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

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Tulare Lake Basin and Salinas Valley Pilot Studies

Prepared for:

California State Water Resources Control Board

July 2012

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Suggested Citation:

Viers, J.H., Liptzin, D., Rosenstock, T.S., Jensen, V.B., Hollander, A.D., McNally, A., King, A.M., Kourakos, G., Lopez, E.M., De La Mora, N., Fryjoff-Hung, A., Dzurella, K.N., Canada, H.E., Laybourne, S., McKenney, C., Darby, J., Quinn, J.F. & Harter, T. (2012) Nitrogen Sources and Loading to Groundwater. Technical Report 2 in: *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*. Center for Watershed Sciences, University of California, Davis.

An electronic copy of this Final Report is available from the following website:

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Acknowledgments

The authors acknowledge that much of this work would not have been possible without the significant data and model compilations performed as part of the California Nitrogen Assessment (http://nitrogen.ucdavis.edu) supported by the David and Lucile Packard Foundation. The authors are also grateful to the following individuals for providing extensive insight, knowledge, data, time and advice in helping make this technical report possible: Adam Asquith, Lisa Babcock, Daniel Benas, Jacquelyn Bjorkman, Bret Boatman, John Borkovich, Scott Bradford, Beverly Briano, Jim Butler, Marsha Campbell Mathews, Paul Charpentier, Ron Cole, Carol Collar, Dawn Collings, Mark Commandatore, David Crohn, Maria de la Paz, John Dickey, Danielle Dolan, Erik Ekdahl, Graham Fogg, Lauren Fondahl, Cindy Fosi, Wayne Fox, Carol Frate, Rob Gailey, Sarge Green, Tom Hardy, Michael Hickey, Glenn Holder, Jim Holiday, Chris Johnson, Mary Kaneshiro, Jonathan Kaplan, Matthew Keeling, William LaBarge, Carol LaRussa, John Letey, Harold Leverenz, Victor Lopez, Michael Louie, Michael McGinnis, Zachary Meyer, Gretchen Miller, Cynthia Moffett, Erin Mustain, Gavin O'Leary, Doug Parker, Clay Rodgers, James Ross, Nathaniel Roth, Yoram Rubin, Blake Sanden, Cheryl Sandoval, Rosa Staggs, Sonja Thiede, Jim Thorne, John Troiano, Darren Verdegaal, Frank Wendland, Dennis Westcot, Craig Wolff, Xiaoming Yang, and Janice Zinky. The contents of this document are solely the responsibility of the authors and do not necessarily represent the views of the individuals contacted, nor the official views of supporting agencies or supporting organizations.

Contributions: JHV, DL, TSR, and TH edited this report. JHV, DL, TSR, VBJ, ADH, JD, JFQ and TH conceptualized and designed portions of the study. JHV, DL, TSR, VBJ, ADH, AMcN, AMK, GK, NDLM, AFH, HC, SL, CMcK, and TH produced data tables, conducted analyses and synthesized results. VBJ, ADH, AMcN, AMK, NDLM, AFH, and TH developed figures and maps. VBJ, NDLM, HC, SL, and CMcK performed surveys. JHV, DL, TSR, and TH wrote portions of Section 1. JHV, DL, EML, and TH wrote portions of Section 2. JHV, DL, TSR, ADH, AM, AFH, KND, JFQ and TH wrote portions of Section 3. TH, AMK, and GK wrote portions of Section 4. DL and JHV wrote portions of Section 5. VBJ, AMK, NDLM and TH wrote portions of Section 6. DL and TH wrote portions of Section 7. DL and JHV wrote portions of Section 8. TH, NDLM, and AMK wrote portions of Section 9.

This work was funded by the State Water Resources Control Board under agreement number 09-122-250.

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

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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

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

NPDES	National Pollutant Discharge Elimination System
N_r	reactive nitrogen
N_{runoff}	N in surface runoff
$N_{TotalNorm}$	totalized recommended fertilizer application
$N_{WWTP\text{-}FP}$	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

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

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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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

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.

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):

- 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).

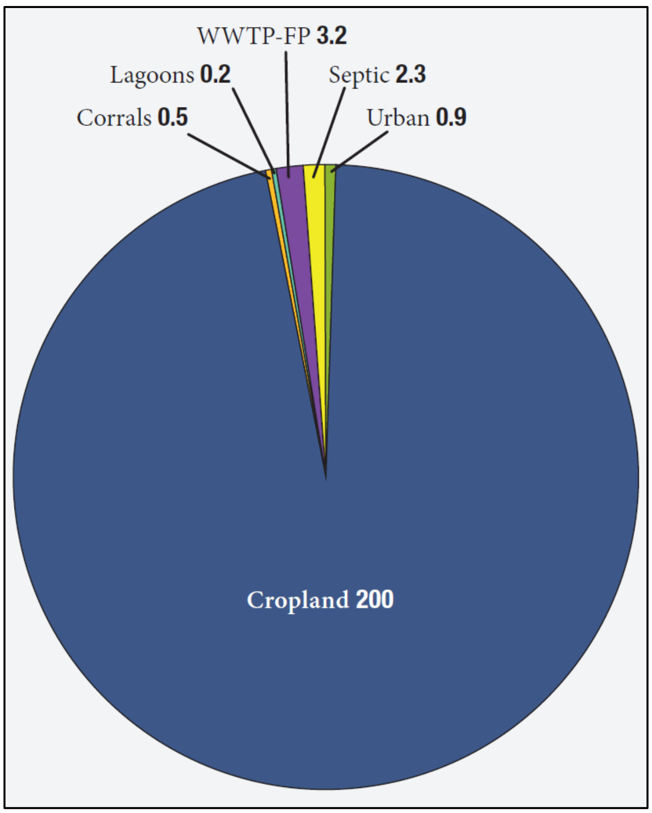

Figure 1. Estimated groundwater nitrate loading from major sources within the Tulare Lake Basin and Salinas Valley, in Gg nitrogen per year (1 Gg = 1,100 t).

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;

- 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.

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 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

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.

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

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[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).

(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.

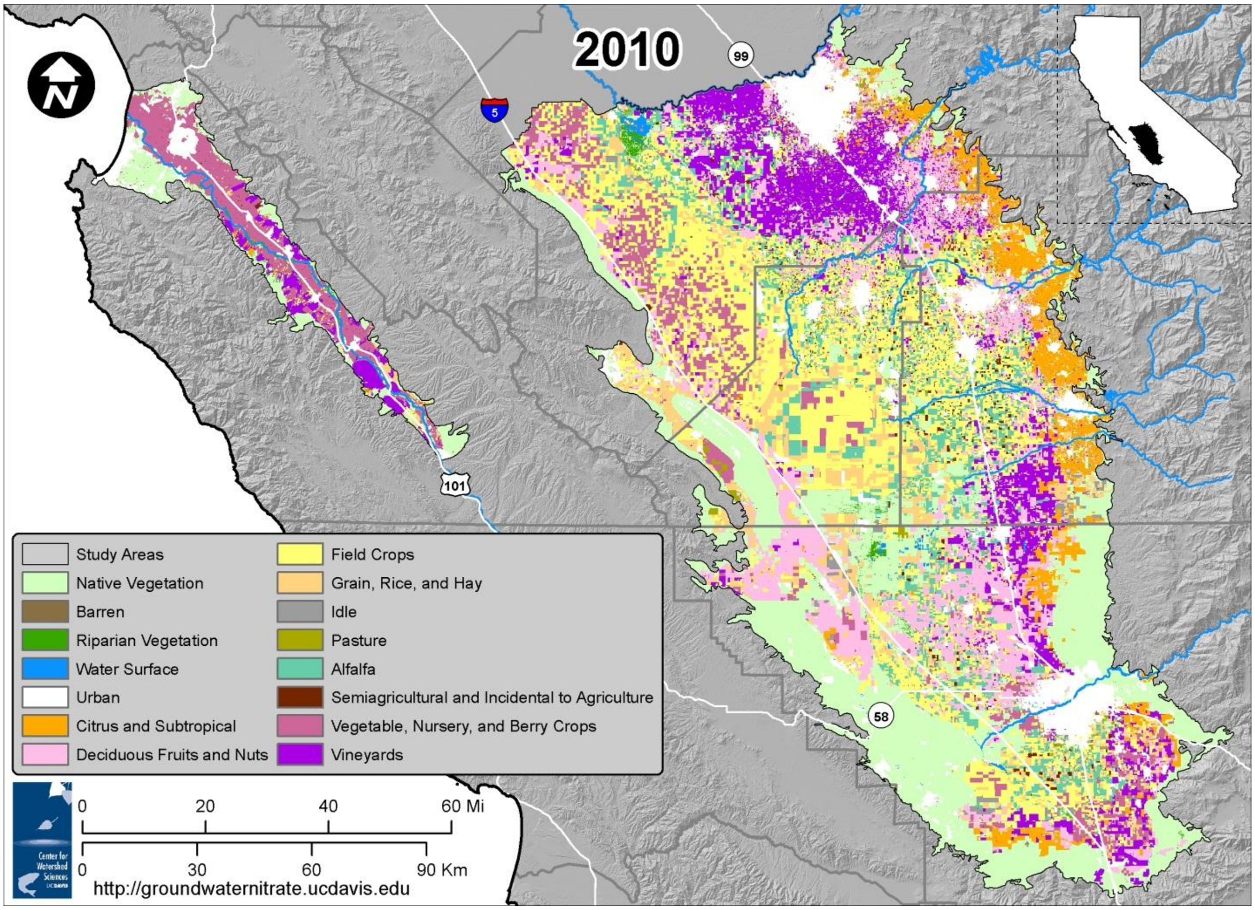

Figure 2. The Tulare Lake Basin (TLB) and Salinas Valley (SV) are the focus of this study. The study area represents 40% of California's diverse irrigated agriculture and more than half of its confined animal farming industry. It is home to 2.6 million people, with a significant rural population in economically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

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 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.

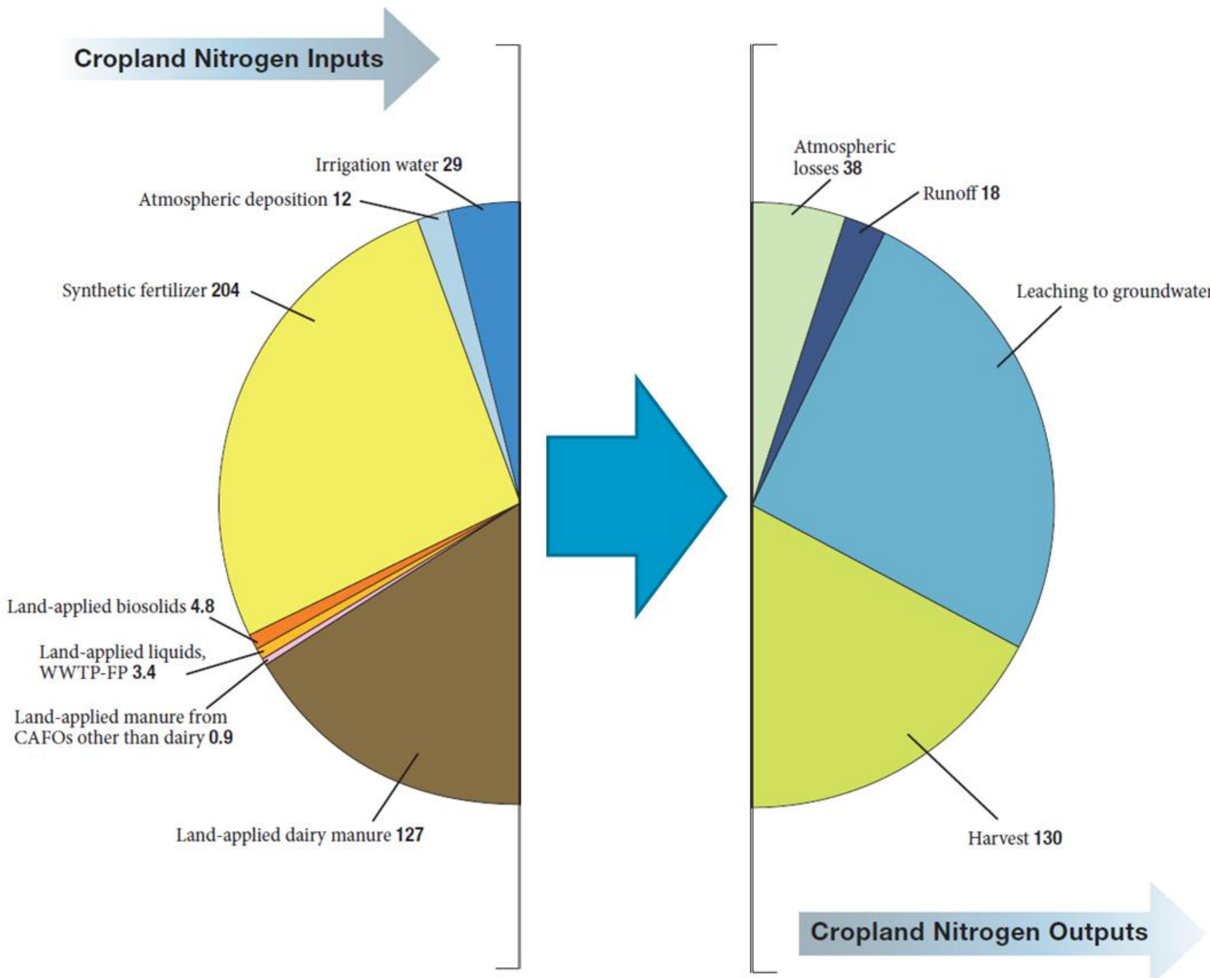

Figure 3. Overview of cropland input and output (Gg N/yr) in the study area (Tulare Lake Basin and Salinas Valley) in 2005. The left half of the pie chart represents total nitrogen inputs to 1.27 million ha (3.12 million ac) of cropland, not including alfalfa. The right half of the pie chart represents 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).

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).

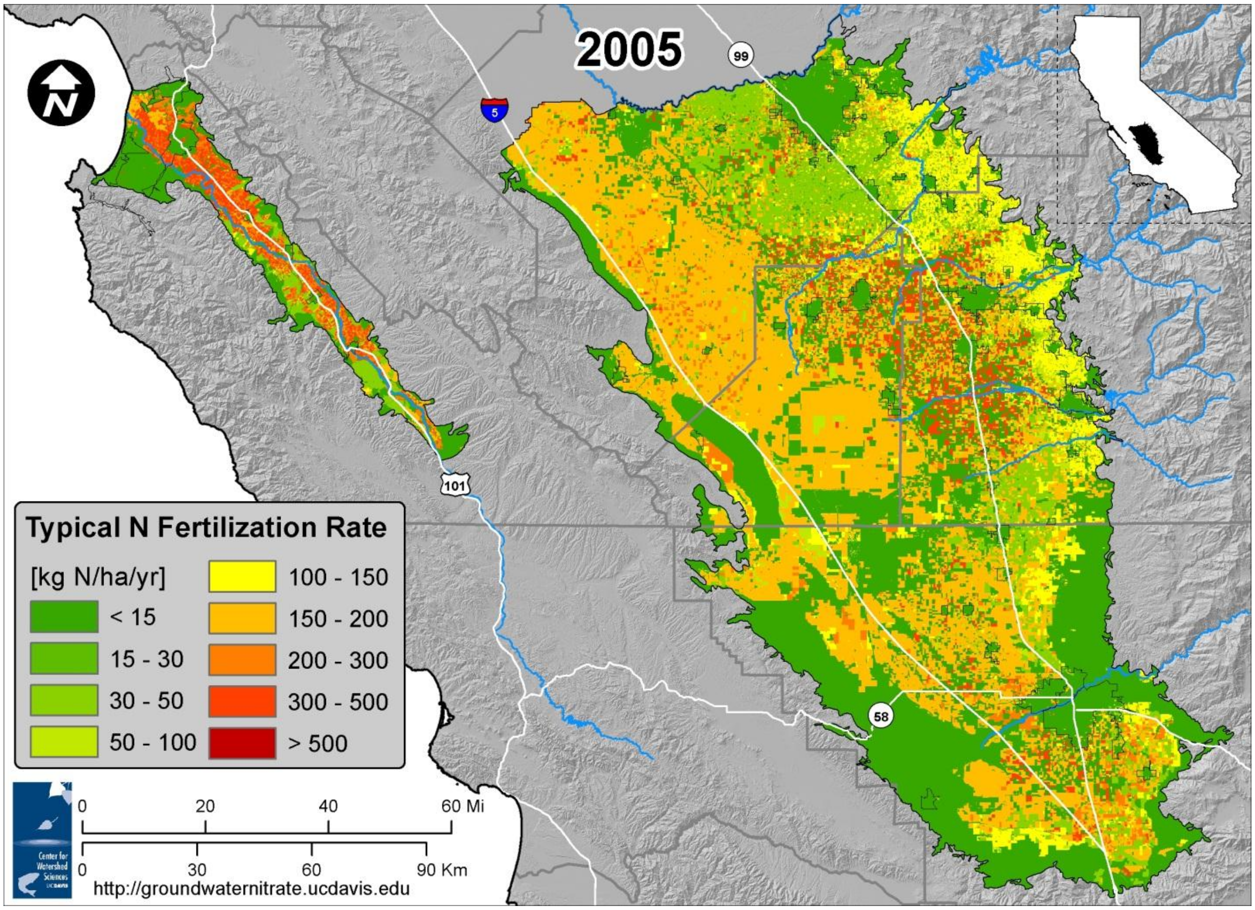

Figure 4. Current typical annual fertilization rates (1 kg/ha/yr = 1.1 lbs/ac/yr) in irrigated agricultural cropland of the study area derived from the literature, United States Department of Agriculture (USDA) Chemical Usage Reports, and agricultural cost and return studies for each of 58 crop categories (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 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).

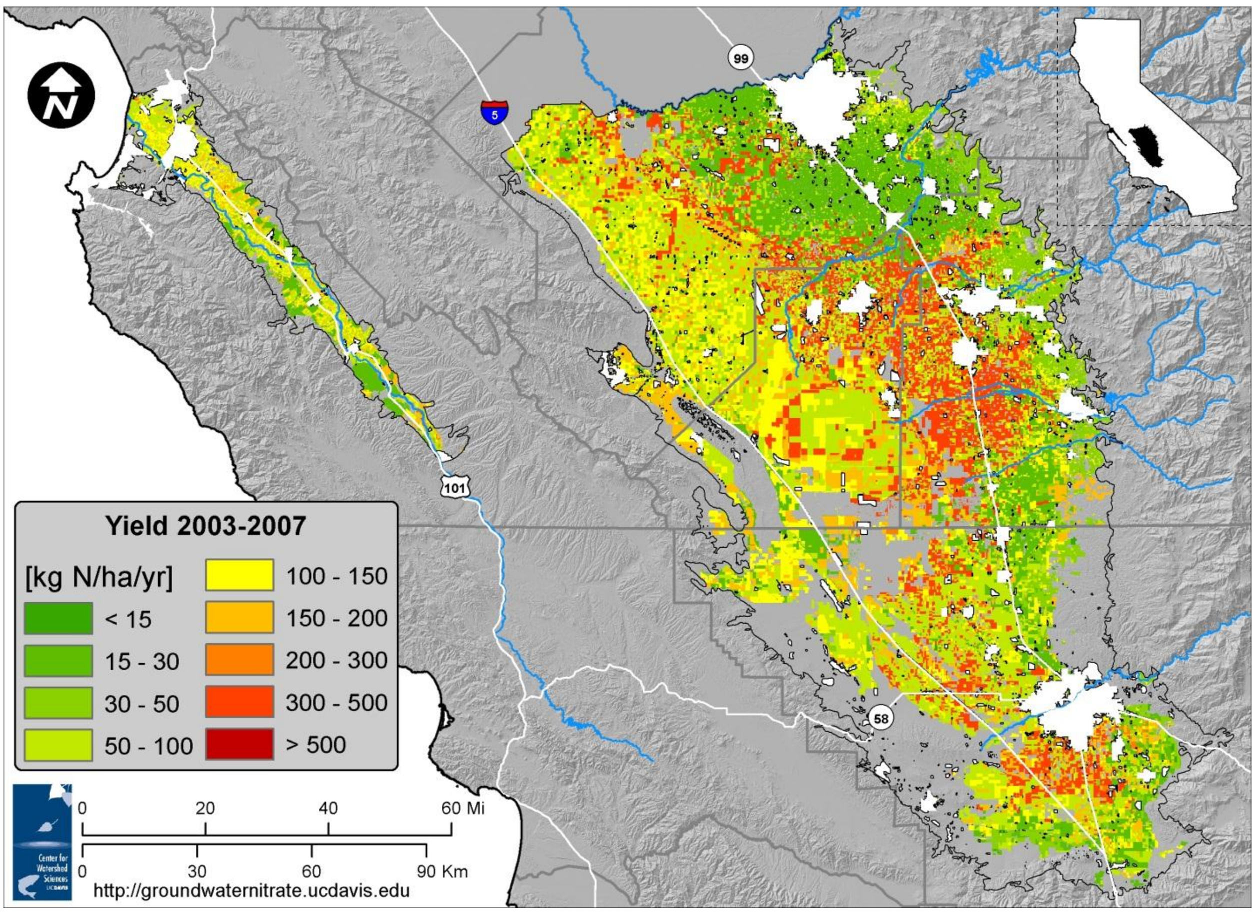

Figure 5. Current annual nitrogen removal rate in harvested materials (1 kg/ha/yr = 1.1 lbs/ac/yr) derived from county reports of harvested area and harvested tonnage for each of 58 crop categories. Rates account for multi-cropping in some vegetable crops and double-cropping of corn and winter grain.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

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.

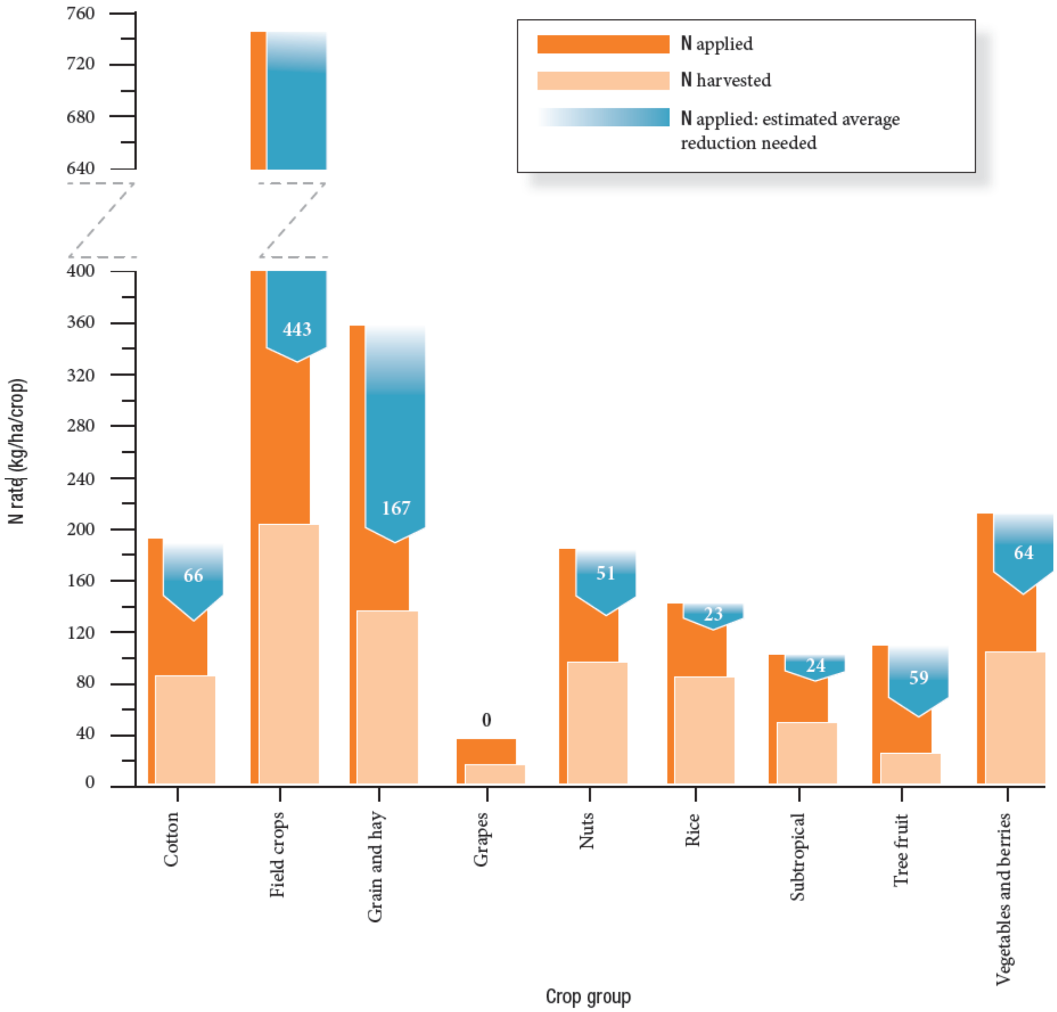

Figure 6. Nitrogen application reduction needed to reduce groundwater nitrate loading to less than 35 kg N/ha/crop, compared with average nitrogen applied (synthetic fertilizer and manure) and nitrogen harvested (all units in kg N/ha/crop). Rates are given per crop, and the required reduction does not 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.

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	Manure Appli-cation	Land Applied Effluent and Biosolids	Harvest	PNB[1]	PNB_0 [2]	Ground-water Loading	Ground-water Loading Intensity
	1,000 t N/yr	1,000 t N/yr	1,000 t N/yr	1,000 t N/yr	%	%	1,000 t N/yr	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

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.

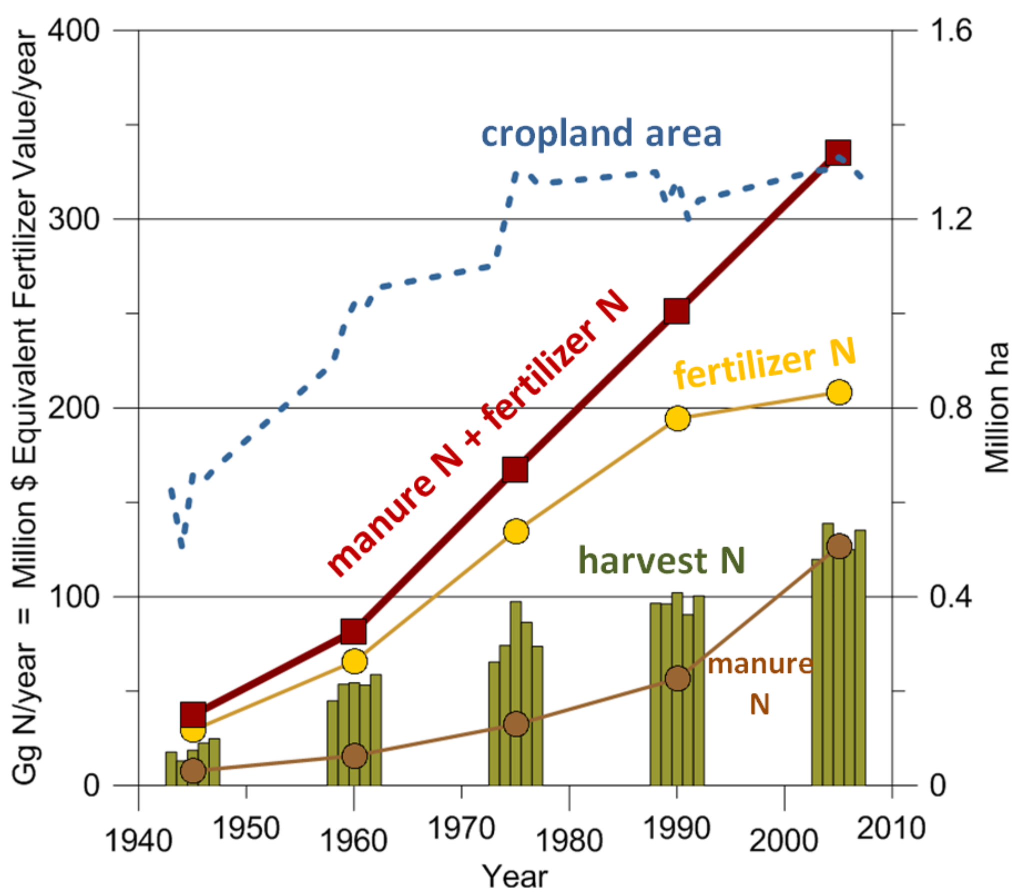

Figure 7. Estimated historical agricultural development in the study area (not including alfalfa): total harvested area, total harvested nitrogen in fertilized crops, fertilizer applied to cropland (5 year average), manure applied to cropland (5 year average), and sum of manure and fertilizer applied 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.

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

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**

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).

