1-1-2013

Vegetation Response After Invasive Tamarix Spp. Removal in the Riparian Zone and Semi-arid Rangeland Ecosystems

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VEGETATION RESPONSE AFTER INVASIVE TAMARIX SPP. REMOVAL IN
THE RIPARIAN ZONE AND SEMI-ARID RANGELAND ECOSYSTEMS

A Dissertation

Presented to

The Faculty of Natural Sciences and Mathematics
University of Denver

In Partial Fulfillment
of the Requirements for the Degree
Doctor of Philosophy

By
Hisham Nagi El Waer

August 2013
Advisor: Anna Sher Simon
ABSTRACT

Removal of Tamarix spp. (a.k.a. tamarisk, saltcedar, Athel) invasion is often involved in restoration of Western, riparian habitat; however monitoring of vegetation after removal is often neglected and thus opportunity for adaptive management lost. To address this need, I have conducted three and half years of monitoring vegetation response after invasive Tamarix removal in twenty-five sites on the East and Western Colorado, starting fall 2009. I am also comparing six different methodologies: Point intercept, line transect, nearest neighbor, meter-square quadrats, nested Whittaker plots, and densitometer with the objective of developing monitoring protocols that can be used by scientists and land managers alike. This project is in collaboration with Branson Trinchera Conservation District (BTCD), The Nature Conservancy (TNC), the Colorado Water Conservation Board (CWCB), US Fish and Wildlife Service (USFWS), and the Bureau of Land Management (BLM).
My intent is that this monitoring project will help to answer the controversial questions about the ecological impact of *Tamarix* removal, including testing the prediction that removal of *Tamarix* will increase native cover, and that an increase in the cover and diversity of desirable species will also prevent secondary invasion of introduced and noxious species. Overall, the project will help to better understand the ecological impact of the invasive species on the invaded native habitat and whether or not restoration efforts are valuable.
ACKNOWLEDGEMENTS

First, I would like to express my deepest appreciation and gratitude to my supervisor Dr. Anna Sher, whose efforts and knowledge constituted the basis for this dissertation. Many thanks to our funders and supporters: BTCD, TNC, CWCB, USFWS, NRCS, DBG, DU, CSFS, and BLM. Special thanks are directed to all of the land owners and land management agencies, especially the Doherty, Wooten, and Larson families, for permitting their properties to be the locations for this project. I am grateful to the DU Ecology Group and to my committee members: Dr. Martin Quigley, Dr. Shannon Murphy, and Dr. Rebecca Powell who supported and encouraged me with their right suggestions and recommendations. Many thanks to all who contributed to data collection and data entry for this project especially, Mr. David Stahl, Mr. Rob Anderson, Katie Merewether, Claire Slattery, Ben Peters, Stuart Coles, Ryan Whitney, Francesca Aguirre, Eliot Jackson, and Court Ballinger. Special thanks are directed to all of the DU Department staff especially Randi Flageolle and Rheagan Hollenbeck, and to the DU Writing Center, Graduate Office, and Penrose Library for providing me with all possible guidance. Finally, I am so appreciative for the support and encouragement that my father, mother, brothers, sisters, and my children, especially my son Ibrhem, during my study. Finally, I am forever grateful to my wife, Mahasen, who, single handedly, provided me with the time, support, and energy for this study.
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CHAPTER 1

INVASIVE SPECIES REMOVAL AND MONITORING

General Concept of the Invasive Species:

Worldwide, invasive species threaten natural ecosystems and biodiversity (Vitousek et al. 1996, Mack et al. 2000, Firn et al. 2010, Marchante et al. 2011). Ecosystem degradation associated with the proliferation of invasive species has led to an increase in the efforts and cooperation of researchers, landowners, and government agencies both globally and locally to restore invaded areas. Understanding the response of invasive species to changing environmental conditions can greatly reduce their negative impact on ecosystems and thus mitigate the damage and economic losses. Increasing our understanding in the relationships between plant community responses to invasive species removal can enhance restoration efforts in the future.

There are several theories about why invasive species proliferate where they do. The traditional theory of invasive species is that communities with high diversity will be more resistant to invaders under the assumption that habitat resources are limited and
competition is the key factor of species diversity (Huston 1979, Levine and D'Antonio 1999, Huston 2004). However, more recently researchers found that a habitat with high native plant diversity can also include numerous invasive plant species (Levine and D'Antonio 1999). Biotic factors such as herbivore and enemy free space, commonly known as the enemy release hypothesis (ERH), and abiotic factors such as floods play extreme roles in invaded ecosystems (Keane and Crawley 2002). For instance, invasive *Tamarix spp* (a.k.a. tamarisk, saltcedar, Athel) proliferates in enemy-free space environments unlike native species in the same habitat (Di Tomaso 1998, Keane and Crawley 2002, Colautti et al. 2004, Liu and Stiling 2006, Hultine et al. 2010); but may be invasive primarily because of abiotic factors such as lack of overbank flooding (Taylor and McDaniel 1998, Sher and Hyatt 1999, Stromberg et al. 2007b).

There is an ongoing debate amongst scientists as to whether the invasive species is the cause (driver) or the consequence (passenger) of ecosystem degradation (Didham et al. 2005, MacDougall and Turkington 2005, Bauer 2012, Johnson 2013). Recently a new model described invasive plants as ‘back-seat drivers’, in which the invasive species will react to disturbance factors and then cause the decline in native species and contribute to ecosystem degradation (Bauer 2012). Understanding whether an invasive species is the driver or passenger of change in an ecosystem has important implications for its management. *Tamarix* is one invasive species that may fit both models (Johnson 2013).

The invasive *Tamarix* tends to dominate dammed and de-watered sections of rivers in the Western U.S. Under natural river flow conditions, native *Populus spp.* are highly competitive over invasive *Tamarix* (Sher et al. 2000, Sher et al. 2002, Huston 2004).
The ecosystem alteration via river flow changes the natural rules for competition giving invasive Tamarix a competitive advantage that depresses the native species. In this way, *Tamarix* is not the cause (driver) of the ecosystem change but rather the result (passenger) of the ecosystem alteration (Johnson 2013). However, *Tamarix* can also act as the driver of change by modifying the ecosystem in its favor. If *Tamarix* colonizes an empty space where native species are absent, then it can act as a driver by increasing fire risk, lowering water tables, changing the morphology of the streambank, and increasing the soil salinity, among other effects (Didham et al. 2005, MacDougall and Turkington 2005, Sher 2006, Johnson 2013). It is essential from a restoration perspective to understand whether the invasive is the passenger or the driver in the specific ecosystem in order to determine the restoration approach (Bauer 2012). For example, removal of the invader is key if it acts as the driver of the ecosystem. For passenger invaders, the underlying causes such as fires, overgrazing, and flooding need to be addressed.

The spread of *Tamarix* in the southwest United States has been attributed to several causes related to its growth habit, reproduction, water usage, response to fire, capability to tolerate highly saline conditions, and relocation of salt from deep in the soil profile to the soil surface (Glenn et al. 2012, Ohrtman et al. 2012, Cleverly 2013, Drus 2013, Zavaleta 2013). Once *Tamarix* has established and colonized, it begins to modify the habitat by increasing the soil salinity in its immediate surroundings; this allows *Tamarix* to grow in places where willow and cottonwood trees cannot (Sher et al. 2000, Sher et al. 2002, McDaniel et al. 2004). *Tamarix* has increased soil salinity in riparian zones
throughout the western United States (Di Tomaso 1998). Soil salinization however, can also happen as a result of soil capillary rise, which serves to pull up salt water from deep aquifers into shallow soil horizons. This is important because removal of *Tamarix* may actually increase soil salinity if there is no replacement vegetation to shade the soil and thus reduce capillary rise from surface evaporation (Glenn et al. 2012, Ohrman et al. 2012, Ohrman and Lair 2013). Elevated salinity in the surface of the soil can prevent seed germination and the growth of native species (Busch and Smith 1993, 1995, Ohrman and Lair 2013).

*Tamarix* as an invasive impacts also the rangeland ecosystems and wildlife refuges by displacing forage grasses and competing with desirable plants; *Tamarix* can also access and use aquifers and groundwater that would otherwise be available to grow forage or crop species (Taylor and McDaniel 1998, DeLoach et al. 2000, Glenn et al. 2012, Bateman et al. 2013, Cleverly 2013). The Bonneville Unit of the Central Utah Water Project estimated cost of water loss along the Colorado river by *Tamarix* to be about $27 million annually (McDaniel et al. 2004). On the other hand, the total cost for *Tamarix* eradication and re-vegetation in the U.S is estimated to be about $11.2 billion (De Waal 1994). Thus, regardless of the management decisions, *Tamarix* establishment and proliferation as an invasive species, leads to a major cost to public and private institutions.
Tamarix as Invader in The U.S:

During the 1800s, eight species of Tamarix were first introduced to the United States from Europe, Asia and North Africa mainly to decrease erosion and slow down water flow in the riparian zone; 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, Di Tomaso 1998, Gaskin and Schaal 2002, McDaniel et al. 2004, Chew 2013, Sher 2013) and it now occupies about 800,000 ha (SWTRG 2010). Tamarix is listed as a noxious weed in several states including Colorado, Montana, Nevada, New Mexico, Washington, Oregon, North Dakota, and Wyoming, and almost there are no riparian systems in those states where Tamarix is absent (McDaniel et al. 2004). Along rivers in arid zones of western North America, Tamarix trees are replacing native riparian plants. The most widely-naturalized species are Tamarix ramosissima, Tamarix chinensis and their hybrid (Gaskin and Schaal 2002, Friedman et al. 2005).

