Approaches for Groundwater Management in Times of Depletion and Regulatory Change

By

Robert Michael Gailey

Dissertation

Submitted in partial satisfaction of the requirements for the degree of

DOCTOR OF PHILOSOPHY

In

Civil and Environmental Engineering

In the

OFFICE OF GRADUATE STUDIES

of the

UNIVERSITY OF CALIFORNIA

DAVIS

Approved:

____

Jay R. Lund, Chair

____

Thomas Harter

____

Yueyue Fan

Committee in Charge

2018

Robert Michael Gailey
June 2018
Civil and Environmental Engineering

ABSTRACT

New legislation and related regulations in California mandate groundwater management to avoid undesirable results from excessive groundwater pumping and other human activities. Required aspects of management planning that will occur in well over 100 alluvial groundwater sub-basins across the state include scientific foundation and public transparency. While the details of individual groundwater management approaches will be particular to conditions in each management area, there will be common elements based on similar aspects of many groundwater systems in the state.

The process of beginning to implement groundwater management for a large part of the state poses challenges. Use of scientific knowledge and methods of analysis developed in academia and industry will be helpful to the overall effort. This dissertation considers aspects of groundwater management that are expected to be common among many planning efforts.

- Inactive water supply wells acting as conduits for contaminant migration is considered through field examples and use of recently available databases regarding existing well constructions and water quality. Nonpoint source pollution effects on groundwater quality from conduit wells are investigated and selective regulatory action is explored as an option for addressing an issue that is too large for complete solution with available resources.

- Economic impacts to domestic well operations from heightened agricultural pumping during drought are evaluated by drawing on information from a recently available database regarding well constructions. Analysis is performed regarding the economic tradeoff between stakeholders for different policies that limit the groundwater drawdown and a compensation scheme is identified that allows both parties to benefit.

- Options for managed aquifer recharge are explored for a combined urban-agricultural area through use of a recently released land use database. A hydro-economic analysis is performed using linear programming to evaluate the efficacy of an on-farm recharge approach.

Future work may include development of additional insights within the context of the study areas presented as well as extensions that consider new geographic areas and additional related considerations.

ACKNOWLEDGEMENTS

I turned 51 years old during my initial quarter of doctoral work in fall 2014. This is remarkable – not in a self-congratulatory way but because of what it says about those who contributed to my success. A professor supported (actually suggested) my application to the doctoral program. His department did not resist the plan. And my wife agreed that I would commute to Davis on a regular basis while she continued to work hard at a demanding job and we raised teenaged children. Remarkable. I cannot thank those who supported me enough; however, I should note some of their contributions.

Jay Lund roped me into this venture and acted as my primary advisor. I thank him for his interest in me as well as sharing time, knowledge, ideas, editing suggestions, enthusiasm and funding. Upon entering Jay's research group, I noticed the community of capable, positive people with which Jay surrounds himself and quickly came to learn that it is a result of the honest and positive way he works. Josué Medellín-Azuara has also been very generous with his time, ideas and funding. I thank him for seeing value in my ideas and capabilities as well as his willingness to collaborate and good humor while working very hard. Graham Fogg deserves special mention for not kicking me out of (or running from) his office when, early on in our acquaintance, I revealed to him that I had been a distant groupie for decades since reading his work for the Texas Bureau of Economic Geology in the mid-1980's. Graham has provided detailed and insightful comments on my papers, research collaboration and funding. Tim Ginn acted as Chair for my qualifying exam, assisted with administrative processes, gave counsel and, perhaps most importantly, took me to lunch where he quelled my concerns about the open-ended nature of the pending oral exam. He also served as a frequent source of intellectual excitement during my first year in the program. Completing all of my coursework over a short period was tiring and I often stopped by Tim's lab to recharge.

Thomas Harter has been enthusiastic about my ideas and potential contributions from the first day I arrived on campus. He has also provided valuable comments on manuscripts as I prepared to send them out for journal review. I also thank him for serving on both my qualifying exam and dissertation committees. Yueyue Fan too served on both my qualifying exam and dissertation committees. She also rigorously introduced me to different ways of approaching mathematical programming formulations. However, her biggest gift to me was an expression of confidence in my technical abilities during one of our discussions. Sam Sandoval Solis welcomed me during my first few weeks on campus and provided thoughts about my research ideas over coffee. I thank him for also serving on my qualifying committee. And then there are the many fellow graduate students and members of both the Center for Watershed Sciences and Hydrologic Sciences Graduate Group who I thank for including me in their conversations and excitement about many things related to water.

Finally, I thank my family. My wife Sara provided me with selfless support beyond what I could have hoped for or imagined. To her, I promise that this is my last degree. I thank my son Alex for good naturedly joining in a "race" to see who would obtain their degree first (my Ph.D. or his high school diploma) even though he knew, no matter how hard he worked, his would take exactly four school years. And I thank my daughter Elle for often finding humor at times when I stop conversing and blankly stare into space. In those moments she says, as she has since quite little, "Oh, Daddy is doing math in his head again!"

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1 - INTRODUCTION

Background and Motivations

It is an exciting time to work on groundwater issues in California. After more than two decades of small steps towards groundwater management and spurred by the recent drought that lasted from 2012 through 2016, California's legislature passed the Sustainable Groundwater Management Act (SGMA) of 2014 establishing substantive requirements for groundwater management in the state[1]. SGMA defines undesirable results from excessive groundwater pumping[2], requires management to avoid "significant and unreasonable" effects, and specifies a schedule for implementation and achievement of sustainable management. This legislation and the regulations that follow from it require long-term (50-year) groundwater resource use planning that is both science-based and publicly transparent (CADWR 2018). Detailed analysis and planning for well over 100 alluvial groundwater sub-basins across the state has been set in motion with final management plans due between 2020 and 2022 depending on prioritization that is tied to conditions of resource depletion.

---

[1] None of the previous legislation actually required groundwater management to occur in a basin that was experiencing chronic depletion. Assembly Bill 3030, passed in 1992, made provisions for voluntary groundwater management by existing local agencies. Senate Bill 1938, passed in 2002, required public agencies seeking certain state funding for groundwater related construction projects to have groundwater management plans in place. Assembly Bill 359, passed in 2011, required that groundwater management plans include identification of recharge areas. As evidenced by continuing groundwater depletion, these pieces of legislation were not sufficient to effect groundwater management in most places where it was required. SGMA was passed in 2014 because there was a historically rare combination of 1) a group of legislators and water thought leaders particularly dedicated to establishing the type of management needed in the state for many decades and 2) public concern over an ongoing drought.

[2] There are six undesirable results under SGMA: 1) chronic lowering of groundwater levels, 2) reduction in groundwater storage, 3) seawater intrusion, 4) degradation of water quality including migration of contaminant plumes, 5) land subsidence and 6) depletion of surface water.

While the details of individual groundwater management approaches will be particular to conditions in each management area, successful plans will address hydrologic, economic, engineering and financial facets of management as well as legal, regulatory, social and political considerations. Most approaches will likely include a combination of supply enhancement and demand reduction, address balancing the economic benefits from and costs of use, and consider issues of social equity. Moreover, there will be many common elements based on similar aspects of most groundwater systems in the state. These similarities include:

- Access to groundwater often buffers water supply against surface water shortage. It also creates water supplies and land use activities where they might not otherwise occur. This access has existed since the early-1900's when well construction and pumping technologies made it possible to extract large quantities of groundwater from the subsurface.
- The benefits from conjunctively using groundwater with surface water occur annually during California's dry season (generally occurring during May through October). However, the benefits from increased groundwater pumping are greatest during severe drought when statewide use of the resource can rise from approximately 30 to 60 percent of total water demand and local agricultural use can account for up to 80 percent of groundwater consumption (CADWR 2014).
- Water supply wells are ubiquitous with agricultural and domestic uses being the most numerous. These structures were developed over many decades with varying technologies and under regulations and inspection/enforcement programs that were generally lax until recently. Pumping from wells, distributed over tens of thousands of locations within the state, creates one of the largest hydrologic outputs from groundwater systems.

- Wells exist as both assets to be protected and a liabilities to be managed. Their value is directly related to the groundwater access provided. Challenges posed by wells include 1) the impact of extracting large volumes of water on groundwater systems and other wells and 2) the potential to spread nonpoint source and other pollution when idle during the wet season.
- The areas requiring management can be quite large and include many thousand parcels. Much of the land is privately-owned and a wide range of hydrologic stresses result from the varying land uses.
- Various forms of managed aquifer recharge are likely to play a role in balancing withdrawals from groundwater. The availability of water for recharge operations will not be distributed smoothly over time but will be concentrated around infrequent precipitation events, surface water reservoir releases and seasonal runoff from snowmelt. The timing of these events is always uncertain and the anticipated effects of climate change increases uncertainty. Because seepage into the subsurface is slow relative to surface water flow, large areas of land will be required to capture water that becomes available for recharge.
- There are many stakeholders with different requirements of groundwater resources and a balancing of needs will be required.

This last point warrants elaboration. The term "groundwater resource" is only a general description for the target of management. In fact, there will be many different management considerations based on the variety of groundwater uses. Quantity will be the focus in some cases, while the concern will be quality in others. But these subdivisions require even further clarification for effective management planning. A significant quantity of water may be present in a groundwater system such that some pumping needs continue to be met, yet undesirable results can still occur. The groundwater levels may not be high enough to support baseflow to streams or they may be so deep that surface water depletion occurs, domestic wells run dry and land subsidence develops. Also, the water quality in a particular area may be acceptable for agricultural irrigation or industrial use but not for drinking water. SGMA provides an initial framework for striking a balance between land uses, water rights, environmental flows, the state-declared human right to water[3] and other considerations.

The process of beginning to implement groundwater management for much of the state over a fairly short period poses challenges that include: obtaining adequate-quality information necessary to develop approaches, securing the funds required to implement and track management programs, and allocating limited technical, management and regulatory human resources to achieve goals efficiently. It will be necessary to perform "management triage" along the way by streamlining and decentralizing some of the decision making. Use of scientific knowledge and methods of analysis developed in academia and industry will be helpful to the overall effort. While there will always be additional questions, as well as needs for new approaches and more data, there is more than enough information available to move forward keeping in mind that adaptive management will occur. This dissertation considers some examples of managing to different aspects of groundwater management.

There are four technical chapters and concluding thoughts in this dissertation. Chapter 2 describes the phenomenon of inactive water supply wells acting as conduits for contaminant migration and provides several field examples with data from across the state. This issue is not widely enough understood in groundwater practice and its consideration is quite important for

[3] https://www.waterboards.ca.gov/water_issues/programs/hr2w/

aspects of groundwater quality management. Chapter 3 builds off of Chapter 2 making use of recently available databases to investigate nonpoint source pollution effects on groundwater quality from conduit wells in the southern Central Valley. The technical analysis is used as a basis to explore performing selective regulatory action for an issue that is too large to be completely addressed with available resources. Chapter 4 also draws on information from a recently available database and evaluates the economic impacts to domestic well operations caused by heightened agricultural pumping during drought in the southern Central Valley. Looking towards regulatory application, analysis is performed regarding the economic tradeoff between stakeholders (agricultural pumpers and domestic well users) for different policies that limit the drawdown from groundwater pumping and a compensation scheme is identified so that both parties benefit. Chapter 5 considers implementing managed aquifer recharge for two sub-basins in the Central Valley just south of Sacramento. A hydro-economic analysis is performed to investigate the efficacy of conducting an on-farm recharge program. Chapter 6 then provides some thoughts for future work.

Publications and Copyrights

Each of the technical chapters of this dissertation exist as stand-alone documents prepared for publication. Chapters 2 and 3 have recently been published in Hydrogeology Journal (Gailey 2017 and 2018). Because the materials were published as Open Access, copyrights for the articles are retained by the author. Chapters 4 and 5 have been submitted for publication to the same journal and are intended to be published as Open Access. I have confirmed with the journal that, whether the articles are published as Open Access or not, there are no impediments to including the raw text, tables and figures used to prepare the journal articles in this dissertation.

References

CADWR (2014) California Water Plan Update 2013. http://www.water.ca.gov/waterplan/cwpu2013/final/index.cfm Accessed 31 January 2018

CADWR (2018) Sustainable groundwater management website. California Department of Water Resources. http://www.water.ca.gov/groundwater/sgm/index.cfm Accessed 31 January 2018

Gailey RM (2017) Inactive supply wells as conduits for flow and contaminant migration: conditions of occurrence and suggestions for management. Hydrogeol J 25(7):2163-2183, doi: 10.1007/s10040-017-1588-y

Gailey RM (2018) Using geographic distribution of well-screen depths and hydrogeologic conditions to identify areas of concern for contaminant migration through inactive supply wells. Hydrogeol J (initially on-line only), doi:10.1007/s10040-018-1734-1

CHAPTER 2 – INACTIVE SUPPLY WELLS AS CONDUITS FOR FLOW AND CONTAMINANT MIGRATION: CONDITIONS OF OCCURRENCE AND SUGGESTIONS FOR MANAGEMENT

Abstract

Water supply wells can act as conduits for vertical flow and contaminant migration between water-bearing strata under common hydrogeologic and well construction conditions. While recognized by some for decades, there is little published data on the magnitude of flows and extent of resulting water quality impacts. Consequently, the issue may not be acknowledged widely enough and the need for better management persists. This is especially true for unconsolidated alluvial groundwater basins that are hydrologically stressed by agricultural activities. Theoretical and practical considerations indicate that significant water volumes can migrate vertically through wells. The flow is often downward, with shallow groundwater, usually poorer in quality, migrating through conduit wells to degrade deeper water quality. Field data from locations in California, USA, are presented in combination with modeling results to illustrate both the prevalence of conditions conducive to intraborehole flow and the resulting impacts to water quality. Suggestions for management of planned wells include better enforcement of current regulations and more detailed consideration of hydrogeologic conditions during design and installation. A potentially greater management challenge is presented by the large number of existing wells. Monitoring for evidence of conduit flow and solute transport in areas of high well density is recommended to identify wells that pose greater risks to water quality. Conduit wells that are discovered may be addressed through approaches that include structural modification and changes in operations.

Introduction

It has long been known that vertical flow and contaminant migration can occur through inactive water supply wells when they interconnect water-bearing zones with different hydraulic heads and water qualities (Johnson, 1961; California Department of Water Resources [CADWR], 1962; Davis et al., 1964). Observations of flow include 1) the sound of water cascading down well screens located above standing water levels (Davis et al., 1964) and 2) downhole videos showing particles entrained in water moving swiftly within well casings and across well screens (observations and consulting reports by field hydrogeologists including the author) as well as measurements of vertical flows through casings (McCombs and Fiedler, 1927; Davis et al., 1964). Evidence of contaminant migration includes wells that must be pumped to waste after periods of inactivity in order to reestablish acceptable water quality (Mayo, 2010). Such flow and transport can also occur in abandoned supply wells (Gass et al., 1977) as well as improperly destroyed wells (CADWR, 1981). While some research and publication on intraborehole flow and transport has occurred, relatively little of the available field data has been documented beyond private reports to well owners and only a few detailed case studies support research findings. Consequently, the prevalence of groundwater supply wells acting as vertical conduits and the extent to which water quality impacts stem from this phenomenon may not be widely understood.

Given the significant reliance on groundwater both globally (Wada et al., 2012) and in specific areas such as California (CADWR, 2014), effective resource management should include an understanding of the threat to water quality posed by wells that act as conduits. If the threat is appreciated at a sufficiently high regulatory level, approaches to minimize groundwater flow and the spread of contaminants through inactive, abandoned and improperly destroyed wells may be further developed. This work 1) quantifies the effects of intraborehole flow and transport through inactive wells in unconsolidated alluvial groundwater basins and 2) suggests management approaches for reducing impacts. As discussed in the following section, these contributions will add to a developing body of knowledge regarding wells that act as conduits with results from field investigation for several wells in California, USA (Figure 1).

Figure 1: Study locations in California, USA. *Black letters* indicate approximate locations of field sites cited in the literature review. *Black numbers* indicate approximate locations of case examples.

Previous Research

Table 1 summarizes research on water supply wells acting as vertical conduits. Both flow and contaminant transport are addressed in most cases. Many contributions are theoretical modeling studies. Some include modeling and field data while others evaluate field data without modeling. The MODFLOW Multi-Node Well package (Halford and Hanson, 2002; Konikow et al., 2009) was used for many of the studies that model intraborehole flow (Hanson et al., 2004; Konikow and Hornberger, 2006; Zinn and Konikow, 2007; Clark et al., 2008; Johnson et al., 2011; Yager and Heywood, 2014) although some researchers represented flow through the inside of the well casing with an equivalent porous medium (Lacombe et al., 1995; Mejia e al., 2012). Some works consider a small spatial scale (the well and immediate vicinity) while others focus on larger areas (wellfields and regions). The locations of most field examples are in the USA. The hydrogeology consists of sedimentary basins in all but one case and unconsolidated sediments in all but four cases. Of 27 citations, there are 16 unique field sites.

Table 1 Research contributions regarding water supply wells acting as vertical conduits for flow and contaminant migration

Reference	Scope: Theoretical	Scope: Field Data	Scale: Well	Scale: Regional	Processes: Flow	Processes: Transport	Setting: Location	Setting: Geology
Bexfield et al., 2012	-	x	x	x	x	x	Albuquerque, New Mexico	Unconsol. alluvial
Bexfield and Jurgens, 2014	-	x	x	x	x	x	Modesto, California, USA	Unconsol. alluvial
	-	x	x	x	x	x	Albuquerque, New Mexico	Unconsol. alluvial
Clark et al., 2008	-	x	x	x	x	x	York, Nebraska, USA	Unconsol. alluvial
Corcho Alvarado et al., 2009	-	x	x	-	x	x	Paris Basin, France	Consol. sedimentary
Davis et al., 1964	-	x	x	-	x	-	San Joaquin Valley, California, USA	Unconsol. alluvial
Dragon et al., 2009	-	x	x	x	x	x	Wielkopolska, Poland	Glacio-fluvial and fluvial
Hanson et al., 2004	-	x	-	x	x	-	Santa Clara Valley, California, USA	Unconsol. alluvial
Hart et al., 2006	-	x	-	x	x	-	Southeastern Wisconsin, USA	Consol. sedimentary
Jimenez-Martinez et al., 2011	-	x	-	x	x	x	Campo de Cartagena, Spain	Consol. sedimentary
Johnson et al., 2011	x	-	x	x	x	-	-	Multi-layered porous medium
Jurgens et al., 2008	-	x	x	x	x	x	Modesto, California, USA	Unconsol. alluvial
Jurgens et al., 2014	-	x	x	x	x	x	Albuquerque, New Mexico	Unconsol. alluvial
Konikow and Hornberger, 2006	x	-	x	x	x	x	-	Multi-layered porous medium
Lacombe et al., 1995	x	-	x	x	x	x	-	Multi-layered porous medium
Landon et al., 2008	-	x	x	x	x	x	York, Nebraska, USA	Unconsol. alluvial
Landon et al., 2009	-	x	x	x	x	x	Modesto, California, USA	Unconsol. alluvial
	-	x	x	x	x	x	York, Nebraska, USA	Unconsol. alluvial
Lu et al., 2008	-	x	-	x	x	x	Chianan Plain, Taiwan	Unconsol. near-shore
Mayo, 2010	-	x	x	-	x	x	San Luis Valley, Colorado, USA	Unconsol. Lacustrine and fluvial
McCombs and Fiedler, 1927	-	x	x	-	x	-	Oahu, Hawaii, USA	Vesicular basalt
	-	x	x	-	x	-	Pecos Valley, New Mexico, USA	Consol. sedimentary
Mejia et al., 2012	x	-	x	x	x	x	-	Multi-layered porous medium
Santi et al., 2006	-	x	-	x	x	x	Merced, California, USA	Unconsol. alluvial

Reference	Scope		Scale		Processes		Setting	
	Theoretical	Field Data	Well	Regional	Flow	Transport	Location	Geology
Silliman and Higgins, 1990	x	-	x	-	x	-	-	Multi-layered porous medium
Sokol, 1963	x	-	x	-	x	-	-	Multi-layered porous medium
Viers et al., 2012	x	x	-	x	x	x	San Joaquin Valley, California, USA	Unconsol. alluvial
	x	x	-	x	x	x	Salinas Valley, California, USA	Unconsol. alluvial
Williamson et al., 1989	-	x	-	x	x	-	Central Valley, California, USA	Unconsol. alluvial
Yager and Heywood, 2014	-	x	x	-	x	x	Modesto, California, USA	Unconsol. alluvial
	-	x	x	-	x	x	Albuquerque, New Mexico	Unconsol. alluvial
Zinn and Konikow, 2007	x	-	x	x	x	x	-	Multi-layered porous medium

Theoretical considerations of intraborehole flow include Sokol (1963) who developed an analytical representation of steady-state flow between multiple confined aquifers. Silliman and Higgins (1990) later developed an analytical approach for representing steady-state flow between two aquifers that included head loss across the well screens and allowed the shallower water bearing zone to be unconfined. Lacombe et al. (1995) developed a numerical approach to simulate transient flow and solute transport across an aquitard via a leaky borehole under a variety of conditions. The authors concluded that rapid migration and widespread distribution of contamination is possible and the potential impacts of leaky boreholes on groundwater quality should be further evaluated. Konikow and Hornberger (2006) addressed flow and solute transport through hypothetical wells to consider potential effects on groundwater systems. Their results indicated that, under some conditions, intraborehole flow can occur even when the well is pumped. Zinn and Konikow (2007) explored the effects of intraborehole flow on groundwater age in a hypothetical regional system. They demonstrated that the location of a conduit well in a flow system can determine direction of flow (upwards or downwards) and the depth where age changes as a result of the flow. Changes in pumping stresses within the groundwater system also affected direction and magnitude of flow in conduit wells as well as the depth of impact by transport. Johnson et al. (2011) reenforced the conclusions of others noting that 1) proximity to pumping wells as well as seasonality of pumping influenced the magnitude of vertical flows and 2) bypassing transport through both shallow water-bearing zones and lower hydraulic conductivity sediments reduced protection to deeper groundwater since less solute attenuation occurred.

Early field research documented flow at the well scale. McCombs and Fiedler (1927) may provide the earliest documented measurements of intraborehole flows. The most extensive dataset on vertical flows through inactive water supply wells may be by Davis et al. (1964). They presented ambient flow (well not pumping) data, collected as a profile along the well length with an impeller flowmeter, from 11 idle agricultural supply wells located in the western San Joaquin Valley of California (see A on Figure 1 for location and Table 2 for data). The wells penetrated the deeper water-bearing zone beneath a regional aquitard (Corcoran Clay). Flow measurements ranged from 48 to 478 gallons per minute (g/min), or 187 to 1,852 liters per minute (L/min), at depths within the range of 485 to 1,350 feet below ground surface (ft bgs), or 148 to 411 meters below ground surface (m bgs). Flow profiles increased with depth in the wells as groundwater entered the wells, largely through holes in the casing, and decreased as water exited the wells through the screen perforations. As indicated by the Reynolds numbers, turbulent flow occurred in the wells. The authors noted that the observed intraborehole flow rates were consistent with intensive groundwater pumping for agricultural irrigation that occurred throughout the groundwater system for years before the data were collected. In this area, pumping from depth and return flows from irrigation at the ground surface created vertical head differences as great as 300 ft (91 m) that drove flow down through the wells.

Table 2 Flow profiling conducted by Davis et al., 1964

Well Number	Screened Interval (bgs)	Conduit Flow: Depth (ft bgs)	Depth (m bgs)	Cumulative (cfs)	Cumulative (L/min)	Incremental (L/min)	Well Diameter (in)	Well Diameter (cm)	Flow Velocity (m/s)	Reynolds Number (-)
14/12-12N1	558-1,709 ft (170-521 m)	550	168	0.00	0	0	12.75	32	0.00	0.0E+00
		600	183	0.20	340	340	12.75	32	0.07	2.0E+04
		650	198	0.23	391	51	12.75	32	0.08	2.3E+04
		700	213	0.13	221	-170	12.75	32	0.04	1.3E+04
		800	244	0.16	272	51	12.75	32	0.06	1.6E+04
		900	274	0.12	204	-68	10.75	27	0.06	1.4E+04
		1,000	305	0.19	323	119	10.75	27	0.09	2.3E+04
		1,100	335	0.16	272	-51	10.75	27	0.08	1.9E+04
		1,200	366	0.16	272	0	10.75	27	0.08	1.9E+04
		1,300	396	0.11	187	-85	10.75	27	0.05	1.3E+04
14/13-17E1	Unknown	600	183	0.00	0	0	18.00	46	0.00	0.0E+00
		700	213	0.00	0	0	18.00	46	0.00	0.0E+00
		734	224	0.12	204	204	18.00	46	0.02	8.5E+03
		750	229	0.13	221	17	18.00	46	0.02	9.2E+03
		760	232	0.12	204	-17	12.75	32	0.04	1.2E+04
		800	244	0.15	255	51	12.75	32	0.05	1.5E+04
		900	274	0.19	323	68	12.75	32	0.07	1.9E+04
		1,000	305	0.19	323	0	12.75	32	0.07	1.9E+04
		1,100	335	0.00	0	-323	12.75	32	0.00	0.0E+00
14/13-29Q1	621-1,803 ft (189-550 m)	800	244	0.00	0	0	12.75	32	0.00	0.0E+00
		900	274	0.12	204	204	12.75	32	0.04	1.2E+04
		1,000	305	0.14	238	34	10.75	27	0.07	1.7E+04
		1,050	320	0.18	306	68	10.75	27	0.09	2.1E+04

Well Number	Screened Interval (bgs)	Conduit Flow: Depth (ft bgs)	Conduit Flow: Depth (m bgs)	Conduit Flow: Cumulative (cfs)	Conduit Flow: Cumulative (L/min)	Conduit Flow: Incremental (L/min)	Well Diameter (in)	Well Diameter (cm)	Flow Velocity (m/s)	Reynolds Number (-)
15/13-8N1	639-1,798 ft (195-548 m)	650	198	0.00	0	0	12.75	32	0.00	0.0E+00
		675	206	0.49	832	832	12.75	32	0.17	4.9E+04
		707	215	0.44	747	-85	12.75	32	0.15	4.4E+04
		750	229	0.44	747	0	12.75	32	0.15	4.4E+04
		800	244	0.47	798	51	12.75	32	0.16	4.7E+04
		1,000	305	0.47	798	0	10.75	27	0.23	5.6E+04
16/15-26N3	Unknown	700	213	0.00	0	0	12.75	32	0.00	0.0E+00
		800	244	0.00	0	0	12.75	32	0.00	0.0E+00
		910	277	0.19	323	323	12.75	32	0.07	1.9E+04
		1,000	305	0.26	442	119	12.75	32	0.09	2.6E+04
		1,100	335	0.00	0	-442	12.75	32	0.00	0.0E+00
17/15-27K1	912-2,130 ft (278-649 m)	925	282	0.00	0	0	12.75	32	0.00	0.0E+00
		975	297	0.18	306	306	12.75	32	0.06	1.8E+04
		1,025	312	0.48	815	510	12.75	32	0.17	4.8E+04
		1,100	335	0.48	815	0	10.75	27	0.23	5.7E+04
		1,200	366	0.48	815	0	10.75	27	0.23	5.7E+04
		1,300	396	0.48	815	0	10.75	27	0.23	5.7E+04
17/16-4E1	416-1,580 ft (127-482 m)	475	145	0.00	0	0	10.75	27	0.00	0.0E+00
		485	148	0.54	917	917	10.75	27	0.26	6.4E+04
		500	152	0.36	612	-306	10.75	27	0.17	4.3E+04
		525	160	0.46	781	170	10.75	27	0.22	5.5E+04
		549	167	1.09	1,852	1,070	10.75	27	0.53	1.3E+05
		600	183	0.96	1,631	-221	10.75	27	0.46	1.1E+05

Well Number	Screened Interval (bgs)	Conduit Flow Depth (ft bgs)	Conduit Flow Depth (m bgs)	Conduit Flow Cumulative (cfs)	Conduit Flow Cumulative (L/min)	Conduit Flow Incremental (L/min)	Well Diameter (in)	Well Diameter (cm)	Flow Velocity (m/s)	Reynolds Number (-)
17/16-8L1	553-1,800 ft (169-549 m)	550	168	0.00	0	0	12.75	32	0.00	0.0E+00
		650	198	0.54	917	917	12.75	32	0.19	5.4E+04
		700	213	0.14	238	-679	12.75	32	0.05	1.4E+04
		800	244	0.13	221	-17	12.75	32	0.04	1.3E+04
		900	274	0.27	459	238	12.75	32	0.09	2.7E+04
		1,000	305	0.20	340	-119	12.75	32	0.07	2.0E+04
		1,100	335	0.16	272	-68	12.75	32	0.06	1.6E+04
		1,200	366	0.00	0	-272	10.75	27	0.00	0.0E+00
19/17-19D1	721-2,063 ft (220-629 m)	700	213	0.00	0	0	12.75	32	0.00	0.0E+00
		800	244	0.12	204	204	12.75	32	0.04	1.2E+04
		900	274	0.12	204	0	12.75	32	0.04	1.2E+04
		1,000	305	0.00	0	-204	12.75	32	0.00	0.0E+00
19/17-19P1	665-2,030 ft (203-619 m)	400	122	0.00	0	0	16.00	41	0.00	0.0E+00
		450	137	0.51	866	866	16.00	41	0.11	4.1E+04
		700	213	0.21	357	-510	12.75	32	0.07	2.1E+04
		900	274	0.16	272	-85	12.75	32	0.06	1.6E+04
		1,100	335	0.16	272	0	10.75	27	0.08	1.9E+04

Well Number	Screened Interval (bgs)	Conduit Flow					Well Diameter		Flow Velocity	Reynolds Number
		Depth		Cumulative		Incremental				
		(ft bgs)	(m bgs)	(cfs)	(L/min)	(L/min)	(in)	(cm)	(m/s)	(-)
19/18-33N2	632-2,025 ft (193-617 m)	575	175	0.00	0	0	16.00	41	0.00	0.0E+00
		600	183	0.00	0	0	16.00	41	0.00	0.0E+00
		625	191	0.42	713	713	16.00	41	0.09	3.4E+04
		650	198	0.47	798	85	12.75	32	0.16	4.7E+04
		750	229	0.40	679	-119	12.75	32	0.14	4.0E+04
		850	259	0.38	646	-34	12.75	32	0.13	3.8E+04
		950	290	0.38	646	0	12.75	32	0.13	3.8E+04
		1,050	320	0.43	730	85	12.75	32	0.15	4.3E+04
		1,150	351	0.39	662	-68	12.75	32	0.13	3.9E+04
		1,240	378	0.38	646	-17	12.75	32	0.13	3.8E+04
		1,350	411	0.37	629	-17	12.75	32	0.13	3.7E+04

Screened intervals, conduit flow depths, cumulative conduit flows and well diameters from data presented by Davis et al. (1964). *Incremental conduit flows, flow velocities, Reynolds numbers* and SI unit entries calculated from those data by the author.

Kinematic viscosity used to calculate Reynolds numbers: 1.1092 mm^2/s at 16^o C

Reynolds number values greater than 2.1E+03 in pipes indicate departures from laminar flow (Vennard and Street, 1982)

ft bgs = feet below ground surface; m bgs = meters below ground surface; cfs = cubic feet per second; L/min = liters per minute; in = inches; cm = centimeters; m/s = meters per second

More recent research documented and evaluated well-scale transport occurring in the field. Corcho Alvarado et al. (2009) analyzed vertical profile data for tritium activity and temperature from an inactive long-screened supply well in an area where downward vertical hydraulic gradients were anticipated and found that the results were consistent with intraborehole transport. Landon et al. (2009) concluded that a long-screened public supply well in Modesto, California (B on Figure 1) acted as a conduit partly based on 1) local downward vertical hydraulic gradients and 2) correlation between periods of pump inactivity and concentration increases in deeper water-bearing zones of constituents typically found in shallow water-bearing zones. Bexfield et al. (2012), Bexfield and Jurgens (2014), Jurgens et al. (2008) and Jurgens et al. (2014) documented vertical groundwater flow and contaminant migration through municipal supply wells in two different parts of USA (Modesto, California and Albuquerque, New Mexico) during periods of nonoperation. Ambient flow was downwards in one well and upwards in the other. Effects on water quality, including elevated arsenic, nitrate and uranium concentrations, changed seasonally with vertical hydraulic gradients. The authors indicated the potential for similar occurrences in other alluvial basins with stratified water quality and vertical hydraulic gradients. Yager and Heywood (2014) modeled these two wells and showed that flow and contaminant migration during nonoperation degraded water quality. They showed that pumping during the off-season could reduce the undesired flows and water quality degradation but noted that the costs of the additional pumping should be weighed against costs of other options such as structurally modifying the wells to prevent connection of separate water-bearing strata.

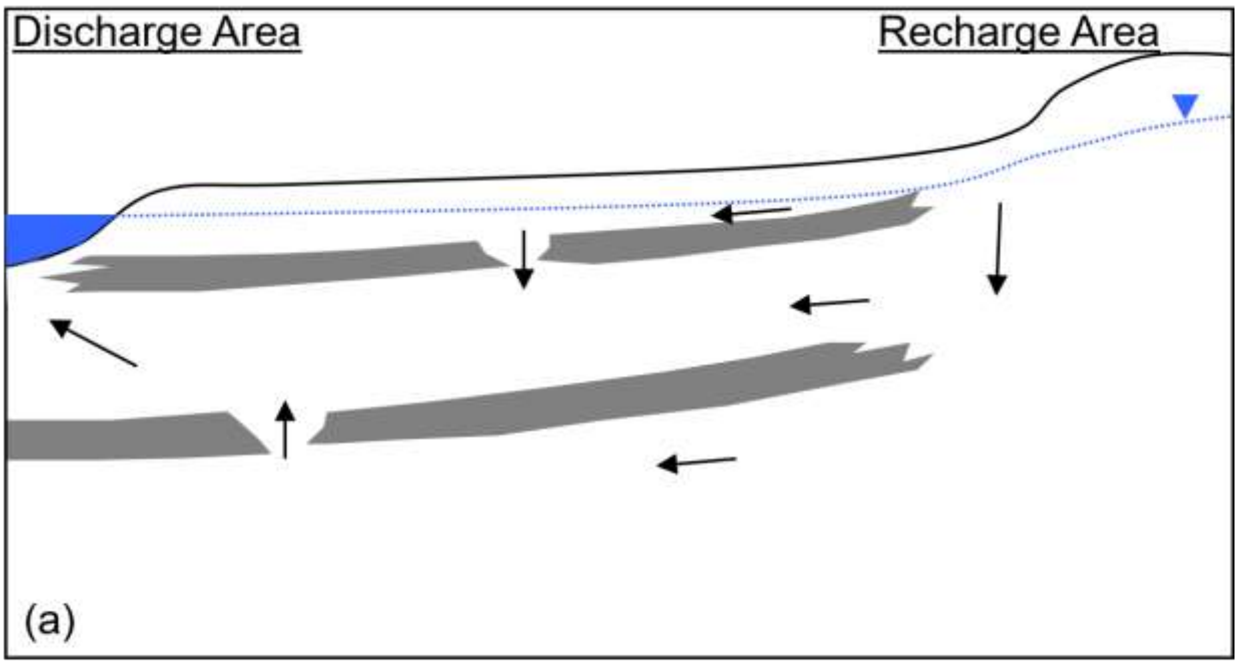
Others have evaluated the effects of flows through inactive wells at the regional scale. For the Central Valley of California, Williamson et al. (1989) estimated through numerical modeling that, in areas where the density of multi-aquifer wells equaled or exceeded one per square mile, the effective vertical hydraulic conductivity of confining layers in the groundwater system could increase by a factor of seven as a result of flow through well casings. Similarly, while modeling the western portion of the San Joaquin Valley, California, Phillips and Belitz (1991) and Belitz et al. (1993) found that the calibrated value for vertical hydraulic conductivity for the regional aquitard was higher than expected from permeameter data and reasoned that flow through the many multi-aquifer wells in the study area was the cause. Bertoldi et al. (1991) and Faunt (2009) provide additional support for these findings. Hanson et al. (2004) modeled groundwater flow in the Santa Clara Valley of California and estimated that approximately 19 percent of flow from shallower to deeper strata occurred through wells. Hart et al. (2006) investigated potential flow mechanisms responsible for a discrepancy between core-scale and regional estimates of vertical conductivity for an aquitard in Wisconsin, USA. Using a combination of numerical and analytical modeling, they found that flow through a single well could be approximately 700 L/min and a spacing of one well every ten kilometers (km) would be sufficient to account for the difference in flow implied by the different estimates of hydraulic conductivity. For a public supply well near York, Nebraska, a combination of field and modeling studies indicated that approximately 25 percent of the water that migrated from shallow to deeper groundwater crossed the aquitard by flowing directly through wells or boreholes that connected water-bearing strata separated by the aquitard (Clark et al, 2008, Landon et al., 2008 and Landon et al., 2009). Viers et al. (2012) and Mejia et al. (2012) evaluated the potential for nitrate migration from shallow groundwater to deeper strata over broad geographic areas using algebraic and numerical modeling methods, respecitvely. The potential effects of intraborehole flow and transport on a regional scale have also been considered in Poland (Dragon et al., 2009), Spain (Jiménez-Martínez et al., 2011) and Taiwan (Lu et al., 2008).

There are three significant limitations of the currently available literature. First, only three studies present actual measurements of intraborehole flow. McCombs and Fiedler (1927) measured a maximum flow equivalent to 520 g/min (1,970 L/min). Davis et al. (1964) measured a maximum flow rate of 478 g/min (1,852 L/min). Bexfield et al. (2012) measured a maximum flow rate of 204 g/min (772 L/min). Second, only one study presents information regarding the extent of water quality impact. Mayo (2010) reported that, after a well had been out of operation for 71 days, reversing the water quality impact from migration of total dissolved solids required running the well at its operational pumping rate of 175 g/min (660 L/min) for approximately 15 days (approximately 3.8 x 10^6 gal or 14.3 x 10^6 L extracted). Third, only one study suggests management approaches. Yager and Heywood (2014) suggested off-season pumping as a possible approach for limiting water quality impacts for specific wells. This current work provides additional field information on intraborehole flow measurements and the extent of water quality impacts as well as suggests several management approaches.

Overview of Flow and Transport Processes

Conditions when Intraborehole Flows Occur

Figure 2a depicts a sedimentary groundwater basin with multiple permeable water-bearing zones common in many parts of the world. Vertical head gradients, both upward and downward, will be present in basins as a natural consequence of recharge and discharge (Fetter, 1994; Phillips et al., 2007). Downward gradients also can result from hydraulic stresses induced by human activity such as groundwater extraction from deeper strata and deep percolation associated with crop irrigation (Figure 2b). In alluvial aquifers and other stratified sedimentary groundwater systems, pronounced bulk anisotropy of hydraulic conductivity ($K_{\text{vertical}}/K_{\text{horizontal}}$), perhaps on the order of 10^{-3} to 10^{-4}, often limits vertical flow and vertical hydraulic gradients can exceed horizontal gradients by orders of magnitude (Fogg, 1986). For conditions that exist in agriculturally stressed alluvial groundwater systems, horizontal and vertical hydraulic gradients can be on the order of 10^{-3} and 10^{-1}, respectively (see field examples #2 and #4 below). When an interconnecting vertical pathway exists, the vertical gradients are sustained regionally by the above-referenced hydraulic stresses and drive vertical flows on a continual basis.



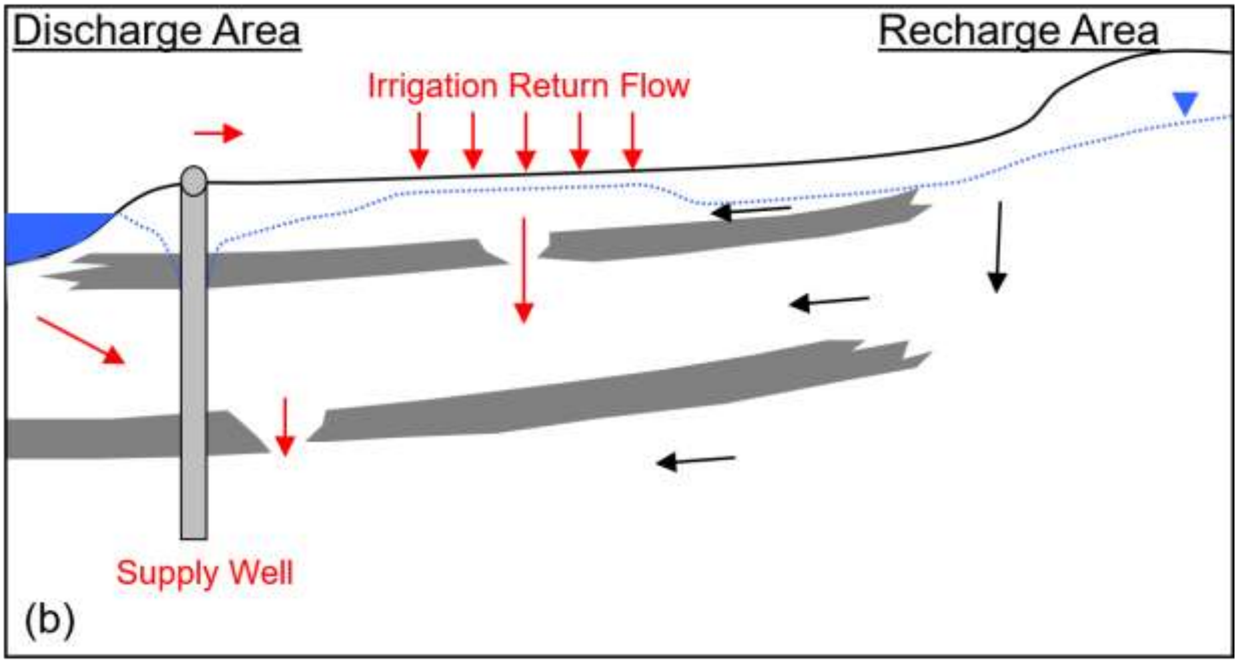


Figure 2: Simplified depiction of unconsolidated sedimentary groundwater basin: a) natural system and b) changes in groundwater stresses and flows from human activities. *White areas* indicate water-bearing zones. *Gray areas* indicate aquitards. Spatial variation in hydraulic conductivity within hydrostratigraphic units not shown. *Black arrows* indicate direction of groundwater flow. *Red arrows* indicate changes induced by human activities.

Interconnecting vertical pathways may exist naturally within the sedimentary structure (gaps in aquitards from erosion before burial or discontinuities in silts and clays deposited by spatially variable processes) or artificially as a result of well construction. Figure 3 shows a range of common well designs and preferential pathways for groundwater flow when vertical gradients are present. A properly designed and maintained well that provides little potential for vertical flow is depicted at the left of Figure 3 and serves as a basis for comparison. This well consists of 1) a well casing and screen installed within a borehole, 2) a cement seal filling the annular space between the borehole wall and the well casing that extends from ground surface down into the aquitard, 3) a gravel pack filling the annular space that extends from above the top to below the bottom of the screen and 4) a gravel fill tube for replenishing the gravel pack in the event of any settlement resulting from operations. Common elements that create the potential for wells to act as vertical conduits include long screens, multiple shorter screens separated over long vertical distances and long gravel packs (Santi et al., 2006). Another common well condition leading to a well becoming a vertical conduit is localized failure of the casing or other steel component (tool port or gravel fill tube) from corrosion or failure of a welded joint. Such a failure allows hydraulic communication between shallower and deeper strata separated by relatively long distances through the interior of the well. The term "long" is used within the context of the hydrogeology at a particular well site. Well elements are considered long if they cross hydraulic barriers created by finer-grained strata where differences in hydraulic head occur (see Figure 3). Conditions that result in flow through the well casing instead of the gravel pack will generally result in higher velocities and larger volumes of water transferred between strata because, as discussed below, resistance to flow will be many orders of magnitude less inside the casing.

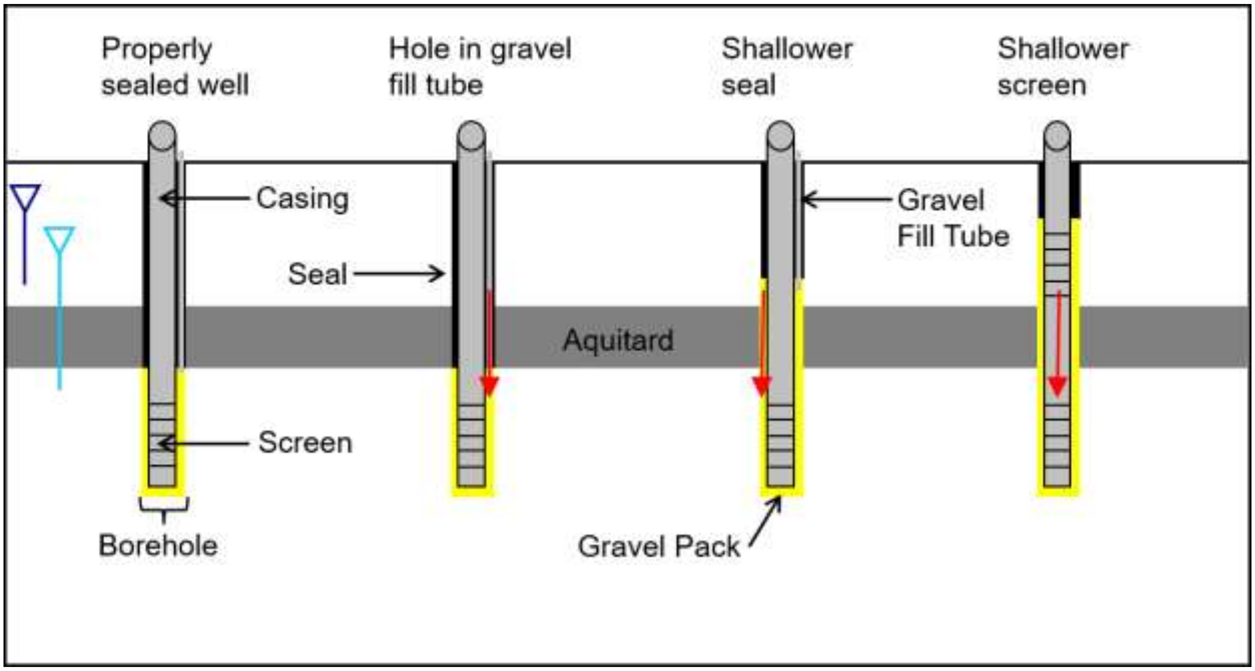


Figure 3: Range of well conditions. *Red arrows* indicate potential vertical flow pathways.

Consistent with Darcy's Law, the potential magnitude of intraborehole flow depends upon both resistance along the flow path and the hydraulic gradient that drives the flow. Typical vertical hydraulic conductivity values for sediments range from approximately 10^{-7} to 10 centimeters per second (cm/s; Freeze and Cherry, 1979; Fetter, 1994) while effective pipe conductivities for well casings or screens and gravel fill tubes are much higher at approximately 10^4 to 10^6 cm/s. Pipe conductivities may be calculated using the Hagen-Poiseuille equation for assumed laminar flow conditions: $K_{well} = r^2\rho g/8\mu$, where r is the inner radius of the well casing or screen, ρ is the fluid density, g is the gravitational acceleration and μ is the fluid viscosity (Reilly et al., 1989 and Lacombe et al., 1995). Because the hydraulic conductivity values for well components are orders of magnitude greater than those for sediments, it is energetically more favorable for water to flow through the well, either the gravel filled borehole annulus or the open casing, than across the lower hydraulic conductivity strata. The range in vertical hydraulic gradients are discussed in the field examples below, but experience suggests that they can be at least as high as -0.8 (see Field Example #2 below). As indicated above and further discussed in the field examples below, flow rates can range up to thousands of liters per minute.

Based on field data, Mayo (2010) found that vertical head differences of 0.1 m could be significant for intraborehole flow where the higher-head stratum is contaminated. However, even smaller head differences can drive intraborehole flows that are quite high since the effective hydraulic conductivity for a well is many orders of magnitude greater than for a natural porous medium. A head difference of as little as 0.01 m over a vertical distance of 10 m would create a vertical gradient on the same order of magnitude as typically observed for horizontal gradients, but the flow through a conduit well would be many orders of magnitude greater than through the natural system given the higher hydraulic conductivity of the well. Because head differences of 0.01 m are not easily measured in the field, many intraborehole flows may not be apparent unless measurements of flow in the wellbore are made or water quality data are collected over time.

Transport Impacts from Intraborehole Flows

Areas where flow through wells is investigated are often located in groundwater systems significantly affected by human activities that create or enhance downward gradients. In these areas, shallow water that is also often affected by activities at ground surface and poorer in quality (i.e., nitrate, pesticides and other organic compounds from anthropogenic sources; Burow et al., 2007) migrates downward. Migration through wells bypasses lower hydraulic conductivity sediments and reduces water quality protections afforded by 1) longer travel times of perhaps 100's to 1,000's of years and 2) beneficial chemical reactions such as denitrification (McMahon et al., 2008). Deeper water quality is degraded in these cases. However, shallow groundwater quality impacts may also occur where flow is upward through wells and deeper strata contain naturally occurring undesirable constituents such as arsenic, chloride or hexavalent chromium (Izbicki et al., 2005a, 2005b, 2006, 2008 and 2015).

In addition to problems from well construction, well management practices contribute to water quality impacts from vertical flows through wells. Extended well inactivity (months to years) allows ongoing flows to accumulate larger volumes of water and masses of dissolved constituents moving between strata. Irrigation and some municipal supply wells are commonly left inactive each year during the winter when demands decrease. Common naturally occurring constituents that can be mobilized include the inorganic constituents mentioned above as well as dissolved gasses such as oxygen. Anthropogenic constituents that can be mobilized include nitrate, volatile organic compounds, total dissolved solids including chloride, fumigants, pesticides and uranium (if mobilized by changing redox conditions with agricultural irrigation, see Jurgens et al., 2010). When intraborehole flow occurs, contaminated water displaces better quality water at depth when the well is inactive (Figure 4a). As depicted in Figure 4b, the quality of produced water then includes a less favorable mix upon restarting the well (Bexfield and Jurgens, 2014).

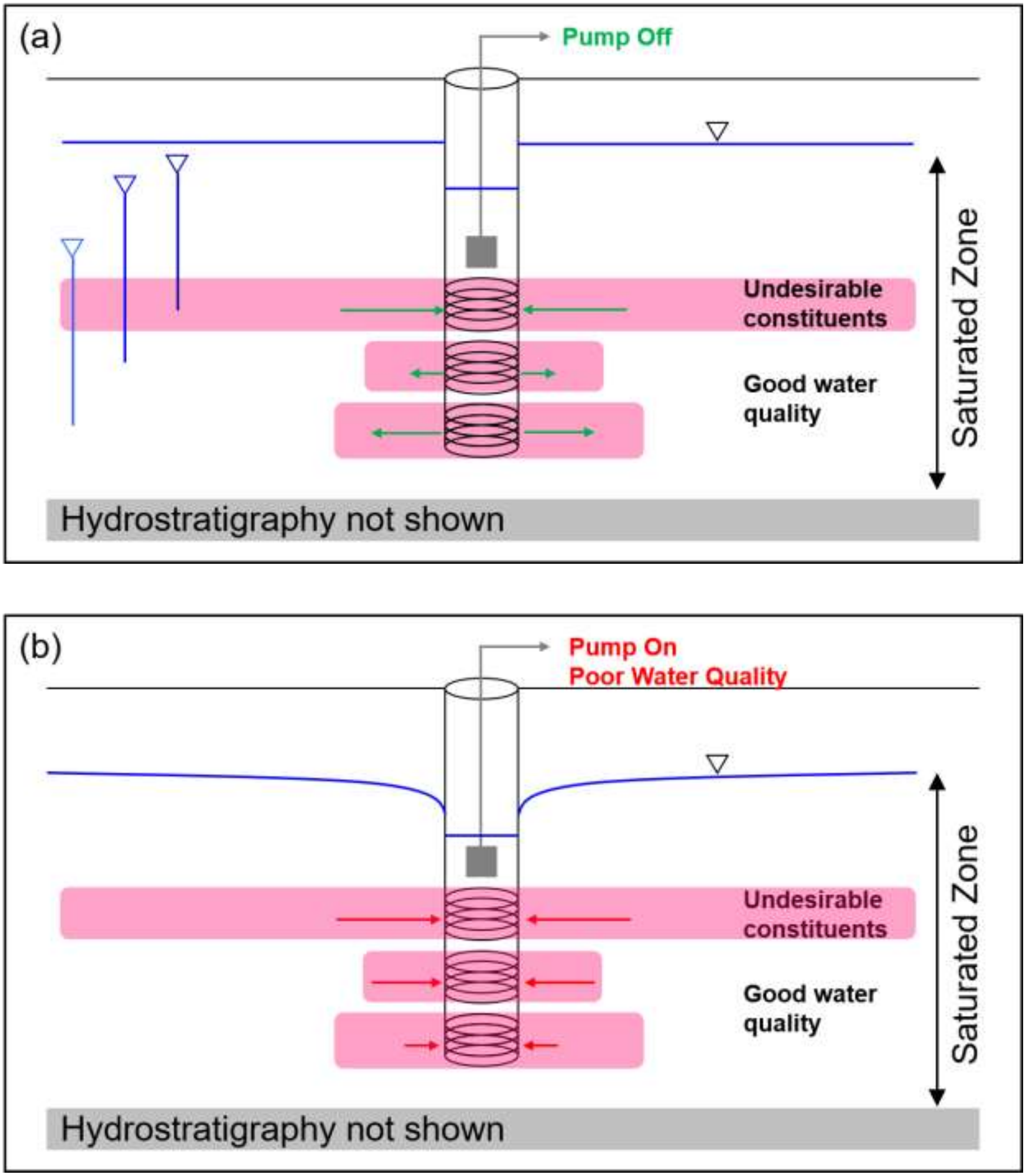


Figure 4: Schematic of well that acts as a conduit: a) vertical flow and contaminant migration through an inactive well and b) water quality degradation during pumping after contaminant

migration through conduit. Lengths of *arrows* indicate flow magnitudes in a heterogeneous hydraulic conductivity field.

The hydraulic performance of a well can also be reduced by conduit flow and transport. Mixing of geochemically dissimilar waters from different strata often causes precipitation of inorganic chemical species that clog well screens (Houben and Treskatis, 2007; van Beek, 2012). Oxygenation of deeper waters can spur bacterial growth that also clogs well screens (Mansuy, 1999). In both cases, suspended sediment from the aquifer material that is often entrained in the flow can become adhered to well components by the inorganic precipitates and bacterial growth resulting in greater volume of clogging material and exacerbating hydraulic performance problems (Houben and Treskatis, 2007). The well pump can become fouled and inefficient through the same processes (Houben and Treskatis, 2007; Smith and Comeskey, 2010).

It should be noted that monitoring wells can also act as vertical conduits depending upon factors including length of well screen and location within the flow field. Such occurrences can be important for contaminated site cleanup and research has focused on understanding concentration data collected from monitoring wells (Reilly et al, 1989; Church and Granato, 1996; Elci et al., 2001; Ma et al., 2011; McMillan et al., 2014; Vitale and Robbins, 2016). However, the differences in construction between water supply wells and monitoring wells, primarily casing diameter and length of well screen, create the potential for greater flows through supply wells where there is less resistance to flow as well as greater head difference between shallow and deep strata. In general, the flow magnitudes and migration distances, both vertical and horizontal, are expected to be greater for water supply wells than monitoring wells. Vertical flow through monitoring wells is not considered in this work.

Methods of Investigation

Field Work

The general locations of wells investigated in the field are indicated on Figure 1. Field examples #1 and #2 present intraborehole flow measurements while field examples #3 and #4 address water quality impact evaluations. For Field Example #1, intraborehole flow under non-pumping conditions was measured on a project conducted by the author using a dye tracer method derived from the method of Izbicki et al. (1999). Similarly, Field Example #2 involved measuring intraborehole flow under non-pumping conditions on a project conducted by others (see discussion of field example below) using an impeller flowmeter (Welenco, 1996; Keys, 1997). The above-referenced measurement methods entail recording dye travel times and impeller rotation rates at different points along well screens to estimate intraborehole flows. For Field Example #3, water quality impact in the vicinity of a well that acted as a vertical conduit was evaluated on a project conducted by the author through 1) pumping the well for an extended period while monitoring nitrate concentrations in the discharge and 2) profiling flow into the well along the well screen with a dye tracer and obtaining depth-specific water samples for analysis using the previously referenced method of Izbicki et al. (1999). Finally, Field Example #4 entailed a hydrogeologic investigation on a project conducted by the author. Methods used included flow and concentration profiling along the well screen of a municipal well using the previously referenced method of Izbicki et al. (1999) as well as subsurface characterization by routine methods (monitoring well installation, water sampling and performing a pumping test).

Management Analysis

Well-Specific Approaches

Options for managing intraborehole flow and transport on an individual well basis were explored through modeling a hypothetical two-aquifer system penetrated by a single well as in Figure 3. The model was implemented as a half-space similar to Konikow and Hornberger (2006) using the MODFLOW-2000 (Harbaugh et al., 2000) and MT3DMS (Zheng and Wang, 1999) modeling codes. Horizontal finite difference grid dimensions were 10 by 10 ft (3 by 3 m) in an area surrounding the simulated well (Figure 5). Grid refinement to 0.25 by 0.25 ft (8 by 8 cm) occurred for the well itself and grid coarsening was implemented towards the domain boundaries. Vertical grid dimensions were 10 ft (3 m) in the shallow aquifer and the intervening aquitard and 5 ft (1.5 m) in the deep aquifer. The shallow and deep aquifers were simulated as 100-ft (30 m) thick sands with horizontal hydraulic conductivity of 2.8 x 10^{1} ft/d (10^{-2} cm/s) and vertical hydraulic conductivity of 2.8 ft/d (10^{-3} cm/s). The intervening aquitard was simulated as a 50-ft (15 m) thick clay with horizontal hydraulic conductivity of 2.8 x 10^{-2} ft/d (10^{-5} cm/s) and vertical hydraulic conductivity of 2.8 x 10^{-3} ft/d (10^{-6} cm/s). A spatially uniform storage coefficient value of 10^{-4} was used. Recharge was made horizontally uniform at a value of 3 x 10^{-3} ft/d (9 x 10^{-4} m/d). Specified-head boundary conditions that remained constant over time were set on the 1) up-flow and down-flow edges of the model grid in the aquifers to establish a horizontal hydraulic gradient of 0.001 and 2) bottom of the model grid to establish a downward vertical hydraulic gradient. The combination of the head boundary conditions and recharge resulted in a head difference across the aquitard of 7.5 ft (2.3 m) in the absence of the conduit well, which corresponded to a vertical gradient of -0.15. The concentration of a conservative solute was set at 100 milligrams per liter (mg/l) in the shallow aquifer and 10 mg/l in the deep aquifer. Advective solute transport was simulated using a porosity of 0.3 and implemented using the Method of Characteristics.

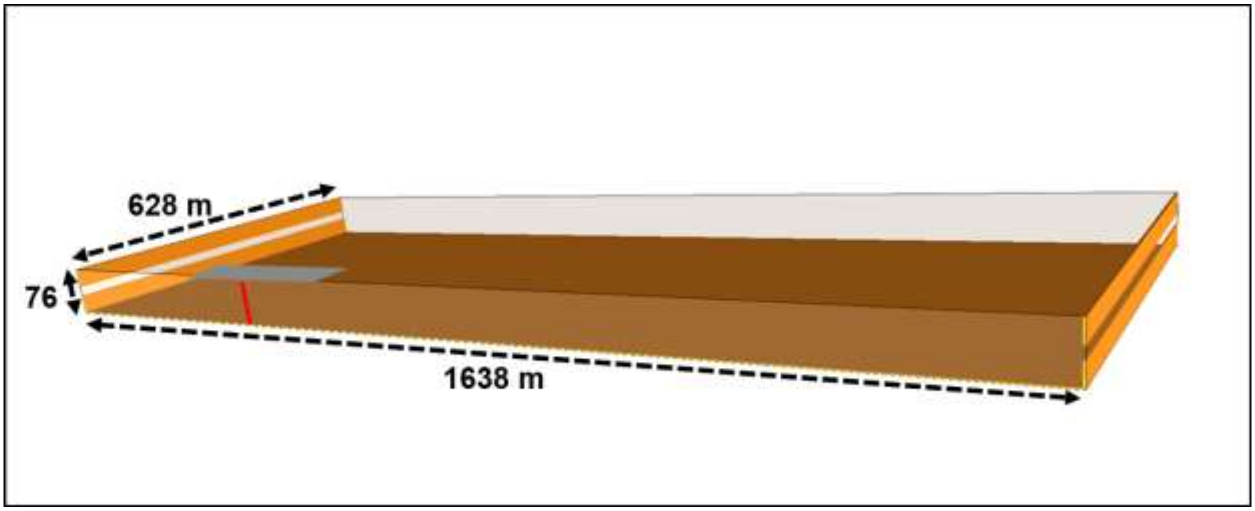


Figure 5: Groundwater model configuration. *White arrows* indicate model domain dimensions. The *grey area* indicates the location of the higher density finite difference grid. The *red line* indicates the conduit well location. *Orange areas* indicate specified head boundaries.

Twelve-inch (in), or 30 cm, diameter well casing and screen were simulated within a 24-in (61 cm) diameter borehole. An effective daily average pumping rate of 125 g/min (473 L/min), equivalent to 500 g/min (1,893 L/min) for six hours each day, was applied; however, only half of this value was used since the model was implemented as a half-space. The well consisted of 1) a 50 ft (15 m) long well seal from 0 to 50 ft bgs (0 to 15 m bgs), 2) a 50 ft (15 m) long well screen in the shallow aquifer placed immediately above the aquitard from 50 to 100 ft bgs (15 to 30 m bgs), 3) a 50 ft (15 m) long well screen in the deep aquifer placed immediately below the aquitard from 150 to 200 ft bgs (45 to 60 m bgs) and 4) one continuous gravel pack from 50 to 205 ft bgs (15 to 60 m bgs). The well was simulated with an assemblage of model grid cells (Horn and

Harter, 2009; Houben and Hauschild, 2011). To simulate the well seal, inactive cells were placed at grid locations that corresponded to the borehole annulus and well casing from 0 to 50 ft bgs (0 to 15 m bgs). The gravel pack was simulated with horizontal and vertical hydraulic conductivities set at 2.8×10^3 ft/d, or 1 cm/s, generally consistent with Houben and Hauschild (2011). To simulate flow in the well, a value one hundred times that of the gravel pack (2.8×10^5 ft/d, or 100 cm/s) was used for cells at grid locations that corresponded to the inside of the well This value was, however, lower than that predicted for laminar flow conditions by Hagen-Poiseuille (2.5×10^6 cm/s) in order to simulate enhanced head loss within the well during pumping as a result of turbulence. This approach was reasonable given that the Reynolds number for the specified casing diameter and pumping rate would be well beyond the upper range for laminar flow (1.2×10^5 compared to 2.1×10^3; Vennard and Street, 1982).

Simulating the unmanaged case entailed modeling transient flow and transport with a pumping schedule as follows: inactivity from day 0 through 180, pumping from day 181 through 360, inactivity from day 361 through 540, pumping from day 541 through 720. Consistent with typical pump settings for water supply wells, the pump intake location was simulated as located in the well casing above the shallower screen. Management options were evaluated by changing the pumping schedule and pump intake depth.

Representing the well as an equivalent porous medium, rather than using the MODFLOW Multi-Node Well Package (Konikow et al., 2009), was consistent with previous approaches by some researchers (Lacombe, 1995; Halford, 2009; Horn and Harter, 2009; Halford et al., 2010; Houben and Hauschild, 2011; Mejia et al., 2012; McMillan et al., 2014) and allowed simulation of head loss as water flowed vertically inside the well during pumping. Because the MODFLOW Multi-Node Well Package assumes no head loss inside the well, it may over allocate pumping flows along the well screen for portions of the screen located farthest from the pump. In such cases, variation in the areal extent of the groundwater capture zone with depth and the potential migration of solutes beyond the capture zone would not be properly simulated. One of the management approaches addressed in this work, changing the pump depth to adjust the capture zone extent with depth, required the simulation of head loss inside the well; therefore, the Multi-Node Well Package was not used. Since this work also simulates intraborehole flows under non-pumping conditions, the selected approach may underestimate those flows because of the lower hydraulic conductivity used inside the casing. As a result, expected volumes transferred between aquifers could be larger and water quality impacts could be greater than those simulated.

Regional Approaches

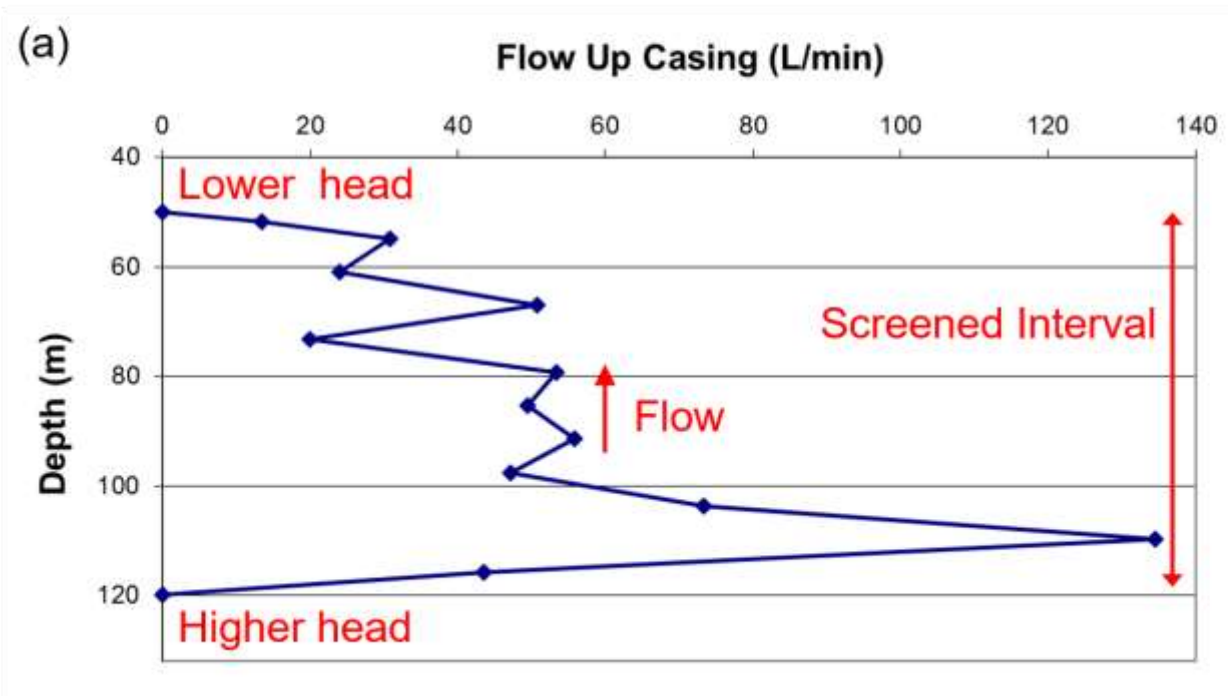
Options for managing intraborehole flow and transport by focusing limited resources on a regional basis were explored using geographical analysis of well construction log data. The northern portion of California's Central Valley (Figure 1) was used for demonstration based on data availability but the concepts apply more broadly. The number and diameter of wells in the study area were tabulated on a 1 km grid spacing then the increase in vertical hydraulic conductivity was estimated using the method presented in the appendix. When combined with a threshold for factor increase in natural vertical hydraulic conductivity, the results can be used to target areas of potential high impact for investigation.

Results and Discussion of Field Investigations

Measurments of Intraborehole Flow

Field Example #1: Upward Flow in a Regional Discharge Area

A 14-in (36 cm) diameter well near the Sacramento River (see #1 on Figure 1) was investigated to determine the cause of water quality impact. Part of the investigation involved measuring intraborehole flow. Figure 6a presents a cumulative flow profile under ambient conditions during the winter. (Cumulative flows are measured inside the well along its length.) Upward flow occurred over the entire 229-ft (70 m) screened interval. Upward hydraulic gradients and groundwater flow are common in areas where the saturated zone is in contact with the river and seasonal pumping stresses do not prevent natural groundwater discharge. The maximum flow along the well was approximately 35 g/min (132 L/min).



