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Article

Russian Olive (*Elaeagnus angustifolia*) Removal in the Western United States: Multi-Site Findings and Considerations for Future Research

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Abstract: *Elaeagnus angustifolia* (Russian olive) is an introduced tree that has become one of the dominant species in many watersheds in the American West. Although it is a target of restoration efforts, very little is known about vegetation response after removal of this exotic species. To address this gap we surveyed 25 sites in Colorado, Wyoming, and Montana where *E. angustifolia* was removed. We collected information regarding plant cover and richness, climate, soil characteristics, management history, and geography. We analyzed these data using regression tree modeling. Our results indicate that moisture and temperature are key environmental factors relating to restoration success as measured by abundance of native cover; lower temperatures and greater availability of water were generally associated with more native cover. These results have important implications for selection of restoration sites, and for understanding the consequences of removing this species.

Keywords: Russian olive; *Elaeagnus angustifolia*; riparian; restoration; regression tree modeling; invasive species removal

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1. Introduction

Restoration of riparian ecosystems in the Western United States increasingly includes removal of *Elaeagnus angustifolia* (Russian olive), an invasive, nitrogen-fixing, non-native tree, however there is currently little research that investigates the resulting ecosystem impact. Invasive species are those that expand dramatically beyond their native range and thereby effect changes to historic ecosystem structure or function [1]. As a nitrogen-fixer, *E. angustifolia* has the potential to significantly alter soil nitrogen dynamics not only when alive [2], but perhaps especially when it is killed and the unused nitrogen is released into the soil [3]. This pulse in soil nitrogen has the potential to change vegetative communities in sites where this tree is being removed. Because many invasive plants are associated with elevated nitrogen levels, there is particular concern that *E. angustifolia* removal sites may be prone to high exotic cover [4]. *E. angustifolia* is the fourth most dominant riparian tree species in the western United States [5,6]. Introduced in the early 1900's from Eurasia, it is a shade-tolerant, climax species, spread primarily through bird dispersal of seeds [7].

In contrast, native *Populus* spp. are pioneer species requiring disturbance for germination success [8,9]. Due to changes in stream flow and geomorphology, some riparian corridors lose their dynamic disturbance patterns and shift from an early to late successional overstory. They eventually become more like the adjacent upland habitats, which are predominantly agricultural wastelands and/or xeric steppe where exotic species coverage is more abundant than native species coverage [10–12].

The majority of research in the western US in the field of riparian restoration focuses on invasive *Tamarix* spp. restoration, especially in terms of water storage and removal from the watershed [13–15]. *Tamarix* spp. are early successional invaders that are now more dominant than native cottonwood (*Populus* spp.) [5,16]. Two studies report on a quantitative synthesis of vegetation response across sites and over time after *Tamarix* spp. removal [17,18]. Still others report on the restoration of mixed native and invasive stands [12,19,20], but none have synthesized vegetation response across sites after removal of *E. angustifolia* alone.

Ecological statistical models can be used to synthesize the results of riparian restoration projects by evaluating the relative importance of various site characteristics (e.g., soil type, elevation, rainfall) for predicting restoration success (e.g., relative native plant cover and richness). Previous research to identify patterns in riparian revegetation where *Tamarix* spp. was removed has used ordination analysis [17] and regression trees [18]. While no significant pattern was found among abiotic and biotic variables in the first study, the later study was able to identify several, easy to measure environmental variables associated with higher native cover across the 28 *Tamarix* removal sites surveyed, including moisture availability and soil characteristics [18]. In light of the complexity of riparian ecosystems, regression tree analysis appears to be a useful way to deal with many variables across highly variable sites.

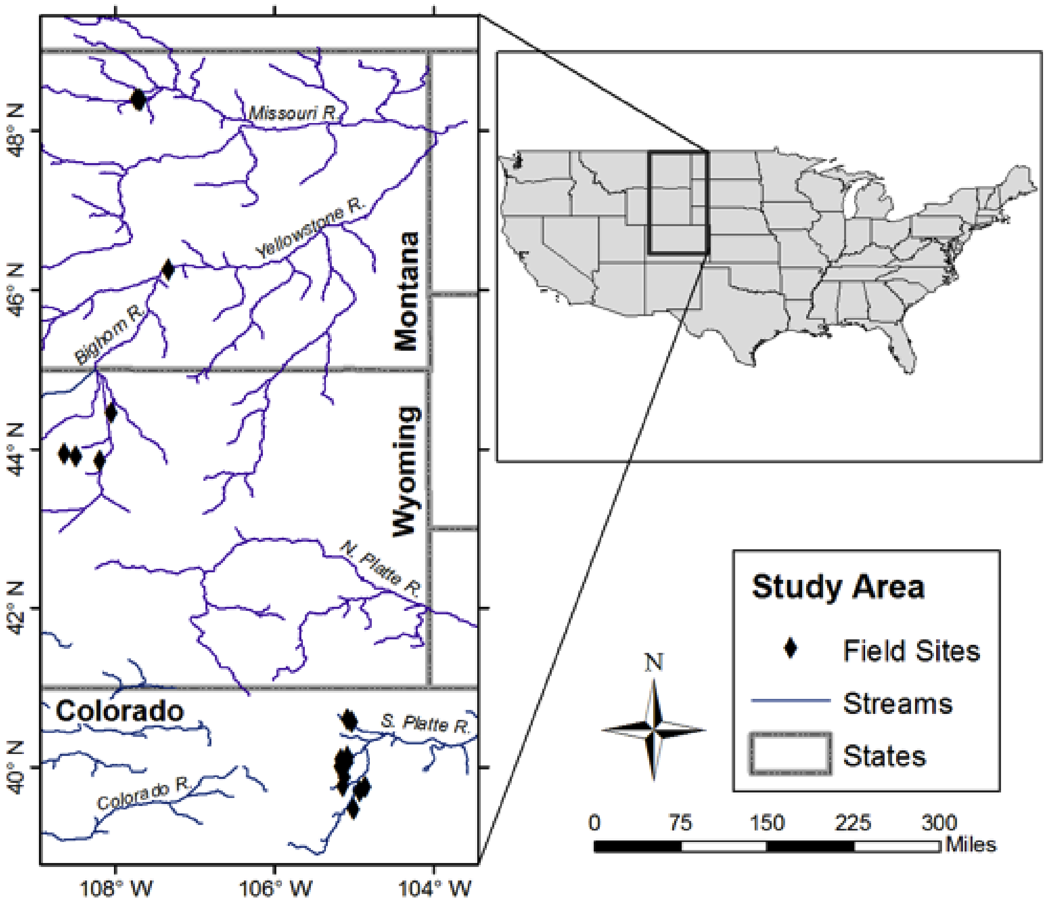
To determine if similar factors were important for determining restoration success following *E. angustifolia* removal, we sampled vegetation and soils at 25 riparian sites in Colorado, Montana, and Wyoming where *E. angustifolia* was invasive and was removed. Given that maximizing native species cover and overall species diversity are common goals in restoration [21], for the purposes of this research, success was evaluated by several different measures including total cover, total exotic (*i.e.*, non-native) cover, total native cover, relative exotic cover, relative native cover, richness, and diversity. It is important to note that "exotic" is not synonymous with "invasive", as not all introduced (*i.e.*, non-native) species cause harm, even if maximizing native cover (at the expense of exotics) is usually considered desirable. Our interest was in determining which, if any, site characteristics such as those associated with the hydrology, soil, or management of a site were associated with success factors in sites where *E. angustifolia* had been removed. Given previous research on *Tamarix* removal sites, we expected that greater moisture availability (as measured by hydrological and weather patterns) would correspond significantly with success of native species over exotics.

