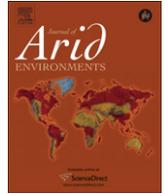




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Using existing agricultural infrastructure for restoration practices: Factors influencing successful establishment of *Populus fremontii* over *Tamarix ramosissima*

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ABSTRACT

Dryland riparian restoration often requires the use of large amounts of surface water. Retired agricultural fields located on floodplains associated with shallow water tables may minimize water requirements for restoration because irrigation could be ceased when target species become phreatophytic. Using existing agricultural infrastructure, we implemented two irrigation treatments over two years to promote native Fremont cottonwood (*Populus fremontii*) establishment over non-native tamarisk (*Tamarix ramosissima*): (1) frequent shallow irrigation; and (2) infrequent deep irrigation. We assessed how these treatments: (a) contributed to above- and below-ground growth of native and non-native species; (b) promoted native species survival; and (c) supported a competitive advantage of *P. fremontii* over *T. ramosissima*. Overall growth rates were highest one year after seeding, and average *P. fremontii* growth rates were higher than *T. ramosissima* in both years. Overall *P. fremontii* mortality was lower than *T. ramosissima* in both years. *P. fremontii* crown cover consistently increased and was ultimately the most abundant species in the overstory. Our results suggest that *P. fremontii* can out-compete *T. ramosissima* in these retired agricultural settings. In addition, this study suggests that irrigation frequency can be further decreased while reducing irrigation input, thus minimizing both effort and surface water required for restoration.

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1. Introduction

Altered hydrologic regimes due to dam construction and water diversion in much of the western United States (US) are associated with declines in native riparian forests (Rood and Mahoney, 1990, 1991). Flow alterations can diminish overall cottonwood (*Populus* spp.) recruitment and consequently prevent the establishment of large stands (Merritt and Poff, 2010; Mortenson and Weisberg, 2010). This is largely because, unlike floods in the past, regulated flows do not scour vegetation to create conditions favorable for cottonwood recruitment and survival (Howe and Knopf, 1991; Taylor et al., 1999). Instead, current altered hydrology has increased soil salinity and decreased soil moisture, now typical characteristics of many western river systems (Pataki et al., 2005; Sprenger et al., 2002). These new conditions provide an opportunity for invasive

species such as tamarisk (*Tamarix* spp.) to replace native species such as *Populus* spp. because invasive species are often better adapted to abiotic stresses such as high soil salinity and low soil moisture (Jackson et al., 1990; Nagler et al., 2008).

The success of invasive species establishment in traditional native habitat is also attributed to phenology and seed dispersal. Both *Populus fremontii* and *Tamarix ramosissima* produce wind- and water-dispersed seeds that only remain viable for a few weeks (Horton et al., 1960). Typically, *Populus* spp. are limited because they disperse seeds within a short window (late spring to early summer) while *Tamarix* spp. disperse seeds multiple times throughout the growing season enabling germination whenever soil conditions are suitable (Harris, 1966; Horton et al., 1960; Ware and Penfound, 1949). While this may give *Tamarix* spp. a recruitment advantage, studies have indicated a superiority of *Populus* spp. over *T. ramosissima* during establishment when water is not limiting (Everitt, 1995; Marler et al., 2001; Sher et al., 2000, 2002). As a result, many restoration efforts have aimed to regenerate riparian vegetation by providing supplemental surface water during native seed dispersal to promote moist soils for native seed

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germination (Friedman et al., 1995; Sprenger et al., 2002; Taylor et al., 1999).

Restoring native riparian habitat for the benefit of native wildlife species is an objective of the Lower Colorado River (LCR) Multi-Species Conservation Program (MSCP). Of particular interest is successful restoration that minimizes costs and resources while minimizing or excluding non-native species, such as *Tamarix* spp. Current restoration typically consists of establishing native species by one of four methods: (1) planted pole cuttings (Dreesen et al., 2002; Landis et al., 2006); (2) out-planted nursery stock (Landis et al., 2006; Pinto and Landis, 2004); (3) natural seed dispersal (Bhattacharjee et al., 2006; Johnson, 1965; Taylor et al., 2006); or (4) native seed collection and distribution (Bhattacharjee et al., 2006; Friedman et al., 1995; GSA, 2009; Raulston, 2003; Sprenger et al., 2002; Taylor et al., 2006). Because each method has advantages and disadvantages, the method or combination of methods chosen likely will depend on how site characteristics (e.g. soil texture and salinity, groundwater quality, above and below groundwater availability) and location affect overall feasibility and likelihood of success.

Minimal drought tolerance and high water use of *Populus* spp. and *Salix* spp. coupled with high evapotranspirative demand in the southwestern US results in substantial water requirements when these keystone species are targeted for riparian restoration. However, restoration sites in areas with shallow groundwater may require less irrigation water than other restoration locations because irrigation could be decreased or ceased when target species become phreatophytic. As many agricultural fields are located on traditional floodplains along rivers in the southwestern US, one promising restoration method is to convert retired agricultural fields into areas that support native riparian species. By using existing canal infrastructure for controlled irrigation practices, watering schemes can be designed to promote the ability of native keystone species such as *Populus* spp. and *Salix* spp. to co-exist with or out-compete invasive species such as *Tamarix* spp. While successful seedling establishment has been demonstrated using these practices (GSA, 2009), the long-term success of this restoration method remains uncertain.

Inter-seasonal water availability (e.g. total seasonal water, pulse size, length between recharge events, number of dry days) has been shown to differentially affect both growth rate and survival among species (Sher et al., 2004). In the context of global climate change, larger and less frequent rainfall events are expected in the southwestern United States (Seager et al., 2007) thus research addressing pulse size and frequency is important to understand how species will react to temporal variations in water supply (Easterling et al., 2000). Riparian restoration that aims to restore native mesic species while accounting for potential non-native species should consider how surface water inputs will impact the establishment of both functionally similar species. Here we examine two irrigation treatments implemented in 2008 using existing agricultural infrastructure at a restoration site, one year after initial seeding: (1) frequent shallow irrigation; and (2) infrequent deep irrigation. We assess how these treatments: (a) contribute to above and below-ground growth of Fremont cottonwood (*P. fremontii*) and tamarisk (*T. ramosissima*); (b) promote *P. fremontii* survival; and (c) support a competitive advantage of *P. fremontii* over *T. ramosissima*.

Specifically, this research addresses three main hypotheses: in an irrigation treatment applying a large amount of water less frequently versus less water more frequently, (1) we will observe higher *P. fremontii* and *T. ramosissima* terminal growth rates; (2) we will observe higher *P. fremontii* and *T. ramosissima* overall mortality; and (3) we will observe higher mortality in smaller trees versus larger trees. An infrequent irrigation regime introduces a more stressful environment and should provoke more competition for

resources compared to a frequent watering regime. However, an infrequent irrigation regime may also have increased oxygen availability to promote enhanced root metabolism. We anticipate that lower near-surface plant-available water imposed by an infrequent irrigation schedule should negatively impact smaller trees, which have a less-extensive rooting system compared to larger trees. In addition, we predict that *P. fremontii* will dominate both crown cover and total cover percentage regardless of irrigation treatment. Depending on restoration objectives, these results can be used to provide management recommendations for optimal irrigation depth and frequency when *P. fremontii* is the target species of interest.

