Sources of groundwater pumpage in a layered aquifer system in the Upper Gulf Coastal Plain, USA

Yun Huang · Bridget R. Scanlon · Jean-Philippe Nicot · Robert C. Reedy · Alan R. Dutton · Van A. Kelley · Neil E. Deeds

Abstract Understanding groundwater-pumpage sources is essential for assessing impacts on water resources and sustainability. The objective of this study was to quantify pumping impacts and sources in dipping, unconfined/confined aquifers in the Gulf Coast (USA) using the Texas Carrizo-Wilcox aquifer. Potentiometric-surface and streamflow data and groundwater modeling were used to evaluate sources and impacts of pumpage. Estimated groundwater storage is much greater in the confined aquifer (2,200 km^3) than in the unconfined aquifer (170 km^3); however, feasibility of abstraction depends on pumpage impacts on the flow system. Simulated predevelopment recharge (0.96 km^3/yr) discharged through evapotranspiration (ET, ~37%), baseflow to streams (~57%), and to the confined aquifer (~6%). Transient simulations (1980–1999) show that pumpage changed three out of ten streams from gaining to losing in the semiarid south and reversed regional vertical flow gradients in ~40% of the entire aquifer area. Simulations of predictive pumpage to 2050 indicate continued storage depletion (41% from storage, 32% from local discharge, and 25% from regional discharge capture). It takes ~100 yrs to recover 40% of storage after pumpage ceases in the south. This study underscores the importance of considering capture mechanism and long-term system response in developing water-management strategies.

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Introduction

Increasing reliance on groundwater resources to meet rising demands from growing populations and increasing food production through irrigation underscores the need to better understand water budgets of aquifers (Molden 2007). Groundwater is also being promoted as a buffer to improve reliability of water supply within the context of increasing climate extremes with longer, more intense droughts interspersed with infrequent floods (Kundzewicz et al. 2008). With rising groundwater demands, it is important to quantify how much groundwater is available for production and the impacts of such production on the system.

Unlike oil production where the primary objective is to produce all available oil from a reservoir, groundwater production is often limited by adverse environmental impacts caused by production such as streamflow reduction, water-quality degradation, and land subsidence. For example, much of the groundwater in the High Plains aquifer (central USA) is stored in the northern part, in Nebraska (~65%); however, most of this groundwater cannot be abstracted because depletion of only 1% of groundwater storage caused up to 50% reductions in baseflow to the Platte River (Luckey et al. 2007). Groundwater abstractions resulted in up to 10 m of subsidence in the California Central Valley (Williamson et al. 1989) and up to 3 m of subsidence in the Houston (Texas) area (Kasmarek and Robinson 2004). In addition, it is important to distinguish renewable from non-renewable or fossil groundwater resources. Groundwater pumpage of renewable resources is limited by fluxes or recharge rates, whereas pumpage of non-renewable resources is limited by groundwater storage (Oki and Kanae 2006, Alley 2007, Gleick and Palaniappan 2010).

What impact does pumpage have on the water budget of the system? Johnston (1997) evaluated impacts of pumpage in 11 major aquifer systems in the US based on groundwater modeling by the US Geological Survey (USGS) Regional Aquifer System Analysis (RASA)

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program. Prior to groundwater pumpage or development, there is generally a long-term dynamic equilibrium between initial or predevelopment aquifer recharge (*Ro*) and initial discharge (*Do*), with essentially no change in groundwater storage ($Ro = Do$ and $\Delta S=0$). Predevelopment recharge and discharge rates ranged from very high in dynamic karstic and basaltic aquifers (e.g. Floridian aquifer, 19,000 km^3/yr; Colombia Plateau aquifer, 6,000 km^3/yr) to very low in the Great Plains system and southern High Plains (~300 km^3/yr). Theis (1940) pointed out that during aquifer development "all water discharged by wells is balanced by a loss of water somewhere". Pumpage is initially accounted for completely by reductions in aquifer storage; however, with time pumpage (*Pu*) can be accounted for by increased recharge (ΔR), and/or decreased discharge (ΔD), termed 'capture', as follows (Theis 1940, Bredehoeft 2002, Devlin and Sophocleous 2005):

$$(Ro + \Delta R) - (Do + \Delta D) + \Delta S = Pu, \tag{1}$$

The RASA program showed that impacts of pumpage on water budgets were variable, with pumpage being accounted for primarily by reductions in aquifer storage in the High Plains, California Central Valley, and Coastal Plain in the Mississippian Embayment. In contrast, pumpage was accounted for by increased recharge related to surface-water irrigation return flow (Central Valley and High Plains), land-use change such as cultivation (High Plains), or transfer of water from local outcrop zones to regional confined zones (Coastal Plain aquifers). Decreases in natural discharge or capture of water occurred primarily in the Coastal Plain systems. Because of very coarse discretization in RASA groundwater models, with grids ranging from 40 to 660 km^2, Johnston's analysis was limited to regional flow in large aquifer systems.

What are the time scales of pumpage impacts on the system? Dissipation of hydrologic stresses caused by pumpage is controlled primarily by aquifer diffusivity, defined as the ratio of aquifer transmissivity to storage coefficient. Karst systems are very dynamic and can quickly reach a new dynamic equilibrium; however, porous media systems may not stabilize for long times after new pumping stresses. While pumping may exert little impact on water resource at short time scales (e.g. impact on streamflow or riparian ET over years), decadal to century-scale impacts may be large. Planning horizons for water resources generally range from 10 to 50 yrs. Long-term system response needs to be considered in current water-management plans.

How do unconfined and confined aquifers differ in terms of typical water storage volumes? In unconfined or water-table aquifers, such as the High Plains aquifer, water is derived from drainable porosity resulting in water-table declines (change in head, ΔH). Converting water-table declines to water volumes (*V*) depends on the aquifer storage coefficient, termed specific yield ($V = Sy \times \Delta H \times A$), where A is aquifer area and values of specific yield (Sy) range from 0.01–0.30 (Freeze and Cherry 1979). In contrast, confined aquifers provide water through the compressibility of water and the matrix. Matrix compressibility is subdivided into elastic and inelastic compressibilities, with inelastic compressibility related to compaction of fine-grained sediments resulting in land subsidence. Typical values of confined aquifer storage coefficients are orders of magnitude less than those for unconfined aquifers; therefore, similar head declines yield much less water from confined aquifers relative to unconfined aquifers. The water source in confined aquifers is initially derived from storage but later may be derived from overlying and underlying confining units (Konikow and Neuzil 2007), overlying aquifers, and/or connected unconfined outcrop zones. Water storage in fine-grained confining layers may be volumetrically much greater than that in confined aquifers (Konikow and Neuzil 2007).

Are impacts of pumpage reversible? In some cases, groundwater may be required for a certain time period, e.g., 20–40 yrs for shale gas development in Texas (Nicot and Scanlon 2012), and it is important to understand whether reductions in pumpage can reverse the impacts. Examples are provided by the California Central Valley where pumpage was reduced by diverting surface water from more humid regions and the potentiometric surface recovered by up to 90 m in some areas (Faunt 2009). The potentiometric surface in the major cone of depression in the Houston (Texas) area has recovered by up to 30 m as a result of reduced pumpage and increased surface water use to reduce subsidence (Kasmarek and Robinson 2004).

How can we manage the system to reduce impacts of pumpage? Over time, the contribution of aquifer storage to pumpage should approach zero and the system should reach a new equilibrium. The concept of sustainable pumpage equates to negligible changes in groundwater storage; however, sustainable pumpage does not equate to the much broader concept of sustainability, which includes minimizing adverse environmental effects and social and economic effects (Alley et al. 1999, Alley and Leake 2004). To reduce pumpage impacts on streamflow and riparian zones, pumpage should be located as far away from discharge areas as possible. Many techniques are available to mitigate negative effects of groundwater pumpage. For example, aquifer storage and recovery using spreading basins at the surface or injection wells has been widely used (Pyne 2005). Aquifer storage and recovery in the Central Valley in California provides an approach to store excess surface water in aquifers that is available for use during droughts; thus, evening variations in water supply and demand associated with floods and droughts (California Department of Water Resources 2005).