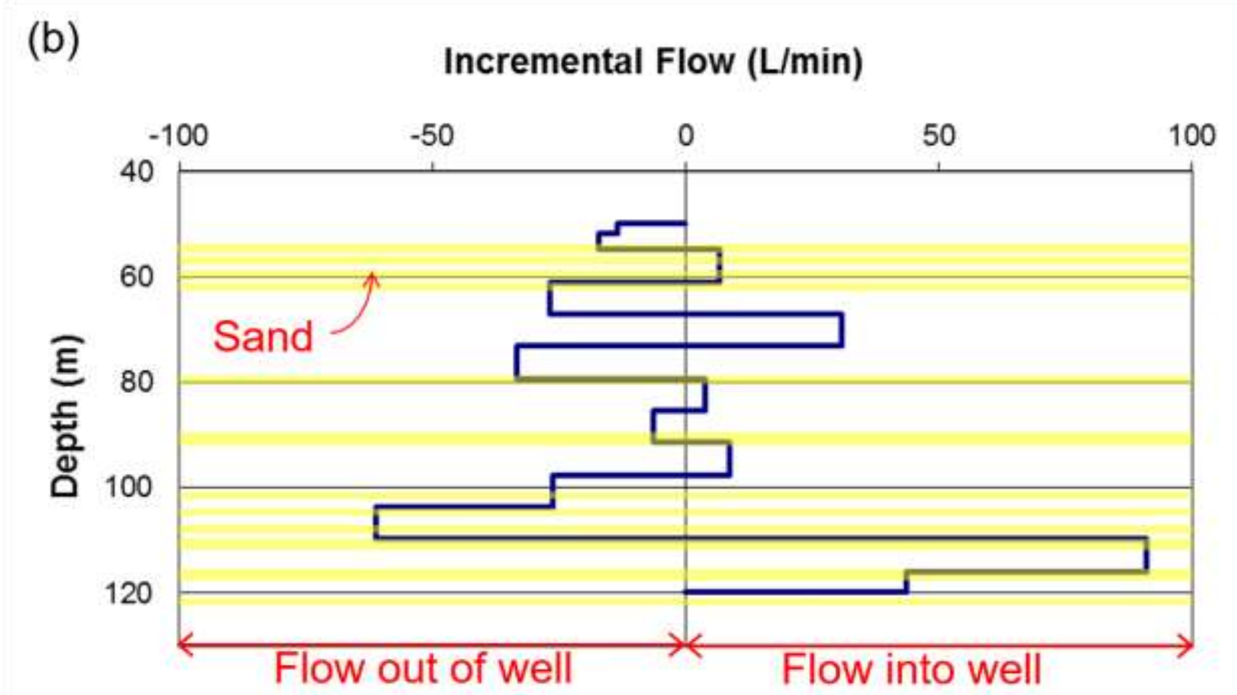
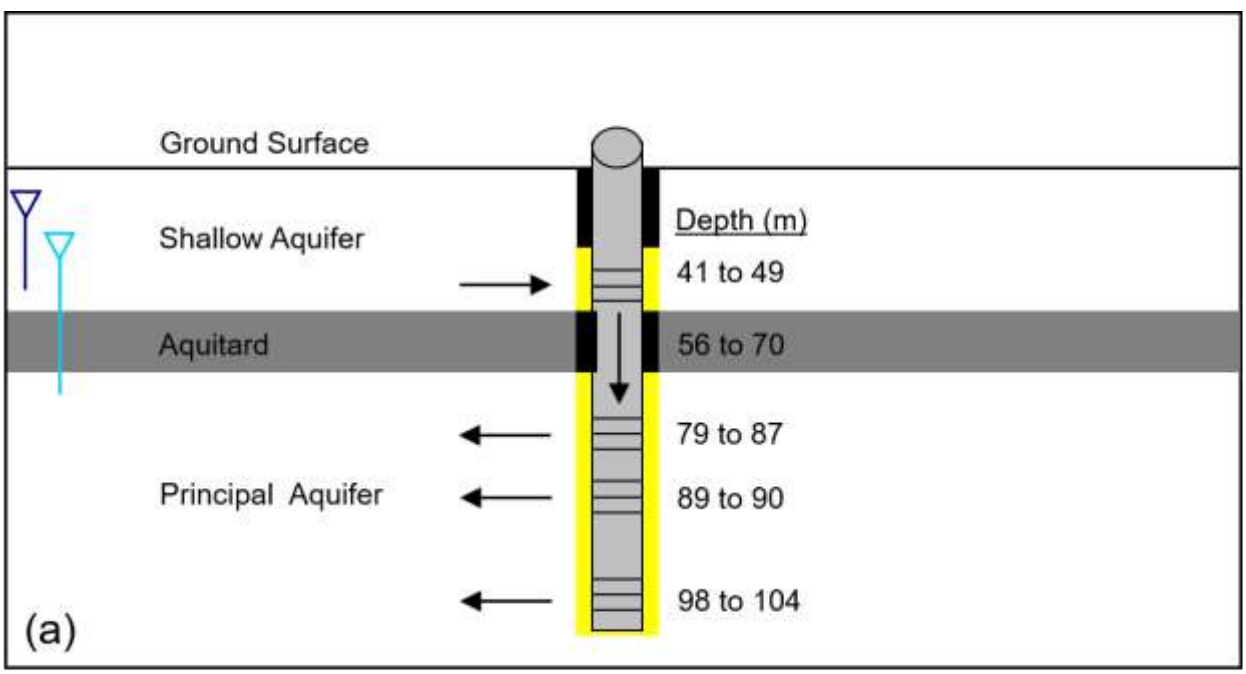


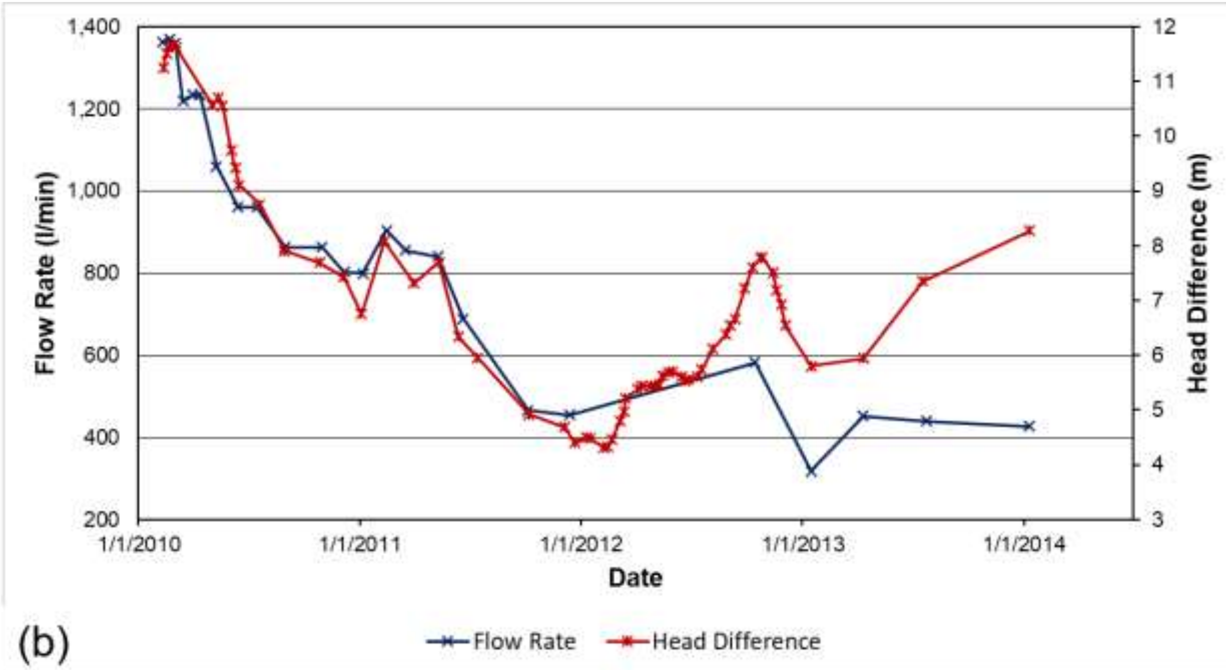
Figure 6: Field Example #1: a) cumulative flow profile conducted under ambient (nonpumping) conditions and b) incremental flow profile derived from the cumulative flow profile.

Figure 6b presents the incremental flow profile for the data presented in Figure 6a as well as the depths and thicknesses of sand layers identified in the lithologic log that was recorded when the wellbore was drilled. (Incremental flows are flows into and out of the well calculated as differences between vertically adjacent flow measurements inside the well with upstream subtracted from downstream.) Depths of the larger inflows and outflows generally coincided with the sand layers. Depths where flows occurred without indications of sand could result from inaccuracies in the lithologic log (sand actually present) or interaction of flows inside the well with the gravel pack. Even though the entire screened interval for this well was within the same water-bearing zone, stratification of the alluvial sediments created conditions conducive to the long-screened well acting as vertical conduit for flow (development of vertical hydraulic gradients and a preferential flow path through the well as a result of the contrast in hydraulic conductivity between the fine-grained strata contained in the alluvial sequence and the open well structure).

Field Example #2: Downward Flow in a Heavily Pumped Basin

Figure 7a presents a schematic for an experimental well installed in Southern California by the Orange County Water District (OCWD) to evaluate the possibility of conveying recharge water to depth through wells (see #2 on Figure 1). A significant amount of water enters the groundwater system as recharge to the Shallow Aquifer through artificial ponds and exits the system through water supply wells pumping from the deeper Principal Aquifer. However, an aquitard present from approximately 185 to 230 ft bgs (56 to70 m bgs) impedes the flow of water between aquifers. The well was constructed of 8-in (20 cm) diameter casing and wire-wrapped screen. There was one 25-ft (8 m) screened interval in the Shallow Aquifer and three screened intervals with a combined length of 49 ft (15 m) in the Principal Aquifer.





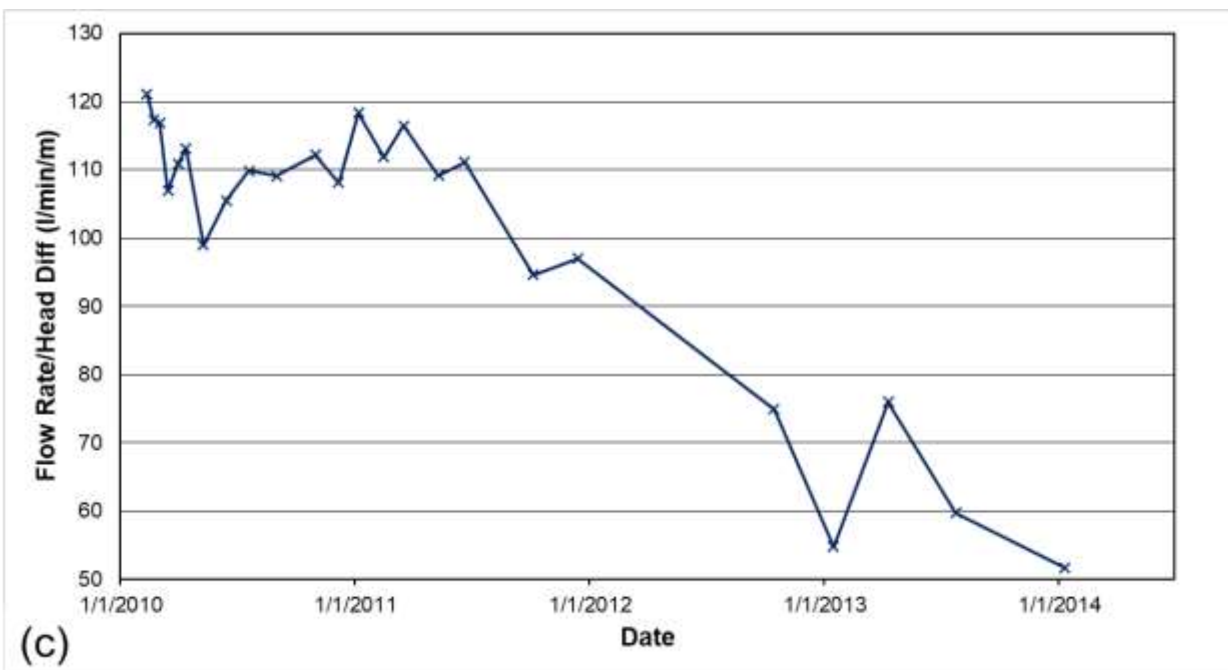


Figure 7: Field Example #2: a) schematic of experimental transfer well, b) flows in well under ambient conditions and head differences driving flows and c) **c**hange in flow response to driving head difference over time.

Figure 7b presents results of 24 flow measurements collected by OCWD with an impeller flowmeter between the shallowest and deeper screened intervals over 47 months. Flows ranged between approximately 80 and 360 g/min (300 and 1,360 L/min). Based upon measurements from shallow and deep short-screened monitoring wells located approximately 38 ft (12 m) away from the experimental well, the head difference across the aquitard varied between approximately 14 and 38 ft (4 and 12 m; Figure 7b). Using these head differences and the aquitard thickness, the vertical hydraulic gradient driving flow down the well was calculated as ranging from approximately -0.3 to -0.8. The flow measurements indicate that approximately 350 x 10^6 gal (1.3 x 10^9 L) of water flowed down the well over the 47-month monitoring period.

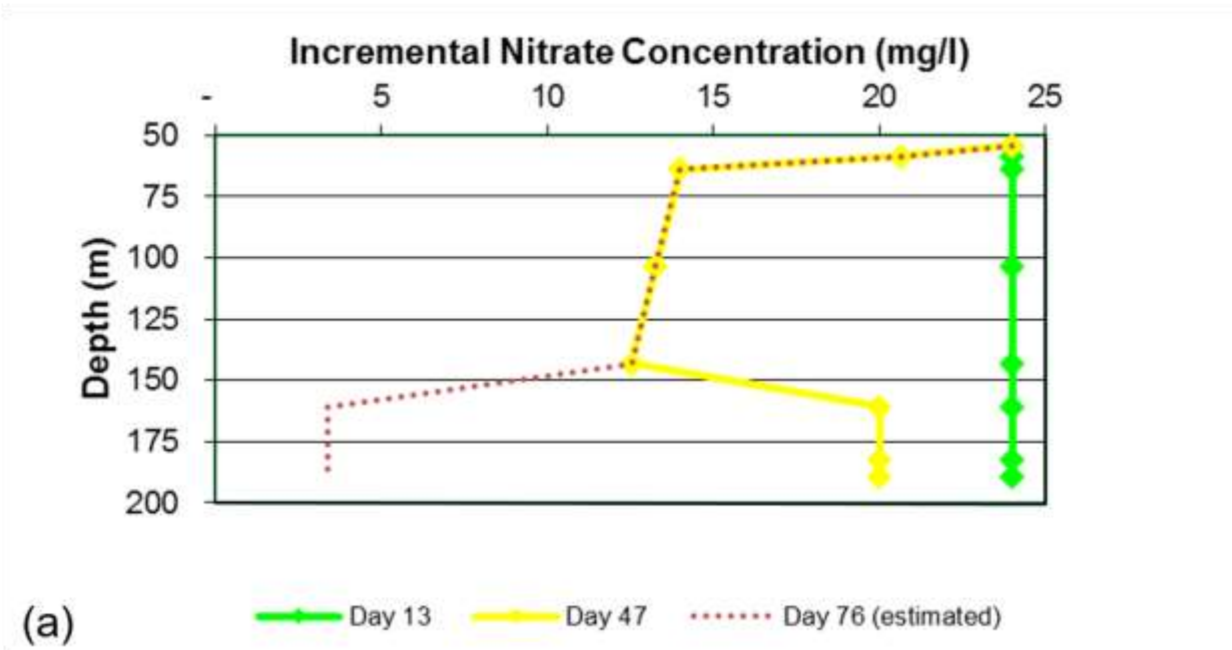
Figure 7c plots the decrease over time in rate of flow divided by the observed head difference across the aquitard. This trend is taken as an indication that the deeper well screens, gravel pack and, possibly, the sediments in the near-well environment began to clog. The causes of clogging were not investigated by OCWD but could have included several processes discussed above (accumulation of suspended sediments as well as development of inorganic precipitation and/or bacterial growth resulting from mixing of geochemically different waters).

Extent of Water Quality Impact

Field Example #3: Water Quality Degradation at the Well-Scale

A 16-in (41 cm) diameter well in the Sacramento Valley was investigated to determine the cause of nitrate contamination (see #3 on Figure 1). The lithologic log for the well and other available information regarding the local geology indicated that the sediments in the groundwater basin comprised a stratified sequence of finer and coarser materials. Significant groundwater pumping and agricultural irrigation occurred in the area and downward hydraulic gradients had been measured in the greater vicinity of the well. The seven screened intervals for the well spanned a significant vertical distance (451 ft, or 137 m; Figure 8a) and the well had not been pumped for over one year. Together, these conditions created the potential for the well to act as a vertical conduit for downward flow and migration of shallow contaminants to depths where better quality water was present.

Figure 8a presents incremental nitrate water quality profiles for the well collected under pumping conditions at different times as the well was continuously purged for an extended period. The later-time profiles indicate that nitrate concentrations decreased with depth. This is consistent with 1) the constituent source being at the ground surface, 2) concentration attenuation with distance from a solute source and 3) increased application rates over time along with long transport times to the deeper part of the groundwater system. Figure 8b presents the wellhead concentrations over a 76-day period when the well was pumped continuously at 600 g/min (2,270 L/min). The initial concentration from the pumping period (slightly less than 24 mg/l on Figure 8b) was consistent with a mix of concentrations from the two shallowest depths sampled for the profile (21 and 24 mg/l on Figure 8a). These observations are consistent with the expectation that the well acts as a vertical conduit for downward flow and contaminant migration during periods of no pumping. Referring back to Figure 4a, groundwater that contained nitrate concentrations from the shallower strata migrated down through the well, displaced water at depth that contained lower concentrations, and created a zone near the well containing relatively uniform water quality along the entire well screen. Consistent with Figure 4b, water quality at the beginning of pumping reflected shallow water quality since the water located along the entire length of the well screen originated in the shallower strata. As pumping progressed, the higher concentration water that had migrated to depth was removed from the deeper strata and the well ultimately pumped from the heterogeneous distribution of water quality representative of conditions in the greater vicinity of the well (day 76 on Figure 8a).



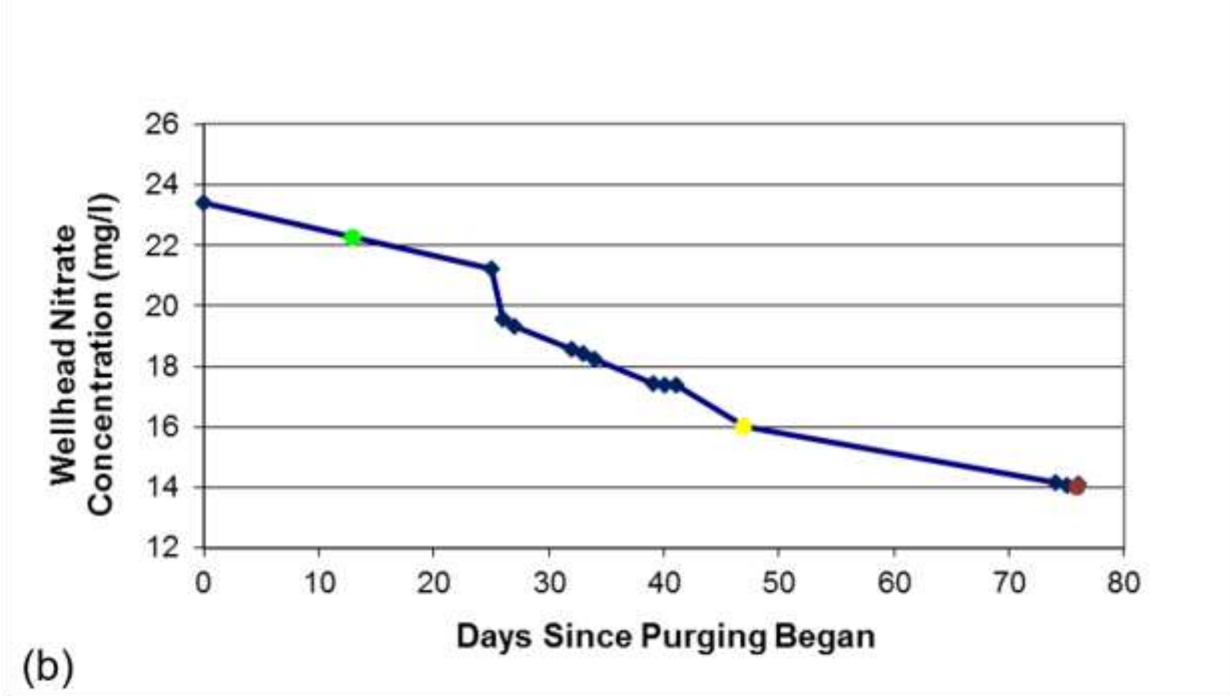


Figure 8: Field Example #3: a) evolution of an incremental nitrate concentration profile with extended purging (A cumulative water quality profile consists of concentrations associated with samples collected from inside the well, while an incremental water quality profile is the result of differencing the flow-weighted cumulative concentration profile as described by Izbicki et al., 1999. The incremental water quality profile indicates the depths and concentrations of dissolved constituents in the strata adjacent to the well screen. *Day 76 concentration profile* at depths greater than 150 m estimated based upon day 76 *wellhead concentration*, shallower portion of *Day 47 concentration profile* and the incremental flow profile.), b**)** nitrate concentration decrease at wellhead during extended period of well pumping. *Colored diamonds* on Figure 8a indicate the incremental concentrations. *Colored circles* on Figure 8b correspond to profiles presented on Figure 8a. The applicable drinking water quality standard is 45 mg/l.

Dynamic (pumping) flow and water quality profiles for the well at 13 and 47 days into the 76-day pumping period document the well purging process. The incremental water quality profiles for these data collection events are presented on Figure 8a. After 13 days of pumping, the water quality profile was homogeneous with a single concentration along the well screen that was consistent with shallow water quality. At this point, with a pumping rate of 600 g/min (2,270 L/min), over 11 x 10^6 gal (42 x 10^6 L) of water containing a relatively high nitrate concentration had been extracted. Clearly, a larger volume of water had migrated down the well. After 47 days of pumping, concentrations decreased with depth, as would be expected once the well was purged, for all but the deepest portion of the water quality profile. The effects of shallow water migrating to depth through the well had been reversed in the sandier strata. Only the deeper strata containing clay, as indicated by the lithologic log, retained poorer quality water that had migrated from the shallower strata. After another 29 days, the effects of contaminated water flowing down the well while it was inactive appeared to have been fully reversed. These data clearly illustrate two points: 1) purging a well after long periods of inactivity can require weeks to months and 2) adequate purging of water supply wells before concentration profiling is necessary to obtain accurate information.

A total of more than 65 x 10^6 gal (246 x 10^6 L) of water was pumped to reverse the effects of the well acting as a vertical conduit. In this case, the vertical flow did not create a water quality problem because the nitrate concentration in the migrating water (21 to 24 mg/l) was below the drinking water standard (45 mg/l) and no action was required. However, shallower groundwater concentrations of nitrate, and possibly other constituents, exceed drinking water standards in many areas. The final example considers such a case.

Field Example #4: Water Quality Degradation at the Wellfield-Scale

The wells considered in this example were near the Sacramento-San Joaquin River Delta (see #4 on Figure 1). Groundwater pumping and agricultural irrigation are prevalent in the area. Lithologic logs for nearby wells plus other information on the local geology indicated a stratified sequence of finer and coarser sediments within the groundwater system. Consistent with these conditions, a downward vertical hydraulic gradient had been documented locally. A pressure transducer survey was conducted for 13 months using monitoring wells with 20-ft (6 m) screens. Two shallow and two deep wells were installed to approximately 35 (shallow) and 180 (deep) ft bgs (11 and 55 m bgs) and equipped with pressure transducers. The data collected showed downward vertical hydraulic gradients throughout the year with seasonal variation that coincided with pumping. Gradients were between 0 and -0.01 during the winter and roughly doubled to between -0.01 and -0.02 during the irrigation season. A 30-day pumping test on a nearby municipal well screened in the deeper portion of the groundwater system at approximately 2,000 g/min (7,570 L/min) increased the gradients to between -0.03 and -0.04 at a distance of approximately 1,000 ft (305 m) from the pumping well. Horizontal hydraulic gradients varied somewhat from shallow to deep and between seasons but were less than or equal to 0.001.

An 8 5/8–in (22 cm) diameter industrial supply well constructed in 1974 to a depth of approximately 200 ft bgs (61 m bgs) had an annular seal that extended to the minimum regulatory standard of 50 ft bgs (15 m bgs) and a gravel pack that extended from approximately 150 to 200 ft bgs (46 to 61 m bgs). It was not clear what type of material was used to fill the annular space from 50 to 150 ft bgs (15 to 46 m bgs). Possibilities included gravel pack as well as a collapse of the borehole and infilling with the surrounding mix of sediments. A 20-ft (6 m) screen was present at the bottom of the well. This well was on the grounds of a fertilizer formulation facility where shallow groundwater was highly contaminated with nitrate. Proximity of the industrial supply well to shallow groundwater contamination and the short annular seal created potential for nitrate migration to depths with better water quality. Decades after the industrial supply well was constructed, the above-referenced municipal supply well capable of producing approximately 2,000 g/min (7,570 L/min) was installed approximately 1,100 ft (335 m) from the industrial supply well. The proximity of the high-capacity municipal well increased downward vertical gradients in the area during pumping operations.

The higher nitrate concentrations in shallow groundwater near the industrial supply well ranged from approximately 1,000 to 4,000 mg/l nitrate (compared to a drinking water standard of 45 mg/l). The industrial well was pumped only intermittently and seasonal variations in nitrate concentration at the wellhead ranged from approximately 20 to 900 mg/l. Ten years of water quality data indicated that the highest concentrations occurred in the well during summer when downward gradients were the greatest. Investigation of the groundwater nitrate distribution, as well as groundwater age dating (Singleton et al., 2010), indicated impacts to deeper water quality (Figure 9). The concentration at the industrial well (917 mg/l on Figure 9) was higher than in surrounding wells screened at similar depths by a factor of ten or more. Also, the vertical distribution of nitrate in the municipal supply well did not decrease with depth, as was the case for Example #3 above, which would be expected for a constituent where the source is at ground surface. Furthermore, the groundwater age distribution indicated younger water at depth near the industrial supply well (average of 27 years on Figure 9) relative to the surrounding groundwater by a factor of as much as 100 percent. These observations indicated that the industrial supply well acted as a vertical conduit for downward flow and nitrate migration and that the impact to groundwater quality extended more than 1,000 ft (300 m) from the well.

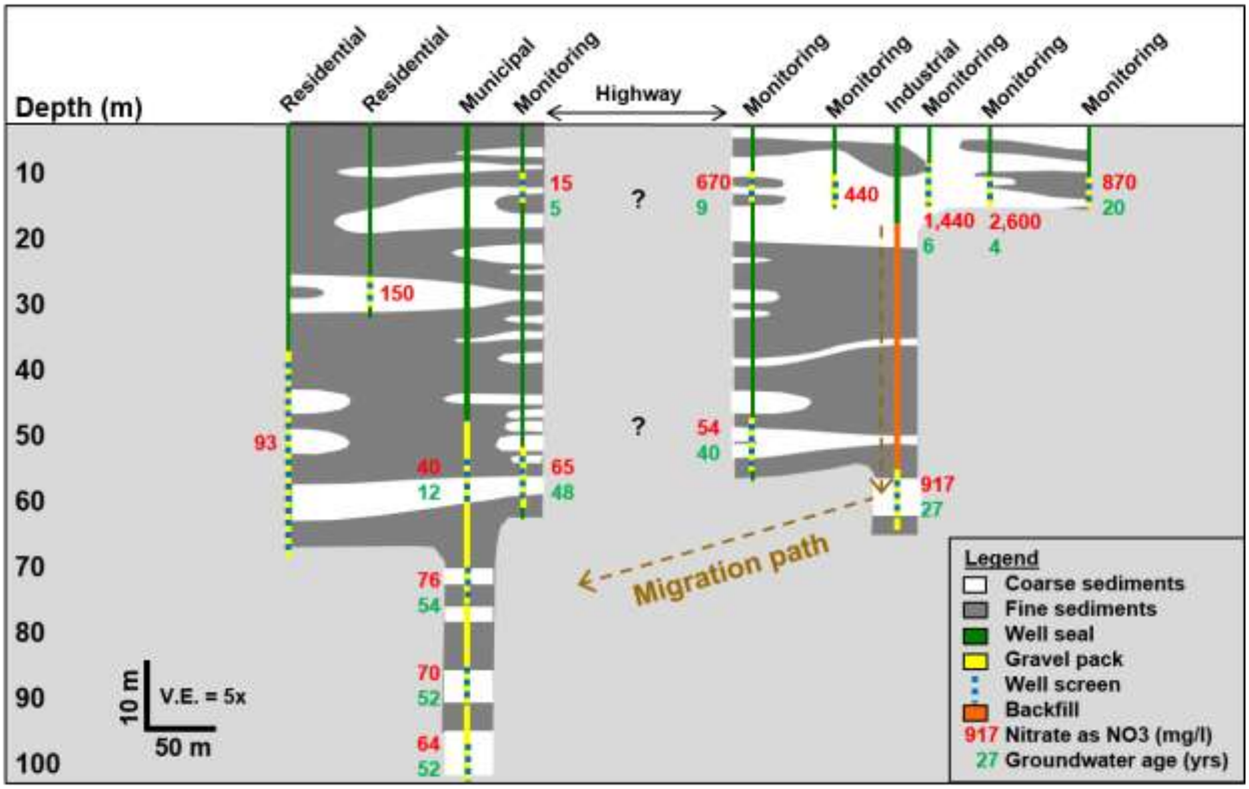
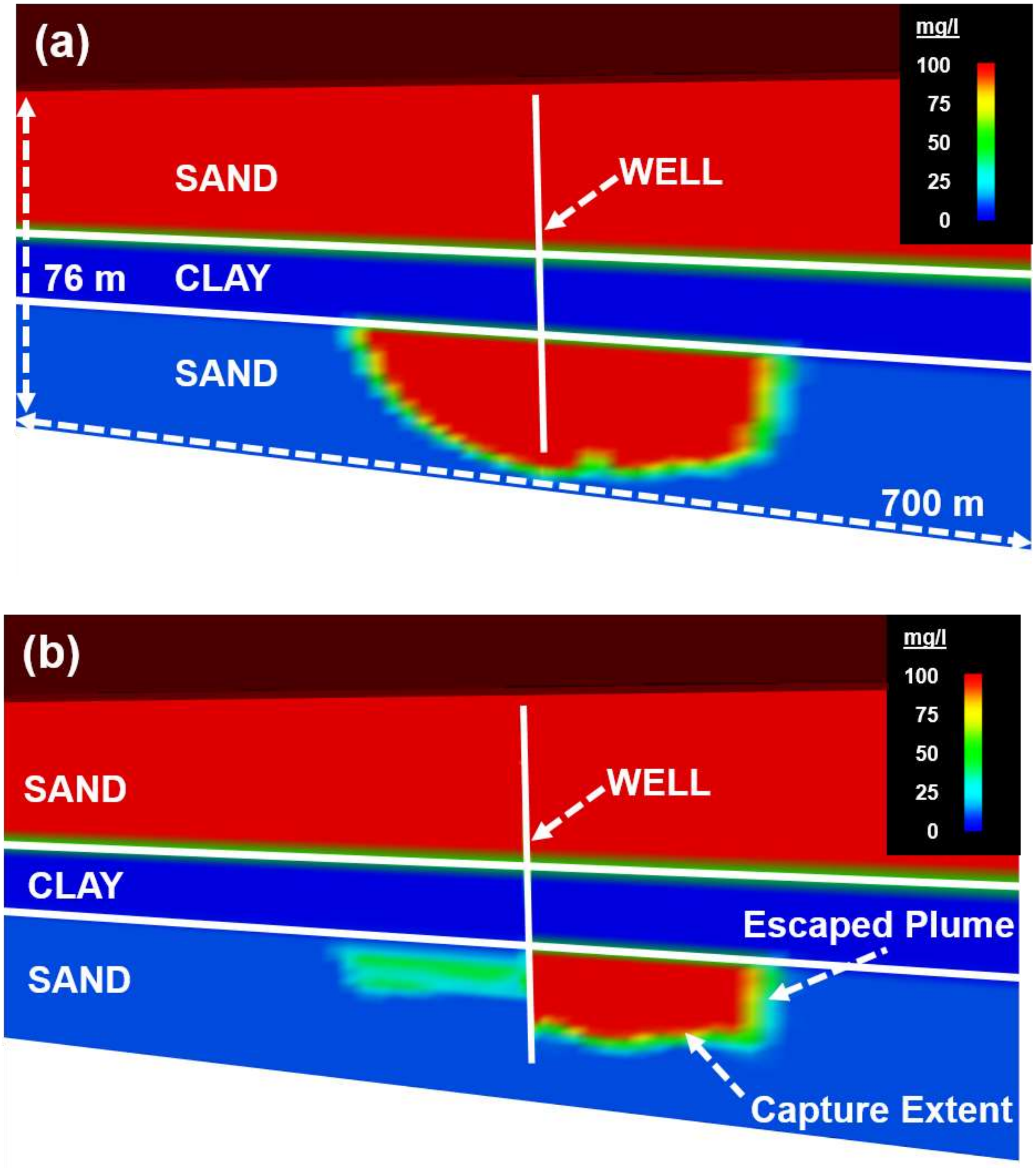


Figure 9: Field Example #3: Nitrate concentrations and groundwater age in the vicinity of an industrial supply well acting as a vertical conduit. Data from Singleton et al. (2010) and this study.

Results and Discussion of Management Analysis

Well-Specific Approaches

The volume of water transferred between strata through a well that acts as a conduit depends upon magnitude and duration of flow. A sustained flow of only one g/min (4 L/min) becomes 1,440 gal (5,450 L) in a day and over 260 x 10^3 gal (992 x 10^3 L) in a six-month period of inactivity (a representative duration of inactivity for wells used seasonally). Figures 10a through 10d depict simulated results of flow and contaminant migration down a typical well that is pumped only seasonally in a layered sand and clay system.


(a)
mg/l
100
75
50
25
0
SAND
WELL
76 m
CLAY
SAND
700 m
(b)
mg/l
100
75
50
25
0
SAND
WELL
CLAY
Escaped Plume
SAND
Capture Extent

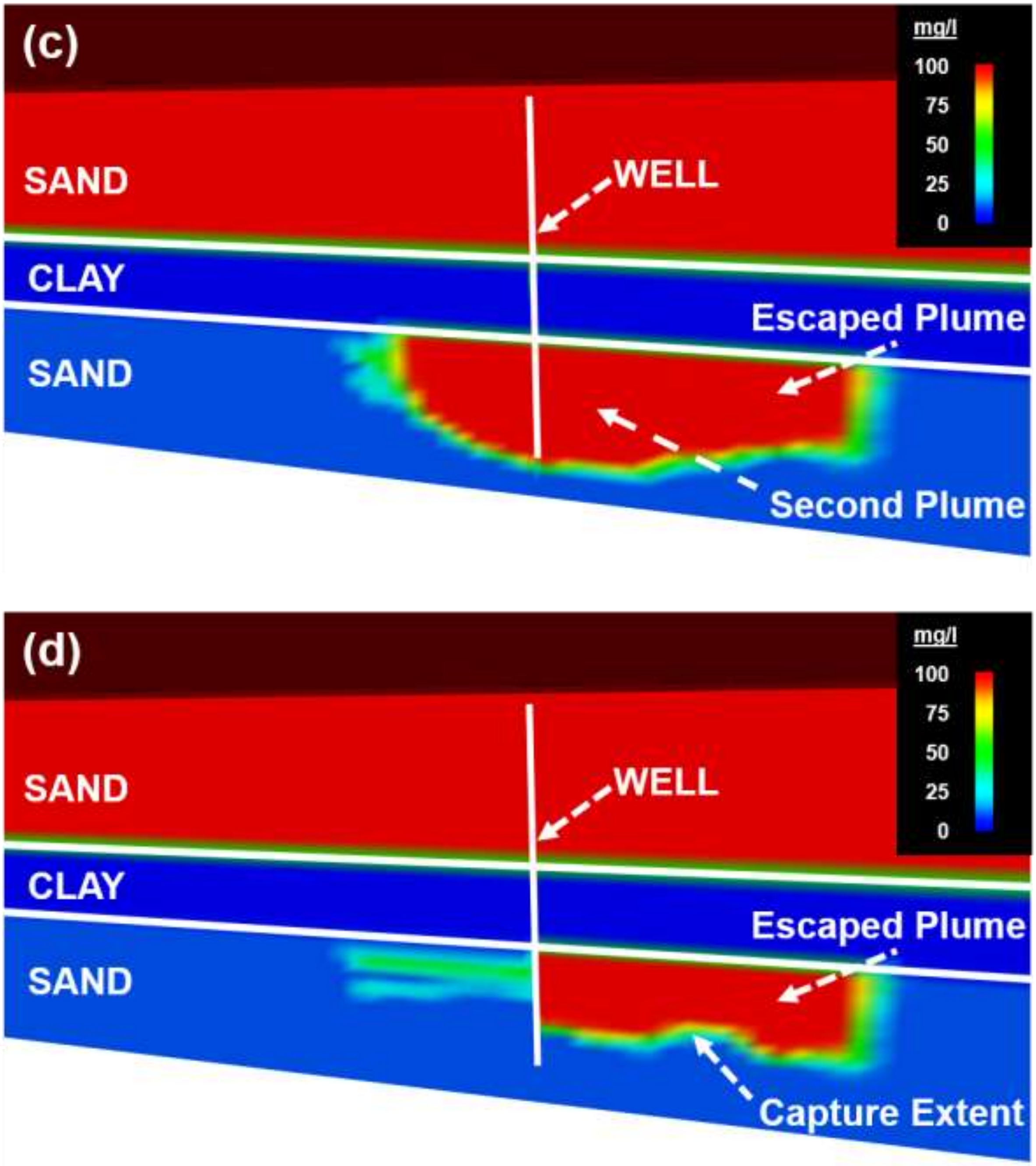
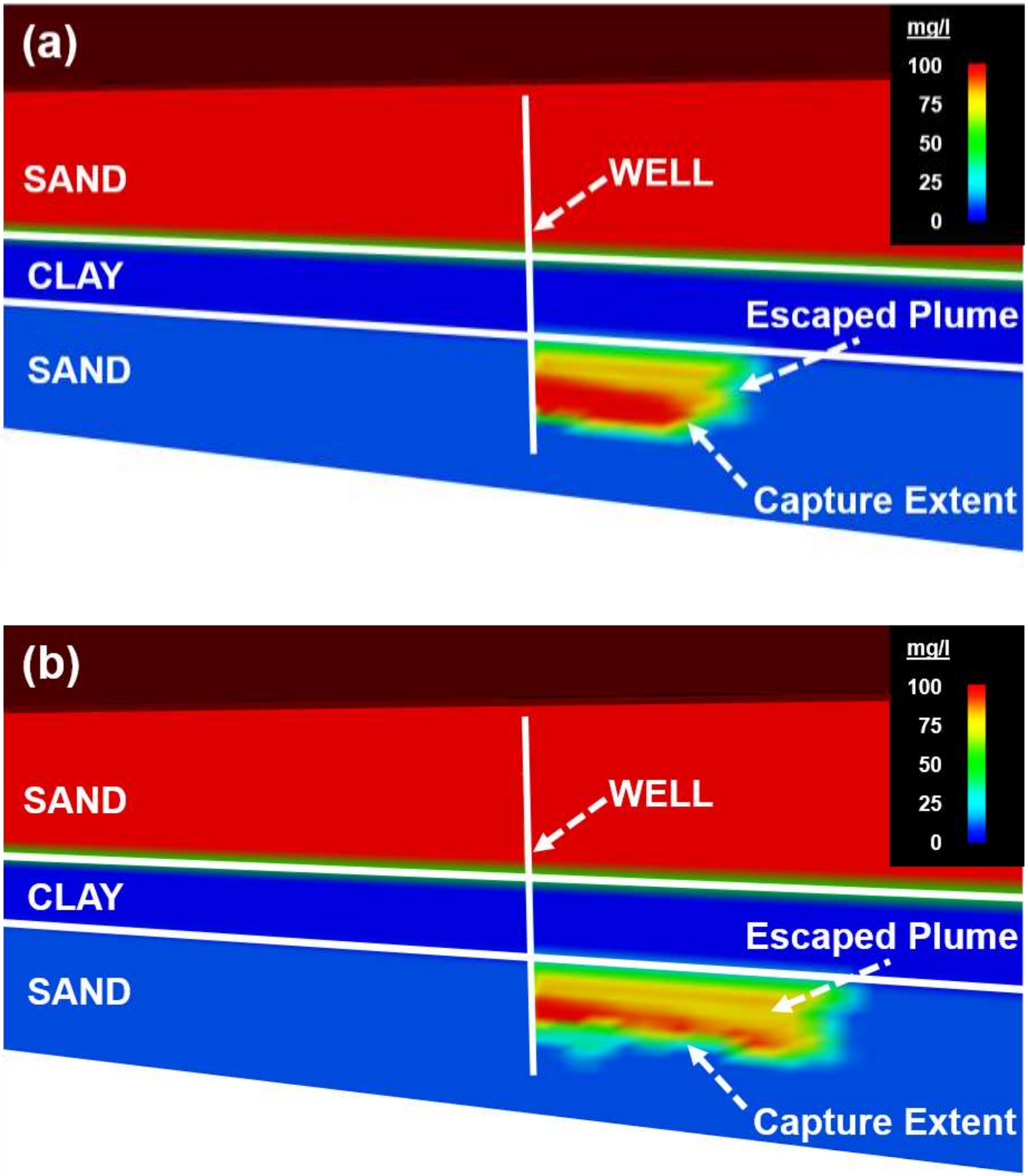


Figure 10: Simulated flow and transport in a conduit well: a) contaminant migration to the deeper zone while the well is idle for 180 days (days 1 through 180), b) pumping the well for 180 days (days 181 through 360), c) a second 180-day period when the well is idle (days 361 through 540) and d) a second 180-day period of pumping the well (days 541 through 720).

Over the twenty-four month simulation period, inactive and pumping periods of six months each occur in a series. Approximately 4.3 x 10^6 gal (16 x 10^6 L) of groundwater is redistributed from the shallow to deep aquifer during each inactive period and approximately 33 x 10^6 gal (124 x 10^6 L) is pumped from the groundwater system during each active period. Because the well pump is positioned above the shallower screen, head loss occurs in the well casing and there is a downward hydraulic gradient, only 29 percent of the volume extracted comes from the deep aquifer. Moreover, the vast majority of flow to the well in the deep aquifer is delivered from the up flow direction by regional flow. Only 6% of the volume extracted comes from the down flow direction in the deep aquifer. This limits the down flow capture zone extent and the regional flow system transports some of the water transferred during inactive periods away from the well. This uncaptured water transfers contaminant mass to the deep aquifer where it mixes with better quality water and degrades the resource.

To prevent such contamination, local regulatory requirements and industry standards for well construction should be followed when constructing new wells (i.e., CADWR, 1991 and 2003). Because regulatory criteria often provide only minimum requirements and are prepared in the absence of site-specific details, more stringent design criteria, such as shorter screened intervals and gravel packs, may be warranted in some cases. The hydrogeology for each new well site should be considered during design development. Key information to consider includes sediment stratigraphy and depositional environment, variations in vertical hydraulic gradient over the year, water quality near the well site and potential changes in conditions over the life of the well (well condition, aquifer stresses and water quality). In addition, best methods should be applied when abandoning test holes since a test hole also can act as a vertical conduit. This includes isolating vertically separate water-bearing strata using sealing materials that will remain stable.

Where significant water quality impacts occur from existing wells acting as vertical conduits, operational and/or structural changes should be considered. Operational changes could include implementing intermittent pumping during periods of low demand to control migration through a well. Comparison of Figures 11a and b with Figures 10b and d demonstrates the improvement in water quality from more frequent pumping, by a factor of two in this case, to limit contaminated water migration out of the well capture zone. This approach would only be possible when at least some water demand exists during the off-season and rotation of pumping among wells is possible. Structural changes could include changing the depth of the pump intake (moving the pump deeper in the case of water flowing downward through a well). Comparison of Figures 11c and d with Figures 10b and d conceptually demonstrates the improvement in water quality from moving the pump deeper in the well to reduce the amount of contaminant mass that escapes the well when pumping. Approximately 48 percent of the extracted volume comes from the deep aquifer, as opposed to 29 percent when the pump was higher in the well, and much less of the plume escapes. Other structural changes could include installing patches or packers to prevent contaminated water from entering and migrating through the well and injecting cement into the gravel pack within some intervals to prevent migration along the outside of the well. Finally, destroying wells should include application of methods that target the migration pathways of concern and collection of sufficient confirmatory information so that wells do not act as conduits for vertical contaminant migration after their useful lives have ended. For example, where a long gravel pack exists above the screen outside the well casing, it may be appropriate to perforate the casing to allow injection of sealing material.


(a)
mg/l
100
75
50
25
0
SAND
WELL
CLAY
Escaped Plume
SAND
Capture Extent
(b)
mg/l
100
75
50
25
0
SAND
WELL
CLAY
Escaped Plume
SAND
Capture Extent

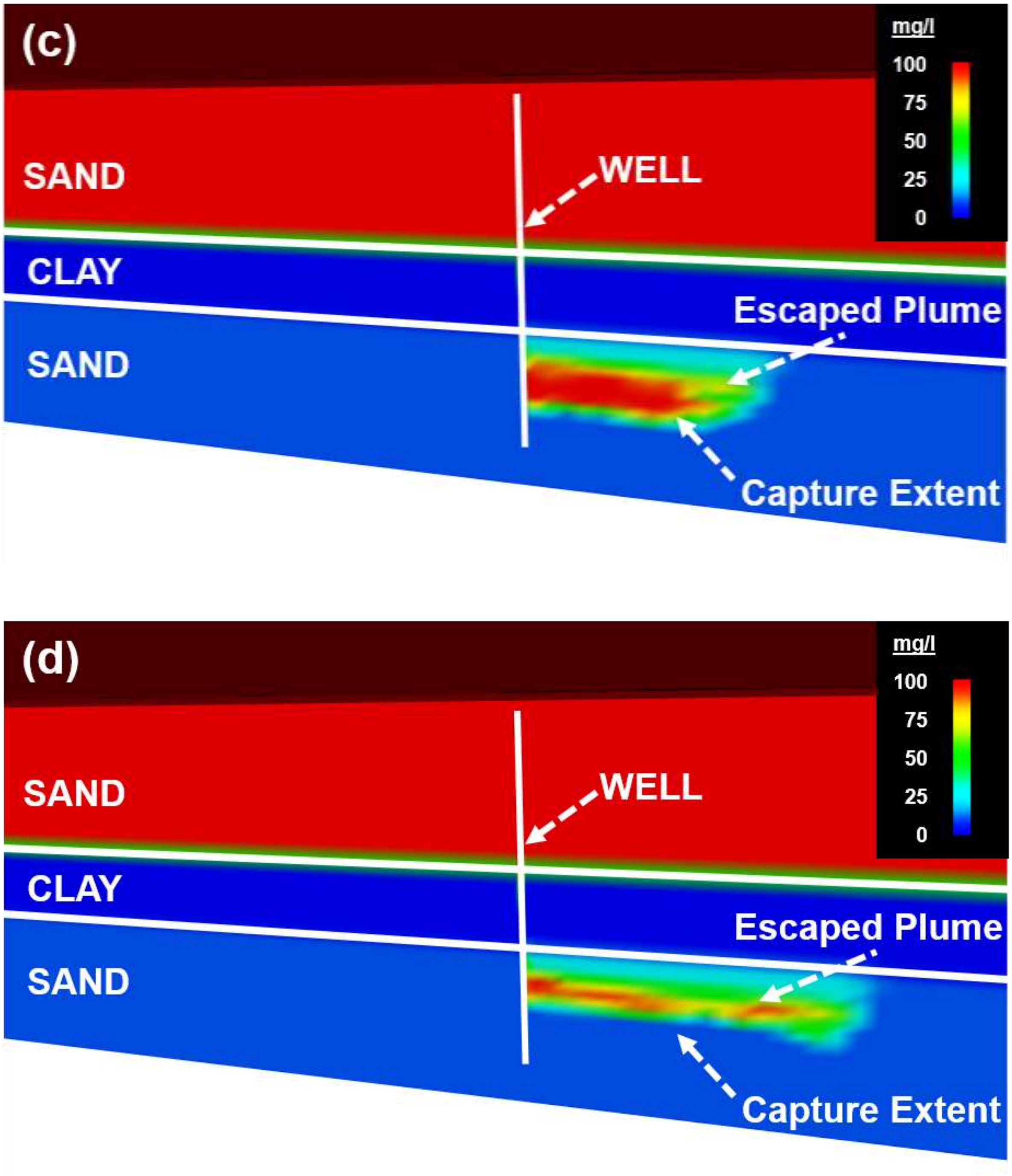


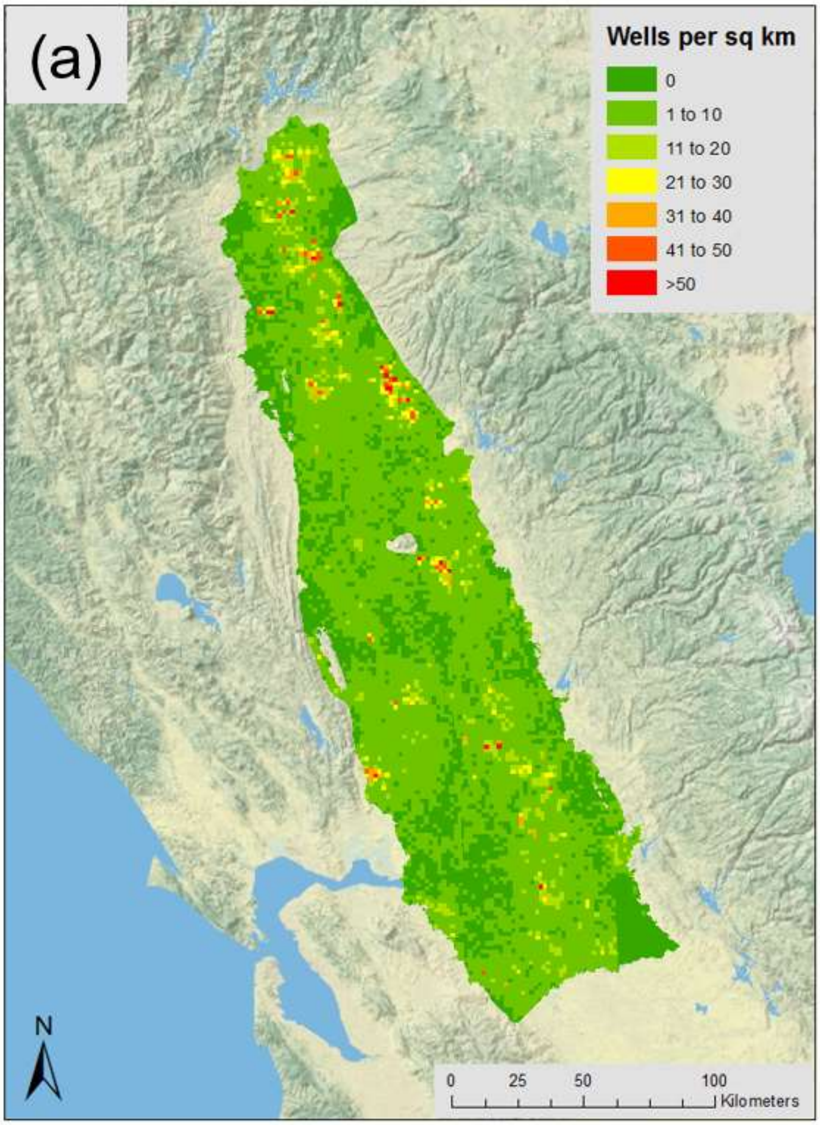
Figure 11: Simulated flow and transport in a modified well: a) resting and pumping the well every 90 days over a period of 360 days, b) resting and pumping the well every 90 days over a period of 720 days, c) moving pump deeper in well so that it is adjacent to the deeper water-bearing unit then resting and pumping the well every 180 days over a period of 360 days and d) moving pump

deeper in well and pumping the well every 180 days over a period of 720 days. The results for 360 and 720 days should be compared with Figures 10b and d, respectively. All other details of the simulations are as described in Section '*Well-specific approaches'*

Regional Approaches

Considering the great number of wells that exist in many groundwater basins and the length of time they remain open in the subsurface (from decades to perpetuity unless the well is properly destroyed), contaminant migration similar to that described here is likely to occur at many locations and on a variety of spatial scales (well, wellfield and region). In many cases, the risks are not fully understood because important information regarding well operations and local hydrogeology is absent. The challenge of protecting groundwater quality under such circumstances is great and efforts to focus limited resources for investigation and corrective measures are needed. An available data set for a portion of northern California provides the opportunity to outline a possible approach.

Figure 12a summarizes a dataset recently constructed by the CADWR for geographic occurrence of water supply wells in the northern portion of California's Central Valley. There are 75,974 total wells of which 56,781 are domestic; 16,354 are for irrigation; 1,913 are for public supply and 926 are for industrial use. These wells vary in age from one year (2%) to more than 60 years (13%), with a median age of 36 years, and likely exist in a wide range of maintenance conditions. Depths of completion vary from 40 ft (12 m, 1%) to over 500 ft (152 m, 5 %) with a median of 180 ft (55 m). The combination of hydrogeologic and well conditions (stratified alluvial sediments, pumping at depth, irrigation water applied at ground surface, poorer shallow groundwater quality in places and long well constructions) creates the potential for these wells to act as conduits for vertical migration of contaminants as described above. Figure 12b summarizes a worst-case calculation of the potential increase in effective vertical hydraulic conductivity for the groundwater system from water supply wells penetrating lower hydraulic conductivity strata. Details of the calculation are presented in the appendix.



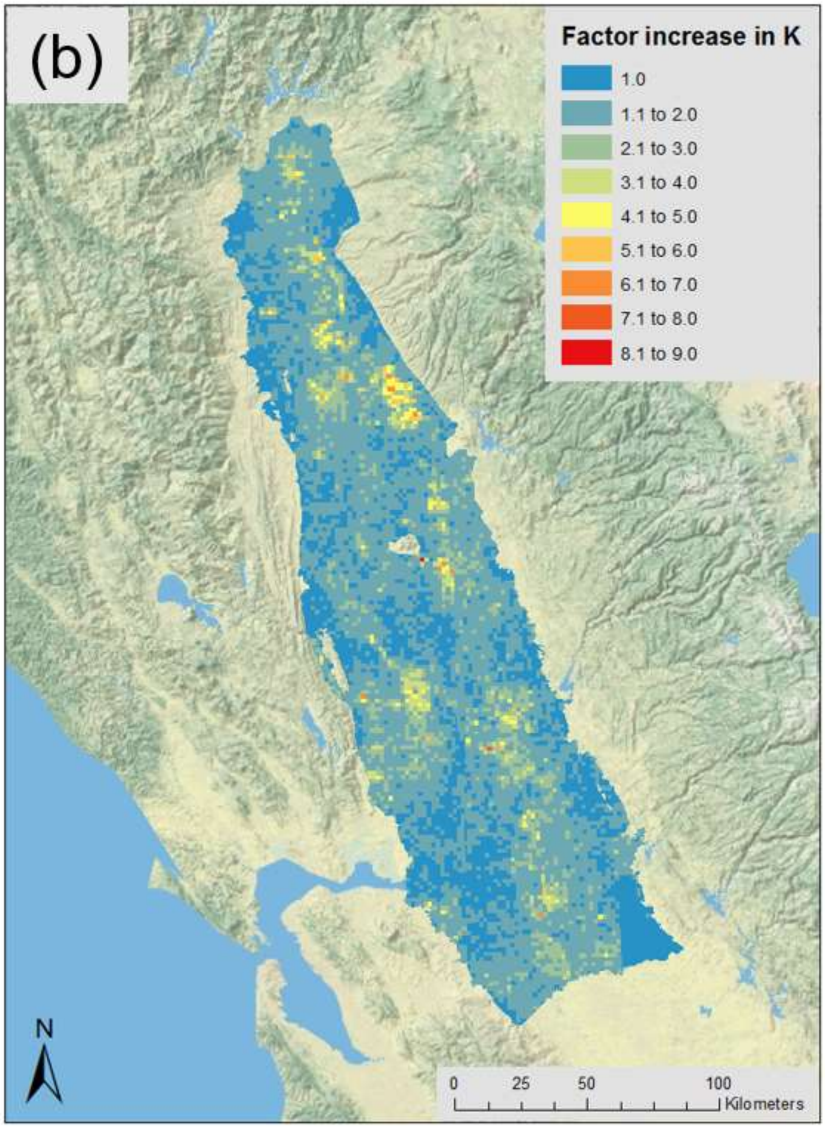


Figure 12: Example screening approach for assessing potential impact of wells that act as conduits: a) water well density in the northern Central Valley of California and b) worst-case assessment of increase in vertical hydraulic conductivity of groundwater system resulting from water wells acting as vertical conduits for flow. A factor increase of 2 indicates a doubling of the vertical hydraulic conductivity.

In areas of higher well density, the vertical hydraulic conductivity could increase by a factor of five to ten. Results similar to those presented on Figure 12b could be combined with information on shallow water quality so areas that pose a higher risk to groundwater quality (high density of wells and poor shallow water quality) could be prioritized for investigation. In the high-priority areas, the effects of vertical flow through wells might be evaluated through routine water quality monitoring upon renewed pumping after periods of inactivity. Further evaluation, such as video logging and ambient flow measurement, could be performed as circumstances warrant. Responses to discovered problems, possibly repairs to casings or changes in pumping operations as discussed above, could be implemented accordingly.

Conclusions

Table 3 summarizes the field examples presented above. While most cases discussed here are in or near agricultural settings, the observations presented on the potential for wells to act as vertical conduits and degrade groundwater quality are also pertinent to urban or urbanizing (converting from agricultural to urban use) areas. Whenever the combination of hydrologic stresses and stratigraphy in a groundwater basin creates vertical hydraulic gradients and there is also vertical heterogeneity in concentrations of undesirable water quality constituents, the potential exists for contaminant migration through idle wells that are improperly constructed or maintained.

Table 3 Summary of case examples for wells acting as vertical conduits

Field Example	**#1**	**#2**	**#3**	**#4**
Hydrostratigraphy	Alternating fine- and coarse-grained strata	Sand/aquitard	Alternating fine- and coarse-grained strata	Alternating fine- and coarse-grained strata
Hydrologic Stress	Regional discharge to river	Deep pumping Recharge operations	Deep pumping Irrigation	Deep pumping Irrigation
Vertical Gradient[a]	Upward	-0.3 to -0.8	Downward	0 to -0.04
Screen Length (m)	70 continuous	Upper: 8 Lower: 15	18 total across 137	6 continuous
Gravel Pack Length (m)	106	Upper: 11 Lower: 29	none	Confirmed: 15 Likely: 46
Casing Diameter (cm)	36	20	41	22
Flow Rate (L/min)	132	300 to 1,360	Unknown	Unknown
Volume Transferred (L)	Unknown	1.3×10^9	$\geq 246 \times 10^6$ Impacted	Unknown but regional
Timeframe	May reverse seasonally	47 months	Unknown	Seasonal over decades
Contaminant Present?	Yes	No	Yes	Yes

Negative *vertical gradient* indicates downward flow.

Wells that act as conduits may create water quality impacts for themselves as well as for the surrounding water-bearing zones and nearby wells. Therefore, special consideration should be given to the construction, maintenance and management of water supply wells when the conditions for intraborehole flow exist or are reasonably expected to exist. The information necessary for designing wells against vertical flow and water quality impacts has been available since at least the early 1960’s (Johnson 1961; CADWR, 1962); nevertheless, examples of the phenomenon appear to be fairly common. Continued and additional efforts are needed to protect groundwater quality from improperly constructed and maintained wells.

Acknowledgements

The author is grateful to the Orange County Water District for sharing data and results presented in Field Example #2 and to the California Department of Water Resources (Data Collection and Management Section, Northern Region Office) for sharing data regarding the geographic distribution of water supply well constructions in the northern Central Valley of California upon which the results in figures 12a and 12b are based. Graham Fogg, Thomas Harter and Jay Lund are also appreciatively acknowledged for discussions and editorial comments that significantly improved this manuscript. Christoph Weidner also provided valuable comments on the final manuscript. This work was supported by UC Water Security and Sustainability Research Initiative funded by the University of California Office of the President (UCOP; Grant No. 13941) and by the S.D. Bechtel Jr. Foundation through the UC Davis Center for Watershed Sciences.

Appendix

A screening approach was used to estimate the potential increase in effective vertical hydraulic conductivity discussed in Section '*Regional approaches'*. It is based upon summation of vertical flows in the region of consideration:

$$Q_{\text{regional}} = Q_{\text{strata}} + \sum_{j=1}^{n} (Q_{\text{well}})_j \tag{1}$$

Where $Q_{regional}$ is the total vertical flow through the groundwater system in the region,

Q_{strata} is the vertical flow through strata in the region,

Q_{well} is the vertical flow through a well acting as a conduit, and

n is the number of wells acting as conduits in the region.

Substitution of Darcy's Law into Equation 1 yields:

$$K_{effective}\, i_{regional}\, A_{regional} = K_{strata}\, i_{regional}\, \left(A_{regional} - \sum_{j=1}^{n} (A_{well})_j\right) + \sum_{j=1}^{n} (K_{well}\, i_{well}\, A_{well})_j \qquad (2)$$

Where $K_{effective}$ is the effective vertical hydraulic conductivity for the groundwater system in the region,

$i_{regional}$ is the average vertical hydraulic gradient for the region,

$A_{regional}$ is the total area of the region,

K_{strata} is the average vertical hydraulic conductivity of the strata in the region,

A_{well} is the cross sectional area of an individual well acting as a conduit,

K_{well} is the effective vertical hydraulic conductivity of a well acting as a conduit, and

i_{well} is the vertical hydraulic gradient for a well acting as a conduit.

The value of i_{well} is expected to be less than that of $i_{regional}$ since there will be groundwater level drawdown where flow enters the well and mounding where flow exits the well.

Rearranging Equation 2 provides an expression for the effective vertical hydraulic conductivity for the groundwater system:

$$K_{effective} = [K_{strata}\ i_{regional}\ (A_{regional} - \sum_{j=1}^{n} (A_{well})_j) + \sum_{j=1}^{n} (K_{well}\ i_{well}\ A_{well})_j\ ]\ /\ (i_{regional}\ A_{regional}) \qquad (3)$$

K_{well} in Equation 3 is calculated using the Hagen-Poiseuille equation:

$$K_{well} = r^2 \rho g / 8\mu \qquad (4)$$

Where r is the inner radius of the well casing or screen

ρ is the groundwater density

g is the gravitational acceleration, and

μ is the groundwater viscosity.

The values used for variables in the in the calculations were as follows:

$i_{regional}$ 0.4 as an example based upon author's experience,

A_{total} 1.0 square miles (2.6 km^2) based upon data presented in Figure 12a,

$K_{regional}$ 10^{-5} cm/s as an example based upon author's experience,

A_{well} varied based upon the database related to Figure 12a,

K_{well} 1.1x10^{6} cm/s for a 20 cm diameter well and 2.5x10^{6} cm/s for a 36 cm diameter well,

i_{well} 7x10^{-6} as an example based upon author's experience (inferred from flows measured in wells), and

n varied between 0 and 232 based upon the data presented in Figure 12a.

Importantly, the gradient in the regional strata ($i_{regional}$) results from the hydraulic stresses (pumping and recharge) and the resistance to flow caused by the layered sediments. The value used is consistent with observations in the field by the author. The gradient in the well (i_{well}) is much smaller and departs from that in the sediments as a result of head losses across the drawdown cone near the shallow screen, the mound up cone near the deep screen and the well screens themselves. These considerations reduce the hydraulic gradient that drives flow through the well.

A more detailed approach, possibly similar to Silliman and Higgins (1990), might be applied if the necessary additional information is available and the hydrogeology can be represented as a simple layered system. However, data are often lacking and the goal of performing the calculations presented above is to identify areas where additional investigation and information collection would occur.