Tamarix is classified as deciduous with either tree or shrub growth habit. It is also a paradoxical plant from the selection theory perspective as it uses both r and k strategies with both a large number of offspring and high longevity (Sher 2013). Tamarix live for more than 100 years, and one large tree produces about 500,000 seeds per year (McDaniel et al. 2004, Stromberg et al. 2007b). This life-history strategy helps both explain it is impact and justify efforts to control it.
Vegetation Response to *Tamarix* Removal:

Invasive plant species threaten the function of natural ecosystems and removal is extremely expensive. Thus the likelihood of success and outcome of removal efforts should be evaluated economically and environmentally before applying removal on a broad-scale. Experimental or small projects can help to design, evaluate and apply the removal and restoration approach on broad-scale (Flory and Clay 2009, Marchante et al. 2011, Bay 2013, Shafroth et al. 2013).

While invasive plant species removal has been a priority in restoration of river systems, relatively little is understood about the ecological impact of the removal of invasive species from the ecosystem (Shafroth et al. 2005, Cuevas and Zalba 2010). Recently, invasive species control has become an important portion of land managers’ responsibilities, along with those of several United States government institutions (Harms and Hiebert 2006, Dennison et al. 2009). Concern over the spread of exotic riparian plants in the western United States has led to congressional proposals to speed up removal efforts, but debate over these proposals is weakened by limited information of exotic species distribution and abundance (Gaskin and Schaal 2002).

Researchers and scientists are still unsure as to whether or not removal of woody invaders such as *Tamarix* will result in a positive vegetation response (Shafroth et al. 2005, Stromberg et al. 2009, Bay 2013). Although absolute restoration of the ecosystem after invasive species control is impractical, it should be possible in long-term projects to restore ecosystem function to that of the original habitat (Marchante et al. 2011, Shafroth
et al. 2013). After removal of invasive *Acacia longifolia* (Sydney golden wattle) in long- and recently-invaded ecosystems in Sao Jacinto Dunes Nature Reserve, species richness was higher in plots where the litter was also removed compared to control sites where *Acacia* was still present (Marchante et al. 2011). In this case, removal of the invasive species’ biomass from the system encouraged rapid recovery of native species. Most species that appeared in the treated plots were natives accounting for more than 70% of absolute cover. The removal of *Acacia* improved ecosystem health both in terms of richness and native cover.

However, results of research concerning invasive species control vary widely in the response of invaded communities to the control of target species (Denslow and D’Antonio 2005, Cuevas and Zalba 2010, Gardener et al. 2011, Douglass et al. 2013). Cuevas and Zalba (2010) found a gradual increase of native species after the removal of invasive Aleppo pine. There was a temporary increase in cover of other exotic species after invasive Aleppo pine removal, however, after 4 years, exotic species cover was down to levels equivalent to non-invaded sites. Removal of woody invasive hill raspberry on Santiago Island resulted in a significant decrease in both density and seed bank of invasive species (Renteria et al. 2012). However, after five years, plant community and vegetation structure in removal areas was dominated by herbaceous species, unlike the woody composition of native control sites (Renteria et al. 2012). Gaddis (2008) found that plant communities were dominated by exotic species after Russian olive removal, but it is unclear if the greater cover of exotic species was a response of invasive removal since no control or pre removal data were established. In Australia, after 20 years of
Martynia annua L. eradication efforts, researchers found that efforts had failed due to re-invasion because of that species long distance dispersal strategy and recurrent seed bank; it was therefore recommended that more widespread eradication efforts are needed to prevent seed production and dispersal (Gardener et al. 2010b). In a review of 30 invasive removal project including 23 invasive species, Gardener et al. (2010a) found that only four projects led to positive impact on the native plant communities. Most projects were unsuccessful due to insecure or non-continuous funds or from denial of landowners to land access. For this reason, it is essential to plan up front and have clear policies, goals, and secure funding for any restoration project to reach the best outcome and avoid failure (Shafroth et al. 2013). Incomplete restoration projects can even further harm the ecosystem.

Research on plant community response to Tamarix removal is similarly 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 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, a similar study but with active re-vegetation, Bay and Sher (2008) found increases in native cover after tamarisk removal under particular conditions, including that the relative cover of planted species was greater in the sites when the removal period was greater than 8 years. Recovery of native species was associated with several abiotic site characteristics and
correlated to *Tamarix* cover, with a greater response of native species in less dense tamarisk.

Many studies suggest that *Tamarix* removal is beneficial to the ecosystem, however, Harms and Hiebert (2006) found that active revegetation is needed after removal to increase the native cover. They also found that different ecosystems differ in their response to invasive removal. However, this and other past studies involved plant surveys at a single point in time, and combined restoration sites where *Tamarix* had been removed in different years. Thus, these results must be interpreted with caution because 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 coincide with single time monitoring) or variation in removal period can confound analyses and guide to misinterpretation of results. Multiple years must 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. Furthermore, the monitoring approaches have been extremely rapid and imprecise, with a large capacity for error.

Another gap in the research is that different *Tamarix* removal techniques have been used in different studies, such as cut and spray, aerial application of herbicide, controlled burns, and biological controls, however, only a few removal studies directly compared the vegetation response to these various *Tamarix* removal techniques (Sher et al. 2002, Harms and Hiebert 2006, Bay and Sher 2008, Sher et al. 2008, Hultine et al. 2010). Harms and Hiebert (2006) compared two different removal techniques, cut stump
and burning, followed by chemical spray and found no significant difference in percent
cover, richness, or species diversity between the two removal techniques, however no
published study compares results of removal activities at multiple sites the same number
of seasons after application.

To address the question of plant community response to *Tamarix* removal in a
study that tracks multiple sites over the same time period, I monitored vegetation twice a
year for three and a half years in the sites where *Tamarix* was concomitantly removed.
This approach has a better capacity than past studies to compare and detect changes in
plant community composition after *Tamarix* removal. Therefore, it will be able to draw
more accurate conclusions about vegetation response over time, including the status of
exotic and native cover, species richness, and density of *Tamarix* after removal.

Long-term monitoring of riparian ecosystems is needed for both management and
research to address the long-term impact of *Tamarix* removal on the ecosystem (Scott and
Reynolds 2007). Removal of invasive woody vegetation can increase indigenous species
diversity and richness; however, other noxious species can proliferate after the removal
of target woody invasive species (Webb et al. 2001, Hartman and McCarthy 2004, Ogden
and Rejmanek 2005). Colonization by noxious species and secondary invasion after
*Tamarix* removal is a substantial subject of concern (Shafroth and Briggs 2008, Sher et
al. 2008). However, there is a lack of research in this area. The goal of this study is to
investigate these knowledge gaps by monitoring vegetation response after *Tamarix*
removal in a long-term study.
Sampling Methodologies and Invasive Removal Techniques:

Different scientific methodologies can be used for quantitative measurement of organisms and communities in general. Various field methods have been used to monitor plant species including vegetation cover, frequency, and density. Line-point intercept, line transect, quadrat, and nested quadrat method are the most common methods that have been used for vegetation measurements or to detect vegetation response after invasive removal (Heady et al. 1959, Mueller-Dombois and Ellenberg 1974, Floyd and Anderson 1987, Brady et al. 1995, Sher et al. 2000, Anderson et al. 2005, Scott and Reynolds 2007, Sher et al. 2008). Measurements from line point intercept and line interception were very comparable; point intercept accomplished about the same level of precision as line interception in one-third less sampling time (Floyd and Anderson 1987). They also found that a point intercept method in the native sagebrush ecosystem is the most efficient and capable method if estimates for most of the species and richness in a community are needed. Line transect and quadrat method are most commonly used to measure riparian vegetation before or after Tamarix removal (Elzinga et al. 1998, Anderson et al. 2005). However, to the best of my knowledge, there are no published studies comparing the six different monitoring methodologies for accuracy and efficiency in the particular riparian ecosystem that I studied.

Limited research currently exists to guide land managers seeking to monitor plant communities, and much of it is contradictory or not done using real plant populations.
Comparisons between line transect and quadrat method in artificial populations (i.e. 2-dimensional simulations of plant communities) suggest that line intercept method is more precise and requires less time than quadrat method (Bauer 1936, 1943, Heady et al. 1959). The size and shape of sampling units can be determined by considering several factors which describe the study area, environment, density, frequency, cover and diversity or plant growth characteristics. However, given that most plant species grow in clumps, the spatial distribution of the species being sampled is the most important factor, and it has been suggested that oblong plots should capture more species (Mueller-Dombois and Ellenberg 1974, Elzinga et al. 1998). Elzinga et al. (1998) found that changing the plot shape from square to rectangular, while keeping the total area of each plot equal, results in more normal distribution of data and a decrease in the population standard deviation. Scott and Reynolds (2007) found that using larger quadrats captured higher species frequency, and species diversity than using smaller quadrats, given equal sample size. However, two different studies found no significant difference in species diversity and richness between rectangular and square plot shape with the same area (Laurance et al. 1998, Keeley and Fotheringham 2005). My research will compare methods, plot sizes and shapes using real populations to determine which of these recommendations are relevant to riparian systems.

