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Source: Invasive Plant Science and Management, 2(4) : 309-317

Published By: Weed Science Society of America

URL: https://doi.org/10.1614/IPSM-09-009.1

Water Loss and Salvage in Saltcedar (*Tamarix* spp.) Stands on the Pecos River, Texas

William L. Hatler and Charles R. Hart*

Water use by saltcedar, an invasive phreatophyte, is of significant concern in many riparian zones in the western United States. Diurnal groundwater fluctuations were analyzed to estimate evapotranspiration and water salvage (water available for other ecological functions) in saltcedar stands over a 6-yr period on a site along the Pecos River in Texas. Seasonal stand-level saltcedar water loss at an untreated control site ranged from 0.42 to 1.18 m/yr. Seasonal water salvage following application of imazapyr ranged from 31% 4 yr after treatment to 82% 2 yr after treatment. Significant water savings may be achieved by chemical saltcedar control, dependent upon water use by replacement vegetation and saltcedar regrowth. A regrowth management strategy is essential to maintain long-term water salvage.

Nomenclature: Imazapyr; saltcedar, *Tamarix* spp.

Key words: Saltcedar water use, water salvage, riparian evapotranspiration.

Saltcedar (*Tamarix* spp.), a phreatophytic tree native to Eurasia, was introduced into the United States by nurserymen in the 1820s (DiTomaso 1998). Subsequent to being planted as an ornamental and for erosion control, saltcedar escaped cultivation in the 1870s and was recognized as an environmental concern in the 1920s (Robinson 1965). Since the 1920s, saltcedar infestations have increased at a rate of approximately 3 to 4% per year in the southwestern United States, and the plant now dominates more than an estimated 600,000 ha (1,500,000 ac) (Robinson 1965). The plant thrives in arid southwestern climates and is commonly found in riparian areas in Utah, Nevada, Arizona, California, New Mexico, Colorado, Oklahoma, and Texas (DiTomaso 1998).

Saltcedar establishment in riparian areas causes several significant environmental concerns. Saltcedar has a marked advantage over native woody species because of its ability to produce seeds almost continually. These have the capability of germinating in conditions unfavorable for most natives (DiTomaso 1998; Sala et al.1996). This contributes to the creation of saltcedar monocultures in areas once inhabited by cottonwoods (*Populus* spp.), willows (*Salix* spp.), and other riparian species. In addition, mature saltcedar recovers quickly following fires and can tolerate extreme drought and flooding conditions (DiTomaso 1998; Hart et al. 2005; Robinson 1965). Soil surfaces beneath the saltcedar canopy may exhibit increased salinity due to the plant's ability to use more saline water than other species. Excess salt is excreted from leaves and drops onto the soil surface beneath the canopy, leading to drastically reduced plant diversity in saltcedar stands (Hart et al. 2005; Robinson 1965; Shafroth et al. 2005). As native species such as willows and cottonwoods are replaced by saltcedar, wildlife preferring the native vegetation for cover and food are usually displaced (DiTomaso 1998).

Recent studies have pointed out that water use by individual saltcedar trees is comparable to that of native species it commonly replaces, such as cottonwoods and willows (Glenn and Nagler 2005; Nagler et al. 2003). However, when making comparisons at the stand level, saltcedar often has a substantially higher leaf area index than natives, allowing it to transpire considerably more water (DiTomaso 1998; Sala et al. 1996). Opinions also vary regarding the economic and ecological value of saltcedar removal (Shafroth et al. 2005; Stromberg et al. 2009). Nevertheless, water use by saltcedar remains a large concern for many land managers. Its deep roots give it access to water at depths of up to 10 m (33 ft), and saltcedar has been shown to negatively affect spring flow and surface water levels due to its high evapotranspiration