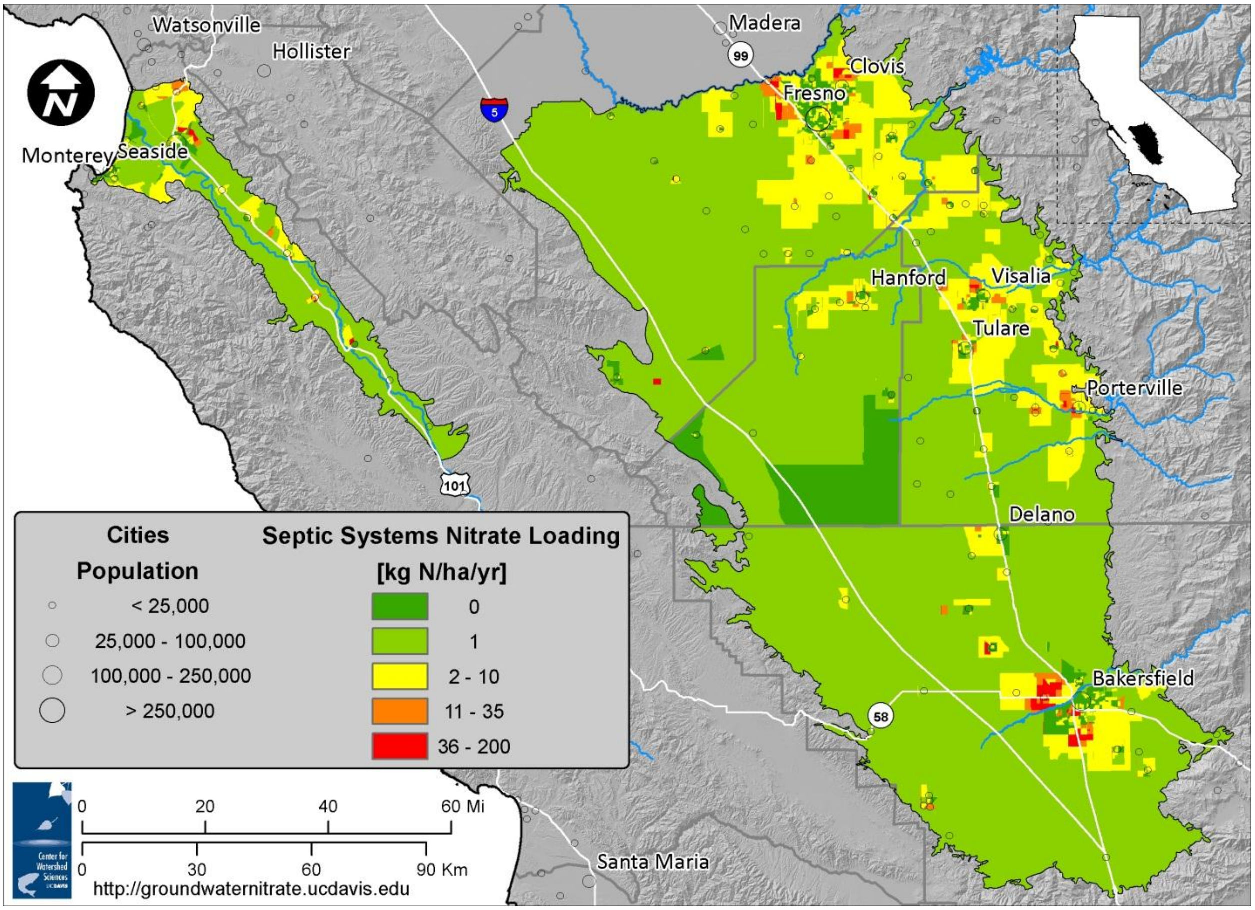

Figure 8. Septic-derived nitrate leaching rates within the study area.

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).

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

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[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,

(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

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).

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.

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 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 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 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.

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 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.

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.

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).

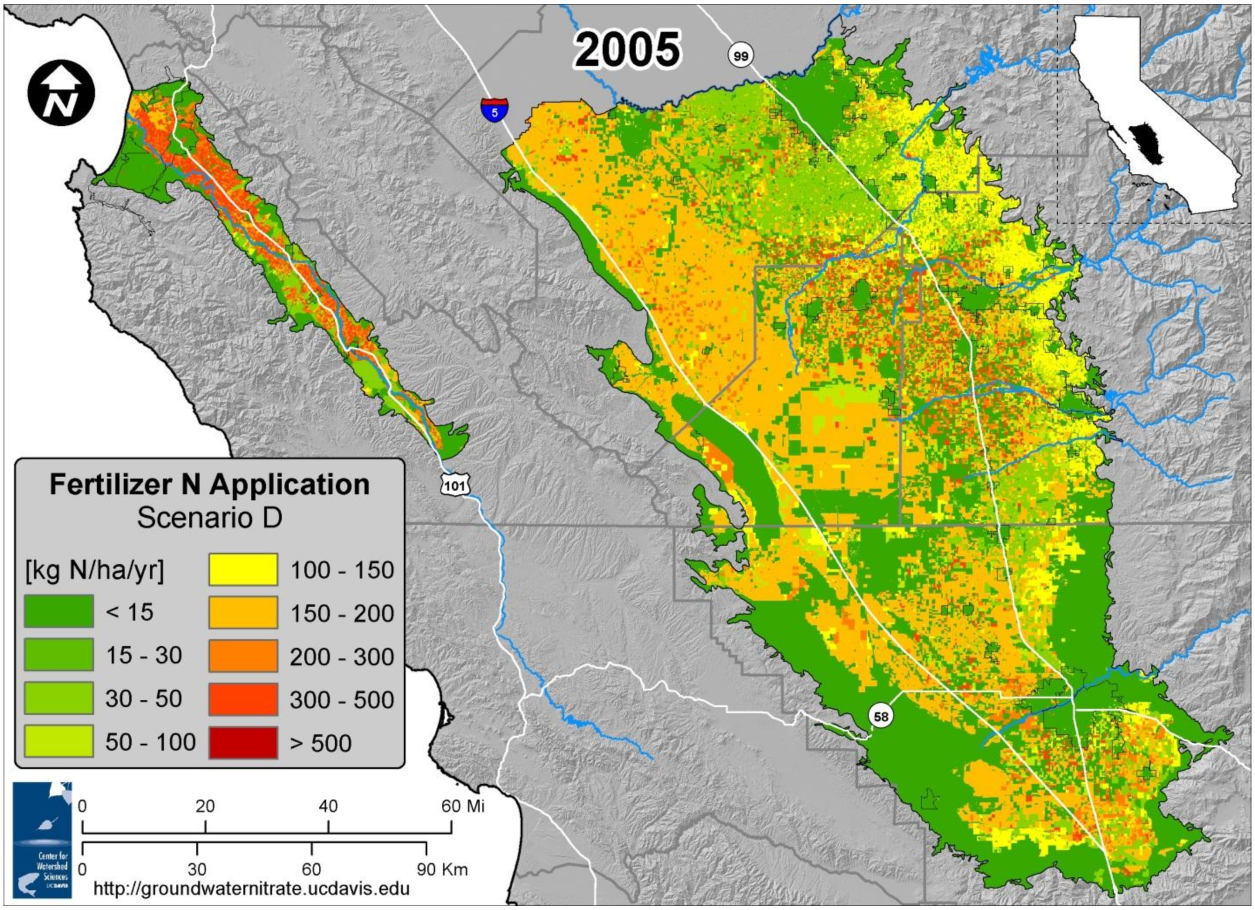


Figure 9. Simulated synthetic fertilizer application on cropland, including alfalfa, after accounting for the application of manure as fertilizer on dairy cropland (manure may make up to 50% of the applied nitrogen need). This map represents results for "Scenario D" (no manure exports from dairies). 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.

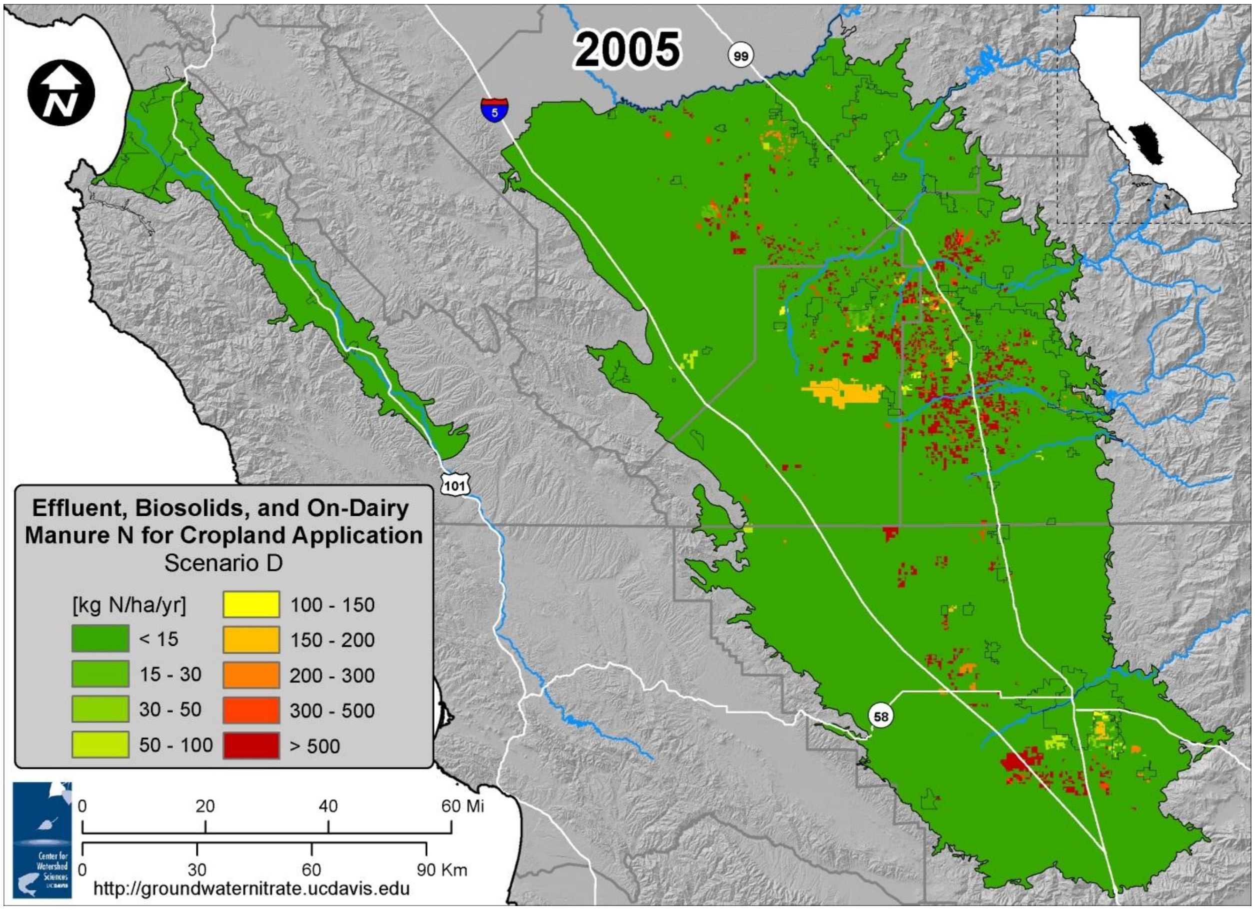


Figure 10. Simulated land applied N from dairy manure (land applied on dairy cropland), and from WWTP/FP effluent and biosolids. This map represents results for the hypothetical "Scenario D" (all manure is land applied on dairies, representing 62% of animal N excretion, no manure is exported from dairies).

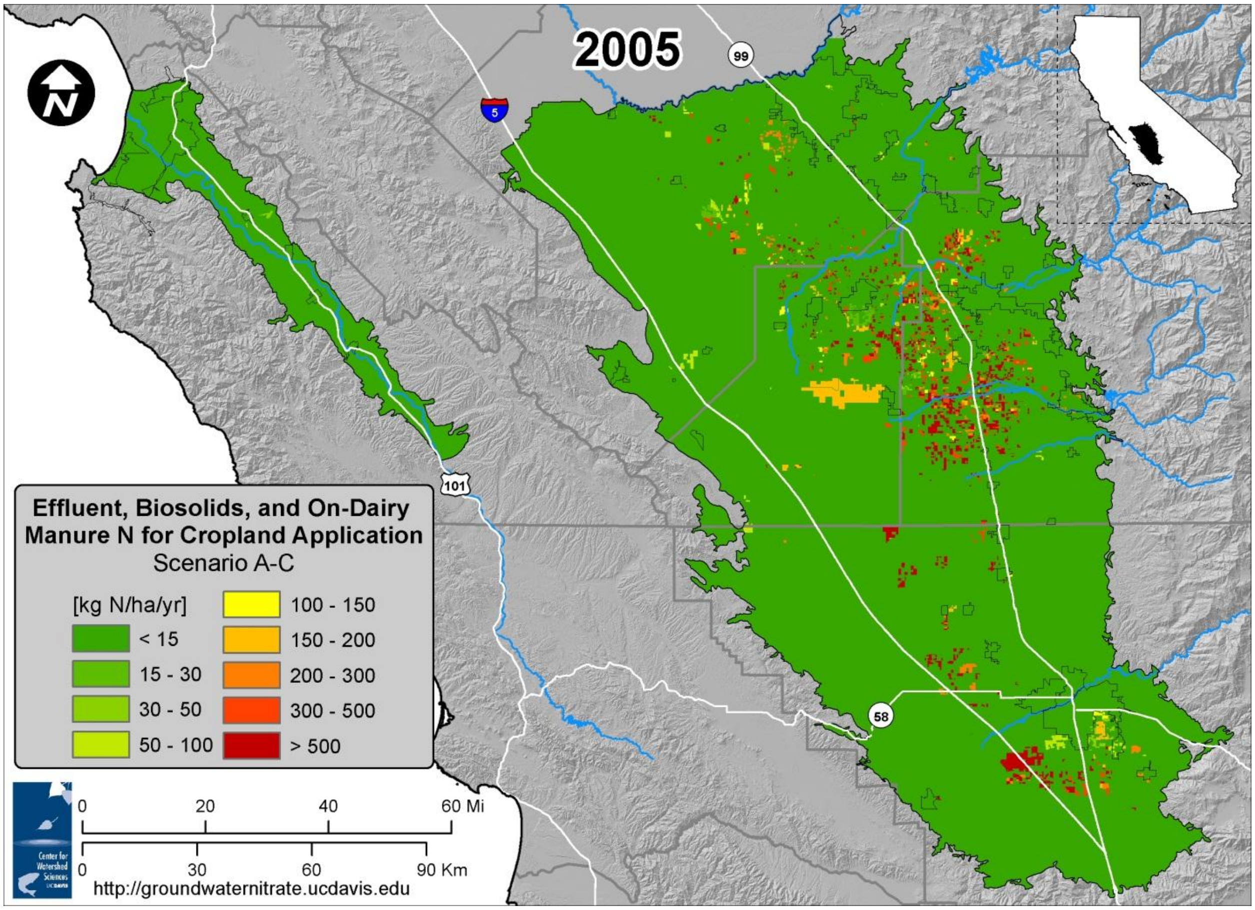


Figure 11. Simulated land applied N from dairy manure (showing only land applied N on dairy cropland, but not exported dairy manure N applications), and from WWTP/FP effluent and biosolids. This map represents results for Scenario A-C (a total of 24% of animal N excreted is land applied on dairies).

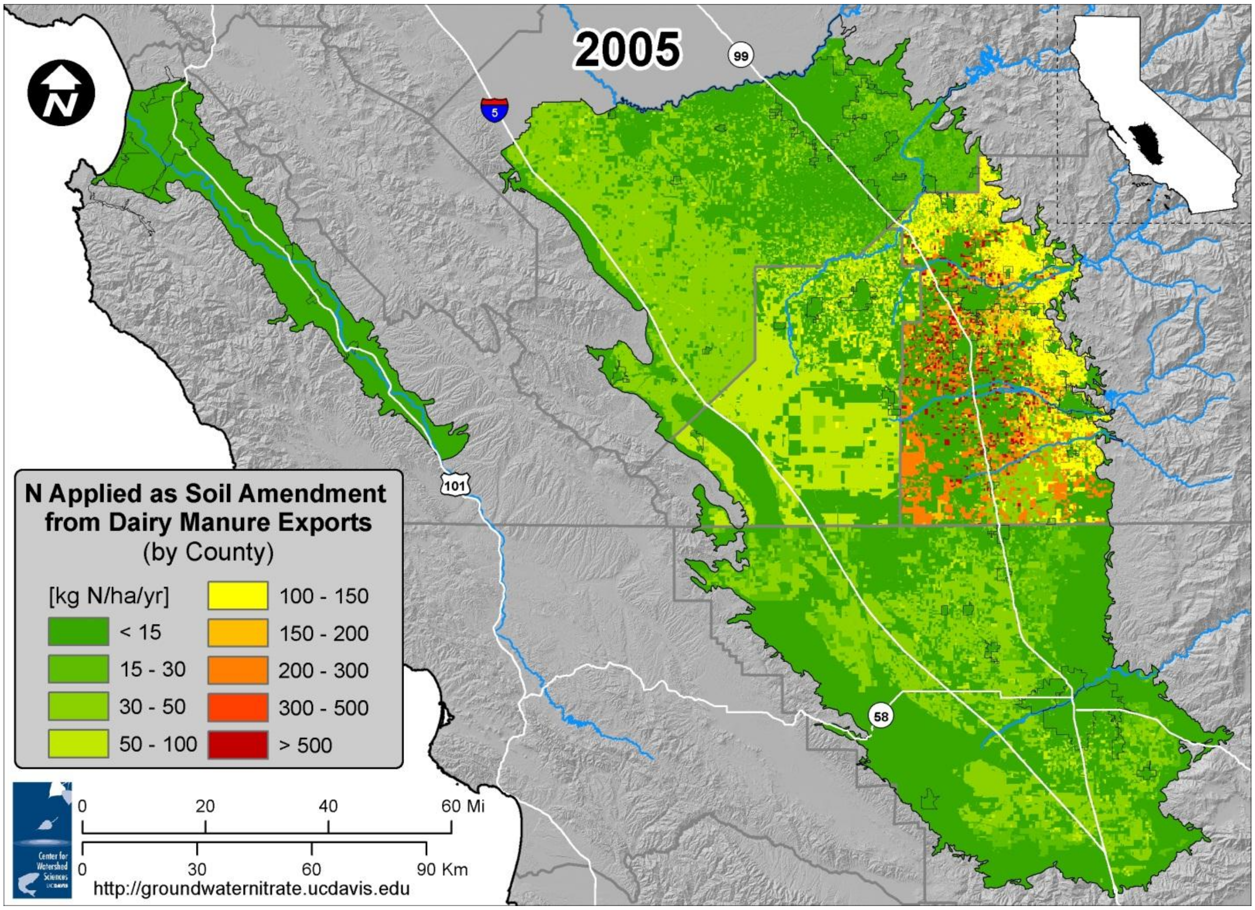


Figure 12. Simulated land application of dairy manure N exported from dairies and land applied within the county of origin at rates proportional and in addition to typical N fertilizer needs in each crop (see Figure 4). Total manure exports represent 38% of animal N excreted ("Scenario C (by County)").

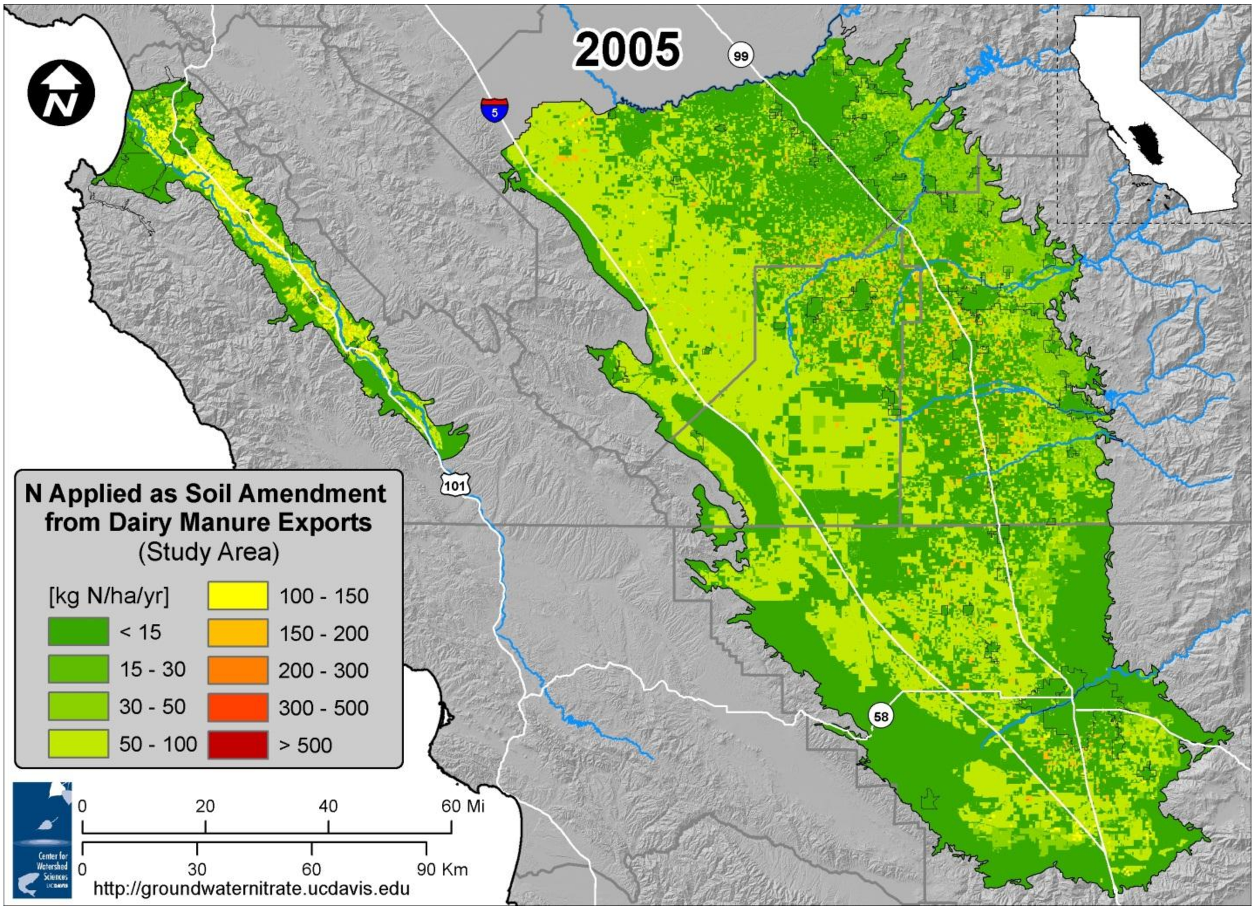


Figure 13. Simulated land application of dairy manure N exported from dairies and land applied across the study area (not restricted to the county of origin) at rates proportional and in addition to typical N fertilizer needs in each crop (see Figure 4). Total manure exports represent 38% of animal N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

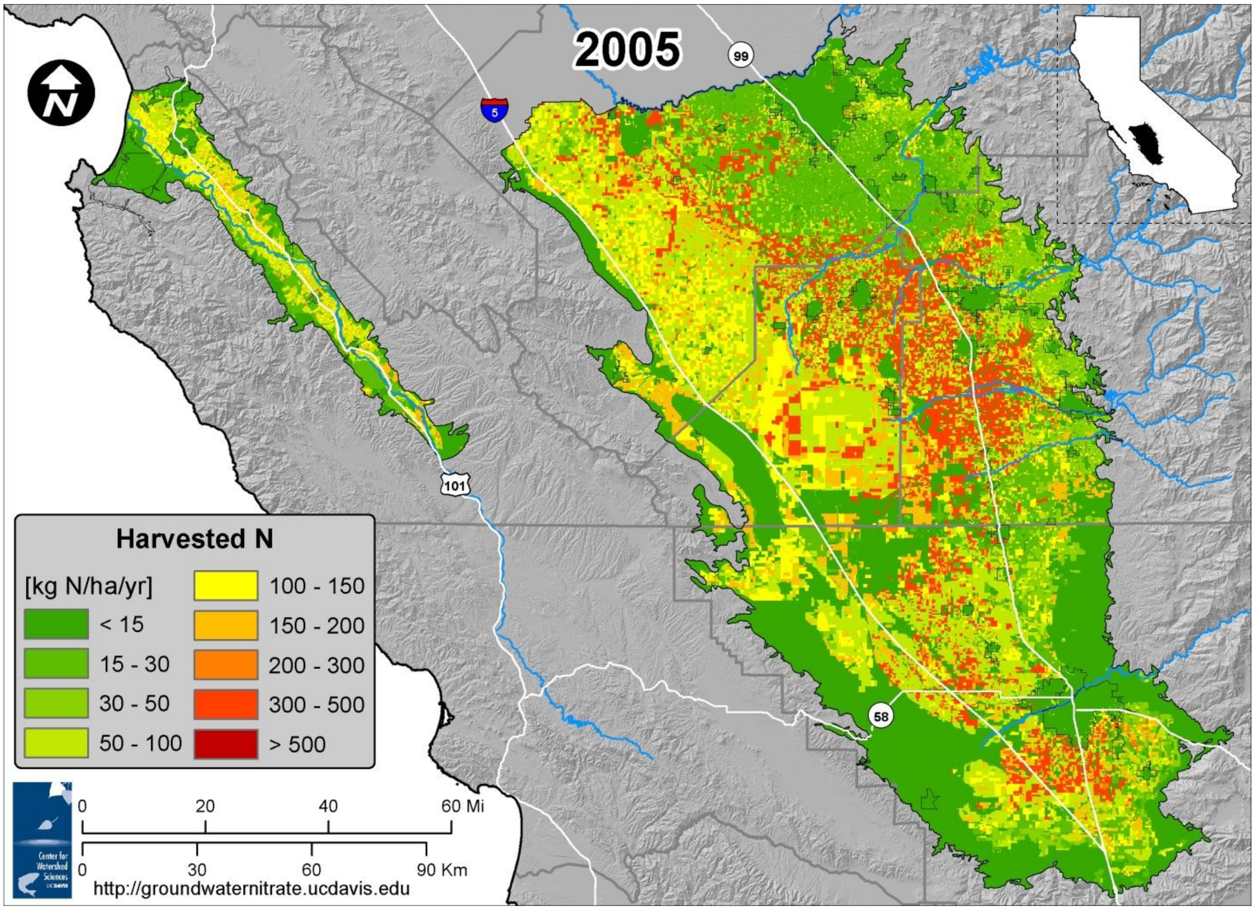


Figure 14. Simulated nitrogen harvested in 2005 from all cropland including alfalfa. The simulation assumes 1.7 annual crops in seven vegetable crops (broccoli, brussels sprouts, cabbage, cauliflower, celery, lettuce, spinach) and double-cropping of all corn acreage with winter grain.

