USING EXISTING AGRICULTURAL INFRASTRUCTURE FOR RESTORATION PRACTICES: FACTORS INFLUENCING SUCCESSFUL ESTABLISHMENT OF COTTONWOOD OVER TAMARISK

By

Daniel P. Bunting
School of Natural Resources and the Environment
The University of Arizona

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And last but not least, thank you *Tamarix ramosissima* for invading our beautiful desert riparian ecosystems and paving the way for graduate students like myself to study watershed resources in the desert. It is through your stubbornness and our failed attempts toward your eradication that doors are now open for peaceful co-existence.

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ABSTRACT

In May 2007, Bureau of Reclamation seeded small-scale restoration plots with native riparian vegetation. Willow species (*Salix* spp.) established poorly and Fremont cottonwood (*Populus fremontii*) established abundantly, but tamarisk (*Tamarix ramosissima*) also established abundantly as a volunteer species. As a result, a monitoring program was initiated to assess native versus non-native species competition.

Two irrigation regimes were implemented during 2008 to provide two treatments (i.e. small versus large estimated depletion of plant-available water). Optimal irrigation depth and frequency was researched to produce draw-down rates promoting root elongation toward groundwater with a goal to eventually cease irrigation. Vegetation characteristics were monitored at the beginning and end of the growing season to obtain growth rates and mortality and to document changes in vegetation cover and density.

Overall growth rates were highest in 2008, one year after seeding. Average *P. fremontii* growth rates were significantly higher than *T. ramosissima* in both growing seasons. Overall *P. fremontii* mortality was also lower than *T. ramosissima* during both growing seasons. *P. fremontii* was the only species in which crown cover percentage increased every survey and, as a result, was the most abundant species in the overstory beginning in spring 2008. Results at our study site indicate that cottonwood can out-compete tamarisk with respect to growth rate, cover percentage, survival, and density over time.

1. INTRODUCTION

1.1 Historical and Current Conditions of Western River Systems

Riparian zones in the southwestern United States have drastically been altered in the past century due to disruption in the natural flow regime (Stromberg 2001). Many large rivers are now regulated to control water storage and to manage flows for energy demands, recreation, and to provide water necessary for agricultural practices and supplemental municipal use. Traditionally, natural pulse floods washed salts from the soil and replenished and freshened aquifers (Ohmart et al. 1988, Poff et al. 1997, Mahoney & Rood 1998, Glenn & Nagler 2005, Pataki et al. 2005). Increased soil salinity and decreased soil moisture are now typical characteristics of many western river systems (Sprenger et al. 2002, Pataki et al. 2005, Nagler et al. 2008). Pulse floods were also responsible for sediment erosion and deposition, which created sites suitable for native species recruitment (Scott et al. 1996). Altered native flood patterns are associated with the decline in native riparian cottonwood (*Populus* spp.) forests (Rood & Mahoney 1990, 1991).

Current conditions provide an opportunity for invasive species such as tamarisk (*Tamarix* spp.) to replace native species such as cottonwood and willow (*Salix* spp.). Invasive species can take advantage of current soil conditions because they are often better adapted to abiotic stresses such as high soil salinity and low soil moisture (Jackson et al. 1990, Nagler et al. 2008). Overbank flooding necessary for native species recruitment does not occur because many river channels are now disconnected from natural floodplains due to channel incision or channelization. In addition, water diversion

and ground water pumping has resulted in groundwater decline, reducing the recruitment of *Populus* spp. (Busch & Smith 1995). *Tamarix* spp., on the other hand, has proliferated along southwestern rivers due its ability to disperse seeds throughout the growing season and establish when conditions are suitable (Ware & Penfound 1949, Horton et al. 1960).

1.2 Restoration Implications along the Lower Colorado River

An objective of the Lower Colorado River Multi-species Conservation Program is to provide habitat along the Lower Colorado River (LCR) that can be used successfully by native fish and wildlife (Raulston 2003). Managers will face major challenges in the restoration of the LCR because the aquifer underneath much of the native riparian floodplain is now saline, with salinity increasing in proportion to distance from the river channel (Nagler et al. 2008). Greenhouse studies have suggested that the only viable areas for successful *Populus* spp. restoration is likely to be near the natural river channel (Nagler et al. 2008). Further complicating efforts are highly saline vadose zones associated with capillary rise of water and salt accumulation. Due to low salt tolerance of *Populus* spp. and *Salix* spp., native mesic tree species can no longer establish on riverbanks without overbank flooding or supplemental irrigation water.

Many current restoration efforts aim to regenerate native riparian habitat by providing supplemental water during native seed dispersal (Freidman et al. 1995, Taylor et al. 1999, and Sprenger et al 2002). Although these practices have resulted in successful germination and establishment, they may not promote long-term stability of *Populus* spp. stands because survival through early stages is infrequent in natural forest populations (Peet and Christensen 1987, Taylor et al. 1999, 2006). For instance, *Populus*

spp. and *Tamarix* spp. both can establish under surface flooding conditions in high salinity soils and shallow water tables, but *Populus* spp. likely would not continue to survive under these conditions (Sprenger et al. 2002, Bhattacharjee et al. 2006). Clearly, analysis of long-term trends from existing restoration plots will provide valuable information to managers when choosing methods that will optimize long-term native species response.

1.3 Current Restoration Methods

Current riparian restoration typically consists of irrigating native species by one of four methods: (1) planted pole cuttings (Dreesen 2002, Landis 2006), (2) transplanted nursery plants (Pinto 2004, Landis 2006), (3) natural seed dispersal (Bhattacharjee et al. 2006, Taylor et al. 2006), or (4) native seed collection and distribution (Friedman et al. 1995, Sprenger et al. 2002, Raulston 2003, Bhattacharjee et al. 2006, Taylor et al. 2006, GSA 2009). There are benefits and disadvantages among different methods associated with cost, labor intensity, and environmental factors. Cutting and collecting poles may be labor intensive and specific protocols need to be followed for successful establishment while maintaining genetic diversity (USDA 2005). Costs are also driven up if nurseries or glass houses are required for establishment before transplanting individual trees. Pole cutting and native seed dispersal techniques can, however, ensure genetic diversity if multiple source trees are used from different locations near the restoration site (Schalau 2000). While the success of each restoration method is dependent on site characteristics (soil characteristics, depth to the water table, grazing history; Dreesen et al. 2002), all methods will require a substantial amount of water during the establishment phase.

1.4 Research Objectives

Restoration sites in close proximity to rivers associated with shallow water tables may require less water than other restoration locations because irrigation could be decreased or ceased when target species become phreatophytic. As many agricultural fields are located on traditional floodplains along rivers in the southwestern United States, one promising restoration method is to convert retired agricultural fields into areas that support native riparian species. By using existing canal infrastructure for controlled irrigation practices, watering schemes can be designed to promote the ability of native species to coexist with or out-compete invasive species. While successful seedling establishment has been demonstrated using these practices (GSA 2009) the long-term success of this restoration method remains uncertain.

As part of a small-scale restoration project in May 2007, the Bureau of Reclamation seeded 36 test plots on Cibola National Wildlife Refuge (CNWR) with native *P. fremontii* and two native willow species, *S. gooddingii*, and *S. exigua*. Willow establishment was minimal while *P. fremontii* establishment was high. Additionally, *T. ramosissima*, a volunteer non-native species established alongside *P. fremontii*. The co-establishment of *P. fremontii* and *T. ramosissima* resulted in the implementation of a long-term monitoring program.

This program was the basis of our study to assess competition between non-native and native growth success over time. This research examines two irrigation treatments: (1) frequent shallow irrigation; and (2) infrequent deep irrigation. We assess how these treatments: (a) contribute to above and below-ground growth of native and non-native

species; (b) promote native species survival; and (c) support a competitive advantage of Fremont cottonwood (*Populus fremontii*) over tamarisk (*Tamarix ramosissima*). These results will be used to provide management recommendations for optimal planted or seeded riparian tree density and irrigation depth and frequency depending on restoration objectives.

1.5 Site Description

CNWR was established in 1964 in order to restore and protect historic habitat and wintering grounds for migratory birds and other wildlife. CNWR is located between Yuma, AZ and Blythe, CA and straddles the lower Colorado River in La Paz County, AZ and Imperial County, CA (Figure 1).
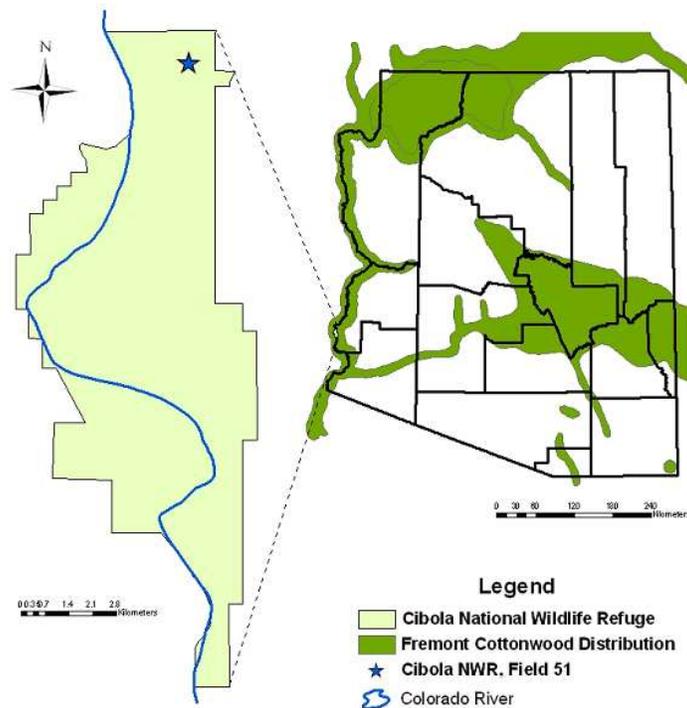

Figure 1. Study site, Cibola National Wildlife Refuge, AZ.

Annual temperature ranges from 4°C to 49°C and rainfall averages under 100 mm yr^{-1}. Over 4000 ha is made up of riparian vegetation, over half is near-monoculture *Tamarix* spp. (i.e. >90% *Tamarix* spp.) with less than half consisting of mixed native species including *P. fremontii*, Goodding's willow (*Salix gooddingii*), and coyote willow (*Salix exigua*). Other native species include arrowweed (*Pluchea sericea*), quailbush (*Atriplex lentiformis*), and fourwing saltbush (*Atriplex canescens*) while upland shrubs include honey mesquite (*Prosopis glandulosa*) and screwbean mesquite (*Prosopis pubescens*).

Our study site, Field 51 on the CNWR, is located 1.8 km east of the Colorado River (33°22'03" N, 114°40'50" W; 71 m elevation) in La Paz County, AZ. Field 51 is an 8-ha retired agricultural field located on the native floodplain terrace and is managed by the Bureau of Reclamation (BOR) for restoration practices. The northern, southern, and eastern units adjacent to Field 51 are also currently managed for native riparian restoration while the land west of Field 51 was recently cleared in 2009 for large-scale native habitat restoration.

