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Quantity and Fate of Water Salvage as a Result of Saltcedar Control on the Pecos River in Texas

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Executive Summary

This report presents results for the Subtask 3.3 of the Pecos River Basin Assessment Project sponsored by the U.S. Environmental Protection Agency (EPA) and the Texas State Soil and Water Conservation Board (TSSWCB). The overall objective of Subtask 3.3 is to examine the hydrologic impacts of *Tamarix spp*. (saltcedar) control along a 5 km segment of the Pecos River near Mentone, Texas. This report is also based on work supported in part by the Cooperative State Research, Education, and Extension Service, U.S. Department of Agriculture, under Agreements No. 2005-34461-15661 and No. 2005-45049-03209, Texas Cooperative Extension (TCE), and Texas Agricultural Experiment Station (TAES).

As part of the deliverables of this project, an existing monitoring network of 8 wells was examined and enhanced with 9 additional wells equipped with water level loggers. Land surface profile and piezometric surface profile were developed to characterize interaction of surface and groundwater for different seasons as well as for verification of monitored water levels. Flow measurements were conducted during a release of water from Red Bluff Reservoir in March 2005 to determine losses or gains within the selected reach. Continued water level monitoring data provide more detailed information about water exchange between surface water and groundwater under different flow conditions. Correlation analyses of river stage and groundwater levels in monitoring boreholes provided further insight.

Results show that the river is hydraulically connected with shallow groundwater for this 5 km segment, which is comprised of Sites A and B, near Mentone, Texas in Loving County. Generally, the river is losing water to the aquifer at both sites. A gentle hydraulic gradient exists on the east bank of the river while a steeper gradient occurs on the west bank probably due to different hydrological properties of soils. Seepage from the river not only recharges the shallow aquifer, but also creates groundwater flow parallel to the channel, which may eventually discharge back to the river downstream. The reversed hydraulic gradients also demonstrate complexity of the dynamic relationship between the river and the aquifer. Water loss at the treated Site A decreased dramatically following saltcedar control in 2001, and remained very low through 2004. This study conservatively estimates water salvage of 0.5 – 1.0 acre feet per acre from control of saltcedar at this particular site.

Salvaged water most likely contributes to aquifer recharge rather than increased streamflow. Vegetation return in the form of native grasses and saltcedar re-growth at Site A may be the cause of corresponding increases in water loss in 2005 and 2006. Site A may also be affected by the untreated adjacent upriver segment (Site B), resulting in over-estimated water loss. Although the saltcedar water loss and salvage estimates presented here are believed to be conservative, the extreme differences in yearly site conditions throughout the study made it difficult to compare pre and post treatment calculations with confidence. It is recommended that additional flow measurements for longer reaches, enhanced monitoring of surface water and groundwater interaction, and further studies on hydrological impacts of saltcedar control be conducted. For future studies using the paired plot method, it is recommended that both sites be logged for at least 3 years prior to treatment. To reduce the potential for upriver treatment affect on downriver study areas, it is recommended that hydrological and ecological conditions immediately upstream of each plot be alike.

Introduction

Dams, groundwater pumping and saltcedar (*Tamarix* spp.) have altered the hydrology and ecology of many river systems (Blackburn et al. 1982, Fleishman et al. 2003; Glenn & Nagler, 2005; Graf 1978; Graf 2005; Lite and Stromberg, 2005; Shafroth et al., 2005). The distribution of saltcedar in North America includes waterways and reservoirs throughout the greater southwestern United States and portions of Mexico. Saltcedar reproductive attributes and greater tolerance to stressors such as water table fluctuation and salinity facilitated the conversion from more diverse native plant communities to saltcedar dominated communities (Glenn and Nagler 2005; Lite and Stromberg 2005). Saltcedar produces many seeds during the entire growing season, utilizes both groundwater and vadose zone moisture, and its roots respond relatively quickly when the water table declines (Horton and Clark 2001). Diminished streamflows result in the accumulation of salts and organic debris, which leads to increased soil salinity and susceptibility to fires (Glenn and Nagler 2005; Lite and Stromberg 2005; Stromberg 2001).

The Pecos River is impacted by saltcedar invasion and flow regulation (Hart, et al. 2005). Originating in the Sangre de Cristo Mountains in northern New Mexico, the Pecos River flows southward through eastern New Mexico and West Texas and discharges into the Rio Grande. It is the largest river sub-basin flowing into the Rio Grande in Texas. The Rio Grande is relied upon by both Mexico and the United States for drinking water, irrigation and industry. As such, it depends heavily upon its major Texas tributary – the Pecos River. The decreasing water quality in the Pecos River has negatively affected the Rio Grande. According to the data of U. S. Section, International Boundary and Water Commission (USIBWC), the Pecos River contributes to the flow of the Rio Grande at an average annual rate of 8.6 billion ft^3, which accounts for 9.5 percent of the stream inflow into Lake Amistad. However, it also contributes to salt loading into Amistad at an annual rate of 0.43 million tons or 26 percent of the total salt loading (Miyamoto et al. 2006). Therefore, understanding of history, biology and hydrology of the Pecos River is important to the future of the Rio Grande basin.

To have a better understanding of the Pecos River system, Texas Cooperative Extension (TCE) and Texas Agricultural Experiment Station (TAES) scientists developed this project supported by U.S. EPA and TSSWCB. This project will assess the physical features of the Pecos River basin, facilitate communications with stakeholder groups and landowners throughout the watershed, and monitor the water quality of the Pecos River. A Watershed Protection Plan will be developed to assess current management practices and identify appropriate voluntary management strategies for future water quality enhancement or protection in the Pecos River basin. Subtask 3.3 focuses on the characterization of surface and groundwater interaction, and hydrologic impacts of saltcedar control.

Literature Review

Saltcedar is a woody phreatophyte imported from Eurasia in the late 19th century for stabilization of railroad rights of way, rivers, irrigation canals, and as an ornamental plant (Glenn and Nagler 2005). Saltcedar may occur as dense monospecific stands typical along the Upper Pecos River in Texas or as small patches within a vegetation mosaic representative of the Middle Rio Grande in New Mexico. "Variability in the degree of saltcedar invasion and the broad range of sites that it occupies complicate generalizations about its impacts and potential for successful control and restoration" (Shafroth et al. 2005).

Until recently, it was widely accepted that saltcedar used significantly more water than native trees. However, individual tree studies suggest saltcedar evapotranspiration rates are comparable to native woody species (Glenn and Nagler 2005; Sala et al. 1996). Although these individual tree evapotranspiration rates are comparable, stand level or regional water use by saltcedar may be substantially higher due to its ability to dominate the vegetative environment of riparian systems (Sala et al. 1996).

Saltcedar water use is site specific, controlled mainly by depth to water table, salinity and stand characteristics (Devitt et al. 1997; Glenn et al. 1998; and Hays 2003). Various techniques that include meteorological data, eddy correlation, sap flow, stomatal resistance, water budget, lysimeters, and monitoring wells have been employed to quantify saltcedar water use at tree, stand and landscape spatial scales. Still, there are discrepancies among estimates that lead to a lack of consensus concerning the amount of water used by saltcedar.

Sensors are used to quantify the dissipation of a heat pulse or the change in temperature between two points on a plant stem. This is known as the stem heat balance method. The temperature flux is a function of water (sap flow) movement. These data are extrapolated from the tree scale to stand scale by multiplying sapwood area and daily sap flux.

The stem heat balance method was used to estimate evapotranspiration (ET) by saltcedar and three native, riparian woody species along the Virgin River in Nevada (Sala et al. 1996). Water loss, based on leaf area, was comparable among the species. However, saltcedar had the highest leaf area index (LAI) and therefore the potential to use more water than the other species. In fact, saltcedar stand transpiration at one site exceeded potential evapotranspiration (PET) by a factor ranging from 1.6 to 2.0 during the summer of 1993. This study was conducted under high water table conditions. Saltcedar transpiration may have further exceeded that of the native species if the water table was lowered sufficiently to have a negative impact on the native species (Sala et al. 1996). Nagler et al. 2001 and Nagler et al. 2004) demonstrated *Populus* (cottonwood) and *Salix* (willow) root cuttings had higher LAI than saltcedar. Further work is needed to assess variability of LAI in saltcedar and the implications on water use.

Glenn et al. (1998) measured water consumption of saltcedar and other riparian species in pots grown in a greenhouse. Saltcedar and cottonwood exhibited significantly higher water loss than *Allenrolfea occidentalis* (pickleweed) and *Baccharis salicifolia* (willow baccharis). *Pluchea sericea* (arrowweed) and willow water use was similar to saltcedar and cottonwood.

Anderson (1982) measured net photosynthesis and transpiration on intact saltcedar twigs with an open gas exchange system. Saltcedar transpired relatively large amounts of water, but similar to other riparian plants measured *in situ* with the same technique. Stomatal resistance was controlled mainly by temperature, light and humidity rather than simply plant water status. The authors reported that stomatal resistance should be considered when meteorological data is used to predict ET and that a failure to do so may result in significant overestimates.

Micrometeorological techniques such as eddy covariance and correlation as well as energy budget computations have been used to estimate water use within riparian corridors. Failure to meet requirements for adequate fetch distance as well as ‘edge effect’ resulting from narrow bands of lush vegetation in surrounding desertic landscapes, reduce the reliability of these measurements. Although there were discrepancies between the two methods, results indicated that saltcedar used about 0.3 meters more water than replacement vegetation. Evapotranspiration was substantially less than PET, whereas Sala et al. (1996) found that saltcedar ET could exceed PET by a factor of 1.6 to 2.0.

Water use estimates in saltcedar thickets and replacement vegetation were derived using the eddy correlation technique and a combined eddy correlation- energy budget technique (Weeks et al. 1987). Saltcedar was cleared from 19,000 acres of the Pecos River floodplain between Acme and Artesia, New Mexico. Initially the site was cleared by bulldozing in 1967, and follow up root plowing took place in 1974. Estimated saltcedar water use minus replacement vegetation water use should have yielded approximately 20,000 acre-feet of water (Weeks et al. 1987). However, no increase in baseflow that could be directly attributed to saltcedar clearing was detected. Any increase in base flow may have been masked by groundwater pumping or unusually dry and wet years following saltcedar clearing (Weeks et al. 1987).

Depth to water regulates plant water use. Saltcedar seedlings were grown in lysimeters along the Virgin River in southern Nevada. Sapflow, soil moisture content, and plant water relations were monitored. Daily sapflow was higher in plants growing along the river's edge compared to those plants in an open stand and at the edge of the desert (Devitt et al. 1997). Sap flow as high as 4.3 lb H_2O ft $^{-2}$ leaf area day $^{-1}$ in plants located along the bank, while at the same time sap flow in the open stand and desert edge lysimeters was less than 0.82 lb H_2O ft $^{-2}$ leaf area day $^{-1}$ (Devitt et al. 1997).

The water balance is another technique used to estimate water use at the landscape scale. Multiple components should be measured including inputs (precipitation, groundwater discharge, change in soil moisture, and surface runoff) and outputs (seepage and evapotranspiration). However, it is challenging to accurately quantify water balance inputs and outputs at the landscape scale due to temporal and spatial heterogeneity of these attributes. In addition, because precipitation and evapotranspiration in arid environments are on the same order of magnitude, differentiation is complicated. For example, twelve components of the water budget were measured for periods of two to three weeks from 1963-1971 along a segment of the Gila River in Arizona (Culler et al. 1982). Phreatophyte clearing along a 14 mile segment of the Gila River in Southeastern Arizona began in December 1964 and was completed in March 1971 (Culler et al. 1982). Discharge was the largest and most variable component. Ground water inflow was relatively small throughout the year. Similar to a study on the Pecos River in New Mexico (Welder 1988), soil moisture was a major source of ET water. Soil moisture accumulated during the winter then rapidly declined from May-July as ET increased. ET losses were small compared to other components of the water budget and generally less than pan evaporation, which averaged 92 inches. Average annual ET was only 3% of the volume of water moving through the system prior to clearing and only 1% after clearing. Culler et al. (1982) reported annual ET depths averaged 43 in prior to clearing and 19 in after clearing. Depths varied depending on plant density. The post clearing estimates do not include any ET by replacement vegetation that would subsequently establish in the cleared areas. Accuracy of ET measurements was dependent on the volume of inflow and outflow. Sampling and bias errors for water budget components were substantial and far exceeded ET; Errors were 15% and 25%, pre- and post clearing, respectively. Estimates of ET were obscured when inflows and outflows were relatively large, compared to measurement periods when rainfall was low and ET was high. The fact that potential estimate error is greater than the ET component raises questions about the accuracy of these estimates.