2. Methods

2.1. Study site

Cibola National Wildlife Refuge (CNWR) was established in 1964 in order to restore and protect historic habitat and wintering grounds for migratory birds and other wildlife. CNWR is located between Yuma, AZ and Blythe, CA and straddles the LCR in La Paz County, AZ and Imperial County, CA (Fig. 1). Annual temperature ranges from 4 °C to 49 °C and rainfall averages under 100 mm yr⁻¹. More than 4000 ha consists of riparian vegetation, over half of which is near-monoculture *Tamarix* spp. (i.e. >90%) with less than half consisting of mixed native species including *P. fremontii*, Goodding's willow (*Salix gooddingii*), and coyote willow (*Salix exigua*; Nagler et al., 2008). Other native species include facultative phreatophytic arrowweed (*Pluchea sericea*), quailbush (*Atriplex lentiformis*), fourwing saltbush (*Atriplex canescens*), honey mesquite (*Prosopis glandulosa*) and screwbean mesquite (*Prosopis pubescens*, Nagler et al., 2008). Our study site, Field 51 on Farm Unit 1 of CNWR, is located 1.5 km east of the Colorado River (33°22'03" N, 114°40'50" W; 71 m elevation) in La Paz County, AZ. Field 51 is an 8-ha retired agricultural field located on the historic floodplain terrace and is managed by the Bureau of Reclamation (BOR) and CNWR for LCR MSCP research and demonstration.

Near-surface (i.e. 0–50 cm below ground surface) soil at Field 51 is predominantly silt and silty loam, but sand and sandy loam strata are prevalent at depth (~1 m below ground surface) toward the southern and central portion of Field 51 (GSA, 2008). Prior to irrigation on Field 51 for restoration purposes, average surface soil salinity was less than 2.0 dS m⁻¹ (GSA, 2008). Depth to groundwater, measured with on site well-point piezometers ranged seasonally from 2.2 to 2.6 m below ground surface prior to irrigation (GSA, 2008).

2.2. Study design

As part of a small-scale research project implemented in May 2007 on Field 51, 36, 6-m × 12-m, test plots were seeded with *P. fremontii*, *S. gooddingii*, and *S. exigua*. *Salix* spp. establishment was minimal (0.4 stems m⁻²) while *P. fremontii* establishment was high (17.6 stems m⁻²). Additionally, *T. ramosissima* volunteer establishment was higher than all seeded species (25.0 stems m⁻²). The co-establishment of *P. fremontii* and *T. ramosissima* created an opportunity to monitor intra- and inter-specific competition and the long-term survival of desired riparian species.

Beginning in March 2008, two different watering regimes were chosen to test effects of different irrigation depth and frequency on growth and survival of *P. fremontii* and *T. ramosissima* yearlings. The first irrigation treatment (*T*₁) consisted of 7-cm applied water once per week while the second irrigation treatment (*T*₂) consisted of 21-cm applied water once per three weeks. Beginning in March, 2009, *T*₁ and *T*₂ were altered to 5.5 cm approximately once per

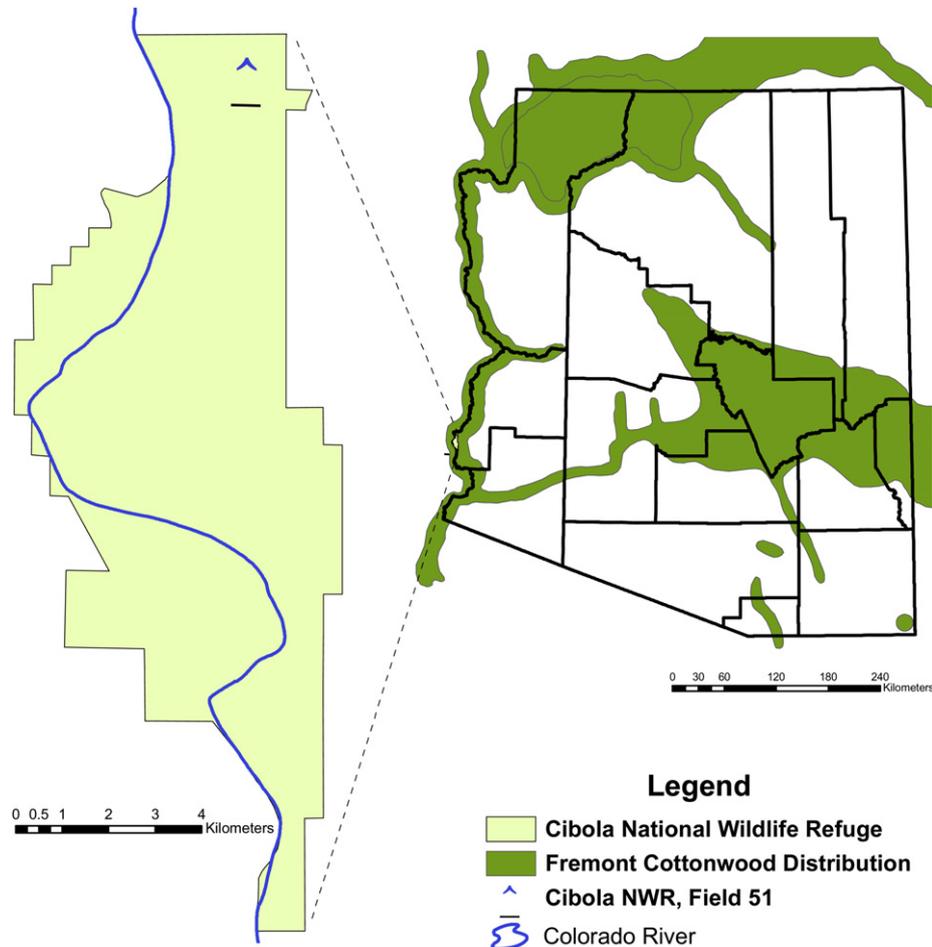


Fig. 1. Site map of Cibola National Wildlife Refuge, La Paz County, Arizona. Field 51 is a retired agricultural field located near the northern refuge boundary 1.5 km east of the Colorado River.

week and 22 cm once every four to five weeks, respectively, to promote greater depletion of available soil water between irrigation events. Irrigation was applied according to irrigation treatment specifications between March 1 and October 31 of each year to correspond with the growing season.

The 36 test plots were grouped into four blocks, and irrigation assignments were staggered to account for potential variation across the length and width of the study site. The northeast and southwest blocks received T_1 while the northwest and southeast blocks received T_2 (Appendix 1, Fig. 1, electronic version only). Vegetation surveys were conducted in each study plot (nine per block, 18 per irrigation treatment) for the duration of the study (Appendix 1, Fig. 1, electronic version only).

2.3. Measurements

2.3.1. Vegetation measurements

All above-ground measurements were made in two field campaigns for two consecutive years (2008–2009). In both years, the first campaign was initiated early in the growing season (i.e. May–June) and the second was initiated near the end of the growing season (i.e. Sept–Oct) before fall leaf-drop. The point-intercept method (RMWU, 2005) was used to document changes in overstory (crown) and total cover percentages through time (i.e. $\text{Cover \%} = (x/n) \times 100$, where x is the number of “hits” of a given cover type, and n is the number of sample points per plot). Three 6-

m transects were randomized within each plot (Appendix 1, Fig. 1, electronic version only). Every cover type below a sample point, spaced at 30.5-cm intervals along each transect, was recorded a maximum of once per point (a “hit”). The first cover type below the sample point represented the crown cover whereas total cover included both overstory and understory; crown cover indicates the tallest species in the observation area, whereas total cover represents total abundance.

