 17. Schematic of the brine treatment system developed by Ionex SG Limited. 116

Figure 18. Schematic of an ion exchange system with brine regeneration coupled with catalytic treatment of brine for reuse. 116

Figure 19. Raw water nitrate levels exceeding the MCL 122

Figure 20. Raw water high nitrate wells with high arsenic levels. 123

Figure 21. Raw water high nitrate wells with high perchlorate levels. 124

Figure 22. Raw water high nitrate wells with pesticides detected. 125

Adminsitrative Record
Page 35275

Figure 23. Salinas Valley [As] versus total well depth (deepest water) and [NO_3^-] versus depth to top of screen (shallowest water). 127

Figure 24. Tulare Lake Basin [As] versus total well depth (deepest water) and [NO_3^-] versus depth to top of screen (shallowest water). 127

Figure 25. Tulare Lake Basin: Incidence of nitrate and arsenic MCL exceedance with well depth. 128

Figure 26. Location of nitrate impacted abandoned, destroyed, and inactivated wells. 132

Figure 27. Digital survey distributed to drinking water systems treating and/or blending to address high nitrate levels. 134

Figure 28. California drinking water systems treating and/or blending for nitrate. 135

Figure 29. Drinking water systems treating or blending for nitrate in the Tulare Lake Basin and Salinas Valley. 136

Figure 30. Wells of a blending system in Tulare County. 138

Figure 31. Average cost comparison of nitrate treatment technologies. 142

Figure 32. Costs of anion exchange for nitrate treatment. 143

Figure 33. Costs of reverse osmosis for nitrate treatment. 143

Figure 34. Costs of biological denitrification in drinking water treatment. 144

Figure 35. Cost curve of IX (blue) and RO (red) for nitrate removal. 145

Figure 36. Decision Tree 1 - Options to address nitrate impacted drinking water sources 153

Figure 37. Decision Tree 2 - Anion exchange. 154

Adminsitrative Record
Page 35276

Acronyms and Abbreviations

AF	Acre Feet
AFY	Acre Feet per Year
AWWA	American Water Works Association
BMP	Best Management Practice
BD	Biological Denitrification
BV	Bed Volume
CD	Chemical Denitrification
CIP	Clean-In-Place
DBP	Disinfection Byproduct
DL	Detection Limit
DO	Dissolved Oxygen
EBCT	Empty Bed Contact Time
ED	Electrodialysis
EDR	Electrodialysis Reversal
FBR	Fluidized Bed Reactor
FXB	Fixed Bed Reactor
GAC	Granular Activated Carbon
GHG	Greenhouse Gas
GPM	Gallons per Minute
GWUDI	Groundwater Under Direct Influence (of Surface Water)
HERO™	High Efficiency Reverse Osmosis
HLR	Hydraulic Loading Rate
ISEP®	Ion Exchange Separation System
IX	Ion Exchange
LSI	Langelier Saturation Index
MBfR	Membrane Biofilm Reactor
MBR	Membrane Bioreactor
MCL	Maximum Contaminant Level
MGD	Million Gallons per Day
MIEX®	Magnetic Ion Exchange
NDMA	N-nitrosodimethylamine
O&M	Operations and Maintenance
PICME	Permits Inspection Compliance Monitoring and Enforcement
POE	Point-of-Entry
POU	Point-of-Use
PRB	Permeable Reactive Barrier
RO	Reverse Osmosis
SBA IX	Strong Base Anion Exchange
SBR	Sequencing Batch Reactor

Adminsitrative Record
Page 35277

SDI	Silt Density Index
SDWA	Safe Drinking Water Act
SED	Selective Electrodialysis
SMI	Sulfur Modified Iron
TDS	Total Dissolved Solids
TSS	Total Suspended Solids
ULPRO	Ultra-Low Pressure Reverse Osmosis
USBR	Up-flow Sludge Blanket Reactor
U.S. EPA	United States Environmental Protection Agency
VSEP	Vibratory Shear Enhanced Process
WAC	Weak Acid Cation Exchange
WBA IX	Weak Base Anion Exchange
WQM	Water Quality Monitoring
ZVI	Zero Valent Iron

Adminsitrative Record
Page 35278

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day
Nitrate Units			
*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-). To convert from:			
Nitrate-N (NO_3-N) →	Nitrate (NO_3^-)	multiply by 4.43	
Nitrate (NO_3^-) →	Nitrate-N (NO_3-N)	multiply by 0.226	

Adminsitrative Record
Page 35279

Summary

Objective

The purpose of the first three sections of this document is to provide a detailed guide to the current state of nitrate treatment alternatives that can be used as a reference tool for the drinking water community. The remainder of this document focuses on nitrate treatment of drinking water in California and specifically in the Tulare Lake Basin and the Salinas Valley.

Background

Nitrate contamination of potable water sources is becoming one of the most important water quality concerns in California and across the United States. The maximum contaminant level (MCL) for nitrate is 45 mg/L as nitrate (NO_3^-), which is approximately equivalent to 10 mg/L as nitrogen (N). The major health concern of nitrate exposure through drinking water is the risk of methemoglobinemia, or "blue baby syndrome," especially in infants and pregnant women. Due to the nature of the infant digestive system, nitrate is reduced to nitrite which can render hemoglobin unable to carry oxygen (SWRCB 2010). Nitrate is naturally occurring at low levels in most waters, but it is particularly prevalent in groundwater that has been impacted by certain agricultural, commercial, or industrial activities. Of specific concern are crop fertilization activities and discharges from animal operations, wastewater treatment facilities, and septic systems. Small rural communities are particularly impacted by nitrate (Pacific Institute 2011). Nitrate presents unique water treatment challenges. The United States Environmental Protection Agency (U.S. EPA) lists only anion exchange (IX), reverse osmosis (RO), and electrodialysis reversal (EDR) as accepted potable water treatment methods for nitrate removal (U.S. EPA 2010). Due to the production of high-strength brine residuals, sustainable application of these three technologies is often limited by a lack of local residual disposal options and the challenge of increasing salt loads. The lack of affordable and feasible nitrate treatment alternatives can force impacted utilities to remove nitrate-contaminated sources from their available water supply. In many instances, this action can severely compromise a water utility's ability to provide an adequate supply of safe and affordable potable water.

The need for additional nitrate treatment technologies has driven the drinking water community to begin developing alternative options to effectively remove nitrate while limiting cost and brine production challenges. Promising treatment options include weak base anion (WBA) exchange and improvements in strong base anion (SBA) exchange such as low brine residual technologies; biological treatment using fluidized bed, fixed bed, and membrane biofilm (MBfR) reactors; and chemical reduction using media such as zero valent iron (ZVI) and sulfur modified iron (SMI). A summary of the options to address nitrate contamination of drinking water is presented in Figure S.1. In this diagram treatment options are classified in terms of their ability to either remove nitrate to a residual waste stream or transform nitrate to other nitrogen species through reduction.

Adminsitrative Record
Page 35280

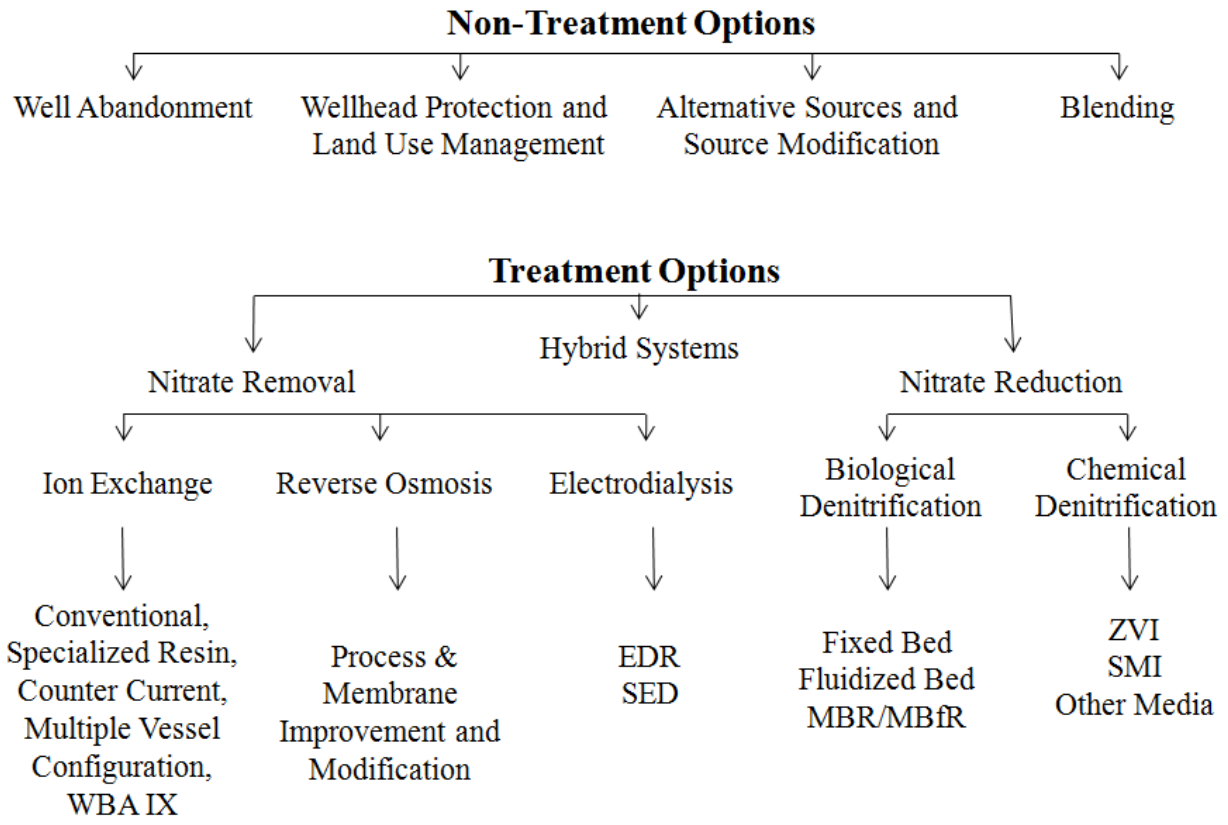

Figure S.1. Summary of nitrate management options.[4]

Approach

This report includes a comprehensive literature review and case studies of specific systems across the range of nitrate treatment alternatives. The literature review is intended to provide background information about current and emerging potable water treatment alternatives to address nitrate contamination. In addition to peer-reviewed literature, information found in the "grey papers" of conference proceedings has been included to assure capture of the most recent technology developments. For each of the major treatment technologies, subsections of the literature review detail the following:

- Design considerations including water quality, system layout, and site considerations; residuals management and disposal; and maintenance, monitoring, and operational complexity,
- cost considerations,
- selected research, and
- a summary of advantages and disadvantages.

[4] For the purposes of this discussion, blending has been categorized as a "non-treatment" option; however, in practice, blending is sometimes referred to as "treatment." Treatment options throughout this report refer to treatment technologies available for the removal or reduction of nitrate in drinking water. Blending can sometimes be used to cost-effectively address the nitrate problem through dilution, but has been categorized separately from treatment options for simplicity.

Adminsitrative Record
Page 35281

Information is summarized in tables whenever appropriate, including a summary table of selected research studies for each of the major treatment technologies (Appendix).

A survey was conducted to collect detailed information about the application of nitrate treatment. A subset of utilities, currently treating for nitrate and/or in design for future treatment, was included in the survey. The survey was developed to gather information with respect to the benefits and limitations of the various nitrate treatment technologies and was conducted via phone and in-person when applicable. The list of utilities included in the survey was developed with the intention of covering a range of utilities with respect to geographic location, treatment type, population size and residual handling techniques (Table S.1). Detailed case studies have been compiled for each of the treatment technologies where full-scale facilities have been in operation or are moving ahead with design. This survey was conducted through collaboration with Jacobs Engineering in the completion of the associated assessment of nitrate treatment alternatives for the American Water Works Association (AWWA) and is complemented by a parallel survey of nitrate treatment systems in California. Details from the initial survey are included as examples following a discussion of each of the treatment technologies. Details of the complementary survey of California systems are included in the second half of this report.

Adminsitrative Record
Page 35282

Table S.1. Utilities included in the case studies.

Case #	Treatment Type	Location	Capacity (gpm)	Avg. Influent Nitrate mg/L as NO_3^- (mg/L as N)
	Ion Exchange			
1	Conventional ion exchange with blending	California	400	31 – 53 (7 – 12)
2	Conventional ion exchange with blending	California	400	~45 (~10)
3	Counter Current Ion Exchange (MIEX®)	Indian Hills, CO	50	53 – 71 (12 – 16)
4	Multiple vessel ion exchange	California	500 – 900	35 – 89 (8 – 20)
5	Multiple vessel ion exchange	Chino, CA	5000	40 – 200 (9 – 45)
	Reverse Osmosis			
6	Reverse osmosis and blending	Bakersfield, CA	120	75 – 84 (17 – 19)
7	Reverse osmosis, exploring biological reduction	Brighton, CO	4600	49 – 89 (11 – 20)
8	Reverse osmosis and blending	Arlington Desalter, Riverside, CA	4583	44 – 89 (10 – 20)
	Combined Reverse Osmosis and Ion Exchange			
9	Reverse osmosis, ion exchange and blending	Chino Desalter I, Chino, CA	4940 (RO), 3400 (IX)	148 – 303 (33 – 68)
10	Reverse osmosis, ion exchange and blending	Chino Desalter II, Mira Loma, CA	4167 (RO), 2778 (IX)	70 – 224 (16 – 51)
	Electrodialysis/Electrodialysis Reversal/Selective Electrodialysis			
11	Electrodialysis Reversal (EDR)	Spain	3,260 (each, 2 systems)	~80 (~18)
12	Selective Electrodialysis (SED)	Israel	310	84 – 89 (19 – 20)
	Biological Denitrification			
13	Implementing fluidized bed biological reduction	Rialto, CA	2000 – 4000	17 – 19 (~4 – 5)
14	Implementing fixed bed biological reduction	Riverside, CA	1670	44 – 89 (10 – 20)

Adminsitrative Record
Page 35283

Findings

Non-Treatment Options

The focus of this assessment is the current state of nitrate treatment alternatives. However, in practice, non-treatment options are generally considered first as they can often be more sustainable and less costly. Non-treatment options include wellhead protection, land use management, well inactivation, source modification, development of alternative sources (including consolidation/connection to a nearby system), and blending. Blending was found to be the most common method to address nitrate contamination. When a low nitrate water supply source is available, dilution of high nitrate sources to produce water with nitrate levels below the MCL is typically more cost-effective than installing treatment.

Treatment Options

Nitrate treatment technologies were categorized into five major types. Ion exchange (IX), reverse osmosis (RO), and electrodialysis/electrodialysis reversal (ED/EDR) remove nitrate to a concentrated waste stream, while biological denitrification (BD) and chemical denitrification (CD) transform nitrate to other nitrogen species through reduction. Common concerns in the application of the removal technologies include waste management costs and treatment interference from other water quality parameters (e.g., hardness and sulfate). Pretreatment is often required to avoid fouling or scaling of the resin for IX and the membranes for RO and ED/EDR. Due to the destruction of nitrate, both biological and chemical denitrification have the potential for more sustainable treatment without brine residuals, but also have limitations to consider. Full-scale application of these nitrate treatment options is currently limited.

The selection of the most appropriate treatment option depends on various key factors specific to the needs and priorities of individual water systems. A brief comparison of fundamental design considerations, and advantages and disadvantages of these treatment options is listed in Table S.2. It is important to note that the contents of Table S.2 are not intended to provide a comprehensive set of criteria for treatment options. Other important criteria in determining the best treatment option, which are site specific and cannot be broadly generalized, include capital and operations and maintenance (O&M) costs, system size (capacity), and system footprint. Overall, there is no single treatment option that can be considered the best method for nitrate removal across all water quality characteristics and for all systems.

Adminsitrative Record
Page 35284

Table S.2. Potable water treatment options for nitrate management (adapted from WA DOH 2005).

	Ion Exchange	Reverse Osmosis	Electrodialysis	Biological Denitrification	Chemical Denitrification
Full-scale Systems	Yes	Yes	Yes	Yes	No
Treatment Type	Removal to waste stream	Removal to waste stream	Removal to waste stream	Biological reduction	Chemical reduction
Common Water Quality Design Considerations	Sulfate, iron, manganese, total suspended solids (TSS), metals (e.g., arsenic), hardness, organic matter	Turbidity, iron, manganese, SDI, particle size, TSS, hardness, organic matter, metals (e.g., arsenic)	Turbidity, iron, manganese, TSS, hydrogen sulfide, hardness, metals (e.g., arsenic)	Temperature and pH, anoxic conditions	Temperature and pH
Pretreatment Needs	Pre-filter, address hardness	Pre-filter, address hardness	Pre-filter, address hardness	pH adjustment, nutrient and substrate addition, need for anoxic conditions	pH adjustment
Post-treatment Needs	pH adjustment	pH adjustment Remineralization	pH adjustment Remineralization	Filtration, disinfection, possible substrate adsorption	pH adjustment, iron removal, potential ammonia control
Waste/Residuals Management	Waste brine	Concentrate	Concentrate	Sludge/biosolids	Waste media, Iron sludge
Start-up Time	Minutes	Minutes	Minutes	Initial plant startup: Days to weeks After reaching steady state: Minutes	Minutes
Water Recovery	Conventional (97%) Low brine (Up to 99.9%)	Up to 85%	Up to 95%	Nearly 100%	Not demonstrated full-scale
Advantages	Nitrate selective resins, common application, multiple contaminant removal	Multiple contaminant removal, desalination (TDS removal)	Multiple contaminant removal, higher water recovery (less waste), desalination, unaffected by silica	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, high water recovery, and potential for multiple contaminant removal	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, and potential for multiple contaminant removal
Disadvantages	Potential for nitrate peaking, high chemical use (salt), brine waste disposal, potential for disinfection byproduct (DBP) formation (e.g., NDMA)	Membrane fouling and scaling, lower water recovery, operational complexity, energy demands, waste disposal	Energy demands, operational complexity, waste disposal	Substrate addition, potentially more complex, high monitoring needs, possible sensitivity to environmental conditions, risk of nitrite formation (potential incomplete denitrification), post-treatment to address turbidity standards and 4-log virus removal (state dependent)	Inconsistency of nitrate reduction, risk of nitrite formation (potential incomplete denitrification), reduction to ammonia, lack of full-scale systems, pH and temperature dependence, possible need for iron removal

Adminsitrative Record
Page 35285

Ion Exchange (IX)

The most commonly used nitrate treatment method is IX. Anion exchange for nitrate removal is similar to a water softener, with nitrate ions removed rather than hardness ions. Nitrate is removed from the treatment stream by displacing chloride on an anion exchange resin. Subsequently, regeneration of the resin is necessary to remove the nitrate from the resin. Regeneration is accomplished by using a highly concentrated salt solution resulting in the displacement of nitrate by chloride. The result is a concentrated waste brine solution high in nitrate that requires disposal. The most significant drawback of this treatment option is the cost for disposal of waste brine, especially for inland communities. The brine volume is largely dependent on the raw water quality and the configuration of the system.

Key factors in the consideration of IX include the pretreatment requirements to avoid resin fouling, the potential need for nitrate selective resin, the frequency of resin replacement, the possible post-treatment requirements to address corrosion or other product water quality concerns (e.g., the potential for NDMA formation), and the management of waste brine. If waste brine disposal options are not limiting, IX can be the best option for low to moderate nitrate contamination and removal of multiple contaminants (including arsenic, perchlorate, and chromium). Application of IX may not be feasible for extremely high nitrate levels due to salt use and waste volume. Current research on brine treatment alternatives may lead to the development of technologies capable of effectively addressing the disposal concern; however, the costs for full-scale implementation of this are unknown at this point.

Modifications to conventional IX have emerged in recent years offering low brine alternatives with improved efficiency. The efficiency of IX systems is dependent on the raw water characteristics. It is important to note that there can be cases where conventional IX systems yield greater water efficiency than a modified system that is implemented at a location with lesser water quality.

Another promising alternative to consider for the future is weak base ion exchange (WBA IX). This emerging technology is more operationally complex than conventional IX, but may offer the advantage of recycling waste as fertilizer.

Reverse Osmosis (RO)

As the second most common nitrate treatment alternative, RO can be feasible for both municipal and Point-of-Use applications and can be used simultaneously for desalination and removal of nitrate and many co-contaminants. Following pretreatment to prevent membrane fouling and scaling, water is forced through a semi-permeable membrane under pressure such that the water passes through, while contaminants are impeded by the membrane.

Key factors in the consideration of RO are the pretreatment requirements, the trade-off between water recovery and power consumption, the management of waste concentrate, and the typically higher costs relative to IX. One deciding factor favoring the selection of RO over IX for nitrate removal would be the need to address salinity.

Adminsitrative Record
Page 35286

Recent advancements in membrane technology and optimization of pre- and post-treatment have led to increases in the efficiency of RO treatment systems. For example, the use of Ultra-Low Pressure Reverse Osmosis (ULPRO) membranes enables lower power consumption.

Electrodialysis (ED, EDR, SED)

The use of ED in potable water treatment has increased in recent years, offering the potential for lower residual volumes through improved water recovery, the ability to selectively remove nitrate ions, and the minimization of chemical and energy requirements. ED works by passing an electric current through a series of anion and cation exchange membranes that trap nitrate and other ions in a concentrated waste stream. To minimize fouling and thus the need for chemical addition, the polarity of the system can be reversed with electrodialysis reversal (EDR). By reversing the polarity (and the solution flow direction) several times per hour, ions move in the opposite direction through the membranes, minimizing buildup.

Key factors in the consideration of EDR are the pretreatment requirements, the operational complexity of the system, the limited number of system manufacturers, the management of waste concentrate, and the lack of full-scale installations for nitrate removal from potable water in the United States. Like RO, EDR is commonly used for desalination and can be an alternative for nitrate treatment of high TDS waters. In contrast to conventional RO, EDR is unaffected by silica. EDR costs are similar to RO and evidence suggests that EDR can be the preferable option as the Silt Density Index (SDI) increases. For very small particle sizes, robust pretreatment can be necessary for RO. It is important to note that the EDR process does not directly filter the treatment stream through the membranes; contaminants are transferred out of the treatment stream and trapped by the membranes. This generally minimizes membrane fouling, decreasing pretreatment requirements in comparison to RO.

Biological Denitrification (BD)

Biological denitrification in potable water treatment is more common in Europe with recent full-scale systems in France, Germany, Austria, Poland, Italy, and Great Britain. To date, full-scale drinking water applications in the United States are limited to a single plant in Coyle, OK (no longer online). However, two full-scale systems are anticipated in California in the next couple of years. Biological denitrification relies on bacteria to transform nitrate to nitrogen gas through reduction. Substrate and nutrient addition is necessary and post-treatment can be more intensive than for the removal processes. Biological denitrification offers the ability to address multiple contaminants and the avoidance of costly waste brine disposal.

Key factors in the consideration of biological denitrification are the chemical requirements, the need for anoxic conditions, the level of operator training, the robustness of the system, and the post-treatment requirements. State regulations are expected to vary and, until more experience with the application of biological denitrification for potable water treatment is obtained in the United States, pilot and demonstration requirements may be intensive. Typically, biological treatment is thought to have a larger footprint; however, with the latest design configurations, the system footprint may be comparable to that of RO or EDR systems.

Adminsitrative Record
Page 35287

With reduction of nitrate to nitrogen gas, the lack of a problematic brine waste stream is a clear advantage of biological treatment over the removal processes. Biological treatment has the potential to provide a sustainable nitrate treatment option for the long term. More will be known with the completion of the anticipated full-scale systems in California; cost estimation suggests that biological treatment can be economically competitive with IX.

Chemical Denitrification (CD)

Chemical denitrification uses metals to transform nitrate to other nitrogen species. As an emerging technology, no full-scale chemical denitrification systems have been installed in the United States for nitrate treatment of potable water, and application for nitrate treatment has been strictly limited to bench- and pilot-scale studies. A significant body of research has explored the use of zero valent iron (ZVI) in denitrification. Several patented granular media options have also been developed including sulfur modified iron (SMI) media, granular clay media, and powdered metal media.

Key factors in the consideration of chemical denitrification are the reliability and consistency of nitrate reduction, the lack of full-scale installations, the type of media, and the dependence on temperature and pH. Chemical denitrification has the potential to become a feasible full-scale nitrate treatment alternative, with the advantage of reducing nitrate to other nitrogen species and avoiding the need to dispose of a concentrated waste stream. However, currently this option is an emerging technology in need of additional pilot- and full-scale testing. Due to the potential benefits, further research and optimization of chemical denitrification systems will likely make this a competitive option in the future, especially for multiple contaminants (e.g., arsenic and chromium).

Conclusions

- Current full-scale nitrate treatment installations in the United States consist predominantly of IX and RO. While EDR is a feasible option for nitrate removal from potable water, the application of EDR is generally limited to waters that have high TDS or silica. The use of biological denitrification to address nitrate contamination of drinking water is more common in Europe than in the U.S. However, this option is emerging in the U.S. and two full-scale systems are expected in a few years. Chemical denitrification may become a feasible nitrate treatment option in the future; however, the lack of current full-scale implementation suggests the need for further research, development and testing.

- Brine reuse and treatment are vital to the continued reliance on IX for nitrate treatment of potable water. The low brine technologies offer a minimal waste approach and current research and development of brine treatment alternatives seem to be lighting the path toward future progress.

- In regions with declining water quality and insufficient water quantity, the need to address multiple contaminants will increase in the future, suggesting the future dominance of technologies capable of multiple contaminant removal. In this context, for any individual water source or system, the most appropriate technology will vary with the contaminants requiring

Adminsitrative Record
Page 35288

mitigation. Although complex, analysis of the optimal treatment option for pairs and groups of contaminants will assist in the treatment design and selection. In such scenarios, the best treatment option for nitrate may not be the most viable overall.

- Currently and into the future, selection of the optimal and most cost-effective potable water treatment options will depend not only on the specific water quality of a given water source, but also on the priorities of a given water system. If land is limited, the typical configuration required for biological treatment may not be feasible. If brine waste disposal options are costly or limited, implementation of denitrification treatment or development of brine recycling and treatment may be the most suitable option.

- When deciding on nitrate treatment, the characteristics of the water system must be taken into account as well. With consideration of economies of scale, many rural small water systems cannot afford to install treatment. Even with financial assistance to cover capital costs, the long term viability of a treatment system can be undermined by O&M costs that are simply not sustainable. For such systems, treatment can become more affordable through consolidation of multiple small water systems into larger combined water systems that can afford treatment as a conglomerate. With a continued decline in water quality, non-treatment options alone, like blending groundwater sources or drilling a new well, may become insufficient measures for a water system to provide an adequate supply of safe and affordable potable water. Especially in rural small communities, perhaps the most promising approach will be consolidation of multiple nearby water systems and the installation of a single centralized treatment plant. Alternatively, separate small treatment facilities can be consolidated under a single agency. For additional discussion on the comparison of alternative water supply options and associated costs see Technical Report 7 (Honeycutt et al. 2012).

- While current cost considerations are commonly the driving force in selecting nitrate treatment, it is essential to consider the long term implications of current industry decisions. For example, it may be cost-effective for a particular system to utilize conventional IX currently, but future water quality changes (e.g., increasing nitrate levels, co-contamination, high salt loading), discharge regulations, or disposal fees may lead to an unmanageable increase in costs. Environmental sustainability in drinking water treatment is being addressed with brine treatment alternatives and denitrification options. It is important to approach the future of drinking water treatment with the mindset that environmental sustainability and economic sustainability are tightly interwoven.

- Centralized treatment may not be feasible for widespread rural communities; another approach to consider is centralized management (e.g., design, purchasing, and maintenance) to minimize costs.

- Point-of-Use (POU) and Point-of-Entry (POE) treatment equipment is an important option to consider, especially for the provision of safe drinking water from private wells. Unless connecting to a nearby public water system becomes an option, users relying on domestic wells

Adminsitrative Record
Page 35289

have two main alternatives: drilling a new well to attain safe drinking water or installing a POU or POE device for the treatment of contaminated water. The use of POU and POE treatment equipment by small public water systems is currently only a temporary option in California and reliance on these devices for the long-term would require regulation changes. While POU and POE treatment equipment has been shown to effectively address nitrate and other contaminants, it is important to properly maintain these devices to ensure the supply of consistently safe drinking water.

- Within the drinking water community, the options typically considered to address nitrate contamination are IX and RO. Alternative technologies are available or emerging (EDR, BD, CD) because, under some circumstances, they offer advantages over IX and RO. New technologies will continue to be investigated and developed because no single option is ideal for all situations. There is not a nitrate treatment option currently available that can affordably address all possible scenarios. The following diagram is a rough guide for treatment technology selection based on water quality concerns and possible priorities for a given water source or system (Table S.3). This diagram includes generalizations and is not intended to be definitive. In the selection of nitrate treatment technologies the unique needs of an individual water system must be assessed by professional engineers to optimize treatment selection and design.

Table S.3. Comparison of major treatment types.[1]

Concerns	IX	RO	EDR	BD	CD
High Nitrate Removal					
High TDS Removal					
Arsenic Removal					
Radium and Uranium Removal					
Chromium Removal					
Perchlorate Removal					

Priorities	IX	RO	EDR	BD	CD
High Hardness Not a Major Concern					
Reliability					
Training/ Ease of operation					
Minimize Capital Cost					
Minimize Ongoing O&M Cost					
Minimize Footprint					
Industry Experience					
Ease of Waste Management					

Good	→	Poor	Unknown (blank)

[1] Ion Exchange (IX), Reverse Osmosis (RO), Electrodialysis Reversal (EDR), Biological Denitrification (BD), Chemical Denitrification (CD). This table offers a generalized comparison and is not intended to be definitive. There are notable exceptions to the above classifications.

Adminsitrative Record
Page 35290

1 Introduction

Nitrate contamination of potable water sources is becoming one of the most important water quality concerns in California and across the United States. The maximum contaminant level (MCL), 45 mg/L as nitrate (NO_3^-) (10 mg/L as nitrogen (N)), is currently being approached or exceeded in potable water supply sources at locations throughout the United States (Nolan et al. 2002; Chen et al. 2009; U.S. EPA N.D.). A major source of nitrate contamination is fertilizer. Application of fertilizer in excess of the amount taken up by crops leads to leaching into the groundwater. Leakage from livestock feedlots and waste storage also contributes to the nitrate problem (LLNL 2002). Additional sources include wastewater treatment discharge, faulty septic systems, and various industrial applications. Due to the typical sources, nitrate contamination is more common in rural agricultural areas. The major health concern of nitrate exposure through drinking water is the risk of methemoglobinemia, especially in infants and pregnant women. Due to the nature of the infant digestive system, nitrate is reduced to nitrite which can render hemoglobin unable to carry oxygen (SWRCB 2010).

1.1 Management Options for Nitrate in Potable Water

To meet the nitrate MCL in the provision of potable water, both non-treatment and treatment options are considered. Source management with non-treatment can sometimes provide less costly solutions through wellhead protection, land use management, well abandonment, source modification, development of alternative sources (including consolidation or connection to a nearby system), or blending.[5] The feasibility of non-treatment options can be limited by various factors including location, budget, source availability, and variability of water quality (i.e., fluctuations in nitrate levels), resulting in the need for treatment to remove or reduce nitrate.

Current treatment methods include ion exchange (IX), reverse osmosis (RO), electrodialysis/ electrodialysis reversal (ED/EDR), biological denitrification (BD), and chemical denitrification (CD). These nitrate management options are examined in detail to assess research findings, capital and O&M costs, typical limitations, and the latest improvements. Design and cost considerations will be addressed with the development of guidelines for determining the most appropriate treatment option based on source water quality and other water system characteristics.

Point-of-Entry (POE) and Point-of-Use (POU) treatment equipment should also be considered as part of a comprehensive examination of nitrate treatment. The Safe Drinking Water Act (SDWA) [Section 1412(b)(4)(E)(ii)] (U.S. EPA 1998) identifies both POE and POU treatment units as options for compliance technologies for small systems; California regulations governing the use of POU and POE devices for water system compliance currently restrict their use to a temporary basis and only for systems having particular characteristics (Section 3.7).

[5] For the purposes of this discussion, blending has been categorized as a "non-treatment" option; however, in practice, blending is sometimes referred to as "treatment." Treatment options throughout this report refer to treatment technologies available for the removal or reduction of nitrate in drinking water. Blending can sometimes be used to cost-effectively address the nitrate problem through dilution, but has been categorized separately from treatment options for simplicity.

Adminsitrative Record
Page 35291

Lastly, hybrid systems are explored. The combination of multiple treatment technologies, including several developing brine treatment alternatives, can maximize the advantages of each option. The goal of this investigation is to provide an overview of management strategies and treatment options, highlighting the most recent advances and elucidating costs and common problems in application.

Adminsitrative Record
Page 35292

2 Non-Treatment Options for Nitrate Contaminated Potable Water

2.1 Well Abandonment, Inactivation, and Destruction

With adequate capacity from other sources, the simplest option for management of nitrate contaminated potable water sources is well abandonment and proper destruction. However, the lack of sufficient alternative water supplies often rules out well abandonment as an option. Based on a recent survey conducted by the American Water Works Association (AWWA), 30.4% (17/56) of survey participants with wells impacted by nitrate selected well abandonment as the implemented option for addressing nitrate contamination (Weir & Roberson 2010; Weir & Roberson 2011). It is important to determine the local requirements for safely removing a well from service. For proper abandonment, local requirements can include covering, sealing, and plugging of the well to prevent contamination and to avoid hazardous conditions. Inactivation or abandonment of a well differs from well destruction. Through inactivation or temporary abandonment, the well can be brought back online in the future (e.g., when treatment is installed). In contrast, well destruction involves the filling of a well, making it no longer viable. The costs for proper well abandonment and destruction can be substantial and vary with well depth, diameter, location, and local standards for well destruction. Analysis of public supply well abandonment and destruction in the Tulare Lake Basin and Salinas Valley, as well as across California, is included below in the Section 5.1 Well Abandonment, Destruction, and Inactivation. Additional information on private well abandonment and destruction is provided in Technical Report 2, Section 9 (Viers et al. 2012).

2.2 Wellhead Protection and Land Use Management

While limiting current nitrate contamination of groundwater will not immediately remove the need for treatment, over time, load reduction will minimize source water nitrate levels. Agricultural practices, management of dairies, control of wastewater treatment plant discharge, and monitoring and remediation of septic tank discharges can be improved to minimize nitrogen loading (for a full discussion of source load reduction see Technical Report 3, Dzurella et al. 2012). For example, a project addressing well head protection and land use management performed by the University of Waterloo (Rudolph 2010) successfully decreased groundwater nitrate levels within a two year travel time from 17 to 7 mg/L total stored nitrogen. Reduced nitrogen loading was accomplished by purchasing agricultural land and implementing Best Management Practices (BMPs).

2.3 Development of Alternative Sources and Source Modification

With adequate information about the nitrate distribution and movement in the subsurface, a new well can potentially be developed to access higher quality source water. Due to the anthropogenic nature of the contamination, nitrate concentration typically decreases with depth (Nolan et al. 2002). If suitable

Adminsitrative Record
Page 35293

water quality exists, drilling a deeper well can remove the need for nitrate treatment. However, the quality improvements must be balanced by a potential decrease in source capacity. Due to drilling and pumping requirements, capital and operational costs increase with the depth to uncontaminated water. When considering the installation of a deeper well to avoid nitrate contamination, it is important to be aware of the risk of encountering other water quality concerns at greater depths (e.g., arsenic) (see Section 4.2 Water Quality - Co-contaminants).

Connecting to a nearby water system that is not impacted by nitrate or to a larger system that can afford nitrate treatment is often the best option for smaller systems. For example, since 1995, the City of Modesto, CA, has been in charge of providing compliant water to the residents of Grayson, using an ion exchange plant for nitrate removal (Scott 2010). Similarly, consolidation of multiple nearby small systems can decrease the cost of treatment per customer to more reasonable levels. Additional alternative source options include purchasing water rights, trucking in potable water, or temporarily relying on bottled water. Reliance on hauled and/or bottled water is only an interim solution for use in emergencies or while an effective compliance option can be implemented. Technical Report 7 provides a comprehensive discussion of alternative water supply options and associated costs (Honeycutt et al. 2012).

Modification of impacted source wells can allow for withdrawal of water with lower nitrate levels by limiting screened intervals to regions of better water quality. Down hole remediation requires characterization of the water quality profile to determine the screening depth range of the higher water quality. Specialized monitoring equipment and techniques are available that can be used without removing pumps (BESST Inc. 2008). With water profile characterization, existing wells can be selectively screened using a packer/plug to limit withdrawal from unwanted regions (Figure 1). The effective application of such well modification techniques is dependent on the subsurface characteristics in the vicinity of the well. Vertical migration of nitrate through the surrounding porous media can lead to increasing nitrate levels in the water withdrawn from the well.

Adminsitrative Record
Page 35294

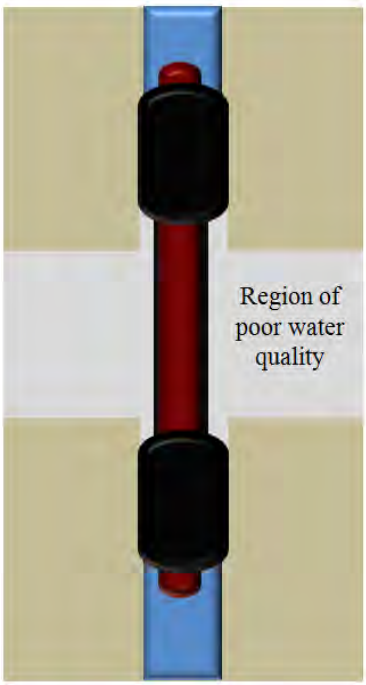

Figure 1. Selective well screening using a packer/plug.

The City of Ceres, CA, is in the process of drilling new wells, in part to avoid the need for nitrate treatment; well modification has also been implemented to avoid water with high nitrate levels (Cannella 2009).

2.4 Blending

The dilution of a nitrate impacted source with an alternate low nitrate source can be a cost-effective option to produce compliant water; this is known as blending and can be applied independently or with treatment.[6] Blending is a common practice for the production of compliant water, but relies on the availability of a low nitrate source and the consistency of nitrate levels to avoid exceedances. High nitrate groundwater can also be blended with surface water when a surface water source is available; however, surface water treatment requirements would increase costs. One drawback of implementing blending to address nitrate contamination is that reliance on blending can limit operational flexibility. If the source used for dilution were compromised, then production would need to be stopped from both wells. Water can also be trucked in for blending purposes when a low nitrate source is unavailable locally; however, hauling water for blending purposes is a temporary solution. Based on the recent AWWA survey, 51.8% (29/56) of respondents with nitrate impacted sources selected blending as the option to address nitrate contamination (Weir & Roberson 2010; Weir & Roberson 2011). Likewise,

[6] For the purposes of this discussion, blending has been categorized as a "non-treatment" option; however, in practice, blending is sometimes referred to as "treatment." Treatment options throughout this report refer to treatment technologies available for the removal or reduction of nitrate in drinking water. Blending can sometimes be used to cost-effectively address the nitrate problem through dilution, but has been categorized separately from treatment options for simplicity.

Adminsitrative Record
Page 35295

nitrate contamination of drinking water in Germany is often addressed by blending, avoiding the costs of treatment (Dördelmann 2009). When feasible, blending is a simple alternative to treatment that avoids disposal concerns and the certification requirements of treatment (WA DOH 2005). However, disadvantages include the capital investment for accessing an alternative source and monitoring requirements to ensure consistent supply of compliant water (WA DOH 2005). Analysis of water systems utilizing blending to address nitrate contamination in the Tulare Lake Basin and Salinas Valley, as well as across California, is included below in the Section 5.2 Survey of Blending and Treating Systems.

Adminsitrative Record
Page 35296

3 Treatment Options for Nitrate Contaminated Potable Water

IX, RO, and ED/EDR transfer nitrate ions from water to a concentrated waste stream that requires disposal. The United States Environmental Protection Agency (U.S. EPA) lists these three processes as accepted potable water treatment methods for nitrate removal (U.S. EPA 2010). In contrast, through biological and chemical denitrification, nitrate is converted to reduced nitrogen species, rather than displaced to a concentrated waste stream that requires disposal.

A survey of nitrate treatment systems was conducted to assess the current state of nitrate treatment. The list of surveyed utilities was developed with the intention of covering a range of utilities with respect to geographic location, treatment type, population size, and residual handling techniques (Table 1). Detailed case studies have been compiled for each of the treatment technologies where full-scale facilities have been in operation or are moving ahead with design. This survey was conducted through collaboration with Jacobs Engineering in the completion of the associated assessment of nitrate treatment alternatives for AWWA and is complemented by a parallel survey of nitrate treatment systems in California. Details from the initial survey are included as examples following a discussion of each of the treatment technologies. Details of the complementary survey of California systems are included in the second half of this report.

A brief comparison of fundamental design considerations, and advantages and disadvantages of the treatment options examined herein is listed in Table 2. It is important to note that the contents of Table 2 are not intended to provide a comprehensive set of criteria for treatment options. Other important criteria in determining the best treatment option, which are site specific and cannot be broadly generalized, include capital and O&M costs, system size (capacity), and system footprint.

IX is the most commonly used nitrate treatment method, with full-scale systems in use throughout the United States. Full-scale application of biological denitrification in potable water treatment is mainly limited to Europe and chemical denitrification methods have been investigated only at the pilot-scale. Others have provided thorough reviews of available nitrate treatment technologies (Kapoor & Viraraghavan 1997; Soares 2000; Shrimali & Singh 2001); however, a recent comprehensive review of the state of nitrate treatment is absent from the literature.

Adminsitrative Record
Page 35297

Table 1. Utilities included in the case studies.

Case #	Treatment Type	Location	Capacity (gpm)	Avg. Influent Nitrate mg/L as NO_3^- (mg/L as N)
	Ion Exchange			
1	Conventional ion exchange with blending	California	400	31 – 53 (7 – 12)
2	Conventional ion exchange with blending	California	400	~45 (~10)
3	Counter Current Ion Exchange (MIEX®)	Indian Hills, CO	50	53 – 71 (12 – 16)
4	Multiple vessel ion exchange	California	500 – 900	35 – 89 (8 – 20)
5	Multiple vessel ion exchange	Chino, CA	5000	40 – 200 (9 – 45)
	Reverse Osmosis			
6	Reverse osmosis and blending	Bakersfield, CA	120	75 – 84 (17 – 19)
7	Reverse osmosis, exploring biological reduction	Brighton, CO	4600	49 – 89 (11 – 20)
8	Reverse osmosis and blending	Arlington Desalter, Riverside, CA	4583	44 – 89 (10 – 20)
	Combined Reverse Osmosis and Ion Exchange			
9	Reverse osmosis, ion exchange and blending	Chino Desalter I, Chino, CA	4940 (RO), 3400 (IX)	148 – 303 (33 – 68)
10	Reverse osmosis, ion exchange and blending	Chino Desalter II, Mira Loma, CA	4167 (RO), 2778 (IX)	70 – 224 (16 – 51)
	Electrodialysis/Electrodialysis Reversal/Selective Electrodialysis			
11	Electrodialysis Reversal (EDR)	Spain	3,260 (each, 2 systems)	~80 (~18)
12	Selective Electrodialysis (SED)	Israel	310	84 – 89 (19 – 20)
	Biological Denitrification			
13	Implementing fluidized bed biological reduction	Rialto, CA	2000 – 4000	17 – 19 (~4 – 5)
14	Implementing fixed bed biological reduction	Riverside, CA	1670	44 – 89 (10 – 20)

Adminsitrative Record
Page 35298

Table 2. Potable water treatment options for nitrate management (adapted from WA DOH 2005).

	Ion Exchange	Reverse Osmosis	Electrodialysis	Biological Denitrification	Chemical Denitrification
Full-scale Systems	Yes	Yes	Yes	Yes	No
Treatment Type	Removal to waste stream	Removal to waste stream	Removal to waste stream	Biological reduction	Chemical reduction
Common Water Quality Design Considerations	Sulfate, iron, manganese, total suspended solids (TSS), metals (e.g., arsenic), hardness, organic matter	Turbidity, iron, manganese, SDI, particle size, TSS, hardness, organic matter, metals (e.g., arsenic)	Turbidity, iron, manganese, TSS, hydrogen sulfide, hardness, metals (e.g., arsenic)	Temperature and pH, anoxic conditions	Temperature and pH
Pretreatment Needs	Pre-filter, address hardness	Pre-filter, address hardness	Pre-filter, address hardness	pH adjustment, nutrient and substrate addition, need for anoxic conditions	pH adjustment
Post-treatment Needs	pH adjustment	pH adjustment Remineralization	pH adjustment Remineralization	Filtration, disinfection, possible substrate adsorption	pH adjustment, iron removal, potential ammonia control
Waste/Residuals Management	Waste brine	Concentrate	Concentrate	Sludge/biosolids	Waste media, Iron sludge
Start-up Time	Minutes	Minutes	Minutes	Initial plant startup: Days to weeks After reaching steady state: Minutes	Minutes
Water Recovery	Conventional (97%) Low brine (Up to 99.9%)	Up to 85%	Up to 95%	Nearly 100%	Not demonstrated full-scale
Advantages	Nitrate selective resins, common application, multiple contaminant removal	Multiple contaminant removal, desalination (TDS removal)	Multiple contaminant removal, higher water recovery (less waste), desalination, unaffected by silica	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, high water recovery, and potential for multiple contaminant removal	No waste brine or concentrate, nitrate reduction rather than transfer to a waste stream, and potential for multiple contaminant removal
Disadvantages	Potential for nitrate peaking, high chemical use (salt), brine waste disposal, potential for disinfection byproduct (DBP) formation (e.g., NDMA)	Membrane fouling and scaling, lower water recovery, operational complexity, energy demands, waste disposal	Energy demands, operational complexity, waste disposal	Substrate addition, potentially more complex, high monitoring needs, possible sensitivity to environmental conditions, risk of nitrite formation (potential incomplete denitrification), post-treatment to address turbidity standards and 4-log virus removal (state dependent)	Inconsistency of nitrate reduction, risk of nitrite formation (potential incomplete denitrification), reduction to ammonia, lack of full-scale systems, pH and temperature dependence, possible need for iron removal

Adminsitrative Record
Page 35299

3.1 Ion Exchange (IX)

As the most commonly used method for the removal of nitrate in potable water treatment, IX has been widely researched, with numerous full-scale installations in operation. With the potential for multiple contaminant removal, IX can also be used to address other water quality concerns including arsenic, perchlorate, selenium, chromium (total and chromium-6), and uranium (AWWA 1990; Boodoo 2004). Selected IX installations used for nitrate treatment in the United States are listed in Table 3.

Table 3. Selection of full-scale ion exchange installations for nitrate removal.

Locations	Year Installed	Influent nitrate (mg/L as NO_3^-)	Capacity (MGD)	Reference
Ellsworth, MN	1994	-	0.047	MN Dept. of Ag. (N.D.)
Clear Lake, MN	1995	-	0.047	MN Dept. of Ag. (N.D.)
Adrian, MN	1998	-	0.129	MN Dept. of Ag. (N.D.)
Edgerton, MN	2002	-	0.137	MN Dept. of Ag. (N.D.)
McCook, NE	2006	Up to 125	6.8	Contaminant Removal News (2007)
McFarland, CA, Well 2	1983	60	1	Guter (1995), See also Guter (1982)
McFarland, CA, Well 4	1987	64	1	Guter (1995), See also Guter (1982)
La Crescenta, CA	1987	70 – 100	2.7	Guter (1995)
Grover City, CA, 3 wells	-	80 – 130	2.3	Guter (1995)
Des Moines, IA	1992	Up to 55	10	Des Moines Water Works, Rash (1992)
Glendale, AZ: full-scale pilot	2010	Spiked up to 177	10	Meyer et al. (2010), See also Clifford et al. (1987)
Indian Hills, CO	2009	53 – 71	0.072 (design)	See Case Study

Due to its common application, the investigation of IX for the removal of nitrate is prevalent in the literature (Yoon et al. 2001; Chabani et al. 2006; Samatya et al. 2006; Clifford 2007; Meyer et al. 2010; Clifford et al. 2010). Kapoor & Viraraghavan (1997) provide an extensive review of IX research up to 1997. Modifications of conventional IX have led to the emergence of more efficient IX processes including multiple vessel configurations, counter current configurations, the use of specialized resins, improved hydraulics, and weak base anion exchange (WBA IX).

3.1.1 Conventional Ion Exchange

Conventional IX utilizes a strong base anion (SBA) exchange resin. In accordance with Figure 2, raw water passes through pretreatment to remove suspended solids and to address other constituents capable of fouling the resin. The nitrate laden treatment stream then enters the ion exchange vessel. Upon contacting the resin, nitrate displaces chloride at surface sites, removing nitrate from the water.

Adminsitrative Record
Page 35300

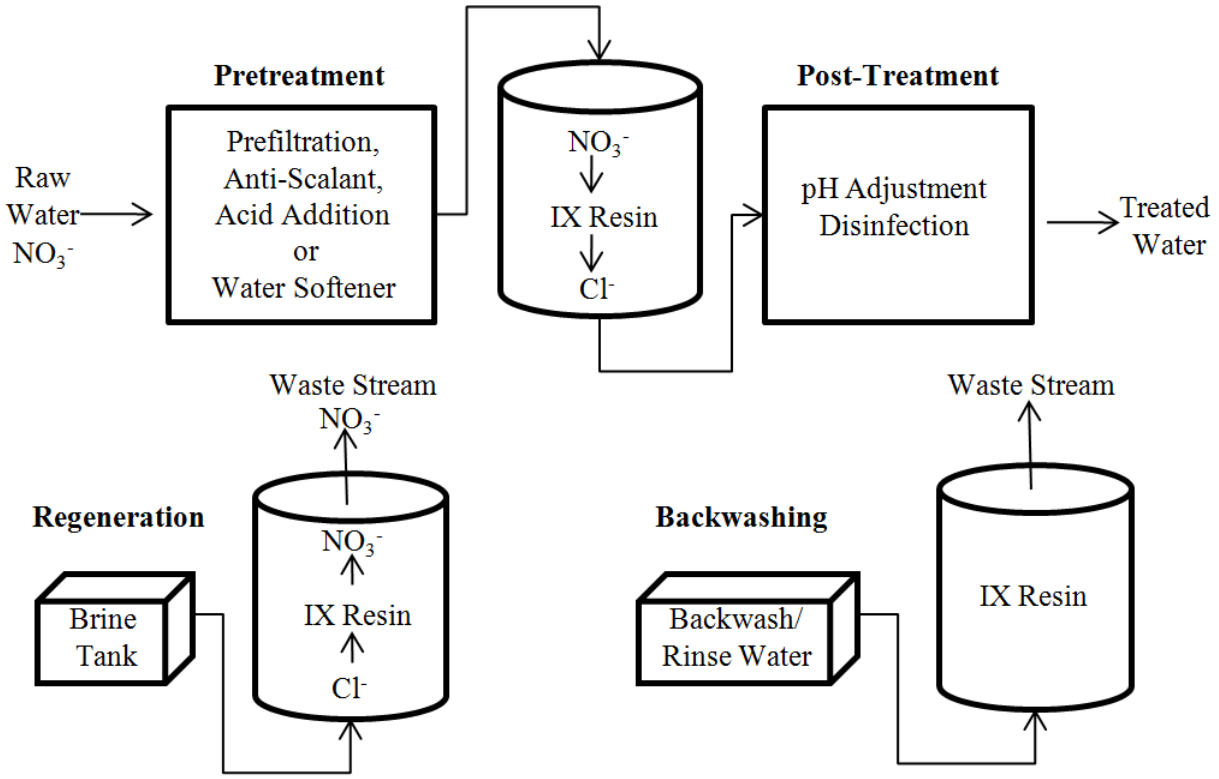

Figure 2. Conventional ion exchange schematic.

This technique is similar to a water softener, which replaces the divalent cations of hard water (Mg^{2+} and Ca^{2+}) with a monovalent cation (Na^+). Eqn. 1 depicts the transfer of ions, with R as the resin surface site. Leaving nitrate behind, treated water exits the ion exchange vessel and passes on to post-treatment for stabilization and disinfection.

$$R\text{-}Cl + NO_3^- \rightarrow R\text{-}NO_3 + Cl^- \qquad \text{(Eqn. 1)}$$

To prevent nitrate breakthrough, regeneration is necessary when the resin is exhausted of chloride ions (chloride has been displaced at the majority of surface sites). The media is backwashed with a high salt solution (0.5 – 3 M, Clifford 2007) to reverse the process, resulting in a brine waste stream high in nitrate and other concentrated ions (Eqn. 2).

$$R\text{-}NO_3 + Cl^- \rightarrow R\text{-}Cl + NO_3^- \qquad \text{(Eqn. 2)}$$

The relative affinity of common anions for conventional anion exchange resin is $SO_4^{2-} > NO_3^- > Cl^- > HCO_3^-$ (Bae et al. 2002; Clifford et al. 2010). If generic resins are not regenerated soon enough, sulfate displacement of nitrate in the resin can lead to nitrate release from the resin to the treatment stream (Eqn. 3). This is known as nitrate dumping, nitrate peaking, or chromatographic peaking and is further discussed below.

$$2R\text{-}NO_3 + SO_4^{-2} \rightarrow R_2\text{-}SO_4 + 2NO_3^- \qquad \text{(Eqn. 3)}$$

Adminsitrative Record
Page 35301

Due to the stronger affinity of the sulfate ion for generic anion exchange resins, nitrate selective resins have been developed for which the order of affinity is NO_3^- > SO_4^{2-} > Cl^- > HCO_3^- (Guter 1982; Guter 1995). Important factors in resin selection are the exchange capacity[7] and selectivity coefficient[8] of the resin and the rate of ion transfer (kinetics[9]).

Detailed case studies of conventional IX plants are included in Section 3.1.7 Ion Exchange - Case Studies.

3.1.2 Ion Exchange - Design Considerations

Various tools are available to assist with IX system design including Dow's CADIX (Computer Assisted Design for Ion Exchange) (Dow 2010b) and Lenntech's Ion Exchange calculator (Lenntech 2009b.) Table 4 summarizes key design considerations in the application of conventional IX to nitrate removal from potable water.

[7] Exchange capacity: The exchange capacity is a measure of how many ions the resin can capture per unit volume.
[8] Selectivity coefficient: The selectivity coefficient of a resin refers to the relative affinity of resin surface sites for a particular ion, in this case nitrate.
[9] Kinetics: The term kinetics refers to the rate of a reaction. The rate that nitrate displaces chloride on the resin is important for efficient treatment and can be affected by competing ions.

Adminsitrative Record
Page 35302

Table 4. Summary of design considerations for conventional IX.

Resin Selection	• Generic SBA resins for maximum exchange capacity (for low sulfate)
	  ○ Less expensive than nitrate selective resins
	  ○ Less frequent regeneration due to higher capacity (in the absence of co-contaminants)
	  ○ Nitrate dumping potential
	• Nitrate selective resins to avoid nitrate dumping (for high sulfate)
	  ○ More expensive than generic resins
	  ○ Longer bed life
	  ○ More nitrate removed per unit of waste brine

Pretreatment	• Filtration to remove iron, manganese, TSS, and organic matter to prevent resin fouling • Water softening (anti-scalant, acid, or water softener) to prevent scaling • Dechlorination to prevent resin oxidation[10]
Post-Treatment	• Chloride : alkalinity ratio and dezincification[11] • Chloride : sulfate ratio and galvanic corrosion[12] • Potential pH adjustment and restoration of buffering capacity to avoid corrosion
Chemical Usage	• pH adjustment (caustic soda or soda ash) • Regenerant brine, salt consumption
O&M	• Frequency of regeneration depends on water quality and resin type • Fresh brine preparation and waste disposal • Resin loss and replacement: 3 – 8 year lifetime (WA DOH 2005; Dow 2010c) • Continuous or frequent monitoring of nitrate levels • Backwashing to dislodge solids

System Components	• Fixed bed versus continuous regeneration
	• Key system configuration parameters are system flow rate, bed swelling, bed depth, backwash flow rate, and rinse requirements
	  ○ Vessels in parallel or in-series
	  ○ Co-current or counter-current regeneration

Waste Management and Disposal	• Significant cost of waste brine disposal is of greatest concern for inland systems • Close proximity to coastal waters is beneficial for brine disposal • Management options can include sewer or septic system, drying beds, trucking off-site, coastal pipeline, deep well injection, and advanced treatment • Disposal options can be limited by waste brine water quality (e.g., volume, salinity, metals, and radionuclides) • Optimization of recycling and treatment of waste brine is desirable
Limitations	• Need to manage resin fouling ○ Hardness, iron, manganese, suspended solids, organic matter, and chlorine • Competing ions (especially sulfate) • Disposal of waste brine • Possible role of resin residuals in DBP formation

[10] The resin can be degraded by oxidation; the functional amine groups on the resin surface are susceptible to oxidation which can lead to diminished capacity (Dow 2010d).

[11] As nitrate and other anions displace chloride on the resin, chloride is released to the product water, leading to the potential for taste issues and dezincification (Kapoor & Viraraghavan 1997). Dezincification refers to the ability of product water to dissolve zinc from brass and is dependent on the ratio of chloride to alkalinity (> 0.5 can be problematic). By restoring alkalinity, the dezincification potential can be minimized.

[12] Galvanic corrosion can result in the release of lead from brass and galvanized solder-copper connections and is associated with a high ratio of chloride to sulfate (> 0.58 can be problematic) (Edwards et al. 1999; Edwards & Triantafyllidou 2007).

Adminsitrative Record
Page 35303

Water Quality

Raw water quality is a key factor in the efficiency of an IX system, impacting resin selection, pretreatment and post-treatment needs, regeneration efficiency, chemical usage, and waste disposal. Important water quality parameters include alkalinity, hardness, iron, manganese, and potential competing ions (predominantly sulfate).

Selection of the appropriate resin for a given system depends directly on source water quality. In the presence of high levels of co-contaminants, nitrate selective resins may be necessary rather than generic resins. Both strong base anion exchange (SBA) resin and weak base anion exchange (WBA) resin can be suitable for nitrate removal from potable water. The latter will be addressed separately. The two standard types of SBA resins deviate in their functional groups. Anion exchange is dependent on the trimethylamine groups of the SBA Type I resin and the dimethylethanolamine groups of the SBA Type II resin (Helfferich 1995).

Nitrate selective resin was invented in the early 1980's by Gerald Guter (Guter 1982, see related patent: Guter 1983). Clifford & Weber (1978 and 1983) contributed to the development and characterization of these resins with their research on resin selectivity (Clifford et al. 2010). Nitrate selective resins rely on different functional groups than Type I and Type II SBA resins. Nitrate selectivity is attributed to the increased hydrophobicity and site spacing of exchange sites due to the triethyl, tripropyl, and tributyl functional groups of nitrate selective resin (Clifford & Weber 1978; Guter 1982; Clifford et al. 2010). The use of nitrate selective resin avoids the problem of nitrate peaking (i.e., nitrate dumping or chromatographic peaking), typically caused by the greater affinity of generic resins for sulfate.

It is important to note the distinction between nitrate peaking and nitrate breakthrough. As nitrate displaces chloride on the resin, the nitrate capacity of the resin is gradually exhausted leading to increasing effluent nitrate levels until influent levels are reached. This is known as breakthrough and can occur regardless of resin type. Nitrate peaking can also occur upon exhaustion of the resin capacity for nitrate. However, with nitrate peaking, the nitrate on the resin is displaced by sulfate, thereby increasing the effluent nitrate concentration to levels above that of the raw water (Clifford et al. 2010). The peak in nitrate concentration is due to the combination of the influent nitrate ions and the nitrate ions that are coming off of the resin via sulfate displacement.

Under low sulfate conditions, the use of generic SBA resins is preferred, due to their larger exchange capacity. As the ratio of sulfate to nitrate increases, the use of nitrate selective resins avoids possible nitrate peaking, minimizing the risk of MCL exceedance and the need for more frequent regeneration. However, with the highest nitrate selectivity, regeneration efficiency can decrease, increasing chemical use. With a stronger affinity for nitrate, the removal of nitrate from the resin in regeneration is more difficult because nitrate is more strongly bound (Dow 2010c). Nitrate selective resins are more costly than the generic option (Water Quality Products 2003), but under appropriate conditions the use of regenerant can be minimized and bed life can be increased significantly with their use. Different system configurations have been implemented as an alternative to the use of nitrate selective resins to address the problem of co-contaminants and the risk of nitrate peaking (Clifford et al. 2010), for more

Adminsitrative Record
Page 35304

information, see System Layout and Site Considerations, below. In the consideration of IX for multiple contaminant removal, the selection of the most appropriate resin depends in part on the type and concentration of co-contaminants. Modeling and column or pilot testing is important to determine appropriate design parameters and to design treatment based on a full life cycle analysis of costs.

Upstream of the IX vessels, pretreatment of the source water may be necessary to avoid resin fouling (Water Quality Products 2003; WA DOH 2005). Potential constituents of concern include organic matter, turbidity, total suspended solids (TSS), sand and metals, primarily iron and manganese (Kapoor & Viraraghavan 1997; Water Quality Products 2003; WA DOH 2005). Pre-filtration is typically used to remove these constituents; however, additional pretreatment methods may be used. For example, coagulation/flocculation and filtration may be necessary for surface waters. Pretreatment may be needed for waters with a total concentration of metals (e.g., iron and manganese) above 0.1 mg/L (Ten States Standards 2007). Hard, alkaline water can lead to resin scaling, due to calcium and magnesium accumulation; pH adjustment or water softening can be used to prevent resin scaling (Water Quality Products 2003). Created by Wolfgang Holl, the carbon dioxide regeneration of ion exchange (CARIX) process combines anion and cation exchange (Holl 1995). The CARIX process enables simultaneous removal of cations (for hardness reduction) and anions through the combination of a weak acid cation exchanger and a strong base anion exchanger.

Resin exposure to disinfectants (chlorine and chloramines) should be avoided to prevent resin oxidation and the possible release of disinfection byproducts, specifically nitrosamines (Kemper et al. 2009). Disinfection byproducts (DBPs) are potentially cancer-causing compounds that can be formed through a reaction of disinfection chemicals like chlorine and chloramines with organic matter. Due to the amine functional groups of anion exchange resins, "these resins may serve as precursors for nitrogenous disinfection byproducts, such as nitrosamines, nitramines, and halonitromethanes" (Kemper et al. 2009, p. 466). Recent research suggests that the risk of DBP formation is higher with the use of new resin; however, precursors can be a problem with downstream chloramine use or with upstream disinfection (see Kemper et al. 2009 in Table A.1 of the Appendix). Magnetic ion exchange resin is an exception as its primary purpose is to remove organic carbon and limit DBP formation (Boyer & Singer 2006) (see the MIEX® process in Section 3.1.7 Ion Exchange - Case Studies).

IX can reduce alkalinity and pH due to removal of bicarbonate. To prevent corrosion in downstream pipes, the product water pH and buffering capacity may need to be increased.[13] Soda ash can be added to the regenerant brine to load a portion of the resin sites with bicarbonate rather than chloride. This can restore some alkalinity to the water as bicarbonate is released from the resin when displaced by nitrate and other anions in the treatment stream (Water Quality Products 2003). To minimize the need for caustic addition in post-treatment, an upstream atmospheric degasifier for carbon dioxide removal can be added during pretreatment (Dow 2010).

[13] It is important to note the relationship between alkalinity and pH. Alkalinity is a measure of buffering capacity or the resistance to changes in pH. Demineralized water or water with a low buffering capacity will be susceptible to more dramatic pH changes and is considered unstable. The pH of acidic product water should be adjusted and the buffering capacity of demineralized product water should be restored to avoid corrosion downstream and the potential for lead and copper challenges.

Adminsitrative Record
Page 35305

As nitrate and other anions displace chloride on the resin, chloride is released to the product water, leading to the potential for taste issues, dezincification, and galvanic corrosion. Dezincification refers to the ability of product water to dissolve zinc from brass and is dependent on the ratio of chloride to alkalinity (as $CaCO_3$)(> 0.5 can be problematic, according to Kapoor & Viraraghavan (1997)). By restoring alkalinity, the dezincification potential can be minimized. Galvanic corrosion can result in the release of lead from brass and galvanized solder-copper connections and is associated with a high ratio of chloride to sulfate (> 0.58 can be problematic) (Edwards et al. 1999; Edwards & Triantafyllidou 2007). It is important to consider the potential downstream impact of subtle water quality changes caused by treatment.

System Layout and Site Considerations

The IX system can be operated using a fixed bed or as a continuous system. Details of modifications to conventional fixed bed systems are provided below. Key parameters in vessel sizing and system configuration are system flow rate, bed swelling, bed depth, backwash flow rate, and rinse requirements (Dow 2010). Regeneration can be designed in a co-current or counter-current configuration. Vessels can be operated in parallel or in-series for redundancy, to maximize removal efficiency per regeneration cycle, to address nitrate peaking and to consistently produce water with limited variation in water quality parameters (Clifford et al. 2010).

Residuals Management and Disposal

Management of waste brine can be costly. Options include discharge to a sewer or septic system, waste volume reduction using drying beds, trucking to an off-site approved disposal location, ocean discharge through a coastal pipeline, deep well injection, and advanced treatment. Water quality characteristics of the waste brine (e.g., volume, salinity, metals, and radionuclides) can affect the feasibility and costs of disposal options. Proximity and access to coastal waters can be a significant factor in determining the burden of brine disposal. Generally, disposal is of greatest concern to inland communities. Although other removal technologies (RO and ED) also require concentrate disposal, this is an issue of particular concern with IX. Because IX requires the addition of salt for resin regeneration, the waste stream consists of not only the nitrate and other ions that have been removed from the water, but also the spent brine solution used in regeneration. The high cost of nitrate laden brine disposal has led to research into optimization of recycling and treatment of this waste stream. Partial regeneration and reuse of treated brine for multiple regeneration cycles can minimize the volume of waste while maximizing regeneration efficiency (Clifford 2007; Clifford et al. 2010). Application of IX systems coupled with biological, chemical, or catalytic denitrification enables removal of nitrate from waste brine, with reduction to nitrogen gas. The electrochemical destruction of nitrate in waste brine is also being explored. Several combined configurations of interest are discussed below in Section 3.6 Brine Treatment Alternatives and Hybrid Treatment Systems.

Adminsitrative Record
Page 35306

Maintenance, Monitoring, and Operational Complexity

Regeneration frequency will depend on pretreatment measures, water quality, and the type of resin used (WA DOH 2005). The typical amount of brine waste compared to water produced can range from ~3.0% for conventional systems (Clifford 2007) to ~0.5% for low brine systems (Calgon Carbon Corporation 2007, discussed below). A constant supply of brine must be available for resin regeneration and waste brine requires appropriate storage and disposal. Resin loss can be controlled by adjusting the backwash flow rate and adding screens (Keller 2000). Resin life will also depend on water quality and pretreatment measures; resin replacement may be required after 3 – 8 years (WA DOH 2005; Dow 2010c). To ensure the production of compliant water, continuous or frequent monitoring of nitrate levels is necessary. In addition to resin regeneration, backwashing is used to dislodge solids and "resin fines" (Dow 2010). In comparison with alternative treatment options, IX requires limited O&M with high feasibility of automation and low operational complexity.

3.1.3 Ion Exchange - Cost Considerations

For optimal operation of an IX system, the fundamental objective is to maximize regeneration efficiency, while meeting necessary potable water guidelines. Factors affecting system cost include facility size (flow rate), source water quality (including nitrate concentration), environmental factors (temperature), and target effluent nitrate concentration. Disposal of waste brine is commonly a significant portion of O&M costs.

Capital costs for IX include land, housing, piping, storage tanks, O&M equipment, vessels, resin, preliminary testing (pilot studies), permits, and training. O&M costs include resin replacement (due to loss or degradation), resin disposal, brine disposal or treatment, chemical use (salt, anti-scalant, pH adjustment), repair and maintenance, power, and labor.

Published cost information from existing IX installations is listed in Table 5. Costs have been adjusted to 2010 dollars, unless indicated otherwise. Costs can be difficult to assess due to inconsistencies in how cost information is reported. Comparison of IX costs is not always valid due to differences in influent water quality parameters, system size, waste management options, and system configuration. Published costs do not always include comparable information. It would be inappropriate to compare the O&M costs of a facility that excludes disposal costs with others that include this information. The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing IX may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the questionnaire includes detailed costs associated with the unique case studies included in this analysis. A detailed discussion of treatment costs is included below in Section 6 Treatment Cost Analysis.

Adminsitrative Record
Page 35307

Table 5. Selected published costs* of ion exchange systems for nitrate removal.

System Flow**	< 0.5 MGD	0.5 – 5 MGD	5+ MGD
Annualized Capital Cost ($/1000 gal)	0.37 – 1.21 [1]	0.28 – 0.94 [2, 3]	0.28 – 0.61 [3, 4, 5]
O&M Cost ($/1000 gal)	0.60 – 4.65 [1]	0.46 – 1.25 [2, 3]	0.37 – 0.87 [3, 4, 5]
Total Annualized Cost ($/1000 gal)	0.97 – 5.71 [1]	0.74 – 2.19 [2, 3]	0.65 – 1.44 [3, 4, 5]

*Costs have been adjusted to 2010 dollars with 7% interest over 20 years, unless indicated otherwise (below).
**When available, costs are based on actual system flow rather than design capacity.
[1] Minnesota Department of Agriculture (N.D.), not adjusted to 2010 dollars, 20 year amortization without interest. [2] Guter (1995). [3] Conlon et al. (1995). [4] Meyer et al. (2010). [5] Drewry (2010).

3.1.4 Ion Exchange - Selected Research

A large body of research has focused on IX. Table A.1 of the Appendix is a list of recent studies relevant to nitrate removal from potable water. Given the many years of successful full-scale operating experience with IX, current applied research is focused on brine recycling efficiency, the optimization of waste management, and improvements in resin capacity and selectivity, to improve efficiency and reduce costs.

3.1.5 Ion Exchange - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of IX in comparison with other treatment options is listed in Table A.6 of the Appendix. Significant advantages of IX include years of industry experience, multiple contaminant removal, selective nitrate removal, financial feasibility, use in small and large systems, and the ability to automate. Disadvantages include the costly disposal of waste brine, the potential for nitrate dumping and resin fouling, the possible need for pH adjustment, the potential for hazardous waste generation (i.e., brine with traces of co-contaminants like arsenic and chromium), and the possible role of resin residuals in DBP formation (Kemper et al. 2009).

3.1.6 Modifications to Conventional Ion Exchange

Modifications of conventional IX have led to the emergence of low-brine IX processes including magnetic ion exchange (MIEX®), ion exchange separation (ISEP®), Envirogen (formerly Basin Water) systems, and weak base ion exchange (WBA IX). Despite their potential advantages, it is important to note that proprietary technologies may have inherent disadvantages, such as a lack of flexibility to use better technology when it becomes available, vulnerability if the manufacturer goes out of business or discontinues supporting the product, and a lack of competition to keep O&M costs down.

Counter Current Flow with Specialized Resin

Magnetic ion exchange technology (MIEX®), developed by Orica Watercare, offers a low brine alternative to conventional IX, using a unique SBA Type I resin. The MIEX® process (Figure 3) differs from

Adminsitrative Record
Page 35308

conventional IX in that the resin is fluidized in a contactor with spent resin removed from the contactor for regeneration outside of the process water stream and then returned to the contactor. This is in contrast to conventional IX, where the resin is stationary. Unlike conventional IX, the MIEX® fluidized bed process is tolerant of suspended solids and low levels of oxidants.

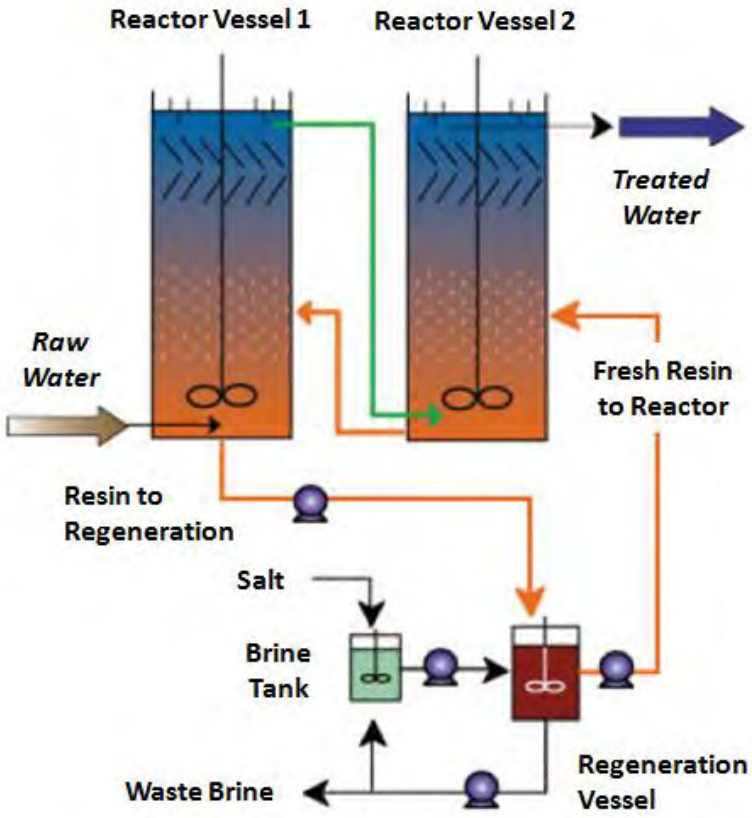

Figure 3. Process flow diagram for counter current MIEX® process. (Source: reprinted with permission, Orica Watercare 2008b.)

The minimization of waste brine is accomplished through frequent batch regeneration. The magnetized resin encourages agglomeration of loaded resin particles, resulting in faster settling. Loaded resin is removed from the bottom of the IX vessel and is passed to regeneration, while regenerated resin is added at the top of the IX vessel. This configuration removes the risk of nitrate peaking because clean resin, added at the end point of the system, captures any displaced nitrate, while competing ions, such as sulfate can be removed early on in the resin vessel. The MIEX® process has been proven to effectively address various water quality concerns including nitrate, organic carbon, arsenic, chromium-6 and perchlorate (Seidel et al. 2004; Humbert et al. 2005; Boyer & Singer 2006; and Watercare 2008).

A detailed case study of a MIEX® treatment plant in Indian Hills, CO is included in Section 3.1.7 Ion Exchange - Case Studies.

Adminsitrative Record
Page 35309

Improved Hydraulics and Nitrate Selective Resins

The Layne Christensen Company and Rohm and Haas offer their Advanced Amberpack® system which utilizes nitrate selective resins and their patented Fractal Distribution Technology to increase the treated water volume and decrease the waste brine. Nitrate selective resins have a greater affinity for nitrate than non-selective resins, resulting in an improvement in removal efficiency, especially in waters where the sulfate to nitrate ratio is greater than one.

The distributor system is designed to provide uniform flow through the ion exchange vessels in both treatment and regeneration modes. As a result of the uniform flow, the exchange capacity of the resin can be maximized while the brine and rinse values can be minimized, thus increasing the water efficiency of the system (Rohm and Haas Company 2007).

Multiple Vessel Carousel Configuration

Calgon Carbon's Continuous Ion Exchange Separation System (ISEP® System) utilizes a carousel configuration. This configuration has the potential to avoid downtime for regeneration, minimize the amount of resin needed, and maximize regeneration efficiency. Illustrated in Figure 4, multiple resin vessels are rotated from active treatment to resin regeneration and rinsing and back to active treatment. The vessels rotate in the opposite direction of the water movement (Figure 5). The counter-current and counter-flow system can provide consistent product water, operating uninterrupted with up-flow regeneration and down-flow treatment. This configuration results in four zones within the system (Calgon Carbon Corporation ISEP® for Nitrate Removal Brochure 2003). In the Adsorption Zone, the feed stream is passed through 14 ports in parallel for single pass flow. Nitrate and other anions are removed from the feed water as they transfer to the resin. In the Displacement Zone, softened water is used to displace the hard feed water to prevent scale build-up in the regeneration zone. In the Regeneration Zone, a combination of brine and rinse is directed through the beds for a true counter-current regeneration to maximize the regeneration efficiency. In the Rinse Zone, a small amount of softened feed water is used to prevent any of the salt from the regeneration zone from reaching the product water.

A detailed case study of an ISEP® treatment plant in Chino, CA is included in Section 3.1.7 Ion Exchange - Case Studies

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Adminsitrative Record
Page 35310

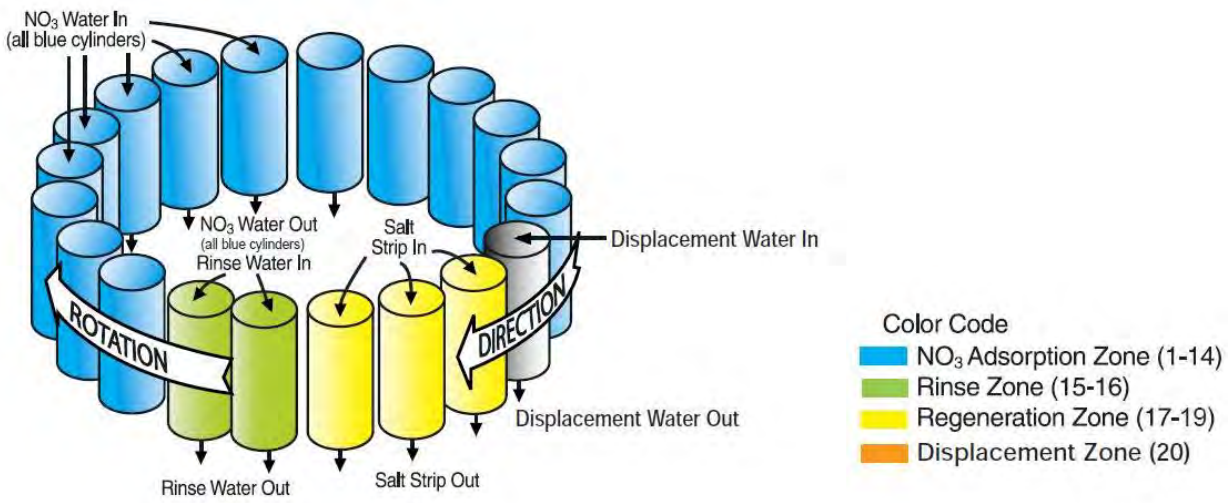


Figure 4. Vessel rotation in Calgon Carbon countercurrent ISEP® system. (Source: Calgon Carbon Corporation 2003.)

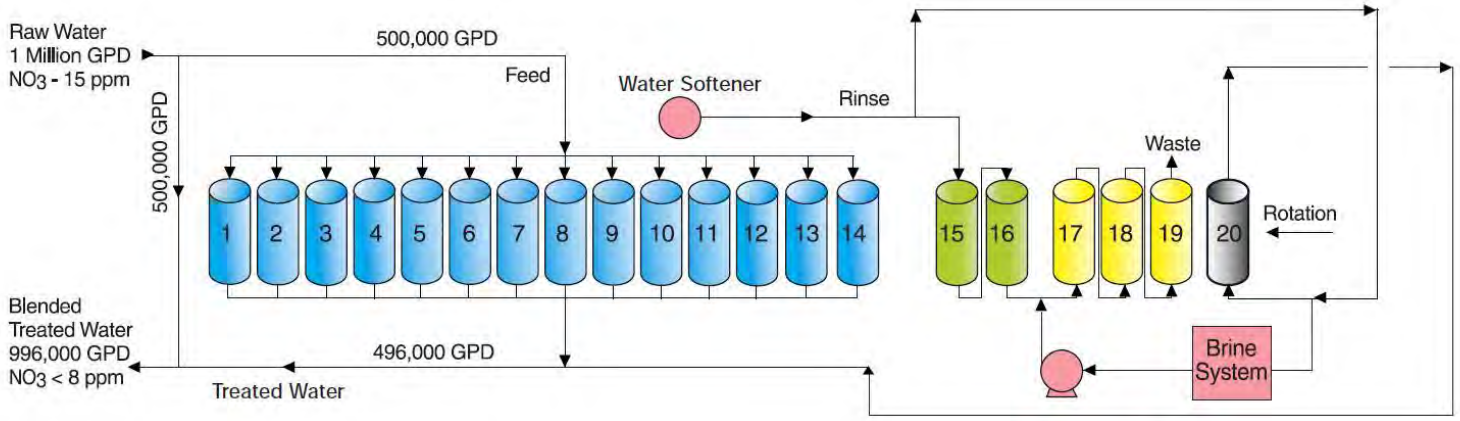


Figure 5. Example of flow through the ISEP® system. (Source: Calgon Carbon Corporation 2003.)

Adminsitrative Record
Page 35311

Multiple Vessel Staggered Configuration

Envirogen Technologies, Inc. offers proprietary ion exchange (IX) systems using multiple beds operated in a staggered design (Figure 6). Such a configuration has the potential to maximize resin capacity and minimize waste and chemical use.

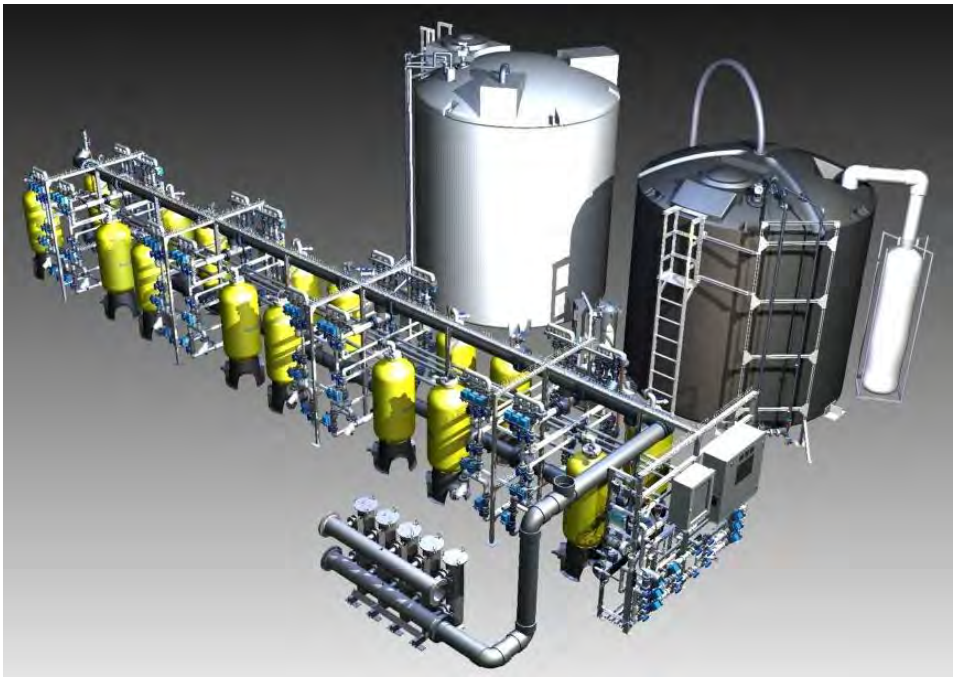


Figure 6. Example of an Envirogen multiple bed proprietary anion exchange system. (Source: reprinted with permission, Envirogen 2009.)

Envirogen's low brine IX system has been successfully implemented for nitrate and uranium removal in San Bernardino County, CA (Envirogen 2010). Delivering 2 MGD with nitrate levels reduced from 50 – 60 mg/L as nitrate (11 – 14 mg/L as N) to 35 mg/L as nitrate (8 mg/L as N) and a system footprint of 50' X 50', the installation resulted in recovery of a well that had been previously decommissioned. Envirogen is contracted to handle all operation and maintenance, including waste disposal.

In Yucca Valley, CA, nitrate treatment was required due to over pumping and the resulting intrusion of septic system contaminated waters. An Envirogen IX system was installed to provide 2.8 MGD. With treatment, nitrate concentrations are decreased from 58 mg/L as nitrate (13 mg/L as N) to 20 mg/L as nitrate (4.5 mg/L as N) with 50% blending and a 0.3% waste rate (0.15% net) (Envirogen 2002).

A detailed case study of an Envirogen low-brine IX treatment plant in California is included in Section 3.1.7 Ion Exchange - Case Studies.

Adminsitrative Record
Page 35312

Weak Base Anion Exchange (WBA IX)

Weak base anion exchange (WBA IX), an emerging technology, can be an effective option for nitrate removal from potable water. Highly pH dependent, nitrate removal using WBA IX is governed by Eqn. 5 (Applied Research Associates N.D.). While SBA IX resin can remove nitrate by splitting nitrate salts, WBA IX resin removes strong acids (Remco Engineering N.D.). First, acid addition protonates the WBA resin (Eqn. 4). Next, the positively charged resin sites remove anions, like nitrate, from the treatment stream (Eqn. 5).

$$R\text{-}NH_2 + H^+ \rightarrow R\text{-}NH_3^+ \qquad \text{(Eqn. 4)}$$

$$R\text{-}NH_3^+ + NO_3^- \rightarrow R\text{-}NH_3\text{-}NO_3 \qquad \text{(Eqn. 5)}$$

Resin regeneration occurs by neutralizing the resin, in accordance with Eqn. 6. Rather than the high salt solution used to regenerate SBA resins, weak bases are used to neutralize the WBA resin.

$$R\text{-}NH_3\text{-}NO_3 + Na^+OH^- \rightarrow R\text{-}NH_2 + HOH + Na^+NO_3^- \qquad \text{(Eqn. 6)}$$

As discussed in previous sections, the use of SBA IX resin results in a high-nitrate brine waste stream. Due to the high salt content, the nitrate in the waste stream generally cannot be beneficially reused. However, with the use of alternative weak bases for regeneration (Eqns. 7 and 8), the waste stream does not have a high salt content and could potentially be recycled as fertilizer (NH_4NO_3 and $Ca(NO_3)_2$) (Clifford 2007).

$$R_3N\text{-}HNO_3 + NH_4OH \rightarrow R_3N\text{-}HOH + NH_4NO_3 \qquad \text{(Eqn. 7)}$$

$$2\ R_3N\text{-}HNO_3 + Ca(OH)_2 \rightarrow 2R_3N\text{-}HOH + Ca(NO_3)_2 \qquad \text{(Eqn. 8)}$$

The use of WBA resins is more operationally complex than the use of SBA resins. Chemical use includes acids and bases, the system is susceptible to corrosion, and pH adjustments are more significant. Adjustment of influent pH to between 3 and 6 is necessary, with subsequent product water pH adjustment as well (Clifford 2007). WBA resins can also be more sensitive to temperature, with one resin having a maximum operating level of 95°C (Dow 2010), but this should not impact their use with municipal drinking water treatment. Regeneration of WBA resin is more efficient than that of the SBA resin of conventional IX; regenerant waste volumes are minimized and waste products can sometimes be recycled as fertilizer (Clifford 2007 and ARA & Purolite N.D.a).

Weak Base Ion Exchange for Nitrate, or the "WIN" Process, was developed by Applied Research Associates, Inc. (ARA) and The Purolite Company (ARA & Purolite N.D.b) as a treatment option that can be less costly and more efficient, with significantly lower waste volumes, than conventional SBA IX (Figure 7).

Adminsitrative Record
Page 35313

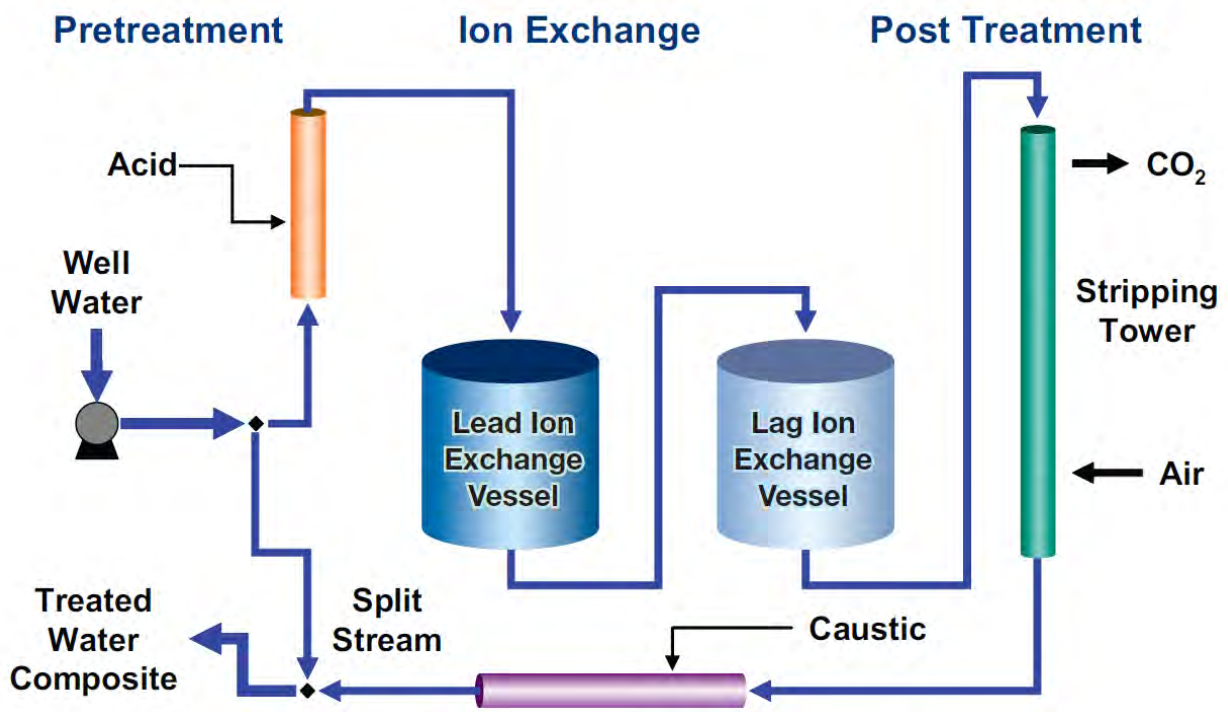


Figure 7. Process schematic of weak base ion exchange for nitrate ("WIN" process). (Source: reprinted with permission, ARA & Purolite N.D.b.)

As an emerging technology, available information is limited to that provided by the manufacturers. The process consists of a pretreatment step to decrease pH, followed by ion exchange vessels in series, and post-treatment to increase pH. The manufacturer states, "The volume of nitrate-containing effluent from the WIN process is typically less than 0.2% of the treated water and, in some cases, may be land applied as fertilizer" (ARA & Purolite N.D.b). As discussed above, the use of SBA IX resin results in a high-nitrate brine waste stream. Due to the high salt content, the nitrate in the waste stream generally cannot be beneficially reused. However, with the use of alternative weak bases for regeneration, the waste stream does not have a high salt content and could potentially be recycled as fertilizer.

3.1.7 Ion Exchange - Case Studies

Conventional Ion Exchange - Case Studies

The following case studies provide detailed information on the design and operation of full-scale conventional IX treatment plants for nitrate removal. Conventional IX is also used by the Chino Basin Desalter Authority in combination with RO. Detailed case studies for the Chino Desalter are listed separately in the RO case study section.

Adminsitrative Record
Page 35314

System Description

Treatment Type	*Ion Exchange and Blending*
System Capacity	*400 gpm*
Raw Water Nitrate	*31 – 53 mg/L as nitrate* *7 – 12 mg/L as N*

A California utility is responsible for a system that has two groundwater supplies, one of which has nitrate at levels that exceed the MCL. The impacted well has a production capacity of 400 gpm and historical nitrate concentrations ranging from 31 mg/L to 53 mg/L of nitrate as NO_3^- (7 – 12 mg/L as N). The utility has implemented a blending program and installed a conventional IX treatment system in 2007 for nitrate control and treatment.

The nitrate impacted sources also have arsenic levels above the MCL of 10 ug/L, which influenced the decision to install IX. IX can simultaneously remove nitrate and arsenic. The treatment system is comprised of three pressure vessels. Two vessels contain Purolite A300E for arsenic removal and the third vessel contains the nitrate selective Purolite A520E resin. The system was originally installed in 2007 and was further modified in 2009. The system is designed to decrease nitrate levels to less than 22 mg/L as nitrate (5 mg/L as N) prior to blending. The maximum distribution system water goal for nitrate is 35 mg/L as nitrate (8 mg/L as N). To assure the system maintains the treatment goals, online nitrate analyzers have been installed at two locations, after the IX system and after blending.

Source Water Quality

- Nitrate - mg/L as NO_3^- (mg/L as N)
  - Average – 35 (8)
  - Minimum – 31 (7)
  - Maximum – 53 (12)
- Co-contaminants
  - Arsenic (15 ug/L)
  - Sulfate (66 mg/L)

Treatment Technology Selection

The conventional IX system was selected to treat both nitrate and arsenic. The decision was further influenced by the ability to discharge the waste brine to a municipal sewer, a cost-effective disposal option. Often technology selection is limited by the costs of brine management. If a utility has the option to dispose of the waste brine to a municipal sewer it can significantly decrease the operations and maintenance cost of the system. No other technologies were pilot tested prior to installation of the IX system.

Operational Notes

Since nitrate is an acute contaminant, the reliability of the treatment system is an utmost concern. The system is equipped with online nitrate analyzers which causes a shutdown of the treatment system when nitrate is at or above 35 mg/L as nitrate (8 mg/L as N). Additionally, the treatment system has experienced arsenic breakthrough resulting in concentrations above the MCL. The treatment system has also experienced premature nitrate breakthrough as a result of faulty distributors in the ion exchange vessels which have since been replaced.

Adminsitrative Record
Page 35315

Treatment System Parameters

- Design Capacity
  - 400 gpm maximum capacity
- Pretreatment
  - None
- Treatment system footprint
  - Treatment system: 30' x 35'
  - Building footprint: None
- Ion exchange pressure vessels
  - Number of vessels: 3
  - Diameter of vessels: 6'
  - Height of vessels: 6'
- Design Loading Rate
  - 5.2 gpm/ft^2
- Bed volumes prior to regeneration
  - 345 – 470 (approximately 220,000 – 300,000 gallons treated)
- Resin Type: Purolite A200E and Purolite A520E nitrate selective
- Previous resins used: None
- Salt Consumption: 1,700 lb/week (May – Sept) and 600 lb/week (Oct – April)
- Volume of brine generated
  - 800 gal/vessel/backwash
  - 99.7% water efficiency
- Monitoring:
  - Online nitrate analyzer
  - Laboratory samples

Residuals Management

The waste brine is discharged to the sewer and sent to a municipal wastewater treatment facility, with an annual cost of $12,000. There have not been any unexpected residuals that have impacted the disposal option.

Technology Benefits and Drawbacks

Benefits

- Ease of regeneration
- Direct-to-sewer brine disposal
- Simple, manually operated system

Drawbacks

- System has potential for breakthrough of nitrate or arsenic
- Time intensive operations
- Required increase in operator certification (California T-3)

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total:	$350,000
Annual O&M Costs (Total with explanation or component costs)	
Total:	$66,500
Resin:	$13,000
Brine Disposal or Treatment:	$12,000
Chemicals:	$5,500
Repair/Maintenance (not including Labor):	$4,500
Power:	$2,500
Labor ($):	$28,000

Adminsitrative Record
Page 35316

System Location: California

CASE #2 System Type: Community Water System

Treatment Type: Conventional Ion Exchange
Startup Date: 2006

Treatment Type	*Ion Exchange*
System Capacity	*400 gpm*
Raw Water Nitrate	*44 mg/L as nitrate*
	10 mg/L as N

System Description

A California water utility operates a system that has three groundwater supplies with varying amounts of nitrate contamination. Two of the wells require treatment as the nitrate concentration is at or above the 45 mg/L as nitrate (10 mg/L as N) MCL. The third well has nitrate ranging from 22 mg/L as nitrate (5 mg/L) as N to 31 mg/L as nitrate (7 mg/L as N) and does not require treatment, but is blended with the IX treated water prior to distribution.

In 2006, a conventional IX system was installed. The treatment system is comprised of three pressure vessels that contain the Rohm and Haas HP 555 ion exchange resin. Since the nitrate concentration in the impacted wells is typically at the MCL, but above the established water quality goal of 35 mg/L as nitrate (8 mg/L as N), a sidestream treatment approach is utilized. In sidestream treatment a portion of the flow is passed through the treatment system while the remainder is bypassed. The IX treatment results in very low nitrate concentration and allows the two streams to be combined to achieve the water quality goal. Sidestream treatment offers capital and operational costs savings.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
  - Average – 44 (10)
- Co-contaminants
  - Sulfate

Treatment Technology Selection

IX was selected for this system because the utility was familiar with the technology from other installations. No technologies were pilot tested prior to the installation of the IX system.

Treatment System Parameters

- Design Capacity
  - 400 gpm maximum capacity
- Pretreatment
  - 100 mesh strainer
  - 10 micron screen
- Treatment system footprint
  - Treatment system: 20’ x 100’
  - Residual handling : 40’ x 100’
  - Total footprint: 60’ x 100’
- Ion exchange pressure vessels
  - Number of vessels: 3
  - Diameter of vessels: 4’
  - Height of vessels: 6’
- Design Loading Rate
  - 10 gpm/ft^2
- Bed volumes prior to regeneration
  - 309 BV
  - Regeneration occurs every 12.4 hours
- Resin Type: Rohm and Haas HP 555
- Previous resins used: None
- Salt Consumption: Estimated to be 32,000 lbs of salt per month
- Volume of brine generated
  - 1,000 gal/vessel/backwash
  - 99.2% water efficiency
- Monitoring:
  - Online nitrate analyzer
  - Laboratory samples

Adminsitrative Record
Page 35317

Residuals Management

The spent brine is held in a storage tank and ultimately disposed off site. Brine volumes are minimized by using a partial flow treatment strategy where only 400 gpm is treated by the IX system while the remaining 500 gpm bypasses the system. The untreated flow is recombined with the treated water prior to entering the distribution system.

Technology Benefits and Drawbacks

Benefits

- Familiarity with technology
- Simple, manually operated system

Drawbacks

- No onsite brine disposal
- Design flaws
- Inconsistent operations
- Potential for nitrate breakthrough

Treatment Technology Costs

Cost information was not included with the survey response.

Operational Notes

This particular IX system has had a series of operational challenges, many of which can be attributed to faulty engineering. Sections of exposed Schedule 80 PVC failed due to freezing, while other sections of pipe failed due to prolonged UV exposure. The brine reclaim tank experienced algal growth and was ultimately replaced with an opaque, UV-resistant tank.

The plant has experienced shutdowns due to an exceedance of the nitrate MCL. Routine sampling showed distribution system nitrate concentrations above the MCL. The system utilizes an online nitrate analyzer to prevent MCL violation; however, a calibration error prevented the system from shutting down as programmed. It is believed a second failure occurred as a result of the brine saturator having a low salt concentration resulting in incomplete regeneration of the resin.

It should be noted that, due to concerns of nitrosamine release (which can be common for IX systems), the effluent of each vessel and the entry point to the distribution system are tested before the system is placed in service.

Adminsitrative Record
Page 35318

Modifications to Conventional Ion Exchange - Case Studies

The following case studies provide detailed information on the design and operation of full-scale low-brine IX systems including MIEX®, ISEP®, and Envirogen (formerly Basin Water) systems.

System Name: Indian Hills
System Location: Indian Hills, CO
PWSID: CO 0130065

CASE #3

System Type: Community Water System

Treatment Type: Counter Current Magnetic Ion Exchange
Questionnaire completed by: Operations staff and Orica Water Care Representatives
Startup Date: 2009

Treatment Type	*Counter Current Ion Exchange*
System Capacity	*50 gpm*
Raw Water Nitrate	*53 – 71 mg/L as nitrate*
	12 – 16 mg/L as N

System Description

The Indian Hills Water District (District) utilizes a groundwater well with a production capacity of approximately 50 gpm. The well has historical nitrate concentrations ranging from 12 mg/L to 16 mg/L of nitrate as N. The District has implemented the counter-current ion exchange process developed by Orica Water Care. Unlike traditional packed bed IX, the Orica process uses a Magnetic Ion Exchange (MIEX®) resin in a series of fluidized beds which allows for a reduction in brine generation.

The minimization of waste brine is accomplished through frequent batch resin regeneration. The magnetized resin encourages agglomeration of loaded resin particles, resulting in faster settling. Loaded resin is removed from the bottom of the IX vessel and is passed to regeneration, while regenerated resin is continuously added at the top of the IX vessel. This configuration reduces the risk of nitrate spiking/chromatographic peaking because clean resin, added at the end point of the system, captures any displaced nitrate, while competing ions, such as sulfate, can be removed early on in the resin vessel.

Regeneration is performed continuously in small batches. Loaded resin is passed to regeneration tanks through the bottom of the IX vessel. The loaded resin is regenerated and then returned to the contactor vessel, thus maintaining a consistent ion exchange capacity in the contactor vessel. Numerous regenerations are performed on a daily basis, with the actual number of regenerations depending on the system's flow rate.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
  - Average – 62 (14)
  - Minimum – 53 (12)
  - Maximum – 71 (16)
- Co-contaminants
  - None noted in survey

Adminsitrative Record
Page 35319

Treatment Technology Selection

The counter-current magnetic ion exchange system was selected based on the expected low levels of brine when compared to conventional packed bed IX. Prior to full-scale implementation, the MIEX® process was pilot tested at Indian Hills. Indian Hills did not pilot test any other technology prior to implementation.

Treatment System Parameters

- Design Capacity
  - 50 gpm maximum capacity
  - 25 gpm typical
  - 11 – 14 gpm actual
- Pretreatment/Post-treatment
  - 1 micron bag filters
  - Chlorination
- Treatment system footprint
  - Treatment system: 9.75' x 5.5'
  - Regeneration system: 13.3' x 6.5'
  - Residuals handling system:
    - 2,000 gal storage tank
  - Total system footprint: 20.5' x 9.75'
  - Building footprint: 30' x 50'
- Number of contactors
  - (2) – 3' diameter; 6.75' height
- Design Loading Rate
  - 7.27 gpm/ft^2 at design flow
- Bed volumes prior to regeneration
  - 125 BV
- Resin Type: MIEX DW 1401- Strong base anion exchange resin in chloride form
- Previous resins used: None
- Salt Consumption: 3,500 lbs/MG treated water
- Volume of brine generated
  - 2,800 gal brine/MG treated water
  - 99.7% water efficiency
- Monitoring:
  - Online nitrate analyzer
  - Laboratory samples
  - Field colorimeter

Residuals Management

The waste brine is sent to a 2,000 gallon underground storage tank. The brine is periodically pumped from the storage tank and ultimately land applied. Indian Hills is investigating deep well injection as an alternative disposal mechanism. The waste brine was analyzed in the pilot portion of the project and no unforeseen residuals were identified that would further limit the brine disposal options.

While not specifically analyzed at this site, waste brine from other MIEX® installations has not had detectable nitrosamine concentrations.

For the MIEX® System, Bed Volumes are defined as the volume of water treated per volume of resin regenerated. BV = gal water treated/gal resin regenerated. For example, if 5 gal of resin were regenerated for every 1000 gal of water treated, the regeneration rate would be: BV = 1000gal/5gal = 200BV.
The design regeneration rate for the MIEX® System is 125 BV, meaning that 8.0 gal of resin are regenerated per 1000 gal of water treated. The system regeneration rate can be adjusted through the control system. There is approximately 1 – 2 gallons of resin attrition per 1 MG of treatment. Small amounts of resin are periodically added to the regeneration system manually.

Adminsitrative Record
Page 35320

Technology Benefits and Drawbacks

Benefits

- Consistent treatment performance
- Relatively low volumes of waste brine
- No nitrate dumping

Drawbacks

- Generation of waste brine
- Resin lost in the treatment must be removed prior to distribution

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Operating and Monitoring Equipment:	Approximately $150,000 (IX treatment system, including initial resin fill)
O&M Costs (Total with explanation or component costs)	
Resin:	$0.08 – $0.15/1000 gallons treated
Resin Disposal:	N/A
Brine Disposal or Treatment:	N/A
Chemicals:	Salt (based on an estimated salt cost of $100/ton): $0.15 – $0.20/1000 gallons treated

Sources*

Vaughan, F., Orica Watercare. (2010) Personal Communication. August, 2010.
Martin, B. (2010) Completed questionnaire. September, 2010.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35321

System Location: California

CASE #4 System Type: Community Water System

Treatment Type: Ion Exchange, Well Modification (proposed), Well Abandonment
Startup Date: 2003 – 2009

System Description

Treatment Type	*Ion Exchange and Blending*
System Capacity	*500 – 900 gpm*
Raw Water Max Nitrate	*35 – 89 mg/L as nitrate* *8 – 20 mg/L as N*

A California district operates 27 groundwater sources, 15 of which contain nitrate at or near the MCL. The district has considered and implemented a variety of solutions including IX, well modifications, and well destruction. The nitrate impacted wells range in capacity from 500 gpm to 1000 gpm with nitrate ranging from 35 – 89 mg/L as nitrate (8 – 20 mg/L as N). It should be noted that a water quality goal of 35 mg/L of nitrate as NO_3^- (< 8 mg/L of nitrate as N) has been implemented in the district, and all sources are treated to below this level.

The district began actively treating the nitrate contaminated sources in 2003 and the most recent system was installed in 2009. The district currently has 6 wells with active IX systems, 7 wells have been destructed or made inactive, 2 wells are being considered for well modifications, and one well has an enhanced control system where there will be an automatic shut down if the nitrate levels exceed a predetermined level. The range of historical nitrate concentrations of the wells is shown below.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
  - Well A- 7 to 58 (1.5 to 13.1)
  - Well B- 15 to 48 (3.4 to 10.8)
  - Well C- 4 to 47 (0.9 to 10.6)
  - Well D- 13 to 88.4 (2.9 to 20.0)
  - Well E- ND to 40.7 (ND to 9.2)
  - Well F- 11 to 53.8 (2.5 to 12.1)
  - Well G- 2 to 52 (0.5 to 11.7)
  - Well H- 8.5 – 72.4 (1.9 to 16.3)
  - Well I- 15.0 to 40.4 (3.4 to 9.1)
  - Well J- 6 to 33.1 (1.4 to 7.5)
  - Well K- 51 (11.5)
  - Well L- 87 (19.6)
  - Well M- 37 (8.4)
  - Well N- 37.4 (8.4)
  - Well O- 64 (14.5)

Treatment Technology Selection

The first nitrate treatment system the district installed was in 2003. At the time of installation IX was deemed the best available technology as it was the most economical with respect to the well's nitrate concentrations and flow rates. When possible, the district elected to install similar systems on the wells that required treatment in an effort to establish operational parallels between their systems. When the footprint of the system or excessive nitrate concentrations made IX treatment infeasible, the district has elected for well destruction. In recent cases where treatment is necessary, the district has evaluated well modifications to determine if it is feasible to reduce the nitrate concentrations without the need for active treatment.

Adminsitrative Record
Page 35322

The following section shows the typical parameters of the district's individual IX systems with the exception of the salt consumption which represents the total salt use for the entire district.

Treatment System Parameters

- Design Capacity
  - 500 – 900 gpm
- Pretreatment
  - None
- Treatment system footprint
  - Two 35' x 20' cement slabs
  - Treatment system: 35' x 10'
  - Housed in a cargo container
- Residuals handling system
  - Three 12' x 12' dia. tanks
- Ion exchange pressure vessels
  - Number of vessels: 16
  - Diameter of vessels: 3'
  - Height of vessels: 6'
- Design Loading Rate
  - 12 gpm/ft^2
- Bed volumes prior to regeneration
  - 300 BV
- Treatment System Manufacturer
  - Envirogen Technologies
- Resin Type: Conventional type 1 ion exchange resin
- Previous resins used: None
- Salt Consumption: 25 tons/ week combined for all systems
- Volume of brine generated
  - 1.3 bed volumes per vessel regenerated
  - Approximately 99.6% water efficiency
- Monitoring
  - Online nitrate analyzer
  - Laboratory samples

Residuals Management

The waste brine is disposed of at an offsite facility. This decision has been impacted by elevated selenium and NDMA in the waste brine.

Technology Benefits and Drawbacks

Benefits

- Allows wells that would normally be offline due to nitrate contamination to be used for potable purposes
- The treatment system vendor provides on-site technical support and operations

Drawbacks

- Treatment system failure poses an acute health risk to the potable water system
- Time intensive operations if there is a treatment disruption
- The systems have numerous valves and moving parts. If there is a mechanical failure it can be difficult to identify the source
- High operating and brine disposal costs
- The district pays the system vendor for stand-by fees and service charges if the system is in stand-by mode during times of low water use

Adminsitrative Record
Page 35323

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total:	$360,000 per unit – Electrical, Piping, Setup, and Sampling/testing (construction costs) The district does not own the treatment plants, tanks, resins, etc.
O&M Costs (Total with explanation or component costs)	
Total:	$59,239.41 per month, per unit
Brine Disposal or Treatment:	$34,016.75 per month, per unit
Repair/Maintenance (not including Labor):	$3,525.00 (service fee) per month, per unit
Salt:	$13,541.41 (salt) per month, per unit
Other:	$7,500.00 (stand by fee) per month, per unit $656.25 (tax) per month, per unit

Adminsitrative Record
Page 35324

System Name: City of Chino
System Location: Chino, CA
PWSID: CA3610012

CASE #5 System Type: Community Water System

Treatment Type: Ion Exchange (IX)
Questionnaire completed by: Gilbert Aldaco, Water Utilities Supervisor, City of Chino Public Works
Startup Date: 2006

System Description

Treatment Type	*Ion Exchange and Blending*
System Capacity	*5000 gpm*
Raw Water Nitrate	*~40 – 200 mg/L as nitrate* *~9 – 45 mg/L as N*

The City of Chino, CA, operates 13 groundwater sources and 3 GWUDI (groundwater under direct influence of surface water) sources. All of the GWUDI and 8 of the groundwater sources are impacted by nitrate contamination. IX and blending are used to address high nitrate and perchlorate levels. One of the wells is inoperable due to perchlorate contamination. IX using the ISEP® system is used for the treatment of 3 wells ranging in capacity from 1100 to 2300 gpm and is being considered for a 4th well. Additionally, an Envirogen (formerly Basin Water) IX system is used for the treatment of an 800 gpm well. Treated surface water from the Metropolitan Water District of Southern California (MWDSC) and treated groundwater from the Chino Basin Desalter Authority is used for blending. The blend ratio is 3:1. The ISEP® system was built in 2005, with a capacity of 5000 gpm (7.2 MGD), and approved for operation in 2006.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
  - 40 to 100 (9 to 45)
- Co-contaminants
  - Perchlorate

Treatment Technology Selection

The costs and feasibility of several alternatives for nitrate treatment, including reverse osmosis, biological processes, and conventional (fixed bed) IX were investigated. Biological treatment and fixed bed IX were pilot tested prior to installation of the current system. The ISEP® system was selected based in part on the potential to simultaneously address perchlorate contamination and due to the efficiency of operation.

Additional Information

- Operators were not required to increase their level of certification to operate the treatment plant.
- When asked about the areas in which the nitrate treatment technology has exceeded expectations, the response was, “Reliability and ability to effectively remove contaminants at varied flow rates (i.e., 1500 gpm to 5000 gpm).”
- There has been no incidence of a plant shutdown due to an alarm or exceedance of an MCL; however, the plant was shut down for approximately 1 month due to theft of computer control equipment.

Adminsitrative Record
Page 35325

Treatment System Parameters

- Design Capacity
  - 5000 gpm
- Pretreatment: Filtration
- Footprint
  - Treatment system: 4400 sq.ft.
  - Residuals handling: 2300 sq.ft.
  - Total system: 6700 sq.ft.
- Ion exchange vessels
  - Number of vessels: 30
  - Diameter of vessels: 3'
  - Height of vessels: 6'
- Max. nitrate concentration goal for delivered water
  - ~36.3 mg/L NO_3^- (~8.2 mg/L N)
- Nitrate concentration goal for the treatment system (before blending)
  - ~19.0 mg/L NO_3^- (~4.3 mg/L N)
- Manufacturer: Calgon Carbon Corporation
- Resin Type: Purolite SBA
- Previous resins used: None
- Bed volumes prior to regeneration
  - Continuous regeneration
- Salt Consumption (@ 2400 gpm)
  - Nitrate mode – 4.9 tons/day
  - Perchlorate mode – 18.6 tons/day
- Volume of brine generated (@ 2400 gpm)
  - Nitrate mode: 12.7 gpm
    - 99.5% water efficiency
  - Perchlorate mode: 31 gpm
    - 98.7% water efficiency
- Monitoring
  - Weekly effluent testing for nitrate and perchlorate
  - Monthly effluent testing for nitrite, sulfate, and total Coliform
  - Monthly raw water testing for nitrate, perchlorate, nitrite, sulfate, total Coliform, HPC
- Grab Samples for resin byproduct testing were negative for nitrosamines

Residuals Management

Waste brine is discharged to a non-reclaimable waste pipeline leading to the LA County Sanitation District, with a total disposal cost of ~$50,000/yr.

Technology Benefits and Drawbacks

Benefits

- Flexibility of operation and High efficiency
- Reliability and Ease of O & M

Drawbacks

- None listed

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total:	~ $4.6 million
Housing:	~ $492 K
Piping:	~ $1.1 million
Resin:	~ $350 K
Testing:	~ $20 K
Other:	~ $2.4 million for ISEP Equipment and Engineering; ~ $350 K for electrical upgrade; ~ $200 K for pumps and associated equipment.
Annual O&M Costs (Total with explanation or component costs)	
Total:	Not reported
Brine Disposal or Treatment:	~ $50 K
Chemicals:	~ $364 K for salt, ~ $50 K for hydrochloric acid

Adminsitrative Record
Page 35326

3.2 Reverse Osmosis (RO)

Reverse osmosis can be a feasible option for nitrate removal in both municipal and Point-of-Use applications (Cevaal et al. 1995; Black 2003; Howe 2004). The first commercial application of RO for potable water treatment was in Coalinga, CA, in 1965 (National Academy of Engineering 2008). RO can be used to address multiple contaminants simultaneously including ionic (e.g., nitrate, arsenic, sodium, chloride, and fluoride), particulate (e.g., asbestos and Protozoan cysts), and organic constituents (e.g., some pesticides) (Dvorak & Skipton 2008). Water is forced through a semi-permeable membrane under pressure such that the water passes through, while contaminants are impeded by the membrane. A typical process schematic of RO for nitrate removal from potable water is illustrated in Figure 8.

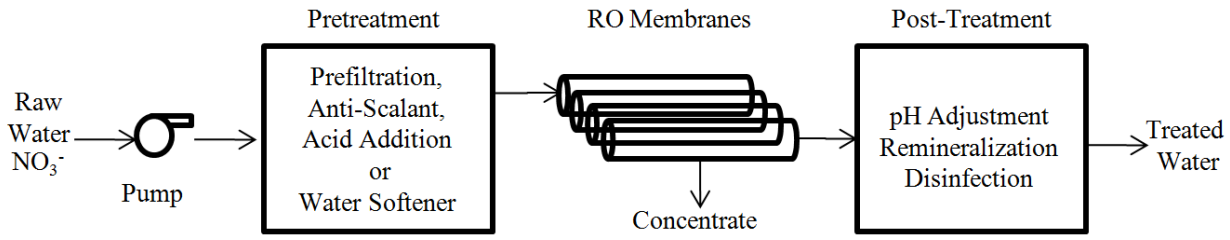


Figure 8. Reverse osmosis schematic.

The required pressure will be dependent on the concentration of solute in the feed water. The collected concentrate is high in nitrate and other rejected constituents (salts) and requires appropriate disposal. The extent to which the RO membrane removes constituents from the water is called the rejection rate. Rejection rates for sodium chloride and sodium nitrate can be as high as 98% and 93%, respectively (Elyanow & Persechino 2005). The water recovery rate of an RO system refers to the "maximum percentage of permeate produced from a given feed flow," and the flow/flux rate is "the maximum flow of permeate through a sq. ft. of the membrane" (Cevaal et al. 1995, p. 107). Modifications and improvements to standard RO have led to the emergence of more efficient RO processes including High Efficiency Reverse Osmosis (HERO™) and Ultra-low Pressure Reverse Osmosis (ULPRO) systems.

3.2.1 Reverse Osmosis - Design Considerations

Table 6 summarizes key design considerations in the application of RO to nitrate removal from potable water.

Adminsitrative Record
Page 35327

Table 6. Summary of design considerations for reverse osmosis.

RO Membranes	• Thin film membranes - Higher rejection rates, lower pressures than CTA membranes • Cellulose triacetate membranes (CTA) - Tolerant of low chlorine levels • Hollow fiber membranes - Compact configuration • Ultra-low pressure RO membranes (ULPRO) • Consider rejection rate, water recovery, and frequency of cleaning • Multiple contaminant removal

Pretreatment	• Prevent membrane damage, scaling and biological, colloidal, and organic fouling
	• Scaling
	  ○ Acid (e.g., sulfuric acid) and/or anti-scaling agents (e.g., poly-acrylic acid)
	  ○ Water softening
	• Biological fouling
	  ○ Upstream disinfection, but dechlorination to prevent membrane damage
	  ▪ Reducing agent (e.g., sodium bisulfate) or activated carbon
	• Colloidal fouling
	  ○ Pre-filtration to remove suspended solids
	  ○ Chemical treatment to keep suspended solids in solution

Post-Treatment	• Avoid corrosion
	  ○ Adjust pH, restore alkalinity for buffering capacity (see remineralization) and/or add corrosion inhibitor (e.g., poly-orthophosphate blend)
	• Remineralization
	  ○ Blending, pH adjustment, addition of caustic soda, bicarbonate, sodium carbonate, phosphates, and/or silicates
	• Blending, disinfection, and storage

Chemical Usage	• pH adjustment, up and down (acids and bases) • Anti-scalants • Cleaning chemicals (acids and bases)

O&M	• Frequency of membrane cleaning depends on water quality and membrane used
	  ○ Typically once a month for 1 hour
	• Management of chemicals and pre-filtration system
	• Waste storage and disposal
	• Membrane replacement/membrane life
	  ○ Up to 20 years or more with appropriate pretreatment and maintenance
	• Monitoring of nitrate levels and membrane flux rate
	• Automation can be feasible
	• Low operational complexity (though higher than IX depending on pretreatment needs)

System Components	• Maximize water recovery while minimizing energy use
	  ○ Pressure range of 100 to 200 psi
	  ○ Based on system size and feed water quality
	• Key system configuration parameters are system flow rate, number of membranes/stages, system footprint, flux rate, water recovery rate, pump selection and sizing, pressure requirement, cleaning frequency

Adminsitrative Record
Page 35328

Waste Management and Disposal	• Significant cost of waste brine/concentrate disposal of greatest concern for inland systems • Management options include sewer or septic system, drying beds, trucking off-site, coastal pipeline, deep well injection, and advanced treatment • Disposal options can be limited by waste brine/concentrate water quality (e.g., volume, salinity, metals, and radionuclides) • Optimization of recycling and treatment of waste brine/concentrate • Higher water recovery (more costly) to minimize waste volume (tradeoff between energy costs and disposal costs)
Limitations	• Need to prevent membrane scaling, fouling, and damage ○ Hardness, iron, manganese, suspended solids, silica, and chlorine • High energy demands • Disposal of waste concentrate • Complete demineralization (no control over target constituents)

Water Quality

While regulated by operational pressure, the water recovery rate depends largely on feed composition. Problematic constituents include sulfate, calcite, calcium sulfate dehydrate (gypsum), silica, colloids, and microorganisms (Cevaal et al. 1995; Elyanow & Persechino 2005; Tarabara 2007). Filtration upstream of the RO membranes is required to remove suspended solids. The life of the RO membranes and prefilters, and the frequency of membrane cleaning are also directly dependent on water quality and the efficiency of pretreatment measures.

