ATTACHMENT 15

A copy of recent studies on the TSSWCB or any of its functions conducted by outside management consultants or academic institutions

Bacteria Total Maximum Daily Load

Task Force Report

June 4, 2007

Prepared for:

Texas Commission on Environmental Quality

and

Texas State Soil and Water Conservation Board

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Executive Summary

In September 2006, the Texas Commission on Environmental Quality (TCEQ) and Texas State Soil and Water Conservation Board (TSSWCB) charged a seven-person Bacteria Total Maximum Daily Load (TMDL) Task Force with:

- examining approaches that other states use to develop and implement bacteria TMDLs,
- recommending cost-effective and time-efficient methods for developing TMDLs,
- recommending effective approaches for developing TMDL Implementation Plans (I-Plans),
- evaluating a variety of models and bacteria source tracking (BST) methods available for developing TMDLs and I-Plans, and recommending under what conditions certain methods are more appropriate, and
- developing a roadmap for further scientific research needed to reduce uncertainty about how bacteria behave under different water conditions in Texas.

The Task Force, assisted by an Expert Advisory Group of approximately 50 stakeholders and agency staff, held two two-hour meetings/teleconferences and developed two drafts of the report. These drafts were shared by e-mail and on a Web site and feedback received from the Expert Advisory Group was also made available on the Web site.

The Task Force report describes the characteristics, as well as some of the strengths and weaknesses of several models that have been used and/or are under development to assist bacteria TMDL and I-Plan analysis. These include:

- load duration curves (LDC),
- spatially explicit statistical models, including Arc Hydro, SPARROW and SELECT,
- the mass balance models BLEST and BIT, and
- the mechanistic hydrologic/water quality models HSPF, SWAT, SWMM and WASP.

The Task Force report also describes and makes recommendations for effective use of BST methods that have been used in Texas and elsewhere for TMDL development. These include ERIC-PCR, Ribotyping, PFGE, KB-ARA, CSU and *Bacteroidales* PCR. Based on recent experience in Texas and elsewhere, the Task Force recommends using library-independent methods like *Bacteriodales* PCR for preliminary qualitative analyses and more expensive and time-consuming library-dependent methods if more quantitative data are required for TMDL or I-Plan development.

Based on the discussions of bacteria models and source tracking, as well as extensive input from the Expert Advisory Group, the Task Force recommends a three-tier approach to implementing bacteria TMDLs and I-Plans.

Tier 1 is a one-year process that includes the formation of a representative stakeholder group, development of a comprehensive geographic information system (GIS) of the watershed, a survey of potential bacterial sources, calculation of load duration curves from existing monitoring data and analysis by agency personnel and stakeholders of data collected for Tier 1. After reviewing information from Tier 1, the group may choose to complete and submit a draft TMDL for agency approval, request an evaluation of the designated use of the water body (an use attainability analysis) or proceed to Tier 2.

Tier 2 is a one-to-two-year effort designed to collect targeted monitoring data to fill gaps in previously collected data, conduct qualitative library-independent BST data to determine whether humans and/or a few major classes of animals are sources and develop simple spatially explicit or mass balance models of bacteria in the watershed. After analysis of Tier 1 and Tier 2 data, the group may chose to complete and submit the draft TMDL (or I-Plan if a TMDL was developed after Tier 1), request an evaluation of the designated use (an use attainability analysis), or initiate a “phased TMDL” and proceed with Tier 3 analysis.

Tier 3 is a two-to-three-year process designed to continue strong stakeholder involvement, implement more extensive targeted monitoring, conduct quantitative

library-dependent BST analysis and develop a detailed hydrologic/water quality model for the watershed. Tier 3 should be implemented only when this level of detailed analysis is needed for I-Plan development or for TMDL development for particularly complex watersheds for which consensus cannot be reached after Tier 2.

The Task Force emphasizes that the agencies and stakeholders may choose to deviate from these recommendations if they reach consensus that a more time- and cost-effective approach is feasible.

The Task Force concludes its report by summarizing a number of research activities needed to strengthen the scientific tools available for TMDL and I-Plan development. The needed research falls into the following categories: characterization of sources, characterization of kinetic rates and transport mechanisms, enhancements to bacteria fate and transport models and bacteria source tracking, determination of effectiveness of control mechanisms and quantification of uncertainty and risk.

Finally, the report includes a number of references to relevant scientific literature and studies, as well as five appendices:

Appendix 1	Bacteria TMDL Task Force Members and Expert Advisors
Appendix 2	Models Used in Bacteria TMDLs as Described in EPA Publications
Appendix 3	EPA Bacteria TMDL Guidelines
Appendix 4	State Approaches to Bacteria TMDLs
Appendix 5	Comments from the Expert Advisory Group

Introduction

As of January 2006, 197 water bodies in Texas were impaired because they did not meet bacteria criteria established by the state to protect contact recreation use (freshwater and saltwater) and/or oyster water use. The freshwater contact recreation use criterion used to determine impairment includes both a geometric mean for *Escherichia coli* (*E. coli)* of 126 colonies per 100 ml and a single sample maximum of 394 colonies per 100 ml. The saltwater contact recreation use criterion includes both a geometric mean for *Enterococci* of 35 colonies per 100 ml and a single sample maximum of 89 colonies per 100 ml. Finally, the oyster water use criterion includes a median fecal coliform concentration of 14 colonies per 100 ml and no more than 10% of samples may exceed 43 colonies per 100 ml. TCEQ is currently considering a variety of potential changes to Texas water quality standards. According to the TCEQ Web site, these "revisions to the standards and Standards Implementation Procedures are needed to:

- incorporate recently developed site-specific standards for individual water bodies,
- incorporate new research on the toxicity of specific chemicals,
- improve the way that standards are used and applied."

As required by Section 303(d) of the Clean Water Act, Texas has committed to complete TMDLs for these bacteria-impaired water bodies within 13 years of the listing date (i.e. 2017 for new bacteria impairments listed on the 2004 list). In order to identify the best and most cost- and time-effective methods to develop bacteria TMDLs and TMDL Implementation Plans (I-Plans), TCEQ and TSSWCB established a joint technical Task Force on Bacteria TMDLs on September 27, 2006. The Task Force was charged with:

- examining approaches that other states use to develop and implement bacteria TMDLs,
- recommending cost-effective and time-efficient methods for developing TMDLs,
- recommending effective approaches for developing I-Plans,

- evaluating the variety of models and bacteria source tracking (BST) methods available for developing TMDLs and I-Plans, and recommending under what conditions certain methods are more appropriate, and
- developing a roadmap for further scientific research needed to reduce uncertainty about how bacteria behave under different water conditions in Texas.

Task Force members are Drs. Allan Jones, Texas Water Resources Institute; George Di Giovanni, Texas Agricultural Experiment Station–El Paso; Larry Hauck, Texas Institute for Applied Environmental Research; Joanna Mott, Texas A&M University–Corpus Christi; Hanadi Rifai, University of Houston; Raghavan Srinivasan, Texas A&M University; and George Ward, The University of Texas at Austin. Dr. Allan Jones was named Task Force Chair by TCEQ and TSSWCB.

Approximately 50 Expert Advisors (Appendix 1) with expertise in bacteria-related issues have also provided significant input to the Task Force during the process. Included in this group are university scientists, environmental consultants and representatives of local, state and federal agencies with jurisdictions impacting bacteria and water quality.

Recommendations from the Task Force are intended to be used by the State of Texas, specifically TSSWCB and TCEQ, to keep Texas as a national leader in water quality protection and restoration.

Bacteria Fate and Transport Models

This section, coordinated by Drs. Hanadi Rifai and Raghavan Srinivasan, describes the strengths and weaknesses of several bacteria fate and transport models that have been used for TMDL and I-Plan development. The discussion is not intended to be a comprehensive review of the numerous models that have been or could be used for bacteria TMDL and I-Plan development. Instead, it briefly characterizes a few of the models that have been developed and/or used in Texas to assist in bacteria TMDL and I-Plan assessments.

Bacterial pollution in surface water bodies is difficult to model because bacterial sources and their fate and transport are complex. In addition, several fate and transport processes control their growth, decay and movement on the land and in streams. Indicators such as *E. coli, Enterococcus spp.*, and fecal coliform bacteria, although typically nonpathogenic, are used to identify the potential for the presence of other disease-causing organisms. These bacteria typically originate from mammalian and avian sources and are released into water bodies via point sources (such as wastewater treatment plant [WWTP] effluent and runoff from stormwater drainage networks) as well as dispersed (or nonpoint) sources (such as direct runoff from residential yards and streets, on-site sewage disposal, deposition from non-domestic animals and livestock, and re-suspension of bacteria from stream sediment). Bacteria are present in both water and sediment and experience survival, growth and decay within a water body. Furthermore, bacteria concentrations in streams vary spatially and temporally because of flow variability within the stream network and loads entering the streams from various sources at different times. Because of this complexity, most states use mathematical models to understand bacteria dynamics when developing bacteria TMDLs and I-Plans. Since numerous water quality models are available, selecting an appropriate model or combination of models for bacteria TMDLs is a challenge. Both the characteristics of each watercourse and the nature of its pathogen loads should be considered. Selection of a model or models is an important and critical step that should be undertaken early in the TMDL and I-Plan development processes in consultation with stakeholders and modeling experts.

Since bacteria TMDLs must estimate the maximum bacteria load that a water body can receive and still meet water quality standards, TMDL development involves estimating both existing and allowable loads, the instream water quality effects due to these loads, as well as the reductions that would be required to meet standards. I-plan development, however, involves designing realistic bacteria reduction strategies for different bacteria sources and examining the effects of these strategies on water quality. The different goals of TMDL development and I-Plan development may require the use of different bacteria models with different levels of complexity.

Efforts to formulate useful TMDLs have led to the development of many predictive tools for the estimation of necessary reductions to meet water quality goals. States required to develop TMDLs as part of consent decrees have been under a great deal of pressure to produce TMDLs quickly and cost-effectively to comply with federal law. States faced with budgetary constraints need an economical means to develop TMDLs to restore impaired waters. As a result, relatively simple and cost-effective approaches to identify sources of pollution and allocate loads are needed to identify bacteria load reductions to meet water quality standards.

These approaches include load duration curves, spatially explicit statistical models and mass balance models that rely on available flow and water quality data. In other cases states have chosen to use more complex mechanistic (process or physically based) hydrologic/water quality models combined with landscape-loading models. Some of the most common models used for bacteria TMDLs and I-Plans are described below. Other modeling tools described in EPA publications are summarized in Appendix 2.

Load Duration Curve (LDC) Models

Load Duration Curve (LDC) methodologies are acknowledged as a cost-effective, useful tool for addressing bacteria impairments because they are easy to understand, produce reasonable results and have minimal data requirements (Cleland 2002 and 2003, Bonta 2002, Stiles 2001).

LDCs graphically represent streamflow data in terms of pollutant loadings. The analysis begins with a flow duration curve where the x-axis is based on the frequency of exceedance of specific flows (y-axis) during the entire period of record represented in the data (Figure 1). The resulting graph depicts the range of flows (expressed as an exceedance frequency) experienced at a single monitoring over time.

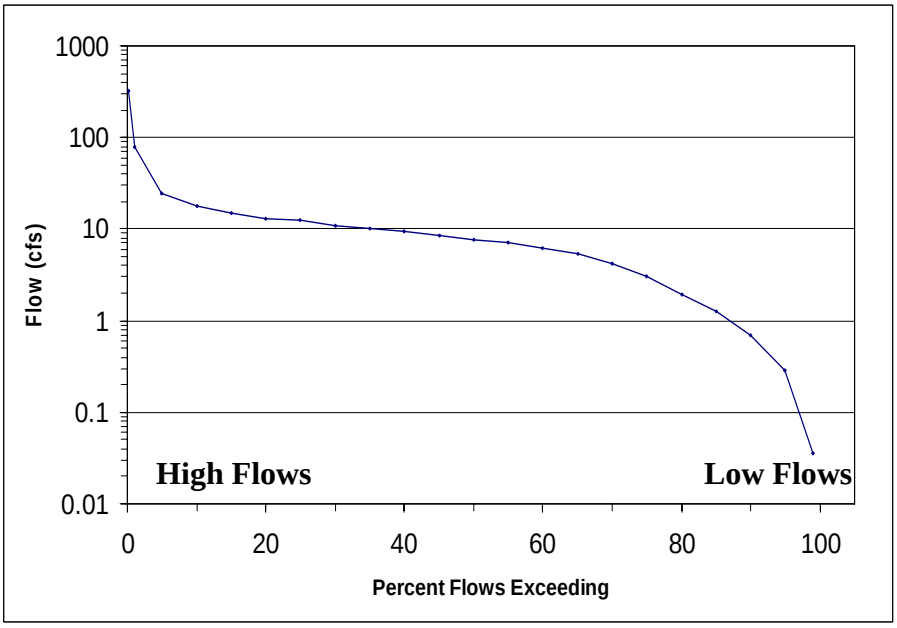

Figure 1: Example flow duration curve

In order to make this approach useful for TMDL purposes, it is necessary to convert the flow duration curve to an LDC. This conversion is accomplished by multiplying the flow (at each frequency interval) by the water quality criterion (Figure 2), in this case the single sample criterion. The resulting plot represents the maximum pollutant load for every flow experienced in the specific stream, in essence the total maximum daily load. The plotted line in Figure 2 is equal to the water quality criterion (in this case, for a single sample) times the flow.

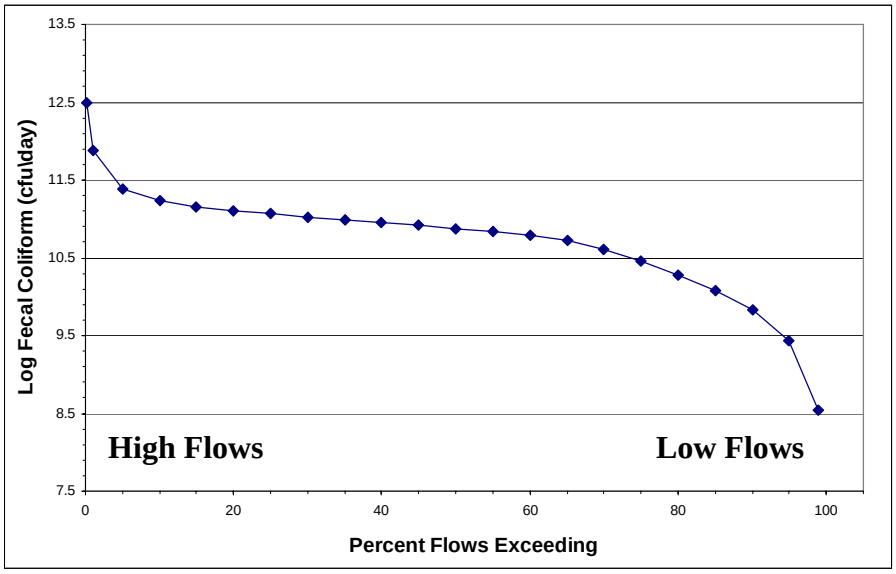

Figure 2: Example load duration curve using the single sample criterion

Monitoring data can be added to the graph to identify those flow conditions where pollutant levels may be above allowable loadings. This step is done by multiplying the water quality sample bacteria concentrations by the daily average flow that occurred when the sample was taken. The data points can then be plotted at the appropriate flow frequency and compared directly to the relevant water quality criterion obtained from the LDC (Figure 3).

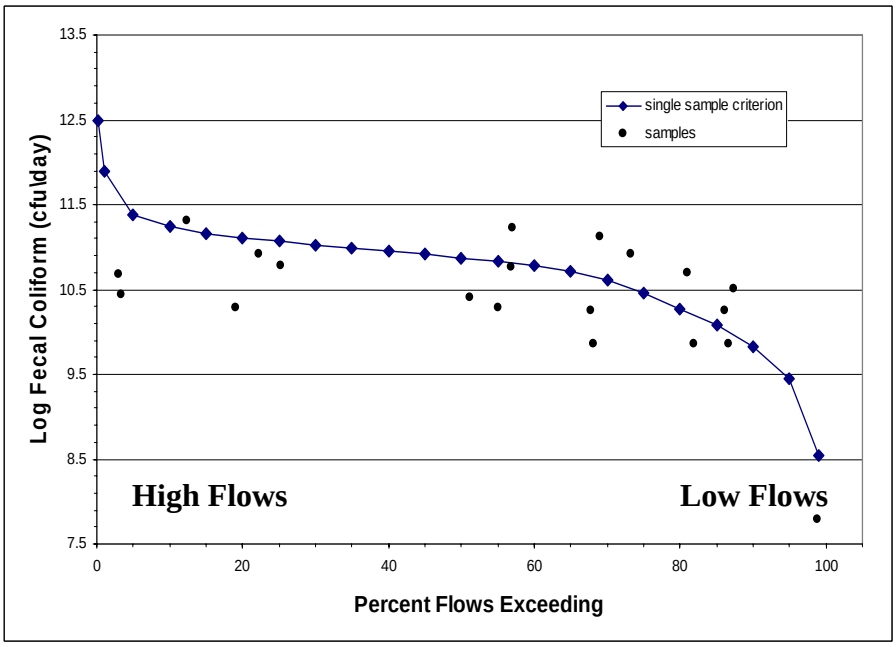

Figure 3: Data assessment using load duration curve

LDCs provide several types of useful information for the TMDL development process. An LDC can differentiate between point and nonpoint sources of bacteria related to different flow conditions (Figure 4). Generally speaking, monitored loads that exceed the allowable load at low flows (i.e., high frequency of exceedance) are likely to be the result of point source discharges, such as a WWTP discharging bacteria into a low flow stream. Resident populations of waterfowl could also contribute to chronic exceedances during low flows. In contrast, monitored loads exceeding the criteria at the mid-range and high flows are typically caused by nonpoint source inputs or point sources (such as stormwater systems) that channel rainfall runoff into water bodies. An example might be surface runoff carrying bacteria from livestock or non-domestic animal sources into a high flow stream. Finally, resuspension of indicator bacteria surviving in streambed sediments could produce exceedances under both high and low flow conditions.

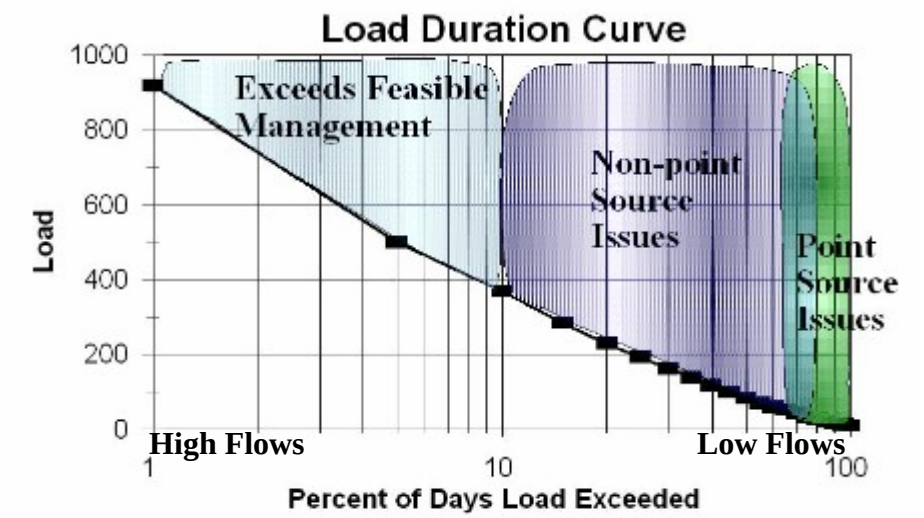

Figure 4: Source identification using load duration curve (Cleland 2003)

Developing load reduction scenarios based upon the LDC requires the examination of relationships between the data and the allowable loads expressed in the curve. This usually requires comparisons between statistical representations of the data (means,

regression lines, confidence intervals) and the LDC, with the difference between the two values representing the required reduction. Statistical estimates of the data may include all of the loads or just those that exceed the TMDL line.

The segmentation of the LDC allows for the development of appropriate implementation strategies that target specific flow conditions (Cleland 2002). Exceedances occurring at the low flows may require regulatory actions to control point sources. At the mid range and high flows, management measures directed towards nonpoint sources could be developed. At some point in the flow frequency, control of pollutant sources becomes unfeasible. Pollutant loadings at these high flow events typically exceed design specifications for control actions. For this reason, it may be reasonable to exclude data and loadings that occur at flooding conditions.

Several states have developed bacteria TMDLs using LDCs and have produced reasonable scenarios to address impairments. Oklahoma developed TMDLs for bacteria in the Upper Canadian River in several nonpoint source-dominated watersheds. In Maryland, a TMDL for bacteria was developed for Cabin John Creek. Maryland also used BST to further refine source loadings. Kansas used the load duration methodology to develop a bacteria TMDL for the Kansas River. Texas is currently developing several TMDLs using LDCs to address bacteria impairments (Upper Oyster Creek, Gilleland Creek, Guadalupe River above Canyon Lake, among others).

Recognizing the importance and practical use of LDCs, the U.S. Environmental Protection Agency's Watershed Branch in the Office of Wetlands, Oceans and Watersheds is producing a detailed technical document on how to use LDCs for establishing TMDLs. The draft of "An Approach for Using Load Duration Curves in Developing TMDLs" is available on the Task Force's Web site at: http://twri.tamu.edu/bacteriatmdl/.

As with all predictive tools used in TMDL development, the LDC approach has both strengths and weaknesses. The primary strengths of this approach are the minimal data

requirements (although large datasets are preferable in all cases), its simplicity and its usefulness as an illustrative model. The model's simple nature can also be considered a weakness in that very few inputs are considered for estimating resulting bacteria loads, potentially increasing uncertainty, though this could also be considered a strength in that other uncertainties are reduced by using fewer assumptions. This method also does not work very well in tidally influenced areas, and intermittent streams tend to produce truncated curves. Though the load duration approach can help differentiate between point and nonpoint sources, more mechanistic models provide more detailed assessments of specific sources of bacteria loads. On the other hand, specific source identification may not be necessary since a TMDL, in its most basic form, only requires differentiation between point (Waste Load Allocation [WLA]) and nonpoint (Load Allocation [LA]) sources.

Further refinements may be desirable for developing measures to control sources in the I-Plan following the TMDL. Additional tools (targeted monitoring, bacteria source surveys, BST and more complex models) used in conjunction with load duration methods have the potential to significantly refine source identification and increase the power of this analytical tool.

Spatially Explicit Statistical Models

The three models described below estimate the spatial distributions of potential pollutant sources throughout a watershed, estimate loadings from each source into water bodies and calculate the change in pollutant concentration within the water body. Key parameters are calibrated to mimic pollutant concentrations measured in the watershed.

Arc Hydro Model (Maidment)

The Arc Hydro Model, developed by Dr. David Maidment of the Center for Research in Water Resources at The University of Texas at Austin, can be defined as a geographic database containing a geographic information system (GIS) representation of a hydrological information system. Arc Hydro uses a case-specific database design that is

extensible, flexible and adaptable to user requirements. It takes advantage of the next generation of spatial data in relational database management systems (an RDBMS-based GIS system), the geodatabase model. Conceptually, it is a combination of GIS objects enhanced with the capabilities of a relational database to allow for relationships, topologies and geometric networks.

This model has been used to develop bacteria TMDLs for Galveston and Copano Bays (Gibson, C.J. et al., 2006) The Copano Bay TMDL model input was developed within a GIS framework. Point sources were located in the GIS and accounted for in the model. Nonpoint source loadings were estimated from livestock, non-domestic animals and domestic animal populations per county. These populations were assumed to be equally distributed within their appropriate land use categories within each county. Loadings were then calculated by combining population, geographic and precipitation data. This information was used to estimate bacterial loadings for each watershed.

Septic systems were accounted for in a similar manner. It was determined, however, that systems located immediately adjacent to the bay were of particular interest. Increased efforts were made, therefore, to accurately account for systems in these areas.

A first order decay rate was assumed for bacterial degradation as pollutants moved through the watershed, into streams/channels and eventually into the bay. Once in the bay, degradation was modeled as four independent continuous flow, stirred tank reactors.

Once the loading to the bay was obtained, a Monte Carlo simulation was used to determine the necessary load reductions. Monte Carlo simulations use random number probability distributions to simulate random behavior, providing estimates of uncertainty in a given outcome. Accurate estimates of measured bacteria concentrations only required calibration of stream residence times and a multiplication factor.

SPAtially Referenced Regressions On Watershed attributes (SPARROW)

SPARROW is maintained by the U.S. Geological Survey. It spatially references various watershed components, such as stream monitoring data, pollutant sources, etc., to surface

water flow paths that are defined by a digital drainage network. It then imposes mass balance constraints to empirically estimate terrestrial and aquatic rates of pollutant flux. Applications of SPARROW include estimation of the spatial distributions of pollutant yields, pollutant sources and the potential for delivery of those yields to receiving waters. This information can be used to:

- predict ranges in pollutant levels in surface waters,
- identify the environmental variables that are significantly correlated to the pollutant levels in streams,
- evaluate monitoring efforts for better determination of pollutant loads, and
- evaluate various management options for reducing pollutant loads to achieve water-quality goals.

SPARROW has been used previously to estimate the quantities of nutrients delivered to streams and watershed outlets from point and nonpoint sources over a range of watershed sizes (Smith et al., 1997; Alexander et al., 2000, 2001; Preston and Brakebill, 1999). This approach can be used for bacteria TMDLs because it not only uses process-based models to simulate transport of pollutants, but it also uses the actual historical monitoring data and known predictor variables to predict the various model input parameters. In this manner, a more realistic model can be developed that closely describes the conditions of the particular watershed (Schwarz, et al., 2006).

Spatially Explicit Load Enrichment Calculation Tool (SELECT)

SELECT (Teague 2007a, 2007b) spatially references the sources of bacteria contamination and is being developed under an ArcGIS 9 environment. SELECT will calculate and allocate bacteria loading to a stream from various sources in a watershed. All loads will be spatially referenced. In order to allocate the bacteria load throughout the watershed, estimations of the source contributions will be made. This, in turn, allows the sources and locations to be ranked according to their potential contribution. The populations of livestock, non-domestic animals and domestic pets will be calculated and distributed throughout the watershed according to appropriate land use. Furthermore, point sources will be identified and their contribution quantified based on flow and

outfall concentration. Septic system contribution will also be estimated based on criteria including distance to a stream, soil type, failure rate and age of system. Once the watershed profile is developed for each potential source, the information can be aggregated to the sub-watershed level to identify the top contributing areas.

Mass Balance (MB) Methods

These methods, as the name implies, calculate a mass balance between bacteria loads entering the water body and bacteria loads within the stream. Sources are typically inventoried, quantified and compared to existing and allowable in-stream loads at specified points within the stream (typically, where the TMDL is sought) for different flow conditions. MB methods require more data than the LDC method, but are more amenable for use in TMDL implementation. These methods have typically been developed using spreadsheets. The main advantages of the MB methods are that they can be used for tidal and non-tidal water bodies, including both TMDL and I-Plan development. In addition, they can be used for watersheds where both point and nonpoint sources appear to contribute at both low flow and high flow conditions. The main disadvantage is that MB methods, like the LDC method, are static and do not consider temporal variations in loading. The MB method accounts for spatial variations since it estimates the various sources within the watershed. However, estimates of nonpoint source loadings may be difficult to obtain. The output from these methods can often be used in the development of more complex, mechanistic models discussed in the next section.

Bacteria Load Estimator Spreadsheet Tool (BLEST)

In Texas, one of the more recent MB applications is described in Petersen (2006) in which BLEST was used to calculate bacteria loads from all sources and land uses on a subwatershed basis for Buffalo and White Oak Bayous. The loads were accumulated by segment and calculated for low, median and high flow conditions in a stream. Sources include WWTPs, septic tanks, nonpoint source runoff, sanitary sewer overflows and bypasses, sewer leaks and spills, in-stream sediment and non-domestic animals and

domesticated animals. BLEST was used to calculate existing loads and allowable loads and to estimate the load reductions that would be required to meet the standard. The limitations of this tool include the extensive amount of inventory data required, the use of literature values for some of its variables (for example, the rate of failure for septic systems) and the absence of attenuation or loss mechanisms that might affect loads before they reach the water body. Additionally, and due to its static nature, temporal variations in the source loads are not modeled.

Tools similar to BLEST have been developed elsewhere; for example, the Bacteria Source Load Calculator (BSLC) from the Center for TMDL and Watershed Studies at Virginia Tech (http://www.tmdl.bse.vt.edu/outreach/C85/).

Bacteria Indicator Tool (BIT)

Another MB tool is the BIT provided by EPA (http://www.epa.gov/waterscience/ftp/basins/system/BASINS3/bit.htm). The BIT is a spreadsheet that can be used to estimate the monthly accumulation rate of fecal coliform bacteria on four land uses (cropland, forested, built-up and pastureland). The tool also estimates the direct input of fecal coliform bacteria to streams from grazing agricultural animals and failing septic systems. Output from BIT can be used as input to the Hydrological Simulation Program Fortran (HSPF) water quality model in BASINS (Better Assessment Science Integrating Point and Nonpoint Sources). BIT has been used in Texas.

Mechanistic Hydrologic/Water Quality Bacteria Models

A number of simulation models have been used to describe (in mathematical form) the mechanisms of water movement as well as loading and transport of pollutants. Both researchers and managers desire to have a means to create scenarios to simulate environmental outcomes in response to specific management practices. Models are used to predict the water quality in a water body based upon changes in pollutant loading and various allocation strategies. Current models have been adapted for use in bacteria TMDLs from other models originally designed for more conventional pollutants

(suspended sediments, nutrients, dissolved salts). These models can be used for both TMDL and I-Plan development to evaluate spatial and temporal variation of bacteria loading within a watershed. These models, however, have extensive data requirements and their level of sophistication requires a significant investment of resources for parameterization, calibration and training of personnel to use the models effectively.

In-stream water quality models are either steady-state or transient and are hydrologically driven (via rainfall) or hydrodynamically driven (via velocities in the water body). A steady-state model does not allow for variations over time and averages water quality over time. A dynamic or transient model, on the other hand, allows for changes over time and can be used to estimate bacteria loads and concentrations at different points in time anywhere in the stream.