Growth rates were measured in three permanent 1-m \times 0.5-m quadrats within all plots. The placement of each quadrat was randomized along the three transects used for conducting cover percentage surveys (Appendix 1, Fig. 1, electronic version only). Every *P. fremontii* and *T. ramosissima* within the quadrats was assigned a unique identification number using an aluminum tag attached to the base of each trunk with bailing wire. Terminal height (i.e. the total height from the base of the tree to the end of the main branch or stem) of all target trees was measured using a 1-m or 3.5-m measuring stick before and after each growing season to obtain terminal growth rates ($\text{cm growing season}^{-1}$).

Mortality each growing season was determined using data collected within the permanent quadrats where any tagged target tree that was dead upon subsequent surveys was documented (i.e. $\text{Mortality \%} = (d/n) \times 100$, where d is the total number of deaths during the growing season and n is the total number of trees documented in the spring survey). Species-specific stem densities (stems m^{-2}) were also calculated using the permanent quadrats

(i.e. Stem density = (s/A) where s is the number of trees of a particular species and A is the area within each quadrat (0.5 m²).

2.3.2. Below ground measurements

In each plot, continuous estimates of volumetric water content (θ_{15}), soil temperature (T_{soil}), and pore water salinity (EC) were determined at 15 cm below ground surface using ECH₂O-TE sensors (Decagon Devices, Inc., Pullman, WA). Additionally, volumetric water content was monitored at 46 cm (θ_{46}) and 91 cm (θ_{91}) below ground surface in each plot using EC-10 sensors (Decagon Devices, Inc., Pullman, WA). Groundwater depth and salinity were measured at five well-point piezometers each instrumented with a WL16 Water Level Logger (Global Water Instrumentation, Inc., Gold River, CA) programmed to monitor groundwater elevations every 4 h. Groundwater salinity was determined by testing bailed water samples with a field EC m and laboratory geochemical analysis. Soil texture and salinity were characterized in November 2007 for 1-m soil core samples taken from the center of each plot using a JMC Environmentalist's Subsoil Probe (ESP; Clements Associates Inc., Newton, IA). Soil texture was determined using a hydrometer and soil EC was determined using the saturated paste extract method (Rhoades, 1986).

A qualitative root survey was performed in October 2008. Trenches were excavated to 1.5 m below ground surface in a total of 10 plots chosen to represent the range of variability in *P. fremontii* and *T. ramosissima* density, to examine root size, growth, and penetration toward groundwater. This survey resulted in destructive sampling through 10 permanent quadrats, thus 2009 growth rates, cover percentages, and mortality were normalized with the exclusion of those quadrats.

2.4. Data analysis

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

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

3. Results

3.1. Growth rates

Overall growth rates were highest in 2008, one year after seeding (i.e. the second growing season). Average *P. fremontii*

Table 1

Size class breakdown where classes are separated by grouping trees into relative size categories based on their ratio to the tallest tree within each species sampling distribution. Total sample numbers (n) are given for both T_1 and T_2 and the range of sizes are given in each category for both 2008 and 2009 for both *P. fremontii* and *T. ramosissima*.

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

growth rates were significantly higher than *T. ramosissima* in both growing seasons. On average in the 2008 growing season *P. fremontii* grew 57.3 cm, but growth rates decreased by 70% in 2009. *T. ramosissima* grew 7.9 cm on average in 2008, but growth rates decreased by 43% in 2009 (Fig. 2a). Average growth rates within each species were not significantly different between T_1 and T_2 in either growing season for either species (Fig. 2a). In fact, excluding *T. ramosissima* growth rates in the 60–80% size category in 2009, T_1 and T_2 did not yield disparate growth rates within any size classes for either species (Fig. 2b). Larger size classes exhibited significantly greater growth rates during 2008. However, during 2009, growth rates were highly variable and larger size classes did not always have in greater growth rates (Fig. 2b).

3.2. Cover percentages

Cover percentage surveys were conducted to examine overall trends at the study site. Because they did not result in significant differences, T_1 and T_2 were combined for both crown cover and total cover percentage analyses. *P. fremontii* was the only species in which crown cover percentage increased during every survey and, as a result, was the most prevalent cover type in the overstory beginning in spring 2008. *T. ramosissima* crown cover percentage reached a plateau at 12% in 2009 while *P. fremontii* increased to 70% by fall of 2009 (Fig. 3a). Grasses and sedges decreased in crown cover from 81% in the fall of 2007 to 15% in the fall of 2009 while shrubs and forbs crown cover was reduced to 0% by fall 2009 (Fig. 3a).

P. fremontii and grasses and sedges were the most common cover types within the total cover after three growing seasons at 72% and 62%, respectively (Fig. 3b). *T. ramosissima* cover increased every season, except for a small decrease in the spring of 2009. Shrubs and forbs cover experienced a reduction from 47% in fall 2007 to less than 1% in 2009 (Fig. 3b).

3.3. Mortality

Overall *P. fremontii* mortality was lower than *T. ramosissima* during both 2008 and 2009 growing seasons, although it was only significantly lower in 2008 (Fig. 4a). *P. fremontii* mortality was similar between 2008 and 2009 (5.9% and 5.4%, respectively), however, *T. ramosissima* experienced higher mortality in 2008 (11.6%) compared to 2009 (7.6%, Fig. 4a). No significant differences were observed in same species *P. fremontii* or *T. ramosissima*