Ecological monitoring requires long-term collection of data, and is relatively expensive. However, providing these baseline data and the use of subsequent adaptive management techniques can greatly increase the success of restoration projects (Spellerberg 2005), thus it is critical that efficient methods be determined for the specific
restoration situation at hand. To address this need for riparian systems, I monitored vegetation response after invasive *Tamarix* removal in several sites on the East Plains and West Mountain slopes of Colorado using five different monitoring methodologies and four removal techniques for three and half years beginning in fall 2009. This project, in collaboration with and funded by the Branson Trinchera Conservation District (BTCD), The Nature Conservancy (TNC), the Colorado Water Conservation Board (CWCB), US Fish and Wildlife Service (USFWS), Natural Resources Conservation Service (NRCS), Denver Botanic Garden (DBG), and the Bureau of Land Management (BLM), aims to establish baseline data for long-term monitoring, and develop the best practices to make recommendations for monitoring by land managers.
CHAPTER 2

Similarity of plant communities in eastern and western Colorado

INTRODUCTION

Understanding the similarity of ecosystems, plant communities, and species distributions are essential for making conservation and restoration decisions (Mueller-Dombois and Ellenberg 1974, Abido and ALkory 1989). In any conservation and restoration project, understanding species diversity is one of the most important factors of the success (Margules et al. 2002, Tobler et al. 2007, Loiselle et al. 2008). Thus, collecting specimens to be positively identified and stored appropriately for future reference or further investigation is essential. Traditional specimen collection has been used since the Italian Renaissance and is still one of the most important sources to study plant distribution, genetics, medicinal uses and phenology (Liston et al. 1990, Ladio et al. 2007). The specimens that were collected in my study are important for comparisons between sites and future study. This chapter will comprise the species lists for eastern and
western Colorado and summarize the most speciose plant families in the riparian zones. It also compares the plant community composition in eastern and western Colorado and each region’s respective percent cover by nativity and functional group. Better understanding of these similarities and differences will help prioritize and direct restoration efforts.

In general, the two regions have similar but distinct abiotic conditions. The average elevation in the western study area is 5500 ft, while it is only 4500 ft in the east. Temperatures are similar for the two regions, although somewhat hotter in the east; when maximum temperatures during the study period from 2010 to 2012 are averaged over a month period, the highest temperature in the west was 91.4°F in July 2010. By comparison the highest temperature recorded in the east was 96.7°F in July 2011 (PRISM 2012). The annual average precipitation is 10 to 15 inches in both east and west sites (PRISM 1990).

In terms of vegetation, generally, most of the understory vegetation cover in western Colorado are native shrubs while the most common understory cover in the eastern sites are exotic herbaceous species. In the overstory, *Tamarix spp.*, *Populus spp.* and *Salix exigua* are the most common species in both regions. However, no formal comparisons of the species in these regions have yet been conducted and so it is unclear how similar or different they might actually be. Even though the method of *Tamarix* removal can be similar, effective active restoration approaches will require an understanding of the plant community composition and species distribution that were and can exist in each particular ecosystem. In other words, the same species may or may not
be appropriate to use for revegetation depending on the ecosystem factors. Also, understanding the regional differences between the compositions of nativity and functional groups will help distribute the restoration efforts more effectively. For example, my study found that eastern sites have more exotic understory cover and thus require more extensive revegetation efforts.

To evaluate the similarity of plant communities in the study area (Figure 1A and Figure 1B), both eastern and western sites were monitored to survey and compute the plant species richness to answer the following questions: 1) What are the species that are present in eastern and western Colorado, 2) what is the level of similarity between the eastern and western sites, and 3) does vegetation cover of natives vs. exotics differ between these two areas?

METHODS AND SITE LOCATIONS

A total of 25 sites were monitored for vegetation: nine sites in three reaches located in western Colorado (Figure 1A) and sixteen sites in five reaches located in the eastern Colorado (Figure 1B). 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 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 nearly all on private land, selected by land
managers for our group to survey, either because they were candidates for tamarisk removal, or they represented an uninvaded ecosystem. Although they were not randomly selected, they represent a range of representative riparian ecosystem conditions, from degraded to fairly pristine.

The following methods were used to intensively survey the plant communities in each site: nested Whittaker plots, modified Whittaker, (Stohlgren et al. 1995), one-meter square quadrats, and line point intercept (See chapter 3 for more details about the methods). All methods were used to sample the sites in both spring of 2010, and in spring and summer in 2011. Sites in the east were also sampled in the summer of 2010, and both east and west were sampled in spring and summer of 2012 using point intercept method. Because overlapping methods were used within the same 20m x 50m area, and because each site was re-sampled between 2 and 6 times over up to three years, there can be a high degree of confidence that most, if not all, species present at each site were recorded.

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. At least two representative specimens for each unidentifiable species were collected with intact leaves, flowers, fruits, seeds, and roots whenever possible. The information written down for each specimen included its presumed name, date of collection, GPS coordinates, small description of the specimen morphology and environment, place and site name, and collector name(s) in each specimen file. A total of 412 specimens were collected by the end of the last season of data collection on August 13th, 2012. All specimens collected in the field were pressed instantly and kept
in a warm and dry place (cabin, cars, or tents). Specimens were checked after two to three days, and the cardboard and newspaper covers were replaced as needed. All specimens were taken out of the pressers after fully drying (four to seven days). 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 double-checked using the following: United States Department of Agriculture Plant Database (USDA 2010), the Colorado State University (CSU Herbarium 2001), and Southwest Environmental Information Network (SWEIN 2012).

To evaluate the similarity of plant communities between the east and west study area, three approaches were used. To determine similarity of the plant species, the Jaccard Index of similarity was used:

\[
J_{\text{JACCARD \ index \ of \ similarity}} = \frac{C}{A + B - C} \times 100 = \% \]
The Jaccard index of similarity has been widely used to address the similarity of different ecosystems (Mueller-Dombois and Ellenberg 1974, Real and Vargas 1996, El waer and Csányi 2006). To determine the difference between east and west sites in presence of functional groups, chi-square was used. For this analysis, functional group presence was included from all sampling periods where there were data for both east and west sites (spring 2010, spring and summer 211, spring and summer 2012). This was done because different species (and therefore functional groups) become apparent in different years and different seasons. Finally, to determine if there was a difference between east and west sites in % cover of native versus exotics, ANOVA was used, using site as replicate. For each of the previous tests, site was the replicate, with all values within a category averaged.
RESULTS AND DISCUSSION

I identified a total of 145 different species within the 25 sites from 2010-2012. The Eastern plains sites had greater species richness than the western slope sites during this study period; 111 species on the eastern plains compared to only 53 species on the Western Slope (Figure 3). 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 the East. 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*. A total of 45 plant families were recorded in both regions. On the eastern plains, 41 families were observed while only 25 families were observed in the west (Table 1). 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).

Differences between the two regions of Colorado may be due to both biotic and abiotic factors, particularly land use. Land use is a key difference between the two regions, as eastern sites have been used for decades to raise cattle, whereas grazing on the West Slope is primarily by wildlife.
The two areas of Colorado had different plant communities. Jaccard Index of similarity indicated that there was 36% similarity between eastern and western sites using all vegetation types, and 50% similarity in tree and shrub communities (Figure 6). East and west sites differed significantly in frequency of different functional groups (Chi-square, Pearson; N=699, DF=8, \( X^2 = 26.34, p<0.0009 \)). Perennial trees had the highest cover in both eastern plains and western slopes, but that shrubs and perennial forbs are more dominant in the west and annual grasses are more abundant in the east (Figure 5).

Although West Slope sites had lower diversity, they started with higher understory relative native cover and fewer exotic species compared to eastern sites (Figure 4, Table 2). This is an indication that more diverse ecosystem do not necessarily translate to healthier ecosystems. This also supports the finding by Levine and D'Antonio (1999) that more diverse ecosystems include also more invasive plant species. On the other hand both regions had similar starting absolute cover of Tamarix at about 30 percent, and less than 10 cover percentage of native woody vegetation.

Plant community structure, species distribution, and functional group are the most important elements for active restoration plans. Results demonstrate that eastern plains have less native cover than the west and the plant communities from each region are not very similar overall. The most important functional groups also differ between the regions. Thus, if active restoration is desired following these four years of passive restoration, it is essential to: 1) have a different plan for each different ecosystem, 2) plant more native species in the eastern sites, and 3) monitor the functional balance to
meet restoration goals, e.g. more grass and forbs for rangeland or more shrubs for wildlife habitat.

It is essential to identify and list all the species that occur in any area being restored. This will help to monitor species distribution for future research and further investigation. For instance, the species list will track any increase in cover percentage of secondary invasives. I also recommended taking species and functional group distribution into account in order to keep the functions of restored ecosystems.
CHAPTER 3

COMPARING DIFFERENT VEGETATION SAMPLING METHODOLOGIES IN A SEMI-ARID, RIPARIAN ECOSYSTEM

INTRODUCTION

Monitoring ecosystems always involves long-term data collection, and therefore can be relatively costly. However, the outcome of these baseline data and the use of subsequent adaptive management techniques are essential for understanding the trajectory of the ecosystem and can significantly increase the success of restoration projects (Zavaleta et al. 2001, Spellerberg 2005). Different ecosystems may be better suited by one method over the other; also, time efficiency of these scientific methods may differ depending on the type or density of the plant community being measured (Floyd and Anderson 1987, Leis et al. 2003, Toledo et al. 2010). Labor and time involved in each method can also play an extreme role and impact the decision with regard to which
method should be used (Goldsmith and Harrison 1976). Because of this, there is great value in determining the fastest, most accurate sampling method for different ecosystems or for different types of data collections and parameters.

