Addressing Nitrate in California's Drinking Water

TECHNICAL REPORT 4:

Groundwater Nitrate Occurrence

With a Focus on Tulare Lake Basin and Salinas Valley Groundwater

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

California Nitrate Project,
Implementation of Senate Bill X2 1

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

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Acronyms and Abbreviations

ADE	Advection Dispersion Equation
AMBAG	Association of Monterey Bay Area Governments
ANOVA	Analysis of Variance
APN	Assessor's Parcel Numbers
AW	Applied Water
BMPs	Best Management Practices
BOR	Bureau of Reclamation
BTC	Breakthrough Curve
C2VSIM	California Central Valley Groundwater-Surface Water Simulation Model
Ca^{2+}	Calcium
CADWSAP	California Drinking Water Source Assessment Program
CAML	California Augmented Multisource Landcover
CASTING	California Ambient Spatio-Temporal Information on Nitrate in Groundwater
CDPH	California Department of Public Health
CDS	Compliance Discharge Surface
CFC	Chlorofluorocarbon
CIMIS	the California Irrigation Management Information System
CIP	Canal Integration Program
Cl^-	Chloride
CO_3^{2-}	Carbonate
COMSOL	Multiphysics Finite Element Analysis Simulation Software
CVHM	Central Valley Hydrologic Model
CVRWB	Central Valley Regional Water Board
CV-SALTS	Central Valley Salinity Alternatives for Long-Term Sustainability
CWS	Community Water System
DAU	Detailed Analysis Units
DBCP	Dibromochloropropane
DD	Domain Decomposition
DGW	Depth to Groundwater
DIP	Distribution Integration Program
DLPA	DWR Division of Local Planning and Assistance
DOC	Dissolved Organic Carbon
DPR	Department of Pesticide Regulations
DSR	Data Summary Reports
DWR	California Department of Water Resources
ecdfs	Empirical cumulative distribution functions
E-Clay	Corcoran Clay
EPA	(U.S.) Environmental Protection Agency
ET	Evapotranspiration

Eto	Evapotranspiration
FEFLOW	Simulation model for subsurface flow and transport processes
FEM	Finite Elements Method
FKC	Friant-Kern Canal
FREDSIM	FRiant Economics-Driven SIMulation model
GAMA	California State Water Board's Groundwater Ambient Monitoring and Assessment
GIS	Geographic Information system
GLEAMS	Ground Water Loading Effects of Agricultural Management Systems
GPS	Geographic Positioning System
GWQB	Ground Water Quality Bureau (New Mexico)
HCO_3^-	Bicarbonate
HR	Hydrologic Region
ICE	UC Davis Information Center for the Environment
IGSM	Integrated Groundwater and Surface-Water Model
IPCC	Intergovernmental Panel on Climate Change
IrrEff	Irrigation Efficiency
K^+	Potassium
KCWA	Kern County Water Agency
KDWCD	Kaweah Delta Water Conservation District
KRCD	Kings River Conservation District
K_C	standardized crop coefficient
L	liter
LPA	Local Primacy Agency
LR	Leaching Requirement
MAF	Million Acre-feet
MassDEP	Massachusetts Department of Environmental Protection
MCEM	MCWRA Agricultural Environmental Monitoring Program
MCL	Maximum Contaminant Level
MCWRA	Monterey County Water Resources Agency
MCDEH	Monterey County Department of Environmental Health
mg	milligram (one-thousandth of one gram)
µg	microgam (one-millionth of one gram)
Mg^{2+}	Magnesium
Na^+	Sodium
NASS	National Agricultural Statistics Service
NAWQA	National Water-Quality Assessment
ND	Non-detect
NMED	New Mexico Environment Department
NO_3^-	Nitrate
NOAA	National Oceanic and Atmospheric Administration
NPS	Nonpoint Source

NPSAT	Non-Point Source Assessment Tool
NTNCWS	Non-Transient Non-Community Water System
NWIS	National Water Information System (USGS)
PA	Principal Aquifers
Pdfs	Probability Distribution Functions
PICME	Permits, Inspections, Compliance, Monitoring, and Enforcement Database, CDPH
PLSS	Public Land Survey System
ppm	Parts per million
Precip	Precipitation
PS-Code	Primary Station Code
PWS	Public Water System
PWTA	Private Well Testing Act
RK	Regional Kendall
RWQCB	Regional Water Quality Control Board
SDWA	Safe Drinking Water Act
SID	State ID
SJV	San Joaquin Valley
SO_4^{2-}	Sulfate
SOM	Soil Organic Matter
STORET	"STOrage and RETrieval", EPA's database of water quality information
SV	Salinas Valley
SVB	Salinas Valley Basin
SVIGSM	Salinas Valley Integrated Ground Water and Surface Model
SWAP	Source Water Assessment Programs
SWN	State Well Number
SWP	State Water Project
SWRCB	State Water Resources Control Board
SWS	Small Water System
TDS	Total Dissolved Solids
TLB	Tulare Lake Basin
TNCWS	Transient Non-Community Water System
U.S. EPA	United State Environmental Protection Agency
URF	Unit Response Functions
USBR	United States Bureau of Reclamation
USDA	United States Department of Agriculture
USGS	United States Geological Survey
WGA	Well Generation Algorithm
WWD	Westlands Water District
YR	year

Unit Conversions

Metric to US		US to Metric	
Mass		*Mass*	
1 gram (g)	0.04 ounces (oz)	1 ounce	28.35 grams
1 kilogram (kg)	2.2 pounds (lb)	1 pound	0.45 kilograms
1 megagram (Mg) (1 tonne)	1.1 short tons	1 short ton (2000 lb)	0.91 megagrams
1 gigagram (Gg) (1000 tonnes)	1102 short tons	1000 short tons	0.91 gigagrams
Distance		*Distance*	
1 centimeter (cm)	0.39 inches (in)	1 inch	2.54 centimeters
1 meter (m)	3.3 feet (ft)	1 foot	0.30 meters
1 meter (m)	1.09 yards (yd)	1 yard	0.91 meters
1 kilometer (km)	0.62 miles (mi)	1 mile	1.61 kilometers
Area		*Area*	
1 square meter (m^2)	10.8 square feet (ft^2)	1 square foot	0.093 square meters
1 square kilometer (km^2)	0.39 square miles (mi^2)	1 square mile	2.59 square kilometers
1 hectare (ha)	2.8 acres (ac)	1 acre	0.40 hectares
Volume		*Volume*	
1 liter (L)	0.26 gallons (gal)	1 gallon	3.79 liters
1 cubic meter (m^3) (1000 L)	35 cubic feet (ft^3)	1 cubic foot	0.03 cubic meters
1 cubic kilometer (km^3)	0.81 million acre-feet (MAF, million ac-ft)	1 million acre-feet	1.23 cubic kilometers
Farm Products		*Farm Products*	
1 kilogram per hectare (kg/ha)	0.89 pounds per acre (lb/ac)	1 pound per acre	1.12 kilograms per hectare
1 tonne per hectare	0.45 short tons per acre	1 short ton per acre	2.24 tonnes per hectare
Flow Rate		*Flow Rate*	
1 cubic meter per day (m^3/day)	0.296 acre-feet per year (ac-ft/yr)	1 acre-foot per year	3.38 cubic meters per day
1 million cubic meters per day (million m^3/day)	264 mega gallons per day (mgd)	1 mega gallon per day (694 gal/min)	0.0038 million cubic meters/day

Nitrate Units

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

To convert from:

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

Summary

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

Of the 19,000 wells, approximately 2,500 are frequently sampled public water supply wells (over 60,000 samples). Apart from the recently established Central Valley dairy regulatory program, which now monitors about 4,000 domestic and irrigation wells in the Tulare Lake Basin, there are no regular well sampling programs for domestic and other private wells. These latter are sampled sporadically by county agencies and through research programs.

From 2000 to 2011, the median nitrate concentration in the Tulare Lake Basin and Salinas Valley public water supply well samples was 23 milligrams per liter (mg/L) and 21 mg/L (as nitrate), respectively, and in all reported non-public well samples, 23 mg/L and 20 mg/L (as nitrate), respectively. In public supply wells, about one in ten raw water samples exceed the nitrate maximum contaminant level (MCL) of 45 mg/L (as nitrate). Nitrate concentrations in wells vary widely with location and well depth. More domestic wells and unregulated small system wells than public supply wells have high nitrate concentrations due to their shallow depth. The highest nitrate concentrations are found in wells of the alluvial fans in the eastern Tulare Lake Basin and in wells of unconfined to semi-confined aquifers in the northern, eastern, and central Salinas Valley. In the Kings, Kaweah, and Tule River groundwater subbasins of Fresno and Kings County, and in the Eastside and Forebay subbasins of Monterey County, one-third of domestic or irrigation wells exceed the nitrate MCL. Consistent with these findings, the maximum nitrate level, measured in any given land section (1 square mile) for which nitrate data exist between 2000 and 2009, exceeds the MCL across wide portions of these areas. Low nitrate concentrations tend to occur in the deeper, confined aquifer in the western and central Tulare Lake Basin.

Nitrate levels have not always been this high. While no significant trend is observed in some areas with low nitrate (e.g., areas of the western TLB), USGS research indicates significant long-term increases in the higher-nitrate areas of the Tulare Lake Basin, which is consistent with the CASTING dataset. Average nitrate concentrations in public supply wells of the Tulare Lake Basin and Salinas Valley have increased by 2.5 mg/L (±0.9 mg/L) per decade over the past three decades. Average trends of similar magnitude are observed in private wells. As a result, the number of wells with nitrate above background levels (> 9 mg/L) has steadily increased over the past half century from one-third of wells in the 1950s to nearly

two-thirds of wells in the 2000s. Due to the large increase in the number of wells tested across agencies and programs, the overall fraction of sampled wells exceeding the MCL grew significantly in the 2000s.

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

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

Denitrification (the natural attenuation of nitrate) is most likely to occur in fine-grained anoxic clay layers, the most prominent of which is the Corcoran Clay separating the upper semi-confined to unconfined aquifer from the lower confined aquifer along and next to both sides of the trough of the Tulare Lake Basin; the several thousand feet thick clay and silt units underlying the former bed of the Tulare Lake south of Hanford; and clay units confining the Pressure aquifer system in the northwestern Salinas Valley. These clay layers have limited effect on most groundwater production wells, which generally obtain most of their water through coarser sand and gravel aquifer sediments that are connecting recharge areas with wells. In very shallow groundwater discharge areas to surface water, denitrification may occur where significant sources of organic carbon are present, e.g., in riparian and marshland areas along the valley trough of the Salinas Valley and Tulare Lake Basin. Some removal of nitrate by denitrification may also occur in deeper reducing aquifer sediments occurring typically at more than 500 to over 1,000 feet depth throughout the Tulare Lake Basin. However, due to the large age of groundwater in deep anoxic zones, it is currently uncertain, to which degree anoxic conditions may slow down or prevent future nitrate pollution at that depth.

1 Introduction

Ninety-eight percent of the population in the Tulare Lake Basin and Salinas Valley rely on groundwater as a source of drinking water. Most of the study area's residents are provided drinking water through public water supply systems or small water systems that own and operate groundwater wells. About one in ten residents obtain their drinking water from private domestic wells. An assessment of current and future quality of groundwater used as drinking water is of critical interest to the public, and to local, state, and federal agencies charged with protecting water resources and providing safe drinking water.

Nitrate has long been known to be a widespread groundwater pollutant in California's groundwater basins. It is associated with fertilizer use, land application of manure and organic wastes, and disposal of urban and domestic sewage. The objective of Technical Report 4 is to review our current understanding of nitrate in groundwater in the Tulare Lake Basin and the Salinas Valley, and to provide a comprehensive assessment of past, current, and future distribution of nitrate.

This report begins with a review of the general characteristics of the Tulare Lake Basin and Salinas Valley physiography, hydrology, hydrogeology, and water quality (Sections 2 and 3). These sections describe and summarize the known occurrence and trends of nitrate in groundwater as described in research studies and government reports over the past half century.

In this study, we also made an effort to collect a comprehensive inventory of data on nitrate occurrence described in reports, housed in county and districts offices, available from state and federal databases, or through individual research groups. In Section 4, we describe the methods used to assemble the CASTING (California Ambient Spatio-Temporal Information on Nitrate in Groundwater) database. The database provides an efficient interim electronic dataset platform for the research team to perform spatial and trend analysis, to map nitrate occurrence, and - after further completion and quality control - for the transfer of the data to the State Water Resources Control Board's publicly accessible Geotracker GAMA groundwater quality database.

The database includes data from publicly accessible data sources as well as data that have never before been part of a thorough regional water quality evaluation, providing an opportunity to update previous studies and assessments, many of which have focused on local areas. With the CASTING dataset, we perform a statistical evaluation of historic and current nitrate occurrence across the study area (Section 5).

Field data are useful to understand current water quality and historic trends, where such data exist. But models are needed to assess groundwater quality where measurements have not been made, and to assess and predict future dynamics of groundwater nitrate, which result from past, current, and future nitrate loading to groundwater. Technical Report 2 (Viers et al. 2012) describes a century of nitrate loading, 1945 to 2050, distributed across the entire study area, reflecting the large spatial and temporal variability in nitrate loading. In this report, we use these data, in conjunction with a newly developed groundwater modeling tool to predict groundwater quality across the study area over the next 40 years. In Section 6 we develop a method to estimate nitrate travel times from the root zone (the point of reference for the nitrate loading calculations described in Technical Report 2, ibid.) through the vadose zone and into groundwater. Section 7 describes the development of a new groundwater modeling tool, specifically to evaluate pollution from diffuse non-point sources in large number of wells distributed over a large aquifer system such as the Tulare Lake Basin. The model simulates the temporal dynamics of groundwater nitrate at thousands of irrigation and drinking water supply wells, from 1945 to 2050. Simulation results for the last 60 years are compared to data, where available to validate the modeling tool.

The appendices contain a general overview of the potential role of denitrification in naturally attenuating groundwater nitrate as well as information on domestic well monitoring programs across the United States. Denitrification is an important natural attenuation process. Further research is needed to determine the long-term (multi-decadal) influence of denitrification on groundwater nitrate distribution, particularly in the confined aquifer system of the central and western Tulare Lake Basin.

2 Tulare Lake Basin Hydrogeology and General Water Quality

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2.1 Physical Setting

2.1.1 Location

The Central Valley of California ("Central Valley") covers a large and central portion of the state of California. The total area is approximately 58,000 km^2 (22,500 square miles), with a length of over 700 km (450 miles) and a width of 60 to 100 km (40 to 60 miles). It is characterized by minimal topographic expression, an essentially flat region with surface slopes less than 1% throughout most of its area. The Central Valley is further divided into its two major watersheds, the Sacramento Valley in the north and San Joaquin Valley in the south. The San Joaquin Valley is further divided into the northern externally-draining San Joaquin River Basin and the southern internally-draining Tulare Lake Basin (TLB). The San Joaquin Basin drains into the Sacramento – San Joaquin River Delta (the Delta), by way of the San Joaquin River. Rivers entering the endorheic[2] TLB terminate in Tulare Lake, which has been drained and reclaimed for irrigated agriculture since the mid-20th century. The TLB is the widest part of the Central Valley (Figure 1).

[2] Endorheic refers to a closed basin which has no outflow to other bodies of water.

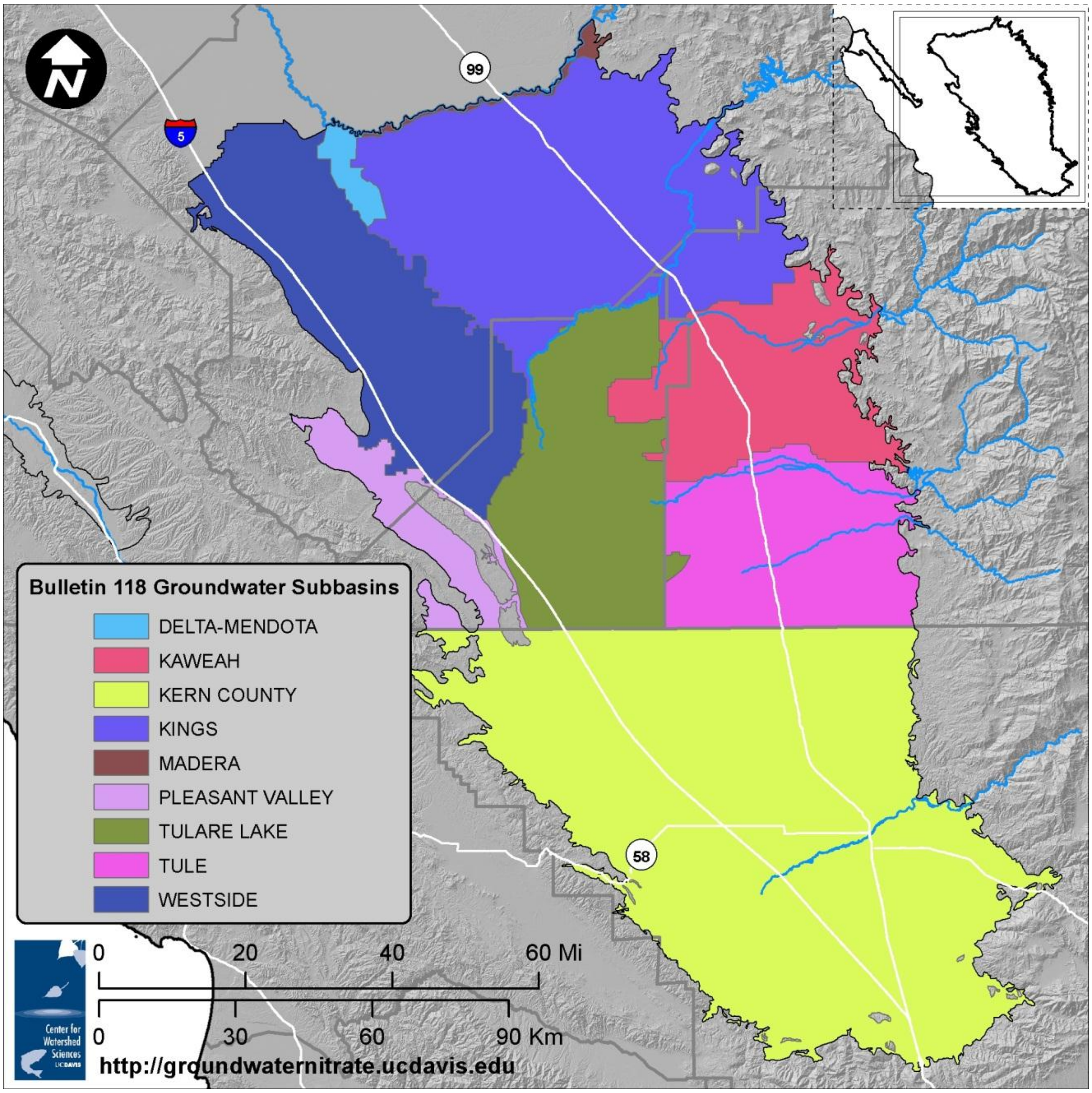

Figure 1. Hydrologic subbasins of the TLB, California. (Source: DWR 2003.)

2.1.2 Hydrologic Subbasins

The TLB comprises over 20,000 km^2 (8,000 square miles) of the valley portion of the Tulare Lake Hydrologic Region (HR) as defined in California Department of Water Resources (DWR) Bulletin 118 (California Department of Water Resources 2003). The northern boundary of the TLB is defined by the westward flowing San Joaquin River upstream and east of the city of Firebaugh and by a shallow watershed divide to the west of Firebaugh. The Kettleman Hills and the Temblor Range of the Coast Ranges of California form the western boundary of the TLB. To the south, the TLB is bordered by the Tehachapi Mountains. The Sierra Nevada foothills form the eastern boundary. The crystalline bedrock formations of the Sierra Nevada slope in a southwesterly direction beneath the sediments of the TLB, forming its lower boundary. The TLB is divided into subbasins based on geology, hydrologic barriers, and institutional boundaries. The groundwater subbasins within the study area are listed in Table 1 and shown in Figure 1. Over the past century, all major rivers in the TLB have been regulated with dams and reservoirs located along the Sierra Nevada foothills. Since the mid-20th century, most stream runoff has been used for irrigation and lakes have been drained and used for irrigated agricultural production.

Table 1. Subbasins within the TLB, California. (Source: DWR 2003.)

Subbasin Name	Subbasin Number	Subbasin Area km^2 [mi^2]
Kings	5-22.08	3,950 [1,530]
Westside	5-22.09	2,590 [1,000]
Pleasant Valley	5-22.09	588 [227]
Kaweah	5-22.11	1,803 [696]
Tulare Lake	5-22.12	2,117 [818]
Tule	5-22.13	1,898 [733]
Kern County	5-22.14	7,874 [3,040]

2.2 Hydrology

2.2.1 Overview

The region is characterized by a semi-arid to arid Mediterranean climate with most precipitation occurring during the winter months and nearly no precipitation during the hot summer months. Much of the water supply in the TLB has its origins in the Sierra Nevada snowpack, and to a much lesser extent in precipitation that falls in the Coast Ranges and Tehachapi Mountains. Winter storms deposit large quantities of snow in the Sierra Nevada and spring melt water discharges into the Valley by way of the TLB's major rivers.

The major rivers of the TLB are the Kings River, the Kaweah River, and the Kern River. The three rivers originate from steep, mountainous watersheds of the snow-covered Sierra Nevada Mountains east of the TLB where the highest elevation (Mt. Whitney) is nearly 4,500 m (15,000 ft). In the past, the Kings, Kaweah, and Tule Rivers, under natural conditions, drained directly into Tulare Lake, while the Kern River drained into Kern and Buena Vista Lakes. All three lakes were historically terminal lakes that occasionally connected via surface sloughs and, at very high water stage, drained into the San Joaquin River via Fresno Slough. Now that the lakes have been drained, and development of groundwater has lowered the water table, the basin is largely considered to have no natural outflow of water from the basin (except for evapotranspiration).

2.2.2 Water Budget

Spring runoff is captured in large reservoirs along the Sierra Nevada foothills, and subsequently released into a network of natural stream channels and canals that serve irrigation and natural water needs and recharge groundwater. The largest canals in the TLB are the Friant-Kern Canal (FKC) along the entire eastern edge of the TLB, and the State Water Project (SWP) and San Luis Canal along the western and southern edge of the TLB. The canal networks associated with the FKC and SWP connect in the southern TLB. Together they are part of the backbone of California's modern water transfer network: The FKC transfers San Joaquin River (and some Kings River) runoff from their respective reservoirs to irrigation districts along the east- and southeast-side of the TLB. The SWP imports water from the much wetter Sacramento Valley via the Delta to irrigation districts in the western TLB and to other southern California regions.

Ideas to transfer "excess" water from the Sacramento Valley to the San Joaquin Valley were formulated as early as 1870s (California Department of Water Resources 1994). Over time, various private and government schemes have transformed the natural hydrology of the Central Valley. Surface water was used for irrigation needs as early as the 1700s (Bertoldi et al. 1991), but as agricultural efforts intensified, it was realized that this water had to be supplemented. The Miller and Lux agricultural enterprise formed in the mid 1800's and by 1900 several canals had been constructed for the purpose of delivering water to the southern San Joaquin Valley (Igler 2001). Groundwater resources started being developed in the 1880s (Bertoldi et al. 1991) and allowed cities to begin to flourish in the San Joaquin

Valley. As early as 1900, groundwater levels had fallen, requiring the development of larger extraction pumps to withdraw the deeper water. Around 1930, the improved deep well turbine pump was developed concurrently with the expansion of rural electrification (Galloway et al. 1999). As a consequence, water could be pumped from an even greater depth, and larger yields could be attained. Shortly thereafter, the invention of the Haber-Bosch process (which created nitrate from nitrogen gas present in the atmosphere) led to large-scale production of industrial nitrogen fertilizer. As a result, the mid-20th century began a period of quickly expanding and intensifying irrigated agricultural production. Groundwater pumping was excessive, particularly along the western TLB and in the southern TLB, due to the lack of significant surface water inflows to these areas. Importation of northern California water via the Sacramento River, the Delta, and the SWP began in the late 1940s which alleviated groundwater overdraft temporarily. In the 1990s and 2000s, restrictions on water transfers from the Sacramento Valley to the SWP, due to ecological concerns in the Delta combined with a series of droughts, led to a reduction in surface water imports and a resurgence of groundwater overdraft.

Today the Central Valley aquifer system, if treated as a single aquifer, is the second most highly pumped aquifer in the United States (Faunt 2009). The TLB accounts for approximately 35% of California's total annual groundwater withdrawal, roughly 5 cubic kilometers (4.34 million acre-feet (MAF)) (California Department of Water Resources 2003).

Several models have been created to understand the water budget for the region. Two established models for the Central Valley are the Central Valley Hydrologic Model (CVHM) and the California Central Valley Groundwater-Surface Water Simulation Model (C2VSIM). Each model also calculates water budgets for sub-regions within the Central Valley. Table 2 shows the groundwater budgets for the Central Valley as a whole, and for the TLB. The results from both models show that the TLB and the Central Valley as a whole have been losing groundwater (groundwater storage) for the past several decades. For the Central Valley, the results shown for CVHM indicate an average annual loss of 1.604 km^3 (1,300,000 acre-ft/year) over the years 1962-2003 (Faunt 2009), and the results shown for C2VSIM indicate an annual average loss of 2.491 km^3 (2,020,108 acre-ft) for the years 1962-2003 (Table 2) (Brush 2012). The final report for C2VSIM has not been published and values shown should be considered preliminary results.

For the TLB, the models calculate similar losses of storage. For the TLB, CVHM estimates an annual average loss in storage of -2.005 km^3 (-1,626,000 acre-ft) (Faunt 2009) and C2VSIM estimates an annual loss of -2.022 km^3 (-1,639,582 acre-ft) (Table 2) (Brush 2012). According to both models, the TLB has the greatest average storage loss out of all the subbasins for the years modeled. Of course, the reason the models can show a greater net loss in storage for the TLB, compared to the Central Valley as a whole, is due to other subbasins having net groundwater gains through time.

Four additional models have been recently developed for modeling water budgets of smaller regions within the TLB; the models are the Kings Integrated Groundwater and Surface Water Model (Kings IGSM), the Kaweah Delta Water Conservation District Model, the FRiant Economics-Driven SIMulation model (FREDSIM), and a model of the Kern Water Bank (Mellier et al. 2001; Ruud et al. 2003; Wrime, Inc. 2007).

Table 2 provides water budgets for the Kings IGSM, FREDSIM, and Kaweah Delta model. Model results show an average net loss of water for all three regions. The FREDSIM model has an economic framework, and the range of average storage loss shown is based on different economic scenarios. A water budget for the Kern County Water Bank model was not provided in the modeling report; however, Kern County Water Agency (KCWA) reports yearly water budget estimates for the water bank (PBS&J Inc. 2007). From 1995 to 2005, estimates show an annual average increase in storage of almost 100,000 acre-ft.

Table 2. Annual average groundwater budgets from hydrologic models in the Central Valley.

Model	Average Annual River Recharge km^3 [acre-ft]	Average Annual Recharge (except river percolation) km^3 [acre-ft]	Average Annual Groundwater Pumpage km^3 [acre-ft]	Average Annual Net Storage Change km^3 [acre-ft]
Central Valley Hydrologic Model (CVHM), USGS	0.3700 [300,000]	9.374 [7,600,000]	-11.471 [-9,300,000]	-1.604 [-1,300,000]
CVHM (Tulare), USGS	0.865 [701,000]	3.932 [3,188,000]	-6.968 [-5,649,000]	-2.005 [-1,626,000]
California Central Valley Groundwater-Surface Water Simulation Model (C2VSIM)**, DWR**	-0.173 [-140,095]	8.886 [7,203,690]	-11.157 [-9,045,364]	-2.491 [-2,020,108]
C2VSIM (Tulare Basin)**, DWR**	0.127 [102,606]	4.306 [3,491,054]	-6.459 [-5,236,202]	-2.022 [-1,639,582]
Kings Integrated Groundwater and Surface Water Model, KRCD	0.397 [321,700]	1.626 [1,318,500]	-2.223 [-1,802,000]	-0.200 [-161,900]
Kaweah Delta Water Conservation District Model, KDWCD	0.083 [67,000]	-0.104 [-84,000*]	-	-0.021 [-17,000]
FRiant Economics-Driven SIMulation model (FREDSIM], UC Davis	-	-	-	-0.155 to -0.355 [-126,000 to -288,000**]
Kern Water Bank, DWR	-	-	-	0.118 [95,525***]

*For the Kaweah model, the Average Annual Recharge value listed represents the net recharge which includes groundwater pumping. **The range of estimated groundwater storage change is based on different economic scenarios (e.g., pumping costs and surface water delivery costs). ***Storage change is based on KCWA estimates. ****Values shown for C2VSIM should be considered preliminary results.

2.2.3 Groundwater Levels

Groundwater levels generally fluctuate annually. This is largely due to groundwater pumping in the summer, causing a lowering of water levels, and subsequent recharge in the winter and spring, providing an increase in storage and raising groundwater levels. Levels also change through time on longer scales, due to periods of drought when groundwater is heavily pumped, and conversely, during periods of high precipitation when greater amounts of surface water are used and groundwater is not relied on as much. The Department of Pesticide Regulation, while investigating groundwater vulnerability to surface contamination, required an estimated depth to groundwater (DGW) coverage for the Central Valley (Spurlock 2000). For the purpose of their study, average DGW coverage was determined for California using spatial DGW measurements. Taken in the months of January through May, 260,000 later winter-spring DGW measurements were selected to represent water table levels. Kriging was used to interpolate between data points, and interpolation was restricted to distances less than 3.2 km (2 miles) from raw data boundaries. A map of the DGW coverage is shown in Figure 2. The primary source of the DGW data was the DWR Division of Local Planning and Assistance (DLPA). Other sources included the United States Geological Survey, San Benito County Water District, Santa Clara Valley Water District, and the Monterey County Water Resources Agency (Spurlock 2000). Although groundwater levels fluctuate through time, the map in Figure 2 shows areas where DGW is generally shallow, and other areas where it is considerably deeper. This has a direct impact on nitrate travel times to wells, as the hydraulic conductivity of unsaturated soils (above the water table) is much lower when compared to their saturated equivalent. The reader is referred to Section 6 *Nitrate Travel Time Through the Vadose Zone in the Tulare Lake Basin and Salinas Valley* for a more in-depth discussion on vadose zone transport.

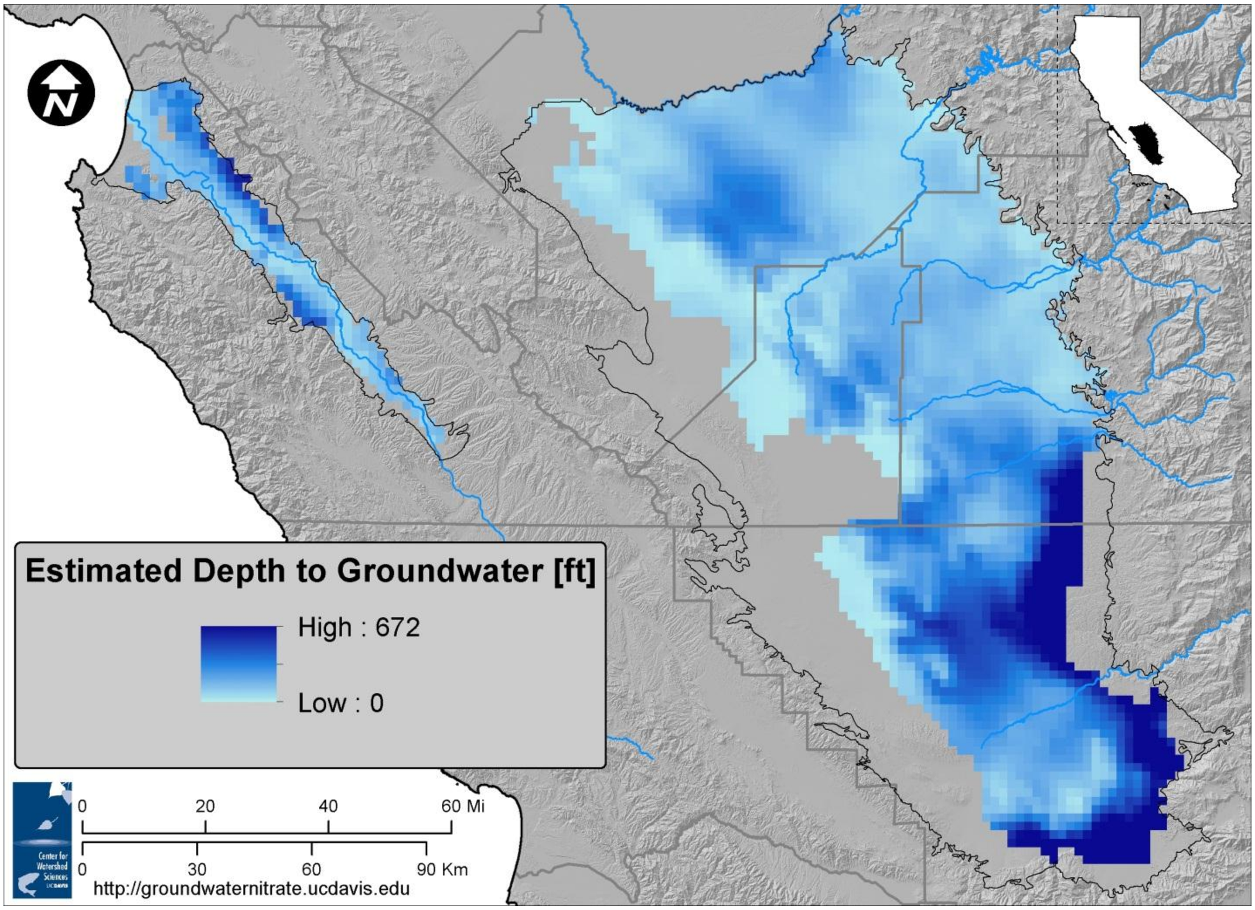

Figure 2. Depth to groundwater for the TLB and SV. (Source: Spurlock 2000.)

2.3 Hydrogeology

2.3.1 Geology

The Sierra Nevada Mountains to the east of the TLB are formed mainly of pre-Tertiary rocks of igneous and metamorphic origin. Sloping southwesterly from the foothills, the granitic bedrock lies beneath the fluvial and alluvial sediments which comprise the Central Valley aquifer systems. The Coast Ranges, to the west of the TLB, were formed by complex folding processes due to the convergent plate boundary to the west, and are both marine and volcanic in origin. The consolidated sediments of the Coast Ranges are of Jurassic, Cretaceous, and Tertiary age (Croft & Gordon 1968; Croft 1972; Planert & Williams 1995; Faunt 2009).

The Central Valley is a structural trough, with largely unconsolidated to semi-consolidated alluvial and fluvial sediment covering the basement complex (Faunt 2009). These sediments comprise the framework which stores the Valley's groundwater supplies. The aquifer systems of the TLB are generally composed of coarser sandy and gravelly sediments within a framework of finer-grained sediments (Weissmann et al. 2005). Through time, rivers and smaller streams emerging from the foothills of the Sierra Nevada and Coast Ranges have carried sediments from the mountains and deposited them on the valley floor, forming alluvial fans consisting of complexly arranged stream bed deposits (mostly sands, gravels); overbank deposits (sands and silts); and flood basin deposits (silts and clays), with inter-bedded lacustrine lakebed sediments (clays) (Weissmann et al. 2005). The coarse grained sediment bodies within the subsurface aquifers are the portions which conduct water the easiest and provide the majority of the water to pumping wells.

Throughout the San Joaquin Valley and the TLB, the shallower aquifer systems are generally composed of alluvial deposits derived from the Coast Ranges and Sierra Nevada Mountains, and flood-basin deposits near the valley trough (Lauden & Belitz 1991). Coast Range alluvium (derived from marine-origin sedimentary formations) skirts the western-most portion of the TLB, while the central and eastern portion of the TLB is dominated by the alluvial fans and plains formed by the streams discharging sediments from the granitic Sierra Nevada mountains (Figure 4) (Miller et al. 1971; Belitz & Heimes 1990; Weissmann et al. 2005; Faunt 2009). Coast Range alluvial deposits interfinger with Sierra Nevada alluvial deposits beneath the valley trough to form a heterogeneous organization of sediments. The Sierra Nevadan alluvial sediments are generally coarser in texture than the Coast Range alluvium, due to their granitic origin, as compared to the marine sediments derived from the Coast Range (Miller et al. 1971; Belitz & Heimes 1990).

Fine-grained lacustrine, paludal, and flood-basin deposits also underlie the valley trough. These finer-grained deposits are hydrologically important, clay and silty-clay deposits, and have been labeled as the A- through F-Clays. The E, C, and A-Clays comprise a significant portion of the TLB aquifer system framework (Croft & Gordon 1968; Croft 1972). The Corcoran Clay (E-Clay) is a large confining unit in the western portion of the valley and its spatial distribution is shown in Figure 3 as it is modeled in CVHM (Faunt 2009). It forms the upper confining boundary for many of the aquifers in the western Central

Valley. Historically, the Corcoran Clay was thought to be a continuous impermeable layer which separated the upper unconfined aquifer and the lower confined aquifer. Figure 3 shows that, although it is relatively continuous over a large area of the Central Valley, it has significant spatial variability within it. Areas containing larger fractions of coarser grained material likely provide an exchange of water between the upper and lower aquifers compared to areas with no coarse grained sediments. More importantly, studies suggest that the development of groundwater has likely increased the connection between the aquifers via well bore holes, creating unimpeded pathways through the confining layer (Williamson et al. 1989; Belitz & Phillips 1995).

Weissmann et al*.* (2005) performed a study of the role that alluvial and fluvial fans have played in the creation of San Joaquin Valley sediment fill. In common with the rest of the San Joaquin Valley, the eastern flanks of the TLB are characterized by many large fluvial fans that coalesce before terminating at the valley trough. In the TLB, the major fans are those that originate from the larger rivers of the Sierra Nevada Mountains.

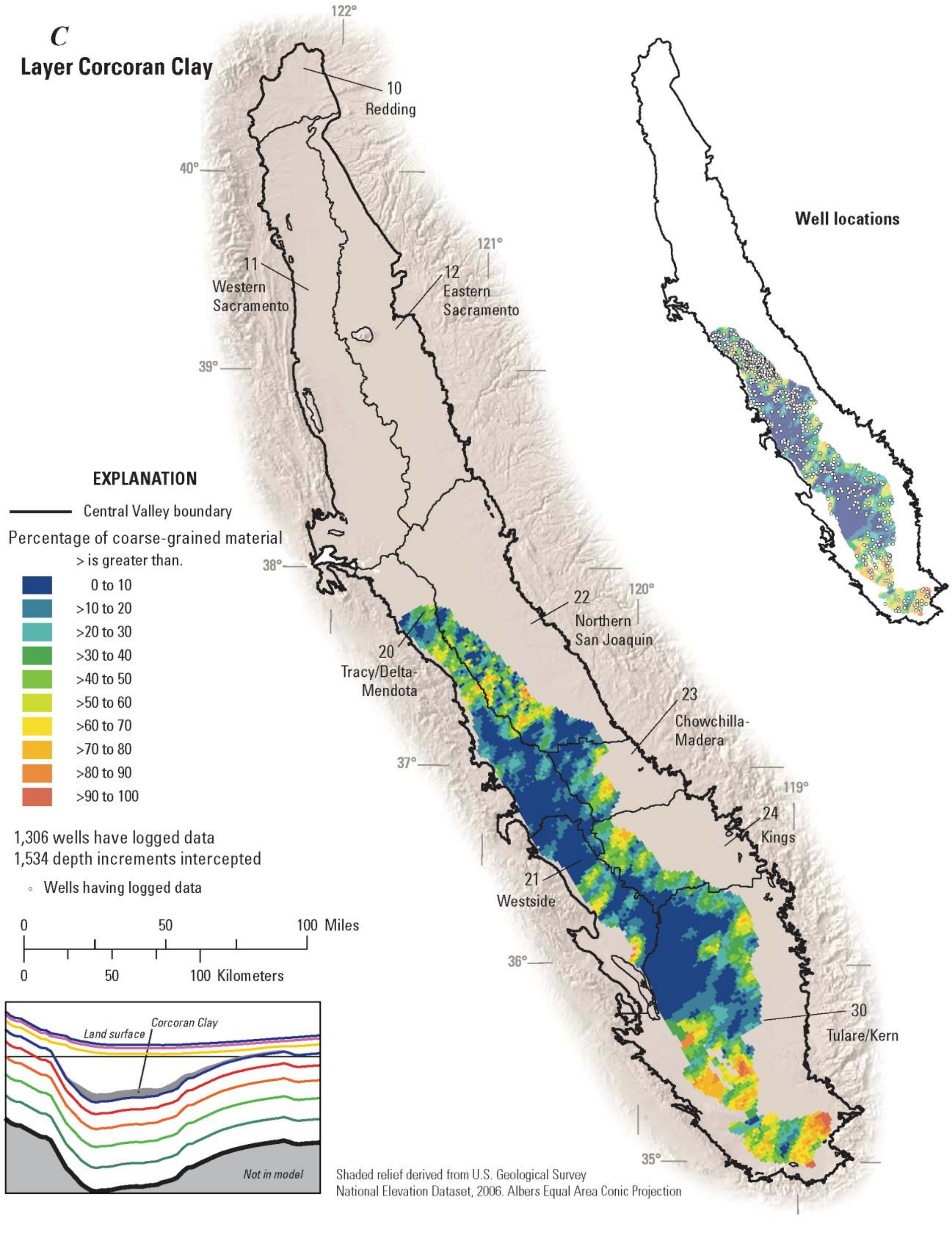

Figure 3. Spatial distribution of sediments of the Corcoran Clay as modeled in the Central Valley Hydrologic Model (CVHM). (Source: Faunt 2009.)

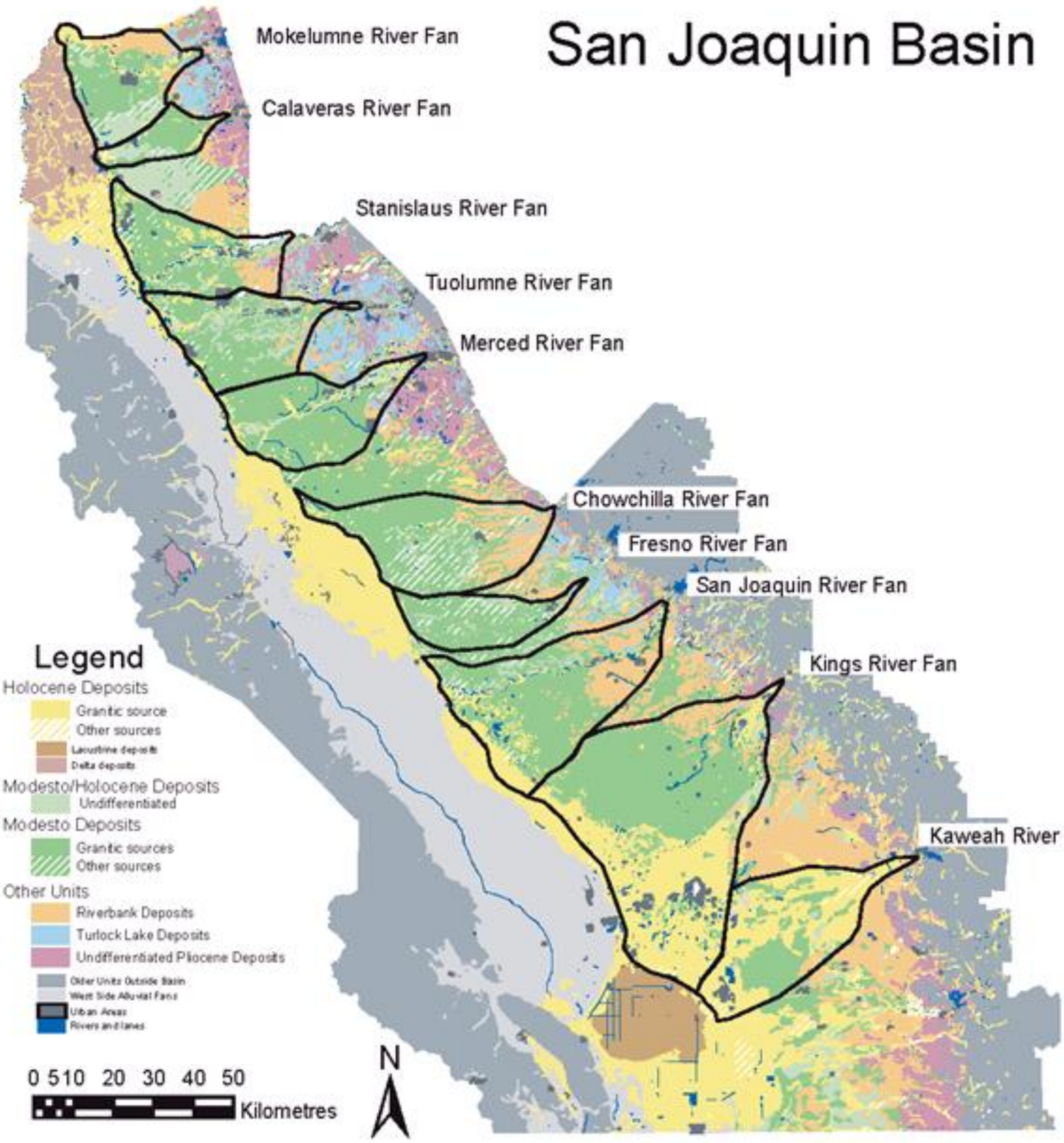

Figure 4. Fans of Northern and Central San Joaquin Valley from *Alluvial Fans: Geomorphology, Sedimentology, Dynamic*s, Geological Society of London Special Publication 251, Geological Society, London. The Kings and Kaweah River Fans are in the TLB. (Source: Weissmann et al. 2005.)

The thickness of aquifers in the TLB varies throughout the basin. The aquifers that contain fresh groundwater are generally thickest in the southern portion of the TLB, in the vicinity of Bakersfield. The thickness of aquifer material bearing freshwater is on average 600m (2,000 ft), but can locally be greater than 4,000 ft (Planert & Williams 1995). Beneath the continental deposits are sediments of marine origin (Faunt 2009). Figure 5 from Faunt (2009) shows a conceptual diagram of the aquifer system.

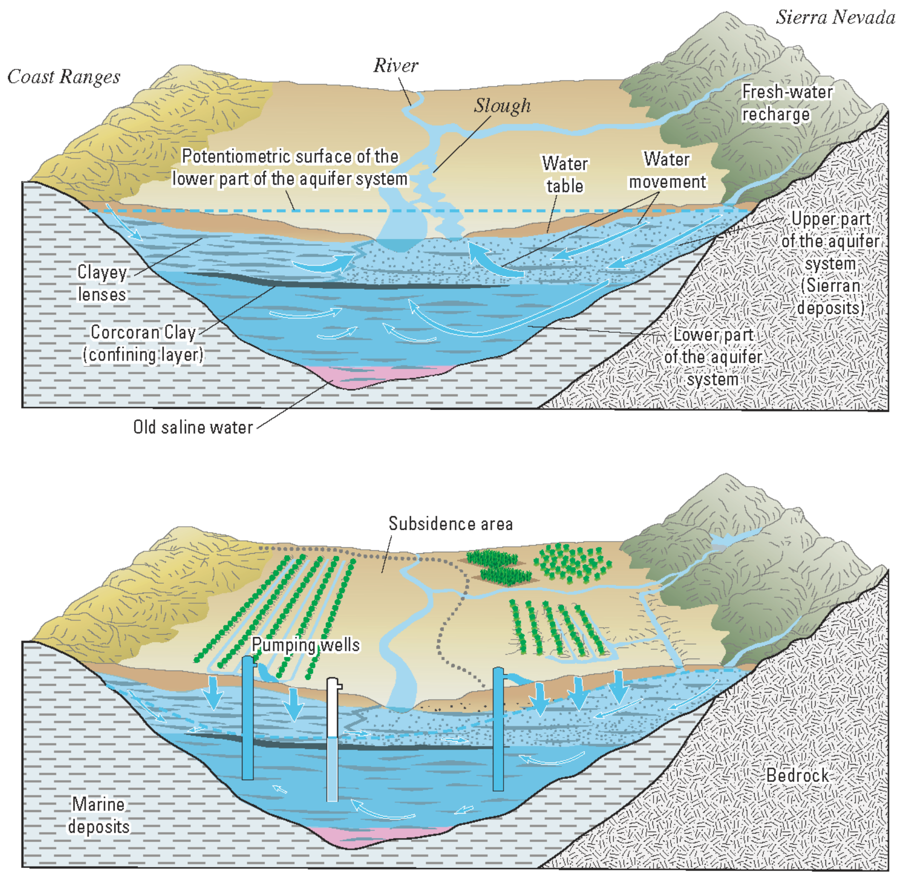

Figure 5. Conceptual diagram of the San Joaquin aquifer system. The first image (top) shows pre-development conditions where groundwater typically discharged to surface water bodies. The second image (bottom) shows that the mechanism for discharge from the aquifer is now dominated by groundwater pumping. (Figure from: Faunt 2009, as modified from Belitz & Heimes 1990; Galloway et al. 1999.)

The marine sediments, deposited in an ocean environment, often contain old saline water. In general, the freshwater portion of the aquifer system is comprised of much younger water than that of the deep marine aquifers. Although freshwater is generally found in continental deposits, in certain areas it can be hard to delineate between the marine and continental deposits (Bertoldi et al. 1991). In the western portion of the TLB, the regionally elevated-TDS groundwater found in the freshwater aquifer systems is the result of the water flowing through the Coast Range alluvial fans composed of marine sediments (Deverel & Gallenthine 1988).

For the regional nitrate analyses in Section 5, the TLB was divided into three separate regions based on sediment origins. The three groups, referred to as the Eastside Alluvial Fans, Basin, and Westside Alluvial Fans groundwater regions, were delineated by the USGS and refer to the origin of the alluvial sediments present in these regions (Burow et al. 1998). The role of denitrification (the natural attenuation of nitrate) in these regions is further explained in Appendix B of this report.

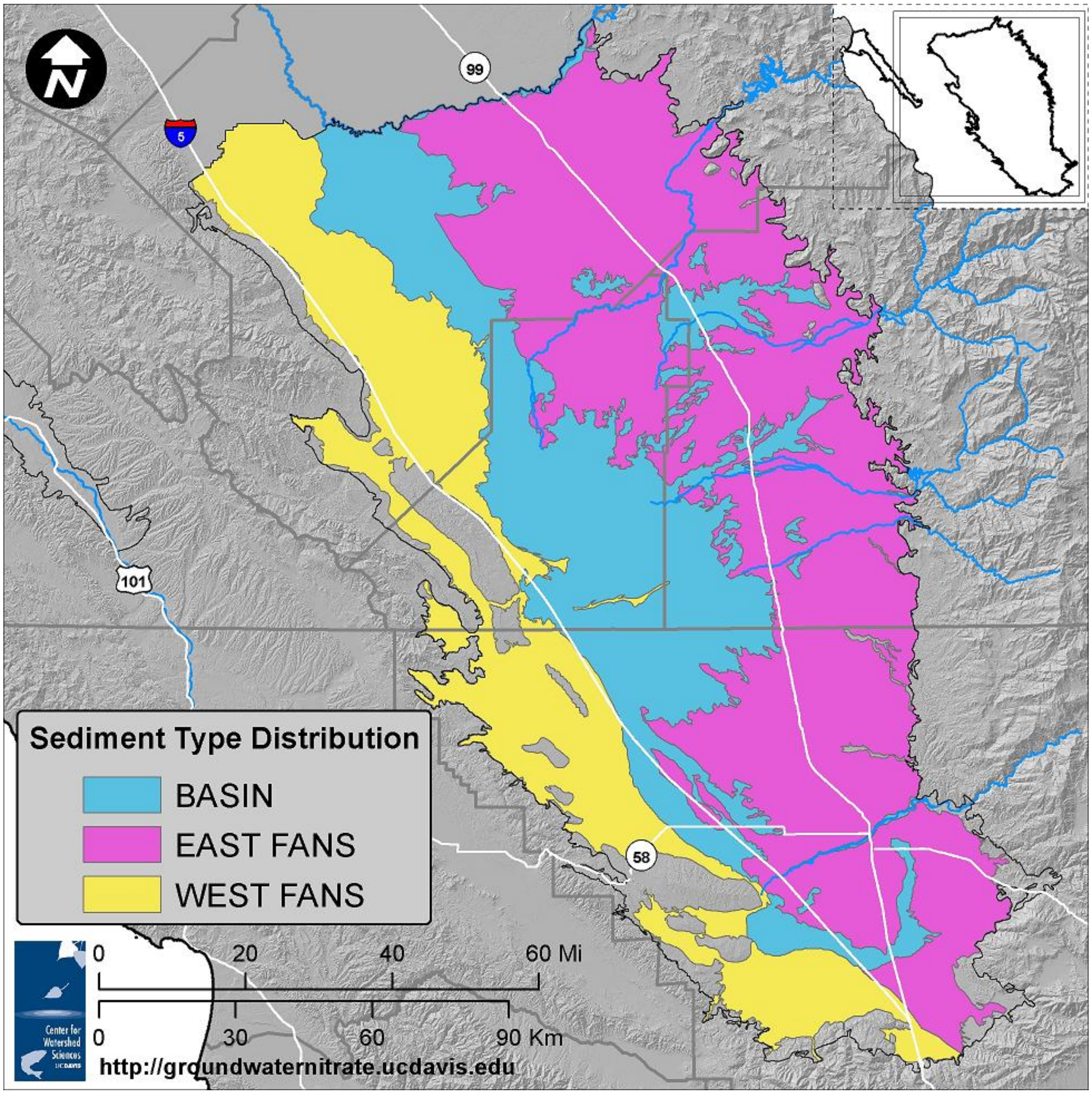

Figure 6. Based on the sediment origins, the TLB is divided into three groundwater regions: the Eastside Alluvial Fans, the Basin, and the Westside Alluvial Fans region.

2.3.2 Groundwater Recharge

Groundwater recharge in the study area comes mainly from irrigation return flow and surface sources such as streams and managed recharge projects. The majority of surface water reaches the groundwater by first being applied as crop irrigation. Groundwater pumping is the main outflow from the groundwater basin. Estimates of groundwater recharge and discharge for the TLB can be found in Table 2 of Section 2.2.2 *Water Budget*.

Some cities and water districts have attempted to counteract declining groundwater storage through artificial recharge programs. These are usually situated over areas within the TLB having comparatively higher-permeability sediments in the shallow subsurface. Kern County, which has a highly valuable agricultural economy while also being very arid, uses surface water from sources such as local streams, the California Aqueduct, and the Friant-Kern Canal for "groundwater banking" (Mellier et al. 2001). When "excess" surface water is available during the wet season, it is transported to suitable areas and allowed to seep into the subsurface to be withdrawn later from the aquifer when it is needed. Major artificial recharge projects are currently operated by the Arvin-Edison, Kern, and Semitropic Water Storage Districts. These groundwater banking operations have a combined storage capacity of approximately 3 million acre-feet (more than 5 times greater than Millerton Lake[3]) (California Department of Water Resources 2003). The success of these projects has led to other districts proposing their own, such as the Madera Ranch Project to the north. The City of Visalia already has its own groundwater recharge program.

2.3.3 Flow Modeling

Section 2.2.2 *Water Budget* provides the water budget results from several groundwater models which encompass all or a portion of the TLB. As a part of this study, a steady state model was developed using input stresses (e.g., groundwater recharge and pumping) from the CVHM model for the purpose of modeling the transport of nitrate in the subsurface. The reader is referred to Section 7 *Nitrate Occurrence: Groundwater Transport Modeling* for more information on the development of the steady state model.

2.3.4 Transport Modeling

To date, there is no groundwater model that models nitrate transport for the TLB. A Non Point Source Assessment Tool (NPSAT) was developed as a part of this study to model nitrate transport at the basin scale and to investigate variable source loading scenarios in regards to nitrate. The reader is referred to Section 7 *Nitrate Occurrence: Groundwater Transport Modeling* for more information.

[3] Millerton Lake is an artificial reservoir constructed on the San Joaquin River near the town of Friant for surface water storage.

2.4 Water Quality

Generally, the groundwater found within the aquifer system as outlined above is suitable for the majority of municipal and agricultural uses, though local water quality impairments can be found. The primary groundwater quality impairments are commonly due to high TDS, nitrate, arsenic, selenium, boron, pesticides and herbicides, and organic compounds (Planert & Williams 1995; California Department of Water Resources 2003). The source of these constituents of concern varies; some are naturally occurring and others originate from anthropogenic activities.

2.4.1 Natural Contaminants

Fresh water, suitable for water supply needs, is defined as having a TDS measurement of less than 2,000 mg/L. Higher levels are defined as brackish and saline. Saline and connate[4] water can be found within the fresh water-bearing continental deposits; most saline and connate water is below the fresh water. This saline water comes from a variety of potential sources, including upward migration of old marine water (present during the deposition of the marine sediments) or through the process of evaporative concentration (Farrar & Bertoldi 1988).

The marine sediments that make up the Coastal Ranges naturally contain a variety of constituents that are of concern in surface and groundwater within the Central Valley. Through dissolution of the marine sediments, minerals and ions are released, and water flowing through such sediments increases in TDS. This has consequentially lead to elevated TDS levels in much of the west-side of the TLB (Deverel & Gallenthine 1988). This is in contrast to the east-side, where water originates in the Sierra Nevada Mountains. The granitic rocks of the Sierra Nevada Mountains dissolve at a much slower rate than the marine sediments and therefore contribute much less mineral content (TDS) over time (Planert & Williams 1995).

In the vertical direction, TDS readings generally start high at the water table, due to the infiltration of high-TDS agricultural return water from the surface, and decrease with depth, until relatively pristine pre-modern (<1900’s) age water is reached. Below this, in the deeper portions of the subsurface, salinity again increases due to the presence of old water with high concentrations of dissolved minerals, as well as connate seawater present from when the marine sediments were deposited.

High TDS water can be a problem in deep wells, which are commonly found in the western and southern portions of the TLB. These deep wells, while trying to avoid pumping the shallow contaminated water, may penetrate deep enough to withdraw high-TDS groundwater or water that has been affected by high salinity water. In addition to the naturally accumulated TDS in waters found in the valley trough, evaporation close to the land surface concentrates the groundwater leading to further elevated TDS. In the western and central portions of the TLB, accumulation of solutes in shallow groundwater has had negative effects on agricultural production. The Central Valley Salinity Alternatives for Long-Term Sustainability (CV-SALTS) coalition, a non-profit organization, was formed in 2008 to address salinity

[4] Water present when the aquifer sediments were deposited.

issues in the Central Valley. For more information on salinity in California, visit the CV-SALTS website at http://www.cvsalinity.org/.

Another natural contaminant, selenium, is also found within the marine Coast Ranges sediments. Selenium can be concentrated by evapotranspiration in western and central portions of the TLB when this water is used for irrigation (Deverel & Gallenthine 1988). Selenium is toxic to many living creatures, and has the potential to bioaccumulate. A well-documented case of selenium contamination occurred at the Kesterson Wildlife Refuge further to the north, where selenium caused harm to waterfowl, fish, insects, plants, and algae (Ohlendorf et al. 1990).

Arsenic is an additional contaminant of recent concern. Arsenic concentrations are often the result of water with iron- or manganese-reducing conditions, which provide conditions that dissolve iron and manganese oxides present on sediments containing sorbed arsenic. Oxic alkaline (high pH) water also has the potential to desorb arsenic from oxides present on sediments (Belitz et al. 2003; Welch et al. 2006). The dry lake-beds within the TLB, namely Tulare, Kern, and Buena Vista are known to have elevated arsenic.

2.4.2 Groundwater Chemistry

As discussed before, the sediments through which groundwater travels can affect the chemistry of the water through dissolution of the minerals present. The following three piper diagrams plot groundwater samples from California Department of Public Health (CDPH) wells for the three geological regions of the TLB; the Eastside Fan, Westside Fan, and Basin sediments (Figure 6, Section 2.3.1). Although there is considerable scatter to the data, due to the large physiographic region of our study area and the natural heterogeneity of the aquifer, the samples generally reflect the aquifer sediments present in these regions.

The major ions present in water are calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), chloride (Cl^{-}), sulfate (SO_4^{2-}), carbonate (CO_3^{2-}), and bicarbonate (HCO_3^{-}). The relative amounts of each constituent, when plotted on a piper diagram (Figure 7-9), provide information on regional flow paths and origins of the water. For example, Ca^{2+} and Mg^{2+} concentrations are often elevated in shallow groundwater and decrease along flow paths due to exchange for Na^{+} present on clay-rich sediments, resulting in more Na^{+} and less Ca^{2+} and Mg^{2+} the longer water has spent traveling through the subsurface. Additionally, groundwater tends to continuously dissolve carbonate and other minerals found naturally in geological materials as it travels through the aquifer system. This results in greater amounts of HCO_3^{-} in older groundwater.

Samples from Eastside Alluvial Fans sediments, while covering the broad spectrum of water types, are concentrated in the left portion of the figure, meaning that the water contains little sulfate, and higher portions of calcium and magnesium (Figure 7). This is a reflection of the granitic sediments coming from the Sierra Nevada Mountains.

Samples from Westside Alluvial Fans sediments show the opposite. While the number of samples is significantly less than those from the Eastside Alluvial Fans sediments, the majority of the samples plot in the upper portion of the piper diagram. This signifies that the groundwater samples contain elevated levels of sulfate compared to the samples taken from the Eastside Alluvial Fans sediments. This reflects the origin of the Westside Alluvial Fans sediments, having primarily a marine origin.

The Basin sediments represent a mixture of both west and east fans sediments . The TLB as a whole, however, is dominated by Eastside Alluvial Fans sediments. The samples taken from the basin sediment region shows that the water chemistry is more similar to the water found in the east fans as compared to the water found in the west sediments. Groundwater from this region generally contains more calcium and magnesium and less sulfate.

In all three regions, nitrate is also plotted with the major ion chemistry. As mentioned before, as water travels through aquifer sediments, it generally exchanges calcium and magnesium for sodium. Water containing more sodium compared to calcium and magnesium can generally be assumed to have been present in groundwater longer than water containing little sodium, compared to calcium and magnesium. In the basin and Eastside Fans sediments the piper plots show that the "older" water generally contains lower nitrate than the "young" water. Section 3.4 provides more in depth discussion of groundwater chemistry, age, and nitrate contamination in the context of the Salinas Valley. Appendix B discusses the role of denitrification.

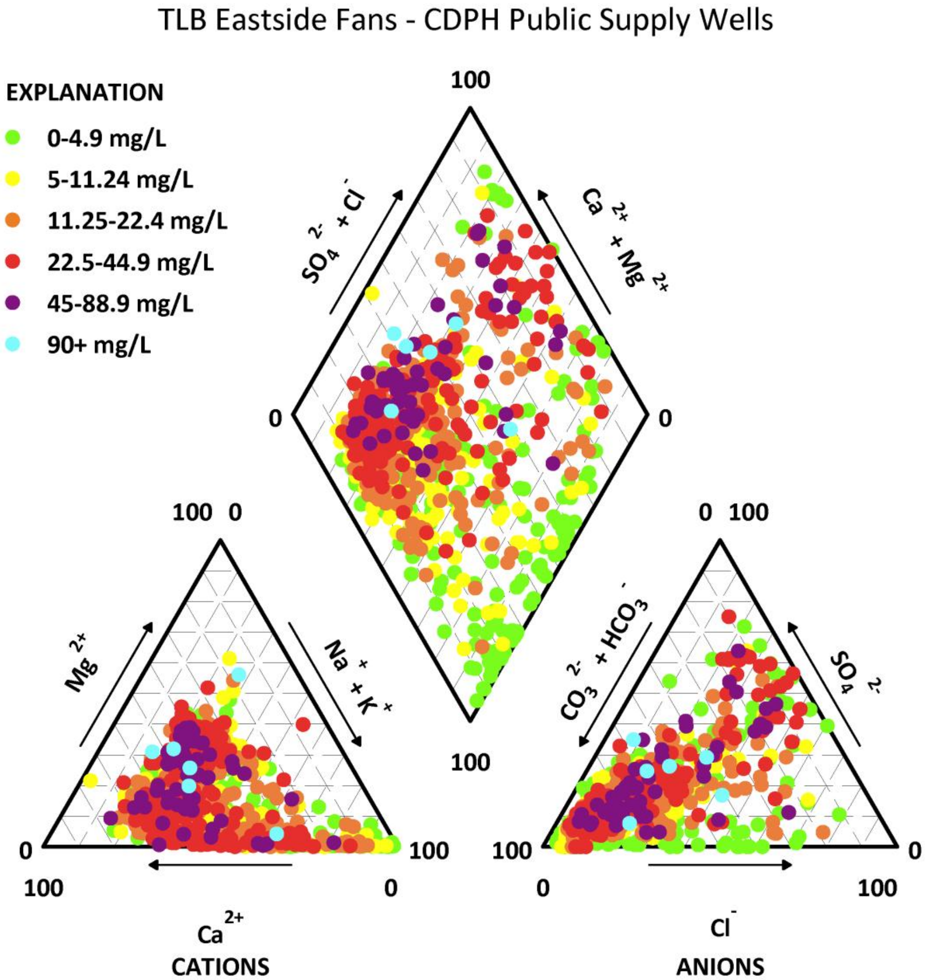

Figure 7. Piper diagram of groundwater samples from Eastside Fans sediments. Concentrations are of nitrate as nitrate.

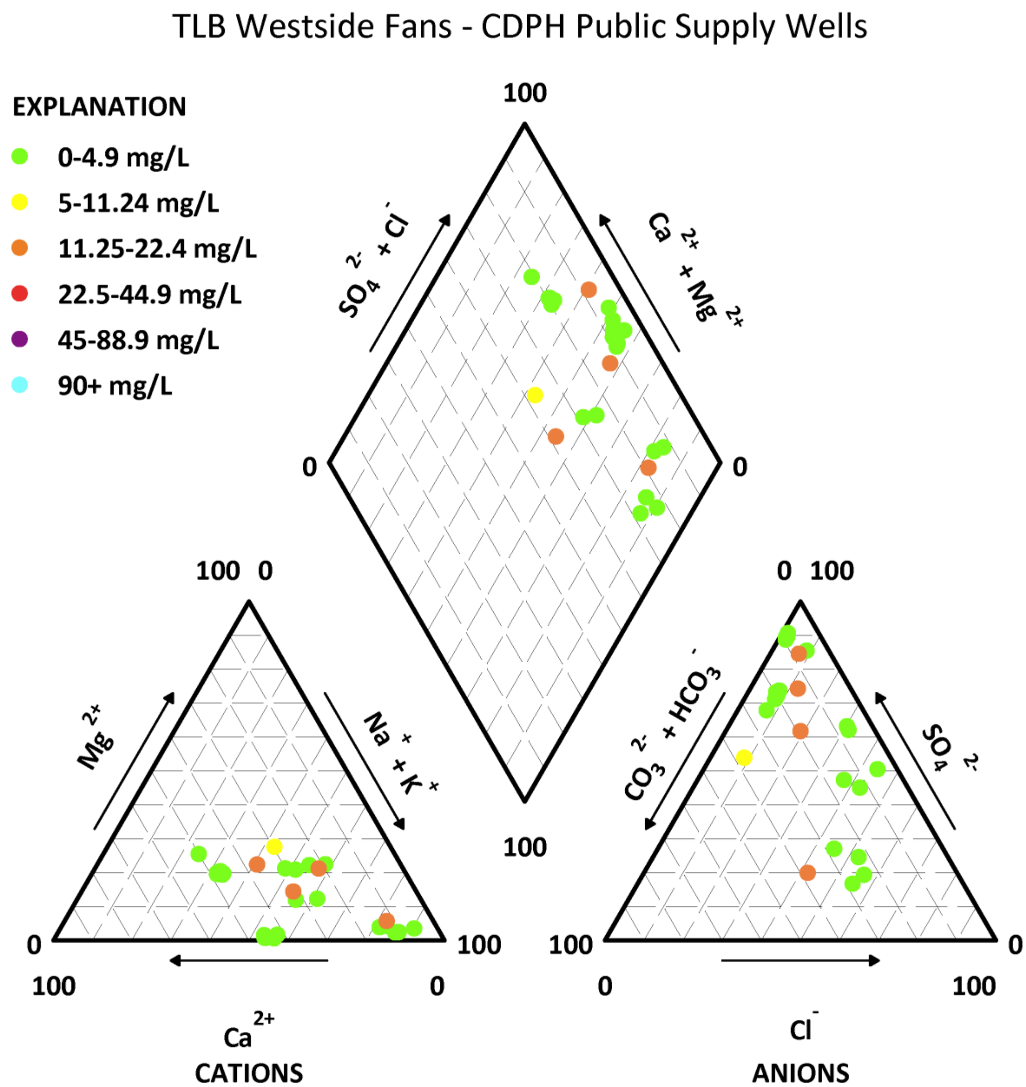

Figure 8. Piper diagram of groundwater samples from Westside Fans sediments. Concentrations are of nitrate as nitrate.

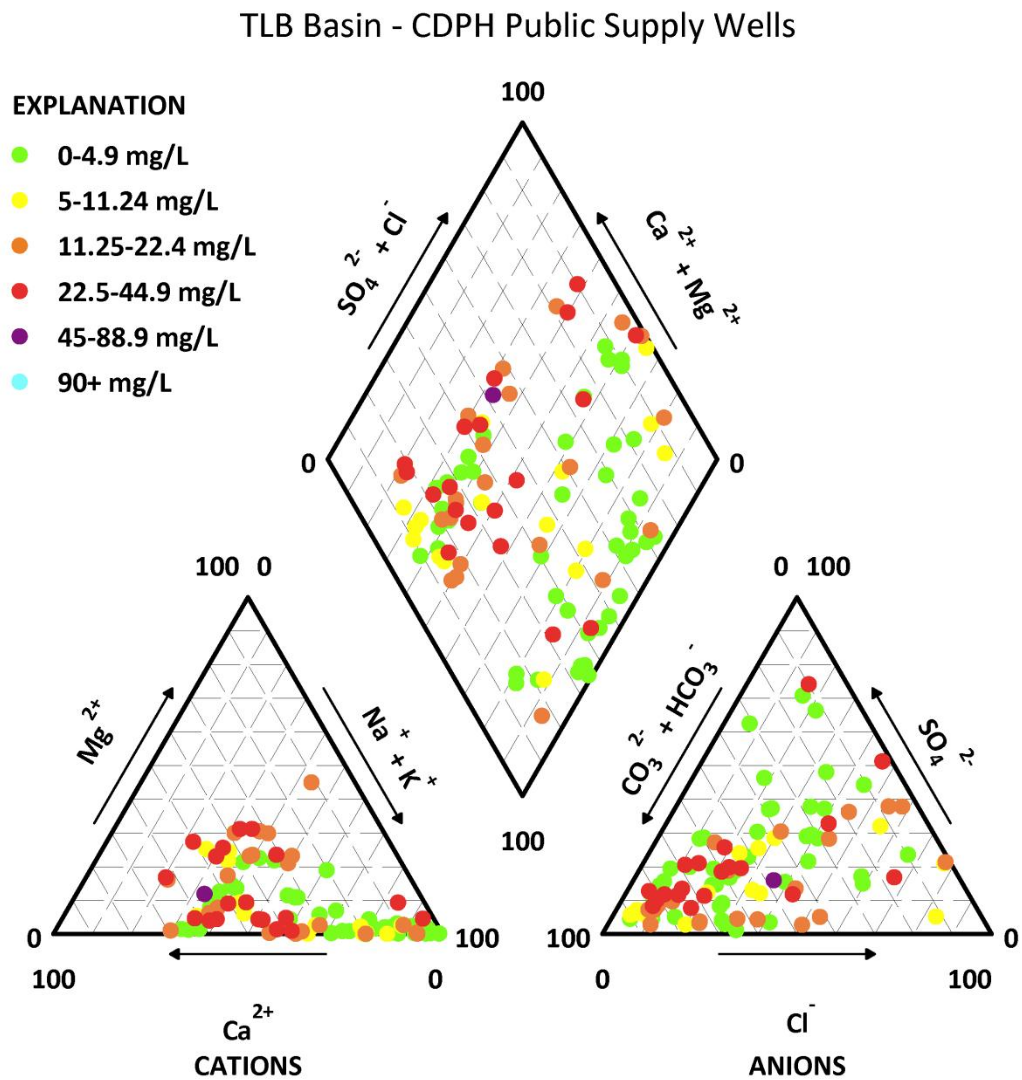

Figure 9. Piper diagram of groundwater samples from Basin sediments. Concentrations are of nitrate as nitrate.

2.4.3 Anthropogenic Contaminants

Nitrate is the most wide-spread pollutant in the TLB and it has been studied extensively. Anthropogenic sources include fertilizer applied to crops; animal operations, such as dairies and feedlots; and human sources, such as wastewater treatment plant effluent and septic tanks. Nitrate contamination is prevalent in every subbasin within the TLB. While nitrate is a naturally-occurring constituent of groundwater, concentrations of nitrate are being detected in the TLB that are well above what is considered natural or "background" concentrations. Levels of nitrate below 9 mg/L (as nitrate) are generally considered background (Mueller & Helsel 1996), and levels above 18 mg/L (as nitrate) are thought to reflect water that has been impacted, or contaminated from anthropogenic activities (Nolan et al. 2002). The maximum contaminate level (MCL) for nitrate, as established by the U.S. EPA, is 45 mg/L (as nitrate).

Anton et al. (1988) presented the first regional-scale examination of nitrate in water wells in a report to the California State Legislature. The San Joaquin Valley (including the TLB) and the Central Coast (including the SV) geographic regions were examined as part of that study. The U.S. EPA STORET database was the primary source of data for the project, but most of the discussion centers on reportage of conclusions from previously unidentified studies. The Anton report characterizes the state of nitrate in groundwater for each of the Tulare Basin counties (Fresno, Kings, Tulare, and Kern). Fresno County is described as having elevated nitrate mostly in the eastern portions of the valley, with lower nitrate concentrations in the mid-valley, and slightly elevated concentrations in the western edge of the valley. Kings County is characterized as being relatively free of high nitrate problems in groundwater. The high nitrate region of eastern Fresno County is described as extending through the eastern valley portion of Tulare as well, and into Kern County (discontinuously from the city of Fresno to the city of Bakersfield). Anton states that 33 small and one large water system in that band have been in violation, but the timing of those violations is not mentioned. The report goes on to describe Kern County as having some of the highest nitrate concentrations in wells, with 34 small systems in violation. Anton also mentions a study from 1982 by Kern County Water Agency that found that the area affected by nitrate concentrations near or exceeding the MCL had expanded from 127 km^2 (49 mi^2) in 1958 to 963 km^2 (372 mi^2) in 1979.

A 1998 study by the USGS investigated nitrate and pesticide contamination of groundwater below several crop types grown in the TLB (Burow et al. 1998). Sixty monitoring wells were installed to sample nitrate and pesticide levels in groundwater in 1994-1995. The wells were placed along approximate groundwater flow paths beneath three land uses in the eastern San Joaquin Valley. The crop types included almond orchards, vineyards, and corn/alfalfa/vegetable crops. The three land use settings were thought to be representative of the range of crops grown in the San Joaquin Valley.

The high levels of fertilizer application, along with the rapid infiltration rate, led to many of the samples obtained from almond orchards to be very high in nitrate levels. About 40% of the samples contained nitrate levels above the MCL of 45 mg/L (as nitrate) (Burow et al. 1998). For vineyards, nitrate levels were much lower, representing approximately 6% of the Central Valley land area. About 15% of samples taken from a vineyard land use setting had nitrate levels greater than the MCL. For the alfalfa, corn, and vegetable land use, nitrate levels were closer to levels found in almond orchards. Alfalfa, corn, and vegetable cover approximately 12% of the Central Valley, and these crops tend to be planted in fine-grained sediments with low dissolved oxygen measurements and a slow infiltration rate. Nitrate levels were higher than the MCL in 35% of samples obtained from these crops. Two wells had concentrations below the detection threshold of 0.05 mg/L. Nitrate levels exceeded the background level (defined in the report as 13.2 mg/L (as nitrate) in 75% of the samples and exceeded the MCL in 30% of the well samples.

Overall, Burow et al. (1998) also found that 68% of the wells tested had detectable levels of pesticides. The main pesticides found were simazine, dibromochloropropane (DBCP), atrazine, desethyl atrazine, and diuron found in 37, 30, 25, 25, and 15 percent of the samples, respectively. All of the pesticides except DBCP were below respective U.S. EPA MCL limits for drinking water (if a limit existed). DBCP exceeded its MCL of 0.2 micrograms per liter (µg/L) in 25% of the wells tested.

USGS Circular 1159 (Dubrovsky et al. 1998) details the findings from investigations conducted under the National Water-Quality Assessment (NAWQA) for the San Joaquin-Tulare Basins. From September 1992 to August 1995, data were collected from a total of 88 domestic wells. Sixty of the domestic wells were located (20 each) in 3 different landuse settings: almond orchards, vineyards, and corn/alfalfa/vegetable crops. These are the same 60 wells that were examined in Burow et al. (1998), referenced above. The remaining 28 wells, plus 2 of the 60, were chosen as representative of the regional aquifer, and had a median nitrate concentration of 20.3 mg/L (as nitrate). Five of the 30 (17%) exceeded the MCL. Taken all together, 25% of the 88 domestic wells sampled for the project exceeded the MCL, and 77% exceeded the accepted background level (defined as 2 mg/L nitrate as nitrate). Elevated nitrate in shallow groundwater (i.e., domestic wells) was associated with agricultural land uses and coarse-grained sediments.

Returning to the same area in 2003, Burow et al. (2007) resampled the 60 monitoring wells for nitrate and pesticide contamination. The study showed (through chlorofluorocarbon (CFC) age dating) that water less than 10 m (33ft) below the water table had mean ages generally less than 15 years old. Water samples from depths greater than 60 m were shown to have mean ages greater than 45 years old. Burow et al. compared nitrate concentrations measured in their localized sampling with regional monitoring networks and found comparable results at the regional scale. Concentrations were the highest and most variable in the more shallow waters with median values and variability decreasing with depth. Comparing the 2003 samples to previous samples taken in 1994-1995, nitrate was found in have increased from 35.3 mg/L to 101.4 mg/L (as nitrate) in samples taken from the shallow groundwater. Analysis also determined the regional waters to be generally oxic and therefore little natural attenuation was expected to be taking place. Given the increases in nitrate concentrations, the oxic conditions, and vertical gradients present in the aquifer system, Burow et al. theorized that the shallow contaminated water would eventually make its way to deeper depths where public wells generally withdraw water.

Burow et al. (2008) examined historical data in the San Joaquin and Tulare Basins using data from 1,437 National Water Information System (NWIS) wells, 3,216 U.S. EPA STOrage and RETrieval (STORET) wells, and 1,689 Permits, Inspection, Compliance, Monitoring, and Enforcement (PICME) wells. In addition, the 2008 study included data from resampling of most of the wells sampled in the 1998 studies (Burow et al. 1998; Dubrovsky et al. 1998). The 2008 study concluded that nitrate concentrations were increasing over time and decreased with depth, and that drinking water supplies were significantly degraded by nonpoint source (agricultural) nitrogen.

In 2008, as part of the California State Water Resources Control Boards' Groundwater Ambient Monitoring and Assessment (GAMA) Priority Basin Project, the USGS sampled 99 public supply wells and irrigation wells in Kings, Kaweah, Tule, and Tulare Lake regions (Burton & Belitz 2008). These public supply wells tend to be drilled to depths that do not have elevated nitrate, are located predominantly in urban and peri-urban areas, and have much deeper screens than the monitoring well network used by

Burow et al. (2007). To obtain spatially unbiased data, public supply wells were sampled using an equal area grid, with a single well selected for each equal area cell. The results showed that six wells (~6%) were found to contain nitrate above the MCL. More than half of the samples contained at least one detectable pesticide, and seven pesticides were present in more than 10% of the samples. However, almost all of the samples' concentrations of pesticides were well below drinking water standards for the constituent.

Under the same GAMA program, a report for Kern County was published in 2008 by the USGS (Shelton et al. 2008). Using the same methodologies for unbiased spatial sampling, 2 out of 17 samples analyzed for nitrate contained concentrations above the drinking water standard, and all but one sample had detectable (>0.26 mg/L as nitrate) concentrations of nitrate. At least one pesticide was detected in 29 out of the 47 wells analyzed, and 5 pesticides were found in more than 10% of the samples. All pesticide concentrations, however, were below drinking water thresholds, with the majority of the concentrations being less than one-thousandth of the thresholds.

In 2010, the USGS National Water Quality Assessment Program published a national assessment of nutrient impacts on groundwater (Dubrovsky et al. 2010). The assessment included a national statistical model to predict groundwater nitrate concentration in the shallow-most groundwater based on land use data, general hydrogeologic information, and soils information. For large portions of the TLB, but also for the SV, the national USGS statistical nitrate model simulates shallow groundwater nitrate values that exceed the drinking water limit (Figure 10).

It is important to remember that nitrate contamination found in the TLB is not from a single land use or process. Nitrate detected in a single groundwater sample is often from a combination of sources. LLNL, using isotopic data of groundwater samples from a domestic well survey program, concluded that although the most contaminated samples were associated with an organic source (i.e. manure, septic, etc.), the majority of the samples containing elevated levels of nitrate had a signature that indicated the contamination was due to multiple sources, meaning a combination of organic and inorganic sources (Singleton et al. 2011). High concentrations of nitrate were found in groundwater associated with all land use categories (Ibid.).

The GAMA reports discussed here and in section 3 are those which specifically pertained to our study area. For more information on SWRCB's GAMA program, and additional reports, including reports on nitrate occurrence and fate, the reader is referred to SWRCB's GAMA website http://www.waterboards.ca.gov/gama/.

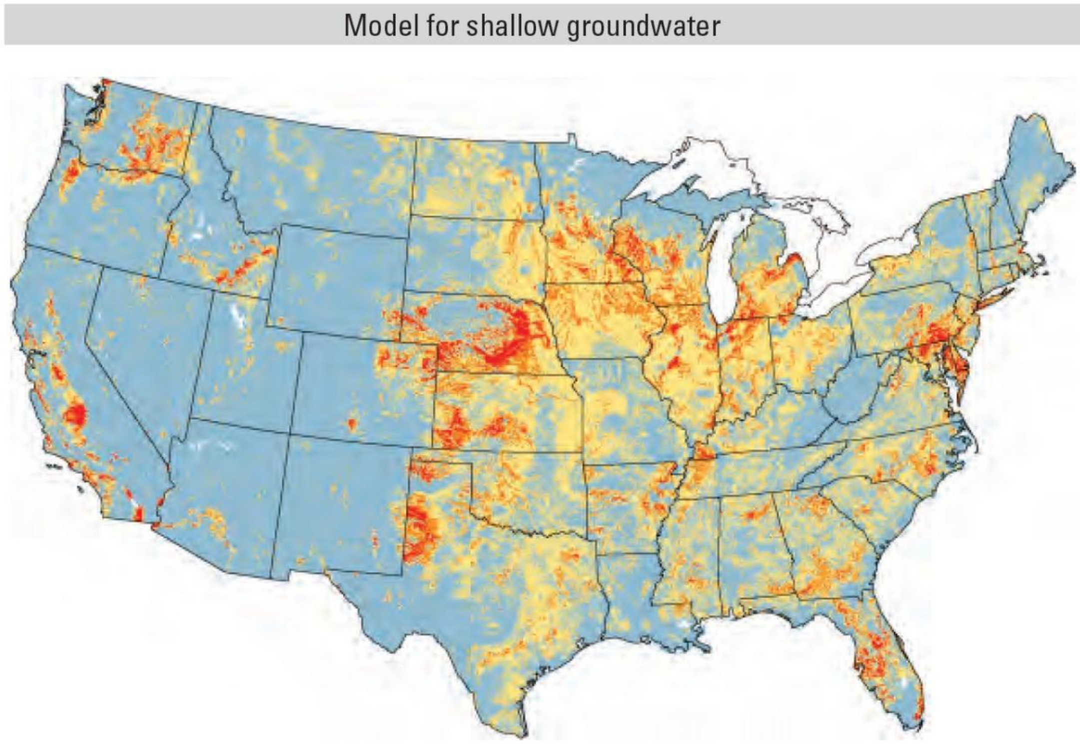

Figure 10. Nitrate levels in the uppermost 3 m of groundwater, simulated by statistical models based on general land use, groundwater, vadose zone thickness, and soils information. Red indicates concentrations above the drinking water MCL. (Source: Dubrovsky et al. 2010).

3 Salinas Valley Hydrogeology and General Water Quality

Prepared by:

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3.1 Physical Setting

3.1.1 Location

The Salinas Valley (SV), located about 160km (100 miles) south of San Francisco, CA, is an intermontaine valley which contains an aquifer system composed of fluvial and alluvial fan sediments. It is bound to the northeast by the Gabilan and Diablo Ranges, to the southeast by the Sierra de Salinas and Santa Lucia Ranges, and to the northwest by Monterey Bay. The valley floor is approximately 85 miles long, ranging from 3 to 10 miles wide containing roughly 91,134 hectares of agricultural land and 22,835 hectares of urban land, according to California Augmented Multisource Landcover (CAML) in 2010 (Viers et al. 2012). The study area consists of the SV groundwater basin from Monterey Bay, southeast to the town of San Ardo. The basin extends beyond San Ardo and is hydrologically connected to the Paso Robles Basin; however, the study area for this report was concerned with the major subareas of the Salinas River watershed (Figure 11) that lie within Monterey County. Almost all of the land is used for agriculture: lettuce (19,512 hectares or 48,215 acres); vineyards (19,234 hectares or 47,528 acres); and truck, nursery, and berry crops (17,165 hectares or 42,415 acres) make up the majority of the crops grown. Viers et al. (2012) provides a more detailed discussion on current and historic land use in the SV.

3.1.2 Subareas

The SV has been divided into four main subareas by the DWR (Durbin et al. 1978). These areas are known as the Pressure, Eastside, Forebay, and Upper Valley subareas (Figure 11). The four subareas do not correspond to subbasins or watersheds, but were based on the differences in hydrogeologic properties and sources of groundwater recharge (GW Recharge) (Durbin et al. 1978; Salinas Valley Ground Water Basin Hydrology Conference 1995). Durbin et al. (1978) discuss three important characteristics that differentiate the areas; confining conditions, specific capacity of wells, and the source of groundwater recharge (Table 3).