Using a combination of base flow analysis and water level monitoring, Welder (1988) attempted to detect and quantify water that was salvaged from a saltcedar control effort along 132 km stretch of the Pecos River between Acme and Artesia, New Mexico. Saltcedar stands were an average of 2800 ft wide, covering about 19,000 acres. Depth to the water table was 10 ft or less. Initial saltcedar control began in March 1967 and continued until May 1979, with complete removal in 1975. Weeks et al. (1987) projected water salvage to be between 10,000 and 20,000 acre feet per year. From 1967-1982, an increase equivalent to less than 1 foot of water per acre of cleared saltcedar, was detected (Welder 1988). The discrepancy may have been attributed to reduced transpiration as a result of clearing, increased precipitation recorded in the mountain recharge zone, a decrease in irrigation pumping, or some combination of these factors (Welder 1988). Failure to detect a substantial increase may also be due to the fact that much of the water formerly transpired by saltcedar was in fact precipitation that percolated into the upper soil layers, not groundwater. This water would not have contributed to base flow prior to treatment. Furthermore, if saltcedar removal resulted in a substantial rise in the water table, as much as 50 cm per year could be lost to evaporation from the soil surface via capillary rise given certain soil conditions. This also would negate any increase in baseflow. Welder (1988) concluded that some water was salvaged, but it was probably less than the average annual base flow gain of 19,110 acre feet.

Analysis of diurnal fluctuations in wells screened across the water table has been used as a method to determine groundwater consumption by phreatophytes (White 1932, Troxell 1936, Gerla 1992, Hays 2003, and Loheide et al. 2005). As plants transpire during the day the water table declines if water use is significant. During the night when transpiration decreases or stops completely, the water table recharges (Loheide et al. 2005). This pattern was recognized by White (1932), who developed a method for analyzing well hydrographs to estimate plant water use. This approach can be readily implemented at relatively low costs, and provides continuous data, which is difficult to obtain with other methods. As the previously mentioned water table fluctuations are very small (often less than 0.4 in.), high-resolution and high-accuracy equipment is necessary for detection. Soil specific yield is also a critical element of the White method, and pains must be taken to ensure the appropriate values are used in the equation (Loheide et al. 2005).

There is still much to learn about the interactions between ET and groundwater recharge. Water savings resulting from reduced ET may be stored in the vadose zone or shallow aquifer and undetectable with the baseflow separation technique (Shafroth et al. 2005).

Objectives of the Project

A study was initiated in 1999 using shallow groundwater monitoring wells equipped with water level loggers to estimate net drawdown or recharge within saltcedar stands along the Pecos River (Hart et al. 2005). Wells were installed at two sites within a study area and monitored for one growing season before herbicide was applied to saltcedar at one site. Pre-treatment and post-treatment water level data from both sites was used to assess the impact of saltcedar control.

The objective of this study is to further explore the effects of saltcedar control on the fate of salvaged water and determine amount of water released to downstream flow and groundwater recharge by completing the following tasks: (1) characterize the aquifer beneath treated and untreated sites with borehole exploration; (2) install additional monitoring wells to configure subsurface flow patterns, (3) conduct flow measurements with designated releases from Red Bluff Reservoir, and (4) estimate annual groundwater use by saltcedar to determine potential water salvage from control.

In general, the river receives precipitation, runoff, groundwater discharge and release from Red Bluff Reservoir as inflow. River outflows include seepage into aquifers, evaporation, transpiration, and irrigation diversion. The proposed tasks will allow us to evaluate flows between the river and the aquifer. Ultimately, data collected will be used to predict the effect of saltcedar control on water quantity as well as quality in the Pecos River basin. Data can also be used for the development of a simulation model of stream flow and salinity.

Study Area

Climate

The climate of the study area is arid, characterized by low precipitation, high evaporation rates, and large variations in daily temperature. Most of the rainfall in the region occurs from May through September and is strongly correlated with elevation (Schuster 1997). The average annual precipitation rate ranges from 9.5 in at Toyah (elevation 2,916 ft) to 13.3 in at Balmorhea (elevation 3,205 ft). Pan evaporation data collected at Balmorhea between 1940 and 1990 indicate evaporation rates of 115.7 in, which is more than five fold greater than the local annual precipitation.

Mean annual precipitation recorded at Red Bluff Reservoir is 10.73 in ± 5.02 in (NCDC 2004). Much of the rainfall occurs during the summer monsoon season as intense, short duration thunderstorms. Mean monthly minimum and maximum air temperatures recorded at Red Bluff Reservoir range from 28.4°F in January to 69.8°F in July and 62.6°F in January to 98.6°F in July, respectively (NCDC 2004).

Regional Hydrogeology

The Pecos River is a perennial stream interrupted by intermittent stretches of subsurface flow. The Upper Pecos River in Texas flows through the Delaware Basin, which is in the western part of the larger Permian Basin. The study area is located above the Cenozoic Pecos Alluvium aquifer, more than 1,500 feet thick (Figure 1). The aquifer is composed of Tertiary and Quaternary unconsolidated to poorly cemented alluvium such as sands, silts, gravels, clays, and caliche (Boghici 1999; Jones 2001; White 1971). These deposits accumulated in two troughs, the Pecos Trough in the south-central portion and the Monument Draw Trough along the eastern edge of the Delaware Basin (Ogilbee et al. 1962).

Hydraulic communication between the troughs is very limited and as such they should be regarded as two separate groundwater systems (Ashworth 1990). The aquifer is unconfined, although clay beds may produce local artesian conditions (Ashworth and Hopkins 1995). This aquifer overlies, and in some places is hydrologically connected to, adjacent aquifers including the Edwards-Trinity (Plateau) aquifer in Pecos and Reeves counties, the Dockum Group in Ward and Winkler counties, the Tertiary volcanics in Reeves County, and the Rustler aquifer (Ashworth and Hopkins 1995).

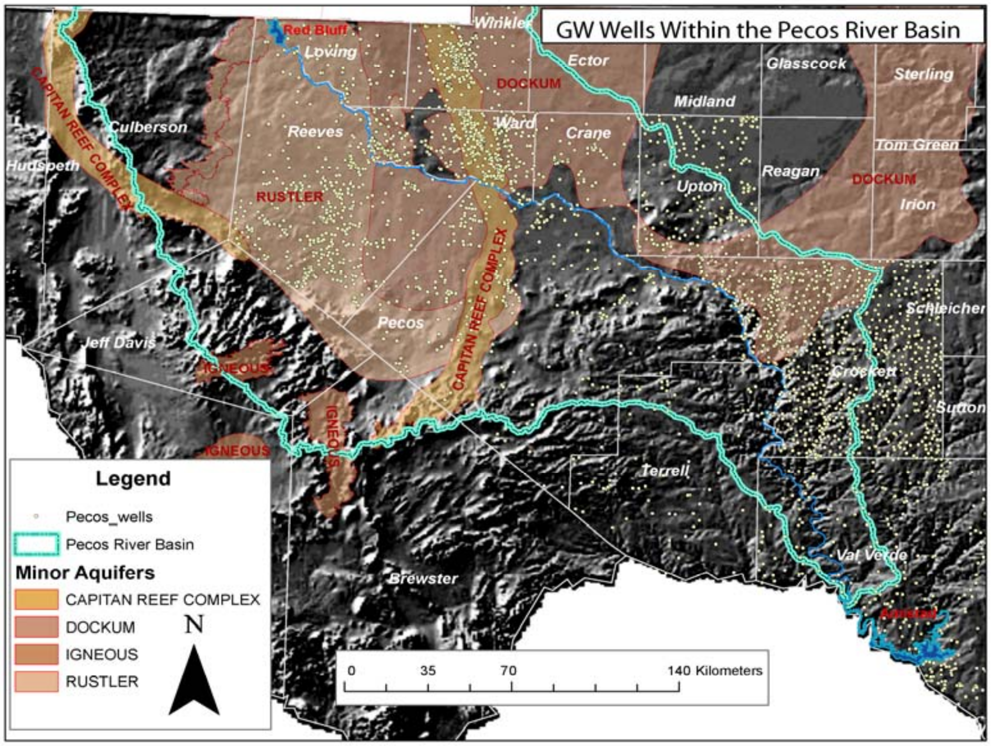


Figure 1 Pecos River Basin and underlying aquifers

The alluvial aquifer receives recharge from the infiltration of precipitation, seepage from ephemeral streams, cross-formational flow from adjacent aquifers, and irrigation return-flow (Ashworth 1990). Recharge only occurs after soil moisture is high enough to overcome the effects of surface tension that would otherwise adhere the water to sand grains. High soil moisture allows water to infiltrate through to the water table (Ashworth 1990). Consequently, natural recharge is episodic and associated with heavy rainfall. Recharge is only likely to occur during long-duration rainfall events or periods of frequent smaller rainfall events. Otherwise, the high evaporation rate is likely to preclude much of the rainwater falling on the alluvium from reaching the aquifer (LaFave 1987). Precipitation falling on the fractured volcanics of the Davis Mountains runs off into the local creeks and also likely infiltrates the alluvium (Schuster 1997). Recharge due to infiltration from ephemeral streams is also episodic, requiring sufficient precipitation to generate runoff through these streams. Cross-formational flow primarily enters the alluvium aquifer in the south and west where the aquifer is hydrologically connected to Permian (Rustler Formation), Triassic (Dockum Group), Cretaceous (Edwards-Trinity aquifer), and Tertiary volcanic aquifers (Ashworth 1990). Seepage from irrigation canals and irrigation return-flow also contributes water to the alluvium aquifer. Estimates of losses due to seepage from irrigation canals range from 30 percent to 72 percent of the diversion attributed to high permeability of sandy soils, while irrigation return-flow is estimated to be 20 percent of applied irrigation water (Ashworth 1990). Work by Grozier et al. (1968) concluded that the Pecos River reach was losing (including ET) up to $4.17 ft^3/s$ per mile between the gauging station at Orla and the Ward County Irrigation District No. 1 canal. Natural discharge from the alluvium aquifer includes ET adjacent to the Pecos River in the form of uptake by phreatophytes such as saltcedar and mesquite, and direct discharge into the Pecos River (White 1971). The major human-induced discharges are withdrawals for irrigation and municipal uses. Irrigation pumping peaked in the 1960s (339,397 acre-feet in Pecos County and 402,017 acre-feet in Reeves County in 1964) and has been declining since then (TWDB 1998).

Groundwater flow in the alluvium aquifer generally converges on the Pecos River, the main discharge zone, except where pumping creates cones of depression. In the Pecos River valley, depths to groundwater are 10 to 20 feet increasing to about 50 feet away from the river (Boghici 1999). Large water-level declines of more than 200 feet have been observed in pumping centers in parts of south-central Reeves and northwest Pecos counties, which has reduced the baseflow of the Pecos River and even reversed the flow direction of water between the river and the aquifer.

Water quality of the Cenozoic Pecos Alluvium aquifer varies from fresh to moderately saline. Pecos Trough groundwater is generally slightly to moderately saline while Monument Draw Trough groundwater varies from fresh to moderately saline. Sources of salinity vary, but quality tends to decrease with depth. Sulfate concentrations are higher in the northern and western portions of the Pecos Trough due to the presence of evaporite beds. Knowles and Lang (1947) stated in their report that the Salado formation is responsible for salt contamination in the Pecos River below Carlsbad.

