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Groundwater in Urban Areas

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Cover photos: Top: *Channel in the Newport Bay Watershed, Orange County, California*, Credit: John J. Dodge; Bottom Left: *California State University graduate students collect water samples from passive groundwater dewatering system in an urban stream channel*, Credit: Barry J. Hibbs; Bottom Right: *Groundwater monitoring*, Credit: Alfredo Granados-Olivas
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Groundwater in Urban Areas

Barry J. Hibbs

California State University, Los Angeles, USA

Kuprianov (2009) defines urban hydrology as "a science, part of land surface hydrology investigating the hydrological cycle, water regime and quality in urbanized territory." Urban hydrogeology is that part of the hydrological cycle focusing on urban groundwater processes and groundwater exchange with other parts of the hydrological cycle. By the year 2000, about half of the Earth's population lived in urban areas (Kuprianov 2009). Urbanization is projected to increase even more proportionally in the next fifty years, and urban development and urban sprawl will impact and increase dependence on groundwater systems while altering groundwater processes locally and regionally (Hibbs and Sharp 2012).

Physical changes associated with urbanization include increasing recharge from leaky utility systems, changes in irrigation and delivered water return flow, and modification of subsurface flow paths due to underground urban infrastructure. Chemical changes include groundwater and soil contamination from point and non-point sources, water quality impacts from storm-water control systems, and both modern and legacy pollution from urban development (Johns and Pope 1998; Rose 2007). Understanding these changes, their causes, and effects is necessary to address the critical and growing environmental and water resource issues of urban areas (Hibbs and Sharp 2012).

In an early paper on urban groundwater, Lerner (1990) discussed effects of urban infrastructure on groundwater budgets (Table 1). Water budgets in urban areas are modified by installation of "impervious" cover, widespread use of import water, construction of soakways and drainage swales, concentration of septic tanks, over-irrigation, and groundwater leakage to and from sewer and water pipes. These groundwater flow terms and associated water budgets in urban areas are difficult to quantify (Lerner 2002; Vazquez-Sune et al. 2010). Pipe leakage and delivered water return flow, for example, are frequently extrapolated from coefficients taken from often limited studies in other urbanized basins.

Modification of natural landscapes by urban sprawl provides many avenues of new and expanded research for groundwater scientists. Considerable work needs to be done to develop more reliable groundwater budgets in urban settings. Groundwater scientists are as important now as ever for providing these and other practical and theoretical contributions in urban hydrogeology. This is especially true where urban infrastructure is being retrofitted. Many new initiatives in stream restoration, for example, have been proposed in recent years in areas where concrete stream linings are targeted for removal and replacement with unlined channel. Classical engineering design posits that concrete stream linings protect urban areas by sending urban runoff quickly downstream away from urban developments (Farassati 2016). Experience has shown that the high velocity of runoff along lined channels sometimes causes unlined channel below the lined section to be damaged and destabilized due to erosion, leading to loss of land along the streambank (Colosimo and Wilcock 2007; Farassati 2016) (Figure 1). As a result, multi-objective planning in urban storm water control now considers a different, more comprehensive engineering approach. With appropriate and comprehensive engineering retrofit, removal of channel lining can be achieved as part of a comprehensive watershed restoration

effort. Simultaneous placement of on-site retention facilities in urban areas, such as dry wells, green belts, or local recharge basins reduces the volume of urban runoff while providing distributed points of recharge to aquifers. The urban retrofit creates a partial return to more natural recharge conditions.

Stream restoration projects also open up many avenues of research for groundwater scientists. An emerging area of research is the study of biogeochemical cycling during hyporheic zone interactions while the stream is recovering after restoration (Hester and Gooseff 2010). Another area of research requiring extensive hydrogeological expertise relates to large-scale artificial recharge projects that are developing in urban centers. Evaluation of sites for artificial recharge factors in many levels of analysis, such as characterizing aquifer permeability, storage properties, and continuity of aquifer strata. Hydrochemical analysis is equally important and includes studies on the possible impacts of mixing waters of different sources on trace element mobilization in aquifer sediments. Evaluating proximity of recharge facilities to legacy contaminant sources is another area of concern, as is persistence of emerging contaminants where treated recycled water is the recharge source.

Numerous opportunities are available for groundwater scientists to engage in these and other levels of hydrogeological analysis in urban centers. Accordingly, the goal of this theme issue is to bring together a collection of papers to showcase ongoing research and policy as these relate to groundwater flow systems in and near urban areas. I am indebted to the authors of this theme issue for their efforts in this endeavor. The general scope and diversity of their eight papers is summarized herein.

Several papers focus on water budget analysis and modeling in and around urban centers. Schnaar et al. describe integrated water budget analysis to quantify groundwater inputs and outputs to San Diego Creek and Newport Bay Watersheds, California. Hauwert describes stream gain/loss studies to recalculate forms of recharge in the Barton Springs Segment of the Edwards Aquifer, Texas. Granados et al. perform rainfall/runoff modeling of desert catchments in and near Ciudad Juarez, Mexico, to support groundwater recharge augmentation projects in the Hueco Bolson aquifer.

Other papers focus on environmental tracers and hydrochemical analysis for evaluating recharge sources, groundwater residence time, and trace element mobilization. Eastoe and Gu use stable water isotopes and radioisotopes to evaluate long-term changes in perched and regional aquifers in the Tucson Basin, Arizona. Larsen et al. use multiple environmental tracers and integrated modeling to evaluate modern water contributions to the Memphis aquifer, Tennessee. Darling evaluates possible arsenic release from aquifer sediments due to mixing of native groundwater and recharge water at an artificial recharge site near Clearwater, Florida.

The remaining papers emphasize policy and methodology. Hodder discusses regulatory compliant decommissioning of water wells in urban areas in California and elsewhere, while Hibbs et al. discuss stream/aquifer interactions along grout-lined channels in urban catchments and describe opportunities for future research along lined channels.

I wish to acknowledge the reviewers of the papers published in this theme issue. Several reviewers have more than two decades of experience conducting groundwater assessments in urban zones. The reviewers include, in no particular order, Peter Van Noort, Stephen Osborn, Bruce Darling, Benjamin Hagedorn, Thomas Meixner, Claudia Espinosa Villegas, Michael Harrison, M. Hassan Rezaie Boroon, Anna Doro-on, John Sharp, Carlos Gutiérrez Ojeda, and Radu Boghici. A few reviewers asked to remain anonymous. Karl Williard and Jackie Crim, co-editors of *Journal of Contemporary Water Research and Education*, provided many helpful suggestions and assistance throughout the process.

This theme issue is dedicated to my mentors for their many years of study and for training students in the field of urban hydrogeology. Foremost among them are David Kreamer, Darryll Pederson, Jerry Ayers, John Sharp, Bob Huber, Alan Dutton, John Ashworth, and Ernie Weber. These dedicated scientists and engineers conducted many prominent groundwater studies in urban areas, and involved dozens of mentees in hydrological training and research.

Table 1. Urban controls on groundwater recharge (adapted from Lerner, 1990).

Urban Factors Increasing Recharge	Urban Factors Decreasing Recharge
Impervious cover reducing evapotranspiration[1]	Impervious cover limiting rain recharge
Changing urban microclimate (+)	Changing urban microclimate (-)
Urban runoff drywells and catch basins	Infiltration to sewer and water pipes
Artificial recharge wells and basins	Infiltration to storm sewers
Concentrated density of septic tanks	Extraction and export to other basins
Exfiltration from sewer and water pipes	
Exfiltration from storm sewers	
Import water return flow	

[1]Wiles and Sharp (2008) noted that leakage through impervious cover must also be considered.



Figure 1. Erosion of unlined channel below a lined section of Barranca Channel, San Diego Creek Watershed, California (photo credit, Henry Jones).

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Comprehensive Groundwater Balance Development to Characterize Selenium Loading to Surface Water Channels in Orange County, California

*Gregory Schnaar[1,2], John J. Dodge[1,3], and Stephen J. Cullen[1,4]

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Abstract: Selenium concentrations are elevated above water quality criteria in several tributaries of Newport Bay. A watershed-scale hydrogeologic study was performed to identify processes driving selenium transport in the former "Swamp of the Frogs" area (cities of Santa Ana, Irvine, and Tustin). The former Swamp of the Frogs area was a depositional environment that historically received regional surface water runoff and accumulated naturally occurring selenium-bearing sediments from the larger watershed. Development beginning in the early 20th century included surface water channel installation that lowered the water table and drained the wetland, mobilizing selenium. A comprehensive groundwater/surface water balance was conducted in order to constrain hydrologic processes that continue to drive selenium mobilization into the surface water channels. The groundwater budget included precipitation and irrigation percolation modeling and groundwater flow mapping from thousands of publicly-available groundwater level measurements. Groundwater balance results indicated that lateral groundwater flow from upgradient alluvial-fan recharge areas accounts for most of the groundwater input to the former wetland area with additional input supplied by deep percolation of precipitation within the former wetland area itself, deep percolation of irrigation, and sewer/water line leakage. Overlaying selenium concentration data from groundwater wells and surface water sampling locations confirmed that selenium concentrations are similar in groundwater and nearby groundwater-to-surface water discharge points (e.g., weepholes). Results supported a Best Management Practice (BMP) Strategic Plan developed in compliance with a regulatory order prescribing numeric limitations on groundwater-related discharges of selenium in the watershed.

Keywords: Monterey formation, selenium, groundwater balance, distributed parameter watershed model

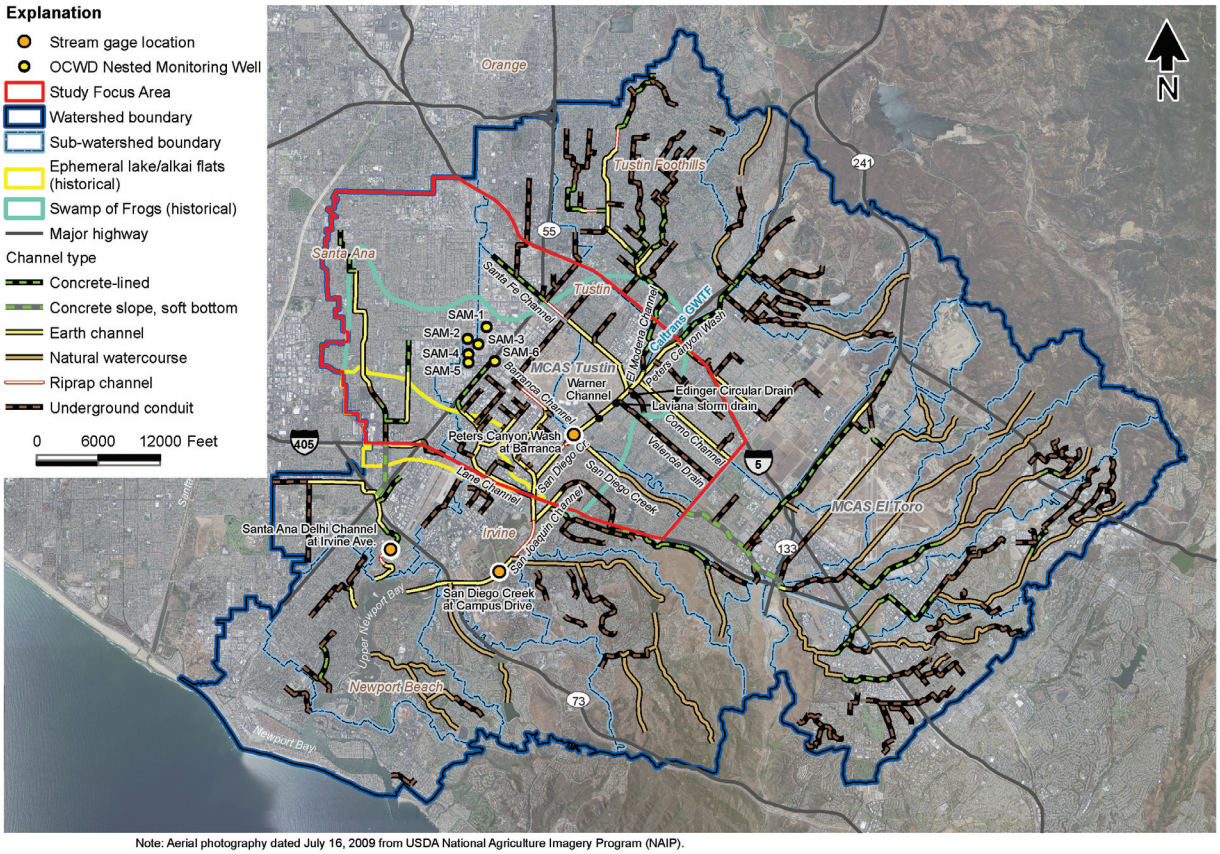
Agricultural and urban development has modified and destroyed wetland environments throughout the United States, often with unintended consequences (e.g., Winter 1988; Dahl 2011). One result of wetland drainage is oxygenation of previously anaerobic environments, which fundamentally alters aqueous redox-geochemistry and in some cases contaminant mobility (e.g., Borch et al. 2010). Selenium is an essential micronutrient in animals; however, it can have toxic effects at elevated concentrations (Hamilton 2004). In highly reducing anaerobic environments selenium occurs primarily as selenide (-II), which is tightly bound to sediments and relatively immobile. In aerobic systems selenium occurs primarily as the more oxidized selenate (Se[VI]), which is not tightly bound to sediments. Given these differences, in cases where selenium occurs, one consequence of wetland drainage is selenium mobilization into groundwater and surface waters.

Central Orange County, California was a wetland named "La Cienaga de las Ranas", or "Swamp of the Frogs," until urbanization of the area occurred in the early 20th century. The former wetland was a depositional environment, evaporative alkali salt flat, and groundwater discharge area (Meixner et al. 2004). The area received regional streamflow

and surface water runoff for millennia until urbanization starting around the turn of the 20th century led to significant hydrological changes. The wetland area was largely drained when land was developed for agricultural use. By the mid-20th century, with increased development, the entire wetland area was fully drained and surface water flow was captured by a network of interconnected surface water and flood control channels extending across central Orange County (Trimble 1998). The surface water channels exit the former wetland area as San Diego Creek on the east and Santa Ana Delhi Channel on the west, both of which flow south into Upper Newport Bay. Peters Canyon Wash is also a significant channel in the area that directs surface water flow from northern Tustin to the south through Irvine, and then merges with San Diego Creek (Figure 1).

This project focuses on the former wetland, ephemeral lake, and surrounding areas where selenium has been detected at elevated concentrations in surface water and includes areas primarily within the cities of Santa Ana, Irvine, and Tustin. Selenium concentrations elevated above applicable toxicity-standards have previously been observed in the surface water channels in this area (e.g., Hibbs and Lee 2000), which is of concern for fish, waterfowl, and other fauna. Selenium concentrations in surface water are often greater than the California Toxics Rule (CTR) chronic water quality criterion for selenium in surface water of 5 µg/L. Groundwater-related discharges have previously been recognized as a dominant source of selenium within the surface water channels (e.g., Hibbs and Lee 2000; Meixner et al. 2004; Hibbs et al. 2008).

Selenium occurrence has been determined to be likely due to marine geologic formations located in upgradient areas of the watershed that are naturally high in selenium content. The Monterey Formation in particular is well documented to contain elevated levels of selenium (USGS 2002). Over an extended time period, selenium-bearing sediments from these formations were eroded and deposited in the



Note: Aerial photography dated July 16, 2009 from USDA National Agriculture Imagery Program (NAIP).

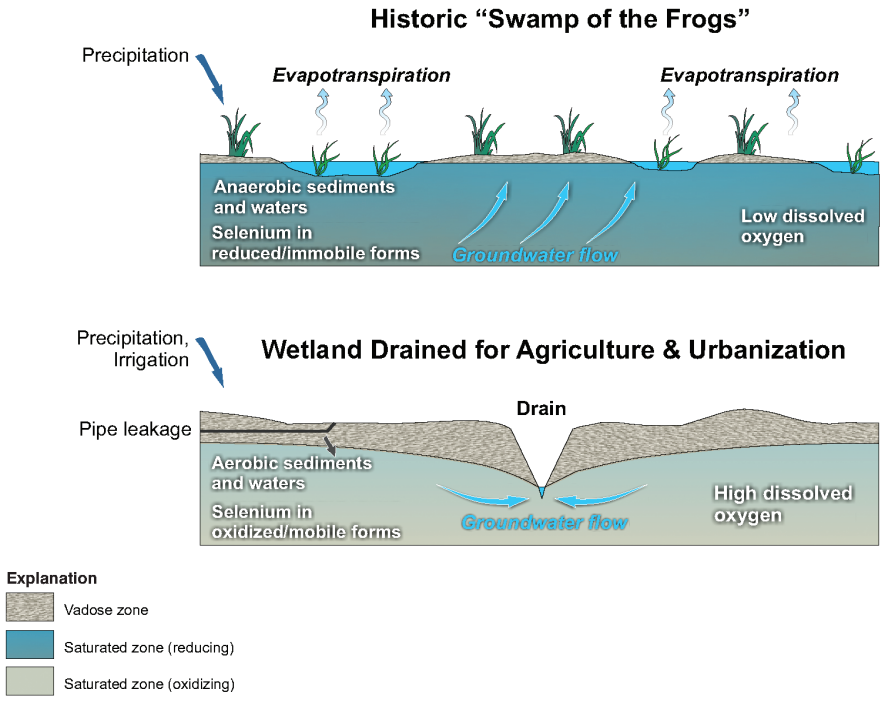
Figure 1. Project location and study focus area.

downslope wetland areas. In addition, selenium was leached by floodwaters and transported downslope via overland runoff to the wetland. The wetland areas were moderately to highly reducing anoxic environments, within which selenium would be reduced primarily to selenide (-II) (e.g., Meixner et al. 2004), and therefore was relatively immobile. Cyclic flooding and evaporation may have played a role in concentration of selenium deposits. Selenium may have been deposited preferentially on the edges of the wetland, where it would be first subject to immobilization by vegetation and anoxic sediments (Hibbs et al. 2008). Land use changes and channelization of the watershed in the 20th century fundamentally altered the redox geochemistry of the shallow groundwater. Lowering of the groundwater table introduced oxygen to previously hydric soils, which likely oxidized selenium to more mobile forms, including selenate (Se[VI]). Figure 2 presents a conceptual diagram of hydrologic processes and selenium transport in the historic "Swamp of the Frogs" and the same area following wetland drainage.

The Santa Ana Regional Water Quality Control Board (RWQCB) issued Order No. R8-2004-0021 (amended later by the Order R8-2007-0041) prescribing, in part, limitations on groundwater-related discharges of selenium in the watershed. Parallel to the effluent limitations, the RWQCB order allows for an alternative compliance approach, which led to the formation of the Nitrogen and Selenium Management Program (NSMP) Working Group to develop and implement a comprehensive Work Plan to address nitrogen and selenium loading. The Working Group consists of the County of Orange, Orange County Flood Control District (OCFCD), the watershed cities, individual dischargers, major land developers, water districts, RWQCB, and environmental groups.

The NSMP developed a draft Best Management Practice (BMP) Strategic Plan (NSMP 2011) based on the work products generated by many subtasks of the Work Plan. The BMP Strategic Plan recognized that a better understanding of groundwater in the watershed is necessary, and therefore proposed a phased approach for further watershed studies. The hydrogeologic characterization and water balance discussed in this report is an "Early Action Item" of the BMP Strategic Plan.

The present study supported policy recommendations regarding mitigation of selenium impacts by improving general understanding



Source: After Hibbs and Lee, 2000

Figure 2. Conceptual model of selenium deposition and remobilization in historic "Swamp of the Frogs."

of present-day drivers for continued selenium mobilization to the surface water channels. An understanding of selenium loading and drivers of selenium mobilization is necessary for effective mitigation planning. Pertinent questions posed by stakeholders and regulators that the water-balance study sought to address included the following:

1. What are the sources of water that cause perennial dry-weather flow in the surface water channels within the study area?
2. To what extent does over-irrigation contribute to dry-weather flow and selenium mobilization?
3. How can we expect selenium mobilization to vary in wet and dry precipitation years?
4. Where are selenium "hot-spots" in groundwater and surface-water that could be targeted for remedial action?
5. Are previous estimates of selenium loading to the surface-water channels consistent with the quantitative water balance and available groundwater selenium sampling data?

Methods

This study focuses on estimating the groundwater balance for the former wetland areas based on available data and hydrogeologic analyses. The analysis starts with estimating the magnitude of groundwater inputs and outputs for the surficial unconfined aquifer (Shallow Aquifer) within the bounds of the study focus area, which is 31 square miles and is shown on Figure 1. The thickness of the Shallow Aquifer was based on hydrogeologic interpretation conducted by the Orange County Water District (OCWD) and included in their regional groundwater model (OCWD 2012a). OCWD stratigraphic data are the most comprehensive available data source for aquifer thickness and are based on review of regional geologic reports, and hydrogeologic interpretation of lithologic logs, geophysical logs, depth-specific water level data, and water quality data.

This study analyzes water years 2005 through 2011 (i.e., October 1, 2004 through September 30, 2011). This timeframe was chosen to focus on recent conditions, while also covering a period with a range of climatic conditions and a statistical profile similar to the historical record. The average annual precipitation over the time period 2005 to 2011 (13.30 inches, as measured at Santa Ana Fire Station) is similar to the average of 1917 to 2011 (13.26 inches), while the median is slightly lower (9.81 and 11.67 inches, respectively). Water years 2005 and 2011 represent relatively wet periods (28.8 and 19.4 inches rainfall, respectively), while water year 2007 represents a relatively dry period (2.77 inches rainfall).

Groundwater inputs for the study area include deep percolation of precipitation and irrigation water (P_p, P_i), lateral groundwater inflow from upgradient portions of the Newport Bay watershed (GW_i), and deep percolation of water leaking from sewer and water lines (S). Groundwater outputs include production discharged to the sanitary sewer (E_r), production discharged to storm drains/ surface water channels (E_d), lateral groundwater outflow to downgradient areas (GW_o), vertical flux of groundwater downwards out of the Shallow Aquifer into deeper aquifer units (V), and passive discharge to surface water channels, storm drains, and subdrains (SW_o), through weeps, seeps, springs, and via the channel bottom. Mathematically, the net groundwater balance, or change in groundwater in storage within the study area, (ΔGWs), is represented by the following equation:

$$\Delta GW_s = [P_p + P_i + GW_i + S] - [E_r + E_d + GW_o + V + SW_o] \quad \text{(Equation 1)}$$

The annual average magnitude of each of the parameters listed on the right side of Equation 1 was estimated using standard hydrologic methods or obtained from literature references, as detailed in the following sections. The groundwater balance was estimated individually for each water year during the study time period (2005 through 2011), as well as on an average basis for the entire study time period. It was assumed that for each water year of our analysis, the sum of all groundwater inputs should approximately equal the sum of all groundwater outputs. This approach assumes that the annual change in groundwater storage within the study area (ΔGW_s) is zero, which is reasonable based on the observation of only minor and temporary changes in groundwater levels as measured in study area groundwater wells (Figure 3).

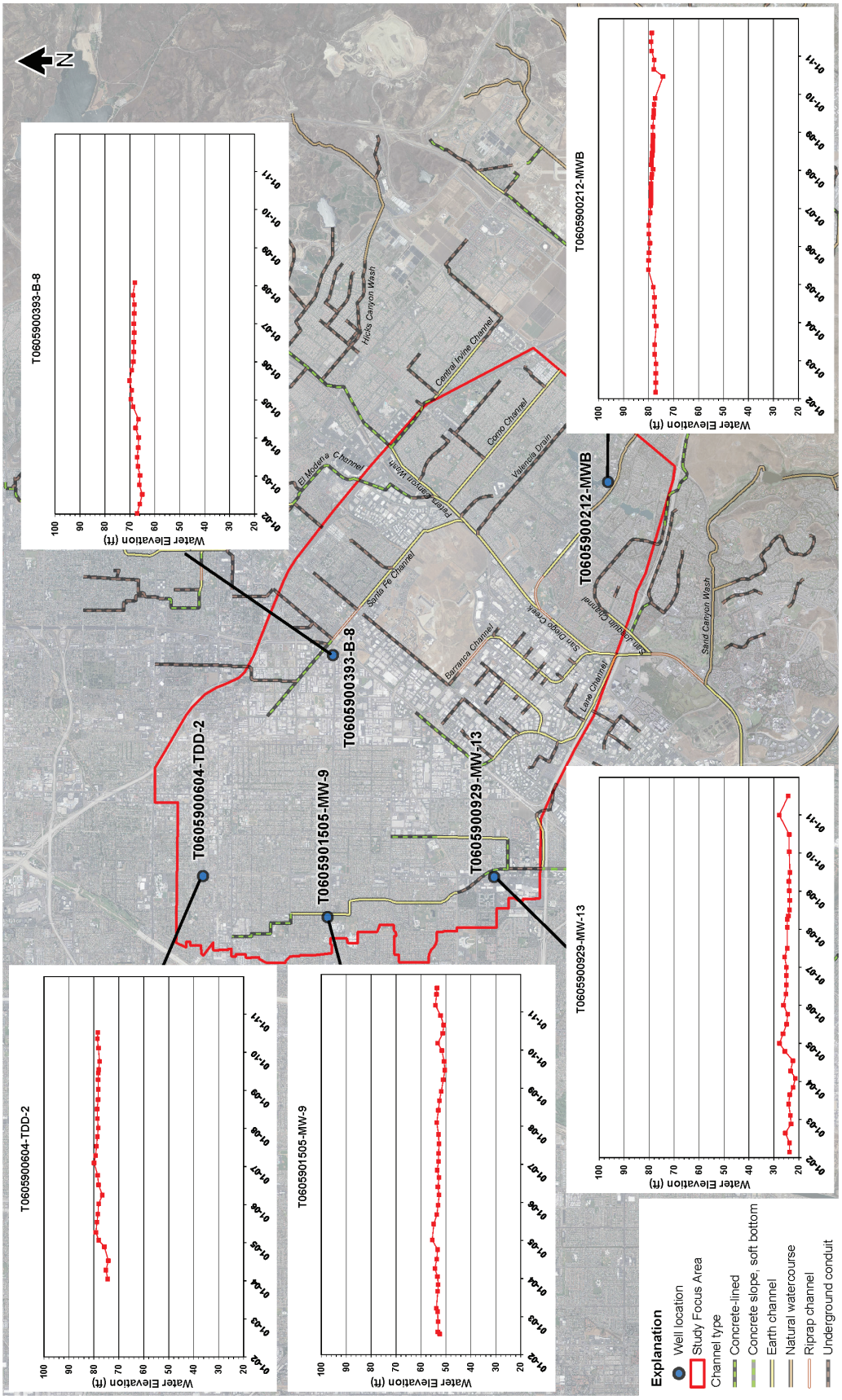


Figure 3. Time-series hydrographs for selected shallow wells.

Lateral Groundwater Inflow and Outflow

A groundwater potentiometric surface contour map was developed as a basis for quantifying groundwater inflow and outflow within the study area. Hundreds of active monitoring wells are screened in the Shallow Aquifer within the study area. The majority of these wells are associated with State-regulated groundwater remediation activities (e.g., dry-cleaner sites, industrial sites, gasoline stations), and quarterly water level and water quality data are largely publically available on-line from the State Water Resources Control Board (SWRCB) Geotracker website.

Depth-to-water and measuring-point elevation data from all available individual remediation sites within the study area were accessed from Geotracker. Typically, several monitoring wells are located at each Geotracker site. Data from a subset of the Geotracker sites located within the study area were excluded because measuring-point elevation values were anomalous, and apparently not based on a surveyed elevation benchmark. Following exclusion of these sites, 44 sites had depth-to-groundwater data available for inclusion in the database. Irvine Ranch Water District (IRWD) provided data for several additional shallow monitoring wells located east of Peters Canyon Wash.

The well data were filtered to include data only for wells with a shallow screen (less than or equal to fifty feet below ground surface), and with groundwater level data available during our study time period (water years 2005 to 2011). Approximately 148,000 individual well measurements were available that fit these criteria, and data were processed with a GIS-based computer program (Milbert and Zilkoski 2003), so that all water level data would be in a consistent vertical datum. Groundwater elevation was calculated for all the available data by subtracting the depth to water from the referenced measuring point elevation.

Many surface-water channels within the study area are recognized as gaining reaches (e.g., Meixner et al. 2004). Under non-storm flow conditions, the elevation of the top of the surface water column at any one location in a channel is therefore the same as the adjacent groundwater elevation at that location. Surveyed channel elevations for a 1.5 mile stretch of Peters Canyon Wash north of the confluence with San Diego Creek were provided by Orange County (OC) Watersheds. Surveyed surface-water elevation data were not available for the remainder of the study area, and this was recognized as a key data gap.

Average groundwater elevations for the study time period (October 2005 to October 2011) were determined for each Geotracker site within the study area (Figure 4). For sites with multiple individual monitoring wells, the site average values represent an average of all measurements made from all wells at that site. Data from nested monitoring wells indicate that groundwater elevations can vary vertically within the Shallow Aquifer. The calculated Geotracker site average groundwater levels therefore represent an average over the vertical screened interval of the monitoring wells used in each calculation (generally between the water table and fifty feet below ground surface (bgs)), a horizontal spatial average (across the horizontal extent of the site), and a temporal average over the study time period. The standard deviation of the values used to calculate averages for each site are plotted, and in most cases are relatively small (i.e., less than 10 percent of average groundwater elevation).

Lateral groundwater inflow was estimated based on Darcy's Law of groundwater flow:

$$Q = K \cdot i \cdot A \qquad \text{(Equation 2)}$$

where Q is groundwater discharge (L^3/T), K is hydraulic conductivity (L/T), i is the hydraulic gradient (L/L), and A is the cross-sectional area (L^2).

The groundwater elevation contour map (Figure 4) was used to estimate groundwater flow direction and the hydraulic gradient. Five "slice planes" were interpolated perpendicular to groundwater inflow flow direction at the upgradient edge of the study area (ENT-1 through ENT-5), and two slice planes were interpolated generally perpendicular to the groundwater direction at the downgradient boundary (EXT-1, EXT-2). Equation 2 was solved for each slice plane in order to estimate the lateral groundwater inflow and outflow.

For each segment, the cross-sectional area (A) is estimated as the product of the length of the slice plane and the vertical saturated thickness. The saturated thickness was determined from

the difference in the groundwater elevation and the elevation of the top of the aquitard unit that separates the Shallow Aquifer and underlying aquifer units. Aquitard elevation data were provided by OCWD (2012a). The hydraulic conductivity for each slice plane was initially based on hydraulic conductivity data for the study area also provided by OCWD (2012a). The hydraulic conductivity values of slice planes ENT-1, ENT-2, and ENT-3 were adjusted during finalization of the groundwater balance, as discussed below. The original values of 175 to 200 feet per day were adjusted to 93 feet per day.

Groundwater Recharge from Precipitation and Irrigation

Recharge from precipitation and irrigation was estimated using a soil-moisture balance model, the Distributed Parameter Watershed Model (DPWM), developed previously by Daniel B. Stephens & Associates, Inc. Application of the DPWM allows for quantitative estimates based on site-specific climatological, geologic, soils, land use, and vegetation factors. DPWM was developed based on the Mass Accounting System for Soil Infiltration and Flow (MASSIF) model developed by Sandia National Laboratories (2007). The DPWM is similar in concept to water balance models used by the USGS (e.g., Net Infiltration Model (INFIL) Hevesi et al. 2003). The model relies on the widely accepted United Nations FAO-56 procedure for computing actual evapotranspiration (AET) from the reference evapotranspiration (ET_0), which is estimated using the Penman-Monteith method (Allen et al. 1998, 2005). Water budget components accounted for in the DPWM model include precipitation, irrigation, bare soil evaporation, transpiration, runoff, run-on, soil water storage, and deep percolation (recharge).

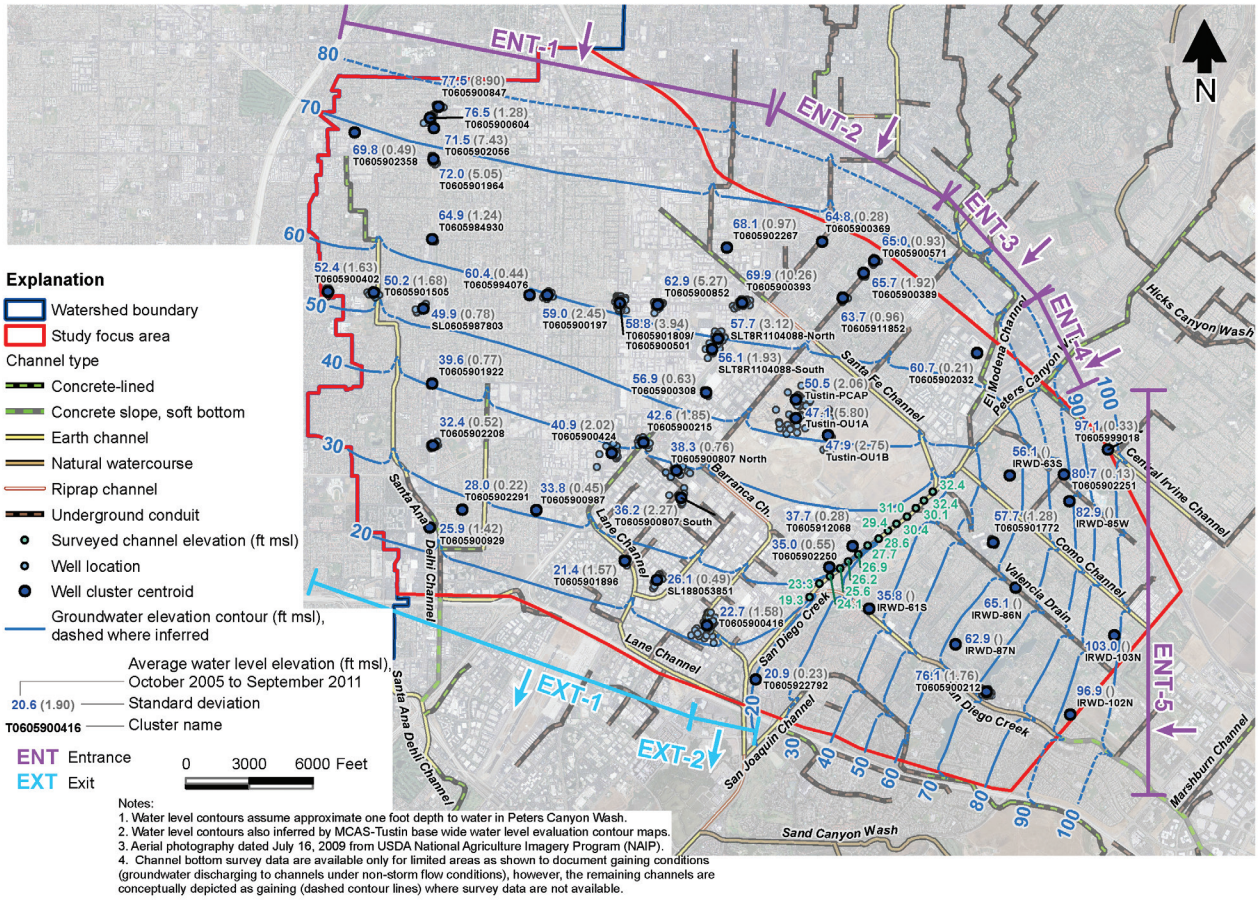


Figure 4. Groundwater elevation of the Shallow Aquifer.

The DPWM model domain covers the study area and entire upgradient watershed, and consists of uniform grid cells 295 feet (90 meters) on a side. The model was run for the complete study time period, water years 2005–2011, using a daily time-step routine. Model data were collected from weather stations, site-specific literature sources, and general literature sources, and/or were estimated from other properties. Key model parameters and data sources are summarized in Table 1.

Routing of precipitation within the study area is significantly impacted by urbanization, including covering of the land surface with impervious surfaces (i.e., pavement, asphalt, rooftop). The presence of impervious surfaces is generally understood to increase runoff and concomitantly decrease infiltration and deep percolation to groundwater. For the purpose of DPWM modeling, a "percent impervious surface" was assigned to each model grid cell (Homer et

Table 1. Key parameters used in Distributed Parameter Watershed Model (DPWM).

DPWM Parameter	Data Source	Notes/Methods
Precipitation Rate [L/T]	CIMIS; PRISM	• Based on hourly precipitation data from local gage Irvine #75 • Extrapolated over entire model area using PRISM
Irrigation Rate [L/T]	Irvine Ranch Water District (IRWD)	• 2,800 gal/ac-d = 37.7 in/yr for vegetated areas of developed cells • Median of Irvine Ranch Water District estimated irrigation requirements for developed areas • CA DWR estimates for agricultural areas (minor) • Monthly irrigation rates calculated from annual averages using monthly fraction of reference ET for CIMIS Zone 16
Soil Hydraulic Conductivity [L/T], Depth [L]	Soil Survey Geographic (SSURGO) Database (USDA NRCS)	• Unmodified for vegetated areas of developed land uses
Soil "Field Capacity" [-]	SSURGO, ROSETTA	• Soil properties related to water retention (i.e., van Genuchten parameters) estimated using ROSETTA based on SSURGO soil property data, model calculates "field capacity"
"Impervious" Surfaces Properties	USGS NLCD GIS data for impervious surfaces (2006); Literature survey (see following slide)	• Fraction impervious land surface from 2006 USGS National Land Cover Dataset (NLCD) • Hydraulic conductivity from literature survey (1·10-7 m/s =0.03 ft/d) • Meant to be generally representative of urban pavements with cracks
Rooting Depth for Grass/Turf [L]	Wu 1985 (Univ. of CA Cooperative Extension)	• "Middle" value for various grass species

al. 2015). The impervious surface portion of each cell in the DPWM model received no irrigation, contained no vegetation, and accepted all runoff from surrounding areas of the cell. Furthermore, runoff on impervious surfaces within each cell was routed directly to the impervious surface portion of the next downgradient cell, and so-on, until reaching the closest mapped surface water channel. DPWM routing of water therefore approximately represented storm flow to storm water drains and then to surface water channels. Because all runoff is routed from one impervious surface to the next when present, "focused recharge" (e.g., from building roofs to vegetated areas via downspouts) is not accounted for. Model results may therefore underestimate the amount of recharge from precipitation as influenced by focused recharge.

A literature review was conducted in order to evaluate the value of hydraulic conductivity assigned to the impervious surfaces section of each model grid cell. Previous studies included double-ring infiltrometer experiments on pavements and asphalts (Wiles and Sharp 2008), weighing lysimeter experiments of infiltration through fresh-laid asphalt (Raimier et al. 2004), laboratory permeameter testing on new asphalt cores (Huang et al. 1999), and previous urban watershed/climate modeling (Dupont et al. 2006). Based on this review, a hydraulic conductivity value of $1 \cdot 10^{-5}$ cm/s (0.028 feet per day) was assigned to the impervious surfaces within the model domain.

Sewer and Water Line Exfiltration/Infiltration

IRWD and the City of Santa Ana provided sewer and water line data used to estimate the amount of pipe exfiltration (loss). Exfiltration was estimated as a percentage of total consumption using a 5 percent loss estimate from Sharp (2010). Pipe infiltration may also occur, which would be considered a groundwater outflow component. Pipe infiltration is particularly relevant when considering sewer lines located beneath the water table, as reportedly occurs within the study area. However, specific data regarding pipe exfiltration and infiltration are currently not available. According to IRWD, sewer pipelines in the IRWD service area range from a minimum of 7 feet deep (below grade) to over 30 feet deep. Many of these pipelines are below shallow groundwater, making them subject to the risk of infiltration, not exfiltration (IRWD 2012). Given the range of groundwater depths (5 to 46 feet bgs) and the range of sewer line depths (7 to 30 feet bgs) in the study area, infiltration from sewer lines in the IRWD was assumed, in the absence of other data, as 2.5 percent loss, which is half of the 5 percent loss assumed by Sharp (2010).

Vertical Groundwater Outflow

The Shallow Aquifer is underlain by an aquitard unit that appears to be continuous throughout the study area, which is in turn underlain by regional aquifer units collectively referred to as the Principal Aquifer. Groundwater flow between the Shallow Aquifer and the Principal Aquifer is governed by differences in hydraulic head between the two units, and is restricted by the low hydraulic conductivity of the interlaying aquitard.

Vertical groundwater flow between the Shallow and Principal Aquifers was estimated based on calculation of the difference in groundwater elevation between the two units. Groundwater levels were generally higher in the Shallow Aquifer compared to the Principal Aquifer, indicating downwards vertical flow. The vertical groundwater flow (Q_v, L^3/T) can be calculated from:

$$Q_v = K_v \cdot i_v \cdot A_v \qquad \text{(Equation 3)}$$

where K_v is the vertical hydraulic conductivity, A_v is the cross-sectional area of vertical flow, and i_v is the vertical hydraulic gradient. Groundwater elevation contour maps of the Principal Aquifer are available from OCWD for each year of the study time period, and cover the entire extent of the study area and surrounding watershed (OCWD 2012b). GIS methods were used to interpolate the difference in groundwater levels in the Shallow and Principal Aquifer throughout the study area for each year, using the yearly OCWD maps for the Principal Aquifer, and the groundwater elevation contour map generated herein (Figure 3) for the Shallow Aquifer.

The study area was discretized into approximately 5,000 cells with dimensions of 416 feet by 417 feet. The vertical gradient

was calculated for each cell as the difference in hydraulic head between the Shallow and Principal Aquifer divided by the representative distance between the two units (500 feet). The vertical hydraulic conductivity was assumed to be $2.8 \cdot 10^{-3}$ feet per day ($1 \cdot 10^{-6}$ cm per second), which is representative of the fine-grained aquitard unit (e.g., Freeze and Cherry 1979).

Shallow Groundwater Production

Groundwater production from the Shallow Aquifer within the study area is limited to remediation and dewatering projects. Remediation systems and some dewatering facilities discharge water to the sanitary sewer system, while other dewatering facilities discharge either to storm drains, which flow into the surface water channels, or directly into the surface water channels themselves. Available data regarding groundwater production rates were compiled from available sources, including the City of Irvine, CalTrans, Nitrogen and Selenium Management Program (NSMP 2009), Geotracker, and Enviro Compliance Solutions Inc. (ECS 2012).

Passive Groundwater Discharge to Surface Water Channels

Groundwater flow to surface water channels in the study area is through direct discharge from dewatering projects, and passive discharge through weeps, seeps, springs, and the channel bottom. Direct monitoring data available to estimate passive groundwater discharge to surface water channels are limited. For example, although there are several surface water gages throughout the watershed, only one permanent surface water gage is located within the study area (Peters Canyon Wash at Barranca, Figure 1).

Passive groundwater discharge to surface water channels was first evaluated by analysis of available downstream surface water gage data. Daily streamflow records were analyzed to estimate the groundwater discharge (baseflow) component of the streamflow hydrograph. The surface water channels exit the study area as San Diego Creek to the east and Santa Ana Delhi Channel to the west. Streamflow gages are located on each surface water channel downstream of the study area (Figure 1), and streamflow data were provided by OC Watersheds. Baseflow in the surface water channels of the study area includes passive groundwater discharge (SW_o), and discharge by dewatering projects (E_d).

Daily streamflow records from the gages for San Diego Creek at Campus Drive and Santa Ana Delhi Channel at Irvine Avenue during the study time period were processed with digital filtering methods to estimate the baseflow component of the surface water flow hydrograph (Lim et al. 2005).

Passive groundwater discharge to surface water channels was estimated as the difference in the sum of groundwater inputs and the sum of remaining groundwater outputs. Equation 1 is rearranged to the following in order to solve for passive groundwater discharge:

$$SW_o = [P_p + P_i + GW_i + S] - [E_r + E_d + GW_o + V] \quad \text{(Equation 4)}$$

The reasoning behind this approach is that for the final groundwater balance, the sum of all groundwater inputs should approximately equal the sum of all groundwater outputs. This assumes that the annual change in groundwater storage within the study area (ΔGW_s) is zero, a reasonable assumption based on the observation of only minor and temporary changes in groundwater levels (Figure 3). Equation 4 was solved for each water year during the study time period, and also on an average basis over all years; it was assumed based on available hydrograph data (Figure 3) that increases in groundwater inflow during wet years is offset by increased outflow (i.e., discharge) within the same water year. A limitation of applying Equation 4 is that errors from previous water-balance calculations are compounded in estimation of groundwater discharge to surface-water channels; therefore, an additional adjustment step was included as described below.

Stream hydrograph results were used to constrain estimates using the groundwater balance method. Values of the total groundwater discharge to surface channels ($SW_o + E_d$) were compared using the two methods. Initially, values of total groundwater discharge to the surface channels from the groundwater balance method were larger than values estimated using the stream hydrograph

method by an average of approximately 43 percent. To minimize the difference in estimated discharge rates using the two methods, the following adjustments were made.

A review of input parameters used in the groundwater balance revealed that hydraulic conductivity parameters used in the groundwater inflow estimation had a significant impact on the overall groundwater balance and were prone to uncertainty. The hydraulic conductivity corresponding to the groundwater "entering" slice planes 1, 2, and 3 (Figure 4) was decreased to lower the value of GW_i, which concomitantly decreased the estimated value of SW_o. The hydraulic conductivity values were decreased to minimize the difference in the average total groundwater discharge ($SW_o + E_d$) estimated using the water balance approach and the stream hydrograph analysis. Initial values of hydraulic conductivity for the region of the entering slice planes 1, 2, and 3 provided by OCWD ranged from 175 to 200 feet per day, and were adjusted to 93 feet per day using this calibration procedure. The value of 93 feet per day is considered reasonable, as it is representative of aquifer sand units (e.g., Freeze and Cherry 1979), and is similar in magnitude to the hydraulic conductivity of other areas of the study area provided by OCWD.

Selenium Data Compilation

NSMP (2009) compiled available selenium data in passive groundwater discharge points ("surfacing groundwater"), surface water, sediments, and biota. We sought to overlay the selenium sampling data collected from surface-water sampling points (NSMP 2009) with data collected from groundwater monitoring wells in order to evaluate similarity in groundwater and surface water selenium concentrations and selenium flux from groundwater to surface water. Selenium data from NSMP (2009) were complemented with selenium data from samples collected from groundwater monitoring wells. Geotracker sites were screened for the availability of selenium data, and additional reported data were compiled from Hibbs and Lee (2000), Meixner et al. (2004), MCAS-Tustin (ECS 2012), IRWD (2012), OCWD (2013), and from a previously developed database provided by OC Watersheds.

Results and Discussion

Groundwater Balance

On an average basis, the principal groundwater input component is lateral groundwater inflow (14,970 ac-ft/yr, or 80 percent of total average input) (Table 2 and Figure 5). The original source of groundwater inflow to the study area is predominantly recharge of precipitation in the alluvial fan head areas along the foothills in the upgradient areas of the watershed. This same water supply likely fed the "Swamp of Frogs" wetland prior to development of the surface water channels.

The groundwater inflow estimate of 14,970 ac-ft/yr compares favorably to estimates of groundwater recharge in the upgradient areas of the watershed. OCWD (2009) estimates groundwater recharge from precipitation along the foothills of the Irvine Subbasin of 14,000 ac-ft/yr, and along the Tustin Hills of 6,000 ac-ft/yr, for a total of 20,000 ac-ft/yr in the areas of the watershed upgradient of the study area. DPWM model runs were conducted for the upgradient area of the watershed in order to validate the estimated groundwater inflow. Recharge estimates ranged from 14,400 to 17,700 ac-ft/yr, which generally agrees with the OCWD estimate of 20,000 ac-ft/yr, and the calculated groundwater inflow to the study area of 14,970 ac-ft/yr. Based on DPWM modeling, approximately 91 percent of the upgradient recharge was due to precipitation, while the remaining 9 percent was due to irrigation in the developed upgradient areas of the watershed.

The principal groundwater output component is passive discharge to surface water channels (12,157 ac-ft/yr, or 65 percent of average output). This is consistent with previous studies, including geochemical investigations demonstrating significant upwelling of groundwater into surface water channels (e.g., Hibbs and Lee 2000; Bechtel 2002; Meixner et al. 2004; Hibbs et al. 2008). For example, Hibbs et al. (2008) report 2006 field monitoring of discharge from four recently installed drains to Peters Canyon Wash. Geochemical signatures of the discharging water indicated it was likely dominated by groundwater flow. A combined discharge rate of 0.4 cubic feet per second (290 ac-ft/yr) was measured from these four drains alone.

Results are consistent with the general understanding that the study area is a groundwater convergence zone (former wetland) which is now drained by the surface channel network, in part, to maintain a lower water table. A strong groundwater convergence zone exists in the direct vicinity of Peters Canyon Wash and San Diego Creek south of its confluence with Peters Canyon Wash, with groundwater flowing into this area from several directions. The groundwater convergence zone is indicated by groundwater elevation contours that appear to "pinch in" around water elevations in the surface water channels (Figure 4). The mapped extent of confirmed groundwater convergence zones are limited by the availability of surveyed surface channel elevation data. If additional surface channel survey channel data were available, similar convergence zones would likely be confirmed along other stretches of the surface channels within the study area.

Comparatively wet and dry water years were evaluated in order to understand the water balance under varying climatic conditions. Over the study time period, the wettest water year is 2005 (Table 2), which is predicted to have a total groundwater input/output of 25,892 ac-ft/yr. In contrast, the driest year during the study time period was 2007, with a total groundwater input/output of 16,680 ac-ft/yr. Intermediate water years (2006, 2008, 2009, and 2010) exhibit a groundwater balance similar to the dry year 2007. These results are explained by the relatively minor recharge from precipitation and irrigation during all but the wettest years, as predicted by the DPWM modeling.

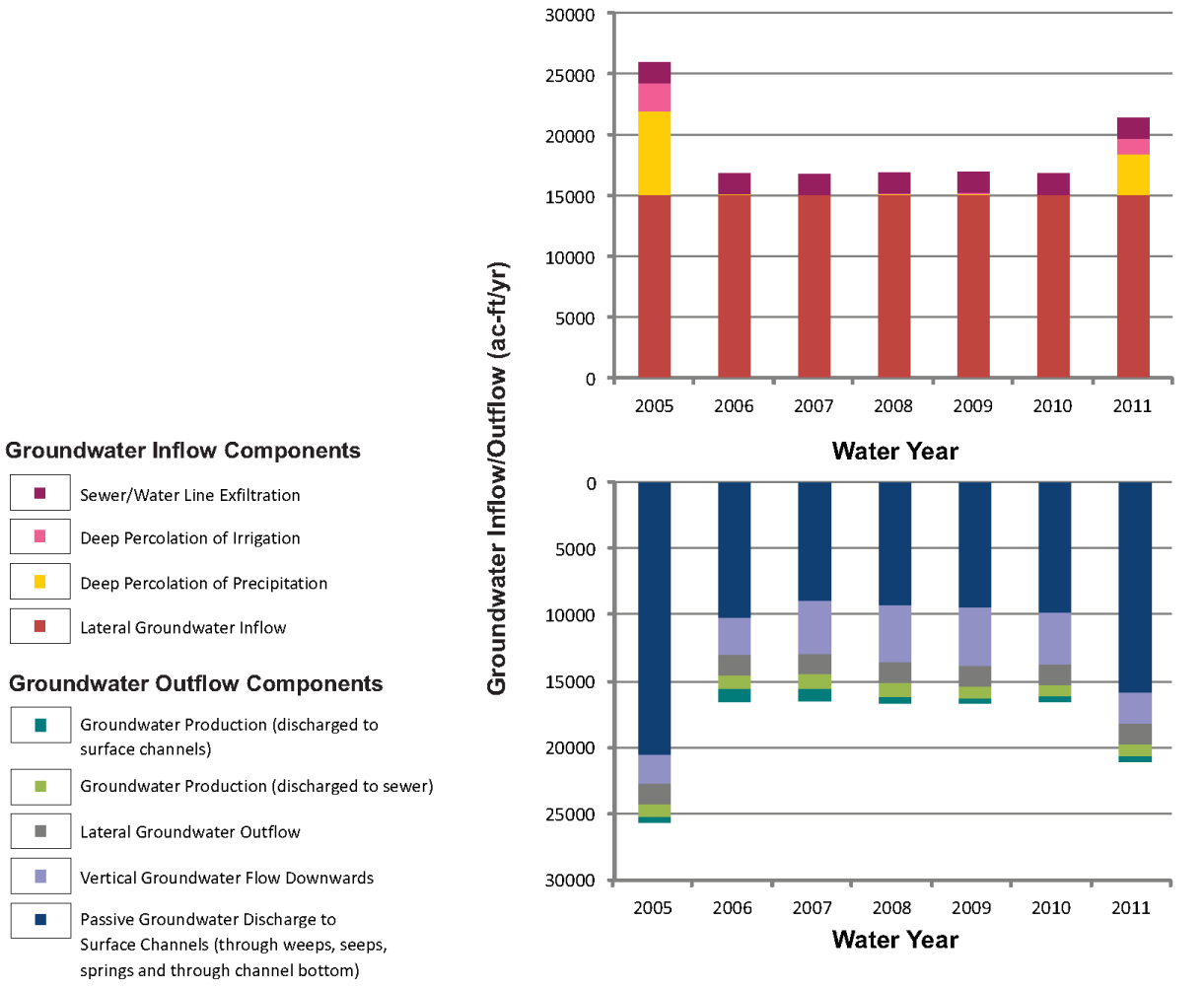


Figure 5. Water balance results.

Table 2. Water balance estimation results.

Category	Component (ac-ft/yr)	Water Year							Average	
		2005	2006	2007	2008	2009	2010	2011	ac-ft/yr	%[a]
Total Precipitation and Irrigation over Study Focus Area	Precipitation	46,214	11,852	2,156	11,370	11,336	8,878	32,497	17,757	-
	Irrigation (IRWD 2012)	21,526	21,526	21,526	21,526	21,526	21,526	21,526	21,526	-
Groundwater Inputs	Deep Percolation of Precipitation	6,893	49	0.4	68	74	29	3,334	1,492	8
	Deep Percolation of Irrigation	2,319	0.5	0.1	72	96	0.4	1,290	540	3
	Sewer/Water Line Exfiltration	1,709	1,709	1,709	1,709	1,709	1,709	1,709	1,709	9
	Lateral Groundwater Inflow	14,970	14,970	14,970	14,970	14,970	14,970	14,970	14,970	80
	Total Groundwater Input	25,892	16,729	16,680	16,819	16,849	16,709	21,303	18,712	-
Groundwater Outputs	Lateral Groundwater Outflow	1,556	1,556	1,556	1,556	1,556	1,556	1,556	1,556	8
	Vertical Groundwater Flow Downwards	2,208	2,812	4,071	4,363	4,461	3,976	2,416	3,473	19
	Groundwater Production (discharged to sewer)	943	1,046	1,119	1,095	866	860	904	976	5
	Groundwater Production (discharged to surface channels)	405	951	907	419	388	379	399	550	3
	Passive Groundwater Discharge to Surface Channels (through weeps, seeps, springs, and through channel bottom)	20,780	10,364	9,027	9,386	9,577	9,938	16,029	12,157	65
	Total Groundwater Output	25,892	16,729	16,680	16,819	16,849	16,709	21,303	18,712	-

[a]Percentage of total groundwater input or output, as applicable.