Confining Conditions

The Pressure area is generally thought to be composed of 3 semi-confined to confined aquifers known as the 180-ft, 400-ft, and Deep aquifers. The names are based on the approximate depths to reach each aquifer (Durbin et al. 1978; Boyle Engineering Corporation 1986; Montgomery Watson Americas, Inc. 1997). The Eastside subarea is considered semi-confined, the Forebay semi-confined to unconfined, and the Upper Valley subarea largely unconfined.

Specific Capacities

Specific capacities of wells generally increase up-valley. The Eastside area has an average of 447 m^2/day (25 (gal/min)/ft), the Pressure area 1073 m^2/day (60 (gal/min)/ft), the Forebay 1788 m^2/day (100(gal/min)/ft), and the Upper Valley 2682 m^2/day (150 (gal/min)/ft) (Durbin et al. 1978).

Groundwater Recharge

Previous work indicates that the Pressure area is recharged largely by irrigation and stream recharge in roughly equal amounts. The Forebay and Upper Valley areas receive recharge from irrigation return water and infiltration from the Salinas River, with estimates indicating that the river provides approximately twice as much recharge as irrigation return. The Eastside area is the only subarea that the Salinas River does not flow through, and the majority of its recharge is from irrigation return water (Durbin et al. 1978; Boyle Engineering Corporation 1986; Montgomery Watson Americas, Inc. 1997).

Table 3. Distinguishing characteristics of the four largest subareas of the SV, California. (Source: Durbin et al. 1978, Montgomery Watson Americas, Inc., final report 1997.)

Subbasin Name	Subbasin Area (acres)	Average Specific Capacity (gal/min)/ft	Confining Conditions	Dominant Source(s) of Recharge
Eastside	74,000	25	Semi-confined and unconfined	Irrigation Return
Pressure	91,000	60	Three confined aquifers	Irrigation Return and Salinas River
Forebay	87,000	100	Semi-confined and unconfined	Salinas River and Irrigation Return
Upper Valley	92,000	150	Unconfined	Salinas River and Irrigation Return

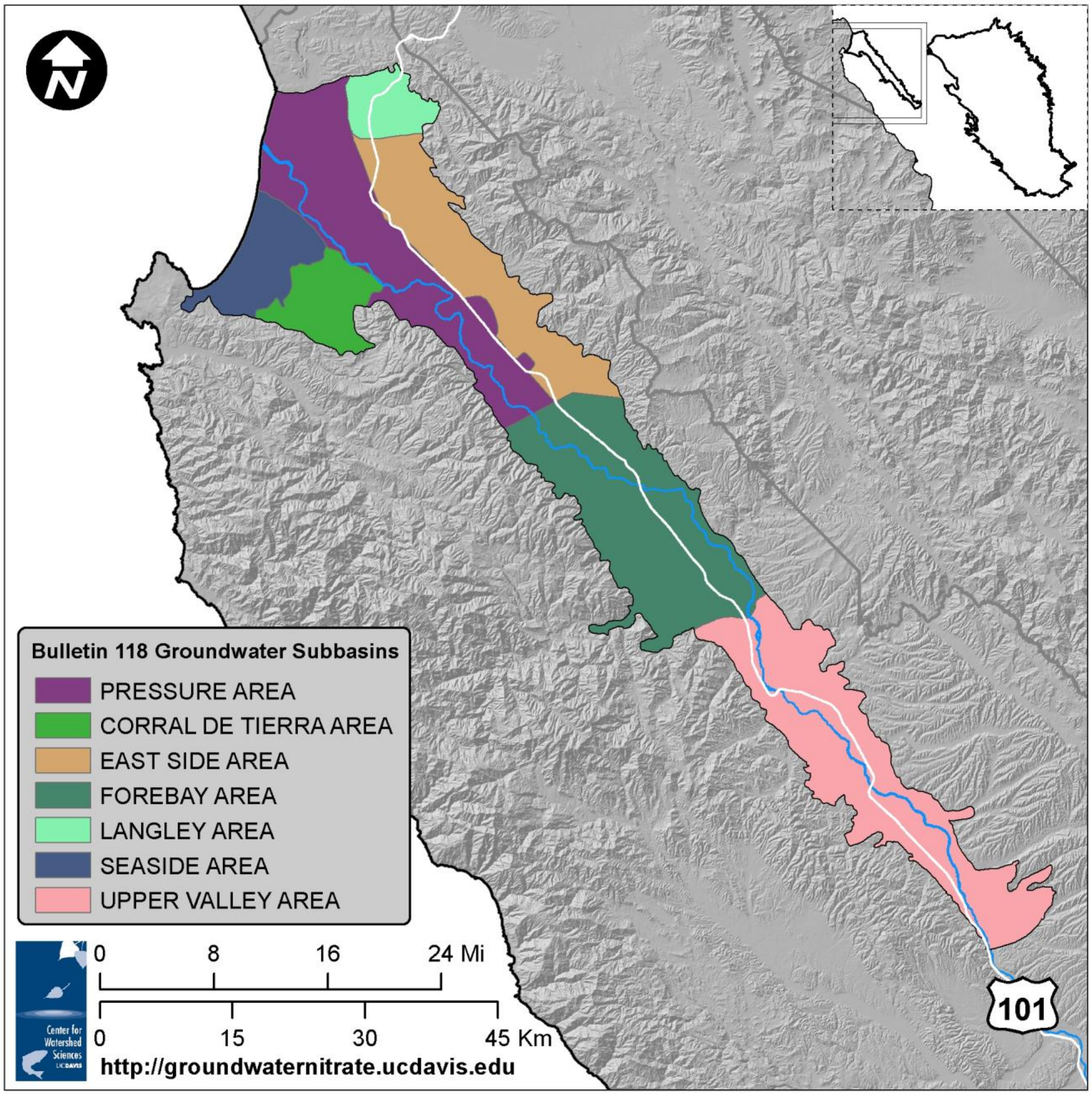

Figure 11. Seven subareas of the SV. The four largest areas considered are the Pressure Area (also referred to as the 180 ft/400 ft aquifers), the Eastside area, the Forebay Area, and the Upper Valley Area (Source: DWR 2003.)

3.2 Hydrology

3.2.1 Overview

The SV has a Mediterranean climate and receives precipitation almost exclusively in the winter months. The months from November to April account for 87% of the annual total, while the summers consist of moderate temperatures and very little rain (Planert & Williams 1995). Annual rainfall varies within the basin, averaging from 10 inches (25.4 cm) in the valley to over 30 (76.2 cm) inches locally at high altitudes (Durbin et al. 1978).

The Salinas River is the largest river in the valley, running the entire length of the SV. The river is also considered to be influent (loses water to the subsurface) for the majority of its length. Historically (before large scale development of groundwater) the river was likely a discharge zone for groundwater in the lower reaches, as is the case in most undeveloped basins. In 1846 the river was described as such: “as it approaches the ocean, is broad and fertile, and there are many fine ranchos upon it. But higher up, [the Salinas River] becomes dry in the summer” (Verardo & Verardo 1989 p.27). This implies that in the summer months, when there was almost no rainfall in the valley, the lower portions of the river still contained water flowing to the ocean, indicating that groundwater was discharging to the surface. In addition, the reclamation ditch (constructed in 1917) was built with the intention of draining the wetlands and lakes present in the lower basin (Casagrande & Watson 2006), again implying that groundwater was historically discharging to the surface. When groundwater pumping began in the valley in the early 1900s, the groundwater system of the valley was changed significantly. By the 1940s, the Salinas River was largely dry in the summer months. Due to its proximity to Monterey Bay, the groundwater basin also historically discharged to the ocean. This discharge mechanism has also been altered as seawater intrusion has been documented in the coastal regions since the 1930s (Montgomery Watson Americas, Inc. 1993).

3.2.2 Water Budget

Due to the seasonal variations in rainfall and river discharge, decreasing groundwater levels, and increasing seaward intrusion on the coast, the Nacimiento and San Antonio reservoirs were constructed (1957 and 1965, respectively) to regulate flow of the Salinas River to facilitate increased recharge to the groundwater basin. By sustaining a perennial flow in the Salinas River through controlled discharge from reservoirs, the groundwater basin has the potential to be recharged throughout year. Since construction of the reservoirs, multiple studies have concluded that stream recharge is the greatest contributor to groundwater recharge for the basin as a whole (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1993; Salinas Valley Ground Water Basin Hydrology Conference 1995). Stream recharge, however, is not distributed equally along the river, with some subareas receiving much more river recharge than others (Table 4). Shallow confining layers in the northern portion of the valley prevent significant infiltration, whereas in the southern valley the semi- and unconfined aquifers facilitate greater amounts of groundwater recharge. Groundwater level responses in each subarea reflect this unequal recharge as seen in Section 3.2.3 *Groundwater Levels*.

Irrigation is the second most significant source of recharge to the valley as a whole (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1993; Salinas Valley Ground Water Basin Hydrology Conference 1995). According to a 2009 Monterey County Water Resources Agency (MCWRA) groundwater summary report, the amount of groundwater currently pumped for irrigation in the valley totals 416,421 acre-ft (Monterey County Water Resources Agency 2011). Groundwater modeling from the Salinas Valley Integrated Ground Water and Surface Model (SVIGSM) indicates that between 1970-1994, an annual average of 189,000 acre-ft was recharged to groundwater as deep percolation (excluding river recharge), the majority as agricultural return. Evapotranspiration by crops and direct evaporation from soils account for the difference, removing roughly two thirds of the total applied. As more irrigation in the SV has been converted to micro-irrigation, recharge from irrigation has potentially decreased (Figure 12) although a significant potential for increasing irrigation efficiency in micro-irrigation often remains to be captured (Tim Hartz, personal communication). Precipitation and boundary flow (from adjacent land outside of the DWR boundaries) contribute to the recharge of the groundwater basin; however, these are likely small compared to irrigation return water and stream recharge (Montgomery Watson Americas, Inc. 1993).

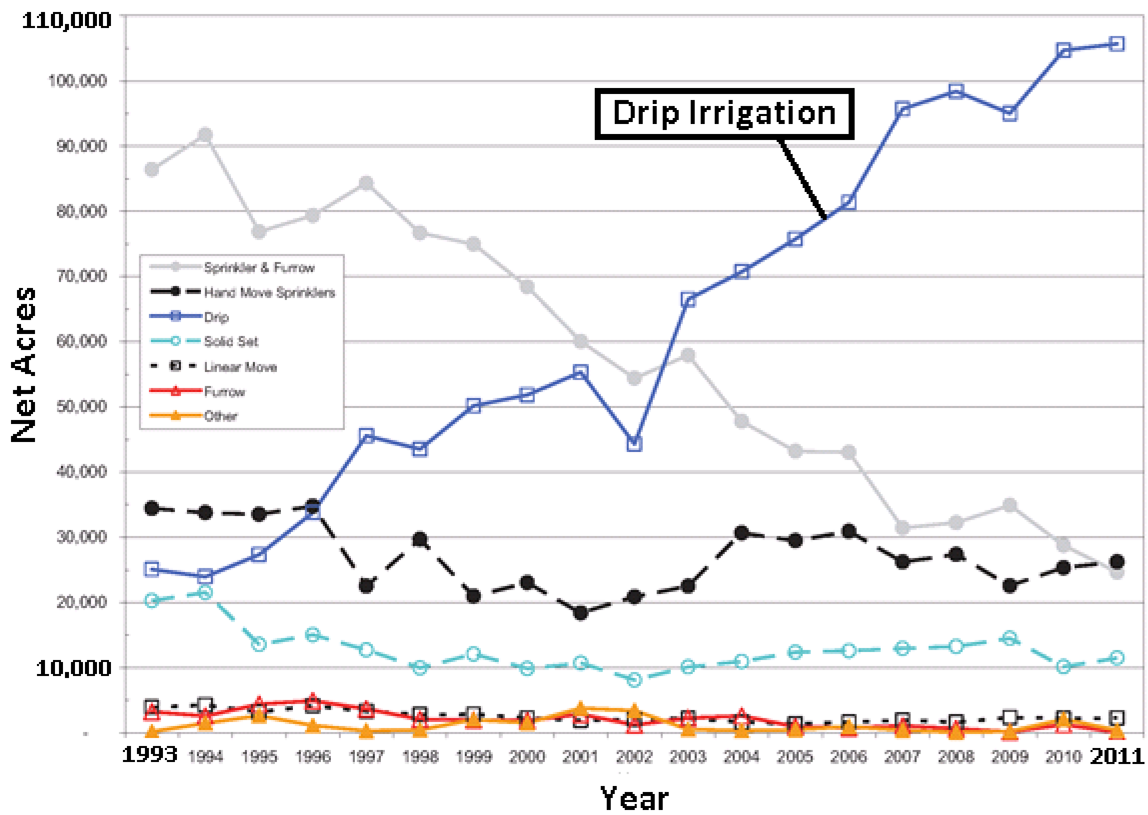

Figure 12. A significant increase in the use of micro irrigation has been observed in the SV. Reprinted with permission. (Source: MCWRA 2011.)

Discharge from the basin occurs mainly from groundwater pumping. According to a MCWRA 2010 Groundwater Summary Report, 460,443 acre-ft of water was pumped in 2010. Agricultural pumping accounted for 90% of all groundwater pumping, while urban pumping amounted to only 10% (Monterey County Water Resources Agency 2011). Portions of the Salinas River may still provide a discharge zone for groundwater in some locations; however, it is likely small compared to the discharge through well pumping.

A water budget for the SV (Table 4) comes from model outputs from the SVIGSM (Montgomery Watson Americas, Inc. 1997) and data from annual groundwater extraction reports published by MCWRA. The breakdown of agricultural pumping and urban pumping in Table 4 is based on MCWRA reported averages during 1995-2010, whereas the remaining values are averages from the SVIGSM (1970-1994). Model results show that the Eastside and Pressure subareas have experienced significant loss in storage during the 25 years modeled. The Forebay and Upper Valley have remained essentially constant, with the Upper Valley showing a slight increase in storage through time. Table 5 shows pumping broken down by city/area and comes from the latest groundwater extraction report provided by MCWRA (MCWRA 2011).

Table 4. Average values of groundwater budget in acre-feet/year.

In acre-feet/year	Pressure	Eastside	Forebay	Upper Valley
GW Recharge[1]	54,337	33,189	44,060	57,753
Stream Recharge[1]	59,958	1,830	102,634	98,956
Seawater Intrusion[1]	15,000	---	---	---
GW Pumping - Total[1]	133,176	84,833	157,585	143,826
GW Pumping–Agricultural[2]	101,203	86,013	140,919	133,389
GW Pumping–Urban[2]	22,227	12,402	7,076	4,255
Net Change in Fresh GW[1]	-11,000	-33,000	0	3,000

[1]Montgomery Watson SVIGSM model averages 1970-1994 (Montgomery Watson Americas, Inc. 1993).
[2]MCWRA extraction reports 1995-2010.

Table 5. Urban groundwater pumping by city or area. (Source: Monterey County Water Resources Agency 2011.)

City or Area	Urban Pumping-2010 (Acre-Feet/year)	Urban Pumping-Avg* (Acre-Feet/year)	Percentage of Total-2010	Percentage of Total-Avg*
Castroville	810	878	1.7%	1.95%
Chualar	121	132	0.3%	0.29%
Former Fort Ord	2,469	2,755	5.7%	5.97%
Gonzales	1,282	1,277	3.2%	2.75%
Greenfield	2,152	1,588	5.1%	3.49%
King City	3,089	3,415	6.3%	7.51%
Marina Coast WD	1,765	2,031	4.3%	4.47%
Other Areas	11,383	7,510	20.3%	16.38
Salinas	16,819	21,992	43.3%	47.79%
San Ardo	100	130	0.3%	0.29%
San Lucas	36	60	0.1%	0.12%
Soledad	2,293	2,329	5.3%	5.04%
Soledad Prisons	1,702	2,330	4.1%	4.93%
Total	44,022	45,960	100.0%	100%
*Averages calculated from 1995-2010 MCWRA extraction reports.				

3.2.3 Groundwater Levels

Development of groundwater in the SV basin initially caused a lowering of groundwater levels throughout the basin; however, the most lasting and significant drops have been observed in the Pressure and Eastside subareas. Since the construction of the Nacimiento and San Antonio reservoirs, water levels have risen to roughly 1944 levels and have become relatively stable in the Forebay and Upper Valley subareas. The Pressure and East Side subareas have shown continued decline, although the East Side has experienced the most significant decline. Figure 13 shows water levels from representative wells for the four subareas.

The differences in water levels over time for each subarea can be explained by the geology, groundwater pumping, and proximity to the Salinas River. According to the SVIGSM and through direct groundwater level measurements, the Eastside and Pressure subareas have experienced considerable overdraft during 1970-1994 (Montgomery Watson Americas, Inc. 1997). The Salinas River flows through every subarea except the Eastside, which is why increased recharge from the Salinas River has had little effect on water levels in this region. Almost all of the groundwater recharge in the Eastside subarea comes from irrigation return flow; however, this is far less than is pumped from the subsurface. River recharge and irrigation return in the Pressure subarea is limited by laterally extensive confining layers that thin out and disappear in the adjacent subareas (Kennedy/Jenks Consultants 2004). This inhibits recharging of the confined and semi-confined aquifers below. Groundwater pumping has generally exceeded recharge in both the Pressure and Eastside subareas, which has resulted in the loss of groundwater storage over time. The Forebay and Upper Valley subareas are for the most part unconfined to semi-confined, and the Salinas River is able to provide significant recharge to the aquifer system. On average, the river provides greater than or equal amount of water to the subsurface in relation to irrigation return in these subareas (Table 4). The combined recharge has matched or exceeded the amount of water pumped from the subsurface, which has allowed groundwater levels to recover. Model results indicate that the Upper Valley has even seen an average increase in storage through time.

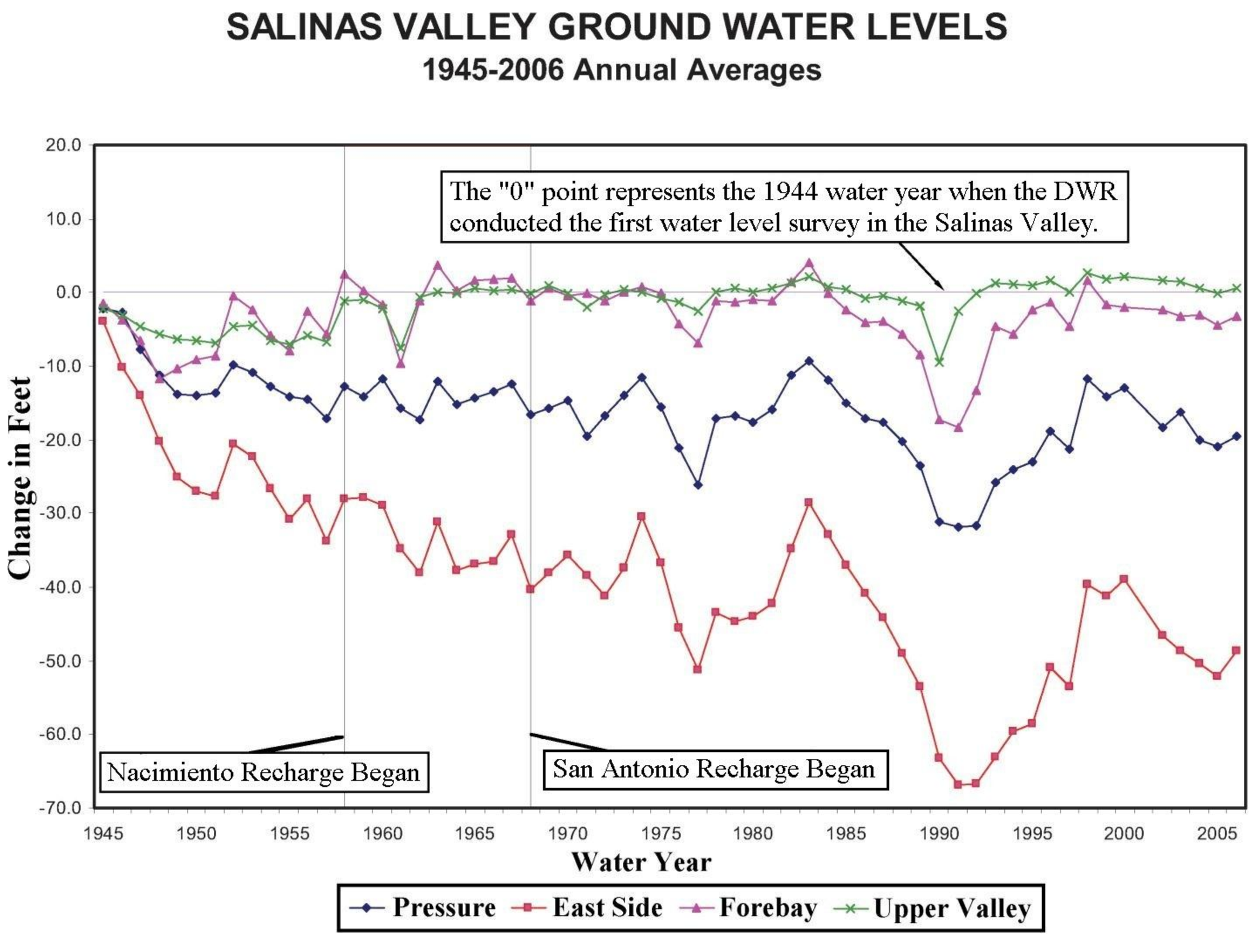

Figure 13. Groundwater levels in the subareas through time. (Modified from Johnson 2009.)

3.2.4 Seawater Intrusion

Seawater intrusion (Figure 14 and Figure 15) has been recognized in the SV since the 1930's (Casagrande & Watson 2006). The intrusion is the result of a reversal of groundwater flow near Monterey Bay.

Historically, fresh water likely discharged into the ocean below sea level where the aquifers outcrop into Monterey Bay; however, groundwater pumping has caused seawater to enter the groundwater basin. The reversal of flow can occur when the groundwater head in the aquifer is lower than that of an adjacent saltwater body (in this case, the Pacific Ocean). This creates a lateral hydraulic gradient, which results in lateral groundwater flow inland. Currently the saline/freshwater transition is operationally defined by TDS concentration of 500 ppm, and the time progression of this front for the 180-ft and 400-ft aquifers show that seawater has moved inland over time (Figure 14 and Figure 15, respectively).

A paper prepared for the MCWRA in 1999 offered a solution to the seawater intrusion problem which consisted of limiting pumping near the Monterey Bay area and delivering water to this area from upgradient (Salinas Valley Ground Water Basin Hydrology Conference 1995). The conclusion of the study was that seawater intrusion was not the result of a supply problem, but a problem of water distribution.

Seawater intrusion continues today; however, its rate of encroachment has been slowed due to the completion of the reservoirs and improved management strategies. In accordance with the recommendation of the Salinas Valley Ground Water Basin Hydrology Conference paper, a surface water diversion system is being built as part of the Salinas Valley Water Project to deliver surface water to the coastal areas near Castroville to replace groundwater pumping (Monterey County Water Resources Agency 2001). These and other basin supply studies can be found in Table 9 through Table 11 in Section 3.4.4.

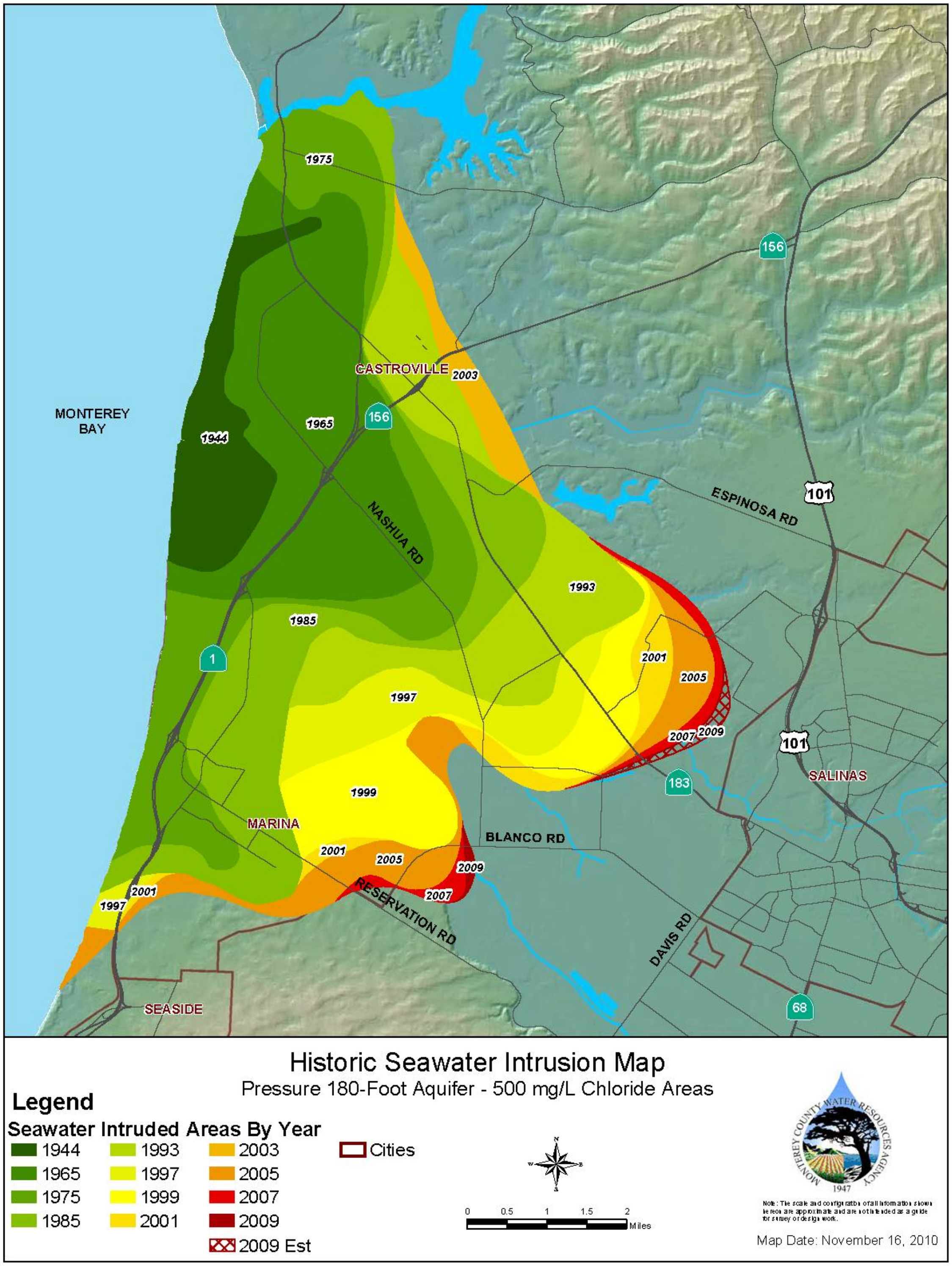

Figure 14. Map of seawater intrusion through time for the 180-ft aquifer. (Source: MCWRA 2010a.)

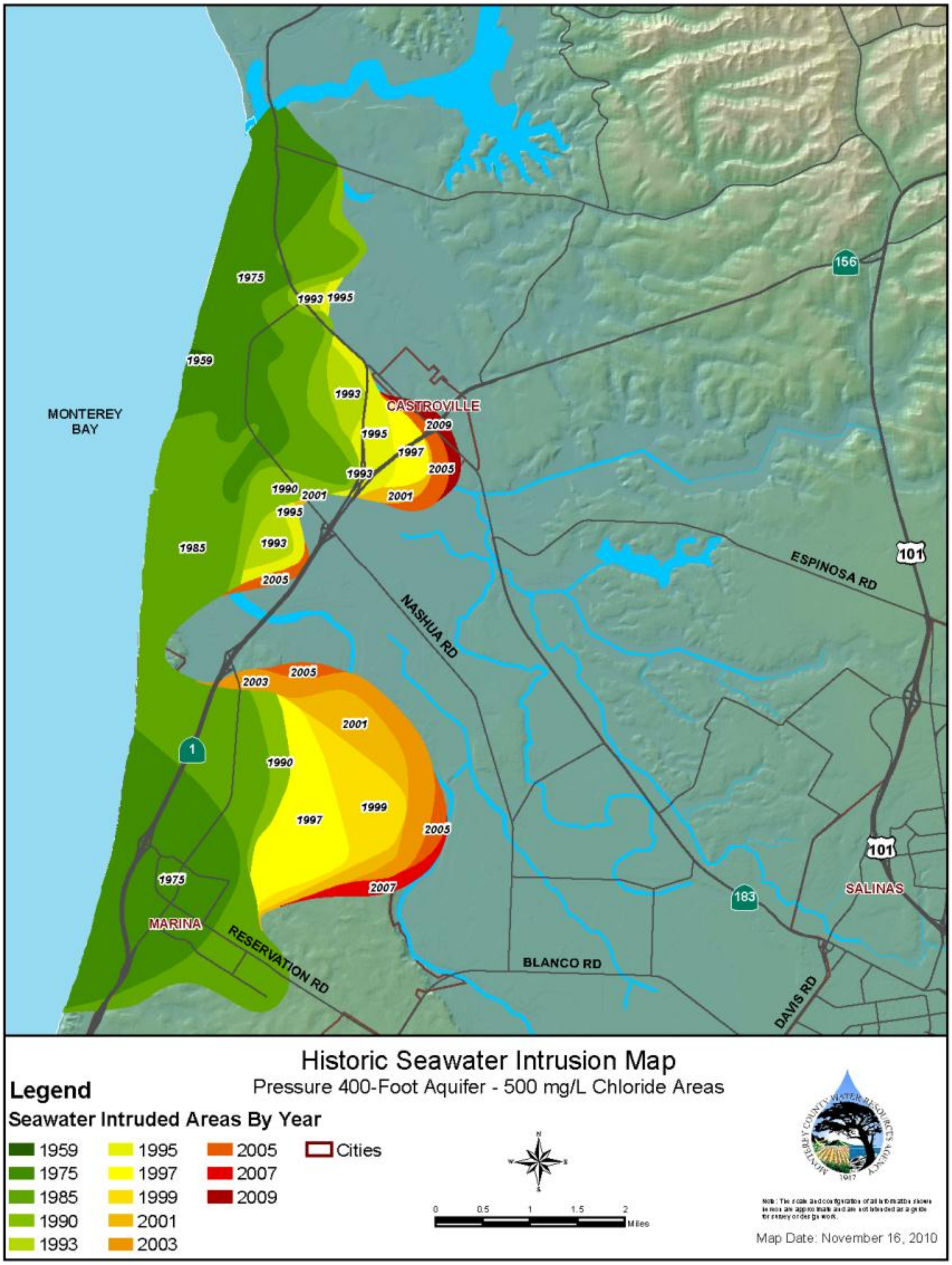

Figure 15. Map of seawater intrusion through time for the 400-ft aquifer. (Source: MCWRA 2010b.)

3.3 Hydrogeology

3.3.1 Geology

The aquifer system in the SV was deposited into a subsiding geologic basin, or trough, by fluvial processes (e.g., the Salinas River) and alluvial fans that mainly occur between the valley walls and the ancestral Salinas River. The bedrock underlying the sediments has an elevation, at its lowest point, of roughly 30m (100 ft) above sea level in the upper portion of the SV, but drops to over 600m (2000 ft) below sea level at its lowest point in the northern valley near Monterey Bay (Durbin et al. 1978). The geologic history of the basin involves a complex interaction with the adjacent sea as it has risen and fallen due to glacial periods. Recessions and transgressions of the ocean combined with alluvial and fluvial processes in the valley have resulted in a highly heterogeneous configuration of sediments. The aquifer system in the lower portion of the SV contains semi-continuous clay deposits thought to have been deposited by the marine transgressions. Such clay layers form the aquitards which create the confined and semi-confined conditions in the Pressure and adjacent subareas.

The general stratigraphic framework of the basin is composed of 3 hydrostratigraphic units, which sit upon a granitic basement and are overlain by recent (Quaternary) unconsolidated fluvial and alluvial sediments. The following geologic descriptions are largely from the 1978 USGS model report and 1997 MCWRA model report (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1997). The Monterey Formation, of Miocene age, is found above the basement granite and is composed of shale and large mudstone layers with thin sandstone beds towards the base. Lying conformably above the Monterey Formation and unconformably above the granitic basement (where the Monterey Formation is nonexistent) are the Santa Margarita/Purisima Formations (Pliocene). Stratigraphically similar, these two formations make up the bulk of the semiconsolidated rocks of marine origin. Both consist of a range of sedimentary rocks from sandstone to siltstone. Above the semiconsolidated marine deposits are unconsolidated materials which consist of the Paso Robles Formation (Pliocene/Pleistocene), Aromas Sand, and recent (Quaternary) alluvium. These unconsolidated sediments are of non-marine origin and make up the majority of the fresh water bearing aquifer material in the SV. The Paso Robles Formation is generally of fluvial origin; however, it contains sediment from alluvial, fluvial, and lacustrian environments. The recent alluvium consists of wind-blown sand, alluvial, and river sediments.

For the nitrate analyses in Section 5, the SV was divided into two regions as per a USGS GAMA 2005 study (Kulongoski & Belitz 2007). The two areas used were the Monterey Bay area, which encompasses the Pressure, Eastside, Langley, Seaside, and Corral de Tierra subareas, and the Salinas Valley area, which contains the Upper Valley and Forebay subareas (Figure 16). For the Monterey bay area, the subbasins were combined based on their similar Quaternary Deposits (Kulongoski & Belitz 2007). The Forebay and Upper Valley subareas were combined based on their similar geology (Kulongoski & Belitz 2007).

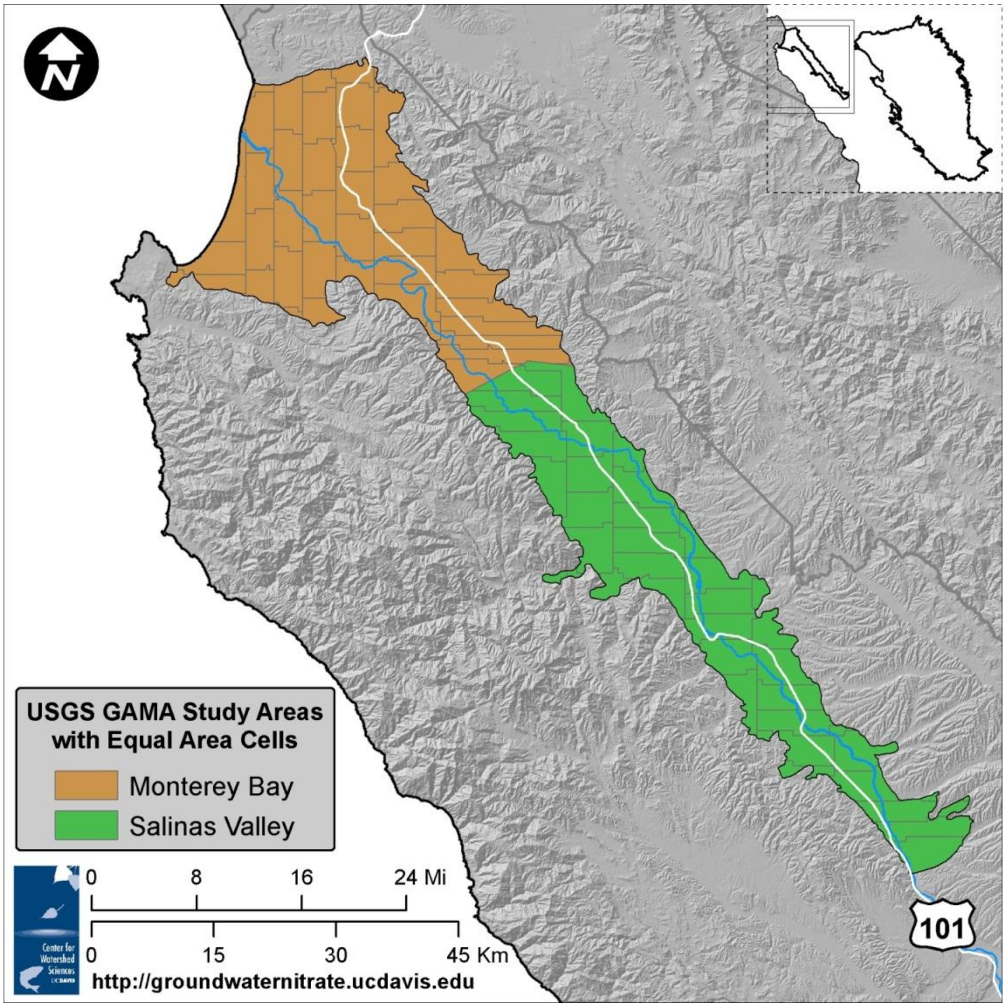

Figure 16. The two major groundwater regions used for the statistical analysis of groundwater nitrate data in Section 5. The equal area cells shown were used in spatial analyses to avoid spatial bias of the inherently clustered well test data (see Section 5) (Source: Kulongoski & Belitz 2007).

3.3.2 Aquifer Heterogeneity

It is well known that, within stratigraphic units that are composed of alluvial and fluvial sediments, the sediments are highly heterogeneous. Further, alluvial sediments that compose aquifers commonly consist of more aquitard sediments (silts, muds and clays) than aquifer sediments (sands and gravels), and the SV is no exception (Figure 17 and Figure 18).

To represent this heterogeneity and its effects on transport of nitrate, Fogg et al. (1999) created three-dimensional, geostatistical models of the dominant sediment textures in the Valley, and two examples are shown in Figure 17 and Figure 18. Geostatistical approaches produce more realistic models of spatially varying subsurface properties than simplified layered models. These models are particularly useful for representing the inevitable windows that breach aquitards (where commonly they are assumed and modeled as perfectly continuous). Fogg et al. (1999) demonstrated that the confining bed above the 180-ft aquifer could be an effective confining bed with respect to the hydraulic conditions in the system while still having enough coarse-grained pathways through them to allow localized nitrate contamination from the surface. This is consistent with the occurrence of localized nitrate contamination in the confined aquifers of the Pressure subarea.

For more information on the geology of the SV, refer to the list of hydrogeological studies in Table 9 through Table 11.

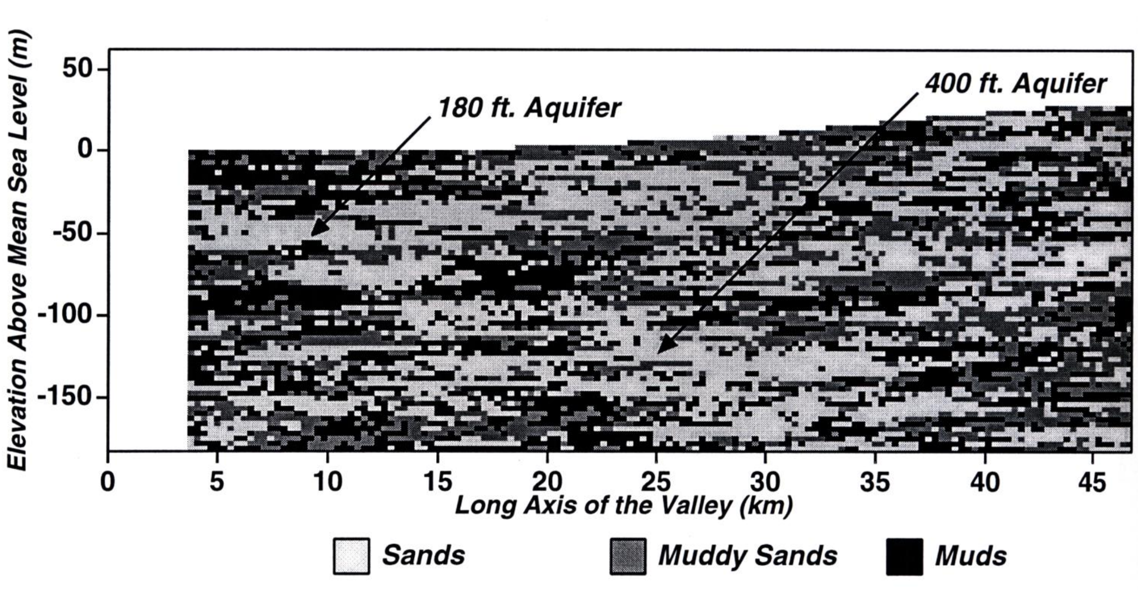

Figure 17. **Vertical cross section through a 3D heterogeneity model of subsurface soils in the Pressure subbasin of the SV. (Source: Fogg et al. 1999.)**

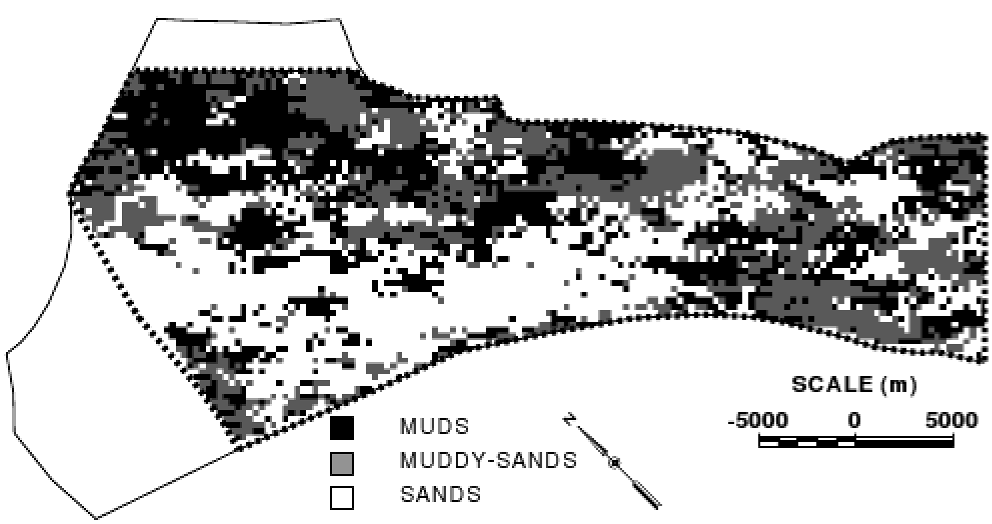

Figure 18. **Horizontal cross section through a 3D heterogeneity model of subsurface soils in the Pressure and Eastside subbasins of the SV. (Source: Fogg et al. 1999.)**

3.3.3 Flow Modeling

In the past, there have been several studies that have modeled all or portions of the SV aquifer system (Table 6). Initially the purpose of these models was to define a water budget for the basin. As models became more sophisticated, they were used to investigate the impacts of groundwater pumping on seawater intrusion. Of the many models, there have been three widely used computer models of the SV: Durbin et al. (1978), Boyle Engineering Corporation (1987), and Montgomery Watson Americas, Inc. (1997) models.

In 1978, the USGS, working with the United States Army Corps of Engineers, developed two transient models: a two dimensional model and a three dimensional model (Durbin et al. 1978). The goal of the model was to investigate the water budget of the aquifer system. The following year, a report was written explaining the results of various management scenarios ran with the model and also qualitatively discussing the results with respect to seawater intrusion (Kapple & Johnson 1979). Following the model's completion, there have been additional modifications to the original version. In 1985, Hydrocomp adapted the USGS model to investigate the effects of groundwater development of the Deep/900-ft aquifer in the Pressure subarea (Hydrocomp, Inc. 1985). In 1988, the model was modified to incorporate new water budget data to simulate basin management options (Yates 1988).

In 1986, Boyle Engineering developed a new model, FEGW-14 for the SV groundwater basin (Boyle Engineering Corporation 1986). The model was used to calculate groundwater budgets and quantify seawater intrusion. A 1987 report by Boyle Engineering provided an alternative analysis to the original report, investigating new water management scenarios. The model was updated again in 1990, incorporating new data and additional basin management scenarios (Boyle Engineering, Inc. 1990).

In the 1990's, Montgomery Watson Americas, Inc. developed an Integrated Ground Water and Surface Model code (IGSM) which was being adapted for several groundwater basins in California at the time. In 1993, the model was applied to the SV groundwater basin to investigate the groundwater budget and include quantitative simulations of seawater intrusion. A final report, an update from the initial 1993 draft report, was published in 1997 and explored various management strategies to increase infiltration and prevent further seawater intrusion (Montgomery Watson Americas, Inc. 1997).

Table 6. Groundwater models of the SV.

Title	Creator	Year	Discretization	SS/Trans.	Purpose
2-d and 3-d Digital Flow Models For The Salinas Valley Groundwater Basin, California	Durbin et al. (U.S. GEOLOGICAL SURVEY)	1978	Finite Element	Transient	Water Budget
Model Simulation of Various Water-Resource Management Alternatives in the Salinas Valley Ground-Water Basin, California	Kapple and Johnson (U.S. GEOLOGICAL SURVEY)	1979	Finite Element	Transient	Water Budget
A Steady - State Finite Difference Model Of The Groundwater Of Salinas Valley	Karen Nelsen	1985	Finite Difference	Steady State	Water Budget
Modeling of the Deep Zone In The Salinas Valley Groundwater Basin	Hydrocomp (Modified ’78 USGS model)	1985	Finite Element	Transient	Seawater Intrusion (Deep/900-ft aquifer)
Salinas Valley Ground Water Model	Boyle Engineering	1986	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Ground Water Model Alternative Analysis	Boyle Engineering	1987	Finite Element	Transient	Water Budget and Seawater Intrusion
Simulated Effects Of Ground-Water Management Alternatives For The Salinas Valley, California	Eugene Yates (Modified ’78 model)	1988	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Groundwater Model: Model Update And Future Hydrologic Analyses	Boyle Engineering	1990	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Integrated Ground Water And Surface Model	Montgomery Watson Americas, Inc.	1993	Finite Element	Transient	Water Budget and Seawater Intrusion
Salinas Valley Integrated Ground Water and Surface Model Update	Montgomery Watson Americas, Inc.	1997	Finite Element	Transient	Water Budget and Seawater Intrusion
Mapping Ground Water Susceptibility to Nitrate and Pesticide Contamination	Monterey County Water Resources Agency	1997	N/A	N/A	Ground Water Susceptibility
Groundwater Vulnerability Assessment: Hydrogeologic Perspective And Example From Salinas Valley, California	Graham Fogg, Eric Labolle, Gary Weissmann	1999	Finite Difference	Steady State	Nitrate Transport

3.3.4 Transport Modeling

Two regional scale groundwater vulnerability studies have been completed in the SV. The first was a Geographic Information System (GIS) based approach, which studied a portion of the SV from Chualar to Greenfield (Zidar 1997). The study used spatial maps of various parameters including agricultural use, soil texture, and depth to groundwater. These data were used with the Ground Water Loading Effects of Agricultural Management Systems (GLEAMS) model. The GLEAMS model is a root zone water quality model that provides an assessment of shallow groundwater vulnerability to contamination. Because the transit times of deeper groundwater from the surface to pumping wells are generally on the order of decades to centuries, the GLEAMS model does not account for effects of deeper hydrogeologic conditions on aquifer vulnerability to contamination. In addition to nitrate, other applied agricultural chemicals, such as pesticides, were investigated as part of the study.

A second vulnerability study (Fogg et al. 1999) used regionally detailed, 3D models of the subsurface heterogeneity to evaluate vulnerability of the deeper groundwater (which supply the irrigation and drinking water wells) to contamination from the surface. Backward-in-time, random-walk particle tracking was used to generate groundwater age distributions for groundwater produced by wells at a depth of 55 m (180-ft aquifer). Age distributions for individual wells were split into young (post WWII industrial production and wide spread use of artificial fertilizers) and old water (pre-industrial production of fertilizers). Wells with higher fractions of "young water" were assumed to represent wells which likely contained contaminated agricultural return water. Using a 50-year simulation, those wells which contained significant portions of young water were compared with current distributions of nitrate throughout the SV. Figure 19 shows that the simulated distribution of young water compared reasonably well with the nitrate spatial distribution in 1988 (concentrations shown are nitrate - as nitrate); wells that contained high fractions of young water were located in areas with known nitrate contamination. This demonstrated that, in addition to investigating surface processes, aquifer heterogeneity is a crucial component to determining the vulnerability of groundwater to contamination from the surface.

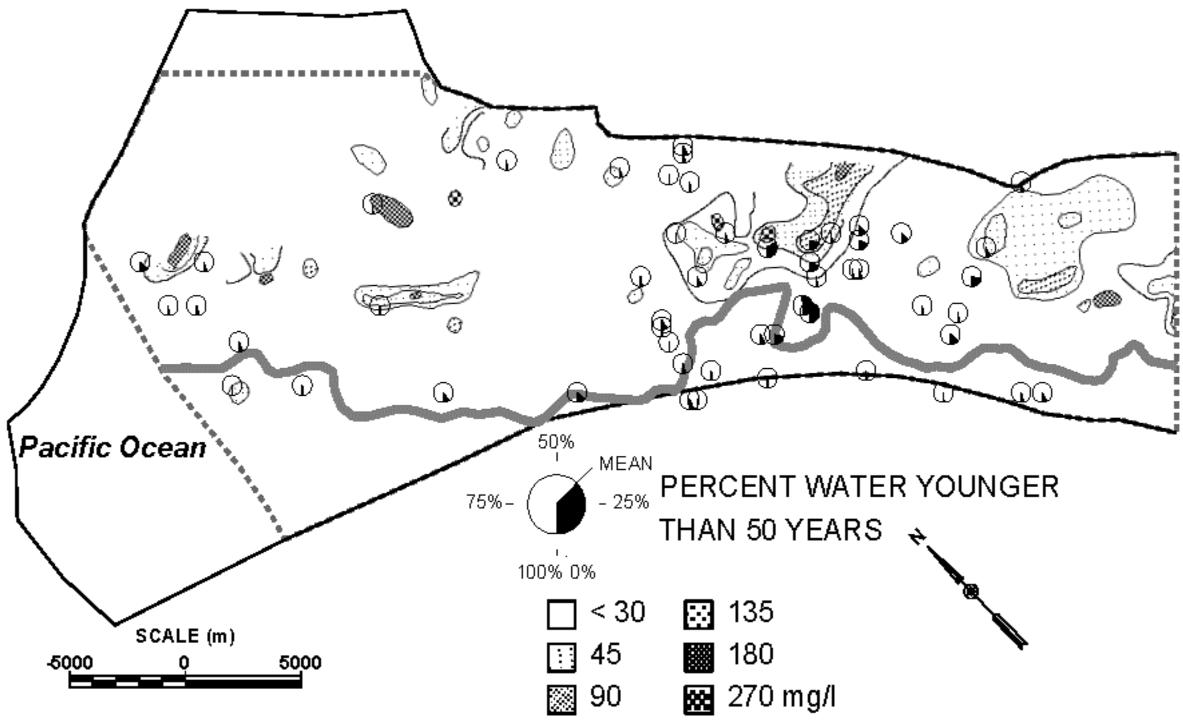

Figure 19. Wells showing (in black) the fraction of simulated young water less than 50 years overlain upon measured nitrate concentration contours. (Source: Fogg et al. 1999.)

3.4 Water Quality

In the SV basin, elevated levels of nitrate have been detected in groundwater since the 1950's (Zidar & Thomasberg 1995). Since the advent of synthetic fertilizers in the post WWII era, intensive agriculture has been the dominant form of land use in the SV. Nitrate travels at the same rate as groundwater and is often the first indication that a well may be compromised by contaminated groundwater recharge. In 1978, a study by the Association of Monterey Bay Area Governments (AMBAG 1978) determined that elevated nitrate levels in wells were primarily the result of agricultural practices at the surface, and recommended establishing a monitoring network for nitrate and other chemicals of interest (Snow et al. 1988). In response to this, MCWRA, working with the DWR developed an ambient monitoring program consisting of approximately 450 wells, the majority being irrigation wells. In addition to MCWRA's monitoring network, nitrate levels are regularly sampled from public wells to ensure that water meeting the U.S. EPA drinking water guidelines is being delivered. The Monterey County Department of Environmental Health (MCDEH) mandates that any water supply system with two connections or more must be tested annually. At the state level, systems with 15 or more connections (or serving more than 25 people for more than 60 days out of the year) are required to be tested annually.

3.4.1 Nitrate Data Collection

There are three programs that currently sample groundwater at the basin scale to assess the state of nitrate contamination in the SV. The first is the synoptic sampling of irrigation wells by MCWRA mentioned above. The second program, also run by MCWRA, samples a network of approximately 40 monitoring wells that are installed throughout the valley. The third is a program where public water systems are required to systematically test their well water, the results which are reported to MCDEH. In the past, a program by DWR (Bulletin 130 series) sampled irrigation, domestic, and public supply wells in the 1960's, the 1970's, and 1985 in 5 hydrologic regions of California (California Department of Water Resources 1963-1974, 1985). The data pertaining to the SV has since been incorporated into MCWRA's database. More recently, in 2005, a GAMA study by the USGS sampled public supply wells throughout the basin as part of a larger Priority Basins project. In 2011, Monterey County domestic wells were sampled for the GAMA Domestic Well program (http://www.swrcb.ca.gov/gama/domestic_well.shtml).[5]

3.4.2 Nitrate Analysis Reports

In the Central Coast region, Anton et al. (1988) describes the SV as severely impacted by nitrate. The report states that 113 of 1,207 wells of small water systems were in violation in 1986, and 6 of 180 wells of large systems were "closed" due to nitrate. In the unconfined aquifer regions of the SV, 48% of monitored wells exceeded the nitrate MCL, according to an unnamed study Anton mentions. The report also mentions that another study had predicted that all the unconfined aquifer regions of the SV would

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[5] Data from this survey were not available until 2012 and are not included in the CASTING dataset described in Section 4.

have nitrate concentrations of 1.9 to 4.4 times the MCL by the year 2000. A study in the Prunedale area of Monterey County found that 27% of the 154 “private and public” wells in the area were over the MCL for nitrate, and that both shallow and deep wells were affected.

The two *Nitrates in Groundwater* studies (Snow et al. 1988; Zidar & Thomasberg 1995), and Technical Memorandum (2010) in Table 7 are from the synoptic well sampling program performed by the MCWRA. The analyses largely consist of irrigation well sampling; however, they also include the 40 dedicated monitoring wells operated by MCWRA. The results from the studies are briefly discussed in Section 3.4.3 *Nitrate in Groundwater,* where the mean concentrations from each report are plotted in Figure 20. The two *Water Resources Data Reports* listed in Table 7 (Monterey County Water Resources Agency 1996, 1997) are from the same sampling program; however, they provide a snapshot of a particular year, rather than analyzing the changes in nitrate concentration over a time period.

As part of the California State Water Resources Control Boards’ GAMA Priority Basin Project, the USGS collected water quality, isotopic, and age tracer samples from public supply wells in 2005. Two reports have been published as a result of this sampling: a data summary report in 2007 (Kulongoski & Belitz 2007) and an interpretive report in 2011 (Kulongoski & Belitz 2011). Water supply systems with two or more connections require annual testing in Monterey County, and the results are reported to Monterey County Public Health. There are currently no reports regarding spatial and temporal trends in public supply wells of the SV; however, data from the reports found in Table 7 and public supply well data from MCDEH are included in the CASTING database. A description of the database and all of the data included in it can be found in Section 4. An analysis of the database is provided in Section 5.

Table 7. Groundwater sampling reports for the SV.

Title	Year(s) Published	Agency
Bulletin 130 Hydrologic Data: Central Coast Area	1963-1975, 1985	DWR
Nitrates in Groundwater 1978-1987	1988	MCWRA
Nitrates in Groundwater 1987-1993	1995	MCWRA
Water Resources Data Report, Water Year 1993-1994	1996	MCWRA
Water Resources Data Report, Water Year 1994-1995	1997	MCWRA
Ground-Water Quality Data in the Monterey Bay and Salinas Valley Basins, California, 2005—Results from the California GAMA Program*	2007	SWRCB
Technical Memorandum 1993-2007	2010	MCWRA
*An additional report (Kulongoski & Belitz 2011) provides an interpretation of the data in this report.		

3.4.3 Nitrate in Groundwater

While some well samples contain nitrate concentrations that are many times greater than the drinking water standard (MCL), the majority of the public supply wells in the SV have concentrations below the MCL (see Section 5 *Analysis of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater Database*). This is to be expected, given that any well with more than 2 connections is tested annually in Monterey County and reported to the MCDEH. If the nitrate concentration is found to be above 50% of the MCL, the well is required to be sampled every 3 months. Once the MCL of a particular contaminant is exceeded, wells are often destroyed, abandoned, or their use is discontinued. Once a well is found to be contaminated, further sampling is often not performed due to its destruction/abandonment. This effectively removes potentially high nitrate samples from the record, maintaining the statistic that the majority of wells sampled are below the MCL.

Of the 34 wells sampled for the GAMA program (31 public supply and 3 monitoring wells), nitrate plus nitrite (as nitrogen) was detected in 24 wells, with two (8%) of the samples being over the MCL (Kulongoski & Belitz 2011). This is consistent with our analysis of state and local public supply wells from the CASTING database, which show that generally one out of ten public supply wells are above the MCL. While an analysis of public supply wells provides information regarding the quality of water which is delivered for the purpose of drinking, a more insightful picture of the overall groundwater quality can be obtained through the synoptic well sampling program of MCWRA. This program samples irrigation wells on an annual basis, indiscriminate of the nitrate concentrations previously found. Some of the wells have been sampled repeatedly for nearly 60 years and contain concentrations of nitrate many times over the MCL.

MCWRA's long-term irrigation well sampling database was unavailable to our research group for reasons of confidentiality, nevertheless three reports analyzing this data and apparent trends have been published by MCWRA (Snow et al. 1988; Zidar & Thomasberg 1995; Monterey County Water Resources Agency 2010c). The mean (average) concentrations are plotted in Figure 20. The data show a general increase in nitrate concentrations through time for the valley, although the latest data for the Upper Valley subarea suggest that, on the average, concentrations did not changed significantly from 1993 to 2007. The Pressure subarea confined aquifers (Pressure 180, Pressure 400, and Pressure Deep) have the lowest average concentrations of nitrate compared to the other subareas of the basin, due to semi-continuous clay layers (aquitards) between the aquifers and the surface, slowing the transport process. The regression lines in Figure 20 show different rates of increase for nitrate concentrations ranging from 0 mg/L/year (Pressure Deep) to 2.23 mg/L/year (Upper Valley). The dip in values for 1993 is possibly explained by the number of wells analyzed for that particular year, roughly twice the amount as other years (Table 8).

While the SV has been home to many different land uses in the past and present (animal grazing, diary farms, vegetable row crops, pasture, etc.) LLNL found that the largest source of nitrate contamination to groundwater was from irrigated agriculture. Geochemical and isotopic results from groundwater and surface water samples containing nitrate above an established background level indicated that the nitrate contamination had a signature consistent with inorganic fertilizers (Moran et al. 2011).

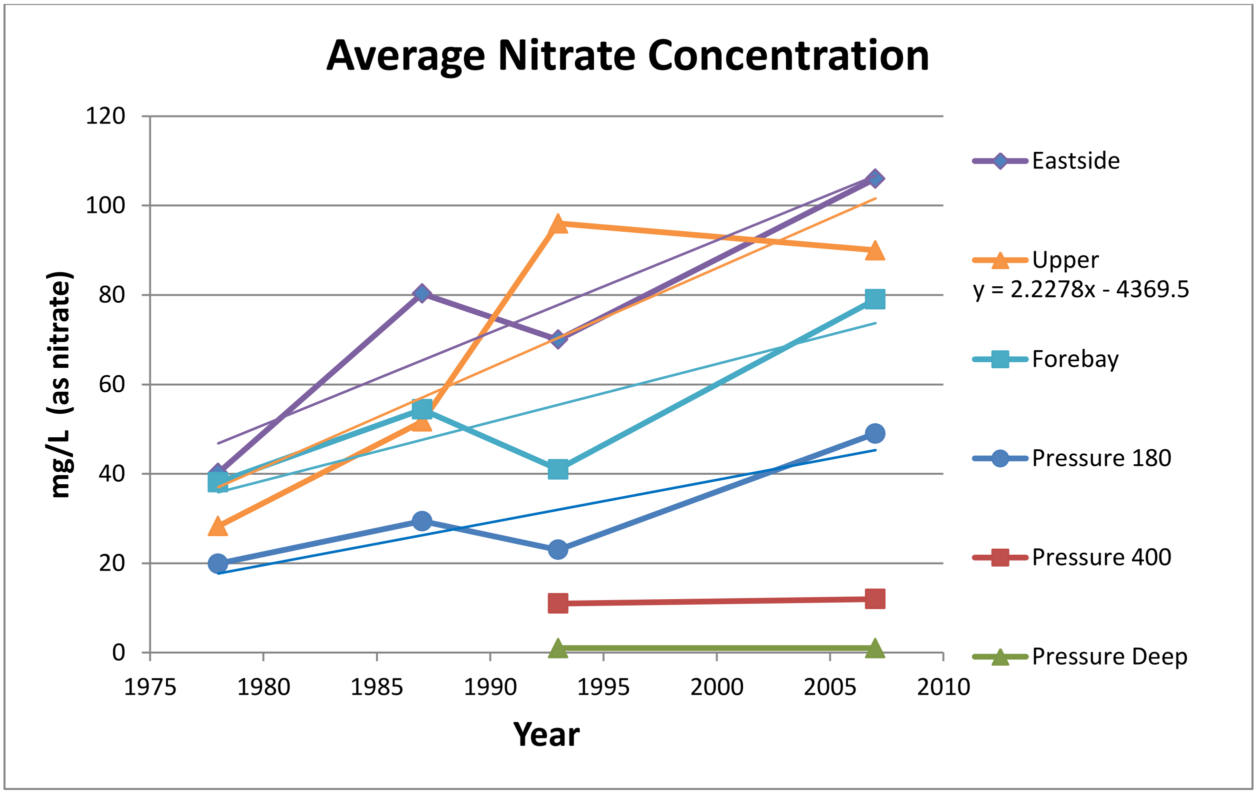

Figure 20. Nitrate trends from MCWRA synoptic sampling program. Values are average nitrate concentrations for each subbasin for 1978, 1987, 1993, and 2007. (Source: Snow et al. 1988; Zidar & Thomasberg 1995; Monterey County Water Resources Agency 2010c)

Table 8. Number of wells sampled by subarea and year as reported in MCWRA's *Nitrate in Groundwater* Reports.

SUBAREA	1978	1987	1993	2007
Pressure – 180 ft aquifer	61	61	95	28
Pressure – 400 ft aquifer	N/A	N/A	98	44
Pressure – Deep aquifer	N/A	N/A	6	5
Eastside	40	40	64	15
Forebay	34	34	73	41
Upper	16	16	34	19
TOTAL	**151**	**151**	**370**	**152**

3.4.4 Additional Reports

Table 9, Table 10, and Table 11 include a list of additional reports provided by MCWRA concerning the geology, water supply, and water quality in the SV. MCWRA has the following reports archived and is the organization to contact for obtaining those documents listed below that cannot be found through conventional sources.

The GAMA reports discussed here and in section 2 are those which specifically pertained to our study area. For more information on SWRCB's GAMA program, and additional reports, the reader is referred to SWRCB's GAMA website http://www.waterboards.ca.gov/gama/.

Table 9. Nitrate/water quality studies for the SV.

Title	Year	Author	Prepared For
Ground Water in California with Particular Reference to Salinas Valley	1949	T.R. Simpson	
CLEAN WATER: Water Quality Management Plan for the Monterey Bay Region	1978	Association of Monterey Bay Area Governments (AMBAG)	Monterey and Santa Cruz Counties
Nonpoint Sources Of Groundwater Pollution in Santa Cruz and Monterey Counties, California	1978	H. Esmaili and Associates, Inc.	
Report Of The Ad Hoc Salinas Valley Nitrate Advisory Committee	1990	Zidar et al. (1990) Monterey County Flood Control & Water Conservation District	
Demonstration Program to Reducing Nitrate Leaching Through Improvements to Irrigation Efficiency and Fertilizer/Cover Crop Management	1993	Varea-Hammond (1993), Monterey County Agricultural Extension Service University of California	MCWRA
Reducing Nitrate Leaching Through Improvements to Irrigation Efficiency And Fertilizer Management: 205 (J) Phase V Final Report to the State Water Resources Control Board	1993	Zidar et al. (1993), Monterey County Water Resources Agency	State Water Resources Control Board
Salinas Valley Ground Water Basin Seawater Intrusion Delineation/Monitoring Well Construction Program	1993	Staal, Gardner, and Dunne Inc.	
Wellhead Protection for Rural Communities Facing Threats from Nonpoint Source Nitrate Contamination, Case Study, King City, Salinas Valley, California	1995	Zidar (1995a), Monterey County Water Resources Agency	MCWRA
Irrigated Agriculture Technical Advisory Committee Report to the California State Water Resources Control Board	1994	Technical Advisory Committee	State Water Resources Control Board
Salinas River Basin Management Plan: BMP Task:2.06.1 Water Quality Assessment Report	1995	Win (1995)	MCWRA
Northern Salinas Valley Watershed Restoration Plan: Nonpoint Source Pollution in Coastal Harbors and Sloughs of the Monterey Bay Region: Problem Assessment and Best Management Practices	1997	Barron (1997), Moss Landing Marine Laboratories	Association of Monterey Bay Area Governments
2001 Nitrate Management Survey Results Report	2002	Monterey County Water Resources Agency	
Implementation of Public Outreach and Education Elements of the Salinas Valley Nitrate Management Plan	2003	Thomasberg (2003), Monterey County Water Resources Agency	Central Coast Regional Water Board State Water Resources Control Board
California GAMA Special Study: Nitrate Fate and Transport in the Salinas Valley	2011	Moran et al.	State Water Resources Control Board

Table 10. SV Basin studies.

Title	Year	Author
Salinas Basin Investigation	1946	Simpson (1946), Department of Public Works, Division of Water Resources
Data Report On Phase I - Salinas Valley Groundwater Basin Management Program	1974	Ott (1974a), Monterey County Flood Control and Water Conservation District
Summary Report On Phase I – Salinas Valley Groundwater Basin Management Program	1974	Ott (1974b), Monterey County Flood Control and Water Conservation District
Groundwater Management Measures for the Salinas Valley Corps of Engineers Urban Studies Report	1979	Army Corps Of Engineers
Optimal Conjunctive Use of Groundwater and Surface Water in the Salinas Valley	1985	Yates
Reconnaissance Report: Salinas Valley Water Transfer Project	1992	Laska
Groundwater Extraction Management Study Report	1992	Hurst (1992), MCWRA
Marina County Water District, Salinas Valley Groundwater Basin Overview	1992 (Draft)	Nolte and Associates
Water Resources Management in the Salinas River Basin. Volume 2: Review of Studies and Plans	1992	Price
Salinas River Basin Water Resources Management Plan study: Task 2.06.1 Report Water Quality Assessment	1992	Silas Snyder Larry O'Hanlon
Hydrogeology and Water Supply of Salinas Valley (White Paper)	1995	Salinas Valley Ground Water Basin Hydrology Conference
North Monterey County Hydrogeologic Study Volume I: Water Resources	1995	Zidar
North Monterey County Hydrogeologic Study Volume II: Critical Issues Report and Interim Management Plan	1996	Zidar
North Monterey County Water Issues Action Plan	1997	Barron
Monterey County Groundwater Management Plan	2006	MCWRA
Salinas Valley Water Project	2007	MCWRA

Table 11. Geological studies of the SV Basin.

Title	Year	Author	Prepared For
Geology of the Lower Portion, Salinas Valley Ground Water Basin	1969	Ford (1969), DWR	
Geology of Southern Monterey Bay and its Relationship to the Ground Water Basin and Salt Water Intrusion	1970	Green (USGS)	
Selected Geological Cross Sections in the Salinas Valley Using GEOBASE	1992	Hall	
Hydrogeologic Assessment - Salinas River Basin Management Plan Salinas Valley Geohydrologic Study	1994	Fugro West, Inc.	Montgomery Watson
Hydrogeologic Assessment. Salinas River Basin Management Plan Salinas Valley Geohydrologic Study: Construction Of Monitoring Well Clusters.	1995	Fugro West, Inc.	Montgomery Watson
Hydrogeologic Investigation of the Salinas Valley Basin in the Vicinity of Fort Ord and Marina Salinas Valley, California	2001	Harding ESE	

3.4.5 Nitrate Transport - Aquifer Heterogeneity & Groundwater Age

Rates of water movement are commonly slower in the unsaturated zone as compared to groundwater because the former have much lower hydraulic conductivity values than their saturated equivalents (Radclifee & Simunek 2010). A modeling study of vadose zone transport at the Wing Ranch in the SV showed a range of modeled travel times for bromide, a conservative tracer. Based on different heterogeneity realizations, travel times from the surface to the water table (35 m or 115 ft) were estimated to be from 4 to 14 years, depending on the framework of the subsurface sediments (Burow 1993). An approximate travel time of 10 years at that location was assumed reasonable. Nitrate is usually assumed to behave conservatively in the subsurface as well, and would be expected to have similar travel times. As a part of this study, advective travel times through the vadose zone were estimated on a regional scale, providing a spatial distribution of minimum and maximum travel times for the entire study area. The study involved 1D transport modeling with program HYDRUS 1D. Rather than explicitly defining the local heterogeneity of the unsaturated soils, the model was run with three homogeneous cases: sand, loam, and clay soil, thought to represent the fastest, intermediate, and longest probable travel times. Average travel times for the entire SV, based on the sand, loam, and clay soils are 5.57 years, 16.97 years, and 23.83 years, respectively. The maps and a description of the methods can be found in Section 6 *Groundwater Nitrate Forecasting: Assessment of Vadose Zone Nitrate Transport*.

After reaching the water table, water can take decades to centuries to migrate to deeper production wells (Fogg et al. 1999). Furthermore, wells draw in a mixture of waters that may have a broad distribution of ages, often ranging from years to centuries. Because of this, groundwater pumped by wells typically consists of both post-agricultural development groundwater that post-dates the introduction of large-scale use of fertilizer and irrigation, and pre-agricultural development groundwater that pre-dates the agricultural contaminant sources. One can think of the pumped groundwater as a mixture of old, pre-1940s-1950s, uncontaminated water and young, post-1950s, potentially contaminated groundwater. As time continues, the fraction of the pumped groundwater that is "young" or post 1950s increases, and the potential for upward trending nitrate concentrations increases if the sources of nitrate contamination have not diminished appreciably. Water found at shallow depths commonly has larger fractions of young water, and thus higher concentrations of nitrate associated with the recent recharged water. Local geology can often have a significant influence in the distribution of ages found in water due to areas of high hydraulic conductivity and areas of low conductivity (Fogg et al. 1999). The interconnectedness of the high conductivity sediments can lead to faster travel times, while areas of low conductivity can impede groundwater and associated contaminants. This explains, to some degree, why some deeper wells can have greater concentrations of nitrate than shallower wells. As discussed previously in Section 3.3.4, the vulnerability map of Fogg et al. (1999) showed that the "percent of young water" at a depth of 55 m (180 feet) compared favorably when overlain on top of a map of actual nitrate occurrence. Wells containing higher fractions of young water appear to roughly match the occurrences of higher nitrate concentrations found from well sampling (Figure 19). Denitrification reduces nitrate concentration. In the Salinaas Valley, denitrification has been found to occur in the immediate vicinity of the Salinas River and its contact with groundwater, and is likely to also occur in the clay layers confining the aquifers of the Pressure area. The potential role of denitrification is further explained in Appendix B of this report.

3.4.6 Groundwater Chemistry

Hydrogeologic characterization of groundwater includes major ion chemistry and its evolution along groundwater flow paths. The major ions present in water are calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), chloride (Cl^{-}), sulfate (SO_4^{-}), carbonate (CO_3^{2-}), and bicarbonate (HCO_3^{-}). The relative amounts of each constituent, when plotted on a piper diagram (Figure 21), provide information on regional flow paths and origins of the water. For example, Ca^{2+} and Mg^{2+} concentrations are often elevated in shallow groundwater and decrease along flow paths due to exchange for Na^{+} present on clay-rich sediments, resulting in more Na^{+} and less Ca^{2+} and Mg^{2+} the longer water has spent traveling through the subsurface. Additionally, groundwater tends to continuously dissolve carbonate and other minerals found naturally in geological materials as it travels through the aquifer system. This results in greater amounts of HCO_3^{-} in older groundwater.

Both trends (exchange of Na^{+} and increasing HCO_3^{-}) can be seen in the water samples taken from municipal wells in the SV. Before the development of groundwater resources in an alluvial-fill basin like the SV, the predominant direction of groundwater flow is usually horizontal, with relatively small vertical flow. Water tables also usually follow the slope of the surface elevation. Therefore, we would expect to find younger waters in the shallow unconfined Upper Valley subarea, and progressively older (and geochemically evolved) water to be found further down the valley.

Piper plots were constructed from samples collected from public supply wells (Figure 21) and reflect the expected geochemical evolutionary trend. In the Upper Valley, all water samples fall in the upper left portion of the diamond graph, meaning higher proportions of Ca^{2+} and Mg^{2+} (relative to Na^{+}) as well as lower proportions of HCO_3^-. The diagram suggests that the water here is relatively "young." Moving into the Forebay subarea, the data display a bit more scatter to the plot (some samples plotting further down and to the right), signifying that this water is more geochemically evolved and has traveled in the subsurface for a longer period of time. However, it is not until the relatively deep Pressure and Eastside aquifers that the more geochemically evolved water appears. The samples in the Pressure and Eastside aquifers, in general, plot much further down and to the right with less Ca^{2+}/Mg^{2+}, more Na^{+}/K^{+}, and more HCO_3^-. These relative proportions suggest older water, which fits our conceptual model of the geochemical evolution of groundwater in the SV where we expect to find older, more evolved water in the lower portion of the valley. The degree of scatter in the data also tells us something of the heterogeneous nature of the groundwater system in the SV. In the semi and unconfined aquifers of the Forebay and Upper Valley subareas we see primarily younger water, with less variability in their chemical evolution, which suggest faster travel times to supply wells. The Pressure and Eastside subareas show more scatter to the data which implies that some wells in these subareas are withdrawing very old water, while others are withdrawing younger water. The presence of confining conditions in these areas tends to cause longer travel times, while the heterogeneous nature of aquifers create a wide range of travel times.

Nitrate concentrations are also shown in Figure 21. Concentrations above 22.5 mg/L nitrate-nitrate (red, purple, and blue), generally plot in the upper portion of the diagram. Concentrations below this (green and yellow) plot further down. Groundwater containing nitrate concentrations over 22.5 mg/L likely has a large fraction of young contaminated water, which is reflected in the location where it plots in Figure 21. The location of groundwater samples containing nitrate concentrations below 22.5 mg/L on the piper diagram suggests that this water contains a higher fraction of older, pre-industrial, recharge water that has not been contaminated by recent (post-1950's) agricultural activities.

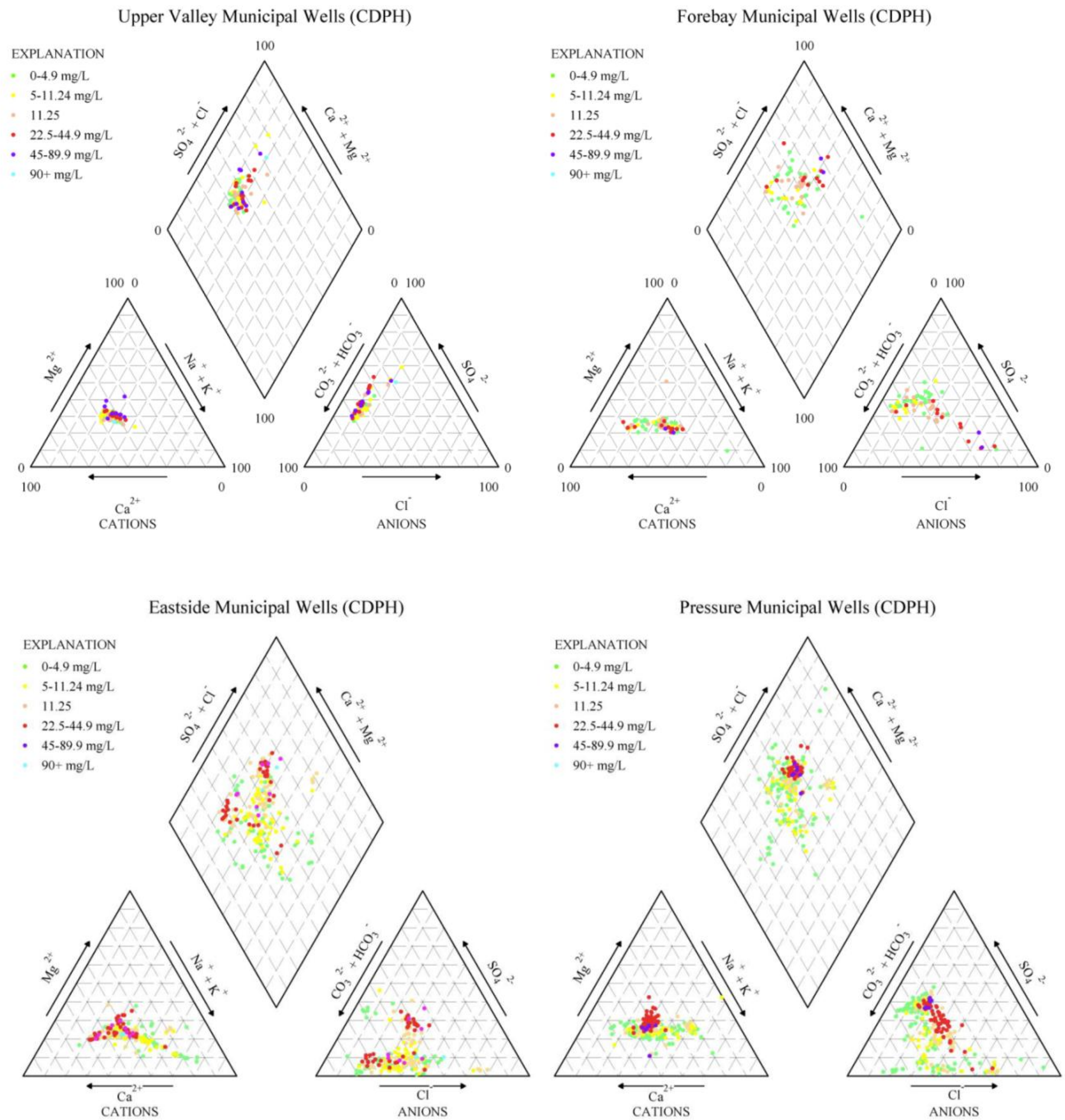

Figure 21. Piper diagrams of public supply water systems in the SV. Concentrations are nitrate as nitrate. Data is from the CDPH STORET database.

4 Development and Description of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database

Prepared by:

Aaron King, Thomas Harter, Dylan Boyle, Giorgos Kourakos

4.1 Introduction

4.1.1 Objectives

The objective of Section 4 is to describe the origins, development, organization, and scope of the database used by this study to examine current, historical, and rate of change of nitrate concentration in groundwater in the study area, the TLB and the SV. We acquired nitrate test data for groundwater in our study regions from federal, state, and many local agencies and reports; attempted to geo-locate every tested well, if at all possible, with an estimate of the spatial precision of the geo-location; used multiple, redundant methods to check for duplication within and between data sources; and created a geographic information system (GIS) database focused on sample x, y, depth, time, and nitrate concentration. In this section we describe the source datasets and document the protocols used to assemble the data into a single database, herein referred to as CASTING (California Ambient Spatio-Temporal Information on Nitrate in Groundwater). A discussion of the spatio-temporal distribution of available data on nitrate in groundwater is presented.

Note that well log information is confidential by current state law. Information from well completion reports and on the exact location of public or private wells will therefore not be made public.

4.1.2 Background

Nitrate test data are collected by federal, state, county, and local agencies for regulatory and scientific purposes. Agencies from which we obtained data for our study area include the United States Geological Survey (USGS), the United States Environmental Protection Agency (EPA), the United States Bureau of Reclamation (BOR), the California Department of Public Health (CDPH), the California Department of Water Resources (DWR), the California State Water Resources Control Board (State Water Board), the Central Coast (Region 3) and Central Valley (Region 5) Regional Water Quality Control Boards (Regional Water Boards), environmental health agencies of Fresno, Kern, Kings, Monterey, and Tulare Counties, Westlands Water District (WWD), the Kern County Water Agency (KCWA), and the Monterey County Water Resources Agency (MCWRA).

Under current drinking water regulations (U.S. Safe Drinking Water Act [SDWA] of 1974, amended 1986, 1996), the state and local regulatory agencies require public water supply system operators to regularly test their raw water and their finished (treated) water provided to households for biological, chemical, and radiological constituents, including a test of nitrate concentration. Water systems report their water quality data to the local agency or directly to CDPH, depending on their size. Nitrate concentration in drinking water is regulated at three levels in California. These regulatory levels are defined by the number of people and number of days served by a system, and by the number of connections served by each water system. Water systems with 15 or more connections or serving more than 25 people for more than 60 days a year are considered public water systems (PWS) under the federal Safe Drinking Water Act, and are regulated by CDPH.

The U.S. EPA distinguishes the following types of PWSs:

- Community Water System (CWS): A public water system that supplies water to the same population year-round.
- Non-Transient Non-Community Water System (NTNCWS): A public water system that regularly supplies water to at least 25 of the same people at least six months per year, but not year-round. Some examples are schools, factories, office buildings, and hospitals which have their own water systems.
- Transient Non-Community Water System (TNCWS): A public water system that provides water to 25 or more people, 60 days or more per year, in a place such as a gas station or campground where people do not remain for long periods of time.

CDPH uses the U.S. EPA definitions as well as further classifying water systems by number of persons and number of service connections (Table 12).

Table 12. Classification of water systems by constituency, connections, and duration of service per year.

<table>
<tr><td></td><td colspan="2">Connections:</td><td><5</td><td>5+</td><td><15</td><td>15+</td><td><200</td><td>200+</td></tr>
<tr><td>Duration of Service</td><td colspan="2">Persons served:</td><td colspan="3"><25</td><td colspan="3">25+</td></tr>
<tr><td>N/A</td><td>Small Water System (SWS)[1]</td><td rowspan="4">Classification defined by</td><td colspan="5">connections</td><td></td></tr>
<tr><td rowspan="2"><60 days/year</td><td>Local Small Water System (LSWS)</td><td>connections and (persons, duration)</td><td colspan="2"></td><td colspan="3"></td></tr>
<tr><td>State small Water System (SSWS)</td><td></td><td colspan="2">connections and (persons, duration)</td><td colspan="3"></td></tr>
<tr><td>≥60 days/year</td><td>Community Public Water System (CPWS)[2]</td><td colspan="3"></td><td colspan="3">connections or (persons, duration)</td></tr>
</table>

[1] Classification as a SWS does not preclude classification as any of the other types. SWSs may be regulated by CDPH or by LPA.
[2] A CPWS is a system for the provision of water for human consumption that has 15 or more service connections OR regularly serves 25 individuals at least 60 days a year (CDPH 2010 b,c).

From Table 12, systems with fewer than 200 connections are considered small water systems (SWSs). Under the provisions of Section 116330 of the California Health and Safety Code, CDPH has delegated regulatory authority over SWSs to 35 counties (called local primacy agencies, or LPAs). In our project area, the county environmental health departments of Kings County, Monterey County, and Tulare County are LPAs, but neither Fresno County nor Kern County have local primacy (i.e., in these counties, CDPH regulates SWSs). Fresno County had LPA status until 2006, when they relinquished that responsibility to CDPH.

Drinking water systems serving between five and 14 connections are referred to as “state-smalls,” and are regulated by the county, by CDPH, or not at all, depending on the county. Systems serving fewer than five connections are referred to as “local-smalls” and are regulated at the discretion of each county.

In general, household (domestic) wells are not regulated, although in the last decade many counties have begun to collect a single nitrate test from each newly drilled drinking water well at the time the well is drilled, including domestic wells (in our study area, this includes Fresno, Tulare, Monterey, and Kern counties).

CDPH maintains the largest portion of the regulatory test result data in the Permits, Inspections, Compliance, Monitoring, and Enforcement system information database (PICME). PICME includes data collected on all PWS sources of drinking water - surface water or groundwater, treated or untreated, active or inactive. Samples are collected by licensed water system operators and taken to certified laboratories that upload the resulting water quality data to PICME. The PWSs listed in PICME serve drinking water to approximately 95% of California’s residents.

Besides the public health regulatory agency data, there are several other sources of nitrate test results for wells in our study area, which are summarized below. For each of these data sources we include information on the specific intention/purpose of the program, the scope of the program (where/how data were collected, the targeted group of wells, timeframe of monitoring), and the types of information included in the database. These data were compiled, reviewed, and assembled into a new database that is here referred to as the CASTING (California Ambient Spatio-Temporal Information on Nitrate in Groundwater) database. The CASTING database serves as an intermediate electronic storage format from which data will be transferred to Geotracker GAMA, the State Water Board’s groundwater quality database. The following sections review the data sources for CASTING, the procedures used for assembling the various data into CASTING, and the quality assurance/quality control procedures that were applied.

4.1.2.2 Privacy Concerns

Currently, neither the locations, nor the physical characteristics of privately owned wells (whether part of public water systems or private domestic drinking water systems or household wells) are public information. Although DWR maintains copies of all well driller logs, that detail the engineering, geology, and location of all drilled wells in the state, these logs are themselves not public information. In the process of our analysis, we obtained permission to access many well driller logs, as well as precise locations and other private information from various agencies. The permission we obtained included license to use the data, but not to disseminate the data. Therefore, precise locations and physical characteristics of particular wells are not disclosed within this report.

4.1.3 Data Sources

CADWSAP (California Drinking Water Source Assessment Program)

Purpose – Under the federal Safe Drinking Water Act (SDWA), source water assessment programs (SWAP) for surface and groundwater were mandated to produce geo-locations (typically as geographic positioning system (GPS) coordinates) and vulnerability assessments for all active drinking water sources. CADWSAP is the California implementation of the federal SWAP program. The CADWSAP database is housed at the Information Center for the Environment (ICE) at UC Davis, and consists of a subset of PICME data that is intended to include all active source waters, with GPS coordinates for each water source location.

Scope – Within our study area, CADWSAP provided 3,616 wells with 60,792 individual nitrate samples dating from the early 1980's to 2010.

