DISSERTATION

ECOSYSTEM IMPACTS OF TAMARISK (*TAMARIX* SPP.) MANAGEMENT IN THE ARKANSAS RIVER WATERSHED, COLORADO: EFFECTS OF DISTURBANCE AND HERBICIDE RESIDUES ON PASSIVE PLANT COMMUNITY RESTORATION

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ABSTRACT

ECOSYSTEM IMPACTS OF TAMARISK (*TAMARIX* SPP.) MANAGEMENT IN THE ARKANSAS RIVER WATERSHED, COLORADO: EFFECTS OF DISTURBANCE AND HERBICIDE RESIDUES ON PASSIVE PLANT COMMUNITY RESTORATION

Tamarisk (*Tamarix* L.) is now one of the most common species of woody plants along waterways in arid and semi-arid areas of the western United States. Tamarisk was intentionally introduced over a century ago for ornamental purposes and erosion control projects, but its expansion since has been influenced by altered hydrologic regimes and global climate change. Approximately sixty years ago the species started to be perceived by federal scientists as noxious and was targeted for control. As the first chapter in this dissertation outlines, management of tamarisk has occurred by many methods, but primarily combinations of herbicides and mechanical tree removal. Successive chapters detail laboratory, greenhouse and field experiments that determined the ecological impacts of currently used tamarisk control strategies, with a particular emphasis on the effects of herbicide residues on plant community restoration patterns following management.

First, an *in vitro* study and high-performance liquid chromatography (HPLC) analysis were used to quantify soil degradation rates for imazapyr and triclopyr from six sites in Colorado. A dose response study was then conducted at two of these sites to determine the relative sensitivity of important restoration plant species to the two herbicides. Exponential decay models estimated imazapyr half-lives (t_{50}) for two soils at 51 and 76 days, and triclopyr half-lives (t_{50}) for all soils averaged 7 days. *Glycyrhiza lepidota* was the only species to demonstrate sensitivity

to triclopyr. *Atriplex canescens*, *Elymus canadensis* and *Sporobolus airoides* were the most sensitive to typical imazapyr residues. Fecundity in *S. airoides* and *Bouteloua curtipendula* were also negatively impacted by the highest rate of both triclopyr (3.92 kg ai ha^{-1}) and imazapyr (0.28 kg ai ha^{-1}). Microbially-mediated degradation of triclopyr was estimated to occur 6.5 times more rapidly than imazapyr.

Second, at three field sites in southeastern Colorado a study was conducted that used three dimensional artificial trees and repeated soil sampling to determine whether tamarisk tree canopies retained aerially-applied imazapyr, and if this retention affected soil residues and degradation. Tamarisk mortality was also quantified using repeated stand and individual tree measurements. The average tree canopy captured 75% of aerially-released imazapyr, resulting in significantly lower soil residues beneath the tree canopy. Although initial imazapyr soil residue levels outside the tree canopy were almost four time greater than those inside, soil degradation occurred more than twice as rapidly in outside soils and resulted in lower residue levels. Helicopter imazapyr applications resulted in 98% tamarisk mortality within two years, but the consistency of treatment effectiveness was reduced by non-linear stand boundaries and tall site obstructions. The same factors also increased variability in the actual quantity of herbicide applied to sites, increasing the probability of substantial non-target ecosystem impacts.

Last, field plots were established at four sites in southeastern Colorado where tamarisk stands were treated with either an aerial imazapyr application or mechanical biomass removal followed by secondary herbicide (imazapyr and triclopyr) or biological control treatments. In the fourth chapter a study conducted at these sites is detailed in which the tamarisk control and cost effectiveness of the different treatments was quantified over a three year period. Whole plant extraction caused 20% higher tamarisk mortality than aerial imazapyr applications or biomass

mulching. Of the secondary treatments evaluated, individual plant treatments (IPTs) of imazapyr caused higher mortality than either triclopyr IPTs or releases of tamarisk leaf beetles (*Diorhabda carinulata*). Aerial imazapyr applications alone were very cost effective, but when the subsequent removal of tree biomass was accounted for, this strategy was less cost effective than primary mechanical treatments followed by biological control releases.

In the final chapter a second study carried out at the same sites is described in which the validity of ecologically based integrated pest management (EBIPM) models for tamarisk management are tested by measuring plant community and ecosystem responses to the different tamarisk control strategies. Plant community dynamics in response to the adjacent treatments were evaluated over three years. Helicopter imazapyr applications severely reduced plant community richness, diversity and abundance and appeared to facilitate invasion by resistant populations of *Bassia scoparia*. Plant communities did not show a strong response to integrated tamarisk management, which in itself was notable because mechanical tree removal caused soil disturbances that in theory would have promoted secondary invasions of existing ruderal species. Ultimately data suggested that plant community re-vegetation patterns following tamarisk removal were more strongly affected by drought and longer term shifts towards community assemblages dominated by upland plant species. These results provide evidence for the need to integrate state and transition models of ecosystem structure and function into the EBIPM framework in order for this tool to be valuable in managing tamarisk and other woody invaders.

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CHAPTER 1. TAMARISK MANAGEMENT: LESSONS AND TECHNIQUES

History of Tamarisk Control

The first widespread introductions of tamarisk began in the 1800s (Chew 2013), though a few authors say that introduction occurred as far back as the mid-seventeenth century by Spanish settlers and travelers (Hefley 1937, Tellman 2002a). By the 1830s, many prominent US nurseries frequently sold cultivars of several *Tamarix* L. species, including *T. gallica* L., and *T. chinensis* Lour. (Tellman 2002a). In 1870, the United States Department of Agriculture (USDA) released *T. pentandra* (syn. *T. chinensis*) for sale to the public following earlier plantings at the National Arboretum in Washington, DC (Robinson 1952). Into the 1930s tamarisk was still considered by many to have beneficial purposes, particularly for large-scale plantings to enhance erosion control, and in sedimentation projects that were considered necessary at the time to support burgeoning agricultural operations in the region (Chew 2009). For example, after observing that the presence of dense tamarisk in the floodplain of Lake McMillan had prevented silting of the reservoir by slowing inlet flow, Dr. T. Taylor of the University of Texas, Austin, suggested that tamarisk should be used similarly at other reservoirs (Taylor 1930). At the same time, however, others were beginning to label tamarisk as a problematic plant (Taylor 1930).

The oldest cases of ornamental plantings escaping cultivation were in 1880 and 1897, in Utah and Texas, respectively (Tellman 2002a, Shafroth et al. 2005). Not until several decades later, though, were naturalized populations widespread, such as the dense infestations along the beaches and banks of the Pecos River in Texas that were cited by Texas Board of Water engineers as having reduced river flows by as much as fivefold (Taylor 1930). Farmers northwest of Barstow, Texas, began abandoning densely infested irrigation canals, saying that it was cheaper to construct an entirely new canal than attempt to clean the infested ones (Taylor

1930). Dr. Taylor expressed the conundrum posed by tamarisk in the 1930s: "In thirty years the tamarisk in the Pecos Valley has spread from one lone tree near Roswell to a rather heavy growth along the river from Roswell to Barstow. The tamarisk can be a blessing to the reservoir, but it can be a troublesome factor to the canal (Taylor 1930)."

In addition to responding to public pressure to provide water for expanding agriculture in the Southwest, New Deal work projects during the Great Depression brought a renewed focus on water issues throughout the southwest. Dam and reservoir construction that had begun at the turn of the twentieth century progressed unabated, altering natural flow regimes throughout regional river systems (Stromberg and Chew 2002, Chew 2009). New, artificial flow regimes in southwestern waterways contributed to an overall lack of management of, and disinterest in, rapidly expanding tamarisk infestations (Dudley et al. 2000, Stromberg and Chew 2002, Tellman 2002b). For example, along a 170-mile stretch of the Brazos River in north-central Texas, an analysis of historical aerial photos showed that tamarisk acreage increased 52% between 1940 and 1969 (Busby and Schuster 1973). Up until the 1960s, tamarisk was still being planted in the region (West and Nabhan 2002).

Concerted efforts to control tamarisk appear to have begun after 1942, when federal irrigation engineers in the Pecos River Joint Investigation first formally accused the species, correctly or not, of being a disproportionate consumer of water and therefore a target for management (Robinson 1952, Chew 2013). By 1948, United States Bureau of Reclamation (BOR) staff were treating hundreds of acres of tamarisk in the delta of the McMillan Reservoir using 2,4-D herbicide, at a cost of $4/acre (Subcommittee 1970). About the same time, in a project sponsored by the United States Department of Commerce and the Defense Plant Corporation, United States Geological Survey (USGS) staff attempted to control tamarisk along

a tributary of the Gila River in Arizona using bulldozers and military-issued flamethrowers. Unfortunately, within 12 months the treated trees regrew to be six to eight feet tall (Robinson 1952). Tamarisk management within the Gila River and Salt River drainages of Arizona gathered steam in 1951 following construction of the Gila and Salt River Floodway by the United States Army Corps of Engineers (USACE). In the spring and early summer of that year 40 acres of tamarisk were bulldozed and the resulting debris burned. Then in September 1951 the regrowth was used for the first large-scale trial comparing the effectiveness of ten herbicide treatments (Subcommittee 1970). Foliar applications of 2,4-D and 2,4,5-T at two concentrations were tested, as were cut-stump and basal-bark applications of 2,4-D in diesel fuel at three concentrations.

Despite these early control projects, tamarisk continued to spread, and the scale and scope of programs to control tamarisk grew rapidly as well. Now, there are improved chemical options for killing tamarisk trees and a biological control agent has been successfully used to control tamarisk populations. This chapter presents an overview of current, commonly used management strategies, but also outlines important concepts to consider when planning tamarisk management. While a variety of removal methods are detailed, it is important to acknowledge that none of these alone will successfully or sustainably control tamarisk. As with other invasive plant species, tamarisk control will only be effective over the long term if individual, site-specific management strategies are integrated into a comprehensive plan (Shafroth et al. 2008, Shafroth et al. 2013). Toward this end, we also aim to demonstrate that sustainable tamarisk management can address the preservation of ecosystem functioning and biotic integrity, and also ensure that the economic value and productivity of land is maintained and even enhanced.

Management Options

Flooding

One of the primary reasons tamarisk has become so abundant throughout the southwestern United States is the alteration of flooding patterns and reduced peak flows in natural waterways (Stromberg et al. 2007, Auerbach et al. 2013). It has been shown that restoration of natural flooding regime can suppress tamarisk by promoting native species' growth (Sher et al. 2002, Russo 2013). But many sites where tamarisk is problematic are no longer hydrologically connected to a flooding source that would allow this treatment option to be feasible. That said, inundation of mature tamarisk trees seems only to kill them reliably with at least two to three years of flooding (Wiedemann and Cross 1979). On the other hand, flooding of infested sites during seed germination, or during the first year of seedling growth, has been found to cause consistently high (> 90%) tamarisk mortality (Smith and Kadlec 1983, Gladwin and Roelle 1998, Sprenger et al. 2001).

Prescribed fire

A small but growing literature is emerging regarding the effects of fire on tamarisk (Drus 2013). With regard to fire as a management tool, it appears that prescribed burns alone do not effectively control tamarisk, as the species is well adapted to all but the hottest and longest fires (Busch 1995, Dudley et al. 2000, Racher and Britton 2003, Racher 2009). Following fires tamarisk plants will re-sprout vigorously from unaffected roots and can regrow up to several meters in one year (Brock 1994). There are examples of successful prescribed fires for the control of tamarisk, but only in combination with herbicide treatments. For example, Harms and Hiebert (2006) documented a 95% reduction in tamarisk foliar cover at thirty sites in

southwestern states where prescribed burns were followed by herbicide application. Racher (2009) found that although burning mature stands did substantially reduce canopy cover, mortality was very inconsistent and generally low, especially for sites with a history of burns. It has been proposed that prescribed burns are probably best used for biomass or debris removal rather than as a primary control method (Racher and Britton 2003).

Biological control

Biological control in the form of defoliation by leaf beetles in the genus *Diorhabda* is the newest tool for tamarisk management and will likely be increasingly significant in years to come (Bean et al. 2013). It takes several years of defoliation by *Diorhabda* species to kill mature tamarisk trees, and some estimates suggest that it would take at least a decade to ensure 75%–80% mortality in a given stand (Dudley et al. 2000). Damage to tamarisk trees by leaf beetle defoliation tends to be fatal mostly on younger trees and seedlings that do not have large enough root systems to recover from the repeated stress of herbivory (Dudley et al. 2000).

While this rate of impact is slower compared to chemical or mechanical control, it also presents an opportunity to combine biological control with other methods. For example, Brooks et al. (2008) found that tamarisk trees that had been first weakened physiologically by repeated *Diorhabda* beetle defoliation were more likely to be killed by prescribed burns than trees that experienced no herbivory. Similarly, I have observed that establishment of *Diorhabda carinulata* Desbrochers occurs more rapidly on trees regrowing following mechanical removal of aboveground biomass, and that the effects of herbivory are more immediate. Presumably the overall physiological effect of beetle herbivory on these trees, which are already stressed from biomass removal, would also be more severe. This suggests that the integration of aboveground

biomass removal and biological-control releases timed to curtail regrowth could be promising for long-term tamarisk management.

Mechanical treatments

Many mechanical strategies to reduce or remove aboveground tamarisk biomass have been tried over the years, including bulldozing, shredding/mulching (brush mowing), chaining, disking, grubbing, knifing, roller chopping, and root plowing (Subcommittee 1970, Brock 1994, Smith et al. 2002). However, most mechanical strategies that only remove the aboveground stems of mature trees do not kill the plants and resprouting can be vigorous. There is some evidence that young (first-year) tamarisk seedlings are particularly sensitive to mechanical removal, even disking at a shallow depth of 12.5 cm (Smith et al. 2002). For the same reasons that trees respond positively to fire, mature tamarisk plants tend to regrow vigorously after any aboveground biomass removal, unless there is also damage or extraction of the root crown and lateral root system (McDaniel and Taylor 2003b).

Killing tamarisk trees using only mechanical approaches requires the physical removal of trunks, plant crowns, and lateral roots (Hart 2009). Track hoes or excavators equipped with specialized grubbing attachments are effective at removing most above- and belowground biomass, but this must be followed by the raking and sifting of the soil to remove lateral roots. The latter can be done with root rakes mounted on bulldozers (Russo 2013). Other equipment used to clear large areas of aboveground tamarisk biomass are site-preparation tractors or skid steers with front-mounted forestry mulching attachments (Nissen et al. 2010). Smaller versions of these mulching attachments can also be mounted onto the swing arms of excavators, allowing for more selective removal of tamarisk trees, which is useful if native woody species are growing

among the tamarisk. Complete removal of root crowns and lateral roots can cause fairly severe soil disturbances that may require mediation before re-vegetation (Scifres 1980, Sher et al. 2010).

Whole plant removal (e.g., by using an excavator with a thumbed bucket) can successfully kill tamarisk if conditions allow for removal of the actual root mass (Nissen et al. 2010). If excavation is planned for winter months it is important to ensure that the soil is not frozen because this can prevent complete removal of the root crown and enable resprouting (Scifres 1980). With any mechanical removal it is particularly important to plan for follow-up monitoring and any necessary re-treatments because the success of many long-term tamarisk management programs relying primarily on mechanical treatments is incumbent on annually re-treating sites (Brock 1994).

Tamarisk regeneration following removal of top growth alone follows a certain pattern: (sometimes dramatically) increased stem densities; shorter, more erect stems with smaller diameters; and a full, bushy canopy. Tamarisk regrowing after aboveground mechanical treatments often has increased vigor, with plants producing more stems per unit area than is typical of undisturbed plants. The shorter-stature tamarisk regrowth allows for more efficient application of herbicides to individual plants, with reduced risk to other nearby vegetation. Alternatively, prescribed burning can be more effectively used as a follow-up treatment at sites where mature tamarisk is first cleared mechanically and the trees windrowed (Taylor and McDaniel 1998, Racher 2009).

Chemical Treatments

Over the past half century many herbicides have been used in attempts to control tamarisk, though chemical control was not reliably effective until the introduction of imazapyr in the mid-1980s. These included various formulations of 2,4-D; 2,4,5-T (silvex); picloram; dicamba; triclopyr; glyphosate; and finally imazapyr (Scifres 1980, Brock 1994). The only products that are environmentally compatible with application in riparian environments that tamarisk frequently infests are 2,4-D, glyphosate, triclopyr, and imazapyr. Until the introduction of triclopyr and imazapyr the most widely used product was 2,4-D, which did not consistently control tamarisk trees (Kerpez 1987, Brock 1994). Glyphosate is still commonly used but is not very effective when used alone (Fick and Geyer 2010). For example, Duncan (2010) found that foliar applications of glyphosate resulted in only 32% mortality, whereas combinations of glyphosate and imazapyr resulted in 99% mortality. Mixtures of glyphosate and imazapyr are very common and reduce the required amount of the more expensive and environmentally persistent imazapyr (Carpenter 1998, Duncan and McDaniel 1998).

There are various methods of herbicide application to tamarisk, but generally the methods are targeted at foliage ("foliar" treatments), the bark of an intact tree ("basal" bark refers to applications that target the bottom 45–71 cm of a tree's bark), or a freshly cut stump surface. The part of the tree targeted by a given herbicide is important to consider because this determines what time of year treatments can be made (e.g., foliar treatments can only be done when trees have leaves) and can influence the difficulty and cost of treatments. Generally, herbicide applications that target the bark or cut surfaces are more expensive and take longer than foliar treatments because they require the applicator to directly apply the product to individual trees. However, these types of applications greatly reduce negative off-target impacts to desirable

vegetation. All herbicides currently used on tamarisk are systemic, which means that once applied to the foliage, a stump, or after penetrating the tree's bark, the chemicals move (translocate) within the plants' vascular system to the roots.

Imazapyr was registered by the United States Environmental Protection Agency (EPA) for non-crop use in July 1984 and first sold shortly thereafter (EPA 2006). This herbicide belongs to a chemical family called imidazolinones that kill plants by inhibiting the synthesis of an enzyme critical to the production of branched-chain amino acids and ultimately proteins (Shaner and O'Connor 1991). Imazapyr has relatively long-lived residual soil activity (25–142 days) and once in the soil it is broken down mostly by microbial metabolism (Senseman 2007). This means that it can persist in the soil at levels toxic to plants for several months to a year after it is applied and consequently inhibits establishment of other plants, both unwanted and desirable (Sher et al. 2010). The duration of effective weed control from imazapyr soil residues depends on factors that affect soil microbial activity (e.g. soil temperatures and moisture), as well as soil organic matter content and pH levels (Vizantinopoulos and Lolos 1994, Nissen et al. 2010). On the other hand, imazapyr is very water soluble, and will degrade quickly (2–3 days) in water when exposed to sunlight (Mallipudi et al. 1991). Most products containing imazapyr are labeled primarily for use as foliar applications (e.g., Habitat®, Arsenal®, Arsenal® Powerline™), exceptions include Chopper® Generation II, which is formulated for dormant season basal bark applications.

Triclopyr was introduced in the early 1970s and is a synthetic auxin herbicide that mimics the physiological effects of a natural hormone ubiquitous in plants (Howard et al. 1983). At naturally occurring concentrations these hormones are essential to proper plant functioning, but at high concentrations the compounds disrupt plant growth and cause death in susceptible

broadleaved plants (Nissen et al. 2010). Commercial herbicides that contain triclopyr generally have relatively limited soil residual activity (10–46 days) and like imazapyr are rapidly degraded (less than 48 hours) when in water and exposed to sunlight (Woodburn et al. 1993, Senseman 2007). Triclopyr can be formulated for basal bark and cut-stump applications (e.g. Remedy Ultra™, Garlon® 4 Ultra™, Pathfinder II™) or can be formulated for cut surface applications (Garlon® 3A).

Imazapyr is the most consistently effective choice for application to tamarisk by aircraft in aerial applications (Duncan and McDaniel 1998, Nissen et al. 2010). Both fixed-wing aircraft and helicopters are used to aerially treat tamarisk, though fixed-wing planes are only advantageous for treating large, monotypic stands (McDaniel and Taylor 2003a). Helicopters are better for aerial applications because of their slower air speeds, closeness to the ground, and ability to use higher application volumes (Hart et al. 2005). Furthermore, their maneuverability allows for treatment of variably shaped and sized tree stands, and avoidance of desirable natives such as *Populus* and *Salix* species (McDaniel and Taylor 2003a, Nissen et al. 2010).

Ground-based, individual plant herbicide applications can take many forms, but are either foliar and applied during the growing season or targeted to the bark or cut surfaces during the winter when trees are dormant (Hart 2009). Individual plant treatments with imazapyr can be very effective (more than 95% control), will reduce the amount of herbicide needed to treat a site, and minimize the potential for overspray into desirable vegetation (Duncan and McDaniel 1998, Nissen et al. 2010). Application timing is important for all foliar tamarisk treatments, which typically occur in August and September before plants begin senescing and shunting foliar resources to their root systems (Duncan and McDaniel 1998, McDaniel and Taylor 2003a).

Low-volume basal bark or cut-surface applications using higher concentrations of either herbicide can also be effective at controlling tamarisk and such applications can be carried out year round (Howard et al. 1983, Parker and Williamson 2000). This type of application is most effective on smaller trees partly because young trees have smoother, thinner bark that is more easily penetrated by herbicides (Parker and Williamson 2000). Applicators can use backpack sprayers, all terrain vehicles (ATVs), or horse-mounted sprayers. The added value of following prescribed burns of mature tamarisk trees with individual plant chemical control has been demonstrated in a few instances (Fox 2001, Racher and Britton 2003). Alternatively, McDaniel and Taylor (2003a) found more than 93% long-term control with an aerial application of imazapyr followed three years later by a summer burn.

It is important when selecting any herbicidal option for controlling tamarisk to consider the habitat in which applications will be taking place and specifically how close to any perennial stream the application will occur (Carpenter 1998). Also, while it is often desirable to remove dead materials after trees have been killed using chemical controls, it is critical that any follow up treatments be delayed for two years (Duncan and McDaniel 1998, Hart 2009). This period allows for complete translocation of a systemic herbicide throughout the entire root system of a tamarisk tree and will ensure that the entire plant is killed. Dead or dying trees that are disturbed before herbicides are completely translocated will often resprout from surviving root fragments.

It is important to understand the trade-offs when using herbicides to control tamarisk. Imazapyr, glyphosate, and other nonselective chemistries will kill nearly any vegetation they contact, the exception being some weed species such as *Bassia scoparia* (kochia) that have developed resistance to these herbicides (Shafroth et al. 2013). Kochia is a widespread exotic weed in crop fields where herbicides with the same mode of action as imazapyr are frequently

used. The selection pressure applied in these fields led to the development of resistance by kochia, which has since spread widely into natural areas (Primiani et al. 1990). Lastly, the persistence of imazapyr in the soil can pose a challenge for re-vegetation (Sher et al. 2010). Herbicides such as triclopyr and other synthetic auxinic chemicals, which only kill broad-leaved plants, will be more selective and only affect sensitive plants (Howard et al. 1983). However, we should be concerned that shifts in functional groups of native plant communities have been documented in rangelands due to the repeated use of selective herbicides (Pearson and Ortega 2009, Ortega and Pearson 2011).

Costs, Impacts and Trade-Offs

While there are many factors that influence the selection of tamarisk control tools - including characteristics of the tamarisk infestation, site constraints, understory vegetation, and project goals - cost is usually the most important determinant of which strategy will be feasible (Shafroth et al. 2008). Management of any invasive species is inherently expensive, but management of woody invaders such as tamarisk is even more costly because it requires two equally important phases: control (killing trees) and biomass management (removing and disposing or reusing branches, trunks and stumps). Biomass management options can include burning, raking into piles, chipping to create mulch, conversion into wood pellets for stoves, and even use as a fuel for downdraft gasification (Sher et al. 2010, Nielsen et al. 2011). Regardless of the strategy biomass management can be more costly than the control phase itself (Taylor and McDaniel 2004, Coalition 2009). Although occasionally it may not be possible or desirable to remove standing dead trees in most cases such efforts will promote desirable replacement vegetation (Sher et al. 2010).

There are two scales of tamarisk management projects: a larger, watershed or drainage-wide scale; and one that is more localized and site-specific. Many factors determine the scale of a management effort, but cost will likely be the most important and will particularly influence the choice of management tools. For example, large projects usually use lower cost per area, more highly efficient management tools such as biological control or aerial herbicide applications. There are a greater number of options available to smaller projects such as removing a patch of trees from an urban riverside park. At these higher value sites more selective and intensive strategies can justifiably be used to completely remove trees and actively re-establish desirable native plant communities. Such methods can include targeted removal by excavators or cut stump methods that reduce exposure of desirable vegetation to herbicide.

Available funds will often define the area that can be treated and how management will be carried out. There are implicit trade-offs between the financial costs of a management project, its ecological impacts or benefits, and the control tools that are chosen. The goal of tamarisk management is frequently to restore some degree of pre-invasion functioning to a site, but the impacts resulting from removal or control may delay or prevent benefits from ecosystem restoration. For example, while aerial applications of an herbicide such as imazapyr are very cost effective, off-target soil residues mean that relatively expensive active re-vegetation efforts are likely necessary. Conversely, targeted individual tree removal and control strategies - which are generally much more costly in the short term - may enable the preservation of existing understory plant communities and ecosystem functioning, resulting in intensive restoration activities not being necessary.

There have been many reports summarizing the relative costs of many of the management options presented here (Barz et al. 2009, Coalition 2009), and expenses such as fuel

and herbicides vary year to year, almost always increasing. Herbicidal control of tamarisk can no longer be accomplished for $4/acre as it was in 1948, but the cost for brand name formulations of imazapyr has fallen considerably over even the past five years (Barz et al. 2009).

Integrated Tamarisk and Ecosystem Management

It is clear that single-method strategies for managing tamarisk are simply not effective over time. It is vital when designing tamarisk management programs to use the suite of tools that will have the highest chance of both maintaining tamarisk control and conserving the inherent ecological resilience of a treated site (Jorgensen 1996, Pearson and Ortega 2009). Selecting appropriate strategies and implementing them in the proper sequence are keys to long-term success. The use of complementary or even synergistic control methods can actually accelerate the rate of natural ecosystem recovery (Masters and Nissen 1998). To ensure the long-term success of management projects it is essential that monitoring and maintenance are planned for and carried out, meaning in particular that secondary invaders or surviving tamarisk plants are actively controlled (Shafroth et al. 2008).

For a truly sustainable approach to managing tamarisk, we must recognize that (1) management method has a direct impact on the capacity of sites to recover, and (2) that controlling tamarisk is only a small part of the comprehensive program needed to attempt management of an ecosystem (Shafroth et al. 2013). First, management of invasive species begins when a land manager decides that a certain species has a disproportionately negative impact on a site. The techniques that are chosen to remove the targeted species and the order in which they are implemented directly affect ecosystem recovery, both from the management action itself and the sudden absence of the previously dominant plant species (Taylor and

McDaniel 2004, Vincent et al. 2009). Strategies that remove a dominant plant such as tamarisk may indirectly facilitate a secondary invasion of an equally noxious species (Pearson and Ortega 2009), or may cause other negative outcomes such as severe erosion during flooding (Vincent et al. 2009).

Second, invasive species removal alone will not necessarily result in a positive outcome if other underlying problems with ecosystem processes (e.g. modified hydrologic regimes or improperly managed cattle grazing) are not also addressed. The success of tamarisk removal and site restoration has been closely tied to such larger-scale changes that enhance the capacity for sites to self-repair (Stromberg and Chew 2002, Taylor and McDaniel 2004). For example, an important part of the successful management of tamarisk infestations along the Mojave River at Barstow Resource Area in California was the installation of a "riparian management fence" to exclude both grazing and off-road recreational vehicles (Lovich et al. 1994, Chavez 1996). This allowed the understory plant community to regenerate without being trampled and prevented erosion and soil disturbance associated with heavy use. In this context it is important for managers to consider whether larger-scale modifications in watershed hydrology and other ecosystem processes are possible to ensure that tamarisk removal and site restoration projects will be successful (Stromberg and Chew 2002). Sites that can be restored to natural ecosystem processes, have a high re-vegetation capacity or especially desirable understory plant community should be prioritized as especially strong candidates for tamarisk management (Taylor and McDaniel 2004, Parker et al. 2005).

Conclusions

Some methods used today to control tamarisk are very similar to those that were first used eighty years ago, such as mechanical extraction. Others are more recent, most notably the availability of selective and less environmentally harmful herbicides. The use of *Diorhabda* spp. beetles as biological control agents for tamarisk, however, is perhaps the most advanced and modern means we have to manage tamarisk. Arguably, what has changed the most over the years we have been tackling tamarisk - and what will hopefully continue to improve - is our understanding of ecosystems and how our management of individual components affects others. Similarly, our history of managing tamarisk has taught us many lessons, principally the importance of using integrated suites of tools rather than relying on single methods. We have also learned through experience the value of using community volunteers and interagency collaborations, because invasive species such as tamarisk are an issue that we must deal with collectively.

The past has taught us a lot about how the ways in which we manage a plant such as tamarisk can affect the condition, economic value and long-term ecological vitality of invaded sites. Recent research in particular allows us to better understand how ecosystems and organisms such as secondary invaders will respond to specific management tools. By developing comprehensive and adaptive management plans, using integrated and intentional suites of management tools and learning lessons from past efforts we might be able to begin pushing back the tide of tamarisk that has quietly crept across the southwest for almost two hundred years.