2. Methods

2.1. Site Selection

While *E. angustifolia* is present in all 11 western states in the U.S., we contained the study to sites on the east side of the Rocky Mountains between the longitudes of W 104°58.260' to W 108°38.687', with a latitudinal gradient from N 39°28.399' to N 48°22.813' (central Colorado to the northern border of Montana) (Figure 1). Latitude is biologically significant as there is a general trend of increasing *E. angustifolia* invasion from the western United States north to Canada [22,23]. Study sites included the latitudinal extent of *E. angustifolia* dominance in the U.S. While *E. angustifolia* can be found south of the Colorado border, climatic factors favor other invaders like *Tamarix* spp. and thus most of the literature from lower latitudes addresses multi-species stands and monotypic *Tamarix* spp. [12,19,20].

Figure 1. Map of the 25 study sites, as shown by diamonds.



We identified sites by contacting land managers of private and public land (approximately 30) who had engaged in *E. angustifolia* removal. Sites were managed for varying objectives including recreation, grazing, conservation, and wildlife habitat. Historic land use was predominantly agricultural. To be included, sites had to be large enough to fit three plots with the dimensions of 15 m × 40 m, running parallel to a river or lake. We found only one site in all three states where active revegetation had occurred, thus the sites were limited to passive revegetation after tree removal. In all sites *E. angustifolia* trees were removed between 1 and 10 years. We found 25 sites that met our size criteria. The sites are somewhat clustered geographically, a consequence of the patchy nature of *E. angustifolia* removal projects, however sites remain independent. The final site list included 15 in Colorado, 4 in Wyoming, and 6 in Montana (Figure 1).

2.2. Data Collection

Both abiotic and biotic data were collected for each site to determine if there is a relationship between site attributes and restoration success, measured in terms of vegetation cover and richness. Vegetation cover by species and richness were sampled using modified Whittaker plots, which have been shown to describe more accurately measures of diversity than other quadrat or transect designs [24,25]. Three 15 m × 40 m Whittaker plots were surveyed at each site with the following subplots within each: (10) 1 m^2, (2) 6 m^2 and (1) 40 m^2. Plots were oriented with the long side parallel to the water source to maintain proximity to the water source. Where the site was sufficiently large, placement of the Whittaker plots was random, but in many sites the sampling effort covered the entire removal effort.

Plant species richness was recorded in each plot and subplot. Within each subplot percent cover of vegetation types (native *vs.* exotic) was estimated and recorded. Percent cover in 1 m^2 plots was determined using a metal grid frame that subdivided the square meter into 50 squares, to improve cover estimation.

Site characteristics were grouped into geographic, soil, management history, hydrology, and climate categories (Table 1). Slope was determined with a slope meter and elevation with a GarminTM Etrex GPS unit. Soil characteristics for each site were determined from a composite of 10 soil cores 10 cm deep by 2 cm in diameter from a random location in each of the 1 m^2 subplots. Samples were frozen until analyzed by the Colorado State University Soil Testing Laboratory at the end of the field season.

From this soil analysis, we used pH, texture, electroconductivity (a standard measurement of soil salinity), and percent organic matter. These are frequently used edaphic features that are likely to influence revegetation in riparian zones and can have high variability between sites [15]. Although inorganic soil nitrogen (nitrate) was analyzed from these samples, estimates of available N are highly unreliable from field collections due to their rapid temporal fluctuations [26]. It should be noted that in a related study that measured available soil N (nitrate and ammonium) before and after *E. angustifolia* removal, available soil N concentrations were five times higher before *E. angustifolia* removal than after removal. In addition, soil moisture was found to affect available soil N to a greater degree than other abiotic factors, as evidenced by inorganic nitrogen fluctuations over a five-month time period [3]. Since we had neither a time series of measurements nor soil N measurements from before

E. angustifolia removal efforts in these sites, precipitation was favored as more accurate abiotic factor over post-season analysis of nitrate concentrations.

Table 1. Predictor variables that were included in the final models. Those data that were collected but had either sample size, multicollinearity issues, or no independent explanatory power (e.g., elevation, years since last flood event) are not listed here. None of the management data (pre-removal density, time since removal, *etc.*) were included in final models.

Category	Data Collected
Geography	- Slope
Climate [27**Error! Reference source not found.**]	- Mean annual precipitation (in) - Mean annual minimum temperature (F°) - Mean growing season maximum temperature (F°)
Hydrology	- Distance from stream channel (m) - Distance to permanent water (m)
Soil	- electroconductivity (mmhos) - Percent organic matter - Texture

Management history was determined through interviews with land managers. For *E. angustifolia* removal technique, we categorized the responses as "cut" or "cut and treat". "Cut" refers to any number of approaches of mechanically removing above ground biomass including chainsaws and chippers. To prevent regrowth, it is common to follow such mechanical control with a treatment of systemic herbicide ("cut and treat"). Several different chemical herbicide combinations were used, but small sample size prevented division of this category into types of chemicals used or application technique (e.g., to the stump or to foliage). None of the managers interviewed reported using herbicide treatment alone. For pre-removal *E. angustifolia* density, it was necessary to categorize the responses as "less than 50%" or "more than 50%"; too few managers collected quantitative data prior to removal efforts. Although considered for each site where available, none of these variables had explanatory power in our models and thus were not included (Table 1).