References

Belitz K, Phillips SP and Gronberg JM (1993) Numerical simulation of ground-water flow in the central part of the western San Joaquin Valley, California. US Geol Surv Water-Supply Pap 2396

Bertoldi GL, Johnston RH and Evenson KD (1991) Ground water in the Central Valley, California – a summary report. US Geol Surv Prof Pap 1401-A

Bexfield LM, Jurgens BC, Crilley DM and Christenson SC (2012) Hydrogeology, water chemistry, and transport processes in the zone of contribution of a public-supply well in Albuquerque, New Mexico. US Geol Surv Sci Invest Rep 2011-5182

Bexfield LM and Jurgens BC (2014) Effects of seasonal operation on the quality of water produced by public supply wells, Ground Water 52(S1):10-24, doi: 10.1111/gwat.12174

Burow KR, Dubrovsky NM and Shelton JL (2007) Temporal trends in concentrations of DBCP and nitrate in groundwater in the eastern San Joaquin Valley, California, USA, Hydrogeology J 15:991-1007, doi: 10.1007/s10040-006-0148-7

CADWR (1962) Bulletin 74, Recommended Minimum Water Well Construction and Sealing Standards for the Protection of Ground Water Quality, State of California, preliminary edn. July 1962

CADWR (1981) Bulletin 74-81, Water Well Standards: State of California. December 1981. http://www.water.ca.gov/pubs/groundwater/water_well_standards__bulletin_74-81_/ca_well_standards_bulletin74-81_1981.pdf. Cited 6 July 2016

CADWR (1991) Bulletin 74-90, Supplement to Bulleting 74-81, California Well Standards. June 1991. http://www.water.ca.gov/pubs/groundwater/water_well_standards__bulletin_74-90_/ca_well_standards_bulletin74-90_1991.pdf. Cited 6 July 2016

CADWR (2003) California Laws for Water Wells, Monitoring Wells, Cathodic Protection Wells, Geothermal Heat Exchange Wells. March 2003. http://www.water.ca.gov/pubs/groundwater/california_laws_for_water_wells__monitoring_wells__cathodic_protection_wells__geothermal_heat_exchange_wells_2003/ca_water_laws_2003.pdf. Cited 6 July 2016

CADWR (2014) Bulletin 160-14, California Water Plan Update 2013. October 2014. http://www.waterplan.water.ca.gov/cwpu2013/. Cited 6 July 2016

Church PE and Granato GE (1996) Bias in ground-water data caused by well-bore flow in long-screen wells. Ground Water 34(2):262-273, doi: 10.1111/j.1745-6584.1996.tb01886.x

Clark BR, Landon MK, Kauffman LJ and Hornberger GZ (2008) Simulations of Ground-Water Flow, Transport, Age, and Particle Tracking near York, Nebraska, for a Study of Transport of Anthropogenic and Natural Contaminants (TANC) to Public Supply Wells, US Geol Surv Sci Invest Rep 2007-5068

Corcho Alvarado JA, Barbecot F and Purtschert R (2009) Ambient vertical flow in long-screen wells: a case study in the Fontainebleau Sands Aquifer (France), Hydrogeology J 17:425-431, doi: 10.1007/s10040-008-0383

Davis GH, Lofgren BE and Mack S (1964) Use of Ground-Water Reservoirs for Storage of Surface Water in the San Joaquin Valley California, US Geol Surv Water-Supply Pap 1618

Dragon K, Gorski J, Marciniak M and Kasztelan D (2009) Use of mathematical modeling to investigate inter-aquifer contamination by organic-rich water through an unplugged well (central Wielkopolska, Poland), Hydrogeology J 17:1257-1564, doi: 10.1007/s10040-009-0432-4

Elci A, Molz FJ and Waldrop WR (2001) Implications of observed and simulated ambient flow in monitoring wells, Ground Water 39(6):853-862, doi: 10.1111/j.1745-6584.2001.tb02473.x

Faunt CC (ed.) (2009) Groundwater availability of the Central Valley Aquifer, California. US Geol Surv Prof Pap 1766

Fetter CW (1994) Applied Hydrogeology. Prentice Hall

Fogg GE (1986) Groundwater flow and sand body interconnectedness in a thick multiple-aquifer system, Water Resour Res 22(5):1679-694, doi: 10.1029/WR022i005p00679

Freeze RA and Cherry JA (1979) Groundwater. Prentice Hall

Gass TE, Lehr JH and Heiss HW (1977) Impact of abandoned wells on ground water, US Environmental Protection Agency Rep EPA/600/3-77-095

Halford KJ and Hanson RT (2002) Users guide for the drawdown-limited, multi-node well (MNW) package for the U.S. Geological Survey's modular three-dimensional ground-water flow model, versions MODFLOW-96 and MODFLOW-2000. US Geol Surv Open-File Rep 02-293

Halford K (2009) AnalyzeHOLE – An Integrated Wellbore Flow Analysis Tool, US Geol Surv Survey Techniques and Methods 4-F2

Halford K, Stamos C, Nishikawa T and Martin P (2010) Arsenic management through well modification and simulation, Ground Water 48(4):526-537, doi: 10.1111/j.1745-6584.2009.00670.x

Hanson RT, Zhen L and Faunt CC (2004) Documentation of the Santa Clara Valley regional ground-water/surface-water flow model, Santa Clara County, California, US Geol Surv Sci Invest Rep 2004-5231

Harbaugh AW, Banta ER, Hill MC and McDonald MG (2000) MODFLOW-2000, The US Geological Survey Modular Ground-Water Model – User Guide to Modularization Concepts and the Ground-Water Flow Process, US Geol Surv Open File Rep 00-92

Hart DJ, Bradbury KR and Feinstein DT (2006) The vertical hydraulic conductivity of an aquitard at two spatial scales, Ground Water 44(2):201-211, doi: 10.1111/j.1745-6584.2005.00125.x

Horn JE and Harter T (2009) Domestic well capture zone and influence of the gravel pack length, Ground Water 47(2):277-286, doi: 10.1111/j.1745-6584.2008.00521.x

Houben GJ and Hauschild S (2011) Numerical modeling of the near-field hydraulics of water wells, Ground Water 49(4):570-575, doi: 10.1111/j.1745-6584.2010.00760.x

Houben GJ and Treskatis C (2007) Water Well Rehabilitation and Reconstruction. McGraw Hill

Izbicki JA, Christensen AH, Hanson RT, Martin P, Crawford SM and Smith GA (1999) US Geological Survey Combined Well-Bore Flow and Depth-Dependent Water Sampler. US Geol Surv Fact Sheet 196-99

Izbicki JA, Christensen AH, Newhouse MW, Smith GA and Hanson RT (2005a) Temporal changes in the vertical distribution of flow and chloride in deep wells, Ground Water 43(4):531-544, doi: 10.1111/j.1745-6584.2005.0032.x

Izbicki JA, Christensen AH, Newhouse MW and Aiken GR (2005b) Inorganic, isotopic, and organic composition of high-chloride water from wells in a coastal southern California aquifer, Appl Geochemistry 20:1496-1517, doi: 10.1016/j.apgeochem.2005.04.010

Izbicki JA, Metzger LF, McPherson KR, Everett RR and Bennett GL (2006) Sources of High-Chloride Water to Wells, Eastern San Joaquin Ground-Water Subbasin, California, US Geol Surv Open File Rep 2006-1309

Izbicki JA, Stamos CL, Metzger LF, Halford KJ, Kulp TR and Bennett GL (2008) Source, Distribution, and Management of Arsenic in Water from Wells, Eastern San Joaquin Ground-Water Subbasin, California, US Geol Surv Open File Rep 2008-1272

Izbicki JA, Wright MT, Seymour WA, McCleskey RB, Fram MS, Belitz K and Esser BK (2015) Cr(VI) occurrence and geochemistry in water from public-supply wells in California, Appl Geochemistry 63:203-217, doi: 10.1016/j.apgeochem.2015.08.007

Jiménez-Martínez J, Arvenda R and Candela L (2011) The role of leaky boreholes in the contamination of a regional confined aquifer. A case study: the Campo de Cartagena Region, Spain, Water Air Soil Pollut 215:311-327, doi: 10.1007/s11270-010-0480-3

Johnson (1961) Dual-aquifer wells pose problems, Johnson National Drillers Journal, January-February

Johnson RL, Clark BR, Landon MK, Kauffman JL and Eberts SM (2011) Modeling the potential impact of seasonal and inactive multi-aquifer wells on contaminant movement to public water-supply wells, J Am Water Resour Ass 47(3):588-596, doi: 10.1111/j.1752-1688.2011.00526.x

Jurgens BC, Burow KR, Dalgish BA and Shelton JL (2008) Hydrogeology, water chemistry, and factors affecting the transport of contaminants in the zone of contribution of a public-supply well in Modesto, eastern San Joaquin Valley, California. US Geol Surv Sci Invest Rep 2008-5156

Jurgens BC, Fram MS, Belitz K, Burow KR and Landon MK (2010) Effects of groundwater development on uranium: Central Valley, California, USA, Ground Water 48(6):913-928, doi: 10.1111/j.1745-6584.2009.00635.x

Jurgens BC, Bexfield LM and Eberts SM (2014) A ternary age-mixing model to explain contaminant occurrence in a deep supply well, Ground Water 52(S1):25–39, doi: 10.1111/gwat.12170

Keys WS (1997) A Practical Guide to Borehole Geophysics in Environmental Investigations. CRC Lewis Publishers

Konikow LF and Hornberger GZ (2006) Modeling effects of multinode wells on solute transport, Ground Water 44(5):648-660, doi: 10.1111/j.1745-6584.2006.00231.x

Konikow LF, Hornberger GZ, Halford KJ and Hanson RT (2009) Revised multi-node well (MNW2) package for MODFLOW ground-water flow model. US Geol Surv Techniques and Methods 6-A30

Lacombe S, Sudicky EA, Frape SK and Unger AJA (1995) Influence of leaky boreholes on cross-formational groundwater flow and contaminant transport, Water Resour Res 31(8):1871-1882, doi: 10.1029/95WR00661

Landon MK, Clark BR, McMahon PB, McGuire VL and Turco MJ (2008) Hydrogeology, Chemical Characteristics, and Transport Processes in the Zone of Contribution of a Public-Supply Well in York, Nebraska. US Geol Surv Sci Invest Rep 2008-5050

Landon MK, Jurgens BC, Katz BG, Eberts SM, Burow KR and Crandall CA (2009) Depth-dependent sampling to identify short-circuit pathways to public supply wells in multiple aquifer settings in the United States, Hydrogeology J 18(3):577-593, doi: 10.1007/s10040-006-0148-7

Lu H, Liu T, Chen W, Peng T, Wang C, Tsai M and Liou T (2008) Use of geochemical modeling to evaluate the hydraulic connection of aquifers: a case study from Chianan Plain, Taiwan, Hydrogeology J 16:139-154, doi: 10.1007/s10040-007-0209-6

Ma R, Zheng C, Tonkin M and Zachara JM (2011) Importance of considering intraborehole flow in solute transport modeling under highly dynamic conditions, J Contaminant Hydrology 123:11-19, doi:10.1016/j.jconhyd.2010.12.001

Mansuy N (1999) Water Well Rehabilitation, A Practical Guide to Understanding Well Problems and Solutions. Lewis Publishers

Mayo AL (2010) Ambient well-bore mixing, aquifer cross-contamination, pumping stress and water quality from long-screened wells: What is samples and what is not?, Hydrogeology J 18:823-837, doi: 10.1007/s10040-009-0568-2

McCombs J and Fiedler AG (1927) Methods of exploring and repairing leaky artesian wells, US Geol Surv Water-Supply Pap 596-A

McMahon PB, Böhlke JK, Kauffman LJ, Kipp KL, Landon MK, Crandall CA, Burrow KR and Brown CJ (2008) Source and transport controls on the movement of nitrate to public supply wells in selected principal aquifers of the United States, Water Resour Res 44(4) W04401, doi: 10.1029/2007WR006252

McMillan LA, Rivett MO, Tellam JH, Dumble P and Sharp H (2014) Influence of vertical flows in wells on groundwater sampling, Journal of Contaminant Hydrology 169:51-60, doi:10.1016/j.jconhyd.2014.05.005

Mejia A, Cassiraga E and Sahuquillo A (2012) Influence of hydraulic conductivity and wellbore design in the fate and transport of nitrate in multi-aquifer systems, Math Geosci 44:227-238, doi: 10.1007/s11004-012-9388-3

Phillips SP and Belitz K (1991) Calibration of a texture-based model of a ground-water flow system, western San Joaquin Valley, California, Ground Water 29(5):702-715, doi: 10.1111/j.1745-6584.1991.tb00562.x

Phillips SP, Green CT, Burow KR, Shelton JL and Rewis DL (2007) Simulation of Multiscale Ground-Water Flow in Part of Northeastern San Joaquin Valley, California. US Geol Surv Sci Invest Rep 2007-5009

Reilly TE, Franke OL and Bennett GD (1989) Bias in groundwater samples caused by wellbore flow, J Hydraulic Eng 115(2):270-276

Santi PM, McCray JE and Martens JL (2006) Investigating cross-contamination of aquifers, Hydrogeology J 14:51-68, doi: 10.1007/s10040-00400403-8

Silliman S and Higgins D (1990) An analytical solution for steady-state flow between aquifers through an open well, Ground Water 28(2):184-190, doi: 10.1111/j.1745-6584.1990.tb02245.x

Singleton MJ, Moran JE, Roberts SK, Hillegonds DJ and Esser BK (2010) California GAMA Special Study: An Isotopic and Dissolved Gas Investigation of Nitrate Source and Transport to a Public Supply Well in California's Central Valley, Final Report for the California State Water Resources Control Board, Rep LLNL-TR-427957, Lawrence Livermore National Laboratory

Smith SA and Comeskey AE (2010) Sustainable Wells: Maintenance, Problem Prevention, and Rehabilitation. CRC Press

Sokol D (1963) Position and fluctuations of water level in wells perforated in more than one aquifer, J Geophys Res 68(4):1079-1080

Van Beek CGEM (2012) Cause and Prevention of Clogging of Wells Abstracting Groundwater from Unconsolidated Aquifers. IWA Publishing

Vennard JK and Street RL (1982) Elementary Fluid Mechanics, 6th edn. John Wiley and Sons

Viers JH, Liptzin D, Rosenstock TS, Jensen VB, Hollander AD, McNally A, King AM, Kourakos G, Lopez EM, De La Mora N, Fryjoff-Hyng A, Dzurella KN, Canada H, Laybourne S, McKenney C, Darby J, Quinn JF and Harter T (2012) Nitrogen sources and loading to groundwater, Technical report 2 in: Addressing nitrate in California's drinking water with a focus on Tulare Lake Basin and Salinas Valley groundwater, Report for the State Water Resources Control Board report to the legislature, Center for Watershed Sciences, University of California, Davis

Vitale SA and Robbins GA (2016) Characterizing groundwater flow in monitoring wells by altering dissolved oxygen, Groundwater Monitoring and Remediation 36(2):59-67, doi:10.1111/gwmr.12157

Wada Y, van Beek LPH and Bierkens MFP (2012) Nonsustainable groundwater sustaining irrigation: a global assessment, Water Resour Res 48(6) W00L06, doi: 10.1029/2011WR010562

Welenco (1996) Water and Environmental Geophysical Well Logs, Volume 1: Technical Information and Data, 8th edn. Welenco Borehole Geophysical Services

Williamson AK, Purdic DE and Swain LA (1989) Ground-Water Flow in the Central Valley of California. US Geol Surv Prof Pap 1401-D

Yager RM and Heywood CE (2014) Simulation of the effects of seasonally varying pumping on intraborehole flow and the vulnerability of public-supply wells to contamination, Ground Water 52(S1):40-52, doi: 10.1111/gwat.12150

Zheng C and Wang PP (1999) MT3DMS: A Modular Three-Dimensional Multispecies Transport Model for Simulation of Advection, Dispersion, and Chemical Reactions of Contaminants in Groundwater Systems: Documentation and User’s Guide, US Army Corps of Eng Rep SERDP-99-1

Zinn BA and Konikow LF (2007) Effects of intraborehole flow on groundwater age distribution, Hydrogeology J 15(4):633-643, doi: 10.1007/s10040-006-0139-8

CHAPTER 3 – USING GEOGRAPHIC DISTRIBUTION OF WELL SCREEN DEPTHS AND HYDROGEOLOGIC CONDITIONS TO IDENTIFY AREAS OF CONCERN FOR CONTAMINANT MIGRATION THROUGH INACTIVE SUPPLY WELLS

Abstract

Contaminant migration through inactive supply wells can negatively affect groundwater quality and the combined effects from groups of such wells may cause greater impacts. Because the number of wells in many basins is often large and the geographic areas involved can be vast, approaches are needed to estimate potential impacts and focus limited resources for investigation and corrective measures on the most important areas. One possibility is to evaluate the geographic distribution of well-screen depths relative to hydrogeologic conditions and assess where contaminant migration through wells may be impacting groundwater quality. This approach is demonstrated for a geographically extensive area in the southern Central Valley of California, USA. The conditions that lead to wells acting as conduits for contaminant migration are evaluated and areas where the problem likely occurs are identified. Although only a small fraction of all wells appear to act as conduits, potential impacts may be significant considering needs to control nonpoint source pollution and improve drinking water quality for rural residents. Addressing a limited number of areas where contaminant migration rates are expected to be high may cost-effectively accomplish the most beneficial groundwater quality protection and improvement. While this work focuses on a specific region, the results indicate that impacts from groups of wells may occur in other areas with similar conditions. Analyses similar to that demonstrated here may guide efficient investigation and corrective action in such areas with benefits occurring for groundwater quality. Potential benefits may justify expenditures to develop the necessary data for performing the analyses.

Introduction

Contaminant migration through inactive supply wells can negatively affect groundwater quality (Landon et al. 2009; Mayo 2010; Jurgens et al. 2014; Zuurbier and Stuyfzand 2017; see Gailey 2017 for a detailed literature review). With the exception of Clark et al. (2008), studies on wells that act as conduits for contaminant migration address single wells; however, the combined effects of groups of wells may cause greater impacts (Gailey 2017). Because the number of wells is often large and the geographic areas can be vast, approaches are needed to evaluate potential impacts and focus limited resources on investigations and corrective measures where most beneficial.

Recent and developing advances in data accessibility may allow survey-level screening to guide work on groundwater management including addressing conduit wells. Early examples include 1) Perrone and Jasechko (2017) who compiled well depth information from 17 databases in western USA and evaluated the incidence of wells going dry during drought and 2) Jasechko et al. (2017) who used a combination of well depth data and water quality information from across the globe to assess water supply vulnerability to contamination. Ongoing efforts to increase accessibility for data pertinent to groundwater management in California, USA are considered by Cantor et al. (2018). The work presented here considers the geographic distribution of conditions that result in wells acting as conduits and evaluates potential magnitudes of contaminant migration in particular areas. To the best of the author's knowledge, such work has not been performed before. Application to the southern Central Valley of California (Figure 13) is presented as a demonstration example.

Figure 13: Study area in California, USA. *Gray shaded area* indicates approximate extent of Corcoran Clay (study area)

Study Area

California's southern Central Valley supports intensive agricultural activity with heavy reliance on groundwater for irrigation (Hanak et al. 2017). The shallower freshwater part of the groundwater system comprises an interfingered assemblage of alluvial and flood-basin deposits that is approximately 1,000 m thick and exists under semiconfined conditions (Faunt 2009). Groundwater overdraft in the region (CADWR 2016; Hanak et al. 2017) combined with pumping at depth and resistance to vertical flow by numerous silt and clay aquitards including the regionally extensive Corcoran Clay (CADWR 1981; Page 1986; Faunt 2009; Figure 13) has resulted in pronounced and geographically extensive decreases in head with depth for the past many decades (Davis et al. 1964; Belitz and Heimes 1990; Faunt 2009). Shallow groundwater quality is generally poor as a result of agricultural activities while deeper water quality is often better (Faunt 2009; Landon et al. 2009; Moore et al. 2011). Deeper supply wells crossing low-hydraulic conductivity layers including the Corcoran Clay may act as conduits for vertical migration of contaminants under these conditions (Davis et al. 1964; Faunt 2009). Given the many wells, variability in conditions with location, and the large extent of the study area (approximately 6,600 mi^2 or 17,100 km^2), assessment of potential for vertical contaminant migration would provide useful information to evaluate the need for investigation and corrective action in particular parts of the study area.

Data Sources and Methods of Analysis

As discussed by Gailey (2017), wells act as conduits for contaminant migration under the following conditions (Figure 14): 1) differences in head and water quality exist between water-bearing strata over a vertical section, 2) stratification of high- and low-hydraulic conductivity sediments impede vertical flow and solute transport and 3) the screened intervals of wells vertically span the stratigraphy allowing relatively rapid vertical flow between water-bearing strata to occur through the wells (short-circuiting flow). This work focuses on the Corcoran Clay as a point of reference for defining such vertical flows in the study area; however, it is noted that short-circuiting of other low-hydraulic conductivity strata and subsequent contaminant migration may also occur above the Corcoran Clay.

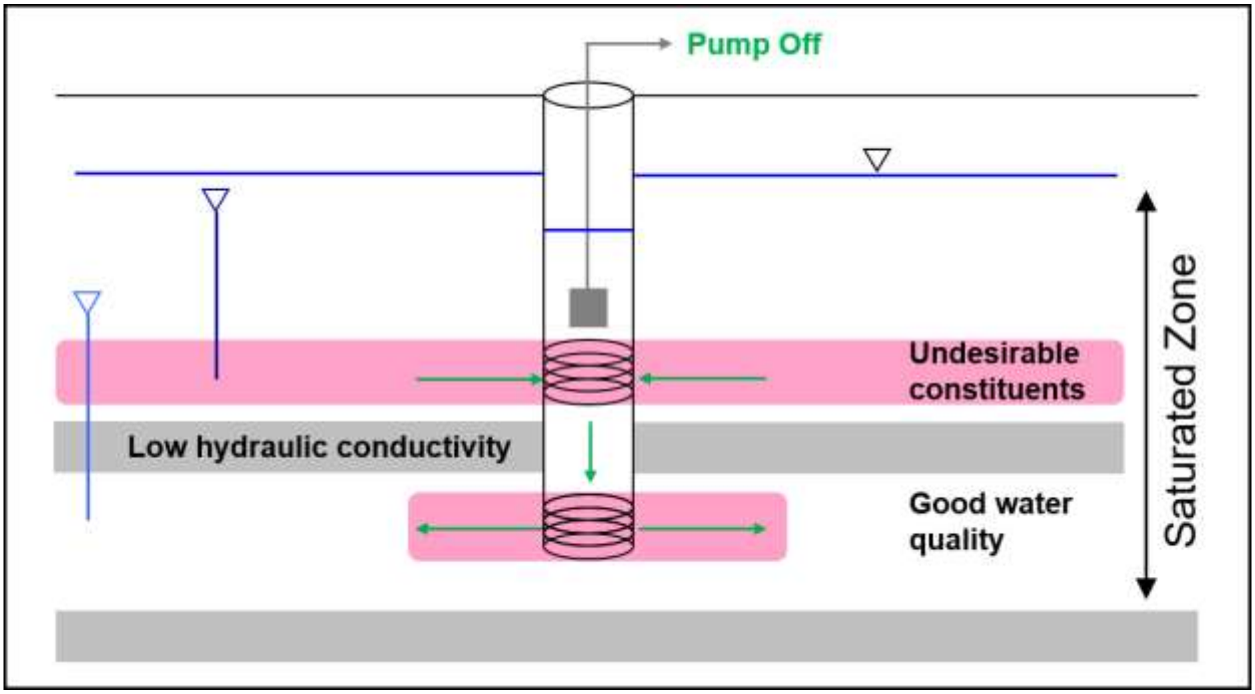


Figure 14: Schematic of well that acts as a conduit for flow and solute transport. *Blue symbols* indicate piezometric levels at different depths in the groundwater system. *Green arrows* indicate relatively rapid flow between water-bearing strata through the well (short-circuiting flow)

Data Sources

Data on the above-referenced conditions are available for locations across the study area from a variety of sources. Decreases in head with depth can be evaluated using groundwater level data available from the California Department of Water Resources (CADWR 2017a). These data were downloaded as spreadsheet files. Variations in shallow groundwater quality (above the Corcoran

Clay) are documented for nitrate and total dissolved solids (TDS) by the Central Valley Salinity Coalition (CVSC 2016). These data were obtained as geographic information system (GIS) layers through an information request to the CVSC. Stratigraphic information, consisting of depths from ground surface to the top and thicknesses of the Corcoran Clay, is documented by Faunt (2012). These data were downloaded as GIS layers. For supply wells, screened intervals, specified by depths from ground surface to the tops of the shallowest and bottoms of the deepest screened sections, casing diameters and approximate locations are documented by the CADWR (CADWR 2017b). These data were obtained as spreadsheet files through an information request to the CADWR.

Methods of Analysis

Two types of analysis are performed for this work. First, information on the geographic distributions of different data related to short-circuiting flow in wells are evaluated, localized areas are identified where conditions for such flow appear to exist, and rates of contaminant transfer to deeper strata are estimated. Second, the potential significance of the estimated flows and contaminant transfer rates through individual wells are evaluated within the context of the much larger groundwater system.

Evaluating Geographic Information

The method presented here identifies geographic coincidence of the different conditions necessary for migration through well conduits. The conditions are evaluated at a horizontal resolution limited to 1 mi^2 (2.6 km^2) as a result of censored well location accuracy in the dataset available for this study. Seven data processing steps use a combination of GIS software and spreadsheets with macro scripts to generate results. The workflow is summarized on Figure 15 and described in detail below.

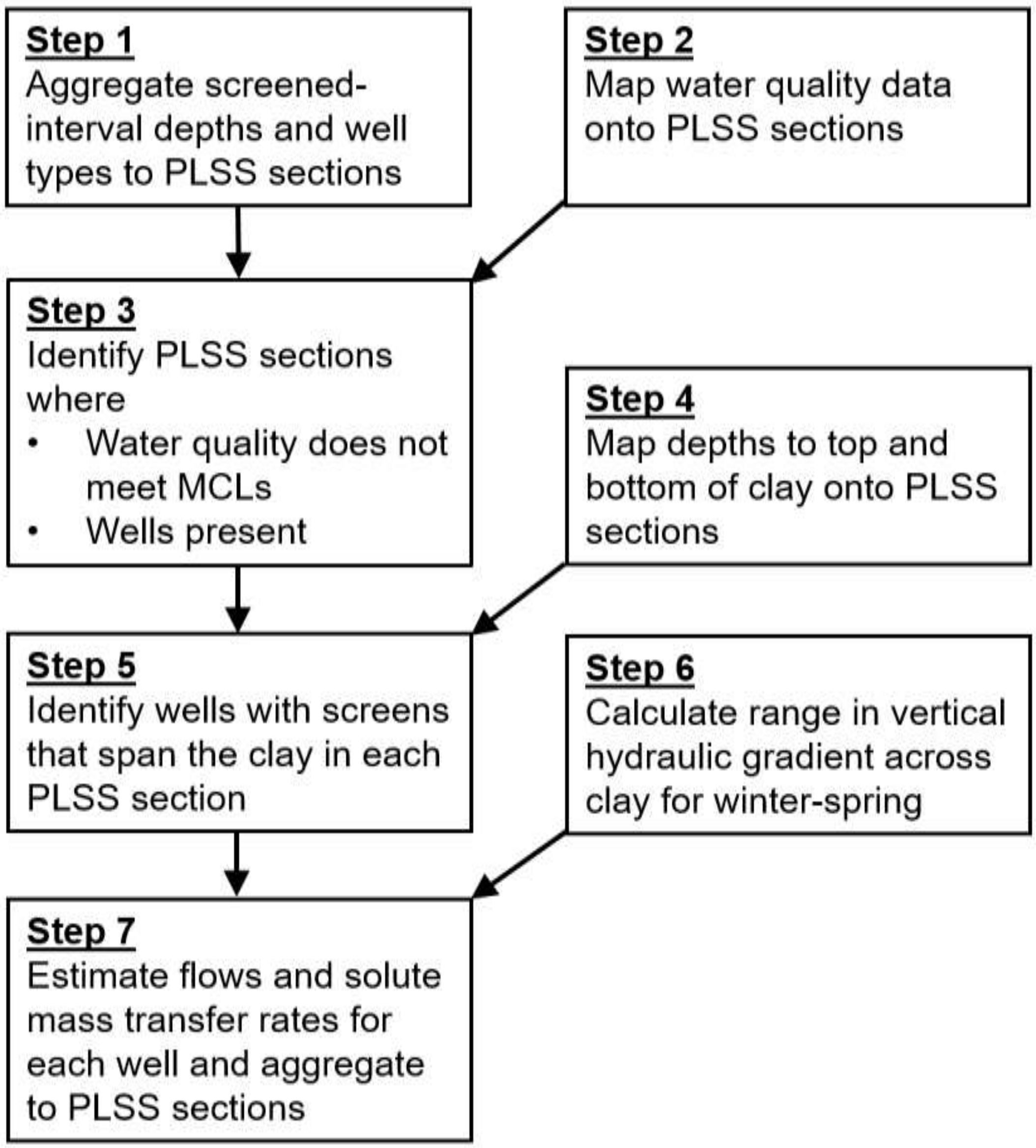


Figure 15: Work flow for evaluating geographic information

1) Screened-interval data for all supply well types in the dataset (agricultural, industrial, municipal and domestic) are available with locations catalogued according to US Public Land Survey System (PLSS) grid at the section level. Because the State of California limits well location information to this 1 mi^2 (2.6 km^2) grid to protect well owner privacy, the maximum horizontal resolution of this study is limited to the same. The screened-interval depth and well type data are aggregated and mapped onto the PLSS section grid.

2) The data for water quality in the shallower water-bearing zone (above the Corcoran Clay) are mapped onto the PLSS grid. Contoured representations of the water quality data are available as GIS polygon shapefiles on a grid of similar horizontal resolution as the PLSS (CVSC 2016). These data are spatially joined to the PLSS grid.

3) Lists of PLSS sections that meet two criteria are compiled.

   a. Water quality constituent concentrations exceed the maximum contaminant level (MCL) for nitrate, TDS or both.
   b. At least one well of any depth is present.

4) Depths to the top and bottom of the Corcoran Clay are mapped onto the PLSS grid. Contoured representations of the depth to top and thickness of the clay, available as GIS polyline shapefiles (Faunt 2012), are rasterized and converted to GIS polygon shapefiles with values specified for each cell on the PLSS grid. Depth to the bottom of the clay is then calculated for each grid cell as the sum of depth to top and thickness of the clay.

5) The number of likely conduit wells is recorded for each PLSS section where water quality does not meet MCLs.

a. Well construction data for each PLSS section in the lists from step 3 are accessed.
b. Depths to top and bottom of screened intervals are compared to depths to top and bottom of clay for the PLSS section.
c. Wells with screens that vertically span the thickness of the Corcoran Clay in each PLSS section are aggregated and mapped to the PLSS grid.

6) Vertical head differences and hydraulic gradients across the Corcoran Clay are calculated for the winter-spring when wells are most often idle and potentially act as conduits. Groundwater levels are initially selected from the available monitoring well dataset if they meet all of the following information requirements: PLSS section location, screened-interval depths and water level data available for winter-spring. The selected data are then culled based upon the following criteria: PLSS section has data from both above and below the Corcoran Clay, data above and below clay occur during winter-spring of the same year, data occur during 2012 to 2016. Because monitoring wells screened above and below the clay are generally not at the same location, data aggregation is necessary. For each PLSS section where data meeting the above-referenced requirements are available, averages are calculated for the following quantities: heads above the clay, depths to bottoms of screened intervals above the clay, heads below the clay and depths to tops of screened intervals below the clay. A vertical gradient is then calculated for each PLSS section:

$$i_{clay} = (H_{above} - H_{below})/(D_{screen\_bottom\_above} - D_{screen\_top\_below}) \quad (5)$$

Where i_{clay} is the estimated vertical hydraulic gradient across the clay

H_{above} is average head above the clay

H_{below} is average head below the clay

$D_{screen\_bottom\_above}$ is average depth to bottom of well screen above the clay

$D_{screen\_top\_below}$ is average depth to top of screen below the clay

7) Flows and mass transfer rates through conduit wells from strata above the Corcoran Clay to those beneath the clay are estimated across the study area.

a. Effective hydraulic conductivity estimates are generated for all wells with screened intervals that span the Corcoran Clay. Well diameter data are used with the Hagen-Poiseuille equation to perform the calculations (Appendix; Gailey 2017).

b. Flow rates are estimated for each well using Darcy's Law with information developed above:

$$Q_{well\_est} = K_{well}\ i_{clay}\ A_{well} \quad (6)$$

Where Q_{well_est} is the estimated flow rate through the well

K_{well} is the effective hydraulic conductivity for the well

A_{well} is the well casing cross sectional area calculated from the well diameter

High and low values for the vertical gradient across the study area (calculated in step 6) are used as the basis to estimate a range in flow rate for each well. These vertical gradients are scaled by a factor of 2 x 10^{-6} based on calculations presented in the Appendix:

$$Q_{well\_est} = K_{well}\, f_{scale}\, i_{clay}\, A_{well} \tag{7}$$

Where f_{scale} is the gradient scaling factor

The scaling is necessary because gradients associated with flow through well casings are typically much lower than vertical gradients in groundwater systems at some distance from the wells as a result of 1) decreased resistance to vertical flow within the casing relative to naturally layered groundwater systems and 2) head losses related to passage through well screens and 3) convergent (divergent) flow to (from) the well through the porous medium. (Compare on Appendix Figure 28 the difference between Hc_U and Hc_L as opposed to the difference between H_U and H_L.) Therefore, over estimation of flow rates occur unless the gradient is reduced (Gailey 2017).

c. Rates of nitrate and TDS transfer across the clay are calculated for each well as the product of flow through the well and concentration in the PLSS section.

d. The flows and transfer rates, initially calculated for the period of one day, are then scaled up to a 180-day period (multiplied by 180). The upscaling is performed to represent the amount of time that wells are generally idle and potentially act as conduits each year during the low demand season. The results are then aggregated for each PLSS section and the study area as a whole.

Comparing Regional and Well-Specific Fluxes

Two numerical groundwater flow and solute transport models presented by Gailey (2017) are used to compare the relative magnitudes of groundwater volumetric flows, fluxes per unit area (Darcy velocities) and extent of water quality impact from solute mass transfer through 1) regional aquitards and 2) wells that act as conduits. The models simulate a hypothetical groundwater system comprising two aquifers and an intervening aquitard. The first model simulates steady-state heads when no conduit well is present. The second model uses the steady-state heads as initial conditions to simulate transient flow and advective transport where 1) no conduit well is present and 2) an idle water supply well cross-connects the aquifers. A single 180-day stress period is simulated. Table 4 summarizes the model parameters. Additional details regarding the model structure are presented in Gailey (2017).

Table 4 Numerical model parameters

Parameter category	Parameter	Value
Half-Space Domain Dimensions	Width	628 m
	Length	1,638 m
	Depth	75 m
Upper and Lower Aquifer Parameters	Thickness	30 m
	Horizontal Hydraulic Conductivity	10^{-2} cm/s
	Vertical Hydraulic Conductivity	10^{-3} cm/s
	Storage Coefficient	10^{-4}
	Porosity	0.3
	Recharge (Upper)	9×10^{-4} m/d
	Initial Concentration (Upper)	100 mg/l
	Initial Concentration (Lower)	10 mg/l
Aquitard Parameters	Thickness	15 m
	Horizontal Hydraulic Conductivity	10^{-5} cm/s
	Vertical Hydraulic Conductivity	10^{-6} cm/s
	Storage Coefficient	10^{-4}
	Porosity	0.3
	Initial Concentration	100 mg/l
Idle Well Parameters	Gravel Pack Hydraulic Conductivity	1 cm/s
	Casing Hydraulic Conductivity	100 cm/s
Hydraulic Gradients	Horizontal in Aquifers	0.001
	Vertical Across Aquitard	-0.15

Given the variation in conditions in the study area (Page 1986; Faunt 2009) and the simplified representation of aquifer system hydrostratigraphy in the modeling, the aquitard hydraulic conductivity is varied to evaluate the potential range of groundwater volumetric flows, fluxes per unit area and solute mass transfer rates. The vertical hydraulic conductivity of the aquitard is changed over four orders of magnitude (with the horizontal hydraulic conductivity remaining one order of magnitude higher that the vertical hydraulic conductivity) and groundwater volumetric flow and flux per unit area are recorded for 1) the aquitard when the well is not present, 2) the aquitard and the well when the well is present. Flow through the aquitard is recorded for the entire model domain (considered two times the simulated half-space dimensions presented in Table 4) and flux is the average over the domain. The length of the solute plume in the lower aquifer resulting from the well acting as a conduit is also recorded (taken as the maximum length parallel to the direction of groundwater flow where concentrations exceed the background value of 10 mg/l presented in Table 4). Sensitivity of the solute transport results to the horizontal hydraulic conductivity of the aquifers is also evaluated by repeating the calculations for an order of magnitude reduction in the values used for the aquifer horizontal hydraulic conductivity (aquifer horizontal hydraulic conductivity reduced from 10^{-2} to 10^{-3} cm/s with the vertical hydraulic conductivity remaining one order of magnitude lower than the horizontal hydraulic conductivity).

Limitations

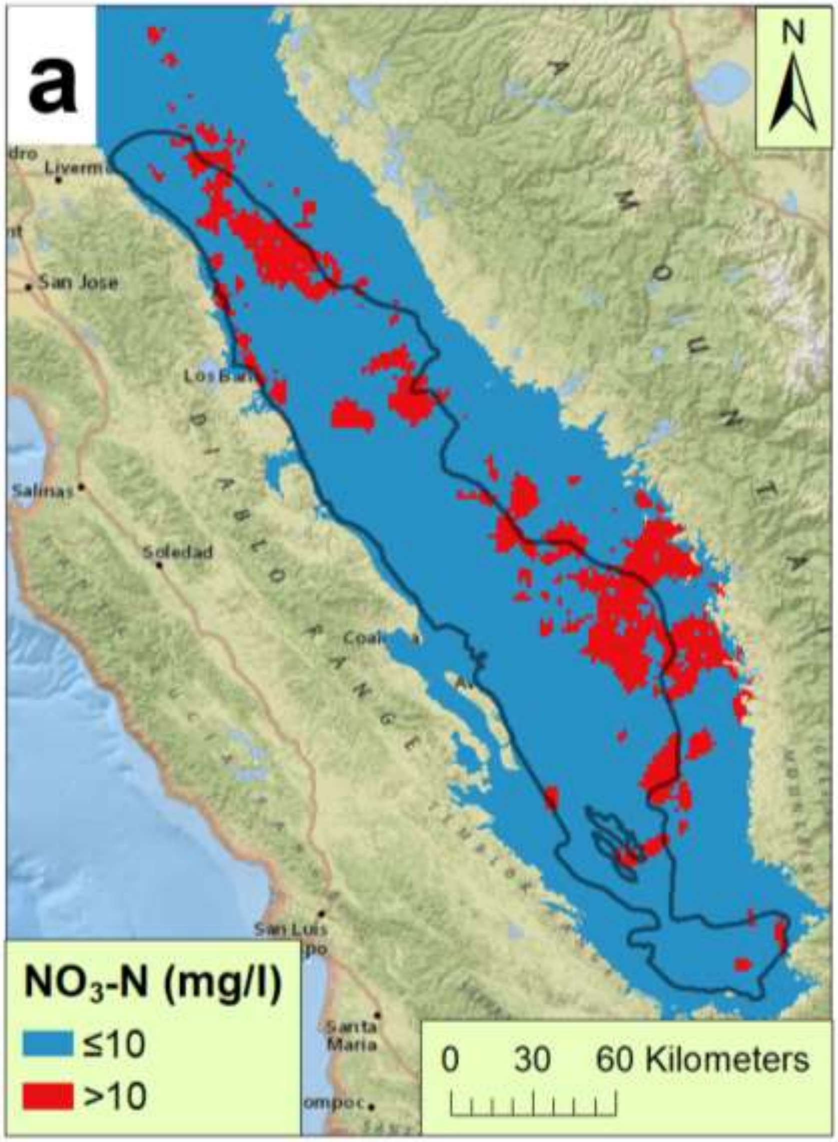
The approach for 'Evaluating geographic information' assumes that the available datasets regarding water quality, stratigraphy and hydraulic head reasonably characterize the hydrogeologic conditions at the scale of PLSS sections (1 mi^2, or 2.6 km^2). These data are available at horizontal resolutions generally lower than the PLSS grid and interpretations, guided by the professional judgement of the author (for representative ranges for vertical hydraulic gradients) and others (for contours of clay depth and thickness as well as water quality constituents obtained for this work), are applied. Additional interpretations are made with regard to the well data. Because no comprehensive well destruction records are available, all wells in the well construction dataset are used in the analysis. This aspect of the approach tends towards a conservatively high estimate of the effects from well conduits as it is possible that some of the wells included in the analysis have been properly destroyed and may no longer exist as potential conduits. However, other wells not identified by this approach may act as conduits because of 1) localized occurrences of nitrate and TDS not included in the water quality information used here, 2) other contaminants not considered and 3) well constructions with long gravel packs that short-circuit the Corcoran Clay. (Transport along gravel packs could be considered if data on this aspect of the well constructions were available.) It should also be noted that short-circuiting flow and contaminant migration relative to other low-hydraulic conductivity strata above the Corcoran Clay may also occur but is not the focus of this work. These potential limitations could be addressed through use of additional data that may become available in the future. Since the purpose of the approach is developing indications of where concern may be warranted so that potential conduit wells can be considered in more detail, the limitations identified here are reasonable.

The approach for 'Comparing regional and well-specific fluxes' assumes 1) simplified stratigraphy, 2) fairly low vertical hydraulic gradient and casing hydraulic conductivity compared to what is possible in the field and 3) only a single well conduit. As a result, the predicted flows and fluxes through the conduit well may be low relative to what could occur in the field. The predicted plume lengths may also be shorter than occur in some field cases as a result of the lower flow rates, lack of simulated heterogeneity in aquifer hydraulic conductivity, and no comingling of plumes from multiple conduit wells. While these limitations may result in underestimation of the flows and water quality impacts from conduit wells, conceptual demonstration of how such wells interact with the greater groundwater system is still achieved.

Results and Discussion

Conditions Creating Potential for Conduit Migration

The results from steps 1 through 3 of 'Evaluating geographic information' provide the initial information regarding conditions that create potential for conduit migration. Figures 16 a and b summarize the distributions of nitrate and TDS concentrations present above the clay. Nitrate concentrations above the MCL (10 mg/l nitrate as nitrogen) are anthropogenic while the TDS concentrations above the MCL (500 mg/l) may result from natural processes in some cases – particularly in the western part of the study area (CVSC 2016). Figures 17 a and b summarize the depth to top and thickness of the Corcoran Clay. In general, the clay is deepest in the central part and on the west side of the study area and also thickest on the west side.


a
N
NO3-N (mg/l)
≤10
>10
0 30 60 Kilometers
Livermo
San Jose
Los Ban
Salinas
Soledad
San Luis
Santa Maria
ompoc
DIABLO RANGE

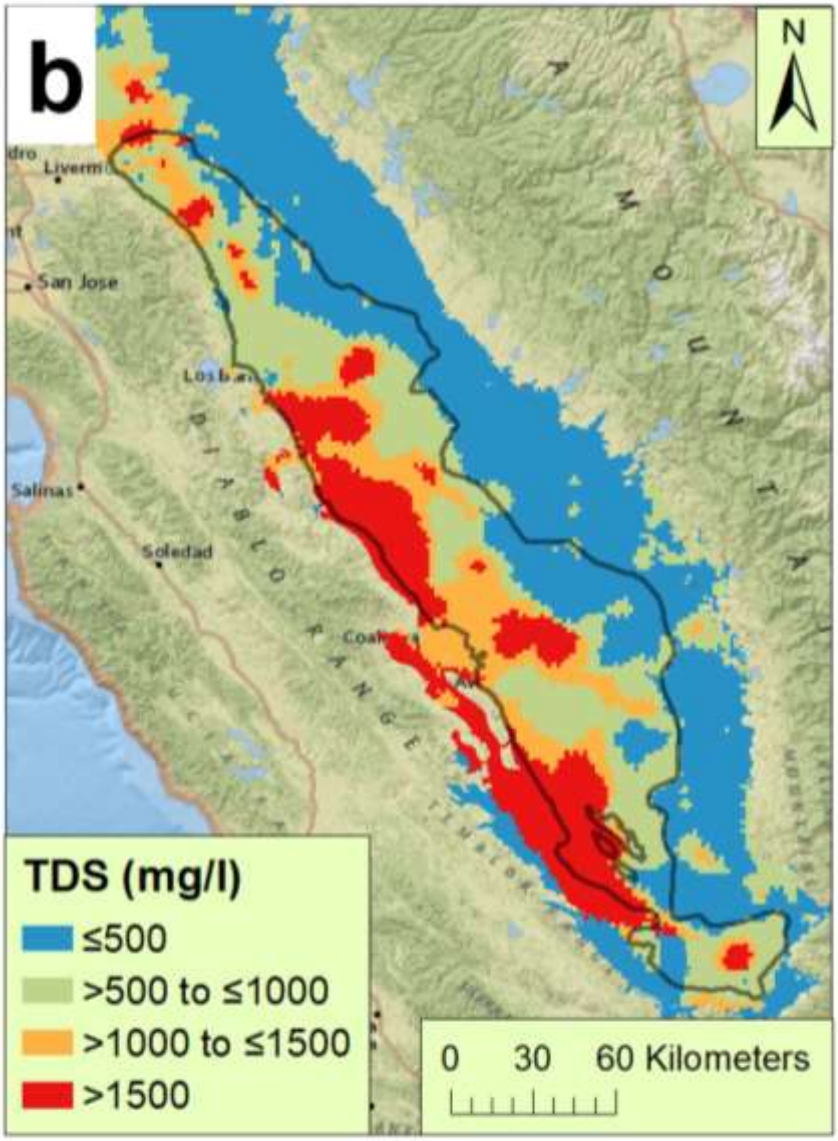
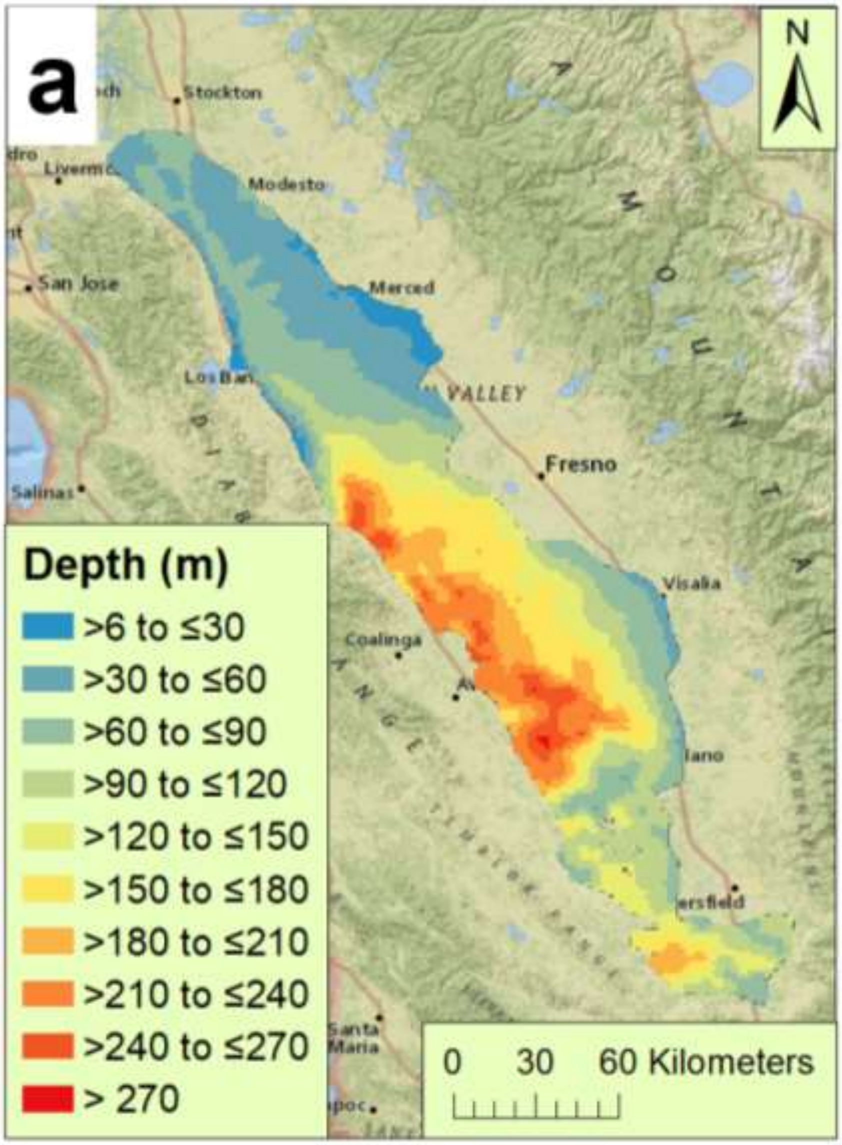


Figure 16: Groundwater quality above the clay: **a** nitrate as nitrogen (NO_3-N) and **b** total dissolved solids (TDS). *Black line* indicates approximate extent of Corcoran Clay. Maximum Contaminant Level for nitrate as nitrogen is 10 mg/l and for total dissolved solids is 500 mg/l. Data from CVSC (2016)


a
Stockton
Modesto
Merced
San Jose
Fresno
Visalia
Coalinga
Salinas
Depth (m)
>6 to ≤30
>30 to ≤60
>60 to ≤90
>90 to ≤120
>120 to ≤150
>150 to ≤180
>180 to ≤210
>210 to ≤240
>240 to ≤270
> 270
0 30 60 Kilometers
N

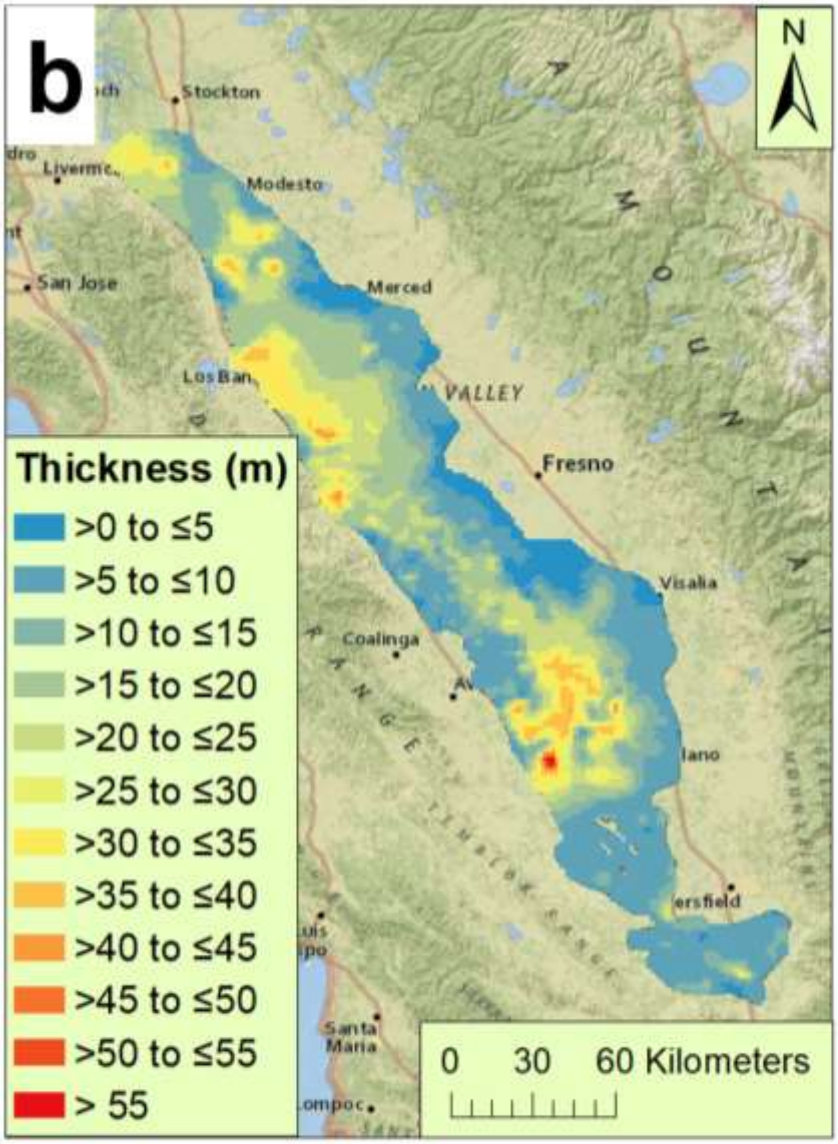
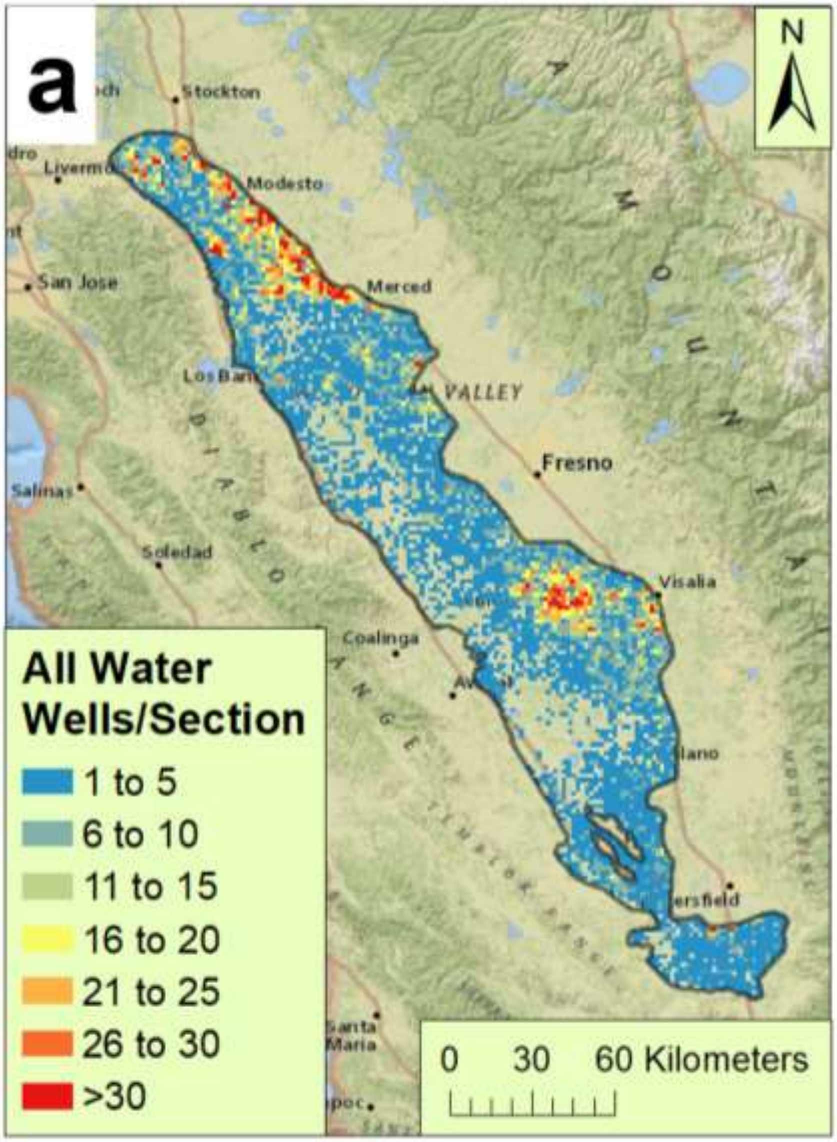
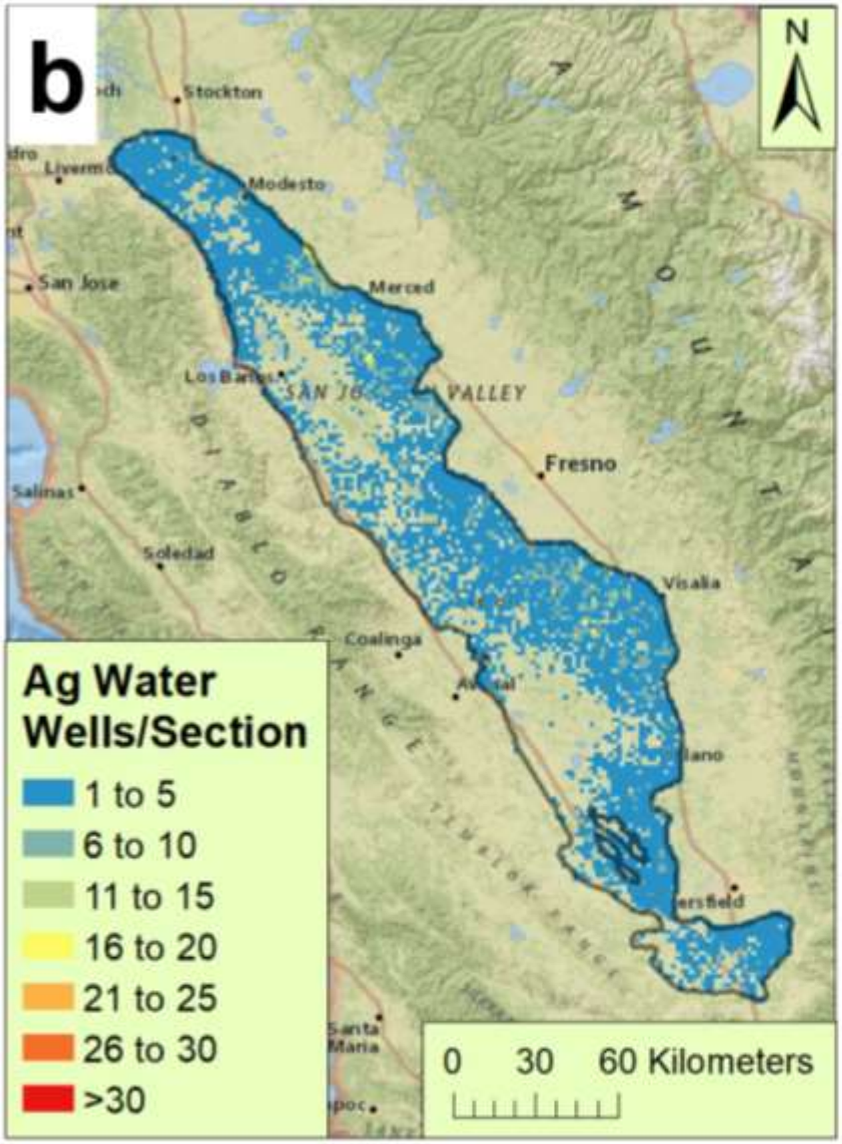


Figure 17: Corcoran Clay: **a** depth to top of the clay and **b** thickness of clay. Data from Faunt (2012)

The well construction data indicate 33,579 supply wells within the extent of the Corcoran Clay (Figure 18a) including 15,024 agricultural; 428 industrial; 804 municipal and 17,323 domestic wells. Many of these wells (22,570 in total) terminate below the clay and decrease heads below the clay when pumped. The vast majority of these wells are agricultural (11,763 or 52%; Figure 18b) and domestic (9,902 or 44%; Figure 18c). Downward vertical gradients drive flows through wells with screened intervals that span the clay thickness (2,693 wells; Figure 18d). These wells tend to be located where the clay is shallower (Figure 17a) and thinner (Figure 17b). Areas where supply wells of any depth are present and water quality does not meet the MCLs for nitrate, TDS or both constituents are indicated on Figures 19 a through c. These results, generated from step 3 of 'Evaluating geographic information', provide the basis for evaluating whether wells in specific areas may act as conduits for contaminant migration. For nitrate, 12,005 wells, or 36% of all wells, are located in PLSS sections where the MCL is exceeded. There are 9,105 wells (27%) that are located in sections with TDS exceedance and 5,557 wells (17%) are located in sections with exceedance of both constituents.


a
N
Stockton
Modesto
Merced
San Jose
VALLEY
Fresno
Salinas
Soledad
Visalia
Coalinga
All Water
Wells/Section
1 to 5
6 to 10
11 to 15
16 to 20
21 to 25
26 to 30
>30
0 30 60 Kilometers


b
N
Stockton
Modesto
Merced
San Jose
Los Banos
VALLEY
Fresno
Salinas
Soledad
Visalia
Coalinga
Santa Maria
Ag Water
Wells/Section
1 to 5
6 to 10
11 to 15
16 to 20
21 to 25
26 to 30
>30
0 30 60 Kilometers

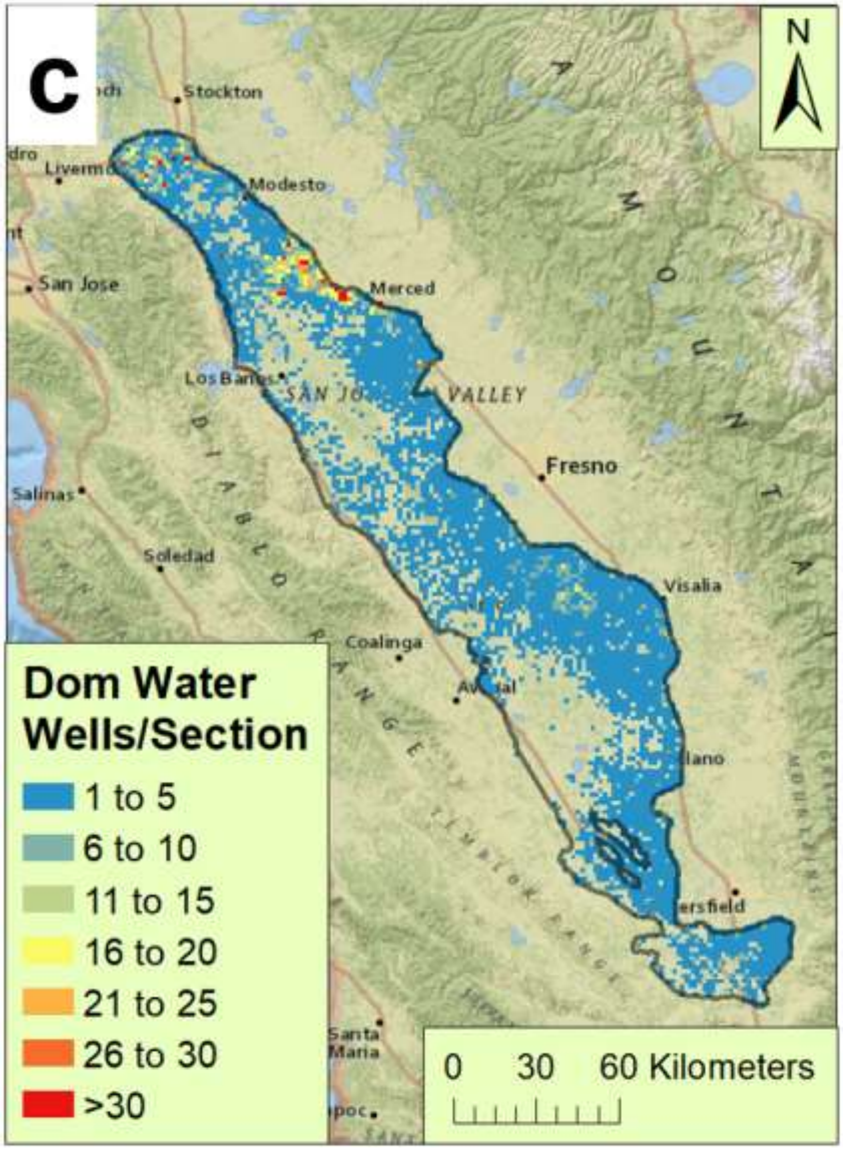


Figure 18: Supply wells in the study area: **a** all wells located within the extent of Corcoran Clay, **b** agricultural (Ag) wells that terminate beneath the clay, **c** domestic (Dom) wells that terminate beneath the clay and **d** total wells with screened intervals that span the clay. *Black line* indicates approximate extent of Corcoran Clay

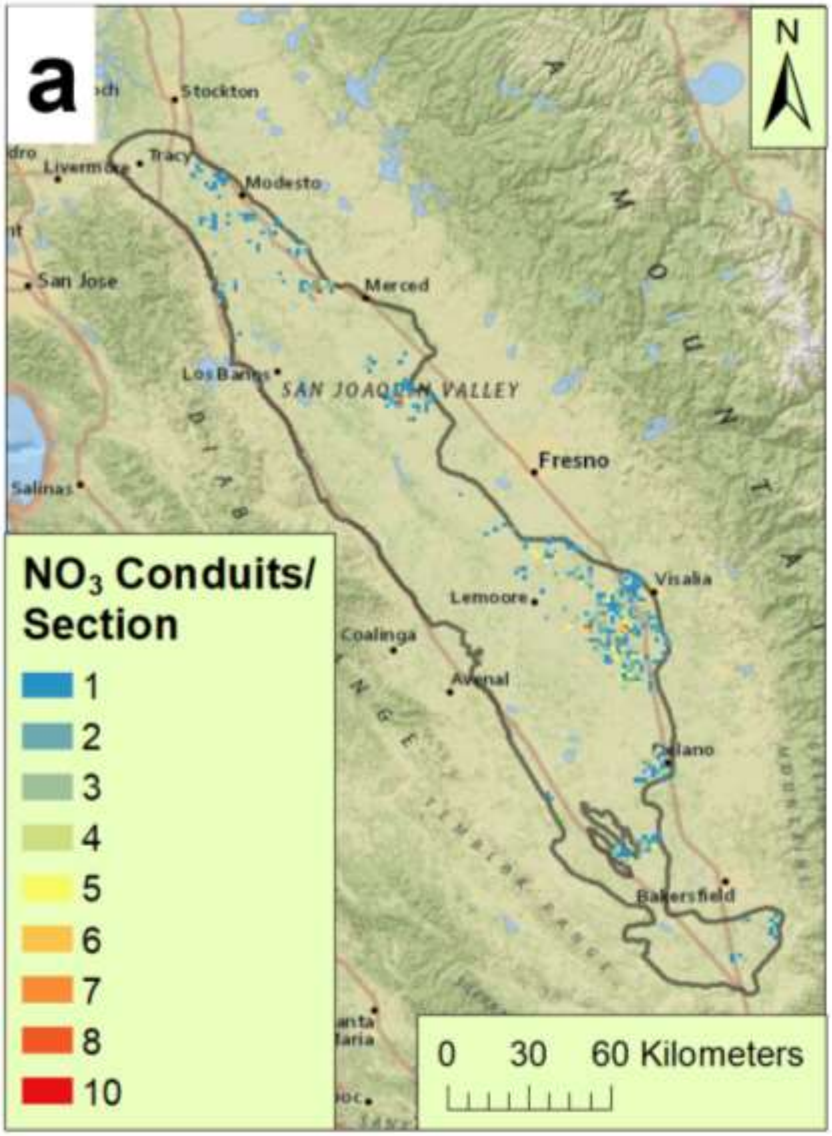
a
N
Stockton
Livermore
Modesto
Merced
San Jose
Los Banos
SAN JOAQ N VALLEY
Fresno
Salinas
Soledad
Visalia
Lemoore
Coalinga
Avenal
Bakersfield
San Luis Obispo
Santa Maria
Lompoc
NO_3 > MCL
0 30 60 Kilometers

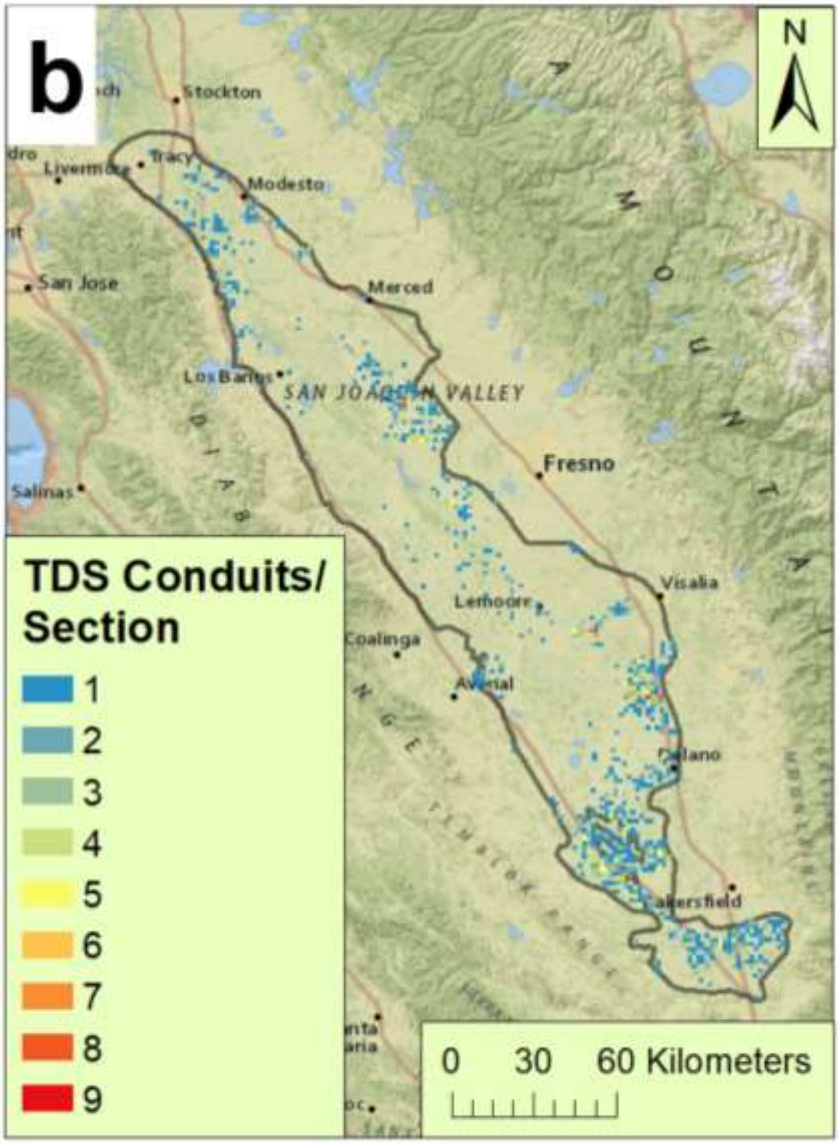

b
N
Stockton
Modesto
Merced
San Jose
VALLEY
Fresno
Salinas
Soledad
Visalia
Coalinga
San Luis Obispo
Santa Maria
Lompoc
TDS > MCL
0 30 60 Kilometers

Figure 19: Areas where supply wells are present and water quality does not meet MCLs: **a** nitrate, **b** total dissolved solids and **c** both constituents. *Black line* indicates approximate extent of Corcoran Clay

Areas of Concern

Results from steps 4 and 5 of ‘Evaluating geographic information’ suggest the areas of concern defined by sections that have 1) concentrations above the MCLs and 2) wells likely acting as conduits. For nitrate, 875 wells located in 430 PLSS sections are constructed such that they may act as conduits for contaminant migration (Figure 20a). This is 3% of the wells within the extent of the Corcoran Clay and 7% of the wells where water quality does not meet the MCL. A notable cluster of wells is present southwest of the city of Visalia and there are smaller clusters throughout the study area. For TDS, 1,505 wells in 817 PLSS sections may act as conduits (Figure 20b). This is 5% of the wells within the extent of the Corcoran Clay and 17% of the wells where water quality does not meet the MCL. While some of the areas of concern overlaps with those for nitrate, the spatial distribution of areas of concern for TDS is different from that of nitrate. Notable clusters of wells are present south of the city of Merced as well as in the southernmost part of the study area. As shown in Figure 20c, 418 wells in 222 sections are constructed such that they may act as conduits for both constituents. This is 1% of the wells within the extent of the Corcoran Clay and 8% of the wells where water quality does not meet both MCLs. There are small clusters of these wells throughout the study area.


a
N
Stockton
Tracy
Livermore
Modesto
San Jose
Merced
Los Banos
SAN JOAQUIN VALLEY
Fresno
Salinas
Visalia
Lemoore
Coalinga
Avenal
Delano
Bakersfield
NO3 Conduits/ Section
1
2
3
4
5
6
7
8
10
0 30 60 Kilometers


b
N
Stockton
Tracy
Livermore
Modesto
San Jose
Merced
Los Banos
SAN JOAQUIN VALLEY
Fresno
Salinas
TDS Conduits/
Section
1
2
3
4
5
6
7
8
9
Visalia
Lemoore
Coalinga
Avenal
Delano
Bakersfield
0 30 60 Kilometers

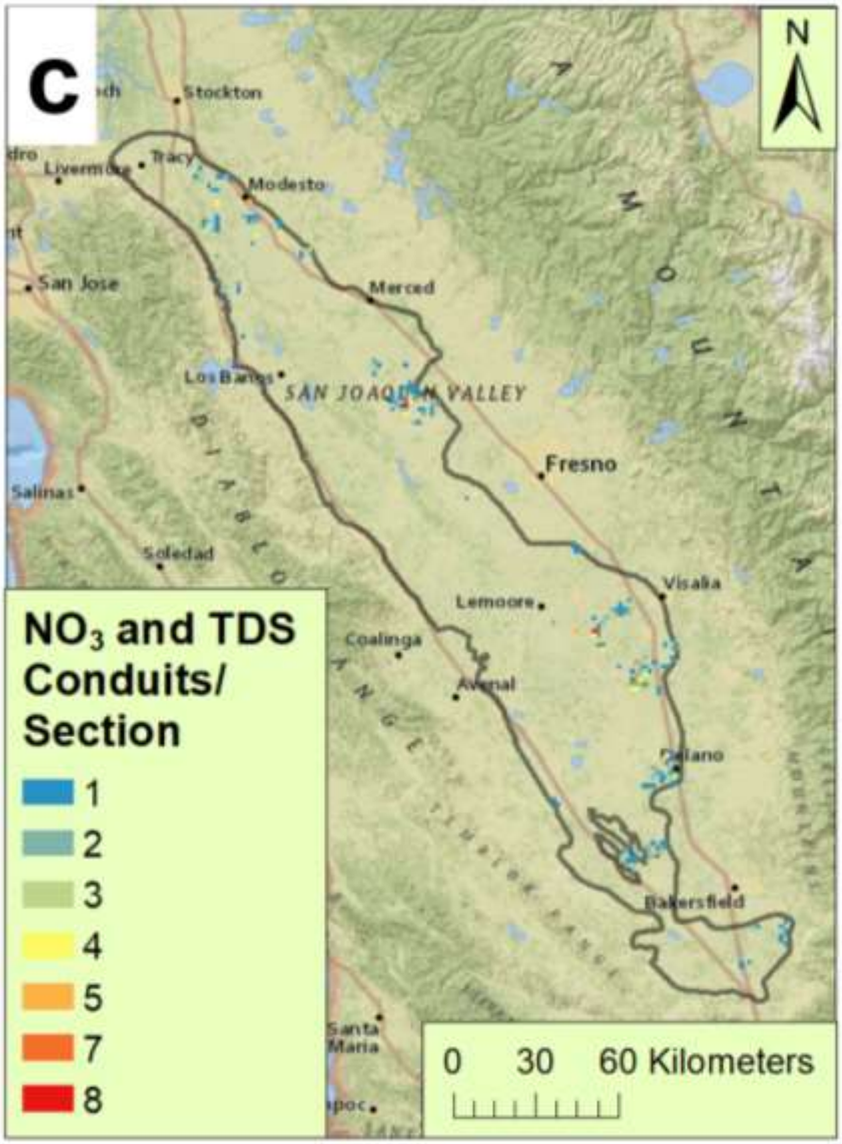


Figure 20: Supply wells that may act as conduits: **a** nitrate, **b** total dissolved solids and **c** both constituents. *Black line* indicates approximate extent of Corcoran Clay

The breakdown of well types that may act as conduits (Table 5) indicates that the majority of wells are agricultural; however, a notable number of domestic wells may also act as conduits. In addition, comparison of Figures 18 b and c with Figures 20 a through c indicates the types of wells potentially impacted by migration through conduits. While both agricultural and domestic wells draw water from beneath the Corcoran Clay throughout the study area and are subject to contamination spreading from wells that act as conduits, water quality requirements are generally more stringent for the domestic wells. The density of domestic wells in the northern portion of the study area near Merced (Figure 18c) may be a particular concern for the spread of nitrate contamination from conduit wells (Figure 20a).

Table 5 Number of wells potentially acting as conduits where MCL exceeded

Constituent	Total	Agricultural	Industrial	Municipal	Domestic
Nitrate	875	711	9	10	145
TDS	1,505	1,280	27	17	181
Both	418	353	7	6	52

Vertical head differences and gradients across the clay during the winter-spring are summarized in Figures 21 a and b. The magnitudes of the head differences are consistent with previous findings (Davis et al. 1964). Likewise, the calculated gradients are supported by previous work (Philips and Belitz 1991). Calculations are made for 29 PLSS sections where sufficient data are available, generally between the years 2013 and 2016. Gradients in the range of -0.5 to -0.1 cover most of the observed variation (69%) and occur throughout the study area. These values are used to estimate ranges in flows for each conduit well.