Ward and Benaman (1999) identified a number of models as being appropriate for use in Texas TMDLs. Their list includes: ANSWERS, CE-QUAL-W2, DYNHYD, EFDC, GLEAMS, HSPF, POM, PRMS, QUALTX, SWAT, SWMM, TxBLEND and WASP. Their assessment categorized these models based on the watercourse type and the scale of resolution for time. For example, HSPF, SWAT, PRSM, SWMM and ANSWERS were characterized as watershed type models that can be used for "slow time variation" and "continuous time variation."

Of the above list of models identified by Ward and Benaman (1999) for use in Texas TMDLs, the most commonly used for bacteria include HSPF, SWAT, SWMM and WASP. HSPF has been the most commonly used of the four, but EPA specifically requested U.S. Department of Agriculture – Agricultural Research Service (USDA–ARS) scientists in Temple, Texas to develop bacteria fate and transport components for SWAT to facilitate its use in TMDL and I-Plan development. These models all require many input variables, a substantial cost and time investment in set-up, calibration and validation time and have a steep learning curve.

Hydrological Simulation Program – FORTRAN (HSPF)

HSPF has been in extensive use since the 1970s, and is distributed by EPA's Center for Exposure Assessment Modeling. This watershed hydrologic model has been commonly used for TMDL development for a variety of conventional water quality parameters in Texas (for indicator bacteria and for dissolved oxygen) and also in other states. The required data include land use; watershed and subwatershed boundaries; location and data for rainfall gages and surface water quality monitoring stations; detailed descriptions of stream geometry and capacity; detailed information about sources within the watershed; sedimentation and re-suspension characteristics and bacteria die-off rates, to name a few. Development of an HSPF model for a given watershed is both complex and time-consuming and involves substantial calibration and validation. HSPF can be used for most types of watersheds (except possibly tidally influenced streams) regardless of the land use, but it requires extensive input from hydrologic and hydraulic models as well as GIS data layers. HSPF allows for a detailed spatial resolution within the watershed and allows for estimation of bacteria loads from runoff from the land surface as well as re-suspension from the streambed and from direct deposition sources. Disadvantages include the inherent difficulty in its application, its poor documentation and inadequate simulation of bacteria fate and transport processes. For example, transport of bacteria associated with sediment, sedimentation and re-suspension, regrowth and die-off processes are simplified and modeled using variables that difficult to measure in field settings. These variables are commonly treated as calibration variables during model development.

Soil and Water Assessment Tool (SWAT)

The SWAT model is a continuation of nearly 30 years of modeling efforts conducted by the USDA-ARS at Temple, Texas. SWAT has gained international acceptance as a robust interdisciplinary watershed modeling tool as evidenced by regular biennial international SWAT conferences (Gassman 2005a), SWAT-related papers presented at numerous other scientific meetings, and dozens of articles published in peer-reviewed journals. The model, like HSPF, has been adopted as part of the EPA Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) software package and is being used by

many federal and state agencies, including the USDA within the Conservation Effects Assessment Project (CEAP). Reviews of SWAT applications and/or components have been previously reported, sometimes in conjunction with comparisons with other models (i.e., Arnold and Fohrer, 2005; Borah and Bera, 2003; Borah and Bera, 2004; Steinhardt and Volk, 2003; Gassman, et. al 2005b).

SWAT was developed primarily to estimate loads from rural and mainly agricultural watersheds; however, the capability for including impervious cover was accomplished by adding urban buildup/washoff equations from SWMM. At the request of EPA, a microbial sub-model was incorporated to SWAT for use at the watershed or river basin levels. The microbial sub-model simulates (1) functional relationships for both the die-off and regrowth rates and (2) release and transport of pathogenic organisms from various sources that have distinctly different biological and physical characteristics. SWAT has been used in Virginia and North Carolina for bacteria TMDL development.

Storm Water Management Model (SWMM)

SWMM was developed primarily for urban areas. It is most commonly used to simulate real storm events based on meteorological and watershed data, although it can also be used for continuous simulations. While SWMM was developed with urban watersheds in mind, it can be used for rural watersheds. The biggest advantage of SWMM is in its ability to model urban infrastructure including drains, detention basins, sewers and related flow controls. However, SWMM does not simulate the in-stream water quality. Linking it to WASP can circumvent this limitation. Perhaps the best application for SWMM is to characterize the bacterial pollution from the urban drainage infrastructure, but this limits its usefulness within a bacterial TMDL context to implementation rather than development.

Water-quality Analysis Simulation Program (WASP)

This model is also distributed by EPA's Center for Exposure Assessment Modeling and is a well-established water quality model incorporating transport and reaction kinetics. Unlike HSPF, however, WASP is driven by flow velocity rather than rainfall, thus it is

usually coupled with a suitable hydrodynamic model such as DYNHYD, EFDC or SWAT. WASP is typically used for main channels, reservoirs and bays and estuaries and not for modeling watershed-scale processes. Problems studied using WASP include biochemical oxygen demand and sources of bacteria, dissolved oxygen dynamics nutrients and eutrophication, organic chemical and heavy metal contamination.

Important Considerations for Bacteria Modeling

- The expectations from using a model for TMDL development or implementation must be realistic and commensurate with the level of data and information available for the watershed in question. The model used will only be as good as the data used to develop it.
- Models should be used as part of the TMDL framework and not as an only tool for decision-making. Models should continually evolve as the knowledge base used in developing them changes.
- In-stream sediment settling and re-suspension processes are not well represented in most models available to date and their roles in bacterial concentrations in water bodies are poorly understood.
- Bacteria regrowth and decay are also not well represented in presently available models. Bacteria death is typically approximated using first-order expressions, and the first-order decay constant is determined from controlled laboratory and/or field experiments.
- Transient (time-varying) models such as HSPF provide bacterial concentrations on a very detailed time scale (minutes or hours), whereas most bacterial measurements are made much less frequently (once a week or once a month or once a quarter) thus complicating calibration and validation of the model.
- The models that are hydrologically driven such as HSPF are biased toward high flow conditions since rainfall is the main driver for flow in the water body. These models have to be fine-tuned to represent bacterial sources in dry weather conditions (under mostly effluent dominated conditions).

- The main advantage of simple models such as LDC, SELECT, BLEST or BIT is in determining required reductions to meet the standard.
- The main value of detailed models is that they allow for spatial and temporal analysis of different reduction strategies (i.e., BMPs) and their effectiveness in improving in-stream water quality.
- Sensitivity and uncertainty in data, parameters and models should be considered and assessed.
- The results of modeling exercises are heavily dependant on the precision of the model as determined by calibration activities. For this reason, calibration specifications for model application should be explicitly stated and standardized throughout all applications.

Table 1 (below) is a matrix describing the applicability and capability of typically used models.

Table 1 Bacteria Modeling Matrix

Model		LDC	Spatial Explicit Statistical Models			Mass Balance Models			Mechanistic/Hydrologic/WQ			
			ArcHydro	SPARROW	SELECT	BLEST	BSLC	BIT	HSPF	SWAT	SWMM	WASP
Watercourse Type	Watersheds		x	x	x	x	x	x	x	x	x	
	River/Stream	x	x	x	x	x	x	x	x	x		x
	Lake/Reservoir		x	x	x	x	x	x				x
	Fresh/Saltwater Estuarine		x	x	x	x	x	x				x
TMDL Phase	Development	x	x	x	x	x	x	x	x	x		x
	Implementation		x			x			x	x		x
Model Type	Analytical	x	x	x	x	x	x	x				
	Numerical								x	x	x	x
Spatial Dimensions	1-D			x	x				x	x	x	x
	2-D											x
	3-D											x
Time Scale	Steady-state			x						x		x
	Time Varying								x	x	x	x
	Single Storm Event				x				x	x	x	
	Continuous in time			x					x	x	x	x
Watershed Characteristics	Rural	x	x	x	x	x	x	x	x	x		
	Urban	x	x	x	x	x	x	x	x	x	x	
	Sediment transport			x	x				x	x		x
In-Stream Processes	Bacteria Regrowth											
	Bacteria Die-off			x					x	x		
	Settling								x	x		
	Re-suspension					x			x	x		
WLA Sources	WWTF			x	x	x			x	x		x
	Storm Sewers			x	x	x			x	x		x
LA Sources	Septic Tanks			x	x	x	x	x	x	x		
	Direct Deposition					x	x	x	x	x		x
	Bed Sediment					x			x			x
Cost		$	$$			$$			$$$			

Notes: 1. Shaded areas: not applicable.

Bacteria Source Tracking (BST)

This section, coordinated by Drs. George Di Giovanni and Joanna Mott, describes the strengths and weaknesses of several BST tools that have been used. The EPA has also issued a microbial source tracking guidance document (USEPA 2005) which provides technical details on many different BST methods, quality control measures, discussion of library size and representativeness, statistical analysis, project design and case studies. No consensus has been reached for most of these issues by the source tracking scientific community, and detailed discussions of these topics are outside the scope of this report. However, where appropriate, brief discussion of these issues have been included in this document and considered in the recommendations presented below.

The premise behind BST is that genetic and phenotypic tests can identify bacterial strains that are host specific so that the original host animal and source of the fecal contamination can be identified. Often *E. coli* or *Enterococcus spp*. are used as the bacteria targets in source tracking, as this provides a direct link with water quality standards which are usually based on one of these two indicators (for example, [Parveen, Portier et al. 1999; Dombek, Johnson et al. 2000; Graves, Hagedorn et al. 2002; Griffith, Weisberg et al. 2003; Hartel, Summer et al. 2003; Kuntz, Hartel et al. 2003; Stoeckel, Mathes et al. 2004; Scott, Jenkins et al. 2005]). While there has been some controversy concerning host specificity and survival of *E. coli* in the environment (Gordon, Bauer et al. 2002), this indicator organism has the advantage that it is known to correlate with a probability of gastrointestinal illness and is used for human health risk assessments. BST of *E. coli,* therefore, has the advantages of direct regulatory significance and availability of standardized culturing techniques for water samples, such as EPA's Method 1603 (USEPA 2005).

Description of Methods

There have been many different technical approaches to bacterial source tracking (Scott, Rose et al. 2002; Simpson, Santo Domingo et al. 2002; Meays, Broersma et al. 2004), but there is currently no consensus on a single method for field application. Genotypic (molecular) tools appear to hold promise for BST, providing the most conclusive characterization and level of discrimination for isolates. Of the molecular tools available, ribosomal ribonucleic acid (RNA), genetic fingerprinting (Ribotyping), repetitive element polymerase chain reaction (rep-PCR), and pulsed-field gel electrophoresis (PFGE) are emerging as a few of the versatile and feasible BST techniques. Antibiotic resistance analysis (ARA), a phenotypic characterization method, also has the potential to identify the human or animal origin of isolates, and variations of this technique have been applied in several BST studies. Carbon source utilization (CSU) is another phenotypic method that has been used in Texas, as well as other states, as a source tracking method.

Four years ago, a review of BST methods was performed before initiating the BST study for Lake Waco and Belton Lake. At that time, and still today, there is no consensus among BST experts as to the best approach or technique. The enterobacterial repetitive intergenic consensus sequence polymerase chain reaction (ERIC-PCR), RiboPrinting, pulsed field gel electrophoresis (PFGE) and Kirby-Bauer antibiotic resistance analysis (KB-ARA) were chosen for the Texas study because they appeared promising from previous BST and other scientific studies, and cover the spectrum in cost, ease of use, and discriminatory ability. This allowed researchers to not only evaluate the practical application of these methods for the identification of human and animal sources of fecal pollution, but also to perform a comprehensive comparison of the methods and composite data sets for further consideration and use in future studies. Additionally, CSU has been used in Texas in conjunction with KB-ARA. Each of these five methods has its strengths and weaknesses, which are described below. A disadvantage of each of these techniques is that reference libraries of genetic or phenotypic fingerprints for *E. coli* isolated from known sources (i.e., domestic sewage, livestock and non-domestic animals) are needed to identify the sources of bacteria isolated from environmental water samples. Thus, the

development of an identification library can be a time consuming and expensive component of a BST study. Libraries of isolates from various parts of Texas have been developed using these methods following the same protocols for comparability.

ERIC-PCR

Enterobacterial repetitive intergenic consensus sequence polymerase chain reaction (ERIC-PCR), a type of rep-PCR, has moderately high ability to resolve different closely related bacterial strains (Versalovic, Schneider et al. 1994). Consumable costs for ERIC-PCR are inexpensive and labor costs for sample processing and data analyses are moderate. ERIC-PCR is a genetic fingerprinting method used in previous BST studies (i.e., McLellan, Daniels et al. 2003; Leung, Mackereth et al. 2004; Casarez, Pillai et al. 2006) as well as many microbial ecology and epidemiological studies. ERIC elements are repeat DNA sequences found in varying numbers and locations in the genomes of different bacteria such as *E. coli*. The PCR is used to amplify the DNA regions between adjacent ERIC elements. This generates a DNA banding pattern or fingerprint which looks similar to a barcode pattern. Different strains of *E. coli* bacteria have different numbers and locations of ERIC elements in their bacterial genomes, and therefore, have different ERIC-PCR fingerprints. ERIC-PCR is useful as a screening technique for library development because of its moderate cost and moderately high ability to resolve different strains of the same species of bacteria. Though rep-PCR banding patterns for isolates tend to be generally stable, differences in fingerprint image processing and PCR protocols between laboratories may result in reduced between-laboratory reproducibility and pose a challenge to generating a composite library in multiple laboratories. Rigorous quality control and quality assurance, standardized protocols for PCR and image processing, and adequate training of personnel is crucial for generation of comparable data. Two analytical strategies that enhance data comparability between laboratories are the use of horizontal fluorophore-enhanced rep-PCR (HFERP; Johnson et al., 2004), or a commercially packaged product such as the DiversiLab system (http://www.bacbarcodes.com/).

Ribotyping

Ribotyping is a genetic fingerprinting method used in previous BST studies (i.e., Parveen, Portier et al. 1999; Scott, Parveen et al. 2003; Moore, Harwood et al. 2005), as well as many microbial ecology and epidemiological studies (i.e., Verduin, Kools-Sijmons et al. 2000; Clark, Kruk et al. 2003), although there is not a consensus as to the best protocol. Ribotyping has a moderate ability to resolve different strains of the same bacteria species. An automated ribotyping system (DuPont Qualicon RiboPrinter) is available, which saves labor costs and requires little training, but the initial investment and the consumable cost per isolate are expensive. The RiboPrinter was originally developed for use in identification and BST of microbial isolates for the food industry. An endonuclease enzyme (i.e., *Hin*dIII) selectively cuts *E. coli* DNA wherever it recognizes a specific DNA sequence. The resulting DNA fragments are separated by size and probed for fragments containing particular conserved ribosomal RNA gene sequences, which results in DNA banding patterns or fingerprints that look similar to barcode patterns. Different strains of *E. coli* bacteria have differences in their DNA sequences and different numbers and locations of enzyme cutting sites, and therefore have different ribotyping fingerprints. By automating the process, the RiboPrinter System can analyze up to 32 samples per day, whereas manual ribotyping methods, which require highly trained and experienced personnel, may require up to several days to complete. All bacteria isolate sample processing is automated using standardized reagents and a robotic workstation, providing a high level of reproducibility. Since the system employs standardized methods and reagents, results obtained from other laboratories using the system are directly comparable.

Pulsed-Field Gel Electrophoresis (PFGE)

PFGE is another leading genetic fingerprinting method used in BST. PFGE has very high resolution and can discriminate between closely related bacteria strains. The entire bacterial genome is fragmented using an infrequent cutting restriction endonuclease enzyme (i.e., *Xba* I) which cuts DNA wherever it recognizes a specific rare sequence. All the DNA fragments are separated by size and stained to visualize the resulting genetic fingerprint that resembles a barcode. Different strains of *E. coli* bacteria have differences

in their DNA sequences and different numbers and locations of enzyme cutting sites and therefore, have different PFGE fingerprints. PFGE is currently being used by the Centers for Disease Control and Prevention (CDC) to track foodborne *E. coli* O157:H7 and *Salmonella* isolates. CDC currently uses this standardized protocol as the basis of its "PulseNet" outbreak surveillance network that allows public health laboratories nationwide to quickly compare their PFGE fingerprints to the CDC central reference library. Although it requires more training and cost, PFGE has very high resolution and can discriminate between closely related strains. While this level of resolution allows higher confidence in the matches made, fewer identifications of water isolates can be made, and an unrealistically large (and costly) library would be needed for field application. In addition, some bacterial strains have genomic DNA in configurations that do not permit effective restriction endonuclease digestions and cannot be analyzed by PFGE.

Kirby-Bauer Antibiotic Resistance Analysis (KB-ARA)

Antibiotic Resistance Analysis was one of the first approaches developed for BST and has been used for many TMDL studies in the United States. It relies on the principle that bacteria from intestinal tract of different animals have been exposed to different antibiotics and will exhibit different profiles of resistance to a panel of antibiotics. Several methods have been used for antibiotic resistance analysis; however all rely on the same principle and the data are usually statistically analyzed using discriminant analysis to categorize isolates. The KB-ARA technique follows procedures used in the clinical laboratory for evaluating the antibiotic resistance of bacteria isolates and has the advantage of strong quality assurance/quality control requirements. The method involves measuring the diameter of the zone of inhibition of bacterial growth around a filter disk impregnated with a specific antibiotic. By comparison to resistant and susceptible control strains, the response of the *E. coli* isolates can be determined. In the procedure used in several studies in Texas (Mott) to further standardize and automate the assay, an image analysis system is used to measure the zones of inhibition and provide electronic archival of data. The KB-ARA profile for an isolate consists of the measurements of the zones of inhibition in response to 20 antibiotics, each at a standard single concentration. Among

ERIC-PCR, RiboPrinting, PFGE and KB-ARA, the KB-ARA method has the lowest ability to discriminate closely related bacteria strains. However, it also has the lowest initial and per sample cost and takes the least time and training.

Carbon Source Utilization (CSU)

CSU is another phenotypic method and is less widely used than ARA. It is based on the principle that bacteria from different animals have been exposed to a range of carbon sources (dietary differences between animals). It is most commonly conducted using the Biolog Microbial Identification System (MIS) (Biolog, Inc., 3939 Trust Way, Hayward, CA 94545). Bacteria are grown on specific media and then transferred to 96 well microplates, with each well containing a different carbon source and an indicator that changes color if the source is utilized. The plates are incubated and read using an automated plate reader, which can provide color intensity data for each well. The profiles of utilization of the carbon sources of unknown source isolates are compared to the profiles of known source isolates (the library) and discriminant analysis is used to categorize isolates. This approach has been used both in Texas, in conjunction with KB-ARA, and in other states as a stand-alone method. In Texas, CSU profiles of *E. coli* were used in conjunction with ARA of the same isolates for a study of sources of contamination in White Oak and Buffalo Bayous (Mott and Lehman, with Dr. Rifai, University of Houston). Combining both profiles provided increased accuracy in terms of rates of correct classification for the library.

Table 2 compares several BST techniques.

Table 2. Relative comparison of several bacterial source tracking techniques

Technique	Acronym	Target organism(s)	Basis of characterization	Previously Used or in Progress in Texas	Used in other states	Accuracy of source identification	Size of library needed for water isolate IDs	Capital cost	Cost per sample (reagents and consumables only)	Ease of use	Hands on processing time for 32*** isolates	Time required to complete processing 32 isolates
Enterobacterial repetitive intergenic consensus sequence polymerase chain reaction	ERIC-PCR	*Escherichia coli* (*E. coli*) and *Enterococcus* spp.	DNA fingerprint	Yes (Di Giovanni)	Yes	Moderate	Moderate	$20,000 ($15,000 BioNumerics software, $5,000 equipment)	$8	Moderate	3 h	24 h**
Automated ribotyping (RiboPrinting)†	RP	*E. coli* and *Enterococcus* spp.	DNA fingerprint	Yes (Di Giovanni)	Yes	Moderate	Moderate	$115,000 ($100K RiboPrinter, $15K BioNumerics software)	$40	Easy	1 h	24 h
Pulsed field gel electrophoresis	PFGE	*E. coli* and *Enterococcus* spp.	DNA fingerprint	Yes (Pillai and Lehman)	Yes	High	Large	$30,000	$40	Difficult	10 h	5 days
Kirby-Bauer antibiotic resistance analysis‡	KB-ARA	*E. coli* and *Enterococcus* spp.	Phenotypic fingerprint	Yes (Mott)	Yes	Moderate*	Moderate	$35,000	$15	Easy	3 h	24 h**
Carbon source utilization	CSU	*E. coli* and *Enterococcus* spp.	Phenotypic fingerprint	Yes (Mott)	Yes	Moderate	Moderate	$15,000	$10	Easy	4 h	24 h**
Bacteriodales polymerase chain reaction	*Bacterio-dales* PCR	*Bacteriodales* species	Genetic marker presence or absence (not quantitative)	Yes (Di Giovanni)	Yes	Moderate to high for only human, ruminant, horse, and pig sources	Not applicable	$5,000	$8	Easy to moderate	3 h	8 h**
Enterococcus faecium surface protein polymerase chain reaction or colony hyb.	*E. faecium* esp marker	*E. faecium*	Genetic marker presence or absence (not quantitative)	Yes (Di Giovanni)	Yes	High for only human	Not applicable	$8,000	$8 to $12	Easy to moderate	3 to 6 h	8 to 24 h**
ERIC and RP 2-method composite	ERIC-RP	*E. coli*	DNA fingerprints	Yes (Di Giovanni)	No	Moderate to high	Moderate	$120,000	$48	Moderate	4 h	24 h
ERIC and KB-ARA 2-method composite	ERIC-ARA	*E. coli*	DNA and phenotypic fingerprints	Yes (Di Giovanni)	No	Moderate to high	Moderate	$55,000	$23	Moderate	6 h	24 h
KB-ARA and CSU 2-method composite	ARA-CSU	*E. coli* and *Enterococcus* spp.	Phenotypic fingerprints	Yes (Mott)	Yes	Moderate to high	Moderate	$50,000	$23	Easy to moderate	7 h	24 h

†A manual ribotyping version is also used by some investigators (i.e. Dr. M. Samadpour with IEH Laboratories and Consulting Group in Seattle), but no detailed information is available for comparison.
‡A variation of this technique using replica plating and +/- scoring of growth on media with different concentrations of antibiotics, called ARA, has been used extensively in Virginia for TMDLs.
*This technique is better for distinguishing broader groups of pollution sources. For example, "wildlife" and "livestock" as opposed to "avian wildlife", "non-avian wildlife," "cattle," etc.
**With sufficient personnel, up to approximately 150 isolates can be analyzed in 24 h.
***Thirty two isolates selected for comparison because it is the maximum throughput per day of the RiboPrinter, which is the only automated system described.

Method Comparisons and Composite Data Sets in a Previous Texas Study

As mentioned above, a BST project was completed for Lake Waco and Belton Lake in which *E. coli* isolates were analyzed using RP, ERIC-PCR, PFGE and KB-ARA. For this study, BST analyses were performed using the individual techniques, as well as composite data sets (Casarez, Pillai et al. 2006). The congruence (concordance) between the groupings of isolates from known fecal sources by individual BST methods and different combinations of composite data sets was determined using the same statistical software (BioNumerics, Applied Maths, Austin, Texas) and Pearson's product moment correlation coefficient. The four-method composite library generated the most desirable BST results in regards to accuracy, blind quality control study results, library quality measures and ability to identify water isolates. However, as few as two methods in combination may be useful based on congruence measurements, library internal accuracy (i.e., rates of correct classification, RCCs), and comparison of water isolate identifications. In particular, the combinations of ERIC-PCR and RiboPrinting (ERIC-RP), or ERIC-PCR and Kirby-Bauer Antibiotic Resistance Analysis (ERIC-ARA) appeared promising. These two-method composite data sets were found to have 90.7% and 87.2% congruence, respectively, to the four-method composite data set. More importantly, based on the identification of water isolates, they identified the same leading sources of fecal pollution as the four-method composite library. ERIC-ARA has the lowest cost for consumables and has high sample throughput, but requires a considerable amount of hands-on sample and data processing. Due to the high cost of RiboPrinting consumables and instrumentation, ERIC-RP has a higher cost than ERIC-ARA. However, ERIC-RP has the advantage of automated sample processing and data preprocessing that the RiboPrinter system provides.

Regulatory Expectations and Capabilities of BST Methods

Regulatory agencies continue to have high hopes and expectations for BST in aiding them to address water quality issues. BST may not be needed, or may not be suitable for all TMDL studies. In some cases, more rigorous monitoring of water bodies and

determination of *E. coli* levels may identify the major sources of pollution. No BST method developed to date can provide 100% accuracy of pollution source identification in field studies. In many cases, the reduced accuracy often does not justify the high cost of some of the analyses.

In addition, the issue of indicator bacteria regrowth in the environment has been raised. There are many uncertainties surrounding this issue and there is a lack of scientific evidence that demonstrates regrowth occurs naturally in subtropical and temperate environments. However, because of the obvious implications for regulatory efforts (particularly for *E. coli*), the survival and regrowth of fecal indicator organisms in the aquatic environment, sediments and soils will need to be addressed not only for source tracking, but the concept of bacterial fecal indicators. While no BST method has specifically been developed to identify environmentally adapted indicator organisms, *ex post facto* analysis of BST data for environmental water isolates may provide some insight into this issue.

Ideally, agencies and stakeholders would prefer identification of pollution sources to the level of individual animal species. Performing a three-way split of pollution sources into domestic sewage, livestock and non-domestic animals source classes would likely be more scientifically justified with the current levels of accuracy and confidence of BST techniques in general. The division of host sources into the seven classes in the Lake Waco study was a compromise between the capabilities of the *E. coli* BST techniques and their practical application. Rates of correct classification (RCCs, measurements of library internal accuracy) are calculated using jackknife statistical analyses in which a single known source isolate is removed from the identification library, treated as an unknown, and is tested against the remaining library isolates to determine the percent of correct source identification. Comparisons of the RCCs for two- or three-way splits of host sources show high accuracy in distinguishing human from animal sources (83% and 95% RCCs, respectively); and distinguishing domestic sewage, livestock and non-domestic animals (83%, 72% and 73% RCCs, respectively). The RCCs for the seven-way split of source classes ranged from 22% to 83%, although those classes of particular

interest (domestic sewage and cattle) were relatively high (83% and 60%, respectively). It is important to note that these RCCs for each of the source classes were three- to seven-fold greater than random chance based on library composition. Identifying pollution loading down to the sampling station level would require collecting hundreds of water samples per sampling site to analyze a statistically significant number of *E. coli*, making this approach not feasible. More realistic for BST is identification of fecal pollution sources at the watershed level. Alternatively, a higher number of *E. coli* isolates (i.e., 50) can be analyzed from fewer water samples to identify statistically significant differences in pollution sources. However, this will only provide pollution source identification on limited time scales, and will not provide an overall assessment of the water body.

Although quantitative allocation of fecal contamination to source categories is a goal of most TMDL projects, uncertainty in classification limits our capacity for absolute quantification. In some cases, library-dependent methods may enable identification of a source that contributes more fecal contamination than other sources, or identification of sources for which there is no credible evidence of substantial contamination. The results of library-dependent classification are conservatively seen as semi-quantitative and suitable for sample-level classification of sources as "contribution not detected" or "contribution detected" with possible refinement to "contribution detected greater than (alternative source)." This information may not be suitable for incorporation into quantitative water-quality models. However, most library-independent methods are even less quantitative than library-dependent methods, and the correlation between the fecal pollution indicators targeted by these methods and regulated contaminants (i.e., *E. coli*) is uncertain.

Another issue is that BST typically only identifies the major sources of pollution for a water body, not the entry pathways of the fecal pollution. This can be an important issue when developing implementation actions for different watershed stakeholders. This emphasizes the importance of sampling station selection and upstream/downstream targeted sampling of suspected pollution sources.

Future Direction

As described above for the EPA CWA §319(h) funded study for Lake Waco and Belton Lake, a cost-effective alternative to using multiple BST methods for each study is to use the two-method combination of either ERIC-PCR and RiboPrinting (ERIC-RP), or ERIC-PCR and Kirby-Bauer antibiotic resistance analysis (ERIC-ARA). Based on other studies in Texas, CSU and KB-ARA may also be a suitable combination. It is important to note that in nearly all cases, no single BST method should be solely relied upon. This is particularly true with the phenotypic methods, such as KB-ARA, which alone cannot provide the resolution needed for most BST studies. A factor to be considered when choosing the methods includes the level of resolution needed for the particular study. In some cases, it may be critical to determine specific animal groups, while in other cases broad categories may provide the level of resolution needed for management decisions. Cost constraints, equipment needs and expertise available are other factors that should be considered in the decision.

Since the Lake Waco study was initiated, there have been significant developments in library-independent BST methods, including bacterial genetic markers specific to different animal sources and humans (i.e., Bernhard and Field 2000; Dick, Bernhard et al. 2005; Scott, Jenkins et al. 2005; Hamilton, Yan et al. 2006). Library-independent methods are cost-effective, rapid and potentially more specific and accurate than library-dependent methods. Concerns with many of the recently developed library-independent approaches include uncertainties regarding geographical stability of markers and the difficulty of interpreting results in relation to regulatory water quality standards and microbial risk, since some target microorganisms are not regulated. More importantly, these library-independent methods can only detect a limited range of pollution sources and are currently not quantitative. For example, the *Bacteroidales* PCR (Bernhard and Field 2000; Dick, Bernhard et al. 2005) can detect fecal pollution from ruminants, humans, dogs, horses and pigs; but no further discrimination is possible. This method involves the concentration of 100 ml water samples, extraction of bacterial DNA, and detection of animal group-specific *Bacteroidales* bacteria genetic markers using the PCR.

This method is qualitative (presence/absence), and does not identify sources of *E. coli*, but rather the potential sources of fecal pollution that may contribute *E. coli*. Despite these limitations, this method may be useful for a rapid assessment of the possible sources of fecal pollution for a water body and are currently being applied to BST studies in Texas. Another example is the detection of optical brighteners used in laundry detergent using fluorometry. This approach can be used in real-time for the detection of leaking septic systems and sewage releases, is ideal for lakes and beach waters but obviously cannot identify other sources of fecal pollution.