Climate and hydrological variables were used in the models. We used the Western Regional Climate Center database of historical and current meteorological measurements for precipitation and temperature records [27]. We considered for inclusion mean annual precipitation, mean annual maximum temperature, mean annual minimum temperature, and mean growing season maximum temperature. Hydrological variables included two-dimensions of water proximity to the study plots. Along a horizontal plane, we measured the distance from permanent water. Along a vertical plane, we measured elevation from the river to the plots (distance to stream channel). The latter was a proxy for water table depth, as we were not able to install wells at each site. Flood regime ("natural" *vs.* "regulated") and time since last flood were determined from manager interviews, but few sites had "nature flow regimes" and/or had flooded since *E. angustifolia* removal, and so this variable could not be included.

2.3. Data Analysis

We employed regression tree analysis to determine the relative importance of each of our site variables for explaining vegetation response at *E. angustifolia* removal sites. Regression tree analysis estimates which site attributes (predictor variables) explained the most variation in the vegetation measurements (response variables) [28]. In ecological studies with necessarily smaller sample sizes, the statistical assumption of normality required for parametric analysis is often difficult to attain. Regression tree analysis is a non-parametric alternative that allows investigators to explore both continuous and categorical data, thus with no assumptions regarding the distribution of the underlying data, only requiring that the data be random and represent independent measures [28]. It is a descriptive tool that compares one response variable, in our case vegetation response, to a suite of predictor variables. We also explored one-to-one relationships between response variables and selected important predictor variables were investigated.

We ran regression tree models using geographic, climatic, hydrologic, and soil variables tested against each vegetation response variable. It is not advisable to incorporate more than nine variables in a regression tree model because of the risk of multicollinearity [29]. Therefore, we first narrowed our list (Table 1) by dropping those that were highly correlated to other variables, and those variables for which there were highly disparate sample sizes between groups.

We used Systat statistical package [29] to build regression tree models, first defining our response variables: richness, total cover, total native cover, total exotic cover, relative native cover, relative exotic cover, and diversity. We then input categories of abiotic site attributes (Table 1), which were the predictor variables. By using least squares, the model computes the proportional reduction of error (PRE) in the response variable due to each predictor variable. The model continues to dichotomously split vegetation data by predictor variables into most-similar groups until there are five or fewer data units in each group. Both the site variables employed in the model and the values at which the data are split are of biological interest. Each regression tree model splits the response data based on divisions in the predictor data that best reduce the error in the response data. The mean value of the response data in each group, the standard deviation of the means, and the sample size are reported. Regression tree results are descriptive rather than purely quantitative; the numbers reported are not as important as the relationship between variables used in each model. We noted which site variables were used first and most in these models and our discussion centers on these variables.

3. Results

Exotic species were present in all 25 study sites with a high degree of variability among sites (Figure 2), the dominance of which was explained at least in part by abiotic conditions. Russian olive accounted for only 5.6% ± 2.6 (SE) of total cover across sites. Moisture availability (e.g., precipitation and distance from water) and/or temperature (e.g., mean annual minimum temperature) were most often selected by the regression tree analysis as the most important descriptors of vegetative response after *E. angustifolia* removal (Table 2). Neither management approach (e.g., removal method) nor geography (e.g., elevation) was important in the models. The strongest models (*i.e.*, with the highest PRE values) were those that predicted total exotic cover, total native cover, and relative native cover.

Figure 2. Mean % cover (with 1 SE) of exotics and natives by site, listed in descending order of latitude for A. Colorado, B. Wyoming, and C. Montana.