The objective of this study was to quantitatively assess sources of groundwater pumpage, impacts of pumpage, time scales of impacts, and reversibility of pumpage impacts in dipping, unconfined/confined aquifers using modeling and monitoring data from the Carrizo-Wilcox aquifer in Texas as an example. The Carrizo-Wilcox aquifer provides an excellent case study because it has been subjected to varying pumpage stresses related to irrigation since the early 1900s and municipal and industrial pumpage since the 1930s. Detailed groundwater models have been developed for the entire aquifer to simulate

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predevelopment conditions and pumpage effects during aquifer development. The aquifer extends across a range of climatic settings from subhumid in the north to semiarid in the south. There is considerable interest in understanding water resources in this system for expanding shale gas production in the southern Carrizo-Wilcox aquifer and for alternative municipal supplies for the city of San Antonio (seventh largest city in the US, population 1.3 million in 2010). Results of this study should advance our understanding of water resources of dipping, mostly confined aquifer systems for future management of these systems.

Materials and methods

Study area and background

The Carrizo-Wilcox aquifer is part of Gulf Coastal Plain aquifer system. The Texas portion of the aquifer extends from the Red River to the Rio Grande. For the purposes of modeling and understanding, the aquifer has been subdivided into northern (Red River to Trinity River), central (Trinity River to Colorado River), and southern (Colorado River to Rio Grande) regions (Fig. 1). Mean annual precipitation (1971–2000) ranges from 1,300 mm in the north to 500 mm in the south (PRISM Climate Group 2004). Land cover varies from pine and hardwood forests in the north to chaparral brush and grasses in the south. The Winter Garden area in the south (30,000 km^2, Fig. 1) is known for year-long vegetable production through irrigation (Turner et al. 1960). Soil texture in the outcrop area generally reflects the distribution of underlying aquifers and aquitards, with more sandy soils over aquifers and more clayey soils over aquitards. The Carrizo-Wilcox aquifer intersects 13 river basins in Texas. Major rivers flow toward the Gulf of Mexico and cross the outcrop of the aquifer formations. Streamflow generally decreases from north to south along the precipitation gradient.

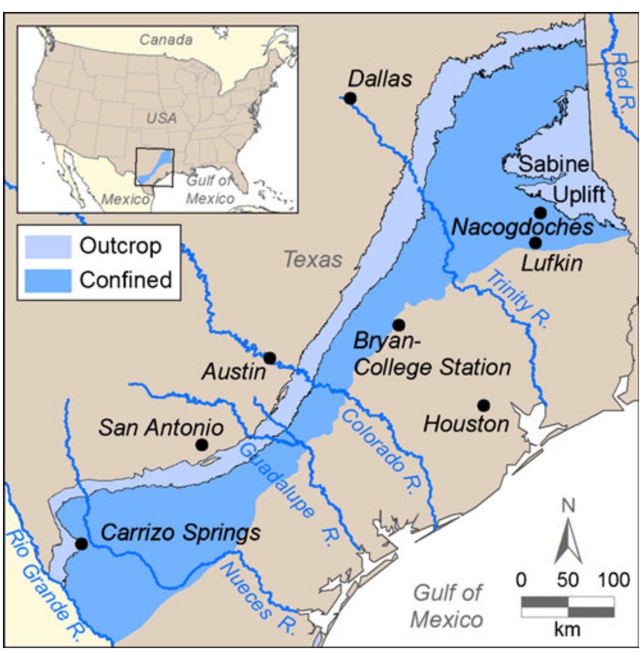

Fig. 1 Location of the Carrizo-Wilcox aquifer, including the outcrop and confined areas. The Winter Garden refers to the aquifer area southwest of the Guadalupe River

The Carrizo-Wilcox aquifer ranks third highest in Texas for water use, after the Gulf Coast and High Plains aquifers (Texas Water Development Board 2007). Rapid shale gas development in the southern Carrizo-Wilcox aquifer has raised concerns about the adequacy of water supply for gas production activities in addition to current heavy pumpage for irrigation in the Winter Garden area in the south. In the Eagle Ford Shale play, fracking is used to create pathways in the low-permeability shale for gas production. Approximately 1,100 oil and gas wells have been drilled through to the underlying Eagle Ford formations in the footprint of the southern Carrizo-Wilcox aquifer as of mid-2011. Current water pumpage for oil and gas production is estimated to be 15×10^6 m^3/yr (Nicot and Scanlon 2012). Pumpage is projected to continue for the next 40 yrs with an expected average amount of 40×10^6 m^3/yr. In addition, there is increasing interest in using the Carrizo-Wilcox aquifer to supplement water supply from the Edwards aquifer for the City of San Antonio. Aquifer storage and recovery (ASR) is currently being conducted, which involves transfer of water from the Edwards aquifer to the Carrizo-Wilcox aquifer, and storage and recovery of water from the Carrizo-Wilcox aquifer. The ASR system was designed such that stored water could be used during droughts or for flow management in the protection of endangered species.

Hydrogeology of the Carrizo-Wilcox aquifer

The Carrizo-Wilcox aquifer is typical of Gulf Coastal Plain dipping aquifers that have generally narrow, unconfined outcrop sections and much wider confined sections. The outcrop (area 27,000 km^2) ranges from 5 to 50 km wide in different regions, whereas the confined aquifer (66,000 km^2) ranges from 40 to 180 km wide (Fig. 1). The aquifer system is up to 1,000 m thick. The downdip extent of the Carrizo-Wilcox aquifer is generally defined on the basis of water quality (3,000 mg/L total dissolved solids). Further downdip is a saline confined section that is generally geopressured and hosts oil and gas reservoirs (Dutton et al. 2006). Although the aquifer is referred to as the Carrizo-Wilcox aquifer, the Wilcox aquifer underlies the Carrizo aquifer. The Carrizo-Wilcox aquifer consists of fluvio-deltaic sediments of the Wilcox Group overlain by Carrizo Sand of the Claiborne Group (Bebout et al. 1982; Hamlin 1988; Fig. 2). In the central region of the Carrizo-Wilcox aquifer, the Wilcox Group is subdivided (from bottom to top) into the Hooper (aquitard), Simsboro (aquifer), and Calvert Bluff (aquitards) formations, and is overlain by the Carrizo Fm. (aquifer; Kaiser 1974; Xue and Galloway 1995). The Simsboro Fm. cannot be distinguished in the southern and northern regions of the Carrizo-Wilcox aquifer and the Wilcox Fm. is subdivided into the Lower, Middle, and Upper Wilcox. The aquitards

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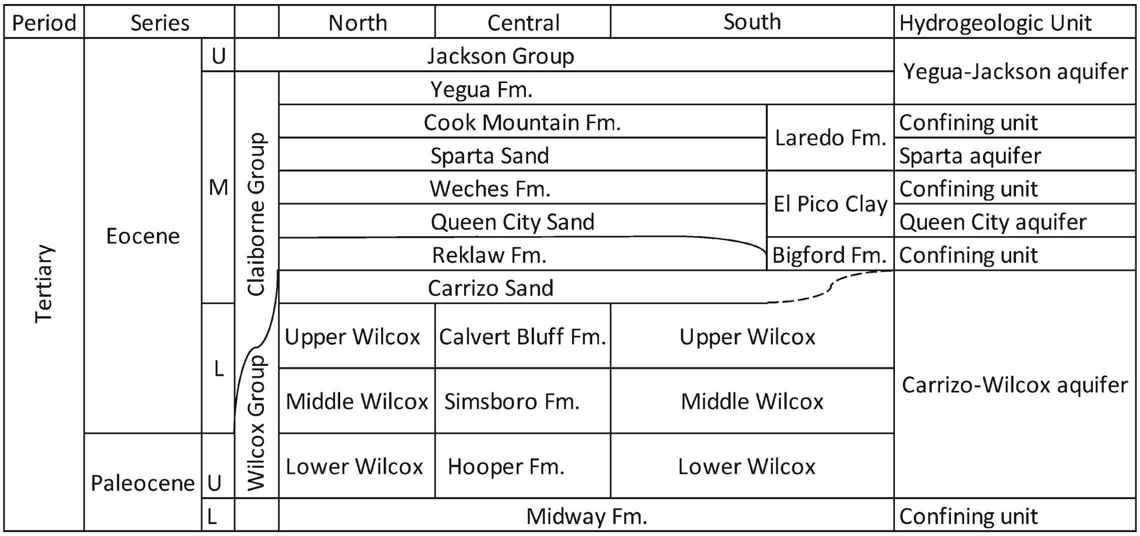