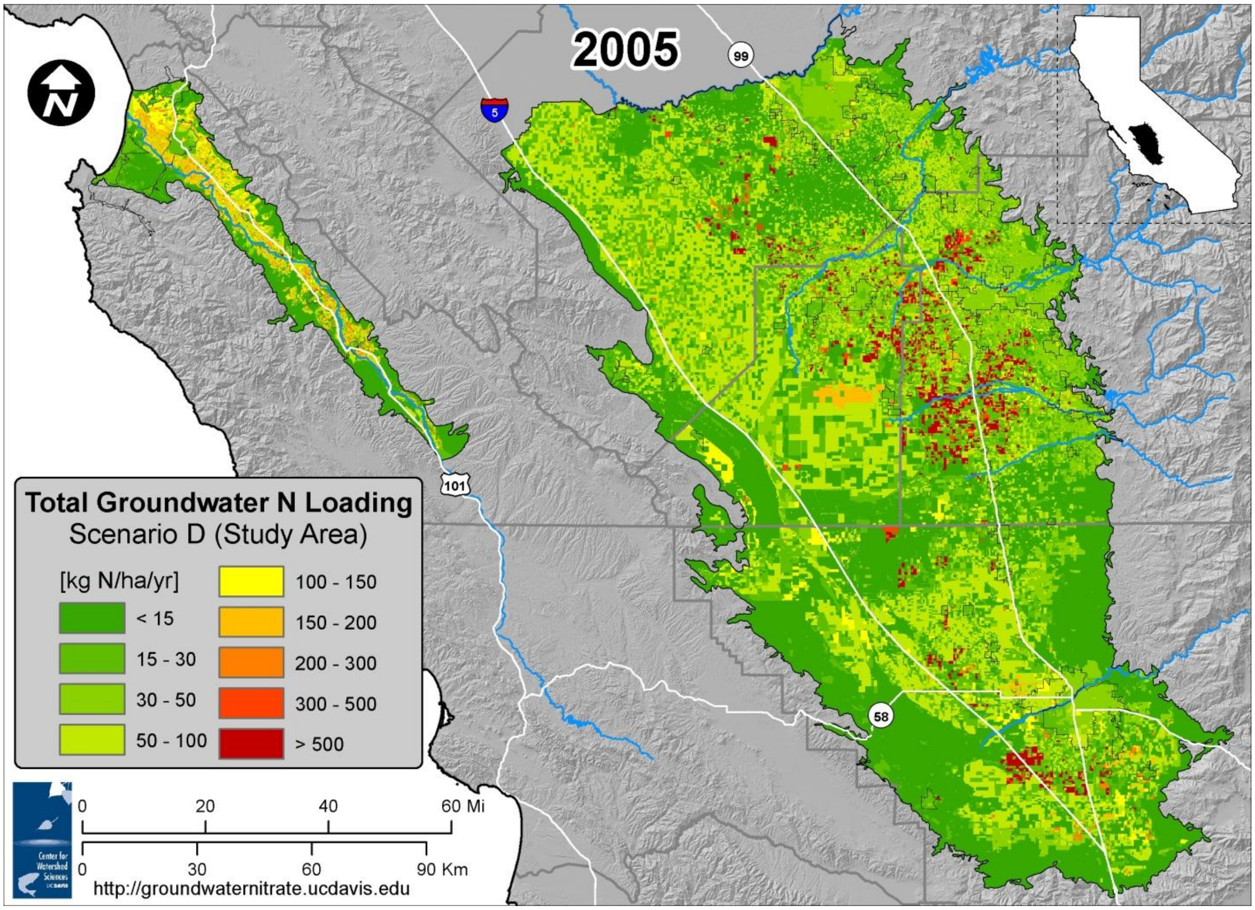


Figure 15. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario D": All land applied manure N (62% of animal N excreted) is applied to corn, grain, and other field crops (not including alfalfa) under the direct management of dairies (no manure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

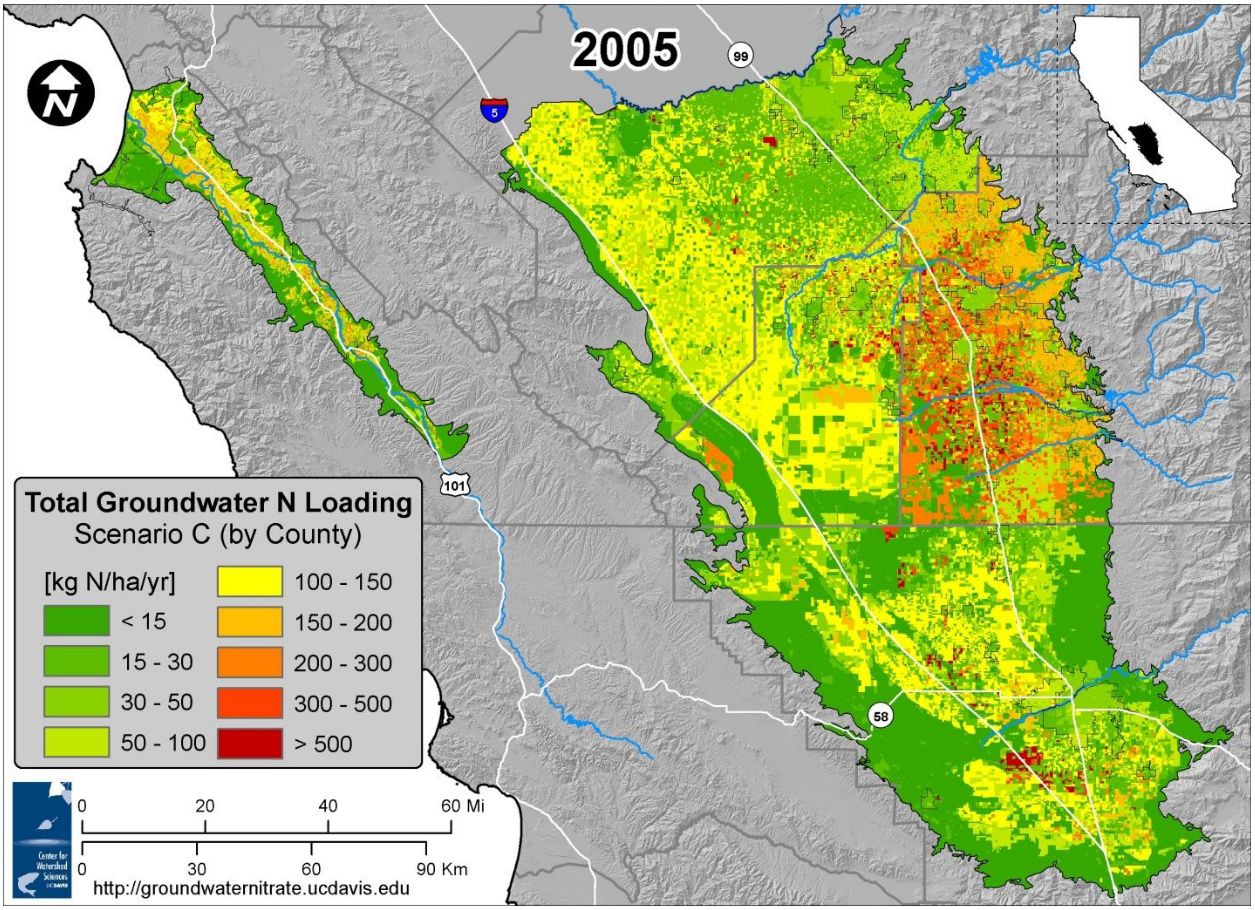


Figure 16. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario C (by county)": hypothetically exported manure N from dairies (38% of animal N excretion) is land applied as soil amendment within the county of origin. Exported manure N applications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

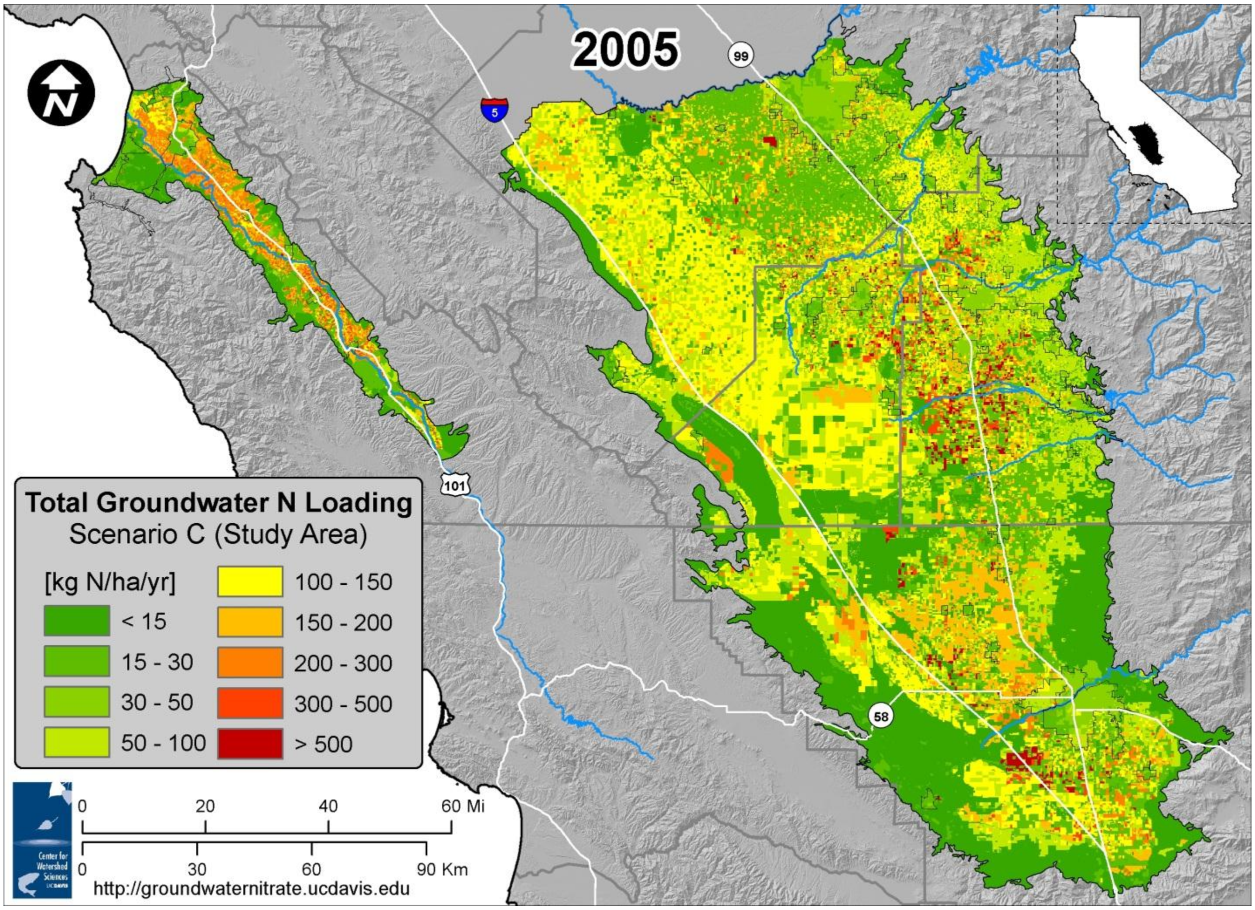


Figure 17. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario C (study area)": hypothetically exported manure N from dairies (38% of animal N excretion) is land applied as soil amendment across the study area (not restricted to the county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

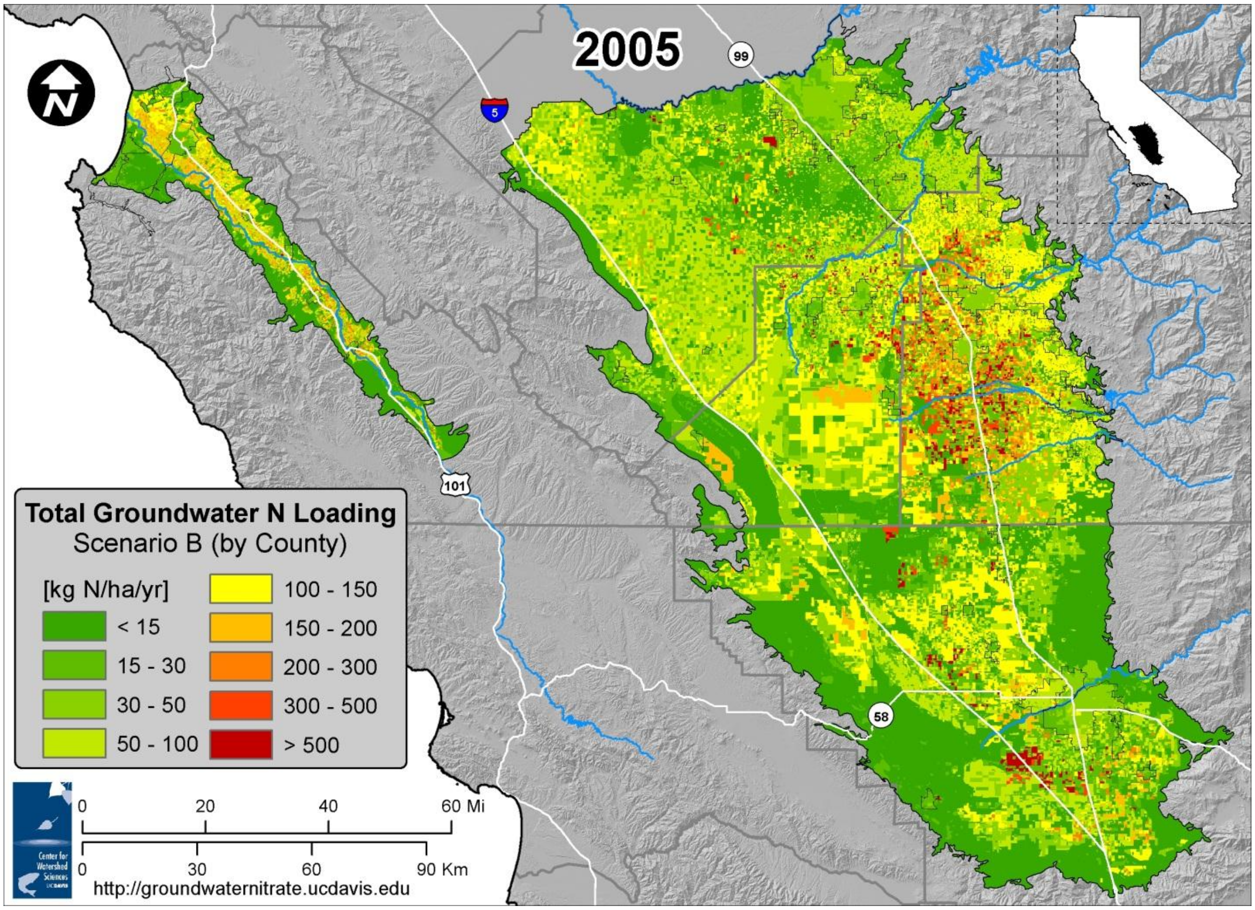


Figure 18. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario B (by county)": half of the hypothetically exported manure N from dairies (19.5% of animal N excretion) is land applied as soil amendment within the county of origin. These manure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

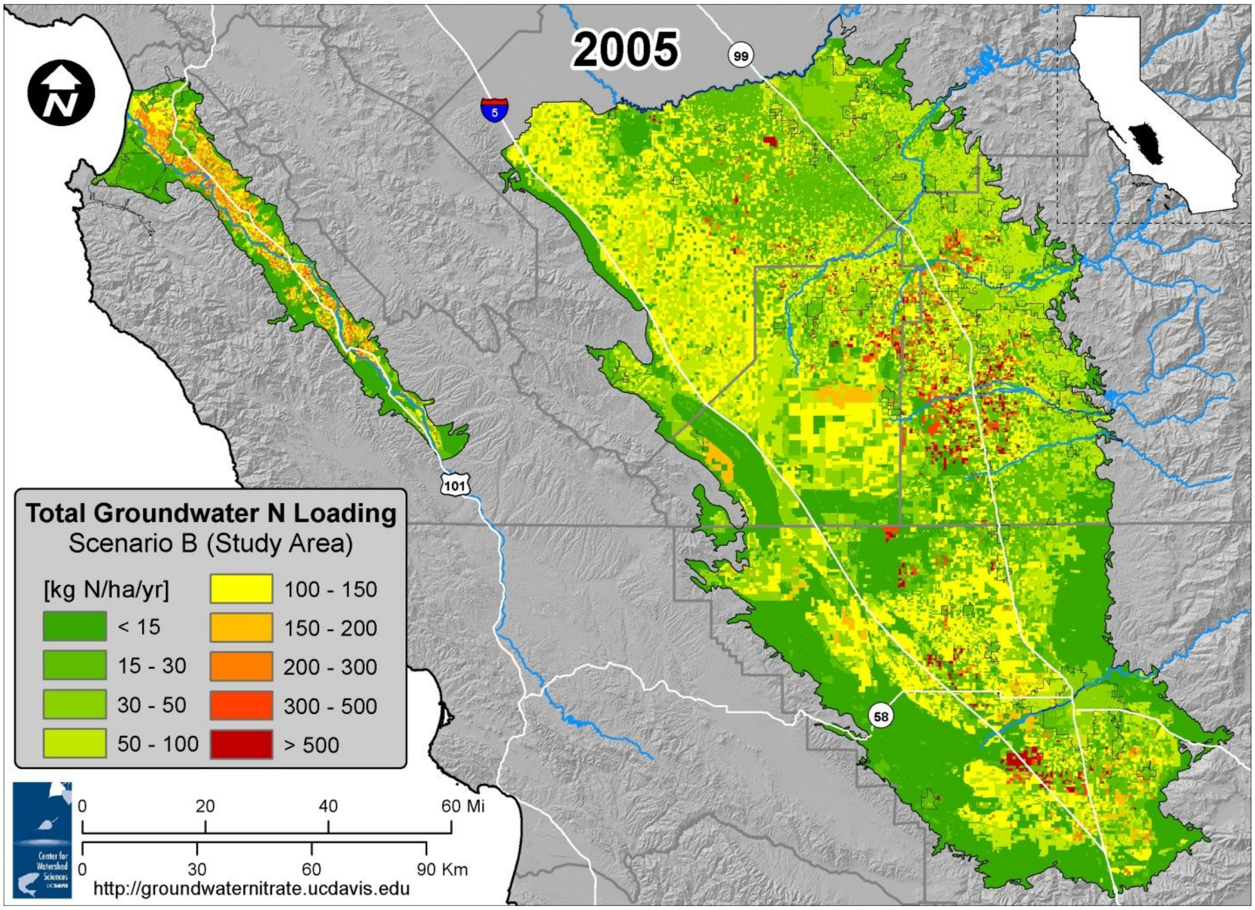


Figure 19. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario B (study area)": half of the hypothetically exported manure N from dairies (19.5% of animal N excretion) is land applied as soil amendment across the study area (not restricted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

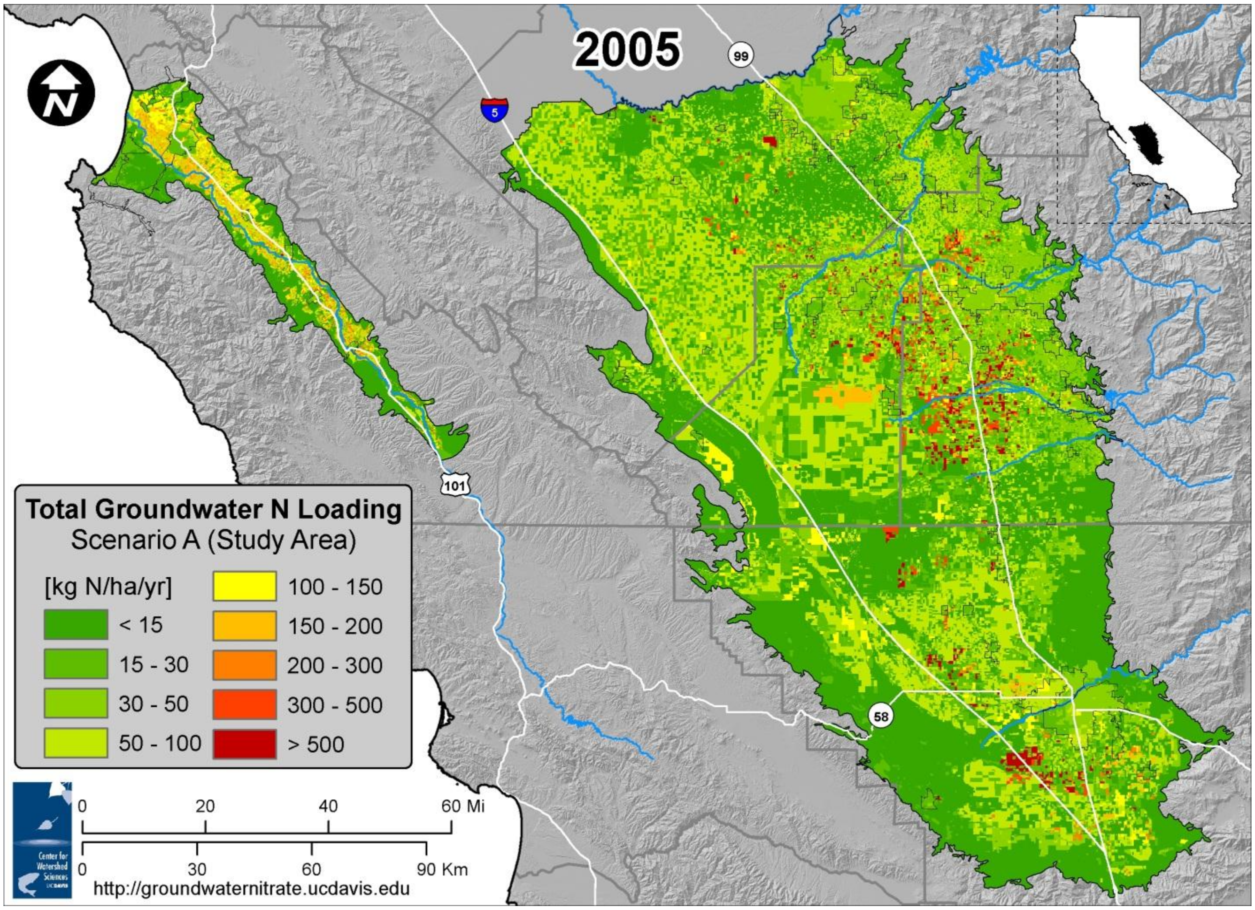


Figure 20. Simulated groundwater nitrate loading from all sources including cropland. Simulation of the hypothetical "Scenario A": All of the hypothetically exported manure N from dairies (38% of animal N excretion) is subject to one of the following conceptual pathways: a) applied to non-dairy cropland, 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.

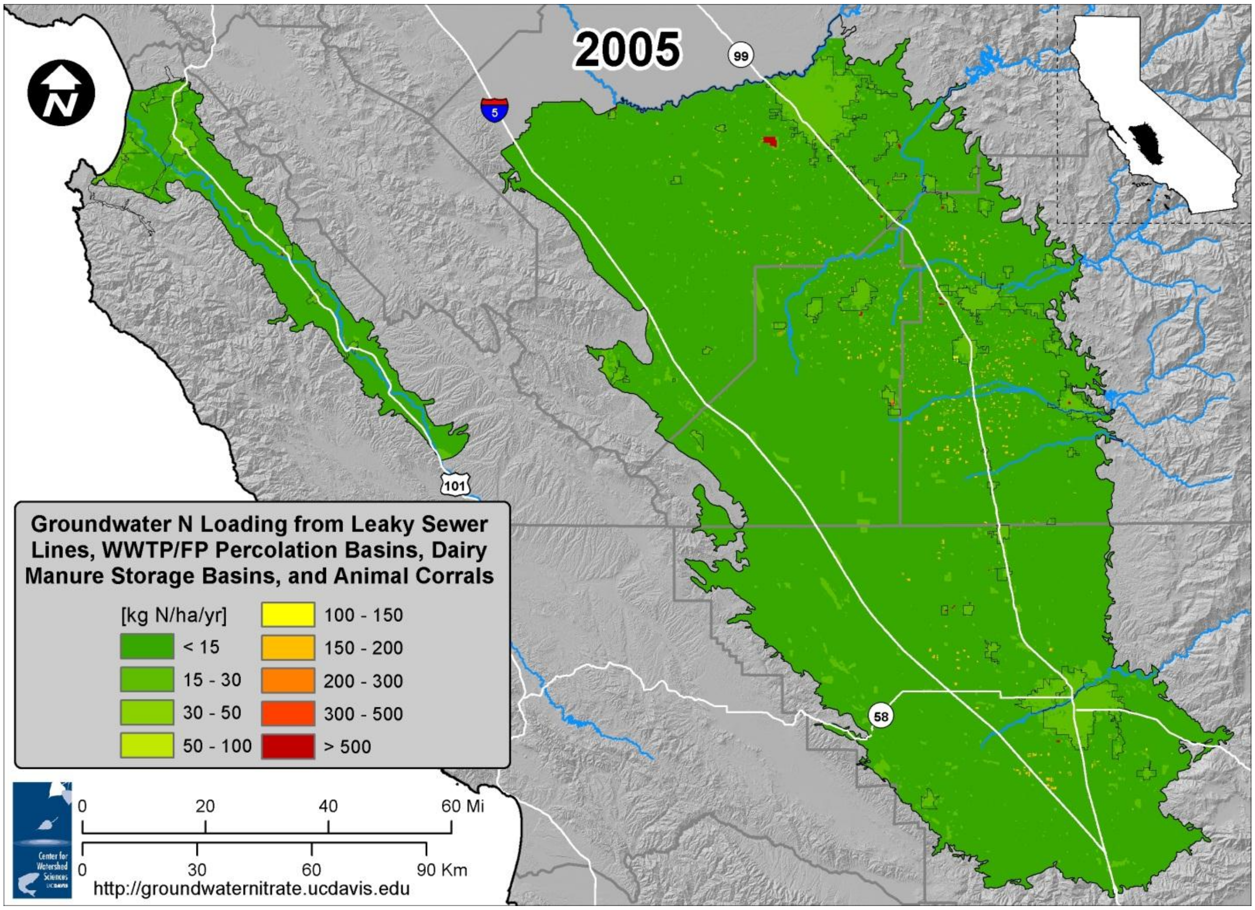


Figure 21. Groundwater loading from non-cropland sources including leaky sewer lines, turf areas, golf courses, WWTP/FP percolation basins, dairy manure lagoons, and animal corrals, in 2005. Septic systems loading (Figure 8) is not included here.

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 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.

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.

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.

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 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).

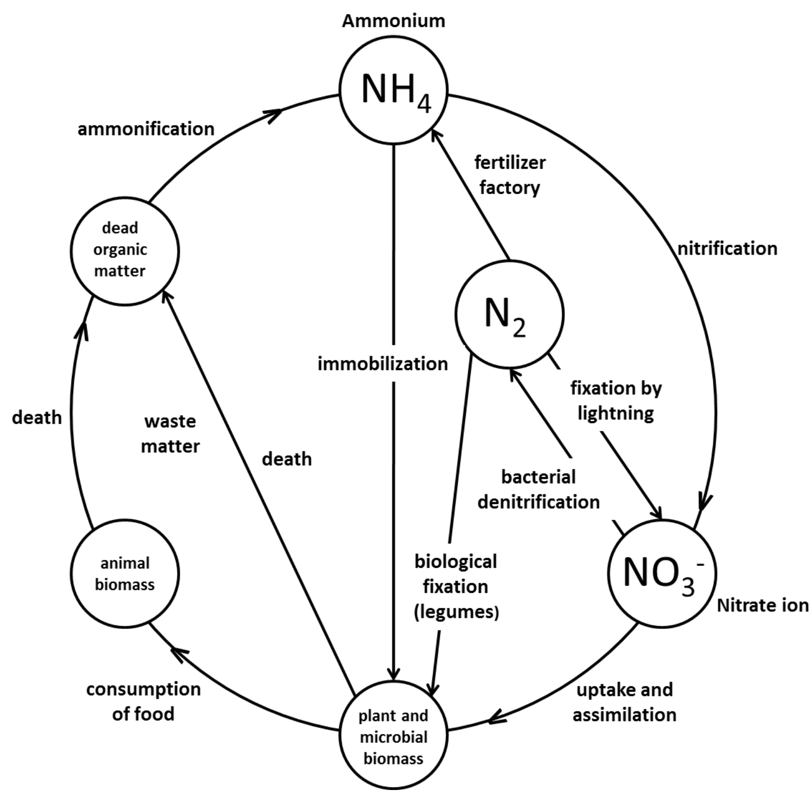


Figure 22. Dominant pathways of the nitrogen cycle (adapted from Stonecypher 2010).

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.

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 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.

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 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.

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.

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 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.

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 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 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.

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.

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.

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, $N_{WWTP\text{-}FP}$

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

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.

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.

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 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}.

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:

$$N_{fertil} = N_{norm} \cdot\bullet\ \text{dairy-}\Sigma N_{fertil} / \text{dairy-}\Sigma N_{norm}$$

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.

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

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.

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.

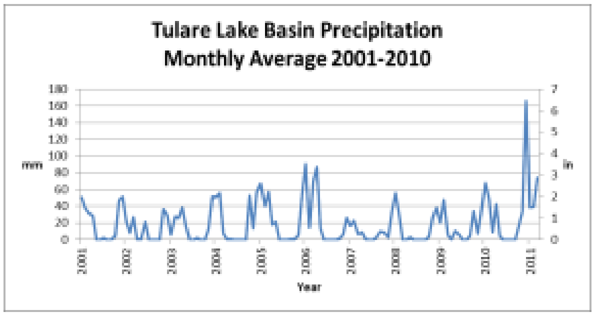
3 Cropland Nitrogen Loading

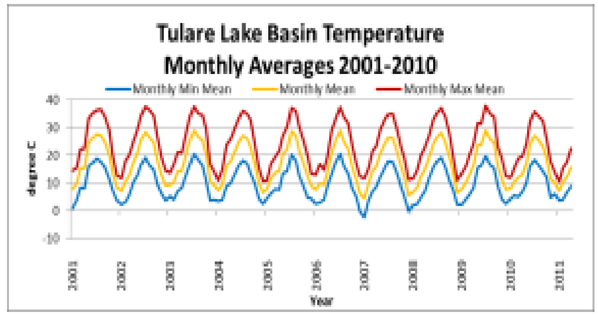
3.1 Introduction

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).

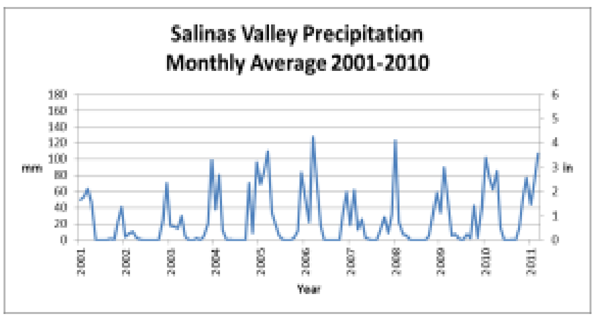
Nitrate leaching from cropland is a well-recognized and well-studied issue globally (Sutton et al. 2011). However, nitrate leaching in California's cropping systems is unique by comparison to other temperate agricultural areas. California's semi-arid climate creates two distinct management periods. During 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).

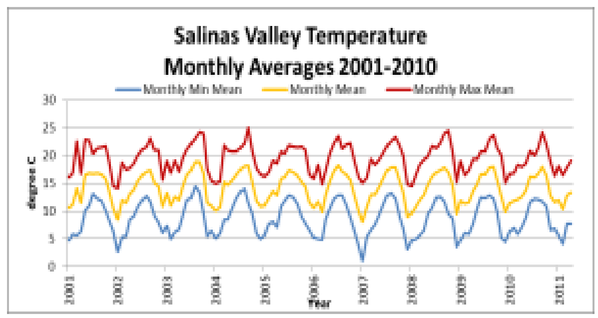
Deep percolation of nitrate laden water does not benefit crop production. Movement of nitrate beyond the root zone represents a financial loss for the farmer and is an environmental concern. However, leaching is sometimes a consequence of the need to control excess salinity. Generally, semi-arid agriculture 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 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.

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.

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 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.

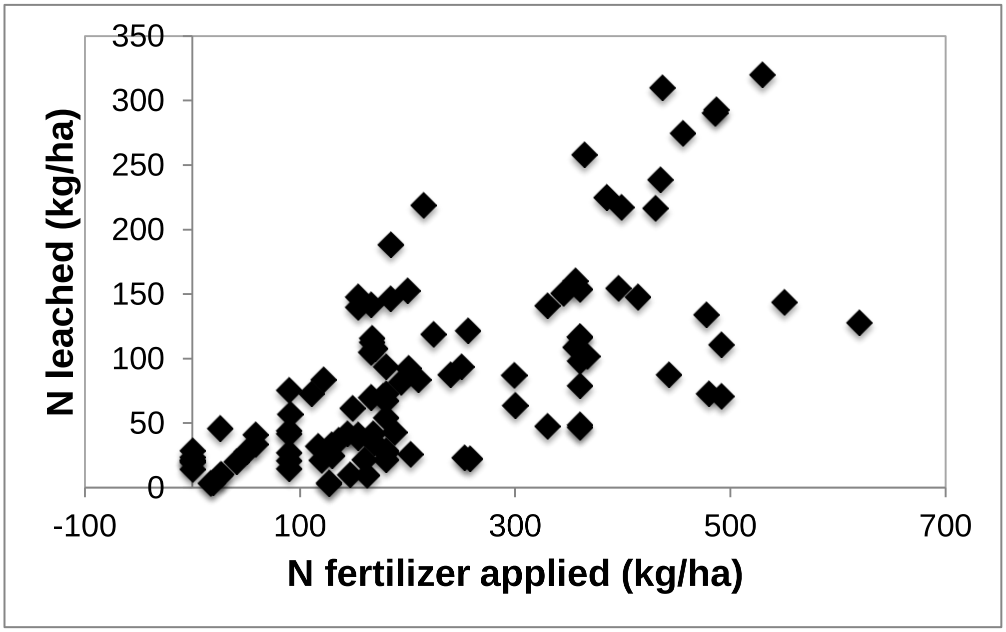


Figure 24. Current evidence of the relationship between N leached and N inputs. Based on a compilation of measurements taken in California, 1970 – 2010. Note: four outliers of high N inputs (> 1000 kg/ha) and high N leached (> 700 kg/ha) were omitted. Source: CNA (In Preparation) & Appendix I.

