Evaluation of the TSSWCB Brush Control Program: Monitoring Needs and Water Yield Enhancement

By

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Beginning in 1998, the estimated water yield potential for the Texas State Soil and Water Conservation Board's (TSSWCB) brush control program was initially set by a series of feasibility studies, that included hydrologic computer simulations of the watershed behaviors before and after brush removal. Removal of brush altered estimated model parameters that would increase surface water runoff and reduce loss of groundwater to the brush-root systems. Large river watersheds were subdivided into smaller subwatersheds in attempts to capture their differences in hydrologic behavior. All simulations also included the assumptions that [1] all brush would be effectively treated or removed at all positions in the subwatersheds, [2] all landowners in the target subwatersheds would willingly participate, and [3] sufficient funds would be available to provide cost share to all landowners. Since the completion of the feasibility studies, the TSSWCB staff has diligently pursued landowners in the selected watersheds, often in cooperation with the Natural Resources Conservation Service (NRCS) Environmental Quality Incentives Program (EQIP). It should be noted that the NRCS EQIP program has identified invasive brush species as a highly ranked concern in 200 of the 254 counties in Texas, and combined recruitment of landowners and cost sharing has greatly influenced TSSWCB funding applications. The three assumptions of the feasibility studies, however, have not yet occurred due to funding limitations and as yet unwilling landowners who reject brush control for privacy or perceived hunting lease issues. In the ten years since the feasibility process began, twelve primary sites have been included in the TSSWCB brush control program, as shown in Table ES.1. The research team visited each of the sites shown in Table ES.1 by helicopter flyover and/or by on-the-ground visits by truck and foot. It should be noted that many of these rural watersheds are difficult to drive through due to roadway limitations and landowner security. Helicopter flyovers were relatively unlimited, except for the Nueces River site that included restricted military air space.

Table ES.1. Sites, Dominant Brush Types (Mesquite, Juniper, or Saltcedar), and Visit Types (Ground, Flyover)

No.	Site	Brush	Visit
1	North Concho River (Grape, Chalk Creeks)	M, J	G,F
2	Twin Buttes Reservoir (Pecan, Spring, Dove Creeks)	M, J	G,F
3	Pedernales River	J, M	F
4	Lake Ballinger	M	G,F
5	Oak Creek Reservoir	J, M	G,F
6	Champion Creek Reservoir	M, J	G,F
7	Nueces River	M	F
8	Hubbard Creek Reservoir	S	F
9	Pecos River	S	F
10	Upper Colorado River	S	G,F
11	Canadian River	S	G,F
12	Wichita River (Lakes Kickapoo and Arrowhead)	M	G,F

Of the sites shown in Table ES.1, all but the Hubbard Creek Reservoir and Wichita River watersheds had already received some brush treatment or removal with TSSWCB funds. Treatment contracts are ready for selected locations in the Hubbard Creek Reservoir area, but high water in the reservoir has delayed treatment, and initial contracts in the Wichita River watershed are currently being pursued. To date, relatively little funding has been available for monitoring, but cooperative efforts between the TSSWCB and other agencies have allowed some data collection work in the North Concho, Canadian River, and Pecos River. In general, there was not time for the TSSWCB to collect and analyze pretreatment data on local scales to see the streamflow vs. precipitation variations, and post-treatment data are not being continuously collected and analyzed to determine runoff or groundwater enhancement. TSSWCB staff tried to include some watershed issues, such as plant density, land use pattern, and proximity to a channel, in selection for treatment contracts, but they have been limited by the number and distribution of willing landowners and the funding appropriations. Without a defined set of scientific site selection criteria, some sites that have been treated may not yield much water to the target river or reservoir. Under these conditions, it is not possible to quantitatively evaluate the effectiveness of brush treatment at the current sites in terms of new water yield.

Based on this finding, our research team demonstrated the application of scientific site selection criteria for choosing locations to pursue for brush control to increase water yield to surface runoff and groundwater. The criteria were provided in September, 2007, to the TSSWCB for their consideration and are included in the appendix. The criteria list is not unique, as other experts in watershed hydrology, hydrogeology, and brush control would likely come up with similar lists. The main direction is to move beyond debating how much water each plant removal might yield to the bigger picture of the watershed's behavior. In general, the selection of a site for water yield enhancement through brush control should include the following considerations:

- Characteristics of the watershed – soils, slope, land use, vegetation and brush distributions, and proximity of the brush to the stream channel or water supply point;
- Local climatic conditions – precipitation amounts, storm intensities, potential evapotranspiration (ET); and
- Interaction of surface water and groundwater – transmission losses or gains in the stream channel, contribution of alluvial aquifers to stream baseflow.

Due to the short time allowed for this study, there was not sufficient time to completely analyze the large watersheds in detail based on our criteria. We did provide example discussions of each of the sites in terms of soil and slope conditions. Much more time and budget would be required to completely map and analyze the brush distributions, groundwater conditions, land uses, and locations of willing landowners within the large watersheds or smaller subwatersheds.

Identification of proper monitoring approaches and upgrades

The hydrologic processes of runoff generation from variable storm events, streamflow gains or losses due to groundwater interactions, and water losses due to ET by nearby vegetation are complex and variable over time, making them difficult to represent accurately with mathematical models. Observation of these processes for sufficiently long periods of time is necessary to see the ranges of streamflows that are caused by dry and wet weather conditions. The best situation would be a pre- and post-treatment paired watershed comparison. Two nearby watersheds with relatively similar sizes, land use distributions, soil and slope variations, groundwater conditions, and brush distributions would be instrumented for pretreatment monitoring with multiple continuous rain and streamflow gauges, as well as multiple groundwater monitoring wells, to allow several years of data collection that establish the range of pretreatment behaviors. One watershed would then receive brush treatment, and several post treatment years of data collection would continue. Qualified hydrologists and engineers would analyze the data to verify the impacts of brush treatment. The second best situation would be to set up a similar pre- and post-treatment monitoring program for a single watershed. The third best situation would be a paired watershed comparison with post-treatment data only. It is recognized that it may never be possible to use one of these approaches at every location that can benefit from brush control, but it should be possible to select a small number of sites with different dominant brush types, geographic locations, soil and slope conditions, and hydrologic characteristics for useful study. Our research team believes that inclusion of funding for pre-treatment and post-treatment monitoring activities along with inclusion of additional technical expertise to design and evaluate monitoring programs will significantly enhance the effectiveness of future brush control programs.

To date, relatively little funding has been spent on monitoring efforts at the brush control treatment sites. In the North Concho watershed, the UCRA and TIAER have three significant programs underway: [1] the paired watershed comparison of the East (treated) and West (untreated) forks of Grape Creek, [2] a paired watershed, one treated and one untreated, comparison of two juniper-infested subwatersheds of Chalk Creek, and [3] a paired site comparison of ET observations from two mesquite-infested plots, one treated and one untreated. While the first two of these programs are steps in the right direction, they do not include the proper complete combinations of continuously operational instrumentation to properly represent the hydrologic behavior of the watersheds.

The basic requirements of an appropriate monitoring system to allow observation of water yield changes in a watershed are relatively straightforward applications of fundamental hydrologic principles. The first requirement is proper delineation of the watershed of interest, which means that a streamflow observation point is identified and the upstream area that can contribute flow to that point is established from the local topography. Next, the areal variability of rainfall events must be compared to the watershed area so that the number rain gauges and their spatial distribution can be selected. Multiple rain gauges with continuously recording dataloggers are necessary to allow accurate estimation of rainfall input to the watersheds. Positions for streamflow measurements and their related configurations must be selected carefully. Continuously recording stream gauges are necessary to capture the short-term flow changes caused by intense rainfall events, with each installation planned to allow observation of high and low flow rates. Multiple sequential stream gauges are necessary to evaluate changes in streamflow between storm runoff events, which indicate whether the stream is gaining or losing flow to the local groundwater. The direction of local groundwater flow, toward or away from the streambed, can be determined with a network of monitoring wells near the stream at appropriately selected locations. The water table level can also be continuously monitored with pressure transducers and dataloggers. Remote weather stations can also be installed to measure local temperature, relative humidity, wind speed, and solar radiation variables to allow calculation of ET.

Estimation of water yield enhancement in areas with salt cedar, mesquite, and juniper

After review of the limited monitoring information from the current TSSWCB sites, it was apparent that data collected to date were not sufficient to allow estimates of water yield enhancement under any of the three dominant brush types. Absence of pretreatment monitoring, incomplete or absent post-treatment monitoring data, and limited areal coverage of brush treatment in parts of the watersheds with best potential for water yield prevent quantification of enhancement effects at any of the sites and extrapolation for predictions at other locations. Under these limitations, the best use of project time and funding was to review the available scientific literature about the water use by the three major brush types.

While often-cited, misinformation has been used to claim that a saltcedar tree can use 200 gal/d, the range of values in peer-reviewed publications has been 0.1 to 15 gal/d for a single tree. Variations depend on available water, plant stand density, and geographic locations. The more detailed studies report saltcedar ET in rates as values of 30 to 60 in/yr, and the conversion to volume of water lost per year requires estimation of the leaf area for the saltcedar stand. After removal or treatment of the saltcedar, the overall reduction in ET is dependent on the replacement vegetation.

Studies of water use by mesquite have reported ET values by mesquite trees of 3 to 44 gal/d, depending on stand density and location. After partial mesquite removal or treatment, it has been noted that the ET rates for the remaining live plants can greatly increase, due to the plant's combination of deep and lateral root systems. After allowing multiple years after treatment for grass to replace the mesquite, the ET for the grassed area can be as large as the ET prior to treatment, resulting in little net increase of runoff or aquifer recharge. Riparian removal/treatment is still worth evaluation based on the impacts of the other site selection criteria.

Juniper changes landscape water balances for a plant community by intercepting a significant proportion of precipitation with its dense canopy and litter. The interception loss associated with the canopies of juniper ranges from 25 to 37 percent of gross precipitation. As an evergreen, it has higher interception potential throughout the year when compared to saltcedar or mesquite. Rainfall that passes through the canopy must also pass through the litter layer prior to entering the soil, and interception losses of 40 percent of precipitation have been noted. These losses leave only 20 to 30 percent of the gross rainfall to reach the ground surface for runoff or infiltration. The amount of water gained by juniper removal is also affected by the pretreatment stand density, with greater water yield potential for removal of denser juniper stands.

Based on the combination of the site selection criteria and the current state of knowledge for water use by saltcedar, juniper, and mesquite, the following conclusions can be drawn.

- Complete treatment or removal of dense stands of saltcedar, juniper, and mesquite in riparian areas near stream channels and lakes has good potential for increasing water yield.
- Based on its rainfall interception ability and its affinity for slopes, control of juniper not only increases rainfall transmission to the soil surface but also enhances runoff.
- As the distance between saltcedar, mesquite, or juniper and the target stream or river channel increases, the potential positive impact of treatment of that stand on water yield may decrease depending on the local soil, slope, land use, and groundwater conditions.
- While complete treatment or removal of mesquite can be effective in increasing water yield, removal or treatment of only selected parts of a mesquite stand may not be as effective, because the lateral root system from the remaining mesquite can spread and increase the water use by the remaining plants.

- Brush control for water enhancement is a complex issue, and all aspects of the site selection criteria and management strategy, not just plant type, must be considered.

Acknowledgments

The research team greatly appreciated the willing cooperation of the staff of the TSSWCB's brush control program in the San Angelo office, the pertinent local Soil and Water Conservation District offices, and the Temple office, as they provided access and guided tours of sites, as well as electronic GIS files and other pertinent available data. Other involved agencies that also provided information included in the Upper Colorado River Authority (UCRA), Canadian River Municipal Water Authority (CRMWA), Texas Institute for Applied Environmental Research (TIAER), and the United States Department of Agriculture-Natural Resources Conservation Service (NRCS). Related information from the Texas A&M AgriLife Research and Extension Service was also obtained. We are also grateful for the time of representatives of the Texas Commission for Environmental Quality (TCEQ), Texas Water Development Board (TWDB), NRCS, UCRA, Texas AgriLife Research and Extension Service, Texas Department of Agriculture (TDA), and the Texas Parks and Wildlife Department (TPWD) who participated in our interviews.

1. Evaluation of Existing Monitoring Programs

1.1 Introduction

The first task in Objective 1 was to collect available information for each site from the Texas State Soil and Water Conservation Board (TSSWCB) and organize this information at the Texas Tech University Water Resources Center (WRC). We received excellent support from the employees of the TSSWCB and its cooperating organizations at the sites. This information is stored as electronic media at the WRC.

The second task was to visit each site, view the current monitoring facilities, and interview local personnel. Initial project contacts were made by visiting with TSSWCB personnel in their San Angelo offices. Other general orientation visits were held with our team members on the Texas Tech University campus. Sites were evaluated by either on-site ground visits to observe the specific management areas, or by aerial site flyovers in a helicopter. Each TSSWCB area was visited by multiple members of our team along with employees of the TSSWCB. Table 1.1 provides a site-specific description of how each site was viewed.

Table 1.1. Sites, Dominant Brush Types (Mesquite, Juniper, or Saltcedar), and Visit Types (Ground, Flyover)

No.	Site	Brush	Visit
1	North Concho River (Grape, Chalk Creeks)	M, J	G,F
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6	Champion Creek Reservoir	M, J	G,F
7	Nueces River	M	F
8	Hubbard Creek Reservoir	S	F
9	Pecos River	S	F
10	Upper Colorado River	S	G,F
11	Canadian River	S	G,F
12	Wichita River (Lakes Kickapoo and Arrowhead)	M	G,F

1.2 Site Evaluations

The third task in Objective 1 was to evaluate the status of projects and include the justification for the target treatment areas. It would be beyond the scope of this project to detail all of the sites and discuss all the possible scenarios relating to brush control issues in the State of Texas. There are, however, critical non-scientific issues that strongly influence the brush control effort within the state. First, we recognize the political issue of the biennial funding allocation set for the program by the Texas Legislature. The TSSWCB's funds are limited by those allocations, which may not be enough to completely treat the brush for any single large watershed, and may be subject to external influences for the selections of which watersheds are treated. A second political issue is directly related to the primary purpose of this report – evaluating the efficacy of brush control in Texas. An individual participating landowner views the effectiveness of brush control based on its local impact to the value and productivity of the treated property, often beyond the water yield enhancement received locally. Urban voters and legislators view the effectiveness in terms of increased downstream flows or decreased reservoir losses, both potential increases in water supply. The time spans required for both rural and urban impacts to be evident may be quite different, and perceptions of drought and flood conditions may cloud their interpretations. A third issue is the critical societal requirement of landowner participation. The TSSWCB must have buy-in and permission from many landowners/managers before they can implement a successful brush control program. Some landowners are averse to government programs in general, some do not want to spend their money with the government cost share, while others may feel that any brush removal could damage hunting and the related lease income. These non-scientific issues can control the successful implementation of brush control for water yield enhancement in the State of Texas. An example of the societal issue is visually displayed in an aerial picture of the Pecos River watershed. One landowner/manager has controlled brush while the other side of the Pecos River owner/manager has not controlled brush (Fig. 1.1). Land owner/manager support for brush control is beyond the direct control of the TSSWCB.

Figure 1.1. A photograph of an uncontrolled brush area on the left and a controlled brush area on the right along the Pecos River.

A critical site-specific component of a successful brush control program is the species of brush that needs to be controlled to enhance water infiltration and/or runoff enhancement. The three primary groups of plants that need to be controlled in Texas are juniper (*Juniperus* spp.), mesquite (*Prosopis* spp.), and saltcedar (*Tamarix* spp.). These three groups are discussed in much further detail in the Objective 3 portion of this report. While these plants are, in general, location specific, recognition of the specific plant species to be controlled should factor into the decisions as to where to allocate the brush control monies.

Each of the twelve areas shown in Table 1.1 was visited by multiple members of our team, most often with TSSWCB employees. While all twelve areas were visited, we focus specific comments in this report on two areas that are representative of mesquite and saltcedar. Specific, detailed information about those two brush species are covered under Objective 3 of this project report. In this section, we evaluate the soils criteria relating to brush control for water yield enhancement on site-specific mesquite and saltcedar subwatersheds. These areas are located in Archer County (Figure 1.2) and in Stephens County (Figures 1.3 and 1.4). The Archer County site, a mesquite-infested area in the Lake Kickapoo watershed, was visited both on the ground and via helicopter flyover. The Stephens County site, Hubbard Creek Reservoir and its drainage area, represents a saltcedar-infested area, and the figures are aerial photographs from two different years, 2004 and 2006, respectively. The free-form white line in these figures represents the helicopter flight lines during the July, 2008 flyovers. Please note that the water level in Hubbard Creek Reservoir in 2006 is higher than that during 2004.

Fish and Rainwater (2007) stated five site-specific conditions for vegetative manipulation to enhance stormwater runoff and associated streamflow. The criteria that they listed were steeper slopes, more uniform slopes with limited soil disturbance, soils with lower infiltration capacities, south and west facing slopes, and closer proximity of contribution area to stream channels. Fish and Rainwater (2007) further enumerated two site-specific conditions for enhanced general baseflow to streams and rivers. These two criteria are soils with higher infiltration capacity near the streambed and higher water table elevation near the streambed. In addition, two criteria associated with the brush itself include brush cover distribution (density and proximity to the stream channel), and the size of the area with brush relative to the watershed area. Alternate, yet similar, criteria have been suggested to quantify stormwater runoff and associated streamflow or baseflow to streams or rivers, such as hydrologic soil type, slope of area, presence of an aquifer recharge zone, and proximity to stream channel.

Using the Fish and Rainwater (2007) soil-specific criteria for the portion of the watershed located adjacent to Lake Kickapoo in Archer County, Texas (Figure 1.2), Table 1.2 was developed for relative potential for runoff enhancement through brush control in those soil zones, assuming that brush is present. The simple scales shown in the notes utilize data from the county soil survey reports for slope, slope uniformity, and permeability. The proximity category is simply based on position of the soil polygon from the soil survey (Daigle, 1995), as shown on Figure 1.2.

The soil with the highest total number in Table 1.2 would have the greatest potential for enchanced runoff if existing brush was reducing runoff. For this area near Lake Kickapoo, it would be the Knoco-Vernon complex, 10 to 45% slope. Please note that the soil zones with the greatest slope, the Knoco-Vernon complex, 10 to 45% slope, would be too steep for mechanical brush removal. The erosion

Figure 1.2. Portion of the Lake Kickapoo watershed located in Archer County, Texas (white line indicates helicopter flight path).

Table 1.2. Relative Runoff Enhancement Potential by Soil Mapping Unit Using the Fish and Rainwater Criteria for the Lake Kickapoo Area as Depicted in Figure 1.2

Map Symbol	Slope Angle[1]	Slope Uniformity[2]	Permeability[3]	Proximity[4]	Total
AsC3	1	2	2	2	7
BeB	1	2	1	3	7
DnA	1	2	2	2	7
GrC	1	2	1	3	7
JoC	1	2	2	2	7
KaA	1	2	2	2	7
KaB	1	2	2	2	7
KvD	1	2	3	3	9
KvE	3	2	3	3	11
Ma	1	2	3	3	9
Mc	1	2	3	3	9
Po	1	2	2	3	8
VeC	1	2	3	2	8
VkD	1	2	3	3	9
We	1	2	2	3	8
WnB	1	2	2	2	7

1. Slope — 1 if <8%, 2 if 8 to 15%, 3 if >15%
2. Slope Uniformity — 1 if undulating, 2 if uniform
3. Permeability — 1 if 2-6 in/hr, 2 if 0.6 to 2 in/hr, 3 if <0.6 in/hr
4. Proximity — 1 if distal, 2 if proximal, 3 if adjacent

potential for 10 to 45% slope of the Knoco-Vernon complex would also preclude treatment of this area by mechanical means. The area could be aerially sprayed if brush exists, but brush control should leave the vegetation in place to minimize erosion potential. Several soils – Knoco-Vernon complex, 2-12% slope; Mangum clay; and Magnum clay frequently flooded – had the next highest numbered total indicating that they can be effectively treated to produce runoff if brush density is sufficient. The soil with the least number in Table 1.2 would be the soil with the least runoff potential. For this area near Lake Kickapoo, many of the soils would fit these criteria, having total scores of seven. Therefore, on this subwatershed, the Knoco-Vernon complex, 2-12% slope; Mangum clay; and Magnum clay frequently flooded would be the soil zones that probably should be treated for mesquite control. Since the Knoco-Vernon complex, 10 to 45% slope soil is so steep, one should maintain the vegetation in place to minimize water erosion and sediment transport into Lake Kickapoo. When we were on site, the lake had a reddish hue indicating sediment transport into the lake.

If baseflow enhancement through increased infiltration to shallow groundwater that drains into the stream, rather than runoff, is the desired result, a differing suite of soils should be treated for mesquite control. The total suite of soils for this subwatershed of Lake Kickapoo and the properties relative to potential baseflow enhancement properties are presented in Table 1.3. It should be noted that if the water table near the stream is presently below the streambed, the stream would lose water through the streambed to the shallow aquifer. It could take a number of years for the water table to rise above the streambed and cause flow from the shallow aquifer to the stream. In this Archer County area, the Bluegrove fine sandy loam, 1 to 5% slope, and the Grandfield fine sandy loam, 1 to 5% slope soils

Table 1.3. Relative Baseflow Enhancement Potential by Soil Mapping Unit Using the Fish and Rainwater Criteria for the Lake Kickapoo Area as Depicted in Figure 1.2

Map symbol	Mapping unit	Permeability[1]	Connectivity[2]	Total
AsC3	Aspermont clay loam, 1-5% slope eroded	2	2	4
BeB	Bluegrove fine sandy loam, 1-5% slope	3	3	6
DnA	Deandale silt loam, 0-1 % slope	2	2	4
GrC	Grandfield fine sandy loam 1-5% slope	3	3	6
JoC	Jolly Rock outcrop complex 2-12% slopes, stony	2	2	4
KaA	Kamay silt loam, 0-1% slope	2	2	4
KaB	Kamay silt loam, 1-3% slope	2	2	4
KvD	Knoco-Vernon complex, 2-12% slope	1	3	4
KvE	Knoco-Vernon complex, 10-45% slope	1	3	4
Ma	Mangum clay, occasionally flooded	1	3	4
Mc	Mangum clay, frequently flooded	1	3	4
Po	Port-Wheatwood complex occasionally flooded	2	3	5
VeC	Vernon clay, 1-5% slope	1	2	3
VkD	Vernon-Knoco complex, 2-8% slopes	1	3	4
We	Wheatwood silt loam, occasionally flooded	2	3	5
WnB	Winters loam, 1-3% slopes	2	2	4

1. Permeability 1 if <0.6 in/hr, 2 if 0.6 to 2 in/hr, 3 if 2-6 in/hr
2. Connectivity 1 if distal, 2 if proximal, 3 if adjacent

should be the soil zones treated for increased baseflow if sufficient brush exists. The Vernon clay, 1-5% slope would be the soil zones least likely to increase baseflow.

Similarly, using the Fish and Rainwater (2007) soil criteria for the area located adjacent to Hubbard Creek Lake in Stephens County (Figures 1.3 and 1.4), Table 1.4 was developed for runoff enhancement potential (Table 1.4). Again, this analysis is based on the assumption that brush is present in sufficient density and location to warrant treatment.

The soil with the highest number exists in the area with the greatest runoff enhancement potential if brush density is sufficient to limit runoff. For this area, the highest number was 11 for the Owens-Harpersville complex hilly, extremely stony soils. There were several soils with the lowest runoff enhancement potential. These include the Bastrop fine sandy loam, 0-1% slope; Bonti-Exray complex, gently undulating; and the Minwells fine sandy loam 1-3% slope soils (Table 1.4). The county soil survey information was taken from Cyprian (1994).

If baseflow, rather than runoff, enhancement is the desired result, a widely differing suite of soils should be treated for saltcedar control (Table 1.5), if the saltcedar is present. In this Stephens County area, the Bastrop fine sandy loam, 0 to 1% slope should be the soil treated for increased baseflow. These soils are high in connectivity and/or in permeability, thus facilitating infiltration potential. The soils with

Figure 1.3. Portion of the Hubbard Creek Lake watershed located in Stephens County, Texas, in 2004 (white line indicates helicopter flight path).

Figure 1.4. Portion of the Hubbard Creek Lake watershed located in Stephens County, Texas, in 2006 (white line indicates helicopter flight path).

Table 1.4. Relative Runoff Enhancement Potential by Soil Mapping Unit Using the Fish and Rainwater Criteria for the Hubbard Creek Lake Area as Depicted in Figures 1.3 and 1.4

Map Symbol	Slope Angle[1]	Slope Uniformity[2]	Permeability[3]	Proximity[4]	Total
BfA	1	2	1	2	6
BgB	1	2	2	3	8
BmB	1	2	2	3	8
BrC	1	1	2	2	6
BxE	2	2	2	2	8
Fr	1	2	2	2	7
Ga	1	2	2	2	7
HsB	1	2	2	3	8
LeA	1	2	3	2	8
MfB	1	2	1	2	6
OcC	1	2	3	3	9
OxE	3	2	3	3	11
ThC	1	2	3	2	8
TrA	1	2	3	3	9
TuB	1	2	2	3	8
WcA	1	2	2	2	7
WcB	1	2	2	2	7

1. Slope 1 if <8%, 2 if 8 to 15%, 3 if >15%
2. Slope Uniformity 1 if undulating, 2 if uniform
3. Permeability 1 if 2-6 in/hr, 2 if 0.6 to 2 in/hr, 3 if <0.6 in/hr
4. Proximity 1 if distal, 2 if proximal, 3 if adjacent

these criteria are the Bastrop fine sandy loam, 0-1% slope; Bluegrove loam, 1-3% slope; Bluegrove flaggy loam, gently sloping; Hensley, gently sloping; and the Truce fine sandy loam, 1-3% slope soils. The soils least likely to produce baseflow are the Leeray clay, 0-1% slope and the Throck clay, 1-5% slope soils due to their low permeability.

These numerical examples are intended for illustrative purposes only. Treating only selected soil polygons in the landscape would produce an odd mosaic for treatment. Brush control treatment areas should be designed in concert with the landscape position. The numerical examples above utilized not only the soil parameters, but also the positional relationship with the water bodies. Soils in the riparian areas were scored higher than those away from the water. This information must be combined with the area's brush density and distribution.

A less intensive soils criterion would be to target the soils in the soil associations that adjoin the lake/reservoir. While the soil association is more general than the use of soil series, it adds the adjoining location aspect to the area to be treated that the soil series do not have. Using the soil association criterion, the soil associations adjoining Lake Kickapoo in Archer County would be the Vernon-Knoco, and Tillman-Vernon associations. The soil association in the riparian zone of the Wichita River that could benefit from brush control, if sufficient brush is present, would be the Wheatwood-Mangum association.