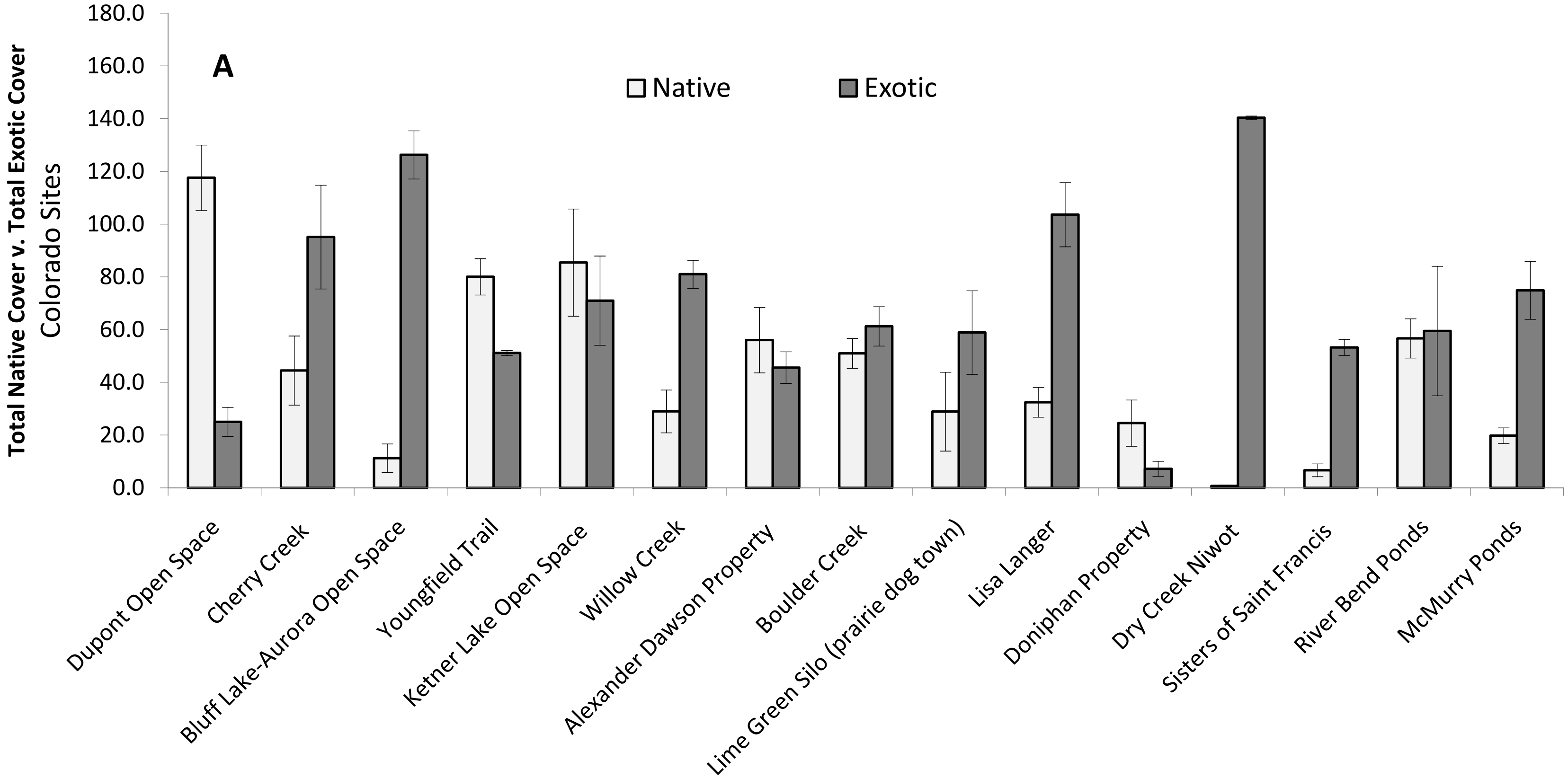


Figure 2. *Cont.*

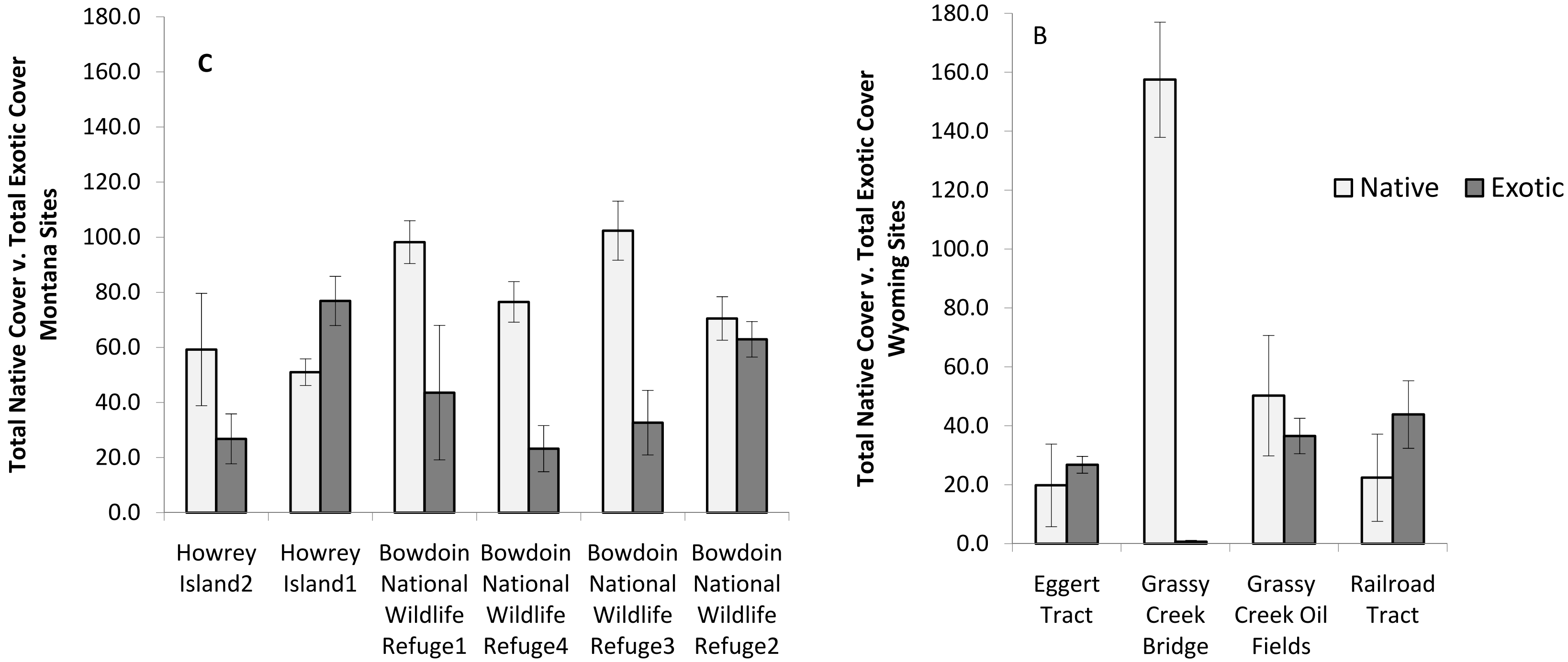


Table 2. Results of regression tree models. Predictor variables are given in order of importance, with first variables having explained the greatest degree of variability.

Response Variable Used	Predictor variables used in the model	PRE
Total Cover	Mean annual precipitation, soil texture, mean annual minimum temperature	0.40
Total Exotic Cover	Mean annual minimum temperature, distance down to stream channel	0.54
Total Native Cover	Distance down to stream channel, distance from permanent water	0.64
Relative Native Cover	Distance down to stream channel, mean annual minimum temperature	0.58
Shannon-Wiener Diversity Index	Soil organic matter, soil texture, mean annual precipitation	0.39
Richness	Electroconductivity, soil organic matter, mean annual precipitation	0.36

On average, when precipitation was more than 35.6 centimeters per year, total cover was 30% higher (Figure 3). Cover significantly increased with precipitation, although the amount of variability explained by this one-to-one relationship was small (Figure 4). Both soil texture and temperature helped explain variability in the regression tree models; in sites with less than 35.6 centimeters per year, total cover typically was higher in loamy soils than in sandy soil types and higher in sites with lower mean annual minimum temperatures. All sites had sandy loam, loamy sand, or loam soils.

Figure 3. Regression tree using total cover as the response variable. The PRE (proportional reduction of error, a measure of fit) of this model is 0.40.