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CHAPTER 2. IMAZAPYR AND TRICLOPYR SOIL DEGRADATION AND RESTORATION SPECIES SENSITIVITY

Summary

Herbicides are frequently used in natural systems to control unwanted plants, but non-target impacts from initial applications and persistent soil residues can result in unintended ecosystem effects. Imazapyr and triclopyr are two herbicides that are widely used in non-crop areas to manage perennial weeds, and especially woody species. We used an in vitro study and HPLC analysis to quantify degradation rates for the herbicides in six Colorado soils and then determined the relative sensitivity of important restoration plant species to the two herbicides in a field dose response study. Exponential decay models estimated imazapyr half-lives (t_{50}) for two soils at 51 and 76 days, and triclopyr half-lives (t_{50}) for all soils averaged 7 days. In field dose response studies *Glycyrrhiza lepidota* was the only species to demonstrate sensitivity to triclopyr. *Atriplex canescens*, *Elymus canadensis* and *Sporobolus airoides* were the most sensitive to typical imazapyr residues. Also, fecundity in *S. airoides* and *Bouteloua curtipendula* were negatively impacted by the highest rate of both triclopyr (3.92 kg ai ha^{-1}) and imazapyr (0.28 kg ai ha^{-1}). In this study microbial degradation of triclopyr occurred 6.5 times more rapidly than imazapyr under different soil textures and the tolerance of common rangeland and riparian plant species to triclopyr was also much greater. These results will allow land managers to minimize non-target effects from herbicides used to control invasive species in natural areas and promote ecosystem recovery.

Introduction

Herbicides can be useful tools for managing weedy or invasive plant species in natural areas, but their use can also result in unintended non-target impacts. In particular, when herbicides are used to control abundant invasive species in natural areas that will subsequently need active re-vegetation, the biological activity and soil persistence of herbicides can negatively affect site rehabilitation (Pearson and Ortega 2009, Sher et al. 2010). Selecting herbicides that effectively control targeted plant species and also have minimal environmental consequences can facilitate ecological restoration efforts.

Methods for removing and controlling invasive plant species include mechanical, biological, chemical, and integrated pest management (IPM) strategies (Hobbs and Humphries 1995, Radosevich et al. 2009). There is also a growing awareness that the weed control method and how carefully and effectively it is implemented can affect both invasive plant control and habitat restoration success (Harms and Hiebert 2006, Flory and Clay 2009). However, very few studies have looked directly at the effects that herbicide applications and subsequent soil residues can have on restoration success, and even fewer have done so under field conditions (Kaeser and Kirkman 2010, Ortega and Pearson 2011). Therefore, this study sought to investigate how commonly used herbicides would influence the establishment and growth of native plant species that might be used for active re-vegetation. These trials were part of a larger project to evaluate the environmental impacts of tamarisk (saltcedar, *Tamarix* L.) management strategies in a Colorado watershed. Plant species included in this study are native to the region and frequently used in riparian restoration projects (Lindauer 1983). We examined the impacts of two herbicides (imazapyr and triclopyr) that are widely used in natural areas to control shrubby or woody invasive plants (Douglass et al. 2013). Imazapyr and triclopyr are commonly used herbicidal

compounds because they are very effective, are non-toxic to a range of micro- and macro-fauna, and under most conditions do not remain biologically active in the environment for long periods of time (Senseman 2007). These characteristics allow the two herbicides to be chemically formulated for use in a variety of habitats, including uplands, grasslands, riparian and aquatic sites (Brock 1994, Douglass et al. 2013).

Imazapyr is a very broad spectrum herbicide, and products containing this compound can be applied using several different methods and timings. In an aqueous solution (i.e. applied to or near standing water) imazapyr photo-degrades within hours after application and so it is used frequently to control invasive plants in wetter habitats (Mallipudi et al. 1991). However, imazapyr residues can be long-lived in the soil (reported soil half-life (t_{50}) = 25-142 days) depending on soil and environmental conditions (Senseman 2007). On the other hand, triclopyr is a more selective compound and is generally only phytotoxic to dicotyledonous plant species. Furthermore, the compound degrades rapidly in the soil after application (t_{50} = 10-46 days) (Senseman 2007).

Soil degradation of both herbicides is known to occur primarily via microbial activity and so environmental characteristics (e.g. temperature and moisture) that promote soil biological activity generally increase degradation rates for both chemistries (Johnson et al. 1995b, McDowell et al. 1997, Newton et al. 2008). Soil moisture levels in particular have a very strong influence on aerobic microbial activity, which is maximized when soil conditions permit an optimal balance of water and oxygen (Skopp et al. 1990, Conant et al. 2004). Edaphic properties that increase adsorption of organic molecules (i.e. herbicides) to soil particles will also reduce an herbicides availability to biological degradation; however, this process also immobilizes herbicide molecules and typically reduces their overall toxicity (Gevao et al. 2000). While

properties such as soil organic matter content will increase adsorption of either compound triclopyr has a greater relative sorption potential than imazapyr, which is more prone to leaching out of the upper horizons of soils where microbial activity is most prominent (Johnson et al. 1995b, Borjesson et al. 2004).

In these studies the aim was to first determine the soil degradation rates for imazapyr and triclopyr in Colorado soils from sites where invasive plant species (*Tamarix* spp. specifically) management was occurring. Second, in field studies the dose response relationship of relevant restoration plant species to appropriate rates of the herbicides were evaluated. Combining data from the degradation study and the field dose response trials the relative sensitivities of important native plant species to imazapyr and triclopyr were established.

Materials and Methods

Laboratory simulation of herbicide soil degradation

Soil was collected from six sites in eastern Colorado (Tables 2.1a & 2.1b), two in the upper Front Range region that were used for the plant dose response studies described below and four in the Arkansas River Watershed where tamarisk management was being conducted. Several liters of soil were collected from the upper 10 cm at each site in a location that had never received herbicide treatment. Soils were air dried for 72 hours and sifted through a 2 mm sieve. A portion was removed and sent to AgSource Laboratories (Lincoln, NE) for chemical and textural analysis and the remainder treated using a handheld spray bottle containing an aqueous herbicide solution. The initial concentrations of treated soils was 1 mg active ingredient (a.i.) kg^{-1} soil each imazaypr and triclopyr. Treatment solutions were made with pure (99.8%) analytical standards (imazapyr: Sigma-Aldrich Inc., St. Louis, MO, Lot No. 7151X, and triclopyr TEA salt:

Sigma-Aldrich Inc., St. Louis, MO, Lot No. 2095X). Water content for the six soils was calculated prior to the experiment and the aqueous volume of the treatment solutions calibrated to bring each soil to 75% of field capacity. The treated soil was transferred to a soil tumbler and homogenized for thirty minutes.

Twenty-four 20 g samples of each soil were weighed into 50 mL polypropylene centrifuge tubes, and held in an incubator at ambient temperature (23-25 °C) and humidity (65-70% RH). At 0, 3.5, 7, 14, 28, 56, 112 and 160 days after treatment (DAT) three tubes containing soil from each of the sites were removed and stored (-20 °C) until analysis. Every other week during the experiment tubes were vigorously shaken and the lid removed momentarily to allow for air exchange. Soil moisture was monitored throughout the experiment and soils brought back to initial starting moisture content when necessary. Each set of time points, soils and herbicides was replicated twice.

Quantification of herbicide concentrations using high performance liquid chromatography (HPLC) with UV detection

To analyze imazapyr and triclopyr residues from soil samples a 5 g sub-sample was transferred to a clean 50 mL polypropylene tube and extracted with 10 mL of deionized water. Samples were shaken for 2 hours, and then centrifuged for ten minutes (4,000 rev min^{-1}; Sorvall Legend XT Centrifuge, Thermo Fisher Scientific, Waltham, MA). A 1 mL aliquot was collected from the supernatant, placed in a 0.45 μm Spin-X® centrifuge tube filter (Costar®, Corning Inc., Corning, NY) and centrifuged for ten minutes (13,000 rev min^{-1}; Sorvall Legend Micro 21 Centrifuge, Thermo Fisher Scientific, Waltham, MA). The resulting filtered liquid was transferred to an HPLC auto sampler vial with a limited volume (400 μL) insert (National

Scientific, Rockwood, TN) for analysis. Quality control (QC) soil samples were prepared for each site by fortifying soil at known concentrations for each compound within the concentration range of the samples. Each analytical run included QC samples prepared at the time of sample preparation.

Samples were analyzed on an Hitachi D-7000 HPLC system with an in line dual UV detector set at 250 nm (imazapyr) and 295 nm (triclopyr). Sample injection volume was 75 µL and a Zorbax Rx C8 4.8 mm X 250 mm column was used (Agilent Technologies, Santa Clara, CA). Mobile phase A was 10% HPLC-grade acetonitrile:90% water and mobile phase B 80% acetonitrile:20% water, both with 0.5% phosphoric acid added. A gradient ranging from 100% A to 100% B in ten minutes followed by a 5 minute re-equilibration time before the next injection was utilized to elute the compounds of interest. Mobile phase flow rate was 1.2 mL min^{-1} at ambient room temperature. Calibration curves for each herbicide were independently obtained by plotting peak areas against analytical standard concentrations for a range of diluted standards (0.005 – 2 µg ml^{-1}). The relationship between chromatographic peak area and herbicide concentration over the range of standards was explained by linear regression (mean $R^2 > 0.99$).

Differences in soil moisture between sites were accounted for by calculating the gravimetric water content for each sample using a separate sub-sample collected and weighed prior to HPLC analysis. Sub-samples used for water content analysis were dried at 110 °C for 24 hours and then re-weighed. Results were used to convert volume-based herbicide concentration data derived from the HPLC analysis into data based on dry soil weight. Mean retention time using this methodology was 7.45 minutes (coefficient of variation (CV) = 4.56%) for imazapyr and 11.50 minutes (CV = 0.59%) for triclopyr in 496 samples. Recovery of imazapyr in 48 fortified soil samples averaged 91.26% (CV = 21.49%) and mean triclopyr recovery was 86.48%

(CV = 13.53%). The limit of quantification (LOQ) was 0.01 mg kg^{-1} in the soils tested and the limit of detection (LOD) was 0.005 mg kg^{-1}.

Quantifying plant species sensitivity to imazapyr and triclopyr soil residues

The following study was replicated at the CSU Agricultural Research, Development and Education Center (AR) and the Horticultural Research Center (HO) (Table 2.1a). The following species (three forbs and six grasses) were selected for use in this study (Table 2.2): *Atriplex canescens* (Pursh) Nutt.; *Bouteloua curtipendula* (Michx.) Torr.; *Elymus canadensis* L.; *E. elymoides* (Raf.) Swezey; *E. trachycaulus* (Link) Gould ex Shinner; *Glycyrrhiza lepidota* Pursh; *Helianthus annuus* L.; *Pascopyrum smithii* (Rydb.) A. Love; and *Sporobolus airoides* (Torr.) Torr. A split-split plot experimental design was utilized with the randomization of plant species planting row restricted by functional group ('forb' or 'grass'). Plant species were seeded north to south across 3 X 23 m plots, and herbicide treatments made west to east within each plot.

The following serial dilutions of imazapyr (Habitat®, 28.7% isopropylamine salt (BASF Corp., Florham Park, NJ) and triclopyr (Garlon 4 Ultra®, 60.45% butoxyethyl ester (Dow Agro Sciences LLC, Indianapolis, IN)) herbicides were applied to plots: 1X, 0.5X, 0.25X, 0.125X, 0.0625X, 0.03125X and 0.015625X (Table 2.3). For imazapyr the starting concentration (1 X) was 0.28 kg ai ha^{-1} and triclopyr dilutions began at 3.92 kg ai ha^{-1}. Herbicide applications were made on 1 June (HO) and 6 June (AR) 2011, using a CO_2-powered backpack sprayer calibrated to deliver 141 liter ha^{-1} volume. At the time of application temperature was 24.7 °C and 29.2 °C, respectively, relative humidity was 21.5% and 15% and wind speeds averaged 4-8 km hr^{-1}. Herbicide applications were made to bare soil and within twenty-four hours of applications the selected plant species were seeded. This experimental design was intended to replicate two

scenarios: 1) passive re-vegetation of natural areas via seed dispersal following herbicide treatments to tamarisk; and 2) active restoration in which re-vegetation seeding would occur on or into herbicide-treated soils.

A modified seed drill was used to plant seeds to a depth of 1 cm in two adjacent rows. Forbs were seeded at a density of 0.39 seeds cm^{-1} and grasses at a density of 1.18 seeds cm^{-1}. An overhead sprinkler supplied supplemental irrigation during the growing season. Total moisture at AR over the study period was 44.5 cm and 49.5 cm at HO. Establishment and growth of the seeded plant species was measured during the growing season in the middle 2.5 m of the 3 m-wide plot. Plant density (stems m^{-2}), plant height (cm) and frequency (%) data were collected for all species. Fecundity (seedheads $plant^{-1}$) was measured for three grasses (*S. airoides*, *B. curtipendula* and *E. trachycaulus*) that flowered. Percent cover was estimated for all species except *Helianthus annuus* L. using a 25 X 100 cm sampling frame marked into 10 cm segments. Frequency (%) was measured as the number of twenty-five 10 cm linear segments in which stems of the given plant species occurred. Finally, aboveground plant biomass was collected 24-25 October (HO) and 1 November (AR), with the exception of *H. annuus* biomass, which was collected two weeks earlier. All biomass samples were oven dried at 70 °C for seven days and then weighed.

Statistical analyses

Model residual data were tested for normality and homoscedasticity prior to all analyses, and the appropriate data transformed if these assumptions were violated. To quantify soil degradation the mean herbicide residue concentrations of each of the three site sub-samples were calculated and re-expressed as percent of initial (0 DAT) values. The results from the two

replicates were then analyzed using regression models. Models of herbicide soil degradation were tested for goodness of fit using the corrected Akaike information criteria (AICc) and AICc weight values (Burnham and Anderson 2002). Regression models were fit separately for the two herbicides to data from each of the six sites. Selected regression models were used to calculate the t_{50} value (the number of days to reach a 50% reduction in herbicide concentration) for imazapyr and triclopyr in the six soils.

The growth measurements from the plant species sensitivity field studies were transformed prior to analysis. Plant density data were square root transformed and biomass, stem height and fecundity data were natural log-transformed with the addition of a small non-zero constant to each value to correct for the presence of zero values in each dataset. Frequency and percent cover were analyzed as proportions and were transformed using the logit ($\log(y/[1-y])$) function (Warton and Hui 2011). For the logit transformed data the minimum non-zero value was added to the numerator and denominator of the function to correct for the bias of sample proportions equal to 0 and 1 (Warton and Hui 2011). Transformations allowed for data from the two sites (AR & HO) to be pooled for analysis, so that for each species X herbicide X dilution treatment there were 8 replicates.

Biomass data were analyzed using Dunnett's Method to compare the values of each plant species and treatment combination against their respective untreated controls and identify the lowest herbicide dilution at which growth was significantly different (at P-value < 0.05). To compare and contrast the relative sensitivity between plant species the log-transformed biomass data were further analyzed using log-logistic dose response models (Seefeldt et al. 1995). From the prediction curves the models generated the herbicide rate at which biomass of each species was reduced 50% (GR_{50}) was estimated. Finally, these GR_{50} values were used in conjunction

with the exponential decay models from herbicide degradation studies to estimate the time period needed for herbicide residues to degrade to the predicted GR_{50} concentration (i.e. T_{50}) in soils from the two study sites where field dose response studies took place. JMP (ver. 10.0.1, SAS Institute, Cary, NC) software was used for all statistical analyses.

Results

Herbicide degradation

Both imazapyr and triclopyr soil degradation were best explained (mean $AICc_{Imazapyr}$ = 116.43; mean $AICc_{Triclopyr}$ = 105.76) by the following exponential decay model:

$$y = a + b*(exp^{c*DAT)}) \quad \text{Eq. 1}$$

where a = minimum asymptote, b = scale, and c = growth rate (Table A.1). Imazapyr degradation rates for four soils (AR, CC, FL & LJ) were too slow to be predicted in the timeframe of this simulation study (Table 2.4). Average triclopyr degradation occurred much faster (mean t_{50} = 8.47 days) than did measurable imazapyr degradation (mean t_{50} = 63.66 days (Table 2.4)). Imazapyr degradation occurred more rapidly in soils from OR (t_{50} = 50.92 days) then from HO (mean t_{50} = 76.39 days), but confidence intervals (95%) for the degradation rates overlapped. Triclopyr degradation occurred most rapidly in soils from HO and OR (mean t_{50} = 5.02 days), followed by AR and CC (mean t_{50} = 6.53 days). Degradation in soils from FL (mean t_{50} = 10.38 days) and LJ (t_{50} = 17.32 days) occurred significantly ($\alpha = 0.05$) slower than that in other soils.

Although we made an effort to standardize soil moisture levels, linear regression (mean $R^2 = 0.59$) indicated that there was a significant ($P < 0.0001$) decline in mean sample soil moisture over time (Data not shown). Losses over time were highest in soils from AR (β = -0.04), followed by HO, LJ & OR (mean β = -0.03) and CC and FL (β = -0.02). Multivariate

correlation analysis suggested that imazapyr degradation was inversely related to organic matter (R^2 = 0.28, P = 0.0057; Table 2.5) and positively correlated with higher sand content (R^2 = -0.21, P = 0.0401). Triclopyr soil concentrations were not significantly correlated with any of the measured soil parameters.

Plant species sensitivity

Imazapyr applications reduced establishment (frequency, percent cover) of *E. canadensis* roughly 50% at 0.07 kg ai ha^{-1}, and *B. curtipendula* and *H. annuus* an average of 82.5% at the highest dose (Table 2.6a). The herbicide impacted the biomass, density and stem height of all species except *G. lepidota* and *H. annuus*. Biomass was reduced an average of 84% in affected species, density 87% and stem height 67%. *S. airoides* and *B. curtipendula* fecundity was reduced an average of 74% at the highest imazapyr dose (0.28 kg ai ha^{-1}).

At a relatively high rate of 1.96 kg ai ha^{-1} triclopyr reduced *E. canadensis* and *B. curtipendula* establishment 64% and also decreased frequency of two of the tested forbs (*G. lepidota* and *H. annuus*) by an average of 89% (Table 2.6b). The high rates (1.96 & 3.92 kg ai ha^{-1}) of triclopyr decreased density of all plant species except *H. annuus* and *S. airoides* by 83%. Biomass and stem height in a number of species were also impacted, most of all for *G. lepidota*, for which 0.98 kg ai ha^{-1} decreased growth 80%. Similar to imazapyr, at the highest dose applied triclopyr reduced *S. airoides* and *B. curtipendula* fecundity by 87%.

Log-logistic dose response models estimated imazapyr GR_{50} values ranging from 0.05 to 0.16 kg ai ha^{-1} for most plant species (Table 2.7; Table A.2). *E. elymoides*, *H. annuus* and *G. lepidota* biomass were not significantly reduced by any rate of imazapyr used in the study. Likewise, *H. annuus* and *E. elymoides* were completely insensitive to any triclopyr rate used in

the trial and three other species (*A. canescens*, *E. trachycaulus* and *P. smithii*) were very tolerant (mean GR_{50} = 2.68 kg ai ha^{-1}). *Elymus canadensis* was moderately tolerant to triclopyr (mean GR_{50} = 1.66 kg ai ha^{-1}) and *B. curtipendula* and *S. airoides* more sensitive (mean GR_{50} = 0.84 kg ai ha^{-1}). *Glycyrrhiza lepidota* was so sensitive to triclopyr that even the lowest rate applied (0.056 kg ai ha^{-1}) decreased biomass by 57%.

Because of the asymptotic exponential decay model used to fit the degradation data and the relatively short timeframe of the experiment the minimum asymptote of the imazapyr model for the sites (AR and HO) where the field dose response study was carried out was 0.54 kg ai ha^{-1}. Given that the GR_{50} values of the tested plant species ranged from 0.06 to 0.25 kg ai ha^{-1} the original exponential decay models to estimate plant species t_{50} values for imazapyr were not used. Instead a non-asymptotic exponential decay model was used to predict species sensitivity, with a minimal decrease in goodness of fit from AICc = 238.73 (3 parameters) to AICc = 249.47 (2 parameters; Table A.3). Based on this simpler model we determined that the species broke into several imazapyr tolerance groups: *B. curtipendula*, *E. trachycaulus* and *P. smithii* (mean T_{50} = 546 days); *A. canescens*, *E. canadensis* and *S. airoides* (mean T_{50} = 742 days). Tolerance of *E. elymoides*, *H. annuus* and *G. lepidota* were too high to be modeled in this study.

Triclopyr soil degradation was estimated to occur rapidly enough in the soils at AR and HO that the herbicide would not remain at biologically relevant concentrations long enough to negatively impact six of the nine tested species. Triclopyr was estimated to remain in the soil for 3 and 5 days at levels that would reduce *S. airoides* and *B. curtipendula* establishment and growth, respectively. G. *lepidota* was sensitive enough to triclopyr that we could not model effects for this species.

Discussion

Herbicide degradation

Overall, this study found a rapid degradation profile for triclopyr relative to imazapyr, which confirms what is known about the two herbicides (Senseman 2007). Imazapyr degradation occurred relatively quickly in soils from Ordway (t_{50} = 51 days) and was also quantifiable in those from the Horticultural Research Center (t_{50} = 76 days). These results concur with previous studies that have reported soil half-lives for imazapyr of anywhere from 25-144 days (McDowell et al. 1997, Borjesson et al. 2004). Degradation of triclopyr occurred in 5-10 days for all sites except La Junta (t_{50} = 17 d). This degradation rate is faster than what has generally been reported (10-46 days), but other studies have found soil half-lives of as little as 5 days in field soils (Johnson et al. 1995a, Senseman 2007). In the only other in vitro triclopyr degradation study in the literature the authors reported a mean half-life of 27.5 days for a soil with slightly lower pH (6.9) and organic matter (0.8%) levels, but similar clay (19%) content (Johnson et al. 1995b). However, the initial soil moisture content for the previous study was only 16%, much lower than soil water content in this study.

Based on previous work with both herbicides we would expect that soil pH, organic matter and cation exchange capacity (CEC) differences between soils would account for varying soil adsorption and consequently affect degradation rates (Pusino et al. 1994, McDowell et al. 1997, Wang and Liu 1999, Thompson et al. 2000). AR, OR, and HO all had low-moderate (18-39%) clay contents and tended to have slightly higher organic matter content than the other sites. The other three soils (CC, FL and LJ) all had very high sand content (61-85%) and correspondingly low (3-8%) clay content. Imazapyr degradation rates were inversely correlated with organic matter levels (R^2 = 0.28; Table 2.5) and positively correlated with relative sand

content (R^2 = -0.21). Neither organic matter (R^2 = 0.11, P = 0.3088) or sand content (R^2 = -0.06, P = 0.5494) was significantly correlated with triclopyr concentrations. CEC varied between sampled sites (mean = 23.23 ± 4.91), but did not strongly affect herbicide residue levels.

Imazapyr adsorption is reportedly very weak in sandy soils, making imazapyr relatively more biologically available (Borjesson et al. 2004). Therefore, if biological activity remained equally high in all the tested soils we would have predicted more dramatic decreases in imazapyr residue levels in the sandier soils tested. Imazapyr also has very low vapor pressure (will not volatilize under ambient conditions) and is extremely stable at neutral pH levels (Jenkins et al. 2000). Because we controlled for the effect of temperature the most parsimonious explanations for reduced imazapyr degradation in soils from AR, CC, FL and LJ sites is reduced biological activity. Lower organic matter coupled with higher sand content might have reduced the water holding capacity of the tested soils, resulting in the observed decreases in soil moisture. Soil moisture levels may have become low enough in sample vials at later time points that aerobic microbial degradation was inhibited (Conant et al. 2004).

Plant species sensitivity

Triclopyr was so short-lived in the specific soils tested (AR & HO) that overall biomass production of most plant species was unaffected by herbicide applications (Table 2.7). Despite being generally safe on the species tested, a typical 'field' rate of triclopyr (1.96 kg ai ha^{-1}) did reduce frequency in some species and density in almost all (Table 2.6b). Fecundity for two of the three grass species that flowered during the study was reduced at the 3.92 kg ai ha^{-1} rate. While triclopyr is not normally considered to be injurious to grasses, our results are supported by prior work. For example, Huffman and Jacoby (1984) reported that high rates of triclopyr (roughly

equivalent to the highest rate we used) decreased germination of *B. curtipendula* by 50% and that even very low rates decreased plumule length. However, these studies were carried out in Petri dishes and so we would not necessarily expect to see the same impacts under field conditions where the herbicide would be subject to microbial degradation and adsorption to soil particles. In another study (conducted in a greenhouse with organic material-rich potting soil), a 1.09 kg ai ha^{-1} pre-plant application of a slightly different triclopyr formulation (Garlon 3A®, 44.4% TEA salt) caused a 67%, 87% and 92% reduction in the biomass of grasses, several legumes, and two composite species, respectively (Kaeser and Kirkman 2010). Again, while this study was not carried out in field conditions, it indicates that there is perhaps more potential than generally recognized for triclopyr residues to injure sensitive plants.

On the other hand, sensitivity of many plants to imazapyr is well recognized, and what is surprising about these results is that species tolerance varied considerably. *E. elymoides, H. annuus* and *G. lepidota* were not at all sensitive to imazapyr at the levels used in this study (the highest of which is approximately one quarter of the normal use rate for Habitat® in non-crop areas). Half of the other grass species in the study (*B. curtipendula*, *E. trachycaulus* and *P. smithii*) were still sensitive to imazapyr concentrations that would be found in the soils 16-19 months after application and the remaining species were sensitive to residues up to 25 months after application (Table 2.7). Fecundity in *S. airoides* and *B. curtipendula* was also significantly reduced by a relatively low rate of imazapyr (0.28 kg ai ha^{-1}). Kaeser and Kirkman (2010) also evaluated a formulation of imazapyr very similar to the one we used (Arsenal®) and found that at the same rate we used, a pre-emergent treatment of imazapyr decreased biomass of test species by 75%. With the exception of the tolerance we observed by *G. lepidota* and *H. annuus*, our

results confirm the work of these and other authors that have found plant species to be broadly sensitive to imazapyr.

The persistence, mobility and ultimate biological availability and activity of herbicides in soils is very site-specific (Ogle and Warren 1954, Gevao et al. 2000). However, this study provides evidence for the important implications of herbicide soil degradation rates on the establishment and growth of desirable plant species. Herbicides such as imazapyr and triclopyr are commonly used in natural areas to control unwanted plant species and can serve an important role in such efforts when properly used. However, those using herbicides as part of natural areas management programs should understand that herbicide residues can strongly affect post-control site recovery capacity. Appropriate herbicide use can favor the establishment of valuable native species, as Masters and Nissen (1998) demonstrated in their work on *Euphorbia esula* L. management in the Great Plains. Chemical weed management can also result in unintended outcomes, such as the facilitation of secondary invasions following the removal of the targeted species (Ortega and Pearson 2011). Also, in a companion study we found that large scale aerial imazapyr treatments of tamarisk led to early successional dominance by the ruderal species *Bassia scoparia* (L.) A.J. Scott, which then inhibited understory plant community recovery (Douglass 2013).

There are often multiple treatment options to control a target plant, including non-chemical weed control strategies. For woody species such as tamarisk there are also typically several application methods that can be used for herbicides such as triclopyr and imazapyr (Douglass et al. 2013). Although often more expensive, application methods targeting individual plants rather than large areas can reduce the amount of an herbicide applied to a site and minimize potential non-target impacts. It is critical that natural resource managers carefully

consider treatment options for a given target species and in particular take into account the possible environmental fates of herbicides and how their persistence may impact re-vegetation.

Finally, further study is needed to develop imazapyr and triclopyr degradation and dissipation models for a wider range of soil types that can also account for inevitable variations in climate conditions. Such models would allow land managers to better predict herbicide soil residue levels and potential environmental impacts following typical herbicide applications.

Table 2.1a. Location of sampling sites in eastern Colorado and descriptions of their dominant soils. Sites marked with an asterisk(*) were those at which the plant species sensitivity studies were conducted.

Site ID	City	Latitude (°)	Longitude (°)	Soil Sub-group	Soil Series
AR*	Wellington	40.64742	-105.00025	Aridic Haplustalfs	Fort Collins
CC	Canon City	38.49080	-105.20189	Ustic Torriorthents	Shingle
FL	Florence	38.37967	-105.03772	Aquic Ustifluvents	N/A
HO*	Fort Collins	40.61179	-104.99386	Aridic Argiustolls	Nunn
LJ	La Junta	37.99278	-103.55008	Ustic Torrifluvents	Glenberg
OR	Ordway	38.18254	-103.74780	Vertic Fluvaquents	Apishapa

Table 2.1b. Soil type, pH, cation exchange capacity (CEC, meq 100 g^{-1}), organic matter (OM, %) and texture (% sand, silt, clay) for sites sampled in this study. Sites marked with an asterisk(*) were those at which the plant species sensitivity studies were conducted. All results from private laboratory analysis.

Site	Soil Type	pH	CEC	% OM	% Sand	% Silt	% Clay
AR*	Loam	8.10	26.60	1.90	39.2	32.0	28.8
CC	Very cobbly sandy loam	7.78	16.30	1.70	85.2	11.6	3.2
FL	N/A	7.90	19.75	1.75	61.2	30.6	8.2
HO*	Clay loam	7.90	31.10	3.00	30.8	30.0	39.2
LJ	Sandy loam	8.00	20.65	1.75	68.2	28.6	3.2
OR	Clay loam	7.80	25.00	3.10	46.2	35.4	18.4

Table 2.2. Plant species used in herbicide sensitivity study; seed source refers to the location of collections. Purveyor codes and locations are as follows: “PB” – Pawnee Buttes Seed, Inc., Greeley, CO; “WN” – Western Native Seed, Coaldale, CO.