Soils & Vegetation

Floodplain soils in the study area are classified as Arno-Pecos-Patrole Association (Jaco 1980). Textures are clay, silty clay, and silt loam, respectively. These soils are typically deep, nearly level, calcareous and moderately saline. They occupy salty bottomland range sites and as such support a relatively limited number of plant species.

Riparian vegetation consists mainly of a saltcedar canopy with a sparse understory of grasses *Chloris crinita* (trichloris), *Cynodon dactylon* (bermuda grass) and *Distichlis spicata* (inland saltgrass). Halophytic species such as *Allenrolfea occidentalis* (pickleweed) are also common along upper reaches of the Pecos River in Texas. The floodplain is inhabited by scattered *Atriplex canescens* (fourwing saltbush), *Isocoma wrightii* (rayless goldenrod), and *Prosopis glandulosa* (honey mesquite) with patches of perennial grasses in depressions.

Methods

Aquifer Characterization

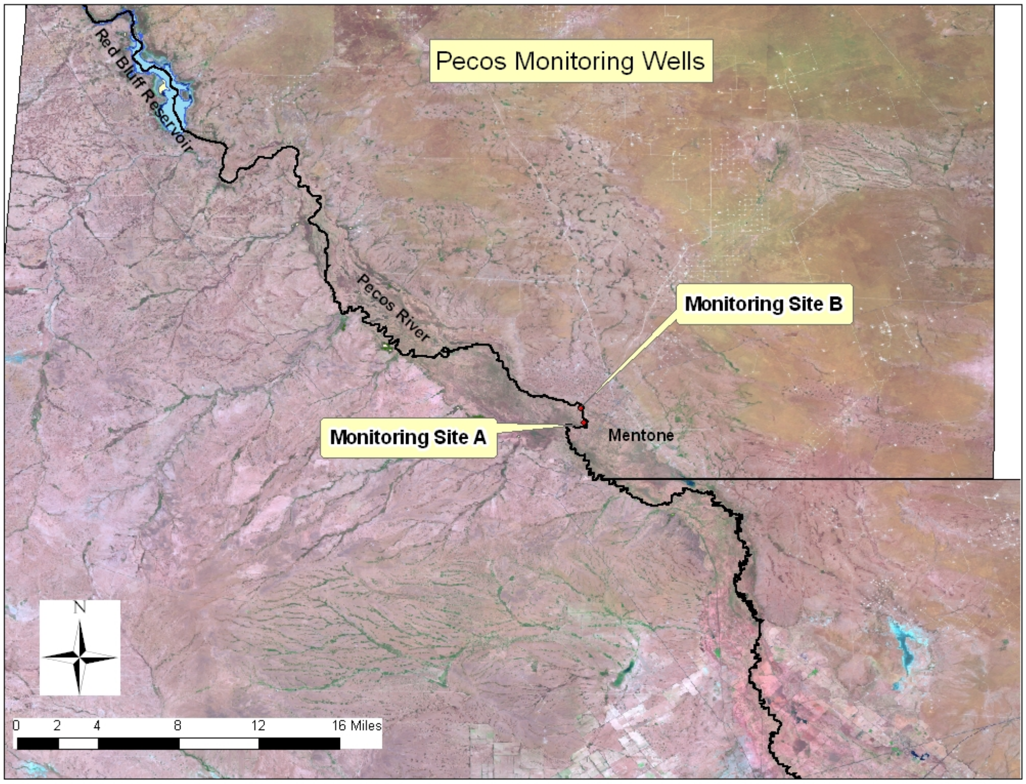
Aquifer hydrologic properties, in part, control the response of groundwater to changes in river flow. Field and laboratory techniques were used to characterize hydraulic properties of alluvial sediments. Slug tests were performed for each monitoring well to estimate hydraulic conductivity (Bouwer and Rice 1976). Particle size distribution of alluvial sediments was determined using a combination of dry sieve and sedimentation methods (Gee and Bauder 1986).

Several methods, such as hydrological profiling, inflow-outflow measurement, ponding tests, measurement of streambed temperature, or injection of conservative tracers (Diiwu 2004; Lee and Cherry 1978; Woessner 2000), can be used to understand hydrological interaction between surface water and groundwater in the underlain aquifer. In this project, both hydrological profiling and flow measurement were used to determine exchange between surface water and groundwater.

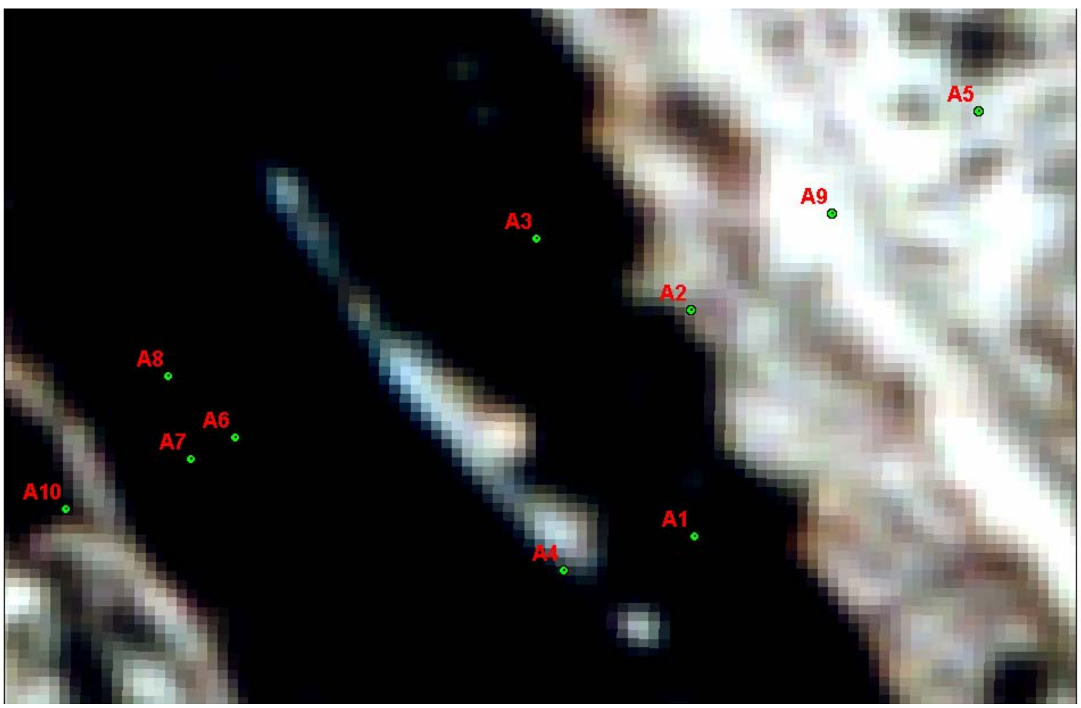
Monitoring Wells and Hydrological Profiles

A total of 17 wells were drilled and equipped with pressure transducers to monitor groundwater level in the shallow aquifer (Figure 2) at Sites A and B. The boreholes were cased with 2 inch diameter PVC pipes. The bottom 3 ft of each borehole was screened. The slots in the well screen were 0.04 in. The annular space around the pipe was filled with blasting sand to limit plugging of the well screen with fine soil particles. Cement was applied around the top of the well to prevent overland flow from entering the annular space around the pipe and entering the well. Seventeen water level loggers (Global Water WL 15) were installed in each well to record water levels hourly. Two additional pressure transducers were installed in the river through PVC pipes to record the river stage at both sites.

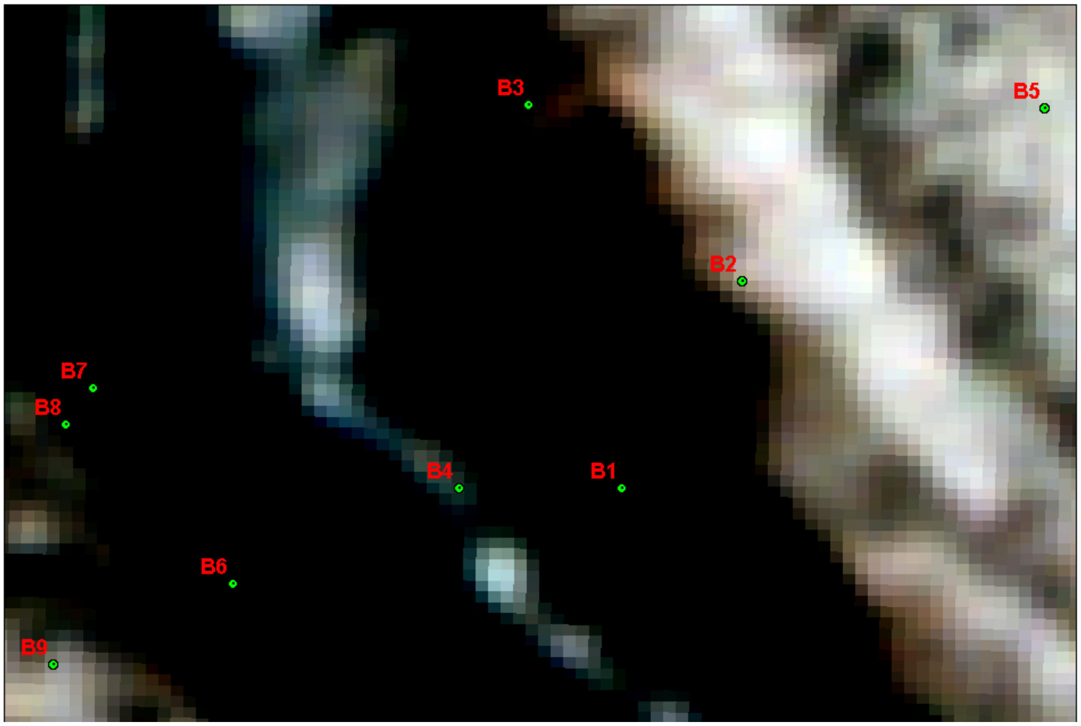
The instantaneous piezometric surface profile was also obtained by measuring the depth to the surface in each well with a Solinst water level indicator. These water level data were used to generate a hydrological profile, which helps us to understand hydraulic gradient between the river and the shallow aquifer as well as flows parallel to the river.



(a) Locations of test sites



(b) 9 monitoring wells and 1 river station at Site A



(c) 8 wells and 1 river station at Site B

Figure 2. Layout of monitoring wells and river stations along the Pecos River

Saltcedar water use and potential salvage from control

Water level data from 6 of the above mentioned wells was analyzed for the years 2001-2006 to estimate annual stand-level water use by saltcedar. Wells A 1-3 and B 1-3, located in the saltcedar stand on the east side of the river were logged for 1 year prior to aerially spraying Site A with Arsenal^{TM} herbicide, and logged throughout the growing season of the following years through 2006.

Hays (2003) developed a modified White (1932) equation to estimate daily water loss for this particular site: Q=((H1-L1)+((H2-L1/T1)xT2))(sy), where H=high water level, L=low water level, T=drawdown and recharge time, and sy=specific yield (Figure 3). Hays then reported saltcedar water use for 2001 (pre-treatment) using specific yield values derived from a soil-texture based specific yield triangle provided in Johnson (1967).