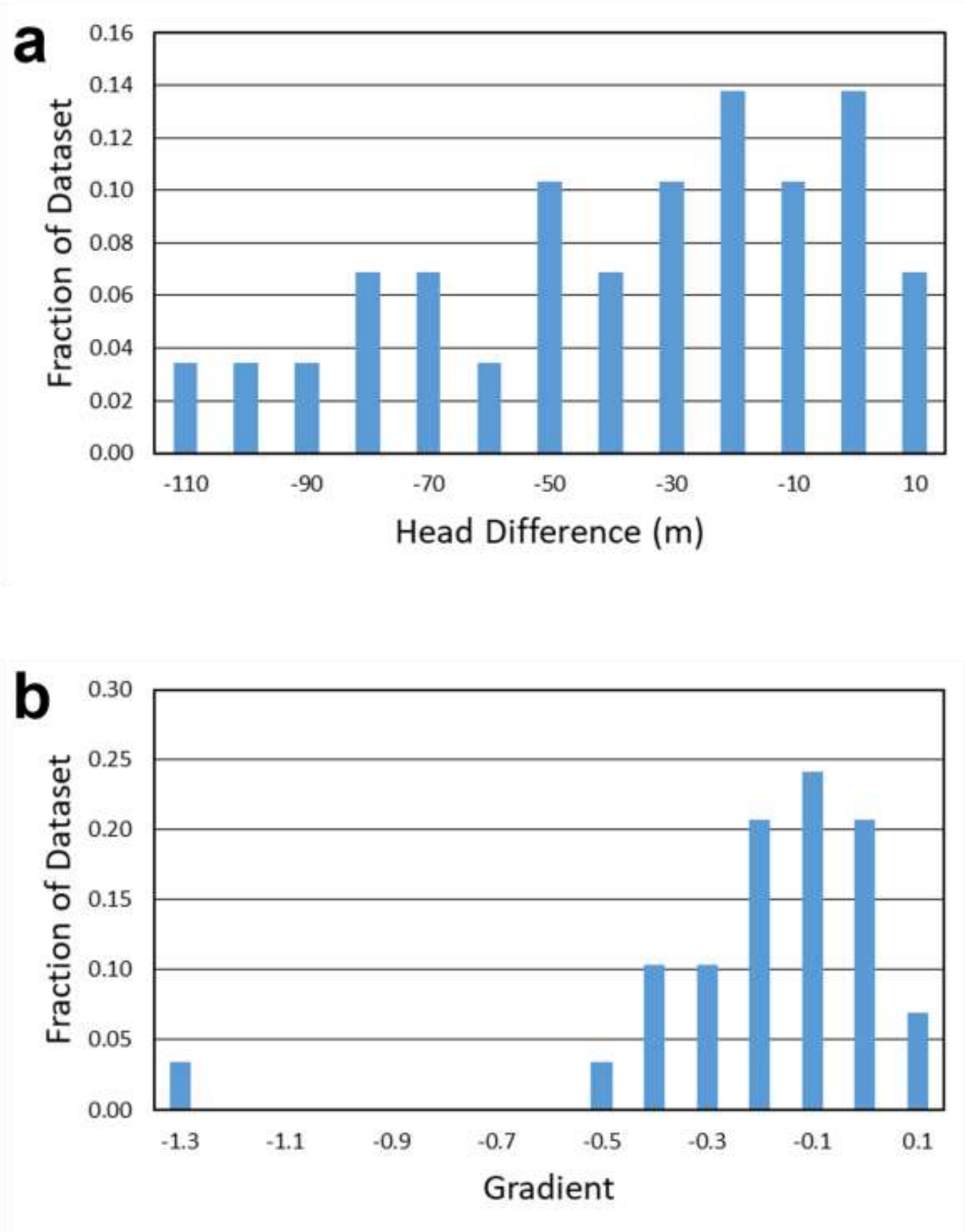
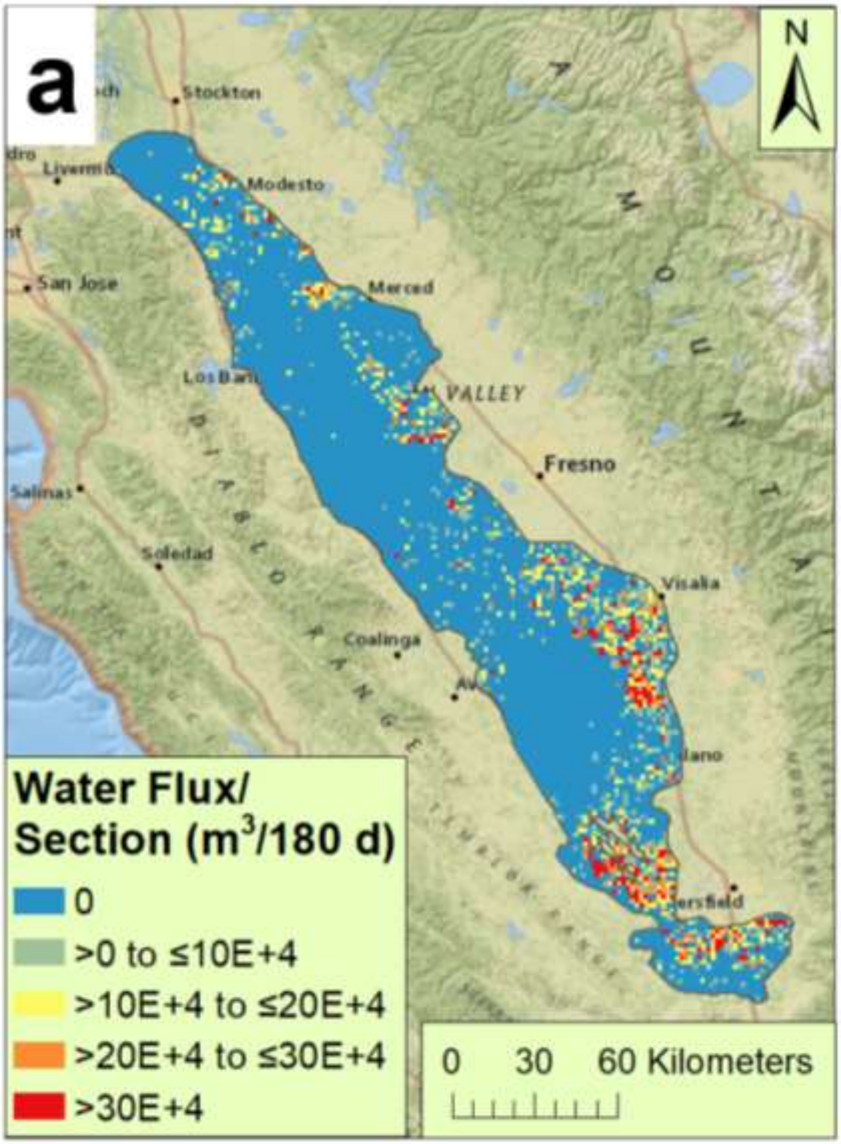
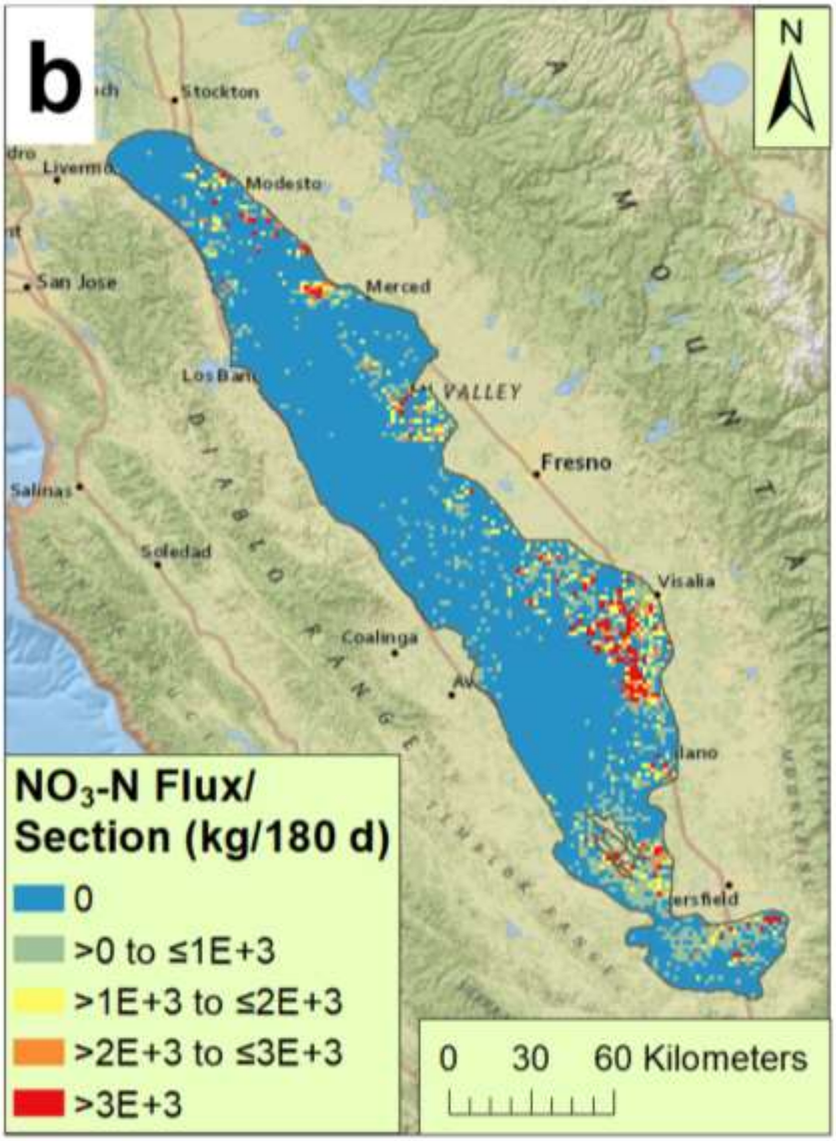


Figure 21: Hydraulic changes across the Corcoran Clay: **a** vertical head difference and **b** vertical gradients. *Negative values* indicate that head is lower beneath the clay than above the clay

Results from step 7 of ‘Evaluating geographic information’ provide estimates for groundwater volumetric flows and contaminant mass transfer rates for the likely conduit wells. Conduit flows range between 5.6 x 10^7 and 2.8 x 10^8 m^3 per 180 days for the study area as a whole. Individual PLSS sections range between 5.0 x 10^0 and 4.8 x 10^6 m^3 per 180 days or approximately 0 to 1.7 percent of the study area total (Table 6 and Figure 22a). The average flow through one of the 2,693 conduit wells on Figure 18d (calculated by dividing the low and high estimated flows for the study area in Table 6 by the number of conduit wells) ranges from 8.1 x 10^1 to 4.0 x 10^2 l/min and is not uncommon for the area (Gailey 2017). The resulting nitrate transfer rates range between 4.2 x 10^5 and 2.1 x 10^6 kg/180 d for the study area. Individual PLSS sections range between 2.3 x 10^{-2} and 3.0 x 10^4 kg nitrate as nitrogen per 180 days or approximately 0 to 1.5 percent of the study area total (Table 6 and Figure 22b). TDS transfer rates range between 5.1 x 10^7 and 2.5 x 10^8 kg/180 d for the study area. Individual PLSS sections range between 9.2 x 10^{-1} and 6.7 x 10^6 kg/180 d or approximately 0 to 2.6 percent of the study area total (Table 6 and Figure 22c). These flows and transfer rates are in addition to those that occur through the Corcoran Clay itself.


a
N
Stockton
Modesto
San Jose
Merced
Los Ban
VALLEY
Fresno
Salinas
Soledad
Visalia
Coalinga
Water Flux/
Section (m³/180 d)
0
>0 to ≤10E+4
>10E+4 to ≤20E+4
>20E+4 to ≤30E+4
>30E+4
0 30 60 Kilometers


b
N
Stockton
Modesto
Merced
San Jose
Los Ban
VALLEY
Fresno
Salinas
Soledad
DIABLO RANGE
Coalinga
Visalia
NO$_3$-N Flux/
Section (kg/180 d)
0
>0 to ≤1E+3
>1E+3 to ≤2E+3
>2E+3 to ≤3E+3
>3E+3
0 30 60 Kilometers

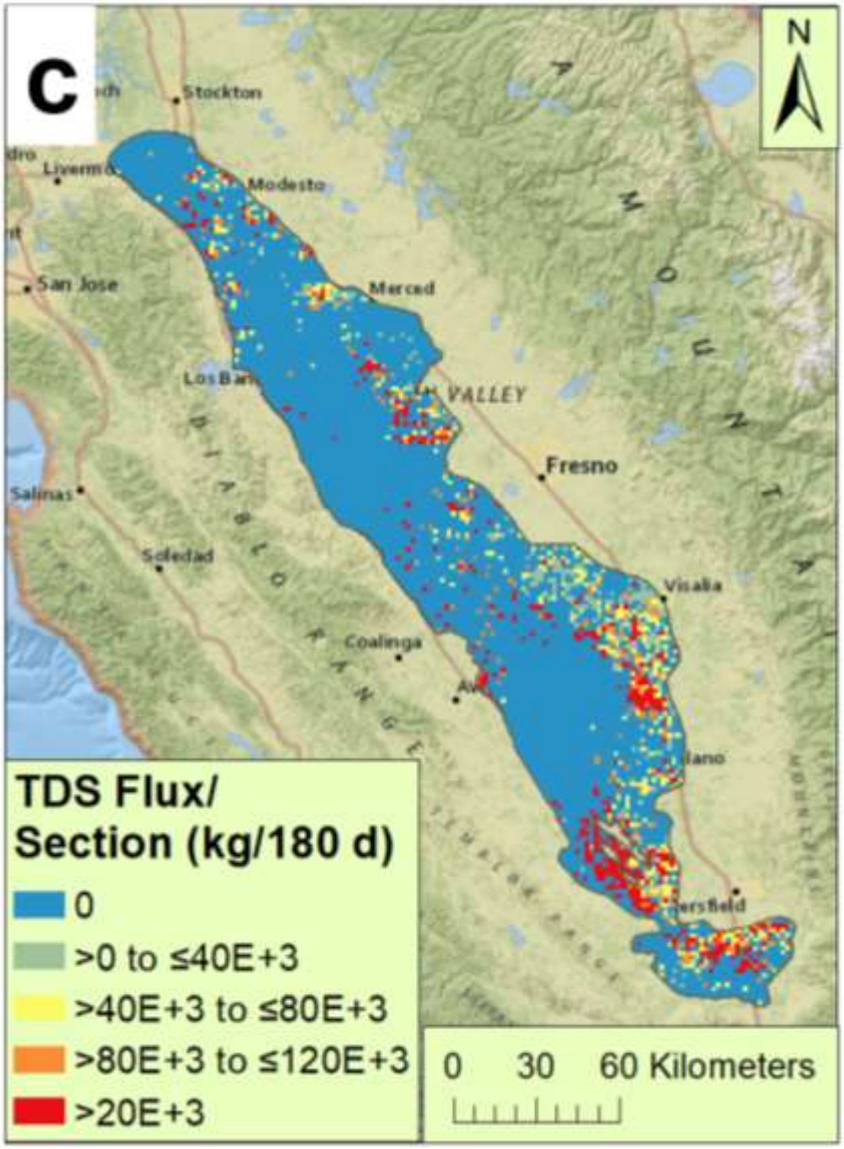


Figure 22: Downward flows and contaminant mass transfer rates through conduit wells: **a** water, **b** nitrate as nitrogen and **c** total dissolved solids. Results correspond to columns on Table 6 labeled *'High'*

Table 6 Estimated transfer rates through potential conduit wells

Location	Water (m^3/180 d)		NO_3-N (kg/180 d)		TDS (kg/180 d)	
	Low	High	Low	High	Low	High
Study Area	5.6×10^7	2.8×10^8	4.2×10^5	2.1×10^6	5.1×10^7	2.5×10^8
Min Section	5.0×10^0	2.5×10^1	2.3×10^{-2}	1.2×10^{-1}	9.2×10^{-1}	4.6×10^0
	0 %	0 %	0 %	0 %	0 %	0 %
Max Section	9.7×10^5	4.8×10^6	6.1×10^3	3.0×10^4	1.4×10^6	6.7×10^6
	1.7 %	1.7 %	1.5 %	1.5 %	2.6 %	2.6 %

Columns designated as *Low* present calculations using a vertical gradient of -0.1.

Columns designated as *High* present calculations using a vertical gradient of -0.5.

Areas for the study area and a section are 17,100 and 2.6 km^2, respectively

Percentages are relative to the *study area values*

Well-Specific versus Regional Fluxes and Impacts on Water Quality

The maximum number of wells with screens that span the Corcoran Clay in a PLSS section is 10 (Figure 18d). Therefore, the area covered by the Corcoran Clay is far larger than the combined cross-sectional area of the conduit well casings. The results of numerical modeling ('Comparing regional and well-specific fluxes') demonstrate that this difference in areas is so great that it is expected to outweigh differences in other Darcy Law terms for flows through the aquitard and well (hydraulic conductivity and gradient). For most conditions, volumetric flow through the aquitard is expected to exceed that through a conduit well (Figure 23). This is true for the aquitard geology in the study area where variations in lithology (Page 1986 and Figure 17b) result in local vertical hydraulic conductivities for the aquitard that likely allow leakage (i.e., $K_v > \sim 10^{-6}$ cm/s on Figure 23). However, comparison of groundwater fluxes calculated by normalizing the volumetric flow rate by the area (Darcy velocities) indicates that flux through conduit wells is many orders of magnitude greater than through the aquitard (Figure 24). While flux through the well is sensitive to the value of aquitard hydraulic conductivity, significant difference between the conductivities of the well and the aquitard remains over a reasonable range of aquitard conductivities and the flux through the well is greater by orders of magnitude.

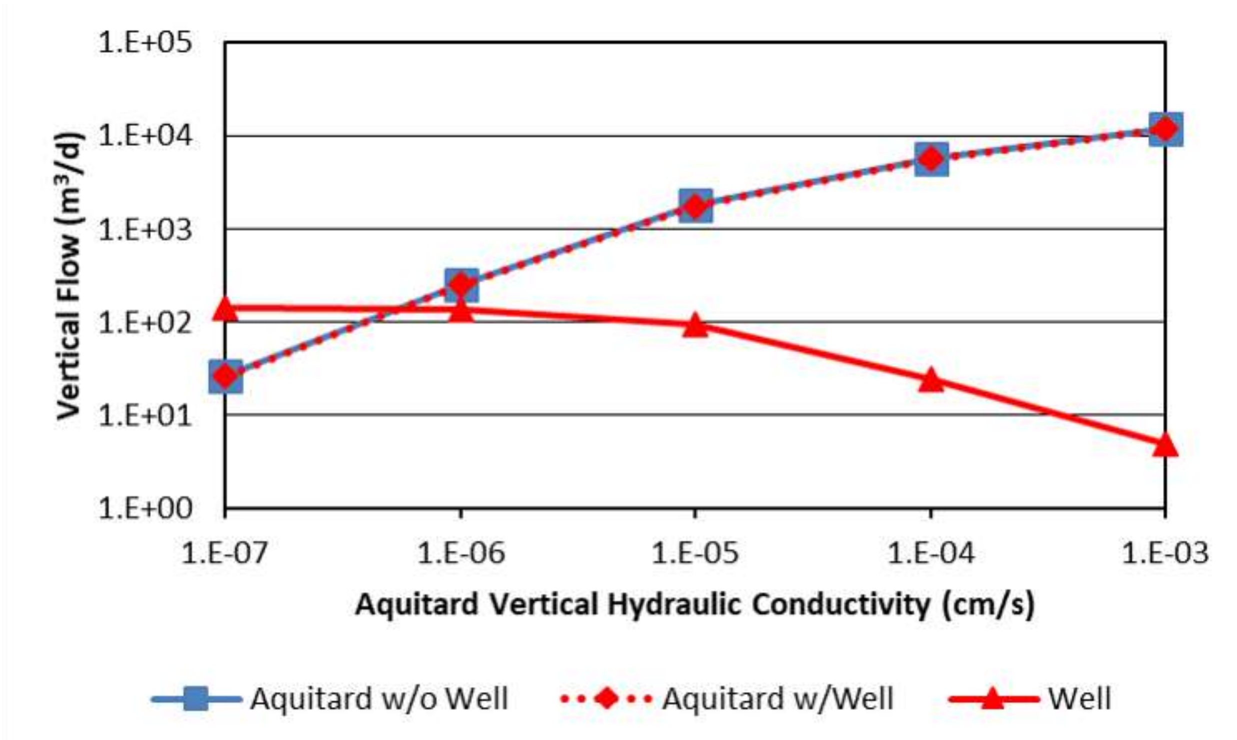


Figure 23: Volumetric water flows through the aquitard and conduit well predicted by the numerical model

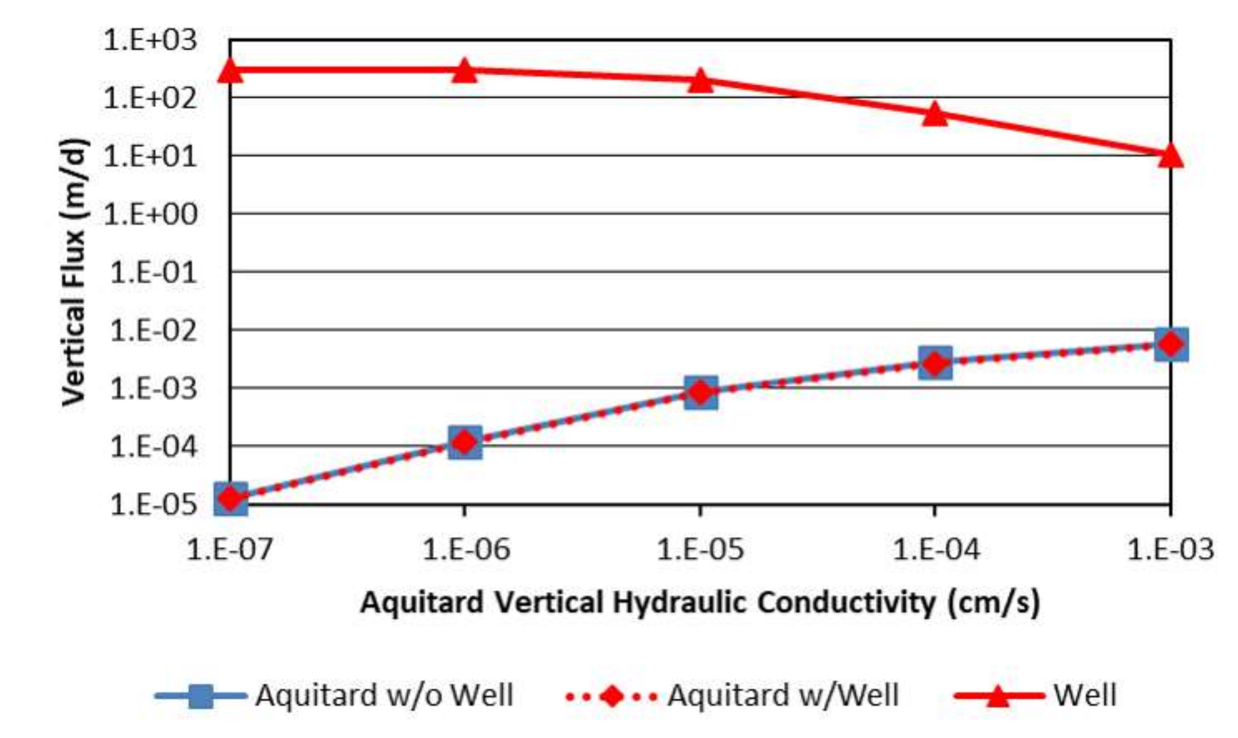


Figure 24: Water fluxes through the aquitard and conduit well developed from volumetric water flows presented in Figure 23

Because the flow of contaminated water through the aquitard occurs over a large area and the flux through the aquitard is relatively low (Figure 24), mass transfer occurs in a dispersed fashion and dilution results from the horizontal flow of cleaner water in the lower aquifer. However, the level of water quality impact in the lower (receiving) aquifer from contaminant mass transfer through a conduit well is greater because it occurs as a result of a localized and higher flux. This point is clarified by considering the ratio of well to aquitard groundwater fluxes and the resulting water quality impact (Figure 25). For cases where the aquitard conductivity is quite low (left-hand side of Figure 25), groundwater flux through the well is much greater than through the aquitard and contaminant plumes are relatively large. Conversely, when the aquitard conductivity is close to that of the aquifers (right-hand side of Figure 25), there is less flux through the well relative to the aquitard and the water quality impact from the conduit well does not migrate as far from the well.

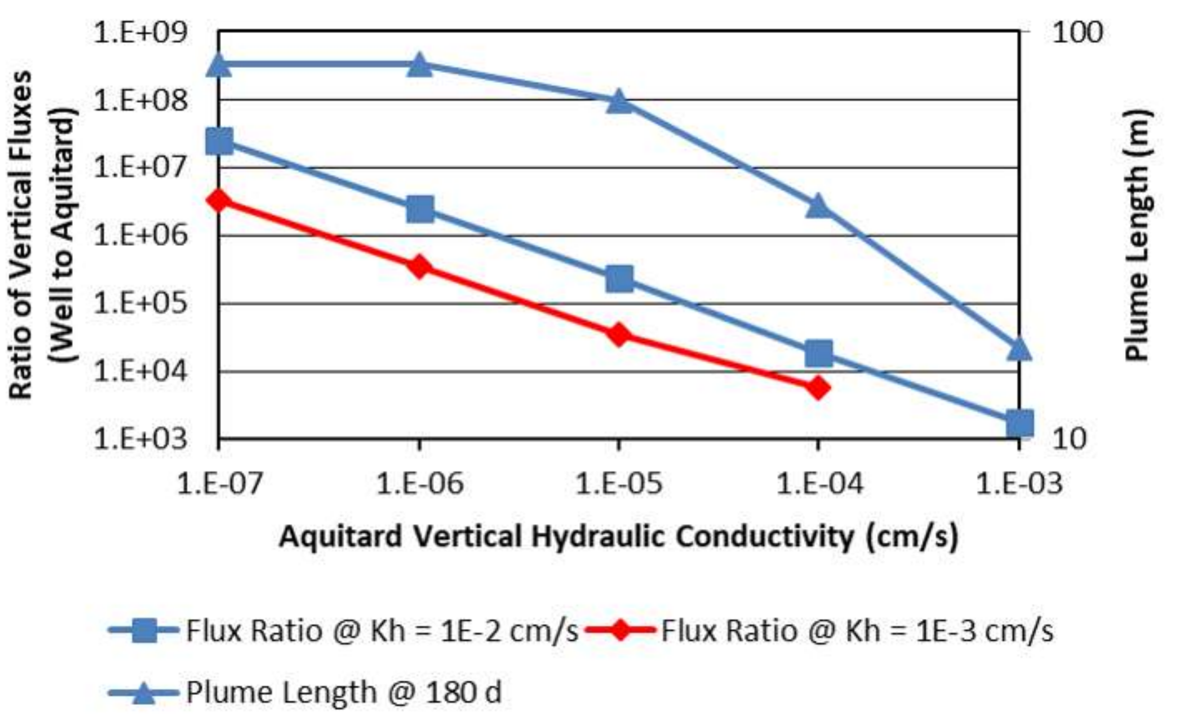


Figure 25: Ratio of groundwater fluxes through the well and aquitard as well as resulting contaminant plume length in the lower aquifer predicted by the numerical model. *Red and blue lines* indicate differences in model results from changing the aquifer horizontal hydraulic conductivity by a factor of ten

Targeted Approach for Improvement

Addressing all, or most, conduit wells in the study area would likely be impossible. Given the many wells and often-forgotten locations of older wells, the costs involved would limit any program of investigation and corrective action. However, the results presented here support use of a targeted approach. Review of Table 6 for each water quality constituent indicates that the maximum transfer rate in a single section is responsible for approximately 1 to 3 percent of the total migration through conduit wells in the study area. Additional analysis of the nitrate transfer rates by section reveals that a significant amount of the area-wide vertical transport for this contaminant occurs in a small number of PLSS sections (Figure 26). Ten percent of the estimated total nitrate migration through conduit wells occurs in only ten sections. Inspection of the geographic data indicates that only 21 wells are involved. Similarly, 30 percent of the transport occurs in 61 sections (225 wells). This information can be used to target limited areas for investigation and potential corrective action.

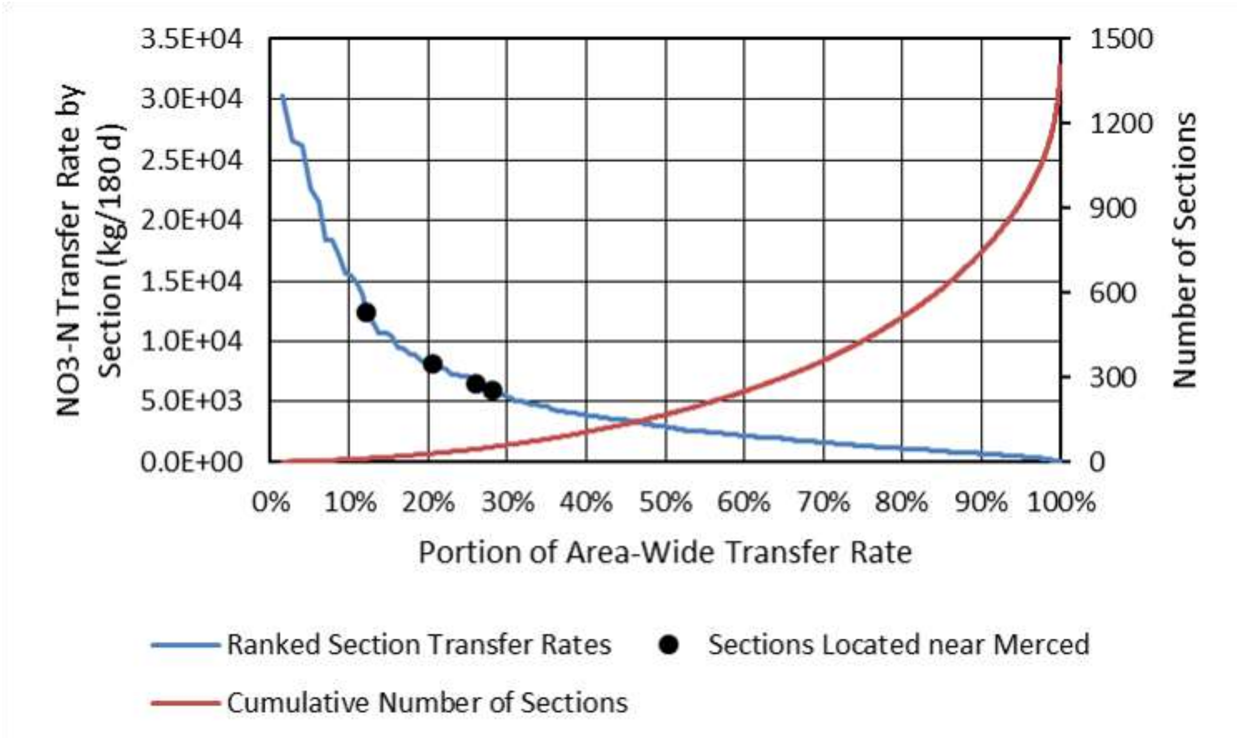


Figure 26: Section contributions for well-conduit transfer of nitrate to strata beneath the clay. Results correspond to *'NO_3-N High'* on Table 6. *Black dots* indicate estimates for PLSS sections located west of Merced (shown in *circled area* on Figure 27)

Consider, for example, that some of the higher nitrate transfer rates (Figure 22b) are close to higher densities of domestic wells in the northern part of the study area (Figure 18c). Moreover, four sections west of the city of Merced contribute notable portions of the total nitrate transport and are close to significant numbers of domestic wells (black dots on Figure 26 and circled area on Figure 27). Useful improvements in groundwater protection and quality might be achieved by investigating this small number of sections where contaminant transfer rates are likely to be high and only 28 wells would be involved.

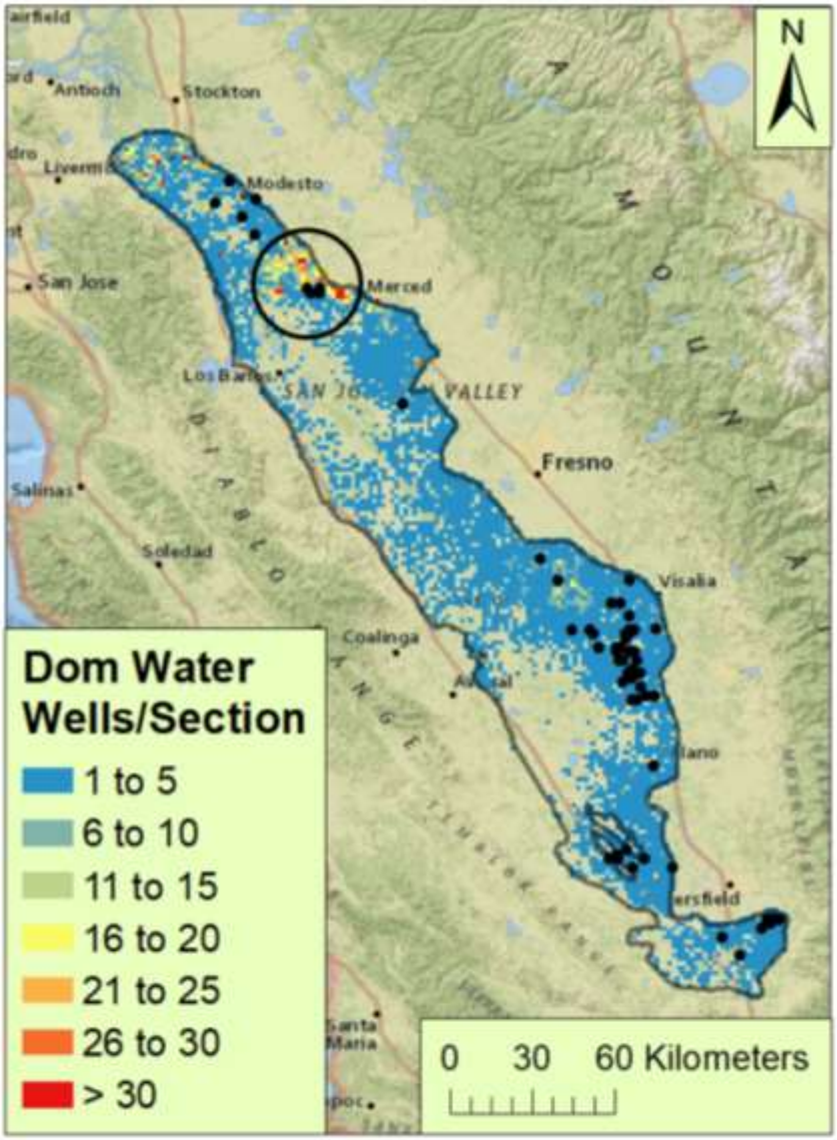


Figure 27: Domestic wells that terminate beneath the clay as on Figure 18c and PLSS section locations with highest nitrate transfer rates. *Large black dots* indicate section locations that together contribute 30% of total well conduit contaminant transfer to strata beneath clay. *Circled portion* contains sections with high nitrate transfer rates (shown on Figure 26 as *black dots*) and nearby areas with higher densities of domestic wells

Conclusions

Identifying areas where supply wells likely act as conduits for contaminant migration is fairly simple if there is access to informative data. In this case, survey-level analysis is performed for a geographically extensive area to identify the co-occurrence of 1) wells screened across a regional aquitard and 2) poor water quality in shallow strata. The number of wells that appear to act as conduits is potentially significant, even though it is a small fraction of all wells in the study area, considering 1) estimates of contaminant transfer rates, 2) proximity to domestic wells and 3) the need to control nonpoint source pollution and improve drinking water quality for rural residents.

These results should be viewed within the context of the analysis performed and assumptions made (see 'Methods of analysis' above). Some false-positive and false–negative results undoubtedly occur for individual wells. Also, contaminant mass transfer rates are approximate. Nevertheless, information from the analysis provides insights regarding areas where investigation and corrective action might be best targeted. Follow-on work guided by the geographic analysis might begin with considering information for specific wells such as 1) construction and lithologic logs prepared at the time of well completion and 2) water quality problems upon start up after idle periods (Gailey 2017). Potential responses to issues that may be revealed include structural and operational changes such as well-screen modification or regularly scheduled pumping (Gailey 2017).

While this work focuses on a specific region, the results indicate that impacts from groups of wells may occur in other areas with similar, and fairly common, conditions (stratified alluvial sediments, irrigation water applied at ground surface, groundwater pumped from depth, poor shallow groundwater quality and long well screens). In these cases, geographic analysis may lead to more detailed evaluation in limited areas of concern. The potential benefits to groundwater quality may justify expenditures to develop the necessary data for performing the type of analysis demonstrated here.

Acknowledgements

The author is grateful to the California Department of Water Resources for sharing data regarding the geographic distribution of water supply well construction details and the Central Valley Salinity Coalition for sharing GIS layers regarding groundwater quality. Jay Lund, Graham Fogg, Thomas Harter and three anonymous reviewers are also appreciatively acknowledged for editorial and technical comments that improved this manuscript. This work was supported by the UC Office of the President's Multi-Campus Research Programs and Initiatives (MR-15-328473) through UC Water, the University of California Water Security and Sustainability Research Initiative and by the S.D. Bechtel Jr. Foundation through the UC Davis Center for Watershed Sciences.

Appendix

The factor for down-scaling vertical gradients across the aquitard to use in calculating flow along well casings (see 'Methods of analysis, Evaluating geographic information' item 7b) is developed through analytical modeling. The analysis is developed similar to Silliman and Higgins (1990) with two exceptions: 1) higher head occurs in the upper aquifer so that flow is downward through the well and 2) head loss that results from flow along the well casing is addressed separately from losses that occur through the well screens. The approach is briefly described here with additional details available in Silliman and Higgins (1990). Physical relationships associated with the mathematical development are shown on Figure 28.

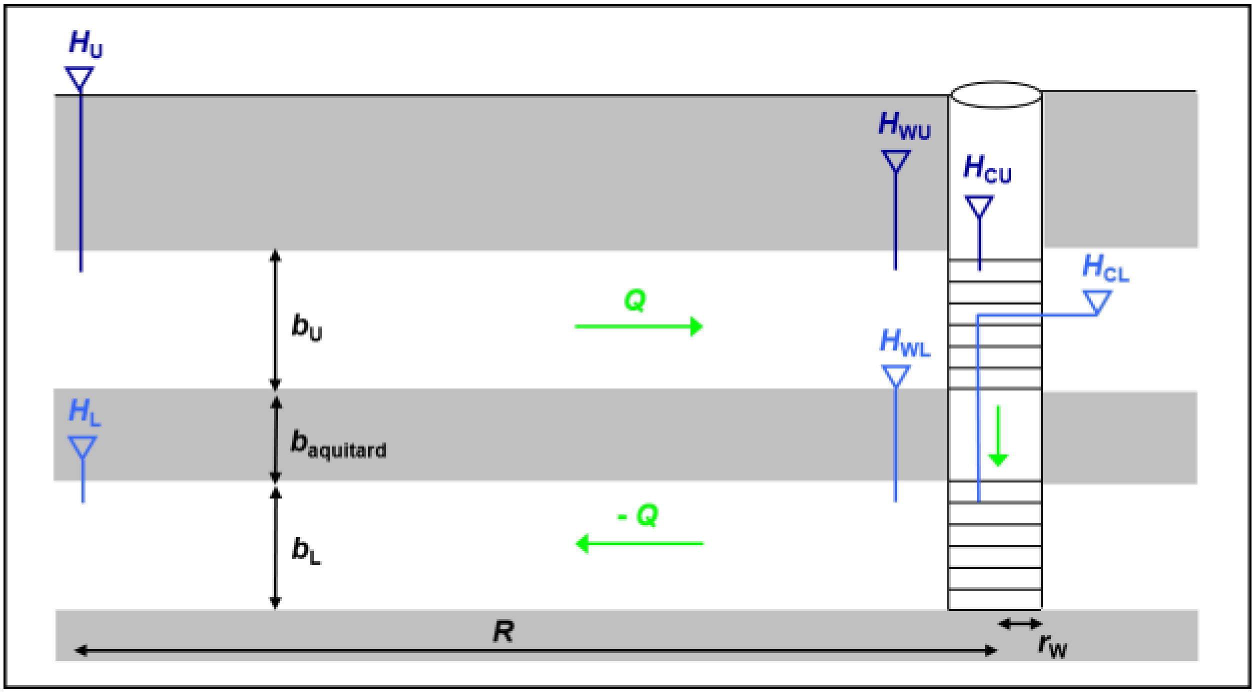


Figure 28: Physical relationships between parameters used to estimate gradient down-scaling factor. Figure not to scale

Confined Conditions

Steady-state groundwater flow to a well under confined conditions may be expressed by the Thiem equation (Bear 1979):

$$H(R) - h(r_w) = [Q/(2\pi T)] \ln(R/r_w) \quad (8)$$

Where $H(\mathrm{R})$ is the head at radial distance R, hereafter referred to as H

$h(r_w)$ is the head at radial distance r_w, hereafter referred to as H_W

Q is the exchange rate of groundwater between the aquifer and the well defined as greater than zero when groundwater flows into well

T is the transmissivity which is further defined as $T = K\, b$

K is the aquifer hydraulic conductivity

b is the aquifer thickness

R is the radius of influence, or the distance from the well beyond which flow in the well does not affect head in the aquifer

r_w is the well radius

Equation (8) is reformulated by rearranging, consolidating variables and applying the subscript U to designate application to the upper aquifer:

$$Q = 2\,\pi\,\alpha_U\,(H_U - H_{WU}) \tag{9}$$

Where $\alpha_U = T_U/\ln(R_U/r_w)$

Equation (9) is reformulated for application to the lower aquifer as equation (11) by applying equation (10) for mass balance and using subscript L to indicate the lower aquifer:

$$Q = Q_U = -Q_L \tag{10}$$

$$Q = 2\,\pi\,\alpha_L\,(H_{WL} - H_L) \tag{11}$$

Head loss that results from flow along the well casing is expressed as:

$$HL = H_{CU} - H_{CL} \tag{12}$$

Where HL is head loss in the casing

H_{CU} is head in the well casing adjacent to the upper aquifer

H_{CL} is head in the well casing adjacent to the lower aquifer

Head losses that occur through the well screens are expressed as:

$$WL_U = H_{WU} - H_{CU} \tag{13}$$

$$WL_L = H_{CL} - H_{WL} \tag{14}$$

Where WL_U is head loss through the upper well screen

H_{WU} is head in the upper aquifer at r_w

WL_L is head loss through the lower well screen

H_{WL} is head in the lower aquifer at r_w

Combining equations (11) – (14) and rearranging results in an expressions for H_{WU}. This expression enumerates the head losses along the flow path between H_L to H_{WU}.

$$H_{WU} = H_L + Q/(2\,\pi\,\alpha_L) + \mathrm{WL}_L + \mathrm{HL} + \mathrm{WL}_U \tag{15}$$

The casing and screen head loss components are represented in terms of the flow rate in equations (12) – (18). HL is represented as in Silliman and Higgins (1990) while the WL terms are represented separately as in de Marsily (1986). These flows can be turbulent (e.g., Gailey 2017).

$$\mathrm{HL} = C\,Q^2 \tag{16}$$

$$\mathrm{WL}_U = A_U\,Q + B_U\,Q^2 \tag{17}$$

$$\mathrm{WL}_L = A_L\,Q + B_L\,Q^2 \tag{18}$$

Substituting equations (16) – (18) into equation (15) and then the resulting expression into equation (9) followed by rearrangement produces a quadratic equation:

$$a\,Q^2 + b\,Q + c = 0 \tag{19}$$

Where $a = (B_U + B_L + C)\,(2\,\pi\,\alpha_L\,\alpha_U)/(\alpha_L + \alpha_U)$ (20)

$$b = [(A_U + A_L)\,(2\,\pi\,\alpha_L\,\alpha_U)/(\alpha_L + \alpha_U)] + 1 \tag{21}$$

$$c = -(H_U - H_L)\,(2\,\pi\,\alpha_L\,\alpha_U)/(\alpha_L + \alpha_U) \tag{22}$$

The flow rate is then found using the quadratic formula:

$$Q^* = [-b + (b^2 - 4\,a\,c)^{1/2}\,]/(2\,a) \tag{23}$$

The factor for down-scaling is then calculated with equation (24) as the ratio of the gradient in the well to the gradient across the aquitard. The gradient in the well is based on 1) an estimate of flow in the well under fairly realistic conditions with the possibility of turbulent flow (equation 23) and 2) a common calculation for effective hydraulic conductivity under the assumption of laminar flow (equation 25).

$$i_{well}/i_{aquitard} = [Q^*/(K_{well}\,A_{well})]/[(H_U - H_L)/b_{aquitard}] \tag{24}$$

Where i_{well} is the vertical hydraulic gradient in the well

$i_{aquitard}$ is the vertical hydraulic gradient across the aquitard

$b_{aquitard}$ is the thickness of the aquitard between the upper and lower aquifers

K_{well} is the effective hydraulic conductivity of the well

A_{well} is the cross sectional area of the well further defined as $\pi\, r_w^2$

The effective hydraulic conductivity of the well is estimated with the Hagen-Poiseuille equation (Gailey 2017):

$$K_{well} = r^2\rho g/8\mu \tag{25}$$

Where r is the inner radius of the well casing or screen

ρ is the groundwater density

g is the gravitational acceleration

μ is the groundwater viscosity.

The value calculated with equation (24) is substituted for f_{scale} in equation (7) of item 7b in 'Methods of analysis, Evaluating geographic information' for estimation of conduit flows. Directly substituting the expression in equation (24) for f_{scale} in equation (7) of item 7b and simplifying reveals that the hydraulic conductivity terms (K_{well}), which involve the assumption of laminar flow, cancel out of the final expression for estimated flow rate through the well:

$$Q_{well\_est} = [(Q^* b_{aquitard})/(H_U - H_L)]\ i_{clay} \quad (26)$$

Therefore, there is no potential conflict in assuming turbulent flow when calculating Q^* and laminar flow when calculating K_{well}.

Unconfined Conditions

For unconfined conditions, Dupuit's solution is used (Silliman and Higgins 1990; de Marsily 1986) and equation (9) is replaced with:

$$Q = \pi\ \alpha_U\ (H_U^2 - H_{WU}^2) \quad (27)$$

Where $\alpha_U = K_U/\ln(R_U/r_w)$

Conservation of mass is expressed by combining equations (27) and (11):

$$\pi\, \alpha_U\, (H_U^2 - H_{WU}^2) = 2\, \pi\, \alpha_L\, (H_{WL} - H_L) \tag{28}$$

Substituting the head loss terms from equations (12) – (14) for H_{WL} and expressing in terms of flow rate according to equations (16) – (18):

$$\pi\, \alpha_U\, (H_U^2 - H_{WU}^2) = 2\, \pi\, \alpha_L\, [H_{WU} - (A_U + A_L)\, Q - (B_U + B_L + C)\, Q^2 - H_L] \tag{29}$$

Substituting equation (27) for terms in Q and rearranging results in a fourth-order polynomial in H_{WU}:

$$D_1\, H_{WU}^4 + D_2\, H_{WU}^2 + D_3\, H_{WU} + D_4 = 0 \tag{30}$$

Where

$$D_1 = -2\, \pi^3\, \alpha_U^2\, \alpha_L\, (B_U + B_L + C) \tag{31}$$

$$D_2 = 2\, \pi^2\, \alpha_U\, \alpha_L\, (A_U + A_L) + 4\, \pi^3\, \alpha_U^2\, \alpha_L\, (B_U + B_L + C)\, H_U^2 + \pi\, \alpha_U \tag{32}$$

$$\mathrm{D}_3 = 2\, \pi\, \alpha_L \tag{33}$$

$$\mathrm{D}_4 = -2\, \pi^2\, \alpha_U\, \alpha_L\, (A_U + A_L)\, H_U^2 - 2\, \pi^3\, \alpha_U^2\, \alpha_L\, (B_U + B_L + C)\, H_U^4 - 2\, \pi\, \alpha_L\, H_L - \pi\, \alpha_U\, H_U^2 \tag{34}$$

Solving for H_{WU} and substituting into equation (27) provides an estimate for flow rate. Equations (24) and (25) in 'Confined conditions' can then be used to calculate the factor for down-scaling.

Estimating the Down-Scaling Factor Value

The downscaling factor from equation (24) was estimated for confined conditions using a reasonable set of parameter values for the study area. Sensitivity analysis was performed by adjusting the parameter values and also considering unconfined conditions. Table 7 summarizes the parameter values used for the base case.

Table 7 Parameter values for the base case

Parameter	Units	Upper Aquifer	Lower Aquifer	Well Loss
H	m	100	50	-
K	m/s	10^{-5}	10^{-5}	-
b	m	60	100	-
T	m^2/s	6×10^{-4}	10^{-3}	-
b_{aquitard}	m	30	30	-
R	m	40	40	-
r_{w}	m	0.2	0.2	-
A_{U}	s/m^2	-	-	100
A_{L}	s/m^2	-	-	100
B_{U}	s^2/m^5	-	-	100
B_{L}	s^2/m^5	-	-	100
e	m	-	-	2.5×10^{-2}
f	-	-	-	0.08
C	s^2/m^5	-	-	17.8

Consistent with the possibility that turbulent flow may occur inside the well casing, the value for head loss in the well casing (C in equation 16) was developed using the Darcy-Weisbach equation (Vennard and Street 1982):

$$\text{HL} = f\,[L/(4\,\pi^2\,g\,r_{\text{w}}^5)]\,Q^2 \tag{35}$$

Where f is the friction factor

L is the length of well casing which is equal to b_{aquitard} in this case (Figure 28)

The friction factor was estimated by implicit solution of the Colebrook equation (Vennard and Street 1982):

$$1/f^{1/2} - 2\log(r_w/2e) = 1.14 - 2\log[1 + 9.28/(\mathrm{Re}\,(2e/r_w)\,f^{1/2})] \qquad (36)$$

Where e is mean height of casing roughness

Re is the Reynolds number defined as follows (Vennard and Street 1982):

$$\mathrm{Re} = (2\,\rho\,Q)/(\pi\,\mu\,r) \qquad (37)$$

The head profiles for upper and lower aquifers using base case parameter values (Figure 29) illustrate the effects of head losses from radial flow, friction in the well screens and friction in the casing. The sensitivity results for confined conditions (Table 8) and the base case result for unconfined conditions (2.3×10^{-6}) indicate that the down-scaling factor is not highly sensitive to reasonable variations in the parameter values. Given that semiconfined conditions exist in the study area, a value between the confined and unconfined base case results was used (2.0×10^{-6}).

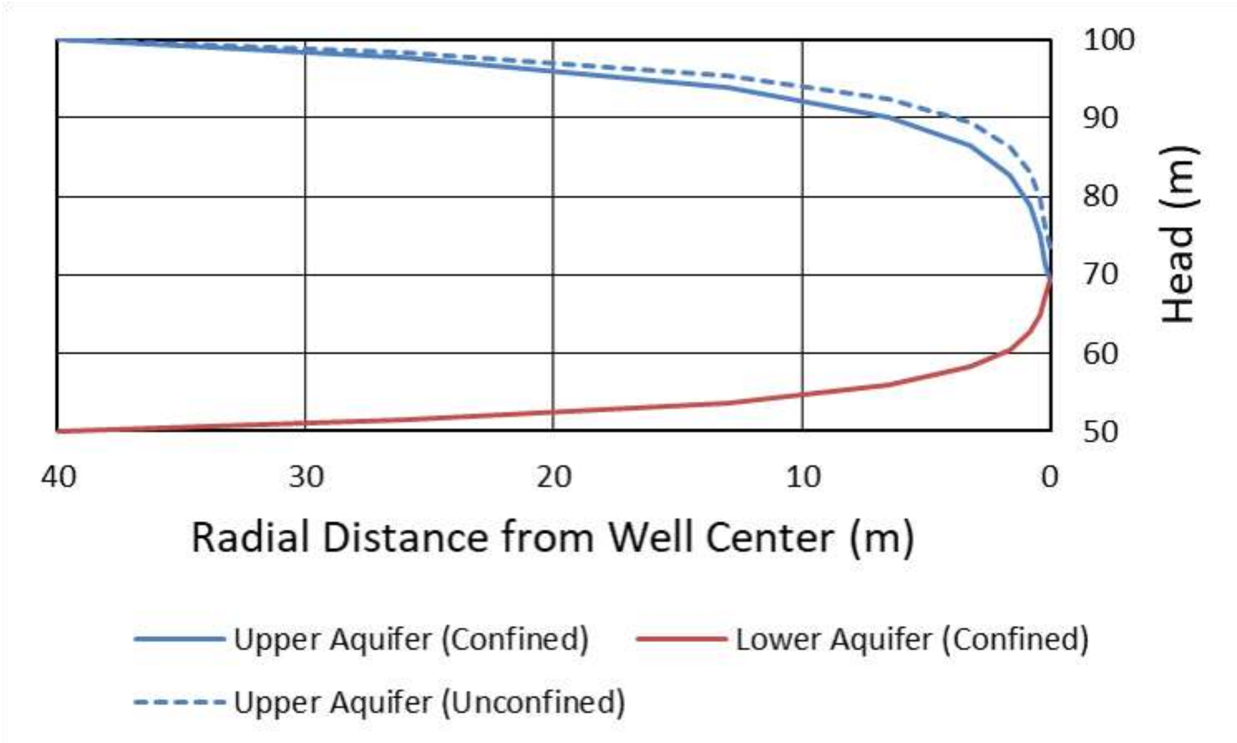


Figure 29: Calculated aquifer head profiles

Table 8 Sensitivity analysis for $i_{well}/i_{aquitard}$ under confined conditions

Parameter	T_U	T_L	R	$b_{aquitard}$	f	A	B
Base Case	1.9×10^{-6}	1.9×10^{-6}	1.9×10^{-6}	1.9×10^{-6}	1.9×10^{-6}	1.9×10^{-6}	1.9×10^{-6}
Increased Result	3.8×10^{-6}	2.7×10^{-6}	1.7×10^{-6}	3.7×10^{-6}	1.9×10^{-6}	1.1×10^{-6}	1.8×10^{-6}
Decreased Result	3.0×10^{-7}	4.6×10^{-7}	2.1×10^{-6}	9.4×10^{-7}	1.9×10^{-6}	2.0×10^{-6}	1.9×10^{-6}

Notes: Factors used to increase parameter values: T_U: 10, T_L: 10, R: 2, $b_{aquitard}$: 2, f: 10, A: 10 and B: 10

Factors used to decrease parameter values: T_U: 0.1, T_L: 0.1, R: 0.5, $b_{aquitard}$: 0.5, f: 0.1, A: 0.1 and B: 0.1

References

Bear J (1979) Hydraulics of groundwater. McGraw-Hill, New York, NY

Belitz K and Heimes FJ (1990) Character and evolution of the ground-water flow system in the central part of the western San Joaquin Valley, California, US Geol Surv Water-Supply Pap 2348

CADWR (1981) Depth to the top of Corcoran Clay – 1981. http://www.water.ca.gov/pubs/groundwater/depth_to_top_of_corcoran_clay_map__1981/depth_to_the_top_of_corcoran_clay-1981.pdf Accessed 11 May 2017

CADWR (2016) Critically overdrafted groundwater basins – January 2016. http://www.water.ca.gov/groundwater/sgm/pdfs/GW_basinsCriticalOverdraft_CA.pdf Accessed 11 May 2017

CADWR (2017a) Groundwater level data reports. http://www.water.ca.gov/waterdatalibrary/groundwater/index.cfm Accessed 11 May 2017

CADWR (2017b) Well completion report map application. https://dwr.maps.arcgis.com/apps/webappviewer/index.html?id=181078580a214c0986e2da28f8623b37 Accessed 28 November 2017

Cantor A, Kiparsky M, Bales R, Hubbard S, Kennedy R, Pecharroman LC (2018) Data for water decision making: informing the implementation of California's Open and Transparent Water Data Act through research and engagement, Center for Law, Energy & the Environment, UC Berkeley School of Law, Berkeley, CA

Clark BR, Landon MK, Kauffman LJ and Hornberger GZ (2008) Simulations of Ground-Water Flow, Transport, Age, and Particle Tracking near York, Nebraska, for a Study of Transport of Anthropogenic and Natural Contaminants (TANC) to Public Supply Wells, US Geol Surv Sci Invest Rep 2007-5068

CVSC (2016) Central Valley salinity alternatives for long-term sustainability, Central Valley region salt and nitrate management plan, final document for Central Valley Regional Water Board consideration, December 2016. https://www.cvsalinity.org/docs/central-valley-snmp/final-snmp.html Accessed 11 May 2017

Davis GH, Lofgren BE and Mack S (1964) Use of ground-water reservoirs for storage of surface water in the San Joaquin Valley California, US Geol Surv Water-Supply Pap 1618

Faunt CC (ed.) (2009) Groundwater availability of the Central Valley Aquifer, California, US Geol Surv Prof Pap 1766

Faunt CC (2012) Contours of Corcoran Clay thickness in feet by Page (1986) for the Central Valley Hydrologic Model (CVHM), United States Geological Survey Water Resources NSDI Node. https://water.usgs.gov/GIS/metadata/usgswrd/XML/pp1766_corcoran_clay_thickness_feet.xml Accessed 11 May 2017

Gailey RM (2017) Inactive supply wells as conduits for flow and contaminant migration: conditions of occurrence and suggestions for management, Hydrogeol J 25(7):2163-2183, doi: 10.1007/s10040-017-1588-y

Hanak E, Lund J, Arnold B, Escriva-Bou A, Gray B, Green S, Harter H, Howitt R, MacEwan D, Meddellin_Azuara Moyle P and Seavey N (2017) Water stress and a changing San Joaquin Valley, Public Policy Institute of California, March 2017. http://www.ppic.org/main/publication.asp?i=1224 Accessed 11 May 2017

Jasechko S, Perrone D, Befus KM, Cardenas MB, Ferguson G, Gleeson T, Luijendijk E, McDonnell JJ, Taylor RG, Wada Y and Kirchner JW (2017) Global aquifers dominated by fossil groundwaters but wells vulnerable to modern contamination, Nature Geoscience 10(6):425-429, doi:10.1038/ngeo2943

Jurgens BC, Bexfield LM and Eberts SM (2014) A ternary age-mixing model to explain contaminant occurrence in a deep supply well, Ground Water 52(S1):25–39, doi: 10.1111/gwat.12170

Landon MK, Jurgens BC, Katz BG, Eberts SM, Burow KR and Crandall CA (2009) Depth-dependent sampling to identify short-circuit pathways to public supply wells in multiple aquifer settings in the United States, Hydrogeol J 18(3):577-593, doi: 10.1007/s10040-006-0148-7

De Marsily G (1986) Quantitative hydrogeology, groundwater hydrology for engineers. Academic Press, San Diego, CA

Mayo AL (2010) Ambient well-bore mixing, aquifer cross-contamination, pumping stress and water quality from long-screened wells: What is sampled and what is not?, Hydrogeol J 18:823-837, doi: 10.1007/s10040-009-0568-2

Moore E, Matalon E, Balazs C, Clary J, Firestone L, De Anda S and Guzman M (2011) The human costs of nitrate-contaminated drinking water in the San Joaquin Valley, Pacific Institute, Oakland, CA

Page RW (1986) Geology of the fresh ground-water basin of the Central Valley, California, with textural maps and sections, US Geol Surv Prof Pap 1401-C. https://pubs.usgs.gov/pp/1401c/report.pdf Accessed 11 May 2017

Perrone D and Jasechko S (2017) Dry groundwater wells in the western United States, Environmental Research Lett 12:104002

Phillips SP and Belitz K (1991) Calibration of a texture-based model of a ground-water flow system, western San Joaquin Valley, California, Ground Water 29(5):702-715, doi: 10.1111/j.1745-6584.1991.tb00562.x

Silliman S and Higgins D (1990) An analytical solution for steady-state flow between aquifers through an open well, Ground Water 28(2):184-190, doi: 10.1111/j.1745-6584.1990.tb02245.x

Vennard JK and Street RL (1982) Elementary fluid mechanics, 6th edn. John Wiley and Sons, New York, NY

Zuurbier KG and Stuyfzand PJ (2017) Consequences and mitigation of saltwater intrusion induced by short-circuiting during aquifer storage and recovery in a coastal subsurface, Hydrol Earth Syst 21:1173-1188, doi: 10.5194/hess-21-1173-2017

CHAPTER 4 - DOMESTIC WELL SUPPLY RELIABILITY: STRESS TESTING FOR GROUNDWATER OVERDRAFT AND ESTIMATING ECONOMIC COSTS

Abstract

Costs from the follow-on effects of groundwater depletion are not borne entirely by high-volume pumpers that largely create the problematic conditions. Ideally, sustainable groundwater management should include addressing economic externalities and social equity, although information quantifying these concerns is often unavailable. This work estimates economic costs to domestic water wells owners from groundwater overdraft, caused mostly by deeper agricultural wells. A study is presented for Tulare County in the southern Central Valley of California, USA where many interruptions in domestic well supplies occurred during a recent drought. The case has unusually well-documented conditions and data available for analysis. A method for evaluating impacts and costs of declining groundwater levels on domestic water supplies is developed in the context of planning for sustainable groundwater management and suggestions are made for application to other areas where less data may be available. This work may provide a basis for evaluating the economic impacts on well owners from over-pumping in basins outside the study area.

Introduction

Although it has long been understood that groundwater pumping can negatively affect aquifer systems and surface water bodies (e.g., Theis 1940; Jenkins 1969; Sophocleous 2000; Bredehoeft 2002), society has not sustainably managed groundwater extraction and depletion is occurring in many parts of the globe (Wada et al. 2012; Döll et al. 2014) including the United States (Konikow 2013) and California (Famiglietti et al. 2011; Farr et al. 2015). The negative and long-lasting results of groundwater overdraft include decreased water levels, reduced surface water flows, seawater intrusion, spreading of contaminants, and land subsidence (Foster and Chilton 2003; Moran et al. 2014; USGS 2017). When groundwater levels decrease over long periods, pumping lifts and costs increase, and wells go dry unless pumps are lowered and wells are deepened or replaced at significant expense (Gailey et al. 2016). Surface water depletion from lower groundwater levels can lead to water rights infringement, agricultural irrigation and municipal supply shortages, ecosystem degradation (Barlow and Leake 2012) and, in many cases, additional groundwater pumping to address lost surface water supplies (Glennon 2002). Water quality degradation occurs as seawater flows into coastal aquifers (Barlow and Reichard, 2010), contaminant plumes migrate horizontally away from sources, degraded quality shallow groundwater flows to deeper water-bearing zones (Gailey 2017 and 2018), and poorer quality groundwater emerges from aquitards (Smedley and Kinniburgh 2002; Erickson and Barnes 2005). Land subsidence causes damage to infrastructure (canals, wells, roads), interferes with the designed gravity flow in canals and reduces groundwater storage (Sneed et al. 2013). Many effects from groundwater overdraft reduce the reliability of both sole-source groundwater supplies and combined water supplies (surface water plus increased use of groundwater as a backstop in times of drought).

Costs from the follow-on effects of groundwater depletion are not borne entirely by high-volume pumpers that largely create the problematic conditions. The economic and public health externalities occur because subsurface flow is not constrained by property lines marked at ground surface, and pumping from a well location draws down groundwater levels at other locations

(Ostrom 1990; Stevenson 1991; Young and Loomis 2014). Social equity also may be a concern when parties that extract much of the resource are less sensitive to increased costs than others who are forced to bear additional costs from overuse (Foster and Chilton 2003; Griffin 2006). This point is clear when considering that groundwater is an important domestic supply for billions of people around the world (UNESCO 2012). Ideally, sustainable groundwater management should address economic externalities and social equity, although information quantifying these concerns is often unavailable.

This work estimates economic costs to domestic water well owners from groundwater overdraft, caused mostly by deeper agricultural wells, for Tulare County in the southern Central Valley of California, USA (Fig. 30) where interruptions in domestic well supplies occurred during a recent drought. The case has unusually well-documented conditions and data available for analysis. A method for evaluating impacts and costs of declining groundwater levels on domestic water supplies is developed within the context of planning for sustainable groundwater management. Suggestions are made for application to other areas where less data may be available. This work may provide a basis for evaluating the economic impacts on well owners from over-pumping in basins outside the study area.

Figure 30: Study area in California, USA. *Gray shaded area* indicates portion of Tulare County located on floor of the Central Valley (study area)

Study Area and Background

California's southern Central Valley supports intensive agriculture which relies heavily on groundwater for irrigation, especially during dry years (Hanak et al. 2017). The groundwater system is extensive with the shallower part comprising an interfingered assemblage of alluvial and flood-basin deposits approximately 1,000 m thick (Faunt 2009). Substantial pumping over many decades resulted in groundwater overdraft and large water level decreases during the mid-twentieth century (Faunt 2009; Hanak et al. 2017). Groundwater declines were reversed by importing surface water for irrigation; however, subsequent long-term reductions in the imported surface water, exacerbated during drought, allowed overdraft to return (Faunt 2009; CADWR 2015; Hanak et al. 2017).

The recent historic drought in California from 2012 through 2016 brought unprecedented groundwater level declines (CADWR 2014a) and reports of dry domestic wells (State of California 2017). Consistent with findings by Perrone and Jasechko (2017), domestic wells likely were impacted more than agricultural wells because of shallower completion depths. The part of Tulare County, California on the valley floor experienced many reported domestic supply well outages during the drought (County of Tulare 2017; Hanak et al. 2017) with many outages in economically disadvantaged communities (Feinstein et al 2017). This area, located in the southeastern Central Valley (Fig. 1), is the focus of this study.

Agricultural production in Tulare County largely coincides with the study area. In the 20 years from 1997 through 2016, land in crop production increased from 1.5 to 1.8 million ac (610 to 720 thousand ha) and total annual agricultural revenue rose from \$2.9 to \$8.1 billion (Tulare County Agricultural Commissioner 1998 through 2017) with acceleration in the second half of this period. Consistent with increasing market prices for agricultural products during these 20 years, inflationadjusted revenue increased over 50 percent, with a peak of almost 90 percent during the recent drought, while land in production increased less than 20 percent. Much of the total revenue is based on demand for agricultural products that involve multi-year investments and require relatively constant water supplies. Fruits and nuts (i.e., citrus, grapes and almonds) are harvested from orchards and vineyards that produce at economically viable levels for decades. Likewise, beef cattle and dairies entail multi-year investments and largely rely upon wet roughage that is locally grown because of transportation costs and required high water content upon delivery. Trees, vines, cattle, and dairies accounted for 85 to 90 percent of county agricultural revenues in recent years (Tulare County Agricultural Commissioner 2012 through 2017). The combined high market prices for agricultural products and the mix of production activities in the study area has hardened demand for water as an input to production.

Institutions in the southern Central Valley and elsewhere in California are beginning to plan for compliance with new state requirements for sustainable groundwater management (CADWR 2018). The new regulations require including a wide range of stakeholder concerns in the planning process. In Tulare County, balancing agricultural pumping with domestic supply reliability during times of drought will likely be an important consideration. Although others have developed information related to the prevalence of domestic wells in California (Johnson and Belitz 2015 and 2017), no work to date appears to provide analysis that supports factoring protection of domestic wells into regional groundwater planning.

Methods of Analysis and Data Sources

Methods of Analysis

The impact of declining groundwater levels on domestic supply well operations is evaluated by comparing groundwater depths to levels required for well pump operation. This is done for each year of the recent drought on a grid across the study area. Supply outage is considered to occur if the groundwater level drops below that required for pump operation at the time the drought begins. The external economic costs from supply outages are estimated by simulating corrective actions required to avoid supply interruption (energy increase for added lift, pump lowering, well screen cleaning and well replacement) and summing these estimated costs for all wells. A well operations impact and cost estimation model is calibrated against observed supply outages, uncertainty in model predictions is assessed and simple future drought scenarios are evaluated.

Impact and Cost Estimation Model

Based on improvements to a model developed by Gailey et al. (2016), impacts to domestic well operations and costs from water level decline is developed from information on well construction details, groundwater levels and other pertinent information for the study area (see 'Data sources' below). Four data processing steps combine output from geographic information system (GIS) software and spreadsheets with macro scripts to estimate impacts (Fig. 31).

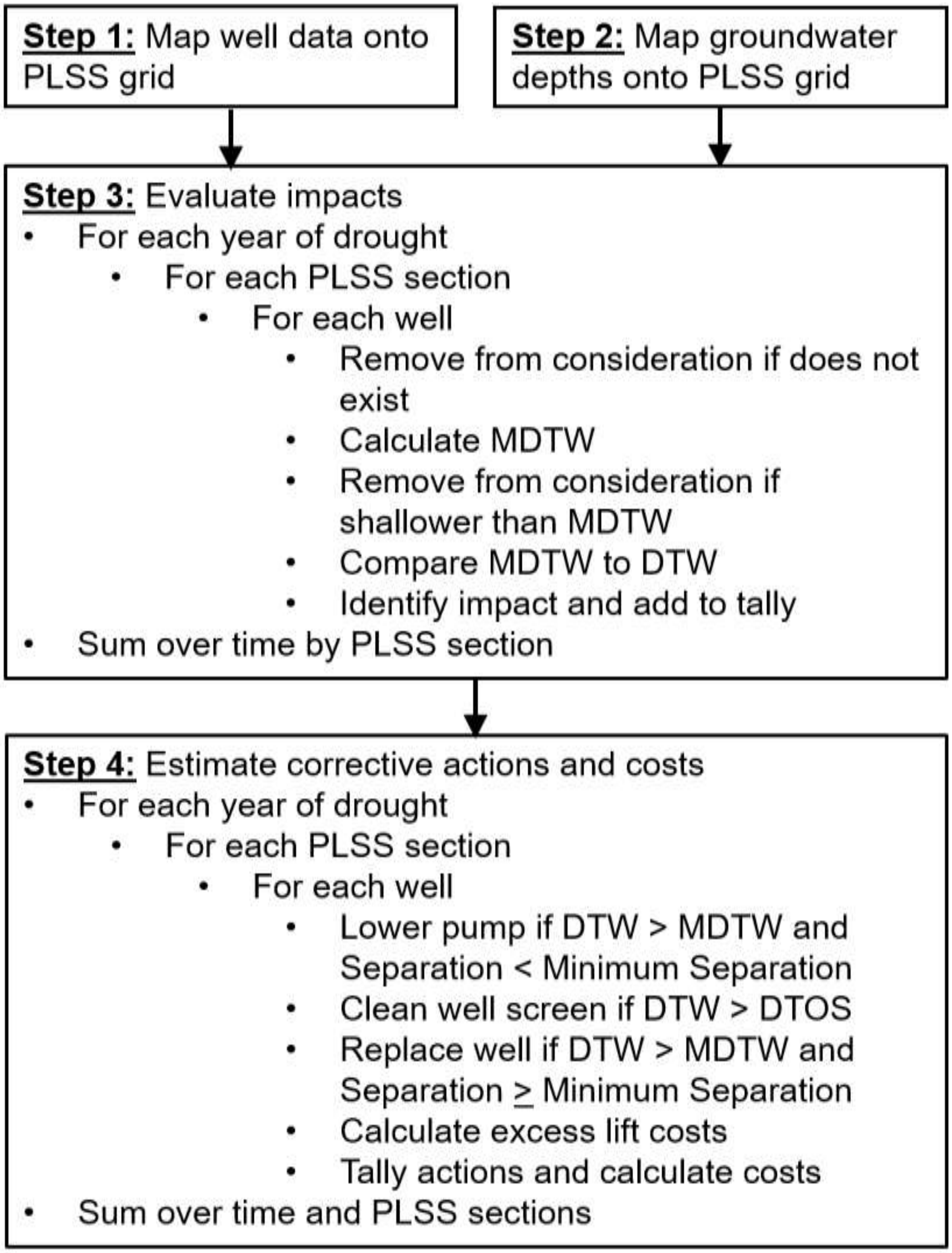


Figure 31: Work flow for data evaluation. PLSS is Public Land Survey System. MDTW is maximum depth to water. DTW is depth to water. DTOS is depth to top of well screen

1) Well data preparation: The historic inventory of domestic well construction data (year constructed, depths to tops of well screens and bottoms of wells, and locations) are aggregated and mapped on the US Public Land Survey System (PLSS) grid. The State of California limits well location information to this 1 mi^2 (2.6 km^2) grid to protect well owner privacy, limiting the maximum horizontal resolution of this study to 1 mi^2.