Treatment efficiency can be compromised by membrane fouling. Anything that decreases available membrane surface area can limit the passage of water through the membrane and decrease water recovery. The four main types of membrane fouling are scaling, colloidal fouling, biological fouling, and organic fouling. When the salt concentration in the feed water exceeds the saturation point at the membrane surface, precipitation of solids on the membrane can diminish the removal efficiency (Elyanow & Persechino 2005). Scale forming constituents,[14] such as precipitates of silica, calcium, barium, and strontium salts pose a significant threat to RO by limiting the membrane surface area

[14] Pretreatment options to prevent scaling include the addition of acid and/or anti-scaling agents and water softening. By decreasing the pH, the prevalent form of the carbonate cycle is bicarbonate rather than the carbonate ion. The precipitation of calcium carbonate will therefore be limited by the concentration of the carbonate ion. Note: the addition of acid helps only if the scale forming constituent is calcite, due to the speciation of carbonate (Lenntech, 2009c). Anti-scaling chemicals can function in three ways: threshold inhibition, crystal modification, and dispersion (Lenntech, 2009c). Threshold inhibition occurs when the anti-scalant increases the solubility of a potential scalant to super saturation, allowing for a greater concentration to remain in solution. Crystal modification refers to the interference of negatively charged functional groups on the anti-scalant with salt crystal formation and membrane attachment. Anti-scaling chemicals can also promote dispersion of crystals by attaching to them and increasing their negative charge. The most economical means of membrane scale control will depend on the system, but typically the use of anti-scaling agents alone or in combination with acid addition is the most financially prudent option. The third alternative to prevent membrane scaling is to remove the problematic constituents from the water entirely. Water softening can be used to replace calcium and magnesium cations with sodium ions. Generally the most expensive option, this requires the addition of a water softener upstream of the membranes and will result in an additional brine waste stream.

Adminsitrative Record
Page 35329

through which water can pass.[15] Development of biomass on the membrane surface can have a similar negative effect on performance.[16] Additionally, RO membranes have limited to zero tolerance for organic species, like grease and oil; organic fouling inhibits membrane performance (Lenntech 2009e). Lastly, suspended solids not removed by pretreatment filtration can inhibit membrane performance through colloidal fouling.[17] Silica can be a particularly problematic constituent for RO membranes due to the potential for colloidal silica fouling and silica scale formation, which can be very difficult to remove. Modifications to conventional RO have emerged to manage high silica source waters (Section 3.2.5 Reverse Osmosis - Improvements and Modifications, below). Common pretreatment measures used to address membrane fouling are included in Table 6.

With acid addition in pretreatment, the permeate will need to be neutralized in post-treatment to avoid corrosion in the distribution system. Alternatively a corrosion inhibitor, such as a poly-orthophosphate blend, can be used (U.S. EPA 2003). Additionally, because RO is not selective in the removal of ions, treated water is demineralized. Thus, alkalinity may need to be added to restore minerals and buffering capacity to avoid corrosion in the distribution system[18] (WHO 2004; Lenntech 2009f). In post-treatment, blending (if used) follows the RO modules, after which water is disinfected and stored (Cevaal et al. 1995).

System Components and Site Considerations

RO systems are operated in stages. Following pretreatment, water is pumped through the membranes with booster pumps (Cevaal et al. 1995). Water recovery can be improved by passing the concentrate through the membranes more than once, but higher water recovery comes at the expense of increased scaling potential. As the water becomes more concentrated, saturation can lead to precipitation on the membrane (Elyanow & Persechino 2005). The membrane flux rate can be decreased to limit scaling and increase membrane life (Cevaal et al. 1995). Key aspects of the system are pressure pumps, membrane configuration, membrane flux rate, number of stages/number of membranes, flow rate, and cleaning and anti-scalant requirements. Pump sizing is based on system size and pressure requirements. The water recovery rate can be regulated by the operational pressure. The necessary pressure is dependent on the concentration of solute in the feed water. For example, with a conductivity of 1550 µS/cm,

[15] The Langelier Saturation Index (LSI) and other corrosion indices are used to characterize the scaling potential of calcium carbonate (calcite), a commonly problematic constituent.

[16] Upstream disinfection can limit membrane biofouling; however, additional measures must be taken to avoid membrane exposure to chlorine "by dosing with a reducing agent (such as sodium bisulfate) or by contacting with activated carbon" (Elyanow & Persechino 2005, p. 6).

[17] The potential for colloidal fouling is typically characterized by the silt density index (SDI). An SDI greater than 3 can indicate the need for further pretreatment to minimize cleaning frequency and membrane damage (Lenntech 2009e; Elyanow & Persechino 2005). Chemical treatment can keep suspended solids in solution. Alternatively, a prefilter can be used to remove solids from the feed water (Remco Engineering N.D.).

[18] It is important to note the relationship between alkalinity and pH. Alkalinity is a measure of buffering capacity or the resistance to changes in pH. Demineralized water or water with a low buffering capacity will be susceptible to more dramatic pH changes and is considered unstable. The pH of acidic product water should be adjusted and the buffering capacity of demineralized product water should be restored to avoid corrosion downstream. Options for the stabilization and remineralization of demineralized water include blending, pH adjustment, and addition of caustic soda, bicarbonate, sodium carbonate, phosphates, and/or silicates (WHO 2004; Lenntech 2009f). Corrosion indices and models can be used to determine appropriate solutions for specific scenarios.

Adminsitrative Record
Page 35330

Panglisch et al. (2005) determined the suitable pressure to be 145 – 174 psi. Similarly, at start-up, the operational pressure at an RO facility in Brighton, CO, used for nitrate removal was 170 psi (Cevaal et al. 1995).

Two types of spiral wound RO membranes are commonly used: polyamide thin film composite membranes (TF) and cellulose triacetate membranes (CTA) (Cevaal et al. 1995 and Remco Engineering N.D.). While TF membranes are capable of slightly higher rejection rates and can be operated at lower pressures, CTA membranes are tolerant of low chlorine levels (AWWA 2011). Hollow fiber RO membranes are also available, which can minimize system footprint, but can be more susceptible to fouling from suspended solids (Hydranautics 2001). Using recently developed ultra-low pressure RO (ULPRO) membranes, operational pressures can be reduced, decreasing power costs (Drewes et al. 2008). For additional information see Section 3.2.5 Reverse Osmosis - Improvements and Modifications.

Residuals Management and Disposal

The volume of the waste stream can be considerable, ranging from 15% to 50% of the starting volume depending on the operational parameters (Howe 2004). The waste stream, or concentrate, can be discharged to a wastewater treatment plant or a septic system (Bilidt 1985; Howe 2004), as long as the system can accommodate an increased salt concentration. Additional disposal options include drying beds, infiltration basins, trucking off-site, a coastal pipeline, deep well injection, advanced treatment, and most commonly, discharge to nearby surface salt-waters (i.e., oceans), when available (Howe 2004). Important water quality characteristics of the waste brine (e.g., volume, salinity, metals, and radionuclides) can affect the feasibility and costs of disposal. Options for inland communities are more limited and costly. Proximity to coastal power plants can be advantageous. Power plants using ocean water for cooling can provide a pre-existing infrastructure for disposal to ocean waters (Black 2003).

The high cost of nitrate laden concentrate disposal has led to research into optimization of recycling and treatment of this waste stream. Coupling of RO systems with biological, chemical, or catalytic denitrification enables removal of nitrate from the waste concentrate, with reduction to nitrogen gas. Additionally, the Vibratory Shear Enhanced Process (VSEP), from New Logic Research, has been explored in the context of treatment of RO concentrate from wastewater treatment. By applying a shear force across the membrane, pore clogging by colloidal particles is minimized, leading to the potential for improved water recovery (Lozier et al. N.D.). Several combined configurations of interest are discussed below in Section 3.6 Brine Treatment Alternatives and Hybrid Treatment Systems.

Maintenance, Monitoring, and Operational Complexity

RO systems are typically highly automated, accommodating the greater operational complexity of RO operation, in comparison with IX. Several operational decisions will be dictated by operator availability and training. For instance, chemical addition in pretreatment can be quite effective, but will require more intensive maintenance. In contrast, opting for the more expensive choice, installing a water softener, will require less operator time. Membrane cleaning frequency varies widely and depends on the efficiency of pretreatment measures and water quality. Interruption of operation is not always

Adminsitrative Record
Page 35331

necessary as the membranes can be isolated and cleaned in place (CIP) in stages. Membranes are typically cleaned with acid or caustic solutions (WA DOH 2005). Cleaning agents are selected based on the cause of membrane fouling. Cleaning and rinsing can take an hour and with effective pretreatment, monthly membrane cleaning should be sufficient (Remco Engineering N.D.; Bates N.D.). With effective pretreatment, cleaning frequency can be significantly minimized. An RO plant in Milan, Italy, using anti-scalants, requires cleaning only once every 18 months (Elyanow & Persechino 2005). Filters should be checked weekly and, if used, the water softener should be maintained with sufficient salt every day. Effluent nitrate concentrations require monitoring to ensure compliance and to assess membrane performance. Over time, membrane degradation will lead to a gradual decrease in removal efficiency. Membrane life varies and can range from 5 to 20 years or more (Remco Engineering N.D.). Waste concentrate management consists of appropriate storage and disposal. More operationally complex than IX, operators of RO systems will typically require more training and system maintenance will demand more time and chemicals. However, with the implementation of appropriate pretreatment measures and the ability for system automation, operational complexity can be minimized.

3.2.2 Reverse Osmosis - Cost Considerations

For the efficient operation of an RO system, the fundamental objective is to maximize water recovery with the minimum amount of energy and chemical usage, while meeting necessary potable water guidelines. Factors affecting system cost include facility size (how much water), source water quality (including nitrate concentration), environmental factors (temperature and pH), and target effluent nitrate concentration (Bilidt 1985). Lower operating pressures are less costly, but result in decreased water recovery. High operating pressures maximize water recovery (decreasing disposal costs), but increase energy demands and the need for "specialized pumps" (WA DOH 2005). Thus, there is a trade-off between the costs of increasing water recovery (increased pretreatment and operational pressure) and the costs of disposal (pumping, storage, and disposal expenses). In pretreatment, the use of anti-scalants rather than acid or a water softener is generally the least expensive. The use of a water softener is the least cost competitive option (Lenntech 2009c). Regarding small water systems, "reverse osmosis is one of the most expensive forms of centralized treatment and will likely not be cost effective unless there are multiple contaminants needing removal" (WA DOH 2005, p. 27).

Capital costs for RO include land, housing, piping, storage tanks, O&M equipment, membranes, preliminary testing (pilot studies), permits, and training. O&M costs include membrane and filter replacement, membrane and filter disposal, concentrate disposal or treatment, chemical use (anti-scalant, pH adjustment, disinfection, etc.), repair, maintenance, power, and labor.

Published cost information, from existing RO installations used for nitrate treatment, is listed in Table 7. Costs have been adjusted to 2010 dollars, unless indicated otherwise. Costs can be difficult to assess due to inconsistencies in how cost information is reported. Comparison of treatment costs is not always valid due to differences in influent water quality parameters, system size, waste management options, and system configuration. Published costs do not always include comparable information. It would be inappropriate to compare the O&M costs of a facility that excludes disposal costs with others that include this information. The listed cost information is provided as an approximate range of costs for

Adminsitrative Record
Page 35332

specific facilities. Costs for implementing RO may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the questionnaire includes detailed costs associated with the unique case studies included in this analysis. A detailed discussion of treatment costs is included below in Section 6 Treatment Cost Analysis.

Table 7. Selected costs* of reverse osmosis systems for nitrate removal.

System Flow**	< 0.5 MGD	0.5 – 5 MGD	5+ MGD
Annualized Capital Cost ($/1000 gal)	3.51 – 5.17 [1, 2]	1.00 – 1.30 [3, 4]	0.95 [3]
O&M Cost ($/1000 gal)	1.46 – 16.16 [1, 2]	1.22 – 2.01 [3, 4]	1.63 [3]
Total Annualized Cost ($/1000 gal)	5.73 – 19.70 [1, 2]	2.52 – 3.21 [3, 4]	2.58 [3]

*Costs have been adjusted to 2010 dollars with 7% interest over 20 years, unless indicated otherwise.
**When available, costs are based on actual system flow rather than design capacity.
[1] Walker (N.D.), costs not adjusted to 2010 dollars. [2] Personal communication with representatives of two small water systems (2010). [3] Conlon et al. (1995). [4] Cevaal et al. (1995).

3.2.3 Reverse Osmosis - Selected Research

Much research has focused on RO; Table A.2 of the Appendix is a list of recent studies relevant to nitrate removal from potable water and several examples of RO application. Current RO research focuses on improvements of membranes and waste management, and decreasing energy use.

3.2.4 Reverse Osmosis - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of RO in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of RO include high quality product water, multiple contaminant removal, desalination (TDS removal), feasible automation, and application for POU applications. According to Elyanow & Persechino (2005), in their comparison of RO and EDR, "…the best economical choice for small capacity systems (<110 gpm or <25 m^3/hr) are simple RO plants, which have less electrical and hydraulic complexity than EDR and other technologies" (p. 7). In waters where salinity is a problem, RO can be better suited than IX due to the ability to remove multiple contaminants (including trihalomethane formation potential precursors (THMFPs)) (Cevaal et al. 1995).

Disadvantages of RO include high capital and O&M costs, membrane fouling susceptibility, high pretreatment and energy demands, and potentially large waste volume (lower water recovery) requiring disposal. The high cost of disposal from inland locations can result in RO treatment becoming cost prohibitive. Howe (2004) presents several alternatives to conventional disposal measures of RO waste brine, including reuse for industrial processes, processing (e.g., for salt production), or use in energy generation ("solar brine pond").

Adminsitrative Record
Page 35333

3.2.5 Reverse Osmosis - Improvements and Modifications

Process Modification

High Efficiency Reverse Osmosis (HERO™) is a patented multi-step process enabling increased water recovery (greater than 90%) and minimizing cleaning requirements. This process limits scaling by incorporating hardness reduction, CO_2 stripping, and pH adjustment (GE 2010b). Raw water is subjected to intensive pretreatment before passing through the RO membranes as follows (Engle 2007):

- Weak acid cation exchange (WAC) is used to remove hardness ions,
- CO_2 stripping is used to remove carbonate and increase pH, and
- Base addition is used to increase the pH to a level of 10.5.

An example flow diagram is illustrated in Figure 9.

With such pretreatment, water fed to the RO membranes is softened and pH is adjusted high enough to significantly increase the solubility of silica. The high chemical usage and multiple steps result in a more complicated process than conventional RO. However, benefits include increased water recovery, decreased waste volume, and the ability to treat severely impaired and poor quality source water containing multiple contaminants (Engle 2007).

The HERO™ process was initially designed to produce ultra-pure water for use in electronics applications and was patented by Debasish Mukhopadhyay with licensing rights for different applications (Engle 2007). The HERO™ process has been implemented for drinking water treatment in the small community of Yalgoo, Australia, to produce high quality drinking water from brackish groundwater high in silica and nitrate (Water Corporation 2007; Water Corporation 2009; Thomson et al. 2009). Higher removal rates result in decreased waste volume. Using the HERO™ process in Yalgoo, waste volumes are as low as "one-tenth of a conventional plant's concentrated brine residue for disposal, eliminating the need for big evaporation ponds" (Water Corporation 2007, p. 8).

Adminsitrative Record
Page 35334

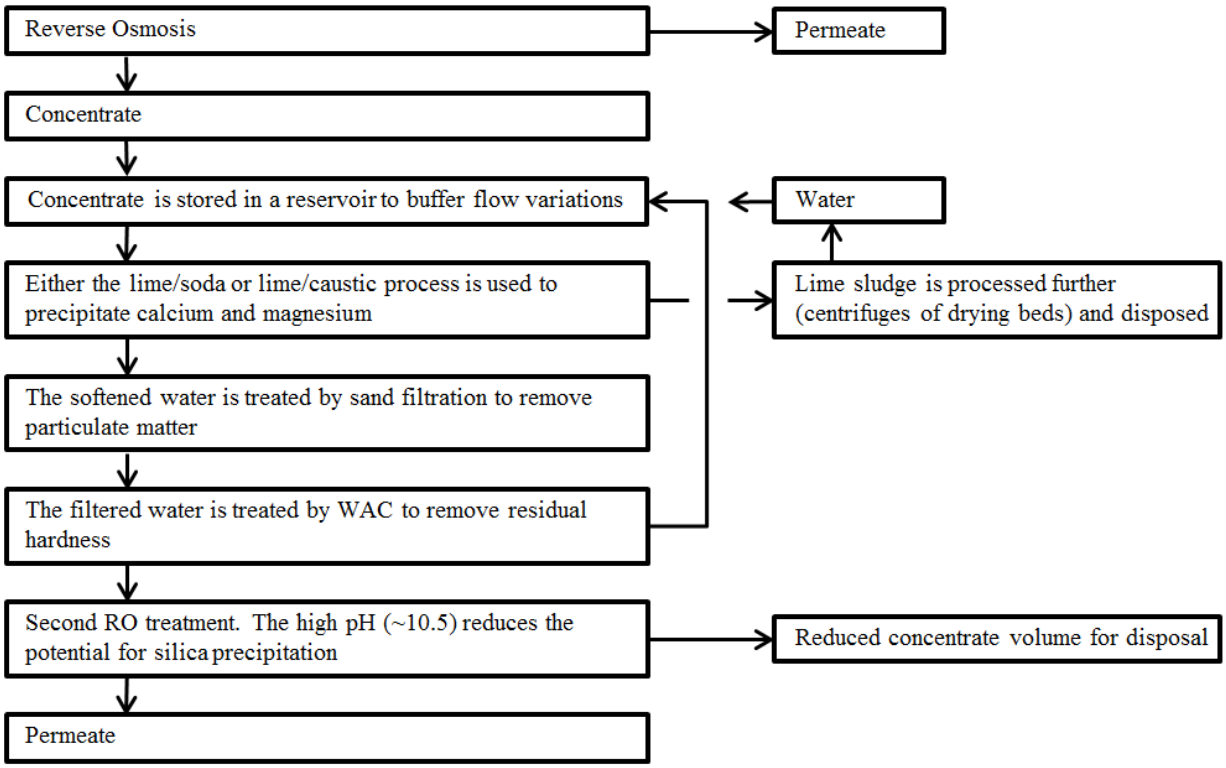


Figure 9. Flow chart of the HERO™ process. (Source: reproduced with permission, Central Arizona Salinity Study 2006.)

Membrane Modification – Low Pressure Membranes

Research and development in membrane technology has resulted in the emergence of Ultra-Low Pressure Reverse Osmosis (ULPRO). In contrast to the high pressures required for conventional RO, use of ULPRO membranes allows for lower operating pressures and improved flux rates. Energy demands can be reduced due to lower operating pressures; however, pretreatment practices to prevent membrane scaling and fouling are similar to those necessary for conventional RO membranes (Drewes et al. 2008). ULPRO membranes are available from several manufacturers. Operating pressures are in the range of 50 to 125 psi, while the pressures required for conventional RO membranes can be over 200 psi (Excel Water 2007; Drewes et al. 2008; Koch Membrane Systems 2008). Drewes et al. (2008) compared the performance of ULPRO membranes and conventional RO membranes, wherein pretreatment included nano-filtration for both RO options. Findings indicate that the ULPRO membranes included in the study were capable of successfully removing nitrate and multiple additional contaminants to potable water standards. "With regard to operating costs, operating pressure is the only TMG [ULPRO membrane] operating parameter considered to deviate from the benchmark ESPA2 [conventional RO membrane] membrane. Pretreatment requirements and recovery rate were the same. Electrical consumption will be directly proportional to the required operating pressure" (Drewes et al. 2008, p. 93). However, in the cost comparison between the two membranes, the benefits of lower operating pressures were overshadowed by the poor recovery of the ULPRO membranes after cleaning.

Adminsitrative Record
Page 35335

The authors suggest that the cleaning of the ULPRO membranes would need to be optimized for an improved cost comparison.

3.2.6 Reverse Osmosis - Case Studies

The following case studies provide detailed information on the design and operation of full-scale RO treatment plants used for nitrate removal. Chino I Desalter and Chino II Desalter are combination systems using both RO and conventional IX.

Adminsitrative Record
Page 35336

System Description

Treatment Type	*Reverse Osmosis and Blending*
System Capacity	*120 gpm*
Raw Water Nitrate	*75 – 84 mg/L as nitrate* *17 – 19 mg/L as N*

A California water utility operates a system that has three groundwater supplies, one of which has nitrate at levels that exceed the MCL. The impacted well has nitrate concentrations that range from 75 to 84 mg/L as nitrate (17 to 19 mg/L as N) and has a typical production capacity of 100 gpm. In 2002, the utility implemented a blending program and installed an RO system for nitrate control and treatment.

In RO, raw water is forced through a semi-permeable membrane under pressure such that the water passes through, while contaminants are impeded by the membrane. The required pressure will be dependent on the concentration of solute in the feed water. The collected concentrate is high in nitrate and other rejected constituents (salts) and requires appropriate disposal. The extent to which the RO membrane removes constituents from the water is called the rejection rate. Rejection rates for sodium chloride and sodium nitrate can be as high as 98% and 93%, respectively (Elyanow & Persechino 2005).

The high nitrate supply is blended with one of the other two groundwater sources prior to RO treatment. The RO system reliably removes nitrate to below 35.4 mg/L as nitrate (8 mg/L as N) and the water delivered to consumers typically has nitrate levels below 13.3 mg/L as nitrate (3 mg/L as N). This system utilizes a leach field type system to land apply the RO concentrate.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N)
  - Average - 75 (17)
  - Minimum - 80 (18)
  - Maximum - 84 (19)
- Co-contaminants
  - Fluoride - 3.3 mg/L
  - Arsenic
  - Radium

Treatment Technology Selection

RO was selected as the most appropriate treatment system as the technology can reliably remove nitrate in addition to the co-occurring contaminants that are present, specifically fluoride, arsenic, and radium.

Adminsitrative Record
Page 35337

Treatment System Parameters

- Design Capacity
  - 120 gpm maximum capacity
- Pretreatment
  - Anti-scalant (Hyposperse MCD 150)
- Treatment system footprint
  - Treatment system: 15' x 30'
  - Total system footprint: 40' x 100'
- RO System
  - System manufacturer: Aria™
  - Membrane manufacturer: Osmonics
  - Number of stages: 4
  - Number of RO elements per stage: 4
- Clean-in-place (CIP)
  - CIP frequency: Quarterly (4x/year)
  - Initiated when there is a 15% decrease in permeate flow or salt rejection or a 15% increase in trans-membrane pressure
  - CIP chemicals: Dilute phosphoric acid
- Water efficiency: 80%
- Monitoring
  - Laboratory samples
- Service life of membranes
  - Approximately 8 years

Residuals Management

The concentrate is disposed to an on-site leach field.

Technology Benefits and Drawbacks

Benefits

- Effectively removes nitrate and other co-contaminants
- On-site concentrate disposal
- Consistent operations

Drawbacks

- Energy intensive
- Relatively low water efficiency (80%)

Treatment Technology Costs

Treatment technology costs are not available for this system.

Operational Notes

The RO system has never had any extended unplanned shut downs or been shut down as the result of an alarm. There has been an exceedance of the fluoride MCL that occurred near the end of the useful life of the membranes. The membranes have since been replaced, resolving this operational issue.

References

Elyanow, D. and Persechino, J. (2005) Advances in Nitrate Removal. GE – General Electric Company, Water & Process Technologies. Accessed June 11, 2010 via < http://www.gewater.com/pdf/Technical%20Papers_Cust/Americas/English/TP1033EN.pdf>.

Adminsitrative Record
Page 35338

System Name: City of Brighton
System Location: Brighton, CO
PWSID: CO 0101025

CASE #7 System Type: Community Water System

Treatment Type: Reverse Osmosis (RO)
Questionnaire completed by: Dave Anderson, City of Brighton RO Chief Plant Operator
Startup Date: 1993

System Description

Treatment Type	*Reverse Osmosis*
System Capacity	*6.65 MGD (4,600 gpm)*
Raw Water Nitrate	*49 – 89 mg/L as nitrate* *11 – 20 mg/L as N*

The City of Brighton (City) utilizes six groundwater wells with production capacities ranging from 900 to 1500 gpm and one groundwater source which has been designated as groundwater under direct influence of surface water (GWUDI), as an emergency well, with a production capacity of 700 gpm. These seven sources are impacted by nitrate with average concentrations ranging from 49 to 89 mg/L of nitrate as nitrate (11 to 20 mg/L of nitrate as N). The City has implemented RO with blending. The design capacity of the RO system is 6.65 MGD of permeate at 80% recovery (1150 gpm/train). Green sand and cartridge filters (Graver) are used to treat the GWUDI source, primarily for the removal of manganese. (Additional sources operated by the City that are not impacted by nitrate are purchased treated surface water and additional GWUDI wells.)

Raw water enters the system with 40% of feed water bypassing the RO system and 60% of feed water passing to pretreatment. After anti-scalant addition, pretreated water is pressurized with boost pumps and passed to the RO skids. Waste concentrate exits the system for disposal and post-treatment of the permeate includes CO_2 stripping and the addition of chlorine and caustic. Post-treated water is blended with bypassed water and sent to storage and ultimately distribution.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N)
  - Average - 46.9 to 90.3 (10.6 to 20.4)
  - Minimum - 20.02 to 69.97 (4.52 to 15.8)
  - Maximum - 78.8 to 112.9 (17.8 to 25.5)
- Co-contaminants
  - TDS: 580 to 1000 mg/L, RO TDS ~34 mg/L, Finished TDS ~280 mg/L
  - Fluoride: 1.3 mg/L
  - TOC: < 2 mg/L
  - Hardness: 370 – 480 mg/L as $CaCO_3$ (historically, Cevaal et al. 1995)

Adminsitrative Record
Page 35339

Treatment Technology Selection

IX and EDR were also considered and pilot tested prior to installation of the RO system. RO was selected due to nitrate levels and hardness. IX could have been less costly; however, the lower salt levels in RO concentrate make it possible to discharge waste to the South Platte River. "By selecting RO, the City hoped to actually reduce the salt load on the river with RO since many Brighton residents currently using home ion-exchange softening units would no longer use them" (Cevaal et al. 1995, p. 102). Biological treatment is also being explored for the treatment of nitrate in the waste brine.

Residuals Management

The waste concentrate is continuously discharged via a brine line to the South Platte River. Biological treatment is being explored for the treatment of the waste concentrate. The biological system would be located on the West side of the RO treatment system and would allow for reduction of nitrate in the waste stream. As mentioned above, the use of RO rather than ion exchange was an effort to decrease salt loading to the South Platte River. Historically, Brighton residents used in-home ion exchange units to soften water.

Technology Benefits and Drawbacks

Benefits

- Consistent treatment performance
- Ease of operation

Drawbacks

- Constant generation of waste stream
- High power consumption

Adminsitrative Record
Page 35340

Treatment System Parameters

- Design Capacity
  - 6.65 MGD of permeate at 80% recovery (1150 gpm/train)
- Pretreatment/Post-treatment
  - Anti-scalant:
    King Lee Technologies
    Pre Treat Plus 0100 phosphonate
  - 5 micron cartridge filters
    - 2.5 inch diameter
    - 90 day replacement
  - pH adjustment
    - caustic soda (NaOH)
    - Air stripping (CO_2 removal)
- Treatment system footprint
  - Treatment system:
    11,000 sq. ft. at installation
    (Cevaal et al. 1995)
- Number of contactors
  - 2 stages, 5 trains
  - RO elements/stage
    - 36 x 18 array
    - 6 membranes/vessel
    - 324 total
- Max. Concentration goal for delivered water
  - 35.4 mg/L as nitrate (8 mg/L as N), (always produce lower)
- Rejection Rate
  - 95 – 98% rejection
  - Nitrate goal (before blending):
    ~4.4 mg/L as nitrate (~1 mg/L as N)
- Flux rate of the RO membranes
  - 13 gpd/sf
- System Manufacturer
  - Hydranautics and Hydrocode
- Membrane Type
  - CPA2 (no others used in past)
- Membrane Life
  - Unknown, none have required replacement (5 yrs. ago, the manufacturer said the membranes should last 3 more yrs.).
- Membrane Cleaning
  - Clean in Place initiated by time rather than decrease in flux. Every ~157 million gals treated (~2x/yr)
  - Chemicals:
    Nalco Product and Citric acid
- Waste
  - Discharge via Brine line to South Platte River
  - Recovery Rate: 80%
- Monitoring
  - Ion chromatography
    - At Source
    - At Point-of-Exit
  - Grab ISE (HACH)
    - At Blending Point
    - At Point-of-Exit
  - Testing once per year is required for compliance

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Operating and Monitoring Equipment:	$8,253,000 (1993) 4MGD RO facility
Annual O & M Costs (Total with explanation or component costs)	
Total:	$2,873,293.00
Membrane:	0
Membrane Disposal:	0
Brine Disposal or Treatment:	0
Chemicals:	Approx. $100,000 year
Power:	Approx. $210,000 year for RO
Labor (Hours per Year):	10 hr/day, 7 day/wk
2 MGD Thornton treated:	$3.60/1000 gallons
COMPLETE Cost (including treatment, distribution, everything):	$3.16/1000 gallons

Adminsitrative Record
Page 35341

Additional Information

- The level of certification required for plant operators is Colorado A treatment.
- During a power outage, there is a pause before the generators start. This requires a manual restart of the system.
- This system has never produced water exceeding the nitrate MCL and has never had an unplanned shutdown exceeding one week.
- The major benefit of the RO system is the rejection rate allowing for removal of regulated contaminants.
- The most significant disadvantages are the high power consumption and the continuous brine discharge.
- The operator also noted that there has been a decreasing trend in nitrate levels in their sources.

References

Cevaal, J.N., Suratt, W.B., and Burke, J.E. (1995) Nitrate removal and water quality improvements with reverse osmosis for Brighton, Colorado. *Desalination*, **103**, 101 – 111.

Anderson, D. (2010) In-person interview and tour of the facility. July 19, 2010.

Anderson, D. (2010) Completed questionnaire. September 15, 2010.

Adminsitrative Record
Page 35342

System Name: Western Municipal Water District - Arlington Desalter
System Location: Riverside, CA
PWSID: CA3310049

CASE #8 System Type: Community Water System

Treatment Type: Reverse Osmosis (RO) and Blending
Questionnaire completed by: Joseph Bernosky, Director of Engineering
Startup Date: 1990 strictly for desalting, Upgraded to drinking water treatment in 2002

System Description

Treatment Type	*Reverse Osmosis and Blending*
System Capacity	*6.6 MGD (4,600 gpm)*
Raw Water Nitrate	*44 – 89 mg/L as nitrate*
	10 – 20 mg/L as N

The Western Municipal Water District (District) operates a system comprised of seven wells, five of which contain nitrate above the MCL. Three of the nitrate impacted wells are treated by a 6.6 MGD RO facility. The permeate, or treated water, from the RO system is blended with the remaining two wells prior to distribution. The RO and blending facilities are collectively referred to as the Arlington Desalter. Approximately 60% of the total flow is treated by the RO system and the remaining 40% is blended with the treated water. The District targets a nitrate concentration of 22 mg/L as nitrate (5 mg/L as N) in the distribution system.

The Arlington Desalter facilities were originally installed in 1990 to address the salt imbalance in the Upper Santa Ana Watershed. High salinity waters withdrawn from the South Arlington Basin were treated by the Arlington Desalter and subsequently discharged to the Santa Ana River for downstream use (and downstream drinking water treatment). The system was upgraded to a drinking water treatment facility in 2002 with the addition of disinfection, a clear well and a pump station used to pump drinking water into the distribution system for the city of Norco, CA.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N)
  - Average - 75 (17)
  - Minimum - 44 (10)
  - Maximum - 89 (20)
- Co-contaminants
  - TDS - 1200 mg/L

Treatment Technology Selection

Due to the original intent of the system and the 2002 conversion for drinking water production, no other technologies were pilot tested prior to the installation of the RO and blending facilities. However, biological treatment of RO bypass water was recently pilot tested with full-scale implementation anticipated. A case study about this fixed bed biological pilot study is listed separately.

Adminsitrative Record
Page 35343

Treatment System Parameters

- Design Capacity: 6.6 MGD
- Pretreatment/Post-treatment
  - Anti-scalant: Y2K Anti-scalant
- Treatment system footprint
  - Treatment system: Approximately 7,500 ft^2 excluding clear well and pump station
- Number of Stages
  - Stage 1: 36 vessels each with 6 units
  - Stage 2: 12 vessels each with 6 units
- Max. Concentration goal for delivered water
  - 22 mg/L as nitrate (5 mg/L as N)
- Water Recovery
  - Original design: 75 – 76%
  - Current: 80%
- Flux rate of the RO membranes
  - 16 gpd/ft^2
- System Manufacturer
  - Hydranautics
- Membrane Type
  - Koch HR400
- Membrane Life
  - > 10 years
- Membrane Cleaning
  - Occurs 2 times per year
  - Chemicals: Low pH solution, hydrofluorosilic acid, high pH solution
- Waste: The RO concentrate is disposed of offshore via the Santa Ana Regional Interceptor (SARI) brine line
- Monitoring
  - Ion chromatography
  - Online nitrate analyzers

Residuals Management

Waste is discharged to the Santa Ana Regional Interceptor (SARI) (Brine Line). The SARI line prevents degradation of natural waters caused by increased salinity. Managed by the Santa Ana Watershed Project Authority (SAWPA), the SARI line is a dedicated interceptor line built to help users meet discharge requirements. In addition to the Arlington Desalter, the SARI line is used by other dischargers including industrial and domestic sources. The District's contribution to the total flow of the SARI line is approximately 5%. The SARI line carries water to the Orange County Sanitary District for wastewater treatment with ultimate offshore discharge.

Having access to the SARI line for brine disposal is a benefit of this system; however, there have been complications with the SARI line. Tremendous scaling problems in Arlington Desalter's reach of the SARI line have resulted in the need to address calcium carbonate buildup and to consider additional cleanout points. Due to multiple discharge sources, the SARI line combines waters having very different water quality characteristics. Reactions within the mixed water can vary based on water chemistry. Several cleaning and maintenance options are being considered.

Technology Benefits and Drawbacks

Benefits

- Reliable
- Access to SARI line for disposal
- The treatment potential of the pre-existing RO system has been maximized through conversion for drinking water treatment

Drawbacks

- Costly brine disposal
- Complications with SARI line (scaling, etc.)

Adminsitrative Record
Page 35344

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)	
Total Capital Costs:	Unavailable for initial installation in 1990
Annual O&M Costs (Total with explanation or component costs) (2009/2010)	
Total:	$2,931,228
Labor and Overhead:	$836,530
Outside Services:	$200,000
Emergency Repairs:	$25,000
General & Administrative:	$85,000
Vehicle and Equipment:	$9,000
SARI Fixed Cost:	$151,800
Materials and Supplies:	$5,500
Permits and Fees:	$27,000
SARI Variable Cost:	$470,000
Chemicals:	$150,000
Energy:	$971,398

Additional Information

Regarding water recovery, the original design recovery rate was 75 – 76%. With modifications to anti-scalant use, the recovery rate increased to 78% and then to the current operational water recovery rate of 80%. To further improve the water recovery rate, additional testing is anticipated which will require engineering work and a cost-benefit analysis.

Currently used membranes are 10 years old and are still performing adequately with respect to operational parameters (flux rate, rejection rate, etc.). The District has budgeted for membrane replacement in this fiscal year; however, due to adequate performance, the current membranes may be used for an additional year. Membranes are actually attaining better water recovery than manufacturer specifications. Membrane life is also exceeding initial expectations.

Sources*

Bernosky, J. (2010) Personal communication. November 5, 2010.
Bernosky, J. (2010) Completed questionnaire. October, 2010.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35345

System Name: Chino Basin Desalter Authority (CDA) – **Chino I Desalter**
System Location: Chino, CA
PWSID: CA31610075

CASE #9 System Type: Community Water System

Treatment Type: Reverse Osmosis (RO) and Ion Exchange (IX)
Questionnaire completed by: Timothy Mim Mack, CDA Coordinator, City of Ontario
René Cruz, Engineering Project Manager, Project Partners Inc., serving the (CDA).
Startup Date: RO in 2000, IX added in 2005

Treatment Type	*Reverse Osmosis, Ion Exchange And Blending*
System Capacity	*RO: 4940 gpm* *IX: 3400 gpm*
Raw Water Nitrate	*147 – 303 mg/L as nitrate* *33 – 68 mg/L as N*

System Description

The Chino Basin Desalter Authority (CDA) in southern California is a conglomerate of the following agencies: Inland Empire Utilities Agency (IEUA), Jurupa Community Services District (JCSD), City of Chino, City of Chino Hills, City of Ontario, City of Norco, Santa Ana River Water Company (SARWC), and Western Municipal Water District (WMWD). The CDA drinking water treatment facilities include two desalters: Chino I Desalter (discussed here) and the Chino II Desalter (discussed in the next case study) to address high TDS levels as well as nitrate contamination. The Chino I Desalter operates 14 source wells, 11 of which have raw nitrate levels well above the MCL. Treatment consists of a combination of RO, conventional anion exchange and blending. Sixty percent of total flow is treated with RO, 27% with IX and 13% passes only through VOC/Air-stripping prior to blending. The RO system was installed in 2000 and the IX system was added in 2005.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N) (of nitrate impacted wells)
  - Average: ~147 – 303 (~33 – 68)
  - Minimum: ~114 – 289 (~26 – 65)
  - Maximum: ~161 – 351 (~36 – 79)
- Co-contaminants
  - TDS: 1100 mg/L

Treatment Technology Selection

RO and IX were selected because, combined, they were determined to be the best mode of technology to adequately treat the high-TDS, high-nitrate source water. No other technologies were pilot tested or considered prior to the installation of the system.

Adminsitrative Record
Page 35346

Treatment System Parameters

- Design Capacity
  - RO: 4940 gpm
  - IX: 3400 gpm
- Pretreatment
  - Anti-scalant: threshold inhibitor
  - Filtration: 1 micron pre-filters
  - pH adjustment: sulfuric acid
- Post-treatment
  - pH adjustment: sodium hydroxide
- Treatment system footprint
  - RO system: 143' X 80'
  - IX system: 190' X 60'
- Number of contactors
  - 4 trains, 2 stages/train
  - RO elements/stage
    - stage 1: 196 elements
    - stage 2: 98 elements
- Ion exchange pressure vessels
  - Number of vessels: 4
  - Diameter of vessels: 12'
  - Height of vessels: 11'
- Design Loading Rate: 1.66 gpm/ft^2
- Max. nitrate concentration goal for delivered water: 36 mg/L as nitrate (8.13 mg/L as N)
- Nitrate concentration goal for the treatment system (before blending): 10 mg/L as nitrate (2.25 mg/L as N)
- RO recovery rate: 80%
- RO membrane flux rate: 0.9 gfd/psi
- System Manufacturer
  - RO: Code-line
  - IX: Hungerford and Terry
- Membrane Type: Dow 400 BW-30
- Membrane Life: 5 years
- Membrane Cleaning
  - Flux decrease initiates CIP
  - Every 6 month
  - Chemicals: for pH adjustment based on manufacturer recommendation
- Resin Type: Rohm and Hass Amberjet 4400 CL SBA
- Volume treated prior to regeneration
  - 700,000 gallons
  - Regeneration once every 12 hrs
- Salt consumption: 75 tons per week
- Volume of brine/backwash
  - 53,000 gallons
  - 92.4% water efficiency
- Resin life: Has not been replaced (online for 5 years)
- Monitoring
  - Online nitrate analyzers
    - Treatment train
    - Blending point
    - POE
  - Laboratory samples
  - Quarterly testing for NDMA

Residuals Management

Concentrate/brine is discharged into a regional brine line called the Inland Empire Brine Line (IEBL) and formerly known as the Santa Ana Regional Interceptor (SARI).

Technology Benefits and Drawbacks

Benefits

- RO provides better removal
- IX is inexpensive
- IX has very low energy demands

Drawbacks

- RO is expensive
- High waste rate of RO
- IX does not address TDS
- Resin replacement will be costly

Adminsitrative Record
Page 35347

Treatment Technology Costs

Capital Costs (Based on projected costs in 2004)	
Treatment Plant Expansion Total (5000 afy expansion):	$6,379,530
Ion Exchange Treatment (4.9 MGD):	$4,031,900
Onsite Modifications:	$1,735,000
SARI Discharge Upgrades & Storm Drain:	$612,630
Additional SARI Capacity Purchase (not included in above total):	$4,140,000
O & M Costs (Based on CDA 2010/11 Budget, for complete plant operation, not just the treatment system)	
Total:	$7,496,315
Chemicals:	$662,257
Electricity, Plant Total:	$2,843,000
Operating Fees:	$1,353,439 (includes SARI fees, permits and other fees)
Labor ($):	$1,370,698

Additional Information

- The RO treatment system is described as falling short of expectations with respect to the high waste rate. 15% of all incoming water is sent to the brine line and delivered to a treatment plant outside of the local watershed at the Orange County Sanitation District.
- Plant shutdown has been required in the past due to high or low pressure, high nitrate, and high TDS.
- In the event of insufficient treatment and the production of water in exceedance of an MCL, the plant has an MOV that closes automatically, sending water to a storm drain.

Sources*

Listed costs are based on:
Chino Basin Desalter Authority. (2005) Presentation: Chino I Desalter Expansion & Chino II Desalter Project Update.
Chino Basin Desalter Authority. (2010) Fiscal Year 2010/11 Budget Adoption.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35348

System Name: Chino Basin Desalter Authority (CDA) – **Chino II Desalter**
System Location: Mira Loma, CA
PWSID: CA3310083

CASE #10 System Type: Community Water System

Treatment Type: Reverse Osmosis (RO) and Ion Exchange (IX)
Questionnaire completed by: Timothy Mim Mack, CDA Coordinator, City of Ontario
René Cruz, Engineering Project Manager, Project Partners Inc., serving the (CDA).
Startup Date: 2006

System Description

Treatment Type	*Reverse Osmosis, Ion Exchange And Blending*
System Capacity	*RO: 4167 gpm* *IX: 2778 gpm*
Raw Water Nitrate	*70 – 224 mg/L as nitrate* *16 – 51 mg/L as N*

The Chino Basin Desalter Authority (CDA) in southern California is a conglomerate of the following agencies: Inland Empire Utilities Agency (IEUA), Jurupa Community Services District (JCSD), City of Chino, City of Chino Hills, City of Ontario, City of Norco, Santa Ana River Water Company (SARWC), and Western Municipal Water District (WMWD). The CDA drinking water treatment facilities include two desalters: Chino I Desalter (discussed above) and the Chino II Desalter (discussed here) to address high TDS levels as well as nitrate contamination. The Chino II Desalter operates 8 source wells, all of which have raw nitrate levels well above the MCL. Treatment consists of a combination of RO, conventional anion exchange, and blending. The combined RO/IX system was installed in 2006.

Source Water Quality

- Nitrate - mg/L as nitrate (mg/L N) (of nitrate impacted wells)
  - Average: ~70 – 224 (~16 – 51)
  - Minimum: ~53 – 190 (~12 – 43)
  - Maximum: ~81 – 260 (~18 – 59)
- Co-contaminants
  - TDS

Treatment Technology Selection

RO and IX were selected because, combined, they were determined to be the best mode of technology to adequately treat the high-TDS, high-nitrate source water. No other technologies were pilot tested or considered prior to the installation of the system.

Adminsitrative Record
Page 35349

Treatment System Parameters

- Design Capacity
  - RO: 4167 gpm
  - IX: 2778 gpm
- Pretreatment
  - Anti-scalant: threshold inhibitor
  - Filtration: 1 micron pre-filters
  - pH adjustment: sulfuric acid
- Post-treatment
  - pH adjustment: sodium hydroxide
- Treatment system footprint
  - RO system: 30'X 188'
  - IX system: 30' X 188'
  - Total system: 60' X 300'
- Number of contactors
  - 3 trains
  - 48 vessels/train
  - RO elements/stage:
    - 7 elements per stage
- Ion exchange pressure vessels
  - Number of vessels: 4
- Design Loading Rate: 10.1 gpm/ft^2
- Max. nitrate concentration goal for delivered water: 25 mg/L as nitrate (5.7 mg/L as N) (goal), 35 mg/L as nitrate (7.9 mg/L as N) (max.)
- Nitrate concentration goal for the treatment system (before blending): 4 mg/L as nitrate (0.9 mg/L as N)
- RO membrane flux rate: 0.30 – 1.70 gfd/psi
- System Manufacturer
  - RO: PROTEC Bekaert
  - IX: Hungerford and Terry
- Membrane Type: Dow/Filmtec Model BW30-400
- Membrane Life: ~5 years
- Membrane Cleaning
  - Flux decrease initiates CIP
  - Every 6 months to 1X per year
  - Chemicals: King Lee (anti-scalant), high/low pH, Silica Cleaner
- Resin Type: Rohm and Hass Amberjet 4400 CL SBA
- Volume treated prior to regeneration
  - 0.8 – 1.4 MGD
  - Regeneration is based on nitrate levels
- Salt consumption: 50 tons per week
- Volume of brine/backwash
  - NA
- Resin life: NA
- Monitoring
  - Online nitrate analyzers
    - At source
    - Blending point
    - POE
  - Laboratory samples

Residuals Management

Concentrate/brine is discharged to an industrial sewer that drains to the Inland Empire Brine Line (IEBL). Waste is transported 45 miles to the Orange County Sanitation District.

Technology Benefits and Drawbacks

Benefits

- RO works well for nitrate removal
- IX is less expensive

Drawbacks

- RO is expensive
- IX does not accomplish contaminant removal as well as RO

Adminsitrative Record
Page 35350

Treatment Technology Costs

Capital Costs (Based on projected costs in 2004)	
Treatment Plant Total (10,400 AFY):	$19,171,837
Ion Exchange Treatment:	$4,346,900
Chino II Desalter:	$14,284,500
RO Membranes:	$540,437
Ion Exchange Land (not included in above total):	$1,730,138
Ion Exchange SARI Fee (not included in above total):	$10,105,000
O & M Costs (Based on CDA 2010/11 Budget, for complete plant operation, not just the treatment system)	
Total:	$6,111,799
Chemicals:	$615,000
Electricity – Plant Total:	$2,331,000
Operating Fees:	$706,154 (includes SARI fees, permits and other fees)
Labor ($):	$1,133,615

Additional Information

- Operator certification levels range from T-3 to T-5.
- Plant shutdown has been required in the past due to chemical pump failure and a high clearwell.
- Overall the combined system is described as working well but at a high price.

Sources*

Listed costs are based on:
Chino Basin Desalter Authority. (2005) Presentation: Chino I Desalter Expansion & Chino II Desalter Project Update.
Chino Basin Desalter Authority. (2010) Fiscal Year 2010/11 Budget Adoption.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35351

3.3 Electrodialysis (ED/EDR/SED)

The use of electrodialysis (ED), including electrodialysis reversal (EDR) and selective electrodialysis (SED), in potable water treatment (Figure 10) has increased in recent years, offering the potential for improved water recovery, the ability to selectively remove nitrate ions, and the minimization of chemical and energy requirements (Kneifel & Luhrs 1988; Kapoor & Viraraghavan 1997; Hell et al. 1998; Koparal & Ogutveren 2002; Midaoui et al. 2002; Sahli et al. 2008; Banasiak & Schafer 2009).

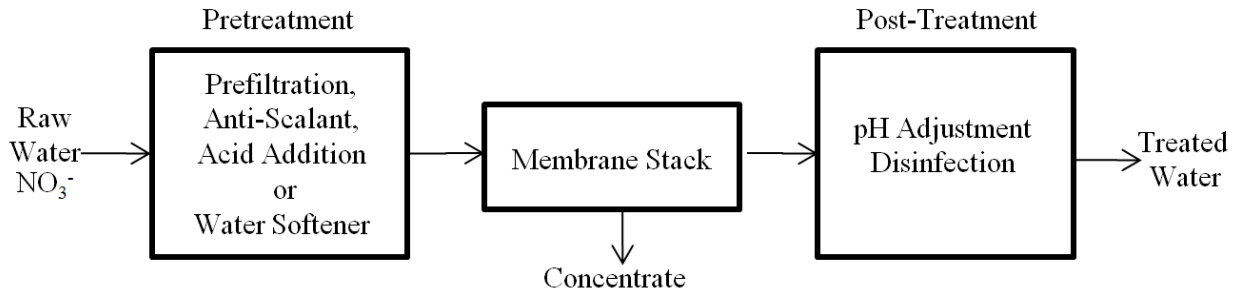


Figure 10. Electrodialysis reversal schematic.

Nitrate removal is accomplished by passing an electrical current through a series or stack of anion and cation exchange membranes, resulting in the movement of ions from the feed solution to a concentrated waste stream. Illustrated in Figure 11, nitrate ions (and other anions) move through the anion exchange membrane toward the anode. Continuing toward the anode, nitrate is rejected by the anion-impermeable cation exchange membrane and trapped in the recycled waste stream. Cations can be removed in a similar manner, migrating toward the cathode through the cation exchange membrane and rejected by the cation-impermeable anion exchange membrane. Nitrate selective membranes allow for treatment without significantly altering the balance of other ions in the water.

The electrical current is passed through the system with the migration of ions across the membranes. For every anion that leaves a compartment, a cation of equivalent charge also leaves, maintaining the charge balance in each compartment. Across the system, the flow of electrons, moving from the cathode to the anode (negative to positive), is governed by the movement of ions through the membrane stack and by the reactions in the electrode compartment. Small levels of gaseous byproducts must be removed. Electrolysis of water generates oxygen at the anode and hydrogen gas at the cathode and chloride can be reduced at the anode, producing chlorine gas (AWWA 1995). The electrode compartment is rinsed to restore ions for current transfer and to remove unwanted reaction products.

Adminsitrative Record
Page 35352

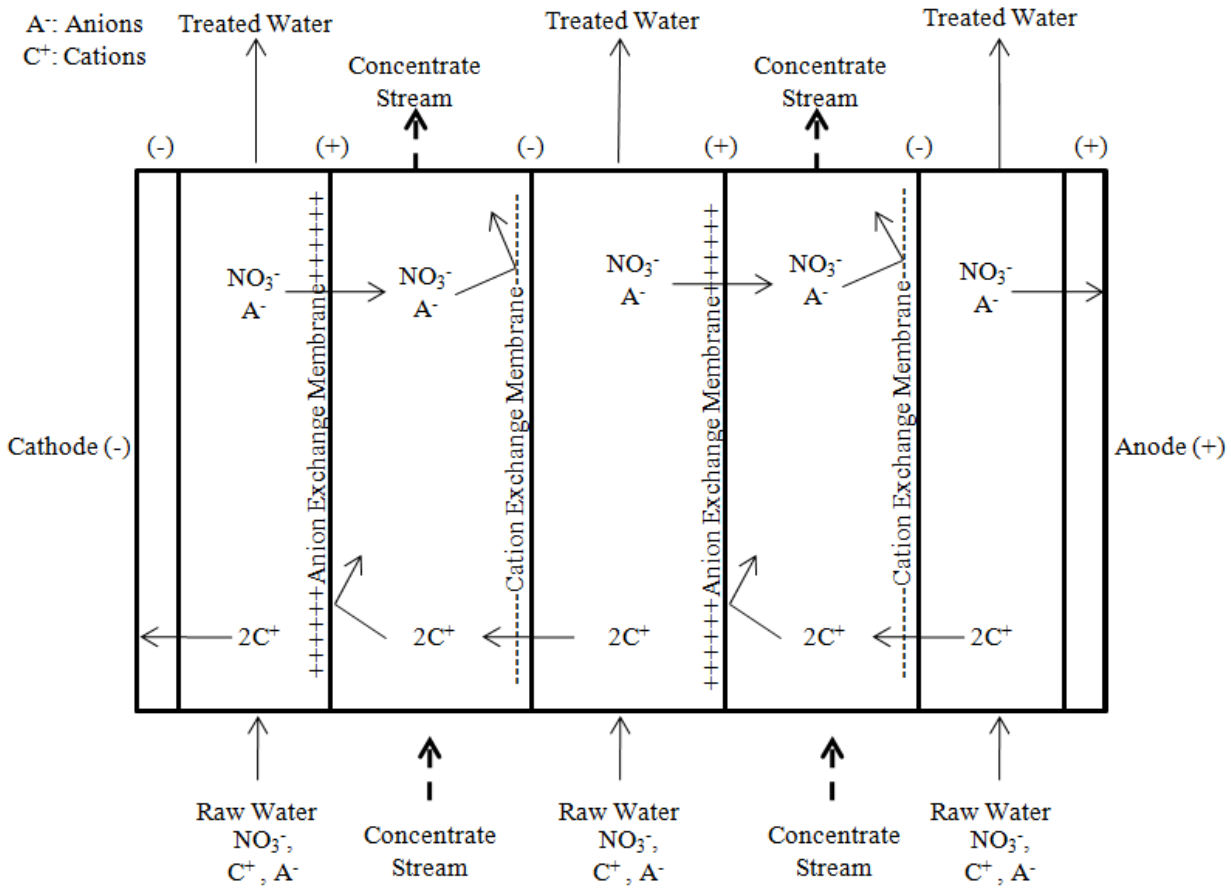


Figure 11. Illustration of electrodialysis membrane stack.

Requiring constant electrical current and low-pressure water, ED has inherent energy demands. However, voltage adjustment enables selective demineralization. "Plants can be designed to remove from 50 to 99 percent of source water contaminants or dissolved solids. Source water salinities of less than 100 mg/L up to 12,000 mg/L TDS can be successfully treated to produce finished water of less than 10 mg/L" (AWWA 1995, p. 7).

A detailed case study of an EDR plant in Spain is included in section 3.3.6 Electrodialysis - Case Studies.

Adminsitrative Record
Page 35353

3.3.1 Electrodialysis - Design Considerations

Table 8 summarizes key design considerations in the use of ED for nitrate removal from potable water.

Table 8. Summary of design considerations for electrodialysis/electrodialysis reversal.

Membranes	• Use of anion and cation exchange membranes • Selective membranes ○ Monovalent versus multivalent • Consider water recovery and frequency of cleaning

Pretreatment	• Lower pretreatment requirements because this is not direct filtration
	• EDR systems can avoid or limit chemical use
	• Prevention of scaling and fouling
	  ○ Filtration to remove suspended solids
	  ○ Treatment for iron and manganese removal
	  ○ Water softening or use of anti-scalants or acid to prevent scaling

Post-Treatment	• pH adjustment to avoid corrosion (if acid used to prevent scaling) • Disinfection
Chemical Usage	• Possible pH adjustment (acids and bases) • Possible anti-scalants • Possible cleaning chemicals

O&M	• Highly automated
	• Frequency of membrane cleaning depends on water quality and membrane used
	  ○ Polarity reversal (electrodialysis reversal) multiple times per hour minimizes fouling
	  ○ ED systems can require weekly cleaning
	• Management of chemicals and pre-filtration system
	  ○ Including electrode compartment rinse solution
	• Waste storage and disposal
	• High monitoring demands
	• Potentially higher operator demand than IX and RO, due to system complexity

System Components	• Maximize water recovery while minimizing energy use • Key aspects of the system are pretreatment requirements, number and configuration of electrodialysis stacks and stages, membrane selection and configuration, operating voltage and pressure, reversal frequency (for EDR), gas venting of anode and cathode compartments, disinfection, “brine loop, electrode rinse loop, concentrate discharge, and dosing station” (Hell et al. 1998, p. 178).
Waste Management and Disposal	• Concentrate disposal of greatest concern for inland systems ○ Close proximity to coastal waters is beneficial for brine/concentrate disposal • Management options include sewer or septic system, reuse for irrigation, drying beds, trucking off-site, coastal pipeline, deep well injection and advanced treatment • Disposal options can be limited by waste brine/concentrate water quality (e.g., volume, salinity, metals, and radionuclides) • Optimization of recycling and treatment of waste concentrate
Limitations	• Need to prevent membrane scaling and fouling ○ Hardness, iron, manganese, and suspended solids • Disposal of waste concentrate • High system complexity

Adminsitrative Record
Page 35354

Water Quality

Membrane life, cleaning frequency, and pretreatment needs are dependent on feed water quality. Pretreatment may be needed for iron levels above 0.3 mg/L, manganese levels above 0.1 mg/L, and hydrogen sulfide levels exceeding 0.3 mg/L (WA DOH 2005). Specifications for an example EDR system from GE indicate feed water turbidity levels should be < 0.5 nephelometric turbidity units (NTU) with typical TDS levels between 100 and 3,000 mg/L (maximum 12,000 mg/L) (GE 2008). SDI limits are generally higher for EDR than for RO, with typical limits of 12 and 4 – 5, respectively (Elyanow & Pereschino 2005). Softening may be necessary to reduce hardness, while pre-filtration diminishes suspended solids. The potential for scaling increases with increasing TDS and is exacerbated by increased solids precipitation with higher water recovery goals. To minimize fouling/scaling, membranes can be treated with anti-scaling chemicals and cleaned with acid (AWWA 1995). However, in comparison with other membrane processes, fouling is minimal because the membrane is subjected to the transfer of ions (directed by the electrical current), rather than the transfer of the entire feed stream. Unfortunately, because ED does not serve as a filter (the water does not pass through the membrane), ED fails to remove microbial contamination (AWWA 1995). Pre-filtration in pretreatment and disinfection in post-treatment address these concerns.

To further minimize fouling and thus the need for chemical addition, the polarity of the system can be reversed with electrodialysis reversal (EDR). By reversing the polarity (and the solution flow direction) several times per hour, ions move in the opposite direction through the membranes, minimizing buildup and the need for chemical addition to control scaling. Biological fouling concerns are lower than other separation processes due to development of membranes that are "more organic resistant and chlorine tolerant" (Elyanow & Persechino 2005, p. 8). ED depends on the transfer of an electrical current and is therefore more efficient when used for brackish waters. In low conductivity feed waters, the ion removal efficiency declines. In contrast to conventional RO, EDR is unaffected by silica.

System Components and Site Considerations

ED and EDR systems are operated in stages. Water recovery can be improved with stages operated in series while capacity can be increased with stages operated in parallel. Key aspects of the system are pretreatment requirements, the number and configuration of electrodialysis stacks and stages, membrane selection and configuration, operating voltage (based on desired removal), reversal frequency (for EDR), gas venting of the anode and cathode compartments, disinfection, "brine loop, electrode rinse loop, concentrate discharge, and dosing station" (Hell et al. 1998, p. 178).

The membranes used in ED/EDR are anion and cation exchange membranes. Membranes have been designed for selective removal based on valency (monovalent versus multivalent) to screen for particular constituents (AWWA 1995). Alternating different selective membranes in the membrane stages can avoid precipitation in the concentrate stream. For example, one stage can remove calcium and a second stage can remove sulfate (to an alternate concentrate stream), this prevents calcium sulfate precipitation (AWWA 1995).

Adminsitrative Record
Page 35355

Residuals Management and Disposal

Waste management requirements are similar to RO and IX; however, the burden of disposal in ED/EDR systems is not as significant due to higher water recovery, selective removal, and the lack of direct filtration (Reahl 2006). Disposal options include sewers, septic systems, drying beds, off-site trucking, coastal pipeline, deep well injection, reuse for irrigation, and advanced treatment. Important water quality characteristics of the concentrate (e.g., volume, salinity, metals, and radionuclides) can affect the feasibility and costs of disposal options. Several combined configurations of interest are discussed in Section 3.6 Brine Treatment Alternatives and Hybrid Treatment Systems.

Maintenance, Monitoring, and Operational Complexity

Although ED/EDR systems are amenable to automation, operator demands can be higher than other separation processes (AWWA 1995). While ED systems have greater pretreatment demands and can require membrane cleaning once a week, EDR systems minimize pretreatment demands and scaling issues, but can still have higher maintenance demands than RO, due to the complexity of the system (Kapoor & Viraraghavan 1997). Appropriate gas venting is important to avoid hazardous conditions (AWWA 1995). Membrane life will depend on water quality and pretreatment measures. However, due to the lack of direct filtration and operation under low pressure, membranes are long lasting, and do not require frequent replacement.

3.3.2 Electrodialysis - Cost Considerations

For the efficient operation of an ED system, the fundamental objective is to maximize water recovery with the minimum amount of energy and chemical usage, while meeting necessary potable water guidelines. Factors affecting system cost include facility size (how much water), source water quality (including nitrate concentration and other contaminants), target effluent nitrate concentration, and disposal options.

Capital costs for ED/EDR systems include land, housing, piping, storage tanks, O&M equipment, cation and anion exchange membranes, preliminary testing (pilot studies), permits, and training. O&M costs include membrane replacement, membrane disposal, concentrate disposal or treatment, chemical use (limited: anti-scalant, etc.), repair, maintenance, power, and labor.

Very little published cost information from existing ED systems used for nitrate removal is available in the literature, due to the limited number of full-scale systems. Costs of ED systems are most comparable to RO. However, in some instances, ED can be the less costly choice due to the greater pretreatment and post-treatment demands (higher chemical use and post-treatment pH adjustment) of RO (Reahl 2006). EDR can be chosen over RO when high water recovery is a priority, especially if land must be purchased for concentrate ponds. “New technology has also reduced the capital and operating cost of EDR nitrate removal by increasing the hydraulic efficiency of the EDR stacks and pumping system” (Elyanow & Persechino 2005, p. 8). Costs listed here have been adjusted to 2010 dollars, unless indicated otherwise. In a technical paper from GE Water & Process Technologies (the primary supplier

Adminsitrative Record
Page 35356

of EDR systems in the U.S.), Werner & Gottberg (2005) present O&M costs of an electrodialysis plant in Suffolk, VA (not specifically for nitrate treatment) (Table 9). It is unclear how disposal costs were included in this study. High water recovery in comparison with other removal processes and disposal of waste concentrate to a nearby estuarine tributary would maintain low disposal costs (Werner & Gottberg 2005). According to Ameridia, the American division of Eurodia Industrie (a manufacturer of EDR systems), the capital investment for a nitrate treatment EDR unit for a ~0.5 MGD system (in 2005) was $475,000 or $0.94 per gallon of daily capacity ($559,653 or $1.11 in 2010 dollars, respectively) (Ameridia). However, additional capital costs are likely not included in this figure. A detailed discussion of treatment costs is included in Section 6 Treatment Cost Analysis.

Table 9. Sample EDR O&M costs (from Werner & Gottberg 2005).

O&M Category	EDR (/1000 gallons)[1]
Fixed	$0.72 ($1.07 adjusted)
Professional Services	$0.06 ($0.09 adjusted)
Chemicals	$0.02 ($0.03 adjusted)
Utilities	$0.21 ($0.31 adjusted)
Maintenance	$0.17 ($0.25 adjusted)
Membrane Replacement	$0.23 ($0.34 adjusted)
Production (1997)	827,339,440 gallons
Total O&M Cost	$1.41 ($2.09 adjusted)

[1] Costs adjusted from 1998 dollars to 2010 dollars.

The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing ED may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation.

3.3.3 Electrodialysis - Selected Research

Much research on ED has focused on desalination applications. Table A.3 of the Appendix lists recent studies relevant to nitrate removal from potable water and several examples of ED application. Research is focused on the influence of co-contaminants on system performance and improvements in exchange membranes, including nitrate selectivity.

3.3.4 Electrodialysis - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of ED in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of ED/EDR systems include low chemical usage, long lasting membranes, selective removal of target species, flexibility in removal rate (through voltage control), good water recovery rate, feasible automation, and multiple contaminant removal (Prato & Parent 1993; AWWA 1995; Hell et al. 1998; WA DOH 2005). With the ability to selectively remove multiple contaminants, ED/EDR systems can be used to address the following constituents: TDS, total chromium, chromium-6, arsenic, perchlorate, sodium, mercury, chloride, copper, sulfate, uranium, fluoride,

Adminsitrative Record
Page 35357

nitrate/nitrite, iron, selenium, hardness, barium, bicarbonate, cadmium, and strontium (AWWA 1995 and GE 2010). Using current reversal, EDR offers additional advantages, improving system performance by "detaching polarization films, breaking up freshly precipitated scale or seeds of scale before they can cause damage, reducing slime formations on membrane surfaces, reducing problems associated with the use of chemicals, and cleaning electrodes with acid automatically during anodic operation" (AWWA 1995, p. 9, 10). Additionally, in comparison with RO systems, EDR can treat waters with higher SDI, silica, and chlorine levels (Elyanow & Persechino 2005).

Disadvantages of ED/EDR systems include the possible need for pretreatment to prevent membrane scaling and fouling, waste disposal, high maintenance demands, costs (comparable to RO systems), the need to vent gaseous byproducts, the potential for precipitation (especially for high water recovery), high system complexity, and limited manufacturers with U.S. experience (e.g., GE is the primary source of EDR systems for drinking water in the U.S.). Additionally, unlike RO, ED does not remove uncharged constituents in the water.

3.3.5 Modifications to Electrodialysis

Selective Electrodialysis (SED)

Since 1997, selective electrodialysis (SED), developed by Shikun & Binui, formerly Nitron, Ltd., has been successfully implemented throughout Israel, reducing national water costs by 55%. The manufacturer indicates that SED offers high water recovery (up to 95%), thereby minimizing waste volume (Nitron 2010). The SED system is accepted by the U.S. EPA as a nitrate treatment option for large plants (Nitron 2009). While similar to traditional ED processes, SED utilizes nitrate selective membranes which have been shown to increase operational performance when used for nitrate treatment.

The nitrate selective membranes used in the SED process have been shown to remove up to 70% of nitrate from solution. At the same time, sulfate ions and carbonate ions, which have a tendency to cause scaling issues in the concentrate stream of traditional ED/EDR and RO technologies, are more readily rejected by the nitrate selective membranes used in the SED process. As a result, the scaling potential is reduced in the concentrate stream. Since scaling problems are minimized, membrane cleaning frequency, maintenance costs, and down time are reduced compared to traditional EDR installations. Another important aspect of membrane selectivity is the energy efficiency of the process. Energy efficiency is related to the extent of ion transfer in ED/EDR and SED technologies. In traditional ED/EDR, energy use is less focused, resulting in the removal of many ions, including ions that do not need to be addressed. SED specifically targets nitrate ions, avoiding energy use for the removal of non-target ions and improving energy efficiency.

Pretreatment can be limited to filtration, energy efficiency is maximized due to low pressure operation (2 – 4 bars, ~30 – ~60 psi), chemical use is limited to concentrate treatment, and low maintenance demands are possible due to automation, remote monitoring and control, and infrequent cleaning. In the SED process there is no change in the pH of the product water. This avoids the need for pH adjustment or remineralization in post-treatment (Nitron 2009; Nitron 2010). Membranes are cleaned

Adminsitrative Record
Page 35358

in place (CIP) for 1 hour every 4 – 6 months and membrane life is typically 7 – 10 years (Nitron 2010). Additional advantages of SED include constant membrane performance, no chemical contact with potable water, the simplicity of the system consisting of pre-filtration and membrane stacks (UV can be added for disinfection), and a small footprint (Nitron 2009b). Potential drawbacks of SED include the lack of full-scale application in the U.S. for nitrate removal from drinking water and, unlike RO, ED does not remove uncharged constituents in the water.

A detailed case study of the use of SED for nitrate removal at locations in Israel is included in the following section.

3.3.6 Electrodialysis - Case Studies

The following case studies provide detailed information on the design and operation of full-scale EDR and SED treatment plants used for nitrate removal.

Adminsitrative Record
Page 35359

System Name: Gandia EDR
System Location: Valencia, Spain
System Type: NA

CASE #11

Treatment Type: Electrodialysis Reversal (EDR)
Questionnaire completed by: GE Water & Process Technologies
Startup Date: 2007

Treatment Type	*Electrodialysis Reversal*
System Capacity	*2 systems at 4.7 MGD each*
Raw Water Nitrate	*80 mg/L as nitrate*
	18 mg/L as N

System Description

Gandia is a tourist area on the Mediterranean coast of Spain. The area sees peak demand during the summer months when the population almost triples. The Spanish Legislature released royal Decree 140/2003 which changed the nitrate limit to 50 mg/L as nitrate (11.3 mg/L as N). This new law required treatment of the existing system to achieve the new nitrate limits. Additionally, the existing well systems had deteriorated over time, forcing the municipality to find alternate wells to feed the community.

Upon analysis of the wells (old and new), it was determined that the nitrate levels were too high to meet the drinking water standard. The well samples had up to 80 mg/L as nitrate (18.1 mg/L as N). Treatment was necessary to produce acceptable levels of nitrate in the product water. An evaluation was conducted and EDR was selected as the technology of choice for the Gandia treatment plants. EDR offered high recovery while effectively reducing the nitrate levels below 25 mg/L as nitrate (5.6 mg/L as N).

EDR was piloted on the wells to verify the nitrate removals and operating cost estimates for power requirements and chemical consumption. The pilot study was successful, and the final systems were designed around 90% water recovery with the overall nitrate removal of 73%.

Source Water Quality

- Nitrate (mg/L N)
  - < 80 mg/L as NO_3^-

Treatment Technology Selection

EDR, Selective ED, and Reverse Osmosis were considered for treating the Gandia Wells. EDR was eventually selected for the high recovery and reduced operating costs compared to RO. SED was ruled out based on the high capital costs of the system.

Adminsitrative Record
Page 35360

Treatment System Parameters

- Design Capacity
  - 3,260 gpm
- EDR System
  - No. of modules: 4
  - Lines per module: 5
  - Stages per line: 2
- Water recovery rate
  - 94.3%

Water Quality Results

The table below summarizes the water quality of the raw water, finished water, concentrate stream, and the total waste from the Gandia EDR facility. Total waste includes concentrate blowdown, electrode waste, and off-spec product from the system.

Ion	Raw Water	Treated Water	Percent Removal	Concentrate Stream	Total Waste
Ca	82	24.9	70%	772.6	544.3
Mg	24	8.3	65%	213.7	151.0
Na	23	10	57%	180	128.1
K	1.0	0.3	70%	9.2	6.5
HCO_3	250	99.1	60%	2074.1	1471.2
SO_4	58	12.7	78%	605.2	424.4
Cl	29	7.5	74%	289.3	203.3
NO_3	60	16.6	72%	584.4	411.1
TDS	527	179.3	66%	4728.6	3339.8
pH	7.5	7.1		8.3	8.1

Technology Benefits and Drawbacks

Benefits

- EDR membranes are chlorine tolerant, providing means to control biological growth.
- Relatively low operational expenditures:
  - Membrane life expectancy is 15 years.
  - Low chemical consumption compared to other technologies.
  - Lower energy consumption compared to RO.
- High water recovery, small concentrate stream for disposal compared to other technologies.

Drawbacks

- Higher capital cost than RO.
- System footprint larger than competitive technologies.

Adminsitrative Record
Page 35361

Operating Costs

Capital costs for the EDR system were not provided.

O & M Costs (Total with explanation or component costs)		
	Unit	Cost
Labor:	$/1,000 gal	0.17
Energy:	$/1,000 gal	0.15
Maintenance:	$/1,000 gal	0.03
Chemicals:	$/1,000 gal	0.10
Consumables:	$/1,000 gal	0.19
Overhead:	$/1,000 gal	0.04
Total:	$/1,000 gal	0.67

Operational Notes

Operating costs were based on estimates prior to plant start-up. After four years of operating, the plant has not replaced any membranes. In 2010, another facility (L'Eliana) was commissioned in the Valencia area for 2.9 MGD production rate for nitrate removal using the EDR technology.

Sources*

Cháfer , V.S., Carbonell, J.S., and de Armas Torrent, J.C. Nitrate and Hardness Removal with Electrodialysis Reversal (EDR) in Gandia, (Valencia, Spain).

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35362

System Name: Weizmann Institute
System Location: Rechovot, Israel
System Type: NA

CASE #12

Treatment Type: Selective Electrodialysis (SED)
Questionnaire completed by: Shikun & Binui Environmental Group
Startup Date: 2008

Treatment Type	*Selective Electrodialysis*
System Capacity	*310 gpm*
Raw Water Nitrate	*84 – 89 mg/L as nitrate*
	19 – 20 mg/L as N

System Description

The Weizmann Institute of Science (Institute), located in Rechovot, Israel, is one of the top-ranking multidisciplinary research institutions in the world. In 2007, the nitrate MCL in the Israeli National drinking water regulations changed from 90 mg/L (20.3 mg/L as N) to 70 mg/L (15.8 mg/L as N). Because of this change, two of the Institute's wells were removed from the drinking water supply and the Institute had to rely on external water suppliers. Over time, municipal and national water costs increased. The Institute's management looked for solutions to solve nitrate problems and enable them to reopen the Institute's wells.

Prior to treatment, the wells were used for potable purposes and the Institute's irrigation needs. Selective Electrodialysis (SED) was identified as the Institute's most appropriate treatment technology. An SED system was implemented for the 310 gpm well. The Institute opted for treatment of one well and uses the second well as a dedicated irrigation supply source. The nitrate enriched concentrate from the SED process is fed into the non-potable irrigation system where the nitrate enhances plant growth.

SED has been successfully implemented throughout Israel since 2007. Developed by Nitron, Ltd., SED offers high water recovery (up to 95%), thereby minimizing waste volume. Pretreatment can generally be limited to filtration, energy efficiency is maximized due to low pressure operation (30 – 60 psi), chemical use is limited to concentrate treatment (no need for chemical addition to feed or product water), and low maintenance demands are possible due to automation, remote monitoring and control and infrequent cleaning. Membranes are cleaned in place (CIP) for 1 hour every 4 – 6 months and membrane life is typically 7 – 10 years.

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
  - Average – 84 (19)
  - Minimum – 84 (19)
  - Maximum – 89 (20)

Treatment Technology Selection

SED and RO were considered for treating the Weizmann Institute well. A 10 year life cycle cost analysis that included capital and operations costs identified SED as the more cost-effective solution.

Adminsitrative Record
Page 35363

Treatment System Parameters

- Design Capacity
  - 310 gpm
- Pretreatment
  - Cartridge filtration
- Post-treatment
  - Chlorination
  - Acid addition (pH 4.5 – 5) to concentrate to prevent the precipitation of calcium carbonate and calcium sulfate in the concentrate cells
- Treatment system footprint
  - Treatment system: 8' x 5'
  - Building footprint: Butler building - 50' x 13' as per client request
- Water recovery rate
  - 94.3%
- SED unit information:
  - Number of SED units: 1
  - Membrane pairs: 240
  - 59% Nitrate reduction
  - 30% TDS reduction
  - Energy consumption: 2.3 KWh/1,000 gal
- Monitoring:
  - Online nitrate analyzer
  - Laboratory nitrate samples
  - Online pH meters
  - Online turbidity
  - Online conductivity meters

Water Quality Results

The table below summarizes the water quality of the raw water, finished water, and concentrate stream from the Weizmann Institute.

Ion	Raw Water	Treated Water	Percent Removal	Concentrate Stream
Cl	194	105	45.9%	2051
SO_4	95	92	**3.0%**	140
HCO_3	232	165	28.7%	442
NO_3	92	44	**52.2%**	835
Na	109	76	30.4%	627
Ca	118	73	37.8%	808
Mg	24	15	36.6%	160
K	3.6	2.5	30.6%	20.5
Ba	0.149	0.093	37.8%	1.020
Sr	0.84	0.52	37.8%	5.75
TDS	869	574	**33.9%**	5090
pH	7.8	7.8		6.5

Residuals Management

The concentrate from the SED process is fed into the Weizmann Institutes non-potable system which is used for irrigation purposes. Since the SED process selectively removes nitrate, the Total Dissolved Solids (TDS) of the concentrate is less than an RO system would be if it were treating the same water, which allows the concentrate to be used for irrigation without the salinity adversely affecting plant growth. This management approach is also beneficial since the concentrated nitrate solution has limited the amount of fertilizer applied by the Weizmann Institute.

Adminsitrative Record
Page 35364

Technology Benefits and Drawbacks

Benefits

- Ease of regeneration
- Fewer chemicals than comparable technologies
- Membranes 7 – 10 year life span
- Energy consumption less than that of RO
- High water recovery
- Concentrate solution has relatively low TDS which may increase disposal options

Drawbacks

- Capital intensive technology
- Requires specific operator training

Treatment Technology Costs

Capital Costs (Total with explanation or component costs)		
	$	Comments
Total:	650,000	
Housing:		Light building 50X13 feet. According to customer demands.
Piping:	50,000	Pipes, electric valves, storage tanks (the largest would be with a volume of about 180 cu. ft. for product water).
Storage Tanks (include description of uses):		See above.
Operating and Monitoring Equipment:	90,000	Control system, remote assistance for the control system, on line nitrate measurement, conductivity, pH, turbidity.
Membranes Modules:	300,000	Complete SED membrane stack.
Permits:		According to local regulations.
Other:	210,000	Design; Electricity boards – design, manufacturing & installation; Pumps and blowers.

O & M Costs (Total with explanation or component costs)		
	Unit	Comments
Membrane:	%/year	10% per year is the replacement rate.
Concentrate Disposal or Treatment:	Gallons/year	Site specific; 4 – 5% of the SED system capacity.
Chemicals:	Total, [lb/1000 gallon]	0.77
Specific acid consumption	[lb/1000 gallon]	0.72
Specific chlorine consumption	[lb/1000 gallon]	0.04
Specific caustic soda consumption	[lb/1000 gallon]	0.01
Repair/Maintenance (not including Labor):	$/year	7,000
Specific power consumption	[kWh/1000 gallon]	1.4
Labor ($):	$	16,000 (40$ per hour basis)
Labor (Hours per Year):	hours	400

Adminsitrative Record
Page 35365

Operational Notes

The SED system is highly instrumented and the PLC has over 200 monitored inputs. As a result there have not been any failures that have resulted in MCL violation. While there have been failures resulting in alarms and shutdowns, the control system has been robust enough to shut down the system and prevent water with high nitrate entering the distribution system. Typical shutdowns can be rectified in a matter of hours and normal operation is resumed.

Sources*

Merhav, Neta. (2010) Completed questionnaire. October, 2010.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35366

3.4 Biological Denitrification (BD)

Commonly used in wastewater treatment, biological reactors are emerging as a method for denitrification of potable water with the potential to address multiple contaminants including nitrate, chromate, perchlorate, and trace organic chemicals (Brown 2008). Biological denitrification (BD) in potable water treatment has been implemented in Europe since 1804 (Lenntech 2009), with recent full-scale systems in France, Germany, Austria, Poland, Italy, and Great Britain (Meyer 2009; Dördelmann 2009). To date, full-scale drinking water applications in the United Sates have been limited to a single plant in Coyle, OK (no longer online). However, two full-scale biological denitrification systems are anticipated in California within the next couple of years.

Denitrification occurs naturally in the environment as part of nitrogen cycling. Application of biological denitrification to potable water treatment (Figure 12) utilizes denitrifying bacteria to reduce nitrate to innocuous nitrogen gas in the absence of oxygen (anoxic conditions).

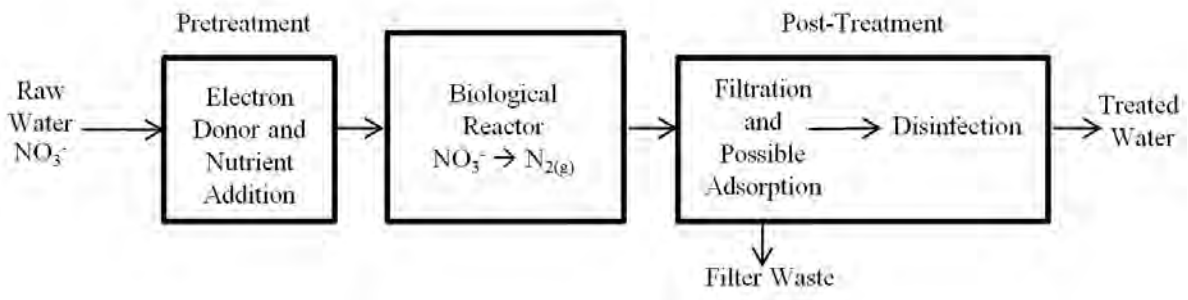


Figure 12. Biological denitrification schematic.

The reduction of nitrate proceeds stepwise in accordance with Eqn. 9. In contrast with the separation processes of IX, RO, and ED/EDR, nitrate is reduced and thereby removed from the system rather than simply being displaced to a concentrated waste stream.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad \text{(Eqn. 9)}$$

Denitrifying bacteria require an electron donor (substrate) for the reduction of nitrate to nitrogen gas. In conventional wastewater treatment, substrate addition is not typically needed, because the wastewater contains sufficient carbon for denitrification to occur. However, depending on the source, substrate addition is often required for the biological denitrification of potable water. The addition of a carbon substrate in potable water treatment is somewhat counter intuitive. In fact, one principal objective of potable water treatment is to minimize dissolved carbon in the water to minimize growth of microbes (e.g., biofilms) and production of disinfectant byproducts (e.g., THMs). Feed water composition may need to be further augmented with the addition of nutrients required for cell growth (phosphorus for example). Autotrophic bacteria utilize sulfur or hydrogen as an electron donor and inorganic carbon (typically carbon dioxide) as a carbon source for cell growth (Eqns. 10 and 11), while heterotrophic bacteria consume an organic carbon substrate, like methanol, ethanol, or acetate (Eqn. 12) (Mateju et al. 1992; Kapoor & Viraraghavan 1997).

Adminsitrative Record
Page 35367

Eqns. 10 through 12 illustrate the overall denitrification reaction defining the stoichiometric relationship between electron donor, carbon source, and nitrate in the production of cells and the conversion of nitrate to nitrogen gas. Not all nitrogen is converted to nitrogen gas. Some nitrogen is required for cell growth. The governing stoichiometric equation indicates the necessary dose and varies with the substrate used. For example, the stoichiometric factor for acetic acid is 0.82 moles of acetic acid per mole of nitrate (Dördelmann et al. 2006).

$$11\ S^0 + 0.5\ CO_2 + 10\ NO_3^- + 2.54\ H_2O + 1.71\ NH_4^+ \rightarrow 0.92\ C_5H_7O_2N + 11\ SO_4^{2-} + 5.4\ N_2 + 9.62\ H^+ \quad \text{(Eqn. 10)}$$

$$H_2 + 0.35\ NO_3^- + 0.35\ H^+ + 0.052\ CO_2 \rightarrow 0.010\ C_5H_7O_2N + 0.17\ N_2 + 1.1\ H_2O \quad \text{(Eqn. 11)}$$

$$1.08\ CH_3OH + NO_3^- + H^+ \rightarrow 0.065\ C_5H_7O_2N + 0.467\ N_2 + 0.76\ CO_2 + 2.44\ H_2O \quad \text{(Eqn. 12)}$$

Various species of bacteria are responsible for denitrification including *Thiobacillis denitrificans, Micrococcus denitrificans, Pseudomonas maltophilia* and *Pseudomonas putrefaciens* (Kapoor & Viraraghavan 1997). Due to slower bacterial growth rates, autotrophic denitrification offers the advantage of minimizing biomass accumulation; however, autotrophic denitrification requires alkalinity to supply the inorganic carbon source (carbon dioxide) for cell growth (Della Rocca et al. 2006). BIODEN® and DENICARB® are heterotrophic biological denitrification processes, while DENITROPUR® is an autotrophic option. Selected full-scale biological denitrification systems are listed in Table 10 (Dördelmann 2009).

Table 10. Full-scale biological denitrification systems for potable water treatment.[1]

Location	Reactor Configuration	Substrate, Denitrification type	Flow rate m^3/h (MGD)
Germany			
Neuss	Fixed bed, down-flow	Acetic acid, Heterotrophic	150 (0.95)
Frankfurt Airport	Fluidized bed, up-flow, DENICARB®	Ethanol, Heterotrophic	320 (2.03)
Aschaffenburg	Fixed bed, up-flow, DENITROPUR®	Hydrogen and CO_2, Autotrophic	1600 (10.14)
Föhr Island			90 (0.57)
Austria			
Obersiebenbrunn	Fixed bed, down-flow, BIODEN®	Ethanol, Heterotrophic	180 (1.14)
Poland			
Czestochowa	Fixed bed, down-flow, BIODEN®	Ethanol, Heterotrophic	500 (3.17)

[1] Dördelmann 2009.

In their review of potable water treatment methods for the removal of nitrate, Mateju et al. (1992), Kapoor & Viraraghavan (1997), Soares (2000), and Shrimali & Singh (2001) discuss previous research and applications of biological denitrification. Problems associated with the application of conventional biological denitrification to potable water treatment include additional post-treatment for removal of biomass and dissolved organics, the potential for incomplete denitrification, increased capital costs, and sensitivity to environmental conditions (Kapoor & Viraraghavan 1997). To address these concerns, several treatment configurations using biological denitrification have been developed.

Adminsitrative Record
Page 35368

3.4.1 Biological Denitrification - Design Considerations

Table 11 summarizes key design considerations in the application of biological denitrification.

Table 11. Summary of design considerations for biological denitrification for nitrate removal in potable water.

Pretreatment	• Substrate and nutrient dosing • pH adjustment

Post-Treatment	• Carbon adsorption for organic carbon removal
	  ○ Carbon adsorption is not always required
	  ○ Residual substrate removal can be accomplished via biological filtration
	• Aeration
	• Filtration
	• Disinfection

Chemical Usage	• Possible pH adjustment • Substrate and nutrient addition • Coagulant/polymer use to meet turbidity standards • Disinfection

O&M	• Historically operator intensive
	  ○ Operator demands can be minimized with latest design configurations
	• Constant monitoring required to assure efficient removal, microbe health, etc.
	• Monitoring of nitrite and ammonia will also be necessary due to the potential for incomplete denitrification
	• Management of chemicals
	• Waste sludge storage and disposal
	• New plants can be highly automated
	• Historically viewed as operationally complex
	  ○ More unit processes than IX
	  ○ New design configurations can minimize complexity (e.g., fixed bed)

System Components	Important process considerations include (Dördelmann 2009): • Dosage requirements of substrate and nutrients • Reactor configuration and governing equation of the biological process • Aeration to remove nitrogen gas and provide oxygen • Filtration to remove particulate matter • Activated carbon may be used to remove substrate residual and avoid DBP formation (for heterotrophic systems) • Disinfection
Waste Management and Disposal	• Sludge disposal – Biological solids and residual organic matter • No waste brine or concentrate as in separation processes
Limitations	• Requires anoxic conditions • Chemical management • Few examples for nitrate removal in the U.S. • Post-treatment requirements • Operator training • Intermittent use of wells may be challenging due to the need for acclimation of microorganisms

Adminsitrative Record
Page 35369

Water Quality

Anoxic conditions are required for denitrification to occur. In the presence of oxygen (> 0.1 mg/L) bacteria preferentially reduce oxygen rather than nitrate, diminishing the efficiency of the process. For all configurations, the optimal growth of microbes must be considered. Control and monitoring of water quality characteristics including temperature, pH, salinity, and oxidation reduction potential (ORP) can be fundamental to the stability and efficiency of the biological denitrification system (WA DOH 2005). For biological denitrification, near neutral pH is preferred (7 – 8) and temperatures below 5°C/41°F can inhibit denitrification (WA DOH 2005).

Pretreatment will include addition of substrate and nutrients in the appropriate dose while post-treatment requirements can include coagulant addition, filtration, gas exchange, and disinfection for the removal of biomass, particulates, and substrate residuals (Panglisch et al. 2005; Dördelmann et al. 2006).

System Components and Site Considerations

Important process considerations in the design and operation of BD systems include (Dördelmann 2009):

- Dosage requirements of substrate and nutrients
- Reactor configuration and governing equation of the biological process
- Aeration to remove nitrogen gas and provide oxygen
- Filtration to remove particulate matter
- Activated carbon to remove substrate residual and avoid DBP formation (for heterotrophic)
- Disinfection

Conditions should be optimized to ensure complete denitrification. In addition to nitrate reduction to meet the nitrate MCL (45 mg/L as NO_3^-, 10 mg/L as N), effluent *nitrite* levels must not exceed the *nitrite* MCL of 1 mg/L as N. Incomplete denitrification, which can be associated with higher dissolved oxygen (DO) levels, can result in the production of greenhouse gases (GHG) such as NO and N_2O. Dissolved oxygen levels can be decreased using reducing agents or through the provision of sufficient electron donor to enable depletion of DO (Meyer et al. 2010).

System configurations of biological denitrification include: fluidized bed reactor, fixed bed reactor, membrane biofilm reactor, and bio-electrochemical reactors. In situ options (including bank filtration) have also been explored in the research.

Fixed Bed

Fixed bed biological reactors can be in up-flow or down-flow systems in a pressurized or open flow configuration (Brown 2008). Typical options for growth support media include sand, plastic, and

Adminsitrative Record
Page 35370

granular activated carbon (Brown 2008). Accumulation of biomass in the media leads to head loss requiring periodic backwashing. Post-treatment requirements can include filtration, gas exchange, and disinfection for the removal of biomass, particulates, and substrate residuals (Panglisch et al. 2005; Dördelmann et al. 2006). The fixed bed configuration "is often coupled with pre-ozonation to improve the removal of organic material, which reduces regrowth potential and DBP formation in distribution systems" (Brown 2008, p. 137). (See Soares (2002), Panglisch et al. (2005), Aslan (2005), Dördelmann et al. (2006), Carollo Engineers (2008), Upadhyaya (2010), Meyer et al. (2010), and City of Thornton (2010), in Table A.4 for research examples of the fixed bed configuration.)

A detailed case study of the Fixed Bed configuration of biological denitrification in Riverside, CA, is included in Section 3.4.5 Biological Denitrification - Case Studies.

Fluidized Bed

The fluidized bed reactor operates in an up-flow mode, resulting in granular growth support media expansion. Fluidizing the granular media offers several advantages over the fixed bed configuration. Flow resistance is minimized and the system does not need to be taken off line for backwashing because accumulated biomass is removed by the fast flowing feed water and/or "in-line mechanical shearing devices" (Brown 2008, p. 139). "The biofilm is detached from the support material only upon strong mechanical effect, thus the excess biomass can be intermittently removed from the reactor, independently of the purified water" (Holló & Czakó 1987, p. 418).

Maintenance of the sufficient up-flow velocity can be achieved through recycled flow and reactor volumes are designed for a typical bed expansion of 25 to 30% (Brown 2008). (See Kurt et al. (1987), Holló & Czakó (1987), and Webster & Togna (2009), in Table A.4 for research examples of the fluidized bed configuration.)

A detailed case study of the Fluidized Bed configuration of biological denitrification in Rialto, CA is included in Section 3.4.5 Biological Denitrification - Case Studies.

Membrane Biological Reactor (MBR)/Membrane Biofilm Reactor (MBfR)

With the addition of membrane technology to conventional biological denitrification, common concerns of biological treatment can be minimized through physical separation of biomass and substrate from the treated water. In a comprehensive review of membrane bioreactors, McAdam & Judd (2006) present the pros and cons of a variety of MBR configurations (Table 12). MBRs can be designed for autotrophic or heterotrophic denitrification. Pressurized systems have been explored using submerged ultrafiltration membranes or external (sidestream) MBRs, while pressure neutral diffusion systems have been implemented with ion exchange membranes and microporous membranes (McAdam & Judd 2006; Brown 2008). Membrane types include hollow fiber, ion exchange, microporous, and flat sheet. (See Mansell & Schroeder (2002), Ergas & Rheinheimer (2004), Nerenberg & Rittman (2004), Chung et al. (2007), Meyer et al. (2010), and City of Thornton (2010) in Table A.4 for research examples of MBR configurations.)

Adminsitrative Record
Page 35371

Applied Process Technology, Inc. (APT) has developed an autotrophic membrane biofilm reactor (MBfR) (Gormly & Borg N.D.) using hydrogen gas as the electron donor rather than a carbon substrate. Early pilot and bench-scale studies of the APT MBfR raised several key challenges requiring optimization of the system, including the need to avoid build-up of excess biomass (Rittmann 2007). As part of a long term pilot-scale study[19] in Glendale, AZ, APT's MBfR was examined to address high nitrate levels in groundwater. This autotrophic biological denitrification system successfully reduced nitrate levels to below the MCL. Three types of hollow fiber membranes were examined for substrate delivery. Operational concerns highlighted by this study include (Meyer et al. 2010):

- Problems with leaking fibers,
- hydrogen sulfide formation due to excessive hydrogen gas pressure,
- ammonium generation from biomass decay due to operational interruption and insufficient electron donor, and
- nitrite levels above the 1 mg/L nitrite as N limit (incomplete denitrification).

To address these concerns, the authors suggest:

- The use of the latest optimized membranes,
- consistent and adequate nutrient and electron donor supply,
- oxidation of nitrite in post-treatment if necessary,
- stable loading and continuous operation to avoid system upset, and
- parallel reactors to allow for maintenance and repair.

The MBfR pilot was one of three biological configurations examined in Glendale. Two up-flow heterotrophic fixed bed bioreactors were also examined, each with a different media type. Post-treatment included filtration using biologically active carbon and ozonation. The two most promising biological treatment options, the MBfR and the up-flow fixed bed bioreactor with plastic media, were compared with each other and also with an IX system. Overall, using a multi-criteria analysis with consideration of sustainability, the MBfR scored the most favorably regarding benefits, but the least favorably regarding life cycle costs. The life cycle costs of both the IX and the fixed bed bioreactor options were lower than that of the MBfR. The Glendale study highlights several key areas of future research including "a shut-down test where biological treatment processes sit dry for a period of time and then re-start at optimal hydraulic loading rates [and] a re-acclimation test where systems are re-initiated after losing all viable biomass" (Meyer et al. 2010, p. 164).

As a promising technology, with further research and design optimization to reduce costs, the MBfR may become a more feasible treatment option. As of 2008, a demonstration project of the MBfR for nitrate and dibromochloropropane (DBCP) removal was under consideration for the City of Fresno, CA, to test performance of the MBfR as an alternative treatment option (City of Fresno 2008).

[19] Project partners and participants: City of Glendale, Arizona, Water Research Foundation, Arizona State University, CH2M HILL, Applied Process Technology, Inc., Intuitech, Inc., KIWA Water Research, and Layne-Christensen.

Adminsitrative Record
Page 35372

Table 12. Membrane biological reactor configurations.

Diffusive Extraction Microporous Membranes	Relying on diffusion for nitrate transfer through the membrane, extractive MBRs (i.e., Fixed MBRs) do not directly filter the treatment stream, but they do provide separation of the denitrification chamber (substrate, biomass, and associated residuals) and the treatment stream. Key issues are the transfer of biomass or substrate across membrane, the potential for fouling/scaling, and the need for a high transfer rate of nitrate to the denitrification compartment.
Diffusive Extraction Ion Exchange Membranes	With the use of an ion exchange membrane rather than a microporous membrane, selectivity for nitrate and decreased mass transfer from the denitrification compartment are facilitated; however, capital and maintenance costs of ion exchange membranes can be significant and the need to manage membrane fouling persists (McAdam & Judd 2006).
Gaseous Substrate Delivery Hollow Fiber Membranes	In autotrophic systems, membranes can be used for gaseous substrate delivery, (i.e., hydrogen gas) (McAdam & Judd 2006). Previous problems with limited hydrogen gas transfer have been addressed with the use of hollow fiber membranes for delivery. Substrate passes through the membrane to the biofilm on the outer membrane surface. The membrane does not separate the biomass from the treatment stream and does not directly filter the treatment stream (McAdam & Judd 2006); the presence of "sloughed biomass" and biological residuals in the treatment stream requires further treatment downstream.
Pressure Driven Direct Filtration MBRs	Pressurized MBRs provide the advantage of direct filtration. Pressure is used to draw the denitrified water through the submerged membrane, leaving behind biomass and other undesirable constituents. However, the use of this configuration for denitrification is complicated by the fact that aeration is typically used for mixing and to minimize fouling of the external membrane surface (Brown 2008).

In Situ Denitrification

Bank filtration refers to the withdrawal of surface water through an embankment. The porous media (soil) of the bank serves as a biological reactor providing treatment through "filtration, dilution, sorption, and biodegradation processes" (Brown 2008, p. 140). Bank filtration was employed in water treatment as early as 1870 along the Rhine River in Germany (Brown 2008). As a recent example, a full-scale study in Aurora, CO, demonstrated effective nitrate removal with bank filtration of surface water from the South Platte River (Waskom, Carlson & Brauer N.D.). Bank filtration has also been implemented to address nitrate impacted waters in Saxony, Germany, and Des Moines, IA (Jones et al. 2007; Grischek et al. 2010).

Biological denitrification for in situ removal of nitrate from groundwater was explored by Hunter (2001), Haugen et al. (2002), Schnobrich et al. (2007), and many others. Through in situ denitrification, the subsurface acts as the porous media through which water is filtered. Residual organics and biomass from denitrifiers can thus be removed naturally. (See Hunter (2001), Haugen et al. (2002), and Schnobrich et al. (2007) in Table A.4 for research examples of in situ application.) See Technical Report 5, Section 2 for a discussion of in situ denitrification, permeable reactive barriers, phytoremediation, and other remediation methods (King et al. 2012).

Adminsitrative Record
Page 35373

Post-Treatment Requirements - Filtration/Taste & Odor/Disinfection

With the use of microorganisms and the addition of a carbon substrate, post-treatment is essential to meet turbidity standards, to remove biomass and residual organic matter, and to address taste and odor concerns. Post-treatment must include disinfection to address biological contamination and can also include dual media filtration and/or activated carbon filtration and aeration (individual state regulations will need to address local requirements).

Residuals Management and Disposal

In contrast to the concentrated waste stream from removal processes, biological denitrification has limited waste demands due to the conversion of nitrate to nitrogen gas. Waste sludge, consisting of biological solids and residual organic matter, requires appropriate disposal; however, with nearly 100% water recovery, the low waste volumes are not a significant burden (Kapoor & Viraraghavan 1997).

Maintenance, Monitoring, and Operational Complexity

Because biological denitrification is microbially mediated, to maximize performance, systems should be run continuously, with a consistent supply of substrate and nutrients at the appropriate dosage (Dördelmann 2009). An initial startup period may be necessary for development and acclimation of the microorganisms (Holló & Czakó 1987; Aslan 2005). This may be problematic for intermittent use of wells and wasting may be required for acclimation to occur. Backwashing, consistent maintenance, and regular monitoring of product water quality are also essential. Constituents that should be monitored frequently are nitrate, nitrite, pH, oxygen, turbidity, conductivity, dissolved organic carbon, and bacterial count (Dördelmann 2009). Operation and maintenance demands of biological denitrification systems typically exceed those of alternative treatment technologies. However, these systems are more sustainable because nitrate is reduced to innocuous nitrogen gas rather than concentrated in a waste stream that requires costly disposal (Dördelmann 2009).

3.4.2 Biological Denitrification - Cost Considerations

For efficient operation of a biological denitrification system, maintaining optimal conditions for the bacteria is essential, as is balancing the appropriate substrate and nutrient dose and managing pre and post-treatment while meeting necessary potable water guidelines. Factors affecting system cost include facility size (flow rate), source water quality (including nitrate concentration), environmental factors (temperature and pH), target effluent nitrate concentration, and possible wasting due to intermittent use of wells and associated acclimation of microorganisms.

Capital costs for biological denitrification include land, housing, piping, storage tanks, O&M equipment, preliminary testing (through extensive pilot studies), permits, and significant operator training. O&M costs include post-treatment, sludge disposal, chemical use (pH adjustment, substrate and nutrient dosing), repair, extensive monitoring and maintenance, power, and labor. Costs can be higher in certain states, depending on post-treatment requirements.

Adminsitrative Record
Page 35374

Very little published cost information from existing biological denitrification systems for drinking water is available in the literature, due to the limited number of full-scale systems (Table 13). Costs have been adjusted to 2010 dollars, unless indicated otherwise. The listed cost information is provided as an approximate range of costs for specific facilities. Costs for implementing biological denitrification may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the questionnaire includes detailed costs associated with the individual case studies included in this analysis. A detailed discussion of treatment costs is included in Section 6 Treatment Cost Analysis.

Table 13. Cost information* for biological denitrification of potable water.

System Flow**	< 0.5 MGD	0.5 – 5 MGD	5+ MGD
Annualized Capital Cost ($/1000 gal)	0.83 [1]	0.61 – 0.80 [2, 3]***	0.51 – 0.62 [4]
O&M Cost ($/1000 gal)	0.30 [1]	0.33 – 0.46 [2, 3] ***	0.74 – 0.94 [4]
Total Annualized Cost ($/1000 gal)	1.13 [1]	1.03 – 1.13 [2, 3] ***	1.25 – 1.56 [4]

*Costs have been adjusted to 2010 dollars with 7% interest over 20 years, unless indicated otherwise.
**When available, costs are based on actual system flow rather than design capacity.
***Listed costs are based on biological treatment for perchlorate and should be considered only as a rough estimate of similar systems for nitrate treatment.
[1] Silverstein (2010), not adjusted to 2010 dollars. [2] Webster & Togna (2009). [3] Carollo Engineers (2008). [4] Meyer et al. (2010).

3.4.3 Biological Denitrification - Selected Research

Table A.4 of the Appendix lists recent research studies relevant to the use of biological denitrification in potable water treatment.

3.4.4 Biological Denitrification - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of biological denitrification in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of the use of biological denitrification for nitrate removal from potable water include high water recovery, no brine or concentrate waste stream (reduction of nitrate rather than removal to a concentrated waste stream), low sludge waste, less expensive operation, limited chemical input, multiple contaminant removal, and increased sustainability (WA DOH 2005; Brown 2008; Upadhyaya 2010).

Disadvantages of nitrate removal using biological denitrification are post-treatment requirements for the removal of biomass and dissolved organics, high capital costs, potential sensitivity to environmental conditions (although recent pilot tests indicate robust newer designs), system footprint larger than typical IX systems, high system complexity (may be simplified with new configurations), lack of full-scale systems in the U.S., potential for incomplete denitrification and GHG production, pilot study requirements, and slow start-up (Kapoor & Viraraghavan 1997; WA DOH 2005).

Adminsitrative Record
Page 35375

3.4.5 Biological Denitrification - Case Studies

The following case studies provide detailed information on the design and operation of a pilot-scale and planned full-scale biological treatment systems that can be used for nitrate removal.

System Name: West Valley Water District
System Location: Rialto, CA
PWSID: CA 3610004

CASE #13 System Type: Demonstration System, Full-scale Installation is under construction

Treatment Type: Biological Denitrification, Fluidized Bed Bioreactor (FBR)
Questionnaire completed by: Todd Webster from Envirogen Technologies, Inc., Director of Sales & Bioreactor Applications.
Tom Crowley from the West Valley Water District.
Demonstration Dates of Operation: 2007 – 2008, Full-scale Installation Proposed Startup Date: 2012

System Description

Treatment Type	*Fluidized Bed Bioreactor*
System Capacity	*2,000 – 4,000 gpm (for proposed full-scale treatment)*
Raw Water Nitrate	
(abandoned well)	*17 – 19 mg/L as nitrate*
	~4 – 5 mg/L as N
(pilot)	*27.5 – 27.9 mg/L as nitrate*
	6.2 – 6.3 mg/L as N
Raw Water Perchlorate	
(pilot)	*50+ ug/L*

The West Valley Water District (District) utilizes 2 surface water sources and 5 groundwater wells. Of these supplies, one groundwater well is impacted by nitrate contamination above the MCL, with an average nitrate concentration of 18 mg/L of nitrate as nitrate (~4 mg/L of nitrate as N). The well source is the Chino Basin which has an estimated capacity of 300 – 400K ac. ft. For the immediate future this well has been abandoned due to nitrate contamination, while feasible treatment options are being explored. The primary water quality concern in the District is perchlorate contamination. Biological denitrification has been explored principally to address perchlorate levels typically in the range of 50 – 53 µg/L. However, simultaneous nitrate reduction in the biological denitrification process makes this system an appropriate example as a nitrate treatment alternative.

Unlike the removal processes of IX, RO, and EDR, biological denitrification allows for the destruction of nitrate through reduction to innocuous nitrogen gas. The fluidized bed configuration maximizes media surface area for the growth of denitrifying bacteria. After a successful 1-year demonstration study on well Rialto #2, a full-scale Fluidized Bed Bioreactor (FBR) is expected to be online in early 2012.

Adminsitrative Record
Page 35376

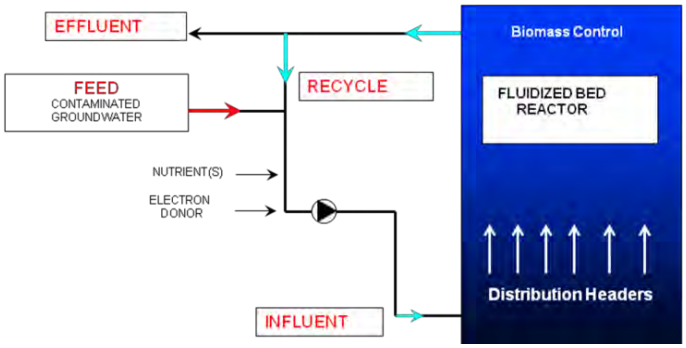


Figure 13. FBR configuration. (Source: reprinted with permission, Webster & Togna 2009.)

"The system is the fluidized bed bioreactor (FBR). The contaminated feed water is pumped from the wellhead and fed directly into a recycle line of the reactor. The feed and recycle water enters the vessel through an inlet header at the bottom of the reactor and is distributed through lateral piping and nozzles (Figure 13). The fluid passes upward through the media, causing the media to hydraulically expand approximately 28% of the settled bed height. Through a self-inoculating process from the contaminated feed water, microorganisms attach on to the fluidized media. Adequate quantities of electron donor (i.e., acetic acid) and nutrients are added to the reactor. Utilizing this electron donor and the nutrients, the attached microorganisms perform an oxidation/reduction reaction in consuming all of the dissolved oxygen, nitrate, and perchlorate. As the microorganisms grow, the amount of attached microbes per media particle also increases. Since the microbes primarily consist of water, the volume of the microbe/media particle increases, but the specific density decreases. This allows the media bed to expand and fluidize further such that longer hydraulic retention times can be achieved for contaminant removal. The treated fluid flows into a submerged recycle collection header pipe and the effluent collection header pipe at the top of the reactor. A portion of the fluid exits the FBR system to a post-aerator while the balance is recycled back to the suction of the influent pump. An in-bed biomass separation device controls bed height growth by physically separating biomass from the media particles. Typically, a bed expansion of 40 – 60% of the settled bed height is targeted. Any excess biomass that is separated from the media exits the system through the effluent collection system" (Webster & Togna 2009, p. 6).

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L as N)
  - Demonstration well: 27.5 – 27.9 (6.2 – 6.3)
  - Abandoned nitrate impacted well
    - Average: 18 (~4)
- Co-contaminants: Perchlorate
  - Demonstration well: 50 – 53 µg/L, spiked to 1000 µg/L (with appropriate substrate and nutrient adjustments)
  - Abandoned nitrate impacted well: a few ppb

Treatment Technology Selection

The fluidized bed bioreactor was selected for perchlorate removal; however, nitrate reduction is also accomplished. Due to the levels of nitrate and perchlorate, biological denitrification was deemed to be more cost-effective than alternative physico-chemical technologies. Generally, with nitrate levels well above the MCL (e.g., 100 mg/L as nitrate), removal processes can become more costly. Biological denitrification can be the more feasible option for source water containing co-contaminants and/or high levels of nitrate.

For the Rialto #6 and West Valley Water District #11 wells, the fluidized bed reactor and IX were considered. Two additional pilot studies were performed to assess nitrate and perchlorate treatment using a packed bed bioreactor and zero valent iron.

Adminsitrative Record
Page 35377

Demonstration System Parameters

- Design Capacity
  - 50 gpm capacity
- Pretreatment
  - Substrate and nutrient addition
    - Acetic Acid:
      16.2 mg/L as C (+20 – 25%)
      15 mL/min, 50% acetic acid
  - Nutrient addition
    - Phosphoric Acid:
      0.3 mg/L as P
      10.5 mL/min
- Post-treatment
  - Aeration tank
    - Increase dissolved oxygen
  - Clarifier/multimedia filter
  - GAC filtration
  - UV disinfection versus chlorination
- Water Recovery: > 99%
- Bed Expansion
  - 28% of settled bed height
  - Max: 40 – 60%
- Media: Sand or GAC
- Cleaning requirements
  - Biomass separator
  - In-bed cleaning eductor
- Manufacturer:
  Envirogen Technologies, Inc.
- Monitoring: Throughout the demonstration using online nitrate and perchlorate analyzers

Demonstration System Cost Estimation - Scaling up from 50 gpm to 1000 gpm

Capital Costs (2008 dollars) (1000 gpm)	
Total Equipment Costs:	$1,966,000
Total Contractor Costs:	$570,140
Total Home Office Costs:	$664,000
Total Installed Capital Costs (1000 gpm):	$3,200,140
Amortized Capital Cost ($/AF):	$128 (30 years, 4.9% bonding rate)
O & M Costs (2008 dollars) (1000 gpm)	
Electricity ($/yr):	$87,600
Chemicals ($/yr):	$133,187
Maintenance ($/yr):	$20,000
Total Operating Costs ($/yr):	$240,787
Operating Costs ($/AF):	$149
Total Cost (2008 dollars) (1000 gpm)	
Total Annualized Cost ($/AF):	$277

Adminsitrative Record
Page 35378

Proposed Full-scale System Parameters - See Figure 14

- Design Capacity
  - 2000 gpm capacity
  - Expandable to 4000 gpm
- Pretreatment
  - Substrate and nutrient addition
    - Acetic Acid: 10 – 15 mg/L as C
- Post-treatment
  - Aeration tank
    - Increase dissolved oxygen
  - Clarifier/multimedia filter
  - Chlorine disinfection
- Treatment system footprint
  - Treatment system: 2 FBRs
    - 14' diameter x 24' height
  - Residuals handling system
    - DAF for solids removal
  - Total system footprint
    - For 4000 gpm
      - 180' x 130'
    - For 2000 gpm
      - 25% less than above
- Water Recovery: > 99% expected
- Bed Expansion
  - 28% of settled bed height
  - Max: 40 – 60%
- Media: Sand or GAC
- Cleaning requirements
  - Biomass separator
  - In-bed cleaning eductor
- Manufacturer:
  Envirogen Technologies, Inc.
- Monitoring: N/A
- Waste Volume:
  0.3 gpm waste per 2000 gpm treated

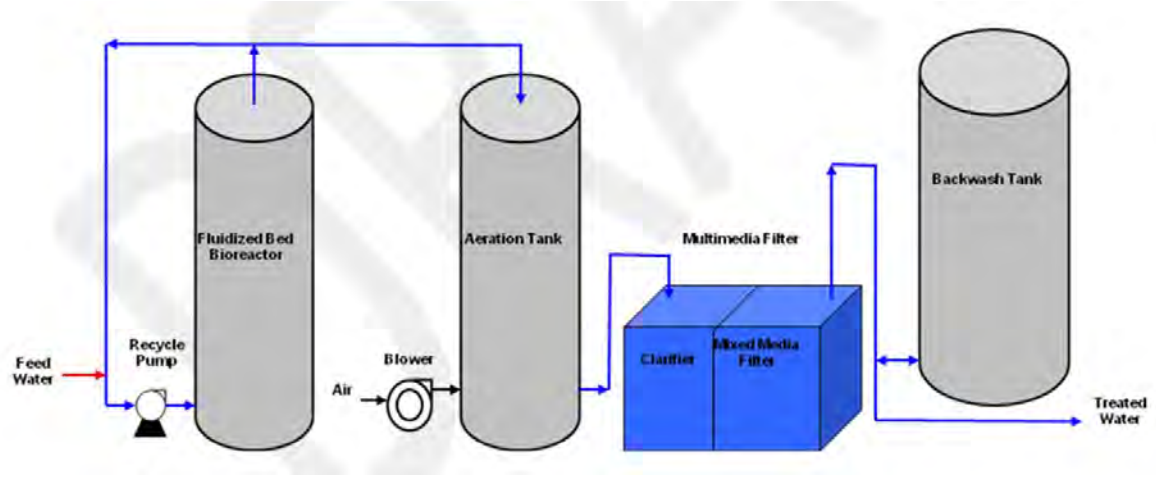


Figure 14. FBR treatment system schematic. (Source: Webster et al. 2009, reprinted from *Journal AWWA* (May 2009) by permission. Copyright © 2009 by the American Water Works Association.)

Adminsitrative Record
Page 35379

Proposed Full-scale System Cost Estimation

Capital Costs (Total with explanation or component costs)	
Treatment and Monitoring Equipment:	$1.8 million for the 2 FBR vessels, the 2 post-aeration vessels, nitrate and perchlorate monitoring, 1 DAF, chemical feed systems, pumps, valves, additional components
Filtration System:	$800,000 for the 2 clarifier and multimedia filters
O & M Costs (Total with explanation or component costs)	
Unavailable because system is not yet operating. Please see above for estimated O & M costs from the full-scale demonstration study.	

Residuals Management

Unlike removal technologies, the use of the FBR results in destruction of nitrate and perchlorate, rather than the transfer of these constituents to a concentrated waste stream requiring disposal.

Technology Benefits and Drawbacks

Benefits

- Capable of perchlorate removal and handling high nitrate levels
- Reduction of nitrate rather than just removal
- High water recovery and limited waste
- Lower operating costs than removal technologies

Drawbacks

- Limited application in the U.S.
- Large system footprint
- More complicated permitting
- Extensive pilot study necessary
- Start-up time (up to a month)
- Increased operator attention
- Sensitivity to system interruption

Additional Information

The full-scale system is being planned specifically to address perchlorate contamination, but will reduce nitrate as well. Construction is currently underway and the system should be fully operational with discharge to groundwater by Mid-2012. With approval from the California Department of Public Health (CDPH) it is expected that distribution of treated drinking water will begin in Early-2013. Regarding operator training, for this system, the operators already held the required certification due to operation of a surface water treatment plant; however, for other utilities lacking this pre-existing experience, additional training would likely be needed. Regarding permitting, the 97-005 process from the CDPH Policy Memo 97-005, ("Guidance for the Direct Domestic Use of Extremely Impaired Sources") was followed as guidance for the FBR installation. The 97-005 process is strenuous and involved, requiring analysis of failure and worst-case scenarios and a 6 month demonstration of proper operation of the treatment plant. The system sizes for which an FBR would be appropriate vary with the concentration of contaminants. Feasible application of an FBR can be more dependent on load than on flow capacity. The FBR has been implemented or tested for systems as small as 7 to 12 gpm and across perchlorate concentrations from 12 to 13000 ppb. Typical flow rates are 250 to 5000 gpm, but with high loads FBRs can become more feasible for lower flow rates.

Adminsitrative Record
Page 35380

Source*

Webster, T.S., Guarini. W.J. and Wong, H.S. (2009) Fluidized bed bioreactor treatment of perchlorate-laden groundwater to potable standards. *Journal of the American Water Works Association*, **101** (5).
Webster, T.S. and Togna, P. (2009) Final Report: Demonstration of a Full-Scale Fluidized Bed Bioreactor for the Treatment of Perchlorate at Low Concentrations in Groundwater. ESTCP Project ER-0543.
Webster, T.S. and Crowley, T. (2010) Completed questionnaire and personal communication.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35381

System Name: Western Municipal Water District (WMWD) - Arlington Desalter
System Location: Riverside, CA
PWSID: CA3310049

CASE #14 System Type: Pilot Study, Full-scale System is currently in the Design Phase

Treatment Type: Fixed Bed Bioreactor (FXB)
Questionnaire completed by: Joseph Bernosky, WMWD, Director of Engineering
Jess Brown, Carollo Engineers, Carollo Research Group Manager
Startup Date: Proposed 2013

Treatment Type	*Fixed Bed Bioreactor*
System Capacity	*2.4 MGD (1670 gpm)*
Raw Water Nitrate	*44 – 89 mg/L as nitrate*
	10 – 20 mg/L as N

System Description

The Western Municipal Water District (Western) operates the Arlington Desalter as a drinking water supply facility. The original RO water treatment facility was constructed in the late-1980s for salt management in the Arlington groundwater basin. In 2002 the facility was upgraded and subsequently approved as a potable water supply by the California Department of Public Health (CDPH). Three wells supply raw water to the RO process and two additional wells supply bypass water for blending with the RO permeate to produce the finished water (product water).

The current capacity of the Arlington Desalter is 5 million gallons per day (MGD) of RO permeate and approximately 1.3 MGD of untreated bypass for a total product water capacity of 6.3 MGD. Western's water supply reliability program, developed through the Integrated Regional Water Management Plan (IRWMP), includes plans for expansion of the Arlington Desalter to 10 MGD capacity.

A pilot study was conducted at the Arlington Desalter from April through November 2007 that demonstrated the feasibility of nitrate removal using fixed bed biological treatment (biodenitrification) of the RO bypass stream. After completion of the initial pilot study, CDPH proposed a turbidity standard for the biodenitrification process. Supplemental pilot testing conducted between May and September 2008 demonstrated compliance with the new turbidity standard by adding a nitrogen degasification step followed by coagulation and polishing filters.

The FXB biological process utilizes a stationary bed of granular activated carbon (GAC) on which biofilms containing nitrate-reducing bacteria develop. Raw water is drawn from a well amended with an electron donor such as acetic acid. The water is then pumped through the GAC bed. Bacteria in the bed convert the nitrate to nitrogen gas and water. A one-time acclimation period is required to develop the nitrate-reducing biological activity, which is done by contacting virgin GAC with raw water and acetic acid for two to three weeks. The denitrifying bacteria used in the system are indigenous to the natural groundwater, meaning the system is naturally seeded with bacteria present in the groundwater.

During the pilot, a clone library analysis was performed on the bacteria within the biofilter to classify the various types of denitrifying bacteria present. The analysis revealed a diverse community of bacteria. At least 10 different denitrifying genera were identified, including Acidovorax, which comprised approximately 37 percent of the total bacteria in the FXB biofilters. The bacteria identified were gram negative, suggesting that they would be particularly sensitive to chemical disinfection, as gram-negative bacteria tend to have thin cell walls.

Adminsitrative Record
Page 35382

Source Water Quality

- Nitrate – mg/L as nitrate (mg/L N)
  - Average – 75 (17)
  - Minimum – 44 (10)
  - Maximum – 89 (20)
- Co-contaminants
  - Perchlorate: 6 µg/L
  - DBCP: 0.025 µg/L

Treatment Technology Selection

The FXB process has been used successfully for removal of nitrate from drinking water supplies in Europe for decades, but has not yet been used full-scale in the United States. Reasons for using the FXB process at the Arlington Desalter include the following:

- Nitrate is not concentrated in a waste stream, as in RO or IX treatment, but is converted to nitrogen gas, which is released to the atmosphere—a harmless emission because the atmosphere is 78 percent nitrogen.
- Nitrate removal efficiencies are high. Typically, greater than 90 percent removal was achieved during the Arlington pilot studies and removal to non-detect levels was possible.
- The biodenitrification process results in simultaneous destruction of some anthropogenic contaminants. For example, perchlorate, found in the Arlington Desalter supply, was reduced to non-detect levels in the pilot study.

Historically, IX treatment has been the process of choice for nitrate removal in this country; however, it results in the replacement of nitrate with chloride in the drinking water supply. It is estimated that the equivalent IX system installed at the Arlington Desalter would add approximately 3 million pounds of salt to the basin annually. The removal of salt was the purpose for construction of the Arlington Desalter in the first place.

Treatment System Parameters

- Design Capacity
  - 1,670 gpm capacity
- Pretreatment
  - Substrate and nutrient addition
- Post-treatment
  - Degasification of N_2
  - Filtration for compliance with SWTR
- Water Recovery: expected 95%
- Media: GAC
- Treatment system footprint: TBD

Residuals Management

The residuals from this process, primarily biological growth, are accumulated in the filtration process. Once a predetermined pressure loss is experienced in filtration, the filters are backwashed to remove the accumulated solids. The solid laden backwash is then sent to the Santa Ana Regional Interceptor (SARI) line and ultimately disposed of off shore.

Technology Benefits and Drawbacks

Benefits

- Less expensive than other technologies
- No disposal of waste brine
- High water recovery rate: > 95%
- Can remove co-contaminants

Drawbacks

- No full-scale applications in operation
- More complicated permitting

Adminsitrative Record
Page 35383

Treatment Technology Costs

Detailed cost estimates are being developed. Detailed design of this system may occur in 2012, and full-scale construction of this system may be in 2013.

As an approximation, the below costs are based on a fixed-bed demonstration system for the removal of perchlorate with a similar empty bed contact time (EBCT) (Carollo Engineers 2008).

Capital Costs (2008 dollars)		
Flow Rate:	**1000 gpm**	**2000 gpm**
Total:	$4,193,000	$7,395,000
Direct Installed Costs:	$2,373,000	$4,200,000
O & M Costs (2008 dollars)		
Total Estimated Annual O&M Costs:	$175,000	$348,000
Chemicals ($/yr):	$161,000	$323,000
Other (GAC and Filter Sand) ($/yr):	$9,000	$17,000
Power ($/yr):	$5,000	$8,000
Total Costs (2008 dollars, 2.8% discount rate, 30-year lifecycle)		
Amortized Project Costs:	$209,000	$368,000
Estimated Annual Budget:	$384,000	$716,000
Total Treatment Costs ($/1000 gal):	$0.73	$0.68
Total Treatment Costs ($/AF):	$238	$222

Source*

Bernosky, J. (2010) Personal communication.
Brown, J. (2010) Personal communication.
Carollo Engineers. Final Report: Fixed-Bed Biological Nitrate Removal Pilot Testing at the Arlington Desalter Facility.
Carollo Engineers. (2008) Final Report: Direct Fixed-Bed Biological Perchlorate Destruction Demonstration. ESTCP Project ER-0544.