DOI: 10.1614/IPSM-09-009.1

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Interpretive Summary

This study concludes that up to 82% water salvage (water available for other ecological functions) was achieved in one growing season by chemical control of saltcedar along the Pecos River in Texas. The volume of water savings is ultimately dependent upon site-specific environmental conditions and, more importantly, replacement vegetation. Assuming net water salvage is the goal of saltcedar control, the Pecos River in Texas is amenable to this practice based on results from this study indicating significantly lower ET rates by native vegetation. This study also concludes that water salvage following chemical control of saltcedar will be short-lived if a strategy for regrowth maintenance is not implemented. It is recommended that sites be inspected and, if needed, saltcedar regrowth treated no later than the third year post-treatment if optimum long-term water salvage is the objective. Based on the agreement of results from this study with others in the literature, reasonably accurate estimates of ET can be calculated by analysis of diurnal groundwater fluctuations coupled with accurate soil specific yield values. Adverse site conditions affected the methodology used in this study and produced conservative estimates of saltcedar water loss and salvage from chemical control. Site conditions in 2001 and 2006 were the most representative of "normal" environmental conditions for the study area and, had conditions been as such throughout the study, saltcedar water loss and salvage from chemical control estimates may have been significantly higher. For future studies using well data to calculate ET, it is recommended that pretreatment baseline data be established for a minimum of 3 yr. The differences in environmental conditions between years in this study made it more difficult to make comparisons between the baseline and post-treatment data.

(ET) potential (DiTomaso 1998; Hart et al. 2005; Robinson 1965; van Hylckama 1974).

Analysis of diurnal fluctuations in wells screened in shallow aquifers has been used as a method to determine groundwater consumption by phreatophytes (Butler et al. 2007; Gatewood et al. 1950; Gerla 1992; Hays 2003; Loheide et al. 2005; Rosenberry and Winter 1997; Schilling 2007; Shafroth et al. 2005; Troxell 1936; White 1932; Zhang and Schilling 2006). As plants transpire during the day, the water table lowers 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 (the White method) for analyzing well hydrographs to estimate plant water use. Soil specific yield is the most critical element of the White method, and care must be taken to ensure the appropriate values are used in the equation (Loheide et al. 2005).

The Pecos River Ecosystem Project (PREP) was begun in 1999 to address the issue of saltcedar infestation along the banks of the Pecos River and its tributaries in Texas. Local, state, and federal entities collaborated to chemically treat a total of 5,462 ha of saltcedar between 1999 and 2005 in an effort to achieve more efficient irrigation deliveries and water conservation within the Red Bluff Water and Power Control District (Hart 2005). Clayton (2002) found that substantial amounts of water released from Red Bluff Reservoir are lost between the release and delivery points, ranging from 39 to 67% on a monthly basis.

This study was initiated to examine the effects of saltcedar control on water loss along the Pecos River in Texas. Specific objectives were to (1) estimate the amount of water lost seasonally through ET in saltcedar stands along the Pecos River near Mentone, TX; and (2) estimate potential stand-level water salvage achieved by chemically treating saltcedar. Water salvage within the context of this study refers to water no longer used by saltcedar that may be available for other ecological functions. It is not assumed that water savings estimated by this study were made available for human consumption.

Materials and Methods

Study Site. The study area was located in a stand of riparian saltcedar along approximately 3 km (1.9 mi) of the Pecos River near the town of Mentone in Loving County, TX (31.692°N, 103.622°W). The area is located in the Chihuahuan Desert and is characterized by an arid climate, with approximately 28 cm (11 in) of precipitation annually, most of which falls in the summer during short thunderstorms (NCDC 2004). Soils were predominantly fine sand, sandy loam, and clay loam with a clay layer present at varying depths in the soil profile. Riparian vegetation in the area was dominated by dense saltcedar growth, with fourwing saltbush [*Atriplex canescens* (Pursh) Nutt.] and honey mesquite (*Prosopis glandulosa* Torr.) commonly occurring in the upper floodplain.

The study site was located approximately 48 km to the southeast of Red Bluff Reservoir, which is situated just below the Texas–New Mexico state line. Red Bluff Reservoir stores water for scheduled releases to irrigation districts within the Red Bluff Water and Power Control District. The streamflow in this section of the Pecos River is highly regulated by releases from Red Bluff, and the river bed has been known to go dry if sufficient water releases are not maintained.

Site Design. Overall project design for this study site was developed by Hays (2003) and in place when this study began. The study site was set up following the EPA (1993) paired plot–study design to allow comparison between two sites (A and B) under different saltcedar treatment scenarios. Eight groundwater monitoring wells, four at each site, were installed along the river before the growing season in 2001. The wells were hand drilled and cased with 5.08-cm-diam polyvinyl chloride (PVC) pipe with a 1.22-m-long well screen attached to the bottom of the casing, and approximately 0.9 m of casing extended above the soil surface. Wells 1, 2, and 3 at both sites were placed in a triangular formation on the bank immediately adjacent to the river, with wells 1 and 3 located inside the riparian saltcedar stand and well 2 situated at the edge of the stand. Well 5 at both sites, located outside the saltcedar stand in the floodplain, was used to compare water loss of floodplain vegetation with that of saltcedar.