2) Groundwater depth data preparation: Depths to groundwater are mapped onto the PLSS sections. This is done for each autumn, when water depths are typically the greatest at the end of the dry growing (irrigation/pumping) season, starting the year before and extending through the drought (2011 through 2016). Contoured representations of the water levels, available as GIS polyline shapefiles, are rasterized and converted to GIS polygon shapefiles with values for each PLSS grid cell.

3) Well operations impact evaluation: For each autumn from 2011 through 2016, the depth to groundwater and well construction details are evaluated for each PLSS section in the study area where all the necessary data are available. Conceptually, the impacts are assessed by dividing the distribution of wells for each PLSS section into impacted and non-impacted categories based on a depth metric (the Maximum Depth to Water discussed below) and the depth-to-water (Fig. 32). The analysis steps are:

    a. Wells are excluded from the analysis if 1) year the well was constructed after the drought year being considered and 2) the well age exceeds the retirement age. Retirement age is a calibrated parameter discussed below.

b. Estimate the maximum depth to water (MDTW) in each well. This screening criterion is calculated as the depth to well bottom minus an adjustment factor that accounts for pump operational water depth requirements. The adjustment factor is the sum of pumping drawdown, pump submergence and separation distance of pump from the bottom of well (Figure 33). The pumping drawdown is the product of pumping rate and specific capacity. Values for the pumping rate, specific capacity and pump submergence included in the adjustment factor are specified in Table 9 based on experience of the author. The separation distance, a calibrated parameter discussed below, may be large given the extra cost to position pumps deeper in wells than required to maintain pump submergence. It is also expected to have some minimum (non-zero, positive) value based on the common inability to place pumps at the bottoms of wells because of sediment accumulation. The separation distance value is updated for each well as the water level declines and the pump is lowered in subsequent years of drought as described below.

c. Eliminate wells where the adjustment factor used to estimate MDTW exceeds the well depth, indicating the well is too shallow to be used for current conditions. The well is assumed to have been abandoned at some point in the past. This step helps account for a lack of available well abandonment records.

d. For each well, if the depth to water in the groundwater system (DTW) exceeds MDTW, the water level in the well is too low for pump operation and there is an impact. A well is recorded as impacted the first time the water level drops below that required for pump operation. All such impacts during the drought are summed for each PLSS section and the sum over all PLSS sections is an estimate of well outages for the area.

Table 9 Model parameter values

Parameter	Value
Well retirement age	Calibrated value
Well pumping rate	10 gallons per minute (38 liters per minute)
Well specific capacity	5 gallons per minute per foot (62 liters per minute per meter)
Pump submergence	5 ft (1.5 m)
Pump separation from well bottom	Calibrated value (minimum of 20 ft or 6 m)

4) Corrective action cost estimation: For each autumn during the drought, DTW is compared to each of three screening conditions (listed on Fig. 31) for each PLSS section where the necessary data are available. The corrective actions needed for continued well operation are tallied and costs are calculated. Figures 5 a through c show the conditions that would result in each corrective action. Unit costs for the calculations (Table 10) are based on interviews with drillers and experience of the first author.

    a. Pump lowering is needed when DTW exceeds MDTW and there is enough separation distance to lower the pump. Lowering occurs in increments of 20 ft (6 m), the standard length of discharge piping, and repeated each year needed as long as the minimum separation distance from the pump to the well bottom (taken here as 20 ft or 6 m) is maintained. Separation is reduced by the amount the pump is lowered (Fig. 34a). Pump depth is initially estimated as the depth-to-water plus the pumping drawdown, required pump submergence and an additional operating margin (Fig. 34a). The operating margin (taken here as 20 ft or 6 m) reduces the frequency of required pump depth adjustments.

b. Well screen cleaning, also called rehabilitation, is considered necessary when DTW exceeds the depth to top of the well screen (DTOS, Fig. 34b). While well screen clogging may occur even when screens are fully submerged, exposure is likely to accelerate physical, geochemical and biological clogging of the well screen and require cleaning (Mansuy 1999; Houben and Treskatis 2007; Smith and Comeskey 2010; van Beek 2012). This corrective action is assumed to be done once during the drought period but is likely a deferred maintenance item.

c. Well failure occurs and replacement is needed whenever DTW exceeds MDTW and there is inadequate separation distance to lower the pump (Fig. 34c). A replacement well is assumed to be 100 ft (30 m) deeper than the original well. No additional corrective actions are considered after a well has been replaced.

5) <u>Increased energy cost estimation:</u> Costs from added pump lift as groundwater levels decline during the drought are calculated for each well and summed across both the study area and over the drought period. The pumping cost estimated for the year before the drought (2011) is used as a baseline and costs in later years that exceed the baseline cost are taken as economic costs from overdraft. The unit cost for additional pumping lift is presented in Table 10 and details of the lift cost calculations are presented in the Appendix.

Table 10 Unit costs

Action	Unit	Cost
Pump lowering	20 ft (6 m)	$ 2,000
Well screen rehabilitation	Well	$10,000
Well replacement	1 ft (0.3 m)	$ 100
Increased lift energy	1 ft (0.3 m)	$ 0.32

Lift energy is based on supplying enough water for a family of four for one year. See appendix for details

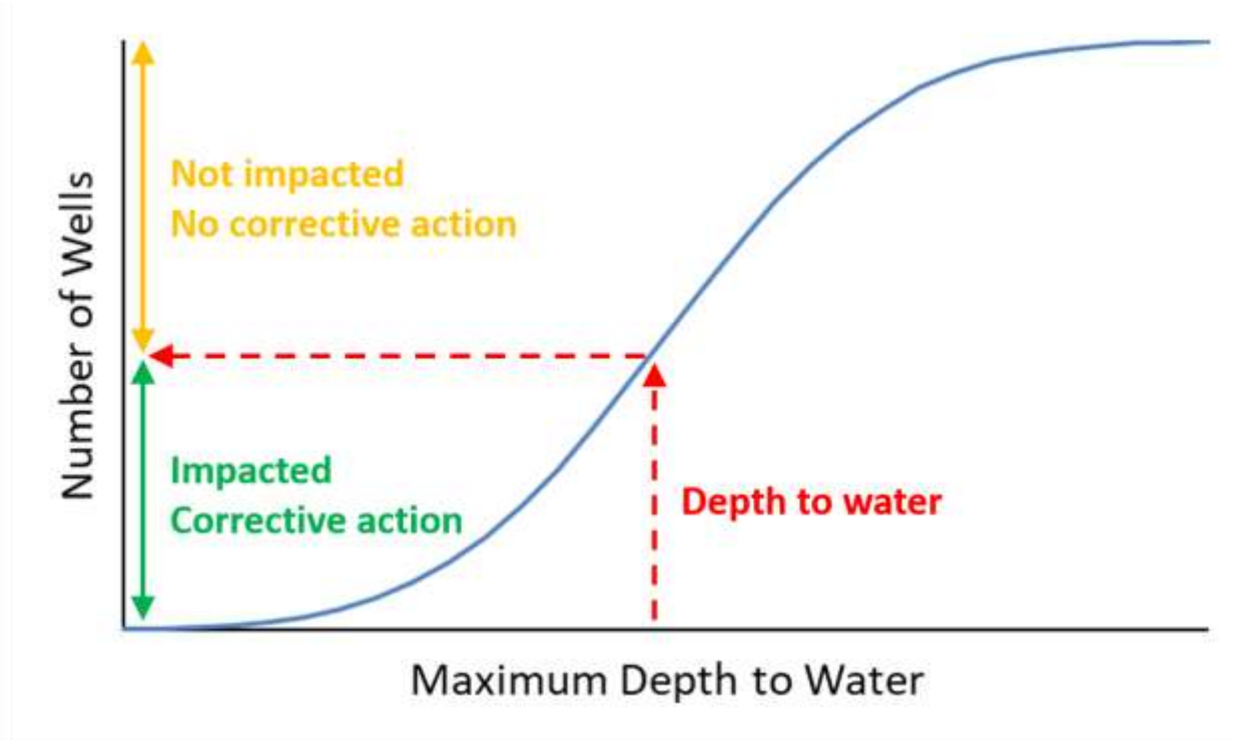


Figure 32: Conceptual representation of impacts and corrective action evaluation

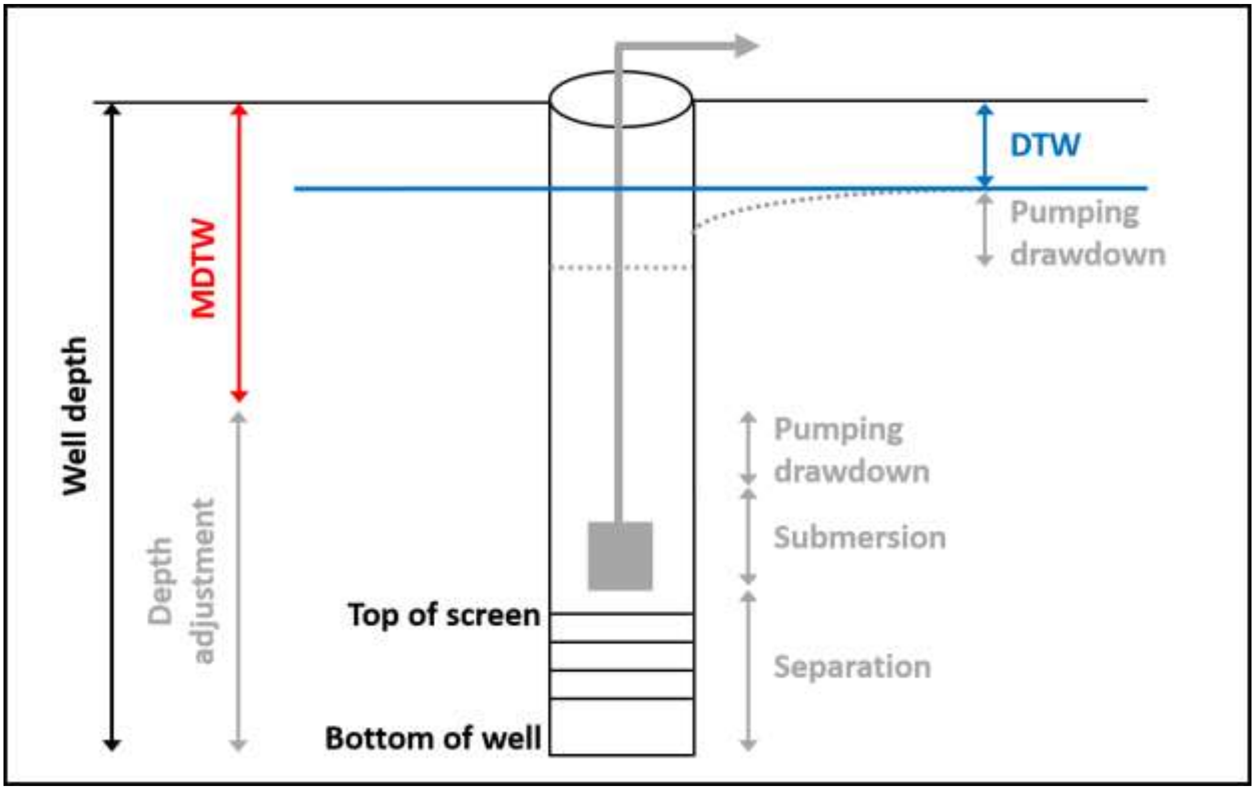
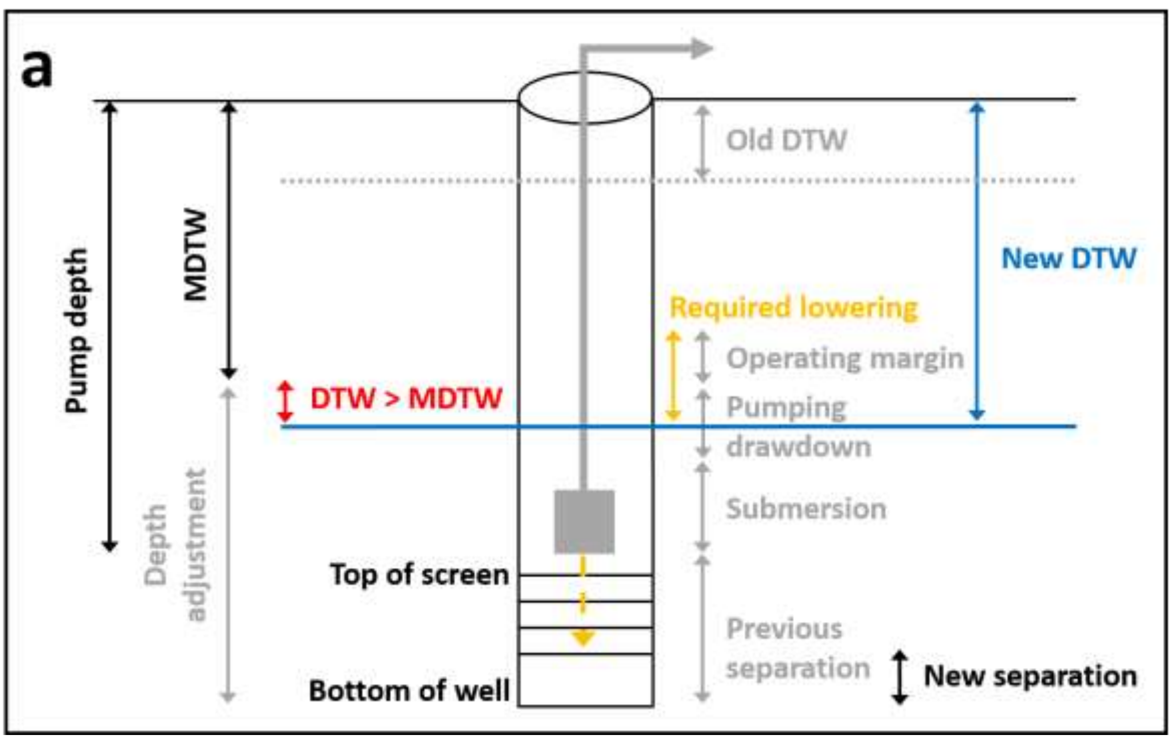
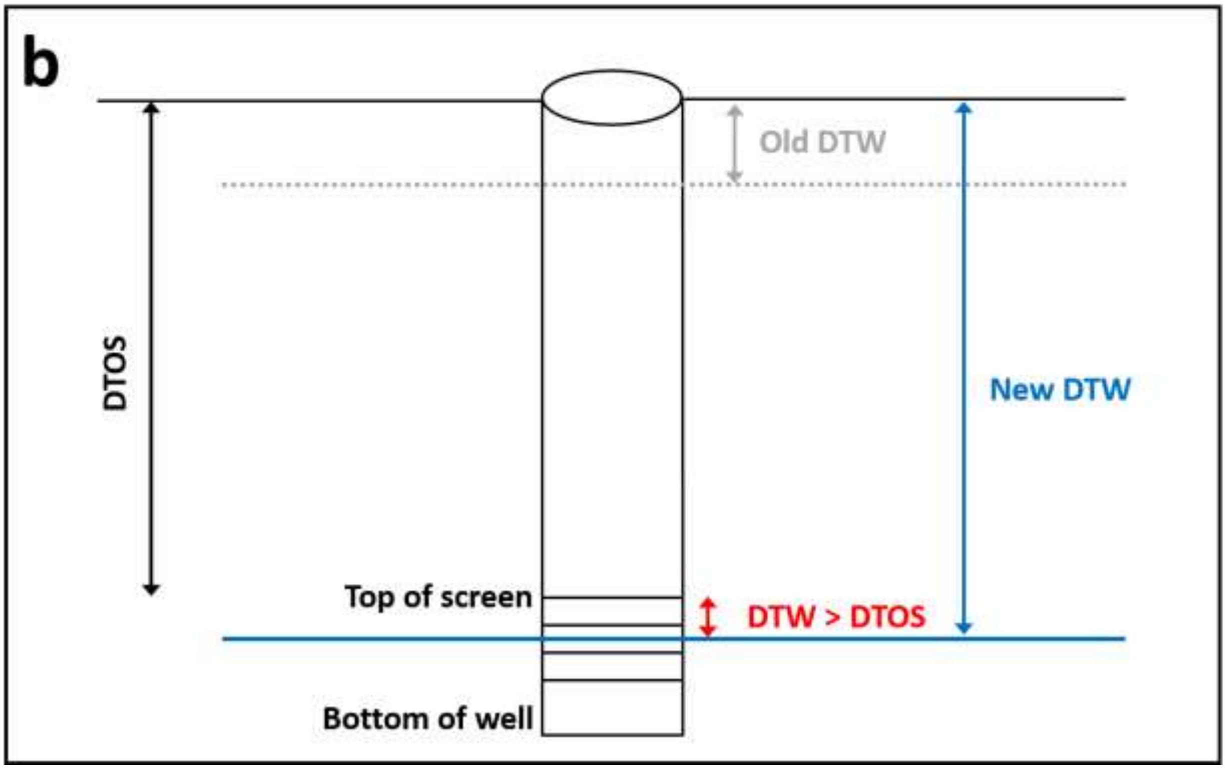


Figure 33: Maximum depth to water (MDTW) calculation shown relative to depth to water (DTW)




b
Old DTW
DTOS
New DTW
Top of screen
DTW > DTOS
Bottom of well

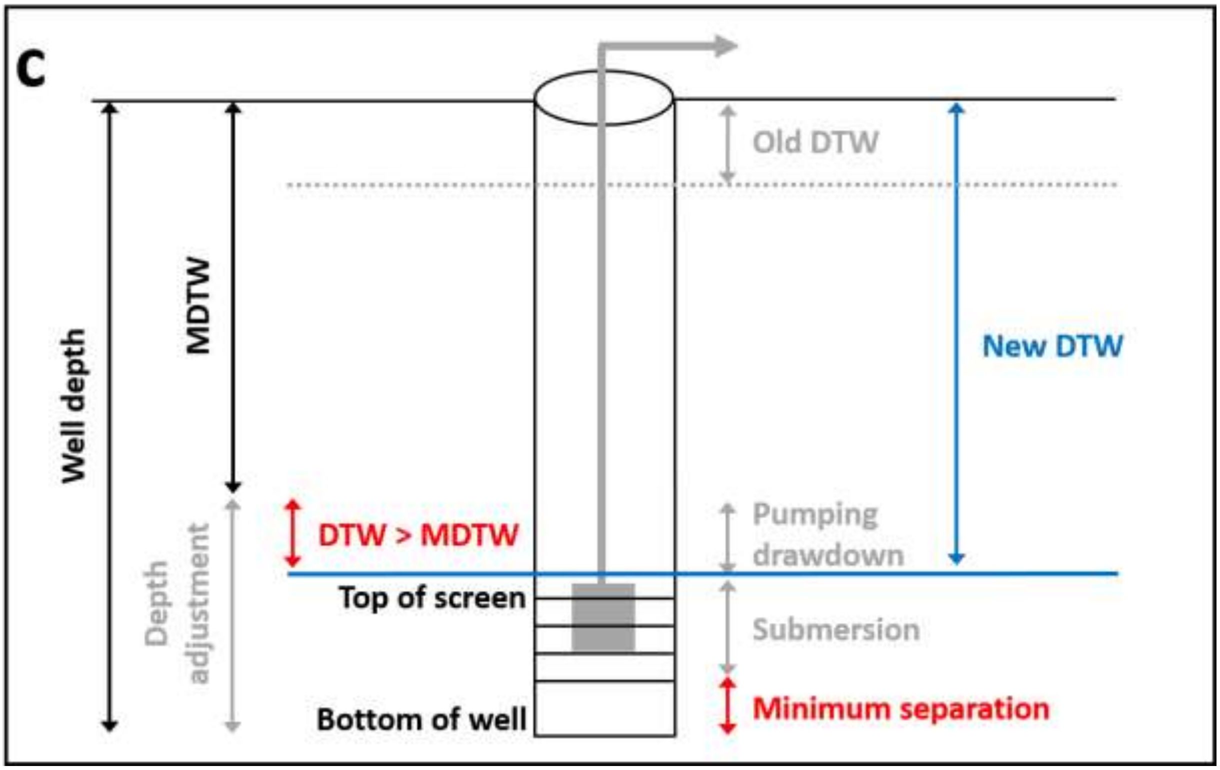


Figure 34: Conditions resulting in corrective actions: **a** pump lowering when depth to water (DTW) is greater than maximum depth to water (MDTW) and there is room to lower the pump, **b** well screen rehabilitation when DTW is greater than depth to top of screen (DTOS) and **c** well failure and replacement when DTW is greater than MDTW and there is no room to lower the pump (condition of minimum separation exists)

Model Calibration

Information needed for the impacts estimation steps are: 1) depths to tops of each well screen, 2) well depths, 3) depth to groundwater in each PLSS section for each autumn during the drought, 4) well retirement age, 5) well pumping rate, 6) well specific capacity, 7) pump submergence and 8) pump separation from the well bottom. As indicated below ('Data sources'), data are available for the depths of well screen tops, well bottoms and groundwater. The remaining variables are regarded as model parameters. The least well-quantified parameters are well retirement age and initial separation of the pump from the well bottom. Retirement age affects the inventory of wells potentially affected by declining groundwater levels (see 'Impact and cost estimation model' items 3a). Separation affects both the inventory of wells considered in the analysis (see 'Impact and cost estimation model' items 3c) and sensitivity of wells to changes in water level. Values for these two parameters are estimated through model calibration. Because the depth adjustment (Fig. 32a) is a combination of items 5 through 8 listed above, calibration of the separation distance acts as a surrogate for adjusting the other parameters.

The total number of domestic supply wells that experience supply interruption from declining groundwater level during the entire drought period are matched by PLSS section using the impact estimation model and weighted least-squares regression. As described above, supply interruption can occur because the water level drops too close to the pump or a well fails and needs replacement. Differences between model predictions and observations for each PLSS section form the population of regression residuals used to minimize the calibration objective function.

$$\min_{\mathbf{B}} \quad \mathrm{RSS}(\mathbf{B}) = \mathbf{R}^{\mathrm{T}}\,\mathbf{W}\,\mathbf{R} \tag{38}$$

Where **B** is the set of model parameters to be optimized (retirement age and separation)

RSS(**B**) is the residual sum of squares as a function of the parameter values

R is the n x 1 vector of regression residuals calculated as $\mathbf{Y} - \mathbf{Y'}(\mathbf{B})$

W is the n x n residual weighting matrix *n* is the number of PLSS sections in the study area with all data requirements filled (well constructions, water level data and observed supply interruptions)

Y is the *n* x 1 vector containing the observed total number of domestic well supply interruptions over the drought period for each PLSS section

Y'(**B**) is the *n* x 1 vector containing the model predicted total number of domestic well supply interruptions over the drought period for each PLSS section

Equation (38) is a standard minimization of error variance formulation where the weighted errors between model predictions and observations are assumed to be normally distributed with zero mean and no correlation (Draper and Smith 1981; Bates and Watts 1988). Parameter values that minimize the residual sum of squares (**B**) are considered optimal. In some cases, the formulation shown above may be simplified by substituting a diagonal residual weighting matrix for the full matrix (Draper and Smith 1981; Hill and Tiedeman 2007).

$$\min_{\mathbf{B}} \quad RSS(\mathbf{B}) = \sum_{i=1}^{n} w_i \, r_i^2 \qquad (39)$$

Where w is the weight for residual i

r is residual i calculated as $y - y'(\mathbf{B})$

y is the observed total number of domestic well supply interruptions over the drought period for PLSS section i

$y'(\mathbf{B})$ is the model predicted total number of domestic well supply interruptions over the drought period for PLSS section i

Because the impacts estimation model is based on well inventory, results are output as integers (no fractional supply interruptions are predicted). Therefore, the RSS response surface is discontinuous with many local minima and automated gradient search methods cannot perform the calibration. Enumerated exploration of the RSS response surface is readily performed through graphical interpretation since only two parameters are considered in the calibration. The enumerated search begins with a coarse-grid search using increments of 10 years and 10 ft (3 m) for the retirement age and separation distance parameters and then refines to increments of 1 year and 1 ft (0.3 m) in the vicinity of the optimal parameter values.

Initially, a diagonal residual weighting matrix of 1's (identity matrix) is used to obtain an approximate set of optimal parameter values using equation (39). Then a full weighting matrix is developed based on analysis of the initial regression residuals, and the regression is performed again with the resulting matrix using equation (38). To develop the full weighting matrix, spatial correlation in the initial residuals is modeled with a covariance function. The correlation likely stems from structural model errors introduced during creation of the model, such as when estimating the depth to groundwater for each PLSS section (see step 2 of 'Impact and cost estimation'). The covariance function is developed using established methods (Bentley 1997).

$$C(h) = C(0) - \gamma(h) \quad (40)$$

$$\gamma(h) = k\ [\ 1 - \exp(-3h/a)\ ] \quad (41)$$

Where $C(h)$ is the regression residual covariance at a lag distance h

$C(0)$ is the regression residual covariance at a lag distance of zero

$\gamma(h)$ is an exponential semivariogram for the regression residuals for lag distance h

k is an estimated parameter based upon fit to the regression residuals

a is an estimated parameter based upon fit to the regression residuals

The weighting matrix is generated by inverting a covariance matrix created with the covariance function (Draper and Smith 1981; Hill and Tiedeman 2007).

$$\mathbf{W} = \mathbf{C}^{-1} \quad (42)$$

Where $\mathbf{C}$ is the covariance matrix

Uncertainty Analysis

Sources of error in the calibrated model include inaccuracies in the data and model misspecification. Obvious inaccuracies in the data are discussed below ('Potential inaccuracies in the data'). Model misspecification errors stem from simplifications in the impacts and cost estimation model and can include 1) using single values for calibrated and non-calibrated parameters instead of incorporating variations that may occur over space and time and 2) applying a single depth to groundwater at each point in time for each PLSS section.

The likelihood ratio method is used to estimate model prediction uncertainty as nonlinear confidence intervals (Cooley and Vecchia 1987; Vecchia and Cooley 1987; Hill and Tiedeman 2007). The method identifies the upper and lower confidence limits of a prediction by separately maximizing and minimizing the prediction within a probabilistically defined region of the calibration RSS response surface that contains the optimal point:

$$\text{Confidence interval} = [\min_{\mathbf{B}} g(\mathbf{B}), \max_{\mathbf{B}} g(\mathbf{B})] \quad (43)$$

Subject to:

$$RSS(\mathbf{B}) \leq f\ RSS(\underline{\mathbf{B}}) \quad (44)$$

$$f = [p/(n-p)\ F_{\alpha}(p, n-p) + 1] \quad (45)$$

Where $g(\mathbf{B})$ is the model prediction using a set of adjusted parameter values within a probabilistically defined region around the calibrated parameter values

$RSS(\mathbf{B})$ is the residual sum of squares for the adjusted parameter values

$RSS(\underline{\mathbf{B}})$ is the residual sum of squares for the calibrated parameter values

p is the number of calibrated parameters

n is the number of observations as for equations (38 – 39)

F	is the statistical F distribution
α	is the significance level

The process of determining the confidence interval for the model prediction is performed manually because, as discussed above, the RSS surface is discontinuous. It entails plotting the probabilistically defined region of the enumerated RSS response surface defined by equation (7), overlaying the RSS response surface with a contour plot of the prediction, and identifying the extreme prediction values along the boundary of the probabilistically defined RSS region (Fig. 35).

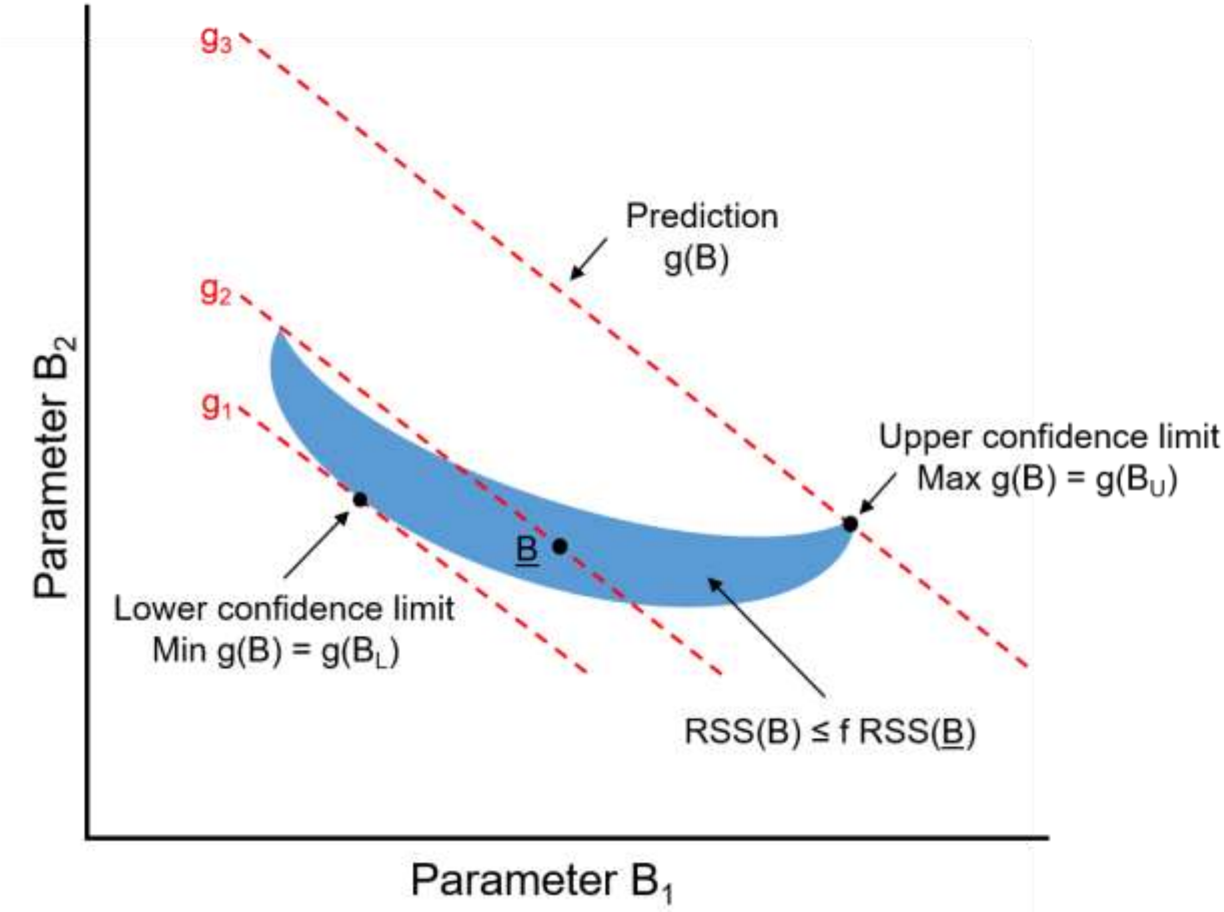


Figure 35: Process for defining model prediction confidence intervals (after Hill and Tiedeman 2007). *Blue shaded region* indicates probabilistically-defined set of parameter value combinations on the RSS response surface over which the search is conducted for minimum and maximum model predictions. *Red dashed lines* represent model prediction values for different combinations of parameter values. For this example, $g_3 > g_2 > g_1$. $\underline{B}$ is the optimal (calibrated) set of parameter values

Drought Scenario Evaluation

The calibrated impacts and cost estimation model is used to evaluate the effects of potential future droughts on domestic well water supplies in the study area. For 2012 to 2016, changes in depths to groundwater since 2011 are scaled by a dimensionless factor as a proxy for the potential effects on groundwater levels from future droughts (surface water shortfalls and subsequent increased groundwater pumping across the county), then impacts and costs are estimated as described above (see 'Impact and cost estimation model'). Values for the scaling factor are specified between 0.5 and 2.0 in increments of 0.5. The estimates are adjusted downward to account for outages permanently addressed during the recent drought and, as a result, would not be expected to experience another impact soon. The adjustment is based on records of resolved supply outages (see 'Data characteristics' below), with a portion of the supply interruption and cost predictions from the calibrated model removed from the drought predictions. Uncertainty in the predictions is evaluated using the likelihood ratio method described above. No discounting of estimated costs is performed.

Limitations

Limitations of this approach include 1) errors in the calibrated model, 2) assumption that future groundwater level declines will resemble those that occurred during the 2012 to 2016 drought, 3) consideration of only the valley floor in the study area (omitting wells completed in fractured-rock groundwater systems in the foothills and mountains) and 4) assumption of estimated costs for corrective actions as economic impacts. The uncertainty analysis attempts to address issues resulting from errors in the model. The spatial distribution of future groundwater level declines and, therefore, domestic supply well impacts and costs could differ with future management actions (e.g., changes in cropping patterns, induced recharge projects and demand management). However, the approach presented here is easily modified for other representations of future groundwater levels such as numerical model output. Any specification of changes in groundwater depth that addresses each PLSS section can be incorporated in the approach. Likewise, use of groundwater levels extending outside the study area could allow wider geographic regions to be considered. With regard to using costs for maintaining well operations as a surrogate for economic impact, another approach could be to use estimated costs for connection to centralized water system; however, this approach may not be realistic where residences are remote and continued water service by wells is required. A consideration not addressed in estimation of economic externalities is the potential for water quality to worsen with declining water levels. The concentrations of naturally-occurring and anthropogenic contaminants in the discharges of existing wells may increase (or decrease) as the relative mass contributions from different strata respond to water level declines. Also, replacement wells may penetrate deeper strata with higher concentrations of undesirable constituents (i.e., arsenic – a common contaminant in the area). It may be possible to add estimated water treatment costs to this approach. Nevertheless, this approach provides analysis where none was previously available, estimates uncertainty in the results, allows for flexibility in future application and adds reasonable insight on the topic.

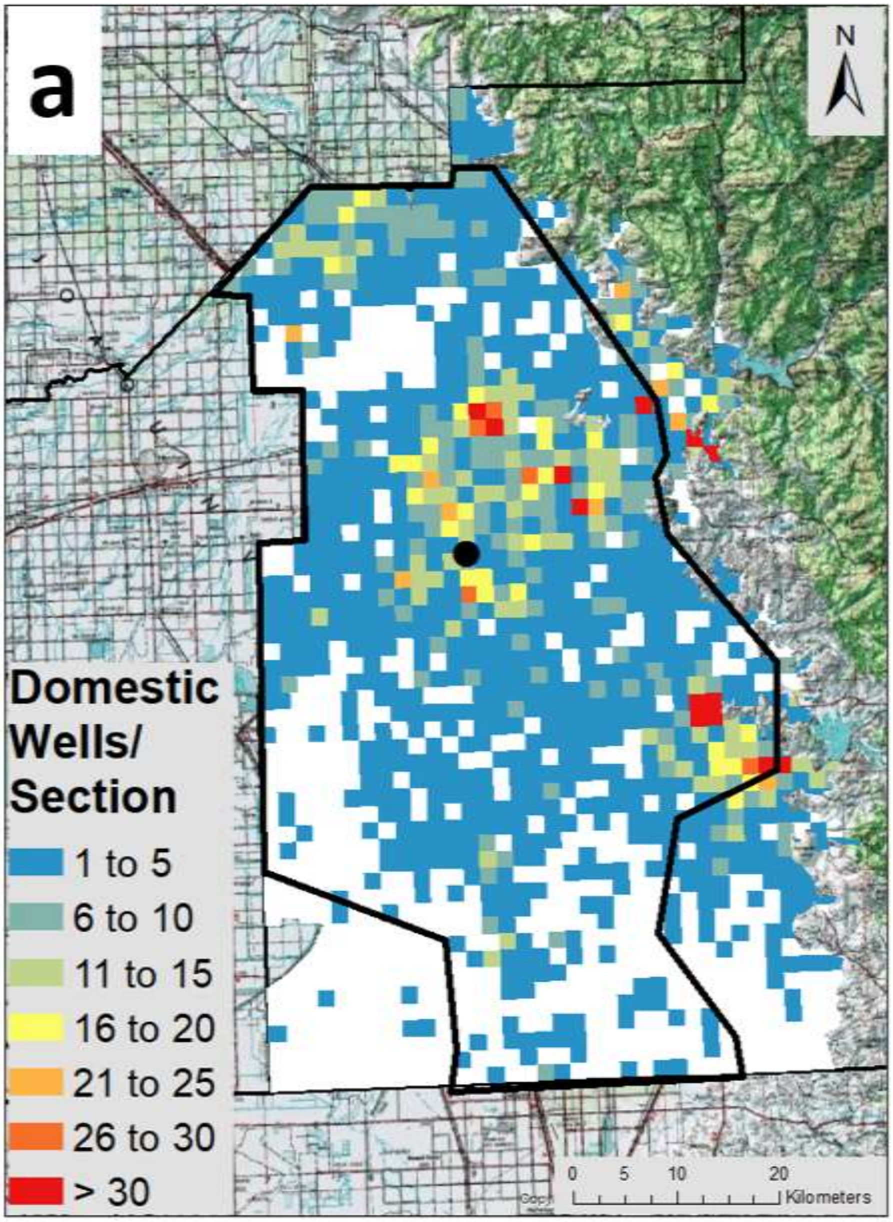
Data Sources

Data for the methods described above are available for the study area. Domestic and agricultural supply well locations, depths to top of screens, well depths, construction dates and drilling methods are recorded at the CADWR (CADWR 2017b). The well locations are based on PLSS sections and the tops of screens, as well as bottoms of wells, are specified by depths from ground surface to the tops of the shallowest screened intervals and bottoms of the wells. Additional information on the intended construction of domestic supply wells, in the form of construction permit applications, over time are available from the Tulare County Health and Human Services Agency, Environmental Health Services Division, with well locations based on PLSS sections. Depth to groundwater data are available from the California Department of Water Resources (CADWR 2017a). These data were downloaded as GIS layers. Domestic supply outages in Tulare County are documented by the Tulare County Office of Emergency Services (County of Tulare 2017), with well locations based on PLSS sections.

Results and Discussion

Data Characteristics

Of the 1,724 PLSS sections in the study area, well construction data needed for the analysis is available for 1,108 sections with a total of 5,774 domestic wells, and groundwater level data throughout the drought period is available for 1,314 sections. Spatial overlap in the well construction and water level datasets creates a model domain of 923 sections (Fig. 36a). During the drought, 1,382 domestic service outages were recorded in 310 of the sections in the model, with no service outages reported in the remaining sections (Fig. 36b).


a
N
Domestic Wells/ Section
1 to 5
6 to 10
11 to 15
16 to 20
21 to 25
26 to 30
> 30
0 5 10 20 Kilometers

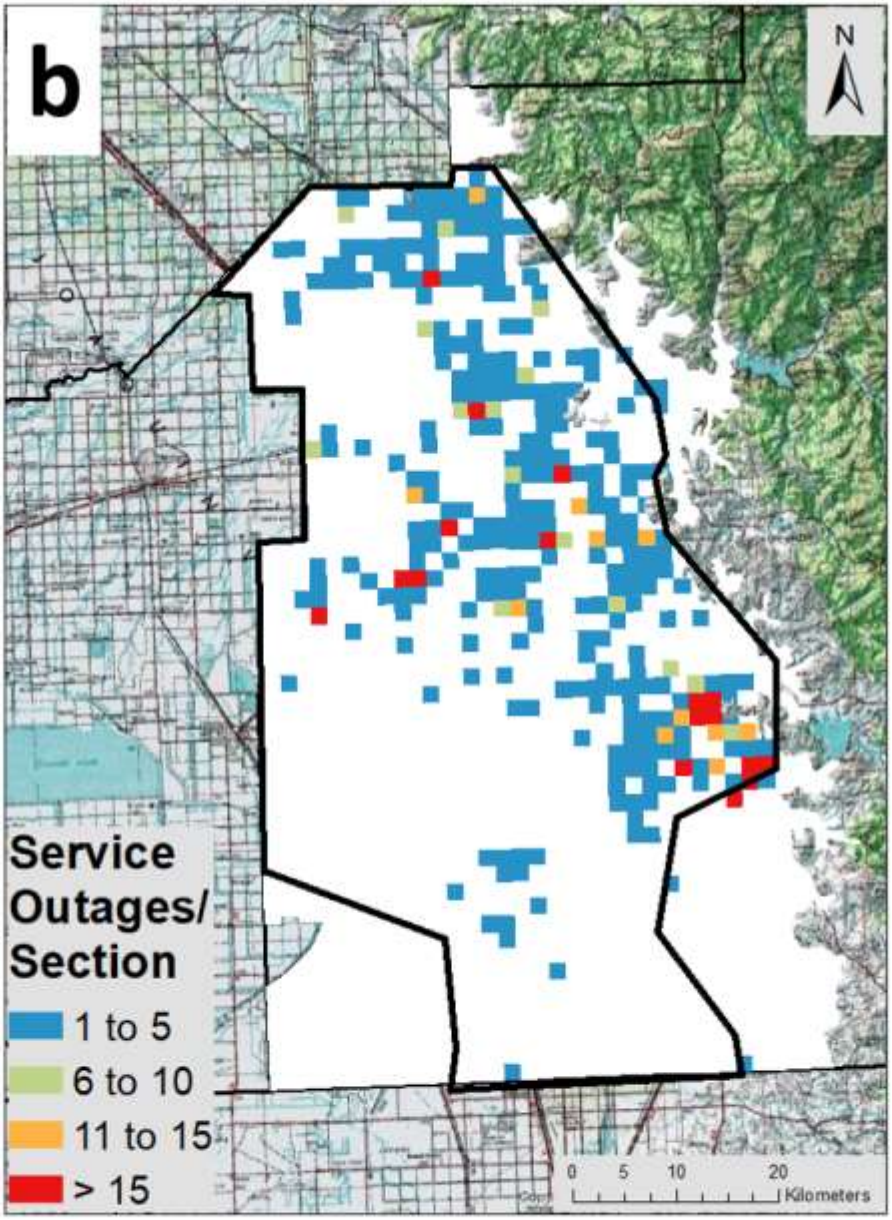
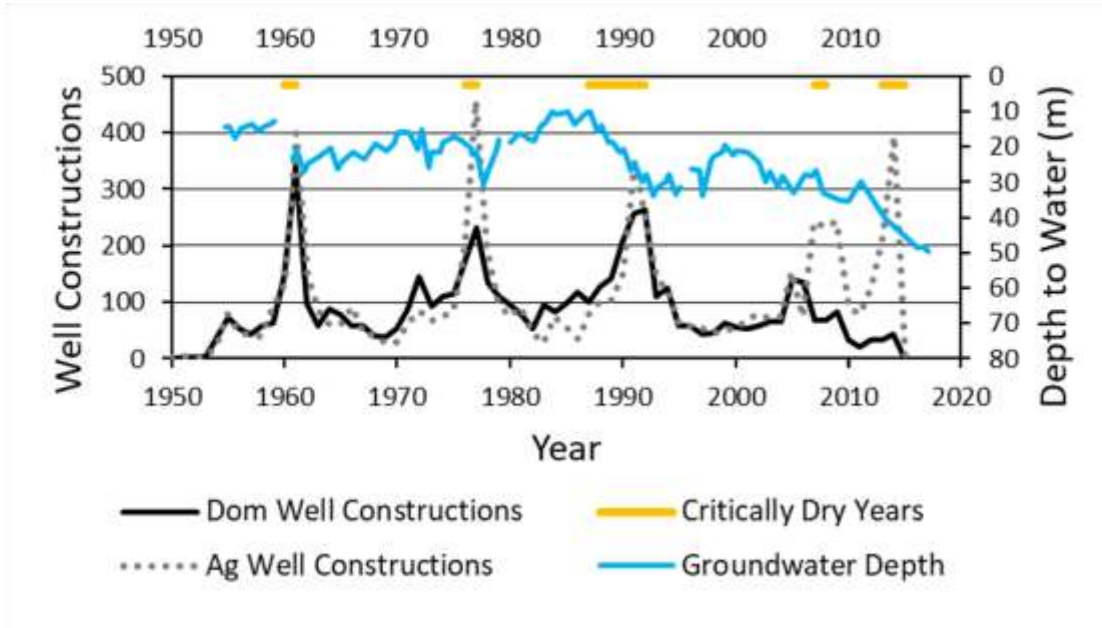


Figure 36: Primary data: **a** number of domestic wells per section and **b** number of service outages per section. *White area* indicates portion of Tulare County located on valley floor. *Black outline* indicates area where groundwater level data are available. *Black dot* on Fig. 7a indicates location of hydrograph well for Fig. 37a

The well construction data through 2016 generally shows that domestic well construction rises in critically dry years with decreases in groundwater levels (Fig. 37a) and suggests that domestic supply outages occurred during previous droughts. However, these data do not show a peak in domestic well construction during the recent drought, while larger, more expensive and profitable agricultural well constructions increase markedly as during previous dry periods. These data may indicate that the supply of well construction labor and equipment was insufficient for the combined demand for domestic and agricultural well replacement during the drought. This hypothesis is supported by 1) higher numbers of domestic well construction permits than recorded constructions and 2) observations of the first author while working in the study area indicating a skilled labor shortage and agricultural wells being serviced first.

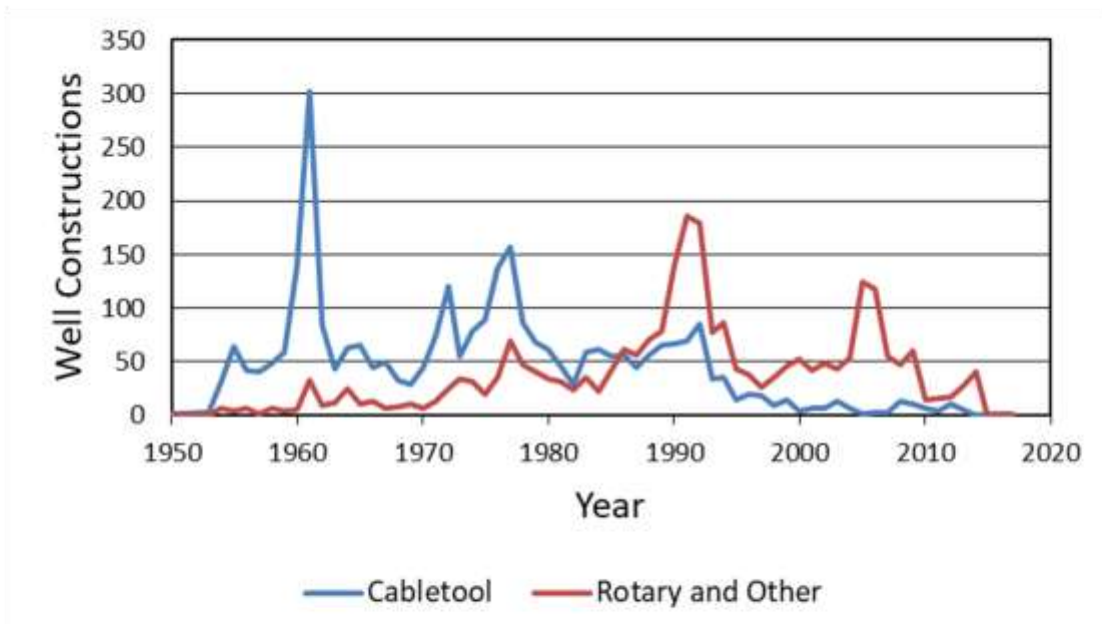
The data also show that well drilling technology shifted over time, with the rotary method largely replacing the cable tool approach (Fig. 37b), allowing wells to be completed to greater depths (Fig. 37c). While the increasing trend in average well depth with time is slight, most deeper wells have been constructed in the past 30 years (Fig. 37d). This is approximately when rotary drilling became more prevalent than cable tool (Fig. 37b). Therefore, older domestic wells are generally shallower and more susceptible to declining groundwater levels than newer deeper wells. Likewise, comparison of depths between domestic and agricultural wells of all ages (Fig. 37e) indicates that domestic wells are shallower than many agricultural wells and more susceptible to groundwater level declines.

a



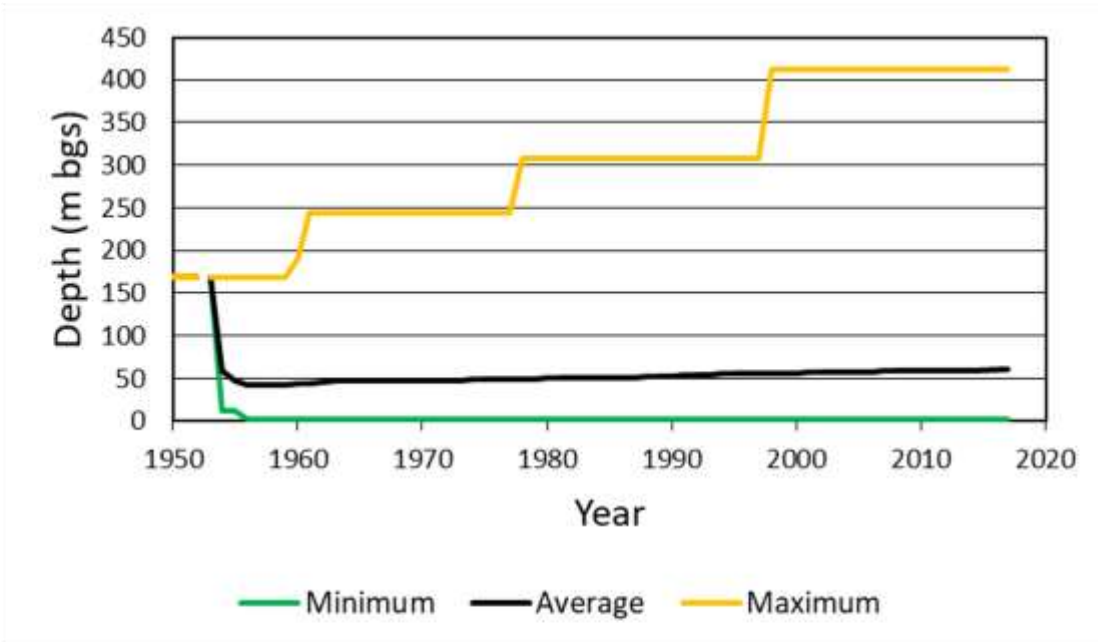
1950
1960
1970
1980
1990
2000
2010
Well Constructions
500
400
300
200
100
0
Depth to Water (m)
0
10
20
30
40
50
60
70
80
2020
Year
Dom Well Constructions
Ag Well Constructions
Critically Dry Years
Groundwater Depth

b



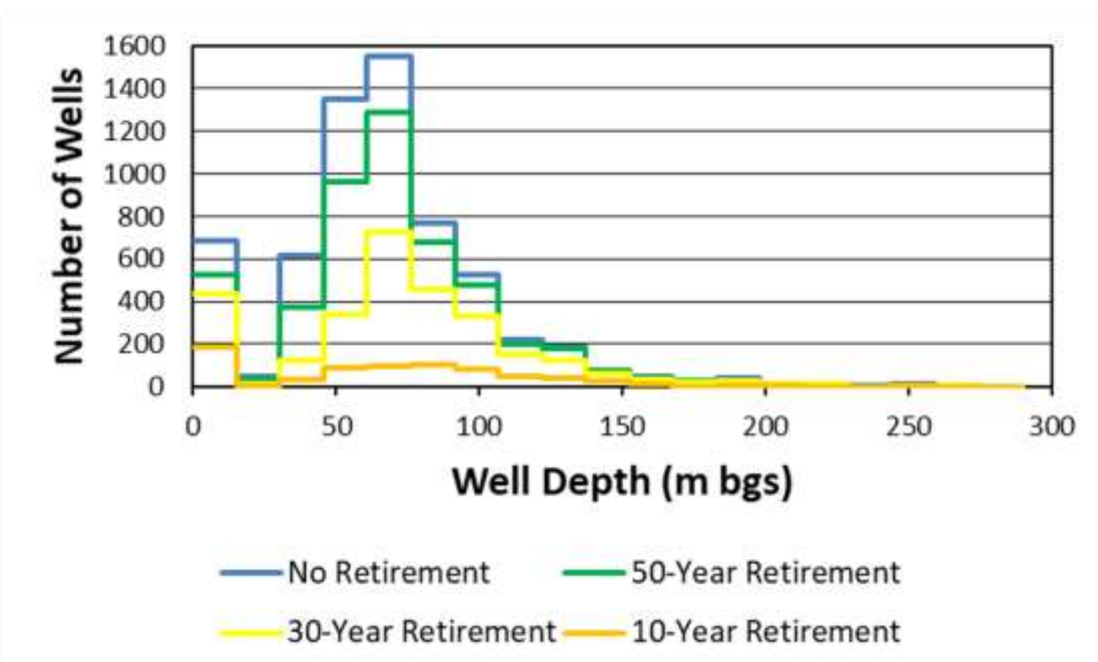
Well Constructions
350
300
250
200
150
100
50
0
1950
1960
1970
1980
1990
2000
2010
2020
Year
Cabletool
Rotary and Other

c



450
400
350
300
250
200
150
100
50
0
Depth (m bgs)
1950
1960
1970
1980
1990
2000
2010
2020
Year
Minimum
Average
Maximum

d



1600
1400
1200
1000
800
600
400
200
0
Number of Wells
0
50
100
150
200
250
300
Well Depth (m bgs)
No Retirement
50-Year Retirement
30-Year Retirement
10-Year Retirement

e

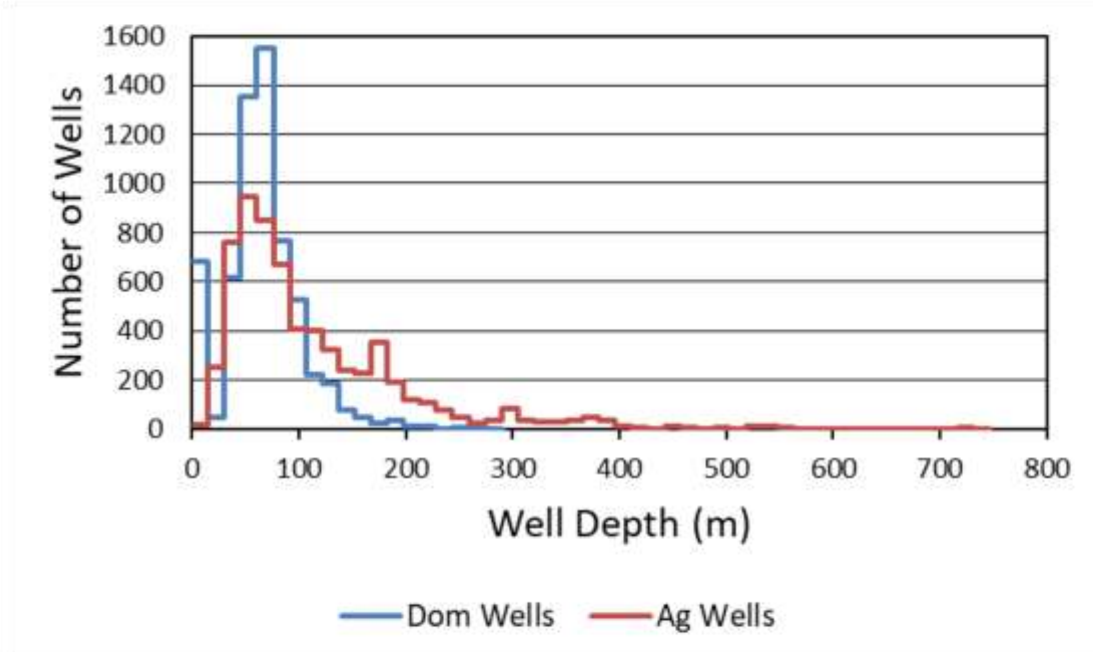


Figure 37: Domestic well construction characteristics: **a** construction history, **b** change in drilling technology, **c** increase in well depths, **d** change in depth distribution and **e** depth distribution relative to agricultural wells. Data location for *groundwater depths* in Figure 37a shown on Figure 36a. *Well construction* data source: CADWR Well completion report map (https://dwr.maps.arcgis.com/apps/webappviewer/index.html?id=181078580a214c0986e2da28f8623b37). *Groundwater depth* data source: Well 362539N1193051W001 CADWR Water Data Library (http://www.water.ca.gov/waterdatalibrary/). *Critically dry year* data source: CADWR California Data Exchange Center (http://cdec.water.ca.gov/cgi-progs/iodir/WSIHIST). Figure 37e compares all available well construction logs: 5,774 domestic (Dom) and 6,480 agricultural (Ag) wells

Reported supply outages (Fig. 36b) are cumulative results for houses on the valley floor over the drought period, with 1,382 reported outages in 310 PLSS sections between 2012-2016. Additional outages (224) occurred just east of the study area in the foothills but are omitted from the analysis since water level data necessary for these locations are not readily available. In some PLSS sections, the reported service outages exceed the number of wells partly because some wells serve several houses. Reported service outages are converted to well service interruptions for each PLSS section to be used as model calibration data. This is done by comparing the number of wells in a section with the number of service outages. If the number of outages exceeds the number of wells, the number of well service interruptions is taken as the number of wells in the section since it is not possible for well service interruptions to exceed the number of wells (Fig. 38). The estimated number of well failures considered for model calibration is 978 in 292 sections. The data also state that slightly more than half (56 percent) of the service outages were permanently addressed during the drought (i.e., new wells constructed deeper or alternative water sources) while other were not addressed.

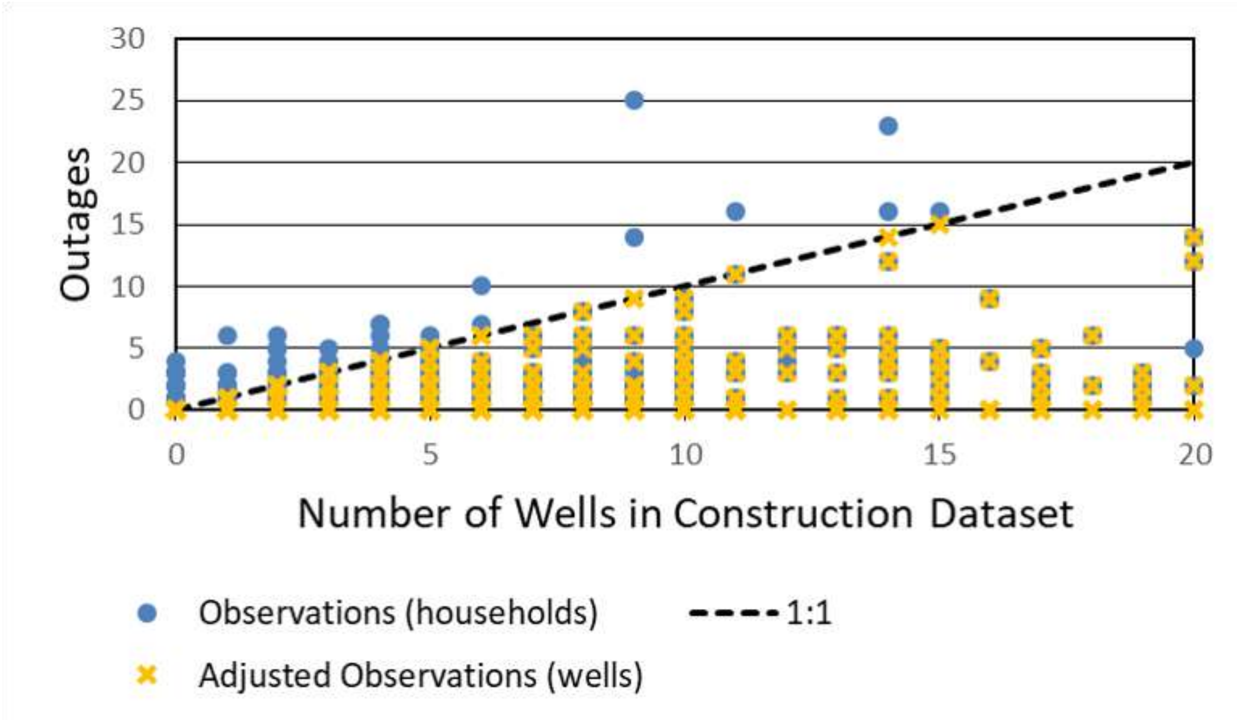


Figure 38: Adjustment of house supply outages to well supply outages

Potential Inaccuracies in the Data

There are potential inaccuracies in each of the three primary datasets used here that motivate application of the estimation and uncertainty analysis described above.

1) Well construction information is based on a historic requirement by the state that details of all well construction activity be self-reported. Reporting occurred on paper forms until 2016 and the dataset for this work was constructed by the CADWR from the old paper files. The combination of self-reporting and data transcription likely resulted in some actual well constructions not being included in the dataset and transcription errors. Missing data may be related to the reductions in the observed service outages described above.

These same limitations on data acquisition may also create a time lag in updating the dataset which could explain the sharp decrease in agricultural well construction activity for 2015 while the drought was still in process (Fig. 8a). An attempt is made to address the errors through the regression procedure and uncertainty analysis.

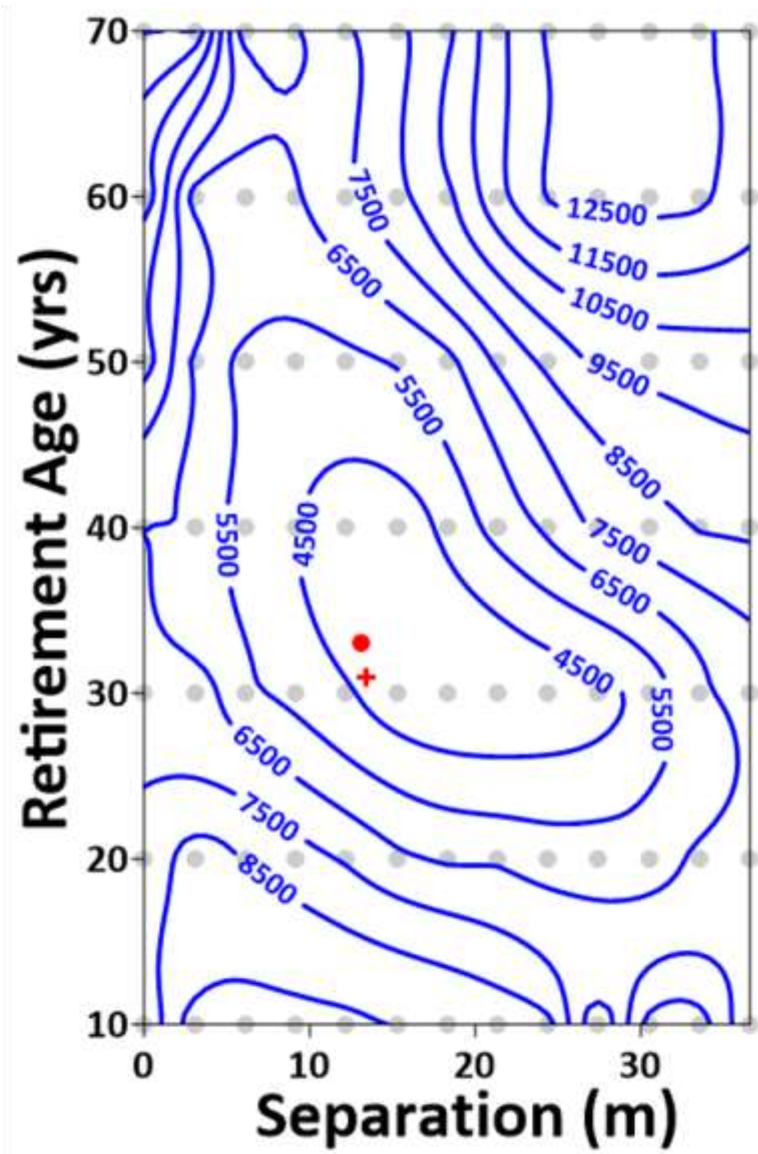
2) Groundwater levels are derived from contours prepared by the CADWR. The contours are themselves based on measurements that are sparse, both spatially and temporally, as well as distributed throughout a groundwater system that exhibits notable variations in the three spatial dimensions as well as in time. Aggregation and interpretation were applied in preparing the contours (personal communication with CADWR staff). Differences are expected between the contoured representations of the groundwater levels and actual groundwater levels. The anticipated data errors likely manifest as correlation in the regression residuals. An attempt is made to address this source of parameter estimation bias and prediction error through use of a full regression weighting matrix as well as the uncertainty analysis.

3) Service outages are also self-reported and missing data are expected. An additional factor that may lead to under-reporting of service outages is the potential correlation between domestic well service interruptions and residents who have complications related to US immigration status. As a result, reports may be lower than actual in sections with some reported outages and sections with no reports may actually contain some outages. As a result of these potential errors in the data, only sections with reported outages are used in the calibration. Sections with no reported outages are not considered as representing a value of zero for residual calculation in the regression analysis.

