Restoration Treatments to Improve Native Riparian Tree Establishment Following Environmental Flow Releases in the Colorado River Delta, Mexico

By

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Thesis directed by Professor Sharon K. Collinge

Abstract

In the mid-1900s, the Colorado River Delta in Northwest Mexico stopped regularly receiving freshwater flows due to the diversion of Colorado River water for human consumptive use. Despite the highly altered hydrologic regime and resulting desiccation of 90% of Delta habitat, sporadic flood flows in the 80s and 90s regenerated riparian habitat along the river corridor in Mexico, suggesting ecosystem resilience. Since 2001 however, ongoing drought has led to a steady decline in ecosystem conditions in the Delta. In 2014, an experimental pulse flow of water was delivered to the Colorado River in Mexico as part of the U.S.-Mexico binational agreement, Minute 319, to determine if a designed flow could improve biologic and hydrologic conditions with a relatively small amount of water. Laguna Grande, a target restoration site in the central Delta, was not expected to receive high-magnitude flows needed to create conditions for riparian tree recruitment. To improve recruitment, areas were cleared of tamarisk (*Tamarix* spp.) and arrowweed (*Pluchea sericea*) prior to the pulse flow, and some areas were seeded with Goodding's willow (*Salix gooddingii*) and Fremont cottonwood (*Populus fremontii*) during flow recession to promote riparian tree establishment.

A controlled, replicated experiment was implemented to assess effects of four restoration treatments on riparian tree species recruitment: 1) control – no inundation from pulse flow, 2) inundated, 3) inundated and cleared, and 4) inundated, cleared, and seeded. Vegetation was monitored in March, May, and October of 2014 (before, immediately after, and one growing season following the pulse flow). Only cleared (3) and seeded (4) treatments had successful cottonwood-willow establishment, and establishment was affected by site conditions and flow delivery hydrology. Mixed tamarisk and native species established in cleared areas, suggesting the persistence of novel plant communities following restoration treatments and flow deliveries in Laguna Grande.

In this highly altered riparian system, components of resilience such as cross-scale interactions and connectivity greatly depend on water management decisions and restoration actions. Studies such as this one provide a quantitative assessment of restoration outcomes that can inform decision makers and feed into adaptive management of restoration sites to achieve riparian resilience.

Dedication

To my husband Joel,
for supporting me every step of the way even from 5,000 miles away.

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Chapter 1: Introduction

Historically, the Colorado River delta was a several-million acre expanse of marsh wetlands, riparian forest, mesquite bosque, estuarine lagoons, and mudflats located at the river's mouth in northwestern Mexico (MacDougal 1904). Set in one of the most arid regions in North America, the diverse habitats of the Delta supported a variety of birds, mammals (such as the jaguar), endemic marine animals, and diverse plant species, as well as native indigenous human populations (Glenn et al. 2001). Much of this diversity was a product of a highly dynamic system.

River systems in arid western North America are dynamic due to their variable and often extreme hydrologic regimes; moderate and large-scale flood events caused by spring rains and snowmelt can result in extensive fluvial-geomorphic change in river channels and surrounding floodplains (Mahoney and Rood 1998). Prior to large-scale damming and associated flow diversion projects on the Colorado River, the Colorado River Delta was an exceptionally dynamic region: an average of 16.7 billion cubic meters (13.5 million acre-feet) of water (Stahle 2003), 160,000 tons of sediment, and a high nutrient load was deposited annually in the region by the Colorado River (Glenn et al. 2001) (see Figure 1 for location of Delta within the Colorado River Basin). There was high seasonal and inter-annual variability in river discharge (Glenn et al. 1996), and additionally, river flows interacted with tidal flows from the Gulf of California, which has one of the most extreme tidal ranges in the world (up to 9 meters difference in high and low tide elevation; Nix 2013).

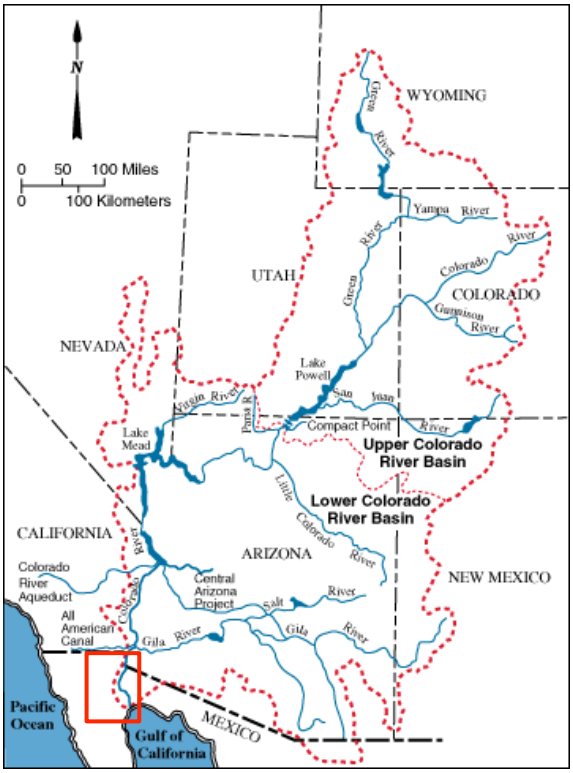

Figure 1. Map of Colorado River Basin (US Bureau of Reclamation 2015), with Colorado River Delta region in solid red box.

Small diversion projects began on the Colorado River as early as 1896, and major reductions in flows to the Delta were observed with the filling of Lake Mead in 1935 and the later filling of Lake Powell in 1963 (Figure 2; Glenn et al. 1996). The 1944 US-Mexico Water Treaty allocated 1.5 maf of the Colorado River's annual flow to Mexico, and this water is diverted at the US-Mexico border for irrigation and municipal use (Glenn et al. 1996). Very little water reached the Delta in the first 50 years following the construction of Hoover Dam, which resulted in the gradual desiccation of wetlands and subsequent loss of over 90% of original habitat (Glenn et al. 2001). Native cottonwood-willow gallery forests declined, and *Tamarix* spp., an invasive plant first observed in the Delta in 1935, rapidly spread throughout the riparian corridor (Glenn et al. 1996).

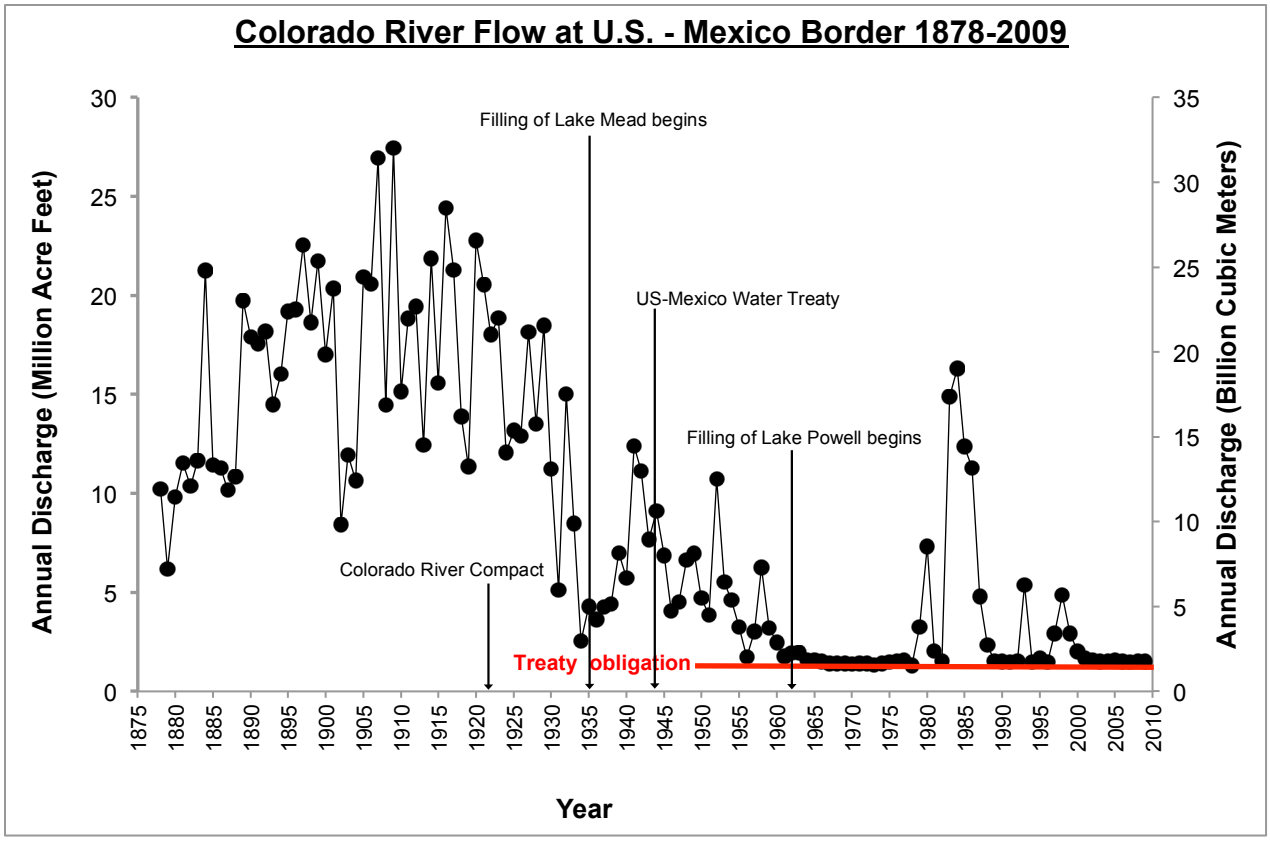

Figure 2. Colorado River annual discharge at the northerly US-Mexico border from 1878-2009. Figure adapted from original by Karl Flessa, University of Arizona.

A period of high Colorado River flows in 1980-1986 led to dam releases similar to pre-dam era flows, which significantly flooded the Delta region and regenerated marsh and native cottonwood *(Populus fremontii*) and willow (*Salix* spp.) habitat along the river corridor (Nagler et al. 2005; Zamora-Arroyo et al. 2001; Glenn et al. 1996). Lower flow volumes released to the Delta in 1993 and 1997-1999 sustained and created additional habitat (Zamora-Arroyo et al. 2001). The ability of the riparian corridor in the Delta to partially recover after half a century of very little flows suggested that the system was surprisingly resilient. Subsequently, the resiliency of the Delta was used as a rallying point to encourage restoration efforts, particularly to promote the dedication of environmental flows to the region (Glenn et al. 1996; Glenn et al. 2001; Nagler et al. 2005; Tiegs et al. 2005; Hinojosa et al. 2013). Environmental flows are loosely defined here as managed flow releases from dams that are designed to benefit the environment by

supporting ecosystem health and providing socially valued ecosystem services (Richter et al. 2006).

In November 2012, the United States and Mexico signed a binational agreement, Minute 319 (an amendment to the 1944 US-Mexico water treaty), to dedicate water for environmental flows to the Colorado River in Mexico for the first time in history (IBWC 2012). A total of 195 million cubic meters (mcm) was allocated to the Delta over the 5-year term (2012-2017) of the agreement (IBWC 2012). A coalition of nongovernmental organizations (NGOs) that make up the Colorado River Delta Water Trust (Sonoran Institute, Pronatura Noroeste, and Environmental Defense Fund) agreed to provide 65 mcm of water for low-magnitude habitat maintenance flows (hereafter referred to as base flow), while the U.S. and Mexico provided a one-time delivery of 130 mcm as a pulse flow (hereafter referred to as pulse flow) to mimic historic springtime floods (IBWC 2012).

Yet, much has changed in the Delta since flood flows inundated the region in the 80s and 90s. Minute 319 and the resulting efforts to scale up restoration along the riparian corridor bring up a number of questions regarding how the current resiliency of the system could affect potential outcomes of re-introduced flows to the system. Firstly, what are the traits and functions of a riparian ecosystem that promote resiliency? Is the Delta riparian corridor as resilient as it was 30 years ago, and thus will Minute 319 environmental flows generate a similar vegetative response as was observed in the 80s and 90s? Will the environmental flows provide a disturbance of sufficient magnitude and duration to generate the desired vegetation response, or are additional management actions required for native species establishment and persistence?

This study investigates themes of resilience and restoration in the Colorado River Delta riparian corridor. A controlled, replicated experiment was conducted to assess the vegetation

response to Minute 319 environmental flows in areas of applied restoration treatments. The experiment compares seedling establishment of native woody riparian tree species *Populus fremontii* (hereafter referred to as cottonwood) and *Salix gooddinggii* (hereafter referred to as willow), and the nonnative shrub *Tamarix* spp. (hereafter referred to as tamarisk), in areas with active management including undesirable vegetation removal and seed application, in combination with inundation from environmental flows.

Arrangement of the Thesis

Following the Chapter 1 introduction, Chapter 2 provides a brief literature review on resilience, riparian vegetation dynamics, and impacts of anthropogenic change on river systems in western North America. Chapter 2 also provides a review of historic and current conditions of the Delta riparian corridor to provide insight on Delta resilience. Chapter 3 presents the experimental design and field and statistical methods of the restoration treatment study. Chapter 4 presents results of the study, while Chapter 5 provides a discussion and interpretation of results, current resilience in the Delta, and lessons learned.

Chapter 2: Literature Review and Background

Resilience in a Riparian Context

Concepts of Resilience

The concept of resilience was first applied to ecological systems by Holling (1973) as "a measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables". Since Holling's (1973) initial application, however, there have been many other definitions and uses of the term in the ecological literature (Gunderson 2000). The assumption that systems have either a single steady state (global equilibrium) or multiple steady states (multiple equilibriums) necessarily affects how resilience is perceived and defined (Gunderson 2000). Under global equilibrium, resilience is defined as how "far" a system has moved from equilibrium following a disturbance and the time required for it to return to that state. This definition, which is based on "return time", is typically referred to as "engineering resilience", which contrasts to what is known as "ecological resilience" (Gunderson 2000; Peterson et al. 1998). Ecological resilience assumes the existence of multiple stable states, and thus resilience is defined as the amount of disturbance that can be absorbed by the system before it transitions into a different steady state (Holling 1973; Gunderson 2000). Ecological resilience is commonly used in the literature for ecosystems with identified multiple stable states such as freshwater lakes, wetlands, forests, savannas, grasslands, and coral reefs (Folke et al. 2004). In addition, ecological resilience as opposed to engineering resilience, is most often used to assess the ecosystem response to large disturbances and/or for predicting when a system will transition from one state to another (Folke et al. 2004). Ecological resilience is most relevant to this discussion on the Colorado River delta riparian

corridor, and the term “resilience” will be used synonymously with ecological resilience throughout the remainder of the paper.

Cross-scale interactions, connectivity, species response diversity, and species functional redundancy shape a system’s resilience to disturbances (Biggs et al. 2012). Response diversity refers to the range of responses of a functional group of species (species that perform the same ecosystem function) to an environmental disturbance (Elmqvist et al. 2003; Folke et al. 2004), while functional redundancy refers to the number of different species groups that perform the same ecosystem function (Walker 1992; Bruno et al. 2016). These elements of biodiversity affect a system’s adaptive capacity and ability to self-organize following a disturbance (Elmqvist et al. 2003). Response diversity and functional redundancy highlight the significance not necessarily of individual species themselves, but rather of the ecosystem functions and services that the *interactions* of species provide within a system (Peterson et al. 1998; Mori et al. 2013). To illustrate these concepts: systems that possess groups of species with overlapping functional roles will possess greater resiliency (functional redundancy); species within a functional group may become locally extinct following a perturbation, but other species within the same group may respond differently and would maintain ecosystem function (response diversity) (Elmqvist et al. 2003; Mori et al. 2013). Under this conceptual framework, an appropriate model of ecological organization is one that assesses the variety and evenness of the distribution of functional roles within the ecosystem, as opposed to earlier models that suggested a linear relationship between stability and species richness (Peterson et al. 1998).

An interesting outcome of response diversity and functional redundancy is that as species are lost over time due to human impacts, the effects may not be noticed immediately because ecosystem function is maintained by other species. However, the overall resiliency of the system

will steadily decrease with additional species loss, leading to increasing vulnerability and the potential for the system to shift into a different state (Folke et al. 2004). As the system becomes less resilient, small disturbances or the introduction of new species may cause a transition, whereas a more resilient system would require a higher magnitude of disturbance to induce a state shift (Folke et al. 2004; Gunderson 2000).

Connectivity and cross-scale interactions can impact the pattern and rate of spread of a disturbance and the ability of the system to recover after disturbance (Biggs et al. 2012). Species functioning at different spatio-temporal scales increases both response diversity and functional redundancy (Elmqvist et al. 2003). For example, regional or landscape-scale biodiversity may provide sources for species recruitment at the local scale following a local disturbance event, given adequate connectivity across scales (Folke et al. 2004). In this example, the response to disturbance of species within the same functional group is different at different spatial scales, which ultimately provides increased ability for the system to reorganize.

Resilience and Riparian Systems

For riparian systems, resilience is highly dependent on connectivity and plant species interactions, although the extent of influence of the latter depends on the specific species present (Reardon-Smith 2011). To investigate the relationship between riparian plant species diversity and resilience, it is first important to understand links between vegetation and ecosystem functions. Riparian vegetation structure and composition affect various ecosystem traits and functions including streambank stabilization, run-off control, soil properties such as porosity, salinity, and organic matter, maintenance of soil moisture, water storage, nutrient cycling, connectivity with backwater areas, and many others related to water quality and the hydrological cycle (Richardson et al. 2007; Tabacchi et al. 2000; Naiman et al. 1997). It has been suggested

that temporal and spatial riparian species diversity can increase nutrient cycling efficiency longitudinally and laterally along the river corridor (Tabacchi et al. 2000), but few studies have actually quantified the relationship between plant diversity and other riparian system functions (Richardson et al. 2007). Generally, such a relationship can vary depending on the process, ecosystem type, species present, and functional characteristics of species; however, high plant species richness has been found to decrease the success of invasive species, indicating a higher resilience of species-rich communities at least to this type of disturbance (Hooper et al. 2005).

Functional riparian systems typically have a high degree of hydrological and biological connectivity across watershed- and reach-scales simply due to the river landform that defines this ecosystem type. Resilience of riparian systems is highly dependent on hydrological connectivity: longitudinal (up- and downstream) connectivity between river reaches, lateral connectivity between the river channel and floodplain, and vertical connectivity between surface water and groundwater (Stromberg et al. 2013; Reardon-Smith 2011). Various ecosystem processes are affected by hydrological connectivity, including sediment transport, erosion, and nutrient transport and cycling, which then affect biological processes such as colonization and succession (Naiman and Décamps 1997). Riparian systems also possess biological connectivity across scales by acting as wildlife corridors and conduits for propagule dispersion (Naiman et al. 1993). Although a high degree of connectivity can enhance resilience, it can also reduce it, due to the often rapid propagation of disturbance along the corridor, particularly with respect to invasive species spread (Richardson et al. 2007; Biggs et al. 2012; Naiman et al. 1993).