**Unpublished sources used in the development of the case studies are not reflected in the References section of this report.*

Adminsitrative Record
Page 35384

3.5 Chemical Denitrification (CD)

Chemical denitrification can be accomplished with reduction of nitrate by metals. Various metals have been investigated for use in nitrate reduction including aluminum and iron (both Fe^{o} and Fe^{2+}), while copper, palladium, and rhodium can be used as catalysts in nitrate reduction (Shrimali & Singh 2001). The advantage of chemical denitrification over the removal technologies is that nitrate is converted to other nitrogen species rather than simply displaced to a concentrated waste stream that requires disposal. Problems with chemical denitrification of potable water are the reduction of nitrate beyond nitrogen gas to ammonia, partial denitrification, and insufficient nitrate removal (nitrite can be converted to nitrate with the use of chlorine in disinfection). No full-scale chemical denitrification systems have been installed in the United States for the removal of nitrate in potable water treatment. A significant body of research has explored the use of zero valent iron (ZVI) in denitrification. Several patented granular media options are also emerging, including SMI-III® (Sulfur Modified Iron), MicroNose™ Technology, and Cleanit®-LC.

Based on lab and pilot-scale studies, there is much variation in the configuration of chemical denitrification systems for nitrate removal from potable water. The generic mechanism of denitrification involves the transfer of electrons from an electron donating metal to nitrate. As in biological denitrification, nitrate is reduced in accordance with Eqn. 9. However, in contrast with biological denitrification, using chemical denitrification, the nitrogen in nitrate is often reduced to the least oxidized form, ammonium (Eqn. 9a) (Huang et al. 1998; Hao et al. 2005).

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \qquad \text{(Eqn. 9)}$$

$$NO_3^- \rightarrow NO_2^- \rightarrow NH_4^+ \qquad \text{(Eqn. 9a)}$$

Nitrate is exposed to an electron donating metal by passing the treatment stream through granular media. Particle size, surface area, and surface chemistry are important media characteristics related to the efficiency of nitrate removal.

3.5.1 Zero Valent Iron (ZVI)

Due to the extensive research focused on the use of zero valent iron (ZVI), ZVI will serve as a preliminary example. There is some variation in the use of ZVI. Forms of application include powdered iron, stabilized iron as nanoparticles, iron filings, and permeable reactive barriers (PRBs). Relevant reactions are listed in Eqns. 13 to 18 (Huang et al. 1998; Hao et al. 2005; Xiong et al. 2009). Nitrate can be reduced to nitrite (Eqn. 13), ammonia (Eqn. 14), or nitrogen gas (Eqn. 15) by ZVI. Following nitrate reduction to nitrite, nitrite can then be reduced to ammonia (Eqn. 16). Nitrate can also be reduced by the hydrogen gas that is produced from corrosion reactions (Eqn. 17) to ammonia (Eqn. 18).

$$Fe^{o} + NO_3^- + 2H^+ \rightarrow Fe^{2+} + NO_2^- + H_2O \qquad \text{(Eqn. 13)}$$

Adminsitrative Record
Page 35385

$$4Fe^{o} + NO_3^- + 10H^+ \rightarrow NH_4^+ + 4Fe^{2+} + 3H_2O \quad \text{(Eqn. 14)}$$

$$5Fe^{o} + 2NO_3^- + 6H_2O \rightarrow N_{2(g)} + 5Fe^{2+} + 12OH^- \quad \text{(Eqn. 15)}$$

$$3Fe^{o} + NO_2^- + 8H^+ \rightarrow 3Fe^{2+} + NH_4^+ + 2H_2O \quad \text{(Eqn. 16)}$$

$$Fe^{o} + 2H^+ \rightarrow H_{2(g)} + Fe^{2+} \quad \text{(Eqn. 17)}$$

$$NO_3^- + 4H_2 + 2H^+ \rightarrow NH_4^+ + 3H_2O \quad \text{(Eqn. 18)}$$

The reduction of nitrate by iron is characterized by an increase in pH and consumption of hydrogen ions. pH is a significant controlling factor for this treatment method (Hao et al. 2005). The kinetics of nitrate reduction by ZVI have been thoroughly covered in the literature to determine the reaction rate under various conditions. For example, Alowitz & Scherer (2002) examined the nitrate reduction rates of three types of iron. Findings indicate that reduction rate increases with decreasing pH. Huang et al. (1998) investigated the use of powdered ZVI for the reduction of nitrate to ammonia. Highly pH dependent, nitrate reduction was kinetically favorable only at a pH below 4. The minimum ratio of iron to nitrate was 120 m^2/mol NO_3^- for complete reduction within 1 hour. Nitrate reduction by ZVI can be optimized through pretreatment of iron particles. High temperature exposure to hydrogen gas and deposition of copper were explored separately as options for pretreatment of the iron surface (Liou et al. 2005). Both methods resulted in improvement of nitrate reduction in almost neutral solutions. The mechanism of improvement is due to the surface chemistry of iron. With a buildup of a surface oxide layer, the availability of sites for nitrate reduction decreases. Hydrogen gas pretreatment reduces the oxide layer, while deposited copper serves as a catalyst for the transfer of electrons. In their investigation of stabilized ZVI nanoparticles, Xiong et al. (2009) found that the end product of denitrification (nitrogen gas versus ammonium) could be controlled by the iron to nitrate ratio and the use of catalysts.

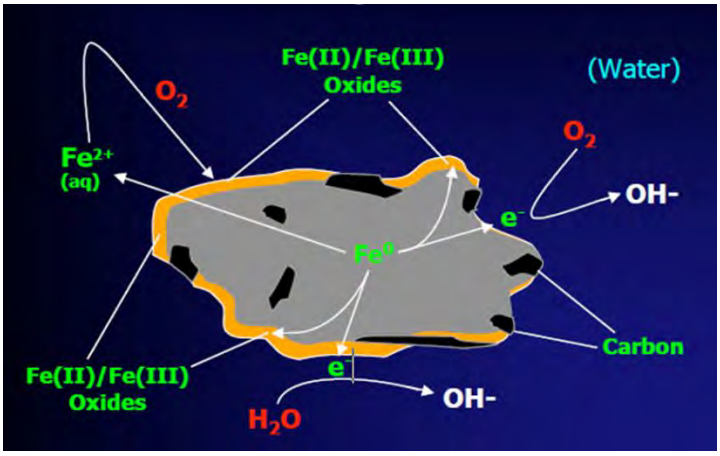


Figure 15. Surface chemistry of ZVI particles. (Source: reprinted with permission, Chiu 2009.)

Examination of the surface chemistry of ZVI particles is of the utmost importance to model and understand its use in the reduction of nitrate. Illustrated in Figure 15, relevant factors include corrosion of ZVI, complexation with water, surface complexation, reduction, precipitation, and adsorption. In the corrosion of ZVI, the formation of "green rusts" and "suspended green particles" is associated with

Adminsitrative Record
Page 35386

stabilization of pH and steady decrease in nitrate (Choe et al. 2004). For nitrate reduction to occur, contact with the reducing agent is required. Reduction of nitrate by ZVI or by any surface bound species requires access to surface sites. Competition for surface sites can impede nitrate reduction; Moore & Young (2005) examined chloride as a potential competitor. Results indicate a minimal impact on nitrate removal; however, other competing ions could be important regarding both competition for adsorption sites and reduction.

3.5.2 Catalytic Denitrification

An extension of chemical denitrification, catalytic denitrification involves metal reduction of nitrate in the presence of a catalyst. Extensive research has investigated catalytic denitrification which may become more readily applicable to potable water treatment with further advances (Reddy & Lin 2000; Pintar et al. 2001; Gavagnin et al. 2002; Lemaignen et al. 2002; Pirkanniemi & Sillanpaa 2002; Chen et al. 2003; Palomares et al. 2003; Pintar 2003; Constantinou et al. 2007; Sun et al. 2010.)

3.5.3 Chemical Denitrification - Design Considerations

Table 14 summarizes key design considerations in the application of chemical denitrification for nitrate removal from potable water.

Adminsitrative Record
Page 35387

Table 14. Summary of design considerations for chemical denitrification.

Pretreatment	• pH adjustment
Post-Treatment	• Filtration for iron removal • pH adjustment • Chlorine addition for disinfection and oxidation of iron • Gas stripping or breakpoint chlorination (for ammonia)
Chemical Usage	• pH adjustment (acids and bases) • Disinfection and oxidation of iron (chlorine)

O&M	• Constant monitoring required to ensure efficient nitrate reduction
	  ○ Nitrate levels
	  ○ Oxidation reduction potential (ORP)
	• Monitoring of nitrite and ammonia will also be necessary due to the potential for incomplete denitrification
	• Management of chemicals
	  ○ pH adjustment
	  ○ Disinfection
	• Waste media and backwash water storage and disposal

System Components/Design Parameters	• Ratio of electron donor to nitrate for desired removal • Hydraulic Loading Rate (HLR), Empty Bed Contact Time (EBCT) • Reactor configuration (up-flow, down-flow, in series, in parallel) • Gas stripping or breakpoint chlorination to remove ammonia (if ammonia is the end-product) • Monitoring equipment • Filtration to remove iron • pH adjustment (decreased in pretreatment and increased before distribution) • Disinfection
Waste Management and Disposal	• Spent media disposal • Iron sludge management • Backwash water • No waste brine or concentrate as in removal processes

Limitations	• No examples of full-scale application for nitrate treatment
	  ○ Unknown reliability for full-scale treatment
	  ○ Unknown costs and operational complications
	• Potential for incomplete denitrification

Water Quality

The performance of chemical denitrification systems can be affected by pH, temperature, potential interference by co-contaminants, and the availability of surface sites. The reduction of nitrate by iron is characterized by an increase in pH and consumption of hydrogen ions. Thus, pH is a significant controlling factor for this treatment method (Hao et al. 2005). Alowitz & Scherer (2002) examined the nitrate reduction rates of three types of iron. Findings indicate that reduction rate increases with decreasing pH. If nitrate in the water does not come in contact with the electron donor, then reduction will not be possible. The build-up of precipitates can negatively impact nitrate reduction. The appropriate iron to nitrate ratio will be based on influent and target nitrate concentrations. Product water quality will require monitoring for nitrate, nitrite, and ammonia.

Adminsitrative Record
Page 35388

System Components and Site Considerations

Major system components include chemical storage (for pH adjustment and disinfection), the column containing the media, and post-treatment disinfection. With reduction to ammonia, post-treatment ammonia stripping may also be necessary. Design constraints include optimal temperature, sufficient EBCT length, avoidance of incomplete denitrification, monitoring requirements (nitrate, nitrite, and ammonia), appropriate iron to nitrate ratio, and adequate chlorine dosing for disinfection and iron oxidation (DSWA 2010). Incomplete denitrification, which can be associated with higher dissolved oxygen (DO) levels, can result in the production of greenhouse gases (GHG) such as NO and N_2O. Dissolved oxygen levels can be decreased using reducing agents or through the provision of sufficient electron donor to enable depletion of DO (Meyer et al. 2010, regarding biological denitrification).

Residuals Management and Disposal

In contrast to IX and the membrane technologies, the burdens of brine and concentrate disposal are minimized because nitrate is reduced through chemical denitrification. There is no concentrated brine solution requiring costly disposal. However, disposal of backwash water, spent media, and iron sludge is necessary.

Maintenance, Monitoring, and Operational Complexity

With the possibility of incomplete denitrification, monitoring is required to ensure that product water does not contain high levels of ammonia or nitrite. Exposure of these nitrogen species to chlorine in disinfection or oxygen downstream can lead to nitrification (oxidation back to nitrate) in the distribution system, unless controlled. Additional O&M demands include management of chemicals (e.g., acids, bases, and chlorine), backwashing the column to maintain flow and performance, and waste management. Despite having no full-scale installation for comparison, overall, chemical denitrification may potentially be less operationally complex than biological denitrification.

3.5.4 Chemical Denitrification - Emerging Technologies

Sulfur-Modified Iron (SMI) Media

Chemical reduction of nitrate has been demonstrated for potable water treatment using sulfur-modified iron granular media (DSWA 2010). Certified to the NSF/ANSI Standard 61 for use in drinking water treatment, SMI-III® is a patented media that is recyclable and offers the advantage of multiple contaminant removal (SMI-PS 2009). Arsenic and metals can be removed via adsorption (hexavalent chromium can also be reduced and precipitated), while nitrate is reduced to ammonia (Prima Environmental N.D.) or nitrogen gas (DSWA 2010). "The SMI-III® manufacturer believes sulfur modification regulates the environment of reactions to achieve greater and a consistent nitrate reduction" (DSWA 2010, p. 9). Nitrate reduction is governed by the following reactions (SMI-PS 2009):

$$4Fe^0 + NO_3^- + 10H^+ \rightarrow 4Fe^{2+} + NH_4^+ + 3H_2O \qquad \text{(Eqn. 19)}$$

Adminsitrative Record
Page 35389

$$5Fe^{0} + 2NO_3^{-} + 12H^{+} \rightarrow 5Fe^{2+} + N_2 + 6H_2O \qquad \text{(Eqn. 20)}$$

Key advantages of SMI-III® are the ability to remove multiple contaminants simultaneously and the limited waste disposal costs relative to other nitrate removal options (no brine waste stream is produced) (DSWA 2010). Some previous research indicates inconsistent and insufficient nitrate removal to meet potable water regulations (DSWA 2010).

Damon S. Williams Associates (DSWA) and the City of Ripon, CA, conducted a pilot study investigating the use of SMI-III® in potable water treatment. Findings suggest that this treatment option may be suitable for source nitrate concentrations slightly above the MCL (up to 70 mg/L as nitrate (16 mg/L as N)). Phase A of the pilot study was operated with the SMI-III® media in an up-flow fluidized bed across a pH range of 6.0 – 6.8 and with an EBCT of 15 to 30 minutes. Phase B tested improved media performance across the same pH range and with an EBCT of 30 minutes. Figure 16 displays a process schematic of the SMI-III® process.

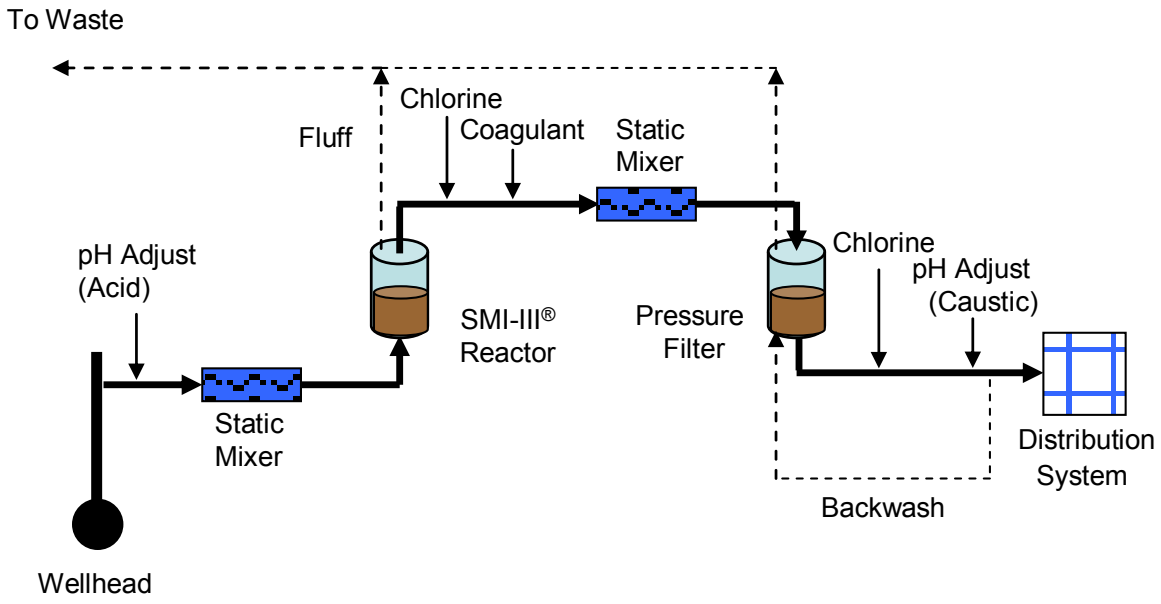


Figure 16. Process schematic for denitrification using SMI-III®. (Source: DSWA 2010.)

The media was fluffed (backwashed in up-flow mode) daily to remove oxidized iron and to avoid media agglomeration. The SMI-III® reactor was followed by coagulation and filtration for the removal of iron, arsenic, and other constituents. In phase A, the greatest nitrate removal (18 mg/L as nitrate, 4 mg/L as N) was insufficient to meet the 20% safety margin of the project (a goal of 36 mg/L as nitrate, 8 mg/L as N), with a starting nitrate concentration of 60 mg/L as nitrate (13.5 mg/L as N). Problems in phase B resulted in operation interruption; however, a maximum nitrate removal of 24 mg/L as nitrate (5.4 mg/L as N) was achieved. When the pH was reduced to 6, the system did produce water with nitrate below the MCL and the project goal nitrate level, on two occasions. Temperature was found to have the most

Adminsitrative Record
Page 35390

significant impact on removal efficiency. Nitrate reduction improved with increasing source water temperatures.

Design constraints include temperature, EBCT, avoidance of incomplete denitrification, determination of the appropriate iron to nitrate ratio, chlorine requirements for released iron oxidation, clogging from precipitates, and pH. Nitrate reduction typically increases with decreasing pH, increasing EBCT, and increasing temperature (DSWA 2010).

SMI - Cost Considerations

Cost analysis in the City of Ripon report included acid, caustic soda, and media and excluded labor and waste management. In both phases, minimal variation in nitrate reduction was found with operation at a pH of 6 versus a pH of 6.8. Operation at the higher pH minimizes costs due to prolonging media life and decreasing chemical input. The production costs for operation at a pH of 6 and 6.8 were estimated to be $2.24/1000 gal. ($729/AF) and $0.88/1000 gal. ($287/AF), respectively (DSWA 2010). Due to dissolution of the media over time, media disposal is not expected to be necessary. Dissolved iron is oxidized and then removed through filtration. In the Ripon pilot study, backwash water was discharged to the sewer; however, when "…direct sewer disposal is not feasible, the backwash water must undergo solid/liquid separation with the decant liquids recycled to the head of the treatment system and the dewatered solids sent to an appropriate landfill for disposal" (DSWA 2010, p. 82). If the waste is deemed hazardous, disposal can be a major cost consideration.

Granular Clay Media

Certified to the NSF/ANSI Standard 61 for use in drinking water treatment, MicroNose™ Technology media is currently being examined in Manteca, California, for nitrate removal in potable water treatment (MicroNose™ 2010). The media consists of "absorbent and permeable pottery granules which function similar to the mucous membrane in the human nose" (University of Hawaii 2006). Limited information is available on this emerging technology. The company states, "MicroNose™ Technology media removes heavy metals, such as arsenic, manganese, and lead as well as nitrates through non-chemical processes" (MicroNose™ 2010). MicroNose™ offers removal of multiple contaminants concomitantly and claims to be cost-effective, suitable for nitrate removal, and a green technology. Additional information is needed to assess the design considerations, costs, and applications of MicroNose™ for nitrate removal from potable water.

Powdered Metal Media

As an emerging technology, Cleanit®-LC (from North American Höganäs) is a metal powder with the potential to achieve 60 – 90% nitrate removal (Lavis 2010). Certified to the NSF/ANSI Standard 61 for use in drinking water treatment, this proprietary iron-based powder could be used for removal of co-contaminants in addition to nitrate, including "arsenic, heavy metals, phosphates and pathogens" (Lavis 2010), and potentially hexavalent chromium. Cleanit®-LC media is characterized by the following: a

Adminsitrative Record
Page 35391

density of 1800 – 2100 kg/m^3, a particle size of 150 – 850 microns, and a porosity of 60%. The powder can be used to adsorb arsenic in less than 10 minutes, with an arsenic capacity of 4 – 8 mg/g powder.

With an up-flow configuration, the treatment stream is pumped through a column containing the media, maximizing surface contact. The most significant consideration is the EBCT. At the particle surface, nitrate is reduced to nitrogen gas with EBCTs of 10 – 30 minutes. Additional key design factors are pH and temperature. In contrast to membrane technologies, the burdens of disposal are minimized because nitrate is reduced rather than removed. There is no concentrated brine solution requiring costly disposal. Preliminary third party results indicate nitrate removal over 7 months to below the MCL (North American Höganäs, data). However, as a new product on the market, further research is required to assess Cleanit®-LC for the removal of nitrate and other constituents in potable water treatment.

3.5.5 Chemical Denitrification - Cost Considerations

For efficient operation of a chemical denitrification system, maintaining efficient nitrate reduction is essential. Optimal performance includes managing pre and post-treatment to provide appropriate environmental conditions, while meeting necessary potable water guidelines. Factors affecting system cost can include facility size (flow rate), source water quality (including nitrate concentration and co-contaminants), environmental factors (temperature and pH), and target effluent nitrate concentration.

Capital costs for chemical denitrification include land, housing, piping, media, storage tanks, O&M equipment, preliminary testing (through pilot studies), permits, and operator training. O&M costs include pre and post-treatment, media replenishment and disposal, backwashing, chemical use (e.g., pH adjustment, chlorine), repair, monitoring, maintenance, power, and labor.

The availability of published cost information for chemical denitrification is strictly limited to pilot-scale studies, due to the lack of full-scale systems. Cost analysis in the City of Ripon report included acid, caustic soda, and media and excluded labor and waste management. In both phases, minimal variation in nitrate reduction was found with operation at a pH of 6 versus a pH of 6.8. Operation at the higher pH minimizes costs due to prolonging media life and decreasing chemical input. The production costs for operation at a pH of 6 and 6.8 were estimated to be $2.24/1000 gal. ($729/AF) and $0.88/1000 gal. ($287/AF), respectively (DSWA 2010). Due to dissolution of the media over time, media disposal is not expected to be necessary. Dissolved iron is oxidized and then removed through filtration. In the Ripon pilot study backwash water was discharged to the sewer. However, when "…direct sewer disposal is not feasible, the backwash water must undergo solid/liquid separation with the decant liquids recycled to the head of the treatment system and the dewatered solids sent to an appropriate landfill for disposal" (DSWA 2010, p. 82). If the waste is deemed hazardous, disposal can be a major cost consideration. A detailed discussion of treatment costs is included in Section 6 Treatment Cost Analysis.

Adminsitrative Record
Page 35392

3.5.6 Chemical Denitrification - Selected Research

Table A.5 of the Appendix lists recent research studies relevant to the use of chemical denitrification in potable water treatment.

3.5.7 Chemical Denitrification - Summary of Advantages and Disadvantages

A summary of advantages and disadvantages of chemical denitrification in comparison with other treatment options is listed in Table A.6 of the Appendix. Advantages of chemical denitrification for nitrate removal from potable water include the conversion of nitrate to other nitrogen species (no brine or concentrate waste stream), the potential for more sustainable treatment, and the ability to remove multiple contaminants.

Problems with chemical denitrification of potable water are the potential reduction of nitrate beyond nitrogen gas to ammonia, the possibility of partial denitrification, and the associated production of GHGs, and the lack of full-scale chemical denitrification systems.

3.6 Brine Treatment Alternatives and Hybrid Treatment Systems

Brine or concentrate disposal can be a great concern with the use of the removal processes, IX, RO, and ED/EDR, especially for inland communities. Brine treatment and recycling alternatives have the potential to address disposal concerns, improving sustainability and decreasing operating costs. Hybrid systems, combining different nitrate treatment technologies, have been explored to include the advantages of multiple treatment options, while avoiding their respective disadvantages (Pintar et al. 2001; Wisniewski et al. 2001; Matos et al. 2006; Kabay et al. 2007; Van Ginkel et al. 2008). Table 15 lists a selection of additional research related to brine treatment alternatives and the use of hybrid systems to improve nitrate treatment performance.

The combination of denitrification methods with removal technologies enables resolution of common problems with each option. The brine waste stream from IX, for example can be treated using biological denitrification to reduce nitrate to nitrogen gas. Biological denitrification of IX waste brine using an up-flow sludge blanket reactor (USBR) was demonstrated at the lab-scale by van der Hoek & Klapwijk (1987). Clifford & Liu (1993) implemented a lab-scale sequencing batch reactor (SBR) for the denitrification of waste brine resulting in the ability to recycle the treated waste brine for 15 regeneration cycles. The SBR process was pilot tested in McFarland, CA (Liu & Clifford 1996). With denitrification of spent brine followed by reuse, a 95% decrease was achieved in salt waste. Use of a membrane bio-reactor in this context has also been explored (Bae et al. 2002; Chung et al. 2007; Van Ginkel et al. 2008). Through the reduction or destruction of nitrate in spent IX brine, disposal needs may be significantly reduced as treated brine can be repeatedly recycled back for use in resin regeneration. Additionally, through treatment of RO or ED/EDR waste concentrate, removal of nitrate from the waste stream may improve disposal options.

Adminsitrative Record
Page 35393

Table 15. Selected research on brine treatment alternatives and hybrid systems for nitrate treatment of potable water.

IX and Catalytic Reduction	Pintar et al. (2001)
RO and EDR	EET Corporation (2008)
IX and EDR	Kabay et al. (2007)
ED and MBR	Wisniewski et al. (2001)
IX and Biological Denitrification	Van der Hoek & Klapwijk (1987), Clifford & Liu (1993), Bae et al. (2002), and Van Ginkel et al. (2008)
RO and VSEP	Lozier et al. (N.D.)
Electrochemical destruction of nitrate (in IX brine)	Yu & Kupferle (2008), Dortsiou et al. (2009), Goltz & Parker, 2010/2011, and Ionex SG Limited (2011)

3.6.1 Electrochemical Destruction of Nitrate in Waste Brine

Research by two different groups is focused on electrochemical destruction of nitrate for the treatment of waste brine.

First, two major companies are collaborating on the development of a system which incorporates IX with electrochemical destruction of nitrate. With the removal of nitrate from spent brine, the brine can be recycled for reuse in regeneration (Goltz 2010; Goltz & Parker 2010/2011). Spent brine is treated following IX treatment. With electricity distributed over a high surface area electrode, upon contacting the electrode surface, nitrate in the brine waste is reduced to nitrogen gas. In the overall reaction, nitrate is reduced and water is oxidized producing nitrogen gas and oxygen. Possible reduction to ammonia/ammonium is accounted for in the process. Laboratory tests have been successful and planning of site tests is underway to examine system performance across a variety of raw water characteristics. The potential benefits of this brine treatment alternative are in the increased sustainability and the decreased disposal costs. The objective is to optimize the process such that the savings on disposal costs, especially for inland communities, outweigh the increased electricity costs of the process.

Second, a company based in the United Kingdom, Ionex SG Limited, has developed and is currently testing a patented brine treatment system utilizing an electrochemical cell for the destruction of nitrate (Ionex SG Limited 2011). Following resin regeneration, spent brine is passed through the cell and nitrate is converted to nitrogen and oxygen gases through interaction with a rhodium catalyst (Figure 17). Treated brine can be subsequently reused for regeneration, minimizing waste brine volume as well as the overall salt consumption of the ion exchange system. The lifespan of the cell has been laboratory tested and is estimated to operate efficiently for at least 15 years through a mechanism of automatic maintenance of the reactive surface (Tucker et al. 2004; Ionex SG Limited 2011). The brine treatment system is currently being pilot tested at a location in California through collaboration with UC Davis. A cost benefit analysis by Ionex indicates that brine disposal costs would be significantly reduced; chemical and power costs of the brine treatment system balance with existing chemical costs for typical ion exchange systems.

Adminsitrative Record
Page 35394

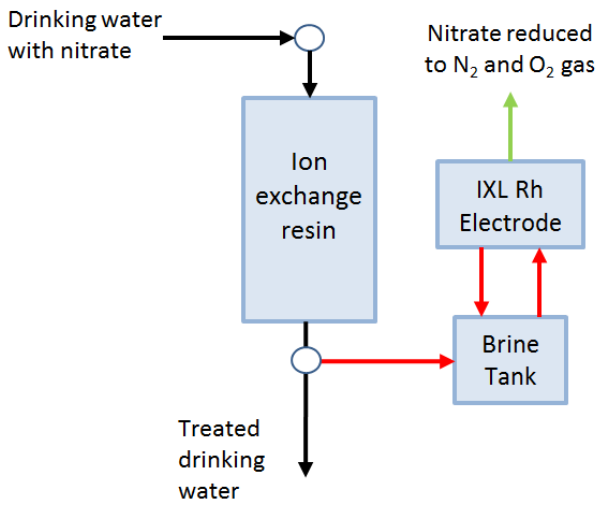


Figure 17. Schematic of the brine treatment system developed by Ionex SG Limited. (Source: Ionex SG Limited 2011.)

3.6.2 Catalytic Treatment of Waste Brine

Calgon Carbon is working with the University of Illinois to develop a brine recovery system with the goal of up to 90% brine recovery for reuse using Pd-based catalytic treatment (Drewry 2010). The following information was provided by Dr. Charles Werth (2010) from the University of Illinois.

As illustrated in Figure 18, "The spent brine solution is equilibrated with hydrogen in a gas-liquid membrane, and subsequently treated to remove nitrate in the packed bed catalyst system containing Pd-In on granular activated carbon (GAC). The treated brine is then put back into a holding tank and reused when ion exchange breakthrough occurs again" (Werth 2010).

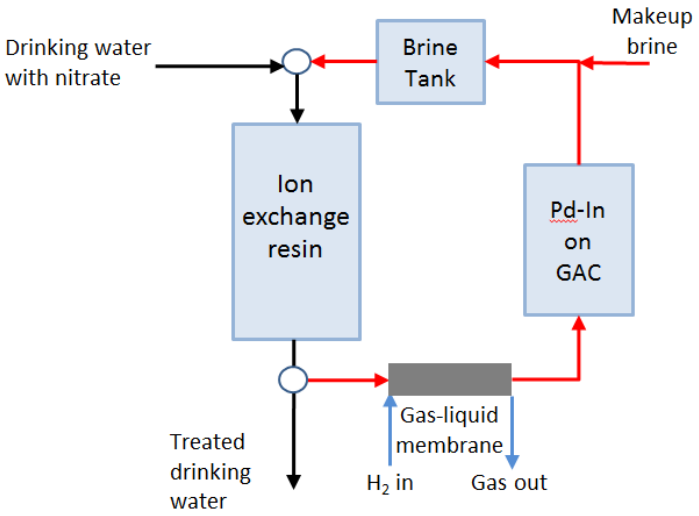


Figure 18. Schematic of an ion exchange system with brine regeneration coupled with catalytic treatment of brine for reuse. (Source: Werth 2010.)

Adminsitrative Record
Page 35395

Advantages of this brine treatment alternative include conversion of nitrate mainly to nitrogen gas, avoidance of the need for brine waste disposal, and rapid reduction of nitrate with the Pd-based catalyst. Disadvantages include the use of flammable hydrogen gas, some conversion of nitrate to ammonia, "the cost of Pd and potential fouling of the catalyst which requires regeneration using a strong oxidant like hypochlorite (Chaplin et al. 2007)" (Werth 2010). Catalytic removal of nitrate has not yet been implemented at the full-scale, but "catalytic systems have been used to remove chlorinated solvents at contaminated groundwater field sites, and they appear to be economically competitive (Davie et al. 2008)" (Werth 2010). Dr. Werth stated, "Catalytic systems show great promise for removing nitrate from IX brines, but pilot-scale studies are needed to evaluate the economic viability."

3.7 Residential Treatment (Point-of-Use, Point-of-Entry)

Point-of-Use (POU) and Point-of-Entry (POE) water treatment devices can be used to address high nitrate levels and other constituents of concern at the residential scale. A POU treatment device is installed for the purpose of reducing contaminants in drinking water at a single tap, typically the kitchen tap. A POE treatment device is installed for the purpose of reducing contaminants in drinking water entering a house or building.

Treatment technologies for POU and POE systems, used to address nitrate contamination, include IX, RO, and distillation (Mahler et al. 2007). IX is generally considered more for POE than for POU and requires disposal of concentrated waste brine. RO systems require more maintenance and have lower water recovery, resulting in a larger waste volume (Mahler et al. 2007). However, according to the New Hampshire Department of Environmental Services (2006), RO "is the most cost-effective method for producing only a few gallons of treated water per day" (NHDES 2006). While distillation can require lower maintenance, energy demands are higher than the other options. Distillation systems are generally intended as POU devices as they remove all minerals and produce water that is aggressive towards plumbing materials.

POU units are installed either under the counter or on the counter top, preferably by a licensed professional. The treatment units generally consist of several stages; for example, a POU RO system can consist of a pre-treatment filter, an RO stage, and a post-treatment filter. The system can also include a storage tank to hold treated water and a conductivity meter to indicate when maintenance is required. POE units, for the treatment of all water entering a building, are larger and require more piping.

Certification to the relevant ANSI/NSF standards by an ANSI accredited third party certifier ensures the safety and performance of the residential treatment systems. In the U.S., the following certifiers have been accredited by ANSI to certify drinking water treatment systems:

- Canadian Standards Association International (www.csa-international.org);
- International Association of Plumbing & Mechanical Officials (www.iapmo.org);
- NSF International (www.nsf.org);
- Underwriters Laboratories Inc. (www.ul.com); and

Adminsitrative Record
Page 35396

- Water Quality Association (www.wqa.org).

Numerous RO devices for nitrate removal are certified to the ANSI/NSF standard specific to RO POU devices: NSF Standard 58 - Reverse osmosis drinking water treatment systems (NSF 2009). Additionally, the Water Quality Association, an accredited certifier, lists two POU ion exchange devices for nitrate removal that are certified to NSF Standard 53: Drinking Water Treatment Units – Health Effects (Water Quality Association 2011). All the technologies listed above are capable of reducing nitrate levels; however, proper maintenance of the treatment equipment is fundamental to ensure the provision of safe drinking water. Additionally, it is important to conduct periodic testing (annually or as recommended by the manufacturer) using an accredited laboratory on both the influent water and the water produced by the treatment system to verify that it is working effectively.

CDPH provides a list of approved POU devices for nitrate treatment consisting predominantly of RO devices (CDPH 2011). Published cost information for POU systems is listed in Table 16. Based on a survey in Minnesota, "the average cost of nitrate removal systems was $800 to install and $100 per year to maintain" (Lewandowski 2008, p. 92A). Providing a more detailed cost analysis, the U.S. EPA has developed a cost estimating tool for the use of POU and POE devices (U.S. EPA 2007; U.S. EPA 2011).

It is important to note that water systems are responsible for meeting federal, state, and local requirements and the allowable uses of POU/POE devices vary by state. Current California regulations enable small public water systems to use POU devices to meet the nitrate MCL for up to three years, with certain restrictions, including the following (California Code of Regulations 2011, p. 2):

"…a public water system may be permitted to use point-of-use treatment devices (POUs) in lieu of centralized treatment for compliance with one or more maximum contaminant levels… if;

(1) the water system serves fewer than 200 service connections,

(2) the water system meets the requirements of this Article,

(3) the water system has demonstrated to the Department that centralized treatment, for the contaminants of concern, is not economically feasible within three years of the water system's submittal of its application for a permit amendment to use POUs,

… no longer than three years or until funding for the total cost of constructing a project for centralized treatment or access to an alternative source of water is available, whichever occurs first…."

Adminsitrative Record
Page 35397

Table 16. Costs of POU treatment for nitrate removal.[1]

	Upfront Investment	Annual Costs	Comments
Ion Exchange	$660 – $2425	Salt costs ($3.30 – $4.40/bag)	Requires disposal of brine waste
Distillation 4-10 gal/d	$275 – $1650	$440 – $550/yr + electricity	Requires scale removal, higher energy use
Reverse Osmosis 2-10 gal/d	$330 – $1430	$110 – $330/yr + electricity	Requires filter replacement, high maintenance, lower water recovery

Costs have been adjusted to 2010 dollars, unless indicated otherwise.

[1] Mahler et al. (2007).

Adminsitrative Record
Page 35398

4 Tulare Lake Basin and Salinas Valley - Water Quality Analysis

As mentioned above in the discussion of treatment technologies, water quality is a key variable in both treatment selection and cost. Table 17 is a summary of water quality data of high nitrate wells across both the Tulare Lake Basin and the Salinas Valley. Constituents that interfere with treatment (e.g., sulfate) or affect treatment technology selection (e.g., TDS) are included as well as co-contaminants. Table 17 lists water quality data for all wells with raw water monitoring data in the CDPH water quality database from 2006 – 2010 having nitrate levels above the MCL, including both abandoned and inactive wells, to enable consideration of the complete range of scenarios. This water quality summary highlights the wide range of water quality characteristics encountered in the region of interest. The GAMA Priority Basin Projects provide additional water quality information including the basins within the study area (USGS 2011).

Table 17. Summary of water quality data of high nitrate wells in the Tulare Lake Basin and Salinas Valley. (Source: CDPH PICME and WQM databases.)

Constituent	Average	Min	Max	# Samples	# PWS	# Sources	Units
Nitrate as NO_3	64.69	45.1	402	1705	159	209	mg/L (as NO_3)
Ammonia	1	-	-	2	1	1	mg/L (NH_3-N)
Sulfate	116.43	3.7	2300	1594	78	111	mg/L
pH (Lab)	7.91	6.38	9.6	1582	87	120	
pH (Field)	7.12	6.52	7.85	1186	3	7	
Temperature	19.71	3	24.4	780	5	10	^{o}C
Hardness	344.7	6.7	5000	1562	88	121	mg/L as $CaCO_3$
Iron	126.34	0	5700	1586	84	117	ug/L
Manganese	11.86	0	400	1559	83	116	ug/L
Chloride	100.04	3.5	16000	2502	78	111	mg/L
Silica	1693.31	28	21000	71	4	5	mg/L
TDS (Conductance)	856.65	23	43600	4176	101	136	
TDS (TDS)	780.04	120	28700	2143	78	111	mg/L
Alkalinity (Total)	203.85	32	340	1572	79	112	mg/L as $CaCO_3$
Arsenic	3.24	0	53	2228	99	137	ug/L
Chromium (Total)	2.9	0	45	1785	96	132	ug/L
Chromium (Hex)	1.19	0.2	3.5	20	2	2	ug/L
Perchlorate	3.09	0	2000	2637	107	138	ug/L
Uranium	15.64	8.4	58	532	3	5	ug/L
TOC	3.61	1.2	6.6	85	1	1	mg/L
Turbidity (Lab)	0.55	0	8.9	1494	74	108	NTU

Adminsitrative Record
Page 35399

4.1 Water Quality - Treatment Interference

The most commonly employed nitrate treatment method, ion exchange (IX), is highly sensitive to competing anions in the treatment stream. With the use of generic anion exchange resin for nitrate treatment, sulfate will out-compete nitrate for resin sites and, as sulfate levels in source water increase, the resin capacity for nitrate will decrease. Under such circumstances, the cost of more frequent regeneration, associated chemicals and disposal may make ion exchange less feasible. Nitrate dumping can also be a problem, resulting in effluent nitrate levels higher than influent levels due to sulfate displacing nitrate on the resin. To address these concerns, nitrate selective resin can be used, which is more expensive, but can maintain a higher capacity for nitrate. At the highest levels in the above listed sulfate range, alternative treatment options would likely become more cost-effective.

High hardness, silica, TSS, turbidity, manganese, and iron can impact most nitrate treatment options requiring additional steps for pretreatment such as filtration and anti-scalant addition to avoid resin (IX) or membrane fouling (RO and EDR). Cleaning and post-treatment requirements can also be affected, thus increasing system complexity and overall costs.

4.2 Water Quality - Co-contaminants

The need for multiple contaminant removal is a key factor in the selection of the most appropriate treatment option. While IX can effectively remove several co-contaminants, this technology cannot address all constituents of concern and alternative options should be considered with particularly poor quality waters. IX has been used for simultaneous removal of perchlorate and nitrate, and arsenic and nitrate; however, specific resins may be required to optimize multiple contaminant removal, resulting in higher costs than IX for nitrate removal alone. Figures 19 – 22 highlight the co-occurrence of nitrate with other constituents of interest in the Tulare Lake Basin and Salinas Valley.

Figure 19 maps raw water nitrate levels above the MCL (45 mg/L as nitrate) including both active and inactive well data from 2006 – 2010. Figure 20 maps high nitrate wells with high arsenic levels for which IX may still be considered or for which RO may be a more suitable treatment option, depending on the priorities of a given system. Figure 21 maps high nitrate wells with high perchlorate levels; again IX may be considered under such circumstances. Alternatively, biological denitrification may be implemented to simultaneously remove both constituents while avoiding the brine disposal problem. Last, Figure 22 maps high nitrate wells in which at least 1 of 4 major pesticides has been detected (bromacil, simazine, atrazine, and DBCP). The co-occurrence of nitrate and pesticides is important on two fronts. With pesticide levels above the MCL, treatment requirements will change. RO may be implemented to address the multiple contaminants or IX could be used for nitrate and activated carbon for pesticides. Regardless of the selected option, treatment for nitrate and pesticides will be more expensive than treatment for only nitrate. Additionally, the co-occurrence may be indicative of the source of nitrate contamination.

Adminsitrative Record
Page 35400

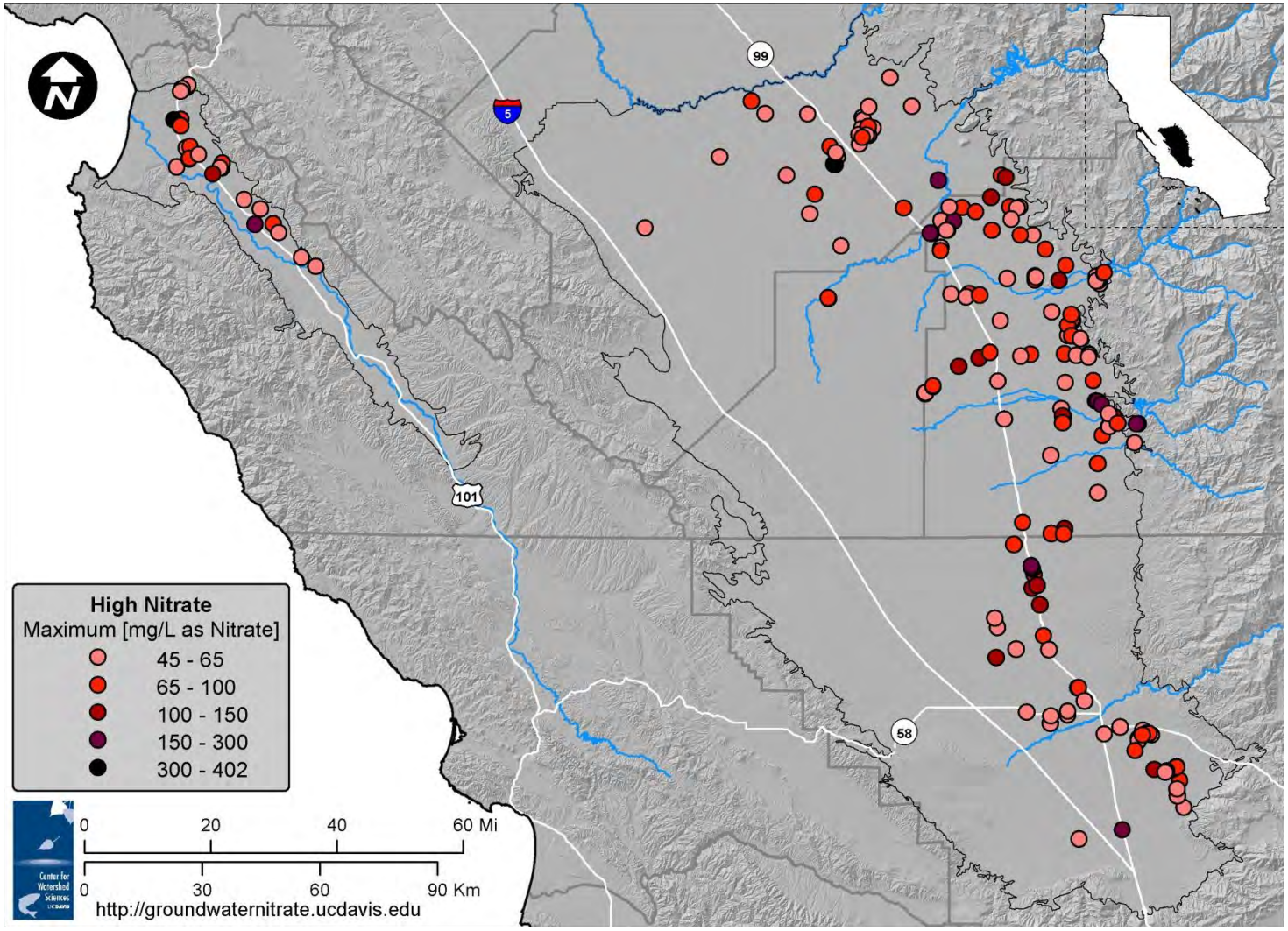


Figure 19. Raw water nitrate levels exceeding the MCL (45 mg/L as nitrate). (Source: CDPH PICME and WQM Databases.)

Adminsitrative Record
Page 35401

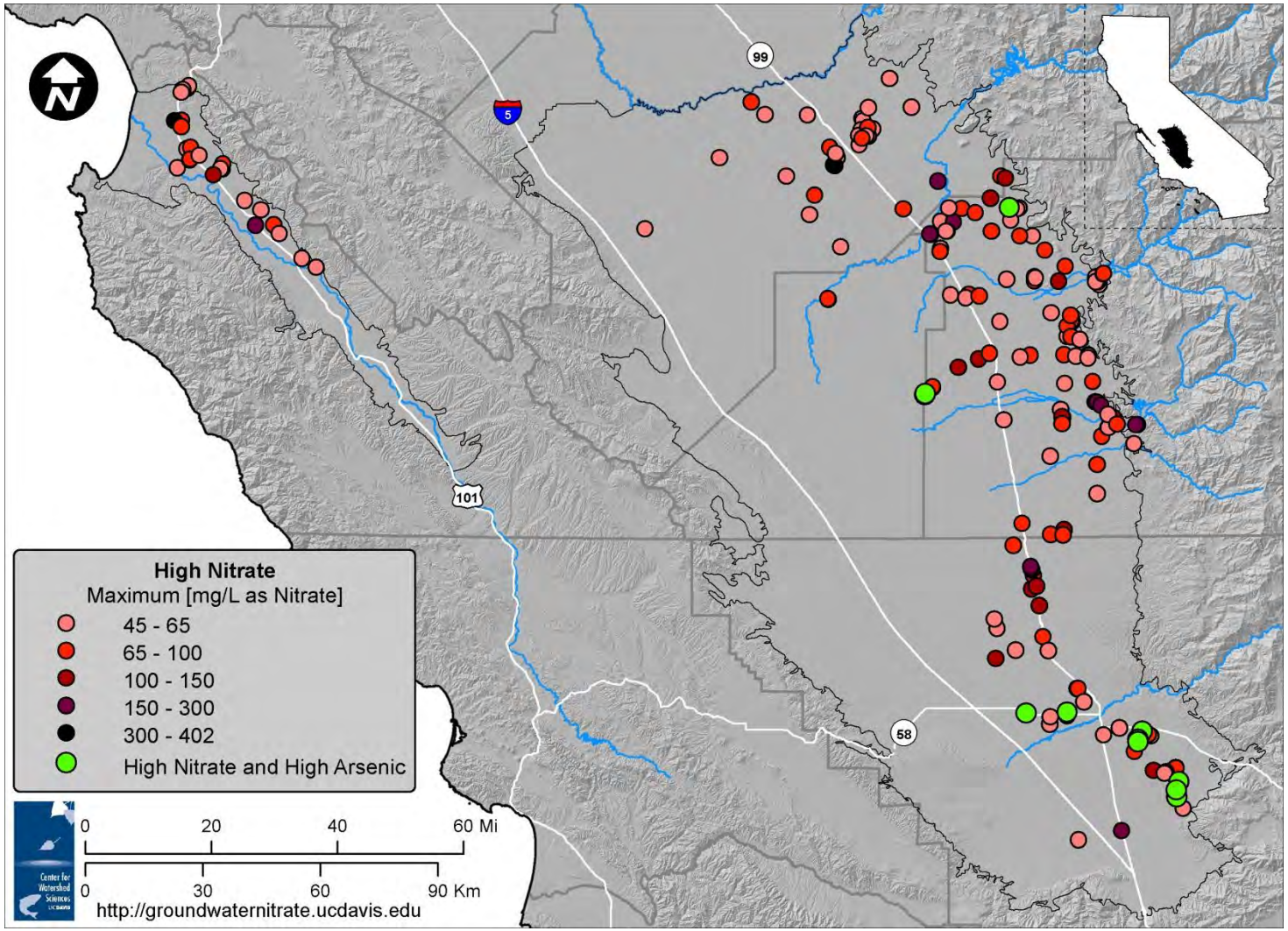


Figure 20. Raw water high nitrate wells with high arsenic levels. (Source: CDPH PICME and WQM Databases.)

Adminsitrative Record
Page 35402

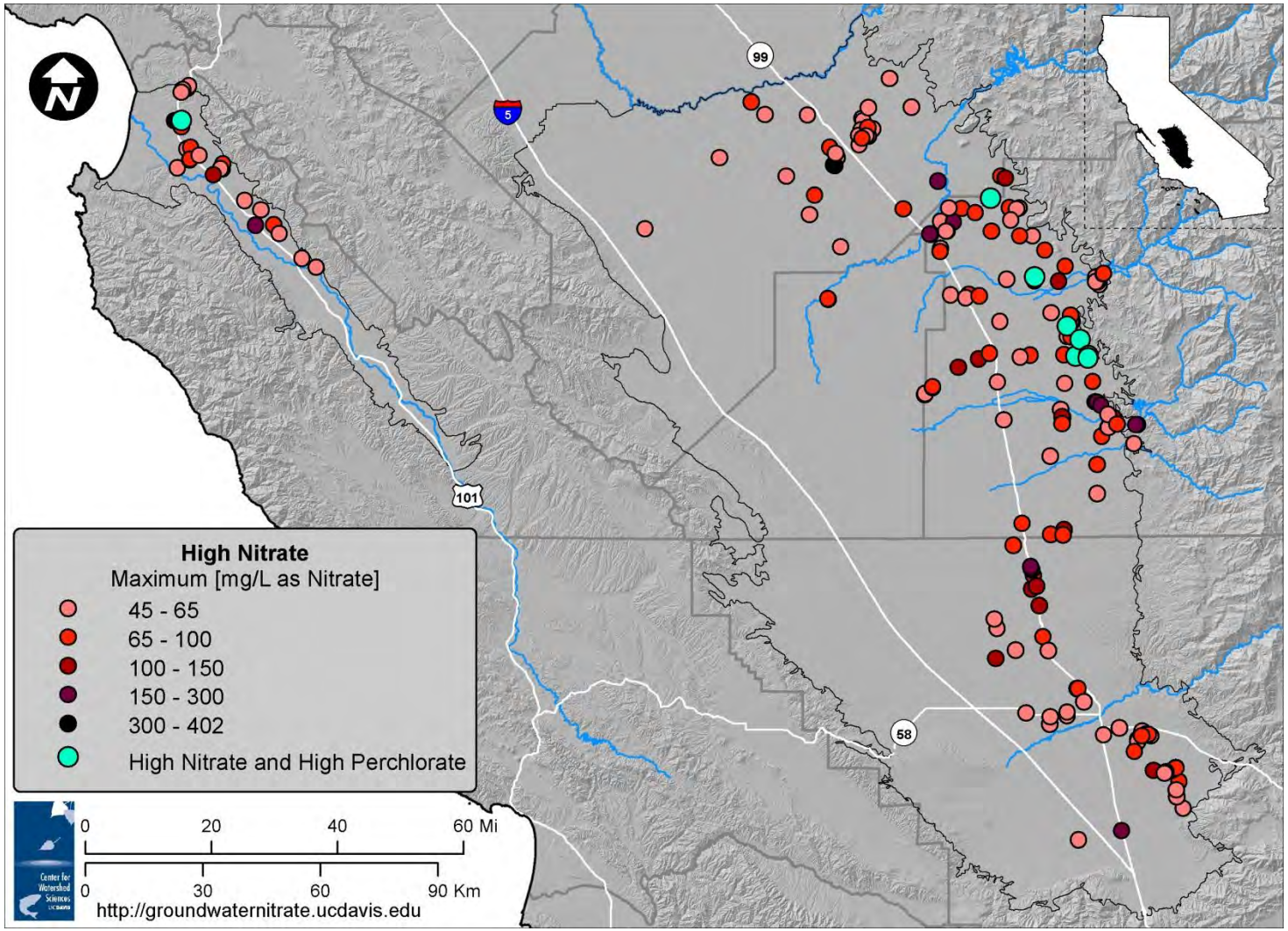


Figure 21. Raw water high nitrate wells with high perchlorate levels. (Source: CDPH PICME and WQM Databases.)

Adminsitrative Record
Page 35403

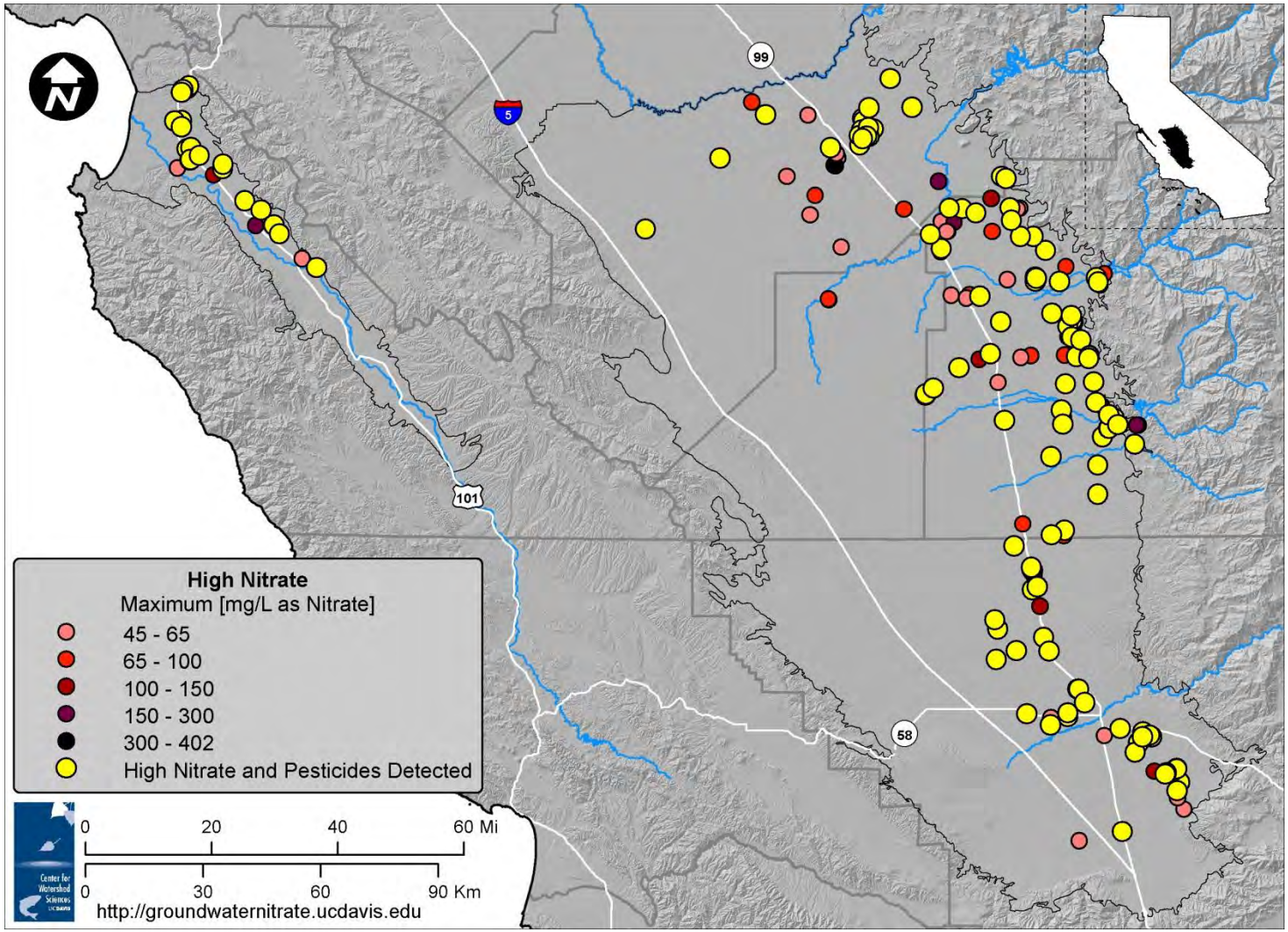


Figure 22. Raw water high nitrate wells with pesticides detected. (Source: CDPH PICME and WQM Databases.)

Adminsitrative Record
Page 35404

Through the investigation of local water quality data, the relationship between well depth and the incidence of nitrate and arsenic emerged as a potential concern for water systems drilling deeper wells to reach groundwater with lower nitrate levels. The incidence of nitrate impacted groundwater was suspected to decrease with well depth while the incidence of arsenic impacted groundwater was suspected to increase with well depth. Further analysis of this option requires that well screen depth is known in addition to well water quality. Unfortunately, for most wells with water quality information, depth information is not available (See Technical Report 4 (Boyle et al. 2012) for additional information on wells data). All wells having depth information and arsenic testing data were included, as were all wells having depth information and nitrate testing data (CDPH PICME [Permits Inspection Compliance Monitoring and Enforcement] and WQM [Water Quality Monitoring] databases). The available dataset leads to a potential bias as it excludes wells with testing data for which depth information was unavailable. For the examination of nitrate and depth, the depth to the top of the screened interval was used; depth categories are based on the minimum screened depth of the well. For the examination of arsenic and depth, well depth was calculated as the sum of the depth to the top of the screen and the length of the screen; depth categories are based on the maximum screened depth of the well. Nitrate and arsenic levels were averaged for each well across all available tests from 2006 – 2010 to avoid bias caused by having many samples for some wells and only few samples for others. Constituent concentrations were subsequently averaged across all wells in each depth category.

No relationship was found in the Salinas Valley (Figure 23); however this may be due, in part, to the more limited sample size of Salinas Valley wells (192 wells for nitrate analysis, 142 wells for arsenic analysis) in comparison with the sample size of Tulare Lake Basin wells (826 wells for nitrate analysis, 741 wells for arsenic analysis). In the Tulare Lake Basin, the variation between individual wells within each depth category and across depth categories leads to inconclusive results lacking statistical significance, despite the suggestion of the expected trend (Figure 24). However, in the context of the nitrate and arsenic MCLs, results in the Tulare Lake Basin suggest an increase in the incidence of arsenic MCL exceedance with well depth and a decrease in the incidence of nitrate MCL exceedance with well depth (Figure 25). Additional data are necessary for definitive confirmation of this trend and local conditions can vary significantly as water quality varies substantially with well location, well design, and subsurface geology.

Adminsitrative Record
Page 35405

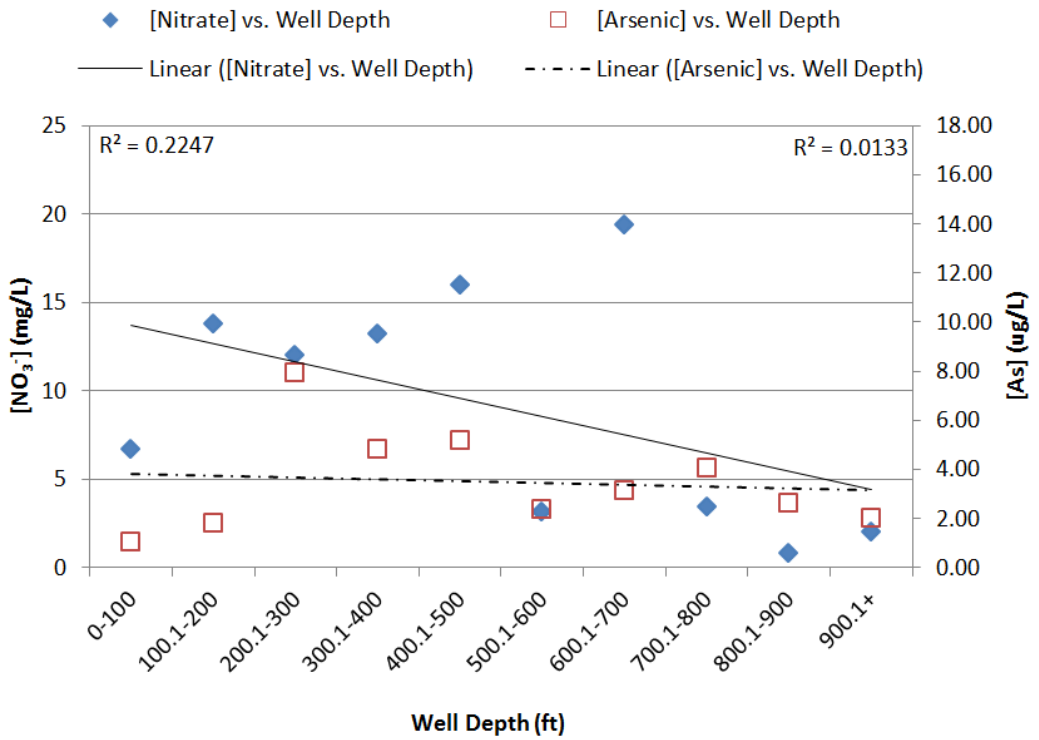


Figure 23. Salinas Valley [As] versus total well depth (deepest water) and [NO_3^-] versus depth to top of screen (shallowest water). [1 ft=0.30 meters]

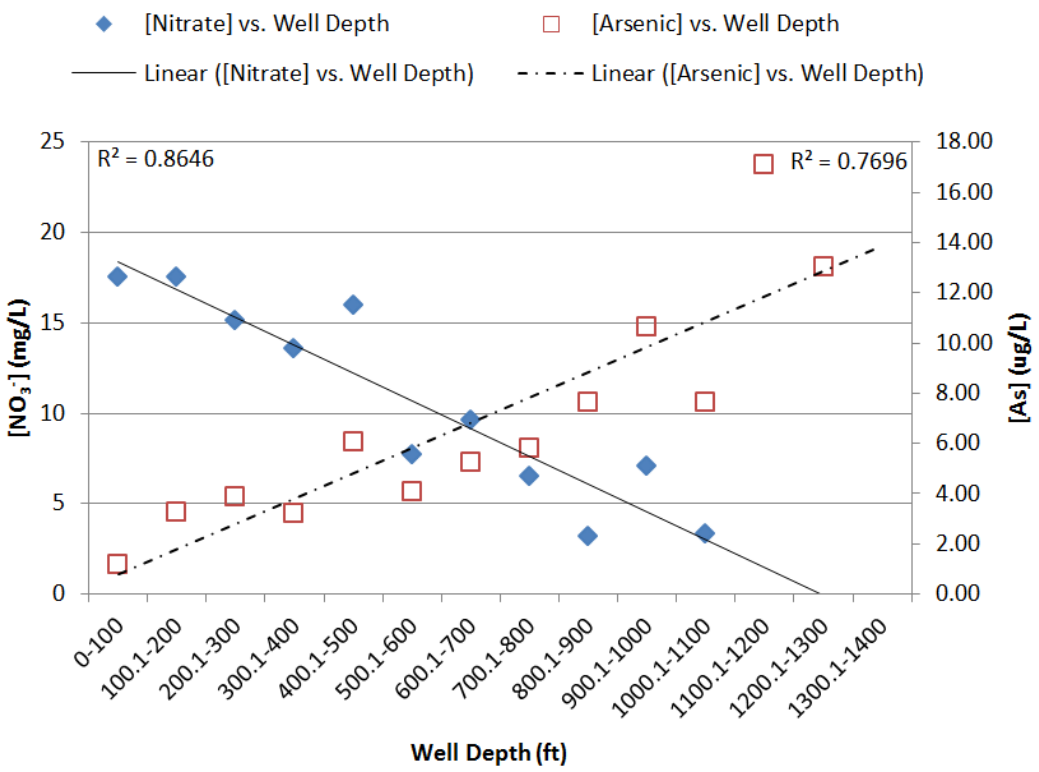


Figure 24. Tulare Lake Basin [As] versus total well depth (deepest water) and [NO_3^-] versus depth to top of screen (shallowest water). [1 ft=0.30 meters]

Adminsitrative Record
Page 35406

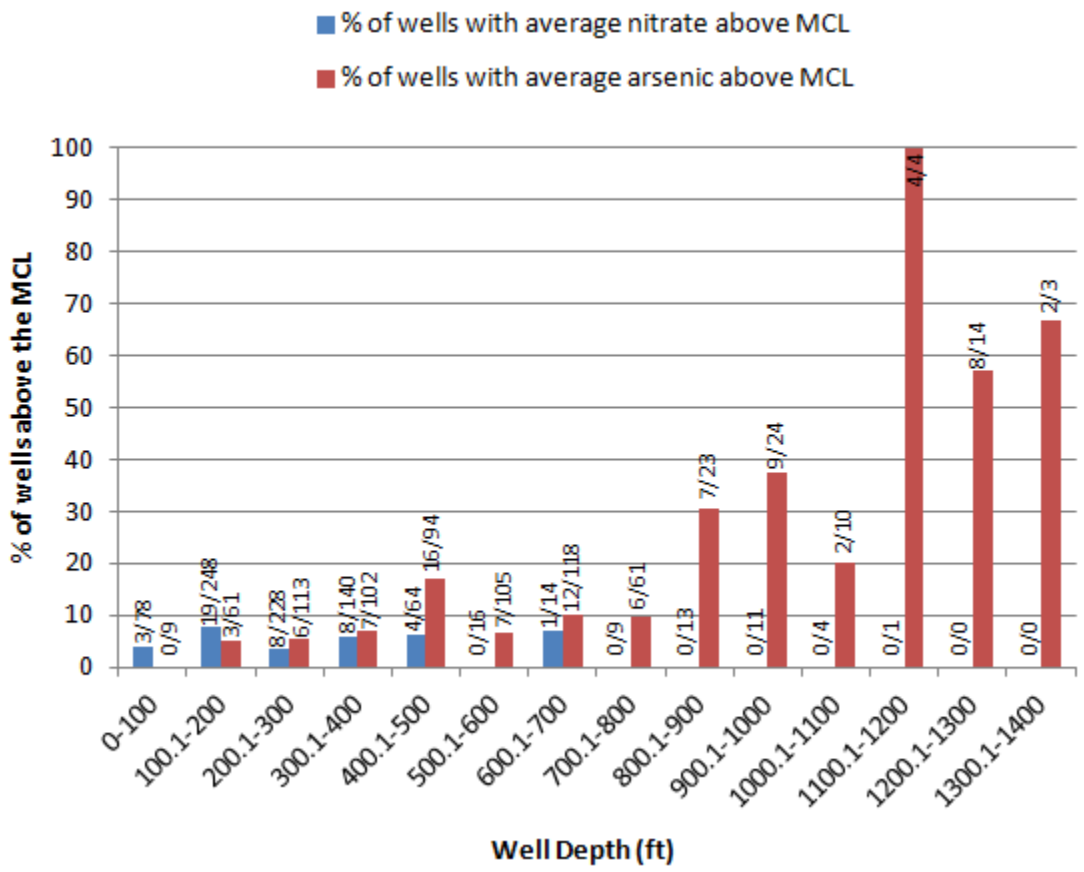


Figure 25. Tulare Lake Basin: Incidence of nitrate and arsenic MCL exceedance with well depth. [1 ft=0.30 meters]

4.3 Water Quality and Treatment Selection

Within the drinking water community, the treatment options typically considered to address nitrate contamination are IX and RO. Other technologies are available or emerging (EDR, BD, CD) because, under some circumstances, the alternatives offer advantages that IX and RO cannot. New technologies will continue to be investigated and developed because no single option is ideal for all situations. There is not a nitrate treatment option currently available that can address all possible scenarios at a lower cost than all other options. The following diagram is a rough guide for treatment technology selection based on water quality concerns and possible priorities for a given water source or system (Table 18). This diagram includes generalizations and is not intended to be definitive. In the selection of nitrate treatment technologies the unique needs of an individual water system must be assessed by professional engineers to optimize treatment selection and design.

As Table 18 shows, the most appropriate method to address nitrate contamination can be influenced by influent nitrate concentrations as well as other water quality parameters. Nitrate levels well above the

Adminsitrative Record
Page 35407

MCL may lead to the selection of one treatment option while nitrate levels just above the MCL may be more cost-effectively addressed with a different treatment option. Table 19 lists several scenarios as an example of appropriate options based on influent nitrate level and water system characteristics.

Table 18. Comparison of major treatment types.[1]

Concerns	IX	RO	EDR	BD	CD
High Nitrate Removal					
High TDS Removal					
Arsenic Removal					
Radium and Uranium Removal					
Chromium Removal					
Perchlorate Removal					

Priorities	IX	RO	EDR	BD	CD
High Hardness Not a Major Concern					
Reliability					
Training/ Ease of operation					
Minimize Capital Cost					
Minimize Ongoing O&M Cost					
Minimize Footprint					
Industry Experience					
Ease of Waste Management					

Good → Poor; Unknown (blank)

[1] Ion Exchange (IX), Reverse Osmosis (RO), Electrodialysis Reversal (EDR), Biological Denitrification (BD), Chemical Denitrification (CD). This table offers a generalized comparison and is not intended to be definitive. There are notable exceptions to the above classifications.

Table 19. Influence of nitrate concentration on treatment selection.[1]

Option	Practical Nitrate Range	Considerations
Blend	10 – 30% above MCL	Dependent on capacity and nitrate level of blending sources.
IX	Up to 2X MCL	Dependent on regeneration efficiency, costs of disposal, and salt usage. Brine treatment, reuse, and recycle can improve feasibility at even higher nitrate levels.
RO	Up to many X the MCL	Dependent on availability of waste discharge options, energy use for pumping, and number of stages. May be more cost-effective than IX for addressing very high nitrate levels.
BD	Up to many X the MCL	Dependent on the supply of electron donor and optimal conditions for denitrifiers. Ability to operate in a start-stop mode has not yet been demonstrated in full-scale application; difficult to implement for single well systems. May be more cost-effective than IX for addressing high nitrate levels.

[1] Based on contact with vendors and environmental engineering consultants.

Adminsitrative Record
Page 35408

5 Addressing Nitrate Impacted Potable Water Sources in California

In the Tulare Lake Basin and Salinas Valley, current and historical methods to address nitrate contamination of potable water supplies include well abandonment, destruction, and inactivation; blending; and treatment. The incidence of well abandonment, destruction, or inactivation in California was explored through analysis of the CDPH PICME and WQM databases. Through collaboration with CDPH, systems treating or blending to address nitrate contamination were identified and a survey was conducted to collect additional information, including the costs of treatment.

5.1 Well Abandonment, Destruction, and Inactivation

If alternative source wells are available, costly treatment is often avoided through abandonment or inactivation of wells. However, wells must be properly destroyed or abandoned, in accordance with local requirements, to avoid hazardous conditions and the potential for groundwater contamination. The cost of proper well destruction and abandonment varies with numerous factors including depth, subsurface conditions, well type, and well construction. The minimum cost to properly destroy a 300 – 400 ft well is ~$15,000; use of best practices would increase cost (Aegis Groundwater Consulting 2011).[20]

To assess the incidence of well abandonment and inactivation due to nitrate contamination, an analysis of the CDPH PICME and WQM databases was performed. Nitrate records from 2006 – 2010 for wells labeled as abandoned, destroyed, or inactive were examined. Wells with at least one nitrate test above the MCL and wells including “NO3” or “Nitrate” in the well description were flagged. Table 20 lists the resulting number of nitrate impacted wells abandoned, destroyed, or inactive in the study area and also across California; locations are mapped in Figure 26. There is evidence of mislabeling in the PICME/WQM database. Wells missing from this analysis which may have been abandoned, destroyed, or inactivated due to nitrate may have records that are not up to date or may be mislabeled. This analysis utilizes exceedance of the nitrate MCL as an indicator of the reason for well status change; however, a portion of these wells may have been abandoned, destroyed, or inactivated for reasons other than nitrate contamination. The purpose of this analysis was to assess the incidence of well abandonment, destruction, and inactivation due to nitrate contamination. However, the small number of wells identified as abandoned, destroyed, or inactivated due to nitrate, relative to the total number of wells in these categories (which were abandoned, destroyed, or inactivated for any number of reasons) leads to two possible conclusions: the reason for well status change is not consistently identified in the CDPH database or there are simply very few wells in these categories. However, comparison of the frequency of abandonment, destruction, and inactivation of wells due to nitrate within the study area

[20] Using best practices to properly destroy wells, “all wells would be perforated top to bottom, a high-grade cement-sand concrete would be used, and the concrete would be pressure grouted into the formation and then allowed to fill the well” (Aegis Groundwater Consulting 2011). Use of best practices for well destruction would increase the cost above the $15,000 minimum.

Adminsitrative Record
Page 35409

with those across California indicates that the study area accounts for ~22% of such wells across the state; a disproportionate number as the total number of wells in these categories accounts for only ~13% of the CA total. Analysis of the frequency of abandoned and destroyed wells and their relevance to nitrogen loading is discussed in Technical Report 2, Section 9, including agricultural wells (Viers et al. 2012).

Table 20. Incidence of abandonment, destruction, and inactivation of nitrate impacted drinking water wells.[1]

	Nitrate Impacted Wells				Total Wells	
	TLB	**SV**	**Study Area Total**	**CA**	**Study Area Total**	**CA**
Destroyed	1	0	1	9	217	2,315
Abandoned	2	1	3	28	494	2,584
Inactive	33	2	35	138	1,001	8,253
Total	***36***	***3***	***39*** ***22% of CA total***	***175***	***1,712*** ***13% of CA total***	***13,152***

[1] Source: CDPH PICME and WQM databases. There is evidence of mislabeling in the PICME/WQM database. Wells missing from this analysis which may have been abandoned, destroyed, or inactivated due to nitrate may have records that are not up to date or may be mislabeled. This analysis utilizes exceedance of the nitrate MCL as an indicator of the reason for well status change; however, a portion of these wells may have been abandoned, destroyed, or inactivated for reasons other than nitrate contamination.

Adminsitrative Record
Page 35410

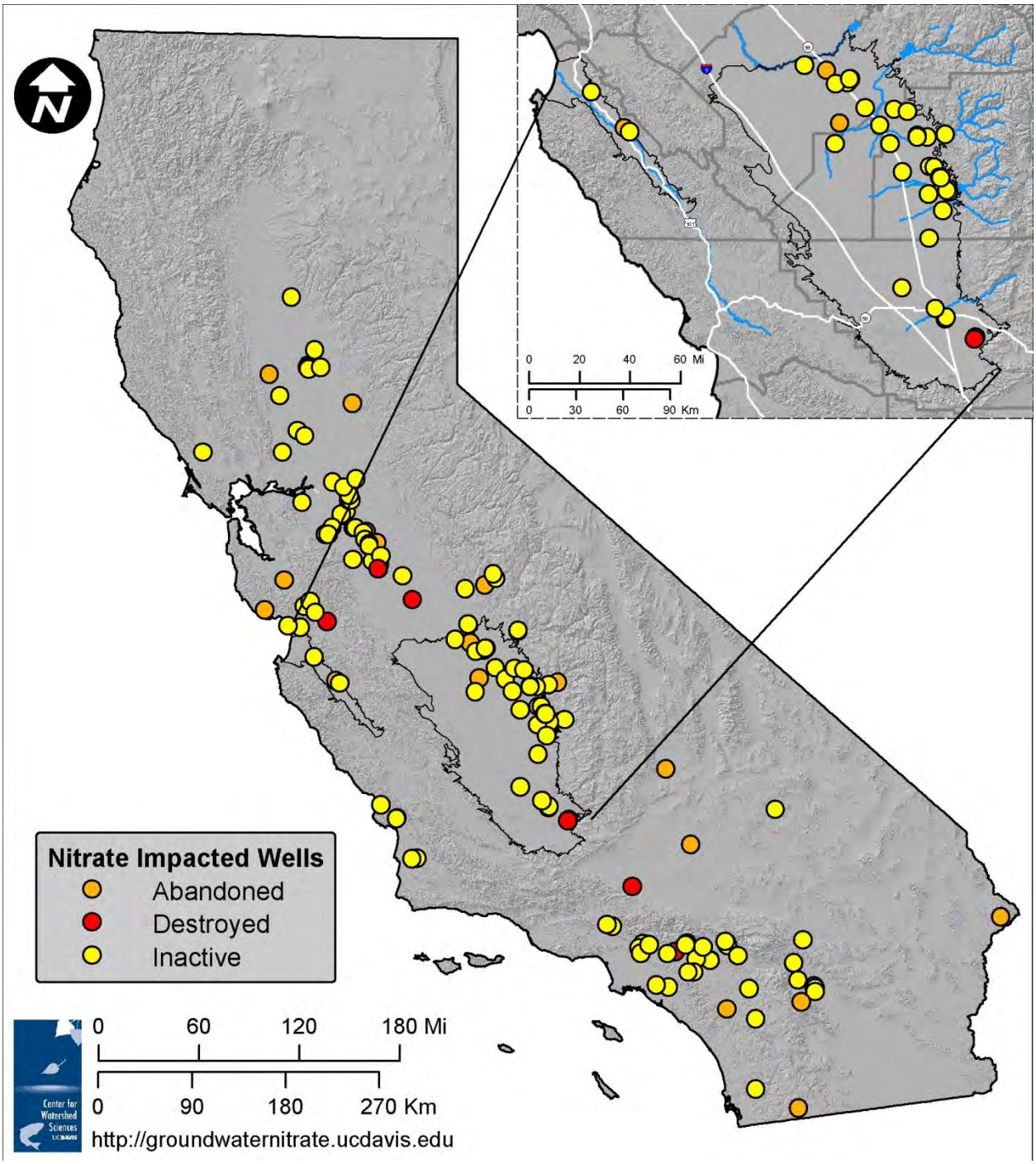


Figure 26. Location of nitrate impacted abandoned, destroyed, and inactivated wells. (Source: CDPH PICME and WQM Databases.)

Adminsitrative Record
Page 35411

5.2 Survey of Blending and Treating Systems

Complementing the detailed cases studies for each of the treatment types (above), a survey was conducted to assess full-scale application of nitrate treatment in California and specifically in the Tulare Lake Basin and Salinas Valley. The survey (Figure 27) was designed to gather detailed information on treatment type, water quality parameters affecting treatment, details of the treatment system, and cost information. Systems were contacted via phone and email and the survey packet was emailed whenever possible for ease of response. The survey packet included a letter of introduction, a brief project description, and the digital survey. Whenever possible, systems were contacted via phone and email for clarification of submitted responses and to gather additional information.

The list of water systems treating and/or blending to address nitrate contamination was developed with assistance from CDPH and analysis of the CDPH PICME and WQM databases. CDPH compiled a list of systems across CA treating and/or blending for nitrate after completing an internal review to ensure the provision of the most comprehensive list possible. Analysis of systems listed in the PICME and WQM databases confirmed treating systems based on nitrate levels and descriptions of individual systems and sources. County regulated systems treating and/or blending for nitrate were subsequently determined through contact with the individual county health departments and added to the list. The survey focus was treating systems, but several blending systems were also included.

Adminsitrative Record
Page 35412

SBX 2 1: Survey of CA Potable Water Systems Treating for Nitrate

Your participation in this survey is greatly appreciated and is essential to our investigation.
The information provided will be highly beneficial in the assessment of nitrate treatment.
Confidentiality Note: Surveyed systems will not be identifiable in any published reports.

Contact Information

Contact Name and Title:	Phone:	Email:
System Engineer or Installer:	Phone:	Email:

General System Information

System No. (PWSID):	District:	County:
Name of System:	# of Connections:	Population Served:
Address, City, State, Zip:		

Nitrate Impacted Sources: Please provide the following information for each nitrate impacted source.

Source Name (Well#)	Source Type (Groundwater, Surface Water, GWUDI)	Source Capacity (gpm)	Status (Active, Inactive)	Method to Address Nitrate Contamination (Abandonment, Blending, Ion Exchange, Reverse Osmosis, etc.)	Raw Water Nitrate Level (mg/L as nitrate)		
					Avg.	Min.	Max.

Please attach information for additional sources (if > 4).

Nitrate Treatment System *(Please attach schematic or sketch if possible)*

Describe Treatment *(manufacturer, design capacity, avg. flow rate, installation year, system footprint, etc.)*:

Describe Pretreatment:

Describe Post-Treatment:

Describe Waste Management:

Describe the influence of water quality parameters on treatment design:

Capital Costs for Nitrate Treatment

Total Estimated Capital Costs (*please describe the component costs included in the total*):

O & M Costs for Nitrate Treatment

Total Estimated Annual O & M Costs (*please describe the component costs included in the total*):

Additional Comments/Information

Figure 27. Digital survey distributed to drinking water systems treating and/or blending to address high nitrate levels.

Of the 42 systems identified as treating for nitrate throughout CA, 26 systems completed the survey. Statewide systems are mapped in Figure 28 and systems in the study area of interest are mapped in Figure 29. Whenever possible, systems are blending to address the nitrate problem, accounting for ~56% of the statewide systems in Figure 28.

Adminsitrative Record
Page 35413

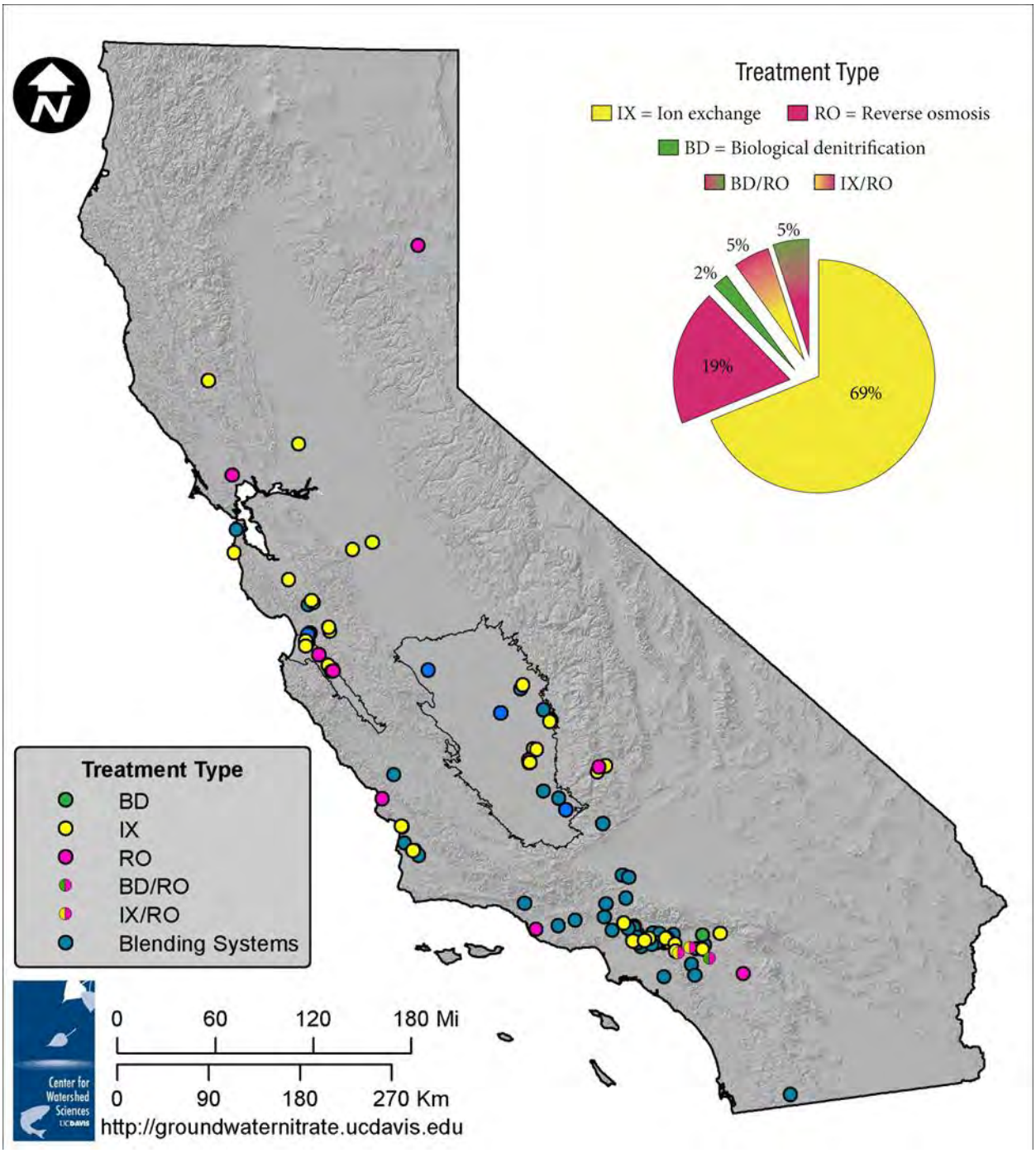


Figure 28. California drinking water systems treating and/or blending for nitrate. (Source: CDPH Internal Review of facilities and contact with facility and county representatives.)

Approximately 70% of treating systems across CA are using IX and ~20% are using RO. Several locations have implemented both RO and IX, primarily to address salinity as well as nitrate. Biological treatment is being implemented at two locations in CA. After successful completion of a one-year demonstration study, a system in Rialto, CA, is installing a biological treatment system primarily to address perchlorate contamination of drinking water. The water system also has a history of nitrate contamination and the

Adminsitrative Record
Page 35414

biological treatment system provides the potential to treat their high nitrate source(s) as well. Construction of the full-scale biological treatment system is underway. In Riverside, CA, biological treatment has been investigated for the treatment of the RO bypass stream to increase total plant capacity. See Section 3.4.5 Biological Denitrification - Case Studies, for detailed case studies of these unique systems.

Focusing on the Tulare Lake Basin and the Salinas Valley (Figure 29), 23 systems were found to be treating and/or blending to address the nitrate problem (10 blending systems, 10 IX systems, and 3 RO systems).

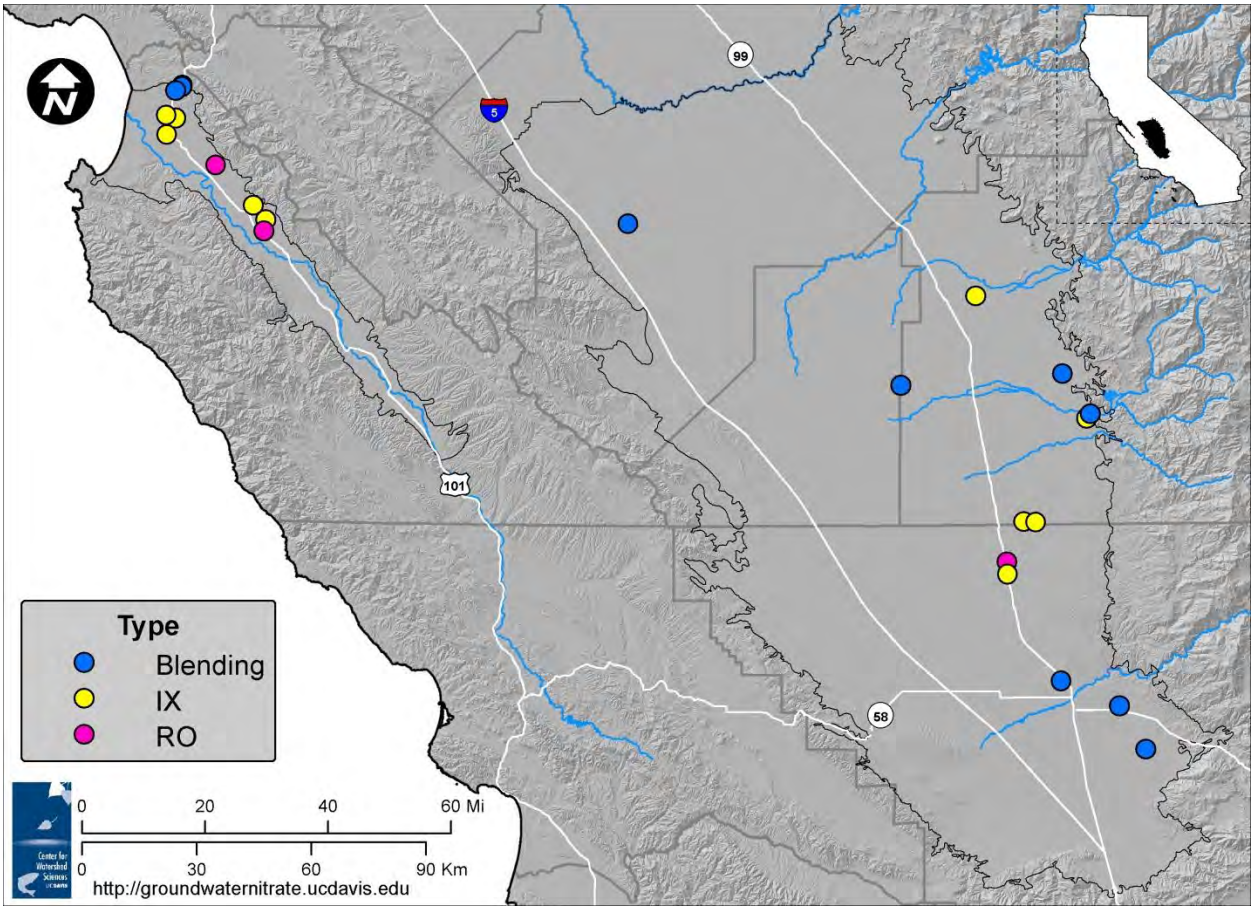


Figure 29. Drinking water systems treating or blending for nitrate in the Tulare Lake Basin and Salinas Valley. (Source: CDPH Internal Review of facilities and contact with facility and county representatives.)

Table 21 lists the population ranges and nitrate levels of blending, IX, and RO systems in the study area. The IX systems cover the widest population range; however, it is important to note that some large systems using IX for nitrate treatment also use blending. For each system the minimum, maximum, and average nitrate concentration across all active wells were determined, then the average of each of those categories across all systems for each of the treatment options was calculated to illustrate the typical maximum, minimum, and average nitrate levels for each treatment type. Of these systems, the average maximum nitrate level of blending systems is slightly lower than that of treating systems; with lower

Adminsitrative Record
Page 35415

nitrate levels and access to a low-nitrate source, the possibility of blending can avoid the need for more costly treatment. Sixty percent of the IX and RO systems in Figure 29 have only a single active well and 40% of the blending systems have only two active wells. Water quality changes and increasing nitrate levels could be particularly problematic for these one- and two-well systems. Even where blending is successfully meeting today's needs, it may be precarious to assume or expect that water systems can rely solely on blending for compliance into the future.

Table 21. Population and nitrate levels of systems in the study area treating and/or blending for nitrate.

		Average Raw Nitrate (mg/L as nitrate)		
	Population Range (Total)	**Max.**	**Min.**	**Avg.**
Ion Exchange	25 – 133,750 (261,200)	71	15	40
Reverse Osmosis	45 – 6,585 (6,760)	75	24	41
Blending	45 – 25,500 (83,475)	64	3	32

An example of a blending system in Tulare County (Figure 30) illustrates the complexity of even a simple blending system. This system has seven wells, two of which are high in nitrate (wells 8 and 11 in red). Most of the year the high nitrate wells are inactive, but with high demand in summer, the system blends a high nitrate source with other wells. Table 22 lists nitrate levels, depth, and capacity of the system's source wells. It is interesting to note that the high nitrate wells have the highest capacity and are actually some of the deeper wells. Increasing nitrate levels in the low nitrate wells would be cause for concern as the system's blending program would be affected. This is one simple example of hundreds of scenarios. Even with this simple blending system, there are several complicating factors including differences in capacity, seasonal variation, and the variability of nitrate levels in wells that are very close together. Extrapolating this concept over the entire study area, the case by case nature of addressing the nitrate problem becomes more apparent.

Adminsitrative Record
Page 35416

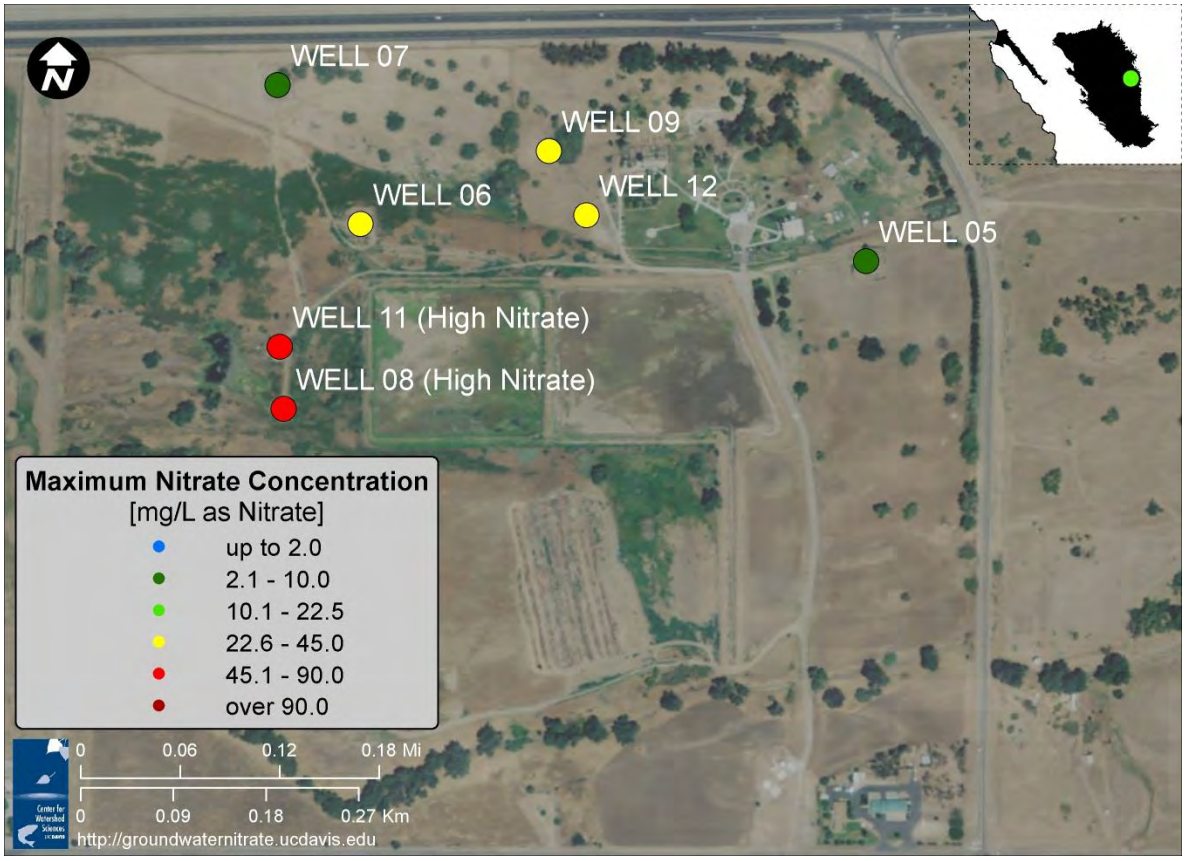


Figure 30. Wells of a blending system in Tulare County. (Source: Contact with facility/survey.)

Table 22. Nitrate level, well depth and well capacity for a Tulare County blending system.

Well #	Max Nitrate (mg/L as nitrate)	Total Depth (ft)	Depth to Top of Screen (ft)	Capacity (gpm)
05	10	255	163	153
06	12	328	30	174
07	7.8	296	94	161
08	**78**	**393**	**250**	**378**
09	11	398	160	150
11	**81**	**400**	**340**	**318**
12	11.8	470	180	170

Case studies of nitrate treatment systems are included above in the various treatment technology sections.

Adminsitrative Record
Page 35417

6 Treatment Cost Analysis

In the estimation of treatment costs, there are two major categories to consider: capital costs and O&M costs. Capital costs refer to the upfront investment required for the design, implementation, and installation of the treatment system. O&M costs refer to the annual costs for operating and maintaining the system. Based on U.S. EPA cost estimating procedures, developed through the Technology Design Panel, capital costs can be further categorized as follows (U.S. EPA 2000; U.S. EPA 2005):

Process Costs
- Manufactured Equipment
- Concrete and Steel
- Electrical & Instrumentation
- Pipes & Valves

Construction Costs
- Sitework & Excavation
- Subsurface Considerations
- Standby Power
- Contingencies
- Interest During Construction

Engineering Costs
- Contractor Overhead and Profit
- Engineering Fees
- Legal, Fiscal, and Administrative Fees

Indirect Costs
- Housing
- Permitting
- Land
- Training
- Piloting
- Public Education

The cost analysis detailed below was performed in accordance with U.S. EPA cost estimation procedures (U.S. EPA 2000). Total capital costs were converted to annualized capital costs ($/kgal) based on Eqn. 21.

Annualized Capital Cost = [Capital Cost ($) * Amortization Factor]

/ [Flow (MGD) * 1000 kgal/mgal * 365 days/year] (Eqn. 21)

An amortization value of 0.0802 was used which corresponds with an interest rate of 5% over 20 years (Eqn. 22).

$$\text{Amortization Factor} = (1+i)^N/((1+i)^N - 1)/i) \quad \text{(Eqn. 22)}$$

Where i = interest rate and N = number of years

Annual O&M costs were calculated based on Eqn. 23 to convert total annual O&M costs to $/kgal.

O&M Cost ($/kgal) = [O&M Cost ($)]

/ [Flow (MGD) * 1000 kgal/mgal * 365 days/year] (Eqn. 23)

Annualized Capital Cost ($/kgal) and O&M Cost ($/kgal) were summed to determine Total Annualized Cost ($/kgal).

Adminsitrative Record
Page 35418

In the U.S. EPA 1979 Cost Estimating Manual (U.S. EPA 1979), cost information is included for anion exchange for nitrate removal; however, for a more recent set of cost indices, cost information reported for anion exchange and membrane processes for arsenic treatment will be used here for comparison with collected cost information (U.S. EPA 2000). Similarly, in a recent AWWA publication examining the national impact of changing the nitrate MCL, cost curves of anion exchange for arsenic removal were used in the estimation of nitrate treatment costs. "USEPA's cost curves were chosen because they are generally used for developing national compliance costs and because arsenic and nitrate use comparable regenerable IX treatment" (Weir & Roberson 2011, p. 49). Based on the U.S. EPA cost curves of IX for arsenic removal, costs by system size using regenerable IX are listed in Table 23, ranging from $0.22/kgal for a 10 MGD system to $4.60/kgal for a 0.01 MGD system. Disposal costs were not included in the U.S. EPA cost estimates of IX for arsenic removal.

Table 23. Cost estimation using U.S. EPA cost curves of IX for arsenic removal.[1]

System Capacity	Annualized Capital Cost ($/kgal)	O&M Cost ($/kgal)	Total Annualized Cost ($/kgal)
10	0.09	0.13	0.22
2	0.13	0.23	0.36
1	0.26	0.46	0.72
0.1	0.21	0.70	0.91
0.01	0.79	3.81	4.60

[1] U.S. EPA 2000.

For the cost analysis detailed herein, treatment cost information was collected from literature, vendors, surveys, and contact with existing systems. Factors affecting system cost include facility size (flow rate), source water quality (including nitrate concentration), environmental factors (temperature), and target effluent nitrate concentration. Disposal of waste brine can be a significant portion of O&M costs for the removal technologies (see Section 6.4 Disposal Costs for a more detailed discussion of disposal costs). Capital costs for treatment can include land, housing, piping, storage tanks, O&M equipment, process equipment (i.e., vessels, resin, membranes, media, etc.), preliminary testing (pilot studies), permits, and training. O&M costs for treatment can include resin, media, or membrane replacement (due to loss or degradation) and disposal; waste residuals disposal or treatment (e.g., brine disposal); chemical use (salt, anti-scalant, pH adjustment); repair and maintenance; power; and labor. Costs can be difficult to assess due to inconsistencies in how cost information is reported. Comparison of costs across different systems is not always valid due to differences in influent water quality parameters, system size, waste management options, and system configuration. Published costs do not always include comparable information. The cost information listed in this section is provided as an approximate range of costs. Costs for implementing treatment may be very different from those listed here. A thorough cost analysis of design parameters for specific locations would be required for accurate cost estimation. The information gathered through the survey includes reported costs associated with treating systems in CA.

Assumptions and sources of uncertainty in this analysis of treatment costs include the following:

- For certain sources of cost information it is unclear if all components are included in capital costs (e.g., preliminary planning, pilot testing, installation, administration fees, engineering fees,

Adminsitrative Record
Page 35419

building cost, storage, etc.) and O&M costs (e.g., pumping, disposal, labor, energy, chemicals, etc.). Whenever possible efforts have been made to ensure inclusion of all relevant costs.

- Many treatment systems blend treated water with a bypass stream; whenever possible costs were calculated on the basis of total produced water to accommodate the blending configuration. For example, an RO system may remove nitrate to very low levels, then blend the permeate with a bypass stream, raising nitrate levels to the distribution goal (and restoring other ions).
- Capital costs generally increase with design capacity, but some systems "over design" with a design capacity significantly greater than the actual flow. The calculation of annualized capital costs are based on average flow rather than design capacity whenever possible to provide capital costs per unit of produced water.
- O&M costs are based on actual average flow, rather than design capacity, whenever possible.
- Costs were adjusted to 2010 dollars.
- Costs of drilling a new well were excluded from the treatment cost analysis to ensure appropriate comparison.
- Costs of systems in the design phase are anticipated costs.
- Several sample costs of electrodialysis and biological treatment systems designed for the removal of other constituents were included; based on communication with water treatment engineers and vendors, the costs for treatment for nitrate removal should be similar.
- Costs were collected for systems with wide-ranging characteristics including variation in system size, nitrate levels, co-contaminants, and other water quality constituents.
- Several systems reported renting treatment equipment and/or contracting O&M services, resulting in very different capital and O&M costs.
- Given only equipment costs (e.g., from vendors), total capital costs were modeled based on U.S. EPA scaling factors (U.S. EPA 2000).

6.1 Costs by Treatment Type

Comparison of the average total annualized cost for IX, RO, EDR, and BD across all system sizes highlights RO as the most expensive option (Figure 31). EDR costs are for a limited number of systems including costs for the treatment of constituents other than nitrate and may not be representative of actual EDR costs for nitrate removal. Based on preliminary estimates, biological treatment has the potential to be cost competitive. Costs for IX, RO, and BD are broken down into three system size categories to illustrate the variability in cost with system size (Figure 32, Figure 33, and Figure 34, respectively). Note the much higher cost for system sizes less than 0.5 MGD for IX (Figure 32) and RO (Figure 33). Preliminary estimates of BD treatment costs do not illustrate the same degree of variability with system size (Figure 34); however, BD for nitrate removal from drinking water is an emerging

Adminsitrative Record
Page 35420

technology and available cost information is limited. The very high O&M costs for small systems (Figure 32 for IX and Figure 33 for RO) are representative of low flow systems making the cost per 1000 gallons quite high. This highlights a problem faced by many small systems lacking the benefits of economies of scale; funding may be available for the initial upfront investment, but with high O&M costs long term treatment can become unsustainable. Additionally, due to insufficient funds for ongoing costs, small water systems can be faced with an inability to retain qualified operators which can lead to MCL violations and insufficient maintenance of the treatment system. (ED/EDR costs are excluded from this comparison due to insufficient cost information.)

The variability in cost information reported here is due to many factors, including variability in water quality parameters, site considerations, and the sources of uncertainty in the cost information, as discussed above. For example, one very small IX system reported the disposal of waste brine to septic, a low cost option, resulting in significantly lower O&M costs in comparison with other systems.

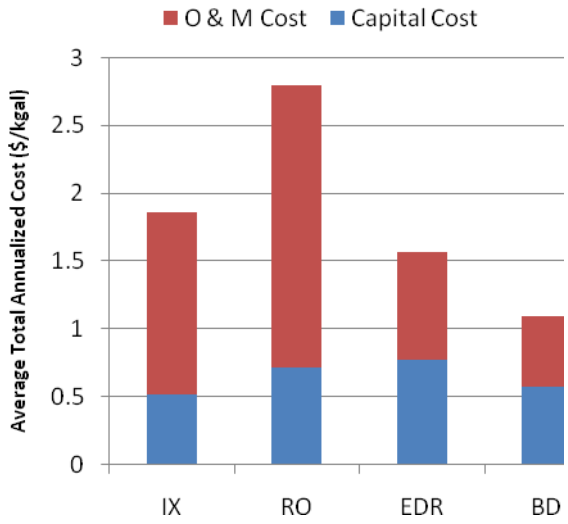


Figure 31. Average cost comparison of nitrate treatment technologies.

Adminsitrative Record
Page 35421

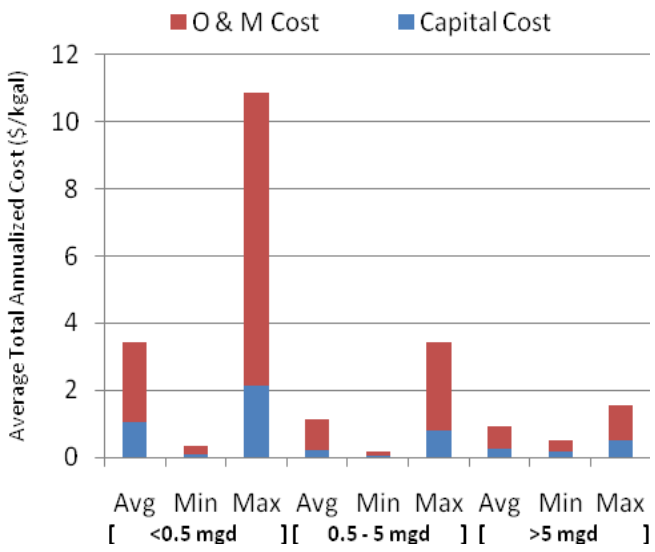


Figure 32. Costs of anion exchange for nitrate treatment.

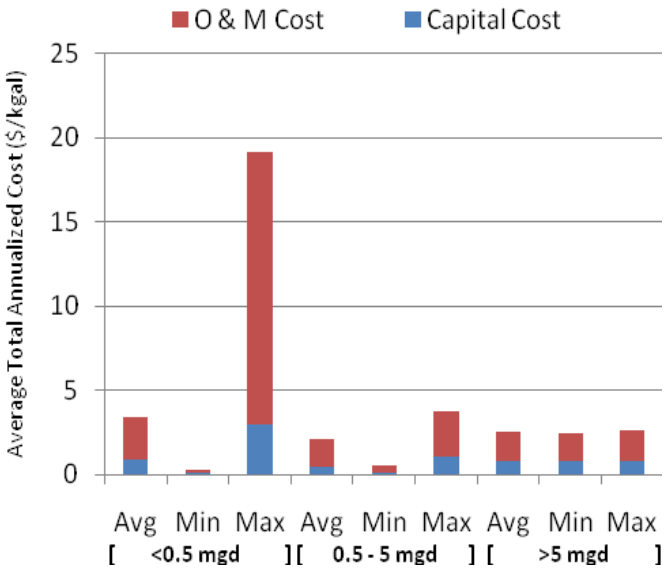


Figure 33. Costs of reverse osmosis for nitrate treatment.

Adminsitrative Record
Page 35422

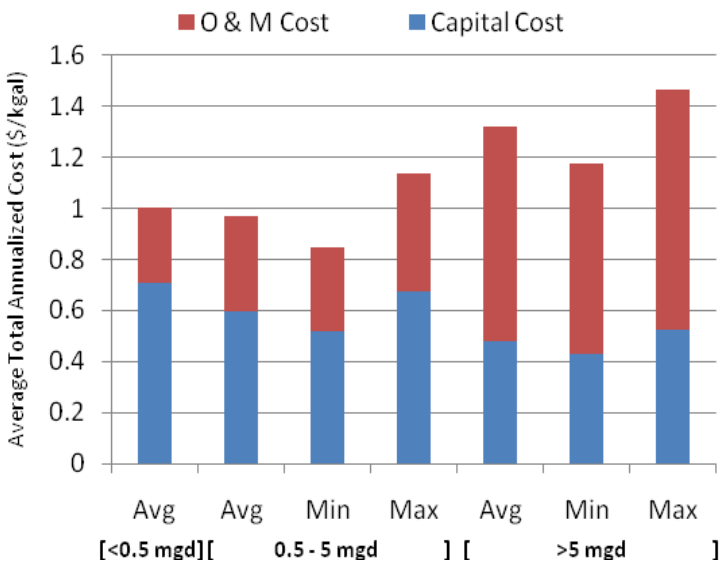


Figure 34. Costs of biological denitrification in drinking water treatment.

6.2 Costs by System Size

As indicated above, system size is a major factor in determining nitrate treatment costs. Larger treatment systems will have higher total capital and O&M costs; however, the cost per unit of produced water generally decreases as system size increases. Large treatment systems have the advantage of economies of scale. Based on cost information collected from vendors, literature, surveys, and treatment systems, treatment costs relative to system size are illustrated in the below cost curves for IX and RO (Figure 35). The development of cost curves for the other technologies was not possible due to insufficient cost information. The higher relative cost of treatment for smaller systems can be seen moving toward the vertical axis, with decreasing system size and increasing cost as the curve sweeps upward. Again, the total cost for RO treatment is higher than that of IX.

Adminsitrative Record
Page 35423

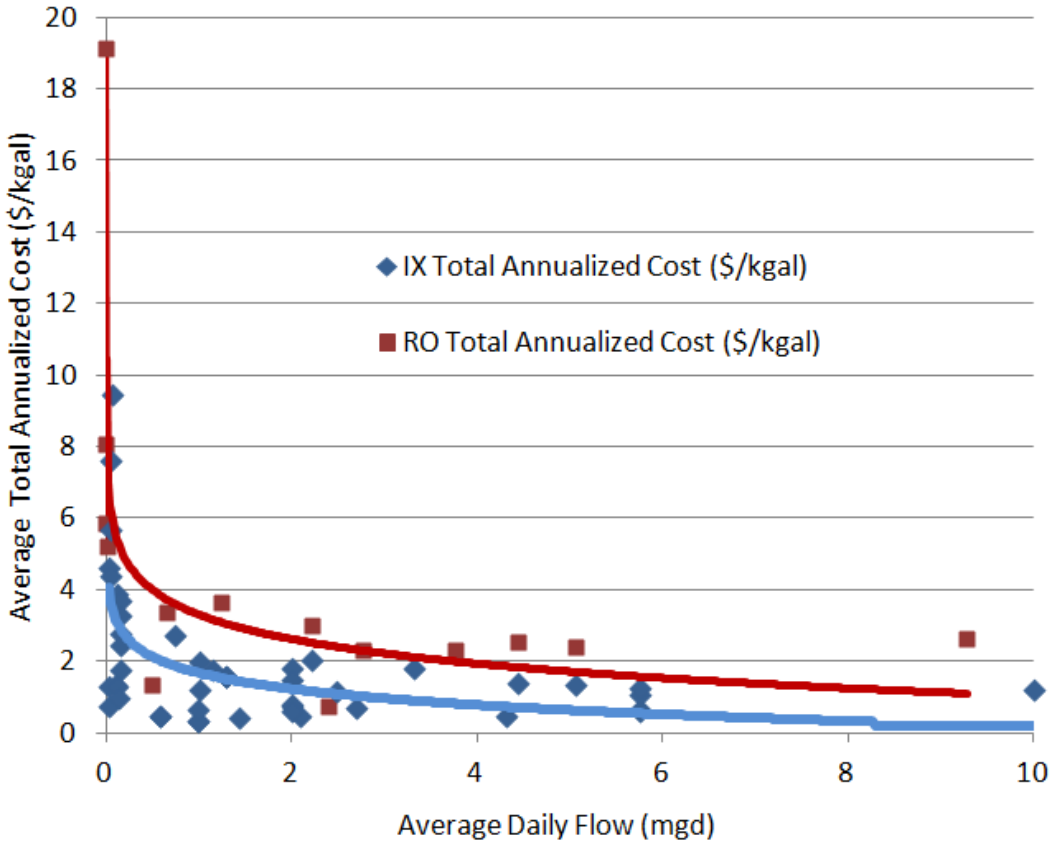


Figure 35. Cost curve of IX (blue) and RO (red) for nitrate removal.

Table 24 includes all of the most reliable treatment cost information collected for comparison of cost ranges across system size categories for IX and RO.

Adminsitrative Record
Page 35424

Table 24. Summary of anion exchange and reverse osmosis cost information by system size.

System Size (people)	Design Flow Range (typical average flow range) MGD	Treatment Type	Annualized Costs in $/1000 gallons		
			Capital Cost Range (Avg.) $/1000 gallons	O&M Cost Range (Avg.) $/1000 gallons	Total Combined Cost Range (Avg.) $/1000 gallons
Very Small (25 – 500)	0.009 – 0.17 (0.002 – 0.052)	Ion Exchange	0.05 – 1.53 (0.75)	0.28 – 3.81 (1.22)	0.62 – 4.60 (1.97)
		Reverse Osmosis	0.47 – 4.40 (2.43)	0.22 – 16.16 (4.22)	0.69 – 19.16 (6.64)
Small (501 – 3,300)	0.17 – 1.09 (0.052 – 0.39)	Ion Exchange	0.08 – 0.25 (0.15)	0.15 – 2.63 (0.87)	0.34 – 2.73 (1.05)
		Reverse Osmosis[1]	0.19 – 1.13 (0.47)	0.23 – 1.15 (0.57)	0.58 – 1.34 (0.93)
Medium (3,301 – 10,000)	1.09 – 3.21 (0.39 – 1.3)	Ion Exchange	0.06 – 0.52 (0.19)	0.12 – 1.69 (0.84)	0.36 – 2.04 (1.06)
		Reverse Osmosis[1]	0.44 – 0.63 (0.53)	0.91 – 2.76 (1.89)	1.35 – 3.39 (2.59)
Large (10,001 – 100,000)	3.21 – 30.45 (1.3 – 15.51)	Ion Exchange	0.09 – 0.41 (0.26)	0.13 – 1.39 (0.66)	0.22 – 1.81 (0.97)
		Reverse Osmosis	0.33 – 1.46 (0.97)	0.40 – 2.21 (1.48)	0.73 – 3.67 (2.38)

[1] Limited data set for the indicated system size and treatment type.

Adminsitrative Record
Page 35425

6.3 Costs by Water Quality Parameters

As highlighted above in the discussion of water quality (Section 4 Tulare Lake Basin and Salinas Valley - Water Quality Analysis), if treatment of multiple contaminants is necessary, treatment costs will generally increase. Similarly, the level of nitrate and water quality parameters that can interfere with treatment can increase O&M costs. Table 25 lists costs by system size with increasing nitrate levels and is intended as an example of nitrate treatment cost estimation based on nitrate concentration in source water. Table 25 is strictly an example and is not intended to be definitive, but only to suggest how treatment costs might change with increasing nitrate levels. The actual costs with increasing nitrate level are wide ranging and vary with numerous factors. The percent increase in O&M costs was modeled based on only two sets of vendor data in which estimates were provided based on given nitrate levels (low and high). Available data were specifically applicable to estimation of O&M increases as the nitrate concentration increases from ~1X the nitrate MCL to 2X the MCL. To extrapolate the exercise further, the same percent increase was used to predict the O&M increase from 2X the MCL to 3X the MCL. It is not possible to accurately estimate or generalize how these costs would translate for other IX systems as the two vendors provided cost estimates specifically for a system using a selective resin and a second unique system designed for low brine. Based on the information herein, O&M costs would be expected to increase even more using conventional IX under the given scenario of increasing nitrate levels. It is important to note that the treatment system could also be designed differently for higher nitrate levels (more or larger vessels, in series/in parallel, different bypass ratios, etc.); this is not included in the table as it would be pure speculation.

Adminsitrative Record
Page 35426

Table 25. An exercise in the estimation of treatment costs based on appropriate technology for various nitrate levels.[1]

System Size (people)	Raw Nitrate Level	Treatment Type	O&M Cost Range (Avg.)[2]	Annualized Combined Cost Range (Avg.)
			$/1000 gallons	$/1000 gallons
Very Small (25 – 500)	1X MCL	Ion Exchange	0.28 – 3.81 (1.22)	0.62 – 4.60 (1.97)
	2X MCL	Ion Exchange	0.35 – 10.48 (2.13)	0.69 – 11.27 (2.88)
	3X MCL	Ion Exchange	0.42 – 17.15 (3.05)	0.76 – 17.94 (3.80)
	3X MCL	Reverse Osmosis	0.22 – 16.16 (4.22)	0.69 – 19.16 (6.64)
Small (501 – 3,300)	1X MCL	Ion Exchange	0.15 – 2.63 (0.87)	0.34 – 2.73 (1.05)
	2X MCL	Ion Exchange	0.19 – 7.23 (1.52)	0.38 – 7.33 (1.70)
	3X MCL	Ion Exchange	0.23 – 11.84 (2.18)	0.42 – 11.94 (2.36)
	3X MCL	Reverse Osmosis[3]	0.23 – 1.15 (0.57)	0.58 – 1.34 (0.93)
Medium (3,301 – 10,000)	1X MCL	Ion Exchange	0.12 – 1.69 (0.84)	0.36 – 2.04 (1.06)
	2X MCL	Ion Exchange	0.15 – 4.65 (1.47)	0.39 – 5.00 (1.60)
	3X MCL	Ion Exchange	0.18 – 7.61 (2.10)	0.42 – 7.96 (2.32)
	3X MCL	Reverse Osmosis[3]	0.91 – 2.76 (1.89)	1.35 – 3.39 (2.59)
Large (10,001 – 100,000)	1X MCL	Ion Exchange	0.13 – 1.39 (0.66)	0.22 – 1.81 (0.97)
	2X MCL	Ion Exchange	0.16 – 3.82 (1.16)	0.25 – 4.24 (1.46)
	3X MCL	Ion Exchange	0.20 – 6.26 (1.65)	0.29 – 6.68 (1.96)
	3X MCL	Reverse Osmosis	0.40 – 2.21 (1.48)	0.73 – 3.67 (2.38)

[1] This table is strictly an example and is not intended to be definitive, but only to suggest how treatment costs might change with increasing nitrate levels. The estimated increase in O&M costs is wide ranging, 25% – 175%, and depends on many factors including water quality parameters, disposal options, resin capacity, resin type, and ion exchange system design. As nitrate levels increase, salt, disposal, and resin costs for IX will increase (O&M). Reverse osmosis costs will increase with increasing TDS, but not at the same rate, this cannot currently be estimated. Depending on other water quality parameters, the costs of IX are predicted to surpass those of RO. In the future, biological denitrification will likely be considered as an option for > 2X the nitrate MCL. Additionally, increasing the number and/or size of resin vessels to address higher nitrate levels would increase capital costs. O&M costs would still increase; in practice the system would be designed to optimize costs. O&M increases were considered here as an example. Actual costs with increasing nitrate levels for specific systems may vary significantly from listed costs and should be assessed by professional engineers.

[2] Increases in O&M are estimated from a limited dataset comprised of vendor cost estimates for IX costs with nitrate levels increasing from just above the MCL to slightly more than double the MCL. All available cost information was included in the 1X MCL scenario as a starting point, including systems with nitrate levels above 1X the MCL.

[3] Limited dataset for the indicated system size and treatment type.

Adminsitrative Record
Page 35427

6.4 Disposal Costs

Brine disposal costs for drinking water systems in CA using IX for nitrate treatment vary with several factors including proximity to a coastal brine line, waste brine volume (e.g., water efficiency), and the water quality characteristics of waste brine (e.g., salinity). The presence of contaminants other than nitrate (e.g., arsenic, selenium, uranium, chromium, and vanadium) in the waste stream can have a significant impact on brine disposal options and costs; disposal to a hazardous waste facility may be required at a greater cost. Methods for disposal of waste brine or concentrate reported in the survey of nitrate treatment systems in CA include discharge to a septic tank and leach fields, to a wastewater treatment plant via a sewer connection, to irrigation ponds (for RO concentrate), to a brine line, and to a wastewater treatment plant via trucking.