Model Calibration

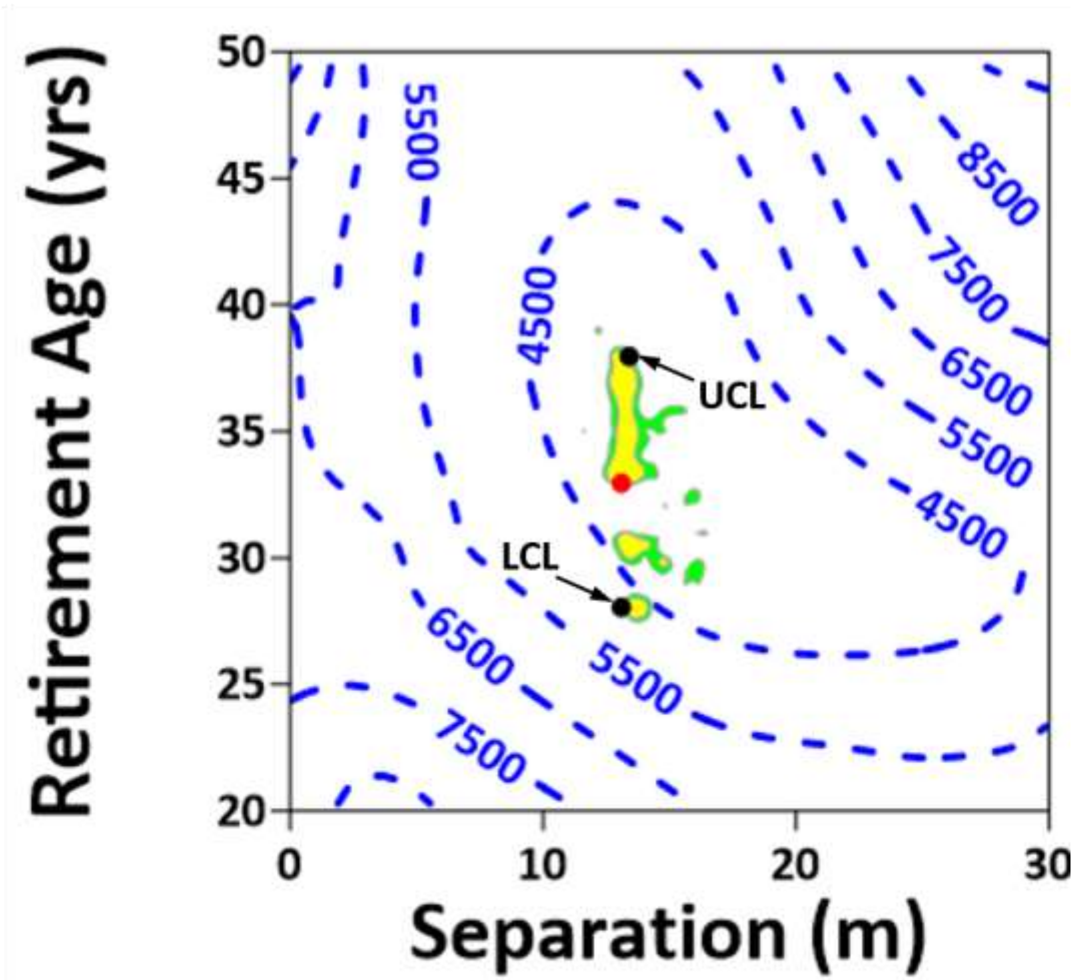
The results of initial regression-based model calibration using a diagonal residual weighting matrix are summarized on Figures 39a through c. Effects that different parameter values have on the topography of the RSS surface are best understood using the RSS surface on Fig. 39a as a reference. Changes in parameter values relative to the minimum point on the surface affect the population of wells considered in the well impact calculations. Too many or too few shallow wells relative to the point increases the objective function in equations (1 and 2) by decreasing the fit to the calibration dataset. As discussed above, the older population of wells includes more shallow wells which are more sensitive to decreases in groundwater levels (Fig. 37c and d). Therefore, for a given decrease in groundwater levels, the number of wells affected increases (relative to the optimal point) with higher retirement age because more shallow wells are considered in the model calculations and more impacts occur than in the calibration dataset. Similarly, lower well retirement age decreases the number of well impacts such that the calibration data are poorly matched and the RSS increases. The value for separation distance also changes the number of shallow wells considered in the well impacts calculations. Large separation values are not physically possible for shallow wells as the pump would be positioned above ground surface. For such values, fewer shallow wells are present in the population, too few well impacts are predicted and the RSS is high. Likewise, placing the pump too low in the well reduces impact sensitivity to declining groundwater levels and the RSS is higher relative to its minimum.

a



70
60
50
40
30
20
10
0
10
20
30
Retirement Age (yrs)
Separation (m)
12500
11500
10500
9500
8500
7500
6500
5500
4500
4500
5500
5500
6500
7500
6500
5500
4500
7500
8500

b



Retirement Age (yrs)
Separation (m)
50
45
40
35
30
25
20
0
10
20
30
5500
4500
8500
7500
6500
5500
4500
UCL
LCL
6500
5500
7500

c

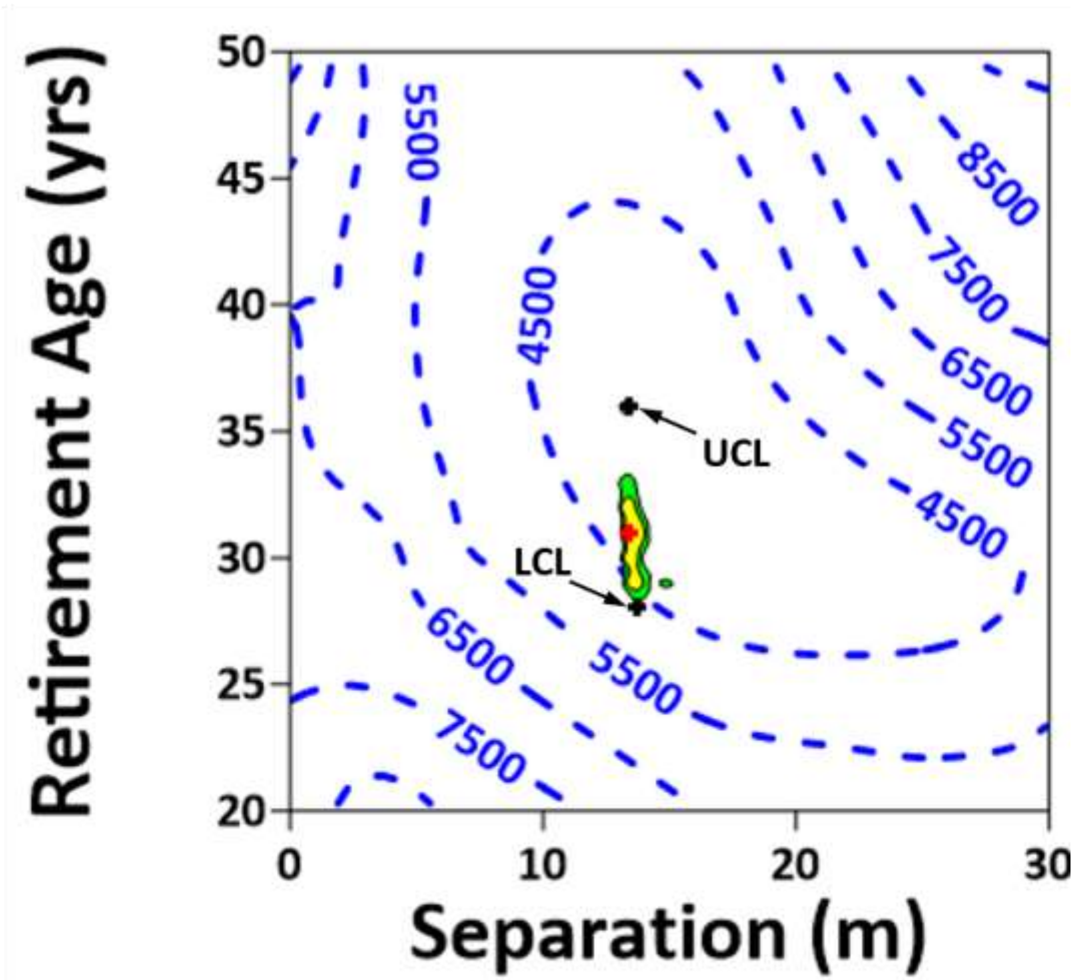
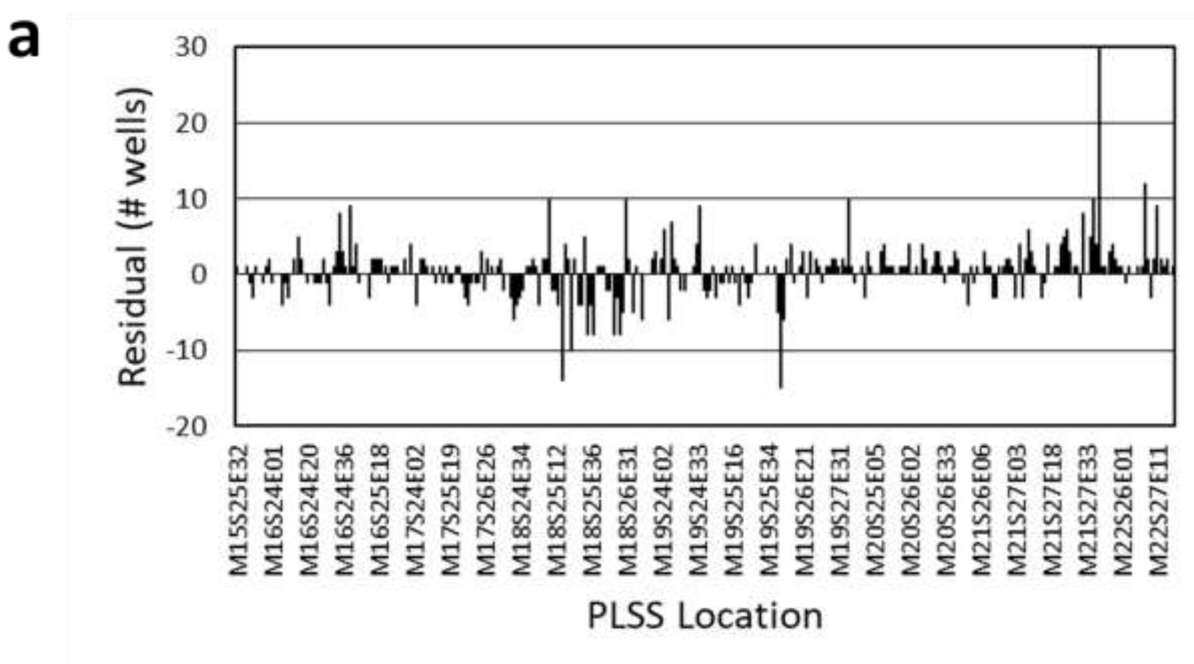


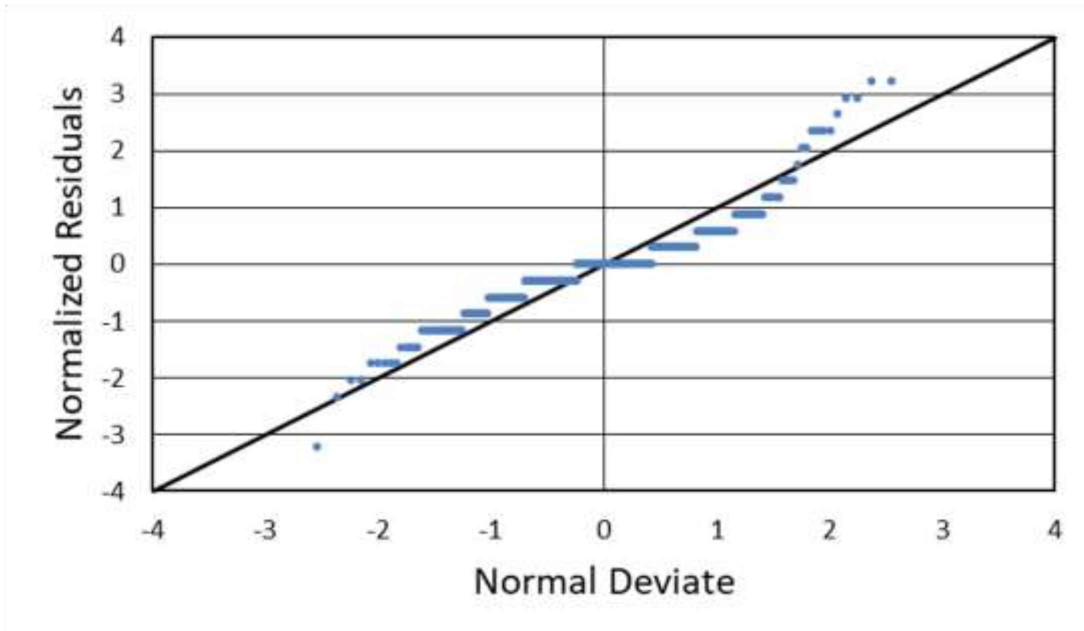
Figure 39: Regression RSS response surface: **a** coarse-grid enumeration, **b** refined-grid enumeration in vicinity of the optimal parameter values using diagonal regression weighting matrix and **c** refined-grid enumeration in vicinity of the optimal parameter values using full regression weighting matrix. *Grey dots* indicate parameter value combinations evaluated for coarse-grid enumerated search. *Red dot* on Figs. 39 a and b indicates optimal parameter values found using the refined-grid and diagonal weighting matrix. *Red cross* on Figs. 39 a and c indicates optimal parameter values found using the refined-grid and full weighting matrix. *Colored zones* on Figs. 39 b and c indicate foot prints of confidence regions. Yellow is the 95%-likelihood region, while yellow and green combined is the 99%-likelihood region. *Black dots and crosses* on Figs.

39 b and c indicate 99%-confidence limits (*UCL* is upper and *LCL* is lower) for the diagonal and full weighting matrices. *Blue contours* are based on results from using the diagonal weighting matrix

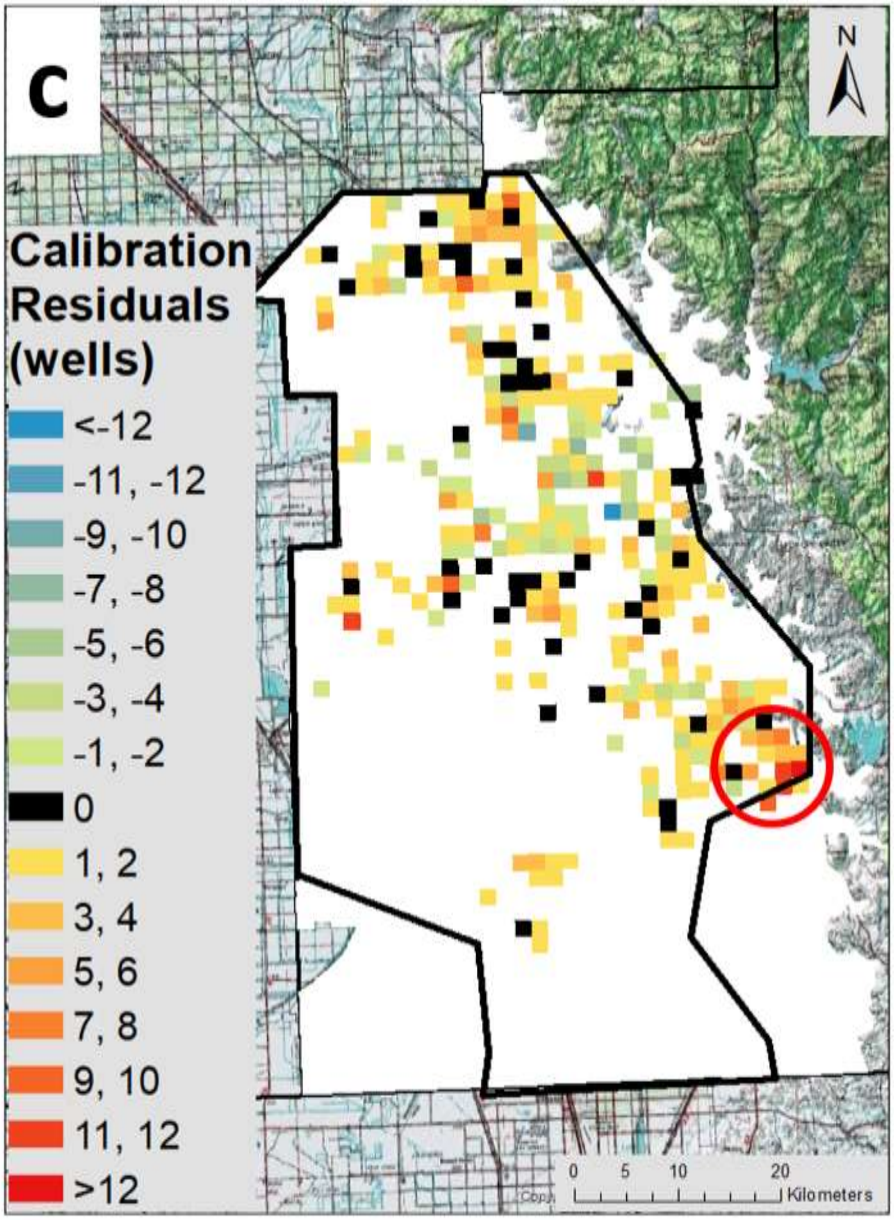
There are multiple minima in the regression response surface (Figs. 39 b and c); however, the enumerated search used for this work is conducted broadly enough that the global optimum appears to have been identified. The optimal parameter values for the diagonal regression weighting matrix are: retirement age of 33 years and separation distance of 43 ft (13 m). These parameter values create a residual population with mean of 1.0 and standard deviation of 3.4 that resembles a normal distribution (Fig. 40a and b). Table 11 summarizes the results. The observed number of well impacts in the calibration dataset is 978 over the 292 PLSS sections while the predicted impact for these calibration data locations is 697 wells.



b



Normalized Residuals
4
3
2
1
0
-1
-2
-3
-4
-4
-3
-2
-1
0
1
2
3
4
Normal Deviate


c
N
Calibration Residuals (wells)
<-12
-11, -12
-9, -10
-7, -8
-5, -6
-3, -4
-1, -2
0
1, 2
3, 4
5, 6
7, 8
9, 10
11, 12
>12
0 5 10 20
Kilometers

d

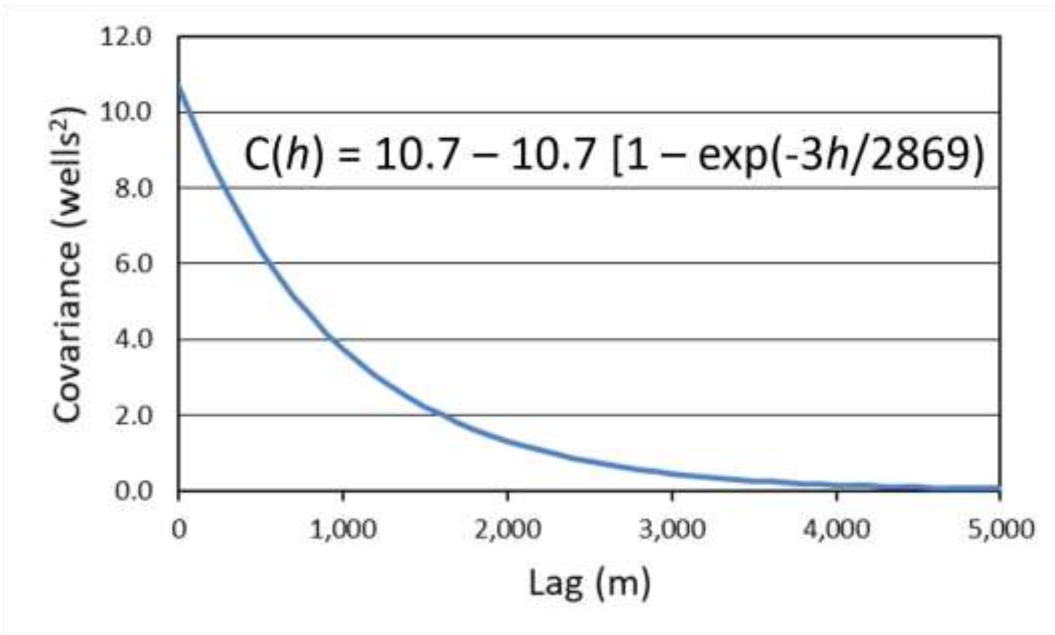


Figure 40: Residual plots: **a** population, **b** normal probability, **c** spatial distribution and **d** covariance function. *White area* on Fig. 40c indicates portion of Tulare County located on valley floor. *Red circle* on Fig. 40c indicates area where model significantly under-predicts impacts to wells

Developing a full residual weighting matrix for final calibration entails modeling the spatial distribution of regression residuals (Fig. 40c) as described above ('Model calibration') using a covariance model that reflects the relatively minor spatial correlation in the regression residuals (Fig. 40d). Use of the full weighting matrix changes the RSS values, as well as smooths and contracts the likelihood region (Fig. 39c). The optimal parameter values for the full regression weighting matrix are very similar to those for the diagonal weighting matrix: retirement age of 31 years and separation distance of 44 ft (13 m). Table 11 summarizes the differences in calibration statistics. Because the residual standard deviations for the two calibrations are comparable but the residual mean is smaller for the diagonal weighting matrix calibration, the parameter values developed with the diagonal weighting matrix are carried forward for additional analysis.

Table 11 Calibration results

Statistic	Diagonal Weights	Full Weights
Calibrated separation	43 ft (13 m)	44 ft (13 m)
Calibrated retirement age	33 years	31 years
RSS	651 wells^2	312 wells^2
Residual mean	1.0 well	1.4 well
Residual standard deviation	3.4 wells	3.3 wells
Observed Well impacts	978	978
Predicted Well Impacts	697	644

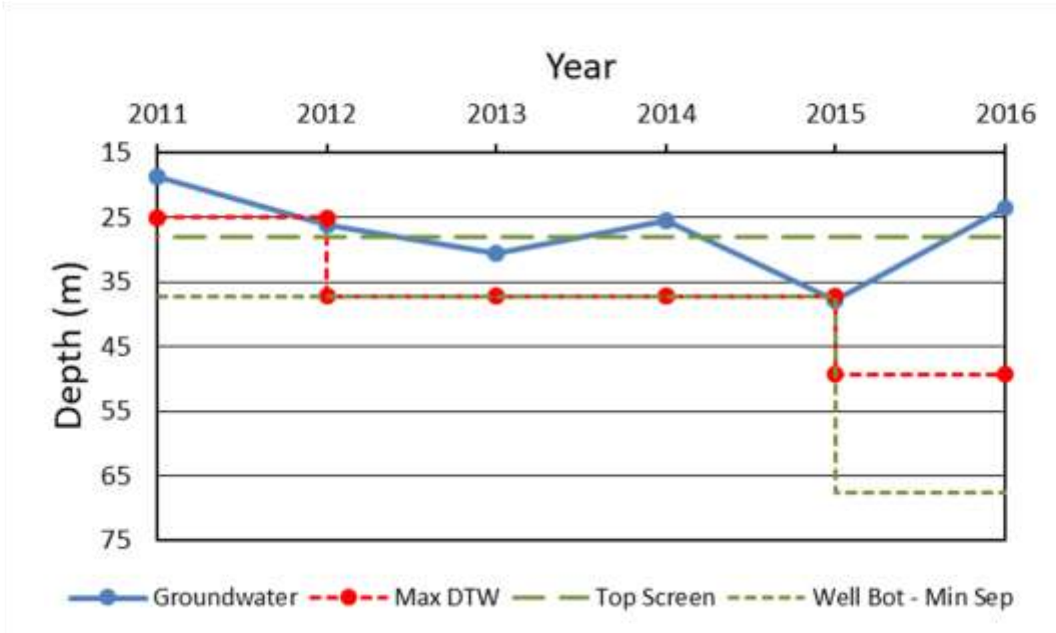
The calibration residuals and statistics indicate that the model under-predicts the level of well impact. This is most evident along the eastern edge of the model (circled area on Fig. 40c) where the model fails to predict impacts in some sections most heavily affected by drought. This model shortcoming results from a structural error imposed by the groundwater level data. The water levels estimated from the available data are all shallower than the well screens obtained from the well construction data. As indicated in 'Potential inaccuracies in the data', the water levels are expected to introduce errors into the modeling. The general problem is exacerbated for the noted location since it is at the edge of the area covered by the data and estimation of the levels are likely less accurate that might otherwise be the case.

Estimation of Economic Externalities from Recent Drought

The calibrated model is used to evaluate potential corrective reactions to groundwater level declines for each well in the study area. Figures 12a through c provide example results for a single well over the drought. The groundwater level starts higher than the MDTW and declines over the drought (Fig. 41a). The MDTW and DTOS criteria are triggered in 2012 and 2013 (Fig. 41a) when corrective actions for pump deepening and well screen cleaning occur (Fig. 41b). The pump lowering in 2012 leads to minimum separation while continued groundwater level decline triggers the MDTW criterion again in 2015 (Fig. 41a). At this point, well replacement occurs and the pump is positioned deeper in the new well (Fig. 41 a and b). The cumulative costs for the corrective actions are plotted on Figure 41c.

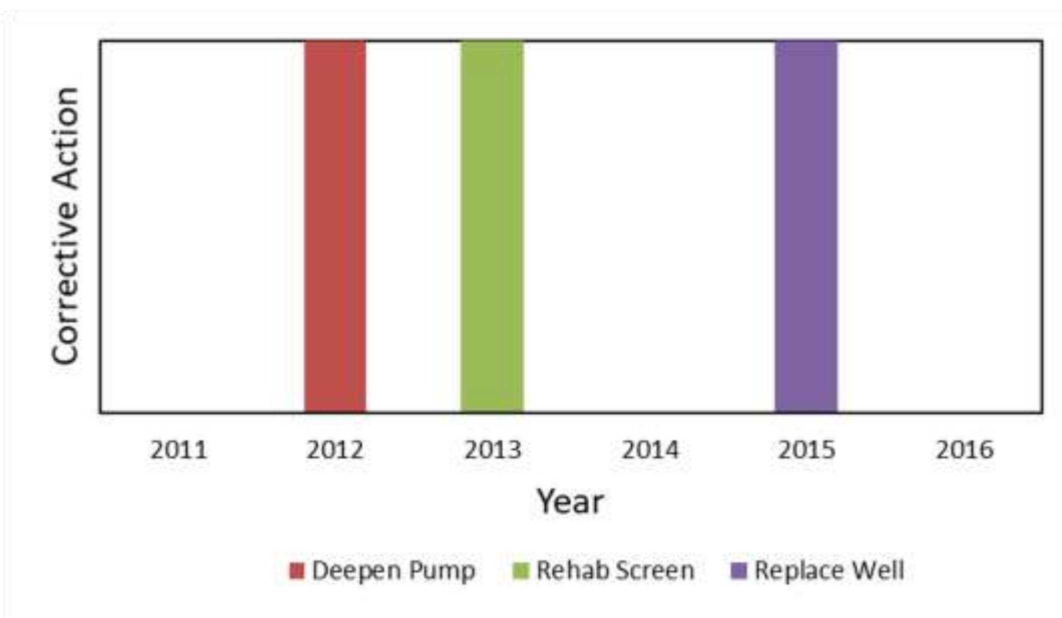
Figure 41d summarizes predicted well impacts during the drought for all PLSS sections with well construction and water level data regardless of whether supply outage observations are available. This is a larger dataset than for calibration (Fig. 36b and 40c). The total number of domestic well supply interruptions predicted for the drought is 1,070 over 404 PLSS sections, of 5,774 domestic wells with complete construction data in the study area. If all impacts to domestic supply wells over the drought period had been addressed, we estimate that the cost would have been $13.9 million (Table 12). This amount does not include additional pumping lift costs in years after the drought ended while depressed groundwater levels were still recovering.

a



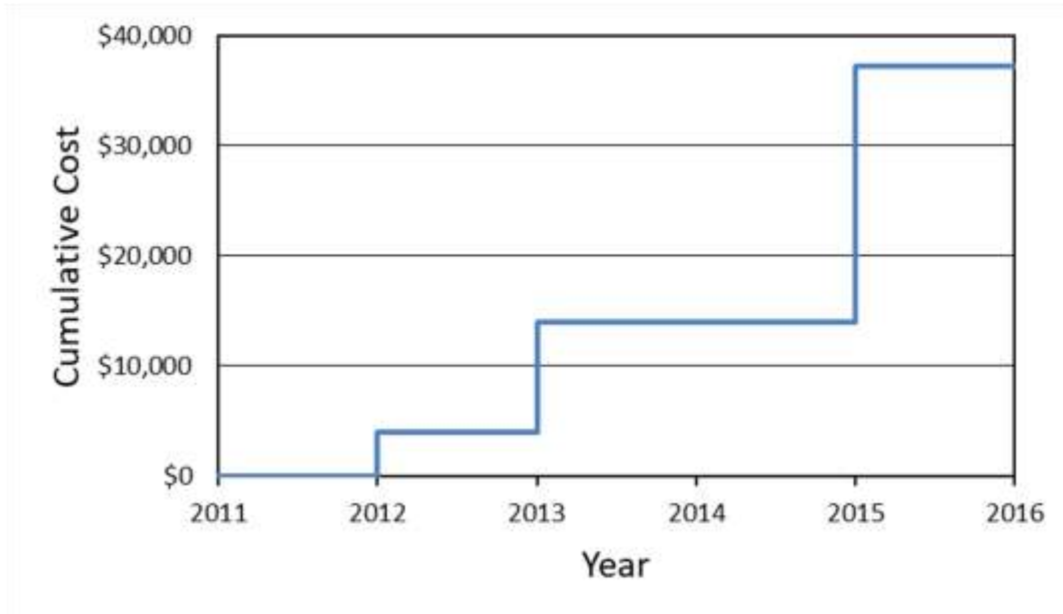
Year
2011
2012
2013
2014
2015
2016
15
25
35
45
55
65
75
Depth (m)
Groundwater
Max DTW
Top Screen
Well Bot - Min Sep

b



Corrective Action
2011
2012
2013
2014
2015
2016
Year
Deepen Pump
Rehab Screen
Replace Well

c


$40,000
$30,000
$20,000
$10,000
$0
Cumulative Cost
2011
2012
2013
2014
2015
2016
Year

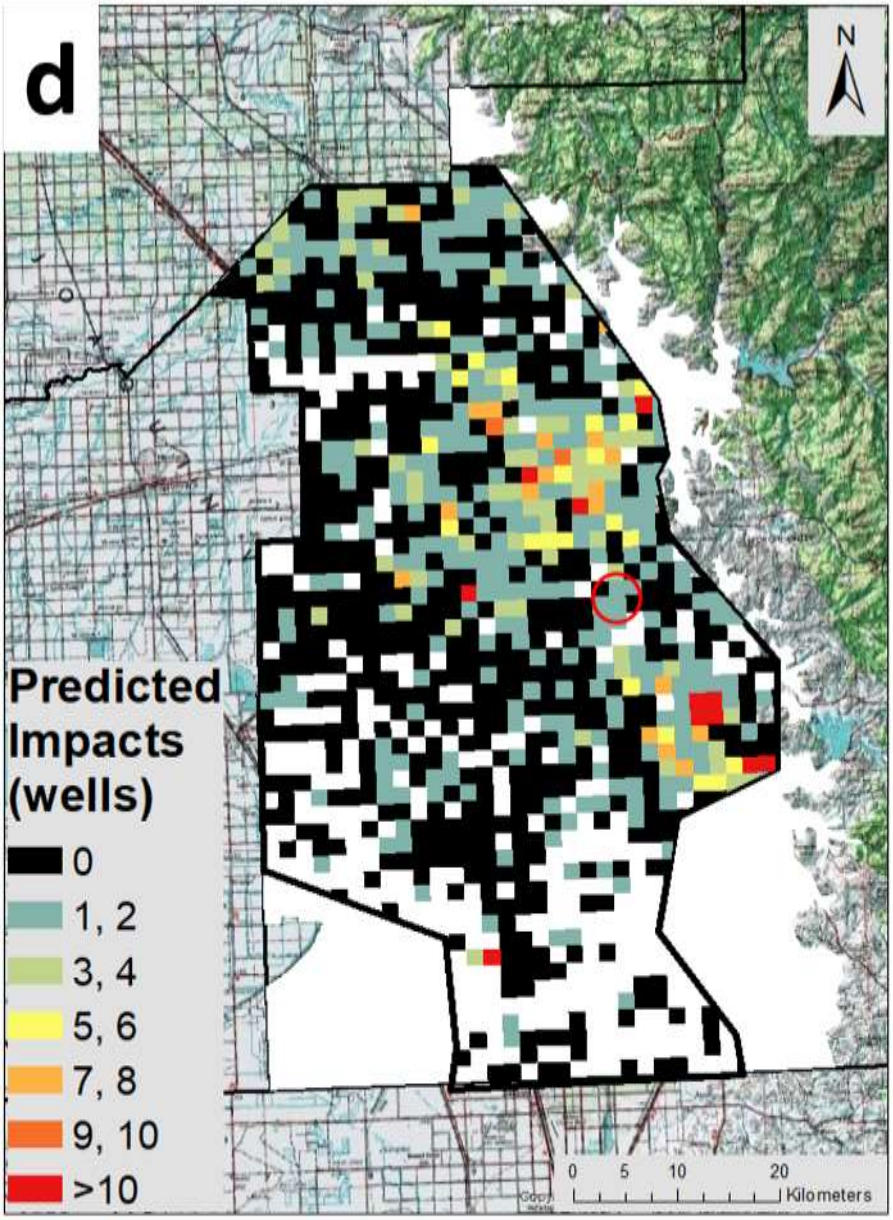


Figure 41: Simulation of recent drought: **a** example results for a single well, **b** timing of corrective actions for the well, **c** estimated cumulative costs for the well and **d** well impacts across study area. *White area* on Fig. 41d indicates portion of Tulare County located on valley floor. *Red circle* on Fig. 41d is the location of example well for Fig. 41 a - c

Table 12 Estimated costs to address 2012-2016 drought impacts to domestic supply wells, Tulare County valley floor

Action	Estimated Costs ($)		
	Mean	Low	High
Pumping energy increase	$8.5x10^4$	$6.8x10^4$ -21%	$9.2x10^4$ +7%
Pump lowering	$3.7x10^6$	$2.9x10^6$ -20%	$4.6x10^6$ +21%
Well screen rehabilitation	$4.2x10^6$	$3.6x10^6$ -14%	$4.8x10^6$ +12%
Well replacement	$6.0x10^6$	$3.7x10^6$ -38%	$8.9x10^6$ +33%
Total	$13.9x10^6$	$10.3x10^6$ -26%	$18.5x10^6$ +24%

Low and *High estimated costs* are based on the 99% confidence interval. *Costs* are in US dollars. *Percentages* are differences from mean costs normalized by the mean costs

Uncertainty Analysis

The likelihood region used in the uncertainty analysis (Fig. 39b) was based on the fine-grid enumeration of the RSS surface and contour values derived from equations (44 - 45). Parameter values for equation (45) were as follows: n = 292 (number of PLSS sections with observed well impacts), p = 2 (number of calibrated model parameters), α = 0.05 for the 95%-likelihood region and α = 0.01 for the 99%-likelihood region. The 99%-likelihood region was used along with the

RSS enumeration results to identify the bounding prediction values for total supply well interruptions and total cost that provide estimates for the confidence limits (UCL and LCL on Figures 39 b and c).

The mean predicted impact is 1,070 wells with upper and lower 99%-confidence limits of 815 (-24%) and 1,461 (+37%) wells. Confidence limits for costs to address drought impacts to domestic wells appear in Table 12. Because the impact and cost estimation model responds nonlinearly to values for calibration parameters (retirement age and separation distance), the confidence intervals are not symmetric. The mean predicted cost of $13.9 million is within a 99% confidence interval of $10.3 to $18.5 million.

Planning for Future Droughts

The prediction model was rerun for calibration parameter values corresponding to the optimal set (based on diagonal weighting matrix) as well as those for the UCL and LCL. Groundwater declines used to simulate potential future droughts were derived by scaling those from the recent drought by a set of factors (0.5, 1.0, 1.5 and 2.0). The results were adjusted by removing an amount for supply interruptions that were permanently resolved during the recent drought (56%). Therefore, simulating a repeat of the same drought (scaling factor of 1) would have less impact since many previously-impacted wells had been made more robust (pumps lowered or deeper wells installed) during the earlier drought. Results are summarized on Figures 42 a and b. These results assume that groundwater levels rebound from the previous drought before the future drought occurs. Impacts would be greater otherwise. (Use of the parameter values developed with the diagonal weighting matrix results in mean and upper confidence limit model predictions that are 10 to 25 percent higher than those obtained with parameter values developed with the full weighting matrix. Also, the lower confidence limit predictions are as much as 13 percent lower than predictions derived from the full weighting matrix.)

The shallowest and most vulnerable wells are impacted for all drought intensity levels while additional deeper wells are impacted for more severe droughts. Because there are fewer deep wells (Fig. 37d), the sensitivity of supply interruptions to drought generally decreases with drought intensity (Fig. 42a). However, costs to maintain supplies during drought and uncertainty in those predictions are more sensitive to drought intensity (Fig. 42b). Progressively deeper wells are impacted as drought intensity increases and these wells are more expensive to address than are shallower wells.

Groundwater level declines can occur seasonally and over multiple years. Because a significant amount of these declines are from resource extraction that exceeds replenishment rates, groundwater management policy choices affect groundwater levels and follow-on effects such as well supply interruptions. The type of results presented here can help guide groundwater management policy setting and planning to address third-party affects. For example, a policy that allows similar amounts of groundwater level decline as the recent drought (drought factor of 1 on Fig. 42 a and b) is expected to produce approximately 500 supply interruptions and approximately $6 million could be required to address the impacted wells.

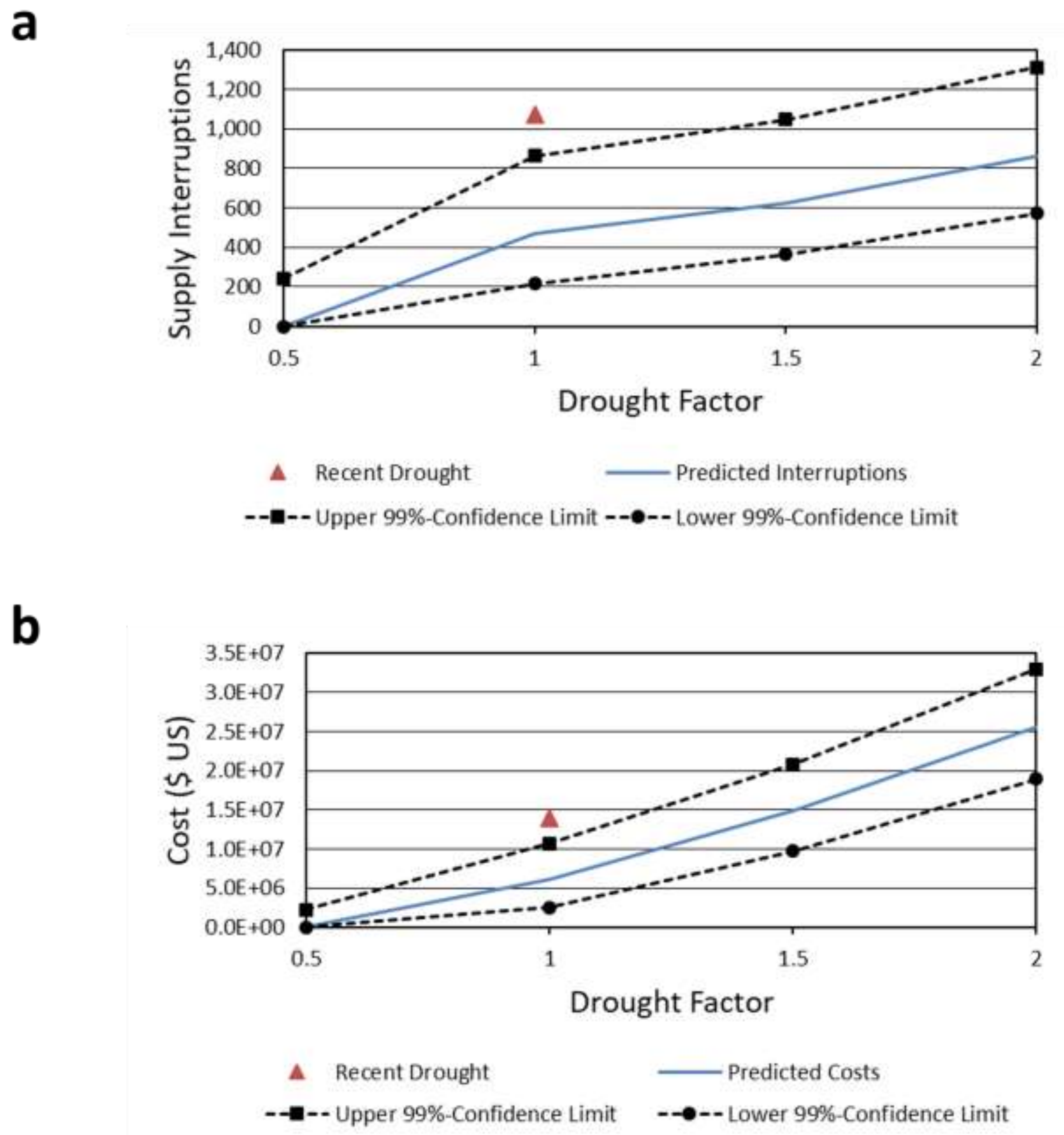


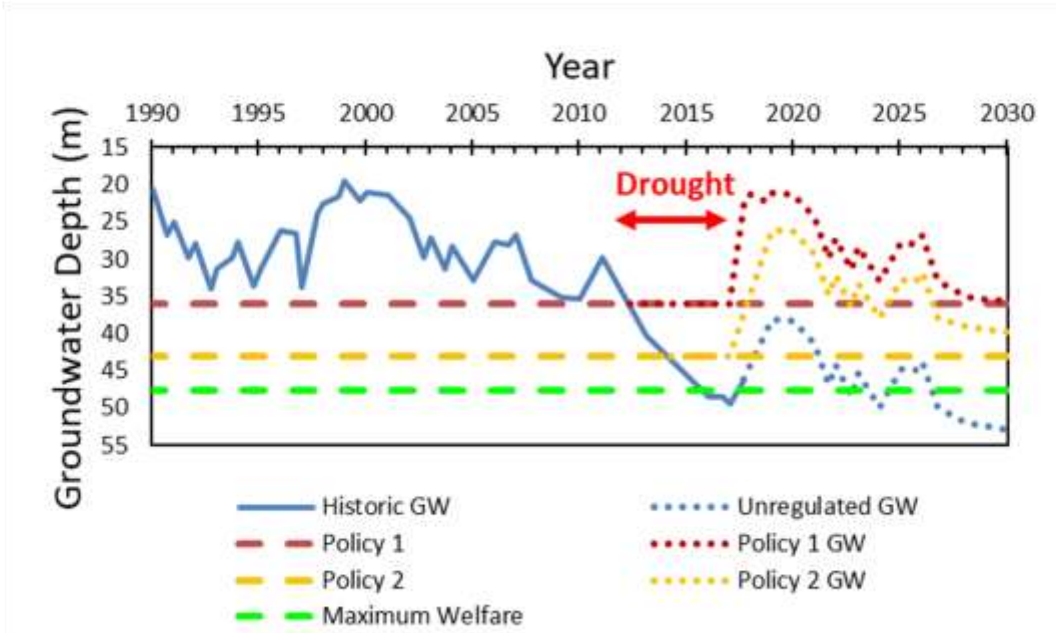
Figure 42: Simulation of future droughts: **a** supply interruptions and **b** costs. *Drought Factor* scales groundwater declines from the recent drought as a surrogate for the range of potential future droughts. The difference between *Recent Drought* and *Predictions for Drought Factor of 1* indicates the adjustment performed to account for actions taken during the recent drought to address supply interruptions

Implications for Groundwater Management

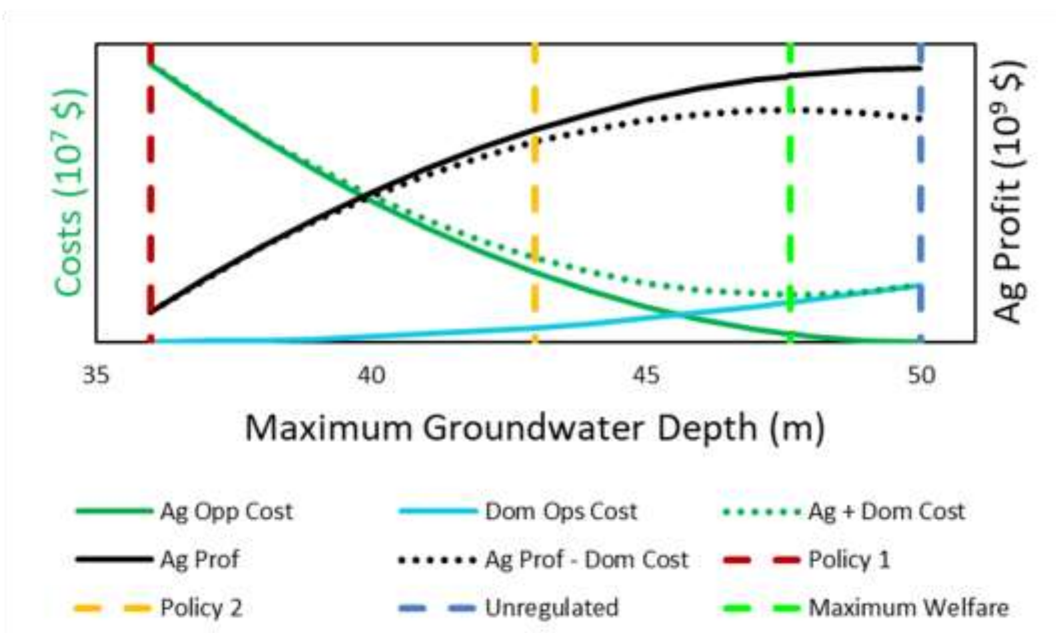
Increases in the depth to groundwater during drought often result from increased pumping to make up for the reduced surface water supply. With newly legislated groundwater management requirements in California, there is a need to develop policies on allowable groundwater level declines during drought. Stringent limits on groundwater level declines will reduce irrigated land, agricultural production, and farm profits. However, more permissive policies will cause economic costs for domestic well owners and users. Successful management must balance these competing interests.

Figures 43 a through c show a generalized example of how the information presented above could help in developing a management approach that considers the interests of both agricultural pumpers and domestic well owners. Using the historic groundwater level record (Fig. 43a), three policies regarding maximum allowable depth to groundwater are retrospectively considered for the recent drought. Policy 1 limits the groundwater level decrease to the previous lowest point (36 m in year 2010). Policy 2 specifies a 43 m limit mid-way between the previous low and the lowest point during the recent drought. The third policy has no regulation and allows groundwater levels to drop enough to meet all agricultural pumping demands (approximately 50 m in 2017). For policies 1 and 2, groundwater levels reaching the regulatory limit would trigger curtailment of pumping so no additional decline occurred and groundwater levels rebounded more quickly after the drought when pumping lessened.

a


Year
1990
1995
2000
2005
2010
2015
2020
2025
2030
Groundwater Depth (m)
15
20
25
30
35
40
45
50
55
Drought
Historic GW
Policy 1
Policy 2
Maximum Welfare
Unregulated GW
Policy 1 GW
Policy 2 GW

b


Costs (10^7 $)
Ag Profit (10^9 $)
35
40
45
50
Maximum Groundwater Depth (m)
Ag Opp Cost
Dom Ops Cost
Ag + Dom Cost
Ag Prof
Ag Prof - Dom Cost
Policy 1
Policy 2
Unregulated
Maximum Welfare

c

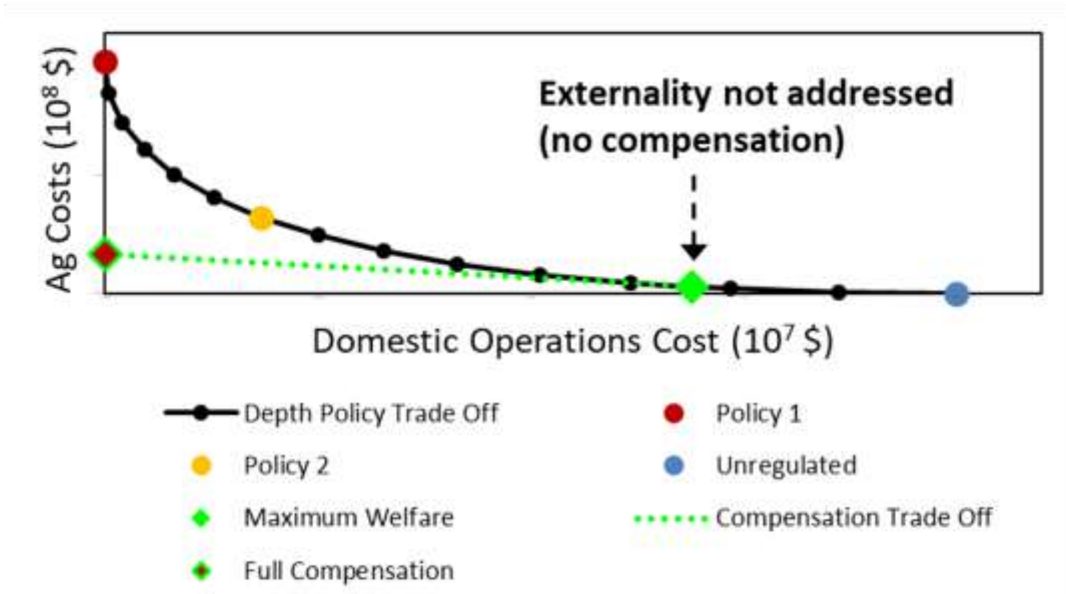


Figure 43: Example groundwater management policy analysis: **a** groundwater levels from potential policies, **b** policy costs and profits relative to maximum groundwater depth policy and **c** depth and compensation tradeoff curves. Data location for *groundwater depths* in Fig. 43a shown on Fig. 36a. *Groundwater depth* data source: Well 362539N1193051W001 CADWR Water Data Library http://www.water.ca.gov/waterdatalibrary/). *Ag* is agricultural. *Opp* is opportunity. *Dom* is domestic well. *Ops* is operations. *Prof* is profit. *Black and colored dots* on Fig. 43c correspond to groundwater depths at 1 m intervals. *Red dot* is 36 m and *blue dot* is 50 m.

The hypothetical policies can be evaluated based on the basic economics principles (i.e., Griffin 2006) by plotting 1) agricultural profit during the drought, 2) agricultural opportunity cost from limiting groundwater pumping through a maximum groundwater depth policy and 3) costs to maintain domestic wells resulting from water level declines (Fig. 43b). The quantities plotted are approximated for discussion while more detailed work is a topic of ongoing research. Agricultural profit is taken as ten percent of the cumulative revenues over the drought (approximately 35 billion dollars; Tulare County Agricultural Commissioner 2013 through 2017) minus an opportunity cost imposed by regulating the maximum groundwater depth. The opportunity cost is fashioned to show an expected diminishing return to scale. The cost for domestic wells is based on the 99-percent upper confidence limit for total cost presented in 'Planning for future droughts' ($18.5 million, Table 4) and shows increasing cost with groundwater level decline consistent with the above-referenced results (marginal cost to maintain supply increases with drought intensity, Fig. 42b). The different groundwater management policies appear on the plot as dashed lines. Intersection of the policy lines with the curves indicates economic impact for each policy. The most stringent Policy 1 decreases agricultural profits significantly and prevents the accrual of costs to domestic well owners while the unregulated case does the opposite (no agricultural opportunity costs other than some increased lift cost and significant costs for domestic wells). The medium-stringency Policy 2 has cost impacts to both agriculture and domestic wells.

Plotting agricultural opportunity costs against domestic well costs (Fig. 43c) shows the tradeoff in costs between interests for different policies (indicated as colored dots). This curve presents the spectrum of potential policies from the perspective of neutral economic efficiency. The policy tradeoff curve is Pareto optimal since moving from one potential policy to another results in gains for one party and losses for the other. A specific policy can be identified if the goal of maximizing total welfare of all stakeholders is assumed. Referring to the dotted lines on Figure 43b, the total cost (agricultural opportunity and well maintenance) is considered and subtracted from agricultural revenue to calculate profit. This approach accounts for the economic externality from agricultural pumping on domestic well owners. A new profit maximizing point results (intersection of dashed green line and dotted black line on Fig. 43b) which corresponds to a specific maximum groundwater depth policy (intersection of dashed green line and horizontal axis on Fig. 43b and green diamond on policy tradeoff curve on Fig. 43c). The new policy occurs between 47 and 48 m as seen in Fig. 43a. The water depth for this policy is near the historic low during the recent drought because the agricultural opportunity cost is so much greater than the domestic well cost. This disparity in costs affects the economic calculations that drive policy selection. Although maximizing the overall economic benefits would do little to ease impacts to domestic wells, future water level declines would be limited (blue dotted line on Figure 43a could not dip below green dashed line).

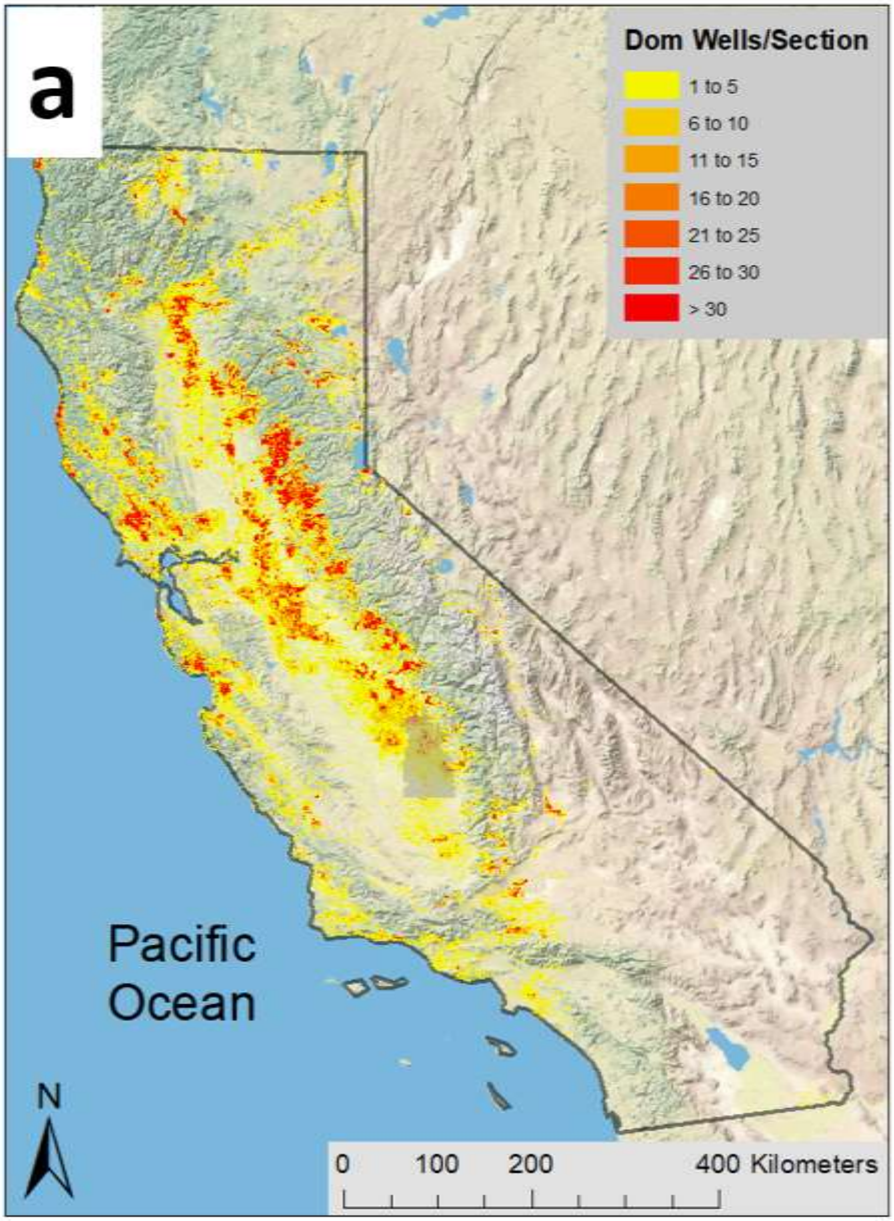
A shortcoming of this approach is lack of accounting for equity concerns such as 1) the distribution of cost among stakeholders relative to benefits received, 2) ability of each stakeholder to absorb costs and 3) impacts on human subsistence (need for drinking water v. value of additional industrial input). These considerations may lead policy development to become more stringent such that the burden on those supplied by domestic wells decrease and opportunity costs for agriculture increase (left shift along depth policy tradeoff curve from green diamond). An alternative approach could be for agriculture to follow the maximum welfare policy approach and provide compensation for domestic well costs.

The green dotted line on Fig. 43c indicates a constant level of maximum welfare (and a single maximum water depth policy) but varies from no compensation (green diamond) to full compensation (red diamond). This is a compensation tradeoff curve that represents a range of negotiated, or regulated, externality cost shifting from domestic well owners back to agricultural producers. The amount of agreed compensation (location along the green dotted line on Fig. 43c) might depend on considerations such as whether some of the groundwater level decline occurs from pumping farther away rather than from nearby agricultural pumpers in the study area. The compensation approach is obviously preferable for domestic well owners and would also be preferable for agricultural producers if it reduced opportunity cost compared to a more stringent policy (cost to agriculture from the compensation curve is below that on the policy curve for the particular policy being considered). The analysis here assumes the maximum groundwater depth policy only addresses cost to domestic wells from agricultural pumping. Other considerations, such as maximum depth limits related to land subsidence or future lift costs, also could be incorporated.

The compensation tradeoff curve (green dotted line on Fig. 43c) includes only economic factors that influence costs to agriculture and well owners; however, a negotiation over compensation could also involve strategic elements. Because groundwater management regulations in California allow for state-imposed curtailment of pumping if groundwater management is not jointly implemented by stakeholders, it may be possible for well owners to withhold agreement on groundwater management plans raising the potential of state intervention, more stringent groundwater management policy and higher costs to agriculture. Under these conditions, the compensation tradeoff curve shown on Figure 43c would provide a lower bound for a negotiated amount of compensation from agriculture to well owners.

Potential for Extending Analysis to Other Areas

Available data show 235,505 domestic and 34,278 agricultural wells across California (Fig. 44 a and b). Clearly, there is significant pumping and potential risk to well operations from groundwater level declines across the state. Application of the approach presented here may support overall groundwater planning.


a
Dom Wells/Section
1 to 5
6 to 10
11 to 15
16 to 20
21 to 25
26 to 30
> 30
Pacific
Ocean
N
0
100
200
400 Kilometers

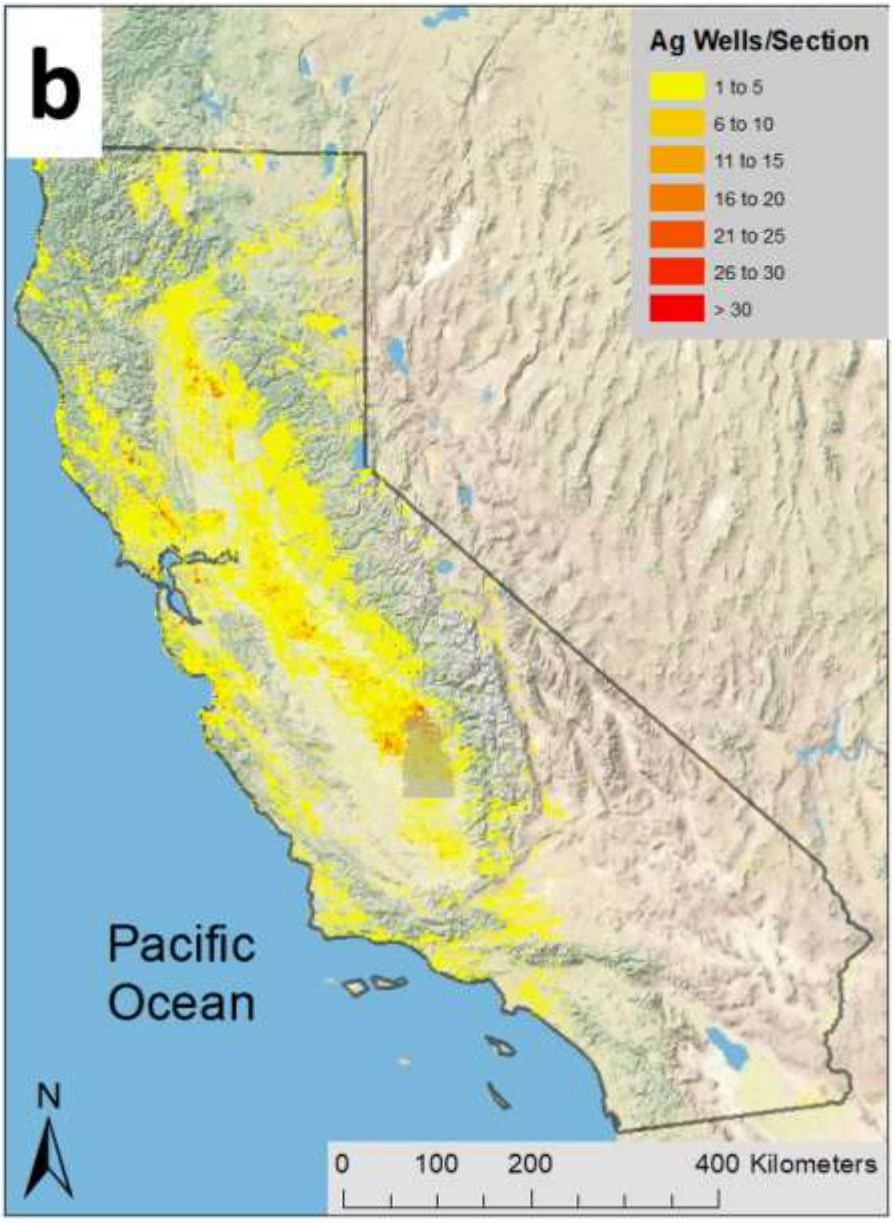


Figure 44: Numbers of wells in California: **a** domestic wells (Dom) and **b** irrigation wells (Ag). *Gray shaded area* indicates portion of Tulare County located on floor of the Central Valley (study area). Data source: CADWR Well completion report map application https://dwr.maps.arcgis.com/apps/webappviewer/index.html?id=181078580a214c0986e2da28f8623b37

Obtaining data for other areas to calibrate the well impacts model parameters (retirement age and separation distance) is a challenge. The most direct approach might be a bounded analysis, using a high and low value for each parameter, based on local knowledge of well conditions. However, the separation distance range would likely be large, causing an overly broad range in predicted impact and providing limited planning value. Another approach to specifying the separation distance value might be as follows:

1) Using a base case (non-drought) distribution of groundwater levels and well construction data for the area of interest, create a histogram of the water column height above the well bottom for the entire well population (no culling on retirement age).

2) Fit a statistical distribution function to the histogram. The function is likely to be a log-normal distribution because the distribution truncates at zero and would not include physically impossible negative values.

3) Pick the water column height at the function mode as the most probable occurrence.

4) Based on Figure 34a and Table 9, subtract from the most-probable water column the values used for pump operation (drawdown, submergence and operating margin). The remainder is an estimate for separation distance.

Figure 45 demonstrates the approach for the domestic wells in the study area. The separation distance estimated with the approach is 35 ft (11 m) while the calibrated value is 43 ft (13 m). If upper and lower bounds for retirement age were set, based on local knowledge, as 30 +/- 5 years (35 and 25 years for the upper and lower values), the range in impacts estimated for the recent drought would have been in the same range as those obtained through calibration (Table 13). While this approximation method for specifying model parameter values may allow useful results to be obtained, it is preferable to obtain calibration data for analyses whenever possible.

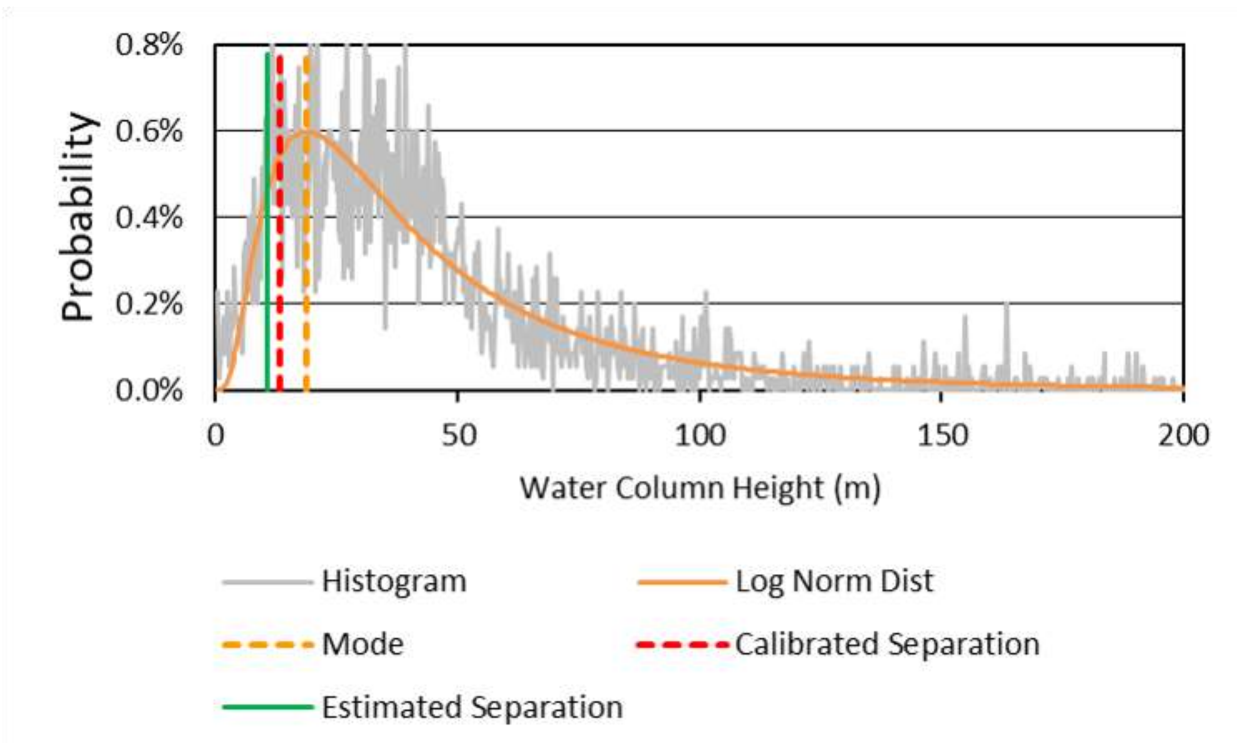


Figure 45: Estimating the separation distance value from calculated water column heights

Table 13 Comparison of model predictions for impacts and costs to domestic supply wells from 2012-2016 drought using calibrated and estimated parameter values

	Estimate	
	Low	High
Supply interruptions		
Calibrated parameters	815	1,461
Estimated parameters	535	1,042
	-34%	-29%
Total Cost		
Calibrated parameters	$\$10.3 \times 10^6$	18.5×10^6
Estimated parameters	$\$6.9 \times 10^6$	$\$13.6 \times 10^6$
	-33%	-27%

Low and *High* entries for the *Calibrated parameters* entries are based on 99% confidence interval calculations discussed in 'Uncertainty analysis'. *Low* and *High* entries for the *Estimated parameters* entries are based on the method described in 'Potential for extending analysis to other areas'. *Costs* are expressed in US dollars. *Percentages* represent differences from Calibrated values normalized by the calibrated values

Conclusions

It is common in arid areas for groundwater levels to fluctuate with seasonal agricultural pumping. More pronounced drawdown and refill cycles also occur when groundwater is used to replace surface water supply shortfalls during drought. However, groundwater management policies, or lack thereof, regarding the permissible range of groundwater level fluctuations has implications for shallower wells (most domestic and some agricultural wells) since they are more sensitive to groundwater level declines.

The full range of groundwater management options when planning for drought may include allowing groundwater uses that impact some users to maximize social welfare. However, the likely impacts of increased pumping on shallow well owners should be addressed in the planning process. One approach for addressing impacts would be to create a fund for maintaining shallow wells during droughts. Some corrective action may entail short-term supply supplementation and well maintenance such as lowering pumps, while other measures could include long-term well replacement. The type of analysis presented here can support this planning.

For the study area considered, pump lowering and well replacement are the most likely corrective actions with additional considerations for well screen clogging and extra pumping lift. Reoccurrence of drought and groundwater management circumstances similar to those of 2012 to 2016 are expected to result in some 500 domestic supply interruptions that would cost approximately $6 million to address. Recognizing that the available information is imperfect and underestimation of the impacts is likely, the upper confidence limits for the estimated impacts (approximately 900 supply interruptions and $11 million in costs) might be used as a more robust basis for planning. These estimated costs are an extremely small fraction of agricultural revenue ($35 billion over the five-year period) and profits for this region. Because the supply of well maintenance and construction labor may be inadequate during drought, at least some of the predicted impacts might be addressed before the impacts occur.

These results may provide useful insights for similar analyses in other areas where there are many domestic and shallower agricultural wells. The approach presented here could supplement measures to balance groundwater budgets according to identified metrics (i.e., groundwater depth and concentration thresholds). Achieving agreement among stakeholders on the metrics could be made easier by applying economic analysis that includes a range of impacts from pumping.

Acknowledgements

The author is grateful to the California Department of Water Resources for sharing an early version of the dataset regarding the geographic distribution of water supply well constructions, the Tulare County Office of Emergency Services for sharing information on the geographic distribution of domestic water supply outages, and the Tulare County Health and Human Services Agency, Environmental Health Services Division for sharing information well construction and destruction permitting. Thomas Harter is also appreciatively acknowledged for editorial and technical comments that improved this manuscript. This work was supported by the UC Office of the President's Multi-Campus Research Programs and Initiatives (MR-15-328473) through UC Water, the University of California Water Security and Sustainability Research Initiative. Additional support was provided by the California Department of Food and Agriculture (UC Agreement 16-0125), S.D. Bechtel Jr. Foundation and US Environmental Protection Agency (Assistance Agreement 83586701) through the UC Davis Center for Watershed Sciences.

Appendix

Following Gailey et al. (2016), information on calculating pump operations costs may be found in Helweg et al. (1983) as well as other references. The cost for operating a pump is calculated as:

$$OC = P\ T\ c \tag{46}$$

Where OC is operating cost for the well pump (\$),

P is pump power (kW)

T is total operation time (hr), and

c is cost per Kilowatt-hour ($/kW-hr)

Pump energy is calculated as:

$P = (0.746\ Q\ H)/(3{,}956\ E)$ (47)

Where Q is pumping rate (gpm)

H is total dynamic head (ft)

E is combined efficiency of the pump bowls and motor (%)

0.746 is a conversion factor from horse power to kW,

3,956 is a conversion factor from flow in gpm and total dynamic head in feet to horse power

H is further defined as:

$H = DTW + Q/SC$ (48)

Where DTW is depth to groundwater under non-pumping conditions (ft)

SC is the specific capacity of the pumped well (gpm/ft)

Total operation time is calculated as:

$T = V/(60\ Q)$ (49)

Where T is total operation time (hr)

V is total volume required (gal)

60 is a conversion factor from minutes to hours.

Substituting equations (47) through (49) into equation (46) yields:

$\mathrm{OC} = 3.14 \times 10^{-6}\ (DTW + Q/SC)\ (\mathrm{V\ c})/\mathrm{E}$ (50)

Information requirements for *DTW*, *Q* and *SC* are addressed in ‘Methods of analysis and data sources’. *V* is taken as the volume of water required for a family of four people during one year (approximately 400,000 gal per year) based on data for the study area region (CADWR 2014b). A location-specific value for *c* ($0.16 per kW-hr) is used (Electricity Local 2018). The value for *E* (0.63) is consistent with the author’s experience (Gailey et al. 2016). Variations in the value for *E* resulting from pumping rate fluctuations with groundwater depth are not considered.

References

Barlow PM and Reichard EG (2010) Saltwater intrusion in coastal regions of North America. Hydrogeology J (18) 2:247-260. doi:10.1007/s10040-009-0514-3

Barlow PM and Leakey SA (2012) Streamflow depletion by wells – understanding and managing the effects of groundwater pumping on streamflow. U.S. Geological Survey Circular 1376

Bates DM and Watts DG (1988) Nonlinear regression analysis and its applications. John Wiley and Sons, New York

Bentley LR (1997) Influence of the regularization weighting matrix on parameter estimates. Advances in Water Resources (20) 4: 231-247

Bredehoeft JD (2002) The water budget myth revisited: why hydrogeologists model. Ground Water (40) 4: 340-345

CADWR (2014a) Public update for drought response: groundwater basins with potential water shortages, gaps in groundwater monitoring, monitoring of land subsidence, and agricultural land fallowing. California Department of Water Resources. http://www.water.ca.gov/waterconditions/docs/DWR_PublicUpdateforDroughtResponse_GroundwaterBasins.pdf Accessed 5 July 2017

CADWR (2014b) California water plan update 2013. California Department of Water Resources. California Department of Water Resources. https://www.water.ca.gov/Programs/California-Water-Plan/Water-Plan-Updates Last accesses 9 April 2018

CADWR (2015) California's most significant droughts: comparing historical and recent conditions. California Department of Water Resources. file:///C:/Users/rob/Desktop/Well%20Impacts/Paper/California_Signficant_Droughts_2015_small.pdf Accessed 17 January 2018

CADWR (2017a) Groundwater information center interactive map application. California Department of Water Resources. https://gis.water.ca.gov/app/gicima/ Accessed 5 July 2017

CADWR (2017b) Well completion report map application. https://dwr.maps.arcgis.com/apps/webappviewer/index.html?id=181078580a214c0986e2da28f8623b37 Accessed 28 November 2017

CADWR (2018) Sustainable groundwater management website. California Department of Water Resources. https://www.water.ca.gov/Programs/Groundwater-Management/SGMA-Groundwater-Management/. Accessed 9 April 2018

Cooley RL and Vecchia AV (1987) Calculation of nonlinear confidence and prediction intervals for ground-water flow models. Water Resources Bulletin (23) 4: 581-599

County of Tulare (2017) Drought effects status updates. http://tularecounty.ca.gov/emergencies/index.cfm/drought/drought-effects-status-updates/ Accessed 5 July 2017

Döll P, Schmied HM, Schuh C, Portmann FT and Eicker A (2014) Global-scale assessment of groundwater depletion and related groundwater abstractions: combining hydrological modeling with information from well observations and GRACE satellites. Water Resources Research 50: 5698-5720. doi:10.1002/2014WR015595

Draper N and Smith H (1981) Applied regression analysis, Second edition. John Wiley and Sons, New York

Electricity Local (2018) Local electricity rates and statistics, Tulare electricity rates.

https://www.electricitylocal.com/states/california/tulare/ Accessed 10 January 2018

Erickson ML and Barnes RJ (2005) Well characteristics influencing arsenic concentrations in ground water. Water Research 39(16): 4029-4039. doi:10.1016/j.watres.2005.07.026

Famiglietti JS, Lo M, Ho SL, Bethune J, Anderson KJ, Syed TH, Swenson SC, de Linage CR and Rodell M (2011) Satellites measure recent rates of groundwater depletion in California's Central Valley. Geophysical Research Letters 38: L03403. doi:10.1029/2010GL046442

Farr TG, Jones C and Liu Z (2015) Progress report: subsidence in the Central Valley, California. Prepared for the California Department of Water Resources. http://water.ca.gov/groundwater/docs/NASA_REPORT.pdf Accessed 5 July 2017

Faunt CC (ed.) (2009) Groundwater availability of the Central Valley Aquifer, California, US Geol Surv Prof Pap 1766

Feinstein L, Phurisamban R, Ford A, Tyler C and Crawford A (2017) Drought and Equity in California, Pacific Institute, Oakland, CA http://pacinst.org/publication/drought-equity-california/ Accessed 28 September 2017

Foster SSD and Chilton PJ (2003) Groundwater: the processes and global significance of aquifer degradation. Philos Trans R Soc B, 358(1440):1957-1972. doi:10.1098/rstb.2003.1380

Gailey RM, Arnold BJ, Kwon AJ, Medellín-Azuara J and Lund JR (2016) The effect of declining groundwater levels on supply well operations, Appendix A of economic analysis of the 2016 California drought on agriculture, a report for the California Department of Food and Agriculture, Center for Watershed Sciences, University of California Davis. https://watershed.ucdavis.edu/droughtimpacts Accessed 26 September 2017

Gailey RM (2017) Inactive supply wells as conduits for flow and contaminant migration: conditions of occurrence and suggestions for management, Hydrogeol J 25(7):2163-2183, doi: 10.1007/s10040-017-1588-y

Gailey RM (2018) Using geographic distribution of well-screen depths and hydrogeologic conditions to identify areas of concern for contaminant migration through inactive supply wells, Hydrogeol J (initially on-line only), doi:10.1007/s10040-018-1734-1

Glennon RJ (2002) Water follies: groundwater pumping and the fate of America's fresh waters. Island Press, Washington, DC

Griffin RC (2006) Water resources economics, the analysis of scarcity, policies, and projects. MIT Press, Cambridge, MA

Hanak E, Lund J, Arnold B, Escriva-Bou A, Gray B, Green S, Harter H, Howitt R, MacEwan D, Medellín-Azuara, Moyle P and Seavey N (2017) Water stress and a changing San Joaquin Valley, Public Policy Institute of California, March 2017. http://www.ppic.org/main/publication.asp?i=1224 Accessed 5 July 2017

Hill MC and Tiedeman CR (2007) Effective groundwater model calibration. John Wiley and Sons, Hoboken, NJ

Helweg OJ, Scott VH and Scalmanini JC (1983) Improving pump and well efficiency. American Water Works Association,, Denver, CO

Houben GJ and Treskatis C (2007) Water Well Rehabilitation and Reconstruction. McGraw Hill, New York

Jenkins CT (1969) Electric-analog and digital-computer model analysis of stream depletion by wells. Ground Water (6) 6: 27-34.