Plant Species		Spp. Code	Purveyor	Source
Atriplex canescens (Pursh) Nutt.	Fourwing saltbush	ATCA	PB	WY
Bouteloua curtipendula (Michx.) Torr.	Sideoats grama	BOCU	PB	TX
Elymus canadensis L.	Canada wildrye	ELCA	PB	CAN
E. elymoides (Raf.) Swezey	Bottlebrush squirreltail	ELEL	PB	WA
E. trachycaulus (Link) Gould ex Shinner	Slender wheatgrass	ELTR	PB	CAN
Glycyrrhiza lepidota Pursh	American licorice	GLLE	WN	CO
Helianthus annuus L.	Common sunflower	HEAN	WN	CO
Pascopyrum smithii (Rydb.) A. Love	Western wheatgrass	PASM	PB	ID
Sporobolus airoides (Torr.) Torr.	Alkali sacaton	SPAI	PB	CO

Table 2.3. Dilutions and corresponding herbicide rates used in field dose response studies to quantify sensitivity of selected plant species to imazapyr and triclopyr.

Dilution	Imazapyr (kg ai ha^{-1})	Triclopyr (kg ai ha^{-1})
1	0.280	3.920
0.5	0.140	1.960
0.25	0.070	0.980
0.125	0.035	0.489
0.0625	0.017	0.245
0.03125	0.009	0.122
0.015625	0.004	0.056
0	0.000	0.000

Table 2.4. Soil half-lives (t_{50}) for imazapyr and triclopyr residues (as % initial 0 DAT values) in selected Colorado soils calculated from exponential decay models of the form $y = a + b(\exp^{c*\text{Time Point(DAT)}})$. See Table A.1 I for model parameters specific to each site.

Site	Imazapyr t_{50} (days)		Triclopyr t_{50} (days)	
	Mean	95% C.I.	Mean	95% C.I.
AR	> 160		6.31	5.52 - 7.09
CC	> 160		6.75	6.16 - 7.34
FL	> 160		10.38	8.72 - 12.03
HO	76.39	28.20 - 124.57	5.18	4.60 - 5.75
LJ	> 160		17.32	14.72 - 19.92
OR	50.92	34.52 - 67.32	4.86	4.17 - 5.55

Table 2.5. Multivariate correlation analysis of imazapyr and triclopyr soil residues (as % initial 0 DAT values), pH, CEC (meq 100 $g^{-}1$), organic matter (%) and texture (% sand, silt, clay). R^2 values are shown below, those marked with an asterisk (*) are significant at $P < 0.05$.

	pH	CEC	% OM	% Sand	% Silt	% Clay
Imazapyr	-0.0624	0.1758	0.2800*	-0.2099*	0.2786	0.1288
Triclopyr	-0.0695	0.0322	0.1050	0.0619	0.1183	0.0174
pH		0.3303*	-0.3365*	-0.3759*	0.4111*	0.2804*
CEC			0.7372*	-0.9704*	0.6485*	0.9578*
% OM				-0.6934*	0.4980*	0.6648*
% Sand					-0.7665*	-0.9314*
% Silt						0.4800*

Table 2.6a. Summary of plant species responses to imazapyr field dose response study. Effective dose (ED) is the lowest dilution at which imazapyr reduced the given growth parameter compared to the untreated control. An asterisk (*) indicates that (using Dunnett's Method of means comparisons) the growth reduction is significant at $P < 0.05$. The decrease in the growth parameter compared to the untreated control (% UTC) is also given. The coefficient of variation (CV) measures the overall dispersion of the data.

Imazapyr

	Log (Per Plant Dry Biomass (g))			Sqrt(No. Plants)			Logit (Frequency (p))		
	ED	% UTC	CV (%)	ED	% UTC	CV (%)	ED	% UTC	CV (%)
ATCA	0.25*	3	182	1	9	84	NS		-31
BOCU	1	10	89	1	13	60	1	25	-117
ELCA	0.5*	16	292	0.5*	20	47	0.25*	54	568
ELEL	1	27	-289	1*	1	55	NS		-56
ELTR	0.5*	22	74	0.5*	34	39	NS		305
GLLE	NS		-310	NS		114	NS		-22
HEAN	NS		8	NS		29	1	10	-40
PASM	0.5	24	142	NS		54	NS		-991
SPAI	0.5*	13	162	1*	1	63	NS		-84
	Log (Stem height (cm))			Logit (Mean Cover (p))			Log (Seedhead No.)		
ATCA	0.25*	23	40	NS		-54			
BOCU	1*	43	27	1*	13	-86	1	20	48
ELCA	0.125*	53	24	0.25	42	-102			
ELEL	NS		35	NS		-38			
ELTR	1*	37	19	NS		1110	NS		234
GLLE	NS		43	NS		-29			
HEAN	1	7	54						
PASM	1*	38	24	1	22	-143			
SPAI	0.5	28	42	NS		-56	1	32	115

Table 2.6b. Summary of plant species responses to triclopyr field dose response study. Effective dose (ED) is the lowest dilution at which triclopyr reduced the given growth parameter compared to the untreated control. An asterisk (*) indicates that (using Dunnett's Method of means comparisons) the growth reduction is significant at $P < 0.05$. The decrease in the growth parameter compared to the untreated control (% UTC) is also given. The coefficient of variation (CV) measures the overall dispersion of the data.

Triclopyr

	Log (Per Plant Dry Biomass (g))			Sqrt(No. Plants)			Logit (Frequency (p))		
	ED	% UTC	CV (%)	ED	% UTC	CV (%)	ED	% UTC	CV (%)
ATCA	1	2	133	1	7	71	NS		-38
BOCU	1	6	178	1*	10	72	1*	24	-78
ELCA	0.5	28	140	0.5*	21	39	0.5*	46	228
ELEL	NS		-300	0.5*	8	44	NS		-46
ELTR	NS		56	0.5	45	35	NS		211
GLLE	0.25	10	-187	0.25	2	134	0.5	13	-18
HEAN	NS		9	NS		32	1	9	-39
PASM	NS		124	1	26	46	NS		525
SPAI	NS		119	NS		71	NS		-88
	Log (Stem height (cm))			Logit (Mean Cover (p))			Log (Seedhead No.)		
ATCA	1*	22	35	NS		-68			
BOCU	NS		33	0.5	35	-68	1*	6	106
ELCA	0.5*	36	20	0.5*	40	-118			
ELEL	NS		35	NS		-34			
ELTR	1*	49	13	NS		3159	NS		-77
GLLE	0.5	29	46	NS		-27			
HEAN	1*	7	55						
PASM	NS		21	1	38	-155			
SPAI	0.5	27	40	NS		-60	1	20	629

Table 2.7. Imazapyr and triclopyr rates (kg ai ha^{-1}) at which plant species biomass was reduced 50% (GR_{50}) estimated from logistic models of the form $y = c + (d-c)/(1 + \exp^{-a*(Rate\ (kg\ ai\ ha-1) - b)})$. Mean soil half-lives ($T_{50}$) for the herbicides from field study sites (AR & HO) calculated from exponential decay models of the form $y = a(\exp^{b*Time\ Point(DAT)})$. Values marked with a '⦸' indicate model parameters that were poorly estimated by the model. See Tables A.1, A.2 & A.3 for relevant model goodness of fit statistics and parameters.

Imazapyr	Est. GR_{50} (kg ai ha^{-1})	95% C.I.	Est. T_{50} (days)	95% C.I.
Atriplex canescens	0.069	⦸	693	488 - 899
Bouteloua curtipendula	0.162	0.082 - 0.244	485	344 - 627
Elymus canadensis	0.051	0.022 - 0.080	767	539 - 995
E. elymoides	> 0.280	⦸	< 0.00	⦸
E. trachycaulus	0.119	0.023 - 0.074	560	396 - 725
Glycyrrhiza lepidota	> 0.280	⦸	< 0.00	⦸
Helianthus annuus	> 0.280	⦸	< 0.00	⦸
Pascopyrum smithii	0.104	0.034 - 0.171	593	418 - 768
Sporobolus airoides	0.051	0.014 - 0.023	767	539 - 995
Triclopyr				
Atriplex canescens	2.696	⦸	N/A	N/A
Bouteloua curtipendula	0.705	0 - 1.443	5	5 - 6
Elymus canadensis	1.657	0.327 - 2.987	N/A	N/A
E. elymoides	> 3.920	⦸	N/A	N/A
E. trachycaulus	2.614	⦸	N/A	N/A
Glycyrrhiza lepidota	< 0.056	0 - 0.483	< 0.00	⦸
Helianthus annuus	> 3.920	⦸	N/A	N/A
Pascopyrum smithii	2.735	⦸	N/A	N/A
Sporobolus airoides	0.964	0.518 - 1.410	3	2 - 3

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CHAPTER 3. EFFICIENCY AND EFFICACY OF HELICOPTER IMAZAPYR APPLICATIONS TO CONTROL AN INVASIVE RIPARIAN TREE SPECIES (TAMARISK, *TAMARIX* SPP.)

Summary

Aerial applications of the herbicide imazapyr are one of the most common and effective methods for controlling the widespread invasive tree tamarisk. However, little research has focused on whether the spatial homogeneity and vertical structure of tamarisk stands and adjacent areas effects the success of control efforts and the extent of unintended outcomes. This study used three dimensional artificial trees and repeated soil sampling to determine whether tamarisk canopies retained aerially-applied imazapyr and how soil degradation was affected. Tamarisk mortality was quantified using multiple, recurring stand and individual tree measurements. The average tree canopy captured 75% of aerially-released imazapyr, resulting in significantly lower soil residues beneath the tree canopy. Although initial imazapyr soil residue levels outside the tree canopy were almost four time greater than those inside, soil degradation occurred more than twice as rapidly in outside soils and resulted in lower residue levels. Helicopter imazapyr applications resulted in 98% tamarisk mortality within two years, but the consistency of effectiveness was reduced by non-linear stand boundaries and tall site obstructions. The same factors also increased variability in the actual quantity of herbicide applied to sites, increasing the probability of substantial non-target ecosystem impacts.

Introduction

There are a number of introduced tree species that have become invasive in North America and one of the most frequently targeted and controversial is tamarisk (Friedman et al. 2005, Stromberg et al. 2009). While there are a half dozen species in the *Tamarix* genus that are

now found in the western United States recent molecular work has established that the most widespread populations are *T. ramosissima* Ledeb., *T. chinensis* Lour. and their hybrids (Gaskin and Schaal 2002, Gaskin and Kazmer 2009). Tamarisk was intentionally introduced into the U.S. beginning in the early 1800s for ornamental purposes and by the 1930s was widely used for erosion and sedimentation control (Tellman 2002, Chew 2009). In the middle of the 1900s tamarisk began to be targeted for removal due to its reportedly disproportionate water consumption, among other factors (Robinson 1965, Chew 2009).

Early efforts to remove and control tamarisk were led by federal scientists who carried out intensive research evaluating the efficacy of chemical and cultural management options (Subcommittee 1970). While there are currently multiple methods for successfully managing tamarisk, herbicides are arguably the most popular (Douglass et al. 2013). One of the most common herbicides currently employed for tamarisk control is imazapyr, which was first registered for use in non-crop areas by the U.S. Environmental Protection Agency (EPA) in 1984 (Douglass et al. 2013). Imazapyr is a broad spectrum herbicide that can be applied using several different methods and generally provides reliable, long term control of tamarisk and other noxious weed species (Brock 1994). Imazapyr also breaks down very rapidly via photolysis in water, resulting in the widespread application of certain formulations in riparian and aquatic environments (Mallipudi et al. 1991, Douglass et al. 2013). However, imazapyr residues can be relatively persistent in soils (reported soil half-life (t_{50}) = 25-142 days) depending on soil texture, organic matter content, pH levels and environmental factors that promote microbial degradation (Vizantinopoulos and Lolos 1994, Senseman 2007, Nissen et al. 2010). Because of its broad spectrum of activity and soil persistence imazapyr can negatively impact non-target plant species (Sher et al. 2010).

Tamarisk infestations often occupy very large (hundreds of hectares) areas and occur in difficult to reach terrain, so aerial imazapyr applications are often the most economical option for effectively killing tamarisk (Douglass et al. 2013). While both fixed wing aircraft and helicopters have been used for this purpose the later are widely used due to their increased versatility, slower air speeds and ability to use higher application volumes (McDaniel and Taylor 2003, Hart 2009). Much of the research on aerial chemical control of tamarisk has focused on modifying application technologies to reduce costs and maintain high mortality (McDaniel and Taylor 2003). To date there has been very little research focused on evaluating how tamarisk tree morphology and canopy structure influences the effectiveness of imazapyr helicopter applications.

This question is important to consider for several reasons. First, tamarisk canopy morphology and other traits vary geographically and seasonally. If tree canopy characteristics influence the efficacy of aerially applied imazapyr it could determine whether or not it is the most suitable application method for given populations (Lesica and Miles 2001, Friedman et al. 2011). Second, it is critical to better understand how tamarisk canopy structure influences the retention of aerially-applied imazapyr. Imazapyr retention by the tree canopy not only facilitates herbicide absorption and translocation, but also reduces non-target site impacts by minimizing soil concentrations beneath the canopy. Finally, tamarisk infestations occur in diverse habitats and it would be beneficial to understand how site and stand attributes affect overall tamarisk control levels.

Our research specifically aimed to: 1) establish how much helicopter-applied imazapyr is retained by the tree canopy; 2) determine the resulting soil concentration and imazapyr

persistence underneath and outside the tamarisk canopy; and, 3) quantify tamarisk mortality from helicopter imazapyr applications.

Materials and Methods

Site information and aerial herbicide applications

Three sites (Canon City (CC), La Junta (LJ) and Ordway (OR)) in the Arkansas River watershed of southeastern Colorado with homogenously dense tamarisk infestations were selected (Table 3.1). A one hectare experimental area at each site was mapped using a handheld GPS unit (Garmin GPSMAP® 60CSx, Garmin International, Inc., Olathe, KS). Aerial herbicide applications were made to each area by a commercial helicopter pilot on 4 September 2009 (CC) and 10 September 2009 (LJ & OR). Imazaypr isopropylamine salt (Habitat® + 1% v/v Dyne-Amic® non-ionic surfactant) was applied at 1.12 kg ai ha^{-1} using CP-03 nozzles (CP Products, Inc., Tempe, AZ). Mean flight speed was 48 km hr^{-1} and applications were made 1-2 m above the tree canopy. Six circular sampling plots (84 m^2) were located in a spatially balanced design in the treated areas using the Reversed Randomized Quadrant-Recursive Raster (RRQRR) algorithm (Theobald et al. 2007) in ArcGIS (Release 9.1, Environmental Systems Research Institute (ESRI), Redlands, CA). This probability-based survey design is statistically rigorous and allowed for the use of standard estimates of population characteristics (Theobald et al. 2007).

Collection of soil, litter and blotting paper samples

To quantify aerially applied imazapyr concentrations at ground level a pair of artificial aluminum ‘trees’ were installed in each sampling plot prior to herbicide applications. Each ‘tree’ consisted of a 1 m tall aluminum rod outfitted with three flat appendages (2 cm x 15 cm). One

artificial ‘tree’ was placed underneath a representatively dense portion of the tamarisk canopy and another directly outside the canopy in a completely open area. Appendages were placed at 15 cm above ground level and oriented at 0, 120 and 240°. A 37.5 cm^2 rectangle of blotting paper (Whatman Grade GB004, Whatman Inc., Florham Park, NJ) was mounted on each appendage immediately before helicopter imazapyr applications. After applications the blotting paper was collected and sealed in a small Ziploc bag. Canopy cover above the inside canopy artificial tree was measured using a LI-COR LAI 2000 canopy area meter (LI-COR, Lincoln, NE). Measurements were taken during both the growing season and the dormant season to accurately quantify leaf area index (LAI).

Adjacent to both the inside and outside canopy artificial ‘trees’ a 10 x 10 x 10 cm soil sample was collected before and immediately following the herbicide application. Soil samples were placed in Ziploc bags, thoroughly homogenized and stored on ice for transport to the laboratory. Soil samples were stored at -20 °C until analysis. The location of the artificial trees was permanently marked and soil samples collected again at the same location 6, 12, 24, and 36 months after treatment (MAT). Before, immediately after and 36 MAT a 100 cm^2 sample of tamarisk leaf litter was collected adjacent to the inside canopy artificial tree by removing a portion of the litter layer down to the soil surface.

Extraction of imazapyr from soil, litter and blotting paper samples and preparation of HPLC analysis samples

To extract imazapyr herbicide residues from blotting paper samples each 37.5 cm^2 piece was cut into smaller sections that fit in a 50 mL polypropylene centrifuge tube (Fisher Scientific Inc., Pittsburgh, PA). Deionized water (20 mL) was added and the tubes were shaken for one

hour before being centrifuged for ten minutes (4,000 rev min^{-1}; Sorvall Legend XT Centrifuge, Thermo Fisher Scientific, Waltham, MA). A 5 mL aliquot was removed from the supernatant and passed through a 0.45 μm Acrodisc® PVDF filter syringe tip (Pall Corp., Port Washington, NY) before a sub-sample was transferred to an HPLC vial for analysis. The three blotting paper samples from each artificial tree were extracted and analyzed separately. An average herbicide concentration value was calculated for inside and outside the canopy at each of six sampling plots at the three field sites.

To extract imazapyr residues from soil samples three separate 5 g sub-samples were taken from each 1 L plot sample. Sub-samples were weighed into individual 50 mL polypropylene centrifuge tubes and extracted with 10 mL of deionized water. The sample was shaken for 2 hours and then centrifuged for ten minutes (4,000 rev min^{-1}). A 1 mL aliquot was collected from the supernatant, placed in a 0.45 μm Spin-X® centrifuge tube filter (Corning Inc., Corning, NY) and then centrifuged for ten minutes (13,000 rev min^{-1}; Sorvall Legend Micro 21 Centrifuge, Thermo Fisher Scientific, Waltham, MA). The resulting filtered liquid was transferred to an HPLC auto sampler vial with a limited volume (400 μL) insert (National Scientific, Rockwood, TN) for analysis. Quality control (QC) samples were prepared with imazapyr analytical standard (Sigma-Aldrich Inc., St. Louis, MO, Lot No. 7151X, 99.9% purity) and included in each HPLC analysis.

To separate bound imazapyr from the surface of collected litter samples a 25 g sub-sample was evenly distributed across a mesh screen (20 X 20 cm) placed above a glass collection pan. An overhead track sprayer was used to apply 1.25 cm of water over 15 minutes. The resulting liquid was strained through cheese cloth to remove debris. An aliquot was removed,

passed through a 0.45 μm PVDF filter syringe tip (Whatman Inc., Florham Park, NJ) and then a sub-sample transferred to an HPLC vial (National Scientific, Rockwood, TN).

HPLC quantification of imazapyr concentrations

Samples were analyzed on an Hitachi D-7000 HPLC system with an in line UV detector set at 250 nm and an injection volume of 100 μL. A Zorbax Rx C8 4.8 mm X 250 mm column was used (Agilent Technologies, Santa Clara, CA). Mobile phase A was 10% HPLC-grade acetonitrile : 90% water and mobile phase B 30% acetonitrile : 20% water, both with 0.5% phosphoric acid added. A gradient ranging from 100% A to 100% B in ten minutes followed by a 5 minute re-equilibration time before the next injection was utilized to elute imazapyr. Mobile phase flow rate was 1.2 mL min^{-1} and all analyses were done at ambient temperature. A calibration curve was obtained for imazapyr by plotting peak areas against analytical standard concentrations for a range of diluted standards (0.005 – 5 μg ml^{-1}). The relationship between chromatographic peak area and imazapyr concentration over the range of standards was described by a linear regression ($R^2 > 0.99$).

Mean retention time for imazapyr was 10.08 minutes (SD = 0.70) for 792 analyzed samples. Imazapyr recovery in 174 fortified soil samples averaged 111.08% with a coefficient of variation (CV) of 37.83%. The limit of quantification (LOQ) was 10 μg kg^{-1} imazapyr in the soils tested and the limit of detection (LOD) was 5 μg kg^{-1} imazapyr. Differences in soil moisture between sites and sampling dates were accounted for by calculating the gravimetric water content for each sample using a sub-sample collected prior to HPLC analysis. Sub-samples used for water content analysis were dried at 70 °C for 24 h. Results were then used to convert

imazapyr concentration data derived from the HPLC analysis into a concentration based on dry soil weight.

Soil and litter bacterial analysis

Prior to herbicide applications a 6.5 cm diameter soil auger was used to collect three 10 cm deep samples from each sub-plot at each site. Samples were pooled, homogenized in a large Ziploc bag and placed on ice in a cooler. From each sub-plot an additional 100 cm^2 sample of tamarisk leaf litter was collected. The litter samples and a portion of the soil samples were submitted to the Environmental Quality Laboratory at Colorado State University, Fort Collins for quantification of aerobic bacteria content. The remainder of the soil samples were air dried for 72 hours and a portion removed and sent to AgSource Laboratories (Lincoln, NE) for analysis of texture, mean soil pH, cation exchange capacity (CEC, meq 100 g^{-1}) and organic matter (%).

Quantifying tamarisk control

Tamarisk mortality resulting from the aerial imazapyr applications was measured inside the six 84 m^2 circular sampling plots at each site. Within each circle tree density (plants were counted separately if primary stems were more than 30 cm apart) and stem density data were collected 0, 12, 24 and 36 months after treatment (MAT). Ten permanent measurement points were also randomly selected from a combination of an angle chosen by dividing the circle into ten equally-distributed wedges and a unique distance from the circle's center. At each point and sampling interval canopy height (m) was measured and canopy cover directly above each point was measured using a LI-COR LAI 2000 canopy area meter, as described previously. Thirty

stems throughout the entire plot were also randomly selected and their diameter (cm) approximately 15 cm above ground level measured at each sampling interval.

Climate data

Climate data were obtained from the US National Oceanic and Atmospheric Administration (NOAA) National Climate Data Center (NCDC 2012). Annual climatological summary data for September 1979 to September 2012 were acquired for the NOAA weather station closest to each site. Data from 1979 to 2008 were used to calculate historical temperature and precipitation averages for the three sites for the time period immediately preceding project implementation. The distance from weather stations to actual treatment areas was 3.48 km for CC, 1.30 km for LJ and 4.72 km for OR.

Statistical analyses

Data were tested for normality and homoscedasticity prior to all analyses using the Shapiro-Wilk W and Levene tests of model residuals, respectively. Imazapyr concentration data met these assumptions, but tamarisk control data was either natural log transformed (plant density, canopy height and stem diameters) or square root transformed. Exponential decay models were fit separately for inside and outside canopy imazapyr soil concentration data derived from HPLC analysis. The regression models were used to calculate the t_{50} value for imazapyr (the number of days to reach a 50% reduction in herbicide concentration) and t_{99} (days to 99% reduction) data. Imazapyr soil concentrations for all decay models were expressed as the percentage of initial (0 MAT) concentrations for each site and canopy location. Tukey's HSD

test was used to compare tamarisk mortality between years and sites. JMP (ver. 10.0.1, SAS Institute, Cary, NC) software was used for all statistical analyses.

Results

Tamarisk canopy retention of aerially-applied imazapyr

Analysis of blotting paper samples indicated that the tamarisk canopy retained a significant ($P < 0.0001$) portion of aerially-applied imazapyr. The mean blotting paper concentration in open areas at La Junta and Ordway was 0.99 ± 0.13 kg ai ha^{-1} and the mean concentration underneath the tree canopy was 0.30 ± 0.05 kg ai ha^{-1} (Data not shown). At Canon City the mean concentration outside the canopy was 2.24 ± 0.33 kg ai ha^{-1}, compared to 0.44 ± 0.09 kg ai ha^{-1} inside the canopy. Blotting paper concentrations were significantly ($P < 0.0001$) correlated with 0 MAT imazapyr soil sample concentrations.

Imazapyr soil and litter persistence

Imazapyr persistence in soils at project sites was best estimated by a non-asymptotic exponential decay model:

$$y = a*(\exp^{b*\text{Time Point(DAT)}})$$

Models explained imazapyr soil degradation inside and outside the canopy at all sites very well (mean AICc = 621 (Table 3.2)). At all sites there was more rapid ($\alpha = 0.05$) initial imazapyr soil degradation outside the canopy (mean t_{50} = 33 days) than inside (mean t_{50} = 82 days (Table 3.2)). Models estimated that longer term soil degradation would occur more rapidly at La Junta, followed by Ordway and Canon City; but differences were not significant. The difference in degradation rates between canopy location were greatest for Canon City (74% more rapidly

outside the canopy), followed by La Junta (65%) and Ordway (46%). Concentrations outside the canopy were reduced substantially (95.5 ± 0.4% (mean ± SE)) within six months of treatment and remained very low throughout the study period but with the exception of La Junta residues did not ever reach zero (Figure 3.1).

Imazapyr concentrations in tamarisk litter immediately following treatment was highest in samples from Ordway (0.38 ± 0.11 kg ai ha^{-1}), followed by La Junta (0.37 ± 0.11 kg ai ha^{-1}) and Canon City (0.16 ± 0.07 kg ai ha^{-1} (Data not shown)). Imazapyr concentrations in leaf litter were positively correlated (P = 0.08) with imazapyr soil concentrations at the same initial time point. There was no imazapyr recovered from litter samples collected in the same locations three years after applications.

Aerobic bacteria concentrations

Aerobic bacteria concentrations inside (2,246,667 bacteria g^{-1} soil) and outside (2,501,910 bacteria g^{-1} soil) the canopy did not differ statistically. Soil bacteria were more numerous outside the canopy than inside at Canon City (69%) and La Junta (27%), but more numerous inside the canopy at Ordway. For all sites bacterial densities in tamarisk leaf litter samples were 3-4 orders of magnitude higher ($P < 0.05$) than densities found in soil samples.

Soil chemistry, texture and moisture

Canon City (CC) and La Junta (LJ) had sandy loam soils with very low (1.73%) organic matter (Table 3.3). Ordway (OR) had a clay loam soil with almost twice the organic matter (3.1%) and more clay and silt. Ordway and La Junta exhibited similar trends in soil moisture, with the wettest soils in spring 2010 (OR: 23.8 ± 1.6%; LJ: 34.9 ± 1.4%) followed by

significantly drier soil the following years (OR: 7.5 ± 0.9%; LJ: 3.6 ± 0.9% (Figure 3.2)). At Canon City soil moisture increased between 2009 and 2010 (21.7 ± 0.8%) and then fell significantly the final two years of the study (4.3 ± 1.3%). Throughout the study soil moisture was higher inside the canopy than outside, though the differences were greatest at lower soil moisture levels. In 2009 mean soil moisture at all sites was 54% lower outside the canopy than inside. The following year when moisture was higher it was almost equal (90%). In the final years of the study moisture levels outside the canopy fell to 43% of those underneath the now bare tree canopy. Imazapyr soil residues were not significantly correlated with soil moisture for Ordway and La Junta, but were for Canon City ($P = 0.0002$).

Precipitation

Canon City had slightly higher historic annual precipitation (34.47 cm) compared to La Junta and Ordway (28.72 cm (Figure 3.3)). The year before this study was initiated precipitation at Canon City and Ordway fell below the historical average (23.82 cm) while that at La Junta increased (37.93 cm). The year after treatments all sites experienced higher than average precipitation (39.04 cm), but in 2011 precipitation fell 58% to historic lows (16.25 cm). In 2012 precipitation at Canon City and La Junta rose back towards the historic normal (29.96 cm), but Ordway remained drier than normal (16.02 cm). There was a significant correlation ($P < 0.0001$) between annual precipitation and soil moisture levels.

Tamarisk control

At Canon City helicopter imazapyr applications resulted in 100% mortality within one year and this control was sustained for the duration of the study (Table 3.4). There was an

average first year plant density reduction of 40% and further declines of 75% (OR) and 53% (LJ) by three years after treatment at the other two sites. At Ordway imazapyr applications reduced the number of living stems per plant by 25% in the first year and 75% by year two. Imazapyr applications did not significantly reduce the number of living stems per plant at La Junta.

Aerial imazapyr applications reduced the mean living canopy height at Ordway 75% the first year and 94% the second. At La Junta the mean canopy height was reduced 45% the first year and an average of 85% three years afterwards. Living tamarisk foliage (measured as LAI) was reduced 44% one year after treatment at Ordway and almost 100% by the third year (Figure 3.4). At La Junta foliar cover declined 66% the first year and 94% by the final year of the study.

Discussion

Tamarisk canopy retention of aerially-applied imazapyr

Analysis of both blotting paper and soil samples suggested that the tamarisk canopy captured and retained a substantial proportion of helicopter-applied imazapyr. Immediately following aerial applications at project sites imazapyr soil concentrations beneath the tamarisk tree canopy (0.53 ± 0.07 kg ai ha^{-1}) were 74% lower than those in adjacent open areas (2.01 ± 0.36 kg ai ha^{-1}). The magnitude of the difference between canopy locations is somewhat skewed due to initial soil concentrations at two sites that were higher than the targeted application rate of 1.12 kg ai ha^{-1}. In particular, at Canon City initial soil residue levels were 2.20 (± 0.70) kg ai ha^{-1} and at La Junta initial residue levels were 2.71 (± 0.57) kg ai ha^{-1}. However, at LJ soil concentrations (0.97 ± 0.33 kg ai ha^{-1}) were not significantly different from the target application rate. Ultimately these soil residues did not necessarily mean that the remaining herbicide was absorbed because only 40% (± 5.8%) of the tamarisk canopy at sites in 2009 was photosynthetic

tissues (Figure 3.3). When applied aerially with a non-ionic surfactant imazapyr absorption by woody species occurs primarily via the foliage and uptake via bark would presumably be minimal (Radosevich et al. 1997).