Disposal options are limited in the Central Valley of California due, in part, to the great distance to the coast. Trucking of waste brine to coastal wastewater facilities, although costly, is sometimes the chosen disposal option. Typical costs for trucking and disposal of spent IX brine at a coastal wastewater plant from the Central Valley can be around $0.15/gallon ($150/1000 gallons of waste brine). East Bay Municipal Utility District (EBMUD), in Oakland, CA, operates a wastewater management program for the disposal of high salinity and high nitrate wastewater.

O&M costs for the disposal of waste brine reported in the survey of nitrate treatment systems in CA range from $0.015 to $ 0.05/1000 gallons of treated water. Assuming a high efficiency of 99.5%, O&M disposal costs range from $3 to $11/1000 gallons of waste brine. This is consistent with the results of a recent research investigation comparing the life cycle costs of several nitrate treatment options. Meyer et al. (2010) discuss the costs of multiple brine disposal options including evaporation ponds, deep well injection, and sewer. Based on vendor estimates, results indicate total brine disposal costs (including capital and O&M costs) ranging from approximately $7 to $27/1000 gallons of waste brine disposal to evaporation ponds and approximately $6 to $11/1000 gallons of waste brine disposal to sewer (Table 26). (In the conversion of costs reported by Meyer et al. to the cost per 1000 gallons, an amortization value of 0.0802 was used which corresponds with an interest rate of 5% over 20 years.) However, it is important to note that the study by Meyer et al. (2010) was focused on the evaluation of nitrate treatment in Arizona; location specific characteristics are an important factor affecting disposal costs.

Adminsitrative Record
Page 35428

Table 26. Brine disposal costs.[1]

	Annualized Capital Cost	O&M Cost	Total Annualized Cost	Total Range
Average Cost by Waste Volume ($/1000 gallons)				
Evaporation Ponds	10.23	5.62	15.85	7 to 27
Solar Ponds	20.48	18.80	39.27	8 to 88
Well Injection	12.00	18.52	30.52	13 to 111
Sewer	2.40	5.51	7.91	6 to 11
Average Cost by Treated Volume ($/1000 gallons)				
Evaporation Ponds	0.046	0.015	0.061	0.03 to 0.14
Solar Ponds	0.063	0.047	0.110	0.07 to 0.20
Well Injection	0.051	0.077	0.128	0.03 to 0.33
Sewer	0.007	0.034	0.041	0.02 to 0.12

[1] Based on Meyer et al. 2010.

Costs of resin disposal can also vary with water quality parameters other than nitrate; IX resin removes not only nitrate, but other contaminants (e.g., arsenic) which can affect disposal options when resin needs to be replaced. High levels of other contaminants on the resin can require disposal at hazardous waste facilities and increase disposal costs, although the impact of co-contaminants is more significant on brine disposal costs than on resin disposal costs. Non-hazardous resin can be land-filled. Using regenerable resin, requiring replacement only once every 3 – 8 years (WA DOH 2005 and Dow 2010c), the cost of land-fill disposal of non-hazardous resin is expected to be minimal compared with the disposal of other waste residuals (waste brine/concentrate). Service contracts are available with various companies to manage resin replacement and disposal.

In the selection of the most appropriate nitrate treatment option, disposal costs are a significant factor; consideration of the pros and cons for the unique conditions of an individual water system is not always straightforward and can be heavily weighted by disposal options. Although other removal technologies (RO and ED) require concentrate disposal, because IX requires the addition of salt for resin regeneration, the waste stream consists of not only the nitrate and other ions that have been removed from the water, but also the spent brine solution used in regeneration. As nitrate levels in source water increase, IX resin will need to be regenerated more frequently, increasing salt use and brine waste volume. In contrast, although the recovery rate for RO is significantly lower than that of IX (~80% and >95%, respectively), the nitrate level that can be addressed with RO is theoretically much higher (in accordance with the membrane nitrate rejection rate). Recall that RO is used for desalination as well. As the nitrate concentration in the treatment stream increases, with appropriate pressure, the RO membrane will continue to reject nitrate, assuming membrane scaling and fouling are properly controlled. In the comparison of IX and RO as nitrate levels increase, theoretically there is a tradeoff between the O&M costs for each technology. With increasing nitrate levels, chemical use and waste volume will increase for IX while power use and membrane maintenance may increase for RO. Excluding all other water quality parameters, the nitrate level at which the cost of IX exceeds the cost of RO requires more research.

Adminsitrative Record
Page 35429

Lastly, several small water systems included in the nitrate treatment survey indicated disposal of waste concentrate to a septic system. This highlights an important tradeoff; while small water systems do not have the advantages of economies of scale, with a low volume waste stream (depending on chemical composition to avoid negatively impacting septic system function or underlying groundwater), disposal to septic can avoid other, more costly disposal options.

Adminsitrative Record
Page 35430

7 Guidance for Addressing Nitrate Impacted Drinking Water

7.1 Checklist for the Selection of Mitigation Strategy

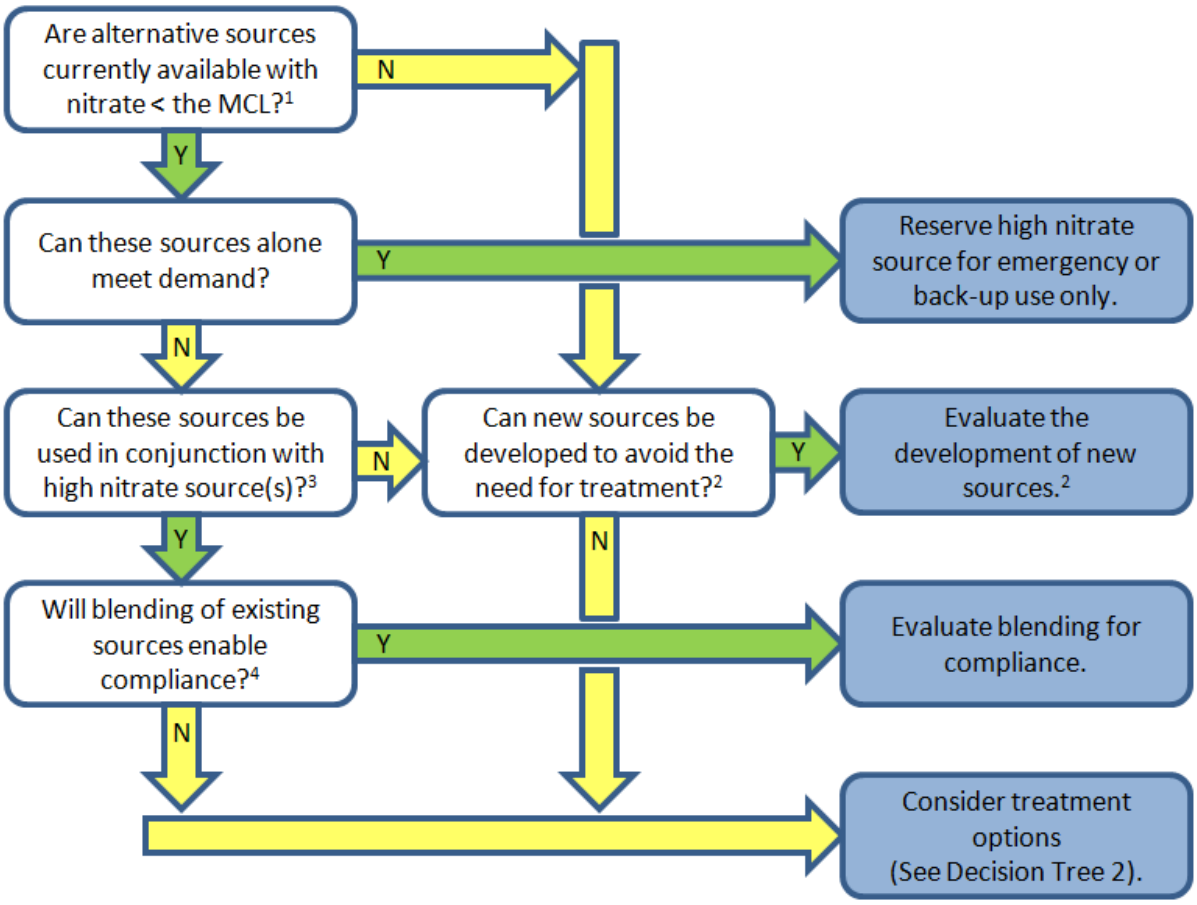
The following checklist is intended as a general guide for the selection of promising mitigation strategies for nitrate impacted drinking water (adapted from U.S. EPA 2003b, WA DOH 2005a, and WA DOH 2005b). Questions regarding mitigation strategy development should be directed to the Department of Public Health.[21]

1. Monitor nitrate concentration of distributed water and determine compliance status. Quarterly monitoring may be necessary and public notification requirements must be met.
2. Develop long-term compliance schedule with the Department of Public Health.
3. Determine all pertinent water quality characteristics (nitrate, arsenic and other co-contaminants, pH, TDS, sulfate, etc.).
4. Assess non-treatment options (e.g., removing well from service, blending, consolidation, development of new sources, etc.). See Decision Tree 1 (Figure 36) and Technical Report 7 (Honeycutt et al. 2012).
5. If non-treatment options are not feasible, determine evaluation criteria for treatment (e.g., effluent nitrate goal, operator certification, water demand, and state and local requirements) and assess treatment options. Choose optimal approach to addressing nitrate impacted source(s). See Decision Trees 1 and 2 (Figure 36 and Figure 37).
6. Develop preliminary or planning-level cost estimates for capital and O&M costs.
7. Assess design considerations. See Table 4 and Table 6 for details on IX and RO, respectively. Considerations for ED/EDR and BD are listed in Table 8 and Table 11, respectively.
8. Pilot test the selected solution (engineering professional required).
9. Develop construction-level cost estimates for capital and O&M costs (engineering professional required).
10. Examine funding options and attain funding (e.g., Drinking water state revolving fund (DWSRF) loan). See Technical Report 8 (Canada et al. 2012).
11. Implement the selected solution. This may include the development of a pre-design report, design, obtaining appropriate permits, construction, inspections, and start-up tasks (engineering professional required).
12. Monitor the system to ensure safe operation and the consistent supply of compliant drinking water (engineering professional may be required).

[21] Additional information can be found at http://www.cdph.ca.gov/certlic/drinkingwater/Pages/default.aspx and http://www.rcac.org/assets/.online%20materials/CA-DrinkingWater_Jan-June%202012.pdf.

Adminsitrative Record
Page 35431

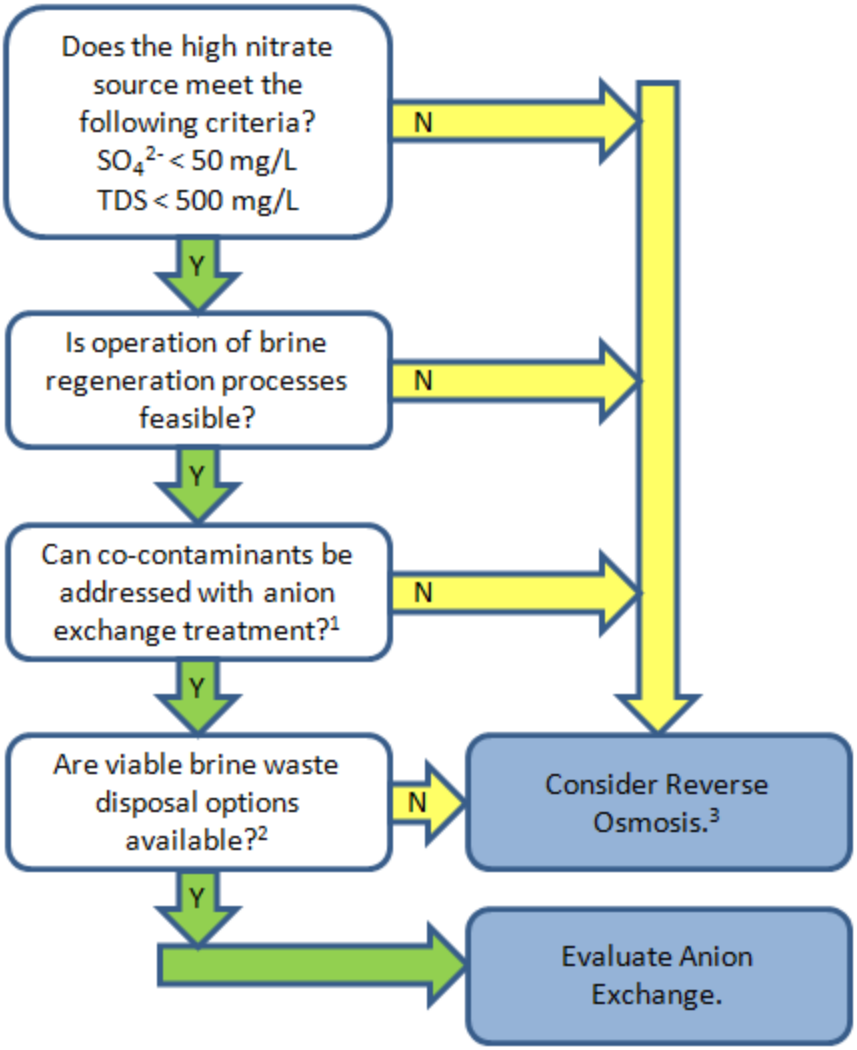
7.2 Decision Trees



1. Currently available sources could include additional existing wells or surface water.
2. Potential new sources could include new well development, surface water, consolidation with nearby system(s) or connection to an existing nearby system.
3. Are there any obstacles to using sources together? For example, is the alternative source a great distance from the high nitrate source?
4. Is there a feasible blend ratio to provide compliant water that would consistently meet demand taking into consideration potential co-contaminants and seasonal variation in demand and/or water quality?

Figure 36. Decision Tree 1 - Options to address nitrate impacted drinking water sources (adapted from U.S. EPA 2003b, WA DOH 2005a, and WA DOH 2005b).

Adminsitrative Record
Page 35432



1. In addition to nitrate, certain co-contaminants can be addressed simultaneously through anion exchange treatment (e.g., arsenic and perchlorate); however, specialized resin may be required for different constituents.
2. Viable options can include disposal to sewer, evaporation ponds, and a brine line.
3. Reverse osmosis can address certain co-contaminants that anion exchange cannot and generally results in a less concentrated waste stream. For systems serving < 500 people, POU reverse osmosis is another option to consider.

Figure 37. Decision Tree 2 - Anion exchange (adapted from USEPA 2003, WA DOH 2005a, and WA DOH 2005b).

Adminsitrative Record
Page 35433

8 Summary and Conclusions

- Current full-scale nitrate treatment installations in the United States consist predominantly of ion exchange (IX) and reverse osmosis (RO). Other technologies are available because, under some circumstances, the alternatives offer advantages that IX and RO cannot. While electrodialysis (EDR) is a feasible option for nitrate removal from potable water, the application of EDR is generally limited to high TDS and/or high silica waters. The use of biological denitrification (BD) to address nitrate contamination of drinking water is more common in Europe than in the U.S. However, this option is emerging in the U.S. and two full-scale systems are expected within a few years. Chemical denitrification (CD) may become a feasible nitrate treatment option in the future; however, the lack of current full-scale implementation suggests the need for further research, development and testing. New technologies will continue to be investigated and developed because no single option is ideal for all situations. A single treatment solution will not fit every community; however, the provision of safe drinking water for all communities can be achieved using currently existing technology.

- Brine reuse and treatment are vital to the continued reliance on IX for nitrate treatment of potable water. The low brine technologies offer a minimal waste approach and current research and development of brine treatment alternatives seem to be lighting the path toward future progress.

- In regions with declining water quality and insufficient water quantity, the need to address multiple contaminants will increase in the future, suggesting the future dominance of technologies capable of multiple contaminant removal. In this context, for any individual water source or system, the most appropriate technology will vary with the contaminants requiring mitigation. Although complex, analysis of the optimal treatment option for pairs and groups of contaminants will assist in the treatment design and selection. In such scenarios, the best treatment option for nitrate may not be the most viable overall.

- Currently and into the future, selection of the optimal and most cost-effective potable water treatment options will depend not only on the specific water quality of a given water source, but also on the priorities of a given water system. If land is limited, the typical configuration required for biological treatment may not be feasible. If disposal options of brine waste are costly or limited, implementation of denitrification treatment or development of brine recycling and treatment may be the most suitable option.

- When deciding on nitrate treatment, the characteristics of the water system must be taken into account as well. With consideration of economies of scale, many rural small water systems cannot afford to install treatment. Even with financial assistance to cover capital costs, the long term viability of a treatment system can be undermined by O&M costs that are simply not sustainable. For such systems, treatment can become more affordable through consolidation of multiple small water systems into larger combined water systems that can afford treatment as a

Adminsitrative Record
Page 35434

conglomerate. With a continued decline in water quality, non-treatment options alone, like blending or drilling a new well, may become insufficient measures for a water system to provide an adequate supply of safe and affordable potable water. Especially in rural small communities, perhaps the most promising approach will be a combination of consolidation and treatment. Alternatively, separate small treatment facilities can be consolidated under a single agency. For additional discussion on the comparison of alternative water supply options and associated costs see Technical Report 7 (Honeycutt et al. 2012).

- While current cost considerations are commonly the driving force in selecting nitrate treatment, it is essential to consider the long term implications of current industry decisions. For example, it may be cost-effective for a particular system to utilize conventional IX currently, but future water quality changes (e.g., increasing nitrate levels, co-contamination, high salt loading), discharge regulations, or disposal fees may lead to an unmanageable increase in costs. Environmental sustainability in drinking water treatment is being addressed with brine treatment alternatives and denitrification options. It is important to approach the future of drinking water treatment with the mindset that environmental sustainability and economic sustainability are tightly interwoven.

- Centralized treatment may not be feasible for widespread rural communities; another approach to consider is centralized management (e.g., design, purchasing, and maintenance) to minimize costs.

- Point-of-Use (POU) and Point-of-Entry (POE) treatment equipment is an important option to consider, especially for the provision of safe drinking water from private wells. Unless connecting to a nearby public water system becomes an option, users relying on domestic wells have two main alternatives: drilling a new well to attain safe drinking water or installing a POU or POE device for the treatment of contaminated water. The use of POU and POE treatment equipment by small public water systems is currently only a temporary option in California and reliance on these devices for the long-term would require regulatory changes. While POU and POE treatment equipment has been shown to effectively address nitrate and other contaminants, it is important to properly maintain these devices to ensure the supply of consistently safe drinking water.

Adminsitrative Record
Page 35435

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Page 35443

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Page 35450

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Adminsitrative Record
Page 35453

10 Appendix

10.1 Tables of Selected Research

Table A.1. Selected research on the use of ion exchange (IX) for nitrate removal.

IX – Nitrate selective resin	Dosing and sorption kinetics of Purolite A 520E (SBA resin). Influent nitrate concentration of 22.6 mg/L as N (100 mg NO_3^-/L). With no sulfate and chloride the treated bed volumes before breakthrough (BV) and resin capacity were 451 BV and 126.4 mg NO_3^-/g resin, respectively. With sulfate and chloride concentrations 10x that of nitrate, BV and resin capacity were 120 BV and 33.6 mg NO_3^-/g resin, respectively.	Samatya et al. (2006)
IX – Adsorption kinetics	Amberlite IRA 400 resin was found to be a suitable choice for nitrate removal with influent nitrate concentration of 0.23 – 4.07 mg/L as N (1 – 18 mg NO_3^-/L) and a 96% removal efficiency.	Chabani et al. (2006)
IX – Sulfate competition	Competition between sulfate and nitrate ions was explored. For the sulfate specific resin examined, findings indicate that sulfate selectivity increased with ionic strength.	Kim & Benjamin (2004)
IX – Co-contaminants	Compared two different anion exchange resins and studied the influence of other ions on nitrate removal.	Yoon et al. (2001)
IX – Resin residuals and DBPs	Quaternary amine groups and nitrosamines carcinogenic DBPs, 3 resins examined. With no disinfectants (chlorine and chloramine) "release 2 – 10 ng/L", max of 20 ng/L nitrosamine. "The lack of significant nitrosamine release in full-scale anion-exchange treatment system after multiple regeneration cycles indicates that releases may eventually subside." Precursors can be a problem with downstream chloramine use. Upstream disinfection releases of 20 – 100 ng/L (type 1) and 400 ng/L (type 2). Possible problem with IX in POU with influent containing chlorine/chloramines.	Kemper et al. (2009)
IX – Pilot study	3-year pilot examined RO, IX and biological denitrification in Mashhad, Iran. Treatment goal was a decrease in nitrate concentration from 26 mg/L as N (115 mg/L as nitrate) to 9 mg/L as N (40 mg/L as nitrate). Raw water with electrical conductivity of 1550 uS/cm, pH of 7.2. Optimal treatment options were deemed to be biological denitrification and RO. In the IX pilot plant pre-treatment consisted of cartridge filtration. Two IX columns containing ~200 L of nitrate selective resins were run in parallel with counter-current regeneration. System characteristics: Column diameter: 0.4 m, Bed Volume (BV): 200 L, Bed Depth: 1.6 m, Flow Rate: 1.0 to 3.0 m^3/h, Specific Flow Rate: 5 to 15 BV/h, Resin replacement every 5 years.	Panglisch et al. (2005) Dördelmann et al. (2006, and N.D.) Dördelmann (2009)
See Also	Clifford & Weber (1978), Guter (1982), Clifford (1987), Rash (1992), Clifford (2007), and Johnston & Heijnen (N.D.).	

Adminsitrative Record
Page 35454

Table A.2. Selected research on the use of reverse osmosis (RO) for nitrate removal.

RO – Pilot Study and Full-scale Installation	Brighton, CO: Use of RO in Brighton, CO, for potable water treatment including nitrate removal began operating in 1993 (11000 sq ft., 4 MGD, total cost of $8,253,000) after the performance of preliminary pilot study (4 membranes were tested). Source water nitrate concentration of 13 to 23 mg/L nitrate-N or 58 to 102 mg/L nitrate as NO_3^- (with TDS 800 – 1140 mg/L, hardness 370 – 480 mg/L as $CaCO_3$). Problems with biofouling of membrane and cartridge filters (slime forming *Pseudomonas*) minimized with anti-scalant use. Pretreatment: acid, anti-scalant and filtration. Post-treatment: CO_2 stripping (degasifier), disinfection, caustic and zinc orthophosphate (ZOP) addition. Blending ratio (untreated%:treated%) 20:80(winter) and 60:40 (summer). Waste discharge to surface water. Additional characteristics: Flux rate < 14.2 gal/sq. ft./day (helps control fouling), 2 stages: 32 : 16 vessels, pressure of 231 psi, each vessel with 6 8" diameter membranes. Cleaning required every 2 months.	Cevaal et al. (1995)
RO – HERO in Australia	Yalgoo, Australia: RO plant, online in 2007 using high-efficiency reverse osmosis (HERO). Water recovery rates as high as 95%. More than 85% waste reduction (a low as 10% the waste of conventional RO.) Removes both nitrate and silica from brackish water.	Water Corporation (2009)
RO – Full-scale uses methane	Inland Empire, CA: Cow Power in Inland Empire, CA: $80 mil. effort is powered by methane gas produced from the high population of cows in that region. Online since 2002, this plant provides 1/5th of regional demand.	Black (2003)
RO – Pilot study comparing IX, RO and BD	Mashhad, Iran: 3-year pilot examined RO, IX and BD in Mashhad, Iran. Treatment goal of 9 mg/L as N (40 mg/L as nitrate) (influent nitrate of 26 mg/L as N, 115 mg/L as nitrate). Results indicate BR and RO were best choices. Automated RO pilot plant with a "capacity [of] 3 m^3/h (RO permeate)." Pre-treatment: acid, anti-scalant, automated filtration (50 um), and cartridge filtration (1um). 2 stages: in the first stage, 2 parallel membranes, second stage, concentrate from stage one to second membrane before mixing with stage 1 permeate. Post-treatment: Blending, CO2 removal. New membranes every 5 years. RO was least expensive despite having highest energy demands (0.6 kWh/m^3 of DW), due to low cost of electricity. Lower operator demand than BD. RO produced better quality water (lower TDS, Ca, Mg, Na, K, Cl, nitrate, sulfate and bicarbonate), but the greatest percentage of waste.	Panglisch et al. (2005) Dördelmann et al. (2006 and ***N.D.***) Dördelmann (2009)
RO – Full-scale installations	Milan, Italy: Influent of 11 – 14 mg/L as N (50 – 60 mg/L as nitrate), target of 9 mg/L as N (40 mg/L as nitrate) with waste concentrate < 30 mg/L as N (132 mg/L as nitrate) (for sewer disposal). Series of 13 1-stage RO plants (7 to 58 m^3/hr permeate), with blending 77 – 88% water recovery (note the disposal limitation). Pretreatment: anti-scalant (2.0 – 3.5 mg/L). Cleaning only every year and a half (SDI<1) and requires only 3 techs.	Elyanow & Persechino, (2005)
RO – Alternative disposal options	Alternatives to conventional disposal measures of RO waste brine, including reuse for industrial processes, processing (e.g., for salt production), or use in energy generation ("solar brine pond").	Howe (2004)
RO – Membrane scaling and fouling, colloidal interaction	Analysis of complex source water, membrane fouling and ways of anticipating changes in flux and rejection rates. Relationship between constituents and how RO treatment is affected by colloids, silica, concentration polarization (higher salt concentration near membrane surface), back diffusion, and scaling. Typical colloidal constituents: "sulfur, silica, and ferric and aluminum hydroxides." Results indicate colloidal fouling may amplify other scaling/fouling factors. "By recognizing possible interferences between rejected salts and colloids deposited on the membrane surface, the work explored the phenomenon of coupling between colloidal fouling, concentration polarization, and scaling as main factors limiting the application of RO membranes" (Tarabara 2007).	Tarabara (2007) Wang & Tarabara (2007)
RO – Influence of upstream processes	Investigation of the interaction of upstream residuals on RO treatment (membrane and additive chemicals). Influence of coagulant residuals on colloidal fouling, and disinfectants on membrane oxidation.	Gabelich et al. (2004)

Adminsitrative Record
Page 35455

Table A.3. Selected research on the use of electrodialysis/electrodialysis reversal for nitrate removal.

ED – Pilot study and Full-scale installation	Austria: 2-year pilot starting in 1990 with 1 m^3/h capacity. Influent nitrate concentration 18 – 23 mg/L as N (80 – 100 mg/L as nitrate), design for 36 mg/L as N (160 mg/L as nitrate). Planning began in 1996 for full-scale installation with seasonal operation commencing in 1997. Disposal options: sewer or irrigation reuse. "Monovalent selective anion exchange membranes." 3 stacks in parallel – 3 stages offer capacity from 48 to 144 m^3/h. Complete automation. Effluent nitrate of 9 mg/L as N (40 mg/L as nitrate) (capable of product water nitrate of 5.7 mg/L as N, 25 mg/L as nitrate) and 23% hardness reduction. Nitrate selectivity: 66% nitrate removal with 25% desalination.	Hell et al. (1998)
ED – Pilot study – Optimization	Morocco: Pilot study using a nitrate selective membrane for nitrate removal from an influent level of 20 mg/L as N (90 mg/L as nitrate) to acceptable levels, in water with a TDS concentration of 800 mg/L. Analysis of operating parameters to minimize precipitation, scaling and associated chemical use. Variation of voltage, flow rate and temperature. Ion removal increased with increasing temperature.	Midaoui et al. (2002)
EDR – Full-scale installation	Delaware: EDR has been successfully employed for the removal of nitrate from potable water in Delaware. The nitrate concentration in treated potable water was 1 mg/L as N (4.4 mg/L as nitrate), just 7.5% of the 13.5 mg/L as N (60 mg/L as nitrate) influent nitrate concentration. 3 stage system, 88% demineralization (TDS reduction), 90% water recovery, and pH decrease from 6.2 to 5.4.	Prato & Parent (1993)
EDR – Full-scale installation	Barcelona, Spain: GE EDR plant with 50 MGD capacity (260,000 households), not specifically for nitrate. Limited to no anti-scalant use. "Compared to a typical RO treatment facility producing 3.8 million gallons of water per day, GE's EDR technology, operating at 83% efficiency, is designed to eliminate the need for over 28,000 pounds of anti-scalant, reducing operating costs by > $100,000 per year at typical 2008 chemical prices." *Previous EDR systems for nitrate removal were installed in Arizona, Delaware, Japan, Italy, Bermuda, and Israel.	GE (2010), GE (N.D.)
EDR – Full-scale installations	Europe: As of ~2005, Ameridia/Eurodia full-scale EDR plants in Italy, France, Switzerland, Netherlands, and Austria. Capacity ranging from 0.032 – 0.925 MGD and water recovery from 93 – 98%. Information on full-scale installations, pilot studies and costs.	Ameridia/Eurodia
ED – Multiple contaminant removal, role of organic matter	Research investigation using synthetic waters to determine the impact of organic matter in the removal of nitrate, fluoride and boron. Fouling led to decreased flux, although nitrate removal was the least affected, due to smaller "hydrated ionic radius." Removal of boron and fluoride was enhanced by the presence of organic matter, while nitrate removal was not enhanced and simply decreased over time with the decrease in membrane flux.	Banasiak & Schäfer (2009)
ED – Pilot study	Research investigation using synthetic waters to assess nitrate removal under different operating conditions. The impact of different voltage (from 40 to 50 V) was examined across several ionic species, including nitrate. Alternating anion and cation exchange membranes were used. Results indicate 94% nitrate removal, with a reduction in the removal rate at 50 V due to back diffusion and fouling.	Nataraj et al. (2006)
ED – Pilot study comparison with adsorption	Morocco: Nitrate removal from brackish water. Comparison of ED using a monovalent membrane and adsorption on chitosan. ED successfully removed nitrate. Adsorption can remove nitrate, but not likely feasible. Adsorption can be used to remove nitrate from ED waste concentrate. Highlights concerns regarding waste concentrate disposal from ED.	Sahli et al. (2008)
	Mashhad, Iran: Comparison of IX, RO, BD, and ED. ED pilot study was started in 2007.	Dördelmann (2009)

Adminsitrative Record
Page 35456

Table A.4. Selected research on the use of biological denitrification for nitrate removal from potable water.

Pilot Testing - Multiple Biological Configurations
The City of Glendale, AZ, has investigated three configurations of biological treatment to address high nitrate levels in groundwater wells (Meyer et al. 2010). An autotrophic MBfR using was compared with two heterotrophic fixed bed bioreactors each with different media (plastic versus granular activated carbon). Hydrogen gas and ethanol were used as electron donor for the autotrophic system and the heterotrophic systems, respectively. Post-treatment included filtration using biologically activated carbon and ozonation. The fixed bed bioreactor with plastic media and the MBfR performed well, with product water meeting or exceeding potable water standards. Multi-criteria analysis found the MBfR to be most favorable regarding benefits, but the least favorable regarding costs. Including comparison with IX, the fixed bed bioreactor with plastic media had the lowest life cycle cost. The MBfR costs were greatest.
An investigation of biological treatment options in the City of Thornton, CO, funded by the WaterRF, examined two packed bed bioreactors and a moving bed biofilm reactor (MBBR™) (City of Thornton 2010, Project # 4202) to address nitrate impacted source water. Nitrate levels were successfully decreased by each of the three pilot systems from an influent concentration of 10 mg/L nitrate as N to an effluent concentration of < 2 mg/L nitrate as N. Operation at high and low temperatures was tested with examination of seeding for low temperatures. The study highlights the need for nutrient and substrate dose optimization in biological treatment systems.
Substrates
Numerous alternative substrate options have been explored in the literature including newspapers, vegetable oil, cotton, and formate (Volokita et al. 1996; Hunter 2001; Killingstad et al. 2002; and Della Rocca et al. 2006.)
Fixed Bed
See Riverside, CA Case Study (Carollo Engineers 2008).
A fixed bed heterotrophic denitrification pilot study was implemented in Mashhad, Iran by Dördelmann et al. (2006) using two parallel fixed beds containing expanded clay media. Acetic acid and ferrous sulfate served as the electron donor and nutrient supply, respectively. Post treatment consisted of "aeration, dual media and activated carbon filtration" (Dördelmann et al. 2006). Influent nitrate levels of 26 mg/L as N (115 mg/L as nitrate) were decreased to < 9 mg/L as N (40 mg/L as nitrate) with a nitrate reduction rate of ~7 kg NO_3/m^3 d (0.43 lb NO_3/ft^3d) (Panglisch et al. 2005 and Dördelmann et al. 2006). Used for flushing and backwashing, 7% of influent volume was wasted. In practice, a final disinfection step would be required (Dördelmann et al. N.D).
An up-flow, fixed-bed, autotrophic, lab-scale system, using granular sulfur as both substrate and growth surface was explored by Soares (2002). Operated over a 5 month period, a maximum denitrification rate of 0.2 kg N/m^3d (0.012 lb N/ft^3d) was achieved with a one hour hydraulic retention time and a loading rate of 0.24 kg N/m^3d (0.015 lb N/ft^3d). Sulfur based autotrophic systems would not be appropriate for the treatment of feed waters high in sulfate.
Aslan (2005) examined a lab-scale packed sand bed system, with ethanol as substrate for the simultaneous removal of nitrate and several pesticides. After 3 days for biofilm development, 93 – 98% nitrate removal was achieved requiring at least a 2 hour residence time. Pesticide removal required longer residence times (up to 12 hours) for efficient removal.
Upadhyaya et al. (2010) investigated the use of a fixed-bed biological reactor with granular activated carbon media for the removal of nitrate and arsenic at the same time. The media was biologically activated from use in a separate bioreactor for the removal of nitrate and perchlorate. Reactors were thus biologically active carbon (BAC) reactors. With acetic acid as the substrate, two in series BAC reactors were used to treat synthetic groundwater. Arsenic levels were reduced from 200 µg/L arsenic in the influent to 20 µg/L in the effluent (still above the arsenic MCL of 10 µg/L) while nitrate levels were decreased from 11 mg/L as N (50 mg/L as nitrate) in the influent to less than 0.045 mg/L as N (0.2 mg/L as nitrate) in the effluent. Used as an electron acceptor by microbes in the oxidation of substrate, nitrate was reduced to nitrogen gas. Arsenic was removed from solution with the formation of arsenic sulfide (solids) and also with adsorption and "surface precipitation on iron sulfides (p. 4958)."

Adminsitrative Record
Page 35457

Fluidized Bed
See Rialto, CA Case Study (Webster & Togna 2009).
Kurt et al. (1987) investigated an autotrophic fluidized sand bed reactor using hydrogen gas as substrate. With an influent nitrate concentration of 25 mg/L nitrate-nitrogen, a maximum nitrate reduction rate of 5 mg/L per hour was attained using a mixed culture. The authors propose multiple stage reactors to address the problem of partial denitrification.
Using a mix of propionic acid and ethanol in a heterotrophic fluidized sand bed reactor, Holló & Czakó (1987) examined denitrification at the lab- and pilot-scale. Post-treatment consisted of cartridge filtration, gas exchange, sand filtration, carbon filtration and disinfection. "Nitrate removal capacity of the reactor was 50 – 60 kg NO_3^-/m^3/day (3.1 – 3.7 lb NO_3^-/gal/d), which could be maintained permanently at temperatures as low as 8 – 10°C as well" (Holló & Czakó 1987).
MBR - Diffusive Extraction and Microporous Membranes (See also Pilot Testing above)
Mansell & Schroeder (2002) assessed hydrogenotrophic denitrification at the lab-scale using a membrane bioreactor (MBR) with a 0.02 micron microporous membrane through which nitrate diffuses to the biological compartment. The membrane prevents mixing of microbes with the water being treated and no carbon substrate was necessary because hydrogen gas was supplied as the electron donor for autotrophic denitrification. Previous issues regarding the transfer of hydrogen gas to the water and safety concerns due to explosive nature of hydrogen gas have been addressed with the development of "membrane dissolution systems" (Mansell & Schroeder 2002). Hydrogen gas was delivered to the biomass with silicone tubing. Results indicated reduction of nitrate levels from a maximum of 40 mg/L NO_3^--N in the feed water to 3.2 mg/L NO_3^--N in the treated water, with 92 – 96% removal. Measured HPC indicated minimal biomass transfer to the treated water compartment.
Ergas & Rheinheimer (2004) studied denitrification of potable water using a membrane bioreactor (MBR) in which feed water is passed through tubular acrylonitrile membranes, nitrate diffuses through the membrane and denitrification occurs on the exterior membrane surface. The mean transfer to the biofilm was 6.1 g NO_3-N/m^2 d (0.6 g NO_3-N/ft^2). The ultimate methanol (substrate) loading rate of 1.1 g/d resulted in a mean concentration of nitrate in the treated potable water of 5.2 mg NO_3^--N/L. A mathematical model of nitrate mass transfer was developed. A removal efficiency of 99% was achieved with a starting concentration of 200 mg NO_3^--N/L. Use of the MBR allows for denitrification with separation of the water to be treated and biological treatment, thereby avoiding post treatment removal of biomass and dissolved organics. The effluent would require additional treatment, because 8% (30 mg/L) of the methanol crossed the membrane; the authors suggest that further development of the biomass could minimize methanol transfer to the effluent stream.
MBR - Gaseous Substrate Delivery – Hollow Fiber Membranes
Chung et al. (2007) explored the use of autotrophic denitrification for the treatment of highly concentrated waste from nitrate removal via anion exchange. Using a hydrogen-based, hollow fiber membrane biofilm reactor, the impact of brine concentration (up to 15%) on nitrate reduction was found to be significant due to microbial inhibition. In the reduction of nitrate, use of hydrogen gas rather than an organic substrate offers an inexpensive alternative for potable water treatment systems. Biomass production is decreased and there is no need to remove substrate residual, as there would be with the use of carbon substrates.
Using nitrate as the primary electron acceptor, hollow fiber MBfRs with hydrogen gas as electron donor can effectively decrease the levels of multiple contaminants including perchlorate, chromate and arsenate (Nerenberg & Rittman 2004). Low levels of these oxidized species without a primary electron acceptor can limit biological reduction; however, in the presence of nitrate, reduction can occur. When the concentrations of nitrate and the oxidized species of interest were 1.1 mg/L as N (5 mg/L as nitrate) and 1 mg/L, respectively, 99% nitrate removal was achieved while removal of perchlorate, chromate and arsenate reached 36%, >75%, and >50%, respectively.

Adminsitrative Record
Page 35458

Bioelectrochemical Denitrification
Ghafari et al. (2008) provide a review of biological denitrification with a focus on bioelectrical reactors (BERs). In a BER, hydrogenotrophic denitrification occurs as hydrogen gas is produced at the cathode and utilized as the electron donor by denitrifiers, while nitrate is reduced to nitrogen gas. Previous BER research is discussed including autotrophic and heterotrophic examples across a range of nitrate levels and generally in synthetic waters. With additional research, BERs may become a feasible alternative for nitrate removal from drinking water.
In situ Denitrification
Permeable reactive barriers (PRBs) can be used to directly treat nitrate contaminated groundwater. Hunter (2001) examined the use of vegetable oil as an electron donor in biological denitrification. The use of an insoluble substrate minimized biomass blockage, a problem common with the use of soluble substrates like ethanol, methanol and acetate. The barrier was composed of soybean oil-coated sand and effectively decreased the nitrate levels from a starting concentration of 20 mg/L nitrate-N to below the MCL for a period of 15 weeks, with a flow rate 1100 L/week. After 15 weeks, insufficient oil remained for denitrification. High chemical oxygen demand, TSS and turbidity in the effluent of the reactor indicate a longer sand bed was needed; however, the author suggests that in situ application of this type of biological reactor would decrease these factors naturally. With a withdrawal point far enough from the barrier, subsequent potable water treatment requirements would be limited to disinfection. The most significant problem encountered in this study was the exhaustion of substrate. An effective means of substrate addition must be found (injection for example), but this was not explored. The estimated life of the PRB was 2.5 – 12.5 years depending on several key factors including flow, nitrate concentration and dissolved oxygen concentration.
With hydrogen gas as the substrate in autotrophic denitrification, Haugen et al. (2002) examined hydrogenotrophic denitrification in a lab scale experiment intended to imitate in situ treatment. Denitrification kinetics, the feasibility and longevity of substrate delivery via tubular membranes and post-treatment water quality were investigated. Delivery of hydrogen gas through tubular membranes minimized the risks associated with utilization of this flammable/explosive gas. The reactor was tested over 155 days. An initial influent nitrate concentration of 8.2 mg NO_3^--N/L was doubled to 16.4 mg NO_3^--N/L. After adjustment of parameters, complete nitrate removal was achieved using the tubular membrane bioreactor. A denitrification rate of 169 mg N/h/m^2 (membrane surface area) was attained with a hydrogen gas pressure of 1.44 atm (lower pressures resulted in incomplete reduction. The greatest hydrogen gas transfer across the membrane (flux) was 1.79 x 10^{-2} mg H_2/m^2s. The simulated groundwater velocity was 0.3 m/d resulting in 14 minutes of membrane contact time. Additional considerations for application of this treatment method include: the lower temperature of groundwater, the need for buffer in the current study, the depth limitation, nutrient requirements, and the difference between aquarium rocks and subsurface porous media. Intermediate denitrification products and end products (ammonia and nitrogen gas) were not measured in this study; however, the authors suggest the high nitrate to substrate ratio would result in reduction to nitrogen gas.
Schnobrich et al. (2007) simulated in situ nitrate removal via hydrogenotrophic Denitrification. With hydrogen gas delivery through a membrane module consisting of a fiberglass membrane wound in a spiral fashion and attached to polyethylene membranes. The study examined the influence of pH, nutrient requirements and the feasibility of appropriate levels of hydrogen gas delivery. To simulate in situ conditions, the porous media was extracted from an aquifer and the system was operated at 10°C. Two up-flow columns were operated in series, only the first of which included a membrane fed with hydrogen gas. Including that required for the reduction of dissolved oxygen, the total concentration of hydrogen gas required for complete denitrification was 11.2 mg H_2/L. Overall, this study demonstrated effective substrate delivery and nitrate removal under conditions more similar to what would be expected naturally.

Adminsitrative Record
Page 35459

Table A.5. Selected research on the use of chemical denitrification (CD) for nitrate removal.

CD using ZVI	Reduction of nitrate to ammonia using ZVI powder was highly pH dependent with optimal kinetics below a pH of 4. The minimum ratio of iron to nitrate was 120 m^2/mol NO_3^- for complete reduction within 1 hour. 50 mg NO_3^-/L, 100% removal.	Huang et al. (1998)
CD – ZVI	Found that the end product of denitrification (nitrogen gas versus ammonium) could be controlled by the iron to nitrate ratio and the use of catalysts.	Xiong et al. (2009)
CD – ZVI	In the corrosion of ZVI, the formation of "green rusts" and "suspended green particles" is associated with stabilization of pH and steady decrease in nitrate.	Choe et al. (2004)
CD – ZVI	Examined the nitrate reduction rates of three types of iron. Findings indicate that rate increases with decreasing pH.	Alowitz & Scherer (2002)
CD – ZVI	Nitrate reduction by ZVI can be optimized through pretreatment of iron particles. High temperature exposure to hydrogen gas and deposition of copper were explored separately as options for pretreatment of the iron surface.	Liou et al. (2005)
CD – ZVI	Examined chloride as a potential competitor. Results indicate a minimal impact on nitrate removal; however, other competing ions could be important regarding both competition for adsorption sites and reduction.	Moore & Young (2005)
CD - SMI	"SMI-III® is a patented, iron-based granular media that has been commercially developed for the removal of nitrate, co-contaminants including uranium, vanadium and chromium, and other compounds from water. It is foreseen that the greatest benefit of this technology is that it does not produce a costly brine stream as do the currently accepted nitrate removal technologies of ion exchange and reverse osmosis."	DSWA and City of Ripon (2010)
CD – Catalytic Denitrification	Reddy & Lin 2000; Pintar et al. 2001; Gavagnin et al. 2002; Lemaignen et al. 2002; Pirkanniemi & Sillanpaa 2002; Chen et al. 2003; Palomares et al. 2003; Pintar 2003; Constantinou et al. 2007; and Sun et al. 2010.	

Adminsitrative Record
Page 35460

Table A.6. Advantages and disadvantages of the five major treatment options for nitrate removal.

	Advantages	Disadvantages
Ion Exchange	• Years of industry experience, • Multiple contaminant removal, • Selective nitrate removal, • Financial feasibility, • Use in small and large systems, and • The ability to automate.	• The disposal of waste brine, • The potential for nitrate dumping specifically for non-selective resin use for high sulfate waters, • The need to address resin susceptibility to hardness, iron, manganese, suspended solids, organic matter, and chlorine, and • The possible role of resin residuals in DBP formation.
Reverse Osmosis	• High quality product water, • Multiple contaminant removal, • Desalination (TDS removal), • Feasible automation, • Small footprint, and • Application for small and POU applications.	• The disposal of waste concentrate, • Typically high capital and O&M costs, • The need to address membrane susceptibility to hardness, iron, manganese, suspended solids, silica, organic matter, and chlorine, • High energy demands, and • The lack of control over target constituents (complete demineralization).
Electrodialysis/ Electrodialysis Reversal	• Limited to no chemical usage, • Long lasting membranes, • Selective removal of target species, • Flexibility in removal rate through voltage control, • Better water recovery (lower waster volume), • Feasible automation, and • Multiple contaminant removal.	• The disposal of waste concentrate, • The need to address membrane susceptibility to hardness, iron, manganese, and suspended solids, • High maintenance demands, • Costs (comparable to RO systems), • The need to vent gaseous byproducts, • The potential for precipitation with high recovery, • High system complexity, and • Dependence on conductivity.
Biological Denitrification	• High water recovery, • No brine or concentrate waste stream (nitrate reduction rather than removal to waste stream), • Low sludge waste, • Less expensive operation, • Limited chemical input, • Increased sustainability, and • Multiple contaminant removal.	• The need for substrate and nutrient addition, • High monitoring needs, • Significant post-treatment requirements, • High capital costs, • Sensitivity to environmental conditions (sometimes), • Large system footprint (sometimes), • High system complexity (sometimes, can be comparable to RO), • Lack of full-scale systems in the U.S., • The possibility of partial denitrification, • Permitting and piloting requirements, and • Slower initial start-up, which could cause challenges for wells with intermittent run time.

Chemical Denitrification	• Conversion of nitrate to other nitrogen species (no brine or concentrate waste stream),
	• The potential for more sustainable treatment,
	• High water recovery (higher than RO according to Cleanit®-LC), and
	• Multiple contaminant removal.
		• The potential reduction of nitrate beyond nitrogen gas to ammonia,
		• The possibility of partial denitrification,
		• The possible dependence of performance on pH and temperature,
		• The possible need for iron removal, and
		• The lack of full-scale chemical denitrification systems resulting in:
		  ○ Unknown reliability,
		  ○ Unknown costs, and
		  ○ Unknown operational complications.

Adminsitrative Record
Page 35461

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 7:

Alternative Water Supply Options for Nitrate Contamination

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
University of California, Davis
http://groundwaternitrate.ucdavis.edu

Prepared for the California State Water Resources Control Board

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Page 35462

Alternative Water Supply Options for Nitrate Contamination

Technical Report 7

Addressing Nitrate in California's Drinking Water

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Report for the State Water Resources Control Board Report to the Legislature

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Contents

Tables v

Figures viii

Acronyms and Abbreviations x

Unit Conversions xii

Summary 1

1 Introduction 3

2 Study Area Background 5

2.1 Tulare Lake Basin (TLB) 5

2.2 Salinas Valley (SV) 7

2.3 Drinking Water Systems 7

3 Susceptible Water Users 9

3.1 Drinking Water Supply Systems 9

3.2 Water Quality Threats 10

3.3 Susceptible Water Users Overview 11

3.4 Water System Infrastructure 14

3.4.1 Low System Vulnerability 16

3.4.2 High System Vulnerability 18

3.4.3 No System Vulnerability 22

3.5 Susceptible Drinking Water Users 23

3.5.1 Highly Susceptible Population 24

3.5.2 Low Susceptibility Population 34

3.5.3 Unknown Susceptibility Population 37

3.6 Major Findings on Susceptible Water Users 37

3.7 Health and Socioeconomic Disparities 42

Adminsitrative Record
Page 35467

3.7.1 Pregnant Women and Infants 43
3.7.2 Disadvantaged Communities (Disadvantaged Unincorporated Communities) 43
3.8 Trends in Affected Populations over Time 45
3.8.1 Accumulating Nitrate and Population Changes 45
3.8.2 Drinking Water System Regulatory Changes 47
4 Alternative Water Supply Options 48
4.1 Improve Existing Water Source 48
4.1.1 Blending 49
4.1.2 Drilling a Deeper or New Well 49
4.1.3 Community Treatment 51
4.1.4 Household Treatment 52
4.2 Connect to Alternative Water Supplies 53
4.2.1 Switch to Treated Surface Water 54
4.2.2 Piped Connection to an Existing System 54
4.2.3 Piped Connection to a Larger Newly Created System 55
4.2.4 Regionalization and Consolidation 56
4.2.5 Trucked Water 60
4.2.6 Bottled Water 61
4.3 Relocate to Area with Better Water Supply 61
4.4 Ancillary Activities 62
4.4.1 Dual Water Distribution System 62
4.4.2 Well Water Quality Testing 62
4.4.3 Rainwater Cisterns 63
5 Evaluation of Options 64
5.1 Economic and Financial Costs 65

Adminsitrative Record
Page 35468

5.1.1 Blending 68
5.1.2 Drilling a Deeper Well 69
5.1.3 Drilling a New Well 70
5.1.4 Community Treatment 71
5.1.5 Household Treatment 72
5.1.6 Connect to an Alternative System 74
5.1.7 Trucked Water 75
5.1.8 Bottled Water 75
5.1.9 Relocate to Area with Better Water Supply 76
5.1.10 Dual Water Distribution System 77
5.1.11 Well Water Quality Testing 78
5.1.12 Summary of Alternative Water Supply Costs 79
5.2 Least Cost Management 82
5.2.1 Self-Supplied Households or Local Small Water Systems 83
5.2.2 Community Public Water Systems 86
5.3 Public Health and Other Considerations 92
6 Basin-wide Costs of Nitrate Contamination 94
6.1 Costs for Community Public and State Small Water System Alternative Solutions 94
6.2 Costs for Household Self-Supplied and Local Small Water System Alternative Solutions 100
6.3 Interim Solutions 101
7 Regulatory and Implementation Implications 103
7.1 Implications of Household Self-Supplied System Alternatives 103
7.2 Implications of Community Public Water System Alternatives 103
8 Conclusions 107
8.1 Major Findings 107

Adminsitrative Record
Page 35469

8.2 Promising Actions 109

9 Literature Cited 111

10 Appendix 125

10.1 Methods 125

10.1.1 Susceptibility Charts 125

10.1.2 Estimating “Delivering” Sources of a System 126

10.2 Rainwater Cisterns 128

10.3 Glossary 129

11 Case Studies 131

Adminsitrative Record
Page 35470

Tables

Table 1. Drinking water system connections and service duration. 8

Table 2. California's drinking water quality responsibilities. 11

Table 3. Community public water systems treating for nitrate in the Tulare Lake Basin. 17

Table 4. Community public water systems treating for nitrate in the Salinas Valley. 17

Table 5. 2010 Estimated self-supplied and local small water system population within the study area based on the parcel use code and DWR land use designation. 19

Table 6. 1990 Census v. 2010 Estimated domestic well (single dwelling unit) population by county. 21

Table 7. 2010 Estimated high susceptibility population served by self-supplied household and local small water systems (referred to as 'domestic well'). 27

Table 8. Single source systems with a high likelihood of nitrate in groundwater. 28

Table 9. Community public water systems in violation of the nitrate MCL (2004-2008). 33

Table 10. 2010 Estimated low susceptibility population served by self-supplied household and local small water systems. 35

Table 11. Single source systems with a low likelihood of nitrate in groundwater. 36

Table 12. Community public water systems with a low likelihood of nitrate in groundwater. 36

Table 13. Assessment of susceptible water users in the study area. 39

Table 14. Long-term trends for nitrate concentrations in public supply wells. 46

Table 15. Estimated time for CPWS sources to exceed the nitrate MCL and total affected population 46

Table 16. Alternative water supply options. 48

Table 17. Influence of nitrate concentration on treatment selection. 51

Table 18. Large systems and the number of nearest small systems identified by distance proximity nearest-neighbor analysis in the Tulare Lake Basin. 58

Table 19. Large systems and the number of nearest small systems identified by distance proximity nearest-neighbor analysis in the Salinas Valley. 59

Table 20. Advantages and disadvantages of options for public water systems. 64

Adminsitrative Record
Page 35471

Table 21. Advantages and disadvantages of options for self-supplied households or local small water systems. 65

Table 22. Summary of approximate alternative water supply option costs. 67

Table 23. Estimated costs for blending. 69

Table 24. Estimated costs for drilling a deeper well. 70

Table 25. Annualized total cost ranges for drilling a new domestic well. 70

Table 26. Annualized total cost ranges for drilling a new public supply well. 71

Table 27. Annualized total cost ranges for public supply well rehabilitation. 71

Table 28. Annualized total cost ranges for groundwater treatment systems. 72

Table 29. Lower bound annualized cost of SCADA infrastructure. 72

Table 30. U.S. EPA reverse osmosis POU device cost estimates. 73

Table 31. Culligan reverse osmosis POU device cost estimates. 74

Table 32. Estimated annualized cost ranges for a system with 1,000 connections to connect to an alternative system 75

Table 33. Average listing prices for homes in study area counties. 76

Table 34. The estimated cost for relocating households. 76

Table 35. Estimated annualized cost ranges for a dual water distribution system. 78

Table 36. Summary of the estimated alternative water supply costs for self-supplied households. 80

Table 37. Summary of the estimated alternative water supply costs for a small water system (1,000 households) 81

Table 38. Estimated O&M and capital cost ranges for alternative water supply options as percentage of total annual costs. 82

Table 39. Estimated household costs for alternative solutions for self-supplied households or local small water systems. 84

Table 40. Low cost ranges for a basin-wide cost analysis. 88

Table 41. High cost ranges for a basin-wide cost analysis. 89

Table 42. Complete table of basin-wide alternative water supply options (minimum value of low range and maximum value of high range). 90

Adminsitrative Record
Page 35472

Table 43. Estimated cost range of the least cost short-term alternative water supply options for susceptible community public and state small water systems based on system size and proximity to a larger system. 97

Table 44. Estimated cost range of the least cost long-term alternative water supply options for susceptible community public and state small water systems based on system size and proximity to a larger system. 99

Table 45. Estimated O&M cost of the least cost short- and long-term alternative water supply options for susceptible community public and state small water systems. 100

Table 46. Estimated cost ranges per household served for interim POU water supplies for households and small water systems serving fewer than 200 connections. 102

Adminsitrative Record
Page 35473

Figures

Figure 1. Bulletin 118 groundwater basins in the study area. 5

Figure 2. Salinas Valley and Tulare Lake Basin counties. 6

Figure 3. The size distribution (by population served) for state small (state-documented, listed in PICME) and community public water systems in the study area. 15

Figure 4. State small (state-documented) and community public water systems treating or not treating for nitrate in the Tulare Lake Basin. 16

Figure 5. State small (state-documented) and community public water systems treating or not treating for nitrate in the Salinas Valley. 16

Figure 6. Estimated location of the area's 74,000 unregulated self-supplied water systems and the 402 regulated community public and state-documented state small water systems. 20

Figure 7. Community public or state small water systems with only one well. 22

Figure 8. Community public water systems with only surface water sources. 23

Figure 9. Maximum raw source water nitrate concentration within shallow wells (≤ 300') as Thiessen polygons. 26

Figure 10. Household self-supplied and local small water systems within a Thiessen polygon having a maximum nitrate concentration value greater than 45 mg/L. 27

Figure 11. Maximum raw nitrate level records in community public and state small water systems. 29

Figure 12. Community public water systems of low vulnerability with delivered water in excess of 45 mg/L as nitrate at least once. 30

Figure 13. System size distribution (by population served) of the low vulnerability state-documented state small or community public water systems exceeding 45 mg/L as nitrate. 30

Figure 14. Community public water systems in the study area in violation of the nitrate MCL 2004 to 2008. 34

Figure 15. Household self-supplied and local small water system wells within a Thiessen polygon of a tested upper aquifer well (<300 ft) with maximum nitrate concentration less than 45 mg/L. 35

Figure 16. System size distribution (by population served) of the community public water systems without water quality information. 37

Figure 17. Classification of susceptible populations based on estimated vulnerability and water quality data for the study area. 40

Adminsitrative Record
Page 35474

Figure 18. 2010 Population and susceptibility characterization for the Tulare Lake Basin based on estimated vulnerability and water quality data. 41

Figure 19. 2010 Population and susceptibility characterization for the Salinas Valley based on estimated vulnerability and water quality data. 42

Figure 20. The relationship between DACs and SDACs and delivered water quality in multiple source CPWSs. 44

Figure 21. The maximum raw source water nitrate concentration for 328 state small and community public water systems in the Tulare Lake Basin and Salinas Valley with MHI data. 45

Figure 22. The minimum distance from a small system (≤ 10,000 people) to a larger system (> 10,000 people) for the entire study area. 55

Figure 23. The minimum distance ranges between all existing small systems (≤ 10,000 people) and larger systems (> 10,000 people) in the Tulare Lake Basin. 59

Figure 24. The minimum distance ranges between all existing small systems (≤ 10,000 people) and larger systems (> 10,000 people) in the Salinas Valley. 60

Figure 25. Annualized total household cost for household self-supplied and local small water system alternatives. 85

Figure 26. Annualized cost ($/kgal) for community public water system alternatives (excluding pipeline costs). 91

Figure 27. Community public water system annualized cost comparison ($/kgal) for alternative water supply option (including pipeline for connecting to a nearby system and excluding SCADA costs). 91

Figure 28. Annualized cost comparison ($/hhld) of alternative water supply options for a 2,000 connection community public water system. 92

Figure 29. SHORT-TERM: Least cost alternative supply option (INCLUDING POU) based on high estimate of option costs for susceptible community public and state small water systems. 96

Figure 30. SHORT-TERM: Least cost alternative supply option (EXCLUDING POU) based on high estimate of option costs for susceptible community public and state small water systems. 97

Adminsitrative Record
Page 35475

Acronyms and Abbreviations

ACR	Annual Compliance Report
Af	Acre-feet
af/year	Acre-feet per Year
ANSI	American National Standards Institute
BID	Background Information Document
CADWSAP	California Drinking Water Source Assessment and Protection Program
CCR	Consumer Confidence Report/California Code of Regulations
CDF	California Department of Finance
CDHS	California Department of Health Services
CDP	Census Designated Place
CDPH	California Department of Public Health
CFR	Calculated Fixed Radius
CPWS	Community Public Water System
CVP	Central Valley Project
CWC	California Water Code/Community Water Center
DAC	Disadvantaged Community
DBCP	Dibromochloropropane
DDWEM	Division of Drinking Water and Environmental Management (CDPH)
DUC	Disadvantaged Unincorporated Community
dus	Dwelling Units
DWP	Drinking Water Program
DWR	Department of Water Resources
DWSAP	Drinking Water Source Assessment Protection Program
EDB	Ethylene Dibromide
EDR	Electrodialysis Reversal
ELAP	Environmental Laboratory Accreditation Program (CDPH)
FDA	Food and Drug Administration
FDB	Food and Drug Branch (CDPH division)
FOB	Field Operations Branch
GAMA	Groundwater Ambient Monitoring and Assessment
gpd	Gallons Per Day
gpm	Gallons Per Minute
hhld	Household
ICE	Information Center for the Environment (UC Davis)
IX	Ion Exchange (Anion Exchange)
kgal	Kilogallon = 1,000 gallons
LPA	Local Primacy Agency
LSWS	Local Small Water System
MCL	Maximum Contaminant Level

Adminsitrative Record
Page 35476

MGD	Million Gallons per Day
MHI	Median Household Income
N	Nitrogen
NAWQA	National Water-Quality Assessment
NGWA	National Groundwater Association
NO_3^-	Nitrate
NSF	National Sanitation Foundation
NTNC	Non-Transient Non-Community
O&M	Operations & Maintenance
OEHHA	Office of Environmental Health Hazard Assessment
PCA	Possible Contaminating Activities
PCE	Perchloroethylene
PHG	Public Health Goal
PICME	Permitting Inspection Compliance Monitoring and Enforcement
PLC	Programmable Logic Controller
POE	Point-of-entry
POU	Point-of-use
ppm	Parts Per Million
PWS	Public Water System
RO	Reverse Osmosis
SB	Senate Bill
SCADA	Supervisory Control and Data Acquisition
SDAC	Severely Disadvantaged Community
SDWA	Safe Drinking Water Act
SSWS	State Small Water System
SV	Salinas Valley
SWB	State Water Resources Control Board (State Water Board)
SWP	State Water Project
SWS	Small Water System
TCE	Trichloroethylene
TCP	Trichloropropane
TLB	Tulare Lake Basin
TMF	Technical, Managerial and Financial
TNC	Transient Non-Community
USBR	United States Bureau of Reclamation
USDA	United States Department of Agriculture
USGS	United States Geological Survey
U.S. EPA	United States Environmental Protection Agency
VOC	Volatile Organic Compound
WQM	Water Quality Management
ZOC	Zone of Contribution

Adminsitrative Record
Page 35477

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-).

To convert from:

Nitrate-N (NO_3-N)	→	Nitrate (NO_3^-)	multiply by 4.43
Nitrate (NO_3^-)	→	Nitrate-N (NO_3-N)	multiply by 0.226

Adminsitrative Record
Page 35478

Summary

The Tulare Lake Basin and Salinas Valley were chosen as pilot study areas because communities in these regions are faced with the need to manage high nitrate loads from agricultural lands and dairies, have a high risk of exposure to nitrate contamination in groundwater, and often cannot afford treatment or alternative water supply options. These factors combine to make the Tulare Lake Basin and Salinas Valley highly susceptible to health effects from nitrate in drinking water.

There are 371 active community public water systems (CPWS) and 30 state small water systems (SSWS) in the study area (281 and 120 in the Tulare Lake Basin and Salinas Valley, respectively). These systems supply water to about 2.4 million people. An estimated additional 245,490 people in these regions get their drinking water from an estimated 74,400 private domestic wells (self-supplied households or local small water systems with fewer than five connections) that are unregulated and largely unmonitored.

The 371 CPWSs are supplied by 3,829 sources and the 30 SSWSs by 31 sources. Of the 3,860 sources overall, 3,682 are wells pumping groundwater; the rest are surface water. According to the California Department of Public Health's (CDPH) Permitting Inspection Compliance Monitoring and Enforcement database (PICME), the Tulare Lake Basin has 8 SSWSs and 172 CPWSs serving very small systems (< 501 people) and 47 community public water systems serving small systems (501 to 3,300 people). About 81% of the Tulare Lake Basin water systems (five or more connections) are very small or small and in total serve 89,125 people (4% of the Tulare Lake Basin population). The Salinas Valley has 22 state small and 73 community public water systems serving very small systems and 11 community public water systems serving small systems. About 89% of the Salinas Valley water systems are very small or small and in total serve 23,215 people (6% of the Salinas Valley population). Thirteen water systems in the study area treat for nitrate. Of these, eight water systems treat by blending with lower nitrate sources, four water systems treat with ion exchange, and one system treats with reverse osmosis.

Approximately 254,000 people in California's Tulare Lake Basin and Salinas Valley have drinking water supplies susceptible or potentially susceptible to nitrate groundwater contamination. Water users that are served by a community public water system with water nitrate concentrations recorded in excess of 45 mg/L in the last five years, or lacking historical nitrate records, account for about 220,000 people. The remaining 34,000 people are estimated to be connected to a self-supplied household or local small water system located in areas with a high likelihood of groundwater in excess of 45 mg/L as nitrate. Assuming unchanging and unabated basin-wide trends in observed nitrate groundwater levels since 1970, the community public water system population facing financial impacts from excess nitrate in raw water supplies is estimated to increase from 57% currently, to 80% by 2050.

Each SSWS and CPWS with high susceptibility (50 and 35 systems in Tulare Lake Basin and Salinas Valley, respectively) will need individual engineering and financial analyses. No single solution will fit every community affected by nitrate in groundwater. In addition, the 34,000 people on approximately 10,000 household self-supplied systems at high risk of contamination remain unregulated and outside any currently existing regulatory framework to provide safe drinking water.

Adminsitrative Record
Page 35479

The most promising options (based on availability and economic considerations and in no particular order) for communities connected to highly susceptible water systems are to: i) consolidate with a nearby larger system; ii) consolidate with nearby smaller systems into a new larger regionalized system; iii) install groundwater community treatment; iv) drill a new well; v) blend sources; and as an interim solution, vi) provide and maintain point-of-use treatment for households. This analysis suggests significant potential for smaller water systems to consolidate with larger water systems. Promising solutions for self-supplied households or local small water systems within a highly susceptible sub-area are to install a point-of-use or point-of-entry treatment system or to drill a new, deeper well.

Consolidation of systems was examined based on system size and the distance from a smaller system to a larger system. In the Tulare Lake Basin and Salinas Valley, about 50% and 15% of smaller systems, respectively, are within five miles of a larger system (>10,000 people), and 88% and 97% of smaller systems, respectively, are within 12.5 miles of a larger system.

The overall cost of providing nitrate-compliant drinking water to the currently affected population in the Tulare Lake Basin and Salinas Valley is estimated to be about $20 million per year for the short-term and about $36 million per year for the long-term. Roughly $17 to $34 million per year of this estimate is for community public water system users for the short- to long-term, respectively, and about $2.5 million is for providing and maintaining point-of-use treatment for household self-supplied or local small water system users. To put this funding need in perspective, the overall costs correspond to $80 to $142 per year per susceptible person, $5 to $9 per study area irrigated acre per year, or $100 to $180 per ton of fertilizer nitrogen applied, for the short- to long-term, respectively.

Adminsitrative Record
Page 35480

1 Introduction

Nitrate is the most common chemical contaminant in the world's aquifers and has significant potential to harm human health (Spalding & Exner 1993). A 2002 report from Lawrence Livermore National Laboratory (Esser et al. 2002) concluded that of all regulated contaminants in drinking water, nitrate contamination poses the greatest threat to California's drinking water supply. High nitrate levels in groundwater in California are primarily from the use of fertilizers on agricultural land and land application of manure at dairies. On average, more than 80 pounds of nitrogen (N) per acre per year may leach into the groundwater from fertilizer application on California farms (Harter 2009 and Dzurella et al. 2012). Other locally important nitrate sources include animal feed lots, wastewater discharges, and septic systems. California's extensive agricultural lands and dairies have greatly increased nitrate loads to groundwater over time. Based on California Department of Public Health (CDPH) statewide data collected since 1980, of the approximately 13,150 public drinking water sources sampled, nitrate levels exceeded the primary drinking water standard (Maximum Contaminant Level or MCL) at least once in 1,077 wells (State Water Resources Control Board (SWB) 2010). A 1988 Report on nitrate in drinking water reported that 10 percent of the California samples in the U.S. EPA database exceeded the MCL, with the highest density of contaminated wells in the Central Valley located close to the Highway 99 corridor, in cities, and near dairies or feedlots (SWRCB 1988). In the San Joaquin Valley, between 2005 and 2008, 92 of the 671 drinking water systems had at least one groundwater well with nitrate levels exceeding the MCL. These 92 drinking water systems serve more than a million residents (Balazs 2011).

Groundwater nitrate concentrations exceeding 9 to 13 mg/L as nitrate (2 to 3 mg/L as N) generally indicate contamination from human-related nitrate sources (Mueller 1995). The drinking water MCL for nitrate was set by CDPH in 1994 at 45 mg/L as nitrate (NO_3^-). This is equivalent to the 1991 federally mandated MCL of 10 mg/L nitrate as nitrogen (N) (CaEPA 1997). In the regulatory literature, nitrate concentrations in water may be reported in milligrams of nitrate per liter (mg/L of NO_3^-) or in milligrams of nitrate-nitrogen per liter (mg/L of N). This report follows the convention of reporting in milligrams of nitrate per liter (as mg/L of NO_3^-).

The Salinas Valley and the Tulare Lake Basin were chosen as pilot study areas for examining solutions to groundwater nitrate contamination for reasons including the high dependence on groundwater for drinking, the high levels of nitrate observed in groundwater in community public water system and domestic wells, and the challenges faced by affected communities in securing safe drinking water in these regions. These factors combine to make the population in the Salinas Valley and the Tulare Lake Basin more at risk to health effects from nitrate in drinking water.

To address nitrate groundwater contamination in the Tulare Lake Basin and the Salinas Valley, the 2010 susceptible population is estimated and available alternative water supply options assessed for recent and long-term conditions. The susceptible population is the population that is currently at risk of drinking water from their tap with nitrate in excess of 45 mg/L, the safe drinking water limit, based on

Adminsitrative Record
Page 35481

the most recent drinking water quality monitoring data available. First, background on each basin and a map of the study area is presented. The area and population are reviewed. To quantify susceptible water users, each water supply system type is first categorized in terms of vulnerability to deliver nitrate contaminated drinking water should excess nitrate in a groundwater source occur, and then the likelihood that groundwater sources or a system's actual delivered water with nitrate concentrations above 45 mg/L is examined. Each identified alternative water supply option is evaluated in terms of financial and economic costs, public health concerns, least cost management, and regulatory implications. Based on the estimate of susceptible water users and the costs and technical feasibility of alternative water supply options, promising alternatives are presented for differing water system types and sizes.

Adminsitrative Record
Page 35482

2 Study Area Background

2.1 Tulare Lake Basin (TLB)

The Tulare Lake Hydrologic region, as defined by DWR in the 2003 update of Bulletin 118, covers 8,000 square miles in the southern Central Valley of California (DWR 2003).

Here, the Tulare Lake Basin Study Area is defined as only the areas within the larger DWR-defined Tulare Lake Hydrologic Region that are a part of the San Joaquin Valley Groundwater Basin. The 8,045 square miles of underlying basin area (DWR 2003) can be defined by the following DWR Bulletin 118 Groundwater Sub-Basins: 5-22.08 (Kings), 5-22.09 (Westside), 5-22.10 (Pleasant Valley), 5-22.11 (Kaweah), 5-22.12 (Tulare Lake), 5-22.13 (Tule), 5-22.14 (Kern County), and the southern tail of 5-22.07 (Delta-Mendota) (Figure 1). The Tulare Lake Basin portion of the study area includes the Central Valley basin parts of Fresno, Kings, Kern, and Tulare counties (Figure 2).

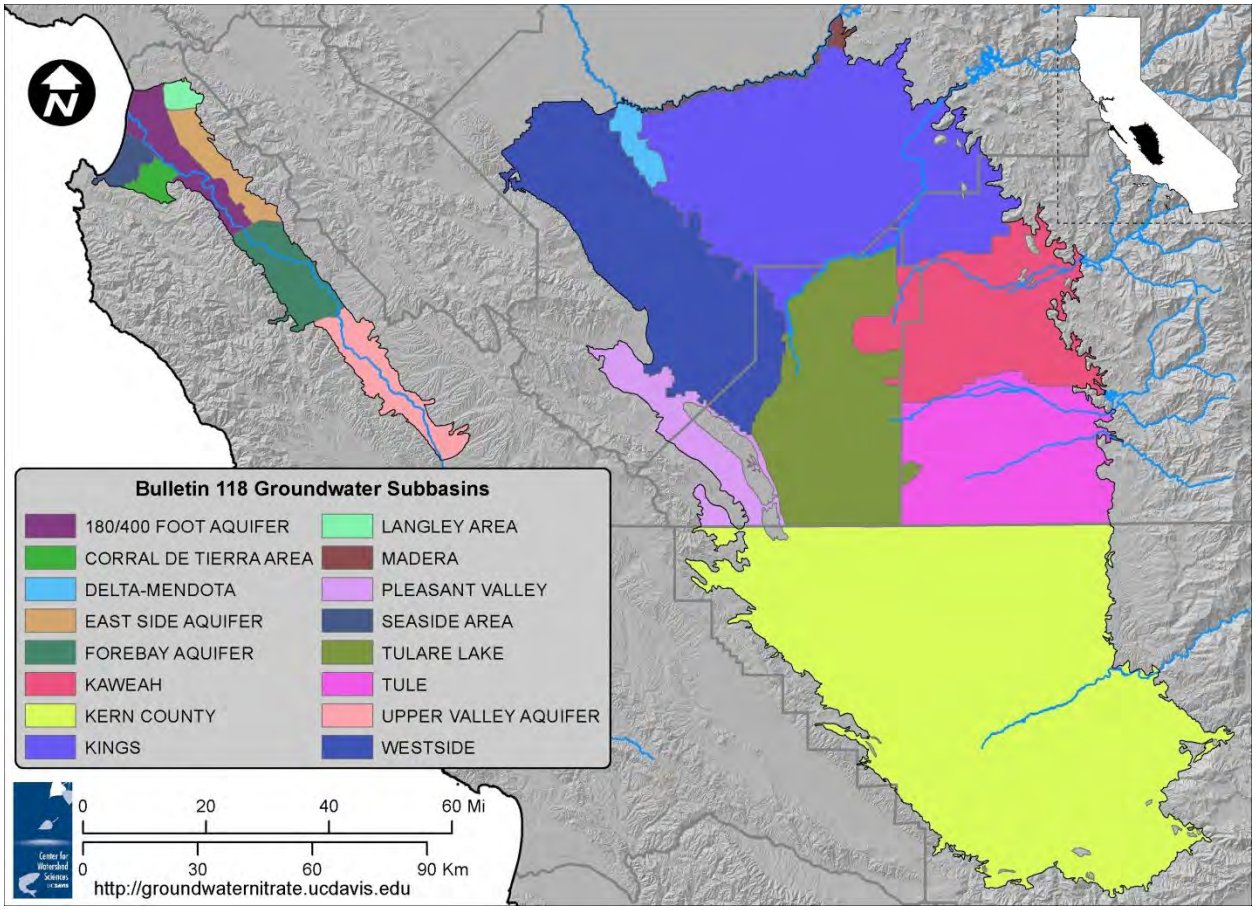

Figure 1. Bulletin 118 groundwater basins in the study area. (Source: DWR 2003.)

Adminsitrative Record
Page 35483

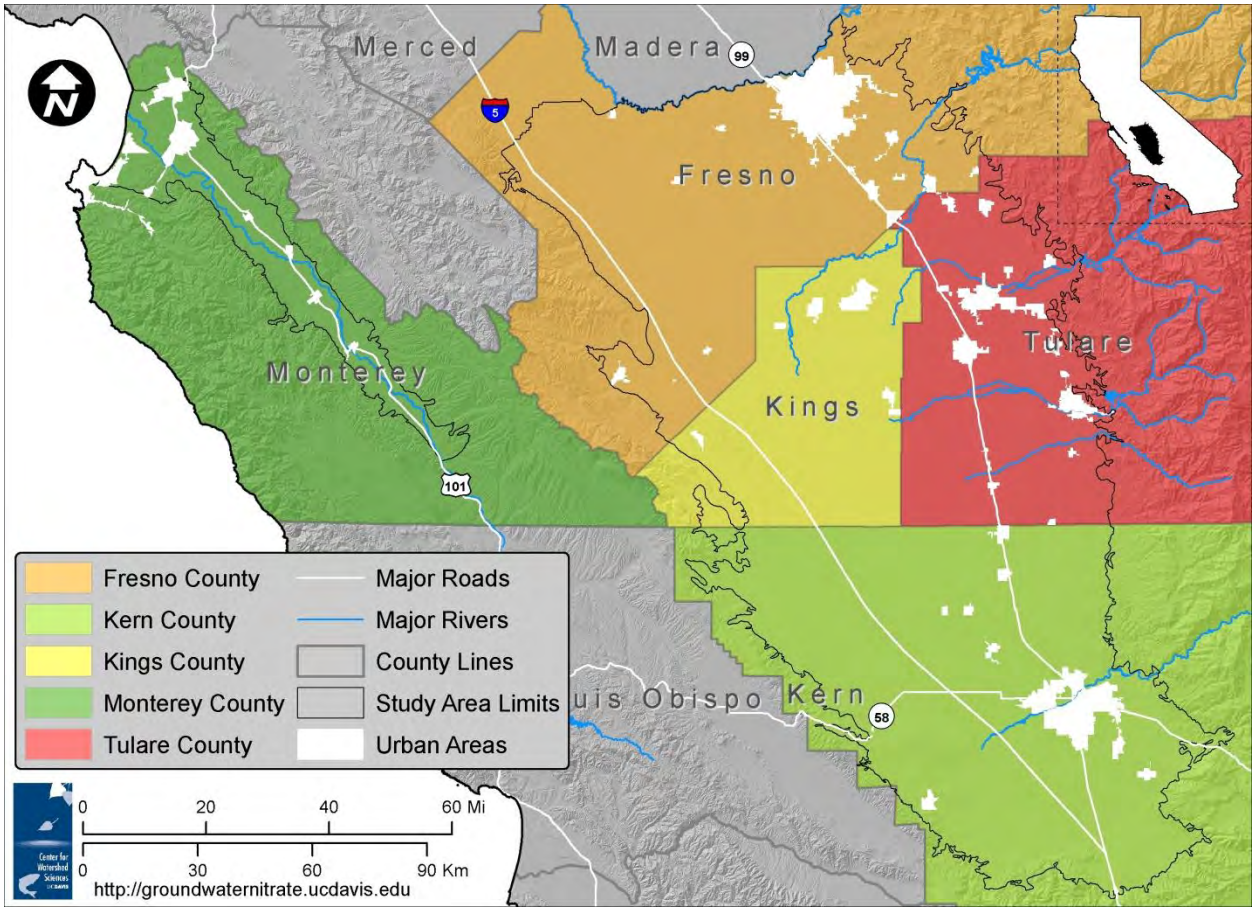

Figure 2. Salinas Valley and Tulare Lake Basin counties. (Source: 2000 Census.)

The Tulare Lake Basin has a Mediterranean climate with hot, dry summers and cool, moist winters. Average rainfall varies from seven inches in central and western parts of the Basin to thirteen inches in eastern parts of the Kaweah and Kern County Sub-Basins (DWR 2003). The Basin's largest city (Fresno) depends almost entirely on local groundwater (DWR 2003). Fresno, Tulare, Kern, and Kings Counties were first, second, third, and eighth among the nation's top agricultural producing counties with gross production values of $5.37 billion, $4.05 billion, $3.61 billion, and $1.76 billion for 2008 (USDA et al. 2008). There are approximately 3.4 million acres of irrigated land in the Tulare Lake Basin; production values are $1,579, $1,191, $1,062, and $518 per irrigated acre, for each county respectively.[2]

In 2000, 11% of the population, or over 200,000 people, lived in areas classified as rural. The other 1.6 million people lived in areas classified as urban.[3] Since 2000, the population of 1.8 million in the Tulare Lake Basin[4] grew to an estimated 2.25 million by 2010.

[2] California Department of Water Resources, Land Use Classification: Irrigated acre totals.

[3] This designation used 2000 census blocks. It follows the 2000 US Census method of defining urban versus rural on the basis of population density: an urbanized area or an urban cluster consists of: 1) core census block groups or blocks that have a population density of at least 1,000 people per square mile and 2) surrounding census blocks that have an overall density of at least 500 people per square mile (US Census Bureau N.D.).

Adminsitrative Record
Page 35484

2.2 Salinas Valley (SV)

With a total drainage area of five thousand square miles, the Salinas Valley (SV) watershed is the largest southern California coastal basin (Kulongoski & Belitz 2005). It is bordered by the San Joaquin Valley and the Pacific Ocean. Boundaries for the SV for this report follow DWR Bulletin 118 Groundwater Sub-Basins: 3-4.01 (180/400 Foot Aquifer), 3-4.02 (East Side Aquifer), 3-4.04 (Forebay Aquifer), 3-4.05 (Upper Valley Aquifer), 3-4.08 (Seaside Area), 3-4.09 (Langley Area), and 3-4.10 (Corral de Tierra Area) (Figure 1) and cover a total drainage area of 650 square miles. The Paso Robles area of the SV watershed is not included in this study. The SV, as considered here, is entirely within Monterey County (Figure 2).

Its climate features warm, dry summers and cool, moist winters. In Monterey, the average annual temperature is 57˚F and average annual precipitation is 20 inches (mostly during the winter and early spring) (Kulongoski & Belitz 2005). Precipitation in the entire Salinas Valley increases with both latitude and altitude (Kulongoski & Belitz 2005). The Salinas Valley depends almost entirely on local groundwater for all water supplies, and the SV supports one of the most productive agricultural industries in California (Monterey County Water Resources Agency 1996). Monterey was fourth among the nation's top agricultural producing counties with a gross production value of $4.03 billion for 2008 (USDA et al. 2008). The Salinas Valley has approximately 200,000 acres of irrigated land with an average production value of $20,150 per irrigated acre.[5]

In 2000, 7% of the population, or approximately 22,600 people, were classified as living in rural areas. The other 300,000 people were classified as living in urban areas.[6]

2.3 Drinking Water Systems

The residential water systems examined in this report include self-supplied households (or domestic wells), local small water systems, SSWSs, and CPWSs (Table 1). A self-supplied water system is not shown in Table 1, but is a system served by a single domestic well and is considered a small water system. Water system definitions and regulations are further discussed in Section 3.1 Drinking Water Supply Systems.

---

[4]The total population living in the Tulare Lake Basin is based on summarizing population values listed in the 2000 US Census blocks (www.census.gov). This is an overestimation of the total population within the study area because blocks extend beyond study area boundaries.

[5] California Department of Water Resources, Land Use Classification: Irrigated acre totals.

[6] This designation uses 2000 census blocks. It follows the 2000 US Census method of defining urban versus rural on the basis of population density: an urbanized area or an urban cluster consists of: 1) core census block groups or blocks that have a population density of at least 1,000 people per square mile; and 2) surrounding census blocks that have an overall density of at least 500 people per square mile (US Census Bureau N.D.).

Adminsitrative Record
Page 35485

Table 1. Drinking water system connections and service duration.

<table>
<tr><td></td><td colspan="2">Connections:</td><td><5</td><td>5+</td><td><15</td><td>15+</td><td><200</td><td>200+</td></tr>
<tr><td>Duration of Service</td><td colspan="2">Persons served:</td><td colspan="3"><25</td><td colspan="3">25+</td></tr>
<tr><td>N/A</td><td>Small Water System (SWS)[1]</td><td rowspan="4">Classification defined by</td><td colspan="5">connections</td><td></td></tr>
<tr><td rowspan="2"><60 days/year</td><td>Local Small Water System (LSWS)</td><td>connections and (persons, duration)</td><td></td><td colspan="3"></td></tr>
<tr><td>State small Water System (SSWS)</td><td></td><td>connections and (persons, duration)</td><td colspan="3"></td></tr>
<tr><td>year-round</td><td>Community Public Water System (CPWS)[2]</td><td colspan="3"></td><td colspan="3">connections or (persons, duration)</td></tr>
</table>

[1] Classification as a SWS does not preclude classification as any of the other types. SWSs may be regulated by CDPH or by LPA.

[2] A CPWS is a system for the provision of water for human consumption that has 15 or more service connections OR regularly serves 25 individuals at least 60 days a year (CDPH 2010 b,c).

Adminsitrative Record
Page 35486

3 Susceptible Water Users

This section examines the existing California drinking water supply systems within the study area and summarizes current threats to groundwater quality in the study area in the context of the established nitrate safe drinking water standard (45 mg/L). A discussion of nitrate susceptible drinking water users is provided, defining system vulnerability and the susceptible population. To identify susceptible water users within the Salinas Valley and the Tulare Lake Basin, all public data pertaining to drinking water systems and water quality were collected and analyzed. Self-supplied households and their domestic well locations were estimated on a land parcel level. The population that relies on domestic wells or local small water systems was deduced from the basin total population and the land parcel level estimates for domestic wells. An overview of the methods and data used for estimating susceptible water users is provided in Section 3.3 Susceptible Water Users Overview.

3.1 Drinking Water Supply Systems

Water system types are defined by the period of water service, the number of people served, and the number of connections. A public water system (PWS) pipes water for human consumption to 15 or more service connections or regularly serves at least 25 people daily for at least 60 days of the year (CDPH 2010b; CDPH 2010c). PWSs include a wide range of system types, both residential and non-residential. A CPWS is a PWS that serves at least 15 residential connections all year or regularly serves at least 25 residents all year (CDPH 2010b; CDPH 2010c). In addition to CPWSs, PWSs include non-transient non-community (NTNC) and transient non-community (TNC) systems. A NTNC PWS serves drinking water to a stable non-residential population of more than 25 people; these systems are often schools and places of business (CDPH 2010b; CDPH 2010c). A TNC PWS serves areas such as campgrounds or restaurants that serve a changing population of 25 or more people, 60 or more days per year (CDPH 2010b; CDPH 2010c). A SSWS is not a PWS and pipes water to five to fourteen connections, and does not regularly serve drinking water to more than an average of 25 people daily for at least 60 days of the year (CDPH 2010b; CDPH 2010c). Systems with two to four service connections are referred to as local small water systems in some counties or as self-supplied households on a domestic well in other counties. Systems with less than two connections and self-supplied households are often referred to together as domestic wells.

Water system regulations depend on water system type. PWSs and CPWSs are state-regulated, SSWSs are county-regulated, and local small and household self-supplied systems are largely unregulated, unless a County ordinance requires well monitoring when well property is sold. Monterey County regulates their local small water systems and requires them to comply with Title 22 of the California Code of Regulations and the Monterey County Code (Monterey County Environmental Health 2011). PWSs and CPWSs are regulated by the state pursuant to Title 22 of the California Code of Regulations and Health and Safety Code Section 131051, et seq. PWSs and CPWSs must also adhere to the National Primary Drinking Water Regulations under the Safe Drinking Water Act.

Adminsitrative Record
Page 35487

Most counties are designated Local Primacy Agencies (LPAs) and are responsible for regulating community public water systems with up to 199 connections. Tulare County, Kings County, and Monterey County are all LPAs (CDPH 2011b; CDPH 2011c). Fresno County relinquished LPA authority in 2007. LPAs are also responsible for regulation or oversight of SSWSs (five to fourteen connections). County-regulated systems serving five to fourteen connections are not explicitly covered by the Safe Drinking Water Act, but they may still be required to treat by a variety of other contractual or development permit terms, local/county ordinances, or anti-pollution laws. However, monitoring or procedures to implement these requirements may not be in place. County-regulated water systems are subject to tort law[7] if they fail to protect the water delivered to consumers. Public water systems are also subject to tort law if they fail to protect the water delivered to consumers.

3.2 Water Quality Threats

Once the water quality of an aquifer has been degraded, the aquifer may no longer be considered a safe drinking water source without treatment; however, all groundwater designated as having municipal or domestic supply (MUN) beneficial use still has that designated use even if the groundwater is contaminated. Threats to groundwater quality can be point or nonpoint source pollution (see Technical Report 2, Viers et al. 2012). Point sources are easier to identify than nonpoint sources because they originate from specific locations, are usually regulated, and are typically discharged from pipes. Nonpoint sources occur from pollutants over a wide area, such as irrigation runoff or infiltration from agriculture. Examples of point sources of nitrate contamination include leaking underground septic systems and discharge from wastewater treatment plants to percolation basins. Nonpoint contamination comes from agriculture, mining, dairies, feedlots, and urban stormwater. Contaminants also enter aquifers directly from surface water, improperly built groundwater wells, and surface water infiltrating through the soil. The primary constituents of concern within California's groundwater are pesticides, nitrate, perchlorate, arsenic, volatile organic compounds (VOCs), microbial agents, and salts (DWR 2003). Contaminated groundwater can also affect the quality of surrounding surface water.

To protect the public from harmful constituents in groundwater and surface water, Congress passed the Safe Drinking Water Act (SDWA) of 1974 to require regular testing of drinking water supplies, set standards for contaminant concentrations, and schedule for development of new standards (U.S. EPA 2011a). The SDWA also requires the Office of Environmental Health Hazard Assessment (OEHHA, within Cal EPA) to adopt Public Health Goals (PHGs), pursuant to California Health and Safety Code Section 116535, based solely on public health considerations (CaEPA 1997). PHGs represent the official level of a contaminant that can be consumed daily for a lifetime without imposing a health risk. The PHGs are based entirely on public health considerations and are used, along with consideration of health, economic cost, and technical feasibility (examined by CDPH), to establish state MCLs. PHGs are developed for chemical contaminants based on the best available toxicological data in the scientific literature.

[7] Tort law is a body of rights, obligations, and remedies that is applied by courts in civil proceedings to provide relief for persons who have suffered harm from the wrongful acts of others (Farlex Inc. 2012).

Adminsitrative Record
Page 35488

The PHG for nitrate is 45 parts per million (ppm), which is equivalent to California's current MCL of 45 mg/L (as NO_3^-). A system becomes legally non-compliant with the nitrate MCL, or "in violation," when there are two successive reporting MCL exceedances or failure to report the results of a follow-up test on an initial reporting MCL exceedance. Water systems that are currently non-compliant with the state MCL must distribute public notifications to all consumers of potential health risks from consumption of their water. When half the MCL is exceeded for nitrate, systems must switch from annual monitoring and reporting to quarterly monitoring and reporting and they must include a health information notice in the consumer confidence report (CCR) discussing public health concerns from consumption of nitrate (CDPH 2008). If a water system exceeds half of the MCL the system must "notify the governing body of the local agency in which users of the drinking water reside" and it is recommended that the systems notify their customers about the occurrence and health concern of consumption of the contaminant (CDPH 2008).

A summary of the state and federal agencies involved in protecting and improving California's drinking water quality appears in Table 2.

Table 2. California's drinking water quality responsibilities.[1]

Department	Key Water Quality Responsibilities
California Department of Public Health (CDPH)	▪ Enforces federal SDWA and state drinking water statues and regulations ▪ Ensures the quality of the state's public drinking water
California Department of Toxic Substances Control	▪ Protects water quality through enforcement, regulation, and pollution prevention
California Office of Environmental Health Hazard Assessment	▪ Performs health-risk assessments related to establishing drinking water standards
California Public Utilities Commission	▪ Ensures reliable service to regulated water utility customers
California State Water Resources Control Board (State Water Board) and California Regional Water Quality Control Boards (Regional Water Board)	▪ Protects the quality of the state's surface water and groundwater for beneficial use

[1] Baass 2011.

3.3 Susceptible Water Users Overview

Susceptible water users are those that could be potentially harmed or affected by consuming drinking water containing contaminants, or by costs related to such contamination. Susceptibility can be classified or defined in a variety of ways. Here, susceptible population is defined in the context of residential consumption of drinking water and the potential or likelihood for that water to have nitrate levels above 45 mg/L as nitrate. The residential users examined in this report are connected to community public, state small, local small, and self-supplied water systems. Previous studies refer to nitrate susceptible population from a human health perspective, such as subpopulations with a history of immunostimulatory conditions or lacking nitrosation inhibitors in the colon (De Roos et al. 2003). In this study susceptible population is defined in terms of consuming nitrate contaminated drinking water and not in terms of specific human health-related conditions as that is outside the scope of our work.

Adminsitrative Record
Page 35489

Balazs (2011) suggested a susceptibility measure based on system water quality and the total number of raw water sources within a community water system. Balazs categorized community public water systems by considering three levels of source water quality: 1) low (< 22.5 mg NO_3^-/L), 2) medium (22.5 mg NO_3^-/L to 44.9 mg NO_3^-/L), or 3) high (≥ 45 mg NO_3^-/L). Balazs then estimated the total population potentially exposed based on the population served by these individual community public water systems (according to CDPH's Permitting Inspection Compliance Monitoring and Enforcement Database (PICME)). For this report, we use a similar approach for defining susceptible water users.

Here, the susceptible population is estimated by examining the water system vulnerability and the recent raw source water and delivered source water (method discussed in Section 10.1.2 Estimating "Delivering" Sources of a System) quality (if available). Specifically we define "susceptible population" as the number of individuals who:

1. are served by a CPWS with multiple drinking water sources that has reported at least one delivered water nitrate record in excess of 45 mg/L in the past five years in CDPH's Water Quality Management Database (WQM), or
2. are served by a CPWS or SSWS with only a single drinking water source that has reported at least one raw water nitrate record in excess of 45 mg/L in the past five years in WQM, or
3. are on domestic wells or local small water systems and located in an area (Thiessen polygon) where shallow groundwater (<300 feet) has exceeded 45 mg/L as nitrate in the past (1989-2010, data from the UC Davis California Spatio-Temporal Information on Nitrate in Groundwater (CASTING) database), or
4. are served by a CPWS or state-documented SSWS (reported in PICME) lacking nitrate water quality data.

Additionally, the Annual Compliance Reports (ACRs) from CDPH were used to find systems in violation of the nitrate MCL from 2004 to 2008, to provide a narrow regulatory violator based estimate of the susceptible population and for comparison with the estimated susceptible population as defined above in item 1, for multiple source CPWSs.

To estimate the population susceptible to nitrate groundwater contamination in the study area, we first categorize the vulnerability of water systems, or system vulnerability. Population susceptibility is then scored as "high", "low", or "unknown" by evaluating historical nitrate water quality data. In other words, the population susceptibility is derived by scoring water system vulnerability and by scoring its respective water quality record.

System vulnerability describes the intrinsic potential for a system to inadvertently deliver drinking water to users with high nitrate levels based on the type of system and based on the number of water sources within the system. System vulnerability is scored as follows:

- Lower system vulnerability is assigned to community public water systems (water systems with > 15 connections) that have more than one source of water (i.e., more than one well) irrespective of whether or not they treat their water to reduce or remove nitrate.
- Higher system vulnerability is assigned to all other water systems (community public water systems with one well, and state small, local small, and household self-supplied water systems).

Adminsitrative Record
Page 35490

- No system vulnerability (to groundwater contamination) is assigned to water systems that are solely supplied by surface water.

Next, the likelihood for a system to encounter adverse drinking water quality conditions (with or without addressing these in the treatment process) is estimated to determine the susceptible population. We examine the water quality history documented for each system, or, if that information is not available, the historical ambient groundwater quality in the vicinity of each source or system. Given the system vulnerability and its water quality history, the susceptible population served by the system is ranked as:

- Low susceptibility if there has been no recent nitrate record in excess of 45 mg/L documented delivered water for multiple source systems, in documented raw water for single source systems, or in ambient groundwater in the vicinity of the source for undocumented systems.
- High susceptibility if there has been at least one recent nitrate record in excess of 45 mg/L in documented delivered water for multiple source systems, in documented raw water for single source systems, or in ambient groundwater in the vicinity of a source for undocumented systems.
- Unknown susceptibility, if a community public water system has no nitrate water quality data available.

The highly susceptible population in this study is considered to be the estimated population served by systems ranked as being of high or unknown susceptibility. The rest of the population is considered to be of low susceptibility to nitrate contamination in groundwater.

The water system populations within the delineated Tulare Lake Basin and Salinas Valley pilot study areas were estimated combining a variety of information sources, including the 2010 Census, California Department of Finance 2010 population estimates, each county's Local Agency Formation Commission, and CDPH PICME data. Once the total pilot area study population was estimated, the PICME population numbers for community public and state small water systems located in each basin were recorded and summed, and the difference used to estimate the remaining population and number of households supplied by domestic wells and local small water systems. This estimation method is inherently imprecise as to absolute populations due to data limitations and inconsistencies, including data coverage (i.e., Census block groups versus county boundaries), population values listed in PICME that are rounded up and tend to exceed the actual population served, and systems within the study area boundaries that may serve households just outside.

Details and discussion of the approach, data and results for classifying system vulnerability and susceptible population, follow in Section 3.4 Water System Infrastructure and Section 3.5 Susceptible Drinking Water Users. These sections include assumptions and methods used to estimate the various categories of classified populations. The final results from the analysis of the population susceptible to nitrate contamination in the pilot study area and each basin are presented in Section 3.6 Major Findings on Susceptible Water Users.

Adminsitrative Record
Page 35491

3.4 Water System Infrastructure

Water system vulnerability is based on a system's ability to protect against nitrate contamination. The system vulnerability classification describes the potential for delivering high nitrate water to users and is a function of system type. All households in the study area are categorized into four types of residential drinking water supply systems: household self-supplied, local small, state small, or community public water system. A household self-supplied, local or state small water system (not already in PICME) has higher vulnerability since they lack multiple sources. A community public water system with multiple sources has less vulnerability and a system using only surface water has no vulnerability to nitrate in groundwater. The CDPH WQM and PICME database provided all community public water system data and some state small water system information to identify system type, locate sources, and determine nitrate levels in raw and distributed water for the vulnerability and susceptibility assessments. The domestic wells were located based on the method discussed in Section 3.4.2.1 Household Self-Supplied or Local Small Water Systems.

In most counties, state small water systems receive little monitoring or regulatory attention, and are typically considerably more vulnerable to ambient pollution than are CPWSs. CPWSs must adhere to the state MCLs for all drinking water contaminants, so households on these systems should have less vulnerability to nitrate contamination. However, community public water systems having only one well have the potential to be more vulnerable since blending cannot be used as a relatively inexpensive solution.

Lower vulnerability is assigned to regulated CPWSs that have more than one well and the opportunity to blend. Systems that rely completely on surface water have no vulnerability to delivering groundwater contaminated with nitrate, though they may be vulnerable to other pollutants.

This report only addresses system vulnerability from community public water systems or water systems that directly serve residences. It is assumed that the non-community systems adequately warn their users if nitrate contamination is a concern; since users are not permanent, we here assume that they are generally able to either avoid use or provide themselves with safe drinking water from another source. Approximately 382 non-transient, non-community public water systems serve about 190,000 people in the study area. These 382 systems are non-residential and serve the same people for at least 6 months, such as schools and businesses. Approximately 318 transient non-community public water systems in the study area serve about 150,000 people. These 318 systems are non-residential and serve a changing population for at least 60 days per year, such as restaurants, hotels, stores and campgrounds.

According to PICME, 401 active community public and state small water systems exist in the study area regions (281 in the Tulare Lake Basin and 120 in the Salinas Valley). These systems supply water to about 2.4 million people. The 371 CPWSs are supplied by 3,829 sources and the 30 state-documented (listed in PICME) SSWSs supplied by 31 sources. Of the 3,860 sources overall, 3,682 are groundwater; the remaining 178 are surface water. The state small water systems in PICME do not account for all state small water systems in the study area. The 30 state small systems were included in PICME as part

Adminsitrative Record
Page 35492

of Assembly bill 1403 and are further referred to as state-documented state small water systems (CDPH 2011). The locations and information on any other state small water systems, not contained in PICME, could not be identified, and thus the population served by these systems was not considered further.

Figure 3 breaks down the number of state small and community public water systems by their U.S. EPA size categories,[8] in PICME and the study area. The Tulare Lake Basin has 8 state small and 172 community public water systems serving very small systems (< 501 people), and 47 community public water systems serving small systems (501 to 3,300 people). About 81% of Tulare Lake Basin water systems (CPWSs and state-documented SSWSs) are very small or small and serve 89,125 people (4% of the Tulare Lake Basin population). The Salinas Valley has 22 state small and 73 community public water systems serving very small systems, and 11 community public water systems serving small systems. About 89% of the Salinas Valley water systems are very small or small and serve 23,215 people (6% of the Salinas Valley population). Figure 4 and Figure 5 show the number of PICME state small and community public water systems treating or blending raw water within each U.S. EPA size category in the Tulare Lake Basin and Salinas Valley.

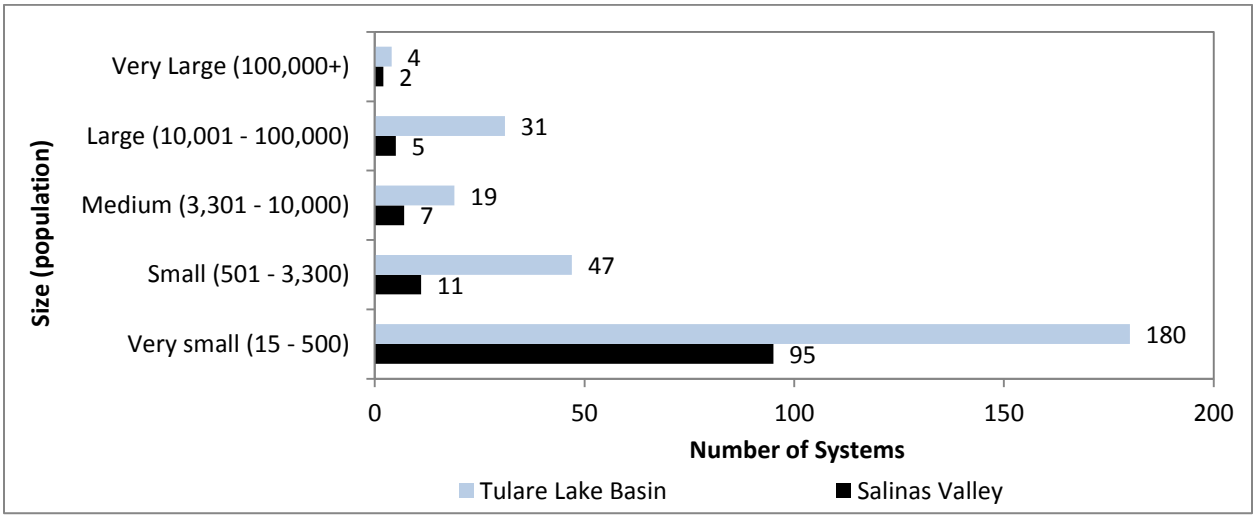

Figure 3. The size distribution (by population served) for state small (state-documented, listed in PICME) and community public water systems in the study area. (Source: CDPH PICME.)

[8] USEPA system size definitions are: (1) very small serves 25-500 people; (2) small serves 501-3,300 people; (3) medium serves 3,301-10,000 people; (4) large serves 10,001-100,000 people; and very large serves greater than 100,000 people (U.S. EPA 2010). Available at: http://water.epa.gov/infrastructure/drinkingwater/pws/factoids.cfm.
Very small in this graph includes some of the SSWSs so the population ranges from 15-500.

Adminsitrative Record
Page 35493

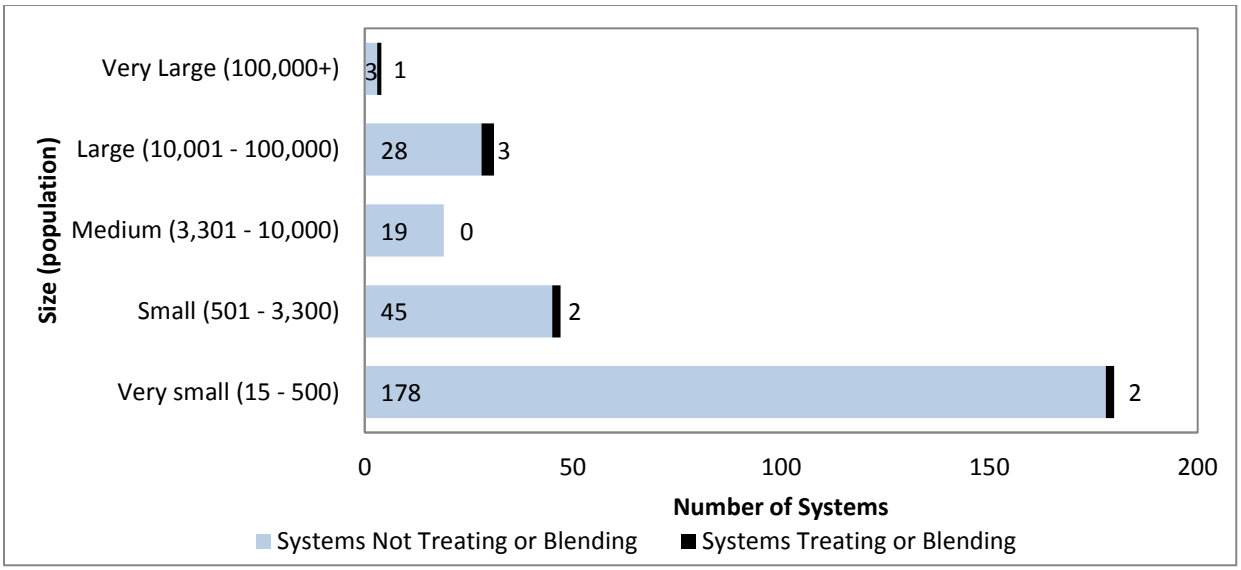

Figure 4. State small (state-documented) and community public water systems treating or not treating for nitrate in the Tulare Lake Basin. (Source: CDPH PICME and Technical Report 6, Section 5.2, Jensen et al. 2012.)

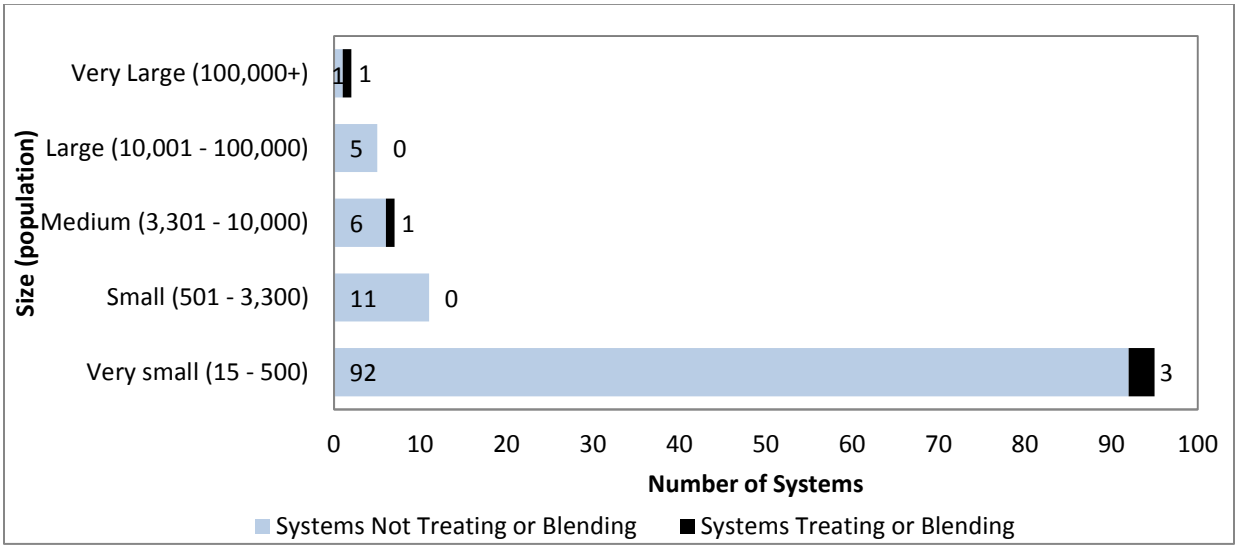

Figure 5. State small (state-documented) and community public water systems treating or not treating for nitrate in the Salinas Valley. (Source: CDPH PICME and Technical Report 6, Section 5.2, Jensen et al. 2012.)

3.4.1 Low System Vulnerability

Theoretically, a CPWS should not deliver water exceeding the nitrate MCL since they must adhere to the SDWA standards (see Section 3.5.1 Highly Susceptible). Where possible, high-nitrate sources can be blended with low-nitrate sources to reduce delivered nitrate levels to a compliant level (although daily monitoring and operations may not always identify an exceedance). Because of the strict regulations and guidelines and the availability of alternate sources, CPWSs with more than one well are considered to have lower system vulnerability.

Adminsitrative Record
Page 35494

The lower vulnerability CPWSs with more than one well include both systems treating for nitrate and not treating for nitrate. Thirteen water systems in the study area treat for nitrate (eight in TLB and five in SV) and eight of these systems treat[9] by blending with lower-nitrate sources (five in TLB and three in SV). Tables of these systems are presented below for the Tulare Lake Basin (Table 3) and the Salinas Valley (Table 4), and their size distributions are shown in Figure 4 and Figure 5, respectively. These systems were identified from inventories of counties in our study area, CDPH, and responses to *The Survey of Nitrate Treatment Systems* discussed in the Drinking Water Treatment Technical Report (See Technical Report 6, Section 5.2, Jensen et al. 2012).

Table 3. Community public water systems treating for nitrate in the Tulare Lake Basin.

UCD System Identifier	Treatment Type or Blending	Number of Sources	Connections	Population
1	Blending	4	81	230
2	Blending	29	7,406	25,500
3	Ion Exchange	15	2,220	12,138
4	Blending	36	7,035	26,860
5	Ion Exchange	2	44	139
6	Blending	8	471	1,904
7	Ion Exchange	190	40,530	133,749
8	Blending	10	89	2,567
TULARE LAKE BASIN TOTAL		**294**	**57,876**	**203,087**

Table 4. Community public water systems treating for nitrate in the Salinas Valley.

UCD System Identifier	Treatment Type or Blending	Number of Sources	Connections	Population
1	Blending	5	66	198
2	Blending	5	19	67
3	Blending	5	70	210
4	Ion Exchange	126	25,451	114,840
5	Reverse Osmosis	8	2,069	6,585
SALINAS VALLEY TOTAL		**151**	**27,680**	**121,945**

Of the total 401 active systems, 264 have more than one source (serving 2.3 million people) and have a lower vulnerability to nitrate contamination in groundwater. Of these, 193 systems are in the Tulare Lake Basin and serve 1.9 million people. The remaining 71 are in the Salinas Valley and serve 400,000 people. The susceptibility level of the population served by these lower vulnerability systems is discussed in Section 3.5 Susceptible Drinking Water Users.

[9] Blending is an alternative that does not require a treatment technology; however blending is sometimes categorized as treatment because water systems are required to monitor and operate the blending process as a permitted treatment facility with a certified operator (CDPH 2008; Commandatore & Collins 2011).

Adminsitrative Record
Page 35495

3.4.2 High System Vulnerability

In addition to CPWSs with just one well source, households not served by a state-regulated CPWS, are considered highly vulnerable because county-regulated systems and individual household wells are usually neither monitored nor treated. If the groundwater source for these households experienced an increase in nitrate levels (above existing elevated anthropogenic levels), these households would not be protected from nitrate contamination. The systems with high system vulnerability are:

1) Household Self-Supplied or Local Small Water Systems (see Section 3.4.2.1)
2) Community Public or State small Water Systems with Only One Well (see Section 3.4.2.2).

3.4.2.1 Household Self-Supplied or Local Small Water Systems

The 1990 and 2010 Census spatial data, DWR land use class designation, and land parcel use code information were used to estimate the current 2010 population on household self-supplied or local small water systems and their approximate geographic locations.

Department of Water Resources (DWR) land use class designations and land parcel use codes were the key data sets used in this analysis. The number of dwelling units affiliated with each parcel was used to develop self-supplied household population estimates, while the parcel locations were used to estimate groundwater nitrate quality for residential susceptibility. It was assumed that 3.3 people reside at one dwelling unit, and parcels with four or fewer dwelling units (dus) were considered self-supplied (1-2 dus) or local small (3-4 dus) water systems. Residential parcels within city limits or water system boundaries were excluded from the count of self-supplied households.

Unlike more recent census data, the 1990 Census asked a sample population about their water systems. These data were collected at the household level and summarized in Attribute H23 of the 1990 Census.[10] Census block groups (for which data are reported) tend to be of small area in urban regions, but relatively large in rural regions so land use parcel code data was used in these areas for estimating self-supplied household and local small water system densities. We then compared the 2010 self-supplied household estimates to the 1990 Census block group numbers. The self-supplied and local small water system population found from parcel use codes and DWR land use designation is shown in Table 5.

The estimated location of these household self-supplied and local small water systems is shown in Figure 6, along with the area's community public and state-documented state small water systems from PICME. The total number of domestic wells in the study area portion of Kern County was difficult to accurately estimate as its parcel use code zoning differs from the other counties. In Kern County more parcels are zoned as having 100 plus dwelling units (i.e., apartment complexes and condominiums) than

[10] Per the 1990 Census definitions, a source that supplies water to five or more housing units is considered a "Public system or private company." This includes any wells that supply water to five or more housing units. If the source serves four or fewer housing units, it is classified as: an "Individual drilled well," an "Individual dug well," or "Some other source." The last distinction, "Some other source," includes springs, creeks, rivers, lakes, cisterns, etc. (US Census Bureau 1999).

Adminsitrative Record
Page 35496

having single dwelling units. Approximately 235,125 people are on self-supplied household or local small water systems in the Tulare Lake Basin and 10,365 in the Salinas Valley (Table 5).

The US Census estimates that 7.6% of residents lived in rural areas in California between 2006 and 2008.[11] Using the counties' rural area definition,[12] an estimated 13% of California residents lived in rural areas in 2009. The self-supplied and local small water system population estimate based on the parcel code and DWR land use data falls within these rural population percentage estimates, accounting for 9.2% of the total study area population. The susceptible population served by these systems is examined in Sections 3.5.1.1 Household Self-Supplied or Local Small Water Systems with a High Likelihood of Nitrate Groundwater Contamination and 3.5.2.1 Household Self-Supplied or Local Small Water Systems with a Low Likelihood of Nitrate Groundwater Contamination.

Table 5. 2010 Estimated self-supplied and local small water system population within the study area based on the parcel use code and DWR land use designation.[1]

Basin	Domestic Wells	Population Served by Domestic Wells
Tulare Lake Basin	71,250	235,125
Salinas Valley	3,141	10,365
STUDY AREA TOTAL	**74,391**	**245,490**

[1] Domestic Wells = Household self-supplied and local small water systems. These are all well systems with fewer than four connections, based on the number of residential dwelling units on a parcel and its location outside of water system and city boundaries.

[11] V. Manual Perez (Chair). Assembly Committee on Jobs, Economic Development and the Economy: *Fast Facts on California Rural Communities* (June 2010). Rural areas contain population densities of less than 500 people per square mile. US Census Bureau. "Census 2000 Urban and Rural Classification".

[12] Counties with 80% or greater rural land mass are generally considered rural.

Adminsitrative Record
Page 35497

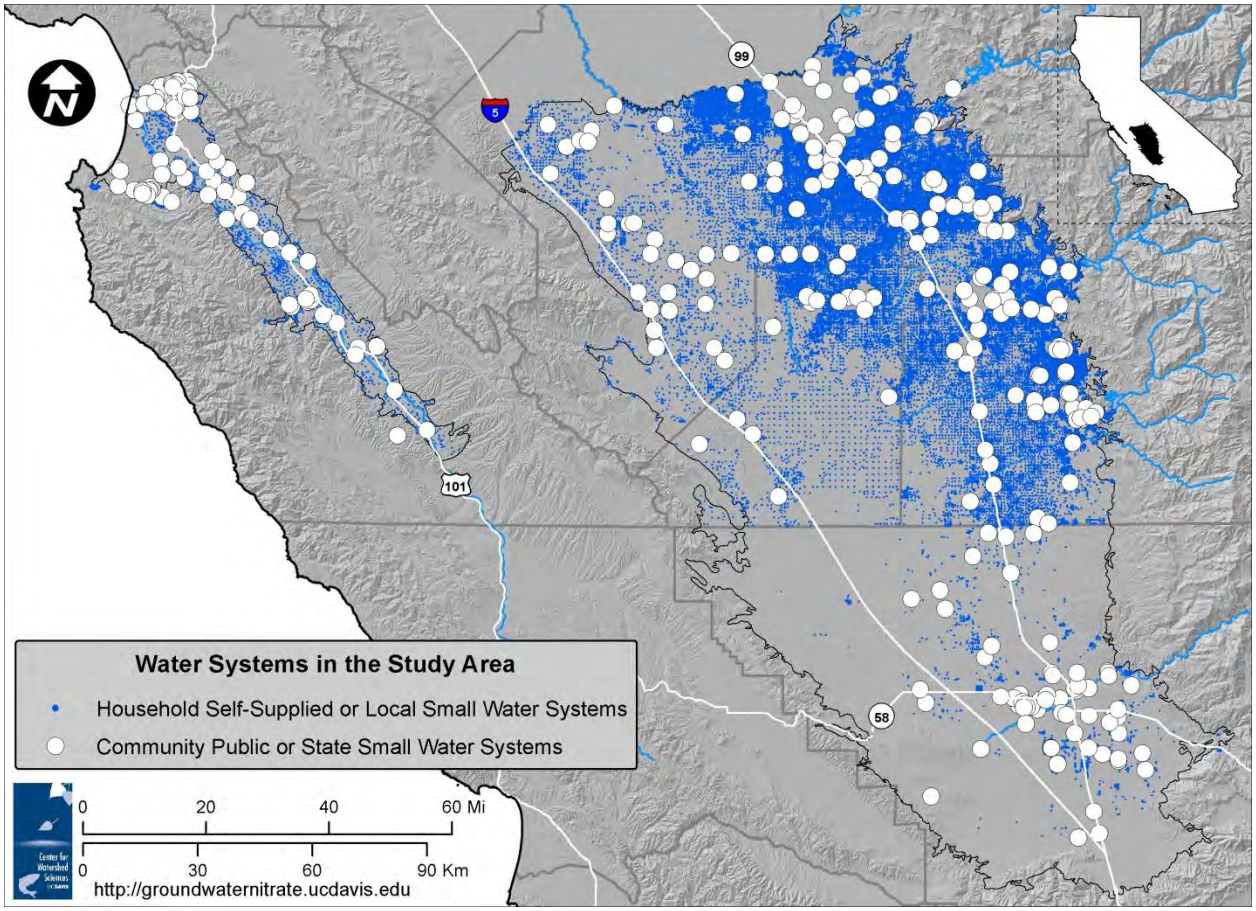

Figure 6. Estimated location of the area's 74,000 unregulated self-supplied water systems and the 402 regulated community public and state-documented state small water systems. (Source: DWR and County Assessor Parcel Land Use Codes and CDPH PICME 2010.)

Since the basin boundaries do not correspond with census or county delineations we have no existing data to effectively compare our domestic well analysis on a basin level, so a comparison on a county level is performed. Table 6 shows the results from applying this land parcel use code method on a county-level, but only regards domestic wells or parcels zoned as having one dwelling unit, located outside of city and water system boundaries. For Fresno, Kings, Monterey and Tulare Counties, the 2010 estimated population is at most one and a half times greater than the 1990 Census block group population. Kern County's 2010 estimate is a little over four times greater than the 1990 Census, which can be attributed to the inclusion of vacant parcels and the lack of water system boundaries outside of the study area and into the outer parts of the county, which would screen out parcels located within existing water systems. Since parcel use codes are from the county assessors they are not a true count of people actually living on the parcel, but are zoning values for distinguishing the number of people that can live there.

Adminsitrative Record
Page 35498

Table 6. 1990 Census v. 2010 Estimated domestic well (single dwelling unit) population by county.

County	1990 Census Block Group Population[1]	2010 Residential Code Population Estimate (Parcel Use Code)[2]
Fresno	110,022	126,968
Kern	40,742	167,274
Kings	15,975	23,354
Monterey	34,528	37,927
Tulare	68,511	91,219

[1] The sum of the 1990 Census Category H0230002 (an "individual drilled well") and H0230003 (an "individual dug well") is the domestic well block group population.
[2] Residential parcels with 1 dwelling unit estimated from County parcel land use codes.

3.4.2.2 Community Public or State Small Water Systems with Only One Well

There are 105 CPWSs or SSWSs in the study area with only one well as a source of drinking water (Figure 7). These systems are classified as having high system vulnerability because they cannot blend with other safe water sources if their source becomes contaminated. The 56 and 49 single source systems in the Tulare Lake Basin and Salinas Valley serve 6,600 and 2,000 people, respectively. Of the 56 one-well systems in the Tulare Lake Basin, 49 are CPWSs and 7 are SSWSs. Of the 49 one-well systems in the Salinas Valley, 27 are CPWSs and 22 are SSWSs.