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.

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.

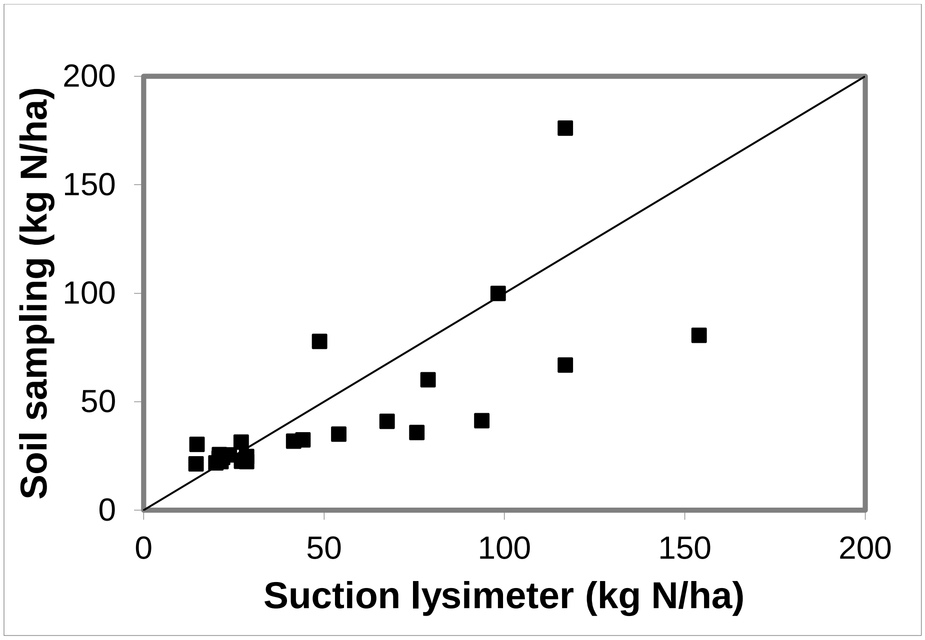


Figure 25. Comparison of nitrate leaching estimates measured by soil sampling and suction lysimeters on a fine sandy loam in California. Line is 1:1. (Source: Broadbent and Carlton 1980.)

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 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 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 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

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.

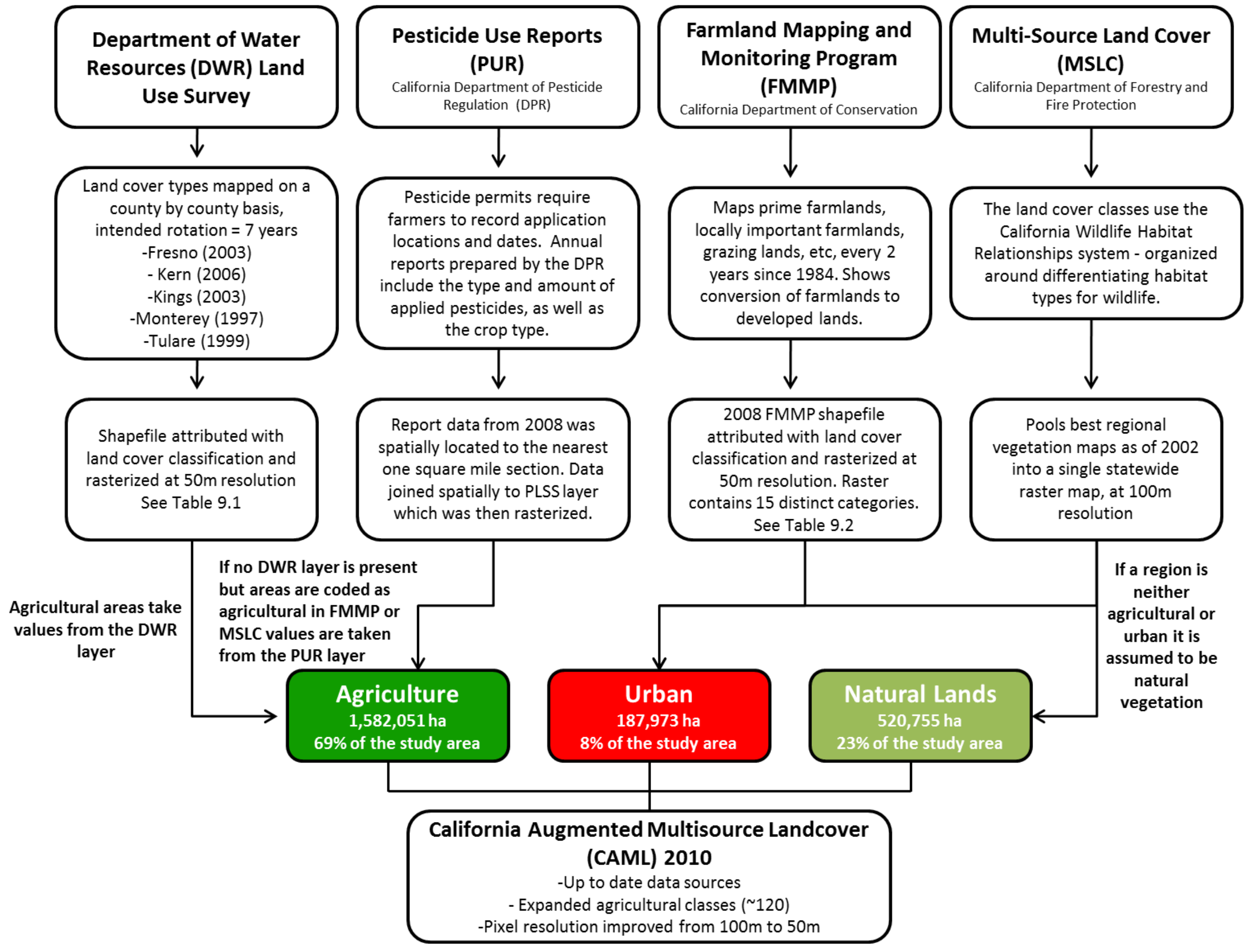


Figure 26. Flowchart of inputs to the CAML raster layer.

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 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).

Table 12. Farmland Mapping and Monitoring Program categories.

List of Categories in Farmland Mapping and Monitoring Program
Confined Animal Agriculture (Cl)
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).

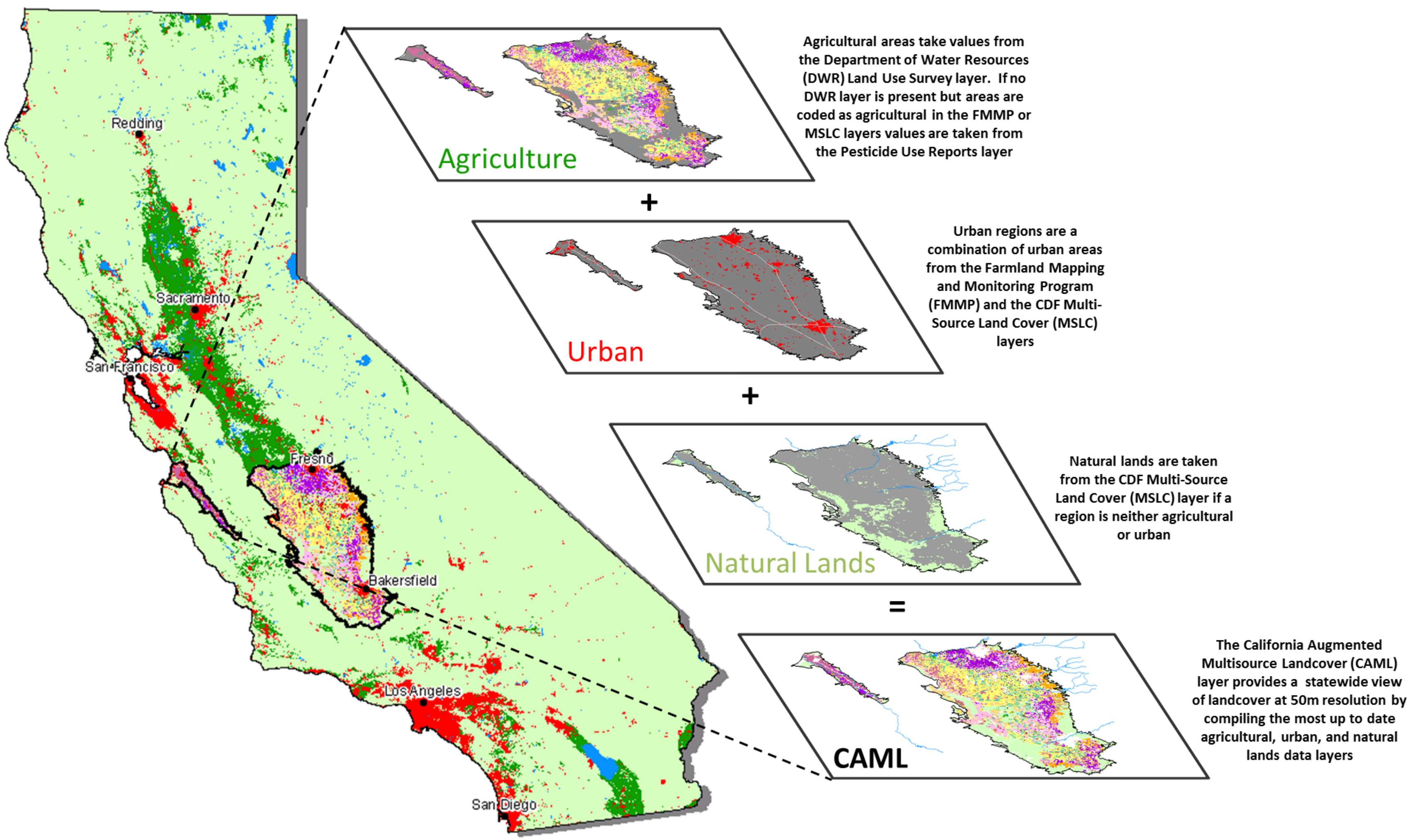


Figure 27. Input layers for the final CAML 2010 raster layer.

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

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.).

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 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 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

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).

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

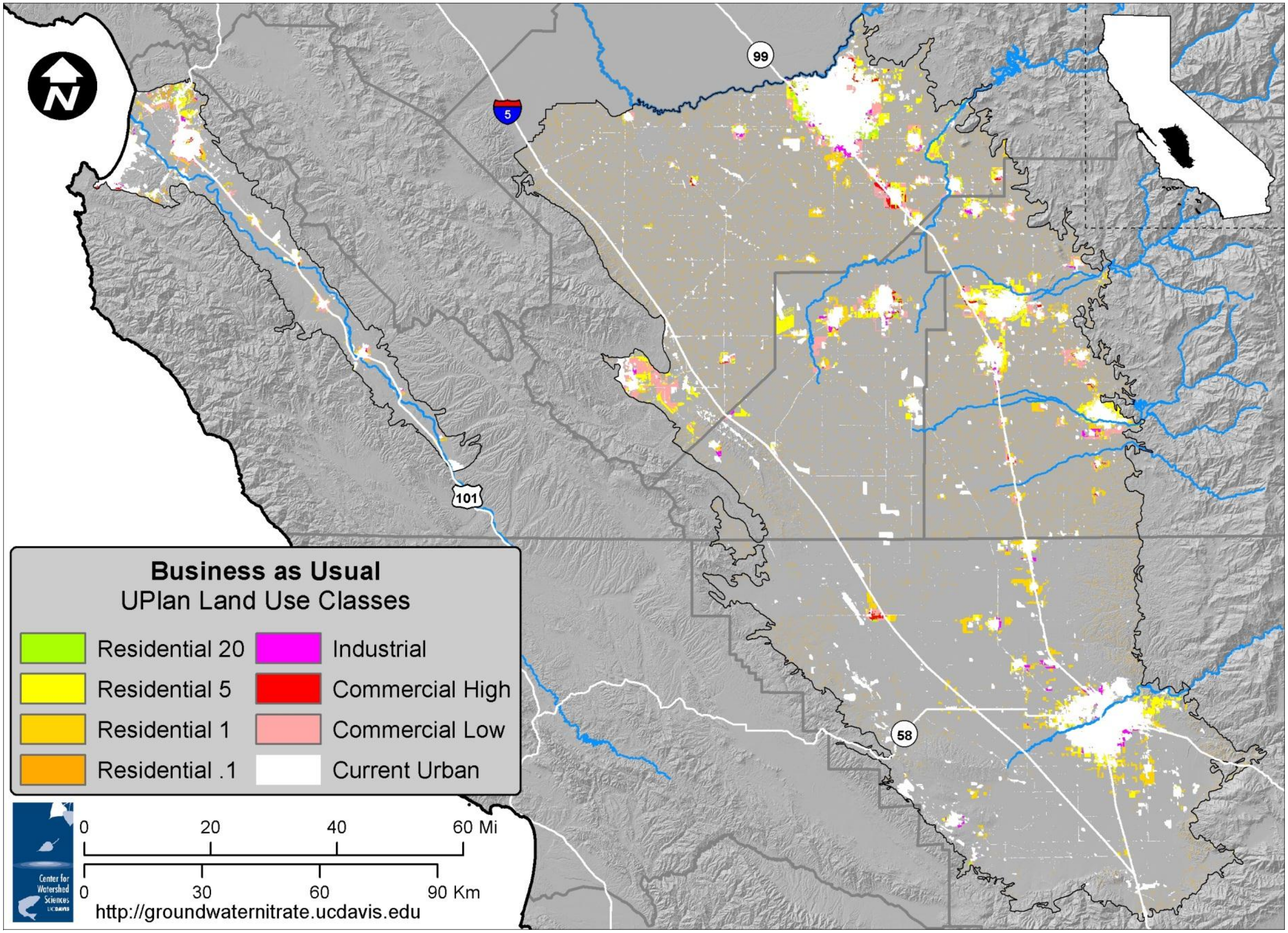


Figure 28. BAU modeling scenario (Information Center for the Environment)

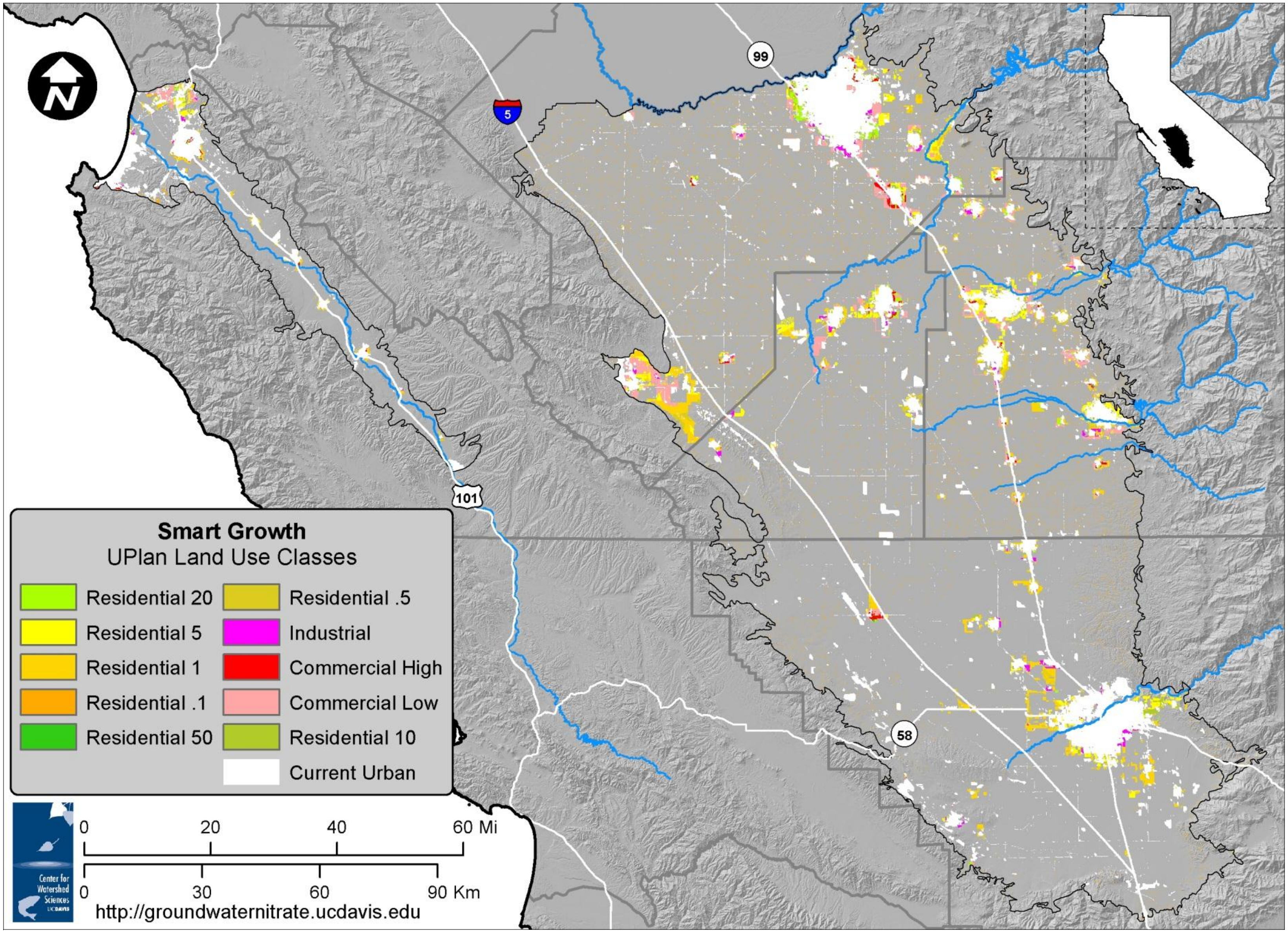


Figure 29. Smart Growth modeling scenario (Information Center for the Environment)

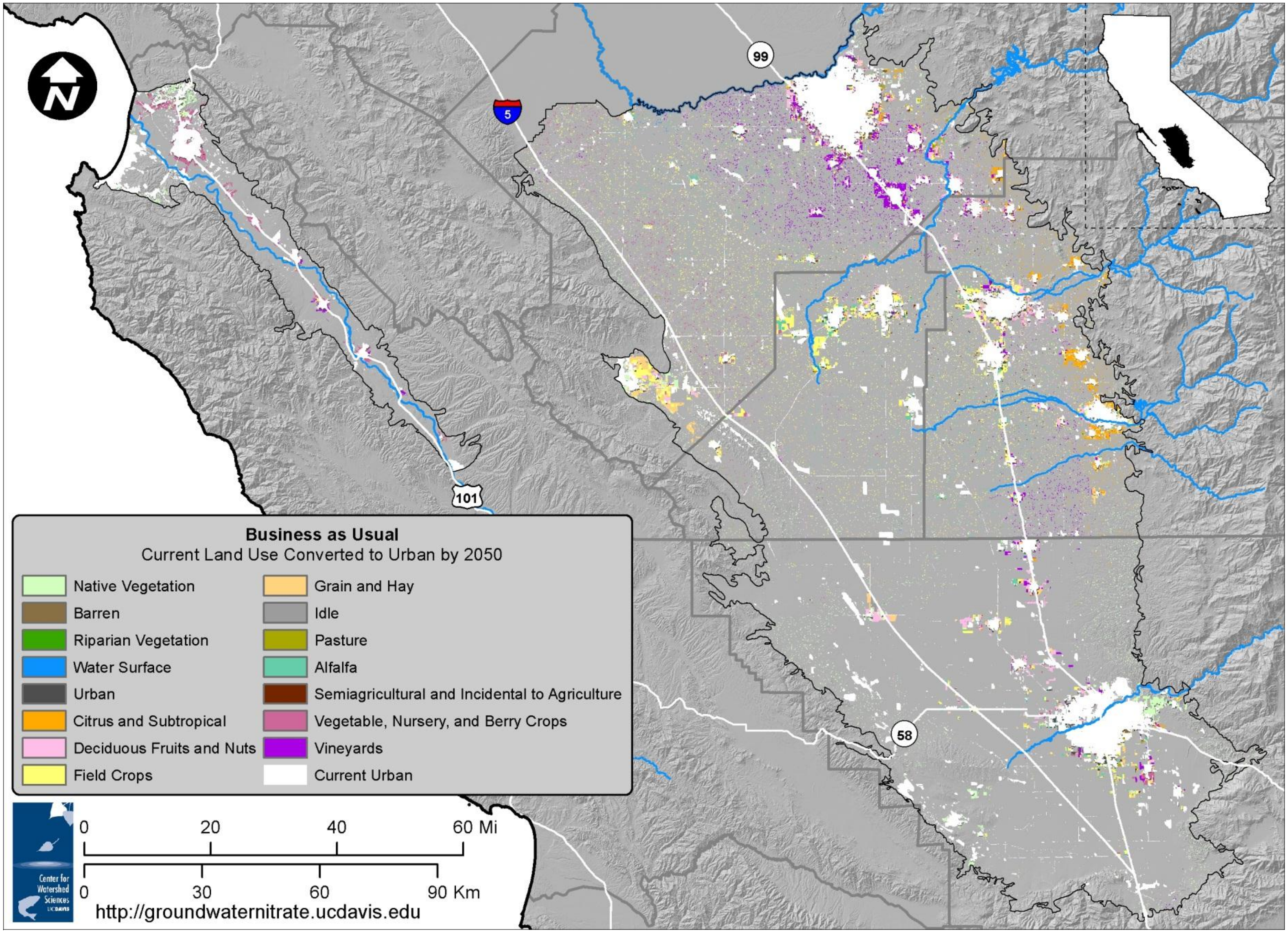


Figure 30. BAU – Agriculture removal modeling scenario (Information Center for the Environment)

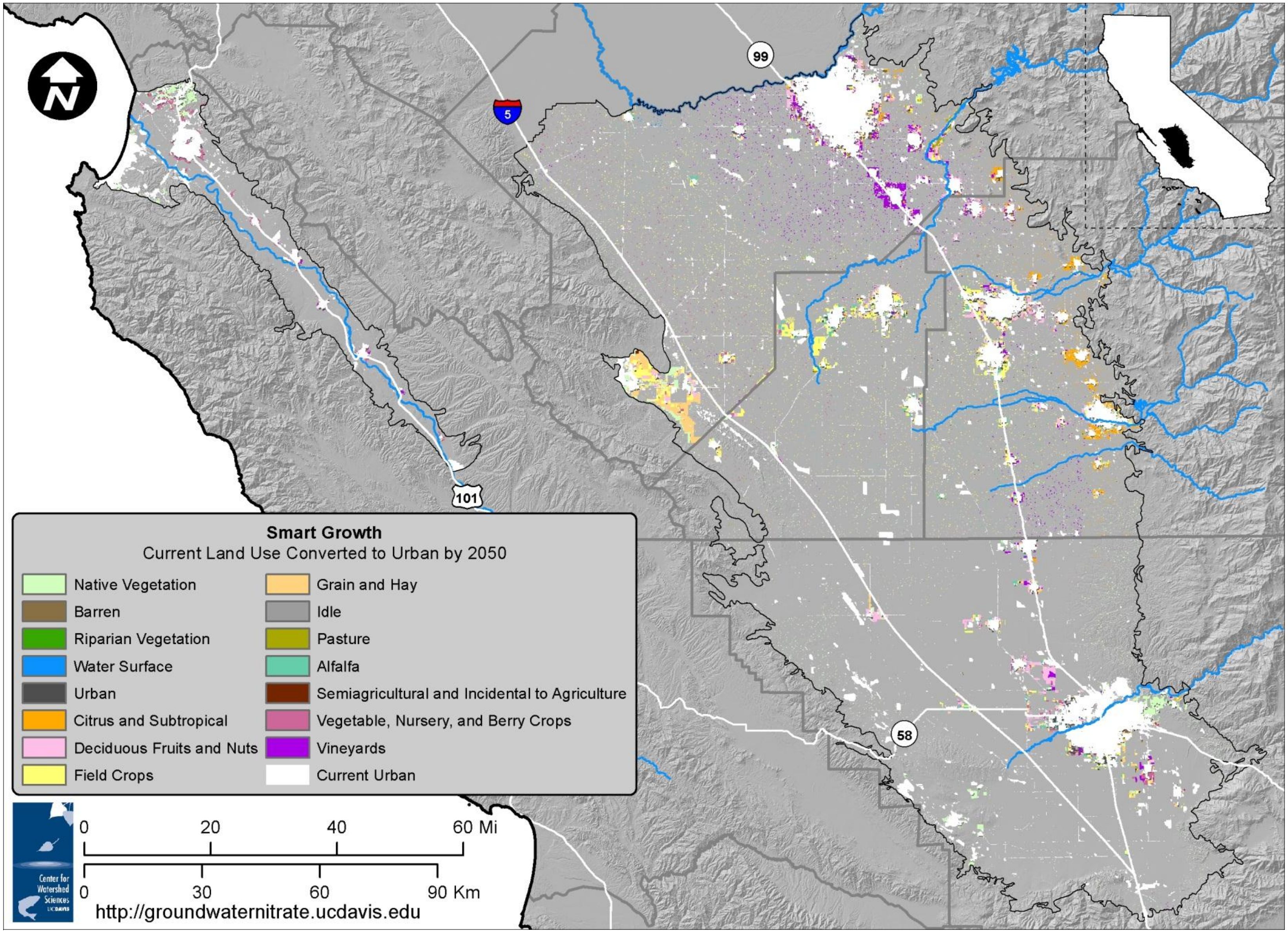


Figure 31. Smart Growth – Agriculture removal modeling scenario (Information Center for the Environment)

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**

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 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

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 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.

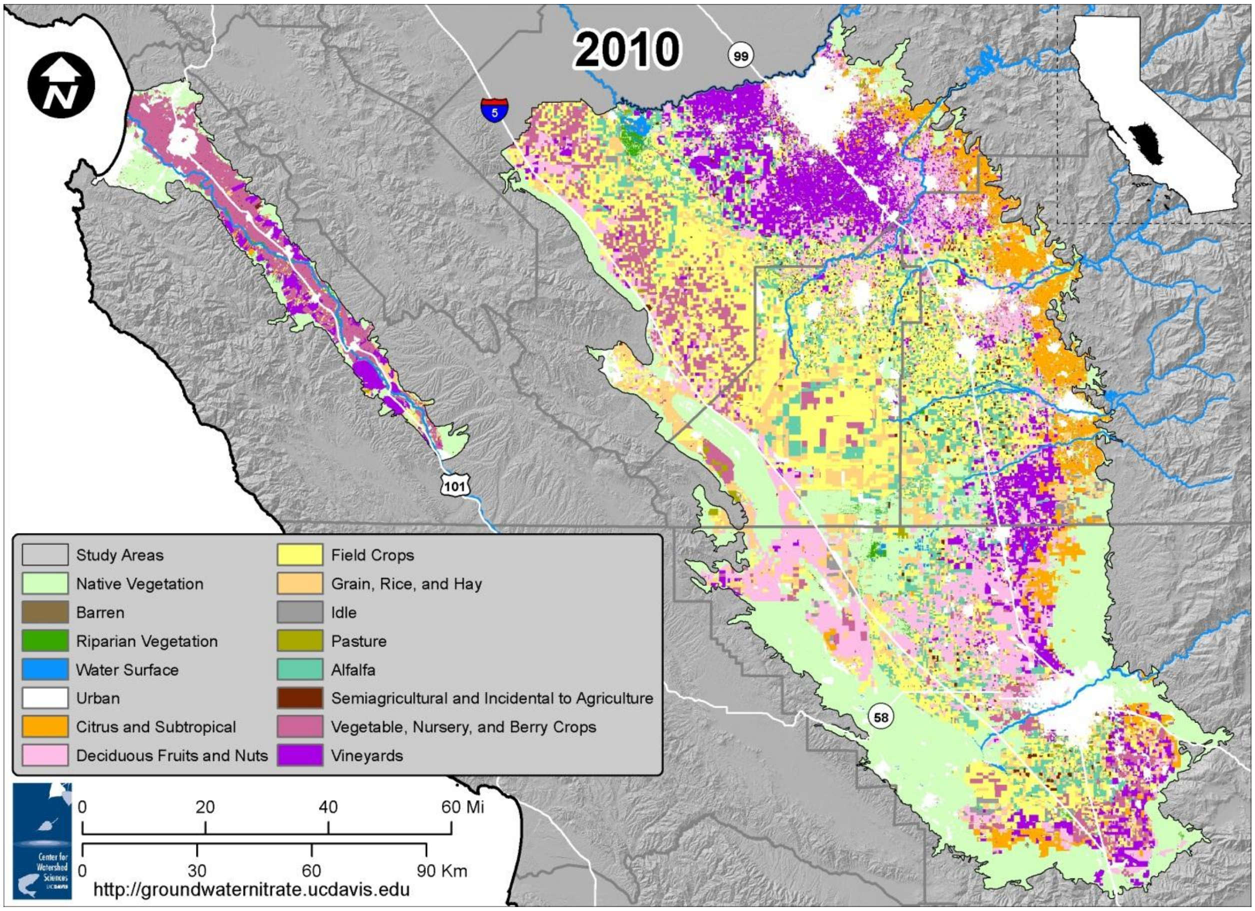


Figure 32. CAML 2010 Land Cover Map (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment)

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).