As alluded to above, riparian systems (particularly in the western U.S.) are dependent on disturbance caused by dynamic hydrologic regimes (Stromberg et al. 2013). Riparian plant species such as cottonwood and willow are adapted to large disturbance events that allow for

rapid colonization and establishment following a flood (Shafroth et al. 2002). Intra- and interannual variability of flood magnitude and timing create conditions in which riparian species must possess inherent resilience due to the range of disturbances they must endure (Richardson et al. 2007; Naiman and Décamps 1997). Yet flood disturbance can also hinder overall system resilience, as rapid shifts in water levels, resource availability, and bare ground can facilitate invasion of nonnative species that eventually can lead to a shift in ecosystem state (Richardson et al. 2007). A particularly high frequency or intense disturbance can lead to loss of plant species resiliency, which can delay recovery and potentially allow for nonnative species to become established (Richardson et al. 2007).

Additionally, anthropogenic-caused alterations in the flood disturbance regime can significantly affect riparian ecosystem functioning, as will be described below. River systems are highly impacted by humans on a global scale; altered hydrological regimes, land-use change near rivers, logging, grazing, and recreation have degraded river systems worldwide, which has led to or occurred in tandem with nonnative species invasions (Poff et al. 2007). To understand how human activities affect riparian system resiliency, and particularly the Delta riparian corridor, riparian vegetation dynamics and anthropogenic change in western North America will be explored in greater detail.

Riparian Vegetation Dynamics and Anthropogenic Change

Cottonwood and Willow Establishment Mechanisms

Cottonwood (*Populus* spp.) and willow (*Salix* spp.) are dominant keystone species in western riparian systems that provide structural habitat diversity, which supports high bird species richness and diversity (Anderson et al. 1983; Rosenberg et al. 1991). However, altered hydrologic regimes and lowered groundwater tables caused by anthropogenic change have led to

extensive decline of cottonwood-willow forests in western North America over the last century (Stromberg 1993; Busch and Smith 1995; Patten 1998; Stromberg 2001). Given these species' significance in riparian systems in arid regions, including the provision of numerous ecosystem services (Richardson et al. 2007), extensive research has been conducted on the ecohydrological interactions that affect cottonwood and willow habitat and prospects for restoration.

Cottonwood and willow seedling recruitment and survival are closely linked to flood disturbance regimes and fluvial change (Rood and Mahoney 1990; Stromberg et al. 1991; Stromberg et al. 1993; Scott et al. 1996; Mahoney and Rood 1998; Shafroth et al. 1998). As pioneer species, they require bare, moist sediment for germination, which is created by vegetation scour and sediment deposition from flooding (Stromberg et al. 1991; Stromberg et al. 1993; Scott et al. 1996). Cottonwood and willow phenology is adapted to the annual hydrograph; they disperse a large number of seeds during the spring/early summer flow recession period (Stromberg et al. 1991; Stromberg 1993; Shafroth et al. 1998; Mahoney and Rood 1998). The cottonwood seed dispersal window is typically one month earlier than that of willow, and thus they tend to become established on slightly higher floodplain terraces, which have an earlier period of groundwater recession (Stromberg et al. 1993). Peak willow seed dispersal occurs in late spring/early summer when surface flows are receding on lower elevation terraces closer to the stream channel, as they require wetter conditions than cottonwood (Stromberg 1993). Both cottonwood and willow seeds have high initial viability, but it is short-lived and diminishes completely over a period of 1-4 weeks (Braatne et al. 1996).

Cottonwood and willow species are phreatophytes; they utilize water from the phreatic zone (capillary fringe) just above the groundwater table during the growing season (Mahoney and Rood 1998). Following initial germination, seedling roots follow soil moisture down the soil

profile as the water table recedes, growing to keep up with the rate of groundwater recession. Various average maximum groundwater recession rates have been reported to support cottonwood seedling survival, from 2.5 centimeters (cm)/day (Mahoney and Rood 1998) to 4.4cm/day (with range of up to 6.1cm/day; Shafroth et al. 1998). Cottonwood and willow trees are typically found in areas with mean depth to groundwater of less than 5 meters (m), and more commonly in the range of 1-3 m (Stromberg et al. 1991; Shafroth et al. 2000).

Given cottonwood and willow species' close relationship with flooding and fluvial processes, it is not surprising that they have significantly declined in western North America, where the majority of river systems are regulated. Consequences of river damming and diversions in the Southwestern U.S. include alterations in the timing and amount of river discharge, reduction of sediment deposition and nutrient transport, channel incision and narrowing, and reduction and salinization of the floodplain area (Busch and Smith 1995; Patten 1998). The loss of processes critical to cottonwood and willow recruitment such as sediment deposition and vegetation scour have all but eliminated new seedling establishment along many rivers in the Southwest (Patten 1998; Stromberg et al. 2007). Additionally, the reduction of surface flows in combination with increased groundwater pumping has led to significant declines in regional and local aquifers in the region, which cottonwood and willow are highly dependent on (Patten 1998). Particularly an issue along the lower Colorado River, the lack of leaching by overbank flows has caused a build up of salts in the soil, which is detrimental to cottonwood and willow establishment and growth as it decreases soil water potential and increases water scarcity for the plant (Busch and Smith 1995; Shafroth et al. 1995).

Following the era of large dam construction on the Colorado River, cottonwood habitat along the mainstem decreased from an estimated 2000 hectares in the 1600s to 200 hectares by

1977 (Ohmart et al. 1977). The expansion of agricultural activities also greatly impacted riparian communities, particularly in the Colorado River delta, where more than 1 million hectares of the river floodplain and surrounding area was converted to farmland (Briggs and Cornelius 1998).

Tamarisk and Western Rivers

As cottonwood and willow habitat declined, an invasive nonnative species frequently took its place, which has dramatically shaped river restoration efforts in western North America over the past 30 years. Tamarisk (*Tamarix* spp.), a plant genus native to Eurasia and Africa, was introduced to North America in the late 1800s as an ornamental and for erosion prevention along stream banks, among other purposes (DiTomaso 1998). Since the 1950s, cover of tamarisk has rapidly increased, with a current estimated cover of several hundred thousand hectares in North America (Sher 2013). It has become dominant in many riparian systems in the western United States, often replacing native species populations with a near-monoculture of tamarisk (Ohmart and Anderson 1988). The dominant species present in the western U.S. include *T. ramosissima, T. chinensis*, and their hybrids, although due to the difficulty in distinguishing these species from one another, they are commonly referred to as their genus name, *Tamarix*, or tamarisk (Sher 2013).

As a facultative halophyte and a facultative phreatophyte, tamarisk is a highly adaptable plant that can thrive under a range of conditions from high salinity and low water availability, to low salinity and high water availability (Ohrtman and Lair 2013, Bagstad et al. 2006). It has the ability to extract water from both the soil and the groundwater table (Nippert et al. 2010), and saline environments do not negatively affect its growth and reproduction (Busch and Smith 1995; Shafroth et al. 1995). Its high water- (Shafroth et al. 2000) and salinity-stress tolerance (Glenn and Nagler 2005; Shafroth et al. 1995), in addition to its reproductive strategy, have

enabled tamarisk to thrive in the post-dam riparian systems of the West. Similar to cottonwood and willow, tamarisk is a pioneer species, but it trumps the native species both in terms of quantity of seeds produced and seed dispersal period length (Merkel and Hopkins 1975). In the warm climate of the Colorado River delta, tamarisk has been observed dispersing seed throughout the growing season (February to November) (K.J. Schlatter, personal observation).

The decline of cottonwood-willow forest and expansion of tamarisk has been linked to a decline in bird diversity and abundance, particularly for riparian obligate species (Ohmart et al. 1977; Hinojosa-Huerta 2013). Tamarisk has also been purported to increase soil and groundwater salinity, use up extensive amounts of groundwater, outcompete native plant species, increase fire frequency, alter above and belowground biota, decrease overall plant and wildlife diversity, and change river channel structure (DiTomaso 1998; Glenn and Nagler 2005; Sher 2013). Although many of these claims have now been refuted, extensive tamarisk eradication efforts along western riparian systems have occurred since the mid-1900s (Stromberg et al. 2009).

There has been much debate as to whether tamarisk is a passenger of change rather than a driver: did the spread of tamarisk contribute to declining water tables, increased soil salinity, and reduction of native species populations along rivers in the West, or does tamarisk just do well in riparian systems that have already been substantially altered by humans (Stromberg et al. 2009; Richardson et al. 2007)? It is often difficult to separate other human impacts from the effects of invasive species, and the cause-effect relationship between native extinction and invasive establishment is often incorrectly inferred (Brown and Sax 2004; Stromberg et al. 2009). However, this is a critical distinction, as it can and has dramatically influenced management decisions regarding tamarisk (Glenn and Nagler 2005; Stromberg et al. 2007). Seemingly,

tamarisk could be a passenger or a driver of change depending on the environmental context; high densities of tamarisk in human-altered systems likely indicate a disrupted flow regime and poor hydrological conditions, in which native species could not persist even if tamarisk was not present (Glenn and Nagler 2005; Stromberg et al. 2007; Stromberg et al. 2009). Once established however, tamarisk has capacity to alter environmental conditions, which may make it difficult for natives to reestablish following removal, even if hydrological conditions were suitable (Glenn and Nagler 2005; Sher 2013).

Novel Ecosystems and Restoration

Changing environmental conditions and the introduction of new species are leading to the establishment of new or "novel" communities with different species compositions and interactions than those previously found (Hobbs et al. 2006, 2009). When such interactions and ecosystem functions have been highly altered to the point that the ecosystem shifts to an alternative state, novel communities can possess "unhelpful resilience", in which the system will maintain its degraded state following disturbance due to altered feedbacks (Standish et al. 2014; Suding et al. 2004). Attempts to restore historic abiotic conditions or vegetation types in novel systems are often unsuccessful due to the failure to reestablish abiotic-biotic feedbacks necessary for ecosystem functioning (Suding et al. 2004).

Many riparian systems in western North America are considered to be novel (Cooper and Andersen 2012; Richardson et al. 2007), as vegetation composition and interactions are different than that of those of the pre-dam era. As such, attempts to remove nonnative vegetation and replace it with native vegetation without restoring abiotic conditions (including the flood disturbance regime) and abiotic-biotic feedbacks can be futile (Richardson et al. 2007; Stromberg et al. 2007; Stromberg 2001). Similarly, attempts to restore flood disturbance regimes

(even at the same magnitude as pre-dam floods), without restoring additional abiotic and biotic mechanisms can and have been largely unsuccessful (Cooper and Andersen 2012).

Environmental Conditions in the Delta

As previously mentioned, recovery of riparian habitat in the Delta in response to flood flows in the 80s and 90s suggested a sustained ecosystem resilience despite a long period with little to no surface flows following dam construction on the Colorado. Habitat recovery over this period of time suggests all or some of the following regarding past Delta resilience: 1) flood flows were of sufficient magnitude to restore critical abiotic conditions and functions such as elevated groundwater levels, sediment deposition, vegetative scour, and wetting of surface soils; 2) existing biotic conditions had not yet surpassed a threshold to an altered ecosystem state, and biotic functions and abiotic-biotic interactions were able to be restored with the return of flood flows to the region. The recent dedication of environmental flows to the Delta under binational agreement Minute 319 leads to the questions of whether such resilience still exists today and if the magnitude of environmental flows is sufficient to restore abiotic conditions. A closer look at historic conditions prior to the 1980s as compared to current Delta conditions provides insight to these questions.

Conditions in the Delta Pre-1980s Floods

Although the hydrologic regime of the Colorado River significantly changed in the mid-1900s with the completion of Hoover Dam and Glen Canyon Dam, research suggests (S.G. Nelson, unpublished data) that abiotic and biotic conditions of the riparian corridor leading up to the 1980s floods still supported a functional, resilient system. Throughout much of the 1970s, there was a low base flow of approximately 6 m^3/s recorded at the Southern International Boundary (SIB) (S.G. Nelson, unpublished data; Figure 3), which is located approximately 34

kilometers from Morelos Dam and the Northern International Boundary (NIB). An analysis of satellite imagery from the 1970s suggests that groundwater conditions were shallow enough at that time that even this low base flow amount connected up to 115km of river channel below the dam with intermittent pulses of water that extended connectivity to the Gulf of California (S.G. Nelson, unpublished data). A small pulse flow in August 1977 (peak around 185 m^3/s with flow release duration of approximately one week) reconnected the entire channel, inundated areas outside of the channel, and traveled rapidly down the corridor (S.G. Nelson, unpublished data), which further provides evidence for the existence of a shallow groundwater table and sparsely vegetated channel at that time. It can be concluded that lateral, longitudinal, and vertical hydrological connectivity, which are critical components of riparian system resilience, still existed along the Delta riparian corridor (with spatio-temporal variability) in the years prior to the flood flows in the 1980s.

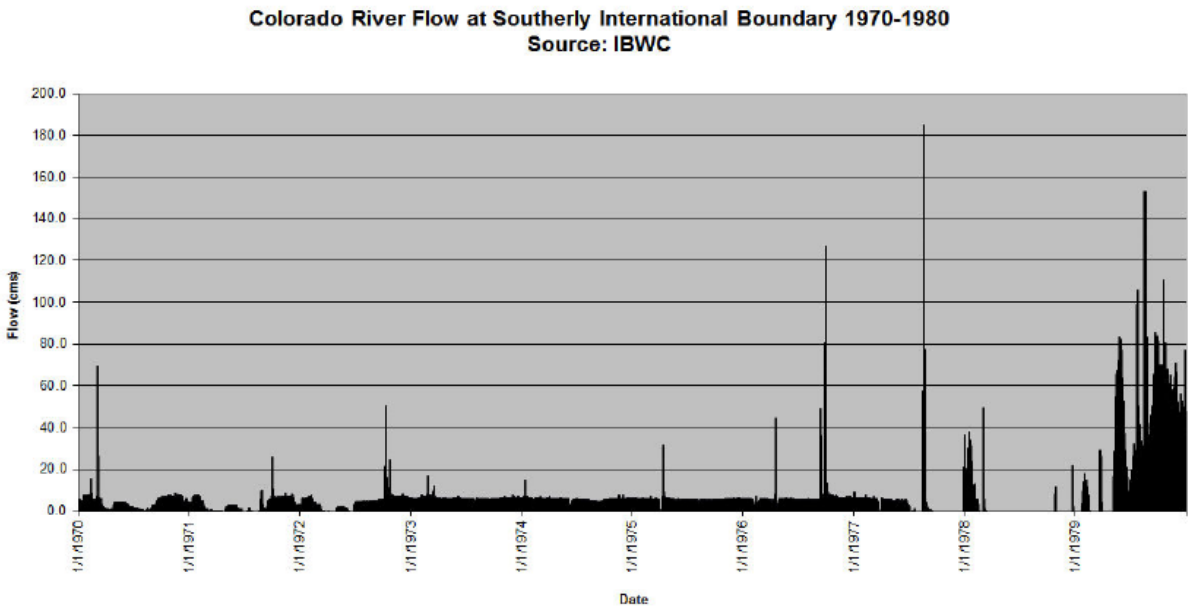

Figure 3. Discharge rate at the Southern International Boundary from 1970-1980 showing low base flow of approximately 6 m^3/s and small pulse flows leading up to 1980s floods.

Biotic conditions pre-1980s were notably degraded from pre-dam conditions: tamarisk was documented in the Delta in the 1930s (Skykes 1937) and had spread throughout much of the corridor by the 1970s, and cottonwood-willow forest declined dramatically during that time (Glenn et al. 1996). However, satellite imagery suggests that groundwater conditions still supported significant patches of native riparian habitat, which would serve as critical seed sources for the 1980s floods, and that tamarisk was not as dense along the corridor pre-1980s as it is currently (S.G. Nelson, unpublished data).

Water releases to the Delta from upstream dams began in 1979 and continued through around 1987, with the highest releases occurring in 1984 at approximately 19 billion cubic meters in that year alone (Figure 2). Researchers in the Delta during and after the 80s flood events verified that high magnitude floods scoured existing vegetation along the river channel and caused significant fluvial-geomorphic change (S.G. Nelson and E.P. Glenn, personal communication, March 2016). Thus, the flood flows in the 1980s likely successfully provided key hydrogeomorphic processes required for native riparian habitat recruitment; restored abiotic-biotic interactions led to the regeneration of native habitat along large portions of the corridor.

Current Conditions in the Delta

Following the period of relatively high flow years in the 80s and 90s, very little flow has reached the Delta since 2001 due to drought in the Colorado River Basin (Figure 4; S.G. Nelson, unpublished data; Hinojosa et al. 2013). Additionally, groundwater pumping has intensified in the region with the expansion of agriculture and increasing water scarcity; pumping in privately owned agricultural wells along the floodplain is not highly regulated in Mexico, and annual groundwater extraction can exceed its legal limit (Carrillo-Guerrero 2013). The lack of surface flows in combination with groundwater pumping in some sections of the floodplain has severely

depleted groundwater levels, particularly in the section of river near the town of San Luis, Rio Colorado. The reduction in hydrological connectivity (vertical, longitudinal, and lateral) has impacted vegetation composition, and tamarisk stands now persist along much of the corridor. Tamarisk vegetative cover along the river corridor in Mexico increased from 2002-2007, while native tree cover decreased by 50% in that same time period (Hinojosa-Huerta 2013).

