

Chapter 4. Invasion and Restoration of Western Rivers Dominated by *Tamarix* spp.

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Introduction

Scientists, land managers, government, and private institutions in the United States have given much attention to invasive control and restoration projects along western rivers; in the West, removal of *Tamarix* spp. (tamarisk, saltcedar) has been a primary focus of these projects (Dennison et al. 2009; González et al. 2017a; Harms and Hiebert 2006; Shafroth et al. 2008). These trees were first introduced to North America during the 1800s from Eurasia mainly to decrease erosion, to be wind breaks, and to slow down water flow in riparian and agriculture areas (DiTomaso 1998). Since that time, this invasive tree has become the third most common woody species and second highest tree cover in the southwestern United States (Friedman et al. 2005). Despite the fact that invasive plant species removal has long been a priority in restoration of riparian ecosystems (González et al. 2015), we are only beginning to understand the ecological impact of the removal of invasive species from such ecosystems. Here we will provide an overview of *Tamarix* ecology along rivers in the western United States and the results of our research monitoring plant communities in 25 riparian sites over 3 years, with a particular interest in the results of removal of *Tamarix* by various methods including biological control by a defoliating beetle.

Tamarix Introduction and Impact

During the 1800s, eight species of *Tamarix* were first introduced to the United States from Europe, Asia, and North Africa. In the 1920s, *Tamarix* spread and occupied about 4,000 ha of riparian habitat in the southwestern United States. By 1987, the area invaded by *Tamarix* increased to about 600,000 ha (Brock 1994; DiTomaso 1998; Gaskin and Schaal 2002; McDaniel et al. 2005; Nagler et al. 2011) and it now occupies approximately 800,000 ha. The most widely naturalized species are *Tamarix ramosissima*, *Tamarix chinensis*, and their hybrid (Friedman et al. 2005; Gaskin and Schaal 2002).

Tamarix is classified as deciduous with either a tree or shrub growth habit. It is also a paradoxical plant from the selection theory perspective as it uses both r and K strategies (sensu McArthur and Wilson 1967) with both a large number of offspring and high longevity (Sher 2013). *Tamarix* individuals can live for more than 100 years, and one large tree produces about 500,000 seeds per year (McDaniel et al. 2005; Stromberg et al. 2007b). It is highly tolerant of drought; its evapotranspiration rates are highly variable and adaptable to water availability, and as a facultative phreatophyte, uses both surface and ground water (Glen and Nagler 2005). Its deep roots can reach declining water tables when other species cannot (Shafroth et al. 2000). *Tamarix* also has adaptations that allow it to tolerate greater salinities than native vegetation (Shafroth et al.

1995). Together, these traits have allowed *Tamarix* to take advantage of the environmental stresses associated with dammed rivers (Sher 2013; Stromberg et al. 2007b).

Under natural river flow conditions, native cottonwoods (*Populus* spp.) are typically competitive over invasive *Tamarix* (Huston 2004; Sher et al. 2000, 2002; Stromberg et al. 2007a), suggesting that *Tamarix* would not thrive under natural conditions. Rather, as a result of widespread flow control, many riparian ecosystems of the Southwest no longer support native vegetation, allowing *Tamarix* to dominate (Sher 2013).

Thus, *Tamarix* dominance is the result of ecosystem change, but it can also be the cause (passenger vs. driver, sensu MacDougall and Turkington 2005; Johnson 2013). Like many invasive species, once *Tamarix* establishes it can also alter the ecosystems in which it occurs, increasing fire risk (Drus 2013), changing the morphology of the stream bank (Auerbach et al. 2013), and increasing the soil salinity (Ohrtman et al. 2012), among other effects (Didham et al. 2005; Johnson 2013; Sher 2006). Such changes add to the inhospitable quality of the riparian area following river flow regulation to native plant communities.

All of these environmental changes mean that *Tamarix*-dominated stands are associated with lower diversity in plant and animal communities (Sher 2013), making *Tamarix* removal a high priority for many private and public institutions, despite its considerable cost (de Wall 1994; Zavaleta 2001). However, the dual role of *Tamarix* as both ecosystem passenger and driver has profound implications for restoration (Bauer 2012). As a driver of ecosystem change, removing *Tamarix* should solve many of the concerns for conserving riparian areas. No doubt, this was the perspective of early aims to remove *Tamarix* (see Chapter 1). However, as a passenger of ecosystem change, its removal alone should affect no positive change; recent research suggests at least in some cases without natural flow regimes, native riparian vegetation cannot reestablish (González et al. 2017a; Merritt and Poff 2010). Furthermore, some changes that *Tamarix* make to riparian ecosystems are difficult to reverse simply by removing *Tamarix* itself, such as soil salinization (Merritt and Shafroth 2012). Thus it is imperative that we carefully monitor the ecological consequences of *Tamarix* removal projects.

Since *Tamarix* removal sites are not selected randomly, monitoring at a single point in time or combining sites with different periods of time since *Tamarix* removal can prove misleading. Variables such as weather patterns (e.g., dry year coinciding with single time monitoring) or variation in time since removal can confound analyses and lead to misinterpretation of results. Multiple years must therefore be sampled to determine whether patterns of recovery are real or a product of confounding variables such as years since removal, drought years, or flood events at a particular location in a particular year. Because of variability associated with geographic location, it is also imperative that sites of the same age but different treatments be sampled at the same time. Rarely have both been done in a single project.

Vegetation Response to *Tamarix* Removal

While invasive plant species removal has long been a priority in the restoration of river systems, relatively little is understood about the ecological impact of the removal of invasive species from the ecosystem (Bay and Sher 2008; Cuevas and Zalba 2010; Ostojia et al. 2014; Shafroth et al. 2005). Previous research on plant community response

to *Tamarix* removal is mixed. Harms and Hiebert (2006) surveyed 33 *Tamarix* removal and non-removal sites where only passive revegetation had been done. They found a decrease in the cover of invasive *Tamarix* compared to the control sites and a significant increase in native foliar cover in the Mojave region.