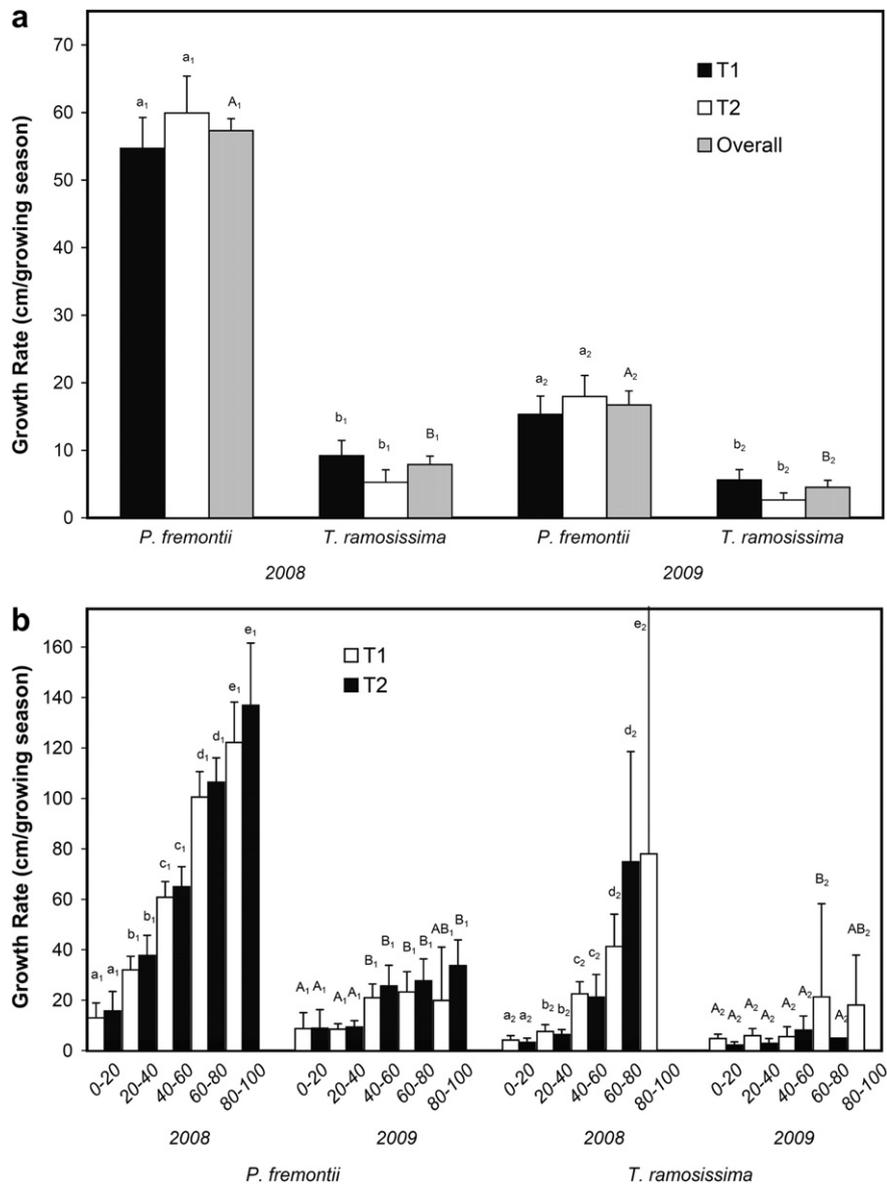


Fig. 2. *P. fremontii* and *T. ramosissima* growth rates documented during the 2008 and 2009 growing seasons. (a) Growth rates documented in T₁ (black) and T₂ (white) as well as overall growth rates (grey). (b) Growth rates by size class where trees were put into size categories according to relative size to the tallest tree in each species sampling distribution (e.g. 0–20% indicates all trees that were 0–20% the total height of the tallest tree). Error bars are one standard deviation and separate tests for significance are indicated by matching caps and subscripts of alphanumeric symbols.

mortality between T₁ and T₂ in 2008 or 2009 (Fig. 4a). The smallest size class, 0–20% of maximum tree height, accounted for the majority of both *P. fremontii* and *T. ramosissima* deaths in both 2008 and 2009 (Fig. 4b). Excluding two *P. fremontii* trees in 2008, no deaths occurred in any classes above 40% of maximum tree height for either species (Fig. 4b). Same species mortality was similar in between T₁ and T₂ in all size classes for both species, excluding the absence of *T. ramosissima* deaths in 2009 within the 20–40% size category for T₁.

3.4. Stem density

Stem density was used to depict overall trends at the study site – again, the differences between T₁ and T₂ were not significant and therefore are not reported here. The Fall 2007 initial survey, four months after seeding, documented a higher abundance in *T. ramosissima* (25.0 stems m⁻²) compared to *P. fremontii* (17.6 stems m⁻²).

Both *P. fremontii* and *T. ramosissima* stem density decreased dramatically during 2008 due to high mortality, and continued to decrease slightly through spring 2009 (Fig. 5). In fall 2009, however, some *P. fremontii*, previously documented as “dead” in earlier surveys, re-sprouted. As a result of *P. fremontii* regeneration and higher *T. ramosissima* mortality, *P. fremontii* stem density was greater than *T. ramosissima* beginning in fall 2009 (Fig. 5).

3.5. Belowground characteristics

Near-surface soil temperature (T_{soil}) followed a typical ambient temperature pattern ranging from 10 °C in the winter to 31 °C in the summer. Salinity measured from 1-m soil cores in winter 2007 was highly variable among plots, ranging from 1.8 dS m⁻¹ to 26.9 dS m⁻¹; soil salinity at 15 cm below ground surface averaged 4.4 dS m⁻¹ and ranged from 1.8 to 22.3 dS m⁻¹; soil salinity at 46 cm was 8.4 dS m⁻¹ and ranged from 2.4 to 22.9 dS m⁻¹; soil salinity at

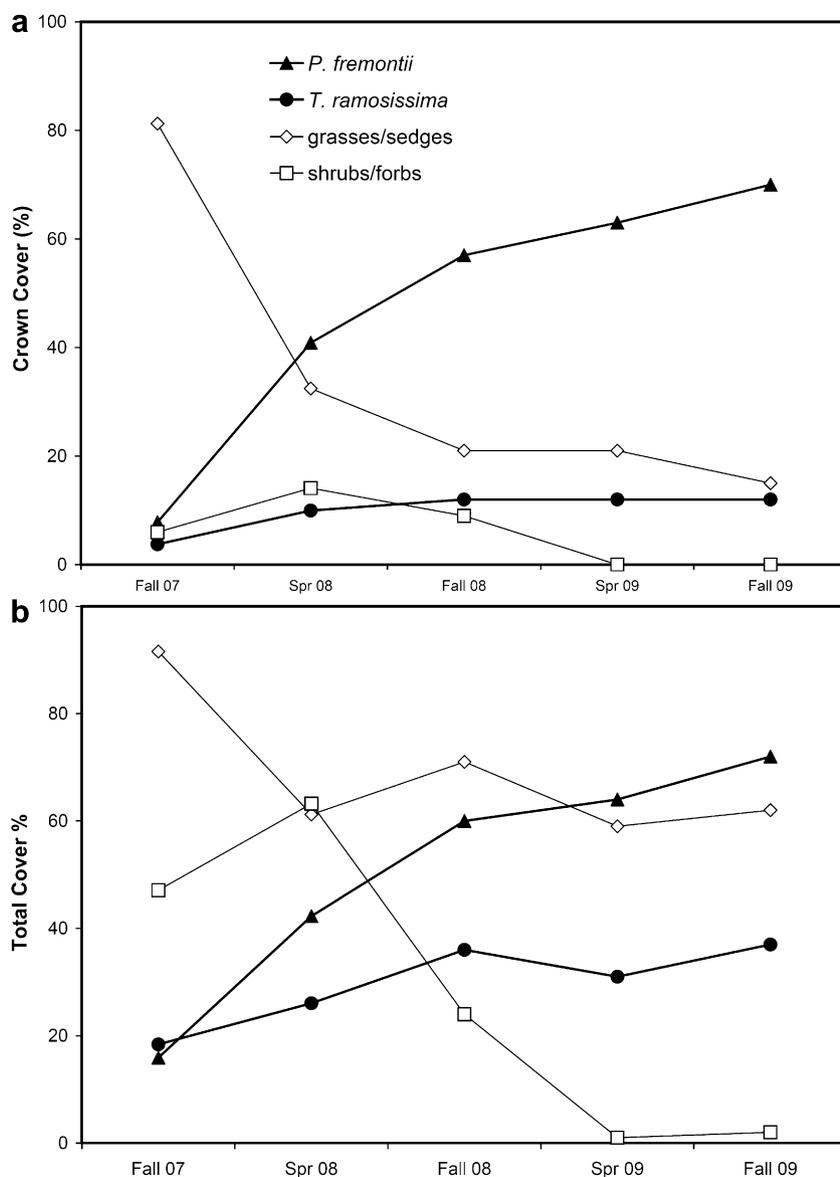


Fig. 3. Cover percentages documented throughout the project, May 2007–October 2009. (a) Crown (overstory) cover for four categories: *P. fremontii* (solid triangle), *T. ramosissima* (solid circle), combined grasses and sedges (Graminoid species, hollow diamond), combined shrubs and forbs (hollow square). (b) Canopy (understory) cover for the same four categories. Error bars for one standard deviation were removed for clarity, but were very small denoting significant differences among all categories due to large sample size.

depth (91 cm) was 6.3 dS m^{-1} and ranged from 0.5 to 26.9 dS m^{-1} . Surface soil salinity tended to be higher near the northern portion of the study site and lower soil salinity was observed at depth in coarser, sandy soils. Groundwater salinity measured in March, April, and June 2009 ranged from 1.6 to 2.7 dS m^{-1} .