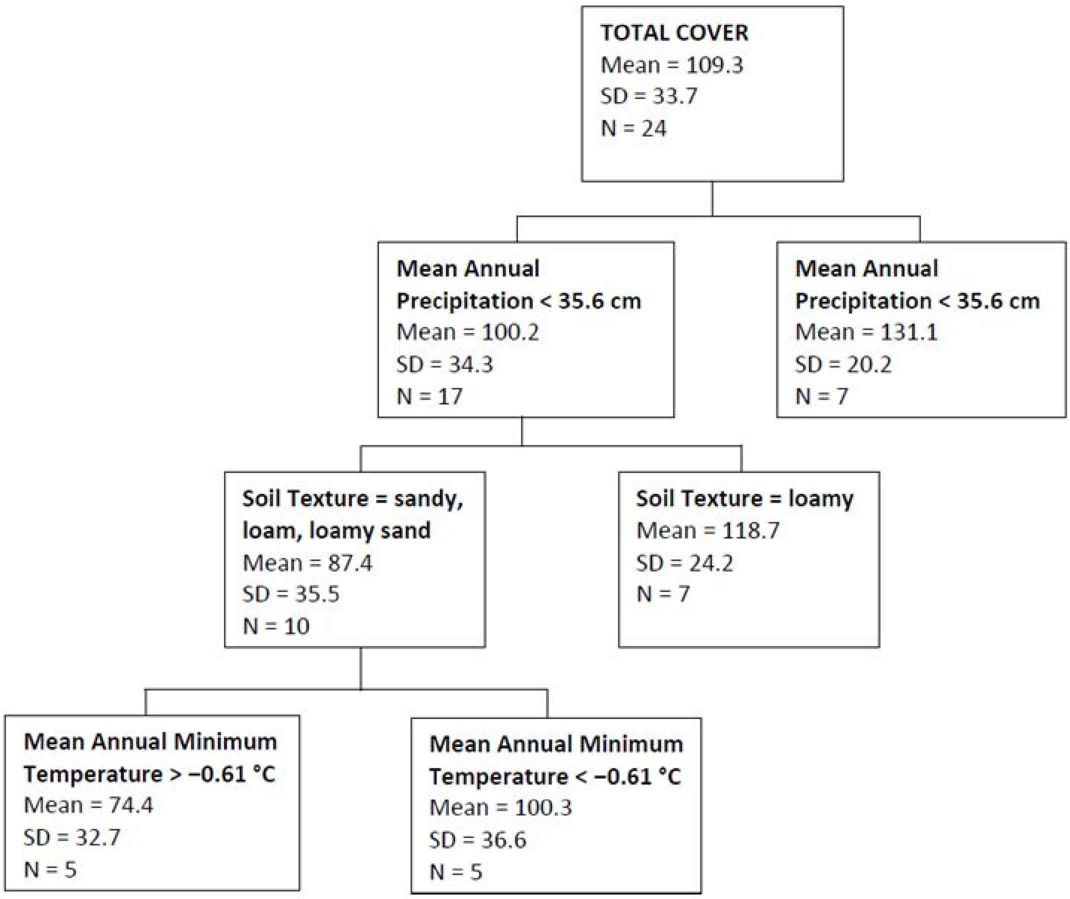
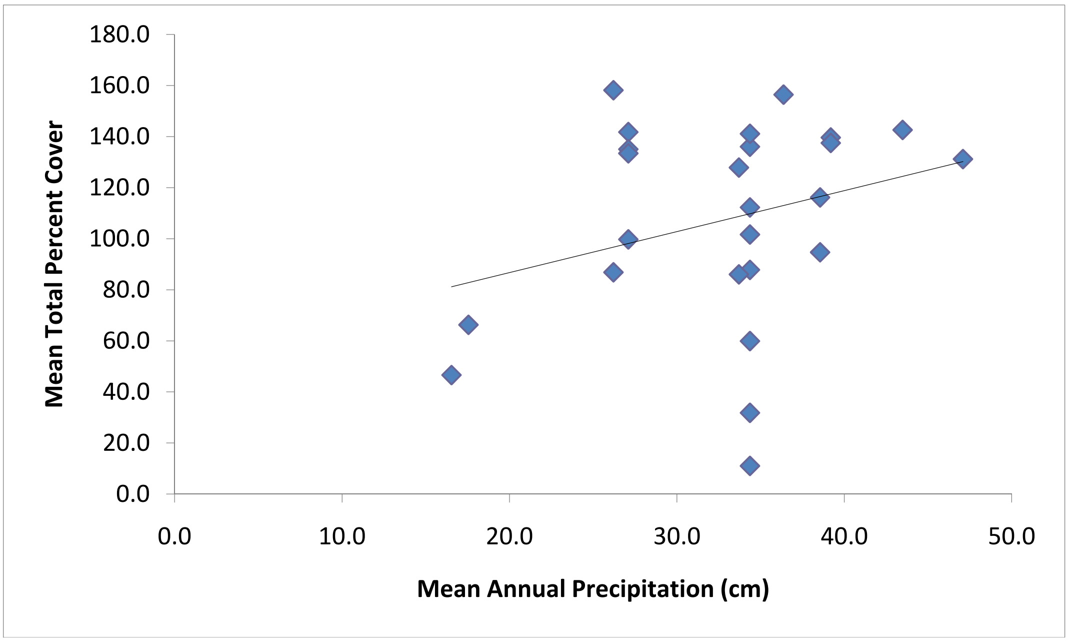


Figure 4. Regression of total % cover against mean annual precipitation ($R^2 = 0.08$, $p = 0.0074$).



On average, when the mean annual minimum temperature was less than 0 °C, the total exotic cover was only 40% (Figure 5) and relative native cover was higher. A regression of these two variables reveals a significantly positive, linear relationship between exotic cover and mean annual minimum temperature (Figure 6). When the mean annual minimum temperature was greater than 0 °C, relative exotic cover was also higher.

Figure 5. Regression tree using total exotic cover as the response variable. The PRE (proportional reduction of error, a measure of fit) of this model is 0.54.

