# THE EFFECT OF TREE SHELTERS ON SURVIVAL OF *SALIX* SPP. IN A SEMI-ARID AND HYDROLOGICALLY IMPACTED WATERSHED

A Thesis

Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Environmental Science

in the

College of Graduate Studies

University of Idaho

by

Allison Cathryne Lutes

Major Professor: Anthony S. Davis, Ph.D.

Committee Members: Andrew Nelson, Ph.D.; Romuald Afatchao, Ph.D.

Department Administrator: J.D. Wulfhorst, Ph.D.

May 2018

# AUTHORIZATION TO SUBMIT THESIS

The thesis of Allison C. Lutes, submitted for the degree of Master of Science with a major in Environmental Science and titled, “THE EFFECT OF TREE SHELTERS ON SURVIVAL OF *SALIX* SPP. IN A SEMI-ARID AND HYDROLOGICALLY IMPACTED WATERSHED,” has been reviewed in final form. Permission, as indicated by the signatures and dates given below, is now granted to submit final copies to the College of Graduate Studies for approval.

Major Professor: ________________________________ Date: ________________
Anthony S. Davis, Ph.D.

Committee Members: ________________________________ Date: ________________
Andrew Nelson, Ph.D.

________________________________ Date: ________________
Romuald Afatchao, Ph.D.

Department
Administrator: ________________________________ Date: ________________
J.D. Wulfhorst, Ph.D.

# ABSTRACT

Many riparian areas in the arid west have become highly altered ecosystems that no longer undergo the underlying physical processes and disturbances that were essential to their formation. With the de-coupling of riparian zones from these natural processes, the loss of complexity in riparian zones has become a widespread problem. Due to the rapid rate of decline, great efforts have been made in the last quarter-century to attempt to restore these altered riparian areas and their plant communities to natural conditions. However, many of these underlying physical processes (over-bank flooding, sediment deposition cycles, etc.) are now controlled by larger systems such as complex water infrastructure projects that cannot be easily un-done given the growing demand for resources. Therefore, novel restoration approaches are needed to aid riparian plant communities in overcoming challenging site conditions that are not conducive to plant survival.

This study tested the use of vented and unvented tree shelters on two species of willow (*Salix lasiolepis* Benth. and *Salix laevigata* Bebb) in a riparian zone within the Shasta River watershed in north-central California to determine the effect of tree shelters on survival. Dormant willow poles were planted into 0.92-meter augured holes and planted into the winter groundwater table. The results of this study indicate that the two shelter treatments varied in their effect on willow survival, and the effect differed between species. Red willows planted in unvented tree shelters were three times more likely to survive than red willows in the control treatment, indicating that this is a successful planting methodology that can improve survival for this species under similar conditions. In comparison, arroyo willows in either tree shelter treatment were not significantly more likely to survive than those in the control treatment. However overall survival for arroyo willows after 24 months was very high across treatments, suggesting that it is robust enough to tolerate the site conditions and that the investment in tree shelters may not be worthwhile for this species.

# ACKNOWLEDGEMENTS

This thesis would not have been possible without the professional guidance, vision, and support from the Shasta River Riparian Working Group: especially Dr. Ada Fowler, Dr. Chris Babcock, Sheri Hagwood, Karen Mallory, David Webb, Brooke Mejica, among others, who helped make this study possible through field work assistance, equipment loans, and early edits. I would like to express a special thank-you to Dr. Ada Fowler who has provided mentorship and friendship throughout this experience, and to whom I look up to greatly.

I am highly indebted to my primary advisor Dr. Anthony S. Davis who has been generous with both his time and knowledge and was willing to take a chance on advising a master's thesis from a distance. I also appreciate immensely the editing and coaching from Dr. Amy Davis-Ross at Oregon State University and Amanda Culley from the University of Idaho Statistics Center, who both provided invaluable feedback at a critical stage for the project.

Thank-you to Dr. Romuald Afatchao and Dr. Andrew Nelson for their review of this work, and guidance through the defense process.

Finally, I would like to thank my partner Matthew Lutes who has been a constant source of support and encouragement in all things, and who I respect and appreciate immeasurably.

# DEDICATION

I would like to dedicate this work to David Webb who has been a champion for the restoration of the Shasta River for many years, and who's unyielding dedication and commitment to the Valley has inspired me greatly both personally and professionally. Thank-you Dave for all that you have done and continue to aspire in the next generation.

# TABLE OF CONTENTS

# LIST OF TABLES

# LIST OF FIGURES

# 1: INTRODUCTION

The terrestrial ecosystem and habitat along streams and rivers, known as the riparian zone, range in width from confined regions along intermittent mountain streams to wide floodplains spanning entire valleys. Natural and intact riparian zones are the most diverse, dynamic and complex ecosystems on the terrestrial landscape (Naiman *et al.* 1992). In a riparian zone with intact natural processes, this inherent diversity and complexity is driven and formed by a natural and frequent disturbance regime such as flood events, debris flows, woody debris transport, fire, etc. (Nakamura 2000). Ecosystem disturbance is an essential driver of change that may destroy or restructure some of the biotic community while providing the essential conditions for the establishment of others (Nakamura 2000). Geomorphic and fluvial disturbances such as flooding can especially result in dramatic changes to riparian zones; often restructuring the channel form and function, the vegetation, soils, nutrients, water availability, and other processes, fundamentally changing the complexity and function of the riparian zone and ecosystem in a single event (Ward 1989, Nakamura 2000).

Riparian zones are tightly linked to their surrounding ecosystems through hydrologic and geomorphic connections as well as by the exchange of energy, nutrients, and plant and animal species (Naimen *et al.* 1992, Sobo *et al.* 2005). Healthy and intact riparian vegetation functions to shade streams for temperature control, provide woody debris for fish habitat and stream complexity, dissipate stream velocity, stabilize stream banks, filter out pollutants and sediments, and provide habitat and food for many terrestrial species (Naimen *et al.* 1992). Riparian vegetation in the floodplain serves to reduce downstream flooding by forcing the river onto its floodplain by slowing the velocity and encouraging energy dissipation (Patten 1998). Therefore, it is of little surprise that riparian ecosystems support unique and essential ecological functions and habitat on the landscape.

Despite only accounting for approximately 2% of the total land area in North America (Svejcar 1997), riparian zones have high species diversity and provide habitat for about one-third of all plant species (Poff *et al.* 2011). This productivity is especially evident in the arid and semi-arid west, where riparian zones with intact ecosystem function and complexity stand in stark contrast to the surrounding shrub-steppe upland ecosystems, which offer little diversity for the biotic community (Patten 1998). Consequently, it is estimated that 70% of all threatened and endangered species in the southwest are characterized as riparian obligates – relying upon the riparian zone for survival (Johnson 1989).

### *Riparian Habitat Threats*

While possessing great intrinsic ecological value, riparian zones also provide many beneficial uses for humans. Access to drinking water, water diversion for land use development such as irrigated agriculture, a source of food, transportation, recreation, and so on, have driven civilization and development to align along rivers. With the advance of agriculture and population growth in the western United States, there has been a significant increase in the use of riparian zones, often impacting the natural function, quantity, and quality (Poff *et al.* 2011). In the United States alone, it is estimated that 66% of riparian zones have been lost or altered to other land uses, primarily agriculture (Swift 1984), and that in the arid southwest this loss is estimated to be up to 95% (Brinson *et al.* 1981). For these reasons, riparian zones and their associated rivers have become among the most degraded ecosystems worldwide (Johnson and McCormick 1979).

The human-driven actions that have the most far-reaching effects are those that primarily involve changes in the physical processes- the hydrology and geomorphology of the riparian zones- which in turn have negative consequences on the biotic components. These actions can limit the natural function of the riparian ecosystem especially by preventing or limiting the timing, duration or intensity of the natural disturbance regime which drives productivity (Poff and Zimmerman 2010). Out of all the threats to riparian zones in the United States, the construction of dams and the land-use changes brought about by wide-spread agriculture can be considered to have the most comprehensive impacts to the function and processes of healthy riparian zones and their capacity to support robust riparian vegetation (Obedzinksi *et al.*2001, Stromberg *et al.* 2007, Stromberg 2001, Poff and Zimmerman 2010). Further, the scale and magnitude of the impact is far-reaching. In the United States alone, there are an estimated 84,000 dams that impound about 17% of all rivers (US Army Corps of Engineers 2016).

In the arid southwest, dams and irrigated agriculture are structurally integrated through complex water projects designed to capture winter snowmelt for the careful release through canals, aqueducts, and other infrastructure to be put to beneficial use throughout the hot, dry summer months. However, the demand for the resource far surpasses the natural supply. In California for example, the current surface water right allocations total 400 billion $m^3$, which is approximately five times the state's mean annual surface runoff (Granthem and Viers 2014). Or more simply: the human demand is 1000% greater than the natural supply for all of California's major river basins (Granthem and Viers 2014). The timing and demand for this water corresponds to the season when ecological demands are also high, underlying the crux of many modern resource- human conflicts (Barnett *et al.* 2005).

While the scale of impact varies depending on the size, purpose, and location of the dam, all dams affect at some magnitude both the upstream and downstream hydrology and geomorphology of a river and consequently its riparian zones (Braatne *et al.* 2007). Dams alter the four aspects of stream flow critical to the underlying fluvial geomorphological processes and forms that drive the composition of the riparian system: the timing of flow (i.e. the season in which flow occurs); the amount of flow; the duration of the flow; and the frequency of high flows (Magilligan *et al.* 2003). These stream flow components are all critical to the movement and deposition of sediment and nutrients, which create conditions for natural recruitment of plant communities in the riparian zone (Bendix and Hupp 2000).

The entrapment of sediment behind dams restricts the deposition of downstream sediment including gravels, silts, and sands that create new geomorphological features such as point bars, gravel bars, and islands (Stromberg 2001). These new sediment deposits brought in by episodic flood events are critical for riparian plant establishment (Bendix and Hupp 2000, Stromberg 2001) and especially for certain riparian species such as cottonwoods that rely upon moist silt, sand, or gravel bars in full sunlight along river floodplains for germination (OSU 2002). The construction of dams has been attributed to changes in riparian vegetation vitality and even the demise of species within watersheds (Stromberg 2001). On the lower Salt River and lower Verde River in Arizona, dam construction and the associated changes to the hydrograph was attributed as the primary cause for the extirpation of cottonwoods from the riparian area due to the lack of sediment deposition needed for cottonwood germination (Fenner *et al.* 1985, McNatt *et al.* 1980).

Additionally, water behind dams is often referred to as “hungry water” as it has additional energy as it is released but does not carry the sediment load to help dissipate this energy as it would in an undammed river system (Kondolf 1997). This excess energy can result in channel incision and downcutting of the streambed which is associated with disconnection from its floodplain, reduced frequency and duration of flooding, decreased sinuosity, and a lowered water table in the adjacent flood plain (Hall *et al.* 2011, Briggs 1996). These hydrological and geomorphological changes can have significant impacts on the riparian vegetation often stranding riparian vegetation from access to the lowered water table (Kondolf 1997, Bendix and Hupp 2000).

Another result of the steady increase in land-use changes associated with irrigated agriculture, dams, and lack of overbank flooding is the increase in salt accumulation in soils (French 1983, Reetz 1983). Salts originate in soils due to weathered bedrock and ancient saline sea-bottoms, and accumulate and redistribute through the soil profile due to water movement (Ogle 2010). Where rainfall is high, most salts are leached out of the soil. In arid regions, high salt levels can accumulate due to increased soil moisture evaporation, and reduced leaching from the lack of precipitation (Ogle 2010). Additional salts can be indirectly accumulated due to irrigation water or groundwater wells that are high in salinity (Ogle 2010). Although salts can have various impacts within the soil profile, the dominant effect of excessive soluble salts on plants is to hinder the ability of roots to absorb soil water even under wet soil conditions (NDSU 2014). Highly saline soils in riparian zones can result in changes to the species composition that can survive and can result in the dominance of invasive species such as *Tamarix* spp. that have a competitive advantage over native species in respect to salt tolerance (Vandersande 2001).

### *Rise of Restoration*

Due to the rapid decline of riparian ecosystems over the past century and an increased awareness in the value of riparian areas on the landscape, riparian conservation and river restoration has become a crucial issue for many federal, state and private organizations (Briggs 1996, NRC 1992, Kauffman 1997). A review of river restoration project records from the National River Restoration Science Synthesis (NRRSS) database from 1990 to 2003 found that $7.5 billion was spent on river restoration efforts in the US during this time (Bernhardt *et al*. 2005). This does not include any projects not reported in the NRRSS database, which is not a mandatory reporting model, or those projects which did not have project reporting costs (Bernhardt *et al*. 2005). Out of the significant investment in restoration, various river and riparian restoration models have developed over time generally representing two major schools of thought: classification versus process based (Doyle *et al.* 1999).

Contemporary restoration ecology models encourage a more holistic process-based restoration approach that would result in re-establishing un-altered rates and magnitudes of physical, chemical, and biological processes that can create and sustain the river and floodplain (riparian) ecosystem (Beechie et al. 2010, Cluer and Thorne 2013). However, not all river restoration projects have the ability in scope to address these physical, chemical, and biological processes or to overcome the complex resource needs (flood control, irrigated agriculture, recreation, etc.) that led to the alteration of the system in the first place. A 2005 review of 37,099 river restoration projects suggested that most river and riparian restoration projects are small in scale (implemented on less than 1 km. of stream length) (Bernhardt *et al.* 2005) which is realistic given that many low- elevation riparian areas in major river basins are privately owned, essentially fragmenting and driving the on-the-ground management approach to restoration (Everest and Reeves 2007).