Annular space around the wells of up to 0.3 m below the soil surface was filled with frac sand to prevent the 0.01-m slots in the well screens from clogging, and capped with concrete at the soil surface. A well (A4 and B4) was also installed in the river at each site to monitor water levels in the stream channel. Surface elevations of the wells relative to each other were determined by using a survey transit and range pole, and depth of the wells was measured by lowering a weighted tape measure to the bottom of each well. Each well was cleaned once per year by scrubbing the well screen with a long-handled brush to remove roots, and flushed with pressurized water to clean any silting that could potentially clog the well screen.

Water Loss Calculation. Each groundwater monitoring well was equipped with a pressure transducer data logger (Model WL15X).[1] The battery powered data loggers measured and recorded hourly groundwater levels with an accuracy of 0.2% or ± 0.0091 m. Loggers were installed prior to the growing season in 2001 to 2006 and remained in the wells throughout the season each year. After the growing season in 2001, site A was aerially treated with imazapyr (Arsenal™)[2] herbicide, and the saltcedar growing at site B was left untreated.

Hays (2003) developed a modified White (1932) equation (Equation 1) to measure water loss at this site, and that equation was used in this study.

$$Q = \{(H_1 - L_1) + [(H_2 - L_1/T_1) \times T_2]\} \times (sy) \qquad [1]$$

The equation uses the first high groundwater water level of the day (H_1), the lowest groundwater level (L_1), the first high groundwater level for the following day (H_2), the number of hours between the second high and first low (T_1), the number of hours between the first high and first low (T_2), and soil specific yield (*sy*) for the 0.3-m increment of the soil profile corresponding to groundwater level.

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. Data were eliminated from any calculations during time periods when water was aboveground due to on-site flooding. Instrument "noise," or variability, was reduced by using a three-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

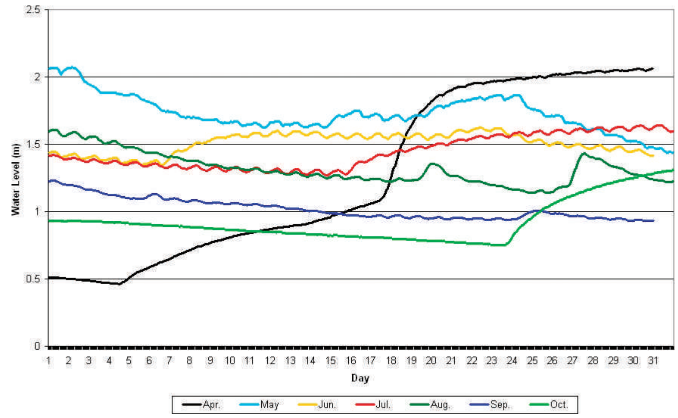

Figure 1. Growing season hourly water levels for well A1 in 2001 (pretreatment).

0.03 m, based on analysis of groundwater fluctuations compared with upstream reservoir releases. Any calculation in which the water table fluctuation exceeded the S value was eliminated, thus reducing the impact of streamflow change from reservoir releases and recharge resulting from significant rainfall events. Calculations with drawdown or recharge times of less than 4 hr or that were negative or equal to zero were eliminated. Remaining daily water loss calculations for wells A1 to A3 (treated) and B1 to B3 (untreated), located within the saltcedar stands, were pooled to get an average daily water loss for each site. Monthly water loss was estimated by multiplying the average daily water loss by the number of days in the month. Average daily water loss was multiplied by the number of days in the growing season of April 1 through September 30 (183 d) to arrive at seasonal water loss estimates. Water loss at wells A5 and B5, located outside the saltcedar zone, was calculated in the same manner.

This method assumes that there was no saltcedar ET at night and that no measurable water losses occurred prior to April or after September. These assumptions were detailed by White (1932) and are discussed in other studies (Cleverly et al. 2002; Gatewood et al. 1950; Gay and Hartman 1982; Loheide et al. 2005). Groundwater hydrographs developed in this study showed that diurnal groundwater fluctuations were generally not detectable until the latter part of April and became undetectable again in October at this particular site (Figure 1).

Specific Yield. The White (1932) method assumes that soil specific yield values used to calculate water loss due to ET can be accurately determined. Soil samples for each 0.3-m increment in the soil profile were collected at each well during the initial drilling and analyzed for texture by Hays (2003). This study used the original samples in analyses to calculate specific yield by the sample saturation and drainage method (Johnson 1967). A 50-cm^3 sample of each 0.3-m increment in the soil profile of each well was

Table 1. Coefficient of determination (R^2) for 2001 sites A and B hourly water level fluctuations (adapted from Hays 2003).