Surface soil (i.e. 0 - 50 cm) at Field 51 is characterized as silt and silt loam. Down to one meter, silt and silt loam are predominant, but sand and sand loam patches are prevalent at depth (~1 m) toward the southern portion of Field 51 (GSA 2008). Prior to irrigation on Field 51, average surface soil salinity was less than 2 dS m^{-1} (GSA 2008). Depth to groundwater, measured with on-site well-point piezometers prior to irrigation ranged seasonally from 2.2 to 2.6 m below ground surface (GSA 2008).

1.6 Structure of Subsequent Chapters

The body of this study is in manuscript form placed in its entirety in Appendix A. This appendix includes detailed methods, results, and discussion. This manuscript will be submitted to the Journal of Restoration Ecology in the spring of 2010 for peer review and publication. Appendix B is a detailed addition that reviews leaf area index (LAI) and outlines how to use a SunScan probe (Delta-T Devices Ltd., Cambridge, UK) for LAI sampling at sites with non-uniform vegetation. This appendix also includes two *in situ* examples outlining detailed methods for specific scenarios that will assist in validating probe readings.

The following section, Present Study, is a compilation of abstracts summarizing Appendix A and Appendix B.

2. PRESENT STUDY

2.1 Abstract to Appendix A

Following Lower Colorado River Multi-Species Conservation Program objectives, the Bureau of Reclamation is converting agricultural fields to restoration areas using existing irrigation infrastructure to support native mesic species. Irrigation practices aim to favor native over non-native species growth as numerous studies have shown that cottonwood (*Populus* spp.) out-compete tamarisk (*Tamarix* spp.) under irrigation that mimics natural flooding.

In May 2007, small-scale restoration plots at Cibola National Wildlife Refuge were seeded with native riparian vegetation. Willow (*Salix* spp.) established poorly and Fremont cottonwood (*Populus* spp.) established abundantly, but tamarisk (*Tamarix ramosissima*) also established abundantly as a volunteer species. As a result, a monitoring program was initiated in 2008 to assess native versus non-native species competition.

Two irrigation regimes were implemented during 2008 to provide two treatments (i.e. small versus large estimated depletion of plant-available water). Optimal irrigation depth and frequency would produce draw-down rates promoting root elongation toward groundwater with a goal to eventually cease irrigation. Vegetation characteristics were monitored at the beginning and end of the growing season to obtain growth rates and mortality and to document changes in vegetation cover and density.

Overall growth rates were highest in 2008, one year after seeding. Average *P. fremontii* growth rates were significantly higher than *T. ramosissima* in both growing

seasons. Overall *P. fremontii* mortality was lower than *T. ramosissima* during both 2008 and 2009 growing seasons. *P. fremontii* was the only species in which crown cover percentage increased every survey and, as a result, was the most abundant species in the overstory beginning in spring 2008. Results at our study site indicate that cottonwood can out-compete tamarisk with respect to growth rate, cover percentage, survival, and density over time.

2.2 Abstract to Appendix B

Leaf area index (LAI) is the one-sided green leaf area per unit ground surface area. LAI is a key variable for global models of biosphere-atmosphere exchanges such as energy, carbon dioxide, and water vapor (Asner 2003), all of which occur at the leaf level. Models often use *in situ* LAI to scale-up measurements to larger landscapes. Hydrological processes are greatly affected by plant physiology and leaf area plays an influential role on the magnitude of impact.

Measuring LAI involves either direct or indirect sampling methods. Direct methods involve destructively sampling a portion or a complete plant/tree and calculating leaf area with a machine or manual method. Indirect methods, on the other hand, are non-destructive, but typically require calibration. Calibration usually involves validating data output values by values acquired through biomass sampling. Detailed methods for acquiring LAI in the field are outlined with two methods of validation: (1) destructive sampling coupled with laboratory processing; and (2) allometry measurements coupled with destructive sampling and laboratory processing.

LAI has many applications in watershed management. LAI data yield valuable information for tree-stand and crop health. Agricultural practices use LAI for measuring primary production and obtaining crop coefficients (Wilhelm et al. 2000). LAI data may also be beneficial for water budgeting, water conservation, base-flow management, and groundwater monitoring because LAI is an important parameter for physical and biological processes associated with photosynthesis, respiration, transpiration, carbon and nutrient cycling, and rainfall interception (Chen and Cihlar 1998), all of which effect the hydrologic process. The utility of LAI measurements are site-specific and many methods are available depending on objectives. While LAI sampling was not conducted for the research herein, methods are included due to the time and effort spent on researching robust sampling methods.

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APPENDIX A: Using Existing Agricultural Infrastructure for Restoration Practices: Factors Influencing Successful Establishment of Cottonwood over Tamarisk

Daniel P. Bunting[1], Shirley A. (Kurc) Papuga[1], and Matthew R. Grabau[2]

[1]School of Natural Resources and the Environment, University of Arizona, Tucson, AZ 85721 USA

[2]GeoSystems Analysis, Inc.,Tucson, AZ, 85745 USA

1. Introduction

Following dam construction and water diversion, altered native flood patterns in much of the western United States are associated with declines in native riparian cottonwood (*Populus* spp.) forests (Rood & Mahoney 1990, 1991). This is largely because regulated flows do not scour back vegetation as floods in the past had done, which had created conditions favorable for *Populus* spp. recruitment and survival (Howe & Knopf 1991, Taylor et al. 1999). Instead, this altered hydrology has increased soil salinity and decreased soil moisture, which are now typical characteristics of many western river systems (Sprenger et al. 2002, Pataki et al. 2005). These new conditions provide an opportunity for invasive species such as tamarisk (*Tamarix* spp.) to replace native species such as *Populus* spp. because invasive species are often better adapted to abiotic stresses such as high soil salinity and low soil moisture (Jackson et al. 1990, Nagler et al. 2008).

The ability of invasive species to establish in traditional native habitat is also attributed to seed phenology and dispersal. Typically, native *Populus* spp. disperse seeds for a short period of time (late spring to early summer) and seeds remain viable for a only few weeks; on the other hand, *Tamarix* spp. disperse seed throughout the growing season which can germinate whenever soil conditions are suitable (Ware & Penfound 1949, Horton et al. 1960). While this may give *Tamarix* spp. a recruitment advantage, studies have indicated a superiority of *Populus* spp. over *T. ramosissima* during establishment when water is not limiting (Everitt 1995, Sher et al. 2000, 2002, and Marler et al. 2001). As a result, many restoration efforts have aimed to regenerate native riparian habitat by

providing supplemental water during native seed dispersal (Freidman et al. 1995, Taylor et al. 1999, and Sprenger et al 2002).

Although these practices have resulted in successful germination and establishment, they may not promote long-term stability of *Populus* spp. stands because survival through early stages is infrequent in natural forest populations (Peet & Christensen 1987, Taylor et al. 1999, 2006). For instance, *Populus* spp. and *Tamarix* spp. both can establish under surface flooding conditions in high salinity soils and shallow water tables, but *Populus* spp. likely would not continue to survive under these conditions (Sprenger et al. 2002, Bhattacharjee et al. 2006). Clearly, analysis of long-term trends from existing restoration plots will provide valuable information to managers when choosing methods that will optimize long-term native species response.

In one of few long-term *Populus* spp. restoration studies, Taylor et al. (2006) found that community composition (i.e. density characteristics) is crucial in the successful restoration of *Populus* spp. forests. In their study, they looked at ten-year-old *Populus* spp. restoration plots in the Middle Rio Grande Valley, New Mexico. Many areas had high woody density after initial establishment. Ten years later these high density areas were shown to be related to high *Populus* spp. density and overstory cover. *Populus* spp. within these high-density areas had greater heights, but were associated with smaller diameters at breast height (DBH) and smaller foliar volumes. In plots where *Tamarix* spp. was initially present, *Tamarix* spp. density negatively influenced *Populus* spp. foliar volume, but did not affect *Populus* spp. density. Initial community composition variables will influence the outcome of restoration practices.

Another recent study relating high initial *Tamarix* spp. density to *Populus* spp. survival found that *Populus* spp. subsequently out-compete *Tamarix* spp. after initial soil conditions favored germination of both species (Bhattacharjee et al. 2008). In fact, as weaker riparian species disappear from the community over time, it has been shown that *Populus* spp. respond with increased growth rates (Taylor et al. 2006). Because initial community composition is likely to affect height, canopy structure, and survival of native riparian species (Taylor et al 2006), managers should choose a method of restoration to match specific restoration goals. High-density native communities with shorter canopies may be desired to support habitat requirements for vertebrates, such as the Southwestern Willow Flycatcher (*Empidonax traillii*), while low initial densities will favor larger *Populus* spp. forests (Taylor et al. 2006).

Restoring native riparian habitat for the benefit of native wildlife species is an objective of the Lower Colorado River Multi-Species Conservation Program (MSCP). Of particular interest is successful restoration that minimizes costs and resources while accounting for non-native species, such as *Tamarix* spp.. Current riparian restoration typically consists of irrigating native species by one of four methods: (1) planted pole cuttings (Dreesen 2002, Landis 2006); (2) transplanted nursery plants (Pinto 2004, Landis 2006); (3) natural seed dispersal (Bhattacharjee et al. 2006, Taylor et al. 2006); or (4) native seed collection and distribution (Friedman et al. 1995, Sprenger et al. 2002, Raulston 2003, Bhattacharjee et al. 2006, Taylor et al. 2006, GSA 2009). Consequently, restoration can be time consuming and often requires a substantial amount of irrigation water.

Restoration sites in close proximity to rivers associated with shallow water tables may require less water than other restoration locations because irrigation could be decreased or ceased when target species become phreatophytic. As many agricultural fields are located on traditional floodplains along rivers in the southwestern United States, one promising restoration method is to convert retired agricultural fields into areas that support native riparian species. By using existing canal infrastructure for controlled irrigation practices, watering schemes can be designed to promote the ability of native species to coexist with or out-compete invasive species. While successful seedling establishment has been demonstrated using these practices (GSA 2009) the long-term success of this restoration method remains uncertain.

Here we examine two irrigation treatments: (1) frequent shallow irrigation; and (2) infrequent deep irrigation. We assess how these treatments: (a) contribute to above and below-ground growth of native and non-native species; (b) promote native species survival; and (c) support a competitive advantage of Fremont cottonwood (*Populus fremontii*) over tamarisk (*Tamarix ramosissima*). These results will be used to provide management recommendations for optimal planted or seeded riparian tree density and irrigation depth and frequency depending on restoration objectives.

Specifically, this research addresses three main hypotheses: (1) an irrigation treatment applying a large amount of water less frequently will promote higher growth rates in *P. fremontii* and *T. ramosissma* versus less water more frequently; (2) regardless of irrigation treatment, *P. fremontii* will continue to dominate crown and canopy cover percentages when water is not limiting; and (3) smaller trees will die off first promoting the tallest trees to establish a stable canopy. In addition, we anticipate higher *P. fremontii*

and *T. ramosissma* mortality in the infrequent, deep irrigation due to lower near-surface plant-available water negatively impacting smaller trees.