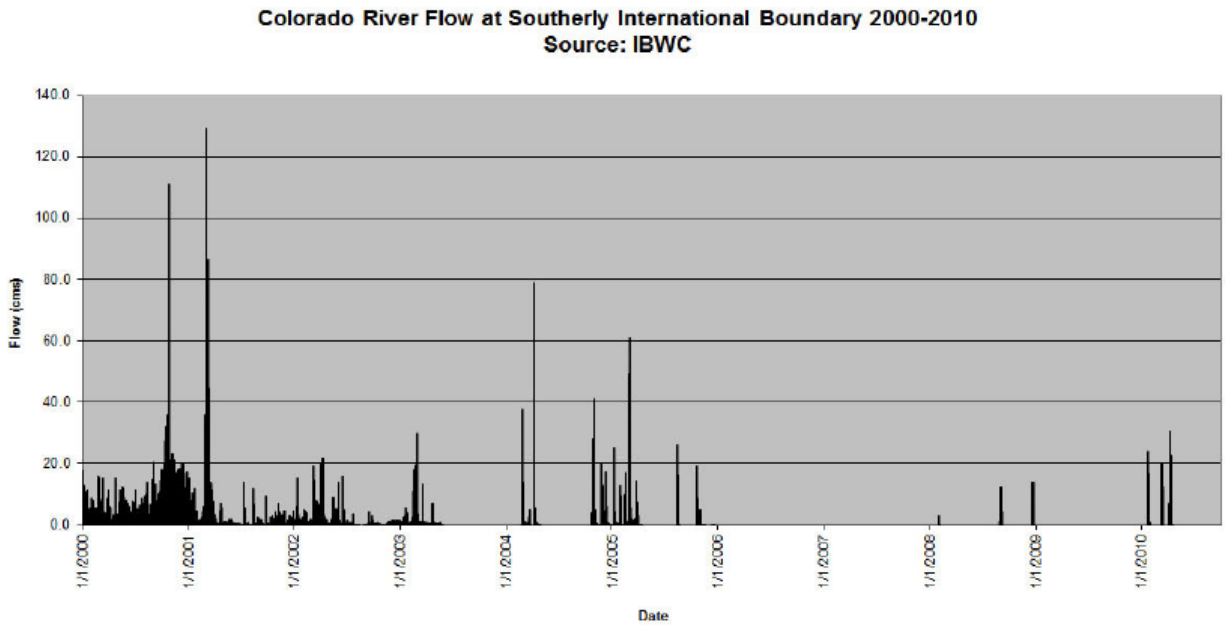

Figure 4. Discharge rate at the Southern International Boundary from 2000-2010 showing the lack of flows to the Delta since the early 2000s.

Research Questions

Given the past 15 years of drought and resulting degraded abiotic-biotic conditions, what are the expected and actual responses of vegetation following the 2014 release of environmental flows to the Colorado River in Mexico? Are abiotic-biotic interactions able to be restored with the Minute 319 environmental flow deliveries, or are additional management actions, such as creation of bare soil and augmentation of the native seed source, required to promote the regeneration of native riparian species? If native riparian species are established by

environmental flows and management actions, can they persist in the highly altered and managed riparian system?

The small-scale study implemented at the Laguna Grande Restoration Area in the Colorado River Delta seeks to answer these questions and provides insight on effective restoration strategies in human-impacted riparian systems.

Chapter 3: Experimental Design and Methods

Study Area

The study was conducted in the "central delta" region in the state of Baja California at the Laguna Grande Restoration Area (hereafter Laguna Grande) (Figure 5). Laguna Grande is approximately 82 river kilometers (km) downstream of Morelos Dam, the last dam on the Colorado River, located approximately 1km south of the U.S.-Mexico Northern International Boundary. The Laguna Grande site spans six river kilometers with a floodplain width that ranges from 1-2km. A series of historic river meanders characterize the restoration area, which were disconnected from the mainstem over time due to lack of river flows. The study was conducted along the mainstem of the river and six historic river meanders within the floodplain. In this section of the river, the dominant vegetation cover is tamarisk (*Tamarix* spp.) and arrowweed (*Pluchea sericea*), but patches of cottonwood-willow habitat established in the late 90s and early 2000s also persist (Glenn et al. 2001). A shallow groundwater table maintained by agricultural return flows from adjacent agricultural fields supports existing riparian habitat and stagnant surface water in this section of river (Glenn et al. 2001; Hinojosa et al. 2013).

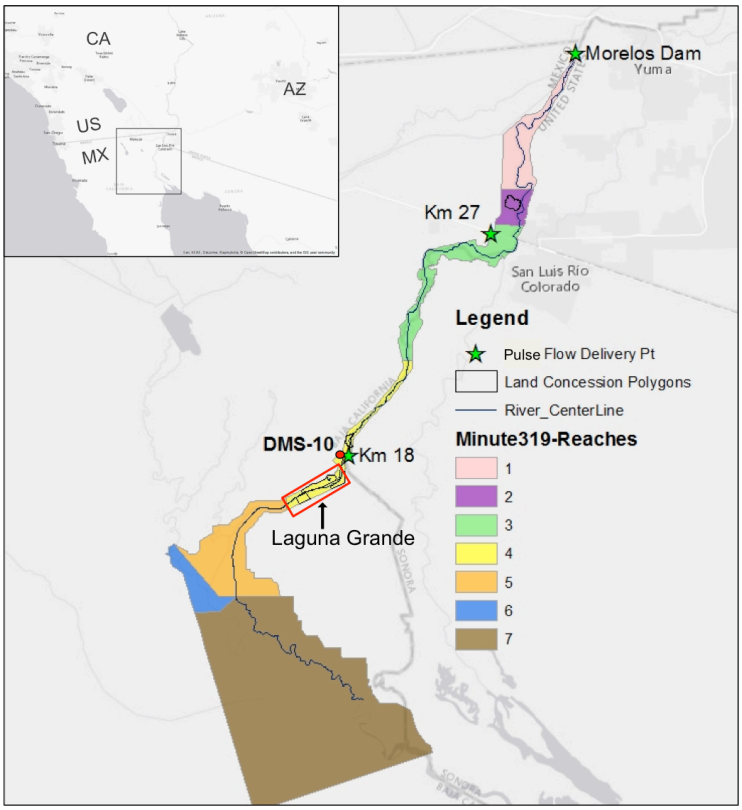

Figure 5. The Colorado River riparian corridor divided into seven reaches, showing the location of Laguna Grande in the central Delta.

Located in the Sonoran Desert, the greater Delta region is hot and arid; between 2003-2014, mean annual precipitation at the Yuma Valley Arizona Meteorological Network station (approximately 60km NE of Laguna Grande) was 58 mm (AZMET 2016). The average high temperature for June, July, and August typically exceeds 40°C, with daily temperatures often reaching over 46°C in July and August (AZMET 2016).

Sonoran Institute, a conservation non-profit organization, has been implementing restoration activities in the Laguna Grande Area since 2006, initially conducting small-scale projects to restore mesquite bosque and cottonwood-willow habitat. In 2008, the Mexican government designated three land areas (567 hectares) for the purpose of restoration, which are managed by NGOs Pronatura Noroeste and Sonoran Institute. The three land areas, named CILA, Cori, and Laguna Larga, make up the Laguna Grande Restoration Area (Figure 6).

Sonoran Institute is actively implementing restoration activities in Laguna Grande with approximately 100 hectares restored at the site to date. To promote cottonwood and willow establishment following inundation from Minute 319 environmental flow deliveries, the Sonoran Institute mechanically and manually removed tamarisk and arrowweed in 130 hectares of land predicted to be inundated with the pulse flow by a one-dimensional, steady-state hydraulic model (IBWC 2014). Additional site preparation included land grading and excavation to reconnect former river meanders with each other and the mainstem.

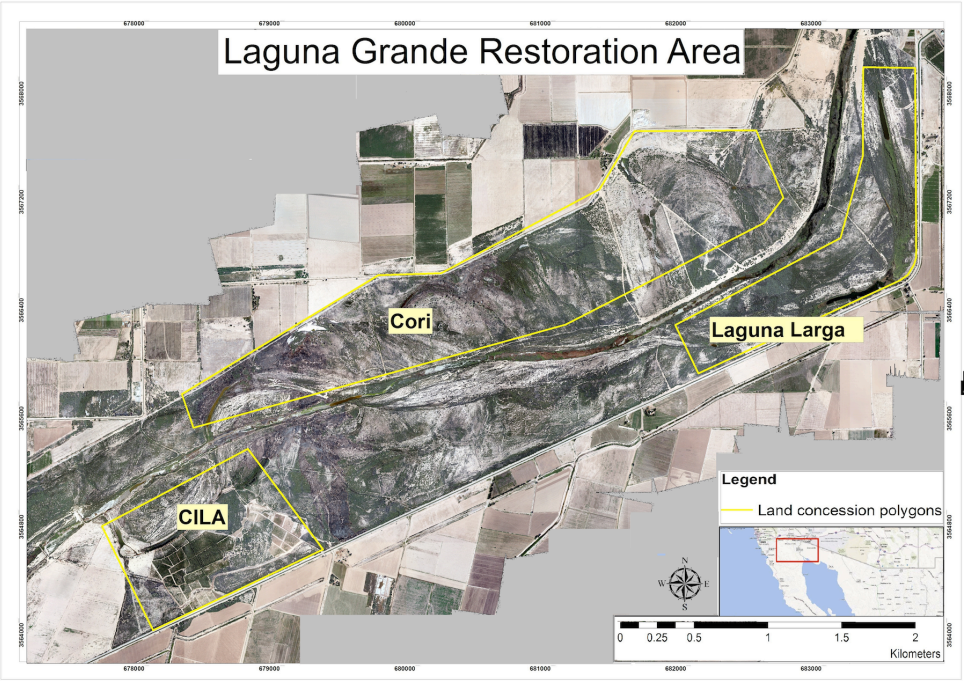

Figure 6. The Laguna Grande Restoration Area. Restoration concession polygons border the Colorado River on both sides and include several sections of historic river channel.

Environmental Flow Deliveries at Laguna Grande

Minute 319 environmental flow deliveries were part of a 5-year pilot program to monitor impacts and assess lessons learned from the implementation of environmental flows (IBWC 2012). Due to the fact that ecological conditions have changed substantially and water had not flowed in the river channel in Mexico for 15 years, the delivery of flows was largely an experiment to determine how water would behave and what the biological response would be

(Flessa et al. 2013). There were no ecological goals specifically outlined in the Minute itself, but the pulse flow hydrograph was designed to promote native riparian vegetation establishment.

A total of 195 mcm of water was dedicated to the Delta as part of Minute 319 (IBWC 2012). The Colorado River Delta Water Trust (Sonoran Institute, Pronatura Noroeste, and Environmental Defense Fund) is providing 65 mcm of water for low-magnitude riparian habitat maintenance flows (base flows) from 2012-2017, while the U.S. and Mexico provided 130 mcm as a one-time pulse flow, which was delivered from March 23-May 18 of 2014 (IBWC 2014).

The majority of pulse flow water was delivered from Morelos Dam in Reach 1 (102 mcm; this included 2 mcm of operational surplus flows beyond the Minute 319 agreement), while smaller amounts were delivered at two points further downstream using irrigation infrastructure (Figure 5 and Figure 7): 21 mcm were delivered to the mainstem via the Km 27 spillway, and 9 mcm were released from the Km 18 spillway. The goal of using delivery points further downstream was to ensure pulse flow water reached the Laguna Grande restoration site in Reach 4. Reach 3 is a dry reach, in which the groundwater table is extremely low, and it was uncertain how much water delivered from Morelos Dam would infiltrate in this reach.

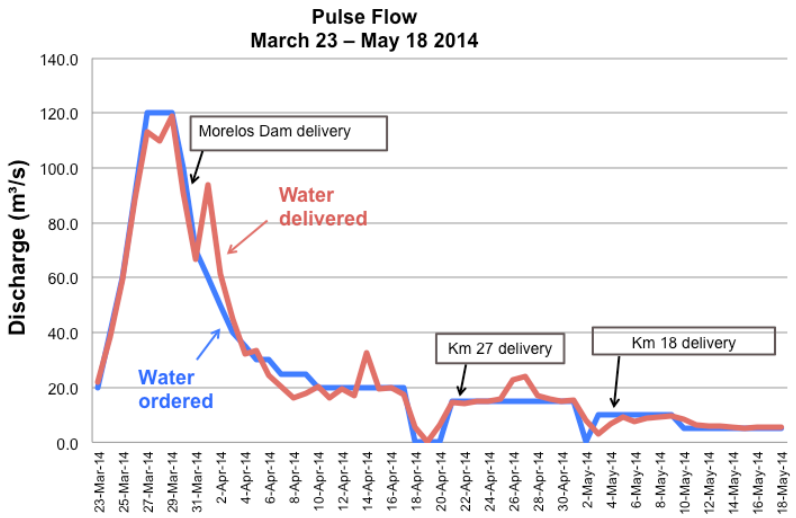

Figure 7. Discharge of the Minute 319 pulse flow showing delivery amounts and timing of releases (planned versus actually delivered). Figure adapted from IBWC 2014.

Figure 7 shows the hydrograph of the pulse flow release (both planned and actual) from the three pulse flow delivery points. As water traveled downstream, a significant amount infiltrated into the dry reaches; less than 10% of pulse flow water arrived to Reach 4 and Laguna Grande, even with the downstream water delivery points (Figure 8). Discharge at DMS-10, located just upstream of Laguna Grande (Figure 5) barely exceeded 9 m3/s (Figure 9).

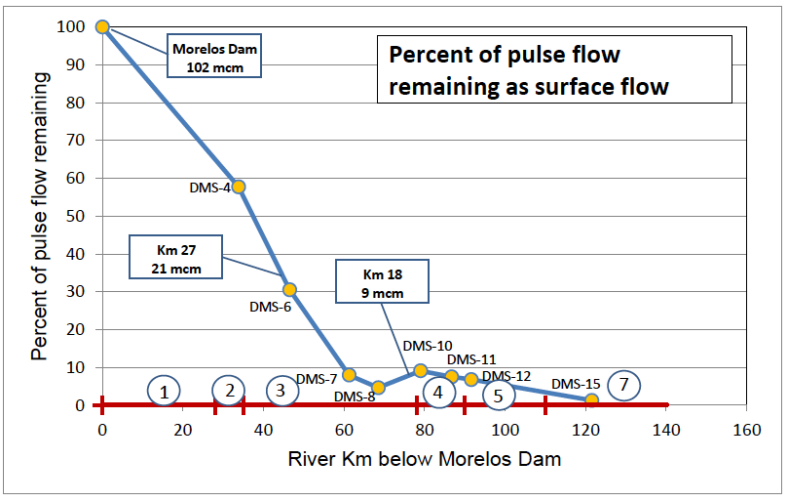

Figure 8. Percent of the pulse flow remaining as surface flow with increasing distance from Morelos Dam. Figure from IBWC 2014.

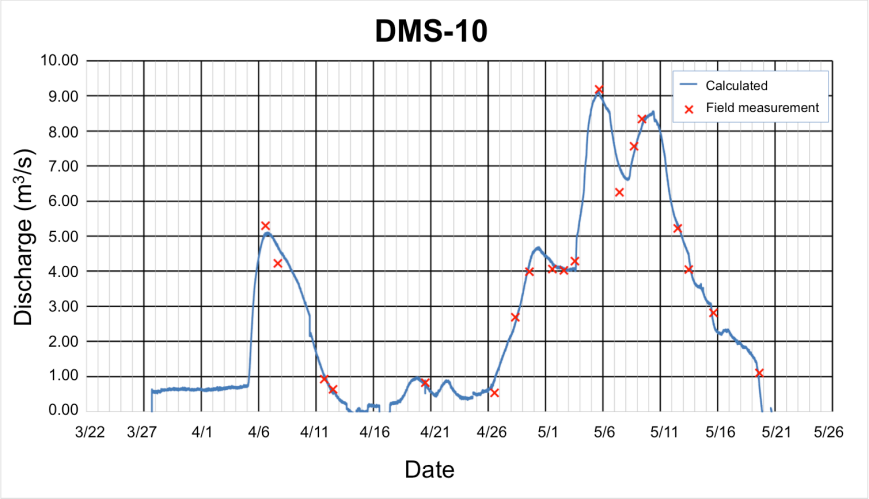

Figure 9. Discharge at DMS-10, located upstream of Laguna Grande. Figure courtesy of Jorge Ramirez and Minute 319 hydrology monitoring team, unpublished data.

Experimental Design and Methods

To determine if increasing levels of management would affect native riparian woody plant recruitment success, a controlled, replicated experiment was implemented with treatments of increasing, additive management activities. The experimental design was generally modeled after Cooper and Andersen's study (2012) to assess tamarisk removal treatments on vegetation establishment following a managed flood release on the Green River. Management treatments were applied as followed. Control plots (C=T1) received no management manipulations and were not inundated from the pulse flow. Inundated treatment plots (I=T2) were in areas inundated during the pulse flow. Removal and inundated treatment plots (R+I=T3) were in areas where tamarisk and arrowweed had been removed prior to inundation from the pulse flow. Seeded, removal, and inundated treatment plots (S+R+I=T4) were in areas where tamarisk and arrowweed had been removed prior to inundation from the pulse flow and then were seeded with cottonwood and willow following pulse flow recession.

For each of the four plot types (T1-T4), six replicate sites were selected in Laguna Grande based on local elevation (topographic position) of the area (note elevation of meanders and river channel shown by digital elevation model; Figure 10), predicted area of inundation, clearing application, and accessibility with hydroseeding equipment. The sites were named CILA1, Cori1, Cori2, Cori3, Cori4, and Cori5 based on their location within the three land areas (Figure 10). Most of the sites were over 500 meters away from one another, with the exception of Cori1 T1/T2 and Cori2 T1/T2. Cori2 T1/T2 had to be placed closer to the Cori1 sites due to accessibility issues. Cori5 was removed from the study because it did not get inundated as predicted. A randomized block design of treatments was not possible due to preexisting locations of cleared areas and consideration of areas predicted to be inundated by the pulse flow.

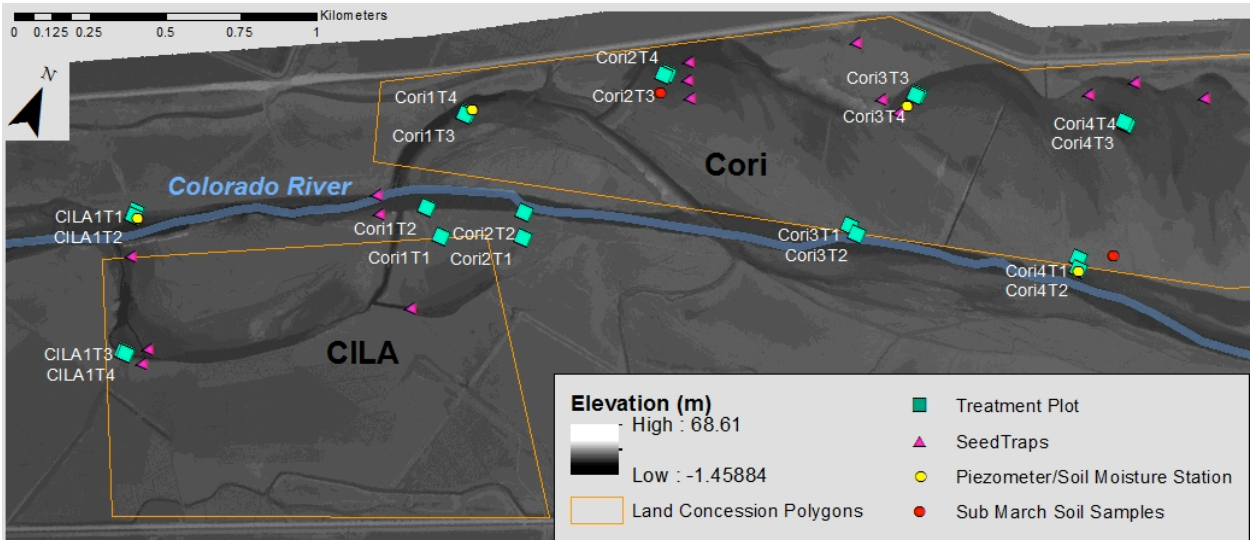

Figure 10. Location of treatment plots, seed traps, and piezometer/soil moisture monitoring stations within the Laguna Grande Restoration Area.