However, there was no consistent change in native cover in the two other regions sampled, and when *Tamarix* was excluded from data analysis, they found no difference regarding species composition across all sites. In contrast, in a similar study but with active revegetation, Bay and Sher (2008) found increases in native cover after tamarisk removal could be explained by several factors, including that the relative cover of planted or native species was greater in the sites when the removal period was greater than 8 years, when there was lower salinity, when there was coarser soil texture, and when closer to a body of water. More recently, Ostojia et al. (2014) examined short-term responses of riparian plant communities to *Tamarix* removal. They found increased species richness and diversity, but still very low absolute diversity and abundance of native species. Most of these studies did not control for the number of years since *Tamarix* removal and did not follow the same sites over time. Furthermore, none examined the response of plant communities to *Tamarix* control using the biological control agent, *Diorhabda* spp.

Diorhabda spp. is a beetle herbivore that defoliates *Tamarix* spp. in its native range. After extensive research on host specificity, it was approved for release as a biological control agent in 2001 (DeLoach et al. 2003; also see Chapter 5). Since its release, *Diorhabda* has moved farther and faster than anticipated (Nagler et al. 2014). Studies so far show that its impact on *Tamarix* as well as riparian plant communities is variable and difficult to predict (González et al. 2017a; Hultine et al. 2015; Kennard et al. 2016).

In the largest review to date of the consequences of *Tamarix* removal, González et al. (2017a) compiled monitoring data for plant communities following *Tamarix* removal across the Southwest in 416 sites, including those with *Diorhabda* spp. biological control. They found that for most removal methods, native cover significantly increased following *Tamarix* removal, but the increases were extremely gradual, averaging less than 2 percent per year. Native cover was not consistently increased with active revegetation, and the community did not typically revert to the mesic plant species that had occurred there historically. Exotic species also increased in cover after *Tamarix* removal (“secondary invasion”), particularly with high disturbance treatment methods, even though exotic cover decreased over time in biocontrol treatment sites (González et al. 2017b). However, the González et al. (2017b) study amassed data from regions all over the western United States, possibly obscuring the importance of regional effects. Nowhere in the literature is there a study that compares regions that were measured at the same times in the same ways with comparable removal dates over time.

In Colorado, biocontrol beetles were released in 2005 on the Western Slope and have become nearly ubiquitous there. At the time of this study (2010–2012), the biocontrol had been released but not yet well-established on the Eastern Slope. This situation provided a unique opportunity to study the impact of *Diorhabda* spp. alone and in combination with other removal techniques on *Tamarix* cover, as well as the consequent effect on the rest of the plant community. This is one of the only studies in which multiple sites in different regions have been sampled by the same methods for multiple years

and multiple seasons within a year, thus representing a unique opportunity to investigate the impact of *Tamarix* removal on plant communities.

Sampling Methodologies and Invasive Removal Techniques

We monitored vegetation response after invasive *Tamarix* removal in sites east and west of the Rocky Mountains in Colorado for 3 and a half years beginning in fall 2009. This project aimed to establish baseline data for long-term monitoring and to develop the best practices to make recommendations for monitoring by land managers.

Our broad interest was understanding the response of plant communities along rivers in Colorado to the removal of a dominant invasive species. Our specific questions were: (1) How do riparian plant communities differ between the Eastern and Western Slope of Colorado that may influence response to restoration? (2) Are removal methods effective in reducing percent cover of *Tamarix*? (3) Does it matter whether the biological control is present or not? (4) How do exotic and native plants respond to *Tamarix* spp. removal in each of these regions? Our intent, by measuring the impact of *Tamarix* removal in the ecosystem via the measurement of vegetation parameters over time, is to help answer some of the controversial questions about the ecological impact of *Tamarix* removal on these ecosystems, including the presence of the biological control.

Methods and Site Locations

We monitored vegetation in a total of 25 sites: nine sites in three reaches located in western Colorado (fig. 7a) and 16 sites in 5 reaches located in eastern Colorado (fig. 7b). The western sites, at approximately 38°1'0" N 108°49'26" W, are located in the Upper Dolores Watershed including Big Gypsum Valley, Disappointment Valley, and Slickrock Canyon. The eastern sites, at approximately 37°33'0" N 103°38'21" W, are located in the Purgatory Watershed including Chacuaco Creek, Plum Creek, and Apishapa River. These sites were mostly on private land in the East and BLM land in the west, selected by land managers for our group to survey either because they were candidates for tamarisk removal, or they represented an un-invaded ecosystem. Although they were not randomly selected, the sites represent a range of representative riparian ecosystem conditions, from degraded to fairly un-impacted. Native-dominated sites are referred to throughout this chapter as positive reference sites (González et al. 2015). The Society for Restoration International (SERI) Primer states that it is imperative to identify sites that represent the goal (positive references) of restoration (SERI 2004). Sites dominated by *Tamarix* where there was no active removal are referred to as negative reference sites. Active removal refers to the removal of *Tamarix* by the following methods: cut stump (chainsaws are used cut down *Tamarix* and herbicide is applied to the remaining stump); track hoe (*Tamarix* is removed using large machinery); or lastly, spraying herbicide by helicopter.

By employing both sampling over time and reference sites, we address potential issues of confounding time and space, referred to as a BACI approach: Before vs. After treatment (within the same site) plus Control vs. Impact (comparing sites with versus without active removal within the same point in time) (Bernhardt et al. 2007; Palmer et al. 2005). In this way, we address the shortcomings of most other studies in which results could be attributed to either time or space rather than the treatment itself.