Well	A1	A2	A3	A5	B1	B2	B3	B5
	R^2							
A1	1	0.96	0.98	0.11	0.95	0.96	0.86	0.96
A2		1	0.86	0.04	0.88	0.93	0.85	0.94
A3			1	0.07	0.97	0.93	0.98	0.88
A5				1	0.12	0.10	0.04	0.06
B1					1	0.98	0.88	0.97
B2						1	0.90	0.98
B3							1	0.92
B5								1

oven dried in a sample can for 24 hr. The sample was then saturated, covered to prevent evaporation, and allowed to drain through a sieve into a holding pan for 24 hr. Specific yield was then 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. This process was repeated three times for each 0.3-m increment and averaged for specific yield calculations.

Water Salvage Calculation. A comparison of hourly water level fluctuations at sites A (treated) and B (untreated) in 2001 (pretreatment) showed a significant correlation between wells 1 to 3 within and between sites (Table 1). This relationship was used to estimate water salvage at site A (treated) in 2002 to 2006 after chemically treating saltcedar.

In order to estimate potential water salvage from saltcedar control, datum from adjacent sites A (treated) and B (untreated) was collected 1 yr prior to treatment (2001) for baseline data and to determine their relationship. After chemical treatment of site A at the end of 2001, the control (site B) was used in subsequent years (2002 to 2006) to predict what the water use calculation would have been under a no-treatment situation. This allowed for a simple comparison between actual and predicted water loss at site A (treated) to arrive at estimated salvage due to saltcedar control (Equation 2).

$$\text{Salvage} = \left[A_1 \times \left(B_y/B_1\right)\right] - \left(A_y\right) \qquad [2]$$

The equation uses the site A pretreatment (2001) seasonal water loss (A_1), untreated site B seasonal water loss for the year in question (B_y), untreated site B seasonal water loss in 2001 (B_1), and site A post-treatment seasonal water loss for the year in question (A_y).

Statistical Analysis. Statistical analyses were performed on water loss calculations for both sites A and B in all years to identify outliers and determine if treatment effects were significant. Daily water loss calculations for each well were analyzed for outliers on a yearly basis using box-plots (Frigge et al. 1989). Values that fell outside 1.5 times the interquartile range were examined in relation to surrounding days and daily water loss estimates found in the literature and were excluded from final seasonal water loss calculations if they were deemed unreasonable. A Levene's test for homogeneity of variances was conducted on average daily water loss calculations on each well by season (Zar 1999). Upon discovering heteroscedasticity, the Welch and Brown-Forsythe corrections were applied after conducting single-factor ANOVA on the average daily water loss calculations at each site to determine if there were significant differences between years of data and treatments (Zar 1999). A Games-Howell–pair-wise post hoc test was then conducted to determine homogeneous subsets of seasonal water loss calculations (Day and Quinn 1989). Standard error of the mean daily water loss for both sites was calculated on a yearly basis for use in displaying variability in the results (Freund 1984). The standard error values for site A (treated) were utilized to display variability in water salvage estimates for each year.

Results and Discussion

Specific Yield. Our results produced a wide range of specific yield values to be used in water loss calculations. Specific yield at the treated site (A) ranged from 3% for clay to 21% for sand. Fine sand was the predominant soil texture at the treated site (A) with calculated specific yield values ranging from 16 to 20%. The untreated site (B) had an overall specific yield range of 3% for silty clay to 22% for sand. The predominant soil texture at the untreated site (B) was fine sandy loam with a calculated specific yield range of 5 to 13%. Johnson (1967) reviewed specific yield values in the literature at that time and compiled a list of average values ranging from 2.5% for sandy clay to 34% for sand. Loheide et al. (2005) offered a similar list with values ranging from 1.5% for sandy clay to 32% for sand.

Water Loss. *Inside the Saltcedar Zone.* Seasonal water loss calculations for wells located within the saltcedar zone at sites A (treated) and B (untreated) are shown in Figure 2. There was no significant difference ($P < 0.05$) in stand-level calculated water loss in 2001 (pretreatment) between site A and B, and as a result, data from both sites were pooled to arrive at a pretreatment baseline seasonal water loss calculation of 1.18 m.