In the southwestern United States, riparian restoration efforts began in the late 1960s supported by federal agencies such as the Bureau of Reclamation looking to compensate for water infrastructure projects (Busch *et al.* 1992, Taylor and McDaniel 1998). Since that time, riparian re-vegetation efforts have evolved using different techniques to overcome the challenges for growth in semi-arid regions where conditions for natural recruitment are poor. One technique commonly employed is to use drip irrigation to aid seedlings during the critical first year of establishment (Taylor and McDaniel 1998). The second method was first documented in New Mexico in 1985 by Swenson and Mullins (1985) and includes the planting of dormant willow and cottonwood poles augured at depth to access lowered water tables (Taylor and McDaniel 1998). Deep planting techniques have proven successful in the semi-arid southwest where dams and flow regulation have resulted in lowered water tables and water-stressed riparian zones (Dreesen and Fenchel 2008). By planting poles into the lowered water table or capillary fringe, plants can draw directly from the groundwater supply (Hall *et al.* 2011, Dreesen and Fenchel 2008). This technique has been favored in areas where soil conditions are too dry to use rooted seedlings, and the cost of irrigation is too great (Swenson and Mullins 1985).

Given the great effort required to establish riparian vegetation in altered sites, especially in the semi-arid southwest, the restoration community has developed different planting accessories such as caging, shelters, soil amendments, watering tubes, etc. aimed at improving survival. Tree shelters, for example, were first introduced in 1979 in the forest silviculture setting to protect planted seedlings from animal browse (Tuley 1985). Although not the original intention, it became apparent to early adopters that tree shelters were improving early seedling survival and growth for some tree species (Potter 1988). Subsequent studies and research indicated that the plastic tree shelters were mimicking a greenhouse environment that protected fragile seedlings during the critical establishment period (Tuley 1985, Devine and Harrington 2008). Non-vented, or solid wall shelters have shown to increase interior daytime air temperatures greater than ambient air temperature (Bellot *et al.* 2002), and vented tree shelters (those with holes or slits for aeration) were later introduced to moderate interior shelter air temperatures (Bellot *et al.* 2002). Tree shelters can also raise the humidity and carbon dioxide ($CO_2$) concentrations within the shelter (Kjelgren *et al.* 1997) due to seedling transpiration and reduced air circulation (Peterson et al 2005). Although results on tree shelter studies have varied with species and study environments, overall improvements in survival and growth during the first year of establishment have seen the rise of tree shelters in the restoration community (Hall *et al.* 2011).

# 2: STUDY OBJECTIVES AND CASE STUDY

In this experimental riparian planting study along the Shasta River in northeast California, the objective was to determine how two tree shelter types (a 3-foot non-vented Protex ProGro® and a 3-foot vented Tree Pro Max Grow Tube®) influence survival of two common native riparian species: the arroyo willow (*Salix lasiolepis* Benth.) and red willow (*Salix laevigata* Bebb). These species were selected because they are native to the Shasta River watershed and are commonly used in riparian restoration planting projects within the region. Red willow is a fast-growing riparian tree native to California and can reach 15 meters in height and 15 meters in width in 10 years with consistent water access (CNPS CalFlora 2018). Arroyo willow is a riparian shrub willow that spreads quickly by root runners in moist areas with a maximum height of 10 meters and 4.5 meters wide (CNPS CalFlora 2018). Both arroyo and red willows can tolerate a variety of soils if sufficient water is available with adequate drainage (CNPS CalFlora 2018). Two shelter types were chosen to assess whether there was a measurable treatment effect between a "vented" option and an "unvented" option when contrasted against a no treatment "control".

The application of tree shelters for the purpose of improving survival and growth of seedlings has been well studied, especially in oak species (McCreary *et al.* 2011, Devine and Harrington 2008, Chaar *et al.* 2008, Bellot *et al.* 2002) and conifer species used in reforestation such as western red-cedar (*Thuja plicata* Donn ex D.Don) (Devine and Harrington 2008) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) (Jacobs and Steinbeck 2001, Jacobs 2011). Few studies have focused on the application for riparian species in semi-arid climates. Hall *et al.* (2001) demonstrated significantly improved planting survival and average growth using a combination of tree shelters and deep planting techniques for cottonwood and willow species in semi-arid eastern Oregon. Although it is unclear whether survival was significantly greater due to the shelter treatment in plantings that did not penetrate the water table with deep planting techniques.

The objectives of this study were to determine the effect of two different tree shelters (vented vs. unvented) on arroyo and red willow survival and plant moisture stress (PMS) and to determine if groundwater drawdown in the Shasta River and associated planting elevation above river stage influenced plant survival.

## *The Shasta River: A Case Study*

The Shasta River in northeast California is chosen as a case-study as it represents the stressors and impaired functions of many semi-arid streams in the southwest. The Shasta River is dammed at river mile 34.6, is over-adjudicated for surface water withdrawals for irrigated agriculture, experiences impaired riparian function exhibited by lack of riparian diversity and complexity, has conflicting resource concerns, and is critical spawning and rearing habitat for the threatened Southern Oregon/Northern California Coast evolutionary significant unit of coho salmon (*Oncorhynchus kisutch)* (Walbaum 1972).

The Shasta Valley is a semi-arid high elevation desert with a warm-summer Mediterranean climate governed by its location within the extensive rain shadow of the Salmon and Marble Mountains, and the 4,321- meter stratovolcano known as Mount Shasta (NRC 2004). The Shasta River watershed is approximately 2,072 kilometers$^2$ and originates within the higher elevations of the Eddy Mountains lying southwest of the town of Weed in Siskiyou County, California and drains a portion of the Cascade province to the east and portion of the Klamath province to the west. The profile of the Shasta River is steep at its headwaters, followed by a large alluvial valley, and then a steep canyon reach before it joins the Klamath River.

The Shasta River exhibits characteristics of both “spring-dominated” and “rainfall/snowmelt run-off dominated” rivers (Mount *et al.* 2007). The hydrograph is manipulated by surface and groundwater irrigation withdrawals primarily during the irrigation season (1 April through 1 October, annually) allowing for irrigation of over 52,000 acres, and the impoundment of the Shasta River in Dwinnell Dam at river mile 34.6. During the low-flow summer period which corresponds with the primary irrigation season, the bulk of base flows of the middle Shasta River are primarily contributed by spring-fed tributaries such as Big Springs Creek (NRC 2004).

Early stream surveys conducted by the then California Department of Fish and Game documented that during the development of the lower Klamath and its tributaries including the Shasta River in the early 1900s, farmers and ranchers cleared much of the riparian vegetation and valley forests for agriculture (NRC 2004, CDFW 1934). In the nearby Scott River, it has been documented that early projects by the U.S. Army Corps of Engineers with the then Soil Conservation Service focused on removing riparian vegetation, draining wetlands, and constructing flood-control structures (NRC 2004). After the gold rush in the late 1800s, most of the land cover in the Shasta Valley was converted for agriculture and open range and although not as thoroughly documented as in the Scott Valley, it can be assumed the conversion to agriculture resulted in clearing of riparian forests (CDWR 1964). According to the 1983 Soil Survey of Siskiyou County, the original vegetation patterns of the Shasta Valley had undergone major change due to cultivation, excessive grazing, and changes to the hydrology and geomorphology of the region (USDA 1983).

The Shasta Valley also exhibits soils typical of the semi-arid south west with issues of high salinity and poor nutrient ratios. The Siskiyou County soil survey indicates that 46% of the Shasta Valley is composed of residual, old-valley filling, and glacial soils, which are shallow with low water-holding capacity and underlain by a hardpan (USDA 1983). Soils with restrictive layers such hardpans or claypans typically experience salinity issues because salts cannot be readily leached from the rooting zone, and therefore can accumulate on the surface (USDA- NRCS 2018). Only a small proportion of soils in the Shasta Valley are free from hardpan and harmful levels of alkali and are of favorable texture that can support crop growth (USDA 1983).

The recorded decline of coho salmon in the Shasta River has been attributed to the development of surface and groundwater diversions to support agricultural activities throughout the Shasta River basin (NRC 2004, Willis *et al.* 2013). The alteration of surface and groundwater hydrology for the purposes of flood-irrigation along the Shasta River has led to reductions in the quantity and quality of cold-water habitat required for salmonids, especially over-summering juvenile coho salmon. Prior to this extensive water development in the Shasta Valley, the river maintained a year-round baseflow of approximately 200 $ft^3$ $s^{-1}$ (NRC 2004) and the extensive cold-water spring system has been attributed as the primary reason for the Shasta River historically being considered one of the most productive streams of its size for anadromous fish in California (NRC 2004).

In 1997, the National Marine Fisheries Service listed the Southern Oregon/Northern California Coast evolutionary significant unit of coho salmon as threatened under the Endangered Species Act (ESA). This listing has resulted in an increased focus on the Klamath River basin and particularly within the Shasta River. The Shasta River is considered high priority within a restoration context because it is the last significant tributary for spawning and rearing coho salmon prior to the five Klamath River dams (NRC 2004).

Active riparian planting and restoration is required in the Shasta River riparian corridor because conditions for natural recruitment and seed germination are poor. The Shasta River is a spring-dominated system below a hydrologically controlled dam, which contributes to a lack of overbank flood events that prevent the natural recruitment of native riparian vegetation. The stage in the Shasta River drops seasonally, corresponding with the start of the irrigation season. Although each reach within the Shasta exhibits specific groundwater function and is characterized as a losing or gaining reach, in general it is expected the water table along the riparian planting corridor drops during the irrigation season, creating water stress on plants during the growing season. Finally, historical cattle grazing activity along the river has denuded much of the riparian corridor and left eroded stream-banks covered in primarily non-native pasture grasses.

## 3: METHODS

### *Experimental Site*

This experimental study was carried out along 167.6 meters of fenced riparian zone of the Shasta River (41.612, -122.476 Decimal Degrees at 772-meter elevation above sea level) within an active cattle ranch currently owned by The Nature Conservancy within the central Shasta Valley in northeast California. The experiment was conducted from November 2015 to November 2017. The NRCS Web Soil Survey classifies the soils at the planting site as Settlemeyer loam, with 0-2% slopes. Settlemeyer loams have alluvium derived from igneous, metamorphic and sedimentary rock parent material with effective rooting depths of ≥ 1.5 meters (NRCS 1983). Permeability of Settlemeyer soils are moderately slow with high available water capacity (NRCS 1983). Precipitation averages 0.3-0.45 meters $yr^{-1}$ and occurs primarily between the months of October and March (USDA AgACIS 2017), with an average growing season of 180 days (Mack 1960).

### *Experimental Design*

A set of 120 *Salix laevigata* poles and 120 *Salix lasiolepis* poles were distributed for planting amongst the 167.6-meter planting zone, and allocated among three treatments (vented tree shelter, unvented tree shelter, and a control with no shelter), with 40 experimental units for each species and treatment. Willow poles were harvested from healthy stands of willows within the Shasta River watershed, using techniques described in the USDA Technical Note Plant Materials No. 23 (USDA 2007). All poles were planted on 4 December 2015 after the 1.8- to 2.4-meter-long willow poles had been cut, stripped of all lateral branches, and completely submerged in water for 12 days to aid transplant survival and improve stem water content (Schaff *et al.* 2002). Planting locations for treatments and species were randomized using a random number generator, and plants were spaced a minimum of 0.9 meters apart to reduce competition for resources. The poles were planted into 0.9-meter-deep auger holes and the holes were filled with a soil-water mixture to ensure good soil to stem contact.

A 1.2 – meter welded wire cage with a 0.46 – meter diameter was placed around each pole and secured to a 1.8 – meter metal t-post with metal cage fasteners. Control (treatment 0) poles received the welded wire caging only. Vented tree shelter (treatment 1) poles received the welded wire caging and a 0.9 – meter Tree Pro Max Grow Tube (Tree Pro, West Lafayette, Indiana) made from Polyethylene plastic with a diameter of 0.1 – meters and with holes or “vents” that begin halfway up the tube. Non-vented tree shelter (treatment 2) poles received the welded wire caging and a 0.9 – meter tall Protex® Pro/Gro Solid wall tree shelter (TerraTech, Eugene, OR) with a diameter of 0.1 – meters constructed out of blue colored type 2 recyclable polyethylene with no vents. Each cage was fastened with a metal identification tag with a unique number identifier.

### *Survival*

Each experimental planting was monitored in spring and fall of 2016 and 2017 for survival (alive or dead) and height. Plants were classified as alive if they had signs of growth such as live shoots and live leaves. Winter flood events eroded a portion of the study site, and 12 of the 240 plantings were swept downstream. This data was removed from the overall survival counts. Poles that were marked as dead but re-sprouted from the base of the original pole at a later monitoring period, were re-classified as alive. Height data was not analyzed due to the high number of re-sprouts that created challenges in comparing growth over time.

Survival was analyzed at the individual plant level using a binary logistic regression with shelter treatment as a categorical covariate using Minitab® 18.1 Statistical Software. Separate analyses were completed for the arroyo willows and the red willows due to the high variability in survival, and between years by looking at survival in fall 2016 and survival in fall 2017 separately.

### *Temperature*

Paired Onset Hobo Temperature loggers (Model: Hobo UTBI-001, Onset, Bourne, MA) were installed inside randomly selected shelters of each treatment type at the bottom (0.15 meters above ground), middle (0.46 meters above ground), and top (0.91 meters above ground) of the shelter. A temperature reading was taken every 30 minutes for a period of 21 months (February 2016 to October 2017). Control temperature loggers were not sufficiently protected from the effects of direct solar radiation and deemed erroneous when compared to ambient air temperature readings from nearby temperature stations (Weed airport, Montague airport, and

Big Springs Creek Meteorological Station). Therefore, ambient air temperature serving as the control was collated from a Meteorological Station located (2.6 miles away) on an adjacent ranch owned by The Nature Conservancy. From this data, daily maximum and daily average air temperature by treatment were determined and graphed by month during the primary growing season.

### *Elevation and Depth to Groundwater*

Three groundwater wells were installed at the site spread within the width of the planting zone at 772.6 meters, 772.7 meters and 773.2 meters above mean sea level (MAMSL). Stilling wells were created using 7.6 cm diameter PVC pipe with a well cap on top, and holes were drilled using an auger with a 7.62 cm diameter drill bit. Onset HOBO U-20 water level loggers (Onset, Bourne, MA) were installed in each stilling well and attached by cable to the well caps. Water level loggers were set to record absolute pressure (psi) every 30 minutes. To compensate for barometric pressure change, a separate above ground HOBO U20 Water Level Logger was used for reference. Data was processed in HOBOware using the Barometric Compensation Pressure Assistant with the data from the stilling wells and the above ground barometric pressure to determine water level below ground.