Flow of water through the vadose zone may contribute to loading of selenium into the Shallow Aquifer (e.g., Meixner et al. 2004; Hibbs et al. 2008). The components of the water balance that flow through the vadose zone within the study area are therefore of particular interest, though they comprise only a small percentage of the overall water balance. These components include deep percolation of irrigation and precipitation, and exfiltration from sewer and water lines. During wet years, vertical flow through the vadose zone appears to be driven primarily by recharge from precipitation. Given the available data, the water balance calculations indicate that during dry and intermediate years, recharge from the vadose zone is driven primarily by sewer and water line exfiltration. However, higher quality data for irrigation inputs and sewer/water line infiltration/exfiltration are needed to better characterize recharge from the vadose zone.

Recharge from irrigation is significant only during wet years, and accounts for less vertical flow compared to recharge from precipitation. These observations are consistent with results reported by Meixner et al. (2004) and Hibbs et al. (2008) that groundwater within the study area in some instances exhibits an isotopic signature similar to local precipitation, while in other instances exhibits an isotopic signature consistent with mixing of local precipitation and imported water that may have leaked from sewer/water lines or percolated from irrigation.

A sensitivity analysis for the overall groundwater balance indicates that hydraulic conductivity is the most sensitive parameter in this analysis (Figure 6). Additional parameters evaluated included irrigation rate, groundwater pumping rate, and pipe exfiltration rate. The most significant parameter evaluated was the hydraulic conductivity of the upgradient areas of the study area (Entering Zones 1, 2, and 3 on Figure 4). As discussed above, the value of hydraulic conductivity for this area was constrained by comparing the overall calculated groundwater balance and stream baseflow. Irrigation rates used in the DPWM, groundwater pumping (discharged to sewer), and hydraulic conductivity of exiting zone 1 (Figure 4) have a relatively minor impact on the total groundwater flow balance.

Selenium Discharge Assessment

The geometric mean selenium concentration for each monitoring well site was plotted on a map with the surface-water sampling data near groundwater discharge points ("surfacing groundwater") data (Figure 7). Selenium concentrations in groundwater are elevated throughout the study area, with concentrations generally exceeding the CTR chronic water quality criteria for selenium in surface water of 5 µg/L, and in many areas exceeding the U.S. Environmental Protection Agency maximum contaminant level for groundwater of 50 µg/L. Overlaying the groundwater data from wells and the surfacing groundwater data confirms that selenium concentrations are similar in groundwater sampled from wells and nearby passive groundwater discharge points. Selenium "hot spots" (>40 µg/L) appear to be in the vicinity of Valencia Drain and Como Channel, MCAS-Tustin, and north of Lane Channel. The groundwater well data also reveals that elevated selenium levels (>10 µg/L) are observed outside of the boundary of the former Swamp of Frogs, along the northern and eastern boundaries of the study area.

NSMP (2009) includes a detailed assessment of selenium loading to surface water channels from various sources, including urban runoff, dewatering facilities, remediation projects, agricultural discharges, atmospheric deposition, open space, and passive groundwater discharge. NSMP (2009) estimates passive groundwater discharge to be the largest source of selenium in the San Diego Creek subwatershed, accounting for an average of 730 lbs/yr of the 1,060 lbs/yr total from all sources (69 percent).

We evaluated this estimate of selenium loading from groundwater to the surface water channels of the San Diego Creek subwatershed to assess if it is consistent with the water balance and selenium data evaluation. The following equation was used to back-calculate the selenium concentration in groundwater (C_{Se}) that would result in an average annual loading of 730 lbs/yr:

$$C_{Se} = \frac{F_{lb/y} \bullet (4.536 \bullet 10^{8})}{SW_{o} \bullet (1.233 \bullet 10^{6}) \bullet 0.4} \qquad \text{(Equation 5)}$$

where F is the reported loading rate in units of

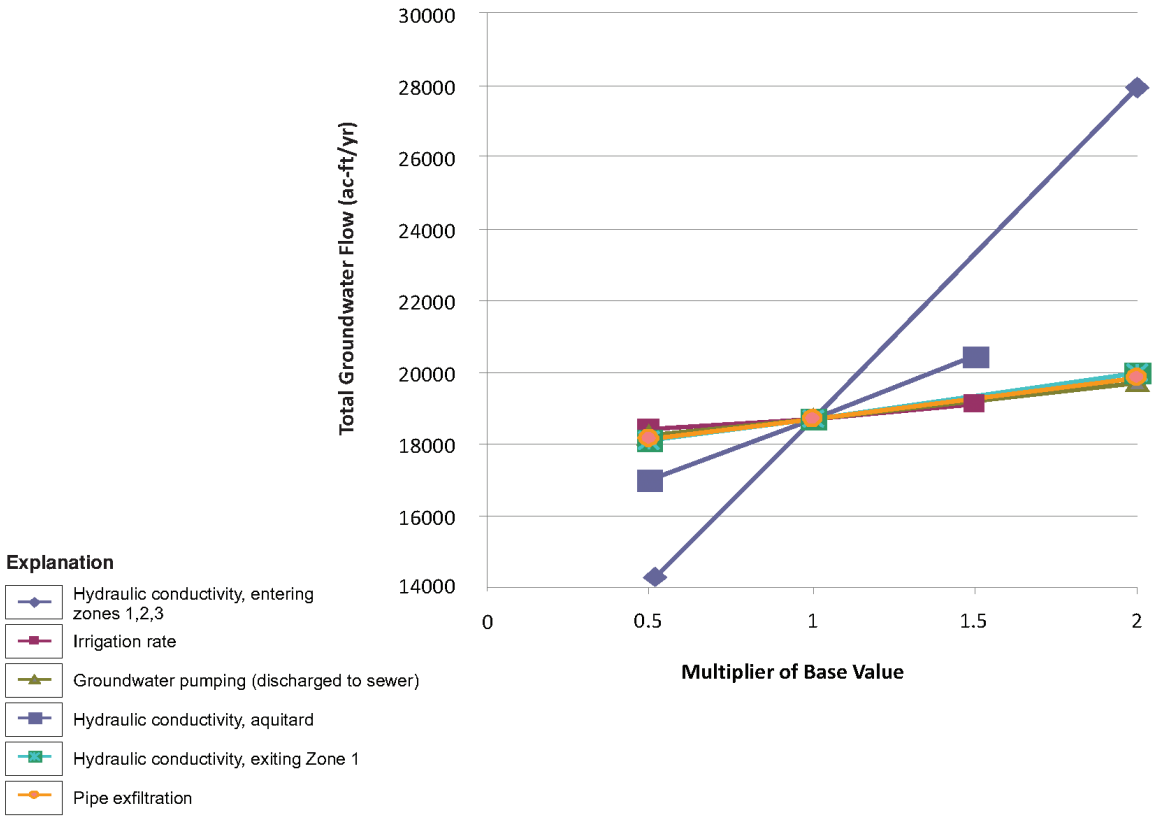


Figure 6. Sensitivity analysis, groundwater balance.

lbs/yr, $4.536 \cdot 10^8$ is a conversion factor from units of lbs to µg, SW_o is the average annual flow rate of passive groundwater discharge to the surface water channels in units of ac-ft/yr, $1.233 \cdot 10^6$ is a conversion factor from units of ac-ft to liters (L), and 0.40 is a factor for the mitigation of selenium concentrations within the surface water channels after discharge from groundwater, as reported by Meixner et al. (2004) and used in the NSMP (2009) calculations.

The value of SW_o for San Diego Creek and its tributaries was obtained from the groundwater balance (Table 2). The calculated value of C_{Se} for the San Diego Creek subwatershed is 56 µg/L, which appears reasonable based on available selenium concentration data in groundwater summarized in Figure 7. The NSMP (2009) estimated loading rates are generally consistent with the groundwater balance and selenium data assessment completed in this study.

Summary

The following summarizes results from the hydrogeologic characterization and water balance development:

- On average, approximately 80 percent of total groundwater input in the study area (18,712 ac-ft/yr) is due to lateral groundwater inflow (14,970 ac-ft/yr). The next largest recharge sources are sewer/water line exfiltration (9 percent or 1,709 ac-ft/yr), infiltrating precipitation (8 percent or 1,492 ac-ft/yr), and infiltrating irrigation (3 percent or 540 ac-ft/yr).
- On average, approximately 65 percent of total groundwater output in the study area (18,712 ac-ft/yr) is due to discharge to surface water channels (12,157 ac-ft/yr). The next largest sinks are vertical groundwater flow (19 percent or 3,473 ac-ft/yr), groundwater outflow (8 percent or 1,556 ac-ft/yr), and groundwater

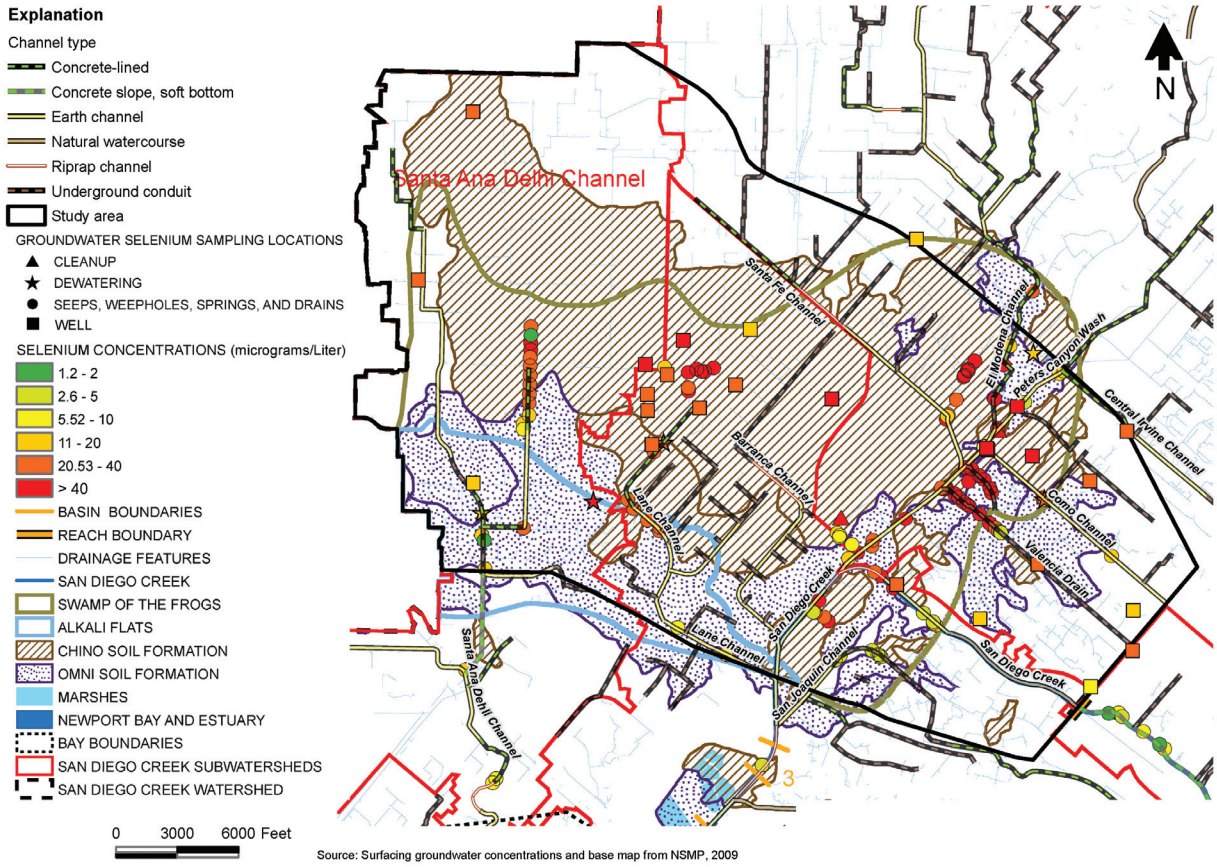


Figure 7. Selenium concentrations in groundwater and surfacing groundwater.

extraction from shallow wells (8 percent or 1,526 ac-ft/yr).

- In wet years (such as 2005), the relative contribution of the various groundwater sources changes to 58 percent lateral groundwater inflow, 27 percent infiltrating precipitation, 9 percent infiltrating irrigation, and 7 percent sewer/water line exfiltration. About 80 percent of the outflow is due to discharge to surface water channels (20,780 ac-ft/yr), 9 percent is due to vertical flow downwards, 6 percent is due to groundwater extraction, and 6 percent is due to lateral groundwater outflow.
- In relatively dry years (such as 2007), lateral groundwater inflow accounts for about 90 percent of the groundwater input. About 10 percent is due to sewer/water line exfiltration and less than 1 percent is due to infiltrating precipitation and/or irrigation. About 55 percent of the outflow is due to discharge to surface water channels (9,027 ac-ft/yr), 24 percent is due to vertical flow downwards, 12 percent is due to groundwater extraction, and 9 percent is due to lateral groundwater outflow.

The water balance was used to verify prior estimates of selenium discharge to the channels and assess proposed engineering solutions to control the discharge. The selenium discharge concentration calculated for the San Diego Creek subwatershed using the water balance is 56 µg/L, which appears reasonable based on available selenium concentration data in groundwater.

Conclusions

Results from our study improve conceptual understanding of the regional hydrologic system and drivers for continued selenium mobilization in the watershed. The majority of groundwater influx into the former wetland area is from groundwater underflow from upgradient portions of the larger watershed (Figure 4); this water originates

primarily as precipitation-related recharge in alluvial-fan head areas north and east of the former wetland area. This conceptual understanding of the local hydrologic system is consistent with the fact that this area was formerly a perennial wetland that was fed by groundwater underflow.

To a more limited extent, shallow groundwater occurs today due to deep percolation of precipitation, irrigation, and sewer/water-line leakage within the former wetland area. Over-irrigation in the former wetland area of residential/commercial turf is not a main driver for the presence of shallow groundwater or shallow groundwater-discharge to the surface water channels. However, selenium mobilization may occur from water percolation through the vadose zone within the former wetland area from deep percolation of precipitation and irrigation and sewer/water-line leakage. Vertical percolation through the vadose-zone in the former wetland area is considered to be significantly greater in wet years as compared to dry years (Figure 5).

Perennial groundwater discharge to the surface water channels occurs due to the presence of shallow groundwater and construction of the surface water channel network below the shallow groundwater table (and in certain instances continued surface-water channel lowering due to erosion). The presence of the surface-water channel network limits long-term increases in groundwater elevation and groundwater-in-storage within the former wetland area, as water readily drains into the surface water channels when groundwater levels rise (i.e., "the bath-tub overflow drain effect"; Figure 3). Groundwater discharge to the surface-water channels is therefore greatest in wet years (Figure 5); however, groundwater elevations do not change markedly from year-to-year.

Selenium hot-spots in groundwater have been identified in certain areas that are also known to be proximal to observed hot-spots within the surface-water network (Figure 7). These areas may be targeted for remedial actions, including surface-water diversion and/or treatment.

Several data gaps were identified during completion of the study, including (1) depth-discrete soil and sediment sampling data to delineate selenium impacts and source areas, and to distinguish vadose-zone leaching as a selenium source separate from leaching from saturated sediments; (2) hydraulic properties of impervious surfaces within the watershed to better constrain percolation estimates in these areas; (3) extensive and depth-discrete selenium groundwater sampling data from throughout the former wetland area have not been collected as part of a consistent and contemporaneous sampling program; (4) sewer/water line exfiltration/infiltration measurements are generally not available to constrain estimates of the importance of this source in the groundwater balance; (5) surface-water channel floor elevation data are lacking to compare to groundwater elevation data in identification of gaining surface water reaches; (6) few surface-water flow gages are available to estimate baseflow and related selenium discharge rates; and (7) local turf irrigation rates are generally not available.

Future work could be focused on additional data collection, including a soil and groundwater sampling program, sewer/water-line infiltration/exfiltration studies, irrigation practices surveys and local irrigation applied water measurements, and the development of feasibility studies that evaluate the merits and/or means of mitigating selenium concentrations in the watershed.

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Stream Recharge Water Balance for the Barton Springs Segment of the Edwards Aquifer

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Abstract: Channel water balance analysis based on streamflow loss coupled with traced groundwater basins allow refinement of upland autogenic recharge values comparable to those derived from more precise site-scaled measurements. Based on total discharge and calculated flow loss in major stream channels, the channel contributions to recharge on the Barton Springs Segment of the Edwards (Balcones Fault Zone) Aquifer are: Onion Creek (32.6%), Barton Creek (less than 10.5%), Slaughter Creek (7.2%), Bear Creek (6.2%), Blanco River (6%), Little Bear Creek (3.4%), and Williamson Creek (1%). Major stream channels contributed from 56% to 67% of total groundwater recharge to the groundwater system, 17% of which was stream autogenic recharge, while upland autogenic recharge and other smaller sources contributed the remaining 33% to 44%. As a percent of precipitation over the Edwards Aquifer outcrop source area, autogenic recharge (upland and tributary recharge in the major stream channels) was 22% to 28%, compared with a recharge value of 28% of precipitation from climate tower water balances in Central Texas. Prior to delineation of groundwater basins using injected tracers, older channel water balance studies determined that upland autogenic recharge was only 15% of the total groundwater recharge (about 1% of precipitation). This study suggests that the channel water balance approach, based on gauged streamflow and spring discharge, is more accurate when one incorporates groundwater basins delineated by groundwater tracing, especially in karst aquifers. Refinement of an appropriate catchment area for the calculation of autogenic recharge also significantly affected the results of a precipitation water balance.

Keywords: *water balance, karst, Edwards Aquifer, Barton Springs, recharge*

The amount of rainfall that recharges an aquifer varies more directly with rock type than soil characteristics, surface morphology, or annual precipitation patterns (Maxey 1964; Weyer 1971; Misstear et al. 2009). Karst terranes are generally composed of limestone and contain internal drainage morphology that allows rainfall to infiltrate rapidly and recharge the aquifers (Jennings 1985). Much surface runoff over karst limestone areas flows only short distances before infiltrating into sinkholes (dolines) or highly permeable soils (Meinzer 1923, 123; White 1977). Autogenic recharge originates from direct precipitation over the recharge zone, a portion of which recharges within the intervening area (upland autogenic recharge) and the remaining recharges via runoff to the major stream channels (stream autogenic recharge). In karst areas of the world, autogenic recharge is typically 20% of precipitation or greater (Hauwert and Sharp 2014). Allogenic recharge sources originate as runoff from upstream of the outcrop of a karst aquifer.

The limitations of recharge measurement techniques are summarized by Lerner et al. (1990); Scanlon et al. (2002); De Vries and Simmers (2002); Taylor and Greene (2008); and Sophocleous (2009). "*Because recharge is not easy to measure directly, estimates of its value are prone to large errors... The most serious and common type of error is an error in the conceptual model. It arises when the recharge process is not fully understood, or when too many simplifying assumptions are made"* Sophocleous (2009, 21). The channel water balance method measures recharge using the difference in upstream and downstream flow stations; flow diversions and additions; stream evapotranspiration; and storage that does not reach the water table (Lerner 1997; Nimmo et al.

2005). Runoff and infiltration occurring between the gauging stations are difficult to distinguish. Furthermore, in areas where adequate direct groundwater tracing is not incorporated, it is not known with certainty where the measured stream recharge actually discharges from an aquifer (Lerner et al. 1990). Recharge and storage characteristics of aquifers are also quantified using hydrograph separation of stream baseflow and spring discharge recessions (Rorabaugh 1964; Hobbs and Smart 1986; Tallaksen 1995; Florea and Vacher 2006), although the results can be misinterpreted in areas where the water sources are not known (Anderson and Burt 1980; Halford and Mayer 2000). For example, stream flow measured in a watershed may originate from recharge in another watershed (Omernik 2003). Springflow recessions are commonly used to distinguish quick and slow components of karst aquifer discharge, but tracing data is rarely available to identify what the water sources are (Long 2009). In water balances of karst systems, traditional methods of delineating groundwater basins such as water-level mapping, hydraulic well testing, geochemical characterization, and modeling alone are not adequate and direct groundwater tracing techniques are necessary (Ford and Williams 1989; White 1993; Taylor and Greene 2008). Historically, channel water balances were the primary method on which groundwater recharge of an area was estimated in Texas (Bennett and Sayre 1962; Garza 1962; Maclay 1995), but these older studies lacked direct tracing of source areas and relied on assumptions of groundwater divides and groundwater sources.

The Barton Springs Segment of the Edwards (Balcones Fault Zone) Aquifer (or Barton Springs Segment, Figure 1) is a karstic, federally designated sole source aquifer that discharges to the Colorado River through Barton Creek. Its largest discharge site is Barton Springs (Figure 2), a set of at least four springs which are the habitat of the federally listed endangered Barton Springs Salamander (*Eurycea sosorum*) and Austin Blind Salamander (*Eurycea waterlooensis*). Cold Springs is another major spring that discharges from the south bank of the Colorado River 2.5 km (1.5 mi) upstream from the mouth of Barton Creek (Figure 1).

The Barton Springs Segment is bounded by the incised Colorado River to the north, the outcrop limit of formations (Small et al. 1996) comprising the Edwards Aquifer to the west, the boundary of fresh water and non-potable Saline Water Zone to the east, and a southern groundwater divide that shifts between the Blanco River to Onion Creek (Figure 1). Hydrologic zones associated with the Barton Springs Segment are the recharge zone, confined or artesian zone, and contributing zone. Within the 243 km^2 (94 mi^2) recharge zone of the Barton Springs Segment, surface water can recharge the aquifer. The recharge zone is based on surface geological mapping (Hill and Vaughn 1898; Guyton and Associates 1964; Slagle et al. 1984; Small et al.1996; Hauwert 2009). Barton, Williamson, Slaughter, Bear, Little Bear, and Onion Creeks and the Blanco River cross the recharge zone and carry streamflow from the upstream contributing zones as well as storm runoff from the upland intervening area across the recharge zone (Guyton and Associates 1964; Slade et al. 1986; Johnson et al. 2012). Swallets are point features, generally sinkholes, within the stream channels where discrete flow loss occurs. The intervening area comprises upland portions of the recharge zone between major stream channels, and includes tributaries to the major stream channels. Runoff from about 10% of the recharge zone does not flow to surface streams under natural conditions but is contained within internal drainage sinkhole basins (Hauwert et al. 2005) that reflect the tendency of the recharge zone to infiltrate runoff (Williams 1972). The confined zone is a 129 km^2 (50 mi^2) area east of the recharge zone where the Edwards Aquifer is overlain by confining clays and limestone. Groundwater levels are below the top of the aquifer along much of the western edge of the confined zone, such that artesian aquifer conditions may not necessarily be present. The upstream contributing zone of Barton, Williamson, Slaughter, Bear, and Onion Creeks is about 673 km^2 (260 mi^2) and is underlain primarily by rocks of the Trinity Group (Mace et al. 2000) that may serve as sources of allogenic inflows to the Barton Springs Segment.

The mean flow of Barton Springs was calculated in Slade et al. (1986) to be 1.5 m^3/s (50 ft^3/s), based on 728 flow measurements taken over 62 years and three years of continuous measurement

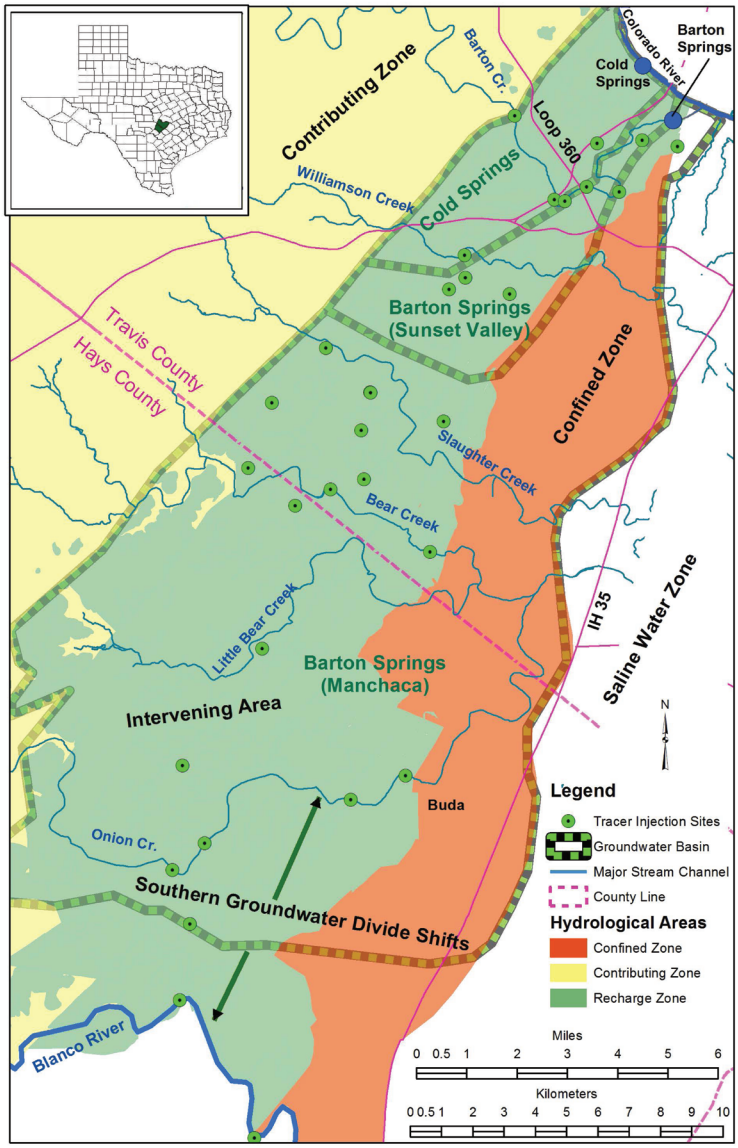

Figure 1. Map of the Barton Springs Segment.

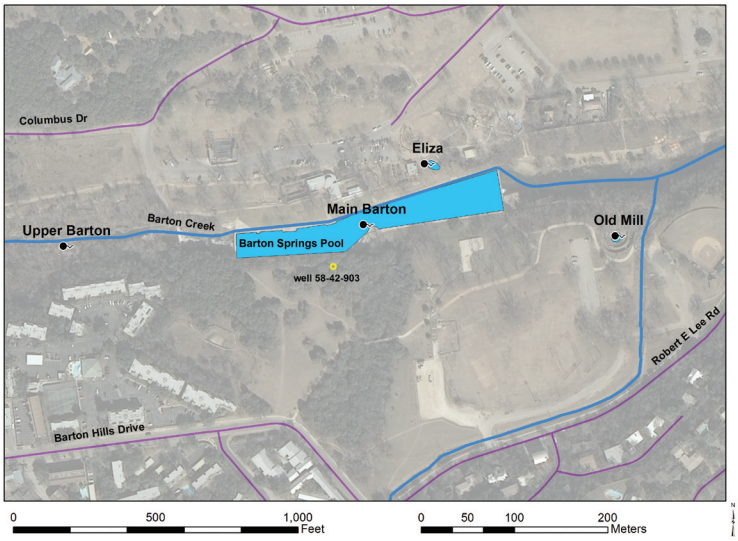

Figure 2. Map of Barton Springs.

based in part on stage discharge rating curve from a nearby well (state well 58-42-903, Figure 2). However, Barton Springs discharge, continuously measured over 36 years from 1978 to 2014 by USGS, show a higher discharge mean of 1.74 m^3/s (62 ft^3/s) and mean pumpage of 0.17 m^3/s (6 ft^3/s), for a mean total discharge of 1.9 m^3/s (68 ft^3/s) (Hunt et al. 2012; Johns 2015). Note that USGS excludes Upper Barton Springs discharge as well as other spring discharge to Barton Creek upstream of Barton Springs pool from its reported Barton Springs discharge. Record low discharges from Barton Springs of 0.27 m^3/s (9.6 ft^3/s) were measured on March 29, 1956 and 0.34 m^3/s (12.1 ft^3/s) on February 26, 1918 (Grover 1922; Twichell 1958). On May 10, 1941 a high discharge of 4.7 m^3/s (166 ft^3/s) was measured (Parker 1943). The long-term average rainfall for Austin is 85.1 cm (33.5 inches, Johns 2015).

Urbanization of the study area expanded during the 1980's, and the USGS, with the support of the University of Texas, the Texas Water Development Board, and the City of Austin, conducted a study of the Barton Springs Segment. This study established permanent stream-flow gauging stations on the upstream side of the recharge zone, continuously gauged Barton Springs discharge, mapped the groundwater-level surface, produced hydrogeological cross sections of the Barton Springs Segment, and evaluated Barton Springs water quality (Senger 1983; Senger and Kreitler 1984; Baker et al. 1986; Slade et al. 1986). Perhaps the most frequently cited aspect of the study was a channel water balance that first calculated the stream contributions to Barton Springs based on continuous and intermittent gauging stations on five of the six streams within the assumed source area for Barton Springs discharge (Slade et al. 1986). Groundwater basins had not yet been directly delineated using injected tracers. Slade et al. (1986) concluded that 85% of Barton Springs discharge was recharged from the major stream channels and only 15% was from upland autogenic recharge attributed to the upland intervening areas. A precipitation water balance calculated 85% evapotranspiration, 6% (allogenic + autogenic) recharge, and 9% runoff from the recharge zone (Woodruff 1984; Slade et al. 1986). Of the 6% recharge to the Edwards Aquifer from precipitation, 5% was calculated as stream (allogenic + autogenic) recharge and 1% as upland autogenic recharge (Woodruff 1984). The calculation utilized precipitation over the entire recharge and contributing zone.

The channel water balance calculation of recharge heavily shaped the growth of Austin over the recharge zone since the 1980's. These relatively low upland recharge and high evapotranspiration values have frequently been cited (URM 1983; Texas DOT 1989; Woodruff 1998; Woodruff 2010) as evidence that these recharge sources outside of the major stream channels are relatively insignificant sources of recharge to the Barton Springs Segment. Common land management practices during the 1980's and 1990's included diverting storm-water runoff into sinkholes; converting sinkholes into water-quality ponds and livestock ponds; building over caves and sinkholes; and irrigating treated wastewater over the Edwards Aquifer (Veni and Hauwert 2015). During this period, the Edwards (Balcones and Edwards Plateau Zones) Aquifer also experienced a number of pathogenic outbreaks and petroleum pipeline ruptures (Rose 1986; Hernandez 2015). In 1986, the Hays County Soil and Water District petitioned the Environmental Protection Agency to protect the Barton Springs Segment for an estimated 22,800 people reliant on water supply wells, and it was designated as a federally-designated sole source aquifer in 1988 (Federal Register 1988). In 1991, citizens of Austin petitioned and voted to enact the non-degradation Save-Our-Springs ordinance, providing Barton Springs with more protection from development. Maintaining the quality and quantity of water in the Barton Springs Segment is deemed important by the City of Austin and over 155 million dollars of voter-approved bonds have been utilized to preserve 22% of the recharge zone (Thuesen 2013). The watersheds of the recharge and contributing zone remain relatively rural, but urbanization continues to expand over the Barton Springs Segment. As recently as 2010, the low (less than 1%) autogenic recharge from precipitation estimate derived from the channel balance method was used to minimize the potential impacts of large-scale wastewater irrigation over the recharge zone (Woodruff 2010). The groundwater basins assumed in the 1980's channel water balance were also incorporated in

groundwater models (Slade et al. 1985; Barrett and Charbeneau 1996; Passarella et al. 2012) including Groundwater Availability Models used by the State of Texas to manage groundwater pumpage (Scanlon et al. 2001; Hutchinson and Hill 2011). Herrington et al. (2010) calculated the density of impervious structures exclusive of roads and some parking lots using 2008 county tax appraisal district records and measured values for Barton (2%), Williamson (11%), Slaughter (4%), Bear (1%), and Onion (0.2%) Creeks, or about 18% total impervious cover.

Beginning in the 1990's, site specific water balances utilized climate towers in Central Texas to measure evapotranspiration directly using eddy covariance or Bowen ratio systems combined with weirs and flumes to measure surface runoff. The autogenic recharge is calculated by subtracting from measured precipitation the measured actual evapotranspiration, measured runoff, and any change in soil moisture storage. A five-year water balance utilizing a Bowen ratio climate tower, rain gauges, and flumes measured that, out of total precipitation, 65% evapotranspirated, 5% flowed downstream into streams, and the residual 30% recharged the Trinity Aquifer of Uvalde County (Dugas et al. 1998). Huang and Wilcox (2005) observed that autogenic recharge was 18% less in precipitation water balance measurements based on channel water balance method when compared to measurements of the site specific climate tower water balance reported by Dugas et al. (1998). Wilcox (2008) also noted a 35% precipitation water balance discrepancy between evapotranspiration calculated by stream gauging in a channel water balance study (Maclay 1995) and direct climate tower measured evapotranspiration (Dugas et al. 1998) within the same Seco Creek watershed. In their compilation of published annualized Central Texas climate tower and runoff data, including data from one station within the Barton Springs Segment, Hauwert and Sharp (2014) note that of average annual precipitation over Central Texas Edwards and Trinity Aquifers, 69% evapotranspirated, 28% recharged, and 3% ran off upland areas. These data also show that recharge and evapotranspiration vary directly with annual precipitation, but the difference in values between sites across Central Texas karst aquifers was not so great as to account for the variation in results derived from channel water balance and climate tower methods. Wilcox (2008) noted that the measurement of precipitation water balance components using site scaled climate towers and flumes is more precise, and questioned why the channel water balance method underestimated recharge and overestimated evapotranspiration.

A long-distance groundwater tracing study by the Barton Springs/Edwards Aquifer Conservation District (BS/EACD) and City of Austin commenced in 1996. The results from 40 groundwater traces with 88% successful recovery rate delineated groundwater basins and flowpaths within the Barton Springs Segment (Hauwert et al. 2004a; Hauwert 2009; Hauwert 2015; Figure 1). Study conclusions are:

1. Rapid groundwater flow rates measured directly by tracing, with mean residence times of one to eight days, contradict some previous estimates of residence times of three to five years (Texas DOT 1989; Alexander 1990) that have been used to plan responses to accidental spills.
2. The entire portion of Barton Creek upstream of the streamflow gauge at Loop 360 (Figure 1), estimated by the channel water balance method to contribute 28% of the recharge to Barton Springs (Slade et al. 1986), was traced, under varying aquifer conditions, to Cold Springs rather than Barton Springs. An attempted groundwater trace by the USGS in 1981 on Barton Creek upstream of Loop 360 also was not detected at Barton Springs (Slade 2008).
3. The upstream 2 km (1.3 mi) of Williamson Creek over the recharge zone lies in the Cold Springs groundwater basin (Hauwert et al. 2004a).
4. The recharge zone of the Barton Springs Segment contains groundwater basins contributing to Barton and Cold Springs that are 212 km^2 and 31 km^2 (82 mi^2 and 12 mi^2), respectively (Hauwert 2009, 94 and 149), although portions of these areas contain the outcrop exposure of some overlying confining units. Excluding isolated internal portions of the recharge zone covered with clays, shales, and other overlying confining units, the

Edwards Aquifer outcrop area recharging to Barton Springs is 176 km^2 (68 mi^2).

5. The tracers flowed as pulses under conduit flow, 1,000 to 60,000 times more than the disperse flow component under non-drought conditions (Hauwert 2015). Based on the tracer breakthroughs, attenuation of tracer transport through the aquifer would be negligible from injections as far as 24 km (15 mi) away.
6. A number of injections from the Blanco River have been traced to Barton Springs in 2008 and 2009, during a period when Barton Springs discharge had declined from 1.08 to 0.54 m^3/s (38 to 19 ft^3/s) (Hauwert 2011; Johnson et al. 2012). The tracers injected in 2008 and 2009 also were recovered in wells to the south and may have also discharged from San Marcos Springs. At an unknown higher flow condition, the Blanco River ceases supplying water to Barton Springs and a portion of Onion Creek recharge diverts to San Marcos Springs (Hunt et al. 2004; Hauwert 2009).

In a channel water balance of the entire Barton Springs Segment, when comparing stream flow loss to aquifer discharge, if Cold Springs discharge is underestimated then the residual upland autogenic recharge would also be underestimated by the same amount (Hauwert 2009). For instance, Slade (1984, 24) stated that Cold Springs "*consistently flows between 3 to 4 ft^3/s*" (0.085 to 0.113 m^3/s) based on Brune (1975), and that its groundwater basin was 10 km^2 (4 mi^2). Slade (2014, 2015) increased the estimate for a long term mean discharge of Cold Springs in 2014 to 0.156 m^3/s (5.5 ft^3/s) and in 2015 to 0.238 m^3/s (8.4 ft^3/s). However, direct measurement of the submerged discharge of Cold Springs into the Colorado River was not possible even prior to the 1960 construction of Town Lake (renamed to Lady Bird Lake in 2007). The USGS (Hill 1892, 94) evaluated Cold Springs prior to the construction of a downstream dam and found "*this group of springs flows a great volume, but inasmuch as they break out in the river's edge, it is impossible to gauge them. The aggregate volume of these springs near Austin is so great that the volume of the river is materially increased by their accession*". Brune (1975) noted that Cold Springs consisted of at least seven spring orifices, although only two orifices are usually exposed. There are no well-documented direct measurements that include discharge from all seven of Cold Springs's known orifices. During lake drawdown conditions on January 29, 2008, I found that four of the (at least) seven Cold Springs orifices were exposed and the combined discharge from these measured to be 0.23 m^3/s (8.2 ft^3/s), and this partial measurement of Cold Springs was 12% of the 1.87 m^3/s (66 ft^3/s) that Barton Springs measured the same day. Other spring outlets from Cold Springs can be detected under some conditions as sites on the mud floor of the river where cool discharging water can be sensed. Tracing results suggest that submerged Cold Springs orifices discharged from recharge sites at least 5 km (3 mi) away, and that, similar to Barton Springs, discharge from the various Cold Springs orifices are derived from different sources (Hauwert et al. 2004a; 2004b). Tracing also showed the Cold Springs groundwater basin is about 31 km^2 (12 mi^2), significantly larger than the previous estimates.

While Cold Springs discharge cannot be measured directly, it has been measured indirectly. Sellards (1934, 366) observed: "*Many of the large springs of Texas are unknown because they are beneath river beds and therefore hidden from view. During dry periods most of the water in surface streams is derived from springs and seepages...The presence of these hidden springs can be determined by gauging stream flow at stations closely spaced in distance*."

Prior to the upstream dam construction in 1940, creating Lake Travis and Lake Buchanan, and downstream dam completion in 1960, the flow of the Colorado River ranged from 0.215 to 6.68 m^3/s (7.6 to 236 ft^3/s) during selected surveys. This would be sufficiently low to measure the discharge indirectly to infer discharge of Cold Springs and other spring discharge between measurement sites. Four discharge surveys of the Colorado River, between Tom Miller Dam and one quarter mile downstream of Deep Eddy, taken on August 1, 1918; August 6, 1918; August 8, 1921; and August 13, 1930, measured increases in discharge of 0.21, 0.12, 0.30, and 0.34 m^3/s (7.5, 4.2, 10.7, and 12.0 ft^3/s) (Grover 1922; Grover and Ellsworth 1923; TBWE 1959). The river gains attributable primarily to Cold Springs discharge were 27% to 54% of Barton Springs discharge measured the same day. A later water balance of Lady Bird

Lake found average gains of 0.82 m^3/s (29 ft^3/s) or roughly half of average Barton Springs discharge in the portion of the lake where Cold Springs is the only known significant spring discharge (Stecher et al. 1992). Using measured flow loss from Barton and Williamson Creeks within the Cold Springs groundwater basin, Hauwert (2009) calculated that Cold Springs has a minimum average discharge greater than 0.42 m^3/s (15 ft^3/s). The combined historical indirect discharge measurements of Cold Springs and other discharge from the Barton Springs Segment along the south bank of the Colorado River suggest an average discharge within the general range of about 0.4 to 1 m^3/s (15 to 35 ft^3/s).

The southern divide for the Barton Springs Segment has also been reevaluated by direct tracing. Prior to direct tracing studies in 2008 and 2009 (Johnson et al. 2012), most studies based on groundwater-level mapping (Petitt and George 1956; Garza 1962; Stein 1995) interpreted that Onion Creek was the southern divide for the Barton Springs Segment under all conditions. The Blanco River was not considered a source to Barton Springs in earlier water budgets. The 2008 and 2009 direct tracing results, short term correlations in flow response, and a shift in Barton Springs recession curve during droughts mirroring Blanco River channel flow loss suggests that the Blanco River may sustain Barton Springs discharge during drought conditions (Hauwert 2011). Because the Blanco recharge is hypothesized to travel to Barton Springs along two groundwater flow paths, one largely confined by overlying clays under artesian conditions and the other largely unconfined, there are both short-term pressure pulses and approximately three month delayed recharge responses (Figure 3). Casteel et al. (2013) included only the portion of recharge from the Blanco River that corresponds with short-term, instantaneous increases in discharge at Barton Springs and quantified a pressure pulse recharge addition of about 0.074 m^3/s (2.6 ft^3/s).

Other potential water sources contributing to the aquifer include: 1) leakage inflows including cross formational groundwater flow from the Trinity Aquifer (Senger 1983); 2) flow from the Saline Water Zone (Senger 1983; Hauwert et al. 2004b); 3) groundwater flow from the San Antonio Segment of the Edwards (Balcones Fault Zone) Aquifer (Hill 1892; Taylor and Schoch 1922; Johnson and Schindel 2008; Land 2010); and 4) municipal water and wastewater that infiltrates and recharges the aquifer (Garcia-Fresca and Sharp 2005; Sharp 2010; Passarella et al. 2012). City of Austin municipal water is derived from the Colorado River and can be considered a type of artificial allogenic recharge. It is not included here in calculations of allogenic recharge, but Passarella et al. (2012) estimated total anthropogenic urban leakage in the Barton Springs Segment to average about 0.098 m^3/s (3.5 ft^3/s). The Saline Water Zone contribution to Barton Springs is calculated to be as much as 3% of Old Mill Springs discharge during drought conditions, but on average contributes a negligible 0.2% to Main and Eliza Springs discharge (Hauwert et al. 2004b).

This study compares measured upstream and downstream streamflow loss to measured total aquifer discharge and average precipitation in order to calculate upland autogenic recharge, stream autogenic recharge, allogenic recharge, and evapotranspiration. It reexamines whether a streamflow loss study can derive recharge values consistent with other recharge measurements once the recharge sources are directly delineated using injected tracers. A water budget for a karst aquifer includes the components of precipitation, discharge through spring discharge and pumpage, downstream runoff, allogenic recharge from upstream contributing areas, autogenic recharge that directly enters the intervening upland outcrop of the aquifer, and autogenic recharge from direct precipitation over the recharge zone that runs off, infiltrates, percolates, and recharges within the major stream channels.

Methods

Streamflow loss sources to Barton Springs were measured using the channel water balance method. Groundwater tracing defined the groundwater basins (Hauwert 2015; Figure 1). Manual low-flow surveys taken from 1998 to 2008 across the recharge zone, combined with similar flow surveys by the USGS from 1980 to 1985, distinguished gaining from losing stream stretches and identified swallets where infiltration is focused. For the first time, beginning in 2003, continuous downstream

and upstream gauging stations were operating on Onion, Bear, Slaughter, Williamson, and Barton Creeks. A 4.3-year time period from May 31, 2003 to September 19, 2007 was selected as the water budget interval containing the best gauging station data from the streams and where starting and ending discharges at Barton Springs were equal. Thus, effects from change in groundwater storage were eliminated.

Total groundwater discharge from the groundwater basins discharging at Barton Springs was calculated as the sum of Barton Springs discharge reported by the U.S. Geological Survey (USGS), lower Barton Creek spring discharge, and pumpage calculated by the BS/EACD. Well pumpage was metered for permitted wells and estimated for unregulated wells (Hunt et al. 2006; Banda et al. 2010). The USGS measures the combined discharge of Main, Eliza, and Old Mill springs, but excludes lower Barton Creek spring discharge, including Upper Barton Springs (Figure 2). On many occasions, where Loop 360 flow gauge reported less than 0.4 m^3/s (15 ft^3/s), lower Barton Creek springflow can be quantified as all stream discharge measured upstream of Barton Springs pool. At the stream gain between Barton Springs pool and a site one mile upstream, discharge was measured manually 19 times between January 1998 and October 2003 by City of Austin (CoA) and BS/EACD staff. These data established a correlation between measured Lower Barton Creek spring discharge and USGS-reported Barton Springs discharge for calculating mean daily lower Barton Creek spring discharge (Table 1).

Because discharge measurement of Cold Springs cannot be directly measured, its discharge was excluded from the total groundwater discharge calculation for this study. Barton Creek downstream of Loop 360 (see Figure 1) generally represents a groundwater divide between Barton Springs and Cold Springs, and the distribution of recharge downstream of Loop 360 to each spring is not yet measured (Hauwert 2009). Consequently, the entire possible range of recharge in Barton Creek channel downstream of Loop 360 was included.

Streamflow loss was calculated from upstream minus downstream daily mean flow at gauging stations in daily time steps to distinguish allogenic and autogenic runoff sources. Gauging stations were located upstream and downstream of the groundwater basins, based on groundwater tracing and low-flow stream surveys (Figure 4). The sensitivity of mean daily intervals was tested on April 2, 2003 by the opening of a single plugged swallet, Crippled Crawfish Cave, resulting in an abrupt increase in potential flow loss from 1.0 m^3/s (35 ft^3/s) up to 1.8 m^3/s (63 ft^3/s) (Figure 5).

Net stream channel recharge was distinguished according to its source, as either intervening area or contributing zone for the major stream channels, except for the Blanco River. Typically, when intervening runoff to the streams is not occurring and there is no autogenic recharge, the allogenic recharge originating from the contributing zone is directly calculated as upstream flow minus downstream flow, and runoff originating from the contributing zone is simply any flow passing the downstream flow station. Downstream runoff is mean daily flow measured at the downstream gauging station.

For relatively short time intervals associated with some precipitation events, the downstream gauging station flow could be higher than the upstream gauging station flow. During such events, short-lived runoff from the intervening area between stream gauges reached the major creek channels and the distinction of recharge and contributing zone major stream channel recharge and runoff

Table 1. Correlation of Lower Barton Creek to Barton Springs Discharge.

Barton Springs Discharge (m^3/s)	Barton Springs Discharge (ft^3/s)	Lower Barton Creek Springflow (m^3/s)
0 - 1.1	0 - 40	0
1.1 - 1.7	40 - 60	0.015*BS
1.7 - 2.8	60 -100	$0.003*(BS)^2 - 0.3336*BS + 0.28$
>2.8	>100	not modeled, but estimated by other means

BS = USGS Reported Barton Springs Discharge.

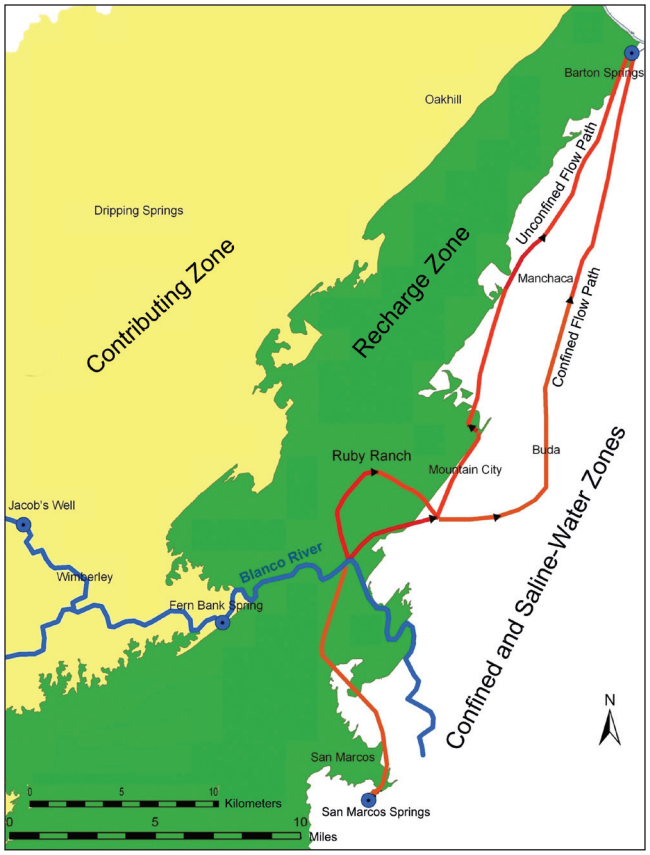

Figure 3. Map of 2008 to 2009 Blanco River Injections and Interpreted Groundwater Flow Paths.

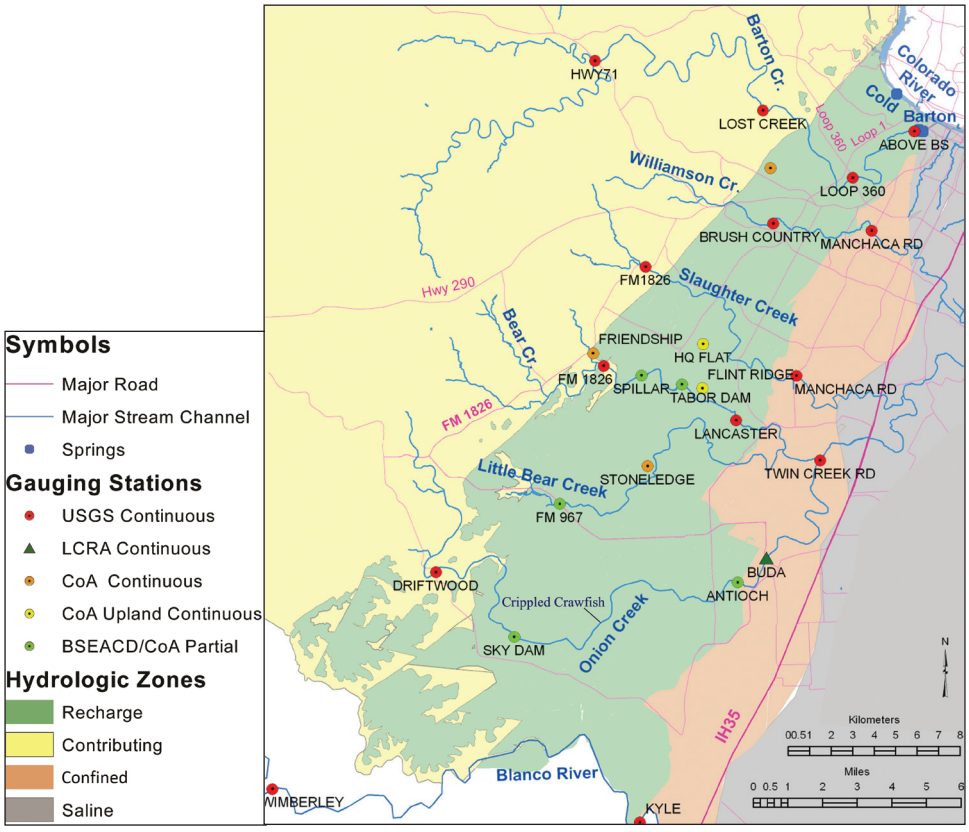

Figure 4. Map of Flow Gauging Stations.

were balanced on daily time steps because:

1. Some intervening area runoff potentially infiltrates in the major stream channels and is not directly measureable from comparing upstream and downstream gauging stations.
2. The amount of contributing zone and intervening area runoff mixtures that runs off beyond the recharge zone cannot be precisely distinguished, although the combination is measured.

Around precipitation events, a time series of potential streamflow loss was plotted versus elapsed time (Figure 6). The corrected flow loss was calculated by extrapolating from declining potential flow loss values back to the start of increasing flow in the stream, using best fit (generally exponential) recession curves. In a few cases, early potential flow loss values did not appear to be greatly affected by intervening runoff and served as data points for extrapolating recession curves. The recession curves were also constrained by the maximum flow loss rate calculated for each stream (Table 2). By comparing upstream minus downstream flows over the period of record, (as applied by Barrett and Charbeneau 1996) the maximum expected flow loss capacity of each stream was estimated and used as a constraint. The amount of intervening area runoff that reached the downstream end of the major stream channels was generally measured as the difference between calculated stream flow loss and potential streamflow loss (Figure 6).

Recession curves were extrapolated for each new runoff event and varied within the same stream due to factors including antecedent rain period, soil moisture, and ground water-levels. For some runoff events on some streams, such as Bear Creek, the recession curve of potential flow loss can be nearly flat over time when the water table is shallow and recharge is temporarily rejected. The effect of correcting potential major streamflow loss increased the net calculated streamflow loss by 27% for Onion Creek, 8% for Bear Creek, 50% for Slaughter Creek, and 67% for Williamson Creek.

Measurements from upland gauging stations (Figure 4) in representative areas at Headquarters Flat (0.19 km^2 or 0.072 mi^2) and Flint Ridge (0.28 km^2 or 0.108 mi^2) were used to calculate the runoff from intervening areas that reach the major creek channels. Both basins are rural former ranchland with oak and juniper-dominated forests mixed with intervening savannahs. The site slopes are generally less than one degree. The runoff is completely contained within internal drainage basins and runs off to cave drains instead of discharge to major streams. Additional details on the upland gauging sites, including topography, soil depths, soil moisture, and vegetation lists are provided by Hauwert (2009). Internal runoff from the upland stations was multiplied by the difference in intervening catchment area for the lower stream gauging station to calculate anticipated runoff. Stream autogenic recharge is calculated around precipitation events on daily time steps to balance runoff to the major stream channels from both the contributing zone and tributaries of the recharge zone minus runoff downstream of the recharge zone.