Various scientific methodologies are used to monitor the change of organisms and communities, including vegetation cover, frequency, density, and diversity. The line-point intercept, line transect, quadrat, and nested quadrat methods are commonly used for vegetation measurements or to evaluate ecosystem restoration efforts (Heady et al. 1959, Floyd and Anderson 1987, Brady et al. 1995, Birdsall et al. 2012). This includes the response of different ecosystems to invasive removal or disturbance, such as grazing, dams, and other human activities. Although a number of studies have compared different monitoring methodologies, none of them evaluated the accuracy and the efficiency of monitoring vegetation concerning over and understory vegetation plus the categories of native and exotic at the same time (Whitman and Siggeirsson 1954, Heady et al. 1959, Floyd and Anderson 1987, Anderson et al. 2005). Comparing different methodologies help to determine if any of these methods over or underestimated the quantitative measurements of any specific categories and specify the most effective method. Further, from my review of the literature, there are no published studies that compared all six of the six monitoring methodologies currently in use in one location. Our study fills this gap with replication over both time and space.

There are few previous studies that compared two or three different sampling methodologies, but they were not consistent in their findings about the accuracy of different methods and they did not address if whether or not these methods can be
suitable for other ecosystems. Comparisons between line intercept and quadrat method sampling of artificial populations (i.e. a 2-dimensional image) suggested that the line intercept is more precise and requires less time than the quadrat method (Bauer 1936, 1943, Heady et al. 1959). In one field experiment conducted in a sagebrush-steppe arid ecosystem that monitored three parameters (shrub cover, bare ground, and litter), the line-point intercept method accomplished a similar level of precision as did the line intercept and the sampling time was less with about one-third compared to the line intercept. Thus, the line-point intercept was the most efficient method, if an estimates for most of the species and richness in the ecosystem needs to be monitored (Floyd and Anderson 1987, McMahan et al. 2002). In contrast Whitman and Siggeirsson (1954) in their study on mixed grass ecosystems South-western North Dakota found that point intercept over-estimated the percent cover by more than ten percent compared to the line intercept method. Scott and Reynolds (2007) evaluated different sampling techniques in the riparian forest ecosystem along small streams on the Colorado Plateau and found no significant difference in the mean percent cover of the shrub vegetation between 10m² and the line intercept method. However, they recommended using quadrat over transect method to monitor the changes in frequencies of common plants and diversity for fairly common species in the riparian forest ecosystem. They also found that 10m² quadrats requires less sampling effort especially at sites with relatively high numbers of species.

In a comparison of the line-point intercept, quadrat, and modified Whittaker methods to measure species richness, a study conducted at Fort Sill Oklahoma in a grassland ecosystem found that modified Whittaker captured more species richness
compared to the other two methods, and that percent cover of bare ground was strongly correlated with time to complete the survey (Leis et al. 2003). Floyd and Anderson (1987) compared three methods (quadrat, line intercept, and line-point intercept) in sagebrush-steppe ecosystem at Snake River Plain in south-eastern Idaho and found that these methods produced similar measurements of cover percentage only for common shrubs, but not for grass and other vegetation. Bonham (1989) found that line intercept had more precision than the quadrat method when the ecosystem comprises different vegetation types. Hanley (1978) states that when the percent cover of a shrub vegetation arranged between eight to about fifty percent, both 350m line intercept and 50 quadrats using 0.1m$^2$ achieved alike levels of precision, still quadrat required almost half time compared to line intercept. Although there is a discrepancy in the literature concerning what is the best method, line and line-point intercept methods appear to be found more consistently precise, although quadrat methods are more frequently recommended when monitoring of only understory vegetation are desired. However, none of these studies repeated their tests in multiple sites or over multiple years.

Determining the optimal size and shape of sampling units will likely depend on environment, density, frequency, cover, diversity, and plant growth habits and characteristics (Greig-Smith 1964, Chapman 1976). It is known that the majority of plant species grow in clumps, thus the spatial distribution of the species being sampled is the most important factor. Generally, it has been recommended that rectangular plots capture more species than square plots when the species are not evenly distributed across space (Mueller-Dombois and Ellenberg 1974, Goldsmith and Harrison 1976, Elzinga et al.
1998). In contrast, other studies conducted in central Amazonia forest ecosystem, and in highly disturbed grasslands, shrublands and forests in the Mediterranean ecosystem of California, U.S.A. found that rectangular plots had only a slight advantage over the square plots while square plots have several advantages including less bias by the edge effect and sampling more homogeneous areas (Elzinga et al. 1998, Laurance et al. 1998, Keeley and Fotheringham 2005). Further, given equal sample size, Scott and Reynolds (2007) found that larger quadrats captured a higher species frequency and diversity compared to smaller quadrats.

With this contradictory finding in plot size, and as none of the previous studies were applied in the riparian ecosystem, my study will test the differences in plot size and shape in this specific ecosystem. Further, this comparison will be done in the same exact location unlike to the other studies when the comparison was taken in the site scale. I tested four different plot sizes, and compared six different monitoring methodologies for four seasons in two years, beginning in spring 2010. I monitored a total of twenty five sites in the East and West regions of Colorado where restoration that involved the removal of a dominant invasive tree was occurring (Tamarix spp., Chapter 1). To identify the most efficient and objective means to monitor vegetation response to restoration efforts, this chapter addresses whether there are differences between monitoring methodologies in terms of accuracy and time efficiency, taking into account different types of vegetation.

My specific questions were: 1) Do monitoring methodologies differ in their estimates of vegetation cover and species richness? 2) Do monitoring methodologies
differ in time needed to measure the same area? 3) Do various plot sizes or shapes differ in their measurement of vegetation cover percentage and richness? By conducting a more direct comparison to answer these questions I aim to illustrate and clarify some of the inconsistency in findings from previous studies. Unlike the previous studies, this research investigates at the vegetation in different categories as overstory and understory vegetation and also to study each of these categories in terms of exotic vs. native species. Also, my study has a repeated measurement for four times in deferent years and seasons. The ultimate intent was to provide recommendations to land managers concerning the best methods or practices that can be used in such ecosystems.

METHODS

A total of 25 sites were monitored, nine sites in three reaches located on the West slope of Colorado and sixteen sites in five reaches located in the East, within the Arkansas River Valley (Chapter 2). Vegetation was monitored two times a year (spring and summer season) at these sites using six different sampling methodologies in 2010 and 2011.

At each site, five transects were established within the boundaries of a 20 x 50m plot established in a representative spot. For each method that required transects (Point intercept, line transect, nearest neighbor, meter-square quadrats, and densitometer) the
same measurements were taken on the same transect for comparison purposes. For each sampling method, data was collected per species, and then summarized in terms of native or exotic (“nativity”) for the purposes of the analysis. Each method was performed as described below:

Nested Whittaker plots (modified Whittaker):

Also known as the nested quadrat method, the main plot area was 1000m$^2$ with dimensions of 20m x 50m, placed perpendicular to the stream (Figure 2). Within the main plot, several smaller plots of various sizes are distributed throughout to measure different vegetative parameters. Ten small quadrats with dimensions of 0.5m x 2m are distributed evenly around the inside edges (three in each of 50m sides and two in each of the 20m sides), for a total area 10m$^2$. Two 2m x 5m quadrats (for a total of 20m$^2$) and one 5m x 20m quadrat (for a total of 100m$^2$) are placed in the center of the main plot. In sites where the 20m x 50m plot would not fit (two sites out of the 25), we used Whittaker plots with the dimension of 25m x 40m (parallel with the stream) in order to keep the total areas consistent. This method provides measurement of percent cover for understory vegetation (herbaceous & shrubs) plus species richness (Stohlgren et al. 1995, Campbell et al. 2002). The 5m x 20m quadrats located in the center of Whittaker plot were also used for a visual estimate of percent cover of overstory species.
Line-point intercept:

In this method, by using a stratified random method, I placed five 50m transects inside the Whittaker plot, one line-transect per interval of four meters (Figure 2). I then recorded all plants that intercept a point on the line-transect every 10cm. Therefore, each of the five transects had a total of 500 points, for a total of 2500 points for each site. In cases where more than one species was present in a single point (if the vegetation overlapped) and when understory and overstory vegetation was present, all plants were recorded in that point. The percent cover was calculated for each transect as the total number of points for plant species (A) that intercepted with the line-transect divided by total number of points along the transect (500) multiplied by 100. This method was used to calculate the percent cover for both over and understory vegetation, as well as for species richness.


Nearest neighbor:

In this method we recorded the trees that intercepted with the line-transects and the distance to the nearest neighbor to each tree intercepted. This method was promoted
during the 1950s (Clark and Evans 1954). This method was used for the comparison of woody species richness and for time efficiency comparison.

Line transect:

This method was used to calculate and compare both species richness and percent cover of woody species. I recorded the width of the canopy for trees vegetation to measure the vegetation cover by species. The percent cover was calculated as the total length of plant species (A) that intercepted with the line-transect divided by the length of the-transect, multiplied by 100. It has been indicated that this method is objective and requires less time and recommended when the monitoring of only woody vegetation is desired (Heady et al. 1959).