The well at 773.2 MAMSL was not drilled deep enough and went dry during the months of July and August in 2016. The battery in the piezometer in the well at 772.7 MAMSL failed part-way through 2017. Data was utilized from the well located at 772.6 MAMSL as it remained in water throughout the study period and tracked closely with spot checks, and with the river stage data. Groundwater data was averaged for each day and graphed over time for the study period.

Elevation measurements were taken at ground level of each pole planting, at ground level for each piezometer, and at five increments and then averaged for surface water elevation of the Shasta River using a Trimble AgGPS 442 GNSS receiver Real Time Kinematic (RTK) with +/- 2.54 cm pass-to-pass and year-to-year repeatable accuracy (Trimble, Sunnyvale, CA). Stage data was collected from the California Data Exchange Center for the nearest flow gage (SPU – Shasta River at Grenada Pump Plant) located approximately 0.6 kilometers upstream of the experimental planting site. Using the stage data on an hourly increment, data was correlated to the known surface water elevations at the planting sites and the change in elevation between the gage and the planting site was used to determine the river stage at the planting site throughout the study period.

### *Soils*

Soil samples were analyzed for organic matter, pH, and nutrient composition. Two representative soil samples were analyzed from a composite of 10 sub-samples. Sub-samples were systematically collected every 16.2 meters across the entire 161.5-meter study area from holes drilled by a mechanized auger at depths of 0.3 and 0.6 meters. Sampling depths were chosen based upon the effective rooting zone. All sub-samples from the same depth were placed in a clean container and mixed thoroughly. Once mixed, a sample from each depth was analyzed by standard laboratory methods at A & L Western Agricultural Laboratories (Modesto, CA).

### *Plant Moisture Stress*

To measure the demand for water within the willow pole plantings, a series of plant moisture stress (PMS) tests was conducted during characteristically hot weekends in July over the two-year study period (July 23 and 24, 2016 and July 28, 2017) using a Scholander Pressure Chamber Model 600 (PMS Instrument Company, Albany, OR). Newer-growth leaves were collected from randomly selected individual willows by cutting the petiole of a single healthy leaf close to the top of the pole, ensuring there was enough petiole to fit within the pressure chamber. For each leaf, a clean cross-sectional slice was made across the petiole of the leaf and the leaf petiole was fit into a rubber stopper that attached to the pressure chamber. The leaf was placed within the pressure chamber with the petiole sticking out of the pressure chamber within view of the operator. Pressure was slowly increased within the chamber until exudate was observed with a hand-lens being forced out of the tip of the petiole, at which point pressure was stopped and the associated pressure (in bars) was recorded. PMS measurements were performed before dawn when stomata were closed and water in the leaf is a function of available soil moisture (McNiesh 1988). Measurements were repeated on the same sub-sample during solar noon when the solar radiation and PMS are at a maximum. Mortality from 2016 to 2017 reduced the sample size per treatment in 2017.

The PMS data was analyzed using a generalized linear mixed model (PROC GLIMMIX) using SAS/STAT® version 9.4 (SAS Institute Inc., Cary, NC) by comparing pressure at solar noon and pre-dawn in 2016 and 2017 between species and among treatments. The data was graphed as medians, with quartiles 1 and 3 because assumptions of normality were not met. The sub-population data was skewed, likely influenced by the small sub-sampling populations (<4 in some sub-sets).

# 4: RESULTS

## *Survival*

Overall survival after 24 months for the red willows was 42% (16/38) in the unvented treatment, 29% (11/38) in the vented treatment, and 18% (7/39) in the control treatment (Table 1). Overall survival after 24 months for the arroyo willows was 89% (32/36) in the unvented treatment, 82% (32/39) in the vented treatment, and 90% in the control treatment (34/38) (Table 1).

After the first winter (as assessed in spring 2016), there was 30% (35/115) mortality across all treatments for red willows, compared to only 2% (2/113) mortality across all treatments for the arroyo willows (Table 1). Additional mortality during summer 2016 (as assessed in fall 2016) reached 48% (38/80) for red willows across treatments, and only 9% (10/111) for arroyo willows across treatments (Table 1). Additional mortality over the second winter (as assessed in spring 2017) was 19% (8/42) across all treatments for red willows, compared to 2% (2/101) across all treatments for arroyo willows (Table 1). There was no additional mortality in summer 2017 for red willows (0/34) but 1% (1/99) mortality was observed for arroyo willows (Table 1).

Binary logistic regression analyses indicated that tree shelter treatment and depth to groundwater did not explain much of the variation in survival of the arroyo willows in 2016 or 2017 (Table 2). Depth-to-groundwater was not a significant factor affecting survival of red willows, though tree shelter was a significant predictor of red willow cumulative survival in 2017. Specifically, red willows planted in unvented shelters had approximately 3 times greater odds of survival (odds ratio = 3.1405, 95% CI = 1.0912, 9.038, Table 2). There was no significant increase in likelihood of red willow survival in 2017 associated with vented shelters relative to control.

## *Plant Moisture Stress*

Pre-dawn PMS measurements in 2016 were similar across all treatments with a median range from 1.5 to 1.8 bars for arroyo willows and 1.5 to 2.3 bars for red willows (Table 3). Median PMS values for arroyo willows at solar noon in 2016 were comparable across all treatments: 8.0 bars (interquartile range (IQR) = 5.1 - 10.7) for control, 8.5 bars (IQR = 7.0 - 10.0) for unvented, and 9.5 bars (IQR = 7.0 - 9.9) for vented treatment. PMS values for red willows at solar noon in 2016 were higher than the arroyo willows: 10.5 bars (IQR = 9.3 - 12.4) for control, 10.0 bars (IQR= 8.5 - 12.0) for unvented, and 10.0 bars (IQR = 8.6 - 12.8) for vented.

Median pre-dawn measurements in 2017 were more varied for arroyo willows than in 2016 and ranged from 2.4 – 3.4 bars across all treatments. Median PMS values for red willows ranged from 2.1 - 2.6 bars in 2017 (Table 3). At solar noon, readings in 2017 were higher for both species than in 2016. Median PMS values for arroyo willows at solar noon in 2017 were comparable across all treatments: 14.8 bars (IQR = 13.0 - 15.6) for control, 13.4 bars (IQR = 12.4 - 15.0) for unvented, and 14.8 bars (IQR = 14.5 - 15.3) for vented treatment. The median PMS values for red willows at solar noon in 2017 were slightly lower than the arroyo willows for the control treatment with 13.3 bars (IQR = 11.8 - 15.0) and for the vented treatment at 13.6 bars (IQR = 12.8 - 14.8), but higher for the unvented treatment at 14.0 bars (IQR = 13.0 - 17.2).

The results of the generalized linear mixed model indicated that arroyo willow had significantly lower pressure at solar noon in 2016 compared to red willow (F = 5.87; p = 0.0185) (Table 4). No other significant differences were found for either species in either year when comparing either pre-dawn and solar noon pressure among treatments (Table 4).

### *Treatment Effect on Temperature*

Differences in mean monthly and maximum daily temperature by month were observed between vented and unvented treatments, between both treatments and ambient air temperature, and at the different monitoring positions within the shelters: bottom (6” above ground), middle (18” above ground) and top (36” above ground). The ambient mean monthly air temperature during the study period peaked in July and August, averaging 35.5 °C in 2016 and 37.2 °C in 2017 (Tables 5 and 6). For each month of the primary growing season (April – September) in both 2016 and 2017, the mean monthly and maximum daily air temperatures were greater within the vented (at all three monitoring positions) and the unvented (at the middle and top monitoring positions) tree shelters when compared to the ambient air (Tables 5, 6, 7 and 8). The vented shelter also had higher mean monthly and maximum daily temperatures by month at all monitoring positions for all study months when compared to the unvented shelter except for the months of March and April in 2016 (Figures 4 and 5).

During July and August, the average mean monthly air temperature for the bottom position in the vented treatments was 5.7°C greater than ambient air temperature and 6.4°C greater than the unvented treatment in 2016 (Tables 5 and 6). This trend continued in 2017, where the mean monthly air temperature for the bottom position in the vented treatment was 1.9°C greater than ambient air temperature and 2.1°C greater than the unvented treatment (Tables 5 and 6). The greatest temperature difference between any shelter type or shelter position versus ambient air temperature occurred in September of 2016 where the maximum daily air temperature for the vented shelter in the 'top' position was reported as 16.3°C hotter than the maximum ambient daily air temperature for the month (Table 6).

### *Depth to Groundwater*

In 2016 groundwater levels and river stage were both lower when compared to 2017. Unlike 2016, which marked the fifth year of drought in California, 2017 was an above-average water year, with significant over-bank flooding during February, March, April and May (Figure 6). Average river stage in 2017 was 0.31 meters higher in February, 0.25 meters higher in March, 0.30 meters higher in April, and 0.24 meters higher in May when compared to the same gage in 2016 (CDEC 2018).

In 2016, the groundwater dropped below the 0.91-meter average planting depth for the majority of the 2016 late-winter/spring and summer, only rising a significant amount (> 0.15 meters) once during a storm event in March (Figure 7). Groundwater drawdown (cm/day) was calculated during the primary growing season for days that it dropped below the average planting depth. In 2016 there were 31 days where drawdown occurred at a rate of 1-3 cm/day, 8 days where it fell 3-6 cm/day and 2 days it fell ≥ 6 cm/day (Table 9).

For the entire 2017 winter and spring, groundwater levels were higher than the average planting depth (0.91 meters, calculated from the average of elevations for all plantings). Groundwater only dropped below the 0.91-meter averaged planting depth at the end of June in 2017 and remained below planting depth for the remainder of the summer (Figure 8). There were no days in April, May or June of 2017 where groundwater drawdown exceeded 1 cm/day and was below the average planting depth. In July there were 10 days with drawdown rates of 1-3 cm/day, 4 days with 3-6 cm/day, and 2 days in August with 1-3 cm/day. There were no days in 2017 where drawdown rates were ≥ 6 cm/day (Table 9).

Depth to groundwater was not a significant factor affecting survival of arroyo or red willows across all treatments (Table 2).

### *Soils*

Soils at the site were characterized as loamy sand at the 0.3-meter depth (80% sand, 12% silt and 8% clay content) and sandy loam at the 0.6-meter depth (76% sand, 14% silt, and 10% clay content). Soils at both sampling depths were alkaline, low in organic matter, low in essential macronutrients (N, P, and K), and high to very high in salts (Mg, Ca and Na) (Table 10; Hangs *et al.* 2011). Macronutrient concentration decreased with increasing soil depth, as did the concentration of Mg and Na, but the concentration of Ca was greater at 0.3 meter compared to 0.6 meter.

# 5: DISCUSSION

The results of this study indicate that the two shelter treatments varied in their effect on willow survival, and the effect differed between species. Red willows planted in unvented tree shelters were three times more likely to survive than red willows in the control treatment, indicating that this is a successful planting methodology that can improve survival for this species under similar conditions. In comparison, arroyo willows in either tree shelter treatment were not significantly more likely to survive than those in the control treatment. However overall survival for arroyo willows after 24 months was very high across treatments, suggesting that the species is robust enough to tolerate poor site conditions and therefore the investment in tree shelters may not be worthwhile for this species.

Vented tree shelters did not have a significant impact on survival for red willows or arroyo willows. This could be due to the high daily maximum temperatures seen in the vented shelters during the 2016/2017 growing seasons. Temperatures in the vented shelters at times exceeded 50 °C and averaged 10.5 °C hotter than ambient air temperatures and were on average 6.6 °C hotter than unvented shelters from April - September at the mid-shelter position. Although certain studies have shown that tree shelters can improve microclimate conditions inside the shelter by reducing transpiration and soil moisture depletion rates (Devine and Harrington 2008), other studies have shown that increased temperatures inside the shelter can become deleterious to plant survival if there is not sufficient water access (Kjelgren and Rupp 1997, Oliet and Jacobs 2007). The results of this study indicated that the 'vented' shelter had higher daily maximum temperatures when compared to both ambient air temperature and 'unvented' treatment shelters. This is an unexpected result, as 'vented' shelters were introduced as a means of over-coming high air temperatures inside tree shelters. Additional temperature monitoring in a controlled greenhouse is recommended to better characterize these findings.

The results of this study can be compared to historic survival data in the Shasta River watershed. Historic riparian planting survival in the Shasta River watershed is considered relatively poor in comparison to statewide standards for riparian plantings projects, that mandate by certain state agencies an 80% survival result after 5 years (Griggs 2009). A 2008 study of 10 historical planting sites in the Shasta River found that survival was very low in most sites with 9 out of the 10 sites having survival less than 50%, and 5 out of the 10 sites have survival less than 10% (Mattson 2008). Sites examined in that study had been planted over a period of years and efforts, and it was hypothesized that survival improved by river mile, with those lower in the basin to have higher planting survival (Mattson 2008). The soils in the lower Shasta basin as it approaches the canyon reach before meeting the Klamath exhibits less hard-pan, which is hypothesized as a contributing factor to the differences in survival (Mattson 2008). In 2012, The Nature Conservancy (TNC) reported survival results from planting approximately 6,000 riparian trees and shrubs on roughly 10 miles within the Shasta River watershed. TNC reported a 3-year cumulative survival of 37% for red willow poles and 79% for arroyo willow poles (Fowler and Babcock 2012). This can be compared to the control treatment cumulative 2-year survival from this study of 18% for the red willows poles in the control treatment and 90% for arroyo willow poles. In both studies the observed cumulative survival varied greatly between the two species, with high survival rates found for arroyo willows (Fowler and Babcock 2012). Favorable spring rains and wet winters in 2010 and 2011 were considered contributing factors affecting survival for the TNC study (Fowler and Babcock 2012).

In addition to different survival response by species, year-to-year survival varied greatly in this study. Mortality for red willows was 30% over the first winter after planting which might be considered high when compared to the 4% mortality for arroyo willows. Survival over the first summer was also poor for red willows with an additional 48% mortality. By comparison, both red willows and arroyo willows did very well over the second summer: with no additional mortality for red willows or arroyo willows in both shelter treatments, and only 1 arroyo control mortality. Poor survival in the first year might be expected as it is the most critical time for physiological stress such as high moisture stress occurs for most plants in the first year of establishment after transplanting (Cleary 1980). Although both seasons experienced approximately the same amount of precipitation over the primary growing season (< 7.6 cm total), the spring hydrographs and corresponding groundwater table levels varied greatly.