T1 and T2 plots were located in areas along the river where no grading or clearing occurred (Figure 11 and Figure 12). T1 plots were located in high terraces not subject to inundation; these areas were primarily dominated by mature, less dense stands of tamarisk and arrowweed (Figure 13). T2 plots were located along the riverbank of the Colorado River mainstem and had a dense mixture of native mature woody species including cottonwood, willow, and Baccharis spp. as well as nonnative species including tamarisk and Arundo donax (arundo) (Figure 13). T3 and T4 plots were located along the banks of former river meanders that had been targeted for mechanical clearing and meander channel grading. These areas were previously dominated by dense mature stands of tamarisk and arrowweed, which were removed prior to the pulse flow releases. Existing stands of cottonwood, willow and Baccharis spp. were left undisturbed in the cleared sites, and were usually within 100-200 meters of T3/T4 plot locations depending on the site. T3 and T4 plots were placed adjacent to each other and parallel to the meander channel (to provide minimal elevation difference) (Figure 11). Cottonwood and willow seed was applied to T4 plots using a 50-gallon hydroseeder in late April 2014 with the

exception of CILA1 site, which remained inundated until mid May; seed was applied to CILA1 T4 on May 15, 2014. The hydroseeding technique had been previously utilized as a revegetation method for cottonwood and willow in Laguna Grande using methods defined in Grabau et al. (2011). Cottonwood and willow seed viability was tested prior to hydroseeding by conducting germination tests on seed collected from source cottonwood and willow trees.

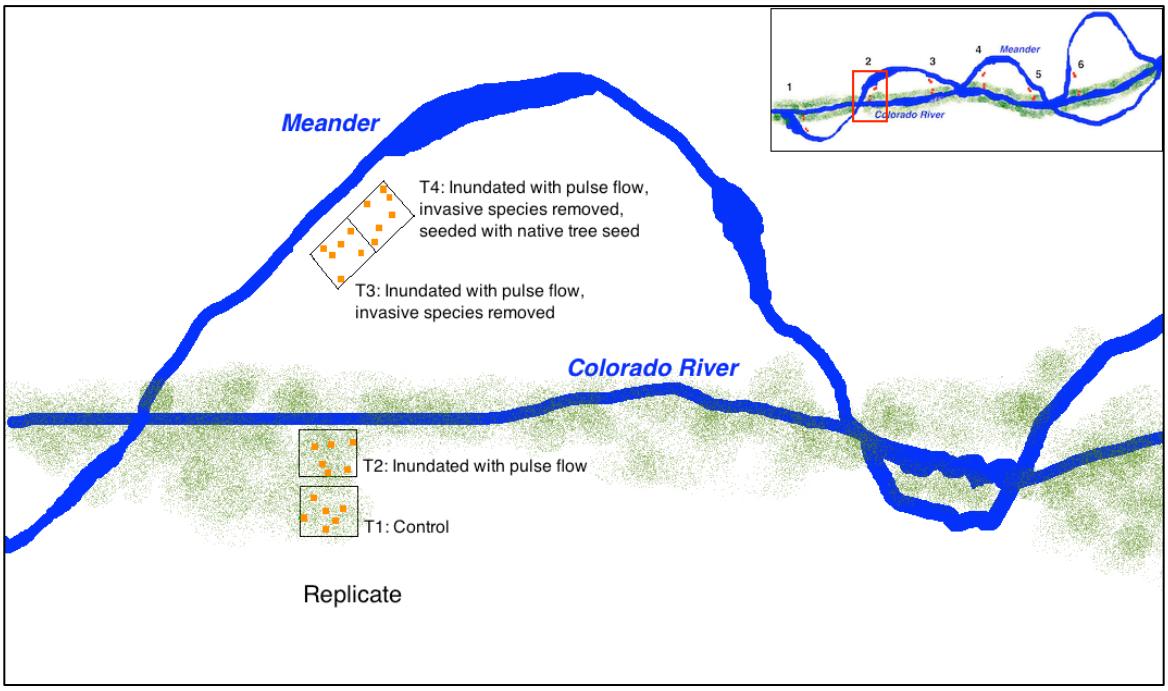

Figure 11. Experimental design diagram with replicates (top right), treatment plots, and quadrats (orange squares).

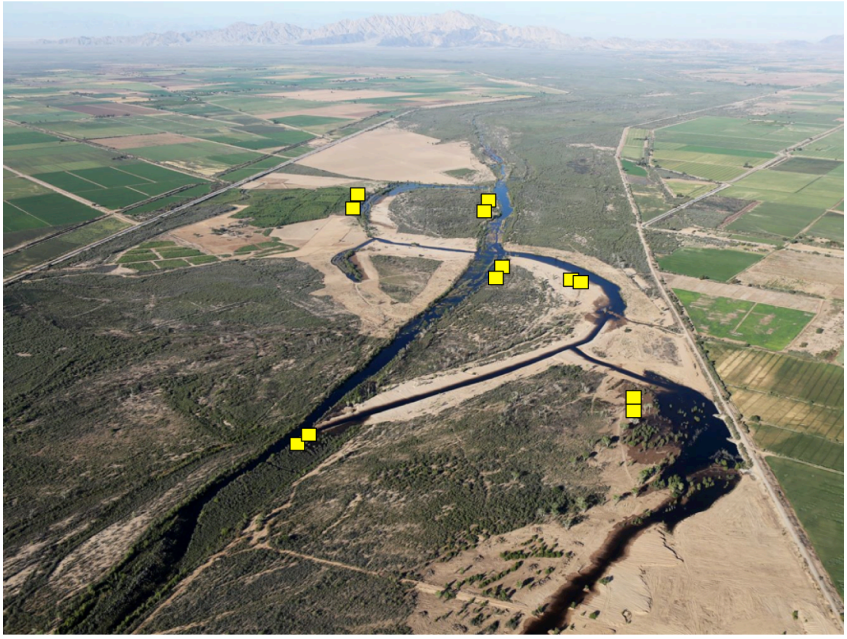

Figure 12. Aerial image of Laguna Grande following tamarisk and arrowweed removal and during inundation from the pulse flow release. Yellow squares show approximate locations of some treatment plots.

Figure 13. Representative photographs of treatment plots showing vegetative conditions.

It was hypothesized that native woody species recruitment would increase with increasing levels of management activities, such that T4>T3>T2>T1. The T4 seeded treatment was predicted to be more successful in areas that were seed limited, but would be equal to T3 in areas that were not seed limited. Tamarisk recruitment was predicted to be higher in the cleared areas than the uncleared areas, such that T4=T3>T2>T1. Tamarisk seed was not applied in T4 plots so T4 tamarisk establishment was expected to equal that of T3.

Each treatment plot was 10x10 meters. Within each treatment plot, six points were randomly selected from a grid, each of which formed the northeast corner of a $1m^2$ quadrat; a total of six quadrats were established in each treatment plot. Within each quadrat the following components were measured: count and cover class by plant species; canopy closure at quadrat corners using a convex densiometer; and groundcover class (groundcover categories: bare, herbaceous, woody debris, leaf litter, water). Cover class by plant species was categorized using the following system: 0 plants present; 1 individual present with less than 1% cover; several individuals present with less than 1% cover; < 5% cover; 5-25% cover; 25-50% cover; 50-75% cover; >75% cover. A similar categorization was used for groundcover classes.

Vegetation surveys were conducted prior to the pulse flow in mid March 2014, again in mid May following the pulse flow recession, and at the end of the growing season in October 2014. If initial seedling density was low, the quadrat was expanded from $1m^2$ to $4m^2$.

A wooden stake was permanently established in the northeast corner of each quadrat and marked with a horizontal line at the soil surface to measure sediment deposition following the pulse flow. Soil samples were taken from T1, T2 and T3/T4 plots (because T3 and T4 plots were adjacent to each other, only one soil sample was taken to represent both plots; n=15) in March and October 2014 at variable depths dependent on soil texture change. Soil textures (percent sand, silt, and clay) were initially recorded from the soil surface to the capillary fringe (zone of soil saturation immediately above groundwater) using visual manual methods (ASTM 2009). Samples were analyzed by Quimical S.A. de C.V. laboratory (Mexicali, Baja California, Mexico) for soil texture (Bouyoucos hydrometer method; Bouyoucos 1962) and electrical conductivity (EC) (1:1 soil:water extract), which is a proxy measurement for soil salinity. A composite of samples from depths of 0-20cm and 20-60cm were used in analysis. Based on the observation of

low magnitude flow at Laguna Grande and geomorphic analysis results (IBWC 2014), significant changes in soil texture between March and October were not likely caused by geomorphic change, but instead were likely caused by variability in field conditions, or lab error. Therefore, samples that had a difference in sand/clay composition of >25% between March and October (n=1) were not used in statistical analyses due to uncertainty in results. Two March sample results (Cori2 T3/T4 and Cori4 T1) were lost by the laboratory. Due to the necessity of having a relatively complete data set for the functioning of the statistical model, the missing sample results were substituted with results from soil samples collected less than 50 meters from the original sampling location and taken at the same time (March 2014; see Figure 10). The substitute samples were within the same range of salinity values as the original locations according to results from a large-scale salinity mapping project (Grabau et al. 2016). The substitute samples had the same texture as the October samples from the original locations.

Soil moisture monitoring stations were coupled with piezometer locations in four sites: CILA1 T2, Cori4 T2, Cori1 T3/T4, and Cori3 T3/T4. Installation of soil moisture stations and piezometers at each treatment plot was not possible due to funding and time limitations. Soil moisture was monitored using Decagon Devices, Inc. (Pullman, Washington, USA) EM50 data loggers wired to EC5 Decagon Devices, Inc. (Pullman, Washington, USA) soil moisture sensors, which measured soil volumetric water content (m^3/m^3). The soil profile was classified from the soil surface to the capillary fringe using visual manual methods presented previously, and soil sensors were installed at representative depths in the soil profile. Data loggers were programmed to record on 2-hour intervals. Three to five sensors were placed at each monitoring station depending on the variability of the soil profile and the depth to the capillary fringe.

Four piezometers were manually installed adjacent to soil moisture monitoring stations. Piezometers were instrumented with an In-Situ, Inc. (Fort Collins, Colorado, USA) Rugged TROLL 100 water level/pressure/temperature data logger, which were programmed to record at 2-hour intervals. An In-Situ, Inc. (Fort Collins, Colorado, USA) Rugged BaroTROLL was placed in a central site in Laguna Grande, which recorded barometric pressure and temperature at 2-hour intervals. The continuous groundwater data were used to determine the window of time in which groundwater recession occurred following the pulse flow release, which then was used to inform seed availability rankings (see below). The recession rate was calculated using a 7-day running average from the start to the end of the groundwater recession period.

The maximum depth-to-groundwater over the growing season was required for the seedling recruitment model at each treatment plot location. However, continuous groundwater and coupled soil moisture data were not available for all plots, so an interpolated groundwater surface was used to determine depth-to-groundwater at the plot scale. Twenty-five additional piezometers located throughout Laguna Grande were manually sounded every week in 2014. Using these data, groundwater depth across the Laguna Grande Restoration Area was determined by overlaying interpolated groundwater elevation results with a Digital Elevation Model (DEM) created from Light Detection and Ranging (LiDAR) remote-sensing data (collected in March 2014; average point densities of 10.7 points/m^2 first-return and 5.8 points/m^2 ground classified) by the U.S. Bureau of Reclamation through the Minute 319 monitoring program. Minimum, maximum, and average depth to water at the center of each treatment plot was determined from the interpolated data for the 2014 growing season (March 1 – November 1). The LiDAR-based DEM was also used to determine the elevation at the center point of each treatment plot.

Seed rain density was monitored every 2-3 weeks from February 19 through June 19 2014 across Laguna Grande (Figure 10) using seed traps made from a 60cm2 plywood board coated with Tanglefoot® and mounted on a 1-meter high post (Cooper et al. 1999). Cottonwood, willow, and tamarisk seed phenology observations were recorded every 2-3 weeks for 3 tagged adult female individuals near each seed trap location to note phases of seed development and peak dispersal periods. Seed rain density and phenology observation data were restricted to the period of groundwater recession in Laguna Grande (May 10 – June 12), as this was considered the ecologically relevant time period. Data were used from four seed trap/phenology monitoring events from 5/7/14 to 6/19/14 to determine seed availability rankings of low, medium, or high for each treatment plot. Some sites (Cori1 T3/T4, Cori2 T1/T2 and Cori3 T1/T2) did not have traps or phenology observations due to accessibility issues, theft, or destruction of seed traps. For these sites, knowledge of the density of mature female individuals and informal observations of seed dispersal were used to inform the ranking. For sites with seed counts from traps, thresholds were established to determine low-medium-high relative rankings for each species. For the native species grouping (cottonwood and willow), less than an average of 5 seeds/60cm^2 per monitoring event received a low ranking; 6-15 seeds was medium; and >15 was high. For tamarisk, 1-50 seeds/60cm^2 per monitoring event was low; 51-100 was medium, and >100 was high.

Statistical Methods

To determine the effects of different environmental variables (random effects) and treatments (fixed effects) on cottonwood-willow and tamarisk germination in May, generalized linear mixed models (GLMM) were used. For the cottonwood-willow species establishment model, the response variable was the combined mean cottonwood and willow May seedling

count per meter squared for each treatment plot (average of quadrats in treatment plot: n=6; total number of treatment plots: n = 20). For the tamarisk species establishment model, the response variable was mean May tamarisk seedling count per meter squared (average of quadrats in treatment plot: n = 6; number of treatment plots: n = 20). Fixed effects in the models included treatment and covariates of treatment: canopy cover and relative bare ground cover. An interaction between treatment and seed availability was included in the cottonwood-willow model to account for seed application in T4, but the model became over-fitted and could not incorporate the additional model complexity given the small sample size. Random effects for both models included seed availability (ordinal data used in model), soil EC, soil texture, and treatment plot (to account for variation at the treatment plot level). Soil EC was log transformed to meet model assumptions of normality of random effects (Thiele and Markussen 2012).

Mean cottonwood-willow seedling count data had a large proportion of zeros (70%) and could not be transformed to normality. Using frequency (presence in x quadrats out of 6 per treatment plot) as the response variable did not assuage the issue. The best-fit distribution for the data was a Poisson distribution, but the data were overdispersed (variance increases at faster rate than the mean); in a Poisson distribution, variance should equal the mean. A penalized quasi-likelihood (PQL) estimation with Poisson distribution and log link function was used to account for overdispersion (Venables and Ripley 2002). Parametric inferential methods including model based standard errors, likelihood ratio tests, and Akaike information criterion are not as reliable in GLMMs, as they only approximate likelihood-based model fitting (Venables and Dichmont 2004; Venables and Ripley 2002). Furthermore, the PQL method computes quasi-likelihood rather than a true likelihood (Bolker et al. 2009). Parameter estimates are thus not true estimates, but are instead predictors that function more like residuals (Venables and Dichmont 2004).

Parameter estimates are provided in the presentation of results, but no interpretation is based on the estimates. For GLMM model selection, it is recommended that the full model be used, keeping biologically meaningful variables in the model, even if they are not significant (Thiele and Markussen 2012). Importance of random effects was based on model validation plots and model fit assessments. Pairwise contrasts between treatments were conducted using Tukey HSD tests. Significance of fixed effects was determined from p-values, although interpretation of significance must include a caveat due to quasi-likelihood estimations. The cottonwood-willow model was fitted using the 'MASS' package in R (Venables and Ripley 2002).

Mean tamarisk seedling count data had a smaller proportion of zeros than that of cottonwood-willow, but still could not be transformed to normality. A LaPlace approximation with a negative-binomial distribution and log link function was used for the tamarisk GLMM to account for overdispersion (Ver Hoef and Boveng 2007). The LaPlace approximation method is preferred to the PQL method in terms of reliability of parameter estimates and its ability to determine true likelihood (Thiele and Markussen 2012; Bolker et al. 2009). Relative importance of random variables was determined by assessing the proportion of variance for each variable; if the proportion was close to zero, the random variable was removed from the model (Starkweather 2010). Pairwise contrasts between treatments were conducted using Tukey HSD tests. Methods defined by Nakagawa and Schielzeth (2013) were used to calculate R^2 values to assess variance explained by the model. Marginal R^2 is the variance described by fixed factors alone, and conditional R^2 is the variance described by both fixed and random factors (Nakagawa and Schielzeth 2013). The tamarisk model was fitted using the 'lme4' package in R (v 3.2.1) (Bates et al. 2015).

Due to the inability to distinguish between seedlings germinated after the May vegetation survey and seedlings germinated before the May survey, survival rate of seedlings from May to October could not be calculated. Difference in seedling density between May and October 2014 was determined by taking the October density divided by the May density and multiplying by 100. To assess importance of environmental factors on the presence of cottonwood-willow seedlings from May to October, comparisons between soil EC, soil texture, maximum depth to groundwater, bare ground, and canopy cover were made between plots where seedlings were present in both May and October versus plots where seedlings had died from May to October. For canopy cover, bare ground, soil EC, and maximum depth to groundwater, paired, one-tailed *t*-tests were used. A one-way ANOVA was used to test for differences in soil texture. Soil EC data were log-transformed and canopy cover was square-root transformed to achieve normality. Summary data are presented for environmental variables in sustained tamarisk present versus tamarisk absent October plots, but t-tests were not conducted due to the small number of tamarisk-absent sites (n=2).

Changes in soil EC, canopy cover, and bare ground from March to October for all treatment plots was determined by using paired, one-tailed *t*-tests. Bare ground data was log transformed, soil EC data was log transformed, and canopy cover data was square-root transformed to meet normality assumptions prior to analysis.

All models were fitted and analyses conducted within the R (v3.2.1) language and environment (R Core Team 2013).

Chapter 4: Results

Sediment Deposition

No sediment deposition was detected at any of the treatment plots following environmental flow deliveries; the sediment level was the same pre-inundation following post-inundation at T1-T4 sites.