Table 1.5. Relative Baseflow Enhancement Potential by Soil Mapping Unit Using the Fish and Rainwater Criteria for the Hubbard Creek Lake Area as Depicted in Figures 1.3 and 1.4

Map symbol	Mapping unit	Permeability[1]	Connectivity[2]	Total
BfA	Bastrop fine sandy loam, 0-1% slope	3	2	5
BgB	Bluegrove loam, 1-3% slope	2	3	5
BmB	Bluegrove flaggy loam, gently sloping	2	3	5
BrC	Bonti-Exray complex, gently undulating	2	2	4
BxE	Bonti-Exray-Truce complex, hilly, very stony complex, gently undulating	2	2	4
Fr	Frio silty clay, occasionally flooded	2	2	4
Ga	Gageby clay loam, occasionally flooded	2	2	4
HsB	Hensley , gently sloping	2	3	5
LeA	Leeray clay, 0-1% slope	1	2	3
MfB	Minwells fine sandy loam 1-3% slope	3	2	5
OcC	Owens clay, 1-5% slope	1	3	4
OxE	Owens-Harpersville complex, hilly, extremely stony	1	3	4
ThC	Throck clay, 1-5% slope	1	2	3
TrA	Thurber clay loam, 1-3% slope	1	3	4
TuB	Truce fine sandy loam, 1-3% slope	2	3	5
WcA	Wichita clay loam, 0-1% slope	2	2	4
WcB	Wichita clay loam, 1-3% slope	2	2	4

1. Permeability 1 if <0.6 in/hr, 2 if 0.6 to 2 in/hr, 3 if 2-6 in/hr
2. Connectivity 1 if distal, 2 if proximal, 3 if adjacent

Using the soil association criterion, the soil association to be treated in Stephens County that is adjacent to Hubbard Creek Lake would be the Bonti-Truce-Bluegrove, Bluegrove-Thurber-Leeray, and Bastrop-Minwells soil associations, if brush density is sufficient. The target soil association within the riparian areas would be the Gageby-Thurber-Frio soil association.

In discussions with the TSSWCB personnel, they mentioned controlling brush in strips that are perpendicular to the water body. This pattern allows for flow across the permeable soils to enhance infiltration into shallow aquifers near the water bodies, or runoff directly into the water bodies. The use of soil associations would allow better descriptions of the treated areas.

Using the TSSWCB sites listed in Table 1.1, the soil associations associated with the riparian areas and/or lakes/reservoirs/impoundment areas are shown in Table 1.6. The data in this table are for illustrative purposes only. Some of these soil associations occur away from the rivers or streams that have been treated. Also, many of these associations occur where there are not water-impounding areas. In those instances, treating and controlling brush would not have the positive water yield enhancement benefits that adjoining areas would have. The utility of the soil associations is shown in the Pecos River flyover Figure 1.5. This picture shows the flight path in white for the visual tour of the area. Terrell County is south and west of the Pecos River, and Crockett County is north and east of the river. The soil association along the river is the Sanderson-Regan soil association. Saltcedar in this area should be treated.

Table 1.6. TSSWCB Sites and the Soil Associations for Riparian and Impoundment Areas

Site	Soil Association	
	Riparian	Impoundment Area
North Concho River (Grape Creek, Chalk Creek)	Rioconcho-Spur	Kimbrough-Mereta-Angelo
Twin Buttes Reservoir (Pecan Creek)	Rioconcho-Spur	Kimbrough-Mereta-Angelo
Pedernales River	Brackett-Purves-Doss and Luckenbach-Pedernales-Heatly	Brackett
Lake Ballinger	Spur-Colorado-Miles	Spur-Colorado-Miles
Oak Creek Reservoir	Potter-Veal-Mereta	Cobb-Cash
Champion Creek Reservoir	Cobb-Miles	Spade-Latom
Nueces River	Coquat-Cochina and Aransas-Sinton	Victoria-Raymondville-Orela and Aransas-Sinton
Hubbard Creek Reservoir	Trinity-Frio	Houston Black-Heiden, Bonti-Truce-Bluegrove and Bluegrove-Thurber-Leeray
Pecos River / Upper Colorado River	Sanderson-Regan	Kimbrough-Olton-Mereta
Canadian River	Acuff-Paloduro-Olton and Mobeetie-Tascosa	Dumas-Dalhart, Mobeetie-Tascosa and Burson-Quinlan-Aspermont
Wichita River (Lake Arrowhead)	Wheatwood-Mangum	Kamay-Bluegrove-Deandale and Bluegrove-Jolly-Weswind

Figures 1.6 through 1.11 illustrate some examples of the variability encountered during our visits to the designated watersheds. These examples were selected for comment and inclusion in this report to specifically document the variability of conditions encountered in brush control activities, and to emphasize the fact that while there are general guidelines applicable to the selection of areas "best suited for treatment," in the final analysis every location tends to be "unique" in one or more aspects of selection criteria. Some of the tonal contrasts, obvious in these pictures, are the result of differing land management practices – primarily grazing intensities. However, others are the artifact of digital image mosaic efforts to "splice" several photographs together for purposes of this report.

The Lake Ballinger watershed is one of the TSSWCB's selected watersheds with the largest complement of land in production agriculture or cropland. This watershed is fairly long and narrow, beginning south of Sweetwater and extending southward to two reservoir impoundments. This watershed also was one of the watersheds with the least amount of treated acreage. In Figure 1.6 one can easily see the helicopter flight path, delineated as a white line, which followed the main drainage channel very closely in the upper portion of the figure. Treated area boundaries as provided for this report by the San Angelo office of TSSWCB are outlined with yellow – some areas are exactly adjacent to the drainage course while others are as much as 2000 meters away from the channel. Several factors are illustrated by the locations of treated areas. Considerations no doubt included proximity to the drainage channel, willingness of the landowner to participate, density of brush to be controlled, and the presence or absence of brush. No brush treatment is needed in the areas in crop production, but it should also be noted that croplands are typically cultivated to reduce runoff potential. The treated area directly west of the first

Figure 1.5. Pecos River channel southeast of Sheffield, Texas (white line indicates helicopter flight path).

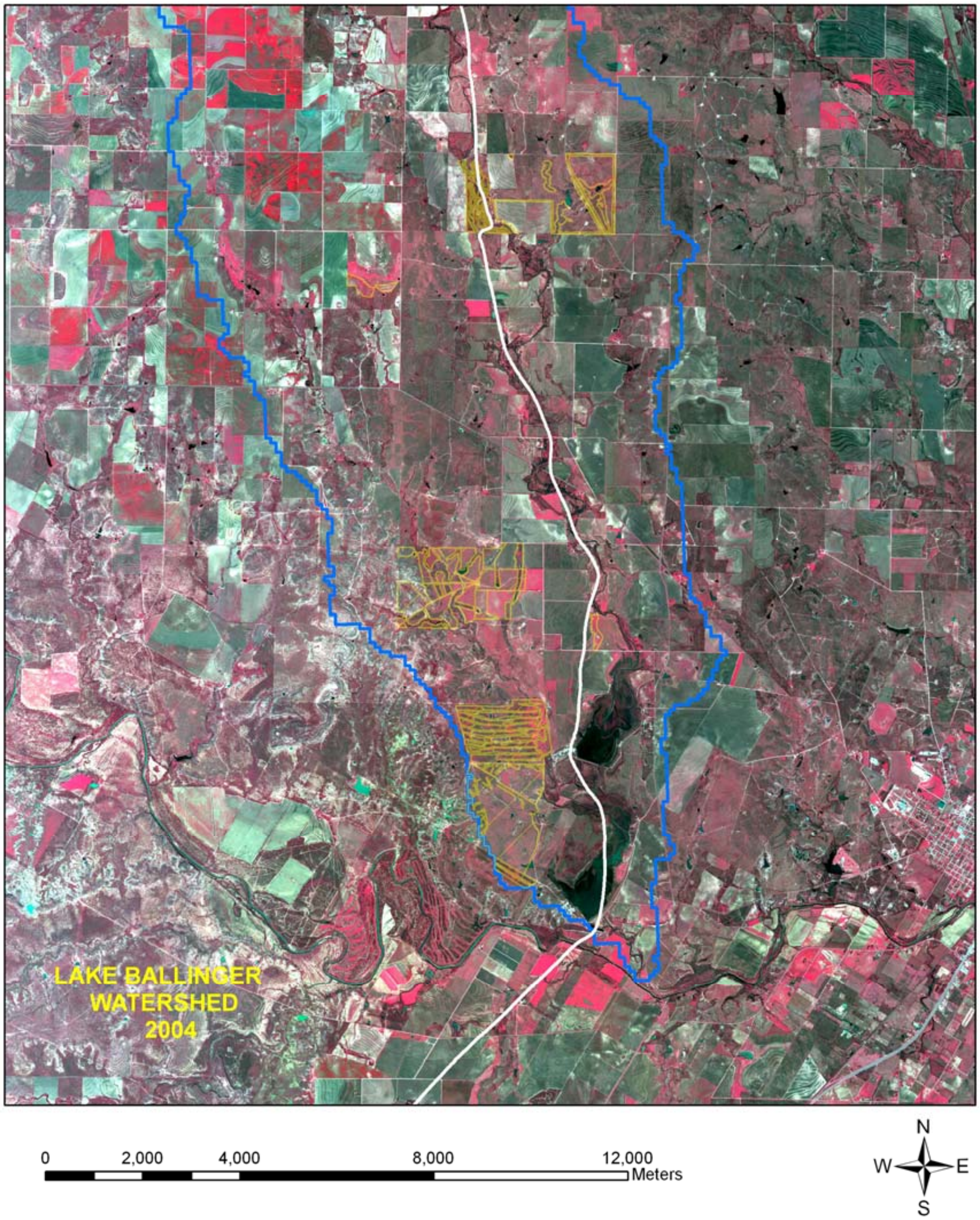

Figure 1.6. Southern end of Lake Ballinger watershed (white line indicates helicopter flight path).

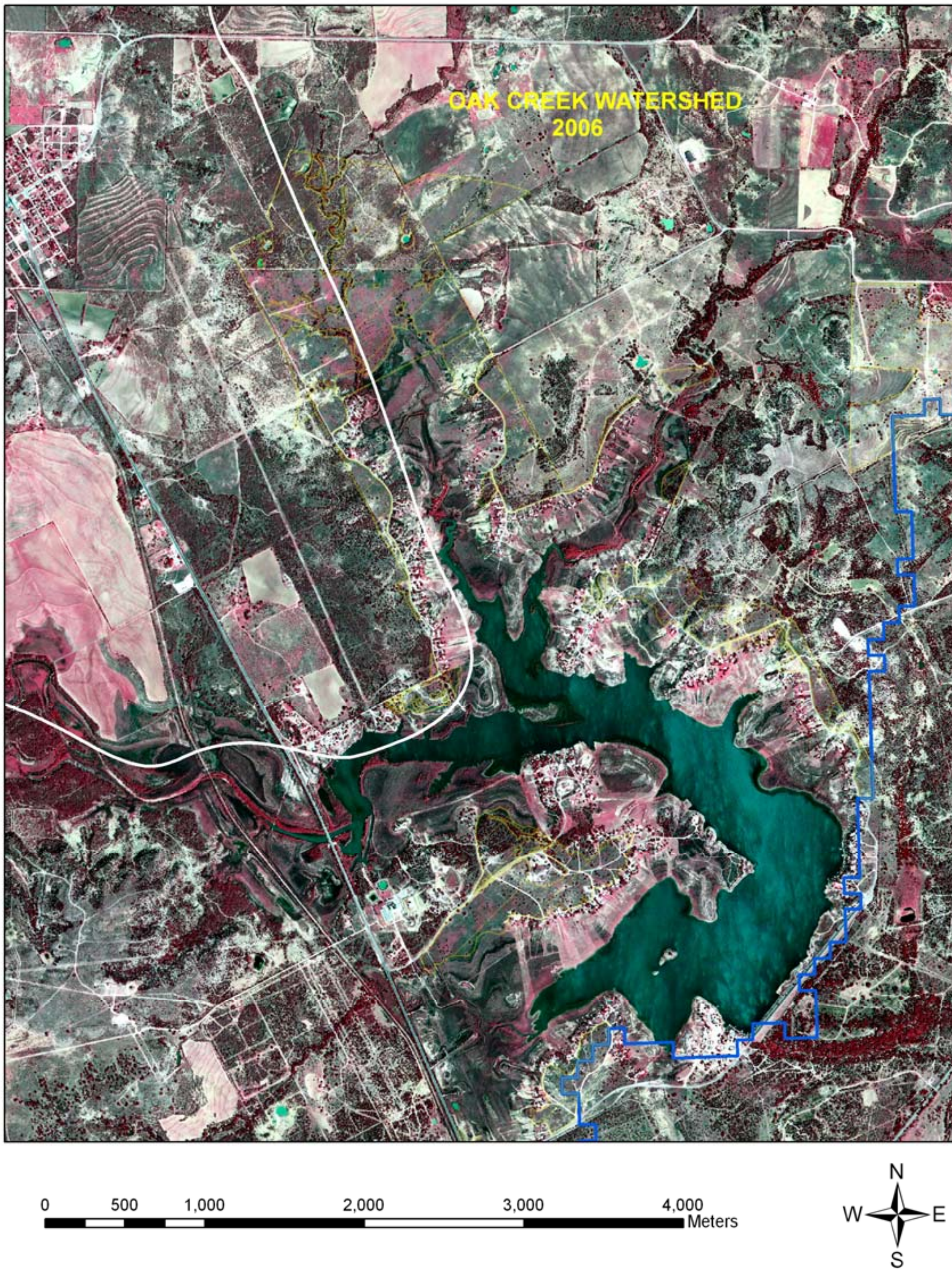

Figure 1.7. Oak Creek watershed area at Oak Creek Reservoir (white line indicates helicopter flight path).

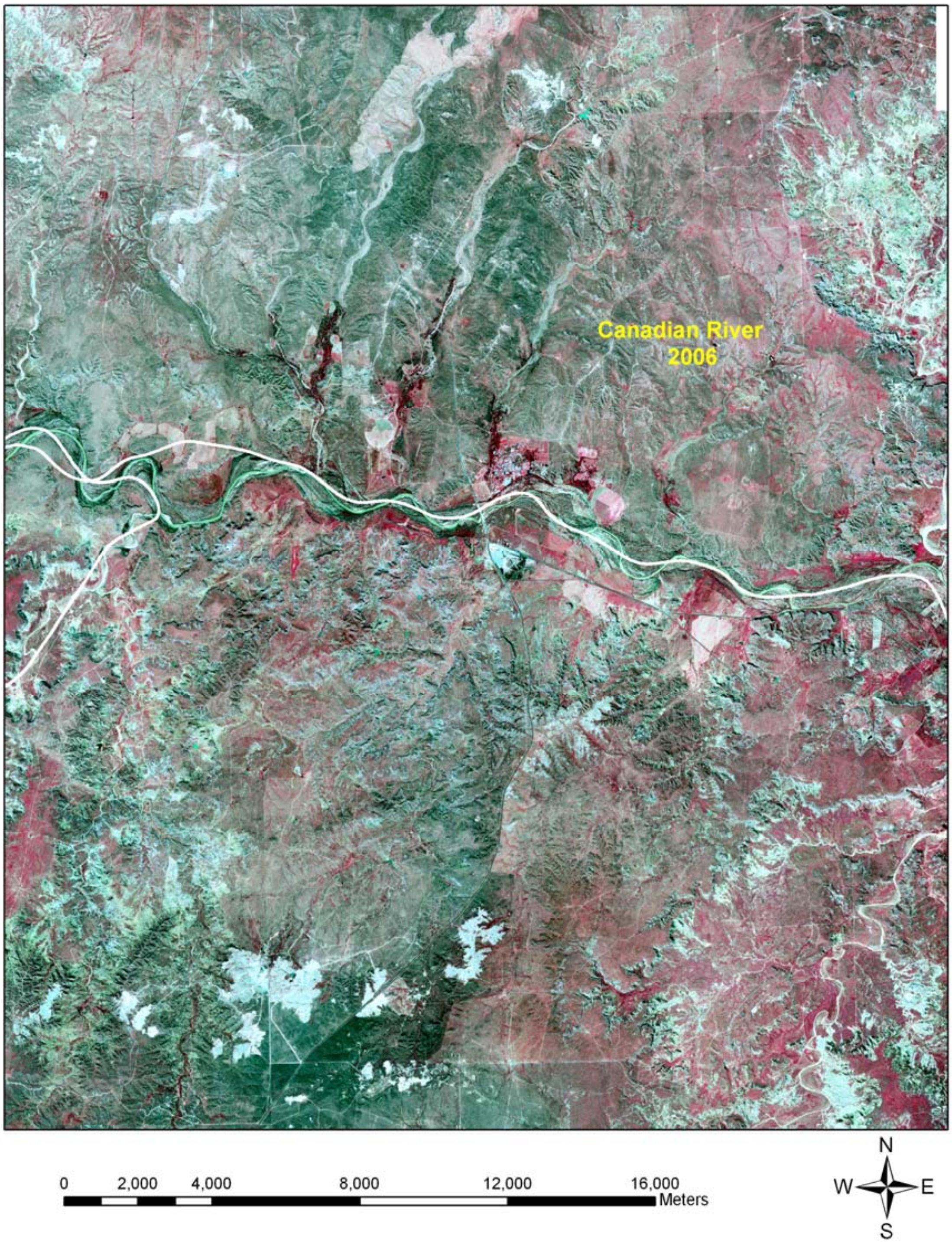

Figure 1.8. Canadian River channel near Boys Ranch, Texas (white line indicates helicopter flight path).

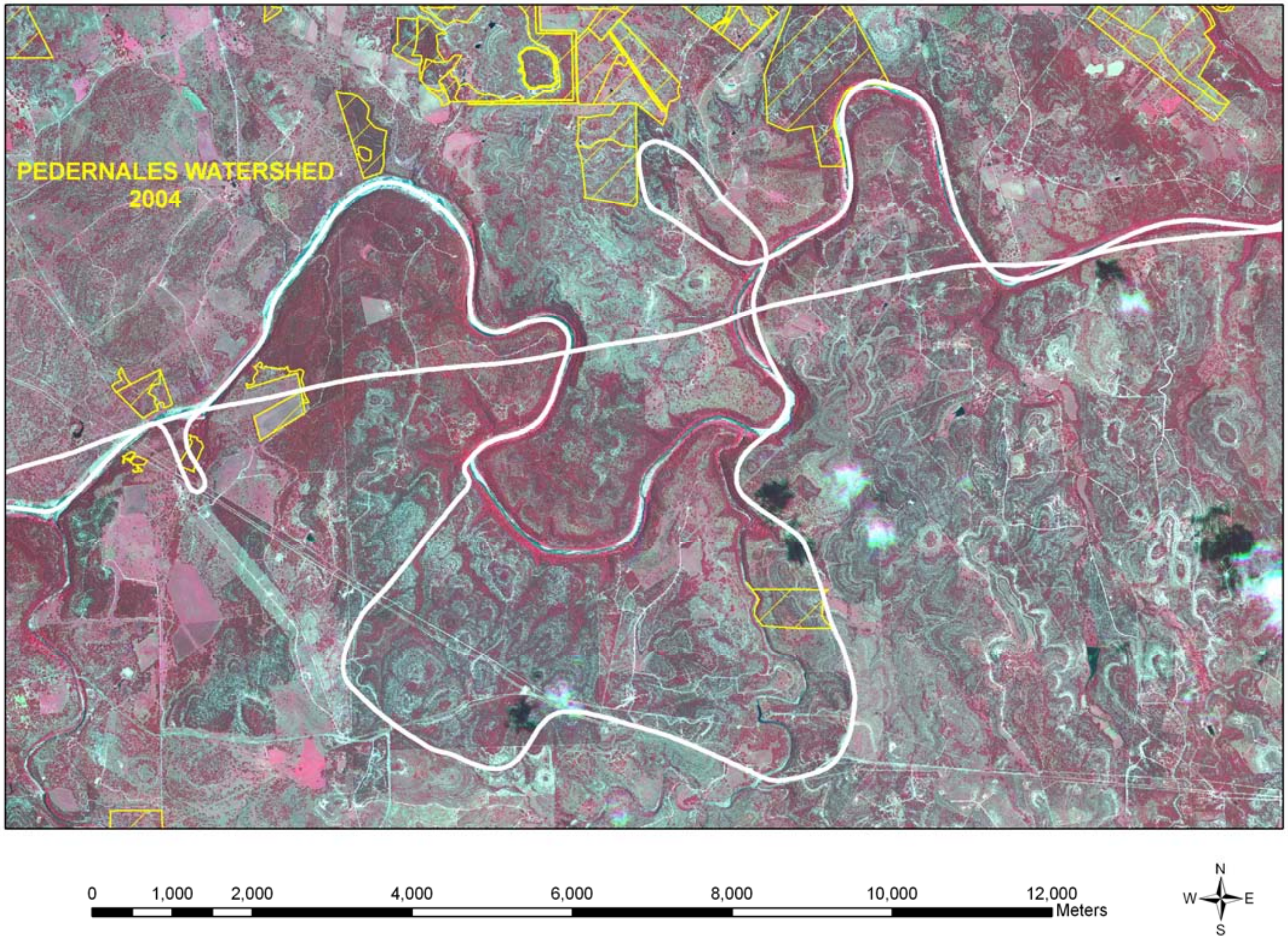

Figure 1.9. Pedernales Watershed with 2004 date image background (white line indicates helicopter flight path).

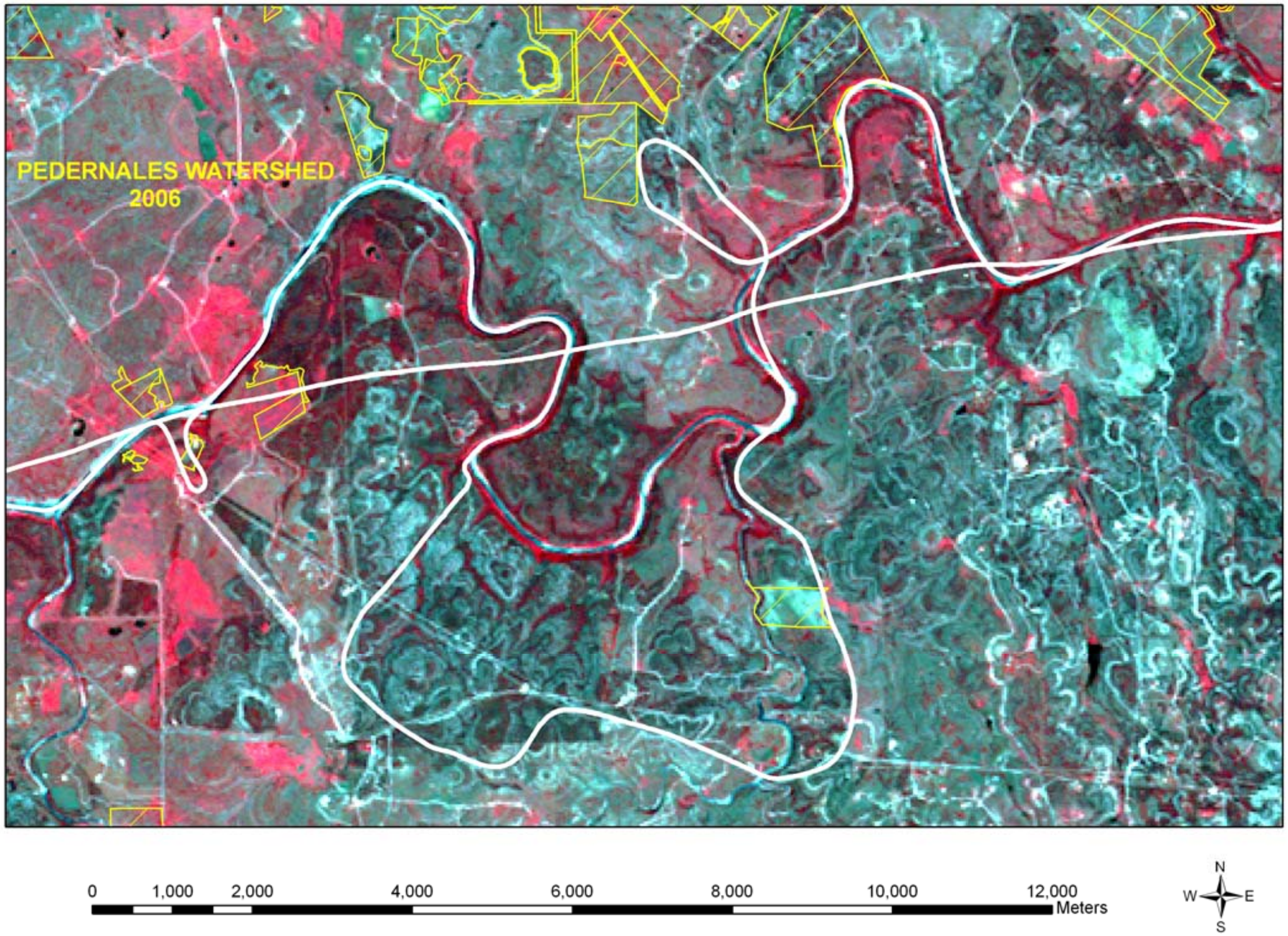

Figure 1.10. Pedernales Watershed with 2006 date image background (white line indicates helicopter flight path).

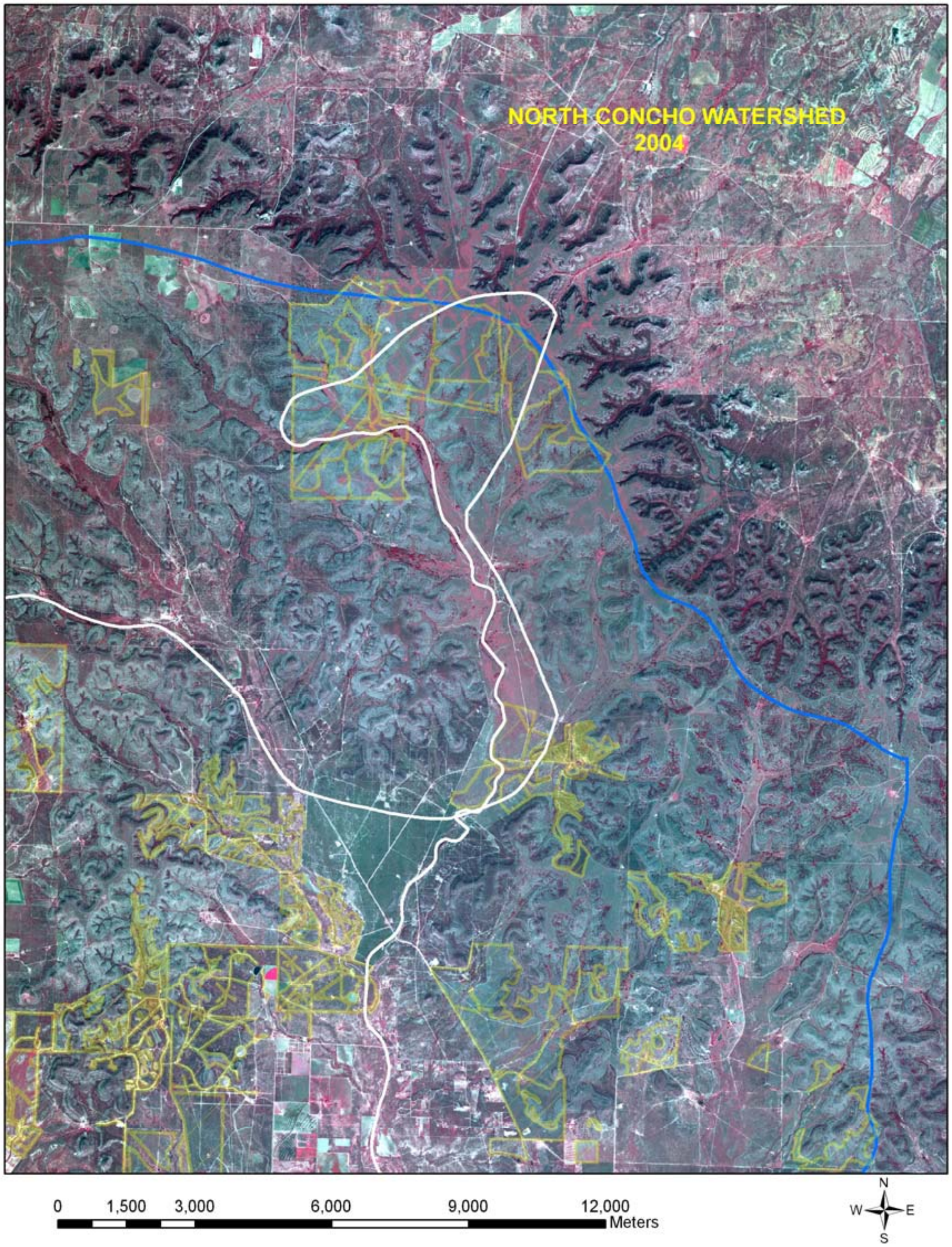

Figure 1.11. North Concho watershed vicinity of Grape Creek (white line indicates helicopter flight path).

impoundment illustrates the use of a strip pattern of brush control, with the long axis of the treatment areas aligned towards the drainage channel or water body in this case. The blue line indicates a portion of the watershed boundary as provided for this report by the San Angelo office of TSSWCB.