2. Methods

2.1 Study Site

Cibola National Wildlife Refuge (CNWR) was established in 1964 in order to restore and protect historic habitat and wintering grounds for migratory birds and other wildlife. CNWR is located between Yuma, AZ and Blythe, CA and straddles the lower Colorado River in La Paz County, AZ and Imperial County, CA (Figure 1). Annual temperature ranges from 4°C to 49°C and rainfall averages under 100 mm yr^{-1}. Over 4000 ha is made up of riparian vegetation, over half is near-monoculture *Tamarix* spp. (i.e. >90% *Tamarix* spp.) with less than half consisting of mixed native species including *P. fremontii*, Goodding's willow (*Salix gooddingii*), and coyote willow (*Salix exigua*). Other native species include arrowweed (*Pluchea sericea*), quailbush (*Atriplex lentiformis*), and fourwing saltbush (*Atriplex canescens*) while upland shrubs include honey mesquite (*Prosopis glandulosa*) and screwbean mesquite (*Prosopis pubescens*).

Our study site, Field 51 on the CNWR, is located 1.8 km east of the Colorado River (33°22'03" N, 114°40'50" W; 71 m elevation) in La Paz County, AZ. Field 51 is an 8-ha retired agricultural field located on the native floodplain terrace and is managed by the Bureau of Reclamation (BOR) for restoration practices. The northern, southern, and eastern units adjacent to Field 51 are also currently managed for native riparian restoration while the land west of Field 51 was recently cleared in 2009 for large-scale native habitat restoration.

Surface soil (i.e. 0 - 50 cm) at Field 51 is characterized as silt and silt loam. Down to one meter, silt and silt loam are predominant, but sand and sand loam patches are prevalent at depth (~1 m) toward the southern portion of Field 51 (GSA 2007). Prior to irrigation on Field 51, average surface soil salinity was less than 2 dS m^{-1} (GSA 2008). Depth to groundwater, measured with on-site well-point piezometers prior to irrigation ranged seasonally from 2.2 to 2.6 m below ground surface (GSA 2008).

2.2. Study Design

As part of a small-scale restoration project in 2007, BOR seeded 36 test plots on Field 51 with native *P. fremontii* and two native willow species, *S. gooddingii*, and *S. exigua*. Willow establishment was minimal while *P. fremontii* establishment was high. Additionally, *T. ramosissima*, a volunteer non-native species established alongside *P. fremontii*. The co-establishment of *P. fremontii* and *T. ramosissima* resulted in the implementation of a long-term monitoring program. This program was the basis of our study to assess competition between non-native and native growth success over time.

Beginning in 2008, two different watering regimes were chosen to test effects of different depths and frequencies on growth and survival of *P. fremontii* and *T. ramosissima* yearlings. The first irrigation treatment (T_1) consisted of a 7-cm water input once per week while the second irrigation treatment (T_2) consisted of a 21-cm water input once every three weeks. In 2009, T_1 and T_2 were altered to 5.5 cm and 22 cm, respectively, to promote more depletion.

The 36 original BOR test plots were divided into four blocks and irrigation assignments were staggered to account for variation across the length and width of the

study site. The northeast and southwest blocks were introduced to the shallow frequent irrigation regime (T_1) while the northwest and southeast blocks were introduced to the deep infrequent irrigation treatment (T_2; Figure 2). Each treatment had exactly one replicate. In addition, nine 12-m x 6-m study plots were established within each block for conducting vegetation surveys (Figure 2).

2.3. Measurements

2.3.1 Above ground measurements

All above ground measurements were made in two field campaigns for two consecutive years (2008-2009). In both years, the first campaign was initiated before the onset of the growing season (i.e. May-June) and the second campaign was initiated after the growing season, but before fall leaf-drop (i.e. Sept-Oct).

Growth success was monitored as a function of overstory (crown) dominance. The point-intercept method (RMWU 2005) was used to document changes in cover percentages over time. Three 6-m transects were randomized within each plot (Figure 2). Every cover type below a sample point, spaced at 30.5-cm intervals along each transect, was recorded a maximum of once per point (a "hit"). The first cover type below the sample point represented the crown cover whereas canopy cover included both overstory and understory cover; crown cover indicates the dominant (tallest) species in the observation area, whereas canopy cover indicates total abundance of a given cover. Plot cover percentage was obtained by dividing the number of hits by the number of sample points as described in Equation (1):

Cover % = (x/n)*100 (1)

where x is the number of "hits" of a given cover type, and n is the number of sample points per plot.

Growth rates were measured in three permanent, 1-m x 0.5-m quadrats within all plots. The placement of each quadrat was randomized along the three transects used for conducting cover percentage surveys (Figure 2). Every target tree (i.e. *P. fremontii* and *T. ramosissima*) within each quadrat was tagged with a unique identification number using an aluminum tag attached to the base of each trunk with bailing wire. Total height of all target trees was measured using a 1-m or 3.5-m measuring stick before and after each growing season to obtain growth rates (cm growing season^{-1}). The second campaigns also included caliper-estimated diameter at breast height (DBH) at 137 cm above ground surface. Additionally, each tree was qualitatively examined using a health index for leaf color and branch senescence and a modified relevé index (MDNR 2007) was used to document all other vegetation within the quadrat, including estimates of crown and canopy cover and average height (not reported here).

Survival was documented using data collected within the permanent quadrats. Any tagged target tree that was dead upon subsequent surveys was documented. Overall species specific mortality percentages were obtained by dividing each species total number of deaths per growing season by each species total number of trees during the spring survey as described in Equation (2):

$$Mortality \% = (x/n)*100 \tag{2}$$

where x is the total number of deaths during the growing season and n is the total number of trees documented in the spring survey. Overall species-specific stem density (stems m^{-2}) was calculated by averaging all stem densities documented within each quadrat. Stem

densities within each quadrat were calculated by dividing the total number of trees from each species by total area within each quadrat (0.5 m^2) as described in Equation (3):

$$Stem\ density = 2(x/A) \tag{3}$$

where x is the number of trees and A is the area within each quadrat. In Equation (3), multiplying by two normalizes the stem densities into stems m^{-2} for clarity.

2.3.2 Below ground measurements

In each plot, continuous measurements of volumetric water content (θ_{15}), soil temperature (T_{soil}), and pore water salinity *(EC)* were measured at 15 cm using ECH$_2$O-TE sensors (Decagon Devices, Inc., Pullman, WA). Additionally, volumetric water content was monitored at 46 cm (θ_{46}) and 91 cm (θ_{91}) in each plot using EC-10 sensors. Groundwater depth and salinity was measured throughout the study using five well-point piezometers. Soil texture and salinity were characterized in winter 2007 for 1-m soil core samples taken from the center of each plot using a JMC Environmentalist's Subsoil Probe (ESP; Clements Associates Inc., Newton, IA).

A qualitative root survey was performed in October 2008. Excavations (1.5 m) were made in a total of 10 plots chosen to represent the range of variability in *P. fremontii* and *T. ramosissima* density, with the objective to examine root size, growth, and penetration toward groundwater. This resulted in destructive sampling through 10 permanent quadrats, thus 2009 growth rates, cover percentages, and mortality were normalized with the exclusion of the outermost quadrat and a portion of the outermost transect.

2.4 Data Analysis

Point-transect data were divided into four groups for analysis: (1) *P. fremontii*; (2) *T. ramosissima*; (3) combined grasses and sedges; and (4) combined shrubs and forbs. Every randomized point-transect within each plot was combined to get an average crown and canopy cover percentage per plot. All averaged plot cover percentages were then combined and averaged to estimate overall crown and canopy cover percentage each growing season. Every randomized quadrat within each plot was combined to compute an average stem-density per plot. All averaged plot stem-densities were then combined and averaged to estimate overall density for *P. fremontii* and *T. ramosissima*.

In addition, all trees were separated into size classes to determine if initial tree size had an impact on growth rates and survival over time. In order to make direct comparisons between species while accounting for disparity in *P. fremontii* and *T. ramosissima* growth rates, tree height data was normalized using five size classes defined by the ratio of individual tree heights to the tallest tree in each species' sampling distribution in the spring survey (Table 1). For example, the first bin was composed of all trees that were only 0-20% the height of the tallest tree during the spring survey (i.e. 2008: *P. fremontii* = 304 cm, *T. ramosissima* = 267 cm; 2009: *P. fremontii* = 500 cm, *T. ramosissima* = 410 cm), whereas the fifth bin was composed of all trees that were 80-100% the height of the tallest tree. JMP8 software was used to compare means and run Student t-tests for determining significant differences in growth rates between irrigation treatments and among size classes.

3. Results

3.1 Growth Rates

Overall growth rates were highest in 2008, one year after seeding. Average *P. fremontii* growth rates were significantly higher than *T. ramosissima* in both growing seasons. On average in the 2008 growing season *P. fremontii* grew 57.3 cm, but decreased by 70% in 2009, whereas *T. ramosissima*, grew 16.7 cm on average in 2008, but decreased by only 43% in 2009 (Figure 3a). Average growth rates within each species were not significantly different in T_1 versus T_2 in either growing season for either species (Figure 3a). In fact, excluding *T. ramosissima* growth rates in the 60-80% size category in 2009, T_1 and T_2 did not yield disparate growth rates within any size classes for either species (Figure 3b). Subsequent size classes ascending from small to large had significantly higher growth rates in 2008 for both species, but were much more variable in 2009 (Figure 3b).

3.2 Cover Percentages

Cover percentage surveys were conducted to examine overall trends at the study site and were not compared between T_1 and T_2. *P. fremontii* was the only species in which crown cover percentage increased during every survey and, as a result, was the most abundant species in the overstory beginning in the spring of 2008 until the termination of the project. *T. ramosissima* crown cover percentage leveled off at 12% in 2009 while *P. fremontii* increased to 70% by fall of 2009 (Figure 4a). Grasses and sedges

decreased in crown cover from 81% in the fall of 2007 to 15% in the fall of 2009 while shrubs and forbs crown cover was reduced to 0% by fall 2009 (Figure 4a).

P. fremontii and grasses and sedges were the most abundant species within the canopy (i.e. understory) after three growing seasons, 72% and 62%, respectively (Figure 4b). *T. ramosissima* canopy cover increased every season, except for a small decrease in the spring of 2009. Shrubs and forbs experienced a reduction in canopy cover from 47% in fall 2007 to less than 1% in 2009 (Figure 4b).

3.3 Mortality

Overall *P. fremontii* mortality was lower than *T. ramosissima* during both 2008 and 2009 growing seasons, although it was only significantly lower in 2008 (Figure 5a). In 2008 and 2009 *P. fremontii* mortality was similar (5.9% and 5.4%, respectively), however, *T. ramosissima* experienced higher mortality in 2008 (11.6%) compared to 2009 (7.6%, Figure 5a) No significant differences were observed in same species *P. fremontii* or *T. ramosissima* mortality between T_1 and T_2 in 2008 or 2009 (Figure 5a). The smallest size class, 0-20% of maximum tree height, accounted for the majority of both *P. fremontii* and *T. ramosissima* deaths (Figure 5b). Excluding two *P. fremontii* trees in 2008, there were no deaths in any classes above 40% of maximum tree height for either species (Figure 5b). Mortality percentage showed no disparity between T_1 and T_2 in any size class for either species, excluding when no *T. ramosissima* deaths were observed in 2009 within the 20-40% size category for T_1.