Average soil moisture was similar between T_1 and T_2 at all depths (Appendix 2, Fig. 2a and b, electronic version only). θ_{15} increased rapidly (maximum = $0.44 \text{ cm}^3 \text{ cm}^{-3}$) in response to irrigation events and then decreased rapidly (minimum = $0.19 \text{ cm}^3 \text{ cm}^{-3}$) between events for both T_1 and T_2 . θ_{46} generally increased rapidly (maximum = $0.37 \text{ cm}^3 \text{ cm}^{-3}$) in response to irrigation events in T_1 and T_2 and then slowly decreased (minimum = $0.30 \text{ cm}^3 \text{ cm}^{-3}$) between irrigation events. Independent of T_1 and T_2 , soil moisture at depth (θ_{91}) remained relatively stable throughout the study but ranged from 0.27 to $0.35 \text{ cm}^3 \text{ cm}^{-3}$. Depth to groundwater during the project ranged from 1.65 to 2.50 m below ground surface and generally increased during the winter dormant season and was variable throughout the remainder of the study depending on

precipitation, irrigation input, and plant water use (Appendix 2, Fig. 2c, electronic version only). In addition, the qualitative root survey revealed many large *P. fremontii* and *T. ramosissima* roots penetrating greater than 1.5 m below ground surface in both T_1 and T_2 .

4. Discussion

4.1. Irrigation effects

Our results did not support our hypothesis that an irrigation treatment applying a large amount of water less frequently would promote higher terminal growth rates in *P. fremontii* and *T. ramosissima*. In fact, T_2 more adequately mimicked natural overbank flooding than T_1 because soil was inundated for a greater period of time. We believed that more extensive soil moisture depletion between irrigation events in T_2 would potentially benefit adventitious roots by promoting penetration toward groundwater and

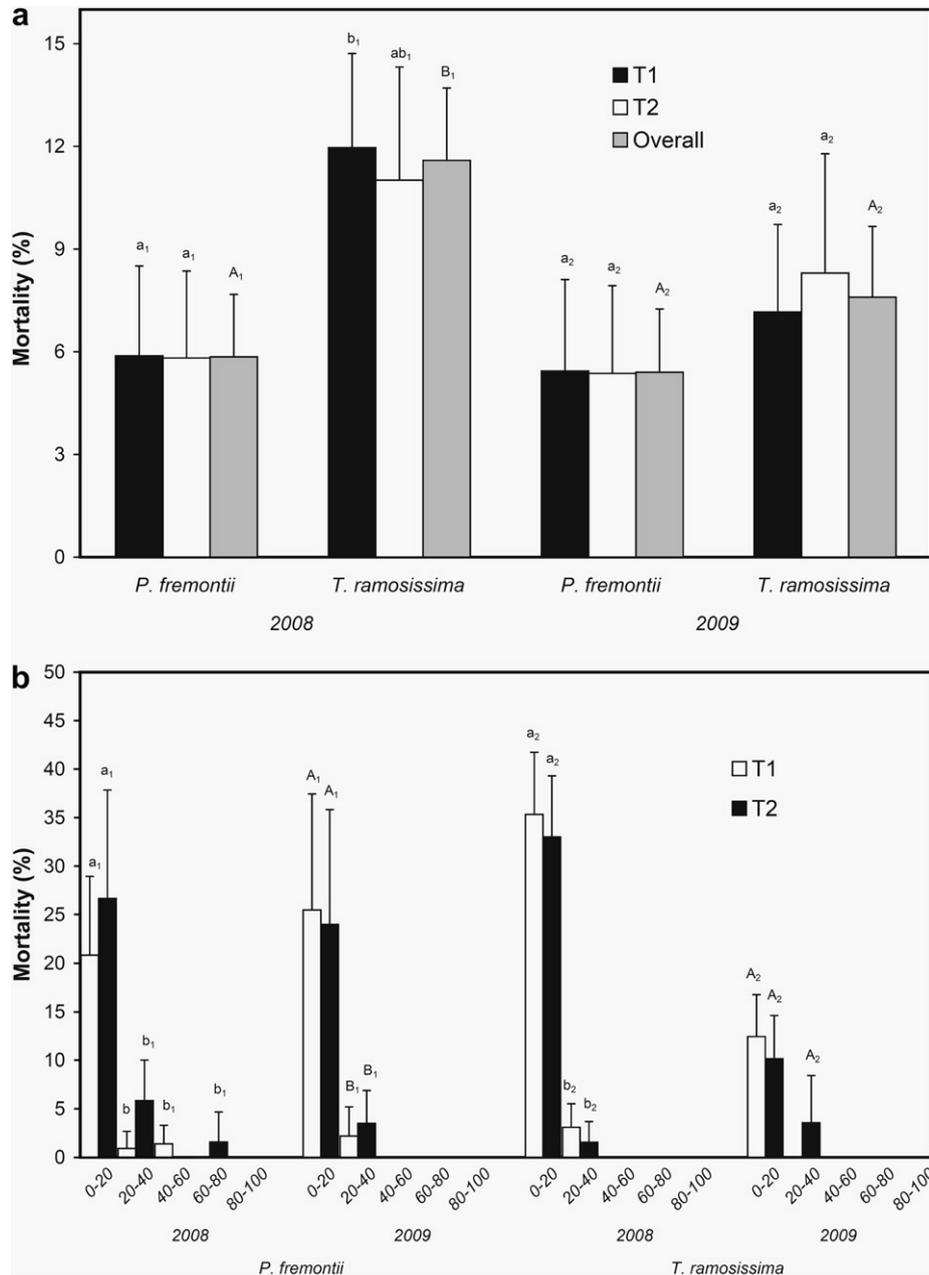


Fig. 4. *P. fremontii* and *T. ramosissima* mortality (expressed as a percentage) documented during the 2008 and 2009 growing seasons. (a) Mortality documented in T_1 (black) and T_2 (white) as well as overall mortality (grey). (b) Mortality by size class where trees were put into size categories according to relative size to the tallest tree in each species sampling distribution (e.g. 0–20% indicates all trees that were 0–20% the total height of the tallest tree). Error bars are one standard deviation and separate tests for significance are indicated by matching caps and subscripts of alphanumeric symbols.

providing greater oxygen for root metabolism. We anticipated that these factors would result in increased above and below-ground biomass. Our results, however, did not show any disparity in growth rates between T_1 and T_2 . In fact, both *P. fremontii* and *T. ramosissima* had similar growth rates in T_1 and T_2 for all size classes.