One-meter square quadrats:

In this method, ten 1m² quadrats were randomly placed in every five-meter intervals on each transect and visual estimation of cover percentage and density of understory species were taken. This method was used to measure and compare the percent cover of understory vegetation and species richness. This method was also used to calculate three parameters for understory vegetation: density, frequency, and cover, then the Importance Value (IV), (the summation of relative density, relative frequency, and relative cover (Mueller-Dombois and Ellenberg 1974). With IV, we assess the most important plant species in the area as a way of defining the plant communities, including
change over time (AOAD 1982). The IV can be simplified as percent importance (out of 100 %) value by dividing IV by three as the IV is summation of relative cover, frequency and the density which make it out of 300 % value (Mueller-Dombois and Ellenberg 1974).

Densiometer:

This method was added in the year of 2012 and combined with the quadrat method for time efficiency comparison and used to estimate the percent of overstory vegetation cover, whereas the quadrat method used to estimate the cover percentage of understory vegetation. Using stratified random method, the center of each 1m² in the quadrat method (a total of 50 quadrats per site) were used to locate and take the estimated cover by spherical convex densitometer instrument (Figure 7).

I compared time efficiency of five different methodologies (Nested Whittaker plots, line-point intercept, nearest neighbor, line transect, and 1m² quadrats) in a natural population of riparian zone ecosystem in 2011. Comparison of time efficiency between these five methods was recorded in six sites, three in the east slope with three levels of diversity (low, medium, and high) and three in the west slope with three levels of density (low, medium, and high).

Some of the above methods were combined to measure the time it took to sample both over- and understory vegetation. Line-point intercept vs. quadrat plus densitometer vs. line-intercept plus quadrat vs. Whittaker plot were compared to determine precision of
measuring total cover percentage and time efficiency comparison. Given that line-point intercept is the only method that measures both over and understory vegetation, I expected that it would be more efficient than a combination of 1m² quadrats plus line intercept methods if the measurement of all vegetation types (e.g. over and understory vegetation) are desired.

Even though the plot size in our study has been determined using the species area curve (Rice and Kelting 1955, Lawrey 1991, Wade and Thompson 1991, Jimenez-Valverde 2012) (Figure 8), this relatively large plot (1m²) was difficult to establish in certain vegetation types (e.g. under very dense New Mexican privet, *Forestiera pubescens*). For that reason, we compared plot size (Large= 1m x 1m, Medium= 0.7m x 0.7, Small= 0.25m x 0.25m) for monitoring the percent cover and richness. Also, we tested the hypothesis that an oblong shaped plot will capture more species compared with the square shape of equal area 0.5m x 1 m (0.5 m²) “Oblong” vs. 0.707m x 0.707m (0.5 m²) “Square”, both the oblong and the square quadrat located at the same location at top right corner.

To make comparisons between sampling methods, repeated measures ANOVA was used with transect as the replicate. In this way, measurements taken at the same time in the same place could be compared between sampling methods, quadrat sizes, and quadrat shapes. Measurement comparisons for understory included reach (i.e. study area) and nativity (native vs. exotic). All data were checked for normality and transformed when necessary.
RESULTS

Method comparison of percent cover:

There was a significant difference between point-intercept, quadrat, and Whittaker sampling methods for measuring understory cover (Table 3); point intercept tended to yield lower cover than quadrat except for two of the reaches in the East (Figure 9). There was a significant interaction between methods and reaches; however, methods did not differ in their measurements of cover percentage between seasons (summer vs. spring measurements) or nativity (measurement of natives vs. exotics) or between sites. While there was a significant linear relationship between line-point intercept and quadrat ($P<0.0001 R^2 = 0.56$), and between line-point intercept and Whittaker plot ($P<0.0001 R^2 = 0.69$), the relatively low $R^2$ values suggest that one is not a good proxy for the other.

The comparison between line-point intercept and line-intercept as two methods to measure overstory shows no significant difference ($F=2.91$, DF= 1/231, p=0.09), and the methods were highly correlated with another (linear regression; $P<0.0001 R^2=0.80$, Figure 10).

Time efficiency comparison:

The five sampling methods did not differ in time efficiency comparison, with the exception of the Whittaker plots, which were faster ($F=5.01$, DF= 4,23, $P<0.01$, Figure 11). There was no significant difference in time efficiency between the combined
methods that measured both over and understory vegetation by transect (excluding Whittaker, which has no transect) (F=2.6, DF= 2/42, P <0.086, Figure 12). However, there was a significant difference between combined methods in time required when Whittaker was included in the site scale (F=7.342, DF= 3,11, P < 0.02) (Figure 13).

Test of plot size and Shape:

A comparison 1m² plots, 0.5m² plots (quadrat or oblong), and 0.25 m² plots taken in the same location showed no significant difference in detecting total percent cover (Figure 14). However, it was found that the oblong plots captured significantly more species than the square plots (Figure 15).

Method comparison of species richness:

The total species richness was compared in both 2010 and 2011 using five different methodologies (nested Whittaker plots, line-point intercept, nearest neighbor, line transect, and 1m² quadrats). Four methods (nested Whittaker plots, line-point intercept, nearest neighbor, and line transect) were used to capture and compare overstory richness. Results showed that all four methods consistently captured equal total number of woody species, which generally included only five or six species (Tamarix spp, Populus spp, Salix exigua, Juniperus scopulorum, Acer negundo, and Celtis reticulata). However, in understory vegetation, Whittaker plots consistently captured more species compared to the line-point intercept and 1m² quadrat methods (Figure 16).
DISCUSSION

My results demonstrated that although measurements will be correlated between methods, they were not the same either in estimates of cover or in the amount of time they took. Only the Whittaker plot differed dramatically, however, which was faster and captured more species. Unlike other methods, the data shows that the centered 100 m$^2$ quadrat did not capture the canopy of the willow (*Salix app.*), which almost always grows in the first five to ten meters from the river bank while the 100 m$^2$ quadrat starts after 15 meters. Given that the line-point intercept is the only method that monitors both over and understory vegetation and did not significantly differ on time required, I conclude that it is the most effective method when all types of plant communities need to be monitored. In addition, this method requires fewer and less expensive tools to carry in the field and fewer office hours in data entry and data analysis compared to any of the combined methods. My results also support findings from Floyd and Anderson (1987) that the measurements from point interception and line interception were very comparable.

Line-point intercept is the only method that can capture both over and understory vegetation and also one of most objective methods (Heady et al. 1959, Jonasson 1988). However, the recommendation regarding what method should be used should not be considered universal. For instance, line intercept method would be recommended over the line-point intercept if the project or research is interested in monitoring only woody vegetation. This method has the same precision and requires less time than line-point intercept. In contrast, if woody vegetation is not part of the ecosystem (e.g. grassland
ecosystem) or only the understory vegetation is what is being monitored, Whittaker plots or quadrat method would be recommended. My results suggest that the Whittaker method is best if the researchers are interested in monitoring species richness and cover percentage, whereas 1m$^2$ quadrats are better for estimates of (IV) as it is more intensive than Whittaker and homogeneously distributed inside the Whittaker plot.

Results from plot size testing show that using a smaller size quadrat will not have an effect on the accuracy of capturing percent cover. Thus it is easier and is suggested to use a smaller (0.25 m$^2$ quadrat) instead of bigger (1 m$^2$ quadrat) quadrats in such riparian ecosystems. Results also demonstrate that oblong 0.5m x 1m is preferred over a square quadrat with equal area when species richness needs to be measured. Line-point intercept was chosen for future monitoring in our project as both over and understory vegetation are desired. My research suggested this method to be used in similar riparian zone and semi-arid ecosystems. However, if the understory is the only type of vegetation being monitored, 0.25 m$^2$ quadrat method is recommended over all other methods including 1m$^2$ quadrat method which was difficult to establish in certain vegetation types (e.g. under very dense New Mexican privet, *Forestiera pubescens*).

This is the first time six different methodologies were tested in multiple sites to recommend the most effective methods taking into consideration what type of data collected are desired. My research notably found that using a smaller (0.25 m$^2$ quadrat) will not affect the accuracy of cover percentage compared to relatively large (1 m$^2$ quadrat) in a riparian ecosystem. This finding will save time and efforts in the future research in this ecosystem.
CHAPTER 4

VEGETATION RESPONSE TO INVASIVE TAMARIX SPP. REMOVAL

INTRODUCTION

The biodiversity and function of many natural ecosystems are at risk by the spread and colonization of invasive species (Vitousek et al. 1996, Mack et al. 2000, Firn et al. 2010, Marchante et al. 2011). In many cases, invasive species removal and control are the first step of ecosystem restoration (El waer and Abido 1995, Zavaleta et al. 2001, Shafroth and Briggs 2008). However, the outcome of such removal is often uncertain. Rapid spread of invasive plants in the western United States has resulted in government approval to speed up weed removal and restoration efforts (Mack et al. 2000, Shafroth et al. 2005). Both land managers and scientists are still uncertain as to whether or not the outcome of invasive removal efforts will lead to an increase to desired species (Shafroth et al. 2005, Stromberg et al. 2009, Bay 2013). Removal of invasive woody vegetation can increase indigenous species diversity and richness; however, other
noxious species or secondary invasive can thrive after the removal of target invasive species (Webb et al. 2001, Hartman and McCarthy 2004, Ogden and Rejmanek 2005). This research addresses the need of long-term monitoring after invasive removal by monitoring the response of plant communities in “passive restoration” efforts after invasive *Tamarix* removal. Passive restoration is an approach when natural processes are allowed to take place after the cessation or removing the cause of ecosystem degradation or preventing the natural recovery process without active restoration such as re-vegetation; no plantings are done (Kauffman et al. 1997).