Eight drought intervals were selected to examine the potential contribution of the Blanco River to Barton Springs during drought conditions. For the selected drought intervals, the daily corrected flow loss of the Blanco River and daily average Barton Springs discharge was summed and compared. By taking entire drought intervals, the effects of both short term pressure pulses with arrival within several days and long-term flow of Blanco River recharge to Barton Springs with roughly three-month delay times can both be quantified.

Corrected and summed streamflow loss was compared to total groundwater discharge in order to calculate the percentage of recharge attributed to each major stream and the Blanco River. The remainder of the total groundwater discharge was assumed to consist of upland autogenic recharge or direct infiltration from the intervening area, with the addition of Trinity Aquifer leakage, Saline Water Zone leakage, urban leakage, and potentially other sources such as groundwater leakage from adjacent segments of the Edwards (Balcones Fault Zone) Aquifer (Figure 7).

The assumptions were that over the water-budget interval:

1. Gauged flow loss in the stream channels completely infiltrated into the subsurface and recharged the aquifer.
2. Evapotranspiration of stream flow in major creek channels was negligible. If the evapotranspiration of stream flow in the major

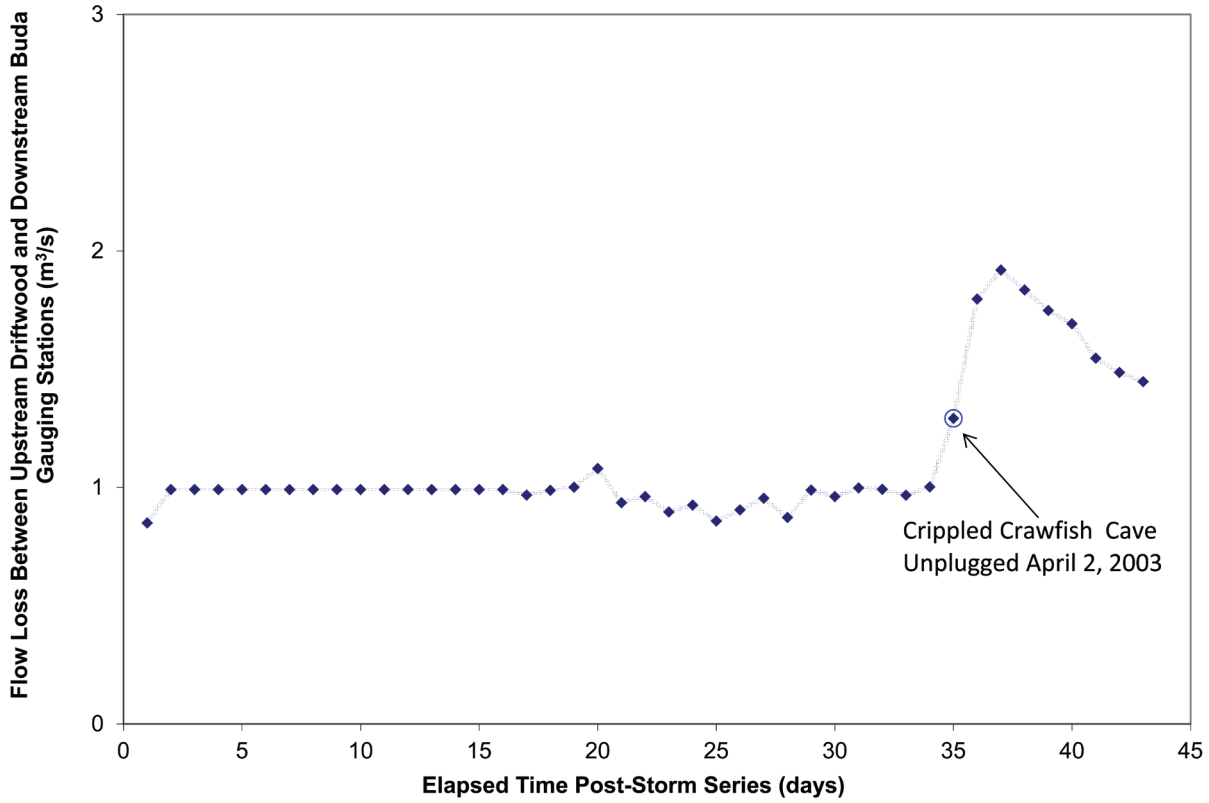

Figure 5. Flow Loss Response of Onion Creek Flow Gauging Stations to Unplugging of Crippled Crawfish Cave.

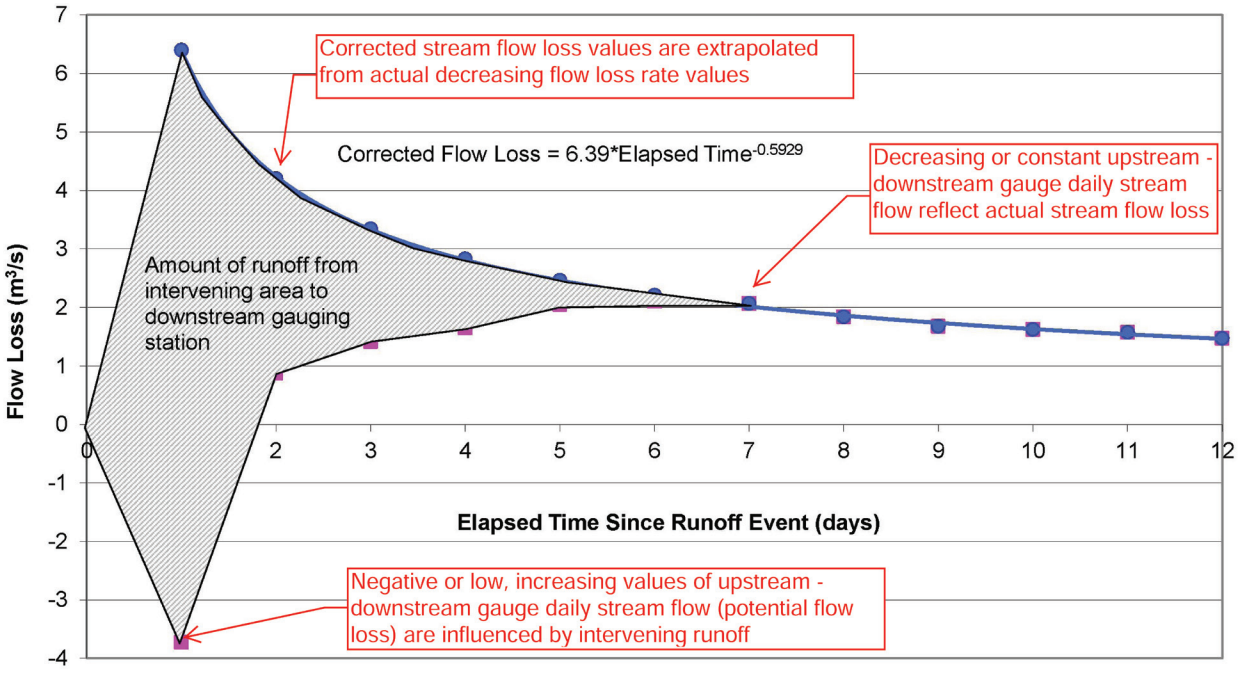

Figure 6. Example Correction of Potential Flow Loss.

Table 2. Maximum Flow Loss for Major Stream Channels.

Source	Maximum Channel Flow Loss Rate							
	Slade et al. 1986		Barrett et al. 1996		This Study			
					Flood		Low Flow	
	(m^3/s)	(ft^3/s)	(m^3/s)	(ft^3/s)	(m^3/s)	(ft^3/s)	(m^3/s)	(ft^3/s)
Barton Upst 360	0.85-2.0	30-70	7.10	251	1.41	50	0.85	30
Barton Dnst 360	---	---	---	---	1.10	40	0.57	20
Williamson	0.37	13	1.00	14	0.57	20	0.37	13
Slaughter	1.47	52	1.50	53	2.40	85	0.79-1.56	28-55
Bear	0.93	33	0.91	32	0.99	35	0.79-0.93	28-33
Little Bear	0.85	30	0.91	32	0.99	35	0.71	25
Onion	3.40	120	3.40	120	7.10	250	0.57-2.80	20-100
Blanco River	---	---	----	---	---	---	0.28-0.51	10-18

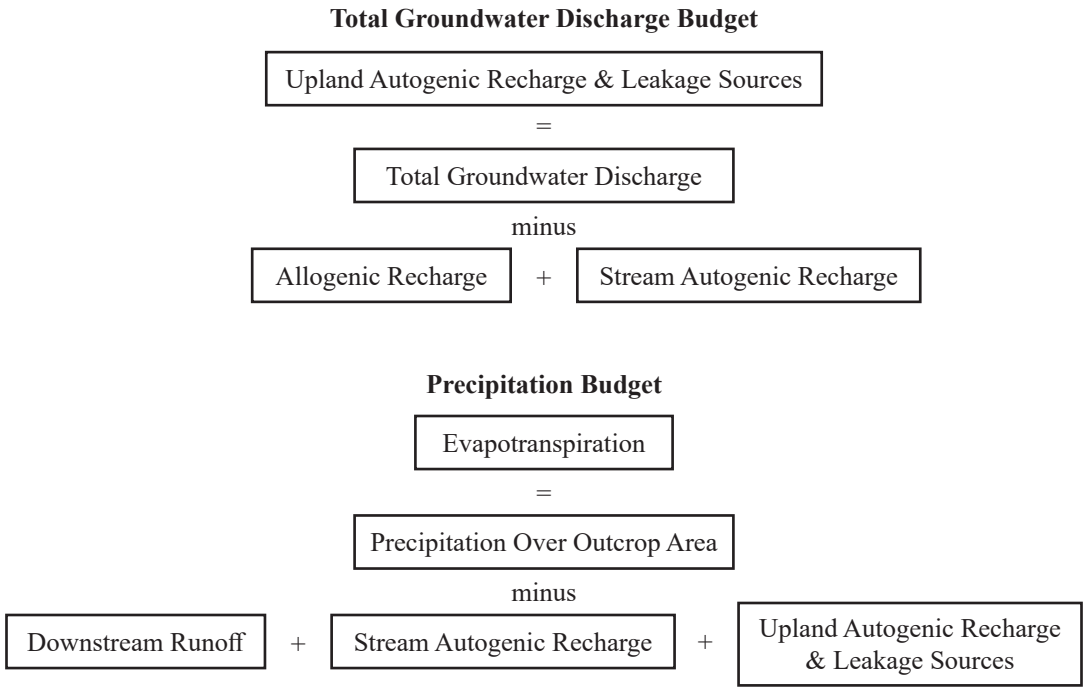

Figure 7. Channel Water Balance Calculations for Upland Autogenic Recharge and Evapotranspiration.

channels was significant, then the portion of major stream channel recharge could potentially be less than calculated in this channel water balance.

3. No stream-channel infiltration in Barton Creek upstream of Loop 360 contributes to Barton Springs. For the three mile stretch of Barton Creek downstream of Loop 360 (Figure 1), a range of potential recharge was calculated with both all and none of recharge in the Barton Creek channel contributing to Barton Springs. This accounts for range of values calculated for recharge components as a percent of total discharge.
4. On Bear Creek, the Spillar Ranch low water crossing (Spillar, Figure 4) was more representative of upstream recharge zone than the FM1826 USGS gauging station site, because of considerable spring discharge between the two sites. We took periodic flow measurements at Spillar, correlated flow gains between the two sites and adjusted FM1826

USGS gauging data to reflect the discharge expected at Spillar. Without this correction the stream flow loss occurring in Bear Creek would be underestimated.

5. All significant recharge to the groundwater system from the channel of Little Bear Creek occurs within the upstream half of the recharge zone between FM967 and the Stoneledge flow gauging stations. A flow loss survey of Little Bear Creek on July 4, 2002 by BS/EACD measured less than 10% total flow loss occurred downstream of the lower gauging station used in this study.
6. The recharge to the groundwater system from Onion Creek is assumed to discharge at Barton Springs. Note that on one occasion under high-flow conditions, tracers injected in Onion Creek were detected at both Barton Springs and at San Marcos Springs. The distribution of recharge from Onion Creek to Barton, San Marcos Springs, and other unknown destinations has not been measured.
7. The flow loss from the Blanco River between the Wimberley and Kyle gauging stations was assumed to discharge completely at Barton Springs for intervals when Barton Springs discharge is less than 1.13 m^3/s (40 ft^3/s). This assumption can underestimate recharge from the Blanco River because of additional spring discharge downstream of Wimberley, such as Fern Bank Springs (Figure 3). As discussed below the direct comparison of Blanco River flow loss to Barton Springs discharge recessions tends to underestimate Blanco River contribution, but this contribution of Blanco River could be overestimated because of diversions of some Blanco River channel recharge to San Marcos Springs or water wells.

Major stream channel flow loss was also compared to precipitation over the 176 km^2 (68 mi^2) outcrop source area to Barton Springs, which excludes 31 km^2 (12 mi^2) of Cold Springs groundwater basin and 36 km^2 (14 mi^2) of overlying confining units within the recharge zone of the groundwater basins discharging to Barton Springs. Monthly average rainfall from May 31, 2003 to September 19, 2007 was summed from four Flood Early Warning System (FEWS) sites distributed across the northern half of the recharge zone. The net precipitation from the four sites, varying from 348 to 371 cm (137 to 146 in) over the 52 months, had an arithmetic mean of 358 cm (141 in). The annual averaged rainfall for the water balance interval of 83 cm (32.5 in) is 97% of the long term average rainfall for the Austin area. After subtracting upland autogenic recharge and runoff downstream of the recharge zone over the 4.3 year water budget interval from rainfall, the remainder of rainfall is assumed to be an approximate measure of evapotranspiration over the recharge zone (Figure 7).

Results

The results of the water budget are presented in Tables 1 and 2. From May 31, 2003 to September 19, 2007 (1,573 days), the total groundwater discharge consisted of 86% USGS-reported Barton Springs discharge of Main, Eliza, and Old Mill Springs, 11% well pumpage, and 3% lower Barton Creek spring discharge upstream of Barton Springs pool (Table 3). Over the 4.3-year water budget interval, total discharge averaged 2.15 m^3/s (76 ft^3/s), which is representative of average discharge. As stated above, discharge from Cold Springs and recharge in its groundwater basin are not included in this analysis. The stream channel recharge sources of total discharge were calculated as 33% from Onion Creek, less than 10.5% from Barton Creek downstream of Loop 360, 7.2% from Slaughter Creek, 6.2% from Bear Creek, 6% from the Blanco River, 3.4% from Little Bear Creek, and 1% from Williamson Creek. Major stream channel recharge (including the Blanco River) accounted for 56% to 67% of total groundwater discharge. The range of values encompasses the potential Barton Creek channel recharge distribution to Barton Springs. The residual 33% to 44% of total groundwater discharge largely originated from upland autogenic recharge, along with possible contributions from other sources including subsurface Trinity Aquifer leakage, Saline Water Zone leakage, urban leakage, and leakage from the San Antonio Segment of the Edwards Aquifer. Intervening area runoff was a higher portion of total streamflow and major stream channel recharge in watersheds having relatively low contributing zone drainage

areas or high relative impervious cover, such as Williamson Creek. The percentages of flow loss of each stream from 2003 to 2007 were within 3% of channel flow-loss estimates for a 1978 to 1982 water budget (Slade et al. 1986) except for Barton Creek and the Blanco River (Table 4).

The portion of the allogenic stream channel recharge originating as runoff from the contributing zone and measured directly from upstream gauging stations is 39% to 50% of total groundwater discharge. A large portion of total groundwater discharge of Barton Springs originates from the recharge zone, including 33% to 44% largely composed of upland autogenic recharge and an additional 17% of total groundwater discharge which runs off the intervening area and becomes autogenic stream recharge within the major stream channels (Figure 8). Consequently, 50% to 61% of Barton Springs discharge is derived as total autogenic recharge from the recharge zone.

The recharge sources measured in this study were compared to precipitation over the water balance interval. Using a 176 km^2 (68 mi^2) outcrop source area to Barton Springs, the rainfall volume over the water budget interval is 633,437,330 m^3 (22,369,628,160 ft^3). The total groundwater discharge between May 31, 2003 and September 19, 2007 was 292,781,028 m^3 (10,339,464,433 ft^3), which is 46% of precipitation over the 176 km^2 (68 mi^2) outcrop source area to Barton Springs. Measured streamflow loss to Barton Springs was 26% to 31% of precipitation over the outcrop source area. The largely autogenic upland recharge over the intervening outcrop area of the recharge zone, constitutes 15% to 20% (18% mean) of precipitation (Figure 9) and 8% of precipitation was estimated to originate as stormwater runoff from the intervening recharge zone that recharged within the major stream channels. The residual of the precipitation water

Table 3. Total Groundwater Discharge.

Start to End Date	Interval (days)	Barton Springs Discharge (Reported by USGS) (m^3)	(%)	Lower Barton Creek Spring Discharge (m^3)	(%)	Total Pumpage (Reported by BS/EACD) (m^3)	(%)	Total Groundwater Discharge (m^3)
5/31/2003 - 9/19/2007	1,573	252,445,005	86	8,159,329	3	32,176,694	11	292,781,028

Table 4. Major Stream Channel Flow Loss Contributions to Total Groundwater Discharge.

Barton Springs Source	----------Recharge----------					Downstream Runoff	
Major Stream Channel	Channel Flow Loss (m^3)	Channel Flow Loss/ Total Groundwater Discharge		Channel Recharge Source		Downstream Runoff Source	
		2003-2007 This Study (%)	1978-1982 Slade 2014 (%)	Contributing Zone (m^3)	Intervening Area (m^3)	Contributing Zone (m^3)	Intervening Area (m^3)
Onion	95,547,515	32.6	34.6	69,279,477	26,298,841	144,567,178	62,540,930
Little Bear	10,057,872	3.4	6.0	0	10,057,872	0	10,057,872
Bear Creek	18,050,834	6.2	6.0	11,978,434	6,084,633	15,203,020	8,254,746
Slaughter	20,948,219	7.2	4.2	13,446,137	7,502,081	3,992,588	7,588,962
Williamson	2,813,569	1.0	2.5	2,679,376	391,452	3,545,088	11,361,897
Barton	<30,877,204	<10.5	25.3	<30,876,715	489	145,085,483	13,961,908
Blanco River (only when Barton Springs discharge is <1.3 m^3/s)	17,566,412	6.0	0	---	---	---	---

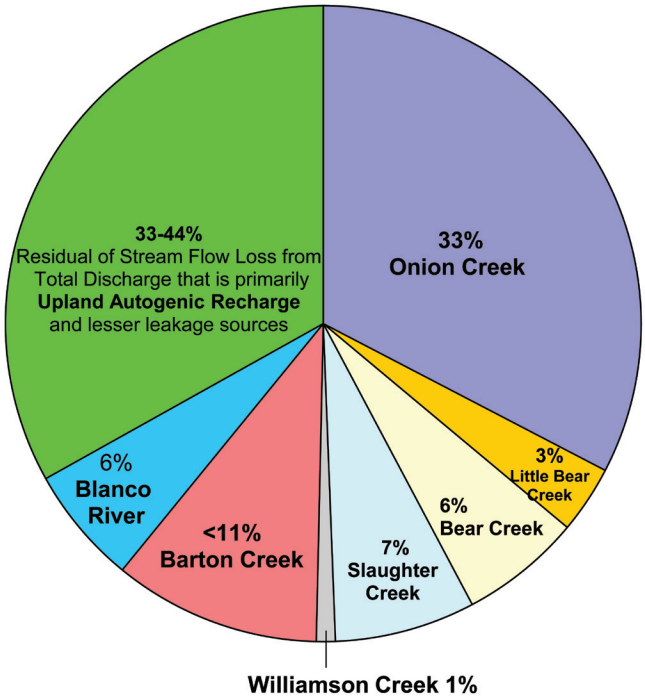

Figure 8. Source Components of Total Groundwater Discharge.

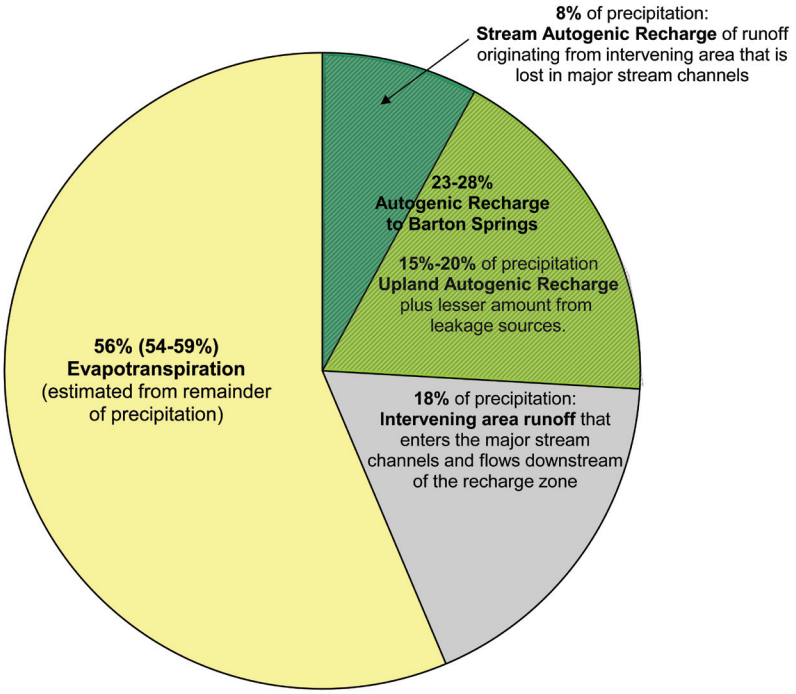

Figure 9. Components of Precipitation over 176 km^2 Outcrop Area.

balance across the 176 km^2 (68 mi^2) outcrop source area to Barton Springs is calculated to be 54% to 59%, averaging 56% of precipitation, that is attributed to evapotranspiration.

For the eight drought comparison intervals between Barton Springs discharge and Blanco River flow loss, Barton Springs discharge recessions experienced abrupt inflections at discharge rates varying from 0.54 to 0.96 m^3/s (19 to 34 ft^3/s), as seen in Table 5 and Figure 10. For the three months prior to each of the eight recession inflection points, Barton Springs discharge generally showed similarity with the Blanco River flow loss. Blanco River flow loss for the three months prior to the

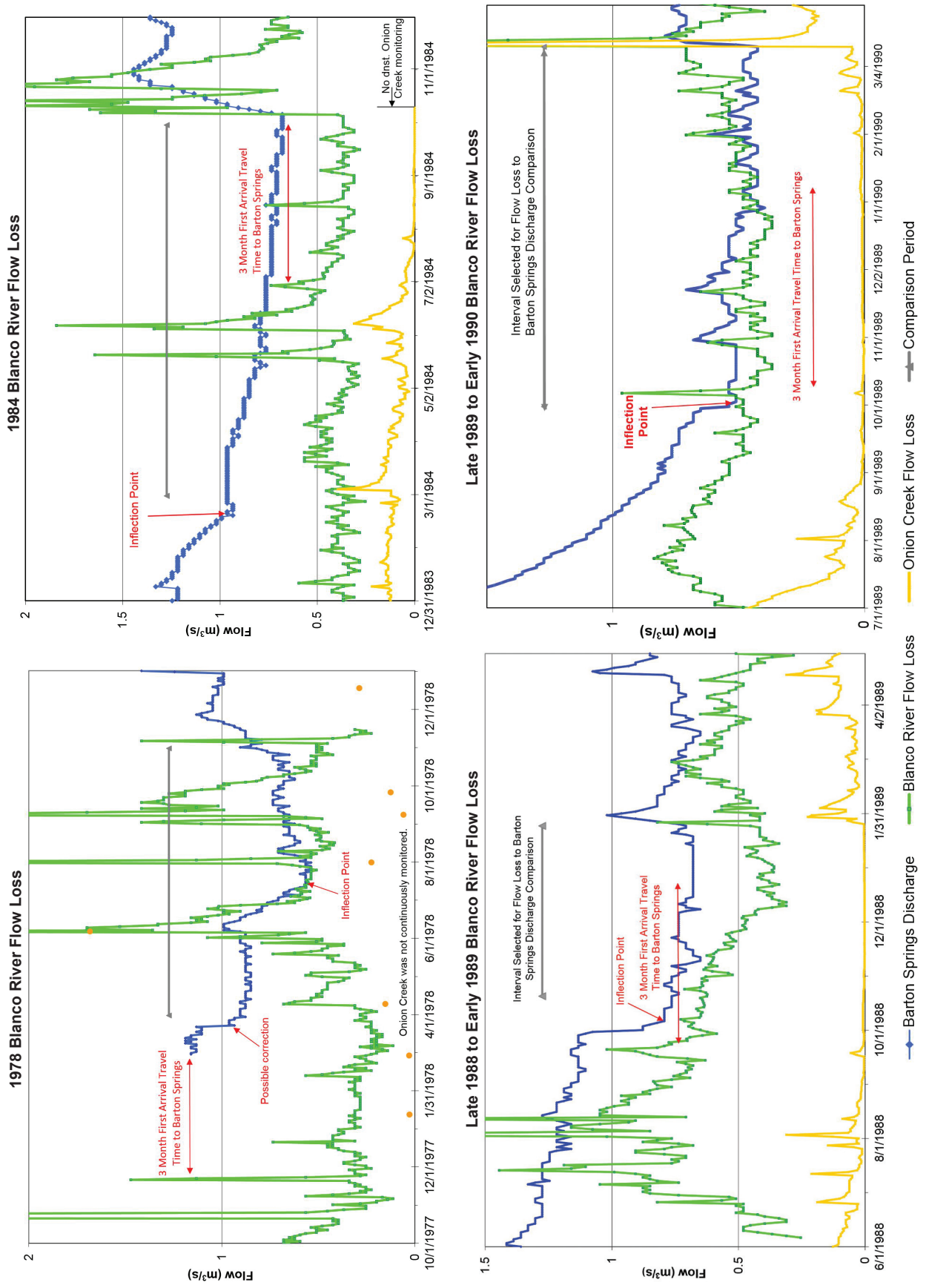

Figure 10. Blanco River Flow Loss and Barton Springs Discharge Recessions.

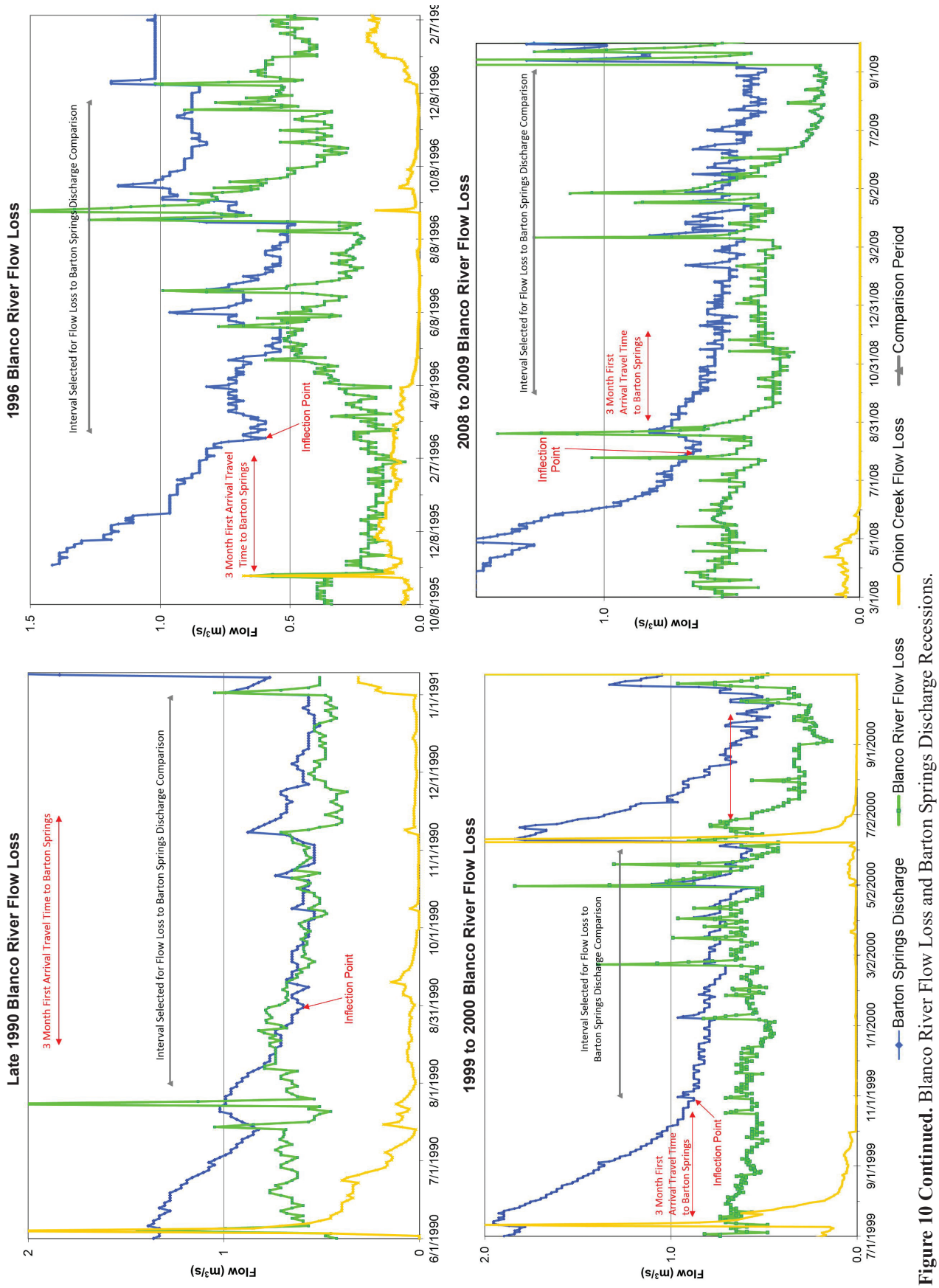

Figure 10 Continued. Blanco River Flow Loss and Barton Springs Discharge Recessions.

characteristic inflection in Barton Springs discharge varied from 27% to 110% of Barton Springs discharge, with an arithmetic mean of 76%. This, together with multiple successful groundwater traces from the Blanco River to Barton Springs, supports the hypothesis that the Blanco River sustains Barton Springs discharge during drought conditions. During the 2008 and 2009 drought, the groundwater tracers required about three months for first arrival. During the eight selected drought intervals, the total recharge on the Blanco River ranged from 49% to 84% of total groundwater discharge from the groundwater basins supplying Barton Springs, with a time-weighted arithmetic mean and median for the eight intervals of 62% with a standard deviation of 0.15. During these eight drought intervals, the flow loss measured from Onion Creek, by far the largest stream source to Barton Springs under average flow conditions, was relatively small, between zero and 7% of total groundwater discharge. Based on this comparison, I estimate that the Blanco River supplies at least half or more of Barton Springs discharge during droughts.

Discussion

The stream channel losses measured in this study were within 3% of 1980's total groundwater discharge estimates for Williamson, Slaughter, Bear, Little Bear, and Onion Creeks (Table 4; Slade et al. 1986; Slade 2014). This suggests that despite different approaches, relatively consistent values can be derived using the channel water balance method on data sets collected decades apart. However, on Barton Creek and the Blanco River, groundwater tracing defined a different fate for recharged water. The measurement differences between this study and the 1980's channel water balances in stream flow recharge compared to total groundwater discharge were greater than about 15% to 25% for Barton Creek and 6% for Blanco River.

Autogenic recharge on the recharge zone is estimated as 23% to 28% of precipitation in this study, with 15% to 20% occurring directly over the intervening area and 8% originating from intervening area runoff that recharges within the major stream channels (Figure 9). For comparison, 26% of rainfall was measured to directly infiltrate through soil-covered areas of an upland research site in the center of the Barton Springs Segment, using a climate tower and flumes (Table 6; Hauwert and Sharp 2014). This 1.4 year water balance period was selected as an interval where starting and ending volumetric soil moisture was 20% to eliminate the variable of changes in soil moisture storage. Although this interval had 34% higher than a mean annual rainfall of 85 cm, a compilation of annualized data from other climate towers over the Edwards and Trinity Aquifers showed that 28% autogenic recharge from precipitation was representative of mean precipitation conditions.

Evapotranspiration is estimated to be 54% to 59% of mean precipitation, calculated as the residual from the precipitation water budget, which averaged 56% in this study. The most precise measurement of actual evapotranspiration, using eddy covariance and Bowen ratio climate towers on karst limestone across Central Texas, measures evapotranspiration as 69% of precipitation under average conditions (Hauwert and Sharp 2014). Prior to groundwater tracing delineation of groundwater basins, evapotranspiration was estimated to be 85% based solely on channel water balance (Woodruff 1984; Slade et al. 1986; Table 6).

The total precipitation budget reported from the 1980's channel water balance differs from those derived from site-scaled climate towers across Central Texas (Hauwert and Sharp 2014) and this study not only because of differences in groundwater basin divides, but also because the 1980's studies used precipitation over the entire recharge and contributing zones. The allogenic source contributing zone was not included for the reference precipitation area in the channel water balance study presented here, since the upgradient Trinity Aquifer is not an impermeable catchment area, but potentially recharges to lower water-bearing intervals (Mace et al. 2000). Furthermore, it has not yet been determined if the springflow that discharges allogenic recharge from the contributing zone streams originates entirely within the watershed boundaries. The outcrop area of Edwards Aquifer within the recharge zone is the hydrologically-relevant catchment area for a precipitation water balance measurement of autogenic recharge. The difference between

Table 5. Comparison of Blanco River Flow Loss to Barton Springs Discharge.

Cumulative Flow Interval Start	End	# of Days	Barton Springs Inflection Point Discharge (m^3)	Blanco River Flow Loss 3 Months Prior Discharge (m^3)	Barton Springs Discharge (m^3)	Blanco River Flow Loss (m^3)	Onion Creek Flow Loss (m^3)	Pumpage (m^3)	Blanco River Flow Loss/ Barton Springs Discharge (%)	Blanco River Flow Loss/ Total Discharge (%)	Onion Creek Flow Loss/ Total Discharge (%)
4/1/1978	10/31/1978	213	0.57	0.54	13,881,870	13,501,354	not gauged	2,130,388	97	84	---
3/1/1984	9/30/1984	213	0.96	0.37	14,801,782	8,453,016	1,292,991	2,476,740	57	49	7
10/20/1988	1/24/1989	96	0.79	0.82	5,945,179	4,000,885	28,258	1,062,354	67	57	0.40
10/1/1989	3/13/1990	163	0.54	0.59	7,163,573	7,065,710	216,130	1,352,779	99	83	2.54
8/1/1990	12/31/1990	152	0.59	0.48	8,467,598	7,718,701	370,412	1,512,893	91	77	3.71
3/1/1996	11/30/1996	274	0.59	0.17	17,422,064	10,330,616	507,151	3,852,665	59	49	2.38
11/1/1999	5/31/2000	212	0.88	0.68	14,427,456	11,652,452	200,644	3,073,280	81	67	1.15
10/1/2008	9/1/2009	335	0.65	0.51	15,391,407	10,141,545	0	5,672,479	66	48	0.00

Table 6. Comparison of Channel and Site-Scaled Preciptation Water Balances.

Water Balance Component of Precipitation	Channel Water Balance				Site Scale Field Meas.
	Slade et al 1986; Woodruff 1984	Slade 2014	Slade 2015	This Study	Hauwert et al. 2014
Catchment Area Used	917 km^2	233 km^2	233 km^2	176 km^2	0.19 km^2
Upland Autogenic Recharge	0.89%	<6.6%	9%	15-20%	28%
Stream Autogenic Recharge	5%*	---	4%	8%	---
Evapotranspiration	85%	---	70%	54-59%	69%
Downstream Runoff	9%	---	17%	18%	3%

*combined stream autogenic and allogenic recharge

comparing recharge of a 917 km^2 (354 mi^2) area versus a 176 km^2 (68 mi^2) area alone would explain that the 1980's autogenic recharge and downstream runoff values would be about 20% of values measured using the smaller outcrop infiltration area used in the study presented here. Slade (2014) recalculated autogenic recharge using his estimated area for the recharge zone without the contributing zone, based on data collected for the 1980's channel water method budget, and derived <6.6% of precipitation.

Limitations in the channel water balance methodology used in this study may also account for the difference between site scaled and aquifer-wide scale evapotranspiration and runoff. From July 31, 2004 until October 11, 2006, which includes most of the water balance interval, the Onion Creek Buda gauging station was not operating due to bridge repairs and a replacement Twin Creek station was the representative downstream station for Onion Creek. The Twin Creek gauging station includes more than 39 km^2 (15 mi^2) more area as well as perennial flowing springs downstream of the recharge zone. About 54% of the total downstream runoff from the recharge zone measured in this study was attributed to Onion Creek. Likely much of the runoff measured at this gauging station was locally derived and not runoff from the recharge zone, although the data were not available to distinguish the two sources. The additional spring-fed area upstream of the Twin Creek gauging station but downstream of the recharge zone may exaggerate downstream runoff and underestimate autogenic recharge and evapotranspiration.

While the Blanco River averaged only a 6% contribution to total groundwater discharge including Barton Springs over the water budget interval, during drought conditions it may sustain a half or more of the discharge of Barton Springs. Because the tracer arrival time from Blanco River to Barton Springs is about three months, the direct flow comparison of any two recessions during a given time interval that was used in this study tends to underestimate the potential contribution to Barton Springs. Tracing conducted during drought intervals suggests that Blanco River recharge arrives at Barton Springs through two separate groundwater flow routes, one primarily under confined artesian pressure capable of transmitting instantaneous pressure pulses to Barton Springs and a separate groundwater flow path under unconfined conditions whose effects are experienced several months later. The new direct tracing and flow comparison results infer that the low-gradient springflow recession that Barton Springs experiences during drought conditions appears to vary with delayed arrival of baseflow from the Blanco River. The Barton Springs discharge recession inflection point varies between droughts, consistent with the hypothesis that the slower rate of discharge diminishment is caused by external factors, such as variability of Blanco River recharge, rather than internal hydrogeological characteristics of the Edwards Aquifer.

Conclusion

Using the channel water balance method, incorporating mean daily flow gauging upstream and downstream of the recharge zone for the Barton Springs Segment, and flow loss in the major stream channels of the Blanco River and Barton, Williamson, Slaughter, Bear, Little Bear, and Onion Creeks accounted for 56% to 67% of the total groundwater discharge of Barton Springs and wells. The remaining 33% to 44% of total groundwater discharge originated as upland autogenic recharge from the intervening area and, to a lesser extent, as unquantified leakage sources from the Trinity Aquifer, the Saline Water Zone, urban water, and possibly adjacent Edwards (Balcones Fault Zone) Aquifer Segment groundwater inflows. Stream autogenic recharge measured 17% of total groundwater discharge. Allogenic recharge from contributing zone runoff measured 39% to 50% of total groundwater discharge, depending on the distribution of recharge in Barton Creek downstream of Loop 360. About 50% to 61% of Barton Springs discharge is autogenic recharge that originates from the recharge zone. Because of limited resources for the protection of water supplies, endangered species habitat, and treasured recreational pools, understanding water sources for the aquifer is critical for focusing management efforts. The findings establish the importance of the recharge zone in providing recharge to wells and Barton Springs. Although both Barton Springs and the Blanco River flowed through the drought on record, the results of this study suggest that

future water diversions that lead to the drying of Blanco River baseflow could significantly affect Barton Springs discharge.

The findings are consistent with the concept of the Edwards Aquifer as a relatively mature karst aquifer with well-developed recharge structure. This study demonstrates the need for intensive data for characterizing recharge in karst areas and shows that streamflow loss water balances can be consistent with site-scaled water balances, but only after source areas are sufficiently delineated using direct groundwater tracing. Streamflow gauging water balances must be evaluated on whether the source areas were sufficiently delineated to eliminate the possibility for alternate interpretations. Between the channel water balance methods applied in the 1980's and in this study, stream flow loss as a fraction of total groundwater discharge differed less than 3% for channels of Williamson, Slaughter, Bear, Little Bear, and Onion Creeks, but 6% to as much as 25% for the Blanco River and Barton Creek, where groundwater basins were significantly redefined with groundwater tracing. In precipitation water budgets, more representative values of autogenic recharge were calculated using the Edwards Aquifer outcrop within the recharge zone area as a reference area, rather than the combined recharge and contributing zones.

In general, evapotranspiration cannot be precisely measured from channel water balances. Water balances could be improved by collecting continuous flow data on Onion and Little Bear Creeks closer to the edge of the recharge zone and monitoring all gauging stations for a longer interval. While the Barton Springs Segment is one of the most intensely traced aquifers in the world, additional tracing studies can help to refine the distribution of recharge across groundwater divides along Barton Creek downstream of Loop 360, Onion Creek, and Blanco River.

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Runoff Modeling to Inform Policy Regarding Development of Green Infrastructure for Flood Risk Management and Groundwater Recharge Augmentation along an Urban Subcatchment, Ciudad Juarez, Mexico

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Abstract: Changes in land use patterns at expanding border cities along the U.S.-Mexico transboundary area have severe impacts on runoff coefficients and flood risk management. Severe rain is the most representative type of precipitation in the Paso del Norte (PdN) region (New Mexico and Texas in the United States and Chihuahua in Mexico), characterized by high intensity, low duration, and high volumes of rain falling in localized, small areas. Rains generate flooding and damage to urban infrastructure, putting at risk people and properties along the arroyos, which lack hydraulic design to control overflowing. While using a Geographic Information System (GIS), we applied the Hydrologic Modeling System (HEC-HMS) to model streamflow at the study site while building the hydrologic domain using ArcGIS with the Flow Area extension. Flood risk analysis was generated to evaluate potential sites for establishment of Green Infrastructure (GI) as a means of reducing risk and induce recharge to local aquifers. A hydrologic model was created using HEC-HMS under GIS tools and later using Flood Area® hydrologic software to evaluate flood risk analysis. For small-scale watersheds (< 10 km^2) runoff can be greatly reduced by using and developing an urban hydrology approach. Furthermore, using GI and applying an urban hydrology approach can generate synergistic benefits by reducing flood risk, enhancing recharge to aquifer formations, weakening urban heat islands, improving habitat for regional species, and generating a common site for social interaction between neighbors. Binational agencies have adapted a new policy to address and promote the generation of such sites while academia, local government, and Non-Governmental Organizations (NGOs) have taken up the challenge of promoting joint collaboration leading to local solutions to the ancient problem of flood risk.

Keywords: *urban hydrology, transboundary watersheds, green infrastructure, flood risk assessment, permaculture, social participation and education policy, aquifer recharge*

Cities are growing all over the world as people move from rural areas to urbanized areas. Demographic projections indicate that by the year 2050, close to 70% of the world´s population will live in cities (Barney 2015). Mass migration towards cities is changing land use/land cover (LULC) patterns, generating increasingly reduced infiltration on paved surfaces causing loss of native vegetation. Resulting changes in runoff behavior, combined with unplanned urban development, place new population centers at risk of flooding, an effect likely to be exacerbated as climate leads to erratic precipitation patterns.

The North American Monsoon (NAM) produces thunderstorms responsible for severe meteorological conditions, including flooding,

hail, wind, dust storms, and lightning in northwest Mexico and the southwest United States (Adams and Comrie 1997; Higgins et al. 2003; Adams et al. 2014). Despite the generally dry climate in semi-desert environments such as the PdN region, floods result from intense precipitation events linked to the summer monsoon. As population in a particular area increases, so do flood risk and degree of damage to urban infrastructure (Gil et al. 2009). Extreme flooding events have occurred in the past within the Paso del Norte (PdN) region (El Paso, Texas in U.S. and Ciudad Juarez, Chihuahua, in MX), where extreme precipitation events exacerbated by changes in LULC have generated loss of property and human life (Figure 1) (Granados et al. 2013a; Granados, et al. 2013b).

The concept of urban hydrology is not new (Leopold 1968; USDA 1986); however, the concept has developed greatly over succeeding years. An understanding of the effects of different types of urban development has emerged. Construction of broad impervious areas leads to increased runoff and decreased recharge in areas where much surface water would have formerly infiltrated (Day and Bremer 2013). Proper urban hydrologic designs, for instance with Green Infrastructure (GI), can manage the movement of water within an urban watershed, and in so doing, help to improve quality of life, reduce contamination, and decentralize water storage (Spatari et al. 2011). GI offers a contemporary approach to the management of landscape resources in urban environments in harmony with newly built infrastructure, or with traditional gray infrastructure. GI has developed rapidly in many southwest North American cities because of the opportunities it has provided in meeting the ecological, economic, and social challenges of spatial planning (Mell et al. 2009; Mell 2010; Zavrl and Zeren 2010; USEPA 2015). The concept is spreading rapidly within large Mexican cities along the northern border. In part, this is because of the support of the Border Environment Cooperation Commission (BECC) which seeks to provide important basic scientific information on the hydrology of the urbanized watersheds (BECC 2016).

In the context of GI policies, Ciudad Juarez, Chihuahua, Mexico, has been an entrepreneur city with vast experience in innovative policy. The maquiladora industry was first established there as a means of producing jobs and expanding the regional economy. Later, construction companies established large units in Ciudad Juarez in response to the need for housing and other urban infrastructure. Furthermore, as the city developed, important governmental institutions such as the Instituto Municipal de Investigacion y Planeacion (IMIP), and NGOs and international cooperation agencies, such as BECC and the North American Development Bank (NADBANK), began to focus some of their important operations on the city. BECC has taken leadership in promoting GI for the city as a policy design of a potential solution for small-scale flood risk and aquifer recharge. This policy approach has led to progress towards flood risk prevention, esthetic enhancement of urban infrastructure, improvement of green areas to promote environmental benefits for native species of flora and fauna, and development of community based collaboration and involvement towards a sustainable city. For example, BECC has promoted two major international conferences in which experts from the U.S. and Mexico presented their work on applied and working GI in different cities of both countries. A third international conference is planned for 2016. Additionally, BECC is promoting major investments in GI in other border cities. Furthermore, BECC has participated in studies of the potential for rainwater harvesting and management for GI and reduction of flood risk. GI policies approach environmental restoration through permaculture and positive feedback. For example, rain will produce runoff; runoff will move downhill and can be collected in basins, both large and small. Soils and other organic materials will also collect in the basins, generating a soil profile adequate for plant growth. In basins where water and nutrients are available, plants will develop the soil profile through root growth, with deep-rooted species providing the greatest benefit. The roots will promote infiltration into deeper layers, and, where the water supply is sufficient, recharge to subjacent aquifers. Additionally, plants will promote establishment of bird species that will enhance positive feedback by sowing native vegetation species from the wider region in their droppings. Communities will participate by adopting these green areas for social interaction among neighbors.

Crime rates are reduced as a result of community participation and involvement (Scott et al. 2013). Positive social outcomes will encourage the development of new GI sites. The development and maintenance of small-scale GI is well within the means of local communities (Figure 2).

The main aim of the study is to apply surface water modeling in order to gain a better understanding of the potential runoff of a small-scale watershed in Ciudad Juarez, Chihuahua, Mexico for 2, 10, and 25 year return period scenarios, under a Type II precipitation regime (USDA 1986). A return period is defined as the recurrence interval estimated from the probability of an extreme precipitation event; for instance, a 25 year return period relates to the largest precipitation event that is likely to occur within an interval of 25 years. Model results are interpreted in terms of flood assessment, and potential for enhanced recharge in an urbanized catchment. A further aim is to examine the potential for cooperation between local communities and government organizations in the development of GI as a means of controlling flooding and augmenting groundwater supply. This study focuses on one small catchment, but has implications for innumerous urbanized sub-watersheds in the semi-desert environment along the border between Mexico and the U.S.



Figure 1. Type II punctual precipitation and its potential for catastrophic flood events. Photos are from the June 2010 storm event in Ciudad Juarez, Chihuahua (Chaparro 2006; Diario de Juarez 2006).

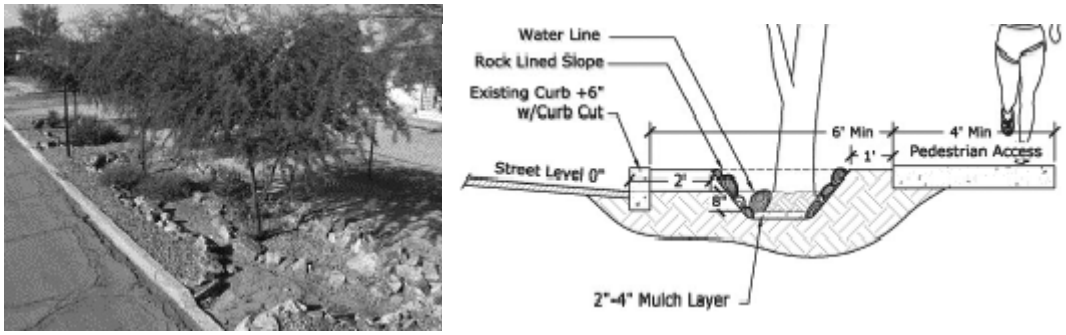

Figure 2. Green infrastructure and design prototype in Tucson, AZ (modified from WMG 2012).

Study Area

The study area is the catchment of an ephemeral watercourse known as Arroyo Tapioca, located at the geographic center in the City of Juarez, Chihuahua, Mexico (Figure 3). Arroyo Tapioca is about 21.78 km long, and the study focuses on an area of 11.65 km^2 in the upper part of the catchment. Slopes in the area range from 0.03% to 0.15%, resulting in laminar and sheet flow runoff.

The urban infrastructure of the study area dominantly features densely populated zones, particularly at the lower (northern) end. Impermeable areas range from 80 to 95% of surface area, which increases the runoff potential. This area is estimated to have a total population of 9,656. Much of the drainage density (Dd ~85 km/km^2) is controlled by major and secondary streets (Figure 4). The Arroyo Tapioca catchment, like most of Ciudad Juarez, is prone to flooding that results in severe localized property damage.

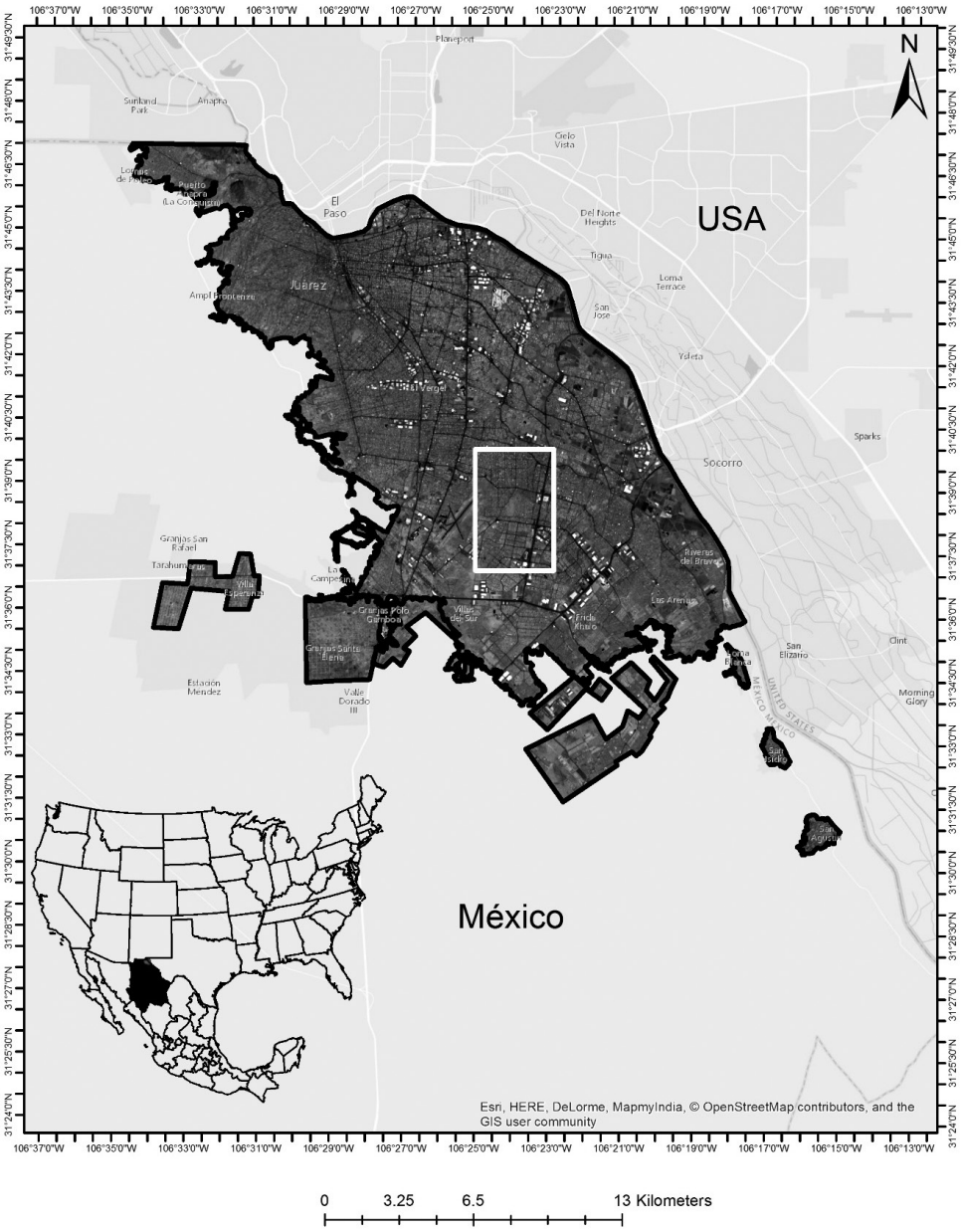

Figure 3. Location of the area of interest at Paso del Norte, Ciudad Juarez, Chihuahua, México.