The Oak Creek Reservoir watershed contained a variety of land use and land cover situations (Figure 1.7). As with the Lake Ballinger watershed, Oak Creek Reservoir's watershed has a fairly large amount of land devoted to production agriculture or cropland. Brush control areas appear to have been associated with willing landowners and areas of land that had not been converted from native vegetation to cropland. The helicopter flight path is shown as a white line generally following two of the main drainage channels feeding Oak Creek Reservoir. The blue line indicates a portion of the watershed boundary as provided for this report by the San Angelo office of TSSWCB. Treated area boundaries as provided for this report by the San Angelo office of TSSWCB are outlined with yellow. Some areas are virtually adjacent to the drainage course, while others are as much as 1000 meters away from the channel. In fact, some of the treated areas could be described as riparian treatments. The bulk of the treated areas in this watershed involved mesquite and juniper. Saltcedar is present immediately adjacent to the lake itself. During our on-site visit in 2008 the lake level was high enough to have inundated many of these plants. Saltcedar is not tolerant of extended inundation, and there are several indications in various publications that 60 to 90 days of continuous inundation is sufficient to kill most saltcedar plants.

The plant of interest in the Canadian River watershed is saltcedar. Therefore, treated areas were locations where saltcedar is the dominant plant directly adjacent to the river channel. The white line in Figure 1.8 indicates the flight path of the helicopter during our flyover of this site. We also visited portions of the Canadian River site on the ground. Boys Ranch is located in the approximate center of this picture, and we examined several locations both upstream (west) and downstream from there on the ground. We were primarily interested in observing those areas in which saltcedar had been controlled via aerial spraying, as well as other areas that had not been sprayed because of landowner unwillingness to participate in the program. Only rarely were there stands of saltcedar in tributary drainages, and when that condition did occur we did not observe any obvious treatment effects. The focus of the brush control program in this watershed has been on saltcedar in the main channel of the Canadian River, although there have been some discussions about attempting to control small populations of the plant that may serve as a "seed source" in areas that are not in the "main channel."

The portion of the Pedernales watershed shown in Figures 1.9 and 1.10 is just west of Lake Travis. The white line indicates the flight path of the helicopter during our aerial observation of this watershed. While we concentrated on areas adjacent to the main channel, we also observed treated areas away from the channel and explored some tributary channels, as can be seen in the lower center of the picture. Ground access was severely limited, and we opted to only visit this location via the helicopter flight. Previously treated locations are shown with yellow boundaries as provided for this report by the San Angelo office of TSSWCB. Some areas are virtually adjacent to the main drainage course, while others are located on tributary channels, and still others are as much as 3000 meters away from the main channel or primary tributary channels. The primary plant of concern in this watershed is juniper. The dark red tone just to the west of the image center is a very dense stand of juniper. Comparison between the 2004 (Figure 1.9) and 2006 (Figure 1.10) dates clearly shows that treatment occurred between the two images for a small area just east of the word "Watershed" in the upper left hand portion of the image. The tone changes from dark red, indicating a high density of juniper, to a grayish green, indicative of more exposed soil with a light cover of grasses and forbs. In all likelihood, the juniper was mechanically removed from this site, and a good bit of soil exposure occurred as a result. The lack of clarity in the 2006 image is due to its lower digital resolution. Enlargement results in "blurring" as individual pixels become obvious in this image.

The portion of the North Concho watershed in Figure 1.11 is a tributary known as Grape Creek. There has been extensive brush control treatment as indicated by the areas outlined in yellow, provided by the TSSWCB office in San Angelo for this report. The area was visited on the ground with members of the TSSWCB and the Upper Colorado River Authority (UCRA) and via a helicopter flight. The white line is the flight path of the helicopter during the June 2008 flight. The blue line is a portion of the watershed boundary of the North Concho as provided by the San Angelo office of the TSSWCB. While all of the treatment areas depicted in this figure are away from the main drainage channel of the North Concho, the majority of them are adjacent to or in near proximity to tributary channels. Plant groups of interest in this area included juniper and mesquite. Juniper generally dominated the steeper slopes and higher elevation areas, while mesquite was dominant on flatter areas and at lower elevations. The predominant land cover in this watershed is native vegetation or rangeland, with the land use being a combination of grazing by domestic livestock and wildlife habitat. Some smaller areas are in production agriculture or cropland, as shown near the bottom center of this figure.

In McMullen County, a portion of the Nueces River watershed has been selected for brush treatment, with mesquite as the primary brush type. Landowner recruitment has been done recently by both the local SWCD and the NRCS. Digital GIS maps of the area and the contracted areas were under development by the local SWCD, but were not yet available to our research team. This area was visited by helicopter flyover only, as road and highway access to the treated sites was limited. Unfortunately, helicopter access to much of the target area was not allowed because of restricted military airspace. Under these conditions, it was not possible to generate useful figures combining treated areas, soil zones, and flyover path as we did for the other sites.

1.3 Agency Interactions

The fourth task was to visit other state and federal agencies with possible interests in brush control or water runoff enhancement. These visits were made in person and by telephone with other state and federal agencies. These agencies included the Texas Commission for Environmental Quality (TCEQ), Texas Water Development Board (TWDB), Upper Colorado River Authority, Texas AgriLife Research, and Extension Service, Texas Department of Agriculture (TDA), United States Department of Agriculture-Natural Resources Conservation Service (NRCS), and the Texas Parks and Wildlife Department (TPWD).

The vision, mission, and philosophy statement of the Texas State government is used to set basic values for state agencies. In the latest version, brush control was a part of a relevant benchmark for Texas' natural resources and agriculture priority goal to conserve and protect the state's natural resources. Brush control was placed under the water conservation heading, along with decreased water usage and increased water reuse. These words are included in the new strategic plans of state agencies, such as the

TSSWCB, TWDB, and TDA. In this section, common brush control or watershed management interests of other state and federal organizations are summarized. The order of presentation does not imply any prioritization of the agencies' roles relative to the TSSWCB brush control program.

1.3.1 Texas Water Development Board

Dr. Barney Austin, Director of the Surface Water Resources Division, described the connections between the TWDB and the TSSWCB's program. First, at the inception of the program, the initial funds for brush control were funneled through the TWDB. Second, the brush control program statute required the TWDB to advise the TSSWCB in this effort, and Dr. Austin meets regularly with Johnny Oswald of the TSSWCB on technical matters. He follows reports of brush management research, such as those by Dr. Brad Wilcox. He also attended a meeting three years ago in which several agencies were invited to give input about prioritizing brush control site selections. Third, the TWDB has assisted with funding for monitoring of brush control impacts through a two-year contract with the Upper Colorado River Authority (UCRA).

Dr. Austin has noted that recent reports on different brush control issues, such as Wilcox et al. (2005), show that there are some differences of opinion among the experts in the field. In this situation, he cautions all to be careful about quantifiable promises of water supply enhancement by brush control. Transmission losses and gains in streambeds should also be addressed as their impacts could be of significance. Opportunities for cooperative work in watershed management exist for the TSSWCB, Lower Colorado River Authority, U.S. Geological Survey, and others in the Pedernales watershed near Johnson City. Finally, Dr. Austin recommends formation of a permanent advisory group for the TSSWCB's brush program.

1.3.2 Texas Commission on Environmental Quality

The TCEQ is the state agency with responsibility for management of appropriative surface water rights allocations as well as protection and restoration of water quality. Laurie Curra, who manages the Clean Rivers program in the Office of Compliance and Enforcement of TCEQ, explained interactions with the TSSWCB. Ms. Curra works with John Foster and T.J. Helton of the TSSWCB to coordinate the Clean Water Act of 1987's Section 319(h) program for nonpoint source pollution prevention and abatement. The Section 319 program currently has about 70 active projects, which are based on watershed protection, rather than service to individual landowners. About half of the projects come from the Section 303(d) list of impaired waters that is updated by the TCEQ and approved by the Environmental Protection Agency (EPA). The other Section 319 projects are at sites that are justifiably in need but not on the 303(d) list. Apparently some water entities do not wish to be classified as impaired, but still want to apply for the program's grants for efforts to improve water quality. The TCEQ and TSSWCB also cooperate on grants for the production of watershed protection plans to set agendas for data collection, data evaluation, and proposal of activities to improve or protect river water quality. It should be noted that the funding for water quality issues managed by the TSSWCB is approximately five times that available for all other programs.

To date, there has never been an expenditure of Section 319 funds for brush control for water yield enhancement, because the program is intended for water quality improvement. Some have confused the application of Section 319 funds for saltcedar control along the Colorado River between Lake J.B. Thomas and Lake E.V. Spence with brush control for water yield. The saltcedar treatment was intended to improve total dissolved solids and sulfate concentrations in the river. Over 11,000 acres of saltcedar, 75 ft on either side of the river channel, have been treated as about 95 percent of the riparian landowners participated. The Colorado River Municipal Water District continues to monitor the water quality and flow rates along this section of the river. Similar work has occurred and could be continued along the Pecos River, pending completion of its watershed protection plan. The TSSWCB has used these and related funds for installation and operation of stream gauging stations by the U.S. Geological Survey.

1.3.3 Texas A&M AgriLife Research and Extension Service, Texas Water Resources Institute

Faculty and staff research and extension scientists affiliated with the Texas A&M University system have been involved with the TSSWCB brush control program since its inception in 1998. Currently, the College of Agriculture and Life Sciences at Texas A&M University, Texas AgriLife Research, and the Texas AgriLife Extension Service are three of the five components of the Texas AgriLife organization. The Texas Water Resources Institute (TWRI) is one of the member institutes of Texas AgriLife. Research and extension centers are scattered across the State, allowing service to landowners and interactions with other university scientists and engineers. As part of a land grant institution, Texas AgriLife has access to federal funding programs for agricultural, water, and environmental concerns, and has also been successful in leveraging their expertise and facilities in cooperative research and applied projects around the state.

As noted in the initial feasibility studies for the TSSWCB brush program, TWRI and other research and extension scientists were directly involved in the original estimates of potential water yields from brush control at the locations considered for treatment. Since the early 1990s, many project reports, white papers, and publications have been generated by AgriLife scientists, and many of these documents are cited in our section on estimation of water yield from treatment of saltcedar, mesquite, and juniper. In some locations, such as the Pecos River, AgriLife research and extension staff have directly led brush control planning, application, monitoring, and reporting. Dr. Allan Jones, Director of TWRI, states that while their past work has contributed to the scientific debates about the challenges of quantifying water yields from brush control, he and his colleagues are committed to assisting the TSSWCB and its cooperators as they insert more scientific hydrologic and geologic criteria into their site selection, treatment, maintenance, and monitoring policies.

1.3.4 Upper Colorado River Authority

Hydrogeologist Scott McWilliams and hydrologist Chuck Brown of the UCRA were visited by our team at their office in San Angelo. The UCRA has been directly involved in the TSSWCB's brush control program since 1999, when they were awarded a contract to perform monitoring and assessment services for work in the North Concho River watershed. The UCRA also received funding from the TWDB and the EPA (through the 319 program managed by the TSSWCB), and the monitoring program was planned for a 10-year duration. A major report on those efforts is nearing completion at this time.

Details of the UCRA monitoring efforts are provided in the monitoring section of this report. The UCRA's efforts have included paired treated/untreated comparisons, such as the East and West forks of Grape Creek, and the mesquite evapotranspiration (ET) observations with researchers from Tarleton State University. Funding levels limited the amount and type of monitoring equipment and placements, but a number of useful lessons are being learned in their studies, as is discussed in the monitoring section.

1.3.5 Texas Institute for Applied Environmental Research

The Texas Institute for Applied Environmental Research (TIAER) at Tarleton State University has worked with the TSSWCB and UCRA since 2000, primarily on monitoring issues. TIAER researchers have been involved in monitoring ET and streamflow impacts of brush control at sites near San Angelo. These projects are discussed in the monitoring section of this report. Drs. Ali Saleh and Larry Hauck were interviewed by telephone for this report.

1.3.6 U.S. Department of Agriculture, Natural Resources Conservation Service

The primary federal agency with interest in brush control is the NRCS. According to Susan Baggett, State Resource Conservationist, brush invasion is the number one resource management problem in Texas. NRCS funding for brush control comes from the Environmental Quality Incentives Program (EQIP) for individual landowners, with average total values for the state of approximately $65,000,000 for each of the last four fiscal years. The primary focus of the EQIP funds in the last four years has been brush control due to its high priority in the criteria used in the application process for 200 of the 254 Texas counties. The funds pay up to 75 percent of the costs of the brush control work. For example, saltcedar treatment can qualify for 75 percent funding. TSSWCB funds have often been used to provide additional support. Cooperation between the NRCS and TSSWCB is especially close, as both agencies work together to serve individual landowners. In some cities, the NRCS and Soil and Water Conservation District staff even share office space.

The NRCS recognizes that brush control improves land for grazing, wildlife, and aesthetics, as well as for increased runoff and possible groundwater recharge. Positive anecdotal evidence of increased stream flows have been reported, but little monitoring for actual quantification has occurred. They recognize the need for pretreatment and long-term maintenance by combining brush removal with a system of practices, such as prescribed grazing. The future of the EQIP program will be affected by the eventual completion of the National Resource Inventory, which is expected next year, and future farm bills.

1.3.7 Texas Department of Agriculture

The TDA was represented by Mike McMurry, Director for Endangered Species, who explained common interests with the TSSWCB's brush program. The TDA has no financial connection to brush control, but rather is involved in policy issues. Through the input of its elected commissioners and the TDA staff, they can encourage rural landowners to consider brush control as a way to improve their lands for ranching, farming, and wildlife enhancement, as well as potential water yield. The TDA also licenses prescribed burn activities, which are sometimes used for brush removal. McMurry and his colleagues also follow the progress of research findings in brush management impacts.

McMurry first stressed the importance of follow-up brush management after initial treatment as part of a long-term maintenance program. While initial treatment is often cost-shared with government programs, the landowner is normally responsible for all later work. Second, both groundwater and surface water impacts must be considered. Third, the role of vegetation in nutrient management in surface water bodies must be included in the overall watershed management approach. Finally, the TDA generally sees treatment of invasive species as a public good.

1.3.8 Texas Parks and Wildlife Department

The Texas Parks and Wildlife Department (TPWD) is primarily interested in ecosystem preservation and wildlife habitats. Dr. Mike Berger, Chief of the Wildlife Division, described the agency’s mandate to not deplete wildlife or the related habitat as potentially supportive of manipulation of vegetation to increase water yields. For example, they see no habitat benefit whatsoever to saltcedar and suggest that it is the number one plant that should be controlled to enhance water yields from various watersheds. Some brush control activities could be considered as restoration of native habitat, which is another desirable goal. The density of juniper in many areas is much greater today than it was historically, therefore a reduction in juniper density would be seen positively. Similar to the TDA, the TPWD does not have funding or regulatory authority relative to brush control, but they do attempt to influence policy issues when given the opportunity.

2. Identification of Proper Monitoring Approaches and Upgrades

2.1 Basic Monitoring Requirements

With typical guidance in hydrologic references, such as Viessman and Lewis (2003), it is possible to identify the basic requirements of an appropriate monitoring system to allow observation of water yield changes in a watershed. The first requirement is proper delineation of the watershed of interest, which means that a streamflow observation point is identified and the upstream area that can contribute flow to that point is established from the local topography. Next, the areal variability of rainfall events must be compared to the watershed area so that the number of rain gauges and their spatial distribution can be selected. Many storm events may cause nonuniform rainfall distributions as they move across a watershed. Three rain gauges would be a minimum to allow for redundancy even if one gauge fails. Rain gauge sites should be selected to allow the installations to be unobstructed and to properly represent the different sections of the watershed. Engineers and hydrologists typically use Thiessen networks (Viessman and Lewis, 2003), a geometric network construct, to place the rain gauges in useful positions, both within and outside the target watershed, to properly observe spatial variations in precipitation. Inexpensive non-recording rain gauges can be used if they can be manually read and maintained within 24 hr of any rainfall event. Affordable continuously recording rain gauges with dataloggers can store the local observations over time, with less frequent data downloads and physical maintenance. At the end of this chapter, we provide more detailed descriptions of the different devices that can be deployed.

Positions for streamflow measurements and their related configurations must be selected carefully. Continuously recording stream gauges are necessary to capture the short-term flow changes caused by intense rainfall events. At a minimum, a stream gauge should be established at the outflow point of the watershed, utilizing a constructed permanent structure that causes a stable relationship, known as a rating curve, between streamflow rate and the water surface elevation, or stage, in the stream cross section. Typical choices include a bridge with a concrete-lined stream channel beneath the bridge, a low-water crossing with concrete or asphalt pavement with stabilized subgrade, a single broad-crested weir, or a culvert and weir combination. Rating curves can be developed for these locations by performing field measurements of water depth and point velocities at different flow conditions or by calculating the relationship from hydraulic equations. Occasional maintenance is necessary to remove sediment and other debris, especially after flash flood conditions pass. The installation must be planned to allow observation of high and low flow rates, so that both storm runoff and baseflow conditions can be accurately represented. A second important issue is the amount of transmission gains or losses within the streambed. If the local groundwater table slopes toward the streambed and intercepts the bank above the water surface elevation of the stream, groundwater is discharging into the stream, and the streamflow rate increases as the water moves downstream. If the local groundwater table is well below the streambed, and if the streambed and geologic material are permeable, part of the streamflow is lost to seepage downward, and the stream is recharging the aquifer. These gains and losses may be of similar or greater magnitude than the losses of water to brush along the same part of the stream channel. Multiple sequential stream gauges are necessary to evaluate changes in streamflow between storm runoff events. The typical cost of a USGS stream gauge installation is approximately $25,000 for construction, followed by additional costs to perform the measurements for to establish the rating curves and maintain the gauge over time.

The direction of local groundwater flow, toward or away from the streambed, can be determined with monitoring wells near the stream at appropriately selected locations. The monitoring wells must be deep enough with a long enough screened interval to allow observation of a sufficient range of elevations to show the relationship between the water table and the stream water surface. At least three wells, in a triangle, not a straight line, on one side of the stream are necessary to allow determination of the magnitude and direction of the local groundwater gradient. It should be noted that the slope of the water table can get relatively steep near the stream. When the lateral extent of the aquifer, its specific yield, and the topography of the base of the aquifer are known, a monitoring well network can be used to estimate the locally available groundwater. When the top of casing elevation has been precisely surveyed, the depth to water can be measured manually as needed and adjusted to establish the water table elevation. The water table level can also be continuously monitored with pressure transducers and dataloggers.

If desired, estimates of local potential evaporation can be made by installing weather stations that collect enough variables, such as temperature, wind speed, relative humidity, and net radiation. These variables can be combined in appropriate theoretical equations to calculate potential evaporation that could occur from a free water surface. These values can then be scaled to estimate ET through different plants at different points in their growing seasons.

An example of a relatively thoroughly instrumented paired watershed comparison is taking place at the Honey Creek State Natural Area (HCSNA) in Comal County (Slattery et al., 2006). The cooperative work, which includes the USGS, USDA-NRCS, the San Antonio Water System (SAWS), and the San Antonio River Authority, was started in 1999 to evaluate the effects of Ashe juniper removal for surface and groundwater enhancement and water quality protection. The intent of the project was to extend the site-specific findings of the Seco Creek project (Dugas et al., 1998) to watershed scale. Juniper removal was planned for a 1.5-km^2 (0.56-mi^2) watershed, adjacent to a 0.93-km^2 (0.36-mi^2) watershed that would be left with juniper stands intact. Figure 2.1 shows locations of continuously recording tipping bucket rain gauges, weir-type streamflow gauges, and sites for observation of net radiation, soil heat flux, soil temperature, soil moisture, air temperature, and air vapor pressure, which allow estimation of ET by the Bowen ratio method. In addition, groundwater levels are monitored in one shallow 15-ft deep well and another deeper, 200-ft well. The wells are intended to indicate infiltration and changes in storage in the Trinity aquifer, but single wells for each purpose may not be sufficient. The pre-brush removal data collection period was 26 months from August, 2001 through September, 2003. The juniper was removed in the treatment watershed in 2004, and monitoring continues. Some of the data are available at the National Water Information System for the Honey Creek Sites near Spring Branch, Texas (*http://waterdata.usgs.gov/tx/nwis/current/?type=flow*).

As the TSSWCB moves forward with its water yield enhancement through brush control, some of the program's funds must be directed toward monitoring efforts to help provide justification for the program. As landowner participation and funding limit the amount of brush that can be treated, it is recommended that small watershed studies be built from existing sites, such as the Grape Creek paired watersheds discussed later in this section, or new small sites be selected to allow affordable instrumentation for hydrologic observations. One approach would be to select one or two sites each to better document the water yield potential from treatment of saltcedar, juniper, and mesquite, respectively, using the Fish and Rainwater (2007) criteria as demonstrated in Section 1. The selected sites would then

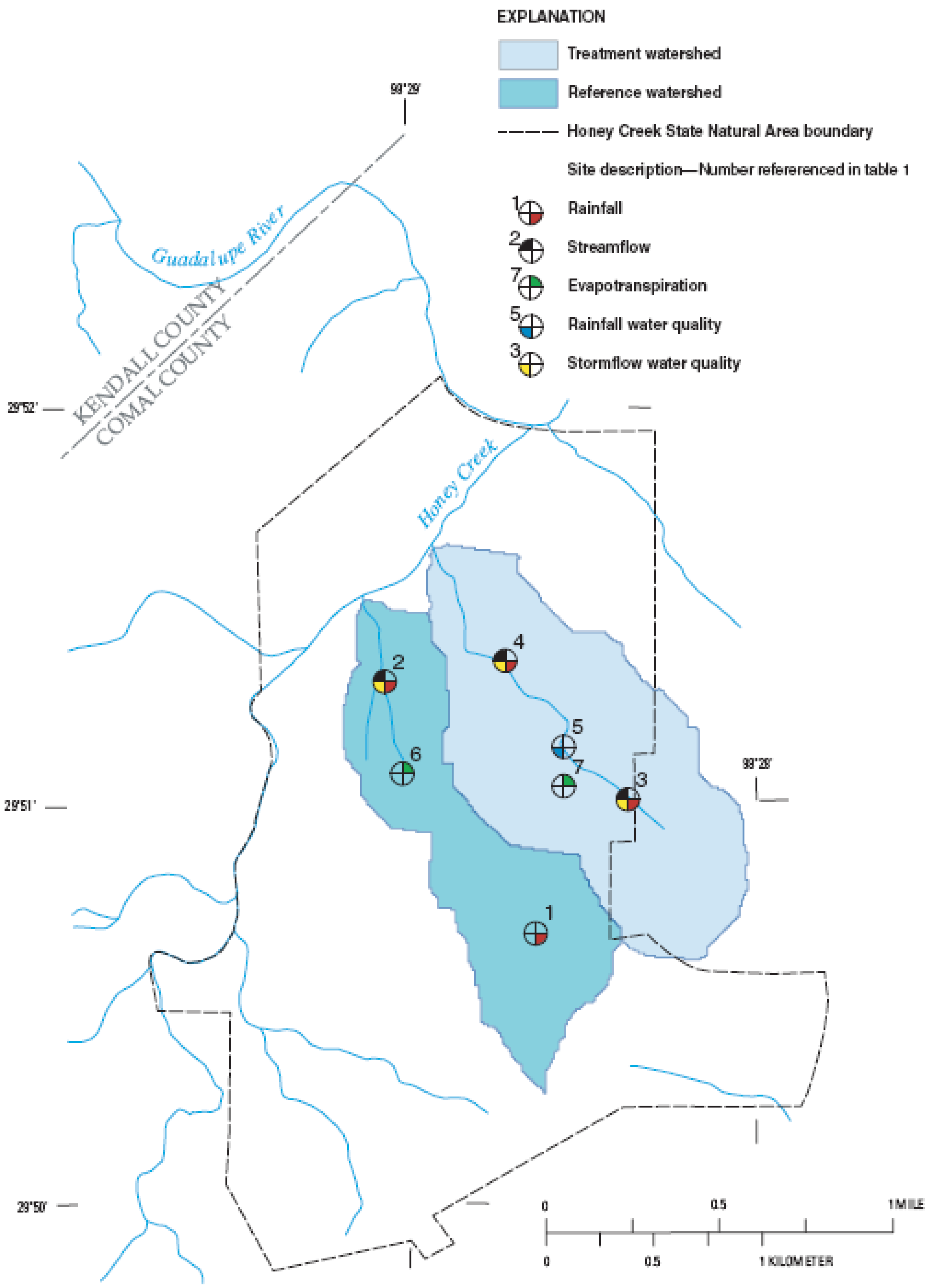

Figure 2.1. Locations of data collection sites in the paired watershed study in the Honey Creek State Natural Area, in Comal County (Slattery et al., 2006).

be instrumented for pretreatment monitoring to establish the hydrologic behavior of the watershed, and monitoring would continue through and after the brush treatment long enough to see a range of responses over varying rainfall events. Paired comparisons, such as the HCSNA approach above, are also just as useful when such conditions are available. Our research team believes that inclusion of funding for pre-treatment and post-treatment monitoring activities along with inclusion of additional technical expertise to design and evaluate monitoring programs will significantly enhance the effectiveness of future brush control programs.