Data – In CADWSAP, nitrate data are reported as NO_3-N, NO_3-NO_3, or NO_3-N + NO_2-N combined, all in mg/L. These wells are all geo-located with GPS and are considered to have high spatial precision (better than 1 m accuracy for spatial location). Many of these well records also include the depth to the top of the well screen and the length of the well screen (perforated interval). CADWSAP wells are identified by a unique Primary Station Code (PS-Code) used only by CDPH and contributing counties. The format of the CADWSAP database provided the template for the CASTING database.

GAMA Domestic Well Project

Purpose – The GAMA program was established under the California Groundwater Quality Monitoring Act of 2001 (AB 599 2001) to provide a statewide comprehensive assessment of groundwater quality. The GAMA program is divided into three projects: the domestic well water quality (this section), priority basin water quality (next section), and special studies. No special studies projects were completed within our study area by the date at which we stopped adding datasets, however, several special studies were completed in 2011[6] that are not included in our database. The GAMA Domestic Well Project, sampled for nitrate (and other constituents) in household wells starting in 2002. The objective of the project is to develop a baseline for assessment of drinking water quality in household wells.

Scope – The domestic well project has collected data in six counties to date, of which Tulare County, (sampled in 2006) and Monterey County (sampled in 2011) are in our study area. As with the special studies, the Monterey County Domestic Well study was not completed before our final cutoff date for

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[6] 1). California GAMA Special Study: Nitrate Fate and Transport in the Salinas Valley Jean E. Moran*, Bradley K. Esser, Darren Hillegonds, Marianne Holtz*, Sarah K. Roberts, Michael J. Singleton, and Ate Visser.
2). California GAMA Program: Impact of Dairy Operations on Groundwater Quality. Bradley K. Esser, Harry R. Beller, Steven F. Carle, G. Bryant Hudson, Staci R. Kane, Roald N. Leif, Tracy E. LeTain, Walt M. McNab and Jean E. Moran. Lawrence Livermore National Laboratory, P.O. Box 808, Livermore, CA 94550.
3). Denitrification in a Shallow Aquifer Underlying a Dairy Farm: New Approaches to Characterization and Modeling: UCRL-PRES-207404

adding data into CASTING, and is not included here. The Tulare County project collected one sample per well and collected samples from 181 wells out of the approximately 26,000 domestic wells in Tulare county. Of these, 141 wells are within the study area.

Data –Highly accurate GPS data exists for these wells; however, for confidentiality reasons, wells are randomized to within ½ mile from the actual well locations. The nitrate concentrations reported by the GAMA program are in units of mg/L as nitrate. Well depths are available for some wells as reported by the well owners. The accuracy of the well depth information is uncertain. In general, domestic wells are drilled to depths less than 100 m (300 ft). GAMA Domestic Well Project wells are identified by a GAMA ID assigned by the State Water Board.

GAMA Priority Basin Project

Purpose – The GAMA Priority Basin Project, conducted by the USGS in coordination with the State Water Board (see previous section), was designed to assess water quality conditions in key groundwater basins (priority basins) that provide over 90 percent of all groundwater use in the State. This project focuses on the quality of ambient groundwater in aquifers used for drinking water supply and to establish a baseline groundwater quality monitoring program.

Scope – The Priority Basin Project has sampled wells from 116 groundwater basins (as defined by DWR Bulletin 118), divided into 35 study units. The Southern San Joaquin Valley study unit that overlapped our study area was completed in 2006. The Monterey/Salinas study unit was completed in 2005. The wells for the Priority Basin Project were sampled one time each, and consist of water supply wells, groundwater monitoring wells, and irrigation wells. In these Priority basin study units, samples were collected from 141 wells, 83 of which are located within our study area. Groundwater quality samples were collected and analyzed, at low detection limits, for a wide variety of natural and anthropogenic chemical (including nitrate), bacteriological, and radiological constituents. The Western San Joaquin Valley study unit was completed in 2010 and the data are not available as of this writing.

Data – These wells were geo-located using field GPS by USGS field personnel and are assumed to be accurate to within 7 m (20 ft). Nitrate concentrations in this dataset are reported in mg/L as nitrate. Well construction information was available for these wells in most cases. The construction information was obtained by the USGS for the Priority Basin Project from DWR well-logs. CADSWAP depth data was used in cases where construction data were not available in the Priority Basins reports. Priority Basin wells are identified by a unique ID code assigned by USGS.

USGS - NWIS

Purpose – The National Water Information System (NWIS) is the general clearinghouse database of locations for the USGS. The purpose of the database is to maintain continuity of identification and attribution of data collections across the many studies conducted by USGS on wells and other locations.

Scope – Within our study area, NWIS included 1,735 wells with 3,850 test results. NWIS identifies wells by the state well number (SWN). About 450 of the NWIS well SWNs matched wells already in the

CASTING database. Since many agencies and programs, notably the CDPH database and the Central Valley Regional Water Board (CVRWB) dairy database, do not use the SWN, an unknown number of the NWIS wells may overlap with other wells in CASTING. The tests are unlikely to be duplicates, as the tests in NWIS are only those collected by the USGS.

Data – NWIS wells are geo-located by various methods but these methods are not recorded in the database. It is therefore not possible to determine the accuracy of the reported geo-locations. The nitrate data are reported in mg/L as nitrate or in mg/L as nitrate-nitrogen. Nearly all of these wells included the depth of the completed well, but the top of the screened interval was not available.

Westlands Water District Irrigation Wells Project

Purpose – In 2008 and 2009, certain groundwater wells within the District were tested for water quality as part of the Canal Integration Program (CIP) and the Distribution Integration Program (DIP). The CIP allowed wells of sufficient water quality to pump into the San Luis Canal for distribution to downstream laterals. The DIP provided the growers an opportunity to pump water into the District's lateral for use on downstream lands served by the lateral. DWR was involved in a one-time sampling program designed to assess the water quality in irrigation wells in Westlands Water District. We obtained data from both Westlands Water District and from DWR.

Scope – The DWR project included a set of 39 wells with one water quality sample each, taken in 2008. All wells are located in Westlands Water District, 36 in western Fresno County and six in western Kings County. These wells were primarily irrigation wells and none were duplicates of CADWSAP wells. The Westlands Water District dataset included 84 water quality records from 71 wells, including 31 of the DWR wells.

Data –Well locations of 42 wells reported by DWR were geo-located with GPS by DWR and are accurate to within 7 m (20 ft). The nitrate concentrations in this dataset are reported in mg/L as nitrate-N, and were converted for our purposes to mg/L as nitrate. The remainder of the wells in the Westlands Water District database were identified by the State Well Number (SWN), defined below. No depth data were collected for these wells.

State Water Board Environmental Monitoring Wells

Purpose – The State Water Board maintains a large number of wells for monitoring of water quality at environmental cleanup sites. These wells are drilled specifically to monitor contaminated groundwater sites, such as leaky underground storage tanks, and other spills.

Scope – There are 537 monitoring wells with nitrate test results in our study area. Many other wells are monitored that do not have nitrate test results and these other wells were not included in our dataset. The monitoring wells are typically placed in clusters around cleanup sites, and are often not representative of regional groundwater quality.

Data – Nitrate concentrations in these wells are often quite high, as nitrate is frequently associated with the cleanup sites. These well locations are geo-located and are accurate to within 7 m (20 ft). The nitrate concentrations in this dataset are reported in mg/L as nitrate-N, and are converted for our purpose to mg/L as nitrate. These wells are identified by a unique code that is only used at the State Board, although a few also referenced the State Well Number.

U.S. EPA STORET Database

Purpose – The U.S. EPA warehouses data from many agency sources in the STORET legacy files database. These data were provided to the STORET facility by EPA's Region 9 water agency, the US Bureau of Reclamation, and California's State Water Board. Data from many different projects are collected in the STORET dataset and the original purpose of the underlying data collection efforts are not retained as an information item in the legacy database that we accessed for this project.

Scope – The U.S. EPA provided us with 16,031 samples from 7,083 wells. These wells were located throughout the TLB study area. For the SV, the Public Lands Survey System (PLSS) coverage of quarter-quarter sections is not available, which prevented us from locating wells using the State Well Number (SWN) system used in that study basin.

Data – These wells were not provided with accurate location data. However, all but 12 of the records included a SWN, so it was possible to locate the wells to the so-called quarter-quarter section, an area of 16.2 hectares (40 acres) comprising one of 16 (4x4) squares within a one-square mile (640 acre) land section. Only those wells with SWNs that matched a quarter-quarter section located within the study area were considered for inclusion in our database.

Without precise locations or any way of cross-referencing to other identification systems, it was unclear how many of the STORET tests and wells are duplicates of data that were already obtained from other databases (e.g., from PICME). To avoid duplication, we included only those wells from STORET into the CASTING database that were located in quarter-quarter-sections for which CASTING did not have a record of nitrate data. The STORET data do not include construction information such as well depth or screened interval. The test results are reported to U.S. EPA in several different concentration units, including mg/L NO_3-NO_3, mg/L NO_3-N plus NO_2-N, and mg/L NO_3-N. For use in CASTING, we converted all values to mg/L as nitrate.

Central Valley Regional Water Board Dairy Wells Monitoring Data

Purpose – Since 2007, the Central Valley Regional Water Board (CVRWB) has collected nutrient and chemistry data from irrigation, domestic, barn, and monitoring wells sampled by dairy owners. This groundwater monitoring and reporting program is the Central Valley Dairy General Order.[7]

[7] http://www.swrcb.ca.gov/rwqcb5/board_decisions/adopted_orders/general_orders/r5-2007-0035.pdf

Scope – We obtained the 2007, 2008, and 2009 data reported by dairies to CVRWB from the CVRWB Fresno office. Results are provided to CVRWB in annual reports provided by each dairy. The dataset includes 11,300 samples from 6,459 presumably unique wells sampled in 2007, 2008, and/or 2009.

Data – Data included dairy name and address, owner-selected names of wells, nitrate data, and date of the sample collection. No information was provided on the well construction details or well use. Well names in some - but not all - cases were indicative of the well use (e.g., "irrigation well #3"). Sample collection is once annually, but not necessarily in yearly intervals. Since well names are provided by the owners or their consultants, naming inconsistencies between years result in uncertainties about the identity of wells between years. For our analysis, we assumed that nitrate records were from the same well only if names between years were identical. These data were provided in spreadsheets. While Regional Water Board 5 also has maps for many of the dairies that identify the well location, these maps were not digitally geo-located for the SBX2 1 project. Instead, wells are associated with the address location and assumed to be within approximately 1 mile of that address. The only spatial reference available is the street address. It is unknown how many of these addresses represent the well locations and how many represent the business office or home of the dairy operator. Unfortunately, some dairies from Tulare and Kings Counties lacked the names of the cities, and therefore were not geo-located to the street address provided. Instead, those wells were located at the county seat – Visalia in Tulare County and Hanford in Kings County. Data provided included results in mg/L NO_3-N, NO_3-NO_3, and total N, all of which were converted to mg/L nitrate prior to inclusion into CASTING. Some samples were described as surface water sources; these were ignored for the purpose of this study. Some samples were reported as “DRY,” and these were also ignored. There were 2,224 samples reported both as NO_3-N and as NO_3-NO_3. Most of these samples were related by the correct mass ratio of NO3:N = 4.4268:1, but 663 were not. Of the 663, 20 were more than 25% different and of those, 12 were more than 50% different. In all 663 cases, the average of the two values was used and included as mg/L nitrate.

DWR Bulletin 130

Purpose – DWR Bulletin 130 reports are produced by DWR in cooperation with USGS to describe hydrologic conditions in surface and groundwater in California. These reports are produced for five regions of the state. Volume 3 covers the Central Coast, including our SV study area. Volume 4 covers the San Joaquin Valley.

Scope – This dataset included 859 samples from 256 wells in the SV, and 2,339 samples from 1,717 wells sampled in the TLB. These wells were sampled during 1963 to 1975 and in 1985.

Data – The data consist of the State Well Number, chemistry test results, the sample date, and in some cases the well use type. These data were digitized from PDF reports. These wells were located to the centroid of quarter-quarter sections, based on their SWN, in the TLB. In the SV, the quarter-quarter section map required to locate these wells is only available through Monterey County Water Resources Agency, which was unable to provide that map to our project. Therefore, SV wells in this dataset (and all other datasets with only SWN for location) are located to the centroid of the section identified in the SWN.

Burow et al. 1998

Purpose – Burow et al. (1998) compiled a set of wells data as part of a study on the occurrence of nitrate and pesticides in the eastern San Joaquin Valley.

Scope – This dataset included 53 samples from 30 wells sampled in 1986 or 1987 and 1995. Exact dates of samples were not available.

Data – The data consist of depths to the top and bottom of the perforated interval, the State Well Number, and the well use (all were household domestic wells). These data were digitized from a report and located by their SWN to the centroid of quarter-quarter sections where possible. Some wells were only possible to locate to the centroid of sections.

MCWRA Environmental Monitoring Wells

Purpose – MCWRA monitors 40 deep wells in Monterey County for monitoring of sea water intrusion and groundwater quality.

Scope – This dataset included 407 samples from 39 wells sampled from 1993 to 2009.

Data – These data were provided with GPS coordinates with accuracy assumed to be on the order of 10 meters. Top and bottom of perforated interval, SWN, sample dates, and chemical parameters were included in the dataset.

MCWRA 93/94 and 94/95 Water Year Reports

Purpose – MCWRA (1994, 1995) produced two water year reports including data on groundwater quality.

S*cope* – This dataset included 769 samples from 105 wells sampled in 1994 and 1995.

Data –These data were digitized from printed reports. They were located using their State Well Numbers to identify section centers. MCWRA did not provide actual locations, or reference maps to identify quarter-quarter sections.

MCWRA 2002

Purpose – MCWRA (2002) was a report on the shallow-well sampling program started in 1998. This program was designed to assess nitrate contamination in drinking water wells in SV.

Scope – This dataset included 243 samples from 76 wells sampled in 1999 and 2001. The report from which these data were acquired does not include precise dates.

Data – These data were digitized from a report and located by their SWN to the centroid of PLSS sections. MCWRA did not provide actual locations, or reference maps to identify quarter-quarter sections.

MCWRA has developed a nitrate well monitoring program with private well owners, that is subject to confidentiality agreements, and that has been operating since the early 1990s. Due to confidentiality agreements, MCWRA did not provide these data to our project, with the exception of the 40 environmental monitoring wells described above. Throughout most of California, the State Well Number can be used to locate a well to its quarter-quarter section (¼ mile by ¼ mile), but in the SV, the PLSS has not been digitized except by MCWRA and Monterey County. Neither MCWRA nor Monterey County has been able to provide UC Davis or the State Water Board with that digitized PLSS. Therefore, all wells in the SV study area that are located by SWN are located only to the section (one square mile).

Department of Pesticide Regulations (DPR)

Purpose - The Department Pesticide Regulations (DPR) maintains a voluntary domestic well sampling program to verify the efficacy of its pesticide regulatory program, particularly in areas that are DPR designated vulnerable groundwater zones.

Scope – The program includes 96 domestic wells of which 71 wells are located within the study area in eastern Fresno County and in Tulare County. Wells have been sampled annually during spring since 2001 (data are available through 2010) and also in the fall of 2001 and 2002.

Data – The data shared between DPR and UCD are subject to a confidentiality agreement. Accuracy of reported well locations is reported as the centroid of the section in which the well resides, roughly within one half mile of the actual location. The dataset includes depth to groundwater, well depth, DPR vulnerability zone, sampling date, and nitrate water quality data. Wells fall into two groups: those located in so-called “runoff” zones and those located in so-called “leaching” zones.[8]

County Data

Each of the five counties in our study area has provided additional well nitrate data that were included in CASTING. All of these data were collected for the purpose of maintaining regulatory compliance either with drinking water regulation or well permit regulation. With the exception of Kings County, all of the counties in the study area conduct one-time nitrate tests at the time each new well is drilled.

Tulare County

Tulare County provided us with assessor’s parcel numbers (APN) of the domestic and local-small wells in the county that have been tested for nitrate. These wells were tested once, at the time they were drilled. The APNs for these domestic wells were used to select and locate those parcels from the county parcel GIS layer. Although APNs can change, it is expected that this will cause only a small number of inaccuracies. For geo-location reporting in CASTING, the centroid of each selected parcel was chosen as the approximate location of the well. The maximum spatial error for these wells is estimated as [½ x (Parcel Area)$^{1/2}$].

[8] http://www.cdpr.ca.gov/docs/emon/grndwtr/

Tulare County also provided several dozen test results for PWS wells that were not duplicates of CADWSAP records. In CASTING, these test results were assigned to their respective CADWSAP wells based on their PS-Code (discussed below).

Kings County

Kings County was able to provide us with documentation of well locations and test results for 20 wells, which we integrated into CASTING. Several of these were duplicates of CADWSAP wells, but included non-duplicate test results. Most of the wells provided by Kings County had only a few or a single nitrate test result. The non-duplicate wells were geo-located to the nearest second of latitude/longitude, (approximately 33 meters).

Fresno County

Fresno County provided us an opportunity to manually digitize well log and initial well water quality test results for household domestic wells. Original data are on paper records located in the County Health Department. The digitization effort covered 437 household domestic drinking water wells in our study area, with one nitrate sample each. These wells are located primarily by geo-coded addresses, with those that did not have satisfactory addresses located by the APN. Although APNs can change, it is expected that this will cause only a small number of inaccuracies. The wells are assumed to be in the centroid of the parcel with the corresponding APN. The precision of each APN-located well's location is calculated as one-half the square root of the area of the parcel. Of these wells, 377 had legible perforated interval top and bottom depth construction information, and 329 had legible estimated yield information.

Kern County

Kern County delegates all relevant data management to the Kern County Water Agency. KCWA has provided us with electronic data for over three thousand wells in the county. Most of these wells are associated with a single test for nitrate. While the bulk of these are irrigation wells, there is also a mix of household domestic, small water system supply, and monitoring wells in the dataset. The Kern County wells are geo-located by State Well Number, with a few exceptions having more precise locations based on manual re-location using aerial photography within a GIS.

Monterey County –scans of paper records

Monterey County provided us with scanned paper copies of well-driller reports and nitrate test results for approximately 1,000 local small and domestic wells. These wells have been sampled since the early 1990's although most records are much more recent. From these records, nearly 8,000 nitrate test results from 634 wells were manually digitized by UC Davis. Of the digitized wells from this effort, 24 were located by GPS coordinates, 350 were able to be geo-coded by street address, 17 were located to the center of PLSS sections, 223 were located to the center of cities, 10 were matches of wells already in the database, and 10 wells fell outside the study area. Although State Well Numbers were available for many of these wells, the lack of any available reference dataset of PLSS quarter-quarter section maps from SV meant that wells were not geo-located as accurately as the SWN would allow. The driller logs include the depth to screen and screen length in almost all cases.

Monterey County – electronic spreadsheets

Monterey County also provided a dataset of well locations and a separate set of nitrate tests for state small and local small water systems and domestic wells. The well location set consisted of 431 unique geo-referenced points, and another 498 wells with addresses associated. We were able to geo-locate 389 of these addressed wells, for a total of 820 unique geo-referenced well locations. The nitrate test results are reported by system name or number and not by a specific well identification. Therefore, for systems with multiple wells, tests could not be associated with specific wells or well locations. For this reason, multi-well systems were ignored. The remaining 680 systems have unique locations (a single well) to which nitrate test results can be matched. Numerous inconsistencies were found in the system naming and ID codes, indicating that the locations and test results associated with certain locations are of uncertain reliability. This dataset is known to overlap the digitized dataset described above; however, no common identification code exists between the two datasets. It is unknown at this time how many of these wells overlap with the previous dataset, which was manually digitized from the scans provided by Monterey County.

Table 13. Tests and wells by data-source within the study area boundaries. Some data-sources provided tests that were duplicates of tests already in the CASTING database, resulting in cases such as the NWIS data-source with more wells than tests; duplicate test results were filtered out of the final database. The total number of wells in the study area in the CASTING version of December 17, 2011, is 20,286.

Source Dataset Code in CASTING	Number of Test Results	Number of Wells	Brief Description of Data-source
CADWSAP	60,792	3616	Subset of PICME dataset with high-quality locations and SWAP implemented
DAIRY	11,271	6459	Regional Board 5 dataset of wells on dairies
MOCODig	5,666	452	Digitized by UC Davis staff: records from MCWRA
EPA	4,871	2883	Legacy STORET data
KeCo	3,412	3007	Supplied by Kern County Water Agency
EnvMon	2,598	537	SWRCB Environmental Monitoring wells
DWRL	2,015	1978	Digitized by UC Davis staff from DWR Bulletin 130 reports
NWIS	1,684	1735	USGS database of tested wells
MoCo	1,572	462	Supplied by Monterey County Environmental Health
MCEM	1,018	229	Digitized by UC Davis staff from water quality reports in Monterey County
DPR	814	71	DPR domestic wells survey
TCEHS	444	444	Domestic wells data supplied by Tulare County Environmental Health
FRCO	368	369	Domestic wells data supplied by Fresno County Environmental Health
GAMA	141	141	SWRCB GAMA Domestic Wells Study
USGS	83	83	USGS/SWRCB GAMA Priority Basin study
KiCo	82	20	Supplied by Kings County Environmental Health
CIP	63	63	Canal Integration Project (Westlands Water District and DWR)
DWR	44	39	Data supplied by DWR from tests in western San Joaquin Valley in 2008
KRB98	36	19	Digitized from Burow, 1998
TuCo	17	579	Supplied by Tulare County GIS staff
DIP	14	14	Distribution Integration Project (Westlands Water District and DWR)

The 21 datasources incorporated into the CASTING database (Table 13) comprise the most comprehensive dataset of well nitrate test results in existence for the TLB and SV. CASTING is intended to be a prototype of a state-wide database for nitrate in groundwater. The varying levels of confidentiality of the datasets incorporated into CASTING require that the database be managed carefully for privacy concerns. All portions of the database that may legally be made public will be as soon as technically feasible.

4.1.4 Well Identification Codes

There are several different identification schemes used throughout the state (Table 14). The State Well Number (SWN) is theoretically applied to every drilled well and is filed with DWR in the well-driller log for each well. The SWN is based on the Public Land Survey System (PLSS), which defines the meridian, township, range, section, and quarter-quarter section that contains each well, and includes a serial number to distinguish each well from other wells in the same quarter-quarter section. As such, it is possible to (approximately) locate a well by its SWN to within an eighth of a mile distance. For many private well owners, such accuracy is cause for privacy concerns. Lack of understanding of the cadastral survey has further led to some problems with the SWN. For example, California has three meridians upon which the township, range, and section numbers can be based (Humboldt, Mount Diablo, and San Bernardino meridians). When this meridian is not included in the SWN, it can be difficult to identify in which region (of the 3) the well belongs, particularly in the TLB. The SWN is the only ID code that is meant to be universal within California. Unfortunately, many agencies do not record the SWN of wells that are sampled in their projects.

The database of drill logs that DWR maintains is the only database that is intended to be comprehensive of all wells in the state, but this database is only for construction information such as depth, screen interval, estimated yield, and drilling date. The database is currently maintained in the form of paper files and scanned images of such files. It is not organized in a manner that allows for convenient access to large numbers of well data files. Although some well records in the database have a SWN recorded on the paper file, the electronic files are identified by a serial number that has no direct relationship to the SWN or any other identification code used by any agency. DWR does not maintain a relational database of this serial number as it relates to the SWN.

The only other commonly used, statewide-applicable code is the so-called State ID (SID). This code consists of the 13 digit latitude/longitude in degrees, minutes, and seconds for the well. This code is far less common than the SWN, and has higher precision, on the order of 30 m (100 ft), (though also more likelihood of error) than the SWN. It also produces confidentiality concerns. Other identification codes are used by specific agencies, projects, or regional organizations, and are generally not cross-referenced to each other, the SWN, or the SID.

Table 14. Well identification codes and agency users.

Code Name	Abbreviation Alternate Name	Example	Agencies/Dataset that use it
PS-Code	PICME Pkey	5400544002	CDPH, Counties
System Number	PWSID	5400544	CDPH, Counties
State Well Number	SWN	011S021E18F001M	SWRCB, DWR, USGS, EPA
GAMA ID		TULE-O7	SWRCB, USGS
State ID	SID	365803119432101	SWRCB, DWR, USGS, EPA
Electronic Data Format Name	EDF	W-5C	SWRCB, DWR, USGS

CDPH and many of the counties use a code called a Primary Source Code, or PS-Code. This code consists of a 7-digit serial number assigned to each water system that provides drinking water to at least two households, followed by a 3-digit serial number that designates the particular source (well, spring, stream, tap, tank, or other point at which water is tested).

The GAMA Domestic Well Project uses well IDs based on the county in which the well is located, followed by a serial number. These codes are unique, but generally not related to the SWN or other identification codes. As a result, although they are convenient for distinguishing between GAMA domestic wells, it is difficult if not impossible to compare distinguish these wells from other wells that have been sampled in the area. The GAMA Priority Basins project uses codes that appear to have been assigned irregularly, with some codes consisting of the county name followed by a serial number, and some others based on groundwater basins or other geographic features. These codes are based on the study area defined in the Priority Basins Studies. Many of these wells are also identified by their SID, thus allowing some cross-referencing with other well sampling projects.

Environmental monitoring wells with information maintained by the State Water Board are assigned names within each separate project by the sampling contractor, without consideration of wells at other monitoring sites that have already been assigned names. Some of these wells have the SWN as the name, but most use a code such as “MW-1” or similar. There are many examples of duplicated well names within that database. To avoid confusion, a project identification code is assigned by the State Board to distinguish between projects. Within a single project, well names are not duplicated. The well name and project ID together provide a unique identification to differentiate between wells with identical well names. A latitude-longitude based identification of reasonably high accuracy is also available for these wells. However, this information is not used as a site ID, because of the possibility of confusion with other, less-accurately defined, coordinate-based ID schema.

4.1.5 Potential Future Improvements to a Nitrate Database

The database represents what has been possible to collect over the 18 months of this project. There are potential future improvements to this database that were outside the scope of this project:

- Tulare County has additional domestic well nitrate data in digital format that we were unable to obtain. It is unclear how many records are not covered in CASTING.
- GPS locations of Tulare County domestic wells that are currently not in CASTING may be available from Tulare County.
- The dairy regulatory program currently does not maintain an electronic database of groundwater nitrate data obtained annually from 1,800 producers. Data are currently entered from paper records into a spreadsheet. However, important information is not collected consistently:
    - Well identification codes may change from year to year.
    - Well locations and well construction information (screen depth) are not known. This is a task that could be done by a single entity for all program participants at once, by working with DWR well log archives.
    - Records are not kept in a suitable database format; significant data manipulations and simulation are needed to bring data records into a format amenable for database archiving and (statistical) analysis.

    - We recommend that the CVRWB work with DWR and State Water Board to set up an internal work process to enter data provided by dairy producers into a high quality database and to follow up with producers on data gaps and data inconsistencies. In principle, this system could also be setup whereby certified consultants enter data collected under the dairy regulatory program directly into an electronic database, including groundwater data.
    - At a minimum, groundwater nitrate data should be associated with:
        - Well identification number
        - Well location
        - Well construction information
        - Sampling date.
- MCWRA has a confidential, private network of wells monitored for nitrate. Data are not included in CASTING, unless they have been published (typically data for which the collection was funded by public grants).
- Fresno County and presumably other counties have a significant number of paper records with water quality data from domestic and other wells that we have not been able to digitize as part of this project.
- Many wells in the CASTING dataset have no information or incomplete information on both well use and well construction. A major effort is required to match these wells with DWR well log information. While this information may not be unambiguously available for all wells, a substantially larger number of wells would have well depth and more precise location information than currently in CASTING.

4.2 Methods

4.2.1 CASTING Database Structure

The CASTING database of wells and test results consists of two large tables. The first table consists of a single record (row) for each well, with attributes (columns) for location information, any identifying codes of names, and any available physical characteristics of the well. This database table is called NO3GW and consists of geo-located datapoints (wells). Each well in this table is assigned an internal unique new ID called "NO3_ID."

The second database table is a tabular set of test results and is designated NO3GT. Each record refers to a single nitrate sample test. The record consists of attributes identifying names and codes, the sample date, the sample result, the agency from which the data were acquired, and the NO3_ID for the well.

The NO3GT dataset can be queried or related to the NO3GW dataset via the attribute NO3_ID. Many organizations maintain databases of NO3 data and of well information, often separately. In many cases, different agencies record separate samples for the same wells. To reduce duplication of data within CASTING that would cause incorrect results from statistical and spatial analyses, new data from outside sources must be compared with wells already in CASTING by the various agency ID codes, the well name, well location, and well test date and results.

4.2.2 Developing the NO3GW and NO3GT Datasets for CASTING

The CADWSAP database was used as the starting point for the development of the CASTING wells database. The dataset is much larger than most of the other well databases used; it is also the most reliable source of water quality testing data and it has the broadest set of recorded parameters. Each well in CADWSAP has a unique ID (PS-Code) from PICME. Any one well may have between one and several hundred test results. It is therefore not practical to store the test results in the same dataset as the well information. PICME, the parent dataset for CADWSAP, uses a database called WQM for storing the water quality information. Each test result has the PS-Code for its associated well, and this code is used to query the WQM database for results associated with wells of interest.

We extracted all raw or untreated water sample data from CADWSAP wells located within our study area. CADWSAP also includes records for all treated or blended sources of drinking water, and for non-well (surface water) sources of drinking water, but we did not include these sources in CASTING. Each well added from the CADWSAP database was then assigned a unique code that is called the "NO3ID" code. This code is simply a serial number preceded by the prefix "NO3_" and is only used for internal record-keeping. NO3ID has no spatial reference incorporated, nor any other identifying characteristics that could trigger confidentiality concerns. The resulting wells table (spreadsheet) is called NO3GW (for nitrate in groundwater - wells) within CASTING.

We used the PS-Codes of these wells to extract the WQM test results for all those wells. The NO3IDs we assigned to each well in the study area were then attached to the test results. The resulting test results database (a spreadsheet or electronic table) is called NO3GT (for nitrate in groundwater - tests) and is the second of two databases in CASTING.

Subsequently, each new well or set of wells obtained from an outside data source was checked against the existing database to find matches, either by codes or well owner name. As new wells were acquired, the ID codes and names associated with those wells were added into the database in order to maintain a record of the source. Additionally, a separate attribute field (column) is created for each data source in both tables (NO3GW and NO3GT), named generally by the data source. All wells from that data source were assigned a single-digit integer value in that attribute. The integer value is associated with the method of geo-location and is described in the metadata of the CASTING database. For example, all wells in the EPA STORET dataset are located by their SWN, so they all receive a "1" in the "EPA" attribute. Similarly, the wells in the Tulare County Domestic Wells dataset provided by Tulare County Environmental Health are located through one of 3 different methods (by Parcel Number, by address, or by GPS coordinates), so each well gets either a"1", "2", or "3" in the "TCEHS" attribute.

Each well might be present in several datasets, but each test record only comes from one dataset. Therefore, in NO3GW it is possible to find many records with values in multiple dataset columns, but in NO3GT each record has only one of these columns filled in. For example, a well that was added from CADWSAP would receive a non-zero value in the CADWSAP column of NO3GW, if it were then found to also exist in the STORET database, it would receive a non-zero value in the STORET column as well, but the new test result would be on a new row in NO3GT. Thus, overlap between data sources can be tracked (in NO3GW), while the source of each test result is retained (in NO3GT).

The process of integrating new data into the NO3GW database is shown schematically in Figure 22. This process adds the well and its associated parameters such as ID codes, data source, depths, screen intervals, or other physical or ownership characteristics to the spatial dataset (NO3GW) of wells in the study area. Once a well was identified and assigned a NO3ID, the test results associated with that well are attached to the NO3GT database with the associated NO3ID.

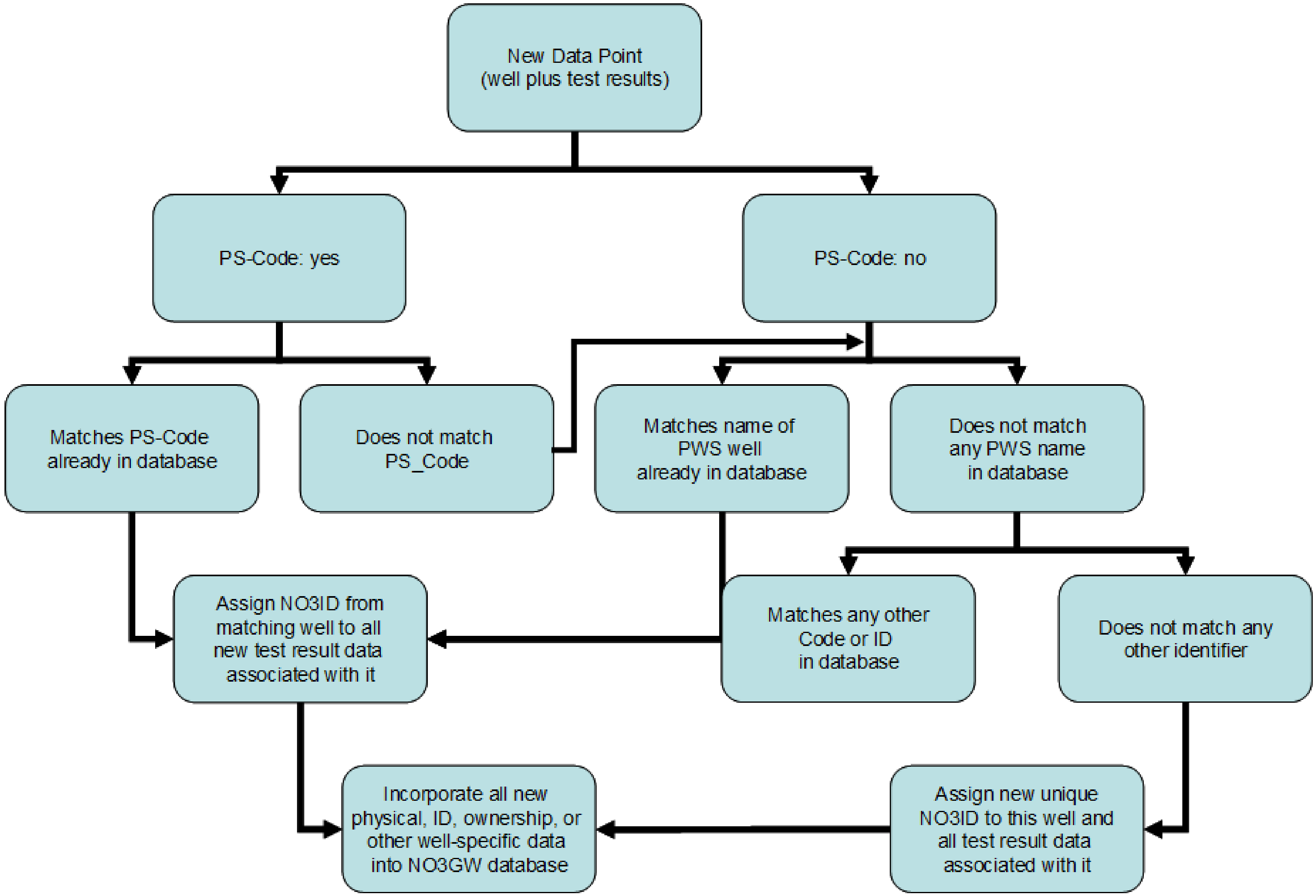

Figure 22. Process diagram for adding new wells to CASTING database.

4.2.3 Adding Data to CASTING

The input databases were developed by different agencies at different times, and thus very few of these databases were in similar formats. Each database was examined for formatting, unit, and precision prior to assimilation into CASTING. The locations of wells were provided in some cases, and these locations were assessed for horizontal precision. For datasets without location data, locations were generally acquired through address geocoding, or based on the SWN.

4.2.3.1 Non-Detect Data

For nitrate test results, all values were converted to nitrate as NO_3. This value is contained in the "FINDING" attribute (column) of the NO3GT database of CASTING. Tests for which "non-detect" (ND), "0" or "less than x" (where x is a detection limit) was reported were assigned the detection limit in the "FINDING" attribute and "<" in the "x_mod" attribute of the NO3GT database. Statistical analyses using these assigned values generally follow the method described in Helsel (2005) of using ½ the detection limit in cases where a non-detect is reported. In cases where the detection limit was not reported by the data provider, it was assumed to be 2 mg/L as NO_3.

4.2.3.2 Precision of Well Locations

In the NO3GW database of CASTING, the attribute "x_y_prcsn_m" gives the likely maximum error in the Northing and Easting location of the well, in meters (m). This value was developed based on the method of location used for each well. When the location of the well was provided by the input database, but the method of location was unclear, this attribute is generally left blank.

For wells located by Assessor's Parcel Number (APN), the SWN, (either to the quarter-quarter (QQ) section or the full section), the value of the precision attribute is equal to ½ the square root of the area of that parcel, QQ section, or section, in meters. When wells were located by their SWN in this manner, it is unavoidable that all wells identified in the same QQ section (or section) will appear in a single location at the centroid of the QQ section (or section). The precision of well locations derived from APNs is variable depending on the size of the parcel. Also, the APN can change when properties change ownership or are subdivided or split, and this will introduce unknown positional error. The precisions of wells located to the QQ section and section are approximately 200 m and 800 m, respectively.

In the SV, no section survey has been conducted, but a virtual PLSS dataset has been built by the Bureau of Land Management (BLM) to the section level. It is our understanding that a similar, though likely not identical, virtual PLSS has been developed by the Monterey County Water Resource Agency to the quarter-quarter section level. This product of the MCWRA was used to assign SWNs to wells that agency tested, and these SWNs were available in public documents recording the sampling data; however, the virtual quarter-quarter section dataset was not provided. Therefore, wells located by SWN in the SV are only located to the section centroid, based on BLM's virtual PLSS, and may have ½ mile or more of Northing and/or Easting error. Wells in several datasets are located by GPS coordinates and these are assumed to be within 20 m of the correct location in most cases. Some wells are located through geocoded addresses, and the precision of these wells is assumed to be 50 m.

4.2.4 Attributing Wells with Depth Information

Some data sources provide well depth, some provide screen depth and/or screen interval length, and some provide no or inconsistent depth data. For the purpose of this study, screen top and bottom depths are acquired wherever possible.

Where well screen top and bottom depths are provided by the originator, we have incorporated those values into the database. When two data sources provide conflicting information about depth for a particular well, priority is given based on the hierarchy shown in Table 15.

Table 15. Priority of screen top data sources.

Priority	Top of Screened Interval Data Source
1	Driller Log
2	USGS data
3	CADWSAP or other agency-sourced data
4	Seal depth from agency source, taken as top of screened interval
5	CADWSAP depth to static water, taken as top of screened interval
6	Regression vs well depth from agency source
7	Regression vs owner reported well depth
8	Unassigned
9	No depth information provided

Where 2 or more agencies reported different depths, we checked for a driller log source and used that if possible. If no driller log source was available or identifiable, we looked for other data points that might invalidate any of the agency-reported depths. If we could not invalidate any data, we choose the deeper value (this only occurred in a very few cases). All values were retained in the NO3GW dataset in case later information needs arise that require us to retrieve these values.

In many cases, the information obtained included well depth, but not the depth to the top or bottom of the perforated interval. A regression was built to model these depths based on well depth using the Priority Basin Project wells. These data include more reliable and comprehensive construction information for the wells than that contained in the CDPH database, and are the only large dataset for which we have the depth to the top and bottom of perforated interval as well as the total well depth.

The Priority Basin Project wells were identified by either a GAMA ID or SID (Table 14). The State Water Board provided a cross-walk between these codes, the SWN and the CDPH PS-Code. We used the GAMA ID, STATEID, and SWN to join these wells with data extracted from the Data Summary Reports (DSR) published on the State Water Board GAMA website for the USGS Priority Basin Project. Based on the information thus joined, we derived an empirical relationship between completed well depth and depth to top of perforated interval (Figure 23). The depth to the bottom of the perforated interval is nearly always the same as the depth of the completed well. The relationship between depth to top of perforations and well depth is nearly linear, but becomes less certain with increasing depth.

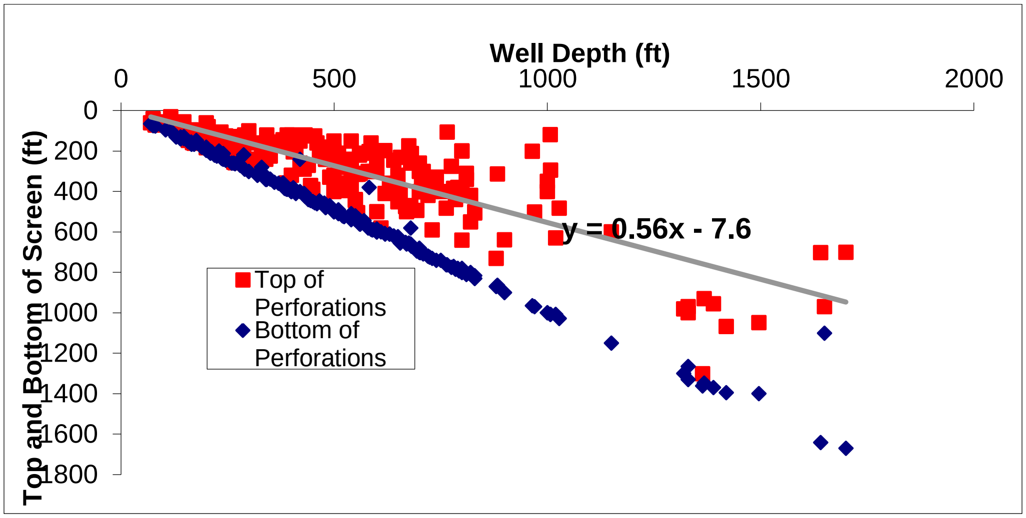

Figure 23. Regression of top of perforated interval from well depth. Based on the USGS GAMA Priority Basins Project dataset.

The linear regression of depth to top of perforation from well depth is:

$$\text{Top of Perforations} = [\text{well depth } (0.56) - 7.6] \qquad \text{(Eqn. 1)}$$

This is based on 185 wells that had top and bottom of perforation depths as well as well depth. These data were contained in the DSR datasets produced by the USGS Priority Basins project.

Of the 4,686 wells with Depth to Top of Perforation in the current version of CASTING, 2,144 are derived from Eqn. 1. The majority of these were wells with only well depth provided, based on Agency reporting.

Each method, or source of information about the depths to top and bottom of perforated interval, was retained in the CASTING database. This allows users to select data for analysis based on the level of confidence in the depth data.

4.2.5 Mapping of Nitrate in Groundwater

ESRI ArcGIS™ 10 was used to develop the CASTING database and to create maps showing the distribution of nitrate in wells in the study area, at different time periods, regional scales, and depths. Nitrate concentration is mapped in relation to rivers, the trough of the TLB, population, and overlying land use characteristics. We used this GIS to produce tables relating the concentration of nitrate to these other variables for statistical analysis. The California Teale Albers NAD83 projection was used for all datasets.

Gradients developed as described above were joined back to the spatially referenced wells and mapped in ArcGIS^{TM}. Nitrate concentrations and gradients of nitrate concentrations were compared across depths and regions. Wells were distinguished by several attributes: depth to the top of the screen, groundwater subbasin, nearness to major natural streams, and well-type.

4.3 Spatial Distribution of Wells Data in CASTING

4.3.1 Spatial Distribution of All Wells in the Database

In the December 17, 2011 version, CASTING consists of 25,214 wells, with 2,109 in the SV area and 18,177 wells in the TLB area. The remaining 4,928 wells are outside the study area, but are retained to provide boundary conditions for modeling purposes such as interpolations of groundwater nitrate and well density.

Well locations are unevenly distributed across the study area (Figure 24). The wells are of relatively low density in the southwestern region of the TLB and in the upper SV. The Tulare Lake bed region in Kings County is effectively devoid of any records of well nitrate, which coincides with a privately-owned, surface water irrigated farming area (Boswell Farms) that has almost no private residences. The highest density of wells in the database is found in the metropolitan areas of Fresno, Hanford, Visalia, Porterville, Delano, and Bakersfield. A high density of wells with nitrate data also exists in areas west and southeast of Bakersfield in the valley portion of Kern County (Figure 24).

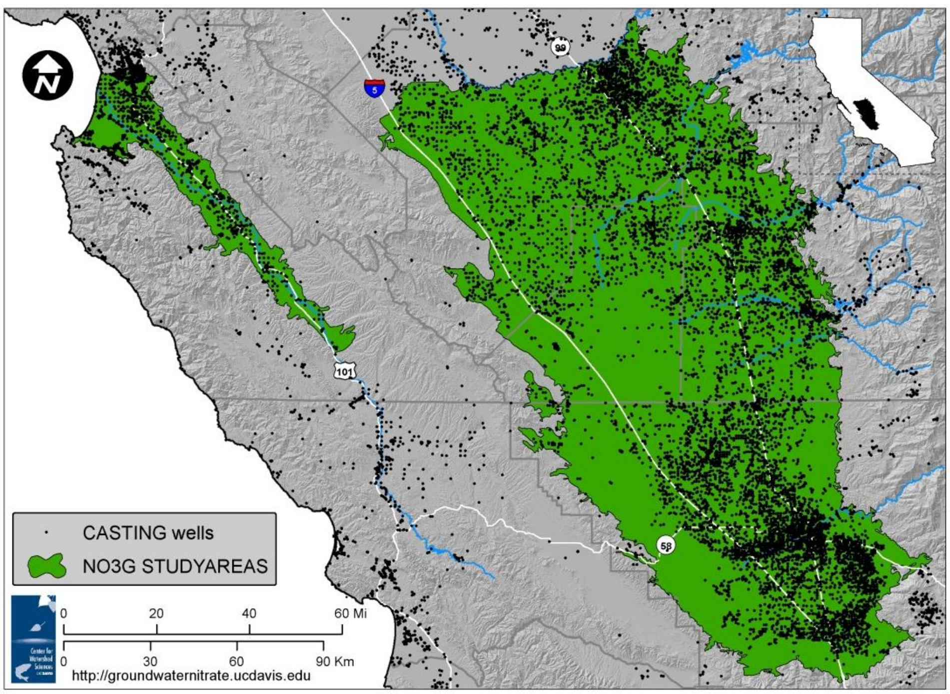

Figure 24. All wells in the database as of December 17, 2011.

4.3.2 Spatial Coverage by Data Source

The CADWSAP database (5,429 wells) had the widest coverage of the various source databases. Even so, CADWSAP lacked good coverage in some regions of the study area (Figure 25). The U.S. EPA STORET database (3,077 wells) is well-distributed in the TLB, but lacks any coverage in the SV portion of the study area (Figure 26). All other datasets were restricted to either very low density coverage (the State Water Board Environmental Monitoring wells, USGS Priority Basins wells – Figure 27) or were very regional (county datasets – Figure 28; DPR, DWR, GAMA Domestic – Figure 29). The Central Valley Regional Board provided 6,459 wells in the study area, most in the eastern portion of the TLB (Figure 30). The following maps show these distributions.

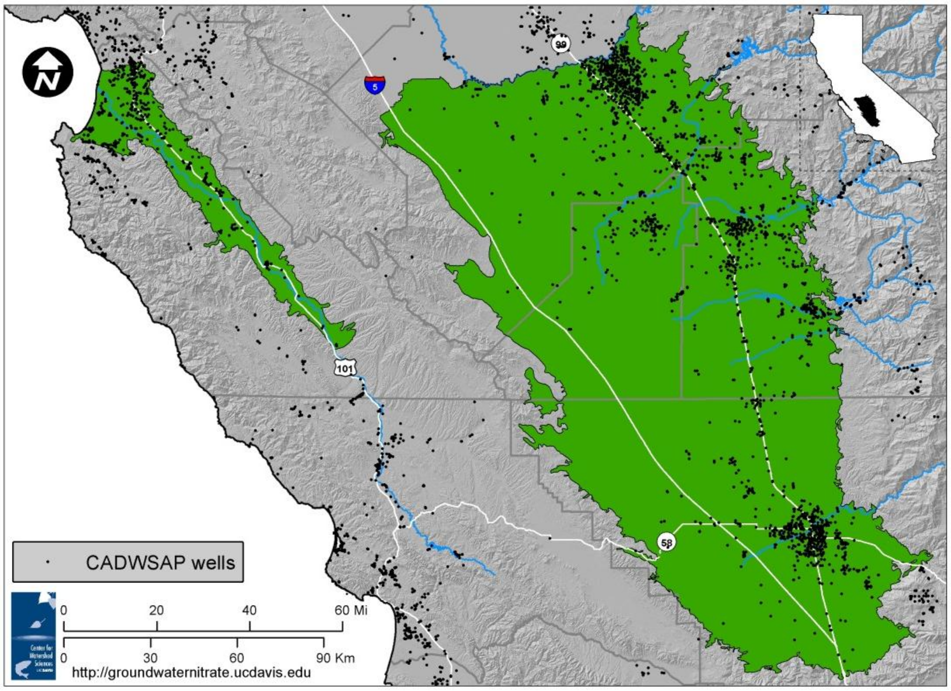

Figure 25. CADWSAP wells.

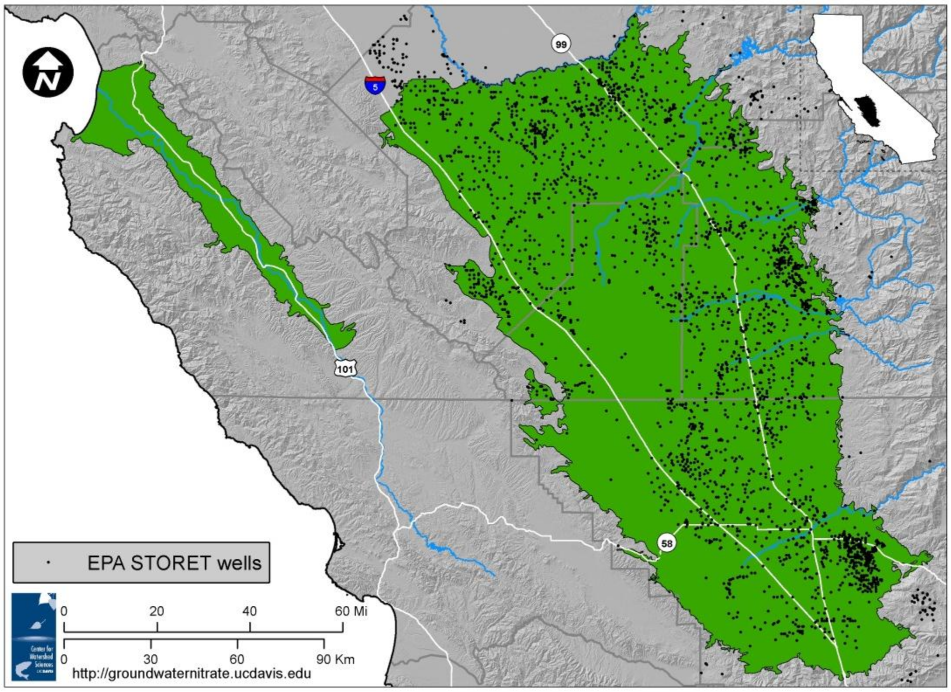

Figure 26. U.S. EPA STORET wells.

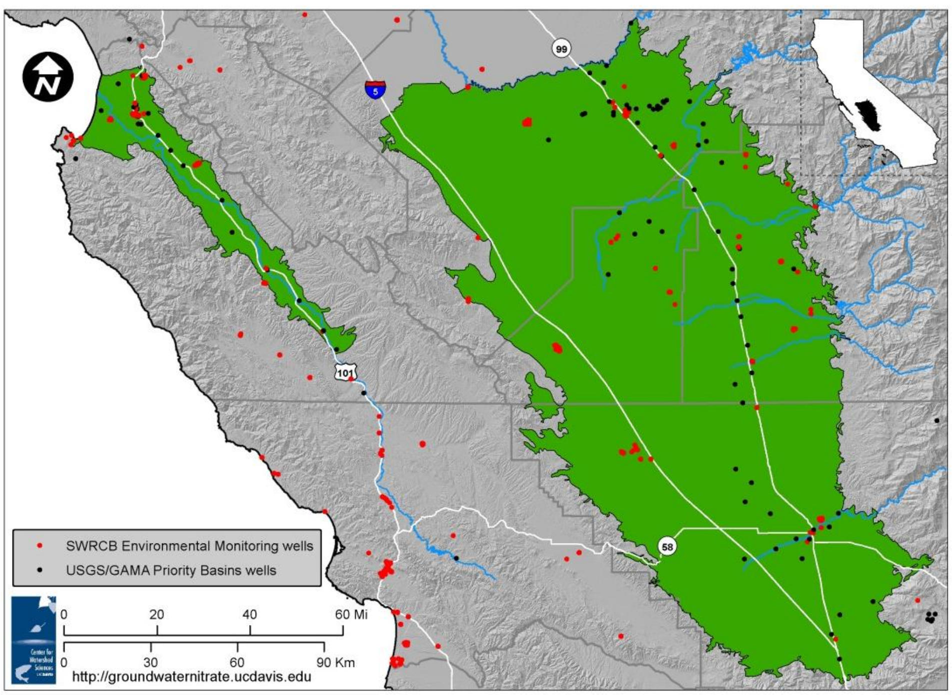

Figure 27. The State Water Board Environmental Monitoring wells and USGS Priority Basins wells.

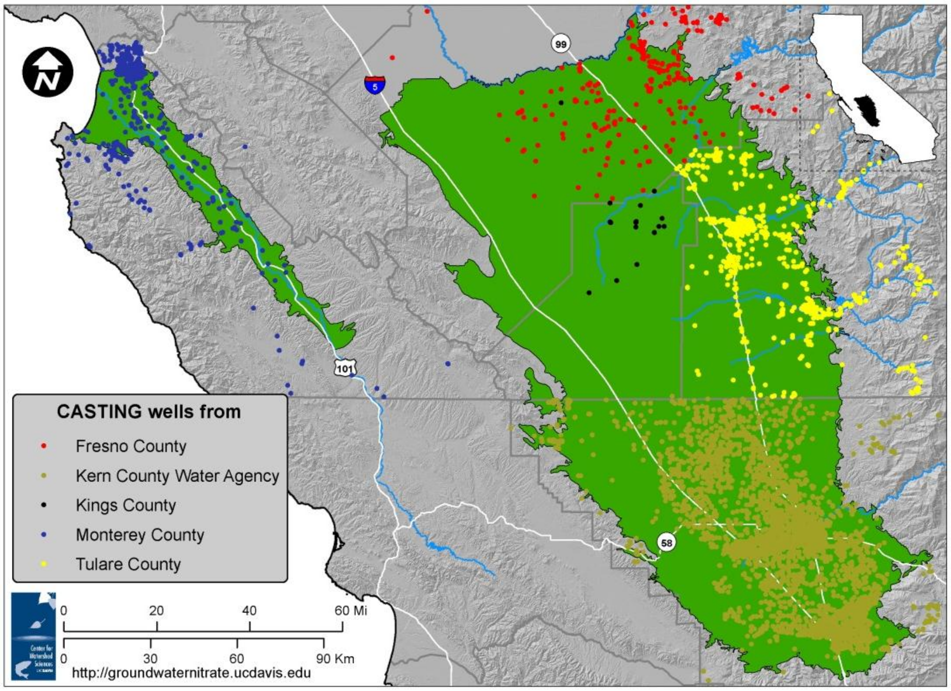

Figure 28. County-sourced wells.

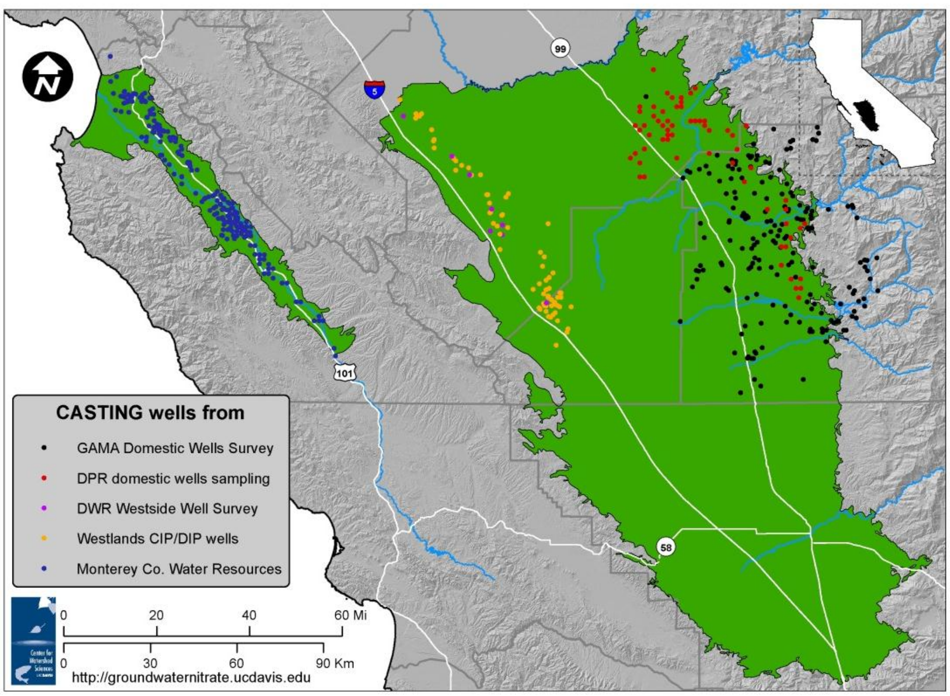

Figure 29. Regional Projects.

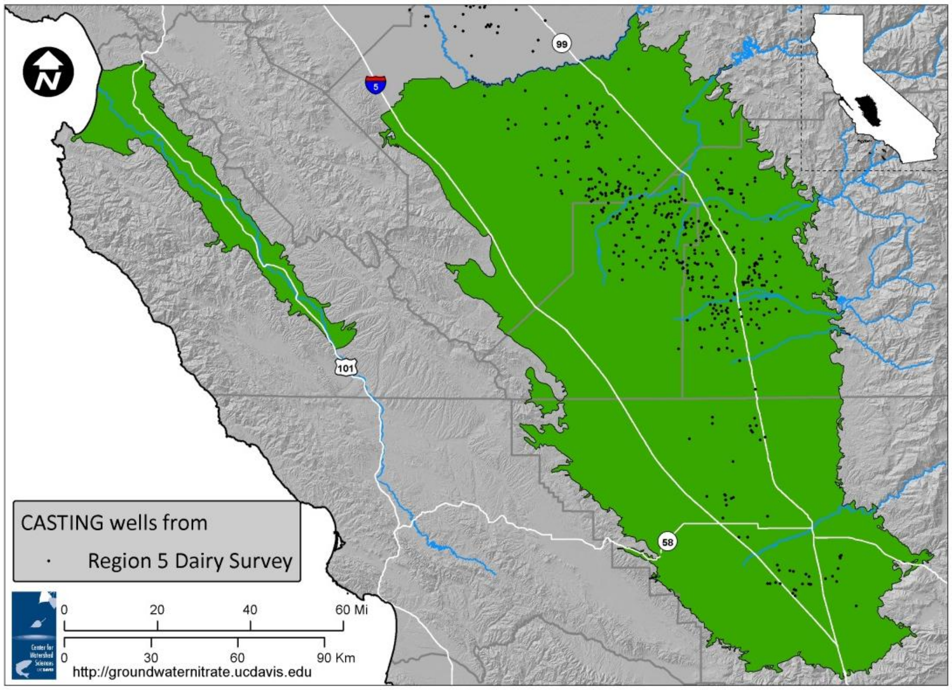

Figure 30. Dairy wells, data from the Central Valley Regional Water Board.

The wells in the CASTING database are distributed similarly to the population of the study area (Figure 31). Wells for which data has been collected tend to be clustered in areas with higher population density and along major roads.

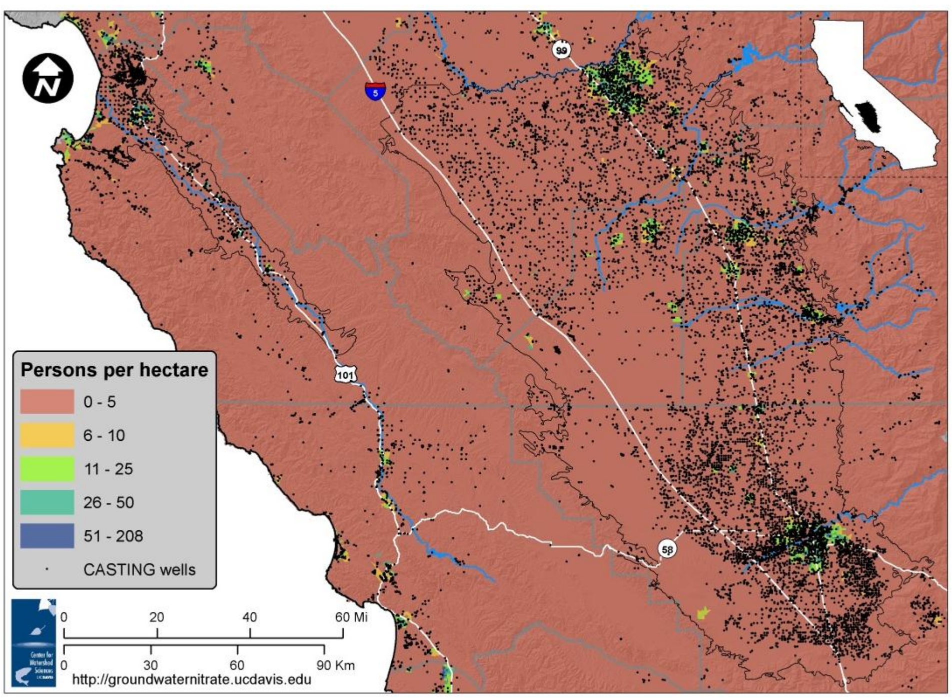

Figure 31. Population density and CASTING well distribution.

4.3.3 Depth of Wells

We were able to obtain or estimate depth to top of perforations for 4,686 of the wells in the database within the study area. Depths to the top of the perforated interval were found to average approximately 79 meters (259 feet) in our dataset, with the shallowest well at 1.8 meters (6 feet) depth, and the deepest well at 741 meters (2,431) feet below ground surface. The standard deviation from the mean of well depth to top of perforation was 72.2 meters (237 feet) for the entire set of wells with depths. The spatial distribution of the wells with depth information was roughly similar to that of the overall database (Figure 32), but with lower density throughout the study area. Depth classes were defined as "Shallow" (0 – 90 m or 0-300 ft below ground surface), "Medium" (90 – 180 m or 300-600 ft), and "Deep" (> 180 m or > 600 ft).

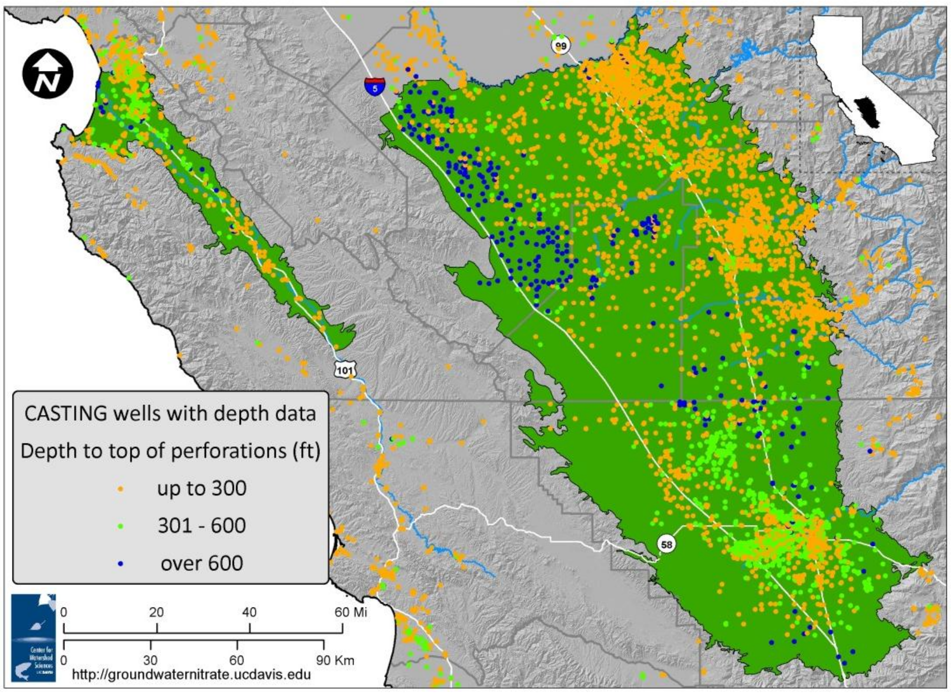

Figure 32. Wells with depth to top of perforation.

For all wells with depths, the distribution of depths is heavily skewed to the higher end (Figure 33). For domestic household wells only (data from Fresno County, GAMA Domestic Wells Survey, CVRWB Dairy Program, and DPR Domestic Wells Survey, Figure 34) this skew is less pronounced. This set of wells consists of wells that are known to be household wells; the mean, max, and standard deviation of domestic well depths to top of perforated interval were 258 ft, 1832 ft, and 97 ft, respectively.

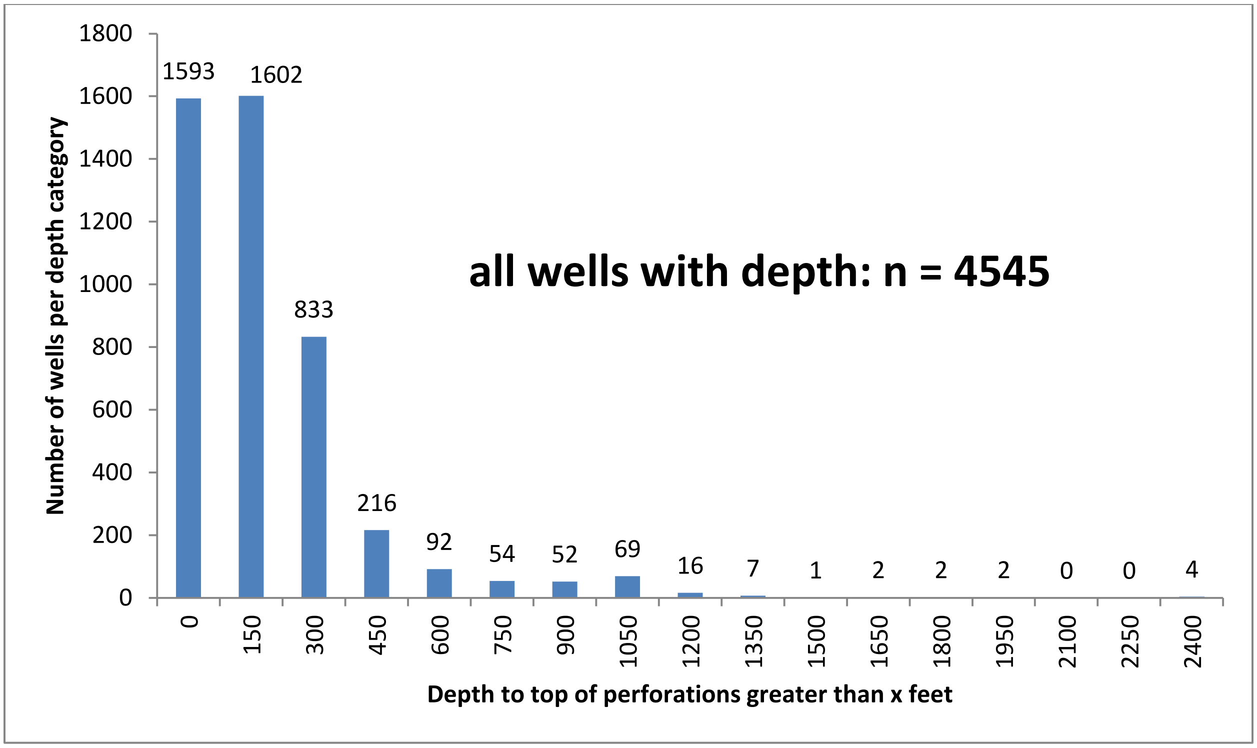

Figure 33. Distribution of depth to top of perforated interval for all wells with available data.

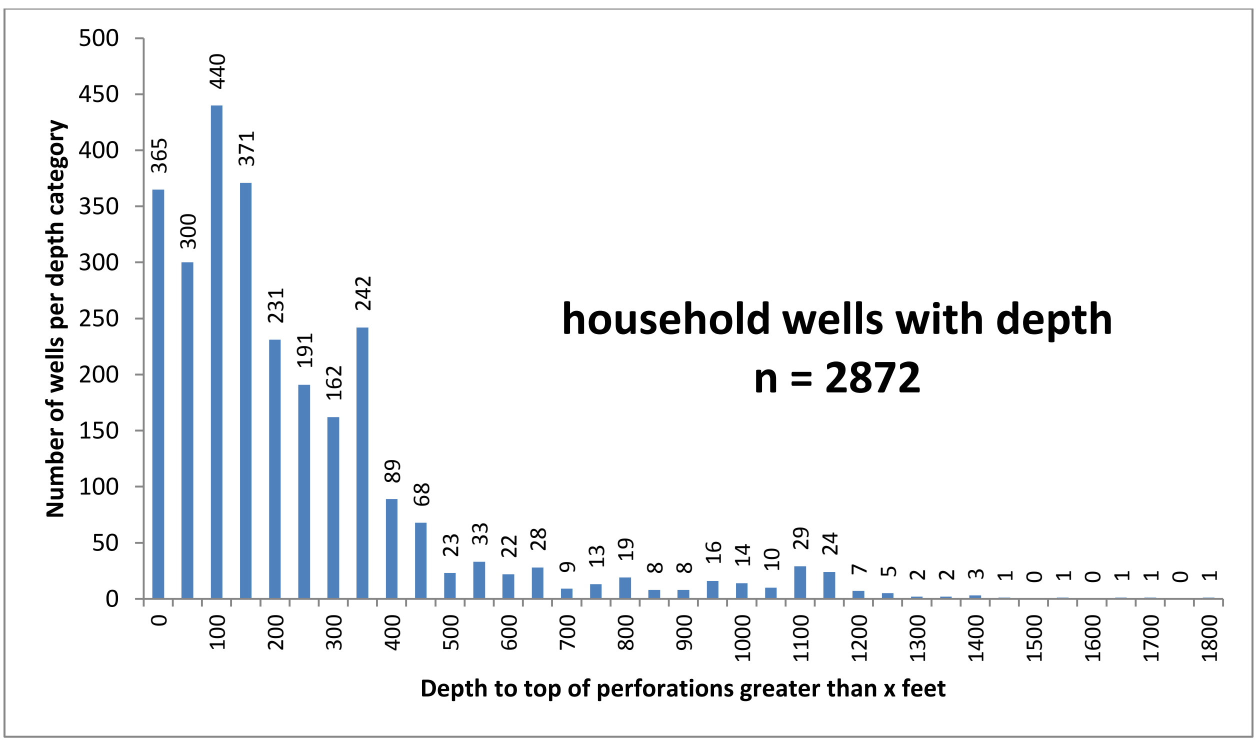

Figure 34. Distribution of depth to top of perforated interval for household domestic drinking water wells with available data.

Wells known to be non-household wells averaged 79.6 meters (261 feet) to top of perforations, with a standard deviation of 191 feet. Most of the household wells in CASTING are located in Tulare, Fresno and Monterey Counties (Figure 35), whereas other types of drinking water wells are more evenly distributed throughout the study area (Figure 25). This spatial bias may reduce the value of this direct comparison between the household and non-household wells.

Between groundwater subbasins, distinct differences in depth distribution arise, making a study-wide comparison of domestic well depths with public supply well depths difficult. Therefore, DWR Bulletin 118 groundwater subbasins were used to group wells for a spatially distributed comparison between household wells and water system wells. The number of wells of each type (household and water system), and their average and standard deviation of depth to top of perforations, paired by subbasin, are shown in Figure 36.

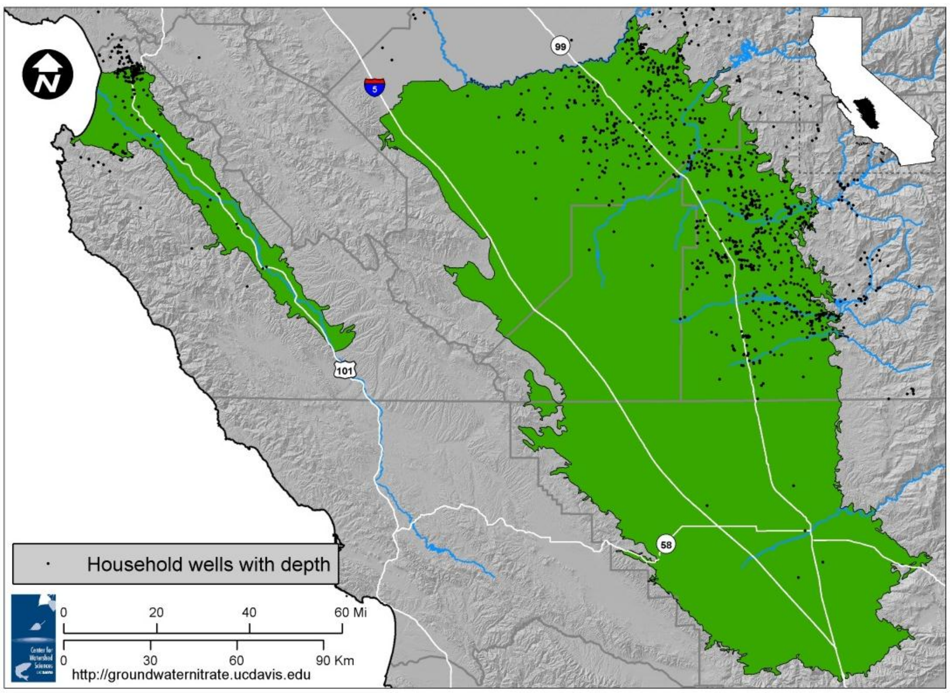

Figure 35. Household domestic wells in CASTING with depth data available.

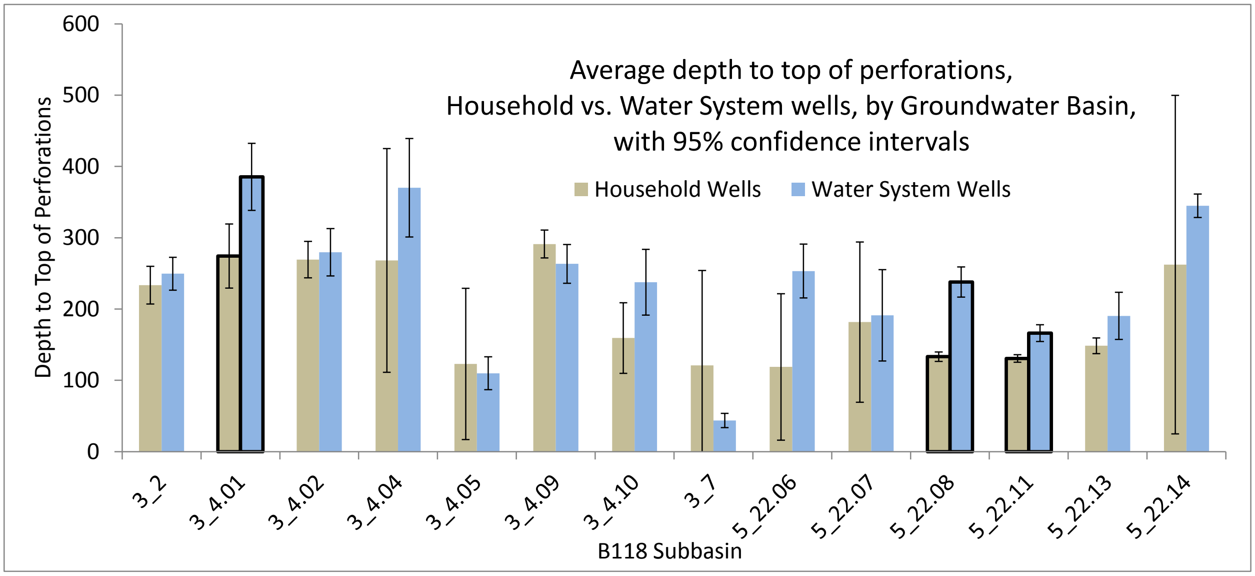

Figure 36. Comparison of mean depth to top of perforations between household wells and water system wells, by groundwater subbasin. Bold outlines indicate significantly (α=0.05) different values.

The three subbasins that produced significantly different values of mean depth to top of perforations all had deeper drinking water system wells than their household wells. However, several subbasins show household wells of essentially the same depth as drinking water system wells. This implies that in certain areas, the assumption that community water system wells are deeper than household wells does not hold.

4.4 Test Origins and Temporal Distribution

We have acquired records for 133,329 test results associated with the wells in and near the study area. Of these tests, 97,005 are on wells in the study area, with 16,663 in the SV and 80,342 in the TLB. We did not differentiate between tests taken by different projects – all tests are treated as equal value. Any test that was performed on any particular well is associated with that well, regardless of the source of the test data. The bulk of the tests in the test database are from the CDPH WQM database. Where test results from other sources were available for CADWSAP wells, these tests are simply included in the tally of tests for those wells. The database of well nitrate samples consists of test results spanning the years from 1900 to 2010 (Figure 37).

Well information is distributed unevenly in time (Table 16). Regular monitoring began in the 1950s with a few hundred records available per year in the study area through the 1980s (a total of approximately 13,000 nitrate test results prior to 1990). During the 1990s, nitrate sampling records collected each year rapidly increased (approximately 15,000 nitrate data are included in CASTING for the 1990s). Over 70 percent of the data in CASTING – approximately 68,000 nitrate records - were collected since the year 2000. The data from 2010 is incomplete.

Table 16. Number of nitrate tests in CASTING, within the study area, by county (study area) and time period. Total for both study area is 97,005 test results.

	Fresno	Kern	Kings	Tulare		(TLB)		Monterey (SV)
1900-1949	30	38	4	44		116		4
1950's	797	2,645	142	521		4,105		0
1960's	872	1,462	205	655		3,194		245
1970's	712	1,840	201	826		3,579		194
1980's	1,059	937	202	671		2,869		933
1990's	3,500	4,045	180	3,080		10,805		4,883
2000's	27,616	9,180	3,593	16,080		56,469		9,563
2010's	1,295	340	50	547		2,232		887
all years	35,881	20,487	4,577	22,424		83,369		16,709

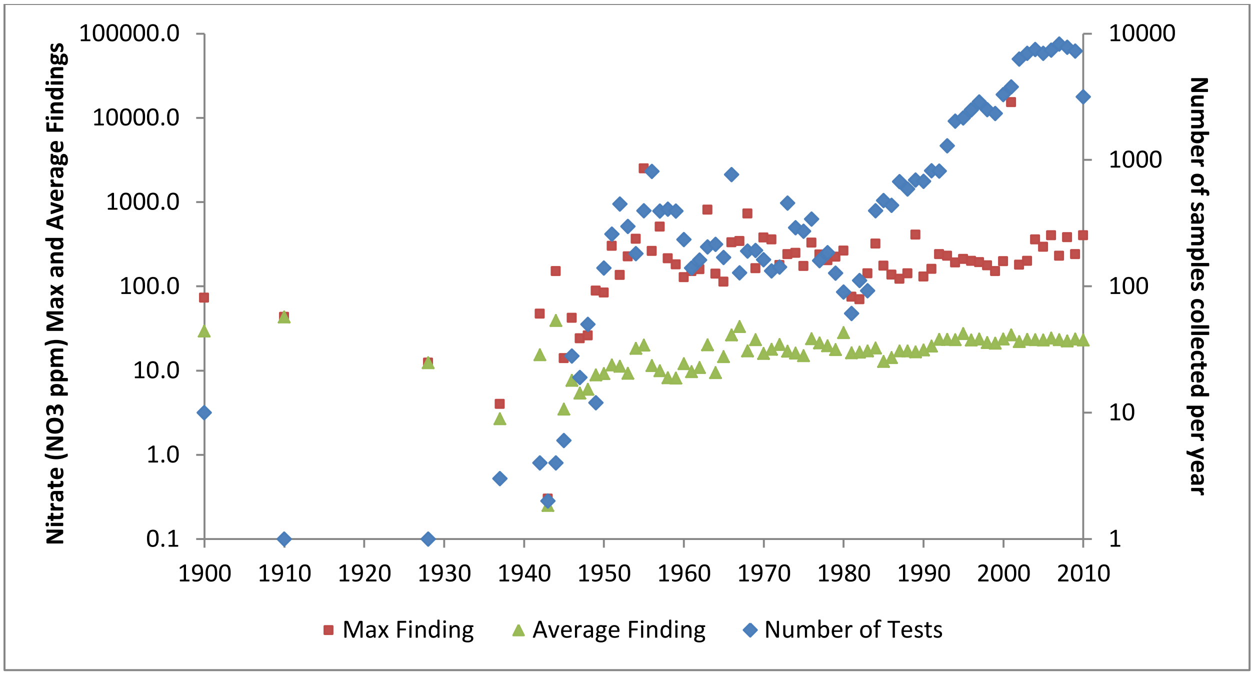

Figure 37. Number of samples collected per year, with maximum and average nitrate (as nitrate) mg/L (each year), in CASTING database.

5 Analysis of the California Ambient Spatio-Temporal Information on Nitrate in Groundwater (CASTING) Database

Prepared by:

Thomas Harter, Aaron King, Dylan Boyle, Giorgos Kourakos

5.1 Introduction

In this section, maps and statistical analyses of the spatio-temporal distribution of nitrate concentration in wells in the study area are presented. Unless mentioned otherwise, all nitrate concentrations [mg/L] are expressed for nitrate as nitrate.

Nitrate concentrations in wells vary widely with location and well depth. More domestic wells and unregulated small system wells have high nitrate concentrations due to their shallow depth (Table 20). Highest nitrate concentrations are found in wells of the alluvial fans in the eastern TLB and in wells of unconfined to semi-confined aquifers in the northern, eastern, and central SV (Figure 41). In the Kings, Kaweah, and Tule River groundwater subbasins of Fresno and Kings County, and in the Eastside and Forebay subbasins of Monterey County, one-third of domestic or irrigation wells exceed the nitrate MCL. Consistent with these findings, the maximum nitrate level, measured in any given land section (1 square mile) for which nitrate data exist between 2000 and 2009, exceeds the MCL across wide portions of these areas. Low nitrate concentrations tend to occur in the deeper, confined aquifer in the western and central TLB. From 2000 to 2011, the median nitrate concentration (the concentration exceeded by half of all samples) in the TLB and SV public water supply well samples was 23 mg/L and 21 mg/L, respectively, and in all reported non-public well samples, 23 mg/L and 20 mg/L, respectively. In public supply wells, about one in ten raw water samples exceeds the nitrate MCL.

Nitrate levels have not always been this high. While no significant trend is observed in some areas with low nitrate (e.g., areas of the western TLB), USGS research indicates significant long-term increases in the higher-nitrate areas of the TLB (Burow et al. 2008), which is consistent with the CASTING dataset. Average nitrate concentrations in public supply wells of the TLB and SV have increased by 2.5 mg/L (±0.9 mg/L) per decade over the past three decades. Average trends of similar magnitude are observed in private wells. As a result, the number of wells with nitrate above background levels (> 9 mg/L) has steadily increased over the past half century from one-third of wells in the 1950s to nearly two-thirds of wells in the 2000s (Figure 57). Due to the large increase in the number of wells tested across agencies and programs, the overall fraction of sampled wells exceeding the MCL grew significantly in the 2000s.

5.1.1 Disparate Data Sources and Collection Methods

The data analyzed in this section were gathered from many different sources (see Section 4). As such, the spatial accuracy and precision of these data are variable, with some wells located to a high degree of certainty and others very poorly located. Nitrate concentration data, in contrast to spatial data, was generally collected with consistent protocols across the various datasets, though detection limits have improved over time. The difference between filtered and unfiltered nitrate concentrations is assumed to be insignificant.

5.1.2 De-clustering of Data Sources

Wells documented in the CASTING database are very unevenly distributed across the study area. In addition, specific data sources target only specific sub-areas within the study area (e.g., public supply wells are dominantly located in urban areas). For some of the statistical analyses described here, we performed a spatial de-clustering of the dataset to estimate nitrate concentrations by aquifer proportions (Belitz et al. 2010). The study area was first divided into five major physiographic groundwater regions based on sedimentological, hydrological, and geological characteristics.

For the SV, the study area was divided into two regions following the USGS GAMA 2005 study (Kulongoski & Belitz 2007). The two areas used were the "Monterey Bay" portion of the SV, which encompasses the Pressure, Eastside, Langley, Seaside, and Corral de Tierra groundwater subbasins, and the "Salinas Valley" area, which contains the Upper Valley and Forebay groundwater subbasins. For the Monterey Bay area, the subbasins were combined based on the similarity in Quaternary deposits (Kulongoski & Belitz 2007). The Forebay and Upper Valley subareas were combined based on their similar geology (Kulongoski & Belitz 2007).

The TLB study area was divided into three separate regions based on sediment origins. The three groups, referred to as the Eastside Alluvial Fan, Basin, and Westside Alluvial Fans regions, were delineated by the USGS and refer to the origin of the alluvial sediments present in these regions (Burow et al. 1998).

Each of the five regions in the study area is further divided into spatially distributed, randomized equal area cells (Figure 38 and Figure 39). The equal area cells for the Monterey Bay and Salinas Valley regions were those used in a 2005 USGS GAMA study of nitrate and pesticide contamination in the SV (Kulongoski & Belitz 2007). For the Monterey Bay region, the study area for this report did not include a northern region included in the 2005 USGS GAMA report, and therefore those cells located outside study area boundary were not included in the analysis provided here. The Monterey Bay region was divided into 48 equal area cells, the Salinas Valley into 31 equal area cells, each measuring 25 km^2 (9.7 sq. miles). The equal area cells within the Eastside Alluvial Fan, Westside Alluvial Fan, and Basin regions of the TLB were generated for this project using the same method by Scott (1990) that was also applied in the USGS GAMA studies. Cell size in the three TLB regions was 81 km^2 (31 sq. miles) : The Eastside Alluvial Fans was divided into 120 equal area cells, the Basin into 65 equal area cells, and the Westside Alluvial Fans into 56 equal area cells (Figure 39). The equal area cells are used to determine representative statistical values for each equal area cell (for example, the mean nitrate concentration in a cell). Further statistical analysis is then performed on the values representative of each cell, thus giving equal weight to each cell area. Details of the statistical analyses performed on equal area cells and the results of the analyses are described in the following sections. A variety of statistical tests are performed. The sections below specifically identify when the equal area cells are used for de-clustering the data (e.g, Table 20, Table 24, Figure 46, Figures 58-60).