Methods

Precipitation data from 1990 to 2010 were acquired from the National Water Commission of Mexico, the International Boundary and Water Commission, and the Autonomous University of Ciudad Juarez (UACJ) Meteorological Lab (UACJ-MetLab). A LIDAR (Light Detection and Ranging) survey undertaken in 2008 by the UACJ Geographic Information Center (UACJ-CIG) was used to prepare a Digital Elevation Model (DEM) with bare ground elevations; the resulting map has a resolution of 2 m per pixel. Physiographic parameters were then inserted into HEC-HMS v3.4 to calculate runoff volumes and other parameters such as watershed areas, main stream length, and average slope (Estrada-Leyva 2013). These parameters were calculated as an extension of a GIS project under ArcView v3.2 software which allowed the insertion of runoff coefficients and concentration and lag times for runoff measurements. Prevailing soil types, LULC, and impermeable areas were mapped from aerial photographs. These serve as the main source of land truth analysis and were used to estimate a Curve Number (CN) from which runoff coefficients were calculated. For this study, soils were considered type B (USDA 1986), characterized as soils that allow very little infiltration (i.e., Vertisols). Once the runoff volumes were calculated, potential flooding areas with estimated buffer zones were obtained using the Flood Area extension of ArcMap v9.3. This enabled determination of the extent of surface water pulses based on calculated return periods for each modeled flood event. The hydrologic models were run for a 24 hour torrential rain event (the

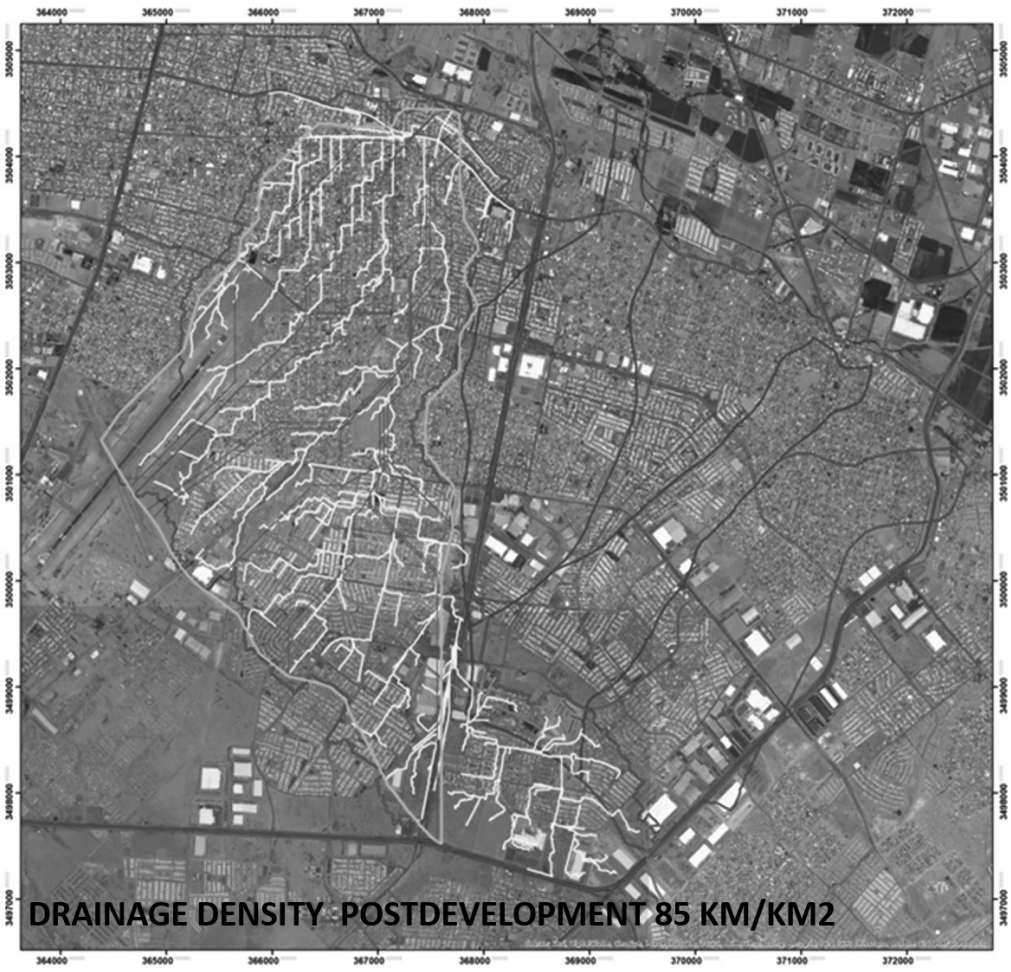

Figure 4. Drainage density (Dd) at area of interest Arroyo Tapioca (Modified from Estrada-Leyva 2013).

modeling criterion used by municipal authorities in Ciudad Juarez, Chihuahua, Mexico), using return periods of 2, 10, and 25 years.

Results and Discussion

The vegetation cover of the study area is less than 30% and the average CN is about 80. Hydrologic modeling results from HEC-HMS and ArcView v3.2 are displayed in Figure 5. The model shows the effects of urban features on the post-development locations of streams, junctions, sub-watersheds, and outlets in the Arroyo Tapioca Watershed. Divides between internal sub-watersheds are governed by major streets and the modified topography resulting from recent dense urbanization. Mainstream arroyos in each sub-watershed follow street paths; nonetheless, runoff at the scale of the entire watershed continues to follow the pre-development northward direction. The street-based drainage pattern has an important hydrologic consequence because flooding hazards are greatly increased at each of the 90° turns imposed by the street grid. Hazards are exacerbated by the changed LULC (removal of vegetation and construction of impermeable surfaces) and the densification of Dd.

Table 1 shows potential precipitation depths for each set of conditions. For a return period of 25 years, the accumulated precipitation depth is 84 mm, representing one third of the average annual precipitation estimated at 250 mm/yr. The model simulations include 2, 10, and 25 year return periods for a 24 hour torrential rain, matching the return periods used by local authorities in Ciudad Juarez, Chihuahua (IMIP 2004). These potential storm events can accumulate important volumes of water because of the transformation from precipitation to runoff linked to impervious areas, low roughness coefficients, length of main streams, and dominant slopes. Slopes and other physiographic parameters are listed for each sub-watershed in Table 2. Points 1 and 2 lie to the north of a paleolake that formerly drained in their direction, but urban topography modification has redirected surface flow to the southeast. Estimated volume and peak discharge were calculated for the junctions and outlets of each of the catchment sub-areas of the Arroyo Tapioca Watershed. Table 3 shows the accumulated

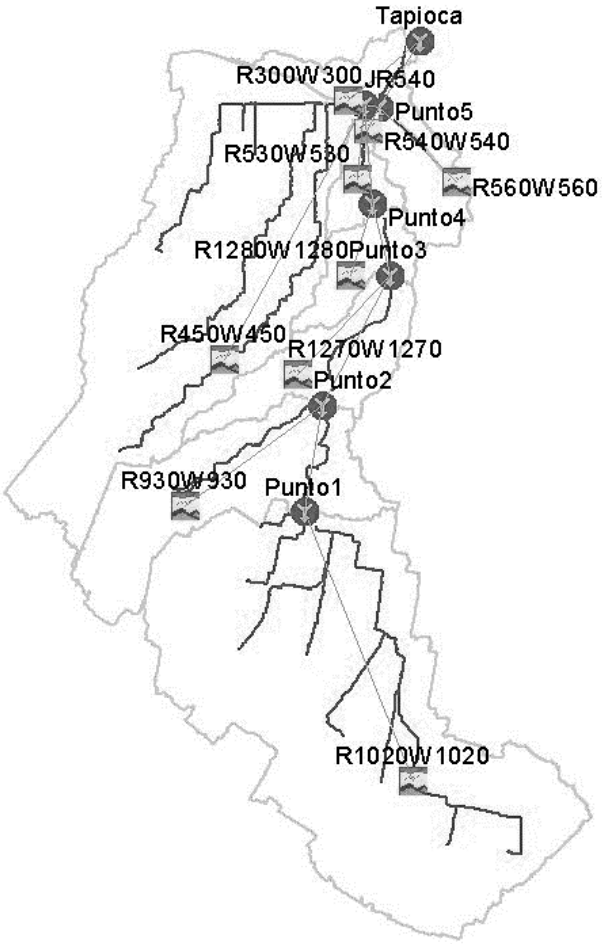

Figure 5. Hydrologic model generated under HEC-HMS showing catchment areas, junctions, and outlets for the area of interest.

Table 1. Potential rainfall depths (mm).

Duration	Return Period		
	2 yrs	10 yrs	25 yrs
5 min	8.30	12.80	15.30
15 min	16.50	25.40	30.40
1 hr	20.70	33.40	42.00
2 hr	24.00	38.90	48.90
3 hr	26.20	42.50	53.40
6 hr	30.50	49.40	62.10
12 hr	35.50	57.40	72.20
24 hr	41.30	66.80	84.00

volumes and peak discharges as functions of the return period. The main stream in Table 3 was defined from (Point 1) upstream to the final outlet at the Tapioca site (Figure 5, also shown as site 6 in Figure 6).

The biggest catchment area, corresponding to Point 1 in Table 3, is about 6.8 km^2 with an estimated impervious area of 80%. It has a long main stream that meanders through the streets for 6.5 km (Table 2). The Tapioca outlet has the most accumulated volume for a return period of 25 years (845,000 m^3), as well as, the maximum estimated peak discharge (44 m^3/s) (Figure 7). The outlet is small (0.5 km^2), but its impervious area is high (95%) and its slope is minimal (1%); hence, the concentration time (TC in Table 2) for runoff accumulation in that area is short, approximately 41 minutes, which consequently generates a higher danger of flooding. Only a small amount of runoff comes from the immediate 0.5 km^2; most originates in the rest of watershed. Therefore, the sub-watershed at catchment area R300W300 (Figure 5), is at higher risk of flood damage during intense storm events. In contrast, the upstream sub-watershed R1020W1020, with an area of approximately 6.8 km^2, has an estimated accumulated volume of 482,000 m^3 for a 25 year return period. In this case, peak discharge reaches about 25 m^3/s over a more extended area; hence, the flood risk could be distributed and significantly reduced.

Enormous quantities of runoff water are generated every year from extreme storm events in the PdN region. It has been demonstrated that flooding potential is a hazard risk within the Arroyo Tapioca, placing residents in danger. Nonetheless, potential solutions to this challenge can be achieved, with added advantages, from the application of engineering alternatives.

Aquifer recharge opportunities

PdN cities are dependent on groundwater for their supply of potable water. Even though rainfall is low (avg. 250 mm/yr.), huge amounts of runoff are generated from impermeable LULC areas resulting from urbanization, as demonstrated by the hydraulic modeling response of the Arroyo Tapioca Watershed. This runoff creates problems and flood risks along mainstream corridors. Aquifer formations and their water tables in the region are falling up to 1.5 m/yr. because of over pumping from deep groundwater wells (Hibbs 2004; Eastoe et al. 2008; Hawley et al. 2009; Granados et al. 2012; Eastoe et al. 2016). The need for innovative facilities for focused artificial recharge is enormous since most of the area has

Table 2. Physiographic parameters for mainstream and sub-watersheds at the Arroyo Tapioca study site.

Catchment Name	Total Area (km^2)	Slope (%)	Length (m)	Roughness Coefficient "n"	TC min	TR min	IS %
Point 1	6.8	0.003	6558	80	114	69	80
Point 2	2	0.007	3616	80	60	36	70
Point 3	1	0.01	3182	80	50	28	95
Point 4	0.5	0.01	1968	80	35	21	95
W530R530	0.3	0.01	1546	80	29	17	95
Point 5	0.02	0.01	568	80	14	9	95
W560R560	0.5	0.01	1957	80	39	23	95
Tapioca	0.5	0.01	2379	80	41	24	95

TC = Concentration Time; TR = Retention Time; IS = Impervious Surface

Table 3. Estimated peak discharge and accumulated volume per return period (RP).

Catchment Name	Hydrologic Elements	Drainage Area (km^2)	Peak Discharge 2 year RP (m^3/s)	2 years Accumulated Volume (*1000 m^3)	Peak Discharge 10 year RP (m^3/s)	10 years Accumulated Volume (*1000 m^3)	Peak Discharge 25 year RP (m^3/s)	25 years Accumulated Volume (*1000 m^3)
Point 1	Catchment Area R1020W1020	6.8	11	215	21	392	25	482
Point 2	Catchment Area R930W930	2	14	272	26	499	32	614
Point 3	Catchment Area R1270W1270	1	16	308	30	562	36	691
Point 4	Catchment Area R1280W1280	0.5	17	326	32	594	38	730
Junction 540	Catchment Area R530W530	0.3	17	338	32	614	39	755
*Point 5	Catchment Area R540W540	0.02	.	.	.	.	.	.
	Catchment Area R560W560	0.5	18	356	34	646	41	793
Tapioca	Catchment Area R300W300	0.5	20	375	37	680	44	835

*Most of the estimated peak discharge and accumulated volume at Point 5 comes from Catchment Area R560W560 (0.5 km^2); Catchment Area R540W540 (0.02 km^2) is not representative.

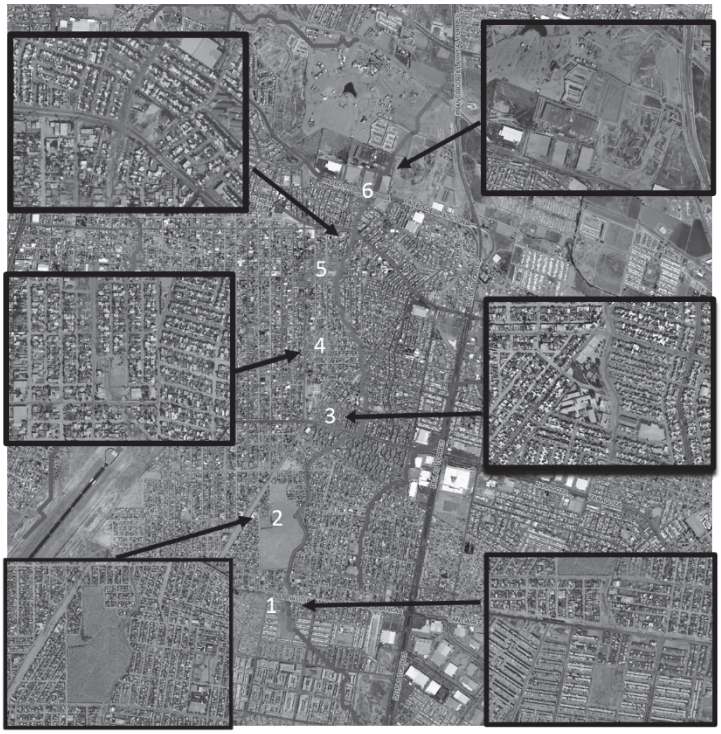

Figure 6. Modeled potential flooding areas within the Arroyo Tapioca for 25 year return.

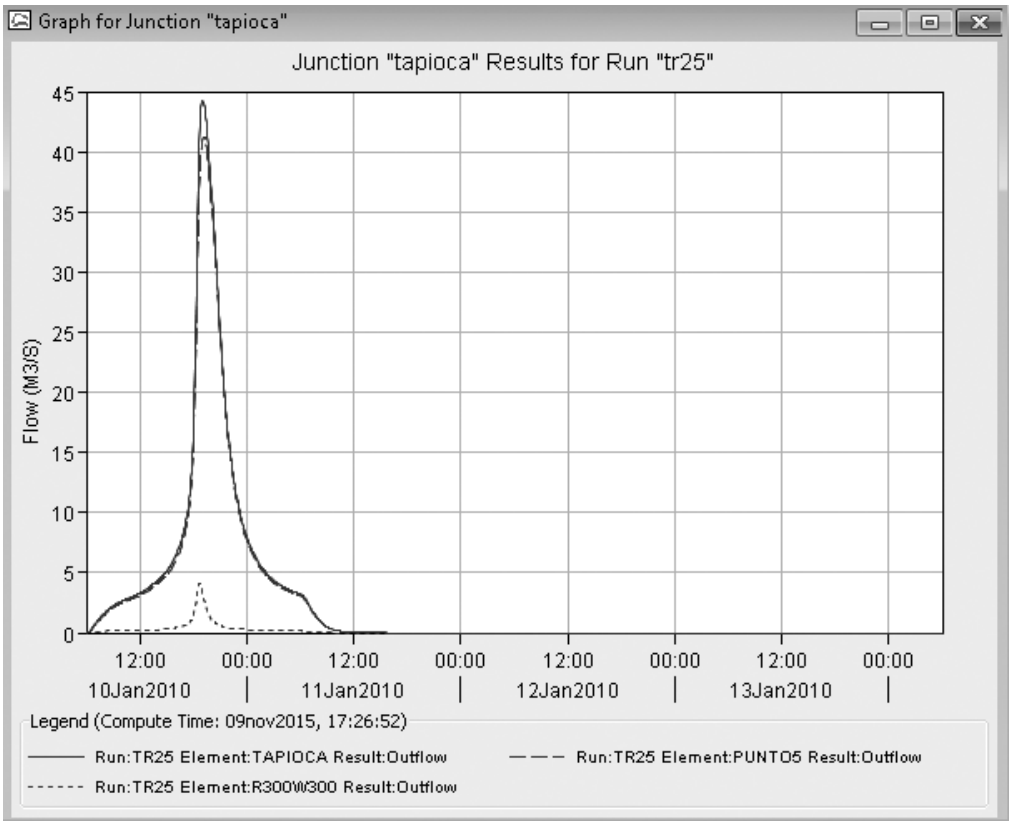

Figure 7. Hydrogram peak discharge for the January 10, 2010 torrential rain event.

been urbanized with high percentages (80 to 95%) of impermeable surfaces. Finding large areas for traditional recharge facilities is almost impossible because the price of land is too high; hence, there is a need for a different approach to reduce flood risk while enhancing recharge. The implementation of GI is a potential solution, and has the added advantage of encouraging and enabling a basic level of community involvement.

Conclusions

An understanding of urban hydrology in small watersheds is becoming more important for flood control and prevention of damage to urban infrastructure in regions where climate change is altering precipitation. For the region of interest in this study, it is clear that extreme precipitation events have already put human life, property, and urban infrastructure at risk. Furthermore, unplanned urban growth has not taken into consideration the importance of urban hydrology in building safe and viable communities. The increase of impervious areas has produced excessive runoff volumes focused into densely populated parts of the lower catchments. If proper runoff control infrastructure is generated, different volumes of water may be put to use in improving the urban environment, while reducing flood risk in lower elevation areas and enhancing recharge. The use of GI to promote flood control is a strategic approach using a permaculture concept in which small local instances multiplied many times can generate a better result than the construction of massive and costly infrastructure. Furthermore, such sites can have a positive impact on the environment, native flora and fauna, and community interaction and wellbeing. All of the findings of this study are applicable throughout urban areas of the PdN region and in other regions of similar climate.

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Groundwater Depletion Beneath Downtown Tucson, Arizona: A 240-Year Record

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Abstract: Beginning with the foundation date of Spanish Tucson in 1775 and continuing through the 1890s, shallow groundwater and related features, including a perched aquifer, stream-bed springs and a short perennial reach in the Santa Cruz River, progressively disappeared from the Downtown Tucson area. The 20th-Century city grew rapidly, and increasing groundwater withdrawals caused large declines in the regional aquifer of Tucson basin, leaving a second perched aquifer with non-aqueous contaminants beneath Downtown. Isotope data show that this fully perched aquifer retains residual water from the regional aquifer, plus recharge derived from a nearby small watercourse. The present regional aquifer beneath Downtown is locally layered and partly confined, and contains water 7-20 Ka years old originating at the northern and southeastern basin margins.

Keywords: *groundwater, urban growth, semi-arid, isotopes, contaminants*

Urban population growth in the semi-arid southwestern region of North America has been rapid since the end of World War II, when population in the USA began to shift to the desirable climates of the "sun belt", and a large population from Mexico moved into border cities in pursuit of economic opportunities afforded at the frontier. Even in cities with access to surface water, the need to supply adequate water for domestic and industrial consumption led to over-exploitation of groundwater. Well-documented examples exist in Albuquerque, New Mexico (Plummer et al. 2004) and the urban area encompassing El Paso, Texas and Ciudad Juárez, Chihuahua (Hibbs et al. 1997), where the static water levels (SWL) fell by tens of meters by the end of the 20th Century. In Tucson, Arizona, where little surface water was available, the population increased from a few thousand in the 1870s to about 1 million by 2012. Figure 1 shows population growth in Pima County, of which Tucson has long been the largest city; in latter years, as the Tucson metropolitan area has extended far beyond the city limits, the county rather than the City of Tucson population has become the better measure of the urban population. The entire metropolitan area depended completely on groundwater until a major engineering project brought Colorado River water to the city by canal in 1992 (Gelt et al. 1999). Small local declines in SWL in Tucson basin in the 1940s (Turner et al. 1943) became large, widespread declines of 15 to more than 50 m by the 1980s (Anderson 1988), and grew locally to as much as 60 m by the 1990s (Gelt et al. 1999).

The effects of over-pumping across the breadth of Tucson basin and the potential for subsidence as a result of dewatering of basin sediments have been reported elsewhere (Anderson 1988; Gelt et al. 1999; Pool and Anderson 2007). Downtown Tucson, the site of the original Spanish settlement towards the west side of the Tucson basin, presents a special case study. The nature of the basin fill (discussed below) differs from that in the broad eastern part of the basin, and there has been little pumping in the area for several decades. Furthermore, historical records offer some insight about changes in hydrologic conditions resulting from two centuries of groundwater exploitation.

In this article, the historical record of groundwater use in Downtown Tucson will

be reviewed and interpreted, and a new set of groundwater isotope data will be presented, with the aim of reconstructing the hydrologic evolution of the area since Spanish settlement.

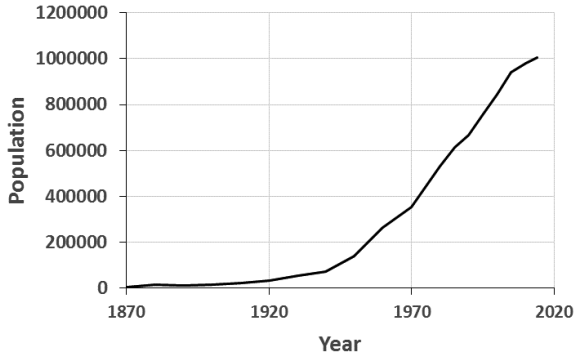

Figure 1. Population of Pima County, 1870-2012. Sources: Pima Association of Governments (2016) and U.S. Bureau of the Census (2016).

Background

Regional Setting

The Tucson basin lies within the watershed of the Santa Cruz River in southern Arizona. The basin is about 30 km (east-west) by 40 km (north-south), and is bounded by mountain ranges of crystalline rock, the Tucson, Santa Catalina, Santa Rita, and Rincon Mountains. Most of the basin floor lies at elevations near 800 meters above sea level (masl). The greater Tucson urban area, which occupies the northern half of the basin, experiences a semi-arid climate with hot summers and cool winters. Rainfall in the central area of the city averages 305 mm annually and is equally distributed between a summer monsoon with brief, localized convectional precipitation, and a fall-winter season of widespread, cyclonic precipitation events associated with Pacific cold fronts or tropical depressions (Gelt et al. 1999).

The basin is of Neogene (post 23 Ma) age, and lies within the Basin-and-Range physiographic province. Regional extension, expressed as detachment faulting followed by high-angle normal faulting, led to rapid deepening of the central part of the basin (Davidson 1973; Eberly and Stanley 1978; Anderson 1987; Dickinson 1999). The basin fill consists of up to 3000 m of alluvial and lacustrine sediments of Oligocene to Holocene age, with fluvial sediments in the flood plains of major streams. Gypsum or anhydrite evaporites occur at depth. Holocene deposits are largely confined to active flood plains. The crystalline basement consists largely of granitic gneiss with Proterozoic and Paleogene protoliths in the Santa Catalina and Rincon Mountains, and of Paleozoic and Mesozoic igneous and sedimentary rocks in the Tucson and Santa Rita Mountains. The principal geomorphic features of the basin include a number of alluvial fans and active flood plains along the major washes, e.g. the Santa Cruz River in the present study area.

Most groundwater in Tucson basin occurs within a single regional aquifer, largely unconfined, hosted in basin-fill alluvium and spanning most of the basin. Static water levels have been strongly modified by decades of pumping, but generally decrease to the northwest, in the direction of the present surface outlet (Gelt et al. 1999). Shallow perched aquifers occur where clayey sediment is common near the surface, and are largely limited to the area immediately east of the Santa Cruz River.

Study Area

This article focuses on the area close to a cross-section of about 5 km, limited to the southwest by Sentinel Peak (a ridge of Oligocene volcanic rock and associated strata jutting east from the Tucson Mountains), and to the northeast by Campbell Avenue (Figures 2 and 3). The area encompasses Downtown Tucson (the commercial center of the city prior to its rapid expansion, and the site of the 18th-Century Spanish settlement), the main campus of the University of Arizona, and part of the flood-plain of the Santa Cruz River. The area straddles the Santa Cruz Fault (Anderson 1987), a high-angle normal fault with the east side downthrown, and one of the principal faults responsible for the deepening of Tucson basin during the Neogene. The basin-fill in this area consists of recent alluvium, about 6 m thick, overlying late Pleistocene Fort Lowell Formation and the Miocene to Pliocene Upper Tinaja Beds. The Fort Lowell Formation truncates the Santa Cruz fault, and is 60 to 90 m thick in the study area. It consists of multiple lenses, varying greatly in thickness and lateral extent, of clay-rich sediment intercalated with silt and sand. The Upper Tinaja Beds consist of gravel, sand, and clayey silt, about

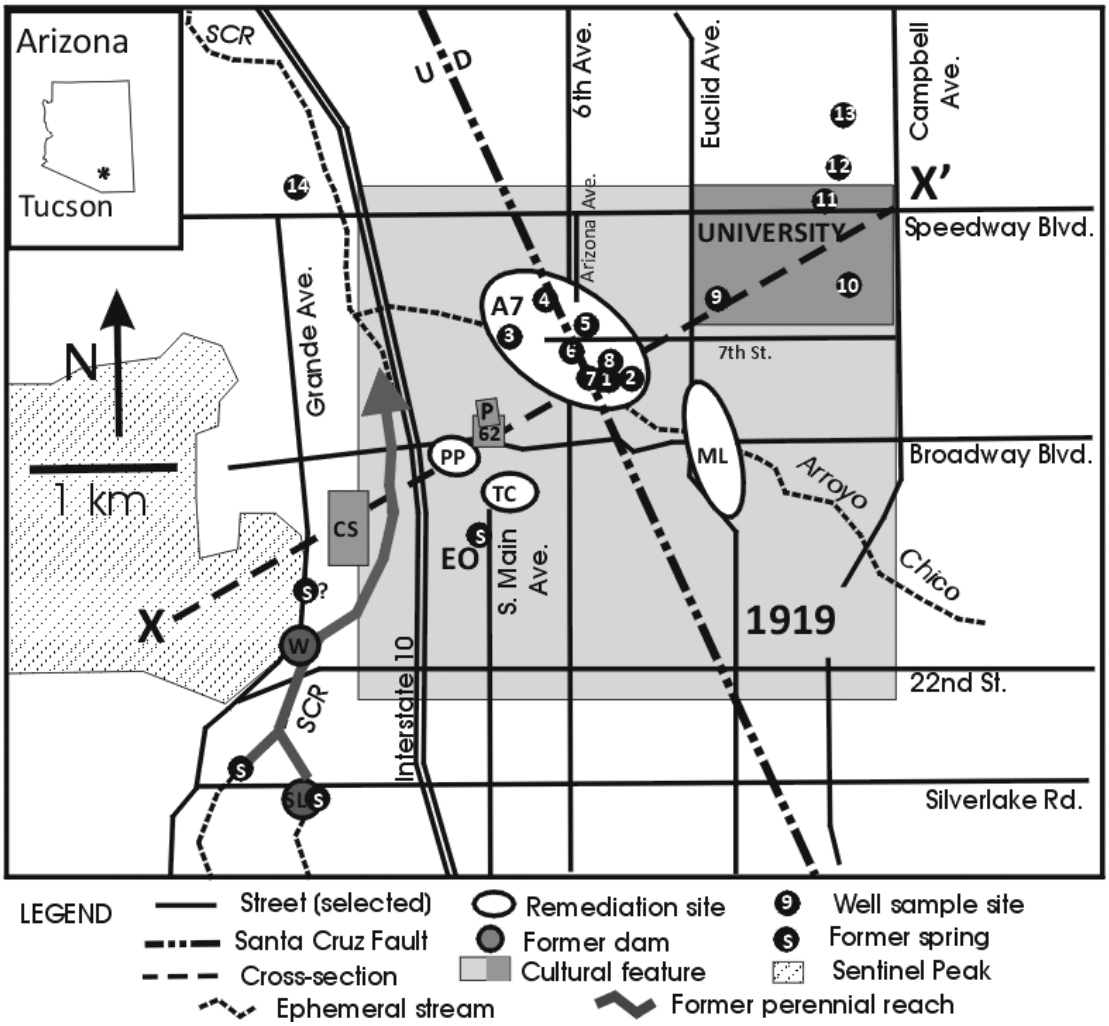

Figure 2. Map of study area. Contaminated sites are Arizona Avenue/7th Street (A7), Mission Linen (ML), Pioneer Paints (PP), and Tucson Convention Center/Police Headquarters (TC). Each contaminated site includes a cluster of wells sampled for the study. Cultural sites are Chuk Shon Village (CS), the Spanish Presidio 1775 (P), and rectangles showing the approximate extent of Tucson in 1862 (62) and 1919. Former dams/reservoirs are Silver Lake (SL) and Warner's (W). SCR = Santa Cruz River. EO = El Ojito Spring. Numbered well sites refer to identifications in Table 1. Information sources: Anderson (1987); Officer (1987); Gelt et al. (1999); Wood et al. (1999); Webb and Leake (2006);. Arizona Department of Water Resources (2016).

30 m thick on the west side of the Santa Cruz fault, and 100 m thick on the east side. (Anderson 1987; City of Tucson Environmental Services 2008). The land surface rises from 710 masl in the entrenched channel of the Santa Cruz River to 750 masl where the cross-section intersects Campbell Avenue. The regional aquifer of the Tucson basin, 3080 m below the surface, is located within the basal Fort Lowell Formation and the Upper Tinaja Beds, and is continuous beneath the study area, with SWLs between 666 and 700 masl. A perched aquifer (or possibly multiple aquifers) 10-30 m below the surface occurs atop clay strata of the Fort Lowell Formation beneath Downtown Tucson, with SWLs at 700 to 711 masl. This fully perched aquifer is locally contaminated with non-aqueous liquid phases (hydrocarbons and chlorinated solvents) discharged in the past from several commercial sites (Figure 2, sites A7, PP, TC, and ML). Both the perched and regional aquifers are at lower

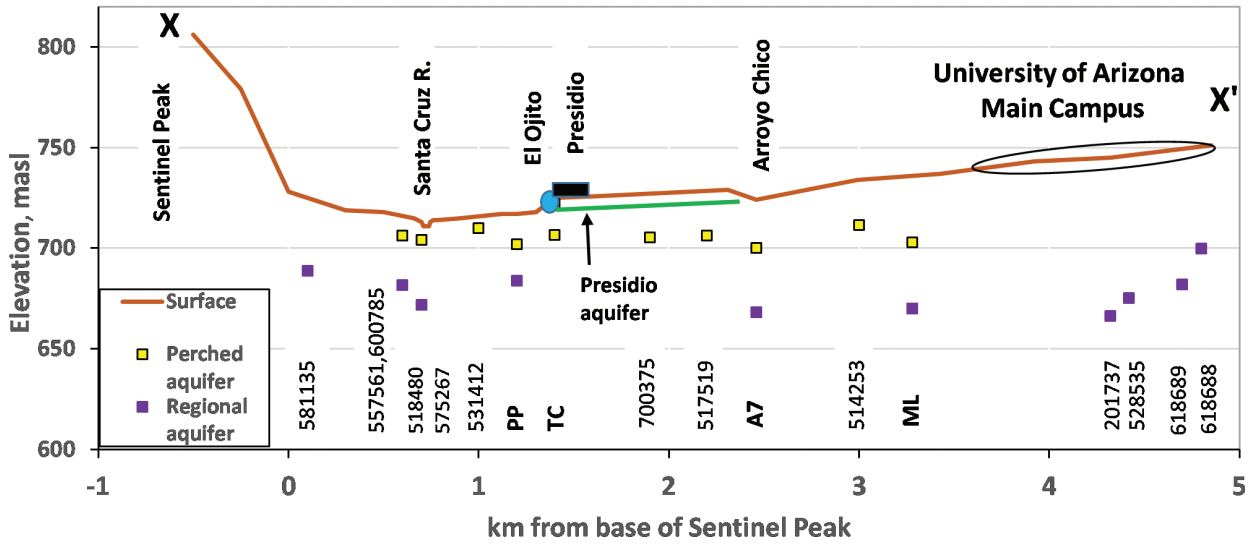

Figure 3. Cross-section of study area (see Fig. 2 for location). Well data are projected on to the section. Well registration numbers (Arizona Department of Water Resources 2016), are given in shortened form as 6 digits (xxxxxx), the full number being 55-xxxxxx.

elevation than the present entrenched bed of the Santa Cruz River. In the study area, water for domestic consumption is pumped only from the regional aquifer at the University of Arizona and from the river flood plain; in other places, samples were available from monitoring wells screened either in the perched aquifer or in the regional aquifer. Figures 3 and 4 show vertical section data derived from drillers' logs available from Arizona Department of Water Resources (2016) using selected logs in which water levels and detailed stratigraphy have been recorded.

Samples for isotope measurement were obtained from five supply wells at the University of Arizona, and from monitoring wells at the following contaminated sites:

1. Pioneer Paint and Varnish, a site contaminated with a variety of organic chemicals (SCS Engineers 2008). Samples were from three wells in the regional aquifer and five in the perched aquifer, all within area PP (Figures 2 and 3).
2. Tucson Convention Center, a site with local gasoline contamination (City of Tucson Environmental Services 2008), with samples from six perched-aquifer wells in area TC (Figures 2 and 3).
3. Arizona Avenue/Seventh Street, a tetrachloroethane (PCE)-contaminated site (Kafura 2007), with samples from two regional-aquifer and six perched-aquifer wells in area A7 (Figures 2 and 3).
4. Mission Linen Superfund site (area ML of Figures 2 and 3, contaminated with chlorinated hydrocarbons), with samples from two perched-aquifer and four regional-aquifer wells.

Previous Groundwater Isotope Research

Previously reported isotope data for groundwater in the study area are available only for wells at the University of Arizona, for the Mission Linen remediation site, and for a few wells in the Santa Cruz flood plain (Kalin 1994; Eastoe et al. 2004; Gu 2005). Eastoe et al. (2004) provided a framework, to be used in this article, for interpreting the origin of Tucson basin groundwater. They presented basin-wide maps of $\delta^{18}O$, $\delta^{34}S$, ^{3}H, and ^{14}C in groundwater, and identified the following: 1.) Localized zones of recent recharge extending several km into the basin from the mountain fronts; 2.) A zone of upward discharge of ancient groundwater, apparently associated with the Santa Cruz fault; and 3.) Distinctive groundwater domains identified by O (water) and S (sulfate) isotope data

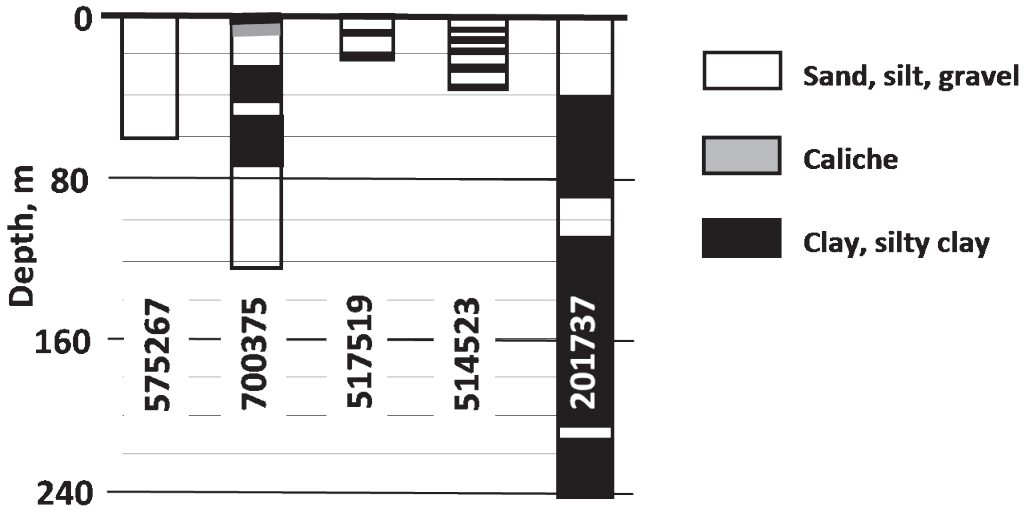

Figure 4. Stratigraphic data from selected drillers' logs (Arizona Department of Water Resources 2016). Well registration numbers are given in shortened form as 6 digits (xxxxxx), the full number being 55-xxxxxx.

in combination, and related to recharge originating from larger streams conveying surface water into the basin. Domain names that will be used here include: A, related to recharge of mountain-derived water from Rillito Creek and its northern tributaries; B, related to the Santa Cruz River; C, related to Cienega Creek; D, related to mountain-derived groundwater possibly discharging upward in the southeastern part of the basin; E, related to Rincon Creek; and F, in the southern part of the basin. The domain map of Tucson basin is shown in Figure 5a, and ranges of $\delta^{18}O$ and $\delta^{34}S$ for domains A to D are shown in Figure 6. Gu (2005) noted that groundwater extraction between the early 1990s (samples of Kalin 1994) and 2000-2005 had led to local replacement of Domain D water by Domain A water (Figure 5b).

Methods

Historical Review

The information presented here is largely drawn from recent historical works in which a review of the primary historical sources is documented.

Sampling and Analyses

Groundwater samples were obtained from active supply wells at the University of Arizona, and during routine sampling campaigns of monitoring wells at several contaminated sites. Isotope measurements were performed at the Environmental Isotope Laboratory, University of Arizona. Stable O and H isotopes of water were measured on a Finnigan Delta S® dual-inlet mass spectrometer equipped with an automated CO_2 equilibrator (for O) and an automated Cr-reduction furnace (for H). Stable S isotopes of sulfate extracted as $BaSO_4$ were measured on a Thermo Electron Delta Plus XL® continuous flow mass spectrometer equipped with a Costech® elemental analyzer for preparation of SO_2. Tritium was measured on 0.18 L water samples after electrolytic enrichment. Carbon-14 was measured on CO_2 extracted from 50 L water samples by acid hydrolysis of dissolved inorganic carbon; the carbon was converted to benzene. Both tritium and carbon-14 were measured in a Quantulus 1220® Spectrometer by liquid scintillation counting. Routine analytical precisions (1σ) are 0.08‰ (O), 0.9‰ (H), 0.15‰ (C), and 0.15‰ (S). Routine detection limits are 0.6 TU for ^3H and 0.2 percent modern carbon (pMC), ^{14}C, undiluted sample.

Correction of ^{14}C Data

The procedure follows Clark and Fritz (1997, 210 ff), using the method based on mixing of stable carbon isotopes, assuming a C_3:C_4 ratio of 3:1 in decaying plant matter in soil, slightly alkaline conditions of initial infiltration of

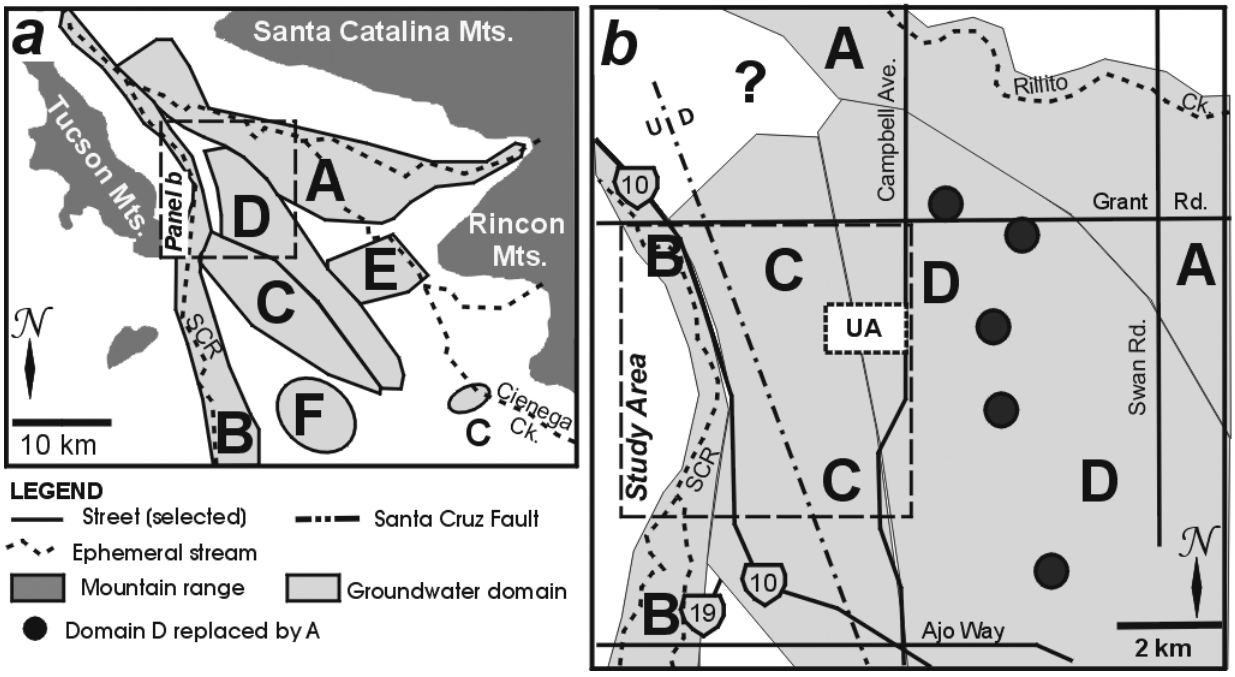

Figure 5. (a) Water domain map of Tucson Basin, after Eastoe et al. (2004); **(b)** Updated water domain map of the area encompassing the study area. UA = University of Arizona; SCR = Santa Cruz River; shaded areas labeled A, B, etc. are domains A, B, etc.

rainwater, and $\delta^{13}C$ values of -1 or -2‰ for rock carbonate (unpublished data of the Laboratory of Environmental Geochemistry). This set of assumptions has been found to yield plausible results for Tucson Basin, with dilution ratios between 0 and 1, and corrected ^{14}C activities less than about 150 pMC.

Review – History of Groundwater Use in Tucson

Father Eusebio Kino, an early emissary of the Spanish Empire, travelled to the area now occupied by Tucson in the 1690s. There he encountered two villages, Wa:k and Chuk Shon, of the Tohono O'odham people (known previously to Europeans as the Papago), several hundred households in total (Sonnichsen 1982; Officer 1987). Each village depended on surface water from short (a few kilometers) perennial reaches of the Santa Cruz River. Water was conveyed from the river to fields around the villages through a system of controlled ditches (Kupel 1986). There is no indication of more extensive perennial reaches in the historical record (Wood et al. 1999; Logan 2002). The water discharged in stream-bed springs from upstream groundwater basins across sills of impermeable rock (Wood et al. 1999). Another spring is reported near Chuk Shon, at the foot of Sentinel Peak (Sonnichsen 1982), but other authors do not mention it; Sonnichsen may have been referring to one of the stream-bed springs. The Chuk Shon perennial reach terminated near the village, or at times a few kilometers downstream (Bourke 1891; Sonnichsen 1982). Native Americans already had a long history of water engineering in the area. About 3000 years ago, canals channeled ephemeral stream water to a settlement known to archaeologists as Las Capas, 15 km downstream of the present city center (Palacios-Fest et al. 2001). Systems of controlled irrigation ditches and shallow dug wells in the flood plain have been excavated in a site dating from 1000 years ago (Logan 2002).

What had been a sufficient water supply for the Tohono O'odham residents became inadequate in the decades following the founding of the Spanish Presidio (fort) on a low mesa opposite Chuk Shon (Figure 2) in 1775. The Presidio was to become the nucleus of Tucson. Any appropriation of local surface water for the domestic and agricultural

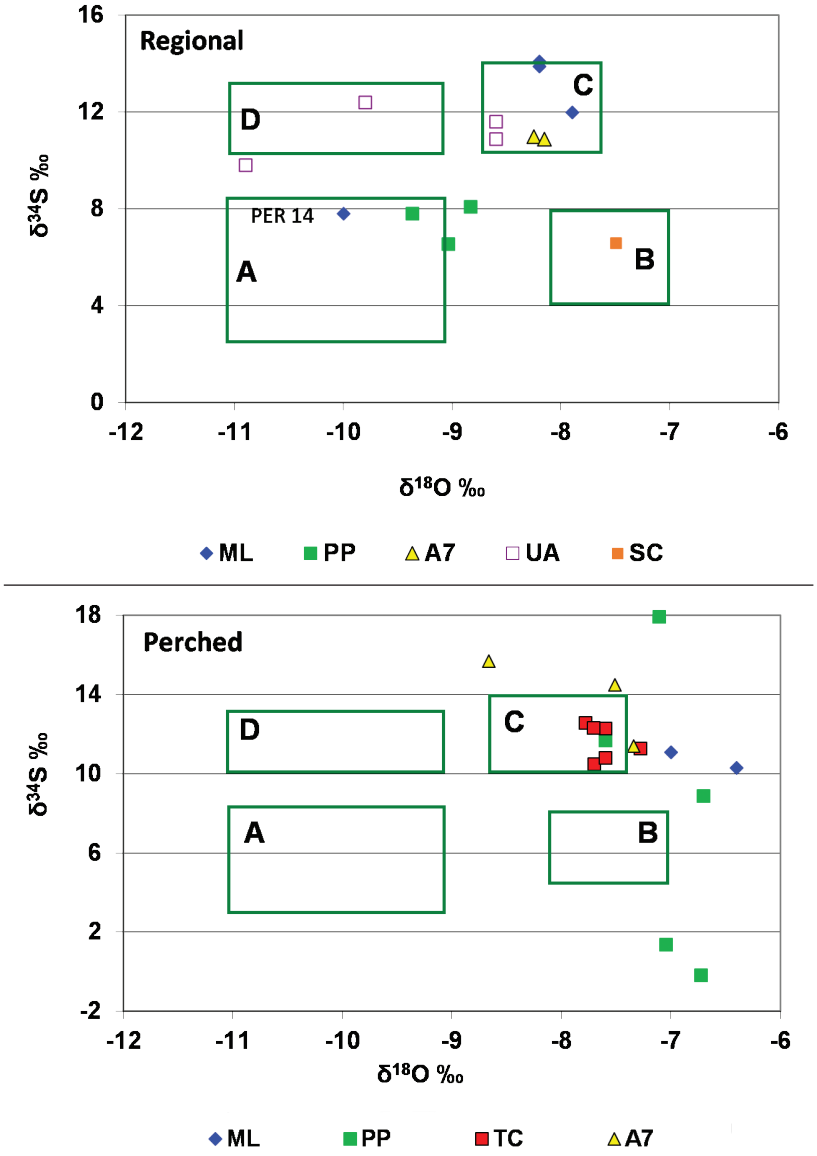

Figure 6. Stable S isotope (sulfate) and O isotope (water) data for the perched and regional aquifers in the study area. Fields A, B, C, and D correspond (with slight modification of field C) to basin-wide groundwater domains as defined in Eastoe et al. (2004).

needs of the few hundred residents of the Presidio inevitably left the Chuk Shon village with insufficient water (Sonnichsen 1982). The site of the presidio was close to El Ojito ("the little spring"), variously described as a spring, a well, and an artesian well (Officer 1987; Logan 2002), from which domestic water was drawn until the 1880s. This may originally have been a spring into which a well was excavated as the supply declined. The water supply was augmented from dug wells within the Presidio and the adjacent Mexican village. Water was encountered within 4 to 5 m of the surface, and it is said to have deteriorated in quality after a brief period of exploitation (Kupel 1986). Local tradition (e.g., Arizona Historical Society 2016) attributes the final drying of El Ojito and the shallow aquifer beneath the village to the 1887 earthquake with epicenter near Bavispe, Sonora. While acute hydrological phenomena correlating with the earthquake are plausibly documented elsewhere in southeastern Arizona, Dubois and Smith (1980) cited no textual reference earlier than Bennet (1977) as evidence of possible hydrological effects in Tucson.

Modest population growth in the latter half of the 19th Century and competition for water between the townspeople and farmers led to the initiation of engineering projects to enhance water supply from surface water and groundwater. Low dams were built on the Santa Cruz River in 1859 (Silver Lake) and 1870 (Warner's) to impound flow from the stream-bed springs supplying the Chuk Shon perennial reach (Figure 2); these provided power to flour mills, and controlled water for irrigated fields, mainly on the flood plain west of Tucson (Kupel 1986; Officer 1987, 289). The river bed adjacent to Tucson was excavated by Sam Hughes in 1887 in an attempt to recover shallow riparian groundwater (Logan 2002). Finally, as steam-driven pumps and drills became available, the community developed well-fields in order to exploit deeper groundwater. The first was about 1 km south of the town-site, and, in 1893, another was installed in the Santa Cruz flood plain 10 km south of the town (Kupel 1986; Logan 2002).

The excavation in the river bed by Sam Hughes was followed by a series of large floods in the late 1880s, and the excavation became the point of initiation of an entrenched river reach that grew to 29 km by 1920. Local tradition (e.g., Logan 2002) holds that the excavation was the principal cause of the entrenchment, which persists to the present. However, similar entrenchments began concurrently elsewhere in the southwestern USA (Bryan 1925) and a number of human-associated and climatic factors may have favored the regional entrenchment of watercourses (Betancourt 1990). The 1-5 m entrenchment along the Santa Cruz River permanently lowered the water table of the flood plain, destroyed the dams, and forced the construction of more complicated diversion canals for irrigation (Logan 2002).

Kupel (1986) outlined the subsequent history of groundwater use in Tucson, summarized briefly here. Artesian water was located at the foot of Sentinel Peak in 1896, but no further such discoveries followed near the town. Evolving pump technology put deeper groundwater resources within reach of the community, and provided it with a water supply through most of the 20th Century. By the 1960s, SWLs in the study area had fallen 15-30 m relative to pre-development levels (Anderson 1987). In the second half of the century, population growth became so rapid, and the demand for potable water so large, that the finite nature of the groundwater supply and the need for water conservation were recognized in civic deliberations. The advent of the Central Arizona Project, channeling Colorado River water to Tucson, enabled the city to provide an alternative water source, and to cease pumping from its central well field by the 1990s. The urban population, however, continues to increase (Figure 1).

Two Tucson place names might suggest a historical association with groundwater. Dunbar/Spring Neighborhood, immediately north of Downtown Tucson, is in fact named for pioneer residents Paul Dunbar and John Spring, and was not the site of a spring (Dunbar/Spring Neighborhood Association 2016). Flowing Wells Irrigation District, 7-8 km north of downtown, began as the Flowing Wells Water Company exploiting the artesian groundwater near Sentinel Peak, and later moved its operations to its present territory where no artesian wells are recorded (Kupel 1986).

Isotope Data

Data are listed in Table 1. Figure 6 shows new S and O isotope data for the perched and regional aquifers in the study area in relation to domain classification of Eastoe et al. (2004). Figure 7 shows O and H isotope data for the two aquifers. The interpretation of δD *vs*. $\delta^{18}O$ plots such as Figure 7 is based on the following general observations (more details can be found in Clark and Fritz 1997, and references therein). At most locations worldwide, annual averages of O and H isotopes in precipitation plot on a line of slope near 8, the Global Meteoric Water Line (GMWL). Partially evaporated meteoric waters plot to the right of the GMWL. Progressive evaporation of a particular sample of water yields residual water having isotope compositions plotting on a line with slope commonly between 3 and 5, depending on relative humidity.

Considering Figures 6 and 7 in conjunction, the following observations emerge:

Perched aquifer

Most samples plot within or very close to the domain C box in Figure 6, indicating the dominance

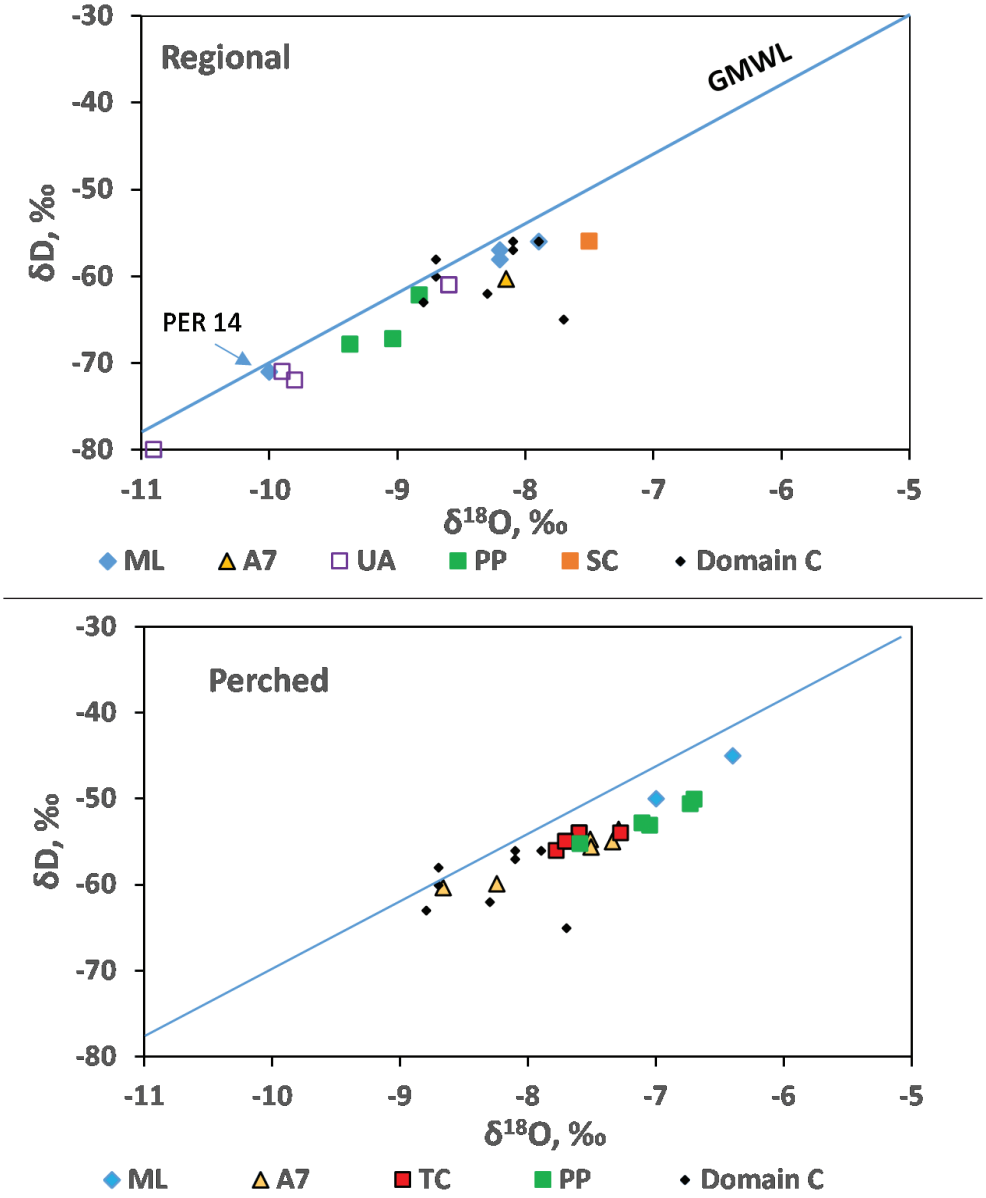

Figure 7. Stable O and H isotope data for the perched and regional aquifers. Data for domain C water from Eastoe et al. (2004).

of sulfate from domain C water. Outliers, from sites ML and PP, to the right of box C can be explained by reference to Figure 7, where the δD and $\delta^{18}O$ data plot on one or more evaporation trends of slope near 4.5; the outliers correspond to the more evaporated samples. The trend emerges from the field of domain C samples in parts of Tucson basin upgradient from the study area (data from Eastoe et al. 2004). Site PP has a very broad range of $\delta^{34}S$, 0 to +18‰. No water from domains A, B, and D was detected. The two samples from the ML site contained elevated tritium (15 and 40 TU) when sampled in 2003.