Adminsitrative Record
Page 35499

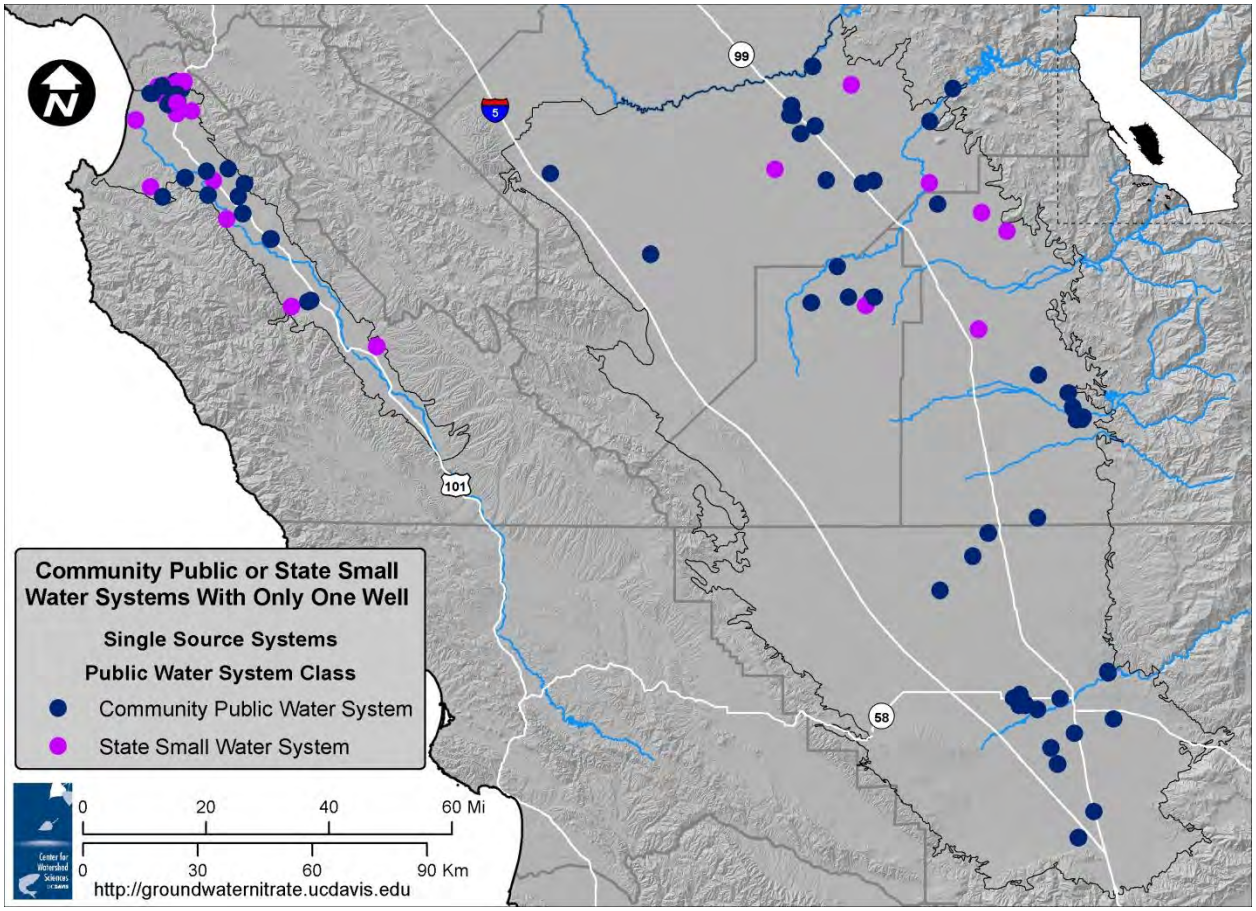

Figure 7. Community public or state small water systems with only one well. (Source: CDPH PICME 2010.)

3.4.3 No System Vulnerability

There are 32 CPWSs in the study area that are recorded in PICME as only having surface water sources (Figure 8). Surface water sources are inherently much less vulnerable to nitrate contamination overall and have essentially no system vulnerability to nitrate contamination in groundwater. The surface water source of most of these 32 systems is the Friant Kern Canal or the Coastal Branch of the California Aqueduct. All 32 surface water systems are in the Tulare Lake Basin.

Adminsitrative Record
Page 35500

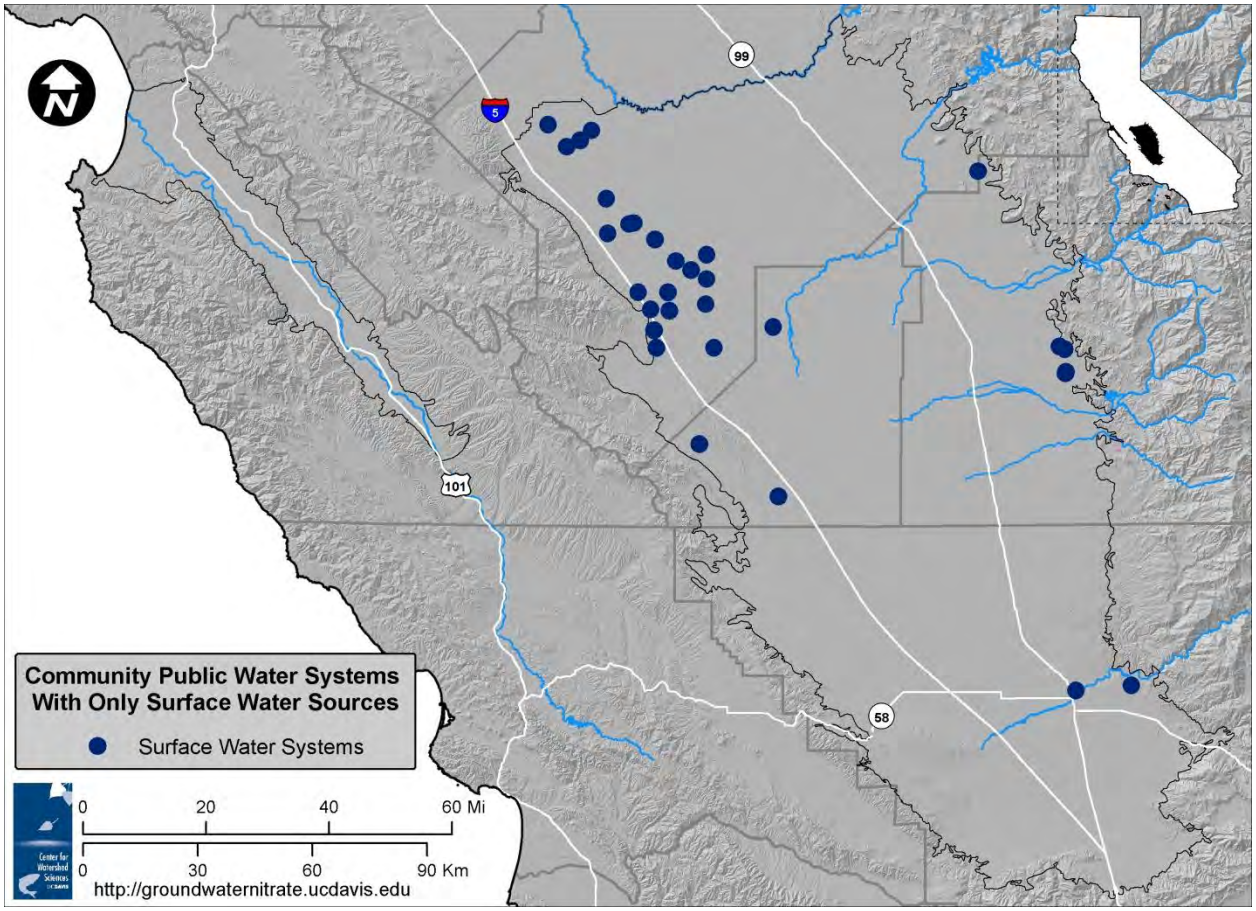

Figure 8. Community public water systems with only surface water sources. (Source: CDPH PICME 2010.)

3.5 Susceptible Drinking Water Users

The level of susceptibility of drinking water users is scored by considering the water system vulnerability and assessing their drinking water quality against a nitrate threshold based on the best historical available water quality information from the PICME WQM database or assembled for this study in the CASTING database (see Technical Report 4, Section 1 and Section 4, Boyle et al. 2012). For all water systems the chosen nitrate threshold for evaluating susceptibility is 45 mg/L. The population susceptibility was scored based on exceeding 45 mg/L as nitrate in delivered drinking water (method discussed in Section 10.1.2 Estimating "Delivering" Sources of Systems), raw source water, or in the case of self-supplied and local small water systems lacking delivered drinking water quality data, in ambient groundwater from historical measurements in the CASTING database from a well in the upper aquifer (depth of less than 300 feet) nearest to a system's estimated source well location.

For community public water systems with low system vulnerability (multiple source community public water systems), the distributed (or delivered) nitrate water quality levels in the PICME WQM database

Adminsitrative Record
Page 35501

were used.[13] If the distributed water for a water system was in excess of 45 mg/L as nitrate at least once from 2006 to 2010, the population served by that water system was classified as having high susceptibility. The population served by systems recorded as distributing drinking water less than 45 mg/L was classified as having low susceptibility. The population served by community or state small water systems with no nitrate water quality data was determined as having an unknown susceptibility, but is included in the total highly susceptible population estimate. For the single source community public and state small water systems (higher vulnerability systems) the raw water quality data from WQM was used to estimate whether or not the system exceeded 45 mg/L as nitrate. For the local small and self-supplied household water systems without nitrate data (higher vulnerability systems), the highest nitrate level in the nearest well (from the CASTING database) was used to estimate whether or not the source exceeded 45 mg/L. The locations of state small water systems that are not accounted for in PICME could not be identified, and so their population of consumers is not considered in this analysis. An estimated 105 state small water systems are located in Tulare and Kern County alone, but there is uncertainty in the total number of state small water systems within the study area.

3.5.1 Highly Susceptible Population

Household self-supplied, local small, state small and community public water systems that have recently exceeded 45 mg/L as nitrate at least once using the relevant water quality data source, are defined as high susceptibility systems. Population considered to be highly susceptible is served by water systems with the following water quality records:

1) Household self-supplied or local small water systems in sub-areas characterized in the CASTING database as having a nitrate concentration in excess of 45 mg/L in shallow (<300 feet) groundwater (see Section 3.5.1.1 Household Self-Supplied or Local Small Water Systems with a High Likelihood of Nitrate Groundwater Contamination)
2) Community public and state small water systems with only one well and that have PICME WQM records of at least one raw source water in excess of 45 mg/L as nitrate since 2006 or lack water quality data (see Section 3.5.1.2 Community Public and State Small Water Systems with Only One Source and Reported Raw Water Nitrate Record in Excess of 45 mg/L or No Water Quality Data)
3) Community public water systems with more than one well and have PICME WQM records of at least one delivered water record in excess of 45 mg/L as nitrate since 2006 (see Section 3.5.1.3 Community Public Water Systems with Reported Delivered Water Nitrate)

For comparison with item 3, community public water systems that have violated the nitrate MCL at least once from 2004 to 2008 are discussed in Section 3.5.1.4 High Susceptibility Community Public Water Systems Evaluated as Violation (versus Exceedance). The difference between an exceedance and a violation is discussed in Section 3.5.1.4 High Susceptibility Community Public Water Systems Evaluated as Violation (versus Exceedance).

[13] The method used for estimating the distributed (or delivered) nitrate water quality levels in the PICME WQM database is discussed further in Section 10.1.2.

Adminsitrative Record
Page 35502

3.5.1.1 Household Self-Supplied or Local Small Water Systems with a High Likelihood of Nitrate Groundwater Contamination

All groundwater wells with nitrate water quality data within the study area were compiled into a comprehensive wells database ("CASTING") that includes nitrate concentrations from 1989 to 2010.[14] Information from the CASTING database was used to evaluate the likelihood of a household self-supplied or local small water system being at risk of nitrate contamination. Each well within the database less than 300 feet[15] in depth was used to geographically seed the creation of a Thiessen polygon or proximal zone. Thiessen polygons represent areas where any location within the polygon is closer to its associated centroid well than any other well (ESRI 2010). Since the true raw source water quality in most of the domestic and local small wells is unknown, the nearest CASTING raw well water quality datum is used to determine whether a self-supplied or local small water system source was likely to be above or below 45 mg/L. The well of a self-supplied or local small system, based on the parcel location, is assumed to have a high likelihood of contamination if the centroid of the parcel is within a Thiessen polygon whose CASTING well nitrate water quality data includes a maximum nitrate concentration value greater than 45 mg/L. The population served by that system is given a high susceptibility rating. Alternatively, a self-supplied or local small water system well is assumed to have a low likelihood of contamination if it is within a Thiessen polygon with CASTING groundwater nitrate concentrations less than or equal to 45 mg/L. The population served by that system is given a low susceptibility rating. This method does not account for the direction of groundwater flow and the actual nitrate concentrations at the true well depth, but provides a reasonable approach to estimate the domestic well and local small water system users potentially at risk of drinking nitrate contaminated water on a geographic basis.

Figure 9 shows the maximum raw source water nitrate concentration from 1989 to 2010 in all wells in the CASTING database less than 300 feet deep. Household self-supplied or local small water systems with a high likelihood of current nitrate groundwater contamination and considered highly susceptible are systems within a Thiessen polygon with a raw water nitrate concentration in excess of 45 mg/L. Figure 10 shows all estimated household self-supplied and local small water systems within high susceptibility Thiessen polygons.

Table 7 provides the highly susceptible population estimated to be served by self-supplied or local small water systems with nitrate in excess of 45 mg/L. Approximately 34,000 people are highly susceptible.

[14] UC Davis CASTING wells database, refer to Technical Report 4 (Boyle et al. 2012) for detailed information on compilation.
[15] Assumed as the average depth for household self-supplied wells.

Adminsitrative Record
Page 35503

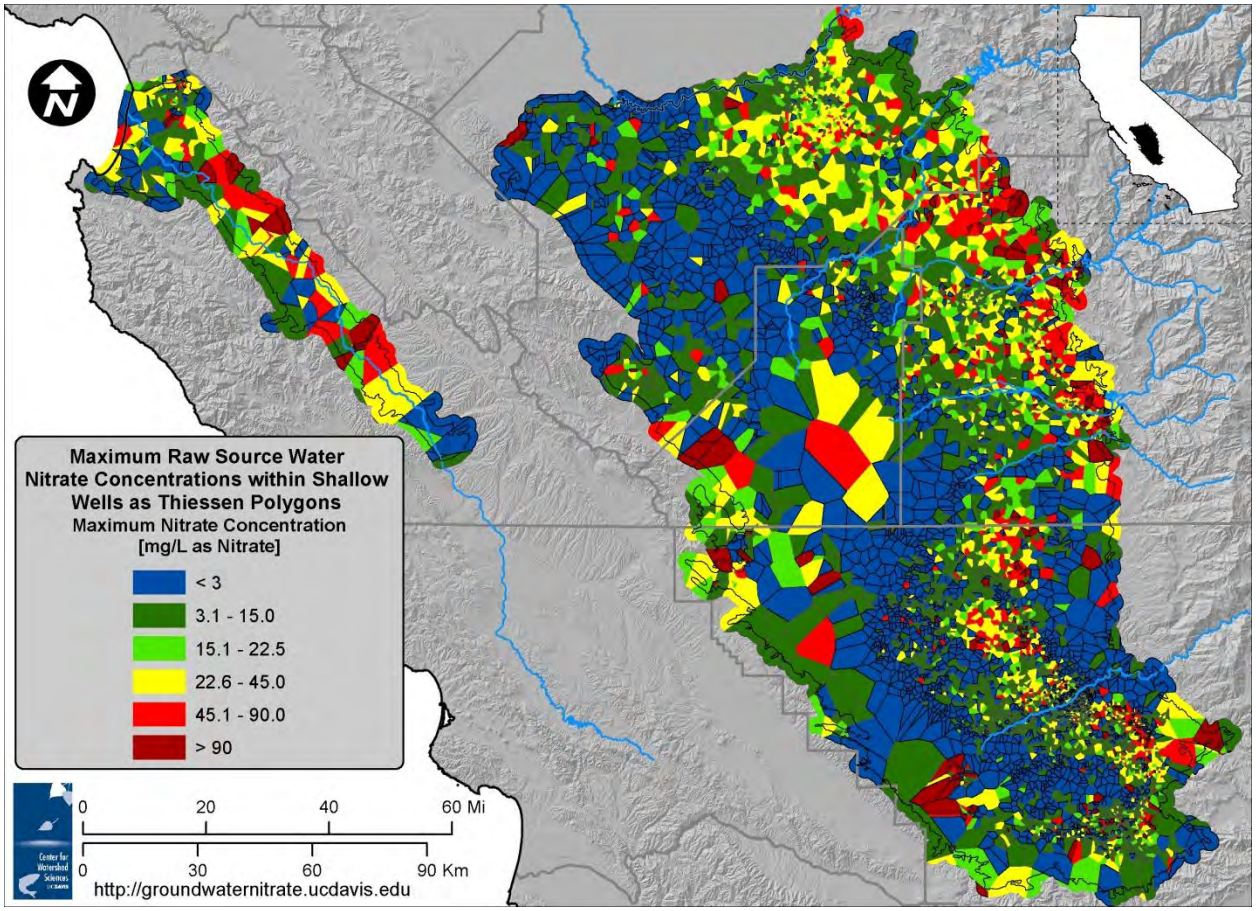

Figure 9. Maximum raw source water nitrate concentration within shallow wells (≤ 300') as Thiessen polygons. (Source: 1989-2010 CASTING Database: GAMA, DWR, SWB, CDPH - CADWSAP, USGS, County Officials, Land Use Parcel Codes and DWR Land Use.)

Adminsitrative Record
Page 35504

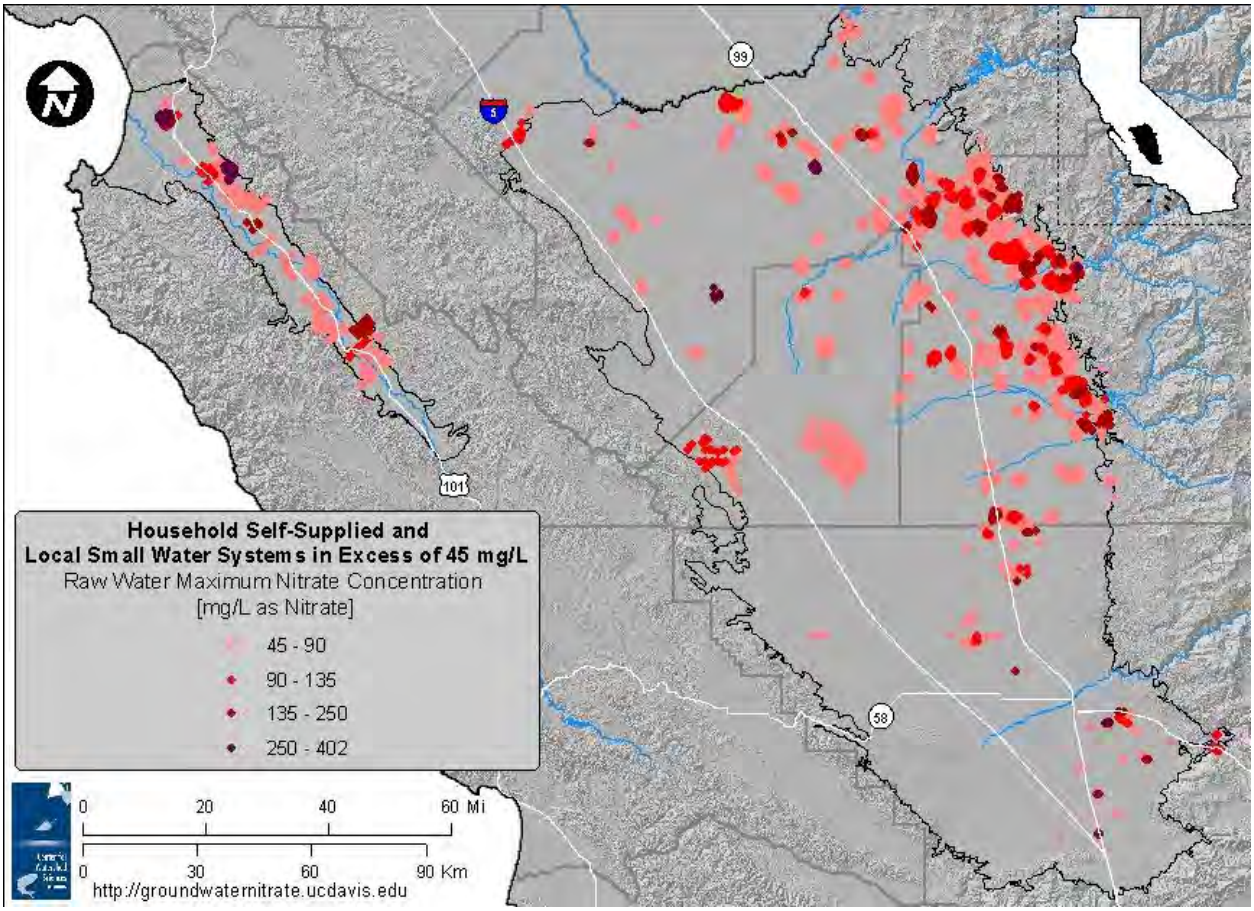

Figure 10. Household self-supplied and local small water systems within a Thiessen polygon having a maximum nitrate concentration value greater than 45 mg/L. (Source: 1989-2010 CASTING Database: GAMA, DWR, SWB, CDPH - CADWSAP, USGS, County Officials, Land Use Parcel Codes and DWR Land Use.)

Table 7. 2010 Estimated high susceptibility population served by self-supplied households and local small water systems (referred to as 'domestic well').

Basin	Population Served by Household Self-Supplied and Local Small Water Systems [1]	High Susceptibility Population Served[2]	% of Household Self-supplied and Local Small Water System Population
Tulare Lake Basin	235,125	32,795	14%
Salinas Valley	10,365	1,294	12%
STUDY AREA TOTAL	**245,490**	**34,089**	--

[1] Household self-supplied and local small water systems estimated from the DWR and parcel use code evaluation. These are all well systems with fewer than five connections, classified as residential dwelling units and located outside of water system and city boundaries.

[2] High susceptibility populations are served by systems with a high likelihood of nitrate contamination, these systems are within a Thiessen polygon that has a maximum raw water nitrate concentration greater than 45 mg/L (as NO_3^-).

Adminsitrative Record
Page 35505

3.5.1.2 Community Public and State Small Water Systems with Only One Source and Reported Raw Water Nitrate Record in Excess of 45 mg/L or No Water Quality Data

The highly susceptible population served by active community public and state small water systems listed in PICME that have only one source is defined as either:

1) have PICME WQM raw source water data in excess of 45 mg/L for nitrate since 2006, or
2) are lacking water quality data.

Of 105 single source systems in the study area, 34 have exceeded 45 mg/L (having a high likelihood of nitrate in groundwater) and are therefore classified as highly susceptible; the population served by these systems is listed in Table 8. If applicable, the highest recorded nitrate measurement per system was used to create conservative estimates. The 3,400 people served by these 34 systems are included in the high susceptibility estimate.

Table 8. Single source systems with a high likelihood of nitrate in groundwater.

Basin	Highly Susceptible Population[1]	Single Source CPWSs or SSWSs that serve Highly Susceptible Population[2]
Tulare Lake Basin	2,424	15
Salinas Valley	1,000	19
STUDY AREA TOTAL	**3,424**	**34**

[1] The highly susceptible population served by single source community public or state small water systems with no nitrate concentration data or systems with raw source water in excess of 45 mg/L (as nitrate) (WQM 2010).
[2] The single source systems that serve the highly susceptible population (PICME 2010).

3.5.1.3 Community Public Water Systems with Reported Delivered Water Nitrate in Excess of 45 mg/L

The maximum nitrate measurement in WQM for each community public and state small water system source in PICME within the study area was mapped. These measurements were taken between January 1st, 2006 and July 13th, 2010. Figure 11 shows a map of WQM raw nitrate data from all sources in the study area and provides an indication of raw nitrate levels in regulated drinking water systems.

To estimate the high susceptibility population served by low vulnerability systems, all active and pending CPWSs and SSWSs (with multiple sources) within CDPH's WQM database were evaluated to determine delivered water nitrate levels. Approximately 15% of the 264 Active/Pending and Community Public/State Small Water Systems (with multiple sources) in the study area delivered water that in excess of 45 mg/L at least once from January 1st, 2006 to July 13th, 2010 (see Figure 12). The method used for identifying the "delivering" source of a system is discussed in the Appendix (Section 10.1.2 Estimating "Delivering" Sources of a System). This includes 39 systems serving 670,000 people (35% of the entire population being served by CPWSs/SSWSs) and suggests potential consumption of water with nitrate levels exceeding the public health standards for an undetermined amount of time. Figure 12

Adminsitrative Record
Page 35506

shows the locations of these exceeding systems. For a system to exceed the 45 mg/L as nitrate, only one sample must be greater than 45 mg/L, versus two consecutive samples in a violation. The difference between exceeding 45 mg/L as nitrate and a CDPH code MCL violation is discussed in Section 3.5.1.4 High Susceptibility Community Public Water Systems Evaluated as Violation (versus Exceedance). This study looks at nitrate trends across the whole region.

Of these 39 systems, three currently blend and one treats with ion exchange (Technical Report 6, Jensen et al. 2012). According to PICME, four other systems are treating, but the type of treatment or reason for treating is not disclosed (i.e., they may be under LPA jurisdiction). Figure 13 shows the system size breakdown (based on established U.S. EPA size categories) for the same systems in Figure 12 that delivered water above 45 mg/L. About 77% of these systems (serving a total of 13,800 people) exceeding 45 mg/L are very small and small systems (serve less than 3,300 people). These smaller systems may find it difficult to comply with the drinking water standards because they lack economies of scale of larger treatment systems and they have a small rate payer base to fund capital expenses (discussed further in Section 4.2.4 Regionalization and Consolidation).

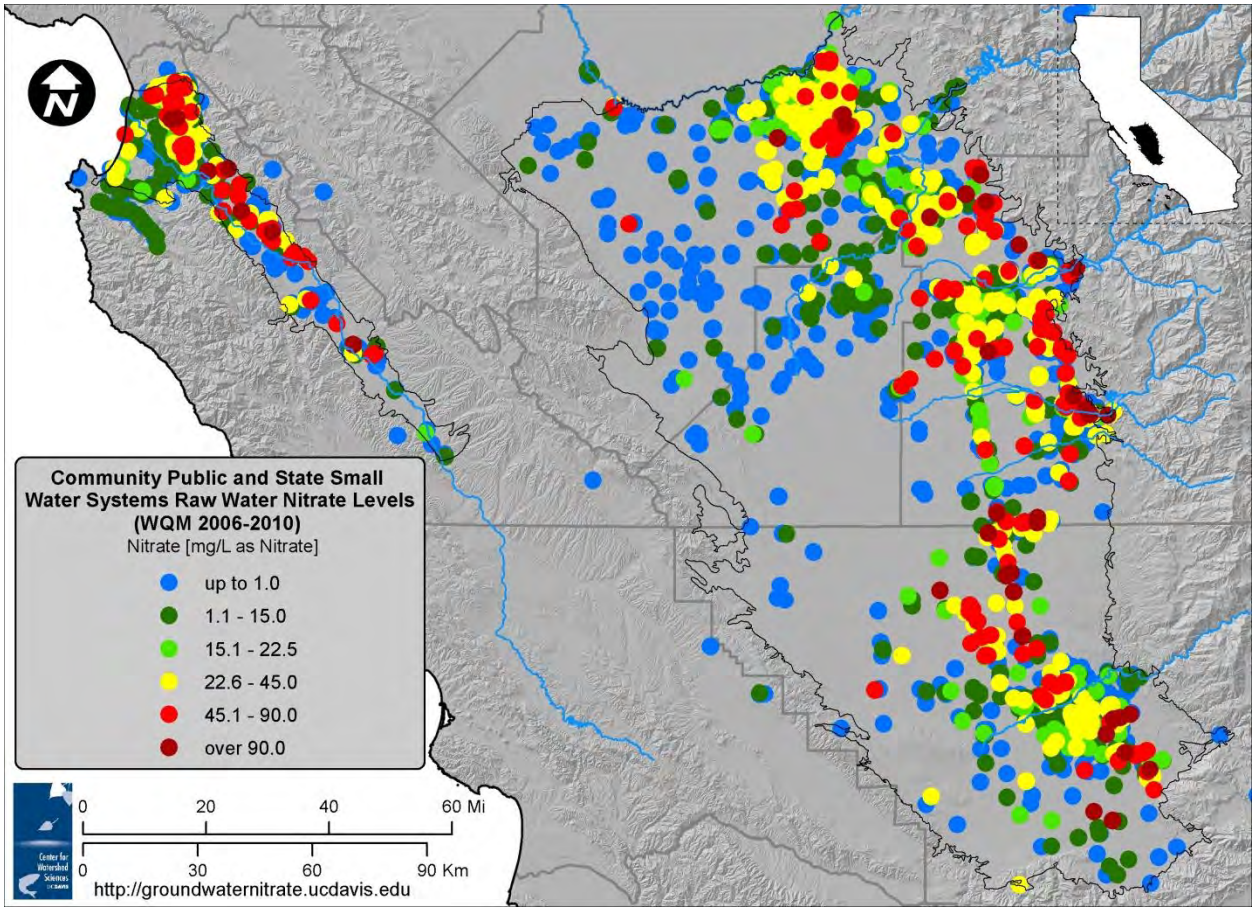

Figure 11. Maximum raw nitrate level records in community public and state small water systems. (Source: CDPH PICME WQM 2006-2010.)

Adminsitrative Record
Page 35507

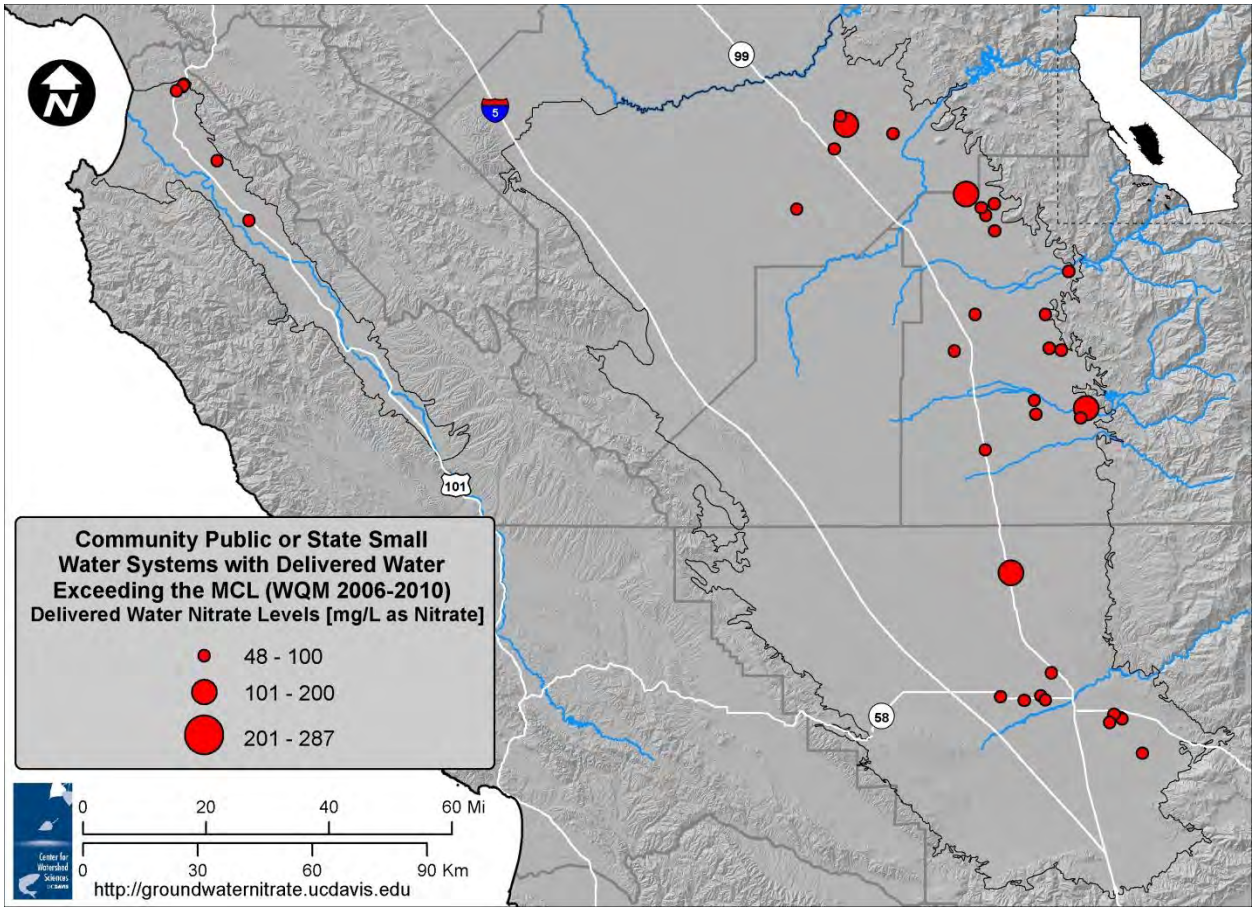

Figure 12. Community public water systems of low vulnerability with delivered water in excess of 45 mg/L as nitrate at least once. (Source: CDPH PICME WQM 2006-2010.) (The highest recorded $NO3^-$ measurement per system is shown.)

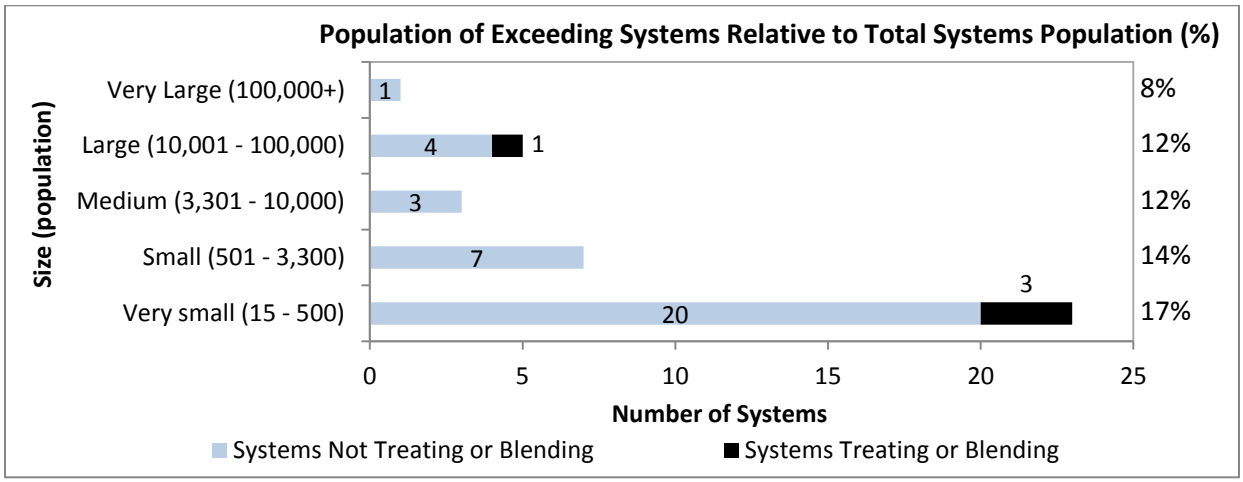

Figure 13. System size distribution (by population served) of the low vulnerability state-documented state small or community public water systems exceeding 45 mg/L as nitrate. (Source: CDPH PICME 2006-2010.)

Within these 39 systems, the City of Fresno's large system was found to exceed the 45 mg/L as nitrate between 2006 and 2010. The City serves approximately 457,000 people served by 250 municipal supply

Adminsitrative Record
Page 35508

groundwater wells and 27.5 mgd (million gallons per day) of surface water. Given the system's large size, we assume that the system has the technical, managerial, and financial capacity and diversification to attend to nitrate contamination in some of their wells, so the population is removed from the multiple-source high susceptibility population total. The City's 2008 Urban Water Management Plan discusses the high nitrate levels found in supply wells (along with other contaminants, i.e., DBCP, EDB, TCP, TCE and PCE) and their plan to mitigate the increasingly poor groundwater quality. As of 2008, the City has constructed wellhead treatment systems and implemented blending plans for several wells as a result of several legal settlements (West Yost Associates 2008). To address the potential increases in groundwater contamination and further decrease their reliance on groundwater supplies, the City plans to expand their surface water treatment capacity by 90 mgd over the next ten years (West Yost Associates 2008). With the increased surface water supplies and treatment capacity the system will be able to decrease groundwater pumpage by 50,000 acre-feet per year (af/year) (West Yost Associates 2008). These factors combine to make the City of Fresno's water system capable of planning for and mitigating future groundwater pollution and the population served is in excess of 45 mg/L. Removing the City of Fresno results in 38 multiple-source systems estimated as high susceptibility systems, serving 213,000 people.

3.5.1.4 High Susceptibility Community Public Water Systems Evaluated as Violation (versus Exceedance)

An alternative to evaluating high susceptibility as the population served by community public water systems with an exceedance is to evaluate high susceptibility based on system violations (per CDPH regulatory language). Systems can sometimes err in reporting contaminant concentrations, and CDPH requires a second lab sample when an MCL is exceeded to verify the accuracy of the original sample. This comparison can only be performed for community public water systems that must submit annual compliance reports (ACRs) to CDPH. The CDPH ACRs were used to identify the CPWSs violating the nitrate MCL. A violation of the nitrate MCL occurs when the MCL is exceeded in two consecutive exceedance reports (CDPH 2008; Commandatore & Collins 2011). When the MCL for nitrate is exceeded once, a secondary, follow-up source sample is required and must be analyzed by an approved CDPH laboratory within 24-hours of notification of the first result. The two results are averaged and if the average exceeds the MCL or if the system fails to collect a confirmation sample, the system is in violation of the nitrate MCL and must contact their regulating agency (the CDPH field office or the local primacy agency) by phone or in writing within 24 hours (CDPH 2008; Commandatore & Collins 2011). The regulating agency then consults with the system to determine the best solution for protecting public health, and the long-term feasibility of complying with the MCL. The regulating agency also helps the system set up a monitoring and reporting schedule to proceed with until deemed necessary. Since the violation of nitrate is a Tier 1 violation, systems must notify customers of the violation within 24 hours and continue communication until the regulator says not to.

It was desired to obtain annual compliance reports for all systems in the Tulare Lake Basin and Salinas Valley from 2006 to 2010 for accurate comparison; however, 2008 is the last year these reports are publicly accessible. To be consistent with the length of record used for the exceedance assessments, a

Adminsitrative Record
Page 35509

five year span evaluation based on ACRs from 2004 to 2008 was used to identify community public water systems violating the MCL. Twenty-six community public water systems (Table 9) violated the nitrate MCL within the Tulare Lake Basin and Salinas Valley, five in Kern, eight in Monterey, thirteen in Tulare, and zero in Kings and Fresno counties. The total population served by violating systems is about 130,000 (Table 9) and is about 83,000 people less than the total population estimated with high susceptibility based on multiple source MCL exceedance systems in Section 3.5.1.3 Community Public Water Systems with Reported Delivered Water Nitrate . This population difference stems from evaluating systems based on an exceedance versus violation. For a system to violate the nitrate MCL, the average of two consecutive samples must be greater than the MCL. For a system to exceed the nitrate MCL, only one sample must be greater than the MCL. CDPH has established the violation definition to avoid any discrepancy in monitoring or reporting at the system or lab level. A conservative approach is taken here in estimating the susceptibility based on an exceedance rather than a violation. The community public water systems in violation of the nitrate MCL are shown in Figure 14, highlighting the total years in violation between 2004 and 2008.

Adminsitrative Record
Page 35510

Table 9. Community public water systems in violation of the nitrate MCL (2004-2008).

County	System Number	System Name	Years in Violation	Population
Kern	1500373	SEVENTH STANDARD MUTUAL	1	110
	1500494	WILSON ROAD WATER COMMUNITY	1	72
	1500544	ENOS LANE PUBLIC UTILITY DISTRICT	1	250
	1500584	GOOSELAKE WATER COMPANY	1	80
	1510001	ARVIN COMMUNITY SERVICES DIST	1	14,500
			KERN TOTAL POPULATION	**15,012**
Monterey	2700665	OAK HEIGHTS W & R CO INC	2	105
	2701036	APPLE AVE WS #03	1	60
	2701904	SAN JERARDO COOP WS	2	249
	2702409	EL CAMINO WC INC	1	90
	2702439	WOODLAND HEIGHTS MWC	1	57
	2702466	SAN VICENTE MWC	1	90
	2710010	CPWSC SALINAS	1	111,135
	2710851	SALINAS VALLEY STATE PRISON	1	5,400
			MONTEREY TOTAL POPULATION	**117,186**
Tulare	5400523	EL MONTE VILLAGE M H P	1	100
	5400550	SEVILLE WATER CO	1	400
	5400567	TOOLEVILLE WATER COMPANY	3	300
	5400616	LEMON COVE WATER CO	4	200
	5400651	BEVERLY GRAND MUTUAL WATER	5	108
	5400663	FAIRWAYS TRACT MUTUAL	5	250
	5400666	WATERTEK - GRANDVIEW GARDENS	1	350
	5400735	RODRIGUEZ LABOR CAMP	3	110
	5400805	SOULTS MUTUAL WATER CO	3	100
	5401003	EAST OROSI CSD	2	106
	5401038	AKIN WATER CO	1	50
	5402047	GLEANINGS FOR THE HUNGRY	5	31
	5403043	YETTEM WATER SYSTEM	2	350
			TULARE TOTAL POPULATION	**2,455**
			STUDY AREA POPULATION VIOLATING THE NITRATE MCL (2004-2008)	**134,653**

Adminsitrative Record
Page 35511

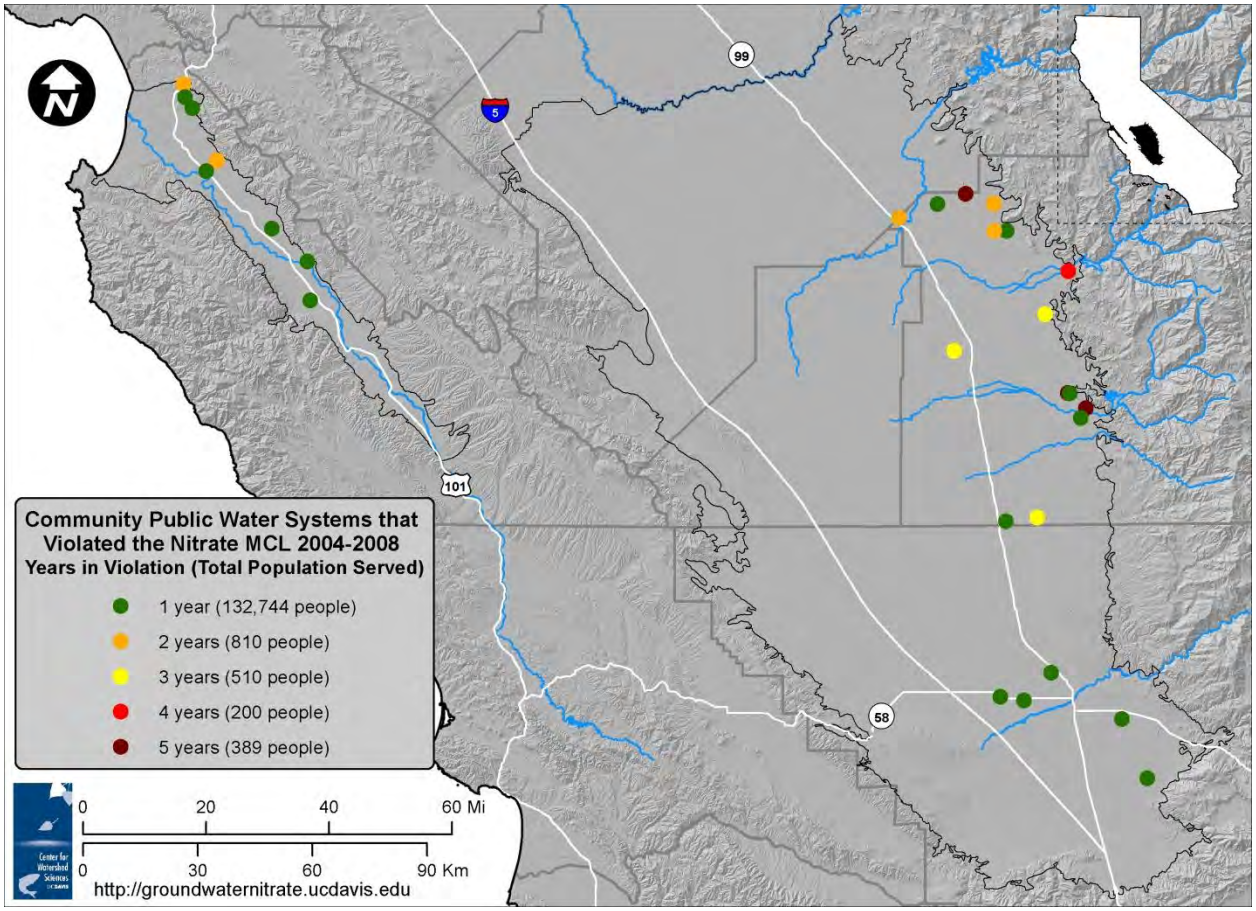

Figure 14. Community public water systems in the study area in violation of the nitrate MCL 2004 to 2008. (Source: CDPH 2010d.)

3.5.2 Low Susceptibility Population

Low susceptibility water users are on systems estimated to have a high vulnerability but with a low likelihood of nitrate contamination in the groundwater, or on systems estimated to have a low vulnerability (CPWSs with more than one well) with no record in excess of 45 mg/L since 2006. We estimate that 1.67 to 1.88 million people in the study area are not currently susceptible to consumption of nitrate-contaminated drinking water within their residences.

3.5.2.1 Household Self-Supplied or Local Small Water Systems with a Low Likelihood of Nitrate Groundwater Contamination

Household self-supplied or local small water systems are assumed to have a low likelihood of current nitrate groundwater contamination when they fall within a Thiessen polygon that has historical raw water nitrate concentrations less than or equal to 45 mg/L. Figure 15 shows all estimated household self-supplied and local small water systems within low-nitrate Thiessen polygons. Table 10 provides population estimates for persons supplied by low susceptibility systems relative to the total basin

Adminsitrative Record
Page 35512

domestic well population. An estimated 212,000 people served by self-supplied household and local small water systems are included in the low susceptibility population estimate.

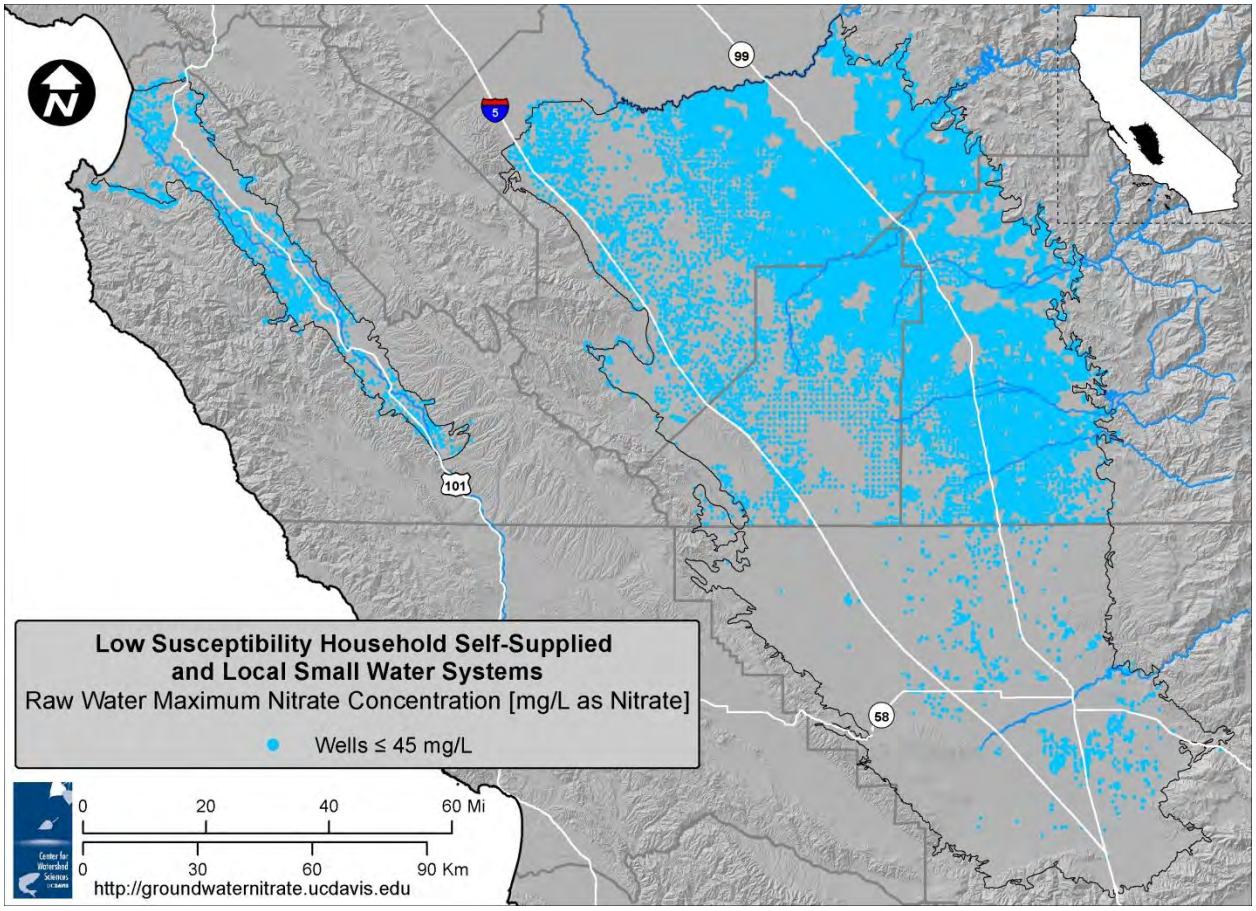

Figure 15. Household self-supplied and local small water system wells within a Thiessen polygon of a tested upper aquifer well (<300 ft) with maximum nitrate concentration less than 45 mg/L. (Source: 1989-2010 CASTING Database: GAMA, DWR, SWB, CDPH - CADWSAP, USGS, County Officials, Land Use Parcel Codes and DWR Land Use.)

Table 10. 2010 Estimated low susceptibility population served by self-supplied household and local small water systems.

Basin	Domestic Well Population[1]	Low Susceptibility Population[2]	% of Domestic Well Population
Tulare Lake Basin	235,125	202,676	86%
Salinas Valley	10,365	9,088	88%
STUDY AREA TOTAL	**245,490**	**211,764**	--

[1] Domestic Wells = Household self-supplied and local small water systems. These are all well systems with fewer than four connections, classified as residential dwelling units and located outside of water system and city boundaries.

[2] The low susceptibility population served by systems with a low likelihood of nitrate contamination. These systems are within a Thiessen polygon that has a maximum raw water nitrate concentration less than or equal to 45 mg/L.

Adminsitrative Record
Page 35513

3.5.2.2 Community Public or State Small Water Systems with Only One Source and No Reported Raw Water Nitrate Record in Excess of 45 mg/L

The active community public and state small water systems in PICME that have only one source (high vulnerability) and have not exceeded 45 mg/L for nitrate since 2006 in their raw source water are considered to have a low likelihood of nitrate in groundwater, and their users are considered to have low susceptibility to nitrate contamination. The raw nitrate levels of these systems are measured against the 45 mg/L as nitrate since the delivered water quality could not be estimated in PICME. Of the 105 single source systems, 71 have a low likelihood of nitrate in groundwater. The low susceptibility population served by these single source systems is given in Table 11 and is included in the low susceptibility estimate.

Table 11. Single source systems with a low likelihood of nitrate in groundwater.

Basin	Low Susceptibility Population[1]	Single Source CPWSs or SSWSs that serve the Low Susceptibility Population[2]
Tulare Lake Basin	4,234	41
Salinas Valley	1,176	30
STUDY AREA TOTAL	**5,410**	**71**

[1] The low susceptibility population served by single source systems with maximum source (raw water) nitrate concentrations less than 45 mg/L (WQM 2010).
[2] Single source community public or state small water systems that serve this low susceptibility population (PICME 2010).

3.5.2.3 Community Public Water Systems with Reported Nitrate Records less than 45 mg/L

There are 212 multiple source CPWSs in the study area with recorded PICME nitrate data having no delivered nitrate records in excess of 45 mg/L from 2006 to 2010. The population of these multiple-source systems is given in Table 12 and is included in the low susceptibility estimate.

Table 12. Community public water systems with a low likelihood of nitrate in groundwater.

Basin	Low Susceptibility Population[1]	Multiple Source CPWSs that serve the Low Susceptibility Population[2]
Tulare Lake Basin	1,259,724	157
Salinas Valley	405,783	55
STUDY AREA TOTAL	**1,665,507**	**212**

[1] The low susceptibility population served by systems with more than one source that has maximum delivered nitrate concentrations less than or equal to 45 mg/L (as nitrate) (WQM 2010).
[2] Community public water systems with more than one source that serve this low susceptibility population (as NO_3^-) (PICME 2010).

Adminsitrative Record
Page 35514

3.5.3 Unknown Susceptibility Population

The WQM dataset is incomplete for 13 multiple source CPWSs that are included in PICME (serving 3,900 people), and are lacking nitrate measurement data. These include three and ten systems in the Tulare Lake Basin and Salinas Valley, serving 2,450 and 1,450 people, respectively. These systems have an unknown water quality level, but are conservatively included in the higher susceptibility water users total for this analysis. This report assumes the lack of nitrate data is from the absence of monitoring and reporting, however, the data could be in the process of being incorporated into WQM. Figure 16 shows the distribution of system sizes for the multiple source water systems with no nitrate measurement data. Eleven of these CPWSs are county-regulated (<200 connections) and may have nitrate data within their respective County health departments.

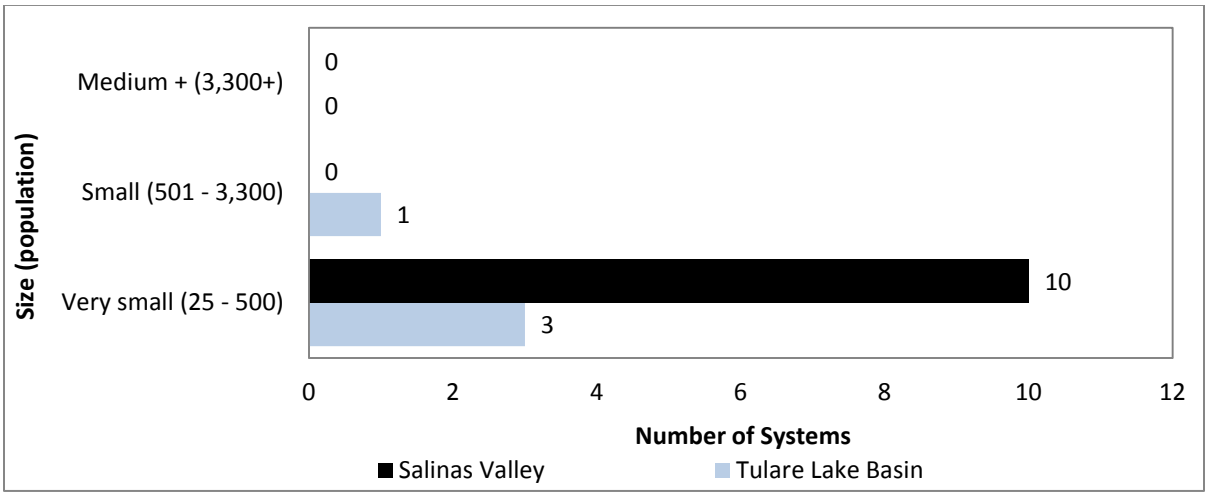

Figure 16. System size distribution (by population served) of the community public water systems without water quality information. (Source: CDPH PICME WQM 2006-2010.)

3.6 Major Findings on Susceptible Water Users

The population connected to each water system type, their system's vulnerability, and the population's susceptibility are summarized below.

- An estimated 246,000 people are served water by self-supplied and local small water systems (about 9% of the total study area population).
  - These systems are considered to have high vulnerability.
  - About 34,000 people, or 14% of the household self-supplied and local small water system population, are within Thiessen polygons with a maximum nitrate concentration in excess of 45 mg/L. This population is considered to have high susceptibility.
  - The remaining 212,000 people are considered to have low nitrate contamination susceptibility.
- Approximately 8,800 people are served by single source state small (listed in PICME) and community public water systems (less than 1% of the study area population).
  - These systems are considered to have high vulnerability.

Adminsitrative Record
Page 35515

    - Approximately, 1,100 people, or 13% of the single source state small and community public water system population, are served by a system with a maximum recorded (raw source water) nitrate level in excess of 45 mg/L; hence, this population is considered highly susceptible to nitrate contamination.
    - Approximately 2,350 people, or 27% of the single source state small and community public water system population, are served by a system having no recorded nitrate data, and so are considered to have high nitrate contamination susceptibility.
    - The remaining 5,400 people are considered to have low susceptibility.
- An estimated 2.3 million people are on multiple source community public water systems (about 88% of the study area population).
    - These systems are considered to be of low vulnerability.
    - Approximately 213,000 people, or 29% of the multiple source community public water system population, have a maximum recorded (delivered water) nitrate level in excess of 45 mg/L, and so have high nitrate contamination susceptibility.
    - Approximately 3,900 people (less than 1% of the multiple source community public water system population), have no recorded nitrate data, and so have high nitrate contamination susceptibility.
    - The remaining 1.7 million people are served by systems with no recorded nitrate levels in excess of 45 mg/L and considered to have low nitrate contamination susceptibility.

A summary of the existing susceptible water systems and the population served, overall and in each basin, is shown in Table 13 and presented visually in Figure 17, Figure 18, and Figure 19. Figure 17 illustrates how we assessed the degree of vulnerability and overall susceptibility for the study area population in year 2010. Figure 18 and Figure 19 show the 2010 population susceptibility assessment for the Tulare Lake Basin and Salinas Valley, respectively.

Approximately 254,000 people have higher susceptibility (about 10% of the study area population, which includes the unknown susceptibility population), 2.34 million have lower susceptibility (90% of the study area population), and 64,000 have no susceptibility (2% of the study area population) to nitrate groundwater contamination.

Finally, an important and cautionary note on the interpretation of the terms “vulnerable” water systems and “susceptible” population. Here, we have defined these terms to provide a tractable scoring system that allows us to estimate the potential for a water system to draw from a nitrate contaminated well and to estimate the magnitude of the population in the study area that may be exposed to nitrate in drinking water from that well. Our analysis does not consider whether the exposure actually occurs. Our analysis also does not consider whether exposure occurs once, occasionally, or regularly. The susceptibility score is merely a qualitative measure of the likelihood that a person is exposed to drinking nitrate contaminated water at the tap at least once, but potentially more than once. We do not know how many people in either the high susceptible population or in the low susceptible population actually drink water that is in excess of 45 mg/L, or how often they do so.

To improve upon our very general susceptibility measure, water quality data would need to be collected on a regular basis from all sources of drinking water, including all local and state small water systems, and all household self-supplied systems (domestic wells). Without these data, a scoring system such as

Adminsitrative Record
Page 35516

the one presented here, provides the best possible practical representation of our current knowledge and available information on nitrate exposure in drinking water in the pilot study area.

Table 13. Assessment of susceptible water users in the study area.

		Population Served		
System Description	**Susc.[1]**	**Salinas Valley**	**Tulare Lake Basin**	**Total Study Area**
Total Basin Population[2]		**397,287**	**2,249,928**	**2,647,215**
Household Self-Supplied or Local Small Water Systems[3]		**10,365**	**235,125**	**245,490**
Max NO_3 45 mg/L Exceedance[3a]	H	1,294	32,795	34,089
Max NO_3 45 mg/L Non-Exceedance[3b]	L	9,088	202,676	211,764
Single Source State small or Community Public Water Systems[4]		**2,176**	**6,658**	**8,834**
Max NO_3 45 mg/L Exceedance or No WQM Data[4a]	H	1,000	2,424	3,424
Max NO_3 45 mg/L Non-Exceedance[4b]	L	1,176	4,234	5,410
Surface Water Community Public Water Systems[5]	No	0	64,501	64,501
Multiple Source Community Public Water Systems[6]		**408,123**	**1,931,267**	**2,339,390**
Treating or Blending for NO_3[7]		121,945	203,087	325,032
Not Treating or Blending for NO_3[8]		286,178	1,728,180	2,014,358
Max NO_3 Distributed Water 45 mg/L Exceedance[9]	H	894	669,101	669,995
Max NO_3 Distributed Water 45 mg/L Non-Exceedance[10]	L	405,783	1,259,724	1,665,507
No NO_3 Data[11]	H	1,446	2,442	3,888
TOTAL HIGHER SUSCEPTIBILITY POPULATION[12]		4,634	249,251	253,885
TOTAL LOWER SUSCEPTIBILITY POPULATION[13]		416,047	1,924,145	2,340,192
TOTAL NO SUSCEPTIBILITY POPULATION[14]		0	64,501	64,501

[1] Susceptibility – Levels: High (H), Low (L), and No.

[2] The total basin population. Estimated from US Census and California Department of Finance data, spatially verified in ArcGIS.

[3] Population on household self-supplied and local small water systems estimated using Parcel Use Codes from County Assessors and DWR land use classification. Household Self-Supplied Water Systems are any residential parcels zoned as having 1-2 dwelling units, located outside of city and water system boundaries. Local Small Water Systems are any residential parcels zoned as having 3-4 dwelling units, located outside of city and water system boundaries. Assumed 3.3 people per dwelling unit.

[3a] Population on household self-supplied and local small water systems located within Thiessen polygons that have a nitrate concentration greater than 45 mg/L (as NO_3). Nitrate concentrations are from DWR, CDPH, USGS, SWB, and all study area counties.

[3b] Population on household self-supplied and local small water systems that are located within Thiessen polygons that have a nitrate concentration less than 45 mg/L (as NO_3).

[4] Population on single source SSWS or CPWS from CDPH's PICME database.

[4a] Population on single source SSWS or CPWS with a maximum raw water nitrate concentration (PICME WQM, 2006-2010) greater than 45 mg/L (as NO_3) or without nitrate data in WQM.

[4b] Population on single source SSWS or CPWS with a maximum raw water nitrate concentration (PICME WQM, 2006-2010) less than 45 mg/L (as NO_3).

[5] Population on CPWSs serving only surface water sources (PICME 2006-2010).

[6] Population on CPWSs with more than one source (PICME 2006-2010).

[7] Population on CPWSs with more than one source treating or blending for nitrate (Technical Report 6, Jensen et al. 2012: Nitrate Treatment Systems Survey and systems approved for treatment by CDPH).

[8] Population on CPWSs with more than one source not treating or blending for nitrate, those systems that did not

Adminsitrative Record
Page 35517

respond to the Nitrate Treatment Systems Survey and CDPH does not have treatment information on.
[9] Population on CPWSs with more than one source with delivered water in excess of 45 mg/L (as NO_3) (PICME WQM 2006-2010).
[10] Population on CPWSs with more than one source with delivered water less than the 45 mg/L (as NO_3) (PICME WQM 2006-2010).
[11] Population on community public water systems with more than one source with no nitrate water quality data (PICME WQM 2006-2010).
[12] Total Higher Susceptibility Population = 3a + 4a + 9 + 11 (Excluding the City of Fresno's water system that serves 457,511 people).
[13] Total Lower Susceptibility Population = 3a + 4a + 10 (Including the City of Fresno's water system that serves 457,511 people).
[14] Total No Susceptibility Population = 5.

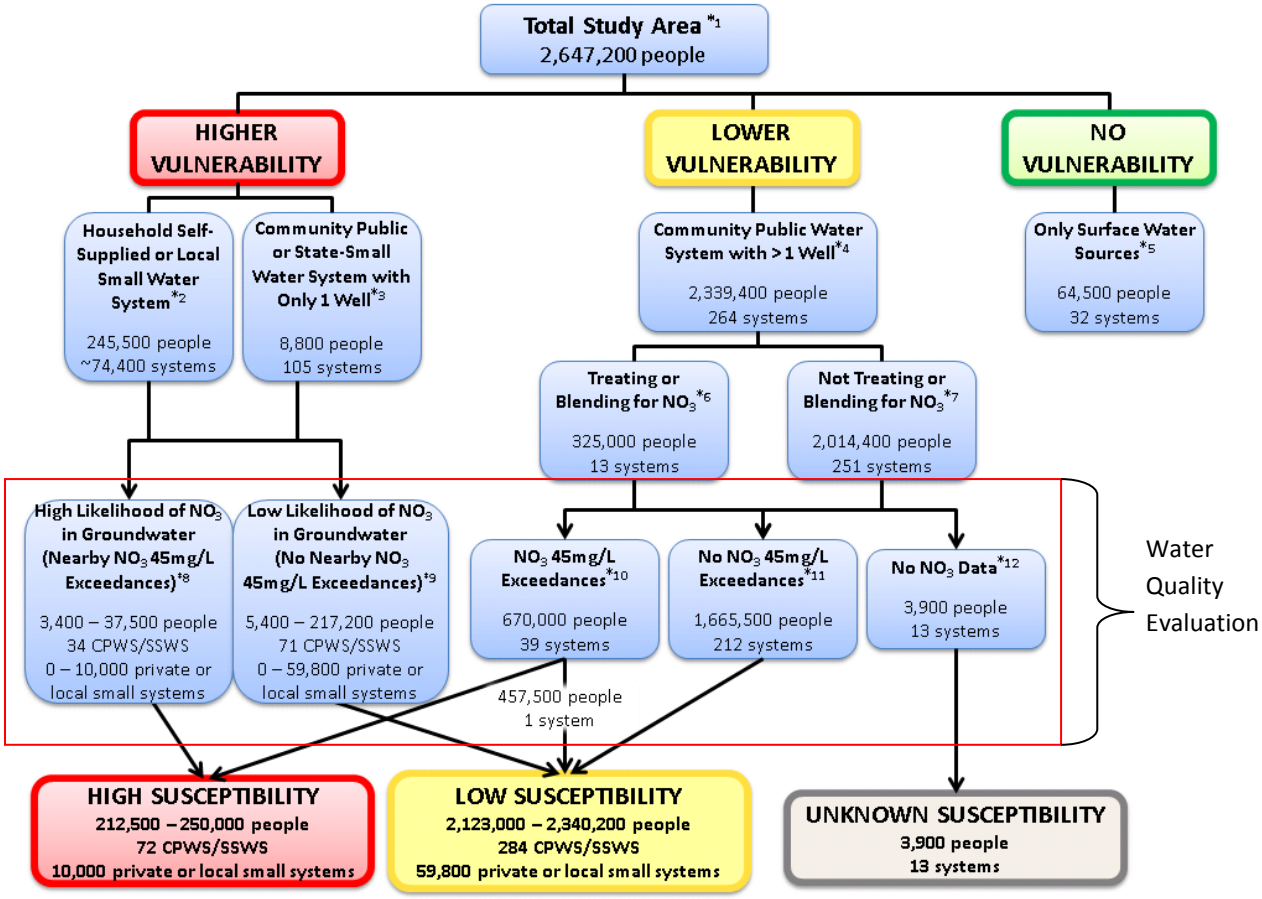

Figure 17. Classification of susceptible populations based on estimated vulnerability and water quality data for the study area.[16]

[16] Due to different sources of data, the summation of the top row does not exactly equal the total study area population. All population and connection information is approximate. The methodology used for creating the susceptibility chart and chart footnotes are further discussed in the Appendix (Section 10.1.1 Susceptibility Charts). Table 13 includes the unknown susceptibility as the highly susceptible population.

Adminsitrative Record
Page 35518

As discussed in Section 3.5.1.3 Community Public Water Systems with Reported Delivered Water Nitrate even though the City of Fresno was detected as delivering drinking water in excess of 45 mg/L they are considered as having low susceptibility to nitrate contamination in groundwater. The methodology used for creating the susceptibility chart and chart footnotes are further discussed in the Appendix (Section 10.1.1 Susceptibility Charts).

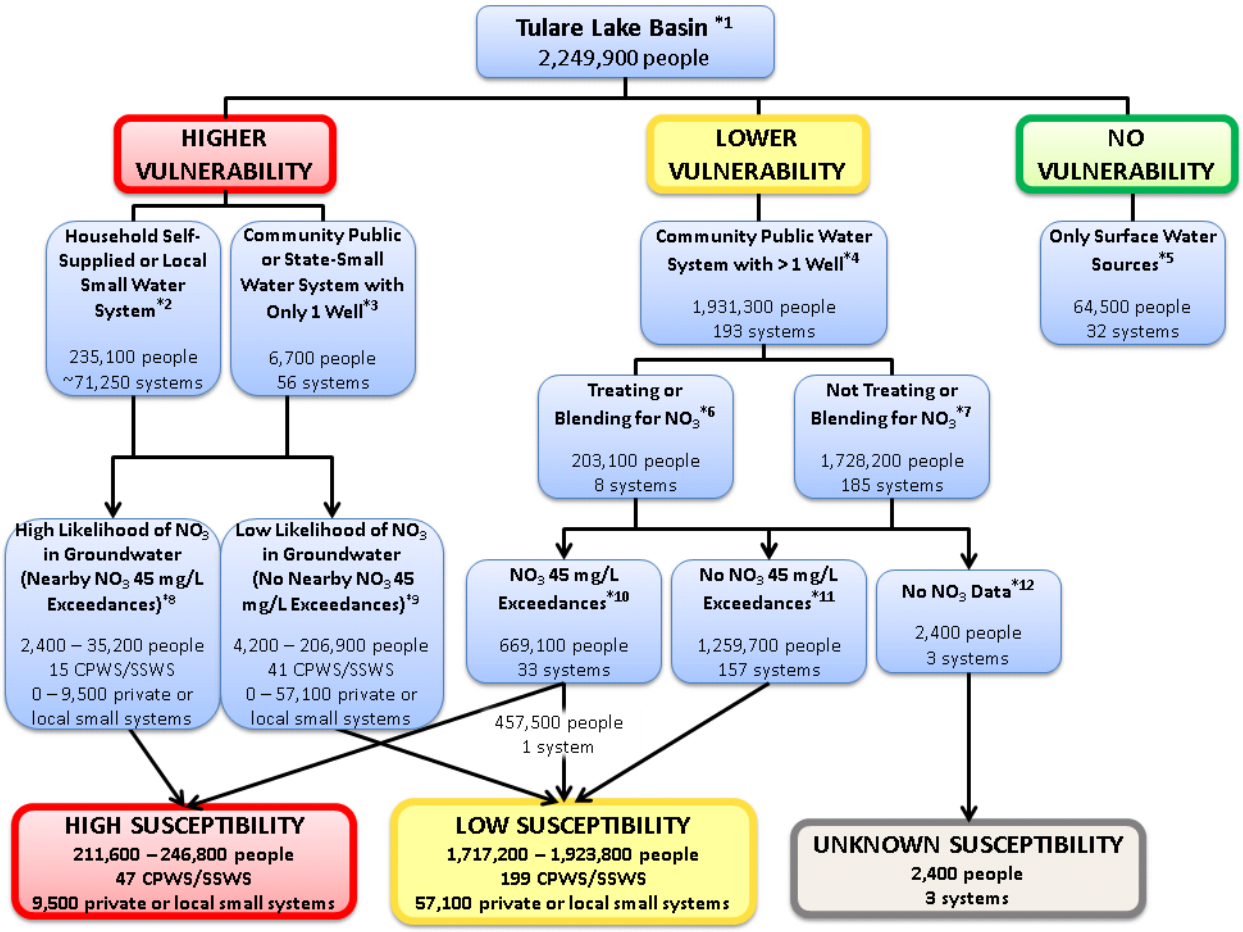

Figure 18. 2010 Population and susceptibility characterization for the Tulare Lake Basin based on estimated vulnerability and water quality data.[17]

[17] See Methods section for detailed explanations. The methodology used for creating the susceptibility chart and chart footnotes are further discussed in the Appendix (Section 10.1.1 Susceptibility Charts). Table 13 includes the unknown susceptibility as the highly susceptible population.

Adminsitrative Record
Page 35519

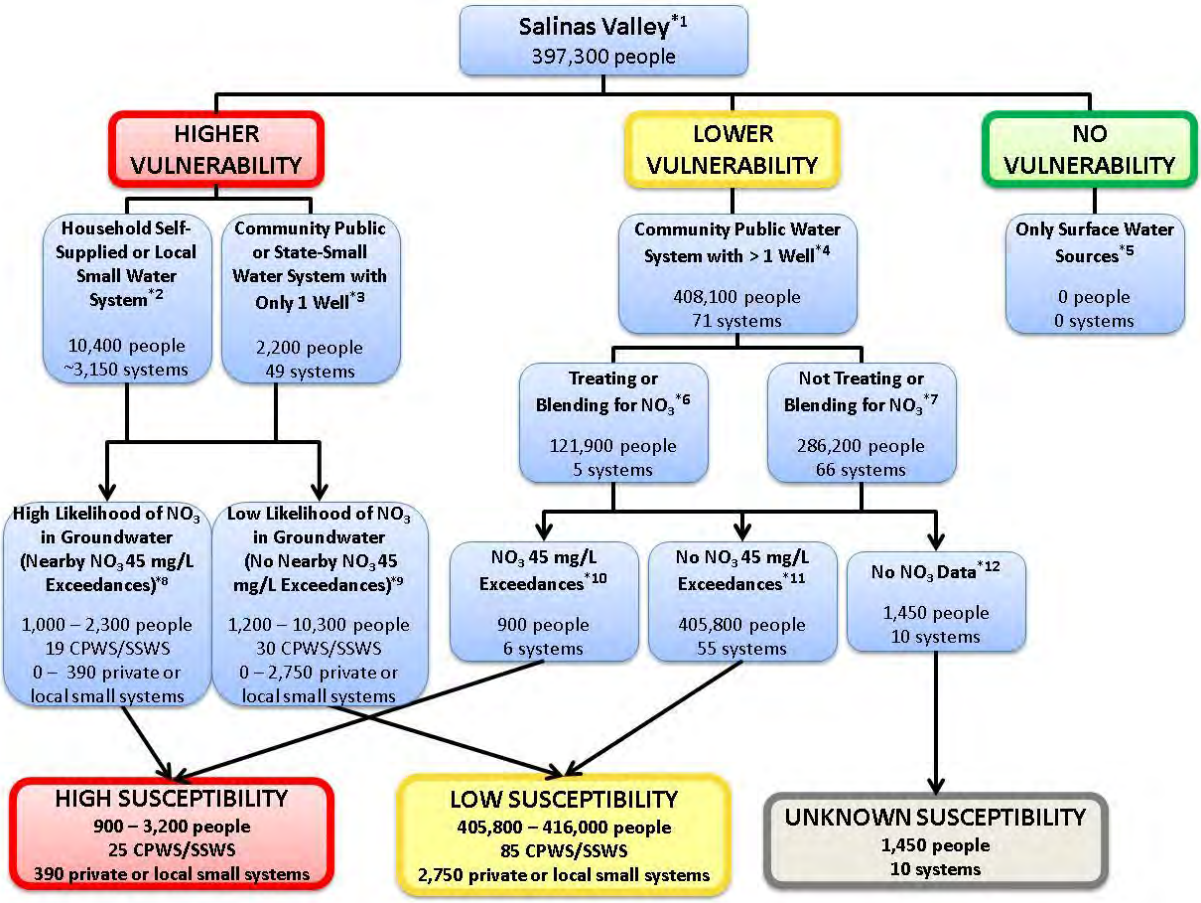

Figure 19. 2010 Population and susceptibility characterization for the Salinas Valley based on estimated vulnerability and water quality data.[18]

3.7 Health and Socioeconomic Disparities

This report estimates susceptibility as a qualitative likelihood of exposure; however, a more common definition of susceptibility to nitrate contamination is based on the health and socioeconomic status of an individual. Among those susceptible to the likelihood of consuming nitrate contaminated drinking water, there are sub-populations that are at financial risk and at a health risk from nitrate contamination of their water sources, but these are much more difficult to quantify. Those that are at a health and financial risk from nitrate contamination of their water sources are:

1) **Pregnant women or infants under six months** are more at risk to the health effects of higher levels of nitrate in drinking water. This is a direct public health concern.

[18] Due to different sources of data, the summation of the top row does not exactly equal the total study area population. All population and connection information is approximate. See Methods section for detailed explanations. The methodology used for creating the susceptibility chart and chart footnotes are further discussed in the Appendix (Section 10.1.1 Susceptibility Charts). Table 13 includes the unknown susceptibility as the highly susceptible population.

Adminsitrative Record
Page 35520

2) **Residents of disadvantaged unincorporated communities** have a more difficult time paying for both the capital and on-going operations and maintenance (O&M) costs of point-of-use or community-wellhead treatment options, or replacement bottled water, if local nitrate contamination becomes a problem. This is primarily a financial impact and financial feasibility concern of higher levels of nitrate in drinking water.

3.7.1 Pregnant Women and Infants

The number of pregnant women and infants within each basin was estimated using the Department of Finance data on a county level. This overestimates the health at-risk population since the boundaries of each county are not fully within in the study area boundaries. Roughly 84,500 and 14,100 pregnant women and infants live in the Tulare Lake Basin counties and Monterey County, respectively (California Department of Finance 2010). However, the location of these pregnant women and infants is unknown, making it difficult to determine if they are currently drinking nitrate contaminated water for incorporation into the nitrate drinking water contamination at-risk analysis. Applying the estimated total at-risk percentages (11% and 1% for the Tulare Lake Basin and Salinas Valley - previous section), approximately 10,000 and 200 pregnant woman and infants within Tulare Lake Basin counties and Monterey may be at risk to health problems from consuming nitrate-contaminated drinking water, respectively. Again, the difference in basin boundaries and county boundaries makes this a conservative estimate (or an overestimation).

3.7.2 Disadvantaged Communities (Disadvantaged Unincorporated Communities)

Title 22 of the CA Code of Regulations defines a disadvantaged community (DAC) as a community whose median household income (MHI) is less than or equal to 80% of the statewide MHI. The MHI for CA was $47,493 in 2000, so for this report, any community with an MHI less than $37,994 is considered a DAC.

DACs that are unincorporated often lack central water and sewer services. These disadvantaged unincorporated communities (DUCs) are at risk of nitrate contamination because they may lack a safe water source and are less financially able to buy bottled water or treat with point-of-use systems if their water source becomes contaminated with nitrate. Since these areas have a large concentration of families with low incomes, community solutions to nitrate treatment or alternative water supply might also be difficult.

Disadvantaged communities within the Tulare Lake Basin and Salinas Valley are shown in Figure 20, along with the delivered water quality of multiple source CPWSs (WQM data from 2006 to 2010). Severely disadvantaged communities (SDACs) have a MHI of less than 60% of the statewide MHI (less than $28,496). Some DACs include areas known as Census Designated Places (CDPs), or unincorporated areas, and Figure 20 includes some disadvantaged unincorporated communities (along with disadvantaged incorporated communities). CPWSs with delivered water quality exceeding the nitrate MCL within severely disadvantaged or disadvantaged communities are shown in hollow blue circles. Those located outside of severely disadvantaged and disadvantaged communities are shown in blue

Adminsitrative Record
Page 35521

points. About 45% of the multiple source systems that have delivered water exceeding the nitrate MCL are located within the severely disadvantaged and disadvantaged communities.

Figure 21 shows a scatter plot that relates the estimated 2000 MHI of the water system based on attributing the Census block group MHI to a water system located within it, with the maximum raw source water nitrate level of the water system (U.S. Census Bureau 2000). Systems are shown as being in an incorporated (non-CDP) or unincorporated (CDP) area. The systems above the red 45 mg/L (MCL) line and to the left of the blue 80% MHI line have a source where raw water has been in excess of 45 mg/L as nitrate at least once since 2006, and are located in a disadvantaged community. There are 51 community public water systems (serving about 714,000 people) in the study area with a raw source exceeding the nitrate MCL; 40 systems (serving about 379,000 people) are located in a disadvantaged community while eleven are located outside of a disadvantaged community. Thirteen of the 40 exceeding systems are CDPs (serving about 167,000 people) and 27 are non-CDPs (serving about 212,000 people). Of all 328 systems shown in Figure 21, 12% of the systems with raw water quality in excess of 45 mg/L as nitrate were located within a disadvantaged community while only 3% were located outside of a disadvantaged community. These numbers are different from Table 13, due to use of a system's raw water sources rather than delivered water quality.

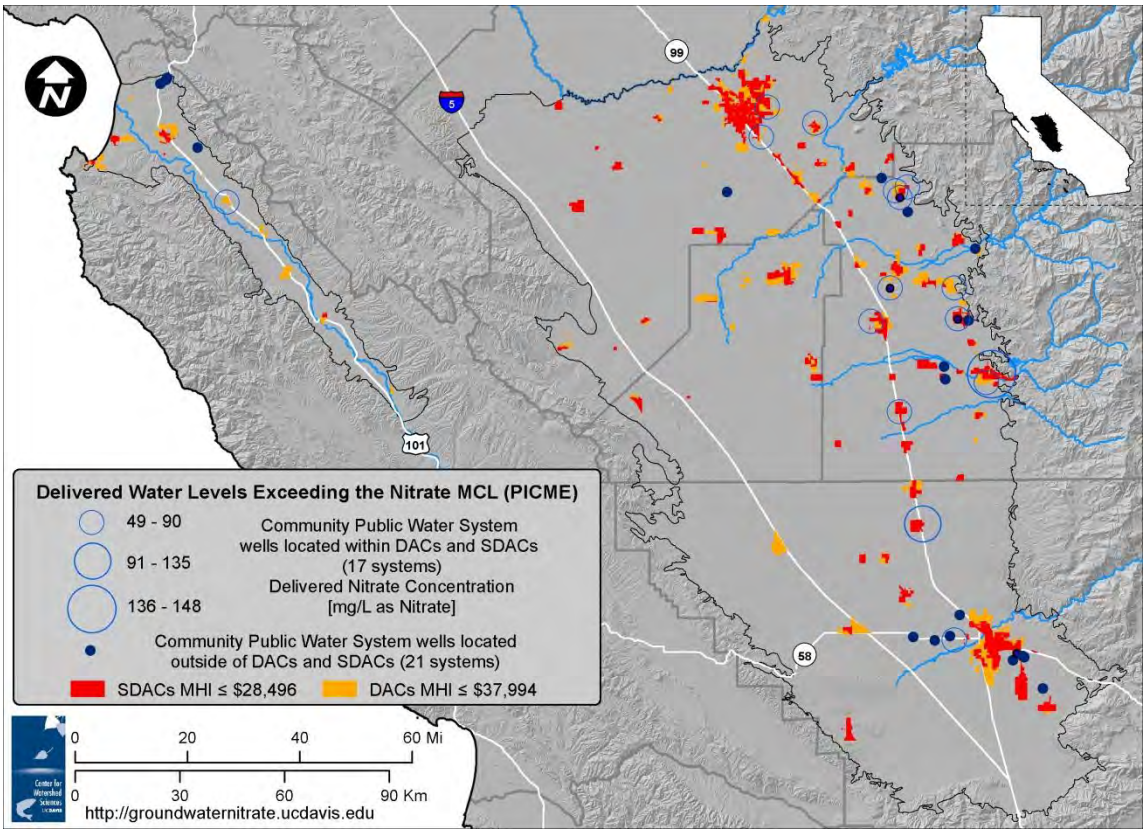

Figure 20. The relationship between DACs and SDACs and delivered water quality in multiple source CPWSs. (Source: CDPH PICME WQM 2006-2010; U.S. Census Bureau 2000; U.S. Census Bureau 2001.)

Adminsitrative Record
Page 35522

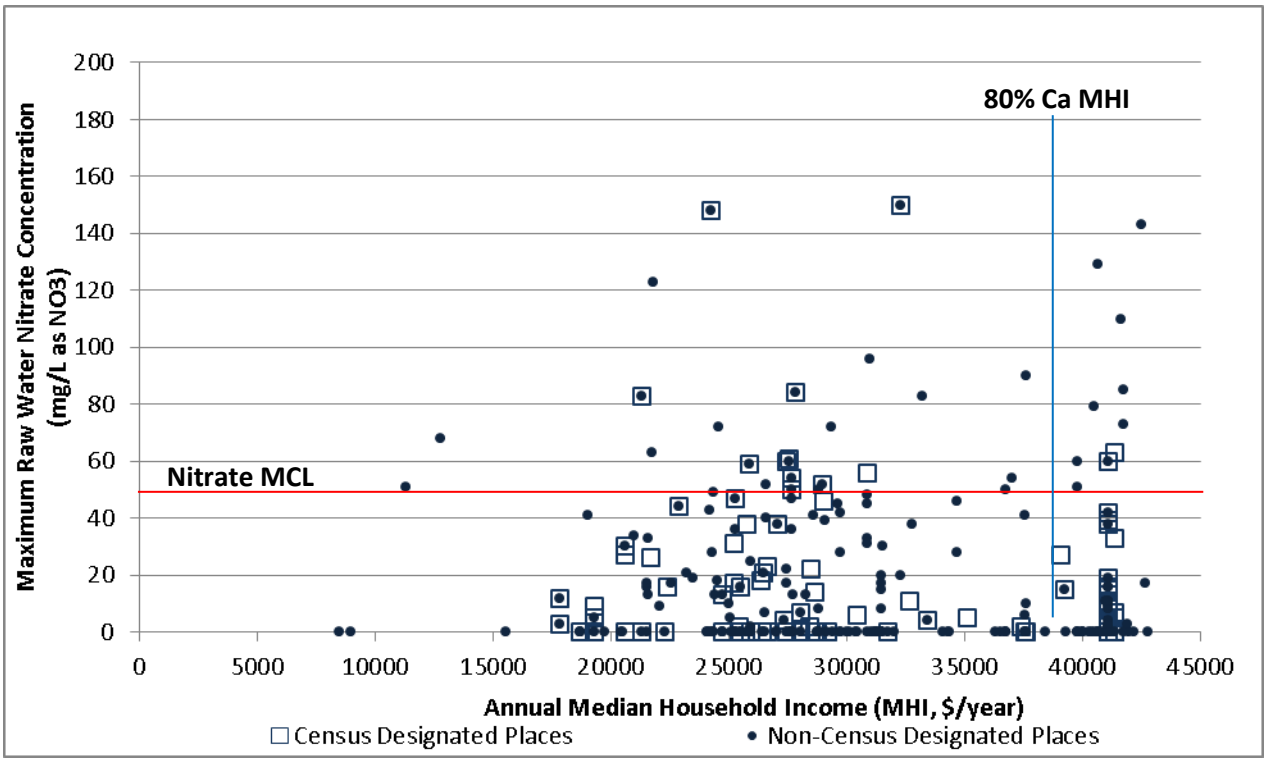

Figure 21. The maximum raw source water nitrate concentration for 328 state small and community public water systems in the Tulare Lake Basin and Salinas Valley with MHI data. (Source: CDPH WQM 2006 – 2010 and 2000 Census.) {Note: CDP = Unincorp. Place and non-CDP = Incorp. Place.}**

Disadvantaged unincorporated communities are smaller communities that are often either not connected to a public water system or are connected to very small systems. The smaller water systems lack technical, managerial and financial capacity to maintain an expensive water treatment facility or to provide alternative water supplies to their customers. Small water systems are typically responsible for more water quality violations and have persistent difficulties with successfully operating and maintaining their systems (Washington State Department of Health 2005).

3.8 Trends in Affected Populations over Time

Legacy nitrate contamination of groundwater will expand in spatial extent in the future and increase the at-risk population. By 2050, the total study area population is projected to grow to about 5.8 million people, with about 5.3 million people in the Tulare Lake Basin, and a little over 500,000 people in the Salinas Valley (California Department of Finance 2010).

3.8.1 Accumulating Nitrate and Population Changes

Current groundwater nitrate contamination is in part a legacy problem, as contamination percolates and spreads in groundwater decades after nitrogen is applied to the soil. In some places, source loading from decades ago is just now reaching the Tulare Lake Basin and Salinas Valley aquifers and nitrate applied to the surface today may not reach drinking water wells for decades. Nitrate accumulates in

Adminsitrative Record
Page 35523

groundwater; given the extent of contamination there is little evidence to suggest that nitrate concentrations in deep groundwater will decrease without intervention (denitrification is very minor in the environments of this study). As the area's population increases, more people in small and household water systems will face health and financial consequences from nitrate contaminated groundwater. As the population increases and agricultural land is converted to residential property, risks from nitrate-laden groundwater will increase, requiring more aggressive and expensive treatment.

We determined the long-term trends (over the past 40 years) of nitrate concentrations in public supply wells in the pilot study area. The average annual change in nitrate concentration was calculated for all public supply wells with at least two nitrate records listed in WQM. Table 14 shows the mean change in nitrate concentration for public supply wells in the Tulare Lake Basin, Tulare County, and the Salinas Valley. Based on this annual trend for nitrate in public supply wells, the total number of community public water systems that will have raw source water exceeding the MCL by 2035 and 2050 is estimated in Table 15 (CDPH's WQM Database from 1970s to current), showing the predicted affected population if no remediation or abatement of source application occurs. This analysis only examines the increasing nitrate trend in existing community public water systems and is applied to the maximum raw source water data in WQM. The affected population is estimated to be approximately 1.9 million people by the year 2050. Approximately 79% of the existing community public water systems' sources could exceed the MCL for nitrate by 2050. This estimate does not consider any treatment implementation or distribution of alternative water supplies and ignores water system population increases.

Table 14. Long-term trends for nitrate concentrations in public supply wells.[1]

	Mean Change [mg/L-yr]	Confidence Interval -95%	Confidence Interval +95%
Tulare Lake Basin {Tulare County} Public Supply Wells, 1970s – current	0.27 {0.41}	0.17 {0.22}	0.36 {0.59}
Salinas Valley Public Supply Wells, 1970s – current	0.53	0.31	0.77

[1] The nitrate trend for all public supply wells listed in CDPH's CADWSAP (WQM) database from 1970s to current.

Table 15. Estimated time for CPWS sources to exceed the nitrate MCL and total affected population.

Time For Maximum Recorded Raw NO_3^- Level to Reach the MCL	Total Number of Affected CPWSs	Total Affected Population	Percent of Total CPWSs Population (Study Area)
0 Years (2010)	77	1,363,657	57%
25 Years (2035)	114	1,836,732	76%
40 Years (2050)	127	1,903,300	79%

Adminsitrative Record
Page 35524

3.8.2 Drinking Water System Regulatory Changes

Tightening drinking water regulations and increasing trends in nitrate concentrations will likely make it more difficult for small local suppliers to comply with regulations for nitrate, arsenic, and other contaminants. With drinking water regulations expected to become more stringent in the future with tighter drinking quality standards and increased contamination occurrences, more water systems will be forced to implement state-of-the-art treatment facilities and monitor more frequently with online monitoring tools. Systems must be prepared to comply with more complex regulations while simultaneously continuing to comply with existing regulations. Compliance will be increasingly difficult as nitrate levels increase over time. Small systems lacking the technical, managerial, and financial capacity for operating and maintaining a treatment facility will struggle with future regulations. The lack of resources and looming safe drinking water challenges should motivate smaller systems to regionalize and consolidate with bigger systems, where possible, to achieve the economies of scale and rate base to be able to assure safe drinking water in an increasingly challenging environment for small systems, or to consult assistance programs for financial, technical and institutional help.

Adminsitrative Record
Page 35525

4 Alternative Water Supply Options

Alternative water supplies and nitrate treatment for drinking water are presented and discussed in this section, including the latest technologies, limiting factors, and capital and O&M costs, with a goal of identifying promising options for providing safe water to high susceptibility populations. Guidelines are developed for selecting promising water supply options and evaluating solution costs as a function of source water quality, system size, and system location. The discussions are based on an inventory and analysis of nitrate drinking water management strategies and treatment options available to the study area population, identifying concerns for each option, including financial and economic aspects. The most promising alternative water supply options for the susceptible population are considered and costs are presented with estimated costs in Section 5 Evaluation of Options.

Alternative water supply and nitrate management options identified in this report are grouped into three categories: improving the existing water source, providing alternative supplies, and relocating households (Table 16). Several ancillary activities can improve the performance of some water supply alternatives. Although each system requires its own engineering design and analysis, generalized cost estimates are discussed in Section 5 Evaluation of Options.

Table 16. Alternative water supply options.

OPTION
IMPROVE EXISTING WATER SOURCE
Blending
Drill Deeper Well
Drill New Well
Community Supply Treatment
Household Supply Treatment
ALTERNATIVE SUPPLIES
Switch to Treated Surface Water
Piped Connection to an Existing System
Piped Connection to a New System
Regionalization and Consolidation
Trucked Water
Bottled Water
RELOCATE HOUSEHOLDS
ANCILLARY ACTIVITIES
Well Water Quality Testing
Dual System

4.1 Improve Existing Water Source

Several source improvement options, including both non-treatment and treatment alternatives, can be used to ensure the provision of water that is compliant with the nitrate MCL. In this report, non-treatment options include blending and drilling a deeper or new well. Both can be limited by well characteristics, quality of available sources, and financial resources. The most common treatment

Adminsitrative Record
Page 35526

options for community public water systems are ion exchange (IX) and reverse osmosis (RO); additional options for drinking water nitrate treatment include electrodialysis/electrodialysis reversal (EDR), biological denitrification and chemical denitrification. Household treatment options include ion exchange at the point-of-entry or reverse osmosis at the point-of-use.

4.1.1 Blending

Blending dilutes a source with higher nitrate levels with a lower nitrate source to produce water that meets safe drinking water standards. Blending typically takes advantage of the differing nitrate concentrations typically found among wells of different locations, districts, and depths. Blending requires at least one nitrate compliant source and cannot occur in systems with only one well, unless additional water is brought from outside.

Blending is often considered a form of treatment because water systems are required to monitor and operate the blending process as a permitted treatment facility with a certified operator (CDPH 2008; Commandatore & Collins 2011). Requirements for blending include daily field monitoring, using continuous online nitrate analyzers to ensure complete mixing and compliant blended water quality; collection of monthly samples to certify a source is uncompromised; and consistent distribution of nitrate compliant water (CDPH 2008; Commandatore & Collins 2011). The compliance point for a blending system shifts from the source water in the well to the blended sampling point (or point-of-entry to the distribution system) (CDPH 2010). Water may be blended from several groundwater wells, or systems can blend groundwater with purchased treated surface water.

According to CDPH, blending is only acceptable as "a treatment process if one of the blended sources exceeds a primary MCL" (CDHS 2005). If a water system decides to blend their sources to comply with drinking water regulations, they must contact CDPH and coordinate with them to create a Blending Program and receive permission to blend. Blending requires two wells to continually operate, ensuring that one is always a low-nitrate source. A blending system must have an operator certified as Grade T2. If the low-nitrate blend water becomes compromised by high nitrate levels, a system automatically loses blending privileges. The maximum blend concentration allowed by CDPH is less than or equal to 40 mg/L (as NO_3^-).

Within the study area, eight CPWSs use blending alone to reduce nitrate in delivered water. Blending is typically the first choice and least expensive option when a nitrate compliant source is available. Estimated costs for blending are presented in Section 5.1.1 Blending.

4.1.2 Drilling a Deeper or New Well

Nitrate slowly follows general groundwater movement down from the surface to the saturated zone. Sometimes it is possible to avoid or defer nitrate contamination by drilling deeper wells. This is often considered a temporary solution because any nitrate contaminating the original shallower well can eventually infiltrate to the deeper well, unless any of the following conditions are met:

Adminsitrative Record
Page 35527

- a strongly chemically reducing aquitard zone, capable of denitrifying downward seeping groundwater, separates the current screen level from the target screen level;
- a rather impermeable layer separates the deeper well from the nitrate contamination; or
- the new well screen is much deeper (several hundred feet) below the current screen level *and* the well is properly sealed to the depth of the well screen.

A deeper aquifer protected by a clay layer could prevent upper aquifer nitrate-contaminated water from entering the new wells withdrawal zone. Depending on the local hydrogeology, source capacity may decrease with deeper wells – however, this is typically not an issue for small production domestic wells or wells with few connections. Other water contaminants (such as arsenic) may emerge at new depths. Jensen et al. (2012) found evidence of a possible increase in the incidence of arsenic MCL exceedance with well depth in areas of the Tulare Lake Basin (see Technical Report 6, Section 4.2, Jensen et al. 2012). New wells might be a feasible option for communities while they await long-term solutions, such as connection with larger systems, a new treatment system, or groundwater remediation. Drilling a deeper or new well takes less time than some construction and remediation projects. New well owners and systems employing new wells should test their well frequently.

Drilling a new or deeper well should employ an experienced well driller who is educated on the local hydrogeology. The driller should be familiar with the nitrate distribution and groundwater gradient at the desired well location. The main costs of drilling a new or deeper well will be drilling and pumping costs; both increase with the depth to uncontaminated water. Well modification of an existing well may limit the screened interval, to capture a region of groundwater with low nitrate concentrations. A packer/plug can be installed to restrict withdrawal from nitrated contaminated regions and installation can occur without removing pumps (BESST Inc. 2008).

A new well should be drilled more than about 30 meters (100 feet) from potential sources of pollution or contamination (such as septic fields). Drilling a well has many costs including:

- drilling a pilot test well and the larger diameter borehole for the production well;
- installing the well, filter pack, borehole seals, and surface completion;
- equipping the site;
- testing for sediment and water quality;
- well development;
- installation of storage and distribution systems; and
- planning, consulting, and engineering services.

Mettler Community Water District, in Kern County, drilled a 700 foot well at a capacity to serve 146 residents and it was estimated to cost $284,000. In 2009, a well comparable in size and depth was built by Plainview Mutual Water Company, in Tulare County, to serve 800 residents and it was estimated to cost $339,000. Ducor Community Service District, also in Tulare County, spent close to $725,000 on a 1,400 foot well for 850 residents. These cost estimates do not include the contingency, escalation, and design costs, that can add an additional 30% to the construction costs. (Self-Help Enterprises 2010)

Installing a new community water supply well can cost from about $300,000 to $1 million, depending on depth and capacity (Newkirk & Darby 2010). Kettleman City Community Services District estimated the costs for drilling a pilot test well to be about $320,000 (Summers Engineering 2011).

Adminsitrative Record
Page 35528

To estimate the costs for drilling a deeper well, a U.S. EPA BID document was used along with a quote from an experienced hydrogeologist. Costs for drilling a deeper or new well are presented in Sections 5.1.2 Drilling a Deeper Well and 5.1.3 Drilling a New Well.

4.1.3 Community Treatment

If an existing CPWS source exceeds the MCL for nitrate, centralized treatment may be implemented. Numerous factors are important in the consideration of community treatment, including the population served, the quantity of water distributed, and the technical and managerial capacity. The U.S. EPA has approved ion exchange, reverse osmosis, and electrodialysis as potable water treatment methods for nitrate removal (see Technical Report 6, Jensen et al. 2012). These three processes remove nitrate ions from the contaminated water and concentrate them into waste brines. The most common nitrate treatment method in the United States is ion exchange. Alternatively, denitrification methods do not transfer nitrate to a concentrated brine waste stream, but convert nitrate to a reduced nitrogen form, such as nitrogen gas. Full-scale application of denitrification treatment of potable water has been limited in the United States; however, two full-scale biological treatment plants, capable of denitrification, are being implemented in California and chemical denitrification has been used at the pilot-scale level. Europe has applied full-scale biological denitrification for potable water treatment (see Technical Report 6, Jensen et al. 2012).

The most appropriate treatment for nitrate contamination can be influenced by influent nitrate concentrations as discussed in Technical Report 6 (Jensen et al. 2012). Table 17 lists several scenarios for influent nitrate level and water system characteristics, with considerations listed for each option.

Table 17. Influence of nitrate concentration on treatment selection.[1]

Option	Practical Nitrate Range	Considerations
Blend	10 – 30% above MCL	Dependent on capacity and nitrate level of blending sources.
IX	Up to 2X MCL	Dependent on regeneration efficiency, costs of disposal and salt usage. Brine treatment, reuse, and recycle can improve feasibility at even higher nitrate levels.
RO	Up to many X the MCL	Dependent on availability of waste discharge options, energy use for pumping and number of stages. May be more cost-effective than IX for addressing very high nitrate levels.
BD	Up to many X the MCL	Dependent on the supply of electron donor and optimal conditions for denitrifiers. Ability to operate in a start-stop mode has not yet been demonstrated in full-scale application; difficult to implement for single well systems. May be more cost-effective than IX for addressing high nitrate levels.

[1]Based on contact with vendors and environmental engineering consultants. Excerpt from Technical Report 6 (Jensen et al. 2012).

Adminsitrative Record
Page 35529

The estimated costs for use of ion exchange and reverse osmosis are presented in Section 5.1.4 Community Treatment; additional information is provided in Technical Report 6 (Jensen et al. 2012). Since the most recent U.S. EPA Cost Estimating Manual is from 1979, collected cost information for arsenic treatment was used for a more up to date comparison of ion exchange treatment (U.S. EPA 2000). Refer to Technical Report 6 on Drinking Water Treatment for more nitrate treatment details (Jensen et al. 2012).

Communities with dual plumbing systems that separate drinking and cooking uses from other water uses, have the possibility to greatly reduce treatment quantities and costs, and reduce production of waste brines requiring disposal. However, dual plumbing systems increase capital and maintenance costs and may raise regulatory issues.

Any CPWS implementing treatment could consider using remote monitoring and management technology to lower operating and maintenance expenses. A remote telemetry or supervisory control and data acquisition (SCADA) system would be very beneficial to small systems lacking resources to support qualified operators on-site. Small water systems are more expensive and challenging to manage well and SCADA allows a skilled operator to supervise several systems remotely. A SCADA system allows real time control of system operation and maintenance of water quality by using a central computer to control mechanical processes and collect data from sensors. Emergency responses can be quick with instant notification of critical system events or episodes automatically sent to the operator. Additionally, the data acquisition component allows utilities to provide statistics on water quality and usage for budget planning, water quality compliance, system improvement, and targeted system expansions. A SCADA system can alert operators to changes in water quality requiring their assistance or automatically modify system operation through preprogrammed control functions not needing operator assistance. A SCADA and Programmable Logic Controller (PLC) control system for a 900 gallon per minute (gpm) surface water treatment plant costs about $75,000 (Summers Engineering 2011).

Based on the U.S. EPA 2007 Drinking Water Infrastructure Needs Survey and Assessment, approximate minimum costs for SCADA are represented for various system sizes in Section 5.1.4 Community Treatment.

4.1.4 Household Treatment

An alternative to community treatment is the use of a household water treatment device either at the point-of-entry (all water entering the home) or at the point-of-use (drinking and cooking water only). Point-of-entry (POE) solutions remove nitrate through reverse osmosis or ion exchange for the entire house (usually only indoor uses). Point-of-use (POU) solutions for nitrate commonly use reverse osmosis for kitchen taps (New Hampshire Department of Environmental Services 2008). Since nitrate is not a concern for non-drinking uses (showering), a POU system that treats drinking and cooking water is more economical than a POE system. POU systems have greater potential public health risks than POE systems, because residents may consume water from bathroom taps without a POU system. In general, reverse osmosis is the cheapest nitrate treatment at the household level (Mahler 2007).

Adminsitrative Record
Page 35530

Certification to the relevant ANSI/NSF standard by an ANSI accredited third party certifier ensures the safety and performance of the residential treatment systems (see Technical Report 6, Jensen et al. 2012). Currently there is only one ANSI/NSF household-level technology standard for nitrate reduction: NSF 58 - Reverse osmosis drinking water treatment systems (NSF 2010). POU devices that claim to reduce a drinking water contaminant must be certified by CDPH (CDPH 2008; CDPH 2010a). CDPH's Certified Residential Water Treatment Devices directory lists approved water treatment devices to reduce nitrate. "Under counter" approved systems reduce nitrate through reverse osmosis or reverse osmosis with carbon. "Counter top" approved systems reduce nitrate through reverse osmosis and granular activated carbon (CDPH 2011b).

POU treatment requires separation of drinking and cooking water supplies from other water uses and potentially increases public health risk, but is much less expensive than treating all household water to remove nitrate, and similarly generates much less waste brine. For a reverse osmosis system, piping from the device and faucet plumbing must be lead-free. This precaution is needed because reduction of the water's alkalinity can increase corrosiveness and leach lead (New Hampshire Department of Environmental Services 2008).

The average rated service flow for the certified residential nitrate treatment devices in the CDPH directory is about 20 gallons per day (gpd), with some devices as low as 7.6 gpd and some as high as 35.5 gpd (CDPH 2011a). The average human uses 0.8 gpd for cooking and drinking (NAS 2004), so these flow rates are appropriate for drinking and cooking needs of most family sizes.

While the residential water treatment devices in the CDPH directory are certified to remove nitrate from drinking water, there is a limit to their effectiveness. For example, many of the reverse osmosis with carbon treatment systems manufactured by Kinetico Incorporated are only acceptable for nitrate levels below 27 mg/L (measured as nitrogen) (CDPH 2011a). However, treatment with these devices can also remove other potential contaminants of concern, such as arsenic.

The estimated cost of a reverse osmosis POU device and an ion exchange POE device is discussed in Section 5.1.5 Household Treatment.

4.2 Connect to Alternative Water Supplies

Alternative water supplies include connection to a better quality water system, trucked potable water from a better source, and purchased bottled or vended water. A piped connection to a better quality water system can take three forms: connecting to an existing system, connecting to a newly created system, and consolidating several small systems into a larger new regional system. Trucking of water would most likely occur for remote, very small communities and businesses as an interim or emergency solution. The use of bottled or vended water is a simple and effective temporary solution for isolated households or small businesses, albeit at some cost and inconvenience.

Adminsitrative Record
Page 35531

4.2.1 Switch to Treated Surface Water

A piped connection to an existing surface water treatment system becomes promising if a community is reasonably near a well-functioning system with capacity. The costs for connecting to an existing surface water treatment system, currently supplied by a source such as the Central Valley Project (CVP) or State Water Project (SWP), would include the pipeline costs for the installed distribution pipe, trenching and excavation, embedment, backfill and compaction, valves, fittings and hydrants, dewatering, sheeting and shoring, horizontal boring, pavement removal and replacement, utility interference, and the fees for each connection to buy into capacity(varies within each system – the City of Davis charges about $9,000 for a residential ¾" meter connection to their system) (City of Davis Public Works Division 2011).

Kettleman City Community Services District is proposing switching from their contaminated groundwater (arsenic and benzene) to a surface water treatment plant. The County of Kings and Tulare Lake agricultural users are allocating some of their State Water Contract (900 acre-feet per year). The estimated cost for a surface water treatment plant is $6.6 million. The plant would supply 1.3 mgd of surface water to 1,500 people. An increase in residential rates is limited to $2.21 per month (1.94% of the MHI) (Summers Engineering 2011). The annual O&M will be subsidized by Kings County.

Although the quantities of water to serve a small community might not be large, legal and contracting issues will arise. When attempting to switch to surface water as a drinking source in the Tulare Lake Basin, operating characteristics of surface water supply systems designed for irrigation use, such as the Friant-Kern Canal, also may pose challenges to small rural communities when such systems are drained for maintenance during the off-season. To prepare for this type of surface water availability limitation, it may sometimes be appropriate to create a surface water treatment system with capacity to treat nitrate contaminated groundwater as well, so the residents have greater supply reliability.

Cutler and Orosi are two unincorporated communities in Tulare County located less than a mile from each other, and less than a mile from a smaller community, East Orosi. Cutler and Orosi each have their own water systems that are contaminated with DBCP and nitrate. The Alta Irrigation District has just completed the first stage of a feasibility study for treating Friant Kern Canal surface water to supplement Cutler and Orosi's contaminated groundwater sources. The project is estimated to cost approximately $17 million in capital costs and approximately $500,000 in annual operating costs (CWC 2011a).

4.2.2 Piped Connection to an Existing System

Costs for a piped connection to an existing water system with safe water would include infrastructure, base installed pipe, trenching and excavation, embedment, backfill and compaction, valves, fittings, and hydrants, dewatering, sheeting and shoring, horizontal boring, pavement removal and replacement, and connection fees to the existing safe water system. Most costs will be for distribution and connection.

Adminsitrative Record
Page 35532

Figure 22 shows the distribution of the minimum distances from a system serving less than 10,000 people (smaller systems) to a system serving more than 10,000 people (larger systems) for all CPWSs and SSWSs in the study area (distances are between sources, not service areas). Within the study area, 306 small systems and 38 large systems are analyzed in Figure 22 for possible interties, disregarding any institutional, political, technical, managerial, or financial barriers and costs. Of the 195 small systems in the Tulare Lake Basin, about 50% are within five miles of a larger system, and 88% are within 12.5 miles (Figure 22). Of the small systems in the Salinas Valley, about 15% are within five miles of a larger system, and 97% are within 12.5 miles (Figure 22). This analysis also assumes safe drinking water quality and available capacity for the systems with more than 10,000 connections. The connection potential for each basin is discussed further in Section 4.2.4 Regionalization and Consolidation.

East Niles Community Services District (ENCSD) charges $5,000 per connection to their system. Installing a 10" PVC pipe and valves was estimated by ENCSD to cost around $85 per foot (Self-Help Enterprises 2010), while U.S. EPA estimated $95 – 142 per foot (U.S. EPA 2007). Rehabilitating a pipeline is slightly less expensive at $73 per foot (U.S. EPA 2007). Any obstructions create additional expenses. For example, a railroad track crossing cost ENCSD $75,000 and crossing a canal cost $25,000 (Self-Help Enterprises 2010).

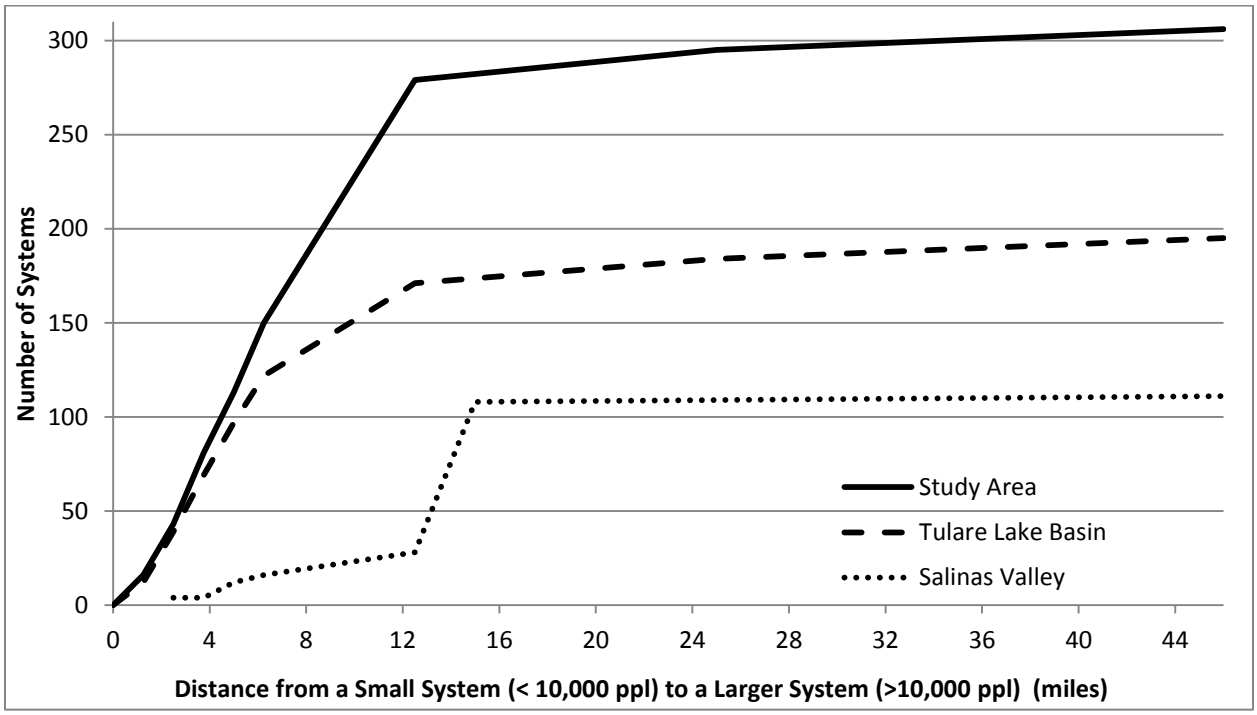

Figure 22. The minimum distance from a small system (≤ 10,000 people) to a larger system (> 10,000 people) for the entire study area. (Source: CDPH PICME 2010.)

4.2.3 Piped Connection to a Larger Newly Created System

Creating a new public water supply system is likely to involve costs for infrastructure, base installed pipe, trenching and excavation, embedment, backfill and compaction, valves, fittings, and hydrants, dewatering, sheeting and shoring, horizontal boring, pavement removal and replacement, utility

Adminsitrative Record
Page 35533

interference, new sources, and treatment equipment. For a new groundwater system, a district or community may need to look outside for safe groundwater. If a new groundwater system is constructed within a district's boundaries there must be property available for a structure to house a nitrate treatment system. It is recommended to construct at least two wells since pumping units can fail within a well and it is prudent to have a backup source. Cost estimates would include costs for two wells, distribution, and treatment.

> The San Jerardo Water System spent over $5 million installing two miles of water transmission pipelines, a 285,000 gallon water tank, a potable water well, and pumping station. They also created an intertie with a nearby potable water supply for emergency service (Monterey County Planning Commission 2009).

4.2.4 Regionalization and Consolidation

Regionalization and consolidation in the drinking water sector can take many forms. Inter-tying systems to create a new larger system combines several small water systems that suffer from diseconomies of scale into one larger water system that serves the desired population and provides treatment to comply with drinking water standards. Regionalization has more formally been defined as "…a creation of an appropriate management or contractual administrative organization or a coordinated physical system plan of two or more community water systems in a geographical area for the purpose of utilizing common resources and facilities to their optimum advantage" (Grigg 1989). Similarly, consolidation has been defined as "one community public water system being absorbed into, combined with, or served by other utilities to gain the resources they lack otherwise" (Raucher et al.2004). Consolidation often refers to giving up control and independence by one entity (or water system) as it is merged into another single entity. This transfer of control and loss of independence does not always occur with regionalization; multiple smaller systems can join together to create a larger system with management distributed evenly among all parties.

> Kettleman City CSD identified potential safe water supplies about 8,000 feet away from their property. The CSD's Engineer estimated drilling two new groundwater wells, with an 8,000 foot distribution pipeline, to cost about $6.6 million (Summers Engineering 2011). The CSD also estimated treating two new groundwater wells (treatment for arsenic and odor) drilled on the existing property to cost about $7.7 million (Summers Engineering 2011).

Regionalization and consolidation combine neighboring water systems to improve service and efficiencies, and to lower costs through economies of scale (Eskaf 2009). Regionalization and consolidation can be considered for drinking water systems struggling to meet regulatory compliance, unable to sustain aging infrastructure, unable to operate the system, or concerned about future water availability. Regionalization and consolidation are especially attractive for small systems that lack population base and access to financial resources and technical expertise. Systems can achieve economies of scale through regionalization without being physically connected by sharing capital equipment and management staff, or by participating in joint business and logistic operations. The least to most collaborative regionalization options range from: 1) create a planning document together; 2) initiate communication to discuss water system issues or to call during an emergency; 3) share inventory or equipment; 4) share an operator; 5) join management or delegate bookkeeping or billing to one

Adminsitrative Record
Page 35534

entity; 6) interconnect systems for emergency purposes only; 7) share water rights or water resources without an intertie; 8) intertie systems, but maintain separate operations; and 9) intertie systems, close current systems, or form a combined system (New Mexico Rural Water Association 2006).

The optimal economic water system size and least-cost service area are estimated by the cost trade-offs between the water source acquisition-treatment component and the transmission-distribution component (Clark & Stevie 1981). Smaller water systems may not have the economies of scale for the acquisition-treatment component, resulting in increased marginal costs for drilling and pumping. Ideally, in consolidation there is a balance between the increasing returns in acquiring and treating the water and the decreasing returns involved in distributing water further and further. Rural small community water systems are generally farther from large urban systems, resulting in high costs for the transmission-distribution component of consolidating that often are unaffordable by low income rural populations (Ottem et al. 2003).

There is significant potential based on size and distance analysis for systems to consolidate in the Tulare Lake Basin. There are 98 large systems and 195 small systems analyzed for connection. The number of smaller systems that are within 15 miles of larger systems is shown in Table 18. The Cities of Fresno, Dinuba and Porterville, as well as the CSU Fresno water systems, are available for ten or more small systems to connect based solely on their spatial proximity. Figure 23 shows the number of smaller systems that are within a specific distance range of a larger system (e.g., there are 12 smaller systems within 1.25 miles of a larger system in the Tulare Lake Basin). Considering only piping costs, about 97 smaller systems could consolidate and join a larger system less than five miles away at a cost of about $1.6 million[19] per system. Spatial proximity is the only consideration in suggesting these interties; other territorial, institutional, infrastructure upgrading, legal, and political barriers are ignored.

Within the Salinas Valley, there are 19 large systems and 111 small systems analyzed for connection (Table 19). The California American Water Company in Monterey and the California Water Service Company in Salinas are close to ten or more small systems for possible connection consideration, based solely on spatial proximity. Figure 24 shows the number of smaller systems that are within a specific distance range of a larger system (e.g., there are 4 smaller systems within 1.25 miles of a larger system in the Salinas Valley). Again, considering only piping costs, about 28 smaller systems could consolidate and join a larger system less than thirteen miles away at a cost of about $4 million per system (Monterey County Environmental Health 2010b).

Assuming sufficient capacity and adequate internal infrastructure of the small systems, there is significant potential for regionalization and consolidation if a system can afford the pipeline and connection costs. The costs considered in this report for connecting to a new or existing system are only those for installing the intertie pipeline and for connection fees. The operating costs for the intertie pipeline, the additional value of the safe water relative to the unsafe water, and the cost for land

[19] This assumes a $61/ft pipe cost from the Granite Ridge Regional Water Supply Project Feasibility Study (Monterey County Environmental Health 2010b).

Adminsitrative Record
Page 35535

easements for transmission are not considered in this report. Various cost scenarios for constructing pipeline to a new or existing system are presented in Section 5.1.6 Connect to an Alternative System.

Table 18. Large systems and the number of nearest small systems identified by distance proximity nearest-neighbor analysis in the Tulare Lake Basin.

System Number	System Name (> 10,000 System)	Connection	Population	Potential Number of Connecting Systems (< 10,000 ppl)[1]	Population of Potential Number of Connecting Systems
1010007	City of Fresno	130,176	457,511	11	11,731
1010019	City of Kingsburg	3,413	11,300	3	653
1010024	California Water Service Co. - Selma	6,078	24,307	8	8,377
1010025	City of Parlier	2,329	12,058	1	1,100
1010027	City of Reedley	5,445	25,584	2	60
1010029	City of Sanger	5,971	25,404	8	2,301
1010339	California State University, Fresno	550	22,000	15	12,094
1510001	Arvin Community Services District	3,446	11,847	1	400
1510005	City of Delano	8,829	53,855	9	14,500
1510006	East Niles Community Services District	7,406	25,500	7	2,560
1510012	Lamont Public Utility District	3,475	13,296	6	6,959
1510015	Oildale Mutual Water Company	7,708	26,000	7	11,307
1510019	City of Shafter	4,090	15,609	3	7,296
1510022	West Kern CWD	7,443	16,630	2	7,464
1510029	Vaughn WC INC F	9,246	28,100	11	1,930
1510031	City of Bakersfield	43,086	147,999	12	1,240
1610003	City of Hanford	15,509	53,320	9	7,157
1610005	City of Lemoore	6,117	24,500	5	3,386
5410002	City of Dinuba	6,025	21,087	13	17,909
5410003	City of Exeter	3,012	10,730	5	8,399
5410004	City of Farmersville	2,420	10,672	3	1,440
5410006	City of Lindsay	2,303	11,450	5	3,773
5410010	City of Porterville	14,896	51,467	17	9,957
5410015	City of Tulare	15,967	57,375	4	3,618
5410016	California Water Service Co. - Visalia	40,530	133,749	9	7,150

[1] Within 15 miles.