The first winter of the study period after transplanting, the region was experiencing year five of persistent drought conditions. This is reflected in the river stage and groundwater levels, which during the critical winter months, showed relatively low groundwater levels and very little over-bank flooding or bank-full conditions at the project site. In January, February, and March of 2016 the average river stage at the project site was 0.25 meters lower than the same time-period in 2017 (CDEC SPU 2018). Corresponding to the lowered river stage and lack of peak winter storms, the groundwater table during the 2016 winter was much lower when compared to 2017. The average groundwater depth during the winter months of February and March in 2016 was at 771.9 meters, or 0.98 meters below the average ground elevation at the planting site. These lowered groundwater depths did not reach such lowered levels in 2017 until the hot and dry summer months of June and July. Further, in 2017 the average planting depth elevation (calculated as the average of the elevations of all the plantings at the site and assuming a 0.91-meter planting depth) at the site would have been fully submerged from January through the end of May. In comparison, the average planting depth in 2016 was submerged or partially submerged for only 19 days from February through the end of May.

In addition to lowered stage and ground-water levels in 2016, there was more rapid groundwater draw-down rates observed in 2016 when compared to 2017. Overall there were 10 days in 2016 where the groundwater draw-down rates were ≥3 cm/day; 2 of those days rates > 6 cm/day. In comparison there were 4 days in 2017 where groundwater draw-down rates were ≥ 3 cm/day, with 0 days > 6cm/days. Very little research exists that calculates specific draw-down rates that impact survival for willow poles planted in auger holes. Generally, a steady water table decline encourages shoot and root growth in plants, whereas a rapid drawdown rate is associated with reduced growth and even mortality (Amlin and Rood 2002). Swenson and Mullins (1985) found that willow poles planted in deep-augured holes in naturally fluctuating water tables had lower rates of survival when compared to those in plots with constant water tables, and that water table fluctuations of < 60 cm over the primary growing season was ideal for pole planting survival in their simulated trials.

Stella *et al.* (2010) examined the effect of simulated water table recession on willow (*Salix gooddingii* C. Ball and *Salix exigua* Nutt.) and cottonwood (*Populus fremontii* S. Watson ssp. *Fremontii)* seedlings and found that drawdown rate had a strong influence on seedling mortality. They reported no survival for seedlings subjected to drawdown rates that exceeded ≥ 6 cm/day, only 12-38% survival for seedlings subjected to 3 cm/day drawdown rates, and comparable survival % to their control seedlings for 1 cm/day drawdown rates (Stella *et al.* 2010). These findings are generally supported by Amlin and Rood (2002) who found that willow and cottonwood seedlings can tolerate drawdown rates as high as 1-2cm/day, though more abrupt declines of > 2 cm/day reduced growth and survival. Although the number of days that experienced drawdown rates ≥3 cm/day during the primary growing season in 2016 and 2017 is not particularly high, there is concern in the number of consecutive days where drawdown rates were ≥ 3 cm/day. In 2016, there were periods in March, April, and July where drawdown rates ≥ 3 cm/day for 4 days consecutively. In 2017, there was a period in July where draw-down ≥ 3 cm/day for 7 days consecutively. Repeated days with draw-down rates ≥ 3 cm/day are likely to intensify the water stress on the willows, relative to if these days were more isolated and spread out over the growing season. The difference in the ability of red and arroyo willows to tolerate different draw-down rates has not been specifically researched, though Stella *et al.* (2010) found variability in tolerance to draw-down rates between the two willow species studied.

The poor water year in 2016 resulted in different levels of water stress between the two willows. One factor contributing to the poor survival of red willows in 2016 was increased plant water stress. The results of the PMS testing indicated that the arroyo willow had significantly lower pressure at solar noon in 2016 compared to the red willow. While a significant difference was found between species, we failed to reject the null hypothesis that treatment affected plant moisture when compared to the no treatment effect. The failure to reject the null hypothesis however does not statistically indicate that no different exists between PMS values from the various treatments and control, and the low sub-population size for plant moisture stress values in 2017 (n>4) due to mortality may have affected the power of the analysis.

Species-specific PMS thresholds do not exist for red or arroyo willows, though general guidelines indicate that as PMS values increase from 5 to 17 bars, the plant experiences greater moisture stress that limits growth ability and photosynthesis, and cell elongation and phloem transport can become reduced (Tyee and Hammel 1972, Cleary 1980). As pressure increases from 20 to 100 bars, most plants will die, and at what pressure and for how long is highly dependent upon the plant and its adaptations (Tyee and Hammel 1972, McNiesh 1986). Some plants were more stressed than others and the maximum PMS value observed for red willows was 17.5 bars in 2016 and 20.3 bars in 2017. By comparison the maximum PMS value observed for arroyo willows was 17.5 bars in 2016 and 18.1 bars in 2017.

PMS values change on a daily basis due to the transpiration demand on the plant and soil (McNiesh 1986) and are greatly impacted by localized weather such as clouds, fog, etc. Therefore, values cannot be easily compared between the years, but can be compared between species within years. Overall, the median results for both species do not indicate that the plants during the sampling period in July were experiencing extreme plant water stress (>20 bars). Further research into drought tolerance and PMS limits for willows could indicate at what PMS value arroyo and red willows begin to experience limited growth ability and lasting physiological harm.

Although both species of willows were subjected to the same poor soil conditions, water drawdown rates, and available soil moisture, physiological differences and adaptation to these conditions is likely to explain the different response and survival rates. Soils at the sampling site tested as alkaline, low in organic matter, low in essential macronutrients (N, P, and K), and high to very high in salts (Mg, Ca and Na). High soil salinity can affect plant growth both physically by increasing the osmotic potential of the soil solution, so the plant needs to use more energy to absorb water, and chemically by limiting nutrient uptake resulting in toxicity (Ogle 2010). Typical plant response includes a progressive decline in growth and yield as salinity levels increase (Magistad 1945). The primary effect of excessive soluble salts on plants is to limit the ability of plant roots to absorb soil water even under wet soil conditions (Ogle 2010).

Salt tolerances for various plant species have been studied and tolerances can vary greatly within a genus and within a species. Hangs *et al.* (2011) studied the salinity tolerance of 37 different native and exotic willows to determine their suitability for use on saline agriculture lands. Their results indicated that while most of the varieties could tolerate moderately saline conditions, some varieties showed no reduction in growth with severe salinity, indicating great variability of salt tolerance within the *Salix* genus. Very little research has been conducted on the specific salt tolerance of red willows and arroyo willows. Kauser (2010) tested the salt tolerance of red willows during the dormant season and during the primary growing season and found that red willows had very low salt tolerance overall across multiple testing thresholds. The apparent salt tolerance was even lower when conducting the experiment during the primary growing season in a greenhouse where the author suspected the plants were also experiencing heat stress (Kauser 2010).

Previous studies have demonstrated that intra-species variation of salt tolerance can exist between sub-populations of a given species such as *Pinus, Acer,* and *Eucalyptus* (Allen et al 1994) and it is dependent upon the species ability to adapt to conditions over time. Ferrus-Garcia (2003) found that salt tolerance varied within sub-populations of arroyo willows by testing salt response to willows from various eco-types and sites. Those willow cuttings taken from the beach along the Pacific Ocean, showed greater tolerance to moderate salinity levels when compared to other sub-populations found more inland (Ferrus-Garcia 2003). Further, it was found that the salt tolerance in the arroyo willow could be enhanced with gradual exposure over time (Ferrus-Garcia 2003). Overall the arroyo willows had moderate to good salt tolerance (Ferrus-Garcia 2003), more research is needed to verify these results.

Although this study did not specifically measure salt accumulation levels within the red and arroyo willows; the highly saline soils, the variability in salt tolerance of arroyo and red willows, and the variability in the survival rates under the same challenging growing conditions indicate that the arroyo willows in this study are likely to be more salt tolerant than red willow. Further, the results of Ferrus-Garcia (2003) suggest that more research is needed to determine whether the sub-population of arroyo willows within the Shasta River has developed a greater tolerance to salts over successive generations. Further investigation to understand the nuances of this potential tolerance are required, although Ferrus-Garcia (2003) and the findings from this study suggest that specific populations of arroyo willow may be better suited for revegetation of salt-contaminated riparian sites within the Shasta River.

# 6: CONCLUSIONS AND RECOMMENDATIONS

Riparian planting projects in the Shasta River watershed have had historically poor rates of survival. The evaluation in this study of the physical, chemical, and biological processes in the Shasta River that contribute to poor riparian plant survival cannot be considered exhaustive. However, findings indicate several site conditions that likely contributed to lowered survival rates in this study, including: very poor soil conditions for plant growth (high salinity, high pH, poor nutrient ratios, low organic matter), groundwater draw down rates that exceed acceptable reported rates for *Salix* spp. seedlings, and lack of springtime over-bank flooding during establishment. While it is expected that there will be differing site conditions throughout the riparian areas of the Shasta River, it is likely that many sites will experience similar conditions at varying degrees as the underlying causes for these poor site conditions can be attributed to altered physical processes throughout the watershed.

In order to overcome these challenging underlying conditions, process-based restoration models would seek to restore the altered physical processes on a watershed scale that could result in the conditions for the ecosystem to self-sustain. Within the context of the Shasta River this could include working with the altered hydrologic regime controlled by irrigated agriculture and the damming of the Shasta River, and the physical processes that have resulted in poor soil conditions. Managers in other dammed watersheds within the semi-arid and arid southwest are working within altered ecosystems to mimic more naturalized flow regimes at critical times of year (Sprenger *et al.* 2002). Successful experimental projects have included timing the dispersal of native plant seeds with controlled flood releases and managing river stage drawdown rates to encourage natural riparian recruitment (Sprenger *et al.* 2002, Bhattacharjee *et al.* 2006).

In addition to replicating naturalized flood regimes, other opportunities may exist to manage the drop in river stage that corresponds with low flow periods and the primary irrigation season on the Shasta River. Staggering the ‘turn-on’ times for irrigators through-out the basin, for example, could result in less rapid changes to the river stage and corresponding groundwater levels in the riparian zone in April, which is a critical time for plant establishment. This may be particularly important in water years with dry spring months because irrigators are more likely to start irrigating on 1 April, instead of delaying turn on times due to spring rains. Consideration of the water table depth in riparian areas during the low-flow periods of July and August will also allow managers to better plan for conditions that riparian plantings will experience during the critical first growth year. Monitoring groundwater levels for one-year prior to planting projects will provide indication of the depth that pole cuttings must be planted to in order to enable the poles to have the best chance of survival.

Although ideal, addressing the altered hydrologic conditions may be beyond the scope and budget of most small-scale riparian restoration planting projects. Therefore, selecting species with adaptations to altered site conditions and using planting techniques to help overcome challenging site conditions becomes a worthwhile investment for managers. The results from this study indicate high variability in survival between the two willow species: red and arroyo. Arroyo willows had much greater survival than red willows when exposed to the same challenging site conditions indicating it is a favorable species for managers to plant in the Shasta River. Further research into the adaptation of arroyo willows in the Shasta watershed when compared to other arroyo willow populations outside of the watershed is needed to reveal whether a salt-tolerance adaptation has developed for the population of Shasta River arroyo willows.

While planting adaptable riparian species is an important planning consideration, the encouragement of a diverse riparian canopy is important for habitat complexity. Therefore, utilizing tree shelters may assist certain riparian tree and shrub species in over-coming challenging site conditions, especially in the critical first year of establishment. In general, there has been little research on the application of tree shelters for riparian species in highly altered semi-arid riparian areas. Though the results of this study indicate that red willows that had poor overall survival, when planted in unvented tree shelters poles were three times more likely to survive than those in the control treatment only. This suggests that unvented tree shelters may be a successful planting methodology that can improve survival for this species under similar challenging growth conditions. The challenging site conditions experienced in the Shasta River are indicative of many rivers and riparian areas in the arid and semi-arid southwest. Given the scale of the problem, further research is needed on the application of tree shelters on different riparian tree species to determine their efficacy in improving survival under challenging and inhospitable site conditions.

# REFERENCES

Abbot, A.G.P. 2002. The Effect of Riparian Vegetation on Stream Temperature in the Shasta River. Master's thesis. U.C. Davis Department of Civil and Environmental Engineering, pp. 1-169.

Allen, J. A., Chambers, J.L. and M. Stine. 1994. Prospects for increasing the salt Tolerance of forest trees: a review. Tree physiology 14: 843-853.

Amlin, N.M. and S. B. Rood. 2002. Comparative tolerances of riparian willows and cottonwoods to water-table decline. Wetlands 22 (2): 338-346.

Barnett TP, Adam JC, and D.P. Lettenmaier. 2005. Potential impacts of a warming climate on water availability in snow-dominated regions. Nature 438: 303–309.

Bellot, J., Ortiz de Urbina, J., Bonet, A. and J.R. Sanchez. 2002. The effects of treeshelters on the growth of *Quercus coccifera* L. seedlings in a semiarid environment. Forestry 75 (1): 89-106.

Bendix, J. and C. R. Hupp. 2000. Hydrological and geomorphological impacts on riparian plant communities. Hydrologic Processes 14: 2977-2990.

Bhattacharjee, J., J.P. Taylor, and L.M. Smith. 2006. Controlled flooding and staged drawdown for restoration of native cottonwoods in the middle Rio Grande Valley, New Mexico, USA. Wetlands 26 (3): 691-702.

Bogener, D. J. 1990. Shasta Valley Water Quality Literature Review. State of California Department of Water Resources, Northern District. Accessed at: https://www.water.ca.gov/LegacyFiles/pubs/waterquality/shasta_valley_water_quality_literature_review/shasta_litreview.pdf. Accessed: 02/11/2018.

Braatne, J. H., Rood, S. B., Goatner, L.A., and C. L. Blair. 2008. Analyzing the impacts of dams on riparian ecosystems: A review of research strategies and their relevance to the Snake River through Hells Canyon. Environmental Management 41: 267-281.