Imazapyr soil and litter persistence

Our study found that imazapyr degradation occurred relatively quickly under field conditions, but also that soil half lives (t_{50}) varied. There did not appear to be any significant relationships between measured soil parameters and imazapyr degradation rates. The most consistent trend revealed by this study was the sustained difference in imazapyr degradation rates underneath the canopy compared to open areas. Although initial imazapyr residues were four times higher in open areas, degradation of those residues within the first six months occurred 2.5 times more rapidly than did imazapyr degradation in soils underneath tamarisk trees (Table 3.2). In this time period soil concentrations outside the canopy were reduced to 2% of initial concentrations (0.04 ± 0.01 kg ai ha^{-1}) while those inside the canopy were reduced to only 17% of initial levels (0.09 ± 0.02 kg ai $ha^{-}1$ (Figure 3.1)). Imazapyr degradation continued to occur more rapidly outside the canopy than inside with models estimating a 99% reduction in concentrations by 221 DAT outside the canopy and 549 days underneath (Table 3.2).

Soil moisture appeared to be an important factor contributing to increased degradation because imazapyr breakdown in the soil is largely due to aerobic microbial activity (Ismail and Ahmad 1994, Conant et al. 2004, Senseman 2007, Nissen et al. 2010). Relatively higher soil moisture levels likely explained rapid early imazapyr degradation outside the canopy and particularly at Canon City and La Junta where moisture levels increased almost three-fold the first year after applications (Figure 3.2). This increase was likely driven by a 20% increase in

precipitation over this time period compared to historic normals (Figure 3.3). Significant decreases in soil moisture levels and annual precipitation the following two years probably negatively impacted soil microbial activity (Figures 3.2 & 3.3).

High precipitation and soil moisture soon after applications may have also resulted in substantial downward movement (leaching) of imazapyr in the soil profile, which we were not able to capture in our shallow soil samples. Rather than reflecting rapid degradation of imazapyr, decreased concentrations that were measured within one year of application in our study could have been due in part to dissipation. Particularly in soils with low organic matter it is not uncommon for imazapyr to move up to 50 cm downwards within months of application (Vizantinopoulos and Lolos 1994, McDowell et al. 1997, Borjesson et al. 2004).

Increased soil persistence of imazapyr underneath the tamarisk canopy was interesting given that average soil moisture levels were higher beneath trees than in open soils for the duration of the study. While aerobic bacteria densities differed between canopy location they did not explain relative degradation rates. For example, bacteria densities were highest underneath the canopy at Canon City where models predicted the greatest imazapyr soil persistence. We hypothesize that increased imazapyr soil persistence underneath the canopy could be a result of treated foliage being shed by deciduous tamarisk trees (Newton et al. 1990). Analysis of canopy cover data confirms that there was a significant loss of foliage by tamarisk trees in 2010 (Figure 3.4). For instance, at La Junta there was a net reduction in foliage equivalent to roughly 30% of the total tree canopy. The capacity of tamarisk foliar litter to retain and release aerially applied imazapyr residues was confirmed by an analysis of concentrations bound to litter (0.39 ± 0.10 kg ai ha^{-1}) collected underneath trees following application.

Tamarisk mortality from helicopter imazapyr applications

Helicopter imazapyr applications resulted in significant tamarisk mortality. Initially tamarisk stands at our sites averaged 0.14 (± 0.03 (SE)) plants m^{-2}, had foliar cover of 0.58 (± 0.09), 6.7 (± 0.91) stems $tree^{-1}$ and a canopy height of 2.45 (± 0.33 m) (Table 3.4). Within one year plant densities were reduced 60% (± 7.1) and by 2011 there were 66% (± 11.3) fewer surviving plants. There was an 87% (± 12.7) reduction in foliage in one year and a 97% (± 2.6) decrease two years after application. The average number of stems per plant decreased 75% (± 10.1) in two years and the mean diameter of living stems was reduced 52% (± 14.1). Tree canopy height was affected more quickly, with a 60% (± 12.3) decrease in only one year. Overall, coarse-scale tamarisk mortality (98% three years after treatment) from our study is comparable to other studies of aerial imazapyr tamarisk control (Duncan and McDaniel 1998, McDaniel and Taylor 2003, Nissen et al. 2010).

Our results points to two important issues that strongly influenced tamarisk mortality from helicopter imazapyr applications: 1) the overall shape of the tamarisk stand; and 2) the presence or absence of overstory obstructions. Of our three project sites, Ordway was an example of an ideal target for aerial application, one with roughly linear boundaries and no overstory. Mean soil concentrations following applications at this ideal site were very close (-12%) to theoretical target rates. In contrast, at Canon City the boundaries and shape of the tamarisk infestation were very irregular. These factors possibly contributed to high initial soil concentrations (+98%) compared to the target rate and relatively high variability (CV_{Soils} = 78%). Interestingly, the mean inside canopy concentrations following applications at these two sites (0.46 kg ai ha^{-1}) was very similar to that at Ordway (0.39 kg ai ha^{-1}). Despite recent improvements to application methods and technologies (e.g. segmented spray booms, fine scale

GPS-based navigation systems) if a tamarisk stand is not homogenously dense or has irregularly shaped boundaries it can clearly be difficult to achieve a uniform application (McDaniel and Taylor 2003, Douglass et al. 2013). Furthermore, to improve accuracy helicopter applications normally take place just above the tamarisk canopy so if there are power lines or tall trees that obstruct helicopter flight lines application precision can be negatively impacted (Thompson et al. 1997).

Conclusions

Initially, three quarters of helicopter-applied imazapyr at sites in southeastern Colorado was retained by the tamarisk tree canopy. Soon after applications, there were significant differences in imazapyr soil concentrations underneath the canopy and in adjacent open areas. The subsequent imazapyr soil degradation profiles inside and outside the tamarisk canopy also varied substantially. Initial concentrations outside the tree canopy were much higher (up to 4x) than those inside, but degradation within the first six months after application occurred much more rapidly in open soils and resulted in lower residue levels.

Helicopter imazapyr applications caused high tamarisk mortality (98% two years after treatment), but the effectiveness of this application method was strongly influenced by the spatial homogeneity of the infestation and by tall site obstructions. Our study provides strong evidence for the importance of considering the suitability of particular sites for aerial imazapyr applications. For tamarisk and other noxious woody species there are a range of effective control strategies available (Douglass et al. 2013). Although clearly effective, helicopter applications of imazapyr may not always be the best choice for controlling non-native woody plant infestations at sites with numerous obstructions or desirable understory plant species.

Table 3.1. Location of tamarisk aerial herbicide application sites in the Arkansas River watershed, Colorado and weather conditions at the time of applications.

Site ID	City	Latitude (°)	Longitude (°)	Temp. (°C)	RH (%)	Avg. Wind Speed (km hr^{-1})
CC	Canon City	38.49174	-105.20243	23	41	11
LJ	La Junta	37.99315	-103.54895	22	38	10
OR	Ordway	38.18259	-103.74740	27	38	10

Table 3.2. Model goodness of fit test results (AICc) for exponential decay models of the form $y = a(\exp^{b*\text{Time Point(DAT)}})$ used to predict imazapyr soil degradation rates following 2009 helicopter applications. The milestones t_{50} and t_{99} represent the time (days) models predict it would take at each site for imazapyr to degrade to 50% or 99% of initial concentrations; 95% confidence intervals (C.I.) for each mean are also given.

	Canopy Location	AICc	t_{50} (Days)	95% C.I.	t_{99} (Days)	95% C.I.
CC	Inside	253.39	110	77 - 142	749	531 - 966
	Outside	121.77	29	22 - 36	195	149 - 241
LJ	Inside	226.92	69	50 - 87	456	339 - 574
	Outside	90.40	24	17 - 31	160	113 - 207
OR	Inside	233.53	77	57 - 98	516	383 - 648
	Outside	210.90	42	25 - 59	280	168 - 392
MEAN	Inside	711.51	82	69 - 96	549	462 - 637
	Outside	528.69	33	25 - 41	221	168 - 275

Table 3.3. Soil type, pH, cation exchange capacity (CEC, meq 100 g^{-1}), organic matter (OM, %) and texture (% sand, silt, clay) for sites sampled in this study. All results from private laboratory analysis.

Site	Soil Type	pH	CEC	% OM	% Sand	% Silt	% Clay
CC	Cobbly sandy loam	7.78	16.30	1.70	85.2	11.6	3.2
LJ	Sandy loam	8.00	20.65	1.75	68.2	28.6	3.2
OR	Clay loam	7.80	25.00	3.10	46.2	35.4	18.4

Table 3.4. Tamarisk mortality as measured by living plant density (plants m^{-2}), living canopy height (m), canopy cover (LAI), living stems per plant and stem diameter (cm) and foliar cover (LAI) at three sites in Colorado for three years following helicopter imazapyr applications. Means sharing the same superscript letters are not significantly different at $P < 0.05$.

Site	Year	Mean Plant Density	Mean Canopy Height	Mean LAI	Mean No. Stem Plant^{-1}	Mean Stem Diameter
CC	2009	0.15^{a}	3.66^{a}	0.77	4.27^{a}	8.39^{a}
	2010	0.00^{b}	0.00^{b}	0.10	0.00^{b}	0.00^{b}
	2011	0.00^{b}	0.00^{b}	0.00	0.00^{b}	0.00^{b}
	2012	0.00^{b}	0.00^{b}	0.00	0.00^{b}	0.00^{b}
LJ	2009	0.15	2.05^{a}	0.55	7.83	3.83^{a}
	2010	0.10	1.14^{b}	0.19	6.07	3.72^{ab}
	2011	0.07	0.24^{c}	0.04	7.00	2.51^{b}
	2012	0.07	0.37^{c}	0.04	5.41	3.63^{ab}
OR	2009	0.11^{a}	1.65^{a}	0.41	8.13^{a}	4.94^{a}
	2010	0.06^{ab}	0.42^{b}	0.27	6.13^{a}	5.13^{a}
	2011	0.03^{b}	0.08^{b}	0.00	1.97^{b}	1.53^{b}
	2012	0.03^{b}	0.08^{b}	0.00	1.50^{b}	3.91^{ab}

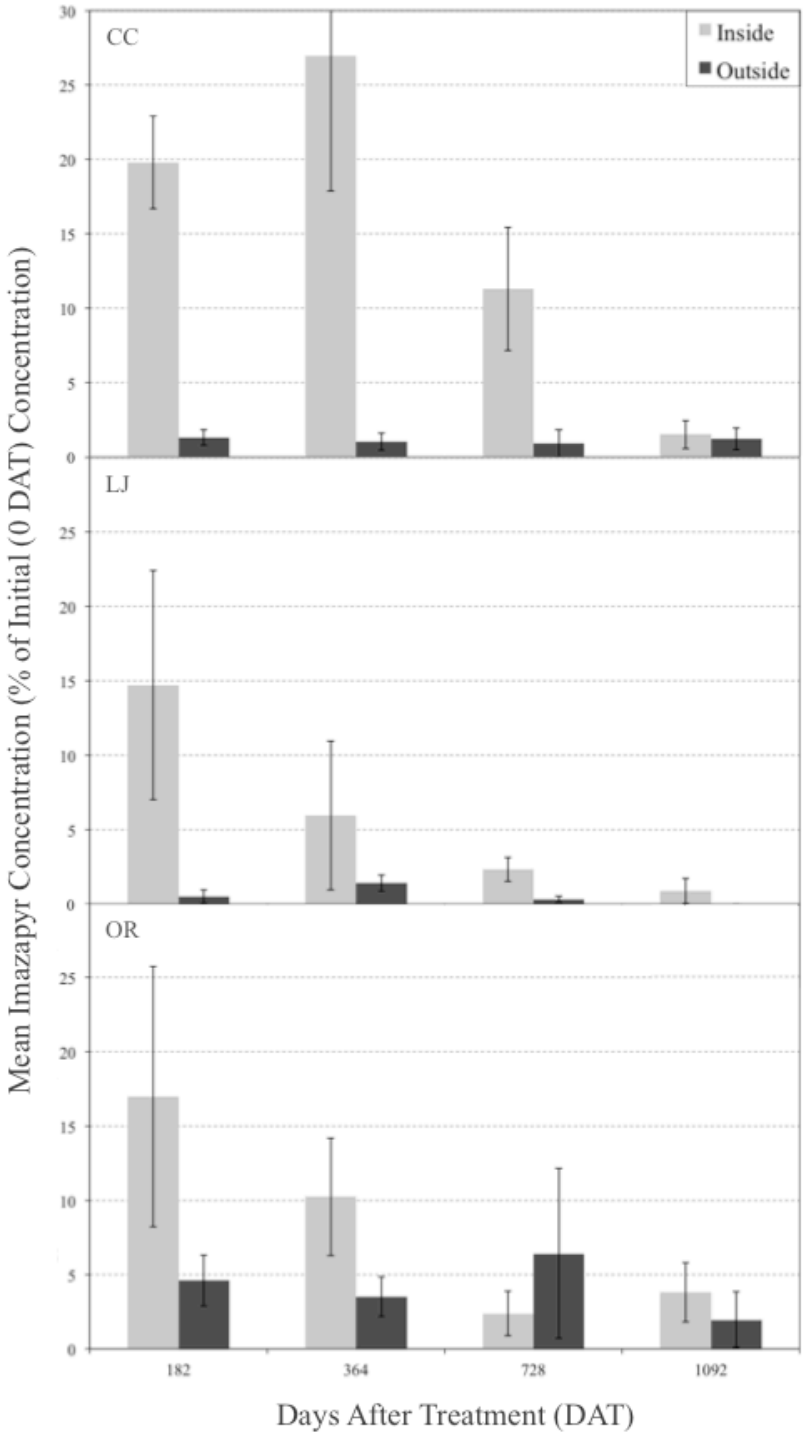

Figure 3.1. Soil imazapyr concentrations (expressed as a percentage of the initial concentration) at several sites in Colorado six months to three years after 2009 helicopter applications. Vertical bars represent ± one standard error (SE). For reference 0 DAT concentrations were as follows (kg ai ha^{-1}, mean ± SE): CC_{Inside} = 0.61 ± 0.19; $CC_{Outside}$ = 2.20 ± 0.70; LJ_{Inside} = 0.46 ± 0.04; $LJ_{Outside}$ = 2.71 ± 0.58; OR_{Inside} = 0.51 ± 0.11; $OR_{Outside}$ = 0.95 ± 0.21.

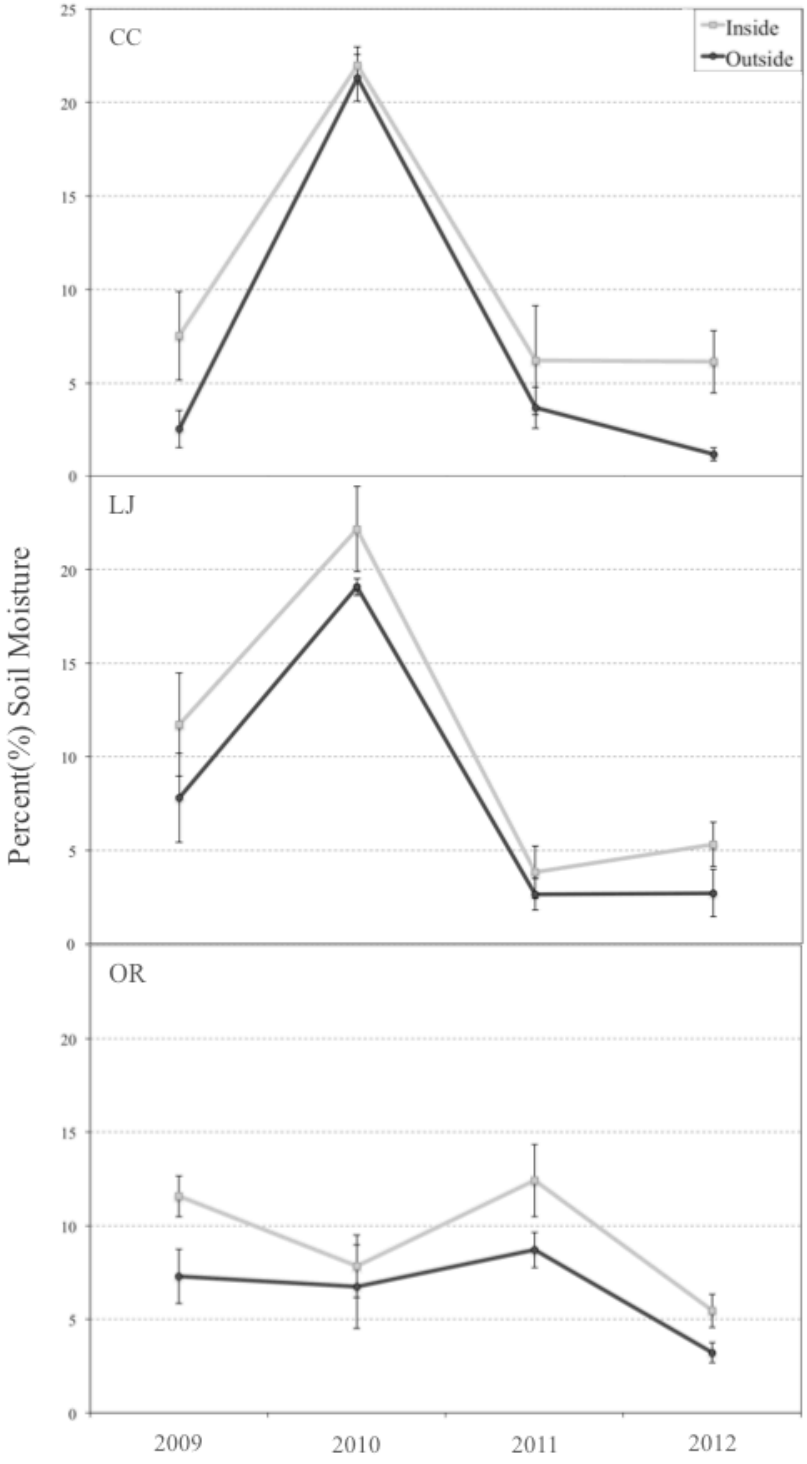

Figure 3.2. Soil moisture (data from gravimetric analysis of collected soil samples) at several sites in Colorado six months to three years after 2009 helicopter applications. Vertical bars represent ± one standard error (SE).

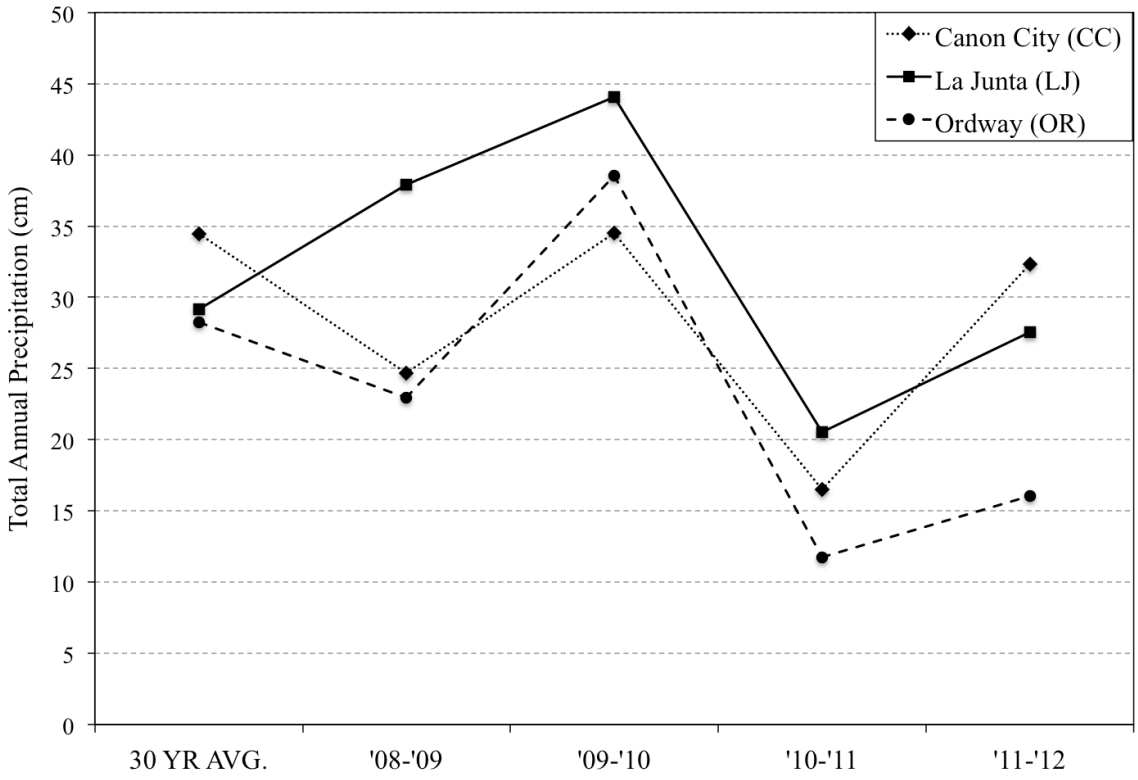

Figure 3.3. Total annual precipitation for project sites in southeastern Colorado for September 2008 – September 2012, twelve month periods that corresponded to sampling intervals for soil imazapyr residue analysis. Thirty year averages calculated using historic precipitation data for each site for September 1979 – September 2008.

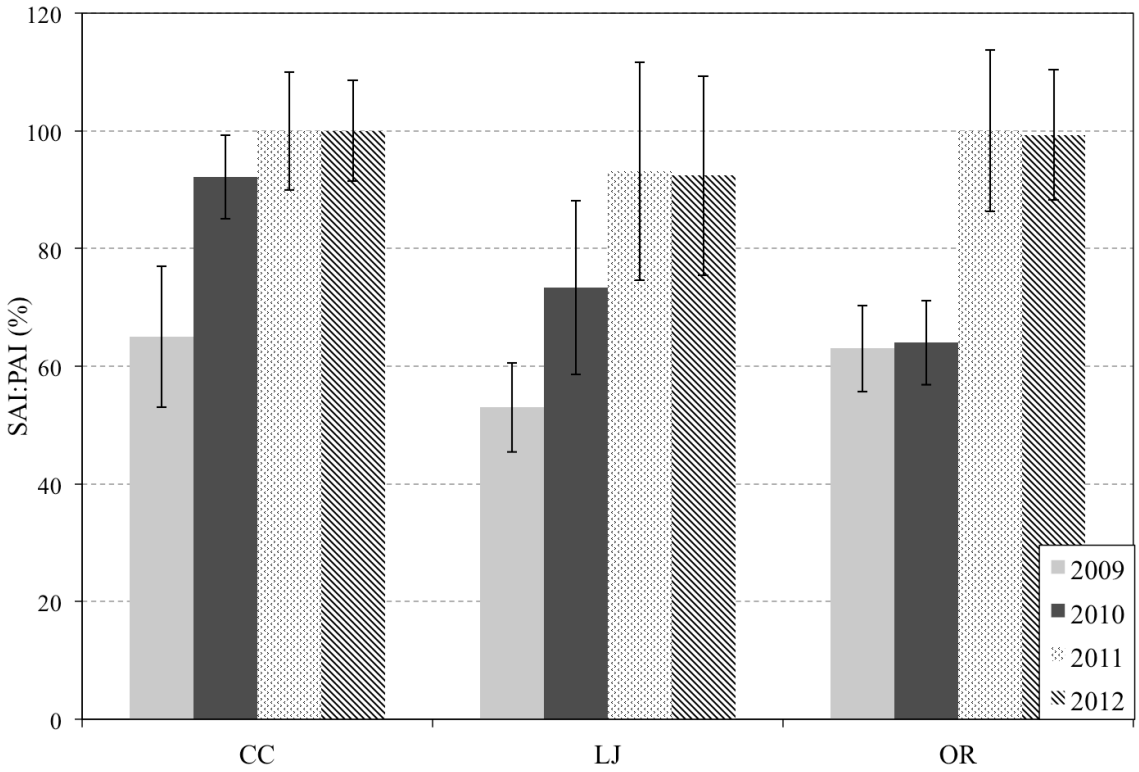

Figure 3.4. Percentage of the tamarisk canopy (plant area index (PAI)) that consists of woody stem materials (stem area index (SAI)). Measurements taken at three sites in Colorado for three years after 2009 helicopter imazapyr applications. Vertical bars represent ± one standard error (SE).

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CHAPTER 4. EFFICACY AND COSTS OF CHEMICAL, MECHANICAL AND BIOLOGICAL STRATEGIES FOR MANAGING THE NON-NATIVE RIPARIAN TREE TAMARISK (*TAMARIX* SPP.)

Summary

Tamarisk is one of the most widespread invasive tree species in arid and semi-arid portions of the United States and is commonly targeted for removal. Intensive tamarisk management began sixty years ago and the most commonly used control strategies have been combinations of chemical and mechanical methods. Despite this history and substantial investments in managing tamarisk little research has focused on the relative efficacy and costs of current control strategies. In this study we established sites in southeastern Colorado where we compared the effectiveness of aerial imazapyr applications to several strategies integrating tree biomass removal with secondary chemical and biological treatments. In the short term whole plant extraction caused 20% higher tamarisk mortality than aerial imazapyr applications or biomass mulching. Of secondary treatments evaluated, individual plant treatments (IPTs) of imazapyr caused higher mortality than either triclopyr IPTs or releases of tamarisk leaf beetles (*Diorhabda carinulata*). Aerial imazapyr applications alone were very cost effective, but when we accounted for subsequent removal of tree biomass this treatment strategy was less cost effective than primary mechanical treatments followed by biological control releases. This study revealed that there are strong trade-offs between treatment effectiveness and ultimate costs. Particularly for woody invasive species such as tamarisk, biomass removal is a critical component of successful long-term management efforts that should not be overlooked.

Introduction

Tamarisk (saltcedar, *Tamarix* spp.) are arguably some of the most abundant and frequently targeted invasive, non-native trees in North America (Friedman et al. 2005). There are roughly six species in this genus that are present primarily in the western United States, but recent evidence suggests that *T. ramosissima* Ledeb., *T. chinensis* Lour. and their hybrids are most common (Gaskin and Schaal 2002, Tellman 2002, Gaskin and Kazmer 2009). Tamarisk was intentionally introduced into the U.S. for ornamental purposes in the early 1800s and in the 1930s its spread accelerated as plants were cultivated and used for erosion control and sedimentation projects (Tellman 2002, Chew 2009). By the middle of the 1900s tamarisk began to be targeted for removal due to its reportedly disproportionate consumption of water, among other factors (Robinson 1965, Chew 2009). The first strategies that were used to control tamarisk were combinations of mechanical tree extraction and herbicides (Douglass et al. 2013). In the sixty years since, cultural practices such as flooding and fire have also been used, though with limited success (Fox 2001, Sprenger et al. 2001, McDaniel and Taylor 2003b). More recently, leaf feeding beetles in the genus *Diorhabda* were introduced as biological control agents and there is growing evidence that they may provide long term tamarisk control (DeLoach et al. 2003, Tracy and Robbins 2009).

Currently, the most common tamarisk management strategies include: applications of glyphosate, imazapyr or triclopyr herbicides (either by large scale aerial applications or targeted individual plant treatments (IPT)); mechanical tree extraction; mechanical mulching; and the release of *Diorhabda* beetles (Douglass et al. 2013). Particular control methods are typically chosen for specific projects based on a number of factors, including cost, availability and ease of accessing the targeted site (Shafroth et al. 2008). Ultimately none of these methods is reliably

100% effective in all cases. Perhaps the only consistent lesson has been that integrated management over several years is more successful than any single strategy (Duncan and McDaniel 1998, Douglass et al. 2013).

While there is strong evidence that tamarisk has negative ecological impacts it has been argued that the singular focus on managing tamarisk is as much due to political and cultural motivations (Duncan et al. 2004, Shafroth et al. 2005, Shafroth and Briggs 2008, Chew 2009, Stromberg et al. 2009). Regardless, over the past decades there have been considerable investments in tamarisk management, though not necessarily with realistic expectations of how tamarisk might respond to various control methods (Representatives 2005, Coalition 2009). Despite the scale of tamarisk control and research only a few studies have empirically evaluated tamarisk responses to management (McDaniel and Taylor 2003b). No published studies have done so using multiple, geographically distinct sites and temporally replicated sampling. Our research aimed to directly compare and contrast tamarisk responses to a suite of commonly used control methods. In this manner we established baseline growth responses and mortality that could be expected by those employing these strategies. Our research tested the following hypotheses: 1) aerial imazapyr applications will result in the highest overall mortality; 2) mechanical tree extraction will result in significantly less tamarisk re-growth than mulching; 3) secondary individual plant treatments using imazapyr will cause higher mortality than those using triclopyr; and 4) biological control releases will not result in significant defoliation or mortality during the study period.

Materials and Methods

Four sites in the Arkansas River watershed of southeastern Colorado were selected for this research (Canon City (CC), Florence (FL), La Junta (LJ) and Ordway (OR) (Table 4.1)). Sites were chosen due to the willingness of private landowners to be involved and the presence of large, evenly dense tamarisk infestations. At each site the perimeter of a 10-12 hectare area of tamarisk was mapped using a handheld GPS unit (Garmin 60CSx, Garmin International, Inc., Olathe, KS) and was then further divided into four equally-sized areas using ArcGIS (Release 9.1, Environmental Systems Research Institute (ESRI), Redlands, CA). Within each of these primary treatment areas six circular sampling plots (84 m^2) were randomly located in a spatially balanced design using the Reversed Randomized Quadrant-Recursive Raster (RRQRR) algorithm (Theobald et al. 2007) in ArcGIS 9.1. This probability-based survey design is statistically rigorous and allowed for the use of standard estimates of population characteristics (Theobald et al. 2007).