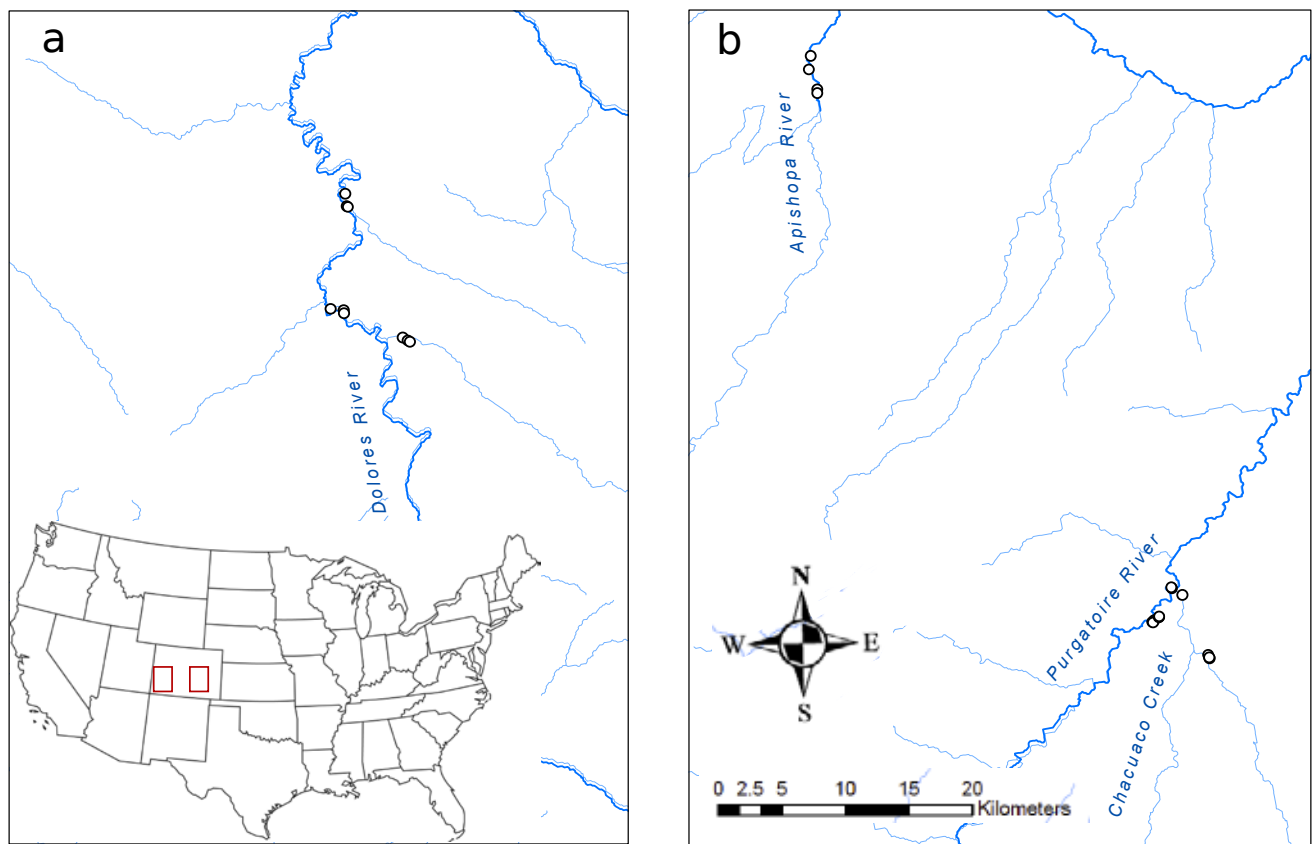


Figure 7—(a) Site locations for west slope of Colorado, a total of 9 sites. (b) Site locations for eastern plains of Colorado, a total of 16 sites (map by Annie Henry).

Within each site, a 20 m x 50 m plot was established, within which five stratified random transects were established that were sampled at 10 cm intervals using the line point intercept method. All sites were sampled in this way in spring of 2010 and in spring and summer of 2011 and 2012. Sites in the East were also sampled in the summer of 2010. The Modified Whittaker Plot method was used to record species richness at each sampling period in each site (Stohlgren et al. 1995).

All specimens that could not be positively identified in the field were collected and taken to Denver Botanic Gardens' Kathryn Kalmbach Herbarium (KHD) for identification. A total of 412 specimens were collected by the end of the last season of data collection on August 13, 2012. Pictures were taken of all specimens and stored in a digital form to serve as a backup for the original.

Specimens were identified with the assistance of local plant experts and the following books: Colorado Flora Eastern Slope (Weber et al. 1996a), Colorado Flora Western Slope (Weber et al. 1996b), Illustrated Key to the Grasses of Colorado (Wingate 1994), Shrubs and Trees of the Southwest Uplands (Elmore 1976), and Weeds of the West (Whitson and Burrill 2000). All specimens identified by dichotomous key were double-checked against stored reference specimens and confirmed by staff at KHD.

Digital plant databases were used to confirm current species information. Geographic distribution, plant morphology, spelling of scientific and common names, nativity status, growth habit, and plant functional group of each specimen were determined using United States Department of Agriculture Plant Database (USDA 2010),

Research Findings and Implications

Regional Differences

We identified a total of 145 different species in 45 plant families within the 25 sites from 2010–2012. The Eastern Slope sites had greater species richness than the Western Slope sites during this study period; 111 species in 41 families on the Eastern Slope compared to only 53 species in 25 families on the Western Slope. The most important families in the study area as indicated by highest numbers of species, in descending order are: (1) Poaceae—35; (2) Asteraceae—34; (3) Chenopodaceae—10; (4) Fabaceae—7; and (5) Brassicaceae—6.