Adminsitrative Record
Page 35536

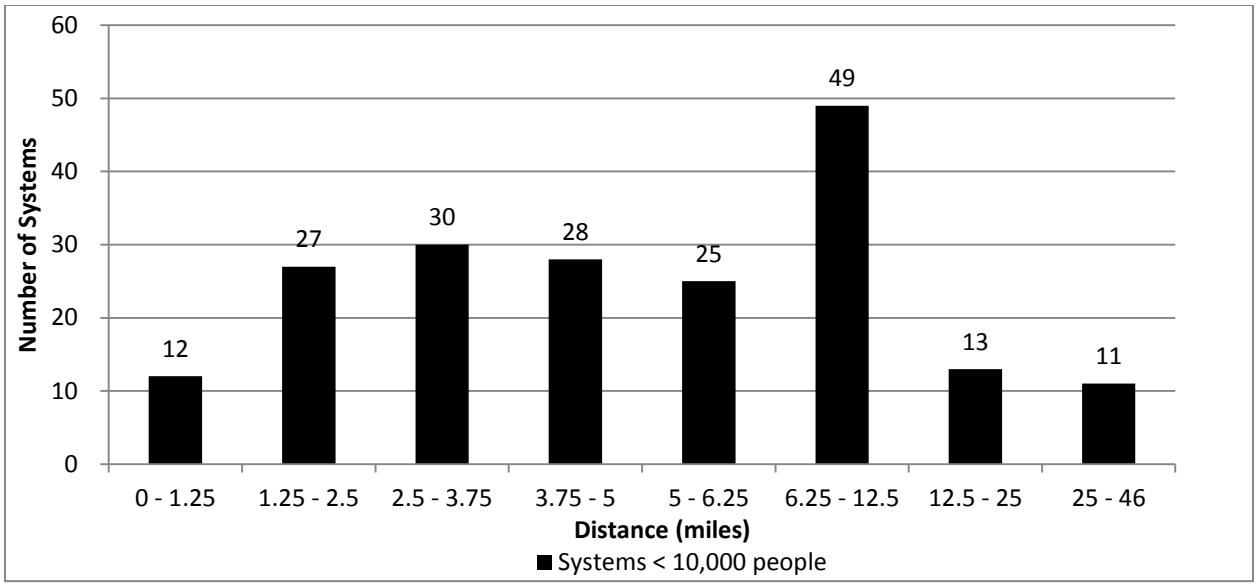

Figure 23. The minimum distance ranges between all existing small systems (≤ 10,000 people) and larger systems (> 10,000 people) in the Tulare Lake Basin. (Source: CDPH PICME 2010.)

Table 19. Large systems and the number of nearest small systems identified by distance proximity nearest-neighbor analysis in the Salinas Valley.

System Number	System Name (> 10,000 System)	Connection	Population	Potential Number of Connecting Systems (< 10,000 ppl)[1]	Population of Potential Number of Connecting Systems
2710004	California American Water Company Monterey	38,701	122,492	12	7,162
2710008	City of Greenfield	3,469	17,547	10	10,990
2710010	California Water Service Co. - Salinas	25,451	114,840	78	25,511
2710011	City of Soledad	4,082	16,146	9	9,411
2710017	Marina Coast Water District	8,357	34,600	2	1,050

[1] Within 15 miles.

Adminsitrative Record
Page 35537

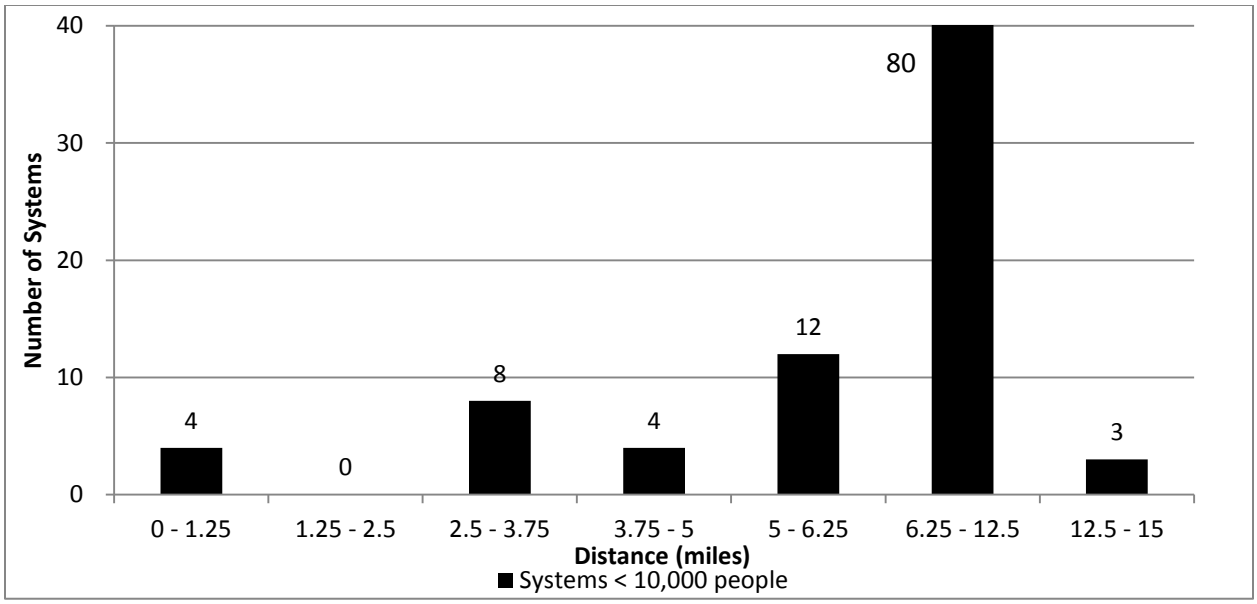

Figure 24. The minimum distance ranges between all existing small systems (≤ 10,000 people) and larger systems (> 10,000 people) in the Salinas Valley. (Source: CDPH PICME 2010.)

4.2.5 Trucked Water

Trucked water is a community or household water supply option that involves hiring a potable drinking water hauler, licensed with the California Department of Public Health, to deliver water to residential and commercial areas. Prior to use, the truck must be cleaned and inspected thoroughly, disinfecting all truck components with chlorine for 24 hours prior to delivery. Trucked water is often used for emergency supplies, but has permitting issues and is not acceptable for new public water systems (CDPH 2011c; CDPH 2011d). CDPH only recommends supplying trucked water for emergency or short-term situations (CDPH 2011c; CDPH 2011d). California water haulers are required to have a Water Hauler License from the Department of Public Health's Food and Drug Branch (FDB) (CDPH 2008). FDB periodically inspects water hauler trucks to ensure compliance with laws and regulations. Historical sanitary and financial concerns have made trucked water problematic.

Trucked water also could be used with a dual system, where only potable water for drinking and cooking is trucked in while the contaminated supply is used for other household needs. A trucked water dual system is infeasible as a long-term solution due to costs and CDPH regulations.

The estimated cost for providing a community or household in Tulare County with trucked water is shown in Section 5.1.7 Trucked Water.

Adminsitrative Record
Page 35538

4.2.6 Bottled Water

Bottled or vended water is often a temporary solution for communities or households with a nitrate problem. While other long-term solutions are being developed, funded, and implemented, bottled water is the quickest solution for low-nitrate drinking water. However, bottled water is considerably more expensive than publicly supplied systems or well water, and is regulated less stringently by the FDA than tap water is by the U.S. EPA (NRDC 2010).

As with any dual system, accidentally drinking water from a faucet served with water exceeding the nitrate MCL is a risk. Bottled water must be either delivered or picked up from the store or distributor, so households might occasionally run out of bottled water and use tap water. Households need to ensure that they order the appropriate amount, inventory their usage, and remember to place new orders. The reoccurring monthly cost is particularly unattractive to low income areas. While long-term purchase of bottled water is expensive, the monthly cost is less than a lump sum of capital-intensive solutions like drilling a new well. Households recently informed of their connection to a contaminated source purchase bottled water as an immediate solution, while still paying for non-potable piped water.

The estimated cost for delivering bottled water to a household in Tulare County is discussed in Section 5.1.8 Bottled Water.

Matheny Tract is a 45-acre disadvantaged unincorporated community outside of Tulare City limits that has been purchasing bottled drinking water since 2007. As of 08/2010 Tulare LAFCO is requiring the City of Tulare to extend water and sewer service to Matheny Tract (Pacific Institute 2011).

Beverly Grand Mutual Water Company (BGMWC), in Porterville, serves 108 people and has been in violation of the nitrate MCL since 2000, with the most recent violation in April 2010 (65 mg/L as NO_3^-). About 50 people are below or near the poverty level. The Pacific Institute surveyed households served by BGMWC and found that, on average, households spend $31.63 on non-tap replacement water per month, while still paying $25.00 per month on contaminated tap water (Pacific Institute 2011).

4.3 Relocate to Area with Better Water Supply

Relocating residents to a place with safe, reliable water supply is a socially and politically difficult extreme option. Susceptible populations would face costs of selling property (often at a loss) or completely abandoning property, moving, loss of jobs, increased travel distance to work, and potential social dislocation. Nearby employers would face higher labor costs and landlords would face reduced rental prospects. The area would also suffer a decrease in economic activity. Some residents currently living in nitrate-contaminated areas are there because rent is inexpensive and employment is nearby. Residents of many small rural communities might also be unable to afford to live in areas with safe water supplies. Nevertheless, under some circumstances relocation may be the most attractive option, particularly when a community faces other economic challenges. The estimated costs for relocating

Adminsitrative Record
Page 35539

households in the study area to a place with a better water supply are discussed in Section 5.1.9 Relocate to Area with Better Water Supply.

4.4 Ancillary Activities

Some options can be improved by simultaneously installing a dual water distribution system or by a well water quality testing program. A dual water system for a self-supplied household can allow for more economical use of POU RO unit or bottled water. A dual water distribution system for community public water system customers will include constructing new plumbing and installing groundwater treatment for the potable portion of the supply, and maintaining the existing distribution of contaminated water for the non-potable supply. Well water quality testing can improve blending efficiency and increase the likelihood of providing low-nitrate water when drilling a new or deeper well.

4.4.1 Dual Water Distribution System

A dual water distribution system has two separate distribution networks, one for the distribution of potable water and another for the distribution of non-potable water. According to Title 22, Section 60301.250 a dual plumbed potable water system has separate piping for potable water (CDPH 2008). A dual water distribution system would rely on water from the current nitrate contaminated supply for non-potable uses while consuming potable water from a POU treatment system (reverse osmosis), bottled water, water trucked from an outside source, or safe drinking water from the existing water system that has been separated, treated, and piped through a secondary distribution network to the household. Total costs would be the current monthly cost for the contaminated supply plus the cost for a potable POU treatment system, bottled water, or trucked water. If the entire water system decided to create a dual plumbed system the costs would include the cost for the contaminated supply, new treatment, installation of a new pipeline to existing service connections, and the re-plumbing of households served. A system could continue to distribute the nitrate-contaminated supply for non-potable uses and install a smaller treatment system for the potable supply with delivery through a new second parallel distribution network. Any water used for irrigation, or other non-drinking water uses, could be delivered through the existing pipeline, with a new plumbing system installed for the delivery of the treated, potable water. This would greatly reduce the water treatment costs for a system since a smaller volume would need to be treated. Costs for dual water distribution systems also include proper maintenance on existing infrastructure, water quality, and water pressure. Costs for dual water distribution options are discussed in Section 5.1.10 Dual Water Distribution System.

4.4.2 Well Water Quality Testing

Well water quality testing is important for all households on non-public water systems to detect high nitrate levels within their water supply, and especially in nitrate affected risk areas. Well water quality testing is recommended for households or state small systems that have not yet tested their water supply for harmful contaminants to determine if they are at risk and need an alternative water supply. The National Ground Water Association (NGWA) recommends domestic well owners test the water

Adminsitrative Record
Page 35540

quality annually for bacteria, nitrite/nitrate, and other constituents of concern. The local health or environmental health department may also provide water quality testing and well cleaning advice. Before blending wells, well water quality testing is required to discover and manage the appropriate blend of sources to produce water below the MCLs for nitrate and other contaminants. Similarly, when considering drilling a deeper well, the well water quality should be tested for other contaminants such as arsenic or manganese, as those contaminants may exceed regulatory levels in deeper wells. When testing the well water quality it is recommended to have a California State-certified drinking water testing laboratory conduct the analyses (a list of drinking water laboratories certified by the CDPH is available at : http://www.cdph.ca.gov/certlic/labs/Documents/ELAPLablist.xls). The costs for residential well water quality testing are discussed in Section 5.1.11 Well Water Quality Testing.

4.4.3 Rainwater Cisterns

Another option considered is implementing rainwater cisterns. However, cisterns are not commonly feasible for the scale of the problem in this arid area. A short description of this option is included in Appendix Section 10.2 Rainwater Cisterns.

Adminsitrative Record
Page 35541

5 Evaluation of Options

Each option is evaluated on a system scale, identifying economic and financial feasibility and addressing any public health concerns. While each water system requires its own engineering analysis to reflect local conditions, here we present a broad general comparison of the costs of various options for policy purposes. The advantages and disadvantages of each alternative are discussed for public water systems in Table 20 and for household self-supplied and local small water systems in Table 21.

Table 20. Advantages and disadvantages of options for public water systems.

	ADVANTAGES	DISADVANTAGES
Blending	▪ Simple non-treatment alternative ▪ Cost-effective, given suitable wells	▪ Capital investment for accessing an alternative source ▪ Relies on availability and consistency of low-nitrate source ▪ Monitoring requirements ▪ Rising nitrate levels may preclude ability to blend
Drilling a Deeper or New Well	▪ Potentially more reliable water supply ▪ Cheaper than bottled water for households using more than 8 gpd	▪ Potential decrease in source capacity ▪ Capital and operational costs increase with depth ▪ Potentially only a temporary "quick-fix"; longevity depends on local hydrogeologic conditions and landuse ▪ Risk of encountering other water quality concerns at greater depths (i.e., arsenic, manganese) ▪ Pipeline costs if source area is far from original source
Community Treatment (IX, RO & EDR)	▪ Multiple contaminant removal ▪ Feasible, safe supply	▪ Disposal of waste residuals (i.e., brine waste) ▪ High maintenance and/or energy demands ▪ Resin or membrane susceptibility
Piped Connection to an Existing System	▪ Safe, reliable water supply	▪ Capital cost of pipe installation ▪ Connection fee ▪ Water rights purchase (surface water)
Piped Connection to a New System	▪ Safe, reliable water supply	▪ Capital cost of pipe installation ▪ High treatment system capital and O&M costs ▪ Water rights purchase (surface water)
Regionalization & Consolidation	▪ Often lower costs	▪ High capital and O&M costs
Trucked Water	▪ Community-wide distribution ▪ No start-up capital cost	▪ Temporary "emergency" solution ▪ Not approved for new water systems
Relocate Households	▪ Safe, reliable water supply	▪ Socially and politically difficult, extreme option ▪ Loss of property value and jobs ▪ Social, familial dislocation
Well Water Quality Testing (Already in place)	▪ Water quality awareness ▪ Beneficial to blending	
Dual System	▪ Hybrid of options ▪ Treating only potable	▪ Possible consumption of water from contaminated source ▪ Cost of contaminated supply plus cost for POU system or trucked/bottled water, or capital dual plumbing costs

Adminsitrative Record
Page 35542

Table 21. Advantages and disadvantages of options for self-supplied households or local small water systems.

	ADVANTAGES	DISADVANTAGES
Drilling a Deeper or New Well	▪ Potentially more reliable water supply ▪ Cheaper than bottled water for households using more than 8 gpd	▪ Potential decrease in source capacity ▪ Capital and operational costs increase with depth ▪ Potentially only a temporary "quick-fix"; the nitrate plume follows groundwater movement ▪ Risk of encountering other water quality concerns at greater depths (i.e., arsenic, manganese) ▪ Pipeline costs if source area is far from original source
Household Treatment (RO)	▪ Multiple contaminant removal ▪ Low-nitrate supply	▪ Unless instructed, risk of improper handling or maintenance of equipment
Regionalization & Consolidation	▪ Cheaper treatment costs on a customer basis	▪ High capital and O&M costs
Trucked Water	▪ Community-wide distribution ▪ No start-up capital cost	▪ Temporary "emergency" solution ▪ Extra potable water storage required if a small community
Bottled Water	▪ Nitrate-free supply ▪ No start-up cost	▪ Inconvenience, monthly expenditure ▪ Temporary solution
Relocate Households	▪ Safe, reliable water supply	▪ Unpleasant, extreme option ▪ Loss of property value and jobs ▪ Social, familial dislocation
Well Water Quality Testing	▪ Water quality awareness ▪ Beneficial to blending	
Dual System	▪ Hybrid of options ▪ Treating only potable	▪ Possible consumption of water from contaminated source ▪ Cost of contaminated supply plus cost for community treatment of potable supply and capital dual plumbing costs.

5.1 Economic and Financial Costs

Affordability and sustainability are key issues for deciding if a solution is appropriate for an area. Small water systems with nitrate contamination will often be unable to support new development with limited safe water sources and unable to increase the number of connections (contributing to local economic decline). Consolidation of small water systems can increase economies of scale, and reduce technical and financial burdens by reducing total cost and distributing costs over a larger population. A community must have the technical, managerial and financial capacity to successfully implement a solution.

The Pacific Institute (2011) and others discuss the cost of avoiding or treating nitrate-contaminated water, suggesting that financing "is typically borne by water users and by local government and water providers, and is indirectly incurred by local and state tax payers, through tax revenues that pay for drinking water improvement projects." Individuals currently connected to an impacted water system

Adminsitrative Record
Page 35543

must pay for their own bottled water, health care services, or point-of-use treatment device. The same costs may be incurred by individuals connected to systems at risk of future contamination. These economic and financial results will be considered when discussing future funding sources and policy options (see Technical Report 8, Canada et al. 2012) for the Tulare Lake Basin and Salinas Valley.

Compared to larger cities, small, disadvantaged, unincorporated communities and self-supplied households often have different economical solutions. Based on the analysis herein, the least expensive option for self-supplied households and local small water systems is often to install point-of-use reverse osmosis devices for all potable uses within their households. If a household can afford drilling a new well and if that well can tap an uncontaminated supply, then a new well can be an attractive alternative; the new well option has fewer potential health concerns from improper handling and accidental consumption of untreated water. The least expensive option for very small community public water systems (serving less than 500 people) is often to install ion exchange treatment within the system configuration. For small community public water systems (serving 500 to 3,300 people) the least expensive option is often to install reverse osmosis treatment. Another economical option for small water systems (serving less than 3,300 people) is often to connect to another system; however, the costs for connection here include pipeline costs and a rough estimation of connection costs. Constructing a new well also may be economical. For medium community public water systems (serving 3,300 to 10,000 people) the least expensive option often is to install groundwater treatment, either ion exchange or reverse osmosis, since the estimated costs overlap. Another economical option for medium systems is to construct a new well or to construct a new well some distance from the existing system location. The economies of scale for pipeline start to prevail for the medium system sizes, and they can pipe their way out of the problem. For larger community public water systems, all options are relatively equal and local conditions become more important. A larger community public water system also has more opportunity to connect to surface water; with the larger population base the costs of connecting, maintaining, and treating the system can be equitably distributed without imposing too much of a financial burden.

For the final basin-wide cost analysis presented in Section 6 Basin-wide Costs of Nitrate Contamination, the following alternative water supply options were excluded from community water supply options: bottled water, trucked water, blending, and a dual water distribution system. The U.S. EPA does not allow a community public water system to distribute bottled water to their consumers as a means of complying with drinking water standards.[20] In addition, new community public water systems are not allowed to have trucked water delivered to their consumers and older water systems are only permitted to use this option in an emergency (CDPH 2005; CDPH 2011c; CDPH 2011d). Lastly, blending was not considered in the basin-wide cost analysis, but is recommended as the first step for a community public water system towards complying with the nitrate MCL. If a water system has an additional, low-nitrate source (at least less than 40 mg/L as nitrate) a blending program should be set-up and permitted by CDPH, as the annual O&M costs are less than a groundwater treatment system. For a small water

[20] National Primary Drinking Water Regulations, Code of Federal Regulations Title 40 §141.101.

Adminsitrative Record
Page 35544

system (less than 3,300 people), the annualized cost of blending and drilling a new well are almost equivalent, however, they both rely on the future fate of the sources available.

The costs of alternative water supply and treatment options vary with numerous factors and the numbers presented here are estimates based on averages of widely variable costs. Costs will be unique to each individual system. For proper cost estimation, a feasibility analysis is necessary to assess the potential solution for each unique system. The following overall assumptions were made to estimate costs for the alternative water supply options in Table 22:

- Twenty year life of product/equipment/materials (except for household treatment – 10 years and bottled/trucked water – no capital)
- 2.15 gallons per household per day of potable water consumption (NAS 2004)
- 3.3 persons per household
- 2010 dollars

Table 22. Summary of approximate alternative water supply option costs.

OPTION	ESTIMATED ANNUAL COST RANGE ($/year)[1]	
	Self-Supplied Household	Small CPWS(1,000 households)
IMPROVE EXISTING WATER SOURCE		
Blending[2]	N/A	$85,000 - $150,000
Drill Deeper Well[3]	$860 - $3,300	$80,000 $100,000
Drill a New Well[4]	$2,100 - $3,100	$40,000 - $290,000
Community Supply Treatment[5]	N/A	$135,000 - $1.09 M
Household Supply Treatment[6]	$250 - $360	$223,000
ALTERNATIVE SUPPLIES		
Piped Connection to an Existing System[7]	$52,400 - $185,500	$59,700 - $192,800
Trucked Water[8]	$950	$350,000
Bottled Water[9]	$1,339	$1.34 M
RELOCATE HOUSEHOLDS[10]	$15,090	$15.1 M
ANCILLARY ACTIVITIES		
Well Water Quality Testing[11]	$15 - $50	N/A
Dual System[12]	$575 - $1,580	$260,000 - $900,000

[1-12] See Table 36 and 37 for references.

The capital and operations and maintenance (O&M) costs were researched for each alternative water supply option. The capital costs for household and community treatment include process, construction, engineering and indirect costs, but do not include planning and preliminary alternative analysis costs (U.S. EPA 2000; U.S. EPA 2005). Alternative supply cost estimates were estimated in accordance with U.S. EPA cost estimation procedures (U.S. EPA 2000) and from historical project estimates.

Capital costs were converted to annualized capital costs ($/1,000 gallons or $/1 kgal) based on the following equation:

$$\text{Annualized Capital Cost (\$/kgal)} = \frac{\{\, Capital\ Cost\ (\$) * Amortization\ Factor \,\}}{\{\, Flow\ (mgd) * \frac{1000\ gal}{Mgal} * \frac{365\ days}{year} \,\}}$$

Adminsitrative Record
Page 35545

An amortization value of 0.0802 was used which corresponds with an interest rate (i) of 5% over 20 years (N), represented by the following equation:

$$\text{Amortization Factor} = \frac{i*(1+i)^N}{((1+i)^N - 1)}$$

Annual O&M costs were converted to annualize O&M costs based on the following equation:

$$\text{Annualized O\&M Cost (\$/kgal)} = \frac{O\&M Cost\ (\$)}{\{Flow\ (mgd) * \frac{1000\ gal}{Mgal} * \frac{365\ days}{year}\}}$$

The total annualized cost of each alternative water supply option equals the sum of the annualized capital and O&M costs.

5.1.1 Blending

To estimate blending costs for nitrate compliance, a report written by Kennedy/Jenks was used that based "bid" cost estimates primarily for nitrate problems (Kennedy/Jenks 2004). They used a blending design case for a 14 inch casing well with a flow rate range of 300-1,200 gpm and a well depth of 700. This report uses the costs corresponding with flow rates of 300 and 600 gpm as the low and high estimate for blending costs for a 1,000 household community. The cost for obtaining a new low-nitrate source is not included in this cost estimate. The original capital and O&M costs are in 2003 dollars and are projected to 2010 dollars using the 2010 ENR CCI. The capital cost incorporates the cost for the basic blending facility and is estimated to cost about $131,000 for each flow rate. Indirect construction costs such as engineering, contingency, and permitting costs were estimated to be about 25 percent of the estimated bid costs (Kennedy/Jenks 2004). The O&M cost for blending water was estimated at $250 per acre-foot (af), costing approximately $50,000 and $100,000 (2003 dollars) annually for each flow rate, respectively. Table 23 shows the resulting blending costs, with annual costs estimated as $83,000 and $148,000, for each flow rate, or $0.52 and $0.47 per kilogallon, respectively. These blending estimates are for a 3,300 person (or 1,000 household) community, as blending is only recommended for public water systems with more than one well and the ability to obtain a low-nitrate source.

Adminsitrative Record
Page 35546

Table 23. Estimated costs for blending.

Itemized Cost	Low Estimate[1]	High Estimate[2]
2003 Capital Cost[3]	$131,000	$131,000
Indirect Construction Cost[4]	$32,750	$32,750
2010 Capital Cost[5]	$216,000	$216,000
2010 O&M Cost[6]	$$65,000	$130,000
Annualized Cost ($/kgal)[7]	$0.52	$0.47
Annualized Cost ($/year)[8]	**$83,000**	**$148,000**

[1] 14" casing well at 700' deep, 300 gpm. A single blending station is assumed for each source. These costs are based on similar projects implemented by Kennedy/Jenks, primarily for nitrate.
[2] 14" casing well at 700' deep, 600 gpm. A single blending station is assumed for each source. These costs are based on similar projects implemented by Kennedy/Jenks, primarily for nitrate.
[3] Capital costs include the construction "bid" costs for constructing the blending facilities.
[4] 25% indirect construction costs.
[5] The 2003 Capital Costs plus 25% Indirect Construction Cost and projected to 2010 costs using the 2010 ENR CCI.
[6] A major O&M cost is the cost of obtaining low-nitrate blending water. These O&M costs assume that there is already an uncontaminated source available for blending. The O&M costs were developed for electrical power, labor, maintenance materials, resin replacement, and monitoring. Labor rates were estimated at an average of $40 per hour, electricity rates were estimate at $0.12/kWh and an annual allowance for maintenance materials was estimated at 1 percent of total capital costs. The average O&M cost within the well range was chosen and projected to 2010 costs using the 2010 ENR CCI.
[7] The cost is annualized over a 20 year period with a 5% annual interest rate and expressed as dollar per kilogallon produced.
[8] The cost is annualized over a 20 year period with a 5% annual interest rate.

5.1.2 Drilling a Deeper Well

The lower bound cost estimates for drilling a deeper well are from a Background Information Document (BID) from U.S. EPA that provides well drilling cost estimates (U.S. EPA 2001). For a domestic well (assumed 10 gpm, 8 inch casing well), the drilling costs are about $50 per foot; and for a public supply well (assumed 700 gpm, 14 inch casing well), the drilling costs are about $110 per foot. Chris Johnson, a principal hydrogeologist, advises that drilling a deeper well can cost almost as much as drilling a new well and provided the upper bound cost estimates of $200 per foot for a domestic well and $1,000 per foot for a public supply well (Johnson 2011). Table 24 shows the estimated costs for drilling a deeper well based on these two references.

Adminsitrative Record
Page 35547

Table 24. Estimated costs for drilling a deeper well.[1]

Itemized Cost	Self-Supplied Household	Public Water System (1,000 hhld)
Drilling Cost ($/foot)	$50 - $200	$110 - $1,000
O&M Cost	$62	$82,000
Annualized Cost ($/kgal)	$6.76 - $25.61	$0.65 - $0.77
Annualized Cost ($/yr)	**$860 - $3,300**	**$84,000 - $98,000**

[1] The lower bound estimates are from the U.S. EPA Yucca Mountain BID Document: *Well Drilling and Pumping Costs* (U.S. EPA N.D.). A domestic well is assumed to be a 10 gpm, 8 inch casing well, originally 300 feet deep and deepened to 500 feet. A public supply well is assumed to be a 700 gpm, 14 inch casing well, originally 500 feet deep and deepened to 700 feet.

5.1.3 Drilling a New Well

The annualized total cost for drilling a new domestic well is based on cost estimates provided by an experienced senior geologist, and is shown in Table 25. The U.S. EPA 2007 Survey and Assessment was used for estimating the costs for drilling a new public supply well, shown in Table 26. New public supply well costs include pump and appurtenances, but do not include well houses. The Survey and Assessment provides the following cost functions for new wells and for rehabilitating existing wells (U.S. EPA 2007):

$$New\ Well\ Cost = e^{13.6502} * D^{0.56445}$$

$$Well\ Rehabilitation\ Cost = e^{11.72961} * D^{1.59738}$$

D is the design capacity of the well in millions of gallons per day (mgd), and *e* represents the exponential function (approximately 2.72) (Goldstein et al. 2006).

The annual O&M cost is assumed to be included in the model. With these functions, rough cost estimates can be made for the lower bound of system categories and a multiplication factor of seven is applied to estimate the upper bound cost. Estimated costs for a new public supply well are listed in Table 26 and estimated costs for public supply well rehabilitation are listed in Table 27.

Table 25. Annualized total cost ranges for drilling a new domestic well.

Option	Drill a New Well[1]
Initial Capital Cost ($/hhld)	$25,000 - $40,000
Annual O&M Cost ($/hhld)	$60
Total Annualized Cost ($/kgal)	$16.17 - $25.59
Total Annualized Cost ($/hhld)	**$2,100 -$3,300**

[1] Initial Capital Cost Estimates from David W. Abbott, Senior Geologist at Todd Engineers (Abbott 2011). Annual O&M Estimate: Assumed 300 foot well depth, 0.6 pump efficiency, and $0.15/kWh. Annualized over 20 years.

Adminsitrative Record
Page 35548

Table 26. Annualized total cost ranges for drilling a new public supply well.

U.S. EPA System Size Classification	Low Cost Range for a New Well ($/kgal)[1]	High Cost Range for a New Well ($/kgal)[2]
Very Small (25 - 500 people)	$0.44 - $1.60	$3.11 - $11.17
Small (501 - 3,300 people)	$0.20 - $0.44	$1.38 - $3.11
Medium (3,301 - 10,000 people)	$0.12 - $0.20	$0.86 - $1.38
Large (10,001 - 100,000 people)	$0.05 - $0.12	$0.32 - $0.86

[1] U.S. EPA 2007 Drinking Water Infrastructure Needs Survey and Assessment.
[2] U.S. EPA 2007 Drinking Water Infrastructure Needs Survey and Assessment multiplied by a factor of 7 to estimate engineering fees, well demobilization, etc.

Table 27. Annualized total cost ranges for public supply well rehabilitation.

U.S. EPA System Size Classification	Low Cost Range for Well Rehabilitation ($/kgal)[1]	High Cost Range for Well Rehabilitation ($/kgal)[2]
Very Small (25 - 500 people)	$0.01	$0.01 - $0.07
Small (501 - 3,300 people)	$0.01 - $0.03	$0.07 - $0.22
Medium (3,301 - 10,000 people)	$0.03 - $0.06	$0.22 - $0.42
Large (10,001 - 100,000 people)	$0.06 - $0.22	$0.42 - $1.62

[1] U.S. EPA 2007 Drinking Water Infrastructure Needs Survey and Assessment.
[2] U.S. EPA 2007 Drinking Water Infrastructure Needs Survey and Assessment multiplied by a factor of 7 to estimate engineering fees and other contingencies.

5.1.4 Community Treatment

The estimated costs for system application of ion exchange and reverse osmosis are shown in Table 28. Cost ranges shown are compiled from detailed literature review and survey responses, as described in Technical Report 6 (Jensen et al. 2012), and represent the lowest and highest cost found for each system size. Any inconsistencies in cost ranges are attributed to the lack of cost information currently available. The economies of scale exist with ion exchange and reverse osmosis treatment systems as the cost per kilogallon decreases with increasing capacity. The cost per unit of produced water decreases as system size increases, however larger treatment systems incur higher total capital and O&M costs. Ion exchange is the cheaper option. Refer to Technical Report 6 (Jensen et al. 2012) for additional details on treatment costs.

Adminsitrative Record
Page 35549

Table 28. Annualized total cost ranges for groundwater treatment systems.[1]

U.S. EPA System Size Classification (population)	Annualized Total Cost Range for Ion Exchange [$/kgal][2]	Annualized Total Cost Range for Ion Exchange [$/yr][2]		Annualized Total Cost Range for Reverse Osmosis [$/kgal]	Annualized Total Cost Range for Reverse Osmosis [$/yr]
Very Small (< 501)	$0.62 - $4.60	$2,100 - $285,000		$0.69 - $19.16	$2,300 - $1.2 M
Small (501 - 3,300)	$0.30 - $2.73	$21,000 - $1.1 M		$0.58 - $1.34	$36,000 - $533,000
Medium (3,301 - 10,000)	$0.36 - $2.04	$143,000 - $2.4 M		$1.35 - $3.39	$537,000 - $4.0 M
Large (10,001 - 100,000)	$0.22 - $1.81	$258,000 - $20.1 M		$0.73 - $3.67	$855,000 - $40.8 M

[1] Cost information is an excerpt from Technical Report 6 (Jensen et al. 2012).

[2] Disposal costs were not included in the U.S. EPA cost estimates of IX for arsenic removal (used to estimate nitrate removal).

U.S. EPA's 2007 Drinking Water Infrastructure Needs Survey and Assessment estimate the cost for a SCADA system by the following equation:

$$Cost\ of\ SCADA = e^{7.7799} * population^{0.48453}$$

The *e* is the exponential function (approximately 2.72).[21] The lower bound annualized cost of SCADA infrastructure was estimated and is shown in Table 29. As the system size increases the cost for SCADA becomes more affordable.

Table 29. Lower bound annualized cost of SCADA infrastructure.[1]

U.S. EPA System Size Classification	Annualized Total Cost for SCADA Infrastructure [$/kgal]	Annualized Total Cost for SCADA Infrastructure [$/yr]
Very Small (25 - 500 people)	$4.99	$11,400
Small (501 - 3,300 people)	$1.06	$49,000
Medium (3,301 - 10,000 people)	$0.42	$116,000
Large (10,001 - 100,000 people)	$0.23	$633,000

[1] U.S. EPA Drinking Water Infrastructure Needs Survey and Assessment (2007).

5.1.5 Household Treatment

The 2010 U.S. EPA POU/POE cost model was used to estimate the costs for installing a reverse osmosis POU device. The cost model assumes an average per capita water consumption of 100 gallons per

[21] Goldstein, Lay, Schneider, and Asmar, *Brief calculus and its applications,* 11th ed., Prentice-Hall, 2006.

Adminsitrative Record
Page 35550

person per day, 2.6 people per household, a discount rate of 7%, and a discount period of 10 years (the lifetime of the unit) (U.S. EPA 2010). The cost model includes equipment installation, laboratory analyses, indirect capital costs, equipment maintenance, and public education and outreach. Table 30 shows the estimated cost per household for installing a reverse osmosis POU unit for one household and for 1,000 households. The lower bound estimate does not incorporate the public education and outreach that is required for systems using a POU device for emergency purposes to comply with drinking water quality standards (discussed in Section 7 Regulatory and Implementation Implications). For a self-supplied household it is cost-effective to not incorporate public education and outreach; however, for a 1,000 household (or connection) public water system it would be beneficial to incorporate public outreach to educate the user and ensure proper procedures are followed.

Table 30. U.S. EPA reverse osmosis POU device cost estimates.

Options	RO POU Cost Estimate (Single Household)[1]	RO POU Cost Range (1,000 Household System)[1]
Initial Capital Cost ($/hhld)	$406 - 1,981	$493 - $494
Annual O&M Cost ($/hhld)	$197 - $1,781	$144 - $145
Total Annualized Cost ($/kgal/hhld)	$3.01 - $24.85	$2.52 - $2.53
Total Annual Cost ($/hhld)	**$250 - $2,038**	**$214 - $215**

[1] Uses the 2010 U.S. EPA Cost Estimate Tool for an NSF/ANSI Certified RO Unit for nitrate treatment. The lower bound estimate includes unit purchase, installation, scheduling time, indirect costs (permitting, pilot testing, legal, engineering, and contingency) and all associated operations and maintenance costs. The upper bound estimate includes all listed above and public education (technical and clerical labor and printed material for all public outreach/education efforts). The tool allows for economies of scale.

For comparison, a quote from Culligan estimates a typical nitrate filter to cost about $360 per year including maintenance for one household, cost details are shown in Table 31 (Culligan 2011). The total annualized cost ($/kgal) is greater than the U.S. EPA cost model estimates because of the difference in maximum potable water consumption. Culligan also rents RO POU devices for about $26 to $36 per month, plus filter replacement and service fees (Culligan 2011). An RO POU device is estimated to cost between$250 and $360 annually discounted over 10 years at a rate of 0.07. For households unable to pay the initial capital, the next best option would be to rent a NSF/ANSI certified RO POU device for at least $430 per year (plus filter replacement and service fees).

Adminsitrative Record
Page 35551

Table 31. Culligan reverse osmosis POU device cost estimates.

Options	RO POU Cost Range (1 hhld)[1]
Initial Capital Cost ($/hhld)	$1,200
Annual O&M Cost ($/hhld)	$191
Total Annualized Cost ($/kgal/hhld)	$39.67
Total Annual Cost ($/hhld)	**$362**

[1] Culligan offers a lifetime warranty on the membrane and pre-filters. It costs about $1,100 to $1,200 to purchase the unit and the pre-filter needs to be replaced every 18 months and the main filter needs to be replaced every 3 years (all factored into the costs listed). The maximum potable water consumption of 25 gallons and a discount rate of 0.07 and discount period of 10 years are assumed.

Currently, there is no POE unit certified for treating nitrate so the U.S. EPA Cost Estimate Tool for an NSF/ANSI Certified POE unit for treating Radium was used to estimate the annual cost of a POE device. The U.S. EPA Cost Estimate Tool provides an annualized cost of $397 for one household to install and maintain a POE device (U.S. EPA 2010). This includes the unit purchase, installation, scheduling time, indirect costs (permitting, pilot testing, legal, engineering, and contingency) and all associated operations and maintenance costs. Future research is necessary to evaluate the true cost of a POE device used for treating nitrate. The costs for POU and POE devices are further discussed in Section 5.2.1 Self-Supplied Households or Local Small Water Systems.

5.1.6 Connect to an Alternative System

Most costs for a system to connect to an alternative system are for installing the pipeline. An estimated pipeline cost of $61 per foot (Monterey County Environmental Health 2010b) and a connection fee on the order of $100,000 are assumed for estimating the costs of connecting to an alternative system. Table 32 represents the annualized cost range estimates for a system with 1,000 connections to connect to an alternative system. The low estimate (or lower bound) is solely pipeline costs and a connection fee and the high estimate (or upper bound) includes pipeline costs, a connection fee, and engineering and administration costs as 43% of the capital (Summers Engineering 2011). These costs are essentially the same if a single household connects to a public water system (estimates shown in Table 22 and Table 36 used a $9,000 connection fee for a self-supplied household); however, the party responsible for the costs would vary by system size and acquisition policies. More involved mergers with a larger system will often require significant additional upgrades to the smaller distribution system.

Adminsitrative Record
Page 35552

Table 32. Estimated annualized cost ranges for a system with 1,000 connections to connect to an alternative system.

Pipeline Distance (miles)	Capital Costs[1]	Engineering & Administration Costs[2]	Total Annualized Cost ($/year)[3]	Annualized Cost ($/kgal)[4]	Annualized Cost ($/hhld)[5]
0.5	$161,040	$69,200	$20,900 - $26,500	$164 - $207	$20.90 - $26.50
2	$644,160	$277,000	$59,700 - $81,900	$467 - $641	$59.70 - $81.90
5	$1,610,400	$692,500	$137,200 - $192,800	$1,074 - $1,509	$137.20 - $192.80
10	$3,220,800	$1,384,900	$266,500 - $377,600	$2,086 - $2,956	$266.50 -$377.60

[1] Pipeline costs at $61 per foot for 1,000 households.
[2] Excerpt from Kettleman City Proposed Surface Water Treatment Plant and Commercial Tank Facility (March, 2011) (Summers Engineering 2011). 43% of the Subtotal: 15% Construction Contingencies; 2% Construction Application, CDPH Information & Labor and Compliance Monitoring; 2% Environmental Documentation & Legal Review; 10% Design & Project Bidding; 7% Project Administration; 5% Project Inspection; and 2% Project Surveying & Geotechnical Testing.
[3] Lower bound includes pipeline and $100,000 connection fee. Upper bound includes pipeline, $100,000 connection fee, and engineering and administration costs. Costs are discounted over 20 years at a 5% discount rate.
[4] Assumes a consumption rate of 350 gallons per household per day.
[5] Total annualized cost divided by 1,000 households.

5.1.7 Trucked Water

An estimate from RMR Water Trucks for providing their water trucks for service is used for estimating the delivered trucked water costs (RMR Trucks 2010):

($100/hr truck driver)* (x hr travel time) + (truck size (gal))*($0.xx/gal of nearby water supply)

Estimates are provided for a small community public water system serving 1,000 households and for a single household located in Tulare County. The estimated cost for providing a community in Tulare County with a single RMR 7,000 gallon water truck (travel time is estimated as 4 hours roundtrip and a nearby water supply is assumed to cost $0.35 per gallon) is about $2,850, or $410/kgal. Assuming one household uses about 2.25 gallons per day,[22] 1,000 households (or a small community water system) would receive water for about three days ($2.85/household). It would cost approximately $350,000 to provide 1,000 households with trucked potable water for one year. To provide one household with water, a 500 gallon RMR truck would cost $575 ($1,150/kgal), and would provide the household with water for 222 days (assuming storage is available). It would cost approximately $950 to provide one household with trucked potable water for one year.

5.1.8 Bottled Water

To estimate the cost of bottled water, the National Academy of Sciences Hydration Study (2004) was used, assuming 3.3 people per household and predicting about 2.25 gallons per household-day needed

[22] NAS Hydration Study Estimate – 3.3 people per household.

Adminsitrative Record
Page 35553

for potable uses. Vended or bottled water can cost $0.25 to $1.30 per gallon, not including transportation costs (Pacific Institute et al. 2010). A common low price for water delivered near the city of Visalia by Alhambra Water is a 5 gallon bottle at $1.63 per gallon (Alhambra Water 2010). The annual cost for a household receiving Alhambra Water is about $1,260. For accuracy and consistency, this cost estimate is used as the cost of bottled water in this report.

5.1.9 Relocate to Area with Better Water Supply

To estimate the costs of relocating a community to an area with a better water supply, the true market value of houses in each respective county is evaluated. The range of average listing prices for houses in each county is shown in Table 33 and the median value of average ranges is used to estimate the cost of relocating a single household (Trulia 2011). To better represent Salinas Valley, the City of Salinas' average listing prices will be used, instead of Monterey County. Cost scenarios for relocating households are shown in Table 34, and the average cost across the study area to relocate one household is $188,000. Using this average, it would cost about $2 billion to relocate the susceptible population on self-supplied and local small water systems (about 10,000 households). It is estimated to cost $37.6 million to relocate 200 households per county. The costs for relocating a household will differ slightly between a homeowner and a renter. Renters should be cheaper to move if the County condemns the property and there may also be less attachment or sentimental value to the home. However, the total loss involved in relocation is probably similar for both homeowners and renters.

Table 33. Average listing prices for homes in study area counties.

County	Range of Average Listing Prices for Houses ($1,000)[1]
Fresno	$158 - $193
Kern	$137 - $167
Kings	$136 - $166
Tulare	$153 - $188
Monterey (City of Salinas)	$525 - $642 ($261 - $319)

[1] Trulia, Inc. *Range of Average House Listing Prices* (August, 2011)

Table 34. The estimated cost for relocating households.

County	Single Household Relocation ($1,000)[1]	200 Household Relocation ($1,000)[1]
Fresno	$176	$35,100
Kern	$152	$30,400
Kings	$151	$30,200
Tulare	$171	$34,100
City of Salinas	$290	$116,700
Study Area Average	**$188**	--

[1] Trulia, Inc. The median of the average range of listing prices (August, 2011).

Adminsitrative Record
Page 35554

5.1.10 Dual Water Distribution System

A dual water distribution system would require a household to continue paying for their contaminated water, using it only for non-potable uses, and to purchase an alternative supply. The four alternatives available for a dual water distribution system are: 1) purchasing bottled or vended water; 2) installing a POU device; 3) trucking in potable water or; 4) installing water system treatment for the potable supply and delivering it through a secondary distribution network. Given the nature of trucked water and the need for storage, it is not a feasible alternative for a dual water distribution system. To estimate the costs of a dual water distribution system, an average monthly residential water rate of $27 is assumed for the non-potable supply cost.[23] A dual system including the purchase of 5 gallon bottles from Alhambra would cost approximately $1,582 annually per household. A dual system including the purchase of a POU device would cost about $574 to $686 annually per household, ranging from the lower bound U.S. EPA POU Cost Model value to the Culligan POU quote.

If a dual water distribution system was installed for an existing 1,000 household water system, the total costs would include the cost for the contaminated supply, treating the potable supply, installing a distribution system to each connection, and re-plumbing each household for the distribution of the new supply to the bathroom and kitchen. Table 35 shows rough estimates for the cost of installing a dual water distribution system for a 1,000 household system. The total annualized cost per household is estimated to cost between $550 and $900.

[23] California Water Company – Visalia, $27 a month for using 6,000 square feet or less per month.

Adminsitrative Record
Page 35555

Table 35. Estimated annualized cost ranges for a dual water distribution system.

Annualized Costs[1]	Cost Range for Dual Water Distribution for 1,000 Household Water System[2]
Contaminated Supply Costs ($/year)[3]	$324,000
Treatment Costs ($/year)[4]	$644,904
Pipeline Costs ($/year)[5]	$130,000 - $260,000
Re-plumbing Costs ($/year)[6]	$200,000 - $320,000
Engineering & Administration Fees ($/year)[7]	$165,000 - $270,000
Total Annualized Cost ($/year)[8]	$550,000 - $900,000
Household Total Annualized Cost ($/hhld-year)[9]	$550 - $900

[1] Costs are discounted at a 5% discount rate, over a 20 year period.
[2] Assumes a water system serves 1,000 households, with 3.3 people per household, and uses 0.20 million gallons of potable water per day, including showering and toilet flushing (112 gallons per capita per day) (Gleick 2003).
[3] Monthly water supply cost for California Water Company in Visalia.
[4] Costs for treating 0.20 mgd of water using an ion exchange treatment system - includes O&M costs (Technical Report 6, Jensen et al. 2012).
[5] Assumes 6" pipe costs $61 per foot and does not include excavation costs. Lower bound estimate uses 5 miles of pipeline and upper bound estimate uses 10 miles of pipeline (Granite Ridge Regionalization Feasibility Study, 2010).
[6] Rough cost estimates provided by ZURN and an experienced plumber for PEX piping to be installed in a 1200 sq. ft. 3 bed, 2 bath house within the kitchen and both bathrooms. The lower bound estimate is for a tract type house with raised wood floors (ease of "popping" pipes through the floor cabinets), and the upper bound estimate is for a slab house (requiring the sheet rock to be cut, patched and painted after installation). This is a rough estimate and each house will vary based on the chosen plumber's site estimate.
[7] Excerpt from Kettleman City Proposed Surface Water Treatment Plant and Commercial Tank Facility (March, 2011). 43% of the Subtotal (treatment, pipeline, and re-plumbing costs): 15% Construction Contingencies; 2% Construction Application, CDPH Information & Labor and Compliance Monitoring; 2% Environmental Documentation & Legal Review; 10% Design & Project Bidding; 7% Project Administration; 5% Project Inspection; and 2% Project Surveying & Geotechnical Testing.
[8] Total annualized cost for all items. Lower bound estimate provides costs for a 1,000 connection system with a service area having a 5 mile distance and consisting of only tract type houses (the economies of scale involved in community tract type housing is ignored here, but could be represented in a true situation where dual plumbing occurs for a whole development). Upper bound estimate provides costs for a 1,000 connection system with a service area having a 10 mile distance and consisting of only slab houses.
[9] Total annualized cost per household.

5.1.11 Well Water Quality Testing

PurTest sells a water test kit for bacteria and nitrate based on U.S. EPA methods for $13 with a basic knowledge booklet that could be used for domestic well users (Home Depot 2011). The Environmental Health Investigations Branch of CDPH estimates certified laboratory water quality tests to cost approximately $50 for testing a private well (CDHS 2000). It is recommended that private wells are sampled once a year, preferably between April and July when nitrate levels are generally the highest (CDHS 2000).

Adminsitrative Record
Page 35556

These two estimates are the lower and upper bound estimates for domestic well water quality testing. County-specific estimates for a State-certified laboratory can be found on the CDPH Environmental Laboratory Accreditation Program (ELAP) website.

5.1.12 Summary of Alternative Water Supply Costs

The economic feasibility of each option will vary based on the effectiveness, benefits, savings, and costs expected from a candidate system. This section only summarizes the expected costs for alternative water supply options, but a true engineering analysis will include economic feasibility studies examined over a project's lifetime. The alternative water supply options for providing a self-supplied household with low-nitrate water all year, ranked from least expensive to most expensive, typically are to:

1) install a POU RO unit;
2) drill a deeper well;
3) install a dual water system;
4) purchase bottled water;
5) drill a new well;
6) relocate the household; and
7) install pipeline and connect to an existing system.

The estimated costs for self-supplied household alternative water supply options are shown in Table 36.

The alternative water supply options for providing a small community public water system (serving 1,000 households) with low-nitrate water all year, ranked from least expensive to most expensive, typically are to 1) drill a new well; 2) install a pipeline and connect to an existing system; 3) drill a deeper well; 4) implement a community groundwater treatment system; 5) blend sources; 6) provide households with POU RO units; 7) construct a dual water distribution system; 8) purchase bottled water; and 9) relocate households. The estimated costs for alternative water supply options for a small community public water system are shown in Table 37. The analysis performed here is general and includes many necessary assumptions. The costs for each option need to be assessed on a system specific basis before selecting the most economical option.

The estimated annual O&M cost ranges for alternative water supply and treatment options that include O&M are shown in Table 38. The cost ranges are shown as percent of the total annualized cost to reflect the wide range of researched and surveyed cost data gathered. For the estimated basin-wide solutions (Section 6 Basin-wide Costs of Nitrate Contamination), the average percentage of O&M is applied to each solution to roughly estimate the basin-wide O&M cost.

The lifetime of each alternative will vary depending on the existing water quality, soil properties, water usage, and existing source supply. An alternative should be evaluated on the least cost and lifetime of the system before choosing the best option for implementation. If a system blends sources to provide nitrate compliant water, the duration of the solution will last only as long as the low-nitrate source remains low and proper source ratios are maintained. Nitrate contamination and the associated groundwater degradation are expected to worsen over the next few decades and it is reasonable to

Adminsitrative Record
Page 35557

expect that, over time, a low-nitrate source will approach the MCL. If an existing well is drilled deeper into a different aquifer, the lifetime of the safe supply will depend on the time it takes for nitrate in shallow groundwater to reach the new depth. There is the risk of the existing shallow nitrate migrating down and there is the risk of increased nitrate found at the new depth. If a new well is properly designed, constructed, developed and completed, it can last for up to 50 years (Harter 2003); however, pumping from a new well can quickly draw down an aquifer and draw nitrate into the existing safe supply. Depending on the type of treatment chosen (IX or RO), a community treatment system can last for 20 years (with the proper maintenance and membrane and resin lifetime will vary with water quality and pretreatment measures). With proper maintenance and replacement of filters, a household treatment system can last up to 10 years. Connecting to an alternative system ensures that future problems of nitrate contamination (or other water quality contaminants) can be managed more easily due to economies of scale. Switching to surface water shifts the problem of nitrate contamination in groundwater to other contaminants (i.e., *giardia lamblia* and *cryptosporidium parvum*) which will need to be addressed with surface water treatment. Constructing a dual water distribution system on a community public water system scale treats less water, expends less energy, and conserves resources.

Table 36. Summary of the estimated alternative water supply costs for self-supplied households.

Option	Estimated Annual Cost Range For a Self-Supplied Household[1]
IMPROVE EXISTING WATER SOURCE	
Blending[2]	N/A
Drill Deeper Well[3]	$860 - $3,300
Drill a New Well[4]	$2,100 - $3,300
Community Supply Treatment[5]	N/A
Household Supply Treatment[6]	$250 - $360
ALTERNATIVE SUPPLIES	
Pipeline and Connection to an Existing System[7]	$52,400 - $185,500
Trucked Water[8]	$950
Bottled Water[9]	$1,339
RELOCATE HOUSEHOLDS[10]	$15,090
ANCILLARY ACTIVITIES	
Well Water Quality Testing[11]	$15 - $50
Dual Water System[12]	$574 - $1,582

[1] All costs are discounted over a 20 year period at a 5% discount rate, except for the RO POU estimate and trucked and bottled water costs.
[2] Blending is only considered for public water systems with more than one source.
[3] The lower bound estimate is from the U.S. EPA Yucca Mountain BID (U.S. EPA N.D.) with estimated drilling costs of $50 per foot. The upper bound estimate is a quote from an experienced hydrogeologist, Chris Johnson; estimated drillings costs of $200 per foot. Annual O&M costs estimated using a pumping well energy equation and assuming $0.15/kWh.
[4] Capital costs estimated from senior geologist David Abbott (Abbott 2011). Annual O&M costs estimated using a pumping well energy equation and assuming $0.15/kWh.
[5] Community supply treatment only refers to community drinking water systems (≥15 connections).
[6] Uses the 2010 U.S. EPA Cost Estimate Tool for 1 NSF/ANSI Certified Reverse Osmosis Point-of-Use Unit. The lower bound estimate includes unit purchase, installation, scheduling time, indirect costs (permitting, pilot testing, legal, engineering, and contingency) and all associated operations and maintenance costs. The upper bound estimate includes all lower bound costs plus public education (technical and clerical labor and printed material for all public outreach/education efforts). Assumes a 10 year lifespan for the unit and is discounted for

Adminsitrative Record
Page 35558

10 years at a 7% discount rate.

[7] Only considers the costs for installing pipeline and the connection fees for connecting to an existing system that has a safe drinking water supply. The lower bound estimate assumes pipeline costs of $61 per foot for a distance of 2 miles, and a $9,000 connection fee per household. The upper bound estimate assumes pipeline costs of $61 per foot for a distance of 5 miles, and a $9,000 connection fee plus engineering and administration costs (43% of the pipeline costs).

[8] Assumes a 500 gallon RMR Water Truck travels from Castaic to Tulare County for a 4 hour roundtrip at $100/hour and purchases 500 gallons of a local, safe drinking water supply at $0.35 per gallon. A one-time 500 gallons cost. Does not include the cost for storage.

[9] Assumes Alhambra in Visalia delivers 5 gallons of "Crystal Fresh" water to a location in Visalia with 3 people per household. Each person consumes about 0.7 gallons per day for 365 days.

[10] The median listing prices for houses in each county (City of Salinas was used instead of Monterey County) were examined and the average listing for a house in the study area is estimated to be $188,000 (trulia.com).

[11] Well water quality test for nitrate and bacteria from PurTest sold at Home Depot ($13) and CDPH estimate of $50 for a private well nitrate sample from a State-certified laboratory. All public water systems (≥ 15 connections) are already required to sample and monitor their water.

[12] Lower bound estimate is the U.S. EPA POU Cost Estimate tool plus the monthly cost of the contaminated supply and the upper bound estimate is the cost for bottled water (Culligan – 5 gallon bottle) plus the monthly cost of the contaminated supply (Visalia Community Water Center is used for the reference, however this is not meant to suggest that Visalia CWC's water is contaminated).

Table 37. Summary of the estimated alternative water supply costs for a small water system (1,000 households).

Option	Estimated Annual Cost Range For a Small Water System (1,000 hhlds)[1]
IMPROVE EXISTING WATER SOURCE	
Blending[2]	$200,000 - $365,000
Drill Deeper Well[3]	$80,000 - $100,000
Drill a New Well[4]	$40,000 - $290,000
Community Supply Treatment[5]	$95,000 - $105,000
Household Supply Treatment[6]	$223,000
ALTERNATIVE SUPPLIES	
Pipeline and Connection to an Existing System[7]	$59,700 - $192,800
Trucked Water[8]	$2,850
Bottled Water[9]	$1.34 M
RELOCATE HOUSEHOLDS[10]	$15.1 M
ANCILLARY ACTIVITIES	
Dual Water Distribution System[11]	$500,000 - $900,000

[1] All costs are discounted over a 20 year period at a 5% discount rate, except for the RO POU estimate and trucked and bottled water costs.

[2] A 14" casing well with flow rates of 300 gpm (lower bound estimate) and 600 gpm (upper bound estimate) and well depth of 700 feet. Does not include the cost of obtaining a low-nitrate source. O&M for blending estimated at $250 per acre-foot and indirect costs are estimated to be about 25% of the estimated bid costs. Kennedy/Jenks (2004) "bid" cost estimates primarily based on nitrate problems.

[3] The lower bound estimate is from the U.S. EPA Yucca Mountain BID (U.S. EPA N.D.) with estimated drilling costs of $110 per foot. The upper bound estimate is a quote from an experienced hydrogeologist, Chris Johnson; estimated drillings costs of $1,000 per foot. Annual O&M costs estimated using a pumping well energy equation and assuming $0.15/kWh.

[4] Costs estimated from the 2007 U.S. EPA Drinking Water Infrastructure Needs Survey & Assessment (O&M assumed to be included in the cost model); projected to 2010 dollars using the 2010 ENR CCI. The upper bound estimate is from applying a multiplication factor of 7 to estimate engineering fees, well demobilization, etc.

[5] Cost estimates from Technical Report 6 (Jensen et al. 2012). Disposal costs were not included in the U.S. EPA cost estimates of ion exchange for arsenic removal (that was used to estimate nitrate removal).

Adminsitrative Record
Page 35559

[6] Uses the 2010 U.S. EPA Cost Estimate Tool for 1,000 NSF/ANSI Certified Reverse Osmosis Point-of-Use Unit. The lower bound estimate includes unit purchase, installation, scheduling time, indirect costs (permitting, pilot testing, legal, engineering, and contingency) and all associated operations and maintenance costs. The upper bound estimate includes all lower bound costs plus public education (technical and clerical labor and printed material for all public outreach/education efforts). Assumes a 10 year lifespan for the unit and is discounted for 10 years at a 7% discount rate.

[7] Only considers the costs for installing pipeline and connection fees for connecting to an existing system that has a safe drinking water supply. The lower bound estimate assumes pipeline costs of $61 per foot for a distance of 2 miles, and a $100,000 connection fee. The upper bound estimate assumes pipeline costs of $61 per foot for a distance of 5 miles, and a $100,000 connection fee plus engineering and administration costs (43% of the pipeline costs).

[8] Assumes a 7,000 gallon RMR Water Truck travels from Castaic to Tulare County for a 4 hour roundtrip at $100/hour and purchases 7,000 gallons of a local, safe drinking water supply at $0.35 per gallon.

[9] Assumes Alhambra in Visalia delivers 5 gallons of water to a location in Visalia with 3 people per household. Each person consumes about 0.7 gallons per day for 365 days.

[10] The median listing prices for houses in each county (City of Salinas was used instead of Monterey County) were examined and the average listing for a house in the study area is estimated to be $188,000 (trulia.com).

[11] Lower bound estimate is the cost of the contaminated supply, the cost for treating 0.20 mgd (Gleick 2003) for 1,000 households assuming 3.3 people per household, a system pipeline distribution distance of 5 miles, PEX plumbing through tract type houses with raised wood flooring (2 bathrooms and 1 kitchen is re-plumbed per house), and a 43% engineering and administration fee. Upper bound estimate is the cost of the contaminated supply, the cost for treating 0.20 mgd (Gleick 2003) for 1,000 households assuming 3.3 people per household, a system pipeline distribution distance of 10 miles, PEX plumbing through slab houses (2 bathrooms and 1 kitchen is re-plumbed per house), and a 43% engineering and administration fee.

Table 38. Estimated O&M and capital cost ranges for alternative water supply options as percentage of total annual costs.

	O&M % of Total Annual Costs[1]		Capital % of Total Annual Costs[1]	
	Range	Average	Range	Average
POU/POE	50% - 70%	60%	30% - 50%	40%
Blending	78% - 88%	83%	12% - 22%	17%
Drilling a New Well	35% - 75%	55%	25% - 65%	45%
Centralized Nitrate Groundwater Treatment	45% - 85%	65%	15% - 55%	35%
Conventional Surface Water Treatment	15% - 50%	33%	50% - 85%	67%

[1] Refer to references shown in Tables 23-37.

5.2 Least Cost Management

Most alternative water supply option costs largely depend on the size of the system. Even if a system receives assistance in financing the capital costs for an alternative solution, such as treatment, the solution's sustainability can be threatened by high annual O&M costs. To assess the lasting viability of each alternative, in this section the previously discussed cost estimates are compared broadly, and least cost management alternatives are highlighted. The least cost management discussion is divided into domestic well water systems (self-supplied households or local small water systems) and public water systems. This section compares costs for self-supplied households or local small water systems on a cost per household and cost per kilogallon scale. Similarly, the costs for community public water systems are given on a cost per kilogallon and cost per household, but are compared on a system size scale.

Adminsitrative Record
Page 35560

5.2.1 Self-Supplied Households or Local Small Water Systems

The estimated costs for alternative solutions for self-supplied households and local small water systems are shown in Table 39 with the total annualized costs displayed per household and per kilogallon of water. The primary feasible options available to a household or local small water system are to purchase bottled water, install a POU device, drill a new well, or deepen an existing well. The least to most expensive cost alternative for households is to:

- install a reverse osmosis POU treatment device, estimated to cost $250 per year (not including any public outreach or education);
- install a POE device; however, currently no POE devices are NSF/ANSI certified for removing nitrate from drinking water (the estimate given in Table 39 is for treating Radium);
- drill a deeper well; however, the user must continually test and monitor the well to make sure the nitrate contamination does not reach the new depth.

Figure 25 shows the annualized cost curves of each option and highlights the cost-effective option for expected water use or consumption. If a household only requires low-nitrate *potable* water, a POU device is the less expensive solution. However, if a household desires to treat more than 18 gallons of water per day, a POU device is no longer cost-effective, and installing a POE device is preferred. The cost curves do not represent the actual maximum potable water consumption per day for filter pumping capacity, which varies by manufacturer (i.e., Culligan has a maximum of 25 gallons per day).

Adminsitrative Record
Page 35561

Table 39. Estimated household costs for alternative solutions for self-supplied households or local small water systems.

Option	Bottled Water[1]	Drill a New Well[2]	Drill a Deeper Well[3]	POE[4]	POU[5]
Initial Capital Cost ($/hhld)	0	$40,000	$25,000	$2,222	$406
Annual O&M Cost ($/hhld)	0	$60	$232	$109	$197
Total Annualized Cost ($/kgal)	$1,630	$25.59	$17.52	$5.12	$3.01
Total Annualized Cost ($/hhld)	**$1,260**	**$3,300**	**$2,238**	**$397**	**$250**

[1] Quote from Alhambra, Visalia for drinking water delivered in 5 gallon bottles (Alhambra Water 2010). Assumed units of water consumption from NAS Hydration Study.
[2] Initial Capital Cost Estimate: Upper bound estimate of David W. Abbott, Senior Geologist at Todd Engineers (Abbott 2011). Annual O&M Estimate: Assumed 300 foot well depth, 0.6 pump efficiency, and $0.15/kWh. Annualized over 20 years.
[3] Initial Capital Cost Estimate: Average estimate of U.S. EPA Yucca Mountain BID (U.S. EPA N.D.) and Chris Johnson, hydrogeologist (Johnson 2011). Annual O&M Estimate: Assumed 500 foot well depth (originally 300 feet, drilled 200 feet deeper), 0.6 pump efficiency, and $0.15/kWh.
[4] Uses the U.S. EPA Cost Estimate Tool for an NSF/ANSI Certified POE Unit for treating Radium. Includes unit purchase, installation, scheduling time, indirect costs (permitting, pilot testing, legal, engineering, contingency) and all associated operations and maintenance costs.**A POE Unit has not been certified for water systems to distribute so the U.S. EPA Tool does not have a Unit capable of treating nitrate for estimating the cost. Future research is necessary to evaluate the true cost of a POE device.
[5] Uses the U.S. EPA Cost Estimate Tool for 1 NSF/ANSI Certified ROU Unit for treating Nitrate. Includes unit purchase, installation, scheduling time, indirect costs (permitting, pilot testing, legal, engineering, contingency) and all associated operations and maintenance costs. Does not include public education/outreach costs.

Adminsitrative Record
Page 35562

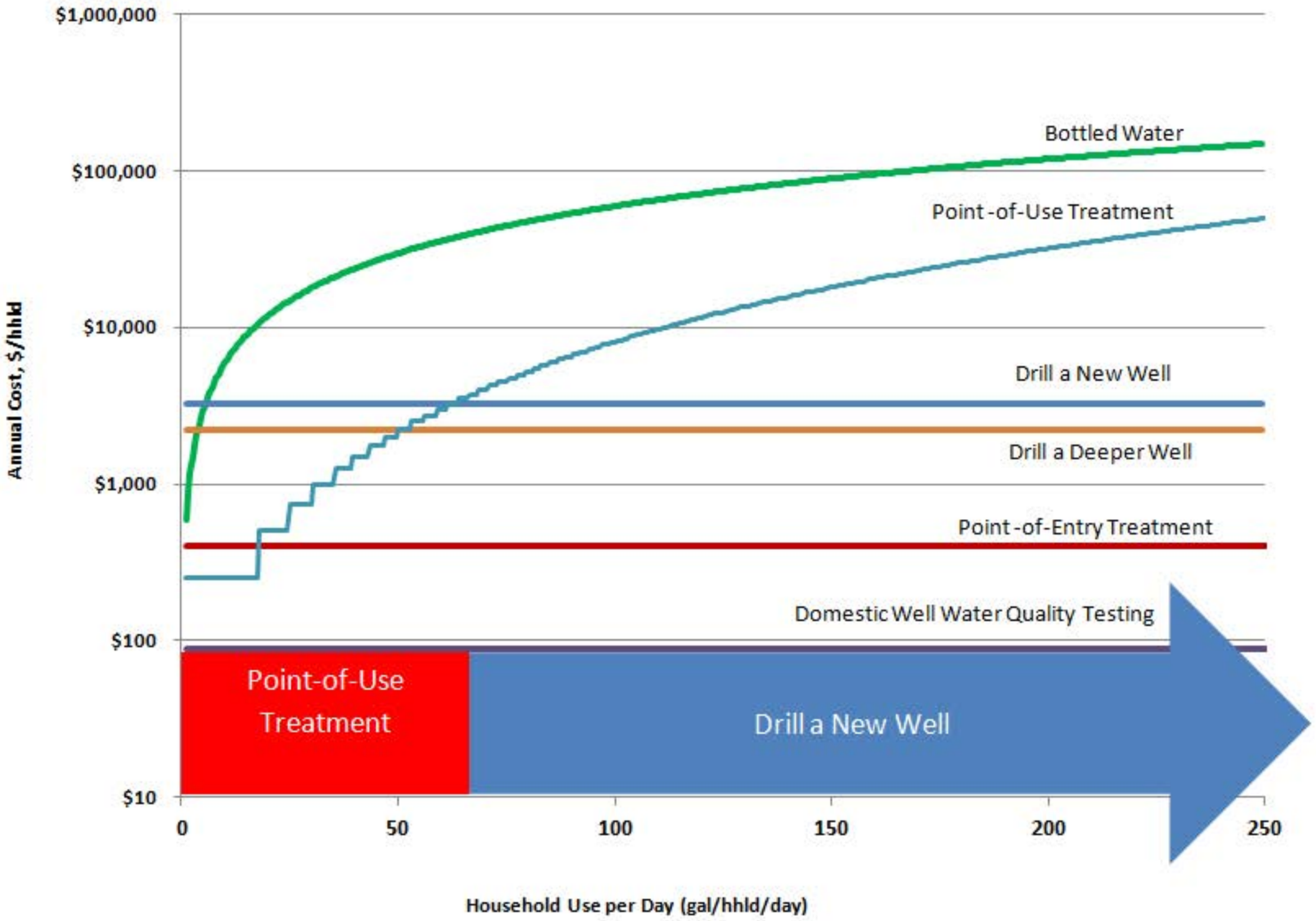

Figure 25. Annualized total household cost for household self-supplied and local small water system alternatives.

Adminsitrative Record
Page 35563

5.2.2 Community Public Water Systems

The minimum and maximum ranges of cost estimates for community public water system categories appear in Table 40 and Table 41, and the compiled cost ranges are shown in Table 42. The annualized low and high cost in dollar per kilogallon ($/kgal) for each alternative is evaluated by U.S. EPA–designated water system size (Table 40 and Table 41). The low cost case is the best case scenario for the least complex system, and the high cost case is more realistic for what the real world average costs would be when all external contingencies are included. For the low cost case, a groundwater treatment facility is the cheapest alternative per kilogallon for very small and small systems. For medium systems, the cost for installing a pipeline to a nearby system becomes very cost-effective, with groundwater treatment and drilling a new well as the next least cost options. POU RO devices are only cost-effective for systems with less than 3,300 people; this coincides well with the 3-year emergency regulations that only allow POU devices to be distributed to systems with less than 200 connections (discussed in Section 7 Regulatory and Implementation Implications). Bottled water costs more than all alternatives (per thousand gallons). Table 40 and Table 41 represent connecting to a better water system only as the pipeline costs and the connection costs are not included in the annualized cost.

For the high cost case, drilling a new well, installing a pipeline to a larger system, or installing a POU RO device are the cheaper options for very small systems. For small and medium systems, a pipeline to a larger system is the cheapest option per kilogallon, but depends on the distance to a nearby system as only two miles of pipeline are assumed. The next best options for small and medium systems are to implement a groundwater treatment facility or drill a new well. For large systems, the cost-effective solutions are to drill a new well or implement groundwater treatment. Bottled water costs more than all alternatives (per thousand gallons).

Figure 26 and Figure 27 graphically show the minimum annualized cost ($/kgal) of alternative options for public water systems; Figure 27 includes pipeline costs and excludes SCADA costs. Similar to the cost table results, Figure 26 shows that for small and medium systems groundwater treatment is the most cost-effective. For medium and large systems an ion exchange treatment facility or a new well are most cost-effective. SCADA costs are shown for treatment and system size comparison. Figure 27 includes the option to construct five miles of pipeline for connecting to another water system or drilling a new well in a low-nitrate location. The economies of scale for pipeline costs can be observed for systems serving more than 3,300 people as pipeline costs decline to less than reverse osmosis costs. A groundwater system with ion exchange remains the cheapest option; however, piping to another system allows a water system to share treatment with a larger entity (i.e., certified operators would already be hired) and the O&M costs would be distributed over a larger population base. Medium and large systems also can search for a low-nitrate source well near the existing system, instead of installing ion exchange treatment.

Figure 28 shows the annualized household cost ($/hhld) of alternative options for public water systems, including consolidation (a small system connecting to a larger system). Costs are shown for a system

Adminsitrative Record
Page 35564

with 2,000 connections (households) and include pipeline costs ($61/foot)[24] and the estimated connection fees (on the order of $150,000), and are discounted over 20 years with an annual discount rate of 0.05. A 50% contingency was attributed to pipeline costs for external expenses, such as permitting activities and legal fees. For a 2,000 connection system, the best option is to drill a new well if the surrounding groundwater quality is acceptable for drinking water purposes. If drilling a new well is not an available option, the next least expensive option is to connect to a larger system that is less than eighteen miles away. If there are no larger systems less than eighteen miles away for a 2,000 connection system to connect to, then implementing an ion exchange system is the cost-effective solution. The maximum distance for connecting to an alternative system will vary with varying system sizes and connection cost estimates. The costs presented and discussed are rough estimates for alternative supply option comparison.

[24] Granite Ridge Regionalization Feasibility Study (Monterey County Environmental Health 2010b).

Adminsitrative Record
Page 35565

Table 40. Low cost ranges for a basin-wide cost analysis.

LOW COST RANGES	Very Small	Small	Medium	Large	[Source]
System Population Range:	**(25 - 500 people)**	**(501 - 3,300 people)**	**(3,301 - 10,000 people)**	**(10,001 - 100,000 people)**	[1]
Assumed Design Rate (mgd):	**0.01 - 0.17**	**0.17 - 1.09**	**1.09 - 3.21**	**3.21 - 30.45**	[2]
Annualized Total Cost ($/kgal):					
Groundwater Treatment	$0.60	$0.30	$0.40	$0.20	[3]
Surface Water Treatment	-	-	-	$0.70 + pipeline	[4]
New Well	$0.44 - $1.60	$0.20 - $0.44	$0.10 - $0.20	$0.05 - $0.10	[5]
Pipeline (2 miles)	$0.80 - $15.70	$0.10 - $0.80	$0.04 - $0.10	$0.01 - $0.06	[6]
Annualized Total Cost for POTABLE USES only ($/kgal):					
POU System for Potable Uses	$2.67 - $3.52	$2.51 - $2.67	$2.51	$2.51	[7]
Bottled Water for Potable Uses	$1,630	$1,630	$1,630	$1,630	[8]

[1] U.S. EPA system size classification.
[2] (U.S. EPA 2005).
[3] (Jensen et al. 2012). These represent the best case scenario of the least complicated system including cheap waste residual disposal and potential blending. Based on the literature and survey data, the average costs for the small and medium system were approximately the same ($0.35/kgal).
[4] (Jensen et al. 2012) (8.25-10 mgd systems).
[5] (U.S. EPA 2007).
[6] (Monterey County Environmental Health 2010b). Does not include connection fee.
[7] (U.S. EPA 2010). Annualized Capital Costs at 7% discount rate over 10 years.
[8] (Alhambra Water 2010).

Adminsitrative Record
Page 35566

Table 41. High cost ranges for a basin-wide cost analysis.

HIGH COST RANGES	**Very Small**	**Small**	**Medium**	**Large**	**[Source]**
System Population Range:	**(25 - 500 people)**	**(501 - 3,300 people)**	**(3,301 - 10,000 people)**	**(10,001 - 100,000 people)**	[1]
Assumed Design Rate (mgd):	**0.01 - 0.17**	**0.17 - 1.09**	**1.09 - 3.21**	**3.21 - 30.45**	[2]
Annualized Total Cost ($/kgal):					
Groundwater Treatment	$4.60	$1.30	$2.00	$1.80	[3]
Surface Water Treatment	-	-	-	$1.20 + pipeline	[4]
New Well	$3.10 - $11.20	$1.40 - $3.10	$0.90 - $1.40	$0.30 - $0.90	[5]
Pipeline (2 miles)	$0.80 - $15.70	$0.10 - $0.80	$0.04 - $0.10	$0.01 - $0.06	[6]
Annualized Total Cost for POTABLE USES only ($/kgal):					
POU System for Potable Uses	$2.67 - $3.52	$2.51 - $2.67	$2.51	$2.51	[7]
Bottled Water for Potable Uses	$1,630	$1,630	$1,630	$1,630	[8]

[1] U.S. EPA system size classification.
[2] (U.S. EPA 2005).
[3] (Jensen et al. 2012).
[4] (Jensen et al. 2012). (8.25-10 mgd systems).
[5] (U.S. EPA 2007).
[6] (Monterey County Environmental Health 2010b). Does not include connection fee.
[7] (U.S. EPA 2010). Annualized Capital Costs at 7% discount rate over 10 years.
[8] (Alhambra Water 2010).

Adminsitrative Record
Page 35567

Table 42. Complete table of basin-wide alternative water supply options (minimum value of low range and maximum value of high range).

	Very Small	Small	Medium	Large	[Source]
System Population Range:	**(25 - 500 people)**	**(501 - 3,300 people)**	**(3,301 - 10,000 people)**	**(10,001 - 100,000 people)**	[1]
Assumed Design Rate (mgd):	**1 - 0.17**	**0.17 - 1.09**	**1.09 - 3.21**	**3.21 - 30.45**	[2]
Annualized Total Cost ($/kgal):					
Groundwater Treatment	$0.60 - $4.60	$0.30 - $1.30	$0.40 - $2.00	$0.20 - $1.80	[3]
Surface Water Treatment	-	-	-	$0.70 - $1.20 + pipeline	[4]
New Well	$0.44 - $10.20	$0.20 - $2.80	$0.10 - $1.30	$0.05 - $0.80	[5]
Pipeline (2 miles)	$0.80 - $15.70	$0.10 - $0.80	$0.04 - $0.10	$0.01 - $0.06	[6]
New Well + 2 Miles of Pipeline	$1.24 - $25.90	$0.30 - $3.70	$0.14 - $1.40	$0.06 - $0.80	[5,6]
Annualized Total Cost for POTABLE USES only ($/kgal):					
POU System for Potable Uses	$2.67 - $3.52	$2.51 - $2.67	$2.51	$2.51	[7]
Bottled Water for Potable Uses	$1,630	$1,630	$1,630	$1,630	[8]

[1] U.S. EPA system size classification.
[2] (U.S. EPA 2005).
[3] (Jensen et al. 2012).
[4] (Jensen et al. 2012). (8.25-10 mgd systems).
[5] (U.S. EPA 2007).
[6] (Monterey County Environmental Health 2010b). Does not include connection fee.
[7] (U.S. EPA 2010). Annualized Capital Costs at 7% discount rate over 10 years.
[8] (Alhambra Water 2010).

Adminsitrative Record
Page 35568

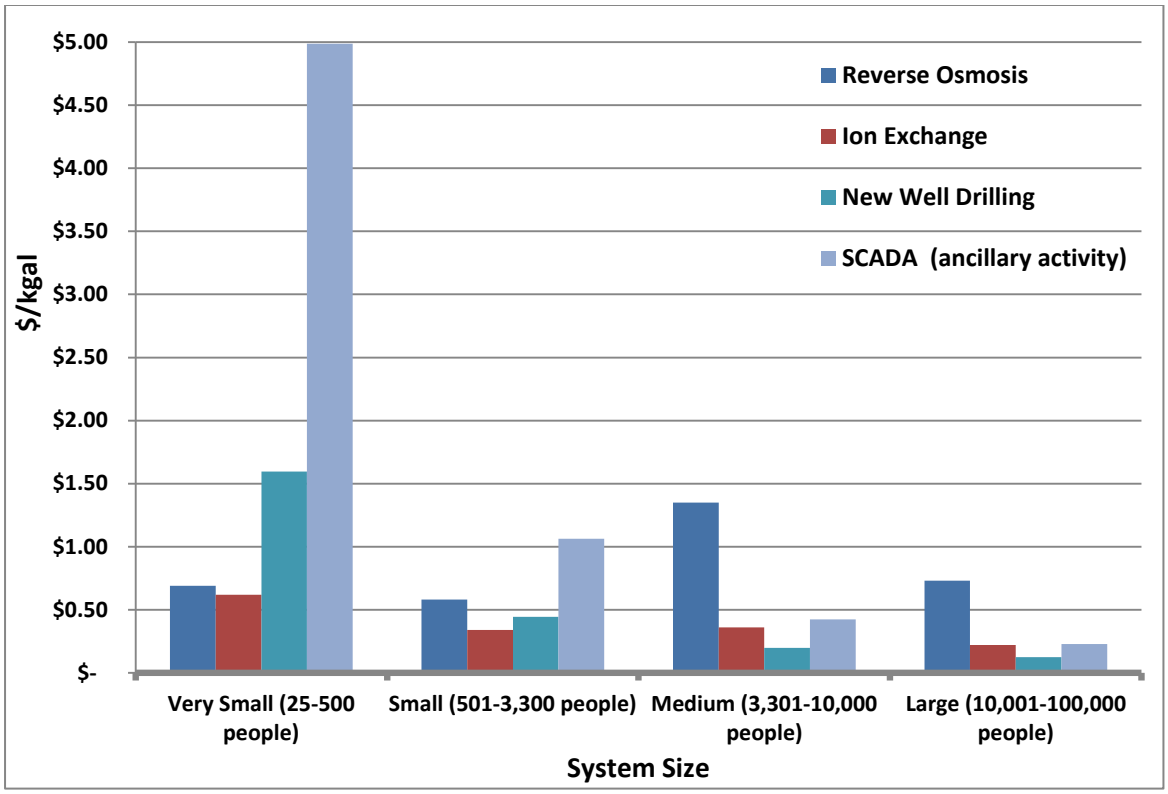

Figure 26. Annualized cost ($/kgal) for community public water system alternatives (excluding pipeline costs).

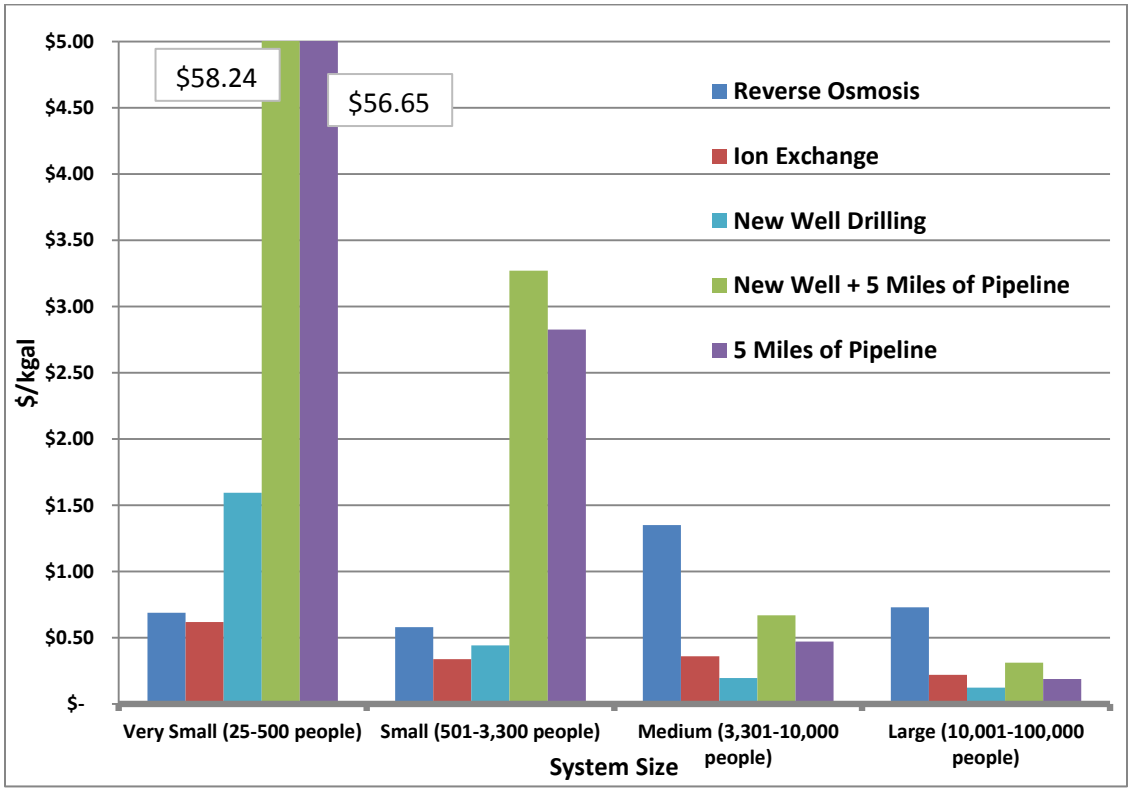

Figure 27. Community public water system annualized cost comparison ($/kgal) for alternative water supply option (including pipeline for connecting to a nearby system and excluding SCADA costs).

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Adminsitrative Record
Page 35569

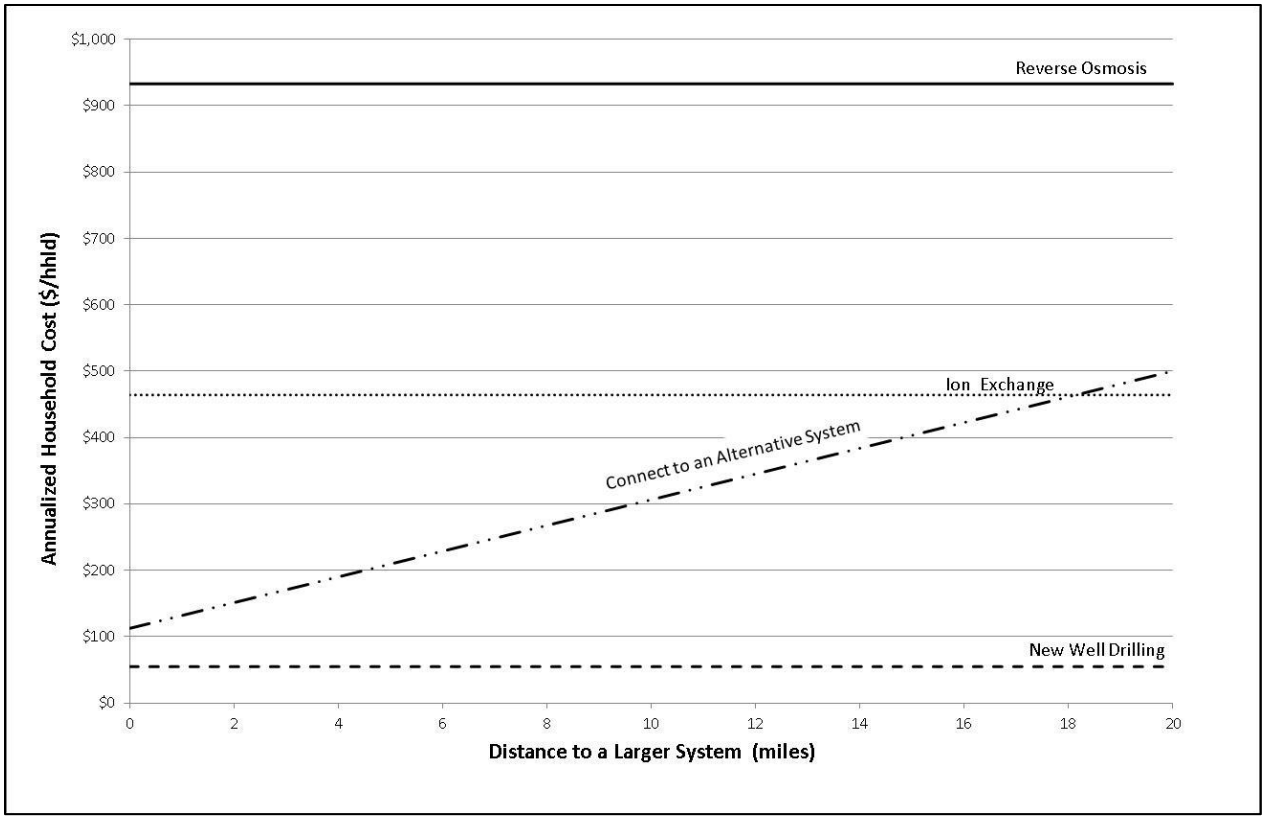

Figure 28. Annualized cost comparison ($/hhld) of alternative water supply options for a 2,000 connection community public water system.

These cost estimates (Table 39, Table 42, Figure 26, Figure 27, and Figure 28) are used to discuss the basin-wide cost estimates for providing alternative solutions to the population susceptible to nitrate contamination in the Salinas Valley and Tulare Lake Basin (discussed in Section 6 Basin-wide Costs of Nitrate Contamination).

5.3 Public Health and Other Considerations

The most recent twelve year survey of waterborne disease in the United States (1991-2002) documented 183 drinking water-related outbreaks, with 76% from groundwater sources (Reynolds et al. 2008). From 2001 to 2002, 92% of outbreaks related to drinking water were from groundwater, and 39% of the groundwater systems were household self-supplied systems not regulated by the U.S. EPA (Blackburn et al. 2004). Drinking water systems have an enormous effect on public health. Ford (1993, 1996, 1998) suggests that "there is reason to be concerned for the future microbiological safety of drinking water because a) source water continue to receive agricultural, industrial, and municipal waters; b) water treatment and distribution systems age and deteriorate; c) water supplies are overwhelmed by excessive demand; and d) there appears to be an increase in diseases, or at least an increased recognition of disease, caused by pathogens with varying degrees of resistance to treatment and disinfection" (Ford 1999).

Adminsitrative Record
Page 35570

A water system struggling with regulatory compliance must manage and plan for the future and prepare for potential tightening of regulations. Reverse osmosis (RO) and ion exchange (IX) are the most cost-effective treatment options to reduce nitrate and to provide the highest quality and reliability of safe water for all households connected to water systems. Furthermore, since nitrate is not the only contaminant of concern within the study area (see Technical Report 6, Jensen et al. 2012) RO treatment may be deemed the most promising option for certain systems because it is effective against many co-occurring contaminants. Until treatment can be afforded or completed, interim solutions include delivery of bottled or trucked water or distribution of POU treatment devices. Bottled water is regulated by the Food and Drug Administration (FDA) and is not required to follow as rigorous regulations as U.S. EPA-regulated tap water; however, bottled water will be better than the current supply of nitrate-contaminated groundwater, albeit at greater expense (particularly since most brands are bottled from U.S. EPA regulated municipal drinking water systems). Water delivered by a truck from a low-nitrate source should be of good quality if proper truck cleaning and transfer procedures are followed; however, CDPH does not allow water systems to serve trucked water to their community water supply customers (CDPH 2008; CDPH 2011c; CDPH 2011d). If POU treatment devices (usually RO) are distributed to households, they must be CDPH approved devices and require households to be properly educated on their use. RO devices require filter replacement. Often, a plumber must be employed to install the device and annual maintenance is advisable.

If a self-supplied household has tested their well for nitrate and found a problem, they can employ a POU or POE device and request assistance on installation and maintenance from a nearby vendor or water system. A POE system allows household members to have the convenience of using any sink in the house instead of only the sink with a POU treatment unit. A properly implemented and maintained POE device supplies a household with the highest quality and reliability of water, compared to a POU device. Drilling a deeper well is promising for households able to access an aquifer with low nitrate water. It would be best to drill to deeper, lower nitrate water, while also testing for arsenic levels.

Adminsitrative Record
Page 35571

6 Basin-wide Costs of Nitrate Contamination

Rough basin-wide costs for nitrate contamination in the pilot study area were estimated for communities, self-supplied households, and local small water systems, using a range of cost estimates from literature, surveys, and researching existing proposal estimates and final project costs as described in detail in Section 5 Evaluation of Options. The highly susceptible population (shown in Table 13) is estimated to be 254,000 people. Of this total, 220,000 people are connected to 85 community public or state small water systems and approximately 34,000 people are served by 10,000 self-supplied households or local small water systems. Overall, the estimated economic least cost for providing nitrate-compliant water to the total highly susceptible population in the study area is roughly $20 million per year ($80/susceptible person/year) for the short-term (excluding one very large system), and is roughly $36 million per year ($142/susceptible person/year) for the long-term (excluding one very large system). Short-term solutions are those that can be used in the interim or as temporary solutions, while a more sustainable, longer-term solution is being planned and implemented. Ideally, there would be a perfect long-term solution for each system; however, in some circumstances a short-term solution is the preferred long-term solution. Smaller water systems may not have the technical, managerial, or financial capacity to incorporate a centralized groundwater treatment system or they may be too far from a larger system to realistically consider a connection. In this case it may be better for small water systems to deliver and maintain POU devices to their customers or to drill a new well that is properly constructed and can avoid future contamination.

Again, the costs of alternative water supply and treatment options vary with numerous factors and the numbers presented here are estimates based on averages of widely variable costs. Costs will be unique to each individual system. For proper cost estimation, a feasibility analysis is necessary to assess the potential solution for each unique system.

6.1 Costs for Community Public and State Small Water System Alternative Solutions

A rough basin-wide lowest cost for solving nitrate contamination of drinking water in the study area was estimated for community public and state small water systems. Only multiple source systems having a recorded level of delivering water above the nitrate MCL, single source systems that have a raw source water level exceeding the nitrate MCL, and systems lacking water quality data within WQM were included in this analysis. The low and high ranges of the cost estimate for each alternative as a function of size of community public water system appear in Table 40 and Table 41, with the compiled cost ranges by size and option in Table 42. The maps in Figure 29 and Figure 30 show the estimated least-cost short-term option for each affected community system based solely on system size. The displayed options are not recommended solutions for specific systems since they only consider system size; individual solutions should be engineered and designed for the local situation of each system. This map merely shows which promising least cost solutions are found in the approximate short-term. The most promising least cost solutions for the approximate long-term are shown in Table 44. By short-term we

Adminsitrative Record
Page 35572

mean temporary or interim solutions that do not address the legacy of worsening nitrate groundwater pollution. By long-term we mean sustainable solutions capable of addressing increasing nitrate levels in groundwater. The short-term least cost for providing nitrate-compliant water to those connected to susceptible community public and state small water systems is roughly $13 – 17 million per year (Table 43). The long-term least cost for providing nitrate-compliant water to those connected to susceptible community public and state small water systems is roughly $34 million per year (Table 44). The estimated annual O&M cost (already included in the aforementioned least cost scenarios) for short-term basin-wide solutions is approximately $6.4 – 8.3 million per year, and about $11.5 million per year for long-term solutions (Table 45).

Short-term Solutions

The alternatives considered for community public and state small water systems (discussed in Table 40 and Table 41) include drilling a new well, installing a pipeline to a nearby system (within 14 miles) that serves more than 10,000 people, delivering and maintaining a POU RO device for potable uses, installing groundwater treatment, or connecting to a nearby new surface water treatment facility. The final short-term cost estimates are based on selecting the high estimate of the least cost alternative (Table 40 and Table 41) as a function solely of system size and proximity to a larger system, and excluding the follow options: allowing systems to provide bottled water to their consumers as a means of compliance, allowing medium and large systems to deliver POU RO devices for compliance, and allowing large systems to install a pipeline to a larger system (only systems with less than 10,000 people are connecting to systems with greater than 10,000 people). The final short-term cost estimates are shown in Table 43, with the first column representing solutions including POU systems and the second column excluding POU systems. Note that the cost to connect to an alternative system does not include the estimated connection fee. Since the costs throughout this report are from literature, surveys and U.S. EPA cost models there is some uncertainty in the accuracy of values to the true system size and capability. To estimate any uncertainty in these costs, all high estimate values (Table 40 and Table 41) are used to estimate the overall basin-wide least cost solutions and a contingency factor is applied based on system size. This contingency factor is meant to account for external factors that may not be included in researched cost estimates.

The options chosen for the final least cost estimate for providing community public and state small water systems with short-term solutions are:

1. drilling a new well;
2. delivering and maintaining POU RO devices for potable uses (only for systems serving up to 200 connections over a three year regulatory time constraint);
3. installing a pipeline to a nearby system (10,000+ system);
4. building a groundwater treatment facility; and
5. connecting to a nearby surface water treatment facility.

Drilling a new well and installing a POU device are the only options considered short-term solutions; however, for cost comparison installing a pipeline, building a groundwater treatment facility and connecting a nearby surface water treatment facility are also included in the short-term cost estimates.

Adminsitrative Record
Page 35573

The susceptible systems included in the community public and state small water system least cost analysis cost model are:

1. all CPWS and SSWSs with multiple sources, delivering water that exceeded the nitrate MCL at least once from 2006 to 2010;
2. all single source CPWS and SSWSs with raw source water that exceeded the MCL for nitrate at least once from 2006 to 2010; and
3. all CPWS and SSWSs with no nitrate water quality data.

A total of 85 susceptible community public and state small water systems serve 220,000 people, shown in Table 43 and Table 44.

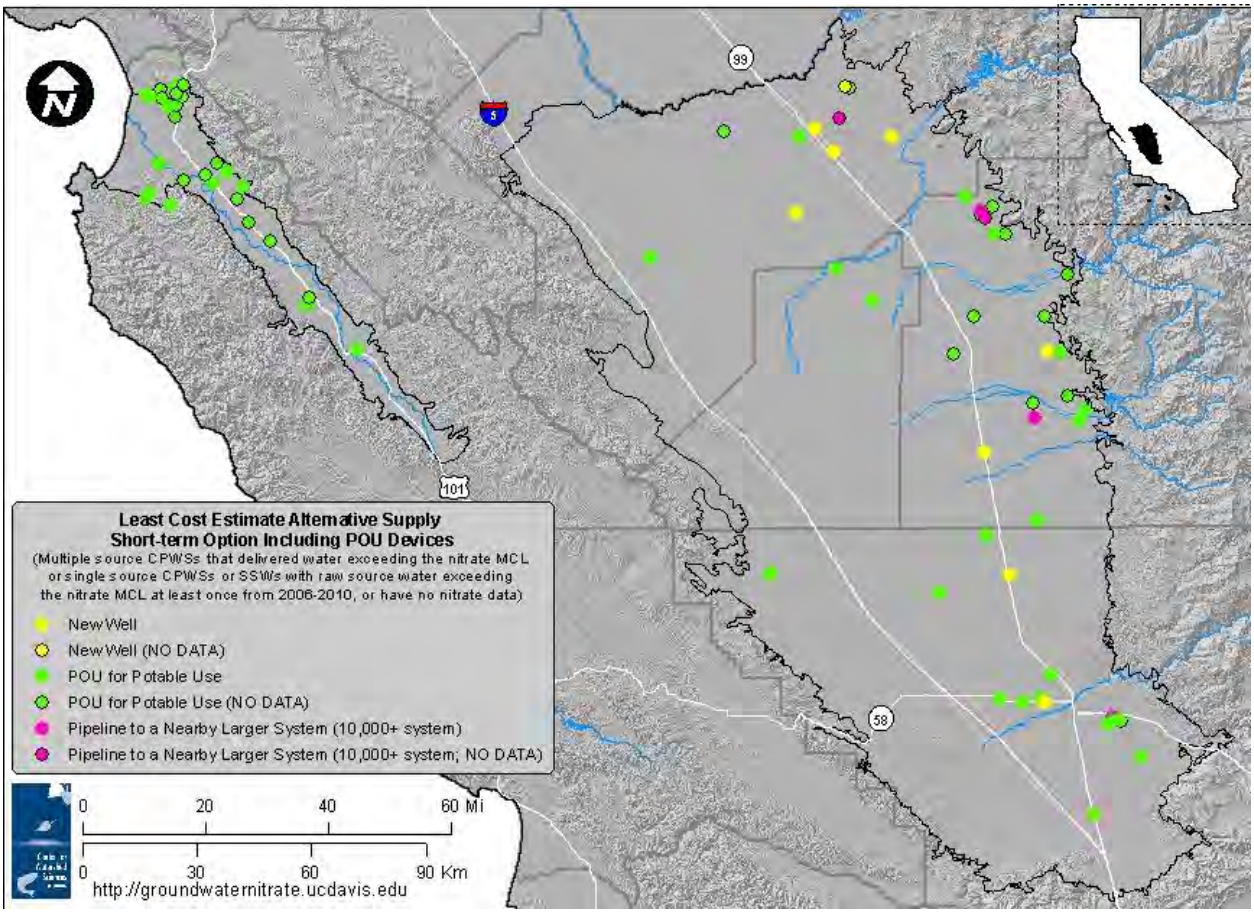

Figure 29. SHORT-TERM: Least cost alternative supply option (INCLUDING POU) based on high estimate of option costs for susceptible community public and state small water systems. (Source: CDPH PICME WQM 2006-2010 and cost estimate sources.)

Adminsitrative Record
Page 35574

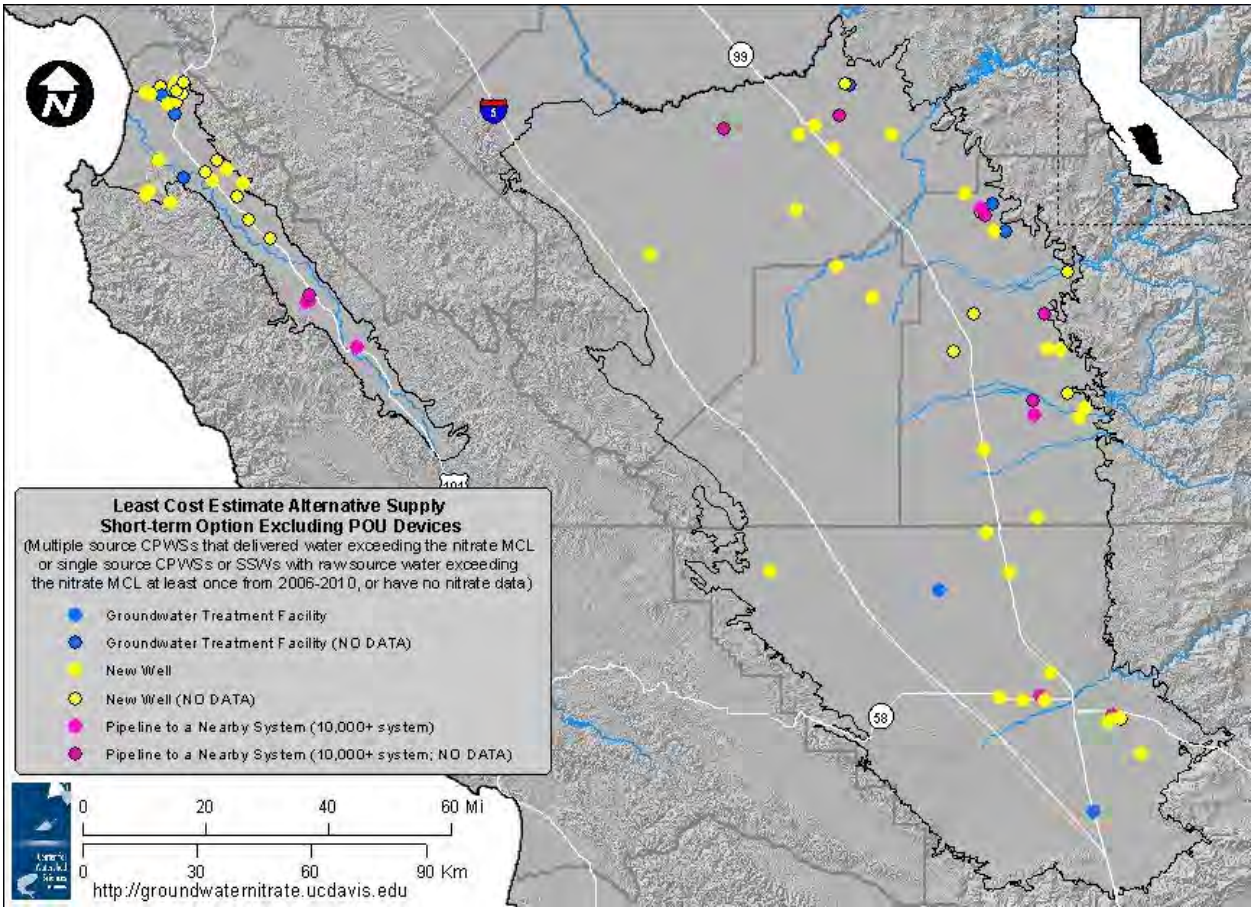

Figure 30. SHORT-TERM: Least cost alternative supply option (EXCLUDING POU) based on high estimate of option costs for susceptible community public and state small water systems. (Source: CDPH PICME WQM 2006-2010 and cost estimate sources.)

Table 43. Estimated cost range of the least cost short-term alternative water supply options for susceptible community public and state small water systems based on system size and proximity to a larger system.

	Number of Susceptible CPWSs/SSWSs		Population		Total Cost ($/year)	
Option	**Including POU**	**Excluding POU**	**Including POU**	**Excluding POU**	**Including POU**	**Excluding POU**
Drill New Well	10	63	184,123	191,715	$10,144,000	$14,500,000
POU Device for Potable Use	70	---	10,515	---	$1,320,000	---
Pipeline to a Nearby System (10,000+ system)	5	13	25,323	27,349	$865,000	$1,463,000
Groundwater Treatment Facility	0	9	0	897	$0	$450,000
Surface Water Treatment Facility	0	0	0	0	$0	$0
TOTAL	85	85	**219,961**	**219,961**	**$13 million**	**$17 million**

The least cost overall alternative water supply cost including POU devices is approximately $13 million per year (annualized), with ten systems drilling a new well, 70 systems delivering and maintaining a POU

Adminsitrative Record
Page 35575

device for potable use, and five systems installing a pipeline to a nearby system. This is graphically displayed in Figure 29 and numerically shown in Table 43. Drilling a new well for ten systems provides a solution for 84% of the susceptible population at a cost of $10.1 million per year. Installing and maintaining a POU device for 70 systems (all serving up to 200 connections) provides a solution for about 5% of the susceptible population at a cost of $1.3 million per year. Installing a pipeline to a nearby larger system for five systems provides a solution for 12% of the susceptible population at a cost of $865,000 per year. Building a groundwater treatment facility was not a least cost option when POU devices were available.

The least cost overall alternative water supply cost excluding POU devices is approximately $17 million per year (annualized), with 63 systems drilling a new well, 13 systems installing a pipeline to a nearby system, and nine systems building a groundwater treatment facility. This is graphically displayed in Figure 30 and numerically shown in Table 43. Drilling a new well for 63 systems provides a solution for 87% of the susceptible population at a cost of $14.5 million per year. Building a groundwater treatment facility for nine systems provides a solution for less than 1% of the susceptible population at a cost of $450,000 per year. Installing a pipeline to a nearby larger system for 13 systems provides a solution for 12% of the susceptible population at a cost of $1.5 million per year. Connecting to a surface water treatment facility was not a least cost option under the short-term least cost analysis solution, for either estimated cost options for the currently susceptible system sizes and locations.

The total estimated range of costs for the least cost alternative option for community public and state small water system short-term solutions is $13 – 17 million per year. However, because there is uncertainty in the amount of time a well can produce nitrate-safe drinking water before it may become compromised by legacy and on-going nitrate leaching, an alternative “long-term” cost scenario is analyzed below and shown in Table 44, in which the option of drilling a new well and installing POU RO devices are excluded as a long-term option.

Long-term Solutions

As groundwater degradation continues over time, nitrate and other contaminants will be of concern (e.g., arsenic, chromium, DBCP, and salts). Drilling a new well does not reliably provide a long-term safe drinking water solution throughout the study area since it may take only a few years for a nitrate plume to contaminate the new well or for other groundwater contaminants to appear in the location chosen for drilling a new well. For this reason, a long-term analysis of alternative water supply costs was undertaken in which more vulnerable options were excluded as potentially unreliable solutions for the longer-term. Excluded solutions included: use of POU RO devices for compliance and drilling a new well. As with the short-term solution cost analysis, the cost of the option to connect to a larger system does not include the estimated connection fee. To estimate any uncertainty in these costs, all high estimate values (Table 40 and Table 41) are used to estimate the overall basin-wide least cost solutions and a contingency factor is applied based on system size. This contingency factor is meant to account for external factors that may not be included in researched cost estimates.

Adminsitrative Record
Page 35576

The options chosen for the low and high option cost estimates for the long-term least cost analysis of alternative water supplies for community public and state small water system solutions are shown in Table 44, and include:

1. installing a pipeline to a nearby system (10,000+ system);
2. building a groundwater treatment facility (RO or IX[25]); and
3. installing a pipeline and connecting to a nearby surface water treatment facility.

Table 44 shows the least cost long-term solutions for the 85 susceptible community public and state small water systems, based on excluding the above indicated options as having only short- to medium-term viability.

Table 44. Estimated cost range of the least cost long-term alternative water supply options for susceptible community public and state small water systems based on system size and proximity to a larger system.

Option	Number of Susceptible CPWSs/SSWSs	Population	Total Cost ($/year)
Pipeline to a Nearby System (10,000+ system)	29	36,577	$5,592,000
Groundwater Treatment Facility	51	8,057	$6,344,000
Surface Water Treatment Facility	5	175,327	$21,532,000
TOTAL	85	**219,961**	**$34 million**

Assuming the viable long-term options, the estimated overall least cost long-term alternative water supply cost is approximately $34 million per year (annualized), with 29 systems installing a pipeline to a nearby larger system, 51 systems building groundwater treatment facilities, and five systems connecting to a surface water treatment facility. Installing a pipeline to a nearby larger system for 29 systems provides a solution for 17% of the susceptible population at a cost of $5.6 million per year. Building a groundwater treatment facility for 51 systems provides a solution for about 4% of the susceptible population at a cost of $6.3 million per year. Connecting to a surface water treatment facility for five systems provides a solution for about 80% of the susceptible population at a cost of $21.5 million per year.

The total estimated least cost long-term alternative water supply solutions for the 85 affected community public and state small water systems is $34 million per year.

For a long-term scenario it should also be mentioned that the groundwater treatment options available each have limitations and environmental considerations. An ion exchange groundwater treatment facility may not be the best long-term solution because of the environmental sustainability and management concerns associated with the disposal of concentrated brine. Additionally, ion exchange can address several co-contaminants, but not all potential contaminants. Reverse osmosis groundwater treatment has similar environmental sustainability and waste brine management concerns, as well as problems with wasting water, but is more capable of treating multiple co-contaminants (see Technical

[25] Reverse osmosis and ion exchange are the most commonly employed nitrate treatment options, but are not the only options for treatment. Other options may become more prevalent in the future (see Jensen et al. 2012).

Adminsitrative Record
Page 35577

Report 6, Jensen et al. 2012, for additional treatment information). The best long-term groundwater treatment will depend on the local characteristics of each system and the technology available at the time of interest.

Estimated O&M Costs for Basin Wide Alternative Water Supply Options

Using the average O&M percentage shown in Table 38, the annual estimated O&M cost associated with each least cost short- and long-term alternative is evaluated (Table 45). The estimated O&M cost related to short-term alternative solutions is approximately $6.4 – 8.3 million per year (including POU and excluding POU). The estimated annual O&M cost related to long-term alternative solutions is approximately $11.5 million. These costs are based on averages of widely variable costs and the true O&M cost for a unique system will require a proper cost estimate performed by a certified professional engineer or a professional cost estimator.

Table 45. Estimated O&M cost of the least cost short- and long-term alternative water supply options for susceptible community public and state small water systems.

	Short-term System Solutions[1]		Long-term System Solutions[1]	Estimated Annual O&M Cost ($/year)[2]		
Option	**Including POU**	**Excluding POU**		**Short-term with POU**	**Short-term without POU**	**Long-term**
Drill New Well	10	63	---	$5,580,000	$8,000,000	---
POU Device for Potable Use	70	---	---	$792,000	---	---
Pipeline to a Nearby System (10,000+ system)	5	13	29	$0	$0	$0
Groundwater Treatment Facility	0	9	51	$0	$295,000	$4,441,000
Surface Water Treatment Facility	0	0	5	$0	$0	$7,106,000
TOTAL	85	85	85	**$6.4 million**	**$8.3 million**	**$11.5 million**

[1] The number of CPWS and SSWSs that were shown in Tables 43 and 44.

[2] The estimated annual O&M cost uses the average O&M percentage from Table 38.

6.2 Costs for Household Self-Supplied and Local Small Water System Alternative Solutions

A rough basin-wide estimate of the least cost water supply solutions for the self-supplied households and local small water systems was developed using researched cost estimates for applicable options and is shown in Table 39. Susceptible self-supplied households and local small water systems are estimated within each Thiessen polygon that exceeds the MCL for nitrate (as discussed in Section 3.5.1.1 Household Self-Supplied or Local Small Water Systems with a High Likelihood of Nitrate Groundwater Contamination) and are estimated to amount to approximately 34,000 people in the study area. The least cost alternative for individual households is to install and operate a reverse osmosis POU device, estimated to cost $250 per year (not including any public outreach or education). The second least expensive alternative for households is to install a POE device; however, currently no POE devices are

Adminsitrative Record
Page 35578

NSF/ANSI certified for removing nitrate from drinking water. Figure 25 (in Section 5 Evaluation of Options) showed the annualized cost curves of each considered option, highlighting the least cost option based on expected water use or consumption per household. If a household only desires to have low-nitrate *potable* water, a POU device is the least expensive solution; however, treating more than 18 gallons of water per day with a POU device is not cost-effective and installing a POE device would be preferred. If installation of a POU or POE device is not preferred by the household user, drilling a new well is the next available option. However, without concurrent reduction in source loading, new wells will risk nitrate contamination over time as pumping and nitrate migration continue.

The expected total annualized costs for 10,000 households (34,000 people) to install a POU RO device is $2.5 million or $4 million for a POE IX device; however, installing a POE device provides the whole house with low-nitrate water and accidental consumption of nitrate-laden water is not of concern. The lifetime of the alternatives are different as well; a POU RO device or POE IX device have a ten year expected lifetime, while drilling a new well has a fifty year expected lifetime. However, if degradation and contamination of the study area continue, the lifetime of a well may be shortened if a nitrate plume spreads and eventually reaches the new well. The POU RO device is annualized over ten years; the cost for drilling a new well is annualized over 20 years.

The range in estimated nitrate costs for self-supplied households is $2.5 to $4 million per year. Using the average O&M percentage shown in Table 38, the estimated annual O&M cost associated with the POU and POE solutions is approximately $1.5 to $2.4 million.

6.3 Interim Solutions

The interim solutions discussed here are specific to domestic well users and small water systems serving less than 200 connections. Small water systems (serving less than 200 connections) may use an interim solution for compliance, such as a POU device, for up to three years under the emergency regulations established by CDPH (further discussed in Section 7 Regulatory and Implementation Implications). Providing POU devices to water system customers is only meant for systems that are in the process of creating or implementing a long-term solution. A small water system may not provide bottled water to consumers as an interim compliance option. Domestic well users may use alternative supplies, such as a POU or POE device or bottled water, at their own discretion and for the length of time they choose, without regulation. However, bottled water for an individual household's potable water needs is more costly over the long-term than installing and operating a POU device. Table 46 shows the estimated costs for interim water supplies for domestic wells and small water systems. The POU costs include capital, O&M, public education, and indirect costs. Cost estimates shown in parentheses exclude the costs of public outreach. For small public water systems serving 15 to 199 households, it is important to provide public education to users to increase proper use and handling. The economies of scale for the POU interim solution occur only in the added costs of public education; for each additional person the per capita price for education decreases. For domestic well users, the cost for U.S. EPA-suggested public education is not cost-effective, and self-education (by researching the POU device chosen) or a low budget, county prescribed public education program is suggested.

Adminsitrative Record
Page 35579

Table 46. Estimated cost ranges per household served for interim POU water supplies for households and small water systems serving fewer than 200 connections.

Options	Bottled Water[1]	POU 15 households[2]	POU 199 households[3]	POU 1 household[4]
Initial Capital Cost ($/hhld)	0	$443 ($406)	$411 ($406)	$1,981 ($406)
Annual O&M Cost ($/hhld)	0	$272 ($166)	$173 ($165)	$1,781 ($197)
Total Annualized Cost ($/kgal)	$1,630	$5.24 ($3.80)	$3.90 ($3.78)	$24.85 ($3.01)
Total Annualized Cost ($/hhld)	**$1,260**	**$435 ($315)**	**$324 ($314)**	**$2,038 ($250)**

[1] Quote from Alhambra, Visalia for drinking water delivered in 5 gallon bottles (Alhambra Water 2010). Assumed units of water consumption from NAS Hydration Study.

[2] Uses the 2010 U.S. EPA Cost Estimate Tool for an NSF/ANSI Certified RO Unit for treating nitrate. Includes unit purchase, installation, scheduling time, public education (technical and clerical labor and printed material for all public outreach/education efforts), indirect costs (permitting, pilot testing, legal, engineering, contingency) and all associated operations and maintenance costs. The tool allows for economies of scale representation assuming 15 households will be receiving units and management will be centralized. Costs in parentheses do not include public education.

[3] Uses the 2010 U.S. EPA Cost Estimate Tool for an NSF/ANSI Certified RO Unit for treating nitrate. Includes unit purchase, installation, scheduling time, public education (technical and clerical labor and printed material for all public outreach/education efforts), indirect costs (permitting, pilot testing, legal, engineering, contingency) and all associated operations and maintenance costs. The tool allows for economies of scale representation assuming 199 households will be receiving units and management will be centralized. Costs in parentheses do not include public education.

[4] Uses the 2010 U.S. EPA Cost Estimate Tool for 1 NSF/ANSI Certified RO Unit for treating nitrate. Includes unit purchase, installation, scheduling time, public education (technical and clerical labor and printed material for all public outreach/education efforts), indirect costs (permitting, pilot testing, legal, engineering, contingency) and all associated operations and maintenance costs. Costs in parentheses do not include public education.

Adminsitrative Record
Page 35580

7 Regulatory and Implementation Implications

Each alternative water supply option has its own regulatory and implementation implications. Since domestic well users are excluded from the statewide drinking water protections, and have no regulatory standards for testing their wells (except for a few local county ordinances), the implications of their water supply options will differ slightly from those for community public water systems.

7.1 Implications of Household Self-Supplied System Alternatives

The large numbers of self-supplied households pose several regulatory challenges. Their small size and unique and varying circumstances result in these small systems bearing significant financial costs and public health risks.

Drilling a New or Deeper Well: Some counties have existing regulations for drilling new domestic wells. In areas with nitrate contamination potential, counties and the state should consider additional monitoring and well construction regulations.

Household Treatment: Since there is little regulatory oversight for domestic well users, there are no specific requirements for POU or POE devices, but purchasing a certified unit is recommended. Households with self-supplied wells that choose to install a POU RO or POE IX device should consider hiring a certified POU/POE distributor to manage and operate the device or should seek training in proper maintenance. Given the often lesser and declining expense of POU/POE systems, the state should continue to examine its regulations to take advantage of these improving technologies.

Connect to Alternative System: If a household wishes to connect to a nearby safe community public water system they must contact the water system or county official and determine if they are within the water system boundary. If a household is outside of water system boundaries they will need to speak with local officials about annexation, an extension of the urban growth boundary, or a separate contractual relationship. Households outside of water system boundaries may find it more difficult to be incorporated into an existing water system.

Bottled Water: There are no regulatory implications for households to start purchasing bottled or vended water; however, the quality of bottled water is an under-regulated industrial activity, and can sometimes be of lower quality than other sources.

7.2 Implications of Community Public Water System Alternatives

Since CPWSs are larger in size and capacity and fewer in number they have many more options, but higher individual consequences. Community public water systems are always in direct contact with the regulators to determine the most effective solution for complying with drinking water standards.

Blending: CDPH has already established a Blending Program; however, the regulatory implications will differ slightly depending on whether the blending source is groundwater or surface water. A

Adminsitrative Record
Page 35581

groundwater source used for blending will not be suitable for use if the groundwater quality eventually exceeds the MCL for nitrate. Once blending is implemented the water system must continually monitor the low-nitrate source to ensure the blending ratio is achieved.

Drilling a New or Deeper Well: California already has a Well Standards Ordinance established for community public water systems for drilling a new or deeper well. Local county ordinances also establish rules for each community water system. Also, CPDH regulates well construction and evaluates the location, source water quality and quantity for drilling a new or deeper well in a community public water system supply.

Community Treatment: CDPH has a Division of Drinking Water and Environmental Management (DDWEM) that provides permitting information for community public water systems and has established a Drinking Water Treatment and California Operation Certification Program that provides the minimum qualifications for a potable water treatment system operator. Furthermore, the respective regulatory agency for each community public water system must verify that the drinking water treatment device employed is consistent with Title 22 California Code of Regulations (CDPH 2008; CDPH 2011a).

Household Treatment: California regulations currently allow small public water systems to provide POU devices to customers as a means of complying with the nitrate standards under the following restrictions (CDPH 2008):
"…a public water system may be permitted to use point-of-use treatment devices (POUs) in lieu of centralized treatment for compliance with one or more maximum contaminant levels… if;

(1) the water system serves fewer than 200 service connections,

(2) the water system meets the requirements of this Article (Title 22 Division 4 Chapter 15 Article 2.5),

(3) the water system has demonstrated to the CDPH that centralized treatment, for the contaminants of concern, is not economically feasible within three years of the water system's submittal of its application for a permit amendment to use POUs,

… no longer than three years or until funding for the total cost of constructing a project for centralized treatment or access to an alternative source of water is available, whichever occurs first…."

According to the emergency legislation (Health and Safety Code 11680(a)(1)) for temporary compliance a POE device may also be employed in lieu of centralized treatment. As of September 22, 2011 a POE system must be considered as a system solution before a POU device. If the POE device is found to be "not economically feasible or not as protective of public health as POUs" then a POU device may be used (CDPH 2011e). . The most significant costs of a POU RO device are in the management and monitoring of the unit. Since the law states that POU units must be centrally-managed by the public water system or by a company hired by the public water system, a fair regulatory policy should be developed. For example, a public water system could work with a private company to create a reasonable contract that allows the company to manage, maintain and monitor all devices within a specific public water system service area. To use a POU device for complying with the SDWA amendments there must be 100% participation within a public water system. If any of the connections deny access to their house it automatically prohibits POU as an alternative for compliance. This is a

Adminsitrative Record
Page 35582

substantial impediment to POU treatment. Other communities have addressed this by passing a local ordinance requiring installation, and employing the authority to disconnect the water supply if installation is refused (U.S. EPA 2006a). A local ordinance was passed in San Ysidro, New Mexico, making water use contingent on POU installation (U.S. EPA 2006a). It is important to provide public education to customers before, during and after implementation of a POU device to ensure success.