Experimental design and treatments

One of four primary treatments – an untreated control, aerial herbicide application, mechanical tree extraction, and mechanical mulching - was randomly assigned to an area. Evidence indicated that mechanically treated tamarisk would re-grow (McDaniel and Taylor 2003b) and so we intentionally included secondary follow-up treatments to the tamarisk re-growth in the experimental design. Three sampling plots in each mechanically-treated area were randomly assigned to one of two secondary treatments: targeted releases of *Diorhabda carinulata* Desbrochers (tamarisk leaf beetles) and individual plant treatments (IPT) of imazapyr

herbicide. In mulched areas we also included a third secondary treatment (IPT triclopyr applications) that was applied to three additional randomly located sampling sub-plots.

Aerial herbicide applications were made by a commercial helicopter pilot on 4 September 2009 (CC and PO) and 10 September 2009 (LJ and GA). Imazaypr (Habitat® + 1% Dyne-Amic® non-ionic surfactant) was applied at 1.12 kg ai ha^{-1} using CP-03 nozzles (CP Products, Inc., Tempe, AZ). Mean flight speed was 48 km hr^{-1} and applications were made 1-2 m above the tree canopy. At the time of application, temperature at the sites averaged 24 °C, relative humidity was 39% and the mean wind speed was 10.3 km hr^{-1}. On 4 January 2010 (CC and FL) and 15-17 March 2010 (LJ and OR) mechanical treatments were carried out. Mulching at all locations was completed by a commercial operator using a Prentice 2664 site preparation machine with a Fecon® Bull-Hog® hydraulic brush cutter head. Mechanical extraction of tamarisk trees was done by private contractors using large excavators (Caterpillar 320 BL (CC and FL), John Deere 690C (LJ and OR)) equipped with 76.2 cm thumbed buckets.

Diorhabda carinulata individuals were obtained from the Colorado Department of Agriculture Insectary at Palisade, CO. Five thousand beetles were released at each site in designated plots on the 4th and 5th of June 2009 and another 6,000 individuals per site released the 8th and 9th of June 2010. IPT foliar imazapyr (1%, V/V Habitat® + 0.25%, V/V non-ionic surfactant) treatments were made to all trees in designated sampling plots 11-15 September 2010 using a CO_2-pressurized backpack sprayer and a single nozzle (Teejet 4003E, Teejet Technologies, Wheaton, IL) handgun applicator. During applications, temperatures averaged 29.9 °C, relative humidity was 9.5% and the mean wind speed was 1.8 km hr^{-1}. IPT basal bark triclopyr (30%, V/V Garlon® 4 Ultra + 70%, V/V basal bark oil (JLB Oil Plus, Brewer International, Vero Beach, FL) applications were made to the bottom 45 cm of the stems of all

trees in designated plots 26 March – 3 April 2011 using the same equipment. At the time of applications, temperatures averaged 16.1 °C, relative humidity was 22.2% and the mean wind speed was 3.2 km hr^{-1}.

Data collection

Tamarisk tree responses to the seven treatments administered (including the untreated control) were measured quantitatively. The total number of plants (trees were considered to be separate if the root crown centers were more than 30 cm apart) and the total number of stems in each circular 84 m^2 sampling plot were counted annually. Distinctions were made between dead and living plants and stems, as well as mature (> 2 cm diameter at ground level) and immature (< 2 cm) stems. Thirty stems were randomly selected throughout the plot at each sampling interval and their diameters (cm) at roughly 15 cm above ground level measured. Additionally, ten points within the circular plot were randomly selected using a combination of angle (circle was divided into ten 36° radians) and distance (the distance between the plot center and the outside edge was divided into ten lengths) and permanently marked. At these permanent points tamarisk canopy height (m) and canopy area were measured. Canopy area was measured using a LI-COR LAI-2000 canopy area meter (LI-COR, Lincoln, NE) and measurements taken at the same point during the growing season (to determine plant area index (PAI)) and the dormant season (to determine stem area index (SAI)). The difference between these two measurements is considered to be a more realistic estimate of the living tree canopy cover (leaf area index, LAI) (Cutini et al. 1998). All data collection described above occurred prior to treatments in 2009 and then again in 2010, 2011 and 2012.

Treatment costs

Primary tamarisk control operations (aerial imazapyr applications, mechanical mulching and tree extraction) were performed by commercial contractors and the costs used for analysis were the actual invoiced charges at the time of treatment (2009-2011). Primary treatment costs did not reflect transportation of equipment to project sites (for the site preparation machine this averaged \$13.56 km^{-1}, for the excavators the average was \$24.75 km^{-1}), only the expense of tamarisk removal. Access to our project sites was relatively easy and only required 1-2 km of travel on maintained secondary roads off paved highways. Secondary treatment (*Diorhabda carinulata* bio-control releases, individual plant treatments (IPT) of imazapyr and triclopyr) costs included labor (2 staff members at \$15 hr^{-1}), herbicides, carrier (surfactant, oil and water) and spray dye at 2011 prices. Total expenses for *Diorhabda carinulata* releases were \$25 ha^{-1} for approximately 10,000 individuals.

Statistical analyses

Prior to analysis, data were checked for the presence of significant outliers and influential points using several methods including Cook's Distance values. A total of three data points were identified using this procedure and excluded. Data from the primary treatment areas (primary treatments, secondary treatments in mechanical removal areas) at each site were treated statistically as sub-samples. Data means from these sub-samples (n = 6 for primary treatments, n = 3 for secondary treatments) were averaged to calculate a single population mean for each treatment at each site. Location (n = 4) was used as the actual replicated unit of measurement for all analyses.

Indices of treatment effect relative to baseline 2009 data were calculated for both primary and secondary treatments for all measurement variables. A composite index of treatment effect was also calculated by averaging these responses (as %) across all seven measurement variables. To evaluate the relationship between treatment effectiveness and cost we calculated the ratio of this composite index of tamarisk growth response to per hectare treatment cost (% tamarisk control $\$^{-1}$). Mixed-effects models were used to test for significant differences due to primary and secondary tamarisk treatments, with 'site' set as a random variable. The restricted maximum likelihood (REML) method was used to estimate model variance components.

Data were tested for normality and homoscedasticity using the Shapiro-Wilk W and Levene tests, respectively, of model residuals. For all measurement variables except mean stem diameter these assumptions were violated and so data were either natural log transformed (plant density, canopy height, LAI and composite metric) or square root transformed (all other variables). If models indicated treatment differences (at $P < 0.05$) a Tukey's HSD test was used as an initial technique to compare means. To compare primary treatment effects a Dunnett's test was used to determine whether any of the treatments caused significant ($P < 0.05$) impacts compared to the untreated control mean. JMP (ver. 10.0.1, SAS Institute, Cary, NC) software was used for all statistical analyses.

Results

Primary treatment effects

Mechanical tree extraction resulted in the highest tamarisk mortality, causing an 80% ($P < 0.05$) reduction in tamarisk growth in one year and an additional 10% control by the third year after treatment (Figure 4.1). By 2010 mechanical mulching and aerial imazapyr applications both

reduced tamarisk growth by roughly 60% ($P < 0.05$). At the end of the study mechanical mulching resulted in 85% tamarisk mortality while aerial imazapyr applications caused 76% mortality. It is important to note that while reductions in tamarisk growth in 2010 alone were due solely to the primary treatment, tamarisk mortality in mechanically-treated plots in 2011 and 2012 were due in part to additional secondary treatments.

Before treatments were initiated there was approximately one tree every five m^2 with 4-7 stems (Table 4.2). Trees were on average 2.12 m tall, had a leaf area index of 0.42 and stems that were on average 4.6 cm in diameter. Aerial imazapyr applications reduced plant density 81% and stem density 61% over three years. Mean canopy height was reduced dramatically and the average diameter of surviving stems by 63% three years after treatment. Initially, mechanical mulching slightly increased plant densities, but with a net decrease in stem density. Re-growth following mulching resulted in a relatively homogenous canopy with 20 cm tall stems that were roughly 1 cm in diameter. Mechanical extraction significantly ($P < 0.05$) decreased average plant and stem densities; re-growing stems were less than 5 cm tall and slightly more than 0.5 cm in diameter.

Secondary treatment effects

Overall, individual plant treatments (IPTs) with imazapyr following either mechanical mulching or extraction resulted in 98% tamarisk control and an average of 89% increased mortality ($P < 0.0001$) over mechanical removal alone (Figure 4.2). IPT triclopyr treatments increased tamarisk control 74% ($P = 0.0003$). Following mechanical mulching, *Diorhabda* beetle releases resulted in 11% greater control in 2011 and 47% by the second year after release. After tree extraction beetles increased mortality only 25% in two years; none of the beetle effects on

tamarisk control were statistically significant. Analysis of changes in specific growth measurements (after being standardized by the relative change in untreated treatments for that year) showed variability in tamarisk responses to secondary treatments. All IPT herbicide treatments significantly ($P < 0.05$) reduced plant and stem densities, but beetle treatments did not further reduce stand densities beyond reductions due to primary mechanical treatments (Table 4.3). *Diorhabda* releases resulted in a 22% decrease ($P < 0.05$) in mean LAI in mechanically mulched plots and 5% in extraction plots. While IPT imazapyr treatments resulted in sustained or increased tamarisk control, IPT triclopyr treatments had less consistent effects. Triclopyr treatments reduced ($P < 0.05$) the density and size of living tamarisk trees, but resulted in only slightly reduced ($P < 0.05$) living stem diameter and an increase in LAI.

Treatment cost effectiveness

Of the primary tamarisk control treatments we evaluated aerial imazapyr applications were 37% less expensive than mechanical extraction and 81% less expensive than mechanical mulching ($P < 0.05$; Table 4.4). *Diorhabda carinulata* releases were substantially less expensive ($P < 0.05$) than any of the other secondary treatment methods we evaluated. Total estimated costs for mechanical biomass removal after aerial imazapyr applications were \$845 ha^{-1} and \$993 ha^{-1} for extraction and mulching, respectively. The cost of IPT herbicide treatments following primary mechanical removal of tamarisk biomass depended largely on the density of the re-growth because this was correlated with treatment duration and labor costs. IPT imazapyr treatments following tree extraction (\$576 ha^{-1}) was half the cost of either IPT treatment following mulching (mean = \$1,054 ha^{-1}). For IPT imazapyr treatments labor accounted for an

average of 82% of the cost and for IPT triclopyr treatments labor was 50% of the average cost (Data not shown).

When initial treatment cost was incorporated into the assessment of treatment effectiveness over time we found that aerial imazapyr applications alone were several times more cost effective than mechanical treatments followed by secondary control (0.07% tamarisk control $\$^{-1}$ (Figure 4.3a)). Mechanical treatments followed by *Diorhabda* bio-control releases were also relatively cost effective (0.01% tamarisk control $\$^{-1}$) despite their comparatively reduced tamarisk control. Mechanical mulching or tree extraction followed by IPT herbicide treatments were found to be the least cost effective (< 0.01% tamarisk control $\$^{-1}$). Using data from our study we extrapolated the additional cost effectiveness of mechanically removing the standing dead biomass of aerially sprayed tamarisk and found that this significantly reduced cost effectiveness (Figure 4.3b). However, the predicted cost effectiveness of aerial imazapyr treatment and biomass removal (0.016% tamarisk control $\$^{-1}$) was still greater than mechanical + IPT herbicide treatments (0.003% tamarisk control $\$^{-1}$).

Discussion

Aerial imazapyr applications resulted in 80-98% tamarisk mortality in three years when measured by any individual growth metric (Table 4.2). This confirms prior studies that have evaluated tamarisk mortality following aerial imazapyr applications using similar metrics (e.g. counts of living versus dead plants (Duncan and McDaniel 1998, Hart et al. 2005)). When mortality was evaluated more holistically as a composite metric of change, aerial imazapyr applications caused a 76% net reduction in tamarisk growth 3 YAT compared to baseline data (Figure 4.1). This composite metric is somewhat biased because we included growth responses

(e.g. stem diameter, canopy height) that were sensitive to aboveground biomass removal involved with mechanical tree control strategies. Also, it has been proposed that tamarisk mortality from aerial imazapyr treatments is highly dependent on tree size and stem densities (Duncan and McDaniel 1998). Similarly, in a companion study we found that variability in tamarisk mortality following aerial imazapyr applications was related to infestation characteristics (e.g. shape, density) and site conditions (e.g. flight hazards such as power lines, presence of desirable overstory tree species (Douglass 2013)).

Helicopter imazapyr applications were considerably less expensive (92% less than mulching, 37% less than extraction) but also resulted in lower overall final mortality (Figure 4.3a). Our aerial application expenses ($346 ha^{-1}) were 25% less than those reported for several projects on the Pecos River in Texas carried out within the past decade, but greater than other published project costs (McDaniel and Taylor 2003b, Hart et al. 2005, Barz et al. 2009). This variation is likely due to numerous factors, not the least of which was the introduction of generic herbicide formulations within the past few years, which has driven down the price of branded imazapyr formulations. In many situations the ultimate management objective for controlling tamarisk requires removing the standing biomass in addition to killing trees (Barz et al. 2009). We estimated that the added costs would be 44-88% of spraying alone, which would still be six times more cost effective than mechanical removal followed by IPT herbicide treatments (Figure 4.3b). It is important to note that after aerial imazapyr applications treated plants should remain undisturbed for a minimum of two years before biomass removal occurs in order to ensure complete translocation of the herbicide throughout the tree's root system (Taylor and McDaniel 1998, Douglass et al. 2013).

Mechanically extracting tamarisk trees resulted in the most immediate and sustained reduction (90%) three years after treatment (Figure 4.1). This level of mortality is slightly less than that reported by McDaniel and Taylor (2003a) for dual phase mechanical tamarisk removal involving a bulldozer followed by repeated root raking. The more extensive mechanical tree removal strategy also cost 25% more than the method we used. We found that extraction later in the winter (March versus January) – when soils were presumably warmer – more significantly reduced tamarisk re-growth. Removing the root crown and remaining root fragments cleanly is important to ensure mechanical control of tamarisk (McDaniel and Taylor 2003b). Trees that re-grew from mechanical extraction were smaller (48% fewer, 24% shorter, and 64% thinner stems) than other treatments (Table 4.2). Consequently, the costs of IPT imazapyr re-treatments for these areas were half of those areas where trees were simply mulched, and 46% less herbicide per tree was used (Table 4.4).

Mechanical mulching did not cause short term tamarisk mortality (Table 4.2), but did significantly reduce tree density and size. Timing was also important for mulching, as treatments carried out earlier in the winter resulted in 50% less tamarisk re-growth. Temporal variability in mulching effectiveness may have to do with the time when solute transport from roots is initiated, which might increase the ability of the root crown to re-sprout (Scifres 1980). Mulching alone was twice as costly as mechanical extraction and in order to ensure a reasonable level of tamarisk mortality a secondary treatment was required (Table 4.2). Costs of mulching at our sites were considerably higher than that reported for some projects on the Pecos River ($754 ha^{-1}) but well within the range estimated by the Tamarisk Coalition in a 2008 meta-analysis (Barz et al. 2009, Coalition 2009). The cost of mechanically removing tree and brush species is strongly

influenced by terrain and access, our sites probably represent 'best case' scenarios in that the terrain was relatively flat and access was easy (Shafroth et al. 2008).

Diorhabda carinulata releases made on trees re-growing from mulching were comparatively more effective, causing a three-fold greater decrease in LAI compared to excavated plots. *Diorhabda* bio-control releases in mulched areas were substantially less expensive than other treatment options, although the final mortality two years after treatment was ten times lower compared to IPT imazapyr treatments. IPT triclopyr treatments (which were as costly as IPT imazapyr re-treatments) were five times less effective two years after treatments, but still resulted in 90% control. The relatively high costs of IPT herbicide treatments appeared to be driven by tree density, size and other factors that increased labor costs. IPT imazapyr treatments made in excavated areas cost half as much as those made to mulched areas and required half as much spray solution. Minimizing the volume of either imazapyr or triclopyr needed for re-treatments not only reduced costs but also decreased the possibility of non-target effects. Negative non-target effects on understory plant communities can be especially pronounced with imazapyr.

It is important to note that we did not conduct any site clean up or seedbank preparation operations and so were not able to evaluate any added ecosystem impacts or costs from these practices. Our mechanical operations resulted in holes left from excavated trees and debris (1-3 cm diameter, 15-25 cm length) scattered throughout treated areas. Our sites are managed for cattle grazing and neither the holes nor the debris significantly impacted this purpose. In fact there is growing evidence that tamarisk debris can favor natural re-vegetation by potentially desirable or useful plant species (Lair 2006, Douglass 2013). For some land managers it would be necessary to conduct more extensive clean up operations, e.g. root raking or other clearing

that would remove debris, ensure more complete removal of tamarisk root fragments from the soil and better prepare the soil seedbed for restoration seeding. These operations would further increase the costs of tamarisk management, and highlight the importance of carefully considering ultimate management objectives when planning and budgeting for tamarisk removal (McDaniel and Taylor 2003b, Shafroth et al. 2008).

Conclusions

Our results largely supported our initial hypotheses regarding tamarisk mortality, but ultimately there were more substantial trade-offs between treatment effectiveness and cost than we had expected. Over the three years treatment responses were evaluated mechanical tree extraction actually resulted in higher mortality than aerial imazapyr applications or mulching of aboveground biomass. In regards to secondary plant treatments, IPT imazapyr treatments resulted in much higher tamarisk mortality than either IPT triclopyr treatments or *Diorhabda* releases. However, by the end of the study we were able to begin to measure biomass reductions due to beetle defoliation.

Ultimately our analyses indicated that in the absence of secondary biomass removal helicopter imazapyr applications alone were not necessarily cost effective. Management strategies that involved first removing tamarisk biomass were more costly in the short term but resulted in more reliable tamarisk mortality. Tree extraction substantially reduced the density and size of tamarisk re-growth and lowered the cost of subsequent individual plant herbicide treatments. Our data also suggested that conducting biomass removal strategies within optimal timing windows – considering both tamarisk physiology and environmental parameters such as soil temperature – was imperative to ensuring treatments are successful.

No single treatment was necessarily better or worse than another, but our study provides evidence that each treatment option has fairly predictable costs and probabilities of tamarisk control. Low per unit cost effectiveness did not correlate with consistent tamarisk control. Depending on a land manager's ultimate management or restoration objectives, one of the first decisions to be made when planning tamarisk management is whether and at what stage tree biomass will be removed. Our study did not test some commonly used methods for biomass removal – the most notable example is prescribed fire – but in general we found that biomass removal could be carried out either before or after treatments that will more directly kill tamarisk trees. Chemical control options were more effective in the 2-3 year time frame we tested, but broad spectrum herbicides in particular have the potential to cause ecologically and ultimately economically costly non-target effects that should be taken into account (Ortega and Pearson 2011, Douglass 2013).

Finally, tamarisk control and removal alone will not inevitably result in an ecologically desirable outcome if related ecosystem processes are not addressed. There is growing evidence that restoration of tamarisk-infested sites in the absence of larger scale modifications to hydrologic regimes and the removal of other environmental stressors (e.g. cattle grazing or recreational uses such as all terrain vehicles (ATVs)) is not likely to be sustainable (Taylor and McDaniel 1998, Stromberg and Chew 2002, Douglass et al. 2013). As with any invasive species management project, controlling tamarisk and rehabilitating affected sites will require careful planning, multi-tiered approaches and a long-term commitment (Shafroth et al. 2008).

Table 4.1. Location of tamarisk management project sites in southeastern Colorado, and brief descriptions of dominant soils.

Site	City	Latitude (°)	Longitude (°)	Soil Series	Soil Type
CC	Canon City	38.49080	-105.20189	Shingle	Very cobbly sandy loam
FL	Florence	38.37967	-105.03772	- - -	Sandy loam
LJ	La Junta	37.99278	-103.55008	Bankard	Sand
OR	Ordway	38.18254	-103.74780	Apishapa	Clay loam

Table 4.2. Tamarisk growth responses to primary plant treatments (UTC = untreated; AXE = mechanical mulching; EXT = mechanical tree extraction; IMZ = aerial imazapyr application) conducted in late 2009. Treatment means are back-transformed values (see text for details). Superscript, lower case letters indicate statistical comparison of primary treatment means by survey year (Tukey HSD test). Means sharing the same letter are not significantly different at $P < 0.05$. Asterisks indicate a comparison (Dunnett's Control test ($P < 0.05$)) of means between treatments and the untreated control within each survey year.

	UTC	AXE	EXT	IMZ
Plant Density (plants m^{-2})				
2009	0.25	0.13^{a*}	0.18^{a}	0.16^{a}
2010	0.18	0.16^{a}	0.08^{b*}	0.03^{b*}
2011	0.17	0.05^{b*}	0.04^{bc*}	0.02^{b*}
2012	0.18	0.06^{b*}	0.04^{c*}	0.03^{b*}
Stem Density (stems m^{-2})				
2009	4.39	6.92^{a*}	5.47^{a}	5.13^{a}
2010	3.54	1.44^{b*}	0.69^{b*}	1.60^{b*}
2011	4.23	0.71^{bc*}	0.38^{b*}	1.02^{b*}
2012	2.90	0.22^{c*}	0.27^{b*}	0.79^{b*}
Canopy Height (m)				
2009	2.18^{a}	1.81^{a}	2.00^{a}	2.50^{a}
2010	1.38^{ab}	0.17^{b*}	0.04^{b*}	0.24^{b*}
2011	1.18^{b}	0.06^{c*}	0.03^{b*}	0.06^{b*}
2012	1.28^{ab}	0.06^{c*}	0.05^{b*}	0.08^{b*}
Stem Diameter (cm)				
2009	4.42	4.52^{a}	4.16^{a}	5.39^{a}
2010	4.14	1.31^{b}	0.84^{b}	2.21^{b}
2011	3.93	0.82^{b}	0.59^{b}	1.07^{b}
2012	4.01	0.82^{b}	0.62^{b}	1.99^{b}
Leaf Area Index (LAI)				
2009	0.46^{ab}	0.31^{a}	0.39^{a}	0.54^{a}
2010	0.54^{a}	0.12^{b*}	0.06^{b*}	0.09^{b*}
2011	0.25^{bc}	0.01^{c*}	0.01^{c*}	0.01^{c*}
2012	0.12^{c}	0.01^{c*}	0.01^{c*}	0.01^{c*}

Table 4.3. Tamarisk growth responses to secondary plant treatments (AXE = mechanical mulching; DIO = *Diorhabda carinulata* release; EXT = mechanical tree extraction; IMA = IPT imazapyr application; TRI = IPT triclopyr application) conducted in 2009-10. Treatment means are back-transformed values (see text for details) relative to baseline 2009 data and standardized as percent (%) of changes in untreated control plots. Asterisks indicate a comparison (Dunnett's Control test ($P < 0.05$)) of means between treatments and the untreated control within each survey year.

	AXE + DIO	AXE + IMA	AXE + TRI	EXT + DIO	EXT + IMA
Plant Density					
2010	122.14	193.48	106.18	32.63*	54.88
2011	131.00	4.93*	18.09*	43.60	6.20*
2012	102.02	4.88*	28.37*	35.35*	6.79*
Stem Density					
2010	16.73	39.85	55.43	6.08*	9.07*
2011	50.16	1.12*	2.24*	7.43*	0.82*
2012	15.26*	1.29*	1.69*	6.86*	1.23*
Canopy Height					
2010	6.59*	13.27*	30.42*	3.47*	3.44*
2011	15.88*	3.14*	4.64*	6.08*	3.14*
2012	9.35*	2.79*	4.50*	9.07*	3.69*
Stem Diameter					
2010	37.16*	41.48*	34.99*	22.99*	26.16*
2011	37.53*	6.02*	19.79*	24.17*	8.80*
2012	36.79*	6.02*	20.60*	26.45*	8.37*
Leaf Area Index (LAI)					
2010	31.66	47.24	1.55*	10.13*	22.09*
2011	5.18*	3.30*	2.60*	3.69*	2.60*
2012	14.09*	6.14*	7.07*	7.14*	5.50*

Table 4.4. Average cost per hectare (mean ($) ± SE) of primary (AXE = mechanical mulching; EXT = mechanical tree extraction; IMZ = aerial imazapyr application) and secondary (DIO = *Diorhabda carinulata* bio-control release; IMA = IPT imazapyr application; TRI = IPT triclopyr application) tamarisk control treatments.

Primary Treatment	Mean Cost ($ ha^{-1} ± SE)	Secondary Treatment	Mean Cost ($ ha^{-1} ± SE)	Estimated Total Costs ($ ha^{-1})
AXE	1,862.79 ± 77.70	+ DIO	33.65 ± 1.24	1,896
		+ IMA	1,010.23 ± 184.44	2,873
		+ TRI	1,096.55 ± 223.41	2,959
EXT	944.78 ± 80.27	+ DIO	33.65 ± 1.24	978
		+ IMA	576.01 ± 108.39	1,521
IMZ	345.80 ± 0.00	+ AXE	647.47 ± 12.58	993
		+ EXT	498.80 ± 13.00	845

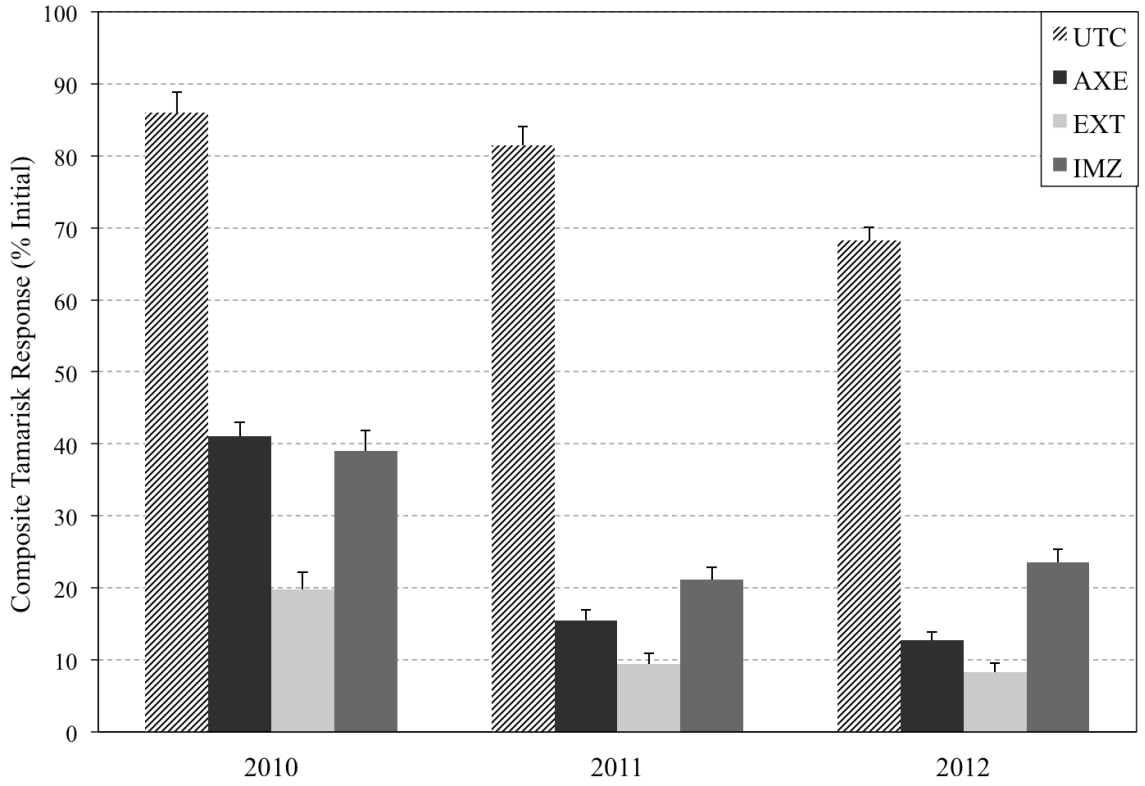

Figure 4.1. Overall effect of primary control methods on tamarisk (measured by a composite index of change relative to baseline 2009 measurements). Vertical bars represent one standard error (SE).