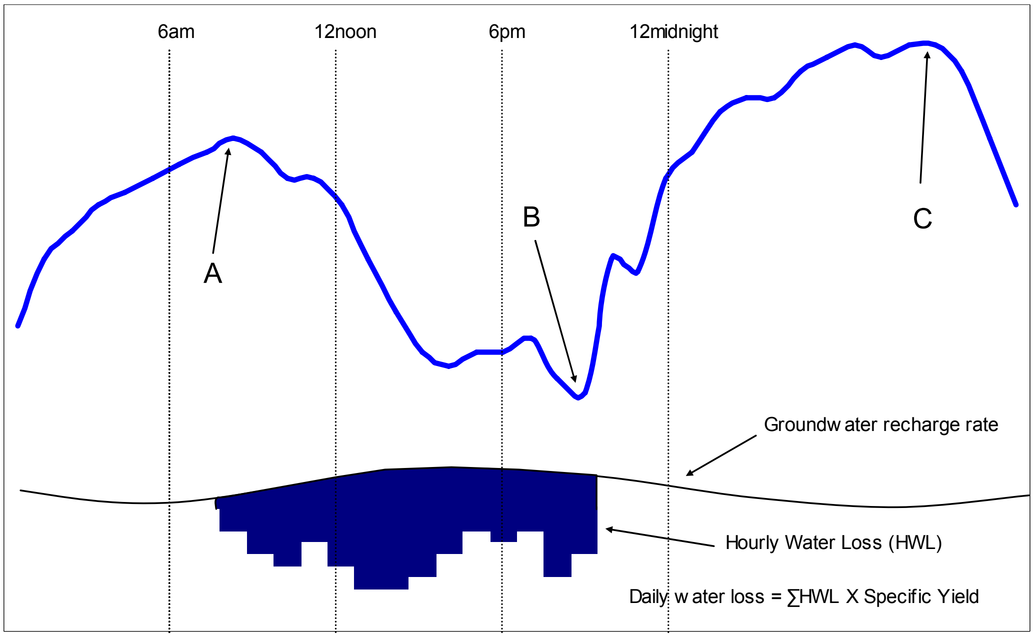


Figure 3. Example diurnal groundwater fluctuation. A=High 1, B=Low 1, C=High 2

This study utilizes the equation reported by Hays, but different specific yield values were arrived at by completing lab analyses on the original soil samples taken while drilling the wells. Specific yield for each 1-foot increment of the soil profile was calculated by subtracting the weight of the gravity drained sample from the weight of the saturated sample, divided by the total volume of water applied.

Use of the modified White equation was complicated by the dynamic nature of upstream releases from Red Bluff Reservoir. In order to account for situations where groundwater fluctuations were influenced by streamflow changes, a procedure for eliminating these events was used. Instrument "noise", or variability, was reduced by using a 3-period running average for each hourly water level reading. The allowable "stable" (S) value for the change in high water level between days was deemed to be 0.1 ft. Any calculation in which the water table fluctuation exceeded the (S) value was eliminated, thus reducing the impact of streamflow change. Calculations with drawdown or recharge time of less than 4 hrs, negative, or equal to zero were also eliminated. Among the remaining days of acceptable water use calculations, outliers were then identified using boxplots for each well on a yearly basis. These values were examined in relation to existing daily saltcedar water use estimates found in the literature, and excluded from final calculations if deemed unreasonable. Remaining water loss calculations for each well were pooled to get an average daily water use for each site.

Upon completion of the above procedures, it was then necessary to calculate values for the missing data in order to arrive at a yearly water loss figure for the site. This was accomplished by generating non-linear polynomial regression equations for each site on a yearly basis, with daily water loss being the dependent factor (y), and day of the year being the independent factor (x). Daily water use values were then totaled for April-September, the growing season, to arrive at total annual water loss. This method assumes there is no evapotranspiration by saltcedar at night, and that there is no significant water use prior to April or after September.

In order to estimate potential water salvage from control of saltcedar, the study site was set up following the EPA (1993) Paired Plot Study Design. The adjacent Sites A and B would be logged for 1 year prior to treatment (2001) for baseline data and to determine their relationship. After treatment of Site A, the control (Site B) would be used in subsequent years (2002-2006) to predict what the water use calculation would have been under a no-treatment situation. This would allow a simple comparison between actual and predicted water loss at Site A to arrive at estimated salvage due to saltcedar control.

Time Series Analysis of Water Level Data

Correlation and time lags between the river stage and the water level in the monitoring boreholes in the shallow aquifer were analyzed using the hourly time series data generated from pressure transducers. Figure 4 shows part of the time series for April 2005 at Site A as an example. The time series constructed from the original data of water levels contains “noise.” This “noise” may be caused by the sensitivity of the pressure transducer to temperature or water salinity rather than hydrological factors. To better interpret correlation between the time series data of the well level and the river stage, the time series model of the nonseasonal damped trend was used to smooth the data and remove the “noisy”data (Gardner and McKenzie1985; Sheng et al. 2006).

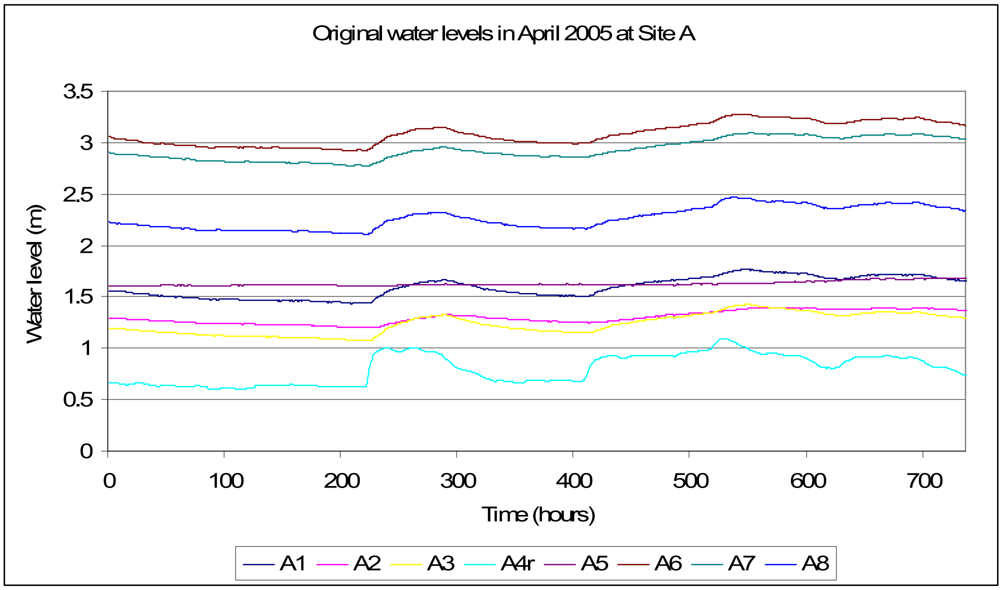


Figure 4. Original water level data in April 2005 at Site A.

The smoothed data are then used to determine correlation and time lags between the water level of the well and the river stage. Correlation, the traditional statistic characteristic is used to find the relationship between the water levels of the river and the wells. However, it should be noted that a change of water level in a well is behind the change of water level in the river. Therefore there is time lag between them even if the time series of the river and a well are correlated. To detect the time lags and the corresponding correlations, an ordered sequence is selected as the candidates of time lags. Then candidates in the sequence are sequentially checked so that a value of correlation can be calculated by assuming the selected values represent the time lags. Finally, the maximum of the computed correlation is selected as the correlation of the river and the well, and the corresponding value from the sequence as the time lag between them.

Inflow-Outflow Measurement

Another method to determine gains or losses for river reaches is inflow and outflow measurement. This involves the measurement of the quantity of water flowing into a particular reach of the river and the corresponding outflow from the same section. The difference between upstream inflow and downstream outflow consists of seepage (gains, discharge into the river or losses from the river) and evaporation losses within the measured reach (Rantz et al. 1982; Nolan and Shields 2000). Current meters were used to measure the speed at which water flows across the cross-section of the river. By multiplying the velocity of water by the cross-sectional area of the river, discharge can then be calculated. The sum of the incremental discharges yields the total discharge of the river channel (Figure 5).

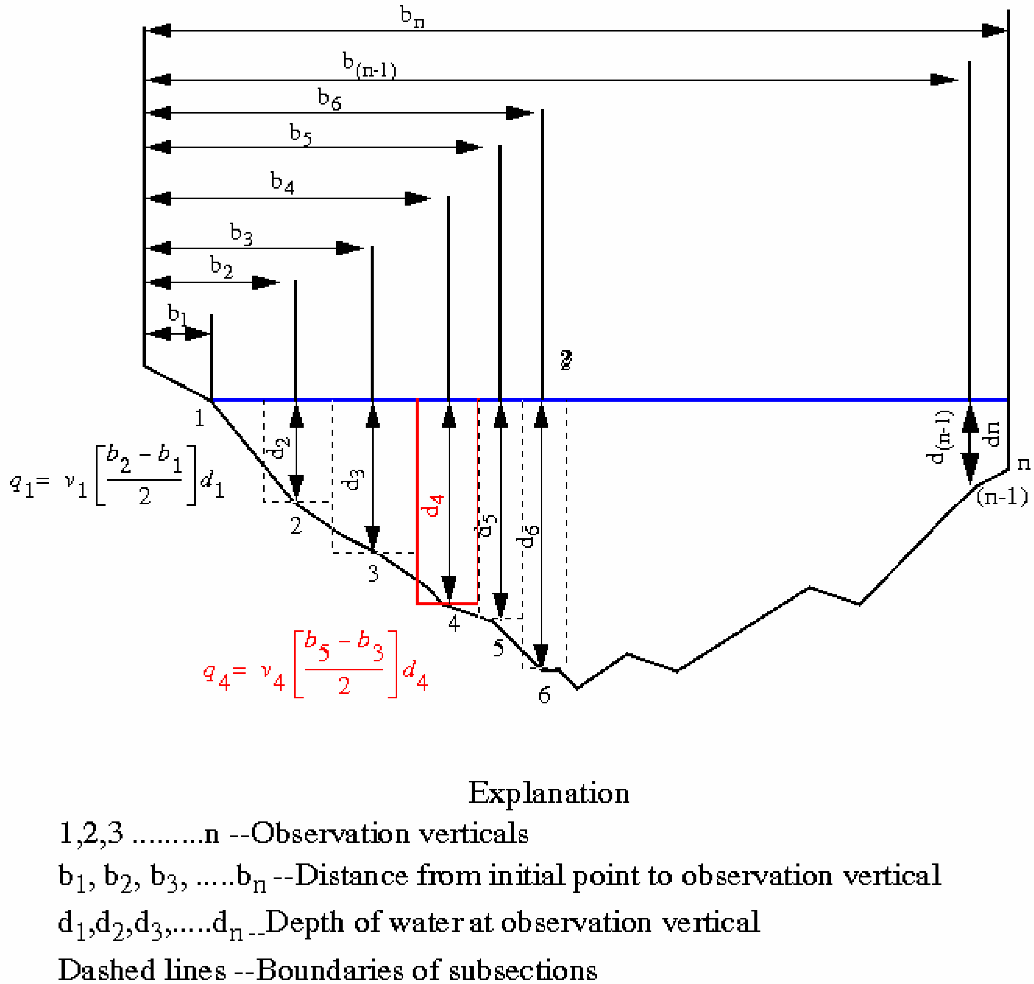


Figure 5. Sketch of discharge computing method (Nolan and Shields 2000)

Results and Discussion

Aquifer Characterization

Soil particle size analyses have been completed for most of the wells and boreholes (Figures 6 and 7). Soil textures at Site A vary with depth and distance from the river. Along the east side of the channel, surface textures are fine sandy loam and sandy loam, becoming coarser with depth. Soils on the west bank are similar, but there are some shallow clay layers present above the loamy and sandy subsoils. At Site B, located upstream from Site A, soils are mostly loamy, with some inclusions of fine sand and sand. Clay content is negatively associated with hydraulic conductivity. Therefore, wells with clayey soils will respond more slowly to changes in surface and groundwater levels. Overall, soil textures are coarser and thus hydraulic conductivity should be higher at Site A. Furthermore, horizontal hydraulic conductivity will also be greater than vertical conductivity.