Johnson TD and Belitz K (2015) Identifying the location and population served by domestic wells in California. Journal of Hydrology: Regional Studies 3(2015) 31-86, doi:10.1016/j.ejrh.2014.09.002

Johnson TD and Belitz K (2017) Domestic well locations and population served in the contiguous U.S.: 1990. Science of the Total Environment 607-608 (2017) 658-668, doi:10.1016j.scitotenv.2017.07.018

Konikow LF (2013) Groundwater depletion in the United States (1900-2008). U.S. Geological Survey Scientific Investigations Report 2013-5079

Mansuy N (1999) Water Well Rehabilitation, A Practical Guide to Understanding Well Problems and Solutions. Lewis, New York

Moran T, Choy J and Sanchez C (2014) The hidden costs of groundwater overdraft, Understanding California's groundwater, Stanford Water in the West. http://waterinthewest.stanford.edu/groundwater/overdraft/ Accessed 5 July 2017

Ostrom E (1990) Governing the commons: the evolution of institutions for collective action. Cambridge University Press, Cambridge

Perrone D and Jasechko S (2017) Dry groundwater wells in the western United States. Environmental Research Letters 12(10) 104002. doi:10.1088/1748-9326/aa8ac0

Smith SA and Comeskey AE (2010) Sustainable Wells: Maintenance, Problem Prevention, and Rehabilitation. CRC Press, Boca Raton, FL

Smedley PL and Kinniburgh DG (2002) A review of the source, behaviour, and distribution of arsenic in natural waters. Applied Geochemistry 17(5): 517-568.

Sneed M, Brandt J and Solt M (2013) Land subsidence along the Delta-Mendota Canal in the northern part of the an Joaquin Valley, California 2003-10. U.S. Geological Survey Scientific Investigations Report 2013-5142

Sophocleous M (2000) From safe yield to sustainable development of water resources – the Kansas experience. Journal of Hydrology 235: 27-43. doi:10.1016/S0022-1694(00)00263-8

State of California (2017) Household water supply shortage reporting system. https://mydrywatersupply.water.ca.gov/report/publicpage Accessed 5 July 2017

Stevenson GG (1991) Common property economics, a general theory and land use applications. Cambridge University Press, Cambridge

Theis CV (1940) The source of water derived from wells: essential factors controlling the response of an aquifer to development. Civil Engineer 10: 277-280

Tulare County Agricultural Commissioner (1998) Tulare County crop and livestock report, 1997. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (1999) Tulare County crop and livestock report, 1998. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2000) Tulare County crop and livestock report, 1999. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2001) Tulare County crop and livestock report, 2000. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2002) Tulare County crop and livestock report, 2001. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2003) Tulare County crop and livestock report, 2002. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2004) Tulare County crop and livestock report, 2003. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2005) Tulare County crop and livestock report, 2004. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2006) Tulare County crop and livestock report, 2005. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2007) Tulare County crop and livestock report, 2006. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2008) Tulare County crop and livestock report, 2007. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2009) Tulare County crop and livestock report, 2008. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2010) Tulare County crop and livestock report, 2009. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2011) Tulare County crop and livestock report, 2010. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2012) Tulare County crop and livestock report, 2011. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2013) Tulare County crop and livestock report, 2012. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2014) Tulare County crop and livestock report, 2013. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2015) Tulare County crop and livestock report, 2014. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2016) Tulare County crop and livestock report, 2015. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

Tulare County Agricultural Commissioner (2017) Tulare County crop and livestock report, 2016. http://agcomm.co.tulare.ca.us/ag/index.cfm/standards-and-quarantine/crop-reports1/ Accessed 17 January 2018

UNESCO (2012) Managing water under uncertainty and risk, United Nations world water development report 4, vol 1. United Nations Educational, Scientific and Cultural Organization, Paris. http://www.unesco.org/new/en/natural-sciences/environment/water/wwap/wwdr/wwdr4-2012/ Accessed 5 July 2017

USGS (2017) Drought impacts website. United States Geological Survey. https://ca.water.usgs.gov/data/drought/drought-impact.html Accessed 5 July 2017

Van Beek CGEM (2012) Cause and Prevention of Clogging of Wells Abstracting Groundwater from Unconsolidated Aquifers. IWA, London

Vecchia AV and Cooley RL (1987) Simultaneous confidence and prediction intervals for nonlinear regression models with application to a groundwater flow model. Water Resources Research 23 (7): 1237-1250

Wada Y, van Beek LPH and Bierkens MFP (2012) Nonsustainable groundwater sustaining Irrigation: a global assessment. Water Resources Research 48: W00L06. doi 10.1029/2011WR010562

Young RA and Loomis JB (2014) Determining the economic value of water, concepts and methods, second edition. RFF Press, New York

CHAPTER 5 – MAXIMIZING ON-FARM GROUNDWATER RECHARGE WITH SURFACE RESERVOIR RELEASES: A PLANNING APPROACH AND CASE STUDY

Abstract

A hydro-economic approach for planning on-farm managed aquifer recharge (MAR) is developed and demonstrated for two contiguous sub-basins in California's Central Valley. The amount and timing of water potentially available for recharge is based on a reoperation study for a nearby surface water reservoir. Privately-owned cropland is intermittently used for recharge with payments to landowners that compensate for perceived risks to crop health and productivity. Using all cropland in the study area would have recharged approximately 3,900 TAF (4.8 km^3) over the 20-year analysis period. Analysis shows that limits to recharge effectiveness are expected from 1) temporal variability in recharge water availability, 2) variations in infiltration rate and few high-infiltration recharge sites in the study area and 3) recharged water escaping from the groundwater system to surface water and adjacent sub-basins. Depending on crop tolerance to ponding depth, these limitations might be reduced by 1) raising berm heights on higher infiltration rate sites and 2) creating dedicated recharge facilities over high-infiltration rate sites.

Introduction

Groundwater is an important water supply for more than two billion people around the world (UNESCO 2012). It also provides more than 40 percent of the irrigation supply for global agricultural production (UNESCO 2015) on approximately 500 million ha of cropland (Portmann et al. 2010; GFSAD30 2017; World Bank 2018). Given such intense use, it is not surprising that depletion of the resource is occurring in many parts of the world (Wada et al. 2012; Döll et al. 2014) including the United States (Konikow 2013) and California (Famiglietti et al. 2011; Farr et al. 2015). Excessive groundwater extraction can decrease water levels, reduce surface water flows, cause seawater intrusion, spread contaminants, and cause land subsidence (Foster and Chilton 2003; Barlow and Reichard 2010; Barlow and Leake 2012; Konikow 2013; Sneed et al. 2013; Moran et al. 2014; USGS 2017).

Sustainable resource management requires a combination of reduced extraction and increased recharge (Scanlon et al. 2016). Some reduced extraction may occur by increasing water use efficiency (Howell 2001; Tindula et al. 2013); however, pronounced rates of extraction in many areas will likely necessitate modifying cropping patterns and fallowing cropland to address problems from over-pumping (Foster and Chilton 2003). Such changes will cause economic distress and likely bring political resistance. While avoiding strong measures to correct groundwater budget imbalances may not be possible, disruption might be reduced by increasing recharge where possible.

Elements for successful artificial recharge projects have been reviewed in detail (Bouwer 2000; Gale 2005; Dillon et al. 2009; Scanlon et al. 2016; Perrone and Rhode 2016; Hank et al. 2018) and may be programmatic or site-specific. Programmatic elements include sourcing, conveyance and placement of recharge water. Sources of recharge water may include urban storm water runoff and recycled water as well as, notwithstanding water rights considerations, stormflows from streams and releases from reoperated surface water reservoirs. Overcoming potential limitations regarding conveyance from source to recharge areas is essential. Considerations include access to either existing canals and ditches, or the land required to construct these structures, as well as routing and capacity specifications. Options for placing water in recharge facilities range from constructing dedicated basins to repurposing existing gravel pits. The recharge water could also be released to lands primarily used for other purposes but available on a seasonal basis, such as sandy-bottomed drainage features, unlined canals and ditches, or croplands. Site-specific details include: 1) location relative to conveyance and favorable hydrogeology, 2) topography of the ground surface and presence of existing berms, 3) type of irrigation technology present, 4) timing of site availability relative to water available for recharge and 5) cost to use the land under purchase, rent or option arrangements.

Site-specific details regarding favorable hydrogeology directly relate to characteristics of the groundwater basin under consideration. Spatial variability of infiltration capacity is heavily influenced by the hydraulic conductivities of the soil and shallow geology (O'Geen 2015) as well as interconnectedness of higher hydraulic conductivity deposits at depth (Fogg et al. 2000; Weissmann et al. 2004). Groundwater storage space is driven by the unsaturated zone thickness and its variations across the basin. The fate of recharged water over time relative to the recharge location can also be important (Niswonger et al. 2017). Recharge at some locations may offset local pumping and increase groundwater storage. At other locations, water entering the subsurface can quickly discharge from the groundwater system to surface water or flow across basin boundaries that are based on governance rather than physical processes.

Data on the performance of managed aquifer recharge (MAR) on croplands is limited and largely focuses on California and western USA. Dokoozlian et al. (1987) conducted a four-year pilot study flooding vineyards in the San Joaquin Valley of California during grapevine dormancy, observed no impact on crop yield, and concluded that the approach was viable for MAR. Bachand et al. (2014, 2016) performed a single-season pilot study for on-farm flood flow capture and recharge, also in the San Joaquin Valley with both perennial (vineyards and orchards) and annual crops. They observed no impacts to crop yield and estimated the unit cost for the on-farm recharge as approximately 3 to 30 times cheaper than surface water storage or dedicated recharge basins. Dahlke et al. (2018) investigated effects of winter flooding on established alfalfa fields at two locations in the Sacramento Valley of California and found that significant amounts of water (2 to 26 ft, or 1 to 8 m) could be applied without decreasing crop yield. Additional unpublished studies indicate that 1) almonds may tolerate at least 2 ft (0.6 m) of cumulative applied recharge water in a season without detrimental effects (H. Dahlke personal communication) and 2) some grapes have shown little to no productivity decline after more than 20 ft (6 m) of recharge in one season (D. Mountjoy personal communication).

Some analysis on scaling up on-farm recharge for larger-scale groundwater management has also occurred. Harter and Dahlke (2014) discussed the potential for on-farm recharge projects to improve conditions in California where groundwater has been stressed by overuse and drought. O'Geen et al. (2015) considered requirements for successful projects and presented a spatially explicit soil agricultural groundwater banking index (SAGBI) for recharge project suitability on agricultural lands in California. Niswonger et al. (2017) examined potential benefits from on-farm MAR (Ag-MAR) for a hypothetical groundwater sub-basin in the semi-arid western USA. They developed an integrated surface water diversion and subsurface flow model to simulate recharge operations and benefits to the groundwater system over a 24-year period. Scenarios considered recharge water from snowmelt in excess of water rights during wet years applied to croplands during two winter months each year. Among other points, the work concluded that increases in groundwater storage from Ag-MAR operations 1) were spatially related to variations in groundwater depth and withdrawals across a basin as well as proximity to natural discharge areas and 2) supported greater pumping supplies for agriculture.

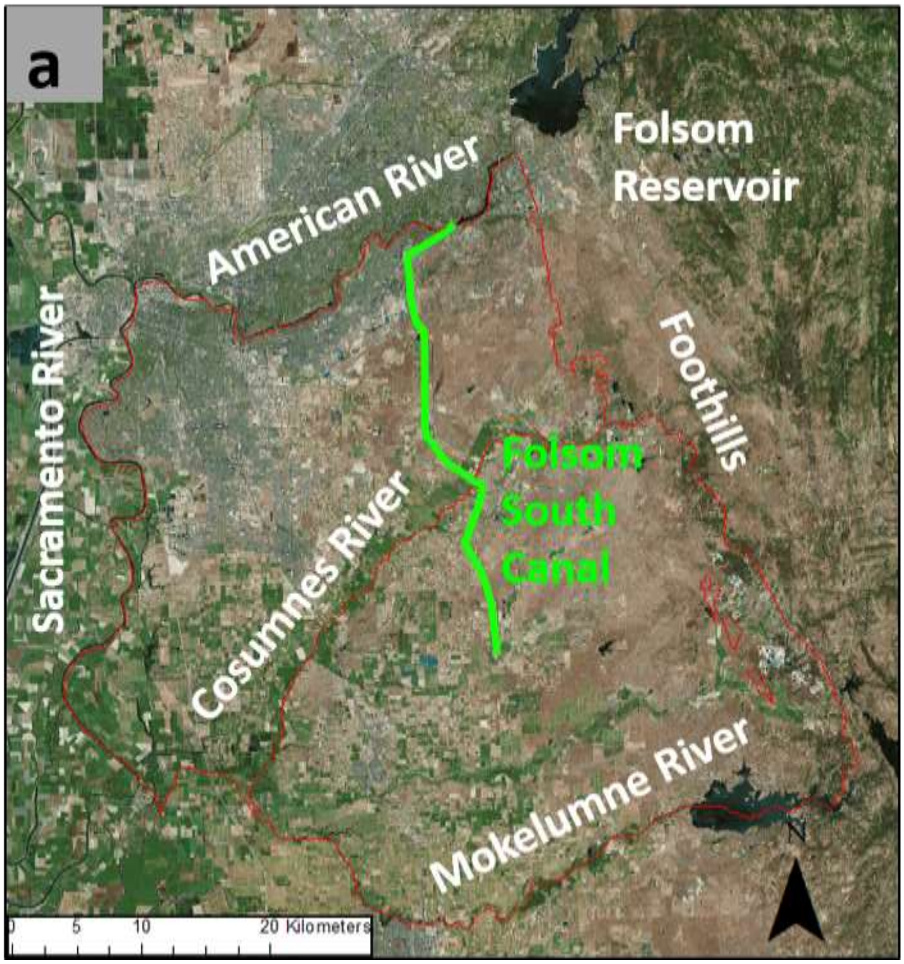
This work addresses planning-level analysis of Ag-MAR using water from reservoir reoperation during wet years for periodic off-season flooding of croplands during winter months. The analysis in the following sections expands on previous work by including 1) consideration of recharge water from reservoir reoperation, 2) evaluation of recharge water sourcing, cropland characteristics and groundwater hydrology for a site-specific setting and 3) demonstrating a hydro-economic optimization approach that simulates separate decisions for land access and water delivery in the performance of Ag-MAR.

Study Area and Background

The regional scale analysis is conducted for a semi-arid part of California, USA (Fig. 46) that has conditions fairly common for many parts of the globe. The two groundwater sub-basins in the study area are part of the much larger Central Valley groundwater system (Bertoldi et al. 1991) with an interfingered assemblage of alluvial and flood-basin deposits of local maximum depth exceeding 1,000 ft (300 m; Faunt 2009; RBI/WRIME 2011). Many of the sub-basin boundaries shown in Figure 47a are arbitrarily based on surface water features, and the southern boundary has recently been adjusted northward to accommodate current groundwater management efforts (CADWR 2016).

The 525,000 ac (212,000 ha) study area has a mix of urban (18%), agricultural (27%), wetland (4%) and undeveloped (51%) land uses (Fig. 47b). Over 90 percent of the total water use in the study area is supplied by groundwater (RBI/WRIME 2011). Moreover, approximately 41 percent of the agricultural acreage is planted as vineyards and orchards (calculations from data presented on Fig. 47b). This investment in perennial crops hardens water demand and intensifies groundwater extraction during droughts.

Fig. 46: Study area in California, USA. *Gray shaded area* indicates study area


a
Folsom
Reservoir
American River
Sacramento River
Foothills
Folsom
South
Canal
Cosumnes River
Mokelumne River
0 5 10 20 Kilometers
N

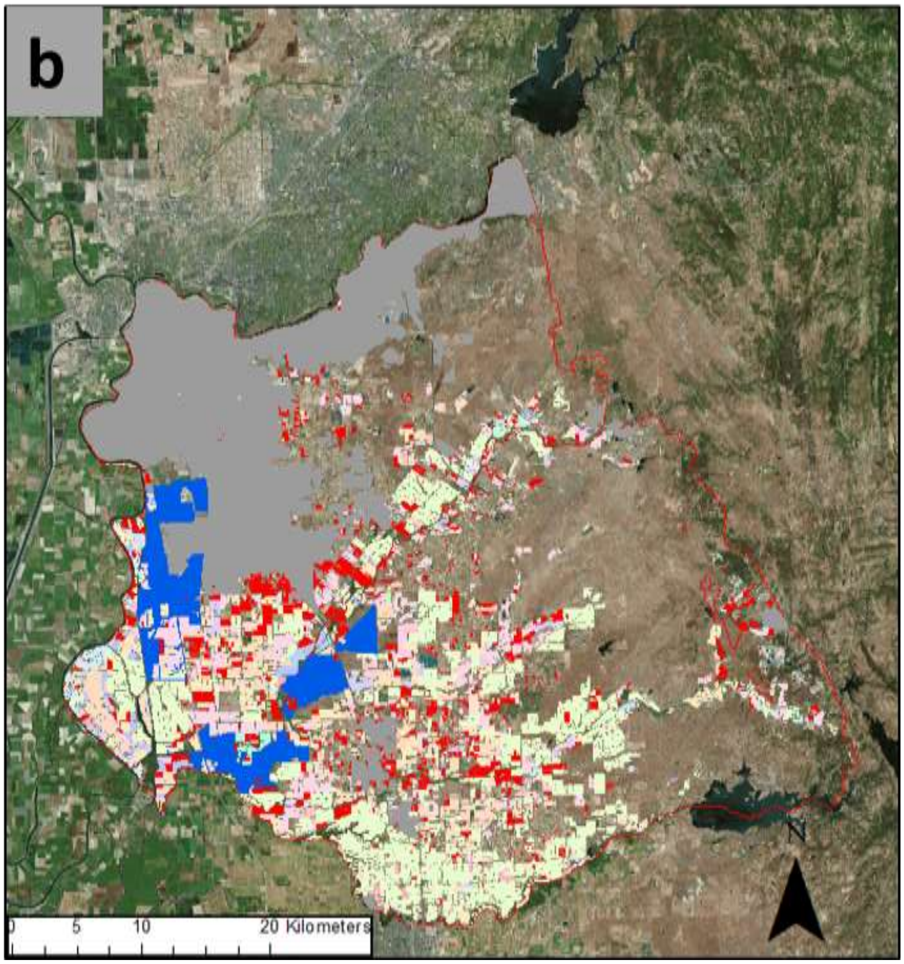


Fig. 47: Sub-basin characteristics: **a** boundaries and **b** land use. Land uses: *gray shading* is urban, *blue shading* is wetland, *lighter colors* are agricultural, *red shading* is idle land during drought in 2014, *unshaded areas* are undeveloped. Data source: https://gis.water.ca.gov/app/CADWRLandUseViewer/

The spatial distribution of recent water levels indicates localized depressions from extractions far exceeding groundwater recharge (Fig. 48). Groundwater levels have dropped as much as 60 ft (20 m) over the past several decades so that surface water frequently becomes disconnected from groundwater system and drains into the subsurface. The lower reaches of the Cosumnes River, in the central part of the study area (Fig. 47a), are dry 85 percent of the time (RBI/WRIME 2011).

New regulations for sustainable groundwater management in California require that this chronic lowering of groundwater levels and depletion of storage be addressed through active measures (Harter 2015; CADWR 2018a). While restoration of surface water baseflow in the study area may not be required by the regulations, there is interest in maintaining, and possibly improving, groundwater support of surface water flows (Hersh-Burdick 2008; RMC 2014).

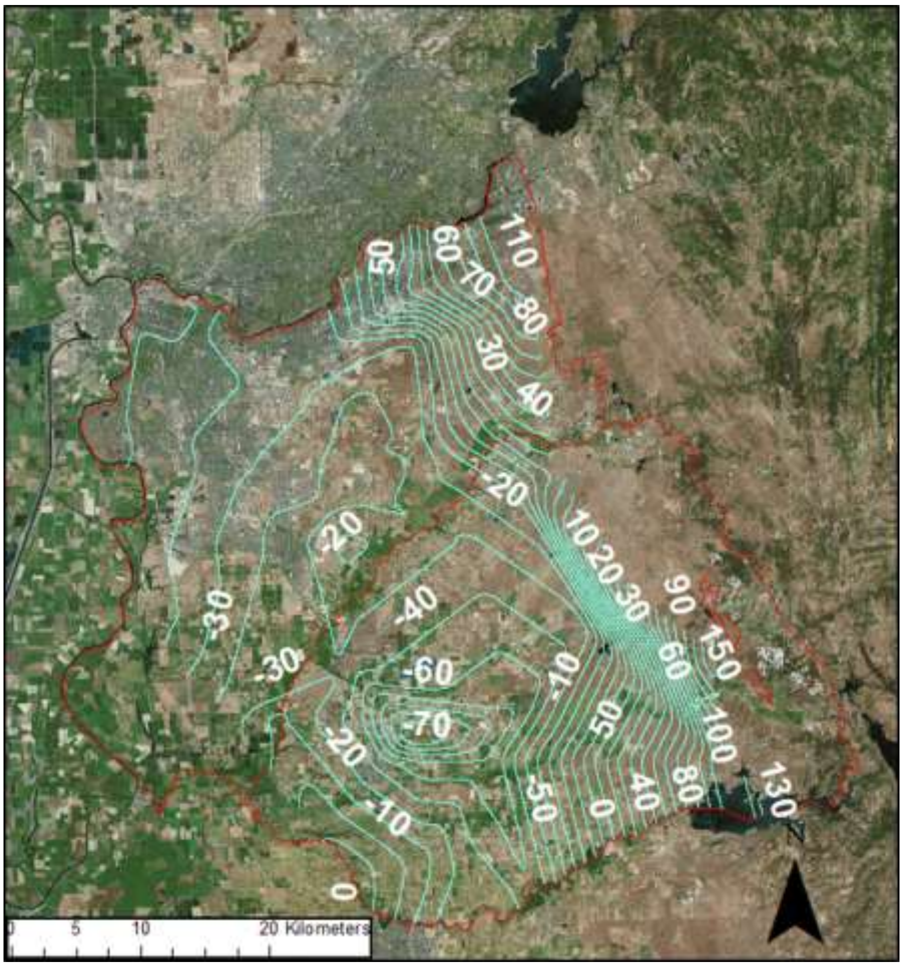


Fig. 48: Groundwater levels in the study area for Fall 2017. *Contours* are in ft msl. Data source: https://gis.water.ca.gov/app/gicima/. Land surface elevation ranges from approximately 0 ft msl in the southwest to 400 ft msl in the northeast

Consistent with recent analysis (Kocis and Dahlke 2017; CADWR 2018b), local stakeholders are interested in harvesting runoff from high-precipitation events for recharging groundwater. One option is reoperation of Folsom Reservoir (Fig. 47a) to release extra water in advance of significant rain events (Goharian et al. 2016). The recharge water might be applied through a portfolio of the options noted above; however, use of on-farm recharge (CADWR 2017; RMC 2015) alone could achieve a potentially significant amount of aquifer recharge using the 140,000 ac (57,000 ha) of croplands in the study area (Fig. 47b). This work presents a planning-level analysis of what might be possible. While infrastructure construction costs are not considered, the results of this work might encourage further evaluation of necessary investments.

Methods of Analysis

A retrospective analysis is conducted to evaluate the range of improvements in groundwater system state (i.e., groundwater elevations and storage as well as baseflow to surface water) that might have occurred for the study area from an Ag-MAR recharge program. Recharge water is from simulated reoperation of Folsom Reservoir with delivery through the Folsom South Canal (Fig. 47a) consistent with capacity limitations (Goharian et al. 2016) over a 20-year period that covers water years 1984 through 2003 (October 1983 through September 2003). The timing and amounts of surface water delivered to croplands for recharge application is prescribed by a linear programming model that combines available information regarding surface water and groundwater hydrology with the spatial distribution of croplands. Groundwater recharge is simulated with a surface water/groundwater model that incorporates existing land uses, surface water deliveries and groundwater demands over the period considered (Brush et al. 2013).

Identifying Recharge Application Schedules

This analysis applies a formulation of simulation-optimization (Singh 2014) to MAR. Previous work includes Mushtaq et al. (1994) who simulated unsaturated flow from individual recharge basins and applied nonlinear programming to identify optimal loading schedules for maximizing recharge volume. Marques et al. (2010) included decisions for recharge area allocation and water volume application as part of a two-stage, quadratic programming analysis that maximized crop profits. Hao et al. (2018) used a genetic algorithm to maximize recharge volume while meeting constraints on groundwater elevations. To the best of the author's knowledge, the approach presented here is new in that it combines elements of recharge basin and groundwater hydraulics with economic considerations at a regional scale. The foundation of the linear programming approach is based on the study area hydrology which is adapted to include economic considerations regarding land use. A hydrologic formulation is presented as an explanatory step in developing the full hydro-economic formulation.

Initial Hydrologic Formulation

Assuming that all cropland would be available to recharge groundwater and ignoring economic considerations, the recharge water application scheduling is determined as follows:

$$\max_{\mathbf{RV}} \quad Z = \sum_{n=1}^{N} \sum_{t=1}^{T} RV_{\mathrm{n,t}} \tag{51}$$

Subject to:

$$\sum_{n=1}^{N} RV_{\mathrm{n,t}} \leq WAR_{\mathrm{t}} \qquad \text{for all } t \tag{52}$$

$$RV_{n,t} \leq UB_n \qquad \text{for all } n, t \qquad (53)$$

$$UB_n = [(K_{scale}\, HB_n\, A_n)(I_n/H_0 + \ln(\varepsilon))]/[1\text{-}e^{-(I_{n,t}/H_0 + \ln(\varepsilon))}] \qquad \text{for all } n \qquad (54)$$

$$GWE_{i,t} \leq GSE_i - FB_i \qquad \text{for all } i, t \qquad (55)$$

$$GWE_{i,t} = H_{i,t} + \sum_{n=1}^{N}\sum_{t=1}^{T} M_{n,i,t}\,(RV_{n,t}/RVu)\,(FD_1/FD_0) \qquad \text{for all } i, t \qquad (56)$$

$$RV_{n,t} \geq 0 \qquad \text{for all } n, t \qquad (57)$$

Where **RV** is the set of recharge volumes to be optimized over space and time

Z is the total recharge volume over the planning horizon

n is the spatial index corresponding to a potential recharge location

N is the total number of potential recharge locations in the study area

t is the temporal index corresponding to the month within the planning period

T is the total number of months in the planning period

$RV_{n,t}$ is the recharge volume at a location and time

WAR_t is the water available for recharge at a time

UB_n is the upper bound on recharge volume at a location

K_{scale} is a scaling factor that accounts for effective vertical hydraulic conductivity of the soil and underlying geology

HB_n is the berm height for a potential cropland recharge location

A_n is the area of cropland at a potential recharge location

I_n is the reference infiltration rate at a potential recharge location

H_0 is the ponding depth associated with the reference infiltration rate

$\mathcal{E}$ is a small increment greater than zero

$GWE_{i,t}$ is the groundwater elevation at a control location and time

GSE_i is the ground surface elevation at a control location

FB_i is the required groundwater freeboard at a control location

i is the spatial index corresponding to a groundwater elevation control location

$H_{i,t}$ is the background groundwater transient head response to unmanaged stresses at a control location and time

$M_{n,i,t}$ is the expected groundwater transient head response (mounding) at control location i and time t in response to potential recharge at location n

RVu is the unit recharge volume used to generate M

FD_1 is the fraction of recharge water delivered net of evaporation during conveyance considered for a particular scenario

FD_0 is the fraction of recharge water delivered net of evaporation during conveyance assumed when generating M

The formulation objective, equation (51), maximizes the volume of water recharged over the planning horizon subject to a set of operational constraints. The total volume of water recharged in any period t cannot exceed the water available for recharge (WAR), equation (52). WAR is derived from a reoperation of Folsom Reservoir to provide additional water during November through March each year (Goharian et al. 2016). The analysis is based on a perfect foresight formulation which provides an upper bound for recharge water available from the reservoir. A static upper bound on the volume of water recharged at a particular location, equations (53 and

54), is based on local infiltration capacity and field berm height through an analytical ponding and drainage model described in the appendix. Equations (55 and 56) dynamically constrain the magnitude of recharge decisions as a result of a cap on groundwater elevation to avoid water-logging of soil. This constraint is tied to the buildup and redistribution of recharged water as a result of groundwater flow and is described further in the appendix. Negative recharge decisions are prevented with equation (57). There are 67 potential recharge locations (N = 67) corresponding to the number of groundwater model elements in the study area, 240 monthly time periods (T = 240) over the 20-year planning horizon and 18 groundwater elevation control locations (I = 18). The groundwater elevation control locations are shown in Figure 49.

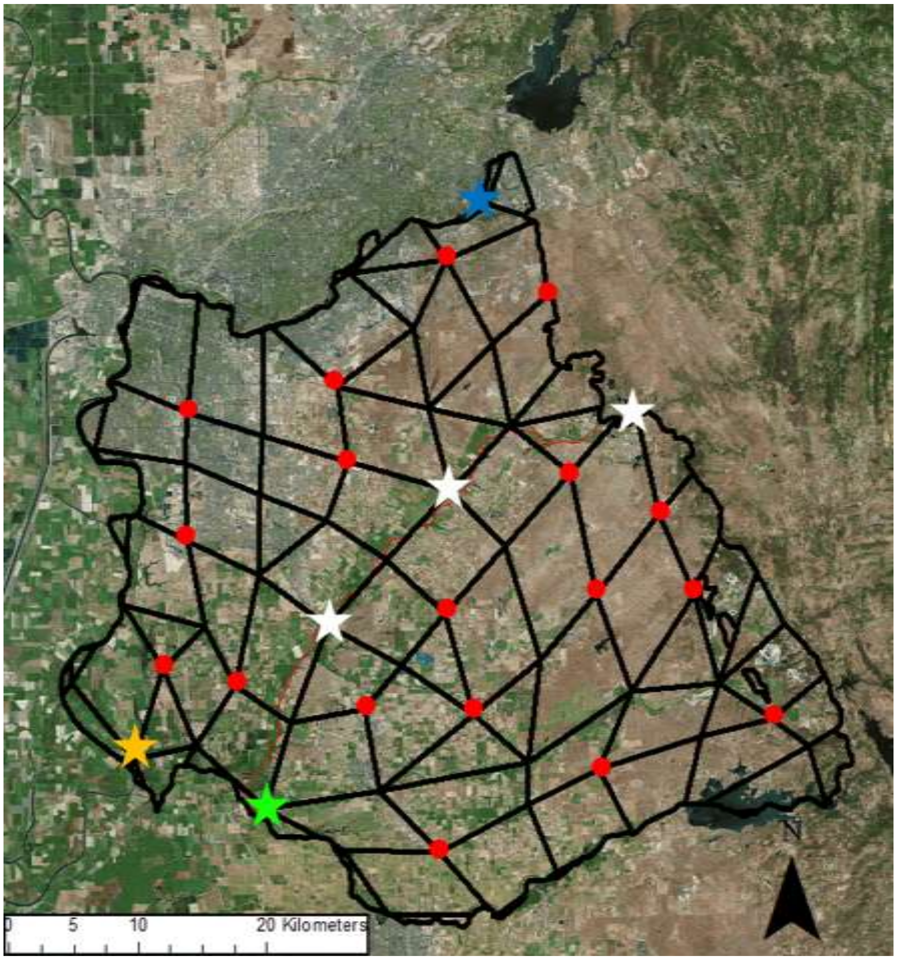


Fig. 49: Groundwater model elevation control and surface water flow locations. *Black lines* are model element boundaries. *Red dots* are groundwater elevation control locations. *Stars* are surface water flow evaluation locations (*blue*: American River, *white*: Cosumnes River, *gold*: Sacramento River, *green*: confluence of Cosumnes and Mokelumne rivers)

Hydro-Economic Formulation

Using cropland for groundwater recharge operations results from two separate sets of decisions made by a groundwater management agency: 1) acquiring access to specific lands for recharge operations and 2) subsequently delivering certain volumes of water to those lands. Land access decisions are made based on costs (rents) required by private landowners, funds available to a groundwater management agency and the infiltration capacity of different parcels. Deliveries of water to specific parcels are decided based on the lands made available through financial agreements and the infiltration capacities of the different lands. From this perspective, the previously noted decision variable RV becomes the product of a constant and two decision variables:

$$RV = A\ RA\ D \tag{58}$$

Where A is the area potentially available for recharge at a location (a constant)

RA is the relative area, ranging in value from 0 to 1, offered for use in recharge operations by the landowner

D is the amount of water, expressed as depth over the area $A\ RA$, delivered to a location by the water agency

The hydrologic formulation is supplemented with economic constraints as follows:

$$\underset{\mathbf{RA},\,\mathbf{D}}{\text{Max}} \quad Z = \sum_{n=1}^{N} \sum_{t=1}^{T} D_{n,t} \sum_{j=1}^{J} RA_{n,t,j}\, A_{n,j} \tag{59}$$

Subject to:

$$\sum_{n=1}^{N} \sum_{j=1}^{J} C_j\, RA_{n,x,j}\, A_{n,j} \leq F_y \qquad \text{for all n, j, t} \quad x \in t: \mathrm{mod}(t,12) = 2 \quad \text{y = 1 to 20} \tag{60}$$

$$RA_{n,x,j} - RA_{n,y,j} = 0 \qquad \text{for all } n, j \quad x \in t: \mathrm{mod}(t,12) = 2 \quad y \in t: x + z,\ z = 1 \text{ to } 5 \tag{61}$$

$$RA_{n,t,j} \leq K \quad \text{for all } n, t, j \quad (62)$$

$$K = 0: \text{mod}(t,12) = 1, 8 \text{ to } 12 \text{ or } A_{n,j} = 0 \quad (63)$$
$$\quad 1$$

$$RA_{n,t,j} \geq 0 \quad \text{for all } n, t, j \quad (64)$$

$$\sum_{n=1}^{N} D_{n,t} \sum_{j=1}^{J} RA_{n,t,j} A_{n,j} \leq WAR_t \quad \text{for all } t \quad (65)$$

$$D_{n,t} \leq [(K_{scale} HB_n)(I_n/H_0 + \ln(\varepsilon))]/[1\text{-}e^{-(I_{n,t}/H_0 + \ln(\varepsilon))}] \quad \text{for all } n, t \quad (66)$$

$$GWE_{i,t} \leq GSE_i - FB_i \quad \text{for all } i, t \quad (67)$$

$$GWE_{i,t} = H_{i,t} + \sum_{n=1}^{N} \sum_{t=1}^{T} M_{n,i,t} (RA_{n,x,t} D_{n,t}/RVu)(FD_1/FD_0)$$

$$\text{for all } i, t \quad (68)$$

$$x \in t: \text{mod}(t,12) = 2$$

$$D_{n,t} \geq 0 \quad \text{for all } n, t \quad (69)$$

Where **RA** is the set of relative areas to be optimized over space and time

D is the set of water delivery depths to be optimized over space and time

j is the crop category for a potential recharge location

J is the total number of crop categories

C_j is the annual cost per unit area to use land containing crop j for recharge operations

F_y is the annual funding available to pay for using land for recharge

As before, the formulation objective, equation (59), maximizes the volume of water recharged over the planning horizon subject to a set of operational constraints. The total expenditure for renting land for recharge during any year y cannot exceed the available funds for that year (F_y), equation (60). This constraint is used for parametric analysis (Wagner 1969) on the total funding available to rent land. The costs for using land in different crop categories are assumed to be determined by farmers bidding in a reverse auction. They vary based on the possibility of increased financial risk from winter recharge operations. However, the selection of land is not based on cost alone, since infiltration capacity influences the land use decisions through the objective function values D.

While the formulation is general enough to allow variation in monthly land use decisions, a practical adjustment is made and the RA terms are tied together for six winter months each water year (months 2 through 7), equation (61). An upper bound on the land use decision, equations (62 and 63), is based on the total land available at particular locations and times. No land is available (K = 0) during the growing season (months 1 and 8 through 12 each water year) or where there is no agricultural land (A = 0). (Cropland could be made available during the growing season for recharge by over-irrigation if there were a water source.) Otherwise, all land is available for recharge (K = 1). Equations (61 through 63) reduce the solution space to the minimum needed for the problem at-hand. Negative land use decisions are prevented with equation (64). Five crop categories (J = 5) may be present in any single groundwater model element. Equations (65 - 69) are a version of equations (52 - 57) modified through substitution of equation (58) and simplification where appropriate.

Because there is a product of decision variables, in the objective function and some constraints, the optimization problem is nonlinear and more difficult to solve. This nonlinear programming formulation can be decomposed into a two-part linear programming formulation and solved by iteration. The same objective function is used for both parts and the constraint set is split between equations (64 and 65).

<u>Part 1: Land allocation</u>

$$\max_{RA} \quad Z = \sum_{n=1}^{N} \sum_{t=1}^{T} D_{n,t} \sum_{j=1}^{J} RA_{n,t,j}\, A_{n,j} \tag{70}$$

Subject to:

$$\sum_{n=1}^{N} \sum_{j=1}^{J} C_j\, RA_{n,x,j}\, A_{n,j} \leq F_y \qquad \text{for all n, j, t} \tag{71}$$

$x \in t$: mod(t,12) = 2
y = 1 to 20

$$RA_{n,x,j} - RA_{n,y,j} = 0 \qquad \text{for all } n, j \tag{72}$$

$x \in t$: mod(t,12) = 2
$y \in t$: $x + z$, z = 1 to 5

$$RA_{n,t,j} \leq K \qquad \text{for all } n, t, j \tag{73}$$

K = 0: mod(t,12) = 1, 8 to 12 or $A_{n,j}$ = 0 (74)
1

$$RA_{n,t,j} \geq 0 \qquad \text{for all } n, t, j \tag{75}$$

The water depths (D) in equation (70) are taken as constant and either assumed as an initial condition on the first iteration (set to the static upper bounds described above) or taken from solution of Part 2 in the previous iteration. They become objective function weights.

Part 2: Water allocation

$$\underset{\mathbf{D}}{\text{Max}} \quad Z = \sum_{n=1}^{N} \sum_{t=1}^{T} D_{n,t} \sum_{j=1}^{J} RA_{n,t,j}\, A_{n,j} \tag{76}$$

Subject to:

$$\sum_{n=1}^{N} D_{n,t} \sum_{j=1}^{J} RA_{n,t,j}\, A_{n,j} \leq WAR_t \qquad \text{for all } t \tag{77}$$

$$D_{n,t} \leq [(K_{scale}\, HB_n)(I_n/H_0 + \ln(\varepsilon))]/[1\text{-}e^{-(I_{n,t}/H_0 + \ln(\varepsilon))}] \qquad \text{for all } n, t \tag{78}$$

$$GWE_{i,t} \leq GSE_i - FB_i \qquad \text{for all } i, t \tag{79}$$

$$GWE_{i,t} = H_{i,t} + \sum_{n=1}^{N} \sum_{t=1}^{T} M_{n,i,t}\, (RA_{n,x,t}\, D_{n,t}/RVu)\, (FD_1/FD_0)$$

$$\text{for all } i, t \tag{80}$$

$$x \in t\text{: mod}(t,12) = 2$$

$$D_{n,t} \geq 0 \qquad \text{for all } n, t \tag{81}$$

The land use decisions (RA) are taken as constants from solution of Part 1. If convergence of optimal objective function values from parts 1 and 2 has not occurred, the decision variables (D) from Part 2 are used in Part 1 as constants and another iteration of the two-part optimization procedure is performed.

This solution scheme is analogous to Benders' Decomposition (Geoffrion 1972):

1) The decision variables in the original problem are separated by forming two sub-problems that each contain only one variable. Where both variables remain in a sub-problem, variable separation is accomplished by converting the additional variable to a constant.

2) Solution of the two sub-problems is performed in series with the results of one sub-problem used to upgrade information used in the next sub-problem. The values of variables held constant are updated based on solution of the preceding sub-problem.

3) Iteration is applied until a satisfactory approximation to solution of the original problem is indicated by convergence of the results between iterations.

The approach used here is not as rigorous as Benders' Decomposition since cuts to the solution space are not represented as functions of the variable held constant. It is also simpler than alternative approaches demonstrated by Cai et al (2001) or Afshar et al. (2010) and uses knowledge about the solution space determined from work with the hydrologic formulation to identify an appropriate initial condition. Because there is more water available for recharge than can be accommodated with the available land, the D values in equation (70) of Part 1 can be set to their maxima at the beginning of the solution procedure based on equation (78). This allows solution for a set of RA values constrained by financial limitations in equation (71). Solution for the D values in Part 2 using the RA values from Part 1 is then possible. This approach converges quickly and encounters no infeasibilities because the original hydro-economic formulation is not so complex.

The information needed to specifying equation constants and coefficients is preprocessed in spreadsheets and passed to a linear programming solver. Experience with the example presented below indicates that convergence of the two-part formulation occurs in two iterations.

Simulating Recharge Application and Evaluating Groundwater System Improvements

After solution of the linear programming model, recharge volume schedules are calculated for each of the 67 groundwater model elements from the values for the decision variables D and RA.

$$RV_{n,t} = D_{n,t} \sum_{j=1}^{J} RA_{n,t,j}\, A_{n,j} \qquad \text{for all } n, t \qquad (82)$$

Unsaturated flow is not simulated because the groundwater model does not address addition of water to ponds during the run period. As a result, a different capability of the model is used and the recharge water is added directly to the saturated zone. Post-processing the model output creates information to evaluate changes in groundwater storage and stream flow relative to a base case of no recharge operations. The locations for stream flow evaluation are indicated on Figure 49.

Limitations

Three limitations of the approach are described here. The first two are related to the linear programming formulation and the third relates to the scope of analysis.

1) Because unsaturated flow cannot be simulated in the groundwater model during recharge operations and all recharge water is applied directly to the saturated zone, no time lags for groundwater elevation responses or partitioning of water between unsaturated and saturated zone storage occurs. This limitation could lead to overestimation of recharge effects from operating decisions. However, this limitation is not expected to be significant since the depth to groundwater is generally less than 100 ft (30 m) near the crop lands.

2) The linearized representation of groundwater head responses to recharge could over-estimate increases in groundwater elevations near rivers that are in contact with groundwater because the discharge of groundwater to surface water is not included in the response matrix. Because final evaluation of improvements to the groundwater system are made with the groundwater model and not the linear programming formulation, this limitation does not affect the ultimate predictions of changes in groundwater system state. Moreover, this final evaluation with the groundwater model shows that the limitation matters most when a groundwater elevation control constraint is binding and that this condition occurs infrequently because the initial depth to groundwater is large.

3) Potential effects on groundwater quality are not considered in this analysis. Studies have shown the presence of potential pollutants in the unsaturated zone beneath a range of land uses, including irrigated agriculture, and considered the potential for groundwater contamination from recharge (Böhlke 2002; Walvoord et al. 2003; Scanlon et al. 2005; McMahaon 2006; Jurgens et al. 2010; Harter and Lund 2012; Ascott et al 2017). In some cases, chemical reactions in the unsaturated zone during recharge reduce the potential for groundwater contamination (Schmidt et al. 2012). Bachand et al. (2014) considered potential groundwater quality impacts for their pilot study by estimating the volume of recharge water required to flush constituents from the unsaturated zone and also dilute resulting water quality impacts in the saturated zone. Gailey (2013) presents data for a different area in the Central Valley where conversion of cropland to a recharge basin caused nitrate concentrations to increase above the maximum contaminant level and more than a decade was needed for water quality impacts to subside. The potential for water quality

impacts from on-farm recharge appears to vary among sites and consideration of the factors involved (Green et al. 2008; Liao et al. 2012) should be part of recharge site selection. Economic incentives may be applied to promote recharge on lands that are less likely to cause impact (i.e., alfalfa; Dahlke et al. 2018). This is an ongoing area of inquiry and additions to the approach presented here may be possible in the future.

Results and Discussion

Data Development and Preliminary Analysis

Water available for recharge is estimated from simulated reservoir reoperation (Goharian et al. 2016) and occurs at some point during each of the 20 years in the planning period (Fig. 50). While a significant total volume is available over the planning period (10.8 million ac-ft or 13.3 km^3), the distribution in time is quite irregular. Simple spreadsheet simulation of WAR capture in recharge basins indicates the total surface water storage capacity needed to capture different amounts of water when it is available (Fig. 51). Capture of all available water would require approximately 205 thousand ac-ft (TAF; 253 x 10^6 m^3) of surface water storage capacity assuming the recharge basins drained every month (solid blue curve on the plot) and double that storage capacity if drainage required twice as long (dashed blue curve on the plot). Because the water is available seasonally and large amounts of water are available only infrequently, the capture curve has diminishing returns to scale and facility utilization is low (green curves on plot).

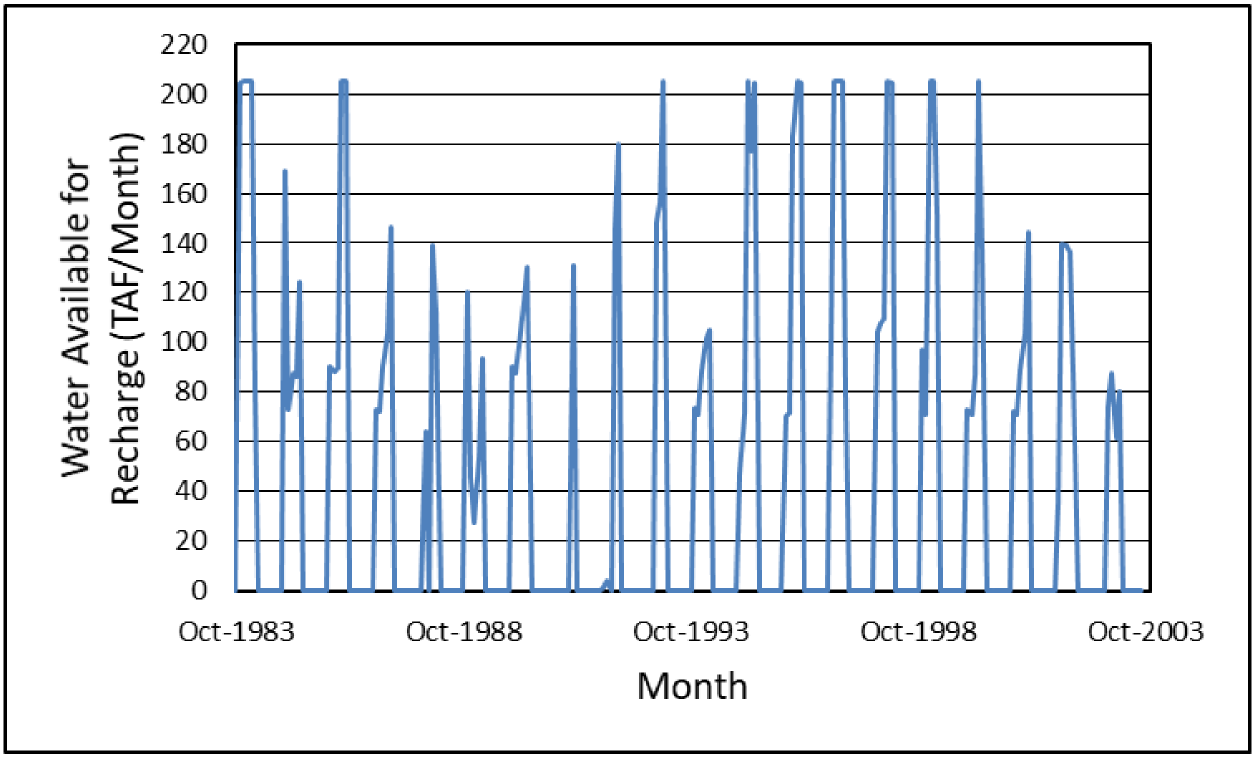


Fig. 50: Water available for recharge

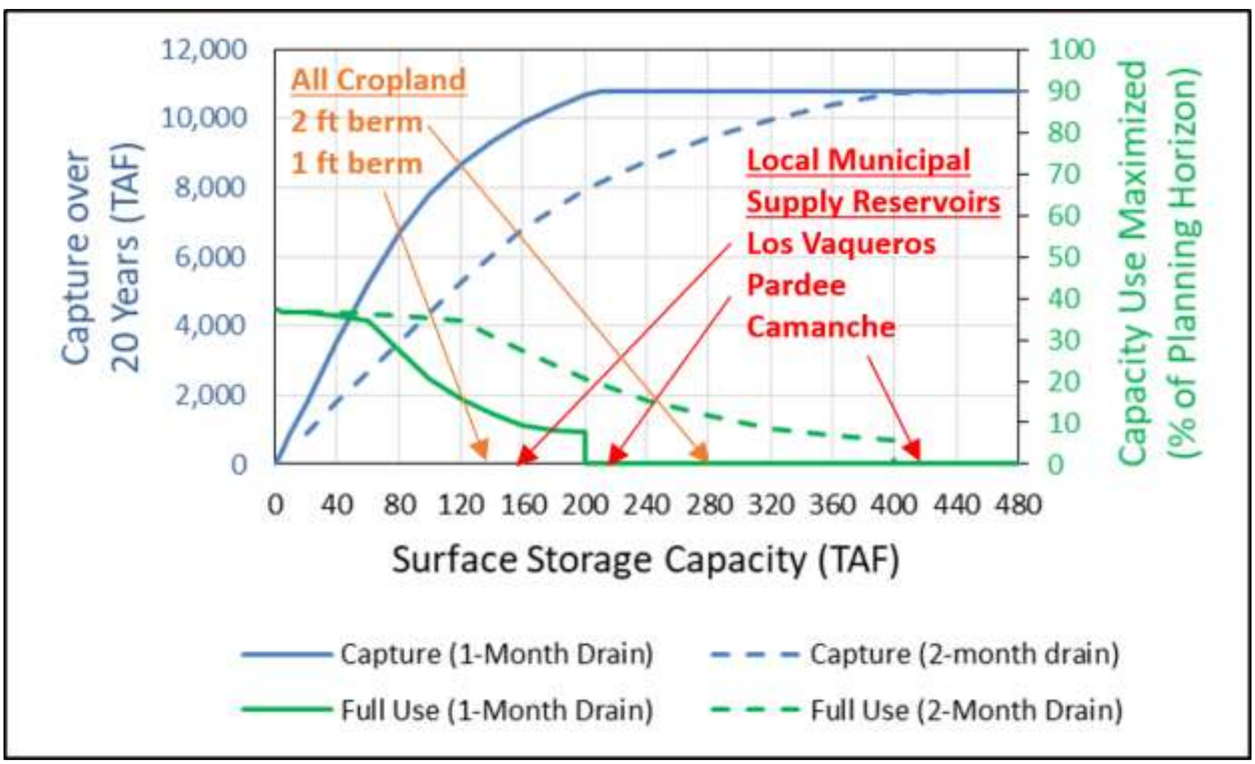


Fig. 51: Conditions for capturing water available for recharge using recharge basins

The significant amount of surface water storage capacity required to implement this traditional approach for groundwater recharge is placed into context by considering the capacities of nearby municipal supply reservoirs (Lake Camanche and Pardee Reservoir operated by the East Bay Municipal Utility District at 417 and 198 TAF, as well as Los Vaqueros Reservoir operated by the Contra Costa Water District at 160 TAF; Fig. 51). By comparison, use of all 140,000 ac of crop land with berms between 1 and 2 ft (0.3 to 0.6 m) high would result in similar amounts of surface water storage capacity (Fig. 51). Therefore, it might be reasonable to evaluate using croplands to meet at least some recharge opportunities for the study area.

Information for infiltration rates in the study area is mapped to the groundwater model elements (Fig. 52). The rates are derived from simulating ponding at ground surface and transient unsaturated/saturate flow into a fine-scale (200 m resolution in the three spatial dimensions) representation of the spatially variable hydrogeology (Maples et al. 2017). Average infiltration rates are calculated over the 120-day simulations for a variety of assemblages of sediments. These values are scaled up for use in the coarser-resolution groundwater flow model used here. It is assumed that any shallow hardpan has been breached consistent with the SAGBI rating for conditions where sites have been modified by deep tillage rating (O'Geen et al. 2015); however, observations of ponded water in parts of the study area suggest that hardpan may be present at some locations.

Crop categories for annual costs to use land in recharge operations are developed to be generally consistent with discussion of risks to crop health and productivity presented by Dahlke et al. (2018) as well as Hanak et al (2018). The categorization in figures 53 a and b is based on discussions with a variety of people working in the study area (i.e., land managers, Sacramento County

Agriculture Commissioner staff, other county staff and researchers). The unit costs applied to the categories (Fig. 54a) are quite preliminary (exploratory) and could be improved with survey data on land manager perceptions of crop risk and attitudes towards financial risk tolerance. Combining the cumulative areas and unit costs for each category provides a view of potential total cost as a function of total land area used for recharge (Fig. 54b). These curves are based on an assumption that land is selected solely on unit price; however, infiltration rates must also be considered when attempting to maximize recharge. To the extent that locations of the cheapest land are not correlated with the highest infiltration rates, the curves will be different than those shown. The linear programming method described above allows performance of the required analysis. Table 14 summarizes the values of the hydro-economic model parameters not addressed elsewhere.

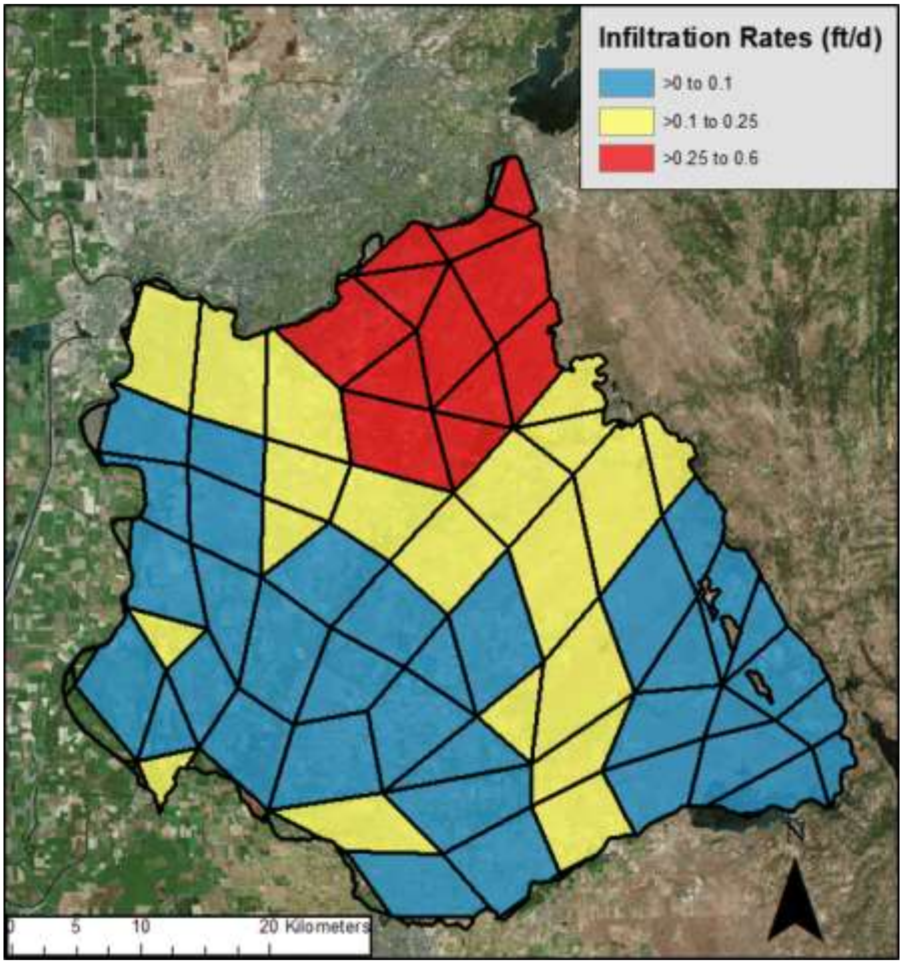
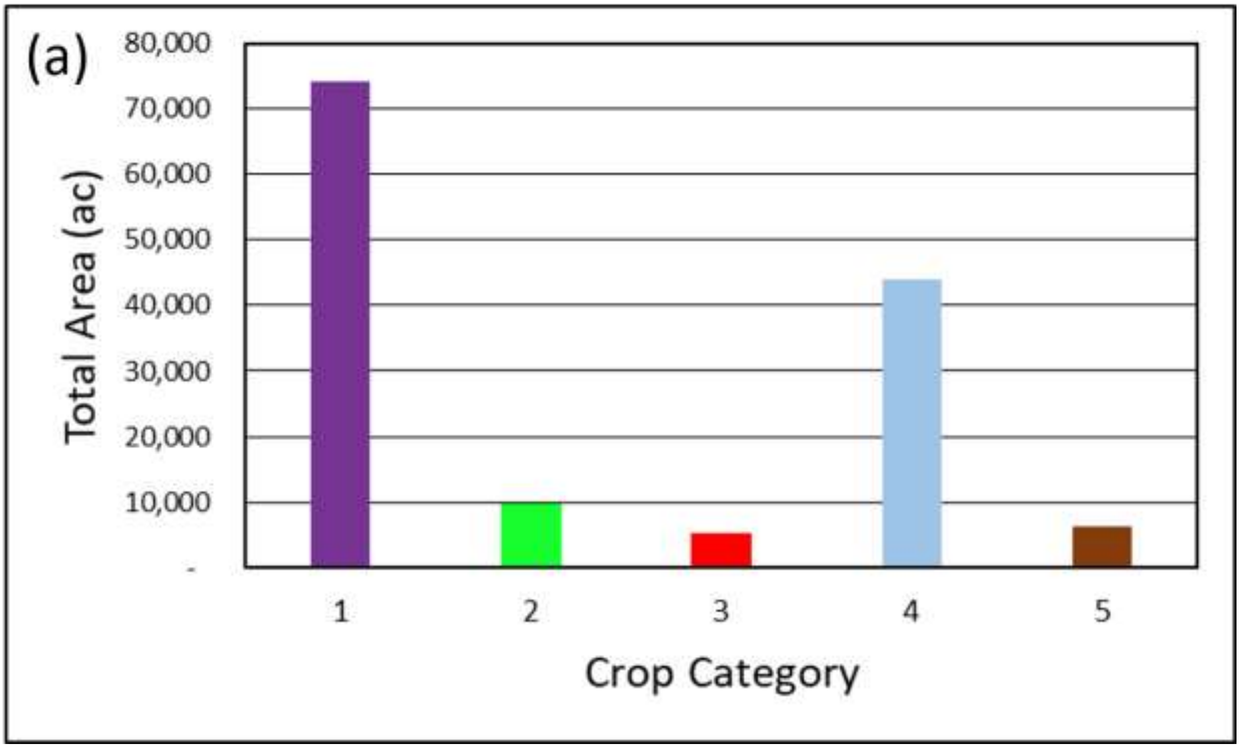


Fig. **52:** Study area infiltration rates


(a)
80,000
70,000
60,000
50,000
40,000
30,000
20,000
10,000
-
Total Area (ac)
1
2
3
4
5
Crop Category

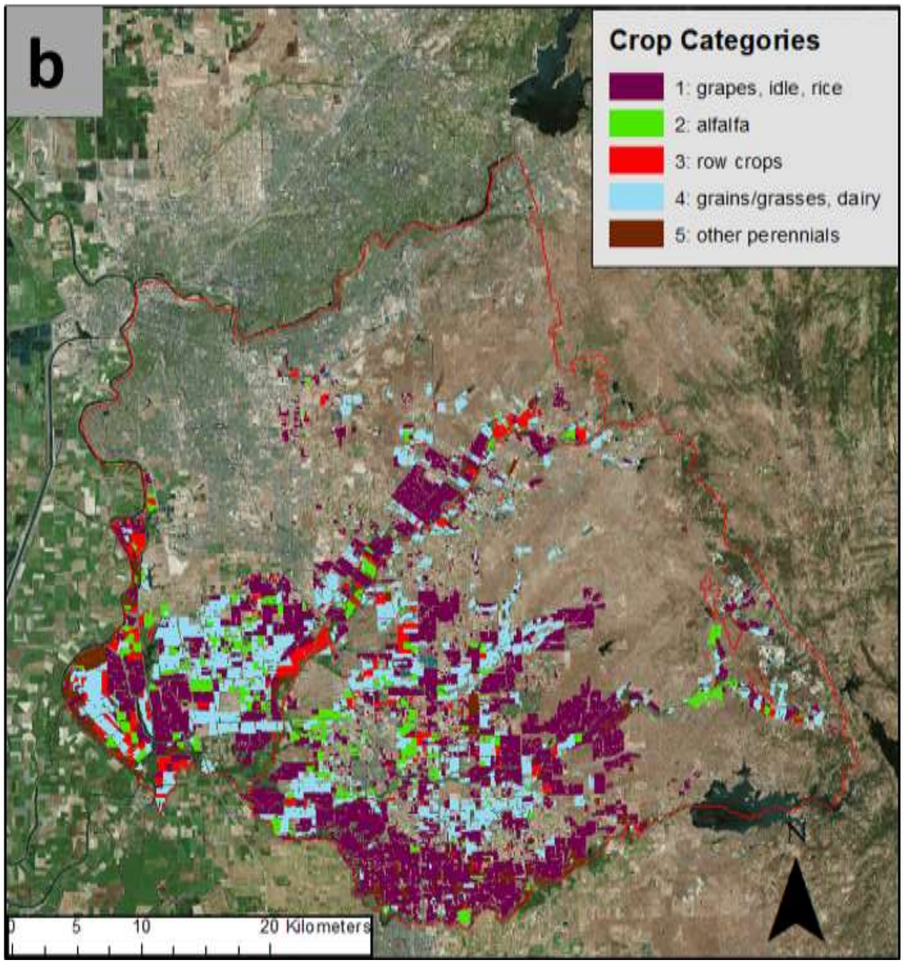
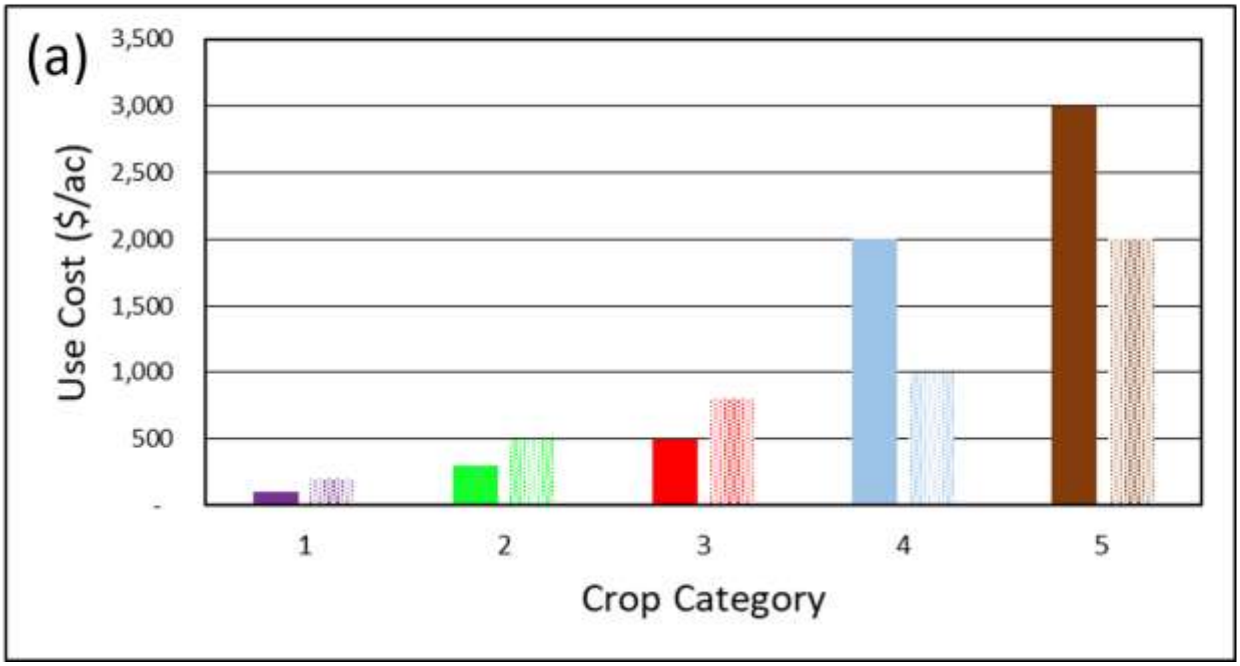


Fig. 53: Crop categories for costs to use land for recharge: **a** categories and total acreages and **b** spatial distribution of categories



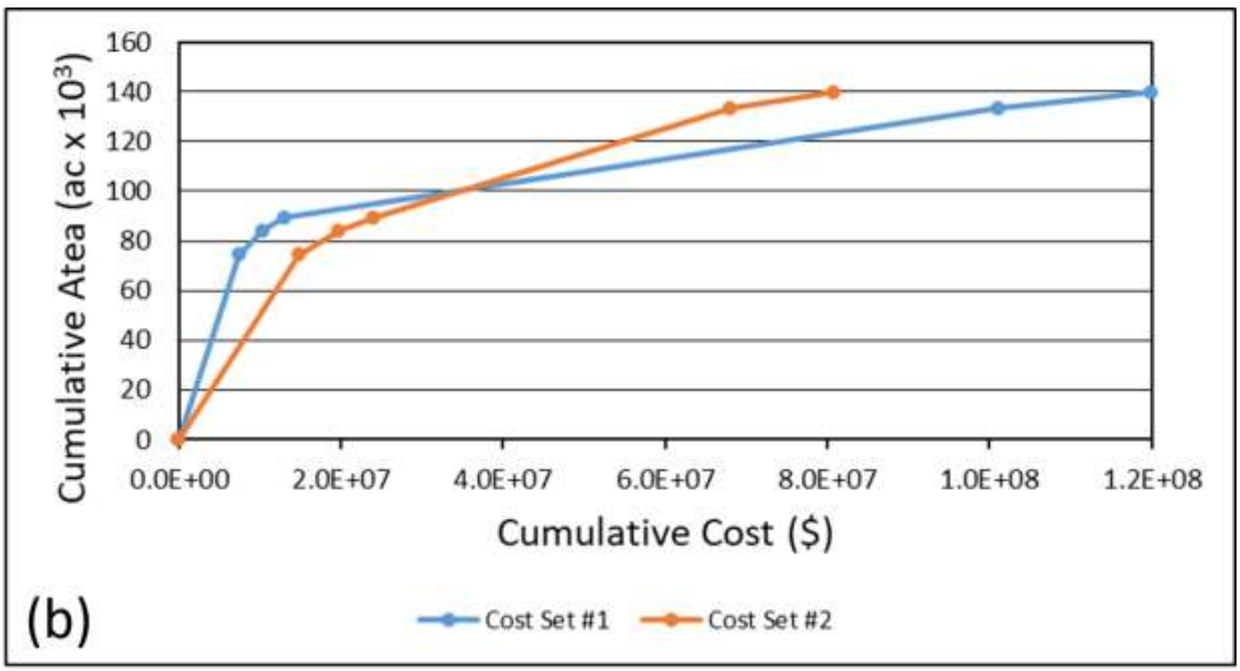


Fig. 54: Cropland use costs: **a** unit annual costs and **b** cumulative annual costs. *Solid colors* on Fig. 54a correspond to cost set #1 and *stippled colors* correspond to cost set #2

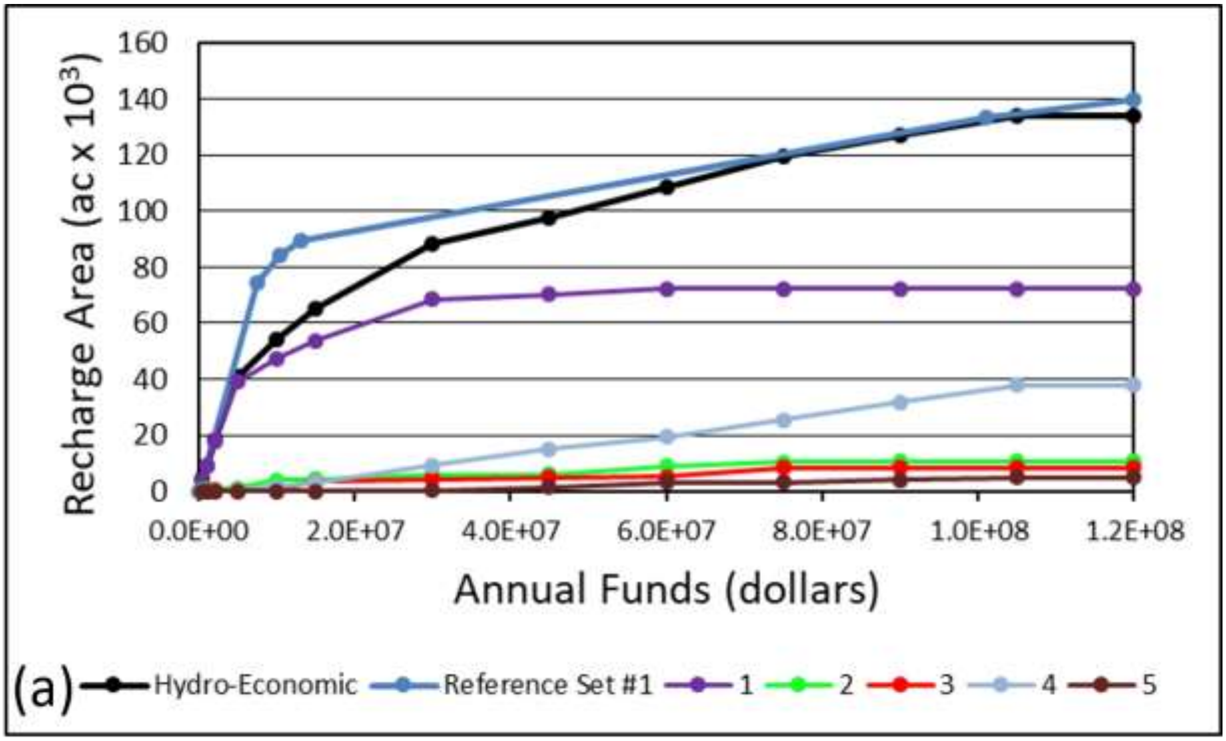
Table 14 Parameter values

Parameter	Value
HB	1 ft (0.3 m)
H_0	0.3 ft (0.1 m)
FB	2 ft (0.6 m)
FD_0	0.95
FD_1	1.0
F	0.5 to 120 million dollars

Results

Cropland area use for recharge as function of funding is presented on Figures 55 a and b. These are the results of parametric analysis using equation (71). Differences between the results for hydro-economic analyses and the reference curves occur because, as indicated by the curves for individual crop categories (Fig. 55a), some of the more expensive land is brought into use before all of the least expensive land has been used. This result is driven by variation in infiltration rate across the study area which is controlled by the shallow geology and the interconnectedness of high conductivity sediments at depth (Maples et al. 2017) used in the ponding model of equation (78). Figure 56 shows the spatial distribution of land use for two different levels of funding. For low amounts of funding, land is brought into use where there is a combination of cheap land and high infiltration rates in an effort to maximize the product of decision variables RA (scaled by A) and D. This observation is consistent with the steep slope of recharge volume as a function of funding for land use at low funding levels (Fig. 57). Spatial distribution of the applied recharge water annual cumulative depth is presented for the maximum funding and land use on Figure 58.