The plant communities differed in composition between the two areas (fig. 8). Understory vegetation cover in western Colorado was dominated by shrubs (*Ericameria nauseosa*, *Chrysothamnus linifolius* Greene, *Artemisia tridentate*, *Atriplex canescens*, *Sarcobatus vermiculatus*, *Forestiera pubescens*, *Rhus trilobata*) while exotic herbaceous species (*Bromus tectorum*, *Bromus japonicas*, *Kochia scoparia*) dominated eastern

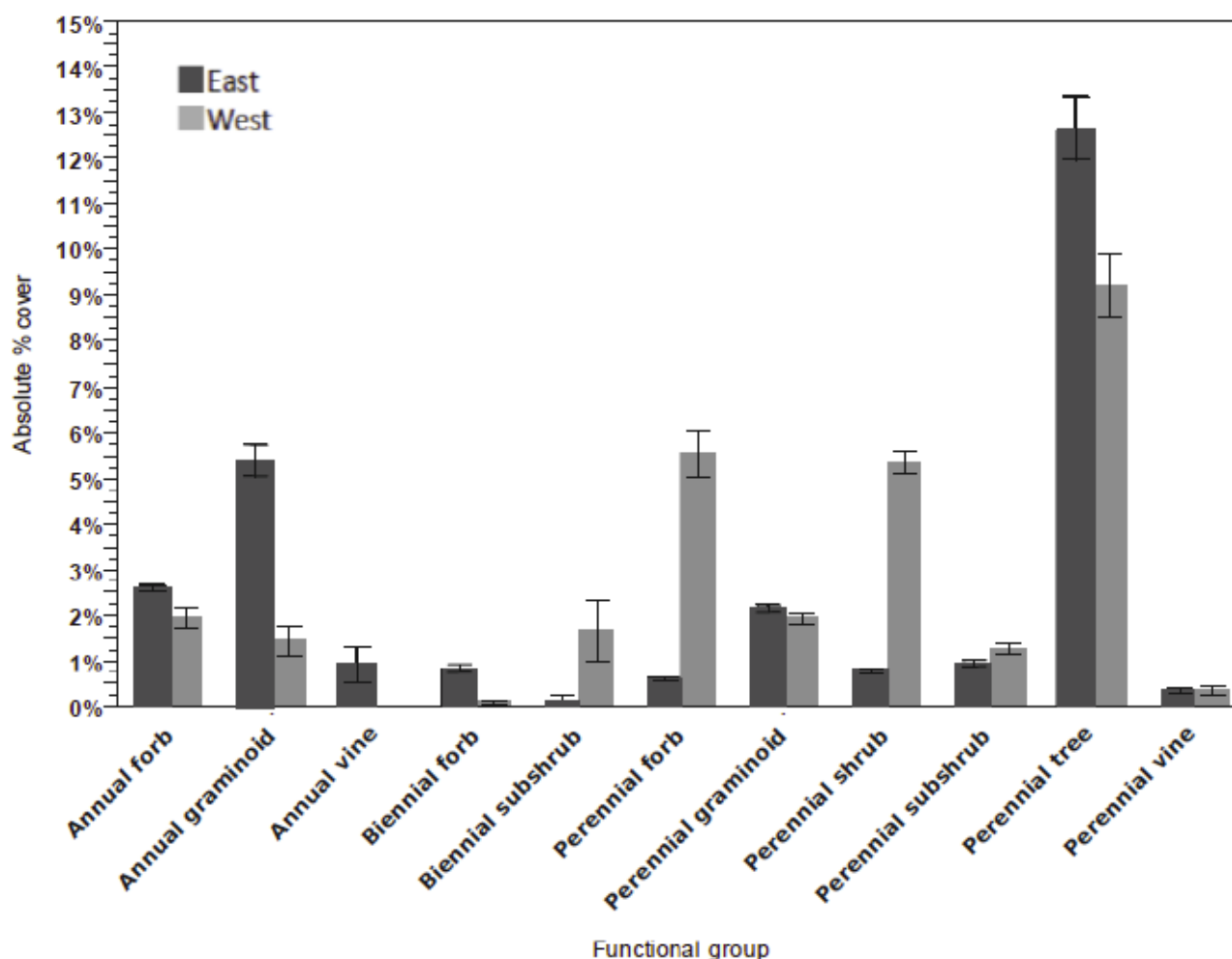


Figure 8—Mean (+/- 1 SE) cover for east and west slope sites by functional group. (Chi-square, Pearson; N = 699, DF = 8, $\chi^2 = 26.34$, $P < 0.0009$.)

Colorado. With regard to trees, both regions were dominated by *Tamarix* spp., *Populus* spp., and *Salix exigua*, but the eastern sites also contained *Juniperus scopulorum* and *Celtis reticulata*, while in the west the only other species was *Acer negundo*.

The Jaccard Index of similarity indicated that there was 36 percent similarity between eastern and western sites using all vegetation types, and 50 percent similarity in tree and shrub communities. East and west sites differed significantly in frequency of different functional groups (Chi-square, Pearson; $N = 699$, $DF = 8$, $X^2 = 26.3$, $P < 0.0009$). While there were distinct regional differences in the understory plant community by functional group, both regions had similar starting absolute cover of *Tamarix* at about 30 percent, and less than 10 cover percentage of native woody vegetation, as well as similar cover of understory natives and exotics.