Regional Aquifer

The O and H isotope data (Figure 7) have domain C water as one end member, but in contrast to the perched aquifer, extend to more negative values typical of domains A and D. Domain C water is found at the UA, ML, and A7 sites. Domain D

Table 1. Well Data and Analyses.

Area Well (screen, mbgs)	ADWR Reg. No. 55-	Aquifer	Collar altitude masl	Depth mbgs	SWL masl	$\delta^{18}O$ water ‰	δD water ‰	$\delta^{34}S$ sulfate ‰	Tritium TU	^{14}C pMC	$\delta^{13}C$ DIC ‰	Corr. age Ka
ML (sampled 2003)												
UAM 1 (63-81)		RC	724.7	82.1	658.3	-8.2	-58	14.1	1.0			
UAM 2 (61-79)		RC	724.7	80.6	660.2	-8.2	-57	13.9	1.9			
PER 14 (143-159)		R	732.7	158.5	681.5	-10.0	-71	7.8	<0.6	15.7	-7.2	12
MLR-1 (79-105)		R	731.6	106.1	677.7	-7.9	-56	12.0	<0.7	20.1	-8.6	8
PEP-11		P	733.7	30.3	704.9	-6.4	-45	10.3	14.8			
PEP-19		P	731.7	30.3	702.9	-7.0	-50	11.1	39.6			
PP (sampled 2008)												
WR250A	535809	P	717	13.6	706.1	-7.1	-53	17.9				
WR251A	541080	P	717	13.6	706.1	-7.0	-53	1.4				
WR252A	542382	P	717	13.6	706.1	-7.6	-55	11.7				
WR269A	556250	P	717	15.2	706.4	-6.7	-50	8.9				
PPM529A	907396	P	717	17.3	701.8	-6.7	-51	-0.2				
WR248A	529533	R	717	42.4	683.7	-8.8	-62	8.1				
WR249A	529534	R	717	53.0	683.7	-9.4	-68	7.8				
WR271A	556258	R	717	42.4	682.2	-9.0	-67	6.6				
TC (sampled 2008)												
CEP-518A	906115	P	726	23.0		-7.7	-55	10.5				
CEP-519A	906116	P	726	21.8	707.5	-7.6	-54	10.8				
CEP-520A	906117	P	726	21.8	707.5	-7.6	-54	12.3				
HQUST531A	907928	P	726	22.4	706.9	-7.8	-56	12.6				
HQUST532A	907927	P	726	22.1	706.9	-7.7	-55	12.3				
HQUST533A	907929	P	726	21.8	707.5	-7.3	-54	11.3				

Table 1 Continued.

Area Well (screen, mbgs)	ADWR Reg. No. 55-	Aquifer	Collar altitude masl	Depth mbgs	SWL masl	$\delta^{18}O$ water ‰	δD water ‰	$\delta^{34}S$ sulfate ‰	Tritium TU	^{14}C pMC	$\delta^{13}C$ DIC ‰	Corr. age Ka
A7 (sampled 2007)												
1 MW-PD-6		P	722.1		700.8	-7.7	-55					
2 MW-PD-7		P	721.9		700.9	-7.3	-54					
3 MW-PD-31		P	715.9		698.6	-8.7	-60	15.7				
4 7AZP-6		P	722.5		697.9	-7.5	-55	14.5				
5 7AZP-7		P	722.8		699.6	-7.3	-55	11.4				
6 YC-6		P	718.9		699.1	-7.5	-56					
7 MW-PD-19 R		R	720.0		667.6	-8.2	-60	11.0				
8 7AZR-1 R		R	720.0		667.1	-8.2	-60	10.9				
UA (sampled 2006-2010)												
9 Park Av. (206-297)	217003	R	737	306.1		-10.9	-80	9.8		4.4	-8	19-20
10 Optical Sciences	201737	R	745	206.0	666.2	-9.8	-72	12.4	<0.5	9.7	-7.2	7-8
11 North cooling	528535	R	747	212.1	674.3	-9.9	-71		<0.9			
12 UMC South	618689	R	748	121.2	681.9	-8.6	-61	10.9	0.7			
13 UMC North	618688	R	743	139.4	697.8	-8.6	-61	11.6	<0.4	14.8	-8.6	11-12
SC (sampled 1999)												
14 Arizona School for Deaf and Blind	600785	R	712	127.3	681.7	-7.5	-57	6.6	11.1			

Explanation: No entries where data are unavailable

ADWR Reg. No. = Arizona Department of Water Resources registration number

mbgs = meters below ground surface

SWL = static water level

masl = meters above sea level

RC = regional, confined

TU = tritium units

R = regional

P = perched

water is found at the UA site, and domain A water at the ML and PP sites (Figure 6). Domain B water was encountered only in the flood plain of the Santa Cruz River. Available tritium measurements (tritium units, TU) are either low (0.7 to 1.9 TU) relative to the 5.2 TU average for Tucson rain (Eastoe et al. 2011) or below detection. Available ^{14}C measurements show low ^{14}C content, 4 to 15 percent modern carbon (pMC). Wells 10 and 13 at the UA site have corrected bulk residence times of 7-8 and 11-12 Ka. The Park Avenue well (9) at the UA site is 300 m deep, much deeper than other wells in the study area, and has unusually low $\delta^{18}O$ values (-10.9‰) for this part of the basin where most water has $\delta^{18}O$ values between -9 and -10‰ (Eastoe et al. 2004). The Park Avenue well (9) also has the lowest pMC measurement, 4.4, of the entire data set for Tucson basin, corresponding to bulk residence ages of 19 to 20 Ka.

Vertical Structure of Regional Aquifer

The ML site, with wells screened at four different depth ranges (Table 1), provides a view of the complexity of the regional aquifer in the area (Figure 8). The upper part of the aquifer, corresponding to screens 61-81 m below the surface, is essentially unconfined (the SWLs lying below those of the perched aquifer), and the samples contained tritium at 1.0 and 1.9 TU. Two deeper wells (MLR-1 and PER 14) intersect a confined, pressurized aquifer with SWLs higher than those of the overlying aquifer, and containing water with bulk residence ages of 8 Ka (screen 79-105 m below the surface) and 12 Ka (PER 14, screen 143-159 m below the surface). The unconfined aquifer and the upper part of the confined aquifer contain domain C water, while sample PER 14 is domain A water. The Park Avenue (9) well 1 km to the north, screened at 206-297 m below the surface, yields water close to domain D composition (Figure 6).

Discussion

Perched Aquifers

Drillers' logs indicate a complex interlayering of clay-rich and coarser clastic units within the Fort Lowell Formation beneath Downtown Tucson. While the continuity of individual clay units is not evident from the logs (Figure 4), the presence of a perched aquifer at 700 to 711 masl beneath most of the study area suggests that at least one such unit is continuous across the study area. This impermeable layer has prevented a variety of contaminants from infiltrating to greater depths below the study area.

Historical descriptions of shallow water beneath the Spanish Presidio in the late 18th Century imply the existence of a higher perched aquifer (Presidio aquifer, Figure 3) at that time, presumably atop one

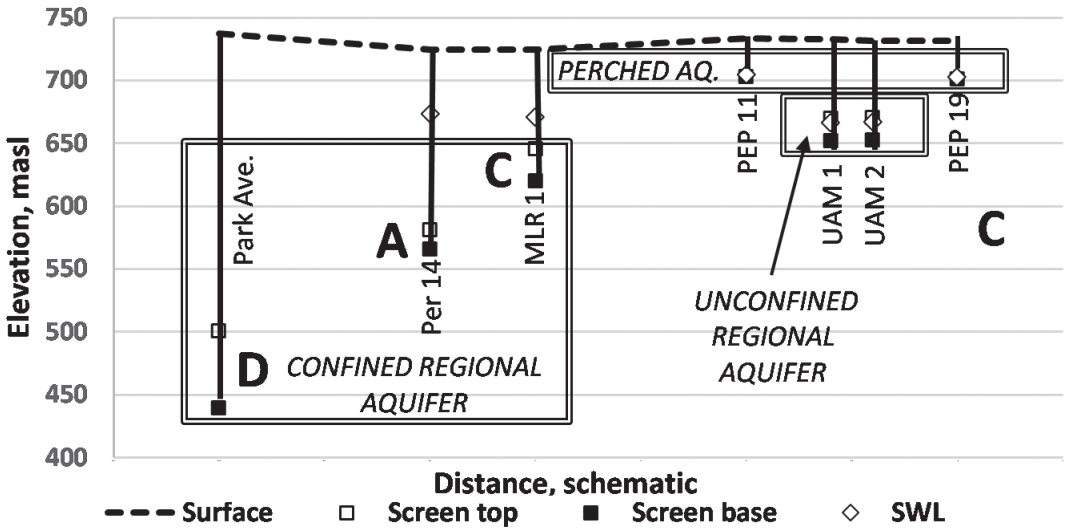

Figure 8. Schematic section of the perched and regional aquifers at the ML site and the Park Avenue (9) well. Bold vertical lines indicate wells. A, C, and D refer to water domain types, as described in the text.

of the shallow clay aquitards in the area (Figure 4). The aquifer was most likely recharged from Arroyo Chico to the east and north of the Presidio or from smaller washes that pass through the study area, and discharged at El Ojito spring. The Presidio aquifer dried up by the late 19th Century, an event that may have coincided roughly with the 1887 earthquake, but may equally have been the first negative consequence of human activities (over-exploitation, or deepening of wells through the aquitard beneath the aquifer).

The present-day perched aquifer receives recharge from Arroyo Chico at the ML site. The high tritium content at that site is either from bomb-spike tritium, 1960-1970, or from tritium released from the nearby American Atomics factory between 1970 and 1980 (Eastoe et al. 2011). The recharge is evaporated, and mixing of such water with domain C water generates the isotope trend in Figure 7. Domain C water originates from Cienega Creek in the southeastern part of the basin (Figure 5a) and is labeled by Permian marine sulfate that is not present in the study area (Eastoe et al. 2004). The presence of domain C water in the perched aquifer shows that the pre-development regional water table was above the present perched aquifer, and that what is now a perched aquifer was a part of the regional aquifer.

The broad range of $\delta^{34}S$ (-0.2 to +17.9‰) at site PP is unusual in Tucson basin, where most groundwater has values between +4 and +15‰ (Eastoe et al. 2004). The higher values may reflect partial sulfate reduction by microbes exploiting organic liquids as a carbon substrate. The lower values remain unexplained, but may be related to chemical contamination at the site, for example by sulfuric acid, which is commonly used as a paint remover and generally has a $\delta^{34}S$ value around 0 ‰.

Regional Aquifer

Prior to development, the regional aquifer beneath Downtown Tucson contained mainly domain C water, as indicated by residual water in the perched aquifer. In its present form, it is a set of unconfined and confined units containing, from upper to lower layers, water of domains C, A, and D. The decline in level of the regional aquifer has been due to basin-wide pumping, and presumably led to the isolation of the perched aquifer prior to the 15-30 m fall in water levels mapped beneath Downtown Tucson by Anderson (1988).

An updated version of the basin-wide water domain map for the regional aquifer (Figure 5b) shows that domain A water appears beneath the Downtown study area at sites PP and ML, remote from its main occurrence near Rillito Creek and its tributaries, but localized near the Santa Cruz fault. At site ML, the domain A water is ancient (Table 1). Upward discharge of ancient domain A water close to the fault also occurs elsewhere in Tucson basin (Eastoe et al. 2004), and appears to be a basin-scale phenomenon. Domain A water has also replaced domain D water outside the Downtown study area at several sites sampled in the early 1990s (Kalin 1994) and repeated in the early 2000s (Gu 2005). These changes have coincided with, and are most likely due to, basin-wide decline in the level of the regional aquifer (Gelt et al. 1999).

Role Of The Santa Cruz River

The original SWL of the regional aquifer was probably limited by the altitude of the pre-entrenchment channel of the Santa Cruz River, into which the aquifer most likely discharged. Eastoe et al. (2004) noted very limited eastward infiltration of domain B water from the Santa Cruz flood plain, but left open the possibility that the Downtown area, at that time unstudied, might form part of domain B. The present study shows that no infiltration from the flood plain has reached either the perched or the regional aquifer at the Downtown sampling sites. Prior to development, river water could not displace domain C water present at higher elevations. The present-day regional aquifer appears to have lower SWLs in the flood plain, where it is heavily pumped upgradient of the study area, than immediately to the east beneath Downtown, where it is not pumped (Figure 3). Infiltrating river water appears to remain in the relatively unconsolidated sediment of the flood plain, rather than moving laterally into the perched and regional aquifers.

Management Implications

The decline of the regional aquifer to levels below those of the present perched aquifer beneath Downtown Tucson has fortuitously isolated a variety of dangerous contaminants above an

aquitard, as noted above. This observation, while not part of the findings of this study, has clear implications for aquifer management in the Downtown area. Wells continue to be drilled in the area for water quality monitoring, or for supply of non-potable water (e.g., well 700375, Arizona Department of Water Resources 2016). New wells must be properly constructed, and old wells carefully decommissioned, so as to prevent downward leakage from the perched aquifer. Groundwater in the contaminated perched aquifer moves slowly (City of Tucson Environmental Services 2008). This study has identified a recharge zone of the perched aquifer along Arroyo Chico at the ML site, and shown that recharge since the separation of the perched and regional aquifers has been insufficient to date to flush old regional aquifer water from the perched aquifer. Such information may be of use in future management of the perched aquifer contaminants.

Conclusions

Since European settlement began in Tucson 240 years ago, groundwater conditions have deteriorated steadily. The small perched aquifer that supported the first settlement dried by the late 1800s. Water levels in the regional aquifer have fallen, in the first instance as a result of stream-bed entrenchment and then as a result of riparian groundwater pumping, leading to the loss of stream-bed springs. Evolution of pumping technology and rapid growth of population during the 20th Century led to accelerating declines in the regional aquifer, and the isolation of a second perched aquifer in the Downtown area.

New isotope data show that domain C groundwater of Eastoe et al. (2004) dominated the regional aquifer in the study area before the rapid decline in water level. Domain C water is ultimately derived from Cienega Creek to the southeast and is estimated to have taken 8 to 12 Ka to flow across Tucson basin to Downtown. Since its isolation from the regional aquifer, the perched aquifer has received a little recharge from Arroyo Chico, but insufficient to flush out residual domain C water. Samples from the lowered regional aquifer include mountain-derived groundwater (domains A and D), with the domain A water supplied by upward discharge along the Santa Cruz fault. No recharge from the Santa Cruz River has been detected.

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Application of Environmental Tracers in the Memphis Aquifer and Implication for Sustainability of Groundwater Resources in the Memphis Metropolitan Area, Tennessee

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Abstract: Fourteen years of results from environmental tracer sampling for ^{3}H, He isotopes, noble gases, and SF_6 in production wells screened in the Memphis aquifer in Memphis, Tennessee, are presented and used to determine the age and mass proportion of modern water (<60 years old) pumped from this regionally important public supply aquifer. The results indicate persistent presence of modern water in water pumped from one or more production wells in most of the well fields sampled. The percentage of modern water is quantified by three methods: tritium-loading history, inverse geochemical modeling, and lumped parameter modeling. The mixing percentages and ages determined from each technique are generally in agreement, but also emphasize the unique information provided by each tracer and associated modeling approaches. The implications of the tracer data for sustainability and vulnerability are also considered, especially in regard to potential water quality threats to the water supply in the Memphis, Tennessee, metropolitan area.

Keywords: *aquifer, environmental tracers, modern water, public supply, sustainability, vulnerability*

Assessing sustainability of groundwater resources in urban areas presents challenges in regard to understanding hydrologic processes, water use, and vulnerability of water resources to contamination or loss of water quality. This contribution presents a case study of application of environmental tracers to assessing sustainability of groundwater resources in Memphis, Tennessee, U.S.A. The Memphis metropolitan area has a population of over 1 million people, for which all municipal and industrial water is pumped from groundwater resources, mostly from the Eocene Memphis aquifer (Figure 1). The Memphis aquifer is a semi-confined unconsolidated sand aquifer, partially confined by a silt and clay confining unit, which is in turn overlain by a sand and gravel shallow aquifer that is in direct hydrologic connection to surface water and near-surface contamination sources (Parks and Carmichael 1990; Parks 1990).

Water production began from the Memphis aquifer in 1886 and increased until 2005 when production peaked at 829,000 m^3/d (Kenny et al. 2009). Groundwater from the Memphis aquifer is a low total dissolved solids (32 to 333 mg/L, median 83 mg/L), mixed cation-bicarbonate water (Brahana et al. 1987), desirable for public supply and industrial applications in chemical manufacturing and food processing. Memphis enjoys some of the lowest municipal and industrial water costs in the United States (Memphis Light, Gas, and Water online), making its water supply a significant asset for economic development. In 2005, the State of Mississippi filed an action lawsuit against the City of Memphis and the major utility, Memphis Light, Gas, and Water (MLGW), in Tennessee, claiming that groundwater withdrawal from the Memphis aquifer by MLGW had caused diversion of groundwater from beneath Mississippi into Tennessee (Hood v. City

of Memphis 2009; Waldron and Larsen 2014). Although the 2005 case was dismissed without prejudice by the U.S. Supreme Court, the State of Mississippi has filed a subsequent lawsuit against the State of Tennessee on similar grounds, which is currently in litigation (State of Mississippi v. State of Tennessee 2015). Recognizing the economic benefit of groundwater resources of the Memphis aquifer as well as the vulnerability of the aquifer resources to legal apportionment, Mayor of Memphis, A.C. Wharton, created a community task force (Blue Stream Task Force - BSTF) in 2014 to determine how to protect, sustain, and best utilize groundwater resources in the Memphis area.

Assessment of sustainability is critical for determining long-term production rates of groundwater from an aquifer. Although sustainability can be equated with safe yield (amount that can be pumped regularly without dangerous depletion of the storage reserve) from an aquifer (Fetter 2001), aspects of water quality may present additional considerations. Considering the specific conditions of groundwater production from the Memphis aquifer in the Memphis area, the BSTF adopted the following operational definition of sustainability: *"the use, development and protection of the groundwater resource at a rate and manner that provides for current and projected needs without damage to*

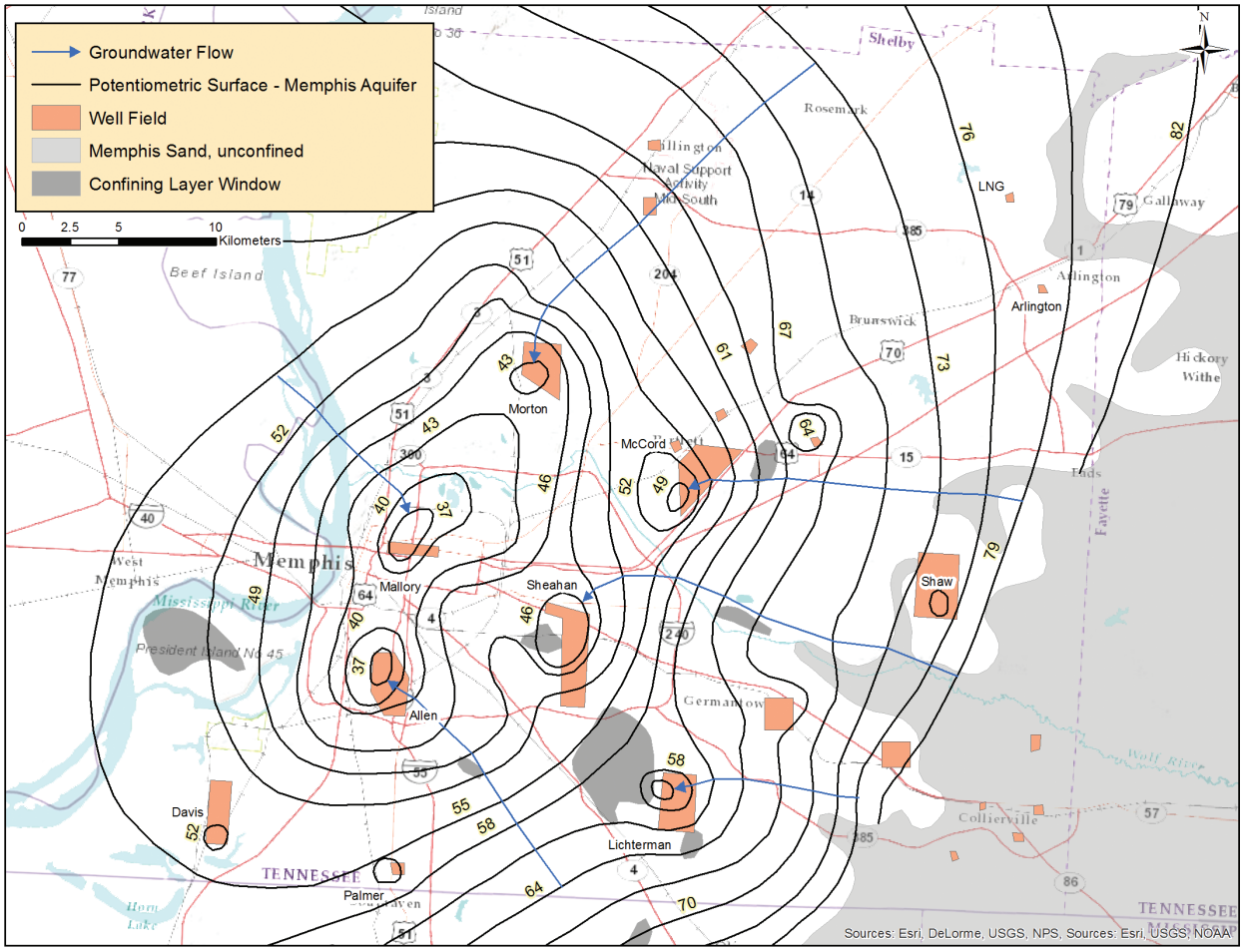

Figure 1. Study area in Memphis and Shelby County, Tennessee. Map shows Memphis and surrounding municipalities, 1995 potentiometric surface for the Memphis aquifer (Kingsbury 1996) (m above sea level, asl), groundwater flow direction, and the locations of public supply well fields for Memphis Light, Gas, and Water (MLGW), Bartlett, Collierville, Germantown, and Millington. Also shown are windows through the upper Claiborne confining unit and the Memphis Sand exposure area as mapped by Parks (1990).

the groundwater system or jeopardizing the ability to meet water needs in the future."

Pertinent aspects of the definition for the present work include determining a rate of production that will not "*damage the groundwater system*," which is interpreted as draining of the aquifer and inducing irreversible compaction or loss of storage, and not jeopardize *"the ability to meet water needs in the future*," which is interpreted as preserving water quality as well as capacity of the aquifer. Thus, the definition of sustainability applied to the Memphis aquifer appeals to balancing production rates while maintaining the desirable characteristics of the aquifer and its stored water resources. Groundwater flow modeling is commonly used to assess various production scenarios assuming known and optimized parameters of the groundwater system. Critical to such modeling endeavors is evaluation of leakage in or out of an aquifer from adjacent aquifers or confining units. In situations where leakage may derive from near-surface water sources, the quantity of leakage also has implications for changes in water quality as well as vulnerability of aquifer resources to contamination.

Environmental tracers are commonly applied to groundwater problems to assess recharge pathways and sources of water in aquifer systems (Cook and Böhlke 2000; Darling et al. 2012; Jurgens et al. 2014; Hagedorn 2015). Environmental tracers, such as chlorofluorocarbons, $^{3}H/^{3}He$, SF_6, and noble gases, are used to determine when groundwater obtained from wells was last exposed to the atmosphere, thus providing a groundwater age based on the atmospheric loading history and, for ^{3}H, radioactive decay. Although the systematics of the isotopic and chemical systems used for environmental tracing are subject to errors and assumptions of application, the resulting data provide independent constraints on both conceptual and numerical models of groundwater flow in aquifer systems. Environmental tracers, specifically combinations of inorganic chemicals, ^{3}H, ^{3}He, and SF_6, have been applied to groundwater systems in the Memphis area with success (Graham and Parks 1986; Parks et al. 1995; Larsen et al. 2003; Gentry et al. 2005; Gentry et al. 2006; Ivey et al. 2008; Koban et al. 2011; Barlow et al. 2012; Larsen et al. 2013), especially in identifying modern water in aquifer systems and leakage pathways between the Shallow and Memphis aquifers. Of particular importance for assessing sustainability, environmental tracers have been used to determine proportions of production well water that have recently been in contact with the atmosphere (modern water) and the likely sources for those modern waters (Larsen et al. 2003; Koban et al. 2011; Larsen et al. 2013).

In this paper we review the hydrogeology of the Memphis aquifer, discuss the application of the $^{3}H/^{3}He$ and SF_6 tracer systems to the Memphis aquifer, explore the results of 14 years of environmental tracer sampling of production wells in Shelby County, and discuss the implications of these data for groundwater sustainability, vulnerability, and policy.

Hydrogeologic Background

Memphis lies within the northern Mississippi embayment, a trough-shaped basin that plunges southward along an axis approximately parallel to the Mississippi River. The Mississippi embayment is filled with more than a thousand meters of sediments of Cretaceous, Tertiary, and Quaternary age (Moore and Brown 1969; Graham and Parks 1986). The early to mid-Tertiary-age geologic units beneath Shelby County dip gently to the west-northwest and include unconsolidated sand, silt, and clay with minor lignite laid down in a Coastal Plain setting. Pleistocene and Pliocene sand and gravel fluvial-terrace deposits, 0 to 20 m thick, unconformably overlie the mid-Tertiary units (Krinitsky 1949; Saucier 1987; Van Arsdale et al. 2008). Excluding the present-day tributary valleys, 3 to 20 m of loess (windblown silt and clay) overlie the fluvial deposits and mid-Tertiary sediments, and mantle the upland topography. Western Tennessee tributary valleys contain late Pleistocene to Holocene alluvium generally comprising as much as 15 m of sand and gravel capped by organic-rich silty sand and silty clay (Carmichael et al. 1997).

The geologic units beneath the Memphis and Shelby County area (Figure 1) are divided into a series of hydrostratigraphic units (Table 1). The loess and the upper part of the alluvium have similar textural and hydraulic properties (1.2×10^{-4} to

Table 1. Geologic and hydrostratigraphic units underlying Shelby County, Tennessee. (Adapted from Graham and Parks (1986) with modifications from Waldron et al. (2011)).

Series	Group	Stratigraphic Unit	Hydrostratigraphic Unit	Thickness (m)	Lithology
Holocene and Pleistocene		Alluvium	Shallow (Alluvial) aquifer	0-53	Sand, gravel, silt, and clay. Underlies the Mississippi alluvial plain and alluvial plains of tributary streams in western Tennessee. Thickest beneath the alluvial plain, where commonly between 30.5 and 45.7 meters thick; generally less than 15.2 meters thick elsewhere.
Pleistocene		Loess	Leaky confining unit	0-20	Silt, silty clay, and minor sand. Principal unit at the surface in upland areas of western Tennessee. Thickest on the bluffs that border the Mississippi alluvial plain; thinner eastward from the bluffs.
Pleistocene and Pliocene (?)		Fluvial terrace deposits	Shallow (Fluvial) aquifer	0-30	Sand, gravel, minor clay, and ferruginous sandstone. Generally underlies the loess in upland areas, but locally absent. Thickness varies greatly because of erosional surfaces at top and base.
Eocene	Claiborne	Jackson Formation Cockfield and Cook Mountain Formations	Upper Claiborne confining layer	0-110	Clay, silt, sand, and lignite. Because of similarities in lithology, the Jackson Formation and upper part of the Claiborne Group cannot be reliably subdivided based on available information. Most of the preserved sequence is the Cockfield and Cook Mountain Formations undivided.
		Memphis Sand	Memphis aquifer	152-271	Sand, clay, and minor lignite. Thick body of sand with lenses of clay at various stratigraphic horizons and minor lignite. Thickest in the southwestern part of the Memphis area; thinnest in the northeastern part.
Eocene?	Wilcox	Flour Island Formation	Flour Island confining layer	49-94	Clay, silt, sand, and lignite. Consists primarily of silty clays and sandy silts with lenses and interbedded fine sand and lignite.
Paleocene		Fort Pillow Sand	Fort Pillow aquifer	38-93	Sand with minor clay and lignite. Sand is fine to medium. Thickest in the southwestern part of the Memphis area; thinnest in the northern and northeastern parts.
	Midway	Old Breastworks Formation	Old Breastworks confining layer	55-107	Clay, Silt, Sand, and lignite. Consists primarily of silty clays and clayey silts with lenses and interbedded fine sand and lignite.

2.1×10^{-3} m/day) (Robinson et al. 1997) and behave as a leaky confining unit, limiting direct infiltration into the underlying sediments. The fluvial-terrace deposits and lower part of the alluvium form the Shallow aquifer (Graham and Parks 1986). The Cockfield and Cook Mountain formations underlie the Shallow aquifer and form the upper confining unit, termed the upper Claiborne confining unit, for the Memphis aquifer. The clay beds in the confining unit range in thickness from 1 to 61 m

and have hydraulic conductivities of 1.5×10^{-6} to 3.0×10^{-4} m/day (Parks 1990; Robinson et al. 1997; Gentry et al. 2006). The Memphis aquifer is a sand-dominated aquifer, ranging in thickness from 122 to 274 m with an average transmissivity of 3.25×10^{3} m^2/day (Parks and Carmichael 1990; Brahana and Broshears 2001) or average hydraulic conductivity of 14.9 m/day (Waldron et al. 2011). The Flour Island Formation forms the lower confining layer for the Memphis aquifer and upper confining unit for the Fort Pillow aquifer, another water resource for the region.

Waldron and Larsen (2014) used historical water level data to reconstruct the predevelopment (prior to 1906) water table and potentiometric surface for the Memphis aquifer. Early studies suggested that the primary source of recharge to the Memphis aquifer was infiltration in upland regions, east and southeast of Memphis (Hosman et al. 1968), including the southeastern part of Shelby County (Criner et al. 1964). The hydraulic head in the aquifer was high enough to create flowing artesian conditions in downtown Memphis during the early years of Memphis aquifer water development. However, flowing artesian conditions soon dissipated and groundwater exploitation between 1886 and 2008 led to significant drawdown within the Memphis aquifer in Shelby and surrounding counties. Thus, prior to groundwater development, a strong upward vertical gradient existed between the Memphis aquifer and the overlying Shallow aquifer; however, groundwater pumping has reversed the gradient such that a downward vertical gradient exists between the Shallow and Memphis aquifers.

Pumping from the Memphis aquifer increased somewhat steadily from 1886 until 2005, but has declined in recent years, in part due to closure of a large food production facility in Memphis (MLGW, personal communication 2015). Pumpage from the Memphis aquifer in Shelby County increased from 0 to 150 million liters per day (MLD) during initial development from 1886 to 1920; however, pumping increased from 1920 to 1975 to reach 720 MLD (Criner and Parks 1976). Pumping remained at approximately 700 to 720 MLD from 1975 to 2005 (Kenny et al. 2009), but dropped to 654 MGL in 2010 (USGS, unpublished data). Long-term data from five observation wells show steady declines in water level until approximately 1975; after that date average yearly water levels have remained largely unchanged except in newly developed areas in the eastern part of Shelby County (Kingsbury 1996).

As groundwater usage increased in the Memphis area and urban growth extended east and southeast from the center of Memphis, evidence for vertical leakage (recharge) through the upper Claiborne confining unit arose from numerous studies (Criner and Armstrong 1958; Criner et al. 1964; Nyman 1965; Criner and Parks 1976; Parks 1990). Groundwater flow modeling of the Memphis aquifer in the Memphis area (Brahana and Broshears 2001) and regionally (Grubb 1998) suggest that substantial vertical leakage is required to accommodate the groundwater withdrawals in southwestern Tennessee. Water-table maps in Shelby County in 1988 (Parks 1990), 2005 (Waldron et al. in press), and 2015 (unpublished data), all show multiple anomalous depressions in the water-level contours, suggesting that leakage, in part, originates from cross-formational flow from the Shallow aquifer. No production wells exist in the Shallow aquifer in the areas of the anomalous depressions; thus, the depressions must originate from another mode of water loss from storage.

Environmental tracer studies have been conducted to establish groundwater age relationships and, specifically, the presence and quantity of modern water (<60 years old) in the Memphis aquifer. Initial studies utilized ^{14}C to demonstrate the dominance of older water (2000 - 3000 years old) in the Memphis aquifer and ^{3}H, which showed some wells where younger water was entering the Memphis aquifer in Shelby County (Graham and Parks 1986). Subsequent hydrogeologic studies showed the presence of numerous hydrogeologic windows in the upper Claiborne confining unit in Shelby County, which allow vertical recharge from the Shallow aquifer or surface water sources to the Memphis aquifer (Parks 1990; Bradley 1991). Hydrogeologic windows are intervals in the upper Claiborne confining unit that are thin due to erosion (e.g., Bradley 1991) or comprise predominantly sand facies rather than silt and clay (e.g., Larsen et al. 2003), such that water may be exchanged between the Shallow and Memphis aquifers. More recently, $^{3}H/^{3}He$ and geochemical data from production wells have shown a clear association of modern water in the

Memphis aquifer with proximity to hydrogeologic windows (Larsen et al. 2003; Gentry et al. 2005; Ivey et al. 2008; Koban et al. 2011; Larsen et al. 2013). Several of these studies using tracers and geochemical mixing models showed that as much as 75% of the water produced from some production wells is modern water, likely from a nearby surface water source (Larsen et al. 2003; Koban et al. 2011; Larsen et al. 2013; Gallo 2015).

The previous studies support a general conceptual model for groundwater flow in the Memphis area in which, prior to development, groundwater recharged into the Memphis aquifer in the hilly exposure belt of the Memphis Sand, 40 to 120 km southeast of the city of Memphis (Brahana and Broshears 2001). Groundwater flow was southeast to northwest toward the city of Memphis and supported flowing, artesian well conditions during the early years of water production from the aquifer in the late 1800's (Waldron and Larsen 2014). This groundwater is relatively old, with residence times ranging from about 1,000 to 3,000 years old based on ^{14}C data (data in Graham and Parks 1986). Following large-scale municipal water-supply development of the Memphis aquifer in the Memphis area, the potentiometric level dropped initially below the land surface and then, with increased pumping, below the base of the Shallow aquifer in the 1960's (Criner and Parks 1976). The drop in the potentiometric surface of the Memphis aquifer created optimal conditions for leakage of young, modern water from storage in the Shallow aquifer and infiltrated stream water through hydrogeologic windows where the upper Claiborne confining unit is thin or absent (Parks 1990). Larsen et al. (2003; 2013), Parks et al. (1995), Bradshaw (2011), Koban et al. (2011), and Gallo (2015) emphasize the importance of saturated alluvium in stream valleys as the long-term water source for leakage into the Memphis aquifer. Furthermore, evidence indicates that pollutants from industrial sites, landfills, and other sites near hydrogeologic windows have entered the Memphis aquifer (Parks 1990; Bradley 1991; Parks and Mirecki 1992; Gentry et al. 2005) and clarifies the need to understand the quantities and pathways by which modern water enters into the Memphis aquifer in the Memphis urban area.

Methods

Sampling

Selected production wells screened in the Memphis or Fort Pillow aquifer in the city of Memphis and Shelby County were sampled between 1999 and 2013. Typical production well design includes a stainless steel casing from the land surface to approximately 100 m depth or more and a screened interval 30.5 m long with a diameter of 0.3 m. Initially, most wells were sampled only for ^{3}H and major solute geochemistry to identify wells that had tritium contents greater than 0.2 tritium units (TU). Wells with ^{3}H contents above this threshold are likely to have a component of modern water (Clark and Fritz 1997), especially considering tritium decay over 1999 to 2013. Starting in 2000, one or more production wells were sampled each year or every other year for major solute geochemistry, ^{3}H, He isotopes, and noble gases. Due to decay of bomb-peak ^{3}H in near-surface groundwater reservoirs and associated loss of tracer utility, sampling of SF_6 was added as a tracer in 2009. Approximately five to seven production wells were sampled for tracers each year; however, no production well sampling was conducted during 2010, 2012, and 2014 due to budgetary limitations. Efforts were made during several of the sampling years to emphasize a specific well field; for example, during 2009 only wells in MLGW Allen well field were sampled. During other sampling years, for example 2011, wells from multiple well fields were sampled, sometimes including wells that had been sampled in previous years.

Water samples for geochemistry, ^{3}H, and SF_6 analyses were obtained from a sampling spigot located on the top of the production wells. Sampling generally commenced after the wells had been pumping for more than twelve hours; however, wells for the smaller utilities in Shelby County were pumped for less than three hours. Given that the pump discharge is approximately 4 m^3/min. and the well volume is generally less than 11 m^3, approximately 1,900 well volumes would be purged prior to sampling for a twelve hour purge period. Field chemistry measurements were made using calibrated probes in a YSI 6600 multi-parameter unit with a flow-through sampling cell:

pH, temperature, specific conductance, dissolved oxygen, and oxidation-reduction potential. In addition, field measurements were made in duplicate for alkalinity (using Hach titration kit) and dissolved oxygen content (using a factory-calibrated Hach 2000 spectrophotometer if probe-measured dissolved oxygen was less than 1 mg/L). As many as four water samples were obtained: 250 ml of raw water in a polypropylene bottle, 250 ml of filtered (0.45 micron filter) water acidified with 1 ml of concentrated ACS-certified nitric acid in a polypropylene bottle, 1 liter of raw water in an amber glass bottle for ^{3}H, and two 1-liter amber glass bottles for SF_6. U.S. Geological Survey protocols were followed for SF_6 sampling (Amber glass bottles were sealed with electrical tape as an added precaution). After collection, waters were stored in a cooler with ice until they were taken to the laboratory and placed in a refrigerator at 4°C.

Dissolved gas sampling required restricting the flow from the pump to the treatment plant to maintain a pressure of 40 to 45 (psi) and, thus, limit degassing during pumping. The gases were sampled in a flow-through cell in duplicate with a diffusion sampler (copper tube with a semi-permeable membrane attached to one end). The diffusion sampler was equilibrated with the flow of production well water for approximately 24 hours, although equilibration was as little as three hours for some of the smaller utilities in Shelby County. Upon retrieval, the copper tube portion of the sampler was sealed with a cold-weld crimper, tips of the tubes were sealed with electrical tape, and tubes were stored for later laboratory analysis. The total gas pressure in the water was measured using a calibrated probe and meter. The raw water samples in the amber glass bottles for ^{3}H and SF_6 and diffusion samplers for noble gas analysis were packed in a cooler and shipped to the Dissolved Gas Laboratory at the University of Utah.

^{3}H Analysis

Tritium activity is determined by the amount of ^{3}He accumulated in a sample over a series of three to four months (Clarke et al. 1976; Bayer et al. 1989). The practical detection limit for the helium in-growth method is 0.05 TU (Solomon and Cook 2000). Under typical conditions, the percent error for lab analysis of Tritium ranges from approximately ± 10% at 1 TU to ± 70% at the detection limit (Alan Rigby, personal communication 2006).

Dissolved Gas Analysis

Samples were analyzed in the Dissolved Gas Laboratory at the University of Utah using the method of Bayer et al. (1989). Reactive gases were removed using a SAES getter (degassing device) and heavy noble gases were sorbed onto charcoal at –180 ˚C for Ar, Kr, and Xe, and at –236 ˚C for Ne. Helium and other noble gas isotopes were then measured on a MAP 215-50 mass spectrometer to a precision of approximately ±0.5%.

Helium-3 dissolved in groundwater arises from four sources: atmospheric equilibration, ^{3}H decay, shallow subsurface nuclear reactions, and the mantle (Solomon and Cook 2000). The atmospheric component can be corrected for by using an excess air model to determine recharge temperature and then applying the appropriate gas-water equilibrium constants. Noble gas composition was used to determine the recharge temperature by applying a modified form of the closed-equilibrium model developed by Aeschbach-Hertig et al. (2000). Shallow subsurface nuclear reactions generate ^{3}He from fission of ^{6}Li produced during U- and Th-series decay. The resulting $^{3}He/^{4}He$ ratio from such reactions is approximately 1×10^{-8}, which is two orders of magnitude less than that of the atmosphere (1.36×10^{-6} at 10°C). Relatively large amounts of ^{4}He from shallow subsurface reactions must be present for this to be an important factor. Nonetheless, ^{3}He from shallow subsurface reactions are accounted for using $^{3}He/^{4}He$ ratio of terrigenic (crustal–produced) helium and using neon to correct for atmospheric ^{4}He contributions. Mantle helium has a $^{3}He/^{4}He$ ratio approximately ten times greater than that of atmospheric helium; thus, small contributions of mantle helium may be difficult to detect. Given the shallow groundwater flow paths inferred by groundwater flow modeling in the Memphis area (Brahana and Broshears 2001) and the cratonic tectonic setting, mantle-sourced helium is not likely to contribute significantly to the helium present in shallow Memphis aquifer waters.

The $^3H/^3He$ age of a groundwater sample is defined as:

$$t = \lambda^{-1} \times \ln(^3He_t/^3H+1) \quad \text{(Equation 1)}$$

where λ is the tritium decay constant, ^3He_t is tritiogenic ^3He, and t is the age in years of the modern (<60 years old) component of the water. Tritiogenic helium is calculated using Ne to correct for the excess atmospheric gas entrained during percolation of water through the vadose zone (Solomon and Cook 2000). The ratio R/R_a measures how closely the measured helium-isotope ratio is to that of air-saturated water. The R/R_a values approach 1 for young apparent age values and are progressively higher for older samples, which is consistent with domination of atmospheric helium contributions. Considering that all possible measures were taken to minimize error and gas loss, it is reasonable to assume, based on past sampling experience and replicate analyses, that the sum of the errors is less than 10% of the age determination.

SF_6 Analysis

SF_6 is analyzed by GC-ECD techniques to a precision of 1-3% at concentrations in excess of 1.0 fg/L to a maximum reporting level of 3.0 pg/L (Wanninkhof and Ledwell 1991; Law et al. 1994). This precision and detection limit permit groundwater dating applications from about 1970 to the present (Busenberg and Plummer 2000). Calculation of groundwater age uses the observed SF_6 concentration, recharge temperature obtained from the noble gas concentrations, recharge elevation approximated for the production well, and recharge salinity of 0 ‰. The partial pressure of SF_6 in groundwater is calculated based on Henry's Law solubility at a given temperature and then used to identify the year of recharge based on measured atmospheric compositions in North America (Busenberg and Plummer 2000). Measured SF_6 values beyond reasonable values for equilibration with atmospheric SF_6 concentrations are considered contaminated and excluded from age calculations. The precision of groundwater age calculated from SF_6 data depends on the analytical errors (generally ± 5%), error in the estimated recharge temperature (± 2°C = 1 to 3 years age difference), and error due to residence time in the unsaturated zone (as much as 8 to 12 years for unsaturated zone thickness of 30 m) (Busenberg and Plummer 2000).

Calculation of the Fraction of Modern Water

Three approaches are used to estimate the fraction of modern water in a sample: tritium loading, lumped parameter modeling, and inverse geochemical modeling. The tritium loading model is similar to that of Manning et al. (2005) in that the ground-water apparent age is used to determine a loading date, assuming that the modern water forms a single-age mode (a mass distribution of modern water centered about a single age). The mean monthly ^3H load for a given year was determined using the synthetic monthly ^3H values in precipitation determined by Ivey et al. (2008), which were calculated for the years 1953-1987 by spatial correlation of measured monthly ^3H values at locations outside the region, and approximated for years subsequent to 1987 using values from Hatteras, N.C. (Ivey et al. 2008). The mixing ratio is then determined by:

$$\%\ \text{modern water} = ((^3H+^3He_t))/((^3H\ load_{yr}))\times 100 \quad \text{(Equation 2)}$$

where ^3He_t is the tritiogenic, ^3He is determined using the noble gas data, and ^3H is the measured tritium activity in the samples. The recharge year is obtained by subtracting the apparent age of the modern water from the sampling year. The ^3H load from the recharge year (^3H $load_{yr}$) is determined from the compiled tritium loading history in Memphis (Ivey et al. 2008). The standard deviation of the ^3H $load_{yr}$ for the recharge year was propagated through the mixing calculations to estimate the uncertainty in the mixing ratio. The approach to uncertainty estimation is different than that of Manning et al. (2005) in that they evaluated uncertainty both by considering hydrodynamic dispersion along a flow path and by considering the maximum deviation of measured monthly ^3H values from the mean ^3H value in precipitation. Despite the difference in approach, the resulting magnitude of the uncertainty is similar (Koban et al. 2011).

Lumped parameter modeling (LPM) has been applied to the interpretation of environmental tracer data since the 1950's (see review in Jurgens

et al. 2014). These models represent simplified aquifer settings and groundwater flow dynamics, accounting for advective transport, dispersion during groundwater flow, and fluid mixing in the borehole. Comparative analysis of LPM and numerical groundwater modeling by Eberts et al. (2012) and Hagedorn et al. (2011) illustrate that apparent or piston-flow model ages from tracers can yield misleading descriptions of the age distribution of waters from some wells, especially public-supply wells with long well screens. Eberts et al. (2012) also show that the age distributions provided by application of an appropriate LPM for the geometry of the groundwater flow system can yield similar results to those from numerical groundwater flow modeling of the system. Models most appropriate for public-supply wells with long screened intervals include exponential piston flow and dispersion models (Ivey et al. 2008; Eberts et al. 2012). Alternatively, binary mixing models of discretely younger and older waters may be best modeled using a binary mixing model describing age distributions of waters flowing from two sources (Jurgens et al. 2014). Various LPM scenarios can be applied to a set of tracer data using the TracerLPM Excel worksheet (Jurgens et al. 2014), which allows for modeling of the age-distributions and percentages of binary mixing components. The worksheet requires applying a conceptual model that best fits the recharge conditions in the study area and iteratively solving the LPM expressions until a minimum relative error is obtained between model results and observed data.

Geochemical modeling is accomplished using PHREEQCi (Parkhurst and Appelo 1999), a speciation and reaction path modeling software package. Inverse geochemical modeling uses chemical species from initial and final water in a hydrologic flow system to predict mixing and mineral and gas transfer reactions in the system. Application of this approach evaluating mixing of modern and older waters is specific to individual wells and depends on the existence of significant chemical differences between the modern water component and the old water component. Successful applications of this approach have been used in several Memphis area well fields where modern water is leaking into the Memphis aquifer and mixing with older Memphis aquifer water (Parks et al. 1995; Larsen et al. 2003; Koban et al. 2011; Larsen et al. 2013; Gallo 2015).

Results

The environmental tracer data and apparent ages from production well waters in Shelby County, Tennessee from 2000 to 2013 are tabulated in Table 2. The data are organized by well field, year of sampling, and sampling date. In addition to samples from Memphis aquifer production wells, ^{3}H data from three wells screened in the Fort Pillow aquifer are presented. Supporting noble gas data are available from the primary author upon request.

Duplicate ^{3}H, $^{3}H/^{3}He$, and SF_6 samples were obtained during multiple sampling years to evaluate data consistency. Tritium duplicate values were generally within 0.10 TU; except when values were near the method detection limit (0.05 TU) and for MLGW 321, sampled in 2006, which had erratic pumping conditions during sampling. Duplicate $^{3}H/^{3}He$ age determinations in MLGW 203A in 2005 yielded age estimates within three years (31.9 and 29.3 years). Duplicate SF_6 age determinations in MLGW 708 in 2011 yielded age estimates within three years (26.8 and 24.3 years).

Tritium contents in excess of 0.2 TU (twice the duplicate activity of 0.1 TU) indicate a component of modern water in the upper production zone of the Memphis aquifer at 100±50 m depth in many well fields in the Memphis area (Figure 2). Production wells sampled from the Fort Pillow aquifer at depths of 320-340 m typically have ^{3}H values near the detection limit except for one well in Millington, which had a value of 0.27 TU. Many of the production wells with ^{3}H content in excess of 0.5 TU were subsequently sampled for $^{3}H/^{3}He$ and yielded apparent ages ranging from 12 to 50 years prior to the sampling date (Table 2). Sampling for SF_6 in production wells post-2009 yielded apparent ages from 6 to 40 years prior to the sampling date; however, these dates do not correspond well with $^{3}H/^{3}He$ apparent ages obtained from the same sampling event (Figure 3).

Table 2. Environmental tracer data and apparent ages.

Well field	Production well	Aquifer	Sample date	Top of screen (m)	R/Ra	^{3}H (TU)	^{3}He* (TU)	Age using Ne (yr)	SF_6 (fMol/L)	SF_6 age-piston flow (yr)
Allen	111A	Memphis	8/11/2004	87.2	1.10	0.64	4.20	36.4	nd	nd
	123	Memphis	8/11/2004	96.6	1.01	0.73	3.40	31.2	nd	nd
	109	Memphis	9/27/2006	96.6	1.08	0.48	4.90	43.4	nd	nd
	111A	Memphis	9/22/2006	87.2	1.08	0.97	2.60	23.3	nd	nd
	107	Memphis	9/22/2006	89.9	1.19	2.05	6.10	24.8	nd	nd
	109	Memphis	11/9/2007	96.6	1.04	0.60	1.70	25.2	nd	nd
	102	Memphis	11/12/2008	89.0	0.96	0.08	<0	nd	nd	nd
	111A	Memphis	11/12/2008	87.2	1.12	0.86	2.90	26.5	nd	nd
	114	Memphis	12/8/2008	90.5	1.46	2.10	13.6	36.1	nd	nd
	117	Memphis	12/8/2008	86.9	0.96	0.08	<0	nd	nd	nd
	103A	Memphis	12/11/2008	82.9	0.99	0.08	0.20	22.3	nd	nd
	114	Memphis	10/28/2011	90.5	1.35	1.94	11.6	34.9	0.13	40.3
	111A	Memphis	10/28/2011	87.2	1.12	0.86	5.00	34.4	0.67	35.8
Davis	432	Memphis	8/11/2004	106.7	1.20	1.62	8.10	32.1	nd	nd
	414	Memphis	8/12/2004	118.3	1.58	3.28	18.8	34.2	nd	nd
	432	Memphis	9/13/2005	106.7	1.17	nd	4.10	nd	nd	nd
	414	Memphis	9/13/2005	118.3	1.56	3.85	17.8	31.0	nd	nd
	432	Memphis	9/27/2006	106.7	1.12	0.47	1.60	26.2	nd	nd
Lichterman	321	Memphis	9/16/2005	69.8	1.06	0.57	0.50	10.7	nd	nd
	311	Memphis	9/16/2005	75.6	1.02	0.44	<0	nd	nd	nd
	311	Memphis	9/28/2006	75.6	0.99	0.04	<0	nd	nd	nd
	321	Memphis	10/1/2006	69.8	1.01	0.07	<0	nd	nd	nd
	324	Memphis	10/5/2006	88.4	1.24	0.62	6.40	43.4	nd	nd
	321	Memphis	11/13/2007	69.8	1.03	0.30	<0	nd	nd	nd
McCord	203A	Memphis	9/22/2005	85.4	1.17	1.10	5.40	31.9	nd	nd
	232	Memphis	9/22/2005	119.2	1.22	1.83	6.80	27.9	nd	nd

Table 2 Continued.