The response of invaded ecosystems to the control of invasive species varies widely (Denslow and D'Antonio 2005, Cuevas and Zalba 2010, Gardener et al. 2011, Douglass et al. 2013). One example of a successful restoration project, after removal of invasive *Acacia longifolia*, found that desired species were the majority in the treated plots with more than 70% of absolute cover, and species richness was higher (Marchante et al. 2011). Moreover, the species richness was relatively higher in the plots when the dead biomass of the invasive was removed. Another successful example of restoration efforts after invasive control address by Cuevas and Zalba (2010). They found a gradual increase of native species after the removal of invasive *Pinus halepensis* (Aleppo pine), and temporary increase in cover of exotic species. However, after four years, exotic species cover was decreased to equivalent levels on non-invaded sites. Thus, the restoration process can be successful; however, it may be slow and therefore requires monitoring beyond two years. Thus, many projects may appear to have failed using short term monitoring.
A review of 30 invasive removal projects including 23 invasive species in the Galapagos reported that only four projects were successful (Gardener et al. 2010a). However another review of 355 papers published during 1960-2009 indicates that long term restoration programs results to higher density and cover of native compared to short restoration projects (Kettenring and Adams 2011). But, out of these 355 papers few evaluated the response after control for more than two years.

Removal of invasives can also result in a decrease in natives and an increase of non-desired species. Kettenring and Adams (2011) found that burning treatments decreased native species and increase invasive species. Results after removal of invasive shrub *Rubus niveus* on Santiago Island showed a significant decline in both density and seed bank of other invasives, five years after removal, plant community and vegetation structure in treated sites was dominated by undesired herbaceous species, unlike the woody plant community of native control sites (Renteria et al. 2012). Another ineffective example of invasive removal indicates that the exotic species dominated the plant communities post removal of *Elaeagnus angustifolia* (Russian olive). However, given that no control or pre removal data were established, it is unclear whether the greater cover of exotic was a response of invasive removal or a result of any other cofounded variable (e.g. drought or overgrazing) (Gaddis 2008). Similarly, 20 years after removal of *Martynia annua* in Australia, restoration had failed due rapid seed production and dispersal strategy. (Gardener et al. 2010b). The failure of many restoration efforts is attributed to limited funds or due to denial of landowners to property access; thus addressing this problems will help to reach the restoration aims (Shafroth et al. 2013).
Recently, scientists, land managers, government and private institutions in the United States have given much attention to invasive control and restoration projects along western rivers (Harms and Hiebert 2006, Dennison et al. 2009). In the west, removal of *Tamarix* spp. (a.k.a. tamarisk, saltcedar) has been a primary focus of these projects. These trees were first introduced to North America during 1800s from southern Europe and the eastern Mediterranean zone mainly to decrease erosion, wind breaks, and slow down water flow in riparian and agriculture areas (Di Tomaso 1998). In several states including Colorado, Montana, Nevada, New Mexico, Washington, Oregon, North Dakota, and Wyoming, where almost no riparian ecosystem exists without *Tamarix*, it’s listed as a noxious-weed (McDaniel et al. 2004). Despite the fact that invasive plant species removal has been a priority in restoration of riparian ecosystems, relatively little is understood about the ecological impact of the removal of invasive species from such ecosystems (Shafroth et al. 2005, Cuevas and Zalba 2010).

Colonization by noxious species and secondary invasion after *Tamarix* removal is a substantial subject of concern (Shafroth and Briggs 2008, Sher et al. 2008). However, more research in this finding is needed. Research on plant community response to *Tamarix* spp. removal is mixed. A survey of 33 *Tamarix* removal and non-removal sites where only passive revegetation had been done by Harms and Hiebert (2006) found a decrease in the cover of *Tamarix* spp. in the removal sites compared to the control 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. When *Tamarix* was excluded from data analysis, there was no difference across all sites in species
composition. In contrast, a similar study with active re-vegetation showed that the recovery of native species depended on site characteristics such as moisture availability and was correlated to Tamarix cover, with a greater response of native species in less dense Tamarix (Bay and Sher 2008). Generally, results indicate that re-vegetation efforts were successful with higher establishment of natives and less cover of Tamarix in area of high water availability and good soil conditions for native species.

The limitation of these research projects, however, is that monitoring was done at a single point in time and in sites where Tamarix had been removed at different times. To draw better conclusion about the ecological impact overtime after Tamarix removal should involve more specific research and long term monitoring.

Even though different Tamarix removal techniques have been used, few removal projects monitored the vegetation response to these various Tamarix removal techniques (Sher et al. 2002, Harms and Hiebert 2006, Bay and Sher 2008, Sher et al. 2008, Hultine et al. 2010). Harms and Hiebert (2006) compared two different removal techniques, cut stump and burning, followed by chemical spray and found no significant difference in percent cover, richness, or species diversity. However, monitoring at a single point in time or combining sites with different periods of time since Tamarix removal can have misleading results because of the variation between years or different periods of time since removal. My research has more power to compare and detect the change in plant community composition and help draw a better conclusion as I monitored the vegetation twice yearly for three and a half years in the sites where Tamarix was concurrently removed.
Long-term monitoring of riparian ecosystems is required to address the long-term impacts of *Tamarix* removal on the ecosystem (Scott and Reynolds 2007). The goal of my study is to investigate these knowledge gaps by monitoring vegetation response after *Tamarix* removal in a long-term study. My broad question is whether vegetation responses post-removal of the invasive *Tamarix* differs over time in terms of cover percentage and density of desired species. My specific questions are: 1) Is *Tamarix* removal effective in reducing percent cover of *Tamarix*? 2) How does the vegetation community respond to *Tamarix spp.* removal? 3) What is the relationship between *Tamarix* and native cover? 4) Do other environmental variables such as grazing or drought explain some of the response of native and exotic vegetation cover?

My intent, by measuring the impact of *Tamarix* removal in the ecosystem via the measurement of vegetation parameters for three and a half years, is to help answer some of the controversial questions about the ecological impact of *Tamarix* removal on these ecosystems. This includes testing the predictions that removal of *Tamarix* will increase native cover, and whether or not the increase in desirable species will also prevent secondary invasion of introduced and noxious species.
METHODS

A total of 25 sites were monitored, nine sites in three reaches located on the Western slope and sixteen sites in five reaches located on the eastern plains (Chapter 2). These sites represented a range of *Tamarix* removal methods including cut-stump, aerial application of herbicide, track hoe, and a biological control (Table 4).

Vegetation was monitored two times annually by the line-point intercept method between 2010 and 2012 during the spring and summer seasons (Chapter 3). Soil was also sampled in all sites twice annually. In each site, five soil samples were collected and homogenized for analysis. Soil was sampled by collecting the top 10 cm of soil from each corner of the Whittaker plot, and one from the plot center. There were a total of 118 samples collected over three years.

In summer 2012, the density of cow patties was monitored to test the impact of grazing on the vegetation cover. This was done by recording the number of cow droppings that occurred with a half meter on either side of the transect.

Statistical Analysis:

To determine if *Tamarix* removal efforts were effective in killing *Tamarix*, I performed two analyses: an ANOVA at the last sampling period (summer 2012, after 3 ½ years) to compare percent cover in removal vs. non-removal sites, and a repeated measures ANOVA from summer 2011 to summer 2012 comparing the same sites over time between removal and non-removal sites. ANOVA was also preformed to detect the
change over time of understory native relative cover within different removal methods. Both importance value and repeated measures ANOVA were used to determine the respond of plant community to *Tamarix* spp. removal.

RESULTS

*Tamarix* in removal vs. non-removal sites:

The absolute percent cover of *Tamarix* is more than ten times less in the removal sites than the non-removal sites. By 2012, however, this difference was greater in the east sites compared to the west (Table 5, Figure 17). Comparing the same sites over time before vs. after removal, there is a significant decrease in *Tamarix* after removal with more dramatic decrease on the West Slope (repeated measures ANOVA, before/after*slope: F=4.13, DF=1/83, p<0.05). On the western slope, there was a dramatic decrease in the total absolute cover of *Tamarix* in the spring 2011; in contrast, *Tamarix* cover was still increasing on the eastern plains in non-removal sites (Figure 17).

There was also a decrease in total density of *Tamarix* in the removal sites compared to non-removal sites from the first to third year; repeated measures ANOVA showed a significant difference in *Tamarix* density between removal and non-removal sites but with a significant interaction with region (Figure 18). On the western slope, there was dramatic increase of *Tamarix* density in 2012 in removal sites. In contrast, there was a continuous decrease of *Tamarix* density on the eastern plains removal sites.
Plant community response to *Tamarix* removal:

Two years after removal, the absolute cover of understory introduced species is less where *Tamarix* has been actively removed (Table 6 and Figure 19). Other than the native control sites, the lowest percent cover of understory exotic species was found where *Tamarix* was removed by the cut stump method (Figure 20). However, the percentage cover of introduced species where herbicide was applied by helicopter increased to exceed even the percentage cover of introduced species under the *Tamarix* control sites. In contrast, results show that 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 (Figure 21).