Seed Dispersal

Cottonwood seed dispersal began in late February, and cottonwood seed was present on seed traps at four sites on March 5 (the first monitoring date following seed trap set up) (Figure 14). Willow seed was present on one trap on March 5, and few individuals were observed to be seeding at this time. Cottonwood seed dispersal peaked around April 14 and ended at the beginning of May. Willow seed dispersal peaked in early June at most sites and was still ongoing at the end of the monitoring period on June 19. Tamarisk dispersed seed in high abundances from the start of the monitoring period through the end of the monitoring period, typically an order of magnitude greater than that of cottonwood and willow.

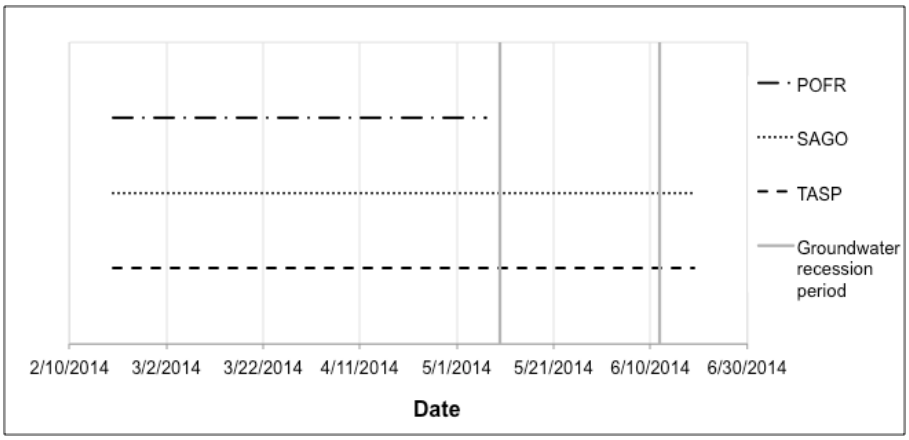

Figure 14. Seed disperal period for cottonwood (POFR= *Populus fremontii*), willow (SAGO = *Salix gooddingii*) and tamarisk (TASP = *Tamarix* species).

Generally, combined cottonwood and willow seed availability ranked low for T1 and T2 sites (Table 1), which were located along the river in areas where native tree species were present in low abundance (Figure 15). T3 and T4 sites had a range of seed availability ranking from low to high, as they were located in areas of varying native tree abundance. Tamarisk seed availability as compared to native seed availability was high across the majority of sites. In comparing relative tamarisk seed availability across sites, T1 and T2 plots had highest seed availability, due to presence of dense tamarisk cover, while T3 and T4 plots had medium to low availability due to the removal treatment application.

Site	Treat-ment	Eleva-tion	Min DtW	Max DtW	Avg DtW	POFR + SAGO Seed Availability	TASP Seed Availability	Canopy Cover		Relative Bare Ground Cover		Soil EC		Soil Texture
								Mar	Oct	Mar	Oct	Mar	Oct	
CILA1	T1	11.91	2.15	3.39	2.78	low	medium	72.50	90.85	7.55	9.69	9.67	10.45	sandy loam
	T2	10.45	0.73	1.97	1.36	low	medium	66.65	80.61	5.18	14.48	8.54	4.05	loamy sand
	T3	10.77	0.91	2.22	1.58	high	medium	0.00	12.50	81.17	51.63	2.43	3.18	sand
	T4	10.73	0.91	2.22	1.58	high	medium	0.00	6.19	74.74	70.37	2.43	3.18	sand
Cori1	T1	12.76	2.47	3.70	3.11	low	high	39.40	54.60	26.46	22.38	18.55	12.00	sandy loam
	T2	10.40	0.39	1.65	1.05	low	high	51.00	79.03	7.78	11.24	4.94	2.49	sand
	T3	11.30	1.00	2.20	1.61	low	low	0.00	0.00	98.87	86.43	5.58	1.94	loamy sand
	T4	11.32	1.00	2.20	1.61	low	low	0.00	0.00	98.87	79.03	5.58	1.94	loamy sand
Cori2	T1	12.56	2.35	3.53	2.97	low	high	0.34	1.01	88.83	83.33	20.24	11.42	clay
	T2	11.25	1.26	2.45	1.89	low	high	87.93	93.67	0.85	5.58	7.56	4.31	loamy sand
	T3	11.37	0.62	1.69	1.15	medium	medium	0.45	0.76	82.13	43.85	35.62	6.29	clay
	T4	11.27	0.62	1.69	1.15	medium	medium	24.88	8.15	77.40	74.64	35.62	6.29	clay
Cori3	T1	13.33	2.92	3.93	3.44	low	high	14.97	13.06	70.89	52.86	-	-	-
	T2	12.43	2.07	3.07	2.59	low	high	40.08	74.64	12.83	18.05	4.35	6.11	sandy loam
	T3	11.23	0.96	1.99	1.47	high	high	18.58	32.09	83.68	17.53	6.13	1.43	silt loam
	T4	11.43	0.96	1.99	1.47	high	high	8.11	8.67	66.94	3.59	6.13	1.43	silt loam
Cori4	T1	13.63	3.59	4.46	4.01	low	high	18.01	28.15	30.82	25.52	28.16	7.41	sandy loam
	T2	11.58	1.37	2.25	1.51	low	high	53.93	82.41	20.11	16.94	23.76	4.74	sandy clay loam
	T3	10.42	0.19	1.18	0.68	low	high	18.13	6.74	14.44	79.67	1.20	1.11	loamy sand
	T4	10.46	0.19	1.18	0.68	low	high	16.44	2.19	26.27	70.61	1.20	1.11	loamy sand

Table 1. Environmental variables summary table. POFR = *Populus fremontii*, SAGO = *Salix gooddingii*, TASP = *Tamarix* spp. (Dtw = Depth to water; EC = Electrical Conductivity)

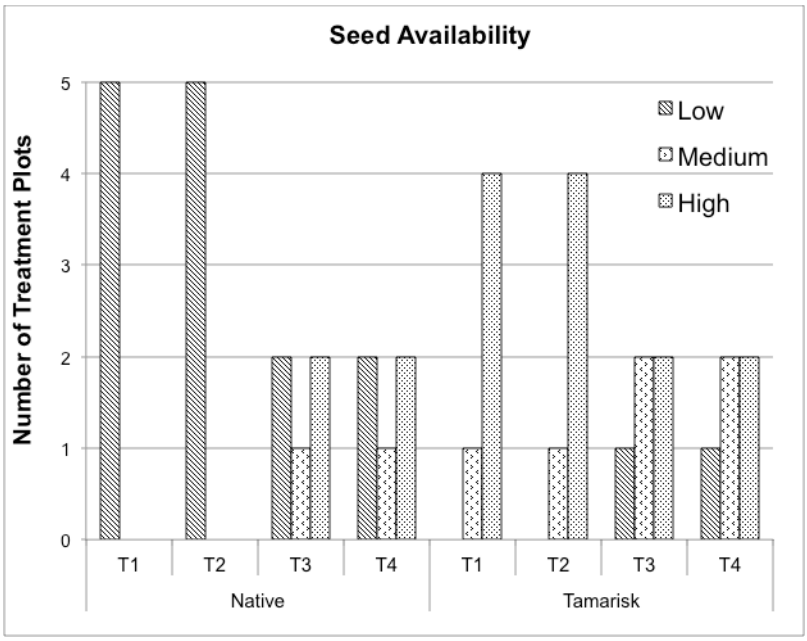

Figure 15. Number of treatment plots with low, medium, and high cottonwood-willow (native) and tamarisk seed availability. N=5 for each treatment.

Canopy Cover and Ground Cover

Canopy cover was generally greatest in T2 sites and lowest in T3/T4 sites. It was variable for T1 sites depending on the size and structure of the tamarisk and arrowweed stands within the control plot. Canopy cover increased in most plots from March to October, likely due to canopy growth, although the difference was not significant (Figure 16; p=0.103). Canopy cover increased the most from March to October in T2 plots, which may have been due to inundation and increased groundwater levels caused by the environmental flow deliveries.

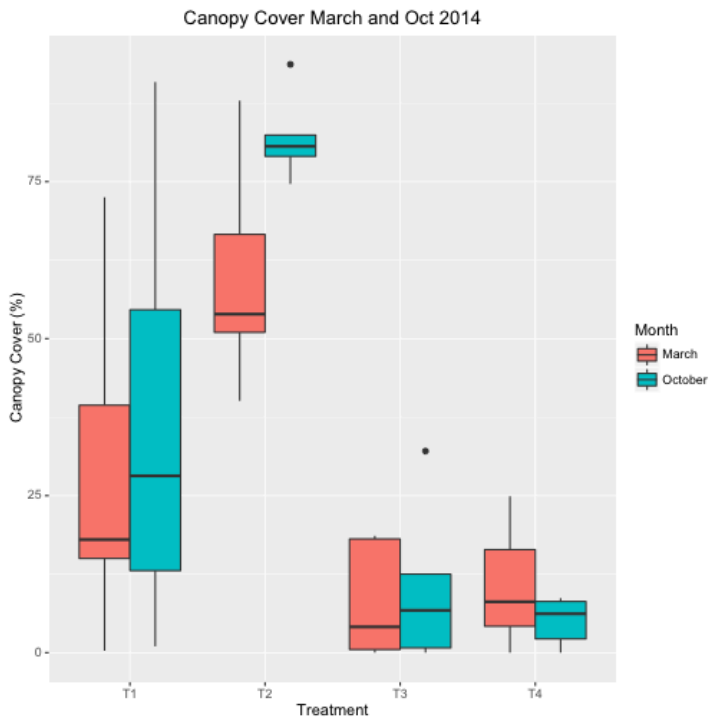

Figure 16. Canopy cover of treatment plots in March and October 2014.

Following a similar trend as canopy cover, groundcover categories of leaf litter, woody debris, and herbaceous were generally highest for T2 sites in March and October due to the presence of dense vegetation and leaf litter build up along the river (Figure 17 and Figure 18). T1 sites generally had variable leaf litter, woody debris, and bare ground cover in March and October. T3 and T4 sites had relative bare ground cover that ranged from 67% to 99% of quadrat cover in March, with the exception of the Cori4 T3/T4 site, which had >60% herbaceous groundcover in March (Figure 17 and Figure 19). This site had been similarly cleared as other T3/T4 sites, but a dense groundcover had already started to establish at the time of March monitoring. From March to October, herbaceous cover increased in T2, T3, and T4 sites likely due to wetted soils from the flow deliveries (Figure 17 and Figure 18). Bare ground cover decreased, but not significantly (p=0.470) (Figure 19). Interestingly, the Cori4 T3/T4 site was secondarily inundated in late August by base flows, which caused herbaceous groundcover and

woody seedling mortality; much of the herbaceous cover that was present in March became either bare ground or dead leaf and woody debris in October (Figure 18).

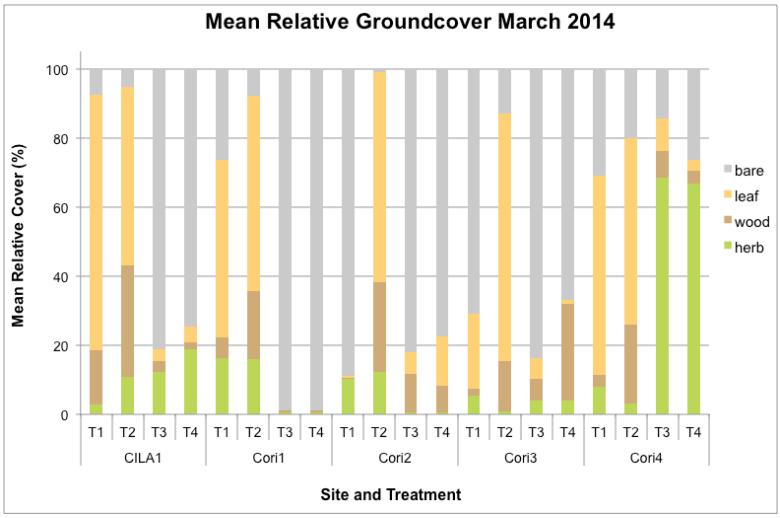

Figure 17. Mean relative groundcover of treatment plots in March 2014

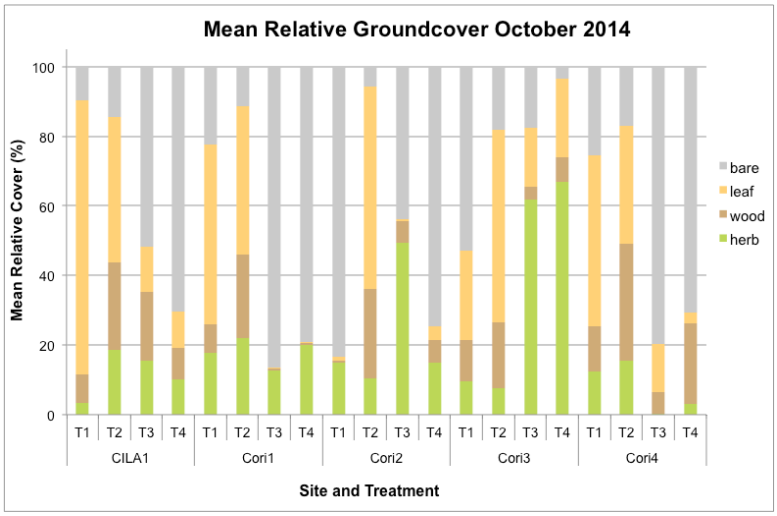

Figure 18. Mean relative groundcover of treatment plots in October 2014.

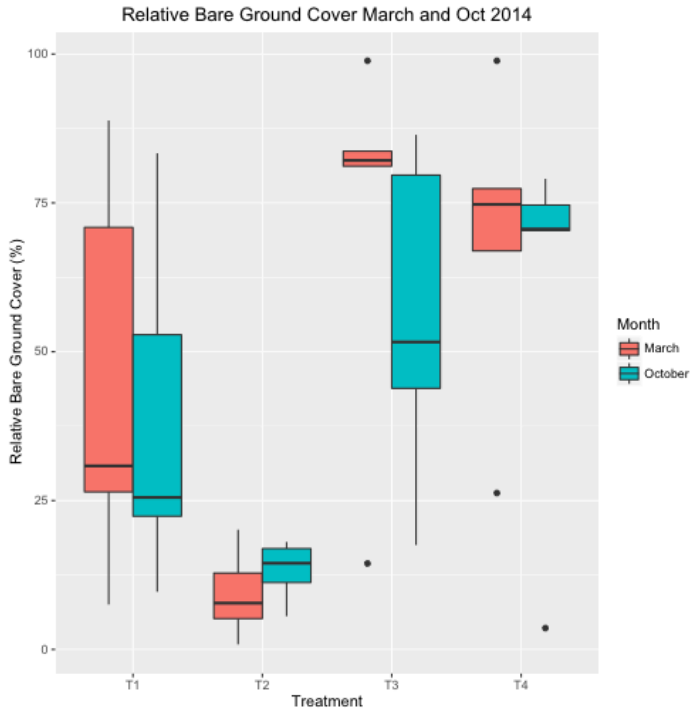

Figure 19. Relative bare ground cover of treatment plots from March to October 2014. The Cori4 T3 and T4 plots are outlier points with low bare ground cover in March.

Soil EC and Texture

Soil EC was highest in T1 plots in March (mean = 19.16 ± SE 3.79 dS/m) and October (mean = 10.32 ± SE 2.04 dS/m), and lowest in T3/T4 plots in March (mean = 3.84 ± SE 1.20 dS/m, excluding Cori2 T3/T4), and October (mean = 2.79 ± SE 0.94 dS/m) (Figure 20; Table 1). Soil EC significantly decreased from March to October ($p = 0.0002$), likely as a result of the flushing of salts from soils from flow deliveries. Soil texture was variable in treatment plots (Figure 21). Soil salinity increased with increasing clay composition (Figure 22), a relationship that is likely due to the high conductivity and cation exchange capacity of clays (Grisso et al. 2009). The Cori2 T3/T4 plots had very high soil salinity in March, which corresponded with a clay soil texture classification. The Cori 2 T3/T4 plot was located in a low elevation area of clayey soils, which infrequently received agricultural return flows from a nearby canal spillway. A disconnected lagoon with no outflow had high evaporation following agricultural flow release

events, which likely led to the build up of salts over time. A large area surrounding the lagoon had visible salts on the soil crust in March, including the Cori2 T3/T4 plots.

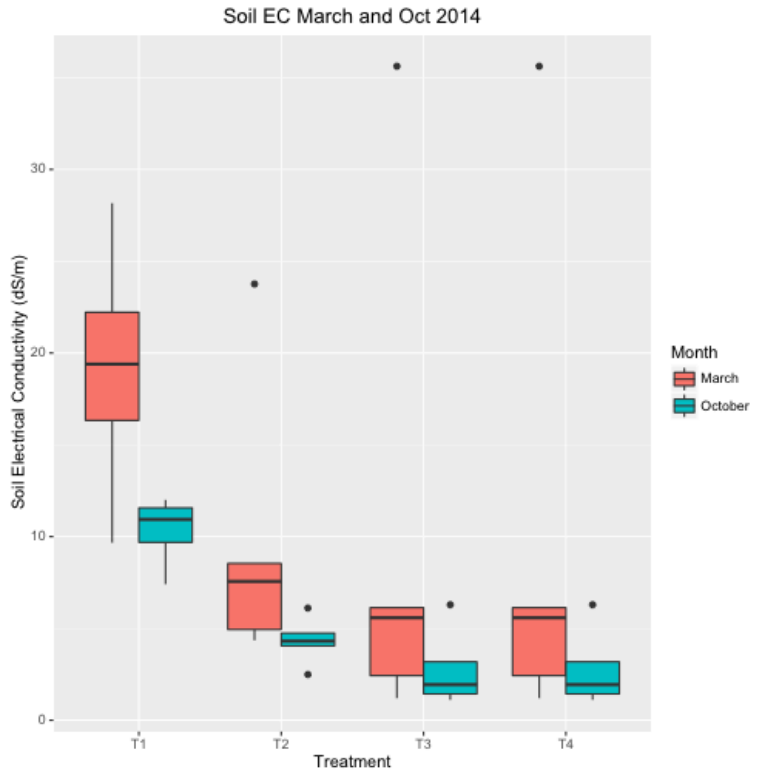

Figure 20. Soil EC in treatment plots from March to October 2014.