Fig. 2 Generalized lithostratigraphic and hydrostratigraphic section for the Claiborne and Wilcox groups in Texas (Modified from Kelley et al. (2004))

consist of interbedded shales and clays, with lignite deposits, whereas the aquifers consist of multistory, multilateral sand deposits (Ayers and Lewis 1985). The Carrizo-Wilcox aquifer is separated from the overlying Queen City aquifer by the Reklaw or Bigford Fm., which is a confining unit (Fig. 2). Underlying the Carrizo-Wilcox aquifer is the Midway Group, which is composed of mostly marine clays. Further downdip, the saline section of the Carrizo-Wilcox formations thicken into a major growth-fault system commonly known as the Wilcox Growth Fault zone (Bebout et al. 1982; Ewing 1990). The growth-fault zone formed as a thick package of Wilcox sediment prograded onto uncompacted marine clay and mud deposited in the subsiding basin beyond the Cretaceous shelf (Bruce 1973). In general, the geologic units of the Carrizo-Wilcox are semi-consolidated and less likely to undergo land subsidence in response to storage depletion. There have been no reported land subsidence issues in the footprint of the Carrizo-Wilcox aquifer.

The conceptual understanding of the groundwater flow system includes recharge to the aquifer in the unconfined outcrop zone and discharge to local and intermediate flow systems in the outcrop through groundwater ET and through baseflow to streams (Fig. 3). Some groundwater moves into the deep confined system and discharges regionally to overlying units as cross-formational flow. The downdip extent of the aquifer is marked by downdip flow from the hydropressured zone and updip flow from the geopressured zone (Dutton et al. 2006).

Aquifer development

Groundwater development in the Carrizo-Wilcox aquifer began in the early 1900s, primarily for irrigation. Irrigation pumpage was dominant in the south with the first flowing artesian well drilled in 1884 to 50 m depth in Dimmit County (Klemt et al. 1976). Peak irrigation pumpage occurred in the 1960s at $\sim 500\times10^6$ m^3/yr (Fig. 4). Since the 1930s municipalities and industry in Nacogdoches and Smith counties in the northern region began to pump groundwater from the Carrizo aquifer (White et al. 1941). The Bryan-College Station well field in the central region was developed in the 1950s (Dutton et al. 2003). Production increased from 9×10^6 m^3/yr in 1950 to 22×10^6 m^3/yr in 1980 and then to 34×10^6 m^3/yr in 2000. In the 1980s, lignite mines began pumping greater amounts of groundwater. Water withdrawal related to all types of mining activities made up an estimated 25% of total production in the central region in 2000 (Dutton et al. 2003). Water production for mining can fluctuate seasonally due to mining activities or longer-term due to mining sites startup or closure. In general, irrigation pumpage exceeded municipal pumpage by up to a factor of two for most years prior to 2000. In contrast to irrigation pumpage that varied over time in response to variability in precipitation and commodity prices, municipal pumpage steadily increased and has exceeded irrigation pumpage since 2005 (Fig. 5).

Total pumpage from the Carrizo-Wilcox aquifer in 1999 was 660×10^6 m^3 (= 0.66 km^3), 25% from the unconfined outcrop zone and 75% from the confined downdip zone. On a regional basis, ~30% of total pumpage is in the north, 20% in the center, and 50% in the south. Breakdown by aquifer shows 57% of total pumpage in the Carrizo aquifer and 43% in the Wilcox aquifer. Summary of pumpage by aquifer varies by region: 50% from the Carrizo aquifer in the north, 10% in the center, and 80% in the south. Most pumpage in the central region occurs in the Simsboro sands in the Wilcox aquifer.

Groundwater flow models of the Carrizo-Wilcox aquifer

There have been many groundwater modeling studies of the Carrizo-Wilcox aquifer, including the regional model

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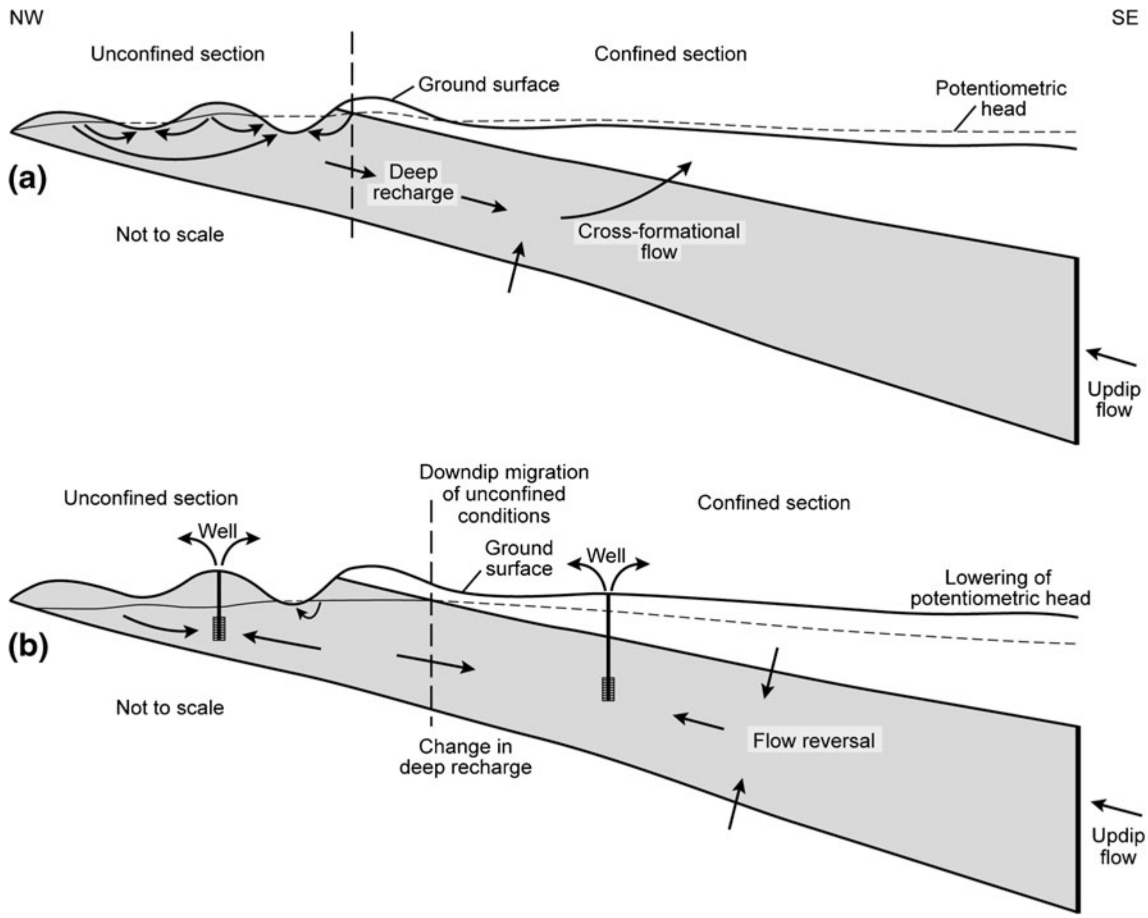

Fig. 3 Conceptual diagrams of groundwater flow components under **a** pre-development and **b** post-development conditions in the Carrizo-Wilcox Aquifer (Modified from Reedy et al. (2009))

from the USGS RASA program (Ryder 1988; Ryder and Ardis 1991). More recent groundwater models were developed for the Carrizo-Wilcox aquifer as part of a statewide Groundwater Availability Modeling program (Mace and Ridgeway 2005, Texas Water Development Board 2011). Individual models were developed for the northern, central and southern portions of the Carrizo-Wilcox aquifer with significant spatially overlapping areas (Deeds et al. 2003, Dutton et al. 2003; Fryar et al. 2003). More recently, these models of the Carrizo-Wilcox aquifer were combined with those of the overlying Queen City