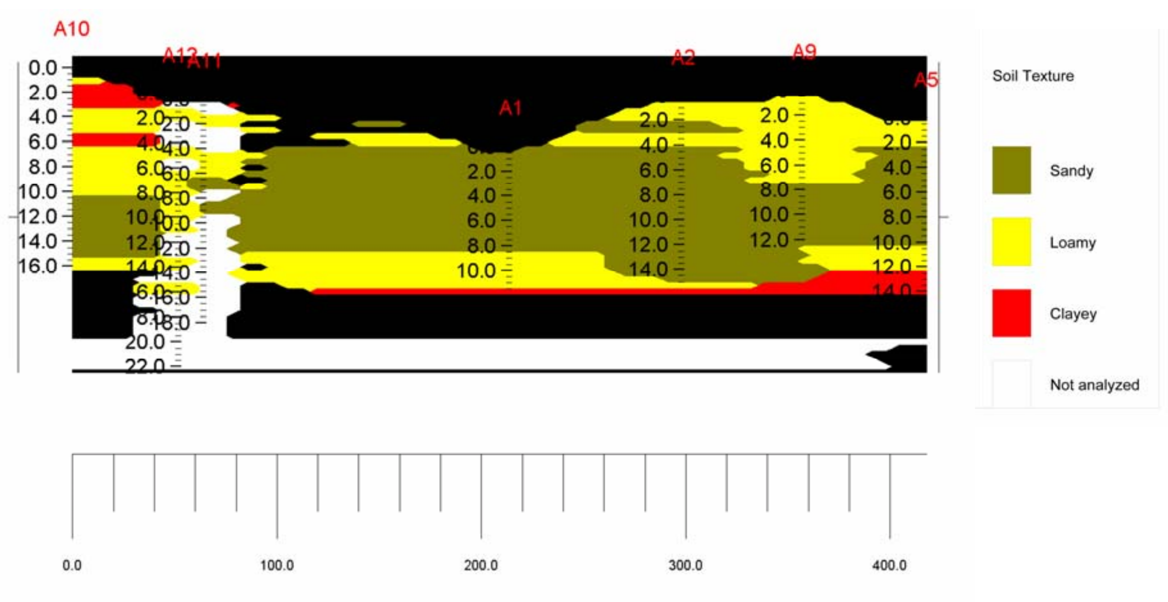


Figure 6. Soil texture from selected wells and boreholes at Site A.

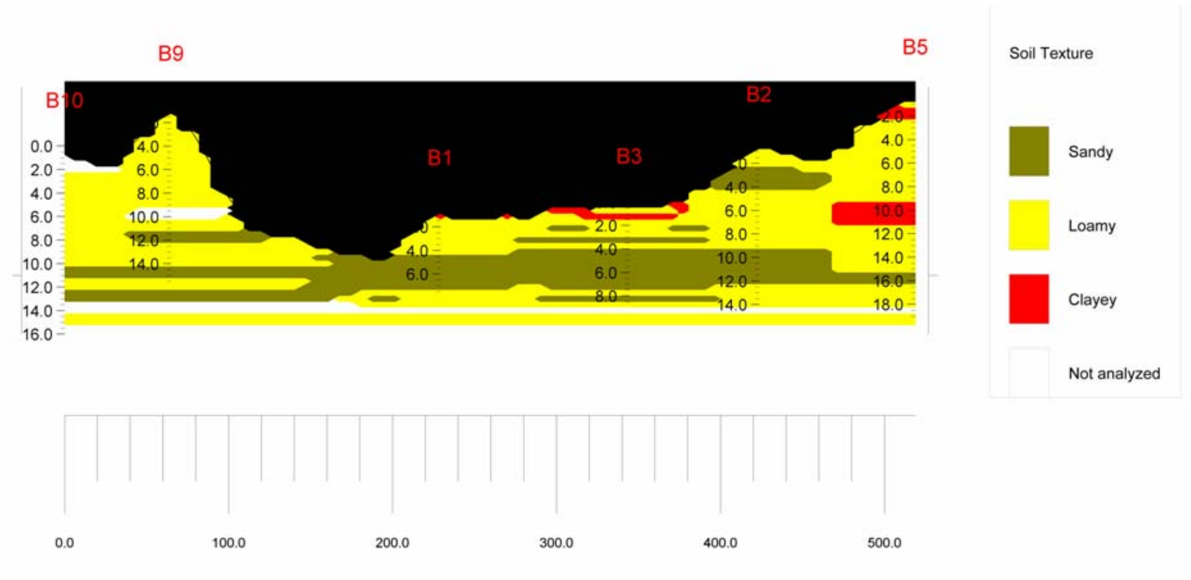


Figure 7. Soil texture from selected wells and boreholes at Site B.

Slug tests were performed in all the wells in December 2006. Only the bottom 3 feet of each well were screened, so slug test results represent only those depths. Saturated hydraulic conductivity (K_{sat}) was generally higher at Site A (Figure 8) than Site B (Figure 9), which an be attributed to thick sand deposits (Figure 6). Within Site A, well 1 had the highest value of K_{sat}. Wells 2 and 9 also have a relatively high K_{sat}, between 5 ft d^{-1} and 7 ft d^{-1}. However, K_{sat} values in wells 6-8, along the west bank, were less than 2 ft d^{-1}. Well 10, which is located on the floodplain, has slightly higher conductivity, nearly 3.5 ft d^{-1}.

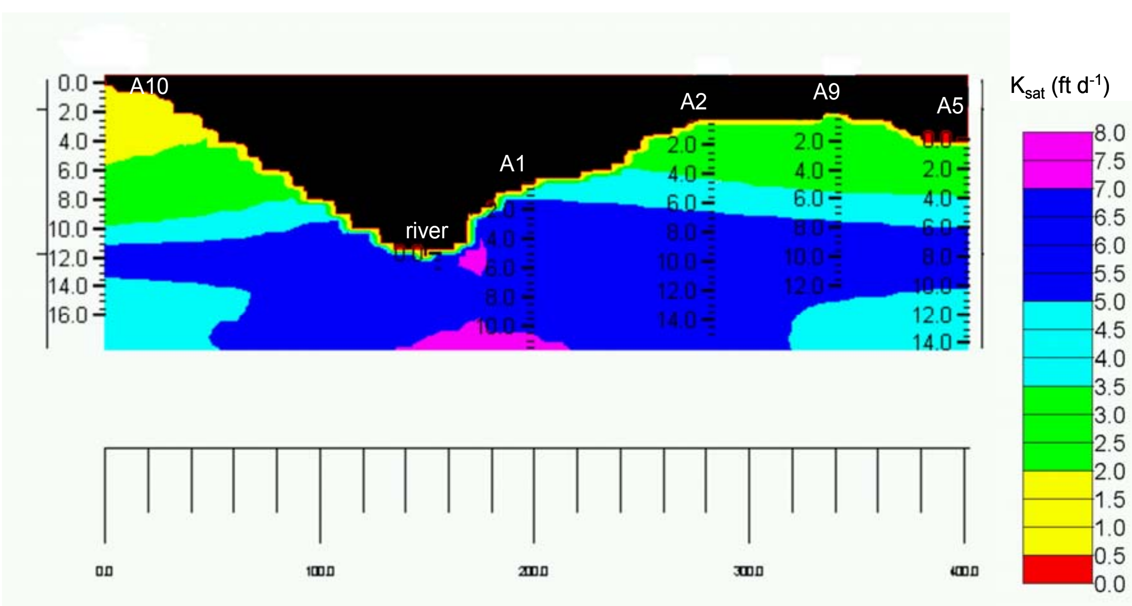


Figure 8. Saturated hydraulic conductivity at Site A.

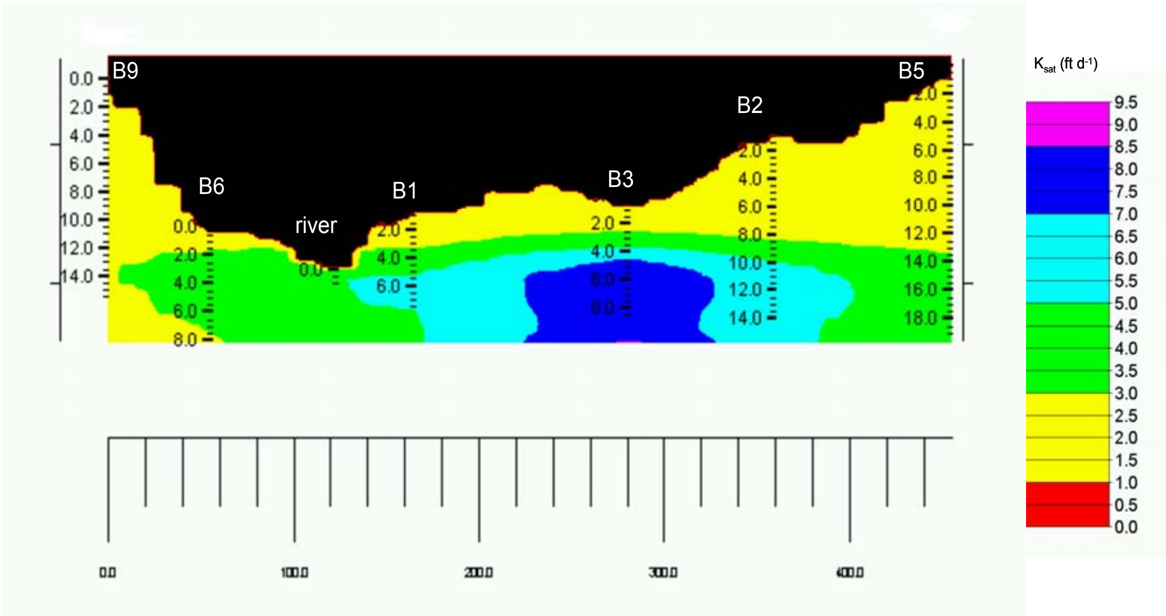


Figure 9. Saturated hydraulic conductivity at Site B.

A similar pattern was observed at Site B. Conductivity was higher on the east bank than the west bank at the observed depths, and values from wells 6–8 were less than 2 ft d^{-1} and increased to 3.47 ft d^{-1} in well B9 located on the floodplain.

Monitoring Wells and Hydrological Profiles

Figures 10 and 11 show water level changes in monitoring wells and the river stage at Sites A and B, respectively, for 2002 irrigation season. Water levels in those wells located close to the river show a strong hydraulic connectivity between the river and the shallow aquifer; however, wells such as A5 and B2 show slower and smaller responses to the river stage than other wells. This is probably due to low hydraulic conductivity of aquifer materials, which isolates these wells from the aquifer material (see Figures 6 and 7). Diurnal fluctuations at the untreated Site B were observed, which may be caused by ET of saltcedar or by the dynamic response of the aquifer to the river stage, or both.

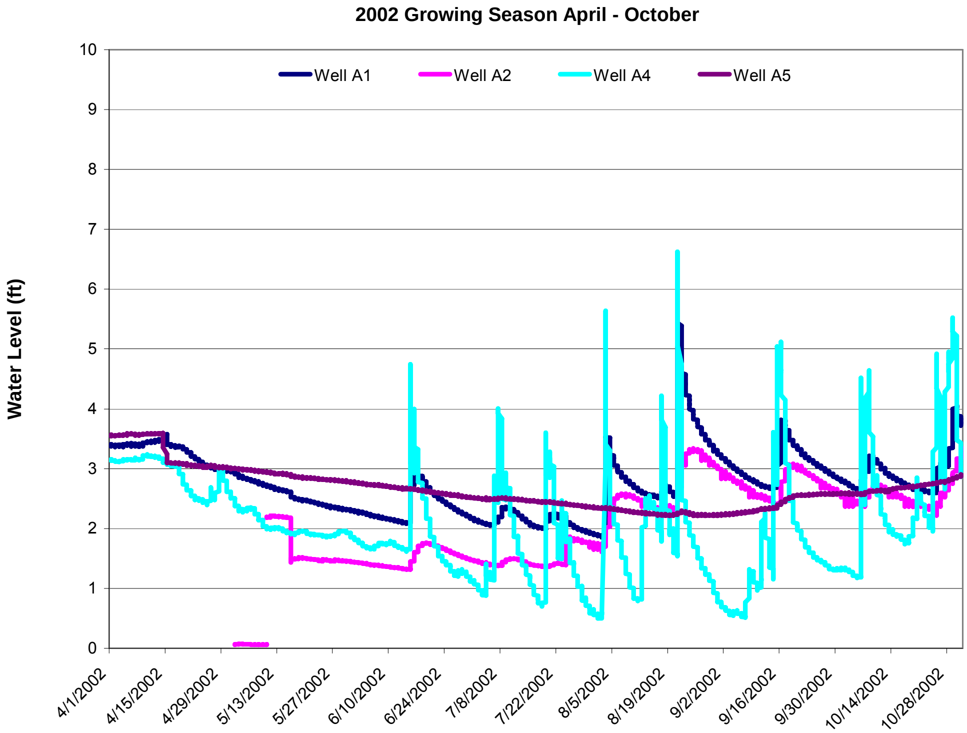


Figure 10. Depth of water in wells and river at Site A along the Pecos River near Mentone, Texas.