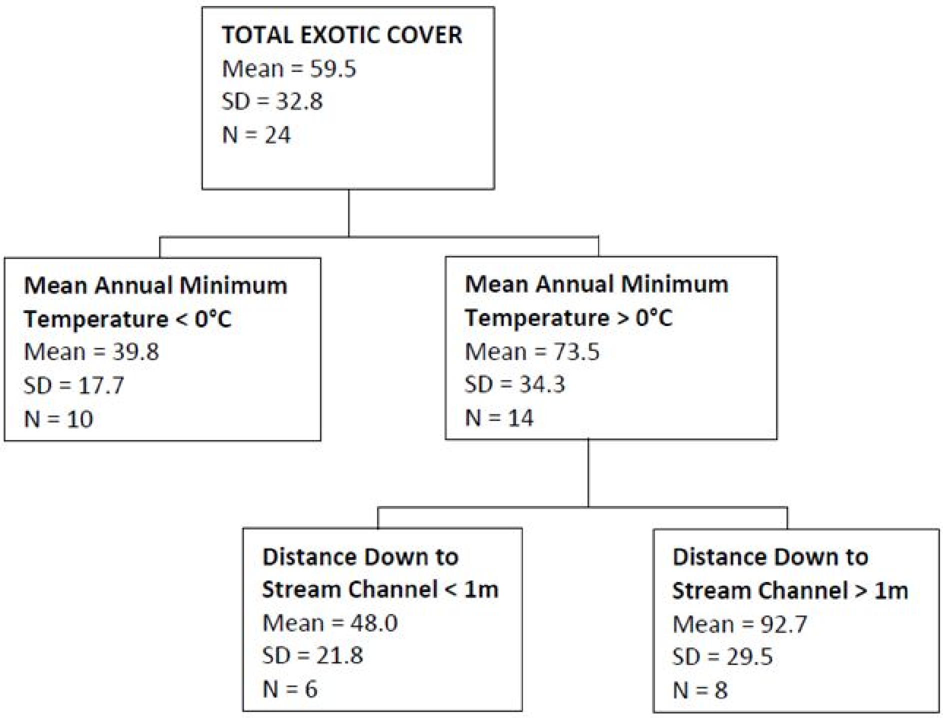
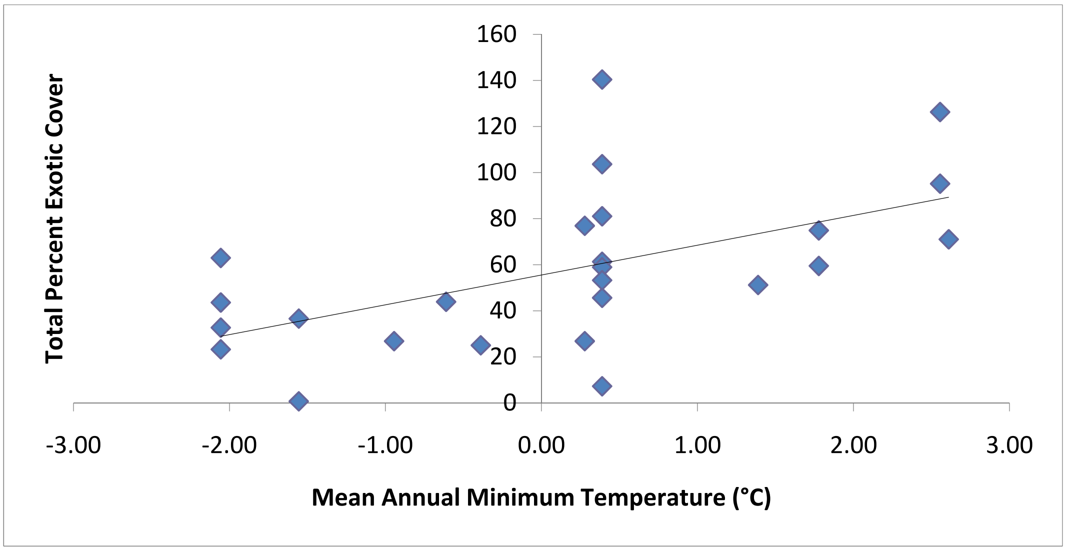
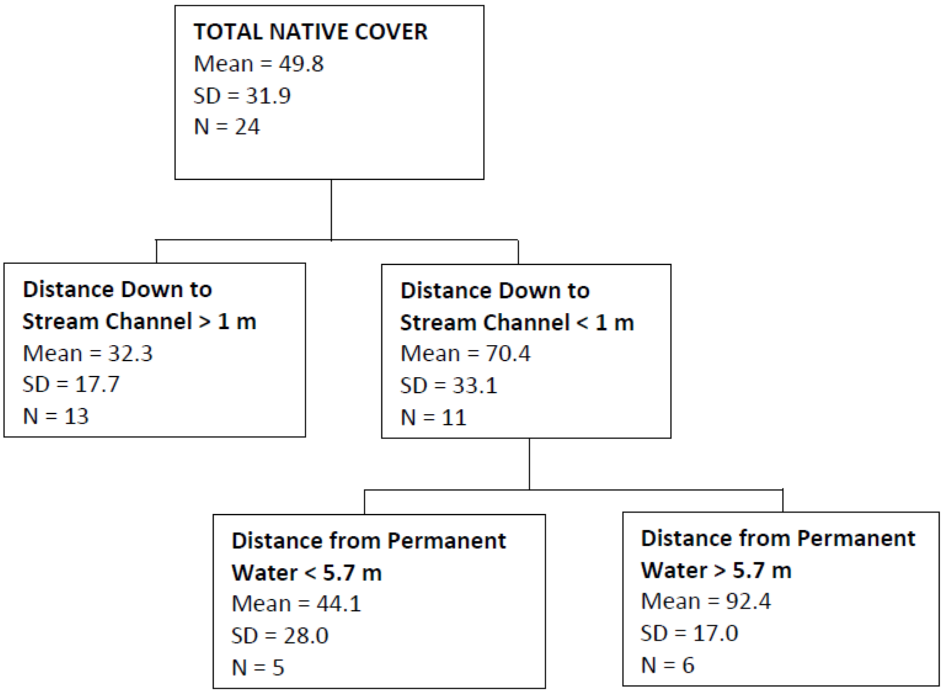


Figure 6. Regression of total exotic cover against mean annual minimum temperature ($R^2 = 0.24$, $p > 0.0001$).



On average, when the distance down to the stream channel from the site was less than 1 meter, native plants cover an average of 70% of a site (Figure 7). On average, when the distance down from the stream channel was greater than 1 meter, total native plant coverage was only 32%. The opposite results were found when total exotic cover was used as the response variable. A distance greater than 1 meter down from the stream channel, roughly doubled total exotic cover (from 48% to 93%). In sites where distance down to the stream channel was less than 1 meter, and when the permanent water was more than 5.6 meters away from the site native coverage was 92%. In other words, in sites without an incised channel native plants thrived, especially beyond the zone of frequent disturbance, *i.e.*, right at the water's edge.

Figure 7. Regression tree using total native cover as the response variable (PRE = 0.64).



When using Shannon-Wiener Index values as the response data, the relative cover of each species was more disparate when organic matter was less than 5% of soil mass. In the group where soil organic matter was less than 5% soil mass, species abundance is more even when sandy soils were present. In the loamy soil sites, species were more even when mean annual precipitation was less than 34.3 centimeters. When electroconductivity was less than 2.35 mmhos, richness was 16 as compared to 13 when electroconductivity was higher than 2.35 mmhos. Among sites with less than 2.35 mmhos of electroconductivity, and when organic matter was less than 5.4%, richness was even higher [18].

4. Discussion

Our survey of *E. angustifolia* removal sites suggests that although there was little evidence of regrowth by the target tree, many sites were still dominated by other exotic (*i.e.*, non-native) species, with evidence that exotic plants are favored over natives in restoration sites with hotter, drier conditions. It is not uncommon for non-native species to dominate when disturbance regimes have been altered and create newly stressful conditions for natives [30–32], as they have been in these riparian systems [33]. Non-native *Tamarix* is the most well-understood example of this, dominating in areas that are more arid [6], generally as a consequence of dam building and channelization of rivers, which dis-favor native trees [9,33]. Previous surveys of riparian restoration sites have similarly found associations between increased exotic cover and low water availability [18,19].

