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Impact of Non-Native Plant Removal on Lizards in Riparian Habitats in the Southwestern United States

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Abstract

Many natural processes in the riparian cottonwood (*Populus deltoides*) forest of the Middle Rio Grande (MRG) in the southwestern United States have been disrupted or altered, allowing non-native plants such as saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) to establish. We investigated reptilian responses to restoration efforts by sampling communities of lizards at 12 study sites invaded by non-native plants along the MRG in New Mexico for 7 years (2000–2006). Sites within three regions were randomly assigned to one of the three treatments to remove non-native plants and woody debris, or as untreated controls. We used pitfall and funnel traps to capture, mark, and release lizards from June to September. Principal components analysis of 15 vegetation variables identified five factors that best explained variation among sites before and after removal of non-native plants. Relative abundances for four of six common species of lizards were associated with vegetation characteristics that significantly changed after plant removal. Species were either positively associated with the more open, park-like understory found in treated sites or negatively associated with debris heaps and thickets of non-native plants found in untreated sites. Eastern fence lizards (*Sceloporus consobrinus*) and New Mexico whiptails (*Aspidoscelis neomexicana*) increased in relative abundance after non-native plants were removed. Overall, removal of non-native plants seems beneficial, or at least is non-damaging, to lizard communities of the MRG forest. Providing information on habitat associations of lizard communities will help land managers balance management objectives with other considerations, such as providing important wildlife habitat.

Key words: invasive/non-native species, lizard, reptile, riparian forest, saltcedar, *Tamarix*.

Introduction

Floodplains and riparian areas are some of the most diverse terrestrial habitats on Earth (Naiman et al. 1993; Kondolf et al. 1996). Over the past 200 years, nearly every major river system has been regulated (Benke 1990; Dynesius & Nilsson 1994; Postel 2002; Tockner & Stanford 2002). Regulation of flow regimes of rivers often disconnects the river from its floodplain (Stanford & Ward 1993; Bayley 1995; Brunke & Gonser 1997; Boulton & Hancock 2006), which results in fewer flood events (Junk et al. 1989) and alters adjacent riparian habitats (Gore & Shields 1995; Nilsson & Berggren 2000). Physical disconnection of rivers from floodplains also can promote establishment of non-native plants (Nilsson & Berggren 2000); therefore, controlling non-native plants is part of repairing native habitats.

In the southwestern United States, saltcedar (*Tamarix* spp.) has established in dense, nearly monotypic, stands along many streams often to the detriment of native plants and habitat quality (DeLoach et al. 1999). Removing saltcedar and other non-native plants can be seen as a step toward the recovery of these critical ecosystems. Therefore, it is vital to understand how saltcedar control methods will impact biotic communities (Shafroth et al. 2005). The Middle Rio Grande (MRG) in central New Mexico is an example of a regulated river altered by human activity, resulting in fewer spring floods and the establishment of non-native invasive plants such as saltcedar and Russian olive (*Elaeagnus angustifolia*; Howe & Knopf 1991). Prior to regulation, the MRG had a variety of riparian plants, a diversity of height classes, and a mix of open and dense vegetation (Farley et al. 1994). Today, regulation has led to a reduction in cottonwood regeneration and created dense stands of non-native vegetation (Howe & Knopf 1991; Farley et al. 1994). Non-native plants and woody debris associated with them have increased the risk of fire. Wildfires are a serious threat to native cottonwood (*Populus deltoides*) forests because these forests are not fire-adapted (Busch & Smith 1995). Many studies have focused on the use of saltcedar habitats by arthropods, birds, and small mammals (Ellis 1995; Ellis et al. 1997, 2000; Shafroth et al. 2005; Finch et al. 2006), as well as amphibians and reptiles (Jakle & Gatz 1985; Jones 1988; Griffin et al. 1989; Lovich & DeGouvenain 1998); however, few studies have addressed the impact of removing

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non-native plants on herpetofaunal abundance. Although several restoration studies have focused on reptilian communities (sensu, prescribed burning; Mushinsky 1992; Litt et al. 2001; Cunningham et al. 2002; Pilliod et al. 2006; Wilgers & Horne 2006, planting native riparian vegetation; Queheillalt & Morrison 2006), our research represents one of the first experimental studies to document the impact of saltcedar control methods on communities of lizards.