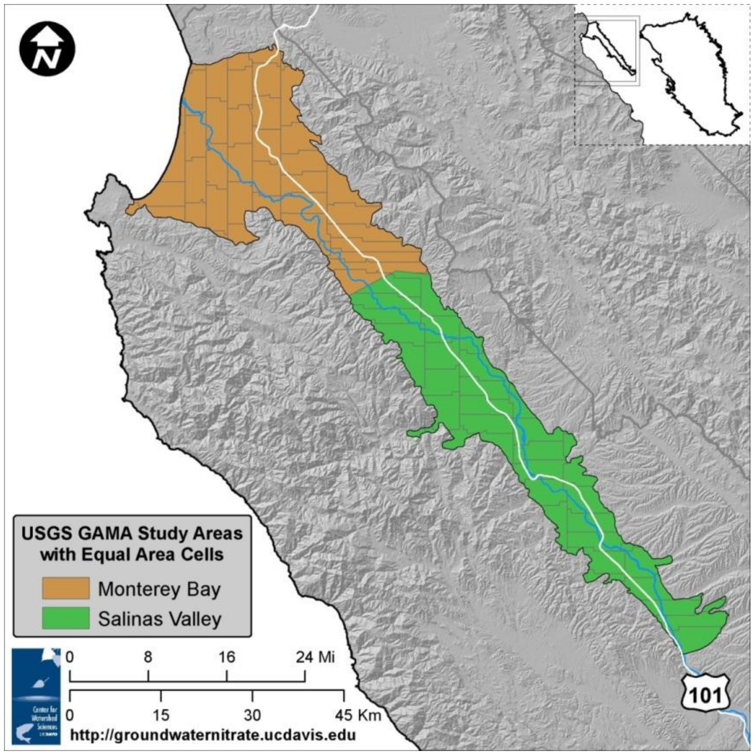

Figure 38. The two physiographic groundwater regions in the SV and their equal area cells used for spatial de-clustering. (Source: Kulongoski & Belitz 2007.)

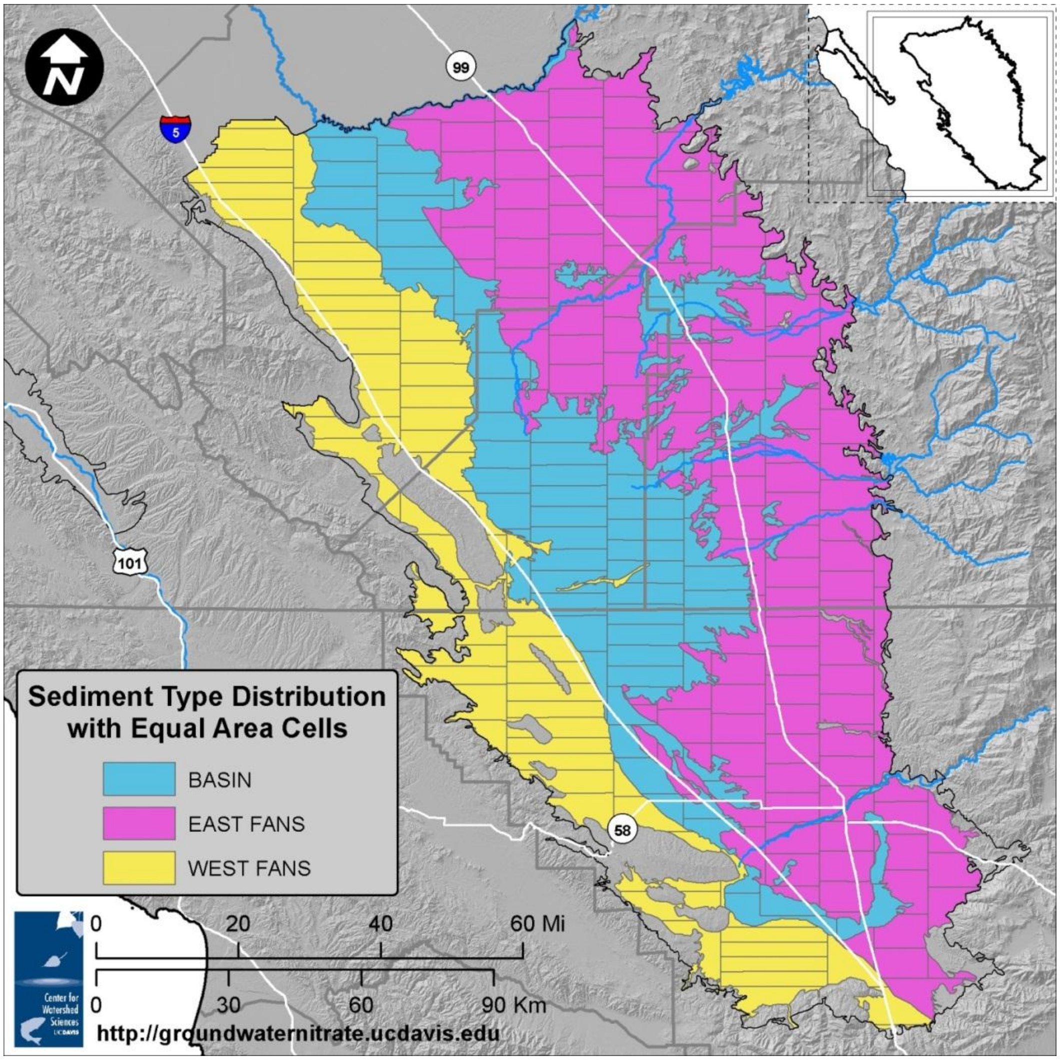

Figure 39. The three physiographic groundwater regions of the Tulare Basin study area and their corresponding equal area cells, which we generated for this report using the procedure of Scott (1990).

5.2 Current Spatial Distribution of Nitrate in Wells

5.2.1 Dataset Overview

The CASTING database version used for the analysis in this section is versioned 2012-01-29. The database contains a total of 100,084 samples from 19,095 wells within the study area and analyzed for nitrate. The dataset combines nitrate concentrations from 16,709 individual samples taken at 1,890 wells in the SV and from 83,375 individual samples taken at 17,205 wells in the TLB collected from the 1940s to 2011. Almost 70% of these samples were collected from 2000 to 2010; only 15% of the samples were collected prior to 1990. Half of all wells sampled had no recorded samples prior to 2000.

Of the 19,000 wells, approximately 2,500 are frequently sampled public water supply wells (over 60,000 samples) located often in urban or peri-urban areas. The Central Valley dairy regulatory program, established in 2007, annually monitors about 4,000 domestic and irrigation wells in the TLB, located mostly in agricultural and rural areas. Apart from these two programs, there are no existing other regular well sampling programs for domestic or other private wells. The exceptions are a ten year old monitoring program on 71 domestic wells in eastern Fresno and Tulare County (DPR), sampled once or twice annually; and Monterey County Water Resources Agency's private well monitoring program data, ongoing for two decades, but with most data confidential and not accessible to this study.

The distribution of nitrate values among the 100,084 samples is highly skewed with most values (86%) less than the drinking water limit of 45 mg/L. One quarter of all samples measure less than 6.8 mg/L, half of all samples measure 19 mg/L or less, and one quarter of all samples measure more than 32.3 mg/L. One in twenty samples exceeds 79 mg/L and one in a hundred samples exceeds 194 mg/L (more than four times the drinking water standard).

5.2.2 Exceptionally High Values and Treatment of Outliers

A very small fraction of samples, 202 records or two-tenths of one percent of all records, indicate nitrate concentrations of 500 mg/L or higher (more than one order of magnitude above the drinking water limit). Of these samples, most are associated with environmental monitoring wells (166 samples) with the highest recorded concentration being nearly 26,000 mg/L. All these environmental monitoring well samples were taken in the 2000s (after 1/1/2000). Another 26 samples are associated with USGS research projects (NWIS database). It is likely that these wells also sample shallow-most groundwater. Among the 36 wells with unknown well use or not designated as environmental monitoring wells (Well Use is *not* "Q"), one sample is 2,500 mg/L, 7 samples are between 1,000 and 1,700 mg/L, and 28 samples are between 500 and less than 1,000 mg/L. Only five of these 36 samples were taken since January 1, 2000 and among these five samples, the maximum concentration is 903 mg/L.

Unless explicitly mentioned, we did not remove any of these values as outliers. Instead, for many analyses, samples taken from wells designated as environmental monitoring wells (except the "monitoring wells" in the Monterey County Water Resources Agency agricultural environmental monitoring program, “MCEM”) were considered separately or excluded. Among the 1,018 MCEM monitoring well samples (from 239 wells), only one sample exceeded 500 mg/L (value: 681 mg/L), and it was taken prior to 1/1/2000. MCEM wells that are designated as “monitoring wells” were not considered part of the “environmental monitoring well” set, since these are not installed to monitor existing site contamination. MCEM wells are included in the analyses below, where the analysis is designated as excluding environmental monitoring wells.

5.2.3 Frequency Distribution

The frequency distribution (histogram) of nitrate values is best fitted with a log-normal distribution. Statistical tests (Chi-Square test and Kolmogorov-Smirnov test, Haan, 1977) for goodness-of-fit with the log-normal distribution are significant at the 1% level. The same goodness-of-fit tests, at the 1% significance level, suggest that the following aggregated nitrate data are also log-normal distributed:

- The 19,095 mean nitrate values computed for each well, with the mean computed for the entire period of record.
- The 23,402 mean nitrate values computed for each well for each decade with well sample records (“decadal well means”).
- The 55,541 mean nitrate values computed for each well for each year with well sample records (“annual well means”).

The $\log_{10}(c)$ of all samples, where c is the sample nitrate value, and the mean of $\log_{10}(c)$ for each well for each decade statistically fit a normal distribution, although the histogram is slightly skewed (Figure 40). The sample mean of $\log_{10}(c)$ is 1.116 (13.1 mg/L of nitrate) and the standard deviation is 0.631, more than a half order of magnitude. The decadal well means of $\log_{10}(c)$ have a mean of 0.929 (8.3 mg/L of nitrate) and a standard deviation of 0.753, nearly one order of magnitude. Because the 2,500 public supply wells make up more than half of all samples, but only one-eighth of all wells, statistics on the total set of all 100,000 samples will be highly biased toward measurement in public supply wells. The distribution of means computed for each well is a better representation of the nitrate distribution in the study area than the distribution of the full sample set. An even less biased analysis is obtained by applying statistical measures to the equal area cells described in the previous section (de-clustered analysis).

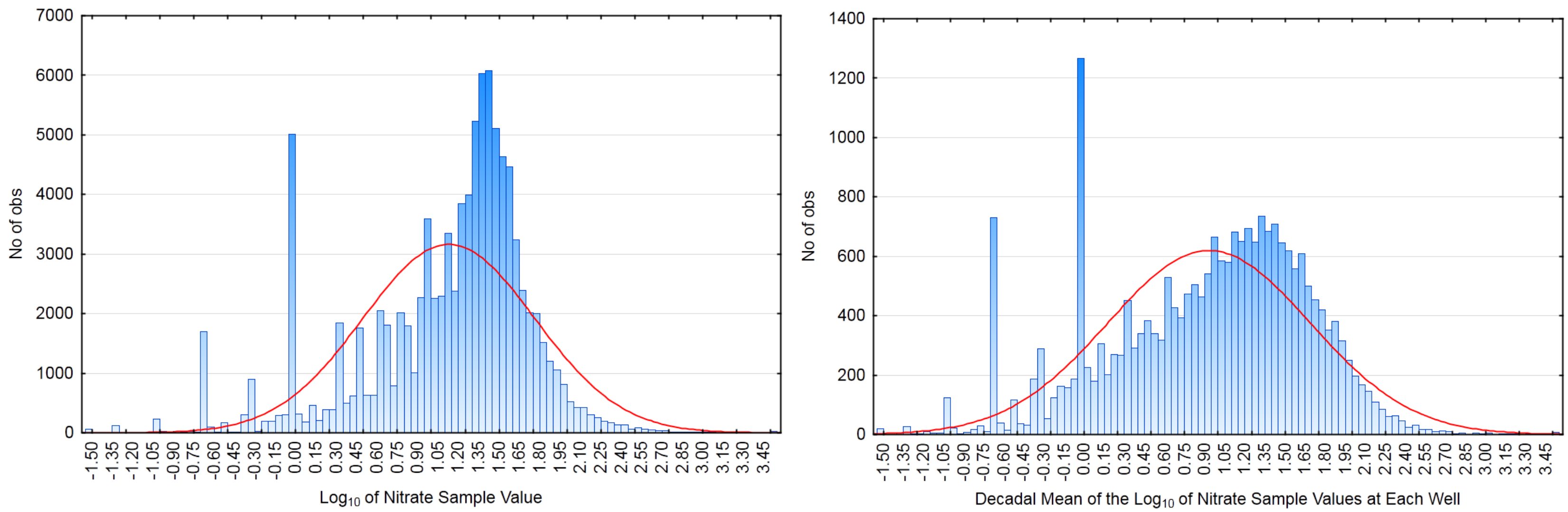

Figure 40. Histogram of $\log_{10}(c)$ (x-axis), where c is the sample nitrate value, for all 100,029 non-zero individual samples (left), and for all 23,377 non-zero decadal means of $\log_{10}(c)$ at individual wells (right). The distributions statistically fit a normal distribution, although they are slightly skewed. A large number of samples are recorded at half of often reported detection limits (detection limits were converted to actual concentrations at half of the detection limit). In logarithmic units, the MCL for nitrate is 1.65. X-axis values of -1, 0, 1, 2, and 3 correspond to nitrate values of 0.1, 1, 10, 100, and 1000 mg/L, respectively.

Due to the skewed distribution of nitrate, much of the remaining analysis focuses on statistical measures that are not dependent on distributional assumptions and that are robust against outliers or high values. In particular, we use medians (exceeded by exactly half of the sample populations) and exceedance probabilities relevant to nitrate. The exceedance probability is the probability that nitrate exceeds a certain value, that is, the fraction of wells in which nitrate exceeds a certain value. In particular, we consider the exceedance probabilities for the following four thresholds:

- 9 mg/L nitrate, which is the level that is often used to separate background nitrate concentrations from anthropogenically influenced nitrate concentrations
- 22.5 mg/L nitrate, which is half of the maximum contaminant level (MCL)
- 45 mg/L nitrate, the drinking water MCL
- 90 mg/L nitrate, twice the drinking water MCL

We focus our analysis on nitrate data collected since January 1, 2000. We consider the 2000s period representative for overall current conditions. Older data are considered for the time trend analysis (see below). Furthermore, we consider the following sample populations for which we compute statistical values:

- All raw sample values ("sample values")
- The annual median nitrate concentration at each well ("annual well median")
- The annual mean nitrate concentration at each well ("annual well mean")
- The decadal mean nitrate concentration at each well ("decadal well mean")
- The decadal mean of $\log_{10}(c)$ at each well, where c is the nitrate concentration ("decadal well mean log nitrate")

5.2.4 Spatial Distribution of Nitrate Concentration

There are 10,120 wells in the TLB study area sampled at least once during the 2000's decade. The back-transformed median concentration[9] of the decadal well means of log nitrate is 12 mg/L. The median of the annual well means is 15 mg/L. About one-quarter (24%) of all wells have a decadal well mean of log nitrate exceeding the MCL, while nearly 20% of annual well mean concentrations exceed the MCL. There are 1,768 wells in the TLB that have been sampled at least 4 times. Of these, 406 (23%) exceeded the MCL for nitrate at least once in that decade.

In the SV, there are 1,474 wells sampled at least once during the 2000's decade. The back-transformed median concentration of the decadal well means of log nitrate is 9.4 mg/L. The median of the annual well means is 14 mg/L. About 13% of all wells have a decadal well mean of log nitrate exceeding the MCL, and 15% of annual well mean concentrations exceed the MCL. In the SV, 729 wells have been

[9] The "backtransformed median" of a log-transformed variable is equal to 10 to the power of the median of the log10-tranformed variable. For example if the median of the log10-transformed nitrate concentrations is 1.3, then the backtransformed median is $10^{1.3} = 19.95$.

sampled at least 4 times during the 2000s, of which 215 (29%) exceeded the MCL for nitrate at least once in that decade.

The above represent different statistical measures that illustrate the range of “typical” or median nitrate in groundwater and of the rate at which the MCL is exceeded across various statistical approaches. The decadal well means represent one mean for each well, regardless of the number of samples from that well or in how many years during the decade it would have been sampled. In the computation of the median of decadal well means, each well weighs equally. In contrast, for each well there may be one or several annual well means per decade. Therefore, wells with multiple years of sampling weigh proportionally more in the computation of the ‘median of annual well means’ than wells with a single sample during the decade. In the median of annual well means, for example, public wells weigh generally ten times more than private domestic wells, because most public wells are tested at least once each year (one annual well mean per year per well), while most domestic and other private wells in the database have only been sampled once (except those in the “dairy” dataset). In the TLB and the SV, the median of annual well means is higher than the back-transformed median of decadal well means of log nitrate, indicating that wells with multiple samples tend to have higher nitrate values. In contrast, in the TLB, the MCL exceedance rate among decadal well means of log nitrate is higher than among annual well means. To further investigate these differences, we will analyze subsets of these data (e.g, public supply wells, domestic wells) further in the following sections. Importantly, the diverse measures given here all demonstrate the significant percentage of wells that exceed the MCL and provide a measure of the magnitude of median nitrate concentration in groundwater.

In the TLB, wells in the eastern and central portions of the valley tend to have higher average concentrations of nitrate than those in the western part of the valley (Figure 41). The Westside Alluvial Fans region of the TLB has fewer wells, but the data that are available suggest lower average levels of nitrate in these wells compared to the eastside. In the SV, the higher average concentrations are observed in the northeastern, central, and southern portion of the SV. The map of maximum values observed in each well (Figure 42) appears visually very similar to average values (Figure 41) due to the large number of wells with a single or few measurements.

Due to the density of wells in some areas of the study area, it is difficult to visualize all wells directly. To better understand the distribution of nitrate concentration in the study area, the average concentration of nitrate in wells for each Public Lands Survey Section for the years 2000 through 2009 were calculated (Figure 43). In any given section, there may be anywhere from one well with a single sample to many wells with one or multiple samples per well. The average nitrate concentration in each section was computed from the average nitrate concentration at each well in that section in 2000 – 2009. Thus, for the average nitrate concentration in each section, each well was counted only once, regardless of the number of samples. A map showing the maximum nitrate concentration observed anywhere within each section between 2000-2000 was similarly prepared (Figure 44).

When the median of all annual well mean nitrate values are computed for each equal area (see Section 5.1 for maps of equal areas), the de-clustered median nitrate concentrations over all equal areas (not including environmental monitoring wells) are 16 mg/L (TLB Eastside Fans), 2.6 mg/L (TLB Basins), 1.5

mg/L (TLB Westside Fans), 13 mg/L (Monterey Bay region), and 12 mg/L (Salinas Valley region) (see Figure 45). For a de-clustered analysis of exceedance probabilities, we compute the MCL exceedance probability for each equal area separately, based on the number of annual well means exceeding the MCL within each equal area during the 2000s (not including environmental monitoring wells). The de-clustered average MCL exceedance rate (average of the exceedance rate in each equal area) in each of the five groundwater regions is 20% (TLB Eastside Fans), 11% (TLB Basins), 14% (TLB Westside Fans), 14% (Monterey Bay region), and 22% (Salinas Valley region) (see Figure 46).

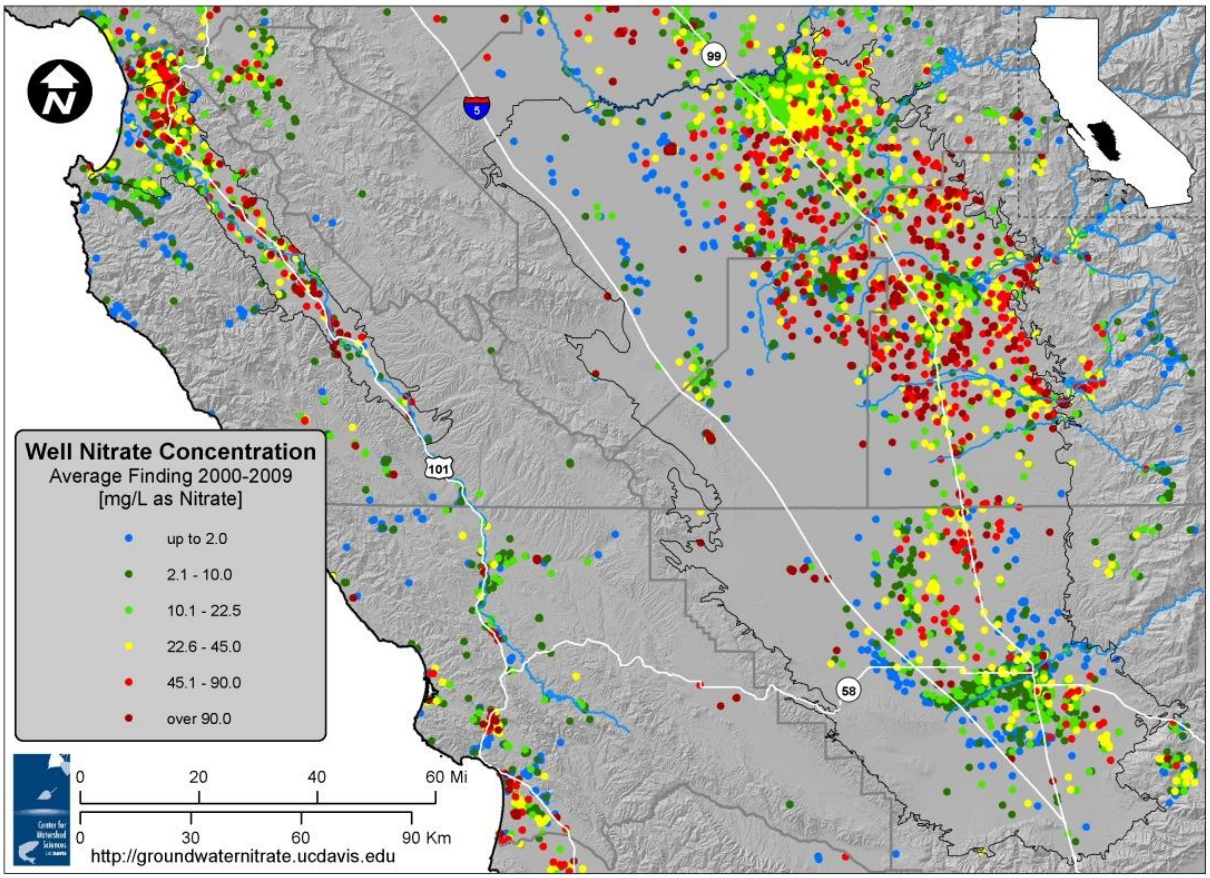

Figure 41. Average concentration of nitrate (mg NO3/L) in wells with at least one sample during the decade 2000 to 2010 (not including SWB Geotracker environmental monitoring wells). Red and dark red indicate wells with average concentration above the MCL. Yellow indicates wells that are above one half the MCL, but below the MCL.

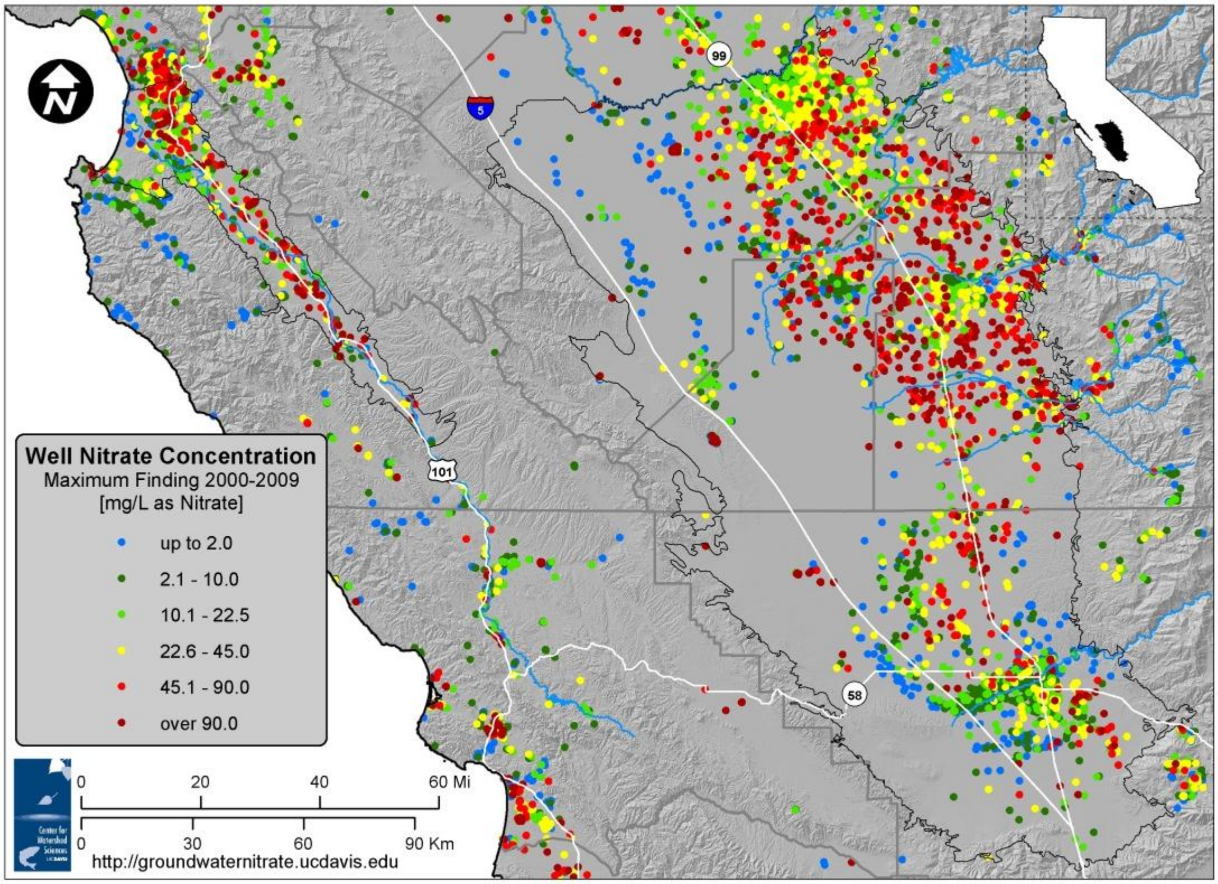

Figure 42. Maximum concentration of nitrate in wells with at least one sample during the decade 2000 to 2010 (not including SWB Geotracker environmental monitoring wells). Red and dark red indicate wells with a maximum measured concentration over the MCL. Yellow indicates wells with a maximum measured concentration that is above one half the MCL but below the MCL.

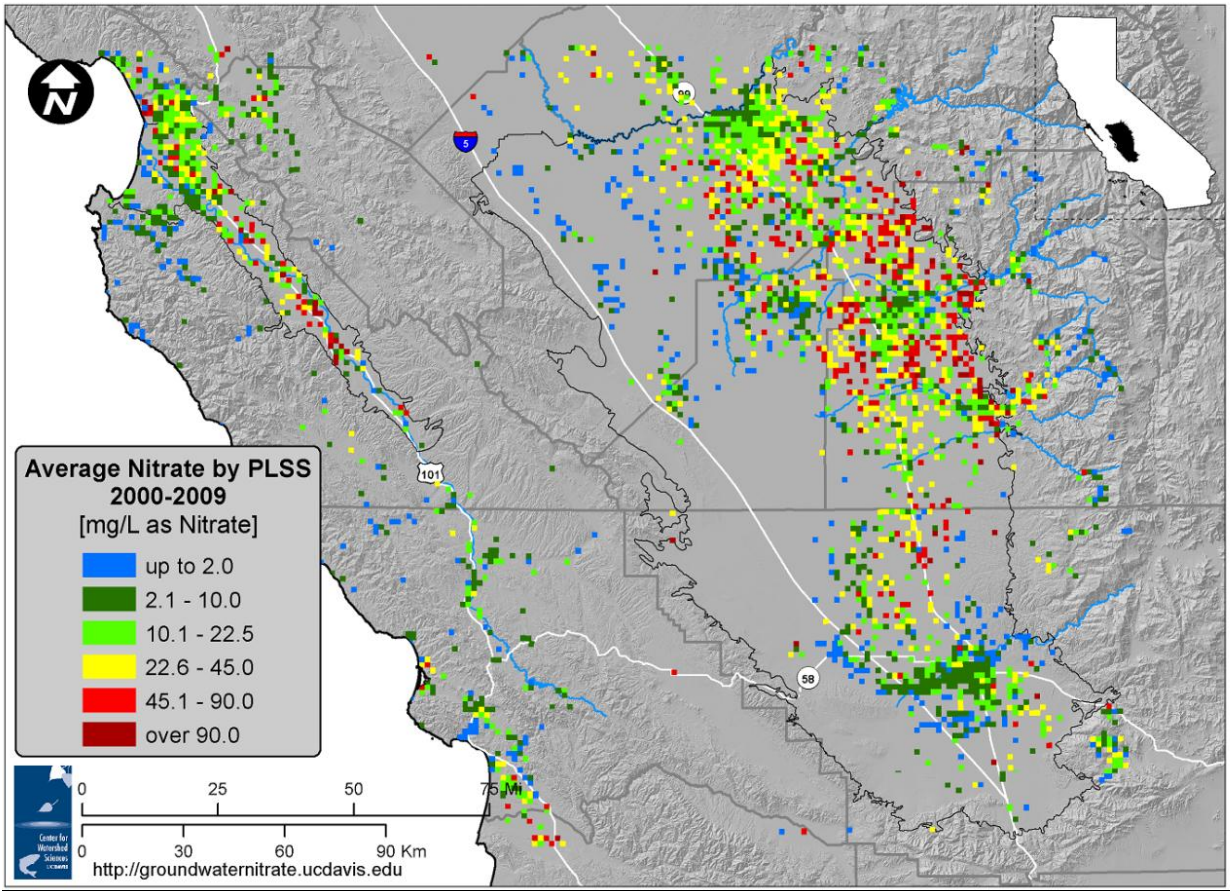

Figure 43. Average nitrate by PLSS section, approximately 2.59 km^2 (1 mi^2), computed from the average at each well of all samples obtained during the years 2000 to 2009 (not including samples from SWB Geotracker environmental monitoring wells). Thus, each well is only counted once toward the average nitrate in each section, regardless of the number of samples obtained for each well. Red and dark red sections are those where the average of all samples in a PLSS section is over the MCL. Yellow indicates PLSS sections where the average of all known samples in this time period is above one half of the MCL.

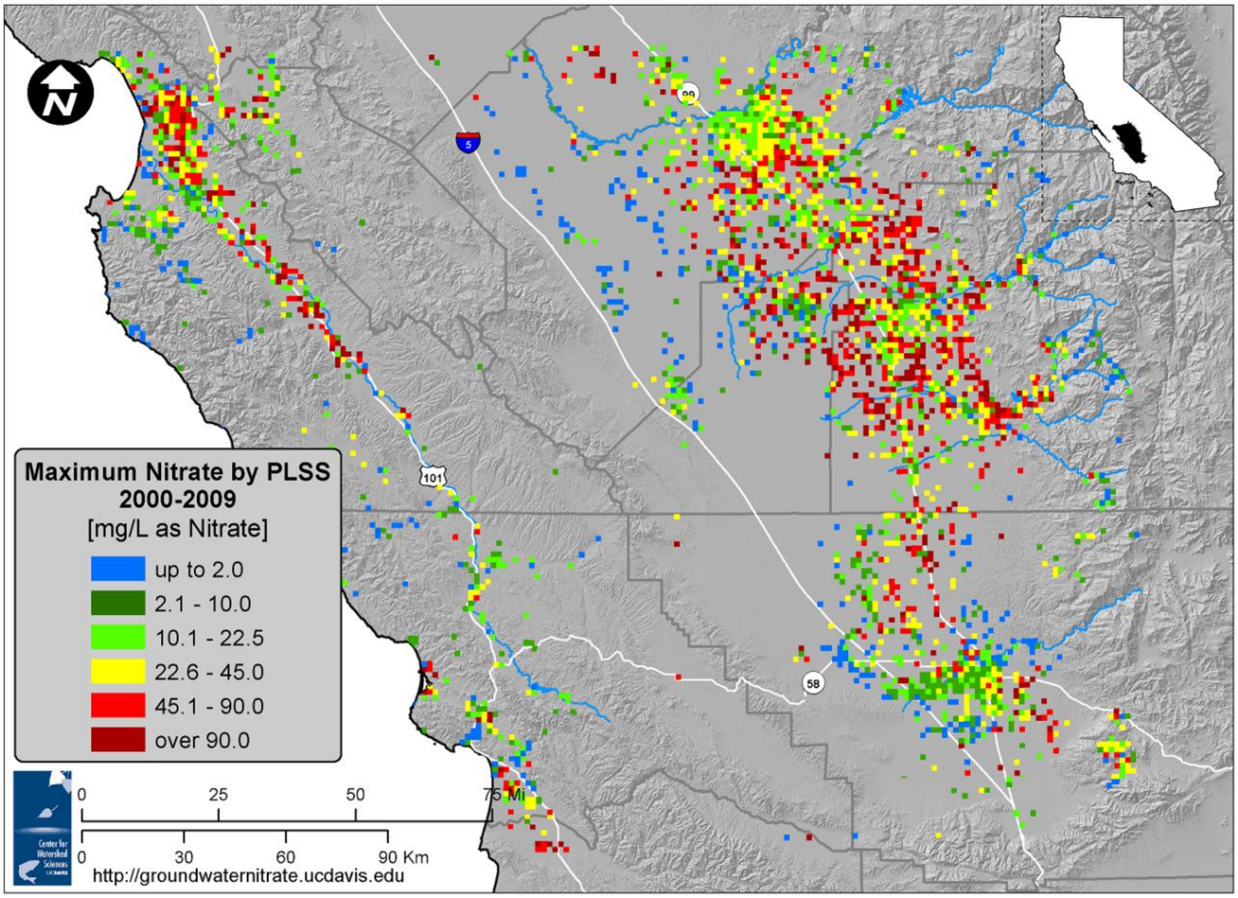

Figure 44. Maximum nitrate concentration measured between 2000 and 2009 in each PLSS section (not including samples from SWB Geotracker environmental monitoring wells), where each section is approximately 2.59 km^2 (1 mi^2).

We de-cluster the data by first taking the annual well means for the 2000s (except environmental monitoring wells), then computing the median of all annual well means in each equal area, and finally taking the mean of the logarithm of equal area medians across all equal areas within each groundwater region (Figure 46). The logarithm is taken because equal area medians are log-normally distributed across the equal areas of the study area. When the data are de-clustered by equal area, the TLB (Central) Basin area has a lower back-transformed mean nitrate (3.2 mg/L) than before de-clustering (7.8 mg/L). All other groundwater regions have similar mean nitrate with or without de-clustering (compare Figure 45 with Figure 46).

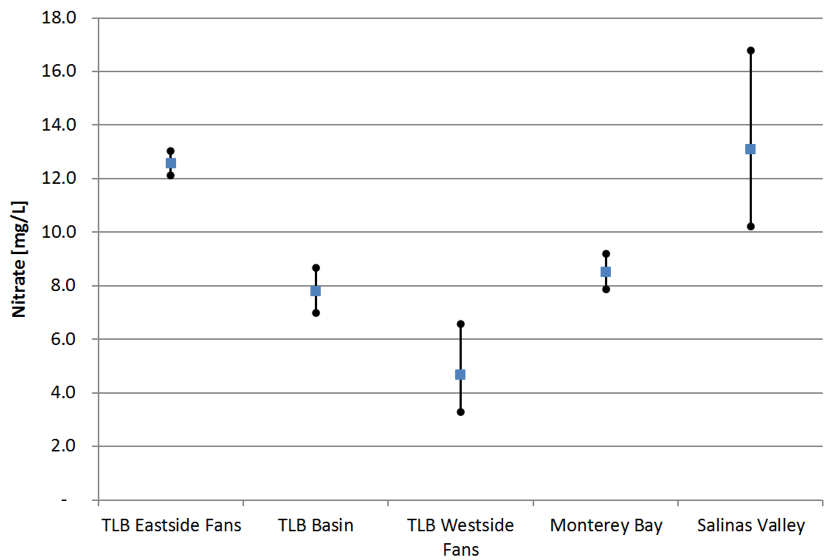

Figure 45. Back-transformed mean and confidence interval of decadal well means of log nitrate [y-axis: in mg/L nitrate] for the five major groundwater regions in the study area without equal area de-clustering. Includes all wells sampled in the 2000s, except environmental monitoring wells.

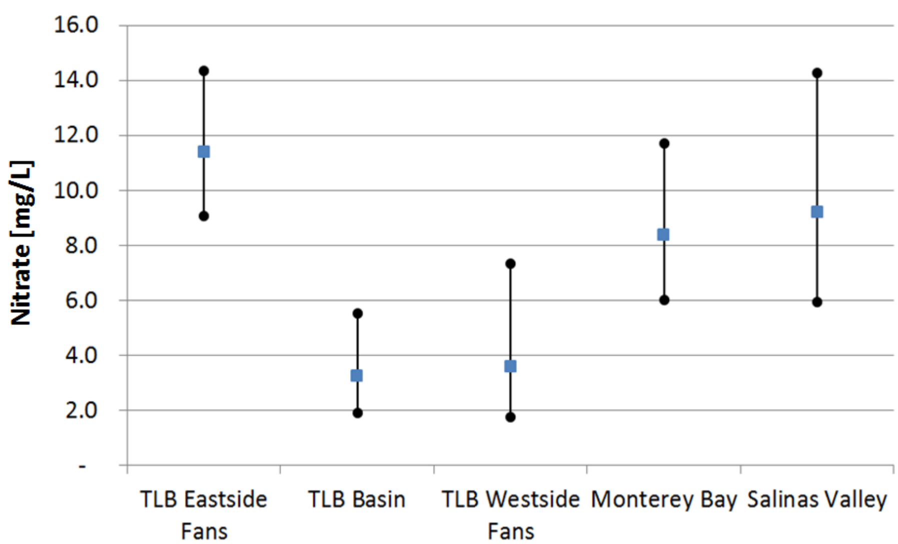

Figure 46. Back-transformed mean and confidence interval of the mean over all equal areas of the logarithm of the median in each equal area of annual well means [y-axis: in mg/L nitrate], separately for the five major groundwater regions in the study area. Note that the medians of equal area annual well means are log-normally distributed. Includes all wells sampled in the 2000s, except environmental monitoring wells. Confidence intervals are broader due to the smaller number of data points (number of equal area cells) when compared to the previous figure's number of data points (number of wells) used for the statistical analysis.

5.3 Effect of Data Source, Well Type, and Well Depth on Groundwater Nitrate

5.3.1 Effect of Data Source

Many of the individual datasets that comprise the CASTING database for the 2000s represent groundwater nitrate samples taken for specific purposes, from specific well types, from a specific region, or under a specific program. This includes the CADWSAP dataset (public water supply wells, mostly located in urban areas, sampled frequently) and the dairy dataset ("Dairy", representing irrigation, domestic, and monitoring wells on dairies and sampled annually since 2007, under the dairy general order). It also includes the various county datasets collected from individual wells once, typically after construction of new domestic wells (Tulare and Fresno Counties, some of the Kern County records), on local small community water systems (Monterey County), or for regional monitoring purposes (DPR, GAMA, MCEM). Two large datasets in CASTING cover only historic (pre-2000) water samples and did not continue into the 21st century and were not considered here: U.S. EPA STORET and DWR legacy data.

We consider the data separately by major data source (Table 17 and Table 18) to illustrate the variability between different groundwater sampling programs. Nitrate samples were aggregated by well and year: for each well *i*, we first obtained the average nitrate concentration, $<c>_{ij}$, in each year *j* ("annual well mean"). We then performed an exceedance probability analysis on the ensemble of all annual well means, $<c>_{ij}$, using all wells *i* (of a particular grouping) and all years *j* (2000 – 2011) within the dataset. Table 17 shows, by data source, the number of wells for which samples were available, and the number of samples, the median nitrate value [mg/L], and the exceedance probability for the key threshold levels. Due to the significantly different nitrate concentrations in the five major groundwater regions, separate analyses were also performed for the major groundwater regions of the study area. Data for 2010 were incomplete, but included. Only few data were available for 2011 (not included in all analyses). Table 18 breaks down the total number of annual average well nitrate values available between 1/1/2000 and 2011 (number of well-years), by data source and by region.

Table 17. Data sources with the total number of samples recorded, total number of sampled wells, location of wells, type of wells, and for the last decade (2000-2010) in the TLB and SV: number of wells measured, median nitrate concentration, and percentage of MCL exceedance for the TLB and the SV.[2]

Data Source[1]	Total				TLB (2000 – 2010)			SV (2000 – 2010)		
	# Wells	# Samples	Well Locations	Well Type[3]	# Wells	Median[2] [mg/L NO_3^-]	% > MCL[2]	# Wells	Median[2] [mg/L NO_3^-]	% > MCL[2]
CDPH	2,421	62,153	throughout study area	PS	1,769	12	6%	327	8	5%
CVRWB DAIRY	6,459	11,300	dairies in TLB	D, I, M	6,459	22	31%			
DPR	71	814	eastern Fresno and Tulare Counties	D	71	40	45%			
DWR	26	44	Westlands Water District	I	28	1	0%			
DWR Bulletin 130	685	2,862	throughout study area	D, I, PS						
ENVMON	537	2,601	throughout study area	M	357		52%	180	27	44%
EPA	2,860	4,946	throughout study area							
Fresno County	368	369	Fresno County	D	349	18	15%			
GAMA	141	141	Tulare County	D	141	38	43%			
Kern County	2,893	3,825	Kern County	D, I	361	5	7%			
Monterey County, Reports	239	1,018	Monterey County	I, M				98	14	36%
Monterey County, Geospatial	388	1,574	Monterey County	LS				431	18	15%
Monterey County, Scanned	452	5,674	Monterey County	LS				427	17	14%
NWIS	1,028	2,151		Misc.	76	35	36%	4	0	0%
Tulare County	444	444	Tulare County	D	438	22	27%			
Westlands Water District	48	77	Westlands Water District	I	31	4	0%			

[1]Data Source: CDPH: public supply well database; CVRWB Dairy: Central Valley RWB Dairy General Order; DWR Bulletin 130: data reports from the 1960-1970s, 1985; ENVMON: State Water Board Geotracker environmental monitoring wells with nitrate data (does not include data from the CVRWB dairy dataset); EPA: STORET dataset; Fresno County: Public Health Department; GAMA: State Water Board domestic well survey; Kern County: Water Agency; Monterey County, Reports: data published in reports by MCWRA; Monterey County, Geospatial: Health Department geospatial database; Monterey County, Scanned: Health Department scanned paper records; NWIS: USGS National Water Information System; Tulare County: Health and Human Services; Westlands Water District: district dataset. Some smaller datasets are not listed. Individual wells that are known to be monitored by multiple sources are here associated only with the data source reporting the first water quality record.

[2] Median and percent MCL exceedance were computed based on the annual mean nitrate concentration at each well for which data were available (not spatially de-clustered). Depending on the number of years (between 2000 and 2010) for which data are available for a specific well, a well may be represented by one to eleven datapoints. See Table 18 for details.

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

For the various datasets, the public water supply wells (CADWSAP dataset) generally have among the lowest rates of MCL exceedances, and the by far lowest median of annual well means (less than 13 mg/L). Other datasets with less than 10% exceedance rates of the MCL are the Kern County wells (except the group of wells on the Westside Alluvial Fans), and the Westlands Water District irrigation wells dataset provided by WWD and DWR. Except for the public supply wells, these wells were mostly measured just once. The Westlands Water District wells are of unknown depths, but, based on their location, are likely deep wells pumping from the lower portions of the upper, semi-confined aquifer or from the confined aquifer below the Corcoran Clay. The Kern County dataset, which also has relatively low nitrate median, includes irrigation wells, many of which may tap deeper portions of the aquifer.

Many of the public supply wells (CADWSAP dataset) are located in urban or semi-urban areas and are impacted by urban recharge, particularly in the city of Fresno. The CADWSAP dataset includes many wells that are screened only at depths exceeding 100 m (330 ft). Wells with chronically elevated nitrate levels are often taken off-line by the water purveyor. Data representing water quality conditions after the well is taken off-line would not be sampled and therefore not be included in the dataset.

Monitoring wells (those in the State Water Board Environmental Monitoring Wells dataset and monitoring wells identified, by name, as monitoring wells in the Dairy dataset) have the highest exceedance rates for the MCL and also for the 90 mg/L threshold. More than half of these monitoring wells, deployed typically to track a known or suspected contamination problem, exceed the nitrate MCL and most of these latter wells measure more than 90 mg/L, twice the MCL. Monitoring wells are typically constructed to measure first encountered groundwater, with well screens at or immediately below the water table. They represent the water quality of recent recharge at the location of monitoring. Many of these are deployed specifically to locations with suspected or known contamination problems.

The datasets that consist exclusively of domestic wells (Fresno and Tulare County, DPR, GAMA Tulare project, and some of the Dairy dataset wells, where identified) show varying rates of nitrate MCL exceedance, ranging from about 30% to 45%. Wells monitored by the Monterey County Water Resources Agency and the Monterey County local system wells (serving less than five connections) show similar exceedance rates in the SV. Only the local systems wells in the northern SV (“Monterey Bay”, which includes the Pressure Aquifer and Eastside Basin sub-regions of SV) have lower MCL exceedance rates (less than 15%). Median values in domestic wells of the DPR and GAMA dataset, at about 40 mg/L, are highest in the Eastside Alluvial Fans region of the TLB. Median concentrations are lower in the central basin and Westside Alluvial Fans region of TLB

Table 18. Summary of database records for nitrate samples for the current period (after 1999) including the data source, the number of wells measured by each data source, the number of wells-years (the sum of all wells, each multiplied by the number of years during which it was sampled), the median nitrate value (exceed by 50% of all samples), and the percent of annual well means that exceeded 9, 22.5, 45, and 90 mg/L nitrate.[1]

Region	Data Source	# of Wells	# Well-Years	Median [mg/L]	> 9 mg/L	> 22.5 mg/L	> 45 mg/L	> 90 mg/L
TLB Eastside Fans	CADWSAP	1,597	11,867	12.5	62.0%	26.1%	5.7%	0.8%
	DAIRY	5,310	9,078	23.0	67.5%	50.3%	30.8%	11.1%
	DPR	70	666	40.5	89.2%	72.8%	45.9%	12.8%
	FRCO	325	325	19.3	73.8%	45.8%	15.4%	1.2%
	GAMA	124	124	41.0	83.9%	68.5%	44.4%	19.4%
	KECO	248	248	5.1	29.8%	12.5%	8.5%	2.8%
	KICO	9	14	10.0	50.0%	28.6%	14.3%	7.1%
	NWIS	75	128	34.9	75.8%	60.9%	35.9%	5.5%
	TCEHS	394	395	22.2	75.7%	49.9%	27.1%	5.1%
	USGS	6	14	32.9	85.7%	57.1%	42.9%	14.3%
	Mon. Wells	298	553	53.8	77.0%	66.0%	52.8%	37.6%
TLB Central Basin	CADWSAP	147	912	8.0	47.0%	17.1%	6.7%	1.5%
	DAIRY	995	1,776	17.7	60.1%	44.8%	26.5%	8.7%
	FRCO	23	24	1.0	25.0%	12.5%	12.5%	0.0%
	GAMA	15	15	32.4	86.7%	66.7%	33.3%	13.3%
	KECO	91	91	4.4	35.2%	13.2%	3.3%	0.0%
	TCEHS	34	34	23.2	82.4%	50.0%	26.5%	5.9%
	Mon. Wells	114	323	122.2	74.0%	66.6%	61.3%	56.7%
TLB Westside Fans	CADWSAP	20	108	1.5	26.9%	5.6%	0.0%	0.0%
	DWR	27	27	1.0	22.2%	3.7%	0.0%	0.0%
	KECO	11	11	6.4	45.5%	45.5%	27.3%	18.2%
	WWD	30	54	3.4	33.3%	9.3%	0.0%	0.0%
	Mon. Wells	29	61	62.0	62.3%	59.0%	52.5%	45.9%
SV-Pressure Aquifer, Eastside, and Monterey Bay[2]	CADWSAP	270	1,511	8.1	48.6%	21.6%	4.6%	0.3%
	MCEM	31	126	6.7	47.6%	37.3%	31.7%	18.3%
	MOCO	395	1,319	18.0	70.6%	40.9%	12.4%	1.7%
	MOCODig	412	1,839	17.0	65.6%	41.3%	13.8%	1.5%
	Mon. Wells	170	570	25.6	60.0%	51.2%	43.0%	27.9%
SV-Forebay and Upper Valley[3]	CADWSAP	57	296	7.0	42.9%	18.9%	4.4%	0.3%
	MCEM	67	72	38.3	66.7%	56.9%	44.4%	23.6%
	MOCO	36	123	24.0	69.9%	51.2%	39.8%	22.0%
	MOCODig	15	56	22.5	60.7%	50.0%	35.7%	23.2%
	Mon. Wells	10	22	77.2	81.8%	72.7%	59.1%	45.5%

[1] Higher nitrate concentrations are exceeded by fewer samples. Only datasets with at least 5 wells and at least 10 well-years are listed. Monitoring well ("Mon. Wells") records are analyzed separately and consist of those from the State Water Board's Geotracker and those in the "DAIRY" dataset.
[2] The "Monterey Bay" area, designated by the 2005 USGS GAMA basin study, encompasses the following Bulletin 118 subbasins: Pressure (180/400 foot aquifer), Eastside, Langley, Seaside, and Corral de Tierra areas.
[3] This area is called the "Salinas Valley" area in the 2005 USGS GAMA basin study.

5.3.2 Depth-Dependency of Nitrate

Given the diversity of well sampling programs, the lack of consistent information on well depth or screen length, and the wide range of the number of samples taken at individual wells, a quantitative regression to determine the depth-dependency of nitrate is not possible. Instead, we compared five groups of wells (not exclusive of each other, in other words, some wells may occur in multiple groups) to qualitatively determine, whether there is a potential effect with well-depth:

- Monitoring wells: State Water Board environmental monitoring wells and monitoring wells labeled as such in the CVRWB dairy dataset. Thought to be screened at depths of less than 30 m (100 ft) below the water table;
- Shallow private wells:
  - Domestic wells or wells of local water systems with less than five connections, designated as such in CASTING. Mostly (but not exclusively) screened at depths of less than 100 m (330 ft);
  - Private wells, including explicitly designated domestic wells, with known depth to the top-of-screen of 60 m (200 ft) or less;
- Shallow private and public wells: All wells, including public supply wells and those identified as domestic wells, with known depth to the top-of-screen of 60 m (200 ft) or less;
- All public supply wells (CADWSAP dataset), including those with unknown screen depth and those with deeper screens, and all wells from the Westlands Water District dataset (WWD and DWR).

Medians and exceedance probabilities of annual well means for these five groups were computed as described in the previous section (for Tables 17 and 18) and are shown in Table 19. As discussed above, the monitoring wells have the highest exceedance rate for both the MCL and the 90 mg/L (twice the MCL) threshold. Their median concentrations range from less than the MCL in the northern SV to over 120 mg/L in the central TLB.

Shallow private wells located in the eastern TLB and in the southern portion of the SV (Forebay and Upper Valley subbasins) have MCL exceedance rates of more than 30% and nearly 45%, respectively, in agreement with exceedance rates found in previous studies (i.e., Burow et al. 1998). Lower exceedance rates are observed in domestic wells elsewhere. Median concentrations are as high as 27 mg/L or as low as 16 mg/L.

Along the central basin of the TLB, the MCL exceedance rates in shallow private wells are more moderate, about one-quarter. Less than five such wells are located on the Westside alluvial fans of the TLB, not sufficient for a significant determination of exceedance rates in that groundwater region.

In the northern SV (Monterey Bay area, Pressure and Eastside subbasins), the exceedance rate is between 10% and 15% for domestic and shallow private and public supply wells, but only 5% for all public supply wells. In the northern SV, the Eastside subbasin is considered more vulnerable than the Pressure subbasin due to overlying agricultural activities and the lack of a confining layer. There, domestic wells and local water supply wells (less than five connections), but also public supply wells with less than 60 m (200 ft) to the top-of-screen have a somewhat higher MCL exceedance rate of 17% (not shown in Table 19). In comparison, the MCL exceedance rate in the Pressure subbasin specifically is 12% for domestic and local supply wells and 0% for shallow public supply wells (not shown in Table 19). Shallow private wells known to have less than 60 m (200 ft) to the top-of-screen in either of these two subbasins have an exceedance rate of about 14%.

For further statistical analysis, we simplify the depth classification into two groups: we compare the group of wells known to either be a household or local small systems supply well or known to have less than 60 m (200 ft) to the top of the well screen ("shallow" wells), against the group of wells that either have more than 60 m (200 ft) to the top of the well screen or are not designated as monitoring, household, or local small systems wells ("non-shallow" wells). Wells that have neither any depth information nor any well use classification are not included in this comparison. The mean of the decadal well means of the log of nitrate are significantly higher in the "shallow" wells than in the "non-shallow" wells (Table 20). Even after spatial de-clustering, distinctly higher means of nitrate are obtained in the "shallow"equal area zones of each groundwater region than in the "non-shallow" equal area zones (Table 20).

Table 19. Median nitrate and exceedance probability of annual well means as a function of well depth and location on nitrate concentration.[1]

Region	Well Depth Category	# of Wells	# Well-Years	Median	> 9 mg/L	> 22.5 mg/L	> 45 mg/L	> 90 mg/L
TLB Eastside Fans	Monitoring	298	553	53.8	77.0%	66.0%	52.8%	37.6%
	Domestic	1,749	2,879	27.4	75.3%	55.9%	33.0%	9.6%
	<200', priv.	785	1,143	27.2	78.8%	56.6%	31.7%	7.5%
	<200', all	1,241	4,682	16.5	69.1%	38.1%	15.4%	2.8%
	public	1,597	11,867	12.5	62.0%	26.1%	5.7%	0.8%
TLB Central Basin	Monitoring	114	323	122.2	74.0%	66.6%	61.3%	56.7%
	Domestic	257	387	21.3	63.8%	47.8%	26.9%	8.0%
	<200', priv.	70	71	19.0	67.6%	42.3%	23.9%	4.2%
	<200', all	118	404	16.5	67.3%	34.2%	14.6%	3.7%
	public	148	913	8.0	47.0%	17.1%	6.7%	1.5%
TLB Westside Fans	Monitoring	29	61	62.0	62.3%	59.0%	52.5%	45.9%
	<200', priv.	2	2	24.1	50.0%	50.0%	0.0%	0.0%
	<200', all	3	9	1.5	11.1%	11.1%	0.0%	0.0%
	public	77	189	1.5	28.0%	6.3%	0.0%	0.0%
SV-	Monitoring	170	570	25.6	60.0%	51.2%	43.0%	27.9%
Pressure Aquifer,	Domestic	530	1,970	16.0	65.0%	38.3%	14.0%	2.0%
Eastside,	<200', priv.	108	458	16.0	65.3%	37.6%	13.1%	2.2%
and Monterey Bay	<200', all	146	678	14.6	63.6%	36.3%	10.0%	1.5%
	public	270	1,511	8.1	48.6%	21.6%	4.6%	0.3%
SV-Forebay	Monitoring	10	22	77.2	81.8%	72.7%	59.1%	45.5%
and Upper Valley	Domestic	33	105	22.0	67.6%	49.5%	40.0%	22.9%
	<200', priv.	62	84	36.3	61.9%	56.0%	42.9%	21.4%
	<200', all	79	193	10.0	52.8%	34.2%	22.3%	9.8%
	public	57	296	7.0	42.9%	18.9%	4.4%	0.3%

[1] Monitoring wells ("Monitoring") are typically measuring the shallow-most groundwater (upper 15 m – 30 m [50 ft – 100 ft]). Domestic wells ("Domestic") are typically screened in the upper 100 m [330 ft], but can be as deep as 150 m [500 ft], rarely more. Some wells have an actual or estimated record of screen depth and length. Here we selected wells with the top of the screen at less than 60 m [200 ft], not including public supply wells ("<200', priv."); those that include public supply wells, often located in urban areas ("< 200' "); and finally all public supply wells and a few deep irrigation wells in Westlands Water District ("public").

Table 20. Mean and 95% confidence interval, by groundwater basin and well depth category, of the mean of decadal well means of the log of nitrate.[1]

Groundwater Region	Depth	Number of Wells *(# equal areas)*	-95% CI	Mean	+95% CI
TLB Eastside Alluvial Fans	shallow	2,294 (*95*)	13.9 (*11.4*)	14.8 (*14.0*)	15.7 (*17.2*)
TLB Eastside Alluvial Fans	deep	5,804 (*119*)	11.4 (*7.7*)	11.9 (*9.5*)	12.4 (*11.8*)
TLB Central Basins	shallow	317 (*33*)	9.6 (*2.4*)	11.6 (*4.1*)	14.1 (*7.2*)
TLB Central Basins	deep	956 (*46*)	6.4 (*1.7*)	7.3 (*2.9*)	8.3 (*4.7*)
TLB Westside Alluvial Fans	shallow	3 (*3*)	0.1 (*0.1*)	6.2 (*6.2*)	741.9 (*741.9*)
TLB Westside Alluvial Fans	deep	86 (*29*)	3.0 (*2.2*)	4.2 (*4.1*)	5.9 (*7.9*)
Monterey Bay	shallow	462 (*31*)	8.6 (*7.8*)	9.7 (*10.8*)	10.8 (*14.9*)
Monterey Bay	deep	612 (*34*)	6.8 (*4.0*)	7.6 (*5.9*)	8.5 (*8.7*)
Salinas Valley	shallow	106 (*23*)	12.5 (*8.1*)	17.0 (*15.0*)	23.2 (*27.6*)
Salinas Valley	deep	67 (*21*)	5.6 (*5.5*)	8.4 (*7.8*)	12.6 (*11.0*)

[1] In parentheses, for comparison, are the de-clustered data, obtained from the mean of the log-transformed equal area medians, where the medians are obtained from all annual well means. Data shown here are back-transformed from their logarithmic values and shown in [mg/L] nitrate. Includes all wells sampled in the 2000s, except environmental monitoring wells.

5.4 Effect of Soil Type and Vadose Zone Thickness

To examine the effect of soil type and depth to groundwater, we categorize the location of wells by the groundwater protection zones that the Department of Pesticide Regulations (DPR) has designated[10] to protect against groundwater contamination. Groundwater protection zones occur only where depth to groundwater is less than 20 m (70 ft) and where soils are either highly permeable ("Leaching" zones) or where a shallow hardpan occurs ("Runoff" zones).

In the TLB, concentrations are significantly higher within groundwater protection zones than outside of groundwater protection zones (Table 21). Wells in "Runoff" zones and "Runoff or Leaching" zones have significantly higher nitrate than those in "Leaching" zones. In "Runoff" zones of the TLB central basin and the TLB Eastside alluvial fans, more than one third (33%) of wells (not including monitoring wells) exceeds the MCL, while only 12% – 16% of wells outside of DPR groundwater protection zones exceed the MCL in these areas of the TLB (Table 21).

In contrast, fewer than 40 out of over 1,200 wells in the SV are within groundwater protection zones and no significant difference is observed between nitrate in wells located within DPR groundwater protection zones and those outside of those protection zones.

Table 21. Median nitrate concentration and exceedance probability in wells, grouped by Department of Pesticide Regulation (DPR) groundwater protection zone. Includes all wells sampled in the 2000s, except environmental monitoring wells.

Region	DPR Groundwater Protection Zone	# of Wells	# Well-Years	Median	> 9 mg/L	> 22.5 mg/L	> 45 mg/L	> 90 mg/L
TLB Eastside Fans	(Outside)	6,661	17,770	13.3	61.4%	33.9%	15.7%	5.0%
	Leaching	647	2,330	16.6	69.4%	40.4%	14.0%	3.8%
	Runoff	814	2,626	32.3	86.3%	64.0%	35.3%	9.7%
	Runoff or Leaching	40	140	20.9	86.4%	45.7%	21.4%	10.0%
TLB Central Basin	(Outside)	903	2,013	6.0	41.6%	24.3%	11.5%	2.8%
	Leaching	7	23	17.0	82.6%	39.1%	39.1%	17.4%
	Runoff	390	800	33.1	88.0%	60.8%	37.5%	13.4%
	Runoff or Leaching	8	19	50.0	89.5%	57.9%	57.9%	26.3%
TLB Westside Fans	(Outside)	89	201	1.8	29.4%	9.0%	2.0%	1.5%
SV- Pressure Aquifer, Eastside, and Monterey Bay	(Outside)	1091	4,716	14.0	61.2%	35.0%	11.0%	1.6%
	Leaching	21	73	12.0	61.6%	27.4%	9.6%	0.0%
	Runoff	4	15	5.0	6.7%	0.0%	0.0%	0.0%
SV-Forebay and Upper Valley	(Outside)	160	508	10.0	53.5%	34.1%	20.9%	10.6%
	Leaching	15	39	12.0	59.0%	38.5%	20.5%	10.3%

[10] http://www.cdpr.ca.gov/docs/emon/grndwtr/index.htm

Given the small number of wells in DPR protection zones of the SV, statistical significance of DPR zones with respect to nitrate concentration in wells is tested only on the entire datase (wells measured in the 2000s, not including environmental monitoring wells) and not by groundwater region. We also consider wells in “Runoff or Leaching” zones as belonging into the “Runoff” category, since median nitrate values of that small group of wells resemble more closely those in the “Runoff” category than those in the “Leaching” category. Analysis of Variance (ANOVA) tests on the decadal well means of log nitrate show that the DPR zonation (outside, “Leaching,” and “Runoff”), which represents the depth to groundwater and soil type classification, has a statistically highly significant influence on the fate of nitrate in groundwater. The back-transformed mean of decadal well means of log nitrate is 9.8 mg/L (9.4 – 10.1 mg/L) for wells outside of DPR protection zones; 12.1 mg/L (10.8-13.6 mg/L) in “Leaching” zones; and 29.4 mg/L (27.7 – 31.3 mg/L) in wells located in “Runoff” zones or “Runoff or Leaching” zones (parentheses indicate back-transformed 95% confidence intervals of means, Figure 47).

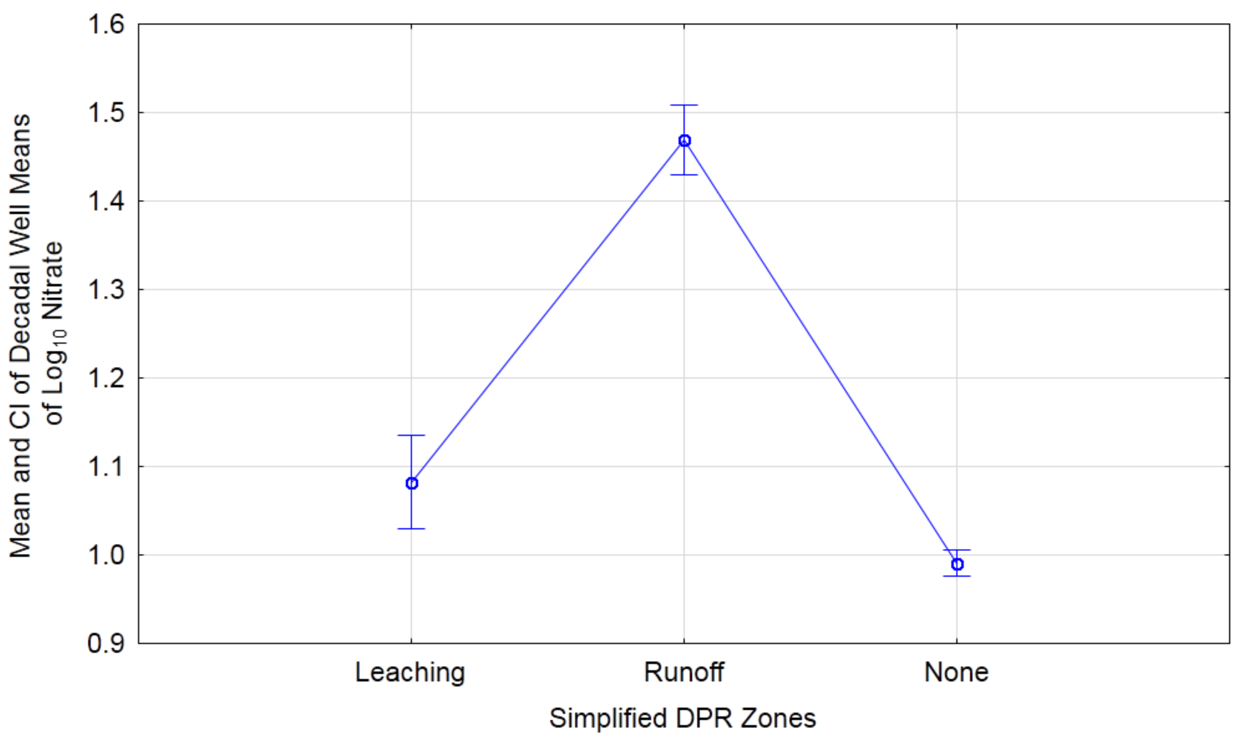

Figure 47. Means and 95% confidence interval of means for DPR groundwater protection zones. Here, “Runoff” refers to wells in either “Runoff” DPR zones or “Runoff or Leaching” DPR zones. Includes all wells sampled in the 2000s, except environmental monitoring wells.

Most “Leaching” zones are located in central Fresno County (south, southwest, and southeast of the city of Fresno) where the dominant crop is vineyards, some are located in the SV. Most “Runoff” zones are located in eastern Fresno County and Tulare County, which have a wide variety of crops. Kings and Kern County have few areas designated as protection zones. Vineyards (40%) and urban areas (13%) make up more than half of the “Leaching” zones, another 20% are deciduous fruit and nut. In contrast less than 20% of “Runoff” and “Runoff or Leaching zones” are vineyard (11%) or urban (8%). It is possible that the relatively lower nitrate concentration in “Leaching” zones, which are dominated by vineyard land use, is due to relatively lower nitrogen use in vineyards, when compared to other crops (see Technical Report 2, Viers et al. 2012). Other potential causes for the difference between these three groups are: differences in temporal (time) delay in the transport of nitrate to groundwater through the vadose zone; different irrigation water use efficiencies due to presence/absence of hardpan or due to soil texture (light soil vs. heavy soils); and differences in denitrification rates between these soils.

5.5 Nitrate Above and Below the Corcoran Clay

The Corcoran Clay underlies the Central Valley from San Joaquin County to Kern County in the western two thirds of the Valley for most of that extent. In the study area, the depth to the Corcoran Clay varies from approximately 10 to 250 meters (32.8 to 820.2 feet) below ground surface. This aquitard unit is between 6 and 55 meters thick. Historically, the Corcoran Clay has been recognized as a regionally extensive barrier that prevents flow between an upper semi-confined aquifer system and a lower confined aquifer system. Recent (>1950's) development of groundwater in the region has likely altered the effect of the barrier due to the many large bore wells that have been drilled through the confining feature to access the lower aquifer (Panert 1995; Bertoldi 1991; Benito 2008). With strong pressure gradients present below and above such confining layers, abandoned and active wells likely function as conduits between groundwater systems which would otherwise be separated (see Viers et al. 2012, Section 9). Flow through the clay aquitard itself is subject to strong denitrification due to the anoxic conditions within the aquitard.

Although development of groundwater may have degraded the ability of the Corcoran Clay to prevent contamination of deeper groundwater from shallow groundwater, we test here at the regional scale if there is a significant difference between wells screened above and below the Corcoran Clay. Of the wells in the study area for which we were able to obtain screened interval values (both top and bottom of screened interval), 825 are screened below the Clay, and 642 are screened above it (Figure 48). CASTING contains 7,358 tests for these wells. Figure 49 shows the distribution of these samples, for wells screened above and below the Corcoran Clay, by decade.

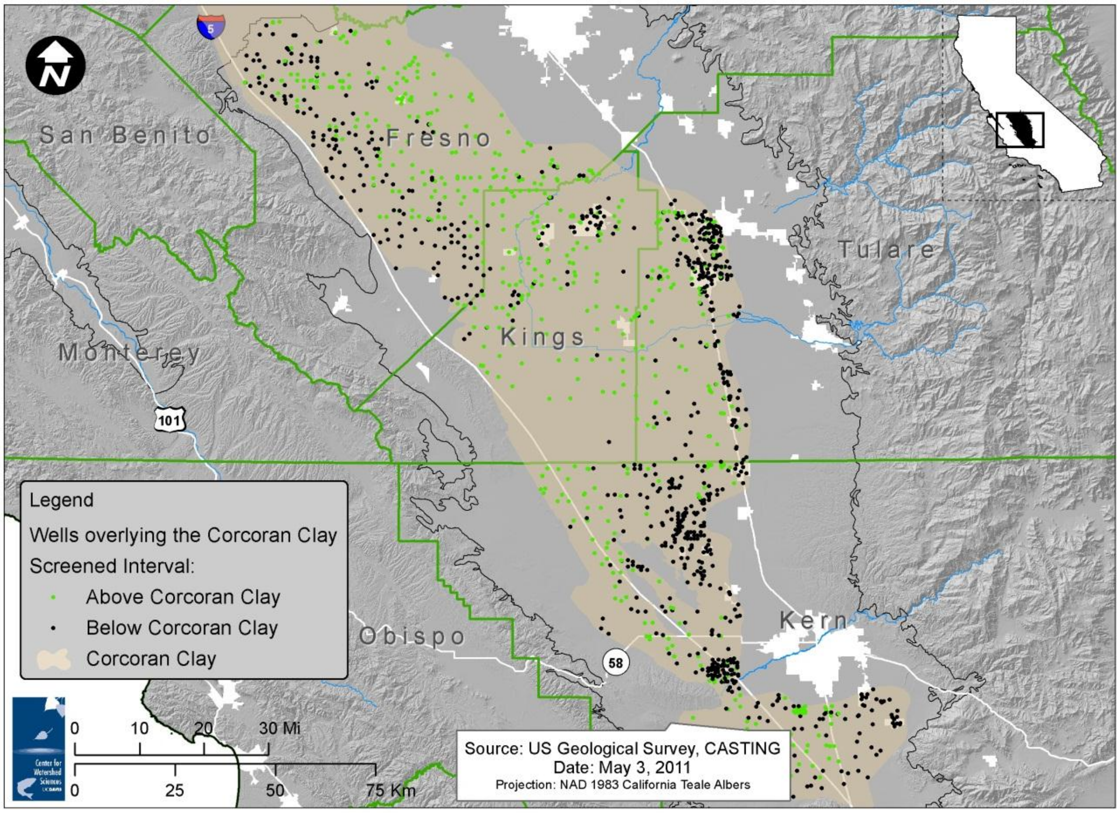

Figure 48. Locations of wells underlain by the Corcoran Clay, and the position of their screens relative to the Clay.

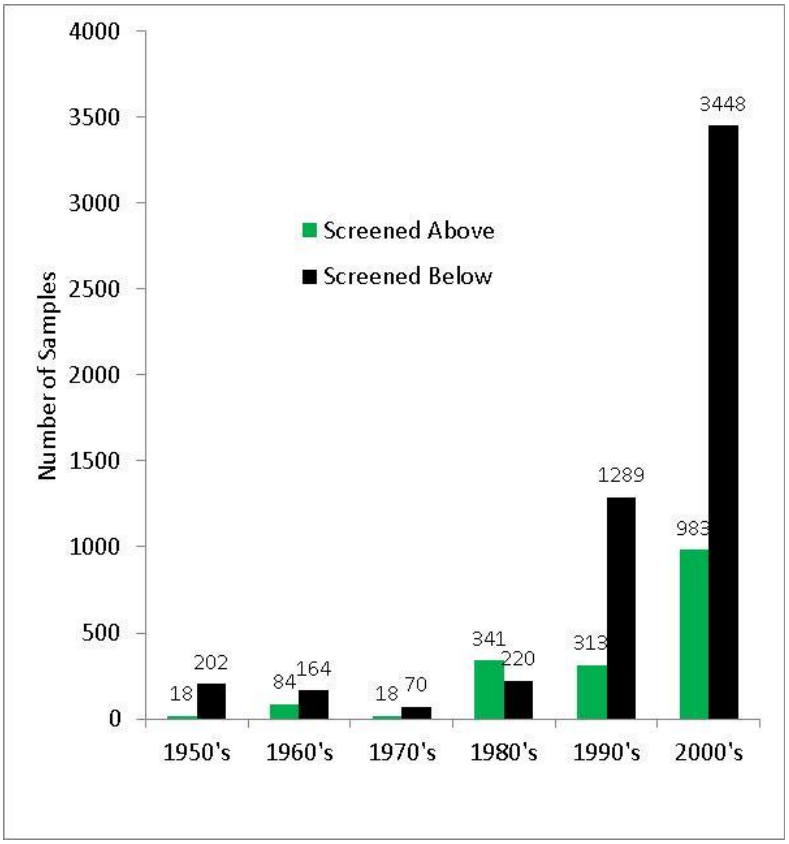

Figure 49. Number of nitrate samples each decade in wells screened above and below the Corcoran Clay.

The decadal median nitrate finding for each of the 1,467 wells with perforated interval information and within the horizontal boundaries of the Corcoran Clay was used for comparison between decades (1950's through 2000's) and between those wells screened above and those screened below the Clay. For each well, the median nitrate finding was found for each decade for which CASTING has nitrate data.

Each well is either screened above or below the Corcoran Clay, therefore direct comparison of the nitrate concentrations above and below the aquitard, in a single borehole, is not possible. To avoid introduction of bias, the area of the Corcoran Clay was divided into equal area hexagons (approximately 21,400 acres, or 8,660 hectares, each), such that the centroid of each is within the outline of the Corcoran Clay, as defined by Claudia Faunt in the Central Valley Hydraulic Model (Faunt 2009). Within each hexagon, for each decade, the median of the decadal median of nitrate concentration in wells above and below the Corcoran Clay was determined. The set of equal area medians above the Corcoran Clay was compared to the same set below, using the Mann-Whitney test for significance at $\alpha = 0.05$ (results in Table 22). This is a non-paired test, meaning that all of the above-clay findings were compared to all of the below-clay findings, including those hexagons that had only one or the other.

Table 22. Median values of the median of decadal well median nitrate concentration (mg/L as nitrate) in hexagonal equal area sample sets for wells screened above and below the Corcoran Clay, by decade. Non-paired comparison.

	Median NO_3 concentration by decade					
Screen Position	**2000's**	**1990's**	**1980's**	**1970's**	**1960's**	**1950's**
Above	7.1	1.4	15.0	6.7	1.2	1.1
Below	2.1	2.6	4.0	2.0	1.5	1.2
Significance at $\alpha = 0.05$	NO	NO	YES	NO	NO	NO

No significant difference between wells screened above and wells screened below the Corcoran Clay was found for any decade except the 1980’s. Although the 1980’s decade did produce a significant result, examination of the source data for that decade showed that the 94 highest nitrate concentration findings were all derived from the NWIS input dataset in the last 4 years of the decade. Each of these 94 findings was the result of a single test on a well, and none of these wells was ever tested before, or since. Unfortunately, NWIS does not provide any information regarding the project objectives for particular data points. As these nitrate concentrations were 1 to 2 orders of magnitude higher than all other nitrate values produced during that decade, excluding known environmental monitoring wells, it is likely that these were samples taken from wells specifically targeting high nitrate concentrations as part of a particular USGS study. Of the 442 samples collected in the 1980’s above or below the Corcoran Clay, 301 were from the NWIS database.

From this non-paired analysis of the entire Corcoran Clay within the study area, we might infer that, at the regional scale, the Corcoran Clay does not separate aquifers of significantly different nitrate concentration. However, a paired analysis reveals more information.

Since some parts of the Corcoran Clay were sampled in some decades and not in others, the non-paired method presents an uncontrolled view in that each sample (hexagon) does not have a relevant comparison sample. Without pairing, the test compares values above the Corcoran Clay in one area to values below the Corcoran Clay in another area, and large-scale regional trends may bias those differences.

To find area-specific differences, those hexagons with values both, above and below the clay in a given decade were considered. Too few data were available from the 2010s to analyze that decade. The following series of figures shows the hexagonal equal area cells used in both the non-paired and paired analyses for the decades from 1950 to present (Figure 50 through Figure 55). Note that there are no cells for which data exist both above and below the Corcoran Clay for all decades.

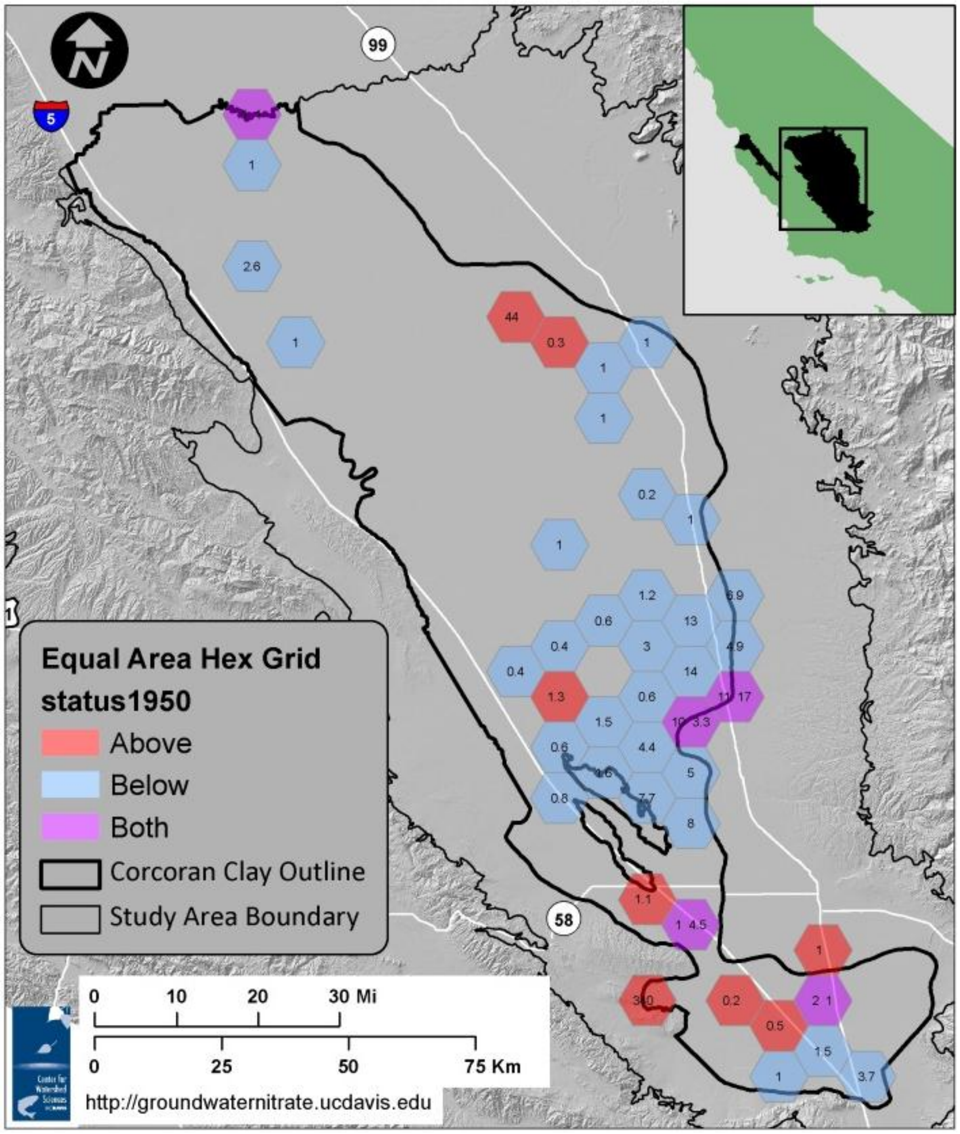

Figure 50. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1950's. Red hexes represent regions with data available from wells screened above the Clay only; blue represents areas with data from below the Clay; purple areas have data from above and below the Clay. The values printed on each hexagon indicate the median of all decadal median nitrate concentrations from the wells in that hexagon – in purple cells, the value on the left is from the above-clay wells and the value on the right is from the below-clay wells.

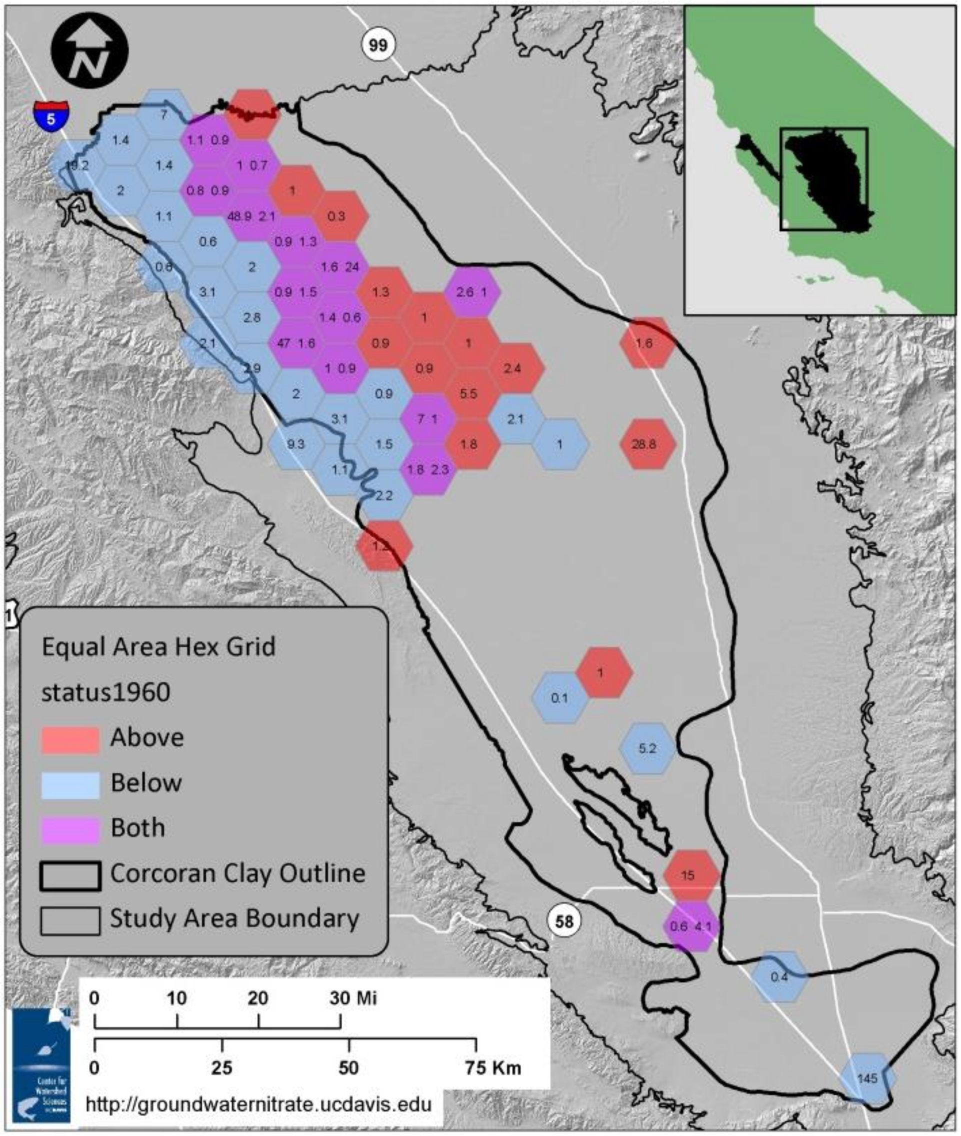

Figure 51. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1960's.

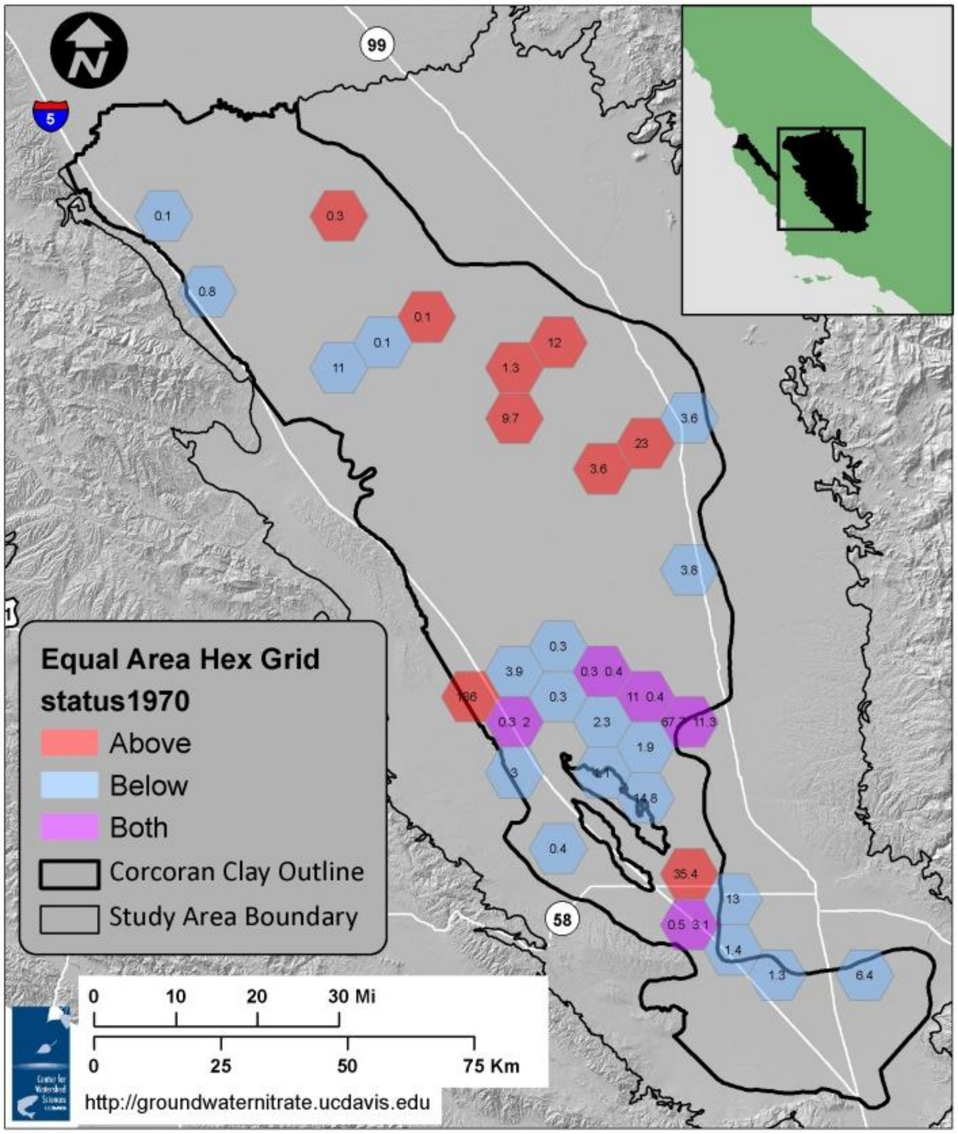

Figure 52. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1970's.