Briggs, M. K. 1996. Riparian ecosystem recovery in arid lands: strategies and references. The University of Arizona Press. Tuscon, AZ, pp. 1-155.

Brinson, M.M., Swift, B.L., Plantico, R.C., and J. S. Barclay. 1981. Riparian ecosystems: their ecology and status. United States Fish and Wildlife Service, Washington, D.C. OBS-81/17, pp. 1-173.

Busch D., Herbranson, L., Johns, E., Pinkney, F., and D. Sisneros. 1992. Vegetation management study: lower Colorado River. Phase I report to the U.S.D.I.-U.S. Bureau of Reclamation, lower Colorado region, pp. 1-103.

Devine, W. and C. Harrington. 2008. Influence of four tree shelter types on microclimate and seedling performance of Oregon white oak and western redcedar. Res. Pap. PNW-RP-576. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, pp. 1-35.

California Data Exchange Center (CDEC). Department of Water Resources. River Stages. Accessed at: http://cdec.water.ca.gov/. Accessed on 02.21.2018.

California Department of Conservation (CA DOC). 2016. Farmland Mapping & Monitoring Program. Accessed at: http://www.conservation.ca.gov/dlrp/fmmp. Accessed 12/10/2017.

California Department of Fish and Wildlife (CDFW). 1934. Stream Survey: Scott River, Tributary to the Klamath River. California Department of Fish and Wildlife, Sacramento, CA, pp. 1-3.

California Department of Water Resources (CDWR). 1964. Shasta Valley Investigation. Bulletin No. 87. Sacramento, CA: Dept. of Water Resources, pp. 1-324.

California Native Plant Society. 2018. Calscape: Red willow and Arroyo Willow. Accessed at: http://calscape.org/Salix-laevigata-(Red Willow). Accessed 02/11/2018.

Chaar, H., Mecherguis, T., Khouaja, A. and H. Abid. 2008. Effects of treeshelters and polyethylene mulch sheets on survival and growth of cork oak (*Quercus suber* L.) seedlings planted in northwestern Tunisia. Forest Ecology and Management 256: 722-731.

Cleary, B.D. and J. B. Zaerr. 1980. Pressure Chamber Techniques for Monitoring and evaluating Seedling Water Status. New Zealand, Journal of Forestry Science. Vol. 10 (1): 133-141.

Department of Water Resources, State of California. 2011. Shasta Valley, Siskiyou County Groundwater Data Needs Assessment. Accessed at: http://svrcd.org/wordpress/wpontent/uploads/2016/07/Shasta_Final_Draft_5_31_1.-mt-reduced-images.pdf. Accessed 02/11/2018.

Doyle, M. W., Miller, D.E., and J. M. Harbor. 1999. Should river restoration be based on classification schemes or process models? Insights from the history of geomorphology. International Conference on Water Resources Engineering, Seattle, Washington, pp. 1-9.

Dreesen, D.R. and G.A. Fenchel. 2008. Deep-planting methods that require minimal or no irrigation to establish riparian trees and shrubs in the Southwest. Journal of Soil and Water Conservation. 63:129A-133A.

Everest, F. E. and G. H. Reeves. 2007. Riparian and aquatic habitats of the Pacific northwest and southeast Alaska: Ecology, management, history, and potential management strategies. United States Forest Service – Pacific Northwest Research Stations, General Technical Report PNW – GTR – 692, pp 1-148.

Fenner, P., Brady, W.W. and D.R. Patton. 1985. Effects of regulated water flows on regeneration of Fremont cottonwood. Journal of Range Management 38 (2): 135-138.

Fowler, A. and C. Babcock. 2012. Survival of native tree species planting on TNC's Shasta Big Springs Ranch between 2010 to 2012. ARRA-NOAA Monitoring Report. Unpublished, pp.1-9.

Gremmen, N. J.M., Reijnen, M.J.S. M., Wiertz, J. and G. Van Wirdum. 1990. A model to predict and assess the effects of ground-water withdrawal on vegetation in the Pleistocene areas of the Netherlands. Journal of Environmental Management 31: 143-155.

Griggs, T.F. 2009. California riparian habitat restoration handbook. Second Edition. Accessed at:https://www.water.ca.gov/LegacyFiles/urbanstreams/docs/ca_riparian_handbook.pdf. Accessed on 03/24/2018.

Harris, R.R., C.A. Fox, and R. Risser. 1987. Impacts of hydroelectric development on riparian vegetation in the Sierra Nevada region, California, USA. Environmental Management 11: 519-527

Hall, J. Pollock, M. and S. Hoh. 2011. Methods for successful establishment of cottonwood and willow along an incised stream in semiarid eastern Oregon, USA. Ecological Restoration 29 (3): 261-269.

Hall, J.E., Pollock, M.M., Hoh, S., Volk, C., Goldsmith, J. and C.E. Joran. 2015. Evaluation of dryland riparian restoration with cottonwood and willow using deep-planting and herbivore protection. Ecosphere 6 (12): 1-12.

Jacobs, F.D. and K. Steinbeck. 2001. Tree shelters improve the survival and growth of planted Engelmann spruce seedlings in southwestern Colorado. Western Journal of Applied Forestry 16 (3): 114-120.

Jacobs, D. F. 2011. Reforestation of a salvage-logged high-elevation clearcut: Engelmann spruce Seedling response to tree shelters after 11 growing seasons. Western Journal of Applied Forestry 26 (2): 53-56.

Jaleel, C.A., Manivannan, P. Wahid, A. Farooq M., Somasundaram, R and R. Panneerselvam. 2009. Drought stress in plants: a review on morphological characteristics and pigments composition. International Journal of Agriculture & Biology 11: 100–105

Jansson, R., Nilsson, C. and B. Renofalt. 2000. Fragmentation of riparian floras in rivers with multiple dams. Ecology 81: 899–903

Jarvis, P.G. and M. S. Jarvis. 1963. The water relations of tree seedlings I. Growth and water use in relation to soil water potential. Plant Physiology 18. 215-235.

Johnson, A.S. 1989. The thin green line: Riparian corridors and endangered species in Arizona and New Mexico. In: Mackintosh, F. ed. Preserving Communities and Corridors. Washington D.C. Defenders of Wildlife: 35-46.

Johnson, R. R. and J. F. McCormack. 1979. Strategies for protection and management of floodplain wetlands and other riparian ecosystems. General Technical Report WO-12. U.S. Forest Service, Washington, DC.

Kjelgren, R. and L. A. Rupp. 1997. Establishment in treeshelters I: shelters reduce growth, water used, and hardiness, but not drought avoidance. Horticultural Science 32 (2): 1281-1283.

Kjelgren, R., Montague, D.T., and L.A. Rupp. 1997. Establishment in treeshelters II: Effect of shelter color on gas exchange and hardiness. Horticultural Science. 32 (7): 1284-1287.

Kondolf G.M. 1997. Hungry water: effects of dams and gravel mining on river channels. Environmental Management 21: 533–551.

Magilligan, F.J., Nislow, K.H., and B.E. Graber. 2003. Scale-independent assessment of discharge reduction and riparian disconnectivity following flow regulation by dams. Geology 31:569–572

Magistad, O.C. 1945. Plant growth relations on saline and alkali soils. Botanical Review 11 (4): 181-230.

Mattson, K. 2008. Effectiveness Monitoring of Restoration Projects in the Shasta Basin. Final Project Report P0410328 for the California Department of Fish and Game, Fishery Restoration Grant Program. Yreka, CA. Accessed at: http://ecosystemsnorthwest.com/wp-content/uploads/2015/12/Effectiveness-Monitoring-Shasta-without-appendices.pdf. Accessed 02/09/2018.

McCreary, D., Tietke, W., Davy, J., Larsen, R., Doran, M., Flavell, D., and S. G. Emirtus. 2011. Tree shelters and weed control enhance growth and survival of natural blue oak seedlings. California Agriculture 65 (4): 192-196.

Minitab 18.1 Statistical Software (2017). Computer Software. Minitab, Inc.

Nakamura, F., Swanson, F.J. and S. M. Wondzell. 2000. Disturbance regimes of stream and riparian systems – a disturbance-cascade perspective. Hydrologic Processes 14: 2849-2860.

Naiman, R. J., Decamps, H. and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3 (2): 209-212.

Nichols, A. and J. F. Mount. 2008. Geological Mediation of Hydrologic Process, Channel Morphology and Resultant Planform Response to Closure of Dwinnell Dam, Shasta River, California. MS Thesis Office of Graduate Studies, University of California Davis: 1-68.

Nilsson, C. 1982. Effects of stream regulation on riparian vegetation. Regulated Rivers. Columbia University Press, New York, pp. 93-106.

North Dakota State University Extension Service (NDSU). 2014. Saline and Sodic Soils. https://www.ndsu.edu/soilhealth/wp-content/uploads/2014/07/Saline-and-Sodic-Soils-2-2.pdf. Accessed 02/11.2018.

National Research Council (NRC). 2004. Endangered and Threatened Fishes in the Klamath River Basin: Causes of Decline and Strategies for Recovery. Board on Environmental Studies and Toxicology, Division of Earth and Life Studies. National Academies Press: Washington, DC. Pp. 1-425.

Obedzinski, R. A., Shaw, C.G. and D. G. Neary. 2001. Declining woody vegetation in riparian ecosystems of the western United States. Western Journal of Applied Forestry 16 (4): 169-181.

Ogle, D. and L. St. John. 2010. Plants for saline to sodic soil conditions. USDA- Natural Resources Conservation Service – Technical Notes. TN Plant Materials No. 9A, pp. 1-10.

Oliet, J.A. and D. F. Jacobs 2007. Micro-climate conditions and plant morph-physiological development within a tree shelter environment during establishment of *Quercus ilex* seedlings. Agricultural and Forest Meteorology 144: 58-72.

Oregon State University – Extension Service. 2002. Cottonwood establishment, survival, and stand characteristics. OSU Extension Catalog: Accessed at: https://catalog.extension.oregonstate.edu/em8800. Accessed 01/15/2018.

Patten, D.T. 1998. Riparian ecosystems of semi-arid North America: diversity and human impacts. Wetlands 18 (4): 498-512.

Poff, B., Koestner, K.A., Neary, D.G., and V. Henderson. 2011. Threats to riparian ecosystems in western North America: An analysis of existing literature. Journal of the American Water Resources Association: 1-14.

Poff, N.L. and J.K.H. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55: 194-205.

Sabo, J. L., Sponseller, R., Dixon, M., Gade, K., Harms, T., Heffernan, J., Jani, A., Katz, G., Soykan, C., Watts, J. and J. Welter. Riparian zones increase regional species richness by harboring different, not more, species. Ecology 86 (1): 56-62.

Schaff S.D., S. R. Pezeshki, F. D. Shields. 2002. Effects of pre-planting soaking on growth and survival of black willow cuttings. Restoration Ecology 10 (2): 267-274.

Sprenger, M. D., L. M. Smith, and J. P. Taylor. 2002. Restoration of riparian habitat using experimental flooding. Wetlands 22:49–57.

Stella, J.C., J.J. Battles, J. R. McBride, and B.K. Orr. 2010. Riparian seedlings mortality from simulated water table recession, and the design of sustainable flow regimes on regulated rivers. Restoration Ecology 18 (2): 284-294.

Stromberg, J. C. 2001. Restoration of riparian vegetation in the south-western United States: importance of flow regimes and fluvial dynamism. Journal of Arid Environments 49: 17-34.

Stromberg, J.C., Beauchamp, V. B., Dixon, M.D., Lite, S.J., and C. Paradzick. 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid south-western United States. Freshwater Biology 52: 651-679.

Stromberg, J.C., Tiller, R. and B. Richter. 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: The San Pedro, Arizona. Ecological Applications 6 (1): 113-131.

Stromberg, J.C. and D. T. Patten. 1990. Riparian vegetation instream flow requirements: a case study from a diverted stream in the eastern Sierra Nevada. Environmental Management (14): 185-194.

Svejcar, T. 1997. Riparian Zones: 1) What They Are and How Do They Work. Rangelands 19: 4-7.

Swift, B.l. 1984. Status of riparian ecosystems in the United States. Water Resources Bulletin 20: 223-228.

Swenson, E.A. and C.L. Mullins. Revegetating riparian trees in southwestern floodplains. Paper presented at the Interagency North American Riparian Conference, Riparian Ecosystems and their Management: Reconciling Conflicting Uses, Tucson, Arizona. April 16-18, 1985.

Taylor, J. P. and K. C. McDaniel. 1998. Restoration of saltcedar (Tamarix sp.) Infested Floodplains on the Bosque del Apache National Wildlife Refuge. Weed Technology 12: 345-352.

Tuley, G. 1985. The growth of young oak trees in shelters. Forestry. 58:181-195.

Tyee, M.T. and H.T. Hammel. 1972. The Measurement of the Turgor Pressure and the Water Relations of Plants by the Pressure Bomb Technique. Journal of Experimental Botany (23): 267-282.

United States Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). 2017. Agricultural Applied Climate Information System (AgACIS). https://efotg.sc.egov.usda.gov/treemenuFS.aspx Accessed: 12/07/2017.

United States Army Corps of Engineers (US ACOE). 2016. National Inventory of Dams. http://nid.usace.army.mil/cm_apex/f?p=838:12 . Accessed. 12.09.2017.

United States Department of Agriculture, Soil Conservation Service (USDA- NRCS). 2007. Technical Note: How to plant willows and cottonwoods for riparian restoration. Technical Note Plant Materials Center No. 23, Aberdeen, Idaho, pp. 1-21.

United States Department of Agriculture, Soil Conservation Service (USDA- NRCS). 1983. Soil Survey of Siskiyou County California, Central Part, pp. 1-116.

United States Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). 2018. Conservation Plant Characteristics, *Salix lasiolepis*. Accessed at: https://plants.usda.gov/java/charProfile?symbol=SALA6. Accessed. 02/11/2018

United States Department of Agriculture, Natural Resource Conservation Service (USDA-NRCS). 2018. Soil Health for Educators. Accessed at: https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/health/assessment/?cid=nrcs142p2_053870. Accessed. 02/11/2018.