3.4 Stem Density

　　Stem density was also used to depict overall trends at the study site and were not compared between T_1 and T_2. Four months after seeding native species, *T. ramosissima* was the dominant woody species recorded in the initial fall 2007 survey. Despite *T. ramosissima* propagating as a volunteer species, it established at 25.0 stems m^{-2} while *P. fremontii* established at 17.6 stems m^{-2} (Figure 6). Both *P. fremontii* and *T. ramosissima* stem-density decreased dramatically over the first year of the project due to high mortality, and continued to decrease slightly through spring 2009 (Figure 6). In fall 2009, however, *P. fremontii* stem-density increased resulting in a stem-density greater than *T. ramosissima* for the first time since the onset of the study (Figure 6).

3.5 Belowground Characteristics

　　Near-surface soil temperature (T_{soil}) followed a typical ambient temperature pattern ranging from 10°C in the winter to 31°C in the summer. Salinity measured from 1-m soil cores in winter 2007 was highly variable among plots, ranging from 2.7 dS m^{-1} to 10.9 dS m^{-1}. Surface soil salinity tended to be higher near the northern portion of the study site and lower soil salinity was observed at depth in coarser, sandy soils. Pore water *EC* measured from sensors at 15 cm below ground surface ranged from 0.40 dS m^{-1} to 0.69 dS m^{-1} in T_1 and 0.30 dS m^{-1} to 0.63 dS m^{-1} in T_2. Average soil moisture recorded with sensors was similar between T_1 and T_2 at all depths (Figure 7a, 7b). θ_{15} increased rapidly (maximum = 0.44 cm^3 cm^{-3}) in response to irrigation events and then decreased rapidly (minimum = 0.19 cm^3 cm^{-3}) between events for both T_1 and T_2. θ_{46} generally increased rapidly (maximum = 0.37 cm^3 cm^{-3}) in response to irrigation events in T_1 and

T_2 and then slowly decreased (minimum = 0.30 cm^3 cm^{-3}) between irrigation events. Soil moisture at depth (θ_{91}) remained relatively stable throughout the study independent of T_1 and T_2 but still ranged from 0.27 cm^3 cm^{-3} to 0.35 cm^3 cm^{-3}. Depth to groundwater during the project ranged from 1.65 to 2.50 m below ground surface and generally increased during the winter dormant season and was variable throughout the remainder of the study depending on precipitation, irrigation input, and plant water use (Figure 7c).

A qualitative root survey conducted in fall 2008 revealed many large *P. fremontii* and *T. ramosissima* tree roots penetrating greater than 1.5 m below ground surface. Based on measurements of depth to groundwater, trees with roots penetrating depths greater than 1.75 will likely have access to groundwater for a portion of the growing season.

4. Discussion

4.1 Irrigation Effects

Our results did not support our hypothesis that an irrigation treatment applying a large amount of water less frequently would promote higher growth rates in *P. fremontii* and *T. ramosissma*. In T_2 we aimed to provide a slower draw down rate than T_1, which would benefit adventitious roots, promote penetration toward groundwater, and result in increased above and below-ground biomass. Our results, however, did not show any disparity in growth rates between T_1 and T_2. In fact, both *P. fremontii* and *T. ramosissima* had similar growth rates in T_1 and T_2 for all size classes, suggesting that irrigation had no treatment effect. In general, shoot growth has been shown to decrease with increasing rates of water table decline, from 1 to 10 cm day^{-1}, resulting in an

increased root:shoot ratio (Mahoney & Rood, 1991, 1992; Kranjcec et al. 1998; Amlin &

Rood 2002). Studies have also shown greater *P. fremontii* survival in slower draw down

rates (Bhattacharjee et al. 2006) and poor *P. fremontii* establishment with reduced growth

under fast ($\geq$10 cm day^{-1}) draw down rates (Kranjcec et al. 1998, Sprenger et al. 2002).

Analysis of volumetric water content at multiple depths allowed us to examine

how irrigation treatments affected plant-available water along the soil profile. θ_{15}

indicated rapid responses to irrigation events with subsequent rapid depletion due to

drainage and near-surface evaporation. θ_{46} had intermediate responses to irrigation

events with slower depletion likely due to drainage and plant water use. θ_{91}, however,

was rarely affected by irrigation events and remained relatively stable. A sharp decline in

θ_{91} was observed in November 2008, but this was independent of irrigation events and

was likely associated with a fluctuation in the water table or capillary fringe. Compared

to T$_1$, decreases in θ_{15} and θ_{46} were higher in T$_2$ but θ_{91} decreased very little in either

treatment. In fact, during irrigation, θ_{91} did not fluctuate significantly from the non-

growing season volumetric water content despite observations of roots at depths greater

than 1.5 m. These results suggest that water was being supplied to the root zone at a

greater rate than evapotranspiration. This could be due to: (1) irrigation water applied at

a higher rate than plant water use of vadose zone soil moisture; and/or (2) capillary rise of

groundwater that contributed to vadose zone soil moisture; and/or (3) tree use of both

groundwater and vadose zone soil moisture such that dry conditions were never observed

at depths below 90 cm. Likely, plant-available water was present at depths greater than

90 cm in plots of both treatments throughout the study period which may in part explain

why both irrigation treatments resulted in similar growth rates and mortality.

4.2 Community Composition

In this study, we intended to analyze density effects on growth rate and mortality; however analysis of variances rarely indicated any significant correlations. The presence of *T. ramosissima* did not affect *P. fremontii* growth rate or survival and vice versa. Furthermore, trees within plots of high stem-density compared to low stem-density did not have significantly larger tree heights or reduced DBHs. These findings are not consistent with other studies in which high initial woody species densities correlated with smaller DBHs, taller trees, and decreased foliar volumes (Taylor et al. 2006). However, trees of both species evidently experienced edge effects in which some trees along outer edges grew much taller and were associated with higher DBHs and foliar volumes. Our study did not allow for analysis of edge effects on tree establishment nor did we have a control to compare how a solitary tree would grow without intra- or inter-species competition. Both *T. ramosissima* and *P. fremontii* may behave functionally similar and the manner in which abiotic or biotic factors influence them may also be similar, which may explain the lack of density affects on inter-species competition. Understanding which biotic and abiotic factors influence competitive superiority will be an important area of future research.

Our results support our hypothesis that smaller trees would die off first. The two smallest size classes accounted for the majority of woody species die-off. Regulated irrigation coupled with high initial *P. fremontii* seeding rates and an abundant *T. ramosissima* seed bed produced seedlings at greater densities than would be expected naturally. Natural thinning was anticipated because both *P. fremontii* and *T. ramosissima*

are known to be shade intolerant (Johnson et al. 1976, Taylor et al. 1999) and are considered high water users (Weeks et al. 1987, Busch & Smith 1995, Sala et al. 1996, DiTomaso 1998), both of which would be affected by high stem densities. Smaller trees within the understory were presumably negatively impacted by reduced light resources and inferior root systems likely could not compete with larger, established trees.

High mortality of smaller trees was responsible for shifting the overall stem density throughout the study. *T. ramosissima* initially established at a higher density than *P. fremontii* in 2007, but *P. fremontii* ultimately had a higher stem-density in 2009. This shift in dominance supports a competitive superiority of *P. fremontii* over *T. ramosissima*. Our results show that *P. fremontii* has successfully outgrown and outnumbered *T. ramosissima* at this restoration site.

4.3 Restoration Implications

Restoration managers aim to minimize costs and resources while preventing tree stress and mortality. Importantly, an objective in this study was to research an optimal irrigation depth and frequency, but did not intentionally introduce conditions that would lead to tree stress and mortality. Our study suggests that more water was used in our two irrigation treatments than was necessary to maintain healthy *P. fremontii* stands. Additional studies that alter irrigation amount and frequency could offer an optimal watering schedule to further conserve water resources during establishment while still maintaining healthy riparian vegetation.

This project serves as an example of how retired agricultural fields may be successfully converted to riparian restoration sites. Although inferences may only be

drawn for this particular research site, our study area is typical of most regions along the lower Colorado River. Both sensor and lab results confirmed the presence of saline soils that often approach or exceed the tolerance of cottonwood and willow species. The ability of *P. fremontii* to survive under high soil salinity and shallow water tables has been challenged by other studies (Sprenger et al. 2002, Bhattacharjee et al. 2006). This particular study has documented the successful establishment of *P. fremontii* and *T. ramosissima* trees in saline soils in a field setting. Furthermore, larger trees have become phreatophytic after three growing seasons. Because both species are known to have facultative roots, further research is necessary to understand how groundwater will affect the long-term success of these trees.

Specific restoration objectives must be addressed before restoration methods are established. As stated earlier, high-density native communities with shorter canopies may be desirable for supporting vertebrate habitat, such as habitat for *E. traillii*. Low initial densities, however, are likely to favor larger *Populus* spp. forests (Taylor et al. 2006). Initial restoration objectives at this research site were to establish a dense, mixed riparian stand that would limit non-native recruitment, promote healthy invertebrate communities, and provide migratory bird habitat. However, understory vegetation diversity decreased throughout the study most likely due to shading, resulting from the establishment of a stable *P. fremontii* canopy throughout much of the site.

In many cases, completely eradicating *Tamarix* spp. may be unrealistic, undesirable, or prohibitively expensive (Shafroth et al. 2005, van Riper et al. 2008). All vegetation was mechanically cleared at our research site prior to seeding, but a larger scale project would require a larger financial investment. Studies have observed lower

avian abundance in *T. ramosissima* stands compared to native vegetation (Anderson &
Ohmart 1977, Cohan et al. 1978, and Engel-Wilson & Ohmart 1978), while other studies
suggest that a mixed native/*Tamarix* spp. habitat can improve avian abundance and
diversity (van Riper et al. 2008). Restoration practices should take into account the
feasibility of eradicating a portion of or entire *Tamarix* spp. monocultures and decide
whether an investment is necessary to meet project objectives. Furthermore, restoration
decisions are site-specific and site factors such as soil characteristics and above and
below-ground water availability will affect the successful establishment of replacement
vegetation (Briggs 1996, Hughes & Rood 2003).

4.4 Main Conclusions

The limiting similarity concept suggests that functionally similar species are more
likely to compete and will not co-exist due to competitive exclusion (McArthur & Levins
1967, Abrams 1983). However, our study supports a level of co-existence between both
P. fremontii and *T. ramosissima*. Figure 8 shows how rapid *P. fremontii* growth rates can
persist within dense tree stands inter-mixed with *T. ramosissima* at our study site.
Extended monitoring will be beneficial to document longer-term response to initial
establishment. Recent research on *P. fremontii* and *T. ramosissima* interactions supports
the ability of *T. ramosissima* to replace, but not invade, traditional native riparian habitat
(Nagler et al. 2008). Our research agrees with this statement as we did not observe new
T. ramosissima recruitment during our study, whereas *P. fremontii* re-sprouts and
recruitment was common.