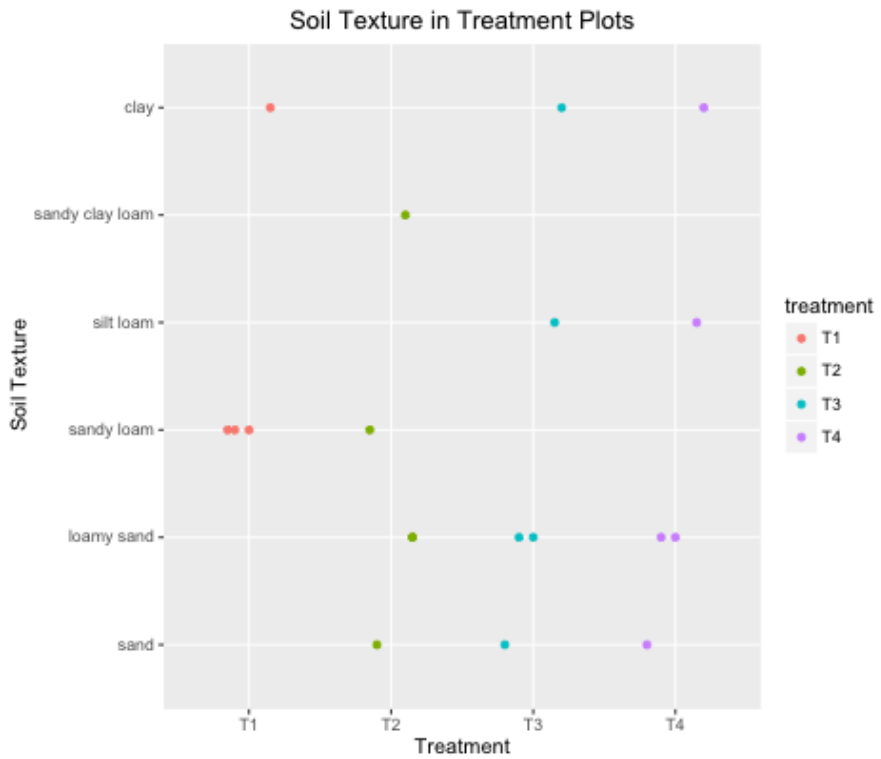

Figure 21. Soil texture classifications by treatment plot.

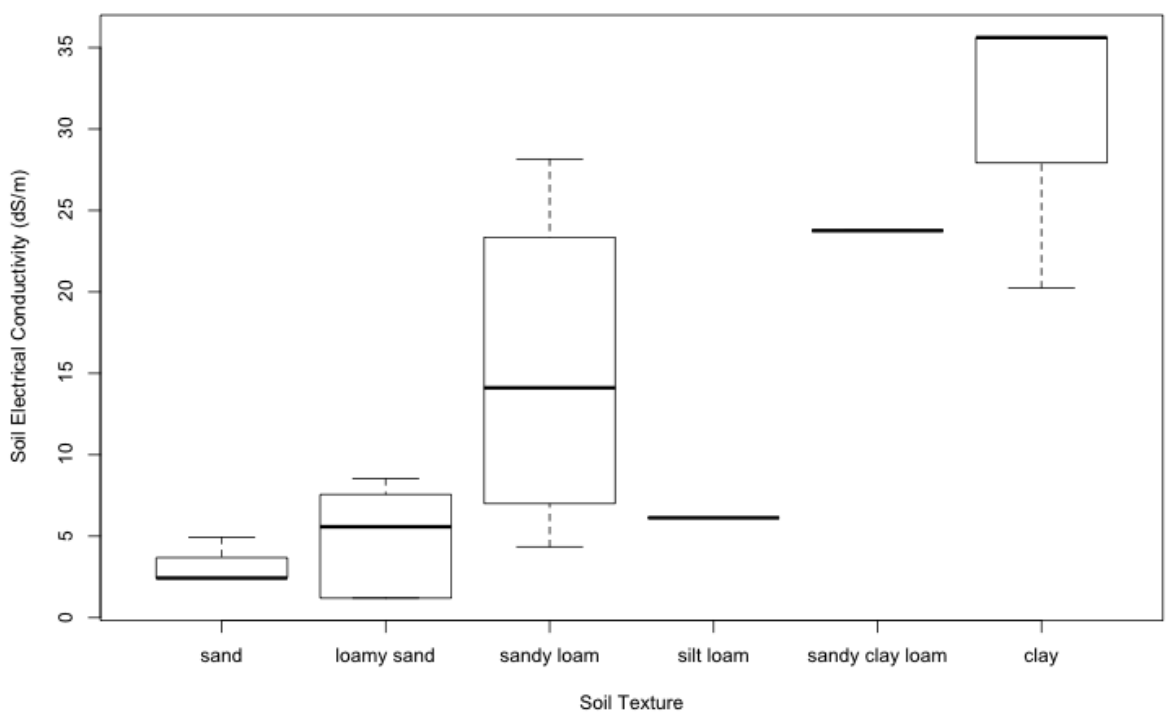

Figure 22. March soil texture and EC showing a general increase in EC with increasing clay content.

Soil Moisture and Groundwater Dynamics

Groundwater

Groundwater levels were shallowest in piezometers located along the main river channel (CILA1 T2 and Cori4 T2), a trend that was consistent throughout the growing season. Baseline depth to water at these two sites was less than 1 meter (0.99m and 0.86m on March 28, respectively). Groundwater was deeper at the Cori1 T3/T4 and Cori3 T3/T4 piezometers, with baseline levels of around 2 meters (2.17m and 1.90m, respectively). The pulse flow arrived at the Laguna Grande Restoration Area on April 4, as can be seen in the rapid increase of groundwater levels in piezometers throughout the site on April 4 and 5 (Figure 23). Groundwater levels initially declined, and rose again around April 25 with the delivery of flows from delivery point Km27 and again on May 6 with delivery of flows from Km18 (see Figure 5 for map of delivery points and Figure 7 for discharge rates). Starting on May 10, groundwater had a long recession period lasting until June 12. After this date, groundwater fluctuations were primarily due to base flow deliveries to the Laguna Grande meander areas, and a base flow delivery from Km18 to the main channel in August (Figure 23). The general groundwater decline from the baseline level pre-pulse flow to the end of the growing season reflects the seasonal pattern caused by agricultural return flows, which are greatest in spring and lowest in fall/winter.

Maximum depth to groundwater from March 1 to November 1 was greatest in T1 sites (mean = 3.80 ± SE 0.19 m) and most shallow in T3/T4 sites (mean = 1.86 ± SE 0.19 m) (Figure 24; Table 1).

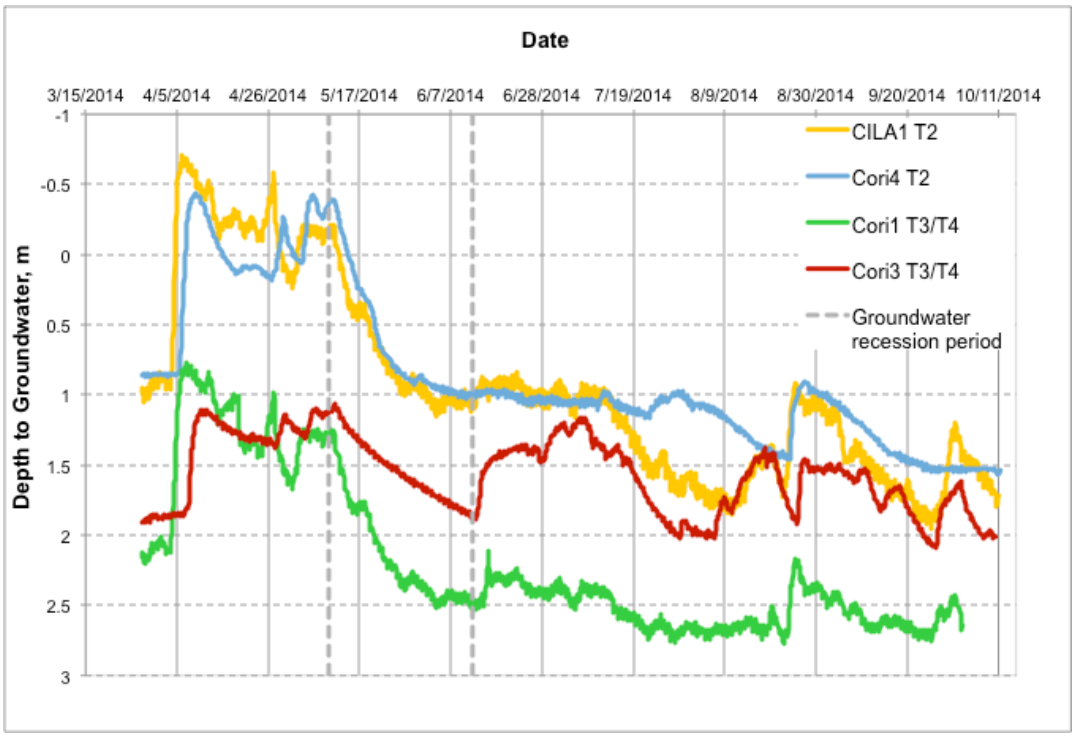

Figure 23. Depth to groundwater from May 28 – October 11, 2014 at CILA1 T2, Cori1 T3/T4, Cori3 T3/T4, and Cori 4 T2.

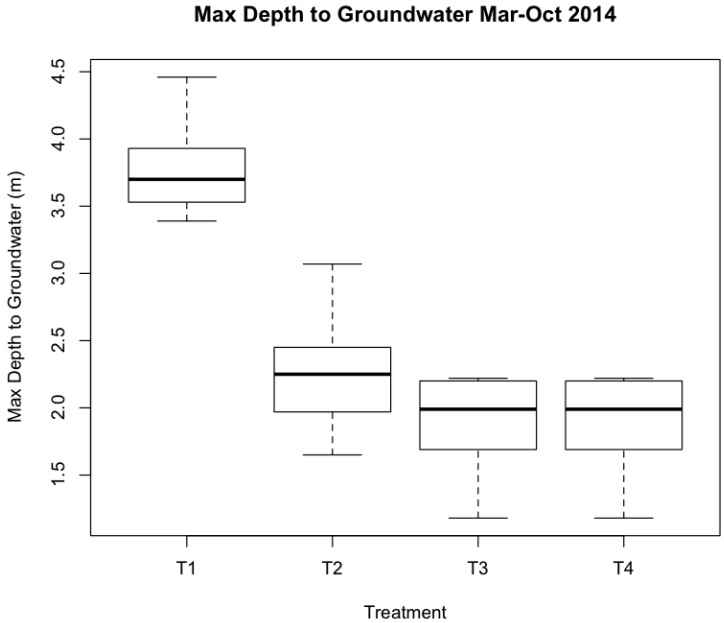

Figure 24. Maximum depth to groundwater from March – October 2014 at treatment plots.

Baseline groundwater levels and changes in levels over the growing season are closely linked to the piezometers' locations along meanders and the main channel. CILA1 T2 and Cori4

T2 piezometers are close to the river channel; groundwater levels are shallower there than areas further from the channel in the floodplain due to infiltration of surface flows in and adjacent to the river channel. The response of groundwater at T2 sites is largely related to flows delivered via the main channel, and there are minimal or no groundwater fluctuations due to deliveries of base flows to meanders. The Cori 3 T3/T4 piezometer is located near a meander base flow delivery point; thus, the groundwater response shows increases and declines caused by base flow deliveries over the growing season. The increase of groundwater levels at Cori3 T3/T4 after mid-June shows that areas nearby the baseflow delivery point had potential for secondary inundation following the initial period of groundwater decline, as groundwater levels at this site rose to nearly the same level in response to baseflow deliveries as the pulse flow delivery. The Cori1 T3/T4 piezometer is located in a meander area that is closer to the main channel, but also has some influence from base flow deliveries. At this piezometer, the groundwater response to both main channel deliveries and meander base flow deliveries is evident, but somewhat muted.

Soil Moisture

Soil moisture (volumetric water content) followed trends in groundwater, and the relationship was more closely linked the deeper the soil moisture sensor from the soil surface (the closer it was to the groundwater table) (Figure 25). The pulse flow arrived to areas outside of the river channel via groundwater first: as groundwater levels rose, soil became wetted in the capillary fringe and eventually became saturated as groundwater continued to rise. Soil water content at 5cm below the soil surface peaked at around 70% for T2 sites and then dropped to 10% or less at the end of the growing season. In the Cori1 T3/T4 site, soil water content at 5cm depth peaked at around 45% following the arrival of the pulse flow, and dropped back down to around 0% before the end of the groundwater recession period. The difference in peak soil water

content between these sites was likely due to textures of surface soils, which were sandier in the T3/T4 sites and more clayey in the T2 sites, particularly Cori4 T2. In the Cori3 T3/T4 site, water content at 5cm depth did not show a response to the pulse flow or to base flow deliveries, which indicates that at the elevation of that soil sensor, inundation did not occur. Similar to the groundwater response at Cori3 T3/T4, soil water content at the lower depths of 50cm and 110cm fluctuated greatly in June-September due to base flow deliveries.

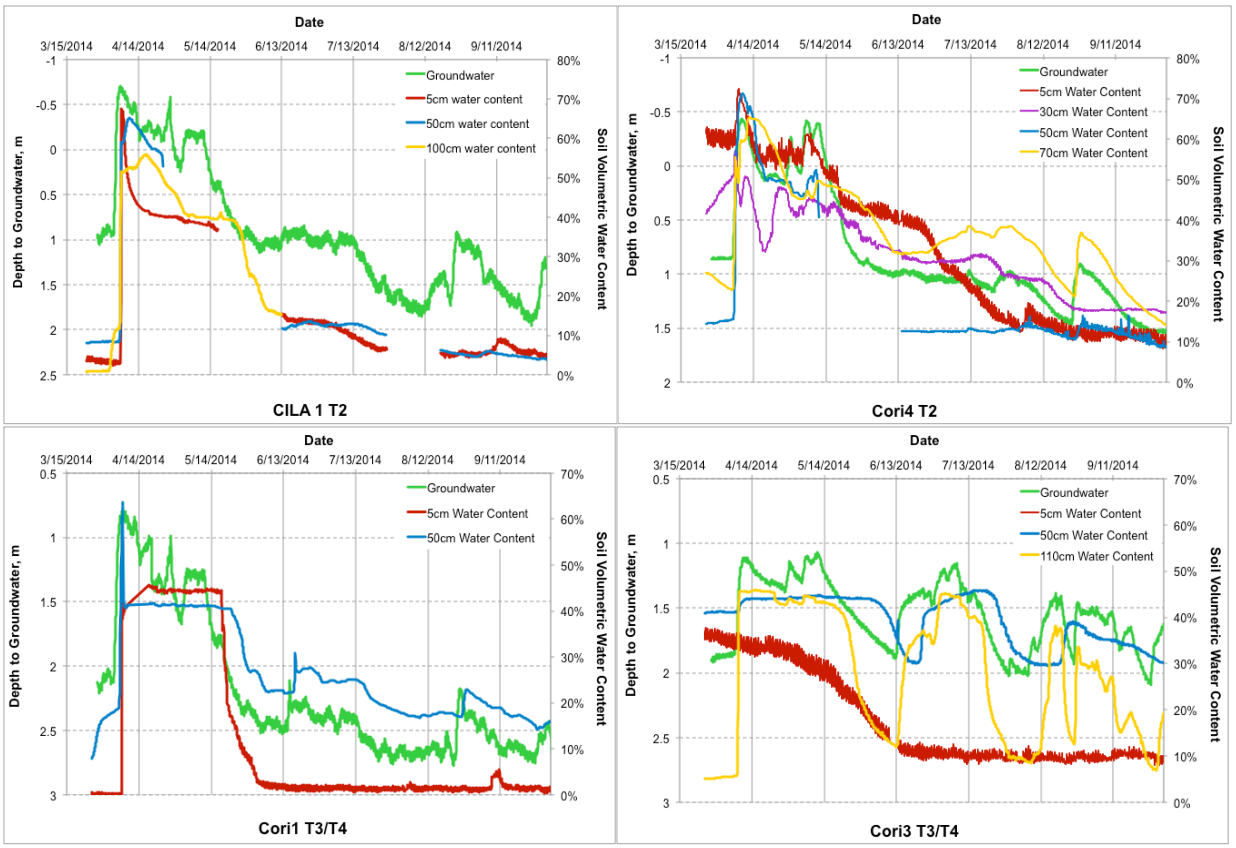

Figure 25. Coupled groundwater and soil moisture response at CILA1 T2 (top left), Cori4 T2 (top right), Cori T3/T4 (bottom left), and Cori3 T3/T4 (bottom right).

Groundwater Recession

The pulse flow groundwater recession period at Laguna Grande was approximately 1 month in duration from May 10- June 12 2014. For the first several days following the start of the recession, recession rates were high (5-9cm/day) at all sites except for Cori3 T3/T4. Starting

on May 22 through the end of the recession period, the recession rate was less than 3cm/day at all sites. This indicates that from May 22 until June 12, recession rates were favorable for cottonwood and willow establishment. For site Cori3 T3/T4, the recession rate was below 3cm/day starting on May 13; this site had a longer period of suitable recession rate for native tree establishment. Although the groundwater recession rate was favorable after May 22 at Cori1 T3/T4, there was a steep decline in soil moisture content at the 5cm depth (likely due to sandy soil composition) (Figure 25), which could have caused high initial seedling mortality.

Figure 26. Groundwater recession rate from May 10 – June 12 2014.

Seedling establishment

Cottonwood-Willow Establishment Model

Cottonwood and willow seedlings germinated in cleared, inundated (T3) and cleared, inundated and seeded treatments (T4) in May (Figure 27). The treatment covariates of canopy cover ($p<0.001$) and bare ground cover ($p=0.0051$) were significant predictors (using penalized quasi-likelihood estimation) of cottonwood-willow May 2014 establishment (Table 2). Establishment in the control treatment (T1) did not differ from the inundated treatment (T2)

(p=1), which was expected since cottonwood-willow seedling count was zero for both. Establishment in the control (T1) and inundated (T2) treatments differed from the cleared inundated (T3) and the cleared, inundated, and seeded treatments (T4) (p-values of <0.001). According to the model, establishment in the cleared, inundated treatment was not significantly different from the cleared, inundated, seeded treatment (p=0.594), which suggests that seed application had no effect on establishment. However, the model could not incorporate an interaction term between treatment and seed availability (the model became over fitted); without the interaction term, the significance of seed availability may be confounding treatment effects of T4 and vice versa. The T4 treatment may have affected seedling establishment in areas of low cottonwood-willow seed availability (see comparison of establishment in T1-T3 plots and T4 plots versus seed availability Figure 28). The cottonwood-willow establishment model results suggest that T1=T2<T3=T4, which is a slight modification to the hypothesis previously stated.