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

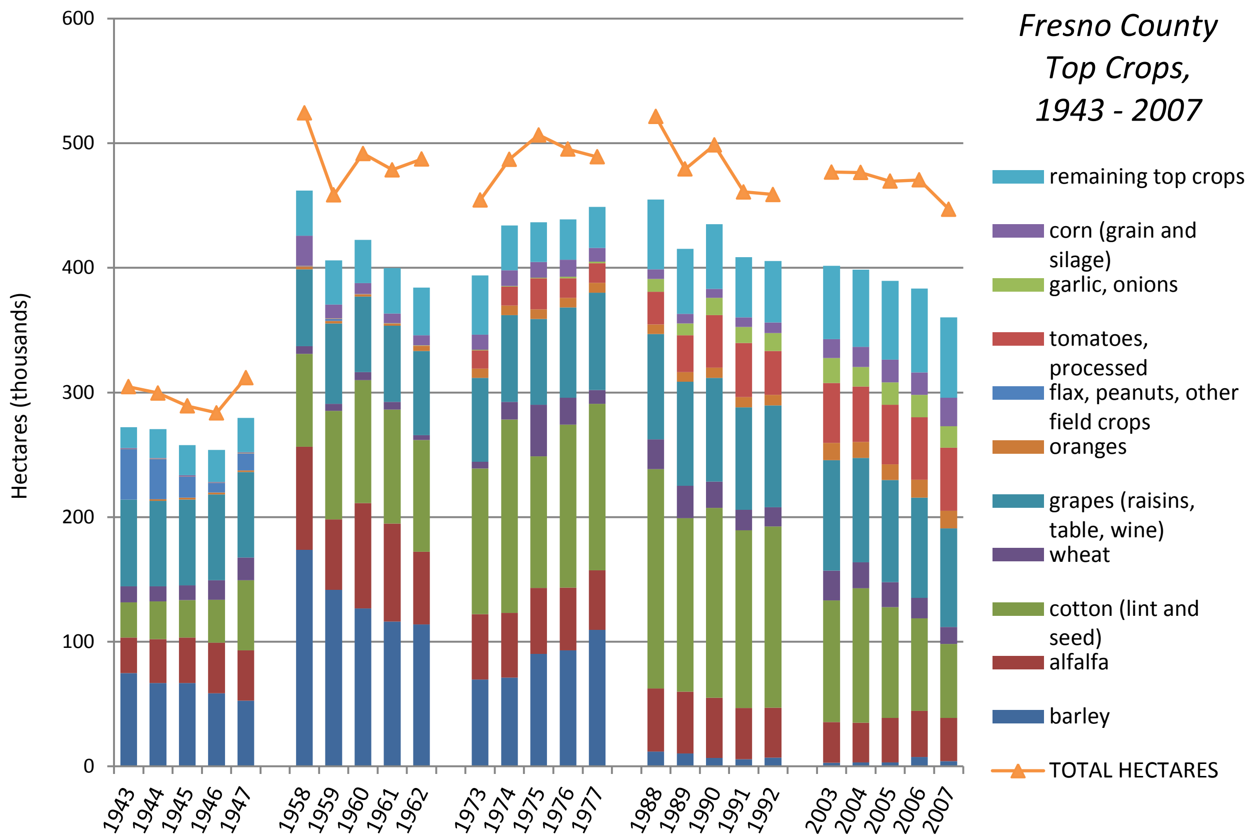


Figure 33. Fresno County top crops in hectares. Source: Fresno County Department of Agriculture Annual Crop and Livestock Reports

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).

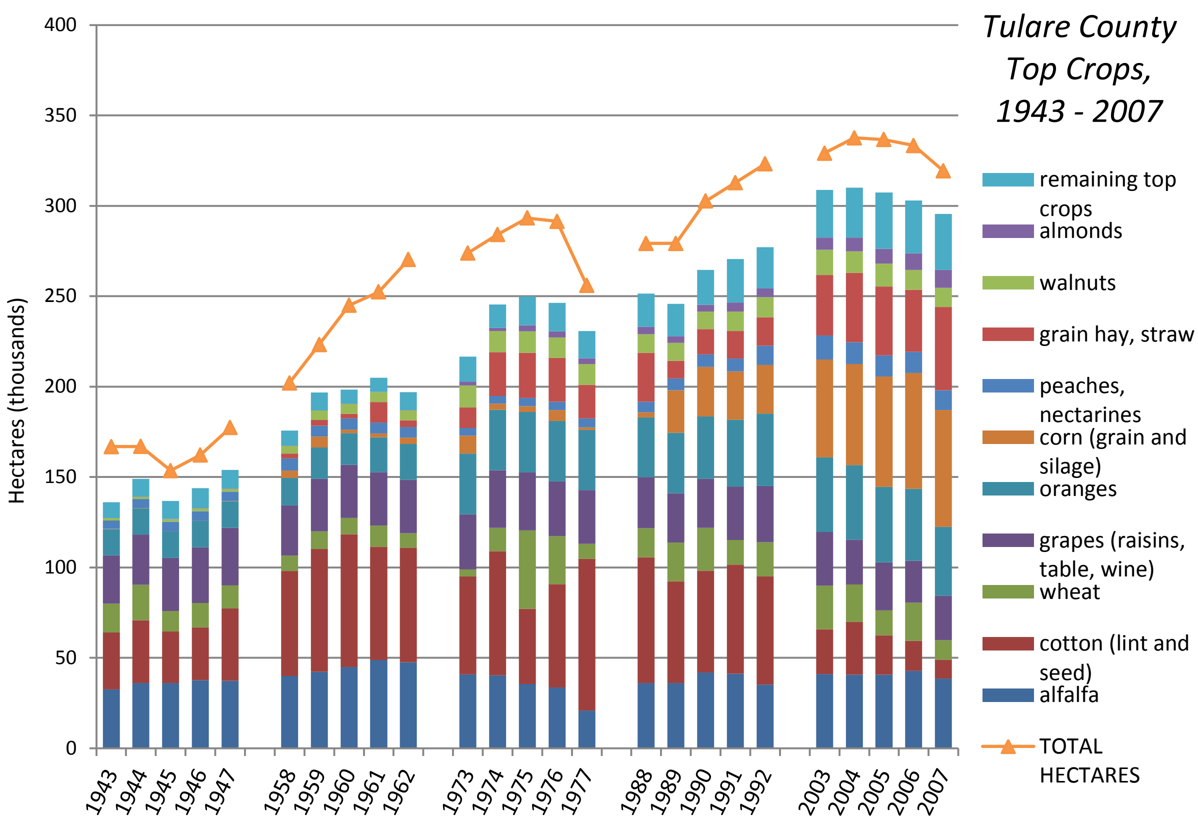


Figure 34. Tulare County top crops in hectares. Source: Tulare County Department of Agriculture and Office of the Agricultural Commissioner/Sealer Annual Crop and Livestock Reports

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).

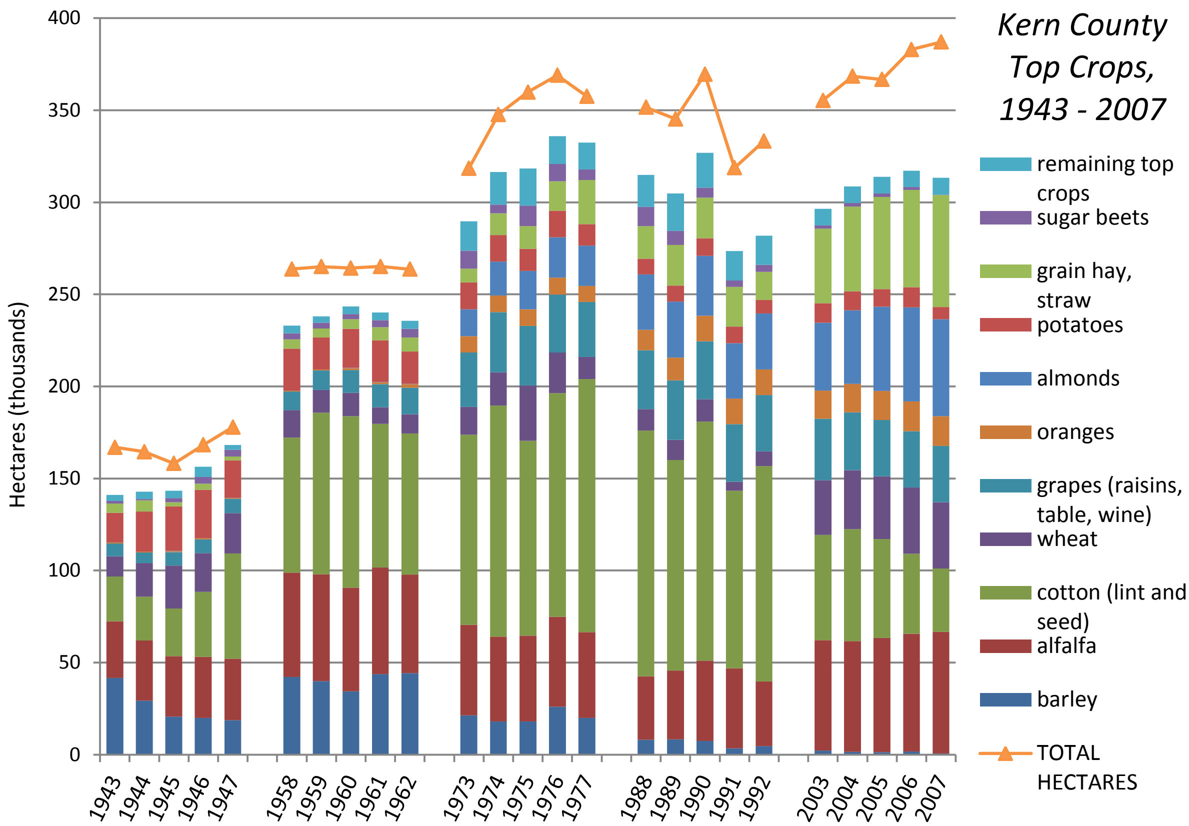


Figure 35. Kern County top crops in hectares. Source: Kern County Department of Agriculture and Measurement Standards Annual Crop Reports.

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).

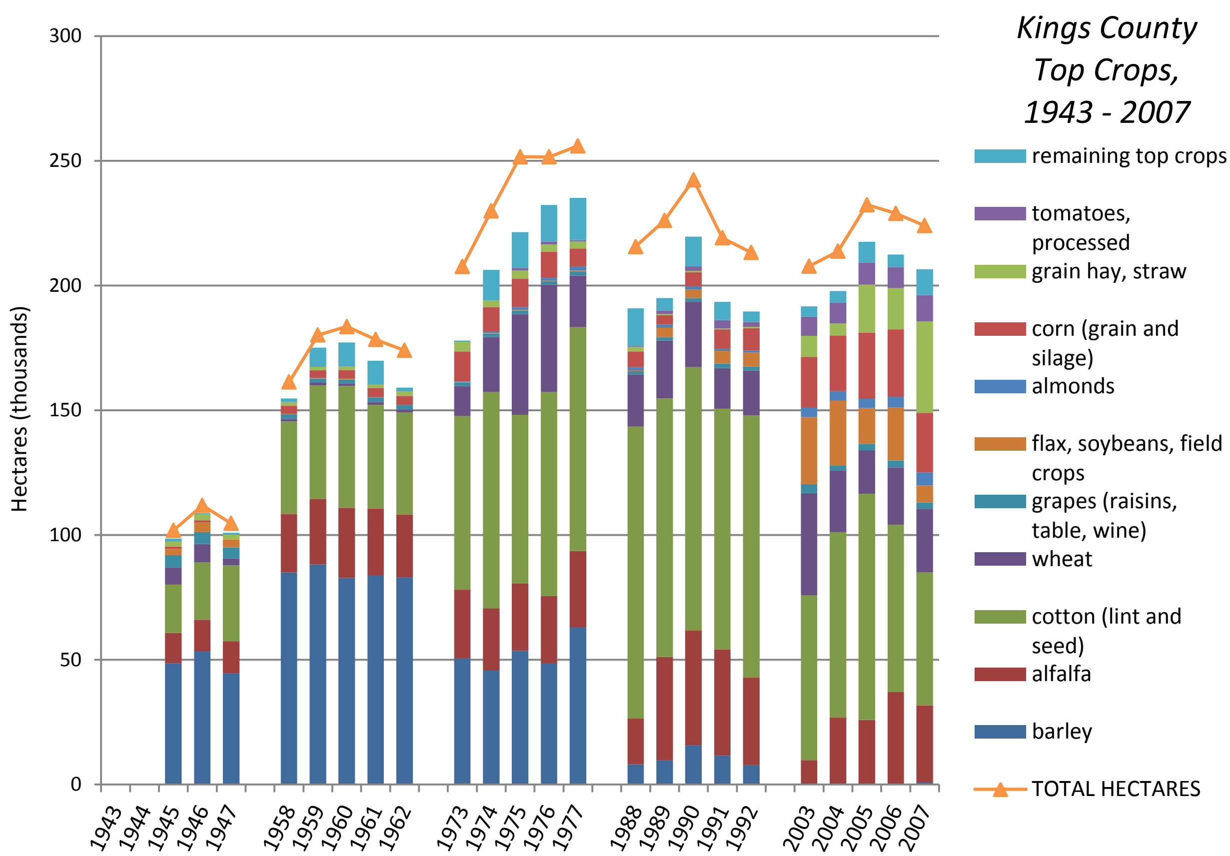


Figure 36. Kings County top crops in hectares. Source: Kings County Department of Agricultural Measurement Standards Annual Crop and Livestock Reports.

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.

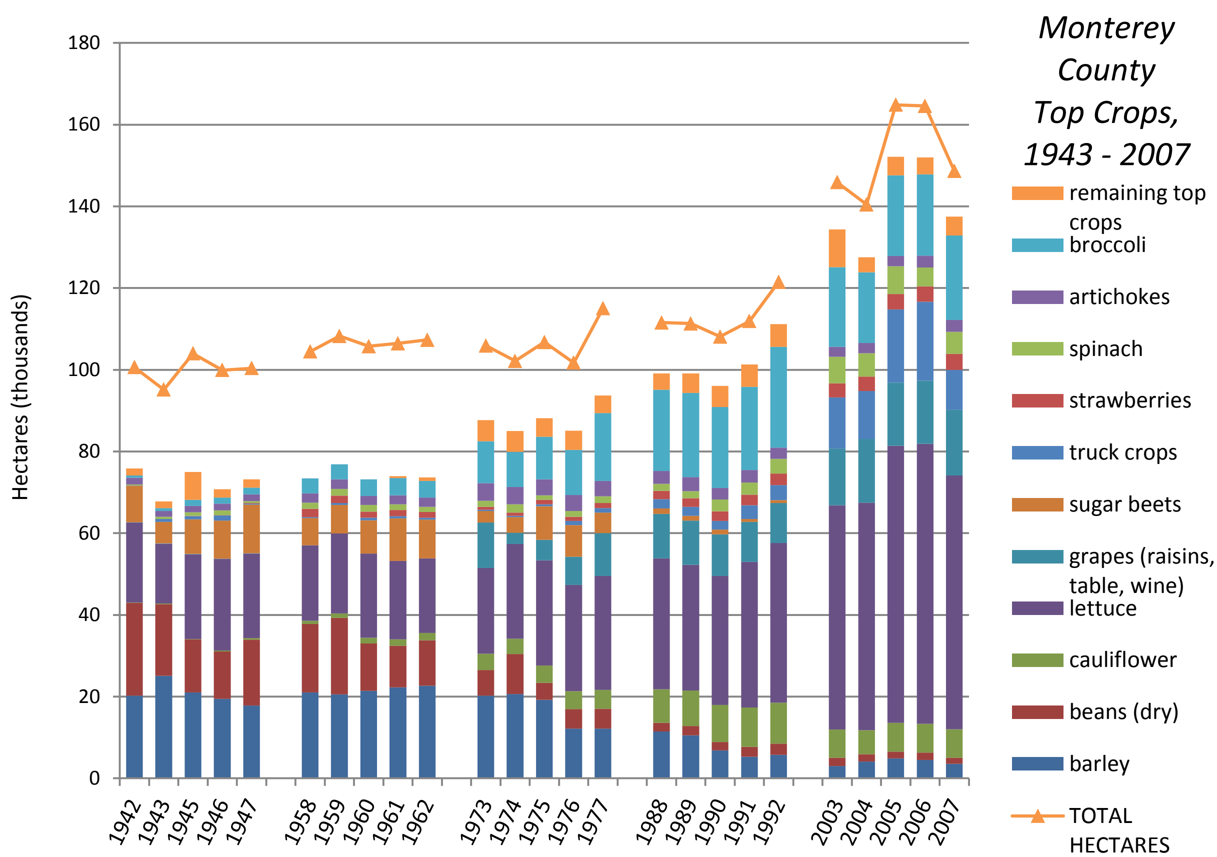


Figure 37. Monterey County top crops in hectares. Source: Monterey County Office of the Agricultural Commissioner Annual Crop Reports.

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

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 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.

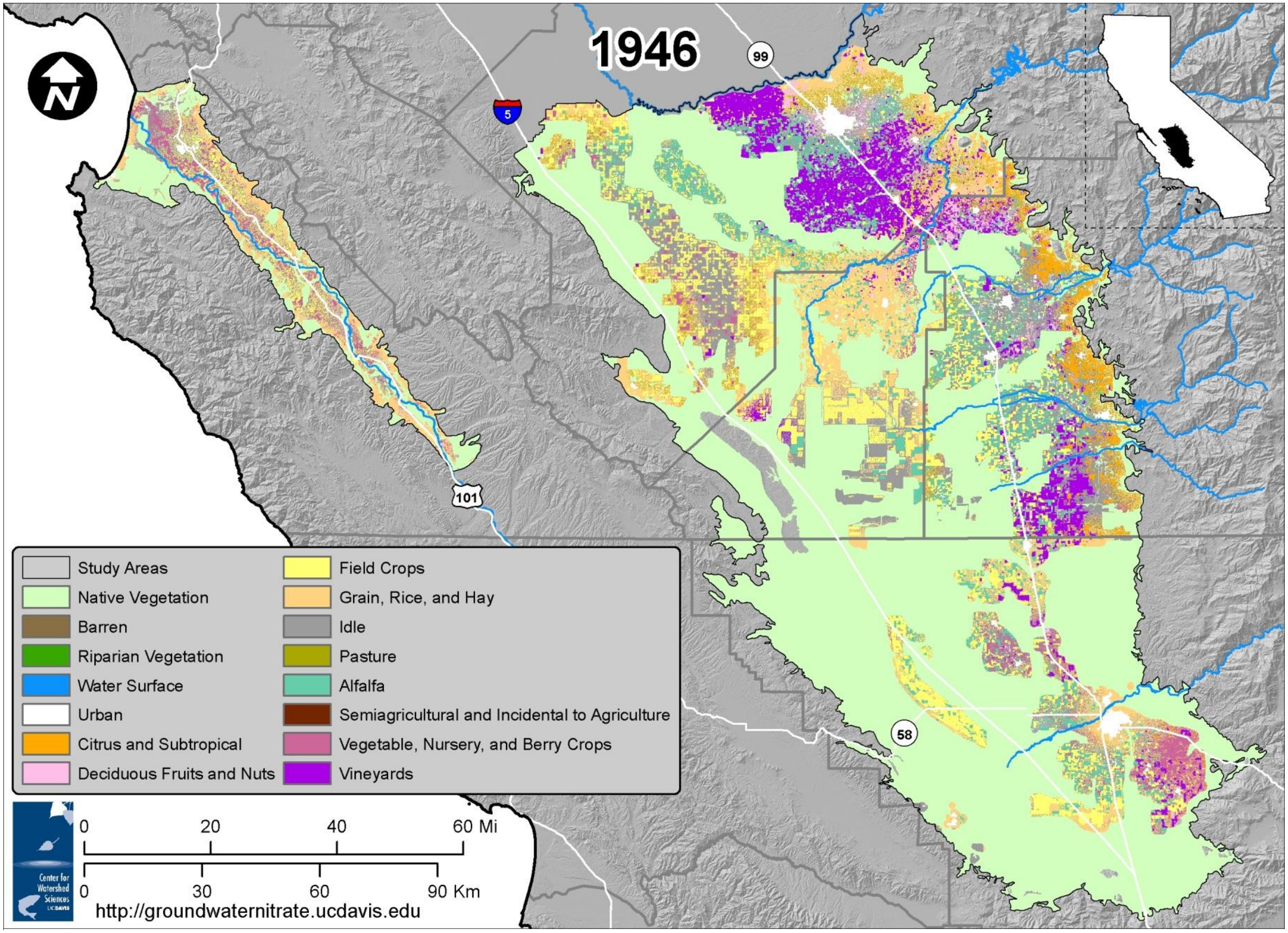


Figure 38. 1946 Land Use from DWR Mapping (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment).

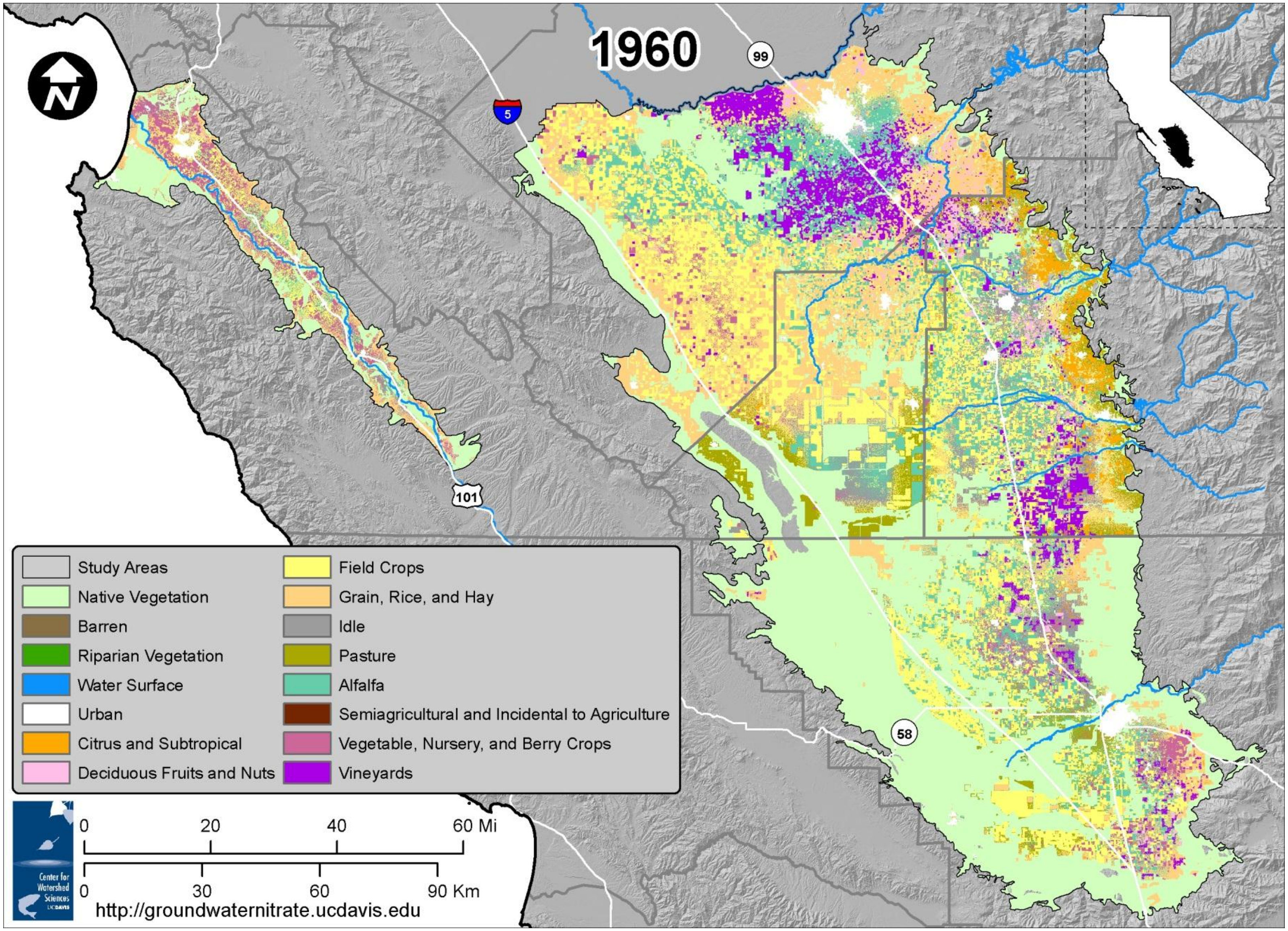


Figure 39. Backcasted 1960 Land Cover Map (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment)

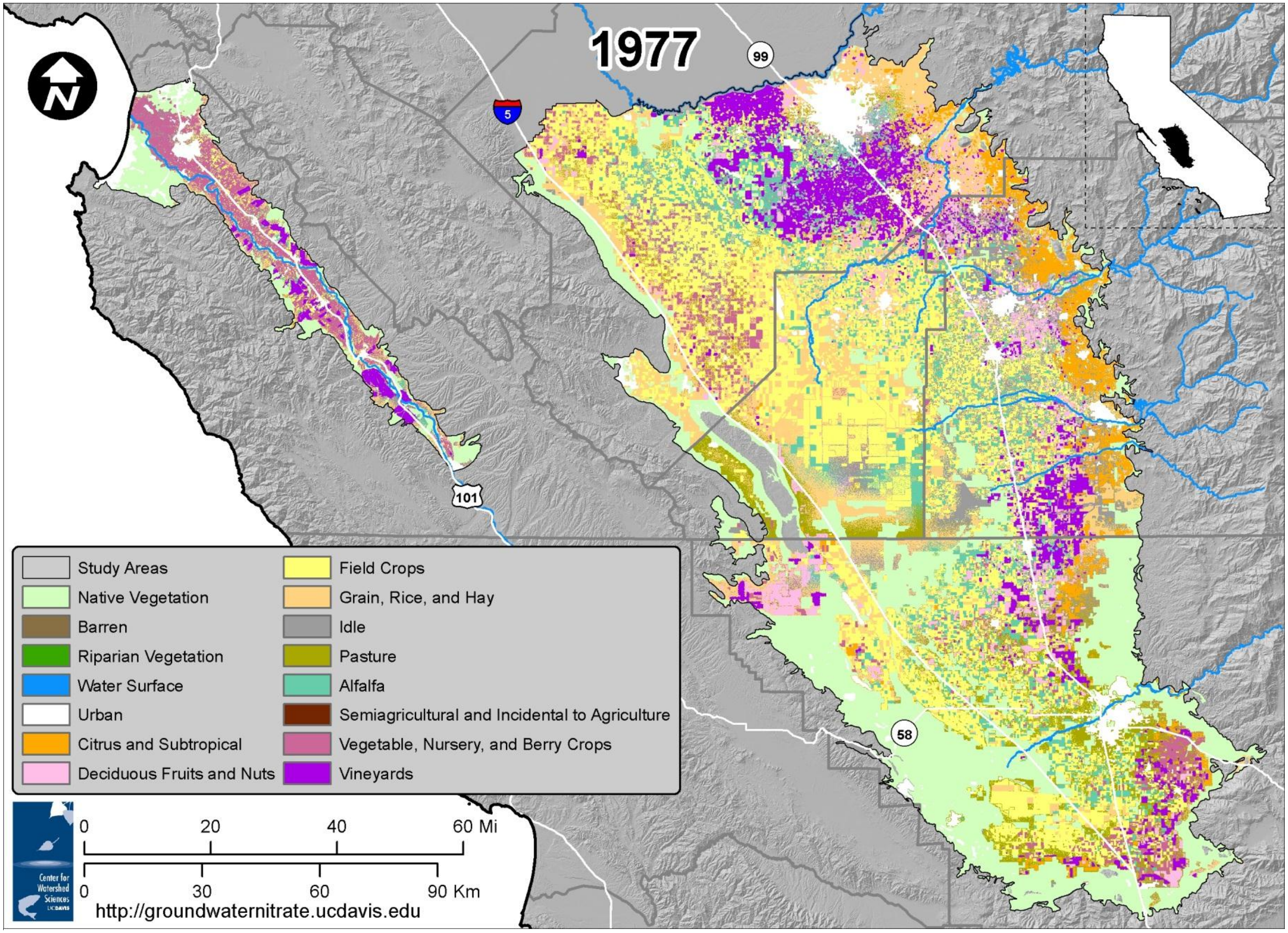


Figure 40. Backcasted 1977 Land Cover Map (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment).

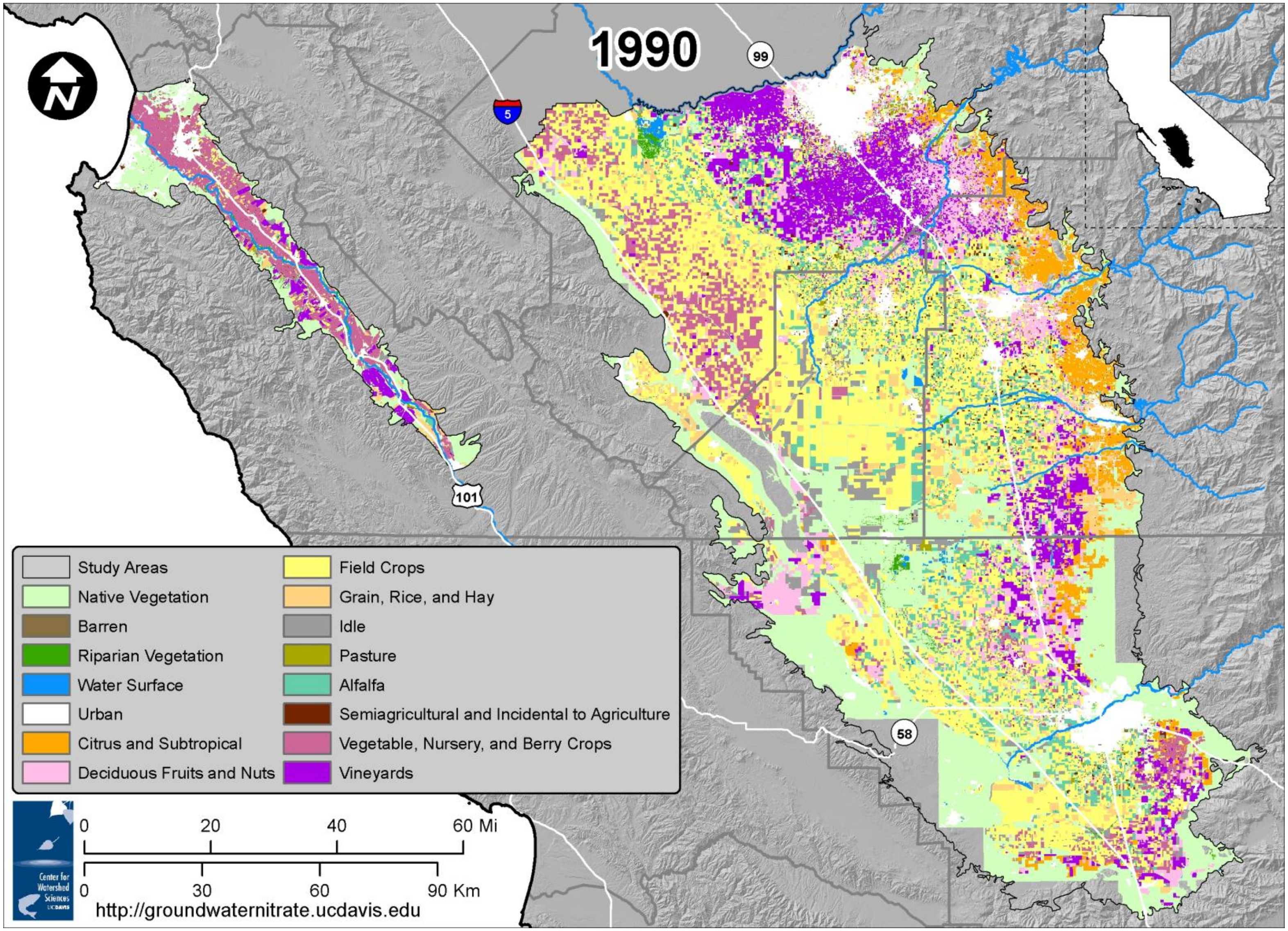


Figure 41. 1990 Land Use from DWR Mapping (CA Agricultural Commissioner, CA Department of Water Resources, Information Center for the Environment).