Mechanisms of community assembly other than competition can alter abiotic and biotic conditions in a manner that also alters competitive interactions between plant species (Firn et al. 2010). Despite these potentially confounding factors, measuring competitive interactions of an invader and more desirable native species may provide key insights for restoration efforts, including the conditions needed to provide a competitive advantage to native species (Firn et al. 2010). Our research supports studies demonstrating the ability of *Populus* spp. to successfully out-compete *Tamarix* spp. during establishment when water is not limiting (Everitt 1995, Sher et al. 2000, 2002, and Marler et al. 2001). With respect to our treatment conditions, *P. fremontii* had higher growth rates, lower mortality, and dominated crown and canopy cover percentages after three growing seasons since establishment. Current trends suggest that *P. fremontii* will continue to dominate *T. ramosissima* over time.

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Tables

Table 1.

| (%) | 2008 | | | 2009 | | |
| | *Populus fremontii* | | | | | |
	size (cm)	$T_1 (n)$	$T_2 (n)$	*size (cm)*	$T_1 (n)$	$T_2 (n)$
0-20	0-59	76	44	0-99	38	38
20-40	60-120	109	113	100-199	89	110
40-60	121-181	142	104	200-299	79	68
60-80	182-242	52	62	300-399	46	45
80-100	243-304	9	14	400-500	9	21
	Tamarix ramosissima					
	size (cm)	$T_1 (n)$	$T_2 (n)$	*size (cm)*	$T_1 (n)$	$T_2 (n)$
0-20	0-52	139	144	0-81	197	159
20-40	53-105	188	127	81-163	120	54
40-60	106-159	91	20	163-245	38	7
60-80	160-212	17	4	246-327	6	1
80-100	213-267	3	0	328-410	2	0

Figures

Figure 1.
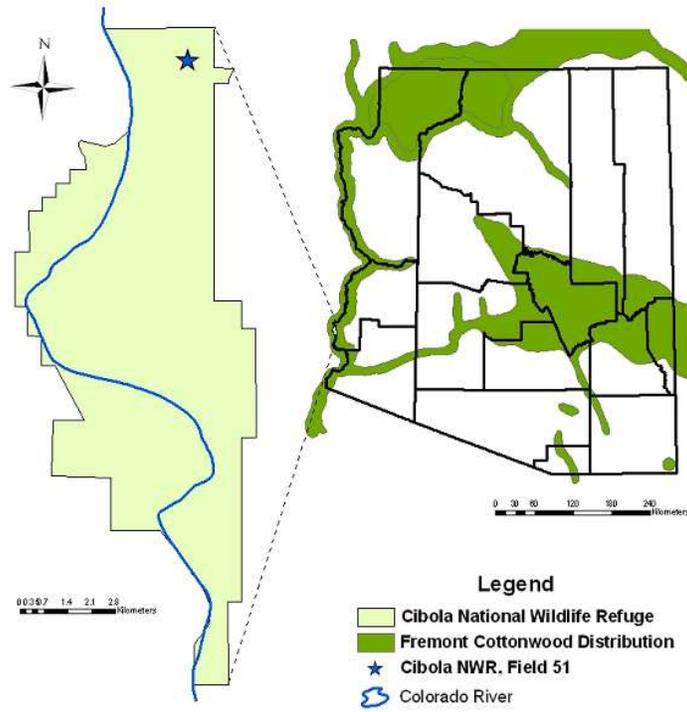
Figure 2.
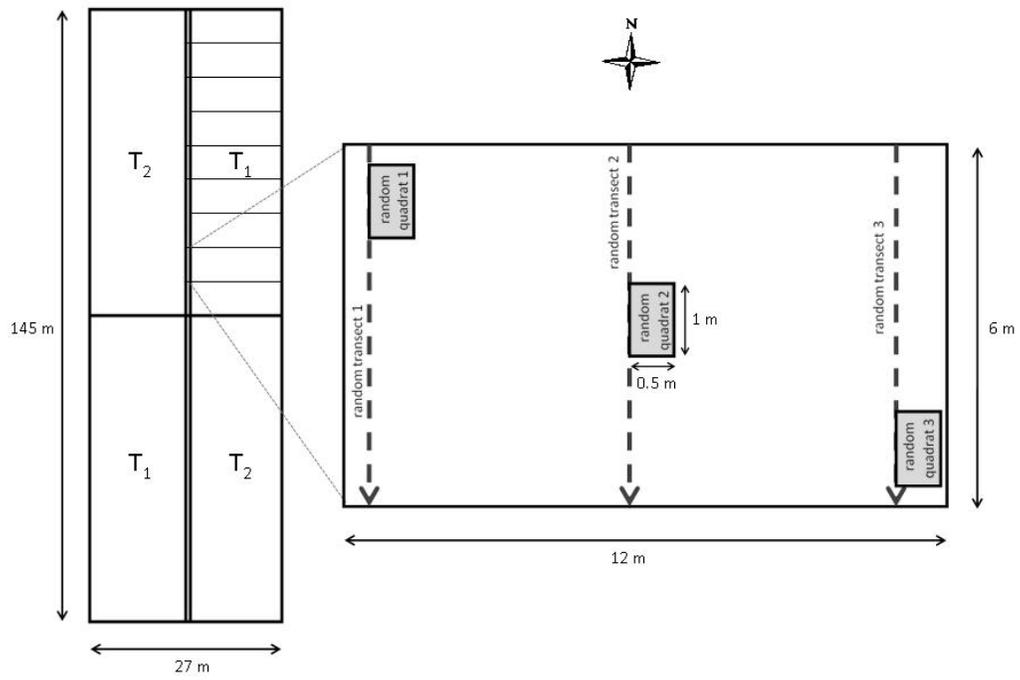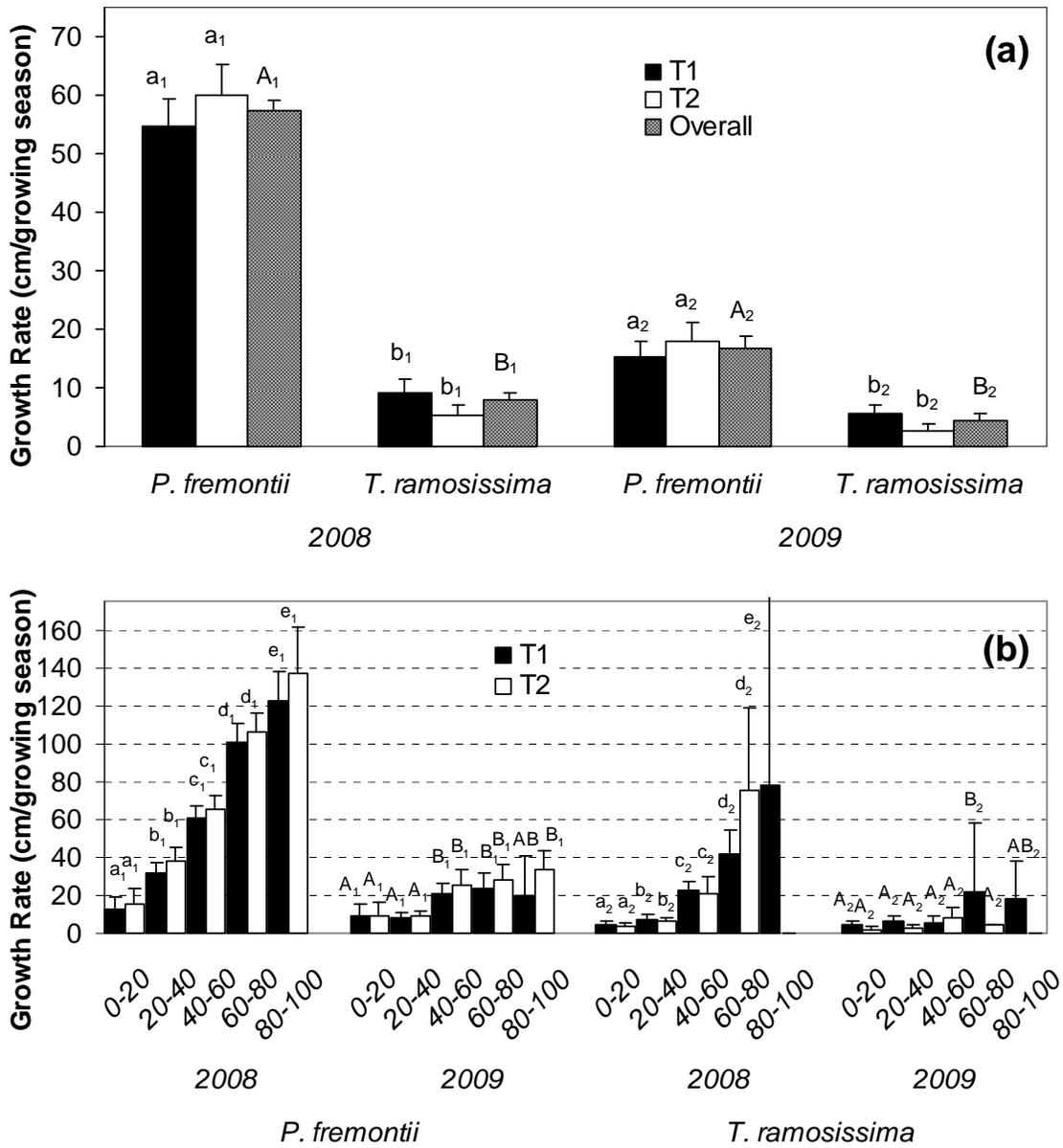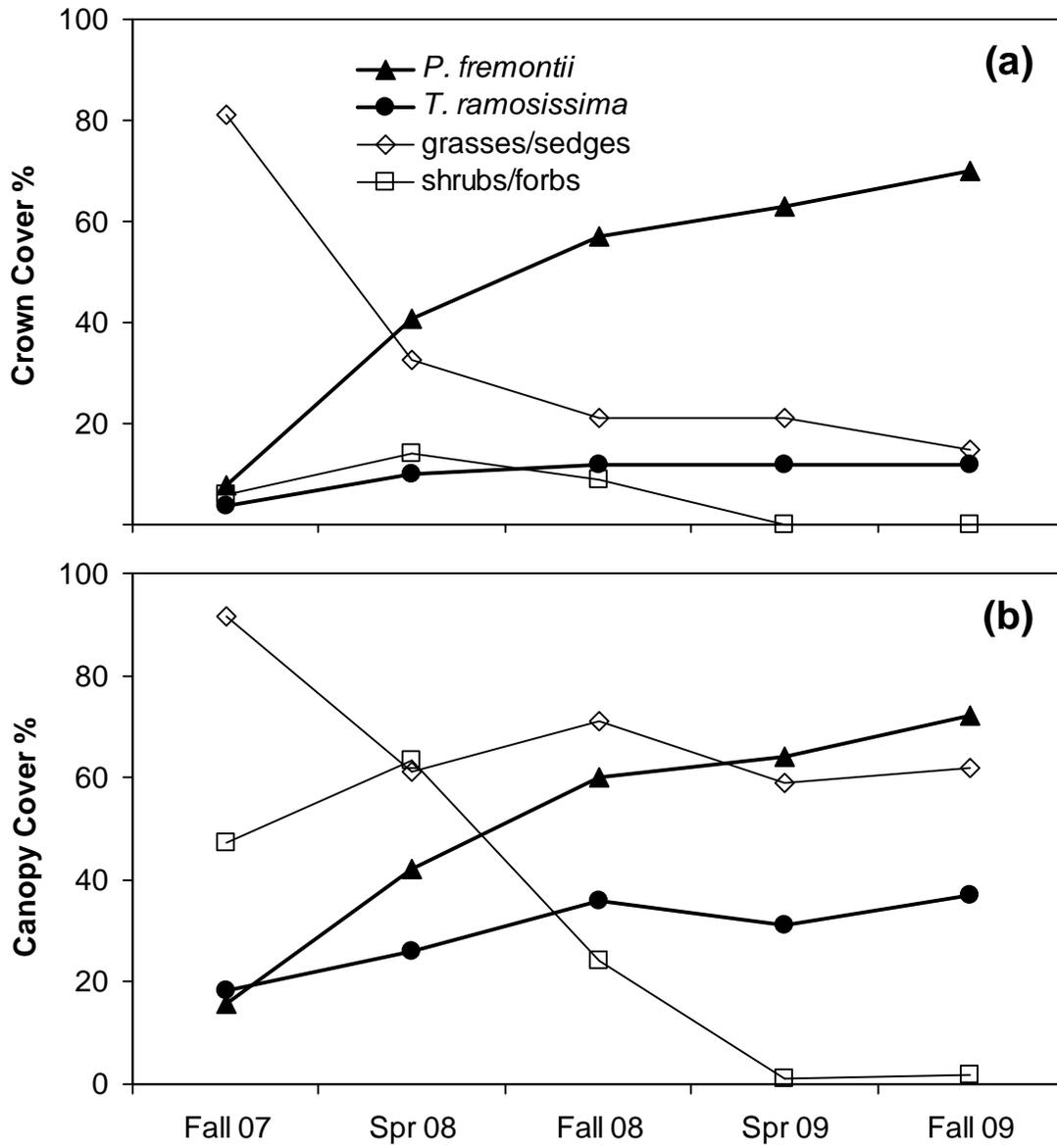
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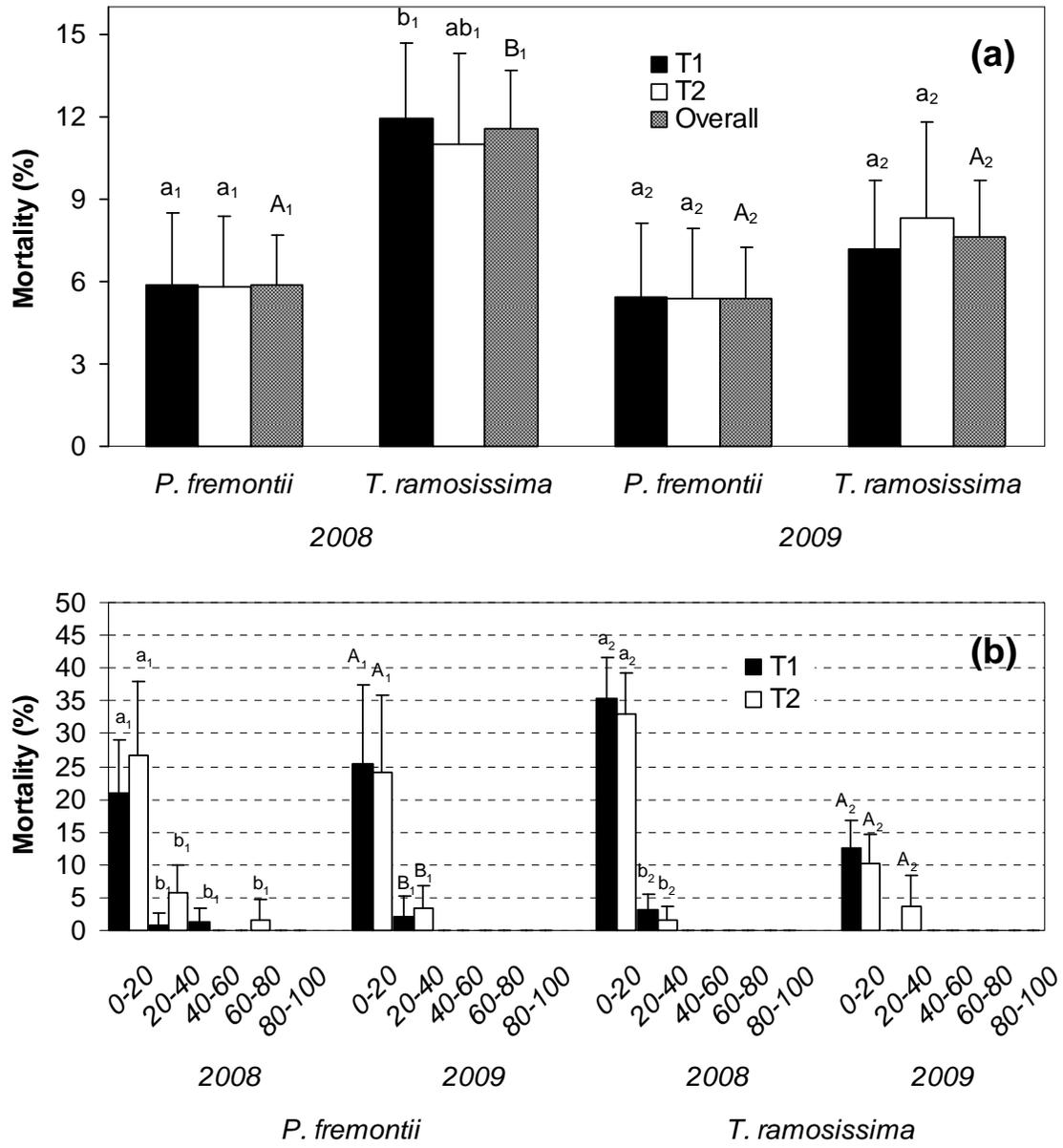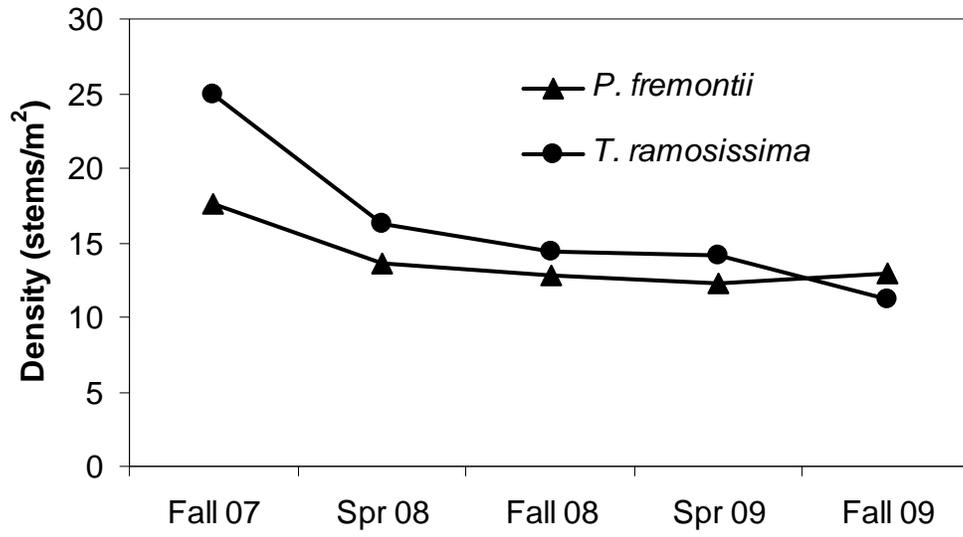**Figure 4.**