United States Environmental Protection Agency (US EPA). 1995. Ecological Restoration: A tool

to Manage Stream Quality. EPA, Office of Water, Washington DV. EPA 841-F-95-007, pp. 1-168.

Vandersands, M.W., Glenn, E.P., and J. L. Walworth. 2001. Tolerance of five riparian plants from the lower Colorado River to salinity drought and inundation.

Ward, J.B. 1989. The four-dimensional nature of lotic ecosystems. Journal of North American Benthological Society 8 (1): 2-8.

Willis, A.D., Nichols, A.L., Jeffres, C.A., Deas, M.L, 2013. Water Resources Management Planning: Conceptual framework and case study of the Shasta River. Unpublished. Report prepared for: National Fish and Wildlife Foundation. Accessed at: https://watershed.ucdavis.edu/library/water-resources-management-planning-conceptual-framework-and-case-study-shasta-basin. Accessed 01/15/2018.

# TABLES

Table 1 - Raw survival (# of alive trees) for each species and treatment measured spring and fall 2016 and 2017. Fall 2015 represents n at the time of planting

| | **Red willow** | | | **Arroyo willow** | | |
|---|---|---|---|---|---|---|
| | Control | Unvented | Vented | Control | Unvented | Vented |
| Fall 2015 (n) | 39 | 38 | 38 | 38 | 36 | 39 |
| Spring 2016 | 28 | 27 | 25 | 38 | 36 | 37 |
| Fall 2016 | 10 | 19 | 13 | 36 | 32 | 33 |
| Spring 2017 | 7 | 16 | 11 | 35 | 32 | 32 |
| Fall 2017 | 7 | 16 | 11 | 34 | 32 | 32 |

Table 2- Effects of depth to groundwater and shelter treatment on survival of red and arroyo willows in 2016 and 2017.

| | | **Red Willow** | | **Arroyo Willow** | |
|---|---|---|---|---|---|
| | | 2016 | 2017 | 2016 | 2017 |
| Deviance $R^2$ (%) | | 3.26 | 3.64 | 8.72 | 2.59 |
| Constant* | | 0.570 | 0.142 | 0.001 | 0.010 |
| Depth to groundwater* | | 0.554 | 0.905 | 0.052 | -/287 |
| Treatment | Control* | - | - | - | - |
| | Unvented* | 0.054 | 0.034 *(Odds Ratio: 3.1405, 95% CI: 1.0912,9.0385)* | 0.376 | 0.962 |
| | Vented* | 0.456 | 0.266 | 0.219 | 0.413 |

p – value associated with binary logistic regression coefficient; for categorical data (i.e., treatment) p-value is relative to the control. Odds Ratio reported only for significant value.

Table 3 - Median, q1 (first quartile) and q3 (third quartile) plant moisture stress (bars) for each species and treatment. Sub-population (n) changes between year are due to mortality.

| | **Red Willow** | | | | **Arroyo Willow** | | | |
|---|---|---|---|---|---|---|---|---|
| | n | Q1 | Median | Q3 | n | Q1 | Median | Q3 |
| **Pre-Dawn 2016** | | | | | | | | |
| Control | 8 | 1.9 | 2.3 | 2.5 | 10 | 1.5 | 1.8 | 2.4 |
| Unvented | 11 | 1.5 | 2.0 | 2.5 | 9 | 1.2 | 1.5 | 1.8 |
| Vented | 11 | 1.0 | 1.5 | 2.3 | 10 | 1.4 | 1.8 | 2.5 |
| **Solar Noon 2016** | | | | | | | | |
| Control | 8 | 9.3 | 10.5 | 12.4 | 10 | 5.1 | 8.0 | 10.7 |
| Unvented | 11 | 8.5 | 10.0 | 12.0 | 9 | 7.0 | 8.5 | 10.0 |
| Vented | 11 | 8.6 | 10.0 | 12.8 | 10 | 7.0 | 9.5 | 9.9 |
| **Pre-dawn 2017** | | | | | | | | |
| Control | 4 | 1.7 | 2.1 | 3.3 | 8 | 2.5 | 3.0 | 4.0 |
| Unvented | 7 | 2.4 | 2.6 | 3.6 | 4 | 2.8 | 3.4 | 4.2 |
| Vented | 8 | 2.3 | 2.5 | 2.7 | 6 | 1.9 | 2.4 | 2.6 |
| **Solar Noon 2017** | | | | | | | | |
| Control | 4 | 11.8 | 13.3 | 15.0 | 8 | 13.0 | 14.8 | 15.6 |
| Unvented | 7 | 13.0 | 14.0 | 17.2 | 4 | 12.4 | 13.4 | 15.0 |
| Vented | 8 | 12.8 | 13.6 | 14.8 | 6 | 14.5 | 14.8 | 15.3 |

Table 4 - Type III tests of fixed effects from analysis of variance of plant moisture stress data (df = degrees of freedom).

| **Effect** | **Pre-dawn 2016** | | | **Pre-dawn 2017** | | | **Solar Noon 2016** | | | **Solar Noon 2017** | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | df | F | Pr > F | df | F | Pr > F | df | F | Pr > F | df | F | Pr > F |
| Species | 1/53 | 0.35 | 0.558 | 1/32 | 0.60 | 0.446 | 1/58 | 5.87 | 0.019 | 1/32 | 0.09 | 0.771 |
| Shelter | 2/53 | 1.66 | 0.200 | 2/32 | 2.24 | 0.123 | 2/58 | 0.05 | 0.948 | 2/32 | 0.09 | 0.911 |
| Species x Shelter | 2/53 | 0.56 | 0.575 | 2/32 | 0.54 | 0.588 | 2/58 | 0.21 | 0.815 | 2/32 | 0.35 | 0.711 |

Table 5 - 2016 Mean monthly air temperature (°C) for treatments and control (ambient). V= vented, U= unvented. Bottom = 0.15m, Mid = 0.46m, Top = 0.91m. Numbers in brackets indicate °C hotter (+) or cooler (-) than ambient air temperature.

| **Treatment/Shelter Position** | **Temperature (°C)** | | | | | |
|---|---|---|---|---|---|---|
| | Apr | May | Jun | Jul | Aug | Sep |
| V bottom | 36.8 (+ 10.6) | 38.3 (+ 10.4) | 39.1 (+ 6.9) | 41.2 (+ 6.4) | 41.2 (+ 4.9) | 37.3 (+ 5.5) |
| U bottom | 35.1 (+ 8.8) | 36.0 (+ 8.1) | 37.5 (+ 5.3) | 36.0 (+ 1.3) | 33.6 (-2.7) | 31.4 (- 0.4) |
| V mid | 37.9 (+ 11.6) | 38.7 (+ 10.8) | 43.6 (+ 11.4) | 47.4 (+ 12.7) | 48.2 (+ 11.9) | 44.2 (+ 12.3) |
| U mid | 32.4 (+ 6.1) | 34.4 (+ 6.4) | 37.7 (+ 5.5) | 40.3 (+ 5.6) | 40.4 (+ 4.1) | 38.1 (+ 6.3) |
| V top | 35.1 (+ 8.8) | 36.9 (+ 9.0) | 44.2 (+ 12.0) | 48.4 (+ 13.6) | 50.9 (+ 14.6) | 46.6 (+ 14.8) |
| U top | 32.9 (+ 6.6) | 34.4 (+ 6.5) | 39.4 (+7.2) | 42.6 (+ 7.9) | 44.6 (+ 8.3) | 41.7 (+ 9.9) |
| Ambient (control) | 26.3 | 27.9 | 32.2 | 34.7 | 36.3 | 31.8 |

Table 6 - 2017 Mean monthly air temperatures (°C) for treatments and control (ambient). V= vented, U= unvented. Bottom = 0.15m, Mid = 0.46m, Top = 0.91m. Numbers in brackets indicate °C hotter (+) or cooler (-) than ambient air temperature.

| **Treatment/Shelter Position** | **Temperature (°C)** | | | | | |
|---|---|---|---|---|---|---|
| | Apr | May | Jun | Jul | Aug | Sep |
| V bottom | 28.8 (+ 6.0) | 37.3 (+ 8.4) | 34.0 (+ 1.8) | 39.2 (+ 1.9) | 39.2 (+ 2.1) | 32.9 (+ 2.0) |
| U bottom | 23.0 (+ 0.2) | 30.7 (+ 1.8) | 32.7 (+ 0.6) | 37.5 (+ 0.3) | 38.3 (+ 1.1) | 32.9 (+ 2.0) |
| V mid | 31.7 (+ 8.9) | 40.8 (+ 11.9) | 41.8 (+ 9.6) | 48.1 (+ 10.8) | 47.4 (+ 10.3) | 40.3 (+ 9.5) |
| U mid | 25.1 (+ 2.3) | 32.6 (+3.7) | 35.7 (+ 3.5) | 40.7 (+ 3.4) | 41.7 (+ 4.6) | 36.1 (+ 5.3) |
| V top | 30.2 (+ 7.4) | 36.8 (+ 7.9) | 40.5 (+ 8.4) | 46.8 (+ 9.5) | 48.2 (+ 11.0) | 43.0 (+ 12.1) |
| U top | 25.0 (+ 2.2) | 33.6 (+ 4.7) | 39.1 (+ 6.9) | 42.7 (+ 5.4) | 42.0 (+ 4.9) | 36.0 (+ 5.2) |
| Ambient (control) | 22.8 | 28.9 | 32.2 | 37.3 | 37.1 | 30.8 |

Table 7 - 2016 Maximum daily air temperature by month for treatments and control (ambient). V= vented, U= unvented. Bottom = 0.15m, Mid = 0.46m, Top = 0.91m. Numbers in brackets indicate °C hotter (+) or cooler (-) than ambient air temperature.

| **Treatment/Shelter Position** | **Temperature (°C)** | | | | | |
|---|---|---|---|---|---|---|
| | Apr | May | Jun | Jul | Aug | Sep |
| V bottom | 44.7 (+ 10.6) | 44.8 (+ 7.9) | 45.5 (+ 6) | 44.8 (+ 2.9) | 43.9 (+ 2.6) | 44.1 (+ 6.5) |
| U bottom | 44.8 (+10.7) | 42.6 (+ 5.7) | 43.5 (+ 4) | 40.9 (- 1) | 36.3 (- 5) | 37.7 (+ 0.1) |
| V mid | 46.6 (+ 12.5) | 47.6 (+ 10.7) | 50.4 (+ 10.9) | 53 (+ 11.1) | 52.2 (+ 10.9) | 50.6 (+ 13) |
| U mid | 41.6 (+ 7.5) | 42.6 (+ 5.7) | 44.4 (+ 4.9) | 46.1 (+ 4.2) | 43.8 (+ 2.5) | 44.4 (+ 6.8) |
| V top | 43.7 (+ 9.6) | 48.8 (+ 11.9) | 51.6 (+ 12.1) | 56.3 (+14.9) | 55.5 (+ 14.2) | 53.9 (+ 16.3) |
| U top | 41.2 (+ 7.1) | 44.8 (+ 7.9) | 46.7 (+ 7.2) | 49.4 (+7.5) | 48.8 (+ 7.5) | 48.5 (+ 10.9) |
| Ambient (control) | 34.1 | 36.9 | 39.5 | 41.9 | 41.3 | 37.6 |

Table 8 - 2017 Maximum daily air temperature by month °C for treatments and control (ambient). V= vented, U= unvented. Bottom = 0.15m, Mid = 0.46m, Top = 0.91m. Numbers in brackets indicate °C hotter (+) or cooler (-) than ambient air temperature.

| **Treatment/Shelter Position** | **Temperature (°C)** | | | | | |
|---|---|---|---|---|---|---|
| | Apr | May | Jun | Jul | Aug | Sep |
| V bottom | 39.8 (+9.5) | 46.0 (+ 9.3) | 40.9 (- 0.3) | 42.9 (+ 0.4) | 44.7 (+ 0.7) | 40.1 (- 0.6) |
| U bottom | 30.2 (-0.1) | 38.2 (+ 1.5) | 39.7 (- 1.5) | 41.2 (- 1.3) | 42.9 (- 1.1) | 40.7 (0) |
| V mid | 40.6 (+ 10.3) | 50.8 (+ 14.1) | 49 (+ 7.8) | 51.8 (+ 9.3) | 54.0 (+ 10.0) | 48.8 (+ 8.1) |
| U mid | 32 (+ 1.7) | 41.1 (+ 4.4) | 44 (+ 2.8) | 44.9 (+ 2.4) | 47.1 (+ 3.1) | 44.3 (+ 3.6) |
| V top | 38.4 (+ 8.1) | 46.0 (+ 9.3) | 50.7 (+ 9.5) | 51.3 (+ 8.8) | 53.9 (+ 9.9) | 51.2 (+ 10.5) |
| U top | 31.4 (+ 1.1) | 44.4 (+ 7.7) | 49.0 (+ 7.8) | 46.1 (+ 3.6) | 47.5 (+ 3.5) | 44.8 (+ 4.1) |
| Ambient (control) | 30.3 | 36.7 | 41.2 | 42.5 | 44 | 40.7 |

Table 9 - Days per month during primary growing season and irrigation season (April - September) for 2016 and 2017 where the average 24-hour groundwater rate exceeded 1 cm/day.

| **2016** | **Apr** | **May** | **Jun** | **Jul** | **Aug** | **Sep** | **Season Total** |
|---|---|---|---|---|---|---|---|
| 1-3 cm/ day | 4 | 7 | 9 | 6 | 5 | 0 | 31 |
| 3-6 cm/day | 2 | 3 | 0 | 3 | 0 | 0 | 8 |
| ≥ 6 cm/day | 1 | 1 | 0 | 0 | 0 | 0 | 2 |
| **2017** | **Apr*** | **May*** | **Jun*** | **Jul** | **Aug** | **Sep** | **Season Total** |
| 1-3 cm/ day | n/a | n/a | n/a | 10 | 2 | 0 | 12 |
| 3-6 cm/day | n/a | n/a | n/a | 4 | 0 | 0 | 4 |
| ≥ 6 cm/day | n/a | n/a | n/a | 0 | 0 | 0 | 0 |

* Data excluded when groundwater peak and drawdown was associated with storm event in April 2017, or when groundwater levels were above average planting depth.