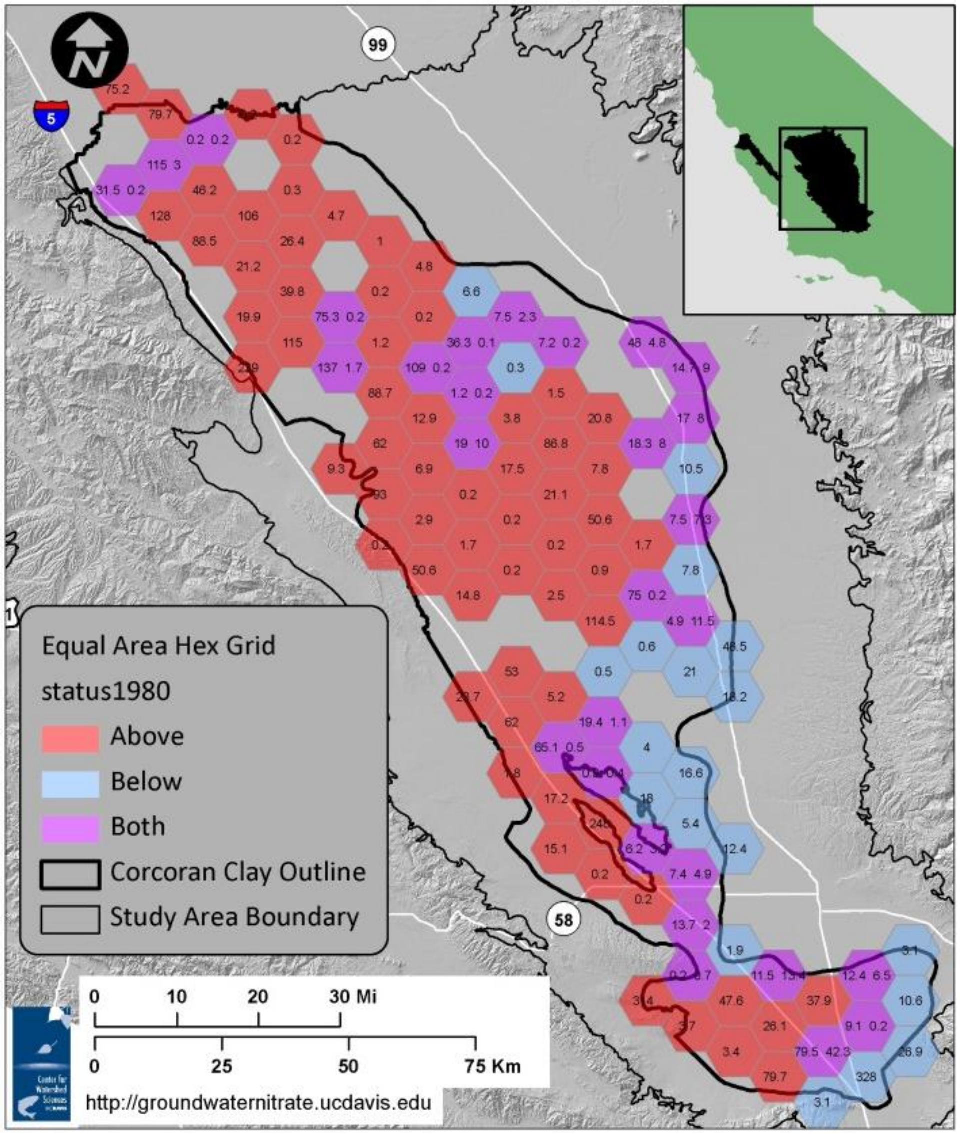

Figure 53. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1980's.

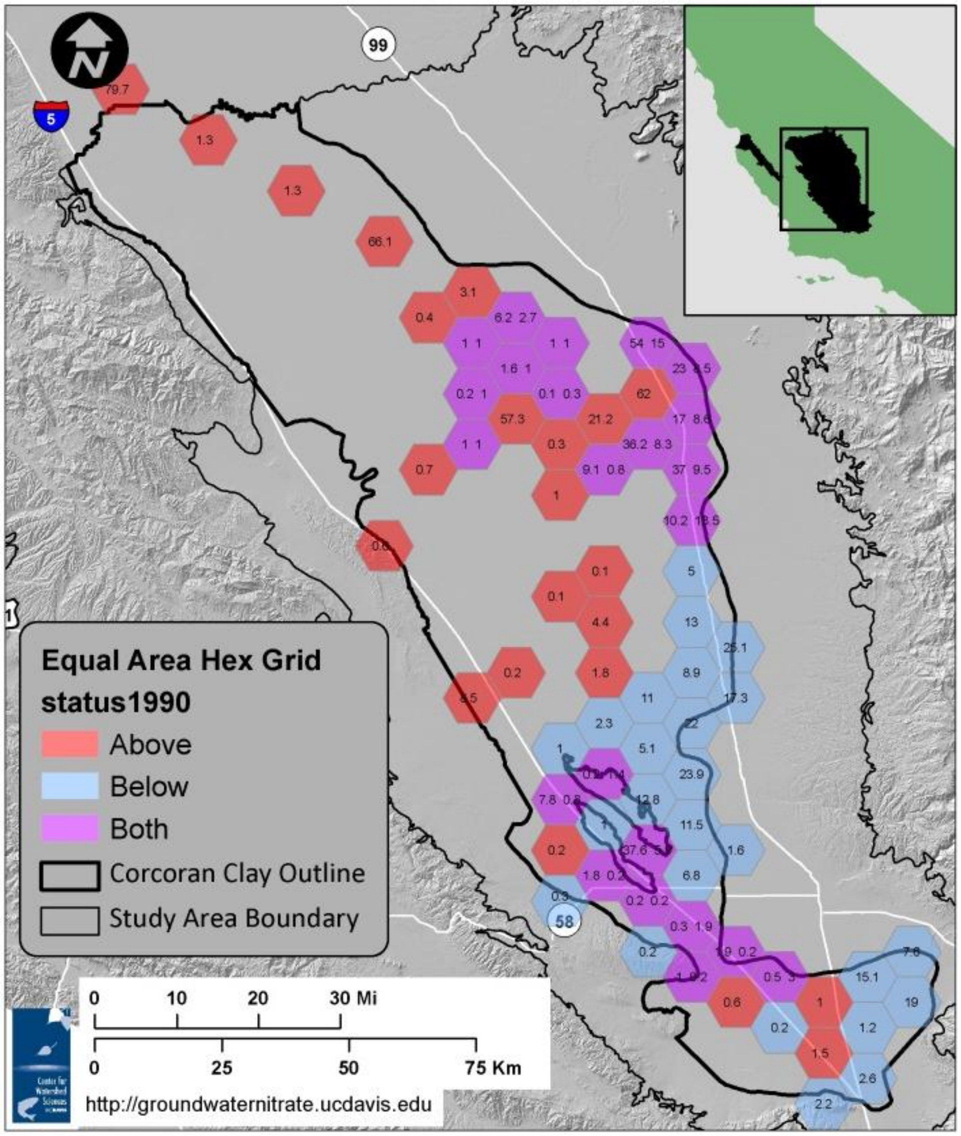

Figure 54. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 1990's.

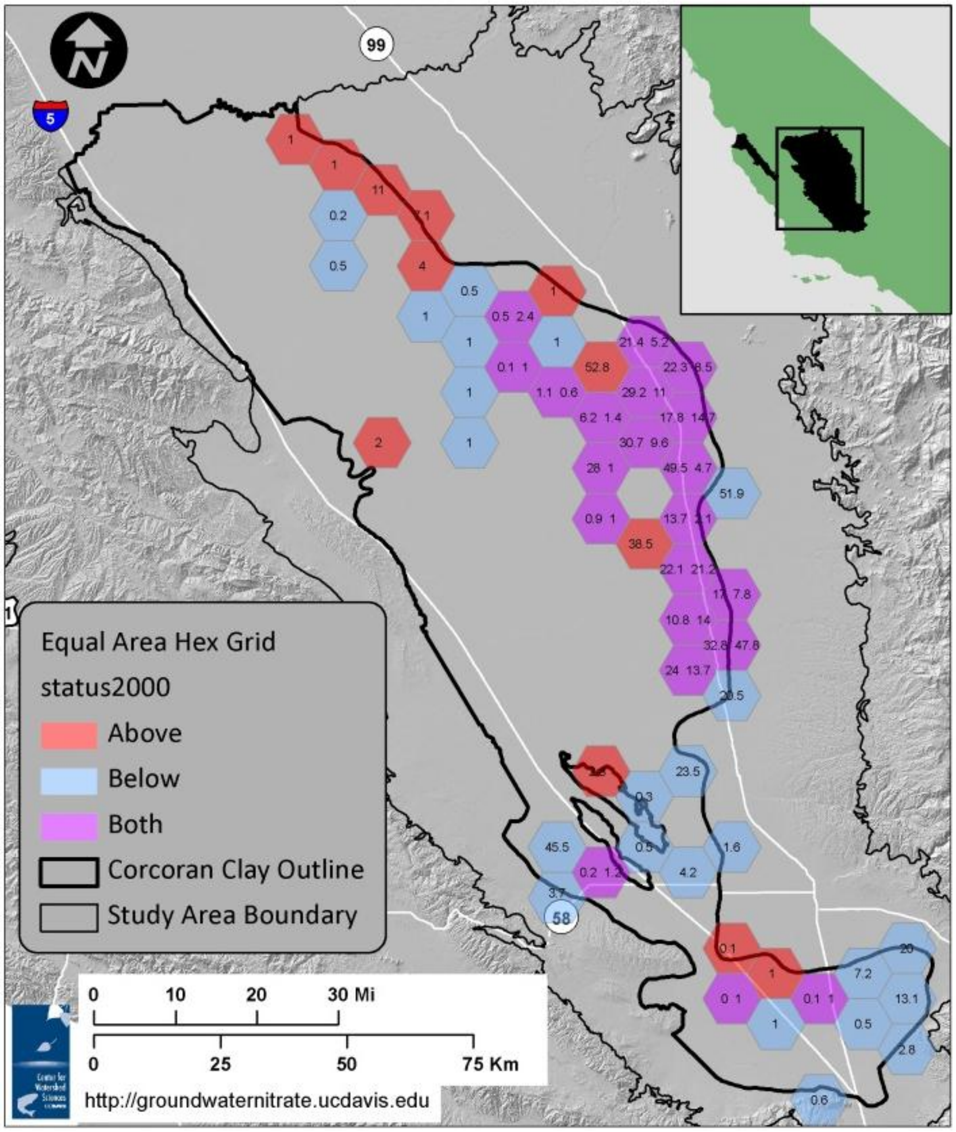

Figure 55. Equal Area Hexagon used in statistical analyses of nitrate concentrations above and below the Corcoran Clay, for the decade of the 2000's.

For hexagon/decade combinations with values both above and below the Corcoran Clay, the below value was subtracted from the above value. A positive result implies that for that decade, in that hexagon, the nitrate concentration was higher above the Corcoran Clay than below it. Median values were used in this analysis as well.

The paired Wilcoxon rank sum test produced very different results from the non-paired analysis. In the 1950's, 1960's, and 1970's, no significant difference was found between the above and below datasets. However, in the 1980's, 1990's and 2000's, the above-clay concentrations were significantly higher than the below-clay data. Not counting the 1980's (due to the bias of the NWIS dataset), the difference in the mean nitrate above and below the Corcoran Clay in the recent two decades has been on the order of 7 – 8 mg/L, nearly twice as much as the difference between "shallow" and "non-shallow" wells found across the study area (see previous section).

Table 23. Results of Paired-Sample Wilcoxon Signed Rank tests.

Decade	p-value	hex count	Above = Below ?	Mean of Above - Below
2000's	0.018	21	Reject	7.5
1990's	0.0076	20	Reject	7.1
1980's	0.000011	29	Reject	27.9
1970's	0.34	5	Do Not Reject	12.5
1960's	0.27	14	Do Not Reject	5.3
1950's	0.5	4	Do Not Reject	-0.4

Based on this analysis, the Corcoran Clay appears to be a significant water quality boundary. Lower nitrate concentration below the Corcoran Clay are the result of multiple factors: groundwater below the Corcoran Clay is significantly older than above the Corcoran Clay, significant portions of the confined aquifer below the Corcoran Clay are geochemically reducing and therefore favorable for in situ denitrification (Burow et al. 1998), and the aquitard represents a significant barrier to downward flow within the aquifer system. On a regional level, this analysis seems to confirm these notions.

On the other hand, a number of wells screened below the Corcoran Clay have nitrate concentrations above typical background levels (more than 9 – 18 mg/L). Also, we caution that a relatively small area of the Corcoran Clay is represented by this analysis because few equal areas, mostly near the eastern boundary of the Corcoran Clay, have sufficient data for a comparison of nitrate values above and below the Corcoran Clay (see Figures above). To the degree that we find nitrate concentrations above background levels in wells screened below the Corcoran Clay, even in the Basin and Westside Alluvial Fans, the dataset indicates that wells may be significant conduits of nitrate, from the upper aquifer system to the confined lower aquifer system below the Corcoran Clay.

5.6 Nitrate in Wells near Streams

In addition to the obvious groundwater quantity benefits, groundwater banking could become an important strategy for improving the regional groundwater quality. Figure 56 shows the average nitrate concentration found in pumping wells by Public Land Survey System (PLSS) section. The circles on the map highlight areas near rivers that have lower nitrate concentrations as compared to wells further away from the rivers. It is likely that these wells have relatively low nitrate due to the fact that the water they are withdrawing consists of a high fraction of uncontaminated river recharge, and a small amount of contaminated recharge via irrigation return water. Additionally, areas near rivers are often irrigated with low nitrate surface water. The resulting leachate from these crops is potentially less than that from crops grown further away from rivers as those fields are often irrigated with pumped groundwater containing elevated levels of nitrate.

Within the city of Fresno, south of the San Joaquin River, large amounts of surface water and storm water are intentionally recharged. Wells that are further from major rivers generally appear to have higher nitrate concentrations. The water withdrawn from these wells likely has a higher fraction of contaminated irrigation return water. At the basin scale, managing recharge by increasing the amount of uncontaminated recharge versus contaminated recharge through artificial recharge basins, as well as managed reservoir releases to maintain stream flow (and therefore stream recharge), has the potential to positively affect the long term trends in water quality within the TLB.

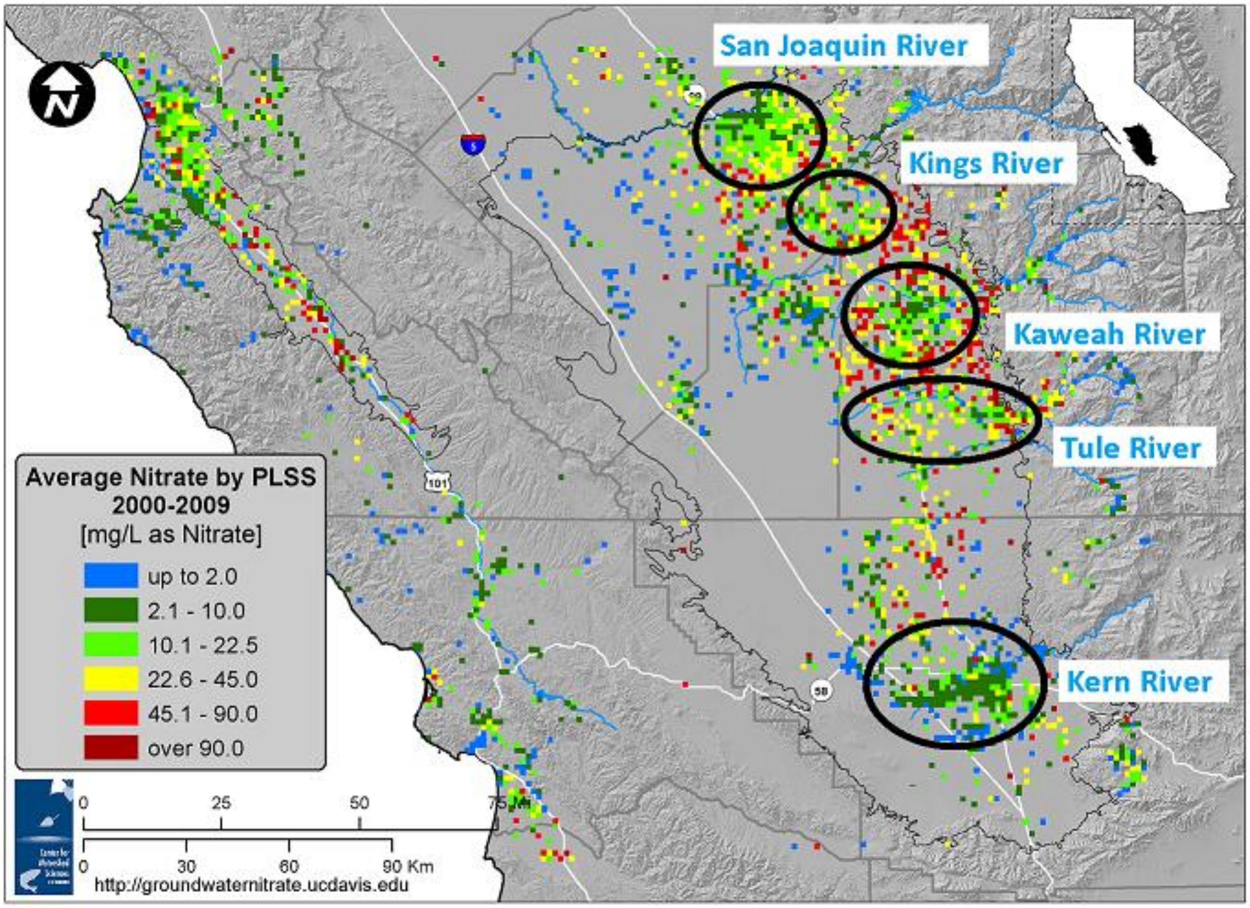

Figure 56. Average nitrate concentration by PLSS section. Circles highlight areas where major rivers enter the study area. These are also areas characterized by wells low in nitrate.

5.7 Historic Trends

The CASTING database represents a large dataset of primarily the current period (since 1/1/2000) with wide regional representation across the study area. CASTING also represents a view back in time. In the study area, only a few tens to a few hundred wells were tested each year for nitrate during much of the 1950s, 1960s, 1970s, and 1980s (Figure 57). Many of these wells, especially after the 1960s, represent testing in public water supply wells. In the late 1980s, the number of wells tested annually increased to over 500, exceeding 1,000 wells in the early 1990s, and exceeding 2,000 wells by the early 2000s. In 2007 – 2010, the dairy regulatory program in the TLB added about 4,000 wells, thereby more than doubling the number of wells tested annually (Figure 57). Over the next decade and beyond, the dairy dataset will provide an opportunity to obtain a much better understanding of groundwater nitrate and long-term trends in agricultural regions of the San Joaquin Valley and TLB, if implemented properly. Currently missing information includes consistent sampling protocols, collection of well construction information (particularly screen location), and consistent data management.

For the period from the 1950s to current, public water supply wells are the only set of wells with long-term records of groundwater nitrate, although those wells exceeding the MCL will often be abandoned or destroyed, which potentially introduces a statistical bias into a trend analysis. Apart from public supply wells (and the recent dairy well program), only the DPR dataset on 71 domestic wells in Fresno and Tulare County, operated for the last decade, has consistently sampled from the same set of wells. Many wells have been sampled once or few times, not allowing for long-term trend analysis. Monterey County Water Resouces Agency administers a confidential set of private well samples monitored for nitrate over the past two decades. Only selected published records, funded from public grants, have been available for this study.

First, we consider the long-term changes in exceedance probabilities across the entire set of wells sampled each year, whether these are sampled once or multiple times. Figure 57 shows that the fraction of well samples exceeding the background threshold level of 9 mg/L has steadily increased over the past sixty years, albeit with significant interannual variations. Exceedance rates have increased from about one-third of wells in the 1950s to more than two-thirds of wells exceeding the background level in the late 2000s. In the late 1970s and 1980s, the exceedance rate seemed to hold steady at about 55%. That was also during a period with relatively few samples taken (Figure 57).

The half-MCL (22.5 mg/L) threshold exceedance rate increased from near 10% in the 1950s to about 30% in the early 1980s, with a spike in the late 1960s. During the 1980s and 1990s, the exceedance rate held steady, but began to increase again in the late 1990s, with a large jump in the late 2000s, when the exceedance rate increased to 45%. On the other hand, the exceedance rate of the MCL has held relatively steady ranging from 5% to 15% over the second half of the 20th century, but increasing significantly since the late 1990s.

The uneven distribution of samples across space, time, and data sources (monitoring purposes), and the relatively low number of samples prior to 1990, make it very difficult to obtain a highly accurate estimate of long-term trends in the study area that also accounts for differences due to variation in land use, soils, vadose zone thickness, well depth and aquifer stratigraphy, and other factors potentially influencing nitrate concentrations in wells. Furthermore, most of the samples taken prior to 2000 are focused on urban and peri-urban areas of the study area. Any analysis of long-term trends must therefore be carefully conducted and interpreted. Here we perform a number of additional tests to further elucidate and interpret apparent trends seen in Figure 57.

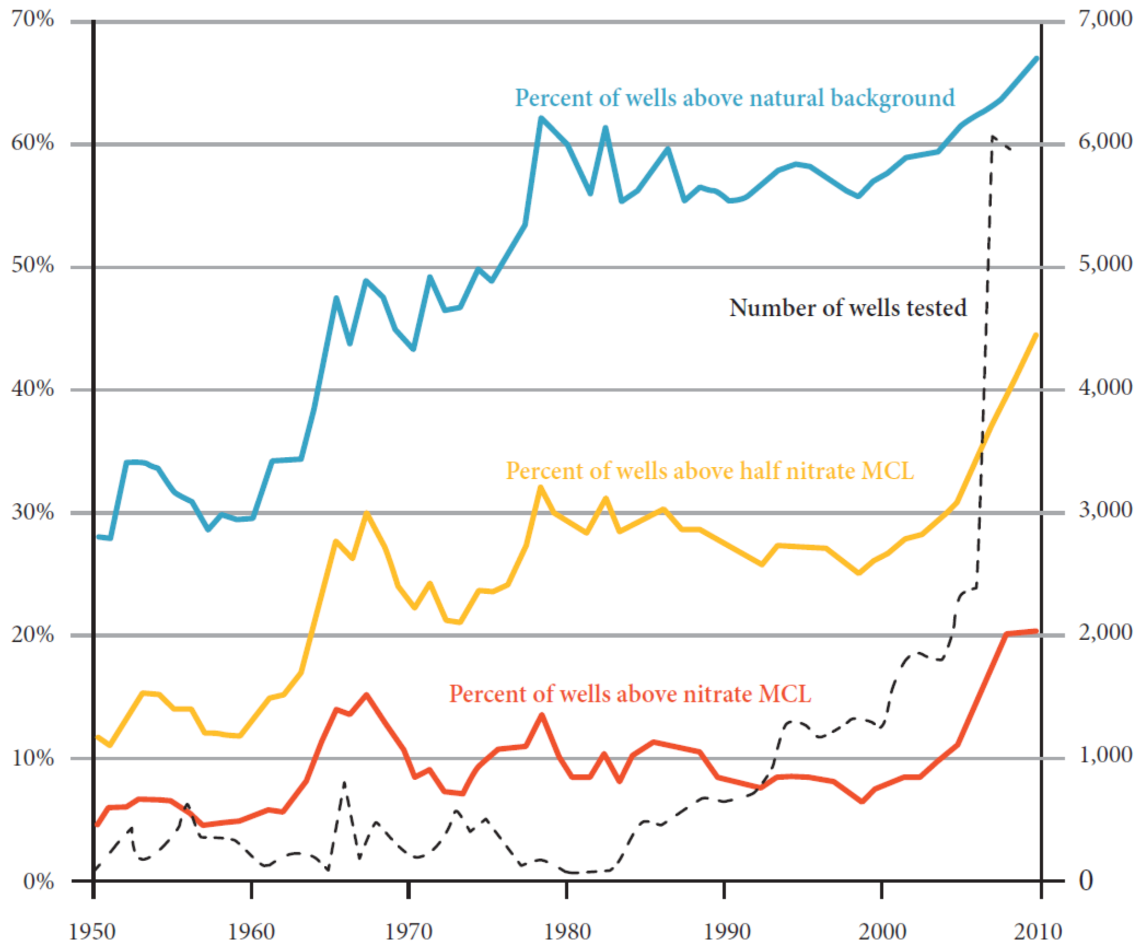

Figure 57. Five-year moving average of the percentage of wells for which the average annual measured concentration exceeded 9 mg/L (background), 22.5 mg/L (half of the MCL), and 45 mg/L (MCL) in any given year. Since the 1990s, an increasing number of wells other than public supply wells have been tested. In 2007, Central Valley dairies began testing their domestic and irrigation wells on an annual basis.

For an investigation of historical trends, we again consider the five physiographic groundwater regions of the study area described above and use de-clustered nitrate values to illustrate the development of nitrate concentration distribution over the past sixty years. We first consider the median and exceedance probabilities of annual well means in each equal area and each decade, not including environmental monitoring wells (see Section 5.1 for a description of equal areas). The medians are log-normally distributed across equal areas. We calculated the mean and confidence interval of the log of medians in each of the five groundwater regions, for each decade (Figure 58). We computed the mean across all equal areas of each equal area's exceedance probability, based on annual well means (Figure 59) and also computed 95% confidence intervals of these means (Figures 58, 59).

Increases in mean nitrate concentrations are observed mostly in the Eastside Alluvial Fans region of the TLB and the Monterey Bay region of the Salinas Valley (Figure 58). Between the 1950s and the 2000s, with all areas equally weighted, mean nitrate levels have increased about 7 mg/L. The 1980s saw significant spikes in mean nitrate in the TLB Basins, TLB Westside Alluvial Fans, and the Salinas Valley regions. However, the number of samples taken during the 1980s was relatively low, and the confidence interval is wide. In the TLB, much of the high nitrate samples collected in the 1980s were associated with a sampling program archived in the USGS NWIS dataset.

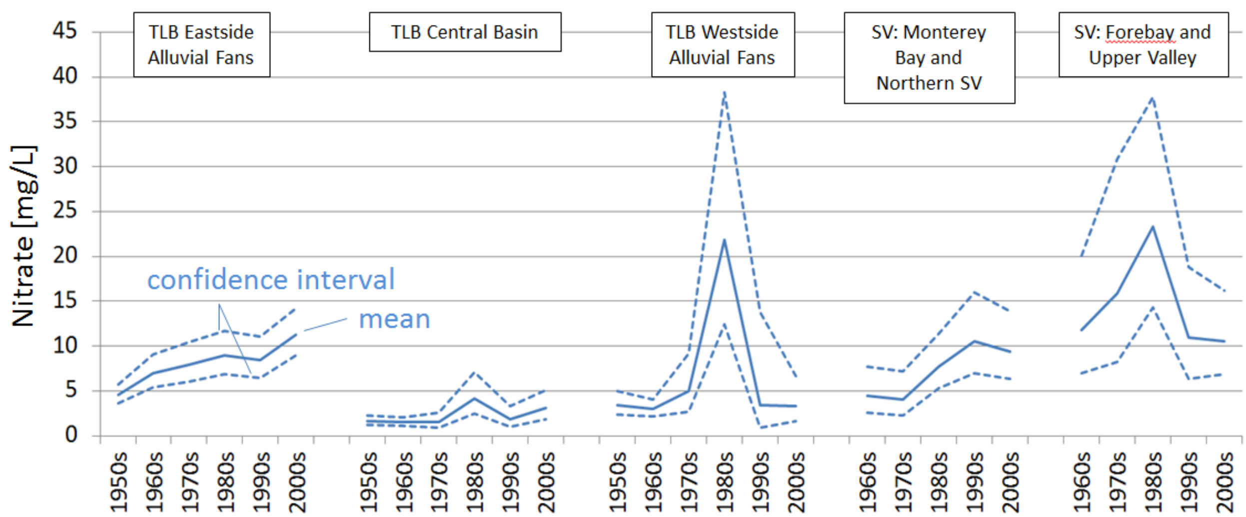

Figure 58. De-clustered, back-transformed mean of the logarithm of equal area decadal medians that were computed from ten years of annual well means. From left to right: TLB Eastside Alluvial Fans, TLB Central Basin, TLB Westside Alluvial Fans, Northern SV and Monterey Bay, SV Forebay and Upper Valley. Confidence intervals of the mean are at the 95% level (back-transformed).

Exceedance probabilities for all three thresholds considered (9 mg/L, 22.5 mg/L, and the MCL at 45 mg/L) show long-term increases in the TLB Eastside, the TLB Basin, and the Monterey Bay and Northern Salinas Valley groundwater regions. Again, the spike of high nitrate samples collected in the 1980s is apparent, if not as strong as in the medians (Figure 59). The decadal changes seem to indicate a decrease in exceedance probabilities between the 1990s and 2000s for both Monterey County regions (Monterey Bay and Salinas Valley, Figure 59). A more detailed look at annual (rather than decadal) medians in each equal area of those two regions (approximately log-normal distributed) reveals that the decrease is mostly limited to the mid-1990s, when sample size was relatively small (and confidence intervals are wide). During the 2000s, no significant decrease in median nitrate concentration was observed in the two Monterey County regions (Figure 60) with a potential upward trend in the past five years in both regions.

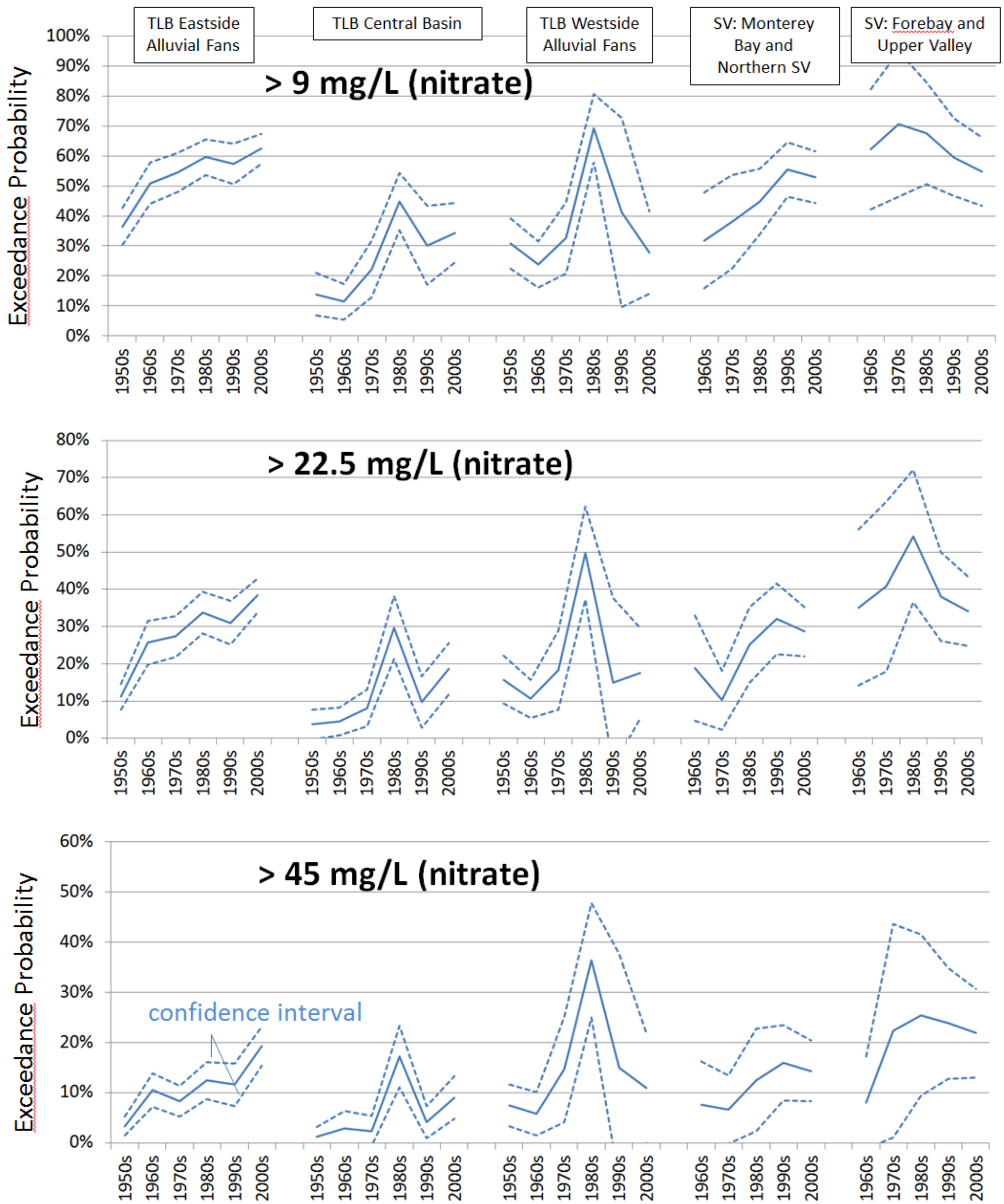

Figure 59. De-clustered mean of the equal area decadal exceedance probablility computed from ten years worth of annual well means in each equal area. From left to right: TLB Eastside Alluvial Fans, TLB Central Basin, TLB Westside Alluvial Fans, Northern SV and Monterey Bay, SV Forebay and Upper Valley. From top to bottom: exceedance probability of nitrate above 9 mg/L (top), nitrate above 22.5 mg/L (center), and nitrate above the MCL of 45 mg/L (bottom). Confidence intervals of the mean are at the 95% level.

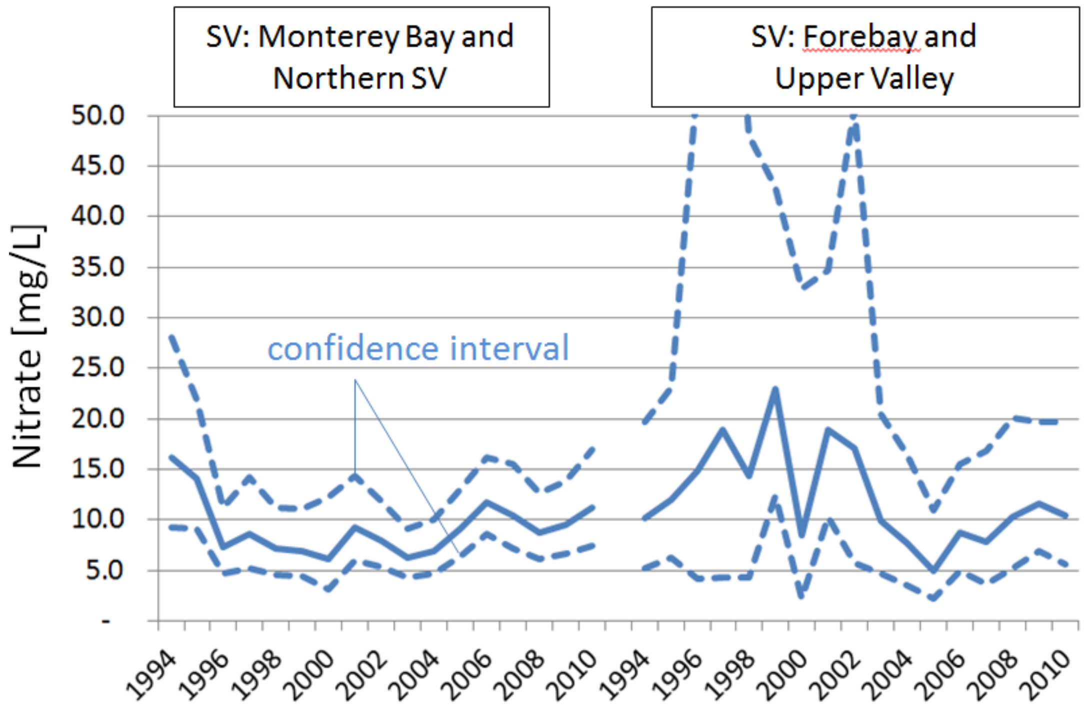

Figure 60. De-clustered back-transformed annual mean of the logarithm of equal area annual medians that were computed for one year of annual means of each measured well. Monterey Bay and Northern Salinas Valley (left), Salinas Valley Forebay and Upper Valley (right). Confidence intervals of the mean are at the 95% level (back-transformed).

To quantitatively assess the changes in nitrate levels through time, a regional statistical test called the "Regional Kendall" (RK) test (Helsel & Frans 2006) was used to test for long-term nitrate concentration trends in the study area. The RK tests whether a regional (or temporal) trend is present in a dataset, and whether that trend is significant or not. In the RK test, the tau value is a correlation coefficient that ranges from -1 to +1. A value of +1 or -1 would indicate a perfect positive or negative correlation. A zero would signify no correlation. The p-value represents the probability that the null hypothesis is true, where the null hypothesis that there is no significant trend in the data. Typically a p value of 0.01 or 0.05 (signifying 99% confidence and 95% confidence) are used as a threshold to conclude whether or not the trend is significant. For this trend analysis, a 99% confidence threshold was used to signify a significant trend ($p \leq 0.01$). The RK test operates on concentration differences between all pairs of nitrate samples taken over time from the same well (or equal area), for wells (or equal areas) with at least two measurements (e.g., each year, each decade, etc.). The test is based on the counts of sample pairs with increasing concentration trend and on the counts of pairs with decreasing concentration trend over time.

To test for trend, the Regional Kendall test was performed on the yearly medians of nitrate for the equal area cells (Helsel & Frans 2006). First, the median value of all tests within a single year for a given well was found. Next, the median value of all wells for each year, for each equal area cell was found. The result is a time series of nitrate data for each equal area cell, where each cell has at most one value per year (if no data were available for a particular year within a cell it would have no value for that year). Given the uneven distribution of samples over time, the dataset was broken into two time periods: 1949–1999 and 2000–2011. The analysis on the first time period represents historical trends while the analysis on the second time period represents recent trends. The results of the RK test are shown in Table 24.

Three tests resulted in a p-value of greater than 0.01 (no significant trend): the TLB Westside Alluvial Fans area for the years 2000–2011, and for both periods of the "Salinas Valley" region of the SV. All other regions and time periods resulted in positive trends that were statistically significant. All significant trends show that nitrate is increasing, with some areas increasing faster than others. For the TLB, the Eastside Alluvial Fans region was found to be increasing the fastest. The granitic and generally coarse sediment coming from the Sierra Nevada Mountains provides for rapid transport of water applied at the surface to the groundwater below. The TLB Westside Alluvial Fans, and the TLB Basin sediments are characteristically much finer compared to the TLB Eastside Alluvial Fans sediments, and we see slower rates of nitrate increases in these regions.

The greatest trends for the study area were observed in the Monterey Bay and TLB Eastside Fans regions. From 2000–2011, the RK test shows that the median values of the equal area cells were increasing by 0.4 (mg/L)/year in the Monterey Bay region, and 0.19 (mg/L)/year in the TLB Eastside Fans region. The Salinas Valley sub-region (Forebay and Upper Valley subbasins) did not result in any significant trend. River recharge is greatest in this portion of the SV. Aquifer conditions also range from semi-confined to unconfined aquifer conditions. This setting likely creates a very heterogeneous system which might obfuscate a trend from showing in the analysis.

The values shown are also very small in magnitude. It is important to remember that the test was performed on the median values of a dataset consisting of mostly drinking water wells. The test is insensitive to outliers (e.g., contaminated wells) and is therefore more a measure of the general water quality of the region than any trend observed in individual wells. A positive trend documents that nitrate concentrations for the region as a whole are increasing, and that nitrate contamination of wells is more of a regional process rather than a localized phenomenon. The RK test results are in agreement with the more qualitative observations made on the long-term changes in nitrate concentration shown in Figure 58, Figure 59, and Figure 60. The magnitude of the upward trend is also consistent with the observed changes in mean nitrate, particularly in the TLB Eastside Alluvial Fans region and the Monterey Bay and Northern Salinas Valley region.

Table 24. Regional Kendall test results on the de-clustered equal area statistics for nitrate. Trends shown in blue are statistically not significant.

	Tau	P	Annual Change in NO_3^-
TLB Eastside Fans, 2000-2011	0.148	0.000	0.1900
TLB Eastside Fans, 1949-1999	0.136	0.0000	0.07561
TLB Basin, 2000-2010	0.141	0.0011	0.01214
TLB Basin, 1950-1999	0.117	0.0000	0.02146
TLB Westside Fans, 2000-2010	0.089	0.3496	0.000
TLB Westside Fans, 1949-1999	0.097	0.0029	0.03984
Monterey Bay, 2000-2011	0.175	0.0011	0.4000
Monterey Bay, 1962-1999	0.163	0.0001	0.1164
Salinas Valley, 2000-2011	-0.143	0.1472	-0.8750
Salinas Valley, 1962-1999	0.097	0.1392	0.1750

To further illustrate the long-term trend of nitrate values in wells, we pursue an investigation of the statistics of the average trend in nitrate values at each well that has at least two measurement data. We investigate the average trend at each well over the entire period of observation and, separately, for each decade of observation. To obtain an average trend of nitrate concentration at a well, we performed a simple linear regression between measured nitrate values and time for the entire period over which a well was sampled and, separately, for each decade at which a well was sampled at least twice. The regression was performed using a least-square estimation procedure to obtain the regression slope. The regression slope was used as a measure of the average trend in nitrate at a given well over the time period of interest. The regression slope expresses average nitrate changes in a well in units of concentration change per year [mg/L/yr]. Here, we refer to the regression slope at a well as the "well trend" or the "decadal well trend," if the regression slope was performed for a specific decade.

Well trends and decadal well trends are distributed nearly symmetrically around zero with a large number of samples showing very small or zero gradients. The number of non-zero well trends (positive or negative) decreases hyper-exponentially as the absolute value of the well trend (or decadal well trend) increases. We computed the mean and its 95% confidence interval, the median, and the lower and upper quartiles of the well trends across all wells within each groundwater region (Table 25). Data were analyzed separately for the major groundwater regions. Well trends and decadal well trends larger than 20 mg/L/yr or smaller than 20 mg/L/yr were considered outliers and not included in the analysis. Results of the statistical analysis are shown in Table 25.

The mean of well slopes over the entire period of measurement is positive in all five regions. The lower confidence interval for the mean slope in all five regions is also larger than zero indicating that the mean is significantly different from zero. This confirms the results of the RK test and would suggest that even the Salinas Valley (no significant trend according to the RK test), over the entire period of measurement, had an increasing trend in nitrate concentrations.

For the study area as a whole, the average well trend is 0.34 mg/L/yr. Half of all wells have a trend larger than 0.08 mg/L and 25% of all wells have a long-term trend of more than 1.11 mg/L/yr.

In the TLB, the Eastside Alluvial Fans region and the Basins region have the fastest increasing well trend. The mean in the Eastside Fans area is at 0.3 mg/L/yr; there, half of all wells increase at a rate higher than 0.08 mg/L/yr and one-quarter of all wells increase at a rate higher than 1.25 mg/L/yr. In the Basins region, the mean well trend is 0.45 mg/L/yr (median: 0.03 mg/L/yr, upper quartile: 1.22 mg/L/yr). Lower rates of increase are found for the Basins regions, where the mean well trend is 0.16 mg/L/yr (median: 0 mg/L/yr). A quarter of wells there exceed a rate of increase of 0.5 mg/L/yr.

Table 25. Statistics of the least-square estimates of the regression slope (mg/L per year) of nitrate concentrations versus time for each well with at least two sample values, grouped by groundwater region and decade.[1]

GW Region	Period	Number of Slopes	CI -95%	Mean Slope	CI +95%	Median Slope	Lower Quartile	Upper Quartile
Study Area	**total period**	7,694	0.23	0.34	0.44	0.08	-0.40	1.11
TLB Eastside Fans	**total period**	5,128	0.15	0.30	0.44	0.09	-0.48	1.25
	1950s	226	-0.74	-0.15	0.43	0.00	-0.49	0.78
	1960s	190	-0.72	0.05	0.83	0.01	-1.36	2.12
	1970s	366	0.01	0.33	0.66	0.13	-0.46	1.36
	1980s	521	-0.48	-0.14	0.20	0.00	-1.13	1.04
	1990s	1,142	-0.16	-0.01	0.14	0.00	-0.59	0.49
	2000s	3,756	0.14	0.32	0.51	0.09	-0.79	1.61
TLB Basin	**total period**	850	0.08	0.45	0.81	0.03	-0.29	1.22
	1950s	92	-0.50	0.08	0.65	0.00	-0.29	0.23
	1960s	63	-1.15	-0.50	0.15	0.00	-0.58	0.40
	1970s	33	-1.99	-0.14	1.71	0.00	-0.07	0.61
	1980s	31	-1.70	-0.40	0.90	0.08	-0.25	0.59
	1990s	81	-0.56	0.13	0.83	0.02	-0.19	0.50
	2000s	575	0.28	0.80	1.31	0.22	-0.73	2.74
TLB Westside Fans	**total period**	298	-0.33	0.16	0.65	0.00	-0.59	0.50
	1950s	140	-1.30	-0.46	0.37	0.00	-1.16	0.45
	1960s	57	-0.97	0.16	1.29	0.08	-0.40	0.75
	1970s	9	-0.67	-0.06	0.56	-0.39	-0.70	0.24
	1980s	11	-1.77	2.03	5.83	0.60	0.00	5.86
	1990s	8	-1.81	-0.31	1.18	0.03	-1.31	0.94
	2000s	42	-2.13	-0.14	1.84	-0.11	-2.23	1.21
Monterey Bay, Pressure Aquifer, and SV Eastside	**total period**	1,185	0.07	0.25	0.42	0.07	-0.22	0.72
	1960s	39	-0.41	0.44	1.29	0.19	0.02	0.63
	1980s	35	-1.46	0.58	2.62	0.00	-0.94	0.99
	1990s	220	-0.99	-0.32	0.35	0.00	-2.45	1.35
	2000s	556	0.16	0.44	0.71	0.13	-0.21	0.95
Salinas Valley, Forebay and Upper Valley	**total period**	1,024	0.13	0.29	0.45	0.04	-0.26	0.79
	1990s	233	0.85	1.50	2.14	0.54	-0.22	3.78
	2000s	20	-0.24	0.76	1.76	0.45	-0.07	1.56

[1] Regression slopes for each well are computed for the entire period of records (“total period”) and, separately, for each decade with at least two measurements on one well. Environmental monitoring wells are excluded.

In the SV, the Monterey Bay region experiences long-term nitrate increases similar to those in the eastern TLB, as expected from the RK test and the change in average regional nitrate levels, discussed above. The mean well trend is 0.25 mg/L/yr with half of all wells exceeding 0.07 mg/L/yr and one quarter of all wells increasing at a rate of 0.72 mg/L/yr or more. In contrast, the Forebay and Upper Valley subbasins of the SV have a mean rate of nitrate increase of 1.5 mg/L/yr, half of all wells increase at a rate of 0.54 mg/L/yr or more, one quarter of all well trends are 3.78 mg/L/yr or more.

Table 25 also lists the mean, medians, and quartiles of the decadal well trends, within each region. When analyzed by groundwater region and decade, half of all wells or more have increasing decadal well trends (median is zero or greater than zero), except in few cases (TLB Westside in the 1970s and 2000s, and Salinas Valley region in the 1970s).

Specifically for the most recent decade (the 2000s), mean decadal well trends were 0.3, 0.8, -0.14, 0.29, and -0.36 mg/L/yr for the TLB Eastside, TLB Basin, TLB Westside, Monterey Bay, and Salinas Valley regions. The two negative trends are associated with the by far smallest sample sets (42 wells in the TLB Westside region and 116 wells in the Salinas Valley) when compared to the other three regions. The decadal well trends are consistent with the RK test and with the changes in average nitrate across wells observed in these regions (Figure 58 and Figure 59). In the five regions, for the last decade, half of all wells (median) had decadal well trends of 0.09, 0.22, -0.11, 0.04, and 0.00 mg/L/yr, or more, respectively. One quarter of all wells (upper quartile) in these five regions had nitrate concentration increases at a rate of 1.61, 2.74, 1.21, 0.79, and 0.83 mg/L/yr or more, respectively.

The study area has four major continuous nitrate monitoring programs: public supply wells (since the 1980s); a private and confidential monitoring program in the SV administered by the Monterey County Water Resources Agency (MCWRA) since the 1990s; the Department of Pesticide Regulations (DPR) domestic well monitoring program in groundwater protection zones of Fresno and Tulare Counties; and – most recently – the dairy monitoring program for which CASTING contains the 2007 through 2009 records.

For comparison, we performed the well trend and decadal well trend analysis separately for these datasets (Table 26). The number of public supply wells with multiple data steadily increased from less than 600 in the 1980s to 1,300 in the 1990s and nearly 2,000 in the last decade. The number of wells monitored in the other programs is relatively steady over time, although we only had access to some published MCWRA records. The long-term mean well trend in public water supply wells, in the MCWRA monitoring wells, and (albeit over only 3 years) in the Dairy Program wells was comparable at 0.27, 0.41, and 0.45 mg/L/yr, respectively. Within each monitoring program, half of all wells (median) had trends of 0.08, 0.07, and 0.11 mg/L/yr or more, respectively; and one-quarter of wells (upper quartile) increased at a rate of 0.54, 3.32, and 3.78 mg/L/yr or more. The mean upward trend is significantly different from zero for the public supply wells and for the dairy wells, but not for the MCWRA monitoring well program, which is only about one-tenth in size when compared to either the public water supply well or dairy well sampling program.

The 69 domestic wells in the DPR program, on average, have a decreasing well trend of -0.27 mg/L/yr. More than half of the wells (median) have a decreasing trend (the median is -0.32 mg/L/yr). One quarter of all wells (lower quartile) decrease at rates of -1.46 mg/L/yr or faster, while one quarter of wells (upper quartile) increase at rates of 1.08 mg/L/yr or faster. About half of these wells are located in "runoff" protection zones, while the other half is located in "leaching" protection zones. Well trends in both zones cover a similar range. Hence, the type of protection zone does not appear to affect the rate of change in nitrate concentration.

Table 26. Statistics of the least-square estimates of the regression slope (mg/L per year) of nitrate concentrations versus time for each well with at least two sample values, grouped by data collection programs and decade.[1]

Data Source / Collection Program	Period	Number of Slopes	CI -95%	Mean Slope	CI +95%	Median Slope	Lower Quartile	Upper Quartile
Public Supply Wells (CADWSAP)	**total period**	2,190	0.18	0.27	0.35	0.08	-0.14	0.54
	1980s	558	-0.20	0.11	0.41	0.02	-0.92	1.24
	1990s	1,311	-0.10	0.04	0.18	0.00	-0.56	0.50
	2000s	1,927	0.21	0.31	0.41	0.11	-0.18	0.66
MCWRA Monitoring Program	**total period**	206	-0.57	0.41	1.38	0.07	-2.67	3.32
	1990s	171	-0.06	1.12	2.30	0.55	-2.12	5.14
	2000s	59	-2.62	-0.64	1.35	0.00	-5.89	2.77
DPR Domestic Wells – all	2000s	69	-0.91	-0.27	0.38	-0.32	-1.46	1.08
DPR - Leaching zones only	2000s	30	-1.33	-0.20	0.92	-0.47	-1.17	1.08
DPR - Runoff zones only	2000s	39	-1.12	-0.32	0.48	-0.31	-1.49	1.12
Dairy General Order	2000s	2,600	0.17	0.45	0.73	0.11	-2.35	3.78

[1] Regression slopes for each well are computed for the entire period of records ("total period") and, separately, for each decade with at least two measurements on one well. Environmental monitoring wells are excluded.

In summary, on average across the study area, groundwater nitrate concentrations are increasing at rates of about 1 to 10 mg/L per decade, particularly in the eastern TLB and in the northern Salinas Valley. The RK test provides the statistically most robust measure of groundwater nitrate trends. The trends observed in the RK test are consistent with our statistical analysis of well trends in the five groundwater regions, and within the four existing continuous groundwater nitrate programs. Locally (in a well), nitrate concentrations may vary significantly over time. Some wells experience a downward trend in nitrate, while other wells experience an upward trend in nitrate. The variability in the dynamics of nitrate at wells is very large. Not all areas of the study area are seeing groundwater nitrate increases. At the TLB Westside and in the Forebay and Upper Valley of the SV, trends in groundwater nitrate are ambiguous.

6 Groundwater Nitrate Forecasting: Assessment of Vadose Zone Nitrate Transport

Prepared by:

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6.1 Introduction

Best Management Practices (BMPs) in agriculture are developed for a number of reasons, often with the intent of minimizing the release of chemical pollutants to surface and groundwater. A key challenge in evaluating BMPs, in the context of groundwater contamination, is the lag time from when a BMP is enacted, and when its effects can be seen in the water withdrawn from a well. The goal of Section 6 is to show spatially, areas within the TLB and SV where the effect of BMPs will be observed the soonest at the water table, and to identify which areas will experience significant lag. Although heterogeneity can cause preferential flow through the vadose zone (Harter et al. 2005), resolving detailed soil structure for 2D and 3D simulations is not feasible at the scale of thousands of square miles. In light of this, three maps were created based on three homogeneous soil types: sand, loam, and clay soil, representing the quickest, intermediate, and slowest probable travel times of nitrate to the water table, respectively. HYDRUS 1D was used to model travel times to the water table by specifying daily leachate fluxes, depth to the water table, and soil type. Fluxes of agricultural return water were determined by mass balance using the differences between calculated evapotranspiration (ET) from a field and the amount of water applied through natural precipitation and irrigation (including various irrigation technologies and their associated efficiencies). We show that travel time is essentially driven by the amount of water that infiltrates past the root zone of a crop, the depth of the water table, and the hydraulic properties of a soil. The hydraulic conductivity of unsaturated soils is a function of their water content, and therefore the rate in which water (and dissolved solutes) travels in the subsurface is influenced by the quantity of water infiltrating past the root zone of a crop. Factors such as irrigation efficiency, annual precipitation, and crop evapotranspiration contribute to this flux.

First, a small modeling exercise will show that annual water budget information is sufficient for estimating travel time. Six representative crops grown in the study area (alfalfa, citrus, cotton, almonds, corn, and grain) are used in a 1D soil column model to simulate solute travel time of a conservative (non-reactive and non-sorbing) solute to the water table. Daily water budgets for each crop are calculated to determine the amount of water leaching past the root zone. These fluxes are used to estimate travel time to the water table using the software HYDRUS 1D, a flow and transport model for unsaturated soil. This initial study shows that travel time to the water table can be approximated from annual water budget information for a particular field. The results from the modeling are then applied to agricultural fields in the SV and TLB to produce regional maps of nitrate travel time to the water table (figures 64-66). The modeling results provide a helpful tool for regional planners by distinguishing where adjusted BMPs can have a relatively quick impact to water quality at the water table, and areas where a significant lag will take place.

6.2 Modeling with HYDRUS 1D

6.2.1 Calculating Leachate Fluxes

The first step in calculating leachate fluxes is determining crop ET, the amount of water needed by a specific crop. Daily water needs for a crop are calculated by using a standardized crop coefficient (Kc) multiplied by reference evapotranspiration (ETo)(Snyder et al. 1989). In California, evapotranspiration from a standard/well maintained grass (reference crop) is monitored at hundreds of locations and data are provided through the California Irrigation Management Information System (CIMIS). Multiplying the ETo by a crop coefficient gives a daily estimate of the water used by a crop through evapotranspiration, storage in the tissue, and evaporation from the adjacent soil. Annual crop coefficient curves, which provide a daily Kc value for a crop, are shown in Figure 61 for the representative crop cycles chosen. Crop coefficient curves were generated from methods outlined in University of California's Cooperative Extension *Leaflet 21427* (Snyder et al. 1989) for a typical annual growing season. For portions of the year during which a particular crop is not being cultivated, an estimated "crop coefficient" for bare soil was used (Snyder et al. 1989). It is evident from the figure that each crop requires different amounts of water at different times of the year. The crop coefficient curves were combined with average monthly ETo and average monthly precipitation for the Visalia area, in the central-eastern TLB, to determine irrigation requirements for the crop cycles.

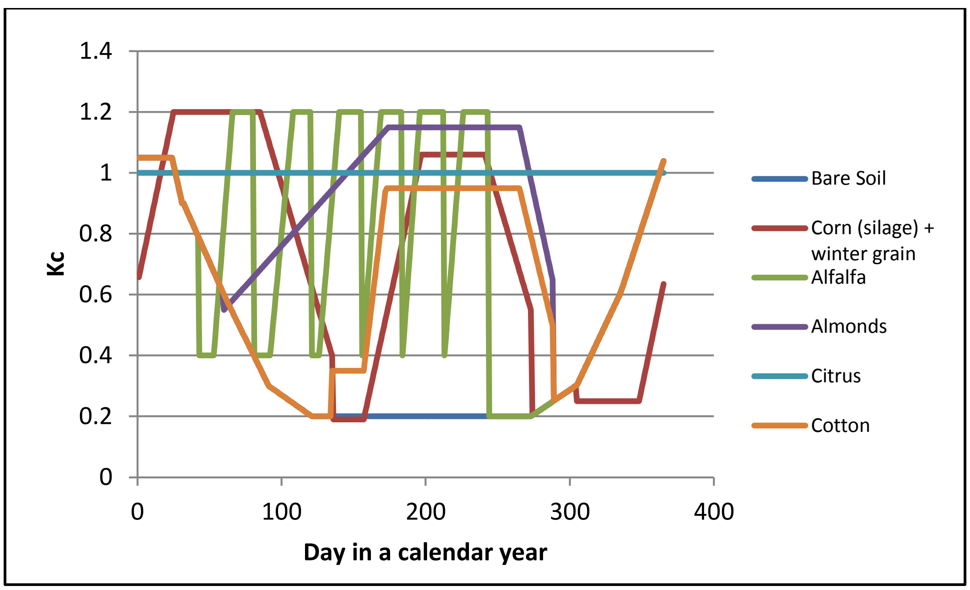

Figure 61. Annual crop coefficient curves.

Irrigated fields generally require more water than will be transpired by the plant. The two main reasons for this are the need to prevent salt buildup at the root zone, and to accommodate irrigation inefficiencies (non-uniform application of water). This results in water traveling past the root zone into the deeper vadose zone. Calculation of this flux is done by subtracting the amount of water transpired, held in the plant tissue, and evaporated from adjacent soils from the amount of water applied via irrigation and natural precipitation.

For each day of the year, one the following calculations was made for each crop:

Precip - (ETo x Kc) + ((ETo x Kc x LR) - Precip) / IrrEff = Daily Leachate under Irrigation (Eqn. 2)

Precip - (ETo x Kc) = Daily Leachate without Irrigation (Eqn. 3)

ETo is the potential evapotranspiration, Kc is the crop coefficient, LR is the leaching requirement needed to control salt buildup in the root zone, IrrEff is the irrigation efficiency, and Precip is the daily average precipitation. Daily Leachate is the amount of water passing the root zone each day assuming a well maintained crop and daily irrigation applications. Eqn. 2 was used if precipitation was not sufficient to fulfill the daily water requirement for a particular crop and additional irrigation was necessary. Eqn. 3 was used if precipitation was sufficient to meet the daily water requirement including the salt leaching requirement.

In Eqn. 2, the daily ETo value (obtained by dividing the long-term monthly average by the number of days in the month) was multiplied by the daily crop coefficient to determine the amount of water required by the crop for a particular day. The water required was multiplied by a leaching requirement (1.1 or 10%), which was assumed for all crops for salt control. Average precipitation (based on monthly long-term averages divided by number of days of the month) was subtracted from the water required by the crop to determine the water deficit needed to be supplemented by irrigation. Irrigation, however, is never 100% efficient due to differences in drip line pressure, sprinkler distribution patterns, fields not being perfectly sloped, and spatial variability in soil permeability. Because of such factors, more water must be applied to a field to ensure that every location on a field receives the minimum requirement of water. Three irrigation efficiencies (90%, 80%, and 70%) were used to represent modern efficiencies of drip, sprinkler, and furrow/flood irrigations, respectively. After irrigation efficiency was included, a daily amount of applied irrigation water was obtained. Subtracted from this number was the amount of water required by the crop (note: a crop’s Kc also accounts for surrounding soil evaporation) that was not being met by precipitation. The excess amount of water was specified as the daily flux of water passing the root zone, referred to here as the Daily Leachate.

In the case where precipitation was sufficient to meet the daily needs of the plants (including a leaching requirement), Eqn. 3 was used and the excessive precipitation was specified as the daily flux of water. There was assumed to be no runoff.

Daily leachate values were calculated for a 365 day calendar year and can be seen in Figure 62. Fluxes shown are based on 70% irrigation efficiency. Table 31 shows total annual leachate fluxes calculated for each crop and efficiency.

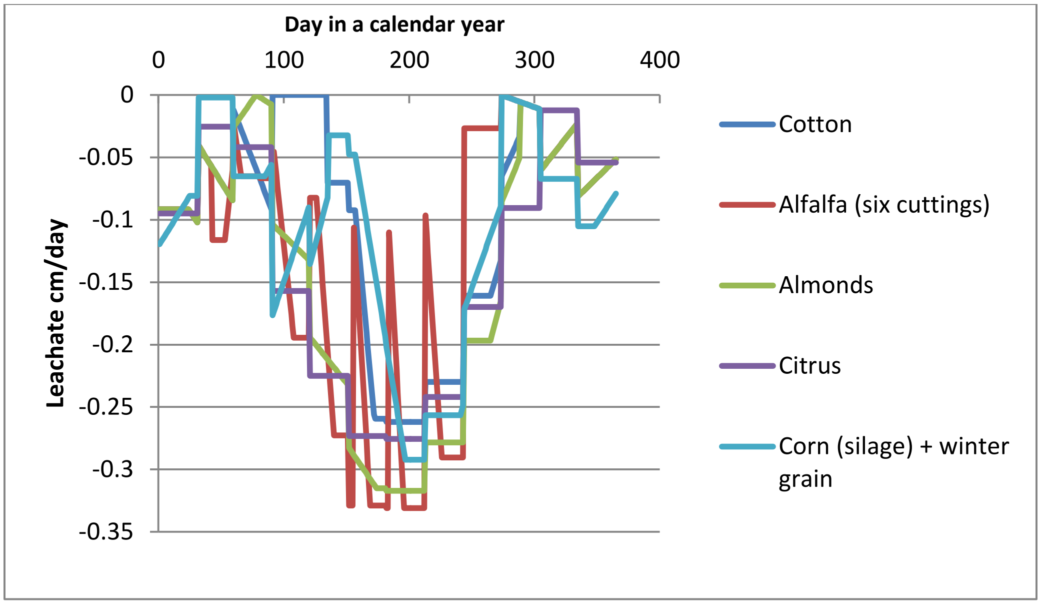

Figure 62. Daily flux of leachate based on 70%irrigation efficiency for the 5 modeled crop cycles. Negative values imply a downward flux in HYDRUS 1D.

6.2.2 Soils

Heterogeneity of subsurface soils usually results in preferential flow (Burow 1993; Harter et al. 2005). Every location in the study area has a unique soil profile, but defining field scale heterogeneity on the scale of thousands of square miles is not possible. Instead of detailed soil profiles, three homogeneous soil profiles were used: sand, loam, and clay soil. Hydraulic parameters for the soil types were obtained from the Rosetta Soil Catalog, included in the HYDRUS 1D software (Simunek et al. 2005). Sandy soil was used to represent the fastest probable travel time of a solute as it has the highest saturated hydraulic conductivity. Clay soil was assumed to represent the longest probable travel time as it has the lowest saturated hydraulic conductivity. Loam soil is thought of as the average soil type, as most soils fall between clay and sandy soil. Table 27 shows the respective parameters for each soil. The Van Genuchten – Mualem single porosity model was used to represent each soil type in HYDRUS 1D.

Table 27. **Hydraulic parameters for the three soil types used.**

Parameter	Sand	Loam	Clay
Residual soil water content	0.05	0.08	0.07
Saturated soil water content	0.43	0.43	0.38
Parameter *alpha* in soil water retention function [cm^{-1}]	0.15	0.04	0.01
Parameter *n* in the soil water retention function	2.68	1.56	1.09
Saturated hydraulic conductivity [cm/day]	712.8	24.96	4.8
Tortuosity parameter in the conductivity function	0.5	0.5	0.5

6.2.3 Soil Column Configurations

A total of 18 different soil columns were modeled using six different depths and three soil types. Nodal spacing was set at 4 cm for each soil column and initial solute concentrations were set to zero (Table 28). Initial pressure heads for each node were set to -100 cm except the last (bottom), which was specified with a head equal to zero in order to simulate the water table and resulting capillary fringe. The default temperature of 20^{o} C was used.

Table 28. Soil column construction.

Column depth	Node Spacing	Number of Nodes	Initial Column Head [cm]	Bottom Node Head [cm]	Temperature (C)	Initial Concentration
2m	4cm	51	-100	0	20	0
5m	4cm	126	-100	0	20	0
10m	4cm	251	-100	0	20	0
20m	4cm	501	-100	0	20	0
30m	4cm	751	-100	0	20	0
40m	4cm	10001	-100	0	20	0

6.2.4 Modeling Nitrate as a Conservative Solute

Nitrate is generally assumed non-reactive when present in oxic unsaturated soils. For the purpose of this study, nitrate was assumed to behave conservatively and was modeled as a non-reactive solute.

Table 29 displays the soil/solute properties used to model nitrate transport in the unsaturated zone (U.S. Department of Agriculture 2011). Longitudinal dispersivity generally changes with the soil type and column depth, but was held constant at 10 cm in this study. Assuming constant longitudinal dispersivity has little to no effect on simulation results, because travel time is determined here as the time until the concentration at the water table is 50% of the input concentration. This travel time is primarily controlled by the advective velocity and not dispersion.

Table 29. Solute transport parameters.

Soil Type	Sand	Loam	Clay
Bulk Density [g/cm^3]	1.69	1.43	1.25
Longitudinal Dispersivity [cm]	10	10	10
Diffusion in water [cm^2/day]	1.55	1.55	1.55

6.2.5 Climate Data

Average monthly precipitation and ETo data were used when calculating daily leachate fluxes. Standard monthly ETo was obtained from the California Irrigation Management Information System (CIMIS) for the city of Visalia, and daily Eto was calculated by dividing the monthly average by the number of days in the month (California Department of Water Resources a). Similarly, average monthly precipitation was obtained from the National Oceanic and Atmospheric Administration (NOAA) for the city of Visalia, and daily precipitation was calculated by dividing the monthly average by the number of days in the month (Nation Oceanic and Atmospheric Administration). Table 30 below shows the monthly values used.

Table 30. Monthly average precipitation and reference evapotranspiration for the city of Visalia.

Precipitation														
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
City Visalia	**mm**	51.56	49.53	54.61	20.32	9.40	3.56	0.25	0.51	6.35	16.51	29.71	37.84	280.16
	inch	2.03	1.95	2.15	0.80	0.37	0.14	0.01	0.02	0.25	0.65	1.17	1.49	11.03
Reference Evapotranspiration (Eto)														
		Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**	**Total**
Station Visalia-33	**mm**	22.09	42.41	84.83	130.30	172.21	194.82	199.64	175.51	125.22	82.04	38.35	21.08	1288.542
	inch	0.87	1.67	3.34	5.13	6.78	7.67	7.86	6.91	4.93	3.23	1.51	0.83	50.73

6.2.6 Modeling

HYDRUS 1D (Simunek et al. 2005) was used to model the transport of nitrate in the vadose zone. HYDRUS 1D solves Richards' Equation (Eqn. 4) for water flow in the vadose zone and Fickian based advection-dispersion equations for solute transport (Radclifee & Simunek 2010).

$$\frac{\partial\theta}{\partial t} = \frac{\partial}{\partial z}\left[K(\theta)\left(\frac{\partial\varphi}{\partial z} + 1\right)\right] \qquad \text{(Eqn. 4)}$$

Crops were simulated for a period of 27 years. Daily leachate values were modeled as a specified flux for the upper boundary condition. If, for example, 2 mm was leached in a particular day, a 2 mm/day flux would be assigned for the entire day. The lower boundary condition was specified as a constant head equal to zero. This allowed the water table to be specified, and for the creation of a capillary fringe.

Fluxes for the initial four years were simulated with a concentration of zero so that a normal wetting pattern could be established. After this, fluxes were assigned a concentration of 1. This was done to "cycle up" the soils and generate adequate initial soil moisture content before the solute was applied. Travel time to the groundwater table was defined as the time when the relative concentration at the water table began to exceed a value of 0.5, reflecting the advective velocity of the solute.

6.2.7 Results

Table 31 below shows the results of the modeling, with travel time values shown in years. Figure 63 shows that there is a linear relationship between the water table depth and travel time for a particular annual flux rate. Using this linear relationship, intermediate depths may be interpolated based on the regression line, which is the downward velocity. The downward velocities were then plotted against their respective annual fluxes to determine the relationship between annual leachate flux and the downward velocity of a solute. This was performed on the model results from the three soil types and is shown in Figure 64. In the next section, these relationships were used to estimate downward solute velocities in lieu of detailed modeling to estimate travel times to the water table.

Table 31. Advective travel time to the groundwater table in years for three soils and six depths. Spaces are left blank when solute transport to a particular depth was not achieved in 27 years.

Advective Travel Time to the Groundwater Table in Years																
		Alfalfa (six cuttings)			Almonds			Citrus			Cotton			Corn (silage) + winter grain		
Irrigation efficiency		70%	80%	90%	70%	80%	90%	70%	80%	90%	70%	80%	90%	70%	80%	90%
Total Leached (cm/yr)		46.6	31.0	18.8	52.8	34.2	19.7	50.8	31.8	17.1	37.1	25.6	16.6	39.8	26.5	16.2
Soil column depth (m)		Alfalfa (six cuttings)			Almonds			Citrus			Cotton			Corn (silage) + winter grain		
Sand	2	0.45	0.54	0.86	0.47	0.56	0.85	0.46	0.57	1.01	0.58	0.68	1.07	0.31	0.63	1.07
	5	0.78	1.48	2.22	0.76	1.35	2.15	0.79	1.51	2.39	1.24	1.73	2.39	1.17	1.5	2.54
	10	1.8	2.78	4.35	1.73	2.69	4.13	1.76	2.78	4.79	2.43	3.29	4.82	2.3	3.13	4.99
	20	3.87	5.44	8.55	3.46	5.16	8.33	3.6	5.4	9.41	4.79	6.53	9.66	4.44	6.27	9.81
	30	5.76	8.39	12.8	5.18	7.69	12.3	5.39	8.08	14.0	7.13	9.86	14.4	6.6	9.49	14.8
	40	7.75	11.1	17.1	7.04	10.2	16.5	7.25	10.9	18.8	9.5	13.2	19.2	8.87	12.7	19.5
Loam	2	1.32	1.68	2.78	1.08	1.63	2.73	1.15	1.68	3.14	1.61	2.15	3.19	1.31	2.08	3.33
	5	2.85	4.32	6.84	2.71	3.94	6.59	2.75	4.18	7.5	3.78	5.17	7.66	3.42	4.95	7.9
	10	5.96	8.58	13.5	5.25	7.95	13.0	5.46	8.39	14.8	7.32	10.3	15.1	6.8	9.92	15.5
	20	11.8	17.2		10.6	15.8		11	16.8		14.7	20.4		13.6	19.7	
	30	17.8			16.0			16.5			21.9			20.5		
	40				21.3			22.1								
Clay	2	1.49	2.35	3.64	1.45	2.08	3.57	1.46	2.34	4.08	1.79	2.69	4.15	1.69	2.6	4.32
	5	3.73	5.74	9.6	3.52	5.46	9.2	3.57	5.66	10.6	4.82	7.13	10.8	4.39	6.78	11.1
	10	7.77	11.7	19.4	6.94	10.7	18.6	7.43	11.6	21.4	9.88	14.4	21.9	9.25	13.8	22.5
	20	15.8			14.1	21.7		14.6			19.9			18.5		
	30				21.2			22.0								
	40															

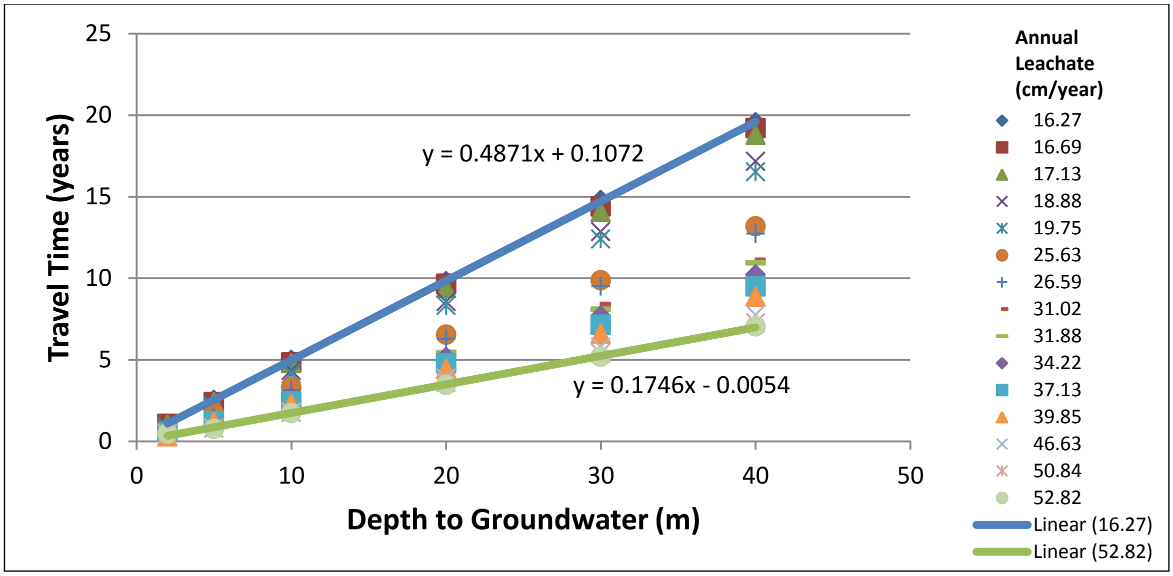

Figure 63. Travel times for a conservative solute in sandy soil.

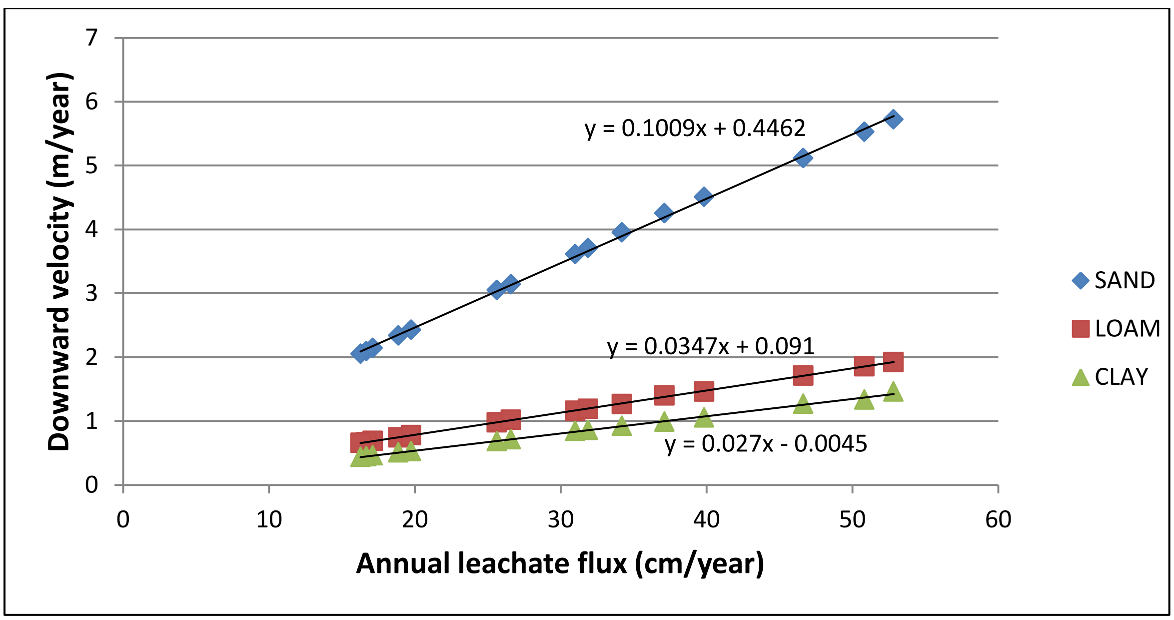

Figure 64. Relationship between annual flux and downward velocity for three soil types.

6.2.8 Modeled Irrigation

Daily fluxes, as previously modeled, are not representative when compared to actual irrigation scheduling. Farmers generally irrigate periodically, often several days to weeks apart. Does this make a difference in travel times? For Alfalfa (six cuttings), daily leaching fluxes were summed every seven days to simulate weekly irrigation to see if this changed the travel times significantly. Table 32 shows that travel times did not change significantly given the time scale under investigation (years to decades). As can be seen in Table 32, the largest difference of 0.2 years was observed at 90% efficiency and 50 m depth to the water table, which is insignificant on the decadal time scale.

Table 32. Travel times compared between simulated daily and weekly irrigation. Values are in years.

	Alfalfa (Six Cuttings), Travel Time in years					
Depth	**Daily Irrigation**			**Weekly Irrigation**		
Efficiency	**90%**	**80%**	**70%**	**90%**	**80%**	**70%**
2m	0.75	0.55	0.48	0.7	0.5	0.43
5m	2.12	1.42	0.77	2	1.37	0.75
10m	4.22	2.77	1.81	4.07	2.72	1.76
20m	8.48	5.42	3.87	8.37	5.36	3.84
30m	12.79	8.33	5.75	12.66	8.17	5.68
40m	17.09	11.08	7.75	16.9	10.99	7.65
50m	21.33	13.72	9.71	21.13	13.62	9.62

6.2.9 Verifying Downward Velocity Estimates

In order to confirm the downward velocities calculated in Figure 64, annual fluxes were divided by average soil water content. This calculation computes an effective vertical velocity that should correspond to the distance traveled in a year according to the mass balance. Cotton and Alfalfa, 70% and 90% irrigation efficiencies, and sand and clay soils were used to verify the results. To establish average soil water content, an observation node was placed at a depth of 5 m in a 20 m soil column. Average soil water content was calculated by averaging the soil water content values recorded at every time step after the conservative solute had reached the observation point (when the concentration at the node exceeded 50%). This was done to ensure that values of water content before the bulk of the water had reached the node (which would reflect the arbitrary initial condition for water content used in the simulation) did not bias the calculated average water content. The results show that our calculated downward velocities using regression analysis (of the relationship between breakthrough time and depth to water table) compare very well with those calculated using annual flux rates and average soil water content. The results are shown in Table 33 and Table 34, below. The results demonstrate that the key variables controlling downward movement of nitrate are the annual water flux rate below the root zone and the effective moisture content of the unsaturated zone between the water table and the root zone.

Table 33. Downward velocities using linear regression of modeled travel times and velocities based on the annual flux divided by the average water content.

Efficiency	Annual Flux (m/yr)	Velocity in sand soil		Velocity in clay soil	
		Using regression (m/yr)	Annual flux/Avg. Water Content (m/yr)	Using regression (m/yr)	Annual flux/Avg. Water Content (m/yr)
Cotton - 70%	**0.3713**	4.193	4.240	0.998	0.990
Cotton - 90%	**0.1669**	2.130	2.080	0.446	0.450
Alfalfa - 70%	**0.4663**	5.151	5.208	1.255	1.238
Alfalfa - 90%	**0.1888**	2.351	2.365	0.505	0.508

Table 34. Average soil water content based on 70% and 90% irrigation efficiencies for cotton and alfalfa grown in sand and clay soils.

Efficiency	Annual Flux (m/yr)	Sand soil average water content	Loam soil average water content	Clay soil average water content
Cotton - 70%	**0.3713**	0.088	0.272	0.375
Cotton - 90%	**0.1669**	0.079	0.249	0.371
Alfalfa - 70%	**0.1888**	0.089	0.277	0.377
Alfalfa - 90%	**0.4663**	0.081	0.252	0.372

6.3 GIS Map Creation

In the previous section, it was verified that three parameters could be used to estimate travel time to the water table: 1) soil type 2) total annual amount of water leached and 3) depth to water table. This section takes the previous modeling and applies the results to the SV and TLB. Three maps (Figure 66, Figure 67, and Figure 68) were generated showing estimated travel time of a solute to reach the groundwater table based on a sand, loam, and clay soil.

6.3.1 Land Use

Detailed land use for the study area was constructed as part of the Technical Report 2 (Viers et al. 2012) and is a combination of Monterey, Kern, Tulare, Kings, and Fresno County surveys created by the DWR (see Technical Report 2, for methods used for land use map construction, Viers et al. 2012). Counties are not surveyed for land use each year, so the map used to define current land use is a collection of several surveys from 1997–2006, with most data coming from the year 2000. The land use surveys cover many land uses such as agricultural, urban, waterways, and natural vegetation. Only those fields classified as agricultural were used in this study.

6.3.2 Field Water Budgets

DWR divides the state into 400+ spatial units for the purpose of hydrological budgeting, referred to as Detailed Analysis Units (DAU) (Calfornia Department of Water Resources). The land use maps were spatially joined to DWR's Detailed Analysis Unit map, to assign each agricultural field the corresponding DAU in which it lies. Each DAU has water budget data estimated for 20 major crops, and is based on local climate and irrigation technologies. Two of the attributes provided by DWR's Annual Water Use spreadsheets are applied water (AW) and evapotranspiration (ET). Applied water is the total amount of water applied via irrigation taking into consideration local irrigation technologies and local climate (precipitation, ETo, etc.). AW includes only water that must be applied *in addition to* precipitation. ET provides an estimate to the amount of water that is taken up into the plant tissue, transpired, and evaporated from the soil surface. The DWR provides this detailed water budget information for the years 1998–2001. This study utilized the average AW and ET values for these four years. The average of the four years was used as it likely represents the long-term average for the study area, given that, in California, these years represent a transition from a wet year (1998) to a dry year (2001).

Precipitation was obtained from Oregon State University's PRISM Climate Group, and is based on average annual precipitation from 1971–2000 (PRISM Climate Group 2006). Precipitation was added to AW to obtain the net amount of water applied to a particular field. ET was subtracted from this to determine the net amount of water which remained. This was assumed to be the annual amount of leachate and was used in combination with depth to the water table and soil type to determine the travel times.

6.3.3 Depth to Water Table

Depth to the water table (Figure 65) was obtained as a point file, which defines depth to groundwater values at the center of PLSS sections in the study area (Spurlock 2000). Fields were assigned a depth to water table value based on the nearest point. If a field contained multiple points, the field was assigned the average value of the points. Although this file provides the best coverage of depth to the groundwater table for our study area, it does not cover it in its entirety. For this reason, if a field was more than 2.5 km (1.6 mi) from the extent of the data coverage, it was not included in the study.

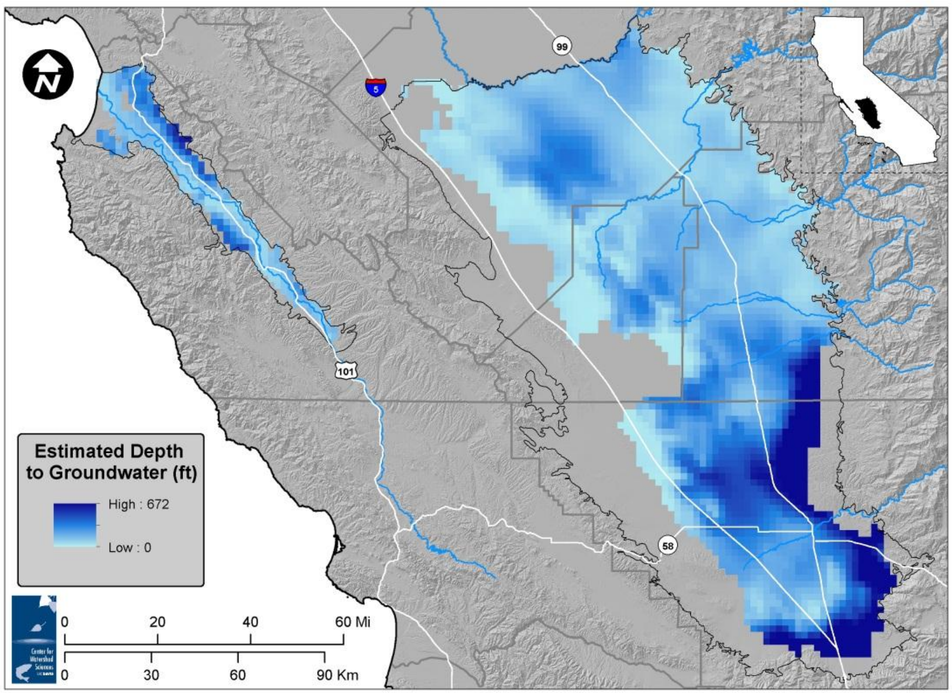

Figure 65. Depth to the water table. (Source: Spurlock 2000.)

6.3.4 Results and Discussion

Three maps are shown in Figure 66, Figure 67, and Figure 68 and represent the estimated travel time from the surface to the groundwater table for three soil types: sand, loam, and clay.