Identification libraries consisting of thousands of isolates from different geographical regions in Texas have already been established for ERIC-PCR, PFGE, RiboPrinting, CSU and KB-ARA patterns. In addition, several thousand more *E. coli* isolates from source samples have been archived and are available to researchers. Library development is one of the most costly components of BST studies. It would be most economical to build upon the libraries already established in Texas. It is recommended that agencies use contractors that use BST methods that will strengthen and expand the current Texas library and follow previously approved standard operating procedures (SOPs) for performing analyses. However, further research is needed to determine if a regional library built from different projects using the same protocols may be useful for the identification of water isolates from other watersheds. That is, geographic and temporal stability of the existing *E. coli* library will need to be explored. Currently, the Di Giovanni laboratory at the El Paso Agricultural Research and Extension Center is cross-validating the libraries generated in the Lake Waco study and the San Antonio watershed study in an attempt to explore issues of geographical and temporal stability of BST libraries, refine library isolate selection and determine accuracy of water isolate identification. By selecting *E. coli* source isolates that are correctly identified in both watershed libraries, we hope to find more geographically universal and host-specific isolates, resulting in more accurate source tracking.

The infrastructure for BST work in Texas needs to be expanded for both library dependent as well as library independent methods. More laboratories with trained

personnel and equipment are needed. This could be facilitated by the use of automated systems for BST analyses. In addition to the RiboPrinter, an automated system for repetitive element PCR (repPCR) using BOX primers (Diversilab, Athens, Georgia) has been developed and may be an alternative to ERIC-PCR. For just the *E. coli* isolates selected through cross-validation of existing libraries, funding is needed to develop a complete data set using BST techniques selected for use in Texas.

For future TMDL studies, an assessment phase using a "toolbox" approach is recommended. The assessment phase should include targeted monitoring of suspected pollution sources, use of library-independent methods to identify the presence of domestic sewage pollution and screening of water isolates from the new watershed against the existing library to determine the need for collection of local source samples and expansion of the library.

Recommended Approach for Bacteria TMDL and Implementation Plan Development

With the background provided by the preceding sections on BST and modeling, as well as extensive input from the expert advisory group, the task force recommends that TCEQ and TSSWCB implement the following integrated approach to the TMDL and I-Plan development.

The task force recognizes that no single process is appropriate for all bacteria impairments, given the variations in: (1) the severity of the bacteria impairments, (2) the complexity of watershed land uses and development, and (3) the interests of stakeholders and the regulated community affected. For these reasons, we propose a tiered approach that identifies alternate procedures to accommodate the site-specific factors that will emerge in each assessment. The three-tiered approach to developing bacteria TMDLs and I-Plans described in this section incorporates adaptive management, phased TMDLs and phased implementation to the extent allowable by EPA. The objectives of Tiers 1 and 2 are to ensure that each TMDL is developed using a scientifically credible, cost-effective process with strong stakeholder involvement. Tier 3 is designed to develop a feasible I-Plan, and, for some complex TMDLs, expands the information available for TMDL development. Figure 5 summarizes the three-tier approach.

Tier 1 should be required of all bacteria TMDLs to provide the minimum level of technical analysis and stakeholder input needed to make sound decisions regarding load reductions and allocations. Tier 2 provides a more in-depth and extensive assessment of the water body in question. We anticipate that it will provide sufficient scientific content and stakeholder involvement to develop most bacteria TMDLs. Tier 3 should provide additional information regarding bacteria sources and impacts of best management practices. We anticipate that for most contaminated water bodies Tier 3 activities will be limited to I-Plan development, though in some cases Tier 3 may be required for TMDL development.

TMDLs requiring Tier 3 could include: (1) those requiring large reductions in bacteria loads (e.g. >75 %), (2) those with highly uncertain and conflicting results, (3) those with large implementation costs or (4) those that have not obtained stakeholder acceptance. The decision to move to Tier 3 is strictly at the discretion of the TCEQ and TSSWCB, but with adequate scientific and stakeholder input.

The tiered framework presented here is intended to be flexible to ensure it best fits the complexity of the watershed sources, availability of data, degree of impairment and level of accuracy required.

TIER 1

Tier 1 is a one-year analysis required for all TMDLs. It provides for early stakeholder involvement, development of a comprehensive geographic information system of watershed information, surveys of potential bacteria sources, development of a load duration curve and analysis of the assembled information by agency staff, agency consultants and stakeholders.

1.1 - Stakeholder Involvement

TCEQ, with the TSSWCB and other key agencies, have been developing TMDLs in consultation with watershed-based stakeholder groups. The process includes extensive efforts to identify and involve diverse and sometimes competing interests. The task force believes it is essential to preserve and enhance these processes. Leaders within the watershed should be enlisted to participate at the outset of the project (after listing on the 303(d) list) and collaborate on all aspects of the TMDL development. County commissioners courts; soil and water conservation districts; community leaders; non-governmental environmental, agricultural and civic organizations; TPDES permit holders; cities and water districts (especially with MS4 permits); watershed landowners; and other potential contributors should form the basis for the stakeholder group. In most cases, the TCEQ should initiate a TMDL project with a planning meeting of state and local entities (such as TSSWCB, Texas Department of Agriculture [TDA], Texas Parks and Wildlife Department [TPWD], Texas Cooperative Extension, key municipalities, county commissioners and soil and water conservation districts) and the appropriate river

authority to develop an initial outreach strategy whereby agency representatives will solicit the commitment and involvement at the local level of key leaders.

Figure 5: Recommended Three-Tier Approach for Bacteria TMDL Development

<u>Tier 1 Analysis (T1)</u> (one-year)
Required for all bacteria TMDLs.

1. Form TMDL stakeholder advisory group.
2. Develop comprehensive GIS inventory for watershed.
3. Implement source survey for watershed.
4. Calculate load duration curves (LDCs).
5. Analyze Tier 1 data with stakeholder advisory group.

Decision 1 (D1) *Are data and analysis adequate?*
Yes → Go to D2.
No → Go to T2.

Decision 2 (D2) *Are needed load reductions socially and economically attainable?*
Yes → Complete and submit draft TMDL for agency approval.
No → Complete and submit a draft TMDL that includes a recommended change in designated use (i.e. Use Attainability Analysis).

<u>Tier 2 Analysis (T2)</u> (one-to-two years)_
Implemented for most bacteria TMDLs. May be adequate for I-Plan development for non-controversial TMDLs.

1. Implement targeted monitoring to fill data gaps.
2. Perform library-independent BST and limited library-dependent BST analysis.
3. Develop simple LDC, GIS and/or Mass Balance Models.
4. Analyze Tier 2 data with stakeholder advisory group.

Decision 3 (D3) *Are data and analysis adequate?*
Yes → Go to D4.
No → Initiate a "phased TMDL" and go to T3.

Decision 4 (D4) *Are needed load reductions socially and economically attainable?*
Yes → Complete and submit draft TMDL (or I-Plan) for agency approval.
No → Complete and submit a draft TMDL that includes a recommended change in designated use (i.e. Use Attainability Analysis).

<u>Tier 3 Analysis (T3)</u> (two-to-three years)
Normally used for I-Plan development. May be required for development of complex "phased TMDLs."

1. Assure extensive stakeholder involvement.
2. Implement extensive targeted monitoring.
3. Perform extensive library-dependent BST analysis.
4. Complete mechanistic modeling.
5. Analyze Tier 3 data with stakeholder advisory group.

Decision 5 (D5) *Are needed load reductions socially and economically attainable?*
Yes → Complete and submit draft I-Plan (or revise "phased TMDL") for agency approval.
No → Complete and submit a draft TMDL that includes a recommended change in designated use (i.e. Use Attainability Analysis).

The identified stakeholder participants and agencies should be invited by TCEQ (or, when appropriate, TSSWCB and other participating organizations) for a kick-off meeting. The meeting will discuss the project goals, the existing water quality, relevant water quality standard(s) not met, the time line and process for TMDL development and what implementation strategies are typical for addressing bacteria impairments. Stakeholder participants should understand the complexity of the watershed, the uncertainty associated with bacteria monitoring and analysis and what may be required of the regulated community during implementation, as well as the typical voluntary or incentive-based strategies needed for implementation. Either at a kick-off meeting or soon after, the TCEQ (and TSSWCB) should discuss with stakeholders and seek consensus on the overall project's design, including the tools, models and monitoring that would be used in Tier 1 of the TMDL assessment. Attentive efforts to receive broad input, involvement and consensus should pay off in the end, particularly with groups representing entities that are not regulated point sources, to improve the water quality. Stakeholders can provide insight into sources and solutions that may not be apparent to those living outside the watershed. Since stakeholders will ultimately be responsible for helping achieve TMDL goals, it is appropriate that they be involved throughout TMDL development.

1.2 - Comprehensive Geographic Information System

A comprehensive geographic information system should be developed for the watershed, including available land use, elevation, soil, stream network, reservoir, road, municipality and satellite or aerial photographic information. In addition, locations of other relevant information (such as wetlands, sewage treatment plants, subdivisions, confined animal feeding operations, etc.) should be included for use by scientists, agencies and other stakeholders. In most cases, almost all data layers required for Tier 1 analysis can be obtained from readily available public sources. Locations of potential bacteria sources (identified in 1.3 below) should be incorporated into the GIS.

1.3 - Surveys of Potential Bacteria Sources

Conducting a potential source survey is an important Tier 1 activity. Texas and other states have conducted these surveys to better characterize the pollutant sources that contribute loadings that cause impairments. As with many efforts of this nature, there is a substantial range of costs and investigative activities that could occur, dependent on the project design, complexity of land uses and geographic size of the target watershed. For instance, in the Buffalo and White Oak Bayous in the Houston area, the TCEQ and its contractors spent several years and substantial funds to assure the watershed (with its intensive land use, dense population and hundreds of point sources) was adequately characterized.

The potential source survey should be representative of warm and cool seasons and, if possible, low and high flow conditions. The survey should be conducted by an integrated team including stakeholders, TPWD biologists, TDA/TCE personnel, Texas Department of State Health Services, TCEQ Regional Office staff, local governments and SWCD and TSSWCB personnel in order to identify all potential contamination sources. Point sources like wastewater treatment plants and industrial outfalls, nonpoint sources, stream sediment sources (especially those below wastewater outfalls) and other possible bacteria sources should be evaluated. According to a study by Harris County *E. coli* in fully disinfected WWTP effluent can regrow a thousand-fold over the course of four days. It is possible that in many effluent-dominated streams, regrowth is a major source of *E. coli.* To complete this evaluation, it will be important to examine permit compliance issues, residences and businesses on central sewage collection systems, septic systems and livestock and non-domestic animal concentrations.

1.4 - Data Analysis / Load Duration Curves

Note: For bacteria impairments in reservoirs or estuaries, the load duration curve method is not usually applicable.

The fourth component of Tier 1 includes analysis of ambient bacteria concentrations using statistical methods and LDCs. LDC development at this point in the process will use existing data and provide stakeholders insight into when exceedences occur, potential reductions needed, and potential sources. This will help stakeholders develop monitoring methodologies needed for the next tier. LDCs should be developed (using the methods described earlier in the document) for all sites having at least 24 bacteria data points (including at least three high flow points). For those sites with large data sets, only the most recent five years of data should be used. As additional data become available following targeted monitoring in Tier 2, this initial LDC should be updated. The analysis of the data should include an assessment of the degree of impairment, expressed as a percentage exceedance of the standard. This will help identify the course of action needed to address the impairment. In addition, this will provide some indication of the need for a UAA, particularly in areas where large exceedances of the criteria are unlikely to be resolved by revised criteria.

1.5 - Analysis of Tier 1 Information

Components 1.1 through 1.4 should be implemented simultaneously during the first year of TMDL development. As information from components 1.1 through 1.4 is collected, it should be shared with interested stakeholders, scientists and agency personnel. In addition to organizational meetings at the beginning of Year 1, at least one meeting with stakeholders should be held in the watershed approximately halfway through Year 1 to share information collected to that point. As the end of Year 1 approaches, accumulated data and recommendations of agency staff and consultants should be collected and shared with stakeholders. A meeting should be held in the watershed to discuss the data and recommendations of experts, and consensus should be sought regarding future activities.

At the meeting, consensus should be sought concerning whether data and analysis are adequate to develop a draft TMDL, including target reductions in bacteria loads and waste loads. If data and analysis from Tier 1 are not adequate, Tier 2, a one- to two-year period of additional data collection and analysis, should begin. If Tier 1 data and analysis

are sufficient, the group must consider the question of whether the required reductions in bacteria waste loads and loads are socially and economically attainable. If they are attainable, a draft TMDL can be completed and submitted to the agencies for approval. However, in some cases, reaching the required waste load and load reductions may not be socially and economically attainable. In those cases, the group may reach consensus that the agencies should consider changing the designated use of the water body in order to meet alternative standards (see Figure 5).

A provisional TMDL should be drafted when a UAA is under way. A final TMDL may be required to comply with the new standard. In this case, the provisional TMDL should be adopted by the TCEQ and TSSWCB to meet the water quality standard in effect at that time and remain in place while standards issues are addressed. However, I-Plan development and implementation for these TMDLs should not proceed until the standards evaluation is completed. Once the standards are resolved, the provisional TMDL would be replaced by the final TMDL and an I-Plan developed.

TIER 2

If Tier 1 does not produce a consensus that data and analysis are adequate, the process should move to Tier 2, a one- to two-year effort to develop sufficient information to implement the needed TMDL or change the designated use of the water body. We recommend that Tier 2 include targeted monitoring to complement data collected in Tier 1, library-independent bacterial source tracking, statistical and mass balance modeling and development of a consensus draft TMDL.

2.1 - Targeted Monitoring

Agency personnel, expert consultants and stakeholders should develop a targeted monitoring plan to fill data gaps (for example, provide additional high flow samples or sample additional locations in the water body), evaluate the magnitude and impacts of possible waste loads or loads identified in Tier 1, and/or evaluate bacteria fate and

transport within the water body. For example, point sources could be sampled at their outfalls. Additional sampling points more representative of the watershed or stream segment than public access points could be sought. Measurements of waste loads and loads could be made and substituted for literature values or estimates used in Tier 1. More intensive (biweekly or monthly) monitoring could be implemented for stream segments at greatest risk for fecal pollution.

2.2 - Library-Independent BST

To further assess bacteria sources, BST analysis should be conducted in conjunction with the targeted monitoring to determine if livestock, humans and/or non-domestic animals are contributing bacteria to the water body. Fifty to 100 samples should be analyzed using the library-independent PCR genetic test for the *Bacteroidales* markers for human, ruminants, horse and swine sources.

If sufficient funds are available, *E. coli* isolates from 50 to 100 different water samples should be analyzed using the BST methods described in the Tier 3 BST discussion and compared with isolates from the previously developed Texas *E. coli* source library to determine the need for development of a local source library (see component 3.2 below). These analyses are needed to confirm that the sources of *E. coli* and *Bacteroidales* are comparable. If these *E. coli* analyses are not done in Tier 2, they should be included in subsequent Tier 3 activities.

2.3 - Spatially Explicit Statistical and Mass Balance Modeling

Spatially explicit statistical models and/or mass balance models should be used in combination with the LDC developed in Tier 1 to provide multiple lines of evidence for bacteria sources. This report does not recommend specific spatially explicit statistical models or mass balance models. We consider that the quality of model calibration, input data quality and the experience of the modeling team are as important as the choice of model used. Therefore, we recommend that agency staff and their expert consultants

evaluate the characteristics of the water body, its watershed and the data available for analysis, and then select the most appropriate model and modeling team.

The main value of LDC and mass balance models such as BLEST and BIT is in determining load reductions required to meet the water quality standard. Spatially explicit models such as Arc Hydro, SELECT or SPARROW should be used to estimate contaminant loads from various sources in a watershed.

Models should be run using the highest quality, readily available input data for the watershed. Every effort should be made to use measured loadings from wastewater treatment plants and other point and nonpoint sources. Data collected through the targeted monitoring in Tier 2.3 should be used to update the LDC as well as for providing the data needed to initialize, calibrate and validate the model(s) used. Updated LDCs should be developed (using the methods described earlier in the document) for all sites having at least 24 bacteria data points (including at least three high-flow points). At each step, confidence intervals for model inputs and outputs should be estimated and reported.

The principal reason for recommending that simpler spatially explicit statistical models and mass balance models be used in Tier 2 rather than more complex hydrologic/water quality models is that the simpler models should be more economical to implement. This may not be the case in the future as baseline simulations are developed by USDA with SWAT. If such baseline, partially calibrated models are available, agency personnel may choose to implement them for Tier 2 analyses.

2.4 - TMDL Development

Components 2.1 through 2.3 should be implemented simultaneously. As during Tier 1, at least one meeting with stakeholders should be held in the watershed approximately halfway through Tier 2 activities to share information about progress to date. As the end of Tier 2 approaches, a meeting should be held in the watershed to discuss the data and recommendations of experts, and consensus should be sought regarding future activities.

We anticipate that for most impacted segments, consensus will be reached by the end of Tier 2 to complete development a draft TMDL, including target reductions in bacteria loads and waste loads. If consensus is reached that Tier 2 data and analysis are sufficient, the group must consider whether the required reductions in bacteria waste loads and loads are socially and economically attainable. If the required reductions are attainable, a draft TMDL can be completed and submitted to the agencies for approval. However, if the required waste load and load reductions are not socially and economically attainable the agencies should consider conducting a Use Attainability Analysis (UAA) and changing the designated use of the water body in order to meet alternative standards (see Figure 5).

A provisional TMDL should be drafted when a UAA is under way. A final TMDL may be required to comply with the new standard. In this case, the provisional TMDL should be adopted by the TCEQ and TSSWCB to meet the water quality standard in effect at that time and remain in place while standards issues are addressed. However, I-Plan development and implementation for these TMDLs should not proceed until the standards evaluation is completed. Once the standards are resolved, the provisional TMDL would be replaced by the final TMDL and an I-Plan developed.

If Tier 2 data and analysis are not sufficient to reach consensus, a "phased TMDL" should be completed. TCEQ staff (with TSSWCB, as appropriate) will be responsible for deciding when consensus cannot be achieved.

The phased TMDL should consist of draft "phase 1" TMDL based on data gathered during Tiers 1 and 2. In addition, the draft "phase 1" TMDL should include a commitment to collect and analyze additional data during Tier 3 I-Plan analysis. This approach allows implementation of the TMDL (beginning with BMPs most socially and economically acceptable) while additional data are collected to reduce uncertainty. The TMDL may be reopened to consider new information that suggests that the WLA or LA should be modified. If standards are not met by target dates established by the stakeholders, increasingly stringent measures should be implemented in a phased manner over a period of years.

TIER 3

Tier 3 analyses will normally be required for development of I-Plans and, if necessary, phased TMDLs. Tier 3 should normally consist of a two- to three-year effort consisting of extensive watershed monitoring, library-dependent bacteria source tracking, mechanistic hydrologic modeling and I-Plan or phased TMDL development.

We recommend the use of "adaptive management" for the I-Plan process. During the multi-year course of development and execution of an I-Plan, changes in water quality, infrastructure and agency policies can be expected. As a result, the I-Plan or TMDL should be subject to revision as conditions change. Agency personnel and stakeholders should also remain sensitive to the need for identifying and implementing cost-effective BMPs. Costly BMPs can have adverse economic impacts on stakeholders and taxpayers, and they should be avoided if possible, especially when good scientific evidence of their effectiveness is lacking.

3.1 - Watershed Monitoring

Extensive watershed monitoring should be initiated to identify bacteria sources, quantify loading and provide the data needed for calibration and validation of watershed scale mechanistic models. Watershed monitoring should be targeted to fill information gaps identified in Tiers 1 and 2. At a minimum, monthly sampling for one year should be conducted, although greater frequency (weekly) and duration (two years) is preferred. Monitoring should include measurement of bacteria levels and flows from all wastewater treatment plants and other point sources that are suspected of contributing significant bacteria loads. Sampling sites should be carefully selected, and samples should represent spatial variability, seasonal fluctuations and typical (low, medium and high) flows. A minimum of 100–200 isolates from approximately 40 water samples should be collected in conjunction with the watershed monitoring effort for library-dependent BST analysis. These samples should be collected at carefully selected and representative sites, seasons and flow conditions (see 3.2 below).

3.2 - Library Dependent Bacteria Source Tracking

The isolates collected during the watershed monitoring described in component 2.2 (above) should first be analyzed using ERIC-RP, ERIC-ARA or CSU-ARA combination methods. Isolates should first be identified using the Texas Source Library. If the Texas Source Library does not provide 80% identification then a local library should be developed using *E. coli* isolates from known sewage and animal sources. The sampling design should be determined using data from the component 2.2 assessment.

At least 300 *E. coli* colonies from approximately 100 known fecal samples should be characterized by the selected combination method (i.e. ERIC-RP, ERIC-ARA, CSU-ARA). The *E. coli* isolates obtained from ambient water samples should be characterized using the selected method. DNA patterns of those isolates should be compared to the individual watershed *E. coli* source library as well as Texas library and identified to cattle, other livestock, avian and non-avian non-domestic animals, domestic sewage and pet sources. Identified sources and unknown sources should be expressed as percentages of total isolates with appropriate confidence intervals.

3.3 - Mechanistic Hydrologic / WQ Modeling

For Tier 3 analyses, a mechanistic hydrologic water quality model or combination of models (such as HSPF, SWAT, SWMM and/or WASP) should be used to model watershed hydrology and fate and transport of *E. coli* within the watershed. The main value of these detailed models is that they allow for spatial and temporal analysis of the effectiveness of different best management practices in improving in-stream water quality. The selected model(s) should be run using the highest quality, readily available input data for the watershed.

The model(s) should be calibrated and validated for flows using long-term weather and stream flow gauges within (or near) the targeted water body. Once the model(s) are calibrated and validated for flow, baseline *E. coli* loads and concentrations should be

calibrated and validated using monitoring data previously collected from the impacted segment. Finally, recommended best management practices (BMPs) identified by the stakeholder committee and/or cooperating agencies should be evaluated for their impacts on bacteria loads. At each step, confidence intervals for model inputs and outputs should be estimated and reported.

Every effort should be made to use measured loadings from wastewater treatment plants and other point and nonpoint sources for model calibration. This caution is particularly important if the effectiveness of BMPs is not well understood or monitored. Where effluent limits are specified as BMPs rather than loads that can be verified (as in the case of NPDES-regulated municipal and small construction storm water discharges), monitoring or other scientifically acceptable BMP performance data should be obtained to assess if the expected load reductions attributed to the BMP are achieved. Otherwise, adaptive management will be limited by our understanding of BMP effectiveness.

3.4 - IP-Plan and Phased TMDL Development

We anticipate that in most cases Tier 3 data collection and analysis will be used for IP-Plan development. However, for some complex and/or controversial water bodies, Tier 3 assessments will be required for phased TMDL development.

As pointed out above, we recommend the use of "adaptive management" for the I-Plan process. During the multi-year course of development and execution of an I-Plan, changes in water quality, infrastructure and agency policies can be expected. As a result, the I-Plan should be subject to revision as conditions change. We recommend that TCEQ identify a watershed coordinator to facilitate the communication, education and coordination needed during I-Plan development, revision and execution.

Adaptive management of I-Plans should facilitate progress toward achieving water quality goals while using new data and information to reduce uncertainty and adjust implementation activities. Adaptive management can include immediate implementation

of BMPs, success monitoring, use of improved monitoring data and modeling, new experimental results and revision of the implementation plan.

Phased TMDLs are an example of the adaptive implementation approach because each new phase utilizes new information to reevaluate the original TMDL. However, even for TMDLs where there is little uncertainty regarding the loading capacity of the water body and the necessary load reductions, an adaptive implementation approach can be a useful tool. Implementation of TMDLs can take many years, and uncertainty about the effectiveness of implementation activities usually exists. Follow-up monitoring addresses uncertainty in the efficacy of implementation actions and can provide assurance that implementation measures are succeeding in attaining water quality standards, as well as in inform the ongoing TMDL implementation strategy. If adaptive implementation activities reveal that TMDL loading capacity needs to be changed, the revision may require EPA approval. In most cases, however, adaptive implementation is not anticipated to lead to the re-opening of a TMDL. Instead, it is a tool used to improve implementation strategies.

The I-Plan describes the actions that the TCEQ, TSSWCB and the project's stakeholders will undertake to achieve restoration of the water body's use. Implementation strategies specify actions to meet the load allocations assigned to all point sources and nonpoint sources identified in the TMDL report. Action strategies may be selected from a menu of possible measures based on an evaluation of feasibility, costs, support, timing and other factors. Activities may be implemented in phases with evaluation of progress and success before proceeding to a subsequent phase.

In addition to the actions taken to reduce pollutant loads from sources, an I-Plan includes provisions to track the progress of the plan using both implementation and water quality indicators. The I-Plan identifies in-stream monitoring at specific locations and targets constituents that will be used to evaluate whether the water quality criterion is achieved. It includes tracking using a schedule identified in the I-Plan for evaluating whether administrative actions undertaken to improve water quality actually occurred.

An I-Plan includes a review strategy that will consider when and if the I-Plan needs to be revised, or as mentioned earlier, whether a subsequent phase of implementation is needed. The review strategy implements adaptive management into the planning process by providing decision points in the process for such consideration. The review strategy accounts for unexpected changes in implementation of controls. In some special circumstances, the I-Plan may include strategies to evaluate the underlying water quality standard and its appropriateness as a goal driving implementation. Recommended changes to a standard could drive the need to re-evaluate and modify the TMDL itself.

Communication is necessary to ensure stakeholders understand the I-Plan and the progress that is being made to restore water quality conditions. The TCEQ disseminates information derived from tracking I-Plan activities to watershed stakeholders and others.

Agency personnel and stakeholders should also remain sensitive to the need for identifying and implementing cost-effective BMPs the effectiveness of which can be verified. Costly BMPs can have adverse economic impacts on stakeholders and taxpayers, and should be avoided if possible, especially when good scientific evidence of their effectiveness is lacking.

Research and Development Needs

This section, coordinated by Drs. Larry Hauck and George Ward, summarizes research and development needed in the next three to five years to improve the tools and methods available to TCEQ and TSSWCB for bacteria TMDL and I-Plan development. These research and development needs may be generally considered to be motivated by the need to quantify and reduce uncertainties in the measurement, identification and modeling of bacterial contaminants.

In previous sections, the diversity of bacteria sources within watersheds and the complexity of their proliferation and movement on the landscape and in receiving waters have been discussed. There are unavoidable inaccuracies in the depiction of the origin, fate and transport of environmental indicator bacteria within fate and transport models, and these represent common sources of uncertainties associated with the load allocation process of a TMDL and its subsequent I-Plan development. Broadly, uncertainties may be reduced by efforts along two parallel paths: (1) refinement of kinetic- and transport-type input parameters used in the preferred models to be applied in Texas and (2) reformulation of kinetic processes in these same models to better represent the present state of understanding of bacteria fate and transport. Laboratory and field studies in combination with model enhancements will be needed in these efforts to reduce uncertainties. BST methods can strongly support the TMDL process, and additional research and development in this arena is also anticipated to be important in reducing uncertainties and providing for improved tools and methods.

Finally, it is recommended that TCEQ and TSSWCB form a work group, in addition to the Task Force, whose focus would be to create a blueprint for a successful stakeholder process. Such a blueprint could be used by staff of both agencies statewide in TMDL, I-plan and watershed protection plan processes. This work group would address stakeholder group membership, attendance at meetings and communication to stakeholders. In addition, it would evaluate the inclusion of state and local agencies on

stakeholder groups in order to ensure TMDL and other regulatory issues are addressed in a manner appropriate to the relevant authority.

The specific research and development needs reported herein were accumulated based on the collective expertise of Task Force members and Expert Advisors, recommendations from the Texas A&M Bacteria Roundtable Discussion of August 10, 2006, studies reported in Rifai et al. (2005), and an overview report on bacteria fate and transport modeling (Benham et al., 2006).

Characterization of Sources

Often a high level of uncertainty exists in the specification of bacteria sources within fate and transport models. Because of the numerous sources of fecal bacteria and the diversity of delivery mechanisms of the bacteria to receiving waters, a broad spectrum of research items will provide useful information.

- Studies to quantify species-specific bacteria production in feces and to measure the variability of this production. There are numerous mammalian and avian species (human, pet, livestock and non-domestic animals), all of which shed fecal bacteria. Within these proposed studies, focus is suggested on dominant and relevant species. While bacteria content of feces has been reported in literature for some species and has been summarized in some reports used in TMDL development, often this information has not been the focus of the reported research and therefore has not undergone thorough peer review. Further, some indications exist that fecal bacteria content may be related to latitude, elevation, diet and water source, which increases the anticipated variability of the data.

- Studies to quantify behavioral patterns of important mammalian and avian species, such as feral hogs. These studies would entail investigations for important species of the fraction of time spent in the water, preferential defecation locations (herding and flocking behavior), and other factors that impact location of feces deposition.

- Studies to improve methods of estimating animal densities in watersheds. Estimations of livestock and non-domestic animal densities in a watershed are often based on limited data and county-level data, and potentially there may be no existing data for some important sources of fecal bacteria (e.g., feral hogs, coyote, rodents and migratory birds). Improved methods are needed to optimize use of existing data to quantify source densities and to provide means of estimating pet, livestock, and non-domestic animals densities for species for which there is no or limited information. These improved methods could include integration of GIS land use and land cover with various types of census data, enhanced stakeholder involvement and detailed sanitary surveys, and other methods that optimize use of existing data and watershed specific resources and knowledge.
- Studies to improve characterization of bacteria loadings from WWTPs and sanitary sewer collection systems. WWTPs are required to disinfect their effluents using chlorination or other equivalent methods (e.g., ultraviolet radiation), and under conditions of proper operation bacteria loads are typically low from such facilities. However, under less than optimal conditions, WWTPs can directly discharge significant bacterial loads directly into receiving waters. Less than optimal conditions may occur because of wet weather, upsets in operating conditions, etc. Regrowth of bacteria relative to the completeness of disinfection is another area of focus worthy of more study. Regrowth and reactivation of bacteria after the disinfection process of WWTPs effluent appears to be a research topic of some concern within Texas and, in particular, Harris County where existing research is occurring on this subject in effluent dominated receiving streams. Quantification of fecal bacteria concentrations in the effluent of WWTPs will require sampling of facilities of various permitted capacities, different operational conditions and varying ages over a wide range of environmental and operational conditions. A related facet of bacteria loading is associated with sanitary sewer collection systems. Studies to quantify frequency of and conditions causing overflows (e.g., wet weather, grease and rags) from sanitary sewer systems and associated bacteria loadings can provide important information to models of urban areas.