Table 10 - Organic matter (OM), pH, and nutrient levels of study site soil at depths of 12” and 24”.

| **Depth** | **pH** | **OM (%)** | **$NO_3$-N (ppm)** | **P (ppm)** | **K (ppm)** | **Mg (ppm)** | **Ca (ppm)** | **Na (ppm)** |
|---|---|---|---|---|---|---|---|---|
| 12” | 9.1* | 1.4** | 4** | 10** | 34** | 984*** | 1152*** | 447*** |
| 24” | 8.7* | 1.3** | 2** | 4** | 17** | 895*** | 1701*** | 166*** |

* Strongly alkaline to very strongly alkaline

** Very low

*** High - very high

# FIGURES

Figure 1 – Map of study area. Adapted from CDFW Water Branch Shasta Valley Map (2018)

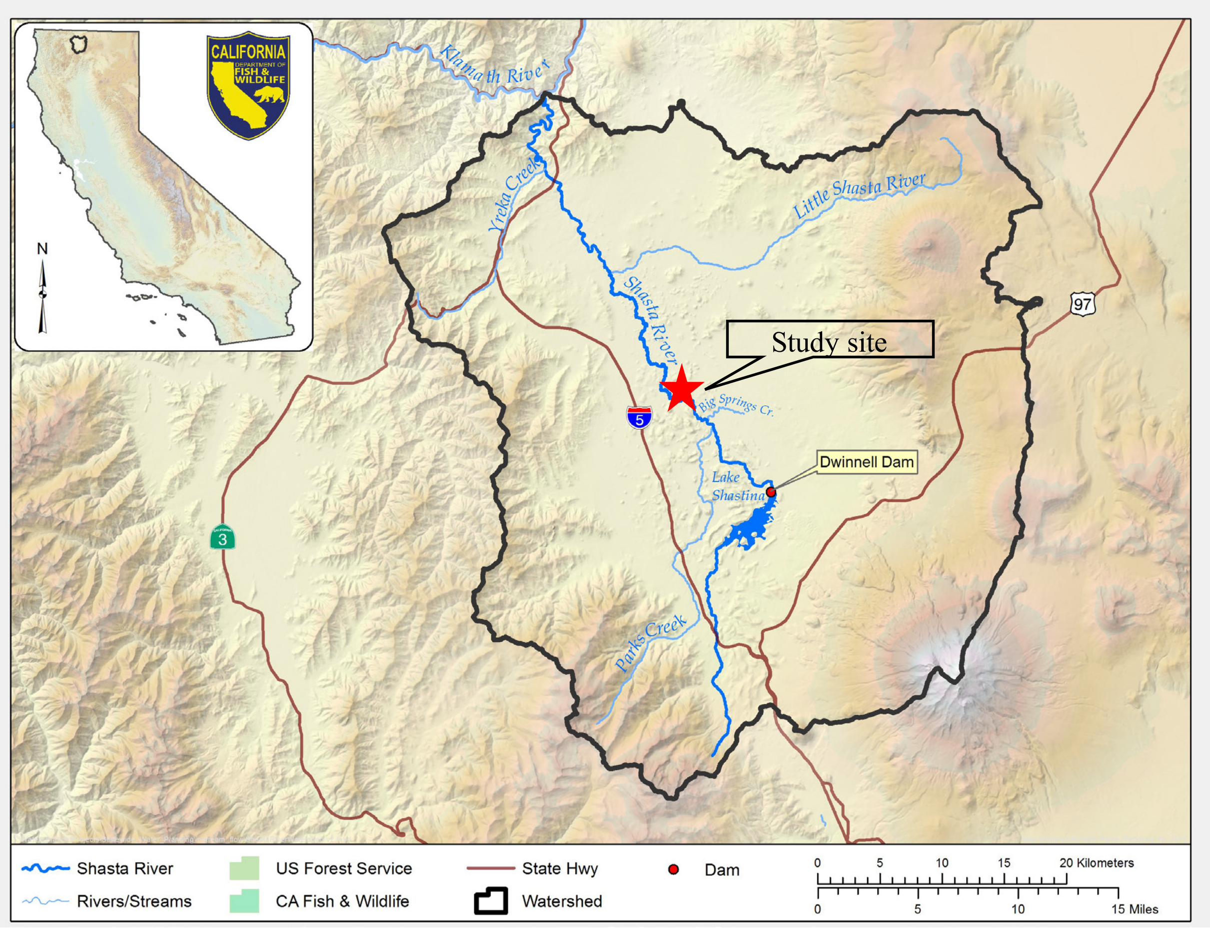

Figure 2 – Percent survival for control (no treatment), vented, and unvented treatments. Survival is grouped by species (arroyo and red willow) and by monitoring period (spring 2016, fall 2016, spring 2017, fall 2017).

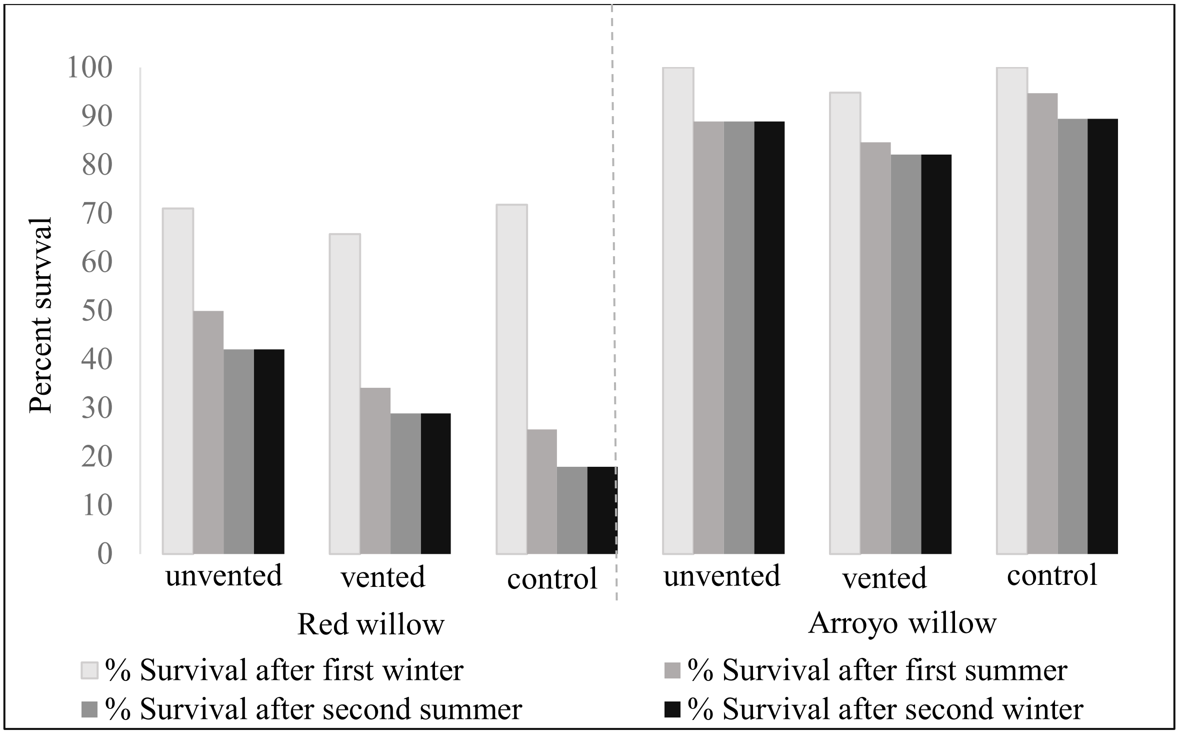

Figure 3 - Interquartile distribution of median plant moisture stress results by year and time (pre-dawn, solar noon). Boxes indicated 25th, 50th, 75th percentile distributions and black dots indicate outliers. Results are displayed by treatment: C = control, U.

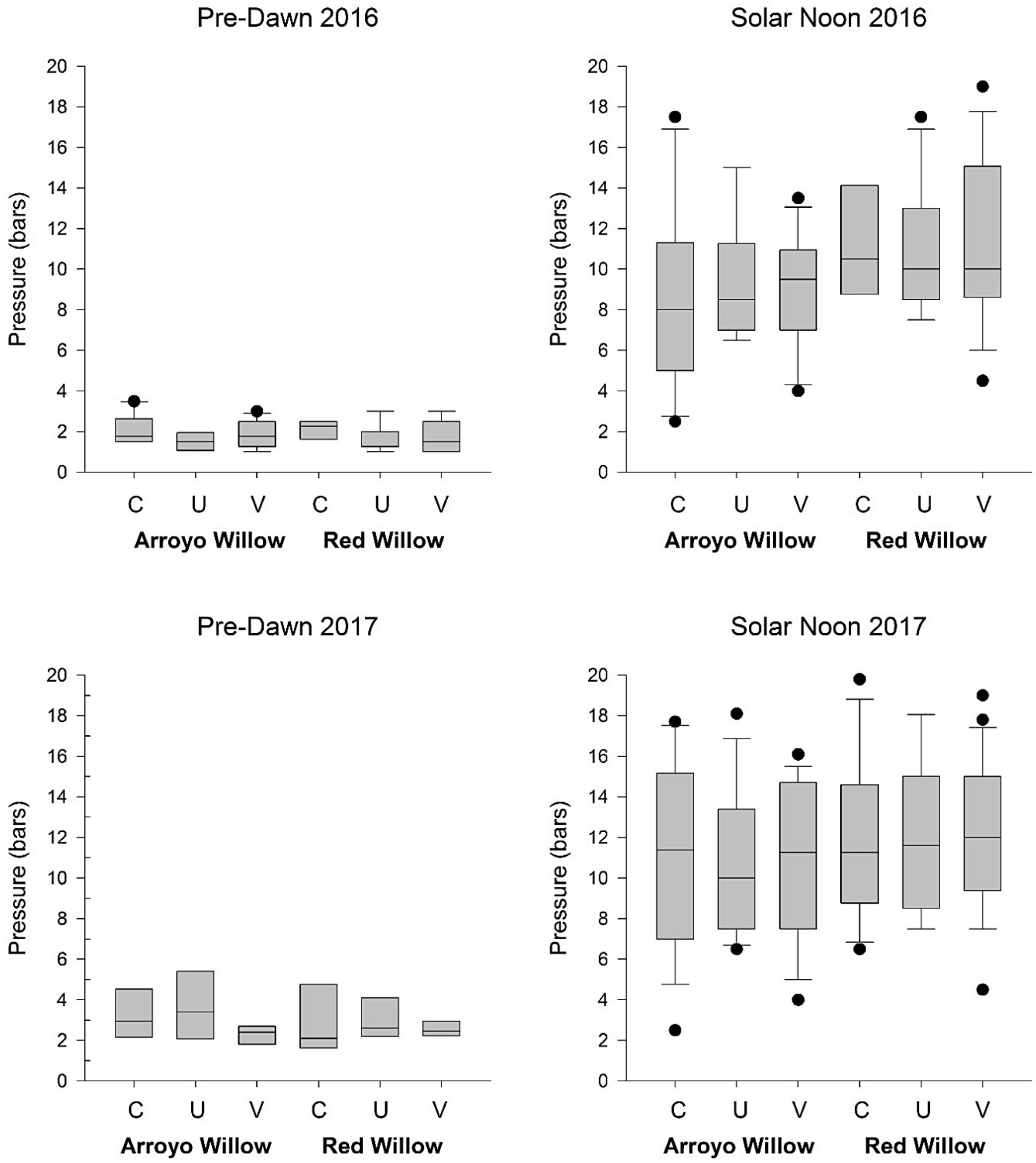

Figure 4 - Maximum daily air temperature by month during the 2016 -2017 study periods for vented, unvented and ambient air temperature at different positions in the shelters (bottom, mid, high).

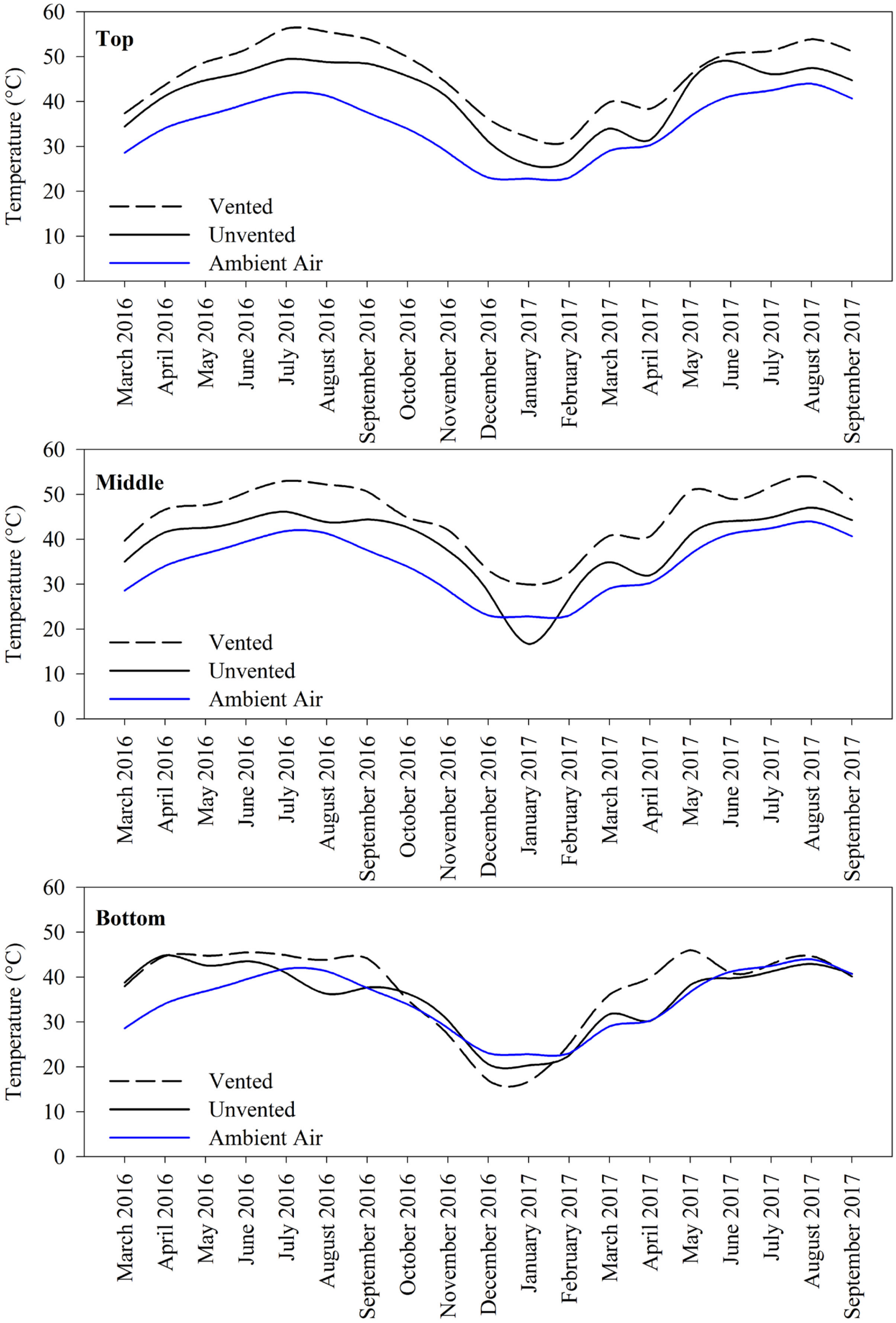

Figure 5 - Mean daily air temperature by month during the 2016 – 2017 study periods for vented, unvented and ambient air temperature at different positions in the shelters (bottom, mid, high).