Hart et al. (2005) reported water loss figures at this site for 2001 to 2003 with notably different results than those reported in this study. Hart et al. (2005) reported an average water loss of 2.34 m in 2001 pretreatment. Water loss for 2002 and 2003 was reported as 1.94 and 2.03 m at site B (untreated), and 0.17 and 0.04 m at site A (treated), respectively (Hart et al. 2005). This study calculated 2002

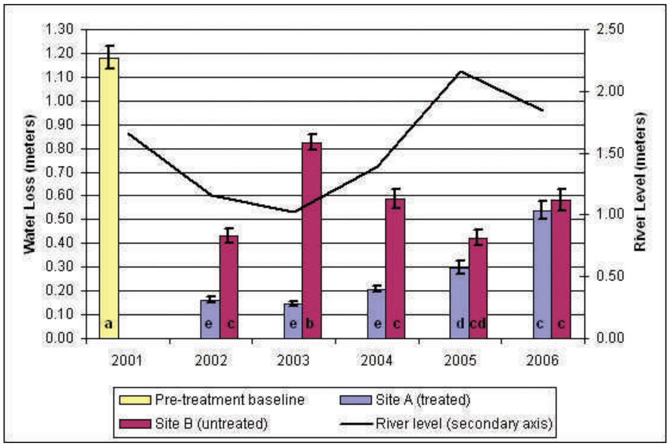

Figure 2. Seasonal water loss inside the saltcedar zone at sites A and B 2001 to 2006. Error bars indicate ± 1 standard error. Letters within bars indicate significant difference ($P < 0.05$).

and 2003 water loss as 0.43 and 0.86 m at site B (untreated), and 0.16 and 0.15 m at site A (treated), respectively. Discrepancy between studies is attributed to differences in specific yield values used in the equation to calculate water loss. Hart et al. (2005) utilized specific yield values for this site given in Hays (2003), whereas this study used specific yield values derived from laboratory analyses.

Gatewood et al. (1950) and Inglis et al. (1996) also used the White (1932) method to calculate saltcedar stand-level ET, and reported a notably wider range of daily water use than this study. Gatewood et al. (1950) reported a range of 1.0 to 14.4 mm/d water use by saltcedar in the Lower Safford Valley in Arizona. Inglis et al. (1996) estimated a saltcedar ET range of 3.9 to 10.4 mm/d in the Lake Mead National Recreation Area of Nevada. The daily average water loss calculated in this study ranged from 2.3 to 6.5 mm/day.

Saltcedar initial mortality rate at site A in 2002 following chemical control was estimated at 90%, leaving virtually no apparent living vegetation present in the riparian zone and producing barely detectable diurnal groundwater fluctuations. Post-treatment water loss at site A dropped to 0.16 m in 2002, or 14% of the baseline, and site B (untreated) also dropped to 0.43 m, or 37% of 2001. Site A (treated) water loss remained significantly ($P < 0.05$) lower than site B (untreated) through 2005. In 2006, differences in water loss were no longer significant and although water loss never again equaled that of the baseline data, the relationship between sites was similar.

The data raises a number of unexpected questions. Water loss at the untreated site B dropped significantly in 2002, although only the treated site A would be expected to do so. Severe drought in 2002 and 2003 resulted in no water releases from upstream Red Bluff Reservoir in those years, and consequently, some of the monitoring wells periodically went dry. The water table dropped approximately 0.9 m from 2001 to 2002, and was likely below the saltcedar root zone at site B (untreated) in 2002, drastically reducing ET. Decreased saltcedar water use related to a declining water table has been noted in other studies (Butler et al. 2007; Cooper et al. 2006; Devitt et al. 1997). 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 (untreated), even though the water table did not recover. In contrast, water loss at site B (untreated) 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 aboveground for extended periods of time at much of the site. Readily available soil moisture in 2004 and 2005 potentially reduced diurnal groundwater fluctuations and the ability to detect saltcedar water loss at the rates seen in 2003 and 2001. Gatewood et al. (1950 p. 143) encountered a similar scenario, noting that "the hygrograph from a well for a day when the soil moisture is plentiful will show a low use of water." Environmental conditions in 2006 were the most similar to 2001, and site B (untreated) water loss increased to 0.58 m, but never recovered to the pretreatment baseline of 1.18 m. Riparian saltcedar was treated both up- and downriver of site B (untreated), leaving it to be the only living stand of trees in the area. Treatment of the saltcedar directly upriver may have had a direct effect on site B (untreated) resulting in underestimated water loss.