The Response of *Tamarix* to Removal Efforts

There was a significant decrease in *Tamarix* cover after active removal with a more dramatic decrease on the Western Slope (repeated measures ANOVA, before/after*slope: $F = 4.13$, $DF = 1/83$, $P < 0.05$). Over time, we can see the decrease in the total absolute cover of *Tamarix* immediately following active removal in the spring 2011, but the trajectories differ between east and west sites after that (fig. 9). Western Slope *Tamarix* cover remained low, whereas *Tamarix* increased slightly in the eastern

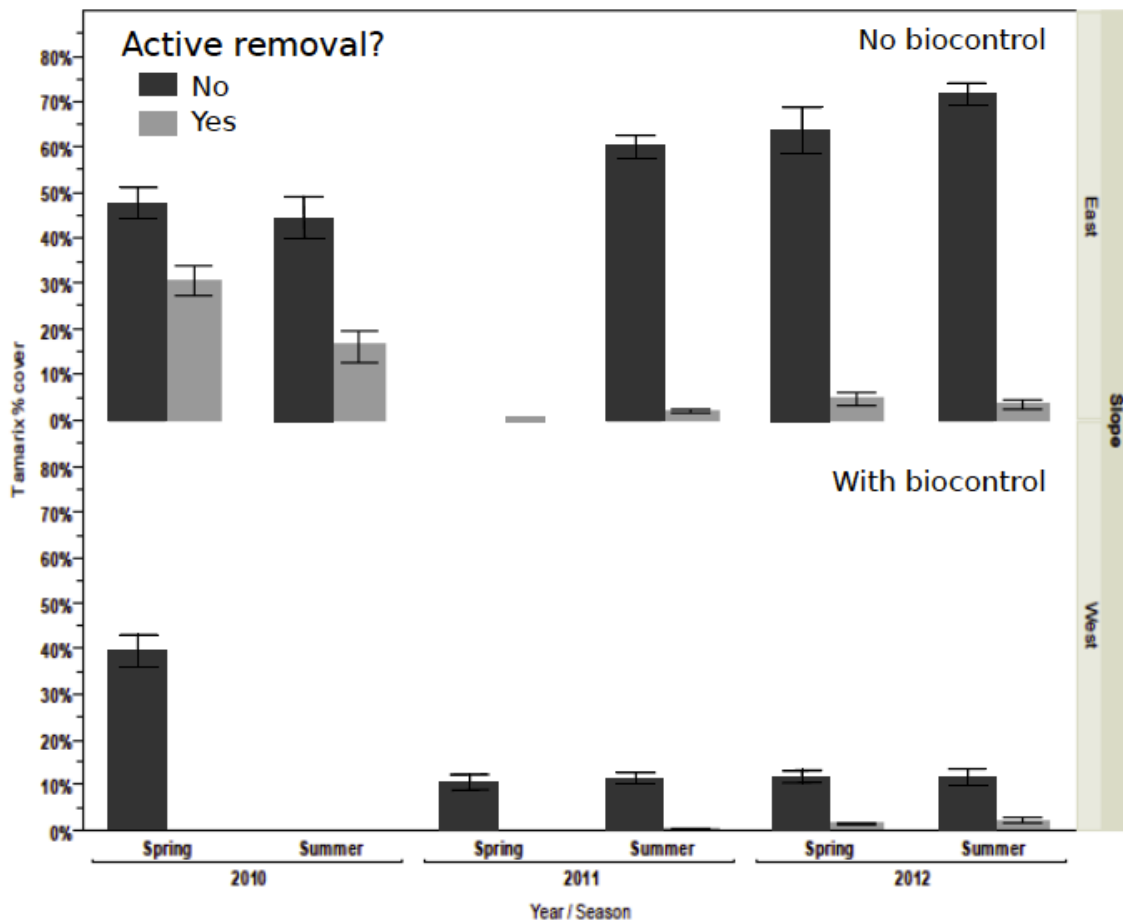


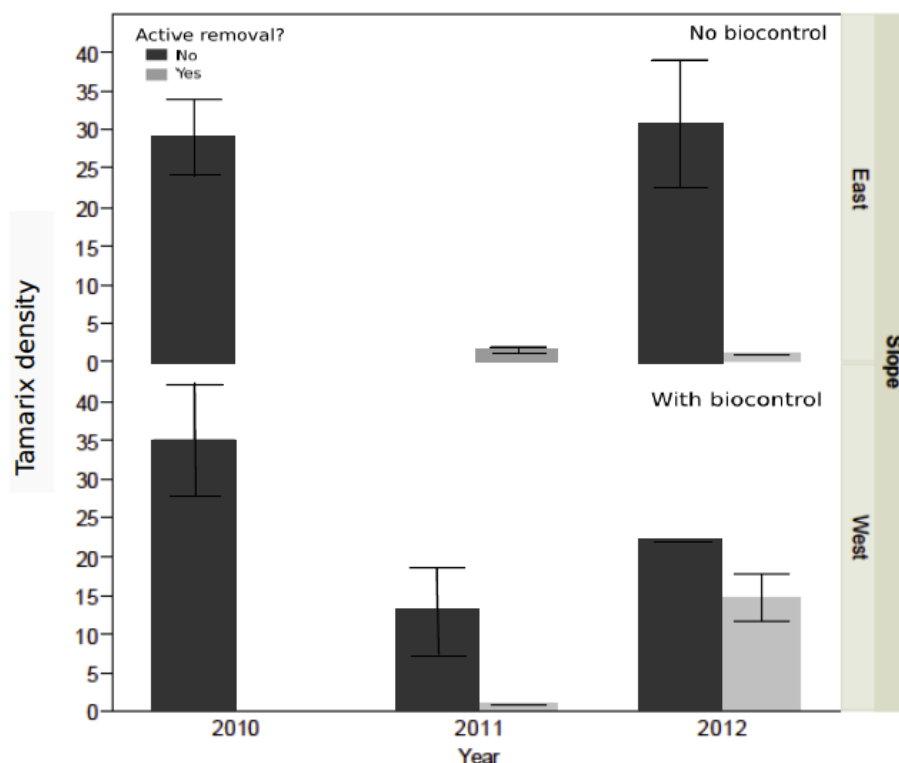
Figure 9—Mean (± 1 SE) cover of *Tamarix* spp. of active removal (light grey) or no active removal (dark grey). West slope sites have beetles present whereas east slope sites do not.

sites in non-active removal sites. When we compare active removal sites with non-active removal at the same point in time, we see even more dramatic evidence of the impact of removal. By 2012, however, this difference was much greater in the east sites compared to the west where *Diorhabda* spp. is known to occur.