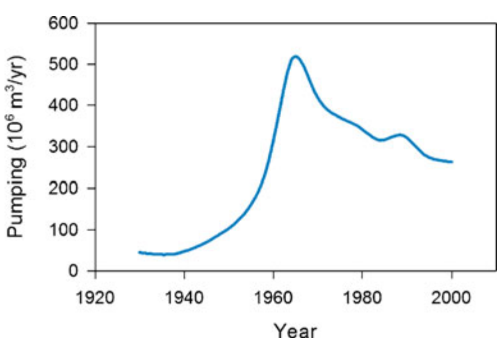

Fig. 4 Irrigation pumpage in the Winter Garden area. The estimates for the years before 1958 were derived from Klemt et al. (1976), while data for later years were obtained from the TWDB irrigation survey (TWDB 2001)

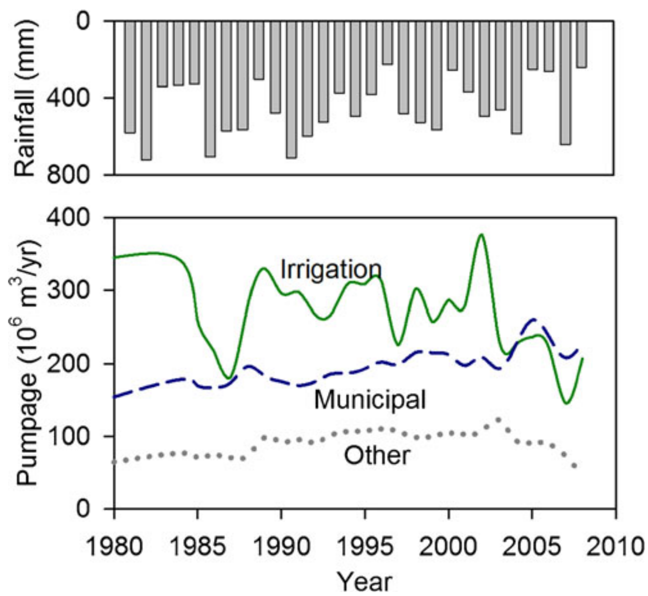

Fig. 5 Total groundwater pumpage from the Carrizo-Wilcox aquifer by water-use category. Pumpage data are from TWDB water-use survey. 'Other' category includes manufacturing, power generation, mining, and livestock use. Rainfall data are from the National Oceanic and Atmospheric Administration station in Dimmit County (station ID=411528)

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and Sparta aquifers (Kelley et al. 2004). The updated groundwater models include steady-state simulations representing predevelopment (before 1900) and transient calibration/verification simulations with annual stress periods from 1980 through 1999 (Kelley et al. 2004). The models are vertically partitioned into eight layers, representing formations or their equivalents: four model layers representing the Carrizo-Wilcox aquifer, overlain by the Reklaw (confining), Queen City (aquifer), Weches (confining), and Sparta (aquifer). The grid size is 1.6×1.6 km^2, equivalent to 1×1 mile^2. The downdip boundary (southeast boundary) is specified as a no-flow boundary and corresponds to the updip limit of the geopressured zone and/or the Wilcox growth-fault zone (Figs. 1 and 3). The bottom boundary is also specified as no flow, assuming no flow between the Carrizo-Wilcox aquifer and the underlying Midway Fm. The top boundary outside of the outcrop areas is a general head boundary on the overlying Sparta aquifer to simulate interconnection between the Sparta aquifer and younger units. The models were calibrated to measured water levels and potentiometric surface data and baseflow to streams. In most cases, calibration achieved a root mean square error (RMSE) ≤10% of the range in measured heads in the simulated aquifer. Leakage from stream segments was compared with stream gain-loss studies where available. Simulated groundwater travel times were compared with results from a previous groundwater age-dating study (Pearson and White 1967) to further assess model results.

Recharge rates were estimated as a function of precipitation, soil and geologic properties, and topography. Recharge was estimated using the chloride mass balance approach applied to groundwater chloride data from the outcrop zone and was related to precipitation (Scanlon et al. 2003). Upland locations or formations with relatively higher hydraulic conductivities were assigned a higher recharge rate. In the transient simulation, recharge was varied annually with precipitation. Groundwater ET was modeled as a step function of water-table depth, with maximum ET at or above a specified elevation (ET surface), linearly decreasing ET with water-table depth decreasing to zero at the extinction depth. Groundwater ET rate and extinction depth were estimated using the Soil Water Assessment Tool (SWAT) hydrological model (Arnold et al. 1993; Arnold et al. 1998) by simulating soil-water balance in river basins. Interaction between groundwater and surface water was modeled as a head-dependent boundary condition using the Streamflow Routing Package (Prudic 1989). Flow for a river reach is calculated as a linear function of the head difference between the stream and aquifer and a conductance term, which is a function of hydraulic conductivity and is generally a calibration parameter.

Groundwater pumpage was estimated from a water-use-survey database developed by the Texas Water Development Board (TWDB). The water-use survey was conducted annually from 1980 and includes municipal, manufacturing, power generation, mining, livestock, and irrigation water-use categories. Water for domestic use was estimated based on population size. Pumping for municipal, manufacturing, mining, and power water uses was distributed to model cells based on actual location of wells whenever possible. Distribution of irrigation and livestock pumping was based on land-use type and distribution of domestic pumping was based on population density.

The primary data source for hydraulic properties of the Carrizo-Wilcox aquifer is Mace and Smyth (2003). Depositional environment and lithology of the hydrostratigraphic units were considered in the process of assigning hydraulic properties to the model domain. In both the northern and southern models, hydraulic conductivities were assumed to decrease with depth (Deeds et al. 2003; Fryar et al. 2003). In the central model, hydraulic conductivity was assumed to be greatest in the thickest part of the fluvial channel axes (Dutton et al. 2003). Aquifer hydraulic properties (hydraulic conductivity and storativity) are summarized on the basis of calibrated parameter values in Kelley et al. (2004) models (Table 1). In general, the Carrizo aquifer has higher mean hydraulic

Table 1 Hydraulic parameters used in the models. Statistics (minimum, maximum, and geometric mean) based on summary of cell values. Hydraulic conductivity *K* in m/day. Storativity *S* dimensionless. The Calvert Bluff, Simsboro and Hooper are equivalent of the Upper, Middle and Lower Wilcox units

Region	Unit	Minimum		Maximum		Mean	
		K	*S*	*K*	*S*	*K*	*S*
North	Carrizo	0.07	1.0×10^{-5}	18.1	1.6×10^{-3}	1.93	3.5×10^{-4}
North	Upper Wilcox	0.30	8.9×10^{-5}	2.1	9.4×10^{-3}	0.56	2.2×10^{-3}
North	Middle Wilcox	0.30	9.0×10^{-5}	3.1	1.3×10^{-2}	0.58	2.9×10^{-3}
North	Lower Wilcox	0.46	9.0×10^{-5}	9.2	7.2×10^{-3}	0.64	6.4×10^{-4}
Center	Carrizo	0.05	1.0×10^{-5}	47.9	3.9×10^{-3}	3.03	7.7×10^{-4}
Center	Calvert Bluff	0.01	3.2×10^{-5}	4.7	1.0×10^{-1}	0.24	1.5×10^{-4}
Center	Simsboro	0.01	3.2×10^{-5}	12.8	1.5×10^{-1}	1.23	7.8×10^{-5}
Center	Hooper	0.00	3.2×10^{-5}	6.5	1.0×10^{-1}	0.38	9.5×10^{-5}
South	Carrizo	0.03	1.0×10^{-5}	45.6	4.4×10^{-3}	1.95	7.7×10^{-4}
South	Upper Wilcox	0.09	6.0×10^{-5}	0.9	5.8×10^{-3}	0.44	4.3×10^{-4}
South	Middle Wilcox	0.09	6.0×10^{-5}	21.2	5.4×10^{-3}	0.20	1.4×10^{-3}
South	Lower Wilcox	0.30	6.0×10^{-5}	20.2	6.0×10^{-3}	0.78	1.9×10^{-3}

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conductivity than the Wilcox aquifer. The Simsboro Fm. (Middle Wilcox) in the central region has higher mean hydraulic conductivity than other Wilcox units. The range in hydraulic parameters in each region is large, suggesting high spatial variability. Specific yield of each unit ranges from 0.10 to 0.15.