As expected, water loss at the treated site A decreased dramatically following saltcedar control and remained very low through 2004. Pretreatment vegetation transects completed by Hays (2003) show that in 2001 (pretreatment) site A had approximately 1,280 saltcedar plants/ha providing 80% canopy cover, and that only 8% herbaceous understory was present. As previously stated, saltcedar initial mortality rated 1 yr after treatment (2002) was estimated at 90%, and little to no green vegetation was evident. Vegetation began returning to site A (treated) in the form of grasses and forbs the second year (2003) after saltcedar treatment. Weeks et al. (1987) used the eddy correlation method to measure daily water use by replacement vegetation after clearing saltcedar on the Pecos River in New Mexico and found that grasses and forbs were using 0.5 to 1.4 mm/d. Comparably, daily water loss at site A (treated) was calculated at 0.9 to 1.1 mm/d post-treatment in 2002 to 2004. Water loss in the year after treatment (2002) is attributable to evaporation from the bare soil, as there was minimal green vegetation on the site at that time. Water loss at site A (treated) did not increase significantly in 2003 and 2004 even though grasses and forbs had returned to the site, whereas water loss did increase significantly at site B (untreated) in 2003 due to the ability of saltcedar to tap the deeper water resources. Notable saltcedar regrowth began to take place in 2005 and increasingly so in 2006, and is likely the cause of corresponding increases in water loss at site A (treated) in those years. Transects completed at site A (treated) in 2006 revealed that saltcedar canopy had regrown to about 25% of pretreatment estimates, and herbaceous understory vegetation had also increased from 8 (pretreatment) to 25% cover. In addition, approximately 50% of saltcedar at site A (treated) showed foliage regrowth and were counted as live trees, reducing the effectiveness of the treatment on saltcedar mortality.

The range of water loss attributable to saltcedar calculated at site B (untreated) throughout the course of this study agrees quite well with other estimates found throughout the literature. The minimum seasonal ET estimate found in the literature was 0.43 m/yr, and the maximum was 3.0 m/yr. This study calculated an average seasonal minimum of 0.42 m/yr and a maximum of 1.18 m/yr in years 5 and 1, respectively, and was highly dependent upon site specific variability. The highest water loss estimate was calculated during optimum conditions for the methodology employed by this study, leading to the conclusion that the lowest estimates may underestimate the true level of water loss in those years. Recent sap flow studies (Moore et al. 2008) have also shown that saltcedar does transpire at night, although at a much lower rate, which is not considered in the White (1932) method used in this study. Taking these factors into consideration, it is believed that the water loss figures calculated in this study are conservative.

Outside the Saltcedar Zone. Two wells (A5 and B5) were located outside the saltcedar zone at both the treated (A) and untreated (B) sites, respectively. Vegetation surrounding these wells, predominantly fourwing saltbush and honey mesquite, received no chemical treatment at either site throughout the course of the study. Seasonal water loss outside the saltcedar zone is compared with water loss within the saltcedar zone for both sites in Figure 3.

Water loss at well B5 was significantly ($P < 0.05$) lower than the average water loss of wells located in the saltcedar zone at site B (untreated) in all years. Water loss at well B5 displayed much lower water loss in 2002 and 2003 in response to drought conditions and no releases from upstream Red Bluff Reservoir. Of particular interest was the continued decrease in water loss at well B5 from 2002 to 2003, as opposed to the sharp increase in water loss at wells B1 to B3 (saltcedar zone) in 2003. This is indicative of the ability of saltcedar to access deeper groundwater than the native species located in the floodplain. In contrast to the 2002 to 2003 scenario, water loss at well B5 increased in 2004 and 2005, while water loss at wells B1 to B3 (saltcedar zone) decreased. Periodic site flooding and water aboveground at wells B1 to B3 (saltcedar zone) in those years affected the ability to detect diurnal groundwater fluctuations. More readily available water in the floodplain,