There was also a decrease in total density (i.e., number of individuals) of *Tamarix* in the removal sites compared to non-active removal sites from the first to third year, also with a significant region effect, but the opposite pattern from what was observed with cover (fig. 10). Both regions saw a decrease in *Tamarix* individuals with active removal, but by 2012 there was an increase in the number of individuals in the west sites.

Differences between regions in successful control of *Tamarix* are multifaceted. Although *Tamarix* cover remained low in the Western Slope sites where biological control is ubiquitous, the *Tamarix* density was almost the same by the end of our 3 years of sampling. This suggests that even though there was new recruitment of *Tamarix* seedlings after active removal in the Western Slope, defoliation by the beetles was likely keeping overall *Tamarix* cover lower than in the Eastern Slope. The lower recruitment of seedlings in the east is likely because there was less overbank flooding than in the west, which is generally associated with regeneration by seed for *Tamarix*; especially high flows were observed by our team in several areas along the Dolores River in 2011, but not in the eastern sites. Water availability is the primary determinant of *Tamarix* establishment by seed (Sher and Marshall 2003). It is also possible that in the eastern sites new individuals were killed with follow-up treatments or by cattle grazing or trampling, given that all of the eastern sites were on private land and thus subject to more intense use and management.

Figure 10—Mean ($\pm$ 1 SE) number of *Tamarix* (i.e., density) per site of *Tamarix* spp. on removal vs. non removal sites, in the eastern plains and west slope of Colorado before active removal in 2010 and for 2 years following; spring season, cut stump method. ANOVA test ($F = 0.340$, $DF = 2$, P value < 0.004).



The Response of Plant Communities to *Tamarix* Removal Efforts

After *Tamarix* removal, we observed slightly greater cover of natives and a decrease in introduced species relative to before removal, but with different magnitudes between the two regions (fig. 11). Overall, we observed much more dramatic positive changes to the understory in the western sites.

Removal method mattered as well. Other than the positive reference sites, the highest relative percent cover of native species was found in the cut stump sites (fig. 12). Relative understory native cover decreased over time only in *Tamarix* control sites and helicopter spray treatment sites, while it dramatically increased in native control, cut stump, and track-hoe treatment.

Species richness was also strongly impacted by *Tamarix* removal, but a different picture is painted whether we consider before vs. after or control vs. impact. In the east, the number of species decreased after removal, likely due to negative effects of the disturbance of the *Tamarix* removal methods (fig. 13). This is in contrast with the increase in richness after removal in the west. However, when we compare species richness at the same point in time, east and west active removal sites have very similar species richness, but this is low relative to positive reference sites in the east and high for those in the west (fig. 14). In the west, but not in the east, *Tamarix*-dominated stands (negative reference sites) were associated with much lower species diversity than positive