Well field	Production well	Aquifer	Sample date	Top of screen (m)	R/Ra	^{3}H (TU)	^{3}He* (TU)	Age using Ne (yr)	SF_6 (fMol/L)	SF_6 age-piston flow (yr)
McCord	203A duplicate	Memphis	9/22/2005	85.4	1.16	0.97	4.00	29.3	nd	nd
	217	Memphis	11/17/2009	122.0	1.03	0.19	2.26	45.9	5.84	4.5
	233	Memphis	11/17/2009	115.9	1.13	0.70	4.12	34.6	3.37	15.4
	206	Memphis	11/18/2009	97.6	1.06	0.45	3.76	40.1	4.26	11.6
	201	Memphis	11/20/2009	102.4	0.97	0.02	1.18	>55	1.02	28.4
	202	Memphis	11/20/2009	89.3	1.09	0.45	4.33	42.4	4.77	9.4
	203A	Memphis	8/7/2013	85.4	1.18	0.85	5.91	37.1	0.41	29.6
	209	Memphis	8/7/2013	86.6	1.21	0.75	6.68	41.2	0.48	27.6
	218	Memphis	8/8/2013	125.0	0.96	nd	0.12	nd	nd	nd
Morton	602	Memphis	11/28/2003	111.0	0.90	0.0	0.60	>40	nd	nd
Shaw	704	Memphis	11/1/2011	76.2	1.05	0.25	2.57	43.3	1.96	6.3
	708	Memphis	11/1/2011	91.5	1.10	0.69	4.11	34.9	0.50	26.8
	708 duplicate	Memphis	11/1/2011	91.5	nd	0.59	nd	nd	0.63	24.3
Sheahan	99	Memphis	11/19/2002	108.2	1.01	0.29	4.87	51.6	nd	nd
	88	Memphis	11/20/2002	103.4	1.04	1.0	1.73	18.0	nd	nd
	87A	Memphis	11/20/2002	94.8	1.11	1.38	4.24	25.2	nd	nd
	78B	Memphis	11/20/2002	125.3	0.94	0.04	<0	nd	nd	nd
	80A	Memphis	11/20/2002	126.5	0.97	0.18	5.11	60.6	nd	nd
	87A	Memphis	11/14/2007	94.8	1.14	1.0	4.80	32.0	nd	nd
	86R	Memphis	11/3/2011	129.6	1.02	0.36	2.17	34.9	0.67	23.6
	99	Memphis	11/3/2011	108.2	1.05	0.32	2.62	39.9	1.61	10.6
Bartlett	P3, W1	Memphis	10/4/2006	125.0	1.05	0.93	4.80	32.7	nd	nd
	P4, W3	Memphis	11/14/2007	149.4	0.98	0.20	<0	nd	nd	nd
Collierville	402	Memphis	12/9/2003	68.6	1.07	0.40	2.20	34.7	nd	nd
Germantown	J2	Memphis	12/10/2003	62.8	1.06	0.40	2.20	34.7	nd	nd
	10	Memphis	12/11/2003	94.5	0.98	0.10	<0	nd	nd	nd
	J2	Memphis	11/10/2011	62.8	1.05	0.42	2.82	36.8	0.81	22
	J10	Memphis	11/10/2011	172.3	0.98	0.05	1.03	>60	0.06	nd

nd = not determined

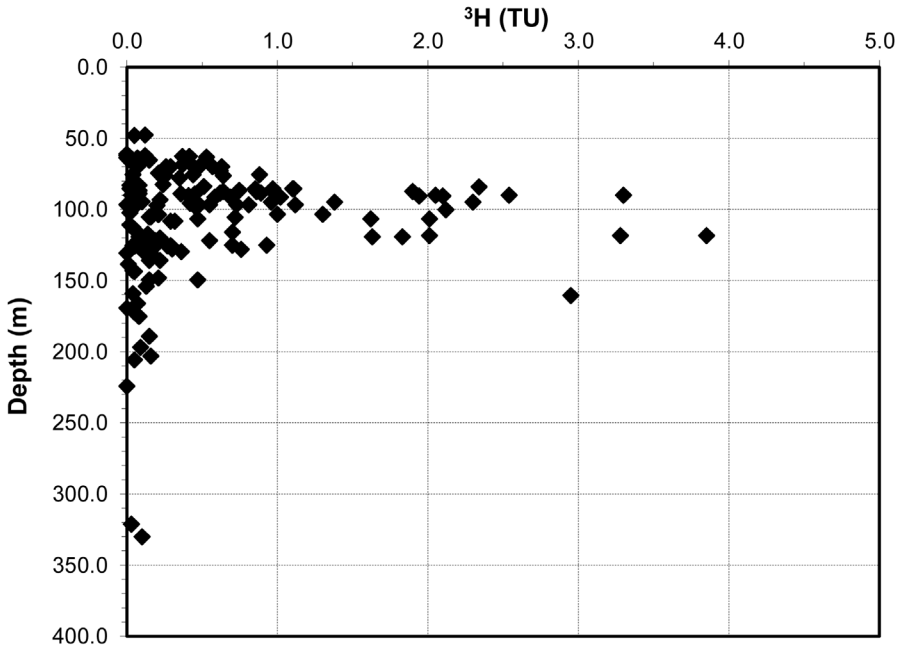

Figure 2. ^{3}H contents in production wells in Shelby County sampled between 2000 and 2013. The depth value plotted is the top of the screened interval, which is 24 to 30 m long for most wells.

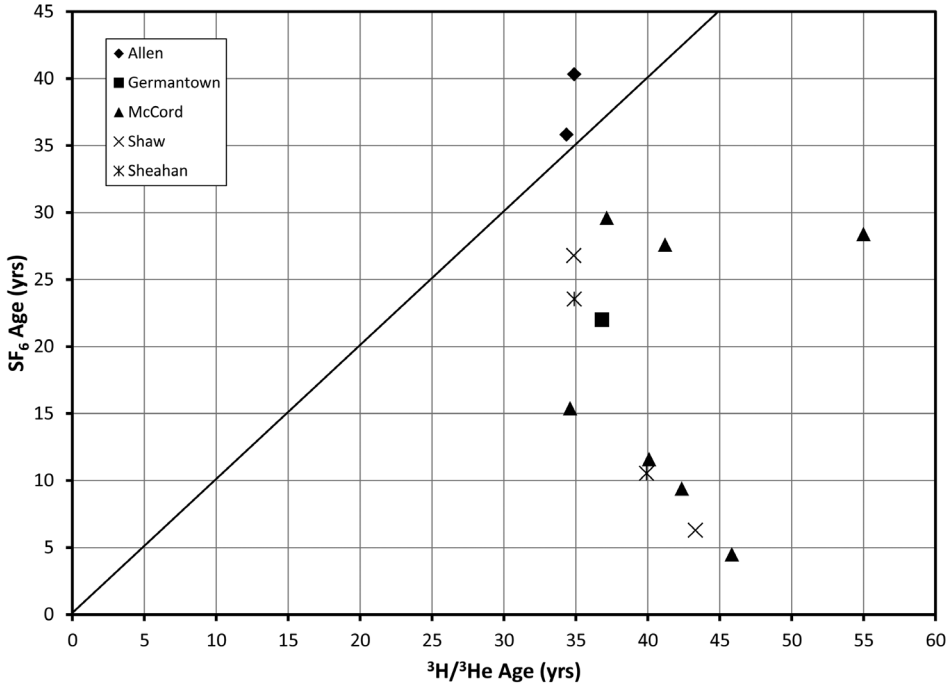

Figure 3. Crossplot of apparent ages determined by ^{3}H/^{3}He and SF_6 for water samples from Memphis aquifer production wells.

Discussion

Data Validation

A scatter plot of N_2 and ^4He is used to evaluate whether the gas data follow the expected water solubility relationships in groundwater equilibrated with the atmosphere (Solomon and Cook 2000). Waters from production wells in the Memphis aquifer plot in a region consistent with excess air saturation and equilibrium recharge temperatures ranging from 5 to 25˚C (Figure 4), with most data indicating equilibration temperatures between 8 and 15 ˚C. The mean annual temperature in the Memphis area is 17.6 ˚C (NCDC 2010) and most recharge is believed to occur during the cooler winter months. The values of R/R_a are generally between 1.0 to 1.6, which are consistent with atmospheric helium values augmented by bomb-pulse tritiogenic ^3He (Solomon and Cook 2000).

The SF_6 values for water from production wells are generally low, close to the lower limit of detection (0.01 fMol/L), but above background concentrations (0.05 fMol/L) of waters equilibrated with atmospheric SF_6. The Memphis aquifer is primarily composed of quartz sand (Lumsden et al. 2009; Larsen and Brock 2014) and is not likely to have significant SF_6 accumulation from igneous and metamorphic rocks (e.g., Busenberg and Plummer 2000). Based on these observations and evidence for the noble gas equilibration with the atmosphere at temperatures reasonable for the Memphis area, the SF_6 data appear to be useful for assessing residence time of young groundwater.

Apparent Age of Modern Water and Mixing with Old Water

The apparent age of groundwater sampled from public-supply wells with long screen intervals is known to comprise a mixture of old and young waters, commonly from one of more transmissive zones within the screened interval (Nelms et al. 2003; Manning et al. 2005; Eberts et al. 2012. The mixing fractions determined by the tritium loading model yield as much as 75% modern water in some wells producing from the Memphis aquifer, with most samples containing between 10 and 40% modern water (Table 3 and Figure 5). Not surprisingly, the apparent ages of these young water components tend to cluster between 20 and 45 years before the date of sampling, an age region

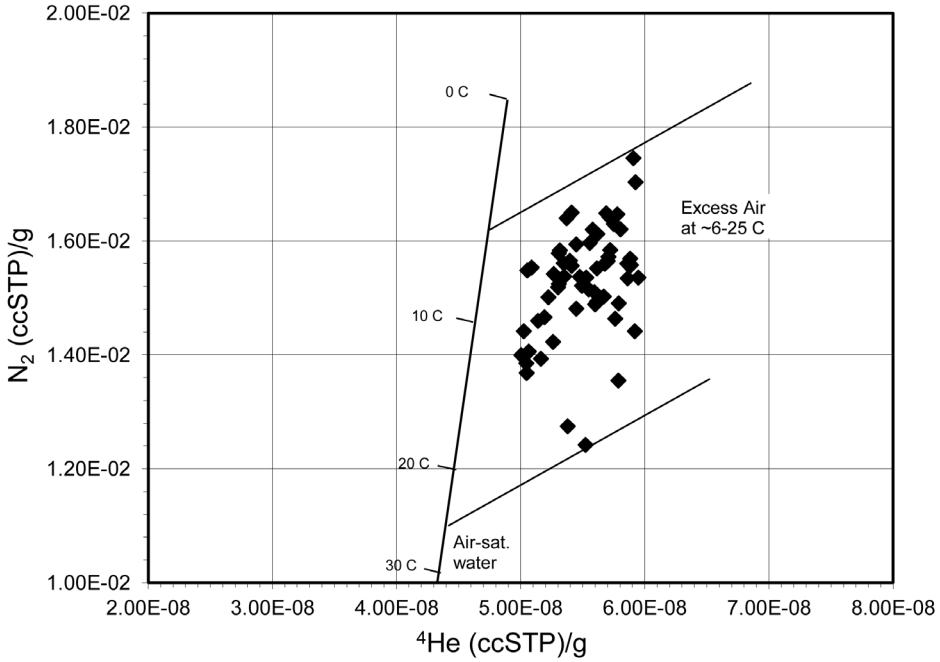

Figure 4. Plot of N_2 and ^4He dissolved gas contents of water from Memphis aquifer production wells.

for which the $^3H/^3He$ system is most sensitive to bomb peak atmospheric tritium loading (Solomon and Cook 2000). Because the piston-flow model assumes no dispersion or multimodality of the tracer loading history, the mixing fractions of modern water determined by tritium loading may be overestimates because of the strong influence of small quantities of bomb-pulse tritium.

Mixing fractions determined from inverse geochemical modeling of production well waters in the Sheahan (Larsen et al. 2003; Ivey et al. 2008), Davis (Koban et al. 2011), and McCord (Gallo 2015) well fields are plotted in Figure 6 with mixing fraction determined by the piston-flow $^3H/^3He$ model. Although the mixing fractions determined from the two independent methods crudely correlate, some of the results (for example, 2009 samples from McCord well field) yield significantly different mixing fractions. Because of the non-unique nature of the inverse geochemical model results, slight changes in water chemistry or choice of end-member wells can produce significantly different mixing fractions (Gallo 2015). Another consideration is that geochemical mixing model results may represent the mixtures of modern water with differing chemistry and sources. For example, Larsen et al. (2013) showed modern waters from two sources may be contributing to leakage in the Sheahan well field: modern water drained from storage in the Shallow aquifer and infiltrated modern stream water flowing through the Shallow aquifer to a window location.

Hydrogeologic and geochemical data from several studies of leakage near Memphis area public supply well fields (Parks et al. 1995; Larsen et al. 2003; Ivey et al. 2008; Koban et al. 2011; Larsen et al. 2013) provide hydrogeologic justification for a conceptual model of vertical leakage required for LPM analysis. Groundwater flow for the modern component is modeled using a range of models from a piston flow model (PFM) (the simplest case) to an exponential piston flow model (EPM), in which recharge enters the Memphis aquifer through a hydrogeologic window of width (x) and nearest distance (x*) (Cook and Bohlke 2000; Ivey et al. 2008). The EPM parameter (EPMP), x*/x, constrains aspects of the geometry of the window and is used in the model with idealized

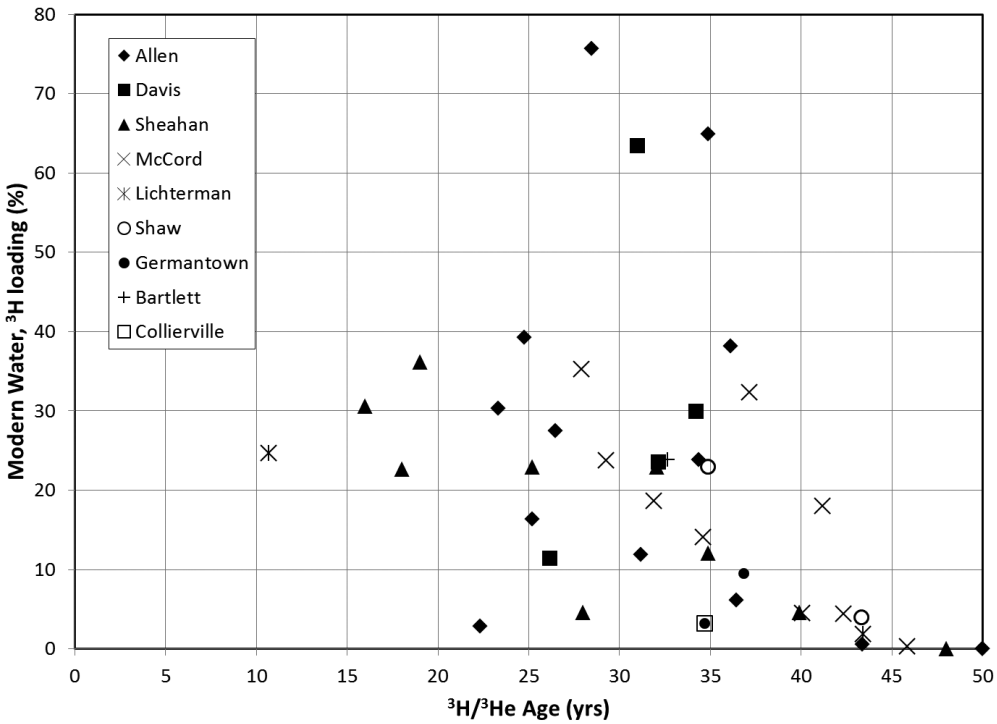

Figure 5. Plot of mixing percentage of modern water, determined by ^3H loading method, and $^3H/^3He$ apparent age for water samples from Memphis aquifer production wells.

tracer loading histories to match the observed tracer data (Jurgens et al. 2014). Groundwater flow for the pre-modern component in the Memphis aquifer is interpreted to follow a dispersion flow model (DM), in which hydrodynamic dispersion broadens the age distribution of a volume of water as the water flows downgradient from a distant recharge source. The age of the older water component was set at 2000 years for wells in all well fields except Shaw and Germantown (which were set to 1000 yrs), based on ^{14}C dating results of Memphis aquifer water sampled from observation wells throughout Shelby County (Graham and Parks 1986). The dispersion parameter, DP, is the ratio of the dispersion coefficient and advective flow over a distance, x (Zuber and Maloszewski 2001). The DP was fixed at 0.5 for all models. Given that advection likely dominates solute transport in the porous sand media of the Memphis aquifer and the 10s of km distance of travel from the outcrop region to the well fields, hydrodynamic dispersion is assumed to be about half as effective as advection in solute transport. Random variation of DP in a typical model resulted in little sensitivity in the model results. The source of most modern water contributions to the Memphis aquifer is closely tied to streams (Bradley 1991; Larsen et al. 2003; Bradshaw 2011; Larsen et al. 2013; Gallo 2015); thus, the travel time in the vadose zone is considered to be negligible.

The LPM results generally indicate a younger mean age of the modern water component than those from the $^{3}H/^{3}He$ and SF_6 apparent ages (Tables 3 and 4), but similar mixing fractions of modern water to those determined by the tritium loading model (Figure 7). The younger value of the mean modern water age is expected from the EPM due to contributions of modern waters from the 1960s and younger to model the observed apparent ages (e.g., Cook and Bohlke 2000). Although the EPMP is based on the conceptual model, in practice the model value is based on fitting the results to the specified model with the least error. The primary issue is that the width of the window, x, is poorly constrained, even in situations where the distance to the window, x*, is constrained by other data (e.g., Sheahan well field, Larsen et al. 2003; 2013; Ivey et al. 2008). However,

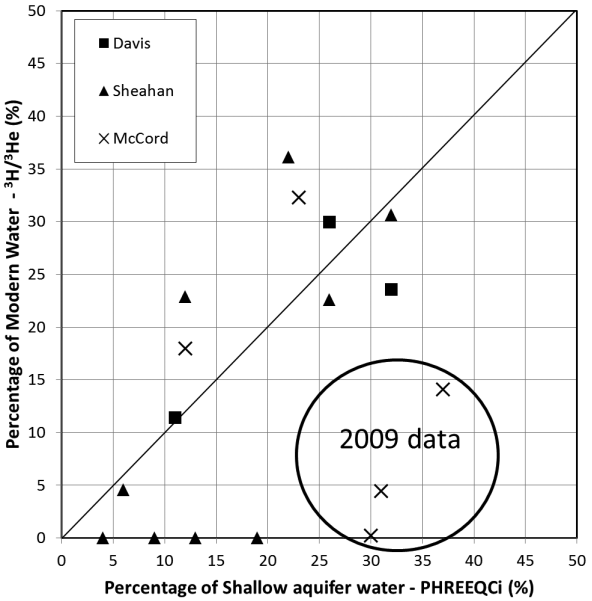

Figure 6. Plot of mixing percentage of modern water, determined by ^{3}H loading method, and mixing percentage of Shallow aquifer water, determined by inverse geochemical modeling with PHREEQCi, for water samples from three Memphis aquifer pubic supply well fields.

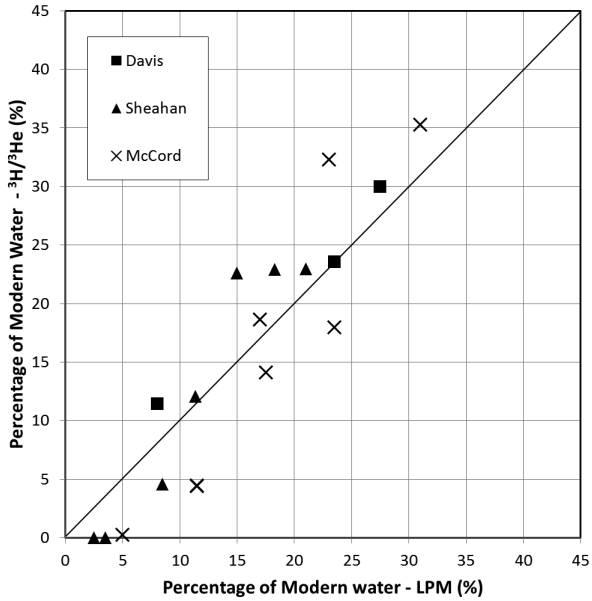

Figure 7. Plot of mixing percentage of modern water, determined by ^{3}H loading method, and mixing percentage of modern water, determined by LPM, for water samples from three Memphis aquifer public-supply well fields.

Table 3. Percentage modern water and adjusted SF_6 apparent age.

Well field	Production well	Aquifer	Sample date	$^3H+^3He^*$ (TU)	Age using Ne (yr)	Recharge year	Recharge yr ^3H load	% Modern ^3H load	% Modern Geochem	% Modern LPM	SF_6 age LPM mix (yr)
Allen	111A	Memphis	8/11/2004	4.9	36.4	1968	79.6	6	nd	nd	nd
	123	Memphis	8/11/2004	4.2	31.2	1973	35.0	12	nd	nd	nd
	107	Memphis	9/22/2006	8.2	24.8	1981	20.8	39	nd	nd	nd
	111A	Memphis	9/22/2006	3.6	23.3	1983	11.8	30	nd	nd	nd
	109	Memphis	9/27/2006	5.4	43.4	1963	937.5	1	nd	nd	nd
	107	Memphis	11/9/2007	12.4	28.5	1979	16.4	76	nd	nd	nd
	109	Memphis	11/9/2007	2.2	25.2	1982	13.7	16	nd	nd	nd
	114	Memphis	12/8/2008	15.7	36.1	1972	41.3	38	nd	nd	nd
	103	Memphis	12/11/2008	0.3	22.3	1986	9.8	3	nd	nd	nd
	111A	Memphis	11/12/2008	3.8	26.5	1982	13.7	28	nd	nd	nd
	114	Memphis	10/28/2011	13.6	34.88	1976	20.9	65	nd	25.6	29.8
	111A	Memphis	10/28/2011	5.9	34.4	1977	24.6	24	nd	21.4	22.3
Davis	432	Memphis	8/11/2004	9.7	32.1	1972	41.3	24	32	23.5	nd
	414	Memphis	8/12/2004	22.1	34.2	1970	73.8	30	26	27.5	nd
	414	Memphis	9/13/2005	21.7	31.0	1974	34.2	63	7	39.8	nd
	432	Memphis	9/27/2006	2.0	26.2	1980	17.7	11	11	8	nd
Licherterman	321	Memphis	9/16/2005	1.0	10.7	1994	4.2	25	nd	nd	nd
	324	Memphis	10/5/2006	7.0	43.4	1962	372.9	2	nd	nd	nd
McCord	203	Memphis	9/22/2005	6.5	31.9	1973	35.0	19	nd	17	nd
	232	Memphis	9/22/2005	8.7	27.9	1977	24.6	35	nd	31	nd
	203 duplicate	Memphis	9/22/2005	5.0	29.3	1976	20.9	24	nd	nd	nd
	217	Memphis	11/17/2009	2.4	45.9	1963	937.5	0	30.0	5.0	4.6
	233	Memphis	11/17/2009	4.8	34.6	1974	34.2	14	37.0	17.5	15.4
	206	Memphis	11/18/2009	4.2	40.1	1969	94.1	4	31.0	11.5	11.6
	201	Memphis	11/20/2009	1.2	>55	1954	4.0	nd	nd	nd	nd
	202	Memphis	11/20/2009	4.8	42.4	1967	108.3	4	nd	11.5	9.4
	203A	Memphis	8/7/2013	6.8	37.1	1976	20.9	32	23.0	23.0	7.6
	209	Memphis	8/7/2013	7.4	41.2	1972	41.3	18	12.0	23.5	4.1

Table 3 Continued.

Well field	Production well	Aquifer	Sample date	$^3H+^3He^*$ (TU)	Age using Ne (yr)	Recharge year	Recharge yr ^3H load	% Modern ^3H load	% Modern Geochem	% Modern LPM	SF_6 age LPM mix (yr)
Shaw	704	Memphis	11/1/2011	2.8	43.3	1968	79.6	4	nd	7.0	17.3
	708	Memphis	11/1/2011	4.8	34.9	1976	20.9	23	nd	19.5	24.8
Sheahan	87	Memphis	10/20/2000	7.5	nd	nd	nd	nd	22	nd	nd
	80	Memphis	10/20/2000	1.6	nd	nd	nd	nd	9	nd	nd
	78A	Memphis	10/21/2000	1.9	nd	nd	nd	nd	6	nd	nd
	88	Memphis	10/21/2000	3.7	nd	nd	nd	nd	32	nd	nd
	55B	Memphis	10/21/2000	1.3	nd	nd	nd	nd	4	nd	nd
	99	Memphis	11/19/2002	5.2	51.6	1950	4.5	nd	19	3.5	nd
	80	Memphis	11/20/2002	5.3	60.6	1941	4.5	nd	13	2.5	nd
	87A	Memphis	11/20/2002	5.6	25.2	1977	24.6	23	12	18.3	nd
	88	Memphis	11/20/2002	2.7	18.0	1984	12.1	23	26	15	nd
	87	Memphis	11/14/2007	5.7	32.0	1975	25.0	23	nd	21	nd
	86R	Memphis	11/3/2011	2.5	34.9	1976	20.9	12	nd	11.4	23.1
	99	Memphis	11/3/2011	2.9	39.9	1971	64.6	5	nd	8.5	19.6
Bartlett	P3, W1	Memphis	10/4/2006	5.7	32.7	1973	35.0	24	nd	nd	nd
Collierville	402	Memphis	12/9/2003	2.6	34.7	1968	79.6	3	nd	nd	nd
Germantown	J-2	Memphis	12/10/2003	2.6	34.7	1968	79.6	3	nd	nd	nd
	J-2	Memphis	11/10/2011	3.2	36.8	1974	34.2	9	nd	11	24
	J-10	Memphis	11/10/2011	1.1	>60	1951	4.0	nd	nd	nd	nd

LPM = Lumped Parameter Model
nd = not determined

Table 4. Lumped Parameter Modeling results.

Well field	Production well	Sample date	LPM model	Vadose zone travel time (yr)	Mean age of modern water (yr)	EPMP	Mixing % of modern water	Mean age of pre-modern water (yr)	Tracers optimized	Relative error (%)
Allen	114	10/28/2011	BMM-EPM-DM	5.0	26.1	6.0	25.6%	2000	^3H, ^3He_t, SF_6	0.0%
	111A	10/28/2011	BMM-EPM-DM	0.0	20.0	16.0	21.4%	2000	^3H, ^3He_t, SF_6	101.7%
Davis	432	8/11/2004	BMM-EPM-DM	0.0	20.0	10.2	23.5%	2000	^3H, ^3He_t	0.0%
	414	8/12/2004	BMM-EPM-DM	0.0	23.2	3.2	27.5%	2000	^3H, ^3He_t	0.1%
	414	9/13/2005	BMM-EPM-DM	0.0	24.2	5.8	39.8%	2000	^3H, ^3He_t	0.0%
	432	9/27/2006	BMM-EPM-DM	0.0	14.8	9.8	8.0%	2000	^3H	0.0%
McCord	203A	9/22/2005	BMM-EPM-DM	0.0	6.0	17.1	17.0%	2000	^3H, ^3He_t	0.0%
	203A	*9/22/2005*	*BMM-EPM-DM*	*0.0*	*5.9*	*5.0*	*17.0%*	*2000*	*^3H, ^3He_t*	*0.0%*
	232	9/22/2005	BMM-EPM-DM	0.0	15.8	6.3	31.0%	2000	^3H, ^3He_t	0.0%
	217	11/17/2009	BMM-EPM-DM	0.0	6.6	23.3	5.0%	2000	^3H, ^3He_t	155.8%
	217	*11/17/2009*	*BMM-PFM-DM*	*0.0*	*4.7*	*na*	*4.5%*	*2000*	*^3H, ^3He_t*	*155.9%*
	233	11/17/2009	BMM-EPM-DM	0.0	5.7	10.6	17.5%	2000	^3H, ^3He_t	26.5%
	206	11/18/2009	BMM-EPM-DM	0.0	6.1	13.2	11.5%	2000	^3H, ^3He_t	46.1%
	202	11/20/2009	BMM-EPM-DM	0.0	6.2	10.1	11.5%	2000	^3H, ^3He_t	27.0%
	203A	8/7/2013	BMM-EPM-DM	0.0	7.1	6.1	23.0%	2000	^3H, ^3He_t, SF_6	14.9%
	209	8/7/2013	BMM-EPM-DM	0.0	9.9	14.9	23.5%	2000	^3H, ^3He_t, SF_6	45.7%
Shaw	704	11/1/2011	BMM-EPM-DM	0.0	7.9	15.3	7.0%	1000	^3H, ^3He_t, SF_6	204.2%
	704	*11/1/2011*	*BMM-EPM-DM*	*0.0*	*7.7*	*16.8*	*7.0%*	*2000*	*^3H, ^3He_t, SF_6*	*202.6%*
	708	11/1/2011	BMM-EPM-DM	0.0	7.9	24.3	19.5%	1000	^3H, ^3He_t	27.5%
	708	*11/1/2011*	*BMM-EPM-DM*	5.0	*14.0*	*20.0*	*12.3%*	*2000*	*^3H, ^3He_t, SF_6*	*25.9%*

Table 4 Continued.

Well field	Production well	Sample date	LPM model	Vadose zone travel time (yr)	Mean age of modern water (yr)	EPMP	Mixing % of modern water	Mean age of pre-modern water (yr)	Tracers optimized	Relative error (%)
Sheahan	99	11/19/2002	BMM-EPM-DM	0.0	5.5	9.2	3.5%	2000	^3H, ^3He_t	21.1%
	80	11/20/2002	BMM-PFM-DM	0.0	7.9	na	2.5%	2000	^3H, ^3He_t	16.4%
	87A	11/20/2002	BMM-EPM-DM	0.0	3.2	8.4	18.3%	2000	^3H, ^3He_t	21.9%
	88	11/20/2002	BMM-EPM-DM	0.0	2.1	8.5	15.0%	2000	^3H, ^3He_t	201.7%
	87A	11/14/2007	BMM-EPM-DM	0.0	3.3	6.5	21.0%	2000	^3H, ^3He_t	3.3%
	86R	11/3/2011	BMM-PFM-DM	0.0	9.8	na	11.4%	2000	^3H, ^3He_t, SF_6	228.8%
	99	11/3/2011	BMM-EPM-DM	0.0	6.9	11.0	8.5%	2000	^3H, ^3He_t	114.9%
Germantown	J-2	11/10/2011	BMM-PFM-DM	0.0	6.6	na	10.5%	1000	^3H, ^3He_t	153.5%
	J-2	*11/10/2011*	*BMM-PFM-DM*	*0.0*	*6.8*	*na*	*10.7%*	*2000*	*^3H, ^3He_t, SF_6*	*176.7%*

LPM = Lumped Parameter Model
EPMP = Exponential Piston Flow Model Parameter
na = not applicable
italics = repeated model run for sample above with either different EPMP value, LPM model, or mean age of pre-modern water

the actual sensitivity of the EPM results on the EPMP is minimal in some cases. For example, model runs with different initial values for mean age, EPMP, and mixing % in well 203A sampled in 2005 (Table 4) converged with similar relative error yielding almost identical mean modern water ages and mixing percentages (6.0 and 5.9 yrs; 0.0% for both), but different values of EPMP (17.1 and 5.0). In addition, the effect of using a PFM or EPM for the modern water component is minimal in some cases. For example, a PFM for well 217 sampled in 2009 yielded similar relative error to the EPM and a slightly younger mean modern water age and mixing % (Table 4). Some wells did not fit any model using $^{3}He_{t}$ very well, resulting in high model error (>100%), commonly because observed $^{3}He_{t}$ values are less than all model values. Good examples of this behavior are wells 217 and 704, which yield excessive error for all models with $^{3}He_{t}$. The model sensitivity to the age of the pre-modern water component is generally minor for the Germantown and Shaw wells sampled (Table 4).

Unlike the apparent age results from the $^{3}H/^{3}He$ system, SF_6 data do not provide independent information of mixing fractions of modern and pre-modern waters. Thus, the poor correlation of apparent ages for production well waters based on $^{3}H/^{3}He$ and SF_6 is expected (Figure 3). Correction of the SF_6 apparent age data for mixing fractions using the LPM mixing percentages in Table 4 does not improve the correspondence to the $^{3}H/^{3}He$ apparent age data, but generally produces younger ages for the modern water component. Several explanations can be considered, including error in the mixing quantification, erroneous $^{3}H/^{3}He$ or SF_6 apparent age data, or differences in the sensitivity of each tracer to the age of modern water. Although errors in the dating and mixing calculations may obscure relationships, the bias of the SF_6 system toward younger apparent age relative to the $^{3}H/^{3}He$ system seems to be the most reasonable explanation for the general shift toward younger SF_6 apparent ages.

Implications for Wellhead Protection and Policy

The results of the past 14 years of environmental tracer analysis in production wells in Memphis and Shelby County, Tennessee, provide strong evidence for vertical recharge of modern water to the Memphis aquifer. However, each tracer provides somewhat different information. The $^{3}H/^{3}He$ system provides the greatest information through the apparent age determination and coupling of assumed atmospheric ^{3}H loading and ^{3}He daughter product retention to calculate mixing relationships (Manning et al. 2005). However, the decay and dispersion of bomb-peak ^{3}H in recently recharged groundwater limits the utility of this system for the youngest recharge. Furthermore, tracer-tracer plots of $^{3}He_{t}$ and ^{3}H or SF_6 commonly show apparent depletion in $^{3}He_{t}$, suggesting the potential for helium loss during sampling. Both of these issues bias the $^{3}H/^{3}He$ apparent age results toward older dates. Because of the increasing atmospheric concentration of SF_6 and resiliency to retarding or decay processes (Busenberg and Plummer 2000; Darling et al. 2012), SF_6 should provide the more useful tracer for the youngest recharge. However, correction for mixing with pre-modern water (lacking a terrestrial source of SF_6) may present challenges, especially in the absence of independent verification. Inverse geochemical models for mixing of modern and pre-modern waters may provide independent verification of mixing fractions under ideal conditions whereby only two chemically distinct water sources are mixing in the subsurface.

Despite the potential errors in calculated mixing proportions of modern water produced from production wells in Shelby County, it is clear from the current studies as well as regional modeling that vertical leakage of modern water occurs at windows in the upper Claiborne confining unit near streams and is a significant component of long-term sustainability of water supplies in the Memphis aquifer. The tracer data verify and detail the results of regional groundwater flow modeling of Brahana and Broshears (2001) that show as much as 50% of the water withdrawn from the Memphis aquifer in 1980 was derived from vertical leakage. Specific areas of prominent leakage are from the Shallow aquifer along the Wolf and Loosahatchie rivers and Nonconnah Creek watersheds, and along the Mississippi River alluvial valley. Based on the tracer results, the percentage of modern water produced is likely to be much less than 50%, except for the cases of the Allen and Davis well fields (Koban et

al. 2011). The LPM modeling results in Table 4 show that most wells affected by contributions of modern leakage have mixing proportions between 5 and 30% modern water. From the standpoint of sustainability of groundwater availability, the Memphis aquifer has great potential to continue to show stable groundwater levels, as demonstrated by Kingsbury (1996), and provide a steady water supply at current usage.

The prominence of vertical leakage of modern water in the Memphis aquifer mainly presents a challenge to water quality. United State Geological Survey studies of water quality in the Shallow aquifer and in urban streams in the Memphis area (Coupe 2000; Gonthier 2002; Barlow et al. 2012) demonstrate widespread loss of water quality from additions of nitrate, volatile organic compounds, and pesticides. In addition, specific areas of contamination of the Shallow aquifer are known in many parts of the Memphis area (see Parks 1990 for review), some of which clearly impact water quality in the underlying Memphis aquifer by vertical leakage through windows in the upper Claiborne confining unit (Parks and Mirecki 1992; Gentry et al. 2005; Doudrick 2008). Figure 8 shows a map of potential and historical contaminant sources in Shelby County assembled from U.S. Environmental Protection Agency (USEPA) facility registry services and publically available data from the Tennessee Department of Environment and Conservation. Potential sources of contamination are proximal to almost all of the public supply well fields and some are adjacent to hydrogeologic windows, which may provide a contaminant pathway to the Memphis aquifer.

The USEPA (EPA 1987) recommends that states implement wellhead protection area (WHPA) criteria for public water supplies. In compliance with USEPA and Safe Drinking Water Act regulations, the State of Tennessee established wellhead protection regulations for all public water supplies utilizing groundwater (TNPR 0400-45-01-.34). A wellhead protection plan (WPP) is developed by public water suppliers and must delineate primary and secondary zones of protection, or time of travel (TOT), zones for public supply wells. In practice, the duration of the TOT zones is agreed upon by the state regulators and utility or municipality so as to protect the groundwater supply. Most utilities utilize the USEPA WHPA code (Blanford and Huyakorn 1991) or the updated version, WhAEM (Wellhead Analytical Element Model) (Kraemer et al. 2007), to evaluate the TOT zones by employing simplified analytical models for determining the radius of effective protection of pumping wells under varying conditions.

Because of the simplified nature of the groundwater flow model in the USEPA WHPA and WhAEM codes, the complexities of the local hydrogeologic setting, such as windows in the confining unit (Parks 1990) or paleovalley channels in the Shallow aquifer (Larsen et al. 2013), cannot be incorporated readily into the modeling environment. The sources of modern water established by several environmental tracer and hydrogeologic studies of public supply wells in the Memphis area are streams or shallow aquifers connected to streams (Larsen et al. 2003, 2013; Bradshaw 2011; Koban et al. 2011; Gallo 2015), most of which are outside of the TOT zones developed from the WPP. Furthermore, it is unclear whether more complex modeling approaches such as applying MODFLOW (McDonald and Harbaugh 1988) would predict the degree of modern water influence, mainly because of limited hydraulic head and aquifer property data in the areas near the well fields (e.g., Sheahan well field). Thus, we argue that environmental tracers of modern water, such as $^{3}H/^{3}He$ and SF_6, provide essential information in any assessment of the WHPA, especially in areas where hydrogeologic complexities and limited hydraulic data diminish the resolution of analytical or numerical models.

The $^{3}H/^{3}He$, SF_6, and associated tracer data gathered during the past 14 years clearly show that the Memphis aquifer at many of the well fields in Shelby County is more vulnerable to contamination than is suggested by wellhead protection modeling. Several aspects of the hydrogeology and groundwater development history of the Memphis aquifer present intrinsic susceptibility to contamination through the existence of fast recharge pathways and a prominent downward vertical gradient between the Shallow and Memphis aquifers. Detailed environmental tracer case studies of the Sheahan (Larsen et al. 2003; 2013; Ivey et al. 2008), Davis (Parks et al. 1995;

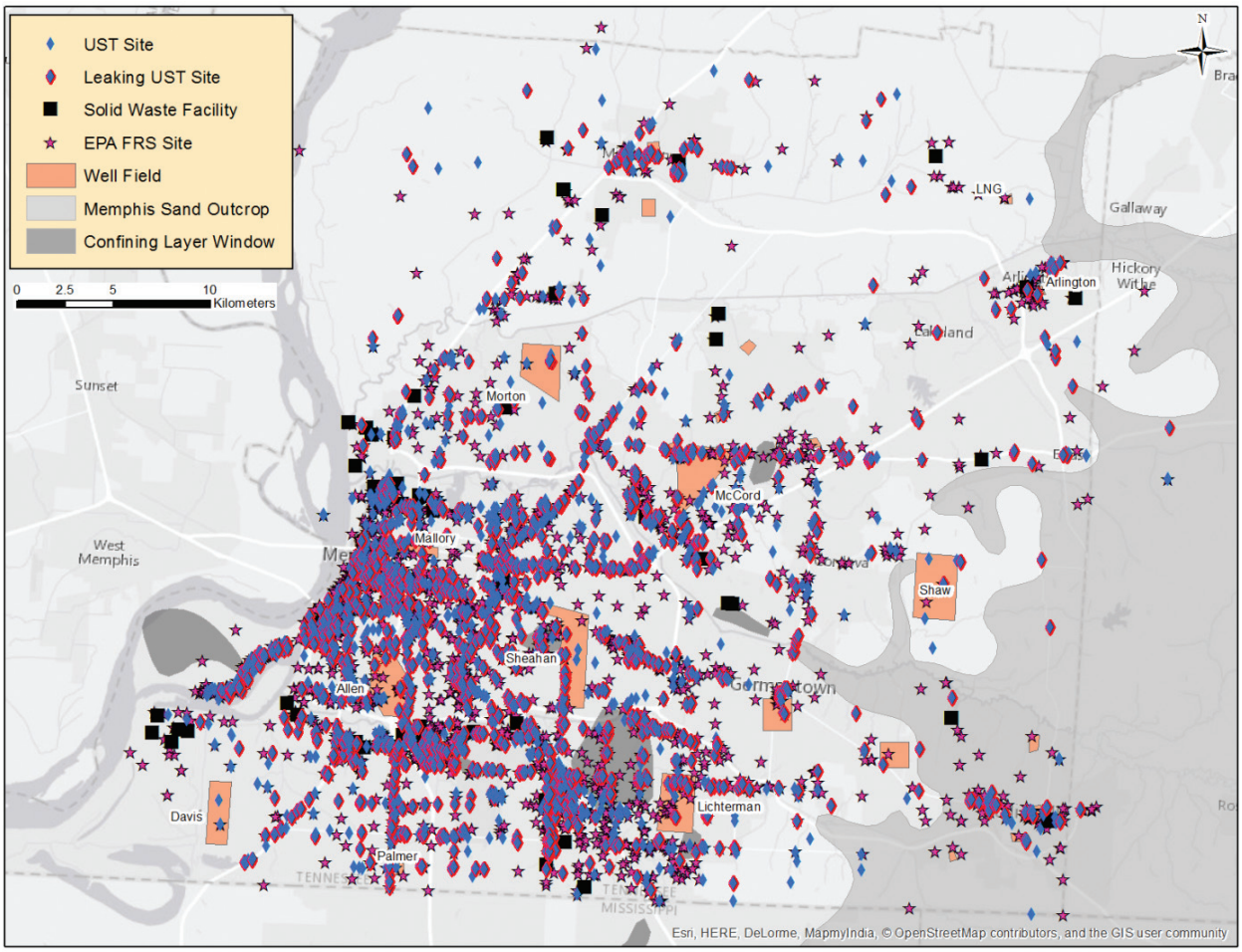

Figure 8. Map of Shelby County showing public supply well fields, hydrogeologic windows, and potential contaminant sources. UST is underground storage tank and FRS is federally registered site.

Koban et al. 2011), Allen (Bradshaw 2011), and McCord (Gallo 2015) well fields all indicate that substantial modern water fractions, mainly from nearby streams, are being drawn into the Memphis aquifer through hydrogeologic windows in the confining unit. However, the results also argue for a more comprehensive and continuous sampling plan rather than the piecemeal sampling approach so far utilized. As shown in figure 8, the number of potential contaminant sources adjacent to these well fields and nearby hydrologic windows creates significant aquifer vulnerability and concerns for the sustainability of water quality.

Conclusions

Environmental tracers (^{3}H, He isotopes, noble gases, and SF_6) have been sampled in production wells in Shelby County, Tennessee, to evaluate the presence of modern water in the semi-confined Memphis aquifer. The Memphis aquifer is a regionally important public supply source that has been shown to be locally susceptible to contamination. Use of groundwater from the Memphis aquifer since the late 1800's has caused the original upward hydraulic gradient in the aquifer to reverse in direction, such that modern water from the Shallow aquifer and local streams recharge the Memphis aquifer locally. Environmental tracers were used to specifically evaluate the age distribution of the modern water and quantity being withdrawn from public supply wells.

Tritium data from a large number of production wells show persistent presence of modern water in the Memphis aquifer, primarily in the upper 100 m. The ^{3}H/^{3}He and SF_6 apparent ages from affected wells in the Memphis aquifer show ages

of 20 to 45 years. The percentage of modern water in these wells using the tritium-loading method ranges from 0 to 75%, although most are less than 40% modern water. Inverse geochemical mixing models of a group of the wells from three well fields generally produce mixing percentages similar to those derived from the $^{3}H/^{3}He$ system; however, poor correspondence is indicated in some wells. Lumped parameter models (LPM) for binary mixtures of vertical recharge following an exponential piston flow behavior into long-flow path of deep Memphis aquifer water generally show mixing percentages similar to those from the tritium-loading method. However, the LPM mean ages of modern water are substantially younger than the apparent ages, mainly due to the application of the exponential piston flow model. Application of the LPM mixing percentages to the SF_6 apparent ages as a correction for dilution by pre-modern, non-tracer bearing waters results in younger SF_6 apparent ages than those from the $^{3}H/^{3}He$ system. The difference in the $^{3}H/^{3}He$ and SF_6 apparent ages is interpreted to reflect the differing sensitivities of these dating systems, with the $^{3}H/^{3}He$ system being more sensitive to older modern water and SF_6 being more sensitive to the youngest modern water.

The $^{3}H/^{3}He$ and SF_6 data and associated modeling verify and clarify the results of groundwater flow modeling in the Memphis aquifer by Brahana and Broshears (2001) that suggested as much as 50% of the water withdrawn from the aquifer in Memphis is from vertical recharge. The tracer-based mixing percentages indicate that quantities between 5 and 30% of modern water are being withdrawn from the Memphis aquifer in affected well fields. The tracer data combined with hydrogeologic information from case studies indicate the modern water source is mainly derived from local streams or shallow aquifers in hydraulic connection to the streams. Wellhead protection areas (WHPA) of the individual wells with significant percentages of modern water do not extend to the local streams, suggesting that the wells have greater inherent susceptibility to contamination than is indicated from WHPA modeling. Furthermore, the proximity of many potential contaminant sources to well fields and nearby hydrogeologic windows suggests significant vulnerability to contamination. The environmental tracer data provide essential information for evaluating WHPA that may not be gathered solely from traditional groundwater flow modeling, especially in cases where complexities in the hydrogeologic setting may be difficult to capture in modeling.

Although detailed groundwater flow modeling will undoubtedly provide greater insight to the flow dynamics and sensitivity of the local hydrologic systems, the environmental tracer data collected in the present study clarify the need for concern and further study. In addition, the connection between surface water and groundwater resources of the Memphis aquifer demand that a more holistic approach to water resources management be employed to ensure comprehensive sustainability.

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Geochemical Factors Controlling the Mobilization of Arsenic at an Artificial Recharge Site, Clearwater, Florida

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Abstract: Aquifer storage and recovery and artificial recharge projects are common methods of developing and managing groundwater resources for future use by cities that face pressure to maintain sustainable supplies of water for growing populations. Such projects, however, are often accompanied by water-quality problems that can be mitigated by careful attention to geological and geochemical factors that account for the occurrence and mobilization of contaminants such as arsenic in storage zones. Investigations of artificial recharge systems in Florida, for example, have identified oxidative dissolution of pyrites by treated injectate waters as the principal factor causing the mobilization of arsenic from the matrix of an aquifer. This paper describes major-ion chemistry and associated reduction-oxidation factors that account for the occurrence of arsenic in groundwater at an artificial recharge site under development by the City of Clearwater, Florida. The results of this investigation are fully consistent with those of previous studies. Of particular significance is the apparent sensitivity of arsenical sulfides to slight changes in the redox conditions under which those minerals are otherwise stable.

Keywords: *aquifer storage and recovery, artificial recharge, arsenic, reduction-oxidation potential, oxidative dissolution*

As many cities consider the need to develop sustainable sources of water for growing populations, strategies such as aquifer storage and recovery (ASR) and artificial recharge (AR) have become central components of long-term water-resource management plans. Without thorough characterization of local geology and groundwater chemistry, such strategies are often accompanied by problems with the potential to limit the effectiveness of ASR and AR systems. One problem associated with such unconventional water-supply programs is the potential for the liberation of arsenic from minerals that are otherwise stable in subsurface environments before the injection of treated water.

Arsenic concentrations greater than the U.S. Environmental Protection Agency's prescribed maximum contaminant level (MCL) of 10 micrograms per liter (µg/L) have been detected in recovery water at ASR sites in Florida (Arthur, Dabous, and Cowart 2002; Price and Pichler 2006; Jones and Pichler 2007). Investigations of the association of arsenic and pyrite have consistently identified oxidation of iron sulfides, specifically pyrite (FeS_2), as the principal factor accounting for the occurrence of arsenic in recovery water (Arthur, Dabous, and Cowart 2002; Jones and Pichler 2007).

Jones and Pichler (2007) studied this association in the Suwannee Limestone of the Upper Floridan Aquifer. Their findings are summarized as follows:

> *Geochemical modeling was employed to examine pyrite stability in limestone during simulated injections of surface water into wells open only to the Suwannee Limestone with known mineralogy and water chemistry. The goal was to determine if aquifer redox conditions could be altered to the degree of pyrite instability. Increasing amounts of injection water were added to native storage-zone water, and resulting reaction paths were plotted on pyrite stability diagrams. Native*

storage-zone water plotted within the pyrite stability field, indicating that conditions were sufficiently reducing to allow for pyrite stability. Thus, arsenic is immobilized in pyrite, and its groundwater concentration should be low. This was corroborated by analysis of water samples, none of which had arsenic concentrations above 0.036 microg/L. During simulation, however, as injection/native storage-zone water ratios increased, conditions became less reducing and pyrite became unstable. The result would be release of arsenic from limestone into storage-zone water (Jones and Pichler 2007, 723).

Clearwater Groundwater Replenishment Project

Geosyntec Consultants (Geosyntec) conducted a groundwater geochemical modeling investigation (Darling 2014) as part of the Clearwater Groundwater Replenishment Project (Project) to evaluate the suitability of the Suwannee Limestone (Floridan aquifer) to be an injection zone for up to three million gallons per day (MGD) of purified reclaimed water from the City of Clearwater Northeast Water Reclamation Facility (NEWRF). This paper is a highly condensed summary of the geochemical investigation (Darling 2014). The work was conducted for Leggett, Brashears & Graham, Inc. (LBG) as part of the hydrogeologic services conducted for the Project on behalf of the City of Clearwater, Florida (City). The Southwest Florida Water Management District (SWFWMD) was a source of cooperative funding for the Project.

A principal matter of concern to the City and SWFWMD is the potential for the mobilization of arsenic and other metals within the injection zone. The primary objectives of this investigation were to describe the major-ion chemistry and hydrochemical facies of the groundwater system at the Project site, ascertain whether arsenic is a naturally occurring solute in groundwater, and identify probable mineralogical associations and geochemical controls on the occurrence of arsenic.

A parallel and complementary component of this investigation involved an assessment of rock cores to describe the bulk mineralogical makeup of the aquifer matrix and to identify associated metals that might be released to groundwater through rock-water interactions. Additionally, bench-scale column tests were conducted to determine whether arsenic and other metals can be leached from the matrix and associated minerals under variable pH and reduction-oxidation (redox) conditions (Norton et al. 2014). The results of the investigation by Norton et al. (2014) are summarized in the discussion section of this paper.

Groundwater Sample Collection and Laboratory Analyses

The investigation began with the collection of samples of groundwater from four monitor wells and one water supply well at the NEWRF. The four monitor wells are divided into two upper-zone (UZAMW-1 and UZAMW-2) and two lower-zone (LZAMW-1 and LZAMW-2) wells, all of which are located within the southern half of the Project area (Figure 1). The depth of each upper-zone well is approximately 150 feet below land surface (ft BLS), and the depth of each lower-zone well is approximately 340 ft BLS (Trommer 2014). The source well drilled to supply water for a long-term injection test is also a lower-zone well, but at a depth of approximately 360 ft BLS, and located near the northeast corner of the property (Figure 1). Water from the source well is pumped through a pipeline to a recharge well (RW-1) located north of the UZAMW-1 and LZAMW-1 monitor wells (Figure 1). The injection zone is designated "Lower Zone A" of the Suwannee Limestone (Figure 2). In this paper, the upper zone wells are referred to as UZA1 and UZA2, and the lower zone wells as LZA1, LZA2, and RW1.

Between April 9 and July 2, 2014, 13 sets of samples were collected from Wells UZA1, UZA2, LZA1, LZA2, and RW1, for a total of 65 samples. The sampling method involved the recording of temperature, conductivity, pH, dissolved oxygen (DO), and oxidation-reduction potential (ORP), along with the time at which the variables were recorded. At each well, water was diverted through a flowline to a closed flow cell with probes attached to a calibrated pH/ORP/temperature/conductivity meter. The above variables were recorded on field data sheets at five-minute intervals until the measurements changed by no more than five percent (plus or minus) over two consecutive five-minute intervals.

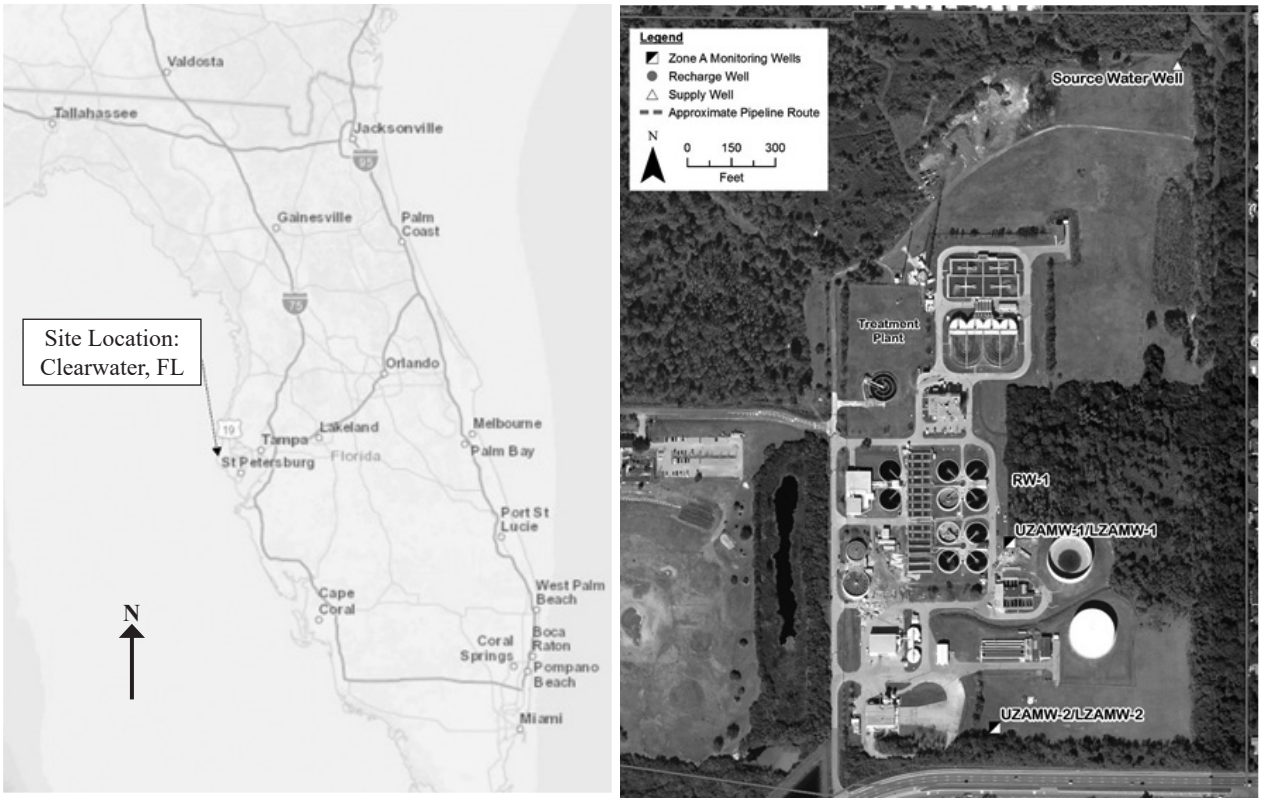

Figure 1. Location of Groundwater Replenishment Project and Project Site. Source: Darling 2014.