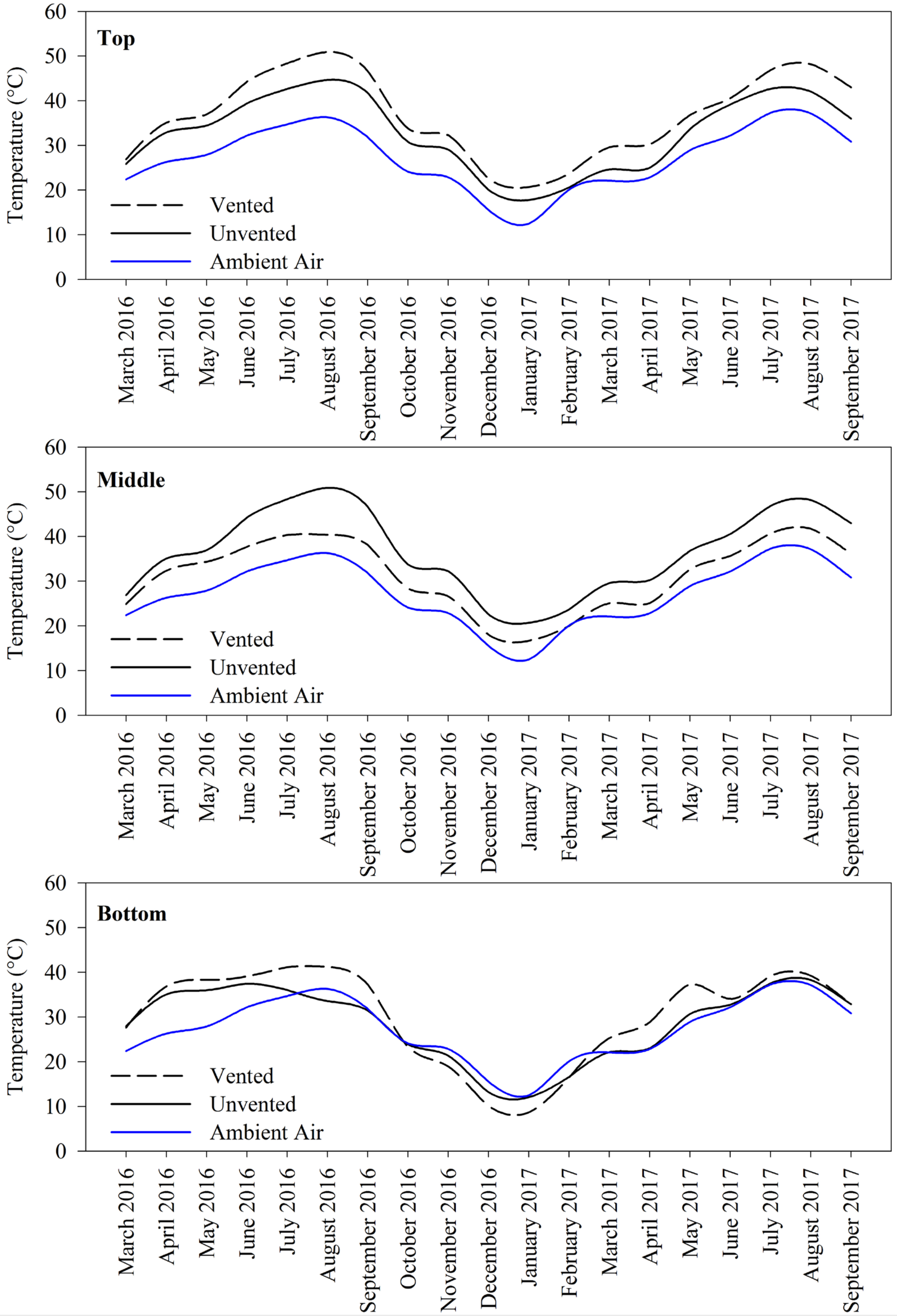

Figure 6 - 2016 and 2017 river stage and average bank-full elevation (meters above sea level) at the study site.

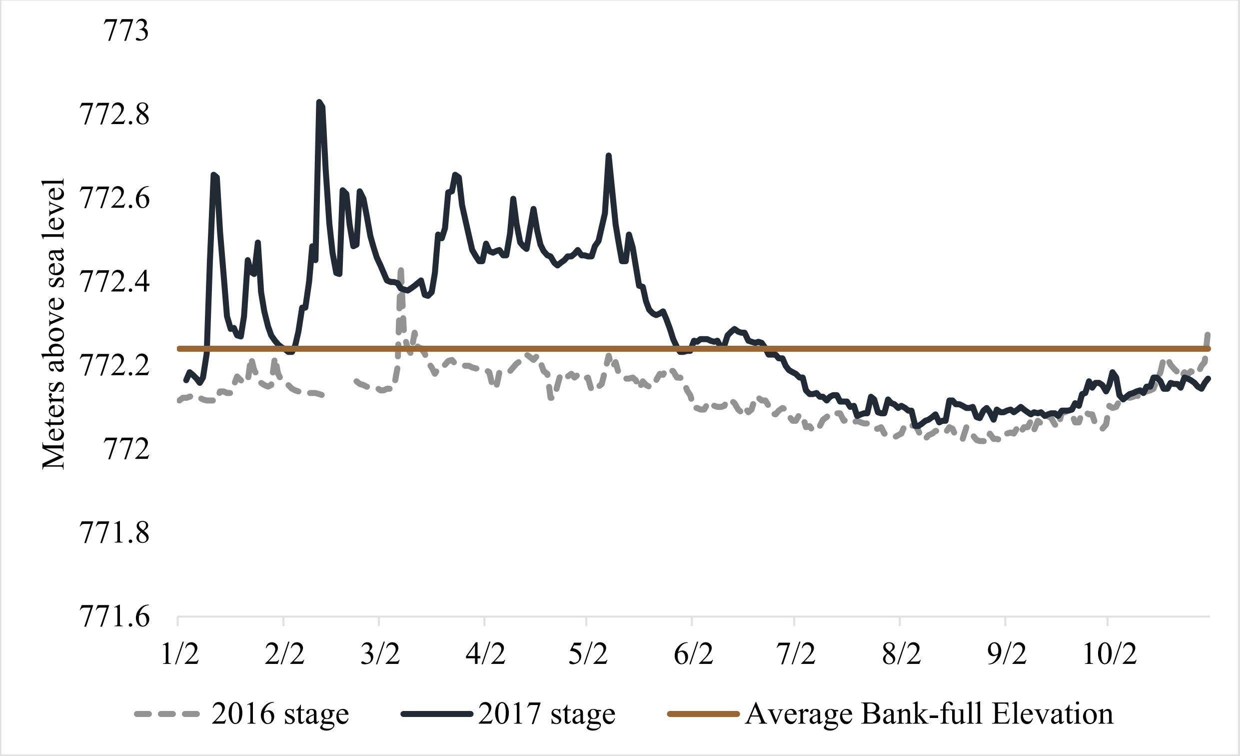

Figure 7 - 2016 groundwater levels (meters) on the primary axis. Groundwater drawdown (cm/day) on the secondary axis. Planting depth = 0.91 meters below the average planting elevation at the study site. Ground elevation is averaged for the study site.

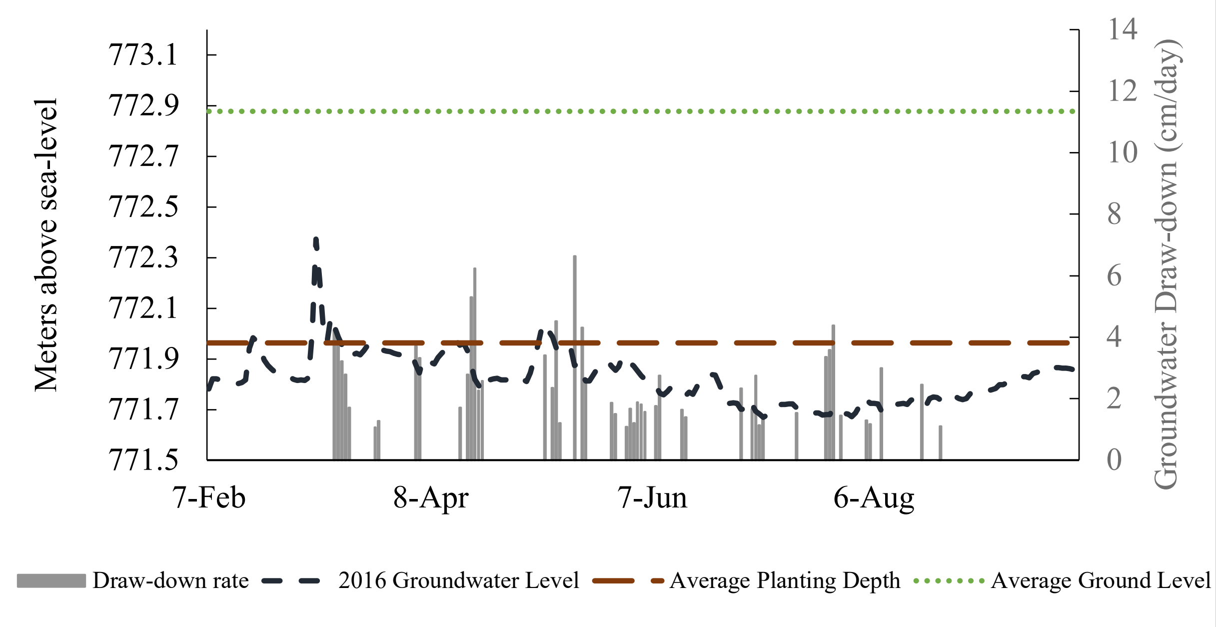

Figure 8 - 2017 groundwater levels (meters) on the primary axis. Groundwater drawdown (cm/day) on the secondary axis. Planting depth = 0.91 meters below the average planting elevation at the study site. Ground elevation is averaged for the study site.

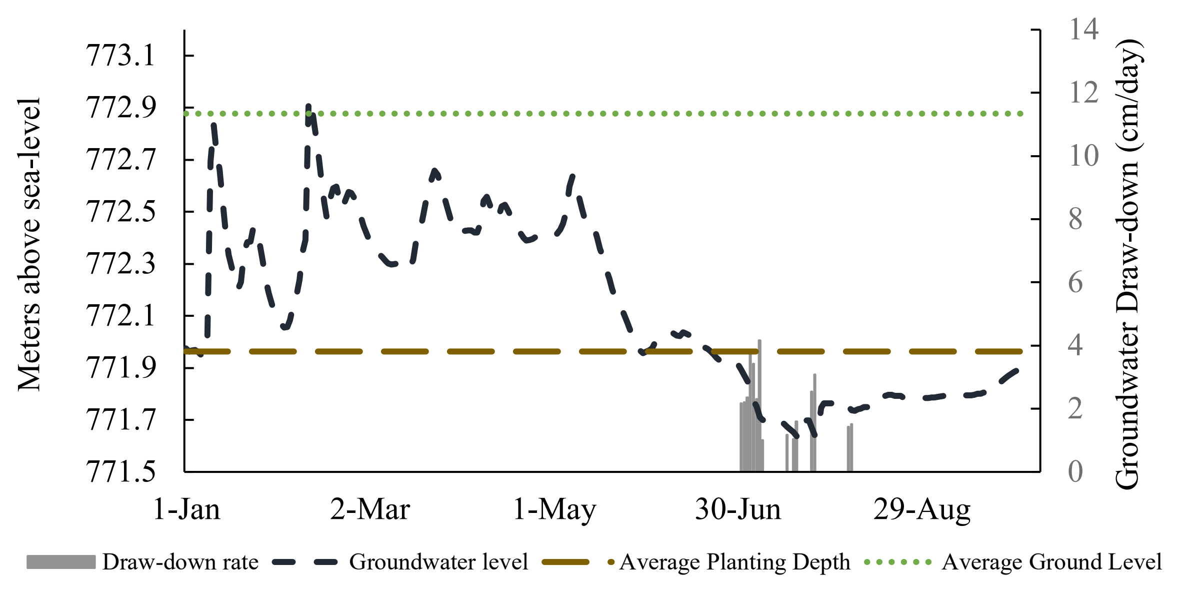

Figure 9 - Photo 1 (left) of ‘vented’ Tree Pro Max Grow Tube. Photo 2 (right) of ‘unvented’ Protex ProGro.

Figure 10 - Photo of study area facing east including plantings and the Shasta River during primary growing season in 2017.