Connect to Alternative Supplies: All public water systems must submit an amended permit application to the local CDPH drinking water field office prior to changing their source or method(s) of treatment (CDPH 2011c; CDPH 2011d). If a water system switches from groundwater to surface water, the water treatment requirements change as specified in the State and Federal Surface Water Treatment Rule. Also, surface water treatment is almost always more expensive than groundwater treatment. However, all water served to the public for drinking water purposes is subject to the same nitrate drinking water standard. A switch to surface water could require purchasing existing surface water rights, contracting with another entity, petitioning for new surface water rights, or getting an existing surface water right owner, such as an irrigation district, to amend their use permit and expand activities to provide treated drinking water services.

Regionalization and Consolidation: Regionalization and consolidation allow systems to increase the levels of service by taking advantage of economy of scale benefits and complying with stringent regulations. Service duplications across management and operational functions can be eliminated, while achieving regulatory compliance and improving financial accountability. Rourke and Smith (1997) estimated that approximately 40 – 45% of community public water systems will experience financial instability from rising operational and future regulatory compliance costs; the larger population base found in regionalization can support increases in operational costs and future regulations. Legislation or CDPH should consider providing larger systems with financial and ratemaking incentives to encourage the acquisition of smaller systems. Regulatory incentives for regionalization have been considered in many policy areas, and "some states have enacted legislation authorizing the use of mandatory 'takeovers'…but, many water utilities would prefer positive incentives to mandatory takeovers" (Beecher 1996). Furthermore, if there was an incentive for larger systems to acquire smaller systems there would be fewer systems for regulating agencies to monitor, reducing administration costs.

Several studies have examined the advantages and disadvantages of regionalization, but the physical implementation depends on the unique needs and barriers in each region. Successful implementation of regionalization and consolidation requires planning in a regional context, along with strong public and political participation. Comprehensive planning that establishes public policy and resource planning on a regional scale will help meet the objective goals involved in consolidation (Beecher et al. 1996). Some implementation issues must be considered before consolidating systems, including: "(1) system income and expenses, (2) level of contributions in aid of construction, (3) rate base, (4) condition of facilities, (5) reasonableness of price and terms, (6) impact on customers, (7) required additional investments, (8) alternatives to sale and impacts of no sale, (9) ability to operate facilities, and (10) public interest assessment (Cloud 1994)" (Beecher 1996). Public participation is essential for regionalization, to properly educate the public and assure their involvement in the project, and for the future fate of the system. CDPH supports consolidation efforts through funding programs and low interest loans to

Adminsitrative Record
Page 35583

construct facilities for the physical consolidation of water systems, but does not have an explicit program to support the planning needed for moving towards regionalization.

To be considered for Safe Drinking Water State Revolving Funds, a water system must prove its technical, managerial and financial (TMF) capability.[26] A water system must also submit an assessment that identifies all public water systems located within a five mile radius and determines the feasibility of consolidation (Newkirk & Darby 2010).

Trucked Water: Hauled or trucked potable water is often allowed by CDPH and used for public water systems in emergency situations when a water source is interrupted for an extended period of time and if there are no other alternatives. A water system that needs to provide trucked potable water to customers must contact CDPH and ensure that a CDPH Food and Drug Branch licensed water hauler is hired for service delivery. The water must be obtained from another regulated public water system and a "boil water advisory" may be given to each consumer as a precautionary measure to account for possible contamination during the delivery.[27]

Dual Water Distribution System: The permitting agency for dual water distribution systems or recycled water systems is the local Regional Water Board (Central Valley and Central Coast). A water system that wants to install two distribution systems, one for potable and one for non-potable or recycled water, must file a report with the appropriate Regional Water Board (CWC section 13522.5). CDPH reviews and evaluates recycled water proposals to verify the protection of public health and ensures that a system has the correct backflow protection established to prevent non-potable water from entering the drinking water system. CDPH encourages the use of recycled water in urban areas where recycled water or irrigation water is available (CDPH 2011d). To mitigate the concerns of inter-connection of the two sources or improper plumbing, water system personnel must be trained and a cross-connection control program must be implemented (CDPH 2011d). This program must also include annual testing of the backflow prevention devices and periodic shut-down tests (CDPH 2011d).

If a dual water distribution system is implemented on a community public water system level there must be full public acceptance among consumers. Along with customer acceptance and approval, there must be a consensus within the City or County about the quantity and quality of water expected for delivery, and communication with the state about existing regulations. If a system desires to install a dual water distribution system there must be consideration of the health effects, treatment, storage and distribution demands (American Water Works Association 1994).

[26] California Health and Safety Code Section 116540.
[27] Boiling water is not an option to address nitrate contaminated water.

Adminsitrative Record
Page 35584

8 Conclusions

8.1 Major Findings

1) **A total of 254,000 people in California's Tulare Lake Basin and Salinas Valley have drinking water supplies susceptible or potentially susceptible to nitrate groundwater contamination.**
   a) Highly susceptible water users fall into one of three identified situations. These comprise: i) being served by a community public water system with multiple sources and at least one reported nitrate record in excess of 45 mg/L since 2006, ii) being a single source community public water system with at least one reported raw source water nitrate record in excess of 45 mg/L since 2006, or iii) being a self-supplied household or served by a local small water system located in the vicinity a shallow nitrate groundwater concentration in excess of 45 mg/L. Approximately 213,000 people in 38 multiple-source systems have at least one reported delivered nitrate record in excess of 45 mg/L. Approximately 3,400 people in 34 single-source systems have at least one, reported raw water nitrate record in excess of 45 mg/L or no nitrate water quality data in WQM. In addition, approximately 10,000 rural households (serving approximately 34,000 people) using domestic wells or on local small water systems of four or less connections are in the vicinity of shallow groundwater where the nitrate concentration exceeds 45 mg/L.
   b) Other water users considered highly susceptible include approximately 3,900 people in 13 multiple source water systems in the study area that have no recorded nitrate concentration data in the statewide water quality database (WQM).
2) **Nitrate contamination problems will grow.** According to recorded *raw* groundwater data in WQM, **57% of the current population of these regions uses a community public water system with recorded raw nitrate levels above 45 mg/L**. Assuming unchanging continued basin-wide trends observed since 1970 in nitrate groundwater levels, this number is expected to increase to almost 80% by 2050. Nitrate groundwater contamination problems will increase, treatment costs will rise, and there is growing potential for public health impacts.
3) **Each community public and state small water system with high susceptibility (50 and 35 systems in Tulare Lake Basin and Salinas Valley, respectively) will need individual engineering and financial analyses. No single solution will fit every community affected by nitrate in groundwater.**
4) **From a physical and engineering cost perspective, there is significant potential to consolidate systems.** The potential for consolidation is based on system size and the distance from a smaller system to a larger system. About 81% and 89% of the Tulare Lake Basin and Salinas Valley water systems are very small or small (defined as serving < 3,300 people) and serve 89,125 and 23,215 people (4% and 6% of the Tulare Lake Basin and Salinas Valley population), respectively. In the Tulare Lake Basin and Salinas Valley, respectively, about 50% and 15% of smaller systems (<10,000 people) are within five miles of a larger system (>10,000 people), and 88% and 97% of smaller systems are within 12.5 miles of a larger system. Consolidation permanently addresses nitrate problems, as well as many other small system safe drinking water problems typically encountered by these systems now and in the future.

Adminsitrative Record
Page 35585

5) **Options for communities connected to highly susceptible systems** are:
   a) consolidation with a larger system that can provide safe drinking water to more customers;
   b) consolidation of nearby small systems into a single larger system, with a larger rate payer base and economies of scale;
   c) ion exchange community water treatment;
   d) interim point-of-use treatment systems until a more long-term and sustainable solution can be implemented;
   e) drilling a new well, at least in the short-to-medium term;
   f) blending of contaminated wells, at least temporarily; and
   g) switching to surface water.

   Some of these options are less viable solutions for the long-term because they may fail to sustainably deliver nitrate-safe water. Those considered less viable in the long-term include POU, drilling a new well, and blending.
6) **Solutions for self-supplied household or local small water systems within a highly susceptible area** are bottled water (as an interim solution), reverse osmosis point-of-use treatment systems, point-of-entry ion exchange systems, and drilling a new or deeper well.
7) **Better data collection and management will improve knowledge of the extent of nitrate contamination.** Improvements in state and local management of data regarding state small wells, water system boundaries, domestic wells, and systems with unknown risk can facilitate understanding and more effective management of nitrate contamination.
8) **The overall estimated least cost, including capital and operating costs, for providing short-term nitrate-compliant drinking water to the Tulare Lake Basin and Salinas Valley is $15 – 20 million per year for the currently susceptible population. The overall estimated least cost, including capital and operating costs, for providing long-term nitrate-compliant drinking water to the Tulare Lake Basin and Salinas Valley is $36 million per year for the currently susceptible population.**

Roughly $13 – 17 million per year will be needed to provide safe drinking water for short-term solutions for multiple source community public water systems exceeding the nitrate MCL, single source community public water systems (and the small fraction of state small systems included in CDPH's database) with nitrate records in excess of 45 mg/L, and community public water systems (and the state small water systems included in CDPH's PICME database) lacking nitrate records in WQM, that together currently serve an estimated 220,000 people (85 systems). Of this cost, approximately $6.4 to 8.3 million is estimated as annual O&M costs. The provision of safe drinking water with long-term solutions for these 85 systems will cost roughly $34 million per year. Of this cost, approximately $11.5 million is estimated as annual O&M costs.

The annualized cost of providing nitrate-compliant drinking water to the estimated 34,000 people (10,000 rural households) using domestic wells or local small water systems (< 5 connections) highly susceptible to current or future nitrate contamination ranges from $2.5 million per year for point-of-use (POU) treatment for drinking purposes only, to $4 million per year for point-of entry (POE) treatment for each household for all water needs. Costs for both could be lower if a manufacturing discount for bulk purchase of POU/POE systems were available. Approximately $1.5 to $2.4 million is estimated as annual

Adminsitrative Record
Page 35586

O&M costs associated with providing POU/POE systems to the susceptible population served by domestic well or local small water systems. The least cost POU option is included in the total project study area estimated least cost of alternative water supply solutions in both the short- and long-term.

The cost to fund alternative water supplies for highly susceptible, nitrate-affected water systems amounts to $60 – 80 per susceptible person per year, $4 – 5 per irrigated acre per year for the four million acres of agriculture in these regions, or $75 – 100 per ton of fertilizer nitrogen (assuming about 200,000 tons of fertilizer is applied in these regions) in the short-term. Allowing for only long-term more viable and sustainable alternative drinking water solutions for affected communities, the total cost would amount to $142 per susceptible person per year, $9 per irrigated acre per year for the four million acres of agriculture in these regions, or $180 per ton of fertilizer nitrogen (assuming about 200,000 tons of fertilizer is applied in these regions) in the long-term.

8.2 Promising Actions

1) **A feasibility analysis should occur for individual systems to determine the most suitable alternative water supply option, in which consolidation and regionalization are actively explored and investigated as the best longer-term options for small systems.**
2) **For any solution, there are long-term considerations, such as:**
   a) **The need to remove additional contaminants.** Alternative water supplies or treatment options selected should be capable of addressing multiple contaminants if these currently exist or there is good evidence that additional contaminants are likely to emerge in the near-to-medium term.
   b) **The drawdown involved in pumping a new or deeper well.** Over time, as pumping continues, a new well could draw nitrate into the well.
   c) **Brine waste from treatment systems.** The low brine technologies in groundwater treatment offer a minimal waste approach, and future research and development of brine treatment alternatives seem promising for greatly reducing brine waste from treatment systems. Systems should consider the sustainable management and disposal of the brine in the selection of the groundwater treatment.
   d) **Environmental impacts of reliance on bottled water.** Manufacturing and transporting bottles uses a lot of energy and causes negative environmental impacts, and the disposal of these bottles stresses solid waste systems.
3) **Advance household treatment for community public water system compliance.**
   a) Create NSF certifiable POE devices for community public water systems to provide to customers.
   b) Allow community public water systems to provide POU devices to customers for more than three years under reasonable conditions.
4) **Regionalize and Consolidate.**
   a) **Fund non-structural regionalization/consolidation of drinking water systems.** Programs in California fund physical consolidation activities like the construction of new pipelines, the installation of water meters, or the expansion of treatment systems. Regionalization efforts

Adminsitrative Record
Page 35587

should be expanded to convene pilot projects that bring together communities of water systems to encourage informational, managerial, institutional, and future planning collaboration to lay the groundwork and foundations for future consolidation with respect to institutional, political, and shared needs. This could be done without necessarily having to start with historically, politically sensitive, and difficult consolidation, since appropriate regionalization projects can help communities begin to collaborate on information and other types of exchanges to address shared nitrate problems, as well as many of the other shared problems of small water systems.

b) **Provide incentives to large safe drinking water systems to consolidate with or initiate regionalization projects with surrounding smaller systems.** To encourage larger systems to take on the risks of a smaller system, incentives should be offered. For example, it may be beneficial to increase the points given to large systems on State Revolving Funds' Project Priority Lists who help bring small systems up to the same technical standards as larger systems, often a prerequisite for consolidation.

5) **Construct, populate, and maintain a statewide publicly accessible comprehensive water quality database for groundwater and public water supply systems.** To facilitate accessibility of groundwater quality data throughout the state, one agency should manage a comprehensive database and create a simple graphical user interface for easy extraction of groundwater quality data.

6) **Create a Small System Water and Wastewater Task Force for integrating water and wastewater treatment projects and efforts.** Small water and wastewater systems have many chronic and common problems, and tend to have more frequent contamination problems and higher costs than larger systems. A Task Force should examine the state's effectiveness of overseeing and aiding such systems, with particular attention to possibilities for better integrating water and wastewater utility functions. This would require an inventory of existing and future system concerns with a special focus on very small and small systems' current infrastructure conditions, problems, and future needs.

7) **Require domestic wells water quality monitoring.** Collection of shallow domestic well water quality data is a promising management practice for identifying and protecting groundwater quality especially for domestic self-supplied households' drinking water needs in areas identified as being at risk for nitrate contamination.

Adminsitrative Record
Page 35588

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Adminsitrative Record
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Adminsitrative Record
Page 35602

10 Appendix

10.1 Methods

10.1.1 Susceptibility Charts

Explanations are given below for the population/connections estimates given in the susceptibility breakdown charts (Figure 17, Figure 18, and Figure 19). The number below corresponds to the labeled boxes in the susceptibility breakdown charts.

[1] Total Population: California Department of Finance (CDF) estimates for city-wide populations were combined with US Census population estimates for Census Designated Places (CDP). Average annual county growth rates from the CDF 2007 report were applied to project the CDP population to 2010.

[2] Household Self-Supplied or Local Small Water Systems: All parcels with a parcel use code designated as residential and having one to four residential dwelling units outside of city and water system boundaries and 3.3 people were assumed to be inhabiting each dwelling unit.

[3] Community Public Water System with Only One Well: Public water system information is from the California Department of Public Health's PICME water system database. Data were pulled for all active community and state small systems in the study area with only one well.

[4] Community Public Water System with More than One Well: Public water system information is from the PICME database. Data were pulled for all active community and state small systems in the study area with more than one well delivering water directly to individuals. A system's source was assumed to be a "delivering" source following the method described below. Systems with one well and a treatment plant were also included.

[5] Only Surface Water Sources: Public water system information is from the PICME database. Data were pulled for all active community and state small systems in the study area where all "delivering" sources were surface water. A system's source was assumed to be a "delivering" source following the method described below.

[6] Treating or Blending for Nitrate: A list of systems that treat or blend specifically for nitrate was compiled through personal communication with County Environmental Health departments, individual water systems, and the California Department of Public Health.

[7] Not Treating or Blending for Nitrate: Population and connections from Box #6 was subtracted from Box #4.

[8] High Likelihood of Nitrate in Groundwater: A population range is presented based on the water quality analysis for populations listed in Box #2 and #3. The population estimated to reside in areas where there was exceedance of the nitrate threshold chosen (22.5 mg/L as NO_3^-). The UC Davis Wells Database (CASTING) was used to examine raw nitrate groundwater levels from 1989 to 2010 in all self-supplied

Adminsitrative Record
Page 35603

household and local small water systems. The CDPH Water Quality Management (WQM) database was used to examine all raw nitrate groundwater levels from 2006 to 2010 in all community public and state small water systems.

[9] Low Likelihood of Nitrate in Groundwater: A population range is presented based on the water quality analysis for populations listed in Box #2 and #3. The population estimated to reside in areas where there was not exceedance of the nitrate threshold chosen. The UC Davis Wells Database (CASTING) was used to examine raw nitrate groundwater levels from 1989 to 2010 in all self-supplied household and local small water systems. The CDPH WQM database was used to examine all raw nitrate groundwater levels from 2006 to 2010 in all community public and state small water systems.

[10] Nitrate MCL Exceedances: Public water system water quality information is from the CDPH WQM database. A system was assumed to exceed the nitrate MCL (i.e., deliver water to customers that exceeded the nitrate MCL) if the maximum recorded nitrate level from 2006-2010 for any "delivering" source in a system was greater than 45 mg/L as NO_3^-. A system's source was assumed to be a "delivering" source following the method described below.

[11]No Nitrate MCL Exceedances: Public water system water quality information is from the CDPH WQM database. A system was assumed to not exceed the nitrate MCL (i.e., no deliveries of water to customers that exceeded the nitrate MCL) if the maximum recorded nitrate level from 2006-2010 for all "delivering" sources in a system were less than or equal 45 mg/L as NO_3^-. A system's source was assumed to be a "delivering" source following the method described below.

[12] No Nitrate Data: This box contains the community water systems with more than one well (Box #4) that did not contain any water quality data on nitrate levels in PICME's CDPH database from 2006-2010.

10.1.2 Estimating "Delivering" Sources of a System

Often, the "sources" field listed in CDPH's PICME database simply refers to a water quality sample point along the treatment/distribution line, and not necessarily a well. Samples can be taken at the beginning, end or middle of the distribution line. A source can even refer to a treatment plant. There are various methods to determine which sources are actually delivering the recorded water quality to customers and which are merely intermediary points along the treatment/distribution line. The method used in this report is by no means infallible, but it uses a column in PICME that is present and consistent for most systems, and can therefore be used as a rough way to understand the bigger picture. CDPH's PICME database contains a column labeled "ENTITY_INFO" that describes the source. All sources (with the exception of inactive sources) that are labeled as "Treated" were considered to be delivering sources because this designation refers to a point along the distribution system after treatment has occurred. Similarly, all sources (with the exception of inactive sources) that are labeled as "Untreated" were also considered to be delivering because these sources refer to points along the distribution system where treatment has not occurred, but will NOT occur in the future. The sources labeled as "Raw" were not included because these sources will be treated in the future, and are therefore not the final entry point into the distribution line before the water reaches customers. Sources with the

Adminsitrative Record
Page 35604

following specific codes in the "ENTITY_INFO" column were considered to be delivering sources and their water quality data was assumed to reach customers as listed in PICME:

- AT = Active Treated. Active source after treatment.
- AU = Active Untreated. Active Source that is not treated and will not be treated.
- CM = Combination/Blend Mixed. Blended sources included in this station are both treated and raw or untreated.
- CT = Combination/Blend Treated. Blended sources all treated prior to sample point.
- CU = Combination/Blend Untreated. Blended sources are all untreated and will not be treated using any method prior to delivery.
- DT = Distribution Treated. Sample point within the distribution system, after treatment.
- PT = Purchased Treated. Purchased source water that was treated by the seller.
- PU = Purchased Untreated. Purchased source water that has not and will not be treated.
- ST = Standby Treated. Emergency source that is used less than 15 calendar days per year, with periods not to exceed five consecutive days, and that receives treatment when in use.
- SU = Standby Untreated. Emergency source that is used less than 15 calendar days per year, with periods not to exceed five consecutive days, untreated (CDPH's PICME Documentation).

A few active community or state small water systems in the study area did not have any sources labeled with the above designations. In these cases, all sources were maintained for the system and were considered to be delivering sources, even if they were labeled as "Raw".

Adminsitrative Record
Page 35605

10.2 Rainwater Cisterns

A rainwater cistern is an underground basin or an above ground barrel or tank that collects and stores rainwater from rooftops or other catchments. Rainwater harvesting has been used for centuries to supply water for household, landscape, and agricultural uses and is currently being applied in locations including Hawaii, Africa, Asia and Australia. Rainwater harvesting relies on dependable rainfall and runoff and is suitable for locations where the average rainfall exceeds 400 mm/year (Lye 2002). A rainwater harvesting system has the following six components: a catchment area or roof, gutters and downspouts, leaf screens and roofwashers, cisterns or storage tanks, conveyance, and water treatment. Within the study area the rainwater will be applied to potable uses that will require proper filtration and disinfection prior to distribution. A rainwater cistern used for potable uses should have durable, watertight exterior and a clean, smooth interior sealed with a non-toxic joint sealant with all materials labeled as FDA-approved. Cisterns may be constructed of plastic, metal, concrete and masonry, or wood. Cistern design depends on the rainfall within the region, the catchment area, and the household's daily water use. The cistern needs to be properly located to avoid sunlight penetration, maintain a minimum distance of 50 feet from septic fields, and have the proper foundation and support.

The average construction cost is estimated to be $1.48 per gallon of collection capacity; a potable water case study of a 5,000 gallon above ground fiberglass cistern with a 5 micron sediment filter, a carbon cartridge filter and UV light costs about $6,200 (Texas Water Development Board 2005).

Air pollution due to crop dusting and agricultural practices would create water quality problems, as the chemicals and debris left on rooftops would wash off into the cistern with the first rainfall. Another concern would be in the reliability, timing, and volume of the rainfall. As previously mentioned, the Salinas Valley annually receives about 20 inches of rain, and the Tulare Lake Basin annually receives between 7 and 13 inches of rain. The Texas Rainwater Harvesting Manual estimates a production of 600 gallons of water for every inch of rain over a 1,000 square foot catchment area, that would yield an annual amount of 12,000 gallons of water per household (32 gallons per day) within the Salinas Valley and an annual amount between 4,200 and 7,800 gallons of water per household (11 to 21 gallons per day) within the Tulare Lake Basin, assuming the average catchment area of 1,000 square feet. The inconsistency and unreliability of the distribution pattern of this source would not be a sufficient supply for a household to depend on for potable water.

Overall, public health and water quality seem likely to be the greatest impediment to the use of cisterns as a replacement water supply. Costs can be high, even though water yields are likely to be adequate for drinking and cooking water.

Adminsitrative Record
Page 35606

10.3 Glossary

Census Block	The smallest geographic unit used by the US Census for tabulating data collected from all households within a region. They are formed by streets, roads, railroads, streams and other bodies of water, other visible physical and cultural features, and the legal boundaries shown on Census Bureau maps. (US Census)
Census Block Group	A cluster of census blocks and a subdivision of a census tract. Census block groups generally have between 600 and 3,000 people. On average there are 39 blocks in a block group. (US Census)
Census Tract	Small, relatively permanent statistical subdivisions of a county delineated for most metropolitan areas and other densely populated counties by local census statistical areas committees. Census tracts usually have between 2,500 and 8,000 persons and, when first delineated, are designed to be homogeneous with respect to population characteristics, economic status, and living conditions. (US Census)
Census Designated Place (CDP)	Areas delineated for each decennial census as the statistical counterparts of incorporated places. CDPs are created to provide data for settled concentrations of population that are identifiable by name but are not legally incorporated under the laws of the state they are located. (US Census)
Community Public Water System (CPWS)	A public water system that serves at least 15 service connections used by yearlong residents or that regularly serves at least 25 yearlong residents of the area. (CDPH)
Disadvantaged Community (DAC) – Block Group	A block group that has a Median Household Income (MHI) of less than 80% of the State of California's Median Household Income.
Household Self-Supplied Water System	A water system that is not connected to a public water system, is assumed to be 1 to 2 dwelling units (or connections), and is considered a domestic well.
Local Small Water System	A water system with 2 to 4 connections.
Local Primacy Agency (LPA)	County environmental health jurisdiction that has applied for and was granted regulatory authority over small community and non-community public water systems in their county. (CDPH)
Maximum Contaminant Level (MCL)	Enforceable drinking water regulations established to protect the public against consumption of drinking water contaminants that present a risk to human health. (U.S. EPA)
Median Household Income (MHI)	The amount that divides the income distribution into two equal groups, half having income above that amount, and half having income below that amount. It is the sum of money received in the calendar year by all household members 15 years of age or older,

Adminsitrative Record
Page 35607

	including unrelated household members. (US Census)
Non-Transient Non-Community public water system (NTNC)	A public water system that is not a community public water system and that regularly serves at least 25 of the same persons over 6 months per year. (CDPH)
Permits, Inspection, Compliance, Monitoring, and Enforcement (PICME)	The PICME database maintained by the Drinking Water Program of the California Department of Public Health and containing information related to the regulation of public drinking water systems subject to the federal and California Safe Drinking Water Acts. (CDPH)
Public Water System (PWS)	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. PWSs are regulated under the Safe Drinking Water Act. (CDPH)
State Small Water System (SSWS)	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. SSWSs are not regulated under the Safe Drinking Water Act.
Susceptibility	The potential for the residential population to consume drinking water above the nitrate MCL.
Transient Non-Community public water system (TNC)	A non-community public water system that does not regularly serve at least 25 of the same persons over 6 months per year. (CDPH)
Vulnerability	The potential for a system to deliver water with high nitrate levels if nitrate contamination occurs at a source. A function of the type of system. Classified as higher, lower, or no vulnerability depending on the source of water and the number of system water sources.
Water Quality Management (WQM)	The WQM database contains one record for each CPWS per quarter and year with average concentrations of nitrate as well as the frequency of sampling, the number of sampling stations, and the date of the last sample.

Adminsitrative Record
Page 35608

11 Case Studies

CITY OF LINDSAY

Phone Conversation with Public Works Director, Mike Camarena (5/19/11)

The City of Lindsay's main water supply today is treated surface water from the Friant Kern Canal. They have some groundwater wells, but they prefer to use the surface water due to groundwater quality issues (groundwater sources contain nitrate, but do not exceed the MCL). The City has a long-term contract with the United States Bureau of Reclamation (USBR) for a set amount of water to be delivered costing $225/af (the contract was signed in 2006). The City is currently helping Paige-Moore Tract by supplying them with water, but the current water treatment plant was built at a specific capacity and the City is starting to have issues with maxing out capacity. They have to chlorinate the raw water from the canal prior to filtration and this initial and final chlorination process is causing disinfection byproducts (DBPs) to contaminate the supply. The facility needs to be expanded to allow the water to sit for a long enough detention time and allow the chlorine to properly disinfect the water. The City of Lindsay is applying for SRF funding for either a new, bigger contact tank or an alternative disinfectant, the total cost is estimated at $300,000 to $400,000. The City will not be funding the distribution or pipeline costs for El Rancho or Tonyville, but they have to apply for SRF funding to connect to the system. The Tonyville application had just been rejected. It is hard for the City to incorporate neighboring communities, even though they want to help because they need to do what is best for their future growth and they must preserve their best interest. Currently within the City of Lindsay, it is policy to charge double the cost of water to anyone served outside of the city limits. This double charge allows for the City to help fund their system and prepare for future growth. **13.04.300 (City Code) – Service Outside City: All water services outside the city limits are subject to council approval, and shall pay twice the applicable monthly rates. (Ord. 329 § 5-5.1974)

Since some of these smaller communities outside of city limits cannot afford to pay double rates, they may have to obtain council approval to try to lower that value. The City is a metered water system and anyone who is connecting must be metered as well. They do not have block water rates. Also, for the City to include Tonyville and El Rancho they must alter their contract with USBR and increase the allotment of Friant Kern Canal water they receive.

Adminsitrative Record
Page 35609

LEMON COVE WATER COMPANY

Phone Conversation with Bill Pensar (5/23/11)

They received Prop 84 Safe Drinking Water funding back in 1991 for a new well at Mateas Point; they have been experiencing nitrate fluxes over the past 20 years. The nitrate concentrations have gone up to 100 mg/L as nitrate and then down to below the MCL and then back up to 100 mg/L. In late October of 2008, they applied for funding for a Feasibility Study to drill a new well. They were accepted for loans, but want grant money. Their application was just recently re-submitted. The feasibility study will cost about $200,000 and drilling a new well will cost about $100,000. They are hoping that the study will also cover expenses for a new tank that can be built up on a hill and can pump at night utilizing the cheap energy costs and gravity driven distribution. The new well will be in a location that is closer to them than the existing well so they will not have to install pipeline or increase the distribution mains. They believe drilling a new well is the best option, avoiding the need for brine disposal associated with treatment, purchasing or filing a new license, hiring an operator or participating in a lot of O&M activities. They are under the impression that RO is outrageously expensive and they are worried about disposing the brine back into the TLB. The cost of trucking the brine to a remote location is too expensive as well.

PLAINVIEW MUTUAL WATER COMPANY: Leaky Distribution Lines and Contaminated Back-Up Well

Plainview Mutual Water Company provides drinking water to around 800 people in the unincorporated area of Plainview, Tulare County (CWC 2011b). When one of their wells was shut down because nitrate levels started to exceed the MCL, Plainview was forced to rely on their only other well. This second well had recorded concentrations of DBCP (CWC 2011b). The distribution mains for this area were installed in 1941 and severe rusting and leakage issues caused bacterial contamination of the drinking water being supplied to the homes (Doan 1995). A flat rate of $25 per month was charged to these households (Doan 1995) whose median income in 1997 was only $12,000. Funds raised by the water company were not enough to adequately maintain the water system or to protect the water; many households were left to struggle to finance their own in home chlorine treatment for the bacteria (Doan 1995). Plainview Mutual Water Company was able to secure $2.3 million from federal and state sources to replace their distribution system and build a new well. This tremendous sum could have never been financed by such a small disadvantaged community.

Doan, Lynn (1995). “Towns Thirst For Safe Water”. Visalia Times-Delta <http://www.lynndoan.com/Towns_Thirst.html>

Community Water Center (CWC) (2011b). “Plainview”. <http://www.communitywatercenter.org/water-valley.php?content=Plainview>

Adminsitrative Record
Page 35610

CITY OF TUCSON: "Brown Water" Produced After Switching from Groundwater to Surface Water

With the enactment of Arizona's Groundwater Management Act in 1980, the City of Tucson was required to achieve a safe yield by appropriately balancing groundwater withdrawals with recharge (University of Arizona 1998). This mandate led to the city's adoption of a 110-year water plan that mapped out a strategy to serve 95% of the population with surface water from the Central Arizona Project (CAP) by 1995 (ADWR 1999). By 1992, the surface water treatment plant was constructed and the city began delivering treated CAP water to 60% of its municipal customers (ADWR 1999). The salts, which had built up inside of the pipes over the years that groundwater was distributed, were released due to the sudden change in pH (attributable to the surface water). Suddenly, households were receiving brown water. Municipal delivery of the surface water was halted in 1994 due to the poor water quality and concerned residents. Later, it was revealed that the utility was only replacing pipes at a rate of 0.5 percent per year (typical rates are 3 – 5%) (ADWR 1999).

Arizona Department of Water Resources (ADWR) (1999). "Third Management Plan for Tucson Active Management Area", Chapter 8: Augmentation and Recharge Program. <http://www.adwr.state.az.us/azdwr/WaterManagement/AMAs/documents/ch8-tuc.pdf>

University of Arizona (1998). "Safe Yield Goal Proving Elusive". Arizona Water Resource, Vol 7, No 1. <http://ag.arizona.edu/AZWATER/awr/sept98/feature1.html>

Adminsitrative Record
Page 35611

Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 8:

Regulatory and Funding Options for Nitrate Groundwater Contamination

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

Report for the State Water Resources Control Board Report to the Legislature

California Nitrate Project,
Implementation of Senate Bill X2 1

Center for Watershed Sciences
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Prepared for the California State Water Resources Control Board

Adminsitrative Record
Page 35612

Regulatory and Funding Options for Nitrate Groundwater Contamination

Technical Report 8

Addressing Nitrate in California's Drinking Water

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California Nitrate Project, Implementation of Senate Bill X2 1

Tulare Lake Basin and Salinas Valley Pilot Studies

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Adminsitrative Record
Page 35613

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Adminsitrative Record
Page 35615

Contents

Tables vii

Figures viii

Acronyms and Abbreviations ix

Unit Conversions xi

Summary 1

Major Findings 2

Regulatory Programs and Options 2

Funding Programs and Options 4

Promising Actions 5

Regulatory Programs and Options 5

Funding Programs and Options 6

1 Introduction 9

1.1 Study Area Description 9

1.2 Regulatory History of Nitrate in Drinking Water 10

2 Current Planning and Regulatory Programs That Address Nitrate in Groundwater 12

2.1 Overview of Current Planning and Regulatory Programs 12

Immediate safe drinking water actions 16

Longer-term source reduction and data management actions 16

2.2 Major Findings: Current Planning and Regulatory Programs 17

2.3 Promising Actions: Current Planning and Regulatory Programs 18

3 Future Regulatory Options to Manage Nitrate 19

3.1 Assumptions, Limitations, and Analytical Criteria 19

3.2 What Can Be Regulated? 21

3.3 Direct Regulatory Options 23

Adminsitrative Record
Page 35616

3.3.1 Technology Mandates 23
3.3.2 Performance Standards 23
3.4 Taxes and Fees 23
3.4.1 Nitrate Leachate Tax 24
3.4.2 Fertilizer Tax 24
3.5 Cap-and-Trade 24
3.5.1 Fertilizer Cap-and-Trade 24
3.5.2 Nitrate Leachate Cap-and-Trade 25
3.6 Hybrid Solutions 25
3.7 Information Disclosure 25
3.8 Liability Rules 26
3.9 Negotiation or Payment for Drinking Water Quality 26
3.10 De-designation of Beneficial Use 27
3.11 Comparing Regulatory Instruments 28
3.11.1 Comparing Direct Regulatory Instruments to Market-Based Approaches 28
3.11.2 Feasibility of Liability Rules and De-designation 29
3.11.3 Role of Information Disclosure 30
3.11.4 Choosing Among Market-Based Approaches – Fees v. Cap-and-Trade 31
3.12 Major Findings: Future Regulatory Options 31
3.13 Promising Actions: Future Regulatory Options 32
4 Current Funding Programs 34
4.1 Information on Current Sources of Funding 34
4.2 Summary of Current Funding Programs 35
4.3 Current Funding Example: Safe Drinking Water State Revolving Fund 39
4.4 Major Findings: Current Funding Programs 41

Adminsitrative Record
Page 35617

4.5 Promising Actions: Current Funding Programs 42
5 Future Funding Options 44
5.1 Overview and Summary Table 44
5.2 Assumptions and Limitations 46
5.3 Funding Sources 46
5.3.1 Water Fees 46
5.3.2 Nitrate Contamination-Related and Agricultural Fees 47
5.4 Comparing Funding Sources 48
5.4.1 Agricultural vs. Non-Agricultural Fees 48
5.4.2 Choosing Among Agricultural Fees 48
5.4.3 The Advantage of a Fertilizer Fee 49
5.5 Major Findings: Future Funding Options 49
5.6 Promising Actions: Future Funding Options 50
6 Summary of Promising Actions 51
6.1 Regulatory Programs and Options 51
6.2 Funding Programs and Options 52
7 Literature Cited 54
Appendix A: Regulatory Options in Practice 58
Watershed-Based Effluent Trading 58
Liability from Groundwater Contamination from Pesticide Use 60
Payment for Ecosystem Services in New York City 61
The Dutch Experience 62
Appendix B: Summary of Current Planning and Regulatory Programs that Address Nitrate in Groundwater 64
U.S. Environmental Protection Agency (U.S. EPA) 64
U.S. EPA: Enforcement Policy Response and Enforcement Targeting Tool (ERP-ETT) 65

Adminsitrative Record
Page 35618

U.S. EPA: Supplemental Environmental Programs 65

United States Department of Agriculture (USDA): National Drinking Water Clearinghouse, West Virginia University 66

California Department of Public Health (CDPH) 66

CDPH: Drinking Water Source Assessment and Protection (DWSAP) 66

CDPH: Expense Reimbursement Grant Program (EPG) 66

State Water Resources Control Board (State Water Board) 66

State Water Board: The Porter-Cologne Water Quality Control Act 67

State Water Board: Groundwater Ambient Monitoring and Assessment (GAMA) Program 68

State Water Board: Recycled Water Policy 69

Regional Water Quality Control Boards (Regional Water Boards) 69

Cleanup and Abatement Order (CAO) 69

Central Coast Regional Water Quality Control Board (Central Coast Regional Water Board) 70

Central Coast Regional Water Board: Irrigated Lands Regulatory Program (ILRP) 70

Central Valley Regional Water Quality Control Board (Central Valley Regional Water Board) 71

Central Valley Regional Water Board: Irrigated Lands Regulatory Program (ILRP) 72

Central Valley Regional Water Board: CV-SALTS' 73

Central Valley Regional Water Board: Dairy Program 73

California Department of Food and Agriculture (CDFA) 74

CDFA: Feed, Fertilizer and Livestock Drugs Regulatory Services (FFLDRS) 74

Assembly Bill 3030 (Groundwater Management Act) 75

Kern County Water Agency (KCWA) 76

Monterey County Water Resources Agency (MCWRA) 76

Monterey County Health Department 77

Southern San Joaquin Valley Water Quality Coalition 77

Tulare County Water Commission 77

Adminsitrative Record
Page 35619

Rural Community Assistance and Partnership (RCAP) 78
Monterey Coastkeeper 78
National Rural Water Association (NRWA) 78
California Rural Water Association (CRWA) 79
Self-Help Enterprises (SHE) 79
Community Water Center (CWC) 79
Appendix C: Description of Current Funding Programs for Safe Drinking Water 81
California Department of Public Health (CDPH) 81
CDPH: Safe Drinking Water State Revolving Fund (SDWSRF) 81
CDPH: Proposition 84 81
CDPH: Proposition 50 82
State Water Resources Control Board (State Water Board) 82
State Water Board: Clean Water State Revolving Fund (CWSRF) 82
State Water Board: Small Community Wastewater Grants 83
State Water Board: Proposition 50 83
State Water Board: Agricultural Drainage Loan Program 83
State Water Board: Dairy Water Quality Grant Program 84
State Water Board: Federal Clean Water Act Section 319: Nonpoint Source Implementation Program 84
State Water Board: Cleanup and Abatement Account 84
State Water Board: Integrated Regional Water Management (IRWM) (Proposition 50) 84
Department of Water Resources (DWR) 85
DWR: Integrated Regional Water Management (IRWM) Financial Assistance (Propositions 50, 84) 85
DWR: Local Groundwater Assistance (Prop 50 & Prop 84) 85
DWR: New Local Water Supply Feasibility Study and Construction Funds (Proposition 82) 85
DWR: Water Use Efficiency Grant Program (Prop 50) 86

Adminsitrative Record
Page 35620

DWR: Agricultural Water Conservation Program (Prop 13) 86
DWR: Infrastructure Rehabilitation Construction Grants (Proposition 13) 86
California Infrastructure and Economic Development Bank (I-Bank) 87
United States Department of Agriculture (USDA) Rural Utilities Service (RUS) 87
USDA-RUS: Water and Waste Disposal Direct Loans and Grants 87
USDA-RUS: Water and Waste Disposal Guaranteed Loans 88
USDA-RUS: Water and Waste Revolving Fund Grants 88
USDA-RUS: Individual Household Water Well Program 88
USDA-RUS: Predevelopment Planning Grants 88
USDA-RUS: Emergency Community Water Assistance Grants 88
USDA-RUS: Technical Assistance and Training Grants 89
US Department of Housing and Urban Development (HUD): Community Development Block Grant Program 89
U.S. Economic Development Administration (EDA) 89
Rural Community Assistance Corporation (RCAC) 90
The Housing Assistance Council (HAC) 90
Cooperative Bank (CoBank) 91

Adminsitrative Record
Page 35621

Tables

Table 1. Summary of directly applicable programs and statutes for reducing nitrate contamination in groundwater. 13

Table 2. Summary of programs and statutes related to groundwater nitrate and drinking water (data collection, information, education, or regulation of drinking water). 14

Table 3. Regulating fertilizer application versus nitrate leachate. 22

Table 4. Summary of regulatory instruments to manage nitrate contamination to groundwater. 29

Table 5. Summary of existing funding sources for safe drinking water. 36

Table 6. Unmet qualified project needs after allocation of 2010-2011 Safe Drinking Water State Revolving Fund. 41

Table 7. Summary of future State funding options. 45

Adminsitrative Record
Page 35622

Figures

Figure 1. Structure of the State Revolving Fund. (U.S. EPA CWSRF Branch)..... 35

Figure 2. Dollar requested per person on proposals made to the 2010-2011 Safe Drinking Water State Revolving Fund (Final Fundable Project Priority List, Oct. 2010)..... 40

Adminsitrative Record
Page 35623

Acronyms and Abbreviations

Abbreviation	Full Name
AB	Assembly Bill
AMBAG	Association of Monterey Bay Area Governments
AQUA	Association of People United for Water
ARRA	American Recovery and Reinvestment Act
AWP	Agricultural Waiver Program
BMP	Best Management Practices
CAA	Cleanup and Abatement Account
CalEPA	California Environmental Protection Agency
CalNRA	California Natural Resources Agency
CCR	California Code of Regulations
CCR	Consumer Confidence Report
CDBG	Community Development Block Grant
CDFA	California Department of Food and Agriculture
CDPH	California Department of Public Health
CoBank	Cooperative Bank
CRWA	California Rural Water Association
CV-SALTS	Central Valley Salinity Alternative for Long-Term Sustainability
CVSC	Central Valley Salinity Coalition
CWA	Clean Water Act
CWC	Community Water Center
CWSRF	Clean Water State Revolving Fund
DPEIR	Draft Program Environmental Impact Report (of the Central Valley ILRP)
DPR	California Department of Pesticide Regulation
DWR	California Department of Water Resources
DWSAP	Drinking Water Source Assessment and Protection
DWSRF	Drinking Water State Revolving Fund
EDA	U.S. Economic Development Administration
EPG	Expense Reimbursement Grant Program
ERP-ETT	Enforcement Policy Response and Enforcement Targeting Tool
FFLDRS	Feed, Fertilizer and Livestock Drugs Regulatory Services
FMIP	Fertilizing Materials Inspection Program
GAMA	Groundwater Ambient Monitoring and Assessment
HAC	Housing Assistance Council
HSNC	Historical Significant Non-Compliers
HUD	U.S. Department of Housing and Urban Development
I-Bank	California Infrastructure and Economic Development Bank
ILRP	Irrigated Lands Regulatory Program
IRWM	Integrated Regional Water Management

Adminsitrative Record
Page 35624

ISRF	Infrastructure State Revolving Fund
KCWA	Kern County Water Agency
LLNL	Lawrence Livermore National Lab
MCL	Maximum Contaminant Level
MCWA	Monterey County Water Resources Agency
MHI	Median Household Income
MUN	Municipal or domestic water supply (beneficial use)
NDWC	National Drinking Water Clearinghouse
NMP	Nutrient Management Plan
NPDES	National Pollutant Discharge Elimination System
NRWA	National Rural Water Association
NWG	Nitrate Working Group
OW	EPA's Office of Water
PES	Payment for Ecosystem Services
PHG	Public Health Goal
Porter-Cologne	Porter-Cologne Water Quality Control Act (California Water Code § 13000 et seq.)
PPL	Project Priority List
PWS	Public Water System
RCAC	Rural Community Assistance Corporation
RCAP	Rural Community Assistance and Partnership
RUS	Rural Utilities Service
SB	Senate Bill
SDWA	Safe Drinking Water Act
SDWSRF	Safe Drinking Water State Revolving Fund
SEP	Supplement Environmental Program
SHE	Self-Help Enterprises
SRF	State Revolving Fund
U.S. EPA	United States Environmental Protection Agency
U.S.C.	United States Code
USDA	United States Department of Agriculture
USGS	U.S. Geological Survey
WARMF	Watershed Analysis Risk Management Framework
WDR	Waste Discharge Requirements
WEP	Water Environmental Program
WMP	Waste Management Plan

Adminsitrative Record
Page 35625

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

*Unless otherwise noted, nitrate concentration is reported as milligrams/liter as nitrate (mg/L as NO_3^-).

To convert from:

Nitrate-N (NO_3-N)	→	Nitrate (NO_3^-)	multiply by 4.43
Nitrate (NO_3^-)	→	Nitrate-N (NO_3-N)	multiply by 0.226

Adminsitrative Record
Page 35626

Summary

In California Senate Bill SBX2 1 (SBX2 1), the Legislature required the State Water Resources Control Board (State Water Board) to prepare a Report to the Legislature to "improve understanding of the causes of [nitrate] groundwater contamination, identify potential remediation solutions and funding sources to recover costs expended by the State … to clean up or treat groundwater, and ensure the provision of safe drinking water to all communities." The State Water Board contracted with the University of California to address the Legislative mandate of SBX2 1. As part of that effort, this technical report reviews existing and potential future funding and regulatory programs for managing nitrate contamination in the groundwater and drinking water supplies in the Salinas Valley and Tulare Lake Basin. Major factual findings are made and several promising directions for regulatory and funding actions are proposed.

Nitrate contamination of groundwater is widespread, and many parts of the Salinas Valley and Tulare Lake Basin have high concentrations of nitrate in drinking water. Current regulatory programs have not effectively controlled groundwater nitrate contamination and water quality in these areas has largely worsened for decades, a trend which seems likely to continue.[2] Looking forward, promising options exist to manage nitrate contamination of groundwater, but it will take years to decades for source control programs introduced today to improve drinking water quality.[2] Existing nitrate contamination will continue to travel, increasing both the area and concentration of groundwater nitrate contamination in the study area. A range of safe drinking water actions (alternative water supplies or drinking water treatment),[3] groundwater remediation,[4] and source reduction actions[5] are needed to provide residents with safe drinking water supplies. The cost of providing safe drinking water to the most susceptible populations in the study area is estimated to be at least $20-$36 million per year.[6]

The slow response of groundwater quality to source reduction implies that the most immediate State efforts should emphasize safe drinking water actions. However, the costs to provide safe drinking water to affected communities in this region are high, due to the large number of groundwater-contaminating nitrate sources, the dispersed population, and the high incidence of elevated nitrate levels in drinking water. As a consequence, communities in the study area have requested more Safe Drinking Water Funds relative to other communities in the State (see Figure 2). Many of these community public water systems[7] are small water systems,[8] which often already face chronic financial problems. They have difficulty in applying for and meeting the eligibility requirements to receive existing State funds for drinking water because smaller systems lack economies of scale and often have less technical,

[2] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)
[3] See Technical Report 6: Drinking Water Treatment for Nitrate (Jensen et al. 2012); and Technical Report 7: Alternative Water Supply Options for Nitrate Contamination (Honeycutt et al. 2012)
[4] See Technical Report 5: Groundwater Remediation and Management for Nitrate (Viers et al. 2012)
[5] See Technical Report 3: Nitrogen Source Reduction to Protect Groundwater Quality (Dzurella et al. 2012)
[6] See Technical Report 7: Alternative Water Supply Options (254,000 highly susceptible people on community water systems or domestic wells) (Honeycutt et al. 2012)
[7] Small community public water system: a water system with 15 or more service connections and regulated by the State
[8] Small water system: a water system that has less than 200 connections (can be regulated or unregulated by the State)

Adminsitrative Record
Page 35627

managerial, and financial capacity than larger systems. State small[9] and local small water systems[10] may not have the resources to form a public entity,[11] so they are ineligible for many State funding sources (State Revolving Fund, State bonds, etc.). When funding is provided, these systems often lack the capacity to manage operation and maintenance costs or make loan repayments. Revising existing funding programs to encourage and guide good long-term local decisions; providing assistance to small systems in the application and management process; and combining successful funding programs may increase the funds available to small systems and increase the effectiveness of these funds in providing safe drinking water. Encouraging regional consolidation of small water systems can also be promising to address nitrate contamination as well as other safe drinking water challenges.

In the long-term, many regulatory options could reduce nitrate contamination from nonpoint sources, but some are preferable to others in terms of abatement costs; monitoring and enforcement costs; information requirements; and the ability to raise revenue. Overall, regulation of fertilizer is easier to design, administer, and enforce than regulation of diffuse nitrate leachate discharges to groundwater. Market-based regulatory approaches (of fertilizer application or of nitrate leachate) often can achieve a water quality target at a lower cost than prescriptive standards. However, no regulatory option is without its drawbacks: e.g., regulation of fertilizer may not achieve the drinking water quality objective at all drinking water systems.

Despite the existence of funding programs for safe drinking water, additional funding sources are needed. A fertilizer fee is a promising funding source that also creates incentives for dischargers to use fertilizer and water more efficiently. Farmers are currently exempt from a sales tax on fertilizer (if applied to land used for food or for feed for food animals) (CA State Board of Equalization 2004), so if this fee were set at a sales tax rate of 7.5% on the cost of nitrogen, it has the potential to raise an estimated $28 million annually, [12] nearly enough to provide alternative water supplies to the highly susceptible populations in the study area.

Major Findings

Regulatory Programs and Options

1. **To date, regulatory programs have been insufficient to control nitrate contamination of groundwater.** Discharges from agriculture (fertilizers and animal wastes) are the largest source of nitrate contamination of groundwater,[13] but there is no current or historic regulatory program that functionally holds these dischargers responsible (with the possible exception of the 2007 Central

[9] State small water system: a water system with 5-14 connections and unregulated by the State.
[10] Local small water system: a water system with less than five connections and unregulated by the State.
[11] A legally-approved public entity is a public water system or another legal entity that has authority to contract and incur debt on behalf of the community.
[12] Assume: 7.5% sales tax and a fertilizer retail price of $0.75 per pound (Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality, Dzurella et al. 2012)
[13] See Technical Report 2: Nitrogen Source and Loading to Groundwater (Viers et al. 2012)

Adminsitrative Record
Page 35628

Valley Dairy General Order, which will see stricter enforcement over the next few years). Overall, nitrate concentrations in groundwater have not decreased in the last three decades; in fact, concentrations have even increased in some areas.[14] Some regulatory programs have recently introduced mandatory monitoring programs, but monitoring alone will not improve water quality.[14]

2. **Many years are needed for regulatory actions to reduce nitrate in groundwater and improve drinking water quality.** The physical properties of nitrate in groundwater mean that regulatory actions on fertilizer or nitrate leachate today will not bring drinking water sources into water quality compliance for years to decades.[14]

3. **Many options exist to regulate nitrate in groundwater, but there is no ideal solution.** The costs of regulatory options vary greatly, and while no option is perfect, some seem preferable to others.

4. **Regulating fertilizer application has lower monitoring and enforcement costs and information requirements than does regulating nitrate leachate, but it may be less effective in achieving nitrate reduction targets.** While the regulation of fertilizer application is easier to implement and enforce than the regulation of nitrate leachate, it is less likely that water quality standards will be met. Due to non-uniform mixing, transport, and dispersion of nitrate in groundwater, it is difficult to quantify the impact of a unit of fertilizer on nitrate contamination of drinking water over time.

5. **Costs to farmers for reducing nitrate contamination can be lower with market-based regulatory actions (fertilizer fees or cap-and-trade programs) than with technology mandates or prescriptive standards because of the additional flexibility farmers have in complying with market-based regulations.** Market-based instruments also encourage the development and adoption of new technologies to reduce fertilizer use, but they may lead to the formation of contamination hot spots.

6. **Well-defined and enforceable regulatory requirements are needed for liability rules to work.** In California, all groundwater is considered to be suitable, or potentially suitable, for municipal or domestic water supply and should be so designated by the Regional Water Quality Control Boards (Regional Water Boards), with a few exceptions.[15] Under existing water code, groundwater is protected from degradation affecting its designated beneficial uses, but this protection is rarely enforced in the case of nonpoint source pollution. Section 13304 of the California Water Code (Porter-Cologne Act) gives the Regional Water Boards authority to force polluters to pay for alternative water supplies for affected users of public water systems and private wells. Legislation might be useful to solidify Regional Water Board authority to apply this provision broadly.

[14] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)
[15] Waters with very high TDS, for example. See Section 4.9 De-designation of Beneficial Use and also, State Water Board Resolution No. 88-63

Adminsitrative Record
Page 35629

Funding Programs and Options

1. **Current funding programs have not met systems' stated need to ensure safe drinking water in the Salinas Valley and Tulare Lake Basin.**[16] These areas face large costs to deal with high nitrate concentrations. Compared to other areas of California, groups in the study area have requested more Safe Drinking Water State Revolving Funds to address nitrate contamination ($29 per person compared to $5 per person statewide, see Figure 2. Providing safe drinking water or alternative water supplies to highly susceptible populations in the study area is estimated to cost at least $20-$36 million per year ($80-$142/year per susceptible person or $5-$9/year per acre of irrigated land).[16]

2. **Most current State funding for nitrate contamination of drinking water is temporary.** State funding for safe drinking water is currently dominated by general obligation bonds for loans through State propositions, Federal economic stimulus package grants, and State revolving fund loans. All are temporary sources except for the State revolving fund loans, which are self-financed long-term sources.

3. **Small water system costs are high**[17] **and these small systems already face chronic financial problems.** In small and rural communities, housing is often low density and spread across a relatively large area; the infrastructure needed to transport water from the source to a household in a rural area is more expensive than for urban areas. Small systems also often lack the economies of scale and the technical, managerial, and financial capacity to repay loans, complete funding applications, and pay both recurring and unexpected operation and maintenance costs. This implies that unit costs of safe drinking water and responding to nitrate contamination will be higher in rural and small communities.

4. **While regionalization**[18] **of drinking water systems is sometimes promising, little funding is provided for facilitating this solution**. California often provides funding to drinking water systems for the physical consolidation[19] of systems, but it provides no support or training for non-construction regionalization activities (e.g., a facilitator brings systems together to assess the feasibility of consolidation, or a technical expert performs asset mapping and financial planning for a group of water systems).

5. **Options exist to raise funds for safe drinking water, but all require that someone bear the cost and many are awkward or insufficient.** Water use fees, groundwater pumping fees, bottled water fees,

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[16] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al. 2012)
[17] EPA. 2011. National Characteristics of Drinking Water Systems Serving 10,000 or Fewer People (U.S. EPA 2011b)
[18] Regionalization: "a creation of an appropriate management or contractual administrative organization or a coordinated physical system plan of two or more community public water systems in a geographical area for the purpose of utilizing common resources and facilities to their optimum advantage" (Grigg 1989)
[19] Consolidation: "one community public water system being absorbed into, combined with, or served by other utilities to gain the resources they lack otherwise" (Raucher, Megan Harrod, & Marca Hagenstad 2004)

crop fees, and fertilizer fees are a few of the many potential sources for funding safe drinking water and source reduction actions.

6. **Some funding options give polluters a useful price signal.** Fertilizer (or nitrate leachate) fees and auctioned permits induce emitters to reduce use of fertilizer or nitrate.

7. **Farmers do not pay sales tax on fertilizer in California**[20].

Promising Actions

Regulatory Programs and Options

1. **Provide immediate safe drinking water to groundwater nitrate affected areas.** Since nitrate source reduction or groundwater remediation will take years to decades to significantly improve drinking water quality,[21] residents currently receiving unsafe drinking water require other alternatives in the immediate future, regardless of source reduction and management.

2. **Identify populations at risk of contaminated drinking water.** California Department of Public Health (CDPH), California Department of Food and Agriculture (CDFA), and State Water Board, in coordination with Department of Water Resources (DWR), issue a report every 5 years to identify populations at risk of contaminated drinking water and monitor long-term trends of the State's success in providing safe drinking water as a supplement to the California Water Plan Update.

3. **Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.**

4. **Convene a State Groundwater Data Task Force.** California Environmental Protection Agency (CalEPA), in coordination with California Natural Resources Agency (CalNRA) and CDPH, convene an independently led State Groundwater Data Task Force to examine the efficacy of current State and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.

5. **Convene a State Groundwater Task Force.** CalEPA, CalNRA, and CDPH maintain a permanent and independently led State Groundwater Task Force to periodically assess State technical and regulatory groundwater programs in terms of effectiveness in addressing California's groundwater quality and quantity problems. These reports could be incorporated into each California Water Plan Update.

6. **Examine successful Department of Pesticde Regulation (DPR) programs for lessons to manage nitrogen.** CalEPA and CDFA examine successful Department of Pesticide Regulation (DPR) data collection, analysis, education, and enforcement programs for lessons to manage nitrogen and

[20] (CA State Board of Equalization 2004)
[21] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)

Adminsitrative Record
Page 35631

consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.

7. **Nitrate dischargers incur the social costs of their discharges. This policy is already implied for drinking water costs under Section 13304 of the California Water Code (Porter-Cologne Water Quality Control Act).** Without regulations on nitrate source loading, nonpoint sources do not pay for the impact of their nitrate contamination to groundwater. Requiring nitrate dischargers to pay for nitrate contamination provides incentives for them to reduce nitrate discharges and ensures that those benefitting from nitrate discharge pay for the cost of contamination.

8. **Regulatory actions focus on nitrogen fertilizer use (including organic fertilizer sources).** Most nitrate discharge to groundwater is from nonpoint sources, particularly cropland, with substantial additional loading from manure.[22] Existing regulatory programs appear to address point sources of nitrate contamination well, but could be extended to include nonpoint sources.

9. **Regulatory actions should focus on controlling fertilizer application rather than nitrate leachate.** The high costs and technical difficulties of field, farm-based, or countywide nitrate leachate regulations could be prohibitive. A fertilizer application regulation that reduces nitrate discharges to groundwater is coarser and less direct, but much less costly.

10. **Market-based instruments are likely to perform better than command-and-control approaches.** Market-based approaches have lower costs than command-and-control approaches because they provide dischargers with more flexibility to comply with nitrate regulations.

11. **A fertilizer fee is a promising form of regulation and funding.** In the short run, a fee on fertilizer use (including organic fertilizer sources) seems preferable to a cap-and-trade system since it could take advantage of existing infrastructure. Since fertilizers are currently exempt from sales taxes in California (CA State Board of Equalization 2004), the State could expand sales taxes to include fertilizer purchases. The impact a fertilizer fee has on nitrate contamination depends on a) the sensitivity of demand for fertilizer to a price change and b) the size of the fee. The long-term feasibility and desirability of a cap-and-trade system for nitrogen management could be evaluated.

Funding Programs and Options

1. **Where appropriate, combine funding programs.** Consolidating funding programs should lower administrative and application costs and improve program effectiveness. A single program will also ease demands on community applicants. For example, water supply and wastewater problems are often intertwined, and linking these sources of funding would reduce upgrade costs for small systems.

2. **Allocate funding to long-term drinking water solutions, particularly regionalization or consolidation of small systems.** Small drinking water systems face many challenges, including

[22] See Technical Report 2: Nitrogen Sources and Loading to Groundwater (Viers et al. 2012)

Adminsitrative Record
Page 35632

nitrate-contaminated groundwater. Connecting to larger systems or consolidating with other small systems can allow such problems to be addressed more effectively and at lower cost. But regionalization and consolidation of systems often requires costly upgrades for smaller systems (e.g., water meters or new pipeline) or requires the larger system to take on an undesirable amount of risk. The State and counties have an interest in encouraging regionalization and consolidation activities to avoid longer-term financial difficulties, water system service inadequacies, and public health problems.

3. **Provide more financial assistance to small systems.** Small systems generally have more difficulty applying for funding programs and repaying loans. A grant or other special assistance program could be established to provide additional help to small systems. Such a program might be part of a larger consolidated effort by CDPH, the State Water Board, DWR, an Integrated Regional Water Management Plan, and the counties to address the problems of small water and wastewater systems.

4. **Create State funding programs for domestic well owners and for State small and local small water systems.** Currently, only water systems served by a public entity can apply for State safe drinking water funds (such as the State Revolving Fund). Domestic well owners and small communities with no recognized water systems thereby lack the funding sources available to public water systems. Assistance may also be provided to these small systems to help them form legally recognized entities.

5. **Increase the current mill assessment rate on nitrogen fertilizer to its fully authorized amount.** CDFA already oversees a mill assessment on fertilizer sales for research and education regarding the use and handling of fertilizing materials (including environmental effects), which is currently only half of its authorized amount. Raising the assessment to the fully authorized amount would raise roughly $3 million per year statewide. $1 million of this could be used for research and education regarding the use and handling of fertilizing materials (including environmental effects), or if current statute is changed, to fund some alternative drinking water supply efforts.

6. **Introduce a special fee on nitrogen fertilizer sales statewide, perhaps equivalent to a sales tax.** This economic signal could both reduce nitrogen applications and help fund safe drinking water solutions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs. Expanding sales tax to include fertilizer could generate $28 million per year in the study area and might reduce applied nitrogen by 1.6%.[23] Similar fees/excise taxes could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

[23] Assume: 7.5% sales tax on the cost of nitrogen and a fertilizer retail price of $0.75 per pound of nitrogen, and 500 million pounds (227 GgN) of fertilizer used in the Salinas Valley and Tulare Lake Basin. See Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality (Dzurella et al. 2012)

Adminsitrative Record
Page 35633

7. **Consider a more comprehensive statewide fee on water use to support many beneficial activities**. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities. Assuming a fee is placed on statewide gross urban and agricultural water use,[24] the fee would need to be $0.5-$0.9 per acre-foot per year to provide safe drinking water to the estimated highly susceptible population of the study area.[25] Likewise, a fee only on statewide gross urban water use[26] would need to be $2.3-$4.1 per acre-foot per year.

[24] Statewide gross urban and agricultural water use is estimated at 41.7 MAF per year. (Hanak et al. 2011)
[25] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al. 2012)
[26] Statewide gross urban water use is estimated at 8.7 MAF per year. (Hanak et al. 2011)

Adminsitrative Record
Page 35634

1 Introduction

This report reviews funding and regulatory options to manage present and future nitrate contamination in California's Salinas Valley and Tulare Lake Basin. In this section, we begin with a short description of the study areas, followed by an overview of how California and the Federal government have regulated nitrate in drinking water. In Sections 2-5, we examine existing State, Federal, local, and nongovernmental regulatory programs and then discuss potential future regulatory options available to the State. The format is then repeated for current funding programs and potential future funding options.

1.1 Study Area Description

Drinking water users in the Salinas Valley and Tulare Lake Basin regions are particularly susceptible to nitrate contamination for three main reasons: lack of monitoring of small water systems, magnitude of contamination, and low socioeconomic status.

Monitoring. Many small communities and rural households are on state small or local small systems (unregulated by the State), or on unregulated wells. While community public water systems (serving 15 or more connections) are required by the California Department of Public Health (CDPH) to monitor quarterly nitrate levels, and state-small systems (5-14 connections) are regulated either locally or by CDPH, smaller systems often have no monitoring requirement. Approximately 12% of the Tulare Lake Basin population and 10% of the Salinas Valley receive drinking water from a groundwater well supplying water to fewer than five households.[27] Technical Report 7, Alternative Water Supply Options (Honeycutt et al. 2012) finds that approximately 254,000 people in the study area are susceptible to nitrate contamination of drinking water. Of this total, 220,000 are on community public water systems (>15 connections) or state-small water systems (4-14 connections); and 34,000 are self-supplied (domestic wells) or local-small water systems (2-4 connections).

Contamination. These regions have more and larger nitrate contamination sources than most other regions in California. Major sources of groundwater nitrate contamination are fertilizer and animal manure use on agricultural lands, with some localized contributions from animal farming operations, wastewater treatment plants, food processing facilities, and septic systems. Four of the study area counties rank among the top five counties for 2007 agricultural sales (crop and livestock sales): Fresno ($3.7 billion), Tulare ($3.3 billion), Kern ($3.2 billion), and Monterey County ($2.2 billion) (USDA 2007).

Socioeconomics. Over 17% of the Tulare Lake Basin population and over 10% of the Monterey County population live in poverty (USDA 2008). Many low-income communities cannot afford adequate drinking water treatment or alternative water supplies when nitrate levels exceed the maximum contaminant level (MCL) for drinking water, particularly when served by unregulated state and local

[27] Census block group data from 1990 (U.S. Census Bureau 2011). Refer to Technical Report 7, Alternative Water Supply Options (Honeycutt *et al.* 2012) for a formal definition.

Adminsitrative Record
Page 35635

small water systems. Increased costs and reduced water quality are often linked to the lack of technical, managerial, and financial capacity in small and disadvantaged communities. A recent study showed that community public water systems in areas of the Central Valley with a higher minority population or a lower socioeconomic status have statistically higher nitrate levels and that this disparity is especially prevalent among smaller public water systems (Balazs et al. 2011).

1.2 Regulatory History of Nitrate in Drinking Water

In 1969, faced with the absence of national water pollution or contamination control legislation, California adopted the Porter-Cologne Water Quality Control Act (Porter-Cologne) (California Water Code § 13000 et seq.). Porter-Cologne grants the State Water Resources Control Board (State Water Board) authority over the protection of State water quality and establishes the Regional Water Quality Control Boards (Regional Water Boards) to carry out these policies at the regional and local level. Subsequently, Congress passed the Clean Water Act of 1972 (CWA) (33 U.S.C. § 1251 et seq.) and the Safe Drinking Water Act of 1974 (SDWA) (42 U.S.C. § 300f et seq.). These acts provide the Federal regulatory framework to manage contaminants in water bodies and drinking water. While the SDWA regulates the quality of delivered drinking water in public water supply systems, the CWA regulates the discharge of contaminants into surface waters of the United States. The CWA does not regulate the contamination of groundwater, which is the focus of this report.

Under the authority of the SDWA, the U.S. Environmental Protection Agency (U.S. EPA) develops and sets drinking water quality standards and oversees State or local implementation of the standards. In 1992, U.S. EPA's Phase II Rule established the Federal maximum contaminant level (MCL) for nitrate in drinking water (U.S. EPA 2011a). California's Office of Environmental Health Hazard Assessment later performed its own risk assessment of nitrate and adopted the Federal MCL in 1997 as a public health goal (PHG), or the level at which no known or anticipated adverse effects on health will occur (costs to comply with this level are not considered) (CDPH 2008).

In California, the Federal framework provided by the CWA and the SDWA are implemented through two separate agencies; the State Water Board implements the CWA, and CDPH implements the SDWA. Currently, only the State Water Board has authority to regulate activities that adversely affect the quality of drinking water sources. Although CDPH does not have authority to regulate sources of contamination to the groundwater, it still maintains groundwater programs with the ultimate goal of protecting the provision of safe drinking water. For example, CDPH's Drinking Water Source Assessment and Protection (DWSAP) program collects monitoring data on possible contaminating activities near drinking water sources (septic tanks, landfills, etc.).

In contrast to the CWA, Porter-Cologne regulates discharges to both surface water and groundwater in California. It requires the State Water Board and the Regional Water Boards (both together, the California Water Boards) to regulate waste discharge to these water bodies from both point sources and nonpoint sources. The Regional Water Boards handle National Pollutant Discharge Elimination System (NPDES) permits for point source discharges to surface water and develop various permit programs (e.g.,

Adminsitrative Record
Page 35636

waste discharge requirements (WDRs), discharge permits, and conditional waivers of waste discharge) for nonpoint and point source discharges to surface water and groundwater. Under the authority of Porter-Cologne, the Regional Water Boards also develop a "basin plan" that explicitly identifies all beneficial uses of individual water bodies (surface water and groundwater) within a Regional Water Board's region and develops measures to protect these beneficial uses. Waste discharge requirements set by the Regional Water Boards must be consistent with the basin plan objectives, including the State Water Board's anti-degradation policy (State Water Board Resolution 68-16), which requires that existing high water quality be maintained to meet beneficial uses (State Water Board 2006). Specifically, any actions that affect surface water or groundwater quality "must (1) be consistent with maximum benefit to the people of the State, (2) not unreasonably affect present and anticipated beneficial use of the water, and (3) not result in water quality less than that prescribed in water quality plans and policies." (State Water Board 2006). Since this anti-degradation policy is a State Water Board resolution, it may be changed (with respect to groundwater)[28] at the discretion of the State Water Board and does not need new legislative authority. Despite the mandates under Resolution 68-16, currently there are no permit requirements placed on agricultural non-point source discharges to groundwater.

In the groundwater basins of the Central Valley and Central Coast regions of California, monitoring of nitrate began long before the national nitrate MCL was established. Since the early 1950s, the Department of Water Resources (DWR) has been gathering nitrate data in areas of the Salinas Valley (Snow, Mills, & Zidar 1988). In 1978, the Association of Monterey Bay Area Governments (AMBAG) published a study concluding that agricultural activities were the primary contributors to the high nitrate levels in the groundwater (AMBAG 1978). Then in 1988, the Monterey County Water Resources Agency (MCWRA) produced the first report documenting nitrate levels in groundwater in the Salinas Valley (Snow *et al.* 1988). Similarly, a 1989 study by the California Department of Food and Agriculture's Nitrate Working Group quantified various sources of nitrate (Nitrate Working Group 1989). These reports were among the first to point out the current and future nitrate trends and how this could harm public health in these areas.

In response to the problem of nitrate in groundwater, planning, regulatory, and funding programs were developed to reduce future contamination and mitigate the health and financial effects of existing contamination. The next section reviews the existing planning and regulatory programs and Section 4 reviews the existing funding programs.

[28] There is a federal anti-degradation policy for surface water, but not for groundwater.

Adminsitrative Record
Page 35637

2 Current Planning and Regulatory Programs That Address Nitrate in Groundwater

Planning and regulatory programs have been implemented at the Federal, State, and local levels to reduce the adverse effects of nitrate-contaminated drinking water. These programs contain monitoring, enforcement, and other guidelines, that could provide the foundation for future programs to address both the public health and economic impacts of nitrate contamination. This section considers the strengths and weaknesses of current programs and recommends future actions to enhance their effectiveness.

2.1 Overview of Current Planning and Regulatory Programs

Many regulatory and planning programs in the study area provide regulatory structure or technical and managerial support to water systems, communities, farmers, dairies, and others who deal with nitrate contamination in groundwater. Statutes also provide a regulatory framework for nitrate contamination of groundwater and drinking water. Current regulatory/planning programs and statutes that have the ability to reduce groundwater nitrate contamination are summarized in Table 1. These programs/statutes have components that target nitrate source reduction or groundwater remediation. Table 2 is a summary of current programs and statutes related to groundwater nitrate and drinking water. These provide for data collection, information, and education on nitrate sources and groundwater nitrate. Some of these programs regulate nitrate in drinking water.

For a more detailed description of all programs refer to Appendix B: Summary of Current Planning and Regulatory Programs. In the study area, there are several Federal programs/statutes (Table 1 and Table 2, blue), State programs/statutes (purple), and nongovernmental programs/agencies (orange) relevant to nitrate contamination and its effects on drinking water.

Adminsitrative Record
Page 35638

Table 1. Summary of directly applicable programs and statutes for reducing nitrate contamination in groundwater.

AGENCY	PROGRAM/STATUTE [year created/passed]	GOAL/PURPOSE
U.S. Environmental Protection Agency (U.S. EPA)	Supplemental Environmental Programs (SEP) [1998]	Environmentally beneficial project that a violator of environmental laws may choose to perform (under an enforcement settlement) in addition to the actions required by law to correct the violation.
State Water Resources Control Board (State Water Board)	Porter-Cologne Water Quality Control Act [1969]	Grants the State Water Board authority over State water quality policy and aims to regulate activities in California to achieve the highest reasonable water quality.
	Recycled Water Policy [2009]	Resolution No. 2009-0011: Calls for development of salt and nutrient management plans and promotes recharge of clean storm water.
Regional Water Quality Control Boards	Cleanup and Abatement Order (CAO)	CA Water Code § 13304: Allows the Regional Water Board to issue a directive to a polluter to require cleanup of waste discharged into waters of the State.
Central Coast Regional Water Quality Control Board	Irrigated Lands Regulatory Program (ILRP) [2004, draft in 2011]	*General Conditional Waiver of Waste Discharge Requirements, 3-Tiered Agricultural Regulatory Program (2004):* Groundwater quality monitoring required to different degrees based on discharger's tier. *Draft (2001):* Requires Tier 3 dischargers with high nitrate loading to meet specified Nitrogen Mass Balance Ratios or implement a solution that leads to an equivalent nitrogen load reduction.
Central Valley Regional Water Quality Control Board	Irrigated Lands Regulatory Program (ILRP) [2003, draft in 2011]	*Conditional Wavier of Waste Discharge Requirements of Discharges from Irrigated Lands:* Interim program to regulate irrigated lands. Does not address groundwater. *Recommended ILRP Framework (2011):* Development of new monitoring and regulatory requirements (includes groundwater).
	CV-SALTS [2006]	Planning effort to develop and implement a basin plan amendment for comprehensive salinity and nitrate management.
	Dairy Program [2007]	*Waste Discharge Requirements General Order for Existing Milk Cow Dairies:* Confined animal facilities must comply with set statewide water quality regulations, and existing milk cow dairies must conduct nutrient and groundwater monitoring plans.
California Department of Food and Agriculture (CDFA)	Feed, Fertilizer and Livestock Drugs Regulatory Services (FFLDRS)	Manages licenses, registration and inspection fees, and a mill tax levied on fertilizer sales, to fund research and educational projects that improve fertilizer practices and decrease environmental impacts from fertilizer use.

Adminsitrative Record
Page 35639

Table 2. Summary of programs and statutes related to groundwater nitrate and drinking water (data collection, information, education, or regulation of drinking water).

AGENCY	PROGRAM/STATUTE [year created/passed]	GOAL/PURPOSE
U.S. Environmental Protection Agency (U.S. EPA)	Safe Drinking Water Act (SDWA) [1974, 1986, 1996]	Mandates EPA to set the drinking water standards and to work with States, localities, and water systems to ensure standards are met.
	Phase II Rule [1992]	Established Federal maximum contaminant level (MCL) for nitrate in public water systems.
	Enforcement Response Policy – Enforcement Targeting Tool	Focuses on high-priority systems with health-based violations or monitoring or reporting violations that can mask acute health-based violations.
U.S. Department of Agriculture (USDA)	Rural Utilities Service: National Drinking Water Clearinghouse [1977]	Provides technical assistance and educational materials to small and rural drinking water systems.
California Department of Public Health (CDPH)	22 CCR § 64431	Established State maximum contaminant level (MCL) for nitrate in public water systems.
	Drinking Water Source Assessment and Protection (DWSAP)	Evaluation of possible contaminating activities surrounding groundwater and surface water sources for drinking water.
	Expense Reimbursement Grant Program (EPG)	Education, training, and certification for small water system (serving <3,301 people) operators.
	Groundwater Ambient Monitoring and Assessment (GAMA)	Improves statewide groundwater monitoring and increases availability of groundwater quality information. Funded by Prop 50 and special fund fees.
Assembly Bill 3030	[1993]	Permits local agencies to adopt programs to manage groundwater and requires all water suppliers overlying useable groundwater basins to develop groundwater management plans which include technical means for monitoring and improving groundwater quality.
Kern County Water Agency (KCWA)	[1961]	Collects, interprets, and distributes groundwater quality data in Kern County.
Monterey County Health Department		Implements a tiered, regular nitrate sampling program based on increasing nitrate concentration for local small water systems and for state-small water systems in Monterey County.
Southern San Joaquin Valley Water Quality Coalition	[2002]	Protects and preserves water quality in the Tulare Lake Basin through surface water quality monitoring and dissemination of collected data. Particular focus is on agricultural discharge areas. Does not currently focus on groundwater.
Tulare County Water Commission	[2007]	Discusses water issues impacting Tulare County and advises the Tulare County Board of Supervisors. Special focus on nitrate in groundwater and improving drinking water in small communities.

Adminsitrative Record
Page 35640

Monterey County Water Resources Agency (MCWRA)	[1947]	Provides water quality management and protection through groundwater quality monitoring (including nitrate levels) and research and outreach efforts to growers to improve fertilizer management and reduce nitrate leaching.
The Waterkeeper Alliance	Monterey Coastkeeper [2007]	Collaborates with the State Water Board to ensure effective monitoring requirements for agricultural runoff and more stringent waste discharge requirements for other nitrate sources.
Rural Community Assistance Partnership (RCAP)	[1979]	Uses publications, training, conferences, and technical assistance to help communities of less than 10,000 people to access safe drinking water, treat & dispose of wastewater, finance infrastructure projects, understand regulations, and manage water facilities.
National Rural Water Association (NRWA)	[1976]	Offers drinking water system technical advice (operation, management, finance, and governance) and advocates for small/rural systems to ensure regulations are appropriate.
California Rural Water Association	[1990]	Provides online classes, onsite training, low cost educational publications, and other forms of technical advice for rural water and wastewater systems.
Self-Help Enterprises (SHE)	Community Development Program [1965]	Provides technical advice and some seed money to small/rural/poor communities for the planning studies and funding applications associated with drinking water system projects.
Community Water Center	Association of People United for Water (AGUA) [2006]	Advocates for regional solutions to chronic local water problems in the San Joaquin Valley. Focused on securing safe drinking water, particularly from nitrate impacted sources.

Despite the long list of programs and statutes related to nitrate contamination (Table 1 and 2), very few can be directly applied to decreasing nitrate contamination to groundwater (Table 1). Additionally, to date, these programs/statutes have been insufficient to control nitrate contamination of groundwater. Overall, nitrate concentrations in groundwater have not decreased in the last three decades; concentrations have increased in many areas.[29] Though Federal law establishes a nitrate MCL, State law has not implemented a regulatory program stringent enough to ensure that groundwater nitrate concentrations are at or below the drinking water standard. While dischargers are supposed to be held responsible for adverse effects to groundwater (under Porter-Cologne), no current or historical regulatory program functionally holds nitrate dischargers responsible. This may develop in the near future with the current development of the Irrigated Lands Regulatory program in the Central Valley and with the 2007 Central Valley Dairy General Order, which will see stricter enforcement over the next few years.

[29] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)

Adminsitrative Record
Page 35641

Some ongoing efforts with potential to reduce nitrate contamination in the future are the Agricultural Regulatory Program by the Central Coast Regional Water Quality Control Board (Central Coast Regional Water Board), the renewal of the Irrigated Lands Regulatory Program by the Central Valley Regional Water Quality Control Board (Central Valley Regional Water Board), and the development of comprehensive salt and nitrate management regulatory programs across California under a State Water Board mandate, the Central Valley Salinity Coalition (CV-SALTS).

Some regulatory programs have recently introduced mandatory monitoring programs. While monitoring programs are essential to understanding nitrate contamination and evaluating the success of nitrate management programs, monitoring alone will not improve water quality. Monitoring programs are more likely to be successful when they are adopted in conjunction with immediate safe drinking water options or longer-term source reduction and data management actions.

Immediate safe drinking water actions

Currently, many details are still unknown about nitrate contamination. Given the physical properties of nitrate in the groundwater, it is difficult to understand how, where, and when a contamination source will affect groundwater, and ultimately, drinking water.[30] We know that it will take years to decades for a nitrate source reduction or groundwater remediation program to significantly improve drinking water quality,[30] so alternative water supply options are necessary immediate actions to ensure safe drinking water.

Organized monitoring is needed to understand who is facing the most risk, but none of the current safe drinking water regulatory programs (see Table 1, Table 2, and Appendix B) have used monitoring data to explicitly identify populations at risk of contaminated drinking water. The "closest" program is CDPH's DWSAP which identifies possible contaminating activities near groundwater sources of drinking water. While this program identifies the contamination to which the drinking water source is most vulnerable, it does not mandate action to help reduce future contamination, nor does it identify the State's most highly susceptible populations.

One option is that CDPH and the State Water Board, in coordination with DWR, issue a report every five years to identify populations at risk of contaminated drinking water and monitor long-term trends of the State's success in providing safe drinking water. This report could supplement each California Water Plan Update.

Longer-term source reduction and data management actions

To ensure long-term protection for sources of drinking water, nitrate source reduction actions will be needed. Many current source reduction efforts such as the Dairy Program and the Irrigated Lands Program (ILRP) include plans for groundwater monitoring so that the success of these programs can be evaluated (see Table 1, Table 2, and Appendix B), but none have gathered information at the regional

[30] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)

Adminsitrative Record
Page 35642

scale to better understand areas at high risk of contamination. The Regional Water Boards could then use these data to officially designate groundwater drinking water sources at risk for nitrate contamination.

Currently, multiple agencies under many planning and regulatory programs hold nitrate monitoring data (see Table 2 and Appendix B). This dispersion of data holdings may lead to a duplication of data collection efforts and make it difficult to gain a comprehensive understanding of nitrate contamination of groundwater. Unfortunately, no feedback mechanism exists to assess current monitoring, planning, or regulatory programs, and therefore no method to identify data gaps or cost-effectiveness. An independently-led State Groundwater Task Force, convened by the California Environmental Protection Agency (CalEPA) in coordination with the California Natural Resources Agency (CalNRA) and CDPH, could evaluate the efficacy and potential overlap of such programs throughout the State.

The California Department of Food and Agriculture (CDFA) currently administers the Feed, Fertilizer and Livestock Drugs Regulatory Services (FFLDRS) program to license, register, collect inspection fees, and manage the mill tax on fertilizer sales (see Table 1 and Appendix B). However, this program does not collect data on fertilizer applications (where, how much, etc.), which could provide information for understanding groundwater nitrate contamination. Currently, the California Department of Pesticide Regulation (DPR) manages the full reporting of agricultural pesticide use (the county agriculture commissioners are required to report all agricultural pesticide use monthly), as required by State regulations (3 CCR sections 6624 – 6628) and conducts groundwater monitoring programs. This type of program is lacking in California for nitrate use, so one option could be for CalEPA to evaluate promising solutions, e.g., the creation of a new program in CalEPA to coordinate with DPR, or the expansion of the current DPR program for the reporting of nitrogen applications (including synthetic fertilizer and any organic sources of land applied nitrogen).

2.2 Major Findings: Current Planning and Regulatory Programs

1. **To date, regulatory programs have been insufficient to control nitrate contamination of groundwater.** Discharges from agriculture (fertilizers and animal wastes) are the largest source of nitrate contamination of groundwater,[31] but there is no current or historic regulatory program that functionally holds these dischargers responsible (with the possible exception of the 2007 Central Valley Dairy General Order, which will see stricter enforcement over the next few years). Overall, nitrate concentrations in groundwater have not decreased in the last three decades; in fact, concentrations have even increased in some areas.[32] Some regulatory programs have recently introduced mandatory monitoring programs, but monitoring alone will not improve water quality.[32]

2. **Many years are needed for regulatory actions to reduce nitrate in groundwater and improve drinking water quality.** The physical properties of nitrate in groundwater mean that regulatory

[31] See Technical Report 2: Nitrogen Source and Loading to Groundwater (Viers et al. 2012)
[32] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)

Adminsitrative Record
Page 35643

actions on fertilizer or nitrate leachate today will not bring drinking water sources into water quality compliance for years to decades.[32]

2.3 Promising Actions: Current Planning and Regulatory Programs

1. **Provide immediate safe drinking water to groundwater nitrate affected areas.** Since nitrate source reduction or groundwater remediation will take years to decades to significantly improve drinking water quality,[33] residents currently receiving unsafe drinking water require other alternatives in the immediate future, regardless of source reduction and management.

2. **Identify populations at risk of contaminated drinking water.** California Department of Public Health (CDPH), California Department of Food and Agriculture (CDFA), and State Water Board, in coordination with Department of Water Resources (DWR), issue a report every 5 years to identify populations at risk of contaminated drinking water and monitor long-term trends of the State's success in providing safe drinking water as a supplement to the California Water Plan Update.

3. **Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.**

4. **Convene a State Groundwater Data Task Force.** California Environmental Protection Agency (CalEPA), in coordination with California Natural Resources Agency (CalNRA) and CDPH, convene an independently led State Groundwater Data Task Force to examine the efficacy of current State and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.

5. **Convene a State Groundwater Task Force.** CalEPA, CalNRA, and CDPH maintain a permanent and independently led State Groundwater Task Force to periodically assess State technical and regulatory groundwater programs in terms of effectiveness in addressing California's groundwater quality and quantity problems. These reports could be incorporated into each California Water Plan Update.

6. **Examine successful Department of Pesticde Regulation (DPR) programs for lessons to manage nitrogen.** CalEPA and CDFA examine successful Department of Pesticide Regulation (DPR) data collection, analysis, education, and enforcement programs for lessons to manage nitrogen and consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.

[33] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)

Adminsitrative Record
Page 35644

3 Future Regulatory Options to Manage Nitrate

Section 3 outlines potential future regulatory options to manage nitrate in groundwater. We first discuss the four dimensions along which we compare proposed regulatory options: 1) abatement costs; 2) monitoring and enforcement costs; 3) information requirements; and 4) their ability to raise revenues. In Section 3.2, we discuss whether future regulations should focus on fertilizer application or nitrate leachate discharge to groundwater. We then summarize an array of available regulatory options and ultimately recommend market-based regulations of fertilizer over command-and-control approaches and regulation of nitrate leachate. Case studies describing regulatory options appear in Appendix A: Regulatory Options in Practice.

3.1 Assumptions, Limitations, and Analytical Criteria

There are several assumptions and limitations of this analysis. First, we look only at nitrate contamination that originates from agricultural sources (including organic fertilizer sources). Agricultural sources are the primary source of nitrate in the Salinas Valley and Tulare Lake Basin with synthetic fertilizer application to crops amounting to 52 percent of annual nitrogen loading and manure application contributing over 31 percent to annual nitrogen loading.[34] While all dischargers are supposed to be held responsible for adverse effects to groundwater (under Porter-Cologne, see Section 1.2 above), there is no current or historic regulatory program that holds agricultural dischargers responsible (with the possible exception of the 2007 Central Valley Dairy General Order, which will see stricter enforcement over the next few years).

Second, our comparison of regulatory options is qualitative (as opposed to quantitative) – each policy option is ranked as high, medium or low for each evaluation criterion. To rank and compare regulatory options, we relied on the economics literature that describes how these instruments should work in theory; discussions with experts (on groundwater nitrate contamination) about the current and future state of knowledge; and case studies that highlight the lessons learned from the actual implementation of these regulations (see Appendix A: Regulatory Options in Practice).

Regulatory options can be compared in many ways. We choose four metrics:

- **abatement costs** (the cost incurred by non-point sources to reduce nitrate)
- **administrative costs** (monitoring and enforcement costs)
- **information requirements**
- **revenue raising**

Depending on the audience, each criterion may take on a range of interpretations. To remain consistent, we define each dimension before beginning our analysis. Based on the ranking of each

[34] See Technical Report 2: Nitrogen Sources and Loading to Groundwater (Viers et al. 2012)

Adminsitrative Record
Page 35645

regulatory option along these four criteria, we determined the most promising options to manage nitrate.

Abatement costs measure the costs to dischargers to reduce nitrogen loading so as to achieve a nitrate standard. This criterion does not include monitoring, enforcement, and information costs. Rather it focuses on the overall nitrate reduction costs incurred by non-point sources to achieve a nitrate goal. For example, an abatement cost could be the cost incurred by a farm to install a technology that reduces nitrate runoff. Historically, abatement costs have been the largest share of costs for pollution control policies (Keohane & Olmstead 2007).[35]

Administrative costs describe the costs to implement, monitor and enforce a policy instrument, and typically are the second largest share of costs for pollution control policies (Keohane & Olmstead 2007). These costs vary across regulations because of differences in technical feasibility. Administrative costs for nitrate contamination of groundwater will depend on "mixing" in the environment, specifically the mixing between the point of discharge into groundwater and the point of use. With uniform mixing, each unit of contamination has the same effect on environmental quality, regardless of where and when the contaminant is discharged. In contrast, with nitrate in groundwater, the effect of a single unit of nitrate leachate on drinking water quality largely depends on where (relative to the drinking water source location) and when nitrate is discharged. We refer to this as "non-uniform mixing."[36] Higher administrative costs from "non-uniform mixing" will influence the choice of a regulatory instrument. For example, it is difficult to measure the amount of nitrate leachate from a farm, so the administrative costs needed to implement, monitor and enforce a regulation on nitrate leachate are likely high. A further complication with a nitrate leachate regulation is that many farmers often affect the regional water quality, so it may be difficult to trace nitrate emissions to a specific nonpoint source. In contrast, it is easy to monitor and measure fertilizer application and therefore administratively cheaper to regulate fertilizer use.

Information requirements describe the information needed to implement a regulation. For example, before establishing a regulation for nitrate leachate, the mixing properties of nitrate and the time horizon for nitrate to reach drinking water sources must be understood.

Revenue-raising describes the ability of a regulatory instrument to generate funding. Some regulatory instruments may both regulate nitrate and raise revenue. We choose revenue-raising as a criterion since Senate Bill SBX2 1 (SBX2 1) also requires an investigation of funding options to support remediation, treatment, or alternative water supplies.

[35] Also refer to Section 3 in Technical Report 3 (Dzurella et al. 2012)

[36] Due to the physical properties of nitrate in soil and groundwater, contamination does not mix evenly in the groundwater and the exact leachate rate to the drinking water source varies depending on when and where nitrate leachate is released.

Adminsitrative Record
Page 35646

3.2 What Can Be Regulated?

Before discussing the various regulatory options, we must discuss what to regulate. We focus on two regulatory units – fertilizer application (including manure) and nitrate leachate (from fields to groundwater). An instrument that regulates fertilizer application is an input regulation, since it regulates an input to nitrate contamination rather than actual nitrate contamination of groundwater. In contrast, a nitrate leachate regulation directly targets nitrate contamination of groundwater.

In theory, the abatement costs of a nitrate leachate regulation should be lower than those from a fertilizer regulation. This occurs because farmers have more flexibility in complying with a nitrate leachate regulation than a fertilizer regulation. As an example, suppose the regulator can choose between a performance standard on fertilizer application or nitrate leachate. A performance standard might establish either a per hectare limit on fertilizer use or on an allowable nitrate leachate. With a per hectare fertilizer limit, farms can only attain the performance standard by reducing their use of fertilizer. By contrast with a fertilizer standard, farms have more flexibility in complying with a nitrate leachate standard, so the cost of abatement will likely be less. A farm can reduce nitrate leachate by: reducing the quantity of fertilizer applied to a hectare of land, reducing irrigation water applications, changing the crop mix, or combining various options.

A second advantage of a nitrate leachate regulation is that the likelihood of a drinking water quality standard being met is higher. With a fertilizer application regulation, there is uncertainty over the impact that a unit of fertilizer will have on nitrate contamination of groundwater over time, making it difficult to assess whether compliance with a fertilizer regulation will lead to compliance with a water quality standard.

A nitrate leachate regulation targets the contaminant itself and in theory, the abatement costs of this regulation are lower. However, the administrative costs and information requirements to implement this direct leachate regulation far exceed those to regulate fertilizer use. A nitrate leachate regulation must account for "non-uniform mixing," which is the complex movements of nitrate to water sources. The effect of a pound of nitrate on water quality within an aquifer varies with the location of discharge, climate, and the characteristics of the underlying aquifer. Before introducing a leachate regulation, we would need to understand how nitrate leached from a given nonpoint source ultimately affects groundwater quality around that specific site. Because of natural and anthropogenic variability, gaining an adequate understanding would likely have prohibitively high information requirements. The non-uniform mixing aspect of a nitrate leachate regulation also makes the administrative costs high.

As a regulation incorporates measures to account for non-uniform mixing, the likelihood of meeting a contaminant standard increases, but so do the administrative costs and the information requirements. Imagine that a basin-wide cap-and-trade system (or a tax or prescriptive standard) on nitrate leachate was introduced. If a regulator established a tradable permit system in which farms were allowed to trade on a one-to-one basis (a pound of nitrate from farm A for a pound of nitrate from any other farm, regardless of the day or the location), then the complexity and variability of nitrate flows might cause

Adminsitrative Record
Page 35647

the nitrate concentration in water sources to exceed the cap. This problem holds true for any regulation on nitrate, whether it is cap-and-trade, fees/taxes, technology standards, or compliance standards.

The establishment of trading ratios, similar to an exchange rate, can allow a permit system to account for non-uniform mixing. Ideally, an individual exchange rate would be set up for all potential trades between sources and across time. Continuing with the earlier example, it may be that farm A can now discharge one pound of leachate if it purchases two pounds of permits from farm B or three pounds of permits from farm C. The exchange rate for permits between A & B, A & C, and B & C is ½, ⅓, and ⅔, respectively. While ideal in theory, a leachate regulation that accounts for non-uniform mixing would be difficult and expensive to implement. The information requirements needed to understand how nitrate leachate from a given field source ultimately affects water quality are large. Even if we could fully define the relationship between source-specific nitrate effluent and groundwater quality, which we cannot, the costs to monitor and enforce this regulation would be high, perhaps prohibitively so. The abatement costs, administrative costs, and information requirements are summarized in Table 3 for fertilizer application and for nitrate leachate.

Table 3. Regulating fertilizer application versus nitrate leachate.

Regulated Entity	Abatement Costs	Administrative Costs	Information Requirements
Fertilizer Application	Higher – regulate input	Low	Low
Nitrate Leachate	Lower – regulate contaminant	High	High

Though the high administrative costs and information requirements of a nitrate leachate regulation make a fertilizer regulation preferable, a regulation on fertilizer application also has its weaknesses. A fertilizer regulation does not guarantee a nitrate leachate standard will be met, because fertilizer use is only an indirect measure of the amount of nitrate leached to groundwater. This weakness may be avoided, or at least decreased, through careful structuring of the fertilizer regulation, e.g., by accounting for the amount of harvested nitrogen expected for each crop type (but in turn, this may increase administrative costs). Regulating both nitrate leachate and fertilizer application might also avoid this problem, but would likely impose the high administrative and implementation costs of a leachate regulation together with the higher abatement costs of a fertilizer regulation. However, if farms already provide detailed accounting of air emissions, pesticide use, water use, salt discharges, and other environmental fluxes, then perhaps the additional costs of nitrate accounting would not be too great.

One potential statutory obstacle to implementing a fertilizer regulation is that Porter-Cologne is currently set up to regulate based on discharge requirements (nitrate leachate), not inputs like fertilizer application. Well-designed regulatory programs may be able to work around this obstacle. For example, a fertilizer application regulation with a “representative” groundwater monitoring program

Adminsitrative Record
Page 35648

could be used to confirm the degree to which the regulation of fertilizer applications can succeed in meeting groundwater quality objectives.

3.3 Direct Regulatory Options

We now discuss various regulatory options to manage nitrate contamination of groundwater, beginning with a description of technology mandates and performance standards. We refer to these regulatory options as direct regulatory options or prescriptive policies, since they directly regulate the behavior of non-point sources.

3.3.1 Technology Mandates

A technology mandate would require all nitrate dischargers to install a specific technology or follow a specific management practice. Technology mandates to reduce the quantity of nitrate entering aquifers could include minimum seal depth for wells and backflow prevention devices.

For example, the State of California requires farmers to follow a pre-determined set of management practices to manage pesticides. Specifically, DPR restricts pesticide-handling practices within 100 feet of unprotected wellheads.[37]

3.3.2 Performance Standards

A performance standard sets a limit on the quantity of nitrate effluent that an individual farm or dairy can discharge. The regulation may specify a maximum effluent rate (pounds of nitrate effluent per hectare) or a maximum allowable quantity of fertilizer application (pounds of nitrogen per hectare) or a nitrogen mass balance ratio.

An example of a performance standard is seen with the WDR General Order for Existing Milk Dairy Cows (No. R5-2007-0035), which requires dairies to maintain a land application of total nitrogen that is less than 1.65 times the total nitrogen removed from the land through harvest and crop removal. The regulation does not outline how a dairy must achieve this standard; rather it sets a standard and allows the dairy to choose how to meet this ratio.

3.4 Taxes and Fees

Market-based instruments, such as cap-and-trade and taxes (fees), regulate fertilizer application (or total nitrate contamination) in the area of study (the Salinas Valley and Tulare Lake Basin), rather than fertilizer application (or groundwater contamination) from each nonpoint source, as is the case with

[37]California Department of Pesticide Regulation. Groundwater Protection Program Regulations. Accessed at: http://www.cdpr.ca.gov/docs/emon/grndwtr/gwp_regs.htm. (CA DPR 2011)

Adminsitrative Record
Page 35649

technology mandates and performance standards. With a fertilizer fee, a price is placed on fertilizer and the quantity of fertilizer application is determined by farmers[38].

3.4.1 Nitrate Leachate Tax

A nitrate leachate tax or fee establishes a price per unit of nitrate leachate in the recharge area of the drinking water source. Since nonpoint sources must pay for the nitrate they discharge, they are forced to internalize (at least in part) the cost that groundwater contamination imposes on society (e.g., health impacts from consumption of contaminated drinking water). Taxes on water contamination have been used to regulate heavy metals and organic discharges in the Netherlands, water pollution in France, industrial polluters in Colombia, and the palm oil industry in Malaysia (Olmstead 2010). A nitrate effluent fee must account for "non-uniform mixing" of nitrate in the water source. Differentiated fees are needed to address this non-uniformity. These fees should be source specific, and reflective of the effect that a unit of nitrate effluent from a given source will have on drinking water quality.

3.4.2 Fertilizer Tax

A fertilizer tax or fee is an input fee since it is placed on an input to nitrate contamination, rather than the nitrate contamination itself. In theory, as the cost of fertilizer use increases, farms use less fertilizer, switch which crops are grown, and adopt more fertilizer-efficient technologies.

3.5 Cap-and-Trade

With cap-and-trade regulation, a total quantity of allowable leachate or fertilizer application is established in a county or aquifer. Tradable permits or allowances are then allocated to the regulated farmers or auctioned off in a market, with each permit corresponding to a unit of allowable nitrate leachate to the groundwater or to a unit of applied fertilizer. A cap-and-trade approach sets the quantity on fertilizer application or nitrate leachate emissions and allows the market to determine the price per unit of effluent or fertilizer. In the United States, Title IV of the Clean Air Act Amendments addressed the problem of acid rain by creating a cap-and-trade system for the regulation of sulfur dioxide (Stavins 2001). For a case study example of cap-and-trade, see Appendix A: Regulatory Options in Practice, Watershed-Based Effluent Trading.

3.5.1 Fertilizer Cap-and-Trade

One potential regulatory option is to set a cap on the quantity of fertilizer use in the study area counties. Under such a cap-and-trade system, initial fertilizer permits could be freely distributed or auctioned off. With an auction, the sale price for fertilizer would reflect the social cost of fertilizer, which includes the private marginal cost to produce fertilizer and the marginal social damages caused by a unit of fertilizer.

[38] See Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality (Dzurella et al. 2012)

Adminsitrative Record
Page 35650

3.5.2 Nitrate Leachate Cap-and-Trade

An alternative policy to regulate nitrate leachate would be to cap the concentration of nitrate in the underlying aquifer and establish a water quality trading program for nonpoint sources within the aquifer. One example of water quality trading for point sources is the salinity trading program in Australia's Hunter River (Olmstead 2010). Under this (surface water) program, point sources were freely allocated permits to discharge saline water into the river and a real-time trading system was implemented, since the impact of saline water on water quality in the river depends on constantly changing flow conditions. An analogous agricultural groundwater quality trading program for nitrate could be employed.

3.6 Hybrid Solutions

Hybrid options are also available to regulate nitrate. For over 15 years, the Netherlands has used a hybrid approach to manage nitrate (for the detailed case study, see Appendix A: Regulatory Options in Practice, The Dutch Experience). Under this system, agricultural sources are regulated using a performance standard and a fertilizer fee. Hybrid regulations may offer a practical alternative to manage nitrate leachate.

3.7 Information Disclosure

In the area of study, approximately ten percent of households receive drinking water from local small water suppliers (2-4 connections) or rely on domestic wells, both of which are unregulated at the State or local level (see Technical Report 7, Alternative Water Supply Options, Honeycutt et al. 2012). Information disclosure about the quality of drinking water from these sources may be a low-cost and viable regulation to manage nitrate for unregulated water systems. Information disclosure will not decrease nitrate in groundwater levels, but it may help encourage the installation of affordable treatment devices or the use of bottled water, thereby reducing the public health risk of nitrate-contaminated groundwater. Recent work suggests that community water suppliers reduced violations of drinking water quality standards after they were required to disclose violations to their customers (Bennear & Olmstead 2008).

Under the SDWA, community water suppliers in the U.S. have been required since 1998 to publish Consumer Confidence Reports (CCRs) that describe a water system's compliance with water quality standards. In California, CDPH extends the applicability of CCRs, requiring community systems serving at least 15 connections to provide annual CCRs. Extending mandatory information disclosure about drinking water quality to smaller systems may lead to improved drinking water quality and a decreased public health risk, at a relatively low cost. Additionally, information disclosure could be expanded to include discharges to groundwater by nitrate dischargers (governed under California's Porter-Cologne Act).

Adminsitrative Record
Page 35651

3.8 Liability Rules

Under liability rules, decision makers can hold either the polluter or the manufacturer of contaminants responsible for contamination. Liability rules could be used to force the polluter to internalize the external costs of nitrate contamination to groundwater. Liability rules for groundwater contamination are already in place, both domestically and in the European Union. In the U.S., the CWA established strict liability of polluters of oil and other hazardous substances. More recently, the European Union has set up The Environmental Liability Directive in which polluters are held responsible for the environmental damage they cause to water, soil or species (Olmstead 2010). For detailed examples of liability rules in action, see Appendix A: Regulatory Options in Practice, Liability from Groundwater Contamination from Pesticide Use.

Section 13304 of Porter-Cologne states that "A cleanup and abatement order issued by the State Water Board or a Regional Water Board may require the provision of, or payment for, uninterrupted replacement water service, which may include wellhead treatment, to each affected public water supplier or private well owner." This provides authority for a Regional Water Board to require landowners contributing to nitrate risk to groundwater drinking water supplies to support drinking water actions for affected public water supplies and private wells. In a sense, this establishes a liability rule and mechanism to regulate nitrate contamination of drinking water sources.

3.9 Negotiation or Payment for Drinking Water Quality

Another regulatory option to manage nitrate in drinking water or groundwater is to establish a market for the payment of drinking water quality, or more generally for environmental or ecosystem services. Environmental services describe benefits provided by healthy ecosystems, including biodiversity conservation, carbon sequestration, watershed protection, and natural treatment systems for drinking water. Property owners benefit from these environmental services; however, most of the benefits from these services accrue to external parties. Since private landowners are not compensated for the social benefits from these services, in the absence of market intervention, these valuable services will be underprovided.

In Mexico, one of the largest Payment for Environmental Services (PES) programs has been established to protect forestland from deforestation and safeguard the hydrologic services that it provides (CONAFOR 2009). In this program, the government of Mexico pays landowners to forego land clearing for agriculture. PES programs provide an incentive-based mechanism to lessen environmental degradation and protect hydrologic services. For a more detailed case study of payment for ecosystem services see Appendix A: Regulatory Options in Practice, Payment for Ecosystem Services in New York City.

An analogous program in the study area counties might have the public pay farmers to alter their farming practices to reduce nitrate leachate or alternatively, the farmers might pay the counties for the right to discharge nitrate. In the Salinas Valley and Tulare Lake Basin, groundwater is currently

Adminsitrative Record
Page 35652

designated for drinking water use. This designation status requires that water quality in the aquifer meet drinking water quality standards. If a nitrate limit is established, then nonpoint source polluters in the region can either choose to meet this limit or pay to provide the affected inhabitants with safe drinking water. If the cost of the latter option is less, then it will be beneficial for farmers to pay for the right to discharge nitrate. Under both scenarios, consumers receive safe drinking water; however, the cost paid by farmers differs. This reflects the famous Coase Theorem[39] in environmental economics.

3.10 De-designation of Beneficial Use

Currently, all water bodies in California are assigned designated or "beneficial uses." Beneficial uses include recreation; agricultural uses; support of water and estuary ecosystems; support of fish migration; industrial uses; municipal and domestic supply; and others (California Water Code § 13050). In most cases, the default designation for groundwater is domestic and municipal water use (under State Water Board Resolution 88-63), though it can be designated for agricultural, industrial, municipal, or domestic use. Currently, the beneficial use designation for groundwater quality in the Salinas Valley and Tulare Lake Basin counties includes drinking water. As a result, the water quality in these groundwater basins must be suitable for drinking water.

Under Porter-Cologne, the Regional Water Boards are required to define the beneficial use for each water body. Water quality objectives must also conform to "anti-degradation" under the CWA section 131.12 (surface water only) and the 1968 State Water Board Resolution 68-16 (*the Statement of Policy with Respect to Maintaining High Quality of Waters in California*). Anti-degradation prohibits the degradation of water quality to levels above the water quality objectives needed to meet beneficial uses. Any actions that can degrade water quality must (1) be consistent with maximum benefit to the people of the State; (2) not unreasonably affect present and anticipated beneficial use of the water; and (3) not result in water quality less than that prescribed in water quality plans and policies. Designated uses and the anti-degradation policy are the basis for regulatory actions to enforce basin plans.

Porter-Cologne recognizes that water quality may be changed without unreasonably affecting the beneficial use. State Board Resolution 88-63 recognizes that in certain instances, exceptions should be made to the designation of groundwater as a drinking water source. The re-designation (or de-designation) of beneficial use refers to the regulatory process that alters the current designated use of groundwater. Currently, under State Water Board policy, where nitrate contamination is high and "cannot reasonably be treated for domestic use using either Best Management Practices (BMPs) or best economically achievable treatment practices" (State Water Board Resolution 88-63), one possible regulatory option is to de-designate portions of a groundwater basin as a source of municipal or domestic supply, which potentially removes certain discharge and groundwater remediation requirements that must otherwise be met. This would shift responsibility for treatment to drinking water users.

[39] Trading of externalities will be efficient if there are no transaction costs. The assignment of property rights is unimportant.

Adminsitrative Record
Page 35653

Since the groundwater anti-degradation policy and the default MUN designation are both State Water Board policies, they may be changed at the discretion of the State Water Board, without the need for new legislative authority, as long as the new resolution is consistent with Porter-Cologne.

3.11 Comparing Regulatory Instruments

We now compare regulatory instruments to determine the most promising ones to regulate nitrate contamination of groundwater. Table 4 below summarizes our qualitative comparison of all regulatory options to manage nitrate across four dimensions (abatement costs, monitoring and enforcement costs, information requirements, and ability to raise revenues). First, we compare direct regulatory instruments (technology mandates and performance standards) to market-based approaches (cap-and-trade and fees). We then discuss the feasibility of liability rules and de-designation as well as the role of information disclosure. Lastly, we compare fertilizer fees to a fertilizer cap-and-trade system, and ultimately recommend a fertilizer fee as the preferred tool to manage nitrate contamination of groundwater.

3.11.1 Comparing Direct Regulatory Instruments to Market-Based Approaches

Technology mandates: Technology mandates are relatively easy to implement from an administrative standpoint, since the regulator simply needs to verify that a technology has been adopted and properly installed. However, technology mandates are disadvantageous when evaluated in terms of abatement costs. In theory, an equivalent reduction in nitrate could be achieved at a lower cost. In practice, technology mandates are often an order of magnitude more costly than market-based approaches (Keohane & Olmstead 2007). Additionally, Porter-Cologne explicitly prohibits technology standards for controlling waste discharge, so existing legislation could not be used to enforce a technology mandate.

Technology mandate vs. performance standard: Performance standards offer farms and other entities more flexibility than technology mandates. For example, a farmer can choose to meet the performance standard by changing the crops grown on a farm, reducing fertilizer use, investing in irrigation efficiency capital, or combinations of these (see Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality (Dzurella et al. 2012) for other examples). The abatement costs of compliance under a performance standard tend to be less than those under technology mandates, since farms may choose how to meet the standard. However, performance standards are more costly in terms of abatement costs than market-based approaches, since each farmer separately determines how to comply with the standard.

Performance standard vs. market regulation: Compared to a performance standard on fertilizer use, market-based instruments (a fertilizer fee or a fertilizer cap-and-trade system) are more cost effective; that is, the abatement costs to achieve a fertilizer target are lower. Under a performance standard, each farmer is required to reduce fertilizer use by the same quantity, though the marginal costs to reduce fertilizer use may differ across farms. In contrast, a fertilizer cap-and-trade system caps the total amount of fertilizer use, but lets the market determine the allocation of fertilizer use among regulated

Adminsitrative Record
Page 35654

farmers. A fertilizer fee places a price on fertilizer and allows farmers to choose how much to fertilizer to apply.

To illustrate the advantage of market-based instruments such as a cap-and-trade, imagine that there are two farms in the Tulare Lake Basin and in the absence of fertilizer regulations each farm applies ten tons of fertilizer. The regulator enters, caps the total amount of fertilizer at ten tons, and allocates five tons of fertilizer to each farm. These farms differ in the marginal costs to reduce fertilizer use, perhaps because they grow different crops, use different amounts of labor, or use different irrigation technologies. It costs farm A $100 to reduce the fifth ton of fertilizer and it costs farm B $50 to reduce the fifth ton of fertilizer use. Farm B would be willing to sell a fertilizer permit for any amount above $50 and farm A would be willing to purchase a fertilizer permit if the price was less than $100. Due to variation in the marginal costs of abatement, it is mutually advantageous to trade fertilizer permits. Trading among farmers is beneficial to both the seller and buyer of fertilizer permits and reduces the costs of meeting a fertilizer cap. With a cap-and-trade system, growers will trade until the marginal costs of fertilizer abatement are equivalent across all farms. Under a performance standard, each farm would be required to reduce fertilizer by five tons, though both would benefit if they were free to trade fertilizer permits.

Table 4. Summary of regulatory instruments to manage nitrate contamination to groundwater.

Regulatory Options	Abatement Costs	Monitoring and Enforcement Costs	Information Requirements	Revenue Raising
Technology Mandate	High	Fertilizer Application: Low Nitrate Leachate: High		No (unless fines)
Performance Standard	Medium			No (unless fines)
Fee	Low			Yes
Cap-and-Trade	Low			Yes (if permits auctioned)
Information Disclosure	Medium	Low	Low	No (unless fines)
Liability Rules	N/A	High	High	Yes
Payment for Water Quality	Low	Low (if payment made to farmers) High (if payment made to State)	High	Yes (if payment made to State)
De-designation of Beneficial Use	N/A	High	Medium	No

3.11.2 Feasibility of Liability Rules and De-designation

As illustrated in the case study on liability rules on pesticide use (see Appendix A: Regulatory Options in Practice, Liability from Groundwater Contamination from Pesticide Use), in the long-run, liability rules are a potentially viable regulation to manage nitrate contamination. However, a handful of concerns

Adminsitrative Record
Page 35655

must be addressed before liability rules pose a credible threat to nitrate polluters. First, in theory, liability rules hold polluters responsible for clean-up costs. However, in practice, liability rules are at times not enforced and when they are, the public sector often shares remediation costs with the polluter, thereby lessening their "bite". A second concern with liability rules is how the regulator defines cleanup costs. The regulator may choose environmental remedies that do not restore polluted drinking water sources to their original state. Another disadvantage of liability rules is that the costs involved in establishing liability are often excessive. This is likely to be further exaggerated with nitrate in groundwater because of the complexities of groundwater flow and contaminant transport processes. Phrased differently, it is difficult to trace groundwater contamination of a drinking water source to a particular non-point source.

Legislation exists to make liability rules a promising management tool in the future. Porter-Cologne Section 13304 provides authority for the Regional Water Boards to require landowners contributing nitrate to groundwater drinking water supplies to support drinking water actions for affected public water supplies and private wells.

The de-designation of a portion of or a whole groundwater basin is a lengthy and costly process because of the expected political and local push-back and the necessary steps required for both State Water Board and Office of Administrative Law approval. The State requires that a basin plan amendment is required to de-designate the basin as a source of municipal and domestic water supply (MUN) (Central Valley Regional Water Board 2011), so an alternative drinking water plan will need to already be in place at the time of de-designation. Further, there are explicit restrictions on de-designation. Specifically, State Water Board regulations (Resolution No. 88-63) only allows MUN de-designation of groundwater with high TDS, insufficient sustained water yield, or contamination that cannot be reasonably treated (State Water Board 2006). Historic examples of de-designation in California have typically been a result of natural water quality concerns. Allowing de-designation of areas degraded by human activity may encourage continued and extensive degradation in other areas because this would remove the liability of dischargers. Currently, the administrative costs required for de-designation are excessive, making other regulatory options preferable to de-designation.

3.11.3 Role of Information Disclosure

In the case of drinking water quality, information disclosure in the form of consumer confidence reports may provide little incentive for small systems to reduce nitrate concentrations. Large systems face costs in the form of customer complaints and CDPH oversight from disclosure of safe drinking water quality violations. The costs of customer complaints increase with the population served simply because there is a larger population to issue complaints. When a water supplier serves a small (2 to 4 connections) and comparatively disadvantaged population, the costs of disclosing a violation may be much lower, thereby limiting (at least in the short-term) the potential of this regulatory option to improve drinking water quality.

Adminsitrative Record
Page 35656

3.11.4 Choosing Among Market-Based Approaches – Fees v. Cap-and-Trade

Weighted along three of our analytical criteria – abatement costs, administrative and enforcement costs, and information requirements – market-based instruments to manage fertilizer are preferred. Market-based approaches have lower administrative costs and information requirements than liability rules or de-designation, and have lower abatement costs than direct regulatory instruments. When choosing between a fertilizer fee and a fertilizer cap-and-trade system, in the near-term, a fertilizer fee has distinct advantages.

First, a fertilizer fee will generate revenues. These revenues could be used for the construction of improved drinking water infrastructure or further treatment of existing sources. Alternatively, fee revenues could be returned to emitters as a refund, used to subsidize technologies to improve management practices, or for research and development of improved nutrient management practices and extension/training programs.

A cap-and-trade system may or may not generate revenues, depending on the allocation of permits. If fertilizer permits are auctioned off, the government will collect revenues from the sale of fertilizer permits. In theory, the revenues collected from the auctioning of fertilizer allowances will equal those collected from a fee. If fertilizer allowances are freely allocated to non-point sources, then non-point sources gain the revenues that the government would collect under a fertilizer fee. Which sources gain the revenues depends on the initial allocation of permits. Though the initial distribution of permits will alter which farms gain revenues, it does not change the overall cost of abatement.

Another reason to prefer a fertilizer fee in the short-run is because the administrative infrastructure already exists to implement this regulation. Since fertilizers are currently exempt from sales taxes in California (if applied to land used for food or for feed for food animals) (CA State Board of Equalization 2004), the State could set a fee on fertilizer at a sales tax rate of 7.5%. The State would not need to develop new infrastructure to implement a fertilizer fee. It could simply extend the sales tax in California to include a fertilizer fee. While a tax would go into the State's general fund, a fee could have a designated purpose such as safe drinking water to areas facing high nitrate concentrations in the groundwater. In contrast, the state would need to develop new administrative infrastructure to implement a cap and trade system for groundwater.

3.12 Major Findings: Future Regulatory Options

1. **Many options exist to regulate nitrate in groundwater, but there is no ideal solution.** The costs of regulatory options vary greatly, and while no option is perfect, some seem preferable to others.

2. **Regulating fertilizer application has lower monitoring and enforcement costs and information requirements than does regulating nitrate leachate, but it may be less effective in achieving nitrate reduction targets.** While the regulation of fertilizer application is easier to implement and enforce than the regulation of nitrate leachate, it is less likely that water quality standards will be

met. Due to non-uniform mixing, transport, and dispersion of nitrate in groundwater, it is difficult to quantify the impact of a unit of fertilizer on nitrate contamination of drinking water over time.

3. **Costs to farmers for reducing nitrate contamination can be lower with market-based regulatory actions (fertilizer fees or cap-and-trade programs) than with technology mandates or prescriptive standards because of the additional flexibility farmers have in complying with market-based regulations.** Market-based instruments also encourage the development and adoption of new technologies to reduce fertilizer use, but they may lead to the formation of contamination hot spots.

4. **Well-defined and enforceable regulatory requirements are needed for liability rules to work.** In California, all groundwater is considered to be suitable, or potentially suitable, for municipal or domestic water supply and should be so designated by the Regional Water Quality Control Boards (Regional Water Boards), with a few exceptions.[40] Under existing water code, groundwater is protected from degradation affecting its designated beneficial uses, but this protection is rarely enforced in the case of nonpoint source pollution. Section 13304 of the California Water Code (Porter-Cologne Act) gives the Regional Water Boards authority to force polluters to pay for alternative water supplies for affected users of public water systems and private wells. Legislation might be useful to solidify Regional Water Board authority to apply this provision broadly.

3.13 Promising Actions: Future Regulatory Options

1. **Nitrate dischargers incur the social costs of their discharges. This policy is already implied for drinking water costs under Section 13304 of the California Water Code (Porter-Cologne Water Quality Control Act).** Without regulations on nitrate source loading, nonpoint sources do not pay for the impact of their nitrate contamination to groundwater. Requiring nitrate dischargers to pay for nitrate contamination provides incentives for them to reduce nitrate discharges and ensures that those benefitting from nitrate discharge pay for the cost of contamination.

2. **Regulatory actions focus on nitrogen fertilizer use (including organic fertilizer sources).** Most nitrate discharge to groundwater is from nonpoint sources, particularly cropland, with substantial additional loading from manure.[41] Existing regulatory programs appear to address point sources of nitrate contamination well, but could be extended to include nonpoint sources.

3. **Regulatory actions should focus on controlling fertilizer application rather than nitrate leachate.** The high costs and technical difficulties of field, farm-based, or countywide nitrate leachate regulations could be prohibitive. A fertilizer application regulation that reduces nitrate discharges to groundwater is coarser and less direct, but much less costly.

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[40] Waters with very high TDS, for example. See Section 4.9 De-designation of Beneficial Use and also, State Water Board Resolution No. 88-63

[41] See Technical Report 2: Nitrogen Sources and Loading to Groundwater (Viers et al. 2012)

Adminsitrative Record
Page 35658

4. **Market-based instruments are likely to perform better than command-and-control approaches.** Market-based approaches have lower costs than command-and-control approaches because they provide dischargers with more flexibility to comply with nitrate regulations.

5. **A fertilizer fee is a promising form of regulation and funding.** In the short run, a fee on fertilizer use (including organic fertilizer sources) seems preferable to a cap-and-trade system since it could take advantage of existing infrastructure. Since fertilizers are currently exempt from sales taxes in California (CA State Board of Equalization 2004), the State could expand sales taxes to include fertilizer purchases. The impact a fertilizer fee has on nitrate contamination depends on a) the sensitivity of demand for fertilizer to a price change and b) the size of the fee. The long-term feasibility and desirability of a cap-and-trade system for nitrogen management could be evaluated.

Adminsitrative Record
Page 35659

4 Current Funding Programs

In addition to the planning and regulatory programs, several State, Federal, and local agencies, as well as nongovernmental organizations, have established funding programs related to nitrate contamination in California's groundwater. This section summarizes existing funding sources available from these agencies to reduce nitrate source loading to groundwater, remediate contaminated groundwater, and provide safe drinking water to affected communities.

4.1 Information on Current Sources of Funding

State funding for safe drinking water is currently dominated by general obligation bonds for loans through State propositions, Federal economic stimulus package grants, and State revolving fund loans.

The most recent propositions that provided loans or grants for drinking water infrastructure or water quality protection/improvement are Propositions 82 [1988], 13 [2000], 50 [2002], and 84 [2006]. All are State general obligation bonds, which are repaid through the general fund. A general obligation bond is a municipal bond secured by the use of State or local government resources to repay bond holders. Often, general obligation bonds levy a property tax or decrease local property tax revenues to meet debt service requirements.

The American Recovery and Reinvestment Act (ARRA) of 2009 created an economic stimulus package that allowed for an increase in the Federal budget deficit. This 2009 stimulus package allocated $160 million in funding to the Federal Safe Drinking Water State Revolving Fund.

State revolving funds also have been a major source of funding for projects that support safe drinking water goals. While water systems still bear most project costs, the State revolving funds subsidize a portion of the costs with low-interest loans. These programs work like environmental infrastructure banks (Figure 1) where the funding program is a self-perpetuating loan assistance authority for water quality improvement projects. It is capitalized by Federal and State contributions and the pot of money is able to continually grow through: investment and interest earnings; principal repayments; and bond proceeds from leveraging. Revenues are recycled back into the program and since grants are not allowed, the funds do not dissipate. More recently, some funding programs, like the State Revolving Fund and the Integrated Regional Water Management Programs, have targeted small and disadvantaged communities by setting aside funds specifically for these populations.