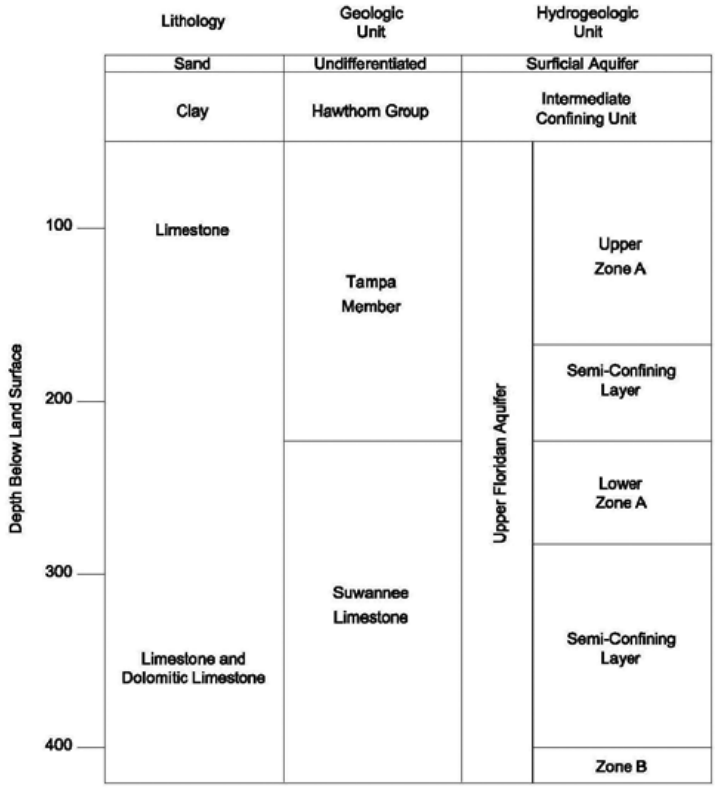

Figure 2. Stratigraphy of Clearwater Groundwater Replenish Project Site.

After the field parameter values were determined to be stable, the flowline was removed from the flow cell and the flow of water to the line was shut off at the outflow valve to which the flowline was attached at the wellhead. A 0.45-micron filter was attached to the end of the flowline and the outflow valve was reopened, allowing water to flow through the line again and to fill the 0.45-micron filter. Filtered water was then discharged to a 0.5-L Nalgene bottle to which nitric acid had been added by TestAmerica Laboratories of Tampa, Florida. That sample was designated for analysis of metals. A second 0.5-L Nalgene bottle without nitric acid was also filled. That sample was designated for analysis of nonmetals. Field measurements and the list of analytes were as follows:

- Temperature
- pH
- ORP
- Dissolved Oxygen
- Total Dissolved Solids (TDS)
- Calcium
- Magnesium
- Sodium
- Potassium
- Iron
- Arsenic
- Sulfate
- Sulfide
- Alkalinity (Bicarbonate)
- Fluoride

Major-Ion Analyses

The major-ion chemistry of all samples is represented by a Durov diagram (Figure 3). The basis of a Durov diagram is percentage plotting, in separate trilinear diagrams of cations and anions in units of milliequivalent per liter (meq/L). Lines from each pair of points in the cation and anion triangles are projected into the central rectangle to form a common point, which represents the composition of the samples with respect to cations and anions. The points in the central rectangle are projected into the TDS and pH fields, which offer additional dimensions for comparison and contrast of samples.

Figure 3 is based on the average concentrations of cations, anions, TDS, and pH for each of the five wells to avoid overlapping of symbols. The groundwater samples form distinct clusters, with hydrochemical compositions that range from calcium-sodium-chloride for the upper-zone wells (UZA1 and UZA2) to sodium-chloride for the lower-zone wells (RW1, LZA1, and LZA2). The clusters are further distinguished from each other, as illustrated by the loci of points within the TDS field. Water from the lower-zone wells has higher TDS than water from upper-zone wells, and the RW-1 samples exhibited the highest TDS. On a plot of sodium versus chloride (Figure 4), the clusters lie along a line with a sodium-to-chloride concentration ratio that is representative of the mixing of fresh groundwater and seawater.

With regard to the occurrence of arsenic in groundwater, there are four distinct clusters formed by the samples from the five wells (Figure 5). Figure 5 is not intended to indicate a causal link between arsenic and chloride. Figure 5 shows different ranges of arsenic for each group of samples. Arsenic concentrations decrease in the following order: UZA2 > RW1 > UZA1 > LZA1 = LZA2. Because arsenic concentrations among the LZA1 and LZA2 samples overlap, LZA1 and LZA2 are regarded as a single group.

Arsenic concentrations among the UZA1 samples and the LZA1 and LZA2 samples are less than the 10-µg/L MCL for arsenic in drinking water. The concentrations for the UZA1 samples are greater than 5 µg/L but less than 10 µg/L, and the concentrations among the LZA1 and LZA2 samples are less than 5 µg/L. Arsenic concentrations among the 13 RW1 samples are distributed around a mean of approximately 14 µg/L.

The association of dissolved arsenic and iron in groundwater samples is illustrated by Figure 6. In samples with iron concentrations less than 0.05 mg/L, arsenic forms three distinct clusters defined by the 26 LZA1 and LZA2 samples, the 13 UZA1 samples, and the 13 RW1 samples. A fourth group is formed by the 13 UZA2 samples. Iron concentrations for the latter group range from 0.05 to 0.19 mg/L.

The highest arsenic concentrations are associated with the UZA2 samples, the group with the largest concentrations of dissolved iron. Figure 7 illustrates that the highest iron concentrations are generally associated with the highest ORP measurements. This is a direct indication that reducing conditions

predominate within the lower zone and within the area of the upper zone in the vicinity of UZA1. Around UZA2, however, the ORP measurements indicate that conditions are more oxidizing and thus more likely to lead to the dissolution of pyrite and to the release of arsenic.

The predominance of reducing conditions is also found in figures of the concentration of sulfide with respect to ORP (Figure 8), dissolved oxygen and iron (Figure 9), dissolved oxygen and sulfide (Figure 10), sulfide and arsenic (Figure 11), and dissolved oxygen and arsenic (Figure 12).

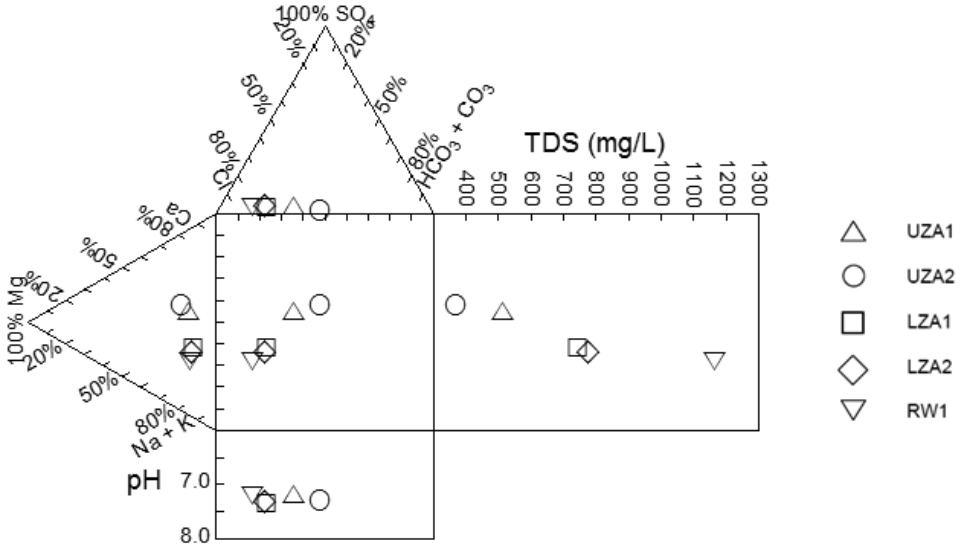

Figure 3. Durov diagram.

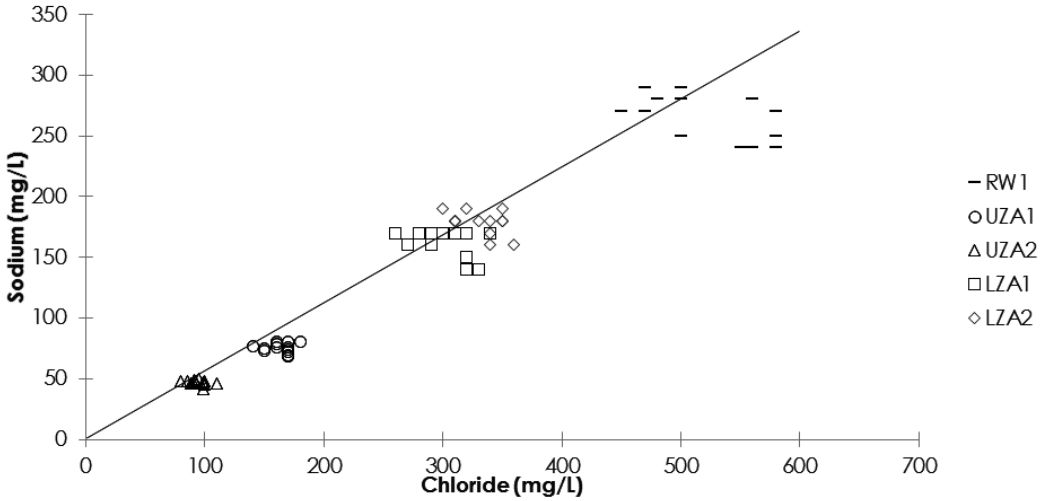

Figure 4. Plot of chloride versus sodium for Upper Zone and Lower Zone groundwaters.

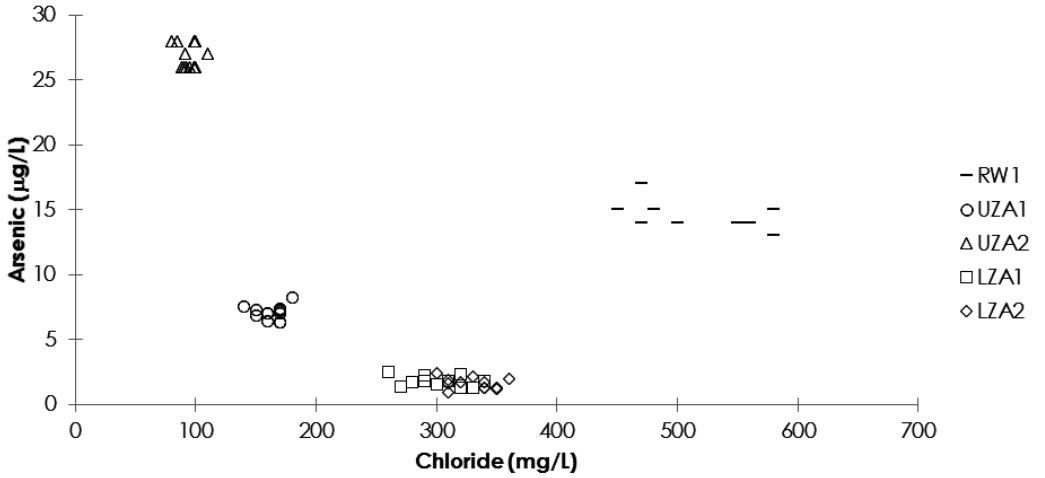

Figure 5. Plot of arsenic clusters and concentrations of chloride for Upper Zone and Lower Zone groundwaters.

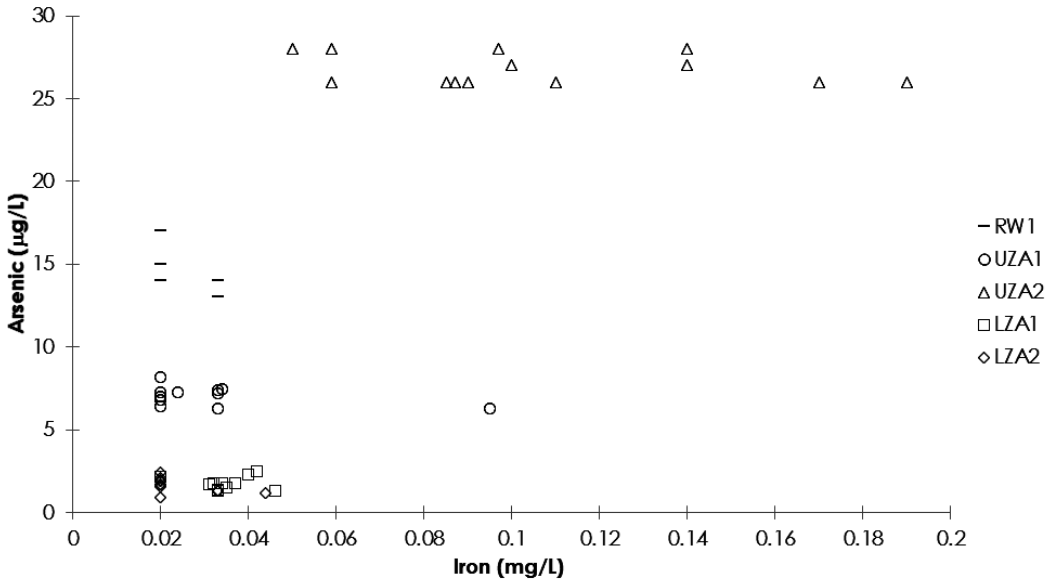

Figure 6. Plot of concentrations of arsenic and iron for Upper Zone and Lower Zone groundwaters.

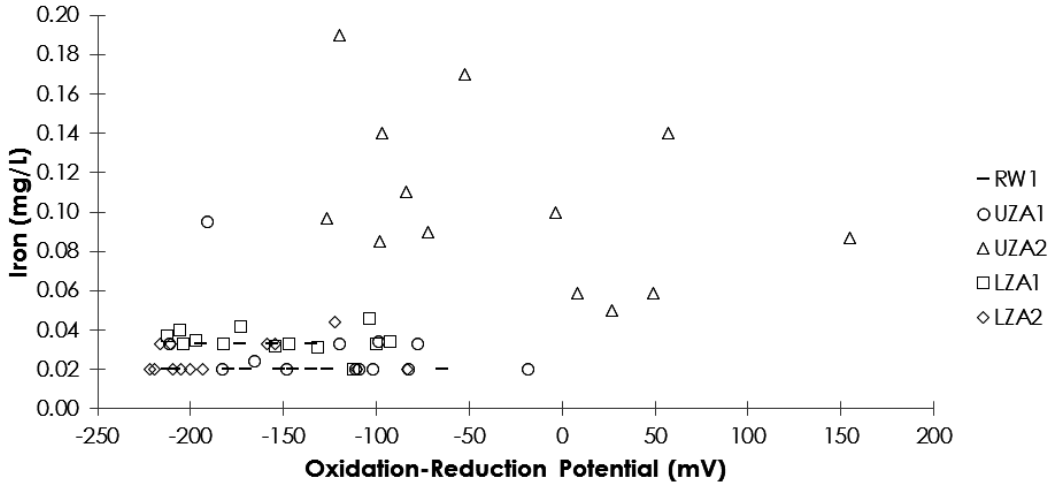

Figure 7. Plot of iron concentrations and measurements of oxidation-reduction potential for Upper Zone and Lower Zone groundwaters.

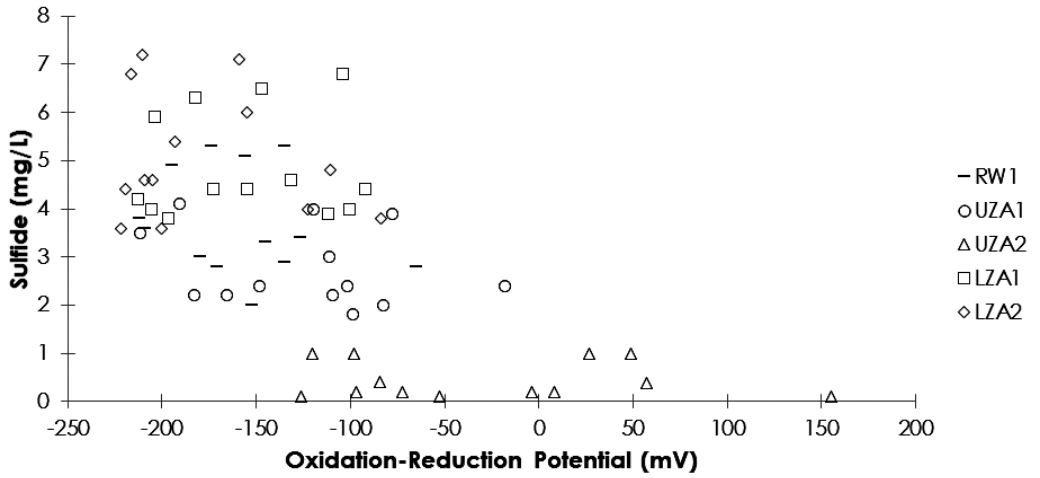

Figure 8. Plot of concentrations of sulfide and measurements of oxidation-reduction potential for Upper Zone and Lower Zone groundwaters.

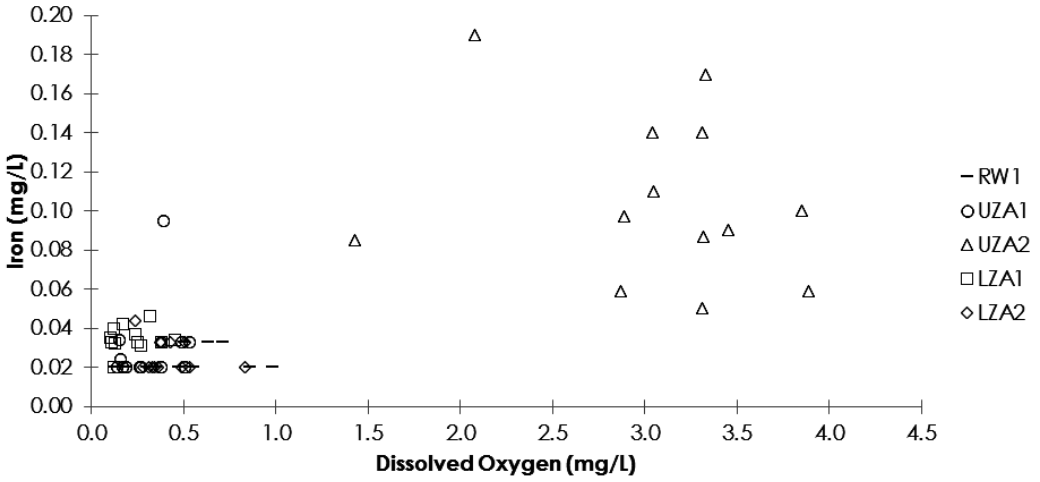

Figure 9. Plot of concentrations of iron and dissolved oxygen for Upper Zone and Lower Zone groundwaters.

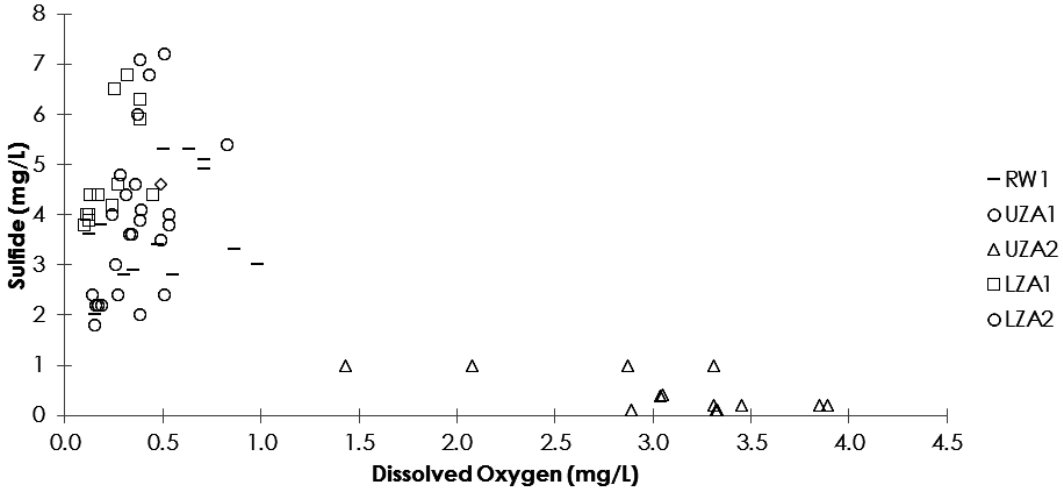

Figure 10. Plot of concentrations of sulfide and dissolved oxygen for Upper Zone and Lower Zone groundwaters.

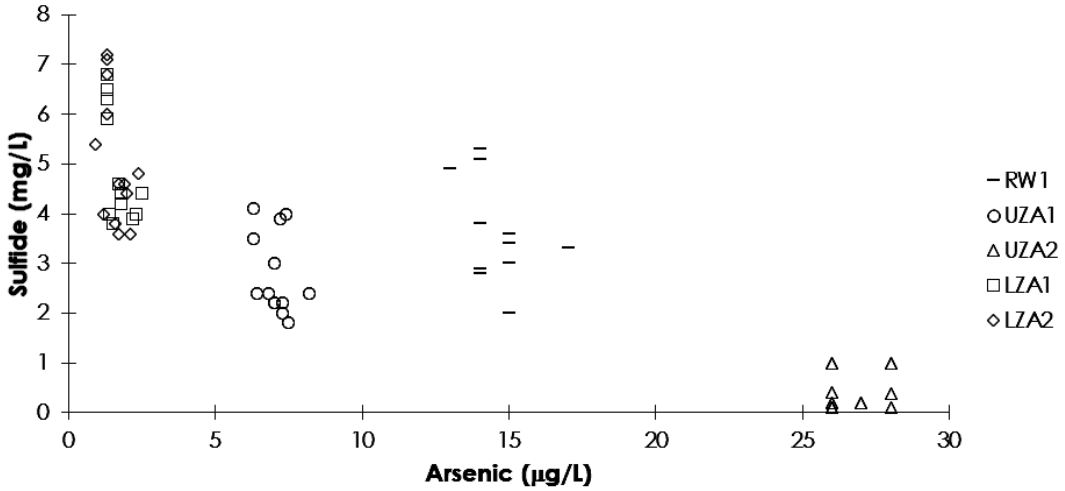

Figure 11. Plot of concentrations of sulfide and arsenic for Upper Zone and Lower Zone groundwaters.

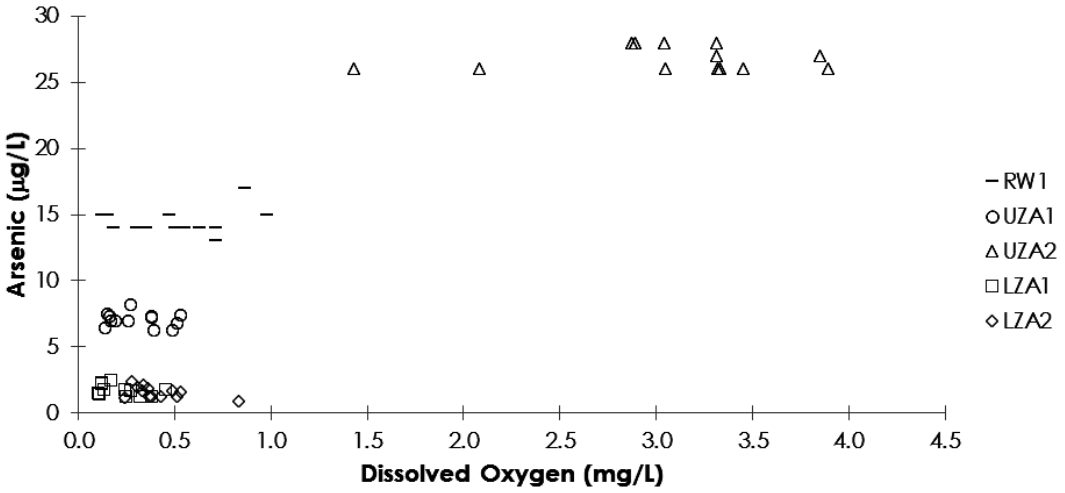

Figure 12. Plot concentrations of arsenic and dissolved oxygen in Upper Zone and Lower Zone groundwaters.

Discussion

Figures 6 through 12 show that anoxic conditions predominate with respect to the upper and lower zones. Anoxic conditions are most pronounced with respect to LZA1 and LZA2, and slightly less so with regard to UZA1 and RW1. Slight differences in ORP might account for much of the difference in arsenic concentrations among these wells, although the higher TDS and chloride concentrations (i.e., higher ionic strength) of RW1 might indicate the potential for a combination of reductive dissolution and competitive desorption as factors controlling the occurrence of arsenic in higher-TDS groundwaters of the area (Liu et al. 2014).

By any objective set of geochemical criteria, the local groundwater system, exclusive of UZA2, is anoxic. Where DO concentrations are reported as 0.50 mg/L or less, iron concentrations are typically less than 0.04 mg/L, ORP is negative, and sulfide concentrations range from 2 to 7 mg/L. It is noted that DO measurements at concentrations less than 0.5 mg/L in the presence of sulfide might not be reliable. However, the ORP measurements and other redox-related data indicate that geochemical conditions are moderately to strongly anoxic, and that conditions as represented by data from LZA1 and LZA2 underscore the high probability that pyrite, the mineral with which arsenic is most commonly associated in the Suwannee

Limestone, is stable and unlikely to yield arsenic at concentrations greater than 3 µg/L.

Additional support for the dominance of anoxic conditions is found in Eh-pH diagrams on which data from the wells are plotted (Figures 13 and 14). Eh is an electrochemical description of the oxidation state for a system in equilibrium (Bethke 2008). It can be derived from ORP by adding a correction factor (typically an average of 200 millivolts (mV)) to ORP. The correction factor is an empirical adjustment to account for the difference between measured and theoretical voltages of electrodes and electrode solutions at different temperatures.

Figures 13 and 14 show the stability fields of ferrous iron (Fe^{2+}) and ferric iron (Fe^{3+}) under different combinations of Eh and pH, within the stability boundaries of water. The figures were constructed with the Act2 module of Geochemist's Workbench©, v. 10, assuming a system temperature of 25 °C. Eh-pH diagrams can be constructed to include different solid mineral phases, such as pyrite and hematite. Solid phases are not included here because the figures are intended to show only the combinations of Eh and pH under which dissolved iron species predominate in aqueous systems at 25 °C.

In Figure 13, the data from UZA1 plot within the Fe^{2+} field – an indication that ferrous iron is the dominant species of dissolved iron. Data from UZA2, however, lie within the Fe^{2}+ and $Fe(OH)_3$ fields – a response to conditions that vary between oxidizing ($Fe(OH)_3$) and reducing (Fe^{2+}) over the narrow pH range for those samples.

With regard to Figure 14, the points from RW1, LZA1, and LZA2 all lie within the Fe^{2+} stability field, over approximately the same narrow range of pH measurements. This indicates that the aqueous environment within the lower zone is reducing and relatively uniform between the LZA1/LZA2 locations and the deeper zone represented by the supply well that provides water for RW1.

Norton et al. (2014) examined potential mineralogical sources of arsenic in conjunction with the Project. The investigation involved examination of petrographic thin sections of 14 core samples from the Suwannee Limestone. The core samples were collected from a depth range of 274-289 feet below land surface, the intended recharge depth. Leaching tests were also conducted on selected samples from the same depth range.

The core samples are described as "peloidal packstone to grainstone with traces of quartz grains and clay" (Norton et al. 2014, 51). The specimens include "sparse grains of pyrite, ilmenite, rutile, iron oxide, chalcopyrite and possible pentlandite (nickel iron sulfide) as well as a few unidentified grains with higher concentrations of the rare earth elements" (Norton et al. 2014, 51). Based on data from associated bulk rock and leaching analyses, Norton et al. (2014) concluded:

- Arsenic concentrations in bulk samples range from below detection limits (< 1 mg/kg) to 2 mg/kg.
- Arsenic concentrations in pyrite range from below detection limits to 1400 (± 350) mg/kg – or near the lower end of the range of 100 to 11,200 mg/kg (the average is 2300 mg/kg) reported by others for the Suwannee Limestone.
- Leaching tests document arsenic concentrations increasing from approximately 2 to 13 µg/L in source water with DO concentrations ranging from less than 10 to 60 µg/L and sharply lower concentrations of arsenic at DO greater than 600 µg/L. Arsenic was detected in solution above 10 µg/L in five of nine leaching tests.
- There is a strong correlation (R^2 = 0.914) between the mass of leached arsenic and source water DO concentration up to 60 µg/L.
- The mass of arsenic released to groundwater is highly correlated with other redox-sensitive metals, especially molybdenum (R^2 = 0.943) and cadmium (R^2 = 0.880).

Conclusions

This paper is a highly condensed summary of work conducted to address a narrow range of questions related to the suitability of the Suwannee Limestone to be an injection zone for purified reclaimed water. The principal geochemical matter of concern is the potential for dissolution of minerals in which arsenic is sequestered either by precipitation/co-precipitation or adsorption. The iron sulfide mineral pyrite has been shown to be the most common source of arsenic at ASR sites in Florida (Arthur, Dabous, and Cowart 2002; Jones and Pichler 2007)

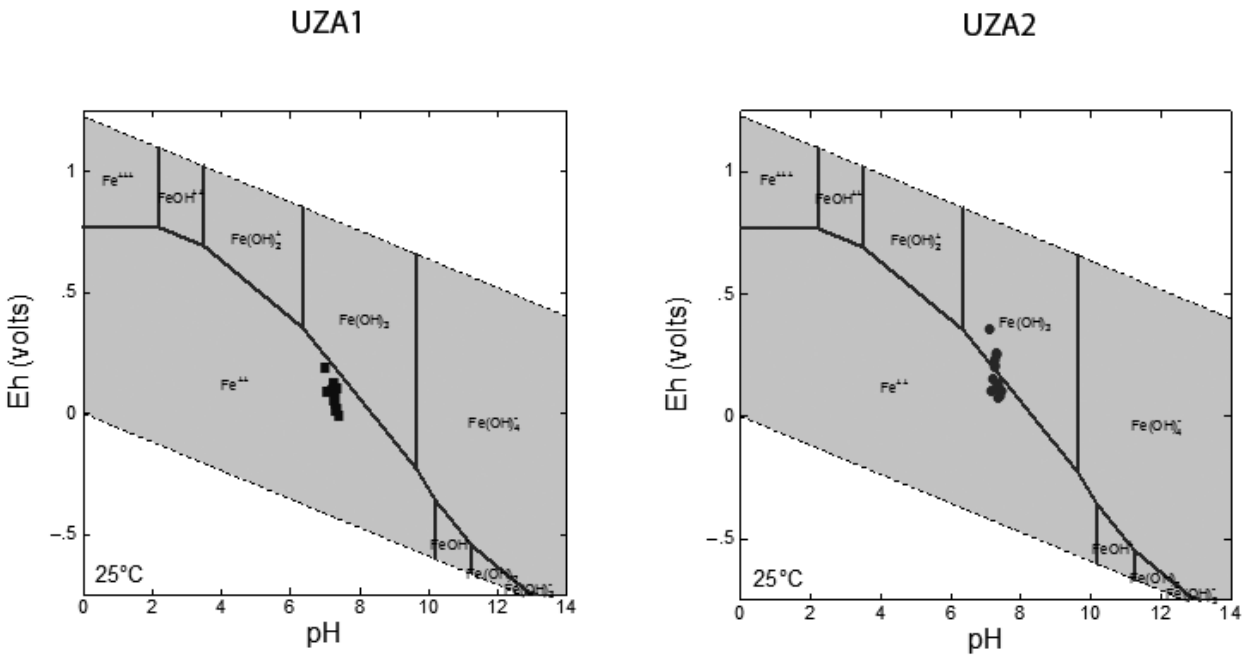

Figure 13. Eh-pH diagrams (iron-oxygen system), Upper Zone.

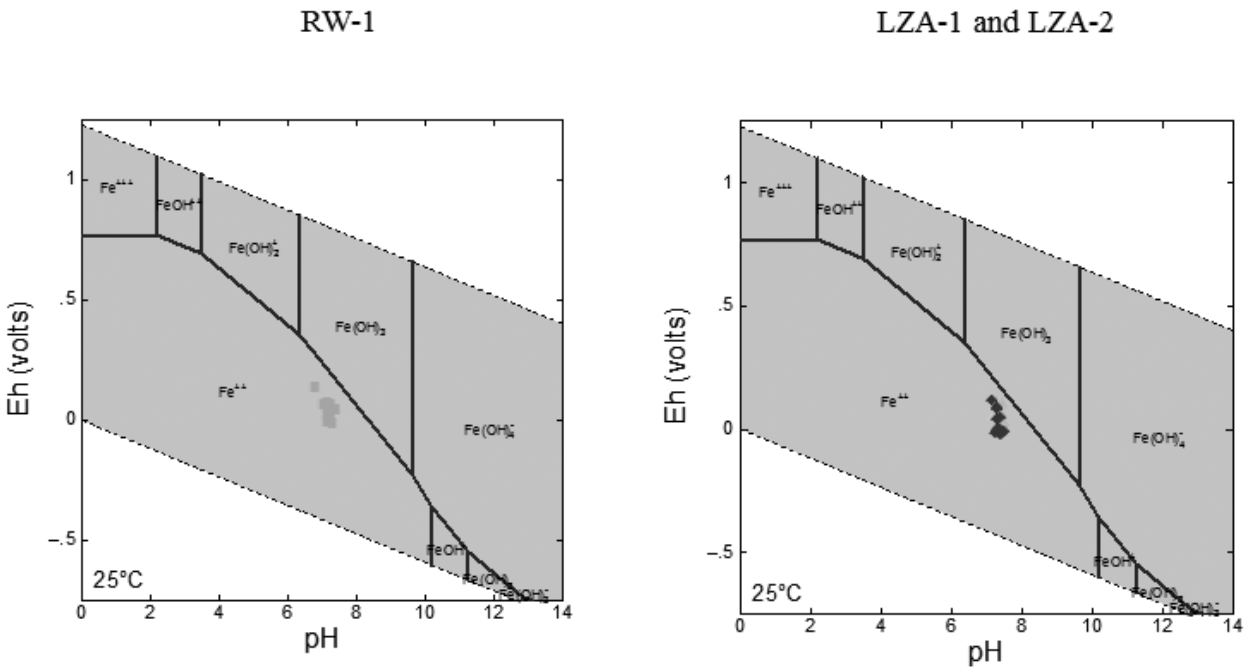

Figure 14. Eh-pH diagrams (iron-oxygen system), Lower Zone.

and at Clearwater (Norton et al. 2014). Other metals of significance at Clearwater are molybdenum and cadmium (Norton et al. 2014). The results of this investigation are consistent with those of other researchers (Arthur, Dabous, and Cowart 2002; Jones and Pichler 2007) and further identify and explain reduction-oxidation factors that control the stability of iron sulfide minerals disseminated within the matrix of the Suwannee Limestone. Significant observations and conclusions from this investigation are the following:

- Upper- and lower-zone groundwaters are calcium-sodium-chloride and sodium-chloride in composition, and all appear to be mixtures of freshwater and seawater. TDS and chloride concentrations are higher in lower-zone groundwaters than in upper-zone groundwaters.
- The lower zone of the Suwannee Limestone is anoxic. This is clearly indicated by negative ORP measurements and DO concentrations less than 0.5 mg/L, sulfide concentrations ranging from 2 to 7 mg/L, and iron concentrations less than 0.04 mg/L.
- Sulfide concentrations greater than 2 mg/L in samples of lower-zone groundwater indicate that the lower zone is anoxic and that DO has probably been exhausted within that part of the groundwater system.
- The upper zone is anoxic in the vicinity of UZA1, but positive ORP measurements and DO concentrations ranging from 1.4 to 3.9 mg/L in samples from UZA2 indicate oxidizing conditions in the vicinity of that well.
- Arsenic occurs naturally in groundwaters of the upper and lower zones. The lowest concentrations of arsenic (0.93 to 2.5 µg/L) are associated with LZA1 and LZA2, wells with consistently negative ORP measurements, elevated sulfide concentrations, and low concentrations of iron. The highest arsenic concentrations (26 to 28 µg/L) are found in samples from UZA2.
- The low iron concentrations (0.020 to 0.046 µg/L) of LZA1 and LZA2 indicate that pyrite, the iron sulfide mineral with which arsenic is most commonly associated in the Suwannee Limestone, is stable in the anoxic environment of the lower zone.
- The low arsenic concentrations associated with LZA1 and LZA2 are probably related to the sequestration of arsenic by pyrite, and the high arsenic concentrations associated with UZA2 are probably related to the release of arsenic through the dissolution of pyrite in that part of the upper zone.
- Arsenic concentrations of approximately 14 µg/L associated with RW1 could be related to competitive desorption in the higher-TDS groundwater produced by the source well. The TDS of this water ranged from 1100 to 1300 mg/L and was higher than the TDS for other lower-zone wells (690 to 860 mg/L).

With regard to the Clearwater AR project, this investigation is one of many components of the City's intensive program to assess the lithological, stratigraphic, hydrogeological, and geochemical characteristics of the Suwannee Limestone and groundwater. The results of this investigation are consistent with those of geochemical studies at other project sites in Florida. As such, the data and associated analyses form a sound basis to help the City's team of hydrogeologists, engineers, and consultants understand the availability of arsenic and the conditions under which it is liberated from the matrix of the Suwannee Limestone, and to design operational procedures to maintain the stability of the sulfide minerals with which arsenic is associated.

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Well Decommission, Policy, Processes, and the Urban Environment

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Abstract: An afterthought in many well construction and design plans is the regulatory compliant destruction or decommission of the well at some point in the future. Agricultural lands that evolved into urban areas in southern California are likely to have thousands of lost or unmanaged wells. For the general public, landowners, and others these wells pose safety, environmental, logistical, and monetary issues. This discussion will cover regulatory guidelines, procedures, and case examples of well decommissioning. In California, regulations for well destruction are rooted in Department of Water Resources bulletins, state water code, legislation, local ordinance, and drilling contractor licensing compliance. In most cases the county departments of public health take the lead role for permitting and enforcing well destruction requirements. Well terminology used can be misleading; for example, by state law, an abandoned well is a noncompliant hazard- not a safely destroyed or decommissioned well adhering to standards. Permitted well decommission typically involves cement-bentonite sealing and casing perforation. With the goal of effective prevention of surface liquid intrusion or aquifer cross contamination, sometimes explosives, over-drilling, and other practices are applied. Challenges are frequent in locating and decommissioning abandoned wells. Lost records lead to reconnaissance, geophysical investigations, and excavation of unintended anomalies. Urban development may surround an abandoned well within a structure, foundation, or utilities. Environmentally sensitive areas will affect standard techniques. With increasing water demand and urbanization, effective well decommission is important for analysis and regulatory emphasis.

Keywords: *destruction, abandoned, shot perforation, California Water Code, permits*

This discussion considers the problem of abandoned or unmanaged wells, reviews common terminology, regulations, well decommission practices, and provides program case examples of well decommission. Emphasis is mainly on water well decommission, as these are the types of wells directly associated with water resources and often have larger casing diameters and open internal designs that can increase potential environmental and/or safety hazards. However, in practice, many other well types in addition to water wells are encountered in an unmanaged condition.

Applicable experiences of the author are mostly within the state of California regulatory environment, thus some definitions, accepted procedures, and compliance matter examples given will not apply directly in other states. However, the practices, issues, and technical aspects of well decommission are universal. Regulatory programs administered in all U.S. states and counties are logically similar or identical in technical implementation. Abandoned well safety, environmental, and water resources issues affect almost all urban areas internationally.

Overall, the importance of well management programs for large land owners, property managers, utilities, municipalities, and water supply operations is encouraged, and awareness is advocated for all those who work with wells. University students and faculty usually do not cover well decommissioning in detail, but it is an extremely important technical topic to consider in limiting contamination between stacked aquifers, in addition to addressing public safety hazards in our growing urban environment. Although well destruction in an ideal case gives the appearance of a simple procedure, most professionals realize that the practical complexities of effective and compliant well destruction in multi-aquifer systems, environmentally sensitive locations, urbanized areas, and other situations are challenging and surprisingly complex.

Unknown wells are commonly discovered during field reconnaissance by trained personnel or from a government records review of large land areas in urban or agricultural areas. These will often be water wells, but also might include oil wells, monitoring wells, grounding wells, cathodic wells, injection wells, and dry wells for water drainage, among others. Regulations for compliant destruction or decommission of wells are in-place, but often are unknown, not enforced, or were not correctly applied by former well owners when property was vacated. When a well ends its useful service life, active regulations require that it be destroyed. The cost of compliant well destruction is another factor that may cause some well owners to forget the unmanaged wells. Aside from the cost of material and drilling contractor expertise, well decommission agency permit fees typically exceed $1,000 per individual well.

With over 800,000 documented well completion records in California Department of Water Resources files, it is likely that thousands of unmanaged deep holes/wells without any discrete known location exist throughout the environment. The lost or unmanaged well problem is present in all states and internationally. As rural areas continue to urbanize, these unmanaged or abandoned wells present safety, environmental, financial, and water resource management challenges.

The urban environment increases the possibility that unmanaged wells will act as conduits for organic and inorganic contaminants into potable groundwater supply. Spills and industrial chemical or domestic waste storage can find rapid transport through damaged abandoned wells and vertical groundwater gradients are increased as deeper wells are installed to avoid surface contaminated waters. The issue has been recognized by the groundwater scientific community for some time and the political structure has been established to mitigate groundwater contamination through programs including well decommission and permitting. Some of these statutes are discussed in the following section.

Environmental Policy and Regulatory Framework

A basis for well management in California is Department of Water Resources Bulletin 74-81 and the updated Bulletin 74-90 (California Department of Water Resources 1981, 1990, 2013). These well standards documents cover well design, construction, renovation, and destruction of wells including water wells, monitoring wells, and cathodic protection wells. The standards given in Bulletins 74-81 and 74-90 have been adopted by the State Water Resources Control Board pursuant to California Water Code Section 13801. For state wide implementation, under the Water Code, a Model Well Ordinance was established in 1989. The judgment defines required minimum standards for wells in California, and it includes permitting and enforcement authorities for state and local agencies. If a local government does not adopt an ordinance that meets or exceeds the standards, the state Model Well Ordinance 13801 will apply and take effect within the local jurisdiction. In the majority of California, the county government has established local ordinance that meets or exceeds the minimum well standards to regulate compliance. In other states, and internationally different but similar codes and practices have been established to protect the public and environment from unmanaged wells (American Water Works Association 1984; Gordon and Koch 1988; Kraemer et al. 1991)

Well permits are required by the regulatory agency for initial well construction, renovation, and final destruction. When planning the construction, renovation, or destruction of a well, the county in which the well is located should be contacted to begin the permitting process. County staff will advise if a separate local agency has adopted an ordinance for jurisdiction. If a local city agency has taken well permit authority by court ordinance, no additional county permit should be necessary. In some less common cases, a city or local governmental authority holds an aspect of well permitting authority via ordinance and may require a well permit in addition to the county. The legal basis for this is unclear, but in some cases it might reflect the addition of more stringent local standards related to business use or zoning that is not captured in the county permit.

In adjudicated groundwater basins in California, the Watermaster may also have a stake in water well decommission permitting in accordance with administrative orders, local

court, or administrative board judgements. For example, the Main San Gabriel Groundwater Basin Watermaster of Los Angeles County follows this process (Main San Gabriel Basin Watermaster 1992). The Watermaster may also require additional approval, installation permits, renovation permits, and field inspection in advance of well destruction activity. With statewide adjudication of every California groundwater basin coming into effect under the Sustainable Groundwater Management Act, by the 2017 implementation date, additional Watermaster regulation of wells and well destruction-decommission is expected.

An important part of the well legislation regulatory framework, especially for those who are trying to locate or determine prior existence of wells on a property, is the drillers license requirement to submit a Well Completion Report to the State Department of Water Resources whenever a well is installed. California Water Code section 13752 was amended in 2015 by SB 83 to allow public access to these Well Completion Reports. Most of the older Well Completion Reports seldom give accurate well locations, but at least give an indication of an address or cross streets. Other agency file well permits are available; however for older well construction information, the Well Completion Report record is the more comprehensive resource.

Oil, natural gas, and geothermal resources wells that may exist on a property are regulated separately from water wells. These fall under jurisdiction of the California Department of Conservation, Division of Oil, Gas, and Geothermal Resources, "DOGGR" (California Department of Conservation, Division of Oil, Gas, and Geothermal Resources 2015). DOGGR maintains a database of these wells and separate permitting, terminology, and reporting requirements apply. Though not the emphasis of this paper, it should be noted that somewhat similar procedures apply to destruction of DOGGR wells, but the complexity of much greater depth, safety, formation pressures, and environmental issues should not be underestimated. A property manager should explore past ownership and other options to avoid destruction expense if such a well is located on the property.

Well Decommission Terminology, Definitions, and Well Classes

Various types of wells/subsurface excavations may be encountered on a property. The surface appearance, subsurface design, and management requirements are not the same for different wells, but are important factors when considering potential safety or environmental risks associated with well management and destruction. For example, properly installed electrical ground wells, cathodic wells, and geotechnical borings are relatively small in diameter. These wells are cement grouted/closed as part of the installation design or are shallow and grouted immediately following soil sampling. If compliantly installed, when left unattended these permitted well types seldom create eminent environmental or safety issues. For definition, the common well types are given in Table 1.

A source of confusion for a well manager is the use of the term well abandonment or abandoned well. In California an abandoned well has the legal definition of, "any well whose original purpose and use has been permanently discontinued or which is in such a state of disrepair that it cannot be used for its original purpose" (California Department of Water Resources 1981, 1990). An abandoned well is in a non-compliant condition and subject to compliance actions and/or fines from the responsible regulatory authority. Regulations require that an abandoned well must be destroyed/decommissioned. Confusion results when individuals and regulatory agencies use the term abandoned well or well abandonment to describe wells that were destroyed or decommissioned in accordance with the appropriate regulatory procedures, or to describe the actual compliant well destruction procedure. However, the terminology is ingrained and will not be corrected easily, and some states differ in legal definition of the term (Nye 1987; Sara 1987). For clarity, the correct terminology and regulatory difference is good to recognize, and that many entities incorrectly use abandoned well synonymously with compliant well destruction or decommission. If a well owner or manager receives a notice of violation for abandoned wells on their property, with fines and action required,

Table 1. Well Types/Classes.

Well Class	Description
Cathodic protection wells	Wells installed to protect equipment or subsurface piping from corrosion.
Grounding wells/electrodes	Wells installed with in-place conductive copper wire or solid electrode to establish electrical ground for high voltage.
Monitoring wells	Water level, water quality, and environmental sampling.
Oil and natural gas wells	Wells installed for extraction of petroleum or petroleum related products.
Open borings that encounter primary groundwater	Historical wells from poorly managed construction, mining, or exploratory operations.
Soil gas monitoring probes or vapor extraction wells	Typically wells mandated for remediation projects or landfill methane recovery.
Water recharge wells/ injection wells/UIC wells	Water resource replenishment, water drainage, or water disposal.
Water wells	Includes: Production/irrigation/agricultural/industrial/non-potable use water wells/ potable water wells/domestic water wells/fire suppression water supply wells.
Remediation/environmental pumping wells	Generally similar to water wells but have additional considerations and regulatory agency involvement for destruction.

the regulatory distinction of terms is understood. Other definitions or terminology applicable to well destruction practices and this discussion are listed and pictured in Table 2 and Figure 1.

Well Decommission-Destruction Compliant Procedures

Well destruction procedures have the goal of effectively sealing the well casing and borehole to prevent water mixing between discrete aquifer zones and to prevent intrusion of surface waters, liquids, or other contaminants. In a simple explanation, steps include removing equipment, pumps, pipe, and obstructions; evaluating, removing, or perforating well casing; and sealing the well or borehole with cement-bentonite grout. Procedures, grout seals, and techniques for evaluating effectiveness have been noted in other studies (Lamb and Kinney 1989; Edil et al. 1992; Christman et al. 2002).

Securing permits, removing surface infrastructure, clearing downhole water pumps, pipe, and other equipment come first. Unfortunately, in older abandoned wells this equipment is sometimes seized in place and can only be partially removed and cut to a certain depth. In these cases, a mitigated solution is reached in collaboration with the permit agency and the later procedure steps are modified in a way that adequately protects the environment.

Before destruction, perforation, or sealing procedures, a cleared well is investigated to determine its condition, details of construction (if unknown), and whether there are obstructions that will cause bridging and interfere with the process of filling and sealing. This may include bailing the well of turbid fluids and video inspection of the well. If no construction records are available for a well, the permit agency will typically require a videolog to locate screen or perforation zones. The videolog assists in determining casing condition and zones that will be targeted for mechanical perforation or penetration with explosives.

The acceptable sealing procedure will depend on well design, depth, casing diameter, and site

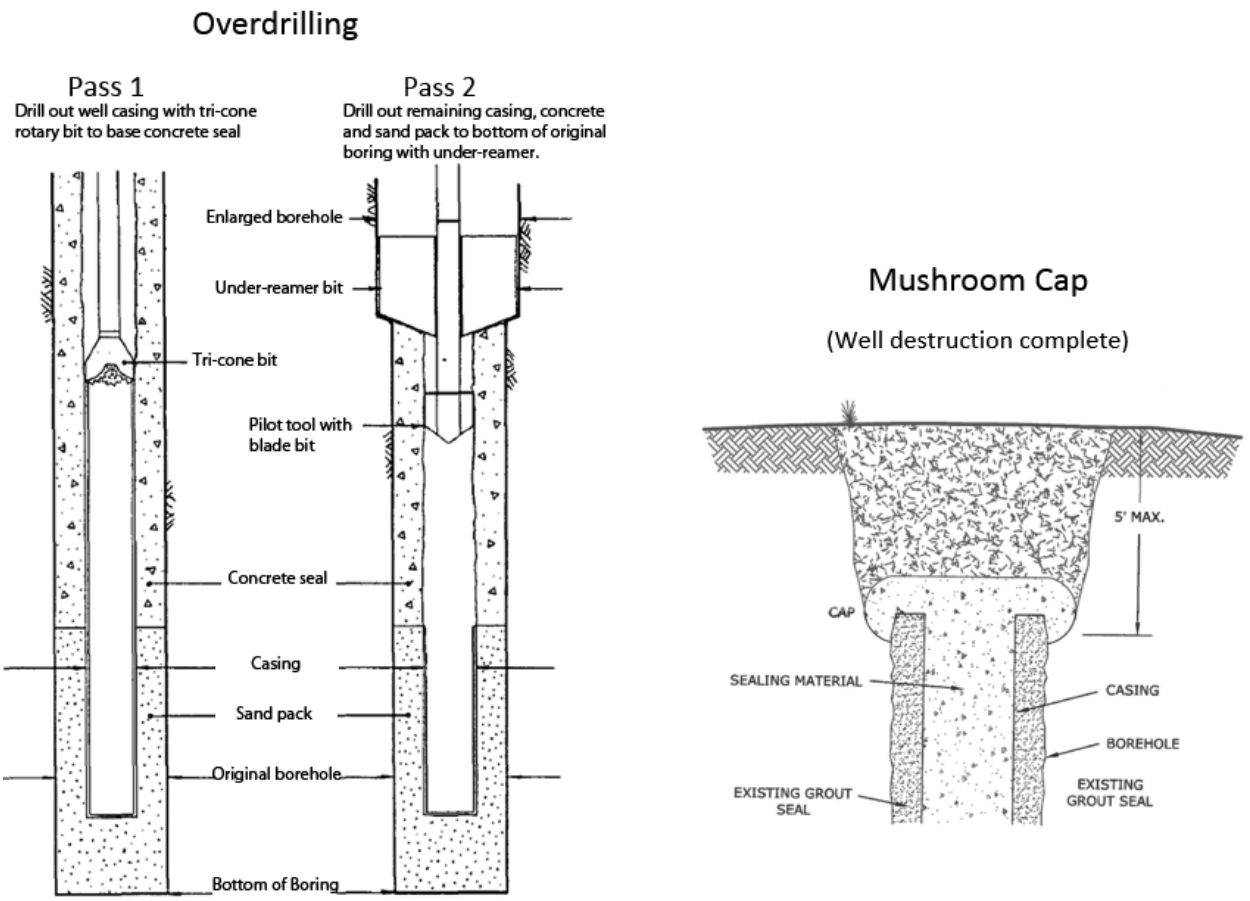

Figure 1. Overdrilling and Mushroom Cap, well destruction step examples. Redrawn from Lamb and Kinney 1989 and California Department of Water Resources 2013.

specific conditions. In advance, the volume of grout needed to fill the well bore and gravel pack zone is calculated. Shallow wells or dry wells are generally acceptable for simple pressure grout techniques or casing pull and pressure grouting of the remaining borehole. In all cases the grout is emplaced from the bottom of the well upward using tremie pipe methods (in other terms, the grout is not allowed to free-fall into the well). The delivery method is appropriate for quality grout seals that need emplacement underwater. Small diameter wells of relatively shallow construction are good candidates for overdrilling techniques in which larger diameter hollow stem augers are advanced. After target depth is reached, the casing is pulled and the remaining borehole is pressure grouted. Overdrilling steel casings with rock bit tools for decommission can be difficult, costly work, but in some situations necessary.

In the current technical, regulatory, and business environment, the preferred sealing procedure for deeper, large diameter water wells involves filling the well bore with grout and applying explosive perforation charges via wireline at set intervals. Use of explosives in well work and decommission is not new but the advantages have been noted (California Department of Water Resources 1990; Cawley 2005). Additional grout is added immediately after the charges are detonated. The grout volume delivered using this technique is usually considerably more than what is calculated to fill void space, so an effective seal is assured. Some agencies require the explosive perforation method in California if no records of well construction details are available. Mechanical casing perforation followed by pressure grouting is applied when required to assure sealing or when other considerations do not allow use of explosives.

Table 2: Well Decommission Procedures and Terminology Definitions.