2.2 Existing Monitoring Efforts

The typical biennial funding for TSSWCB's brush control program is concentrated on cost sharing for actual removal or killing of the target plants on the property of willing landowners. To date, little of the TSSWCB's legislative appropriations have been available for monitoring pretreatment or post-treatment surface water or groundwater conditions in the target watersheds. The feasibility studies that provided the impetus for the program were based on the modeling assumptions that [1] all landowners with brush would be willing to participate, [2] all brush in the watershed would be treated successfully for all time, and [3] future rainfall patterns would be similar to those in the past. If all three of these assumptions are true, then the probability of enhanced water yield from the watershed would be very high, and pre- and post-monitoring of the streamflow and groundwater would not be necessary. Of course, those three assumptions are very unlikely to be true, so monitoring programs are necessary to provide quantifiable observations of watershed behaviors.

The hydrologic processes of runoff generation from variable storm events, streamflow gains or losses due to groundwater interactions, and water losses due to ET by nearby vegetation are complex and variable over time, making them difficult to represent accurately with mathematical models. Observation of these processes for sufficiently long periods of time is necessary to see the ranges of streamflows that are caused by dry and wet weather conditions. The best situation would be a pre- and post-treatment paired watershed comparison. Two nearby watersheds with relatively similar sizes, land use distributions, soil and slope variations, groundwater conditions, and brush distributions would be instrumented for pretreatment monitoring with multiple continuous rain and streamflow gauges, as well as multiple groundwater monitoring wells, to allow several years of data collection that establish the range of pretreatment behaviors. One watershed would then receive brush treatment, and several post treatment years of data collection would continue. Qualified hydrologists and engineers would analyze the data to verify the impacts of brush treatment. The second best situation would be to set up a similar pre- and post-treatment monitoring program for a single watershed. The third best situation would be a paired watershed comparison with post-treatment data only. It is recognized that it may never be possible to use one of these approaches at every location that can benefit from brush control, but it should be possible to select a small number of sites with different dominant brush types, geographic locations, soil and slope conditions, and hydrologic characteristics for useful study. This work could be directed by the TSSWCB if its staff was expanded to include one or two in-house water resources engineers or hydrologist.

The initial brush control feasibility studies (TAES-BRC, 2000) employed the Soil and Water Assessment Tool (SWAT, Arnold et al., 1998) model to predict large-scale, typically over several counties, watershed behaviors on significant rivers and streams, most typically large enough to have previously installed stream gauges managed by the USGS, with each stream gauge recording the effects of large contributing areas. If the effects of brush control are to be quantifiable by the observations at this type of stream gauge, a significant fraction of that contributing area must receive successful treatment. To date, based on the site visits and data review discussed in section 1, the TSSWCB did not typically have sufficient funding or willing landowner participation to treat large portions of large watersheds. The primary exceptions have been riparian saltcedar treatment on the Canadian and Pecos rivers, and the significant treated areas near San Angelo. In recognition of these limitations, the TSSWCB and other interested parties have pursued smaller-scale investigations, often paired comparisons, as are summarized in this section. It should be noted that review of other brush control studies that are happening without TSSWCB cooperation was beyond the scope of this project.

2.2.1 North Concho River Area

As noted in Section 1, the UCRA has been involved in monitoring activities in the North Concho River watershed since 1999 (UCRA and TIAER, 2006). By December, 2007, almost 330,000 ac of the 950,000-ac North Concho watershed had been treated (TSSWCB, 2008), implying that a relatively large fraction of the watershed has been impacted. Two USGS continuous stream gauges, North Concho River near Carlsbad Station and North Concho River at Sterling City Station, existed prior to the TSSWCB brush program. Four additional USGS continuous stream gauges were added, the North Concho River above Sterling City Station, the North Concho River near Grape Creek Station, the Grape Creek near Grape Creek Station, and the Chalk Creek near Water Valley Station. The UCRA staff routinely analyzes the comparisons of flow rates and runoff volumes following significant rainfall events. Review of file memos for comparisons during 2001 to 2004 showed the staff's concerns for sizable transmission losses between the North Concho River near Carlsbad Station and the North Concho River near Grape Creek Station, a distance of about 8 river miles, following extended dry periods. Smaller transmission losses appear during extended wet periods, indicating impacts of alluvial aquifer saturation. The streamflow analyses would have benefited from precipitation data from a local network.

Groundwater elevations have been monitored on a quarterly basis, with up to 23 wells measured by the UCRA staff in Tom Green and Coke counties, and 18 wells measured by the Sterling County Underground Water Conservation District (UCRA and TIAER, 2006). The well locations apparently have changed somewhat over time, and the report did not provide a map to show the well locations or any details about the well depths, making it difficult to establish any local conclusions. The UCRA also had a project funded from 2004 to 2005 by the TWDB to attempt continuous groundwater level monitoring in the North Concho watershed using pre-existing wells and willing landowners. Unfortunately, only two wells were available, both being in the Chalk Creek subwatershed. Both wells were equipped with pressure transducer/datalogger combinations that collected data at 15-min intervals, but data could only be retrieved from one datalogger, even with the manufacturer's assistance. The one 6-mo dataset was compared to precipitation data from the WSR-88D system. The study was able to show that Chalk Creek experiences significant transmission losses to the shallow aquifer.

The UCRA and TIAER are currently cooperating on a paired-site study of reduction in ET by treatment of mesquite, and a report is to be published soon. Two adjacent sites, each approximately 200 ac, were selected in a flat mesquite-dominated area with deep soils in northern Tom Green County.

Mesquite was treated by herbicide in 2002 on site M1, while site M2 was left untreated. Weather monitoring instruments provide continuous wind speed, temperature, and vapor pressure to allow direct determination of sensible and latent heat fluxes, so that local ET can be estimated by the eddy covariance method. After careful quality assurance and quality control in the data analyses, the field observations indicated that during the mesquite growing season the ET for untreated site M2 significantly exceeded the ET at treated site M1. After the mesquite growing season ends, however, the ET at site M1 exceeded the ET at M2 due to the flourishing grass at site M1.

UCRA and TIAER are also currently cooperating on a paired watershed study of redberry juniper brush control. The two selected small watersheds are each approximately 100 ac with shallow soils on hillside slopes. Runoff is monitored continuously at the outlet for each watershed with a combination of two large parallel corrugated pipes with bubbler flow meters rated for large flows and a downstream H-flume with its own bubbler flowmeter to measure low flows. Tipping bucket continuous rain gauges were placed at each site in 2005. Pretreatment data collection continued into 2007, and mechanical removal of juniper at one site is ongoing in 2008.

Finally, the UCRA has worked with the TSSWCB on the paired watershed study of the East and West forks of Grape Creek (UCRA and TIAER, 2006). Both watersheds are approximately 25,000 ac. About 80 percent of the East fork watershed has been treated for mesquite and juniper, while the West fork watershed has had less than 300 ac treated. Since early 2005, the UCRA staff has been making occasional flow measurements with a portable H-flume or a portable current meter at selected sites on both forks following rainfall events. Unfortunately, these episodic measurements are not able to determine baseflow fluctuations and likely miss significant parts of the runoff events following storm events. These paired watersheds could provide a useful comparative study, but more continuous instrumentation for rainfall and streamflow monitoring would be needed.

2.2.2 Canadian River Municipal Water Authority

The CRMWA is primarily concerned with riparian saltcedar along the Canadian River channel and in the upstream end of Lake Meredith. Only one USGS stream gauge exists on the river, the Canadian River near Amarillo Station on Highway 87. The CRMWA staff does have a portable current meter to allow manual measurements at other selected locations. Shallow groundwater monitoring wells have been placed at several locations since 2005, and seven are currently operational. The purpose of the monitoring wells is to observe possible recovery of the water table elevations after saltcedar treatment.

2.2.3 Pecos River

Texas AgriLife engineers and scientists have taken leadership roles in saltcedar treatment along the Pecos River in West Texas. The primary purpose of saltcedar control in the Pecos River is an attempt to improve water quality (Gregory and Hatler, 2008), but studies by Hart et al. (2005) and Sheng et al. (2007) have addressed potential “water salvage” from brush control as well. Only three continuous stream gauges exist on the Pecos River, two operated by the USGS and one by the TCEQ’s Clean Rivers Program. Saltcedar treatment began in 1999 and continued into 2005. Delineating impacts on river flow from saltcedar treatment has been difficult due to uncertainties in flow interpretations as affected by releases from Red Bluff Reservoir. Hart et al. (2005) and Sheng et al. (2007) have estimated "water salvage" of 0.5 to 1 acre-ft/acre of saltcedar treatment from a paired site comparison, but they surmised that the water went to groundwater recharge instead of streamflow. The significance of the alluvial aquifer along the Pecos is conceptually significant based on its storage volume and water quality, and studies continue to try to quantify the interaction of surface and groundwater. Other cooperating agencies include the Environmental Protection Agency and the International Boundary and Water Commission.

2.2.4 Mesquite Creek on the Upper Colorado River

The TSSWCB's Upper Colorado Soil and Water Conservation District and their local NRCS counterparts have been qualitatively observing the effects of saltcedar treatment at the spring site that feeds Mesquite Creek in Scurry County, west of Gail. The headwaters of Mesquite Creek are well known at the edge of the caprock escarpment, where riparian saltcedar existed near the Seep Pond springs. Although the flow from the spring is relatively continuous, it was small enough that the downstream flow disappeared within a mile downstream on Mesquite Creek. About 48 ac of saltcedar at the springs and another 165 ac further downstream were sprayed in August 2005, and since that time the distance travelled by the spring flow down Mesquite Creek has tripled. No continuous hydrologic data are currently being collected.

2.3 Instrumentation for Rainfall and Other Weather Observations

Acceptance of the importance of hydrologic and weather monitoring to improve TSSWCB's water yield enhancement through brush control must also be informed by understanding the types of instrumentation available, their limitations, and their costs. This section was largely provided by Dr. John Schroeder, Associate Professor of Atmospheric Sciences, and Mr. Wes Burgett, Research Associate, who are leaders of the West Texas Mesonet Project at Texas Tech University. No vendors are endorsed, and all costs are estimates and intended for relative comparisons only. The final part of this section is a discussion of the current challenges of converting radar rainfall estimates into actual amounts of water.

2.3.1 Types of Rainfall Measurement Devices

National Weather Service Cooperative Rainfall Gauges (Graduated Cylinders)

These non-automated rain gauges are four inches in diameter and can hold up to eleven inches of rain before emptying. A small amount of lightweight oil and antifreeze keeps collected rainfall from freezing or evaporating. Gauges must be manually read and emptied as frequently as possible. These gauges are made of plastic and can be damaged by larger hailstones. Costs: $40 per gauge. Gauge is mounted on a wooden or metal stake away from any obstructions.

Tipping Bucket Gauges

These automated rain gauges are typically made of metal and can vary in diameter and size. A tipping bucket gauge uses a reed switch with a calibrated bucket assembly that tips with a certain amount of rain (generally 0.01 in or 1 mm). These gauges have a tendency to underestimate rainfall during heavy rain events when a lot of rain falls very quickly. Heating elements are available in some models that will allow snow and ice measurement. Real-time data are available when hooked to a datalogger or communication device. Periodic maintenance is required to remove dirt, leaves, insects, or other obstructions. Costs: $300-600 per gauge, higher with heating element (additional power also required).

Siphon Tipping Bucket Gauges

This gauge is similar to a standard tipping bucket rain gauge with the addition of a siphon feature above the tipping assembly. The metal siphon controls removal of rainfall from the assembly, allowing more accurate readings in heavy rain events. The siphon also helps to prevent debris from entering the rain gauge and blocking measurements. Real-time data are available when hooked to a datalogger or communication device. Maintenance is still required on a periodic basis, but can be extended longer than a standard tipping bucket gauge. Costs: $800-1,000 per gauge, higher with heating element (additional power also required).

Weighing Gauges

These automated rain gauges are larger and more rugged than a tipping bucket gauge. These gauges use the principle that a predefined volume of water weighs an exact amount. This type is the best rain gauge for heavy rain events, but it is less robust in very light rain. All of these gauges come with heating elements and do well in winter precipitation events. Weighing rain gauges are used by the National Weather Service for official rainfall observations at airports and other important climate stations. Older weighing gauges used an ink pen to record rainfall totals on a rotating drum of paper. Newer models have dedicated displays for real-time or historical data access. Maintenance is required, but not as frequently as a standard tipping bucket gauge. Costs: $4,000-9,000 depending on model and accuracy required.

2.3.2 Sensors for Weather Stations in a Watershed

Automated weather stations can provide valuable measurements of potential evaporation from a watershed. These stations are usually placed in remote areas where human measurements of evaporation are limited. A station's size and layout will vary depending on location and data required. A standard automated weather station is a 2-m (6-ft) tall tripod with sensors attached. Power is provided by solar panels that charge external batteries. Official climate stations at airports and mesonets use a 10-m (30-ft) tall steel or aluminum tower. Fire weather interests use a 6-m tall tower (20-ft) with a sensor array geared to fire weather. The following are sensors that are placed on automated weather stations to measure potential evaporation. Data from these sensors can be used to calculate ET for spraying and other agricultural activities.

Anemometer – This device is used to measure wind speed and direction at a certain height. There is a wide variety of anemometer types, ranging from simple three-cup with direction vanes to sonic anemometers. Generally, the higher the wind speed measured, the higher the evaporation rate.

A three-cup anemometer with directional vane is a simple and reliable mechanical device (if maintained) to measure average and peak wind speed with wind direction readings. These anemometers require periodic maintenance to replace bearings. New units are usually made of plastic and can be damaged by hail and ice events during the winter. These devices can underestimate high wind speeds due to mechanical limits of the bearings. Costs: $400-700 depending on model. Most units do not have a heating element, so they will freeze in icing events.

The propeller-type anemometer measures both wind speed and direction with one sensor. This type is a reliable sensor for long-term stability in measuring wind speed and direction, but it will underestimate wind speed in very light winds. Periodic maintenance is required, and bearings must be replaced at least once every several years. This unit is more rugged than a three-cup anemometer, but it can still break in large hail. Units can be purchased with heating element to work in icing events. Costs: $900-$1,100 depending on model (higher for heating element).

Sonic anemometers use changes in sound frequency to measure wind speed and direction. These sensors can sample wind data at a faster rate than a mechanical (bearing-limited) anemometer. Periodic calibration of the unit is required. Sonic anemometers can be damaged by hail or birds. These units are designed to work in icing events, and as such are used at major airports. Costs: $2,500-$9,000 depending on model.

Temperature/Relative Humidity Sensor – The majority of these sensors use a thermistor and a humicap device to give calibrated values of instantaneous or average temperature and relative humidity. These values can be stored for long-term analysis of climate in a location.

Thermistor Probe – A non-aspirated thermistor-type probe measures temperature only. These are low-cost units that have long-term stability. Thermistors are very sensitive to solar radiation and must be placed in radiation shields. These devices are relatively low-maintenance temperature probes, but have a high failure rate due to static or electrical discharges. Costs: $100-$200 depending on sensor response. Radiation shields costs: $150-$350 depending on number of shield plates and size of probe.

Temperature/Relative Humidity Probe – A non-aspirated probe provides temperature and relative humidity (in percent) measurements. A thermistor gives readings of temperature, while a humicap device in the probe provides relative humidity readings. These probes provide excellent stability of measurements with calibration required every two years. Periodic maintenance is required to clear dust and other debris from filters on the probe. Radiation shields are also required to minimize the impacts of solar radiation on the probe. Costs: $600-$800 depending on model type (radiation shields costs: $250-$350).

Aspirated Temperature/Relative Humidity Probe – An aspirated probes has a fan that provides a steady flow of air over the thermistor and humicap sensor. These fans are important in light wind events to help provide better stability in measuring temperature and relative humidity. Fans also require a significant amount of power with additional solar panels and batteries required to keep the sensor functioning. Costs: $600-$800 (aspirated fans run from $180 to $400 plus the radiation shield costs).

Solar Radiation sensors – A pyranometer measures incoming solar radiation. A net radiometer measures incoming solar radiation and outgoing earth radiation. Both of these units are valuable in determining the amount of solar radiation received at a location. A net radiometer is most valuable at night to measure outgoing earth radiation. Data from these sensors is valuable in determining daily ET rates for plants and agricultural interests. Higher ET rates can have significant impacts on brush spraying operations. Pyranometer Costs: $250-$3,500 (pyranometers come in many models. A simple silicon-type pyranometer will provide long-term stability measurements. Net Radiometer Costs: $800-$1,500 (size and stability determine price).

Rain gauges – Please see the earlier section for a detailed breakdown on rain gauges. Location of a rain gauge is as important as the type. Rain gauges mounted on towers or tripods can have missing totals due to obstructions. The best location for a rain gauge is slightly above ground level on a flat surface away from a tower. A full alter-type wind screen around the unit will provide more accurate rainfall measurements. The wind screen allows rain drops to fall into the rain gauge during high winds. Costs: $40-$9,000 depending on type (alter wind screens vary from $100-$600).

Optional Sensors – The following sensors may be useful for monitoring spraying and other interests in a watershed.

Soil Moisture – A water content reflectometer measures volumetric moisture at a pre-determined depth. These units can be valuable in determining how deep rainfall penetrates into the soil. These units have long-term stability, but are high maintenance due to problems with burrowing animals and lightning damage. Costs: $250-$400 depending on size and accuracy.

Soil Temperature – A thermistor-type probe measures soil temperature at a pre-determined depth. These units can be important in determining the depth of freezing temperatures, which would impact plant/brush growth in the growing season. Soil temperatures are also valuable for agricultural interests. Costs: $100-$300 depending on length of cable needed underground.

Leaf Wetness Grid – This low cost device simulates drop formation on a plant. A leaf wetness grid gives estimates of dew or rain drop formation/evaporation and coverage on a plant leaf. This sensor can be valuable in determining whether or not to spray brush. Costs: $100-$150 depending on length of cable needed.

Barometer – A barometer measures atmospheric pressure at a specific location. Barometric pressure readings can be valuable to pilots involved in spraying operations (derived altimeter readings). Costs: Analog Barometers $500-$800 with Digital Barometers from $1,400-$3,000.

Evaporation Pan – An evaporation pan requires human involvement to measure evaporation rates. These can be time consuming and are performed once a day usually in the morning. Estimates of the evaporation rate for the day are then given in hundredths of an inch. Costs are variable depending who is measuring the evaporation rates. Volume of the pan and pre-determined equations are very important in determining the actual rate of evaporation.

2.3.3 Automated Weather Stations

The actual price of a fully instrumented automated weather station will vary depending on the sensors and what communication options are used to send real-time data. In most cases, the communication option is determined by the location of the weather station. Communication options vary from landline phones to wireless internet to a satellite system. Hourly data collection should be considered to provide the best estimates of evaporation in real-time.

The following is a list of communication options and estimated recurring charges.

- Landline Phone: Modem, $250 with installation charges, 3-4 cents per call.
- Cell Phone Modems: Modem, $500 with activation charges, $45-65 dollar monthly fee.
- Wireless Internet, Serial Server, and Wireless Equipment: $800-$1,000, $40-$60 a month if through a private vendor.
- GOES Satellite: Equipment, $3,000-$4,000. No monthly charge for state or federal agencies, but limited data throughput, hourly transmissions only.

Additional options include using internet at schools or county government buildings and/or private sponsorship.

Next follows an estimated price list of fully automated stations with different designs and platform heights. Prices include the mandatory sensors with some optional sensors, and communication devices add a significant amount to the cost of each station.

- Standard 2-meter tripod system - $5,000-$9,000
- Fire-weather 6-meter tower system - $11,000-15,000
- West Texas Mesonet 10-meter complete station - $14,000-$19,000

2.3.4 Using Radar Data to Estimate Rainfall

Operational radar data can be acquired at minimal costs from the National Climactic Data Center and processed to estimate rainfall rates and totals over an area of interest. The National Weather Service operates a network of 10 cm wavelength Doppler radars (i.e. WSR-88D or NEXRAD radars) that provide coverage for almost all locations within the United States. The radars measure various parameters remotely using pre-defined scanning strategies. The typical scanning strategy turns the radar antenna through a complete 360° rotation at a given elevation angle or tilt (e.g. 0.5°). The elevation angle is then changed and another rotation is complete. This process continues through numerous elevation angles until a full volume scan is collected. A volume scan can then be used to evaluate the horizontal and vertical structure of precipitation within the observation domain given other radar limitations. A full volume scan typically takes approximately 5-6 minutes to complete, which means the re-visit time for any particular point in space within the radar domain is 5-6 minutes.

Radar estimates of rainfall are fundamentally based on radar reflectivity, which is a measure of the return energy back to the radar from the remotely intercepted radar targets (e.g. hydrometeors). Reflectivity is a function of many variables, but most importantly the backscatter from an individual spherical hydrometeor is a function of its diameter to the sixth power. Hence, the reflectivity for a particular radar volume is the integration of the backscattered power from all of the hydrometeors contained in that volume. This relationship is useful and serves as the basis for developing empirical relationships between reflectivity and rainfall. Hence, radar data has the advantages of providing good spatial coverage of reflectivity data at approximately 5 minute intervals that can then be related to rainfall rates.

Unfortunately, radar-estimated rainfall rates also have their disadvantages. It is easy to verify that two drop-size distributions will yield the same reflectivity, but maintain different volumes of water and hence different rainfall rates. To deal with this issue, statistical mean drop size distributions are used to develop the empirical reflectivity/rain rate relationships, but it is not uncommon for these relationships to over or under estimate the rainfall rates by a significant amount. The scientific literature is full of case studies comparing the two parameters in specific events using rain gauges as ground truth. In fact, multiple reflectivity/rain rate relationships are used operationally by the national weather service owning to fundamental differences between warm and cool season (i.e. convective versus frontal) rains in different locations. The problem is that no single empirical relationship fits all atmospheric events, even in the same location. One way to help negate this issue is to create feedback between the rain gauge measurements and the radar measured reflectivity. In a sense, allowing the available ground truth to help define a particular event's empirical relationship for a given location.

Radar-estimated rainfall rates also have other issues. While the WSR-88D network of radar provides good spatial coverage of the United States, the tilt of the radar beam and the earth's curvature ensure that as range increases, the measurements become more elevated above the earth's surface. This limitation does not significantly impact measurements near individual radars, but it can adversely impact measurements at larger ranges where a significant portion of the atmosphere resides below the lowest elevation angle.

Also, as the radar beam interacts with hydrometeors, a portion of the energy is backscattered. As this process continues down the radar beam, the energy continues to be attenuated, which reduces the radar's ability to sample distant targets. Attenuation is exacerbated in times of heavy rainfall or during hailfall, and can severely reduce the measured reflectivity in some portions of the radar domain. If this attenuated reflectivity is used to estimate rainfall, it could severely underestimate the true value.

Radars remotely sample volumes of the atmosphere due to their associated beam spreading and gate spacing as dictated by hardware limitations (e.g. antenna size and wavelength.). Hence, the acquired data maintains a specified spatial resolution that becomes coarser with increasing range. While frontal rain events can be rather homogeneous over large areas, convective rainfall events can provide significant gradients in rainfall (e.g. it is raining on one side of the street but not the other) that cannot be fully documented by the radar data given the limitations in spatial resolution.

Radar resolution, both temporal and spatial, and attenuation issues can be mitigated during a field study by installing or deploying shorter wavelength research radars to the location of interest. While this arrangement is advantageous for many reasons, it can be relatively costly to deploy mobile research radars. Research radars typically demand a significant "roll out" fee (~$2,000-$5,000 per month) to deploy, and then additional transmitter fees (~$100-$200 per hour) to operate. While these expenses may sound exorbitant, one must remember that various components on these radar systems can cost well over

$100,000 to replace, and installation of a stationary land-based radar system can easily reach over $500,000 depending on the selected wavelength and other factors. At the same time, data processing and evaluation relative to ground truth must still occur to derive the empirical relationship between reflectivity and rain rate.

In summary, radars provide great tools to study rainfall. The closer the area of interest is to the radar site, the better job the radar can do at estimating rainfall since attenuation and beam elevation issues are mitigated. Regardless, a significant number of ground truth rain measurements will always be needed to calibrate the radar reflectivity data for a given event. Precise estimates of actual rainfall depths and their spatial variation are still difficult and expensive to obtain, even for single events.

3. Estimation of Water Yield Enhancement in Areas with Saltcedar, Juniper, and Mesquite

3.1 Introduction and Summary

Changes in the water delivery characteristics of watersheds resulting from vegetative manipulations have been explored, studied, and reported for many years across a multitude of ecological settings. If there is one common thread in all of these reports, it is the fact that the results frequently are not consistent with expectations (Wilcox et al., 2008). Stated in a more practical vernacular, the response to a question "What will happen if?" should probably be, "It depends."

- It depends on all of the specific physical characteristics of a given watershed (such as geology, soils, topography, and land use),
- it depends on the sequence of meteorological events that may or may not lead to the generation of runoff,
- it depends on the general climatic conditions present on the watershed,
- it depends on the type and species of vegetation that is being manipulated,
- it depends on how the vegetation was manipulated (chemically, mechanically, by fire),
- it depends on the type and species of vegetation (if any) that replaces the one being "manipulated," and
- any reliable conclusions depend on having accurate water yield data before and after treatment upon which to base judgments as to the impact of the particular treatment.

In this report, we have been asked to provide summary information on the potential for water yield enhancement via vegetative manipulations involving three specific plant groups: saltcedar, mesquite, and juniper. Each of the following sections focuses on one of these groups while at the same time attempting to make comparisons between groups as appropriate. Research on saltcedar, as a plant that makes excessive use of water, is probably the most abundant. Research on mesquite, while not as prolific in terms of sheer numbers of studies, has probably been more comprehensive with respect to all of the plant's morphological and ecophysiological aspects. More research on various types of control mechanisms has been conducted on mesquite than on any of the other groups. The least research has been done with respect to water use by juniper, although geographic distribution of juniper is probably more extensive than either of the other groups. Having pointed out the fact that it is critical to consider all factors when making predictions as to water use by specific plants or water savings resulting from their removal, the following selected statements from each of the species-specific sections provide a reasonable summary of our current understanding for saltcedar, mesquite, and juniper. More detail is provided in the sections 3.2 and following in this chapter. Please note that the various authors chose their own units to express distance (ft or m), ET rates as lengths per unit time (such as in/d, in/yr, gal/acre/yr, mm/d, mm/yr), and plant water uses based on leaf area index (LAI) or stand area as volumes per unit time (such as gal/d, gal/yr, L/d, or L/yr), and sometimes mixed English and metric units.