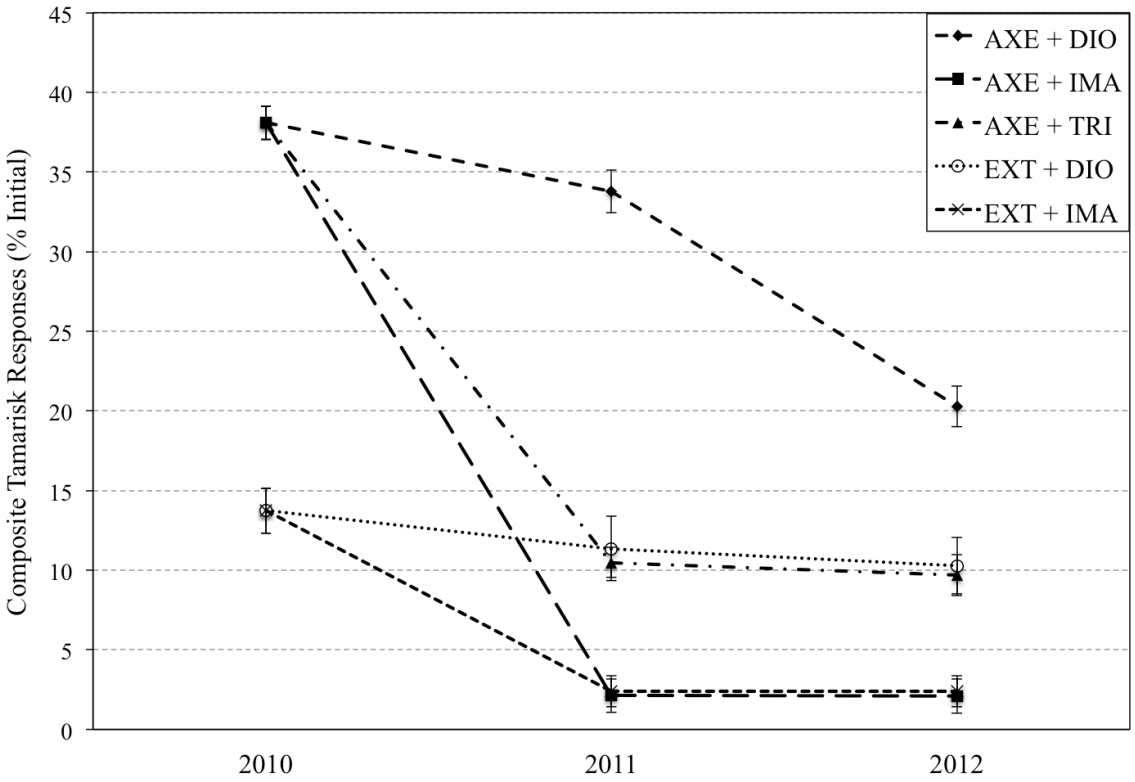

Figure 4.2. Overall effect of secondary control methods on tamarisk growth (measured by a composite index of change relative to baseline 2009 measurements). Note that: AXE = mechanical mulching; DIO = Diorhabda bio-control releases; EXT = mechanical extraction; IMZ = IPT imazapyr treatments; TRI = IPT triclopyr treatments. Vertical bars around each mean represent ± one standard error (SE).

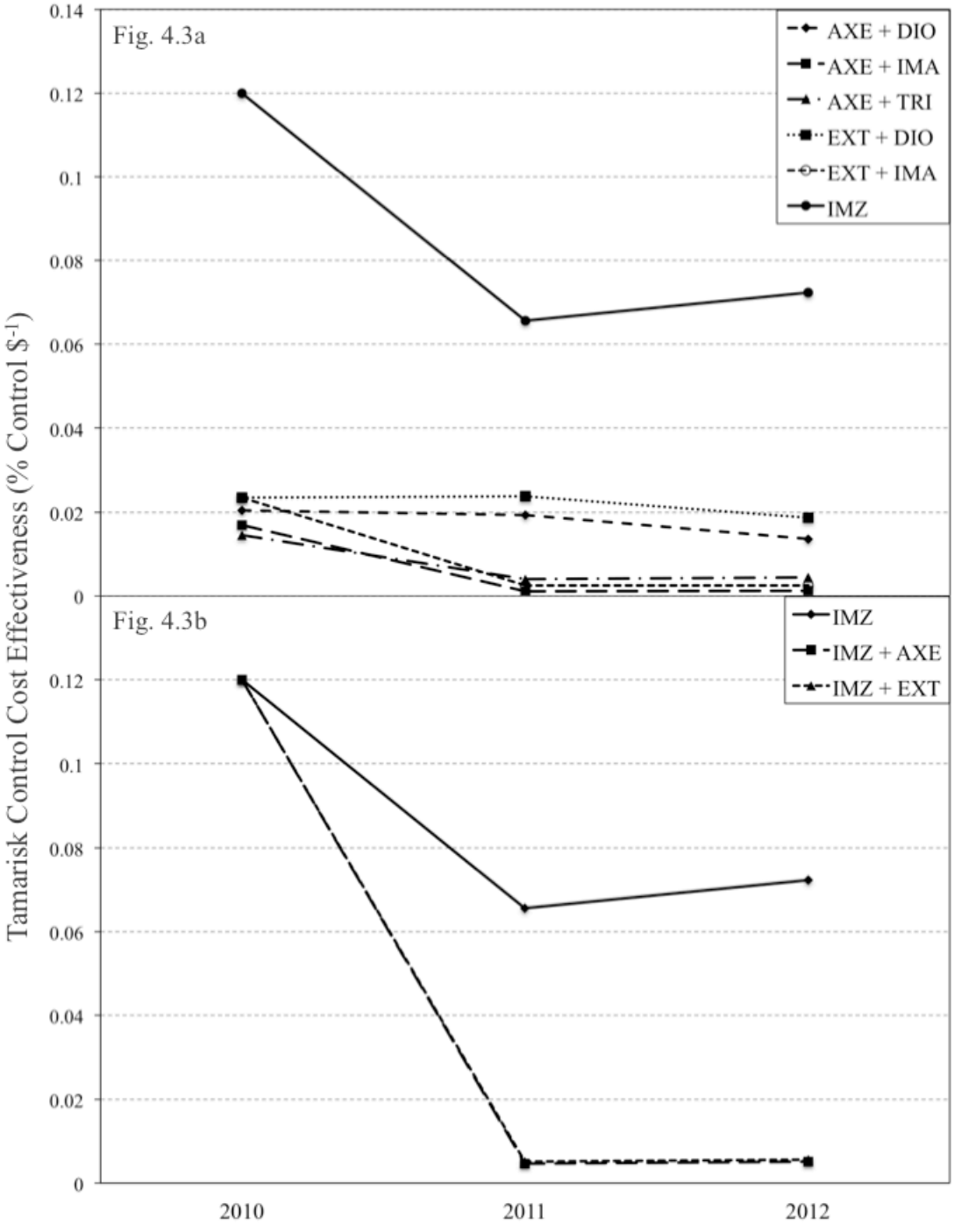

Figure 4.3. Ratio of tamarisk control (measured by a composite metric of tamarisk control (%)) to actual (Fig. 4.3a) and predicted (Fig. 4.3b) treatment costs ($ ha^{-1}). Note that: AXE = mechanical mulching; DIO = Diorhabda bio-control releases; EXT = mechanical extraction; IMZ = IPT imazapyr treatments; TRI = IPT triclopyr treatments.

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CHAPTER 5. PLANT COMMUNITY RESPONSES TO REMOVAL OF THE INVASIVE TREE TAMARISK (*TAMARIX* SPP.) USING CHEMICAL, MECHANICAL AND BIOLOGICAL CONTROL STRATEGIES

Summary

Tamarix species are now some of the most common woody plants along waterways in the western United States. They were intentionally introduced, but tamarisk's expansion has been facilitated by altered hydrologic regimes and increasingly by global climate change. In the 1950s the species started to be perceived as noxious and was targeted for control. Management of tamarisk has occurred by many methods, the more common of which have involved combinations of herbicides and mechanical tree removal. Ecologically based integrated pest management (EBIPM) models have recently been proposed that argue that weed control strategies have the capacity to re-direct successional processes. In this study we aimed to evaluate whether this was a valid model for tamarisk management by empirically testing plant community responses to different tamarisk control strategies.

We established field plots at four sites in southeastern Colorado and treated tamarisk stands with either an aerial herbicide (imazapyr) application or integrated mechanical biomass removal followed by secondary herbicide or biological control treatments. Plant community dynamics in response to the treatments were evaluated over three years. Helicopter imazapyr applications severely reduced plant community richness, diversity and abundance and appeared to promote invasion by herbicide-resistant populations of *Bassia scoparia*. Plant communities did not show a strong response to integrated tamarisk management, which in itself was notable because mechanical tree removal caused soil disturbances that in theory would have promoted secondary invasions of existing noxious plants. Ultimately, data suggested that plant community re-vegetation patterns following tamarisk removal were most strongly affected by climate (i.e.

drought) and shifts in the abundance of wetland plant species in response to watershed-scale hydrologic changes. These results provide some evidence for the need to re-evaluate basic models of ecosystem structure and functioning that are fundamental to the EBIPM framework in order for this tool to be valuable in managing tamarisk and other woody invaders.

Introduction

Tamarisk species (*Tamarix* spp.) were intentionally introduced into the United States from the early 1800s through the mid twentieth century (Tellman 2002, West and Nabhan 2002). *T. ramosissima* Ledeb., *T. chinensis* Lour. and hybrids of these two species are the third most common woody species in riparian areas of the western United States and were estimated to occur on 21% of perennial stream length in arid portions of the region (Friedman et al. 2005, Ringold et al. 2008). By the early 1950s federal hydrologists began to implicate tamarisk as a disproportionate water consumer and widespread efforts began to remove and control these species using chemical, mechanical and other means (Robinson 1965, Subcommittee 1970, Chew 2009). In the ensuing years, dozens of control strategies have been used against tamarisk, including cultural practices such as deliberately timed floods and prescribed fires (Douglass et al. 2013). Most recently, the release of tamarisk leaf beetles (*Diorhabda* spp.) as biological control agents has appeared promising in providing long term, sustainable tamarisk control (DeLoach et al. 2003, Tracy and Robbins 2009). Currently, the most common tamarisk management strategies include: herbicide applications (glyphosate, imazapyr or triclopyr) either by large scale aerial applications or targeted individual plant treatments (IPT); mechanical tree extraction; mechanical mulching or selective removal of aboveground biomass; and the release of *Diorhabda* beetles (Douglass et al. 2013).

Control methods are typically chosen for specific projects based on a number of factors, including cost, availability and ease of accessing the targeted site (Shafroth et al. 2008). Research related to tamarisk management and restoration has frequently focused on optimizing tamarisk mortality and improving the success of efforts to re-establish native riparian phreatophytes (Taylor and McDaniel 1998, Bay and Sher 2008). Much of this research presumes that the targeted ecosystem still has the hydrologic connectivity and overall ecosystem health to support native woody phreatophytes such as *Populus deltoides* W. Bartram ex Marshall and *Salix* L. species. There is growing evidence though that due to climate change, decades of flow modifications and other factors contributing to site degradation, riparian habitats in arid and semiarid western river systems may no longer have the capacity to support native woody species (Briggs et al. 1994, Merritt and Poff 2010, Reynolds and Cooper 2011, Perry et al. 2012).

Tamarisk removal itself is very costly, and the additional expense of active restoration efforts over the large areas where tamarisk is frequently managed often make such projects cost prohibitive (Barz et al. 2009). One strategy cited to promote post-treatment site rehabilitation in the absence of active restoration is to prioritize certain sites for management based on their potential for passive re-vegetation (Taylor and McDaniel 2004, Shafroth et al. 2008, Douglass et al. 2013). Yet there is little empirical evidence that establishes how commonly used tamarisk removal strategies impact plant species functional groups or particular species of interest, such as secondary invasive species. Harms and Hiebert (2006) analyzed the impacts of using prescribed burns and cut stump removal of tamarisk in the southwest U.S. and found very little passive re-vegetation following either treatment compared to reference sites where tamarisk was absent. However, their study did not include common tamarisk removal methods such as aerial herbicide treatments and mechanical tree removal methods.

Therefore, our study sought to determine the effect of tamarisk removal methods (chemical, mechanical or biological) on passive plant community succession following treatments. By understanding these interactions we could more appropriately select integrated tamarisk management strategies that best conserve and promote the ecological resilience of sites targeted for management (Suding et al. 2004, Pearson and Ortega 2009). Our study sought to indirectly test a core principal of ecologically based integrated plant management (EBIPM), namely that adaptive, integrated weed management (IWM) strategies can positively effect succession-mediated ecosystem recovery (Sheley et al. 2006, Sheley et al. 2010). These models and management frameworks were developed in rangelands of the western U.S. and so their applicability to the relatively more diverse habitats invaded by tamarisk (including riparian areas and upland grasslands) is largely unclear. Furthermore, management of invasive woody species, and particularly trees, essentially requires two equally important phases: mortality; and removing the resulting biomass (Douglass et al. 2013). This adds more complexity to ecosystem recovery and arguably more opportunities for perturbations of natural ecosystem recovery, thereby likely reducing the predictably of successional trajectories necessary for the successful application of EBIPM frameworks.

In this study we wanted to quantitatively establish the relative effects of selected tamarisk control methods on associated understory plant communities. We predicted that: 1) aerial imazapyr applications would significantly inhibit any and all re-vegetation; 2) mechanical tamarisk biomass removal would favor the post-treatment recruitment of weedy noxious plant species; 3) that biological control of tamarisk plants would not negatively impact understory plants compared to targeted herbicide applications; and 4) that the targeted application of a non-

selective herbicide (imazapyr) would have a greater deleterious impact on passive re-vegetation than use of a selective herbicide (triclopyr).

Materials and Methods

Study area

This study was carried out on private land at four sites in the Arkansas River watershed of southeastern Colorado (Table 5.1). Two of the sites (Canon City (CC) and Florence (FL)) were located along tributaries in the upper (upstream of Pueblo) portion of the watershed. The Canon City site was roughly 2.5 km north east of town and adjacent to the intermittent Four Mile Creek. The site had a gravelly soil substrate and a native grass (*Sporobolus* R. Br., *Distichlis spicata* (L.) Greene) - *Juniperus scopulorum* Sarg. - *Populus deltoides* plant community. The Florence site was 7 km east of town at a small oxbow in Hardscrabble Creek and had a plant community largely composed of warm season grasses, shrubs (*Atriplex* L.) and native phreatophytes along the creek banks.

The other two sites were located downstream of Pueblo in a portion of the watershed where the hydrology is more strongly affected by water diversions for agricultural irrigation and urban development (Merritt and Poff 2010). The OR site was 3.4 km south of Ordway and 1.25 km west of the Lake Meredith reservoir, and was an arid rangeland site with a salt meadow community dominated by *Sporobolus airoides* (Torr.) Torr., *Pascopyrum smithii* (Rydb.) A. Love and *D. spicata*. La Junta (LJ) was a sandy site in the floodplain on the north bank of the main Arkansas River channel. The site had a mixed understory of *D. spicata* - native forbs (e.g. *Helianthus* L. and *Ratibida tagetes* (James) Barnhart) - weedy annual species (e.g. *Bassia scoparia* (L.) A.J. Scott, *Ambrosia psilostachya* DC.) and scattered *P. deltoides*. All of our sites

were affected by cattle grazing, with the strongest and most regular grazing pressure at LJ and OR, and infrequent grazing at CC and FL.

Published reports indicate that *Tamarix* spp. first appeared along the Arkansas River in Colorado near the town of Lamar between 1905 and 1913 (Lindauer 1983). Tamarisk infestations downstream of Pueblo are generally older than those closer to the Rocky Mountain and Sangre de Cristo foothills (Lindauer 1983, Lovell et al. 2009). According to our observations and anecdotal reports from land owners, the infestations at our sites were established in individual flood events during the past several decades. Tamarisk stands had homogenous densities (2,500 plants ha^{-1}) and there was little evidence of seedling recruitment since the initial population establishment decades ago. The mean canopy height was 2.18 m and average stem (mean diameter = 4.42 cm) density was 4.39 stems m^{-2} (Douglass 2013).

Experimental design and treatments

At each site the perimeter of a 10-12 hectare area of tamarisk was mapped using a handheld GPS unit (Garmin 60CSx, Garmin International, Inc., Olathe, KS). GIS analysis (ArcGIS, Release 9.1, Environmental Systems Research Institute (ESRI), Redlands, CA) was used to further divide the chosen areas into four 2-3 hectare plots. Within each of these plots, six circular, multi-scale sampling sub-plots (84 m^2) were randomly located in a spatially balanced design using the Reversed Randomized Quadrant-Recursive Raster (RRQRR) algorithm (Theobald et al. 2007) in ArcGIS 9.1. Application of this statistically rigorous probability-based survey design allowed for the use of standard estimates of population characteristics (Theobald et al. 2007). The circular, multi-scale sampling sub-plots were modified from Barnett et al. (2007) to better suit the low, dense canopy typical of tamarisk infestations.

One of four primary tamarisk removal methods - an untreated control, aerial herbicide application, mechanical tree extraction, and mechanical mulching - were randomly assigned to each plot. Based on previous studies we expected that mechanically-treated trees would re-grow and so we included secondary follow-up treatments to the tamarisk re-growth in those areas (McDaniel and Taylor 2003). Sampling sub-plots were randomly designated for applications of one of two secondary treatments: targeted releases of *Diorhabda carinulata* Desbrochers (tamarisk leaf beetles); or individual plant treatments (IPT) of imazapyr herbicide. In mulched areas we also included a third secondary treatment (IPT triclopyr applications) that was applied to three additional randomly located sub-plots.

More detailed information on treatments and environmental conditions at the time of treatments are given in Douglass et al. (2012). To summarize, aerial imazapyr (1.12 kg ai ha^{-1} Habitat® + 1% non-ionic surfactant) applications were made by a commercial helicopter pilot in early September 2009. Mechanical treatments were carried out by private contractors in January-March 2010 while trees were dormant. Mulching at all locations involved the use of a "Hydro-axe" (site preparation machine with a Fecon® Bull-Hog® hydraulic brush cutter head) that shredded all aboveground tree biomass. Whole trees were extracted by large excavators equipped with thumbed buckets. Tamarisk leaf beetles were obtained from the Colorado Department of Agriculture Insectary (Palisade, CO), and 5,000 individuals were released in designated sub-plots at each site in June 2009 and 2010. IPT foliar imazapyr treatments (1% V/V Habitat® + 0.25% V/V non-ionic surfactant) were made to all trees in designated sub-plots in September 2010 using a CO_2-pressurized backpack sprayer and a single nozzle handgun. IPT basal bark triclopyr applications (30% V/V Garlon® 4 Ultra + 70% V/V citrus-based basal bark oil) were made to the bottom 45 cm of the stems of all trees in designated sub-plots in early April 2011.

Soil seedbank sample collection

In order to capture the pre-treatment re-vegetation capacity of the understory plant community, soil seedbank samples were collected in all sub-plots in August 2009. Next to each 1 m^2 frame and at the sub-plot center a 10 X 10 X 5 cm soil sample was collected, placed in a Ziploc bag and stored on ice for transport. A modified seedling emergence method was used to study seedbank plant communities (Ter Heerdt et al. 1996). Soils were dark stratified in a cold room (3-5 °C) for six weeks to standardize germination. Each soil sample was coarsely (2 mm) sieved to remove stones and debris and then 'concentrated' by washing the remaining soil through a fine sieve (0.211 mm). The soil slurry that remained was then poured in a layer (3-5 mm) on top of 4 cm of steam-sterilized potting soil in a 15 X 10 X 5 cm tray insert.

Potted seedbank samples were kept in a climate-controlled greenhouse (31.99 ± 5.83 °C, 33.40 ± 13.49 % RH (mean ± SD)) for 6 months and were watered as necessary to keep the soil moist. As seedlings emerged they were identified to species, counted and then removed. Seedlings that could not be definitely identified were allowed to grow until they flowered and identification was possible. Voucher specimens containing reproductive structures were prepared for all plant species and used to verify identifications in consultation with staff of the Colorado Natural Heritage Program and the Colorado State University Herbarium, Fort Collins, CO.

Vegetation data collection

Canopy cover (%), plant density (plants m^{-2}), and absolute cover classes ('vegetation,' 'bare ground' or 'litter') were recorded for three 1 m^2 frames in each sub-plot (Figure 5.1). Species presence was recorded for both the frames and the entire sub-plot. Vegetation surveys took place in 2010, 2011 and 2012. The distance (m) to the nearest tree treated with either

imazapyr or triclopyr was measured from the corners of each vegetation sampling frame. As a coarse measurement of IPT herbicide effects on understory plant species the radius (cm) of the "herbicide impact zone" occurring in 2011 and 2012 was measured from the stem of treated trees in each plot to the nearest individual plant. In imazapyr-treated plots *B. scoparia* plants were excluded from this analysis and in triclopyr-treated plots all monocots were excluded. In both of these cases the species were excluded because they were known to be resistant or tolerant to the respective herbicides.

All plant species nomenclature followed the USDA PLANTS Database (USDA 2012). Species from both field and seedbank studies were assigned taxonomic and life history characterizations (duration, growth habit and native status) according to the PLANTS Database (USDA 2012). Plant species regulated by the Colorado Department of Agriculture as "noxious" were also identified (CDA 2012). For monocotyledonous species the photosynthetic pathway (C3 or C4) was determined using Waller and Lewis (1979). The wetland indicator status (WIS) for each species followed the 2012 National Wetland Plant List indications for the "Great Plains" region (or region 5 in previous lists (Lichvar and Kartesz 2012)). Using this status a per plot weighted average wetland indicator index was calculated by multiplying the assigned scores (one for obligate wetland species, two for facultative wetland, three for facultative, four for facultative upland, and five for obligate upland species) by the relative plant density (field and seedbank studies) and canopy cover (field study) for species within each class (Stromberg 2001, Reynolds and Cooper 2011). For each plot the Shannon-Wiener diversity index (H') was calculated using both stem density (field and seedbank studies) and cover (field studies) as the basis for determining the relative individual proportions (Magurran 2004).

Soil and climate data collection

In August 2009 additional soil samples were collected from each sub-plot at the four sites for determination of mean soil pH, organic matter (%), cation exchange capacity (CEC, meq 100 g^{-1}), salinity (EC, μS cm^{-1}), sodicity (sodium absorption ratio, SAR) and texture. A 6.5 cm diameter soil auger was used to collect five, 10 cm deep sub-samples from random points in each sub-plot. The sub-samples were pooled together, air dried for 72 hours and then a portion removed and sent to AgSource Laboratories (Lincoln, NE) for analysis.

Climate data were obtained from US National Oceanic and Atmospheric Administration (NOAA) National Climate Data Center (NCDC 2012). Annual climatological summary data for July 1979 to July 2012 was acquired for the NOAA weather station closest to each project site. 'Annual' precipitation was calculated based on sampling intervals (i.e. August – July) because sampling took place in July each year between 2009 and 2012. Data from 1979 to 2008 were used to calculate 30 year historical temperature and precipitation averages for the four sites for the time period immediately preceding project implementation. The distance from weather stations to actual treatment areas was 3.48 km for CC, 8.61 km for FL, 1.30 km for LJ, and 4.72 km for OR.

Statistical analyses

Data from the respective treatment sub-plots (primary treatments, secondary treatments in mechanical removal areas) at each site were treated statistically as sub-samples. Means for each plant community measurement (species $plot^{-1}$, $e^{H'}_{Cover}$, $e^{H'}_{Density}$, WIS_{Cover}, $WIS_{Density}$ and relative proportions based on native status, duration, growth habit, regulatory status and monocotyledonous photosynthetic pathway) were averaged across frames and sub-plots to

calculate a treatment plot mean for each site. Primary treatment means were the average of six sub-plots, and for secondary treatments means were the average of three sub-plots. Data within each of the four sites were pooled for statistical analyses and locations were treated as replicates (n = 4).

Data were tested for normality and homoscedasticity by applying Shapiro-Wilk W and Levene tests, respectively, to model residuals. Diversity ($e^{H'}$) and WIS data did not meet these assumptions and were natural log-transformed. Several binomial datasets (proportion of C4 species, proportion of annual species, and proportion of forbs) were logit ($\log(y/[1-y])$) transformed. For photosynthetic pathway data the value 1/(2*n) (where 'n' is the number of individuals in the population) was subtracted from the numerator and denominator of the function to correct for the bias of sample proportions equal to 1 (Warton and Hui 2011). Individual plant species cover (%) and density (number stems m^{-2}) data were weighted by the relative abundance of each species and log transformed before analysis (Klimesova et al. 2011).

Mixed-effects models were used to test for significant differences due to primary and secondary tamarisk treatments, with 'site' set as a random variable. The restricted maximum likelihood (REML) method was used to estimate variance components in the model. If models indicated treatment differences ($P < 0.05$) means were compared using a Tukey's HSD test. To compare primary treatment effects a Dunnett's test was used to determine whether any of the treatments caused significant ($P < 0.05$) impacts compared to the untreated control mean. Principal components analysis (PCA) was used to identify whether the highly correlated edaphic and environmental parameters affected plant species richness and diversity. PCA was conducted on correlations for the five measures of soil chemistry, the three variables measuring soil texture and the four climate variables. The number of principal component axes to retain for analysis

was determined using scree plots of eigenvalues because interpretable components were easily identified for our dataset (Jackson 1993). JMP (ver. 10.0.1, SAS Institute, Cary, NC) software was used for all statistical analyses.

Results

Seedbank and untreated plant communities

One hundred species were identified in the soil seedbank and 153 in field samples (Table A.4). Fifty-five species were found in both seedbank and field samples. *Bassia scoparia* was the most abundant species, followed by the native grass *Distichlis spicata* (Table 5.3). Native species that were abundant in both sample sets included several *Sporobolus* species, *Glycyrrhiza lepidota* Pursh and *Grindelia squarrosa* (Pursh) Dunal. Five of the most abundant species were "noxious" species in Colorado, with the most common being tamarisk, *Acroptilon repens* (L.) DC and *Bromus tectorum* L.

Mean species richness in seedbank samples (12.88 ± 0.42 species plot^{-1} (mean ± SE)) was not significantly different from that of untreated field plots in 2010 (13.13 ± 1.27). Richness in the same field plots decreased significantly ($P < 0.05$) in 2011 to 7.88 (± 1.13) species plot^{-1} and rose slightly in 2012 (9.79 ± 0.98 (Table 5.2)). Plant diversity in seedbank samples ($e^{H'}_{Density}$ = 5.25 ± 0.24) was higher than that of field plots (mean = 3.02 ± 0.34). Weighted average WIS did not differ between the seedbank (3.74 ± 0.04) and field samples in 2010 or 2011 (3.68 ± 0.1), but was significantly ($P < 0.05$) higher in the field in 2012 (3.92 ± 0.11). The relative proportion of native species was higher in seedbank samples (54.6 ± 1.4%) than in field samples for all surveyed years (46.6 ± 4.9%). Annual species were over-represented in seedbank samples (49.4

± 1.4%) compared to the field (30.9 ± 4.1%). Forbs were especially abundant in the seedbank (73.2 ± 1.1%) relative to field plots (20.9 ± 2.8%).

Plant community responses to primary tamarisk management strategies

Compared to pre-treatment seedbank samples aerial imazapyr applications (IMZ) resulted in a 64% decrease in species richness (Figure 5.2). Relative to the untreated control this treatment reduced species richness 68% in 2010 (Figure 5.2). Similarly, vegetative cover (71%) and cover-based diversity ($e^{H'_{Cover}}$: 77%, $P = 0.04$) were reduced in 2010 (Figure 5.3). Density-based diversity ($e^{H'_{Density}}$) was reduced an average of 54% ($P < 0.05$) in 2010 and 2011.

Among functional groups only annual forbs shifted in their relative abundance. In 2010 and 2011 there were no differences among treatment plots in the relative proportion of annual forbs, but by 2012 the relative proportion of annual forbs in plots following the aerial herbicide treatment was double that in other plots (Figure 5.4). This shift appeared to be due largely to *Bassia scoparia*, which became significantly ($P < 0.05$) more abundant in aerial imazapyr plots than elsewhere by 2012 (Figure 5.5).

In 2010 there were five other species (*Opuntia phaecantha* Engelm., *S. airoides*, *Carduus nutans* L., *Iva axillaris* Pursh and *Muhlenbergia asperifolia* (Nees & Meyen ex Trin.) Parodi) with average cover equal to *B. scoparia*, but by the end of the study only *O. phaecantha* and *Tamarix* had similar cover. The relative stem density of *B. scoparia* stems increased 16% over the course of the study (mean = 11.88%) while that of every other species declined (Data not shown). The species' average frequency weighted stem density increased ($P < 0.05$) from 5.13 stems m^{-2} in 2010 to 240.76 stems m^{-2} in 2012. This final density was 17 times greater than the next densest species (*Distichlis spicata* (14.01 stems m^{-2})). Two other species that were

relatively unaffected by aerial imazapyr applications were the native legume *Glycyrrhiza lepidota* and the native grass *D. spicata*. The weighted stem density of the former species increased 12 times over the study period (Table 5.4). Although mean cover of *D. spicata* did not change significantly, the species density doubled by the end of the study.

Many grass species (e.g. *B. tectorum*, *D. spicata*, *S. airoides* and *S. cryptandrus* (Torr.) A. Gray) responded positively to mechanical tamarisk removal. Relative to untreated control plots, *S. airoides* (+33%) and *S. cryptandrus* (+92%) stem densities increased most notably. Mechanical mulching of tamarisk trees resulted in a debris layer with an average cover of 49% and a depth of 1.45 ± 0.36 cm (mean ± SD). The presence of debris initially increased plant species richness 83% ($P < 0.05$), but by the following year plots in which debris had been cleared had 52% ($P < 0.05$) more plants per area than those with remaining debris (Data not shown)).

Plant community responses to secondary tamarisk management strategies

The mean distance from vegetation sampling plots to the nearest herbicide-treated tree was 3.69 ± 1.69 m for imazapyr and 2.75 ± 1.03 m for triclopyr. There was no significant difference in the distance to the nearest treated tree between the two herbicides. The mean "herbicide impact zone" differed (but not significantly) between the two herbicides ($radius_{imazapyr}$ = 109 ± 38 cm; $radius_{triclopyr}$ = 42 ± 20 cm).

When year to year changes in climate were accounted for by calculating a mean index of change in plant species abundance (a composite of percentage changes in density and richness) relative to the untreated control, data indicated that IPT herbicide treatments following mechanical tamarisk removal were more injurious to the plant community than *Diorhabda* releases (Figure 5.6). Beetle releases following whole plant extraction actually resulted in an

overall positive effect on plant richness and diversity when results from 2011 and 2012 were averaged. In contrast, IPT herbicide treatments in 2011 had a substantially negative effect (-0.3X UTC) on plant communities. The effects of IPT triclopyr treatments appeared to be minimized the following year, when the relative magnitude of the negative impact was significantly ($P < 0.05$) less.