In this study, the integrated Carrizo-Wilcox/Queen City Sparta aquifer models of Kelley et al. (2004) were used. Overlapping areas among models were excluded in water-budget summary and non-overlapping regions were used as defined in Deeds et al. (2009).

Methodology

Aquifer storage volumes in a unit area were calculated from drainable volumes in the unconfined portion (product of specific yield and aquifer saturated thickness) and as the sum of compressive volumes for the confined aquifer (product of storativity and pressure head above the top of the confined aquifer) and drainable volumes to represent conditions when the confined aquifer becomes unconfined. This calculation included the Carrizo and Upper Wilcox aquifers for the northern and southern regions and the Carrizo and Middle Wilcox aquifers for the central region. Aquifer storage volume, as used in this text, refers to recoverable storage, rather than total storage (product of saturated thickness and porosity per unit area).

The water budget was quantified for pre- and post-development periods for the entire aquifer and for the northern, central, and southern regions. The pre-development condition is represented by the results of the steady-state groundwater models (Kelley et al. 2004). The water budget for the years 1980 through 1999 was analyzed from the transient models to represent aquifer development. Impacts of pumpage on vertical flow gradients were evaluated based on comparison of simulated water levels/potentiometric surfaces between pre-development and 1999.

Sources of water for pumpage were estimated from the groundwater models of Kelley et al. (2004). Predictive pumpage from Texas State Water Plan (Texas Water Development Board 2002) and long-term average recharge were used to simulate the conditions from 2000 to 2050. Predicted pumpage in 2050 (0.17, 0.30, and 0.18 km^3 for the north, center and south) is about 15% higher in the north, 25% higher in the center, but 15% lower in the south than that in 2010. The prediction was mainly based on increased population growth and land-use change (increases in urbanization in all regions and decrease in irrigation in the south). Sources of water for pumpage were partitioned into change in aquifer storage, capture of local aquifer discharge in the outcrop area from ET and baseflow to streams, and capture of regional groundwater discharge through cross-formational flow from the Carrizo-Wilcox aquifer to the overlying Queen City-Sparta aquifer in the confined zone. In addition, the southern model was run using 1999 pumping (1) until a new steady-state was reached (negligible water storage change) to examine how long it takes for the aquifer to reach a new equilibrium and streamflow status under the new equilibrium and (2) for an additional 100 yrs without pumpage to assess aquifer recovery following 100 yrs of pumpage.

While detailed stochastic analysis would be desirable to assess uncertainty of simulation results to various input parameters, high computation cost and lack of information on probability distributions of input parameters prohibit such analysis. Therefore, sensitivity analyses were conducted for certain parameters to evaluate variability in simulation outputs to these parameters using the central Carrizo-Wilcox model. Key model parameters (recharge, transmissivity, and storativity) were increased and decreased by 20% and variations in the sources of pumpage were evaluated.

To quantify the input of water from overlying and underlying confining units to pumpage in a confined aquifer, a simulation was performed using six wells, each pumping 6×10^6 m^3/yr from the confined section of the Simsboro Fm. (about 50 km from the outcrop) in the central region. The pumping level is approximately equivalent to that from the Bryan-College Station well field. Total storage change in the aquifer and in the underlying and overlying confining units was estimated from the model results.

Sensitivity of water-level change in the outcrop to pumping appears to vary with distance from the outcrop. This sensitivity was evaluated using the central model. A transect of three pumping wells at varying distances parallel to the outcrop was simulated. Total pumping of those three wells was 26×10^6 m^3/yr. A well in the outcrop was used as an observation well. Change in water levels in the observation well with time up to 50 yrs of pumping versus change in distance of pumping from the outcrop zone was evaluated.

Table 2 Groundwater storage volume (km^3) and aquifer area (km^2) for the Carrizo-Wilcox aquifer. The values are representative for circa 2000 and were derived from the groundwater model results for 1999

Region	Unit	Area	Drainable storage	Compressive storage	Total storage
North	Outcrop	9,730	90	0	90
North	Confined	26,700	750	10	760
Center	Outcrop	6,140	60	0	60
Center	Confined	11,500	390	10	400
South	Outcrop	5,220	20	0	20
South	Confined	27,800	1,020	40	1,060
Total	Outcrop	21,000	170	0	170
Total	Confined	66,000	2,160	60	2,220
Total		87,000	2,330	60	2,390

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Table 3 Steady-state water budget for the Carrizo-Wilcox aquifer[a]

Region	Recharge	Streams	ET	X-fm flow
North	0.64	−0.34 (54)	−0.30 (47)	0.00 (0)
Center	0.19	−0.13 (66)	−0.05 (27)	−0.01 (6)
South	0.13	−0.07 (60)	−0.01 (5)	−0.05 (37)
Total	0.96	−0.54 (57)	−0.36 (37)	−0.06 (6)

[a] All values are in km^3/yr. Negative numbers indicate discharge from the aquifer. ET is evapotranspiration. X-fm flow represents cross-formational flow. Numbers in parentheses represent percentages of total outflow. Streams represent all surface-water features

Results and discussion

Groundwater storage

Groundwater storage in 1999 was estimated to be 170 km^3 of drainable storage in the outcrop and 60 km^3 of compressive storage in the confined aquifer, totaling 230 km^3. However, if the confined aquifer transitions to unconfined status when the potentiometric surface declines below the top of the confined aquifer, drainable storage in the confined aquifer would be 2,200 km^3, which is about an order of magnitude higher than compressive storage (Table 2). On a regional basis, groundwater storage is 850 km^3 (36%) in the north, 460 km^3 in the center (19%) and 1,100 km^3 in the south (45%). Low storage in the central region relative to the other regions generally corresponds to the differences in areas of the confined aquifer (Fig. 1). Drainable storage in the unconfined aquifer is highest in the northern region because the outcrop area is greatest in this region.

Groundwater flow dynamics

The steady-state, predevelopment water budget for the entire aquifer indicates that total recharge in the outcrop area is 0.96 km^3/yr and varies from 0.64 km^3/yr in the north, 0.19 km^3/yr in the center, and 0.13 km^3/yr in the south (Table 3). This decrease in total recharge mostly reflects a decrease in aquifer outcrop areas because differences in recharge per unit area are low (28 mm/yr in north and center and 20 mm/yr south). Groundwater discharge balances recharge and totals 0.96 km^3/yr. Discharge to streams represents 55–66% of total recharge. Discharge to riparian zone ET represents 47, 27, and 6% of recharge from north, center, and south regions, respectively. These variations in ET reflect differences in water-table depth which increases to the south. Deep recharge varies from ~0% in the north, 6% in the center, and 37% in the south. Lack of water flow into the deep regional system in the north is attributed to shallower water tables and rejection of recharge through ET and baseflow to streams in this more humid region and to the larger outcrop area in the north. Deep recharge into the confined part of the aquifer is balanced by regional discharge through upward leakage, or cross-formational flow, through the confining layer (Reklaw Fm.) to the overlying Queen City aquifer. Predevelopment conditions can be characterized as baseflow to streams (~57%) and groundwater ET (~37%, more significant in the north) and downgradient flow to the confined aquifer (~6%, more significant in the south).

Simulation results show that major rivers flowing across the outcrop zone of the Carrizo-Wilcox aquifer were predominantly gaining during pre-development. Exceptions include the Leona River and the Nueces River in the extreme southern region of the aquifer, which were losing.

The water budget varies annually in the transient simulation, in response to annually varying recharge and pumpage. A summary of the mean water budget for 1980–1999 is shown in Table 4. In general, mean recharge for 1980–1999 (20 yrs) is similar in magnitude to that during pre-development for all three regions. Mean groundwater pumpage is highest in the south (0.33 km^3/yr, 100% of total outflow) and lowest in the center (0.09 km^3/yr, 33% of total outflow). Overall, streams gain water from the aquifer but the amount has been largely reduced relative to that during pre-development. Streamflow gains in some reaches balance losses in other reaches in the southern region during the 20-yr transient simulation. Compared with pre-development, streamflow losses increased by 0.07 km^3/yr in the south. ET was reduced from predevelopment in all three regions because of water-table declines.