3.1.1 Saltcedar

It has been reported that saltcedar can use 200 gal/d of water (Tribe, 2002), but this number has been questioned by many researchers (Wilcox et al., 2006; Owens and Moore, 2007). The peer-viewed scientific literature cited by Owens and Moore (2007) indicated that daily water use of an individual saltcedar tree is in the range of 0.4 to 57 L, or less than 15 gal/d (Davenport et al., 1982; Sala et al., 1996; Smith et al., 1996; Cleverly et al., 1997; Devitt et al., 1997; Wullschleger et al., 2001; Nagler et al., 2003). A variety of techniques have been used to estimate water use by saltcedar at the stand scale. Dahm et al. (2002) found that saltcedar stands on floodplains had higher ET rates than those in non-flooding areas (1,000 vs. 750 mm/yr). In the Virgin River of southern Nevada, Devitt et al. (1998) reported ET for saltcedar stands of 750 mm/yr during a dry year and 1,500 mm/yr during a wet year. On the landscape scale, Culler et al. (1982) estimated that water consumption by saltcedar stands was about 1,090 mm/yr along the Gila River in Arizona. When the phreatophytes were removed, subsequent measurements revealed that water savings came to 480 mm/yr after the replacement vegetation was established. In the Middle Rio Grande, Cleverly et al. (2006) found that a dense saltcedar stand frequently consumes up to 11.5 mm/d, especially when flooded; ET from other vegetation types seldom spikes so high. Conversion from a dense monoculture of saltcedar to a sparse saltcedar/saltgrass woodland was predicted to save 200 mm/yr (0.7 acre-ft/acre-year), based upon both ET and LAI changes in such a conversion.

3.1.2 Mesquite

It has been estimated that a mesquite tree in Sonoran Desert washes would transpire 15 gal/d (Nilsen et al., 1983). In another study by Ansley et al. (1998), they found that 5 years after mesquite density was reduced from 121 to 32 trees per acre, daily water use per tree increased from 13 to 44 gal/d. By using sap flow techniques, Dugas and Mayeux (1991) determined the total seasonal water use of 1,600 L per mesquite tree, or 2.8 gal/d based on a 150-day growing season. It is interesting to note that the reported value of 44 gal/d water use by a single mesquite tree is far greater than the maximum tree-level daily water use of 32.2 gal/d by saltcedar derived from sap flux measurement (Owens and Moore, 2007). By tracking changes in water content in a 1.5-m soil profile and surface runoff over a period of 7 years, Richardson et al. (1979) reported that following mesquite removal, ET was lower and soil moisture higher by 80 mm/yr, and runoff increased 30 mm/yr. In most Texas rangelands, most of the precipitation is retained in the upper 1 m of the soil profile where mesquite and herbaceous plants have similar root density (Weltz and Blackburn, 1995), and there is little deep drainage. Therefore, water savings from removing mesquite cover from these rangelands would be minimal except in the riparian ecosystems.

3.1.3 Juniper

Juniper changes landscape water balances for a plant community by intercepting a significant proportion of precipitation with its dense canopy and litter (Young et al., 1984; Thurow, 1991; Eddleman and Miller, 1992; Hester, 1996; Thurow and Hester, 1997; Lyons et al. 2006; Owens et al., 2006). The interception loss associated with the canopies of redberry juniper (*J. pinchotii*) and Ashe juniper was 25.9% and 36.7% of gross precipitation, respectively (Hester, 1996). Juniper is an evergreen, and therefore its canopy maintains a high interception potential throughout the year when compared to saltcedar or mesquite. Rainwater that passes through the canopy must also pass through the litter layer prior to entering the soil. The amount of interception loss associated with the litter layer is considerably greater for redberry juniper (40.1%) and Ashe juniper (43%) than for western juniper species (2-27% by Young et al., 1984; Thurow and Hester, 1997). As a result of interception loss via the canopy and litter, only 20.3% and 34% of annual rainfall reaches mineral soil under the canopy of Ashe juniper and redberry juniper, respectively. Owens and Ansley (1997) conducted research at various sites in the Edwards Plateau of Texas, and found that daily water use by redberry juniper and Ashe juniper was 46.8 and 33.1 gal/d, respectively. Hibbert (1979) estimated a 13-mm increase in runoff by controlling pinyon-juniper in the Colorado Basin. Dugas et al. (1998) estimated that removing woody plant cover reduced ET by 40 mm/yr for a period of at least two years. A recent study at the small-catchment scale by Huang et al. (2006) estimated that removal of juniper will increase streamflow by 46 mm/yr, representing about 5% of precipitation. A much higher water savings was reported in a study that was conducted at the Sonora Agricultural Experimental Station (Thurow and Hester, 1997). The soils at their research sites were 6 to 18 inches deep, which overlay a fractured limestore substrate. Their data indicate that substantial water yield can be achieved through conversion of pasture vegetation from juniper to grass dominance. Although the area received an annual precipitation of only 574 mm/yr, deep drainage occurred due to karst geology. The estimated deep drainage was 94 mm/yr in a 100% grass pasture as compared to 0 in a juniper/oak/grass community. This difference was largely caused by a three-fold greater interception loss in the juniper/oak/grass community. The water yield following juniper removal is equivalent to 100,500 gal/acre/yr. There was little runoff from these pastures, because the cut juniper maintained very high infiltration rates after the trees were removed. The moderately grazed pastures also had a good herbaceous cover in the juniper interspaces. Therefore, the added precipitation reaching the soil as a result of reduced interception losses did not runoff of the pasture but was instead channeled into the soil.

3.2 Water Use by Saltcedar

3.2.1 Distribution and Growth Habitats

Saltcedar (*Tamarix* spp.) is an invasive weed that occupies vast areas in New Mexico, Arizona, California, Nevada, and Texas. Saltcedar species are exotic phreatophytes, with deep roots tapping the water tables, that depend on groundwater for their water supply (Anderson, 1982). They grow mainly in riparian habitats, along stream channels and on floodplains. Saltcedar is capable of invading river banks and stream channels, replacing native phreatophytes and other native species, and forming solid dense stands. It is estimated that, in Texas alone, more than a half million acres are infested by saltcedar.

Unlike native phreatophytes such as cottonwoods and willows, saltcedar species also have extensive shallow root systems. Saltcedar seedlings can grow a root system over a meter deep in the first growing season and then grow up to 2 m by the end of the second growing season (Smith et al., 1997). Their adventitious roots easily develop from submerged or buried stems. About 60% of stem tissues produced new shoots/roots under greenhouse conditions (Brock, 1984). The dual root systems enable saltcedar to use soil water wherever it is available, thus they are facultative phreatophytes, or opportunistic water users. Because of rapid root growth and dual root systems, saltcedar seedlings have competitive advantages over seedlings of the native tree in soil water uptake.

Saltcedar species are extravagant water users and compete successfully with the native phreatophytes for limited water supply. In the Rio Grande basin, for example, native cottonwoods are declining in most areas, and half of the wetlands in the drainage were lost in just 50 years. Invasion by non-native phreatophytic trees such as saltcedar and Russian olive have dramatically altered riparian forest composition. Without changes in water management, exotic species will likely dominate riparian zones within half a century (Jackson et al., 2001).

When the water table drops below the root depths of the native obligate phreatophytes (cottonwoods, willows), these plants are severely stressed. For example, cottonwood prefers areas with groundwater less than 6.5 ft from the soil surface (Cleverly et al., 2006a). In contrast, saltcedar's primary taproot can easily penetrate 15 ft, or even grow down as deep as 40 to 50 ft (Tribe, 2002; Wilson et al., 2004), or 75 ft (Morrison, 2003). Once the taproot reaches the water table, secondary root branching becomes profuse (Di Tomaso, 1998). Unlike obligate phreatophytes, such as cottonwoods and willows, saltcedar is often able to survive under conditions where groundwater is inaccessible (Devitt et al. 1997b; Di Tomaso, 1998). Therefore, saltcedar water use is less affected by water table declines from drought or groundwater pumping thanks to its deeper rooting and effective use of summer rainfall by its shallow roots (Devitt et al., 1997a; Mounsif et al., 2002).

Water use by saltcedar trees has been a subject of debate. It has been reported that saltcedar can use 200 gal/d (Tribe, 2002), but this number has been questioned by researchers (Wilcox et al., 2006; Owens and Moore, 2007). It is highly unlikely that saltcedar can use that much water in a single day. Owens and Moore (2007) used three lines of evidence – peer-viewed scientific literature, sap flux rates and sap wood area, and potential ET rates – demonstrate the improbability that saltcedar, or any other woody species, can use this much water per tree on a daily basis. The peer-viewed scientific literature cited by Owens and Moore (2007) indicated that daily water use of an individual saltcedar tree is in the range of 0.4 to 57 L/d, or less than 15 gal/d (Davenport et al., 1982; Sala et al., 1996; Smith et al., 1996; Cleverly et al., 1997; Devitt et al., 1997a; Wullschleger et al., 2001; Nagler et al., 2003). The large discrepancy in daily water use in the literature could be related to plant canopy size, plant age, depth to water table, and environmental conditions. Depending on the habitat and plant age, saltcedar can grow as a small shrub or a big tree. A moderate estimate by Smith et al. (1996) was 15.9 L/d, or 4.2 gal/d. The transpiration rate and sap-flow area were 28% and 30% of that of Nagler et al. (2003), respectively, indicating a fairly good correlation. Based on a limited sample of sapwood area on the Rio Grande and Pecos Rivers and detailed sap flux estimates, Owens and Moore (2007) suggested that maximum tree-level daily water use derived from sap flux measurement would be less than 122 L/d (32.2 gal/d).

Some researchers believe that saltcedar water consumption is twice as much as native phreatophytes. Zavaleta (2000) stated that on average, *Tamarix* stands consume 3000 to 4600 m^3/ha/yr more water than the native vegetation that they replace. According to Zavaleta (2000), marginal water losses to *Tamarix* are comparable to annual precipitation totals, which remain below 4500 m^3/ha/yr (450 mm annual precipitation) throughout the invaded region. Following root plowing in the Pecos River floodplain, Weeks et al. (1987) estimated that water use by saltcedar was 300 mm more than the replacement vegetation. Still, water use comparisons between saltcedar and the native phreatophytes are far from conclusive. A literature review by Hays (2003) summarized daily water use by saltcedar as between 1.6 and 16.3 mm/d with a mean of 7.9 mm/d, which is equal to, or sometimes in excess of, water use for other riparian woody vegetation. Assuming 180-day growing season, the annual ET averages 1422 mm/yr. Dahm et al. (2002) found that ET rates from a dense stand of saltcedar along the Rio Grande were 1100 to 1200 mm/yr, which is comparable to a cottonwood-dominated community with an understory of saltcedar and Russian olive (1200 mm/yr). Cleverly et al. (2002) summarized results from

18 studies in saltcedar research and showed that maximum daily transpiration is 20 mm/d for an individual saltcedar plant, which is within the range for other native woody species (Scott et al., 2004; Nagler et al., 2005). A recent work by Nagler et al. (2003) indicated that saltcedar uses similar amounts of water as the native phreatophytes (cottonwoods and willows) of similar canopy size. The average daily water consumption of a saltcedar tree (4 to 5 m tall) is about 57 L/d, or 15 gal/d (Nagler et al. 2003). The data of Nagler et al. (2003) are consistent with Busch and Smith (1995), who found water uptake of saltcedar to be equal to that of native woody plants along the Colorado River. Cleverly et al. (2006a) reported that both saltcedar and native cottonwood trees in the Rio Grande transpire large quantities of water under favorable environmental conditions. However, in a Mojave Desert floodplain, Cleverly et al. (1997) reported that the native willow trees transpired more water per unit leaf surface area than saltcedar. Sala et al. (1996) indicated that the transpiration rate of saltcedar on a unit leaf area basis was similar to those of native phreatophytes, but transpiration on the whole plant basis was higher because of the larger LAI of saltcedar. Smith et al. (1998) also reported that leaf-level transpiration was not different between saltcedar and native phreatophytes, but sap-flow rates per unit sapwood area in saltcedar were higher than in the natives, suggesting that saltcedar maintains a higher leaf area than the natives.

Higher LAI and stand density would result in higher ET in saltcedar. Davenport et al. (1982) compared ET of saltcedar based on stand density, and found water use of 2 mm/d for a sparse stand and nearly 16 mm/d for a dense stand. It is estimated that annual ET over a saltcedar stand along the middle of Rio Grande reach was 570 mm/yr; and the ET almost doubled in a much denser stand (Dahm et al., 2002).

In riparian ecosystems, does saltcedar always have higher LAI than the native phreatophytes? A recent report by Nagler et al. (2005) pointed out that current evidence does not support the conclusion that saltcedar has unusually high ET rates or LAI that would allow it to desiccate water courses. By combining remote sensing and in-situ measurements to estimate ET from riparian vegetation over large reaches of western US rivers, Nagler et al. (2005) found that cottonwood and willow stands generally had the highest annual ET rates (1100-1300 mm/yr), while mesquite (400-1100 mm/yr) and saltcedar (300-1300 mm/yr) were intermediate, and giant sacaton (500-800 mm/yr) and arrowweed (300-700 mm/yr) were lowest.

Glenn and Nagler (2005) believe that the ecophysiological traits of saltcedar make it a formidable competitor of the native vegetation, and eventually the dominant species and largest water user in the riparian ecosystems. Saltcedar is able to survive severe water deficits (Cleverly et al., 1997; Devitt et al., 1997b; Di Tomaso, 1998; Smith et al. 1998). Cleverly et al. (1997) found that xylem sap flow in saltcedar was higher than three co-occurring native tree species under drought conditions. Saltcedar was the most drought tolerant and willow the least drought tolerant among four phreatophytes. They concluded that as floodplains in the Mojave Desert become more desiccated with age, saltcedar assumes greater dominance due to its superior drought tolerance over native phreatophytes and its ability to produce high density stands and high leaf area.

3.2.2 How Much Water Can a Saltcedar Plant Use?

Water use by saltcedar could vary greatly depending on growth habitats, soil moisture availability and atmospheric demand. Anderson (1982) reported that saltcedar exhibits effective stomatal control of water loss when exposed to high temperature and low humidity. Mounsif et al. (2002) reported that both stomatal conductance and photosynthesis in saltcedar declined significantly during a drought year as compared to a wet year, even when the water table was relatively high (< 2.74 m). During a drought year, ET rates of saltcedar declined dramatically over a 60-day period from 11 mm/d to <1 mm/d (Devitt et al., 1998). Therefore, saltcedar has the potential to be both a high water user and a low water user. The effective stomatal control mechanism enables saltcedar to prevent excessive water loss and increase water use efficiency. Anderson (1982) stated that "failure to treat stomatal resistance as a variable in attempts to predict ET from meteorological data and stand characteristics may result in significant overestimates."

Saltcedar can be subjected to significant soil water deficits and still respond rapidly (within 24 hours) to surface irrigation (Devitt et al., 1997a). Saltcedar has extraordinary ability to take up water from the unsaturated soil profile. Under typical hot, dry summer conditions, saltcedar would effectively utilize water from most summer rainfall events to maintain photosynthesis (Devitt et al., 1997b; Mounsif et al., 2002). In contrast, obligate phreatophytes depend entirely on groundwater and do not use summer precipitation (Flanagan et al. 1992). In riparian habitats, high ET rates of saltcedar can lower the water table in heavily infested areas and make groundwater less available to the native phreatophytes (Di Tomaso, 1998). Nagler et al. (2003) reported that during the non-stress part of their experiment, canopies of saltcedar, cottonwood, and willow had similar rates of ET, but saltcedar maintained higher ET than the native trees on the stress treatment.

Saltcedar water use is affected by the depth to water table. Water use by saltcedar peaks when the water table is less than 2 m below the soil surface, then decreases rapidly and stabilizes at water table depths > 4 m (Great Western Research, 1989). Hays (2003) reported that saltcedar along a Colorado River site was characterized by dense young stands growing in a floodplain with a water table greater than 20 ft deep, and ET was estimated at 500 mm during the growing season. During the same growing season, dense mature saltcedar growing along the banks of Pecos River in a 5 to 10 ft water table depth showed an ET estimate of 2300 mm. Comparatively, a dense, mature infestation of saltcedar and Russian olive along the Canadian River growing into a water table less than 3 ft deep evapotranspired 4100 mm during the year. Van Hylckama (1974) also reported that saltcedar annual water use was 950 mm/yr at the groundwater depth of 2.7 m, and 2150 mm/yr at the depth of 1.5 m.

Although saltcedar water use is generally affected by depth to the water table, studies have shown that some *Tamarix* species can maintain high transpiration rates even with declining water tables. Cleverly et al. (2006a) recently reported that in the Rio Grande basin, ET from a dense *Tamarix chinensis* thicket did not decline with increasing groundwater depth; instead, ET increased by 50%, from 6 mm/d to 9 mm/d, as water table receded at nearly 7 cm/d. LAI of the saltcedar thicket, likewise, increased during groundwater decline. When saltcedar and Russian olive were removed, water salvage through reduced ET was 260 mm/yr in relation to ET measured at a reference site.

Saltcedar trees transpire more water when growing in close proximity to rivers or streams than when growing farther away. Devitt et al. (1997b) found that sap flow on a unit leaf area basis was higher for saltcedar plants growing along the river's edge, with midday hourly values significantly higher when a water table was present. In a dry-down phase, the sap flow decreased in the river's edge and reached zero in the open stand lysimeter as the water table dropped. However, recent reports seem to indicate that saltcedar can maintain high transpiration rates even when grown away from rivers or streams. Mounsif et al. (2002) reported that stomatal conductance in saltcedar trees grown in close proximity to a pond did not differ from the trees growing 60 m away from the pond. Nagler et al. (2006) used remote sensing methods and ground surveys to characterize the stand structure and ET of three large (1 km^2 plots), dense stands of saltcedar on the lower Colorado River in California. The plots were 200 m (Plot 1), 800 m (Plot 2), and 1600 m (Plot 3) away from the river channel, and water table depth was 3 m for Plot 1 and 3.7 m for Plots 2 and 3. LAI for individual trees averaged 5 for all plots. Annual ET was 1600, 1900, and 1800 mm/yr for Plots 1, 2, and 3, respectively. These values are close to annual ET of 1680 mm in dense saltcedar stands along the lower Colorado River in Arizona estimated by Gay (1986). These ET rates compare to annual rates of 800 to 1200 mm/yr measured for the Middle Rio Grande (LAI 3-3.5) and 800 to 1000 mm/yr for mixed arrowweed/saltcedar stands on the lower Colorado River. Thus, by rooting deeper to reach for groundwater, saltcedar can grow farther away from river banks, still have high transpiration rates, and colonize large areas along the river channels. In Nagler et al.'s (2006) study, the water table 1.6 km from the river bank was only 0.7 m lower than in the bank proximity. Once a saltcedar's tap root hits the water table, lateral roots would grow horizontally and spread within the capillary fringe.

A variety of techniques have been used to estimate water use by saltcedar at the stand scale, including sap flow measurements, groundwater monitoring, large-lysimeter measurements, remote sensing, and micrometeorology. By using eddy covariance to estimate season-long ET along the middle of Rio Grande River, Dahm et al. (2002) found that saltcedar stands on floodplains had higher ET rates than those in non-flooding areas (1000 vs. 750 mm/yr). In the Virgin River of southern Nevada, Devitt et al. (1998) reported ET for saltcedar stands of 750 mm/yr during a dry year and 1,500 mm/yr during a wet year.

3.2.3 Stress Tolerance and Extreme Adaptability of Saltcedar

Mature saltcedar plants are tolerant of a variety of stress conditions, including heat, cold, drought, flooding, and high salinity (Di Tomaso, 1998; Smith et al., 1998; Glenn and Nagler, 2005). These ecophysiological traits enable them to develop dense monocultures that replace native vegetation; as stand density and plant size increase, so does water use. Saltcedar accumulates salt in its leaf glands that is then transferred to the soil when plants drop their leaves. Increased soil salinity under saltcedar stands impairs germination and establishment of many native species. In the meantime, saltcedar seedlings can rapidly colonize moist areas after summer rains. Morrison (2003) observed that saltcedar can live in soils 25 times saltier than either willows or cottonwood can stand. It moves salt from the bottom of its 75-ft to 100-ft rooting depth to the soil surface. Over time, these accumulated salts may kill any other plants below or around it. It was reported that saltcedar can tolerate salt content of 8,000 to 10,000 ppm (Di Tomaso, 1998; Nagler et al., 2006), which inhibits growth of competing species. Saltcedar exhibits near maximum photosynthesis and growth up to 36,000 ppm NaCl, whereas willow and cottonwood showed rapid declines in these parameters at only 1,500 ppm (Smith et al., 1997). Robinson (1965) found that salt exudation enables *Tamarix* plants to tolerate saline soils in Death Valley, California, of up to 50,000 ppm of salt. In comparison, salt content of seawater is about 35,000 ppm. Walker and Smith (1997) pointed out that the most single important way the invasion of saltcedar fundamentally alters ecosystems is through salinization of floodplain habitats. Therefore, they suggested that in many ecosystems being reclaimed from saltcedar invasion, only a return of annual floods, which leach the soil of salts, will allow the ecosystem to be re-vegetated with former dominants such as cottonwood and willow. Although dense stands of saltcedar can increase soil salinity, water use of saltcedar is also affected by increased salinity levels. By comparing flushed versus non-flushed evapotranspirometers, van Hylckama (1970) showed that in the treatment where salt was removed by flushing the system with fresh water, saltcedar used 2,290 mm water, twice as much as in the non-flushed system.

Mature saltcedar can survive inundation for 98 days when root crowns were submerged in still water (Warren and Turner, 1975). The ability to survive inundation is an attribute that allows saltcedar to be well adapted to aquatic sites or sites periodically inundated during the growing season. Tallent-Halsell and Walker (2002) found neither saltcedar nor willow could withstand an extended period of inundation (105 days). However, more willow died in the water saturated soil than saltcedar. Saltcedar also grew more rapidly than willow under saturated conditions, suggesting saltcedar is well-suited for vigorous establishment or recovery after flooding recedes. For example, saltcedar biomass was 2.5 and 3.5 times greater than that of willow under saturated and drawn-down treatments, respectively. Tallent-Halsell and Walker (2002) concluded that whenever suitable land is released from flooding, saltcedar seeds and shoots may rapidly colonize the exposed shores because they can outgrow the native phreatophytes.

3.2.4 Water Yield from Saltcedar Control

Water salvage estimates show a significant reduction in system water loss after saltcedar treatment (Culler et al., 1982; Weeks et al., 1987; Hays, 2003; Bawazir et al., 2006; Groeneveld et al., 2006; Cleverly et al., 2006b). Clearing high density saltcedar stands has greater effect on water salvage than treating low density stands (Hays, 2003). Hays (2003) did a paired analysis between herbicide treated and untreated plots in the Colorado River basin and found potential water savings of 400 mm/yr, based on the assumption of 49% mortality with top kill of saltcedar. From a before-after comparison, Groeneveld et al. (2006) estimated water savings of 3.1 acre-ft/acre on approximately 6,000 treated acres along the Pecos River, and the annual salvage came to 18,600 acre-ft. Bawazir et al. (2006) investigated water salvage by chemically controlling saltcedar at the Elephant Butte Delta of New Mexico, and found that estimated ET for non-treated saltcedar was 1002 mm when compared to measured ET of 386 mm at the treated site, a difference of 61% for 189 days. This difference was close to the 57% decline for 83 days determined by direct measurements during the April to May growing season. Water salvage estimates in all of the previous literature were probably overestimates because the studies did not account for ET losses from the replacement vegetation.

More realistic estimations on water salvage should be based on data collected after re-establishment of the native vegetation. On the landscape scale, Culler et al. (1982) estimated that water consumption by saltcedar stands was about 1090 mm/yr along the Gila River in Arizona. When the phreatophytes were removed, subsequent measurements revealed that water savings came to 480 mm/yr after the replacement vegetation was established. In the Middle Rio Grande, Cleverly et al. (2006b) found that a dense saltcedar stand frequently consumes up to 11.5 mm/d, especially when flooded; ET from other vegetation types seldom spikes so high. Conversion from a dense monoculture of saltcedar to a sparse saltcedar/saltgrass woodland is predicted to save 200 mm/yr (0.7 acre-ft/acre-year), based upon both ET and LAI changes in such a conversion.

Some field studies by the USGS indicated that measurable water salvage following saltcedar clearing is only 0 to 1.5 acre-ft/yr due to ET of replacement vegetation, increased evaporation, loss to ground water, or other sinks (Culler et al., 1982; Weeks et al., 1987; Shafroth et al., 2005). A large-scale saltcedar control program, which was initiated in the Pecos River of New Mexico in 1967 and spanned 15 years, revealed no change in streamflow as a result of saltcedar clearing (Welder, 1988). This finding was in sharp contrast to the companion study of Weeks (1987), who found that water use by saltcedar at the stand scale was 300 mm/yr greater than water use by replacement vegetation. It is possible that rapid re-growth of saltcedar and a buffer of untreated saltcedar immediately adjacent to the river was sufficient to maintain ET at high levels following treatment in Welder's (1988) study (Wilcox et al., 2006).

As suggested by Wilcox et al. (2006), the fundamental controlling factor for water salvage from brush control seems to be the availability of groundwater. It is apparent from Hays' (2003) study that brush control on some habitat types will yield considerably more water than others. In Texas, the regions with highest potential for water salvage from brush control are riparian ones dominated by saltcedar (Wilcox et al., 2006). One such region in Texas is along the Pecos River, despite the fact that no change in streamflow after saltcedar treatment was observed by Welder (1988) in the Pecos River basin in New Mexico. Preliminary results from an ongoing project in the Pecos River in Texas (Hart et al., 2004) indicate that there is a great potential for water salvage by saltcedar control, even though no fixed values were obtained for the amount of water salvaged. Once saltcedar is replaced by non-woody native species, given less leaf area, shallow rooting depths, and shorter seasons of active growth, less water would be used (Wilcox et al., 2006). In Nagler et al.'s (2005) study, if a dense saltcedar stand is removed and replaced by giant sacaton and arrowweed, water savings would be about 500-600 mm/yr.

3.3 Water Use by Mesquite

3.3.1 Distribution and Growth Habitats

Mesquite (*Prosopis* spp.) is a group of trees and shrubs that are widespread throughout the world. Mesquite is recognized as a rangeland invader in the southwestern United States. It has a wide distribution, from the semiarid high plains of Texas to the Sonoran, Mojave, and Chihuahuan Deserts. Depending on the growth habitats and local climate, mesquite can grow as a shrub or a tree. On the uplands of the Chihuahuan Desert, it assumes a shrub-like form, whereas in perennial water courses in the Sonoran Desert, mesquite grows as a tree. In the arid shrublands of California, characterized as Mediterranean-type climate (Hibbert, 1983), precipitation occurs mostly in the winter when transpiration is low, allowing for deeper drainage (Seyfried and Wilcox, 2006). In these regions, mesquite exploits groundwater by growing a tap root (Philips, 1963; Nilsen et al., 1981), and it is an obligate phreatophyte. Because mesquite has a permanent water supply, it is presumably able to tolerate drought and maintain high leaf area during the dry season, and it is a "water spender" (Levitt, 1980). However, in the semiarid grasslands of Texas and the warm deserts of Mexico where most precipitation occurs in the summer, and the water table is usually inaccessible, mesquite mostly relies on its lengthy shallow lateral roots to grow (Heitschmidt et al., 1988; Ansley et al., 1990), and it is more like a facultative phreatophyte (Thomas and Sosebee, 1978).