- Studies to quantify failure rates of on-site sewage facilities (OSSFs). Failing OSSFs and even straight pipe discharges can be an important contribution of bacteria loading to some receiving waters. Local information at the county- and city-level will often be important in estimating failed OSSFs in a watershed. However, state- or regional-level statistical analyses relating failure rates to income level, education, age of communities, soil types, rainfall and physiographic region will provide additional support and defense of estimates regarding failure rates and bacteria loadings from OSSFs. As a starting point of these studies, a 2001 study funded by the Texas On-Site Wastewater Treatment Research Council (Reed, Stowe & Yanke, LLC, 2001) provides regional information on reasons for and magnitude of malfunctioning OSSFs.
- Studies to quantify bacteria (especially indicator bacteria such as *E. coli*) production/shedding from non-animal sources, such as mulch, gardens or other organic-rich environments.
- Studies to determine sources of the high concentration of indicator bacteria in rainfall runoff from a spectrum of land uses that might be anticipated to contain low concentrations of bacteria, (e.g., parking lots, streets, building roofs).
- Studies on pathogen sources and forms in stormwater. Research is needed on the sources of pathogenic bacteria in stormwater, the relationship of pathogenic bacteria to indicator bacteria in stormwater as compared to the relationship in such sources as human sewage and the human health significance of the pathogenic bacteria forms found in stormwater.
- Studies to better define indicators and disease risk for water bodies without focused areas of frequent swimming and contact recreation. The original EPA studies to identify indicator bacteria criteria to protect contact recreation use were conducted in lakes with designated swimming areas and nearby wastewater discharge points (EPA, 1986). Criteria development is needed for water bodies without focused or designated areas where swimming and contact recreation use are infrequent. Such water bodies would more likely involve recreational use with incidental contact during boating and wading activities.

Characterization of Kinetic Rates and Transport Mechanisms

Quantification of production, mobilization and movement of fecal bacteria from the landscape to receiving waters and the subsequent fate and transport of these bacteria in receiving waters is addressed by some comprehensive hydrologic/water-quality models as well as simpler *ad hoc* models (e.g., mass balance methods). In many instances, the scientific understanding of relevant processes concerning fate and transport on the landscape and in receiving waters is primitive, and in other instances, this scientific understanding exceeds the representation of those processes in existing models. Note that buildup (i.e., accumulation and proliferation) and mobilization of bacteria on the landscape will also be considered under this loose definition of fate and transport. Within this subsection, the focus will be upon possible studies to enhance the understanding of fate and transport processes, and within the next subsection, potential enhancements to hydrologic/water-quality models will be discussed. This artificial separation, however, should not be taken to mean that the empirical data collection and analyses designed to better understand physical, chemical and biological processes important to landscape and receiving water fate and transport of fecal bacteria can occur in the absence of interaction with model enhancement efforts. On the contrary, such interactions are encouraged and are necessary to the overall success of improving our TMDLs and I-Plans.

- Studies to quantify buildup and mobilization of fecal bacteria from the landscape. The release of fecal bacteria during rainfall-runoff events is a function of many processes antecedent to the event (e.g., rates and timing of manure and feces deposition, *in situ* die-off rates, growth, etc.) and activities during the event (e.g., rainfall duration and intensity and transport processes). Studies that increase understanding of the processes and factors determining bacteria buildup and mobilization and that provide quantification of those processes and factors are needed to enhance our understanding of mechanisms delivering bacteria to receiving waters. These studies can include different land uses (e.g., low-density residential and surface applied manure application fields, to name but two) and could also include investigations of important transport mechanisms such as

surface mobilization and runoff and subsurface movements, including preferential pathways.

- Studies to identify dominant environmental factors that affect bacterial transport in landscapes (e.g., vegetative cover, soil type, temperature, etc.) and persistence and survival of bacteria in soil.
- Studies to quantify bacteria die-off phases in receiving waters. While bacteria die-off is generally assumed to follow Chick's law, experimental data indicate that fecal bacteria often do not follow this law and that die-off occurs in various phases. Further, die-off (and possibly regrowth) during these phases is further complicated by such factors as 1) whether the bacteria are planktonic, attached to suspended sediment, or re-suspended from bed sediments and 2) water temperature, sunlight intensity and levels of nutrients and suspended solids present. Further research to identify and define bacteria die-off phases and to mathematically describe die-off under a variety of conditions will improve our knowledge base on bacteria persistence in receiving waters. Instream regrowth of bacteria under various sediment, nutrient, occurrences of WTTP effluent and water temperature conditions is an additional important consideration under the broad category of bacteria die-off. Harris County is involved in a study on die-off and regrowth of bacteria from WWTP effluents, and any additional studies in this area should be coordinated with that ongoing study.
- Studies to identify dominant environmental factors that affect bacterial transport in streams (e.g., physical and chemical composition of stream waters [pH, total suspended solids, total dissolved solids, nutrients, etc.], temperature, etc.).
- Studies to provide better understanding of spatial and temporal distribution of instream bacteria conditions under various environmental conditions. The State's existing monitoring programs provide insufficient information to quantify spatial and temporal distributions of fecal bacteria at the time scales typically applicable in TMDLs. These time scales may in some instances be less than a day. Intensive spatial and temporal sampling of selected streams under various conditions (e.g.,

rainfall-runoff and dry conditions, high and low water temperatures) could enhance our basic understanding of bacteria issues within the state.

- Studies to quantify suspended bacteria in stream water column and deposited bacteria in stream sediments. Resuspension of bacteria from sediments to water column during peak flows and streambed erosion should be studied as well as, studies to improve understanding of sediment contributions to water-column bacteria levels. Settling of bacteria to sediments during periods of low turbulence and re-suspension of sediments and associated bacteria during conditions of high turbulence are recognized fate and transport processes. Studies are needed to better understand and quantify the interactions of the water column and sediments from the perspective of bacteria fate and transport dynamics in Texas streams and tidal systems.
- Determination of acceptable default inputs which can be used in model development and application. An initial set of inputs representing reasonable estimates of these processes needs to be proposed and agreed upon by the participating agencies. Initially these values should be taken from available literature and as time progresses may be changed to reflect recent research developments.

Enhancements to Bacteria Fate and Transport Models

Hydrologic/water quality models and some of the simpler models that partially incorporate mechanistic processes could benefit from improvements to their fate and transport processes.

- Enhancements to allow better representation of bacteria life cycle and processes. With support from the previously mentioned experimental studies on characterization of bacteria fate and transport, existing models can benefit from improvements and modifications in their representation of bacteria life cycles and processes.

- Enhancements to allow better representation of fate and transport processes. Again, with support from experimental studies, benefits to selected bacteria models would result from improvements in their representations of 1) sorption and bacteria release processes on the landscape and in receiving waters, 2) die-off and regrowth, 3) other fate and transport processes, and 4) effects of various environmental factors (e.g., temperature, light intensity, pH, etc.) on growth and die-off.
- Development of a spatially-explicit tool that will assess bacterial sources, distribute estimated loads to the land as a function of land use and source type, and generate bacterial load input parameters for watershed-scale simulation. This tool can build upon existing and under-development tools, such as SELECT, SPARROW, BLEST and BIT, among others previously mentioned in the Bacteria Fate and Transport Model section.
- Studies to improve linkages of BST and allocation modeling. A stand-alone spatially explicit load allocation tool's results can be validated with BST data or vice versa. Research is needed to determine benefits of linking BST and allocation modeling. At least two questions need to be addressed by these studies. Can BST and modeling be linked in a manner the benefits either TMDL load allocation or I-Plan development? If there were benefits from such linkage, how would that linkage occur?

Bacteria Source Tracking

Within the section on BST, a few areas of research and development were mentioned, and these are repeated below with some additional research items.

- Determine a reasonable expectation for the level of source identification by BST. For example, can BST methods reliably identify to the individual animal species level or is a coarser cut all that is reasonable (e.g., separation into categories of sewage, livestock and non-domestic animals)?

- Investigation and refinement of library-independent BST methods to assist in the TMDL process, and determine which library-independent BST method or methods are best suited for Texas TMDL development and implementation.
- Continue investigations into the most promising library-dependent BST methods, and continue research into promising combinations of such methods.
- Investigate the usefulness of a regional known-source library for BST library-dependent methods. Geographic and temporal stability of BST libraries is not well understood. Additional studies are needed to address these issues and to make library-dependent BST techniques better tools. These studies should build upon and not duplicate ongoing studies in Texas that are cross-validating the libraries generated in the Lake Waco study and the San Antonio watershed study in an attempt to explore issues of geographical and temporal stability of BST libraries, refine library isolate selection, and determine accuracy of water isolate identification.
- Define appropriate ambient water sampling protocol to provide desired statistical confidence with BST findings. For example, in some past studies, 10 to 12 monitoring events are conducted at each station; for each ambient water monitoring event five water samples are collected at a station with one to two minutes between collection of each sample; and two randomly selected bacteria isolates from each cultured sample are subsequently identified using BST methods (e.g., TIAER et al., 2005). Is this level of sampling adequate for the subsequent statistical characterization of sources and determination environmental conditions influencing source contributions (often wet versus dry weather conditions)?
- What is an appropriately sized watershed or sub-watershed for BST sampling design? Beyond what size drainage area do BST results tend to show a wide mix of contributions from all species in a library? Are there factors such as the degree of land development or anthropogenic activity that determine an appropriate drainage size to optimize source discrimination?

These listed items can be developed into a series of studies to advance BST methods and the utility of these methods for Texas bacteria TMDLs.

Determination of Effectiveness of Control Measures

Another research area directly supporting bacteria TMDL and I-Plan development entails determination of effectiveness of various control practices and best management practices to remove and reduce bacteria loadings and concentrations from sources. Success of an I-Plan and willingness of individuals and entities to adopt various control practices and best management practices may be greatly enhanced in the presence of scientifically conducted studies showing efficacy of various practices under a variety of conditions (e.g., rainfall patterns, landscape position, etc.).

- Studies to determine efficiency of agricultural best management practices. Prominent BMPs for bacteria load reduction from agriculture should be evaluated to determine their efficiencies under a variety of environmental conditions. Specifically, those conditions that exceed the ability of these measures to function should be identified.
- Studies to determine efficiency of urban control practices. Prominent control practices for achieving bacteria load reduction from urban and suburban areas should be evaluated to determine their efficiencies under a variety of environmental conditions. Specifically, those conditions that exceed the ability of these measures to function should be identified.

Quantification of Uncertainty and Communication of Risk

Because of the nature of the pollutant, bacteria TMDLs and I-Plans, while using best available information and applying accepted methods of determination, will contain uncertainties. Even if sources of error in field sampling, kinetics modeling and numerical implementation could be eliminated, there is a core uncertainty associated with the "noise" in the bacterial determination methodologies themselves, as indicated by

imprecision in replicate measurements. The spectrum of those involved in bacteria TMDLs (including, to name a few, the regulated community, environmental regulatory agencies, engineers and scientists performing technical evaluations and those providing assistance to unregulated sources) are cognizant of the degree of uncertainty in bacteria TMDL development. Efforts to reduce this uncertainty and to provide heightened defensibility of the process are both worthy and necessary goals, and the research and development studies enumerated above will provide significant advances toward realization of those goals. However, the brutal reality is that over the near future uncertainties — that are sometimes quite large compared to other water-quality parameters — will exist in bacteria TMDL and I-Plan development. Hence, the need exists for one other potential area of research — quantification of uncertainty and the associated communication of risk resulting from the uncertainty associated with TMDLs and I-Plans.

- Studies to quantify uncertainty. A body of literature exists on uncertainty and how to incorporate uncertainty analysis, typically via Monte Carlo techniques, into environmental models. In fact, the latest version of SWAT contains a feature allowing the performance of uncertainty analysis. Some other models employed in TMDL load allocation development do not contain this feature. Knowledge of how to incorporate uncertainty into the TMDL and implementation process can be obtained from existing literature. That knowledge can be transferred to the unique characteristics of the Texas TMDL process for purposes of developing methods to allow quantification of uncertainty.

- Development of proper means to communicate the risks arising from uncertainties in the TMDL process. This communication can assist in overcoming roadblocks that can dampen stakeholder enthusiasm or become excuses for no action in reducing bacteria loadings. Again, a body of literature (e.g., Morgan and Henrion, 1990) and experience exists on risk communication that can be exploited for application to the Texas situation. Successful risk communication can assist in overcoming the uncertainties within bacteria TMDLs and I-Plans and assist in moving the process toward the desired outcome of restored and improved water quality. Areas of study could review lessons learned, examples in the United

States of successful TMDL implementation in spite of uncertainty, and identifying positive paths forward in areas of known difficulties in present and recent bacteria TMDLs that include uncertainty. Results could be implemented to enhance interactions with stakeholders with a focus on the development of best mechanisms of risk communication.

Prioritization of Research and Development Activities

The activities briefly described above are not exhaustive, but these activities do provide indications of major areas where research and development advancements can benefit TMDL and I-Plan development. Additionally, this listing of needs helps to inform the decision makers at the TCEQ and TSSWCB of the current state of the science and techniques. These needs encompass a breadth of activities that most certainly will exceed the resources available for the successful performance of all potential studies. Purposefully, within this section no attempt has been made to prioritize these research and development activities. The selected activities need to be focused to support management decisions, which provide a broad direction to the TMDL and I-Plan processes. For example, such broad direction could include which models are to be supported in the near term for TMDL development. Also, known technical needs can define and inform prioritization of research and development needs that will, over the next several years, improve the tools and methods available to TCEQ and TSSWCB for bacteria TMDL and I-Plan development. Review of existing literature will be an important first step in further defining some of the possible research and development activities enumerated above under several research headings: characterization of sources, characterization of kinetic rates and transport mechanisms, enhancements to bacteria fate and transport models, BST, determination of effectiveness of control measures, and quantification of uncertainty and risk communication.

Some Expert Advisory members voiced certain research preferences during the conference call process used to refine and develop this final report. Those research topics preferentially voiced by certain Expert Advisory members include the following: 1) the

issue of regrowth of bacteria in WWTP effluents and in effluent dominated streams, 2) better quantification of species-specific bacteria production and loadings that used as input to many bacteria fate and transport models, 3) integration of fate and transport models and BST methods within the TMDL and I-Plan process, and 4) improved understanding of the accuracy and precision of BST methods. This list of preferred research topics is included within the more refined lists provided previously within this report section. It should be noted that these four research topics only reflect the views of certain Expert Advisory members and do not necessarily reflect the views of all Expert Advisory members and the Task Force members.

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Appendix 1: Bacteria TMDL Task Force Members and Expert Advisors

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Appendix 2: Models Used in Bacteria TMDLs as Described in EPA Publications

HSPF: Hydrological Simulation Program—Fortran

HSPF is a comprehensive watershed-scale model developed by EPA. The model uses continuous simulation of water balance and pollutant buildup and washoff processes to generate time series of runoff flow rates, as well as pollutant concentration at any given point in the watershed. Runoff from both urban and rural areas can be simulated using HSPF; however, simulation of CSOs is not possible. Because of the comprehensive nature of the model, data requirements for HSPF are extensive and using this model requires highly trained personnel (EPA 2002b).

SWMM: Storm Water Management Model

SWMM is a comprehensive watershed-scale model developed by EPA. It can be used to model several types of pollutants on either a continuous or storm event basis. Simulation of mixed land uses is possible using SWMM, but the model's capabilities are limited for rural areas. SWMM can simulate loadings from CSOs. The model requires both intensive data input and a special effort for validation and calibration. The output of the model is time series of flow, storage and contaminant concentrations at any point in the watershed (EPA 2002b).

STORM: Storage, Treatment, Overflow, Runoff Model

STORM is a watershed-loading model developed by the U.S. Army Corps of Engineers for continuous simulation of runoff quantity and quality. The model was primarily designed for modeling storm water runoff from urban areas, but it also can simulate combined sewer systems. It requires relatively moderate to high calibration and input data. The simulation output is hourly hydrographs and pollutographs (EPA 2002b).

CE-QUAL-RIV1: Hydrodynamic and Water Quality Model for Streams

CE-QUAL-RIV1 is a dynamic, one-dimensional model for rivers and estuaries consisting of two codes — one for hydraulic routing and another for dynamic water quality simulation. CE-QUAL-RIV1 allows simulation of unsteady flow of branched river systems. The input data requirements include the river geometry, boundary conditions, initial in-stream and inflow boundary water quality concentrations and meteorological data. The model predicts time-varying concentrations of water quality constituents (EPA 2002b).

Predicting the response of lakes and estuaries to pathogen loading requires an understanding of the hydrodynamic processes. Shallow lakes can be simulated as a simplified, completely mixed system with an inflow stream and an outflow stream. However, simulating deep lakes or estuaries with multiple inflows and outflows that are affected by tidal cycles is not a simple task. Pathogen concentration prediction is dominated by the processes of advection and dispersion, and these processes are affected by the tidal flow. The size of the lake or the estuary, the net freshwater flow, and wind conditions are some of the factors that determine the applicability of the models. The lake and estuary models are briefly described below (EPA 2002b).

WASP5: Water Quality Analysis Simulation Program

WASP5 is a general-purpose modeling system for assessing the fate and transport of pollutants in surface water. The model can be applied in one, two or three dimensions and can be linked to other hydrodynamic models. WASP5 simulates the time-varying processes of advection and dispersion while considering point and nonpoint source loadings and boundary exchange. The water body to be simulated is divided into a series of completely mixed segments, and the loads, boundary concentrations, and initial concentrations must be specified for each state variable (EPA 2002b).

CE-QUAL-ICM: A Three-Dimensional Time-Variable Integrated-Compartment Eutrophication Model

CE-QUAL-ICM is a dynamic water quality model that can be applied to most water bodies in one, two or three dimensions. The model can be coupled with three-dimensional hydrodynamic and benthic-sediment model components.

CE-QUAL-ICM predicts time-varying concentrations of water quality constituents. The input requirements for the model include 140 parameters to specify the kinetic interactions, initial and boundary conditions, and geometric data to define the water body to be simulated. Model use might require significant expertise in aquatic biology and chemistry (EPA 2002b).

EFDC: Environmental Fluid Dynamics Computer Code

EFDC is a general three-dimensional hydrodynamic model developed by Hamrick (1992). EFDC is applicable to rivers, lakes, reservoirs, estuaries, wetlands and coastal regions where complex water circulation, mixing and transport conditions are present. EFDC must be linked to a water quality model to predict the receiving water quality conditions. HEM-3D is a three-dimensional hydrodynamic eutrophication model that was developed by integrating EFDC with a water quality model. Considerable technical expertise in hydrodynamics and eutrophication processes is required to use the EFDC model (EPA 2002b).

CE-QUAL-W2: A Two-Dimensional, Laterally Averaged Hydrodynamic and Water Quality Model

CE-QUAL-W2 is a hydrodynamic water quality model that can be applied to most water bodies in one dimension or laterally averaged in two dimensions. The model is suited for simulating long, narrow water bodies like reservoirs and long estuaries, where stratification might occur. The model application is flexible because the constituents are arranged in four levels of complexity. Also, the water quality and hydrodynamic routines

are directly coupled, allowing for more frequent updating of the water quality routines. This feature can reduce the computational burden for complex systems. The input requirements for CE-QUAL-W2 include geometric data to define the water body, specific initial boundary conditions and specification of approximately 60 coefficients for the simulation of water quality (EPA 2002b).

QUAL2E: The Enhanced Stream Water Quality Model

QUAL2E is a steady-state receiving water model. The basic equation used in QUAL2E is the one-dimensional advective-dispersive mass transport equation. Although the model assumes a steady-state flow, it allows simulation of diurnal variations in meteorological inputs. The input requirements of QUAL2E include the stream reach physical representation and the chemical and biological properties for each reach (EPA 2002b).

TPM: Tidal Prism Model

TPM is a steady-state receiving water quality model applicable only to small coastal basins. In such locations, the tidal cycles dominate the mixing and transport of pollutants. The model assumes that the tide rises and falls simultaneously throughout the water body and that the system is in hydrodynamic equilibrium. Two types of input data are required to run TPM. The geometric data that define the system being simulated are the returning ratio, initial concentration and boundary conditions. The physical data required are the water temperature, reaction rate, point and nonpoint sources and initial boundary conditions for water quality parameters modeled (EPA 2002b).

BASINS: Better Assessment Science Integrating Point and Nonpoint Sources

BASINS system Version 2.0, with the Nonpoint Source Model (NPSM), can be used to predict the significance of fecal coliform sources and fecal coliform levels watersheds. BASINS is a multipurpose environmental analysis system for use in performing watershed and water quality-based studies. A geographic information system (GIS)

provides the integrating framework for BASINS and allows for the display and analysis of a wide variety of landscape information (e.g., land uses, monitoring stations, point source discharges). The NPSM model within BASINS simulates nonpoint source runoff from selected watersheds, as well as the transport and flow of the pollutants through stream reaches. Through calibration of model parameters and representation of watershed sources, the transport and delivery of bacteria to watershed streams and the resulting in-stream response and concentrations were simulated (EPA 2002a).

Models Used in Bacteria Source Tracking as Described in EPA Publications References

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Appendix 3: EPA Bacteria TMDL Guidelines

This section provides an overview of several EPA guidance documents related to the use of models and BST to develop bacteria TMDLs. Components of a TMDL include (1) Problem Statement, (2) Numeric Targets, (3) Source Assessment, (4) Linkage Analysis, (5) Allocations and (6) Monitoring/Evaluation Plan (for phased TMDLs). Because BST and modeling are primarily used to assist with source assessment, linkage analysis and allocations, this chapter will focus primarily on these components of the TMDL (EPA 2002a).

Overall, EPA allows a great deal of flexibility in bacteria TMDL development as long as the method selected adequately identifies the load reductions or other actions needed to restore the designated uses of the water body in question. There are trade-offs associated with using either simple or detailed approaches. These trade-offs, along with site-specific factors, should always be taken into account and an appropriate balance struck between cost and time issues and the benefits of additional analyses (EPA 2002a).

Source Assessment

Source Assessment involves characterizing the type, magnitude and location of pollutant sources of fecal indicator loading. Source assessments also consider the conditions under which a particular source may have the most influence. For example, nonpoint sources typically predominate during high flow events while point sources predominate under low flows. For this reason, data collection efforts to support source assessment must focus on specific conditions. Monitoring data should be used to estimate the magnitude of loads from the major sources when available. In the absence of such data, a combination of literature values, best professional judgment, BST and empirical techniques/models is necessary. In general, EPA (2002a) recommends the use of the simplest approach that provides meaningful predictions.

EPA (2002a) encourages starting with the assumption that models are not required. If it is determined that models are required, then the following factors should be considered:

- Availability of data and/or funds to support data collection
- Availability of staff
- Familiarity of staff with potential models or other analytical tools
- Level of accuracy required

Depending on the complexity of the sources in the watershed, load estimation might be as simple as conducting a literature search or as complex as using a combination of long-term monitoring and modeling. Analysis of pollutant waste loads from point sources are generally recommended to be based on the effluent monitoring required for the National Pollutant Discharge Elimination System (NPDES) permit or based upon the permit's effluent limitations (EPA 2002a). However, many NPDES (and TPDES permits in Texas) require monitoring of chlorine residual to verify the effectiveness of chlorination but do not include either report or effluent limits for bacteria.

Nonpoint source loads are typically separated into urban and rural categories since runoff processes differ between these environments. Pathogen loads in urban stormwater can be estimated using a variety of techniques, ranging in complexity from simple loading rate assumptions and constant concentration estimates, to statistical estimates, to highly complex computer simulation (EPA 2002a). Examples of techniques for estimating pathogen loads in urban storm water include the FecaLOAD model, constant concentration estimates, statistical or regression approaches and stormwater models, such as SWMM and HSPF.

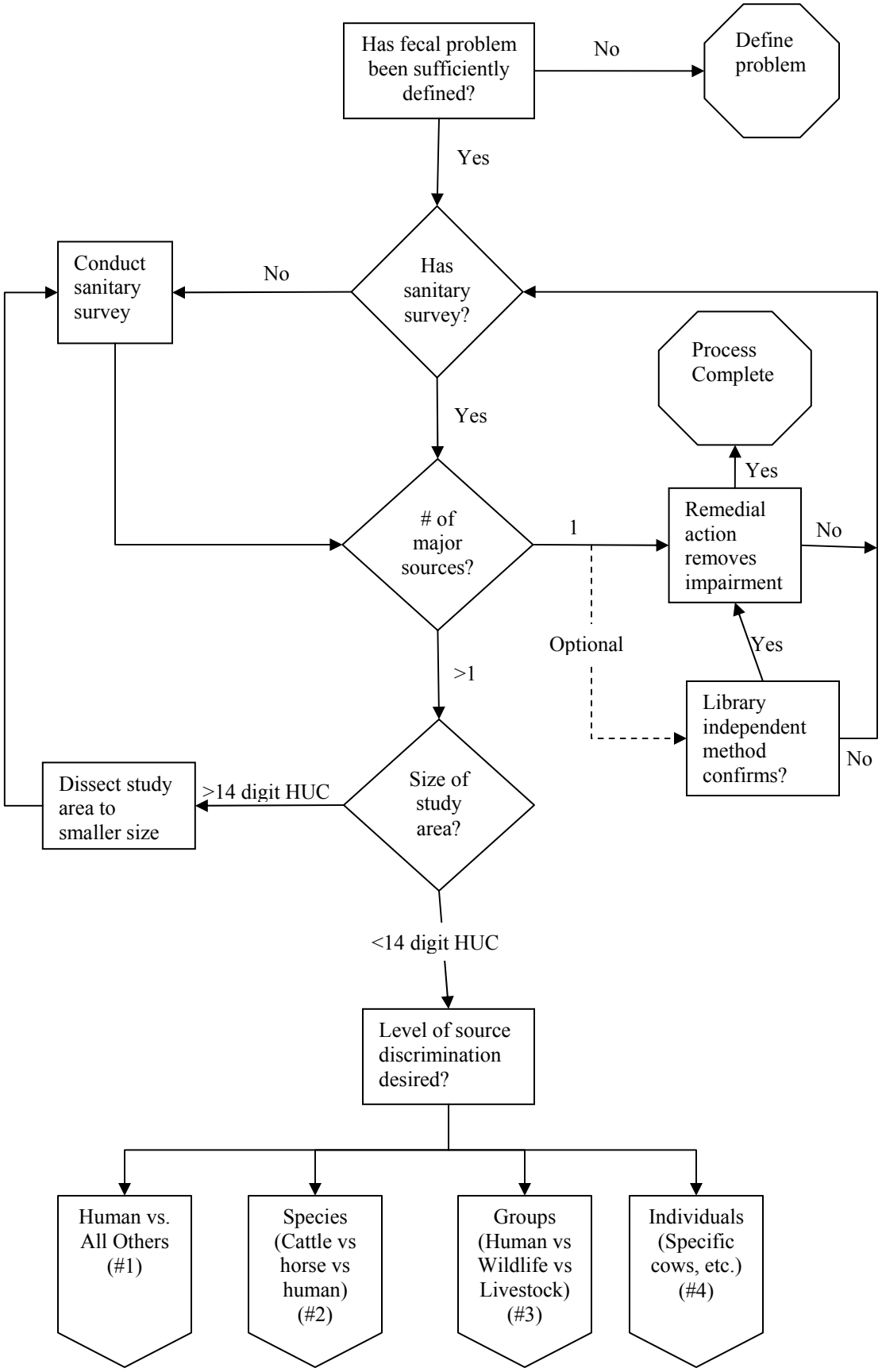
Rural nonpoint source loads may also be estimated using a variety of techniques, ranging from simple loading function estimates to use of complex simulation models. Techniques, such as the loading function approach, site-specific analysis, estimates of time series of loading and detailed models, such as AGNPS (Agricultural Nonpoint Source), may be used (EPA 2002a). Models are discussed in greater depth in the Linkage Analysis section.