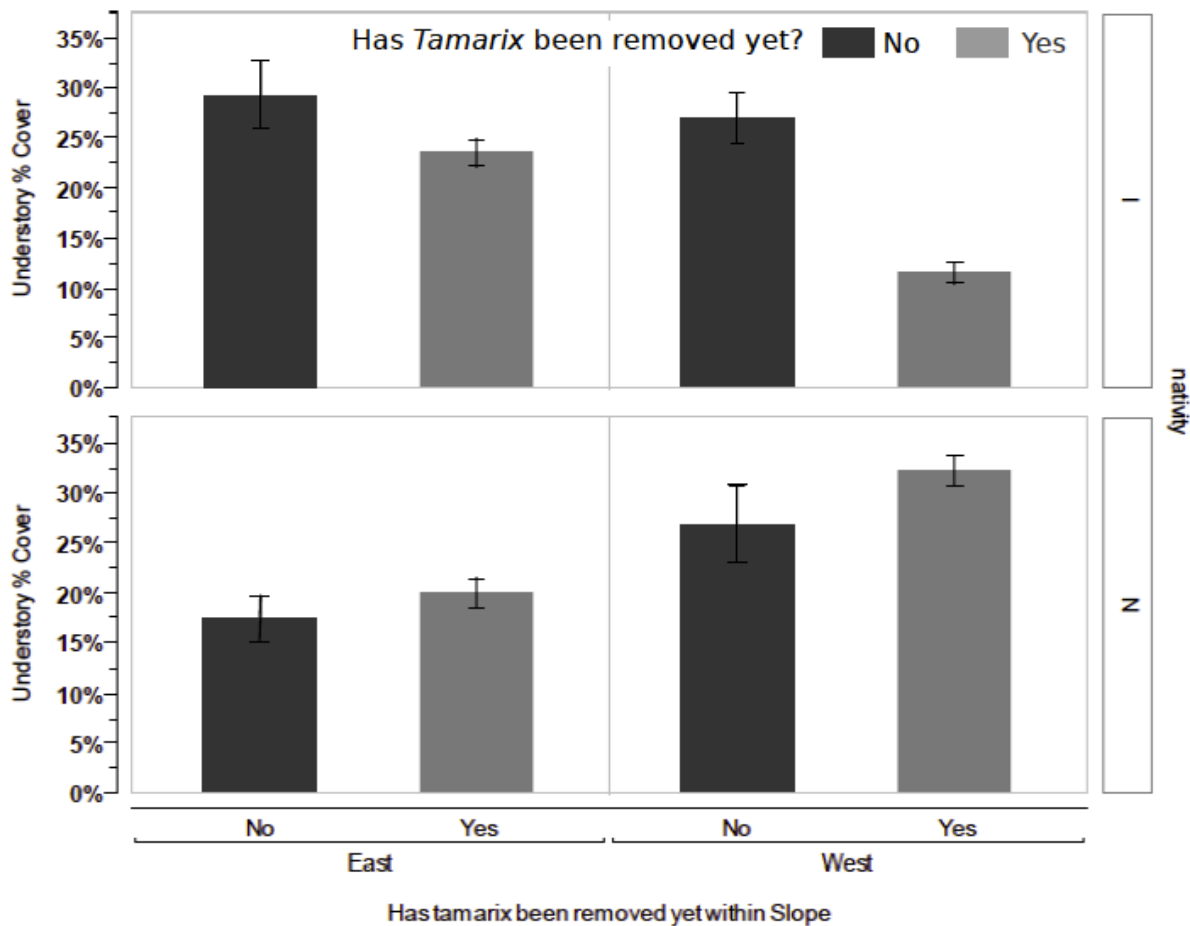


Figure 11—Mean (± 1 SE) cover of understory vegetation by nativity (native “N” and introduced “I”) for east and west sites by active removal (“has *Tamarix* been removed yet”).

reference sites. The difference between regions in the response to *Tamarix* removal is likely due to the difference in initial plant communities; most of understory vegetation were shrubs in the Western Slope whereas the Eastern Slope was dominated by herbaceous species. Herbaceous species are much more likely to be killed by the heavy machinery, use of herbicides, and trampling associated with *Tamarix* removal relative to shrubs. In particular, helicopter-applied herbicides in some of the eastern sites likely negatively affected the plant community.

Given that very different patterns appeared in the sites where helicopter spray was applied, we used another quantitative measurement, the importance value (IV), to investigate the community response in these sites with more detail. The importance value is a combination of the relative cover, density, and frequency, and as such can better explain the change of plant composition than just one parameter (Mueller-Dombois and Ellenberg 1974)³. Results of this analysis showed that the increase in understory exotics in the helicopter spray sites was primarily due to the exotic forb *Bassia scoparia* (a.k.a. *Kochia scoparia*, burningbush). This exotic forb, which ranked third in IV before removal, took first place after removal, shifting the community from native dominated (*Elymus canadensis*) to exotic because of the increase in the relative cover of *B. scoparia*. This is an exotic species that is considered a nuisance but is not a State-listed weed. It is also known to have resistance to commonly used herbicides. Thus, the primary problem we observed with the aerial application of herbicide was its direct negative effect on the native species, which decreased significantly when this approach was used.

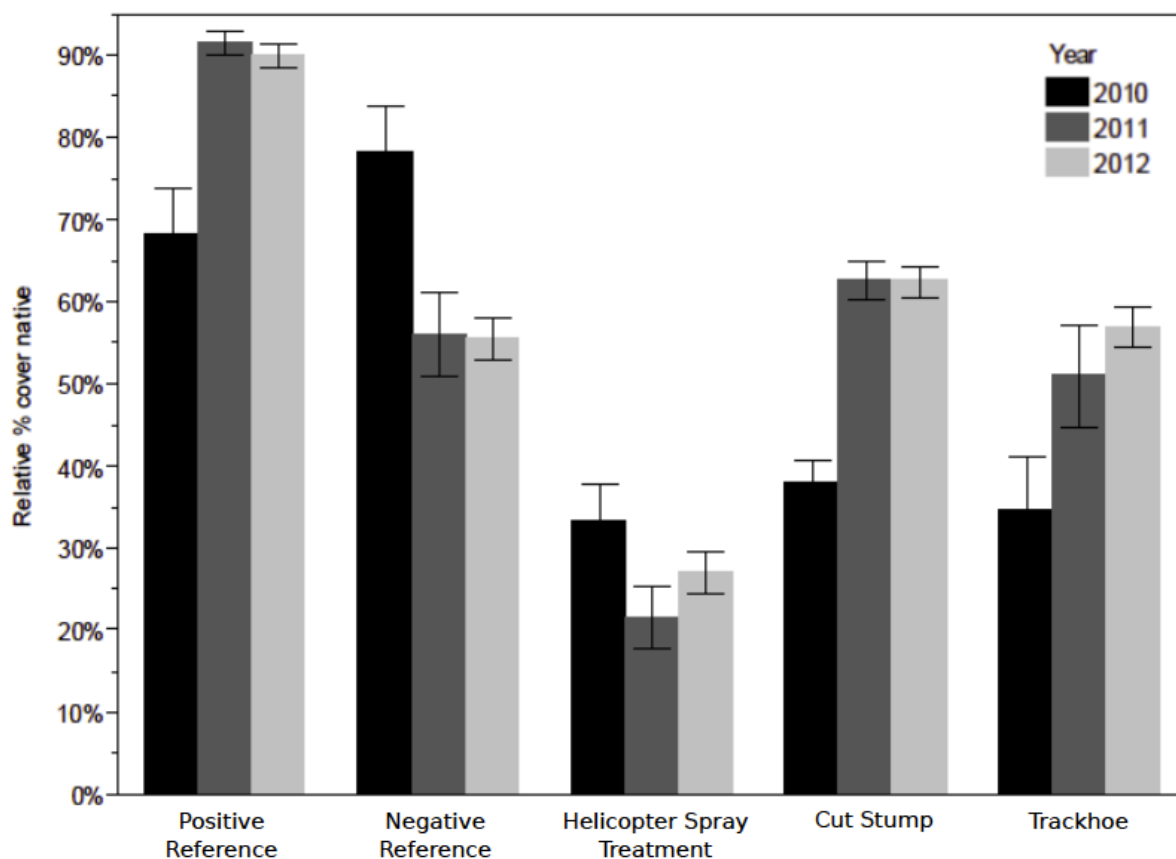


Figure 12—Mean (+/- 1SE) relative native understory cover over time within different removal methods. ($F = 55.9$, $DF = 5,599$, $P < 0.0001$).