The values are generally within a reasonable range based on currently available information on crop inundation tolerance; however, constraints could be added to control cumulative water application as necessary.



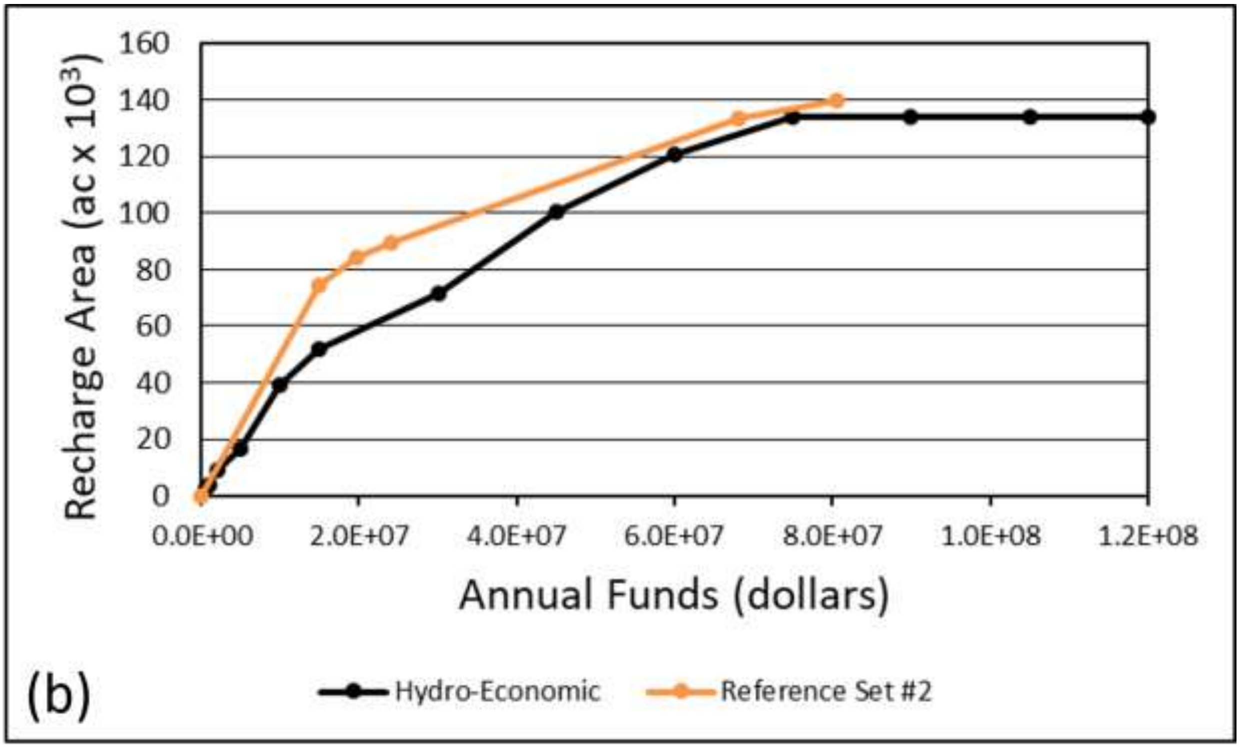
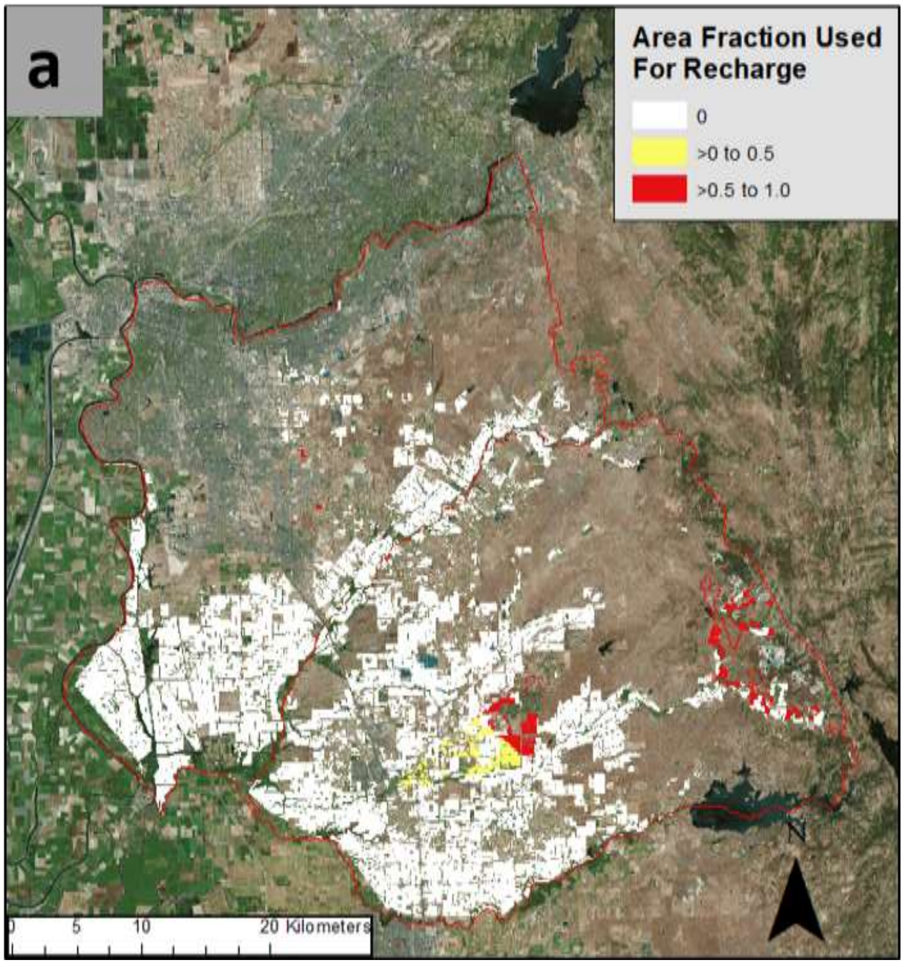


Fig. 55: Variation of cropland used for recharge with annual funding: **a** cost set #1 and **b** cost set #2. Reference curves based on cost only from Fig. 54b are presented for comparison. Numbered curves on Fig. 55a indicate land use for recharge by individual crop categories


a
Area Fraction Used For Recharge
0
>0 to 0.5
>0.5 to 1.0
0
5
10
20 Kilometers
N

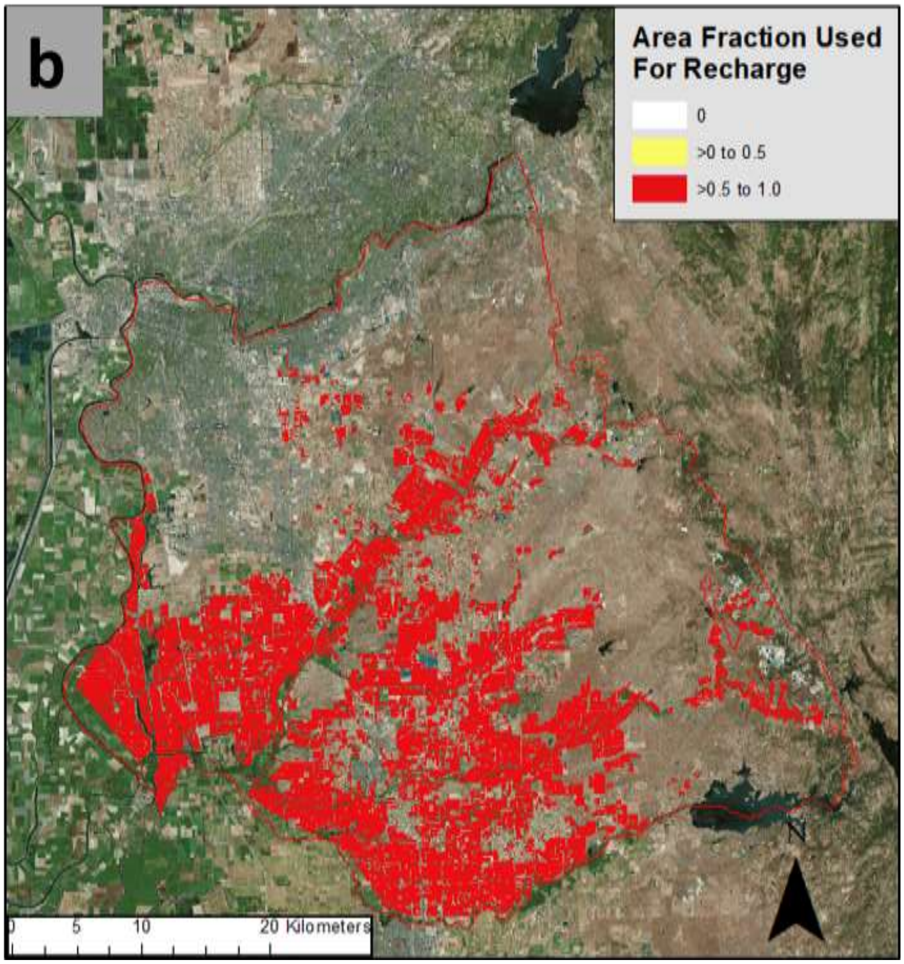


Fig. 56: Spatial distribution of land used for recharge **a** F_y = \$500,000 and **b** F_y = \$120,000,000.

Results for cost set #1. *Area fraction* plotted is the decision variable RA

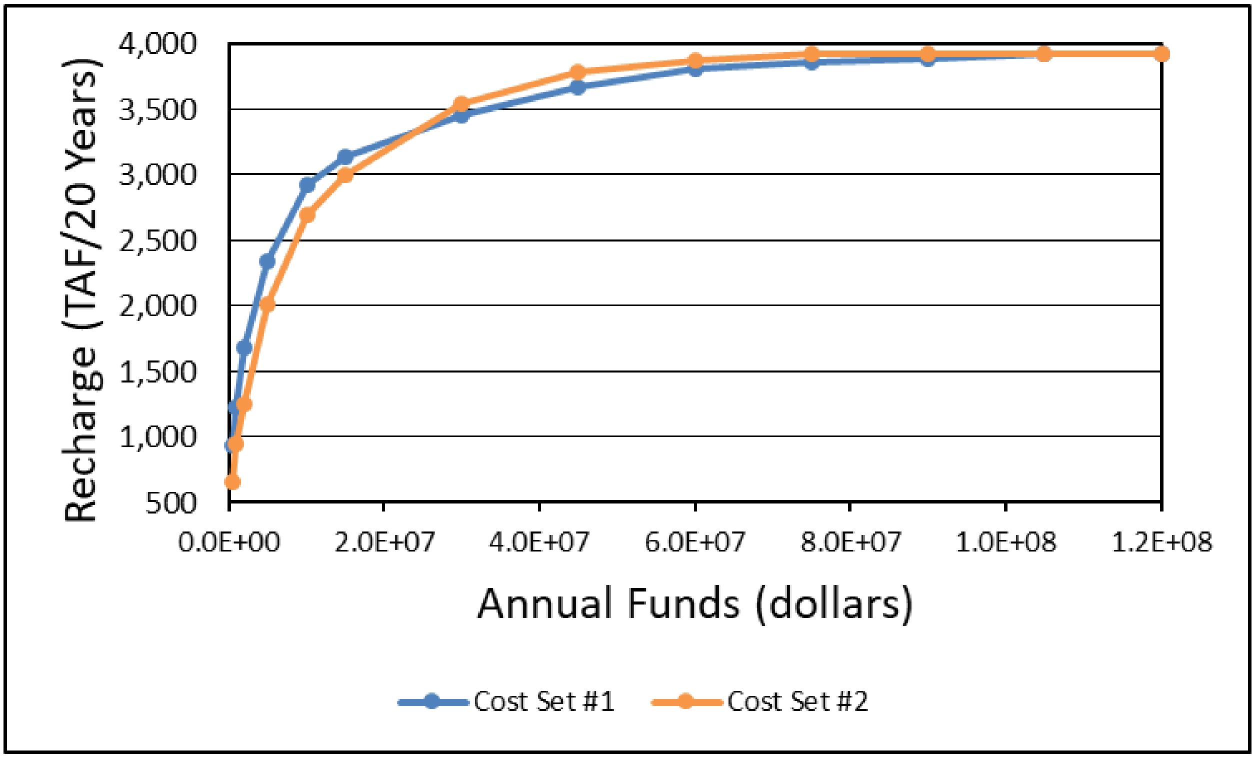


Fig. 57: Variation of recharge volume with annual funding

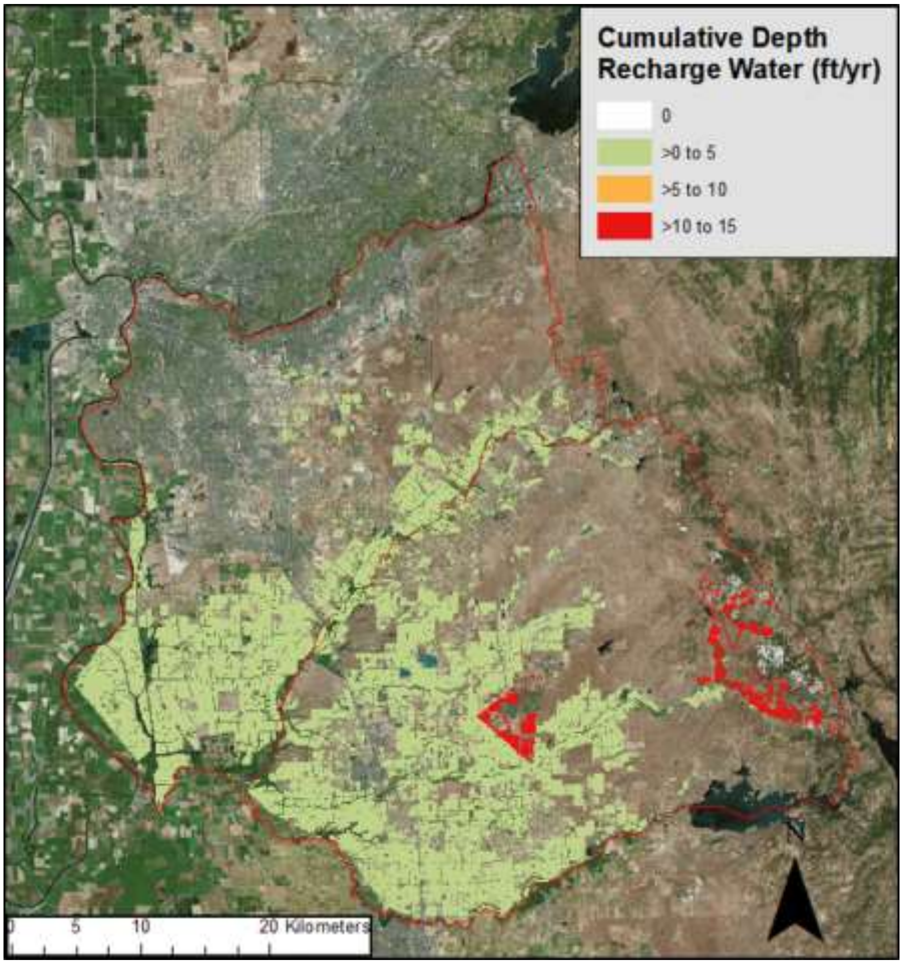
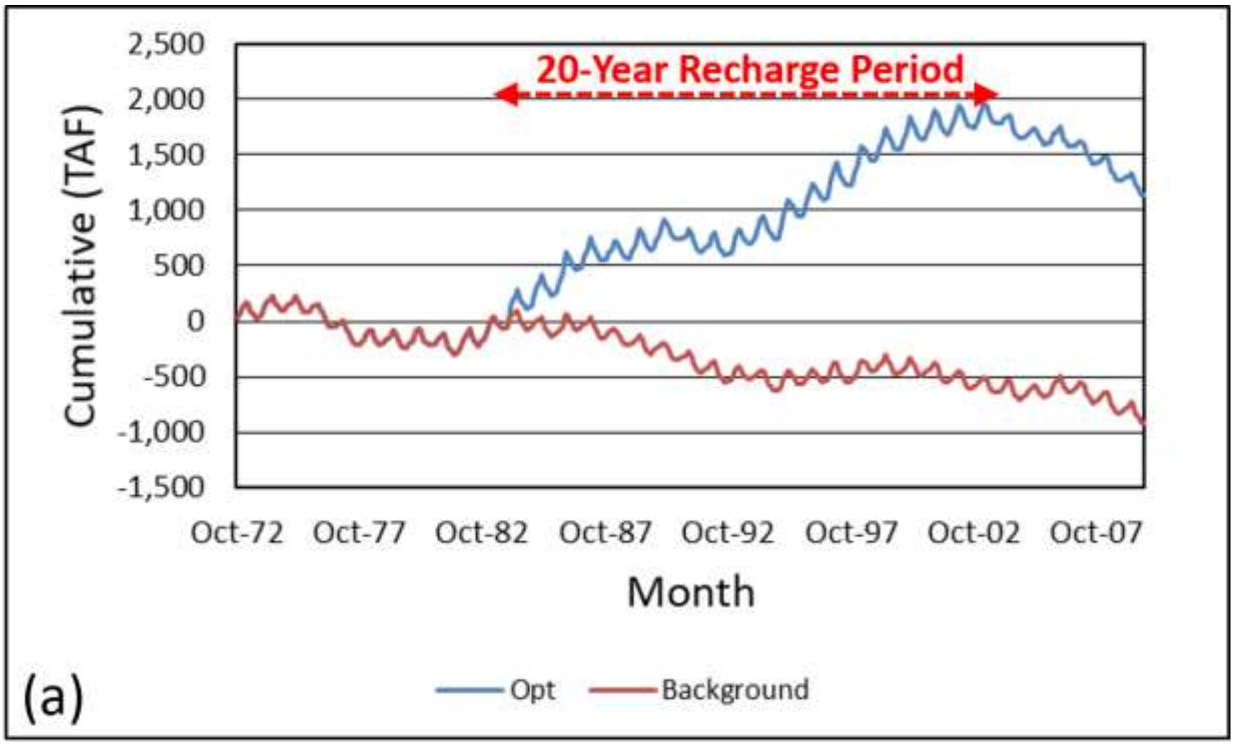


Fig. 58: Cumulative depth of applied recharge water for maximum land use funding using cost set #1

Figure 59 indicates the increase in groundwater storage from recharge using all of the cropland. Recharging over the 20-year planning period used 36 percent of the WAR (3,921 TAF or 4.8 km^3). Most of the water remained in the groundwater system (2,419 TAF or 62% of the total volume recharged); however, appreciable amounts exited to surface water (718 TAF, 18%) or flowed across sub-basin boundaries (764 TAF, 20%). The recharge provided enough baseflow to support flow in the Cosumnes River throughout the 20-year simulation except during a five-year drought from 1987 through 1992. Table 15 presents results for a range of recharge funding levels.

Volumes discharging to surface water and flowing to other sub-basins decrease with the volume recharged since head buildup from adding water to the system (the driving force for groundwater flow) is less pronounced.



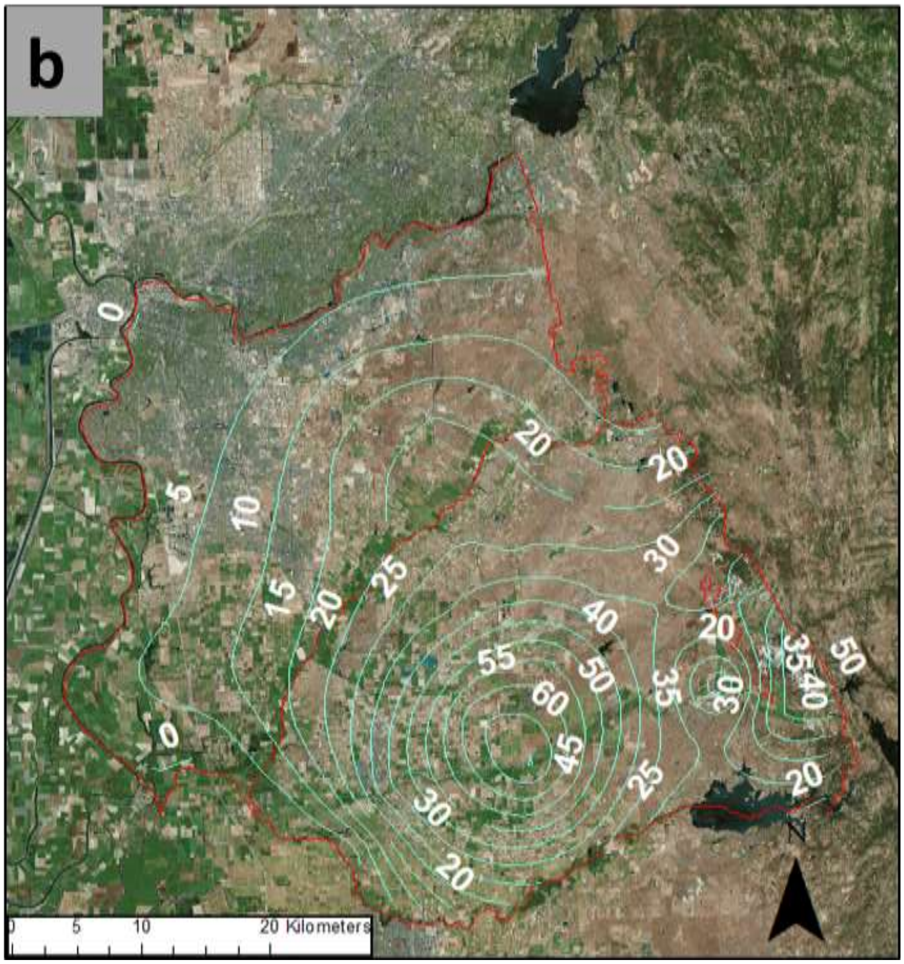
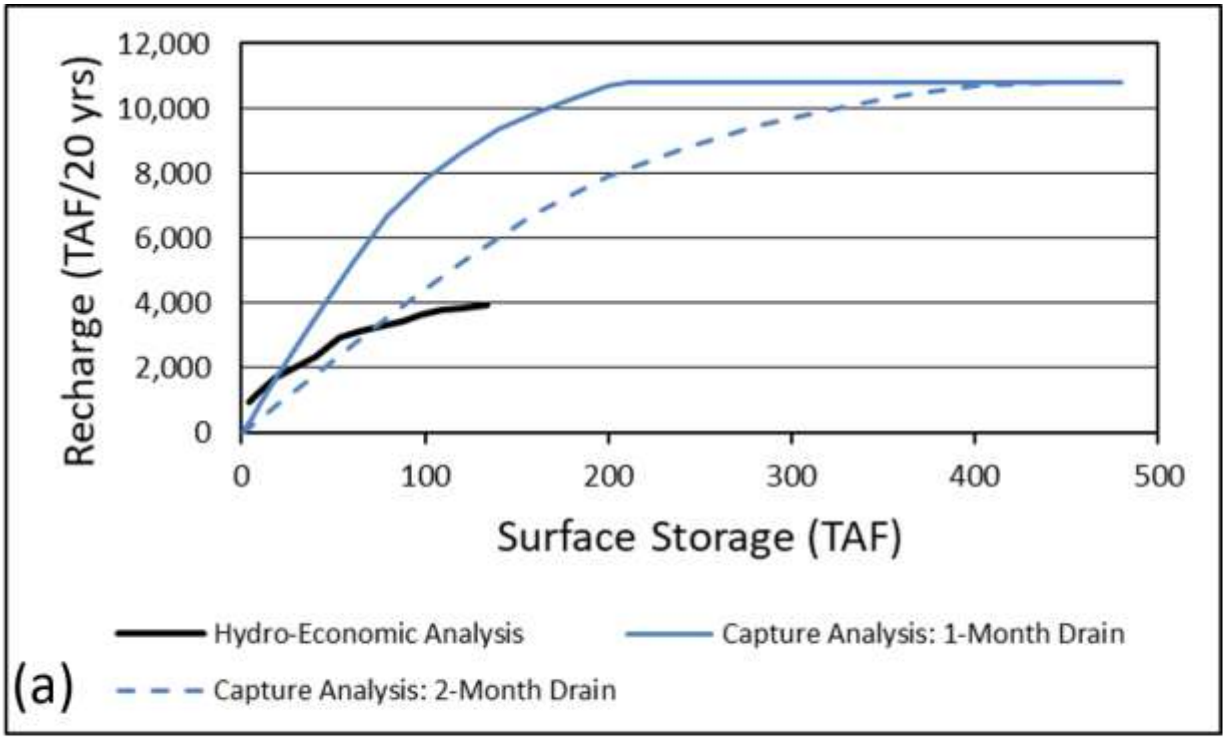


Fig. 59: Increase in groundwater storage using all cropland: **a** storage accumulation over time and **b** spatial distribution of elevation increases

Table 15 Results for different levels of land use funding (cost set #1)

	Funding Level		
	\$500,000	\$5,000,000	\$120,000,000
Land use area	4 x 10^3 ac	41 x 10^3 ac	134 x 10^3 ac
Recharge volume	937 TAF	2,335 TAF	3,921 TAF
Increased gw storage	628 TAF	1,551 TAF	2,419 TAF
Discharge to surface water	203 TAF	359 TAF	718 TAF
Flow to other sub-basins	107 TAF	425 TAF	764 TAF

Comparison of the recharge volume results from the hydro-economic analysis for cost set #1 (Fig. 57) with reference curves from the initial capture analysis (Fig. 51) indicates the effect of including study area hydrogeology (spatial variation in infiltration rate) in the analysis (Fig. 60a). High infiltration rate sites are selected preferentially, even when the amount of recharge area is limited by funding, and plot on the left side of the hydro-economic curve. These sites drain quickly and the results plot above the reference curves. Few of these sites are within the footprint of the cropland and, when greater amounts of land are used for recharge, the additional sites drain slower and plot below one or both of the reference curves. The result is a capture curve for the study area that is shallower in slope than the reference curves. Therefore, the spatial variability in infiltration rate magnifies the diminishing returns to scale already occurring as a results of the temporal variability of the water source.



Recharge (TAF/20 yrs)
12,000
10,000
8,000
6,000
4,000
2,000
0
0
100
200
300
400
500
Surface Storage (TAF)
Hydro-Economic Analysis
Capture Analysis: 1-Month Drain
Capture Analysis: 2-Month Drain
(a)

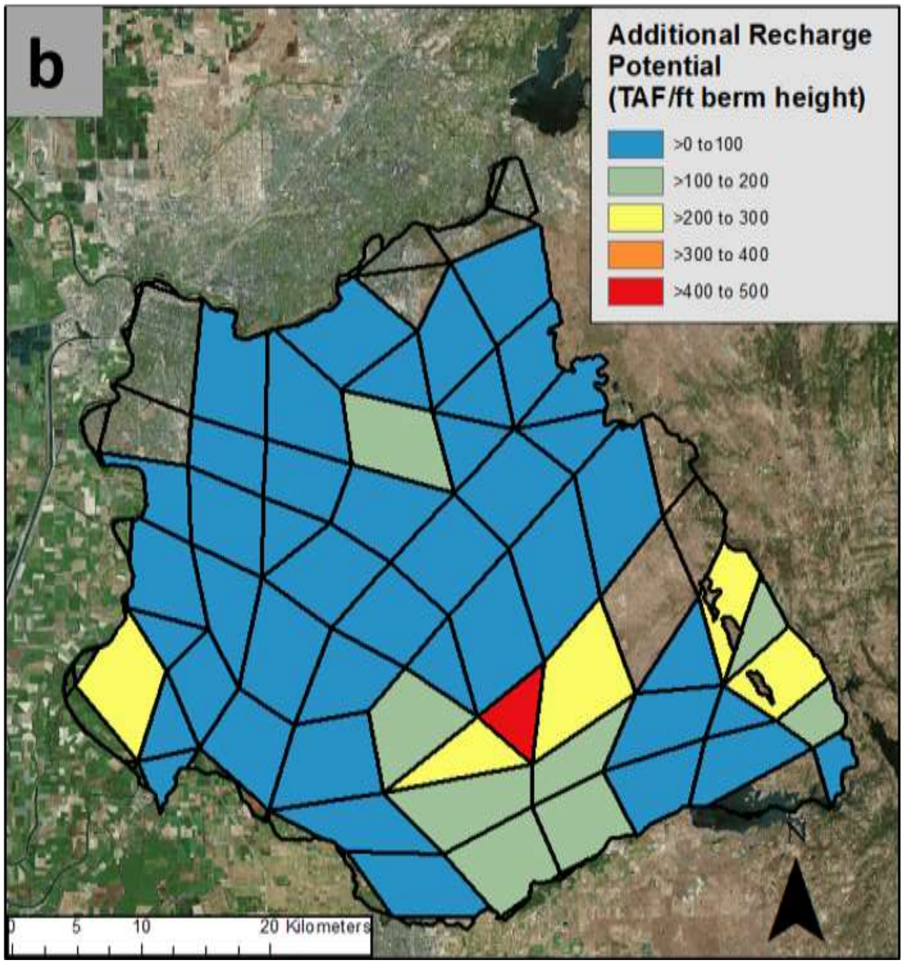
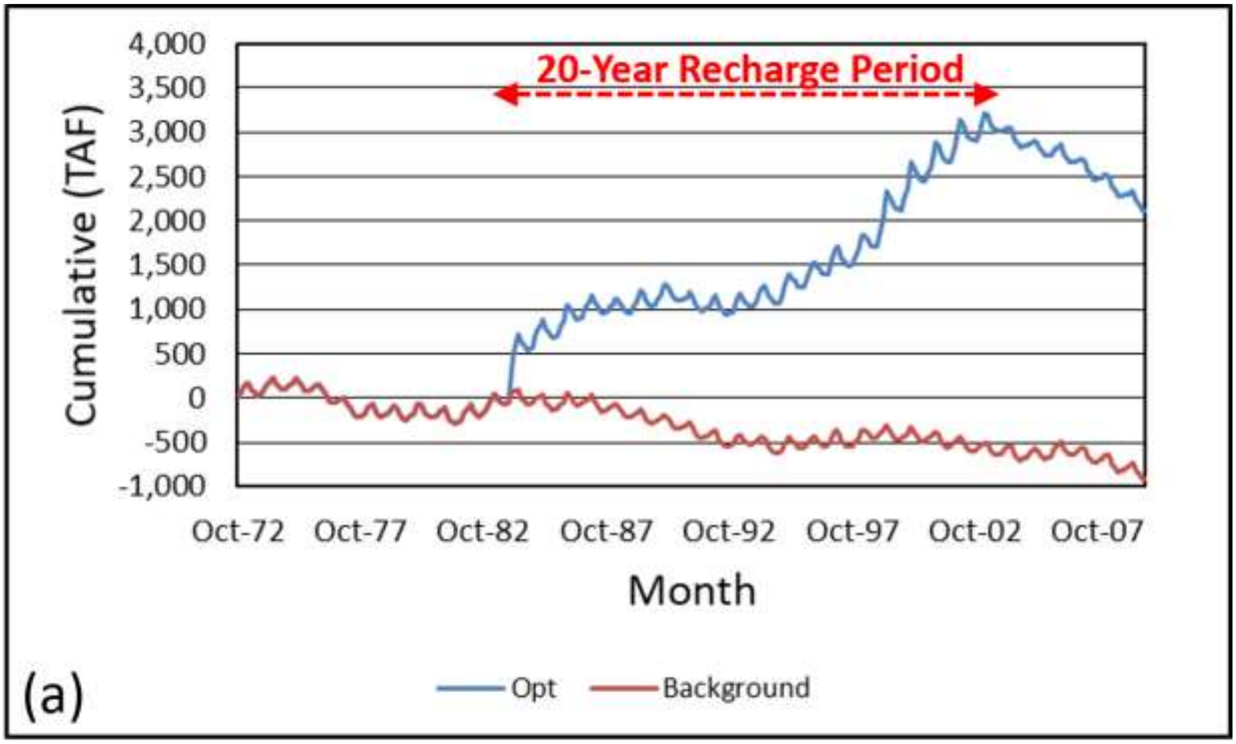


Fig. 60: Effect of spatial variation in infiltration rate on recharge volume potential: **a** capture curves and **b** Lagrange multipliers

More recharge could be achieved, and the study area capture curve moved higher on the plot, if the berm height around the cropland were increased. The linear programming results obtained can help develop guidance on where such capital investment might be most valuable. Reformulating the Lagrange multiplier for equation (78) in terms of the berm height (see appendix) indicates locations and volumes of additional water that would be recharged over the planning horizon if berms were raised from 1 to 2 feet (0.3 to 0.6 m; Fig. 60b). This result provides a high estimate of what might be possible since some perennial crops may be unable to accommodate the increased ponding depth.

The values for Lagrange multipliers based on increasing berm height by 1 ft (0.6 m) are low in the northern portion of the study area (Fig. 60b) because little cropland is present (Fig. 47). Given the high infiltration rates of the deeper geology in the north (Fig. 52), recharge potential would be much better for a gravel pit since it would penetrate the low hydraulic conductivity soil layer included in this analysis. Cropland present in one of the northern model elements with high-infiltration rate was used to simulate the potential effect of repurposing a gravel pit for recharge. A total of 570 ac (231 ha) in crop categories 2, 3 and 4 were used to simulate gravel pits by increasing the hydraulic conductivity of the soil layer to match the underlying geology and increasing the berm height to 20 ft (6 m).

Figures 61 a and b summarize the results of gravel pit simulation at the maximum annual funding level. Recharging over the 20-year planning period used 50 percent of the WAR (5,412 TAF or 6.8 km^3). Most of the water remained in the groundwater system (3,651 TAF or 68% of the total volume recharged) with amounts similar to the previously-presented results exiting to surface water (869 TAF, 16%) and flowing across sub-basin boundaries (889 TAF, 16%). Allocation was skewed towards the gravel pits (31% of the total volume recharged) and provided enough baseflow to support continuous flow in the Cosumnes River throughout the 20-year simulation including during the previously-mentioned five-year drought.


20-Year Recharge Period
Cumulative (TAF)
4,000
3,500
3,000
2,500
2,000
1,500
1,000
500
0
-500
-1,000
Oct-72
Oct-77
Oct-82
Oct-87
Oct-92
Oct-97
Oct-02
Oct-07
Month
(a)
Opt
Background

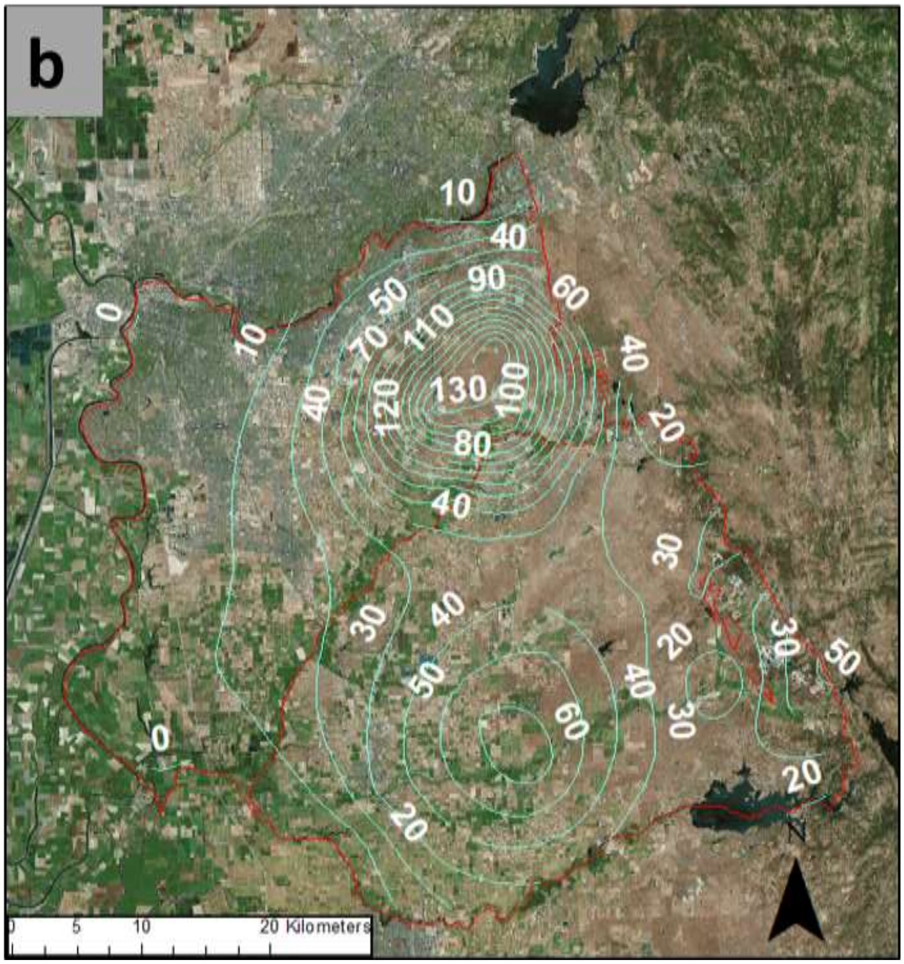


Fig. 61: Increase in groundwater storage using all cropland and repurposed gravel pits in north: **a** storage accumulation over time and **b** spatial distribution of elevation increases

Potential Extensions

The method and analysis for the study area could be extended to include net metering (Kiparsky et al. 2018). This approach could entail representing cropland managers as individual profit-maximizing agents along with a groundwater management agency charging fees for groundwater pumping and providing rebates for recharge. It is unclear if the aggregate effect of net metering with modest pumping fees would significantly differ from the work presented here since the influence on rational profit maximizers of a net rebate, rather than a payment for using land for recharge, may be similar. However, the effect of net metering combined with a cash flow constraint applied to water management operations (revenue from groundwater pumping minus a financial friction for management equal to or exceeding payments for recharge) could impose limits on a program for improving groundwater system conditions and possibly driving pumping fees higher. These changes could influence the behaviors of profit maximizing land managers.

It may also be possible to explore improving groundwater conditions through water banking operations where capital investments (i.e., construction of distribution canals from Folsom South Canal) and operations costs would be paid by a client, or clients, external to the basin. Management policy questions would include: 1) how much water would be left in-place to benefit the groundwater system (recharged but not withdrawn at a later time) and 2) the longevity of withdrawal rights (ability to withdraw water decreases with time since recharge event). Details of the policy decisions would likely have financial implications such as the amount of infrastructure investment a water banking client might be willing to make.

Either the cash flow or water banking approach might be modified to encourage recharge in areas where it is most needed. Lower bound constraints for groundwater elevations at control locations could be added in parts of the basin with the greatest cumulative drawdowns. It might also be possible to evaluate policies to avoid potential water quality degradation from flushing undesirable constituents (i.e., nitrate, pesticides and salts) from the unsaturated zone and shallow groundwater by including subsidies (reducing costs to use lands for recharge) to focus recharge on more desirable lands (i.e., alfalfa fields as suggested by Dahlke et al. 2018).

Recharge at locations important for supporting and increasing surface water baseflow could also be emphasized. The simplest way to achieve this benefit would be to set lower bound constraints for groundwater elevations at control locations near the surface water bodies (i.e., Cosumnes River). However, this approach would require experimentation with lower bound values and locations because, as previously indicated, the linearized approach for representing the groundwater response to recharge does not account for exchange between groundwater and surface water. A more thorough approach would entail reformulating the planning problem as a nonlinear program where the groundwater flow model is called on each iteration of the solver to evaluate the value for a baseflow constraint.

Finally, a portfolio of recharge projects that includes a mix of croplands and dedicated facilities could be considered. This extension would be desirable since croplands in the study area are concentrated in the south while the higher hydraulic conductivity deposits are in the north (compare figures 47 and 52). Not coincidentally, potential properties that could be repurposed as dedicated facilities (gravel excavations) are in the north. To control the number of potential facilities considered, upper bound infiltration capacity constraints for the gravel pits could be manipulated to include/exclude the potential facilities in the formulation by toggling the bound value between zero and an estimated capacity (a form of parametric analysis). Alternatively, the linear programming model could be reformulated as a mixer-integer linear program and parametric analysis could be performed on a funding constraint.

Conclusions

On-farm recharge appears to be promising for the study area. Using all of the 140,000 ac of cropland in the study area would have allowed approximately 3,900 TAF (4.8 km^3) of recharge over the 20-year period considered (October 1983 through September 2003). Analysis indicates that there would be decreasing returns to scale as a result of 1) temporal variability of water available for recharge, 2) variations in infiltration rate and a limited number of high-infiltration rate sites across the study area and 3) recharged water exiting the study area groundwater system to surface water and adjacent sub-basins. Depending upon crop tolerance to ponding depth, these limitations might be reduced by raising berm heights at higher infiltration rate sites. Additional efforts to recharge higher-infiltration rate sediments to the north through pits that penetrate lower-infiltration rate topsoil could significantly increase total recharge volume. Preliminary results indicate approximately 5,400 TAF (6.8 km^3) of recharge could occur over the 20-year period using 570 ac of gravel pits.

The method applied in this work is general enough that it can accommodate additional information such as:

1) Soil hydraulic conductivity values and variations
2) Site-level geology
3) Field tests of infiltration rates
4) Observations from land managers who have detailed knowledge of field drainage rates
5) Details regarding flooding tolerance for different crops
    a. Months that are appropriate
    b. Duration of flooding
    c. Total amount of recharge
6) Details regarding costs to use specific sites

Extensions of the work could readily address:

1) Financial considerations regarding investment and operations
2) Measures to safeguard groundwater quality
3) Support for baseflow to the Cosumnes River
4) Portfolios of recharge facilities and approaches

Such analysis is within reach given this initial work. Continued collaboration with stakeholders in the study area may provide future directions.

Acknowledgements

Yueyue Fan and Thomas Harter are appreciatively acknowledged for editorial and technical comments that improved this manuscript. This work was supported by the UC Office of the President's Multi-Campus Research Programs and Initiatives (MR-15-328473) through UC Water, the University of California Water Security and Sustainability Research Initiative.

Appendix

Details regarding formulation of the linear programming model are presented in the following sections.

Ponded Water Drainage

The upper bound for recharge water applied is specified as equations (54 and 66) and based on a requirement that the water ponded on the field not overtop an assumed perimeter berm of height

HB. A series of steps are taken to develop an expression for the maximum allowable recharge volume.

An ordinary differential equation and initial condition for water mass balance in a recharge pond during filling is formulated and solved:

$$A\ dh/dt = Q - A\ (I/H_0)\ h \tag{83}$$

$$h(0) = 0 \tag{84}$$

$$h = [(Q\ H_0)/(I\ A)]\ [1 - e^{-(I/H_0)t}] \tag{85}$$

Where A is the ponding area

h is the ponding depth

t is time

Q is the rate of inflow

I is the reference rate ponded water infiltrates the subsurface (Maples et al. 2017)

H_0 is ponding depth associated with I (Maples et al. 2017)

The quantity (I/H_0) in equation (83) normalizes the infiltration rate by the ponding depth used to estimate the quantity and allows scaling by h to simulate variation in infiltration rate with ponding depth.

Filling the pond at a constant rate until the maximum ponding depth is reached at a specified time is represented by substituting h = HB, Q = Q_{max} and t = T into equation (85). Rearrangement yields:

$$Q_{max} = (I\ A\ HB/H_0)/[1 - e^{-(I\ T/H_0)}] \quad (86)$$

An ordinary differential equation for water mass balance in a recharge pond during draining is formulated and solved:

$$dh/dt = -(I/H_0)\ h \quad (87)$$

$$h = e^{-(I/H_0)t + K} \quad (88)$$

Substituting t = t – T so that equations (85 and 88) initiate at the same time and rearranging yields:

$$h = K\ e^{-(I/H_0)\ (t - T)} \quad (89)$$

Equating equations (85 and 89) at time t = T, solving for K and substituting into equation (89) yields:

$$h = [(Q\ H_0)/(I\ A)]\ [1 - e^{-(I/H_0)T}]\ e^{-(I/H_0)\ (t - T)} \quad (90)$$

Substituting equation (86) for Q and rearranging yields an expression for filling to time T and then draining thereafter:

$$h = HB\ e^{-(I/H_0)\ (t - T)} \quad (91)$$

Assume that the pond must be filled and drained within one month to allow operational flexibility such that the land could be used for purposes other than recharge the following month. A one-month filling and draining cycle is approximated by introducing a terminal boundary condition $h(1) = \varepsilon\ HB$, where ε is a small increment. Solving for T yields:

$$T = 1 + (H_0/I)\ln(\varepsilon) \quad (92)$$

Substituting equation (92) into equation (91) and the result into equation (86) yields an expression for Q_{max}:

$$Q_{max} = (HB\ A\ I/H_0)/[1-e^{-(I/H_0 + \ln(\varepsilon))}] \quad (93)$$

Multiplying this expression for Q_{max} by the equation (92) for T results in an expression for the maximum recharge volume that can be added to a pond in a single month:

$$RV_{max} = [(HB\ A)(I/H_0 + \ln(\varepsilon))]/[1-e^{-(I/H_0 + \ln(\varepsilon))}] \quad (94)$$

Equation (94) is based on an infiltration rate derived for water ponded on the raw geology. Because a lower hydraulic conductivity soil overlays the geology, the expression is scaled by a factor that accounts for the effective vertical hydraulic conductivity of the layered porous medium:

$$K_{scale} = K_{eff}/K_{geol} \quad (95)$$

$$K_{eff} = (b_{soil} + b_{geol})/[(b_{soil}/K_{soil}) + (b_{geol}/K_{geol})] \quad (96)$$

Where K_{eff} is the effective vertical hydraulic conductivity calculated as the harmonic mean of the conductivities of the soil and geologic layers

K_{geol} is the averaged vertical hydraulic conductivity of the deeper geologic materials (Maples et al. 2017)

K_{soil} is the vertical hydraulic conductivity of the soil (taken as 3×10^{-2} ft//d, or 10^{-5} cm/s, based on Brush et al. 2013)

b_{geol} is the thickness of the unsaturated zone in the geologic materials

(Maples et al. 2017)

b_{soil} is the thickness of the soil layer (taken as 1 ft or 0.3 m)

Applying the scaling factor to equation (94) yields the expression used for equation (54).

$$RV_{max} = [(K_{scale}\ HB\ A)(I/H_0 + \ln(\varepsilon))]/[1-e^{-(I/H_0 + \ln(\varepsilon))}] \quad (97)$$

Dividing equation (97) by A yields an expression for the maximum recharge depth that can be added to a pond in a single month. This is equation (66).

$$D_{max} = [(K_{scale}\ HB)(I/H_0 + \ln(\varepsilon))]/[1-e^{-(I/H_0 + \ln(\varepsilon))}] \quad (98)$$

The formulation is somewhat sensitive to value chosen for ε with smaller values reducing the upper bound. Using a value of 0.01 appeared reasonable for this analysis. Finally, the assumed one-month filling and drainage cycle could be adjusted by extending the approach described here to simulate pulsed flooding for crop root health (Dahlke et al. 2018).

Groundwater Elevation Calculation

The upper bound on groundwater elevation is based on ground surface elevation and an assumed required freeboard to avoid waterlogging of soil. This consideration can be important for down-flow parts of basin where recharge might not be applied but water levels may rise as a result of recharge water redistribution by means of groundwater flow (Niswonger et al. 2017). The groundwater elevation itself is based on a linearized representation of groundwater head response to addition of water to the system at a particular location and time (Reilly et al. 1987, Gorelick et al. 1993, Ahlfeld and Mulligan 2000). The representation is most accurate for confined systems but works well for unconfined conditions when the head change in response to the addition is small relative to the saturated thickness – as is the case for this work.

The groundwater simulation model used in this work (coarse-grid version of C2VSim; Brush et al. 2013) was manipulated to generate the background groundwater heads (H) as well as the mounding response matrix (M) for the control locations. The background heads were based on running the unaltered model. Information for M was generated by 1) stripping all unmanaged hydrologic stresses from the model, 2) making a suite of runs with the altered model separately simulating a managed stress for each potential recharge location using a unit recharge volume (RVu) in the first time step of the model, 3) running the altered model once with no managed stresses and 4) calculating the differences in heads at control locations between the runs from steps 2 and 3. The resulting information for M is a set of vectors containing transient mounding responses at each control location for each potential recharge location. The vectors are then arranged in tableaus as described by Gorelick et al. (1993) to create a matrix M for each control location.

The end result is a groundwater elevation simulator that represents increases in elevation over time as a linear combination of responses to monthly recharge volumes. The responses 1) are produced by recharge events simulated for single time steps in any model element within the study area and any time step over the planning horizon, 2) scale with the magnitude of recharge volume and 3) can be summed to simulate combinations of recharge events over space and time.

Reformulation of Lagrange Multiplier for Berm Height

A generalized form of constraint equation (78) is as follows:

$$D \leq [(K_{scale}\, HB)(I/H_0 + \ln(\varepsilon))]/[1\text{-}e^{-(I_t/H_0 + \ln(\varepsilon))}] \tag{99}$$

When this constraint is binding in the linear programming solution, the Lagrange multiplier will be non-zero and indicate the change in the optimal value of decision variable D for an increase of 1 in the right-hand side (RHS). If HB in equation (99) were increased by 1, the RHS would increase by $[K_{scale}\,(I/H_0 + \ln(\varepsilon))]/[1\text{-}e^{-(I_t/H_0 + \ln(\varepsilon))}]$. Multiplying the Lagrange multiplier value from equation (99) by this quantity converts the original linear programming result, Lagrange multiplier for equation (99), into a Lagrange multiplier for HB. Summing the converted Lagrange multipliers for each model element over all time steps in the planning horizon provides a location-specific estimate for total increase in recharge over the planning horizon for a unit increase in berm height.

References

Afshar A, Zahrael A and Marino MA (2010) Large-scale nonlinear conjunctive use optimization problem: decomposition algorithm. J Water Resour Plann Manage 136(1):59-71. doi:10.1061/(ASCE)0733-9496(2010)136:1(59)

Ahlfeld DP and Mulligan AE (2000) Optimal management of flow in groundwater systems. Academic Press, San Diego

Ascott MJ, Gooddy DC, Wang L, Stuart ME, Lewis MA, Ward RS and Binley AM (2017) Global patterns of nitrate storage in the vadose zone. Nature Communications 8:1416. doi:10.1038/s41467-017-01321-w

Bachand PAM, Roy SB, Choperena J, Cameron D and Horwath WR (2014) Implications for using on-farm flood flow capture to recharge groundwater and mitigate flood risks along the Kings River, CA. Environ Sci Technol 48(23): 13601-13609. doi:10.1021/es501115c

Bachand PAM, Roy SB, Stern N, Choperena J, Cameron D and Horwath WR (2016) On-farm flood capture could reduce groundwater overdraft in Kings River Basin. Calif Agric 70(4):200-207. doi:10.3733/ca.2016a0018

Barlow PM and Reichard EG (2010) Saltwater intrusion in coastal regions of North America. Hydrogeology J 18(2):247-260. doi:10.1007/s10040-009-0514-3

Barlow PM and Leakey SA (2012) Streamflow depletion by wells – understanding and managing the effects of groundwater pumping on streamflow. U.S. Geological Survey Circular 1376

Bertoldi GL, Johnston RH and Evenson KD (1991) Ground water in the Central Valley, California – a summary report. U.S. Geological Survey Professional Paper 1401-A

Böhlke JK (2002) Groundwater recharge and agricultural contamination. Hydrogeology J 10(1):153-179. doi:10.1007/s10040-001-0183-3

Bouwer H (2000) Integrated water management: emerging issues and challenges. Agric Water Manage 45(3): 217-228. doi:10.1016/S0378-3774(00)00092-5

Brush CF, Dogrul EC and Kadir TN (2013) Development and calibration of the California Central Valley Groundwater-Surface Water Simulation Model (C2VSim), Version 3.03-CG. California Department of Water Resources. http://baydeltaoffice.water.ca.gov/modeling/hydrology/C2VSim/download/C2VSim_Model_Report_2016-03_vR374.pdf Accessed 1 April 2018

CADWR (2016) Basin boundary modifications. California Department of Water Resources. https://www.water.ca.gov/Programs/Groundwater-Management/Bulletin-118/Basin-Boundary-Modifications Accessed 30 March 2018

CADWR (2017) Flood-MAR - using flood water for managed aquifer recharge to support sustainable water resources, white paper – discussion draft. California Department of Water Resources. https://www.water.ca.gov/-/media/DWR-DSIWM-SIIB-Website-Files/Flood-MAR-White-Paper-Documents/Flood-MAR_Discussion-Draft_Nov-2017.pdf Accessed 31 March 2018

CADWR (2018a) Sustainable groundwater management website. California Department of Water Resources. California Department of Water Resources. https://www.water.ca.gov/Programs/Groundwater-Management/SGMA-Groundwater-Management/ Accessed 31 March 2018

CADWR (2018b) Water available for replenishment. California Department of Water Resources. https://www.water.ca.gov/-/media/DWR-Website/Web-Pages/Programs/Groundwater-Management/Data-and-Tools/Files/Statewide-Reports/WAFR/Final/Water-Available-for-Replenishment---Final-Report.pdf Accessed 20 April 2018

Cia X, McKinney DC and Lasdon LS (2001) Solving nonlinear water management models using a combined genetic algorithm and linear programming models. Adv Water Resour 24:667-676. doi:10.1016/S0309-1708(00)00069-5

Dahlke HE, Brown AG, Orloff S, Putnam D and O'Geen T (2018) Managed winter flooding of alfalfa recharges groundwater with minimal crop damage. Calif Agric 72(1):65-75. doi:10.3733/ca.2018a0001

Dillon PJ, Pavelic P, Page D, Beringen Hand Ward J (2009) Managed aquifer recharge, an introduction. Australia National Water Commission Waterlines Report Series 13

Dokoozlian NK, Petrucci VE, Ayars JE Clary CD and Schoneman RA (1987) Artificial ground water recharge by flooding during grapevine dormancy. J Am Water Res As 23(2):307-311.

Döll P, Schmied HM, Schuh C, Portmann FT and Eicker A (2014) Global-scale assessment of groundwater depletion and related groundwater abstractions: combining hydrological modeling with information from well observations and GRACE satellites. Water Resources Research 50: 5698-5720. doi:10.1002/2014WR015595

Famiglietti JS, Lo M, Ho SL, Bethune J, Anderson KJ, Syed TH, Swenson SC, de Linage CR and Rodell M (2011) Satellites measure recent rates of groundwater depletion in California's Central Valley. Geophysical Research Letters 38: L03403. doi:10.1029/2010GL046442

Farr TG, Jones C and Liu Z (2015) Progress report: subsidence in the Central Valley, California. Prepared for the California Department of Water Resources. http://water.ca.gov/groundwater/docs/NASA_REPORT.pdf Accessed 5 July 2017

Faunt CC (ed.) (2009) Groundwater availability of the Central Valley Aquifer, California, US Geol Surv Prof Pap 1766

Fogg GE, Carle SF and Green C (2000) Connected-network paradigm for the alluvial aquifer system. In Zhang D and Winter CL. (eds), Theory, modeling and field investigation in hydrogeology: a special volume in honor of Shlomo P. Neuman's 60th Birthday, Geological Society of America Special Paper 348, p 25-42.

Foster SSD and Chilton PJ (2003) Groundwater: the processes and global significance of aquifer degradation. Philos Trans R Soc B, 358(1440):1957-1972. doi:10.1098/rstb.2003.1380

Gailey RM (2013) Comments on hydrogeologic points of concern for the Kern River Watershed Coalition Authority area regarding Monitoring and Reporting Program Tentative Order R5-2013-XXXX waste discharge requirements general order for growers within the Tulare Lake Basin area that are members of third-party group. Prepared for Ernest A Conant, Young Wooldridge, LLP Counsel for Kern River Watershed Coalition Authority by The Source Group, Inc. April 2013. http://www.itrc.org/swrcb/Files/Gailey%20Comments%20on%20Hydrogeologic%20Points%20of%20Concern%202013%20.pdf Accessed 5 April 2018

Gale I (ed) (2005) Strategies for managed aquifer recharge (MAR) in semi-arid areas. United Nations Educational, Scientific and Cultural Organization, Paris

Geoffrion AM (1972) Generalized Benders decomposition. Journal of Optimization Theory and Application 10(4):237-260.

GFSAD30 (2017) Global food security analysis-support data at 30 meters (GFSAD30) project. https://www.croplands.org/app/map/finalmaps#Mosaic Accessed 28 March 2018

Goharian E, Gailey R, Medellin-Azuara J, Maples S, Adams LE, Sandoval-Solis S, Fogg GE, Dahlke HE, Harter T and Lund JR (2016) Whole watershed management to maximize total water storage: case study of the American-Cosumnes River Basin. American Geophysical Union Fall Meeting, San Francisco, CA.

Gorelick SM, Freeze RA, Donohue D and Keely JF (1993) Groundwater contamination, optimal capture and containment. Lewis Publishers, Boca Raton

Green CT, Fisher LH and Bekins BA (2008) Nitrogen fluxes through unsaturated zones in five agricultural settings across the United States. J Environ Qual 37(3):1073-1085. doi:10.2134/jeq2007.0010

Hanak E, Jezdimirovic J, Green S and Escriva_Bou A (2018) Replenishing groundwater in the San Joaquin Valley. Public Policy Institute of California. http://www.ppic.org/wp-content/uploads/r-0417ehr.pdf Accessed 20 April 2018

Hao Q, Shao J, Cui Y Zhang Q and Huang L (2018) Optimization of groundwater artificial recharge systems using a genetic algorithm: a case study in Beijing, China. Hydrogeology J (published online). doi:10.1007/s10040-018-1781-7

Harter T and Lund JR (2012) Addressing nitrate in California's drinking water with a focus on Tulare Lake Basin and Salinas Valley groundwater. Prepared for California State Water Resources Control Board by University of California Davis Watershed Sciences Center. March 2012. http://groundwaternitrate.ucdavis.edu/ Accessed 5 April 2018

Harter T and Dahlke H (2014) Out of sight but not out of mind: California refocuses on groundwater. Calif Agric 68(3):54-44. doi:10.3733/ca.v068n03p54

Harter T (2015) California's agricultural regions gear up to actively manage groundwater use and protection. Calif Agric 69(3):193-201. doi:10.3733/ca.E.v069n03p193

Hersh-Burdick R (2008) Effects of groundwater management strategies on the greater Sacramento water supply. MS thesis, University of California Davis. https://watershed.ucdavis.edu/shed/lund/students/HershBurdickThesis.pdf Accessed 31 March 2018

Howell TA (2001) Enhancing water use efficiency in irrigated agriculture. Agron J 93(2):281-289. doi:10.2134/agronj2001.932281x

Jurgens BC, Fram MS, Belitz K, Burow KR and Landon MK (2010) Effects of groundwater development on uranium: Central Valley, California, USA, Ground Water 48(6):913-928, doi:10.1111/j.1745-6584.2009.00635.x

Kiparsky M, Fisher AT, Hanemann MW, Bowie J, Kantor R, Coburn C and Lockwood B (2018) Recharge net metering to enhance groundwater sustainability. Center for Law, Energy & the

Environment, UC Berkeley School of Law, Berkeley, CA. https://www.law.berkeley.edu/wp-content/uploads/2018/04/CLEE_ReNeM_IssueBrief.pdf Accessed 20 April 2018

Kocis TN and Dahlke HE (2017) Availability of high-magnitude streamflow for groundwater banking in the Central Valley, California. Environ Res Lett 12(8):084009. doi:10.1088/1748-9326/aa7b1b

Konikow LF (2013) Groundwater depletion in the United States (1900-2008). U.S. Geological Survey Scientific Investigations Report 2013-5079

Liao L, Green CT, Bekins BA and Böhlke JK (2012) Factors controlling nitrate fluxes in groundwater in agricultural areas. Water Resources Research 48(6):W00L09. doi:10.1029/2011WR011008

Maples SR, Fogg GE, Maxwell RM, Liu Y (2017) Transitioning groundwater from an extractive resource to a managed water storage resource: geology and recharge in sedimentary basins. American Geophysical Union Fall Meeting, San Francisco, CA.

Marques GF, Lund JR and Howitt RE (2010) Modeling conjunctive use operations and farm decisions with two-stage stochastic quadratic programming. J Water Resour Plann Manage 136(3):386-394. doi:10.1061(ASCE)WR.1943-5452.0000045

Mushtaq H, Mays LW and Lansey KE (1994) Optimum operation of recharge basins. J Water Resour Plann Manage 120(6):927-943. doi:10.1061/(ASCE)0733-9496(1994)120:6(927)

McMahon PB, Dennehy KF, Bruce BW, Böhlke JK, Michel RL, Gurdak JJ and Hurlbut DB (2006) Storage and transit time of chemicals in thick unsaturated zones under rangeland and irrigated cropland, High Plains, United States. Water Resources Research 42(3):W03413. doi:10.1029/2005WR004417

Moran T, Choy J and Sanchez C (2014) The hidden costs of groundwater overdraft, Understanding California's groundwater, Stanford Water in the West. http://waterinthewest.stanford.edu/groundwater/overdraft/ Accessed 5 July 2017

Niswonger RG, Morway ED, Triana E and Huntington JL (2017) Managed aquifer recharge through off-season irrigation in agricultural regions. Water Resources Research 53(8):6970-6992. doi:10.1002/2017WR020458

O'Geen AT, Saal MBB, Dahlke H, Döll D, Elkins R, Fulton A, Fogg G, Harter T, Hopmans JW, Ingels C, Niederholzer F, Sandoval-Solis S, Verdegaal P and Walkinshaw M (2015) Soil suitability index identifies potential areas for groundwater banking on agricultural lands. Calif Agric 69(2):75-84. doi:10.3733/ca.v069n02p75

Perrone DD and Rhode MM (2016) Benefits and economic costs of managed aquifer recharge in California. San Francisco Estuary and Watershed Science 14(2):1-13. doi:10.15447/sfews.2016v14iss2art4

Portmann F, Siebert S and Döll P (2010) MIRCA2000 – Global monthly irrigated and rainfed crop areas around the year 2000: a new high-resolution dataset for agricultural and hydrological modeling. Global Biogeochem Cycles 24:GB1011. doi:10.1029/2008GB003435

RBI/WRIME (2011) South basin groundwater management plan. Prepared by Robertson-Bryan, Inc. and Water Resources & Information Management Engineering, Inc. for the South Area Water Council. October 2011

Reilly TE, Franke OL and Bennett GD (1987) The principle of superposition and its application in ground-water hydraulics. U.S. Geological Survey Techniques of Water-Resources Investigations of the United States Geological Survey, Book 3: Applications of Hydraulics, Chapter B6. https://pubs.usgs.gov/twri/twri3-b6/pdf/twri_3-B6_a.pdf Accessed 5 April 2018

RMC (2014) Groundwater and stream interaction in California's Central Valley: insights for sustainable groundwater management. Prepared by RMC Water and Environment for The Nature Conservancy. June 2014 https://www.scienceforconservation.org/assets/downloads/GroundwaterStreamInteraction_2016.pdf Accessed 31 March 2018

RMC (2015) Creating an opportunity: groundwater recharge through winter flooding of agricultural land in the San Joaquin Valley. Prepared by RMC Water and Environment for The California Water Foundation. October 2015. http://californiawaterfoundation.org/wp-content/uploads/2015/09/Creating%20an%20Opportunity%20On%20Farm%20Recharge%20Final%20Full%20Report%20(00306327xA1C15).pdf Accessed 31 March 2018

Scanlon BR, Reedy RC, Stonestrom DA, Prudic DE and Dennehy KF (2005) Impacts of land use and land cover change on groundwater recharge and quality in southwestern US. Global Change Biol 11(10):1577-1593. doi:10.1111/j.1365-2486.2005.01026.x

Scanlon BR, Reedy RC, Faunt CC, Pool D and Uhlman K (2016) Enhancing drought resilience with conjunctive use and managed aquifer recharge in California and Arizona. Environ Res Lett 11: 035013. doi:10.1088/1748-9326/11/3/035013

Schmidt CM, Fisher AT, Racz A, Wheat CG, Los Huertos M and Lockwood B (2012) Rapid nutrient load reduction during infiltration of managed aquifer recharge in an agricultural groundwater basin: Pajaro Valley, California. Hydrological Processes 26(15):2235-2247. doi:10.1002/hyp.8320.

Singh A (2014) Simulation-optimization modeling for conjunctive water use management. Agric Water Manage 141:23-29. doi:10.1016/j.agwat.2014.04.003

Sneed M, Brandt J and Solt M (2013) Land subsidence along the Delta-Mendota Canal in the northern part of the San Joaquin Valley, California 2003-10. U.S. Geological Survey Scientific Investigations Report 2013-5142

Tindula GN, Orang MN and Snyder RL (2013) Survey of irrigation methods in California in 2010. J Irrig Drain Eng 139(3):233-238. doi:10.1061/(ASCE)IR.1943-4774.0000538

UNESCO (2012) Managing water under uncertainty and risk, United Nations world water development report 4, vol 1. United Nations Educational, Scientific and Cultural Organization, Paris. http://www.unesco.org/new/en/natural-sciences/environment/water/wwap/wwdr/wwdr4-2012/ Accessed 28 March 2018

UNESCO (2015) Water for a sustainable world, United Nations world water development report 2015. United Nations Educational, Scientific and Cultural Organization, Paris. http://unesdoc.unesco.org/images/0023/002318/231823E.pdf Accessed 28 March 2018

USGS (2017) Drought impacts website. United States Geological Survey. https://ca.water.usgs.gov/data/drought/drought-impact.html Accessed 5 July 2017

Wada Y, van Beek LPH and Bierkens MFP (2012) Nonsustainable groundwater sustaining Irrigation: a global assessment. Water Resources Research 48(6):W00L06. doi:10.1029/2011WR010562

Wagner HM (1969) Principles of operations research. Prentice-Hall Inc., Englewood Cliffs, NJ

Walvoord MA, Phillips FM, Stonestrom DA, Evans RD, Hartsough PC, Newman BD and Striegl RG (2003) A reservoir of nitrate beneath desert soils. Science 302(5647):1021-1024. doi:10.1126/science.1086435

Weissmann GS, Zhang Y, Fogg GE and Mount JF (2004) Influence of incised-valley fill deposits on hydrogeology of stream-dominated alluvial fan. In Bridges JS and Hyndman DW (eds), Aquifer characterization. SEPM Vol. 80, p 15-28.

World Bank (2018) World Bank open data. https://data.worldbank.org/indicator/AG.LND.AGRI.K2?end=2015&start=2015&view=map Accessed 28 March 2018

CHAPTER 6 – THOUGHTS FOR FUTURE WORK

The material presented in this dissertation is the product of my long-term interest in groundwater science. Many of the ideas expressed have developed over extended periods of time while I conducted projects in my practice as a consulting hydrogeologist. Insights gained from visiting field sites were quite important for the results presented here. There is simply no substitute for seeing the physical setting, observing some of the physics in action, experiencing the challenges of collecting quality data, and talking to the stakeholders, regulators and contractors involved. The scope of a research problem is usually not fully understood upon first consideration and visiting the field helps to ripen problem formulation.

The work presented in this dissertation has furthered my thinking and set the stage for additional contributions:

- For well conduits, field examples and analysis of specific corrective actions (modifications to individual wells as well as management of well and wellfield pumping schedules) are largely absent from the literature. Data related to these topics are available from my consulting practice and I hope to prepare publications in the relatively near future.
- For economic externalities caused by groundwater withdrawals, there is an opportunity to expand the work presented here to more explicitly address variations in surface water availability, the resulting demands on groundwater, and the tradeoff between domestic well costs and agricultural opportunity costs. This work is underway. The methods and tools presented here might also be applied to other areas in California and beyond. Finally, the general approach might be extended to quantify other types of impact such as land subsidence and surface water depletion.

- For managed aquifer recharge, more detailed work on key physical processes and economic drivers would be beneficial. Also, the work could be updated to include improved groundwater flow models that cover the study area and are about to be released by the state (C2VSim and SVSim). Extensions of the work include those discussed earlier in this dissertation as well as considering application of the economic tradeoff approach considered in Chapter 4.

SGMA provides an opportunity to perform such work. Very specific questions are being asked and the data necessary to address these questions are becoming available. As stated in the introduction, it is an exciting time to work on groundwater issues in California – and perhaps elsewhere.