DNA fingerprinting may also provide information for Source Assessments (EPA 2002a). There are many BST methods available and more are under development. Overall, molecular BST methods may offer the most precise identification of specific types of sources, but are limited by high costs and detailed, time-consuming procedures (EPA 2002c). Costs vary however, based on:

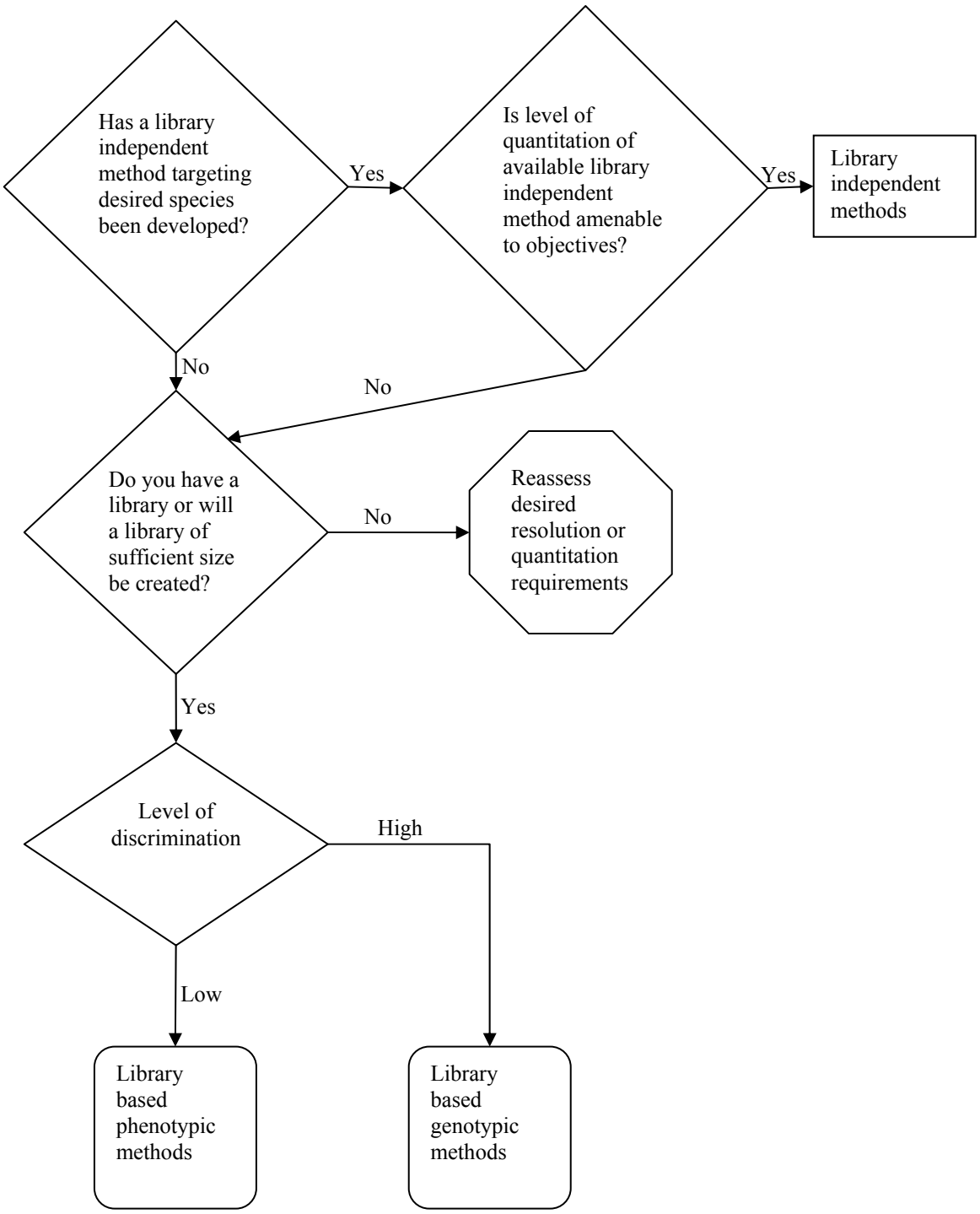
- Analytical method used
- Size of the database needed
- Number of environmental isolates analyzed
- Level of accuracy needed
- Number of subwatersheds and geographical size of the area under study

Comparison studies have shown that no single method is clearly superior to the others. Thus, the decision on which method to use depends on the unique set of circumstances associated with the area in question, the results of sanitary surveys, and budgetary and time constraints. A decision tree was created by EPA to assist in deciding whether BST methods are necessary to determine the sources of fecal pollution in a particular watershed and, if so, which group of methods might be most appropriate (EPA 2005). The decision tree included on the next four pages consists of five steps:

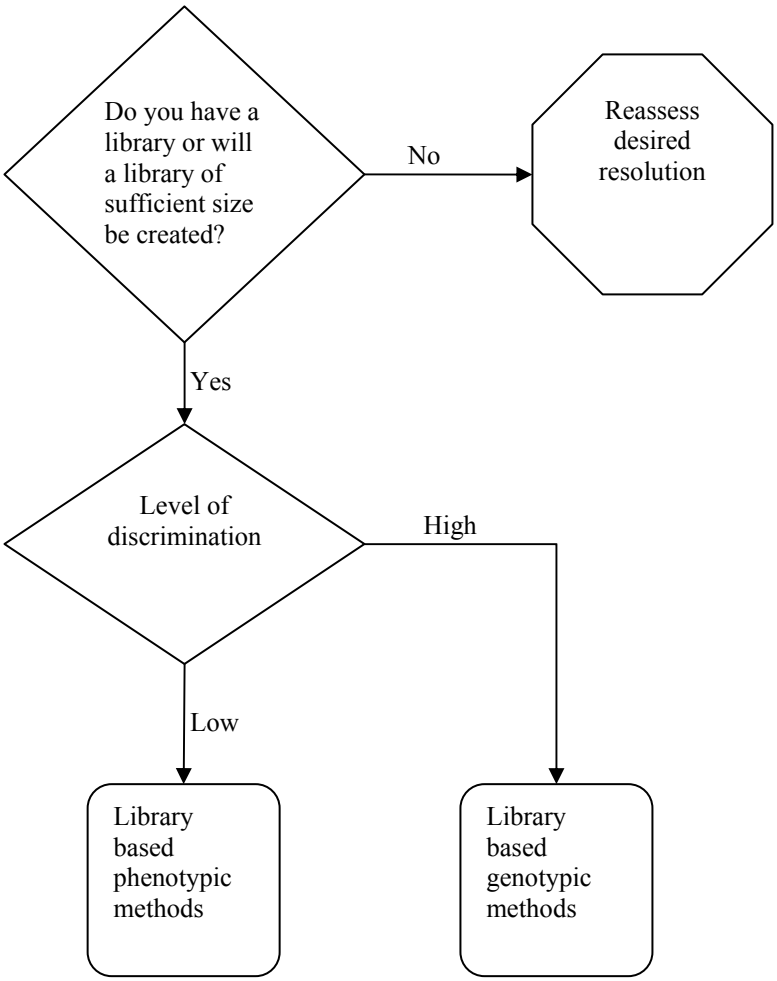
- Adequately defining the problem
- Conducting a sanitary survey
- Determining the potential number of major sources
- Ensuring the watershed/study area is of manageable size
- Determining the desired level of discrimination


Has fecal problem been sufficiently defined?
No
Define problem
Yes
Has sanitary survey?
No
Conduct sanitary survey
Yes
Process Complete
Yes
# of major sources?
1
Remedial action removes impairment
No
Optional
Yes
>1
Library independent method confirms?
No
Size of study area?
>14 digit HUC
Dissect study area to smaller size
<14 digit HUC
Level of source discrimination desired?
Human vs. All Others (#1)
Species (Cattle vs horse vs human) (#2)
Groups (Human vs Wildlife vs Livestock) (#3)
Individuals (Specific cows, etc.) (#4)

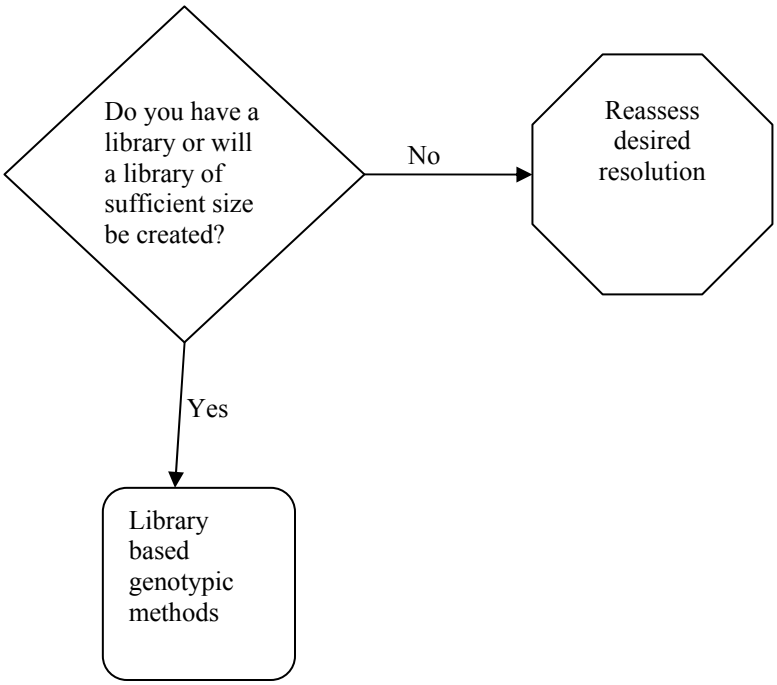
Human only #1 and Species Specificity #2 (EPA 2005)



Define by Groups #3 (EPA 2005)



Define to specific sources by type and location #4 (EPA 2005)



Linkage Analysis and Allocations

EPA (2003) has identified three analytical methods appropriate for calculating loads and linking water quality targets and sources:

- Empirical Approaches – When sufficient observations are available, existing data can be used to determine linkage between sources and water quality targets (e.g., regression approach).
- Simple Approaches – When permitted sources are sole source of bacteria, simple dilution calculations and/or compliance monitoring are adequate.
- Detailed Modeling – When sources of bacteria are complex, a water quality modeling approach (e.g. dynamic or steady-state modeling) is typically used. When detailed modeling is used, different types of models are required for accurate simulation for rivers and streams as compared to lakes and estuaries because the response is specific to the water body.

Steady-state modeling uses constant inputs for effluent flow and concentration, receiving water flow and meteorological conditions and is generally used where insufficient data exists for developing a dynamic model. Steady-state modeling provides very conservative results when applied to wet weather sources. If a state elects to use a steady state model, EPA recommends a dual design approach (e.g., load duration curve) where the loadings for intermittent or episodic sources are calculated using a flow duration approach and the loadings for continuous sources are calculated based on a low flow statistic (EPA 2003).

Dynamic modeling considers time-dependent variation of inputs and applies to the entire record of flows and loadings. In certain situations, EPA (2003) recommends the use of dynamic modeling to calculate loads. The three dynamic modeling techniques recommended are:

- Continuous simulation
- Monte Carlo simulation
- Log-normal probability modeling

Specific models recommended by EPA can be divided into two categories — watershed loading models and pathogen concentration prediction models. Loading models provide estimates of either the total pollutant loading or a time series loadings. The key watershed loading models suited for pathogens include HSPF, SWMM and STORM (EPA 2002b). These are briefly described in Appendix 3 and the previous section about models.

Prediction of pathogen concentration in rivers and streams is dominated by advection and dispersion processes and bacteria die-off. One-, two- and three-dimensional models have been developed to describe these processes. Water body type and data availability are the two most important factors that determine model applicability. For most small and shallow rivers, one-dimensional models are sufficient. However, for large and deep rivers and streams, two- or three-dimensional models that integrate the hydrodynamics of the system should be used (EPA 2002b). The river and stream models are briefly described in Appendix 3 and include the following:

- HSPF: Hydrological Simulation Program–FORTRAN
- CE-QUAL-RIV1: Hydrodynamic and Water Quality Model for Streams
- WASP5: Water Quality Analysis Simulation Program
- CE-QUAL-ICM: A Three-Dimensional Time-Variable Integrated-Compartment Eutrophication Model
- EFDC: Environmental Fluid Dynamics Computer Code
- CE-QUAL-W2: A Two-Dimensional, Laterally Averaged Hydrodynamic and Water Quality Model
- QUAL2E: The Enhanced Stream Water Quality Model
- TPM: Tidal Prism Model

In closing, EPA (2002a) recommends that when developing linkages between water quality targets and sources, states should:

- Use all available and relevant data (specifically monitoring data for associating water body responses with flow and loading conditions).
- Perform a scoping analysis using empirical analysis and/or steady-state modeling to review and analyze existing data prior to any complex modeling. The scoping

analysis should include identifying targets, quantifying sources, locating critical points, identifying critical conditions, and evaluating the need for more complex analysis.

- Use the simplest technique that adequately addresses all relevant factors when selecting a technique to establish a relationship between sources and water quality response.

EPA Bacteria TMDL Guidelines References

EPA (Environmental Protection Agency). 2002a. Protocols for Developing Pathogen TMDLs. EPA 841-R-00-002.

EPA (Environmental Protection Agency). 2002b. National Beach Guidance and Required Performance Criteria for Grants. June 2002.

EPA (Environmental Protection Agency). 2002c. Wastewater Technology Fact Sheet – Bacterial Source Tracking. EPA 832-F-02-010.

EPA (Environmental Protection Agency). 2003. Implementation Guidance for Ambient Water Quality Criteria for Bacteria. DRAFT Document.

EPA (Environmental Protection Agency). 2005. Microbial Source Tracking Guide Document. EPA/600-R-05-064.

Appendix 4: State Approaches to Bacteria TMDLs

This section provides a brief overview of approaches other states are using to develop TMDLs for bacteria and related issues. EPA has allowed much flexibility in developing pathogen TMDLs, as outlined in the agency's 2002 publication "Protocols for Developing Pathogen TMDLs" and 2003 DRAFT publication "Implementation Guidance for Ambient Water Quality Criteria for Bacteria." As a result, states have taken a variety of approaches to developing bacteria TMDLs. A large number of bacteria TMDLs have been approved by EPA since 1996 (Table 1). Most of these have been approved in states in which EPA is under court order or agreed consent decree to establish TMDLs.

Table 1. EPA Approved Pathogen TMDLs in Regions 3, 4, 6 and 7.

EPA Region	**State**	**EPA Approved TMDLs**						**Court Order**	**Litigation Dismissed**
		FC	**Pathogen**	**Bacteria**	**Fecal**	**TC**	***E. coli***		
3	Delaware			25				Yes	No
3	DC	22	9					Yes	No
3	Maryland	57	1					No	Yes
3	Pennsylvania	1	100					Yes	No
3	Virginia	186	94		1			Yes	No
3	W. Virginia	196						Yes	No
4	Georgia	534						Yes	No
4	Florida	21			48	45		Yes	No
4	Kentucky	23						No	No
4	Alabama	26						Yes	No
4	Mississippi	172						Yes	No
4	N. Carolina	38			1			No	Yes
4	Tennessee	62					191	Yes	No
4	S. Carolina	270						No	No
6	Arkansas	2						Yes	No
6	Louisiana	27						Yes	No
6	New Mexico	20						Yes	No
6	Oklahoma	0						No	Yes
6	Texas	0						No	No
7	Kansas	471						Yes	No
7	Nebraska	11					20	No	No
7	Missouri	3						Yes	No
7	Iowa		1					Yes	No

TMDLs that are prepared hurriedly to meet court-mandated deadlines may not be sufficiently specific or achieve adequate stakeholder acceptance to support development of comprehensive I-plans, particularly where multiple sources and types of sources are involved. Accordingly, the approval of a large number of bacteria TMDLs does not necessarily represent a significant improvement in water quality.

A brief overview of the TMDL methods used by states in EPA Regions 4 and 6, along with a few examples from select states in other regions is presented here. No examples were found of any state that has successfully improved water quality by implementing a bacterial TMDL. EPA Region 4 was initially targeted by this survey because environmental conditions (e.g. climate, rainfall, temperature) are most like the eastern portion of Texas where a majority of the 303(d) listings occur. This was later expanded to other regions. Much of the information for this summary was acquired from EPA's TMDL Web site at http://www.epa.gov/owow/tmdl/ and the Web sites of the individual states referenced.

EPA Region 6

In EPA Region 6, a total of 49 fecal coliform TMDLs are reported to have been approved since January 1, 1996. The load duration curve model was the primary model used to develop the approved TMDLs to date. The only exceptions are the two TMDLs developed in Arkansas, which used empirical methods.

Although no bacteria TMDLs are currently approved in Texas, a number of bacteria TMDLs and watershed protection plans are under way. Texas has taken the approach of spending more time in developing TMDLs that are designed to achieve stakeholder buy-in and result in meaningful implementation of water quality improvements. Texas has primarily used the HSPF and load duration curve models for a majority of the TMDLs under development to date (Table 2).

Table 2. Bacteria TMDLs Under Development in Texas.

Project	HSPF	Load Duration	Other Models	Bacteria Source Tracking Method
Upper San Antonio River	❶			ERIC-PCR and RiboPrinting
Leon River	❶			ERIC-PCR and RiboPrinting
Peach Creek	❶			ERIC-PCR and RiboPrinting
Adams and Cow Bayous	❶		RMA2/ACE	No BST
White Oak and Buffalo Bayous	❶			ARA and CSU
Lower San Antonio River		❶		ERIC-PCR and RiboPrinting
Atascosa River		❶		No BST
Elm and Sandies Creeks		❶		No BST
Upper Trinity River		❶		Ribotyping (Institute for Environmental Health, Inc., Seattle, WA)
Guadalupe River above Canyon Lake		❶		Ribotyping (Source Molecular Corporation, Inc., Miami, FL)
Upper Oyster Creek		❶		Ribotyping (Institute for Environmental Health, Inc., Seattle, WA)
Copano Bay and Mission and Aransas Rivers			ArcHydro\Monte Carlo Simulation	ARP and PFGE
Oso Bay and Oso Creek			ArcHydro\SWAT	No BST
Gilleland Creek		❶		No BST
Clear Creek		❶		
Metropolitan Houston (Brays, Greens, Halls and other Bayous)	❶			ARA and CSU
WPP – Lake Granbury				
WPP – Buck Creek			TBD	*E. faecium*, ERIC-PCR, RP
WPP – Bastrop Bayou				
WPP – Plum Creek		❶	SELECT, SPARROW, SWAT	No BST

Unlike other states in Region 6, Texas has supplemented the models utilizing BST. The primary BST methods that have been used include:

- ERIC-PCR conducted at TAES – El Paso
- RiboPrinting conducted at TAES – El Paso
- Antibiotic Resistance Analysis (ARA) conducted at the University of Houston
- Carbon Source Utilization (CSU) conducted at the University of Houston
- Ribotyping conducted at Institute for Environmental Health, Inc., Seattle, WA
- Ribotyping conducted at Source Molecular Corporation, Inc in Miami, Florida
- Antibiotic Resistance Profiling (ARP) conducted at Texas A&M University – Corpus Christi
- Pulse Field Gel Electrophoresis (PFGE) conducted at Texas A&M University-Corpus Christi and Texas A&M University – College Station

EPA Region 4

A total of 1,146 fecal coliform, 191 *E. coli*, 49 fecal and 45 total coliform TMDLs are reported to have been approved in EPA Region 4 since January 1, 1996. As in Region 3, litigation has driven much of the TMDL development. Only Kentucky and South Carolina have escaped litigation.

Georgia has led the way in TMDL approval. EPA Region 4 completed a number of these (e.g., Chickasawatchee Creek) using the BASINS model (HSPF) for both source analysis and for linking sources to indicators. LDCs and the equivalent site approach (as described below) have also been used extensively in Georgia. Georgia has not explicitly published a guidance document describing its methodology for developing bacteria TMDLs; however, their "Analytical Approach" is outlined in TMDLs, such as the "TMDL Evaluation for the Chattahoochee River Basin (Fecal coliform)" which can be found at: www.gadnr.org/epd/Files_PDF/techguide/wpb/TMDL/Chattahoochee/chatt_fc_tmdl.pdf. For those segments in which sufficient water quality data was available to calculate at least one 30-day geometric mean that was above the regulatory standard, the load duration curve approach was used. The method involves comparing the "current" critical

load to summer and winter seasonal TMDL curves. For listed segments that do not have sufficient data to calculate the 30-day geometric mean fecal coliform concentrations, Georgia used an equivalent site approach to estimate the "current" and TMDL loads. This approach involves calculating loads based on the relationship of the stream segments that lack sufficient data to equivalent site(s) that have data. This method provides estimates that can be refined as additional data are collected. The WLA loads are calculated based on the permitted or design flows and average monthly permitted fecal coliform concentrations or a fecal coliform concentration of 200 counts/ 100 mL as a 30-day geometric mean. The LA is calculated as the remaining portion of the TMDL load available after allocating the WLA and the MOS.

The Florida Department of Environmental Protection has developed and published their methodology in a document titled "TMDL Protocol" in June 2006. Source identification in Florida begins with evaluating existing data and developing GIS layers to map probable sources. If the sources cannot be determined with existing data and field inspections, then simple and inexpensive chemical and biological tracers are used for identifying sources during dry and wet weather conditions. If the simple methods do not provide the level of source identification needed, then more complex and expensive BST methods are used. Modeling tools are then used to evaluate flows and loads, define allowable loads, and evaluate BMPs. No watershed models are specifically identified for developing bacteria TMDLs; however, a number of receiving water models are identified (e.g. AESOP, CE-QUAL-R1, CE-QUAL-RIV1, EFDC, EPDRiv1, HEC-5Q, HSPF-RCHRES, SWMM, TWQM, and WASP). A document such as this would be very beneficial in Texas: (http://www.dep.state.fl.us/water/tmdl/docs/TMDL_Protocol.pdf).

Kentucky is one of only two states in EPA Region 4 not under litigation or consent decree to complete TMDLs. Like Georgia, Kentucky has not published specific guidance on development of TMDLs; however, a review of its approved TMDLs shows that, mass balance and load duration curves have been used to develop a large number of their bacteria TMDLs. Their approach to using LDCs is typically outlined under the "Data Analysis" section of their TMDLs, such as the Brush Creek and Crooked Creek TMDL:

http://www.water.ky.gov/NR/rdonlyres/B2EE6B7D-A658-4FA6-A0BA-67C10EB77369/0/TMDLBrush_Crooked.pdf

In this TMDL, where a LDC was used, the allowable loading curves was divided into five zones representing high flows (0-10%), moist conditions (10-40%), median or mid-range flows (40-60%), dry conditions (60-90%), and low flows (90-100%). If more than five sample points plot above the allowable load line, a trendline can be drawn through the data violations for predicting the load at other duration intervals and the correlation factor (r^2) determined. If insufficient numbers of samples are available or r^2 is not high, then no line is drawn. Existing loads are expressed as a range based on the zones where the violations occur (e.g. median flow, moist conditions, etc.). When multiple violations occur within a zone, the existing load is represented as 90th percentile value. The percent reduction required to meet the criterion is calculated based on the 90th percentile of coliform concentrations collected during the recreation season that violate the fecal coliform target and calculated as follows:

Percent Reduction (%) = (existing concentration – target) / existing concentration x 100

Alabama, like most other states, has not published a protocol for developing bacteria TMDLs. A review of approved TMDLs in Alabama on EPA's TMDL website reveals that a variety of approaches have been used including:

- Empirical models
- Loading Simulation Program in C++ (LSPC), Environmental Fluid Dynamics Code (EFDC), Water Quality Analysis Simulation Program (WASP)
- BASINS Watershed Characterization System (WCS) and Nonpoint Source Model (a modified version of HSPF)
- Mass balance
- Load duration curves (LDCs)

A review of approved TMDLs in Mississippi reveals that state has primarily utilized empirical linear regression models, BASINS NPSM, and mass balance. The BASINS NPS Model (NPSM), a modified version of HSPF, was used in the Pearl River TMDL, (http://www.deq.state.ms.us/MDEQ.nsf/pdf/TWB_PearlRivMarionDc00/$File/PearlRBPearlRiv(Marion)Dc00.pdf?OpenElement) for estimating current conditions. The key reason for using BASINS as the modeling

framework was its ability to integrate both point and nonpoint sources in the simulation, as well as its ability to assess instream water quality response.

North Carolina has used a number of models including BASINS HSPF, LDCs and Watershed Analysis Risk Framework (WARMF). Load duration curves are based on the cumulative frequency distribution of flow conditions in the watershed. Allowable loads are average loads over the recurrence interval between the 95th and 10th percentile flow record (excludes extreme drought (>95th percentile) and floods (<10th percentile). Percent reductions are expressed as the average value between existing loads (typically calculated using an equation to fit a curve through actual water quality violations) and the allowable load at each percent flow exceeded.

Tennessee utilizes a variety of models including the BASINS Watershed Characterization System and NPS Model (NPSM); Loading Simulation Program in C++ (LSPC) / Hydrologic Simulation Program –FORTRAN (HSPF) / Watershed Characterization System (WCS) model combination, LDCs and mass balance.

South Carolina has primarily used LDCs. In limited circumstances, they have also used empirical methods, mass balance or the BASINS/HSPF/WSC combo. A "TMDL Talk" on TMDLS.NET titled *Watershed Characterization & Bacteria TMDL's: South Carolina's Approach* may indicate greater use of BASINS/HSPF/WSC in coming years. The use of the Watershed Characterization System (WSC) ensures adequate consideration of the wide array of sources and is a key component of the technical approach toward building bacteria TMDLs and describing allocation options. In evaluating pollutant sources, loads are characterized using the best available information (e.g. monitoring data, GIS data layers, literature values and local knowledge). Pollutant sources are then linked to water quality targets using analytical approaches including WCS and the Nonpoint source Model (NPSM), a modified version of HSPF. Estimates of loading rates are generated by fecal coliform spreadsheet tools included with WCS. These loading rate estimates are then used by NPSM to simulate the resulting water quality response. Allocation for point sources considers discharge-monitoring

information. NPS allocations for significant categories are identified at key points in the watershed from the model analyses. This approach was used for the Rocky Creek TMDL and others.

Other States

Connecticut and Delaware use the Cumulative Frequency Distribution Function Method, developed by the Connecticut Department of Environmental Protection, to develop TMDLs. The reduction in bacteria density from current levels needed to achieve compliance with state water quality standards is quantified by calculating the difference between the cumulative relative frequency of the sample data set (a minimum of 21 sampling dates during the recreational season) and the criteria adopted to support recreational use. Adopted water quality criteria for *E. coli* are represented by a statistical distribution of the geometric mean 126 and log standard deviation 0.4 for purposes of the TMDL calculations. TMDLs developed using this approach are expressed as the average percentage reduction from current conditions required to achieve consistency with criteria. The procedure partitions the TMDL into wet and dry weather allocations by quantifying the contribution of ambient monitoring data collected during periods of high stormwater influence and minimal stormwater influence to the current condition.

In EPA Region 7, a total of 485 fecal coliform, 20 *E. coli* and 1 pathogen TMDLs are reported to have been approved since October 1, 1995. Development in Kansas, Missouri, and Iowa has been driven by court orders. Much like EPA Region 6, LDCs appear to be the method of choice for developing bacteria TMDLs. Bacteria TMDLs approved in Kansas, Missouri and Nebraska primarily used LDCs. Kansas has lead the way in the use of LDCs. Kansas TMDL Curve Methodology can be found at the following website: (http://www.kdheks.gov/tmdl/basic.htm#data).

Use of LDCs in Nebraska is described in the document entitled "Nebraska's Approach for Developing TMDLs for Streams Using the Load Duration Curve Methodology." Only one pathogen TMDL (*E. coli)* has been approved in Iowa. Iowa used the Soil and Water Assessment Tool (SWAT) model to estimate daily flow into Beeds Lake. The SWAT

flow estimates were then used to create a load duration curve. Use of EPA's bacterial indicator tool was used to identify the significance of bacteria sources in the watershed.

Indiana also uses LDCs and provides very good "Duration Curve Information" on their website (http://www.in.gov/idem/programs/water/tmdl/documents.html) and includes helpful tutorials, spreadsheets and publications on developing LDCs.

Nevada Division of Environmental Protection uses Load Duration Curve Methodology for Assessment and TMDL Development and has developed guidance on their website (http://www.ndep.nv.gov/bwqp/loadcurv.pdf) for developing LDCs.

New Jersey's policy is published in a fact sheet titled "Fecal Coliform TMDLs" (http://www.nj.gov/dep/watershedmgt/DOCS/tmdlfactsheet2.pdf). Nonpoint sources and stormwater point sources were identified as the primary contributors to all listed streams while Publicly Owned Treatment Works (POTWs) were considered insignificant and not allocated any loading reductions. The percent reduction was calculated empirically; however, when data was available, LDCs were used. Sources are identified using river assessments, visual surveys, stakeholder input and aerial photography.

New York used a statistical rollback method to estimate reductions needed and the Watershed Treatment Model (WTM) for characterizing the water bodies. The method for determining the geomean rollback factor follows:

$$F_{rollback} = (\text{Observed geomean} - \text{water quality standard})/(\text{Observed geomean})$$

The same method is applied for the 90th percentile values and standards:

$$F_{rollback} = (\text{Observed } 90^{th} \text{ percentile} - \text{water quality standard})/(\text{Observed } 90^{th} \text{ percentile})$$

The most restrictive of the two (i.e., the greatest percent reduction required) is chosen as the target reduction. More information on the WTM is included in section 6 of the "Final Report for Peconic Bay TMDL" (http://www.dec.state.ny.us/website/dow/pecpart56.pdf).

Virginia's approach is outlined through a series of Guidance Memos. "HSPF Model Calibration and Verification for Bacteria TMDLs" is outlined in Guidance Memo No. 03-

2012 (http://www.deq.virginia.gov/waterguidance/pdf/032012.pdf). Guidance Memo No. 03-2015 (http://www.deq.virginia.gov/waterguidance/pdf/032015.pdf) describes the "Method for Representing WLAs in Bacteria TMDLs". These memos were released to achieve greater consistency in their TMDL development. Even though the state is under court order, it is similar to Texas in many respects. Virginia develops bacteria TMDLs primarily using either LDCs or the HSPF model (or a modified version – NPSM); however, in a number of TMDLs, BST has been utilized in conjunction with simplified modeling approaches. One such example is the Little Wicomico River Watershed TMDL and Coan River Watershed TMDL, where Virginia DEQ utilized its point source inventory, a shoreline survey, and antibiotic resistance analysis to determine the potential sources of bacteria and quantify source loadings from humans, livestock and non-domestic animals. In addition, a simplified modeling approach (Tidal Volumetric Model) was used. The BST data was used to determine the relative sources of fecal coliform violations and ambient water quality data used to determine the load reductions needed to attain the applicable criteria. The most recent 30 months of data coinciding with the end of the TMDL study were reviewed to determine the loading to the water body. The geometric mean loading is based on the most recent 30-month geometric mean of fecal coliform. The load is also quantified for the 90th percentile of the 30-month grouping.

The geometric mean load is determined by multiplying the geometric mean concentration based on the most recent 30-month period of record by the volume of the water. The acceptable load is then determined by multiplying the geometric mean criteria by the volume of the water. The load reductions needed for the attainment of the geometric mean are then determined by subtracting the acceptable load from the geometric mean load.

Example: (Geometric Mean Value MPN/100ml) x (volume) = Existing Load
(Criteria Value 14 MPN/100ml) x (volume) = Allowable Load
Existing Load – Allowable Load = Load Reduction

The 90th percentile load is determined by multiplying the 90th percentile concentration, based on the most appropriate 30-month period of record, by the volume of the water. The acceptable load is determined by multiplying the 90th percentile criteria by the volume of the water. The load reductions needed for the attainment of the 90th percentile criteria are determined by subtracting the acceptable load from the 90th percentile load. The more stringent reductions between the two methods (i.e. 90th percentile load or geometric mean load) are used for the TMDL. The more stringent method is combined with the results of the BST to allocate source contributions and establish load reduction targets among the various contributing sources.

The BST data determines the percent loading for each of the major source categories and is used to determine where load reductions are needed. Since one BST sample per month is collected for a period of one year for each TMDL, the percent loading per source is averaged over the 12-month period if there are no seasonal differences between sources. The percent loading by source is multiplied by the more stringent method (i.e. 90th percentile load or geometric mean load) to determine the load by source. The percent reduction needed to attain the water quality standard or criteria are allocated to each source category.