Water relations and water use by mesquite in the Sonoran Desert area have been studied extensively by Nilsen et al. (1981, 1983, and 1987). At a study site with a 4 to 6 m deep water table, mesquite exhibited very high transpiration rates and productivity. Nilsen et al. (1987) thus claimed that mesquite productivity and water use are decoupled from the natural precipitation. In these desert environments, mesquite exhibited extraordinary ability in growing its tap root deep into the soil profile. A tap root as deep as 53 m was reported by Philips (1963). It was estimated that a mesquite tree in the Sonoran Desert washes would transpire as much as 15 gal/d (Nilsen et al., 1983). Because the plant can maintain high transpiration, the mesquite woodlands in the Sonoran Desert had the highest recorded productivity of any vegetation type in the North American deserts (Smith et al., 1997).

In the Sonoran Desert, mesquite exhibits a clear zonation in growth forms depending on depth to the water table (Sharifi et al., 1982). In a lowland area where the water table was 5 m deep, mesquite grew as a tall tree, but it became a shrub where groundwater dropped to 12 m deep. Similar results were reported by Stromberg et al. (1992), in which velvet mesquite responded to a declining water table by growing smaller stature and smaller leaflets, and shed leaves during a drought period. However, even as the water table dropped past 25 m deep, velvet mesquite still grew to 7 to 8 m in height, indicating the extraordinary ability to exploit deep water sources by this phreatophyte.

In areas where most annual precipitation occurs as summer rainfall, deep drainage is unlikely to occur because immediate evaporation from soil surfaces reduces amounts of drainage, and also because of the changes in rooting patterns between woody and herbaceous species. Woody species such as mesquite tend to be more shallowly rooted in climates with summer rainfall regimes, as compared to more deeply rooted in climates with substantial winter precipitation (Schenk and Jackson, 2002). Consequently, mesquite growing in the upland on Texas plains utilizes water from the unsaturated soil horizons. Dugas and Mayeux (1992) compared sap flow of mesquite from west Texas during the wet (after a 20-day period of high rainfall) versus the dry season (after several months without rainfall). They found that sap flow was 62% higher when soil was wet than dry, suggesting these plants rapidly utilized surface moisture when available.

In semiarid west Texas rangelands with an annual precipitation of 450 mm, an argillic horizon has developed in the soil. The argillic horizon is rich in clay content (35-37%), which restricts the depth of water percolation. The wettest soil layers on these rangelands usually occur at depths of 60 to 75 cm during the growing season, and the water table is often more than 10 m deep. These impenetrable argillic horizons also restrict root growth. Therefore, the plants often have less developed tap roots. The majority of mesquite roots grow in the upper 60-cm soil profile, although 40% of roots were distributed below 67-cm depths in regions with higher precipitation (Heitschmidt et al., 1988). In ecosystems where the water table is beyond exploitation of the deep roots, mesquite trees often respond rapidly to moisture in the upper soil layers with their extended shallow lateral roots (Easter and Sosebee, 1975; Thomas and Sosebee, 1978; Brown and Archer, 1990; Wan and Sosebee, 1991; Ansley et al., 1991). Lateral roots of mesquite can extend 30 ft or more from the tree center, and most of them are distributed 30 cm below the surface, a little deeper than grass roots (Ansley et al., 1991). Rapid water uptake by mesquite from the 60-cm soil profile following summer precipitation led to more than three times higher transpiration rates in the rainy season as compared to the dry season (Wan and Sosebee, 1991). This condition suggests that mesquite lateral roots used rainwater very effectively. When lateral roots of mesquite were severed, the whole plant leaf area was reduced by 50% in the first growing season as compared to the non-severed plants (Ansley et al., 1991).

3.3.2 How Much Water Can a Mesquite Plant Use?

How much water can mesquite trees transpire? In a Mojave Desert floodplain, Sala et al. (1996) found that *Prosopis pubescen*s (Benth) transpired 4.5 L/d, or 1.2 gal/d. In the Sonoran Desert perennial water course, daily mesquite transpiration was 15 gal/d (Nilsen et al., 1983). On upland sites at Vernon, Texas, Ansley et al. (1991) found that leaf transpiration rate on a sunny mid-summer day is about 227 grams of water/ft^2 of leaf area/d. A typical 12-ft mesquite tree in a dense stand has about 130 ft^2 of leaf area (11.7 m^2). The calculated water use per day would come to 8 gal/d per tree in a dense stand (200 trees per acre). This result was based on leaf chamber measurements, and was confirmed by sap flow measurement for small stems and calculated on a per tree basis (Ansley et al., 1994). The total water use per year by a mesquite stand represents 32% of annual precipitation (660 mm). When mesquite stand density declined to 120 trees per acre, water use per tree increased to 13 gal/d, and annual water use per acre showed little change, as 31% of annual precipitation was used by mesquite. This finding is in sharp contrast to the water use pattern of saltcedar in a riparian ecosystem by Dahm et al. (2002), who showed annual ET over a saltcedar stand along the middle Rio Grande reach was 570 mm/yr; and the ET almost doubled in a much denser stand. Since the lateral roots of mesquite in west Texas rangelands can extend 30 ft from the tree center, the denser the stand, the less water was available to each individual tree, resulting in lower water use per tree (from 13 to 8 gal/d).

Transpiration data at the leaf level by Ansley et al. (1991) seems to agree with Wan and Sosebee (1991). The latter reported a seasonal average transpiration of 3.28 mmol/m^2/s from trees growing on a sandy loam soil in Lubbock, Texas. That rate is equal to 2.33 L/m^2 leaf area assuming 11 hours transpiration. As those trees were much smaller (about 6 to 7 ft tall) than those in Vernon, Texas, and the leaf area was only about 3.3 m^2, the daily water use per tree would be about one quarter of that reported by Ansley et al. (1991), or 2.04 gal/d. For a mesquite stand of 300 trees per acre, the total annual water consumption would be 85,700 gal/acre, or about 18% of annual precipitation. By using sap flow techniques, Dugas and Mayeux (1991) determined the total seasonal water use of 1600 L per mesquite tree, or 2.8 gal/d based on a 150-d growing season. In their study site in Throckmorton, Texas (annual precipitation 600 mm), the mesquite crown diameter was about 2 m, and the leaf area was 7.5 m^2 for the sap flow measurement. The plant transpiration for 1 m^2 leaf area was 106 L, which was comparable to Nilsen et al. (1991) study.

Compared to saltcedar's high LAI of 3 to 5, mesquite has much less LAI (1 to 1.5). This range may explain higher transpiration rates of saltcedar compared to mesquite on a tree basis (Sala et al., 1996). Mesquite is well adapted to a semiarid climate through adjusting its leaf area and leaf level transpiration rates. Ansley et al. (1991) reported that when lateral roots of mesquite were severed, leaf area was reduced by 50% in the first growing season. However, because the transpiration rates per unit area of the remaining leaves increased significantly, whole plant transpiration was not different between root-severed and non-severed plants before and after leaf abscission during the second growing season. This result may suggest that remaining unsevered lateral roots can regenerate secondary roots rapidly to compensate for the reduced water-uptake capacity after root severing. Because mesquite trees can adjust total leaf area and unit leaf transpiration rates to match the soil water availability, water use at the single tree level is not an important indicator for total water use at the stand level. For example, in Ansley et al.'s (1998) study, a high density stand (200 trees per acre) consumed about the same amount of water as a lower density stand (120 trees per acre), although the daily water use was different on a per tree basis. When stand density dropped to 32 trees per acre, the daily water use per tree was as high as 25 gal/d, which was three times the daily water use in the high density stand. The amount of annual precipitation should be considered when comparing mesquite water use in different environments. Mesquite trees in Lubbock, Texas, transpire much less water than trees in Vernon, Texas, which had 44% more precipitation, and where more rainwater would percolate into the deeper soil layers and could be used by mesquite tap roots.

For upland mesquite trees, daily water consumption of 25 gal/d appears to be very high when compared to 15 gal/d water use by trees growing along desert washes in California where groundwater was within the reach of plant deep roots (Nilsen et al. 1983). Even when the longer growing season in California (214 days) is accounted for, total seasonal water use per tree was still higher for Texas upland mesquite trees based on a 140-day growing season (25x140=3500 gal vs. 15x214=3210 gal). This value is also substantially higher than daily water use (15 gal/d) by similar sized saltcedar (4 to 5 m tall) growing in a riparian habitat (Nagler et al. 2003). For a tree to transpire 25 gal/d, it must have a leaf area of 416 ft^2, or 37.5 m^2. Even for the smaller trees with 130 ft^2 leaf area in Ansley et al.'s (1991) study, the daily water use was still 2.8 times higher than for trees in the 600-mm rainfall region (Dugas and Mayeux, 1991). In west Texas rangelands, mesquite stands usually have much higher density, ranging from 300 to 1000 trees per acre (Sosebee, 1980). Therefore, the reported water consumption of 25 gal/d probably does not represent a "typical" daily transpiration rate for mesquite trees growing in west Texas rangelands.

In another study by Ansley et al. (1998), they found that 5 years after mesquite density was reduced from 121 to 32 trees per acre, daily water use per tree increased from 13 to 44 gal/d. They attributed this change mainly to canopy size increase, although leaf level transpiration was also slightly higher. This finding further indicates how vigorously the plant grew its lateral roots to capture soil moisture. It also suggests that intra-species competition between adjacent trees would get stronger when more trees invade into a pasture. When this competition weakened through stand thinning, the remaining trees can grow much more aggressively. On the other hand, stand thinning created opportunity for grasses to fill in the inter-tree spaces (Dahl et al., 1978). Therefore, after stand thinning, total water use by mesquite on a unit area basis would be less, which was indeed the case in Ansley et al.'s (1998) study in which trees in the low density stand consumed 197,000 gal/acre annually, as compared to 220,000 gal/acre in the higher density stand. It is interesting to note that the reported value of 44 gal/d water use by a single mesquite tree is far greater than the maximum tree-level daily water use of 32.2 gallons by saltcedar derived from sap flux measurement (Owens and Moore, 2007).

3.3.3 Rooting Characteristics and Water Use Patterns in Different Environments

The following statements are a summary of mesquite rooting patterns and transpiration that have been observed in different environmental conditions.

1. <u>Sonoran Desert washes, characterized by a Mediterranean-type climate, where most precipitation occurs in the winter, and there is deep drainage.</u> Mesquite primarily uses groundwater by its taproot from the water table of 5 m or deeper. The trees assume shrubby forms when the water table dropped to 25 m. Transpiration and productivity are decoupled from annual precipitation (Nilsen et al., 1981, 1983, 1987).

2. <u>Chihuahuan Desert environment where annual precipitation is between 200 to 300 mm.</u> Mesquite roots can penetrate through calcic and petrocalcic horizons, accessing soil water from soil horizons where recharge is infrequent but may be crucial in sustaining plants during extended drought (Gibbens and Lenz, 2001). Because mesquite root density is lower than grasses in the upper soil layers, mesquite may not respond to small or moderate summer rainfalls. Montana et al. (1995) reported that plant water potentials of mesquite were little affected by irrigation that percolated to soil depths <40 cm, while the warm-season tobosagrass (*Hilaria mutica*) responded strongly to irrigation and small rain events. However, mesquite trees responded quickly to large irrigation pulses (35 mm), and the magnitude of this response is greater on coarse-textured than fine-textured soil (Fravolini et al., 2005). Molinar et al. (2002) also found that mesquite dominated on deep sandy soils that facilitate downward water infiltration and retain little moisture near the soil surface, whereas black grama (*Bouteloua eriopoda*) was most productive on shallow loamy or shallow sandy soils.

3. <u>Semiarid rangelands in west Texas, on range sites where annual precipitation is about 450 mm, such as Lubbock, Texas.</u> The soils commonly develop argillic or petracalcic horizons at depths of 75 to 100 cm (and sometimes <75 cm). The highest water content throughout the growing season occurred at the 60-cm to 75-cm soil depth, or just above the caliche layers. Precipitation of carbonate begins at a depth that is correlated with the mean annual wetting of the soil profile (Arkley, 1967). Therefore, the 60-cm soil depth represents the typical soil horizon to which most rainwater is percolated and where most precipitation in the winter is stored. Mesquite roots are distributed mainly in the upper 60-cm soil layers, and can respond rapidly to summer rainfalls and compete successfully with grasses for topsoil moisture (Wan and Sosebee, 1991). This root distribution pattern is similar to that in the Patagonian arid grassland where maximum root density of shrubs was found at the 55-cm depth (Fernandez and Paruelo, 1988). The tap root of mesquite is less developed and is only important when plant survival is threatened during an extended drought; and, the plant's transpiration rate is mainly determined by water uptake from the lateral roots (Thomas and Sosebee, 1978).

4. <u>Texas range sites with higher annual precipitation (about 700 mm).</u> Patterson (1990) suggests that long-term mean infiltration depth is approximately 2.3 m in a climate with about 700 mm precipitation. This finding compares well with the geometric mean rooting depth of 2.1 m in a climate with 650-700 mm precipitation (Schenk and Jackson, 2002). Mesquite trees are more

deep rooted on these range sites. About 40% of mesquite roots are below 67-cm depths (Heitschmidt et al., 1988). When mesquite trees were exposed to drought, more large-diameter roots developed in soil layers between 2 and 2.7 m (Ansley et al., 2007). During a drought period, mesquite transpiration was 100% of total ET of the vegetation, and this ratio dropped to 20% after a heavy rainfall (Dugas and Mayeux, 1991), indicating mesquite roots dominated water uptake from the lower soil horizons during a drought. Mesquite water use is usually much higher here than in the lower precipitation west Texas regions. The classic two-layer model proposed by Walter (1974), in which herbaceous plants use moisture from the surface soil and woody species exploit soil water beneath the grass roots, is more appropriate in Texas rangelands and the Chihuahuan Desert rangeland.

5. <u>Semiarid riparian ecosystems where mesquite shrubs or trees mainly exploit groundwater source.</u> Scott et al. (2006) found that growing season ET totals were 407, 450, and 639 mm in the grassland, shrubland, and woodland, respectively, and in excess of precipitation by 227, 265, and 473 mm. This excess was derived from groundwater, especially during the extremely dry pre-monsoon period when groundwater was the only source of moisture available to plants. Like the mesquite trees in the Sonoran Desert washes (Nilsen et al., 1981), water use by mesquite in this riparian habitat also exceeded local precipitation by a wide margin. Decoupling of ET from precipitation was most evident at the woodland site, though all sites showed some degree of decoupling. Similarly, Cleverly et al. (2002) found that on the Rio Grande River in New Mexico, a dense stand of saltcedar used 1016 mm/yr, which exceeded the annual precipitation by four times. In comparison, in Vernon, Texas, mesquite transpiration was only 31-32% of annual precipitation even in the dense stands (Ansley et al., 1994). These riparian ecosystems have thus been described as hydrologically sensitive areas (Wilcox et al., 2006) because woody plants access water at depths beyond the reach of non-woody plants, and they are able to use groundwater that otherwise would supply streamflow. It is probable that if the woody plants were replaced by herbaceous plants, much less water would be lost because of the lower ET of herbaceous plants.

6. <u>Certain regions characterized by deep drainage.</u> These areas have heavy clay soils with shrink-swell cracking and are dominated by mesquite (Wilcox et al., 2006). In these areas, deep drainage does occur and groundwater tables are accessible to woody plants, and therefore they are also strong in hydrological sensitivity. Richardson et al. (1979) reported that in the Blackland Prairie of Texas, the heavy clay soils developed extensive cracking, which allows rapid and deep movement of rainfalls that occur. Once wet, their permeability is reduced and allows slow transit of water. Similarly, Cuomo et al. (1992) reported that in Vernon, Texas, a site near an ephemeral stream had a very clayey soil with clay content of 52%, and the water table was only 1.5 m deep. Mesquite trees growing on this lowland site had leaf-level transpiration rates that were higher during a drought period, but lower during rainy periods as compared to trees on an upland site with a sandy loam soil.

3.3.4 Hydraulic Redistribution and Water Relations of Mesquite

As a phreatophyte, mesquite can survive a prolonged drought. One mechanism involved is hydraulic redistribution of water within the soil profile by its root system. A study by Mooney et al. (1980) in the Atacama Desert in Chile found that *Prosopis tamarugo* used groundwater as its primary water source and moved the groundwater into the surface root mat; the water would then be released into the dry topsoil. Therefore, soil water potential in the near surface root mat was quite high despite the complete absence of rainfall.

Although mesquite has a very extensive root system, its water use is constrained by soil water availability and atmospheric demand. During a rainy period, soil water is more available in coarse textured soils due to higher infiltration rates and rapid soil water recharge, while during a drought period, water is more available in fine textured soils due to higher water holding capacity. Soil water uptake by mesquite trees from a sandy loam site in the rainy season was twice as much as that from a clay loam site, as was the mesquite transpiration rate (Wan and Sosebee, 1991). During a drought, mesquite transpiration was significantly higher on a clay loam site due to higher water content. Overall, transpiration rates of mesquite (based on 130 diurnal measurements throughout the growing season) were 16.5% higher on the clay loam site than on the sandy loam site because the former was capable of storing more water in the soil profile. The depth to water table was over 10 m on both sites, and was beyond reach of mesquite tap roots.

Like other phreatophytes such as saltcedar, mesquite regulates transpiration through stomatal control to curb excessive water loss under high atmospheric demand. Phreatophytes frequently have hydraulic failure in root xylem tissue under drought conditions (Kolb and Sperry, 1999; Sperry and Hacke, 2002). When facing increasing vapor pressure deficits (VPD), mesquite tends to reduce stomatal conductance and to prevent xylem water potential (XWP) from decline (Nilsen et al., 1983). This condition occurred even when soil water was most available in the rainy season. Wan and Sosebee (1991) reported that after a 24-mm rainfall, mesquite trees transpired vigorously in the morning and reached a peak rate before noon. Stomatal conductance and transpiration rates then declined sharply as VPD surpassed 20 mbar and XWP dropped below -3 MPa in the afternoon. Yan et al. (2000) also found that the desert shrub creosotebush (*Larrea tridentata*) responded to small rainfall events (6 to 8 mm) with an increase in xylem water potential, but without noticeable changes in stomatal conductance, suggesting a strong stomatal control of water loss. Cuomo et al. (1992) reported that mesquite trees growing on a lowland site with a water table of only 1.5 m actually transpired less than those on the upland site during the rainy season. Even in the riparian ecosystems, annual ET of the dense mesquite stands (400 to 1100 mm/yr) was less than the obligate phreatophyte cottonwood and willow stands (1100 to 1300 mm/yr, Nalger et. al., 2005). Therefore, mesquite may not be a profligate water spender as previously defined (Levitt, 1980).

3.3.5 Water yield from mesquite control

In regions with strong hydrological sensitivity, removal of mesquite increased water yield. Rechenthin and Smith (1964) estimated that a comprehensive brush control program could save “12,000 million m^3 of water in the Rio Grande Plains of Texas.” They assumed that removal of woody plants would reduce ET, increase grass production and water yield. However, their estimate was based on research conducted mainly in Arizona and California. In the Blackland Prairie of Texas (annual precipitation 860 mm/yr), heavy clay soils develop extensive cracking that allows deep drainage. By tracking changes in water content in a 1.5-m soil profile and surface runoff over a period of seven years, Richardson et al. (1979) reported that following mesquite removal, ET was lower and soil moisture higher by 80 mm/yr, and runoff increased 30 mm/yr. Surface runoff from these high-clay soils is substantial, averaging about 30% of the water budget. Results from Richardson et al. (1979) have not been replicated in other work on mesquite rangelands. Carlson et al. (1990) monitored water balance as influenced by vegetation change in a three-year study using lysimeters. They found that on a Texas Rolling Plains rangeland site where annual precipitation was 646 mm, only 0.5-1.4% of precipitation drained below 3 m regardless of vegetation cover type. There was essentially no net change in deep drainage, ET, or runoff on sites where the herbaceous vegetation increased in response to mesquite removal. A study in an adjacent range site by Dugas and Mayeux (1991) found that total seasonal ET from the non-treated site and treated site were 190 and 176 mm, respectively, a 7% reduction due to mesquite defoliation. Mesquite trees were only 15% of total vegetation cover, but the ET from mesquite was 38% of the total. While mesquite used large amounts of water, ET from the treated rangeland was only slightly lower due to increased herbaceous transpiration. They stated that “under circumstances of low grazing pressure and low runoff potential, honey mesquite removal would provide little if any additional water for off-site uses in the short term,” (Dugas and Mayeux, 1991). Heitschmidt and Dowhower (1991) concluded that increasing water yields in south Texas through vegetation manipulation is marginal and limited to those years when rainfall exceeds potential ET. In a study conducted in a southern Texas rangeland where annual precipitation was 710 mm, Weltz and Blackburn (1995) found little difference in soil moisture storage or ET between adjacent mesquite- and grass-dominated communities. In their study, the root density in the upper 65-cm soil profile was not significantly different between the shrub clusters and grass interspaces. From 0.9 to 2 m, the woody plant root density in a shrub cluster was higher than the density of grass roots in the grass interspaces. There was a significant difference in the patterns of soil water use, but not in total ET, or soil water content between the two sites. A total of only 22 mm of water percolated below 2 m from the grass interspaces during the 18-mo study period.

Wilcox (2002) concluded, “Shrub control on mesquite dominated rangelands is unlikely to affect streamflow significantly for 4 reasons:

- Evaporative demand is high, and typical herbaceous replacement vegetation uses most of the available soil water;
- Soils on these sites are typically deep, effectively isolating the groundwater zone from the surface;
- Runoff is generated primarily as Horton overland flow; and
- Runoff is very flashy in nature, generated by flood producing events, overwhelming other factors.”

3.4 Water Use by Juniper

3.4.1 Distribution and Growth Habitats

The genus *Juniperus*, represented by 17 species in the western United States (Owens and Ansley, 1997), has invaded many semiarid rangelands. Junipers are among the most drought-tolerant of evergreens. When juniper trees invade a rangeland, herbaceous production is generally reduced; when the tree community matures, the herbaceous production is further diminished under closed canopies. This lack of herbaceous biomass reduces livestock production, wildlife diversity, and watershed protection. While juniper may grow over a broad range of habitat types, most juniper populations are found in the upland or non-riparian rangelands.

Juniper trees can strongly impact soil water content and landscape water balance of a plant community. The most direct negative impact is to use more water than the herbaceous vegetation they are replacing. Juniper trees have very large leaf area that transpires large quantities of water. The trees remain green all year long, and can transpire when other plants are dormant. Junipers have deep root systems. The trees proliferate in regions where deep drainage is available. *Juniper ashei* has wide distribution in the Edwards Plateau of central Texas where the geology is characterized as a karst system. Karst geology has two important features, namely, shallow soils, which cannot hold much water, and fractured parent material, which allows rapid, deep drainage of rainfall, and facilitates the presence of springs (Wilcox et al., 2006). These shallow soils are underlain with limestone containing deep fractures and underground caves and streams. In a limestone cave study, long-term sap flux measurements of *Juniper ashei* showed that deep roots, which penetrated 7 m below soil surface, were able to contribute 60% of daily transpiration after prolonged drought (McElrone et al., 2003). One large tap root at 7-m depth supplied a large proportion of daily water use. During periods without rain, upward flow through deep roots was continuous during both day and night. The nocturnal hydraulic lift contributes 20% of water movement from this depth. This night-time water flow from deep roots to shallow roots occurred most often during a drought when water potential gradient from surface to depth was steep. A study by McCole (2003), which was also conducted on the Edwards Plateau of Texas, found that Ashe juniper trees derived 72 to 100% of their water from groundwater during dry periods of the year (late summer and winter). During the wet periods of the year (spring and fall), between 45 and 100% of water use by juniper was derived from soil water. This study indicates that juniper reduce groundwater resources both by lateral roots intercepting potential recharge during the wet season and direct uptake of ground water by deep roots during the dry season. In another study, Leffler et al. (2002) found that Utah juniper (*J. osteosperma*) dried the soil from the surface downward to a depth of about 1 m. The study confirmed that hydraulic redistribution is a significant process in soil water dynamics. Because juniper uses large quantities of soil water, growth of herbaceous plants is suppressed under juniper overstory. Cutting juniper trees was effective in increasing total understory biomass, cover, and diversity; and herbaceous biomass was nine times greater in cut versus woodland treatments in the second year post-cutting (Bates et al., 2000).

Juniper also changes landscape water balances of a plant community by intercepting a significant proportion of precipitation with its dense canopy and litter (Young et al., 1984;Thurow, 1991; Eddleman and Miller, 1992; Hester, 1996; Thurow and Hester, 1997; Lyons et al., 2006; Owens et al., 2006). This intercepted rainfall or snowfall results in high evaporation and sublimation losses directly back to the atmosphere from wetted canopy and litter. This phenomenon has been estimated to reduce winter soil moisture recharge by more than 50% in dense juniper stands (Eddleman and Miller, 1992). The interception loss associated with the canopies of redberry juniper (*J. pinchotii*) and Ashe juniper was 25.9% and 36.7% of gross precipitation, respectively (Hester, 1996). Ashe juniper typically has a very dense canopy and thus more surface area to intercept rainfall and snowfall, which is then evaporated to the atmosphere. Rainwater that passes through the canopy must also pass through the litter layer prior to entering the soil. The amount of interception loss associated with the litter layer is considerably greater for redberry juniper (40.1%) and Ashe juniper (43%) than for western juniper species (2-27% by Young et al., 1984; Thurow and Hester, 1997). The result of Thurow and Hester (1997) was based on a 10-in thick organic soil layer instead of the coarse litter fraction, and the interception by the litter layer may be overestimated. Under a dense juniper canopy, most of the small rainfall events (<5 mm) do not reach the litter layer because of water retention by the foliage. As a result of interception loss via the canopy and litter, only 20.3% and 34% of annual rainfall reaches mineral soil under the canopy of Ashe juniper and redberry juniper, respectively. In contrast, as high as 81.9% and 89.2% of annual precipitation reaches the soil under bunchgrass and shortgrass cover, respectively (Thurow and Hester, 1997).

The magnitude of interception loss also depends on rainfall intensity. Low-intensity storms were defined as storms yielding less than 0.5 in of rain over a 24-hr period (Lyons et al., 2006). These storms occur frequently in semiarid rangelands, but contribute little moisture to the soil surface under juniper canopies. In the Lyons et al. study (2006), at ten research sites with average juniper trees 18 ft tall, and canopy area 230 ft^2, 60% of storms were less than 0.1 in, and they were either intercepted by the canopy (96%) or the litter layer (2%), leaving only 2% of the bulk rainfall to reach the soil surface beneath the juniper canopy. In contrast, only 2.7% of storms were more than 2.5 in, but these large rainfalls contributed more than 27% of the total rainfall. These high-intensity storms can deposit more than 1 in of rain in a very short time. As storm size increased, the proportion of water intercepted by the canopy and lost to evaporation decreased. During a typical high-intensity storm, only about 15% of the rain is intercepted by either the canopy or the litter layer. In semiarid areas, interception loss to coniferous trees ranges from 20 to 48%, which is higher than 9 to 20% for deciduous trees, and 13 to 40% for shrubs. Lyons et al. (2006) found that Ashe juniper canopy and litter intercepted about 40% of the total bulk precipitation over all 10 sites and all intensities of rainfall during a 3-year period. Their results on canopy and litter interception were considerably lower than that (79%) of Thurow and Hester (1997) due to much smaller interception by the litter layer, which was the coarse litter fraction (0.2- to 2.4-in thick) instead of the 10-in thick litter layer. Lyons et al. (2006) argue that between 2.4- and 10-in soil depths, plant roots were prevalent and water use from this layer would be largely impacted by transpiration. Their estimate of the total interception loss compares favorably with 46% by Young et al. (1984) and 40% by Owens et al. (2006), but is considerably higher than 5 to 25% by Utah juniper in Arizona (Skau, 1964). Interception loss is generally small in arid shrublands or savannas because of lower canopy cover, but higher in juniper woodlands or grasslands if the cover is extensive. In a Sonora, Texas, grassland, which was composed of 40% grass, 24% oak, and 36% juniper cover, the interception loss by the plant canopies was 42% of the annual precipitation (Thurow and Hester, 1997).