Figure 13—Average richness over time from before treatment to 2 years after treatment, by slope.

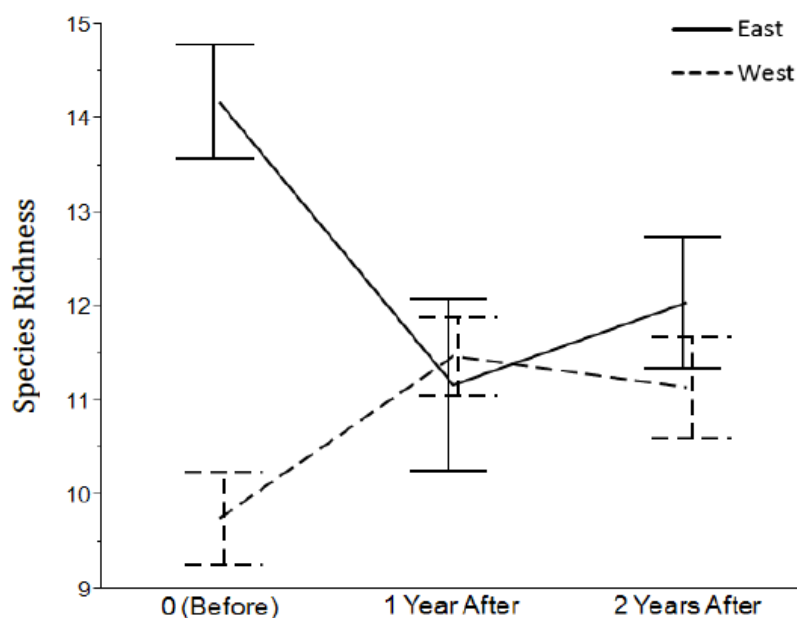
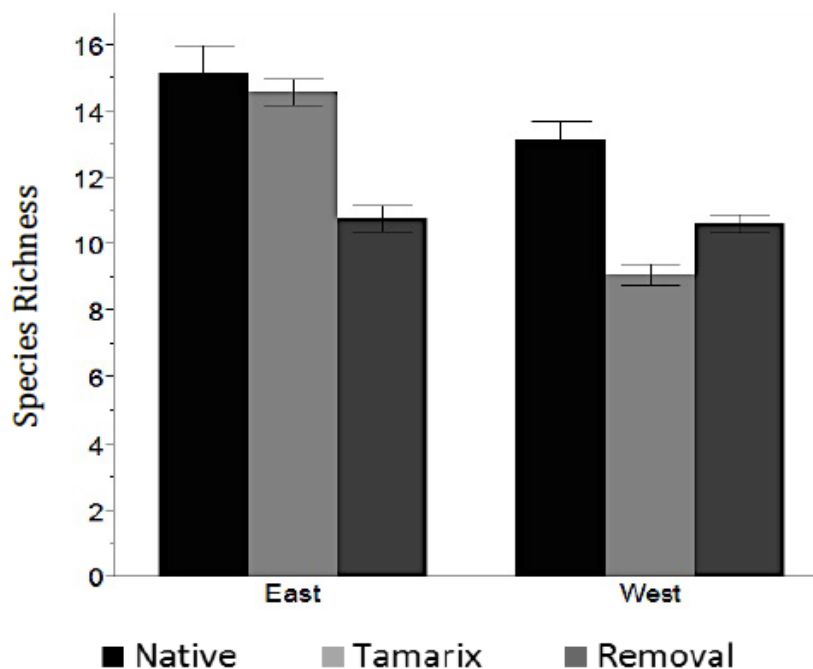


Figure 14—Mean ($\pm$ 1 SE) number of species per transect for positive reference sites (“native”), negative reference sites (“*Tamarix*”), and active removal sites in both east and west slope sites. (ANOVA Treatment*Slope: $F = 19.7$, $DF = 2,380$, $P < 0.0001$.)



Conclusion and Implications

In summary, active removal of *Tamarix* was highly successful and had significant effects on the understory that differed by region. In the western sites where the biocontrol beetle *Diorhabda* spp. was present, *Tamarix* cover decreased over time in both *Tamarix* active removal sites and non-active removal sites, and the former was associated with decreases in other exotics while increasing native species. The eastern sites where there was no biological control and no active removal had very high cover of *Tamarix*, but differences between these sites were not as great for the understory, due to the poor results seen in those sites with aerial application of herbicide. Generally,

Tamarix removal led to an increase in native understory vegetation except when herbicide was applied by helicopter. Spraying herbicide by helicopter was associated with the increased importance of *Bassia scoparia*, an understory exotic. These results are consistent with other studies in which secondary invasions were less notable following biocontrol than other treatment methods (González et al. 2017a; Sher et al., in review), particularly *B. scoparia* (González et al. 2017b).

It is also important to note that we observed positive reference sites to increase in native cover over time, whereas sites with no *Tamarix* removal decreased in native cover, and when the biocontrol was absent, increased in *Tamarix* cover. This suggests a risk of site depredation if nothing is done to reduce this exotic tree.

In conclusion, our results suggest that removal of *Tamarix* by commonly used methods is either neutral or positive for the native plant community. Although 2 years post removal is too short a period to reach any broad conclusions about impact, our results are consistent with those over longer time periods and greater geographic range that suggest that native cover can be promoted through *Tamarix* removal (González et al. 2017a). Although this should not be confused with a return to the mesic riparian forests associated with pre-invasion and pre-damming, it does bode well for improvement of these communities.

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