Analysis of volumetric water content at multiple depths allowed us to examine how irrigation treatments affected plant-available water through the soil profile. Rapid responses of θ_{15} to irrigation events with subsequent rapid depletion likely indicated drainage and near-surface evapotranspiration. Slower depletion of θ_{46} was likely due to drainage and plant water use. θ_{91} , however, was rarely affected by irrigation events and remained relatively stable. A sharp decline in θ_{91} was observed in November 2008, but this was

independent of irrigation events and was likely associated with a fluctuation in the water table or capillary fringe. Compared to T_1 , decreases in θ_{15} and θ_{46} were higher between irrigation events in T_2 whereas θ_{91} did not differ between T_1 and T_2 . In fact, θ_{91} did not fluctuate significantly from the non-growing season volumetric water content despite observations of roots at depths greater than 1.5 m. These results suggest that soil water within the rooting zone was not depleted by evapotranspiration. This could be due to: (1) irrigation water applied at a higher rate than plant water use of vadose zone soil moisture; and/or (2) capillary rise of groundwater that contributed to vadose zone soil moisture; and/or (3) tree use of both groundwater and vadose zone soil moisture such that dry conditions were never observed at depths below 90 cm. Likely,

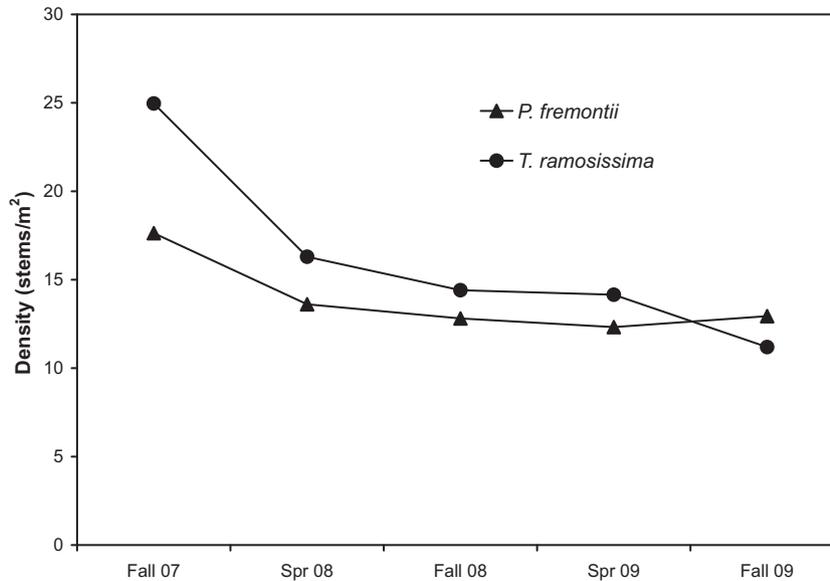


Fig. 5. Stem density (expressed as stems m^{-2}) documented throughout the project, May 2007–October 2009, for *P. fremontii* (solid triangle), *T. ramosissima* (solid circle). Error bars for one standard deviation were removed for clarity, but were very small denoting significant differences among all categories due to large sample size.

plant-available water was present at depths greater than 90 cm in plots of both treatments throughout the study period which may in part explain why both irrigation treatments resulted in similar growth rates and mortality. Furthermore, phreatophytes, such as our target species, are unlikely to compete for water if taproots reach the groundwater (Sher and Marshall, 2003). The root survey revealed roots of larger trees penetrating depths greater than 1.5 m, which likely allows these trees to access water from the capillary fringe or saturated zone, thus complicating the quantification of irrigation effects on growth and survival.

4.2. Community composition

Our results did not support our hypothesis that higher mortality would be observed in the infrequent, deep watering regime. We believed that drier conditions imposed by this treatment would result in greater die-off regardless of tree size. After analyzing soil moisture and depth to groundwater data, it is apparent that this treatment did not stress the target trees as much as we originally intended it to. In fact, the presence of stable year-round soil moisture at depth (θ_{91}), a shallow capillary fringe, and roots extending beyond 1.5 m in both T_1 and T_2 provide evidence that competitive interactions were likely suppressed. The frequency of T_2 was decreased from once every three weeks in the 2008 growing season to once every four to five weeks in the 2009 growing season to further decrease soil-water between irrigation events. However, there was still no treatment effect observed between T_1 and T_2 when observing growth rates or mortality in 2009.

Our results support our hypothesis that smaller trees would experience greater mortality, as the two smallest size classes accounted for the majority of die-off. Natural thinning was anticipated because both *P. fremontii* and *T. ramosissima* are known to be shade intolerant (Johnson et al., 1976; Taylor et al., 1999) and are considered high water users (Busch and Smith, 1995; DiTomaso, 1998; Sala et al., 1996; Weeks et al., 1987), both of which would be affected by high stem densities. Smaller trees within the understory may not have been able to compete with larger, established trees because they were likely impacted by reduced light resources and inferior root systems.

High mortality of smaller trees was responsible for shifting the overall stem density throughout the study. *T. ramosissima* initially established at a higher density than *P. fremontii* in 2007, but *P. fremontii* ultimately had a higher stem density in 2009. This shift in dominance indicates a competitive superiority of *P. fremontii* over *T. ramosissima* under the conditions imposed at our field site—*P. fremontii* has successfully outgrown and outnumbered *T. ramosissima* at this restoration site after three growing seasons.

4.3. Restoration implications

This project serves as an example of how agricultural infrastructure may be successfully utilized to promote high growth rates and survival of riparian phreatophytes where shallow groundwater is present on disconnected floodplains. While this study was performed at a relatively small-scale, our study area is typical of many regions along the LCR. Our objective was to research an optimal irrigation depth and frequency. Low overall mortality and sufficient soil moisture throughout the soil profile suggest that more water was applied in our two irrigation treatments than was necessary to maintain healthy *P. fremontii* stands. Additional studies that alter irrigation amount and frequency could offer an optimal watering schedule to further conserve water resources during establishment while still maintaining healthy riparian vegetation. Furthermore, it was evident that larger trees at this field site were phreatophytic after two growing seasons. Because there is still some debate on whether *P. fremontii* is a facultative or obligate phreatophyte (Busch et al., 1992; Butler et al., 2007; McQueen and Miller, 1972; Snyder and Williams, 2000), further research is necessary to understand how groundwater will affect the long-term success of these trees after irrigation is ceased.

In the future, effects of irrigation regimes on long-term soil salinity should be evaluated because high salinity soils can be detrimental to long-term growth success. Both sensor and lab results confirmed the presence of saline soil profiles that often approach or exceed the tolerance of cottonwood and willow species ($3\text{--}10\text{ dS m}^{-1}$). This study documented the successful establishment of *P. fremontii* and *T. ramosissima* trees in saline soils greater than 10 dS m^{-1} in a field setting, whereas Shafroth et al. (1995)

reported favorable *Populus* spp. germination in soil salinities up to 3.45 dS m⁻¹. Bhattacharjee et al. (2006) showed high recruitment in soil salinity averaging 4.19 dS m⁻¹, Siegel and Brock (1990) reported high *Populus* spp. germination percentage up to 4.6 dS m⁻¹, and Jackson et al. (1990) reported *Populus* spp. germination at soil salinity levels reaching 9.98 dS m⁻¹. Infrequent irrigation might result in increased salinity of both soil water and groundwater used by riparian vegetation, particularly if the irrigation regime results in an insufficient leaching fraction.

Specific restoration objectives must be addressed before restoration methods are established. For instance, many projects are focused on protecting avian communities and restoring or enhancing avian habitat. Avian habitat specialists benefit from vegetative species such as *Prosopis* spp., *Populus* spp., and *Salix* spp. (Meents et al., 1984). Lower avian abundance and diversity has been observed in *T. ramosissima* monocultures (Anderson and Ohmart, 1977; Cohan et al., 1978; Engel-Wilson and Ohmart, 1978), but management converting these monocultures to mixed native/*Tamarix* spp. habitat may improve avian abundance and diversity (Van Riper et al., 2008). Furthermore, high-density native communities with shorter canopies may be desired to satisfy habitat requirements for the Southwestern willow flycatcher (*Empidonax traillii extimus*); however, low initial densities will favor larger *Populus* spp. forests, not preferred by the Southwestern willow flycatcher (Taylor et al., 2006). Initial project goals at this study site were to restore native mesic species on a retired and cleared agricultural field located on a hydraulically-disconnected floodplain; but specifically, a high density of mixed native species were targeted to create avian habitat while suppressing potential *T. ramosissima* recruitment. Unfortunately, *S. gooddingii* and *S. exigua* originally seeded at this site did not establish successfully while *T. ramosissima* proliferated resulting in lower native woody diversity than desired. However, passive recruitment of native species from natural seed dispersal such as *Prosopis* spp., *Pluchea* spp., *Atriplex* spp., and *Baccharis* spp. was common. In many cases, completely eradicating *Tamarix* spp. may be unrealistic, undesirable, or prohibitively expensive (Shafroth et al., 2005; Van Riper et al., 2008). Restoration practices should take into account the feasibility of minimizing versus excluding *Tamarix* spp. to decide whether an investment is necessary to meet project objectives.