Given that very different patterns appeared in the sites where helicopter spray was applied, I 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). Secondary invasive *Kochia scoparia* (burningbush) ranked third in IV before removal took the first after removal, shifting the community from native dominated (*Elymus canadensis*) to exotic because of the increase in the relative cover of *Kochia scoparia* (Figure 22). There was no change in the relative density of *Kochia scoparia* before and after removal; individuals simply got larger.
The response of species richness to *Tamarix* removal:

Species richness on the east and western slope responded differently. In the eastern sites, species richness dramatically decreased in the first season after *Tamarix* removal. In contrast, in the Western slope sites there was a dramatic increase in the second season after *Tamarix* removal. There was a significant change in species richness with seasons since *Tamarix* removal, but did so differently for sites in the east versus west (Figure 23). Four seasons post removal, species richness in the western slope sites increased to exceed the species richness pre removal. Meanwhile, six seasons after removal, there was an increase of species richness in the eastern plains sites to almost reach the same number of species pre removal.

Impact of grazing:

A regression test, performed in transformed data (log+1), shows that there was no significant linear relationship between the grazing and percentage cover of either introduced (N= 114, $p=0.19, R^2=0.02$) or native (N= 114, $p=0.60, R^2=0.01$) species.
DISCUSSION

*Tamarix* in removal vs. non-removal sites:

The absolute cover of *Tamarix* is ten times less in the removal sites compared to non-removal sites in both regions sampled. However, there was a dramatic increase of *Tamarix* density in the western slope sites two years post removal. The *Tamarix* density in other words was almost the same in removal sites compared to non-removal sites, suggesting that it may be a matter of time before the canopy of re-growth of *Tamarix* increases. In contrast, the eastern plains results indicate that the *Tamarix* density is still very low or even slightly decreased over time, likely because all eastern plains sites are on private land and thus the re-growth was regularly checked by the landowner and herbicide was applied multiple times as needed.

Although the west slope sites suffered from re-establishment of *Tamarix* in removal sites, there was dramatic decrease in *Tamarix* cover overall, including in non-removal sites. The dramatic decrease of total absolute *Tamarix* cover in the western slope during the spring, 2011, even in the sites when the *Tamarix* was not actively removed, is likely a result of the defoliation by *Tamarix* beetles, *Diorhabda spp.*

In contrast *Tamarix* cover was still increasing in the eastern plains in non-removal sites where the beetles were absent.
Plant community response to *Tamarix* removal:

Results clearly exhibit that helicopter spray led to an increase in the absolute cover of introduced species and decrease in the relative cover percentage of desired species as reflected in native cover. This is due to secondary invasion by imazapyr resistant *kochia scoparia* (Figure 20 and Figure 22). Thus, as now, this method should not be recommended for restoration efforts in such riparian ecosystems. Cut stump method on the other hand, results in lowest introduced cover and highest relative native cover and thus it should be recommended over all other removal methods.

The response of species richness to *Tamarix* removal:

Four seasons after removal, species richness increased in both eastern and western sites. Accordingly, *Tamarix* removal efforts led to direct-positive impact in species richness (Figure 23). The difference between the slopes’ responses is due to the difference in initial plant communities where most of understory vegetation were shrubs in the western slope unlike the eastern plains with herbaceous.

CONCLUSIONS

Passive restoration had some promising results as species richness and native relative cover was increased over time. Meanwhile, there was a dramatic increase of *Tamarix* density on the western slope sites. Thus it is critical at this point to have a
follow-up treatment to reduce the re-growth of *Tamarix*. However, *Tamarix* beetles are a possibility to do so, but these sites need to be monitored. Cut stump method should be recommended over all other removal methods as it results in lowest cover of introduced species.
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**APPENDIX 1:**

**Table 1.** Species list for East and Western of Colorado that incurred and identified in the study area. Nativity (Nat.) N=Native, I=Introduced, U=Unknown. East and West column indicates the frequency of the species on the transects during the study period.

<table>
<thead>
<tr>
<th>Scientific</th>
<th>Family</th>
<th>Nat.</th>
<th>Functional Group</th>
<th>East</th>
<th>West</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abronia Spp.</td>
<td>Nyctaginaceae</td>
<td>N/A</td>
<td>N/A</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Acer negundo</td>
<td>Aceraceae</td>
<td>N</td>
<td>Perennial Tree</td>
<td>0</td>
<td>25</td>
</tr>
<tr>
<td>Achnatherum hymenoides</td>
<td>Poaceae</td>
<td>N</td>
<td>Perennial Graminoid</td>
<td>15</td>
<td>30</td>
</tr>
<tr>
<td>Aegopilus repens</td>
<td>Asteraceae</td>
<td>I</td>
<td>Perennial Forb</td>
<td>0</td>
<td>215</td>
</tr>
<tr>
<td>Agroptilon cylindrica</td>
<td>Poaceae</td>
<td>I</td>
<td>Annual Graminoid</td>
<td>5</td>
<td>0</td>
</tr>
<tr>
<td>Agropyron cristatum</td>
<td>Poaceae</td>
<td>I</td>
<td>Perennial Graminoid</td>
<td>10</td>
<td>20</td>
</tr>
<tr>
<td>Agrostis gigantea</td>
<td>Poaceae</td>
<td>I</td>
<td>Perennial Graminoid</td>
<td>60</td>
<td>35</td>
</tr>
<tr>
<td>Amaranthus arenicola</td>
<td>Amaranthaceae</td>
<td>N</td>
<td>Annual Forb</td>
<td>65</td>
<td>0</td>
</tr>
<tr>
<td>Amaranthus retroflexus</td>
<td>Amaranthaceae</td>
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<td>Annual Forb</td>
<td>95</td>
<td>0</td>
</tr>
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<td>Ambrosia acanthicarpa</td>
<td>Asteraceae</td>
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<td>Annual Forb</td>
<td>10</td>
<td>0</td>
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<tr>
<td>Ambrosia psilostachya</td>
<td>Asteraceae</td>
<td>N</td>
<td>Annual Forb</td>
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<td>0</td>
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<tr>
<td>Apocynum sibiricum</td>
<td>Apocynaceae</td>
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<td>Perennial Forb</td>
<td>50</td>
<td>35</td>
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<tr>
<td>Arctium minus</td>
<td>Asteraceae</td>
<td>I</td>
<td>Biennial Forb</td>
<td>80</td>
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<tr>
<td>Argemone polyanthemos</td>
<td>Papaveraceae</td>
<td>N</td>
<td>Annual Forb</td>
<td>15</td>
<td>5</td>
</tr>
<tr>
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<td>Poaceae</td>
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<td>Asteraceae</td>
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<td>Perennial Subshrub</td>
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<td>15</td>
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<td>Perennial Shrub</td>
<td>60</td>
<td>94</td>
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<tr>
<td>Atriplex spp.</td>
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<td>Annual Forb</td>
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<td>Scientific Name</td>
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<td>Life Form</td>
<td>Height (cm)</td>
<td>Spread (cm)</td>
</tr>
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<td>-----------------------------------</td>
<td>---------------------------</td>
<td>---------------------</td>
<td>--------------------</td>
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</tr>
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<td>Poaceae</td>
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<td>Code 2</td>
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**Table 2.** ANOVA test for differences in percent cover (log +1) for native vs. exotic (“nativity”) and location in east vs. west sites (“Slope”). Replicate is the site mean.

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<td>nativity</td>
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<td>3.17</td>
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<tr>
<td>nativity*Slope</td>
<td>1</td>
<td>53.23</td>
<td>0.0001*</td>
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<tr>
<td>Whole Model</td>
<td>3/199</td>
<td>23.69</td>
<td>0.0001*</td>
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**Table 3.** Repeated measures ANOVA test for comparisons of understory cover between different sampling methodologies (point-intercept, quadrat, Whittaker) and for interactions with site (nested within reach), reach, nativity (natives vs. exotics), and season (spring vs. summer sampling).

<table>
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<th>Method</th>
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<td>Method*Season</td>
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<td>Method*Nativity</td>
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<td>Method*Site[Reach]</td>
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<td>0.77</td>
<td>0.78</td>
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Table 4. Site names, locations, period of Tamarix removal, and treatment for East and West Colorado. Treatments include two controls where no active removal method has been used (native control and Tamarix control), and four active removal methods: cut stump (tree has been cut to the ground and herbicide applied), track hoe (biomass of tree removed including roots), helicopter spray (herbicide applied using a helicopter), hydroaxe (above ground tree biomass chipped, no herbicide applied).