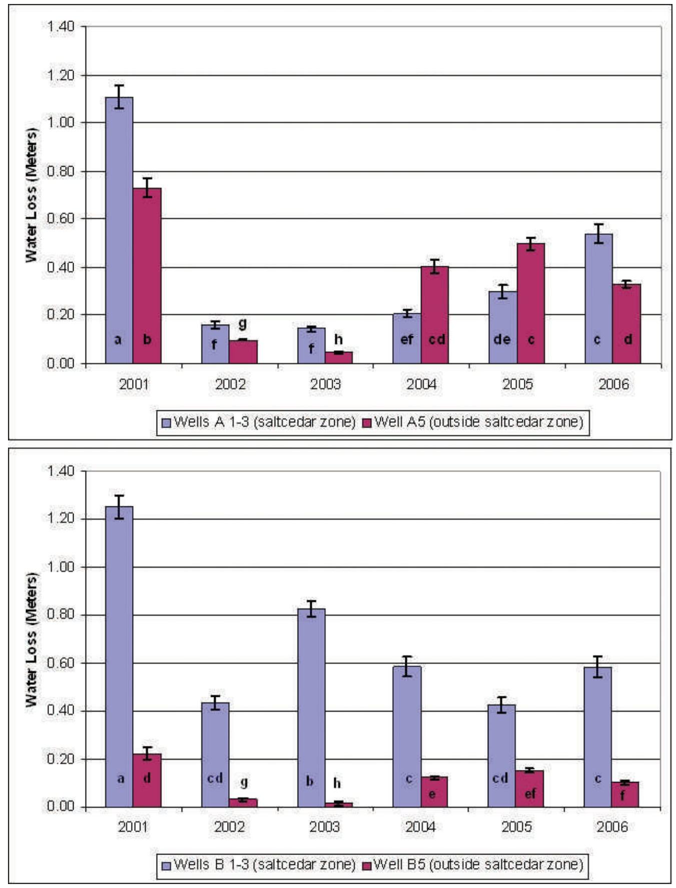

Figure 3. Comparison of seasonal water loss for wells 1 to 3 (inside the saltcedar zone) and well 5 (outside the saltcedar zone) at the treated (A) and untreated (B) sites. Error bars indicate ± 1 standard error. Letters within bars indicate significant differences ($P < 0.05$) across all years within each site.

and the ability to detect increased diurnal fluctuations contributed to the increased water loss at well B5. This indicates that water loss was probably underestimated in the saltcedar zone at site B (untreated) in 2004 and 2005.

Water loss at well A5 was significantly lower ($P < 0.05$) than the average water loss of wells A1 to A3 (saltcedar zone) in 2001 to 2003 and displayed the same downward trend as that of well B5 water loss. The relationship between well A5 and wells A1 to A3 (saltcedar zone) changed dramatically in 2004 and 2005 with well A5 showing significantly higher water loss than those located in the saltcedar zone for those years. Water availability on-site from increased precipitation and upstream reservoir releases possibly made groundwater more accessible to the floodplain vegetation surrounding well A5 in the absence of riparian saltcedar water use. In 2006, wells A5 and A1 to A3 (saltcedar zone) returned to their original relationship with wells within the saltcedar zone showing significantly higher water loss than well A5. This may be attributed to increased water loss at wells A1 to A3 (saltcedar zone) due

Table 2. Seasonal water salvage for site A (treated) 2002 to 2006.

	Water salvage average	Water salvage average	Standard error	Water salvage minimum	Water salvage maximum
	%	m			
2002	63	0.27	0.013	0.26	0.28
2003	82	0.68	0.012	0.67	0.69
2004	65	0.38	0.015	0.37	0.40
2005	31	0.13	0.028	0.10	0.16
2006	7	0.04	0.037	0.00	0.08

to saltcedar regrowth. Caution must be used when making direct comparisons between wells A1 to A3 (saltcedar zone) and well A5 due to very low correlation between hourly diurnal water fluctuations (Table 1) and very low correlation between well A5 and river level.

Water Salvage. The original strategy (Hays 2003) for calculating water salvage from saltcedar control was to use the site B (untreated) water loss calculations to predict what water loss would have been at site A in 2002 to 2006 had it been left untreated. Since no significant difference ($P = 0.94$) between sites existed in 2001 (pretreatment), the data were pooled to arrive at an average pretreatment baseline water loss of 1.18 m. Given the high degree of correlation between sites presented in Table 1 and the pooled pretreatment data, water loss at site A (treated) was simply subtracted from water loss at site B (untreated) on a yearly basis in 2002 to 2006 to calculate an estimated seasonal water salvage. Seasonal water salvage results are presented in Table 2.