Influence of climate and soils

Precipitation in the 2010-11 growing season was 51% lower ($P < 0.05$) than the historic mean (Figure 5.7). FL and OR were the drier of the four sites (mean = 22.6 cm), while LJ was the wettest (31.8 cm). There was little temperature variability among the sites, though CC tended to be cooler during the growing season and OR was colder during winter months (Data not shown). In 2010-11 the mean temperature across the watershed was 20% higher ($P < 0.05$) than historic averages; otherwise mean temperatures were consistent with historic trends.

PCA suggested that soil texture and salinity affected species richness and diversity (Data not shown). The first principal component axis summarized the influence of edaphic variables and explained 51% of the variance for both richness and diversity. The second axis represented climate parameters and explained roughly 25% of the variation for both variables of interest. Relative clay content in soils most negatively influenced species richness, particularly in the 2009-2010 and 2010-2011 growing seasons when precipitation was greater. Soil salinity and mean annual temperature both had strong effects on richness as well. While specific loading scores differed for diversity measurements the general magnitude and direction of trends was identical.

Discussion

Seedbank and untreated plant communities

Overall the soil seedbank and extant vegetation were very similar with respect to common plant species and broad measures of community composition we measured. Perennials and woody species were under-represented in the seedbank, which is likely due to a lack of proper seed treatment methods for these species (Coffin and Lauenroth 1989, Gross 1990). Diversity differed between the seedbank and the field and this was especially the case for stem density-based diversity. This hints at what is perhaps the most significant difference between the seedbank and extant vegetation over the three sampling years – below average precipitation.

When precipitation was similar to the 30 year average the seedbank and field plant communities did not differ. In contrast, extant vegetation differed significantly from that in the seedbank in 2011 when precipitation was considerably below the historic average for all sites (Figure 5.7). This was also the case in 2012 when precipitation at LJ and OR were 36% below historic average. Two consecutive years of drought likely influenced the significant ($P < 0.05$) difference in weighted average WIS scores between seedbank samples and field data from 2012.

Plant community responses to primary tamarisk management strategies

Of all treatments tested, aerial imazapyr applications resulted in the most severe immediate effects on understory plant communities. Species diversity and richness did increase somewhat from severe impacts the year following herbicide treatment, but even in 2012 richness and diversity in aerially treated plots was half that of untreated plots (Figures 5.2 & 5.3). There was essentially no effect on plant communities from the two mechanical tree removal methods we tested, but the removal of debris created through tree mulching decreased long term species

recruitment. Our results differ from Harms and Hiebert (2006), who found increased species diversity following targeted tamarisk removal. Our sites could have a greater degree of abiotic degradation and therefore a lower overall capacity for plant community regeneration (Suding et al. 2004). Also, ongoing cattle grazing at our project sites (especially LJ and OR) could generally decrease plant species abundance and diversity (Fleischner 2002).

Plant community responses to secondary tamarisk management strategies

Of the three secondary tamarisk treatments we used all were relatively benign. There was some indication that biological control releases resulted in a more rapidly recovering plant community. Plant community data suggested that targeted applications of both the non-selective herbicide (imazapyr) and the more selective herbicide (triclopyr) caused initially severe impacts. However, results from surveys in 2012 suggested that plant communities in plots that received triclopyr were recovering significantly more rapidly than those treated with imazapyr. Previous studies have generally found that imazapyr has a higher potential for negative off-target impacts than does triclopyr (Douglass 2013, Kaeser and Kirkman 2010). Admittedly our sampling design and low number of replicates may not have been entirely adequate to accurately measure finer scale community changes in response to secondary tamarisk removal methods.

Conclusions

On the whole, plant species richness and diversity was equally high under the remaining tamarisk canopy in untreated areas and mechanically-treated portions of sites. These results concur with a recent study in Grand Canyon National Park that found plant communities were generally unresponsive to targeted manual removal or cut-stump tamarisk treatments (Belote et

al. 2010). In both studies water availability (e.g. annual precipitation and overall hydrologic connectivity) appeared to be a stronger driver of plant community condition than was the presence or absence of tamarisk (Suding et al. 2004, Stromberg et al. 2009). The lack of response from plant communities in our study, and especially native species, in plots where tamarisk biomass was removed provides some indication that tamarisk dominance is not the sole limiting factor in the ecosystems where this study was conducted (MacDougall and Turkington 2005).

To summarize, we found that: 1) aerial imazapyr applications facilitated establishment and rapid population growth by a single ruderal species, *Bassia scoparia*; 2) mechanical methods did not favor recruitment by weedy noxious species; 3) there was a trend towards biological control benefiting passive restoration; and 4) IPT imazapyr and triclopyr applications did not have different effects on re-vegetation patterns. Over a longer time frame some treatment differences, in particular the beneficial effect of biological control, will likely become more apparent.

Population dynamics of secondary invasive plants

We did not find that soil disturbance associated with mechanical tamarisk management contributed to the expansion of secondary invasive species. However, several noxious species became more abundant throughout our sites over the study period. For noxious forbs in particular, neither tamarisk treatments or soil disturbance affected population dynamics. Instead it seemed that climate and simple population expansion explained recruitment patterns (Lockwood et al. 2005). For example, *Acroptilon repens* and *Carduus nutans* became more abundant across sites where they were already minimally present regardless of tamarisk removal method. Across all sampling plots *Acroptilon* cover increased 4X over three years, cover for

Carduus increased 3.25X, and the mean weighted stem density for both species increased five-fold.

Population growth of both species was restricted to individual sites where they were already present (OR: *A. repens*; FL: *C. nutans*) at the beginning of the study as confirmed by seedbank samples. The average weighted density for *Acroptilon* in the seedbank was actually 20% higher field records, suggesting that establishment of this species in the field may have been limited more by resources than other factors. Harms and Hiebert (2006) did not find increased recruitment of exotic species following tamarisk removal, but did not specifically identify noxious species versus more innocuous non-native species. Likewise, our results contrast with Sher et al. (2008), who found that *B. tectorum* cover was reduced in areas where tamarisk was also removed.

Imazapyr soil residue impacts on plant community re-vegetation patterns

Finally, in a previous study (Douglass 2013) we estimated that imazapyr residue levels in the upper portion (top 10 cm) of soils from these same sites would decay to 95% of initial concentrations (roughly 0.05 kg ai ha^{-1}) in roughly 385 days after initial application. We also estimated that these residue levels would not negatively impact establishment of especially sensitive monocots (e.g. *S. airoides* and *Elymus canadensis* L. (Douglass 2013)). Therefore, depending on soil type and other edaphic factors theoretically by about 13 months after application imazapyr residues are no longer limiting factors in the seedbank horizon of these soils.

So in the case of dominance by *Bassia scoparia* in areas aerially treated with imazapyr, we hypothesize that competition mediated by this species limits recruitment of other forbs and

grasses in the understory. *B. scoparia* is an annual forb common in agronomic systems that is known to be resistant to acetolactate synthase (ALS) herbicides such as imazapyr (Chodova and Mikulka 2000). The species has a prodigious seed output (in a few plots we measured over 1,500 stems m^{-2}) and with adequate moisture can grow several meters tall and effectively out-compete less vigorous species. We propose that the intense selection pressure applied by aerial imazapyr applications facilitates the rapid establishment of *B. scoparia*, which is especially well suited to act as a dominant ecosystem engineer (Crooks 2002, MacDougall and Turkington 2005). *B. scoparia* is a regulated species in several U.S. states (USDA 2012), but not in Colorado where it is a very common plant species. The plant is frequently seen by ranchers in drought-prone parts of the state as being beneficial since it serves as a forage species when native grasses are absent (Sherrod 1971). While it is not widely seen as being a detrimental plant species, in the short time period following tamarisk removal *Bassia* functionally appears to suppress the inherent resiliency and capacity of some natural systems for passive recovery of a diverse plant community.

Our study provides further evidence that it is possible to control tamarisk and other dominant woody invasive species using integrated strategies that do not detrimentally affect the extant plant community. Our data does indicate though that integrating mechanical tree biomass removal with secondary targeted treatments is more effective than widespread herbicide applications in maintaining an ecosystem's inherent capacity for passive plant community restoration. Ultimately, it becomes critical to address underlying causes of site degradation (e.g. modified hydrologic regimes or intensive livestock grazing) if sites are to be expected to re-vegetate naturally following tamarisk removal (Briggs et al. 1994, Shafroth et al. 2008).

Ecologically based integrated tamarisk management

Overall our results suggest that there is some promise for the application of EBIPM models to tamarisk-invaded ecosystems, though our data also highlights some potential issues with the five-step framework proposed by Sheley et al. (2010). The core theory underlying EBIPM is that the ecological causes of a plant invasion can be assessed and this information used to intentionally direct successional processes and community interactions along a trajectory towards a desirable outcome (James et al. 2010). While the EBIPM framework is sufficient to assess and predict two- and three-way community interactions, it does not appear to have the capacity to account for more complex indirect interactions between the climate and plant communities or long-term alterations in hydrologic regimes in associated waterways. The EBIPM framework would likely be appropriate for managing and correcting the roughly linear processes that led to *Bassia scoparia* invasions in plots aerially treated with imazapyr. Arguably though it would not have the capacity to adapt to plant community patterns that were primarily controlled by external factors (i.e. decreased precipitation, stochastic dynamics in invasive plant populations) in areas that received integrated tamarisk management strategies.

Alternatively, we propose that merging state and transition models into the EBIPM framework would more adequately address the challenges of managing tamarisk in complex habitats within a region increasingly affected by climate change (Suding et al. 2004, Cortina et al. 2006, Perry et al. 2012). Doing so would allow for greater flexibility in selecting non-traditional reference ecosystems, which appears valuable in our case study given the evidence for functional shifts in plant community guilds from those typical of riparian sites to those more characteristic of upland habitats (Reynolds and Cooper 2011). Also, the recognition of multiple alternative ecosystem states involving secondary invaders whose casual controls may not be

strongly influenced by the targeted invasive plant species - or conversely were affected by unintended consequences of the strategy chosen to manage the target species - might facilitate a more holistic restoration effort (Suding et al. 2004, Pearson and Ortega 2009). The structure of the EBIPM framework could be very valuable in guiding comprehensive management of degraded ecosystems invaded by species such as tamarisk. We argue though that this framework needs to better incorporate alternative state theory in order to have the flexibility and predictive power necessary to assist land managers in managing a woody invasive species in ecosystems simultaneously undergoing several fundamental functional shifts. Research in the future that investigates management of ecosystems impacted by multiple environmental stressors, and in particular the interaction of climate, natural plant community assembly processes and population expansion by secondary plant invaders will be especially valuable in ensuring sustainable management of invasive woody species.

Table 5.1a. Location of tamarisk management project sites in southeastern Colorado and brief description of dominant soils.

Site	City	Elevation (m)	Latitude (°)	Longitude (°)	Soil Series	Soil Type
CC	Canon City	1,666	38.49080	-105.20189	Shingle	Very cobbly sandy loam
FL	Florence	1,564	38.37967	-105.03772	- - -	Sandy loam
LJ	La Junta	1,236	37.99278	-103.55008	Glenberg	Sandy loam
OR	Ordway	1,297	38.18254	-103.74780	Apishapa	Clay loam

Table 5.1b. Soil pH, organic matter (OM, %), cation exchange capacity (CEC, meq 100 g^{-1}), salinity (EC, mmhos cm^{-1}), sodicity (SAR) and texture (% sand, silt, clay) for project sites in the Arkansas River watershed of Colorado. All results from private laboratory analysis; means are the average of eight (n = 8) spatially distinct samples per site. Means sharing superscript letters are not significantly different at $P < 0.05$ according to Tukey's HSD means comparisons tests.

Site	pH	OM	CEC	EC	SAR	Sand	Silt	Clay
CC	7.79^{c}	1.75^{c}	17.20^{c}	2.29^{b}	0.29^{c}	78.7^{a}	18.4^{b}	2.9^{c}
FL	7.88^{bc}	2.14^{b}	22.95^{b}	2.55^{b}	0.80^{c}	54.5^{b}	39.6^{b}	5.9^{c}
LJ	8.00^{a}	2.14^{b}	25.14^{ab}	5.61^{a}	5.64^{a}	50.5^{b}	38.4^{b}	11.1^{b}
OR	7.98^{ab}	2.99^{a}	25.99^{a}	2.36^{b}	2.98^{b}	36.2^{c}	39.9^{a}	23.9^{a}

Table 5.2. Results for fixed parameters ('Year', 'Treatment', 'Year*Treatment') from mixed effects models testing whether tamarisk removal method caused significant differences in subsequent re-vegetation patterns. Study was carried out at four sites in the Arkansas River watershed of southeastern Colorado in 2009-2012.

		Prob > F		
	Model R^2	Year	Treatment	Year*Treatment
Richness (species $plot^{-1}$)	0.928	0.0025	0.0389	<.0001
Diversity ($e^{H'}_{Cover}$)	0.919	0.2089	0.6903	0.0180
Diversity (log $e^{H'}_{Density}$)	0.819	<.0001	0.0067	0.0148
log WIS_{Cover}	0.893	0.0081	0.0042	0.0040
log $WIS_{Density}$	0.728	0.0860	0.7255	0.2514
p Cover as Vegetation	0.871	0.0091	0.4370	0.0408
p Species: Noxious	0.823	0.1947	0.0694	0.4978
p Species: Native	0.876	0.0807	0.1991	0.0870
logit (p Species: Annual)	0.790	0.0037	0.2321	0.0605
logit (p Species: Forb)	0.479	0.0013	0.0344	0.0326

Table 5.3. Twenty-five most widespread plant species re-vegetating sites where tamarisk were removed, as measured by frequency-weighted cover (%) and density (stems m^{-2}). Note that 'NP' indicates that the respective species was not found in seedbank samples.

					Density (Stems m^{-2})	
		Origin[1]	WIS[2]	Cover (%)	Field	Seedbank
Bassia scoparia	Kochia	I	4	8.53	102.45	184.25
Distichlis spicata	Saltgrass	N	2	4.49	28.91	9.32
Tamarix spp.	Tamarisk	I*	2	2.68	7.52	0.03
Sporobolus airoides	Alkali sacaton	N	3	2.29	6.10	50.26
Bromus inermis	Smooth brome	N	5	1.13	6.69	NP
Acroptilon repens	Russian knapweed	I*	5	1.62	3.09	4.28
Bromus tectorum	Downy brome	I*	5	0.97	4.35	2.11
Atriplex micrantha	Twoscale saltbush	I	5	0.69	5.18	2.17
Sporobolus cryptandrus	Sand dropseed	N	4	0.89	3.12	57.48
Chenopodium album	Common lambsquarters	I	3	0.48	4.50	4.56
Glycyrrhiza lepidota	American licorice	N	4	0.54	3.09	NP
Carduus nutans	Musk thistle	I*	5	0.50	1.51	0.83
Grindelia squarrosa	Curlycup gumweed	N	4	0.39	1.71	0.05
Salsola tragus	Russian thistle	I	4	0.30	1.85	0.29
Ambrosia psilostachya	Western ragweed	N	3	0.44	0.98	0.08
Helianthus annuus	Common sunflower	N	4	0.38	1.05	0.42
Pascopyrum smithii	Western wheatgrass	N	4	0.30	1.16	NP
Iva axillaris	Povertyweed	N	3	0.25	1.20	NP
Mirabilis nyctaginea	Heartleaf four o'clock	N	5	0.29	1.14	1.17
Ericameria nauseosa	Rubber rabbitbrush	N	5	0.77	0.25	NP
Sphaeralcea angustifolia	Copper globemallow	N	5	1.46	0.03	NP
Atriplex canescens	Fourwing saltbush	N	4	0.40	0.53	NP
Cirsium arvense	Canada thistle	I*	4	0.32	0.71	0.57
Bromus arvensis	Japanese brome	I	4	0.28	0.73	0.05
Muhlenbergia asperifolia	Scratchgrass	N	2	0.23	0.91	NP

[1.] Origin as determined by the USDA PLANTS Database (USDA 2012): 'N' – native; 'I' – introduced. Introduced species marked with an asterisk (*) are regulated as 'noxious' species by the Colorado Department of Agriculture.
[2.] Wetland indicator status as determined by the National Wetland Plant List (Lichvar and Kartesz 2012): 1 – Obligate wetland species; 2 – Facultative wetland species; 3 – Facultative species; 4 – Facultative upland species; 5 – Obligate upland species.

Table 5.4. Response of selected species to mechanical (MEX, including both 'AXE' and 'EXC') and chemical (IMZ) tamarisk removal methods, as measured by frequency-weighted cover (%) and density (stems m^{-2}). Means sharing the same superscript, lower case letters were not significantly different ($P < 0.05$) when compared across sampling year. Means sharing the same superscript, upper case letters were not significantly different ($P < 0.05$) when compared across treatments.

		Weighted Cover (%)			Weighted Density (Stems m^{-2})		
		2010	2011	2012	2010	2011	2012
Bromus tectorum	MEX	0.76	0.65	0.39	1.96^{B}	0.42	6.47
	IMZ	0.00	0.00	0.00	0.00	0.00	0.00
	UTC	1.13	0.31	1.48	6.04^{A}	2.51	4.40
Distichlis spicata	MEX	3.20^{b}	4.80^{ab}	6.40^{a}	28.86	26.08	44.25
	IMZ	4.70	3.25	3.74	6.49^{ab}	3.26^{b}	14.01^{a}
	UTC	3.22	4.09	4.27	20.06	27.59	34.97
Glycyrrhiza lepidota	MEX	0.40	0.60	0.69	0.86	1.65	1.19^{B}
	IMZ	0.41	0.38	0.63	0.54	0.36	6.48^{B}
	UTC	0.10	0.85	0.60	1.39	1.22	35.68^{A}
Sporobolus airoides	MEX	3.34	1.84	2.29	6.47	5.51	8.59^{A}
	IMZ	1.55	0.69	0.00	3.38	3.12	0.00
	UTC	2.54	2.38	2.72	8.73	4.48	0.34^{B}
Sporobolus cryptandrus	MEX	$0.67^{b,B}$	1.58^{a}	$0.89^{ab,A}$	2.06	2.93	3.95
	IMZ	2.36^{A}	0.00	0.00	0.57	0.00	0.00
	UTC	0.44^{B}	0.59	0.10^{B}	2.89	8.73	1.04

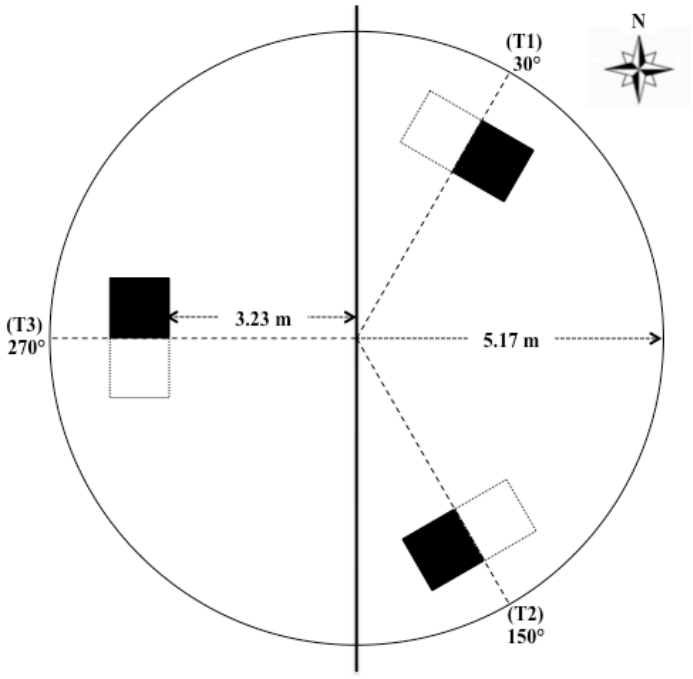

Figure 5.1. The circular, multi-scale vegetation sampling plot (84 m^2) used to sample tamarisk understory plant communities. Solid 1 m^2 subplots were included in all treatment plots, open subplots were only used in mechanically-treated areas to study the impact of tamarisk debris on re-vegetation patterns. Plots adapted from Barnett et al. (2007) to accommodate the lower stature tamarisk canopy.

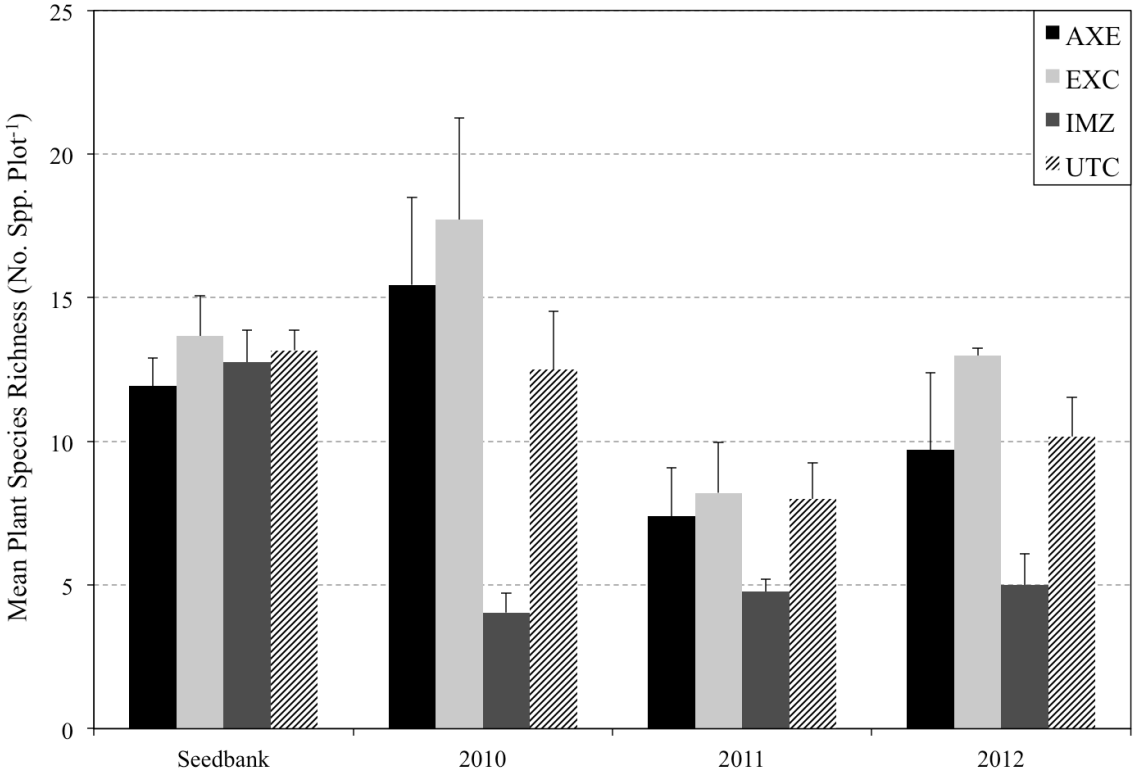

Figure 5.2. Changes in plant species richness (species plot^{-1}) over time due to primary tamarisk tree removal strategies. 'AXE' = mechanical mulching of aboveground tree biomass; 'EXC' = whole tree extraction; 'IMZ' = aerial applications of imazapyr herbicide; 'UTC' = untreated. Vertical bars above treatment means represent one standard error (SE).

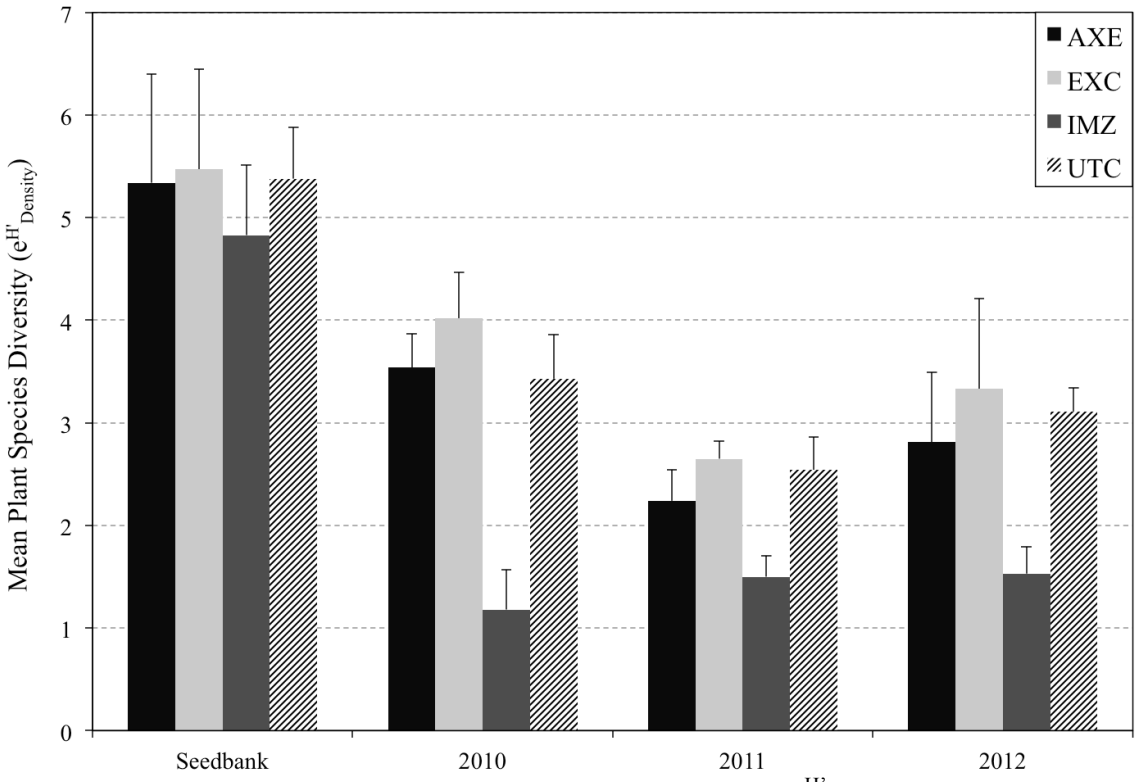

Figure 5.3. Changes in species diversity based on plant densities ($e^{H'}_{Density}$) over time due to primary tamarisk tree removal strategies. 'AXE' = mechanical mulching of aboveground tree biomass; 'EXC' = whole tree extraction; 'IMZ' = aerial applications of imazapyr herbicide; 'UTC' = untreated. Vertical bars above treatment means represent one standard error (SE).

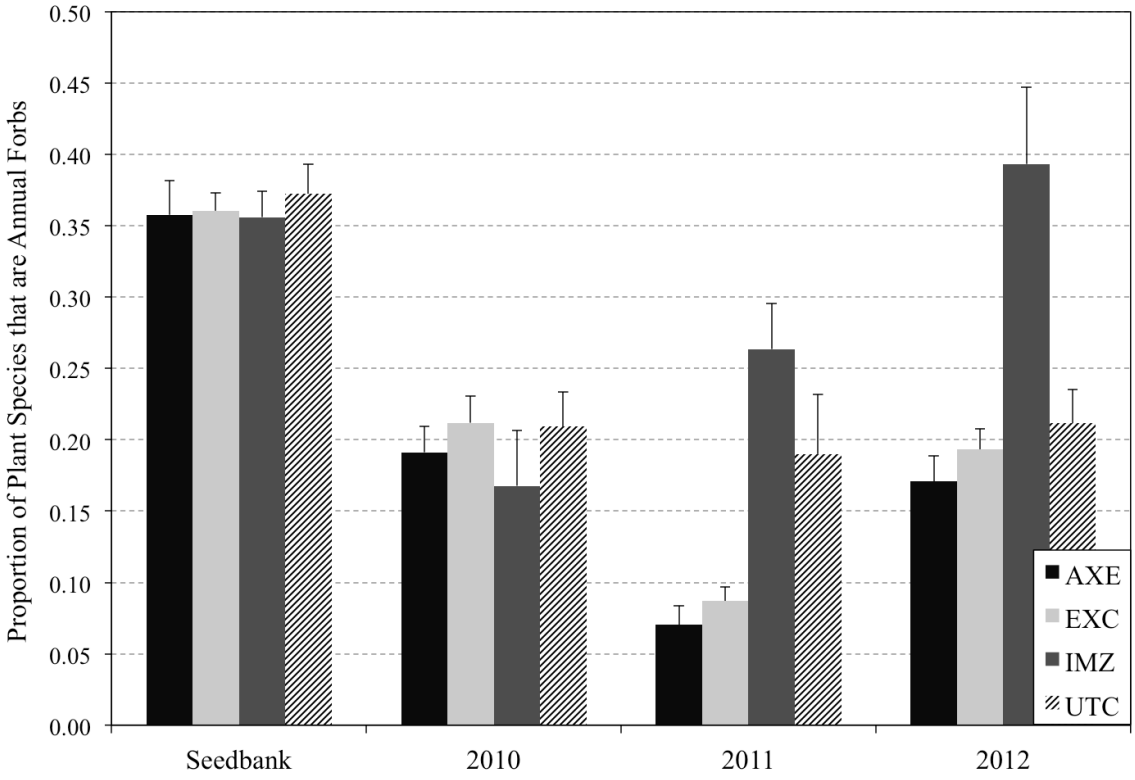

Figure 5.4**.** Changes in the relative abundance (proportion of total species) of annual forbs over time due to primary tamarisk tree removal strategies. 'AXE' = mechanical mulching of aboveground tree biomass; 'EXC' = whole tree extraction; 'IMZ' = aerial applications of imazapyr herbicide; 'UTC' = untreated. Vertical bars above treatment means represent one standard error (SE).