Impact of pumpage on aquifer storage and flow system

Groundwater pumpage depleted aquifer storage, particularly in the south, because pumpage was much greater in this irrigated Winter Garden region (~50% of pumpage in the entire Carrizo-Wilcox aquifer). The potentiometric surface decreased by ≥60 m over 6,500 km^2 in the Winter Garden area (Figs. 1 and 6). This depletion corresponds to ~4 km^3 of water based on cell-by-cell calculations of volume depletion from the model and includes some

Table 4 Water budget for the Carrizo-Wilcox aquifer from the transient simulation (mean for 1980–1999)[a]

Region	Recharge	Streams	ET	Pumping	Storage	X-fm flow
North	0.66	−0.25 (34)	−0.11 (14)	−0.17 (23)	−0.21	0.08
Center	0.20	−0.12 (45)	−0.04 (16)	−0.09 (33)	0.07	−0.02
South	0.13	0.00 (0)	0.00 (0)	−0.33 (100)	0.14	0.06
Total	0.99	−0.37 (338)	−0.15 (14)	−0.59 (53)	0.00	0.12

[a] All values are in km^3/yr. Negative numbers indicate discharge from the aquifer. X-fm flow represents cross-formational flow. Numbers in parentheses represent percentages of total outflow. Streams represent all surface-water features

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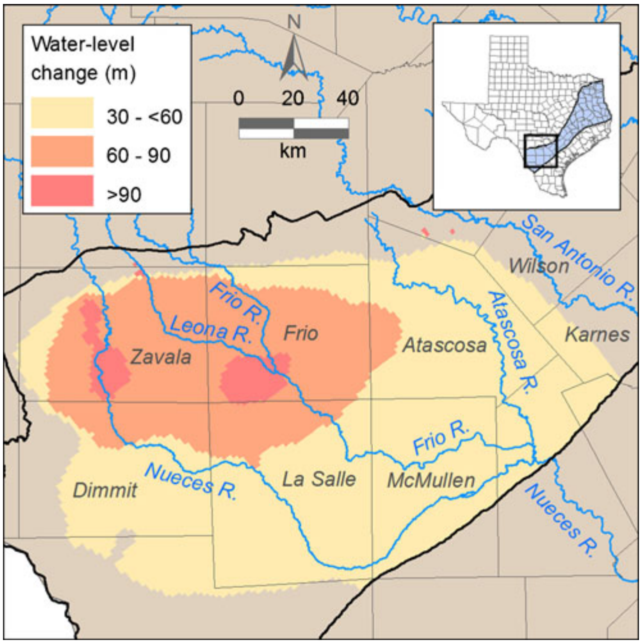

Fig. 6 Change in water-level elevation in the Carrizo Aquifer from pre-development to 1999 in the Winter Garden area. The *thin grey line* indicates county boundary and county names are labeled. Data from simulated pre-development and 1999 water levels using Kelley et al. (2004) model

conversion of confined to unconfined conditions close to the outcrop.

Groundwater pumpage has had large-scale impacts on the flow system. Simulation results indicate that pumpage increased stream leakage by a factor of 1.5 in the two streams in the south that were losing during predevelopment, and changed three additional streams (Atascosa, Frio, and San Antonio) from gaining to losing (Fig. 6). Therefore, five out of 10 major streams in the south were losing by 1999. In-stream flow could be impacted when streams change from gaining to losing. The maximum amount of streamflow loss is limited by the amount of streamflow available and hydraulic connectivity of surface water and the aquifer. Table 5 provides summary statistics on flows of selected major rivers in the central and southern regions (rivers were not included if no suitable flow gages could be located relative to the aquifer outcrop). Flow losses from rivers, due to pumpage, were estimated from the difference between post-development (1999; Table 2) values and predevelopment values (Table 4) and represent 0.2% of total mean streamflow in the central region and 7% in the southern region. The analysis indicates that capture of streamflow by groundwater pumping is a substantial portion of the streamflow budget in the arid south. The situation would be exacerbated in dry years when streamflow is reduced while capture by pumpage continues. Groundwater and surface-water connectivity also plays a role in streamflow capture and could become a limiting factor on how much streamflow can be captured.

Groundwater pumpage also reversed the regional vertical flow gradient from upward (from the Carrizo-Wilcox aquifer to the overlying Queen City aquifer) to downward (Fig. 7). Regional flow reversal occurred over ~44% of the total aquifer area by 1999. The most noticeable flow reversals are in the south in the Winter Garden area. Large flow reversals also occurred in the north, around Nacogdoches and Lufkin cities (Figs. 1 and 7), where pumpage for municipal and industrial purposes has been significant. Water quality could be affected by changes in vertical flow gradients.

Simulated impacts of pumpage on water flow are corroborated by other observational data in the south. The artesian Carrizo Springs (Fig. 1) in the far southern region ceased flowing in 1929 because of heavy pumpage (Brune 1981). Prior to significant groundwater pumpage (before 1900) artesian Carrizo wells flowed at elevations up to 210 m above mean sea level (msl). By the 1930s, flowing artesian wells were limited to elevations ≤150 m above msl, and by 1972 only certain wells flowed at elevations ≤100 m above msl (Hamlin 1988, Reedy et al. 2009).

Model simulations of sources of groundwater pumpage

Sources of water for groundwater pumpage differ by region due to difference in pumping stress levels and aquifer flow dynamics (Fig. 8). After 51 yrs of predictive

Table 5 Streamflow statistics of selected rivers. Streamflow (in km^3/yr) reflects measurements (1980–1999) from the nearest long-term gages before rivers enter the Carrizo-Wilcox outcrop. Site numbers 08178000 and 08180800 have 2 yrs missing data each. Data source: US Geological Survey National Water Information System (2011)

Site number	Site name	Region	Mean	Min	Max
08098290	Brazos River near Highbank, Texas	Center	2.75	0.29	10.12
08106500	Little River near Cameron, Texas	Center	1.60	0.17	6.93
08110325	Navasota River above Groesbeck, Texas	Center	0.10	0.00	0.24
08158000	Colorado River at Austin, Texas	Center	1.67	0.73	6.67
Total		Center	6.12		
08168500	Guadalupe River above Comal River at New Braunfels, Texas	South	0.52	0.08	1.84
08178000	San Antonio River at San Antonio, Texas	South	0.05	0.01	0.18
08180800	Medina River near Somerset, Texas	South	0.19	0.04	0.92
08192000	Nueces River below Uvalde, Texas	South	0.17	0.01	0.61
08197500	Frio River below Dry Frio River near Uvalde, Texas	South	0.05	0.00	0.20
08198500	Sabinal River at Sabinal, Texas	South	0.04	0.00	0.24
Total		South	1.02		

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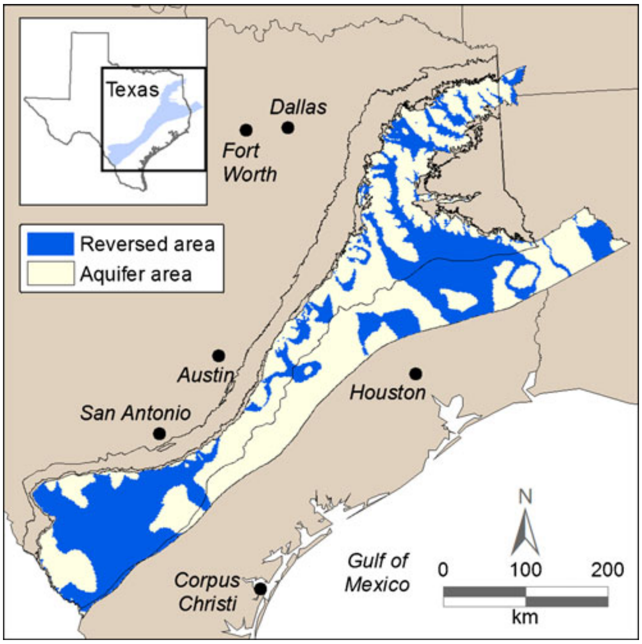