Although some projects have been successful at restoring a small number of species and age classes along regulated rivers, they do not address long-term dynamics and may not restore native species diversity or landscape heterogeneity to pre-impact levels (Stromberg et al., 2007). This is partly because certain methods (e.g. pole cuttings) can bypass the need to restore fluvial processes and the use of supplemental irrigation may bypass the need to recharge aquifers that moisten rhizospheres (Stromberg et al., 2007). The use of existing agriculture infrastructure is likely to continue to become an important component in meeting more large-scale restoration goals along the LCR. River restoration along well-regulated river systems will have to make realistic objectives and decide what outcomes are acceptable when making efforts to return a disconnected floodplain toward its native state.

5. Conclusions

The limiting similarity concept suggests that functionally similar species are more likely to compete and will not co-exist due to competitive exclusion (Abrams, 1983; MacArthur and Levins, 1967). Our study supports a level of co-existence between both *P. fremontii* and *T. ramosissima*, two functionally similar riparian species, through three growing seasons. While high *P. fremontii* growth rates persisted within dense tree stands inter-mixed with *T. ramosissima* at our study site (Appendix 3, Fig. 3a and b, electronic

version only), extended monitoring is necessary to document longer-term response to initial establishment. Our study was unique in that all vegetation was grown from the same start date on a mechanically cleared, retired agricultural field in which no manual thinning or herbicidal treatment was introduced during the experiment (in 2007, precluding our experiment, Arrow 2EC [Arysta LifeScience North America, LLC, Cary, North Carolina] grass-specific herbicide was applied to all plots on four occasions to restrict grass growth and promote seedling emergence through the canopy). Our research is consistent with previous studies that have highlighted the ability of *Populus* spp. to successfully out-compete *Tamarix* spp. during establishment when water is not limiting (Everitt, 1995; Marler et al., 2001; Sher et al., 2000, 2002). Regardless of irrigation treatment, *P. fremontii* had higher growth rates, lower mortality, and dominated crown and total cover percentages after three growing seasons. Current trends suggest that *P. fremontii* will continue to dominate *T. ramosissima* over time.

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Appendix. Supplementary material

Supplementary data related to this article can be found online at doi:10.1016/j.jaridenv.2011.03.013

References

- Abrams, P., 1983. The theory of limiting similarity. *Annual Review of Ecology and Systematics* 14, 359–376.
- Anderson, B.W., Ohmart, R.D., 1977. Vegetative structure and bird use in the lower Colorado River valley. In: Importance, Preservation, and Management of Riparian Habitat: A Symposium USDA Forest Service General Technical Report RM-43.
- Bhattacharjee, J., Taylor Jr., J.P., Smith, L.M., 2006. Controlled flooding and staged drawdown for restoration of native cottonwoods in the Middle Rio Grande Valley, New Mexico, USA. *Wetlands* 26, 691–702.
- Busch, D.E., Ingraham, N.L., Smith, S.D., 1992. Water-uptake in woody riparian phreatophytes of the southwestern United States: a stable isotope study. *Ecological Applications* 2, 450–459.
- Busch, D.E., Smith, S.D., 1995. Mechanisms associated with the decline of woody species in riparian ecosystems of the southwestern U.S. *Ecological Monographs* 65, 347–370.
- Butler, J.J., Kluitenberg, G.J., Whitmore, D.O., Loheide, S.P., Jin, W., Billinger, M.A., Zhan, X.Y., 2007. A field investigation of phreatophyte-induced fluctuations in the water table. *Water Resources Research* 43, W02404.
- Cohan, D.R., Anderson, W., Ohmart, R.D., 1978. Avian population responses to saltcedar along the lower Colorado River. USDA Forest Service General Technical Report WO-12.
- DiTomaso, J., 1998. Impact, biology, and ecology of saltcedar (*Tamarix* spp.) in the southwestern United States. *Weed Technology* 12, 326–336.
- Dreesen, D., Harrington, J., Subirge, T., Stewart, P., Fenchel, G., 2002. Riparian restoration in the Southwest: species selection, propagation, planting methods, and case studies. In: Dumroese, R.K., Riley, L.E., Landis, T.D., Technical Coordinators (Eds.), *National Proceedings: Forest and Conservation Nursery Associations—1999, 2000, and 2001*. USDA Forest Service, Rocky Mountain Research Station, Ft Collins (CO), pp. 253–272. *Proceedings RMRS-P-24*.
- Easterling, D.R., Meehl, G.A., Parmesan, C., Changnon, S.A., Karl, T.R., Mearns, L.O., 2000. Climate extremes: observations, modeling, and impact. *Science* 289, 2068–2074.
- Engel-Wilson, R.W., Ohmart, R.D., 1978. Floral and attendant faunal changes on the lower Rio Grande between Fort Quitman and Presidio, Texas. USDA Forest Service General Technical Report WO-12.