Land managers who are weighing options for improving ecological conditions of degraded bottomland habitat require information on how restoration activities will affect biotic communities. Within the semiarid southwestern United States, removing non-native plants is often seen as a step toward improving bottomland habitat quality. Therefore, understanding the effects of saltcedar control on biotic communities is vital (Shafroth et al. 2005). Our research will help land managers make sound decisions that balance management objectives (e.g., reducing fire risk) with other considerations (e.g., providing important wildlife habitat). The purpose of our study was to determine the impact of removing non-native plants and associated woody debris on microhabitat and abundance of lizard communities along the MRG by comparing sites where non-native plants dominate the understory to sites with non-native plants removed (Fig. 1). We investigated (1) how vegetation characteristics of the riparian forest differed before and after non-native plants were removed; (2) how relative abundance and presence/absence of species of lizards related to changes in vegetation characteristics; and (3) how abundance of species of lizards differed before and after non-native plants were removed.

Figure 1. An example of two sites showing non-native plant removal (top panel) and a site after plant removal (bottom panel) in the cottonwood (*Populus deltoides*) forest along the MRG, New Mexico, U.S.A. Crews used chainsaws to cut non-native plants like saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*). After treatment sites had a more open understory with fewer non-native plants, fewer downed branches, and less litter on the ground compared to before treatment (Top photo from the U.S.D.A. FS Rocky Mountain Research Station; bottom photo by H.L.B.).

Methods

Study Site

This study was conducted in riparian forests along the MRG in central New Mexico. The MRG is in the Basin-Range physiographic province, which is characterized by mountain ranges with deep river troughs and semiarid to arid climate (Tuan 1962). Historically, flooding occurred along the MRG in spring due to snowmelt and in late summer due to monsoonal storms. However, flood control measures, including dams and levees, have reduced the frequency and magnitude of flooding (Molles et al. 1998). Today, the riparian forest contains a mix of native Rio Grande cottonwood (*Populus deltoides* ssp. *wislizenii*), non-native saltcedar (*Tamarix chinensis* and *T. ramosissima*), and non-native Russian olive (*Elaeagnus angustifolia*).

We monitored relative abundances of lizards at 12 sites (approximately 20 ha each) for 7 years in three geographically distinct regions (i.e., north, middle, and south) along the MRG from Albuquerque (lat 35°01′04 N, long 106°40′30 W) to approximately 140 km south at the Bosque del Apache National Wildlife Refuge (lat 33°47′59 N, long 106°52′59 W). Each region had four sites: three sites were treated and one was an untreated control. Each treated site was randomly assigned to one of the three treatments to remove non-native plants and dead woody debris. Hand crews used chainsaws to remove non-native plants. Herbicide (Garlon) was applied to stumps and reapplied 1 and 2 years later to stump sprouts. In addition to plant removal, one treatment consisted of burning slash piles. Another treatment consisted of planting 247 native shrubs per hectare (100 plants per acre), including existing natives (NRCS 2005). All sites had a native cottonwood overstory with non-native plants dominant in the understory. Monitoring was performed from June to September from 2000 to 2006. Treatments were applied during fall or winter to avoid disturbing animals being monitored. Non-native plant removal began in 2003 and was completed in 2005. Not all treated sites were treated simultaneously due to difficulties in coordinating efforts of several agencies.

As a result, the duration of pre-treatment conditions varied from 3 to 5 years, and post-treatment conditions varied from 2 to 4 years. The field season in the first year was 6 weeks shorter than other seasons; therefore, lizard captures are fewer in 2000 compared to other years and are not included in analyses of relative abundances.