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**

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

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 (

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.

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.

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).

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 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).

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.

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 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 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

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, 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.

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

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.

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 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

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 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 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

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.

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 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.

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 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.

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).

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.

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 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

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"

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

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.

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 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).

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

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 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 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.

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).

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 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).

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 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.

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 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

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 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).

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).

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 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

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[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."

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.)

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.

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.

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

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.

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[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).

    - 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.

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 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 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);

- 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.

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.

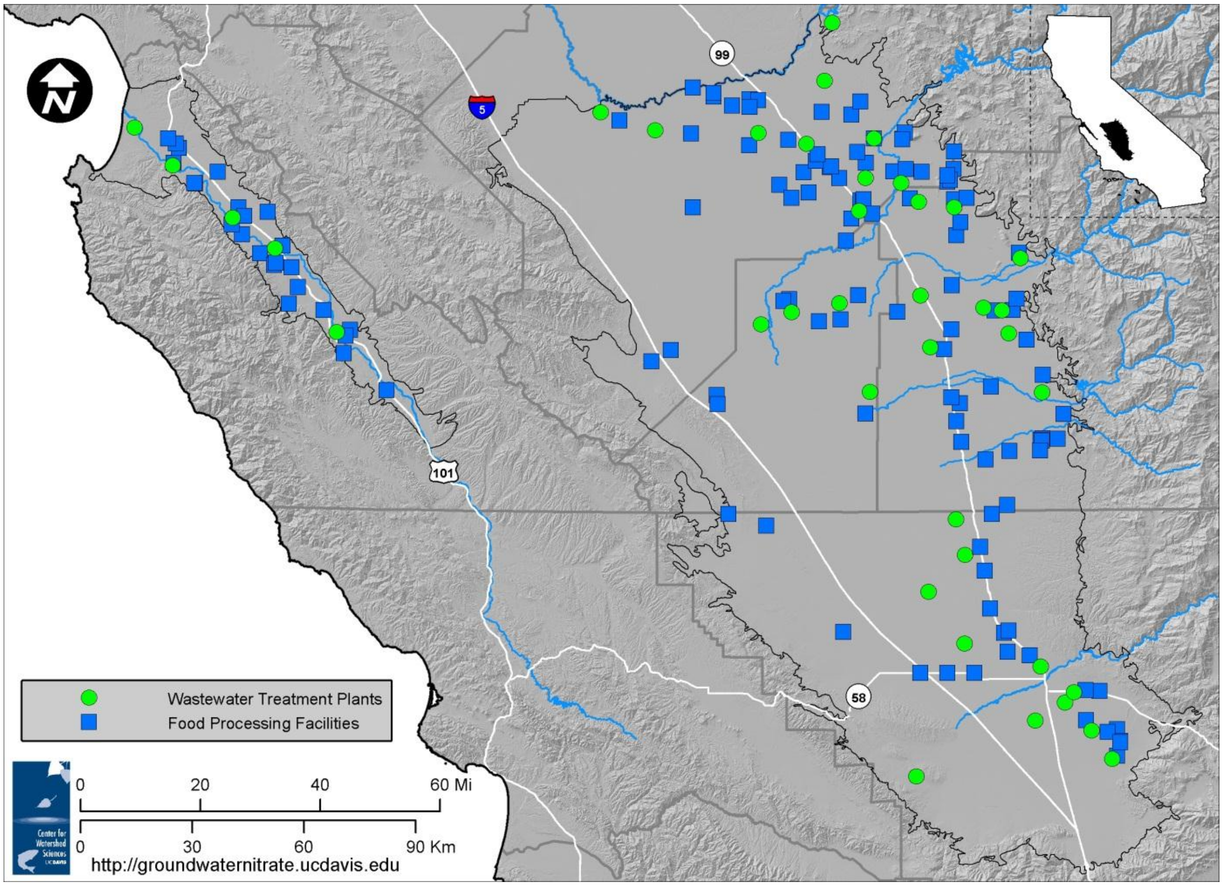

Figure 42. Location of included wastewater treatment plants and food processing facilities in the Tulare Lake Basin and Salinas Valley (FPs with an active discharge permit and the largest WWTPs comprising 90% of design flow in each basin were included in this analysis, see Appendix Table 8 **for facility 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.

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 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.

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

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 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.				

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.

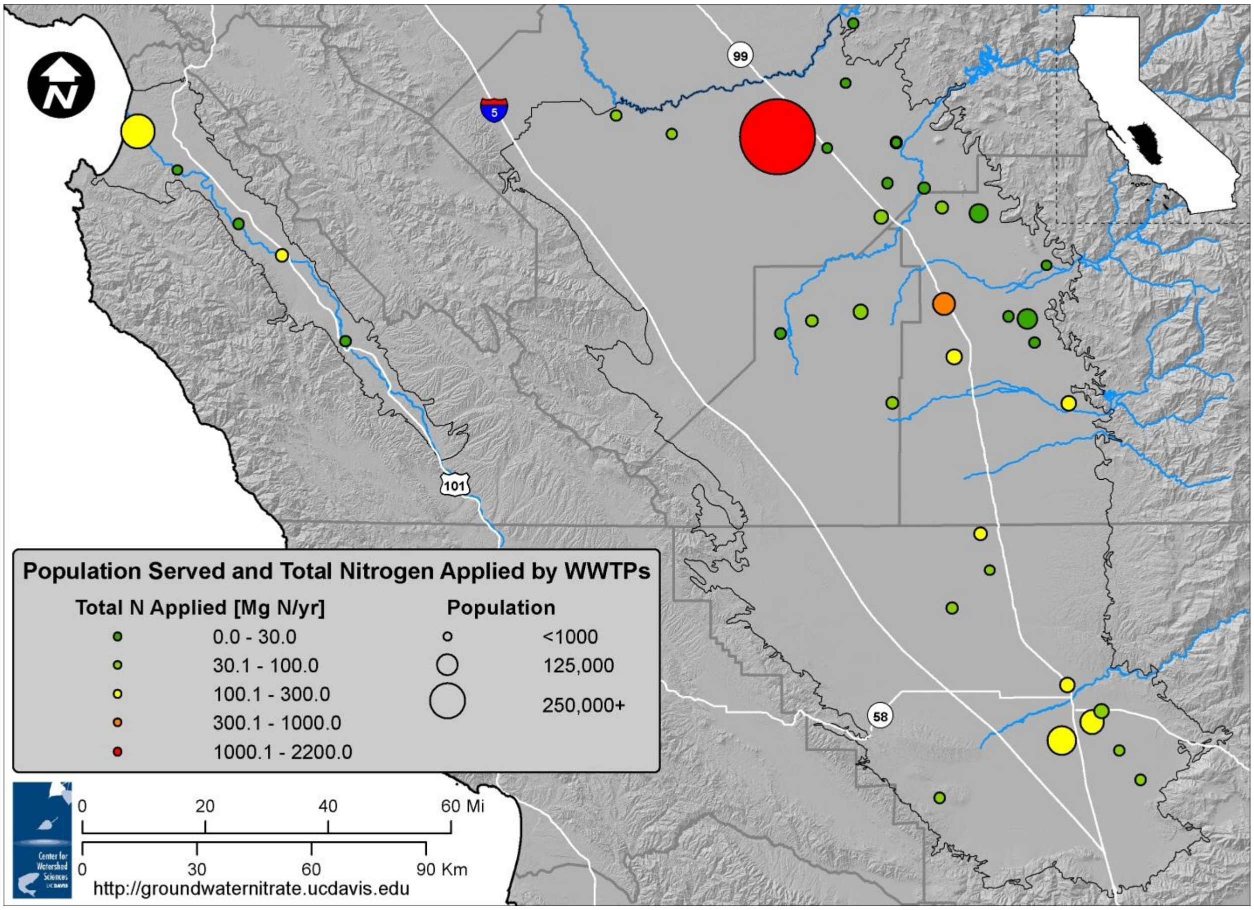

Figure 43. Wastewater treatment plants: Total applied nitrogen (metric tons N/Yr) [by color] and population served [by symbol diameter] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 Mg = 1 metric ton = 1.1 tons.]

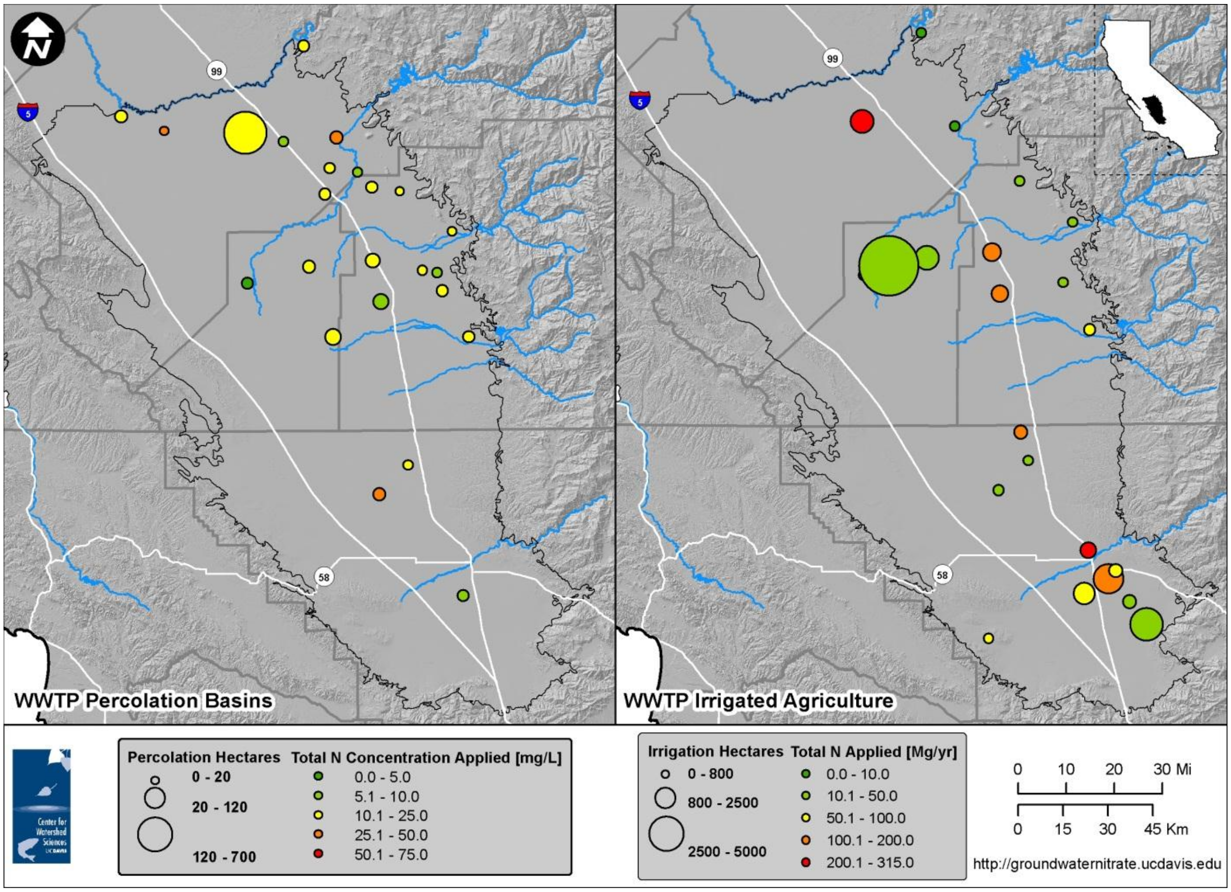

Figure 44. Tulare Lake Basin wastewater treatment plants: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) of total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 Mg = 1 metric ton = 1.1 tons, 1 hectare = 2.47 acres.]

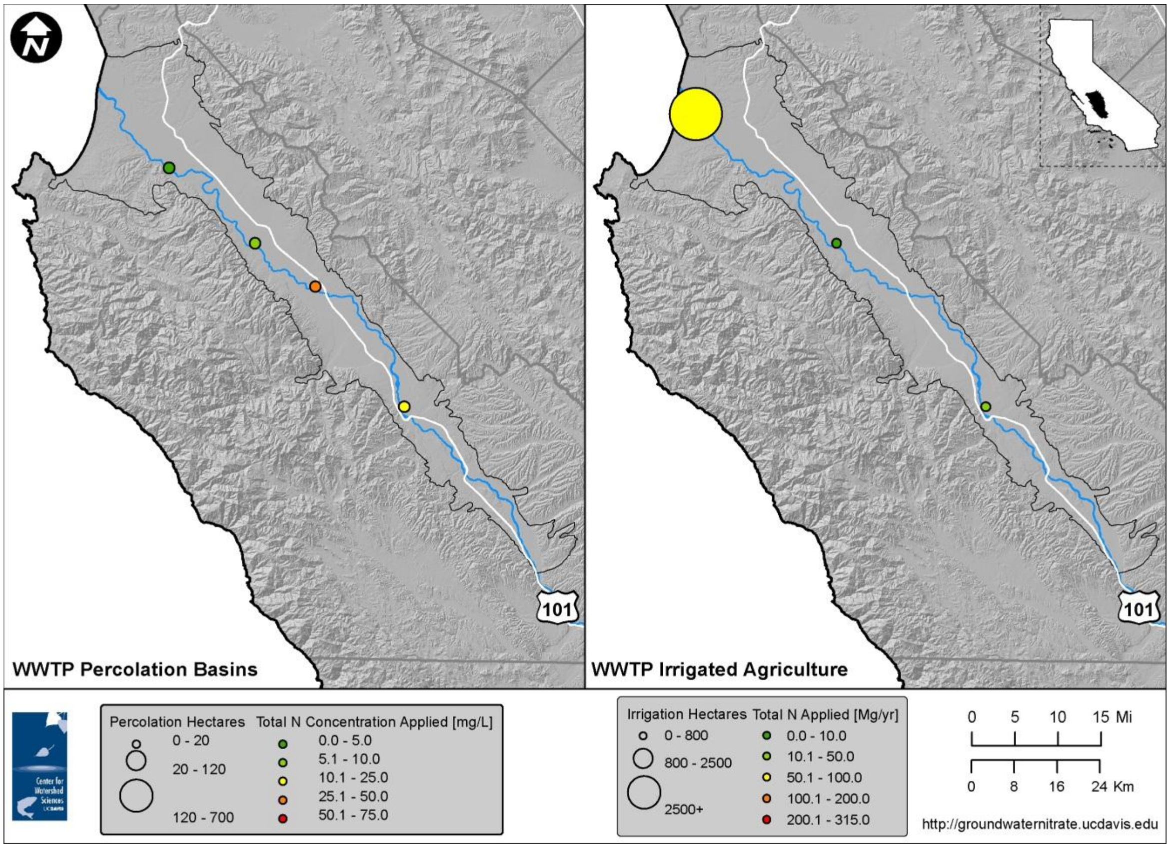

Figure 45. Salinas Valley wastewater treatment plants: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) of total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (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.

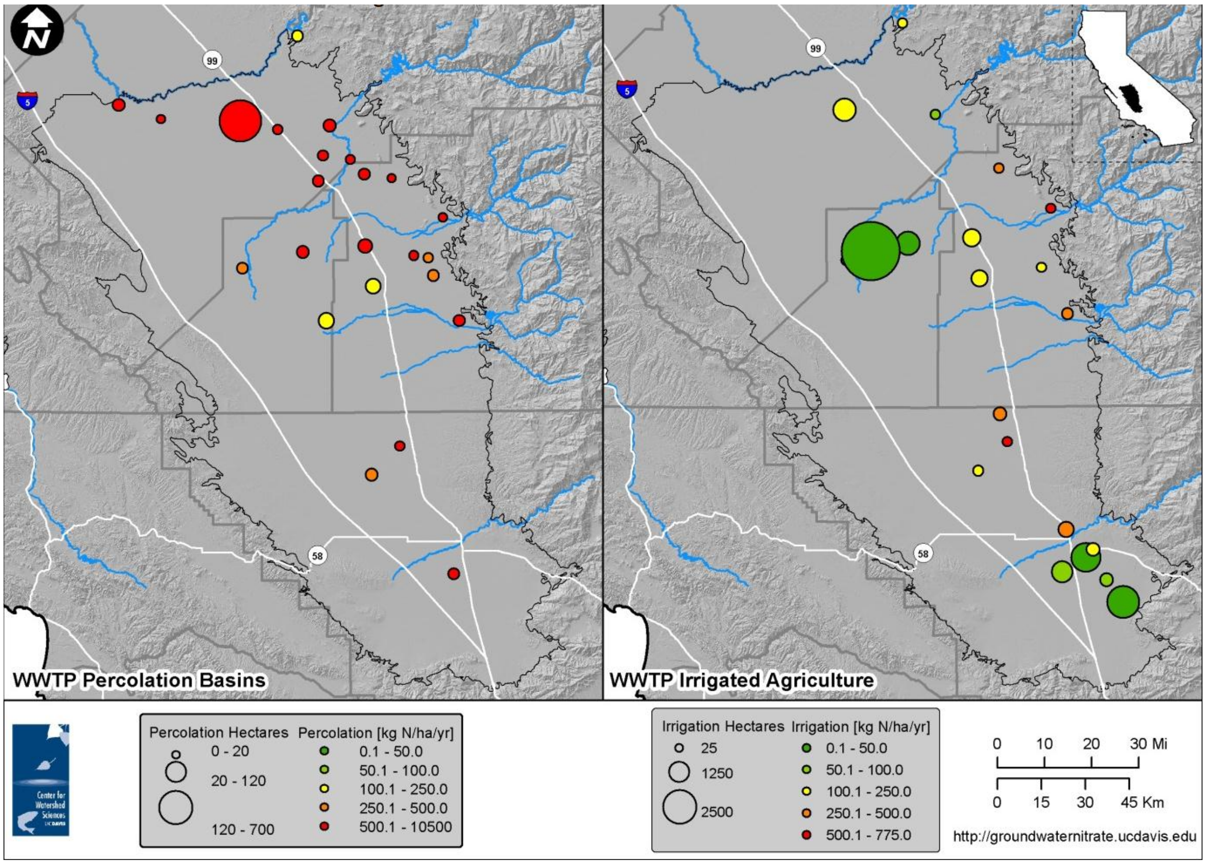

Figure 46. Tulare Lake Basin wastewater treatment plants: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

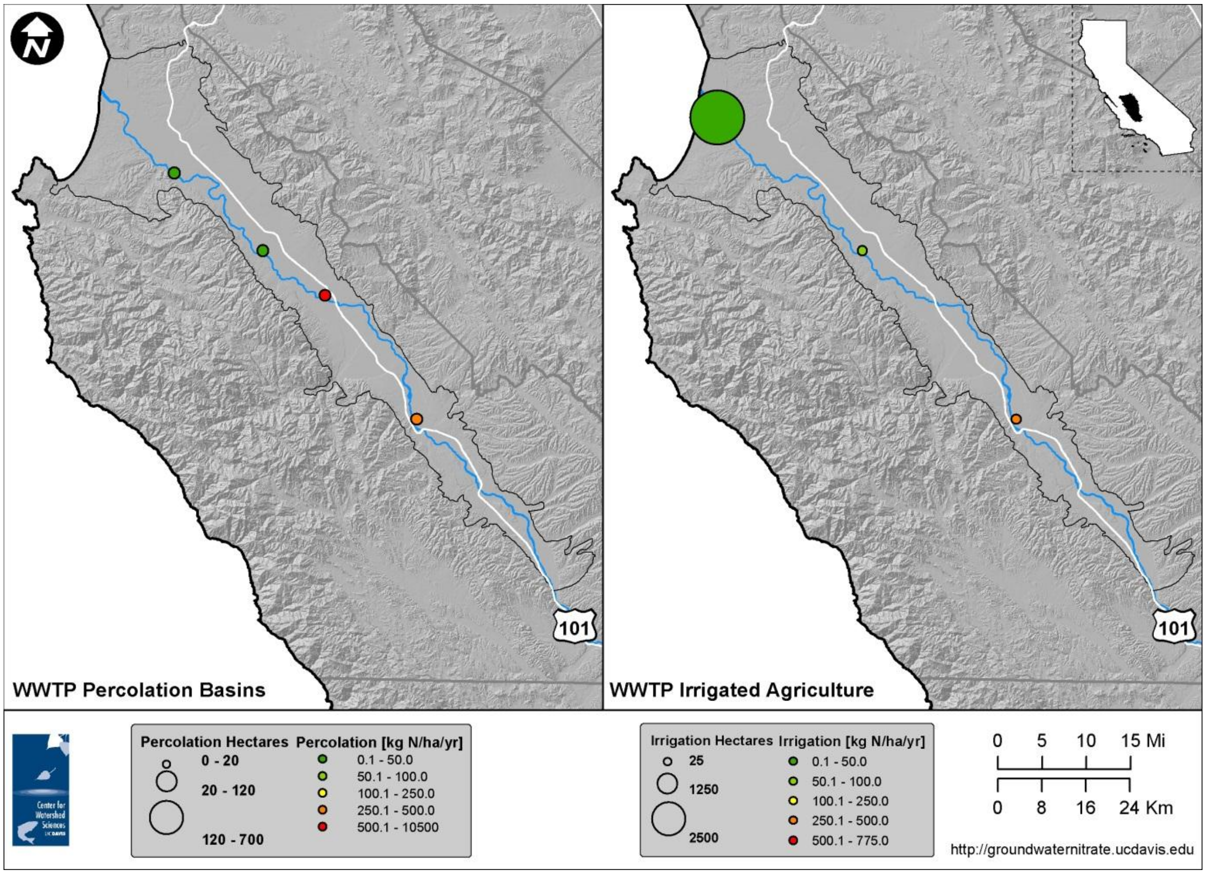

Figure 47. Salinas Valley wastewater treatment plants: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs, SMRs.) [1 kg = ~2.2 lb, 1 hectare = 2.47 acres.]

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).

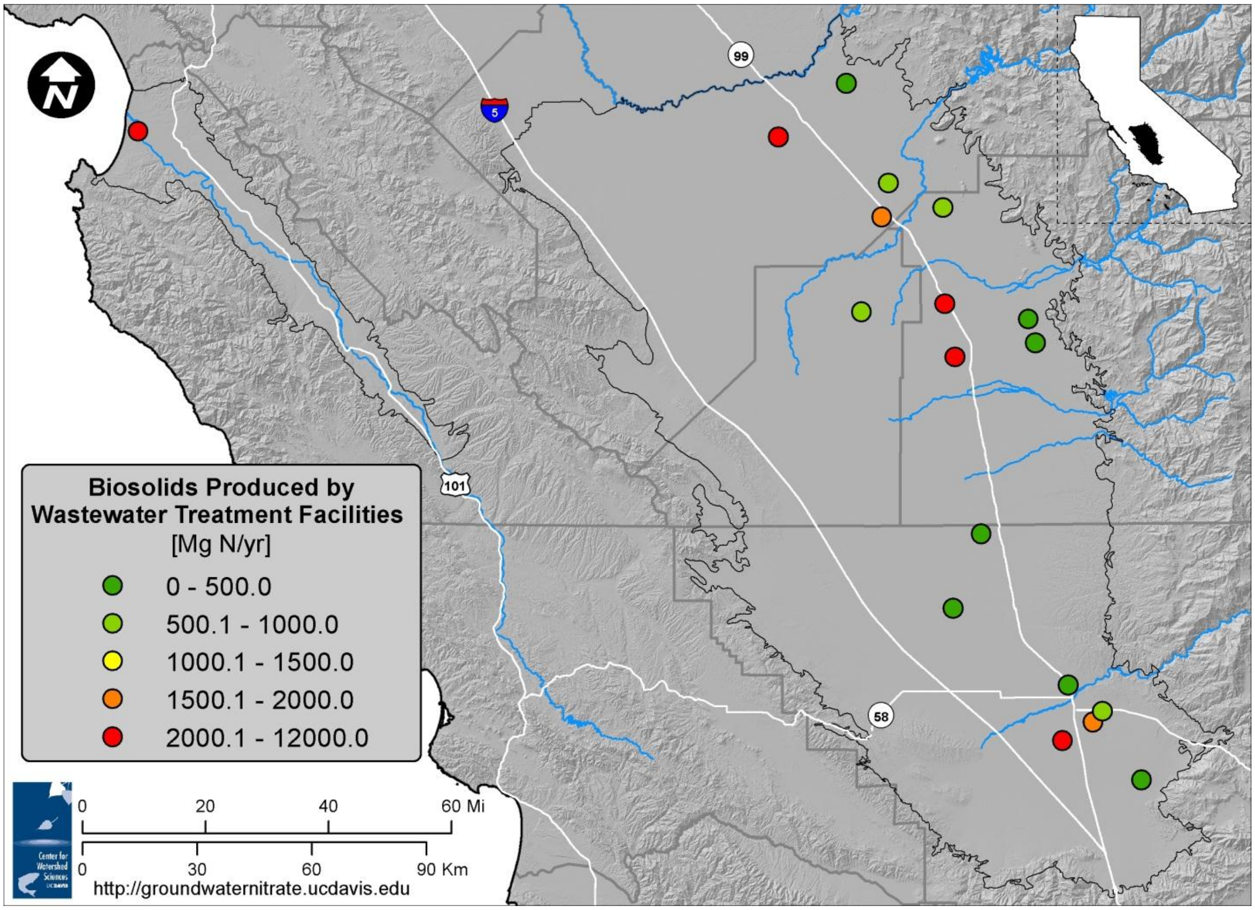

Figure 48. Annual metric tons of biosolids produced by surveyed WWTPs (~50% of facilities reporting, see Appendix Table 8 for facility specific information). (Source: California Water Boards, Contact with Facilities, WDRs.) [1 metric ton = 1.1 tons.]

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.

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).

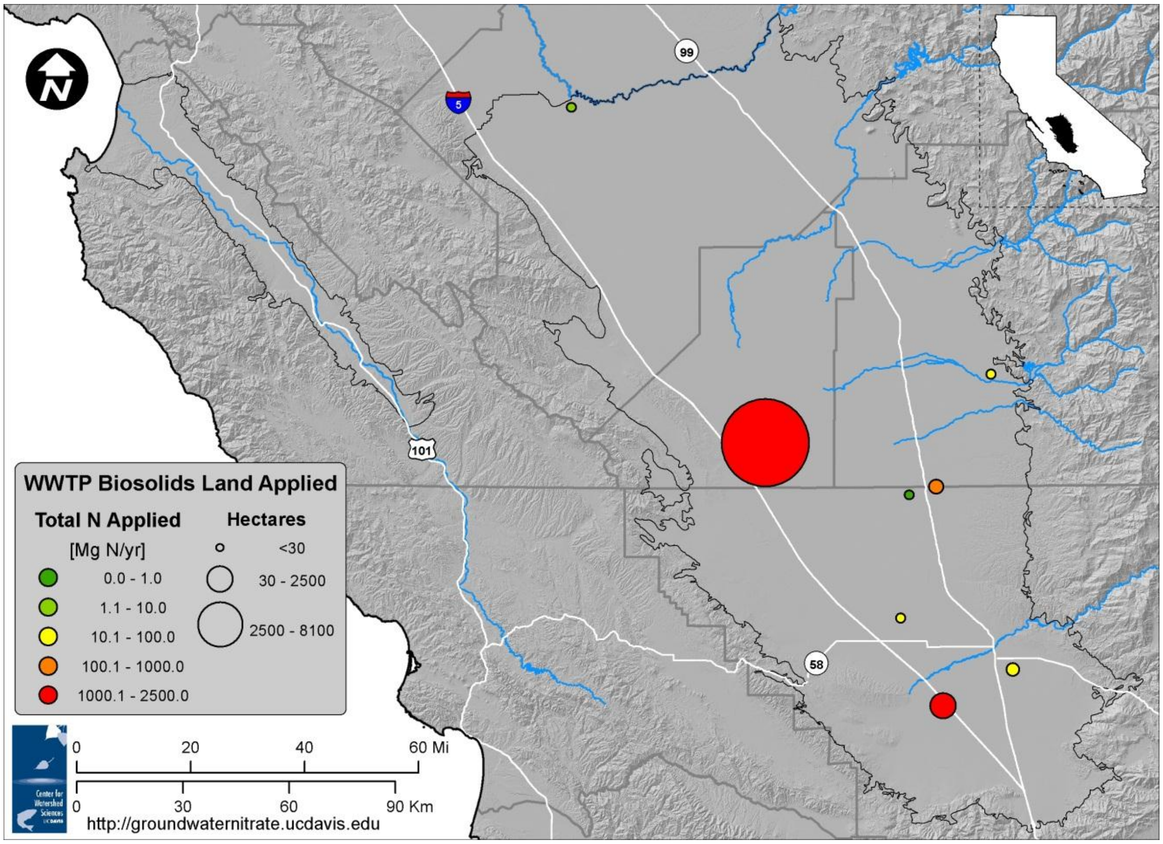

Figure 49. Large land application operations and WWTPs reporting direct land application of biosolids: Total annual mass of nitrogen [color] and hectarage in ha [diameter] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, U.S. EPA Region 9, Contact with Facilities, WDRs.) [1 metric ton = 1.1 tons, 1 hectare = 2.47 acres.]