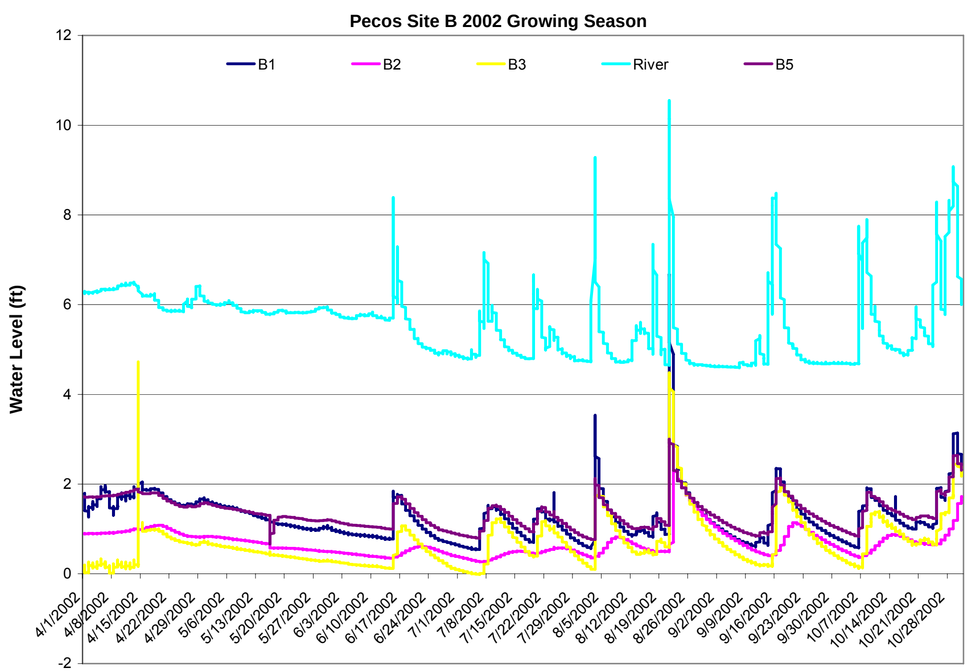


Figure 11. Depth of water in wells and river at Site B along the Pecos River near Mentone, Texas.

Figures 12 and 13 show hydrological profiles for both Sites A and B that consist of instantaneous water level measurements from monitoring wells and river stages. At Site A (Figure 12), water seeps through the river beds and banks to the shallow aquifer. The hydraulic gradient on the west side bank at Site A is greater than that on the east side, probably due to different hydrological properties of soils (Figure 6). This is consistent with slug test results and particle analysis results. The hydraulic gradient also changes with different flow conditions of the river. For a high river stage under a high flow condition in summer, the hydraulic gradient is greater than that under low flow condition in winter. The contours generated using piezometric head measurements also show that a portion of the seepage water flows parallel to the river within banks. This further complicates water exchange between the river and shallow aquifer.

At Site B (Figure 13), the hydraulic gradient on the west side bank is also greater than that on the east side, which is almost flat and even reversed at locations farthest from the river. The reversed flow direction is probably related to a time lag, which will be discussed in the next section of this report. The hydraulic gradient also changes with different

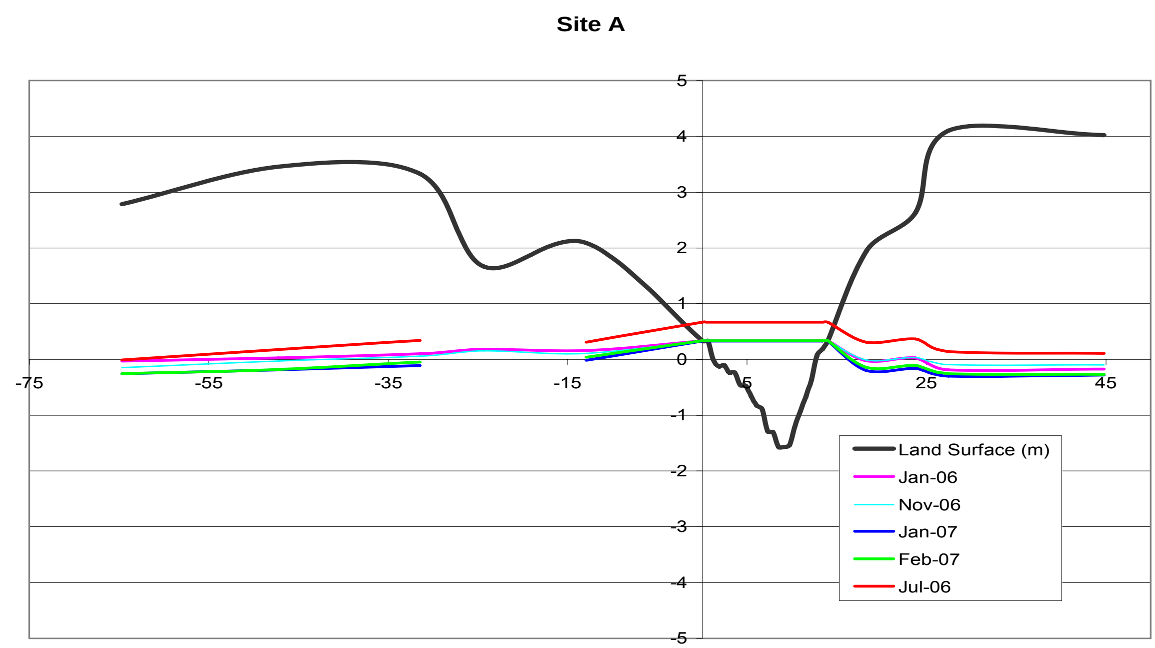


Figure 12. Hydrological profile at Site A (in meters)

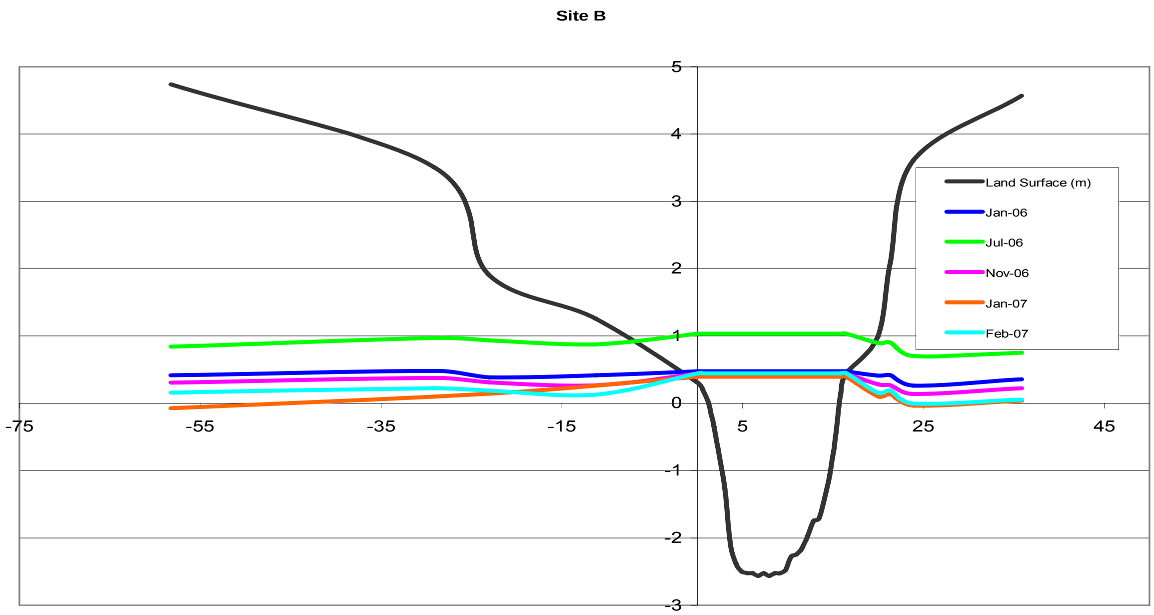


Figure 13. Hydrological profile at Site B (in meters)

streamflow conditions, but not as sharply as Site A. For a high river stage under a high flow condition in summer, the hydraulic gradient is greater than that under low flow condition in winter. The contours generated using piezometric head measurements also show that a portion of the seepage water flows parallel to the river within banks or under floodplains. This demonstrates complexity of the dynamic relationship between the river and the shallow aquifer.

No firm conclusions can be reached with such a small data set, but the results suggest spatial and temporal complexity that may have important management implications and justifies further study. For instance, if phreatophyte control appreciably reduces riparian ET, the result would be enhanced seepage, not streamflow, along losing segments such as this one. Additionally, further analysis could help identify key river stages and associated discharges that would permit parsimonious reservoir releases. That is, volumes large enough to meet delivery while limiting bank overflow and seepage.

Saltcedar water use and potential salvage from control

Yearly water loss and salvage estimates for Sites A and B are shown in Table 1 and Figures 14 and 15. Stand-level calculated water loss in 2001 (pre-treatment) ranged from

3.35 ft at Site A to 4.29 ft at Site B. These figures are significantly lower than those reported by Hays (2003), due to the difference in specific yield values used in this study. Saltcedar mortality rates at Site A in 2002 following control were approximately 90%, leaving virtually no living vegetation present in the riparian zone and barely detectable diurnal groundwater fluctuations. Post-treatment water loss at Site A dropped to 0.61 ft in 2002, or 18% of the 2001 calculation, and Site B (untreated) also dropped to 1.30 ft., or 30% of 2001.

Table 1. Annual water loss and salvage estimates

Year	Site A Loss (ft.)	n	R^2	% of 2001	Predicted A (ft.)	Estimated salvage(ft.)	Estimated % Salvage
2001	3.35	205	0.42				
2002	0.61	85	0.17	18%	1.02	0.41	40%
2003	0.67	209	0.04	20%	1.84	1.17	63%
2004	0.65	98	0.13	19%	1.33	0.68	51%
2005	0.91	66	0.08	27%	1.30	0.39	30%
2006	1.63	171	0.41	49%	1.73	0.10	6%
Year	Site B Loss (ft.)						
2001	4.29	217	0.26				
2002	1.30	157	0.03	30%			
2003	2.35	248	0.18	55%			
2004	1.70	88	0.15	40%			
2005	1.66	109	0.20	39%			
2006	2.21	160	0.34	52%			

Site B yearly percentages of the 2001 calculation were used to predict what Site A water loss would have been if left untreated. Actual Site A water loss values were subtracted from the predicted to derive estimated water salvage resulting from saltcedar control, which ranged from 1.17 ft in 2003 to 0.10 ft in 2006. Severe drought conditions in 2002 and 2003, and flooding in 2004 and 2005 led to equipment failure, calculation difficulties, and lower confidence levels in water loss and salvage estimates for those years.