The travel time results shown in Table 31 (for solute transport in sandy soil) are similar to those obtained in a previous study where models of subsurface heterogeneity were employed. Burow, studying nitrate transport in the vadose zone in the SV, found travel times between 4 and 14 years for a depth to groundwater of 35 m (Burow 1993). Burow used two dimensional statistical realizations of subsurface sediments to model the natural variability of hydraulic parameters often found in alluvial and fluvial deposits. The differences in travel times obtained by Burow were based on different realizations of subsurface heterogeneity. Looking at Table 31, similar travel times are seen for the sandy soil travel times to a depth of 30 m (98 ft)(5.18–14.8 years); however, the differences between travel times are a function of annual leachate fluxes.

It is well known that preferential flow exists in unsaturated sediments (Burow 1993; Harter et al. 2005). Harter et al. demonstrated that travel times through a heterogeneous configuration of sediments are much quicker than those through uniformly distributed sediments due to preferential flow through the coarser grained sediments (Harter et al. 2005). Therefore, it is expected that travel times to the water table are likely closer to the results based on sandy soil (Figure 66). The presence of laterally extensive low conductivity layers (e.g., clays and silts) may impede solute travel times; however, even these hydrologic features are known to have spatially variable hydraulic properties that can lead to faster than expected travel times via preferential flow.

Travel times established by groundwater modeling in the saturated zone may underestimate total travel time of a solute from the surface to a well screen due to the additional time needed for a solutes traveling through the unsaturated zone. The study presented here shows that the amount of additional travel time can be significant, depending on the soils present, crops being grown, and depth to the water table.

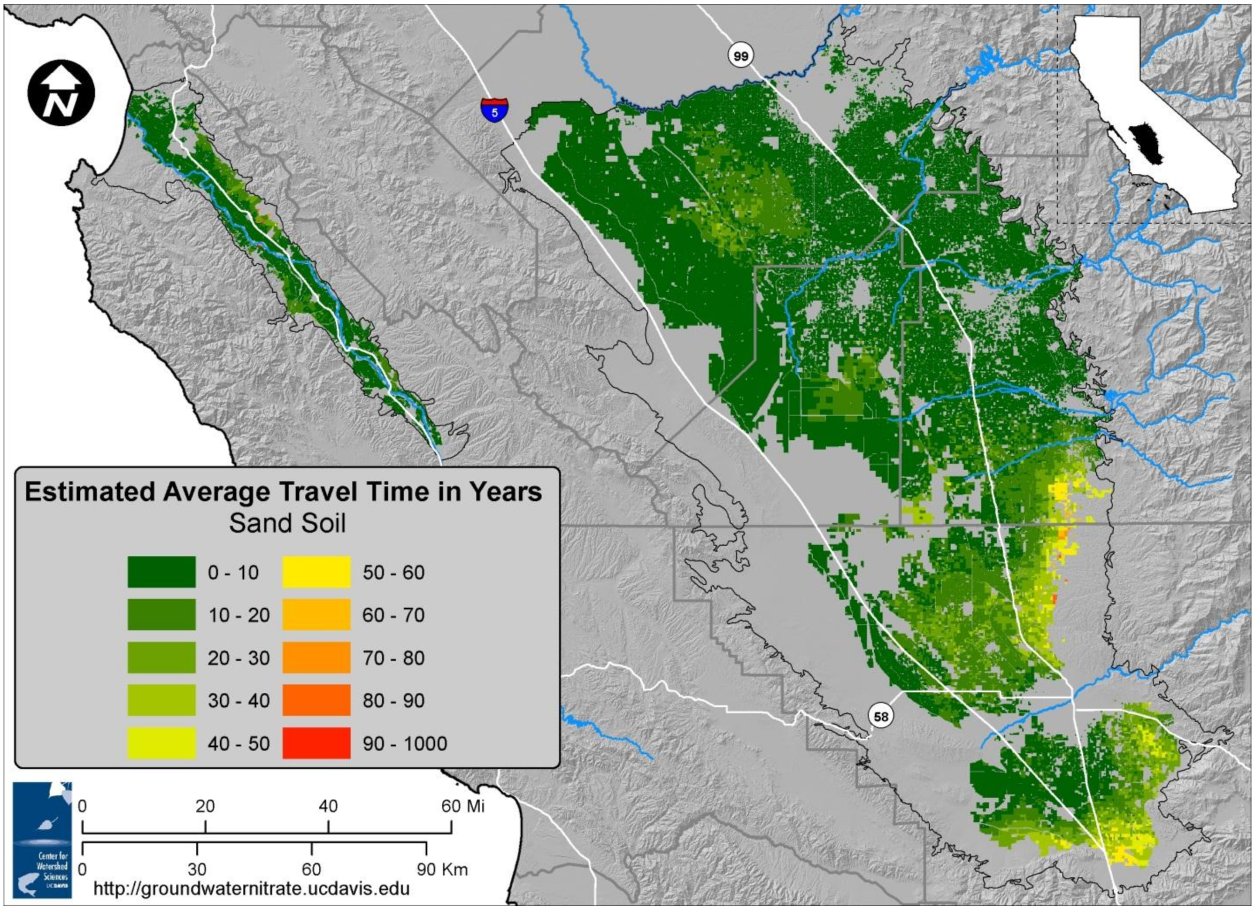

Figure 66. Travel time based on sand soil.

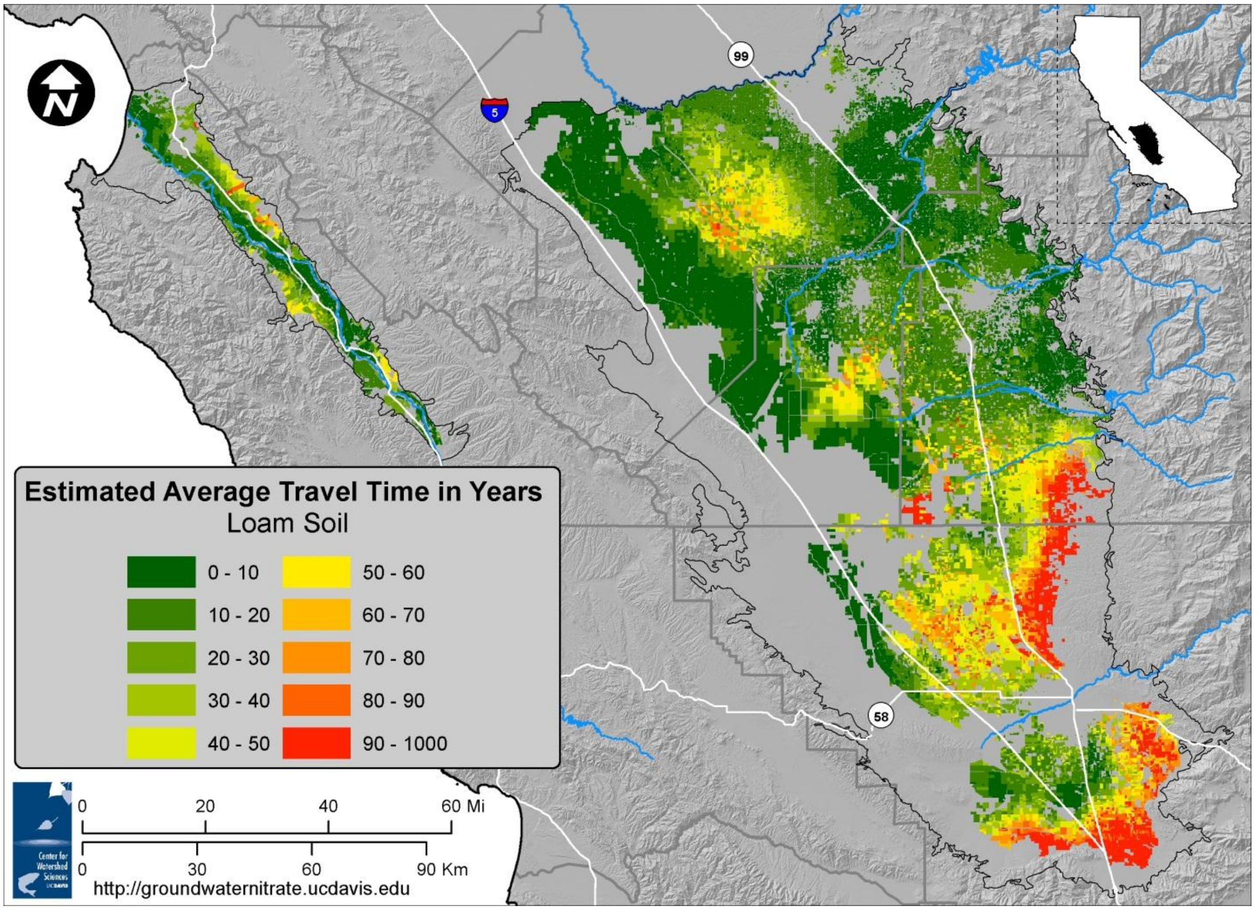

Figure 67. Travel time based on loam soil.

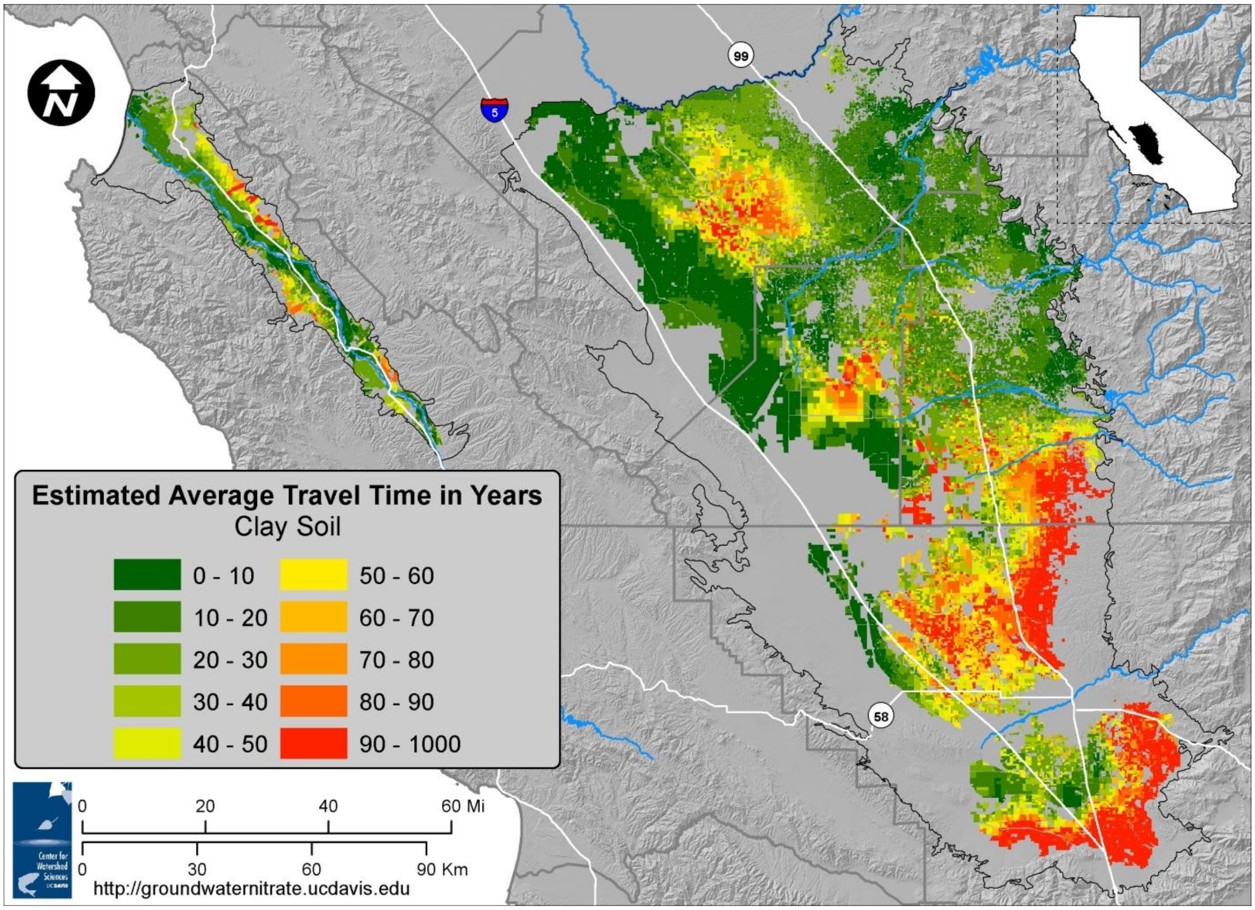

Figure 68. Travel time based on clay soil.

7 Groundwater Nitrate Forecasting/Modeling

Prepared by:

Giorgos Kourakos, Thomas Harter

7.1 Introduction and General Conceptual Overview of the Approach

7.1.1 Nonpoint Source Pollution a Global Groundwater Quality Threat

Nonpoint source (NPS) pollution of groundwater has been recognized as a key water quality problem worldwide (WWAP 2009). Agriculture is considered the most dominant NPS polluter of groundwater, primarily through emissions of nitrogen and salt, but also pesticides and other farm-chemicals (Humenik et al. 1987; Bower 2000; VanDrecht et al. 2003; Vitousek et al. 2010; Watanabe et al. 2010). Nitrate-nitrogen is considered by far the most common type of groundwater contamination associated with agricultural activities (e.g., Spalding and Exner 1993; Harter et al. 2002; Spruill et al. 2002; VanDrecht et al. 2003; Burow et al. 2010). Groundwater nitrate is largely derived from fertilizer nitrogen and animal nitrogen applied in agriculture, where nitrogen is a vital nutrient for plant growth (WWAP 2006, p. 117). The increasingly intensive use of nitrogen-based fertilizers in agriculture has allowed global food production to stay ahead of rapid population growth (Almasri and Kaluarachchi 2007; Stadler et al. 2008; Laftouhi et al. 2003), but at potentially significant cost to current and future water quality in production wells (Corwin and Wagenet 1996). With further growing world population and higher standards of living, food consumption is estimated to increase 70% over the next four decades, while global land and water resources have limited growth reserves. Intensification of agriculture will therefore continue (Molden 2007). Besides nitrate, long-term salinization of groundwater basins from nonpoint sources, particularly in semi-arid and arid irrigated agricultural regions is a second critical threat to groundwater quality around the globe (Burkhalter and Gates 2005; Martín-Queller et al. 2010).

The degradation of groundwater resources not only impacts ecosystems worldwide via return flow of groundwater to surface water (Bouwman et al. 2009), but it affects both irrigation water (salinity) and drinking water quality (nitrate, salinity, pesticides, pathogens). Approximately half of the global population depends on groundwater as a drinking water source (UN WWAP 2003; Giordano 2009). In contrast, most of the global population in intensively farmed agricultural regions such as the California Central Valley, the North-American High Plains and Floridan aquifers, Central Europe's unconsolidated aquifers, the Indo-Gangetic aquifer complex, and the North China plains, relies on – often shallow – groundwater (e.g., Power and Schepers 1989; Chakraborti et al. 2011). Globally, 43% of consumptive water used in irrigation is groundwater (Siebert et al. 2010). Furthermore, particularly in Europe and North America, the need to protect drinking water quality for large populations sectors has driven and continues to drive NPS policy (e.g., Sonneveld and Bouma 2003; Dowd et al. 2008).

Sound policy requires thorough scientific understanding of nonpoint sources and how they work, and of the linkage between nonpoint sources and groundwater discharges to users or affected ecosystems (domestic wells, irrigation wells, urban/municipal wells, springs, discharges to stream reaches). Significant scientific effort has been dedicated to understand, manage, and monitor potential sources, to understand the dynamics of NPS pollutants in the vadose zone and in groundwater, and to assess the environmental and public health consequences of NPS pollution of groundwater (Addiscott and

Wagenet 1985; Corwin et al. 1999; Pavlis et. al. 2010). The spatio-temporal and process complexity of NPS pollution of groundwater on one hand and the number and large diversity of affected stakeholders on the other hand (Figure 69) requires management of large datasets, the bridging of possibly huge datagaps, upscaling, the use of potentially complex models, and – most importantly – that science effectively communicates with policy and decision makers (King and Corwin 1999; National Research Council 1993).

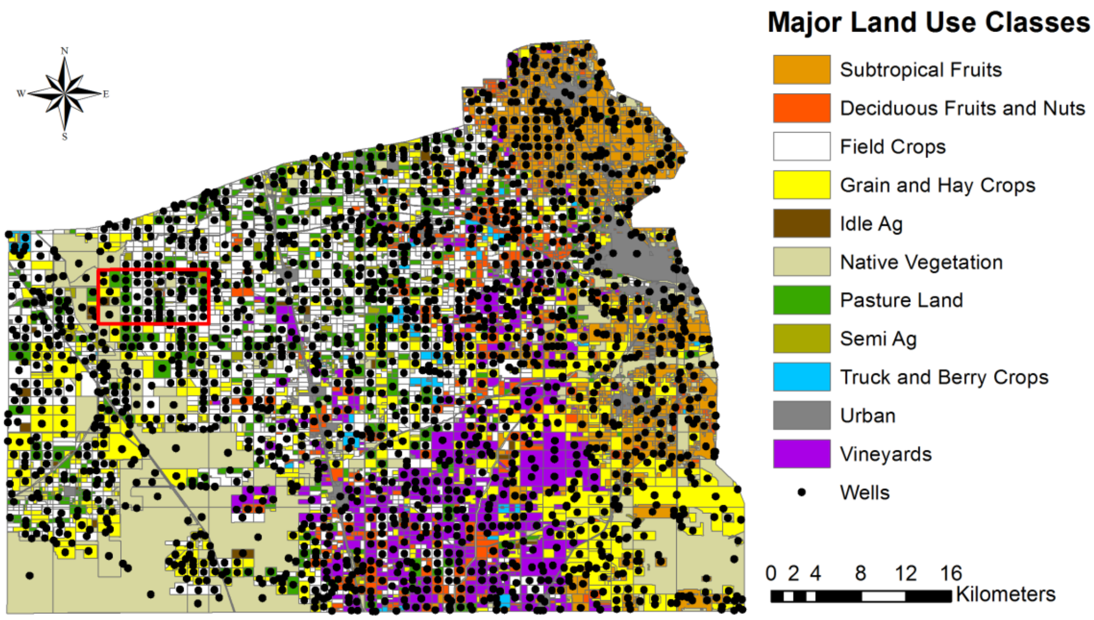

Figure 69. Typical spatial variability of a land use (and implicitly, associated diffuse pollution) in an intensively managed semi-arid agricultural region with significant groundwater pumping for irrigation (black dots), Tule River Groundwater Subbasin, Central Valley Aquifer System, California. (Source: modified from Ruud et al. 2004.)

7.1.2 Key Differences Between Nonpoint Source Pollution and Point Source Pollution Dynamics

Agricultural activities are the dominant nonpoint source of groundwater nitrate and salts. Other significant sources include urban wastewater discharge, septic systems, wastewater holding ponds, and atmospheric deposition. For assessment, planning, and regulatory purposes, nonpoint source pollution, particularly from nitrate and salt, has a distinctly different problem set than point sources:

1. Control measures for point sources have been in development now for four decades (e.g., U.S. Code Title 42 Chapter 103 U.S. Comprehensive Environmental Response, Compensation and Liability Act of 1980), while control and monitoring measures for nonpoint sources of groundwater have only begun to be developed over the last one to two decades (e.g., EU Nitrate Directive[11] and California Salt and Nutrient Basin Plan[12] development).

[11] http://ec.europa.eu/environment/water/water-nitrates/index_en.html

2. Point sources, most commonly, are accidental spills of limited duration contributing a negligible fraction of basin recharge. Nonpoint sources are commonly associated with natural or intentional, managed or unmanaged sources of recharge that provide a significant fraction or even the majority of a groundwater basin's recharge on a time-varying but continuous basis (GWSP 2008; UN/WWAP 2006; Burow et al. 2010).
3. Point sources tend to be of limited spatial and temporal extent. One-time spills may occur over areas from less than 0.1 ha (0.25 acres) to few ha in size (Freeze and Cherry 1979; Bower 2000; Domenico and Schwartz 2008). In contrast, nonpoint source pollution can occur repeatedly across the majority of the land surface area of entire groundwater basins, particularly in agricultural regions. Sources are spatially near-contiguous, while individual source facilities (liable parties) range from ten to several hundred hectares in size and are characterized by large spatial as well as temporal variability and, hence, uncertainty. For example the volume of irrigated water and its concentration in nitrogen vary significantly with crop type and season. In addition there is significant uncertainty related to the estimated amounts of excess nitrate that leach to groundwater (Loague and Corwin 1998). Similar uncertainty exists in rain-fed crops, where nitrate leaching depends on various factors such as rainfall intensity, air temperature (Xin-Qiang et al. In press), the form of nitrogen application (Pilbeam et al. 2004), N application rate, form, and timing (Basso et al. 2010), etc.
4. Point sources of pollution are often very intensive (i.e., the associated pollution level (concentration) at and near the source can be many orders of magnitude above the regulatory limit). Nonpoint source pollution is typically of low intensity (i.e., at concentration levels near the regulatory limit and up to one order of magnitude above the regulatory limit (e.g., nitrate and salts)) (Burow et al. 2010).

These differences between point source pollution and nonpoint source pollution of groundwater require that assessment methods, monitoring approaches, and regulatory frameworks for nonpoint source control do not simply copy the approaches taken in the point source arena, but that methods are developed specifically for nonpoint sources.

7.1.3 Groundwater Nonpoint Source Assessment Tools

Studies have developed various modeling tools for assessing and predicting aquifer pollution impacts or concentration levels in response to land use and management strategies. These models can be generally grouped into three categories:

1. Overlay and index methods, where different parameters of spatially distributed hydrologic, geographic, soils, and source information are combined to give an estimation of the vulnerability in the form of an index (National Research Council 1993; Pavlis et al. 2010) such as DRASTIC (Aller et al. 1987), SINTACS (Civita and De Maio 2004), etc.
2. Statistical methods that estimate the vulnerability by correlating spatial variables with actual occurrence of pollutants in the groundwater (Pavlis et al. 2010) such as regression (Nolan et al. 2002; Worrall et al. 2000), fuzzy logic (Uricchio et al., 2004), etc.

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[12] http://www.swrcb.ca.gov/centralvalley/water_issues/salinity/index.shtml

3. Process based methods to simulate the contaminant transport using mathematical formulas (Fogg et al. 1999).

The majority of these latter models are limited to the simulation of pollutants in the vadose zone, while simple methods such as zero-order mixing models (Mercado 1976; Lee 2007) or vertical plug-flow models (Refsgaard et al. 1999; Hansen 1991; Cho and Mostaghimi 2009) are used for the estimation of the fate of contaminants in the saturated zone. These kinds of approaches are not able to properly capture the spatial and temporal variability of contaminant loading across large aquifer systems.

Detailed spatio-temporal nonpoint source impact assessment in an aquifer requires numerical flow and transport models in two and three dimensions. A few studies developed analytical solutions to governing flow and transport equations (Leij & Dane 1990; Fry et al. 1993; Perez-Guerrero et al. 2009); however, their applicability is limited to simple geometry cases under special boundary conditions. Coupled numerical solution schemes of groundwater flow and transport have been applied in groundwater remediation studies and nonpoint source prediction models at relatively small scale sites (Trowsdale and Lerner 2007) or large scales (Carle et al. 2006), but often using relatively coarse gridded solutions (Almasri and Kaluarachchi 2007; Jiang and Somers 2009; Zhang and Hiscock In Press).

The implementation of a fully three-dimensional flow and transport model for nonpoint source assessment is largely limited by computational resources; typically, current numerical flow and transport models are designed with 10^5 to 10^8 degrees of freedom (particle lines, finite difference cells, finite elements), allowing for 10^2 – 10^3 discretization points per dimension. At typical (point source) contamination sites to which these are applied, the resulting spatial discretization is on the order of 10^{-1} to 10^2 meters (Carle et al. 2006). On the other hand, the simulation of entire groundwater basins affected by, for example, agricultural nonpoint source pollution, being tens to hundreds of kilometers across (e.g., Floridan aquifer system, High Plains aquifer system, Central Valley aquifer system, North China Plain aquifer system, Indo-Gangetic aquifer system), would require spatial grids that are four to six orders of magnitude larger (10^9 – 10^{14} degrees of freedom), at effectively similar discretization. The latter is necessary to properly capture individual sources (e.g., crop fields, lagoons, septic leach fields) and the impacts to individual contaminant sinks or receptors (wells, stream reaches, springs) across a basin (Bloomfield et al. 2006). The application of classic numerical contaminant modeling approaches to model nonpoint sources and their impacts at multiple locations (sinks) across entire basins is generally beyond current computational capacities. This is particularly true if the focus is on the impact to individual production wells (domestic, municipal, irrigation), of which there may be tens of thousands across a single groundwater basin.

To render the computational burden tractable, alternative methods have been proposed. For example Lin et al. (2010) developed a simplified numerical model where the governing equations of 3D groundwater flow and contaminant transport are replaced by a 2D finite element approximation in x-y direction, with a 1D finite difference approach for the vertical direction. Almasri and Kaluarachchi (2007) used surrogate models such as Modular Neural Networks in order to predict nitrate contamination in the Sumas-Blaine aquifer in Washington State, but performance was found inferior to the classical fate and transport model.

A widely used alternative technique is the streamline simulation model, where a multi-dimensional simulation problem is decoupled into multiple one-dimensional problems (Martin and Wegner 1979). Streamline models have been used extensively in petroleum engineering (Blunt et al. 1996; Baker et al. 2002). Jang and Choe (2002) utilized the streamline model to simulate solute transport in fractures, and found that the Breakthrough Curves (BTCs) from simulations matched excellently with experimental data. Bandilla et al. (2009) combined an analytic element based solution of groundwater flow with the streamline method neglecting transverse dispersivity effects. Recently, Herrera et al. (2010) proposed an improved version of the method for simulating reactive solute transport in porous media. Streamline methods have also been used for model calibration (Jang 2007; Jang and Choe 2002, 2004), where the flow domain is decomposed into streamlines and the calibration parameters are adjusted along the streamlines. The efficiency of the streamline method stems from neglecting transverse numerical dispersion. Depending on the modeling objective, the method is computationally far less demanding than a full three-dimensional solution at equivalent high resolution.

Although the streamline method has been established as a reliable alternative solution to simulate transport in aquifers, simulation time for long simulation periods can be exceptionally large. For environmental managers interested in evaluating or optimizing multiple nonpoint source management scenarios, these simulation models are not practical. Tools for efficiently evaluating the long-term impacts of past, current, and (alternate) future nonpoint source loading scenarios at the groundwater basin scale are still lacking.

7.1.4 Proposed Modeling Framework

In this report we developed and use a very efficient, yet highly resolved transport simulation approach that accounts for and takes advantage of the distinct attributes of nonpoint source pollution (as opposed to point source pollution). Our objective is to design a Non Point Source Assessment Tool (NPSAT) applicable to large groundwater basins with a highly heterogeneous, but spatio-temporally continuous coverage of nonpoint sources, that gives stakeholders, decision makers, and environmental managers specific information on the time-dependent statistical distribution of nonpoint source pollutants in a finite-sized ensemble of discrete groundwater discharge surfaces (e.g., well screens and streambeds). The latter comprise a distributed set of discrete compliance surfaces, which can be further categorized (grouped) by significantly controlling factors such as depth, hydrogeologic sub-regions, landscape and land use regions, etc. The concentration history at a discharge surface is controlled by aquifer properties and their spatial distribution, by groundwater pumping and other discharges, and by the spatio-temporally variable, continuous nonpoint source pollution fluxes across the recharge surfaces of the aquifer. Specifically, our objective is to develop an efficient physically-based hydrogeological modeling framework to predict time-dependent pollutant concentration histograms and their probability distributions for compliance surfaces in a groundwater basin under spatio-temporally variable nonpoint source loading.

7.2 Nonpoint Source Assessment Toolbox

7.2.1 Conceptual Approach

Groundwater flow is governed by Darcy's law and the conservation of mass (Bear 1979):

$$\nabla\mathbf{q} + Q_s = \nabla(\mathbf{K}\nabla h) + Q_s = S\frac{\partial h}{\partial t} \qquad \text{(Eqn. 5)}$$

subject to appropriate initial and boundary conditions. Here, $\mathbf{q}$ is the Darcy flux, h is the hydraulic head, $\mathbf{K}$ is the hydraulic conductivity tensor, Q_s represents a vector of sources and/or sinks, S is the storage coefficient, and t represents time. The governing equation of contaminant transport in groundwater is (Bear 1979):

$$R\frac{\partial c}{\partial t} = \nabla\cdot(\mathbf{D}\nabla c) - \nabla(\mathbf{v}c) + G \qquad \text{(Eqn. 6)}$$

where c is the concentration of the contaminant, $\mathbf{v} = \mathbf{q}/\theta$ is the velocity field, θ is the porosity of the porous medium, $\mathbf{D}$ is the dispersion tensor, t represents time, G represents sources and sinks (e.g., via recharge, wells) and R is the retardation factor (Putti et al. 1990). Pollutant concentrations (Eqn. 6) in a groundwater basin are controlled by spatio-temporally variable, dynamic sources and sinks of water and associated (dissolved) pollutants (Figure 69) and by spatially distributed (heterogeneous) aquifer properties.

Of particular interest to groundwater quality protection and management is the pollutant concentration (historical, current, and future) in water discharged from a (finite) set of individual wells or gaining stream reaches (Compliance Discharge Surfaces-CDSs) within a groundwater basin. The pollutant concentration history of the well water or stream reach discharge is here referred to as the breakthrough curve (BTC). The BTC at the CDS is controlled by the pollutant loading history in the source area of the CDS and by the solute reactions and dispersion along the groundwater flow paths between source area and CDS. The CDS source area is defined as the recharge area associated with all groundwater flow discharging into the CDS. Generally, recharge and pollutant loading within the source area may be spatially and temporally variable and not all locations within the source area contribute to the associated CDS at all times, due to transient changes in groundwater flow direction.

To yield the solution of Eqns. 5 and 6 tractable for nonpoint source pollution at the basin scale, yet with sufficiently high resolution, we make three critical simplifications: first, we assume that groundwater flow is steady-state:

$$\nabla\cdot(\mathbf{K}\nabla h) + Q_s = 0 \qquad \text{(Eqn. 7)}$$

Second, we assume that transverse dispersion in Eqn. 6 is negligible (longitudinal dispersion only), and, third, we assume that pollutant reactions are limited to first order degradation, linear sorption, or a combination thereof.

We support the first assumption with the following heuristic consideration: a nonpoint source pollutant entering an aquifer at a specific location, but continuously over a period of time may discharge at multiple proximate CDSs at different times if groundwater flow is sufficiently transient. However, our focus here is on exceedance probabilities and hence on the ensemble set of BTCs across a group of CDSs (and, hence, a group of source areas), which are much less sensitive to transient changes in source area. Source areas of CDSs may partially overlap due to transient flow conditions. Hence, a steady-state flow approximation still allows for capturing both central tendencies (mean travel time) and the degree of variability of travel time within a CDS and between CDSs.

The second assumption is thought to introduce limited error, because the lateral extent of nonpoint sources is large relative to the length scale of transverse dispersivity or transverse macrodispersivity (Neuman 1990; Gelhar et al. 1992; Kim et al. 2004). The third assumption has been found to be applicable to a wide range of nonpoint source pollutants, including salinity, nitrate, and pesticides (Beltman et al. 1995; Lindenschmidt 2006; Almasri and Kaluarachchi 2007).

The steady-state flow problem Eqn. 7 is separable from the transport problem Eqn. 6 and here solved subject to the appropriate aquifer domain and boundary conditions using a finite element method (FEM). The grid resolution is chosen to capture the spatial pollutant loading variability as well as the flow dynamics around individual CDSs with sufficient detail. For example, the average size of individual sources in a typical agricultural region (California, Central High Plains, North China Plains, Central Europe) varies from 10^2 to 10^6 m^2 (10^3 – 10^7 ft^2); hence, the maximum size of a side of an element is in the range from 10 m (~33 ft) to 1000 m(~3300 ft). Near the CDSs, resolution is on the order of 10 m (~33 ft) (e.g., Figure 69 (Pilot study area)) to provide appropriate flow field resolution near the well.

7.2.2 High Resolution Groundwater Velocity Field Computation

Nonpoint contamination sources exhibit significant variability across source types (agricultural crops, septic leach field, ponding basins/lagoons) and among similar sources managed by different landowners and subject to variable land uses (e.g., varying crops). Nonpoint source loading is also highly variable in time. To assess nitrate and other NPS pollution in domestic and public supply wells, spatial variability must be resolved to the scale of individual wells and their source area or finer. For some NPS pollutants, such as nitrate and salinity, which typically vary over a relatively narrow range near regulatory limits (+/- one order of magnitude), temporal variations of source loading at the annual and inter-annual scale drive subsequent well pollutant levels, while shorter-term and very small-scale variations in the pollutant signal are absorbed by the mixing that occurs in typical production well screens. Proper resolution of physical transport processes in groundwater requires high resolution computer models and limits the application of process models to large groundwater basins (10^3 to 10^5 km^2 ($3.8x10^2$ -$3.8x10^4$ mi^2). For example, the average size of individual sources in a typical agricultural region (California, Central High Plains, North China Plains, Central Europe) varies from 10^2 to 10^6 m^2 (10^3 – 10^7 ft^2) (i.e., the maximum size of a side of an element or cell likely is in the range from 10 m (33 ft) to 1000 m (3280 ft)). Process-based model applications have therefore been limited to site-specific studies of pollution, primarily point source pollution, particularly if aquifer heterogeneity is also considered.

In this report the process-based regional modeling is achieved by introducing an automated two-step domain decomposition method where the domain is first simulated using a coarse resolution and then is divided into several overlapping sub-domains for either sequential or parallel high resolution simulation of flow and transport. Boundary conditions for the sub-domains are interpolated from the coarse grid solution, while a weighted average scheme is used to smooth the velocity field across sub-domain boundaries.

In general Domain Decomposition (DD) Methods are special techniques for solving linear or non-linear systems of equations arising from the discretization of partial differential equations (Smith et al. 1996). The majority of the DD methods require intervention at a computing level that existing simulation models either do not provide (COMSOL, FEFLOW) or that is very difficult to implement (MODFLOW, MT3DMS, HYDROGEOSPHERE). Here we use a simplistic domain decomposition method, which has the advantage that it can be combined seamlessly with the majority of the existing simulation models.

According to the simplistic DD, the aquifer is initially treated as a single domain Ω and simulated using a coarse discretization, resulting in a coarse hydraulic head field h^c. Secondly the domain Ω is divided into N sub-domains $\Omega_1, \Omega_2, \dots, \Omega_N$ of very fine resolution. In this report we choose to divide the domain into orthogonal sub-domains, but the same method can be extended to any arbitrary sub-domain shape. We define the boundaries $\partial\Omega$ of domain Ω as external, and the artificial boundaries Γ_n as internal. Note that $\partial\Omega_n$ (i.e., the boundary of sub-domain) consists either of part of $\partial\Omega$ and Γ_n or exclusively of internal boundaries Γ_n. To assign boundary conditions to internal boundaries Γ_n we use an interpolation method $I_{\Omega\to\Gamma_n}$ where the unknown boundary conditions Γ_n are interpolated from the coarse solution h^c and the fine head field h_n^f for each sub-domain n is calculated independently. At the end of this process, h^f is the union of all individual simulation results $h^f = h_1^f \cup h_1^f \cup \dots \cup h_N^f$.

However, based on our simulation results already obtained, it was found that the resultant fine head field h^f exhibits discontinuities across the internal boundaries Γ_n. To alleviate this problem, we propose to offset the artificial sub-domains boundaries at a specified distance l_{of}, thus producing an overlapping zone of width $2l_{of}$. As shown in Figure 70 the aquifer domain can be of any arbitrary shape, and the simulated area of each sub-domain is defined as the intersection of domain Ω and the extended sub-domain boundary. In Figure 70, the solid red lines correspond to the non-extended sub-domain and the green dashed lined to the extended sub-domain after the offset operation.

Let i be a discrete point, located within the j non-extended sub-domain and also within M extended sub-domains (eg., Figure 70, the point i is located within the 2nd non-extended sub-domain and in the 8th extended sub-domain eg., $M = 1$). Each sub-domain returns a slightly different solution $h_i^j \neq h_i^k \neq h_i^{k+1} \neq \cdots \neq h_i^M$ as a result of the different boundary conditions, where $k \in [1, M]$ are the subdomain IDs. In the case of nonconforming meshes, an interpolated value is used for the heads $h_i^k, h_i^{k+1}, \dots h_i^M$.

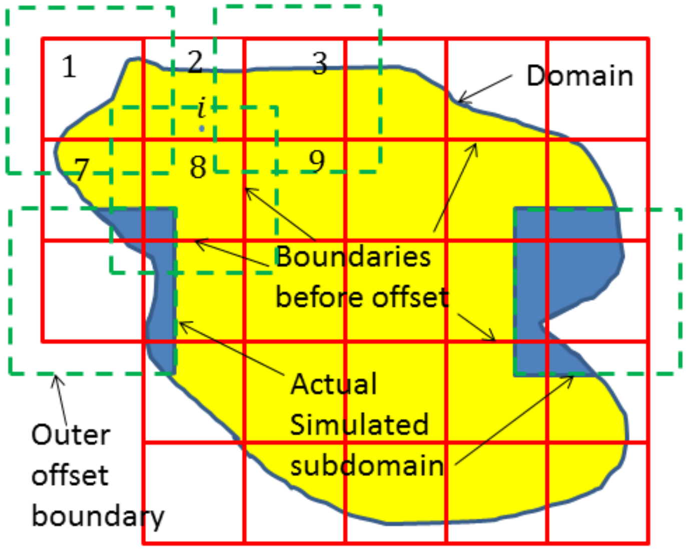

Figure 70. Schematic illustration of Simplistic Domain Decomposition Method.

Subsequently the $M+1$ head values can be averaged to obtain the head for point i. However, it has been observed that points closer to the boundaries have a larger head discrepancy from true head values. To counteract this problem, we propose a weighted average scheme, where the weights are taken proportional to the distance from the barycenter[13] of each sub-domain:

$$\bar{h}_i = \frac{1}{\sum_n^M w_i^n} \sum_{n=0}^{M} w_i^n h_i^n; \qquad n \in [j, k, k+1, \dots, M] \tag{Eqn. 8}$$

where h_i^n is the calculated head of point i based on the solution of sub-domain Ω_n and w_i^n is the weight of point i with respect to sub-domain n, calculated by the following empirical formula:

$$w_i^n = \min\left\{W\left({}^{l_x}/_2, l_{of}, |x_i - x_c^n|\right), W\left({}^{l_x}/_2, l_{of}, |x_i - x_c^n|\right)\right\} \tag{Eqn. 9}$$

where x_i, y_i are the coordinates of the point i, l_x, l_y are the sub-domain lengths before the offset operator along the x and y directions, respectively (see Figure 71 left), x_c, y_c are the coordinates of the barycenter of the orthogonal and W is a geometric function defined as:

$$W(l, d, x) = \frac{x - (l + d)}{-2d} \tag{Eqn. 10}$$

It can be seen from Eqn. 10, that input arguments l and d of Eqn. 10 are constants for each sub-domain. Eqn. 9 returns zero weights for the points that lie on the outer offset edges and weights of ones at the inner offset edges, with weights that vary linearly within the overlapping zone between these two numbers (Figure 71, Right). The negative weights outside the extended orthogonal are not used and

[13] The barycenter of an object is the center of Mass

they are ignored. The result of the above formulation is a smooth velocity field across the artificial boundaries.

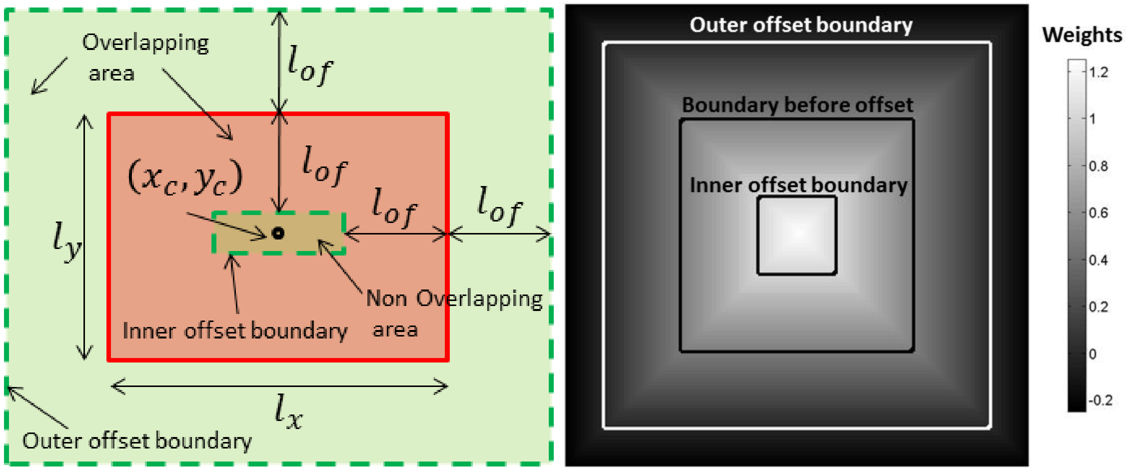

Figure 71. Left) Geometric definition of symbols involved in the weight calculation, Right) Weight function.

The velocity field is then used by transport simulation model of NPSAT for a highly efficient, streamline-based quasi-3D solution of an (arbitrary) solute transport equation yielding a pollutant unit response Functions (URF).

7.2.3 Transport Simulation: Streamlines

Neglecting transverse dispersion, the transport equation (Eqn. 6) is solved through an ensemble of one-dimensional streamline-based solutions focused on the CDSs rather than a fully three-dimensional solution. Obtaining a quasi-3D solution with the streamline approach specifically for CDS locations can be significantly more efficient than computing fully three-dimensional transient solutions over the entire groundwater nonpoint source contamination domain for time-horizons spanning decades to centuries (Thiele 2001).

Here, we define streamlines by their exit points on the CDS and use backtracking to the source area (Figure 72). Exit points of streamlines are distributed across each CDS such that each streamline represents a known fraction of flow into the CDS. Streamline exit points around a well screen CDS are organized in multiple horizontal layers, each with a finite number of exit points per layer: $N_s = N_l \cdot N_{ppl}$ where N_l is the number of layers, N_{ppl} is the number of exit points per layer and N_s is the total number of streamlines exiting on the well screen. Streamline contributions to the CDS BTC are weighted based on the exit velocity at the CDS.

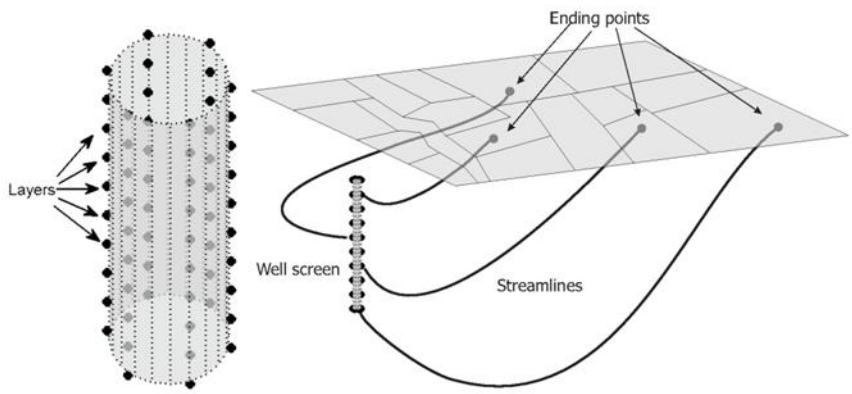

Figure 72. Distribution of exit points around the well screen.

The accuracy of the breakthrough curve simulation at the CDS is determined by the number of streamlines used relative to the (spatio-temporally variable) pollutant loading across the source area (source loading). For computational efficiency, a balance must be sought between accuracy and numerical efficiency. The choice of the number of streamlines used is application-specific and depends on the desired accuracy. Given typical uncertainties and inaccuracies associated with estimating groundwater flow parameters and source loading, a practical simulation goal is to obtain discharge (well) concentrations that have a numerical accuracy within 5% of the true mathematical solution or, alternatively, at the 5%–10% level of a problem-specific contaminant concentration level of interest (e.g., drinking water limits for nitrate, salinity), whichever is larger. Consider a nonpoint source concentration that varies between 10% (background) and 1,000% (intensive source) of the regulatory control level while recharge is relatively uniform. This is typical for nitrate and salinity pollution from agricultural landscapes (Harter et al. 2002; Burow et al. 2010). In this case, the scenario requiring the highest resolution occurs if 1% of the CDS source area has the maximum concentration of 1,000% of the regulatory control level, while the remainder of the source area recharges at a background concentration of only 10% of the regulatory control level. Hence, the high polluter adds ($1\% \cdot 1000\%$=) 10% of the regulatory concentration level to the background concentration at the CDS. The number of streamlines must be sufficiently large to ascertain that the procedure captures the 1% of the source area with high concentration. Generally, from 10^2 to 10^3 streamlines are therefore needed to properly simulate the BTC at such a CDS.

7.2.4 Streamline Computation

To define the streamline associated with each exit point on the CDS, a backward particle tracking is performed for each of the N_S exit points, and the positional and velocity vectors are computed for

$N_s \times N_{CDS}$ streamlines, where N_{CDS} is the number of CDSs included in the simulation area. Backward particle tracking for streamlines has two distinct advantages; transport is computed only for the part of the aquifer that is of interest to the simulation outcome (concentration hydrographs at contaminant sinks), and by using backward particle tracking to define streamlines, we avoid the so-called "weak-sink" problem in numerical solutions of (5) (Zheng and Wang 1999).

For each starting point $i_s \in [1, N_s]$ a backward particle tracking is performed until the particle intersects the water table, yielding a streamline $S_{i_s}^{i_{CDS}}$. In groundwater, the streamlines describe the time t for a particle to travel a certain distance s within the groundwater velocity field. Mathematically the streamlines are expressed by the following first-order, initial-value, ordinary differential equation:

$$\frac{d\vec{x}_p}{dt} = \vec{v}_p(\vec{x}_p, t)$$

(Eqn. 11)

$$\vec{x}_p(t = \tau_p) = \vec{x}_{p_0}$$

where $\vec{x}_p$ is the position vector, $\vec{v}_p$ is the pore velocity vector, and $\vec{x}_p(t = \tau_p)$ is the starting point (e.g., exit point of streamline Figure 72). Eqn. 11 can be solved analytically (Pollock 1994) or numerically using any known numerical methods. In this report we used a hybrid numerical integration method. The method is a mix of the predictor-corrector scheme and the fourth order Runge-Kutta integration method. The method starts by calculating the velocity $\vec{v}_p$ at a given point $\vec{x}_p$. The velocity is computed through interpolation based on the head field. Next we estimate a new point $\vec{x}_{p+1}^{(1)}$ using an explicit Euler formula:

$$\vec{x}_{p+1}^{(1)} = \vec{x}_p + \vec{v}_p \cdot h \qquad \text{(Eqn. 12)}$$

Where h is a predefined step. According to the predictor-corrector scheme, the position $\vec{x}_{p+1}^{(1)}$ is improved iteratively using the corrector:

$$\vec{x}_{p+1}^{(i+1)} = \vec{x}_p + \frac{1}{2}(\vec{v}_p + \vec{v}_p^{(i)}) \cdot h; \quad i = 1,2,3 \qquad \text{(Eqn. 13)}$$

The iterations stop when the discrepancy between the positions of points i and $i + 1$ is smaller than a specified threshold.

Typically, based on our simulations, the predictor-corrector scheme converges after 2–5 iterations. However, it was observed that in some cases (i.e., very irregular geometry of mesh elements) the predictor-corrector scheme either does not converge fast or in very few cases does not converge at all. To counteract this, we employed a fourth order Runge-Kutta integration method when the iterations of the predictor-corrector integration exceed a certain number without converging (e.g., 10). In these cases the new position $\vec{x}_{p+1}^{(1)}$ is calculated using Eqn. 12 where $\vec{v}_p$ is the weighted average of four velocities along the direction of flow.

A critical step in the particle tracking method is the computation of velocity, based on a hydraulic head field. In general, velocity is defined as the negative gradient of head divided by the aquifer porosity (i.e., $v = -\boldsymbol{K}\nabla H/\theta$, where $\boldsymbol{K}$ is the hydraulic conductivity tensor, H is the hydraulic head, and θ is the porosity)**.** According to the finite element method a continuous hydraulic head $H(x, y, z)$ field is approximated as the weighted average of the heads H_i at specific discrete locations $i = [1,2, \dots, N]$ (i.e., the nodes of the finite element mesh) weighted by the shape functions $N_i(x, y, z)$, $H(x, y, z) = \sum_{i=1}^{N} N_i(x, y, z)H_i$. Similarly, the gradient of the hydraulic head is approximated as the weighted average of the heads H_i multiplied by the gradient of the shape functions $\nabla N_i(x, y, z)$. Since the hydraulic heads H_i are computed from the solution of the groundwater flow equation, the calculation of velocity is simplified to the calculation of the shape function derivatives. However in 3D models, shape function derivatives are very complex expressions of x, y, z. In this report we used exclusively prism elements (Figure 73) and isoparametric shape functions, which simplify the derivative calculations.

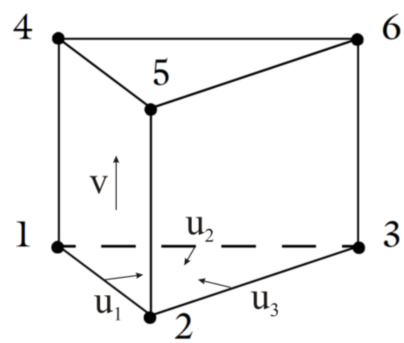

Figure 73. Prism element.

Isoparametric shape functions are defined on an element local coordinate system and in the case of prism elements are expressed as the following:

$$
\begin{aligned}
N_1 &= u_1\,(1-v)/2 \\
N_2 &= u_2\,(1-v)/2 \\
N_3 &= u_3\,(1-v)/2 \\
N_4 &= u_1\,(1+v)/2 \\
N_5 &= u_2\,(1+v)/2 \\
N_6 &= u_3\,(1+v)/2
\end{aligned}
\qquad \text{(Eqn. 14)}
$$

where $u_1 + u_2 = u_3$. Note that shape functions are defined in a local coordinate system $N(u_1 u_2, v)$ and velocity computation requires the derivatives with respect to the physical coordinate system $\partial N(x, y, z)$. However the conversion is straightforward and is obtained by the following equation:

$$\partial\mathbf{N}(x, y, z) = \mathbf{J}^{-1}\mathbf{G} \cdot \mathbf{N} \qquad \text{(Eqn. 15)}$$

where $\mathbf{G}$ is the gradient operator in the local coordinate system $\mathbf{G} = [{}^{\partial}/_{\partial u_1}\ {}^{\partial}/_{\partial u_2}\ {}^{\partial}/_{\partial v}]^T$, $\mathbf{N} = [N_1, N_2, \dots, N_6]$ and $\mathbf{J} = \mathbf{G} \cdot \mathbf{N} \cdot [\mathbf{x}\,\mathbf{y}\,\mathbf{z}]$, where $\mathbf{x} = [x_1, x_2, \dots, x_6]^T$, $\mathbf{y} = [y_1, y_2, \dots, y_6]^T$, and $\mathbf{z} = [z_1, z_2, \dots, z_6]^T$.

In the above analysis we used linear shape functions. Higher order elements can be also used,at the expense of computational complexity and increased CPU runtime.

After the calculation of the streamlines, each one consists of a positional vector $\mathbf{x}_i$, and velocity vector $\mathbf{v}_i$. The positional vector $\mathbf{x}_i$ contains the distance along the streamline $S_{i_s}^{i_{CDS}}$ measured from the initial point i_s, and the velocity vector $\mathbf{v}_i$ contains the velocity magnitude that corresponds to positions of vector $\mathbf{x}_i$. Note that the last element of vector $\mathbf{x}_i$ is the key link of the streamline $S_{i_s}^{i_{CDS}}$ with the nonpoint source loading function L_j; $j \in [1, N_{land}]$, and eventually links each exit point on the CDS with a contamination source L_j.

7.2.5 Unit Response Function Approach

The linearization of the transport problem (Eqn. 6) allows for the application of the principle of superposition (Jury and Roth 1990); the concentration history at any streamline exit point on the CDS, due to a temporally variable source loading history at the associated source boundary, can be computed as a superposition of solutions of the so-called unit response functions (URF). URFs have been widely used for the simulation of rainfall-runoff processes (Saghafian 2006; Jukic and Denic-Jukic 2009), where the URF is known as unit hydrograph. Researchers also employed URFs as transfer functions to simulate solute transport in the unsaturated zone (Jury 1982; Jury et al. 1982, 1986; Jury & Roth 1990; Heng and White 1996; Stewart and Loague 2003; Jaladi and Rowell 2008; Mattern and Vanclooster 2010) and in watersheds (Botter et al. 2006). Here, the transfer function concept explored in Jury and Roth (1990) is interpreted as a transfer function across a finite-sized three-dimensional streamtube linking a fraction of the source area with a fraction of the CDS. Each streamline represents an infinite number of stream-filaments (particle paths) within the associated streamtube (Ginn 2002).

7.2.6 Transport Simulation along Streamlines

For each streamline, a one-dimensional transport model is applied to compute the URF. Generally, any transport model may be applied within the NPSAT framework to compute the URF provided that the superposition principle can be applied (e.g., Continuous Time Random Walk (Berkowitz et al. 2006), Fractional Advection Dispersion equation (Meerschaert et al. 1999), the tempered one-sided stable residence time density (Cvetkovic 2011), and others). Here, we use the one-dimensional Advection Dispersion Equation (ADE) (Jury and Roth 1990):

$$\frac{\partial c}{\partial t} = \frac{\partial}{\partial x}\left(D\frac{\partial c}{\partial x} - vx\right) \quad \text{(Eqn. 16)}$$

subject to:

$$\begin{aligned} &c(x,0) = 0 \\ &c(0,t) = 1\,; t > 0 \text{ (Heaviside step function input)} \\ &\left(\frac{\partial c}{\partial t}\right)_{x=x_{\max}} = 0 \end{aligned} \quad \text{(Eqn. 17)}$$

where D represents the effective macrodispersion, given by $D = \tilde{\alpha}_L v$. The macrodispersivity, $\tilde{\alpha}_L$ intrinsically accounts for the effects of aquifer heterogeneity within the streamtube represented by the 1D streamline. Consistent with field experiments (Gelhar et al. 1992) and numerical experiments (Green et al. 2010), the longitudinal macrodispersivity, $\tilde{\alpha}_L$ is scaled relative to the length of the streamline,

$$\tilde{\alpha}_L = f(L_s) \quad \text{(Eqn. 18)}$$

where L_s is the streamline length. The velocity varies along the streamline, and is calculated from the flow solution by the norm $v = \sqrt{v_x^2 + v_y^2 + v_z^2}$. The solution to Eqn. 16 is obtained numerically for each streamline. The streamline URF is computed from the resulting solution $c(t)$

$$URF = c(t) - c(t-1) \quad \text{(Eqn. 19)}$$

This ensures that the area of the URF is always equal to 1 (Figure 74).

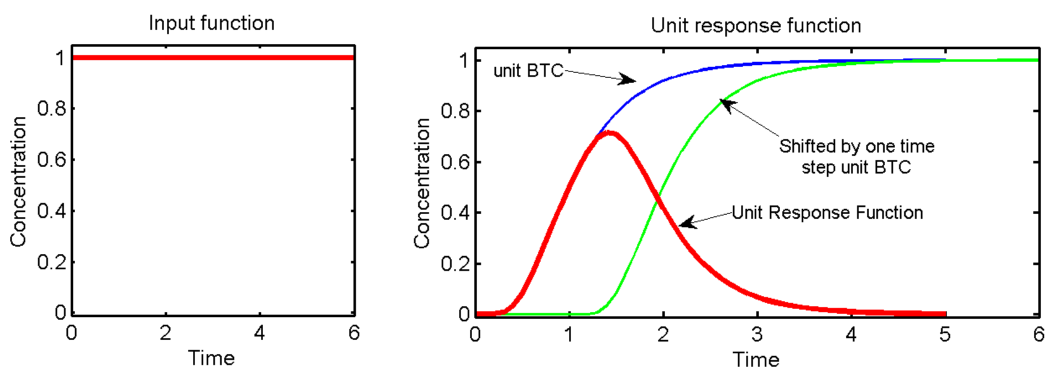

Figure 74. Calculation of Unit Response Function.

7.2.7 Unit Response Function Parameterization

Streamline URFs are archived for retrieval during the forecasting phase of the NPSAT. We found that the shape of the URFs is similar to that of common probability distribution functions (pdfs) found in the statistical literature and in statistical software. Both functions intrinsically integrate to a unit area. We can therefore readily fit the empirically determined URFs to a library of pdf functions using existing software. Pdfs are typically defined by two to three parameters and the function itself. The fitting procedure used consists of a gradient based optimization method aiming at minimization of the error between the empirical and fitted URF. Rather than archiving on the order of 10^4 – 10^5 bytes of data per CDS (10^2 or more datapoints per each of 10^2 – 10^3 streamlines), this procedure reduces the archive to 3 to 4 numbers (2–3 parameters and one index to identify the function) per streamline or between 10^2 and 10^3 bytes of data per CDS. This allows for efficient data storage and forward modeling in applications to large groundwater regions, where the number of CDSs may range from 10^4–10^6 or even higher.

7.2.8 Summary of Construction Phase

Use of the URF approach computationally decouples the transport process from the nonpoint source loading process. URFs can be computed a priori without knowledge of the actual nonpoint source concentration history. We call this the NPSAT construction phase. The NPSAT construction phase requires the following steps (Figure 75):

- Geospatial mapping of the individual land use parcels and their (non-transient, average) recharge, of the CDSs (e.g., wells) and their (non-transient, average) discharge, and of other boundary conditions
- Computation of a detailed three-dimensional steady-state groundwater velocity field using a high-resolution numerical solution to Eqn. 7 with distributed recharge, groundwater pumping, and groundwater discharge to streams
- Geospatial mapping of the desired distribution of streamline exit points on the set of CDSs
- Backward computation of streamlines from their CDS exit points to the water table (source area)
- Computation of a URF as the one-dimensional solution of Eqn. 16 separately for each streamline, given a unit input step function, and fitting a parametric function to this empirically obtained URF at each streamline, a step that drastically reduces the data storage requirements.

The construction phase yields a geospatial database of the location of CDSs (wells, drains, springs, stream reaches), an identification of associated streamlines, the parameters and a code identifying the form-function of the URF for each streamline, and each streamline’s recharge and discharge (beginning and end) location.

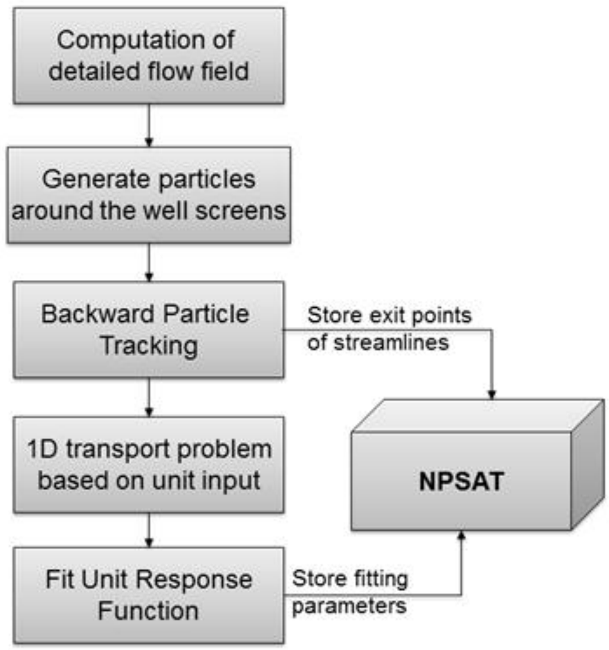

Figure 75. Construction phase of the NPSAT.

7.2.9 Implementation Phase: Computing Breakthrough Curves at CDSs

In the implementation phase of the NPSAT, the BTC of a CDS is computed by convoluting each streamline-specific unit response function with the actual, location-specific nonpoint source loading function, then performing a flux-weighted integration of streamline-output concentrations at time t over all streamlines exiting in a specific CDS (model prediction phase). Suppose that N_s streamlines and associated unit response functions $\mathbf{URF} = \{URF_1, URF_2, \dots, URF_{N_s}\}$ were computed for the $i_{CDS}{}^{\text{th}}$ compliance discharge surface. The source loading functions associated with the $\mathbf{URF}$ are denoted as $L_i; i \in [1, N_s]$. For each streamline, the concentration history is obtained by convolution:

$$BTC_j(t) = \sum_{d=0}^{t} L_j(t-d) \cdot URF_j(d) \qquad \text{(Eqn. 20)}$$

where d increases in the summation at time step intervals and t is the total runtime of the transport model. Eqn. 20 is the numerical approximation of the general convolution operator between two functions f and g expressed as $f * g = \int_0^t f(\zeta)g(t-\zeta)d\zeta$. After the calculation of the concentration history at each streamline, the BTC of the CDS is computed from:

$$\overline{BTC}_{i_{CDS}}(t) = \frac{1}{w_1 + w_1 + \cdots + w_{N_s}} \sum_{j=1}^{N_s} w_j \cdot BTC_j(t) \quad \text{(Eqn. 21)}$$

where w_j is the weight that represents the amount of flow that corresponds to each streamline.

Unlike in watershed NPS modeling, where the CDS output of individual stream reaches or tributaries is effectively integrated at the watershed outlet (Basso et al. 2010), the solute output at individual groundwater well CDSs does not further mix among CDSs (similarly in the case where a large number of low order stream reaches are considered separately). Instead, the BTCs provide the basis for constructing time-dependent pollutant exceedance probability distribution functions (pdfs) across user-specified specific population sets of CDSs (e.g., domestic wells, irrigation wells, drinking water wells, stream reaches) within the modeling domain. This stochastic analysis is the final step in the NPSAT process (Figure 76).

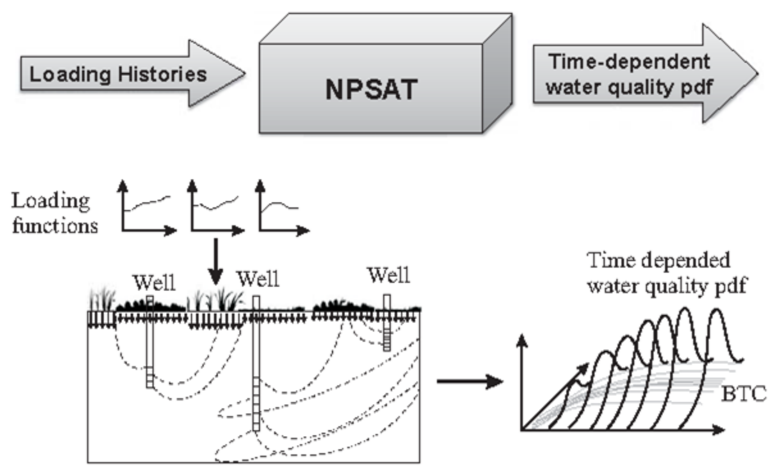

Figure 76. Simulation phase of the NPSAT.

7.3 Approach to Generate Representative Groundwater Velocity Distribution

One of the main assumptions of the proposed modeling approach is the aquifer flow field is steady state. While this assumption might not be valid when a monthly or even seasonal temporal scale for the analysis is used, it is justifiable with the rationale that the model will be used for prediction at the annual temporal scale (see justification in subsection 7.2.1 Conceptual approach).

To choose a representative steady state flow field for the study area we used two criteria: 1) the change of storage volume in the aquifer for the selected period should be as small as possible (i.e., the volume of water that entered the groundwater aquifer must be equal to the losses) and 2) the various stresses for the selected period have to correspond with the average stresses for the study area.

7.3.1 Steady-state Model for Tulare Lake Basin

For the TLB the analysis is based on the Central Valley Hydrologic Model (CVHM) (Faunt 2009). The CVHM is a transient state groundwater model, with a monthly time step, coupled with the FARM process (Schmid et al. 2006) which is used as a water budget tool to estimate the groundwater stresses due to agricultural practices (e.g., groundwater recharge and pumping, etc.). The CVHM simulates the groundwater flow for the entire Central Valley aquifer for the years 1962 – 2003. According to CVHM transient simulation, the storage in the aquifer decreases over the 41-year simulation period. Figure 77 illustrates the cumulative change in groundwater storage for four basins of the Central Valley. The violet line corresponds to the TLB. It can be seen that, despite the general declining trend, there is a two-year period (1995 – 1997) where the change in storage is rather negligible (see red oval Figure 77). The next requirement is that the stresses need to be representative of the study area. Figure 78 shows the simulated water budget from the CVHM. In particular it shows the agricultural pumping, the surface water deliveries, the water delivery requirements for irrigation (groundwater plus surface water deliveries), and the landscape recharge. Note also that three years, 1975, 1990, and 1998, have been designated as typical, dry, and wet, respectively. Interestingly, for the years 1996 and 1997 the simulated agricultural pumping, surface water deliveries, total delivery requirements, and the landscape recharge are very close to those of the typical year (compare the ends of the arrows which point to the amount of acre feet per year for the typical year 1975, and the year 1996 in Figure 78).

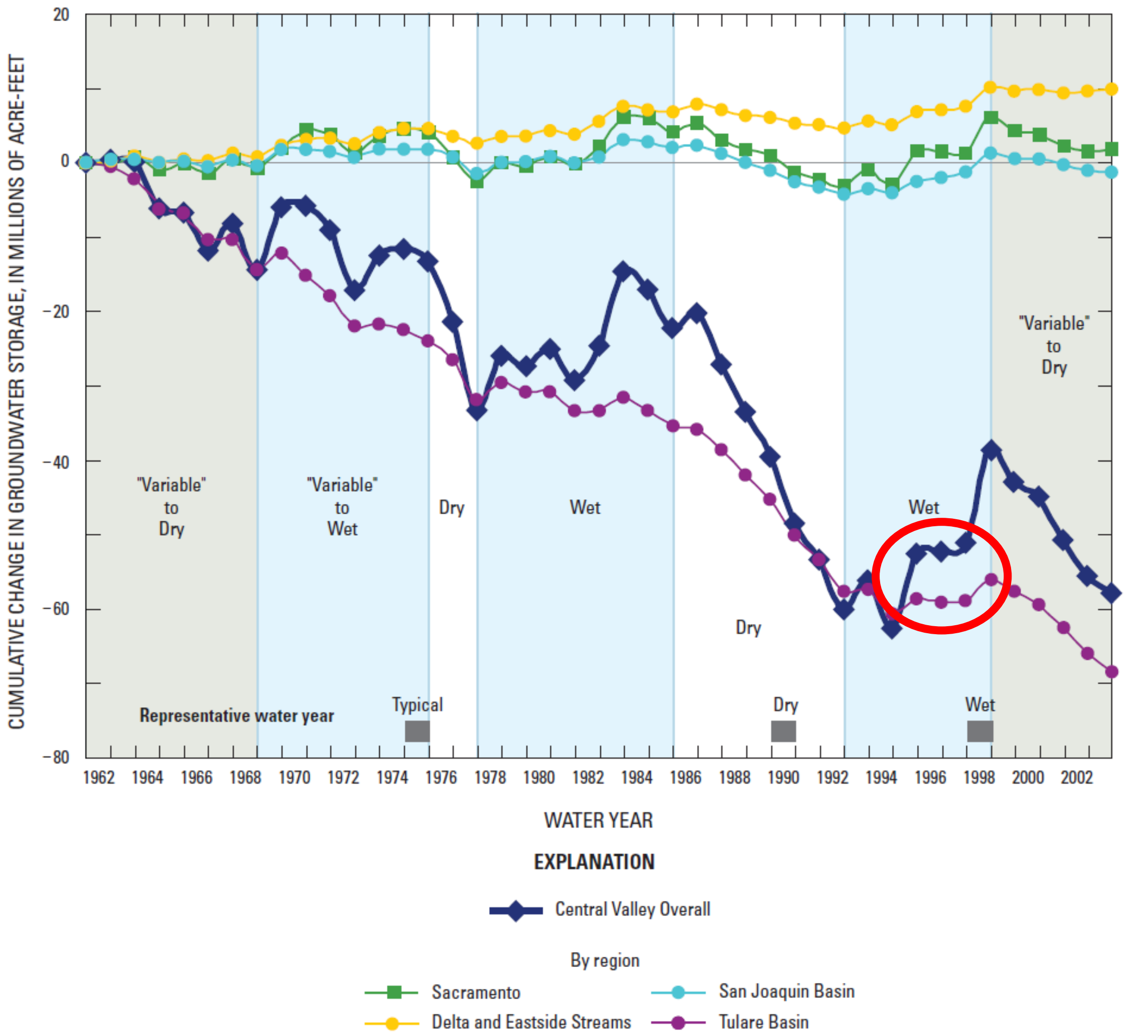

Figure 77. Simulated cumulative annual changes in aquifer storage (reprinted with permission from: Faunt 2009, p. 77, Figure B9.). Red circled area indicates the years chosen to represent steady-state stress conditions.

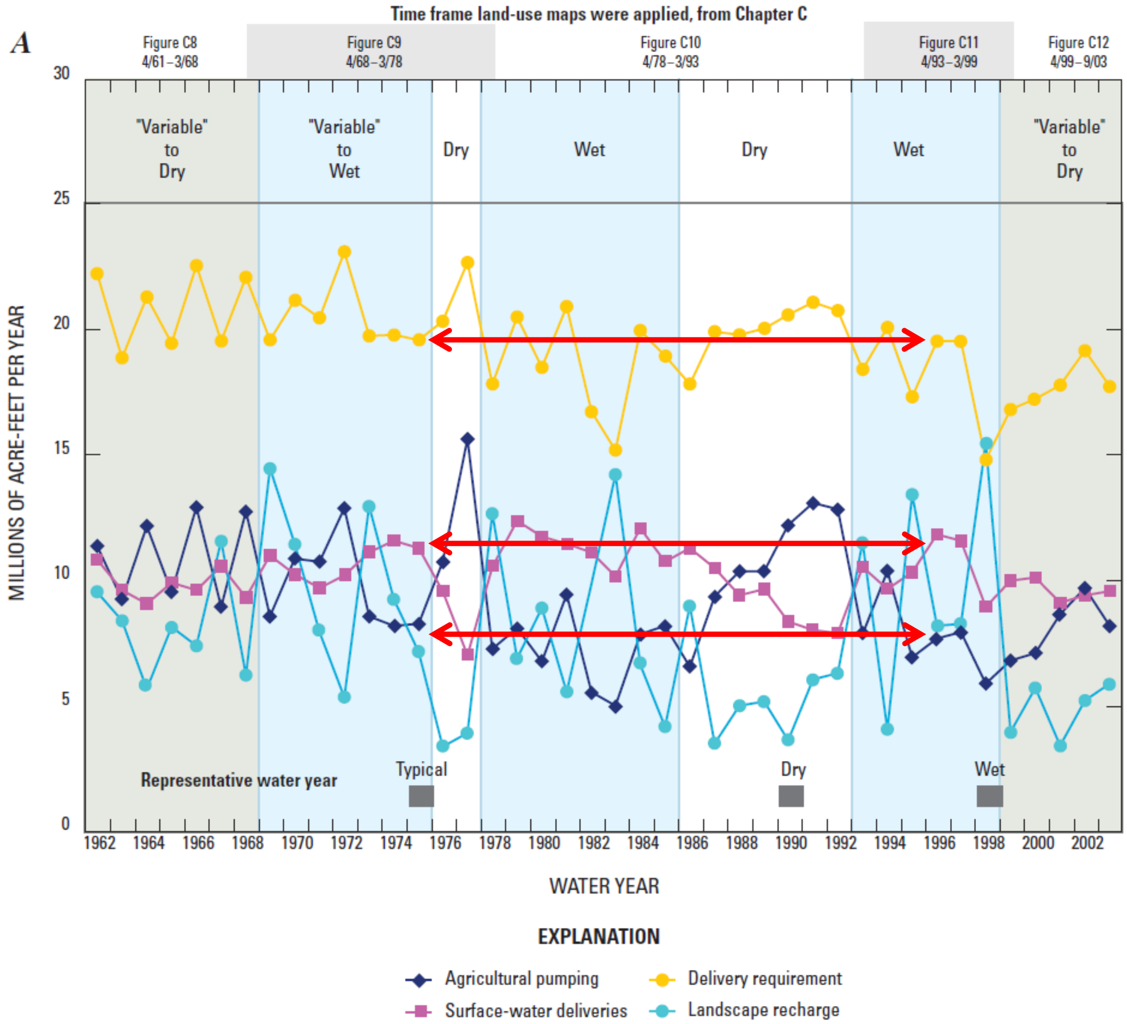

Figure 78. Water budget of CVHM, where "delivery requirement" is the irrigation water requirement from both groundwater and surface water sources. (Reprinted with permission from: Faunt 2009, p. 73, Figure B6.)

For the simulation we chose the water year spanning Oct. 1996 – Sep. 1997 as the stresses remain nearly constant compared to the previous year.

However, the CVHM model is a transient state model with a monthly step, where the head distribution, groundwater recharge, and groundwater pumping, vary over the year. To obtain a steady head field we average the monthly rates for the water year 1996. The mean hydraulic head relative to mean sea level is shown in Figure 79. The heads vary between -10 to 150 m (-33 to 492 ft). The head field is one of the main drivers of the velocity field and how the contaminants are transported through the aquifer. Therefore an accurate representation of head is very important. Figure 80 shows the cumulative distribution of the discrepancy between the averaged head field and each monthly head field of the year 1996. The discrepancy between the transient simulated head values and averaged head field is less than 1 m for 80% of the head values and less than 2 m for 90%.

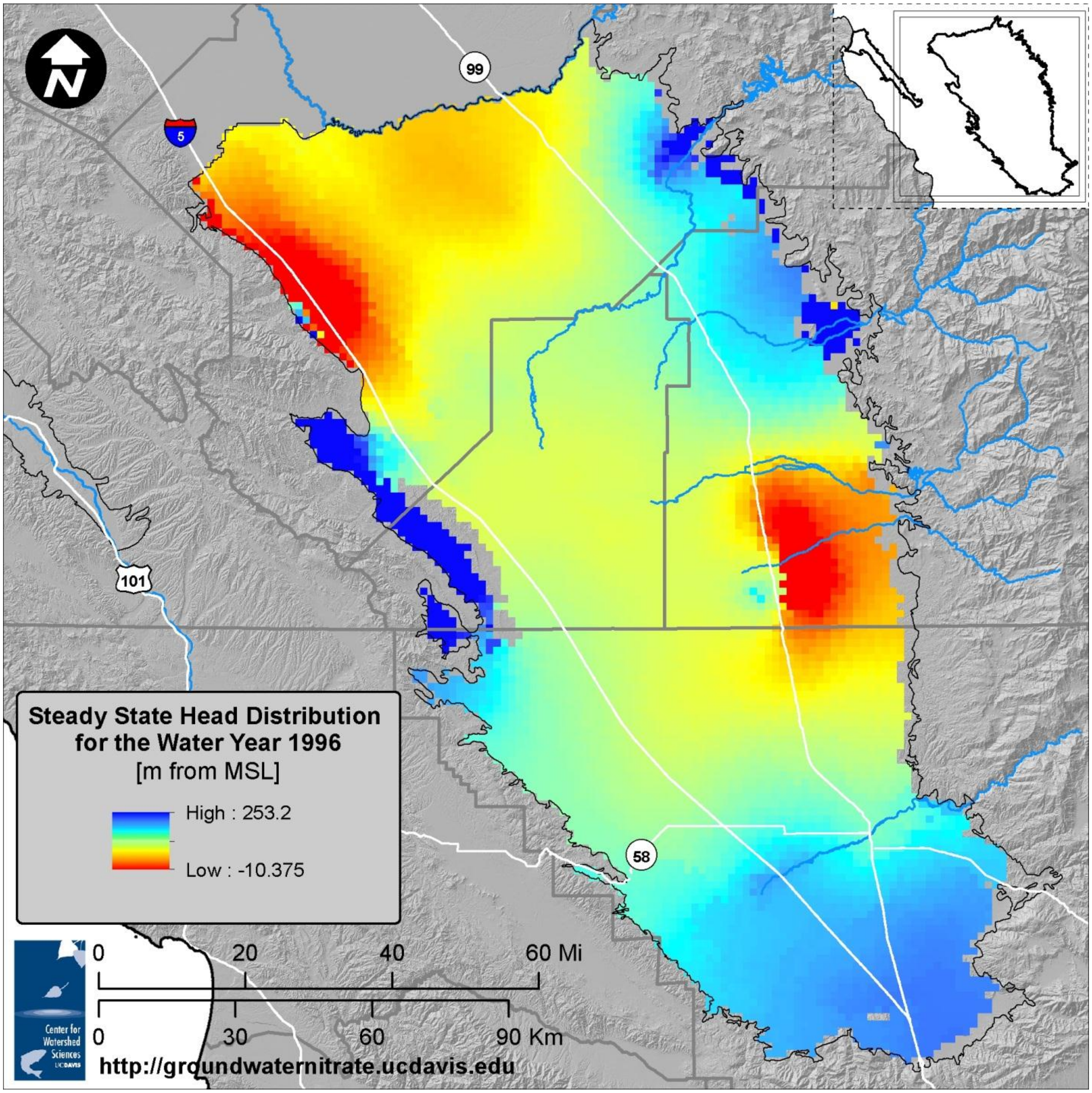

Figure 79. Steady state head distribution for the water year 1996.

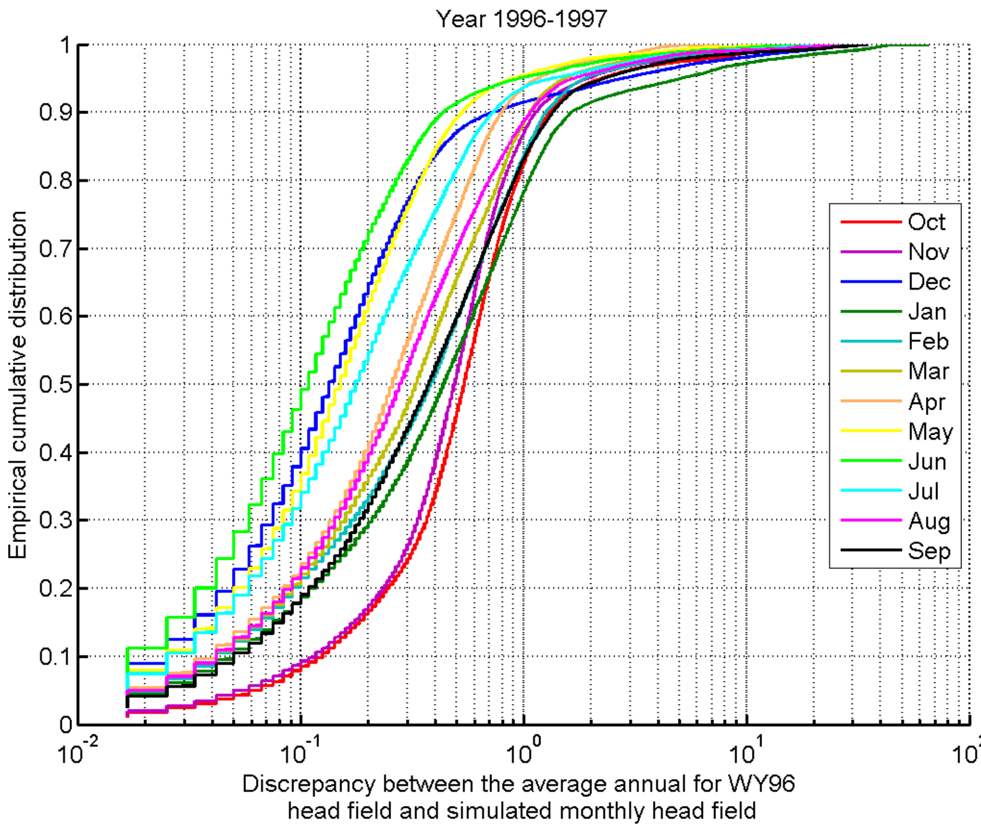

Figure 80. Comparison of averaged heads against the monthly heads used for averaging.

7.3.2 Conceptual Background

Well Generation Algorithm

The CVHM uses finite difference approximation with discretization of one square mile. Therefore various stresses such as groundwater recharge, groundwater pumping, and stream interactions, are assigned to the center of each cell, and the assigned values correspond to aggregation of all stresses located in each cell (e.g., the pumping rates of all the wells that are found in a square mile). To compute BTCs for wells with typical characteristics (i.e., pumping rates, well depths and screens that are closer to reality), in the refined model we developed an algorithm for generating wells with representative pumping rates, screen length, and depth. In fact the Well Generation Algorithm (WGA) is an automated way to distribute the well stresses back to individual wells. Since the location of the real wells is unknown, we use the WGA for generating random locations and random pumping rates, which satisfy the overall mass balance and honor the well characteristics, such as pumping rates, screen depth, and length typical of the study area. The input data for the WGA are the empirical distribution functions (ecdfs), which describe the well pumping rates, screen lengths and depths. The ecdfs used in this report are illustrated in Figure 81, Figure 82, and Figure 83.

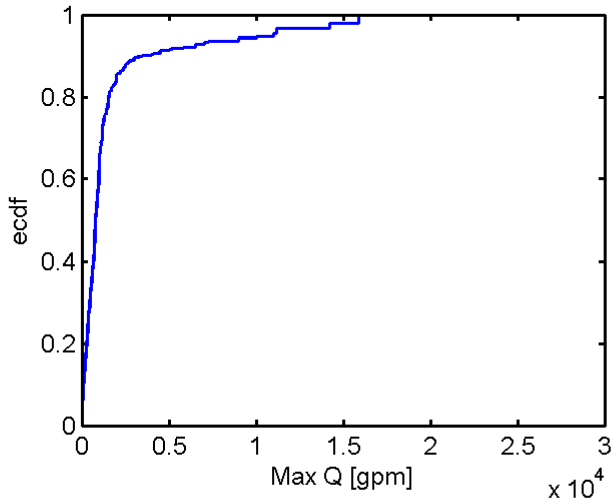

Figure 81. Empirical cumulative distribution function of maximum well pumping rate.

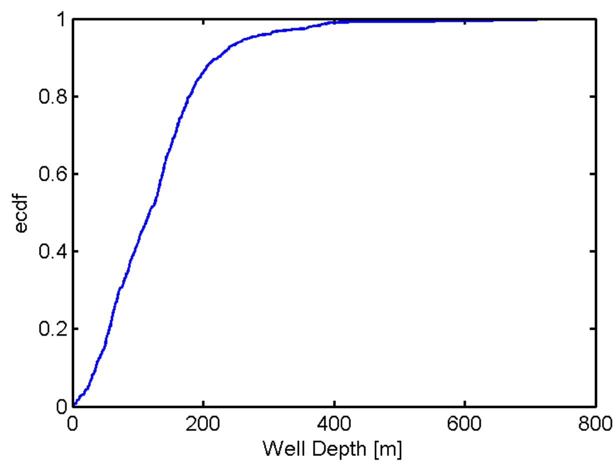

Figure 82. Empirical cumulative distribution function of well depth [m] (distance between water table and bottom of the well screen).

The ecdf of pumping rates is shown in Figure 81. Note that the data correspond to the maximum capacity of each well in the sample and not the actual pumping rate, which is unknown. In addition, note that most of the wells, if they operate at their maximum design capacity, they do so approximately 5 months per year (dry season) while during the remaining months, they either do not operate or operate at reduced rates. Based on expert opinion, it was suggested to sample from a subset of the full distribution, excluding the extremely large rates (>0.9) or negligible rates (<0.05). Note also that, in the simulation, we are interested in the yearly average rates, therefore the sampled rates were further reduced to correspond with yearly rates.

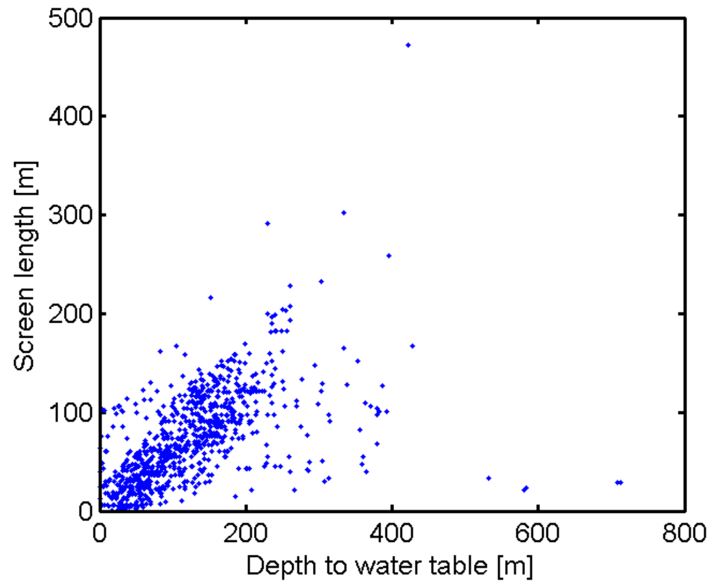

Figure 83. Correlation between well depth and screen length.

The steps of the WGA to generate locations and pumping rates are outlined as follows and explained in more detail below:

I. Choose a cell with assigned pumping rate
    1. Generate a well location
    2. Generate pumping rate and keep track of cumulative pumping
    3. If cumulative pumping is less than the cell assigned pumping rate, repeat steps 1 and 2
    4. Else, subtract the difference from the nearest cell, and proceed to the next cell

II. Generate random depth and screen length based on the distribution

III. If the depth is greater than 100 m assign the depth to a well with large production rate

IV. If the depth is less than 100 m assign the depth to any well

V. Check that there is 3 m (10 ft) of screen for every 100 gpm in coarse material. If not repeat steps II-V.

According to the pseudocode for each cell (square mile) of the CVHM discretization with an assigned pumping rate, we randomly generate wells within the cell using the formula $\mathbf{x} = \mathbf{x}^{min} + \mathbf{r}(\mathbf{x}^{max} - \mathbf{x}^{min})$ which generates uniformly distributed numbers within the specified limits (e.g., cell extent) where $\mathbf{x} = [x, y]$ are the coordinates of the generated well and $\mathbf{r}$ is a vector of two random numbers from the uniform distribution between 0 and 1. In addition, we specified a minimum threshold distance equal to 300 m, so that the wells are not in close proximity. The generation of the wells is a sequential process and each new well is accepted only if the distance with the closest existing well is greater than the threshold. To assign pumping rates, we sample randomly from a subset of the empirical cumulative distribution function. When the cumulative pumping rate of the generated wells per cell exceeds the rate of the cell, which is specified by the CVHM, we proceed to the next cell. However, to maintain the water budget and at the same time to honor the pumping ecdf, the difference between the total cumulative pumping rate and the specified CVHM rate, is subtracted from the nearest cell to the last generated well in the current cell. Based on the above approach the number of wells per cell (i.e., wells per square mile) is dictated first by the cell pumping rate, (e.g., the higher the rate the more wells will be generated) and secondly by the shape of the ecdf.