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.

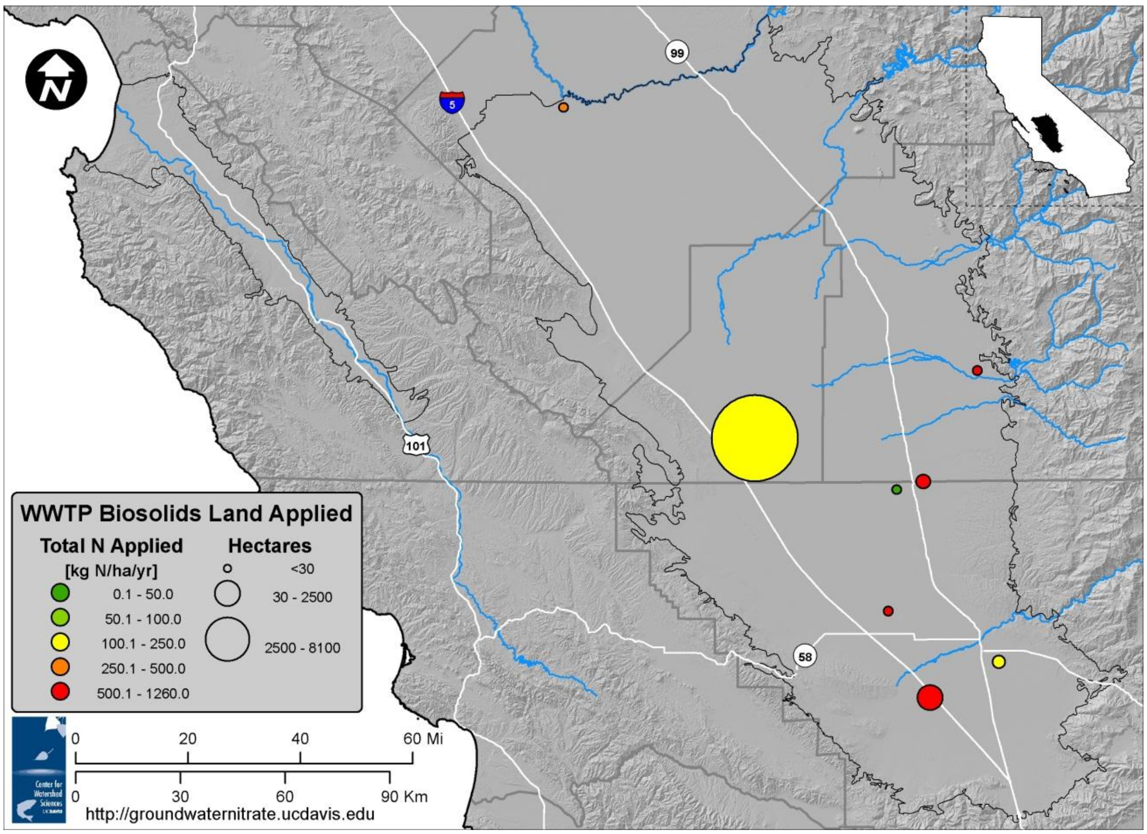

Figure 50. Large land application operations and WWTPs reporting direct land application of biosolids: kg N/ha/yr [color] and hectarage in ha [diameter] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, U.S. EPA Region 9, Contact with Facilities, WDRs.) [1 kg = ~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.

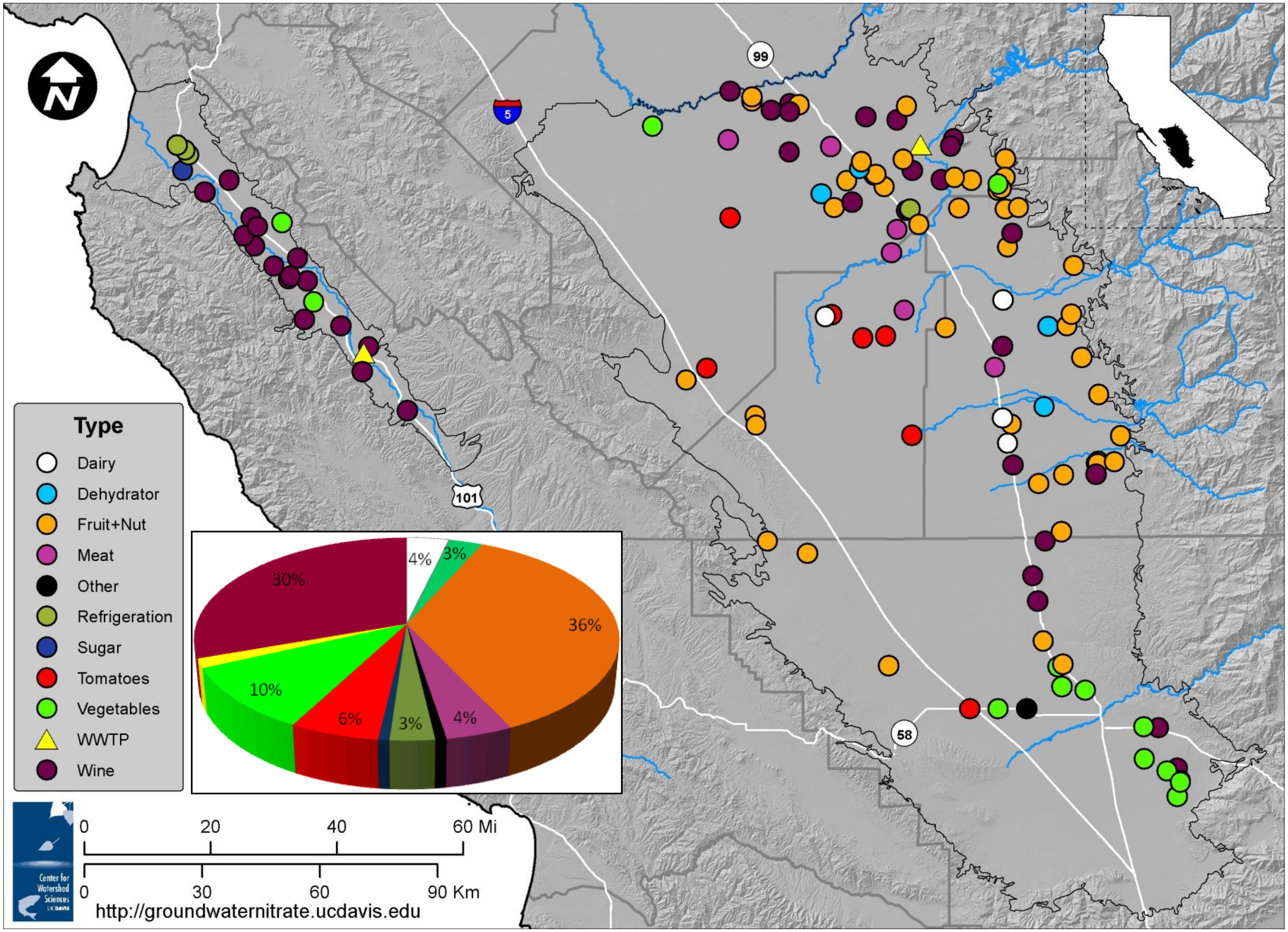

Figure 51. Food processor type distribution. The pie-chart shows the distribution of food processor type as a percentage of the total number of surveyed food processors (Source: California Water Boards, Geolocating by Address, WDRs.)

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).

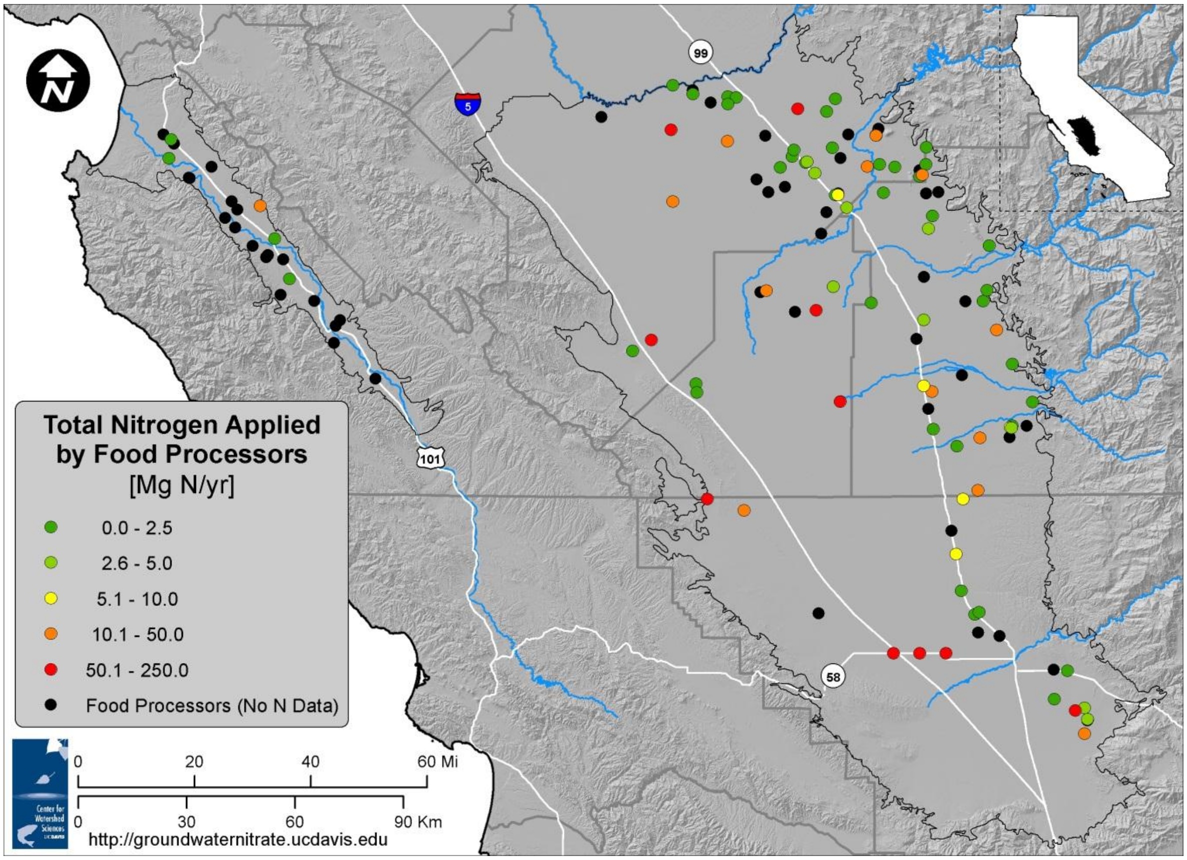

Figure 52. Total metric tons of nitrogen applied annually from food processor discharge (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [1 Mg = 1 metric ton = 0.001 Gg = 1.1 tons.]

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.

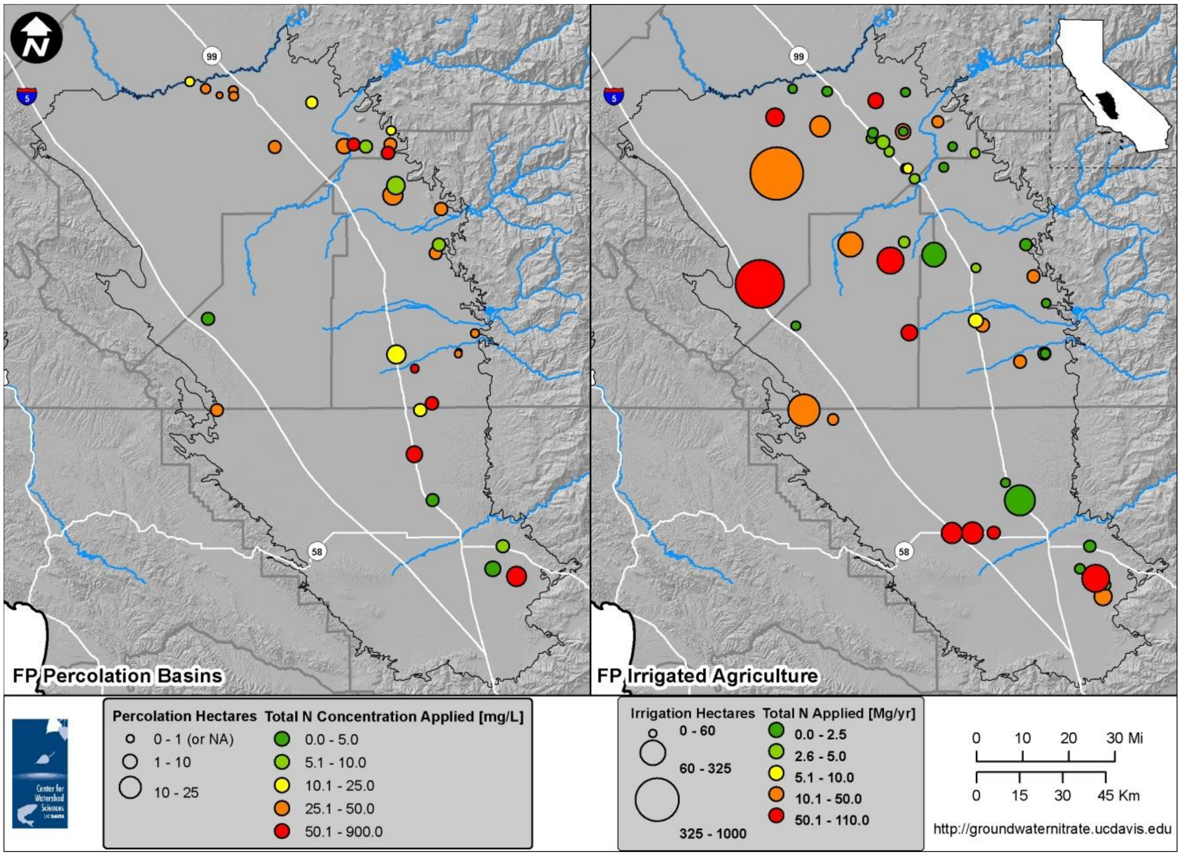

Figure 53. Tulare Lake Basin food processors: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) and total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: 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.]

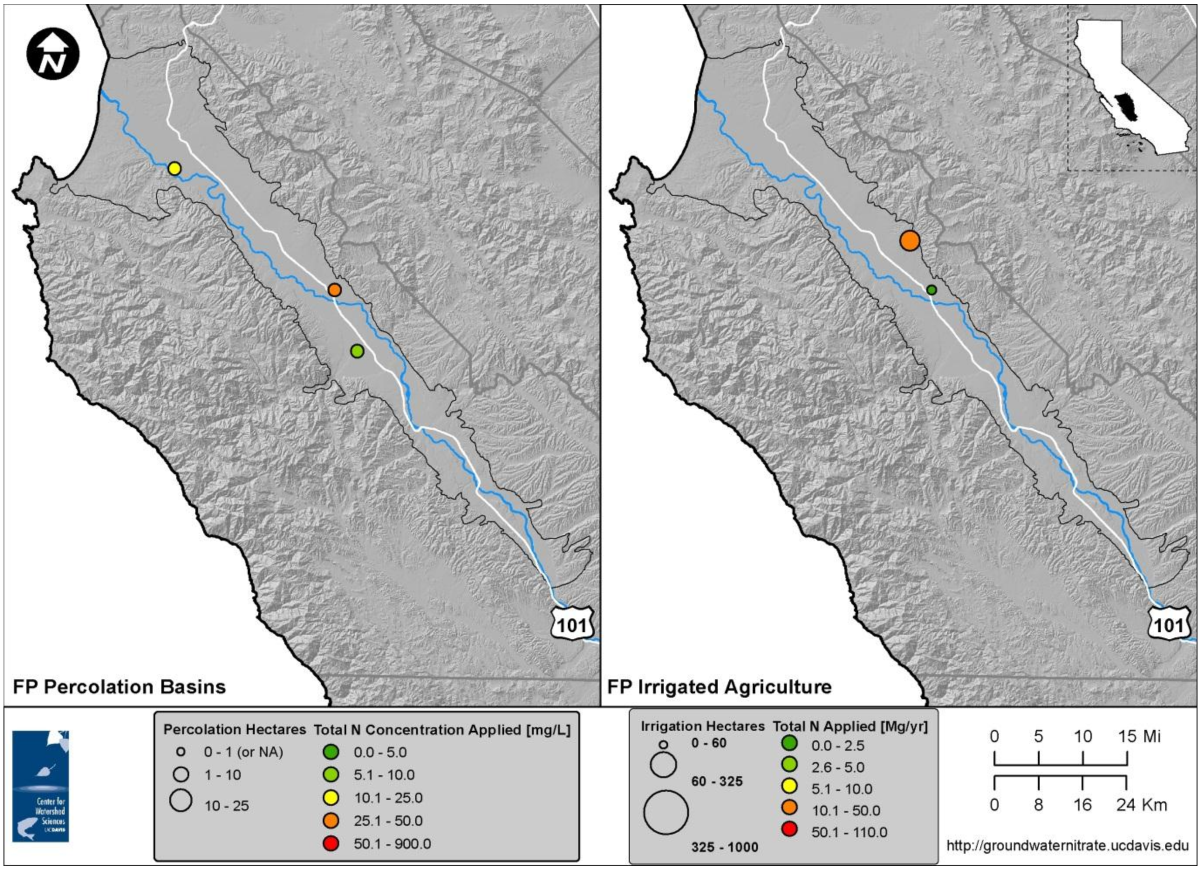

Figure 54. Salinas Valley food processors: Hectarage (ha) and total N concentration (mg/L) of discharge to percolation basins [left] and hectarage (ha) and total nitrogen applied (Mg N/Yr) of discharge to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: 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.]

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.

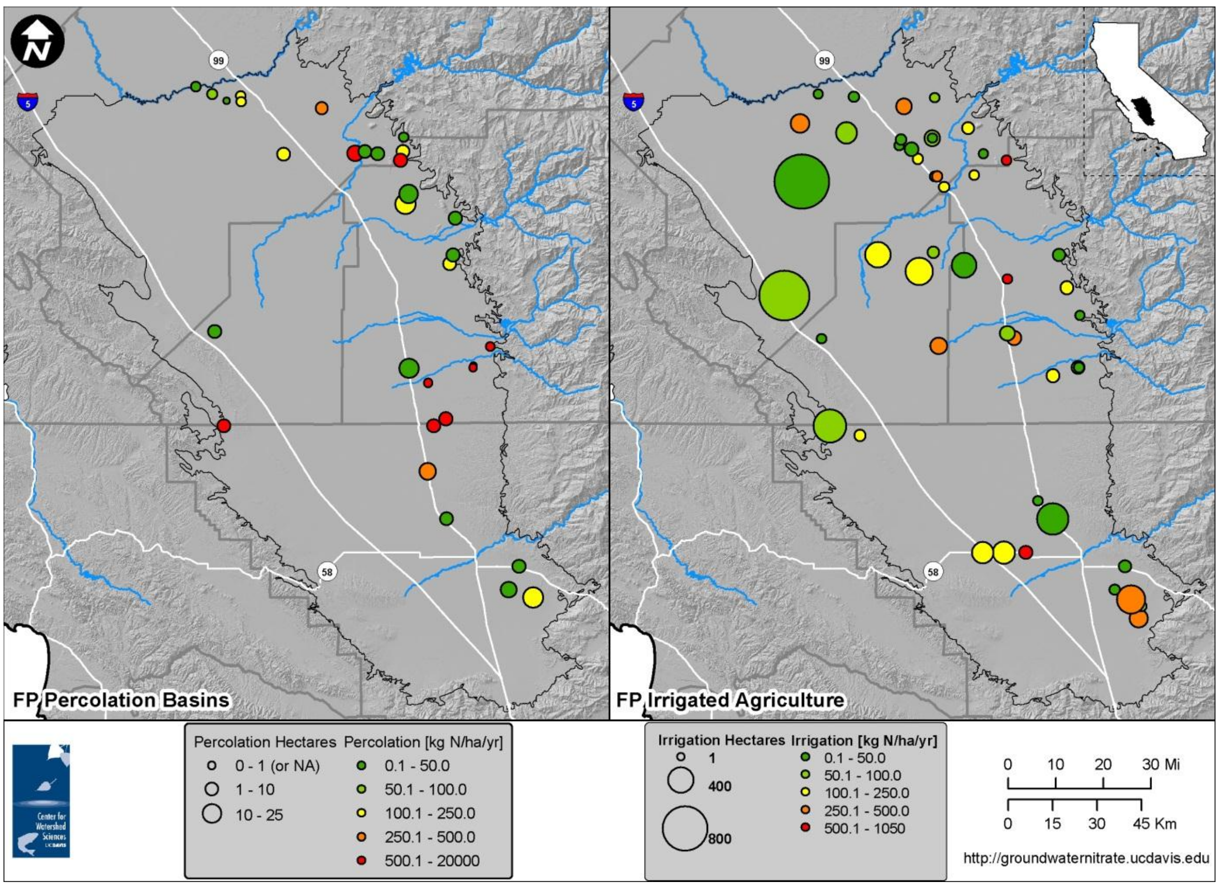

Figure 55. Tulare Lake Basin food processors: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [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.

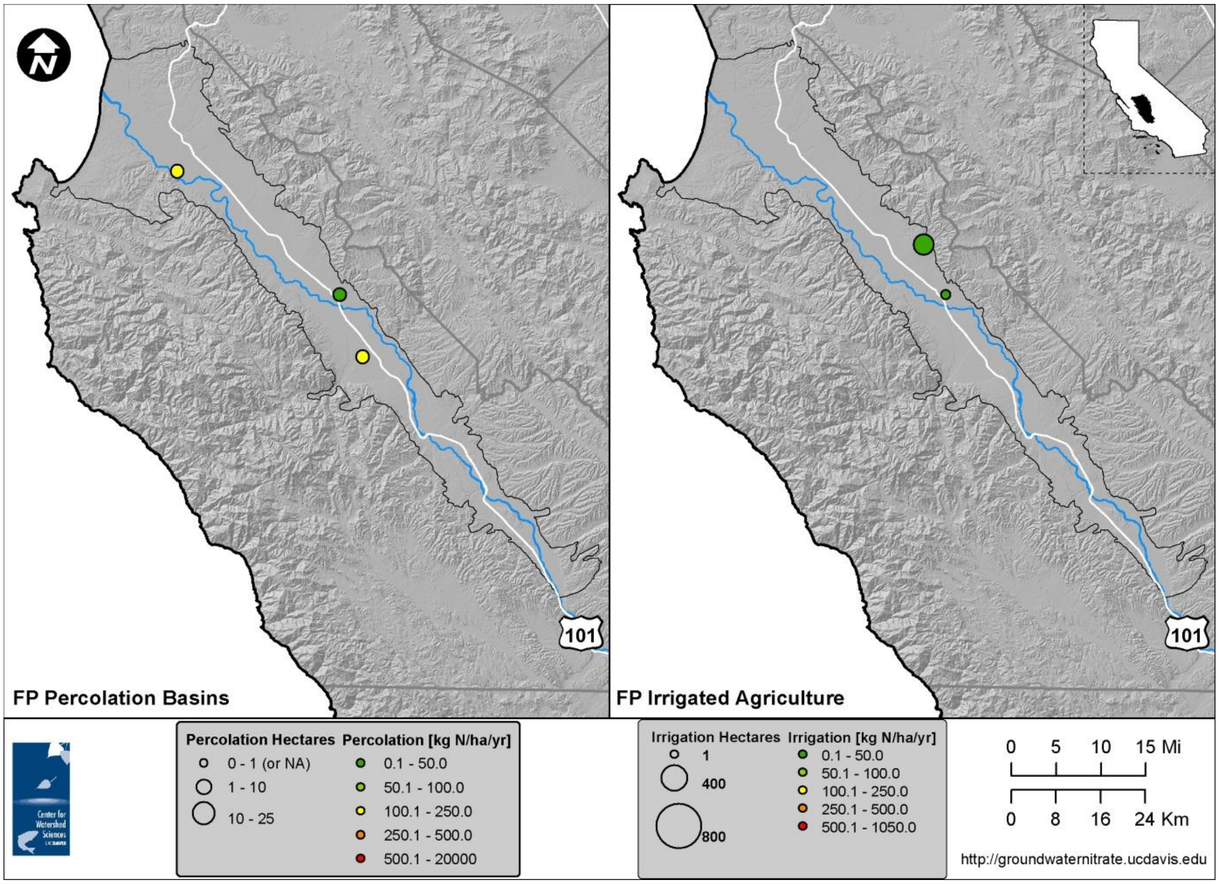

Figure 56. Salinas Valley food processors: Hectarage (ha) and kg N/ha/yr of applied nitrogen to percolation basins [left] and to irrigated agriculture [right] (see Appendix Table 8 for facility specific information). (Source: California Water Boards, Geolocating by Address, WDRs, Hilmar SEP Database.) [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 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).

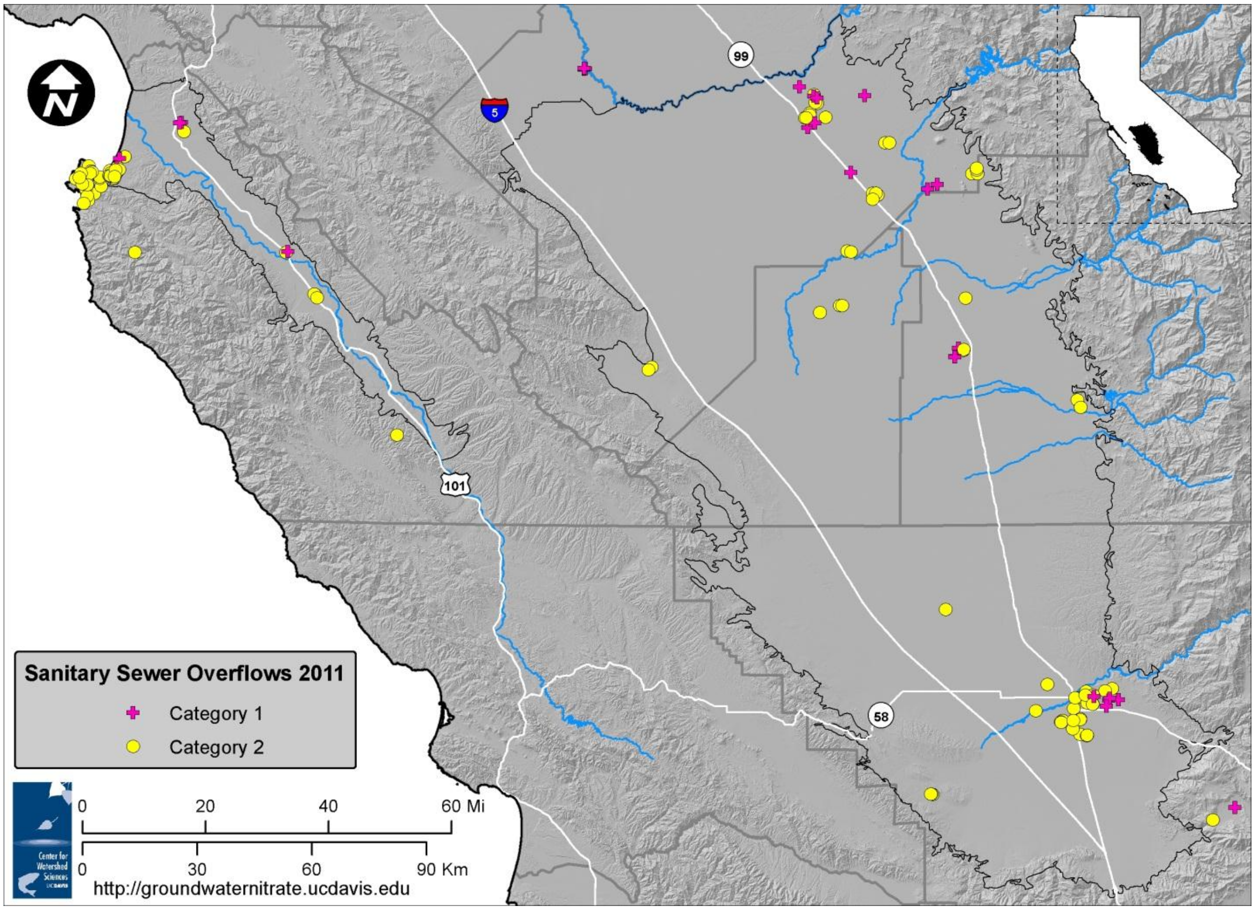

Figure 57. Sanitary sewer overflows in the Tulare Lake Basin and Salinas Valley reported in 2011. (Source: State Water Resources Control Board 2011c.)

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 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.

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 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 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 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.

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 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

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).

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 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):

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 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).