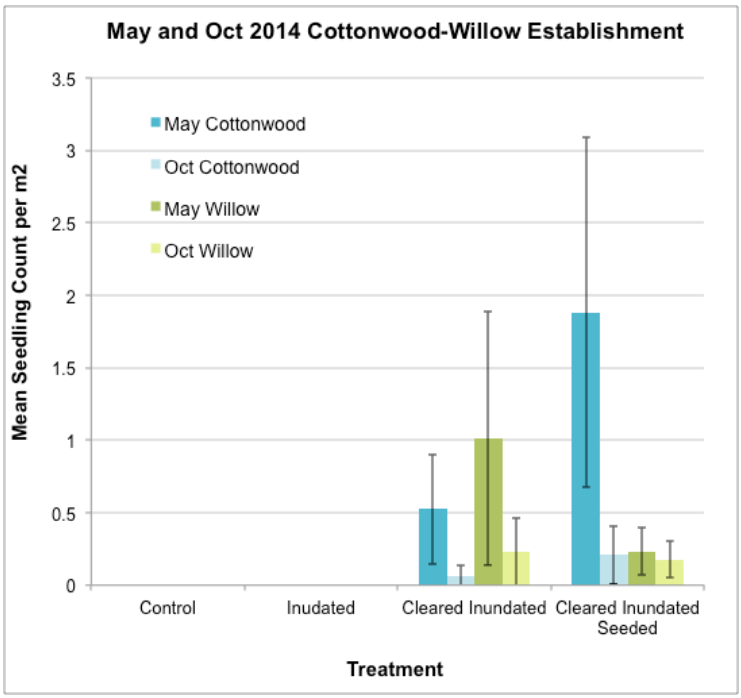

Figure 27. May and October cottonwood willow mean density (m2) per treatment.

Fixed Effects:	Parameter Estimate	Standard Error	t-value	p-value
Canopy	0.1068	0.0210	5.0924	0.0002
Bare Ground	0.0477	0.0142	3.3580	0.0051
Tukey Contrasts:	**Parameter Estimate**	**Standard Error**	**z-value**	**p-value**
T1-T2	-0.1447	2.4618	-0.059	1
T1-T3	23.7308	2.4452	9.705	<1e-05
T1-T4	26.2179	2.3906	10.967	<1e-05
T2-T3	23.8755	3.141	7.601	<1e-05
T2-T4	26.3626	3.0531	8.635	<1e-05
T3-T4	2.4871	2.0135	1.235	0.594

Correlation of Fixed Effects:

	T1	T2	T3	T4	Canopy
T2	-0.148				
T3	-0.375	0.181			
T4	-0.376	0.208	0.653		
Canopy	-0.808	-0.351	0.26	0.239	
Bare Ground	-0.765	0.072	-0.174	-0.162	0.604

Table 2. Results from May cottonwood-willow establishment model with penalized quasi-likelihood estimation.

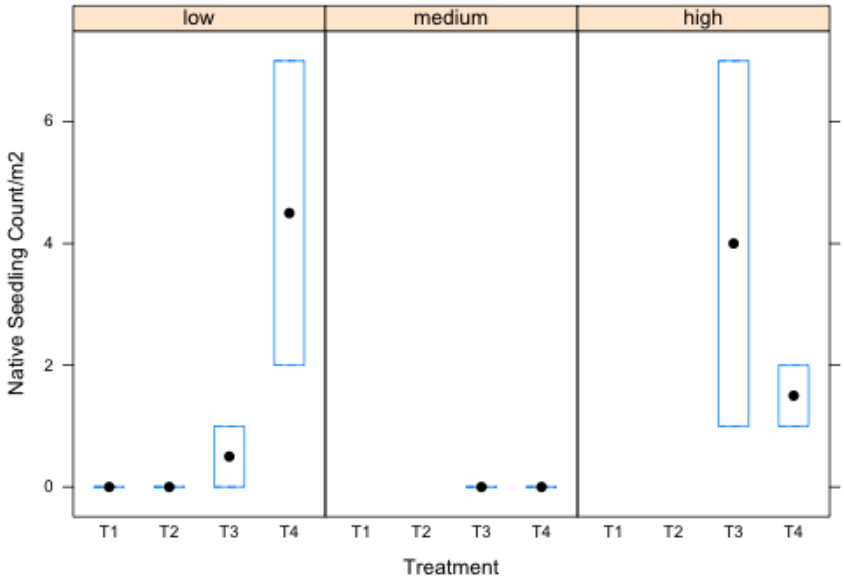

Figure 28. Seed availability and cottonwood-willow seedling establishment in T1-T4 plots. Note the high seedling establishment in T4 plots with low seed availability.

The significance of the random effect environmental variables could not be determined with the penalized quasi-likelihood method. Soil texture was removed from the model to improve model fit (>3 random variables caused model over fitting) and because it was thought that seed availability and soil EC were likely more significant predictors of establishment than texture. The random effect of plot was retained in the model as a way to incorporate environmental variability not included in the monitored factors.

Cottonwood and willow seedlings established in areas of sand, loamy sand, and silt loam textured soils (Figure 29). Seedlings established in treatment plots with mean March soil EC ranging from 1.2-6.13 dS/m (Figure 30 and Table 1). However, mean soil EC in these plots was reduced to a range of 1.11-3.18 dS/m in October; salinity may have been lowered at the time of germination due to the flushing of salts from the soil by surface and subsurface flows.

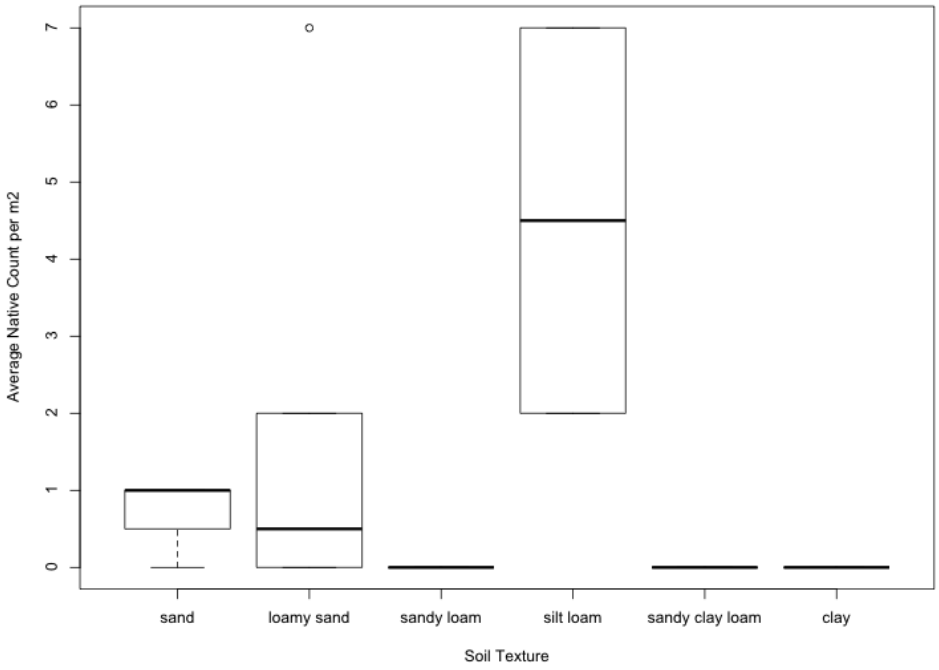

Figure 29. Soil texture of treatment plots where cottonwood-willow seedling establishment occurred in May 2014.

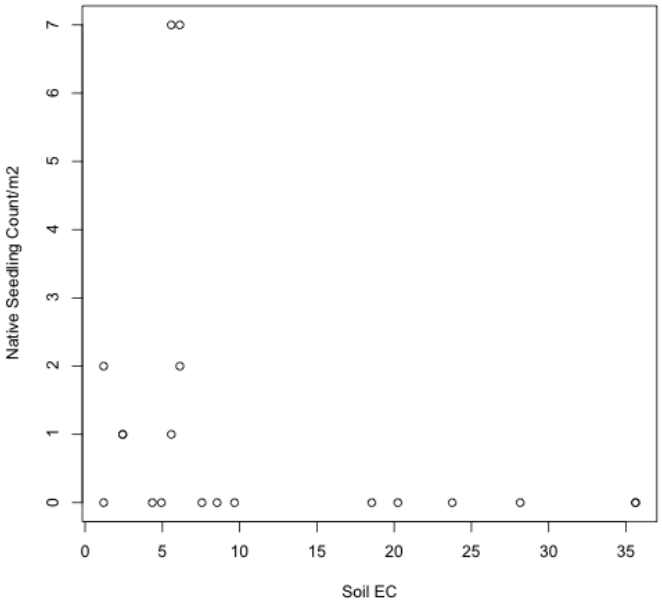

Figure 30. Soil EC of treatment plots where cottonwood-willow seedling establishment occurred in May 2014.

Tamarisk Establishment Model

Tamarisk seedlings established in inundated (T2), cleared inundated (T3), and cleared, inundated, and seeded (T4) plots in May 2014, generally in densities 10 to 100 times greater than cottonwood and willow seedling densities (Figure 31). Treatment covariates of canopy cover ($p=0.00853$) and bare ground cover ($p<0.001$) were significant predictors (using LaPlace approximation method) of tamarisk May 2014 establishment with canopy having a negative relationship, and bare ground having a positive relationship (Table 3). Establishment in the control treatment (T1) differed from establishment in the inundated treatment (T2), the cleared treatment (T3), and the seeded treatments (T4) (p-values of <0.001), with a positive relationship that suggests T1< T2, T3, and T4. Establishment in the inundated treatment (T2) was significantly different than the cleared, inundated treatment (T3) ($p=0.0192$) and marginally significantly different for the cleared, inundated, seeded treatment (T4) ($p = 0.048$) with a negative relationship, suggesting that T2< T3 and T4. Tamarisk seedling establishment in T3 and

T4 was significantly different based on model results (p=0.0083), which was not expected since there was no tamarisk seed application in the T4 plots. However, the significance of the comparison is not reliable due to high collinearity in the model between T3 and T4. The high correlation between T3 and T4 (0.977; Table 3) suggests collinearity, which was not able to be resolved in the model. As such, the p-values of the correlated indicators could be falsely high (Allison 2012). It is unlikely that tamarisk counts of T3 and T4 are truly significantly different, given the mean counts and variance. The tamarisk seedling establishment model suggests that T1<T2<T3?T4, which, aside from the uncertain relationship between T3 and T4, follows the hypothesis previously stated.

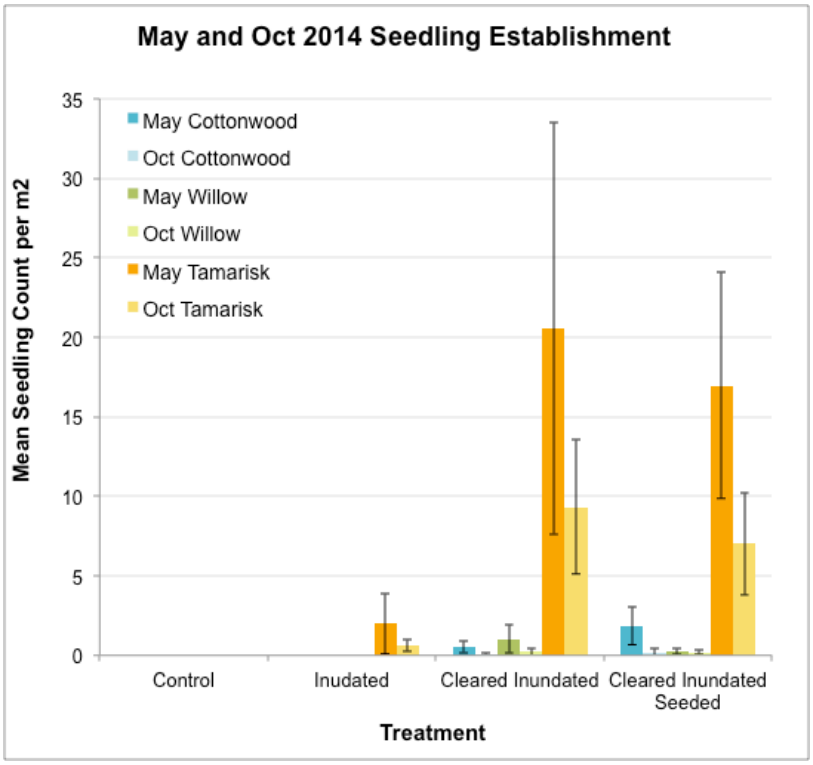

Figure 31. May and October tamarisk, cottonwood, and willow mean density (m2) per treatment.

Fixed Effects:	Parameter Estimate	Standard Error	t-value	p-value
Canopy	-0.04617	0.01755	-2.631	0.00853
Bare Ground	0.11891	0.0191	6.224	4.84E-10
Tukey Contrasts:	**Parameter Estimate**	**Standard Error**	**z-value**	**p-value**
T1-T2	27.0594	1.6214	16.689	<1e-04
T1-T3	21.8069	1.0492	20.785	<1e-04
T1-T4	22.4987	1.0107	22.26	<1e-04
T2-T3	-5.2525	1.8674	-2.813	0.0192
T2-T4	-4.5607	1.8335	-2.487	0.048
T3-T4	0.6918	0.2246	3.081	0.0083

Correlation of Fixed Effects:

	T1	T2	T3	T4	Canopy
T2	-0.232				
T3	-0.104	0.071			
T4	-0.156	0.088	0.977		
Canopy	0.065	-0.212	0.189	0.168	
Bare Ground	-0.29	0.154	-0.349	-0.217	-0.509

Table 3. Results from May tamarisk establishment model with LaPlace approximation.

The significance of the random effect environmental variables could not be determined with the LaPlace approximation method. However, the proportion of variance for random variables of plot and soil texture were close to zero (Table 4) and thus were removed from the model (Starkweather 2010). Soil EC accounted for 79.5% of the variance explained by random variables in the full model, while seed availability accounted for 20.5%. Following removal of plot and soil texture from the model, the respective proportions remained nearly the same as in the full model (Table 4). Using methods from Nakagawa and Schielzeth (2013), the R^2 conditional value of the tamarisk establishment model was calculated at 0.9902, which means that 99% of variance was explained by fixed and random effects of the model. The R^2 marginal value was 0.8831, which signifies that 88.3% of variance was explained by fixed effects alone; 10.7% was explained by random effects. These R^2 numbers are unusually high, and further investigation into their accuracy is needed.

Random Effects: (Full model)	Variance	Standard Deviation	Proportion of Variance
Plot	2.67E-13	5.17E-07	0.000
Soil EC	1.26E+01	3.54E+00	79.481
Soil Texture	1.19E-09	3.45E-05	0.000
Seed Availability	3.24E+00	1.80E+00	20.519
Random Effects: (Plot and Texture Removed)	**Variance**	**Standard Deviation**	**Proportion of Variance**
Soil EC	1.25E+01	3.54E+00	79.468
Seed Availability	3.24E+00	1.80E+00	20.532

Table 4. Relative proportion of variance of random effects in the May tamarisk establishment model.

Tamarisk seedlings established in areas with sand, loamy sand, silt loam, and clay soil textures (Figure 32). Seedlings established in treatment plots with mean March soil EC ranging from 1.20 to 35.65 dS/m (Figure 33 and Table 1). Mean soil EC in these plots was reduced to a range of 1.11-6.29 dS/m in October; similarly as for native species germination, salinity may have been lowered at the time of germination due to the flushing of salts from the soil from surface and subsurface flows. However, tamarisk can continue to extract water out of soils with salinity as high as 38 dS/m (Vandersande et al. 2001), so establishment on soils with EC of 35 dS/m is not unusual for this highly salt tolerant plant.

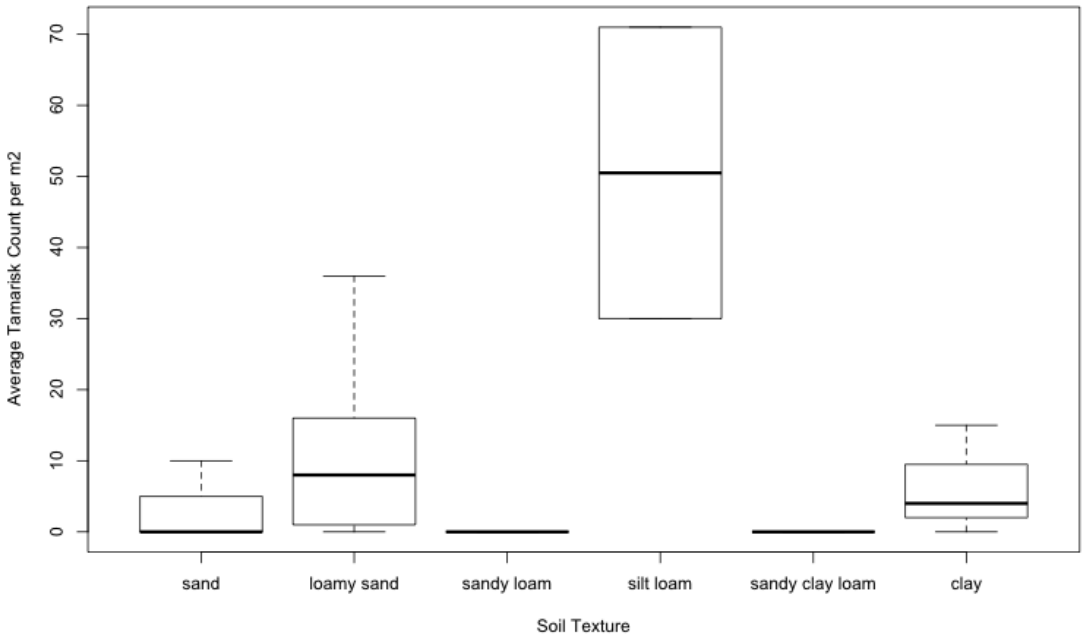

Figure 32. Soil texture of treatment plots where tamarisk seedling establishment occurred in May 2014.

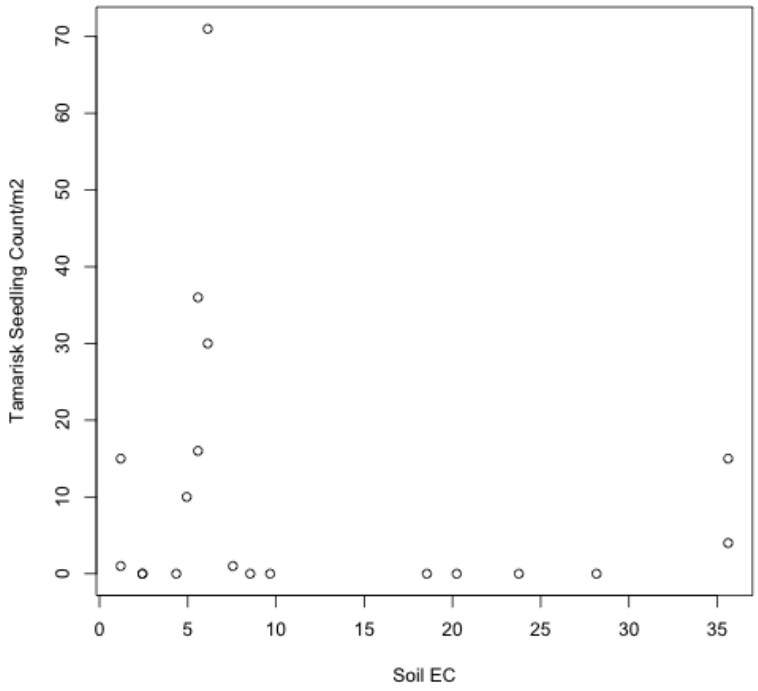

Figure 33. Soil EC of treatment plots where tamarisk seedling establishment occurred in May 2014.