Washington primarily uses LDCs for calculating bacteria TMDLs. To identify nonpoint sources of bacteria, a yearlong (minimum) water quality study of possible source areas is conducted. Once the locations of the bacterial sources are narrowed down, the state works with local interests to identify sources of pollution. Two methods that can be used to identify bacteria sources: (1) pinpointing the location of the source and (2) identifying the types of sources contributing to the problem. One of the most economical methods pinpointing the locations of sources is to conduct intensive upstream-downstream water quality monitoring, including flow measurements, to identify specific stream reaches, land uses or tributaries that are a problem. Dye testing can also be used for pinpointing the locations of sources. BST can be used to determine the types of sources. Most BST techniques are quite costly; thus, it is important to pick the appropriate method and time to use BST. BST does not tell you how much each source contributes to bacterial

contamination, only the different kinds of sources. In addition, it is possible that not all source types will be identified or, with some techniques, that sources will be misidentified. Washington State Department of Ecology's Fact Sheet "Focus on Bacterial Source Tracking" provides an overview of their approach for identifying sources and using BST (http://www.ecy.wa.gov/pubs/0310059.pdf).

Other states using BST for TMDLs include New Hampshire and Maryland. Ribotyping has been used in New Hampshire to determine sources of bacteria for TMDLs. Maryland is using Antibiotic Resistance Analysis (ARA) BST methodology to determine the relative contribution of bacteria from various categories.

EPA Bacteria TMDL Guidelines References

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

By

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August, 2008

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Executive Summary

Evaluation of Existing Monitoring Programs

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
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

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).

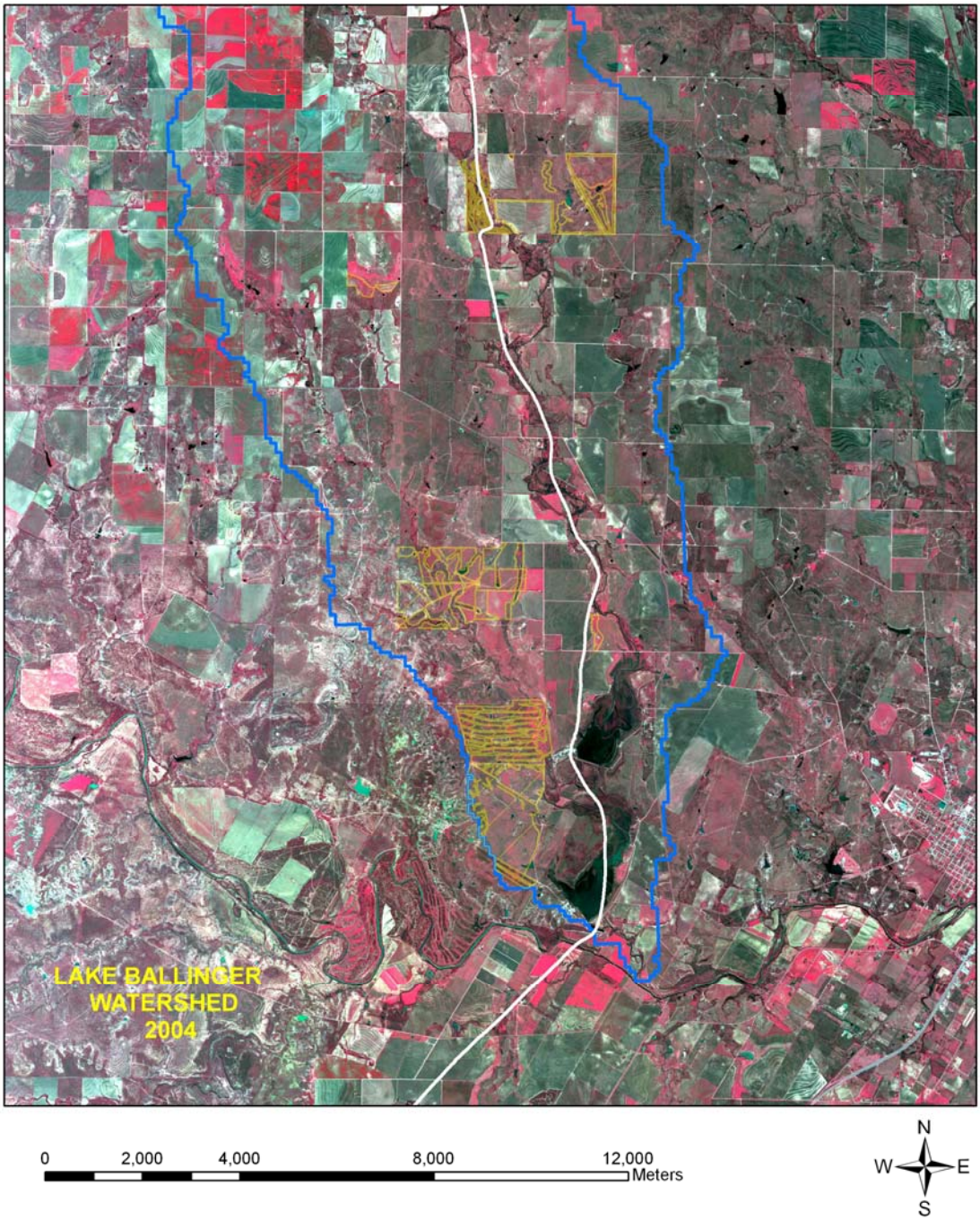

Figure 1.6. Southern end of Lake Ballinger watershed (white line indicates helicopter flight path).

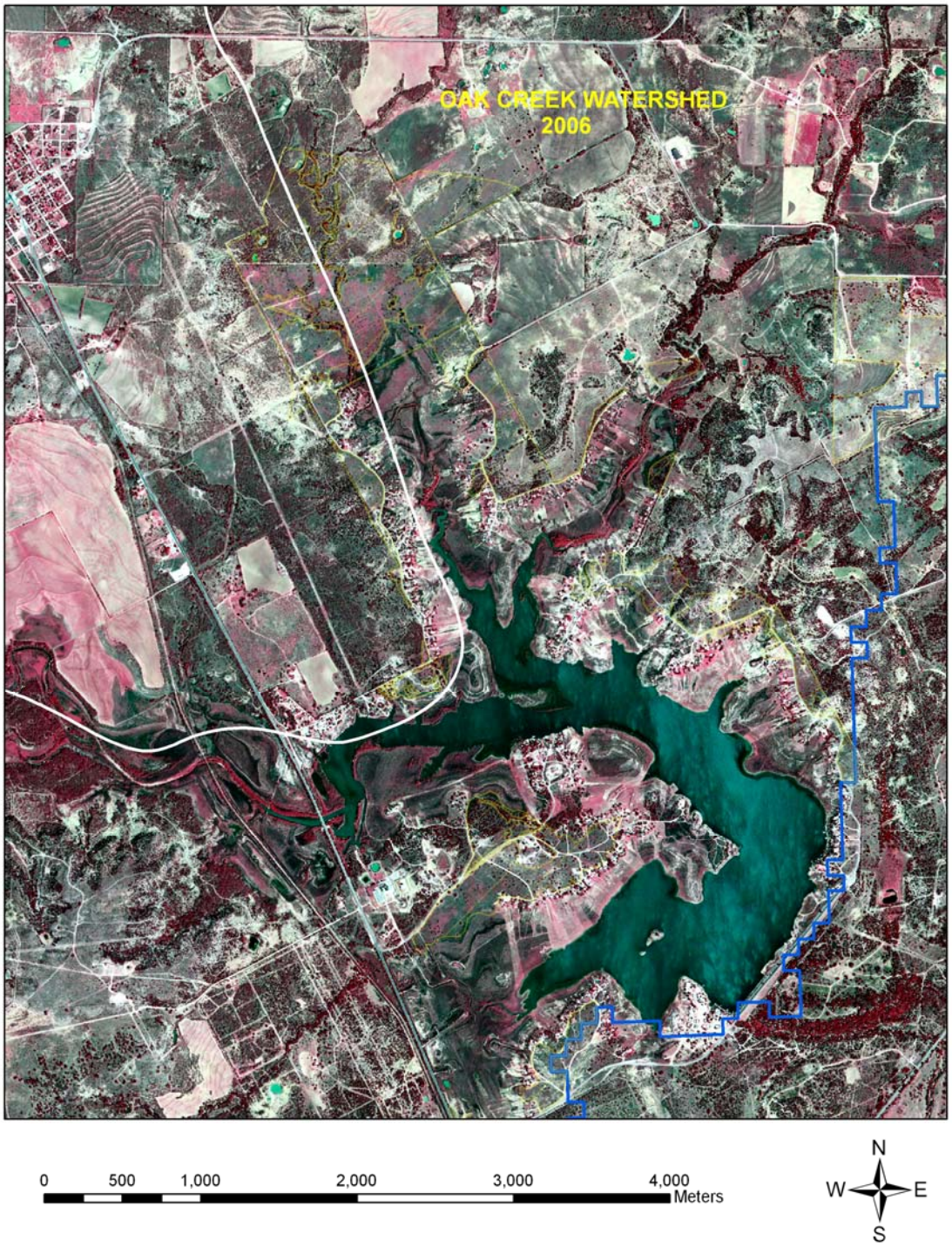

Figure 1.7. Oak Creek watershed area at Oak Creek Reservoir (white line indicates helicopter flight path).

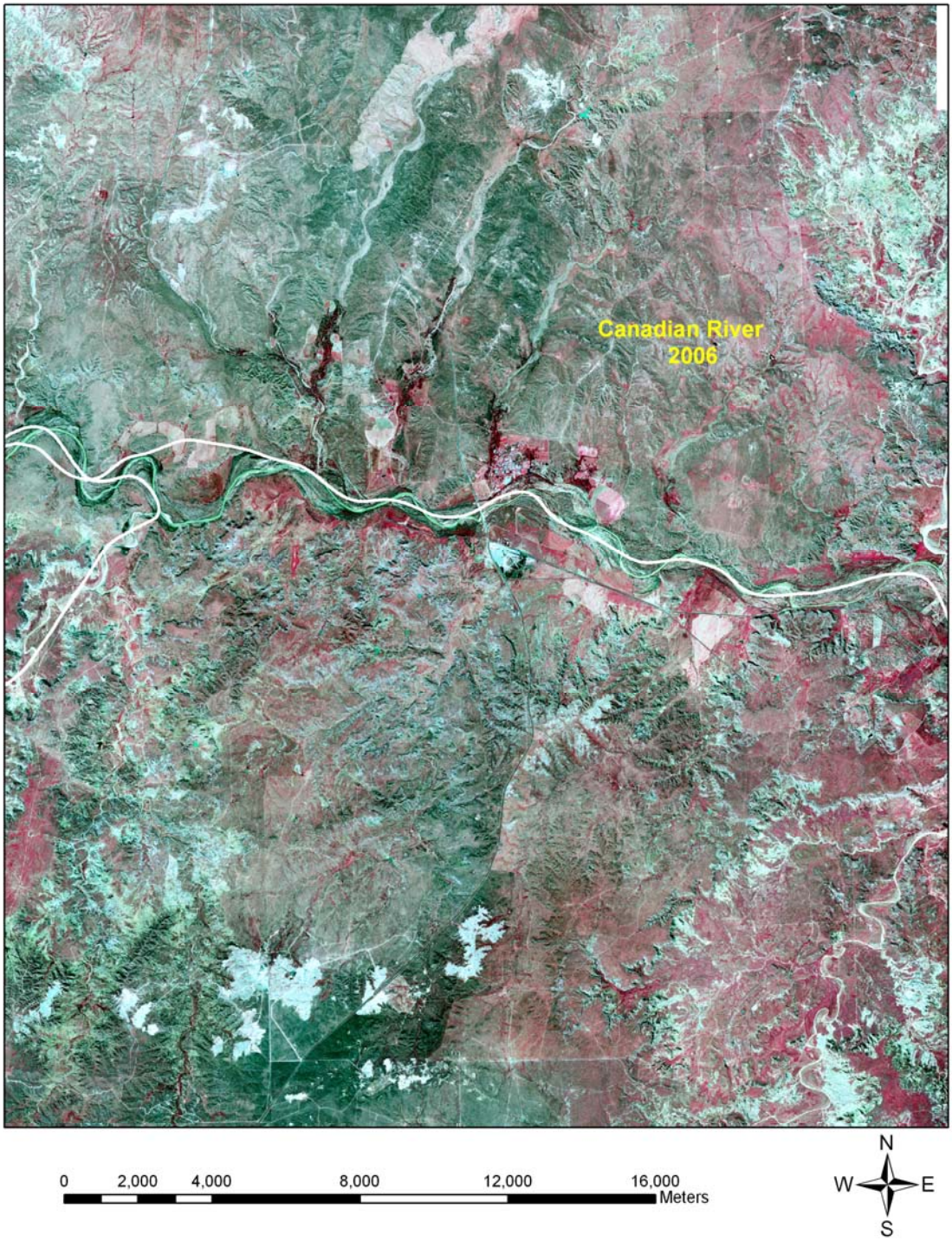

Figure 1.8. Canadian River channel near Boys Ranch, Texas (white line indicates helicopter flight path).

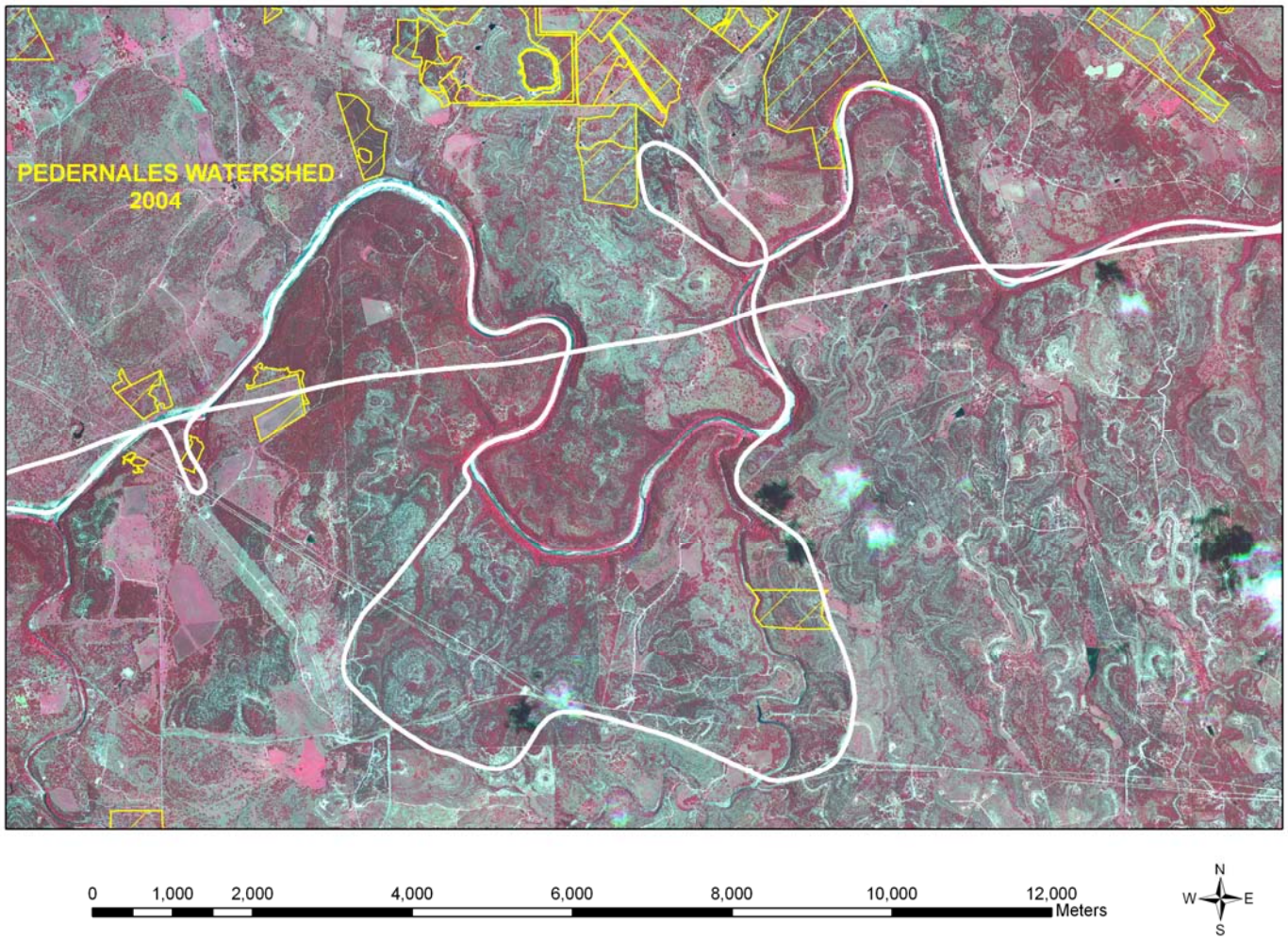

Figure 1.9. Pedernales Watershed with 2004 date image background (white line indicates helicopter flight path).

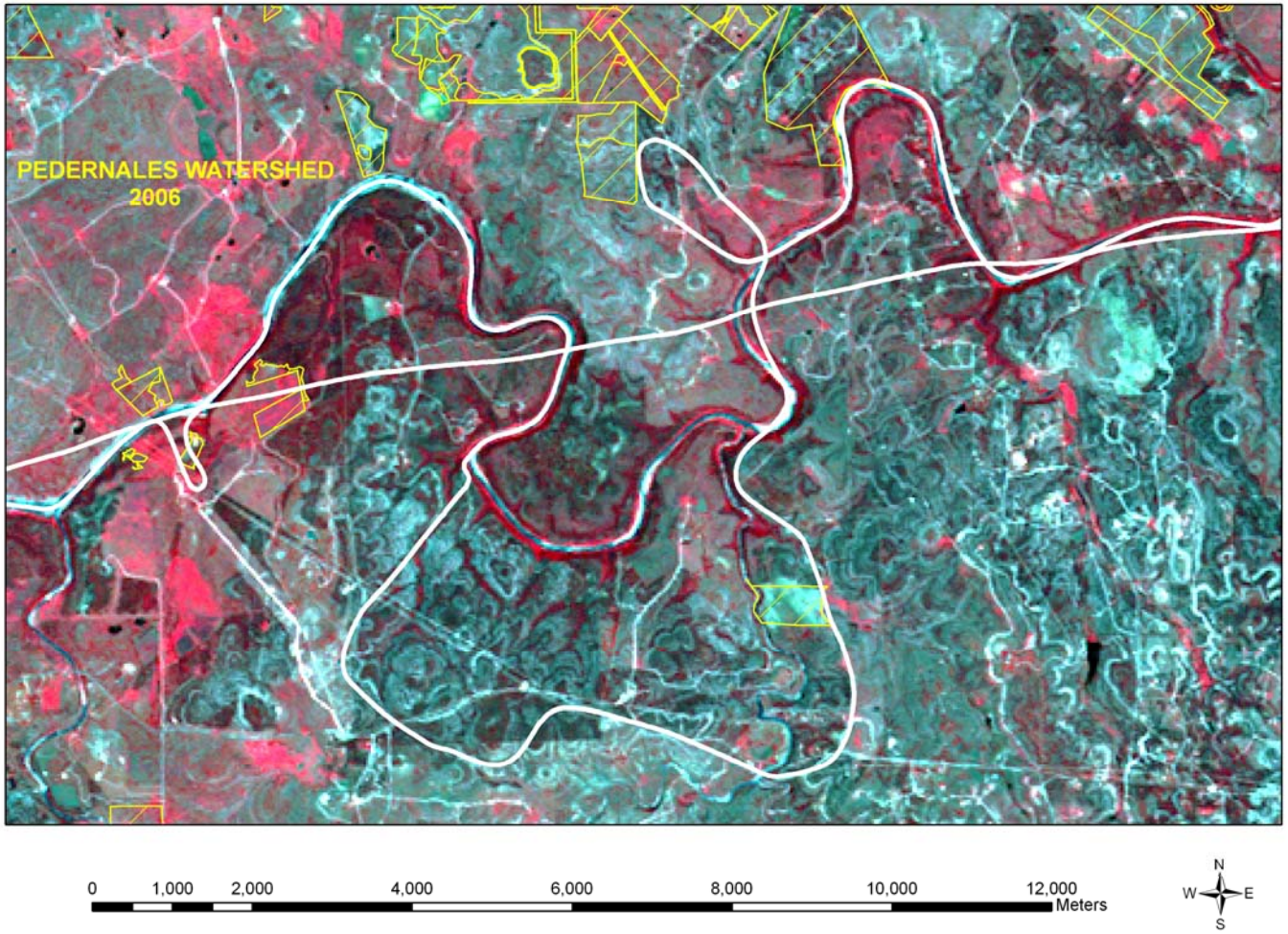

Figure 1.10. Pedernales Watershed with 2006 date image background (white line indicates helicopter flight path).

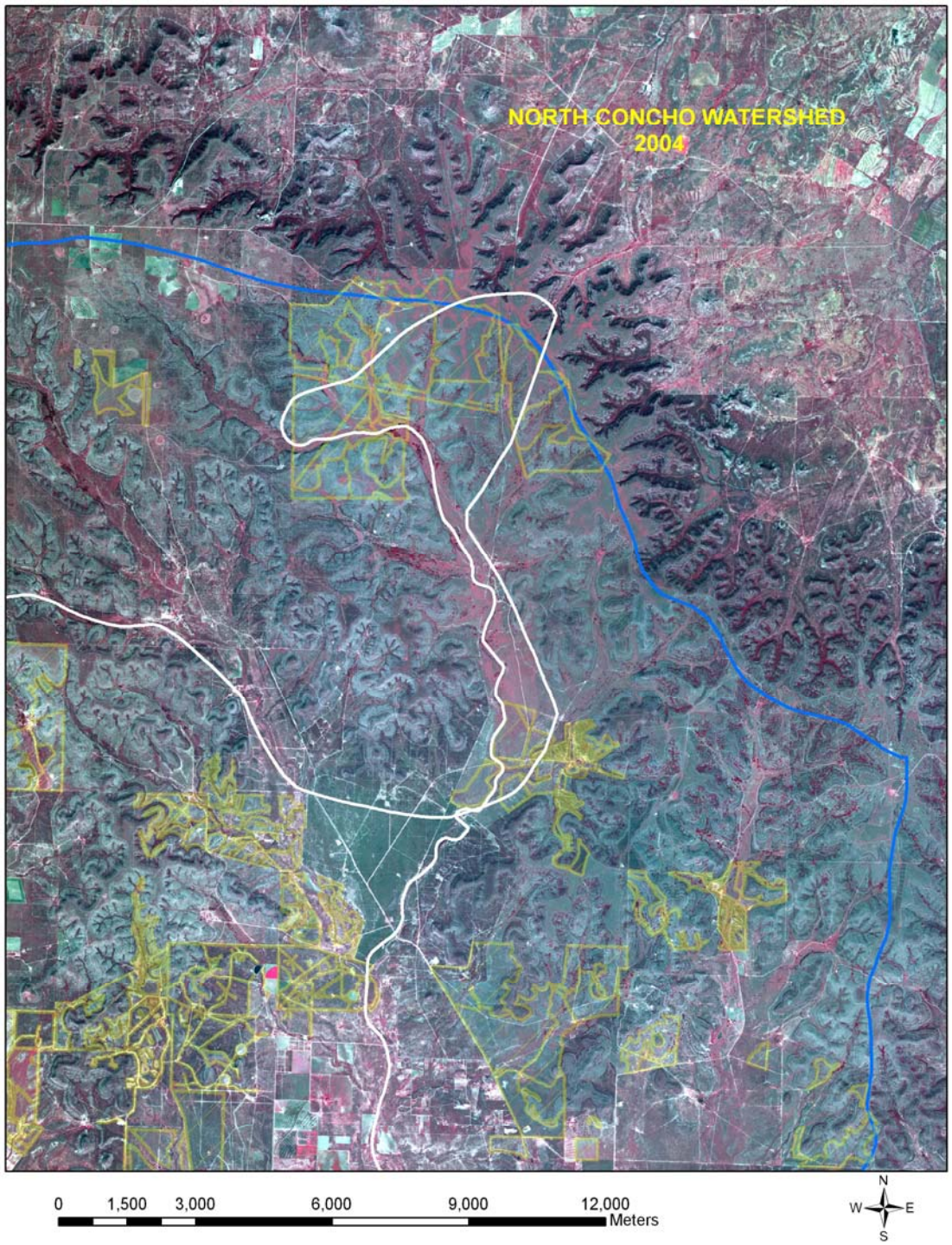

Figure 1.11. North Concho watershed vicinity of Grape Creek (white line indicates helicopter flight path).

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

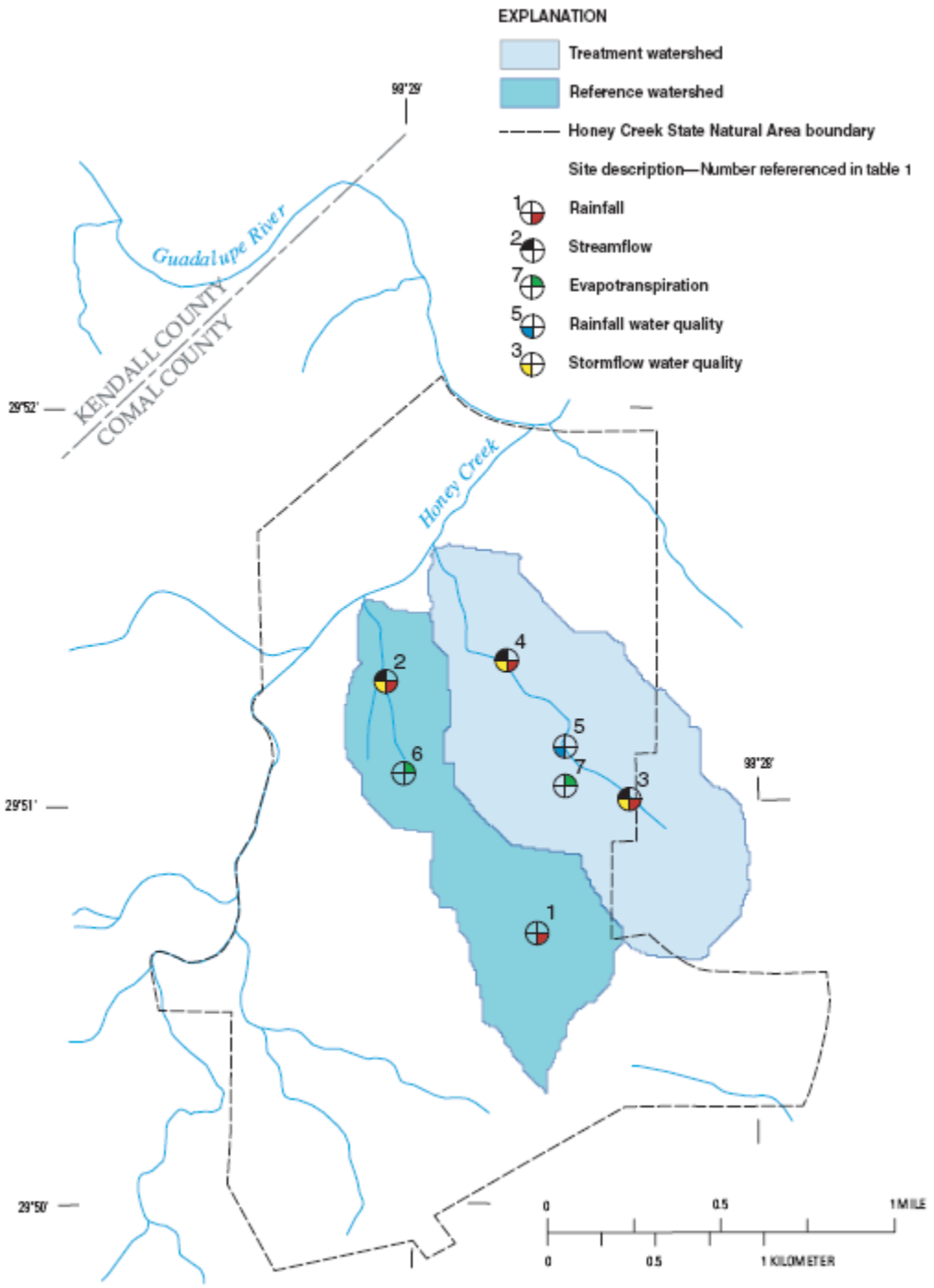

Figure 2.1. Locations of data collection sites in the paired watershed study in the Honey Creek State Natural Area, in Comal County (Slattery et al., 2006).

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. Sonoran Desert washes, characterized by a Mediterranean-type climate, where most precipitation occurs in the winter, and there is deep drainage. 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. Chihuahuan Desert environment where annual precipitation is between 200 to 300 mm. 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. Semiarid rangelands in west Texas, on range sites where annual precipitation is about 450 mm, such as Lubbock, Texas. 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. Texas range sites with higher annual precipitation (about 700 mm). 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. Semiarid riparian ecosystems where mesquite shrubs or trees mainly exploit groundwater source. 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. Certain regions characterized by deep drainage. 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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November, 2007

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.