- Everitt, B.L., 1995. Hydrologic factors in regeneration of Fremont cottonwood along the Fremont River, Utah. *Geophysical Monographs* 89, 197–208.
- Friedman, J.M., Scott, M.L., Lewis, W.M., 1995. Restoration of riparian forests using irrigation, artificial disturbance, and natural seed fall. *Environmental Management* 19, 547–557.
- GeoSystems Analysis, Inc., March 31, 2008. Site Characterization: Cibola National Wildlife Refuge Field 51 Prepared for Bureau of Reclamation, Lower Colorado Region.
- GeoSystems Analysis, Inc., Feasibility Study Using Native Seeds in Restoration, California–Arizona–Nevada (Bureau of Reclamation Contract No. 06CR308057) – 2008 Annual Report. Prepared for Bureau of Reclamation, March 3, 2009.
- Harris, D.R., 1966. Recent plant invasions in the arid and semi-arid southwest of the United States. *Annals of the Association of American Geographers* 56, 408–422.
- Howe, W.H., Knopf, F.L., 1991. On the imminent decline of Rio Grande cottonwoods in central New Mexico. *Southwestern Naturalist* 36, 218–224.
- Horton, J.S., Mounds, F.C., Kraft, J.M., 1960. Seed Germination and Seedling Establishment of Phreatophyte Species. U. S. Department of Agriculture, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO, USA. Paper No. 48.
- Jackson, J., Ball, J.T., Rose, M.R., 1990. Assessment of the Salinity Tolerance of Eight Sonoran Desert Riparian Trees and Shrubs. Final Report to the United States Department of the Interior Bureau of Reclamation, Reno, NV, USA.
- Johnson, R.L., January 1965. Regenerating cottonwood from natural seedfall. *Journal of Forestry* January, 33–36.
- Johnson, W.C., Burgess, R.L., Keammerer, W.R., 1976. Forest overstory vegetation and environment on the Missouri River floodplain in North Dakota. *Ecological Monographs* 46, 59–84.
- Landis, T.D., Dreessen, D.R., Pinto, J.R., Dumroese, R.K., 2006. Propagating native Salicaceae for riparian restoration on the Hopi Reservation in Arizona. *Native Plants Journal* 7.1, 52–60.
- Marler, R.J., Stromberg, J.C., Pattern, D.T., 2001. Growth response of *Populus fremontii*, *Salix gooddingii*, and *Tamarix ramosissima* seedlings under different nitrogen and phosphorus concentrations. *Journal of Arid Environments* 49, 133–146.
- MacArthur, R.H., Levins, R., 1967. The limiting similarity, convergence and divergence of coexisting species. *The American Naturalist* 101, 377–385.
- McQueen, I.S., Miller, R.F., 1972. Soil-moisture and Energy Relationships Associated with Riparian Vegetation Near San Carlos, AZ Professional Paper 655-E. U. S. Geological Survey, Washington D. C., USA.
- Meents, J.K., Anderson, B.W., Ohmart, R.D., 1984. Sensitivity of riparian birds to habitat loss. In: Warner, R.E., Hendrix, K.M. (Eds.), *California Riparian Systems: Ecology, Conservation, and Productive Management*. University of California Press, Berkeley and Los Angeles, California and London, England, pp. 619–625.
- Merritt, D.M., Poff, N.L., 2010. Shifting dominance of riparian *Populus* and *Tamarix* along gradients of flow alteration in western North American rivers. *Ecological Applications* 20, 135–152.
- Mortenson, S.G., Weisberg, P.J., 2010. Does river regulation increase the dominance of invasive woody species in riparian landscapes? *Global Ecology and Biogeography* 19, 562–574.
- Nagler, P.L., Morino, K., Didan, K., Erker, J., Osterberg, J., Hultine, K.R., Glenn, E.P., 2008. Removal of saltcedar (*Tamarix* spp.) will not facilitate the return of native trees nor save water on the lower Colorado River. *Ecology* doi:10.1002/eco.35.
- Pataki, D.E., Bush, S.E., Gardner, P., Solomon, D.K., Ehleringer, J.R., 2005. Ecology in a Colorado River riparian forest: implications for the decline of *Populus fremontii*. *Ecological Applications* 15, 1009–1018.
- Pinto, J.R., Landis, T.D., 2004. Propagating Plant Materials for the Hopi Reservation, Arizona USDA Forest Service Proceedings RMRS-P-33.
- Rangeland Monitoring in Western Uplands, 2005, Collecting Monitoring Data: Point Intercept Method. Part 9, p. 15.
- Raulston, B.E., 2003. Habitat Restoration on the Lower Colorado River, Demonstration Projects: 1995–2002. U.S. Bureau of Reclamation, Lower Colorado Regional Office, Boulder City, Nevada, p. 36.
- Rhoades, J.D., 1986. Soluble salts. Methods of soil analysis, Part 2. In: Page, A.L. (Ed.). *American Society of Agronomy*, Madison, WI, pp. 167–179.
- Rood, S.B., Mahoney, J.M., 1990. The collapse of riparian poplar forests downstream from dams on western prairies: probable causes and prospects for mitigation. *Environmental Management* 14, 451–464.
- Rood, S.B., Mahoney, J.M., 1991. The importance and extent of cottonwood forest decline downstream from dams. In: Rood, S.B., Mahoney, J.M. (Eds.), *The Biology and Management of Southern Alberta's Cottonwoods*. University of Lethbridge Press, Alberta Canada, pp. 1–9.
- Sala, A., Smith, S., Devitt, D., 1996. Water use by *Tamarix ramosissima* and other associated phreatophytes in the Mojave Desert Floodplain. *Ecological Applications* 6, 888–898.
- Seager, R., Ting, M.F., Held, I., Kushnir, Y., Lu, J., Vecchi, G., Huang, H.P., Harnik, N., Leetmaa, A., Lau, N.C., Li, C.H., Velez, J., Naik, N., 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316, 1181–1184.
- Shafroth, P.B., Friedman, J.M., Ischinger, L.S., 1995. Effects of salinity on the establishment of *Populus fremontii* (cottonwood) and *Tamarix ramosissima* (saltcedar) in southwestern United States. *Great Basin Naturalist* 55, 58–65.
- Shafroth, P.B., Cleverly, J.R., Dudley, T.L., Taylor, J.P., van Riper III, C., Weeks, E.P., Stuart, J.N., 2005. Control of *Tamarix* in the western United States: implications for water salvage, wildlife use, and riparian restoration. *Environmental Management* 35, 231–246.
- Sher, A.A., Marshall, D.L., Gilbert, S.A., 2000. Competition between native *Populus deltoides* and invasive *Tamarix ramosissima* and the implication for re-establishing flooding disturbance. *Conservation Biology* 14, 1744–1745.
- Sher, A.A., Marshall, D.L., Taylor, J.P., 2002. Establishment of native *Populus* and *Salix* in presence of invasive nonnative *Tamarix*. *Ecological Applications* 12, 760–772.
- Sher, A.A., Marshall, D.L., 2003. Seedling competition between native *Populus deltoides* (Salicaceae) and exotic *Tamarix ramosissima* (Tamaricaceae) across water regimes and substrate types. *American Journal of Botany* 90, 413–422.
- Sher, A.A., Goldberg, D.E., Novoplansky, A., 2004. The effect of mean and variance in resource supply on survival of annulus from Mediterranean and desert environments. *Oecologia* 141, 353–362.
- Siegel, R.S., Brock, J.H., 1990. Germination requirements of keys southwestern woody riparian species. *Desert Plants* 10, 3–8.
- Snyder, K.A., Williams, D.G., 2000. Water sources used by riparian trees varies among stream types on the San Pedro River, Arizona. *Agricultural and Forest Meteorology* 105, 227–240.
- Sprenger, M.D., Smith, L.M., Taylor, J.P., 2002. Restoration of riparian habitat using experimental flooding. *Wetlands* 22, 49–57.
- Stromberg, J.C., Beauchamp, V.B., Dixon, M.D., Lite, S.J., Paradzick, C., 2007. Importance of low-flow and high-flow characteristics to restoration of riparian vegetation along rivers in arid southwestern United States. *Freshwater Biology* 52, 651–679.
- Taylor, J.P., Wester, D.B., Smith, L.M., 1999. Soil disturbance, flood management, and woody riparian plant establishment in the Rio Grande floodplain. *Wetlands* 19, 372–382.
- Taylor, J.P., Smith, L.M., Haukos, D.A., 2006. Evaluation of woody plant restoration in the Middle Rio Grande: ten years after. *Wetlands* 26, 1151–1160.
- Van Riper III, C., Paxton, K.L., O'Brien, C., Shafroth, P.B., McGrath, L.J., 2008. Rethinking avian response to *Tamarix* on the lower Colorado River: a threshold hypothesis. *Restoration Ecology* 16, 155–167.
- Ware, G.H., Penfound, W.T., 1949. The vegetation of the lower levels of the floodplain of the South Canadian River in central Oklahoma. *Ecology* 30, 478–484.
- Weeks, E.P., Weaver, H.L., Campbell, G.S., Tanner, B.D., 1987. Water Use by Saltcedar and by Replacement Vegetation in the Pecos River Floodplain between Acme and Artesia, New Mexico U. S. Geological Survey Professional Paper 491-G, 33.