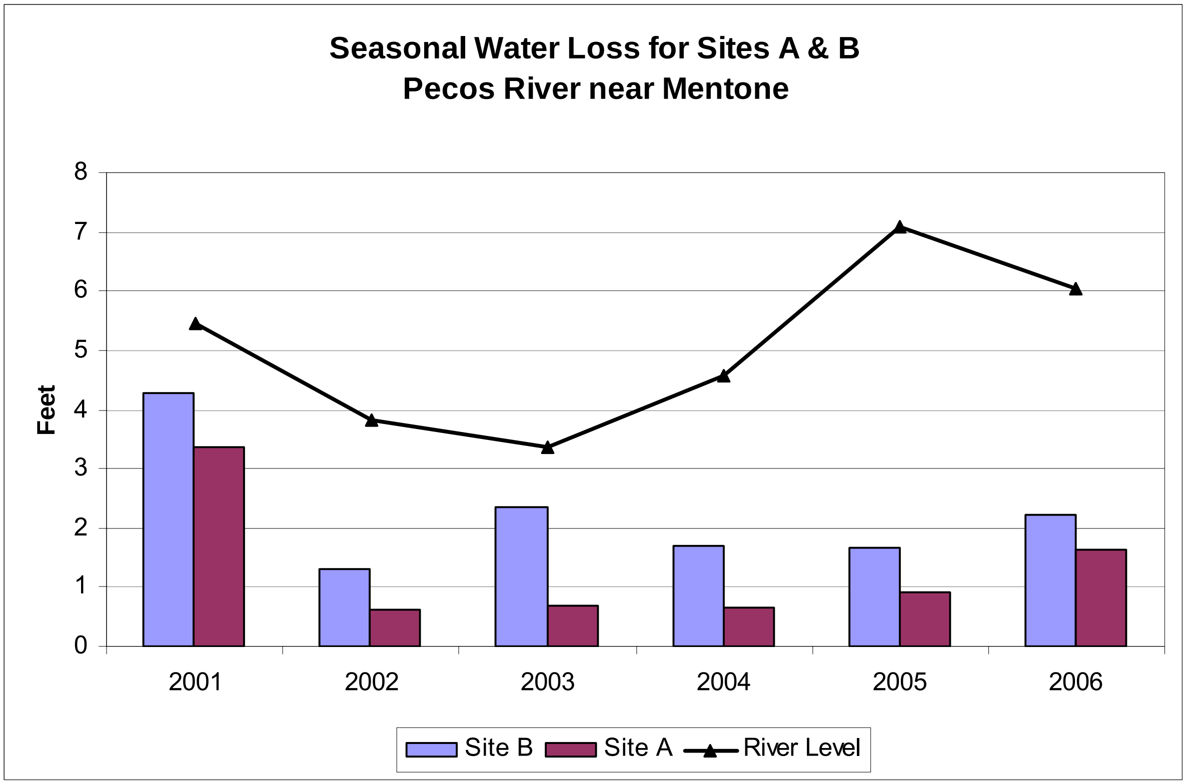


Figure 14. Annual water loss at Site A and B

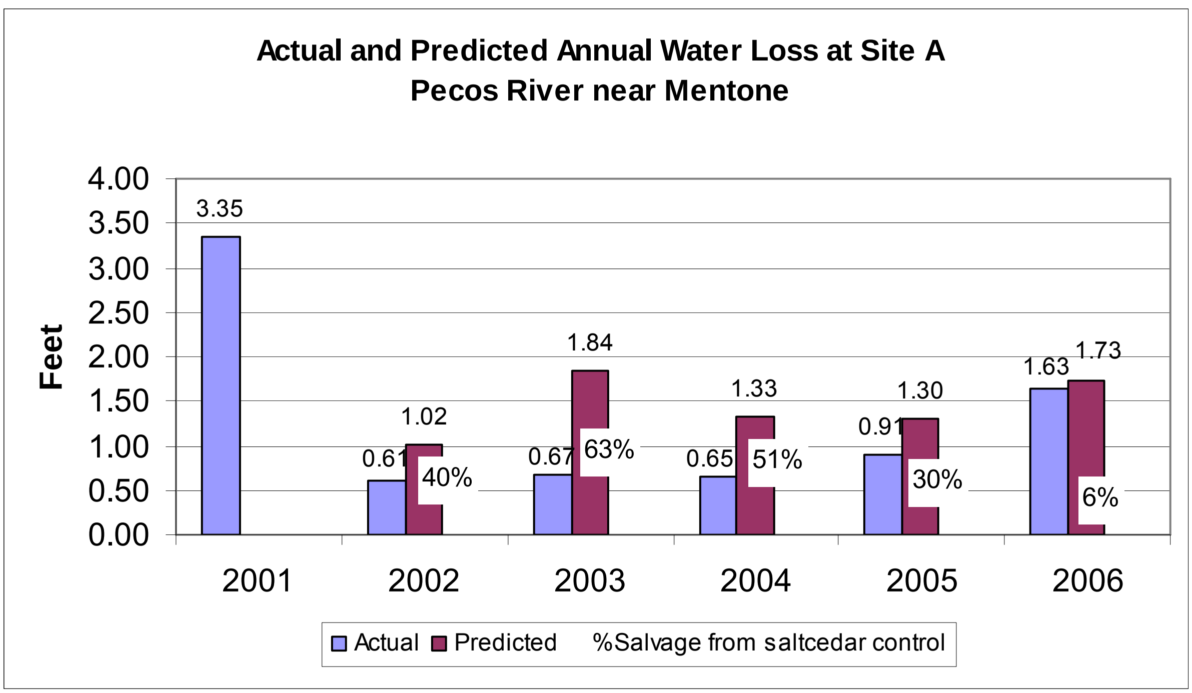


Figure 15. Actual and estimated annual water loss for Site A with % water salvage

The data exhibits a number of unexpected scenarios that arose throughout the course of the project. Water loss at the untreated Site B dropped significantly in 2002, although only the treated Site A was expected to do so. Severe drought in 2002 and 2003 resulted in no water releases from Red Bluff Reservoir in those years, and consequently some of the monitoring wells went dry. It is likely that the water table dropped below the saltcedar root zone at Site B in 2002, drastically reducing evapotranspiration (ET). By 2003, the roots appear to have recovered and again found the water table, as evidenced by the sharp increase in water loss at Site B. In contrast, water loss at Site B dropped significantly again in 2004 and 2005 due to flooding at the site. Heavy rainfall events and large releases from Red Bluff led to saturated soils and water above ground for extended periods of time at much of the site. This scenario may have suppressed saltcedar ET by reducing oxygen available to the root system, inhibiting the respiration process. Also, as discussed in the 'Aquifer Characterization' section of this document, saturated hydraulic conductivity is generally lower at Site B and may have affected the water loss estimates in 2004 and 2005. Site B water loss increases again to 2.21 ft in 2006, but never recovers to the pre-treatment estimate of 4.29 ft. Riparian saltcedar was treated both up and down river of Site B, leaving it to be the only living stand of trees in the area. Treatment of the saltcedar directly upriver is most likely having a direct effect on Site B, resulting in under-estimated water loss.

As expected, water loss at the treated Site A decreased dramatically following saltcedar control, and remained very low through 2004. As no living vegetation was present at site A in 2002 and 2003, it is highly unlikely that the water loss estimates for those years are attributable to plant ET. Pre-treatment vegetation transects completed by Hays (2003) show that prior to 2002, the vegetation at site A was dominated by the saltcedar monoculture, and therefore a portion of the water loss reduction is attributable to saltcedar control. This study conservatively estimates net water salvage of 0.5 – 1.0 acre feet from control of saltcedar at this particular site, which may be contributing to aquifer recharge rather than increased streamflow, as discussed in other portions of this document. Vegetation has begun to return to Site A in the form of native grasses and saltcedar re-growth, and may be the cause of corresponding increases in water loss in 2005 and 2006. Site A may also be affected by the untreated adjacent upriver segment (Site B), resulting in over-estimated water loss.

Although the saltcedar water loss and salvage estimates presented here are believed to be conservative, the extreme differences in yearly site conditions throughout the study made it difficult to compare pre and post treatment calculations with confidence. For future studies using the paired plot method, it is recommended that both sites be logged for at least 3 yrs prior to treatment. To reduce the potential for upriver treatment affect on downriver study areas, it is recommended that conditions immediately upstream of each plot be alike.

Time Series Analysis of Water Level Data

Correlation between the river stage and the water level at each monitoring well as well as time lags are shown in Tables 2 and 3. In general, the closer the well is located to the river, the better its water level correlates with the river stage. At Site A water levels at wells A1 and A6 are impacted by fluctuations in river stage more so than those wells farther from the river, such as A5. The aquifer acts as a buffer zone to reduce dynamic interaction between the river and shallow aquifer. Figures 16 and 17 demonstrate that both wells A1 and A5 reflect the river stress better at higher river stage than at the lower river stage.

At Site B, water levels at wells B1 and B6 are impacted by the river stress greater than those wells farther from the river such as B2. However, B5 has shown a good correlation even though it is the farthest well from the river. Additional analysis is recommended to verify such abnormality. Figure 18 and 19 demonstrate that both wells B1 and B2 reflect the river stress better at higher river stage than at the lower river stage. A comparison of the two sites clearly demonstrates the variability of responses to changes in river flow is influenced by differences in hydrological conditions surrounding each monitoring well.

Table 2. Results of Correlation Analysis for Time Series Data at Site A

Well	Correlation	Time lag (hr)
A1	0.9864	19
A2	0.9734	46
A3	0.9528	24
A5	0.9249	100
A6	0.9860	13
A7	0.9907	26
A8	0.9861	14
A10	0.9790	58

Table 3. Results of Correlation Analysis for Time Series Data at Site B

Well	Correlation	Time lag (hr)
B1	0.9948	7
B2	0.9882	24
B3	0.9948	11
B5	0.9904	17
B6	0.9939	5
B7	0.9982	3

For those wells corresponding to river water level, there are time lags between the river stage and the time series of water levels at each well (Tables 2 and 3). There are two major factors that determine these time lags: the distance between the river and the monitoring wells, and the hydraulic properties of the aquifer material between the river and the observation wells. In general, the time lag is longer for the wells farther from the river than those located closer to the river. However, heterogeneity of aquifer material further complicates the response of the aquifer to the river stresses. Salvaged water contributes more to recharge into the shallow aquifer rather than increase in the river flow. As a result, hydrological impacts of saltcedar