Adminsitrative Record
Page 35660

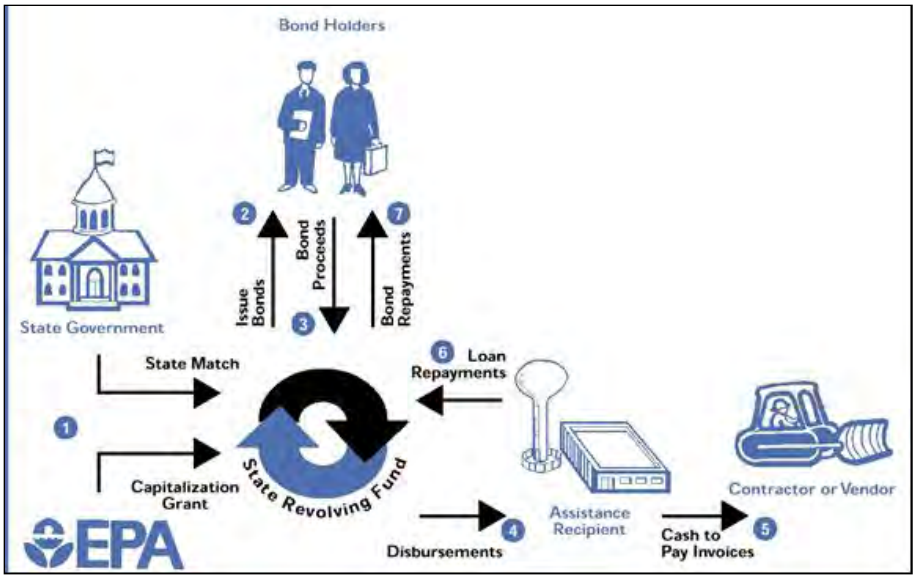

Figure 1. Structure of the State Revolving Fund. (U.S. EPA CWSRF Branch 2006, reproduced with permission)

In addition to these major sources of funding for safe drinking water, some fees exist to help prevent groundwater contamination from nitrate sources. One example is CDFA's mill assessment on fertilizer, which provides funding for research on the use and handling of fertilizer, including environmental effects. Nongovernmental and non-profit funding also exists for drinking water, water quality, and water supply improvements. Typically, these pools of money are significantly smaller and more limited than State and Federal resources.

4.2 Summary of Current Funding Programs

A summary of existing funding sources to address problems related to nitrate in drinking water is shown in Table 5. In general, these programs are structured to provide assistance for activities related to alternative water supplies and nitrate load reduction. California has eighteen relevant State funding programs, administered by four agencies (Table 5, purple). The Federal government manages an additional three funding programs (blue). Three large nongovernmental drinking water funding programs in the study area are highlighted in orange. For a more detailed review of these programs, see Appendix C: Description of Current Funding Programs for Safe Drinking Water. Several State funding efforts for safe drinking water infrastructure are also reviewed by the California Financing Coordinating Committee.[42]

[42] http://www.cfcc.ca.gov/

Adminsitrative Record
Page 35661

Table 5. Summary of existing funding sources for safe drinking water.

AGENCY	PROGRAM [year passed or created]	FUNDING PROVIDED (in millions of dollars)
California Department of Public Health (CDPH)	Safe Drinking Water State Revolving Fund [SDWSRF] [1996] (grants and loans)	**Generally $100-$150:** Low interest loans and some grants to support water systems with technical, managerial, and financial development and infrastructure improvements.
	Proposition 84 [2006] (grants) **(fully allocated)**	**$180:** Small community improvements. **$60:** Protection and reduction of contamination of groundwater sources. **$10:** Emergency and urgent projects.
	Proposition 50 [2002] (grants) **(fully allocated)**	**$50:** Water security for drinking water systems. **$69:** Community treatment facilities and monitoring programs. **$105:** Matching funds for Federal grants for public water system infrastructure improvements.
State Water Resources Control Board (State Water Board)	Clean Water State Revolving Fund (CWSRF) [1987] (loans)	**$200 - $300 per year:** Water quality protection projects, wastewater treatment, nonpoint source contamination control, and watershed management.
	Small Community Wastewater Grants [2004, amended 2007] (grants)	**$86 (fees on the CWSRF):** Loan forgiveness to small disadvantaged communities and grants to non-profits which provide technical assistance and training to these communities in wastewater management and preparation of project applications.
	Proposition 50 [2002] (grants) **(fully allocated)**	**$100:** Drinking water source protection, water contamination prevention, and water quality blending and exchange projects.
	Agricultural Drainage Program [1986] (loans) **(fully allocated)**	**$30:** Addressing treatment, storage, conveyance or disposal of agricultural drainage.
	Dairy Water Quality Grant Program [2005] (grants) **(fully allocated)**	**$5 (Prop 50):** Regional and on-farm dairy projects to address dairy water quality impacts.
	Nonpoint Source Implementation Program [2005] (grants)	**$5.5 per year:** Projects that reduce or prevent nonpoint source contamination to ground and surface waters.
	Cleanup and Abatement Account [2009]	**$9 in 2010:** Clean up or abate a condition of contamination affecting water quality.
	Integrated Regional Water Management (IRWM) [2002] (grants) **(fully allocated)**	**$380 (Prop 50):** Planning ($15) and implementation ($365) projects related to protecting and improving water quality, and other projects to ensure sustainable water use.

Adminsitrative Record
Page 35662

California Department of Water Resources (DWR)	Integrated Regional Water Management (IRWM) [2002] (grants)	**$500 remaining (Prop 84):** Regional water planning and implementation.
	Local Groundwater Assistance Grant [2008] (grants)	**$4.7 anticipated for 2011-2012 (Prop 84):** Groundwater studies, monitoring and management activities.
	Proposition 82 [1988] (loans)	**$22:** New local water supply feasibility & construction loans.
	Water Use Efficiency Grant Program [2001] (grants)	**$15 in 2011 (Prop 50):** Water use efficiency projects for agriculture, such as: wellhead rehabilitation, water and wastewater treatment, conjunctive use, water storage tanks.
	Agricultural Water Conservation Loan Program [2003] (loans)	**$28 (Prop 13):** Agricultural water conservation projects, such as: lining ditches, tailwater or spill recovery systems, & water use measurement.
	Infrastructure Rehabilitation Construction Grants [2001] (grants) **(fully allocated)**	**$57 (Prop 13):** Drinking water infrastructure rehabilitation and construction projects in poor communities.
California Infrastructure and Economic Development Bank (I-Bank)	Infrastructure State Revolving Fund (ISRF) [1994] (loans)	**$0.25– $10 per project:** Construction or repair of publicly owned water supply, treatment and distribution systems.
U.S. Department of Agriculture (USDA)	Rural Utilities Service - Water and Environmental Programs (RUS WEPs) (loans and grants)	**$15.5 :** Development and rehabilitation of community public water systems (less than 10,000 people), including: emergency community water assistance grants, predevelopment planning grants, technical assistance, guaranteed loans, and a household well water program.
U.S. Department of Housing and Development (HUD)	Community Development Block Grant (CDBG) (grants)	**$500 in 2010 for CA:** Community development projects: feasibility studies, final plans and specs, site acquisition and construction, and grant administration.
U.S. Department of Commerce	Economic Development Administration (EDA) (grants)	**Grants up to 50% of project costs:** Supports economic development, planning, and technical assistance for public works projects.
Rural Community Assistance Corporation (RCAC)	Drinking Water Technical Assistance and Training Services Project (loans)	**$1.2 per year:** Administers funds from the US EPA Office of Groundwater & Drinking Water for infrastructure projects, including water.
The Housing Assistance Council (HAC)	Small Water/Wastewater Fund (loans)	**Up to $0.25 per project:** Loans for land acquisition, site development, and construction.
Cooperative Bank (CoBank)	Water and Wastewater Loan (loans)	**$1 per project:** Water and wastewater infrastructure, system improvements, water right purchases, and system acquisitions. **$0.05-$0.5 per project:** Construction costs.

Adminsitrative Record
Page 35663

A variety of funding programs exist for the development of alternative water supply actions. Funding is available for capital investments in new water supplies, safe drinking water treatment, aging infrastructure replacement, water use efficiency, and water meter installation. Funds also are available to educate communities and systems about water quality contamination and to support technical, managerial, and financial capacity building. A very small amount of money is available to help systems prepare funding applications and perform pre-investment planning.

The diversity of funding sources for safe drinking water makes it difficult for drinking water systems or communities to navigate the litany of agencies and programs. Each program has its own funding application to understand and complete. One promising action for the State could be to combine appropriate funding programs to ease demands on community applicants, lower administrative costs, and improve overall statewide funding effectiveness.

Most safe drinking water funding programs do not provide support for operation and maintenance costs; the State of California specifically does not fund operation and maintenance activities. Additionally, not all drinking water funding programs support the regionalization[43] of multiple water systems or the consolidation[44] of smaller districts. For example, while New Mexico's Rural Community Assistance Corporation (RCAC) office receives State funding for regionalization, California's RCAC office only receives State funding for training activities like operator certification. When funding is provided for regionalization and consolidation activities, money is restricted to construction activities such as the installation of a new pipeline or water meters. California's RCAC funding does not support institutional activities such as forming a joint powers authority, hiring a facilitator to organize pilot projects (brings together water systems and evaluate the feasibility of collaboration), or hiring a technical expert to help water systems perform asset mapping and financial planning.

Current funding programs have not met systems' stated need to ensure safe drinking water in the Salinas Valley and Tulare Lake Basin.[45] These areas have disproportionately high costs to deal with high nitrate concentrations. Compared to other areas of California, groups in the study area have requested more Safe Drinking Water Funds to address nitrate contamination ($29 per person compared to $5 per person statewide, see Figure 2). Providing safe drinking water or alternative water supplies to highly susceptible populations in the study area is estimated to cost at least $20-$36 million per year ($80-$142/year per susceptible person or $5-$9/year per acre of irrigated land).[45] Most current State funding for nitrate contamination problems is temporary (general obligation bonds for loans through State propositions and the Federal economic stimulus package grants) and many programs have already been fully allocated (see Table 5). Long-term funding for safe drinking water is needed.

[43] Regionalization: "a creation of an appropriate management or contractual administrative organization or a coordinated physical system plan of two or more community public water systems in a geographical area for the purpose of utilizing common resources and facilities to their optimum advantage" (Grigg 1989).
[44] Consolidation: "one community public water system being absorbed into, combined with, or served by other utilities to gain the resources they lack otherwise" (Raucher et al. 2004).
[45] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al. 2012)

Adminsitrative Record
Page 35664

Small water systems typically have higher per capita funding needs than larger systems. In small and rural communities, households are usually spread out; the infrastructure needed to transport water from the source to a household in a rural area is more expensive than for urban areas. This implies that unit costs of safe drinking water will be higher in rural or small communities. Small water systems also often lack the economies of scale to economically treat nitrate contaminated ground water, and the technical, managerial, and financial capacity to repay loans, complete funding applications, and pay both recurring and unexpected operation and maintenance costs. Further, many State funding programs (State Revolving Fund, State Bonds, etc.) only accept applications from water systems served by a public entity,[46] so domestic well owners and small communities with no recognized water system lack a major State funding source. This policy is inconsistent with the Environmental Justice principles in California's laws and policies, which are based on "the fair treatment of people of all races, cultures, and incomes with respect to the development, adoption, implementation, and enforcement of environmental laws and policies" (California Government Code section 65040.12).

In addition to funding programs for safe drinking water, programs are also available for nitrate source load reduction. There are programs to help convert communities from septic to sewer systems, install wastewater treatment upgrades, monitor groundwater quality, and protect receiving water quality in both surface water and groundwater. Planning activities are funded through programs like Integrated Regional Water Management. Funding for agricultural nutrient management education, training, and research also is available.

The California Department of Food and Agriculture (CDFA) oversees a mill assessment on fertilizer sales (see Appendix B: Summary of Current Planning and Regulatory Programs that Address Nitrate in Groundwater). Revenues are used for research and education of proper use and handling of fertilizing materials (including environmental effects) and to support the program. Currently, the assessment is only set at half of its authorized amount. Raising the assessment to the fully authorized amount would raise roughly $3 million per year statewide. $1 million of this could be used for the described fertilizer-use activities, or if current statute is changed, to fund some alternative drinking water supply efforts.

4.3 Current Funding Example: Safe Drinking Water State Revolving Fund[47]

The Safe Drinking Water State Revolving Fund (SDWSRF), created under the Federal SDWA, is one of the largest funding sources for community water systems in California, so it is presented here as an example program.

The SDWSRF allows CDPH to provide low interest loans and other assistance to public water systems. This fund supports: (1) infrastructure improvements; (2) water system technical, managerial, and

[46] A legally-approved public entity is a public water system or another legal entity that has authority to contract and incur debt on behalf of the community.
[47] (CDPH 2011)

Adminsitrative Record
Page 35665

financial capacity development; and (3) water and energy efficiency projects. As part of this program, CDPH establishes an annual project priority list based on applications from water systems. From this list, they create a "fundable list" of projects they intend to fund, assuming each system can meet all program requirements.

CDPH specifically addresses disadvantaged communities and small water systems in their project priority and fundable lists. Communities with lower median household incomes are given higher priority. Disadvantaged communities can also receive the following additional assistance from the SDWSRF: 1) zero percent interest rates (compared to 3-4%), 2) extended repayment periods of 30 years (compared to 20 years), and 3) forgiveness of up to 80% of the loan principal (CDPH, 2010). With the recent adoption of AB 983, severely disadvantaged communities may now be eligible for up to 100% grant funding (AB 983, 2011).

Derived from the 2010-2011 Final Project Priority List, Figure 2 shows that in dollars per person, more money was requested for drinking water projects in the study area counties relative to all of California. Despite the host of regulated contaminants, the 2010-2011 project priority list reflects the severity of nitrate contamination in the Tulare Lake Basin and Salinas Valley. Approximately 9% of funds sought (derived from the SDWSRF list) for all drinking water projects listed in the study area are related to nitrate contamination (install new treatment facilities, upgrade existing facilities, connect a pipeline to other drinking water systems, etc.). In comparison, only 1.6% of the statewide project costs are listed for nitrate projects. Figure 2 shows that $29 per person in the Tulare Lake Basin counties and $28 per person in Monterey County has been requested for nitrate projects, while the statewide requests are only $5 per Californian.

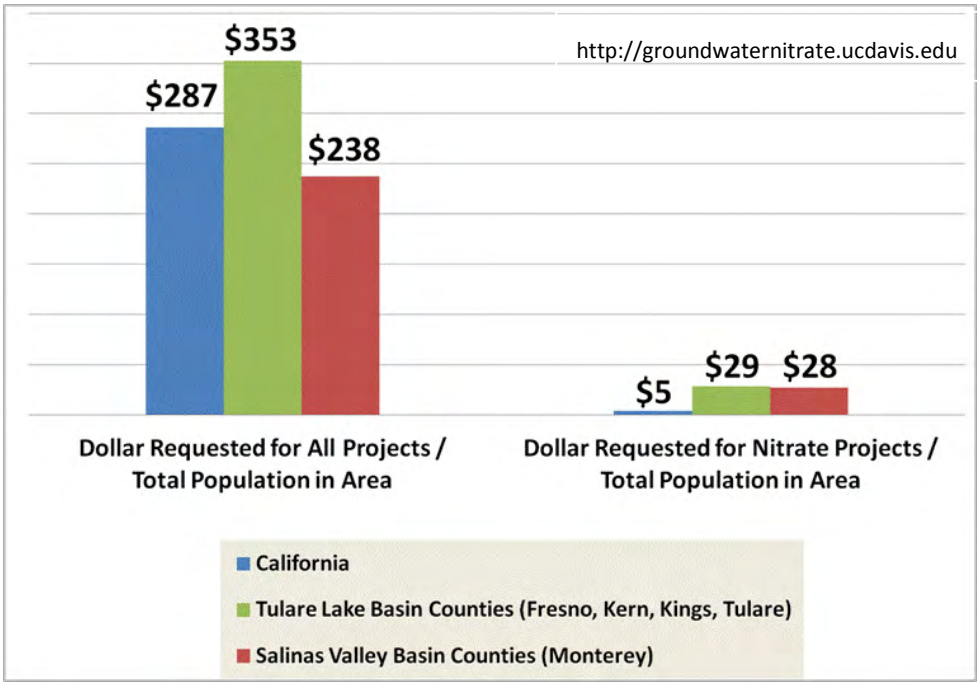

Figure 2. Dollar requested per person on proposals made to the 2010-2011 Safe Drinking Water State Revolving Fund (Final Fundable Project Priority List, Oct. 2010).

Adminsitrative Record
Page 35666

While this analysis only shows data from formal funding requests that have been approved, it is still useful for comparing the study area to the rest of the State. Looking solely at systems that are currently aware of their nitrate problem and who were able to apply to the SDWSRF, California will need at least $4 per person to fulfill the current statewide funding requests and around $27-28 per person in the five study area counties (Table 6, below). The total actual need for nitrate projects could be much higher when accounting for communities who were either unaware of their nitrate problem, unconcerned with the consequences, or unable to formally apply for funding.

Table 6. Unmet qualified project needs after allocation of 2010-2011 Safe Drinking Water State Revolving Fund.

	Total Area Population (Dept. of Finance 2010)	Type of Drinking Water Project	People with Unmet Requested Need (Millions)	Unmet Requested Need (Millions)	Dollar of Unmet Requested Need Per Total Area Population
All of California	38,648,090	All	41.94	10,859	281
		Nitrate	1.29	167	4
Tulare Lake Basin Counties: Fresno, Kern, Kings, Tulare	2,397,451	All	1.17	824	344
		Nitrate	0.12	66	28
Salinas Valley County: Monterey	435,878	All	0.40	103	236
		Nitrate	0.03	12	27

After the allocation of all available funds for the 2010-2011 SDWSRF, study area counties have approximately $78 million in unfulfilled funding requests for nitrate (Table 6). This can be compared to the alternative water supply cost analysis discussed in the Alternative Water Supply Options Technical Report 7 (Honeycutt et al. 2012), which estimated a present value cost of $212-$424 million ($17-34 million *per year*) to provide safe drinking water to the 220,000 "highly susceptible" people on public water systems in the study area. The alternative water supply cost estimate is significantly larger than the requested funding for SDWSRF. This is because the (Technical Report 7) estimate includes all water systems that have recorded delivered nitrate levels above the MCL, as well as those with no recorded nitrate levels. The SDWSRF project priority list only includes projects for which the community is both aware of their nitrate problems and has been able to navigate through the complicated and time-intensive funding application process.

4.4 Major Findings: Current Funding Programs

1. **Current funding programs have not met systems' stated need to ensure safe drinking water in the Salinas Valley and Tulare Lake Basin.**[48] These areas face large costs to deal with high nitrate

[48] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al., 2012)

Adminsitrative Record
Page 35667

concentrations. Compared to other areas of California, groups in the study area have requested more Safe Drinking Water State Revolving Funds to address nitrate contamination ($29 per person compared to $5 per person statewide, see Figure 2). Providing safe drinking water or alternative water supplies to highly susceptible populations in the study area is estimated to cost at least $20-$36 million per year ($80-$142/year per susceptible person or $5-$9/year per acre of irrigated land).[48]

2. **Most current State funding for nitrate contamination of drinking water is temporary.** State funding for safe drinking water is currently dominated by general obligation bonds for loans through State propositions, Federal economic stimulus package grants, and State revolving fund loans. All are temporary sources except for the State revolving fund loans, which are self-financed long-term sources.

3. **Small water system costs are high[49] and these small systems already face chronic financial problems.** In small and rural communities, housing is often low density and spread across a relatively large area; the infrastructure needed to transport water from the source to a household in a rural area is more expensive than for urban areas. Small systems also often lack the economies of scale and the technical, managerial, and financial capacity to repay loans, complete funding applications, and pay both recurring and unexpected operation and maintenance costs. This implies that unit costs of safe drinking water and responding to nitrate contamination will be higher in rural and small communities.

4. **While regionalization of drinking water systems is sometimes promising, little funding is provided for facilitating this solution**. California often provides funding to drinking water systems for the physical consolidation of systems, but it provides no support or training for non-construction regionalization activities (e.g., a facilitator brings systems together to assess the feasibility of consolidation, or a technical expert performs asset mapping and financial planning for a group of water systems).

4.5 Promising Actions: Current Funding Programs

1. **Where appropriate, combine funding programs.** Consolidating funding programs should lower administrative and application costs and improve program effectiveness. A single program will also ease demands on community applicants. For example, water supply and wastewater problems are often intertwined, and linking these sources of funding would reduce upgrade costs for small systems.

2. **Allocate funding to long-term drinking water solutions, particularly regionalization or consolidation of small systems.** Small drinking water systems face many challenges, including

[49] EPA. 2011. National Characteristics of Drinking Water Systems Serving 10,000 or Fewer People. (U.S. EPA 2011b)

Adminsitrative Record
Page 35668

nitrate-contaminated groundwater. Connecting to larger systems or consolidating with other small systems can allow such problems to be addressed more effectively and at lower cost. But regionalization and consolidation of systems often requires costly upgrades for smaller systems (e.g., water meters or new pipeline) or requires the larger system to take on an undesirable amount of risk. The State and counties have an interest in encouraging regionalization and consolidation activities to avoid longer-term financial difficulties, water system service inadequacies, and public health problems.

3. **Provide more financial assistance to small systems.** Small systems generally have more difficulty applying for funding programs and repaying loans. A grant or other special assistance program could be established to provide additional help to small systems. Such a program might be part of a larger consolidated effort by CDPH, the State Water Board, DWR, an Integrated Regional Water Management Plan, and the counties to address the problems of small water and wastewater systems.

4. **Create State funding programs for domestic well owners and for State small and local small water systems.** Currently, only water systems served by a public entity can apply for State safe drinking water funds (such as the State Revolving Fund). Domestic well owners and small communities with no recognized water systems thereby lack the funding sources available to public water systems. Assistance may also be provided to these small systems to help them form legally recognized entities.

5. **Increase the current mill assessment rate on nitrogen fertilizer to its fully authorized amount.** CDFA already oversees a mill assessment on fertilizer sales for research and education regarding the use and handling of fertilizing materials (including environmental effects), which is currently only half of its authorized amount. Raising the assessment to the fully authorized amount would raise roughly $3 million per year statewide. $1 million of this could be used for research and education regarding the use and handling of fertilizing materials (including environmental effects), or if current statute is changed, to fund some alternative drinking water supply efforts.

Adminsitrative Record
Page 35669

5 Future Funding Options

Current funding programs are insufficient to address the long-term issue of nitrate contamination. In this section, we outline an array of potential future funding options to support groundwater remediation, provide safe drinking water to affected communities, and reduce nitrate loading to the groundwater. It will take years to decades for an implemented nitrate regulatory program to affect drinking water quality, so immediate funding for safe drinking water is necessary.

In Section 5.3, we describe the funding options and then compare these sources in Section 5.4. We first compare non-agricultural fees to agricultural fees and recommend that funding sources should focus on agricultural fees. Next, we discuss various agricultural fees, concluding that a fee on nitrate or fertilizer is the preferred funding source. Finally, we recommend a fertilizer fee as the most promising funding option.

5.1 Overview and Summary Table

As discussed in Section 4, there are a number of existing funding programs such as the State revolving loans and grants that support safe drinking water. These programs, however, tend to be temporary in duration and have been ineffective in meeting the needs of at-risk populations, especially those on small water systems. Even if these programs are expanded, more revenue will be needed to support continued safe drinking water in the long-term. There are a number of other funding options to consider.

Table 7 provides a list of potential funding options, ranks the ability of each option to reduce nitrate contamination, lists who bears the burden of payment, and cites a relevant example.

Adminsitrative Record
Page 35670

Table 7. Summary of future State funding options.

OPTION	INCENTIVE TO REDUCE NITRATE	WHO PAYS?	EXAMPLE
Crop Tax	No	Producers and consumers of food	State Sales Tax Rate for Soft Drinks: The State of Maryland charges a 6% sales tax for soft drinks
Fixed Fee on drinking water agricultural water	 No No	 Drinking water users Agricultural users	Federal Communications Commission Universal Service Fee: A fixed fee placed on monthly phone bill to assure universal access to telecommunications for low-income and high-cost rural populations.
Volumetric Fee on drinking water agricultural water	 No Low	 Drinking water users Agricultural users	Gas Public Purpose Program Surcharge: A volumetric fee on gas bills in California to fund assistance programs for low-income gas customers, energy efficiency programs, and public-interest research.
Groundwater Pumping Fee	Medium	Agricultural groundwater users	Pajaro Valley Groundwater Pumping Fee: A per acre-foot charge to secure financing for debt stabilization and to address groundwater overdraft.
Fee/Tax on Bottled Water	No	Consumers of bottled water	California Redemption Value: A refundable fee placed on recyclable bottles at the point of sale.
Agricultural Property Tax	No	Agricultural property owners	CA State Property Tax: A statewide ad valorem tax equal to a percentage of the purchase price is collected from all properties in the State, with some exceptions.
Fertilizer Tax/Fee	High	Consumers of fertilizer	Mill Assessment Program: The State imposes a fee of 2.1 cents per dollar on pesticide sales at the point of first sale into the State.
Nitrate Leachate Tax/Fee	Highest	Nitrate emitters	Duty on Wastewater: In the Netherlands, a tax of approximately $3.60 is imposed on each kg of nitrate in wastewater.
Cap-and-Trade with Auctioned Permits	High/ Highest	Consumers of fertilizer and nitrate emitters	Title IV of the Clean Air Act Amendments: Established a tradable permit approach to control sulfur dioxide emissions. A small portion of permits sold in an auction.

Adminsitrative Record
Page 35671

5.2 Assumptions and Limitations

This analysis presents a wide range of potential funding options. We do not discuss how these revenues should be distributed. Revenues from these options could be used to remediate groundwater, treat groundwater, alter farming practices, or fund safe drinking water (or fund programs unrelated to nitrate in groundwater). Regulatory agencies will have the discretion to best choose how to employ these funds.

We evaluate funding sources along two criteria: 1) the potential of funding sources to reduce nitrate use among non-point agricultural sources and 2) which individuals will bear the burden of payment. Where possible, we also provide a back-of-the-envelope estimate of the per capita cost of each funding source to fund water supply actions in the study area. Future work may compare funding mechanisms along additional criteria; however, such a comparison is beyond the scope of this study.

5.3 Funding Sources

5.3.1 Water Fees

The State could employ a range of water fees to raise funds including a fixed monthly fee on drinking water, a volumetric fee on drinking water, a volumetric fee on irrigated water, a fixed fee on agricultural water, a groundwater pumping fee, or a fee on bottled water. The population base for generating the revenues for these funding sources could also vary, and fee increases could apply to the entire State or only to the Salinas Valley and Tulare Lake Basin.

Fixed fee on water use: An increase in the fixed fee on water use is one funding source option. Under this funding option, a fixed fee on drinking, agricultural, or irrigated water (depending on the funding option employed) would be charged to each consumer regardless of the quantity of water consumed. As the population charged this fixed fee increases, the fee incurred by each individual could be decreased.

Volumetric fee on water use and groundwater pumping fee: With a volumetric fee, the State or County would charge a fee, in addition to the current price, for each unit of water used. For example, California relies on the Gas Public Purpose Program Surcharge to fund gas assistance programs for low-income customers (PG&E 2009). As of January 1, 2011 Pacific Gas and Electric (PG&E) applied a surcharge of $0.08 per therm of natural gas used (PG&E 2011a), or on average, $3.60 per year per household.[50] A volumetric fee on statewide gross urban water use[51] would need to be $2.3-$4.1 per acre-foot per year to provide safe drinking water to the highly susceptible population of the study area.[52] If a fee is placed on both statewide gross urban and agricultural water use,[53] the fee would need to be at least $0.5-$0.9

[50] Average yearly household natural gas usage is 45 therms. (PG&E 2011b)
[51] Statewide gross urban water use is estimated at 8.7 MAF per year. (Hanak et al. 2011)
[52] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al. 2012)
[53] Statewide gross urban and agricultural water use is estimated at 41.7 MAF per year. (Hanak et al. 2011)

Adminsitrative Record
Page 35672

per acre-foot per year to provide safe drinking water to the estimated highly susceptible population of the study area.[54] One acre-foot is approximately the amount of water consumed by one family over two years. Alternatively, this volumetric fee could be limited to groundwater users and could take the form of a volumetric groundwater pumping tax.

Fixed fee on bottled water: A fixed fee or tax could be added to bottled water sales. One can imagine a fee on bottled drinking water that resembles the bottle tax added to the purchase of recyclable bottles. This fee could be collected to fund actions related to the management of groundwater nitrate contamination.

5.3.2 Nitrate Contamination-Related and Agricultural Fees

Revenues can also be generated by imposing fees on a range of nitrate contamination-related or agricultural goods. We discuss the potential of a crop tax, nitrate fees, fertilizer fees, and a cap-and-trade system. In addition to these funding options for managing nitrate contamination, other potential instruments include an agricultural property tax, a fee on septic tank discharge, or a wastewater fee.

Crop tax: A crop tax (which could be extended to include dairies and poultry) could be implemented as an ad valorem tax on the value of the crop, a unit tax on crop weight, or a unit tax based on the nitrate volume (or concentration) used to grow a crop. If a crop tax were imposed throughout the State, the tax would be very small. If consumers and producers do not alter behavior in response to a small crop tax, then the tax would need to equal $4 per thousand dollars of crop sold in the study area to fund the provision of alternative water supplies (assuming that it costs $36 million annually to provide safe drinking water to the 254,000 highly susceptible population,[54] and $9 billion dollars of crops are sold per year (USDA 2007).

Fertilizer or nitrate leachate fee: A fee on fertilizer or nitrate leachate reduces nitrate contamination to groundwater and raises revenue. Currently, there is no sales tax on fertilizer in California (CA Board of Equalization 2009). As discussed in Technical Report 3 (Dzurella et al. 2012), a fee on fertilizer that is approximately equal to the sales tax rate might raise $28 million in revenue annually and reduce nitrogen application by 1.6 percent in the study area.[55]

Cap-and-trade with auctioned permits: The auctioning of fertilizer application permits (the right to apply one unit of fertilizer) or nitrate leachate permits (the right to leach one unit of nitrate to groundwater) under a cap-and-trade system could generate funds. Under certain assumptions, the revenue from the auctioning of permits will equal the amount generated under a fee or tax.

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[54] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al. 2012)

[55] Assume: 7.5% sales tax on the cost of nitrogen, a fertilizer retail price of $0.75 per pound of nitrogen, and 500 million pounds (227 GgN) of fertilizer used in the Salinas Valley and Tulare Lake Basin (see Section 3 of Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality, Dzurella et al. 2012)

Adminsitrative Record
Page 35673

5.4 Comparing Funding Sources

We now compare funding sources to determine the most promising options to raise revenues for safe drinking water supply actions, nitrate source loading reduction, or groundwater remediation. First we compare **agricultural fees** (crop taxes, fixed/volumetric agricultural water fees, groundwater pumping fees, agricultural property taxes/fees, fertilizer taxes/fees, nitrate leachate taxes/fees, and auctioned fertilizer/nitrate permits) to **non-agricultural fees** (fixed/volumetric fees on drinking water and fees on bottled water), and recommend agricultural fees. We then discuss the various agricultural fees and ultimately recommend a fertilizer fee as the preferred funding source.

5.4.1 Agricultural vs. Non-Agricultural Fees

The advantage of an agricultural fee over a non-agricultural fee is that it requires nitrate emitters to internalize the social costs of their behavior. Nitrates leached from agricultural and dairy sources impose a real cost on society in the form of contaminated drinking water. However in the absence of regulation, the producers of nitrate do not bear the costs of their behavior. Rather, residents in the areas with contaminated drinking water incur these costs either by drinking contaminated water, paying for water treatment or purchasing alternative water supplies. An agricultural fee, at least partly, requires nitrate polluters to incur some of the social costs of their behavior. Different agricultural fees vary in the extent to which they require polluters to internalize the social costs of their behavior.

There are advantages to non-agricultural fees. First, as the population incurring the fee increases, as would be the case under a statewide drinking water fee or food tax, the cost borne by each individual declines. Under a statewide drinking water fee, the rate increase would be small, amounting to $2.3-$4.1 per acre, and negligibly impact individual income. Second, since the rate increase is small, there may be little resistance to a statewide fee; however, introducing drinking water fees to fund drinking water actions in the Salinas Valley and Tulare Lake Basin may establish a precedent that quickly leads to the use of drinking water fees to fund many other drinking water-related programs.

Since agricultural fees require nitrate emitters to partly pay for the costs of their behavior, we recommend agricultural fees over non-agricultural fees.

5.4.2 Choosing Among Agricultural Fees

The drawbacks of a fixed fee: Fixed agricultural fees describe funding sources that rely on fixed fees to generate revenues. Under a fixed fee, all nitrate emitters will pay the same fee regardless of their behavior. There are two main disadvantages to a fixed fee. First, nitrate emitters have no incentive to alter their nitrate emissions, since they pay the same fee irrespective of nitrate emissions. Second, the fees borne by non-point sources are not proportional to the nitrates leached; rather, they are uniform across non-point sources. In contrast, volumetric fees provide some incentive for nitrate emitters to alter their behavior, and for this reason are preferred to fixed fees.

Adminsitrative Record
Page 35674

The drawbacks of a crop tax: A small crop tax amounting to $4 per thousand dollars of crop sold in the study area could fund the provision of water supply actions. However, we prefer other agricultural fees to a crop tax for three reasons. First, a crop tax provides a weak signal for non-point sources to reduce nitrate emissions since the tax is on the value of the crop rather than on nitrate emissions. If non-point sources altered behavior in response to a crop tax, they could do so by reducing water use, labor, fertilizer application, etc. A second drawback of a crop tax is that it is regressive, since low-income individuals spend a larger share of their income on food. Finally, a crop tax may also induce individuals to reduce the consumption of fruits and vegetables since these products become marginally more expensive with a food tax.

The drawbacks of water fees: A volumetric water fee describes an indirect fee on nitrate contamination of groundwater. Under this fee, non-point sources face a direct incentive to reduce water use rather than fertilizer use. While water and nitrate use are positively correlated, it is not the case that a 1-unit reduction in water use will cause a 1-unit reduction in nitrate use. A volumetric water fee places the burden of payment for water supply actions on water intensive farms rather than nitrate-intensive farms. In contrast, a nitrate or fertilizer fee places the burden of payment on nitrate emitters. These funding sources also provide a stronger incentive for nitrate emitters to reduce nitrate use.

5.4.3 The Advantage of a Fertilizer Fee

Compared to the other funding sources, nitrate fees, auctioned nitrate permits, fertilizer fees, and auctioned fertilizer permits are preferred since they require nitrate emitters to more wholly internalize the costs of their behavior. Among these funding sources, we prefer a fertilizer fee. We recommend a fertilizer funding option over a nitrate funding source, since the administrative costs and information requirements to implement a nitrate fee or permit are high (refer to Section 3.2 for a detailed discussion). We favor a fertilizer fee over auctioned permits since the administrative infrastructure exists to expand the sales tax to include fertilizer (refer to Section 3.11.4 for a detailed discussion). This fee could either be applied based on the percent nitrogen or with a unit cost per weight of fertilizer. While a fee that accounts for the concentration of nitrogen will better link the fee to groundwater nitrate contamination potential, a cost based fee is easier to implement.

5.5 Major Findings: Future Funding Options

1. **Options exist to raise funds for safe drinking water, but all require that someone bear the cost and many are awkward or insufficient.** Water use fees, groundwater pumping fees, bottled water fees, crop fees, and fertilizer fees are a few of the many potential sources for funding safe drinking water and source reduction actions.

2. **Some funding options give polluters a useful price signal.** Fertilizer (or nitrate leachate) fees and auctioned permits induce emitters to reduce use of fertilizer or nitrate.

Adminsitrative Record
Page 35675

3. **Farmers do not pay sales tax on fertilizer in California**[56].

5.6 Promising Actions: Future Funding Options

1. **Introduce a special fee on nitrogen fertilizer sales statewide, perhaps equivalent to a sales tax.** This economic signal could both reduce nitrogen applications and help fund safe drinking water solutions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs. Expanding sales tax to include fertilizer could generate $28 million per year in the study area and might reduce applied nitrogen by 1.6%.[57] Similar fees/excise taxes could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

2. **Consider a more comprehensive statewide fee on water use to support many beneficial activities**. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities. Assuming a fee is placed on statewide gross urban and agricultural water use,[58] the fee would need to be $0.5-$0.9 per acre-foot per year to provide safe drinking water to the estimated highly susceptible population of the study area.[59] Likewise, a fee only on statewide gross urban water use[60] would need to be $2.3-$4.1 per acre-foot per year.

[56] (CA State Board of Equalization 2004)
[57] Assume: 7.5% sales tax on the cost of nitrogen, a fertilizer retail price of $0.75 per pound of nitrogen, and 500 million pounds (227 GgN) of fertilizer used in the Salinas Valley and Tulare Lake Basin (Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality, Dzurella et al. 2012)
[58] Statewide gross urban and agricultural water use is estimated at 41.7 MAF per year. (Hanak et al. 2011)
[59] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al. 2012)
[60] Statewide gross urban water use is estimated at 8.7 MAF per year. (Hanak et al. 2011)

Adminsitrative Record
Page 35676

6 Summary of Promising Actions

6.1 Regulatory Programs and Options

1. **Provide immediate safe drinking water to groundwater nitrate affected areas.** Since nitrate source reduction or groundwater remediation will take years to decades to significantly improve drinking water quality,[61] residents currently receiving unsafe drinking water require other alternatives in the immediate future, regardless of source reduction and management.

2. **Identify populations at risk of contaminated drinking water.** California Department of Public Health (CDPH), California Department of Food and Agriculture (CDFA), and State Water Board, in coordination with Department of Water Resources (DWR), issue a report every 5 years to identify populations at risk of contaminated drinking water and monitor long-term trends of the State's success in providing safe drinking water as a supplement to the California Water Plan Update.

3. **Regional Water Boards designate areas where groundwater sources of drinking water are at risk of being contaminated by nitrate.**

4. **Convene a State Groundwater Data Task Force.** California Environmental Protection Agency (CalEPA), in coordination with California Natural Resources Agency (CalNRA) and CDPH, convene an independently led State Groundwater Data Task Force to examine the efficacy of current State and local efforts to collect, maintain, report, and use groundwater data for California's groundwater quality and quantity problems.

5. **Convene a State Groundwater Task Force.** CalEPA, CalNRA, and CDPH maintain a permanent and independently led State Groundwater Task Force to periodically assess State technical and regulatory groundwater programs in terms of effectiveness in addressing California's groundwater quality and quantity problems. These reports could be incorporated into each California Water Plan Update.

6. **Examine successful Department of Pesticde Regulation (DPR) programs for lessons to manage nitrogen.** CalEPA and CDFA examine successful Department of Pesticide Regulation (DPR) data collection, analysis, education, and enforcement programs for lessons to manage nitrogen and consider expanding or building upon the existing DPR program to include comprehensive nitrogen use reporting to support nitrate discharge management.

7. **Nitrate dischargers incur the social costs of their discharges. This policy is already implied for drinking water costs under Section 13304 of the California Water Code (Porter-Cologne Water Quality Control Act).** Without regulations on nitrate source loading, nonpoint sources do not pay for the impact of their nitrate contamination to groundwater. Requiring nitrate dischargers to pay for nitrate contamination provides incentives for them to reduce nitrate discharges and ensures that those benefitting from nitrate discharge pay for the cost of contamination.

[61] See Technical Report 4: Groundwater Nitrate Occurrence (Boyle et al. 2012)

Adminsitrative Record
Page 35677

8. **Regulatory actions focus on nitrogen fertilizer use (including organic fertilizer sources).** Most nitrate discharge to groundwater is from nonpoint sources, particularly cropland, with substantial additional loading from manure.[62] Existing regulatory programs appear to address point sources of nitrate contamination well, but could be extended to include nonpoint sources.

9. **Regulatory actions should focus on controlling fertilizer application rather than nitrate leachate.** The high costs and technical difficulties of field, farm-based, or countywide nitrate leachate regulations could be prohibitive. A fertilizer application regulation that reduces nitrate discharges to groundwater is coarser and less direct, but much less costly.

10. **Market-based instruments are likely to perform better than command-and-control approaches.** Market-based approaches have lower costs than command-and-control approaches because they provide dischargers with more flexibility to comply with nitrate regulations.

11. **A fertilizer fee is a promising form of regulation and funding.** In the short run, a fee on fertilizer use (including organic fertilizer sources) seems preferable to a cap-and-trade system since it could take advantage of existing infrastructure. Since fertilizers are currently exempt from sales taxes in California (CA State Board of Equalization 2004), the State could expand sales taxes to include fertilizer purchases. The impact a fertilizer fee has on nitrate contamination depends on a) the sensitivity of demand for fertilizer to a price change and b) the size of the fee. The long-term feasibility and desirability of a cap-and-trade system for nitrogen management could be evaluated.

6.2 Funding Programs and Options

1. **Where appropriate, combine funding programs.** Consolidating funding programs should lower administrative and application costs and improve program effectiveness. A single program will also ease demands on community applicants. For example, water supply and wastewater problems are often intertwined, and linking these sources of funding would reduce upgrade costs for small systems.

2. **Allocate funding to long-term drinking water solutions, particularly regionalization or consolidation of small systems.** Small drinking water systems face many challenges, including nitrate-contaminated groundwater. Connecting to larger systems or consolidating with other small systems can allow such problems to be addressed more effectively and at lower cost. But regionalization and consolidation of systems often requires costly upgrades for smaller systems (e.g., water meters or new pipeline) or requires the larger system to take on an undesirable amount of risk. The State and counties have an interest in encouraging regionalization and consolidation activities to avoid longer-term financial difficulties, water system service inadequacies, and public health problems.

[62] See Technical Report 2: Nitrogen Sources and Loading to Groundwater (Viers et al., 2012)

Adminsitrative Record
Page 35678

3. **Provide more financial assistance to small systems.** Small systems generally have more difficulty applying for funding programs and repaying loans. A grant or other special assistance program could be established to provide additional help to small systems. Such a program might be part of a larger consolidated effort by CDPH, the State Water Board, DWR, an Integrated Regional Water Management Plan, and the counties to address the problems of small water and wastewater systems.

4. **Create State funding programs for domestic well owners and for State small and local small water systems.** Currently, only water systems served by a public entity can apply for State safe drinking water funds (such as the State Revolving Fund). Domestic well owners and small communities with no recognized water systems thereby lack the funding sources available to public water systems. Assistance may also be provided to these small systems to help them form legally recognized entities.

5. **Increase the current mill assessment rate on nitrogen fertilizer to its fully authorized amount.** CDFA already oversees a mill assessment on fertilizer sales for research and education regarding the use and handling of fertilizing materials (including environmental effects), which is currently only half of its authorized amount. Raising the assessment to the fully authorized amount would raise roughly $3 million per year statewide. $1 million of this could be used for research and education regarding the use and handling of fertilizing materials (including environmental effects), or if current statute is changed, to fund some alternative drinking water supply efforts.

6. **Introduce a special fee on nitrogen fertilizer sales statewide, perhaps equivalent to a sales tax.** This economic signal could both reduce nitrogen applications and help fund safe drinking water solutions, nitrate source load reduction efforts, and nitrate monitoring and assessment programs. Expanding sales tax to include fertilizer could generate $28 million per year in the study area and might reduce applied nitrogen by 1.6%.[63] Similar fees/excise taxes could be considered for organic fertilizer sources (manure, green waste, wastewater effluent, biosolids, etc.).

7. **Consider a more comprehensive statewide fee on water use to support many beneficial activities**. Some of such revenues could fund management and safe drinking water actions in areas affected by nitrate contamination, including short-term emergency drinking water measures for disadvantaged communities. Assuming a fee is placed on statewide gross urban and agricultural water use,[64] the fee would need to be $0.5-$0.9 per acre-foot per year to provide safe drinking water to the estimated highly susceptible population of the study area.[65] Likewise, a fee only on statewide gross urban water use[66] would need to be $2.3-$4.1 per acre-foot per year.

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[63] Assume: 7.5% sales tax on the cost of nitrogen, a fertilizer retail price of $0.75 per pound of nitrogen, and 500 million pounds (227GgN) of fertilizer used in the Salinas Valley and Tulare Lake Basin (See Section 3 of Technical Report 3: Nitrate Source Reduction to Protect Groundwater Quality, Dzurella et al. 2012)

[64] Statewide gross urban and agricultural water use is estimated at 41.7 MAF per year. (Hanak et al. 2011)

[65] See Technical Report 7: Alternative Water Supply Options (Honeycutt et al., 2012)

[66] Statewide gross urban water use is estimated at 8.7 MAF per year. (Hanak et al. 2011)

Adminsitrative Record
Page 35679

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Adminsitrative Record
Page 35683

Appendix A: Regulatory Options in Practice

Watershed-Based Effluent Trading

Watershed-based effluent trading programs (also called water quality trading programs or nutrient trading programs) are market-based instruments used to effectively meet water quality goals at a lower cost (Selman et al. 2009). Such trading programs allow point source polluters to purchase contamination discharge reductions from other point or nonpoint source polluters that have lower marginal costs of abatement. Trading between point sources, which are regulated by EPA's National Pollutant Discharge Elimination System (NPDES), and nonpoint sources, which are excluded from State and Federal water quality regulations, could significantly lower the costs to comply with drinking water quality standards. Nonpoint source contamination abatement costs could be up to 65 times less than that of point sources (Bacon 1993) and the inclusion of nonpoint source contamination reduction in lieu of tertiary treatment could save $15 billion in abatement costs (U.S. EPA 1992).

Under a permit system, a measurable target is set for contamination abatement or water quality and sources are allowed to trade contamination permits between each other such that the total target is met. For this target to be met, one must understand the mixing and transport properties of a contaminant, which can be difficult. Kerr and others examined the information required and proposed a level of analysis needed to define allowances and set trading caps for a nutrient trading system to control nutrient inputs to a lake (Kerr, Lauder, & Fairman 2007). They found that caps on inputs from the trading program could not be defined until nutrient flows into the lake were scientifically estimated. They also found that assigning landowners a specific groundwater lag zone would assign nutrient loading values appropriate to the time lag of contaminant transport in groundwater.

In 2003, the EPA officially approved and recommended water quality trading to achieve watershed goals (Chesapeake Bay Foundation 2011), encouraging States to develop water quality programs to mitigate excess nutrient concentrations. At the same time, the agency recognized the limitations of water quality trading and found this approach to be feasible only when 1) there is an incentive for facilities to reduce their effluent contamination; 2) there is variation in abatement costs across sources; 3) contamination of a watershed can be reduced without requiring all sources to abate; and 4) all major stakeholders are engaged in the design and implementation of a water quality trading program (U.S. EPA 2003). The diffuse nature of groundwater and the absence of water quality regulations for nonpoint sources add difficulty in managing groundwater contamination.

Currently, in the U.S. at least 23 water trading programs have completed at least one trade; some examples are provided in *Environmental and Natural Resource Economics* (Tietenberg & Lewis 2003).

Adminsitrative Record
Page 35684

Thirteen statewide water quality trading guidance, policies or rules exist or are in development in the U.S., including (Selman et al. 2009):

- Connecticut Water Quality Trading Legislation,
- Delaware State Trading Initiatives (*under development*),
- Florida Water Quality Trading Rules (*under development*),
- Idaho Pollutant Trading Guidance,
- Maryland State Water Quality Trading Policy (*under development*),
- Michigan Water Quality Trading Rules,
- Minnesota Water Quality Trading Policy (*under development*),
- Ohio Water Quality Trading Policy
- Oregon Final Internal Management Directive of Water Quality Trading,
- Pennsylvania State Water Quality Trading Policy,
- Virginia State Water Quality Trading Rules,
- West Virginia Water Quality Trading Guidance (*under development*), and
- Georgia Water Quality Trading Initiatives (*under development*).

Long Island Sound Nitrogen Credit Exchange Program, Connecticut

In 2001, Connecticut imposed a Total Maximum Daily Load (TMDL) on the Long Island Sound that limits the amount of nitrogen that can be discharged into the estuary. Initially, Long Island Sound point source dischargers were allocated a number of permits such that each source would comply with the TMDL. However, since TMDL compliance was regulated at the estuary level, substantial trading could occur among point sources. Seventy nine point sources engaged in trading (State of Connecticut 2011). In 2002, a Nitrogen Credit Exchange (NCE) was formally created to facilitate all nitrogen credit exchanges within the Long Island Sound (Selman *et al.* 2009). Between 2002 and 2009, about 15.5 million nitrogen credits were exchanged, with a total purchased and sold value of $45.9 million (State of Connecticut 2010). It is estimated that this trading program generated potential savings of 300 to 400 million dollars (compared to more traditional approaches).

Pennsylvania Water Quality Trading Program

In 2006, Pennsylvania issued water quality trading guidance to allow point source trading with point and nonpoint sources to achieve nitrogen and phosphorus loading reductions (Selman *et al.* 2009). The program requires all sources to meet legal baseline requirements before nutrient and sediment reductions are considered eligible for trading. For point sources, this baseline effluent limit is the TMDL set by the NPDES permit and for nonpoint sources the baseline is determined by the existing regulation for each type of nonpoint source. For agricultural operations, this implies that sources must comply with existing State nutrient management laws. Between 2006 and 2009, five point and nonpoint source trades occurred (Selman *et al.* 2009). To address complications that arise due to uncertainty in nonpoint source abatement and the diffuse behavior of contamination in water, the regulator has suggested "establishing a threshold above legal baseline for generation of credits" (State of Pennsylvania 2007).

Adminsitrative Record
Page 35685

Nutrient Trading in the Chesapeake Bay Watershed

In 2001, the Chesapeake Bay Program established principles and guidelines for nutrient trading in the Chesapeake Bay Watershed (Chesapeake Bay Foundation 2011). Of the States bordering the watershed, Maryland, Pennsylvania and Virginia have passed legislation related to nutrient trading and are now in the implementation process. Virginia primarily allows point source trading, and limits nonpoint source trading to only new or expanding facilities. In Pennsylvania, a participant must comply with applicable State and Federal regulations before engaging in water quality trading. The Chesapeake Bay Foundation, the regulatory agency that oversees the Chesapeake Bay Program, established a margin of safety to account for the uncertainty in nonpoint source controls when allowing point-nonpoint source trades by imposing a minimum credit ratio of 2:1 for trades between nonpoint and point sources.

Liability from Groundwater Contamination from Pesticide Use

Throughout the U.S., liability rules have been applied and enforced to regulate pesticide contamination in groundwater. Liability rules vary across States with some States holding polluters strictly liable, others finding polluters liable only if due care was not taken, and still others implementing no liability rules. Under strict liability rules, the farmers and manufacturers identified as the liable parties are responsible for paying the costs to clean up the contaminated source and providing alternative sources of drinking water to the water users. With due care liability rules, a polluter is only liable for damages if a "due care" standard is not met.

The difficulty with liability rules is in identifying the liable party. This is especially true with groundwater pollution due to non-uniform mixing and the time lag between leachate/emissions and drinking water contamination. Some States have enforced liability rules to regulate pesticides in groundwater, suggesting that liability rules could also be applied to regulate nitrate.

In Connecticut, the pesticide ethylene dibromide (EDB) contaminated groundwater wells and the State applied legislation that held agricultural sources liable for contamination (Centner, 1989). The liable parties ultimately settled and farmers and manufacturers agreed to split the cost to provide potable water to affected water users. Prior to this case, farmers in Connecticut were exempt from pesticide contamination liability if they complied with certain requirements.

In California, a different approach was taken in interpreting and applying liability rules. The groundwater sources in many cities and water districts were contaminated with dibromochloropropane (DBCP) and 1,2,3-trichloropropane (TCP). However, it was difficult to prove that the behavior of specific farmers and manufacturers had polluted the water source. Instead, many water districts and cities sued the manufacturers of DBCP and TCP (Berck & Helfand 2010). Since it was known that these chemicals had contaminated the groundwater sources, the manufacturers were held responsible. For example, in 1996 the City of Lodi settled a lawsuit against DBCP manufacturers in which the manufacturers agreed to pay the costs related to DBCP treatment for 40 years (City of Lodi 2011). In 2008, the City of Bakersfield filed a lawsuit against TCP manufacturers, claiming that that these manufacturers

contaminated about 15% of the city's wells (Ferguson 2008). Similarly, the cities of Shafter, Delano and Wasco have also filed suits. These cases are still pending.

Hawaii also has been involved in lawsuits against manufacturers of DBCP and TCP (de Naie & McMahon 2011). In 1996, the Maui Water Board sued companies that supplied DBCP due to contamination found in groundwater wells in Haiku and West Maui. The settlement resulted in the chemical companies paying Maui County $3 million towards capital costs of a new filtration plant and about $200,000 operating costs every year for 40 years. The chemical companies also agreed to assist Maui County in complying with Federal and State standards for DBCP levels within 50 of their wells.

In these cases, liability rules led to funding alternative water supplies or the implementation of water treatment and monitoring. The enforcement of liability rules occurs when a liable party is identified. For nitrate in groundwater, it seems the polluter would be liable if a 'hot spot' was created and the manufacturer would be liable when the polluter is unidentified. For the Salinas Valley and Tulare Lake Basin, a "due care" standard would encourage farmers to reduce the amount of nitrogen applied and would hold them responsible for any contamination over that standard. The application of liability rules for managing pesticides suggests that it is feasible for managing nitrate. However, the long time period between nitrogen application and drinking-water impacts can make liability rules awkward and reduce their effectiveness.

Payment for Ecosystem Services in New York City

Currently, New York City participates in a payment for ecosystem services (PES) program for watershed protection. Under the US Safe Drinking Water Act (SDWA) the City was required to meet the State water quality standards by either constructing a water filtration plant at an estimated cost of $6 billion in capital and $300 million in annual operating costs (Postel & Thompson Jr. 2005) or implementing a much less expensive watershed protection program. The City successfully requested a waiver from the SDWA filtration requirement and negotiated an agreement with upstream landowners and communities within the Catskill-Delaware watershed to establish a watershed protection plan. In 1997, a memorandum of agreement (MOA) was signed by State and Federal officials, environmental organizations and 70 watershed towns and villages to invest $1.5 billion over ten years to restore and protect the watershed (Postel & Thompson Jr. 2005). Program financing comes from bonds issued by the City and increases in residential water bills.

The program's fundamental activities include land acquisition; a program to manage and reduce agricultural runoff; a program for better forestry management; a program for enhanced stream management to reduce erosion and habitat degradation; improvements for wastewater infrastructure in the watershed; construction of an ultraviolet disinfection plant; and new regulation and enforcement of mechanisms to ensure continued water quality protection within the watershed (Postel & Thompson Jr. 2005). As of 2004, the City has put $1 billion into the watershed protection program (Ward 2004). The negotiated partnership creates a watershed that provides high quality drinking water, provides landowners with additional income, and improves recreational usage for nearby communities.

Adminsitrative Record
Page 35687

In this instance, negotiation or payment for ecosystem services led to the provision of safe drinking water at a lower cost than the default water filtration plant. By linking the ecosystem service providers with the beneficiaries, New York City successfully executed a comprehensive watershed protection program that delivers safe drinking water at a relatively low cost. New York City's watershed protection program is an example of a PES program that guarantees the supply of high-quality drinking water and is financed via residential water bills and City bonds. Communities within the Tulare Lake Basin and Salinas Valley should examine potential partnerships amongst State and Federal officials, environmental organizations, and those liable for potential sources of contamination.

The Dutch Experience

(Mayzelle & Harter 2011)

In response to increasingly intensive animal production and a growing awareness of its effects on nitrate concentrations in surface water and groundwater, the European Council Nitrate Directive (ND) (Council Directive 91/67/EEC) was established in 1991 as part of the European Union (EU) Water Framework. The ND imposes a performance standard of 50 mg/L N (as nitrate) on effluent, groundwater and surface water quality levels within all EU countries. Furthermore, each country is required to establish nitrate contamination reduction plans, monitor program effectiveness, and regularly report their findings to the European Council (EC) (EU Publications Office). Compliance with the ND is costly in terms of time, expertise, and money; however, countries that do not meet ND standards face huge fines from the EC. While the ND does very little in the way of explicitly specifying how countries should act in efforts to comply with these requirements, plans that do not propose to regulate manure application at ND standards (i.e., land application rates in the range of 170-210 kg N/ha) have been historically rejected.

As an agricultural hotspot, the Netherlands has struggled to meet the ND requisites. To fulfill the obligatory ND requirements (Ondersteijn 2002), the Dutch government first created the Mineral Accounting System (MINAS) in 1998 (Henkens & Van Keulen 2001). MINAS was a farm-gate policy created to ensure the balance of nitrogen (N) and phosphorus (P) inputs (fertilizer and feed) and outputs (products and manure) on individual farms via balance sheets (Oenema, van Liere, & Schoumans 2005). MINAS resembled a farm-gate performance standard that was enforced by a penalty tax for excess N and P inputs: farms consuming more N or P than could be accounted for via harvest outputs would be fined per kilogram of N or P lost to the environment (as of 2003, fines of € 2.27/kg N [$ 1.40/lbs N] were enforced, more than seven times the cost of N fertilizer at the time). MINAS was popular for its simplicity, and was well supported by government aid. RIVM (Netherland's National Institute for Public Health and the Environment), which monitors N and P soil and water concentrations nationally, reports that nitrogen surpluses in agricultural areas fell substantially beginning in 1998 as a result of its implementation. Nevertheless, the EU declared the Dutch MINAS policy noncompliant with ND requirements, stating that the policy did not directly regulate water nitrate concentrations (Henkens & Van Keulen 2001).

In response to the EU's rejection of MINAS, the Netherlands implemented an additional policy in 2002: the Mineral Transfer Agreement System (MTAS). MTAS was a cap-and-trade system that prescribed manure (not inorganic fertilizer) application rates (as per ND objectives) and allowed farmers to purchase surplus application rights from those farmers applying manure to their land below legal limits. Rather than repealing MINAS, however, the Dutch increased enforceable fines under MINAS to serve as a safety net under the newly implemented MTAS (Ondersteijn 2002). Water nitrate levels continued to fall at pre-MTAS rates (Henkens & Van Keulen 2001; Ondersteijn 2002; Berentsen & Tiessink 2003; Helming & Reinhard 2009), suggesting that the implementation of MTAS in addition to MINAS had little or no effect.

Given the apparent futility of MTAS, and following the repeated rejection of MINAS by the European court of justice in 2003, both MTAS and MINAS were abandoned by the Dutch government by 2006. The two competing regulations were replaced by a composite policy that enforces N as well as P application standards for both manure and inorganic fertilizer, thereby satisfying both ND standards and the unique challenges encountered in Dutch territory, while minimizing administrative and economic costs. The composite policy remains in effect to date.

Adminsitrative Record
Page 35689

Appendix B: Summary of Current Planning and Regulatory Programs that Address Nitrate in Groundwater

Water quality management efforts also occur through many planning and regulatory programs, and statutes. They provide regulatory structure or technical and managerial support to water systems, communities, farmers, dairies, and others who deal with nitrate contamination in groundwater. Some only provide indirect assistance to managing nitrate contamination to the groundwater, and are marked as such.

U.S. Environmental Protection Agency (U.S. EPA)[67]

The Environmental Protection Agency (U.S. EPA) is responsible nationally for protecting human health and natural ecosystems.[68] U.S. EPA implements its authorities through ten U.S. EPA Regions, other Federal agencies, State and local governments, and tribal regulatory partners.

The Clean Water Act (CWA) of 1972 led U.S. EPA to set water quality standards for point discharges to surface waters and to control groundwater contamination by setting industry-wide effluent standards. Section 402 of the Clean Water Act, for example, required U.S. EPA to establish the National Pollutant Discharge Elimination System (NPDES) and to authorize permits under this program. The CWA does not provide a mechanism for regulating discharges to groundwater, so the permits (Waste Discharge Requirements) issued under the State Water Board are the only tool to currently regulate nitrate discharges to groundwater (see description of State Water Resources Control Board, below).

The U.S. EPA's Office of Water (OW) is responsible for implementing the Clean Water Act, the Safe Drinking Water Act, and several other statutes. They protect human health, support economic and recreational activities, and secure sufficient ecological habitat through the management of water bodies and ecosystems. The OW consists of several organizations: the American Indian Environmental Office; the Office of Wetlands, Oceans and Watersheds; the Office of Science and Technology; the Office of Wastewater Management; and the Office of Ground Water and Drinking Water.[68] The Office of Ground Water and Drinking Water maintains several activities to protect groundwater and protect public health through the provision of safe drinking water. U.S. EPA's Enforcement Policy Response and Enforcement Targeting Tool Programs are described below.

[67] http://water.epa.gov/aboutow/org/programs/owintro.cfm
[68] U.S.EPA Office of Water Information, accessed January 2011. Available at: http://water.epa.gov/aboutow/org/programs/owintro.cfm

Adminsitrative Record
Page 35690

U.S. EPA: Enforcement Policy Response and Enforcement Targeting Tool (ERP-ETT)[69]

[Indirect Assistance for Nitrate in Groundwater]

Over more than the past decade, violations of Safe Drinking Water Act regulations have mounted. Often these non-compliers are small rural systems which cannot afford the fine usually imposed after a violation occurs. EPA determined that these fines were just perpetuating the problem. As a result, the EPA recently revised and put in place a new system for tracking violations and their severity for public health for a given system and then monitoring the duration from violation to correction to better identify and then focus attention and effort on those systems that consistently struggle to comply with drinking water regulations.

The goal of the new system of supervision/monitoring is to help return systems to compliance more rapidly and sustainably. These non-compliance systems are dominated by small systems, across the USA. Small community water systems (<3,300 connections) have a 40-60% higher rate of "historical significant non-compliers" (HSNC) compared to large systems (>50,000 connections). While only 2% of California CWSs are classified as HSNCs, over 90% of these HSNC systems are "small" (US EPA definition is a system of less than 3,300 connections or 10,000 people). The new ERP-ETT system assigns a weight (# of points) to each type of violation, based on its threat to public health, and includes points for persistence of the violation (tracking the # of years since the first unaddressed violation occurred), in order to rank systems with health-based violations within the monitoring system. The existing system does not differentiate level of risk of a violation, treating all violations as equal, and does not provide information about trends in violations over time or about other violations over time of the same system. In the Central Valley, the main health-based violations among "small" systems of less than 500 people (there are 771 such sized systems in the Central Valley) are bacteria, nitrate and arsenic (Connie Li, US EPA Reg 9, UC Merced talk).

U.S. EPA: Supplemental Environmental Programs[70]

A Supplemental Environmental Program is agreed to under an enforcement settlement and is an environmentally beneficial project that a violator of environmental laws may choose to perform in addition to the actions required by law to correct the violation. The environmental project must be related to the violation and accomplished in place of other penalties. EPA defines seven specific categories that may be performed by the violating company: pollution prevention, pollution reduction, public health, environmental restoration and protection, assessment and audits, environmental compliance promotion, and emergency planning and preparedness. A company may, for example, pay a community for safe drinking water treatment to resolve a previous contaminating activity that affected groundwater quality.

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[69] http://www.epa.gov/compliance/resources/policies/civil/sdwa/drinking_water_erp_2009.pdf
[70] http://www.epa.gov/oecaerth/civil/seps

Adminsitrative Record
Page 35691

United States Department of Agriculture (USDA): National Drinking Water Clearinghouse, West Virginia University[71]

[Indirect Assistance for Nitrate in Groundwater]

The National Drinking Water Clearinghouse (NDWC) is sponsored by USDA's Rural Development Program Rural Utilities Service. Their engineers and experts provide technical assistance for small and rural drinking water treatment plants. They also inform communities on topics such as available funding options and Federal and State drinking water regulations. The NDWC publishes drinking water newsletters, technical brief fact sheets, and a magazine, all of which are valuable sources of information for small and rural water systems that face nitrate or other contaminants in their drinking water.

California Department of Public Health (CDPH)

CDPH: Drinking Water Source Assessment and Protection (DWSAP)[72]

[Indirect Assistance for Nitrate in Groundwater]

Since 1997, CDPH has used $7.5 million allocated from the State Revolving Fund for evaluating possible contaminating activities (PCAs) surrounding groundwater and surface water sources for drinking water. As of 2003, 94% of public water systems in California had been evaluated. This program helps communities better understand the vulnerability of their drinking water and prioritize use of limited funds towards source reduction and cleanup.

CDPH: Expense Reimbursement Grant Program (EPG)[73]

[Indirect Assistance for Nitrate in Groundwater]

Through a grant from the U.S. EPA, CDPH provides education, training, and certification for small water system operators. CDPH contracts Cooperative Personnel Services (CPS) to carry out these tasks. Eligible systems serve a community or non-transient population of less than 3,301 people.

State Water Resources Control Board (State Water Board)[74]

The State Water Board and each Regional Water Board are the principal State agencies responsible for the coordination and control of a unified and effective water quality control program in the State of California (Water Code § 13001). The State Water Board formulates and adopts State policy for water

[71] http://www.nesc.wvu.edu/ndwc/articles/OT/WI03/SRFandRUS.html
[72] http://www.cdph.ca.gov/certlic/drinkingwater/pages/dwsap.aspx
[73] http://www.cps.ca.gov/tlc/sws/
[74] http://www.swrcb.ca.gov

quality consisting of: a) water quality principles and guidelines for long-range resource planning, including ground water and surface water management programs and control and use of recycled water; b) water quality objectives at key locations for planning and operation of water resources development projects and for water quality control activities; and c) other principles, guidelines, and objectives deemed essential by the State Water Board for water quality control to provide a suitable living environment for California residents (Water Code § 13140 and § 13142).

The State Water Board adopts the water quality control plans (Basin Plans) prepared by each of the Regional Water Boards in California (33 U.S.C. 1313 (a), Water Code § 13170) as part of the California Water Plan. Each Regional Water Board must submit to the State Water Board a Regional Water Quality Control Plan (Basin Plan), except the Central Valley Regional Water Board, which has two plans: the Sacramento and San Joaquin River Basins and Tulare Lake Basin. These Basin Plans define beneficial uses for groundwater and surface water, set water quality levels to protect these beneficial uses, and establish programs for meeting these water quality objectives. Beneficial uses to be protected include, but are not limited to, "domestic, municipal, agricultural and industrial supply; power generation; recreation; aesthetic enjoyment; navigation; and preservation and enhancement of fish, wildlife, and other aquatic resources or preserves" (Water Code § 13050 (f)). The water quality standards must be reviewed every three years under the Clean Water Act and periodically under the California Water Code.[75]

The California Water Code Section 13263 authorizes the California Water Boards to issue Waste Discharge Requirements (WDRs) for projects or activities that discharge waste to waters of the State. The Federal Clean Water Act does not contain a mechanism to regulate discharges to groundwater (only surface water), so WDRs are the only regulatory tool that can be used to ensure that discharges to California groundwater does not exceed water quality objectives. The California Water Boards may find it in the public interest to issue a waiver of a WDR instead of a WDR. A waiver is limited to five years and has explicit conditions for protecting water quality.

In October 2001, the Groundwater Quality Monitoring Act of 2001 (AB 599) was established by the California Assembly. The Groundwater Quality Monitoring Act of 2001 required the State Water Board, an Interagency Task Force, and a Public Advisory Committee to: establish a comprehensive statewide groundwater quality monitoring program; increase the accessibility of groundwater quality data to the public; and allow groundwater basin assessment. Assessment is defined by AB599 as "assessing susceptibility of groundwater to water quality degradation, characterizing current water quality in a basin, and predicting future water quality under various conditions."

State Water Board: The Porter-Cologne Water Quality Control Act

The Porter-Cologne Water Quality Control Act was adopted in 1969 and granted the State Water Board the ultimate authority over State water quality policy (Water Code § 13146). The main goal of the Porter-Cologne Act is to regulate activities in California to achieve the highest reasonable water quality;

[75] Clean Water Act § 303(c) and California Water Code § 13240.

"reasonable" is defined after the demands on the waters and total values involved are considered (Water Code § 13146).

The framework of Porter-Cologne laid out the framework for future regulations and programs, including: the Dairy Waste Discharge Requirements regulatory program (focused on groundwater contamination from dairies), the Irrigated Lands Regulatory Program (initially focused on discharges to surface water only from all irrigation sources, but is now considering discharges to groundwater), and the Central Valley Salinity Alternative for Long-Term Sustainability (focused on surface water and groundwater and considers both salt and nitrate).

State Water Board: Groundwater Ambient Monitoring and Assessment (GAMA) Program[76]

[Indirect Assistance for Nitrate in Groundwater]

The GAMA Program, created in 2000, was officially mandated by the State legislature under the Groundwater Quality Monitoring Act of 2001 (AB 599) and is funded by Proposition 50 and special fund fees. It aims to improve statewide groundwater quality monitoring and increase the availability of groundwater quality information. With 95% of California's population on public water systems, and another 1.7 million on self-supply or very small systems less than 15 connections, reliant on groundwater for some or all of their drinking water, GAMA has a mandate to integrate disparate data collection efforts and initiate new programs as needed, to develop a comprehensive integrated statewide groundwater monitoring program. The USGS and Lawrence Livermore National Laboratory have been collaborators on this initiative, providing scientific expertise for developing the initial monitoring plan in 2003 and leading various groundwater quality assessments.

The GAMA Program consists of:

1) GeoTracker GAMA, an on-line searchable database that standardizes and integrates groundwater quality monitoring data collected by: the California Water Boards, the California Departments of Public Health, Pesticide Regulation, and Water Resources; the United States Geological Survey; and Lawrence Livermore National Laboratory into a unified data information system. GeoTracker GAMA covers over 150,000 locations in California. GeoTracker GAMA has been an important source on historical and current measurements of nitrate levels and co-constituents in groundwater in the Salinas Valley and Tulare Lake Basin study areas.

2) Priority Basin Project, focusing on 116 of 472 DWR groundwater basins in California, organized into study units, and with sampling and assessment of groundwater quality in priority basins for the presence and levels of CDPH regulated contaminants as well as other unregulated contaminants. The UC Davis Nitrate Project Study Area involves 4 GAMA Priority Basin Project Study Units: Kern, SE San Joaquin Valley, W. San Joaquin Valley, and Monterey.

[76] http://www.swrcb.ca.gov/water_issues/programs/gama/

3) Domestic Wells Project, a voluntary groundwater testing program for domestic wells with samples collected and tested at no cost by State Water Board staff. Over 1,000 domestic wells have been sampled so far in 5 counties, including Tulare. Tulare County domestic wells were sampled in 2006 under this project, and nitrate at or above the 10 mg/L nitrogen MCL (45 mg/L as nitrate) were found in over 40% of the sampled wells at levels up to 54 mg/L of N. Four percent of wells exceeded the nitrite MCL.

4) Special Projects, involving specialized research and study projects to measure and understand processes of groundwater contamination.

State Water Board: Recycled Water Policy[77]

In 2009, the State Water Board adopted Resolution No. 2009-0011: Recycled Water Policy. This policy sets goals for increasing the use of recycled water and storm water by 2020, including a goal for all regions to develop salt and nutrient management plans by 2012. These plans are to be managed on a basin-wide or watershed-wide basis. The State Water Board strongly encourages regions to include storm water use and recharge plans in their salt and nutrient management plans because storm water is typically lower in salt and nutrients. Each salt and nutrient management plan will also include a basin/sub-basin wide monitoring program dependent on the site-specific characteristics, but sufficient to determine if water quality objectives are being met.

Regional Water Quality Control Boards (Regional Water Boards)

Cleanup and Abatement Order (CAO)

Under authority of the Porter-Cologne Act, CA Water Code Section 13304 provides authority to the Regional Water Boards to compel known groundwater dischargers to clean up or cease degradation. Section 13304 states: “any person who has discharged or discharges waste into the waters of this State in violation of any waste discharge requirement …or who has caused or permitted….any waste to be discharged or….discharged into the waters of the State….shall upon order of the regional board, clean up the waste or abate the effects of the waste”. If a polluter refuses to comply with a CAO, the Regional Water Quality Control Board can request that the California Attorney General sue the polluter to force it to comply with the CAO.

Water Code Section 13304 also states that “A cleanup and abatement order issued by the state board or a regional board may require the provision of, or payment for, uninterrupted replacement water service, which may include wellhead treatment, to each affected public water supplier or private well owner.” This provides authority for a regional board to require landowners contributing to nitrate risk to groundwater drinking water supplies to support drinking water actions for affected public water supplies and private wells.

[77] http://www.waterboards.ca.gov/water_issues/programs/water_recycling_policy/docs/recycledwaterpolicy_approved.pdf

Adminsitrative Record
Page 35695

Central Coast Regional Water Quality Control Board (Central Coast Regional Water Board)[78]

The Central Coast Regional Water Board has jurisdiction over Santa Cruz, San Benito, Monterey, San Luis Obispo, and Santa Barbara Counties, and parts of Santa Clara, San Mateo, Kern, and Ventura Counties. It regulates activities over approximately three million hectares, with irrigated agricultural lands covering about 435,000 hectares.[79] The Central Coast Regional Water Board aims to complete the following by 2025: (1) 80% of aquatic habitat is healthy; (2) 80% of watershed lands will be properly managed to keep watersheds healthy and functioning well; and (3) 80% of groundwater will test clean according to the the Central Coast Regional Water Board's TDS and nitrate standards.[80] The Central Coast Ambient Monitoring Program (CCAMP) is the Central Coast Regional Water Board's water quality monitoring and assessment program. CCAMP gathers groundwater monitoring data from the U.S. Geological Survey, the Department of Health Services, and DWR and manages the databases to facilitate the Central Coast Regional Water Board's use. Under the Recycled Water Policy they have Salt and Nutrient Management Plan Requirements; stakeholders must develop implementation plans for meeting objectives for salts and nutrients.

To address the nitrate contamination of groundwater, the Central Coast Regional Water Board has created the Irrigated Lands Regulatory Program (also known as the Agricultural Regulatory Program) to regulate discharges from irrigated agricultural lands and encourage best management practices (BMPs), water quality monitoring, and proper implementation of corrective actions.

Central Coast Regional Water Board: Irrigated Lands Regulatory Program (ILRP)

Central Coast Regional Water Board's Irrigated Lands Regulatory Program (ILRP) specifically addresses crucial water quality issues on a priority watershed basis, continually assessing and tracking the progress of improvements in water quality and agricultural land management (Central Coast Regional Water Board, 2011). The priority watersheds are the Salinas River, Santa Maria, and Pajaro watersheds. The Conditional Waiver of WDRs for Discharges from Irrigated Lands was adopted in 2004, known as the Agricultural Waiver Program (AWP), and expired on September 30, 2011. The AWP is a voluntary negotiated agreement between the Central Coast Regional Water Board, growers, and environmental organizations. The AWP uses BMPs and rules for managing on-farm water resources to reach environmental goals, with the threat of mandatory regulatory action as incentive for a discharger to join the program. Farmers are required to complete ambient water quality monitoring, attend fifteen hours of educational classes, create a farm plan, and implement BMPs (Dowd et al., 2008). Waivers last for five years, with the quantity and quality of the reporting designated by the tier classification of the

[78] http://www.swrcb.ca.gov/rwqcb3/
[79] Central Coast Regional Water Board. Conditional Waiver of Waste Discharge Requirements for Discharges from Irrigated Lands. Draft Order No. R3-2011-0006.
[80] San Luis Obispo Science and Ecosystem Alliance. Achieving Management and Conservation Goals through the Application of Ecosystem-based Management of the Central Coast of California. August 27, 2008. http://groups.ucanr.org/HumboldtBayEBM/files/59049.pdf

Adminsitrative Record
Page 35696

discharger. Tier 1 dischargers are are generally considered sustainable (does not use chlorpyrifos or diazinon, is far from an Imparied Surface Waterbody, and any nitrogen discharges are far from public water systems wells); Tier 3 dischargers either have a high potential to discharge nitrogen to groundwater or apply chlorpyrifos/diazinon near an Impaired Waterbody; and Tier 2 dischargers fall in between the other two categories. A Tier 1 discharger provides an updated management-practice list two and a half years through the waiver and a Tier 2 discharger submits annual progress reports identifying current management practices and projections of educational goals completion (Dowd et al., 2008). If a discharger has not completed the educational classes after three years of the waiver, they are at risk of being issued a WDR. To decrease water quality monitoring costs, the Central Coast Regional Water Board and stakeholders agreed on a monitoring program for farmers to cooperatively monitor the main stems and tributaries of the Central Coast Region, sampling nutrients, temperature, orthophosphate, chlorophyll *a*, dissolved oxygen, total dissolved solids, pH, turbidity, and discharge (Dowd et al., 2008). The existing Conditional Waiver of Waste Discharge Requirements (WDRs) for Dischargers from Irrigated Lands (Irrigated Ag Order R3-2010-0040) was extended on July 8, 2010 (Central Coast Regional Water Board, 2011), March 29, 2011, and again on September 30, 2011.

The current order (Agricultural Order No. R3-2004-0117, dated Sept. 30, 2011) has been extended through September 30th, 2012. The Central Coast Regional Water Board is currently making revisions to the 2004 Agricultural Order. The September 1st, 2011 draft of the Agricultural Order requires Tier 3 dischargers with high nitrate loading risk to meet specified Nitrogen Mass Balance Ratios or implement an alternative solution that leads to an equivalent nitrogen load reduction.

Central Valley Regional Water Quality Control Board (Central Valley Regional Water Board)[81]

The Central Valley Regional Water Board has jurisdiction over nearly 40% of the State, including all or part of 38 of the State's 58 counties and about 80% of the State's irrigated agricultural land (Central Valley Regional Water Board, 2010). The Tulare Lake Basin section of the study area is within the Central Valley Regional Water Board's Fresno Boundary of Responsibility. The three major watersheds in the Central Valley Region are the Tulare Lake Basin, and the drainages of the Sacramento River and the San Joaquin River. The Central Valley Regional Water Board supervises the following programs to protect groundwater quality and to clean up contaminated groundwater: (1) Waste Discharge Requirements (WDRs); (2) Land Disposal (Title 27) Program; (3) Underground Storage Tank Program; (4) Cleanup Program; and (5) Federal Facilities Program.

The Central Valley Regional Water Board manages two main programs which affect ground water quality and specifically nitrate concentration: the Irrigated Lands Regulatory Program (ILRP) and the Central Valley Salinity Alternative for Long-Term Sustainability (CV-SALTS) program.

[81] http://www.swrcb.ca.gov/rwqcb5/

Adminsitrative Record
Page 35697

Central Valley Regional Water Board: Irrigated Lands Regulatory Program (ILRP)[82]

The current Irrigated Lands Regulatory Program (ILRP) in the Central Valley was created in 2003 as an interim program to regulate irrigated lands until 2011, when the long-term program will be completed.

The Draft Program Environmental Impact Report (DPEIR) for the long-term Central Valley ILRP was released on July 28, 2010. The Draft PEIR discusses five programmatic alternatives for regulating irrigated agriculture dischargers and their impacts, with Alternative 2[83] designated as the Central Valley Regional Water Board's Staff-Recommended Program (Central Valley Regional Water Board, DPEIR, 2010). A group of nonprofit organizations provided comments on the Draft, noting insufficient protection of water quality objectives or beneficial uses, recommending that the Central Valley Regional Water Board "revisit both its economic and environmental analyses as well as the components of the final program", and use Alternative 4[84] as the foundation of a revised program.[85] The nonprofit organizations believe an effective program can be formed from a fair and balanced analysis which involves: (1) collection of information on baseline parameters such as farm practices and water quality; (2) farm-level education and assistance requirements, and representative monitoring to ensure best management practices; (3) accountability by enforcement to induce compliance; and (4) plans to clean up legacy and continued agricultural contamination.[85]

Water quality plans will be required for areas where problems are known. The groundwater program would specifically look for nitrate, pathogens, and pesticides. Each grower would need to submit a farm evaluation. This plan will facilitate nitrate monitoring, increase the availability of water quality data, and ideally (through awareness) decrease the excessive application of nitrate fertilizer.

The August 2011 Recommended ILRP Framework proposes establishing three tiers: Tier 1 is for constituents that could affect, but do not pose a threat to, water quality; Tier 2 is assigned if the threat from irrigated agriculture is unknown; and Tier 3 will be assigned if irrigated agriculture is causing or contributing to a known water quality problem from a specific constituent. Tier 1 areas will generally not be required to monitor water quality, while Tiers 2 and 3 will be required to submit water quality monitoring and assessment reports. Agricultural operations in Tier 3 areas for which nitrate is the contaminant of concern will be required to prepare and maintain a farm-specific nutrient management plan (Central Valley Regional Water Board, Draft Requirements for Nutrient Management Plans in High Priority Groundwater Areas, Aug. 2011).

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[82] http://www.swrcb.ca.gov/centralvalley/water_issues/irrigated_lands/index.shtml

[83] Third party groups function as legal entities to represent the growers. The regulatory mechanisms will be established by the Central Valley Regional Water Board to be flexible to account for a variety of environmental conditions and agricultural operations.

[84] Growers or other legal entities responsible for waste discharges by a group of growers would apply to the Central Valley Regional Water Board for direct oversight with the option for regional monitoring by a third party.

[85] Community Water Center et al. Comments on the ILRP Staff Report, Economic Analysis and DPEIR, Sept. 2010

Adminsitrative Record
Page 35698

Central Valley Regional Water Board: CV-SALTS[86,87]

In 2006, the Central Valley Regional Water Board, the State Water Board, and affiliated stakeholders initiated an effort to address salinity and nitrate problems in the Central Valley and to establish solutions for improved water quality.[86] The Central Valley Salinity Alternative for Long-Term Sustainability (CV-SALTS) is a collaborative basin planning effort aimed at developing and implementing the policies and science to create sustainable and comprehensive salinity and nitrate management.[86] A primary goal of CV-SALTS is to update the policies and regulations of the Water Quality Control Plans "to facilitate cost effective salinity management while protecting beneficial uses of surface and ground waters".[86] New water quality objectives for salts and nitrate will be established and where the objectives are not met a comprehensive implementation plan will be established to protect water quality and meet the objectives in the near future. In 2008, the Central Valley Salinity Coalition (CVSC) was formed to represent stakeholders collaborating with the Central Valley Regional Water Board to efficiently manage salinity in the Central Valley. All efforts needed to complete the goals of CV-SALTS are administered and financed by the CVSC.

In 2010, Larry Walker Associates and others conducted the first pilot study of the CV-SALTS initiative to address the salt and nutrient management problems in the Central Valley and to guide stakeholders in creating effective salt and nutrient management plans. Input data sets were identified and assembled for the Watershed Analysis Risk Management Framework (WARMF) model to quantitatively relate salt and nitrate sources and sinks within the Yolo, Modesto, and Tule River areas.[88] The study found irrigation and fertilizer/land application to be the principal inputs of nitrate to near-surface groundwater and found nitrate is accumulating in near-surface groundwater.

A draft outline of stakeholder (i.e., polluters, environmental NGOs, etc.) proposed elements for a salt and nitrate management plan is available on the CVSC website.[89] The salinity and nitrate management plans will outline basin monitoring programs, identify salinity and nitrate sources and processes, create a nutrient budget, identify the population affected, analyze and compare trends with beneficial uses and water quality objectives, and develop policies. Basin Plan Amendments will be completed May 2015.

Central Valley Regional Water Board: Dairy Program[90]

To comply with the Porter-Cologne Water Quality Control Act (California Water Code Division 7), the Central Valley Regional Water Board created the Dairy Program, also known as the Confined Animal Facility Program, to regulate confined animal facilities, including dairies, feedlots, poultry facilities, and

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[86] http://www.swrcb.ca.gov/centralvalley/water_issues/salinity/index.shtml
[87] http://www.cvsalinity.org/
[88] CV-SALTS: Salt and Nitrate Sources Pilot Implementation Study Report, February 2010. Larry Walker Associates, and others.
[89] The non-profit coalition of stakeholders, known as the CVSC, accessed December 2010. Available at: http://www.cvsalinity.org/index.php/documents/cat_view/39-docs/49-documents-related-to-salt-and-nutrient-management-planning.
[90] http://www.swrcb.ca.gov/rwqcb5/water_issues/dairies/index.shtml

horse facilities.[91] Under the Dairy Program, dairies, feedlots and other confined animal facilities must comply with set statewide water quality regulations and existing milk cow dairies of all sizes must follow waste discharge requirements. Requirements for the dairy production area and land application area are outlined in the Waste Discharge Requirements General Order for Existing Milk Cow Dairies (General Order) adopted on May 3, 2007.[91] The General Order requires that all domestic and agricultural supply wells and subsurface (tile) drainage systems in the production and/or land application areas be sampled by November 3, 2007, and then annually under the General Order.[92] The General Order requires existing milk cow dairies to conduct nutrient and groundwater monitoring, measuring electrical conductivity, total ammonia nitrogen and un-ionized ammonia nitrogen (NH_3-N). The General Order requires that each dairy implement their Waste Management Plan (WMP) by 2011 and their Nutrient Management Plan (NMP) by 2012. The WMP ensures "the production area of the dairy facility is designed, constructed, operated and maintained so that dairy wastes generated at the dairy are managed in compliance with WDR General Order No. R5-2007-0035 in order to prevent adverse impacts to groundwater and surface water quality".[92] The NMP is created to "budget and manage the nutrients applied to the land application area(s) considering all sources of nutrients, crop requirements, soil types, climate, and local conditions in order to prevent adverse impacts to surface water and groundwater quality".[92]

California Department of Food and Agriculture (CDFA)

The California Department of Food and Agriculture (CDFA) is the lead agency responsible for protecting California's agriculture, enforcing environmental regulations on agricultural production, and ensuring equitable marketing to consumers. In 1988, the CDFA secretary appointed the Nitrate Working Group (NWG) to study California's agricultural nitrate problem. In 1989, the NWG wrote "Nitrate and Agriculture in California", a report which identified California's "nitrate-sensitive areas", recommended a prioritized plan for those areas to start and implement nitrate management programs, and improved farming practices to decrease nitrate loads to groundwater. In 1990, the Director of the CDFA established the Nitrate Management Program (NMP) and tasked them with implementing the report recommendations. The NMP then led to the creation of the Fertilizing Materials Inspection Program (FMIP) and a Fertilizer Research and Education Program (FREP), described below.

CDFA: Feed, Fertilizer and Livestock Drugs Regulatory Services (FFLDRS)

The CDFA Inspection Services Division has a Feed, Fertilizer and Livestock Drugs Regulatory Services (FFLDRS) Branch which runs the FMIP and FREP programs. FMIP regulates the manufacture, distribution, licensing, and labeling of fertilizing materials in California, to provide safety and quality assurance. FMIP has a Fertilizer Inspection Advisory Board comprised of nine members in charge of

[91] Central Valley Regional Water Board Dairy Program, accessed November 2010. Available at: http://www.swrcb.ca.gov/rwqcb5/water_issues/dairies/index.shtml
[92] Order No. R5-2007-0035. Waste Discharge Requirements for General Order for Existing Milk Cow Dairies. May 3, 2007. Available at http://www.waterboards.ca.gov/centralvalley/adopted_orders/GeneralOrders/R5-2007-0035.pdf

recommending proposed regulations to the Secretary of Agriculture. The structure and functions of the FMIP could be expanded to regulate and track fertilizer use applications and to collect fees on fertilizer sales, with little need for legislative action.

FREP funds and researches methods for ensuring fertilizer use is environmentally safe and proper handling practices are followed. FREP was the first attempt at voluntary action to reduce nitrate in response to the 1989 "Nitrates and Agriculture in California" Report. FREP involves funding farmer education and research about nutrient management. FFLDRS programs are funded from a mill tax and license, registration and inspection fees. Currently a total assessment of $0.0015 per dollar of fertilizer sales is collected, however the Food and Agricultural Code allows an assessment of up to three mills ($0.003). This $0.0015 assessment consists of a $0.0005 per dollar sales assessment on all commercial fertilizer to fund research and educational projects that improve farming practices and decrease environmental impacts due to fertilizer use,[93] and a $0.001 per dollar of sales assessment to pay a "fertilizing materials" inspection assessment.[94] The fertilizer research assessment ($0.0005) currently generates about $1 M per year. This funding source could be quickly and easily increased because the code allows up to two mills ($0.002) per dollar of sales to be imposed on all fertilizing materials distributed in California and an additional assessment of up to one mill ($0.001) per dollar of sales to be imposed on all licensees to provide funding for research and education pertaining the use and handling of fertilizing material and any environmental effects.[95] Since the current assessments collected for licensing and research and educational projects is only half of the allowed amount, the assessments could be raised to the full three mill assessment.

Assembly Bill 3030 (Groundwater Management Act)[96]

[Indirect Assistance for Nitrate in Groundwater]

Since 1993, Assembly Bill 3030 has permitted local agencies to adopt programs to manage groundwater. The Central Valley Project Improvement Act further requires that all water suppliers which overlie a useable groundwater basin develop a groundwater management plan under AB 3030 guidelines. AB 3030 lists technical components which may be included, such as the identification of well construction policies, the coordination of land use planning to reduce the risk of groundwater contamination, and the identification of wellhead protection areas. The technical components of AB 3030 provide a means for local agencies to protect their local resources from nitrate contamination. Once a plan is adopted, the local agency must pass rules and regulations which maintain consistency with the plan.

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[93] The California Food and Agriculture Fertilizer Research and Education Program: The California Food and Agricultural Code Section 14611(b), accessed December 2010. Available at: http://www.cdfa.ca.gov/is/fflders/pdfs/2010_FREP_Proceedings.pdf
[94] The California Food and Agriculture Fertilizing Materials Inspection Program: Fertilizing Materials Inspection Assessment, accessed December 2010. Available at: http://www.cdfa.ca.gov/is/fflders/fertilizer.html
[95] The California Department of Food and Agricultural Code: Division 7, Chapter 5, Article 6, Section 14611, accessed January 2011. Available at: http://www.leginfo.ca.gov/cgi-bin/displaycode?section=fac&group=14001-15000&file=14611-14611
[96] http://www.water.ca.gov/groundwater/gwmanagement/ab_3030.cfm

Adminsitrative Record
Page 35701

Kern County Water Agency (KCWA) [97]

[Indirect Assistance for Nitrate in Groundwater]

The Kern County Water Agency (KCWA) monitors and reports groundwater quality and levels in Kern County. This information is primarily used in conjunction with groundwater banking and recharge projects. The KCWA monitors around 240 wells monthly and 800 semiannually. The well data are critical to understanding historical nitrate levels in the county groundwater basins and for catching future changes in water quality before contamination affects public health.

Monterey County Water Resources Agency (MCWRA) [98]

[Indirect Assistance for Nitrate in Groundwater]

The Monterey County Water Resources Agency (MCWRA) (established as the Monterey County Flood Control and Water Conservation District in 1947 and renamed in 1990), provides flood and water quality management and protection to the people of Monterey County. The Board has nine members appointed by the Monterey County Farm Bureau, Monterey Grower-Shipper Association, the Monterey County Farm Advisory Committee, County Supervisors, and the Mayor's Select Committee.[99]

One of their six listed purposes for collecting water quality data is to evaluate nitrate in groundwater. Their nitrate monitoring program existed since 1978, with a 1995 study showing that 28% of 350 sampled wells exceeded the nitrate MCL.[100] The highest concentrations were seen in the Upper Valley, East Side, and Forebay Subareas.

For the local agriculture community, MCWRA has a water quality planning program that includes research efforts and outreach to growers to improve irrigation efficiency and fertilizer management to effectively reduce nitrate leaching. As an educational guide, they provide online access to Nitrate Management Fact Sheets that describe water and fertilizer management techniques, guidelines for handling fertilizers, and methods for accounting for nitrate already present in the soil and water. They also provide instruction on how to properly monitor and sample for nitrogen in the soil and water, as well as resources for ordering nitrogen test kits and strips.

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[97] http://www.kcwa.com

[98] http://www.mcwra.co.monterey.ca.us

[99] An article written by Monterey Coastkeeper on October 21, 2010. Available at: http://yubanet.com/california/Monterey-Coastkeeper-Sues-Monterey-County-Water-Resources-Agency-To-Protect-Water-Quality.php

[100] Water Resources Data Report: Water Year 1995. MCWRA. http://www.mcwra.co.monterey.ca.us/Agency_data/Hydrogeologic%20Reports/WaterResourcesDataReport/Section6_WaterQuality.pdf

Adminsitrative Record
Page 35702

Monterey County Health Department

[Indirect Assistance for Nitrate in Groundwater]

The Monterey County Health Department implements a tiered nitrate sampling program based on increasing concentration for local small water systems and state-small water systems. If the system has recently measured a nitrate concentration at or above the MCL, the system must increase their sampling rate from yearly to quarterly. Monterey County Health Department's nitrate sampling program is more stringent than the State regulations, which only require one-time monitoring for nitrate at the point of initial permit application (CCR Title 22).

Southern San Joaquin Valley Water Quality Coalition[101]

[Indirect Assistance for Nitrate in Groundwater]

The Southern San Joaquin Valley Water Quality Coalition was established in 2002 as a result of a request by the Central Valley Regional Water Board to create a voluntary water quality monitoring program as part of the region-wide consideration of agricultural discharge permitting. The Coalition was formed to serve the Tulare Lake Basin watershed and involves various agencies, including the Kern County Water Agency and Kings River Water Association. The Coalition publishes an annual report which documents their efforts to protect and preserve water quality supplies and water rights in the watershed. Recently, the Coalition was named a member of the stakeholder work group responsible for evaluating and updating the Irrigated Lands Regulatory Program for the Central Valley Regional Water Board.

Tulare County Water Commission

[Indirect Assistance for Nitrate in Groundwater]

The Tulare County Water Commission meets once a month to discuss water issues affecting Tulare County. The Commission includes engineers, water district managers, elected officials and community activists and serves as an advisory body to the Tulare County Board. The Commission, along with the Community Water Center, lobbied for funding to address the drinking and wastewater needs of disadvantaged communities in the Tulare Lake Basin. This $2 million project is currently underway (The Disadvantaged Community Water Study Project). A major focus of this project is identifying the overlap of disadvantaged communities and poor groundwater quality areas. This analysis will be more detailed than the rough analysis in this report (see Alternative Water Supply Report) and will evaluate community-specific solutions.

[101] http://www.ssjwqc.org

Adminsitrative Record
Page 35703

Rural Community Assistance and Partnership (RCAP)[102]

[Indirect Assistance for Nitrate in Groundwater]

The Rural Community Assistance Partnership is a national organization of six regional partners which helps communities of less than 10,000 people, often disadvantaged and frequently with populations of less than 2,500. Through publications, training events, conferences, toolboxes, and hands-on technical assistance, RCAP helps people living in rural communities to: access safe drinking water supplies, properly treat and dispose of wastewater, finance infrastructure projects, understand regulations, and manage water facilities. They do not provide loans or grants to communities, but they provide financial operations assistance and guidance. RCAP receives funding from the U.S. Department of Agriculture Rural Development program, EPA's Office of Groundwater and Drinking Water and Office of Waste Water Management, and the Office of Community Services of the U.S. Department of Health and Human Services.

Monterey Coastkeeper

[Indirect Assistance for Nitrate in Groundwater]

The Monterey Coastkeeper is a non-profit organization serving Monterey and Santa Cruz Counties and portions of San Mateo, Santa Clara and San Benito Counties. The Coastkeeper advocates for effective government, public policy, and active community participation for the protection of water quality. The Monterey Coastkeeper particularly seeks more effective monitoring requirements of agricultural runoff and collaborates with the State Water Board to ensure the success of the Agricultural Waiver Program. In 2010, the Monterey Coastkeeper encouraged the Central Coast Regional Water Board to adopt more stringent waste discharge requirements for the Gallo Cattle facility near Gonzales. As a result, the Central Coast Regional Water Board also required Gallo to create a groundwater management plan.

National Rural Water Association (NRWA)[103]

[Indirect Assistance for Nitrate in Groundwater]

The National Rural Water Association is the largest water and wastewater utility membership organization in the U.S. They focus primarily on systems with less than 10,000 people but they also have representatives from 94% of public water systems overall. They offer technical advice in the areas of operation, management, finance, and governance. NRWA advocates for small and rural systems to insure that rules and regulations are appropriate for everyone and that sufficient funding is available to these systems. Additionally, they have developed a library of free white papers for rural and small water

[102] http://www.rcap.org
[103] http://www.nrwa.org

Adminsitrative Record
Page 35704

and wastewater systems. These reports are valuable to small and rural systems where data and information are often difficult to obtain.

California Rural Water Association (CRWA)[104]

[Indirect Assistance for Nitrate in Groundwater]

Incorporated in 1990, the California Rural Water Association is a state affiliate of the National Rural Water Association. CRWA provides online classes, onsite training, low cost educational publications, and other forms of technical advice for rural water and wastewater systems. They also provide legislative representation, aid in developing new rate schedules, installing new testing methods, understanding government regulations, or updating operator certification requirements specifically for small rural community water and wastewater needs.

Self-Help Enterprises (SHE)

[Indirect Assistance for Nitrate in Groundwater]

Self-Help Enterprises (SHE) helps low-income families in the San Joaquin Valley to help themselves through the improvement of their water systems, among other projects. They can provide seed money to small rural disadvantaged communities for preparing reports and studies needed for funding applications for water system construction and improvements. They also assist these communities in preparing their applications to programs providing water system funding by providing human resources and the technical assistance to develop adequate water delivery and wastewater disposal systems. Drinking water projects involving nitrate contamination have been a major part of their work. SHE partnered with the Community Water Center to offer free well testing and funding advice to the community of Monson who faced high levels of nitrate and DBCP.

Community Water Center (CWC)[105]

[Indirect Assistance for Nitrate in Groundwater]

The Community Water Center (CWC) advocates for providing safe drinking water to all communities in the San Joaquin Valley, regardless of economic condition. They lobby local and State government, support policies, educate local agencies and communities, and organize community projects. Recently, the CWC published a study of nitrate contamination of drinking water and related health effects, specifically in the San Joaquin Valley. This document was designed to educate local communities so they are aware of the nitrate problem and understand their options for obtaining safe drinking water.

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[104] http://www.calruralwater.org/CRWA.htm
[105] http://www.communitywatercenter.org/

Adminsitrative Record
Page 35705

The CWC coordinates the AGUA Coalition (la Asociación de Gente Unida por el Agua, or the Association of People United for Water). AGUA campaigns for regional solutions to enduring water system problems in the San Joaquin Valley. They are currently focused on protecting groundwater sources for drinking water use from contaminants, especially nitrate. Recently, the CWC brought nation-wide attention to the nitrate contamination issue in the Central Valley by organizing the visit of a U.N. representative.

Adminsitrative Record
Page 35706

Appendix C: Description of Current Funding Programs for Safe Drinking Water

This appendix summarizes existing funding programs for safe drinking water in the study area. These resources are available from the national to the local level.

California Department of Public Health (CDPH)

The California Department of Public Health (CDPH) is responsible for implementing Public Resources Code (PRC) Sections 75020 through 75023 and 75025 under Chapter 2: Safe Drinking Water and Water Quality Projects. They administer both State and Federal funds to improve drinking water systems.

CDPH: Safe Drinking Water State Revolving Fund (SDWSRF)[106]

This is one of the State's major forms of funding for local capital improvements. A description of the Safe Drinking Water State Revolving Fund (SDWSRF) is found above in Section 4.3 Current Funding Example: Safe Drinking Water State Revolving Fund.

CDPH: Proposition 84[107]

[Fully Allocated]

Proposition 84 (the Safe Drinking Water, Water Quality and Supply, Flood Control, River and Coastal Protection Bond Act of 2006), allocated $1.5 billion of its $5.4 billion in general obligation bonds to CDPH, DWR, and the State Water Board for safe drinking water and water quality projects. CDPH received funding for four main purposes related to public water systems.

Emergency and urgent projects were allocated $10 million to ensure the immediate health and safety of drinking water supplies. Projects include: emergency interties with larger water systems; tank, pump, and well replacements; design, installation, and initial operation costs for water treatment systems, and the provision of bottled water when necessary. In 2007, this emergency fund provided approximately $81,000 for water districts within Tulare County to replace well equipment.[108]

Small community drinking water system improvements to help meet the safe drinking water standards were allocated $180 million. Priority is given to projects that reduce nitrate and other chemical contaminants in disadvantaged communities. Construction grants are limited to $5 million per project. Funding also is available for feasibility studies and engineering reports, so water systems can meet

[106] http://www.cdph.ca.gov/services/funding/Pages/SRF.aspx
[107] http://www.cdph.ca.gov/services/funding/Pages/Prop84.aspx
[108] CDPH's Proposition 84: Emergency Safe Drinking Water Supply Program, accessed January 2011. Tulare County water districts: Ducor Community Services District, Lanare Community Services District, Tooleville MWC and West Goshen MWC. Available at: http://bondaccountability.resources.ca.gov/plevel1.aspx?id=1&pid=4

Adminsitrative Record
Page 35707

application requirements for construction grants. A small amount of this funding ($5 million) is available for community technical assistance.[109]

Prevention and reduction of contamination of groundwater sources that serve drinking water systems was allocated a total of $60 million. Projects must be ready to begin implementation immediately and must protect groundwater that provides at least one third of water for a community. The maximum grant per applicant is $10 million. Additional points are given to projects that: serve disadvantaged communities, affect a population greater than 100,000, or address contaminants with acute health effects.[109],[110] Additionally, $50 million was allocated to increase the SDWSRF.[109]

CDPH: Proposition 50[111]

[Fully Allocated]

Proposition 50 (the Water Security, Clean Drinking Water, Coastal and Beach Protection Act of 2002) awarded $3.4 billion to various State departments. CDPH was allocated $50 million for water security projects for drinking water systems. These include emergency interties, improvement and installation of treatment facilities, and monitoring programs. $69 million was set aside to help community and small community water utilities meet safe drinking water standards by providing grants for infrastructure improvements, pilot studies, and the improvement of water quality monitoring, treatment, and distribution facilities. An additional $105 million was provided as match funds for Federal grants for public water system infrastructure improvements.[112,113]

State Water Resources Control Board (State Water Board)

The State Water Board has funding programs to reduce contamination of surface and groundwater from point and non-point sources; the main programs are described below.

State Water Board: Clean Water State Revolving Fund (CWSRF)[109]

In 1987, an amendment to the Clean Water Act established the Clean Water State Revolving Fund (CWSRF) for water quality projects. The CWSRF is funded through Federal grants (most recently the American Recovery and Reinvestment Act (ARRA) of 2009), State funds, and revenue bonds (including Prop 84: $73.2 million to reduce or prevent contamination of impaired water bodies). The fund provides low-interest or subsidized loans for construction of publicly-owned wastewater treatment facilities, local sewers, sewer interceptors, and water recycling facilities. Up to $50 million per year is allowed for each applying agency, program, or water system. Funded projects include wastewater

[109] http://bondaccountability.resources.ca.gov/plevel1.aspx?id=2&pid=4
[110] http://www.cdph.ca.gov/services/funding/Documents/Prop84/Prop84Section75025Criteria-09-17-2009.pdf
[111] http://www.cdph.ca.gov/services/funding/Pages/Prop50.aspx
[112] http://www.resources.ca.gov/bond/Prop_50_Summary_of_Programs2.pdf
[113] http://www.resources.ca.gov/bonds_prop50.html

treatment plant upgrades and improvements, water reclamation, plant nitrification and de-nitrification facilities, and sewer replacements. This program and funding provides a source of financing for municipal and septic source loading reductions by facilitating the replacement of septic systems or treating wastewater discharges for nitrate. $200-$300 million total is dispersed annually.

State Water Board: Small Community Wastewater Grants[114]

In conjunction with the CWSRF is the Small Community Wastewater Grant program for disadvantaged small communities. This program was established in 2004 and funds up to 90% of the costs of planning, design, and construction for publicly-owned wastewater treatment and collection facilities in qualifying communities. It specifically addresses the unique needs of small communities with less than 20,000 people and with financial hardships (quantified by a median household income of less than 80% of the statewide MHI). Funds may be used to improve wastewater collection and treatment systems, including the replacement of old pipes and septic systems to reduce contamination to surface and groundwater.

This grant program has been financed with funds from Propositions 40 and 50 in the past and now receives funds from a small fee on CWSRF loan agreements. In 2009, $86 million of obtained ARRA Federal funds of the CWSRF were used to forgive the principal loan amount for 25 small and disadvantaged community wastewater projects. As part of the small community strategy, the State Water Board also makes grants to non-profit organizations (such as RCAC) to provide free technical assistance and training to small disadvantaged communities in wastewater infrastructure management and system operations, and in assessment and preparation of project applications.

State Water Board: Proposition 50[115]

[Fully Allocated]

Proposition 50 awarded $100 million to the State Water Board for grants to public agencies and nonprofit organizations to improve water quality. Funds can be used for: drinking water source protection projects (including well head protection from nitrate and other contaminants), water contamination prevention programs, and water quality blending and exchange projects.

State Water Board: Agricultural Drainage Loan Program[116]

[Fully Allocated]

This loan program is for projects that address treatment, storage, conveyance or disposal of agricultural drainage that threatens waters of the State. $100,000 per project is allowed for feasibility studies with a $5 million per project cap on implementation costs. Overall, $30 million was allocated to this program and there is currently $10.4 million still available.

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[114] http://www.swrcb.ca.gov/water_issues/programs/grants_loans/small_community_wastewater_grant/index.shtml
[115] http://www.resources.ca.gov/bond/Prop_50_Summary_of_Programs2.pdf
[116] http://www.swrcb.ca.gov/water_issues/programs/grants_loans/agdrain/agdrain_mgmt.shtml

State Water Board: Dairy Water Quality Grant Program[117]

[Fully Allocated]

Proposition 50 included $5 million for regional and on-farm dairy projects that address water quality impacts from dairies. Water quality planning and both regional and on-farm projects were allowed. The dairy operator was required to have completed the environmental stewardship short course of CDQAP (California Dairy Quality Assurance Program). The final list of recommended projects was developed in 2006, so applications are no longer being accepted.

State Water Board: Federal Clean Water Act Section 319: Nonpoint Source Implementation Program

Through the Clean Water Act Section 319, the State Water Board provides grants (with a match requirement) for implementation of measures and practices that reduce or prevent nonpoint source contamination to ground and surface waters. Normally, individual requests up to $1 million are accepted and $4.5-5.5 million is allocated per year.

State Water Board: Cleanup and Abatement Account[118]

The Cleanup and Abatement Account (CAA) was established through Water Code Sections 13440-13443. The State Water Board is responsible for the financial management of the program. The CAA may provide funds to the State Water Board, the Regional Water Boards, and other public agencies authorized to cleanup a waste or abate the effects of a waste. The CAA funds are used to clean up a waste, abate the effects of a waste, and remediate an unforeseen water contamination problem. Often, a viable responsible party has not been identified. In 2009, $12 M was provided for 24 projects, four of which were in the study area (Richgrove CSD, Mettler County Water District, North Shafter Wastewater Project, and the Central Valley Salinity Work Group). In 2010, another $9 M was provided to support 15 projects; two of these projects were located in the study area (San Jerardo Wastewater Improvement Project and Central Valley Salt and Nutrient Plan).

State Water Board: Integrated Regional Water Management (IRWM) (Proposition 50)

[Fully Allocated]

The Integrated Regional Water Management (IRWM) program was launched following the passage of the IRWM Act of 2002 (SB 1672) to encourage local agencies to work cooperatively to manage local and imported water supplies to improve the quality, quantity, and reliability. In 2002, Proposition 50 allocated $15 million for planning projects and $365 million for implementation projects related to protecting and improving water quality and other projects to ensure sustainable water use. A 25% cost

[117] http://www.swrcb.ca.gov/water_issues/programs/grants_loans/dairy/index.shtml
[118] http://www.waterboards.ca.gov/water_issues/programs/grants_loans/caa/

share is required of all localities, programs, and other groups looking for funding from the IRWM program.

Department of Water Resources (DWR)

DWR: Integrated Regional Water Management (IRWM) Financial Assistance (Propositions 50, 84)[119]

DWR also has several Integrated Regional Water Management (IRWM) grant program funding opportunities. Proposition 50 (2002) provided $500 million specifically to DWR to fund competitive grants for projects consistent with an adopted IRWM plan (CWC § 79560-79565). Proposition 84 (2006) provided $1 billion total ($60 million to the Tulare Lake region and $52 for the Central Coast region) to DWR for IRWM Planning and Implementation (PRC §75001-75130). All Proposition 50 funds have been allocated and approximately $500 M remains of the Proposition 84 funds.[120]

DWR: Local Groundwater Assistance (Prop 50 & Prop 84)[121]

Under the Department of Water Resources' Local Groundwater Assistance program, financing is provided to local public agencies with authority to manage groundwater resources for projects that involve: groundwater data collection, modeling, monitoring and management studies; monitoring programs and installation of equipment; basin management; development of information systems; and other groundwater related work. Funds were initially available from Proposition 50 ($6.4 million total and limited to $250,000 per applicant). Renewed funding from Proposition 84 was available in 2010/11.

DWR: New Local Water Supply Feasibility Study[122] and Construction Funds[123] (Proposition 82)

Managed by DWR and funded from Prop 82, this program has $22 million of loan money for feasibility study and construction activities, but only towards projects which will improve *existing* water supply (diversion, storage, or distribution) problems. Local public agencies can apply for loans for projects such as canals, dams, reservoirs, groundwater extraction facilities or other construction or improvements to their water supplies. While the maximum loan amount for feasibility studies is small, it does allow systems or communities to evaluate potential new water sources. It permits a loan up to $0.5 million per eligible feasibility study with 5 year repayment limit and up to $5 million per eligible construction

[119] http://www.water.ca.gov/irwm/
[120] California Resources Agency. August 2011. Proposition 84 Allocation Balance Report. Accessed December 2011: http://bondaccountability.resources.ca.gov/Attachments/b1a801cb-36af-44c7-854e-3b9047a4525f/29/Prop84AllocationBalanceReport.pdf
[121] http://www.water.ca.gov/lgagrant/
[122] http://www.grantsloans.water.ca.gov/loans/feasibility.cfm
[123] http://www.water.ca.gov/grantsloans/prop82/

Adminsitrative Record
Page 35711

project with a 20 year repayment limit. The interest rate is the rate of the State's most recent obligation bonds. Applications are processed on a continuous basis.

DWR: Water Use Efficiency Grant Program (Prop 50)[124]

Proposition 50 established $180 million for grants for urban and agricultural water conservation, recycling, and other water use efficiency programs. $105 million of this allocation was awarded to DWR for the Water Use Efficiency Program. This program supports projects to improve agricultural water use efficiency, including: research and development, feasibility studies, training and education, and technical assistance. Specific projects include: wellhead rehabilitation, new storage tanks, water and wastewater treatment, etc. Up to $3 million is available per project. Local cost share is required, but disadvantaged communities may be eligible for a waiver. $15 million will be available for the 2011 funding cycle.[125]

DWR: Agricultural Water Conservation Program (Prop 13)[126]

Loans to agriculture under the Agricultural Water Conservation Program, created by Prop 13, are also available ($28 M total) under DWR's Water Use Efficiency financial assistance programs. These cover capital outlay and construction of up to $5 million per project for agricultural water conservation, including such activities as lining or piping of ditches; automating canal structures; improvements to water distribution system controls; tailwater or spill recovery systems; major improvements or replacement of leaking agricultural distribution systems; purchasing and installing water measurement devices; and capital improvements for on-farm irrigation. Irrigation application improvements are a key way to improving nitrogen application efficiency.

DWR: Infrastructure Rehabilitation Construction Grants (Proposition 13)[127]

[Fully Allocated]

The Infrastructure Rehabilitation Program is a grant program authorized under Prop 13 to provide assistance to poor communities for construction projects and feasibility studies to fix or replace failing water distribution systems that threaten the health, safety, and economy of these areas. The demand for funding far exceeded the amount of money allocated under Proposition 13. Since the program began in 2001, it received more than 71 proposals representing about $124 million during three funding cycles. Of these requests, it awarded $56.4 million for 22 feasibility studies and 20 construction projects. Presently, all program funds have been committed and the applications are closed.

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[124] http://www.water.ca.gov/wateruseefficiency/finance/
[125] California Financing Coordinating Committee. 2011. Funding Fair Handbook. Accessed December 2011 at: http://www.cfcc.ca.gov/res/docs/CFCC_Handbook_WebPosting.pdf
[126] http://www.water.ca.gov/wateruseefficiency/finance/
[127] http://www.grantsloans.water.ca.gov/docs/ircon/awardscon.pdf

Adminsitrative Record
Page 35712

California Infrastructure and Economic Development Bank (I-Bank)[128]

The California Infrastructure and Economic Development Bank (I-Bank) can issue tax exempt and taxable revenue bonds for a variety of projects which promote the revitalization of employment and the overall CA economy. One program, the Infrastructure State Revolving Fund (ISRF), provides low cost loans to public agencies for water supply, treatment and distribution projects. The ISRF funds $250,000 to $10 million per project for a 30 year loan period. Only agencies which are a subdivision of local government may apply; i.e. cities, counties, redevelopment agencies, special districts, assessment districts, joint powers authorities and non-profit corporations formed on behalf of a local government.

United States Department of Agriculture (USDA) Rural Utilities Service (RUS)[129]

The USDA Rural Utilities Service administers the Water and Environmental Programs (WEP) which supports drinking water, wastewater, and storm water facilities for rural communities of less than 10,000 people. Support includes loans, grants, and technical assistance, either directly to the facilities or indirectly through non-profit groups.

In the last eight years, RUS WEP provided more than $13 billion in loans and grants for rural water and waste infrastructure[130]. With additional funding in 2009 from the American Recovery and Reinvestment Act, RUS was able to invest $2.5 million under the Water and Waste Disposal Loan and Grant Program.[130]

The WEP has 7 main funding programs, discussed below.

USDA-RUS: Water and Waste Disposal Direct Loans and Grants

As part of the WEP, USDA-RUS manages the Water and Waste Disposal Direct Loans and Grants program. This program provides funds to public bodies and non-profit organizations to develop water and waste disposal systems in rural areas. Specifically, funds are limited to: construction, land acquisition, legal fees, engineering fees, capitalized interest, equipment, initial operation and maintenance costs, project contingencies, and any other cost needed to complete the project. Grants are given for up to 75% of eligible project costs when the service area MHI is less than 80% of the State MHI.

[128] http://www.ibank.ca.gov/infrastructure_loans.htm
[129] http://www.rurdev.usda.gov/UWEP_HomePage.html
[130] http://www.rurdev.usda.gov/SupportDocuments/Adelstein3-23-10.pdf

Adminsitrative Record
Page 35713

USDA-RUS: Water and Waste Disposal Guaranteed Loans

In addition to direct loans, guaranteed loans are also available for the construction or improvement of water and waste disposal projects. Qualifying applicants will be unable to obtain (at reasonable rates) the necessary credit without the guarantee. Public bodies and non-profit organizations benefit from these loans which are made and serviced by lenders such as banks, savings and loan associations, mortgage companies and other eligible lenders under the Guarantee Loan Program.

USDA-RUS: Water and Waste Revolving Fund Grants

Through the Water and Waste Revolving Fund Program, USDA-RUS issues grants to private non-profit corporations which have previously operated successful revolving loan funds to rural areas. These loan funds may only be issued to communities in rural areas for use with pre-development costs of water and wastewater projects or short-term small capital improvement projects not part of the regular operations and maintenance of current water and wastewater systems. A maximum of $100,000 may be made available per project.

USDA-RUS: Individual Household Water Well Program

Additional grants are provided by USDA-RUS to private non-profit corporations through the Individual Household Water Well Program. These grants establish loan programs for individuals who need capital to construct, refurbish, or service their well system. The maximum loan amount per well is $8,000.

USDA-RUS: Predevelopment Planning Grants

Grants of up to $15,000 or 75% of the total project cost (whichever is lower) may be awarded to communities or water systems for assistance in preparing a water project application. Eligible applicants must provide proof that they are unable to pay the necessary predevelopment costs.

USDA-RUS: Emergency Community Water Assistance Grants

Communities which face or expect to face an abrupt and severe decrease in water quality may apply to USDA-RUS for an emergency community water assistance grant. Typical events which cause the decline in water quality are: a natural disaster such as drought, earthquake, flood, tornado, or hurricane; a disease outbreak; or a chemical spill/leakage/seepage. Projects which relieve a decline in the quantity and quality of drinking water through the construction of a new water source and/or treatment plant are eligible for full project funding up to $500,000. Communities facing emergency repairs or replacements of facilities on existing systems due to acute concerns (e.g. washed out river crossing in a distribution system, or construction of distribution lines to individuals not currently on the system, whose wells have gone dry) may apply for up to $150,000 for distribution waterline extensions, breaks or repairs on distribution waterlines, and operation and maintenance.

USDA-RUS: Technical Assistance and Training Grants

USDA-RUS manages a technical assistance and training program which funds the educational and technical capacity building programs of private non-profit and tax-exempt organizations. Applicable to the study area, organizations receiving funding for these activities are: RCAP, the National Rural Water Association, and the National Drinking Water Clearinghouse. These individual programs are described in Appendix B: Summary of Current Planning and Regulatory Programs that Address Nitrate in Groundwater. In addition to technical assistance efforts, these non-profit organizations may also use the funding to assist communities in preparing funding applications for water or waste projects.

US Department of Housing and Urban Development (HUD): Community Development Block Grant Program[131]

The Community Development Block Grant (CDBG) program, administered by the U.S. Department of Housing and Urban Development, provides grants to States to fund economic development, housing, public improvements, public services, and administrative and planning. Within the public improvement category, funds are available to construct or improve community water and sewer systems, build technical capacity, and assist nonprofit organizations who aid in community development.

HUD has the responsibility to ensure that the States are following Federal regulations and policies. The majority of the program responsibilities fall on the States, including: determining how to allocate funds among communities, formulating community development goals, and ensuring that at least 70 percent of its CDBG grant funds are used to benefit low- and moderate- income persons.

The CDBG program funds cities with less than 50,000 people and counties less than 200,000. The State of California may use up to $100,000 plus 50% of their incurred administrative costs per project. Up to 3% of California's total allocation from HUD may be used on technical assistance. In 2010, HUD provided almost $500 million for CDBG programs in California.

U.S. Economic Development Administration (EDA)[132]

The U.S. Economic Development Administration (EDA) is an agency within the U.S. Department of Commerce. The EDA's Public Works and Economic Development Program provides grants for the construction, expansion, or upgrade of infrastructure (including water and wastewater) in communities facing economic distress, natural disasters, or the depletion of natural resources. The goal of this program is to create and sustain long-term private sector job opportunities in distressed communities. The funds may be used for land acquisition, construction, renovation, expansion, improvement, or

[131] http://www.hud.gov/offices/cpd/communitydevelopment/programs/stateadmin/
[132] http://www.eda.gov/InvestmentsGrants/Investments.xml

Adminsitrative Record
Page 35715

design of a public works facility. The grants cover up to 50 percent of project costs. During 2009, EDA awarded $13 million for public works projects across the United States.

EDA also offers technical assistance to public and nonprofit groups which work with communities on project planning and feasibility studies. During 2009, EDA awarded $135,000 for technical assistance projects in California.

Rural Community Assistance Corporation (RCAC)[133]

The Rural Community Assistance Corporation (RCAC), based in California, is RCAP's western regional partner. They focus on areas with populations of 2,000 or less, minority communities, and disadvantaged communities. Their projects help improve access to safe drinking water supplies, develop and maintain wastewater systems, protect the groundwater, and improve access to financial assistance resources. Their overarching goal is to help small water and wastewater systems to build the technical, managerial, and financial capacity necessary to comply with State and Federal regulations. Often, this assistance is financed by State and Federal contracts, and is thus free to the community.

RCAC also administers $1.2 million per year from the US EPA for water and wastewater construction loan funds. As of September 30, 2010, they have supported over 46,000 individual water and wastewater connections.

RCAC's New Mexico office has initiated and managed the regionalization of a few small water systems into larger organizations with more technical, managerial, and financial capacity. Recently, five New Mexico small drinking water systems (totaling 8,000 people) merged to form the Lower Rio Grande Public Water Works Authority Agency. Their model could be used in California to help solve the deficiencies in functioning and capacity of small community water and wastewater systems.

The Housing Assistance Council (HAC)[134]

The Housing Assistance Council administers the Small Water/Wastewater Loan Fund nationally at interest rates equal to or below the market rate. Small short-term loans of $100,000 to $250,000 are made to local nonprofits, for profits, and government entities that are developing housing for lower income rural communities. These loans can be used to finance predevelopment, land acquisition, site development, and construction phases of a water or wastewater infrastructure project.

[133] http://www.rcac.org
[134] http://www.ruralhome.org

Cooperative Bank (CoBank)[135]

The Cooperative Bank (CoBank) offers a national Water and Wastewater Loan Program for communities of fewer than 20,000 people. These loans, typically around $1 million, are used to finance new water and wastewater infrastructure projects, system improvements, water right purchases, and system acquisitions. Smaller loans of $50,000 to $500,000 are also offered to help cover initial construction costs.

[135] http://www.cobank.com

Adminsitrative Record
Page 35717