Water salvage immediately following chemical saltcedar control in 2002 was calculated to be 0.27 m and was surprisingly low compared to the 2003 and 2004 figures. As stated previously in the water loss discussion, the area experienced drought conditions in 2002, which led to a significant drop in the water table and less water available for saltcedar ET. Consequently, the potential water salvage was decreased significantly due to the substantially lower water loss calculated at site B (untreated) that year. The highest potential water salvage occurred in 2003, resulting directly from the increased water losses calculated at the untreated site B that year. Although site conditions were similar in 2002 and 2003, it appears that the saltcedar root system at site B (untreated) was able to access and begin "pumping" groundwater again, registering higher diurnal fluctuations and water loss in 2003. In the absence of competition from saltcedar, grasses and forbs began returning to site A (treated) in 2003 and were present throughout the growing season in all following years of the study. It is apparent that these plants were incapable of tapping the shallow aquifer in drought years when saltcedar was able to do so. Water salvage remained significant in 2004 and 2005 while decreasing in response to lower water losses at site B (untreated) and higher ET losses at site A (treated) due to saltcedar regrowth. There was no significant difference ($P = 1.00$) in water loss calculations for site A (treated) and B (untreated) in 2006; therefore, the water salvage figure for 2006 presented herein is not significant.

Water salvage resulting from saltcedar control or removal is highly dependent on replacement vegetation and other site-specific conditions. Weeks et al. (1987) predicted water salvage on the Pecos River in New Mexico to be 0.20 to 0.40 m/yr with replacement vegetation consisting of grasses, forbs, and halberdleaf orach (*Atriplex patula* L.). Culler et al. (1982) calculated a water salvage range of 0.36 to 0.66 m/yr on bare ground following saltcedar removal on the Gila River in Arizona. Hays (2003) estimated 0.40 m/yr annual water salvage from chemically controlling saltcedar on the Colorado River in Texas, but did not report the composition of replacement vegetation. Nagler et al. (2008) estimated potential water salvage from saltcedar control to range from −0.2 to 0.6 m/yr, based on site revegetation by cottonwood trees and saltgrass [*Distichlis spicata* (L.) Green], respectively. This study calculated water salvage resulting from chemical control of saltcedar to be 0.13 to 0.68 m/yr, based on natural revegetation of the study site with grasses, forbs, and saltcedar regrowth. This assumption is appropriate for the Pecos River in Texas, as grasses and forbs comprised the natural riparian vegetation along the river prior to saltcedar invasion (Wilcox et al. 2006). The lowest seasonal water salvage (0.13 m) occurred 4 yr after treatment presumably due to saltcedar regrowth at the treated site. Although regrowth at the treated site 4 and 5 yr after treatment did not equal the density of saltcedar growth prior to chemical treatment, the "oasis effect" possibly increased the ET potential of individual trees. Ansley et al. (1998) studied transpiration by honey mesquite trees in high density versus thinned stands and found that water use by individual trees was significantly higher in the thinned stands. It was concluded that within a limited pool of water, intraspecific competition forced mesquite in high density stands to use water more conservatively (Ansley et al. 1998).

Significant localized, temporary water savings were achieved at this study site by chemically controlling saltcedar. Beneficial effects on the shallow aquifer adjacent to the Pecos River were observed for approximately 4 yr, after which the effects appeared to be negligible. A follow-up management strategy to control saltcedar regrowth will need to be implemented if long-term water savings are to be achieved by chemical control. Long-term water savings are not likely to be realized in areas where aggressive and sustained saltcedar regrowth monitoring and treatment are not physically or economically feasible. Others researching saltcedar water use have also expressed the need for a post-treatment management program to achieve sustained water salvage (Glenn and Nagler 2005; Sala et al. 1996). The natural revegetation that occurred on this study site exhibited lower ET potential than the saltcedar it replaced, resulting in net water salvage in the years after treatment and before significant saltcedar regrowth. This study may not be applicable to, and long-term net water salvage may not occur on, other river systems where cottonwoods, willows, or other phreatophytes are likely to be the replacement vegetation following control or removal of saltcedar.

Sources of Materials

[1] Pressure transducer data logger, Model WL15X, Global Water Instrumentation, Inc., 11390 Amalgam Way, Gold River, CA 95670.

[2] Arsenal™ herbicide, BASF Corporation, Agricultural Products, 26 Davis Drive, Research Triangle Park, NC 27709.

Acknowledgments

Our gratitude is extended to Carol Thompson and Roger Wittie, Tarleton State University faculty, for assistance with data analysis. Work supported by the Rio Grande Basin Initiative administered by the Texas Water Resources Institute, Texas AgriLife Research and Extension with funds provided through a grant from Cooperative State Research, Education, and Extension Service, U.S. Department of Agriculture, under Agreement No. 2008-45049-04328. Funding was also provided by the Texas State Soil and Water Conservation Board through the Environmental Protection Agency Clean Water Act Section 319(h) program.

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Received February 19, 2009, and approved July 17, 2009.