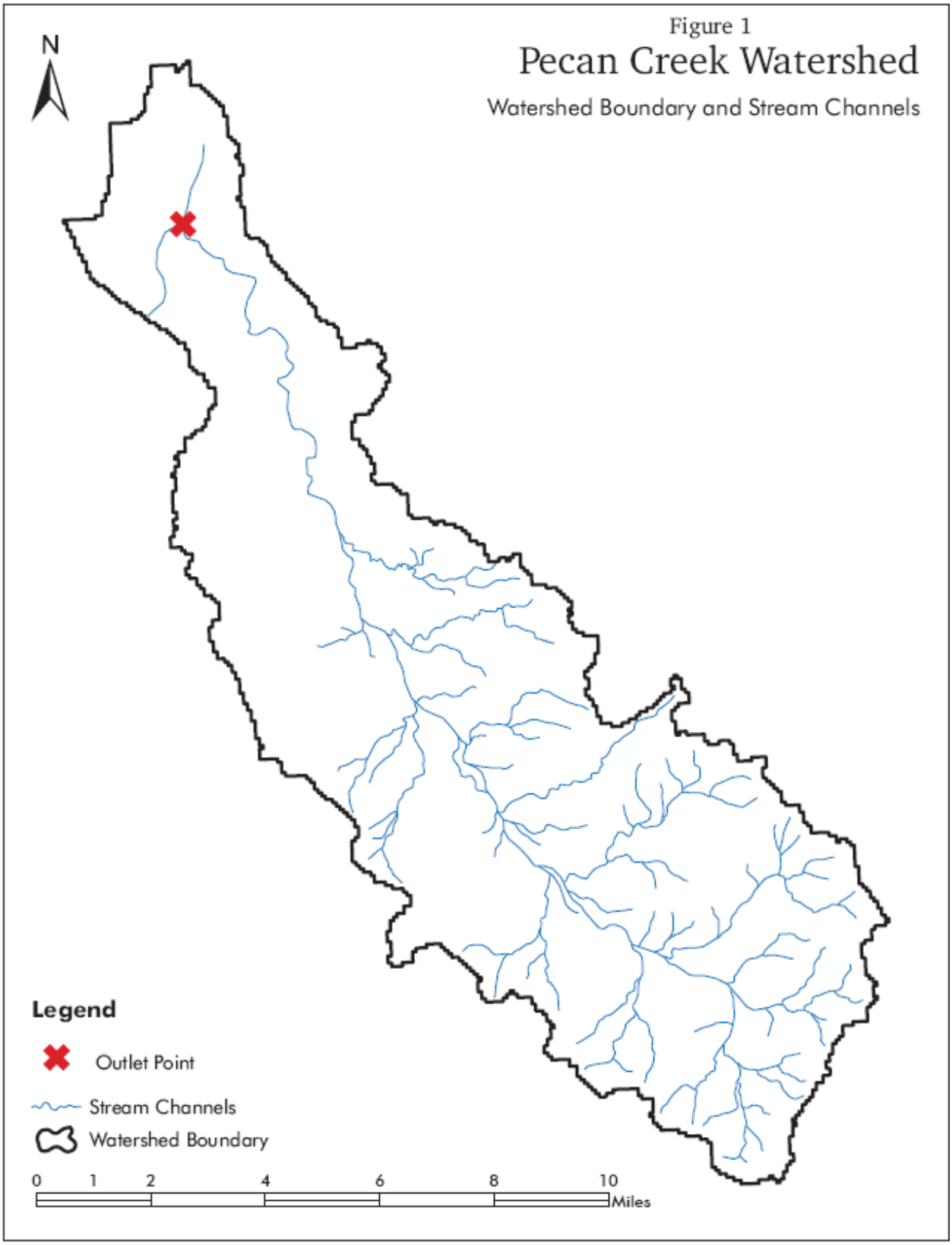
Figure 1
Pecan Creek Watershed
Watershed Boundary and Stream Channels
N
Legend
Outlet Point
Stream Channels
Watershed Boundary
0
1
2
4
6
8
10
Miles

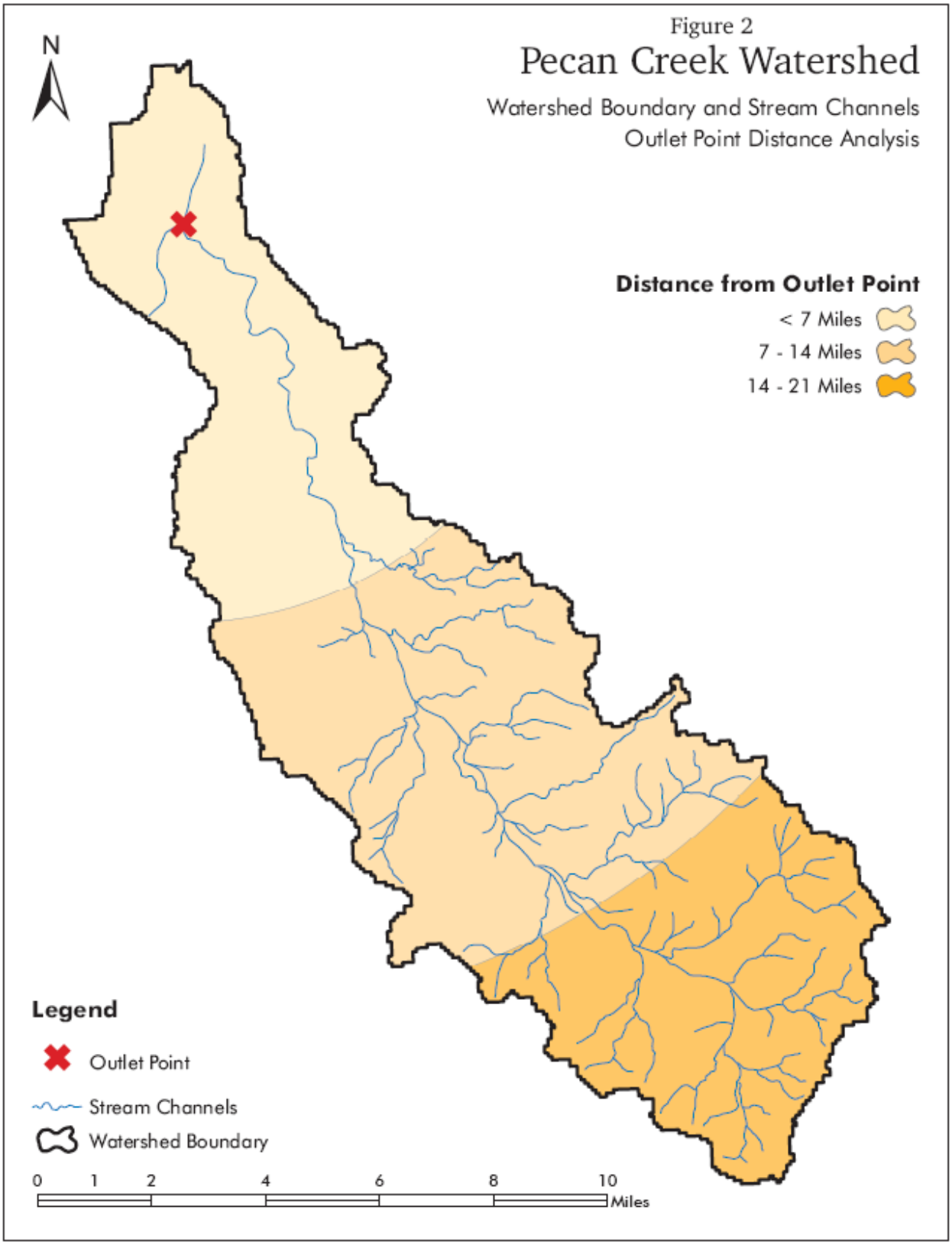
Figure 2
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Outlet Point Distance Analysis
N
Distance from Outlet Point
< 7 Miles
7 - 14 Miles
14 - 21 Miles
Legend
Outlet Point
Stream Channels
Watershed Boundary
0 1 2 4 6 8 10
Miles

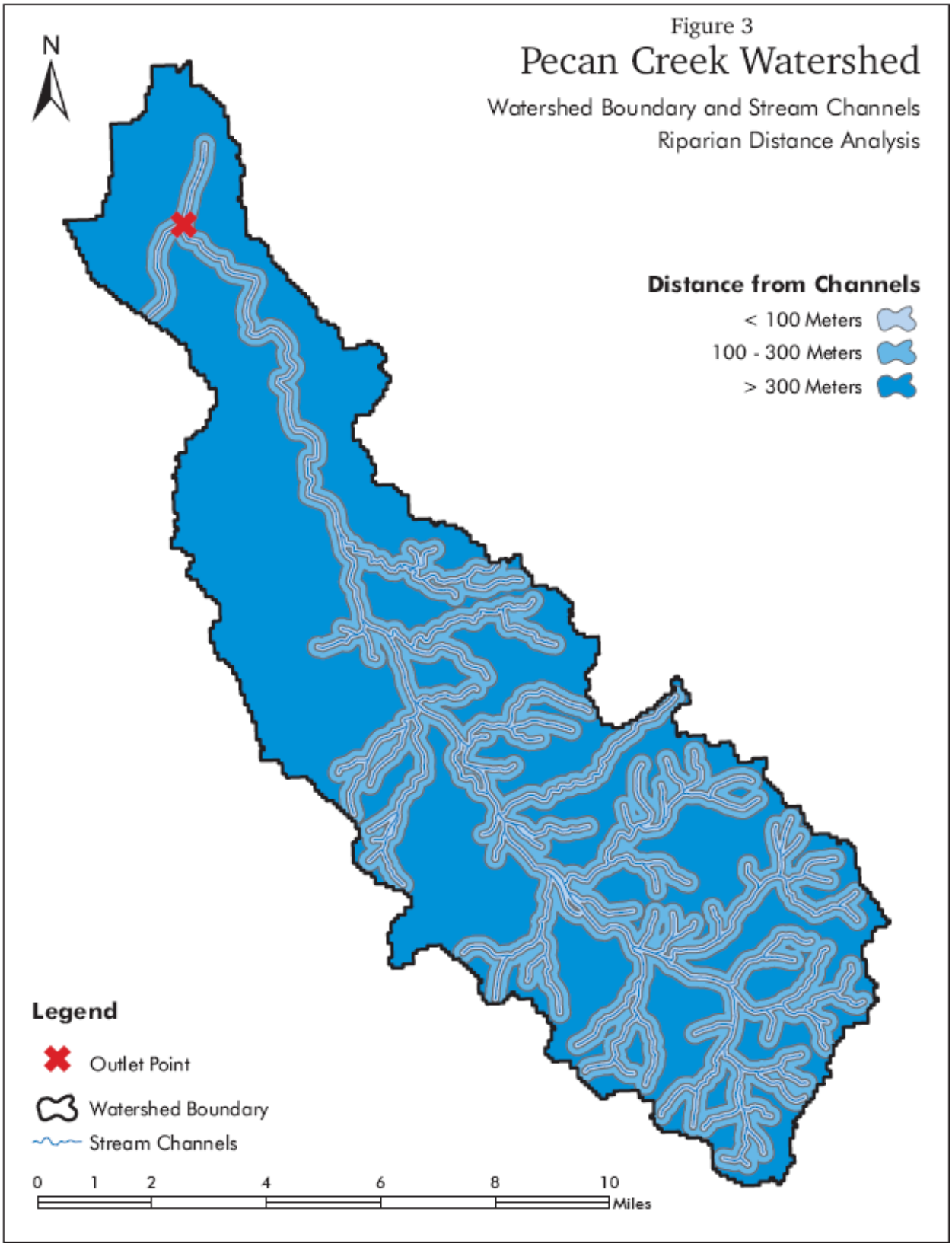
Figure 3
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Riparian Distance Analysis
N
Distance from Channels
< 100 Meters
100 - 300 Meters
> 300 Meters
Legend
Outlet Point
Watershed Boundary
Stream Channels
0 1 2 4 6 8 10
Miles

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

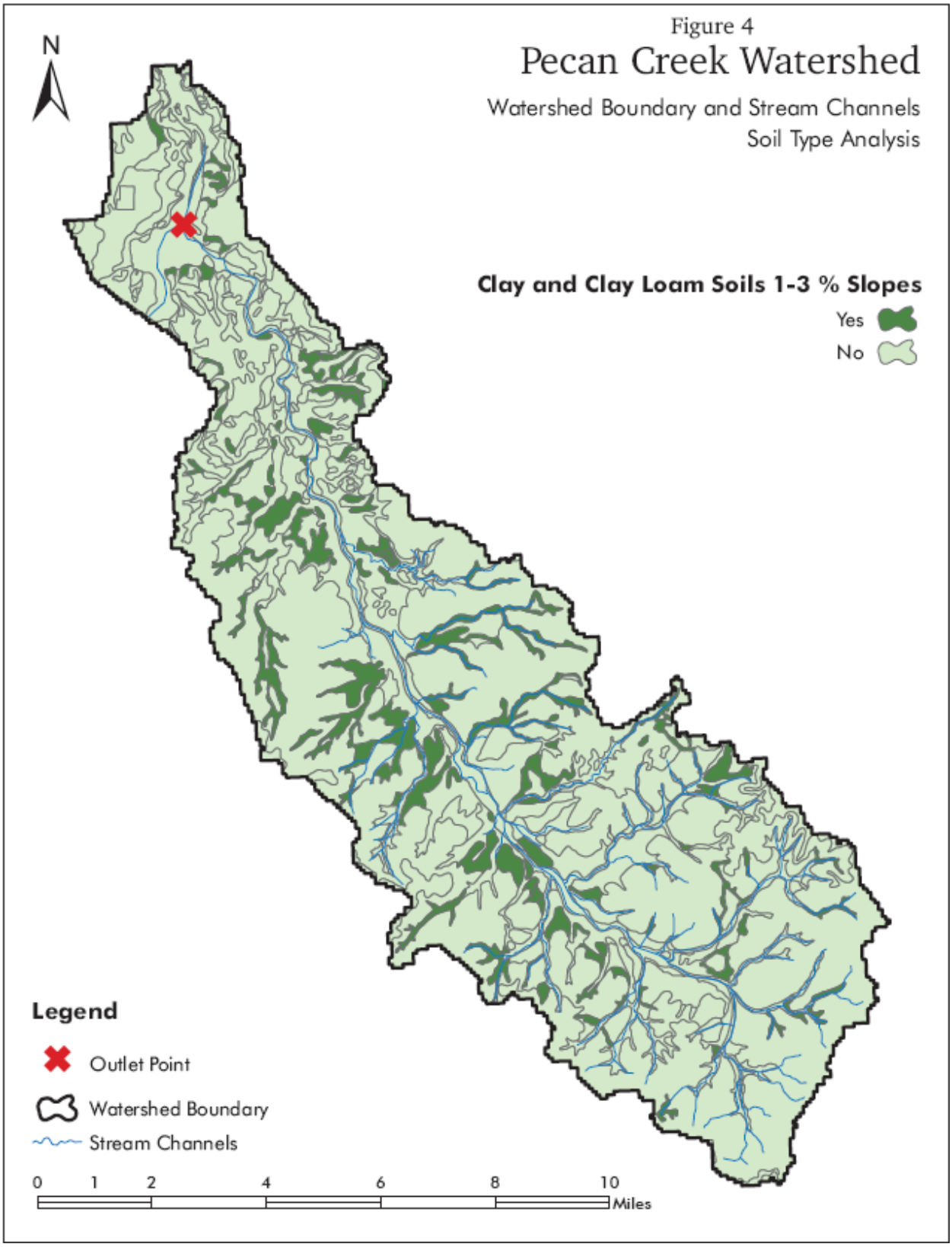
Figure 4
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Soil Type Analysis
N
Clay and Clay Loam Soils 1-3 % Slopes
Yes
No
Legend
Outlet Point
Watershed Boundary
Stream Channels
0 1 2 4 6 8 10
Miles

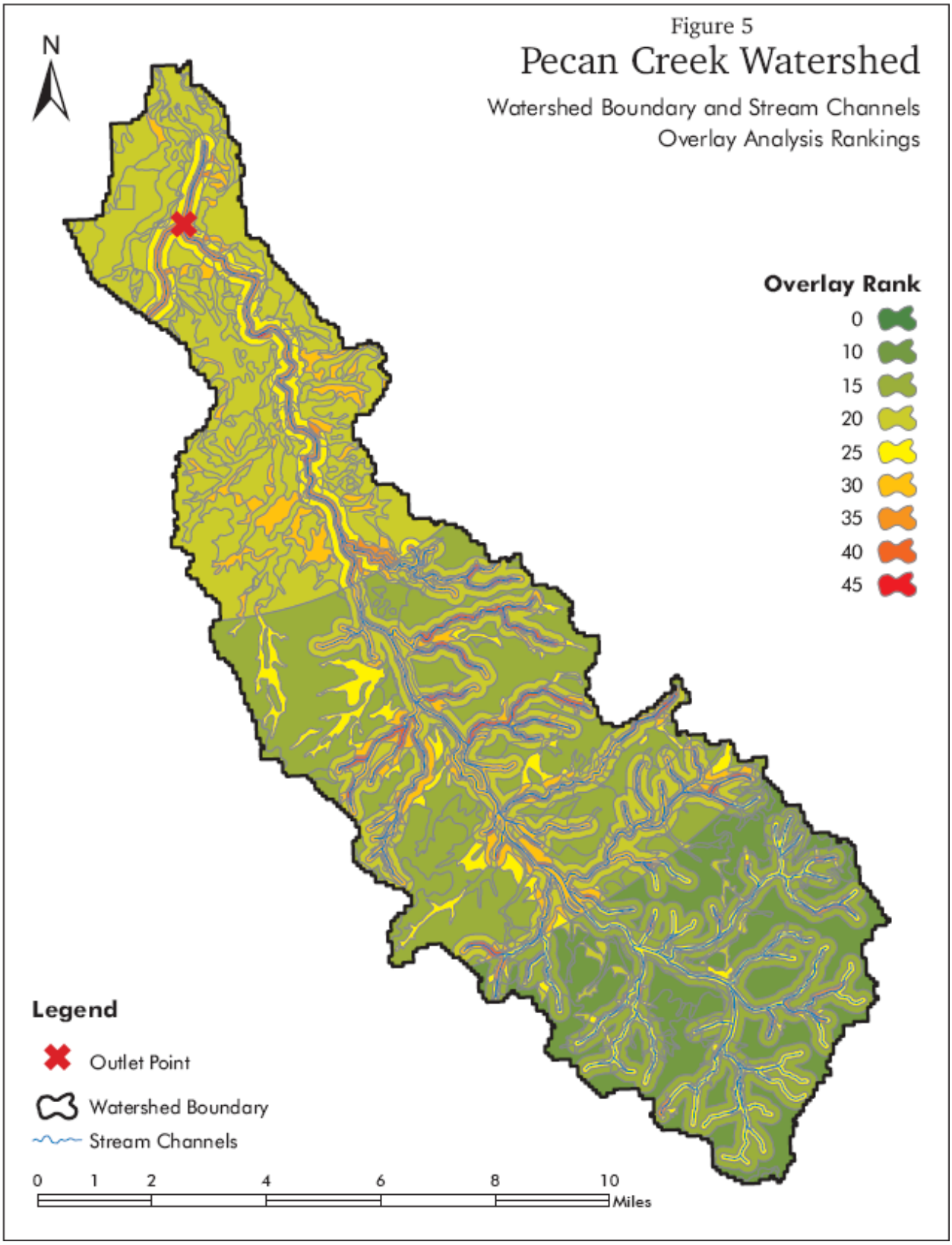
Figure 5
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Overlay Analysis Rankings
N
Overlay Rank
0
10
15
20
25
30
35
40
45
Legend
Outlet Point
Watershed Boundary
Stream Channels
0 1 2 4 6 8 10
Miles

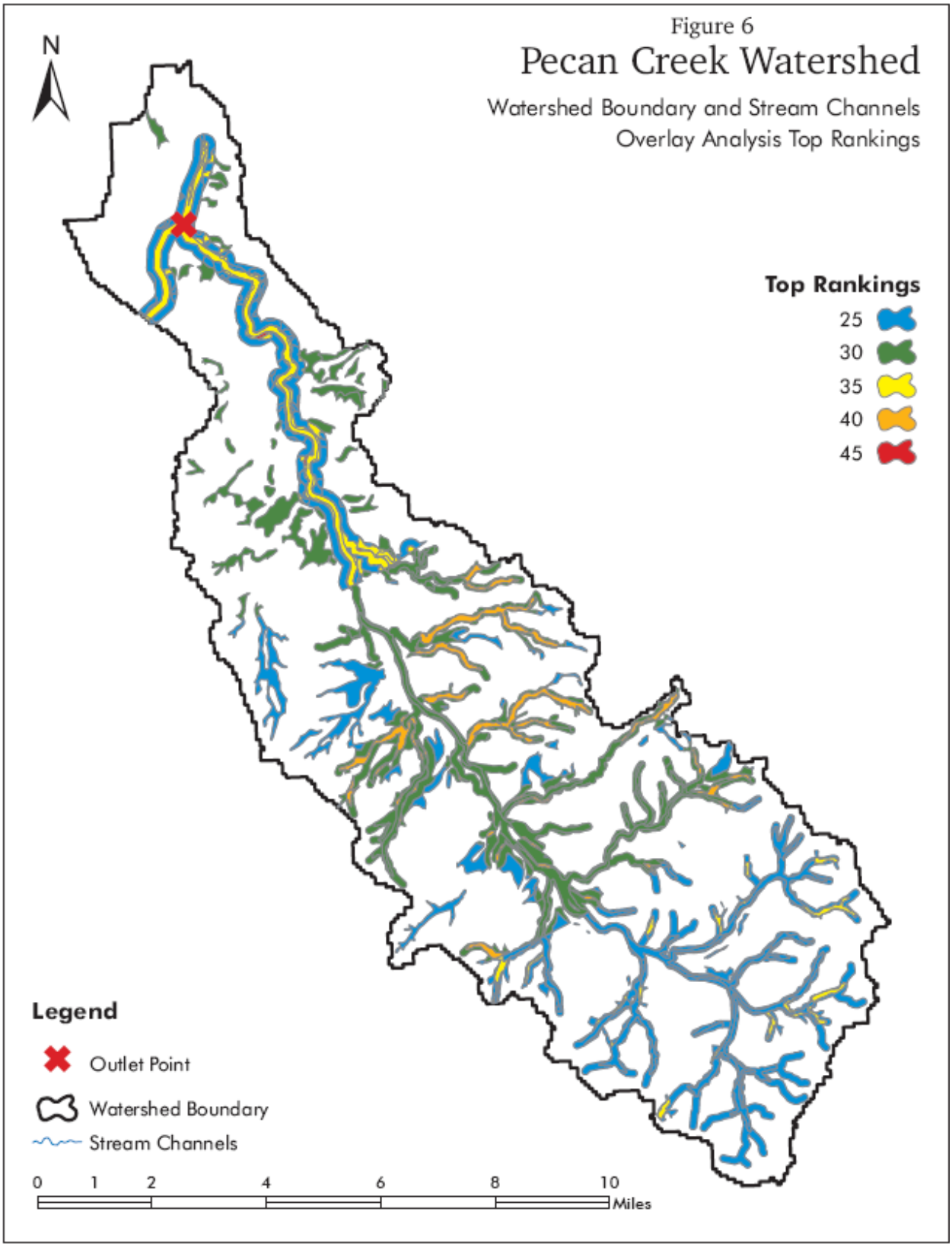
Figure 6
Pecan Creek Watershed
Watershed Boundary and Stream Channels
Overlay Analysis Top Rankings
N
Top Rankings
25
30
35
40
45
Legend
Outlet Point
Watershed Boundary
Stream Channels
0 1 2 4 6 8 10
Miles

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.

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Effects of Brush Management on Water Resources

By:

C. Allan Jones and Lucas Gregory, Texas Water Resources Institute, Texas A&M AgriLife

Texas Water Resources Institute Technical Report
November 2008

Texas Water Resources Institute
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Effects of Brush Management on Water Resources

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Summary

For several decades, land managers have cleared brush species, such as mesquite and juniper (cedar), and observed increases in spring and streamflows. Scientists have also conducted numerous studies in which they have measured the effects of brush removal on different aspects of rangeland hydrology. These include the amount of rainfall that is intercepted and held by the plant leaves, surface runoff, spring flow, water use by individual plants and plant communities, fluctuation of shallow water tables, and streamflows. Considering this very diverse information, many scientists agree on several points:

1. The roots of some brush species extract water from greater depths than do grasses and forbs, and brush control can reduce the total amount of water used by vegetation.
2. Brush and other deep-rooted vegetation growing over shallow aquifers near streams can be expected to use large amounts of groundwater, likely reducing the amount in both the interconnected stream and aquifer.
3. Removal of brush like juniper and live oak from upland areas some distance from streams may increase streamflow and/or recharge aquifers especially when:
    a. The brush canopy is dense and intercepts substantial amounts of rainfall (for example: dense juniper [cedar] or live oak stands), effectively reducing the amount of rainfall reaching the soil surface, and
    b. Soils, subsoils and/or geologic strata are permeable, and streams in the area are fed by seeps and springs. Water can quickly percolate below the roots of grasses and forbs and move through subsurface pathways to local streams or aquifers.
4. Brush control in upland areas is unlikely to increase significantly water yields if soils and geologic formations are not conducive to increased runoff and/or subsurface flows to streams or to aquifers.
5. For brush control to have substantial long-term impacts on water yield, most or all of the woody vegetation in the treated area should be killed, and regrowth of brush

and herbaceous vegetation should be controlled so that it is less dense and more shallow rooted than the pretreatment vegetation.

6. New science-based tools can help pinpoint locations where brush control should substantially increase water flows in streams.
7. A geographically targeted brush control program with careful scientific verification of impacts is needed to guide long-term brush control policies.

Introduction

For many years, brush management has been an important tool in maintaining livestock and wildlife production on rangelands. It has long been recognized that water used by brush is not available to the grasses and forbs, and clearing brush typically stimulates grass and forb growth.

Historically, many brush species have served as a vital part of rangeland habitats in Texas and the southwestern United States. The range and coverage of woody plant species has increased in recent times. Van Auken (2000) describes this conversion of grasslands or native rangelands and savannas to woodlands as woody plant encroachment. He attributes much of the increased density of woody plants to a combination of changing climates, overgrazing, and fire suppression. Overgrazing and fire suppression can easily be linked to the expansion of western settlement (Blackburn, 1983; Archer, 1994; Dugas et al. 1998) and have served as the primary catalyst in the increases in upland woody species, such as Ashe juniper and mesquite. Other species have been introduced to the state and have adapted quite well. Salt cedar was introduced to the United States in the early 1800s as an ornamental and in the early 1900s, it was used as a means for streambank stabilization (Everitt, 1998). Since then, salt cedar has taken over large areas of riparian habitats in the western United States. It has drastically altered the vegetative composition of these plant communities and the hydrology of these areas (Hart et al., 2005). Ultimately, brush has and always will play a vital role in Texas landscapes, but human influences have greatly altered the balance between brush and herbaceous plant communities and rangeland hydrology. This realization took hold in the early 1960s and brush clearing efforts began shortly thereafter in an attempt to correct these ecologic and hydrologic modifications.

In the 1960s, the USDA Soil Conservation Service (SCS) cleared brush, primarily mesquite, from the Rocky Creek watershed near San Angelo and the creek began flowing again for the first time in many years. This was sufficient evidence for SCS and the ranchers. They then began clearing brush, not only to increase forage for their livestock, but also to restore water in West Texas streams (Kelton, 1975).

Over the years following the work in the Rocky Creek watershed, many more scientific studies have demonstrated the impacts that brush has on the various components of rangeland hydrology, including the amount of rainfall intercepted by different species of brush, transpiration of water from plants and from plant communities (evapotranspiration), runoff from the soil surface, infiltration into the soil, and subsurface movement of water to streams via seeps and springs. From this research, scientists have concluded that under certain conditions brush control can substantially increase the amount of water reaching streams and aquifers. Under other conditions, brush control can have little or no effect (Bosch and Hewlett, 1982; Hibbert, 1983; Huxman et al., 2005; Rainwater et al., 2008; Thurow, 1990; Thurow et al., 2000; Wilcox, 2002; Wilcox et al., 2006b). For example, Wilcox et al. (2006b) reviewed available literature and concluded that brush control is most likely to increase water yield in three key areas: 1) riparian areas with accessible groundwater and dominated by invasive riparian species such as salt cedar, 2) upland landscapes with woody species such as juniper and oak on soils that allow rapid deep drainage such as shallow or highly permeable soils over fragmented karst limestones like those in the Edwards Plateau and 3) mesquite growing on deep sandy soils like those in the Carrizo-Wilcox Aquifer recharge zone. They also concluded that areas where brush control is least likely to increase water yields significantly are those with deep soils, deep or absent groundwater, and where no subsurface flow to springs occurs.

Sterling Creek in the North Concho River watershed, once dried, is a perennial-flowing creek after brush control (May 2005).

The purpose of this white paper is to present the scientific literature on this important issue in a form that will be useful to decision makers considering public policies that encourage brush control in Texas for water yield enhancement. For this paper, a working definition of "brush" is "unwanted woody vegetation on range lands, including but not limited to juniper (cedar), mesquite, salt cedar, and oak."

Because the benefits of well-designed brush control programs on rangeland productivity, livestock production, and rural economic activity are widely understood, the focus here will be only on the effects of this conservation practice on the amount of water in our streams and aquifers. To facilitate communication and comparisons among studies, in most cases water savings resulting from brush control are expressed in English rather than metric units, and spring flows, streamflows, aquifer recharge, etc. have been converted to units of inches or feet. For example, if brush control is reported in a particular experiment to have increased streamflow by 2.0 inches per year, it means that the annual streamflow increased by an amount equivalent to 2.0 acre-inches per acre of the watershed (usually per acre of the watershed treated). One acre-inch is equal to 27,154 gallons, and one acre-foot equals 325,851 gallons.

What Do We Know?

In the following sections, we attempt to describe clearly the current state of scientific knowledge about the effects of brush control on rangelands, with emphasis on Texas and the southwestern United States.

The roots of some brush species extract water from greater depths than do grasses and forbs, and brush control can reduce the total amount of water used by vegetation.

Ranchers have long observed that, during dry periods, woody species such as mesquite, juniper, and live oak stay green after most grasses and forbs mature and turn brown. In addition, the roots of brush species are often observed far deeper in the soil (as seen in road cuts and deep gullies) than grass roots. Based on these observations, it has been widely assumed that brush can use water from deeper in the soil than can grasses and forbs.

This assumption was confirmed in work conducted by Jackson et al. (1999) that found roots of woody plants in 14 of the 19 caves they studied in the Edwards Plateau region. By analyzing root DNA, they determined that the roots of six species penetrated at least 17 feet. Roots of Ashe juniper and live oak were found as deep as 27 feet and 73 feet, respectively.

Richardson et al. (1979) studied runoff and calculated evapotranspiration from soil moisture in two mesquite-infested watersheds on deep clay soils of the Blackland Prairie near Riesel. The watersheds were monitored for two years prior to chemical control of the mesquite and for three years after treatment. In this experiment, mesquite control reduced evapotranspiration by 3.1 inches per year. This reduction was the result of decreased use of soil water between 1 and 5 feet below the surface, especially late in the growing season. These results, like those of Jackson et al. (1999), suggest that brush species often use water from deeper in the soil than grasses and forbs.

Scott et al. (2000) measured ET from two plant communities, a perennial grassland and a mesquite-dominated shrubland, on a river floodplain in Arizona. They found that the mesquite shrubland extracted water from deeper in the soil profile than the grassland. Over the course of a year, the grassland used approximately 10.8 inches of soil water, and the mesquite shrubland used approximately 14.8 inches. But there was little indication that the shrubland used a significant amount of water from the shallow aquifer, which was over 30 feet deep at the site.