In total, the algorithm generated 5,486 wells within the TLB study area. It can be seen (Figure 84) that the average number of wells per square mile is 4 while 80% of the simulated wells are found at a density less than or equal to 5 wells per square mile. This is also in agreement with reality, according to expert opinions. The simulated well locations are illustrated in Figure 85. The regional spatial distribution of the simulated wells is dictated by the parent model, here CVHM, and the CVHM pumping fluxes for the representative period chosen for the steady-state flow field (here: 1996). The WGA introduces an artificial variability at the scale of and within the CVHM grid cells. Due to the much finer discretization, the spatial local distribution of the wells obtained from WGA is therefore different and more detailed than with CVHM.

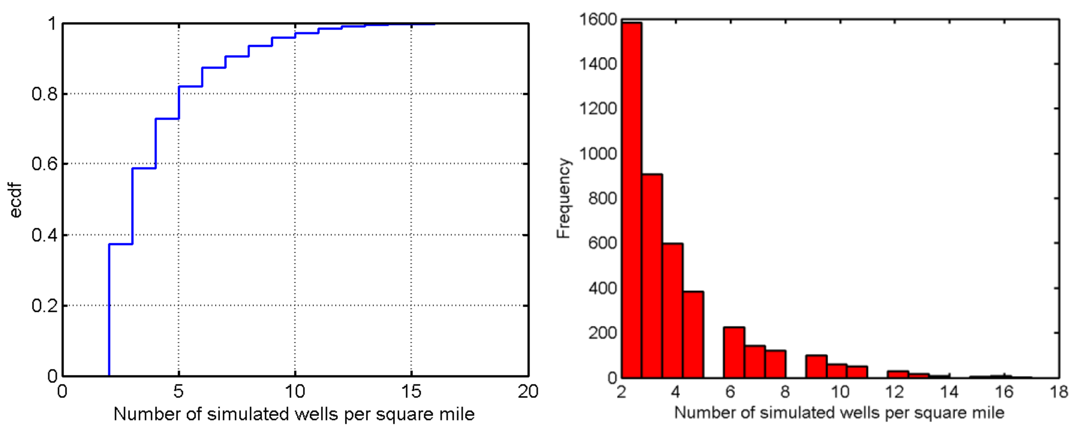

Figure 84. Left) Empirical cumulative distribution function of the number of wells per square mile. Right) histogram of the number of simulated wells per square mile.

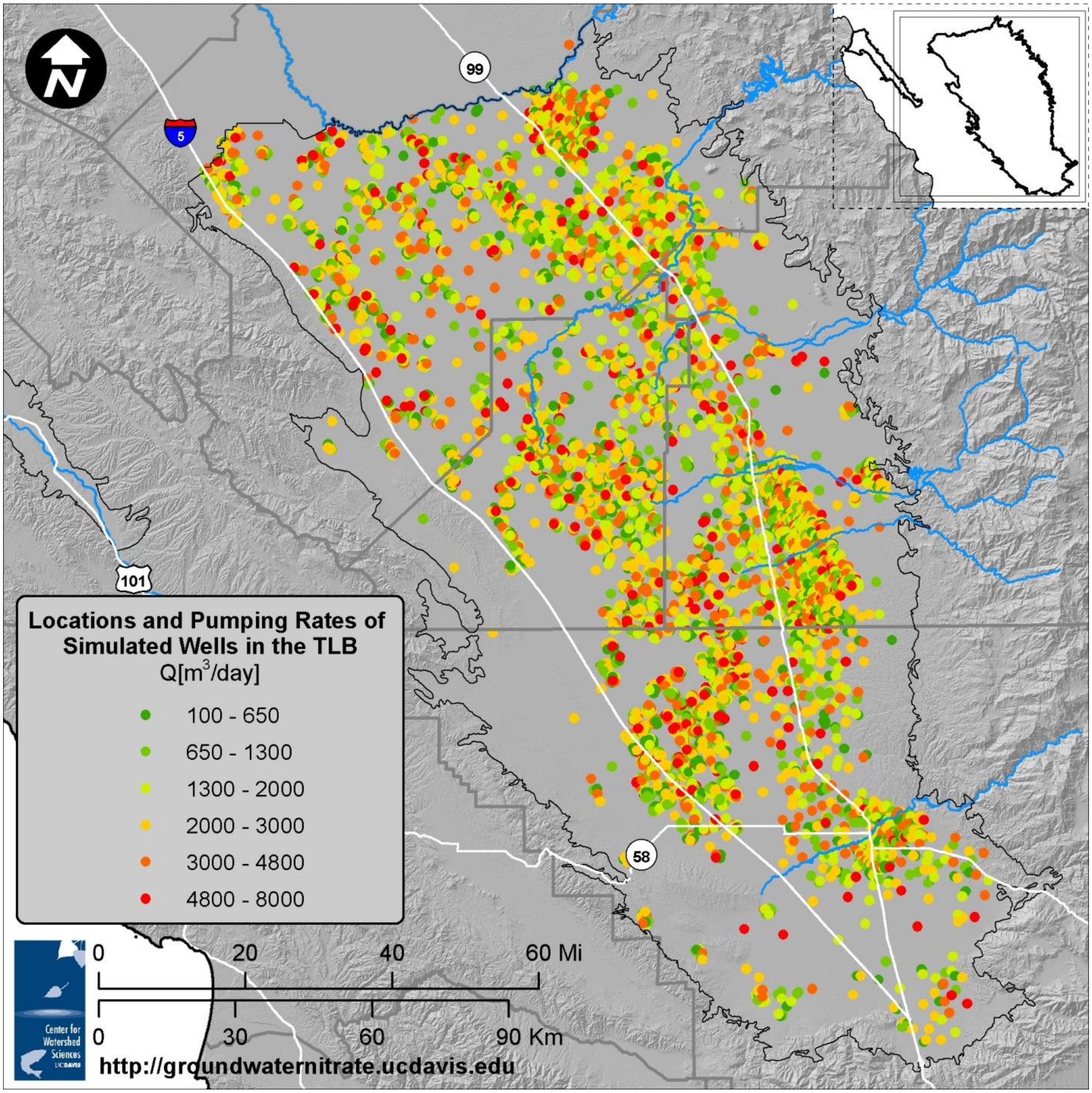

Figure 85. Locations and pumping rates of simulated wells in the TLB study area.

The distribution of pumping wells in Figure 85 reflects long-term average agricultural and urban water extractions, as represented by the year 1996 water budget (see discussion above and Figure 78). Under "normal year" conditions, surface water supplies along the easternmost portion of the TLB are sufficient to supply most irrigation water. Hence, most pumping wells in the easternmost part of the TLB are for urban uses (e.g., City of Fresno, City of Visalia, City of Porterville).

The distribution of pumping rates of the simulated wells is illustrated in Figure 86. Since we used a subset of the original distribution, the simulated distribution does not match the data, yet the median value is approximately equal to 800 gpm, assuming a 5-month pumping period, which is a reasonable assumption for the particular study area.

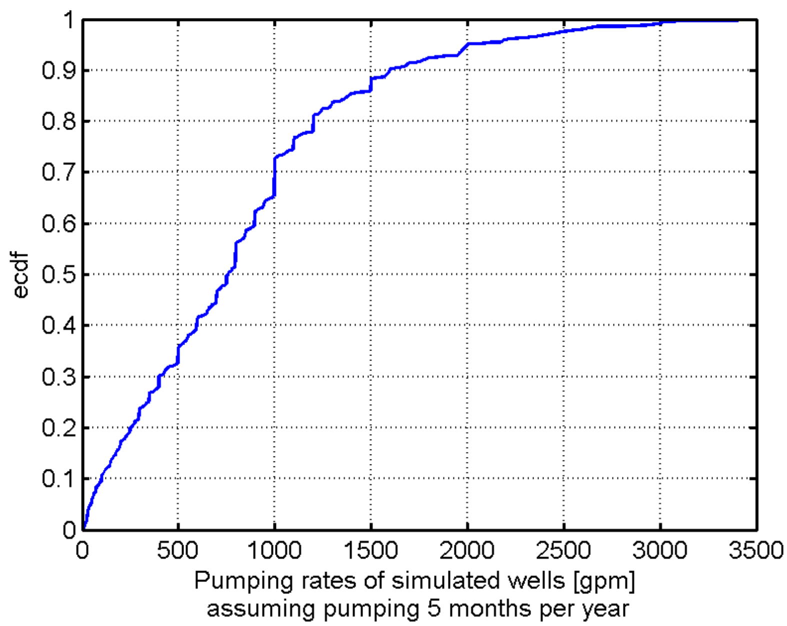

Figure 86. Empirical cumulative distribution function of simulated pumping rates.

After the generation of well locations and pumping rates, we assigned depth and screen lengths. Similarly the ecdf of well depths (Figure 82) and the correlation between screen lengths with depth (Figure 83) are used to generate random values. However special treatment is required to avoid assigning very large pumping rates to shallow wells. To do so we imposed a to sample from the upper half of the depth distribution when the pumping rate exceeds the median pumping rate of the pumping rate ecdf. However, when the rate is less than the median we use the full range of the distribution. In addition to pumping rate generation, special treatment is required for the screen length generation. Typically the shallow wells have shorter screen lengths than the deep wells. Figure 83 shows the depth of the well against the screen lengths. Ideally, an empirical bivariate probability density function is computed; however, this requires a large amount of data. Here we choose a two-step approach where, for each well, we first generate the depth and then the screen length. During the screen length generation, instead of sampling from the full distribution, we sample from a narrow range of the distribution. For example, suppose that d is the depth generated by sampling the depth distribution (Figure 82). To assign screen lengths we identify the samples with depth within the range $[r - d, r + d]$; hence, we use only a limited number of samples instead of the full distribution and construct a local ecdf, which describes the distribution of the well screen lengths in the vicinity of the selected depth d (Figure 87). The range r depends on the density of sample points (wells) around d and it is determined in an automated fashion. For each well, the algorithm starts with a very small value for the range r and increases the range until a sufficient number of sample points are found to construct a local ecdf.

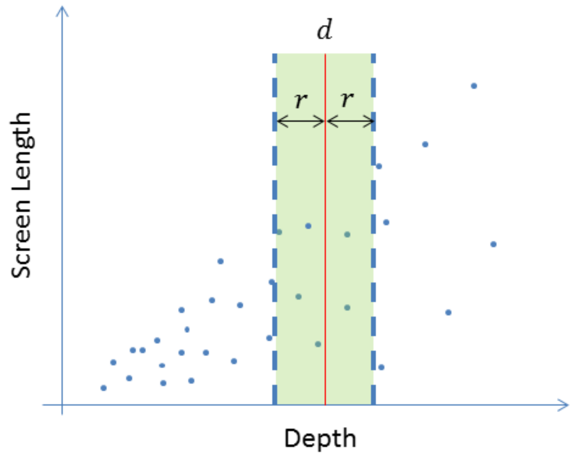

Figure 87. Sampling range for screen length.

Based on the depth and screen length distributions we assigned random well depths and screen lengths to the simulated wells that were generated in the previous step, according to their pumping rates. In order to generate an unbiased depth-screen length distribution, first we split the simulated wells into two groups. One group includes all the wells with simulated pumping rates greater than the median (i.e., 800 gpm), while the second group actually contains all the wells. Then we generate random depth d and random screen length based on the ecdfs. If the generated depth d is greater than 100 m (328 ft) then the well is designated as deep and the depth is assigned randomly to one of the wells that belongs to the first subgroup (of large production wells) and the well is removed from both sets. If the generated depth is less than 100 m then the pair of depth-screen length is assigned arbitrarily to one of the wells of the second group which contains all the wells. It can be seen that it is possible for wells with relatively small pumping rates to have large screen lengths, yet the suggested approach does not allow large wells to have small screen lengths. At the same time, since all the distributions are sampled independently, we avoid introducing bias. This becomes apparent by comparing Figure 82, which shows the ecdf of depths based on the real data and Figure 88 that shows the ecdf of the simulated well depths. The correlation between well depth and screen length is illustrated in Figure 89, which is very similar to the correlation of the real data (Figure 83). Last, we check if the material between the screen lengths can support the assigned pumping rate. We utilized an empirical formula where, for each 100 gpm we require at least 3 m (10 ft) of coarse material (e.g., K> 10 m/day (33 ft/day)). If the well does not meet this restriction, we repeat the depth and screen length generation.

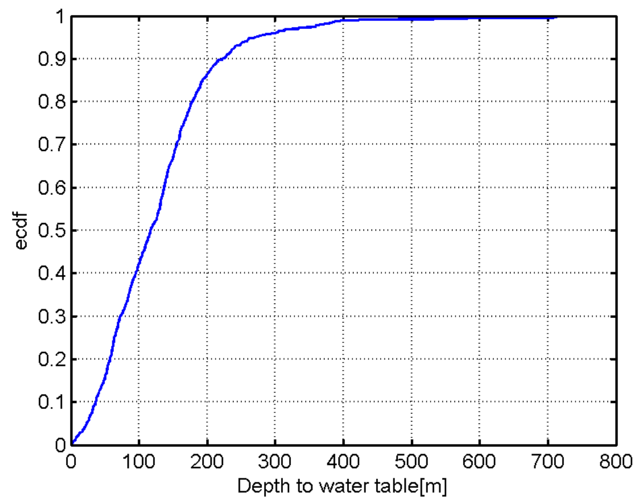

Figure 88. Empirical cumulative distribution function of Simulated well depth (distance between simulated water table and bottom of the well).

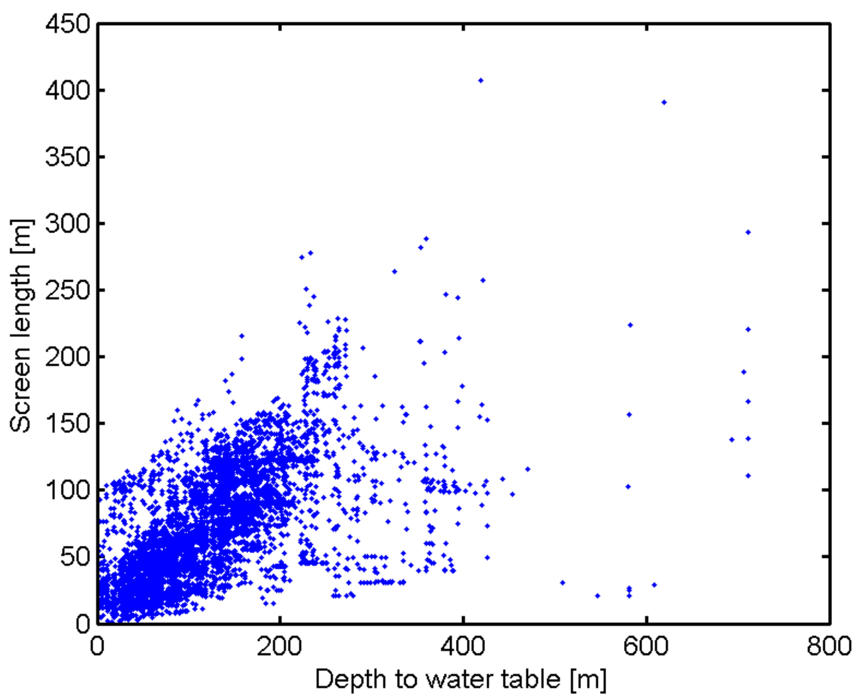

Figure 89. Correlation between simulated well depths and screen lengths.

7.3.3 Groundwater and Transport Simulation

After the generation of wells, all the additional stresses, which were defined as monthly rates, such as groundwater recharge stream leakage interactions, were averaged to obtain representative yearly rates. The hydraulic conductivity values and the ratios between horizontal and vertical conductivities were obtained from the input files of CVHM model.

To simulate the steady state flow field that corresponds to the water year 1996, we use finite element discretization, where the discretization varies from few meters ~10 m (33 ft) near wells to several hundred meters. Figure 90 illustrates an example of the fine discretization that was used in the steady state solution. The red cells correspond to the cells of the CVHM (i.e., each cell is 2.6 km (1 sq.mi)). For the vertical discretization we used 25 layers as opposed to the 10 layers used in CVHM.

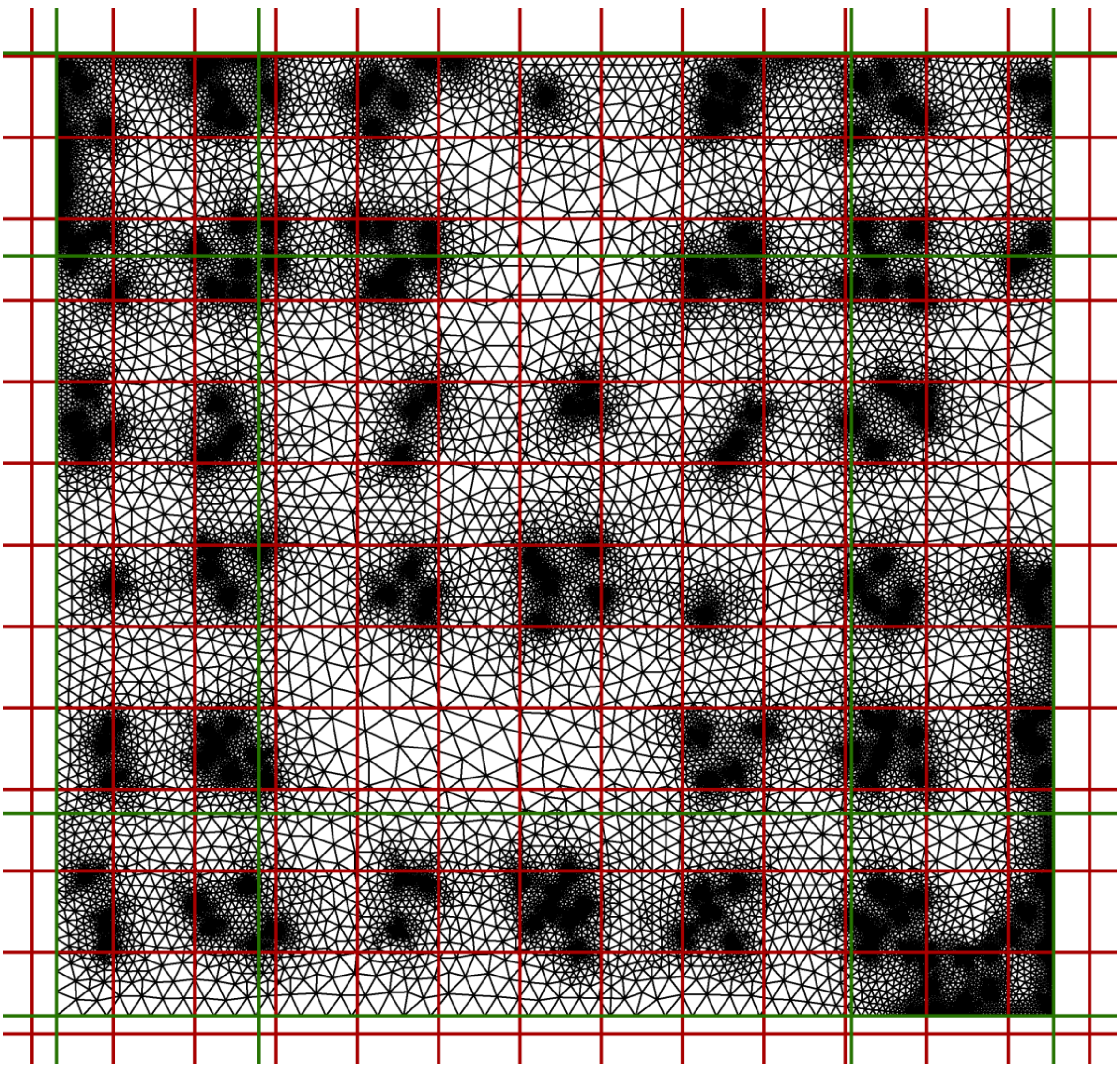

Figure 90. Discretization based on finite elements. The red lines correspond to the cells of the CVHM model. The dense black areas indicate the highly detailed discretization around the wells.

To obtain a steady state solution at this level of discretization, we employed the simplistic domain decomposition method presented above. According to the method, the domain was first divided into overlapping subdomains. Figure 92 shows the division of the study area into overlapping subdomains. In total, the subdomain was divided into 97 overlapping subdomains. The dimension of each subdomain was approximately 20 x 20 km., while the width of the overlapping zone was constant and equal to 2 km.

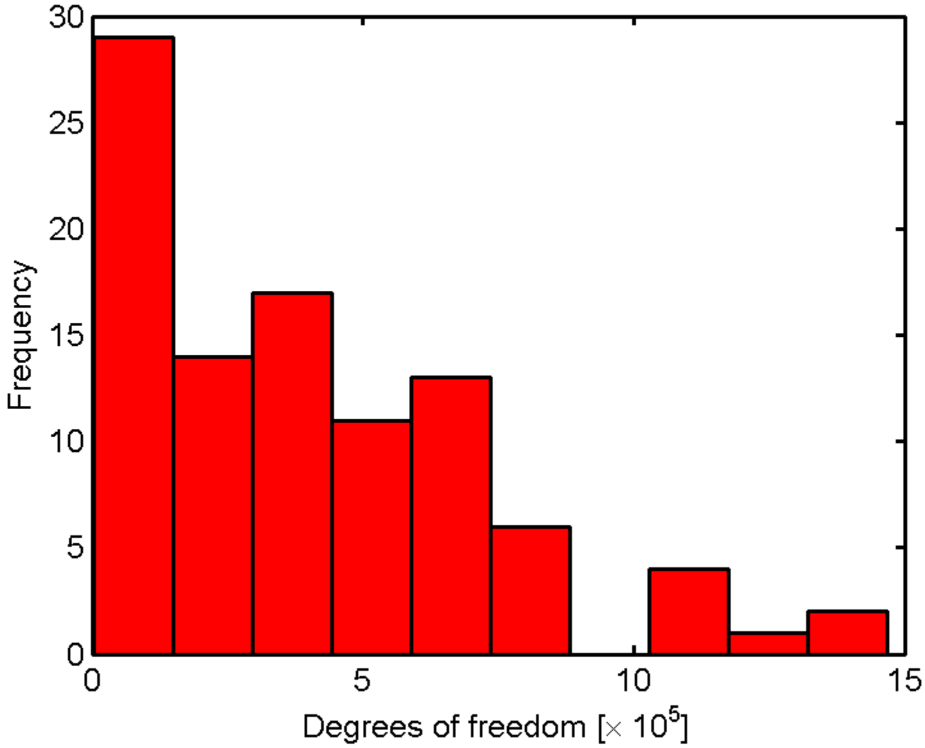

Figure 91. Histogram of degrees of freedom for the simulated subdomains.

The degrees of freedom for each subdomain vary from $3.1 x 10^3$ to $14.8 x 10^5$ (Figure 91). The total number degrees of freedom is $39 x 10^5$. Note that in our approach the meshes of adjacent subdomains were not conforming (i.e., within the overlapping zone the nodes of two subdomains do not coincide). According to the simplistic domain decomposition method, each subdomain was solved independently. The boundary conditions for the inner boundaries of each subdomain were interpolated from the coarse solution of CVHM. To obtain a divergent free velocity field we used the proposed (section 7.2.2) weighted averaging scheme.

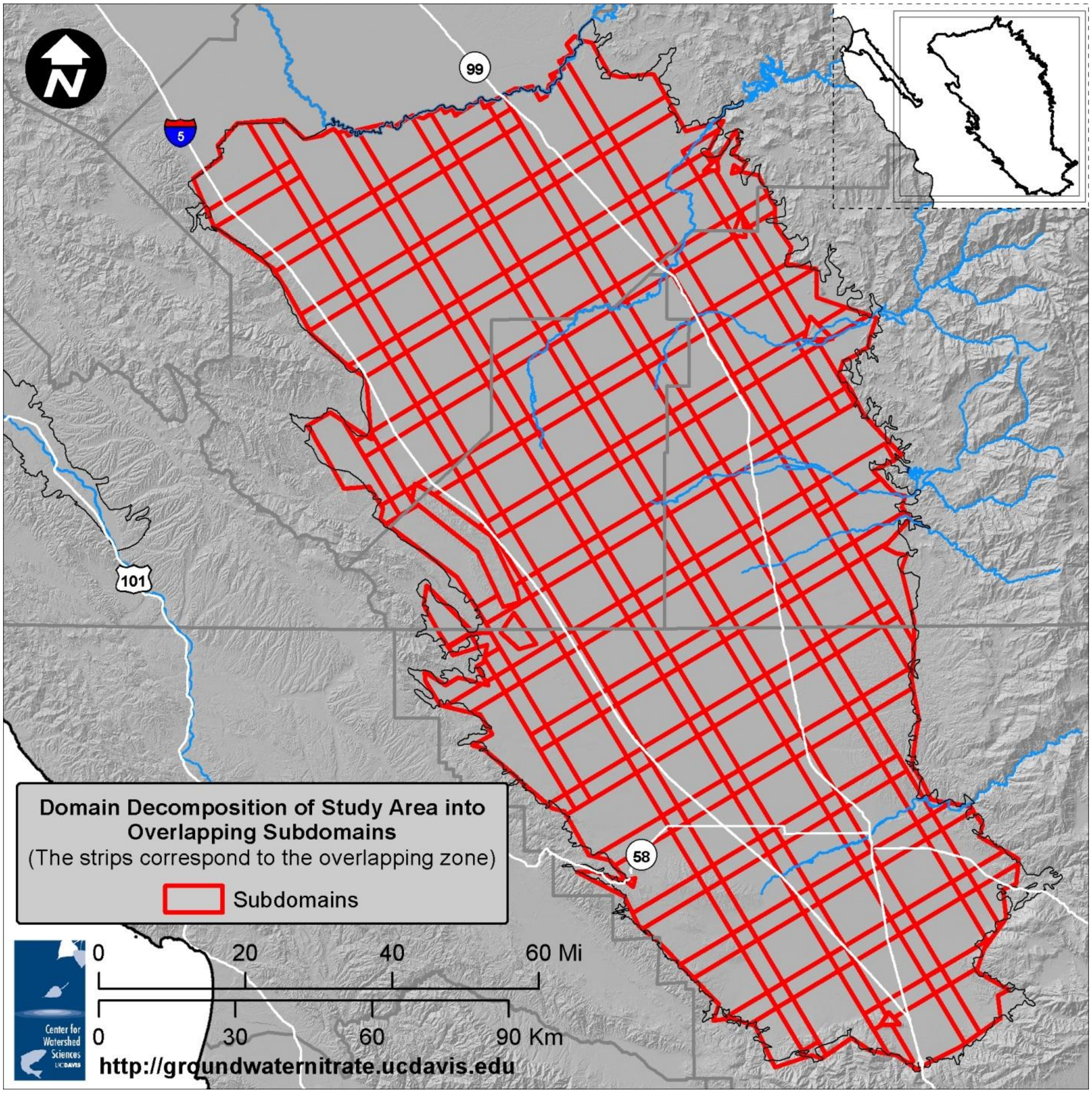

Figure 92. Domain decomposition of study area into overlapping subdomains. The strips correspond to the overlapping zone.

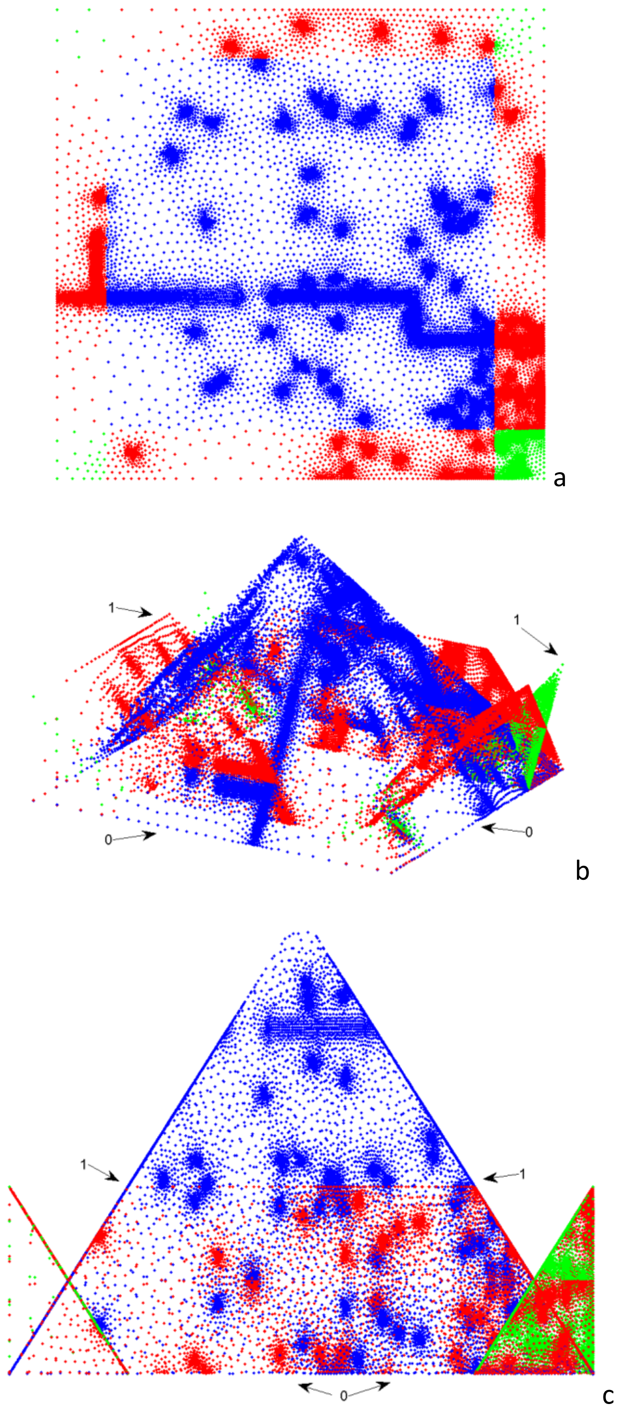

Figure 93. Illustration of weighting scheme between adjacent subdomains.

The hydraulic heads for the nodes within the non-overlapping area were obtained directly from the head solution of each subdomain. For the nodes in the overlapping zones, first, we computed the weights associated with each subdomain and then the node heads were calculated as the weighted average. Figure 93a illustrates the nodes of one of the subdomains. To keep the figure simple the triangularization has been removed from the figure. The blue dots in Figure 93a correspond to the nodes within the non-overlapping area. The nodes with red correspond to the overlapping zone between 2 subdomains and the green to the overlapping -zone between 4 subdomains. In the other two panels, Figure 93b and Figure 93c, the height of each node corresponds to the weight associated with each subdomain. It can be seen that the weight of the main subdomain (blue dots) is zero at the boundaries and increases linearly. Similarly, the weights of the adjacent subdomains are zero at their boundaries and increase linearly as well (see red and green dots). The final head field is the weighted average of the subdomain head distribution. For example, the heads of the main subdomain (blue dots) of Figure 93 at the boundaries will be given zero weight, since the error is expected to be large due to the interpolated (from the coarse solution) boundary constraints. As we move towards to the center of the main subdomain (blue dots) the velocity obtained from the main subdomain will be given higher weight, while the weight from the adjacent subdomains decreases. Hence, the overlapping zone creates a smooth transition zone between subdomains

The head and subsequently the velocity field were used for particle tracking. According to the methodology, for each well we released 100 particles, uniformly distributed, around the well screen and tracked backwards until they exit the aquifer. Since the number of wells is 5,486, the simulated streamlines equal to 548,600 (Figure 94).

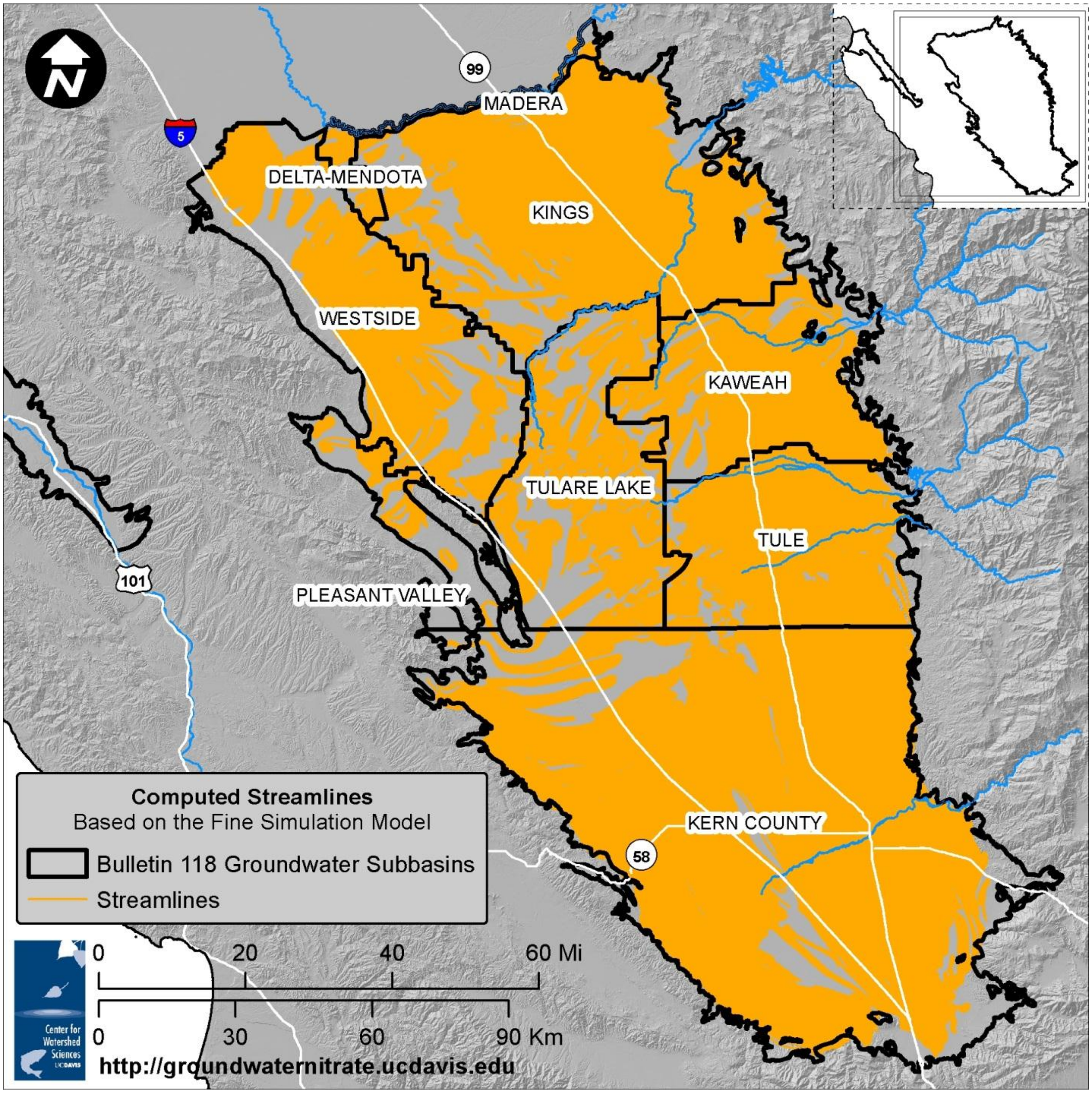

Figure 94. Computed streamlines based on the fine simulation model.

Then, for each streamline the 1D ADE transport problem was solved to obtain unit response functions. Here we used the analytical solution of 1D ADE where the longitudinal dispersivity was a function of streamline length and the velocity was set equal to the advective velocity. Note that, during the 1D transport problem, we assumed unit input loading to obtain a unit response function for each streamline, which was subsequently stored into a GIS database and was used for predictions based on different loading scenarios. Figure 95 illustrates the computed URFs for two of the simulated wells. We choose one shallow and one deep well to point out the different responses one should expect. It can be seen that the travel time distributions for shallow wells are shorter between few years and few decades, as opposed to deep wells where the range of travel time distribution spans from several decades to centuries. In addition, the shape of the URF for shallow wells is spiky, indicating that shallow wells exhibit greater concentrations compared to deep wells.

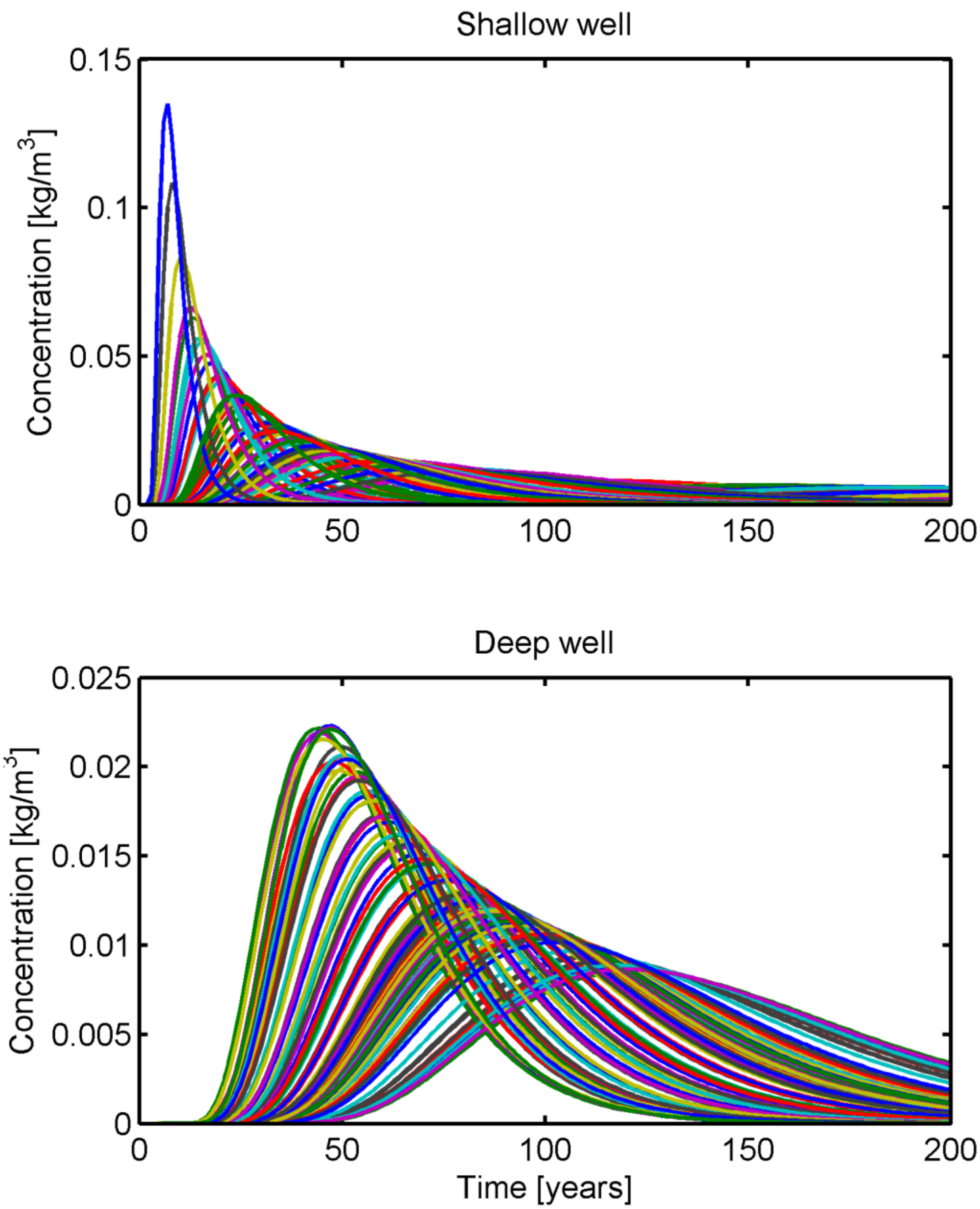

Figure 95. Unit Response Functions.

7.4 NPSAT Implementation Based on Coarse Model

In addition to the highly detailed model, we applied the NPSAT using the CVHM coarse approximation. Similar to the detailed simulation, to obtain an average steady state flow field for the water year 1996, we averaged the monthly flow fields which were obtained from the output of CVHM. Based on analytical particle tracking (Pollock 1994), we perform backward particle tracking until the particles exit from the water table. The particle starting points are identical to those used in the detailed model. Yet, due to coarse approximation, there is no cone of depression around each simulated well, resulting in an unrealistic shape of streamlines around the wells, which can be seen locally (Figure 98).

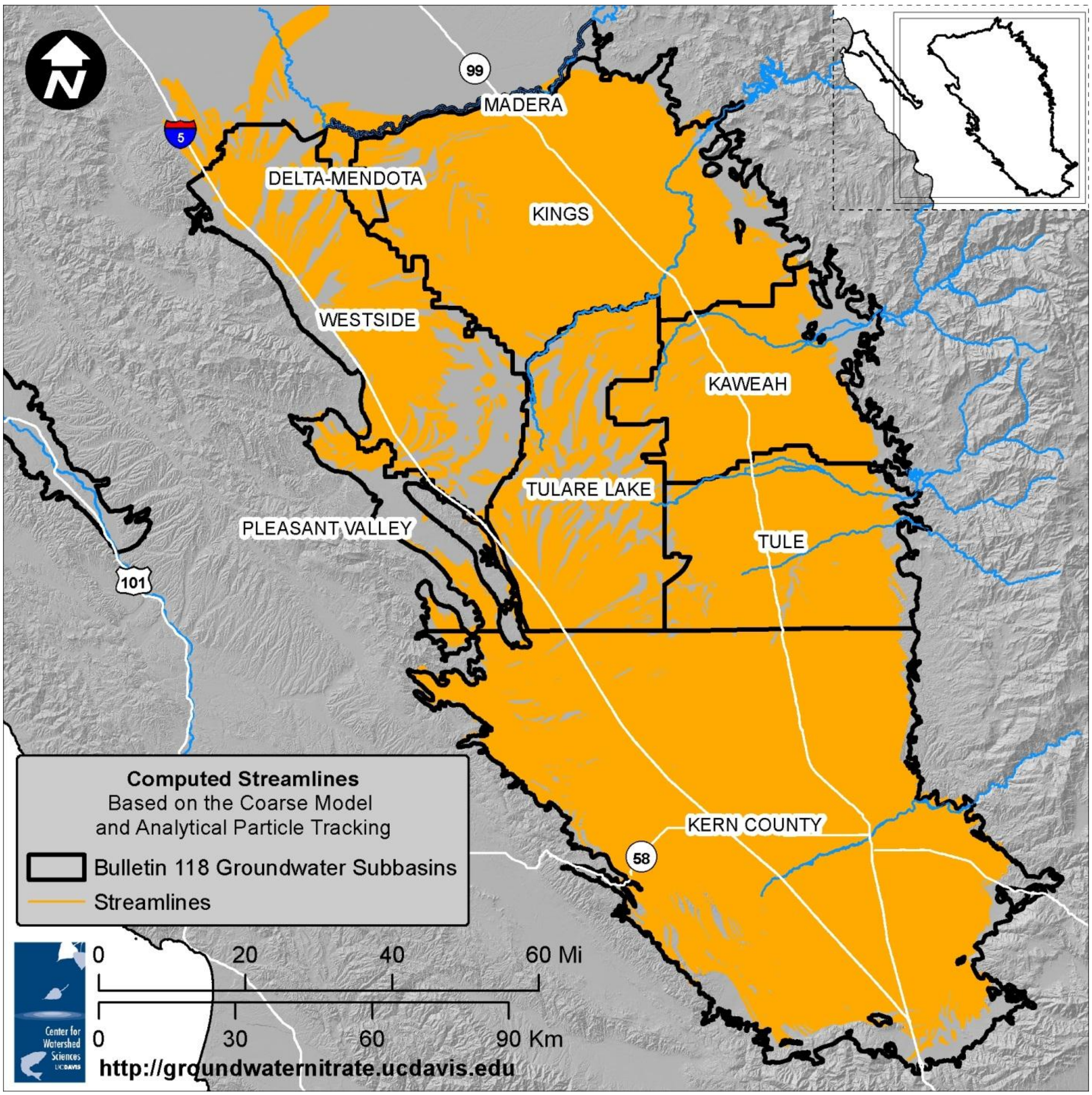

Figure 96. Streamlines based on coarse model and analytical particle tracking.

The streamlines based on the coarse approximation are shown in Figure 96. It can be seen that, in general, the streamlines follow the same pattern as in the highly detailed simulation. In both cases the streamlines cover the study area and in both cases there are fewer streamlines in the basins of Tulare Lake and Westside compared to the other basins. This can be attributed to the fact that there are fewer wells, but also that the particle travel distance is shorter in some cases.

Although the average shape of streamlines seems similar in both cases, the statistics of the streamlines reveal significant differences (Figure 97). One of the major differences is the distribution of the streamline length. In the coarse approximation the logarithm of streamline length appears normally distributed with a mean value of about 15,000 m (15 km), while for the highly detailed simulation, the distribution exhibits two peaks, one close to 6,000 m (6 km) and one close to a few hundred meters. The significant difference in streamline length is reflected also in the age distributions. While the logarithm of age appears normally distributed, the mean value for the detailed model is approximately 1.8 (~65 years), but it is 2.1 (120 years) for the coarse model. In addition there is a significant percentage of streamlines in the detailed model with travel time less than a year.

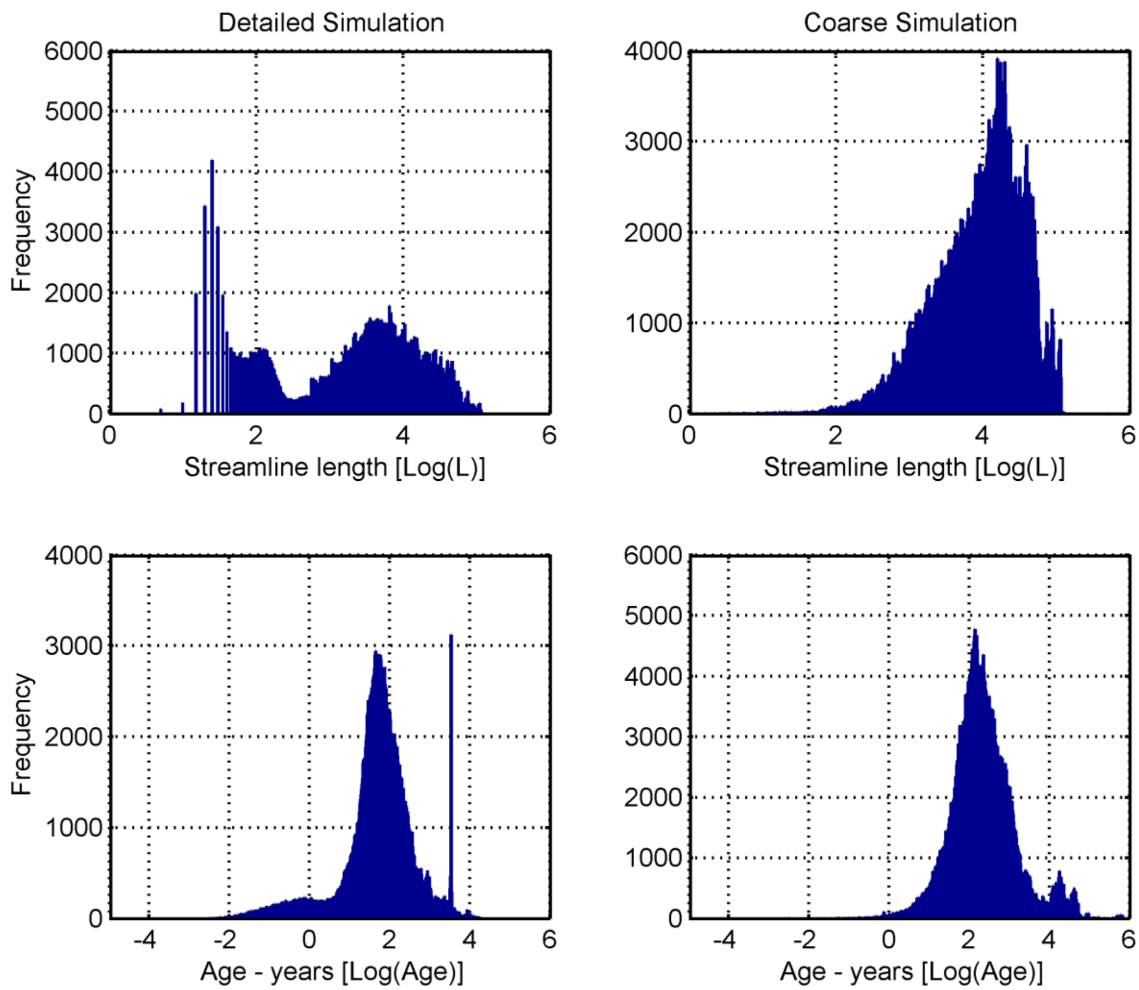

Figure 97. Comparison of the distribution of streamline length [m] and age [years] between the two simulation models. A log of 0, 1, 2, 3, and 4 corresponds to 1, 10, 100, 1000, and 10,000 units (m or years), respectively.

One of the main reasons for these differences is the fact that, in the highly detailed simulation, the hydraulic head field was depressed around the wells, due to fine discretization. Note that in coarse simulation the local cone of depression around the simulated wells was absent. Although in both models the particles originated from the exact same positions, in the fine model the particles are actually closer to the water table, due to the cone of depression, which was absent in the coarse mode. Therefore for the fine model the particle's travel time to exit the aquifer was shorter compared to the coarse model, as well as the length of their streamline. In addition, it was not unusual that part of the screen length was above the water table and the particles above the water table in the detailed model were ignored. To illustrate the difference we have plotted the streamlines for one of the wells (Figure 98 top panel) and zoomed around the well (bottom panels. The blue lines correspond to the coarse model and the red to the highly detailed. The streamlines shown here were selected as representative. Indeed, in most cases the streamlines based on the coarse model where longer and narrower compared to the detailed model, due mainly to the absence of a cone of depression around the well.

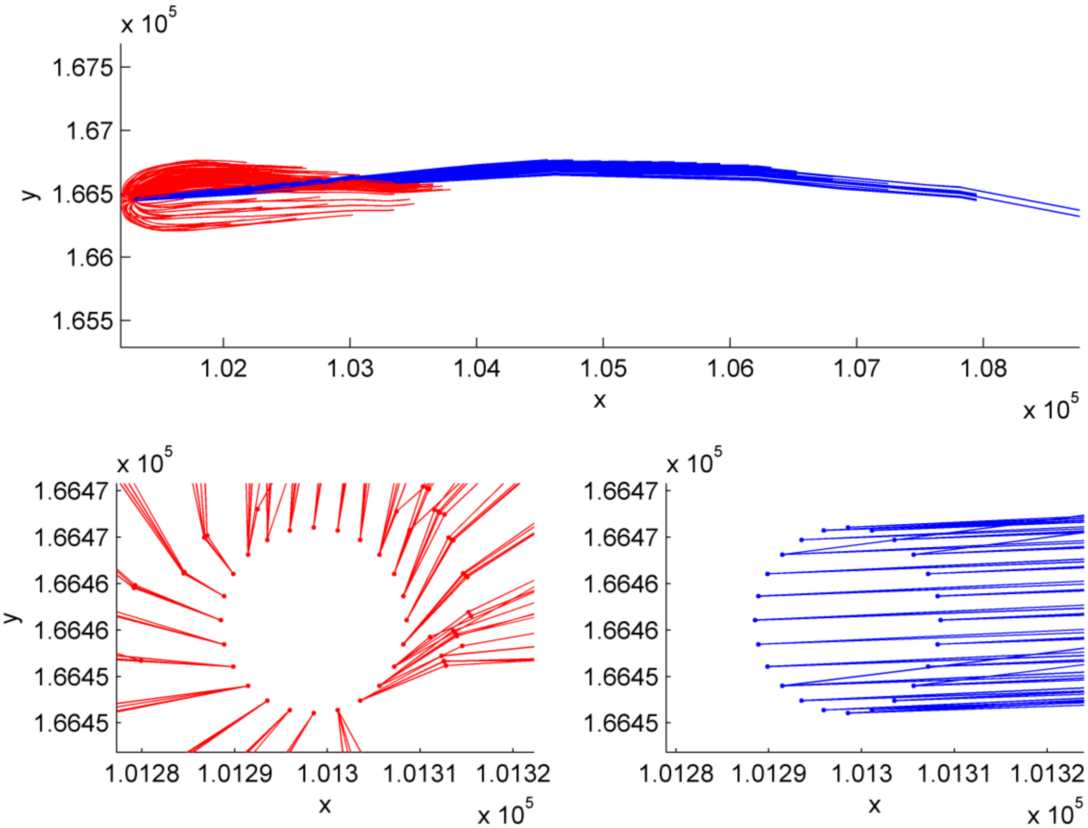

Figure 98. Comparison between streamlines computed with the detailed and coarse model. Red streamlines correspond to detailed model and blue to coarse model. Top panel shows the overall streamline shapes of the 100 streamlines for a particular well. The bottom panels show the orbits of particles around the wells. Due to the absence of a cone of depression on the coarse model (blue lines) the streamlines are actually straight lines.

7.5 Results

7.5.1 Nitrogen Loading Scenarios

The computed Unit Response Functions (URFs) can be rapidly convoluted with alternative loading scenarios to calculate real breakthrough curves for each well. In this report we examined the impact of four alternative nitrogen loading scenarios. A detailed description of the development of N loading scenarios is discussed in Section 1.8 of Technical Report 2 (Viers et al. 2012). Briefly, the four scenarios represent different management of excess manure N (nitrogen) exported from dairies after 1980: In Scenario A, exported manure N either leaves the study area or is incorporated into nutrient management on non-dairy crops such the typical amount of N applied to crops does not change. In scenario B (by study area) and C (by study area), half and all of exported manure, respectively, is applied to non-dairy cropland across the study area as an amendment in addition to typically applied fertilizer N resulting in higher groundwater nitrate loading. In scenario D, all manure N is applied on manured crops within dairies resulting in very high nitrate loading to groundwater on dairies, but elsewhere identical to Scenario A. The approach taken to modeling manure applications is also used to model food processing waste application, and application of biosolids and WWTP effluent nitrogen, whether it is to cropland or directly into recharge facilities.

The output of the N loading algorithm developed by Viers et al. (2012) is four alternative scenarios of the N leaching into into the groundwater table. The leaching rates are given in kg/ha for 8 different years (1945, 1960, 1975, 1990, 2005, 2020, 2035, and 2050). For the simulation purposes we assumed that the recharge is constant over the 106 years of simulation, while the nitrate leaching rates vary linearly between these yearly values. Groundwater water recharge is a very important parameter, not only due to its influence on the flow field, but also because it determines the final leaching concentration rates. The loading histories need to be converted in units kg/m^3 (i.e., concentration). Therefore, prior to convolution of loading functions with the URF, we divided the N rates with the recharge. Figure 99 shows the cumulative distribution of loading rates assigned to each streamline. The left panels shows the N loading leaching rates in kg/ha (e.g., as they were computed by the N loading algorithm), while the right panels show the actual concentration that is assigned as input to each streamline. Note that this plot takes into account the loading rates for each streamline. Therefore, when multiple streamlines originate from the same land parcel, the corresponding rate is counted multiple times. During the prediction phase this does not affect the outcome as the final breakthrough curve is the weighted average of the streamline BTC (eqn. 21)

Similarly, in Figure 100 we plotted the exceedance probabilities of half the MCL, the MCL, and twice the MCL, that is the probability that wells exceed half the MCL, the MCL, or twice the MCL. Note that these three concentration limits will serve as reference values in the prediction phase. It can be seen that, in year 1945, approximately 40% of streamlines (i.e., 40% of the whole well source area) were associated with land parcels that leach with rates of half of the MCL, while 20% and 13% exceed the MCL and twice the MCL, respectively. Having these conditions as starting points, loading increases somewhat linearly until the year 2005, where 83%, 68% and 55% of the streamlines are associated with land parcels that exceed the 22.5 mg/L, 45mg/L and 90 mg/L, respectively. For future predictions, four different loading scenarios are considered (see details in Section 1.8 of Technical Report 2, Viers et al. 2012). Outside of dairies, there is no difference in N loading to groundwater between “Scenario A” and “Scenario D” as manure exports either don’t affect typical N application or there are no manure exports to areas outside dairies. Post-1990, these two scenarios represent significantly less loading than “scenario B (study area)” and “scenario C (study area)”, which apply half and all exported manure to cropland, respectively, in addition to typical fertilizer applications. No manure exports occur in 1975 and prior to 1975 in any of these scenarios. Hence the loading distribution across the landscape is identical between the scenarios for 1945 – 1975.

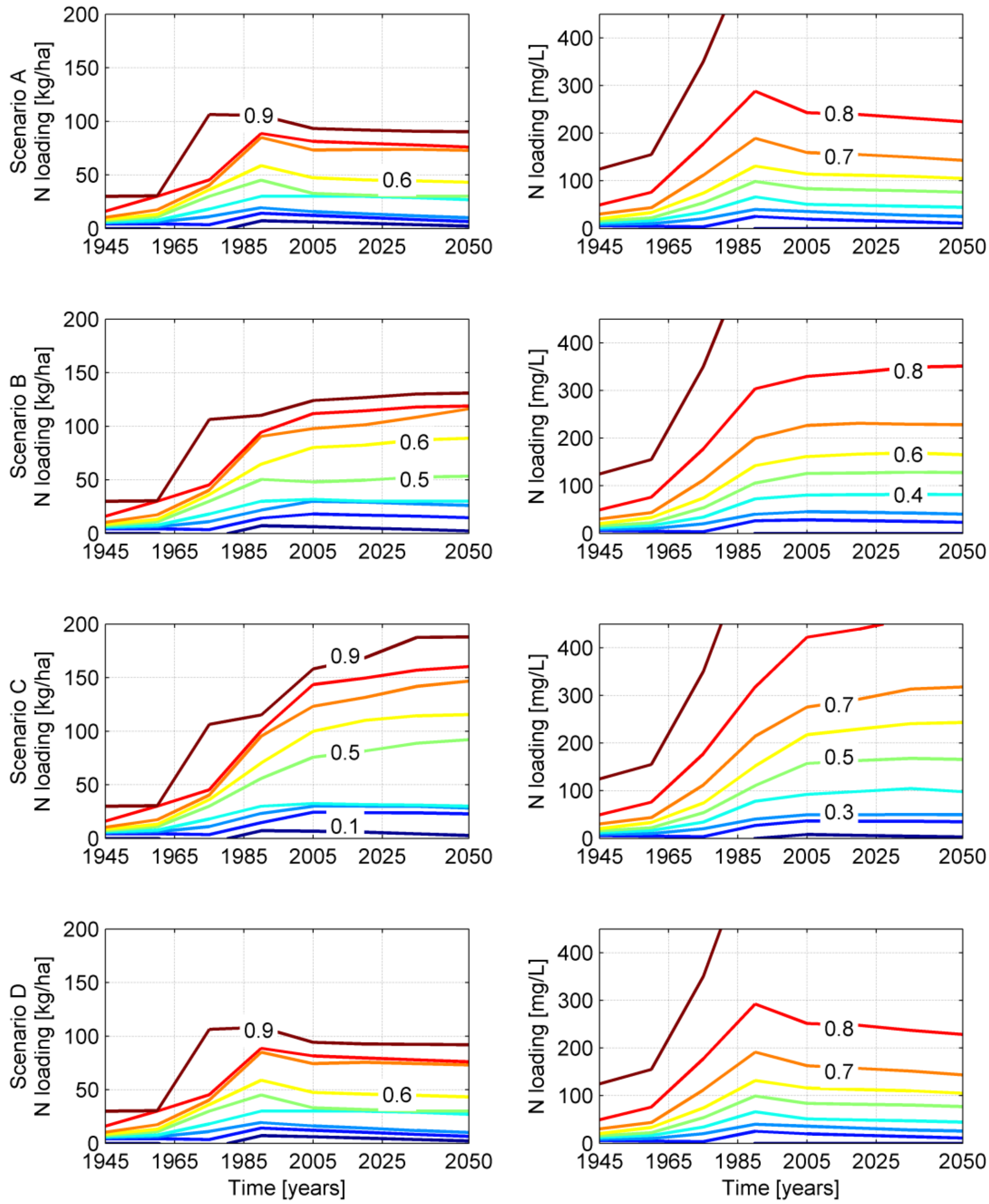

Figure 99. Cumulative probability distribution of N loading per year assigned to each streamline. The left panels correspond to N loading with units kg/ha -output of N loading algorithm Technical Report 2 (Viers et al. 2012), while the right panels correspond to the actual loading of N as nitrate (45 mg/L drinking water limit), which is used as input to NPSAT simulation model. All scenarios here are "by study area" and are explained in Section 1.8 of Technical Report 2 (Viers et al. ,2012).

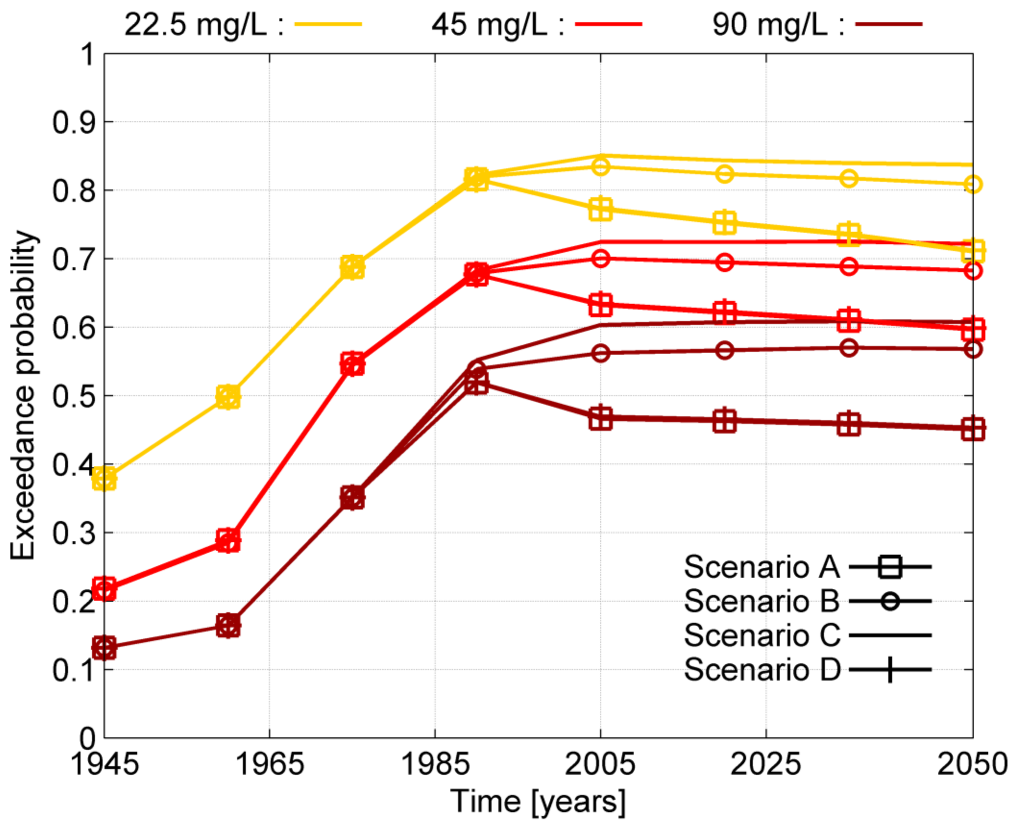

Figure 100. Exceedance probabilities of N loading rates as nitrate. All scenarios used here distribute excess nitrogen "by study area" and are explained in Section 1.8 of Technical Report 2 (Viers et al. ,2012).

The above loading functions were computed based on the detailed simulation model. Yet, when the coarse simulation model is used to convert the rates to concentrations, the figures change significantly. Figure 101 illustrates the cumulative distribution functions of N loading over time based on the coarse model. While the output of the N loading algorithm is identical in both cases (compare left panels in both figures) the loading distribution is primarily a function of the starting points of streamlines (i.e., the point where the particles exit the aquifer during backward particle tracking). The four scenarios exhibit similar behavior as described previously (i.e., in scenario A and D the leaching rates are reduced after 1990, in scenario B [by study area] leaching rates remain nearly constant, and in scenario C [by study area] there is an increase in loading), but the magnitude of the overall loading is less compared to the detailed scenario. The main reason is that, in coarse simulation, the particles were likely to connect wells to the higher recharge areas, therefore, during the conversion from mass kg/ha to mg/L the concentrations appear to be smaller. The main reason for that can be attributed to the nature of the particle tracking algorithm that was used in each case (e.g., analytical particle tracking for the coarse case and numerical particle tracking for the detailed simulation.) Furthermore, the streams for the coarse model are simulated by one square mile cells, while in detailed simulation the streams were assumed line sources with width approximately 50 m. It was found that, in the coarse model, a larger number of streamlines originate from cells associated with streams. Finally, the exceedance probabilities of the three reference concentration limits are shown in Figure 102. By comparing Figure 100 and Figure

102 we see that the N leaching into the groundwater concentration rates for the coarse model are 5% to 10% less than those of the fine model.

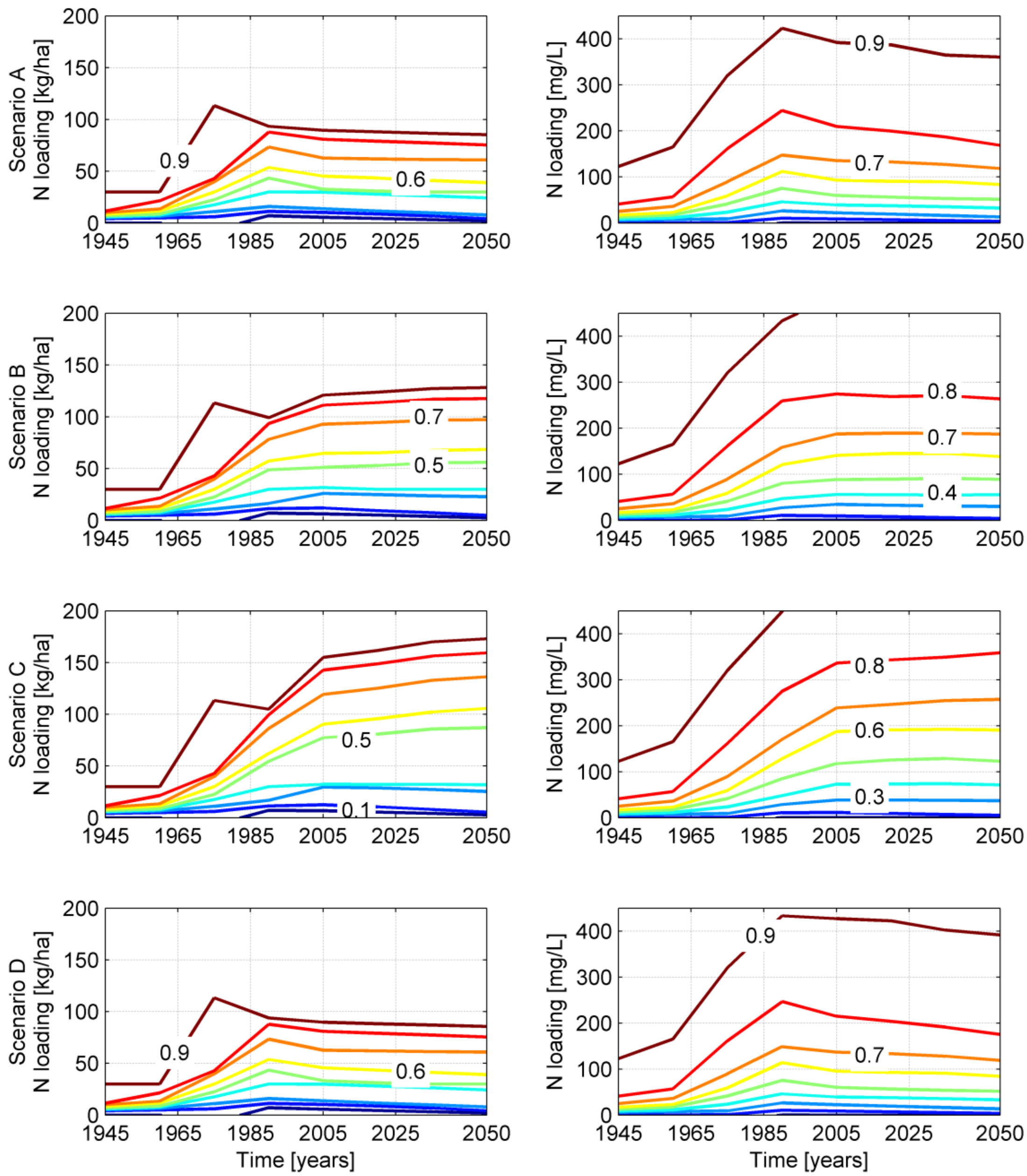

Figure 101. Cumulative distribution function of N loading based on the coarse model. All scenarios used here distribute excess nitrogen "by study area" (see Section 1.8 of Technical Report 2, Viers et al. ,2012).

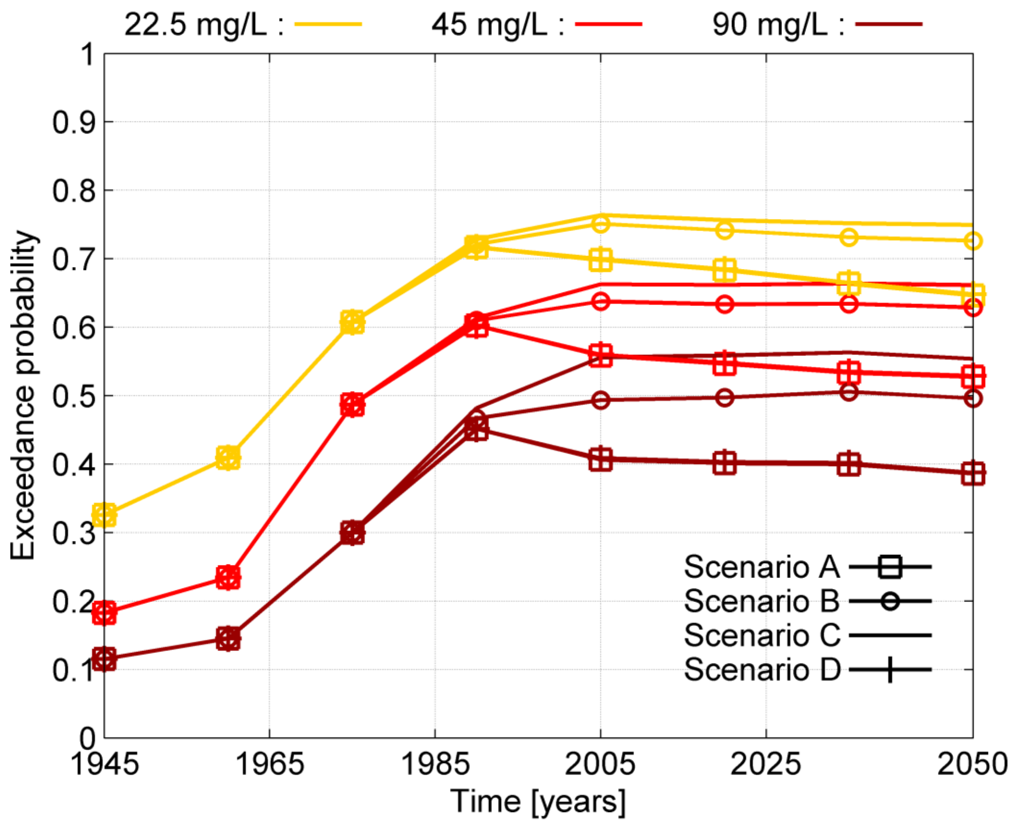

Figure 102. Exceedance probabilities of leaching rates as nitrate based on coarse model.. All scenarios used here distribute excess nitrogen "by study area" (see Section 1.8 of Technical Report 2, Viers et al. ,2012).

7.5.2 Model Validation

Prior to using the model for future predictions we perform a validation step, where the outcome of the model is compared against real data that have been gathered from various sources (section 4.4). The rationale for the data gathering and processing is explained in detail in section 4.4; however, for completeness of the section we summarize the main steps. In the well database there is a large number of samples; however, these samples are not uniformly distributed, neither spatially nor temporally. To alleviate both non uniformities the study area was divided into a number of equal area cells (Figure 39). Then, starting with 1950 and through 2010, for each cell, we compute the median of the annual well concentration means on a decadal basis. At the end of this process we obtain one representative concentration value for each cell for each decade. To calculate a de-clustered decadal average concentration for a particular region we compute the mean and the confidence interval of the mean of the decadal medians of the cells located in that region after they are log transformed. The back-transformed decadal means for the three regions of the TLB (Figure 39) are shown in Figure 103 and Figure 104, with the 95% confidence interval of the mean.

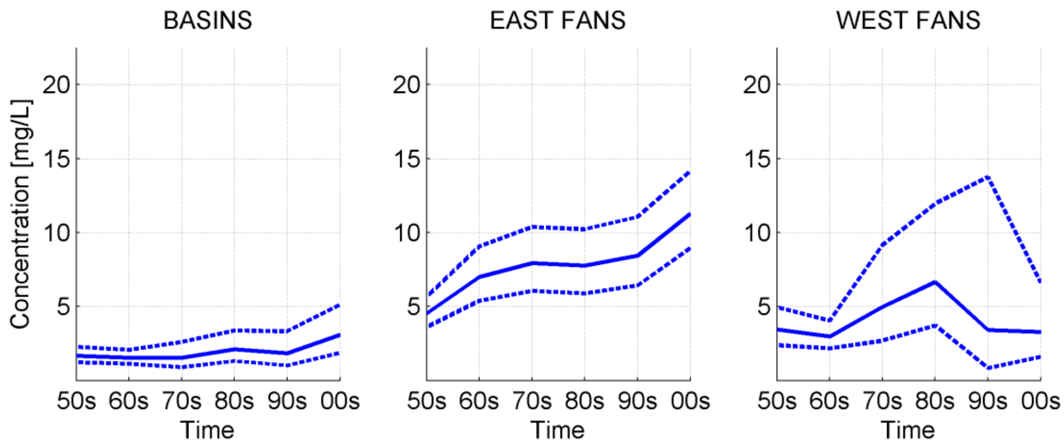

Figure 103. Decadal means of nitrate concentration for the three regions based on measured data. The solid line represents the mean and the dashed line the confidence interval.

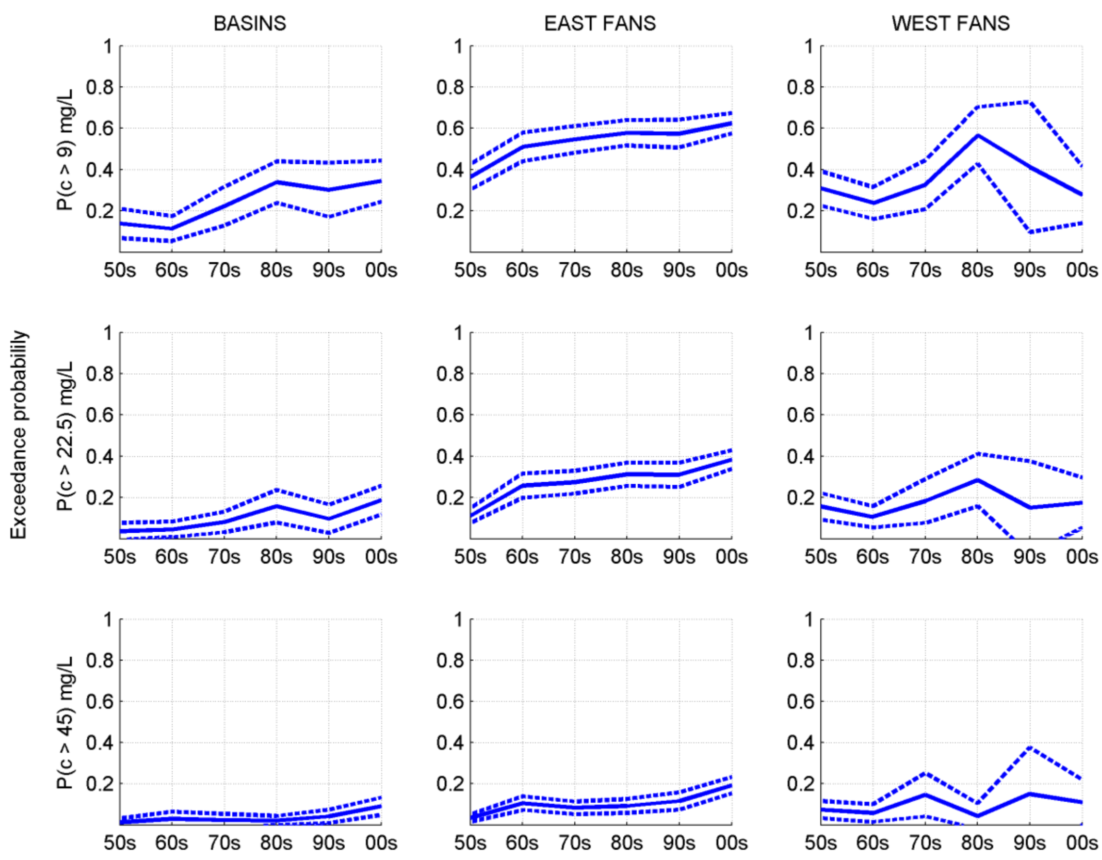

Figure 104. Decadal exceedance probabilities based on measured data. The solid line represents the de-clustered mean exceedance probability within each region and the dashed line the 95% confidence interval.

Next, the measured data are compared against the simulated trends, based on the two modeling approaches, the fine and coarse model. Similar to the measured data, the simulated results were grouped into the same equal area cells and the de-clustered decadal means and confidence intervals of the means were calculated as described previously.

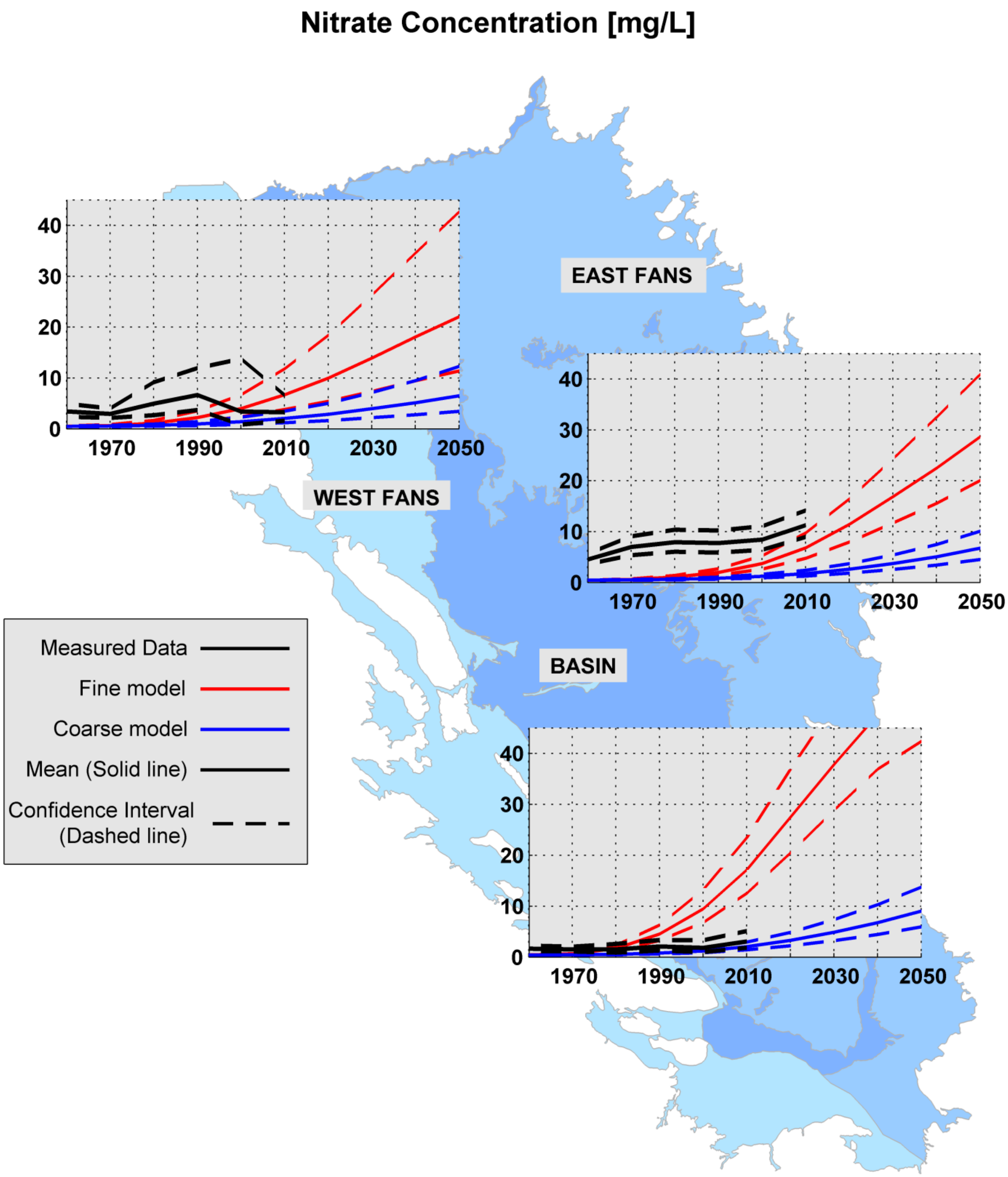

Figure 105. De-clustered, back-transformed mean of equal area log nitrate medians in each region of annual well means. Comparison of mean and 95% confidence interval between measured data and model predictions.

First we compare the de-clustered mean concentration data against the model responses (Figure 105). The solid lines represent the de-clustered, back-transformed mean of log transformed decadal medians of annual well means calculated per equal area cell and the dashed line represents the confidence intervals. In general, both models fail to follow the early decades 1950–1980, with an exception for the Basin region, yet this is due to the very low measured concentrations. Note that, in all regions, the responses of both models are practically identical until the 1970s and both calculate very low concentrations at the beginning of the simulation. This is due to the fact that the groundwater nitrate loading model does not account for nitrate loading from any sources prior to 1945 – groundwater is assumed to be "clean" in 1945. The model also neglects any background nitrate concentration, which is typically on the order of ~4 mg/L nitrate or less. A second reason is that the groundwater model does not include shallow domestic wells with very low pumping rates, which are often affected before the larger production wells with longer screens, simulated here, are affected by higher nitrate concentrations. For the Basin region, the coarse model fits better to the data compared to the fine model. In the Basin region, the fine model predicts a more rapid increase in nitrate concentrations than the measured data suggest. For the Eastside Fans region, early nitrate data are higher than predicted by either model. For the 1990s and 2000s, the fine model, predicting a more rapid increase than the coarse model, comes relatively close to measured de-clustered mean. The coarse model predicts, on average, very low concentrations, less than 10 mg/l, while the upper limit of fine model is approximately identical with the lower limit of the data for the last decade. Note also that the slope for the last decade is somewhat similar to the slope suggested by the fine model. For the Westside Fans region, there is similar discrepancy as in the Eastside Fans region. The fine model predicts a slightly upward trend for the 1980s, which becomes more apparent during the 1990s, then significantly overpredicts the observed average low nitrate concentration measured during last decade (2000-2010). The coarse model underpredicts even further the higher average nitrate concentrations observed during the 1980s in the Westside Fans region, but appears to better predict the more recently observed low average nitrate concentrations. We caution that for the Westside Fans, the measured decadal mean values are based on much fewer measured data than for the other regions, with the source of data and the associated types of wells measured varying significantly over decades (see discussion in Section 5). Overall, it appears that the response of the fine model is closer to what is measured for the Eastside Fans than the coarse model. In contrast, the coarse model, showing a slower rise in nitrate concentrations than the fine model, due to the above discussed inherent conceptual model differences, compared better to the measured average nitrate concentrations in the Basin and Westside Fans regions. Neither model takes into account denitrification, which may be a significant process within the Corcoran Clay separating the upper from the lower aquifer system in the Basin and Westside Fans region. Future modeling efforts may be performed to accommodate denitrification in the Corcoran Clay and other portions of the aquifer system, for which denitrification rates would have to be determined.