<table>
<thead>
<tr>
<th>Slope</th>
<th>name</th>
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<th>Elevation</th>
<th>Tamarix removal</th>
<th>Treatment</th>
<th>Beetles present?</th>
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<td>East</td>
<td>Larson S1</td>
<td>38°2’11”N 104°0’14”W</td>
<td>4461 ft</td>
<td>Spring, 2010</td>
<td>Helicopter spray</td>
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<tr>
<td>East</td>
<td>Larson S2</td>
<td>38°1’59”N 104°0’15”W</td>
<td>4459 ft</td>
<td>Spring, 2010</td>
<td>Helicopter spray</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Larson S3</td>
<td>38°3’18”N 104°0’42”W</td>
<td>4445 ft</td>
<td>Spring, 2010</td>
<td>Helicopter spray</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Larson S4</td>
<td>38°4’2”N 104°0’29.9”W</td>
<td>4348 ft</td>
<td>No</td>
<td>Tamarix Control</td>
<td>No</td>
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<tr>
<td>East</td>
<td>Doherty 1</td>
<td>37°29’36”N 103°36’57”W</td>
<td>4706 ft</td>
<td>No</td>
<td>Native Control</td>
<td>No</td>
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<td>East</td>
<td>Doherty 2</td>
<td>37°29’25”N 103°36’53”W</td>
<td>4620 ft</td>
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<td>Cut Stump</td>
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</tr>
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<td>37°29’28”N 103°36’54”W</td>
<td>4667 ft</td>
<td>Summer, 2010</td>
<td>Cut Stump</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Wooten S1</td>
<td>37°33’24”N 103°39’0”W</td>
<td>4557 ft</td>
<td>Summer, 2010</td>
<td>Cut Stump</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Wooten S2</td>
<td>37°33’0”N 103°38’21”W</td>
<td>4426 ft</td>
<td>Spring, 2009</td>
<td>Trackhoe</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Wooten S3</td>
<td>37°33’28”N 103°39’3”W</td>
<td>4516 ft</td>
<td>No</td>
<td>Tamarix Control</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Pamena Gap W1</td>
<td>37°31’44”N 103°W</td>
<td>4540 ft</td>
<td>Spring, 2012</td>
<td>Tamarix Control</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Pamena Gap W2</td>
<td>37°31’38”N 103°40’30”W</td>
<td>4540 ft</td>
<td>Spring, 2012</td>
<td>Cut Stump</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Pamena Gap W3</td>
<td>37°31’36”N 103°40’28”W</td>
<td>4548 ft</td>
<td>Spring, 2012</td>
<td>Trackhoe</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Pamena Gap E1</td>
<td>37°31’52”N 103°40’3”W</td>
<td>4523 ft</td>
<td>Spring, 2012</td>
<td>Trackhoe</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Pamena Gap E2</td>
<td>37°31’53”N 103°40’1”W</td>
<td>4520 ft</td>
<td>Spring, 2012</td>
<td>Cut Stump</td>
<td>No</td>
</tr>
<tr>
<td>East</td>
<td>Pamena Gap E3</td>
<td>37°31’55”N 103°40’1”W</td>
<td>4522 ft</td>
<td>Spring, 2012</td>
<td>Tamarix Control</td>
<td>No</td>
</tr>
<tr>
<td>West</td>
<td>B.G.V1</td>
<td>38°8’20”N</td>
<td>5381 ft</td>
<td>Spring, 2011</td>
<td>Cut Stump</td>
<td>Yes</td>
</tr>
</tbody>
</table>
Table 5. Tamarix cover for sites to determine differences between sites before vs. after Tamarix removal and slope (east vs. west sites) using ANOVA at the last sampling period (summer 2012, after 3 ½ years).

<table>
<thead>
<tr>
<th>Source</th>
<th>DF</th>
<th>F</th>
<th>P&lt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before vs. After</td>
<td>1</td>
<td>533.03</td>
<td>&lt;.0001*</td>
</tr>
<tr>
<td>Slope</td>
<td>1</td>
<td>288.68</td>
<td>&lt;.0001*</td>
</tr>
<tr>
<td>Slope* Before vs. After</td>
<td>1</td>
<td>270.20</td>
<td>&lt;.0001*</td>
</tr>
<tr>
<td>Whole model</td>
<td>3/114</td>
<td>308.08</td>
<td>&lt;.0001*</td>
</tr>
</tbody>
</table>
Table 6. Understory vegetation response to whether Tamarix has been removed yet, site location (slope), and nativity (native vs. exotic) in the final season (summer 2012) using an ANOVA test. This analysis does not include native control sites (i.e. where Tamarix never was present).

<table>
<thead>
<tr>
<th>Source</th>
<th>DF</th>
<th>F</th>
<th>P&lt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>Slope</td>
<td>1</td>
<td>1.23</td>
<td>0.27</td>
</tr>
<tr>
<td>Has Tamarix been removed yet</td>
<td>1</td>
<td>3.28</td>
<td>0.07</td>
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<tr>
<td>Slope*Has Tamarix been removed yet</td>
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<td>1.07</td>
<td>0.30</td>
</tr>
<tr>
<td>nativity</td>
<td>1</td>
<td>0.53</td>
<td>0.47</td>
</tr>
<tr>
<td>Slope*nativity</td>
<td>1</td>
<td>27.09</td>
<td>&lt;.0001*</td>
</tr>
<tr>
<td>Has Tamarix been removed yet*nativity</td>
<td>1</td>
<td>18.89</td>
<td>&lt;.0001*</td>
</tr>
<tr>
<td>Slope<em>Has Tamarix been removed yet</em>nativity</td>
<td>1</td>
<td>4.05</td>
<td>0.05*</td>
</tr>
<tr>
<td>Whole model</td>
<td>7/249</td>
<td>8.77</td>
<td>&lt;.0001*</td>
</tr>
</tbody>
</table>
LIST OF FIGURES

APPENDIX 2:

Figure 1 A. Site locations for West slope of Colorado. A total of 9 sites.
Figure 1 B. Site locations for Eastern plains of Colorado. A total of 16 sites.
Figure 2. Sampling design used. Each quadrat contains nested 1, 10, 100 and 1000m² Modified Whittaker sampling plots, as well as five randomly placed 50m line transects, each containing 10 1m² quadrats. Transects were used for ---list methods here--- (figure adapted from one created by R. Bay, based on Stohlgren et al. 1995).
Figure 3. Species richness for Eastren and western Colorado during the study period 2009-2012.
Figure 4. (Average with 1 +/- SE) % cover of understory Native and introduced species for east and west Colorado spring, 2010.
Figure 5. (Average with 1 +/- SE) East and Western slope of Colorado, percent cover by functional group. (Chi-square, Pearson; N=699, DF=8, X2 = 26.34, p<0.0009).
Figure 6. Jaccard Index of Similarity showing the similarity of plant community between Eastern plains and Western slope sites in Colorado, using the data from 3 years.
Figure 7. Spherical convex densitometer,
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Soil Conservation Service
U. S. Department of Agriculture
Washington, D.C.
Figure 8. Species - Area Curve. The minimal sample area was determined through the species area curve by increasing the quadrat size starting with 0.5m x 0.5m = 0.25m². The minimal sample area used was 1m x 1m = 1m².
Figure 9. Comparison of methods: Point-intercept vs. Quadrat. In measuring understory cover, quadrats tended to overestimate cover relative to Line-point intercept in most reaches.
Figure 10. Comparison of methods: Line-point-intercept vs. Line-intercept, measuring overstory cover at 8 different reaches. Each point represents an average at site scale.
Figure 11. (Average with 1 +/- SE). Time Efficiency Comparison (average time per method per site). AVE = average and the number 6 or 3 indicates the number of sites where methods were performed. Letters indicate significant differences between Methods in a Tukey post-hoc test. \( F=5.01, \ DF=4,23, P \text{ value} < 0.006 \).
Figure 12. (Average with 1 +/- SE). Time Efficiency Comparison (average time per method per transect). \((F=2.6, \ DF= 2.42, \ P \ value < 0.09)\).
Figure 13. (Average with 1 +/- SE). Time Efficiency Comparison (time per method per site). Letters indicate significant differences between methods in a Tukey post-hoc test. ($F=7.342$, $DF=3,11$, $P$ value < 0.02).
Figure 14. Plot Size Testing. (Average with 1 +/- SE) of total understory cover for each of four plot sizes (Large = 1m x 1m, Medium = 0.7m x 0.7, Oblong = 1m x 0.5 m, Small = 0.25m x 0.25m) at three sites. ANOVA was used to test differences in plot size across sites and found no significant interaction between plot size and site (p>0.254).
Figure 15. Pairwise comparison of oblong versus square plots for species richness, as compared with a 1:1 line. Points above the line were those where oblong plots found more species than square plots sampled in the same area. (pairwise t-test: t=3.24, p<0.002).
Figure 16. Richness comparison of the understory vegetation for the West slope Summer 2011. Total count of species richness.
Figure 17. (Average +/- 1SE) This graph shows total absolute cover of Tamarix spp. of removal vs. non-removal sites on the east and western slope of Colorado.
Figure 18. (Average +/- 1SE) This graph shows total number of Tamarix (density) per site of Tamarix spp. on removal vs. non removal sites. In the eastern plains and western slope of Colorado. Spring season, Cut stump method. ANOVA test ($F = 0.340, DF = 2, P$ value <0.004).
Figure 19. Percent cover of understory vegetation (Average +/- 1SE) by nativity (native “N” and exotic “I”) for east and west sites for sites where yet removed or not (“has Tamarix been remove yet”).
Figure 20. (Average +/- 1SE) %cover of understory introduced species within different removal methods. ($F = 14.6$, $DF = 5,129$, $P$ value $<0.0001$).
Figure 21. (Average +/- 1SE) Change over time of understory native relative cover within different removal methods. ($F = 55.930$, $DF = 5,599$, $P < 0.0001$).
Figure 22. Plant communities as indicated by the Importance value A) before treatment and B) one year after helicopter spray treatment.
Figure 23. (Average + 1SE) This graph shows the mean number of species per transect over time of the removal sites in both east and west slope sites. Repeated measures ANOVA shows a significant difference over time ($F = 5.53$, $DF = 2.52$, $P < 0.0001$).