Saleh et al. (2008) measured evapotranspiration on two 200-acre watersheds within the North Concho watershed near San Angelo. Brush, primarily mesquite, was removed from one of the watersheds and left undisturbed on the other. The watershed on which all mesquite had been removed averaged 11 percent less evapotranspiration over two growing seasons (May-October). The difference was slightly greater (14 percent) during the warmer June-September period, and the difference was greater in a dry June-September period (4.8 inches, 17 percent) than a wetter period (11.8 inches, 12 percent). These results, while still awaiting publication, are consistent with those of Richardson et al. (1979), Dugas and Mayeux (1991), and Scott et al. (2000), who concluded that mesquite removal decreased evapotranspiration in the Blackland Prairie, Rolling Plains, and Arizona, respectively. In these studies, mesquite roots were presumably able to reach and extract water below the rooting depth of grasses and forbs.

Of course, if brush is allowed to re-infest the treated area or other deep-rooted replacement vegetation is allowed to grow so that its leaf area is comparable to the brush prior to treatment, potential water savings from brush control will likely be negated.

Brush and other deep-rooted vegetation growing over shallow aquifers near streams can be expected to use large amounts of ground water, likely reducing the amount in both the interconnected stream and aquifer.

Streams often change course over geologic time, depositing layers of sands and gravels in intricate patterns. The water in these deposits can remain in hydrologic contact with the

water in the stream, creating shallow "riparian aquifers." Water can enter these shallow riparian aquifers in four ways: from flooding of the bottomlands along the stream, from rainfall that percolates downward from soils directly over the aquifer, by subsurface flow from nearby uplands, or by flow directly from the stream when the water level in the stream is higher than that in the aquifer. When the water level in the aquifer is higher than the water in the stream, water returns from the aquifer through springs to the stream. Woody vegetation growing near streams can often use large amounts of water from riparian aquifers. This situation has been clearly demonstrated in New Mexico, where Cleverly et al. (2002) reported that a salt cedar-dominated site that flooded twice a year used 48 inches of water over a 157-day growing season while one that did not flood used 29 inches. This is well within the range of daily salt cedar water use in studies reviewed by Cleverly et al. (2002). Several of these studies measured maximum water use by salt cedar stands in hot and dry areas of over 0.37 inches per day. Nagler et al. (2008) obtained similar results along the Lower Colorado River in the Cibola National Wildlife Refuge. There, salt cedar growing on river terraces where the water table was 10 to 13 feet deep used from 43 to 55 inches of water per year; however, this water use equates to only 1 to 2 percent of annual river flow (McGinly 2008, Nagler et.al,2008).

Salt cedar being sprayed on the Pecos River

Owens and Moore (2007) found similar water use by a young and dense salt cedar stand near the Rio Grande. They used sap flow measurements to estimate a maximum salt cedar transpiration rate of 0.23 inches per day and a total of 41.4 inches over a 180-day growing season. A mature, less dense stand very close to the Pecos River used much less water, a maximum of only 0.01 inches per day, possibly because the trees were growing in very salty soils (G. W. Moore, personal communication).

Hart et al. (2005) measured the daily rise and fall of the shallow riparian aquifer along the Pecos River in Texas before and after and with and without herbicidal control of salt cedar. The daily variation in the water table under salt cedar stands clearly demonstrated that during the daylight hours the plants were using substantial water from the shallow aquifer associated with the stream. At night when plant water use decreased, the water table level recovered. The study demonstrated that water use ceased when the salt cedar was killed with herbicide. Over a three-year period, annual water use at the untreated site varied from 68 inches to 80 inches. For the site where salt cedar was killed, the annual water use was 116 inches the year prior to treatment and declined to only 7 inches the year after.

Although salt cedar is the most well known example of unwelcome riparian vegetation that uses water from Texas streams, mesquite, juniper, giant cane, and other species often invade abandoned croplands and overgrazed rangelands near streams. For example, Unland et al. (1998) measured water use by several vegetation types in a riparian corridor between one-half and one mile wide along the Santa Cruz River in southern Arizona. During a one-year period with a total of 28.1 inches of rainfall, the vegetation, consisting of willow-dominated and mesquite-dominated plant communities (about 2 acres of mesquite for each acre of willow), used a total of 44.7 inches of water. Much of this water use was without doubt from the shallow aquifer associated with the river. Similarly, work conducted by Nagler et al. (2008) found that salt cedar growth in the Cibola National Wildlife Refuge near the Lower Colorado River used an annual average of 43.2 inches of water in an environment where the annual average rainfall is only 3.1 inches. In contrast, short low-density vegetation growing on abandoned agricultural fields nearby used only 6.2 inches of water.

Tromble (1972) also measured the daily rise and fall of a shallow riparian aquifer with a typical depth of 10 to 13 feet below a mesquite woodland in southeastern Arizona. From these data, he estimated water use by the vegetation of up to 0.42 inches per day. Similar values of mesquite and salt cedar evapotranspiration from shallow aquifers have been reported by Gatewood et al. (1950) and Qashu and Evans (1967).

Moore et al. (2008) found significant amounts of salt cedar transpiration (up to 36 percent of total daily transpiration) at night. This nocturnal water use would tend to reduce diurnal variation in water table levels and cause underestimation of total water use; essentially, aquifer levels do not rise as much as they would if no transpiration occurred at night. This suggests that other studies of salt cedar water use that measured diurnal variation in the shallow water table may have underestimated total water use.

Scott et al. (2003) measured water use by both mesquite trees and the grasses and forbs growing in a mesquite woodland within about 500 yards of the Upper San Pedro River in southeastern Arizona. In this location, the mesquite tree roots were observed approximately 33 feet below the soil surface, the approximate depth of the water table. Evapotranspiration measurements of the mesquite canopy and

Giant Cane along the Arroyo Colorado in the Lower Rio Grande Valley

understory plants, as well as measurements of daily fluctuations of the water table, led the authors to conclude that the mesquite trees obtained most of their water from a deep groundwater source, whereas the understory vegetation primarily used recent precipitation stored near the soil surface. In various periods from June through September, water use by the mesquite alone ranged from about 0.04 to 0.12 inches per day.

Giant cane is another exotic invasive species that has colonized many riparian areas across Texas. Though few direct measurements of water use by giant cane have been made in Texas, studies in California found between 3.8 and 4.4 feet of water use per year. These values are within the range of water use measured in salt cedar (Bell, 1997; Jackson et al., 2002). With 60,000 acres of giant cane in the riparian areas of the Rio Grande alone, replacement of this species with vegetation that uses less water could increase available water significantly.

Several of the studies and reviews cited above suggest that salt cedar, mesquite, giant cane, and other woody species growing near streams over associated shallow aquifers can use up to 4 acre-feet of water for each acre of vegetation. Much of this water can be saved or "salvaged" by killing the woody vegetation and replacing it with low-density, shallow-rooted grasses and forbs. These water savings may be reduced or lost if the original vegetation or other woody species like willows are allowed to grow again on the area previously treated (W. Hatler and C. Hart, personal communication). However, even if the long-term effect of salt cedar control is to save only 1 acre-foot of water for each acre treated, the impacts on water flow can be significant. For example, if clearing 13,500 acres of salt cedar on the Pecos River from 1999 through 2005 saves only 13,500 acre-feet of water per year, the increased flow of the river and/or storage in the riparian aquifer is equivalent to 59 percent of the average flow of the river at Girvin (23,000 acre-feet per year) (Hart, 2005; Miyamoto et al., 2005).

Removal of brush like juniper and live oak from upland areas may increase streamflow and/or recharge aquifers, especially when:

a. ***The brush canopy is dense and intercepts substantial amounts of rainfall (for***

example, dense juniper [cedar] or live oak stands), effectively reducing the amount of rainfall reaching the soil surface, and

b. Soils, subsoils and/or geologic strata are permeable, and streams in the area are fed by seeps and springs. Water can quickly percolate below the roots of grasses and forbs, and subsurface pathways can conduct water from the uplands to local streams or aquifers.

Water that is intercepted by and remains in the leaf canopy after a rain event evaporates without ever reaching the soil surface. Water that passes through the leaf canopy and reaches the ground either runs off or percolates into the soil. Once in the soil, the water can be stored in the root zone until it evaporates from the soil surface, it is transpired by the vegetation, or it percolates below the root zone, where it either recharges an aquifer or reaches nearby streams via seeps and springs.

Of course, rainfall intercepted by the leaf canopy has little or no effect when rainfall is not enough to produce runoff or deep percolation below the root zone. However, when rainfall is sufficient, additional water that reaches the soil surface when no brush is present can substantially increase runoff and/or deep percolation. For example, Thurow et al. (1987) estimated that near Sonora live oak mottes intercepted 46 percent of the annual precipitation, compared with 18 percent interception by sideoats grama and 11 percent by curlymesquite grass.

Similarly, Thurow and Hester (1997) summarized studies on the Edwards Plateau near Sonora. They concluded that clearing brush (36 percent juniper and 24 percent oak) increased the amount of rainfall reaching the soil by 6.6 inches per year and increased deep drainage by 3.7 inches per year during a two-year period with 22.6 inches of annual rainfall. In these studies runoff was minimal (0.2 inches per year) due to high soil infiltration rates. This work was reviewed by Thurow et al. (2000) along with other works (Redecker et al. (1998), Carlson et al. (1990), Carlson and Thurow (1996)) and they concluded that water yield from grass rangelands on the Edwards Plateau near Sonora, Texas exceeds those of rangelands with brush cover of 15 percent or greater.

In Texas, controlling juniper (thereby reducing interception) can increase both surface runoff and spring flow. For example, Huang et al. (2006) measured spring flow and total flow at the base of a forty seven-acre watershed on the Edwards Plateau in Comal County for two years prior to juniper control and two years after control. Over the course of the four years, total streamflow (consisting of both storm flow and baseflow) varied from 4 percent of rainfall in the driest year to 34 percent in the wettest year, and averaged 22 percent of rainfall overall. Baseflow (spring and seep flows) contributed approximately half the total. On a per-rainfall event basis, average runoff was 0.22 inches before juniper removal and 0.35 inches afterward, suggesting that juniper removal resulted in a 1.8 inch (60 percent) annual increase in total streamflow.

These results are consistent with the results of two reports from the Seco Creek watershed in Medina County. Dugas et al. (1998) found that for the first two years after juniper was cleared, total evapotranspiration was about 4.3 inches per year less on the cleared site than on the uncleared site. This finding illustrates how reductions in interception and vegetative water use can alter the hydrology of a treated site. In the third year after brush clearing, Dugas et al. (1998) found that both sites produced approximately equal amounts of evapotranspiration, probably due to much greater growth of perennial grasses as well as compensatory growth of other woody plants on the cleared site. The authors suggest that the beneficial effects of juniper control might have continued if compensatory vegetation growth had been suppressed by grazing and follow-up brush control. Similarly, Wright (1996) reported increases in spring flow of approximately 1.6 inches per year as a result of juniper control in the watershed.

Grubber removing mesquite

A number of studies conducted outside Texas are consistent with those cited above. For example, Bosch and Hewlett (1982) reviewed ninety-four watershed experiments from around the world to determine the effects of vegetation changes on water yield. Virtually all the experiments found that reducing vegetation cover (and interception) increases water yield from watersheds. On average, for pine and eucalypt forests water yield increased by an annual average of 1.6 inches for each 10 percent reduction in canopy cover. For deciduous hardwood and "scrub" vegetation, the corresponding increases in water yield were 1.0 inches and 0.4 inches, respectively. Experiments in Arizona, California, and Utah were particularly relevant to the situation in western Texas. In eight experiments, vegetation (oak woodland, chaparral, or juniper-pinyon) was cleared in areas with average rainfall between approximately 18 inches and 27 inches. The average annual water yield increase over several years following clearing varied from "non-significant" to about 5.2 inches with larger increases in the experiments with greater rainfall. The average streamflow for these eight experiments increased by approximately 2.0 inches, more that doubling the mean annual streamflow of 1.8 inches for these sites prior to clearing the vegetation.

In view of the studies summarized above, it seems likely that clearing dense juniper and live oak brush in the Edwards Plateau or similar areas can produce 1 to 4 acre-inches of additional water per year for each acre of brush cleared. Of course, as discussed in the previous section, brush control must be maintained and excessive compensatory growth of herbaceous vegetation must be controlled by grazing or other means in order to sustain these increased water yields.

Brush control in upland areas is unlikely to significantly increase water yields if soils and geologic formations are not conducive to increased runoff and/or subsurface flows to streams or to aquifers.

Brush control in upland areas receiving very little annual rainfall is unlikely to reliably increase water yields because runoff and/or subsurface flows are seldom large. Wilcox et al. (2006b), Ball and Taylor (2003), and Bosch and Hewett (1982) concluded that control

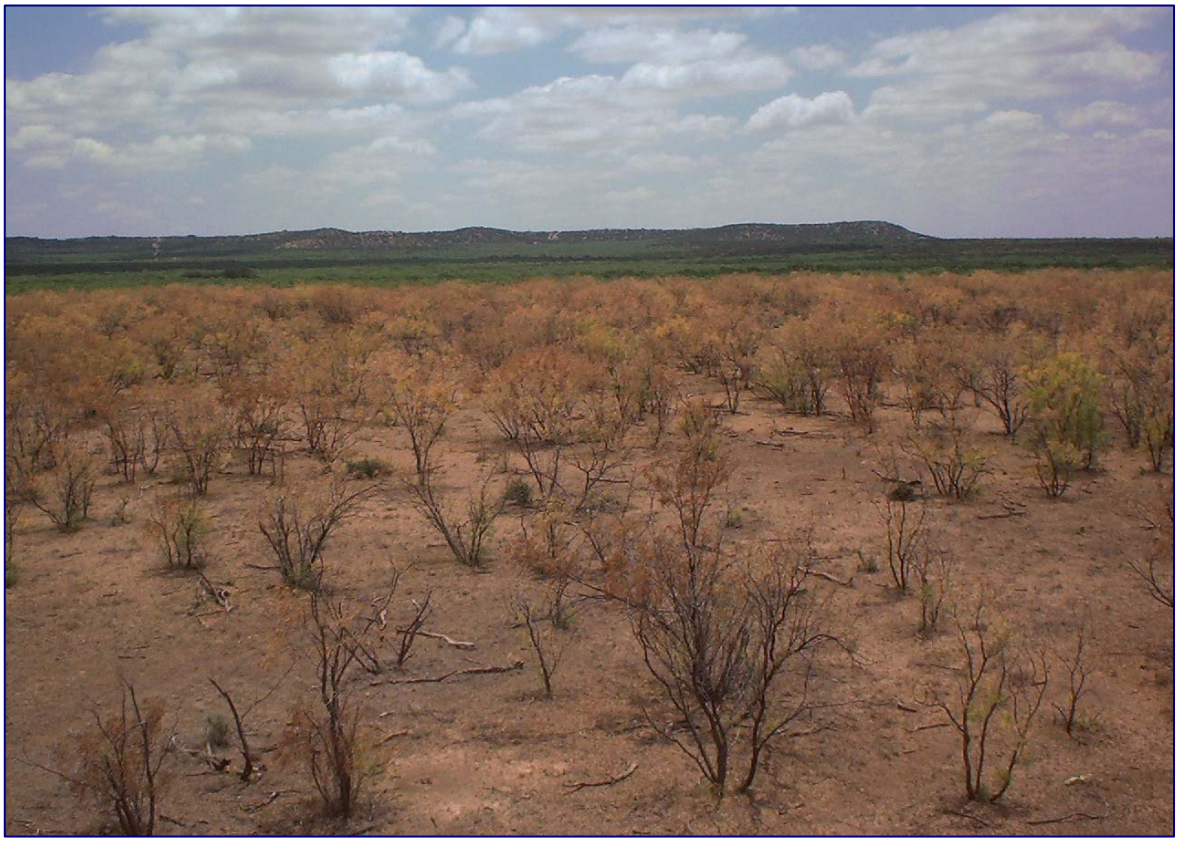
of brush on areas receiving less than 18 inches of rainfall annually is not likely to increase water yields. Similarly, brush control on sites with deep permeable soils and no local springs or shallow aquifers is unlikely to generate significant increases in streamflows or aquifer recharge because these soils and geologic formations are not conducive to increased runoff and/or lateral subsurface flows to streams or to aquifers (Wilcox et al. 2006b).

For example, Carlson et al. (1990) measured evapotranspiration, deep drainage, soil storage, and runoff from lysimeters with and without mesquite on the Rolling Plains near Throckmorton. Over a three-year period, they found little effect of mesquite removal on deep drainage or surface runoff. This response seems to be the case in the Rolling Plains of North Texas where soils are permeable and deep, runoff rarely occurs, and mesquite dominates the woody vegetation. Though brush control in these cases may benefit the rancher by stimulating compensatory forage growth, it does not necessarily increase streamflows or aquifer recharge.

For brush control to have substantial long-term impacts on water yield, most or all of the woody vegetation in the treated area should be killed, and regrowth of brush and herbaceous vegetation should be controlled.

As mentioned above, any effort to control woody vegetation to increase water availability must also consider long-term maintenance of that control. For example, if only the juniper is removed from an oak-juniper woodland, the oaks and other species will often respond to fill in spaces formerly occupied by the juniper, reducing or even eliminating the benefits of the juniper control. The same effects may be seen if brush species are allowed to regrow or if grasses and forbs regrow to much greater biomass and leaf area than they had before the brush was controlled. Their greater leaf development can intercept more rainfall, slow runoff, and transpire more water to offset the otherwise beneficial hydrologic effects of brush control.

For example, Dugas and Mayeux (1991) found that chemical control of mesquite in mid-summer immediately decreased evapotranspiration by up to 40 percent, compared with the



Dead mesquite after brush control

untreated plot. However, the plots were not grazed, and during the subsequent summer, grass and forb growth increased dramatically in the treated plot, and evapotranspiration was only reduced by 7 percent compared with the control. Brush removal in this situation may simply allow more water to be stored in the soil for use by the remaining grasses and forbs. Similar effects were observed by Dugas et al. (1998) in their study of juniper control in the Seco Creek watershed on the Edwards Plateau (see discussion above).

Moore and Owens (2006) also emphasized the importance of completely clearing juniper within an area and maintaining control after initial clearing. Otherwise, juvenile trees remaining after larger trees are cleared will compensate with rapid growth and water use, significantly decreasing the positive impacts of brush clearing on forage production and water yield.

New science-based tools can help pinpoint locations where brush control should substantially increase water flows in streams.

Over the last decade, researchers have developed and used simulation models and decision tools to pinpoint locations where brush control is most likely to increase water yields to streams and/or aquifers.

Redecker et al. (1998) used SPUR (Simulation of Production and Utilization of Rangelands-91) model to simulate the effects of broad-scale brush control on the Cusenbary Draw watershed (80 square miles) on the Edwards Plateau. Based on surveys of ranchers in the watershed, they concluded that landowners would enroll 40 percent of their land in a brush control program, and the results would be a two-fold increase in water yield from the rangelands on which brush was controlled (from 0.80 to 1.60 inches per acre per year). This study did not identify target areas where brush control would yield the most water.

Arnold et al. (2008) used the SWAT (Soil and Water Assessment Tool) model to estimate the effects of brush control on streamflows for eight Texas river basins composed of numerous sub-basins. The studies were conducted to provide guidance for the brush control cost-share program conducted by the Texas State Soil and Water Conservation Board and are reported in detail in Texas Agricultural Experiment Station (2000). Results suggested that little increase in streamflow (baseflow plus surface runoff per unit treated area) can be expected where mean annual rainfall is less than about 18 inches, but substantial streamflow increases (2 and 4 inches per year) could be expected with mean annual rainfall of 24 and 30 inches, respectively.

Afinowicz et al. (2005) also used SWAT to estimate the effects of brush control on water yields in the Edwards Plateau. Considering the area where brush was removed over a ten-year period, this simulated removal of brush reduced evapotranspiration by an average of 1.8 inches per year, increased runoff and lateral flow to the stream by 0.4 inches per year,

increased baseflow to streams by 0.1 inches per year, and increased deep aquifer recharge by 1.2 inches per year.

The hydrologic estimates of these modeling studies are generally consistent with experimental data reported by Bosch and Hewlett (1982), Huang et al. (2006), Dugas et al. (1998), Thurow and Taylor (1995), and Owens and Knight (1992).

The Spatial Sciences Laboratory at Texas A&M University has developed a completely independent method to identify areas of Texas with substantial baseflows, which are indicative of geology and soils that allow precipitation to percolate into shallow aquifers and return to nearby streams via springs (R. Srinivasan, personal communication). This method is based on long-term streamflow data from a large number of U. S. Geological Survey stream gaging stations (Figure 1). Daily streamflow data were analyzed mathematically to distinguish between overland and baseflow components for each gage. The ratios of baseflow to total streamflow for all points were interpolated to generate the map showing the fraction of total streamflow that is baseflow. In general terms, areas with the highest percentage of annual baseflow correspond quite well with major portions of the Edwards Plateau, the Cross Timbers and Prairies, and the East Texas Pineywoods. These are areas where rainfall is sufficient and soils and geology are permeable enough for precipitation to penetrate the soil and move laterally to nearby streams. The green, and perhaps yellow, areas appear to offer the greatest opportunity for upland brush control to produce significant increases in water yield.

Another approach uses rules based on consensus expert opinion to target hydrologically sensitive areas within a watershed (R. Srinivasan, personal communication). An expert system based on a currently available, multi-layered geographic information system has been developed to identify areas within a watershed with the greatest potential for increasing water yield through vegetation management (increased streamflow and/or groundwater recharge). The tool incorporates rules related to: 1) the presence or absence of brush, 2) topography of the watershed, 3) proximity of the brush to a stream or drainage path and 4) the likelihood of precipitation becoming aquifer recharge.

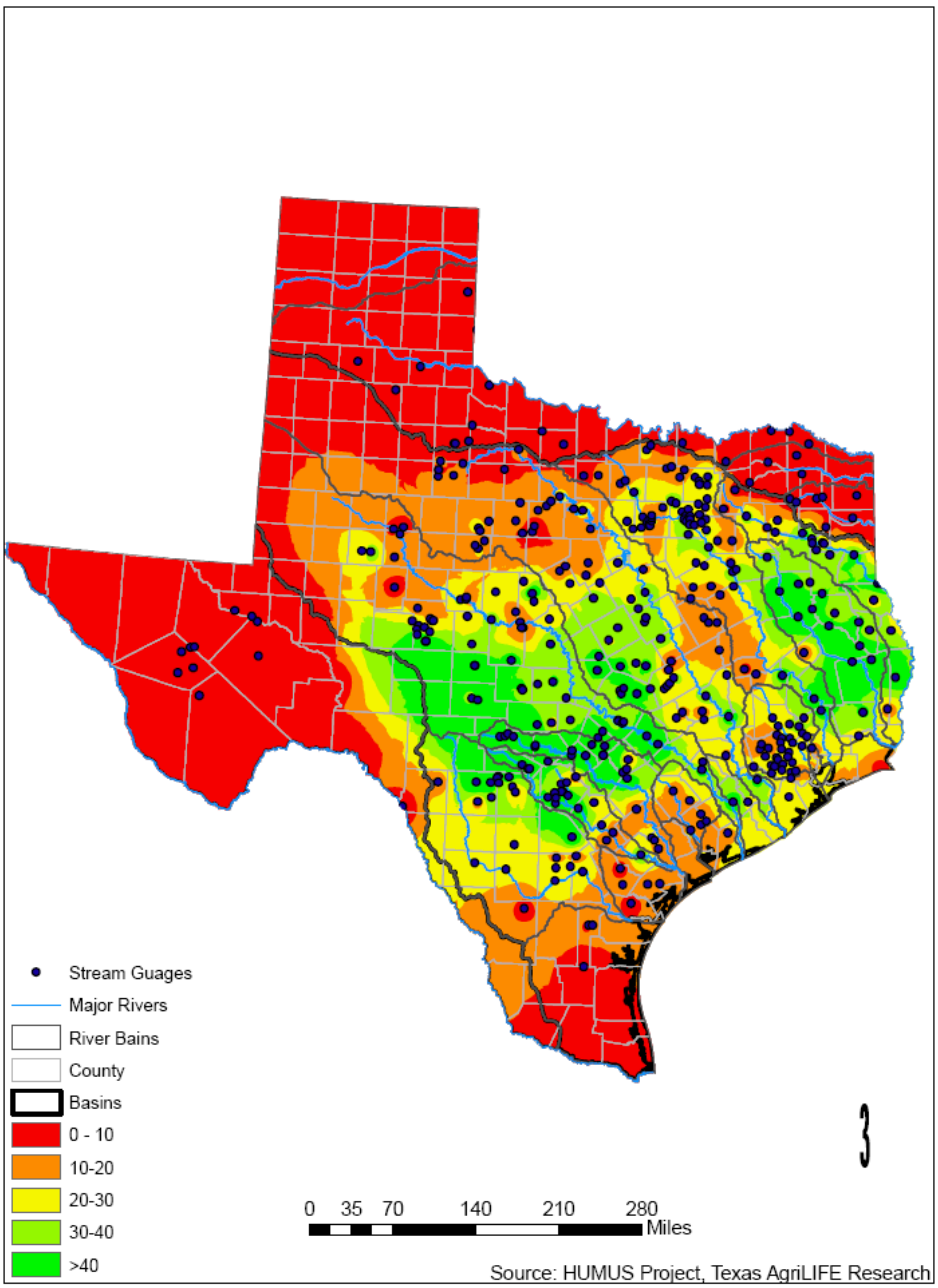


Figure 1. Annual average baseflow as a percentage of total baseflow.

Rainwater et al. (2008) developed a similar approach to select the most appropriate sites for brush control to increase water yields. Evaluation of watersheds should consider: 1) characteristics of the watershed (soils, slope, land use, vegetation and brush distributions,

and proximity of brush the stream), 2) local climatic conditions, and 3) interaction of surface water and groundwater in the area.

Of course, field observations and expert knowledge of the area should also be used in the decision making process. For example, it might be unwise to implement brush control in upland areas that contribute to highly saline springs. Increasing spring flows might simply increase salt loads to downstream rivers and reservoirs. Likewise, land fragmentation or land owner attitudes toward wildlife might reduce the feasibility of implementing large-scale brush control programs in some areas.

These tools are now available to decision makers who wish to estimate the benefits, costs, and most appropriate locations to implement brush management programs in different locations across Texas or within a specific watershed.

A geographically targeted brush control program with careful scientific verification of impacts is needed to guide long-term brush control policies.

Because of large year-to-year variation in rainfall and spatial variation in geology, soils, brush cover, and land ownership, obtaining definitive measurements of the effects of brush control on streamflows and aquifer recharge is

Mechanical-removed brush near San Angelo

very challenging. A comprehensive and definitive watershed-scale hydrologic study measuring all relevant components of the rangeland water balance has never been done in Texas. However, in view of the large amounts of water in question and the expectation of continued brush encroachment, such studies have been strongly recommended (Rainwater et al., 2008; Wilcox et al., 2005).

Our scientific understanding of the benefits and costs of brush control have increased dramatically since the Rocky Creek experience in the 1960s and 1970s or the beginnings of the Texas State Soil and Water Conservation Board brush control efforts a decade ago. The methods recently developed by Rainwater et al. (2008) and Srinivasan (personal communication), combined with the economic and hydrologic results of earlier modeling studies conducted for the Texas State Soil and Water Conservation Board brush control program (Texas Agricultural Experiment Station, 2000) should be used to guide a pilot program in carefully selected areas to demonstrate the efficacy of brush control to enhance water yield. The program should be implemented in areas where brush control is most likely to increase water yields, perhaps on juniper-infested upland areas of the Edwards Plateau and riparian bottomlands covered with salt cedar, mesquite, giant cane, and other undesirable species elsewhere in the state. The program should include rigorous scientific verification of the impacts of brush control on runoff, aquifer recharge, spring flows, in-streamflows, rangeland productivity, wildlife habitat, fisheries, the potential to harvest brush for bioenergy production, and landowner attitudes. This scientific information would provide invaluable guidance for future decision makers interested in expanding brush management programs for both water yields and other benefits they produce.

Concluding Remarks

Reaching scientific consensus about the effects of brush management on rangeland water yield has been challenging, complicated by issues of measurement methods, temporal variation of precipitation, and spatial variation in landscape, climate, vegetation, soils, and geology.

In addition to these scientific and technical issues, scientists, landowners, policymakers, and other stakeholders maintain a host of beliefs, both positive and negative, about brush control's effects on water availability. Furthermore, brush management for grazing land and livestock productivity, ecosystem health, wildlife habitat, endangered species habitat, environmental flows, water supplies, carbon sequestration and the landscape's scenic value are also quite important and should be taken into consideration prior to brush control implementation. Perceptions regarding the long-term impacts of climate change, grazing, and fire on vegetation further complicate discussions of brush management.

A few authors have grappled with these complex issues and explored the relationships between multiple factors. For example, Conner et al. (2001) found that the shift from grasslands to woodlands has contributed to an overall decrease in the total amount of grassland habitat and the loss of ecosystem functions while Teague et al, (2008) concluded that a system of brush control using prescribed fire, rotational grazing and grazing deferment can maximize ecosystem health and function simultaneously with land manager profits. Olenick et al. (2004) were able to quantify the monetary benefits of brush removal and found that the public cost of producing additional water ranged from $32 per acre-foot to $159 per acre-foot depending on the location within the Edwards Plateau. These examples only provide a snapshot of brush management's economic benefit. More extensive evaluations will undoubtedly be able to link the economic impacts of ecosystem services to tax revenues and economic activity.

Ultimately, landowners will manage their property as they desire; however, efforts should be made to convey the importance of achieving and maintaining a healthy balance of brush

and grasslands to the landowner and society. While removing all brush would likely have more profound impacts on the hydrologic cycle, it would be detrimental to many species that depend on these landscapes for critical habitat. Selectively clearing brush using a set of predetermined criteria such as those set forth by Rainwater et al. (2008) and Srinivasan (personal communication) will likely have the most profound and positive impacts on ecosystem health, rangeland condition, and water salvage while maintaining the ecological integrity of the landscape. It must be stressed that proper management and maintenance of these lands after brush control has been carried out is the most important factor in maintaining the long-term balance, function, and health of the landscape.

For the foreseeable future, Texas landowners will manage their properties with multiple economic, aesthetic, and environmental objectives. Scientists and policy makers should strive to provide these landowners with scientifically based information, education, decision tools, policies and programs to achieve both private and public benefits from Texas' private lands. Brush control programs are a very important means of managing the state's private grazing lands to achieve both private and public goals.

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