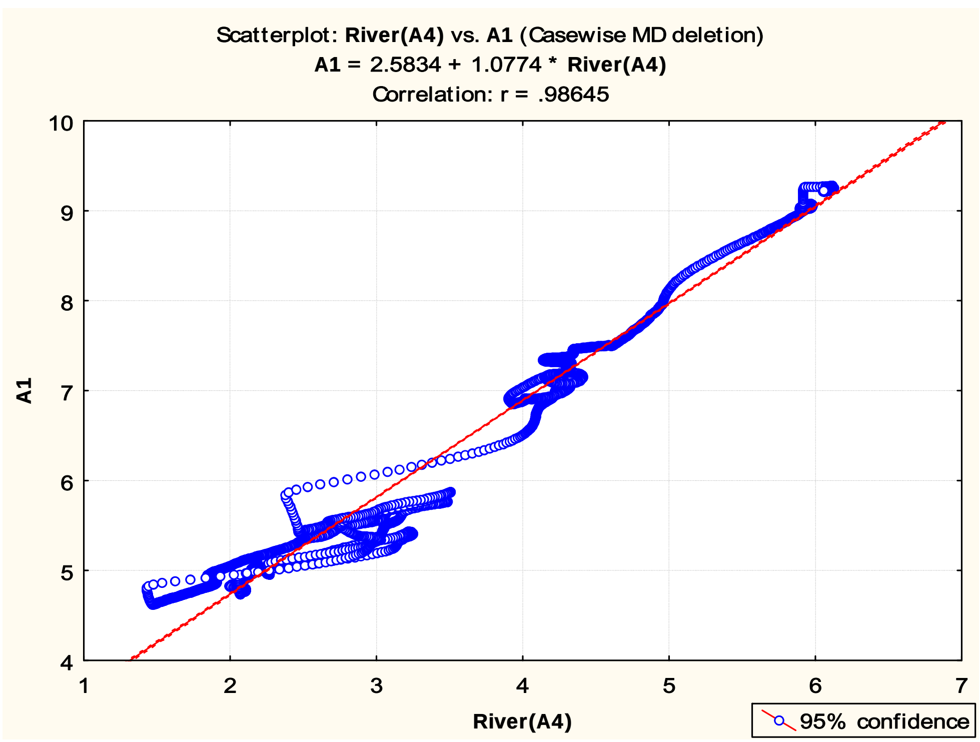


Figure 16. Correlation between the river stage and water level at the well A1

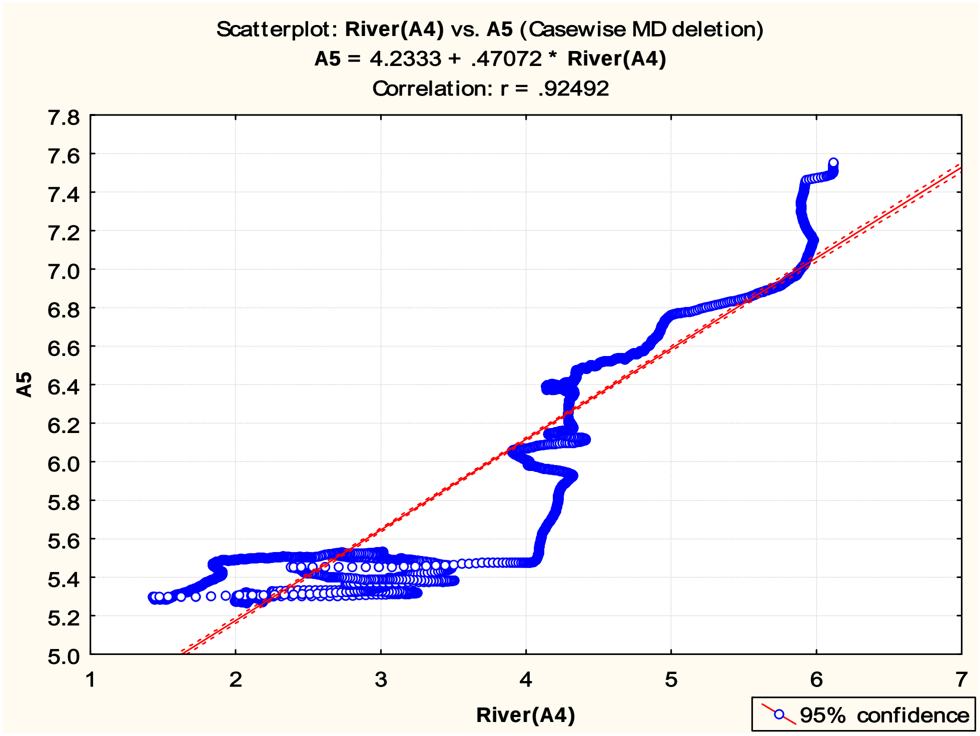


Figure 17. Correlation between the river stage and water level at the well A5

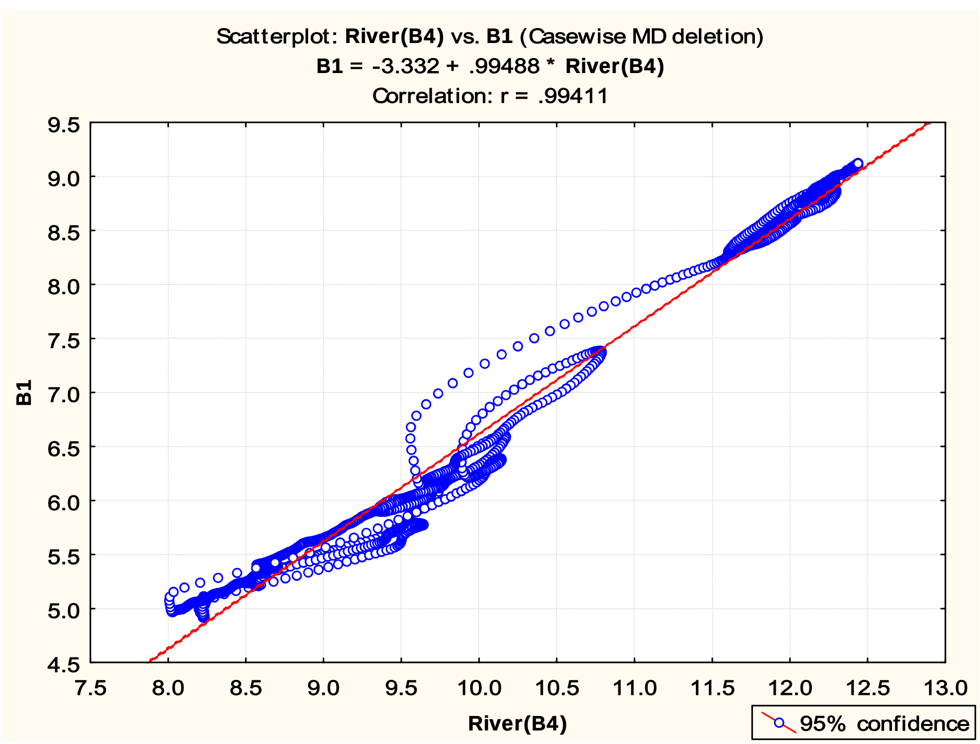


Figure 18. Correlation between the river stage and water level at the well B1

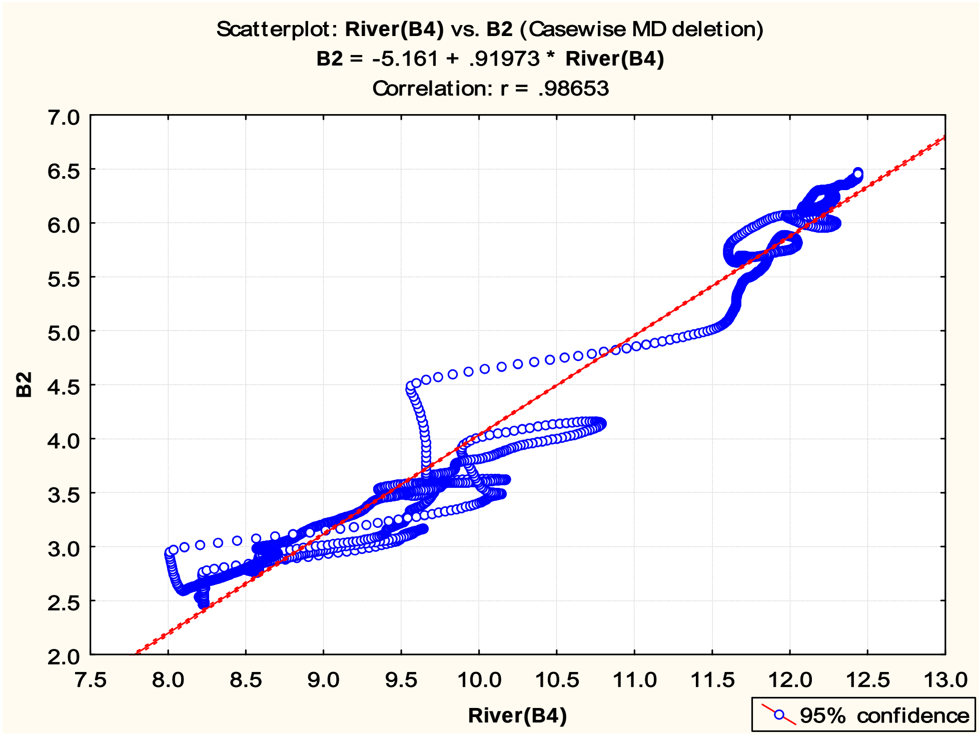


Figure 19. Correlation between the river stage and water level at the well B2

cannot simply be determined by the increase or decrease in the stream flow. Several other factors could also have affected the results. No release from the Red Bluff reservoir in 2002 and 2003 due to drought may have altered the hydrological conditions of the river and the shallow aquifer. Moreover, recovery of native vegetation may have masked the impacts of saltcedar control on the river flow. A better understanding of surface water and groundwater interaction is the key to a reliable assessment of hydrologic impacts of saltcedar infestation and water conservation benefits from its control. It is recommended that long-term impacts of saltcedar control be further evaluated with continued monitoring of the river-aquifer system and additional flow measurement for long reaches.

Inflow-Outflow Measurement

The difference in discharge measurements for the two cross-sections was very small, about 3 percent, and is within the measurement error (5 percent) associated with the equipment and technique. If flow conditions permit, additional discharge measurements will be computed, but comparison between the two sites will require an alternate approach.

A more detailed analysis of the monitoring well data will be required to identify time periods when the river stage is steady, but the hydraulic gradient in saltcedar increases. This change in gradient coupled with soil hydraulic conductivity will provide another estimate of water loss. Depending on depth of groundwater, evaporation can be computed and subtracted from this loss to provide an estimate of ET.

Summary and Discussion

From the observation of the river stage and groundwater levels, it can be concluded that the river is losing water to the aquifer for the selected reach under normal and high flow conditions. The dynamic relationship between the river and groundwater as well as its spatial and temporal variation complicates assessment of hydrological impacts of saltcedar control. Any reduction in ET as a result of saltcedar control may contribute to more recharge into the shallow aquifer rather than increase in streamflow if the selected reach is a losing stream, as depicted in Figure 20, during all observed flow conditions. Such recharge, if significant, may not be treated as immediate benefits to some stakeholders, but may have significant long-term impacts on the regional water resources.

In addition, seepage water also creates flow parallel to the river channel, which may eventually discharge back to the river downstream as hydrological conditions permit, or contribute to recharge into the shallow aquifer downstream. It should be noted that the time lag in the monitoring wells demonstrates the dynamics of flow in the aquifer. The shallow aquifer will take time to respond to changes in river stage. It can be inferred that any changes in the aquifer, either by ET or pumping, may not immediately affect the stream flow. Such a dynamic relationship should be considered in assessing hydrological impacts of saltcedar on the interaction between surface water and groundwater.

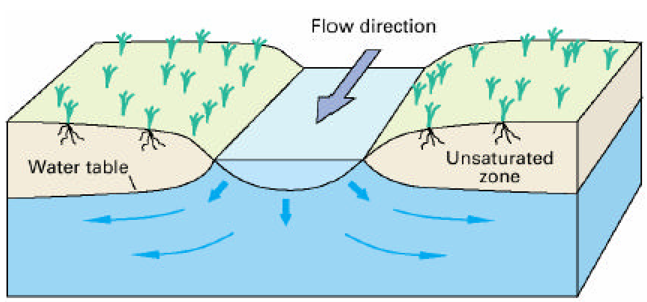


Figure 20. Flow pattern of a losing stream (from Winter et al. 1998)

This test site only includes a segment of the river which is a losing reach, and could not be used to represent hydrological conditions of the gaining reaches. The gaining reaches receive discharge from the shallow aquifer. The flow pattern and hydrological impacts of saltcedar control may eventually manifest in the stream flow if sufficient water salvage is obtained.

Recommendations

Based on preliminary results of the real-time monitoring of water levels, measurements of flow, and analysis of water quality, the following activities are suggested for further studies of the interaction of surface water and groundwater, which will eventually provide guidelines of implementing watershed management plans.

- Expand the study area beyond existing test sites to explore overall interaction of surface water and groundwater through seepage losses assessment and with additional monitoring sites. Conduct inflow-outflow measurement for a longer reach starting from Red Bluff Reservoir and characterize gaining and losing patterns of the river reach under high and low flow conditions. Select a gaining reach and establish a monitoring network to characterize hydrological process between surface water and groundwater.
- Monitor water quality changes within the test sites A and B as well as new monitoring sites, which may provide a deep insight into hydrological process between surface water and groundwater in a semi-arid region, and also help to identify potential sources of salinity.
- Conduct additional field investigation using geophysical methods to characterize the river beds and shallow aquifer accompanying with borehole samples. Use temperature probes to identify zones for interaction of surface water and groundwater, and assess water exchange between the river and the aquifer.
- For future studies using the paired plot method, it is recommended that both sites be logged for at least 3 years prior to treatment. To reduce the potential for upriver treatment affect on downriver study areas, it is recommended that hydrological and ecological conditions immediately upstream of each plot be alike.
- Develop numerical models to evaluate different scenarios for improvement of water delivery efficiency, assess impacts of saltcedar control under different hydrological conditions with emphasis on the interaction of surface and groundwater,

and evaluate strategies for integrated surface and groundwater resources management within the basin or watershed.

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