Although not unexpected, our observed relationship between exotic species and aridity is particularly notable given that we only had indirect measures of moisture availability; precipitation, distance down to stream channel, and distance from permanent water were proxies for moisture availability in the absence of wells or direct soil moisture measurements which were logistically unfeasible in this study. With few exceptions, temperature and moisture predictor variables were most important for understanding vegetation response.

Understanding the mechanism of the relationship between moisture and exotic cover requires that we consider native species distribution. In all models, opposite trends are reported for native and exotic cover. In other words, conditions that promoted native species were associated with a decrease in the coverage of exotic species. This is consistent with similar surveys of *Tamarix* removal sites [18,19,34], and may indicate competitive interactions between native and exotics [35]. This theory offers the explanation that although both exotic and native species are expected to respond positively to increased water availability, exotics will dominate only where native species cannot—suggesting that exotics in this system are competitively excluded where conditions are generally favorable [34]. This mechanism has been repeatedly demonstrated for *Tamarix* trees [34–36], however further research is necessary to determine whether it holds true for other exotic species, particularly those that have been considered invasive [37].

Of the few native species we found in these sites, most were either early succession species adapted to disturbance or salt tolerant grasses. The latter are native species usually found in the more xeric, upland habitats adjacent to mesic riparian habitats. Future research should address the efficacy of restoring to mesic, riparian, native vegetation communities *versus* xeric, native, upland vegetation communities [12]. With reduced flooding to promote disturbance-adapted early succession species, the mechanisms to maintain riparian vegetation communities are absent. Therefore adjacent native xeric communities may be more desirable than exotic riparian species in the riparian zone.

It should be noted that many of our sites were unusually dry during a year of drought. Only five of our sites in Colorado actually had precipitation greater than 32.3 centimeters despite the statewide average of 38.4 centimeters that year. Native plants in the riparian corridor are accustomed to a mesic habitat, thus drought conditions may explain the abundance of exotic species in these sites. There were many plants that did not flower for the duration of our field collection period. All of the sites were in areas categorized as experiencing moderate to extreme drought [38]. Thus, the role of moisture for determining vegetative response may have been especially heightened the year of our study [39].

There are several site descriptor variables that would be expected to be important for explaining vegetative response but could not be used in the final models because of missing information; most managers simply did not have baseline data or good records of site histories. Even if including these data would be unlikely to change the patterns observed, there is no doubt that our models would have been more powerful and informative with the inclusion of site history data. It is our intent that our work and other research on the outcomes of restoration projects will encourage monitoring practices in general, including the collection of baseline information.

Removal of any dominant tree will result in a pulse of resources which are likely to affect vegetative response; these include an increase in light availability at the very least, and likely soil resources as well because of competitive release. It is critical to understand what factors are associated with greater or lesser exotic cover among restoration sites. We found soil factors to be more important for understanding species richness and diversity than for cover estimates, and to be less predictive overall, however we did not have good nutrient measurements and therefore can only speculate about their potential role. Nor was our experimental design intended to study the effects of *E. angustifolia* removal (e.g., relative to sites where it was not removed). As these are important questions, we would recommend their pursuit for future work.

5. Conclusions

This quantitative study incorporated site data from Colorado, Wyoming, and Montana where *E. angustifolia* was removed. The models suggest that moisture and temperature site attributes were the best predictors of plant cover and richness, the response variables measured to quantify restoration success. As was found in *Tamarix* removal sites in an earlier study [18], increased availability of water, as measured by precipitation or by distance to stream channel, was associated with higher percentage of natives both in total, and relative to exotic species, which dominated in hotter and drier areas. This has important implications in the context of climate change, which predicts the latter conditions to increase in the future, and also for restoration site selection by land managers. These results suggest that when revegetation is dependent upon passive means, wetter sites are more likely to recover with native species than drier ones.

It is imperative that ecologists and land managers work together to develop a management and restoration plan for *E. angustifolia*, given the expectation that this species is likely to dominate in the future. We hope this research will promote further study regarding *E. angustifolia* so we can develop a sound management plan for this species.

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Conflict of Interest

The authors declare no conflict of interest.

References and Notes

1. Mack, R.N.; Simberloff, D.; Lonsdale, W.M.; Evans, H.; Clout, M.; Bazzaz, F.A. Biotic Invasions: Causes, Epidemiology, Global Consequences, and Control. *Ecol. Appl.* **2000**, *10*, 689–710.
2. DeCant, J.B. Russian olive, *Elaeagnus angustifolia*, alters patterns in soil nitrogen pools along the Rio Grande River, New Mexico, USA. *Wetlands* **2008**, *28*, 896–904.
3. Gaddis, M.L.; Sher, A.A. Environmental impact of restoration of riparian ecosystems: Fitting Russian olive (*Elaeagnus angustifolia*) into the picture. M.S. Thesis, University of Denver, Denver, CO, USA, 2008.
4. Vitousek, P.M.; Walker, L.R. Biological invasion by *Myrica faya* in Hawai'i: Plant demography, nitrogen fixation, ecosystem effects. *Ecol. Monogr.* **1989**, *59*, 247–265.
5. Friedman, J.M.; Auble, G.T.; Shafroth, P.B.; Scott, M.L.; Merigliano, M.F.; Freehling, M.D.; Griffin, E.R. Dominance of non-native riparian trees in western USA. *Biol. Invasions* **2005**, *7*, 747–751.
6. Nagler, P.L.; Glenn, E.P.; Jarnevich, C.S.; Shafroth, P.B. Distribution and Abundance of Saltcedar and Russian olive in the Western United States. *Crit. Rev. Plant Sci.* **2011**, *30*, 508–523.
7. Katz, G.L.; Shafroth, P.B. Biology, ecology and management of *Elaeagnus angustifolia* L. (Russian olive) in western North America. *Wetlands* **2003**, *23*, 763–777.
8. Shafroth, P.B.; Auble, G.T.; Scott, M.L. Germination and establishment of the native plains cottonwood (*Populus deltoides Marshall* subsp. *Monilifera*) and the exotic Russian olive (*Elaeagnus angustifolia L.*). *Conserv. Biol.* **1995**, *9*, 1169–1175.
9. Friedman, J.M.; Scott, M.L.; Auble, G.T. Water management and cottonwood dynamics along prairie streams. *Ecology* **1997**, *125*, 49–71.
10. Poff, N.L.; Allan, J.D.; Bain, M.B.; Karr, J.R.; Prestegaard, K.L.; Richter, B.D.; Stromberg, J.C. The natural flow regime: A paradigm for river conservation and restoration. *BioScience* **1997**, *47*, 769–784.
11. Stromberg, J.C. Restoration of riparian vegetation in the south-western United States: Importance of flow regimes and fluvial dynamism. *J. Arid Environ.* **2001**, *49*, 17–34.
12. Reynolds, L.V.; Cooper, D.J. Ecosystem response to removal of exotic riparian shrubs and a transition to upland vegetation. *Plant Ecol.* **2011**, *212*, 1243–1261.