In Lyons et al.'s (2006) study, only 50% of a 0.4-in storm reached the soil surface, and 50% held by the canopy and litter was lost to evaporation. High-intensity rainfall was less influenced by juniper canopy, only 20% was lost to interception. On the average, about 60% of rainfall can reach the soil surface, which is considerably higher than the estimate (20 to 34%) by Thurow and Hester (1997). When juniper cover was 20%, the amount of water lost to interception averaged 2.4 in/yr. As tree cover increased from 20 to 100%, the amount of water lost to interception increased to 12.6 in/yr, or 5.2 times higher (Lyons et al., 2006). Some intercepted precipitation may reach the soil as throughfall and stem flow. While throughfall could be intercepted by litter below the tree canopy, stem flow will most often reach the soil profile. Young et al. (1984) and Larsen (1993) documented juniper stem flow to be less than 5% of precipitation, but suggested it still may provide a significant advantage for juniper growth. This additional available water channeled to the base of tree was taken up by the concentration of fine roots adjacent to the trunk in western juniper (Young et al., 1984).

3.4.2 How Much Water Can a Juniper Plant Use?

How much water can a single juniper tree use on daily basis? It depends on the tree size, annual precipitation, depth to water table, density of the stand, and environmental conditions. Generally, juniper trees transpire much more water than herbaceous vegetation because juniper transpires throughout the year, typically has more leaf area, and can access water at great depths. Owens and Ansley (1997) conducted research at various sites in the Edwards Plateau of Texas, and found that daily water use by redberry juniper and Ashe juniper was 46.8 and 33.1 gal/d, respectively. With an average daily water use of 39.8 gal/tree, the juniper transpiration was equivalent to 400 mm/yr. Owens (1996) reported that more than 20 year old Ashe juniper transpired 33 gallons per tree on a daily basis, which is close to that of Owens and Ansley (1997). Compared with other phreatophytes such as mesquite, juniper uses water twice as much on a per tree basis, and has lower water use efficiency. For example, redberry juniper daily water use was 46.8 gal/tree as compared to 20.9 gal/tree for honey mesquite (Owens and Ansley, 1997), which was due to much larger leaf area of juniper. In the regions with lower precipitation such as eastern and central Oregon, western juniper transpired less water than the trees in Edwards Plateau. Jacks (1998) found that on a warm April day, individual trees can use up to 20 gal/d. In central Oregon it was estimated that juniper trees used over 12.6 inches of water in a precipitation zone of 15 in. The site had a density of 480 trees/acre, and the water use by these trees amounted to 84% of annual precipitation, leaving only 2.4 in of water for other plant species. Juniper competition led to fewer plants, less soil cover, lower infiltration rates and more opportunity for overland flow and soil erosion. Gifford (1975) estimated that ET for pinyon-juniper woodland was equivalent to 69-97% of annual precipitation. Lane and Barnes (1987) reported 80-100% of annual precipitation was evapotranspired by *J. osteosperma* and *J. deppeana*. The highest estimate was made by Thurow and Hester (1997) with *J. pinchottii* and *J. ashei*, which had ET equivalent to 100% of annual precipitation. Using density estimates combined with a canopy model, Owens and Ansley (1997) predicted water use by juniper in a non-grazed pasture transpired an average of 1.4 acre-ft/yr (420 mm/yr), in a lightly browsed pasture transpired 0.97 acre-ft/yr, and in a heavily browsed pasture transpired 0.34 acre-ft/yr. It is logical that removal of juniper trees could lead to more water available for herbaceous plants and streamflow.

3.4.3 Water Yield from Juniper Control

There is a potential for water savings by removing juniper. Wilcox et al. (2006) stated at the tree scale, for an area with an average annual precipitation of 750 mm, an individual tree will intercept and transpire virtually all of the available water. Therefore, the hypothetical potential water savings from removal of juniper would be substantial. However, in reality there is limited evidence to support this hypothesis (Belsky, 1996). In a few studies that reported increased water savings, such effects only occurred in selected watersheds (Wilcox et al., 2006), and the magnitude of water savings may not be as great as one would expect (Dugas et al., 1998; Huang et al., 2006). Juniper cover can influence overland flow, streamflow, and/or groundwater recharge. There are, however, conflicting reports on the magnitude of the impact of juniper removal on rangeland hydrology.

Rangeland runoff dynamics are influenced by juniper cover. A widely held view is that overland flow and erosion will be increased by higher coverage of woody plants. Increases in runoff and erosion following juniper encroachment are the result of overgrazing of the diminishing herbaceous cover (Thurow and Hester, 1997). Buckhouse and Mattison (1980) found lower infiltration and higher erosion rates in western juniper woodlands than in other northwestern range and forest communities. Dugas et al. (1998) reported dramatic reductions in Horton overland flow following juniper eradication. On many juniper-dominated sites, tree canopy cover is between 20 and 35%, leaving up to 80% of the area with reduced vegetation or litter cover for protection (Miller et al., 2005). Frederick et al. (2007) reported 15 times higher runoff on juniper-dominated sites. Removal of juniper increased ground cover in the interspaces between trees from 16% to 36%, improved infiltration capacity and reduced runoff by 67%. Cutting juniper also protected the soil surface from large, high-intensity thunderstorms. However, other studies on pinyon-juniper woodlands in the Southwest and Great Basin failed to find lower water infiltration rates or more erosion than in other communities (Gifford, 1973; Clary et al., 1974; Gaither and Buckhouse, 1983; Renard, 1987; Schmidt, 1987). The effects of juniper woodlands on infiltration rates and erosion may be site-specific (Blackburn and Skau, 1974) and depend on slope, soil type, disturbance, vegetation cover, and frost dynamics (Wilcox, 1994).

In contradiction to the widely held view, Blackburn (1975) found that infiltration through surface soil was actually higher in Ashe juniper areas than in grass-covered areas. This view is shared by other authors (Wood and Blackburn, 1981; Thurow et al., 1986; Hester et al., 1997). They reported that because of higher vegetation cover on woody species dominated sites, infiltration rates are often highest under trees and shrubs, followed by bunchgrass and shortgrass sites. The litter also contributes to building better soil structure, which maintains large stable pores in the soil through which water can pass. For these reasons, the surface runoff should be higher following juniper removal. Wright et al. (1976) reported that Horton overland flow was significantly greater for two to three years following removal of juniper by burning; presumably it took this much time for the vegetation to completely recover. Hibbert (1979) estimated a 13 mm increase in runoff by controlling pinyon-juniper in the Colorado Basin. In the north Concho River watershed, Wu et al. (2007) found that when junipers were cleared on two sites, 7.7 and 10.7% of rainfall events produced runoff during the 2005 to 2007 study period. In a 4-year study in the Edwards Plateau, Huang et al. (2006) found that runoff made up 22% of the water budget, with baseflow from the spring accounting for about half of the total flow. The mean runoff after a rainfall event was 5.5 mm for the pre-treatment period and 8.8 mm for the post-treatment period, an increase of 60% after removal of juniper (Huang et al., 2006). Wilcox et al. (2002) pointed out that effects of shrub control on surface runoff depend on how the control method modifies surface conditions. Therefore, shrub control could result in either an increase or decrease in Horton overland flow. Clary et al. (1974) conducted a paired watershed study in which pinyon-juniper woodlands in Arizona were removed by herbicide, chaining, or cutting. Streamflow increased in watersheds where trees were killed by herbicide and left standing, but no changes in streamflow were noted in other treatments where trees were removed by cabling and felled by hand and left in place. The increased water yield in the herbicide treatment may be due to the absence of soil disturbance. Blackburn (1983) also reported that spraying Utah juniper in Arizona significantly increased runoff compared to that in an undisturbed woodland, where hand slashing and chaining had little effect on runoff. Runoff from intense summer thunderstorms was greater from chained with debris-windrowed than from chained with debris-left-in-place, or undisturbed woodlands. In a long-term study on small watersheds in Sonora, Texas, Thurow and Hester (1997) found that removal of juniper had little or no effect on surface runoff. They attributed this result to good grass cover, and high organic matter and porosity in the soil under juniper trees, which helped to stabilize the site and reduce the potential for runoff and erosion when the junipers were cleared. As Wilcox et al. (2002) pointed out, if surface disturbance is minimal, herbaceous cover rapidly replaces juniper and runoff can actually be lower following juniper removal.

Water balance studies on the Edwards Plateau suggest that on average 15% of precipitation ends up as recharge for the underlying Edwards Aquifer, most of it via transmission losses from stream channels that cross the Edwards Aquifer recharge zone (Maclay, 1995). Since juniper trees can access groundwater, it is reasonable to expect that removal of juniper trees would contribute to recharge of groundwater stores. ET estimation based on the Bowen ratio method at the juniper stand suggests the direct recharge in this landscape following juniper removal could be substantial (Dugas et al., 1998). They estimated that removing woody plant cover reduced ET by 40 mm/yr for a period of at least two years. A recent study at the small-catchment scale by Huang et al. (2006) estimated that removal of juniper will increase stream-flow by 46 mm/yr, representing about 5% of precipitation. From these limited studies, it appears that conversion of Ashe juniper woodlands to grasslands or open savannas will translate to increases in spring flow and groundwater recharge at the small-catchment scale.

There are, however, mixed findings in the literature on relations between conversion of juniper woodland and water yield. In a 12-year study of an Arizona watershed with annual precipitation of 510 mm/yr, Collings and Myrick (1966) found no significant increase in annual water yield following juniper removal by cutting and prescribed burning. In a five-year study, Gifford (1975) examined storm runoff volumes from 1-acre sites in southern Utah following juniper control by chaining. Chained and windrowed sites yielded from 1.2 to 5 times more storm flow than did the undisturbed woodland. No change in runoff was observed where downed trees were left on-site after chaining, because the debris detained runoff and enhanced infiltration. Backer (1984) reported on a 14-year study of water yield following Utah juniper control with herbicide on a 363-acre rangeland in Arizona with average precipitation of 463 mm/yr. Backer (1984) found an increase in annual streamflow of 157% in the first two years post-treatment, which was not statistically significant eight years post-treatment. Lack of sustained reduction in ET was probably due to increased growth of herbaceous vegetation following juniper removal. In arid watersheds, the potential to increase streamflow is complicated by high evaporative demand, high percentage of bare ground, and high direct evaporation from the soil surface (Bosch and Hewlett, 1982; Hibbert, 1983; Wilcox et al., 2002). Hibbert (1983) examined studies in arid and semiarid rangelands, and concluded that less than 1% of rangelands in the western United States are conducive to being successfully managed for increased water yield by vegetation conversion.

Amount of precipitation is a key factor to determine whether water yield can be achieved from juniper removal. Through a literature review, Bosch and Hewlett (1982) found no increases in water yield in areas averaging less than 17.7 in/yr (450 mm/yr) of annual precipitation. Hibbert (1979) stated if brush management is expected to increase water supply for an area, the annual precipitation should be greater than 18 in/yr (457 mm/yr). Wilcox (2002) also stressed that there is little prospect of increasing streamflows where mean annual precipitation is less than 19.7 in/yr (500 mm/yr). This notion appears to be confirmed by the available data. For example, studies by Dugas et al. (1998) and Huang et al. (2006) were both conducted on the Edwards Plateau where precipitation was relatively high (920 mm/yr in Huang et al.). In contrast, the studies in Arizona by Backer (1984) were conducted in areas with annual precipitation of less than 510 mm/yr. Thus, no apparent water yield increase was observed or the increase was not sustainable following juniper removal. On the study site of Backer (1984), the 463 mm/yr annual precipitation was probably fully used by increased herbaceous vegetation eight years after juniper removal. Kuhn et al. (2007) conducted research in the Klamath River Basin in California that has significant areas dominated by western juniper. They found that in the areas with annual precipitation of greater than 17.7 in/yr, even the complete removal of western juniper did not significantly increase water yield. However, they did observe small increases in summer flow rates in small tributaries and spring flows that supported wetlands after juniper removal. They concluded that although insignificant in the arena of increasing Klamath River flow, these flows are extremely critical for maintaining aquatic habitat and drinking water for wildlife and livestock.

Another important issue relating to water yield is how much juniper cover is removed. Bosch and Hewlett (1982) proposed that the amount of vegetation cover removed is proportional to changes in water yield and that, for many areas, removing less than 20% of the cover would not yield detectable changes in streamflows. This conclusion is understandable because, as Lyons et al. (2006) pointed out, when juniper cover increased from 20 to 100%, the amount of water lost to interception increased to 12.6 in/yr, or was 5.2 times higher. That amount was just interception by the canopy and the litter layer, which was then evaporated into the atmosphere; if transpiration was taken into account, there would be a huge difference in water consumption between 20 and 100% juniper cover. Thus, when juniper cover is reduced from 100 to 20%, there would hypothetically be a substantial water savings. However, Hibbert (1983) stated that the relationship between percentage of vegetation removal and reduced transpiration is non-linear, and that meaningful reductions in transpiration in arid environments are only achieved at high levels of removal. For instance, removing half of the deep-rooted vegetation may hypothetically result in only a 20% reduction in transpiration. Hibbert (1983) also cautioned that when greater amounts of juniper canopy were removed, it would increase direct evaporation from soil surface. Therefore, reseeding with herbaceous vegetation following juniper removal was recommended. The replacement species should be shallow-rooted, deciduous, or have a low biomass (Hibbert, 1979).

The fundamental controlling factor in determining water yield appears to be the availability of groundwater (Wilcox et al., 2006) as, for example, in riparian environments. For an upland site with a calcic soil horizon, such as in west Texas, the soil water is mainly in the upper 1 m of the profile, and downward flux of water is very small. In regions where junipers are found on deep soils, the subsurface flow does not occur. Eradication in these regions is unlikely to increase water yield or streamflow. For an upland area to be hydrologically sensitive to changes in woody plant cover, there must be a reservoir of water available to deep-rooted plants that is not available to shallow-rooted plants. In rangelands not characterized by groundwater within a few meters of the surface, the geological conditions must allow deep drainage to maintain these reservoirs. These areas are in the relatively mesic rangelands situated in karst geologic settings with shallow soils underlain by fractures of the parent material and underground caves where rapid recharge occurs after rainfall. There are reports of spring flow appearing or increasing after shrub control for juniper rangelands on the Edwards Plateau (Wright, 1996) and for pinyon-juniper watersheds in Utah (McCarthy et al., 1999). Despite higher precipitation and deep drainage, water yield from juniper control in upland environments is meager, only 40 to 46 mm/yr (Dugas et al., 1998; Huang et al., 2006), as compared to 300 mm/yr in riparian areas of dense saltcedar (Weeks et al., 1987).

Much higher water savings were reported in a study that was conducted at the Sonora Agricultural Experimental Station (Thurow and Hester, 1997). The soils at their research sites were 6 to 18 in deep, which overlay a fractured limestone substrate. Their data indicated that substantial water yield can be achieved through conversion of pasture vegetation from juniper to grass dominance. Although the area received an annual precipitation of only 574 mm/yr, deep drainage occurred due to karst geology. The estimated deep drainage was 94 mm/yr in a 100% grass pasture as compared to 0 mm/yr in a juniper/oak/grass community. This result was largely caused by a three-fold greater interception loss in the juniper/oak/grass community. The water yield following juniper removal was equivalent to 100,500 gal/acre/yr. There was little runoff from these pastures, because the cut juniper maintained very high infiltration rates after the trees were removed. The moderately grazed pastures also had a good herbaceous cover in the juniper interspaces. Therefore, the added precipitation reaching the soil as a result of reduced interception losses did not runoff of the pasture but was instead channeled into the soil.

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Appendix

Fish, E. and K. Rainwater. 2007. Subwatershed selection criteria for demonstration of water yield enhancement through brush control. Final Report, Texas State Soil and Water Conservation Board, San Angelo, TX, 15 p.

The following document was prepared by the authors to assist the TSSWCB staff in their internal discussions of site selection for brush control for water yield enhancement, and to facilitate conversations with legislators and their staff members. The document was written in a relatively qualitative fashion to encourage conceptual connections with both technical and non-technical interested individuals. With that intent, it was not written in the same scientific, highly referenced manner as the report to which it is now attached. It is included here because it was expedient to provide the document rather than try to intertwine it within the requirements of the larger monitoring and water yield enhancement evaluation. We recognize that the criteria are not unique, but rather follow fundamental principles that would be presented by many other scientists and engineers with interest and expertise in watershed stewardship.

Subwatershed Selection Criteria for Demonstration of Streamflow Yield Enhancement through Brush Control

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Problem Statement

The Texas State Soil and Water Conservation Board (TSSWCB) manages the State of Texas' program for brush control to enhance surface water flows for urban and rural benefits. To date, the sites selected have been large watersheds with varying coverages of different high-water use plants, such as salt cedar, juniper, and mesquite. Using conventional techniques available in the late 1990s, predictions of increased streamflow through brush control were based on watershed modeling that primarily changed runoff generation and groundwater loss parameters in each watershed. Unfortunately, there has not been sufficient pre- and post-treatment monitoring to confirm the positive impacts of the selected brush control practices. It is now apparent that scientific principles from hydrology must be applied to improve the site selection process to prioritize sites for potential observable enhancement of streamflow yields.

The purpose of this report is to provide a straightforward list of criteria for selection of sites for streamflow enhancement that will maximize the potential for observation of positive impacts of brush control. The report includes a brief presentation of the general hydrologic and watershed management concepts that contribute to possible increases in runoff and streamflow from a watershed. Next, the proposed criteria are listed. Finally, some general observations about the importance of monitoring are summarized. This report is not a detailed, thoroughly referenced critique of the historical brush control program, but is rather intended as a useful reference for decision makers in the TSSWCB and the Texas Legislature. The authors plan to participate in a more detailed review of the TSSWCB brush control program later in this fiscal year.

General Concepts

Runoff generated by a watershed is a reflection of the integrated net effects of all watershed characteristics as they interact with and reflect the hydrologic cycle. One of the primary objectives of watershed management is to maintain or improve water yields. To do so requires an understanding of how a watershed functions in the delivery of water to its outlet. A complete discussion of the factors involved and their functional relationships is beyond the scope of this report, but it is possible to summarize the major components and discuss some of their complexities and interactions.

Probably the most significant relationship impacting water yields is the overall ratio of precipitation to evaporation. In locations where this ratio exceeds 1.0, the likelihood of enhancing runoff is increased; likewise, when the ratio is less than 1.0 runoff enhancement is more challenging. If rainfall and evaporation were both uniformly distributed over time, it would be difficult to have any streamflow at all in locations where evaporation exceeds precipitation. It must be recognized, however, that the intensity, duration, and seasonality of precipitation are more important than the annual totals. Intensity of rainfall refers to the depth of rainfall per unit time, and duration is the time length of the rainfall event. Not all precipitation events are "runoff producing" even when the total amount of precipitation is identical, as the time between storm events can affect the pre-storm soil moisture. It is possible at almost any location in the state (except for large sand dunes) to have short-term storm events that put a lot of water on a watershed quickly enough that runoff can reach streams and rivers faster than it is lost to evaporation. Seasonal trends in rainfall intensity have been noted across the state. For example, the western side of the state receives much of its rainfall from convective thunderstorms in the late spring and summer months, while the coastal regions can receive major rainfall events during hurricane and tropical storm season.

Land use activities that alter the type or extent of vegetative cover on a watershed can increase or decrease streamflow yields. The expected outcome from a planned vegetation adjustment is based on change in evapotranspiration (ET) “losses” from the watershed. Complicating these expectations are variations in precipitation patterns, soil infiltration capacities, groundwater conditions, watershed area, land surface slopes, and other characteristics of particular watersheds. For example, Hibbert (1979, 1983) concluded that vegetative manipulations were likely to enhance water yields only when the watershed received more than 18 inches of annual precipitation. Residual vegetation was expected to consume all available precipitation in areas receiving less precipitation.

After accounting for precipitation patterns and amounts, water yields from watersheds receiving similar inputs are dramatically impacted by soil, elevation, slope, aspect, climate, and vegetation. Selection of areas for vegetative manipulation to enhance water yields should therefore be prioritized using a combination of the following factors. In general, storm runoff and associated streamflow will be greater under these conditions.

- ***Steeper slopes.*** Runoff on steeper slopes can produce more erosion. Steeper slopes can be more difficult to stabilize following vegetative manipulation, as the vegetation likely helped hold soils in place.
- ***More uniform slopes with limited soil disturbance.*** Adding terraces to slopes reduces runoff yield. Mechanical disturbances that leave the soil surface "cratered" can also lose runoff to depression storage.
- ***Soils with lower infiltration capacities.*** Typically soils with finer textures and with poorer structure have lower permeability and accept less vertical infiltration.
- ***South and west facing slopes.*** These aspects tend to be steeper and have less vegetative cover in the northern hemisphere.
- ***Closer proximity of contributing area to stream channels.*** Runoff flow volume is diminished by transmission losses to infiltration and depression storage over long flow routes.

Another important consideration in water yield from a watershed is baseflow, which is the contribution of groundwater discharge to streamflow. For perennial streams, baseflow represents the streamflow that takes place between rainfall/runoff events, without the contribution of storm runoff. It should be noted that ephemeral streams, which have no baseflow, may still be positively impacted by watershed management in the arid and semi-arid parts of the state. It is possible that evapotranspiration near the streambed can lower the local groundwater table and turn a perennial stream into an ephemeral or intermittent stream. In general baseflow to streams and rivers will be greater under the following conditions.

- ***Soils with higher infiltration capacity soils near the streambed.*** Alluvial (eroded and deposited within the stream channel) soils with coarser textures and mostly large particle sizes allow faster movement of water from the aquifer toward the stream.
- ***Higher water table elevation near streambed.*** If the elevation of the groundwater table in the alluvial material near the streambed is higher than the elevation of the water surface in the stream, groundwater will flow toward the “gaining” stream and

contribute baseflow. If the groundwater table elevation is lower than the water surface in the stream, water will from the "losing" stream into the alluvial aquifer.

- ***Replacement of deeper-rooted with shallower-rooted plant species.*** Typically deep-rooted woody species can be replaced with shallow rooted herbaceous species. Concern must also be given to the density of the vegetative cover, as replacing a moderately dense stand of mesquite trees with a heavy cover of switchgrass could in fact reduce alluvial groundwater storage because of increased vegetative demand on available soil water.

Statement of Selection Criteria

The TSSWCB desires to maximize its ability to demonstrate the positive impacts of brush control on streamflow enhancement. The TSSWCB and some of its collaborators have already realized that it is more appropriate to consider smaller subwatersheds rather than large river basins. The subwatershed approach should be continued, whether in the existing brush control treatment sites or in new locations. The following criteria are proposed for consideration in selection of these sites.

- ***Soils – low permeability in the watershed catchment area and leading toward the streambed***
- ***Slope – sufficiently steep to carry runoff to streambed***
- ***Area – large enough to generate measurable flow contribution***
- ***Brush cover distribution – fraction of the area with treatable brush cover and proximity to stream channel***
- ***Land use – vegetation and land management strategies by land owner***
- ***Streamflow observation – proximity to a stream gauging station, whether installed for the brush control project or existing for other agency's purposes***
- ***Groundwater conditions – depth to groundwater table, groundwater flow direction, and aquifer permeability***

It should be noted that this list is not given in order of numerical priority. It may not be possible to maximize all the criteria at the same time in the same subwatershed. Still, the criteria are based on sound hydrologic principles and conditions that can be observed and mapped.

Conceptual Application of Criteria

As an illustration of application of some of these criteria in map form, we used the following illustrations based on the Pecan Creek Watershed, as its delineation was presented by the TSSWCB (Figure 1). This application is only a "demonstration" of the techniques, and the arbitrary choices of elements and categories within elements were for illustration only. Additional elements, additional categories, and different definitions of relative importance are all possible depending on site specific characteristics and management objectives.

In arid and semi-arid environments, streams typically experience "transmission losses" as the water moves longer and longer distances from the point of overland flow generation through the channel to the final watershed outlet. The closer the point of runoff generation is to the watershed outlet, the greater will be the expected streamflow because channel transmission losses to seepage and evapotranspiration will be minimized. In this demonstration, we divided the Pecan Creek Watershed into three zones based on channel travel lengths: closest to the outlet, furthest from the outlet, and "in between" (Figure 2). The zones were arbitrarily delineated by a circular buffer from the outlet point of 7, 14, and 21 miles. Each zone was assigned a numerical value (15, 10, or 5), and higher value is associated with less transmission loss and better streamflow contribution to the outlet.

Proximity to a stream or tributary channel is another variable worthy of consideration. Brush treatments applied closer to a channel are more likely to produce overland flow or increase baseflow that reaches a channel than are treatment located further away from the channel. This concept can also be stated as expecting treatments in riparian corridors to enhance yields more than similar treatments on upland sites. In this demonstration, we divided the Pecan Creek Watershed into three zones based on proximity to a stream channel: closest to the channel, furthest from the channel, and "in between" (Figure 3). The zones were delineated by linear buffers of 100 and 300 meters to create three zones: < 100 meters from a channel, 100-300 meters from a channel, and > 300 meters from a channel. Each zone was assigned a numerical value (20, 10, or 5), and higher value is related to greater proximity to the channel.