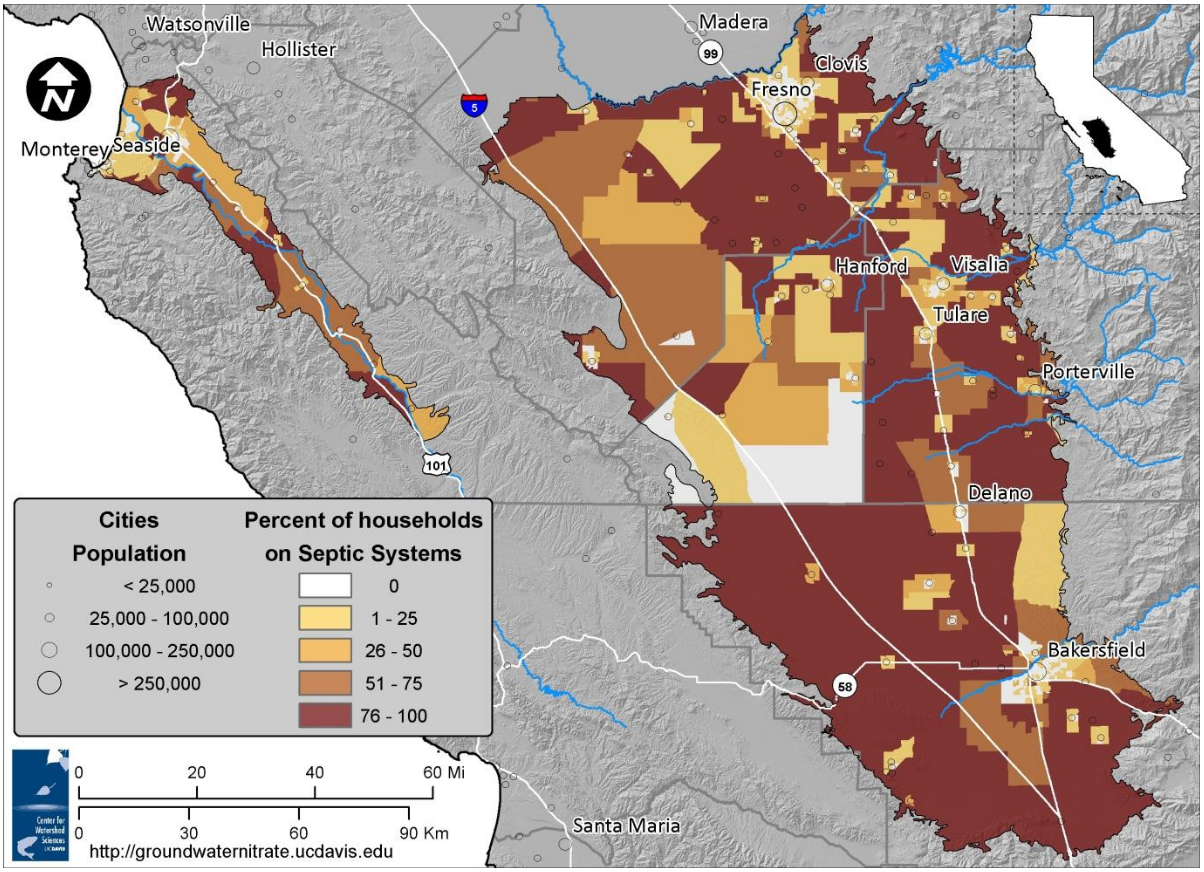

Figure 58. Percent of households on septic systems, by blockgroup, from 1990 census data, in the study area.

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.

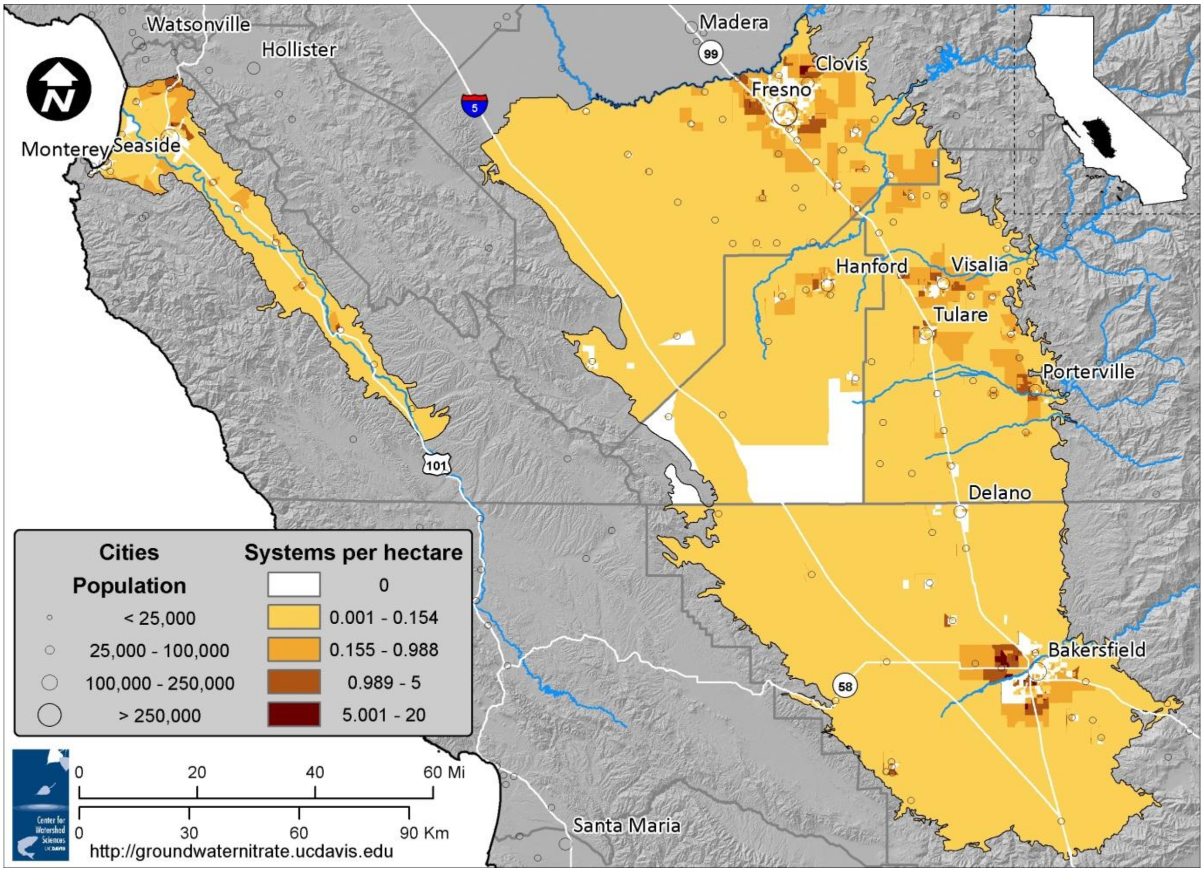

Figure 59. Septic systems per hectare, in the study area. Two thresholds are shown: 0.154 systems per ha (40 systems per square mile) and 0.99 systems per ha (1 system per acre), the maximum density allowed in some counties.

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).

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.

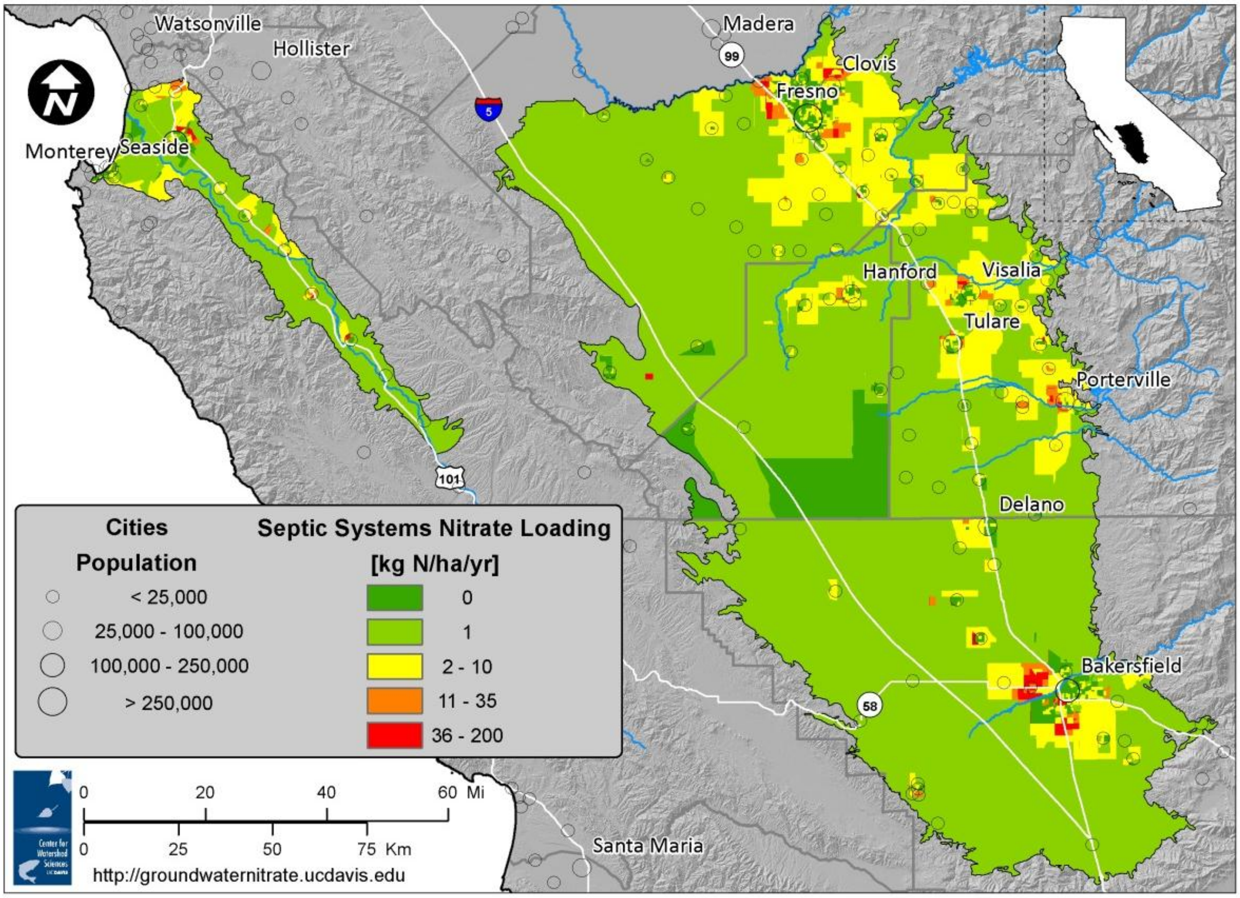

Figure 60. Septic-derived nitrogen leaching rates within the study area.

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.

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 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.

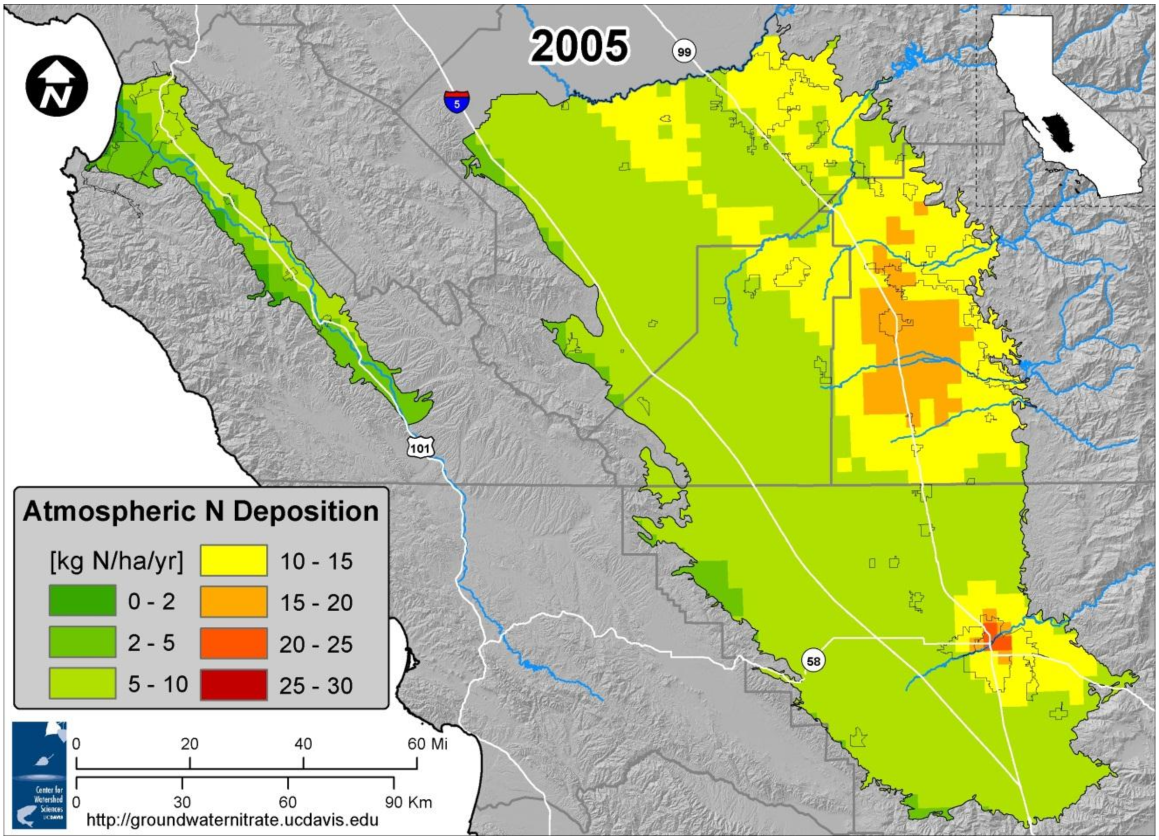

Figure 61. Atmospheric N deposition, spatially distributed across the study area, based on the Community Multiscale Air Quality model developed by U.S. EPA.

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 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.

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.

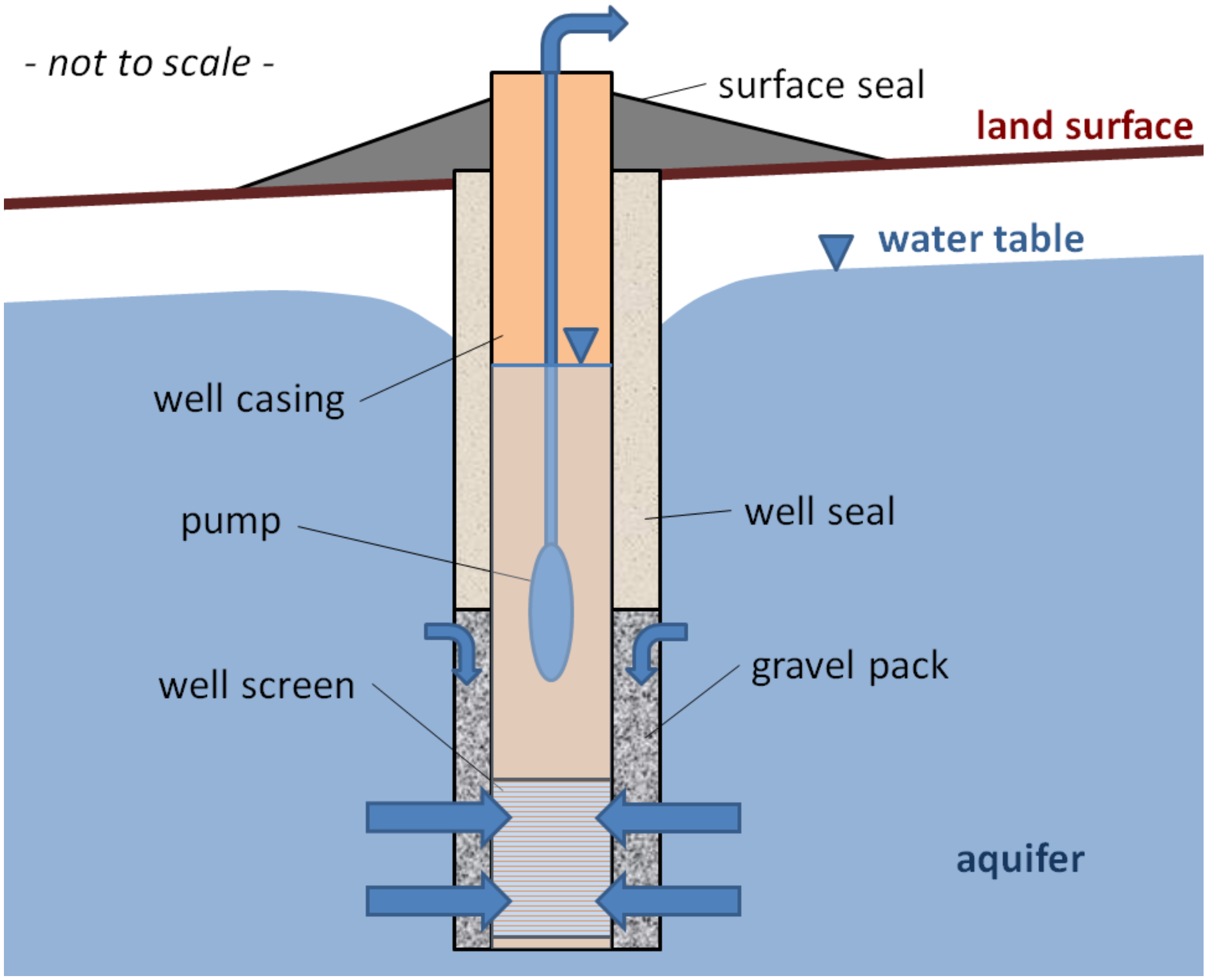

Figure 62. Schematic outline of a well. Well casing and the well screen are surrounded by the borehole annulus, which is packed with gravel materials around the screen and with well seal materials above the gravel pack. Water enters the well from the aquifer through the gravel pack and the well screen.

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 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/

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), 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.

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.

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

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 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 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

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.

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).

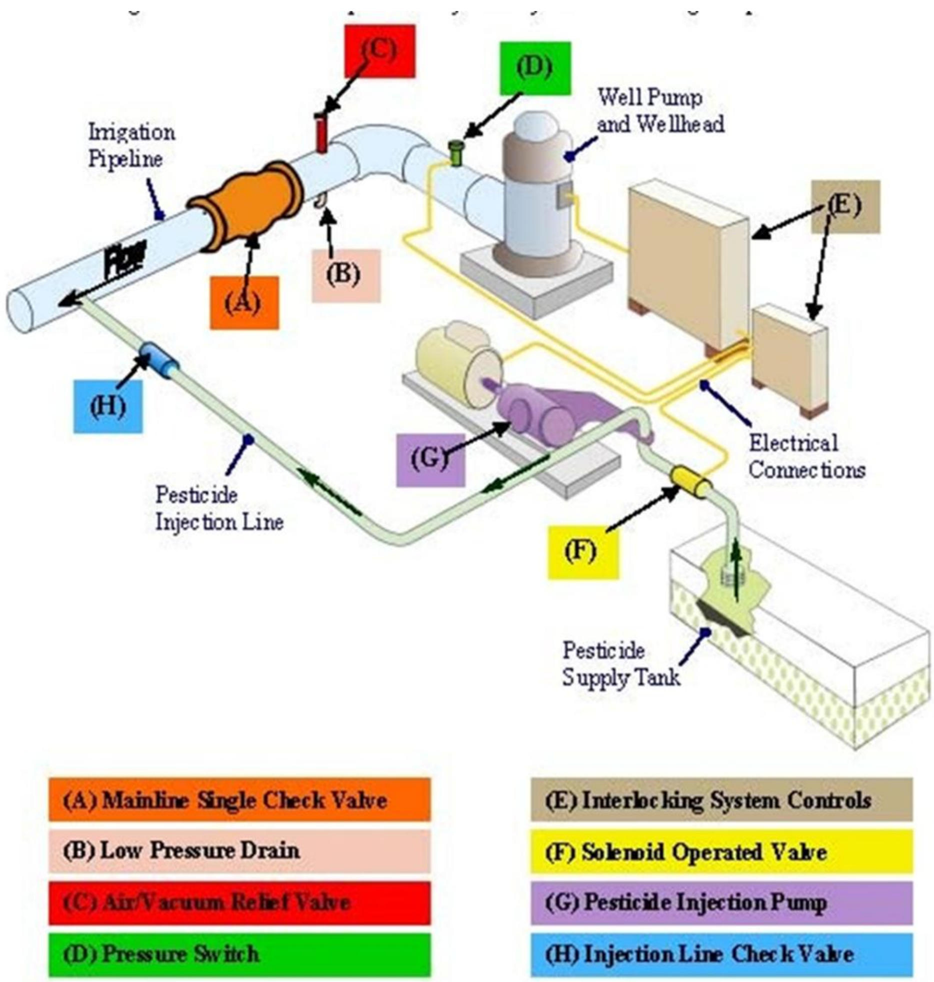

Figure 64. Possible backflow prevention device schematic (Zoldoske et al. 2004).

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 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).

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.

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 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

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, 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.

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 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.

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 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

- 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.

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**

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:

- 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.

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.

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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

	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

									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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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%

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%

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%

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%

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%

		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%

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%

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%

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%

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.

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

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	-	-	-	

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

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	-	-	-	

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	-	-	-	

ID [1]	County	Facility Name	Total [2] MGD	Total [2] mg N/L	Total [2] kg N/yr	To Irrigation [3] MGD	To Irrigation [3] Hectares	To Irrigation [3] kg N/yr	To Percolation [4] MGD	To Percolation [4] Hectares	To Percolation [4] kg N/yr	Notes [5]
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	

ID [1]	County	Facility Name	Total [2] MGD	mg N/L	kg N/yr	To Irrigation [3] MGD	Hectares	kg N/yr	To Percolation [4] MGD	Hectares	kg N/yr	Notes [5]
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.

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

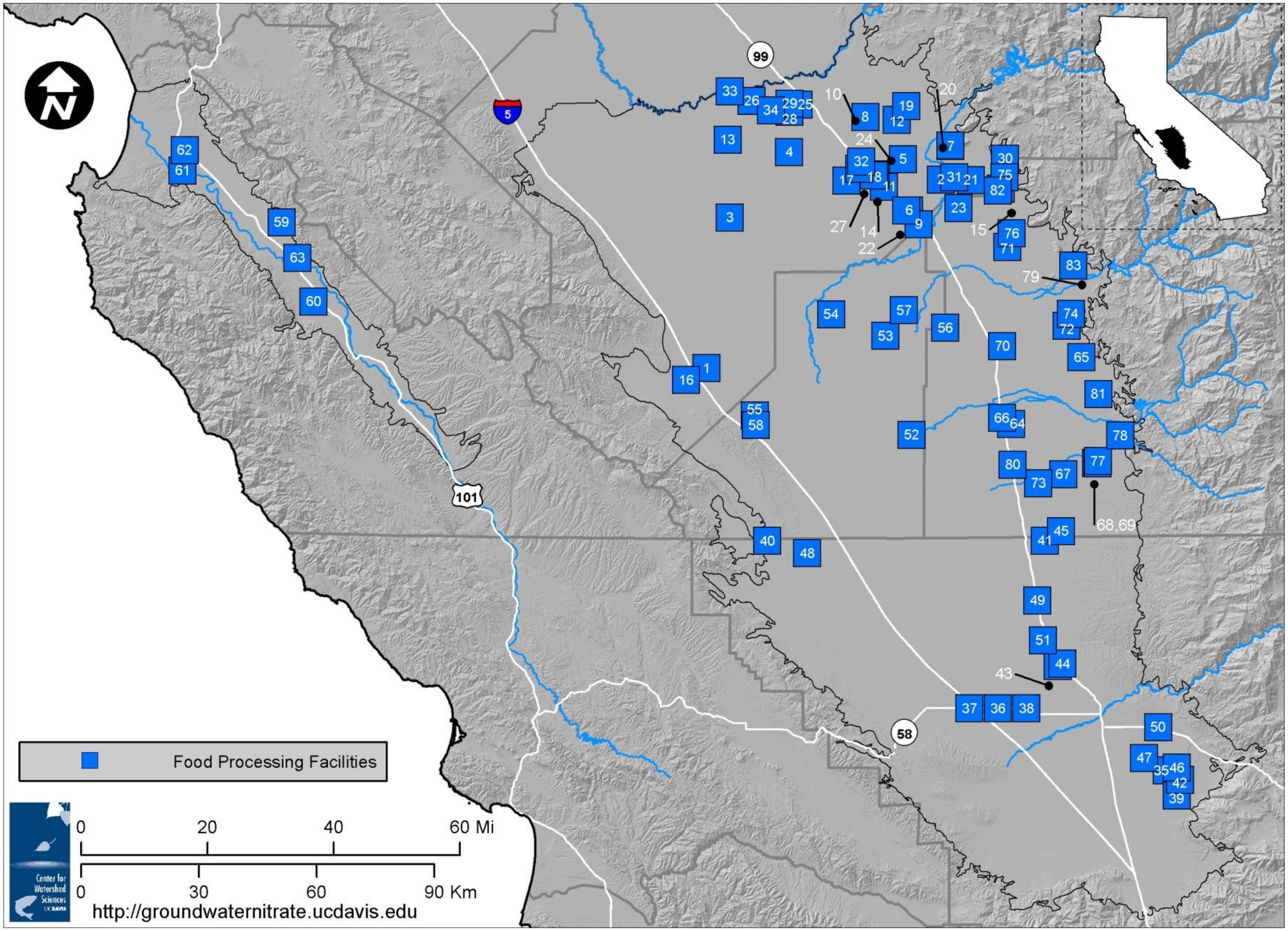
Year	Domestic	Agricultural	Municipal
1977	406	526	7
1978	263	340	5
1979	229	162	4
1980	167	100	3
1981	180	140	6
1982	70	55	4
1983	131	35	5
1984	191	72	5
1985	155	70	10
1986	194	46	8
1987	137	68	8
1988	215	96	10
1989	245	124	7
1990	281	168	21
1991	376	346	9
1992	367	295	13
1993	207	159	5
1994	191	145	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

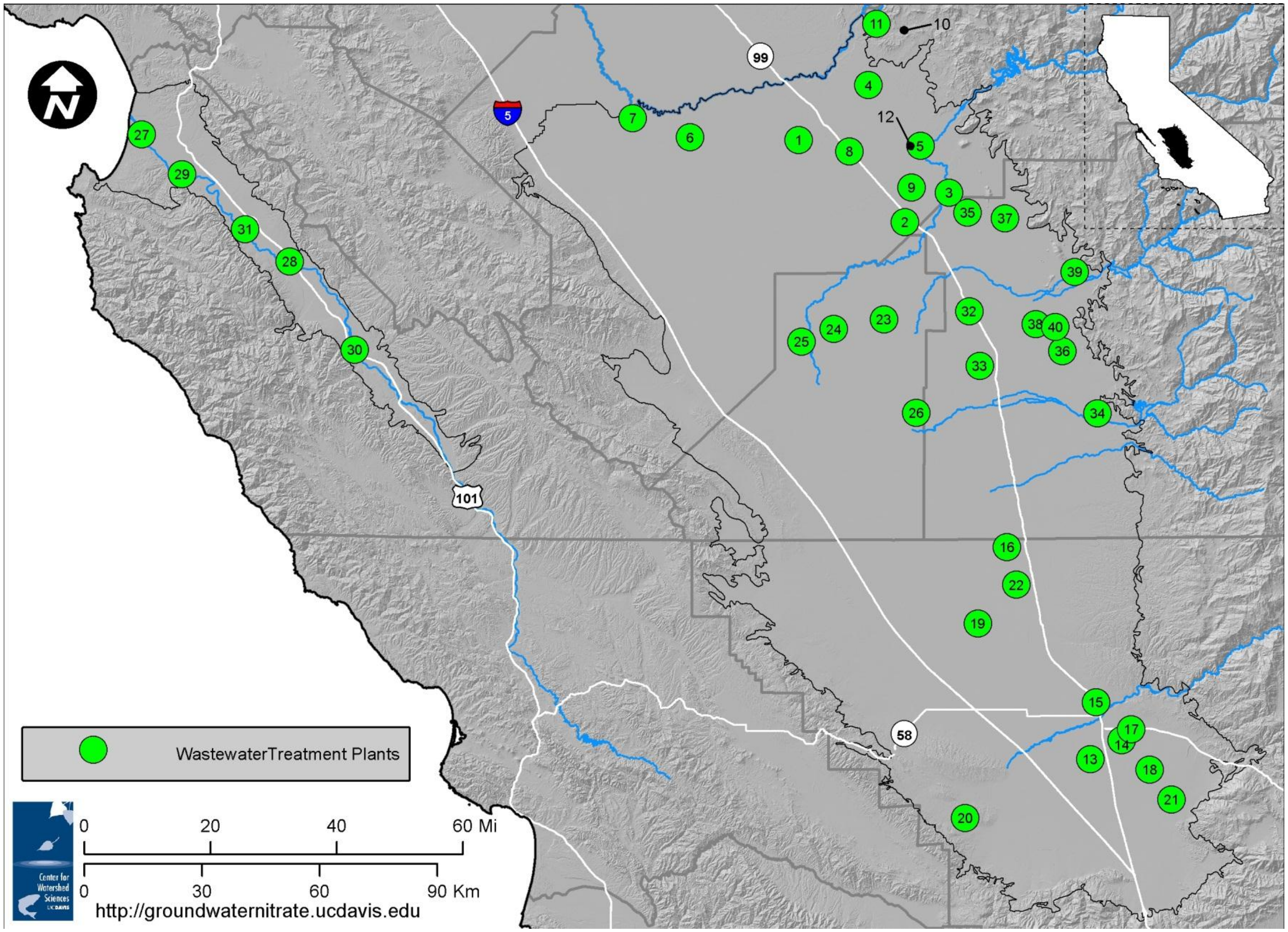
Year	Domestic	Agricultural	Municipal
1977	1,259	975	17
1978	993	609	22
1979	936	234	11
1980	736	188	15
1981	473	119	13
1982	324	96	8
1983	418	45	6
1984	408	55	7
1985	391	57	14
1986	419	34	7
1987	377	70	9
1988	470	99	12
1989	469	151	16
1990	611	200	29
1991	737	313	32
1992	769	223	56
1993	427	129	28
1994	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**



Appendix Figure 1. Food processor locations corresponding with facility numbering in Appendix Table 8. (Source: California Water Boards, Geolocating by Address, WDRs.)



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