Figure 5.

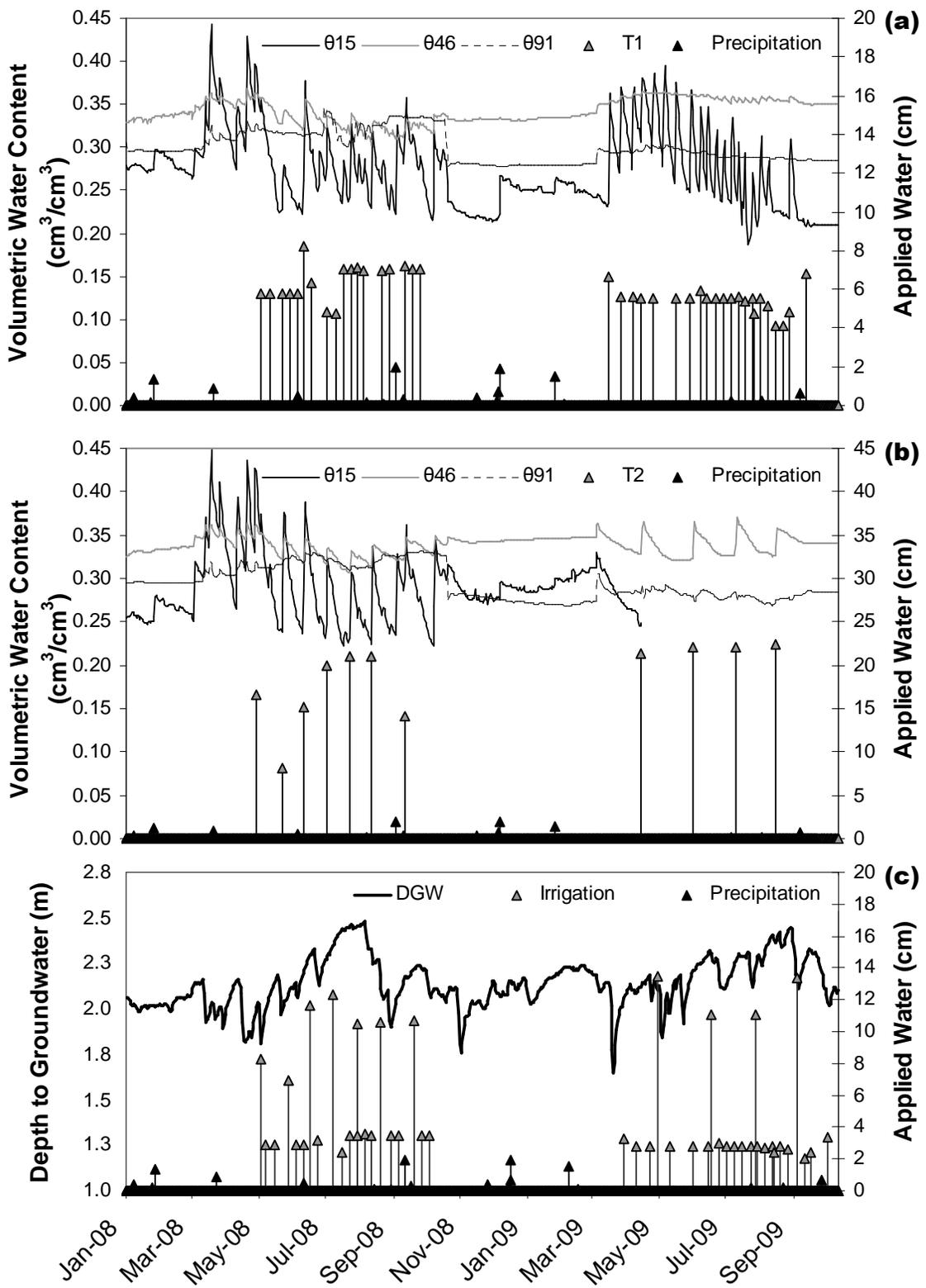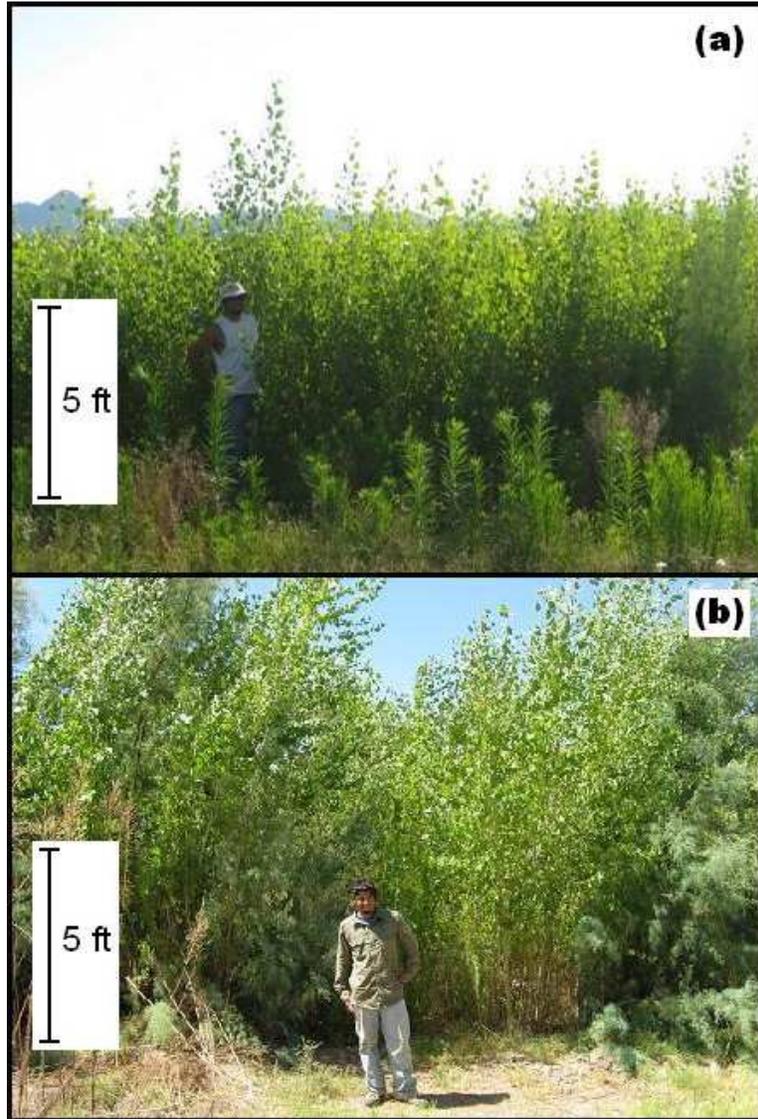**Figure 6.**

Figure 7.

Figure 8.

APPENDIX B: Using Delta-T SunScan for Leaf Area Index Sampling: Methods for Non-uniform Fields of Intermediate Height

Daniel P. Bunting

School of Natural Resources and the Environment, University of Arizona, Tucson, AZ 85721 USA

1. Introduction

Leaf area index (LAI) is the one-sided green leaf area per unit ground surface area. LAI is a key variable for global models of biosphere-atmosphere exchanges such as energy, carbon dioxide, and water vapor (Asner 2003), all of which occur at the leaf level. Models often use *in situ* LAI to scale-up measurements to larger landscapes. Hydrological processes are greatly affected by plant physiology and leaf area plays an influential role on the magnitude of impact.

Measuring LAI involves either direct or indirect sampling methods. Direct methods involve destructively sampling a portion or a complete plant/tree and calculating leaf area with a machine or manual method. Indirect methods, on the other hand, are non-destructive, but typically require calibration. Many indirect methods are available including allometry, remote-sensing, and Lidar, but LAI diffuse light meters/sensors are the most common due to their efficiency. For example, LAI-2000 (LI-COR Inc. Lincoln, Nebraska) AccuPAR (Decagon Devices, Inc. Pullman, WA), and SunScan (Delta-T Devices Ltd., Cambridge, UK) are all sensors that estimate LAI with a probe connected to a data terminal. This method measures solar radiation with optical methods and built-in statistical equations compute LAI. This method is the most efficient in the field as data is collected with a touch of a button, but requires validation by biomass sampling from each plant type. This paper focuses on the use of a Delta-T SunScan device.

LAI has many applications in watershed management. LAI data yield valuable information for tree-stand and crop health. Agricultural practices use LAI for measuring primary production and obtaining crop coefficients (Wilhelm et al. 2000). LAI data may also be beneficial for water budgeting, water conservation, base-flow management, and

groundwater monitoring because LAI is an important parameter for physical and biological processes associated with photosynthesis, respiration, transpiration, carbon and nutrient cycling, and rainfall interception (Chen and Cihlar 1998), all of which effect the hydrologic process. The utility of LAI measurements are site-specific and many methods are available depending on objectives.

2. Objectives

1) Measure leaf area index (LAI) in established vegetation plots
2) Assess effects of community composition (i.e. stem-density) on LAI within different levels of abundance and diversity
3) Measure LAI for whole trees and homogenous tree-stands to understand species specific properties
4) Use LAI to estimate water use (WU) for different vegetation associations across plots

3. Materials and Supplies

3.1 Field equipment
☐ Delta-T SunScan sensor with BF3 sensor and handheld Workabout
☐ large measuring stick (2.5 m) or clinometer depending on tree size
☐ calipers
☐ PVC frame (.25 m^2)

- ☐ hand clippers or saw (small, hand saw)

- ☐ Ziploc bags (large, 1-gallon)

- ☐ paper bags (large, grocery size)

- ☐ garbage bags (medium 13-gallon)

- ☐ garbage bags (large, 30-gallon)

- ☐ cooler (large, 150-quart)

- ☐ ice

- ☐ datasheet (Appendix A)

- ☐ pen

- ☐ black permanent marker

3.2 Laboratory equipment

- ☐ graph paper

- ☐ Xerox machine/Scanner

- ☐ drying oven (to 105C)

- ☐ scale (to one-hundredth of a gram)

- ☐ calculator

- ☐ computer (Delta-T software, MicrosoftExcel)

4. Set-up and Operation

4.1 Preliminary Settings

1) Name the Site and enter its Latitude and Longitude.

2) Set the Time and Date.

3) Set local time GMT (–12 to +13; e.g. USA eastern time = GMT -5), so AZ/MtnStd would be GMT -2 or GMT -3 depending on daylight savings.

4) Set the leaf Adsorption default to 0.85, although can be estimated 0.8 to 0.9. Adsorption is the amount of incident PAR absorbed and should not be changed from the default unless leaves of interest are thick, dark green or thin, transparent.

5) Set Ellipsoidal Leaf Angle Distribution Parameter (*ELADP*) to 1.0. This can also be estimated using $ELADP = (\pi/2)(N_h/N_v)$ where N is the number of leaves in the vertical or horizontal direction taken from a small, representative volume of the species of interest. Most crops have an ELADP of 0.5-2.0 and is defined by the ratio of projected horizontal leaf area to the projected vertical leaf area.

6) Set display format to LAI (this allows interaction between BFS sensor, if connected, in order to determine the sequence of readings).

7) Save configuration and set the type of file to be saved (e.g. if you want to work in a spreadsheet, choose "comma separated" instead of the default A:\DATA.PRN)

4.2 Set-up

Before taking the SunScan sensor to the field, the researcher must take into account the type of field settings they will encounter. This section describes two available options depending on the community composition and height of vegetation that you will be encountering.

For LAI measurements, the BFS (Unique Sunshine Sensor BF3) is connected via cable to the field probe, which is connected to the Workabout (handheld data terminal). The following three options are available for taking *in situ* LAI measurements:

4.2.1 Option 1: Set BF3 in open area

This requires an RS-232 cable long enough to connect the field probe, located at point of measurement, to the BF3 sensor, located in an open canopy unrestricted by vegetation. If vegetation is hindering the BF3 sensor in any direction, this may skew LAI values. Prior to LAI data collection, it is recommended to position the field probe in the open canopy near the BF3 sensor to ensure LAI reads a value near zero.

4.2.2 Option 2: Set BF3 above canopy

If open areas are not available, the BF3 sensor must be lifted above the canopy. To work in higher canopies, this requires innovation as an extension must be used to reach above the canopy. A wooden or aluminum post extension, for example, could be used to attach the BF3 sensor in order to raise it above the tallest tree. This will require at least 2 people to take readings (i.e. 1 person to hold the BF3 sensor above the canopy avoiding shadows and another to take the readings using the field probe with the Workabout in hand to label and store readings).

4.2.3 Option 3: Measurements without BF3

LAI can be estimated without the BF3 sensor connected, but this does not take advantage of the device's efficiency. This method involves using the SunScan probe as its own

reference point by taking an additional reading above the canopy before each

measurement, much like methods involving an LAI 2000. This method is also limited to

"steady-light conditions" (e.g. slow-changing sky conditions, blue sky, evenly overcast,

no intermittent cloud cover) and requires extra calculations after data is downloaded to a

spreadsheet.

4.3 Operation

After preliminary settings and set-up, measurements are ready to be taken.

Keeping the field probe level and ensuring that shadows do not hinder the readings, the

only responsibility for the researcher is to point and click. Readings are logged within

the Workabout and can be sorted and downloaded in the office or lab. The Delta-T

SunScan manual can be referenced if you want to label or add notes to individual

readings as measurements are taken.

It is recommended to design a sampling method that ensures both randomization

and replication. Any vegetation field sampling methods book will ensure a consistent,

replicable design. The rest of this paper outlines examples of detailed methods for LAI

sampling involving specific scenarios.

5. Field Example Samples

5.1 Proposed Methods for Sampling at Cibola National Wildlife Refuge

For detailed background information on past research at this field site, please refer

to APPENDIX A. The most efficient and valuable way to use the SunScan sensor for

measuring LAI at this field site is to use the existing quadrats as reference markers. One

of the objectives for LAI mapping at this study site would be to document what

independent variables may be significantly affected by available light resources.

Previous vegetation surveys at Cibola NWR have consisted of three randomized

permanent quadrats within each of 36 established plots (Figure 1). Each quadrat

documented average height, average growth rate, stem-density, and mortality over time.