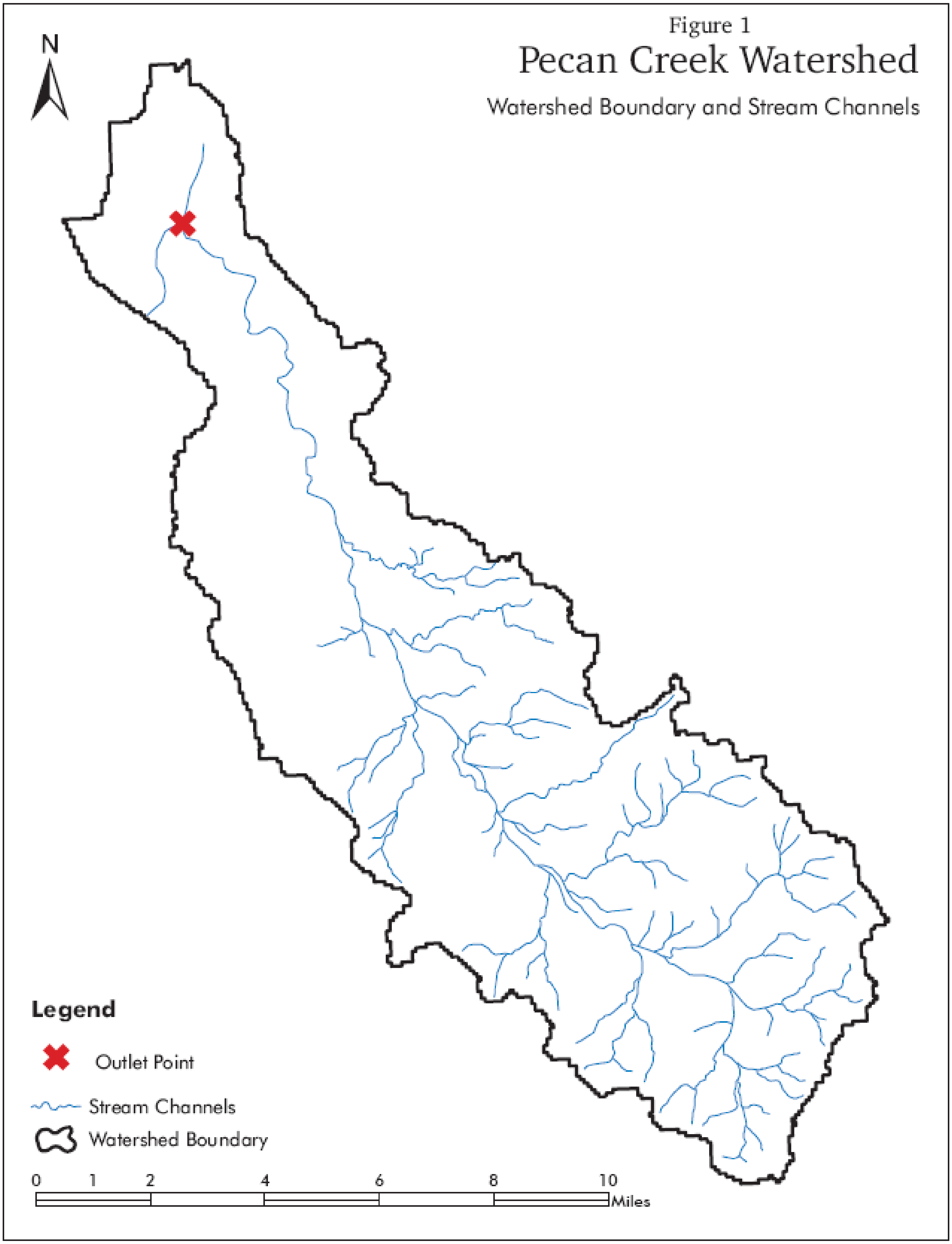
Figure 1
Pecan Creek Watershed
Watershed Boundary and Stream Channels
N
Legend
Outlet Point
Stream Channels
Watershed Boundary
0 1 2 4 6 8 10
Miles

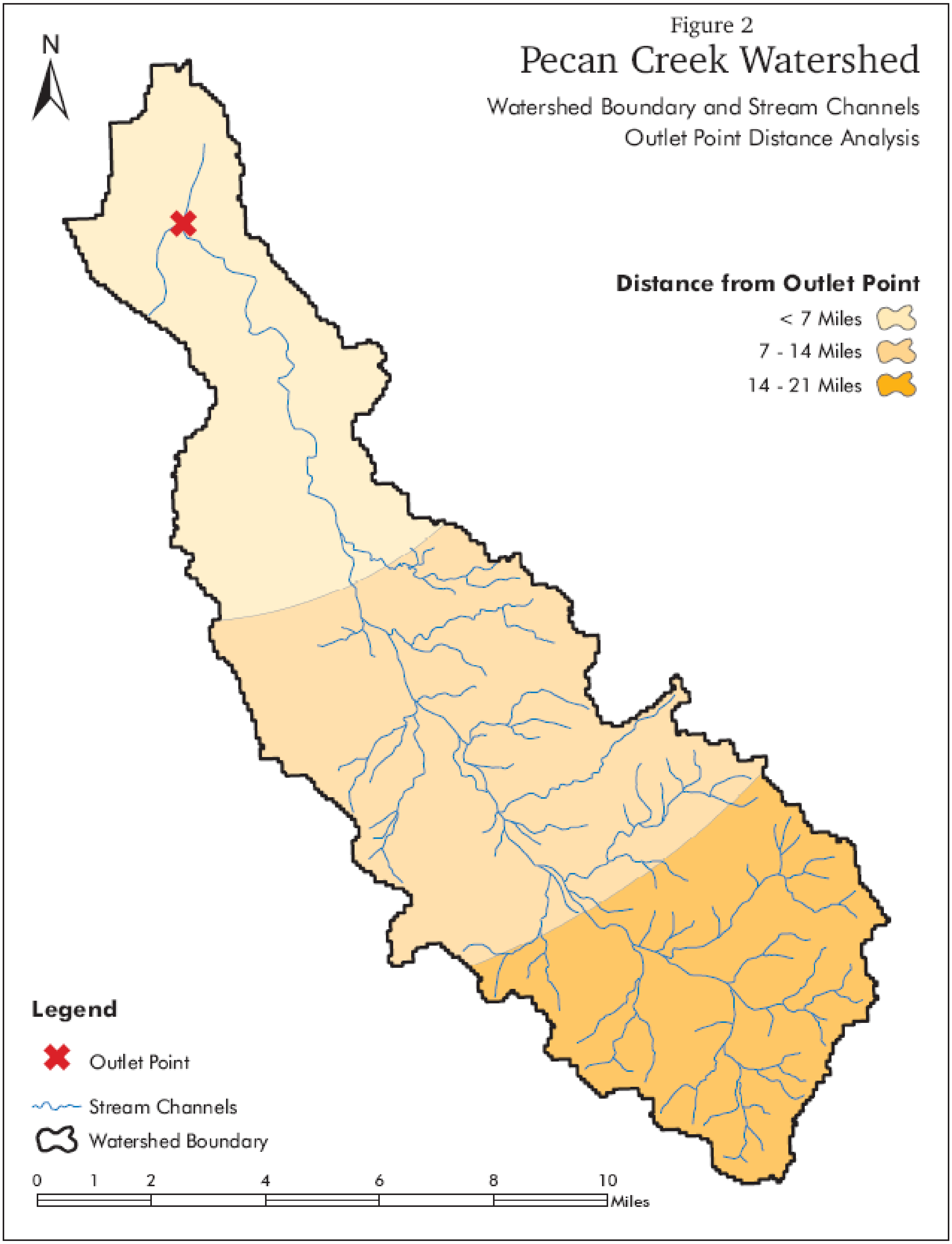
Figure 2
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Outlet Point Distance Analysis
N
Distance from Outlet Point
< 7 Miles
7 - 14 Miles
14 - 21 Miles
Legend
Outlet Point
Stream Channels
Watershed Boundary
0
1
2
4
6
8
10
Miles

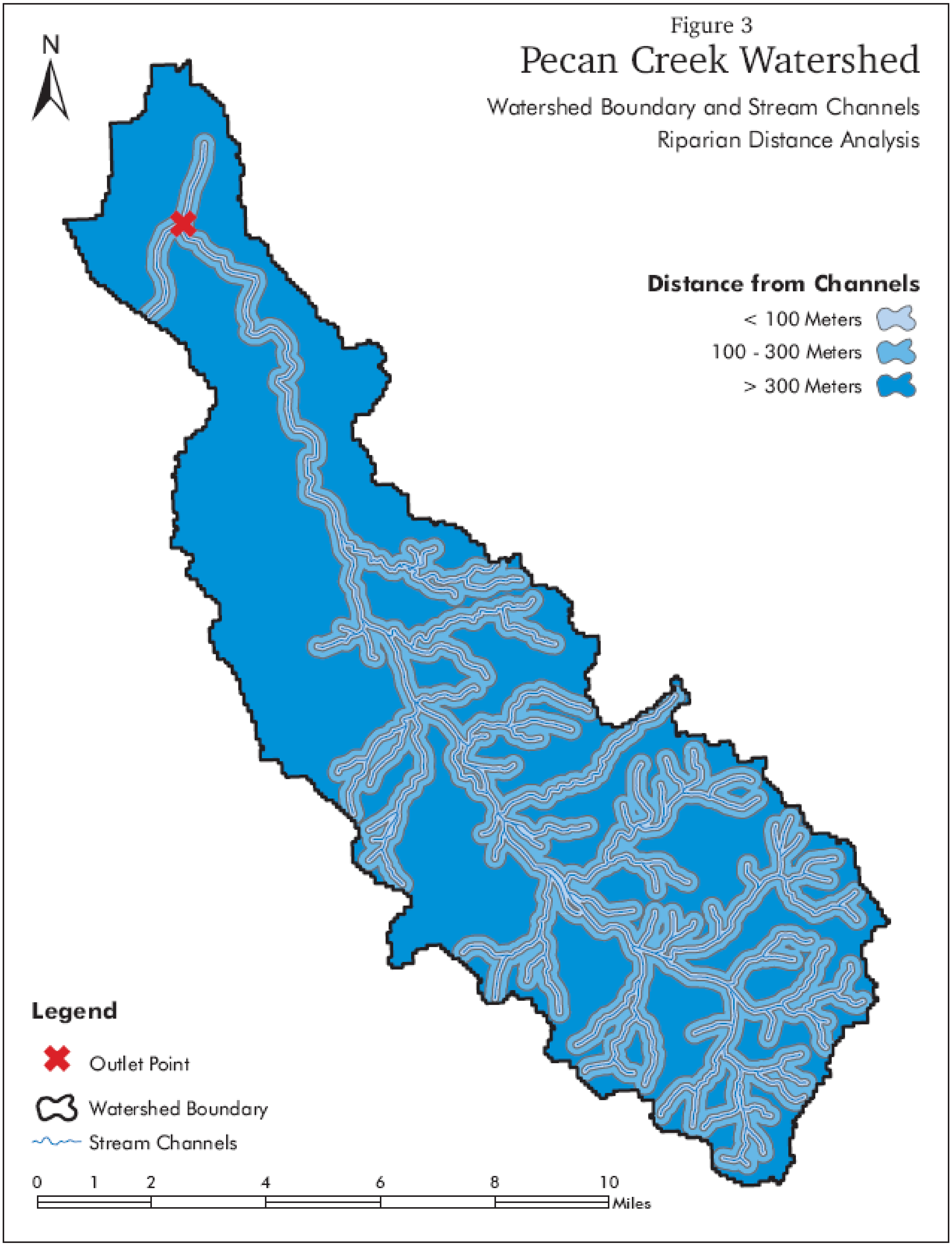
Figure 3
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Riparian Distance Analysis
N
Distance from Channels
< 100 Meters
100 - 300 Meters
> 300 Meters
Legend
Outlet Point
Watershed Boundary
Stream Channels
0 1 2 4 6 8 10
Miles

Soil texture and slope are other variables impacting runoff. For purposes of this illustration, we selected a combination of soil textures and a slope category to create areas with an increased likelihood for generation of enhanced runoff following treatment. In this particular case, soils with textures of clay or clay loam having slopes in the range of 1 to 3 percent were selected to illustrate the methodology (Figure 4). Heavier textured soils with steep slopes would be expected to have lower infiltration rates than coarse textured soils on flat slopes. In this demonstration, we divided the Pecan Creek Watershed into two zones based on a combination variable of soil texture and slope. The zones were delineated by selection of soil mapping units from Natural Resources Conservation Service (NRCS) county-level soil mapping data. Each zone was assigned a numerical value (10 or 0) for being in or out of the preferred soil/slope zone.

While this process could continue for additional variables, we used these three criteria for the current illustration. The result is a classification of the Pecan Creek Watershed into 8 possible zones or regions with varying potential to generate enhanced water yields based on the criteria applied (Figure 5). Numerically, the "scores" range from a low of 10 to a high of 45. The highest priority areas or sites would be within 7 miles of the watershed outlet, on clay or clay loam soils with slopes from 1 to 3 percent, and within 100 meters of a defined channel (Figure 6).

The series of maps was provided to graphically display the process employed, the results of each criterion application, as well as the final result. It must be remembered that this is simply a demonstration of the methodology; the weighting values and the zone definitions within a criteria were arbitrarily chosen to help illustrate the method. Their exact values and definitions would need to be determined for each individual watershed to which the methodology is applied based on the best science available for the specific site.

Monitoring Considerations

While preparing this brief report, the authors reviewed existing data for several of the TSSWCB's treated sites. This information included the reports of the SWAT modeling that estimated the potential added streamflow from proposed brush control, nearby historical streamflow measurements when available, and existing monitoring efforts for streamflow and

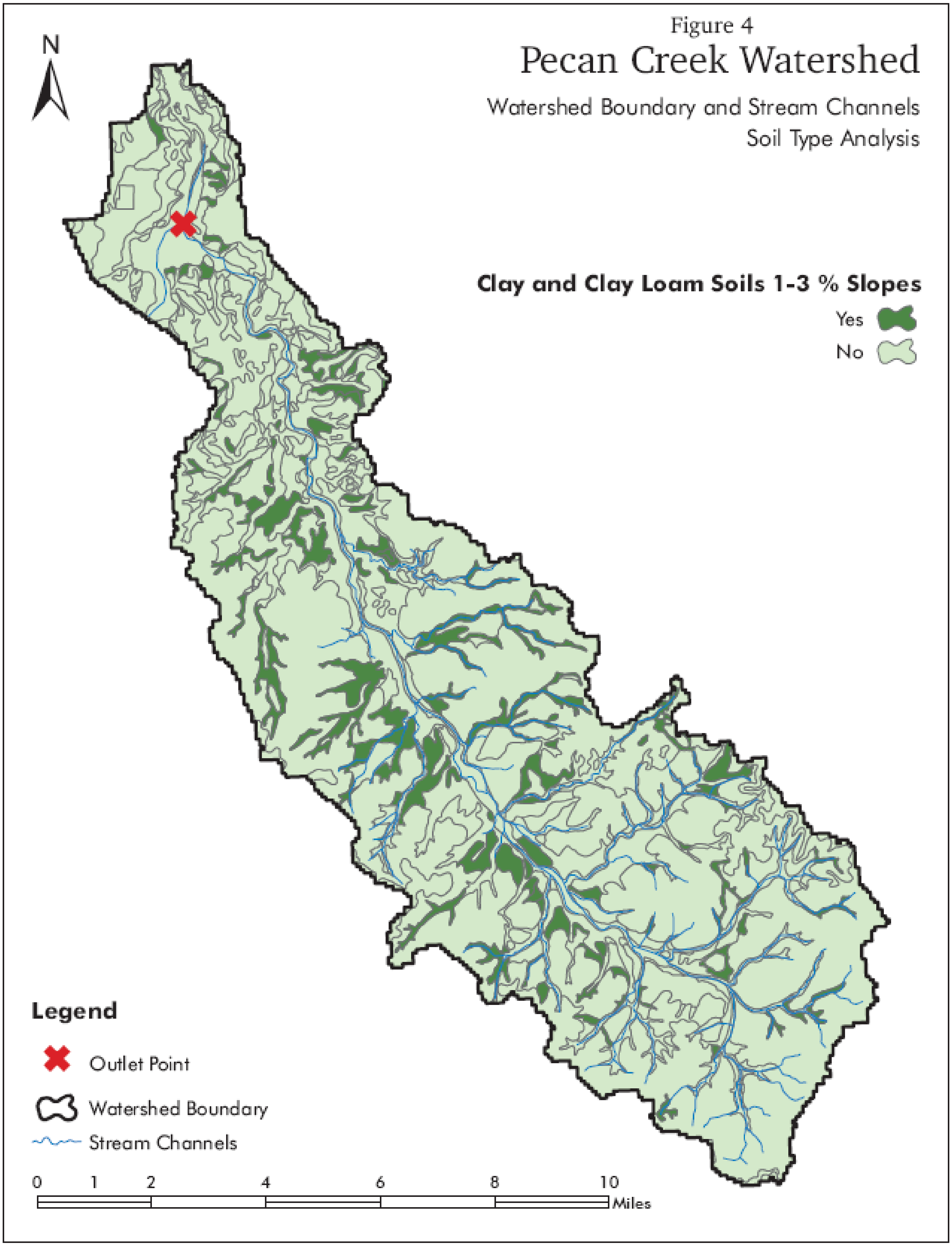
Figure 4
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Soil Type Analysis
N
Clay and Clay Loam Soils 1-3 % Slopes
Yes
No
Legend
Outlet Point
Watershed Boundary
Stream Channels
0 1 2 4 6 8 10
Miles

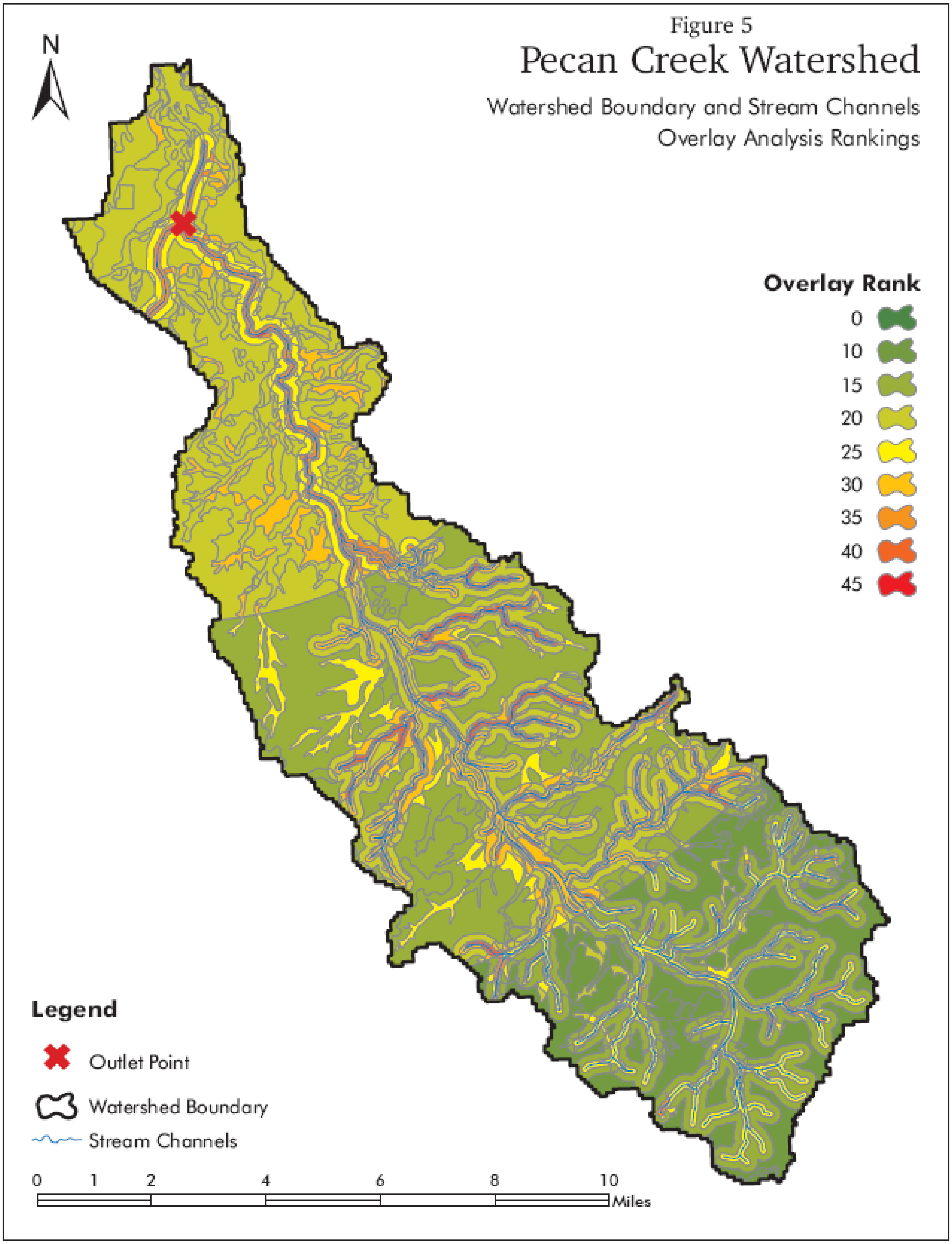
Figure 5
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Overlay Analysis Rankings
N
Overlay Rank
0
10
15
20
25
30
35
40
45
Legend
Outlet Point
Watershed Boundary
Stream Channels
0
1
2
4
6
8
10
Miles

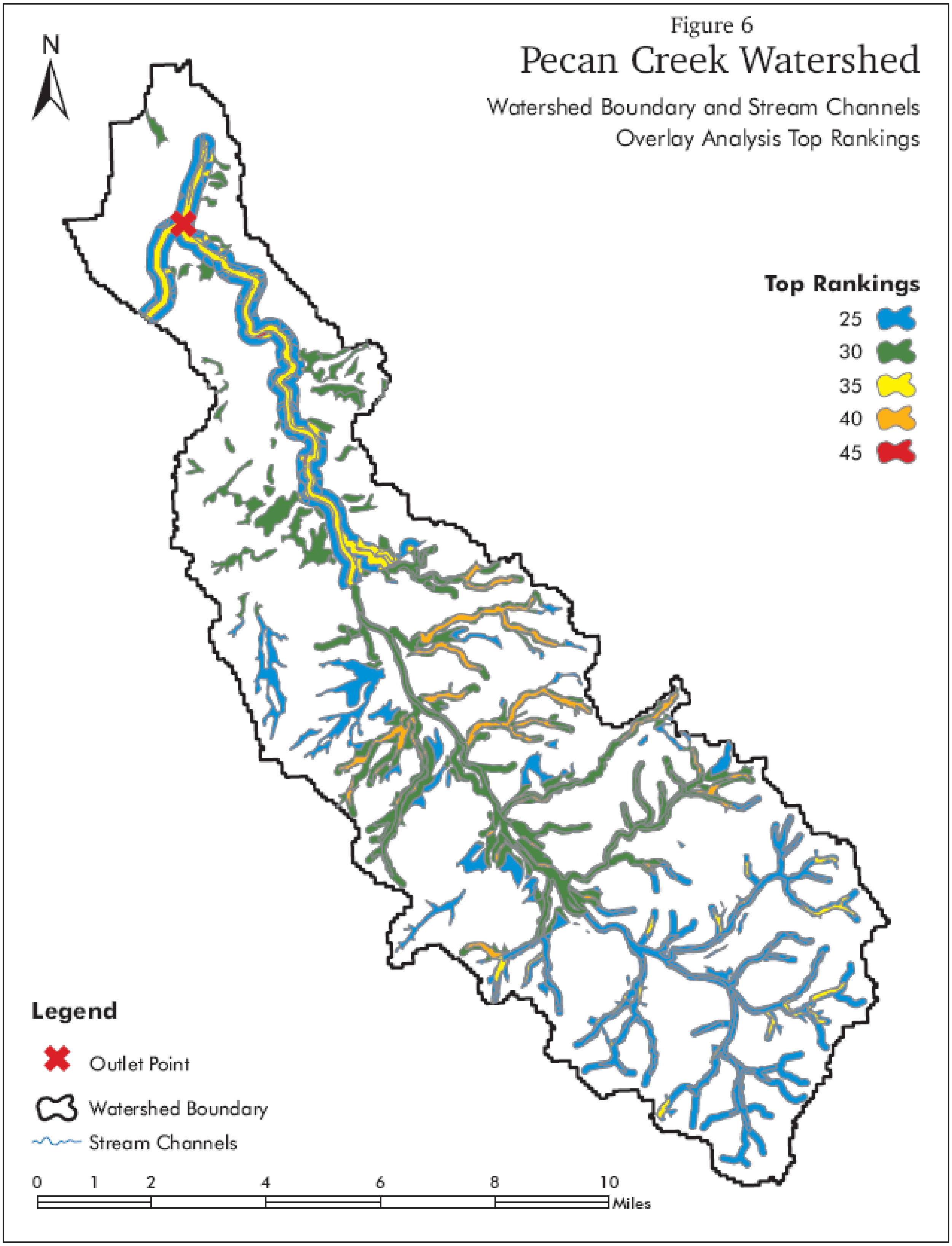
Figure 6
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Overlay Analysis Top Rankings
N
Top Rankings
25
30
35
40
45
Legend
Outlet Point
Watershed Boundary
Stream Channels
0
1
2
4
6
8
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Miles

groundwater impacts after brush treatment. It is apparent that the same type of hydrologic and watershed management concepts used to generate the subwatershed selection criteria are also pertinent for monitoring considerations.

The major concern of the program is enhancement of streamflow. In order to measure such flows, it is necessary to install continuous streamflow recorders at the outlets of the treated subwatersheds. It would be best to have both pre- and post-treatment data to demonstrate the ranges of flow values. The typical flow recording system would most likely be a water level sensor, such as a pressure transducer, installed at a fixed channel cross-section, such as a paved low-water crossing, broad-crested weir, or a fixed measuring flume. The system would have a relationship between water surface elevation in the stream and flow rate, and allow continuous data collection so that baseflow and runoff components could always be observed. Pressure transducers typically come with electronic data loggers that can be downloaded to laptop computers.

Continuous observation of rainfall is just as important as streamflow, so that the source of the runoff can be estimated. Multiple recording rain gauges, such as the tipping bucket type that can sense rainfall to the nearest 0.01 in, should be placed at strategic locations across the watershed to allow estimation of the areal and temporal distribution of rainfall for each storm event. These rain gauges can store data in data loggers for occasional downloading to laptop computers.

Observation of local groundwater conditions should be done through monitoring wells in the shallow alluvial aquifer in and near the streambed. The elevations of the groundwater table in the monitoring wells can be compared each other and to the elevation of the water surface in the stream to demonstrate which way the groundwater is flowing and the changes in groundwater storage over time. The groundwater levels can be continuously monitored with pressure transducers, or manually measured less often if readily accessible.

Estimation of evapotranspiration losses through vegetation within the target areas of the treated subwatershed can be done by using site visits, aerial photography, and satellite imagery to identify the effectiveness of brush management over the treated areas of the subwatershed. Potential ET can be estimated with local weather stations that measure and record wind speed, relative humidity, net solar radiation, and temperature. Actual ET can then be estimated as proportional to the potential ET based on plant type and seasonal variations in water consumption.

The best situation for application of hydrologic monitoring to confirm positive impacts of brush control would be to have several years of pre-treatment data to compare to several years of post-treatment data. Unfortunately, this situation is unlikely for the subwatersheds that have already been or will soon be treated. It is possible that two similar subwatersheds can be selected, instrumented, and observed with one receiving brush treatment and the other left untreated. The hydrologic behaviors of the two subwatersheds over several years could then be later compared to determine the impact of treatment. An example of this type of situation is in the East and West Grape Creek subwatersheds near San Angelo.

An over-riding concern about hydrologic monitoring for streamflow enhancement, or any other purpose, is that the longer the observation period is, the more confident we are in the findings. Installation of equipment to measure streamflow often seems to cause a drought. We encourage all those concerned with streamflow enhancement, whether through brush control or other watershed management techniques, to be patient and allow multiple years of data collection and analyses to observe a reasonable range of weather conditions over time.

References

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