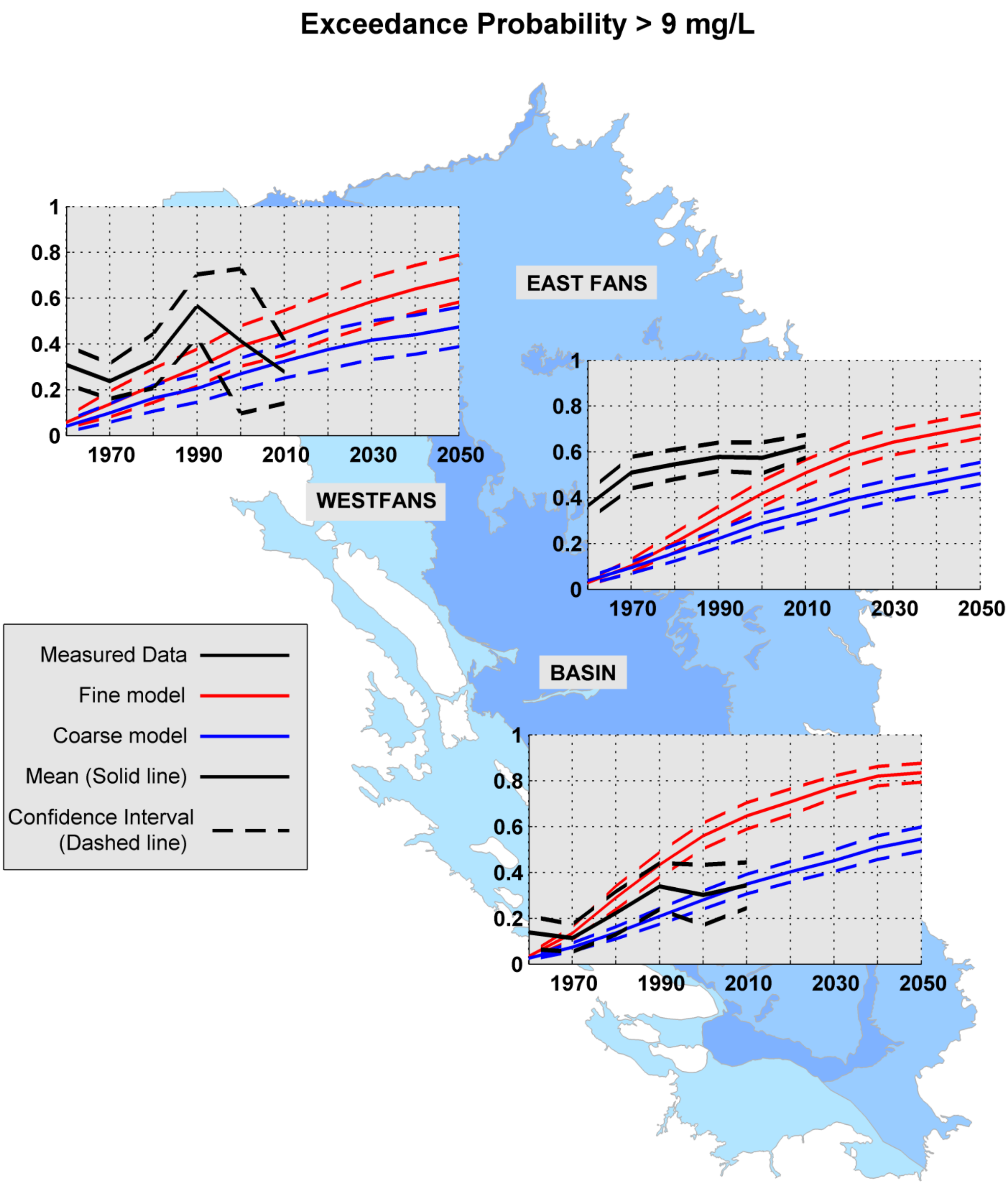

Figure 106. Comparison of decadal trends of exceedance probability of 9 mg/L, between measured data and model predictions.

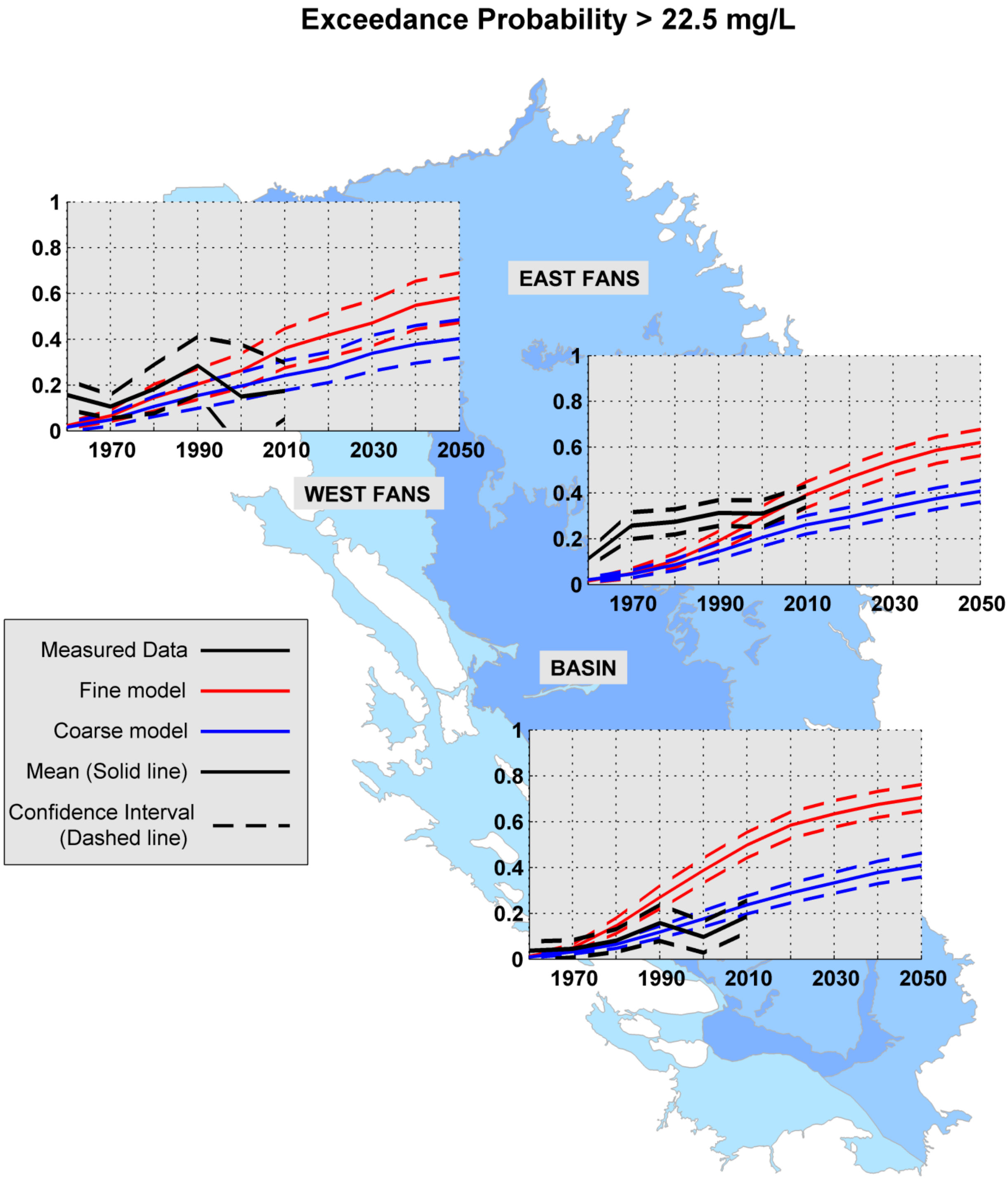

Figure 107. Comparison of decadal trends of exceedance probability of 22.5 mg/L, between measured data and model predictions.

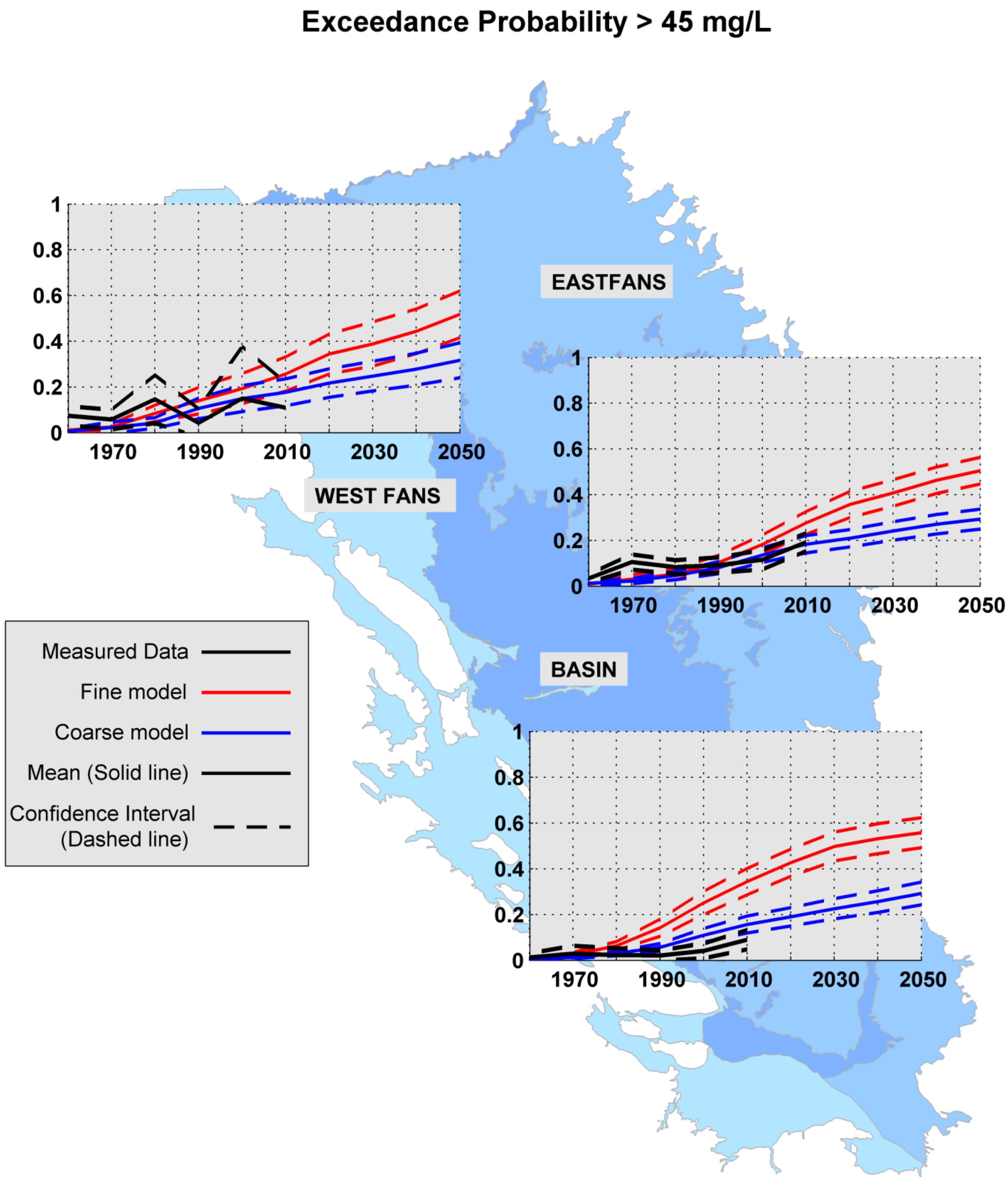

Figure 108. Comparison of decadal trends of exceedance probability of 45 mg/L, between measured data and model predictions.

Another way to study the decadal trends is to examine the exceedance probabilities for a given concentration. To this end we computed the exceedance probabilities for concentration levels 9, 22.5 and 45 mg/L. Note that the drinking water limit is 45 mg/L. Figure 106 illustrates the mean decadal exceedance probabilities of 9 mg/L and the confidence intervals based on the measured data and the responses of the two models per region. Interestingly, for the Basin region the fine and the coarse models follow the upper and lower limit of confidence intervals of the real data for the 70s and 80s, but they fail to reproduce the abrupt change that occurs between 80s and 90s. For the Eastside Fans region both models underestimate their predictions during the first decades, while the response of the fine model are somewhat closer the measured data. Both models behave in a similar manner for the Westside Fans region predicting a continuous increase, yet based on the data there is a significant decreasing trend from the 80s onwards, which is not captured by either model, but maybe due to a sampling bias in the measured data. The response of the models for the exceedance probabilities of 22.5 mg/L (Figure 107) is very similar to the exceedance probabilities of 9 mg/L. Both models fail to capture the decreasing trends for the Westside Fans and Basins regions, while the predictions of the fine model are closer to real data for the Eastside Fans region.

Interestingly, the models are showing better fit for the exceedance probabilities of the MCL (45 mg/L) (Figure 108). In antithesis with the previous figures, the trends in all three regions show a general increasing trend. For the Westside Fans and Eastside Fans regions, both models respond similarly, although the coarse model seems to fit slightly better with the real data. For the Basin region, both models predict a significant upward trend after the 70s; however, based on the real data, the upward trend only occurs two decades later.

In conclusion, the measured nonpoint source pollution system behaves in certain cases very different than what would be expected based on the model. In every case, both models failed to reproduce any downward trend. The fine model predictions seem to fit to the measured data better for the last decade, especially for the Eastside Fans region. Comparison between the models confirms, in certain cases, our expectation that the coarse model tends to underestimate, while the fine model tends to overestimate the actual concentration levels, although there are certain cases where this general “rule” is violated. Accounting for denitrification, with independently measured denitrification rates would provide a tool to improve the average model predictions when compared to average measured data.

7.5.3 NPSAT Modeling Predictions

The N loading functions for each scenario were convoluted with the URFs, a process that involves only analytical calculations; hence, we were able to calculate each alternative scenario A, B (by study area), C (by study area), and D (for detailed description of these scenarios, see Section 1.8 of Technical Report 2; Viers et al., 2012). The BTC for each well are shown in Figure 109. The thin colored lines represent the well BTCs, the red lines show the drinking water limit, and the black thick lines depict the exceedance probabilities. It can be seen that there is a significant number of wells with very fast BTCs (i.e., shallow wells with relatively small screen lengths), and also a significant number of wells that exhibit very slow responses (e.g., deep wells).

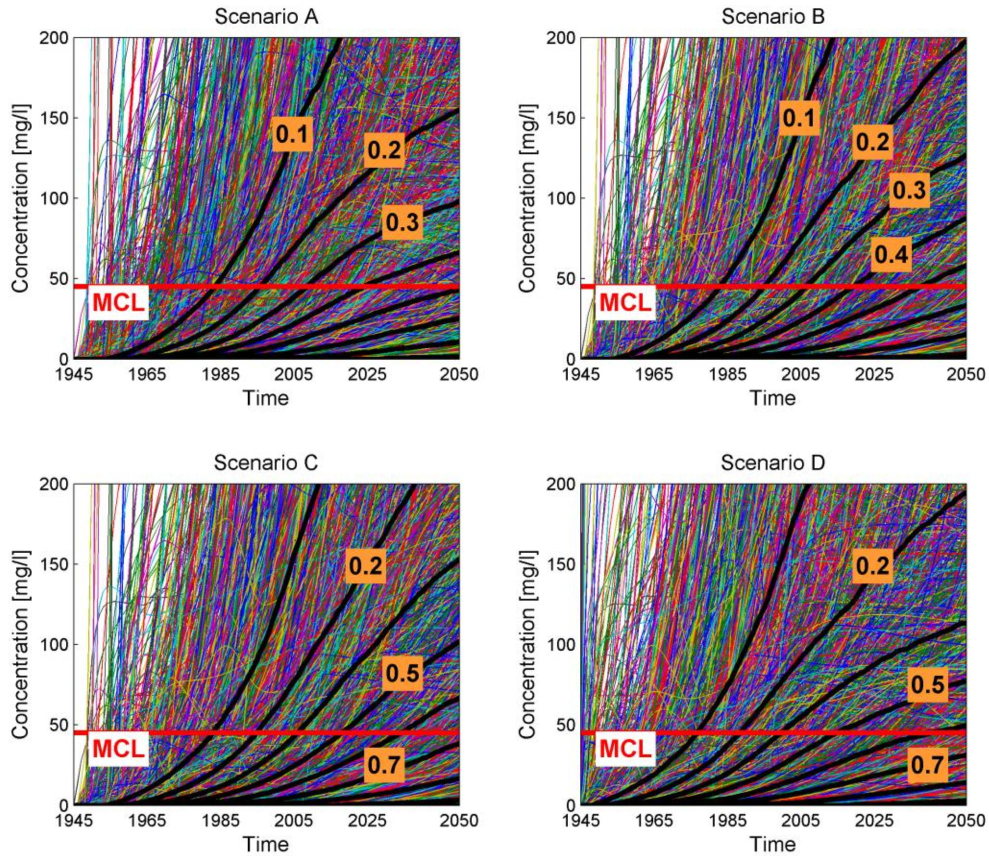

Figure 109. Breakthrough curves based on model simulation for each well for the four alternative (by study area) scenarios (thin colored lines). The black thick lines correspond to exceedance probabilities and the red lines show the drinking water limit (45 m/L). Scenario details are provided in Section 1.8 of Technical Report 2 (Viers et al., 2012).

Although the simulated BTCs are temporally uniformly distributed, their spatial distribution is non-uniform. Therefore we utilized a similar method as in the previous section (4.7.5.2) to process the results, where the TLB area was split into six basins (Kern, Kings, Tule, Tulare Lake, Kaweah, and Westside). Each basin was divided into equal area cells, and, according to the previous method, we computed the means and confidence intervals for each basin. Figure 110 shows the comparison between the fine model and the coarse model using the loading functions of scenario D. It can be seen that in all basins the fine model (red lines) predicts higher concentrations compared to coarse model (blue lines). Note also that the uncertainty is rather high in most basins (e.g., Kaweah, Tule), and in particular for the future predictions. In addition, the two models exhibit large discrepancies, yet a few safe conclusions can be drawn. For example, both models predict that for the Kings and Kern basins the concentrations will remain below the drinking water limit. On the other hand, both models predict high concentrations for the Kaweah basin. By the year 2050 the mean concentration, based on the fine model, is expected to be twice the MCL, and the coarse model estimates concentrations close to MCL with but significant variance in its estimation). This may be attributed to the fact that Kaweah basin contains the majority of dairies. As far as the remaining three basins are concerned, the models exhibit large discrepancies. For example, the coarse model predicts that the concentration in Tule basin will not exceed 10 mg/L by 2050 (the upper confidence limit), while the fine model predicts that the concentration will not exceed, but will be very close to the MCL, yet, this figure is associated with large variance. Note the upper confidence limit is approximately twice the MCL. For the basins on the west side, the coarse model predicts a similar upward trend, where the mean concentration by the year 2050 is expected to be 20 mg/L, approximately. On the other hand, the fine model predicts significantly higher concentrations.

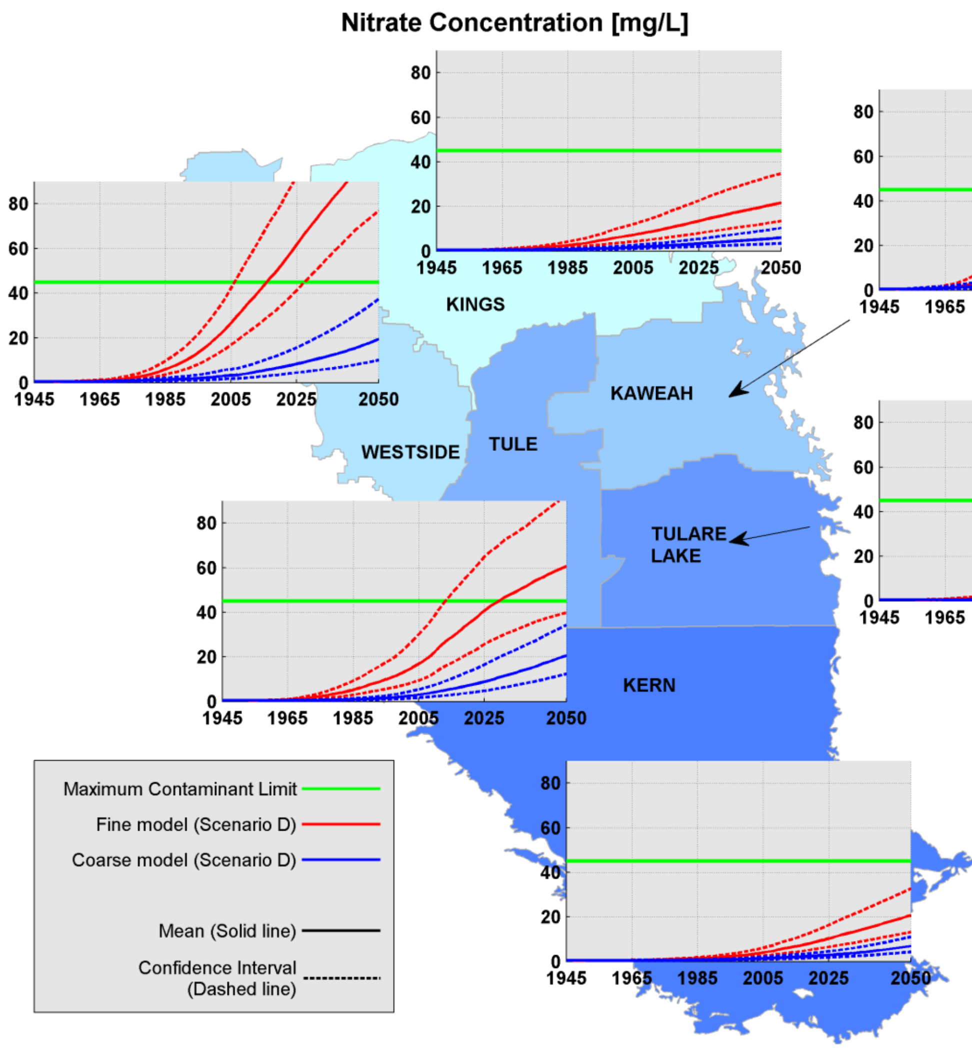

Figure 110. Comparison between coarse model and fine model based on scenario D.

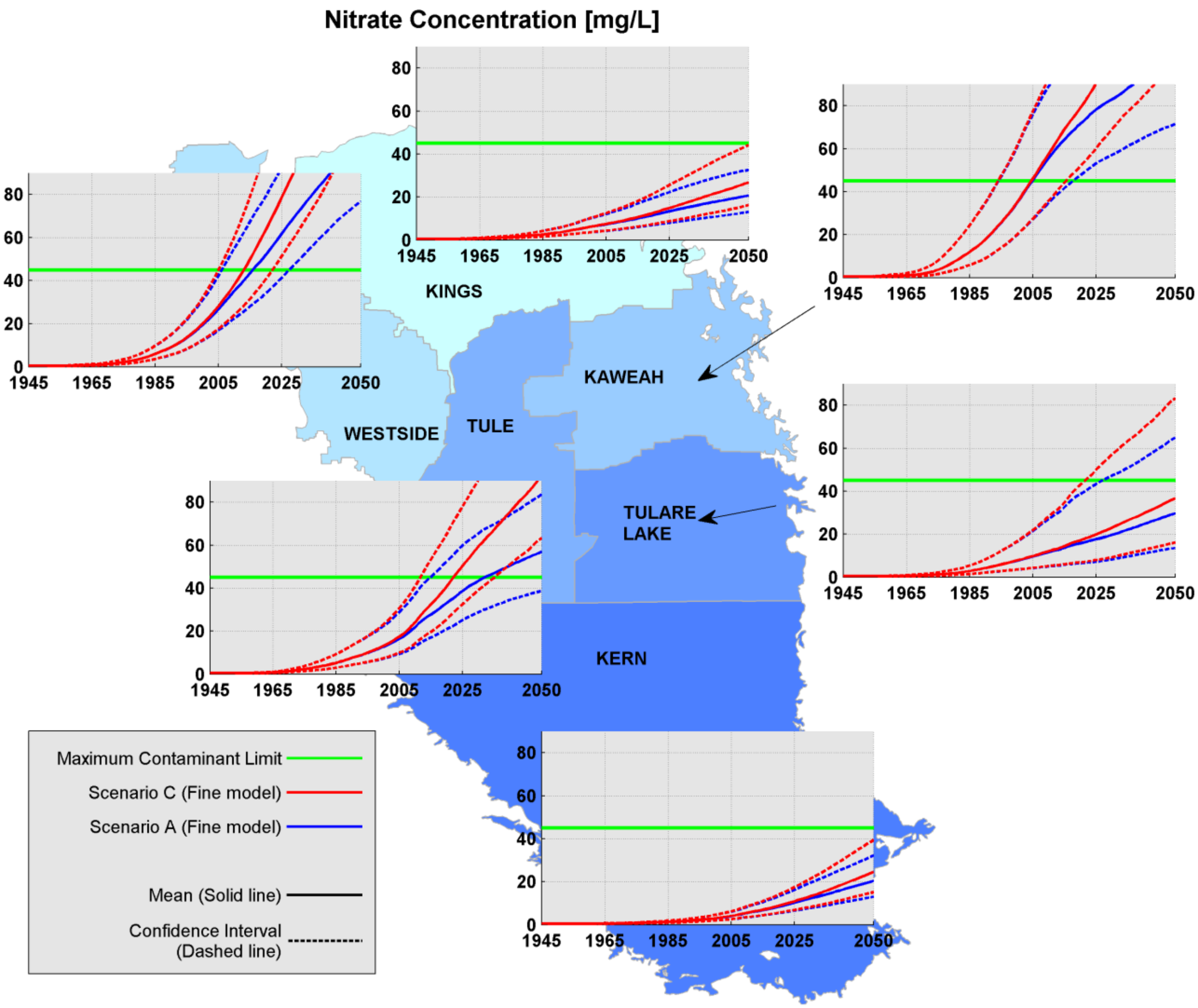

Figure 111. Comparison between scenario A and C based on the fine model predictions.

While the models may not accurately capture the actual responses they can still be useful tools to evaluate the impact of alternative scenarios. Figure 111 plots the predictions of the fine model based on the loading scenarios A and C. Scenario C is considered here the worst scenario. It can be seen that the impact of scenario C for the basins of Kern, Tulare Lake, and Kings is rather negligible. In antithesis, the basins of Kaweah, Tule, and West side are significantly affected.

Last, we compute the exceedance probabilities for each basin, based the fine and the coarse models for three given limits (e.g., 22.5 mg/L (half the MCL), 45 mg/L (MCL), and 90 mg/L (twice the MCL)). Figure 112 shows the exceedance probabilities based on the fine model using scenario D, while Figure 113 shows the exceedance probabilities based on the coarse model and the same scenario D. By comparing the two figures we observe similar discrepancy between the two models. The fine model tends to predict higher exceedance probabilities compared to the exceedance probabilities of the coarse model.

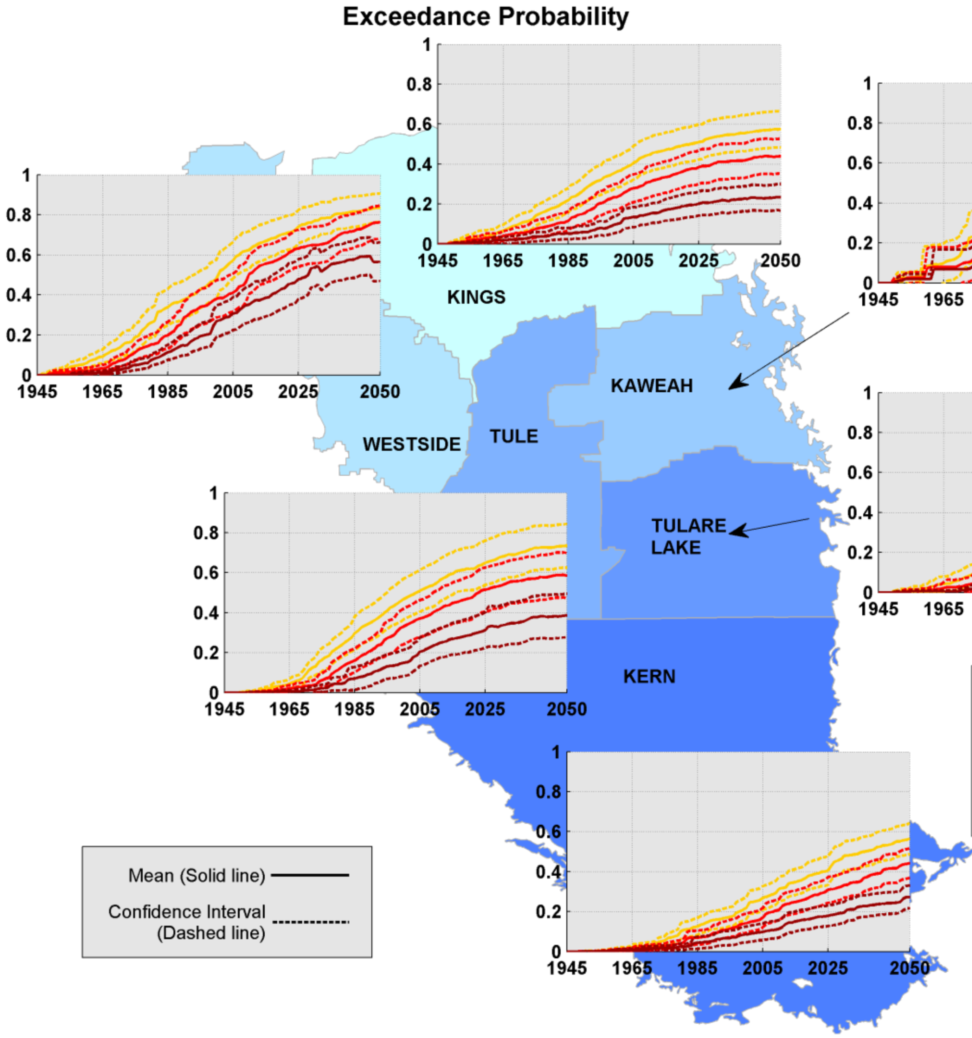

Figure 112. Exceedance probabilities for 22.5, 45, and 90 mg/L based on the fine model.

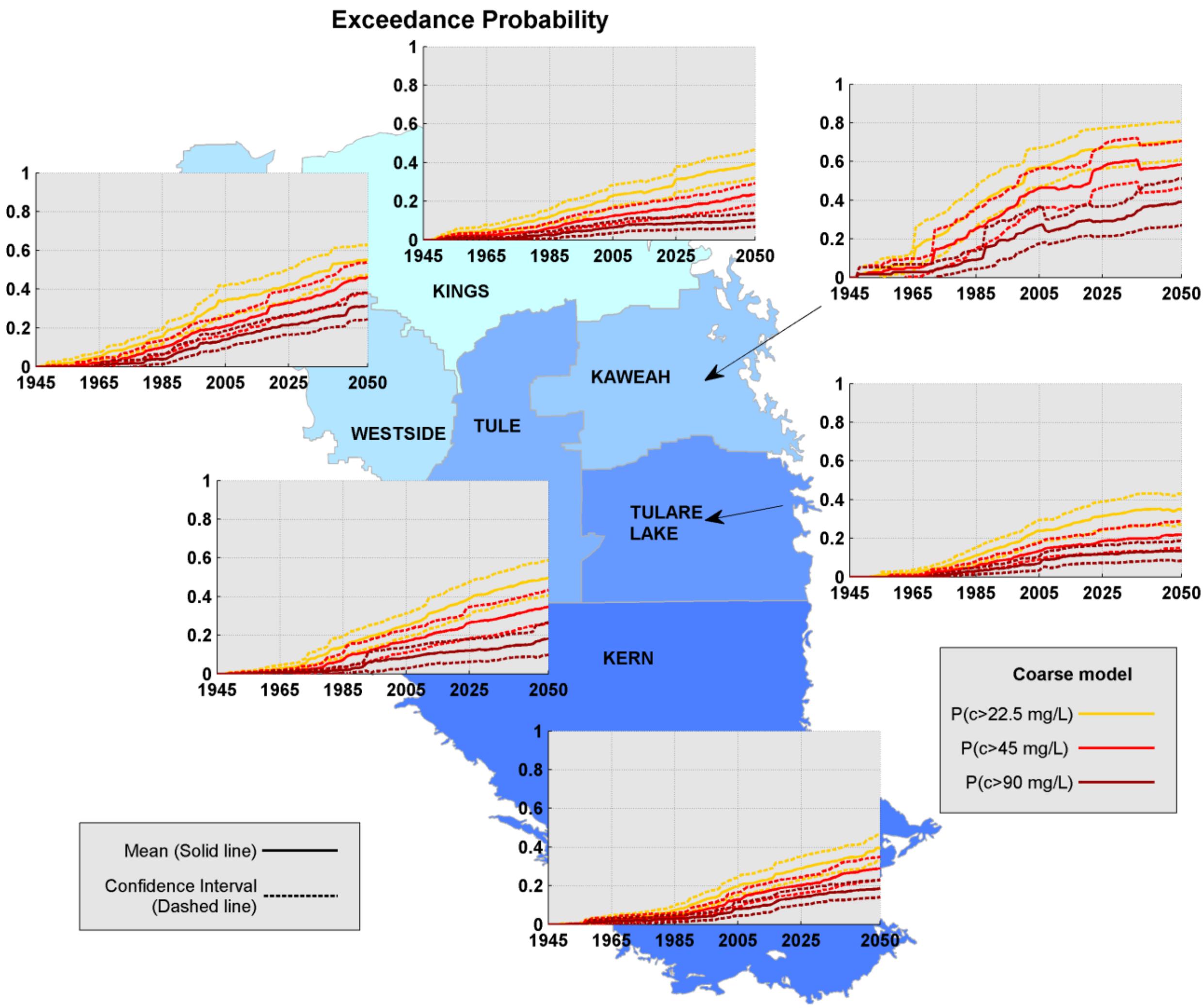

Figure 113. Exceedance probabilities for 22.5, 45, and 90 mg/L based on the coarse model.

For example, by the year 2050, the exceedance probability of the drinking water limit for the basin of Kern, based on the fine model, is approximately 30%, while based on the coarse model the same probability is ~42%. Similarly for the other regions, the two models exhibit a discrepancy on the order of 10% to 20%.

In conclusion, it is apparent that there is a strong disagreement between the model predictions, yet the two models are consistent as far as the impact of the alternative scenarios is concerned. In addition, they are consistent regarding the trends for each basin. The basin of Kaweah exhibits the highest concentration levels among the six basins (e.g., 80% and 60% of wells exceed the drinking water limit based on fine model and coarse model, respectively). For the Kern and Kings basins the two models converge that the concentration and exceedance probabilities will remain low, while a notable increase is expected for the west basins.

The difference between the two models indicates that two similar models (e.g., both are based on the CVHM model) may exhibit very different results. As shown in the previous paragraph, the streamline length distributions were quite different between the two models, with the coarse model having longer streamlines with older ages. Therefore, it is not surprising that the coarse model predicts lower concentrations. Although the fine model captures more precisely the local variability of the head field, one of the main drawbacks of the fine model is that it fails to maintain the general water balance due to the simplistic decomposition method. For example 85% of the groundwater recharge is diffuse recharge and the remaining corresponds to stream recharge. Yet the number of streamlines that originate from the streams, based the fine model, is approximately 6.5%. For the coarse model it appears that 20% of the streamlines originate from the streams. Therefore, these two models can be seen as the two extremes, where the fine model overestimates nitrate contamination and the coarse model tends to underestimate.

Overall the simulation of nonpoint source pollution for large groundwater basins is a subject that requires further research and deeper understanding of the transport mechanisms. The fine model approach, with adjustments to the simplified domain decomposition method and after accounting for denitrification, is the most promising approach. The coarse model approach, while apparently better in predicting average nitrate concentrations in some instances than the fine model, when comparing to actual average regional data, has been shown to have intrinsic weakness in the computation of travel paths that the fine model is able to address better. As a result, the coarse model appears to make better predictions especially in areas, where in fact denitrification is a significant controlling factor. Further improvements to the fine model approach are needed, and independently measured denitrification rates in the TLB region, to improve on the current modeling capability.

8 Overall Conclusions: Groundwater Nitrate Occurrence

- 100,000 nitrate records from over a half century of groundwater nitrate sampling on 19,000 wells were archived into a geospatial database.
- Availability of nitrate groundwater data is spatially highly non-uniform and most data represent only the past one to two decades. Much fewer data are available for previous decades.
- Statistical and geospatial analyses consistently show that nitrate concentrations have been increasing in much of the study area over the past 60 years with the exception of the western TLB, where wells are relatively deep and where denitrification in the Corcoran Clay, separating the upper from the lower aquifer system, may be a significant factor.
- The largest increases and highest nitrate concentrations are observed in the unconfined and semi-confined aquifer systems of the central-eastern TLB and in the northeastern and central SV.
- In many areas of the central and eastern TLB and SV, from 20% to over 30% of domestic wells and wells with relatively shallow screen exceed the nitrate MCL.
- The fraction of shallower wells that exceeds the nitrate MCL is likely to continue to increase for some time to come.
- Travel times in the unsaturated zone range from less than one year to decades, depending mostly on depth to groundwater, and on recharge rates under agricultural operations.
- We developed a new groundwater modeling tool that can be used in conjunction with estimated nitrate loading maps for past, current, and future conditions (Viers et al. 2012) to simulate the development of nitrate exceedance probabilities in sub-regional aquifer systems of interest. The model is consistent with measured nitrate concentration distributions and suggests significant increases in nitrate exceedance rates over the coming years.

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Section 8 Groundwater Nitrate Occurrence: Overall Conclusions

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10 Appendix A - Review of Domestic Well Monitoring Programs for the U.S. 50 States

Prepared by: Katherine Lockhart

10.1 Introduction

Nitrate contamination of groundwater in California's Central Valley may pose a health risk to residents who rely on well water for their drinking and cooking needs. Drinking water supplied by private domestic wells is not currently regulated in California. Well owners are personally responsible for ensuring their water meets drinking water standards and for treatment if water from their well is contaminated. However, many well owners do not regularly test their well water.

There are a variety of reasons why well owners often choose not to test their well water including: cost, lack of information on how/where/what to test for, lack of knowledge about groundwater contamination risks, or choice. Requirements for private domestic well water quality analysis or water quality testing programs and education could help to provide an accurate assessment of the quality of the groundwater for those who rely on domestic wells. California can look to other states that are currently requiring domestic well water quality analysis or that provide testing and educational programs (if any), as an example.

The purpose of this review was to analyze existing requirements for water quality analysis of domestic well water, state-by-state. For this review, links from the Environmental Protection Agency (EPA) website "State Private Drinking Water Wells Web Sites" (United States Environmental Protection Agency 2012) were followed and webpages were reviewed state-by-state. In some cases, the state Health Department was contacted for more information. Educational programs or information for well owners and programs or information provided for water quality testing were also noted (when information was available). These educational and water testing programs and resources are included as examples of the type of assistance available in many states by a multitude of agencies at the local and regional level.

10.2 Summary

This online survey of domestic well water quality regulations and programs demonstrates that some states have only minimal requirements for the testing of domestic well water, and most states have no requirements at all. Among states that do require domestic well water testing, basic analysis requirements vary and include testing upon well installation, repair, maintenance, upon property transfer, or every 5 years for rental properties. This survey found only one state that is currently requiring on-going analysis of private domestic well water: New Jersey requires the owners of rental properties have the properties well water tested once every 5 years and provide the results to the tenants.

Several states had extensive and helpful information online for homeowners concerned about the water quality from their domestic well (Rhode Island as an example) and many states provided basic information such as lists of certified laboratories or recommended analytes. However, many states may benefit from offering better online resources for homeowners on domestic well water quality.

In terms of sampling assistance or financial support for water testing, this survey found several examples. Financial assistance programs include waiving or reducing the testing fee for low income households, providing free analysis for nitrate if the household includes an infant under six months of age, or providing free testing to well owners in certain high-risk areas. For sampling assistance, in many cases, websites mentioned that local or county health departments would collect a water sample for the well owner for a small fee. Several local and county health department websites offered a detailed explanation of sample collection procedures if the well owner called the department and many departments provide test bottle kits for free or for a small fee. New Mexico's Water Fair and Water-Quality Outreach Program is exemplary in providing free testing while simultaneously delivering rural community education.

Information obtained for each state is summarized below (Table 35) alphabetical order by state. States that have an actual or potential requirement on domestic well water quality testing are highlighted in bold and underlined. This survey was completed in the summer of 2011 and more recent information may be available and/or information may have changed since then. The review does not include many educational programs provided by local, regional, or state agencies. These can be readily found through internet searches. Some examples are included in the table.

Table 35. Summary of examples for domestic well regulations, testing programs, and educational resources by state. States with regulatory requirements for domestic wells are in bold and underlined.

State	Summary	Department
Alabama	Resources: The Alabama Cooperative Extension System organizes the Home*A*Syst program that offers educational information for well owners (Alabama Cooperative Extension System 2005).	Alabama Cooperative Extension System
Alaska	Resources: Information is provided for well water testing and interpretation of the results (Alaska Department of Environmental Conservation Division of Environmental Health Drinking Water Program 2007).	Alaska Department of Environmental Conservation Division of Environmental Health Drinking Water Program
Arizona	Resources: The University of Arizona Well Owners Help Program's website offers information on wells, advice on well water testing, and a link to the Arizona Department of Health contact for a list of certified labs. The program is conducting a study of well water quality in Cochise County and has offered Saturday workshops for well owners (Arizona Board of Regents 2009). The University of Arizona's Arizona Wells Program offers a web service where interested persons can search for information on nearby wells. However, due to budget cuts, Arizona Wells is currently not operating (University of Arizona 2010).	Arizona Board of Regents and The University of Arizona
Arkansas	Regulatory: Mortgage companies often require safe drinking water results before closing on a home mortgage. Most mortgage companies require that testing be done in an EPA-certified laboratory. Arkansas has no EPA-certified private laboratories so the testing must be done at the Water Microbiology Laboratory in Little Rock or at one of the certified municipal laboratories that provides the service. Testing is probably for coliform organisms only (Arkansas Department of Health 2011).	Arkansas Department of Public Health
California	Regulatory: Testing of wells may be required upon installation. Resources: The Department of Water Resources webpage provides a list of laboratories, analyte[14] recommendations, test result interpretation information, and water treatment information (California Department of Water Resources 2011). The State Water Resources Control Board's Groundwater Ambient Monitoring and Assessment Program (GAMA) has sampled domestic wells in six county focus areas at no cost to the well owners. The State Water Resources Control Board has published many reports from GAMA program findings (State Water Resources Control Board 2012).	California Department of Water Resources
Colorado	Some monitoring of water use is required in specific basins, however it was not clear who performs the monitoring (Colorado Division of Water Resources 2011).	Colorado Division of Water Resources

[14] "analyte" is an individual chemical to be tested in a water sample.

<u>Connecticut</u>	<u>Regulatory</u>: Upon well installation or property transfer, well owners are required to test the well water for total coliform, nitrate, nitrite, sodium, chloride, iron, manganese, hardness, turbidity, pH, sulfate, apparent color and odor. If pesticides are suspected to have been used in the immediate area, the sample may be required to be analyzed for alachlor, atrazine, dicamba, ethylene dibromide (EDB), metolachlor, simazine and 2, 4-D (Connecticut Department of Public Health 2007). <u>Resources:</u> For additional well water testing, the University of Connecticut Cooperative Extension System offers recommendations on what to test for and provides a link to water testing laboratories University of Connecticut Cooperative Extension System 2012).	Connecticut Department of Public Health and the University of Connecticut Cooperative Extension System
Delaware	<u>Resources:</u> Test kits are available for a fee through the Division of Public Health Office of Drinking Water, with laboratory recommendations provided (State of Delaware, Delaware Department of Natural Resources and Environmental Control, Division of Water 2011).	State of Delaware
Florida	<u>Resources:</u> County health departments will provide instructions on testing and/or collect a sample for the homeowner for a small fee. County health departments provide lists of certified labs (Florida Department of Health, Division of Environmental Health, Bureau of Water Programs 2011).	Florida Department of Health
Georgia	<u>Resources:</u> University of Georgia Cooperative Extension offers Home*A*Syst resources online. Home*A*Syst offers extensive information on water quality risks and helps homeowners decide how to better protect their water quality and determine if their water could potentially be contaminated. County extension offices offer advice on how to collect a sample and where to send it. The Extension also prepares circulars on uranium and arsenic contamination for homeowners and has held workshops in the past covering general water quality, uranium, radon, and arsenic (University of Georgia Cooperative Extension Service, Housing and Environment 2003).	University of Georgia Cooperative Extension
Hawaii	<u>Resources:</u> University of Hawaii Cooperative Extension Service offers a fact-sheet on private well water quality and provides phone numbers for the local health department for various areas (University of Hawaii at Manoa, College of Tropical Agriculture and Human Resources, Cooperative Extension Service 2000).	University of Hawaii Cooperative Extension
Idaho	<u>Resources:</u> The Idaho department of Environmental Quality will provide a list of labs and regional health departments offer to help interpret lab results (Idaho Department of Environmental Quality 2011).	Idaho Department of Environmental Quality
Illinois	<u>Resources:</u> There is no state laboratory fee for the testing of private wells for coliform or nitrate if they meet the following conditions: 1) newly constructed wells, 2) wells that have been affected by flooding, 3) in support of studies related to water borne disease, and 4) wells serving infants under 6 months of age (Dalsin, G.J. 2011). In some cases the local health department will collect the sample for the homeowner. In other cases, the home-owner can request a sample kit that includes sample mailing instructions. Local health departments may charge a testing fee or nominal fee for sample collection (Illinois Department of Public Health, Environmental Health 2011).	Illinois Department of Public Health
<u>Iowa</u>	<u>Regulatory:</u> Well owner is responsible for collecting and testing a well water sample for coliform and nitrate when a well is installed or repaired (between 10-30 days after the well is put into service) (Iowa Department of Natural Resources 2012).	Iowa Department of Natural Resources

Indiana	Resources: "The Private Well Complaint Response Program receives complaints, investigates, and samples at-risk private water wells which are suspected of being contaminated by man-made contaminants." Homeowners can contact their local county health department to see if their well qualifies (Indiana Department of Environmental Management 2011).	Indiana Department of Environmental Management
Kansas	Resources: Some local health and environmental departments provide screening tests. Kansas State University has a brochure that recommends where to look up labs and where to get help with interpreting well water test results (Powell, G.M., Bradshaw, M.H., & Dallemand, B. 1999).	Kansas State University
Kentucky	Regulatory: KAR 5:037 requires domestic well owners to develop a groundwater protection plan that includes testing for fecal coliform and other contaminants of concern once a year. The plan can be the generic form found at the Energy and Environment Cabinet Department for Environmental Protection website (Kentucky Energy and Environment Cabinet, Department for Environmental Protection 2011). Resources: The KY-A-Syst for the Home program run by the University of Kentucky Cooperative Extension offers well educational information and advice on water testing (University of Kentucky College of Agriculture Cooperative Extension Service 2000).	Energy and Environment Cabinet Department for Environmental Protection and the University of Kentucky College of Agriculture Cooperative Extension Service
Louisiana	Resources: The department of health provides a link to analytical labs and can provide recommendations on what to test for (Louisiana Department of Health and Hospitals, Center for Environmental Health Services 2011).	State of Louisiana Department of Health and Hospitals
Maine	Resources: The department of environmental health has a link to labs and can provide recommendations on what to test for. Assistance with water treatment or well repairs may be available to low-income households (Maine Department of Health and Human Services, Center for Disease Control and Prevention, Division of Environmental Health 2012).	Maine Center for Disease Control and Prevention Division of Environmental Health
Maryland	Regulatory: Well testing may be required when a house is sold or refinanced. Local health departments may test well water (Miller, T.H. 2007).	University of Maryland Extension
Massachusetts	Regulatory: Wells are regulated at the local level and rules may vary. Resources: While there is no state requirement to have well water tested (although there may be from mortgage lenders or local Boards of Health), the Massachusetts Department of Environmental Protection (MassDEP) recommends that all homeowners with private wells do so, and use a state certified laboratory. The Department of Environmental Health website provides recommendations on tests and a link for certified labs (Massachusetts Department of Environmental Protection 2011).	Massachusetts Department of Environmental Protection
Michigan	Resources: Well drillers or mortgage lenders may test domestic well water and the homeowner can access the report. Local health departments may collect a sample as a part of the well inspection and permitting process (Michigan Department of Environmental Quality, Water Bureau 2008).	Michigan Department of Environmental Quality Water Bureau
Minnesota	Resources: County health agencies can perform nitrate and bacteria testing and some operate labs. There is a $30-$40 fee. The Minnesota department of health offers recommendations on what to test for and how often (Minnesota Department of Health 2011).	Minnesota Department of Health

Mississippi	<u>Resources:</u> The Mississippi State University Extension website has a checklist well owners can go through to determine whether or not their well is at a low, medium, or high risk for contamination. The extension also provides recommendations for what to test for, how often to test, and where testing samples can be analyzed (Bonner, J. 2010).	Mississippi State University Extension
Missouri	<u>Resources:</u> Missouri Department of Health provides free sample bottles and low cost analysis. St. Charles County Division of Public Health offers recommendations on what to test for and how often to test (St. Charles County Division of Public Health 2001).	St. Charles County Division of Public Health
Montana	<u>Resources:</u> Montana State University Extension Water Quality offers the Well Educated Program. "The Well Educated Program guides private well owners through the process of testing water quality, provides materials to help interpret test results, and offers insight on ways to help protect drinking water resources. The program is offered as a service for Montana well owners to provide affordable well testing services accompanied by test result interpretation, while simultaneously providing a useful water quality data source for managers" (Montana State University Extension Water Quality Program 2009 and 2010).	Montana State University Extension Water Quality Program
Nebraska	<u>Resources:</u> Nebraska Department of Health and Human Services provides information on what to test for and where to have sampled analyzed. The Public Health Environmental Laboratory can provide sampling for a fee. Water sample bottles are available from the Public Health Environmental Lab and many local health departments and county extension agents (Skipton, S.O., Woldt, W., Dvorak, B.I. & Pulte, R. 2008). The Nebraska Department of Environmental Quality helps organize "Test Your Well" events that offer free nitrate testing for private well owners (The Groundwater Foundation 2006).	Nebraska Department of Health and Human Services and Nebraska Department of Environmental Quality
Nevada	<u>Resources:</u> Nevada Division of Environmental Protection, Bureau of Safe Drinking Water provides information on laboratories (State of Nevada, Nevada Division of Environmental Protection, Bureau of Safe Drinking Water 2012).	Nevada Division of Environmental Protection, Bureau of Safe Drinking Water
<u>New Hampshire</u>	<u>Regulatory:</u> Mortgage lenders or specific towns may require testing. <u>Resources:</u> New Hampshire Department of Environmental Services website provides recommendations on when and what to test for and provides laboratory information (New Hampshire Department of Environmental Services 2008).	New Hampshire Department of Environmental Services
<u>New Jersey</u>	<u>Regulatory:</u> In 2001, the Private Well Testing Act (PWTA) passed into law, aiming at information disclosure about private drinking water wells. Under the PWTA, certain wells must be tested before a house can be sold. Buyers or sellers must have the water tested and review the results before the close of title. Landlords of certain properties must also test for certain drinking water parameters every five years and provide a written copy of the result to their tenants. Results must also be reported electronically to the New Jersey Department of Environmental Protection. Sample collection and analysis are conducted by laboratories certified by the NJDEPs Office of Quality Assurance. Individuals are responsible to pay for the tests which cost between $450-$650. Parameters include total coliform, VOCs, nitrate, lead, arsenic, mercury, 48-hour gross alpha particle radioactivity, pH, iron, and manganese. Not all tests are required in all areas of the state (Atherholt, T.B., Louis, J.B., Shevlin, J., Fell, K. & Krietzman, S. 2009).	New Jersey Department of Environmental Protection

New York	Resources: New York State Department of Environmental Health website offers recommendations on what to test for and how often. They provide a link to a list of state certified labs (New York State Department of Health, Bureau of Water Supply Protection 2006).	New York State Department of Health, Bureau of Water Supply Protection
New Mexico	Resources: The New Mexico Environment Department (NMED), Ground Water Quality Bureau (GWQB) conducts free testing of domestic wells throughout the rural areas of the state by holding "water fairs" ten times a year. At the water fairs, domestic well water is tested for free for electrical conductivity, fluoride, iron, nitrate, pH, and sulfate using portable analytical equipment. Domestic well owners are also educated about water quality issues and how they can help preserve or improve water quality in their communities. "This program has proven to be very popular with the general public and continues to provide NMED with valuable information on ground water quality in rural communities. The NMED continues to receive numerous requests for water fairs from community organizations, NMED Field Offices, other State, County and City agencies, and private citizens. The Water Fair and Water-Quality Outreach Program is an important tool for identifying possible non-point source water quality problems" (New Mexico Environment Department, Groundwater Quality Bureau 2012).	New Mexico Environment Department
North Carolina	Regulatory: Wells must be tested when constructed or repaired. The local health department is responsible for the testing and must report the results to the homeowner. Constituents include: arsenic, barium, cadmium, chromium, copper, fluoride, lead, iron, magnesium, manganese, mercury, nitrates, nitrites, selenium, silver, sodium, zinc, pH, and bacterial indicators. Resources: Local health departments can provide information if a homeowner wants to re-test the water (North Carolina Department of Environment and Natural Resources 2012).	North Carolina Department of Environment and Natural Resources
North Dakota	Resources: North Dakota Department of Health offers advice on what to test for and provides a list of labs on their brochure (North Dakota Board of Water Well Contractors 2007).	North Dakota Board of Water Well Contractors and North Dakota Department of Health
Ohio	Resources: The state is leading several nonpoint source pollution investigations that include sampling domestic wells (Ohio Department of Natural Resources, Division of Soil and Water Resources – Ground Water Mapping & Technical Services 2011).	Ohio Department of Natural Resources
Oklahoma	Regulatory: Most lending institutions require a water test before they will approve a loan for purchase or construction of a home. Resources: The Oklahoma Cooperative Extension Service offers educational information on wells and advice on water testing (Oklahoma Cooperative Extension Service 2012).	Oklahoma State University Cooperative Extension Service
Oregon	Regulatory: Homeowners are responsible for testing their water when there is a property transfer. Constituents required are nitrate, total coliform bacteria, and arsenic. Results must be reported to the Department of Human Services Drinking Water Program and to the buyer within 90 days. No financial assistance is available (Oregon Health Authority, Public Health Division 2011).	Oregon Health Authority

Pennsylvania	Regulatory: The state of Pennsylvania does not regulate private domestic wells. Resources: The Department of Environmental Protection offers educational information online concerning well water quality and testing (Pennsylvania Department of Environmental Protection 2011). The Penn State Extension Master Well Owner Network trains volunteers to provide homeowners with assistance for their private water systems (Penn State Extension 2012). The Center for Rural Pennsylvania has conducted at least one study throughout the state that sampled rural domestic wells for a variety of constituents (The Center for Rural Pennsylvania 2009).	Pennsylvania Department of Environmental Protection, Penn State Extension and the Center for Rural Pennsylvania
Rhode Island	Regulatory: State law currently requires testing of new private wells or for certification of occupancy requests (rental properties). State law will be requiring testing for real estate transfers in the near future. Constituents include coliform bacteria, nitrate, turbidity, and chloride. Testing must be done at a state certified lab. Resources: Rhode Island Department of Health provides a web-based interactive map that recommends constituents for well water analysis based on a wells location (Rhode Island Department of Health Private Well Testing Viewer 2011). Rhode Island Department of Health also provides sampling kits, instructions, and a laboratory testing order form. Sampling kits are based on initial/annual testing, 3-5 year testing, and 5-10 year testing, with different kits for each testing period (only the initial testing is required). The price of each test is listed on the form so well owners can pick and choose if they don't want to run the full suite of analytes (State of Rhode Island Department of Health 2011).	Rhode Island Department of Health
South Carolina	Regulatory: Fecal coliform test is required when a new well is installed ($20 extra charge on the well permit). Resources: Additional water quality tests for domestic wells are performed for a small fee and are sometimes waived or reduced depending on the homeowners income or if the well is part of a department groundwater contamination investigation. Constituents include: total or fecal coliform, metals and minerals, other inorganic parameters, volatile organic chemicals, herbicides, pesticides and other synthetic organic parameters. Funds come from funds appropriated for public drinking water oversight monitoring. Free sample kits are provided (South Carolina Department of Health and Environmental Control 2011).	South Carolina Department of Health and Environmental Control
South Dakota	Regulatory: The Centennial Environmental Protection Act of 1989 requires that all new domestic wells drilled in South Dakota are tested for bacteria and several selected chemicals. Resources: South Dakota Department of Environment and Natural Resources provides recommendations for additional well testing and an explanation of why constituents are important. South Dakota Department of Health provides sample bottles (South Dakota Department of Environment and Natural Resources 2011).	South Dakota Department of Environment and Natural Resources
Tennessee	Resources: The Tennessee Department of Environment and Conservation, Division of Water Resources published a Healthy Well Manual that recommends bacteriological testing of well water once every two years and provides advice on well maintenance (Tennessee Department of Environment and Conservation, Division of Water Resources 2012).	Tennessee Department of Environment and Conservation, Division of Water Resources

<u>Texas</u>	<u>Regulatory:</u> The Texas Commission on Environmental Quality requires that a well water sample be analyzed for coliform bacteria when a well pump is repaired or replaced (Texas Commission on Environmental Quality 2004) and they may sample domestic well water for a specific chemical or for coliform bacteria if they receive a request by a doctor (Texas Commission on Environmental Quality 2003). <u>Resources:</u> The Texas Water Resources institute, Texas Well Owner Network offers well sample screening and provides educational information for well owners (Texas Water Resources institute, Texas Well Owner Network 2012).	Texas Commission on Environmental Quality and the Texas Water Resources Institute.
Utah	<u>Resources:</u> The Utah State University Cooperative Extension provides information on how and what to test for (Utah State University Cooperative Extension 2012), and have a results interpretation tool online (Utah State University Cooperative Extension, Agricultural & Water Quality 2012).	Utah State University Cooperative Extension
Vermont	<u>Resources:</u> The Vermont Department of Health offers recommendations and provides test kits. There may be a charge for test kits and there is a charge for testing. Landlords are required to "provide safe drinking water" but the website did not provide details on what that entails (Vermont Department of Health, Agency of Human Services 2011).	Vermont Department of Health
Virginia	<u>Resources:</u> The Virginia Master Well Owner Network (VAMWON) consists of trained citizens who are available to educate citizens about well water quality and well construction. In addition to VAMWON, the Virginia Household Water Quality Program (VAHWQP) organizes drinking water clinics where well owners can have their water tested and get information on test results (Virginia Tech 2012).	Virginia Tech
<u>Washington</u>	<u>Regulatory:</u> "In most counties, when a home with a private well is bought or sold, the county health or planning department, or the lending institution involved, may require the seller to provide water-sampling results to show the water is safe to drink" (Washington State Department of Health, Division of Environmental Health, Office of Drinking Water 2010).	Washington State Department of Health
West Virginia	<u>Resources:</u> The local health departments may preform tests and provide guidance and recommendations, if requested (West Virginia Department of Health and Human Resources, Public Health Sanitation Division 2011).	West Virginia Department of Health and Human Resources
Wisconsin	<u>Resources:</u> The Department of Natural Resources, Bureau of Drinking Water and Groundwater provide recommendations for testing (University of Wisconsin, Stevens Point, College of Natural Resources, UW Extension, Central Wisconsin Groundwater Center 2010).	Wisconsin Department of Natural Resources, Bureau of Drinking Water and Groundwater and the Central Wisconsin Groundwater Center
Wyoming	<u>Resources:</u> Wyoming Department of Environmental Quality has recommendations for what to test for and how often (a tiered system) (Wyoming Department of Environmental Conservation Division of Environmental Health Drinking Water Program 2007).	Wyoming Department of Environmental Quality, Water Quality Division

11 Appendix B – Denitrification in Central California Soils and Aquifers

Prepared by: Megan Mayzelle and Thomas Harter

11.1 Denitrification Processes

Nitrate fate below the rootzone can be affected by four processes: soil retention, assimilatory reduction, dissimilatory reduction, and denitrification. Of these, only denitrification can act as a major N sink; the others only temporarily immobilize it (Korom 1992). Denitrification refers to the four-stage process (Bothe 2007) of converting nitrate (NO_3) to nitrogen (N_2) gas, a gas which composes 80% of the Earth's atmosphere (Beller et al. 2002). Soil and aquifer denitrification can help mitigate nitrate loading, thus reducing the environmental and health risks associated with groundwater nitrate contamination. Quantifying the denitrification capacity of a system and its spatial distribution is thus essential to understanding and predicting the ultimate impact of nitrate loading and to designing appropriate approaches to remediation (Ibid.; McMahon & Chapelle 2008).

Active denitrification is evidenced by a loss of nitrate from the system, an accumulation of N_2 gas, and quantitative evidence that the apparent loss of NO_3 is not entirely accounted for by dilution (Ibid.). Denitrification is traditionally considered a microbially-mediated process occurring under anoxic or anaerobic conditions where sufficient dissolved organic carbon (DOC) is present to serve as an electron donor (Korom 1992; Bothe 2007). Various studies, however, suggest that denitrification can occur under a variety of hydrological conditions.

The onset of anaerobic biological denitrification generally occurs when dissolved oxygen is below 0.25 mg O_2/L H_2O (McMahon & Chapelle 2008). Meanwhile, denitrification by facultative anaerobic bacteria has been shown to occur in soils at O_2 levels approaching that of air-saturated water)—8.6 mg O_2/L at 25°C up to more than 14 mg O_2/L at 0°C (Lloyd 1993; Burt et al. 1993; Water on the Web 2007). Research has demonstrated aerobic denitrification to be widespread in natural environments, and several researchers have suggested that it may even be predominant among denitrifying bacteria (Lloyd 1993). Unlike most anaerobic denitrifiers (Korom 1992), however, aerobic denitrifiers generally only partially mediate the denitrification process, thus producing harmful nitrogen oxide gases (Lloyd 1993) instead of harmless N_2. These gases include nitric oxide (NO), which, when exposed to oxygen, is rapidly converted to nitrogen dioxide, a major air pollutant. Nitrous oxide (N_20), a highly reflective greenhouse gas, can also form (Korom 1992). Of the 1,500 soil denitrifiers identified by Gamble et al. (1977), only 146 are capable of complete denitrification. While some methods, such as molecular markers and genetic fingerprinting, exist for identifying taxonomic groups of bacteria, the current understanding of active denitrification populations is still limited by identification procedures (Bothe 2007). Both groundwater and surface waters have been identified as important contributors of emissions of N_2O (Drecht et al. 2003; Bothe 2007); aerobic denitrification may be one explanation of this contribution.

The presence of heterotrophic versus autotrophic denitrifiers is another possible catalyst for the denitrification process. While many studies demonstrate that denitrification is limited by the availability of dissolved organic carbon (DOC) (Starr & Gillham 1993; DeSimone & Howes 1998), some suggest that denitrification by autotrophic denitrifiers can occur in the absence of DOC (Beller et al. 2002) by using reduced manganese, iron, and sulfides as electron donors (Moran et al. 2011; Korom 1992). There has also been evidence of abiotic denitrification (chemodenitrification) occurring at slightly basic pH under laboratory conditions (Van Hecke et al.1990) as well as in groundwater (Ibid.) and soils (White, R. 2006). Other research shows chemodenitrification producing NO_2 in isolated instances (Ibid.) and only after bacteria have mediated the initial steps of the denitrification process (Brons 1992; Korom 1992). While the tendency of current research is to assume the presence of microbial denitrifiers in anaerobic conditions with DOC as a limiting factor, aerobic denitrification and chemodenitrification may also affect soil and aquifer denitrification (Burt et al. 1993; Bothe 2007).

11.2 Denitrification in Aquifers

The challenges of assessing the occurrence of aquifer denitrification include accurate quantification of hydrogeochemical conditions and spatial tracking (Tesoriero et al. 2007; Rupert 2008). While the segregate and sequential nature of redox processes allow aquifer water quality data to be used to determine denitrification capacity at a given point in the aquifer (McMahon & Chapelle 2008), attributing changes in aquifer NO_3 concentration to denitrification is always accompanied by uncertainties with respect to hydrologic mixing, dispersal, and flow paths (Beller et al. 2002; Rupert 2008 Green et al. 2010), especially in areas with high-capacity pumping wells (McMahon & Chapelle 2008; McMahon et al. 2008). Factors such as recharge rates (Rupert 2008), the depth of the water table (Wright, Belitz, & Johnson 2004; McMahon & Chapelle 2008; Landon et al. 2010), spatial variability in surface-level fertilizer and manure application (Puckett & Cowdery 2002; Puckett et al. 2002; Sobota, Harrison, & Dahlgren 2009), location of the test point within the aquifer (Smith & Duff 1988), and measurement error (Beller et al. 2002) may also contribute to NO_3 concentration variability. Fluctuation in the relative abundance of stable isotopes can provide insight into to the hydrological processes—precipitation, evaporation, mixing, etc.—contributing to the recharge of the aquifer being tested (Moran et al. 2011; Wright et al. 2004), as well as any sources—rocks, manures, plants, etc--of background nitrate levels (Moran et al. 2011). The utilization of various robust indicators is ultimately key to corroborating the predicted denitrification capacity of any given system. While research has pointed toward highly heterogeneous forms of denitrification (described above), anaerobic heterotrophic microbial denitrification is currently the most well-documented (White, R. 2006; Bothe 2007; Rupert 2008) and widely accepted as the dominant form of the process. Thus, when attempting to characterize and predict the denitrification capacity of a system, indicators of environmental conditions permitting this particular denitrification pathway should be considered the most reliable (Bradley, Fernandez, & Chapelle 1992).

Given that oxygen depletion by aerobic microbial respiration occurs over time (Tesoriero et al. 2007), the age of water in a system can often serve as evidence of denitrification potential (Beller et al. 2002). However, this correlation has been shown to be invalid in some studies of the California San Joaquin

Valley and Salinas Valley aquifer systems (Tesoriero et al. 2007; Moran et al. 2011). The presence of almost exclusively anthropogenic compounds, such as tritium, a radioactive by-product of nuclear reactions (US NRC 2011), is used to age-date water, since most of these chemicals have been produced since 1950 (GAMA 2011). In a study of principle North American aquifers, McMahon & Chapelle (2008) further showed that NO_3 concentrations are significantly larger (indicating less denitrification activity) in water samples containing > 0.5 mg O_2/L and concomitant (Beller et al. 2002) Mn^{2+} and Fe^{3+} concentrations of < 0.05 mg/L and < 0.1 mg/L , respectively. Concordantly, Rupert (2008) observed through a national decadal study that wells with anoxic conditions had significantly smaller increases in nitrate concentrations as a consequence of anthropogenic NO_3 loading than mixed or oxidized wells. McMahon & Chapelle (2008) have thus identified the aforementioned concentrations as threshold values for denitrification to occur. Nevertheless, when using oxygen concentration as an indicator of denitrification capacity, the presence of oxygen-rich microenvironments as well as subsequent blending with oxygen-rich waters must be taken into consideration before a system can be characterized as anaerobic (Ibid.).

An aquifer's porous matrix can provide further important clues as to its denitrification potential. A blue, greenish, or gray color indicates anoxic conditions; these sediments generally represent subaqueous deposition and tend to be relatively higher in organic matter and calcium carbonate than oxidized sediments (Croft 1972). Additionally, since most denitrifiers apparently reside in the aquifer's porous sediment matrix, denitrification rates in sediment core samples are generally higher than rates in water samples (Korom 1992; Tesoriero et al. 2007). In fact, denitrification rates in one aquifer varied from 1.2 to 74.4 nmol/g H_2O/day, while denitrification in the saturated sediments surrounding the same aquifer ranged from 3.8 to 233.6 mmol/m^2/day (Bradley et al. 1992). Various samples may thus be necessary when attempting to characterize biological denitrification potential in order to accurately represent the aquifer's total denitrification capacity and biological diversity. A microbial presence can also be determined by amending an aquifer with an organic carbon source, such as sucrose (Hiscock et al. 1991) and observing sucrose degradation and/or CO_2 production. The addition of an organic carbon source can also help identify a situation in which DOC concentrations are a limiting factor of denitrification capacity (Bradley et al. 1992).

Stable isotope ratios of N_2 and O_2 (Böttcher et al. 1990) and N^2/H^2O concentrations (Beller et al. 2002) are also frequently used as indicators of denitrification. These methods are especially advantageous when investigating age-dated aquifers, since N_2 quantity can then be used to determine the rate of denitrification (Ibid.). Nevertheless, measurements have been constrained by the difficulty in determining the atmospheric N_2 component as opposed to that produced by denitrification (Ibid.). These methods of denitrification quantification have been recently improved by new technologies, such as membrane inlet mass spectrometry, which allows precise determination of the dissolved concentrations of N_2 and O_2 (Ibid.). An additional benefit of this field test is that it quantifies dissolved argon concentrations (Ibid.), which further corroborate indications of denitrification activity (McMahon & Chapelle 2008) by determining the amount of N_2 produced by denitrification as opposed to normally-occurring concentrations (Beller et al. 2002). McMahon & Chapelle (2008) establish a framework for assessing redox processes based on such easy-to-measure and relatively inexpensive parameters (Ibid.).

In this framework, favorable conditions for denitrification are identified as: $O_2 < 0.5$ mg/L, $NO_3 \geq 0.5$ mg/L, $Mn^{2+} < 0.05$mg/L, and $Fe^{2+} < 0.1$mg/L. In addition to these parameters, McMahon & Chapelle (2008) recommend that other redox indicators (including MH_4, H_2S, CH_4, and H_2) be measured as additional verification. While the application of this framework will not always result in a single-redox diagnosis, McMahon & Chapelle (2008) assert that even a mixed redox diagnosis will be useful in the context of assessing water quality issues.

Aquifer denitrification is likely to remain a topic of interest both within and beyond the scope of groundwater nitrate contamination. Potential exists for amending anaerobic environments with ethanol and acetate (Lorrain et al. 2004) or even sawdust (Israel et al. 2009) to increase denitrification. The possibility of adding nitrate to aquifers to serve as an electron acceptor in the degradation of hazardous organic compounds has also been explored (Alvarez & Vogel 1995; Pelz et al. 2001), although studies have shown that such organic compounds can coexist with high nitrate concentrations in aquifers without significant nitrate denitrification (Moran et al. 2004) . Site restriction for such systems is also very stringent to ensure that drinking water sources remain uncontaminated (Isosaari et al. 2010). Additionally, the role of denitrification of contaminants in the production of anthropogenic N_2O will remain highly relevant as governments face higher accountablity for their greenhouse gas production (Gon 2005; UNFCCC 2007).

11.3 Denitrification in Soils

Soil denitrification depends on the combined effects of hydrological conditions, soil properties, temperature, fertilizer type and application rate, and crop or plant characteristics (Burt et al. 1993; Drecht et al. 2003). Rates of soil denitrification are highly temporally and spatially variable, and can reach 4.5 – 9.0 kg NO_3/ha/day for short periods (White 2006). As in aquifers, denitrification in soils is best known to occur when denitrifying bacteria are present, DOC is available, water content is high, and oxygen levels are concomitantly low (air < 15% porosity at field capacity) (Burt et al. 1993).

NO_3 is highly mobile in soils, and is hence prone to leaching out of the root zone and into the subsoil (>50 cm) during periods of water and NO_3 application in excess of plant demand (Fageria & Baligar 2005). Leaching is the major source of NO_3 in subsoils (Bothe 2007) and in aquifers (Buczko & Kuchenbuch 2010). Additionally, leaching tendency is greater on arable land than grassland (Barraclough et al. 1983) and is further influenced by crop type (McLay et al. 2001), fertilizer type and application rate (Sobota et al. 2009), soil texture and structure (White 2006), and management techniques (Burt et al. 1993) iincluding irrigation (Burt et al. 1993; White 2006). Although most soil denitrification occurs in the top 10-20cm of soil (White 2006), where organic matter concentrations are high, subsoil denitrification has reportedly denitrified up to two-thirds of agricultural nitrate input to a shallow glacial outwash system (Puckett & Cowdery 2002; Puckett et al. 2002). Nevertheless, prevailing conditions can vary considerably from the root zone to the subsoil, meaning that rootzone denitrification capacity is not a determinant of subsoil denitrification capacity (White 2006).

Humid climates (Drecht et al. 2003) and large annual precipitation (White 2006) result in a high leaching rate and short residence time of nitrate in the soil (Drecht et al. 2003); high leaching and short residence times can be logically correlated to heavy irrigation activities as well (Burt et al. 1993), especially when temporally near fertilizer applications (White 2006). In contrast, dry climates with restricted precipitation have lower leaching rates and a higher residence time of nitrate in the soil (Drecht et al. 2003). Soil properties also affect denitrification by influencing soil water capacity, oxygen status, and leaching tendencies. Fine-texture and low porosity (either due to massive structure or a large percentage of fine micropores) both tend to hold water more tightly than sandy soils (White 2006), consequently reaching anaerobic conditions more easily and maintaining them for longer periods (Drecht et al. 2003). Poorly drained soils also tend more toward being anaerobic (Burt et al. 1993); although denitrification can begin when soil air falls below 15% of soil pore space (White 2006), rates increase exponentially as water-filled pore space moves upward of 90% (Bothe 2007). Inundated fields, such as for wetland rice, are an extreme example of this condition (Drecht et al. 2003). On the other hand, sandy and high porosity soils tend toward leaching and low soil water saturation (high soil air fraction) (White 2006), which can carry residual soil nitrate into deeper soil layers toward groundwater aquifers. Regardless of the soil type, denitrification outside of periods of irrigation (Burt et al. 1993) or rainfall does not generally occur in shallow agricultural soil systems due to aeration by tilling (Bothe 2007). This combination of high fertilizer input and low denitrification rates results in significant levels of nitrate present in agricultural soils compared to other ecosystems (Burt et al. 1993). Short bursts of microbial growth during irrigation are being studied as a potentially important source of N_2O gas (Smart et al. in press).

While agricultural soils themselves do not normally experience extended periods of denitrification, fertilizers applied only to the surface of the soil have significant volatilization capacity (Burt et al. 1993). Animal manures are particularly vulnerable since they provide NO_3 as well as abundant DOC to denitrifiers (Ibid.); manures also have a particular tendency toward partial denitrification (White 2006), resulting in N_2O production (Korom 1992). Animal feed composition and the length and method of storage alter the composition of manure, which subsequently affects soil denitrification capacity and concomitant N_2O emissions following application (Bothe 2007). A strong knowledge of the plant-available-N content of fertilizers and the site-specific mineralization tendencies is additionally necessary to decrease N losses by adjusting application rates to best reflect crop uptake patterns (Ibid.). The Intergovernmental Panel on Climate Change (IPCC) has developed a model designed for large-scale utilization that combines fertilizer and manure application rates, soil type, and number of animals to determine N_2O emission factors (Gon 2005); this model estimates the default emission factor for N_2O fertilizers and manures at 1.25% of N applied to soil, with an additional 2.5% of all nitrate leached to groundwater eventually being released as N_2O gas as well (Bothe 2007). Such models may be useful in determining state-, region-, or even farm-wide N_2O emissions over time; they also indicate, at the regional level, very limited losses of NO_3 to denitrification being assigned in these models relative to the total amount of NO_3 fluxes to groundwater.

Temperature can serve as a general indicator of rate of microbial denitrification, since cool temperatures generally reduce microbial activity (Korom 1992). Processes are very slow below 10°C (White, R. 2006) but can be stimulated by abundant carbon amendments (Burt et al. 1993). The optimum temperature for denitrification is approximately 40°C. Temperature, along with pH, can further affect the ratio of N_2O to N_2 produced by denitrification processes; in cool, acidic conditions, relatively more N_2O is produced, whereas if pH >6 and temperatures 25°C or greater, N_2 is favored (White 2006). This effect of pH and temperature changes on N_2O : N_2 ratios is immediate, and thus a direct effect on N_2O production, as opposed to being as a result of a change in microbial population composition (Burt et al. 1993).

The rate of denitrification over a short period of time can be measured in the field by injecting acetylene, which blocks the reduction of N_2O to N_2, into the soil and collecting N_2O emissions from a limited area (Drecht et al. 2003). This method has a number of disadvantages, including acetylene leaching from the soil, and is logistically inapplicable to subsoil layers since disturbance may aerate the soil (Bothe 2007). N-labeling is an alternative that allows long-term measurements without soil disturbances, but that requires specialized laboratory analyses and expensive materials (Ibid.). Field-scale mechanistic models can also be used in combination with crop production models to simulate denitrification and the events affecting it, such as crop N uptake, changes in soil organic matter (SOM) composition, and leaching (Drecht et al. 2003).

Interesting possibilities exist for exploiting natural soil denitrification processes to mitigate nitrate contamination including overland flow systems, constructed wetlands, and pond systems. Overland flow systems utilize vegetation, microbial communities, and a limited-percolation deep soil layer to facilitate denitrification across a sloped field (Isosaari et al. 2010). While total denitrification in such systems is positively correlated with field length (up to 60 meters) and DOC availability, site-specific evaluations are necessary to determine the sustainable denitrification capacity of a given system (Ibid.). Constructed wetlands, especially those including wetland plants and DOC amendments, offer an indefinite and socially appealing option for catalyzing denitrification. These systems remove a median 18-37% of nitrogen from wastewater; this percentage can be augmented by recycling effluent back into the system (Ibid.). Pond systems, while working on a similar principle, have an inconsistent denitrification rate of 20-80% and require a large land area (Ibid.).

11.4 Study Area Characterization

The Salinas Valley Basin (SVB) extends 120 miles southeast from the Monterey Bay down through the center of Monterey County on the central coast of California (Kulongoski & Belitz 2007). Monterey surpassed Kern County in 2009 to become the 3rd most agriculturally productive county in the nation; agricultural product value for Monterey County now totals more than $4 billion (USDA NASS 2010). Groundwater meets 95% of agricultural, municipal, and industrial water needs in the SVB (Saavedra 2007) and has been historically overdrawn as a result (MCWRA 2007). Most public supply wells and some irrigation wells source from deep groundwater (Moran et al. 2011; Fogg et al. 1999); recharge occurs principally from stream percolation and irrigation return water (California Department of Water Resources (2003).

The SVB has mean annual precipitation of 38 centimeters and a mean annual temperature of 14°C with generally little seasonal variation (Kulongoski & Belitz 2007). The valley is filled with about 800 meters of marine and terrestrial sediments, including as much as 610 meters of saturated alluvium. Water-bearing units are unconsolidated to semi-consolidated, interbedded gravel, sand, and silt lying above a consolidated granitic basement; these include the Monterey Formation, Santa Margarita Formation, the Paso Robles Formation, and recent alluvium at the surface (Durbin et al. 1978; Montgomery Watson Americas, Inc. 1997). The Salinas Valley groundwater basin includes the main aquifers at their respective depths: the 180-foot, the 400-foot, and the deep 900-foot aquifers (Saavedra 2007). While the northeast side and seaside portions of these aquifers are confined and semi-confined (Fogg et al. 1999) by clay aquitards (Moran et al. 2011), the 400 and 180 aquifers become unconfined in the midsection of the Valley (Forebay subbasin) and in the southern portion of the SVB (Upper Valley subbasin), respectively (Fogg et al. 1999; California Department of Water Resources 2003). While anoxic groundwater can be found in the northern portions of the valley (Moran et al. 2011), groundwater samples in the Forebay and Upper Valley subbasins area have an average O_2 concentrations of 3.9 mg/L and an average temperature of 19.3°C (Kulongoski & Belitz 2011). Their combined total aquifer storage capacity is approximately 8,820,000 acre feet (California Department of Water Resources 2003). Groundwater overdraft has reduced water levels by approximately 25 meters, resulting in 3-5 mile wide seawater intrusions in the 180 and 400 aquifers (Moran et al. 2011; California Department of Water Resources 2003; MCWRA 2007).

With the exception of the seaside area, groundwater NO_3 concentrations in the 180 and 400 aquifers have been steadily on the rise since the 1950s, primarily as a result of nonpoint agricultural contamination (Fogg et al. 1999; California Department of Water Resources 2003). Nitrate concentrations in the SV show high spatial and temporal variability, probably due to movement and mixing caused by the draw from well pumps, as well as the varying depths of the tested wells (Moran et al. 2011). The California DWR reports that 9.7% of public supply wells in the SV exceeded the NO_3 maximum contaminant load (MCL) from 1994-2000 (Ibid.). In a 1995 study of irrigation and monitoring wells, 23 of 35 wells (66%) tested in the Upper Valley region and 30 of 81 wells (37%) tested in the Forebay region exceeded the MCL for NO_3; the average NO_3 concentrations were 98 and 45 mg NO_3/L respectively (California Department of Water Resources 2003). In contrast, a 2005 study of the Forebay and Upper Valley Aquifers detected NO_3 in 6 of 19 wells (32%) tested, with an average well NO_3 concentration of only 3.15 mg/L as N (Kulongoski & Belitz 2007) . Most recently, Moran et al. (2011) have reported one deep drinking water well in the San Jerardo area as having NO_3 concentrations ranging from 69-130 mg/L throughout 2010 and sporadic concentrations ranging from 100-681 mg/L throughout the SV.

The Tulare Lake Basin (TLB), comprised of portions of Fresno, Kern, Kings, and Tulare counties, sits in the south of the San Joaquin Valley (SJV) of California. Five of the eight counties in the SJV rank among the top ten agriculturally most productive in the nation, with the market value of agricultural products sold from the SJV totaling $18.3 billion (USDA NASS 2007). Groundwater serves as the primary drinking water source for nearly 90% of SJV residents (Community Water Center's Health and Drinking Water Series 2011). While depth to groundwater in the Central Valley is generally shallow (California Department of Water Resources 2003), water levels in the TLB are comparatively much lower (Faunt 2009). TLB groundwater levels dropped nearly 17 feet from 1970 through 2000, with 7 feet of loss occurring between 1999 and 2000 alone (California Department of Water Resources 2003).

Groundwater in the Central Valley is generally well-oxygenated (Beller et al. 2002), and principally recharged by surface irrigation and stream recharge (Groundwater Management Technical Committee 1999; California's Groundwater Bulletin 2006).

The TLB receives an average of 18 cm annual precipitation and has an average annual temperature of 20°C, with large seasonal variations from 0°C to 39°C (National Weather Service). The TLB is filled with up to 9,700 meters of marine and continental sediments (California Department of Water Resources 2003). The younger alluvium sits largely near and above the water table (ibid.) underlain by the generally poorly sorted deposits of the older alluvium and continental deposits (Croft and Gordon 1968; Croft 1972). These are moderately to highly permeable water-bearing units (Croft 1972; California Department of Water Resources 2003) and form the principle aquifers for the region (Croft & Gordon 1968). The Corcoran Clay, the most prominent of various very low permeable clay layers spreading over the region divides the subsurface into an upper unconfined to semi-confined aquifer and a lower confined aquifer (Croft & Gordon 1968; California Department of Water Resources 2003). Anoxic conditions generally prevail in the Corcoran Clay and other, less prominent clay layers, often designated as the A-clay, B-clay, C-clay, D-clay, and F-clay (the Corcoran Clay is also referred to as the E-clay). Anoxic conditions also prevail in the hundreds to several thousand feet thick lacustrine and paludial clay deposits underneath the lakebed of (former) Tulare Lake west of the town of Corcoran, which yield very little water of poor quality (Croft & Gordon, 1968).

11.5 Denitrification Potential in Tulare Lake Basin and Salinas Valley

A comparison of NO_3 concentrations between various North American principle aquifers (PA) revealed that the Central Valley sand and gravel PAs, along with the western volcanic PAs, have the largest NO_3 concentrations on the continent (McMahon & Chapelle 2008). This finding is consistent with the high N loading from fertilizer and manure recorded in those areas. Electron transfers from Mn^{4+} and Fe^{3+} reduction is relatively small in these aquifers and redox conditions are the least heterogeneously distributed of all North American PAs (Ibid.). Movement of NO_3 into the aquifers would be expected to divert electrons from these compounds since NO_3 sits higher in the succession of terminal electron-accepting processes.

Subsurface conditions are often variable and change on regional and local scales (Green et al. 2008; Green et al. 2010; Landon et al. 2011). In general, a lack of correlation between anaerobic conditions and groundwater age has been observed in the study area (Moran et al. 2011; Tesoriero et al. 2007). On the whole, it is thus likely that denitrification does not affect nitrate concentrations significantly in the production aquifer system of the Central Valley (Ibid.; Tesoriero et al. 2007).

While not the dominant process affecting nitrate concentrations in the Central Valley as a whole, denitrification may be affecting concentrations in certain areas of the valley, in particular in shallow groundwater near the valley trough. Previous investigations in the San Joaquin Valley have characterized the groundwater as being mostly oxic. Anoxic conditions that may lead to denitrification have been found to be more prevalent in shallower parts of the aquifer system where groundwater has historically discharged into wetlands and streams near the central trough of the valley (Davis et al. 1959;

Croft & Gordon 1968; Croft 1972; Bertoldi et al. 1991; Dubrovsky et al. 1993; Burow et al. 1998; Singleton et al., 2007; Landon et al. 2011). There, a shallow groundwater table with historically upward groundwater flow and the presence of large amounts of organic matter provide shallow subsurface conditions with extremely limited oxygen availability but large amounts of dissolved organic carbon as electron donor for microbially mediated redox reactions that create anoxic conditions in the shallow-most aquifer sediments.

Anoxic conditions generally prevail below the oxic zone, in deeper sections of the older alluvium and continental deposits, typically below depths of 500 to more than 1,000 ft throughout the Tulare Lake Basin (Croft 1972). Due to the large age of groundwater in the anoxic zone, it is currently uncertain, to which degree such anoxic conditions may slow or prevent future nitrate pollution (Landon et al. 2011).

Soil denitrification in the TLB and SV plays a potentially significant role. Riparian zone, shallow aquifer, and stream/groundwater denitrification have been shown to remove up to 93 mg NO_3/L from the Merced River, a gaining river (one whose flow increases due to groundwater discharging to the surface) just north of the TLB in the SJV (Domagalski et al. 2008). Similar results have been found for the Parajo River, a losing river (one whose flow decreases due to surface water recharging groundwater) just north of the mouth of the SV at Monterey Bay. Ruehl et al. (2007) report that NO_3 concentrations in the Pajaro River decrease downstream across 11km while other ions concentrations remain unchanged, with denitrification rates peaking where seepage loss and surface-subsurface exchange are the greatest and stream discharge is the lowest (Ibid.). Moran et al. (2011) have further found NO_3 concentrations in monitoring wells near the Salinas River to be significantly lower than those located directly in the Salinas River (about 10 mg NO_3/L). Isotopic signatures of the water samples indicated that denitrification was taking place, and that this process may occur in other areas where similar conditions exist near the river. Denitrification was not, however, found to occur in areas away from the Salinas River where oxic groundwater (high levels of dissolved oxygen) was found.

11.6 Conclusions

The naturally occurring process of denitrification in soils and aquifers is increasingly being recognized for its relevance to current anthropogenic NO_3 and N_2O contamination. Scientific understanding of the various denitrification pathways has been restricted by effective assessment techniques, especially with respect to aquifer and subsoil denitrification potential. Oxygen concentrations, DOC concentrations, and microbial presence have been acknowledged as key conditions of system denitrification potential; other factors, including pH, temperature, fertilizer types and rate, vegetation type, and hydrologic processes such as irrigation and precipitation, have also been identified as important influences. Nonpoint agricultural emissions, including inorganic fertilizers and manures, are the principle sources of NO_3 contamination of groundwater in the SV and TLB of California. Although some initiatives have begun to analyze this issue, the problem of water insecurity in the SV and TLB deserves further attention. While the potential for aquifer denitrification is limited where most groundwater is pumped in these areas, managed surface-level denitrification systems merit additional exploration as a plausible form of NO_3 contamination mitigation.