By sampling LAI in each quadrat, analysis of variances can be computed to find

significant differences. In addition, LAI can be averaged over the entire study area to

compare values at this high-density restoration site to other studies using LAI. Methods

to achieve this objective are as follows:

1) Each quadrat in each plot will be systematically measured with the SunScan on

   the LAI setting (with BF3 attached, time of day should not matter)

   a. The 1st person will be responsible for positioning the BF3 in an open area,

      or if necessary, holding the BF3 above the canopy in such a way to avoid

      casting a shadow over the SunScan Probe below. In addition, the BF3 is

      leveled and positioned due north using the built in leveler and compass,

      which is necessary for setting the shade ring to the appropriate elevation.

   b. The 2nd person will record 3 readings using the SunScan and the

      Walkabout (Note** each reading will be given a unique ID/label; e.g. for

      plot 1, quadrat 1, reading 1 = "Group 1, Plot 1, Sample 1;" there is also an

      option to type a note to any reading).

i. All readings will be taken with the probe pointing northbound. The 1st reading will be taken along the north/south line along the east side of the quadrat, the 2nd along the center of the quadrat, and the 3rd along the west side of the quadrat (Figure 2). The distance between measurements is 0.25 m (Note** quadrats were originally oriented north/south to account for variation along 1-m furrows).

Note** LAI sensors measure light interception of both branches and leaves (plant area index, PAI) and must be calibrated: (1) repeat measurements can be done in the winter after leaf-drop to measure true LAI, or it can be regressed with destructive sampling

2) Direct LAI measurements will be taken to calibrate SunScan readings for homogenous stands of cottonwood and saltcedar trees as well as mixed communities that include other species. At least three samples of each vegetation association should be collected.

   a. Using the methods described above, the SunScan probe and BF3 will be used to gather additional LAI readings (Note** these samples differ from the quadrats as these trees will have to be destructively sampled).

   b. After sensor measurements, a 0.25-m^2 frame will be used to clip all vegetation above the center of the SunScan probe within an imaginary column.

c. All samples will be labeled and brought back to Tucson in a cooler for

   processing following Nagler et al. (2004; Note** all specimens should be

   processed within 24 hours of clipping to avoid inaccurate leaf size)

    i. A sub-set (i.e. the equivalent of filling five sheets of graph paper)

   of the fresh leaf biomass collected will be placed on graph paper of

   known dimension and scanned/Xeroxed (Figure 3; Note**

   scanning fresh leaves permits the calculations to be done anytime

   thereafter). The sub-set is then labeled and oven-dried at 105C.

    ii. Using the printed copy, all intersections (intersecting gridlines)

   that are not overlapped by foliage are counted and leaf area is

   computed by back-calculating (i.e. counting intersections gives

   you the total area not covered by foliage so this total is subtracted

   from the total area of one piece of graph paper to yield leaf area).

    iii. The oven-dried leaves are weighed to the nearest hundredth of a

   gram. The sub-set dry leaf biomass:leaf area relationship is used to

   estimate total leaf area collected in the frame.

    iv. LAI is calculated by dividing each sample's leaf area by the area of

   the frame used (0.25 m^2).

3) In the laboratory, the SunScan data will be downloaded into an Excel spreadsheet.

   The unique ID will allow data to be sorted or averaged within quadrats or plots

   which, depending on the desired objectives, will allow for efficient statistical tests

of significance. Tests may also be run to find disparity within and among

different species densities.

5.2 Proposed Methods using Allometry for LAI and calibration

Species specific allometry can be used to calculate LAI using multiple regression

equations. Depending on the objectives, single tree measurements can be scaled up to

whole tree or homogenous tree-stand estimates for LAI. Sub-samples from different

density associations could also be collected to correlate tree dimensions to leaf area in

order to scale-up to tree-stand LAI following Nagler et al. 2004. This could be valuable

when comparing total leaf area of trees within different densities where trees may vary

from traditional dimensions (i.e. trees in high-density stands tend to have smaller DBH

and greater heights than those in the open; Taylor et al. 2006). The following are

proposed methods to achieve these objectives:

1) For each tree destructively sampled, the following measurements are required:

    a. Basal diameter or diameter at breast height (DBH): This measurement is

       taken using calipers or, if the diameter is too large, a meter stick is used

       for the estimate. DBH is universally defined to be 137 cm above the

       ground surface (Figure 4).

    b. Height of the tree: A large 2.5-m measuring stick is used to measure

       smaller tree heights, but the use of a clinometer and measuring tape may

       be used to estimate larger trees. Using this method, a person will stand a

       known distance away from the tree trunk and use a clinometer to measure

       the slope/degree to the top of the tree (Figure 4). Simple geometry is used

to find total tree height: total tree height = [(distance to the tree trunk)/ (cosine(angle))] + (height from ground surface to observer's eye).

c. Canopy width: This is estimated by averaging north-south and east-west measurements (Figure 4).

d. Total number of leaf-bearing branches: This is a total count of all secondary branches (i.e. branches growing from main trunk/stem) that have leaves growing on them (Note** senesced leaves do not count because they do not contribute to transpiration).

2) A sub-sample of branches (5-15 or ~10% of total) are collected that are representative of the whole tree (i.e. branches with diameters of 0.1-2 cm representing the range of variability of leaf-bearing branches). The sub-sample is used to scale-up to the whole tree because assessing an entire tree is tedious and often times not feasible.

a. The diameter of each chosen branch is measured where it meets the main trunk. Then each branch is clipped, bagged, labeled, and transferred in a cooler until processed.

3) Sample Leaf Area is calculated

a. For each sample, three sub-sets of fresh leaves representing all leaf sizes are put onto graph paper of known dimension and scanned/Xeroxed.

b. All intersections that are not overlapped by foliage are counted and leaf area is computed by back-calculating. All three sub-sets are combined in order to have one representative leaf area for each sample. The sub-sets

are oven-dried at 105C and weighed in order to find the dry weight:leaf area relationship.

   c. All remaining leaves are oven-dried at 105C and weighed for each sample. The dry weight:leaf area relationship is used to scale up from sub-set to sub-sample, thus obtaining a total Leaf Area per sample

4) Whole-tree LAI is calculated

   a. Sample Leaf Area is scaled up to whole tree: (Sample Leaf Area) x (1-"percent sub-sampled"). Note the importance of choosing a representative sub-sample!

   b. Whole-tree Leaf Area is divided by the area of the canopy. (Note** the average of the north/south and east/west canopy widths is used to find the area of the canopy using $A=\pi r^2$)

5.3 Correlating LAI to Water Use

Overall water use can be estimated using LAI readings.

1) Water use will be estimated for each plot to compare vegetation associations and tree sizes (Methods are still a work in progress; Nagler et al. 2008 is the best reference thus far to relate LAI to WU).

2) Water use will be scaled up to estimate total water use by the small-scale study site.

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List of Figures

Figure 1. Sample 6-m x 12-m plot in which three quadrats were randomized along three randomized transects within each third of the plot. Each quadrat was positioned along the transect with the east/west sides facing away from the central irrigation piping and north/south sides perpendicular to furrows.

Figure 2. A sample quadrat with arrows showing the northward placement of the SunScan probe along the west side, center, and east side of the quadrat.

Figure 3. Fresh leaves are first overlaid on graph paper of known dimensions such that over half of the paper is covered with non-overlapping leaves and then is scanned or Xeroxed (Note: this picture is a demonstration using dry leaves, which were not carefully placed to avoid overlapping).

Figure 4. This is a conceptual figure for required field measurements including tree height, basal diameter, diameter at breast height, and canopy width. Total tree height (h) is calculated by summing h1 and h2, where h1 is found using a clinometer for the angle measurement and x is the measured distance from the observer to the base of the tree. Canopy width, needed to calculate canopy area, is measured by averaging the distance of canopy lengths extending along the north/south and east/west directions.
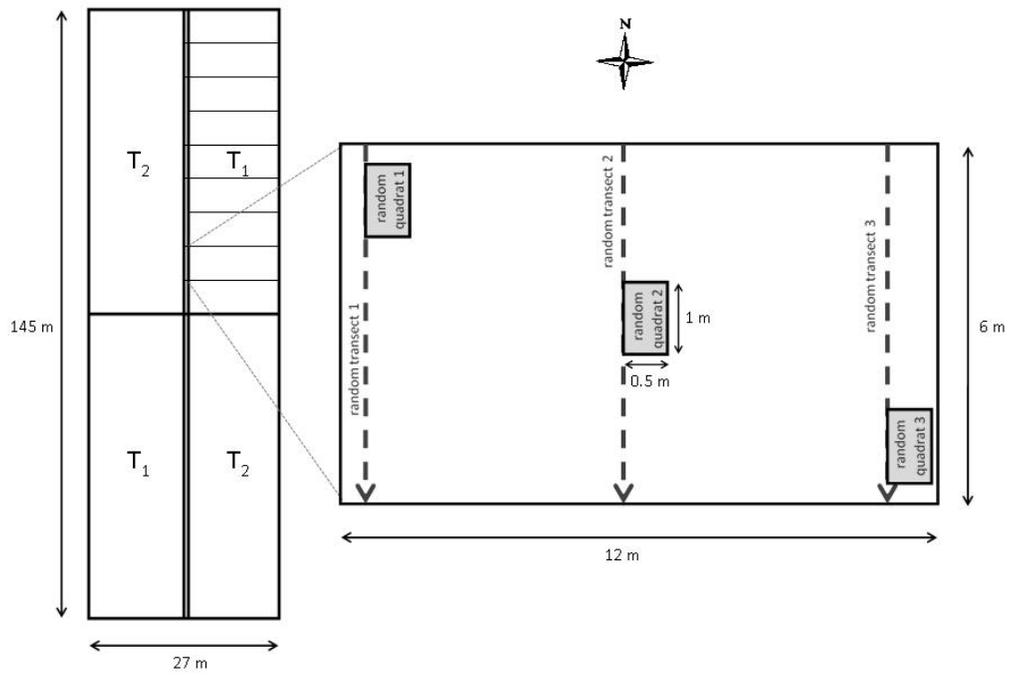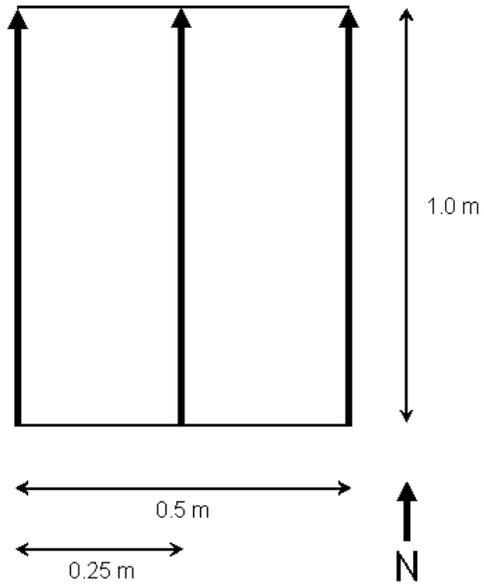
Figures

Figure 1.

Figure 2.

Figure 3.

Figure 4.

total
height

h

h1

angle

h2

x

137 cm

diameter at
breast height
(DBH)

basal
diameter

N