Procedure or Terminology	Definition
Cement-bentonite grout	Well sealing material composed of standard C150 Portland cement and approximately 5% bentonite. It is commonly applied in subsurface well sealing and destruction.
Tremie pipe	The tremie concrete placement method uses a tremie pipe, through which concrete is placed below the water level in a well. The tremie pipe extends to the bottom of the well and is raised as the grout fills the well. The lower end of the pipe is kept immersed in fresh concrete so that the rising concrete from the bottom displaces the water without washing out the cement content, and standing liquid in the well is displaced. The method allows void free sealing of a well.
Pressure grouting	The simplest compliant well destruction method that involves pressure emplacement of wet cement-bentonite grout by tremie pipe.
Shot perforation	Perforation of well casing to allow thorough penetration and grout sealing of well and borehole void spaces. The procedure involves shell shots within a well to be destroyed or synchronized explosive charges.
Mechanical perforation	Mechanical perforation of well casing involving “mills knife” steel cutting tools or similar metal cutting or perforating devices of mechanical means.
Mushroom cap	The inverted conical or mushroom shaped cement grout cap emplaced at the top of a grout sealed well casing. The cap is emplaced after well casing is cut at least three feet or more below existing surface grade. In the subsurface, the shape helps deflect potential polluting liquid accumulation in the area above the former well with an impermeable cap.
Overdrilling	Involves complete removal of a well casing by drilling on top with oversized hollow stem augers or by combinations of hard rock bit pulverization of existing well casing. The overdrilled borehole is pressure grout sealed when casing is removed to the target depth.

In some cases, a well found abandoned may have future value to be reactivated for beneficial water use or monitoring purposes. In this situation, an intent for future use may be filed with the agency and, thus, it is assured that the well is appropriately secured for safety and environmental protection. Exercising this option is usually only feasible when the subsurface construction design is known. California agencies do not allow wells with questionable annular seals to be reactivated or remain in place without action. Reactivation may require relatively expensive neutron logging in addition to video and other geophysical logs to evaluate whether the annular seal emplaced during original construction is adequate.

Case Examples: Well Management

The example case of well management for this study is one adopted by a California electric utility. The author is employed as a project manager with the utility, with responsibilities including managing well standards and well decommissioning. The utility is a large land owner in California and the land is generally associated with electric power transmission corridors and electrical supply operations locations. The utility has an operational history of over 100 years, and urban development in the region has grown significantly. In 2012, the inventory of known wells owned by the utility mostly fell under the categories of electrical grounding wells, cathodic wells, and in some cases water wells where limited infrastructure existed to supply industrial water, irrigation, or drinking water. Environmental monitoring wells installed for due diligence or managed under specific regulatory directives made up another large portion of the utility well inventory. The known wells were managed relatively independently and effectively within the utility operating units.

However, in 2012, fifteen unmanaged water wells were discovered abandoned on utility property by the regulatory agency in active urban locations. The wells were found beneath power lines within transmission line corridors or within adjacent utility-owned land leased to others. Farms, plant nurseries, and similar businesses were often present, or had been operating. The wells were in varying states of condition and their exact origin was unknown. Some of the wells were apparently on the land when the utility purchased it, others may have been installed possibly illegally by others and left behind when the property was vacated. In any case, the finding revealed that the utility owned, by default, far more wells than was previously realized and that along with this ownership came regulatory, environmental, and safety responsibilities. Examples of unmanaged wells are shown in Figure 2.

The finding of previously unknown and unmanaged wells triggered immediate action to secure those that posed potential environmental or safety hazards. Following the temporary securement, assessment, permitting, and qualified contractor assignment, the abandoned wells identified by the agency were destroyed under approved permits in 2013. Concurrently a well management program was established by the utility to centralize the well inventory, evaluate government records, and field check utility lands to locate unmanaged wells. Under the program many additional unmanaged wells of unknown origin were identified to exist or formerly exist on the utility property. Steps were taken to field locate the additional wells, leading to many being found and destroyed/decommissioned. The well management and decommissioning program has continued and wells have been field identified, secured, and decommissioned annually since the initial program work was established in 2013.

Establishing well management and installation policies to prevent occurrence or reoccurrence of future well issues is advisable to land managers. Not granting license to third party lessees to install wells is a mitigating recommendation to land managers. There are offsetting exceptions that might allow a well installation, but they should be considered carefully and with knowledge of the encumbrance to the property and potential environmental dangers associated with insolvent or irresponsible lessees. As is clear from the proceeding and following examples, wells installed on a property come with significant long-term management obligations.

Finding the location of unmanaged wells based on available references can be problematic. Agency records will exist that reference locations that cannot be found in the field or give the same well identification at different locations. Older records seldom give discrete locations and even relatively recent records may be flawed. The well may have been backfilled or grouted without permit record, may have already been destroyed under a different well name, or was buried, or otherwise obscured. In some cases, the well is lost or has dubious existence and all that can be done is to save the reference information in case construction or development encounters it in the future.

In the case study, some of the abandoned buried wells have been successfully located with the aid of geophysical surveys. The surveys considered practical for locating well-type structures include electromagnetic (EM), magnetic, and ground penetrating radar (GPR). Steel casings can be detected using an EM instrument (buried metal detector) or a magnetometer (which measures the strength of the earth's naturally occurring magnetic field). The magnetic anomalies can be investigated by shallow digging. Magnetic techniques are especially effective in unpaved, unimproved areas with good egress. Other buried metal items like light bases or pipes can result in excavation of false anomalies. However, impressive results have been yielded when interference is minor and the well actually exists within the survey area. In other cases, a GPR survey may be used when the well can be of non-metal construction. Due to cost considerations, a GPR survey is most effective when focused on a few key areas of a possible well site (Shortt 2010).

Destruction or decommission of unmanaged wells in urban areas presents other procedural and logistic challenges. Buildings, roads, freeways, and rail lines may be installed near or on top of wells; abandoned wells are commonly looted of metal, vandalized, and other equipment is left in a potentially hazardous state. Homeless camps develop around or near many abandoned wells, increasing public danger and making access difficult

or dangerous for well workers. Increased chances for pollution or waste disposal issues in the area of the well can be expected in these urban situations.

Additionally challenging are abandoned wells found in land that has become a designated critical area for biological habitat, archaeological significance, or those that require special federal waterway permitting before field work may begin. Two 2014 abandoned case well locations for the utility-adjacent urban areas also had this environmental permitting factor to consider. Frequent access to the wells by teens and the transient community aroused public complaints and triggered agency calls for action. The wells, however, were also in a flood zone biologically sensitive area, and required archeological work to meet regulatory requirements before destruction-decommission could proceed. The options were evaluated, reviewed with the applicable agencies, and helicopter equipment and material unloading/loading was selected as the most cost-effective method to accomplish the work. Creation of temporary roads and permits would have resulted in long-term studies, a prolonged schedule, and may have required significant mitigation of separate land areas. Other variations of options proved to have similar issues and were cost-prohibitive or unacceptable from a regulatory perspective. In addition to the equipment drops, the helicopter was applied to directly lifting pump and pipe from the wells (Figure 3).

Conclusions

In summary, well management and regulatory compliant destruction or decommission of wells is worthy of additional attention from the environmental, industrial, technical, and regulatory communities. While in the simplest sense well decommission may be dismissed as filling an open hole in the ground with concrete, in reality it is much more complicated. Deep wells penetrating many aquifer zones and other wells with unknown designs, as well as environmental, archeological, safety, permitting, and surface issues, add to the procedural complexity. New laws and technical procedures have developed in well decommissioning since the 1980's; newer unique procedures including explosives, casing perforation, and applications of helicopters are discussed in the case study of this manuscript.

Larger commercial organizations, landowners, and government land managers could benefit by establishing internal well management programs to alleviate potential issues as the utility in the case study presented has done. Those who install wells or knowingly or unknowingly come to own wells, hold the long-term responsibility to safely and compliantly destroy the wells at a future time. Major oil and gas companies may already have well management programs, but reconsideration is needed based on notable poorly managed wells that have failed recently resulting in catastrophic gas and oil releases with environmental, safety, and financial consequences.

Existing regulations in California and elsewhere provide a good framework and guidance for safe and effective decommission of wells. The California State Water Resources Control Board has established a model ordinance that includes well decommission and management practices. However, in some instances, jurisdiction is not clear and permits overlap between municipalities or are duplicated. Regulatory staff experience with wells and availability to review and consider permit approval may be limited. County governments and environmental or public health departments are the agencies primarily responsible for regulation and enforcement of well standards in California and may be contacted for advice if matters of jurisdiction are in question. The well regulations are not unique; generally they are similar throughout the U.S. and in most cases the compliant procedures can be determined and implemented in collaboration with the agencies.

With increasing urbanization and dwindling surface water resources, groundwater will continue to grow in importance, and with thousands of lost or unmanaged wells in California alone, water quality is at risk. Well decommission is an important component of water resource management and the volume, variety, and surface appearance of well types already present in the environment often is unrecognized. It is the author's desire to raise awareness of the issue, help educate those who may not recognize a well in the field, and advocate well decommission practices and programs that are protective of safety and the environment.



Figure 2. Case study abandoned well examples, 2012.



Figure 3. Well decommission heavy lift helicopter assisted well pump and pipe removal, 2014. On left: Rigging to lift well turbine pump and pipe. On right: Lifting to remove from well.

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Issues of Stream-Aquifer Interactions in Grout Lined Channels in Urban Watersheds

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Abstract: Grout lined storm drains are used worldwide to convey storm runoff away from urban areas. They also convey dry weather flows such as car wash runoff, lawn runoff, and other "nuisance flows" in urban centers. Areas in Southern California and elsewhere have shallow water tables where dry weather flows in grout lined storm drains consist primarily of perennial groundwater baseflows. In these areas, groundwater leaks through cracks, joints, weepholes, and intentional dewatering structures into storm drains and eventually flows into natural channels, coastal wetlands, and estuaries. The "lined" sections of these channels are not always impervious; in fact, the lining of the channels often affords excellent opportunities to collect groundwater samples at discrete points that are usually not present along unlined channels where well control is sparse or where technical or legal issues prevent installation of piezometers. Frequently, as much as 95% of the nutrient and trace element loading to urban catchments during dry weather is from seepage of groundwater that is laden with pollutants. This project presents several examples of our experiences in conducting stream/aquifer studies in lined and semi-lined channels in Southern California. Sub-projects include evolution of groundwater quality along flowpaths, detection of unwanted recharge mounds in shallow groundwater systems due to leaky water main pipes, and comparisons of water quality in lined and unlined sections of a creek. Tools that are utilized and described in the paper include isotopic tracers; trace element and standard inorganic constituents; and innovative sampling methods.

Keywords: *stable water isotopes, urban groundwater, selenium, nitrate, hyporheic zone, liquefaction*

In urban areas many channels are lined with grout concrete. Some creeks are lined nearly 100% along their full extent, while others may be lined only beneath bridge overpasses. These channel linings are installed in order to route stormwater runoff away from urban areas as fast as possible and to protect urban infrastructure from erosion. One does not usually think of conducting research of stream/aquifer interactions along grout lined channels since the channels are often considered to be impervious to fluid flow, but typically they are not completely impervious. In this paper, we document our experiences in evaluating stream/aquifer interactions along grout lined storm drains and channels. The topics include descriptions of conduits as pathways for fluid flow between lined channels and groundwater. Three case studies of stream/aquifer interactions are also described including technical and policy issues related to local water resources management. Pertinent additional topics are addressed, including problems of composite sampling along lined channels, discussion of safety issues for any personnel engaged in this type of work, and discussion of opportunities for conducting research.

Channel Lining and Permeability Conduits along Lined Channels

Lining of channels is known to reduce streambed permeability and is done to reduce infiltration from rivers to underlying aquifers (Leggat and Davis 1966; Meyer 1976; Coachella Valley Water District 2006). Lined channels are not impervious to fluid flow and may in fact be moderately permeable depending on channel construction and

condition/weathering of the grout lining (Hibbs and Sharp 2012). Conditions vary but the flux between groundwater and surface water along lined channels generally depends on: 1) density of cracks and separation voids in the channel lining, 2) abundance and density of weepholes, 3) hydraulic conductivity of aquifer and hydraulic gradient near the channel, and 4) general construction and condition of the lined channel.

Typically, flood flows temporarily reverse hydraulic gradients in normally gaining urban channels, creating brief periods of aquifer recharge through the same weepholes that were installed to direct groundwater into the channel. Where many orifices exist, aquifers connected to lined stream channels can receive moderate amounts of recharge during flood flows due to higher head in the channel.

We have identified many types of permeability conduits along grout lined channels. The most common type is the weephole, installed along channel sides and bottoms to allow groundwater to seep out into the channel, thus lowering the water table (Figure 1). Weepholes reduce pore fluid pressure on the channel lining, thereby protecting the lining from cracking and degradation due to excessive pressure (Colorado Department of Transportation 2004). Weepholes are usually constructed with PVC pipe installed into the grout lining of the channel. The PVC pipe extends back behind the wall of the lining (Figure 1). Weepholes usually flow between 1 to 5 gallons/minute, although some may flow a little more than a fraction of a gallon per minute. Other weepholes occasionally exceed 15 gallons/minute. In watersheds in semi-arid areas such as Los Angeles, many creeks and streams are completely lined and groundwater baseflow along these permeability features reaches 100%. Frequently where channels have intermittent linings, groundwater seeps out in quantity along the lower end of the lining just where a section of unlined channel begins.

Weepholes are commonly found along open rectangular, open trapezoidal, and underground rectangular storm drains and channels (Figure 1). We have not found weepholes in circular grout lined storm drains. Presumably this is because the circular structure of the channel is strength-resistant to fluid pressure. Weepholes are not needed to reduce pressure in a circular drain because any orifice that is installed in a circular drain will weaken the overall integrity of the circular structure. However, we frequently found groundwater seeps along the joints between sections of circular pipe in circular storm drains (Figure 2). Some of these seeps were quite prominent and created a type of artesian groundwater flow emanating from the joint.

Secondary permeability conduits along cracks and cave-ins, created by settling or subsidence near a channel, are other important features in grout lined channels. The flux of water and solute occurs between channels with conduits and shallow aquifers. Permeability voids develop due to collapse features where subdrains merge with main drains in grout lined channels (Figure 3). Groundwater flow through these partings may exceed 0.2 cfs. These estimates were derived in some cases from chemical mass balance of distinctive input parameters in groundwater, and more often from channel gaging directly upstream and downstream of the channel parting input. Additional seepage features include ordinary cracks and voids caused by aging and breakdown of pipes and lined channels. In some cases, the original weepholes no longer function because secondary joints and cracks at channel grade allow groundwater to discharge into the channel (Figure 3).

Groundwater discharging into lined channels also occurs due to temporary and permanent groundwater dewatering infrastructure. Temporary dewatering equipment is used when construction is being done and when water table lowering is required; for example, during subsurface construction. Temporary dewatering systems may be active for weeks to months and may be replaced by passive French drains or other permanent dewatering apparatus when the construction project is completed (Figure 4).

Shallow groundwater dewatering facilities lower the water table near construction sites, bridge underpasses, and high rises. Permanent dewatering structures may be used where basements or road underpasses are installed near urban structures (Figures 5 and 6). Frequently, groundwater dewatering flows contain excessive nutrients, trace elements, and organic contaminants that are discharged to storm drains (Hibbs and Sharp 2012).



Figure 1. Weepholes installed along the sides and bottom of trapezoidal grout-lined channels.



Figure 2. Circular storm drains with groundwater flowing through the joint between sections of pipe.



Figure 3. Groundwater flowing through partings and degraded linings of underground and aboveground storm drains.

Many older dewatering operations that discharge to storm drains are not archived in as-built plans and are detected only when underground inspections are done. Older, permanent dewatering operations are sometimes grandfathered by regulators. In such cases, solute-laden groundwater continues to flow unabated into storm drains (Hibbs and Sharp 2012).

In Figures 5 and 6 we show permanent, high discharge and low discharge dewatering units. These designs incorporate an underground dewatering sump that passively collects groundwater until the sump is filled. The water in the sump is then pumped out into the channel when a check valve indicates the sump is ready to be evacuated. In these types of dewatering systems we have observed pumping cycles occurring every ten minutes to two hours with a five to twenty-minute pumping period. Other systems have been reported to cycle for only a few minutes once or twice daily. The frequency and duration of pumping is a function of the hydraulic gradient toward the sump and aquifer permeability.

Some channel linings provide armoring and are permeable to fluid flow. In one such case a permeable armoring was placed along part of upper San Diego Creek in Orange County, California (Figures 7 and 8). The permeable lining was expected to provide channel stability while offering semi-riparian channel habitat. Unfortunately, the extreme liquefaction potential along this channel led to the destruction of the pervious armoring during a flood when the liquefaction made the stream sediments highly unstable (Figure 7).

Motivation for Studies of Stream/Aquifer Interactions along Lined Channels

Many groundwater sampling orifices in grout lined channels can be integrated in studies of stream/aquifer interactions. Owing to the frequent lack of shallow well control along urban channels, these orifices can provide a means to collect groundwater issuing into surface channels. Orifices emitting groundwater along grout lined channels provide supplemental data as needed when there is limited shallow well control (Hibbs and Lee 2000; Meixner et al. 2004). Sometimes the orifices along lined channels provide the only groundwater data in a study (Hibbs et al. 2012; Bardsley et al. 2015). In addition, well purging and disposal of excess purge water is not required in contaminated settings because weepholes and other orifices discharge groundwater directly into the channel.

Demonstration Case Studies

We describe three of our case studies to illustrate methodology and to demonstrate effectiveness of research projects in grout lined channels (Hibbs and Lee, 2000; Hibbs et al. 2008; Harrison and Hibbs in press). The first case study documents groundwater evolution along a flowpath leading to an important hypothesis of trace element release in a sensitive terrestrial watershed. The second case study documents use of stable water isotopes to detect unwanted leakage from an underground water main. The third case study compares redox processes and pollutant loading in stream/aquifer systems that are both lined and unlined.

Case Study 1: Hydrochemical Evolution along a Groundwater Flowpath

A study in El Modena Channel in San Diego Creek Watershed of Orange County, California demonstrates important stream/aquifer interactions associated with changing land use activity along this lined channel. El Modena Channel flows into the main stem in the watershed, Peters Canyon Wash (Figures 8 and 9). Near-channel groundwater sample points included four weepholes (EMW-1weep, EMW-2weep, EMW-4weep, EMW-Wweep) and a shallow water well in the dividing area between Peters Canyon Wash and El Modena Channel (CW1-2D) (Figure 9). These weepholes and water wells are oriented along a groundwater flowpath oriented subparallel to the direction of surface water flow in Peters Canyon Wash and El Modena Channel (Figure 9). This channel begins above, and later enters a land region that was once occupied by a large regional marsh known as the "Swamp of the Frogs Marsh" (Hibbs 2001). The channel is lined along the whole reach investigated.

Chloride, sulfate, and selenium data collected from weepholes and from the shallow water well are plotted as bar graphs and line graphs (Figures 9 and 10). The graphs indicate that selenium and sulfate increase significantly between Bryan Avenue and Walnut Avenue. In this region, selenium increases from a low value of 20 μg/L (weephole EMW-2)



Figure 4. Circular storm drain with groundwater constantly discharging from a French drain dewatering apparatus that extends laterally beneath a residential area.



Figure 5. Small discharge dewatering operation that cycles every 40 to 50 minutes when an underground sump fills up with groundwater. The sump is then pumped out for 10 to 15 minutes.



Figure 6. Large discharge dewatering operation that cycles every 50 to 60 minutes when an underground sump fills up with groundwater. The sump is then pumped out for 10 to 15 minutes.



Figure 7. A permeable stream bed lining installed, San Diego Creek. The high liquefaction activity in this stream (see quicksand photo) led to the destruction of the permeable lining after a series of storms. Photo credit, Henry Jones.

to a high value of 178 µg/L (weephole EMW-W) along the groundwater flowpath. Sulfate increases simultaneously from a low value of 458 mg/L (weephole EMW-2) to a high value of 1430 mg/L (weephole EMW-W) as groundwater flows down the hydraulic gradient. Chloride concentrations do not increase proportionately to sulfate and selenium concentrations along the hydraulic gradient. Chloride varies from a low of 189 mg/L (weephole EMW-2) to a high of 251 mg/L (weephole EMW-weep) in the other wells and weepholes (Figures 9 and 10).

Hibbs and Lee (2000) and Hibbs (2001) speculated that groundwater evaporation alone could not explain selenium enrichment in groundwater; otherwise, the conservative chloride ion would become similarly enriched. Chloride concentrations are fairly constant in groundwater in the data set. Therefore, evaporation cannot be the primary agent enriching sulfate and selenium along the groundwater flowpath. This led to the hypothesis of selenium remobilization from reduced forms in the Swamp of the Frogs Marsh. Hibbs and Lee (2000, p. 40-41) discuss the geologic source of selenium and sulfur in shallow groundwater in marsh sediments:

> *"During the historic period of the Swamp of the Frogs Marsh, any forms of selenium and sulfur that flowed into the marsh were probably derived from surface runoff from the highlands in the oxidized forms (SeO_4^{2-} and SO_4^{2-}). Once the oxidized forms of selenium and sulfur reached the anoxic waters of the marsh, these oxidized species were probably reduced to SeO_3^{2-}, elemental Se(0), Se^{2-}, H_2S, and HS^-. These forms would be expected to be removed from the waters in the marsh and sorbed onto solid particles, precipitated as crystalline and amorphous polymorphs, or transformed into volatile gases.*
>
> *Once immobilized in solid forms, the selenium and sulfur solids would tend to remain stable in the soil substrate as new sediments and organic matter in the marsh continued to accumulate. When the Swamp of the Frogs Marsh was drained in the late 1800's and early 1900's, the shallow subsurface was transformed into an oxic environment. In fact, the shallow groundwater flow system today is being recharged by oxygen-enriched water that retains some dissolved oxygen as it flows through the old marsh deposits. The groundwater data collected during this study indicate that groundwater contains relatively high concentrations of dissolved oxygen (usually 0.3 to 5 mg/L), very high ratios of dissolved nitrate to nitrite, and the general absence of iron and manganese. These hydrochemical signatures indicate an oxic groundwater flow system.*
>
> *Oxidizing groundwaters flowing through the old marsh sediments today remobilizes the bound forms of selenium and sulfur into their oxidized and mobile forms (SeO_4^{2-} and SO_4^{2-}). Thus, we can account for high sulfate and selenium in groundwaters in large part from remobilization of reduced forms of selenium and sulfur as oxic waters flow through the historic marsh sediments."*

This primary set of observations along El Modena Channel and additional data collected along unlined and lined channels in the catchment resulted in expanded studies in San Diego Creek Watershed. In the subsequent work we investigated the release of toxic trace elements from the Swamp of the Frogs Marsh, including selenium and arsenic. Today the hypothesis of selenium mobilization from the Swamp of the Frogs Marsh is widely accepted and recognized by the technical hydrological community (Larry Walker and Associates and CH2M Hill 2007; Presser and Luoma 2009; Daniel B. Stephens & Associates, Inc. 2013).

Case Study 2: Use of Stable Water Isotopes to Detect Leakage from an Underground Water Main

Another example of a baseflow-fed drain includes Valencia Drain in San Diego Creek Watershed. Valencia Drain flows into Peters Canyon Wash (Figures 8 and 11). Baseflow was confirmed by analysis of hydrochemical constituents (e.g., selenium, nitrate) emitted by the drain, and by underground inspection of the drain where we found many flowing weepholes and flowing seams inside the drain. The discharge from this storm drain is typically 0.75 to 1.25 cfs during dry weather flows (Meixner et al. 2004).

Samples were collected inside Valencia Drain for measurement of selenium and arsenic species, other inorganic species, and stable isotopes. Our discussion in this paper covers stable water isotopes analysis (O, D) only. A large number of flows from seams and weepholes were sampled for water isotopes during dry weather conditions (Figure 11). Sample stations in Figure 11 are labeled by distance from Peters Canyon Wash to the point in the drain where the sample was collected, in feet. Valencia Drain is oriented almost perfectly in-line with a groundwater flowpath in the shallow aquifer. IRWD well 86N is very close to Valencia Drain and was included in the analysis. This well is the upgradient point in flowpath analysis.

Previous samples collected for isotope analysis from springs, weepholes, and water wells near Peters Canyon Wash plot along an evaporation water line of locally derived water (Figure 12). This identified a local groundwater evaporation line. Most groundwater depths in the Peters Canyon Wash and San Diego Creek area are shallow, sometimes only 6 to 8 feet beneath the land surface. In these areas, groundwater evaporates partially

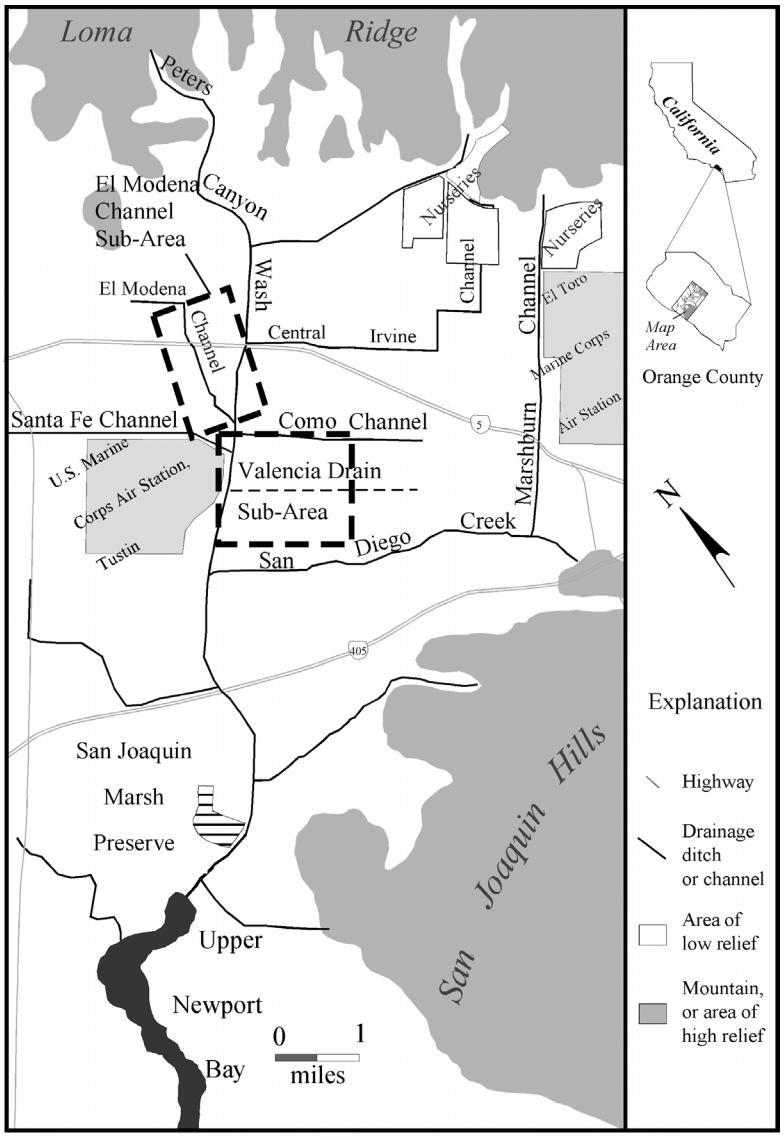

Figure 8. San Diego Creek Watershed showing index subareas for El Modena Channel (Case Study 1) and Valencia Underground Drain (Case Study 2).

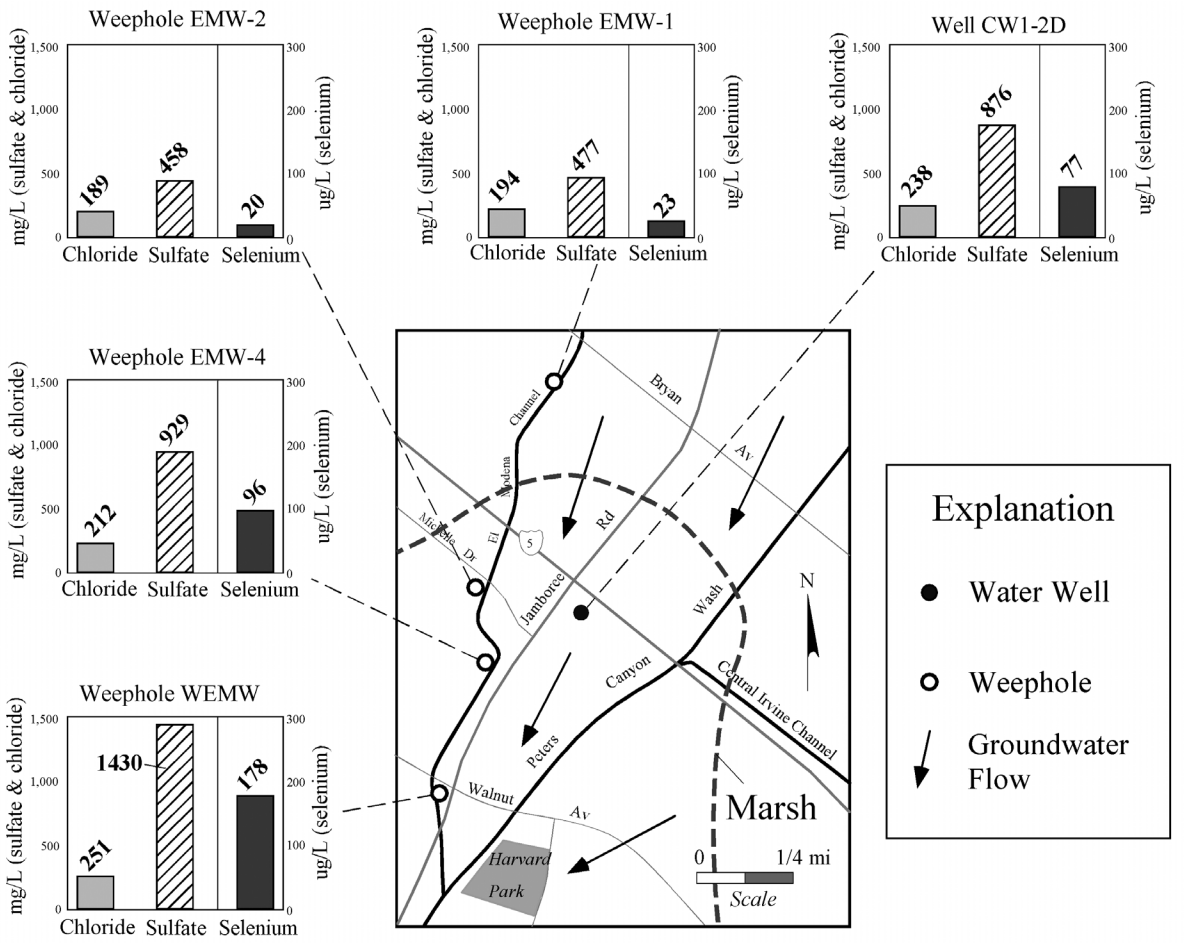

Figure 9. Weepholes and a shallow monitoring well were sampled along a groundwater flowpath along El Modena Channel, showing rapidly increasing sulfate and selenium concentrations down the hydraulic gradient, with little increase in chloride, suggesting a geological source of selenium and sulfate.

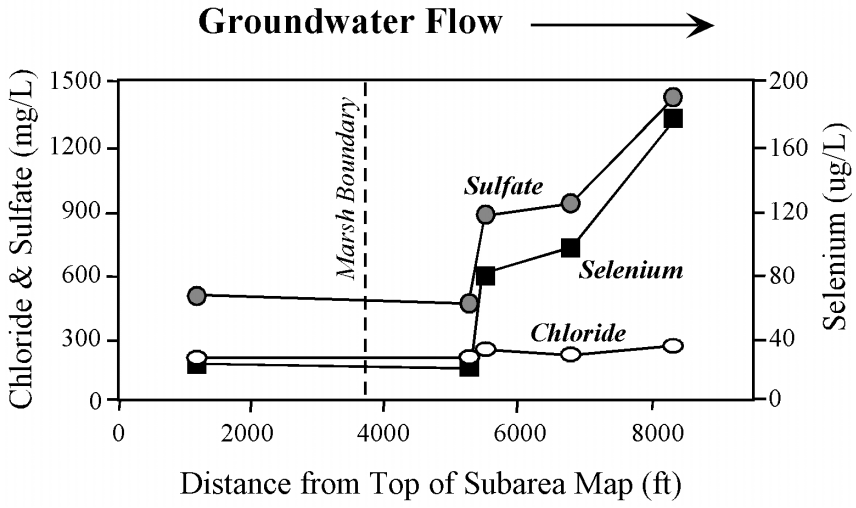

Figure 10. Data from Figure 9 denoting the step increase in sulfate and selenium concentration when groundwater flows across the Swamp of the Frogs Marsh boundary (stipled line). Chloride does not increase significantly along the groundwater flowpath.

by a wicking effect as atmospheric winds create venting of soil vapor, allowing physical evaporation of groundwater. Fine textured soils allow capillary rise up to a few feet above the water table, which can accelerate evaporative effects. Evaporation of soil moisture or irrigation water is also possible prior to recharge, and may explain evaporative effects shown in Peters Canyon Wash and San Diego Creek isotope samples (Figure 12) (Clark and Fritz 1997; Song et al. 2009; Wang et al. 2012).

Most of the data collected from Valencia Drain plot very close to the evaporation line developed for shallow groundwaters in the Peters Canyon Wash area, clustering within the region identified as "Local Water" (Figure 12). Other points plot along a mixing line between the region identified as "Local Water" and "State Project Water." State Project Water is imported from the California State Water Project system in Northern California and is isotopically distinct from local water in the Los Angeles Basin. In recent studies by the U.S. Geological Survey (Reichard et al. 2003), the isotopic signature of imported State Project Water was identified in the Los Angeles Basin. Imported State Project water ranges narrowly about a mean of around -9.5 per mil del ^{18}O and -75 per mil del D (Reichard et al. 2003). Sample point 3534 ft in Valencia Drain is also in this range, identifying this sample as almost entirely sourced from imported State Project Water (Figure 12).

The reason for recharge of State Project Water near a section of Valencia Drain was due to a prominent subsurface water main pipe leak. This fact was established after our findings were reported to Irvine Ranch Water District when the suspected leak was confirmed. The leaky pipe was

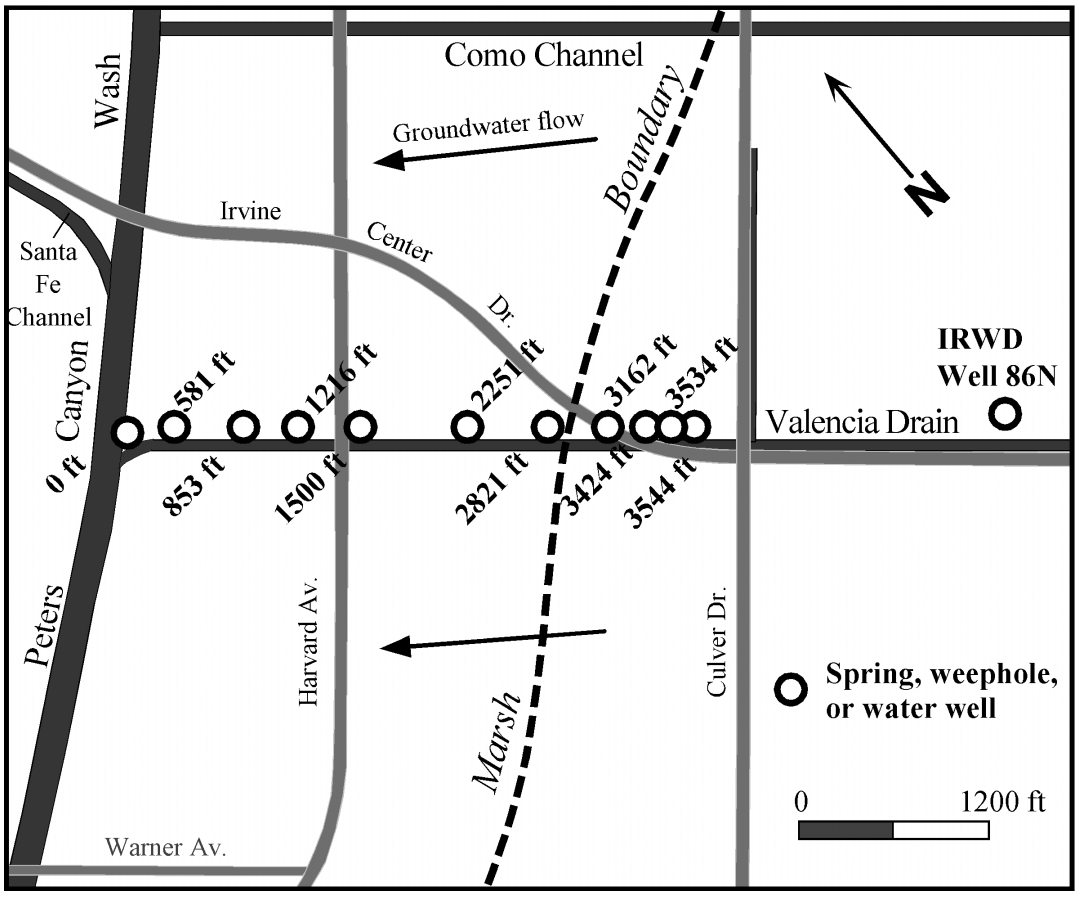

Figure 11. Groundwater sampling points in Valencia Underground Storm Drain include weepholes, channel partings, and a shallow observation well (IRWD Well 86N) installed adjacent to the storm drain.

subsequently repaired (Norris Brandt, personal communication 2006). Other samples have mixed proportions of State Project Water and Local Water (Figure 12). The underground studies identified an area of unwanted leakage from a large underground water main and allowed a water loss to be stopped.

Two years after this leak was identified by isotopic analysis, we returned to sample the weepholes inside the drain. We found that subsequent flows showed little to no imported State Project Water. Apparently after repair of the unwanted leak, most of the imported State Project Water that had recharged the aquifer near Valencia Drain had discharged into the drain and/or mixed with native groundwater. Dilution and outflow accounted for the absence of State Project Water two years after the leak in the water main was first identified and later repaired.

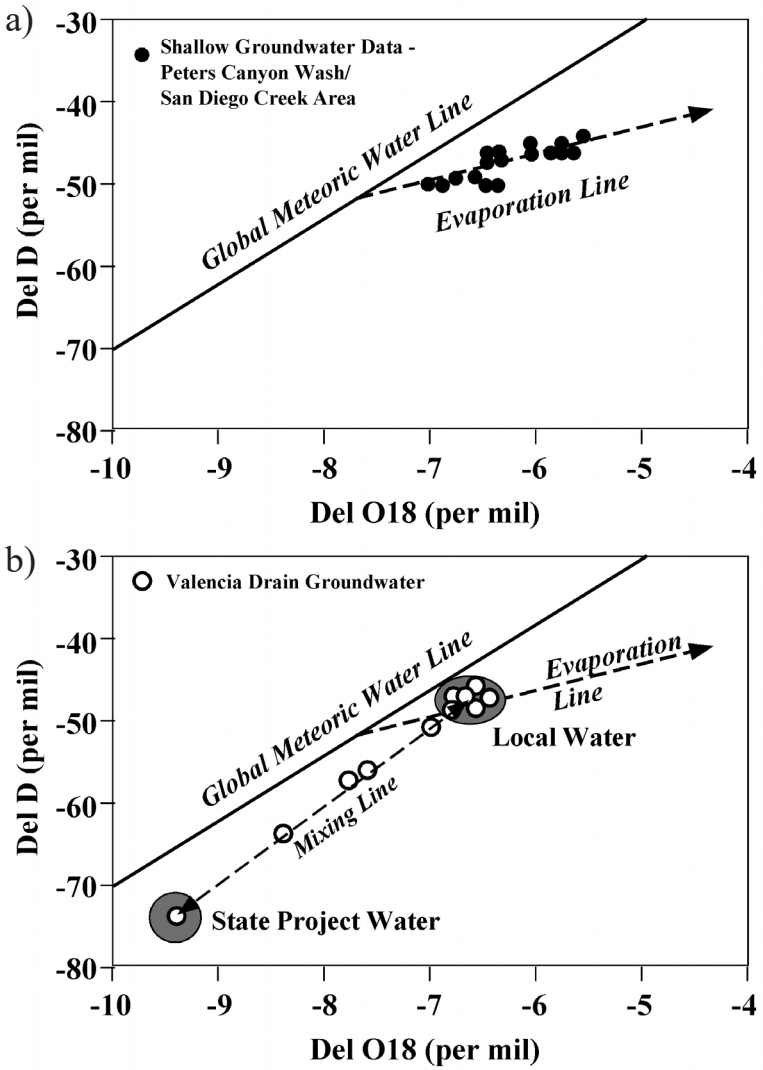

Figure 12. Oxygen and hydrogen isotopes in shallow groundwater in the study area. Figure 12a shows samples collected from springs, weepholes, and shallow water wells in the sub-regional vicinity of Peters Canyon Wash and San Diego Creek, following an evaporation line. Figure 12b shows groundwater data collected from within Valencia Drain, including waters that are derived from local precipitation and waters containing mixtures of imported State Project Water. Most samples cluster along the local evaporation line and are locally derived, but a few samples trend toward the isotopic region defined as State Project Water. Those samples lying along a mixing line with State Project Water obtain their isotopic signature from leakage of a water main pipe into the shallow aquifer adjacent to Valencia Drain.

Case Study 3: Comparative Issues in Lined and Unlined Channels

Frequently channels in urban settings include lined and unlined reaches at various locations. For example, sometimes the rate of groundwater seepage is so great that it is impossible to install a lined channel along a certain reach due to high fluid pressures. This type of a situation is found in part of San Diego Creek in Orange County, California (Sprotte et al. 1980) and along the "narrows" of the Los Angeles River, near Griffith Park (Brown 1975; James M. Montgomery, Inc. 1992). In other cases, unlined channels are retained for aesthetics and for the benefit of wildlife habitat and recreation.

Dynamic redox processes occur during stream/aquifer interactions in unlined channels due to the presence of hyporheic zones. Hyporheic zones are vegetation/carbon rich saturated areas occupying streambeds and streambanks adjacent to unlined open channels. Biogeochemical processes are operative in hyporheic zones due to plant and animal respiration, organic matter decay, and fluid flux caused by fluctuating stream hydraulic head. Such dynamic processes are restricted along concrete-lined channels.

Hyporheic zones are often suboxic to anoxic due to large amounts of decaying organic matter contained in them. While such conditions may be spatially and temporally transitory depending on scour and fill processes, the bulk of the time suboxic to anoxic conditions exist in hyporheic zones remaining favorable for processes such as denitrification, trace element speciation/removal, and organic contaminant degradation. Aquifer flow through hyporheic zone to stream often transitions respectively from oxidizing, to reducing, to oxidizing conditions, providing a robust set of transitioning processes influencing contaminant attenuation and removal.

Differences in redox conditions beneath lined and unlined channels bring up two important topics. First, from a regulatory perspective, grout lined channels do not have the natural bioremediation potential offered by the hyporheic zone as seen in unlined channels. Consequently, lined channels are likely to have greater contaminant loading via groundwater seepage through weepholes and other seepage surfaces along lined sections. This may become an especially significant problem where grout lined channels discharge into sensitive environments such as lakes, marine environments, and marshes. With respect to the main thrust of this paper, the presence of a stream with and without grout linings along alternating reaches offers excellent research opportunities for comparing variable redox processes in proximal lined and unlined sections.

An example is summarized in our current study in El Camino Real Creek in the Malibu Creek Watershed of Southern California. Here we have identified springs and seeps emanating from unlined and lined parts of the creek. Groundwater seeps along lined segments of the channel emit higher nitrate and selenium with low concentrations of manganese and iron (Table 1). This suite of parameters is indicative of oxidizing conditions in the aquifer because dissolved oxyanions of selenium and nitrate are stable in oxic waters, while iron and manganese remain stable as oxide cements that are mostly insoluble

Table 1. Comparison of redox sensitive parameters in unlined and lined groundwater discharge orifices in El Camino Real Creek, Malibu Creek Watershed.

Station	Conductivity (uS/cm)	NO_3-N (mg/L)	Se (ug/L)	Mn (ug/L)	Fe (ug/L)
Weephole 1 lined	4444	4.9	27.0	0.5	10.0
Weephole 2 lined	4436	6.8	36.0	36.0	30.0
Spring 1 unlined	4481	0.3	1.0	19,000	5100
Spring 2 unlined	4680	0.3	1.1	1100	180

in oxidizing waters. The slightly to moderately saline water derives from dissolution of gypsum contained in Miocene marine shale units from which groundwater issues.

Groundwater issuing from springs along the unlined section of the channel flows through a hyporheic zone and contains relatively low concentrations of nitrate and selenium and higher concentrations of manganese and iron (Table 1). This suite of parameters is demonstrably associated with reducing conditions. The reducing environment removes nitrate due to denitrification and dissimilatory nitrate reduction while reducing selenium to low solubility forms (as discussed in case study 1). Furthermore, the iron and manganese oxides in the sediments in the hyporheic zone are reduced into their soluble forms, Fe^{2+} and Mn^{2+}. This suite of data explains why iron and manganese are elevated in springs issuing from unlined sections of the creek (Table 1). Nitrate and selenium concentrations are of greatest concern with respect to pollutant loading to the receiving waters, and these are elevated in springs issuing from weepholes along lined channels (Table 1). Other details on chemistry of streamflow and loading of baseflow in local streams are found in Harrison (2016).

In summary, the hyporheic zone has been found in our research to remove nitrate and selenium through denitrification and trace element reduction to low solubility forms and inert gases. The carbon and vegetation-rich environment beneath unlined channels is demonstrably associated with the removal of nitrate and selenium in groundwater as it flows into open channels. This occurs due to suboxic to reducing conditions created by the organic matter decay and carbon production in the hyporheic zone. Such processes are usually absent along grout lined channels, where conditions often remain oxidizing. In these areas, groundwater seeps through the lining of the channel, unabated by complex biogeochemical removal processes observed where hyporheic zones are present.

Composite Sampling and Sample Reliability Issues

Regarding sample reliability, another concern focuses on the coarse textured material that is frequently constructed as an inert base below concrete drainage infrastructure. This material is installed to prepare a firm, non-corrosive base below the concrete lining of a channel or drain. Coarse textured material is more permeable than most shallow aquifer material and it may act as a sink to groundwater flow, creating lateral flow pathways underneath and alongside a drain or channel (Figure 13). Composite mixing of groundwater (compositing) can occur in such instances, making it difficult to decipher hydrochemical processes developing along groundwater flowpaths. This problem is not uniquely restricted to grout lined channels. Similar compositing issues arise in conventional hydrogeological studies where screens of high capacity water wells penetrate multiple intervals in an aquifer. In such cases wells also provide an integrated composite water quality sample.

In the case of grout lined storm channels, it is necessary to determine if a composite sample might be obtained at a sample point. The chance of this happening is minimized when the water table adjacent to the concrete lined channel is more than a foot above channel grade. In our studies we have been fortunate not to have experienced many cases of severe composite sampling. Weepholes, cracks, and other groundwater sampling orifices provided water quality values that appear to be directly related to the aquifer strata in the immediate vicinity of the orifice sampled (Hibbs and Lee 2000; Meixner et al. 2004; Hibbs et al. 2012).

Losing or gaining stream reaches are easily determined where shallow well control is available near the channel. Hydraulic head differences between stream and interconnected aquifer make this determination possible. Analysts must take care to assure that well control is not collected from a small, localized perched aquifer system however. Where well control is not available, the input from groundwater may be assessed by gaging upstream and downstream of segments of channel and by assuring that any influent waters along the gaged reach are not from urban runoff. Visual inspection for urban runoff conduits and flows are usually done. Where hydrochemical differences exist between urban runoff and shallow groundwater, conservative environmental tracers can make it a routine analysis to determine if the stream

reach in question is a gaining or losing reach. If it is a losing reach, there will be little change in chemistry between upstream and downstream segments of the monitored segment of stream. If it is a gaining reach, the influent groundwater will result in chemistry changes where tracer concentration in groundwater and urban runoff are significantly different, and where the amount of additive baseflow along the characterized reach is more than 5 to 10% of total streamflow.

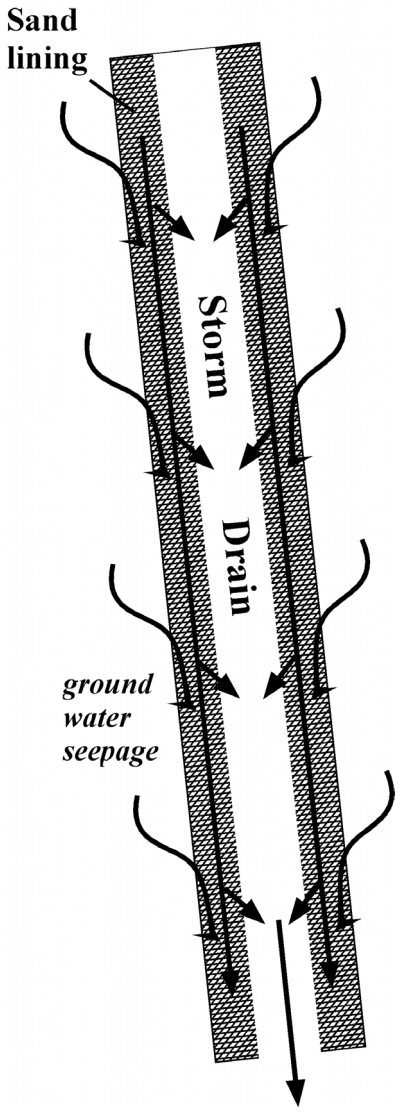

Figure 13. Groundwater compositing along a grout lined channel or storm drain can result in undesirable mixing. Groundwater compositing is limited when the hydraulic gradient of the aquifer is directed sub-perpendicular to the drain.

A question has come up during our discussions of this methodology. Does the grout lining affect the hydrochemistry in the groundwater emitted at the orifice? In our sampling comparisons between shallow wells and adjacent weepholes we have noted no significant irregularities or differences in groundwater quality with respect to general minerals, trace elements, and environmental isotopes. We have observed mineralized surfaces that build up on the walls of the concrete, thus isolating concrete surfaces from direct contact with groundwater (Figure 14). Given the usually high rate of flow that often occurs through the concrete orifices, it is improbable that hydrochemical interactions occur to any noticeable degree between concrete lining and groundwater flowing through the orifice. Accordingly, we feel that weepholes, cracks, and other orifices in lined channels generally provide reliable groundwater chemistry data.

Safety Issues

This work presents several potential hazards. The main hazard is operating in confined spaces due to the possible presence of toxic gases caused by runoff of accidental or purposeful spills of chemicals from industrial and residential areas. Extreme hazard can develop in such cases due to the possibility of respiratory poisoning. While operating in confined spaces it is essential to have gas sensors, air tight goggles, and emergency oxygen breathing equipment available. Only the most experienced personnel should engage in studies in confined spaces.

Another concern includes wildlife that occupies lined channels and tributary subdrains. We have encountered poisonous snakes (Figure 15), poisonous spiders, rats, bats, coyotes, and raccoons in or alongside channels during our studies. Dangers from these wildlife are not specific to urban infrastructure, but in natural habitat there are easier opportunities for wildlife and humans to retreat from a possible confrontation. In urban storm drains avenues of escape are often restricted. Wildlife may react aggressively to a "trapped" situation. Animals nursing young can be especially dangerous. Rabid animals, especially bats, present a constant threat. Fortunately, our encounters with wildlife have proven relatively harmless so far.

Flooding is usually not a serious threat during dry weather studies except in unusual circumstances. Fire hydrants and water pipes may rupture, releasing a torrent of flows into urban channels. Purposeful discharges into storm channels may also occur during construction dewatering. Vertically walled channels are particularly hazardous in such circumstances due to the difficulty of escape. Ladders are often installed at locations in urban channels but are often sporadically placed.



Figure 14. Mineralized surface built up along the seam between two sections of underground circular pipe in a stormdrain. Groundwater flowing along the seam for many years created the mineralized surface due to outgassing of carbon dioxide and precipitation of a calcite cement.



Figure 15. Studies in grout lined channels present risks to investigators. Risks include wildlife, flooding, and toxic gases in closed spaces.

An especially hazardous flooding situation was encountered by the main author of this paper. While performing surveys with a graduate student along the underpass section of Central Irvine Channel at the Interstate 5 Freeway, an estimated 16 to 20 inch wall of water cascaded at a high rate of flow down Central Irvine Channel toward the author and student (Figure 8). We were forced to run downstream for our safety to the outlet of the underpass. When we emerged from the underpass part of the flow had already started to wet our shoes. After escape from this dangerous situation, we stood in a safe zone and witnessed an enormous torrent of water entering the larger section of Peters Canyon Wash. Some hours later we learned that a very rare and very dangerous scenario had occurred upstream, when a 69-inch principal water main supplying large areas in Orange County ruptured, spilling 5 million gallons of water across the upper part of San Diego Creek Watershed (Harris and Roug, Los Angeles Times 1999). The disintegrating pipe under high pressure threw large chunks of concrete into the air from the force of the rupture. Amazingly, the pipe was buried 25 feet underground. This type of incident occurs rarely; it won't be forgotten.

Conclusions

Many of our research studies in urban watersheds and along lined channels have been fruitful, with results identifying leakage of water main ruptures flowing into storm drains via groundwater seepage; infiltration of contaminated stormwater runoff into groundwater; evolution of hydrochemical patterns along groundwater flowpaths; and detection of groundwater mounding areas appearing as rising groundwater at storm drains. This type of research is possible when shallow well control is either not available or where financial or regulatory hurdles undermine the conventional means of evaluating stream/aquifer interactions in urban corridors via installation of shallow groundwater monitoring wells. Safety considerations are paramount in this kind of work. Suggestions are provided to help avoid notable hazards.

The findings from this work reveal many opportunities for collecting groundwater data along grout lined channels. This work is paramount

where loading of solutes is due to groundwater baseflow and where shallow well control is absent. Endmember groundwater concentrations can be collected directly at the point of groundwater issue into streams. Our work has also commented on the hydrochemical differences of groundwater baseflow issuing from grout-linings and organic rich streambed sediments. These differences relate to practical applications of undesirable pollutants where channels are unlined, while emphasizing the value of rehabilitating and restoring urban channels where technical and regulatory issues make stream restoration possible.

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