End of Growing Season Seedling Density

From May to October, the number of treatment plots with cottonwood or willow seedlings went from 7 to 4 (Table 5). The percent difference of seedling count between May and October (Oct/May*100) ranged from 0% to 42.4%. The environmental variables of canopy cover, bare ground, and maximum depth to groundwater were not significantly different in areas with sustained cottonwood-willow presence in October than those without (p-values 0.6098, 0.1361, and 0.0825, respectively) (Table 6). Soil EC was significantly higher in areas of cottonwood-willow absent plots than present (p=0.0194). October seedling density was significantly higher in plots with silt loam soil texture versus sand soil texture (p=0.0344). Interestingly, both cottonwood-willow and tamarisk absent plots in October had sandy soil texture classifications. It's possible that sandy soils had rapid soil moisture draw down rates, which may have led to seedling mortality in these plots.

Site	Treatment	POFR+SAGO Count		POFR+SAGO May to Oct % Difference	Tamarisk Count		TASP May to Oct % Difference
		May	October		May	October	
	T1	0.00	0.00	-	0.00	0.00	-
	T2	0.00	0.00	-	0.00	0.83	-
	T3	0.52	0.00	0.00	0.00	0.00	-
CILA1	T4	0.08	0.00	0.00	0.00	0.00	-
	T1	0.00	0.00	-	0.00	0.00	-
	T2	0.00	0.00	-	9.50	0.00	0.00
	T3	0.67	0.00	0.00	16.33	11.17	68.37
Cori1	T4	6.50	1.17	17.95	36.33	18.50	50.92
	T1	0.00	0.00		0.00	0.00	-
	T2	0.00	0.00	-	0.40	0.33	83.33
	T3	0.00	0.00	-	14.83	13.50	91.01
Cori2	T4	0.00	0.00	-	3.50	1.83	52.38
	T1	0.00	0.00	-	0.00	0.00	-
	T2	0.00	0.00	-	0.00	0.00	-
	T3	6.50	1.50	23.08	70.50	22.00	31.21
Cori3	T4	1.67	0.71	42.40	29.67	7.83	26.40
	T1	0.00	0.00	-	0.00	0.00	-
	T2	0.00	0.00	-	0.00	2.00	-
	T3	0.00	0.00	-	1.00	0.00	0.00
Cori4	T4	2.33	0.06	2.57	15.33	7.00	68.37

Table 5. May and October 2014 cottonwood (POFR), willow (SAGO), and tamarisk (TASP) density comparison (Oct. count/May count *100).

Variable	POFR + SAGO Present (Oct)	POFR + SAGO Absent (Oct)	TASP Present (Oct)	TASP Absent (Oct)
Canopy Cover (%)	10.74 ± 7.35	6.23 ± 3.61	18.19 ± 10.77	42.89 ± 36.15
Bare Ground Cover (%)	42.69 ± 18.85	69.48 ± 10.06	47.66 ± 11.53	45.46 ± 34.22
Soil EC (dS/m)	1.48 ± 0.17	2.77 ± 0.41	3.09 ± 0.74	1.8 ± 0.69
Max Depth to Groundwater (m)	1.84 ± 0.23	2.21 ± 0.01	1.92 ± 0.13	1.42 ± 0.23
Soil Texture	Sand, silt loam, loamy sand	Sand	Sand, sandy loam, loamy sand, silt loam, clay	Sand

Table 6. Summary of environmental variables in plots where cottonwood (POFR), willow (SAGO), and tamarisk (TASP) were present/ absent.

Chapter 5: Discussion

Cottonwood and Willow Establishment

Vegetation establishment results from the study suggest that Minute 319 environmental flow deliveries at Laguna Grande did not provide a disturbance of great enough magnitude to restore key fluvial processes of erosion, sediment deposition, and vegetation scour. The lack of bare soil surfaces along the densely vegetated river channel likely resulted in no cottonwood or willow species establishment in non-cleared areas. In addition, without the excavation and grading implemented in the meander areas prior to the flow deliveries, it is unlikely that most of the backwater areas in meanders would have been inundated at all, since the flows would not have been of great enough magnitude to reestablish connectivity with the main channel. However, the flow deliveries did provide some hydrological functions such as wetted surface soils, increased groundwater levels, and reduced soil salinity. This, in combination with active management of restoration sites, led to at least a temporary recovery of some abiotic-biotic interactions and establishment of cottonwood and willow species in the majority of cleared, inundated areas.

In Laguna Grande, cottonwood and willow seed availability was less of a limiting factor for establishment due to the abundance of mature trees in patches of remnant riparian habitat and planted, irrigated areas. This is likely not the case on other sections of the lower Colorado River in the U.S. and Mexico or in other riparian systems with small native seed source populations, and seed application may be an effective method of increasing cottonwood and willow establishment in these areas.

Additionally, synchronizing the timing of the flow recession with seed dispersal is critical for successful establishment. The cottonwood seed dispersal period of 10-11 weeks reported in

this study was longer than that reported in other studies (Cooper et al. 1999; Warren and Turner 1975), but cottonwood seed dispersal ended one week prior to the start of the groundwater recession period at three out of four instrumented sites. Interestingly, the mean cottonwood count in May and October was much higher in the seeded sites than the non-seeded sites, a trend that was absent for willow seedlings. This suggests that seed application may have had more of an effect on cottonwood establishment than willow due to differences in seed dispersal timing. It's likely that wind- and water-dispersed cottonwood seed was still viable when the recession period started (Braatne et al. 1996), but recession rates of >4cm/day for the first two weeks of the recession period may have led to mortality for cottonwood seedlings established on higher elevation areas (Shafroth et al. 1998, Mahoney and Rood 1998). Willow trees continued dispersing seed during the slowest recession rate period in late May and early June (Stromberg et al. 1993). Unfortunately, seedling densities were too low to conduct separate analyses on cottonwood and willow species; a different sampling method, increased treatment plot size and/or increased number of treatment plots and sub-samples could provide more manageable density data in future studies.

An alternative explanation for higher cottonwood count but not willow count in seeded sites is that applied cottonwood seed had higher viability than applied willow seed, although germination tests were conducted prior to seed application to account for differences in seed viability; an adjusted seed weight was calculated based on viability to achieve desired densities. However, Grabau et al. (2011) reported lower Goodding's willow establishment success than Fremont cottonwood using the hydroseeding method.

Soil salinity was likely a limiting factor for cottonwood and willow seedling establishment in some areas. Shafroth et al. (2008) suggest that soil EC of 0-4 dS/m is suitable

for plant growth, while 4-8 dS/m is considered tolerable but may limit plant growth. Beyond 8 dS/m, only salt-tolerant plants are able to establish and grow. No cottonwood or willow seedlings established in areas with soil salinity above 8 dS/m in this study.

Tamarisk Establishment

Results support previous observations that tamarisk seedlings are more successful at establishing in a wider range of environmental conditions than native woody riparian tree species (Shafroth et al. 1995; Shafroth et al. 2000). Tamarisk seedlings established in areas of higher soil EC and on more soil textural classes than cottonwood and willow seedlings. However, the lack of cottonwood and willow establishment on clay soils was likely due to the high soil salinity of clay soils in the study; a previous study found a preference of both tamarisk and cottonwood seedlings to clay soils over sand (Sher and Marshall 2003). Tamarisk seedlings established in inundated, non-cleared areas with relatively high canopy cover and low bare soil availability in densities that were above that of cottonwood and willow seedlings in cleared areas. Part of this success is likely due to their reproductive capacity and dense distribution in the Delta. The "low" ranking for tamarisk seed availability was <50 seeds/60cm^2/two weeks, which was only observed in areas where tamarisk had been previously cleared. Even at the most abundant seed dispersal sites for cottonwood and willow, the highest *combined* total was 70 seeds/60m^2/two weeks, showing a clear reproductive advantage of tamarisk over native riparian tree species. Seed dispersal monitoring ended in mid June, but informal observations at Laguna Grande suggest that tamarisk continued to disperse seed throughout the entire growing season similar to other studies (Shafroth et al. 1998; Cooper et al. 1999), while willow dispersal ended around the beginning of August (Stromberg 1993).

Maximum depth to groundwater did not appear to affect the persistence of cottonwood, willow, or tamarisk seedlings in October; however, all inundated sites (T2, T3, and T4) had maximum groundwater levels of less than 3 meters, which means it likely was not a limiting factor for seedling establishment (Shafroth et al. 2000).

In cleared areas, tamarisk seedlings established in high densities in combination with cottonwood, willow, *Baccharis* spp., arrowweed, and other native plant species. These results support previous observations that tamarisk seedling sprouts will not outcompete native seedlings at the colonization stage, provided there are adequate abiotic conditions for growth and survival (Sher and Marshall 2003; Sher et al. 2002; Sher et al. 2000). Growth was not measured in the study, but photographs taken from treatment plots demonstrate the ability of cottonwood and willow species to outcompete tamarisk through rapid growth in height and canopy (Figure 34) (Sher and Marshall 2003; Sher et al. 2000).

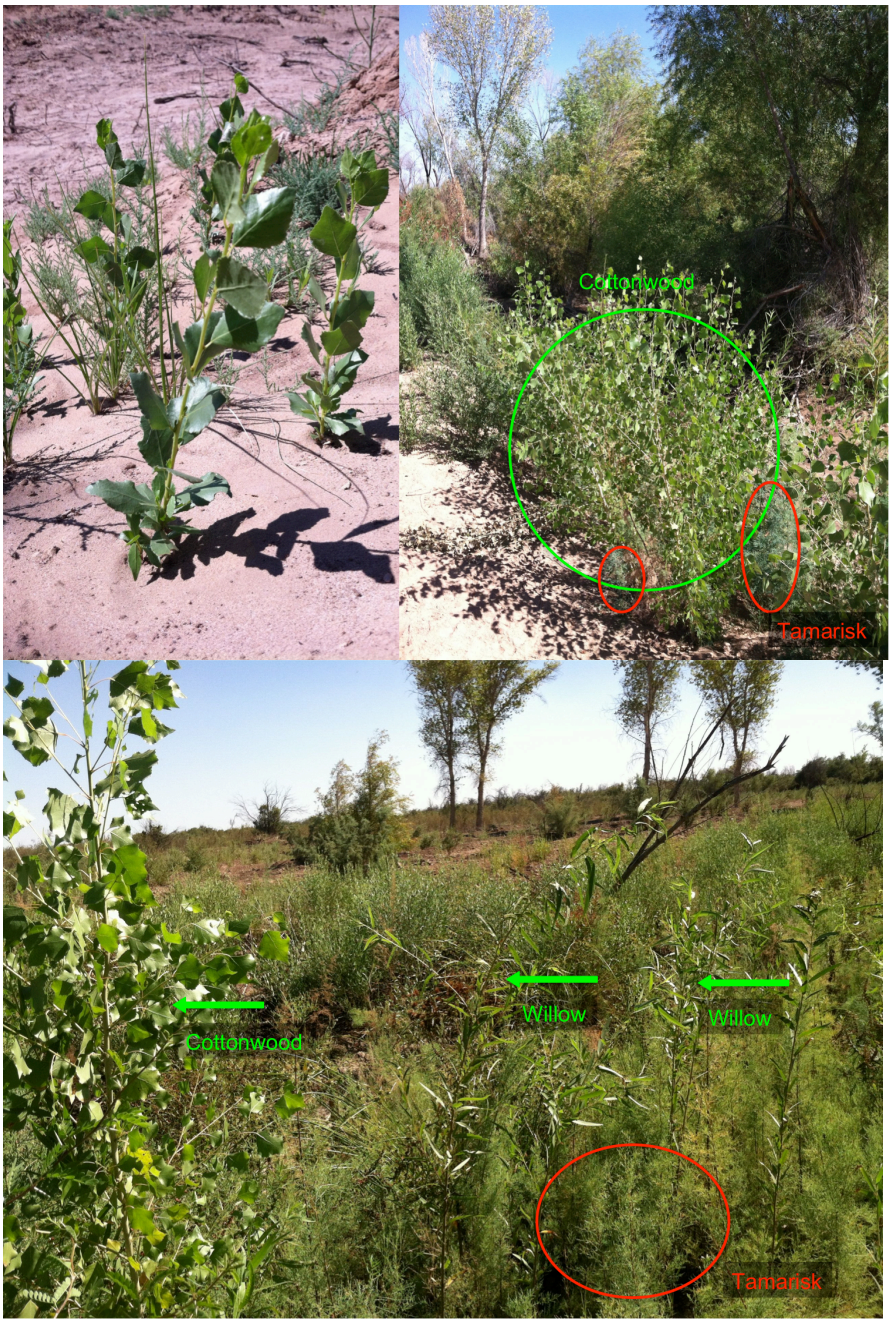

Figure 34. Cottonwood and willow seedling establishment in the presence of a dense tamarisk understory. Photos taken at Laguna Grande in 2014. Top left: June 2014 at Cori3 T3/T4. Top right: establishment along an unmonitored meander area in Laguna Grande, October 2014. Bottom: October 2014 at the Cori3 T3/T4.

Novel Riparian Communities in the Delta

The persistence of novel plant communities in the Delta following restoration treatments and flow deliveries suggests that ecological thresholds have been surpassed and novel

communities are resilient—that is, disturbance is unlikely to shift the system back to a previous historic ecosystem state (Suding et al. 2004). Even with flood flows of similar magnitude and duration as flows in the 1980s, it is unlikely that tamarisk would disappear entirely from the system (it didn't following flood flows in the 80s). The persistence of novel plant communities in the Delta is not necessarily a bad thing, however. Although dense tamarisk stands have been shown to have low bird species richness (Anderson et al. 1983), mixed tamarisk and native plant communities have high structural diversity that can support relatively high bird diversity (Van Riper et al. 2008). In areas along the Colorado River where abiotic conditions can no longer support native riparian communities, tamarisk can provide habitat in what might otherwise be barren soil or sparse desert vegetation (Sogge et al. 2008). Van Riper et al. (2008) found high abundance of many riparian bird species in mixed riparian habitat with 40-60% tamarisk cover, and recommend incorporation of 20-40% native vegetation cover as a feasible and beneficial restoration goal instead of complete tamarisk eradication.

Furthermore, with increasing drought and temperatures predicted in the short- and long-term due to climate change, water scarcity in the Colorado River Basin is inevitable (US Bureau of Reclamation, 2012). In a highly altered riparian environment, tamarisk may be better adapted to novel conditions and can provide ecosystem functions (functional redundancy) that might otherwise be lost in a less resilient native community.

Managing for Resilience

In the case of the Colorado River Delta, environmental conditions are almost entirely dependent on human activities, and consequently, so are changes in ecosystem resilience and potential restoration opportunities. Agricultural water management in the floodplain significantly impacts the hydrology of the river corridor and consequently the condition and extent of riparian

habitat (Carrillo-Guerrero et al. 2013; Hinojosa-huerta 2013; Glenn et al. 2013). In some sections of the river, groundwater is highly depleted due to pumping for agricultural irrigation, and riparian restoration in these areas would require extreme amounts of energy and investment with potential for failure. In other sections of the river, such as Laguna Grande, existing and restored cottonwood-willow habitat is maintained by a shallow groundwater table supported by agricultural return flows (Glenn et al. 2013; Carrillo-Guerrero 2013). Water management decisions including those concerning agricultural deliveries, lining of canals, and the delivery of environmental baseflows and/or pulse flows will all impact the sustainability of riparian habitat on the Colorado River in Mexico in the future. Aspects of ecosystem resilience including connectivity, interaction across scales, response diversity, and functional redundancy will similarly be impacted by societal values and resulting management decisions (Biggs et al. 2012).

Lessons Learned and Adaptive Management

In terms of restoration implementation lessons learned, for this section of river, the distinction between the "pulse" flow and "base" flow was hydrologically meaningless. Restoration managers could essentially achieve similar local hydrological effects with base flow deliveries on site as with a pulse flow of water delivered upstream (at least at the magnitude of the 2014 pulse flow). However, landscape-scale connectivity is highly important for ecosystem functioning, and impacts on resilience and long-term habitat sustainability should be considered when assessing costs and benefits associated with flow delivery decisions.

A less steep meander bank slope likely would have provided a greater area for native species establishment and would lead to less dramatic initial declines in soil water content as flows recede. Additionally, improved water control would allow site managers to control recession rates more effectively and could reduce seedling mortality caused by secondary

inundation, which occurred at the Cori4 T3/T4 plots. Inundation duration was also an issue at the CILA1 site, where flows backed up in the meander channel due to an illegal instream dike erected by a nearby farmer in the river channel. The area was inundated for a long period of time, which resulted in salt accumulation on the surface due to water evaporation. Lastly, cottonwood and willow seed application using the hydroseeding method has the potential to increase native establishment when seed is limited due to lack of seed source or if the timing of seed dispersal is desynchronized with the recession period. However, accessibility with hydroseeding equipment can be a challenge for areas along rivers or meanders with no road access.

This study provides a direct example of adaptive management in restoration through the use of an experimental design to assess outcomes of management actions (Williams 2010). Adaptive management is recommended for management and restoration of novel systems, due to the high degree of uncertainty that exists surrounding abiotic-biotic interactions of such systems and how management actions will impact them (Suding 2011; Seastedt et al. 2008). Adaptive management is also recommended in the context of environmental flow deliveries in order to evaluate benefits of different flow delivery amounts and methods and adjust flow recommendations (Cooper et al. 2012; Richter et al. 2006). Under an adaptive management framework, decisions can incorporate empirical process-based evaluations of species and ecosystems in order to promote resiliency and persistence under new environmental conditions. Monitoring and the incorporation of monitoring results into management plans are critical in understanding current ecosystem conditions and assessing drivers of change. Informed and cost-effective management decisions are more likely to result when quantitative ecosystem assessments are incorporated into restoration actions (Suding 2011).

In summary, management activities and environmental flow deliveries at current groundwater levels and native seed availability can support the establishment and persistence of mixed native-nonnative plant communities along the Colorado River riparian corridor in the Delta. The study presented is a small-scale experiment; corridor-wide vegetation monitoring results from the Minute 319 science team will provide additional insight on the potential to scale-up experimental results for large-scale restoration management.

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