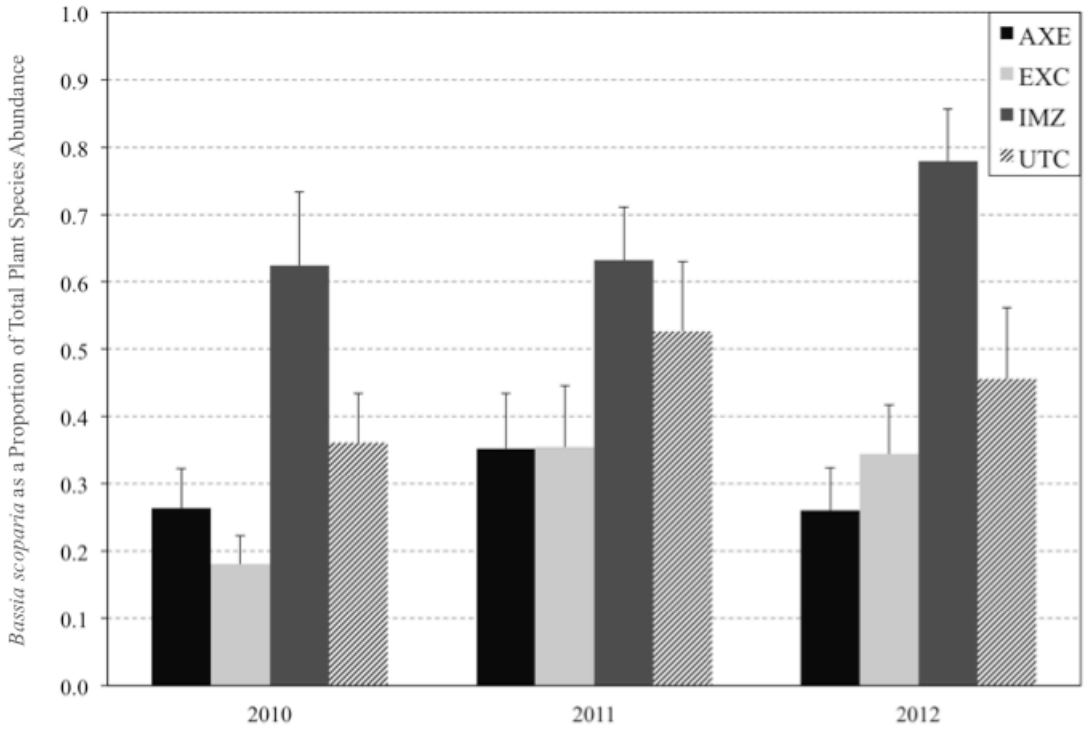

Figure 5.5. Changes in the abundance (averaged proportion of ground cover (%) and density (stems m^{-2})) of *Bassia scoparia* relative to all plant species over time due to primary tamarisk tree removal strategies. 'AXE' = mechanical mulching of aboveground tree biomass; 'EXC' = whole tree extraction; 'IMZ' = aerial applications of imazapyr herbicide; 'UTC' = untreated. Vertical bars above treatment means represent one standard error (SE).

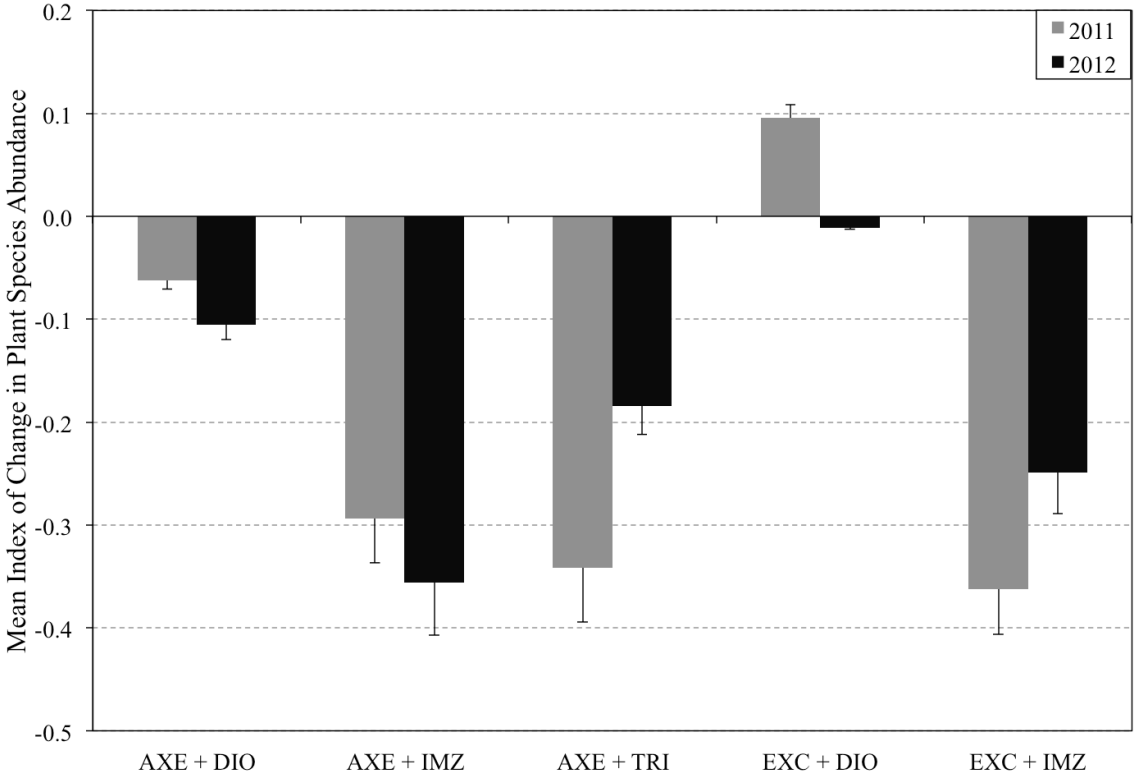

Figure 5.6. Index of year-year (2011 = '10-'11; 2012 = '11-'12) change in plant species abundance in response to secondary tamarisk removal methods, as averaged from plant species richness and diversity data. 'AXE' = mechanical mulching of aboveground tree biomass; 'EXC' = whole tree extraction; 'DIO' = *Diorhabda carinulata* (tamarisk leaf beetle) releases; 'IMZ' = individual plant foliar applications of imazapyr herbicide; 'TRI' = individual plant basal bark applications of triclopyr herbicide. Vertical bars above treatment means represent one standard error (SE).

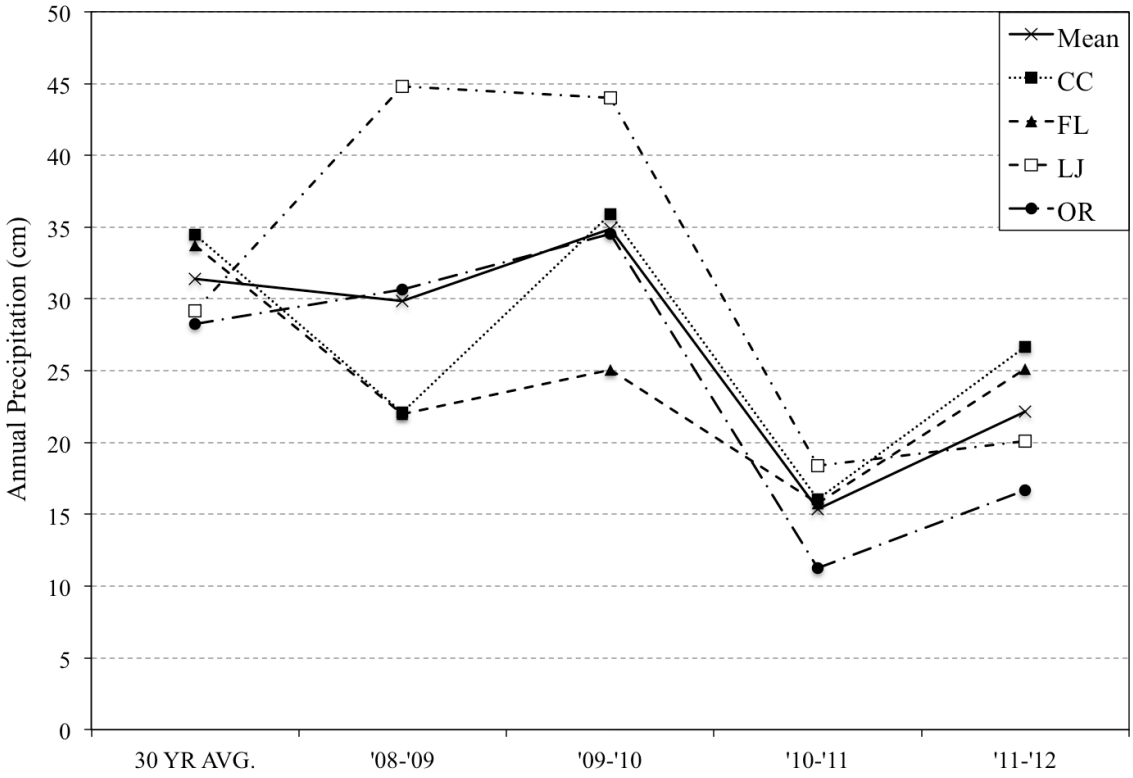

Figure 5.7. Annual precipitation (cm) at tamarisk management sites in Colorado for 2008-2012 and the 30 year average (1979-2007). Note that CC: Canon City, FL: Florence, LJ: La Junta and OR: Ordway. Vertical bars above and below the 'Mean' value represents one standard error (SE).

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APPENDIX

Table A.1. Model parameters for exponential decay models of the form $y = a + b(\exp^{c*\text{Time Point(DAT)}})$ that best fit imazapyr and triclopyr soil degradation (as % initial 0 DAT values) in laboratory simulation studies. Note that a = minimum asymptote, b = scale and c = growth rate.

Imazapyr	AICc	a	b	c
AR	110.06	63.81	35.15	-0.11
CC	113.18	55.66	40.67	-0.04
FL	117.06	50.99	45.05	-0.03
HO	126.81	45.13	51.06	-0.03
LJ	110.84	54.36	46.44	-0.01
OR	120.72	31.29	63.04	-0.02
Triclopyr	AICc	a	b	c
AR	104.87	0.29	99.40	-0.11
CC	93.81	-0.86	99.01	-0.10
FL	111.04	4.91	90.80	-0.07
HO	101.60	-0.99	101.86	-0.13
LJ	114.12	-1.55	106.50	-0.04
OR	109.09	-0.68	102.80	-0.15

Table A.2. Model parameters for logistic models of the form $y = c + (d-c)/(1 + \exp^{-a*(Rate\ (kg\ ai\ ha-1) - b)})$ that best estimated the dose response relationship between plants species dry weight biomass (g) and herbicide rate (kg ai ha^{-1}). Note that a = growth rate, b = inflection point, and c = lower asymptote and d = upper asymptote. Values marked with a '⦸' indicate model parameters that were poorly estimated by the model.

Imazapyr					
	AICc	a	b	c	d
ATCA	294.62	1125.80	0.07	2.75	-0.70
BOCU	260.76	0.51	-28.95	⦸	-16.87
ELCA	169.67	-23.77	0.08	-1.27	1.57
ELEL	73.79	⦸	0.25	-0.12	-1.90
ELTR	198.99	-28.06	0.12	0.12	2.35
GLLE	249.42	27.67	-0.55	⦸	-1.10
HEAN	58.32	32.92	0.40	6.54	-44.68
PASM	205.26	-61.38	0.11	-0.36	1.34
SPAI	95.64	-7.07	-1.93	-1.93	⦸
Triclopyr					
ATCA	326.57	0.74	23.47	3.08	⦸
BOCU	296.99	0.77	-17.42	⦸	-0.58
ELCA	187.07	-5.36	1.73	-0.64	1.32
ELEL	101.01	24.89	1.62	-0.13	-1.03
ELTR	193.35	11.72	2.28	1.98	1.17
GLLE	267.67	1.29	-11.86	⦸	-1.92
HEAN	80.44	-42.40	1.98	6.16	6.74
PASM	223.52	-0.80	3.63	-1.16	1.47
SPAI	119.20	-5.88	0.92	-0.05	2.02

Table A.3. Model parameters for reduced parameter decay models of the form $y = a(\exp^{b*Time\ Point(DAT)})$ used to predict imazapyr soil degradation beyond the 160 day time frame of the study. Note that 'a' = scale and 'b' = growth rate.

Site	AICc	Est. t_{50} (days)	95% C.I.	a	b
AR	122.89	186.02	109.05 - 263.00	83.85	-0.003
CC	122.43	142.70	98.46 - 186.95	87.65	-0.004
FL	124.99	125.56	86.92 - 164.21	88.37	-0.005
HO	129.30	100.09	68.35 - 131.83	88.41	-0.006
LJ	107.89	265.26	202.45 - 328.08	99.70	-0.003
OR	125.79	68.06	50-83 - 85.29	88.80	-0.008

Table A.4. Plant species identified in seedbank samples and field surveys at four sites in the Arkansas River watershed, Colorado 2009-2012. Taxonomic nomenclature and origin designations ("N" = Native; "I" = Introduced) are according to the USDA PLANTS Database[1]. Species marked with an asterisk ("*") are regulated as noxious species by the Colorado Department of Agriculture[2]. Relative frequency was calculated as: (Number of plots/site) X (Number of sites/year) X (Number of years present).

Plant Species	Common Name	Origin	Relative Frequency	
			Field	Seedbank
Achillea millefolium L.	Common yarrow	N		0.06
Achnatherum hymenoides (Roem. & Schult.) Barkworth	Indian ricegrass	N	0.01	
Acroptilon repens (L.) DC	Russian knapweed	I*	0.23	0.24
Agrostis scabra Willd.	Rough bentgrass	N		0.04
Amaranthus spp.	Pigweed	N/I	0.01	
Amaranthus hybridus L.	Slim amaranth	N		0.01
Amaranthus retroflexus L.	Redroot amaranth	N		0.03
Amaranthus spinosus L.	Spiny amaranth	N		0.06
Ambrosia acanthicarpa Hook.	Annual bursage	N	0.08	0.01
Ambrosia psilostachya DC.	Western ragweed	N	0.17	0.04
Ambrosia trifida L.	Giant ragweed	N	0.09	0.14
Anagallis arvensis L.	Scarlet pimpernel	I		0.40
Anthemis cotula L.	Mayweed chamomile	I*		0.03
Apocynum cannabinum L.	Hemp dogbane	N	0.02	
Aristida purpurea Nutt.	Purple threeawn	N	0.02	
Artemisia dracunculus L.	Tarragon	N	0.11	0.17
Artemisia frigida Willd.	Fringed sage	N	0.07	0.03
Artemisia ludoviciana Nutt.	White sagebrush	N	0.06	0.01
Asclepias speciosa Torr.	Showy milkweed	N	0.03	
Asclepias subverticillata (A. Gray) Vail	Horsetail milkweed	N	0.02	0.02
Asparagus officinalis L.	Asparagus	I	0.02	
Astragalus spp.	Milkvetch	N	0.01	
Astragalus bisulcatus (Hook.) A. Gray	Two-grooved milkvetch	N	0.02	
Atriplex canescens (Pursh) Nutt.	Fourwing saltbush	N	0.09	
Atriplex micrantha Ledeb.	Twoscale saltbush	I	0.25	0.16
Atriplex rosea L.	Tumbling saltweed	I	0.03	
Bassia scoparia (L.) A.J. Scott	Kochia	I	0.87	0.89
Bidens frondosa L.	Devil's beggarticks	N		0.03
Brickellia spp.	Brickelbush	N	0.03	
Bromus arvensis L.	Japanese brome	I	0.11	0.01
Bromus inermis Leyss.	Smooth brome	N	0.23	

Bromus tectorum L.	Downy brome	I*	0.24	0.21
Capsella bursa-pastoris (L.) Medik.	Shepherd's purse	I	0.01	
Cardamine oligosperma Nutt.	Little western bittercress	N		0.03
Cardaria draba (L.) Desv.	Whitetop	I*	0.01	
Carduus nutans L.	Musk thistle	I*	0.19	0.13
Carex spp.	Sedge	N	0.03	
Chamaesaracha coronopus (Dunal) A. Gray	Greenleaf five eyes	N	0.02	
Chamaesyce spp.	Sandmat	N	0.06	
Chamaesyce glyptosperma (Engelm.) Small	Ribseed sandmat	N		0.01
Chamaesyce serpyllifolia (Pers.) Small	Thymeleaf sandmat	N		0.13
Chamaesyce prostrata (Aiton) Small	Small prostrate sandmat	N		0.60
Chamaesyce serpens (Kunth) Small	Creeping spurge	N		0.11
Chenopodium album L.	Common lambsquarters	I	0.26	0.44
Chenopodium fremontii S. Watson	Fremont's goosefoot	N		0.06
Chenopodium leptophyllum (Moq.) Nutt. ex S. Watson	Narrowleaf goosefoot	N	0.07	0.11
Chorispora tenella (Pall.) DC.	Crossflower	I	0.01	
Chrysothamnus viscidiflorus (Hook.) Nutt.	Yellow rabbitbrush	N	0.11	
Cirsium arvense (L.) Scop.	Canada thistle	I*	0.09	0.13
Cirsium vulgare (Savi) Ten.	Bull thistle	I*	0.01	
Clematis ligusticifolia Nutt.	Western white clematis	N	0.03	
Conium maculatum L.	Poison hemlock	I*	0.01	
Convolvulus arvensis L.	Field bindweed	I*	0.09	
Conyza canadensis (L.) Cronquist	Mare's tail	N	0.20	0.90
Croton texensis (Klotzch) Mull. Arg.	Texas croton	N	0.08	
Cucurbita foetidissima Knuth	Stinking gourd	N	0.02	
Cyclachaena xanthifolia (Nutt.) Fresen.	Tall marsh elder	N	0.03	
Cylindropuntia imbricata (Haw.) F.M. Knuth	Tree cholla	N	0.12	
Cynodon dactylon (L.) Pers.	Bermudagrass	I		0.01
Cynoglossum officinale L.	Houndstongue	I*	0.04	
Cyperus spp.	Flatsedge	N		0.18
Dactylis glomerata L.	Orchardgrass	I	0.01	
Descurainia incana (Bernh. ex Fisch. & C.A. Mey.) Dorn	Mountain tansymustard	N	0.01	
Descurainia pinnata (Walter) Britton	Western tansymustard	N	0.02	
Descurainia sophia (L.) Webb ex Prantl	Flixweed	I	0.11	0.33

Digitaria californica (Benth.) Henr.	Arizona cottontop	N		0.01
Dipsacus fullonum L.	Common teasel	I*	0.01	
Distichlis spicata (L.) Greene	Saltgrass	N	0.57	0.30
Echinochloa crus-galli (L.) P. Beauv.	Barnyardgrass	I	0.04	0.03
Elaeagnus angustifolia L.	Russian olive	I*	0.12	
Elymus canadensis L.	Canada wildrye	N	0.06	
Elymus elymoides (Raf.) Swezey	Squirreltail	N	0.06	
Elymus repens (L.) Gould	Quackgrass	I*	0.11	
Elymus trachycaulus (Link) Gould ex Shinners	Slender wheatgrass	N	0.09	
Epilobium brachycarpum C. Prel	Tall annual willowherb	N		0.01
Eragrostis spp.	Lovegrass	N/I	0.01	
Ericameria nauseosa (Pall. ex Pursh) G.L. Nesom & Baird	Rubber rabbitbrush	N	0.30	
Erigeron spp.	Fleabane	N	0.05	
Erigeron compositus Pursh	Cutleaf daisy	N		0.02
Erigeron pumilus Nutt.	Shaggy fleabane	N	0.04	
Eriogonum cernuum Nutt.	Nodding buckwheat	N	0.01	
Erodium cicutarium (L.) L'Her. ex Aiton	Redstem filaree	I*	0.02	0.02
Euphorbia spp.	Spurge	N/I	0.01	
Euphorbia dentata Michx.	Toothed spurge	N	0.02	0.03
Euphorbia marginata Pursh	Snow on the mountain	N	0.01	
Euthamia occidentalis Nutt.	Western goldentop	N	0.02	
Festuca arizonica Vasey	Arizona fescue	N		0.02
Frankenia jamesii Torr. ex A. Gray	James' seaheath	N	0.01	
Galinsoga parviflora Cav.	Small flower galinsoga	I		0.01
Galium spp.	Bedstraw	N		0.01
Gaura coccinea Nutt. ex Pursh	Scarlet beeblossom	N	0.03	
Gaura mollis James	Velvetweed	N	0.04	
Glycyrrhiza lepidota Pursh	American licorice	N	0.18	
Grindelia nuda Alph. Wood	Curlytop gumweed	N		0.09
Grindelia squarrosa (Pursh) Dunal	Curlycup gumweed	N	0.19	0.02
Gutierrezia sarothrae (Pursh) Britton & Rusby	Broom snakeweed	N	0.16	
Helianthus annuus L.	Common sunflower	N	0.26	0.08
Helianthus petiolaris Nutt.	Prairie sunflower	N	0.14	0.02
Heterotheca villosa (Pursh) Shinners	Hairy false goldenaster	N	0.02	
Hibiscus trionum L.	Venice mallow	I*		0.04
Hordeum jubatum L.	Foxtail barley	N	0.10	

Ipomopsis laxiflora (J.M. Coult.) V.E. Grant	Iron ipomopsis	N	0.06	0.04
Iva axillaris Pursh	Povertyweed	N	0.03	
Juncus spp.	Rush	N		0.15
Juncus bufonius L.	Toad rush	N		0.15
Juniperus scopulorum Sarg.	Rocky Mountain juniper	N	0.05	
Koeleria macrantha (Ledeb.) Schult.	Prarie junegrass	N		0.11
Lactuca serriola L.	Prickly lettuce	I	0.17	0.03
Lamium purpureum L.	Purple deadnettle	I		0.06
Lappula occidentalis (S. Watson) Greene	Flatspine stickseed	N	0.02	
Leonurus cardiaca L.	Common motherwort	I	0.01	0.01
Lepidium densiflorum Schrad.	Common pepperweed	N	0.09	0.03
Lepidium latifolium L.	Perennial pepperweed	I*	0.16	0.22
Leptochloa dubia (Kunth) Nees	Green sprangletop	N	0.02	
Linaria dalmatica (L.) Mill.	Dalmatian toadflax	I*	0.01	0.01
Linaria vulgaris Mill.	Dalmation toadflax	I*		0.01
Linum rigidum Pursh	Stiffstem flax	N	0.01	
Lolium perenne L.	Perennial ryegrass	I		0.01
Machaeranthera pinnatifida (Hook.) Shinners	Lacy tansyaster	N	0.02	
Marrubium vulgare L.	Horehound	I	0.08	0.05
Medicago lupulina L.	Black medick	I		0.13
Melilotus officinalis (L.) Lam.	Yellow sweetclover	I	0.08	
Mentzelia nuda (Pursh) Torr. & A. Gray	Bractless blazingstar	N	0.01	
Mirabilis linearis (Pursh) Heimerl	Narrowleaf four o'clock	N	0.04	
Mirabilis multiflora (Torr.) A. Gray	Colorado four o'clock	N	0.05	
Mirabilis nyctaginea (Michx.) MacMill.	Heartleaf four o'clock	N	0.22	0.10
Muhlenbergia asperifolia (Nees & Meyen ex Trin.) Parodi	Scratchgrass	N	0.05	
Muhlenbergia racemosa (Michx.) Britton, Sterns & Poggenb.	Marsh muhly	N		0.01
Nasturtium officinale W.T. Aiton	Watercress	I		0.01
Nepeta cataria L.	Catnip	I	0.03	
Oonopsis foliosa (A. Gray) Greene	Leafy false goldenweed	N	0.03	
Opuntia macrorhiza Engelm.	Twistspine pricklypear	N	0.05	
Opuntia phaeacantha Engelm.	Tulip pricklypear	N	0.26	
Opuntia polyacantha Haw.	Plains pricklypear	N	0.18	
Panicum capillare L.	Witchgrass	N	0.05	0.05
Panicum miliaceum L.	Wild proso millet	I*		0.03

Panicum obtusum Kunth	Vine mesquite	N	0.03	
Pascopyrum smithii (Rydb.) A. Love	Western wheatgrass	N	0.17	
Phragmites australis (Cav.) Trin. ex Steud.	Common reed	N	0.02	
Pinus edulis Engelm.	Twoneedle pinyon	N	0.01	
Plantago lanceolata L.	Narrowleaved plantain	I	0.01	0.01
Plantago major L.	Broadleaf plantain	I		0.01
Plantago patagonica Jacq.	Woolly plantain	N	0.01	
Poa compressa L.	Canada bluegrass	I	0.01	
Poa pratensis L.	Kentucky bluegrass	N/I	0.01	0.01
Polygonum spp.	Knotweed	I	0.01	0.02
Polygonum arenastrum Jord. ex Boreau	Oval-leaf knotweed	I	0.05	
Polypogon monspeliensis (L.) Desf.	Annual rabbitsfoot grass	I		0.28
Populus deltoides W. Bartram ex Marshall	Plains cottonwood	N	0.08	0.01
Portulaca oleracea L.	Common purslane	I	0.03	0.30
Portulaca pilosa L.	Kiss me quick	N		0.08
Ranunculus coloradensis (L.D. Benson) L.D. Benson	Colorado buttercup	N		0.06
Ranunculus sceleratus L.	Cursed buttercup	N		0.10
Ratibida tagetes (James) Barnhart	Green prairie coneflower	N	0.02	
Rhus trilobata Nutt.	Skunkbush sumac	N	0.02	
Ribes spp.	Currant	N		0.01
Ribes aureum Pursh	Golden currant	N	0.12	
Rosa woodsii Lindl.	Woods rose	N	0.01	
Rumex crispus L.	Curly dock	I	0.16	0.18
Salix amygdaloides Andersson	Peachleaf willow	N	0.01	
Salix exigua Nutt.	Coyote willow	N	0.03	
Salsola tragus L.	Russian thistle	I	0.24	0.05
Salvia reflexa Hornem.	Lanceleaf sage	N	0.01	
Saponaria officinalis L.	Bouncingbet	I*	0.01	
Schedonnardus paniculatus (Nutt.) Trel.	Tumblegrass	N		0.02
Schkuhria multiflora Hook. & Arn.	Manyflower false threadleaf	N	0.03	
Schoenoplectus maritimus (L.) Lye	Cosmopolitan bulrush	N	0.01	
Schoenoplectus pungens (Vahl) Palla	Common threesquare	N	0.03	
Setaria verticillata (L.) P. Beauv.	Bristly foxtail	I	0.05	
Setaria viridis (L.) P. Beauv.	Green foxtail	I	0.03	0.05
Sisymbrium altissimum L.	Tall tumblemustard	I	0.05	0.09
Sisymbrium irio L.	London rocket	I		0.01

Sisymbrium loeselii L.	Small tumblemustard	I	0.33	0.01
Solanum americanum Mill.	American black nightshade	N	0.01	0.06
Solanum rostratum Dunal	Buffalobur	N	0.01	
Solidago canadensis L.	Canada goldenrod	N	0.03	0.02
Solidago nana Nutt.	Baby goldenrod	N	0.03	
Solidago nemoralis Aiton	Gray goldenrod	N	0.02	
Sonchus spp.	Sowthistle	I	0.01	
Sonchus arvensis L.	Field sowthistle	I*	0.03	0.04
Spergularia salina J. Presl & C. Presl	Salt sandspurry	N		0.01
Sophora nuttalliana B.L. Turner	Silky sophora	N	0.07	
Sphaeralcea angustifolia (Cav.) G. Don	Copper globemallow	N	0.06	
Sporobolus airoides (Torr.) Torr.	Alkali sacaton	N	0.31	0.36
Sporobolus cryptandrus (Torr.) A. Gray	Sand dropseed	N	0.28	0.79
Stanleya pinnata (Pursh) Britton	Desert princesplume	N	0.04	
Symphoricarpos occidentalis Hook.	Western snowberry	N	0.04	
Symphyotrichum ericoides (L.) G.L. Nesom	White heath aster	N	0.06	
Tamarix spp.	Tamarisk	I*	0.41	0.01
Tanacetum parthenium (L.) Sch. Bip.	Common tansy	I		0.06
Taraxacum officinale F.H. Wigg.	Dandelion	N/I	0.09	0.05
Tragopogon dubius Scop.	Western salsify	I	0.08	
Tribulus terrestris L.	Puncturevine	I*		0.01
Ulmus pumila L.	Siberian elm	I	0.03	0.01
Verbascum thapsus L.	Common mullein	I*	0.01	0.28
Verbesina encelioides (Cav.) Benth. & Hook. f. ex A. Gray	Golden crownbeard	N	0.07	0.08
Vulpia octoflora (Walter) Rydb.	Sixweeks fescue	N	0.01	0.02
Xanthium strumarium L.	Common cocklebur	N	0.01	

1. United States Department of Agriculture (USDA). 2012. The PLANTS Database. National Plant Data Team, Greensboro, NC. (http://plants.usda.gov, 17 November 2012).

2. Colorado Department of Agriculture (CDA). 2012. Noxious Weed List. Colorado Department of Agriculture, Conservation Services, Lakewood, CO. (http://www.colorado.gov/cs/Satellite/ag_Conservation/CBON/ 1251618780047).