Fig. 7 Vertical flow gradient reversal from the overlying Queen City Sparta aquifer to the underlying Carrizo-Wilcox aquifer due to pumping. Based on difference between simulated pre-development water levels and 1999 water levels using Kelley et al. (2004) model

pumping (at year 2050), release of groundwater from storage represents ~41% of total pumpage for the entire aquifer (Table 6 and Fig. 8), the largest contributor. Pumpage is also supplied by capture of cross-formational flow (25% of pumpage) from the overlying units, the second largest contributor. Downward leakage (cross-formational flow) from the overlying units is highest in the south where most pumpage is from the Carrizo aquifer, close to the overlying Queen City aquifer (separated by Reklaw or Bigford Fm., an aquitard; Fig. 2). Lower amounts of downward leakage in the center and north is attributed to less pumpage from the Carrizo aquifer. In the central region, most pumpage is from the Simsboro Fm. (Fig. 2), which is separated from the Queen City aquifer by the Calvert Bluff (aquitard) and Carrizo (aquifer), resulting in lower downward leakage. Pumpage water is also supplied by reduced discharge in the outcrop, including reduced baseflow discharge to streams (24%) and groundwater ET (8%). Capture of groundwater ET is negligible in the southern region because ET was not a significant discharge mechanism during predevelopment and, therefore, could not be captured by pumpage. In the central and northern regions, ET and cross-formational flow contributions to pumpage level off over time, whereas stream capture continually increases, indicating that streamflow contribution should eventually become the dominant source of pumpage in these regions (Fig. 8).

Sensitivity analyses indicate that model results are not highly sensitive to variations in input parameters. In each case, percentage changes in input parameters result in much lower percentage changes in various contributions to pumpage (Fig. 9). Storage contribution and ET capture are sensitive to recharge variations. A 20% increase in recharge reduces storage contribution and increases ET capture by up to 3% in 50 yrs. Variations in storage contribution and ET capture increase with time because additional recharge represents more water input to the system which accumulates over time. A 20% increase in transmissivity raises streamflow capture by ~2% and is compensated by a reduction in ET and cross-formational flow capture. A 20% increase in storativity raises storage contribution by ~1% and is compensated by a reduction in cross-formational flow and streamflow capture. In all cases, a decrease in model input parameters has opposite effects with similar magnitude.

When pumping in the confined section of the Simsboro Fm. (the aquifer) in the central region, water is initially derived entirely from aquifer storage (Fig. 10). Developing cones of depression induce water from overlying and underlying confining units. Water from storage change in

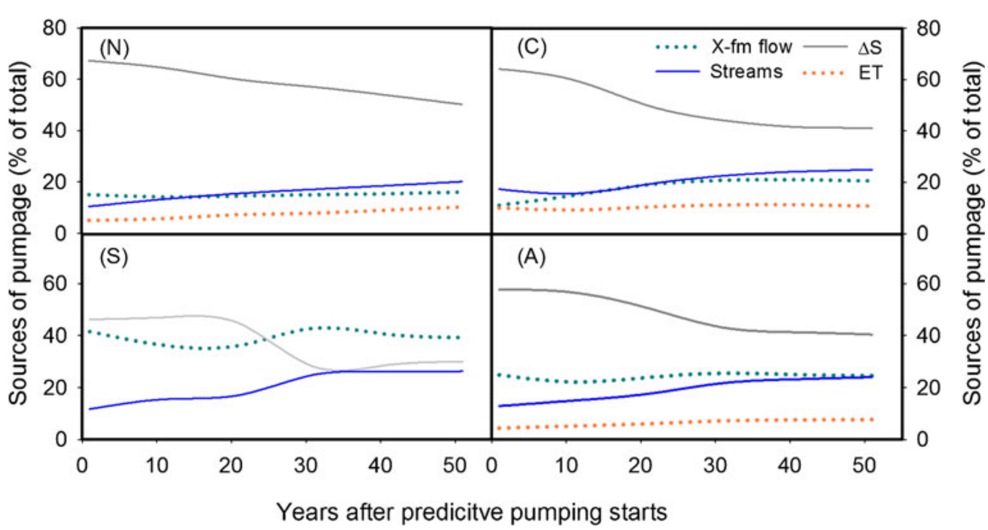

Fig. 8 Sources of water for pumpage in the northern (*N*), central (*C*), southern (*S*) and the entire aquifer area (*A*)

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Table 6 Sources of water for pumpage (percentage of pumpage) in the Carrizo-Wilcox aquifer after 51 yrs (at year 2050) of predictive pumpage for the northern, central, southern regions and the entire aquifer areas[a]

Region	ΔS	X-fm flow	Streams	ET
North	50	16	20	10
Center	41	21	25	11
South	30	39	27	0
Total	41	25	24	8

[a] ΔS represents change in storage. X-fm flow represents cross-formational flow. The total percentage for each region may not add up to exactly 100% due to some inter-regional flow between the overlapping area and the non-overlapping regions and rounding errors

the confining units exceeds that from the storage change in the aquifer within ~10 yrs for the overlying confining unit and ~20 yrs for the underlying confining unit. This analysis points out that leakage may be the dominant source of water in some confined aquifer systems. Neglecting the contribution from confining units would result in significant overestimation of storage change in the aquifer. Similar results were found by Konikow and Neuzil (2007) for the Dakota Sandstone aquifer.

Time scales of pumpage impacts

If a new equilibrium is reached, aquifer storage should not contribute water for pumpage. The southern model was run to approximate steady-state, which took ~500 yrs. In other words, pumping at 1999 rates will continue to decrease aquifer storage for up to 500 yrs. Simulations were also conducted to assess aquifer recovery after pumpage ceases and showed that aquifer storage would recover ~24% in 50 yrs and 38% in 100 yrs in the south (Fig. 11). Projected aquifer pumpage in this region for shale gas development is projected to last ~40 yrs; therefore, this simulation indicates that it would take centuries for the aquifer to fully recover.

The time scale of capture in a groundwater system is dependent on aquifer properties, distribution of pumping, and location of recharge and discharge. Water-level change in an observation well in the outcrop relative to changing distance of the pumping center in the confined aquifer is shown in Fig. 12. Drawdown decreases exponentially as the pumping center moves away from the outcrop zone within a distance of ~20 km. Beyond 20 km, pumping distance has a much reduced impact on water levels in outcrop wells. The trend is similar for pumping durations of 10, 30 or 50 yrs. The example illustrates that both time and pumping distribution should be considered in the capture processes.

Implications for water-resources management

Understanding impacts of future pumpage on water resources is a critical issue for management of this aquifer. The following questions should be addressed relative to future management of the aquifer.

Will pumping dry out the streams? In general, pumping is higher and streamflow lower in the south than in the center and north. The modeling analysis indicates that, on average, between 1980 and 1999, additional streamflow loss due to pumping represented 7% of total streamflow available in the south. Under a new steady-state condition after 500 yrs of pumping at 1999 levels, streamflow loss is about 10% of total available streamflow (approximated by mean streamflow between 1980 and 1999). Therefore, the current level of pumping should not dry out all the streams. However, some streams are in low-flow conditions or already go dry during dry years (Table 5). Capture of baseflow by pumpage could exacerbate the situation. Because of the coarse model grid and limited calibration to streamflow gains and losses in individual streams, use of this model to evaluate pumpage impacts at