13. Pollen-Bankhead, N.; Simon, A.; Jaeger, K.; Wohl, E. Destabilization of streambanks by removal of invasive species in Canyon de Chelly National Monument, Arizona. *Geomorphology* **2009**, *103*, 363–374.
14. Scott, R.L.; Cable, W.L.; Huxman, T.E.; Nagler, P.L. Multiyear riparian evapotranspiration and groundwater use for a semiarid watershed. *J. Arid Environ.* **2008**, *72*, 1232–1246.
15. Jaeger, K.L.; Wohl, E. Channel response in a semiarid stream to removal of tamarisk and Russian olive. *Water Resour. Res.* **2011**, *47*, W02536.
16. Glenn, E.; Nagler, P. Comparative ecophysiology of saltcedar and native trees in western U.S. riparian zones. *J. Arid Environ.* **2005**, *61*, 419–446.
17. Harms, R.S.; Heibert, R.D. Vegetation response following invasive tamarisk (*Tamarix* spp.): Removal and implications for riparian restoration. *Restor. Ecol.* **2006**, *14*, 461–472.
18. Bay, R.F.; Sher, A.A. Success of active revegetation after *Tamarix* removal in riparian ecosystems of the southwestern United States: A quantitative assessment of past restoration projects. *Restor. Ecol.* **2008**, *16*, 113–128.
19. Huddle, J.A.; Awada, T.; Martin, D.L.; Zhou, X.H. Understory response following the removal of invasive woody species from a cottonwood riparian forest depends on overstory type. In Proceedings of the Meeting of the Ecological Society of America, Albuquerque, NM, USA, 2 August, 2009.
20. Hummel, O.; Caplan, T. Evaluating restoration success and applying adaptive management in the Middle Rio Grande Bosque. In Proceedings of the Meeting of the Ecological Society of America, Albuquerque, NM, USA, 2 August 2009.
21. Sher, A.A.; Lair, K.; DePrenger-Levin, M.; Dohrenwend, K. *Best Management Practices for Revegetation in the Upper Colorado River Basin*; Denver Botanic Gardens: Denver, CO, USA, 2010; p. 55.
22. Shafroth, P.B.; Brown, C.A.; Merritt, D.M. Saltcedar and Russian olive control demonstration act science assessment. In *Scientific Investigations Report 2009–5247*; US Geological Survey: Reston, VA, USA, 2010; p. 143.
23. Olson, T.E.; Knopf, F.L. Naturalization of Russian-olive in the western United States. West. *J. Appl. Forestry* **1986** *1*, 65–69.
24. Shmida, A. Whittaker's plant diversity sampling method. *Isr. J. Bot.* **1984**, *33*, 41–46.
25. Stohlgren, T.J.; Falkner, M.B.; Schell, L.D. A Modified-Whittaker Nested Vegetation Sampling Method. *Vegetation* **1995**, *117*, 113–121.
26. Bradshaw, A.D.; Chadwick, M.J. *The Restoration of Land: The Ecology and Reclamation of Derelict and Degraded Land*; Anderson, D.J., Greig-Smith, P., Pitelka, F.A., Eds.; University of California Press: Berkeley and Los Angeles, CA, USA, 1980; p. 116.
27. Western Regional Climate Center: Historical Climate Information. Available online: http://www.wrcc.dri.edu/summary/Climsmco.html (accessed on 29 August 2007).
28. Quinn, G.P.; Keough, M.J. *Experimental Design and Data Analysis for Biologists*; Cambridge University Press: Cambridge, UK, 2002; p. 206.
29. Systat Software Inc. *SYSTAT 11 Statistical Software*. Richmond, CA, USA, 2004.
30. Sher, A.A.; Hyatt, L.H. The Disturbance-Invasion Matrix: A new framework for predicting plant invasions. *Biol. Invasions* **1999**, *1*, 109–114.

31. Hobbs, R.J.; Huenneke, L.F. Disturbance, diversity, and invasion: implications for conservation. *Conserv. Biol.* **1992**, *6*, 324–337.
32. Forcella, F.; Harvey, S.J. Relative abundance in an alien weed flora. *Oecologia* **1983**, *59*, 292–295.
33. Shafroth, P.B.; Stromberg, J.C.; Patten, D.T. Riparian vegetation response to altered disturbance and stress regimes. *Ecol. Appl.* **2002**, *12*, 107–123.
34. Sher, A.A.; Marshall, D.L. Competition between native and exotic floodplain tree species across water regimes and soil textures. *Am. J. Bot.* **2003**, *90*, 413–422.
35. Sher, A.A.; Marshall, D.L.; Taylor, J. Spatial partitioning within southwestern floodplains: Patterns of establishment of native *Populus* and *Salix* in the presence of invasive, non-native *Tamarix*. *Ecol. Appl.* **2002**, *12*, 760–772.
36. Sher, A.A.; Marshall, D.L.; Gilbert, S.A. Competition between native *Populus deltoides* and invasive *Tamarix ramosissima* and the implications of reestablishing flooding disturbance. *Conserv. Biol.* **2000**, *14*, 1744–1754.
37. Stromberg, J.C.; Lite, S.J.; Marler, R.; Paradzick, C.; Shafroth, P.B.; Shorrock, D.; White, J.M.; White, M.S. Altered stream-flow regimes and invasive plant species: The Tamarix case. *Global Ecol. Biogeogr.* **2007**, *16*, 381–393.
38. National Climate Data Center: Palmer Hydrological Drought Index. Available online: http://www.ncdc.noaa.gov/temp-and-precip/drought/historical-palmers.php (accessed on 2 February 2007).
39. Didham, R.K.; Tylianakis, J.M.; Hutchison, M.A.; Ewers, R.M.; Gemmell, N.J. Are invasive species the drivers of ecological change? *Trends Ecol. Evol.* **2005**, *20*, 470–474.