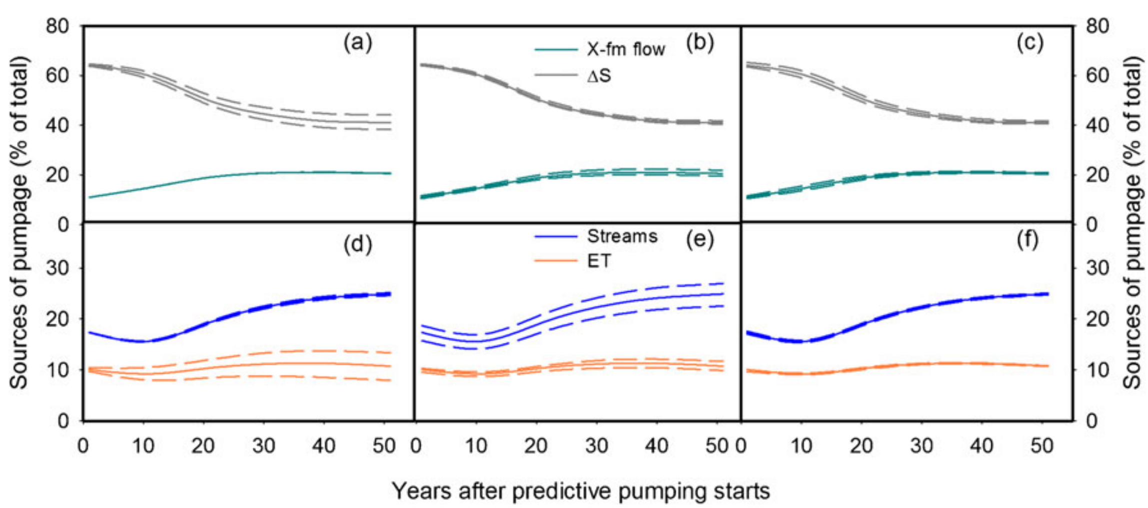

Fig. 9 Uncertainty estimates for sources of water analysis. **a**–**c** Change in cross-formational flow (X-fm flow) and storage release (ΔS) due to 20% change in recharge, transmissivity and storativity, respectively. **d**–**f** Change in streamflow and ET captures due to 20% change in recharge, transmissivity and storativity, respectively

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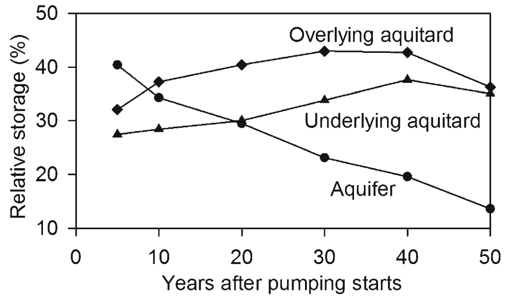

Fig. 10 Storage reduction in the overlying and underlying aquitards (Calvert Bluff Fm. and Hooper Fm., respectively) due to pumping in the aquifer (Simsboro Fm.)

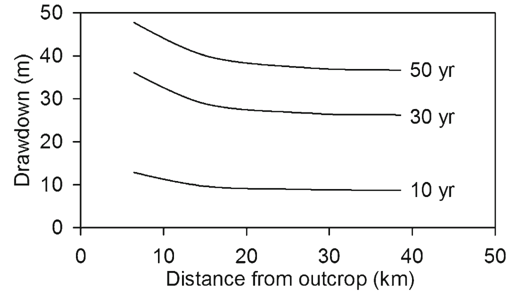

Fig. 12 Simulated drawdown response in an outcrop well due to change in pumping distance from the outcrop

the level of individual streams is inappropriate without local grid refinement at horizontal and vertical levels.

Will pumping affect groundwater quality? Cross-formational flow from confining units overlying and underlying aquifers contributes large amounts of water to pumpage in this system. Water in the adjacent confining units may have different chemistry from water in the pumped aquifer and it should be reflected in the mixed pumped water in the long term. To evaluate mixing of different waters, groundwater chemistry data from different geological units need to be sampled. Limited samples evaluated by Boghici (2009) indicate that there are no systematic temporal variations in total dissolved solids in the wells in the southern region; some show no change, while others fluctuate or show increasing trends. Further studies should be conducted to address this issue.

What are the implications of long time scales of pumpage impacts and recovery on aquifer management? As a dipping, mostly confined clastic rock aquifer system, the Carrizo-Wilcox aquifer has a flow regime that is not as dynamic as highly transmissive carbonate-rock aquifers, such as the Edwards aquifer in Texas. Groundwater flow in the aquifer is defined by a shallow, unconfined system in the outcrop where most recharge discharges to surface water and by a deep, restricted regional flow to the confined section of the aquifer. The regional flow is restricted by recharge that occurs in the narrow outcrop area and by the pressure in the convergence zone downdip. While the dynamic Edwards aquifer responds rapidly, within annual time scales, to stresses related to pumping and climate, it also recovers rapidly. However, the

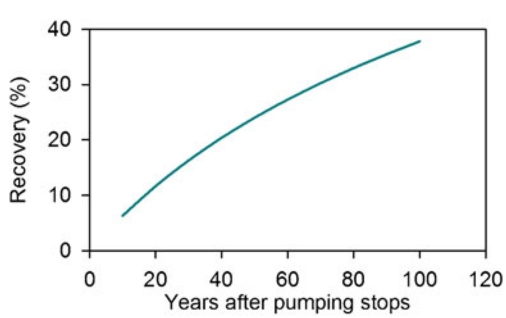

Fig. 11 Storage recovery after pumping stops. The southern Carrizo-Wilcox aquifer was pumped at 1999 level for 100 yrs. Storage recovery was monitored for 100 yrs after pumping stops

Carrizo-Wilcox aquifer takes up to 500 yrs in the southern region to reach a new equilibrium under current pumping stress levels and similar time scales are required for recovery. Therefore, this dipping confined aquifer system could be subjected to large stresses without the impacts of such stresses being immediately evident. These long time scales need to be considered in water-resources management. Approaches to mitigating pumpage impacts such as aquifer storage and recovery should also be considered in water-resource management plans. Because most of the pumpage occurs in the confined aquifer and there is limited flow from the unconfined to the confined aquifer, aquifer storage and recovery programs should focus on the confined aquifer where depletion has occurred.

Conclusions

Monitoring data and modeling analyses in this study provide valuable information on sources of groundwater pumpage, impacts of pumpage on water resources, time scales of impacts, and reversibility of pumpage impacts in a dipping unconfined/confined aquifer system exemplified by the Carrizo-Wilcox aquifer. Estimated groundwater storage includes 170 km^3 of storage in the unconfined aquifer and 60 km^3 of compressive storage in the confined aquifer; however, there is an estimated additional 2,200 km^3 of drainable storage in the confined aquifer if it transitions from confined to unconfined conditions with storage depletion. Although these estimates represent total amounts of groundwater in storage, feasibility of abstraction depends on pumpage impacts on the flow system.

Simulated predevelopment recharge was 0.96 km^3/yr and discharged predominantly to local and intermediate flow systems in the outcrop area as stream baseflow (57%) and groundwater ET (37%) with minor discharge to a regional flow system in the confined aquifer (6%). While irrigation pumpage for agriculture was dominant during historical times, municipal and industrial pumpage increased over time and exceeded irrigation pumpage in 2005. Groundwater pumpage changed three out of 10 major streams from gaining to losing in the southern region and reversed the vertical flow gradients from upwards to downwards in ~40% of the entire aquifer area.

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Simulations of predictive pumpage for 51 yrs (to 2050) showed that sources of water for pumpage are different by region. By the end of 2050, storage contributes ~41% of pumpage, capture of baseflow and ET contributes ~32%, and capture of regional flow accounts for ~25% of pumpage. These simulation results indicate that the aquifer storage will continue to be depleted in 50 yrs. In the central and northern regions, contributions from capture of groundwater ET and regional flow level off over time, whereas stream capture continually increases, indicating that the contribution from streamflow will dominate in the future. Sensitivity analysis suggested that the sources of pumpage results are relatively insensitive to change in key input parameters such as recharge, transmissivity and storativity. Modeling analysis indicates that it will take ~500 yrs to reach a new equilibrium in the southern region of the aquifer. Analysis of recovery times after pumpage ceases indicates that groundwater storage would recover by ~40% in 100 yrs in the southern region of the aquifer.

Management scenarios need to consider the mechanism of pumping capture and the long time scales to reach sustainable pumpage in this system. In addition, long recovery times also need to be considered if pumping is greatly reduced after meeting transient demands such as those for shale gas production or other demands. Future studies should consider approaches to offsetting pumpage impacts using aquifer storage and recovery or other management options. The modeling framework provides a valuable tool to test current conceptual understanding of the system and to project impacts of future pumpage.

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