Field Sampling

We captured lizards during summers 2000–2006 using pitfall and funnel traps set along drift fences made from solid metal flashing and erosion control fence. Three arrays of drift fences were installed at least 320 m apart at each site. Each array consisted of three fences with two pitfalls and two funnels per fence (Fig. 2). Each fence was 6 m long, started 7.5 m from a central point, and positioned at an angle of 120° from other fences. The center of each array was located at a random distance greater than 25 m from the edge of the site. Traps were open continuously from June to mid-September and checked 3 days per week. Lizards were identified to species using characteristics provided by Degenhardt et al. (1996). Lizards were measured, given a unique toe clip, and then released. We defined relative abundance as the minimum number of individuals captured. Sampling techniques were approved by University of New Mexico Animal Care and Use Committee protocol (20415).

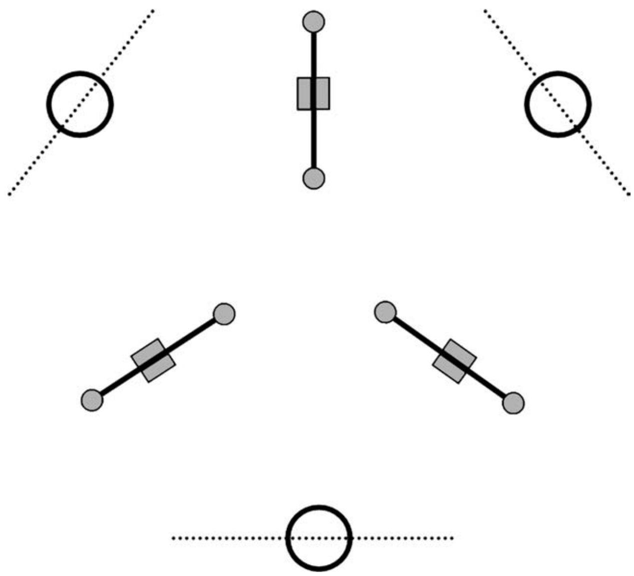

Figure 2. Orientation of a trapping array, vegetation plots, and transects (not drawn to scale). Each trapping array included three, 6-m-long drift fences (black lines), six pitfall traps (gray circles), and six funnel traps (gray rectangles). Fences began 7.5 m from a central point and were oriented at 0, 120, and 240°. The center of each 4-m-radius plot (open black circle) was 25 m from the center point of each array. Vegetation measurements at plots and transects alternated among 60, 180, and 300° from the center of each array (the three positions depicted). Each 50-m transect (dashed black line) ran through the center of each circular plot and was oriented perpendicular to an imaginary line extending from the array's center to the plot's center.

We measured 15 vegetation variables at transects and plots during the summer months once before and once after treatment (pre- and post-treatment measurements were 34 months apart on average). Ground cover was measured at every meter, depth of litter was measured at every fifth meter where litter occurred, and number and size of dead branches were measured along 50-m transects. Tree diameters and numbers of shrubs and trees were measured in 4-m-radius plots. Canopy cover was estimated from two readings (facing north and south) with a spherical densiometer from the center of plots. Basal area of cottonwoods was measured with a wedge prism (basal area factor = 20) from the center of plots. Transect midpoints and plot centers coincided and were located 25 m from the center of arrays and occurred at 60, 180, or 300° relative to fences (Fig. 2).

Statistical Analyses

Some years and sites were eliminated from summary statistics and analyses when wildfires or flash floods that were not part of the experimental design impacted the study (Appendix 1). Years 2001–2006 were used for all sites in North region, except for one treated site that burned in 2006. At the beginning of the project, a fire burned the control site in Middle region; therefore, year 2001 was eliminated and a new control was established in 2002. Years 2001–2006 were used for all sites in South region, except for one treated site that was impacted by flash flooding in 2006.

We summarized variation in vegetation before and after non-native plants were removed with a principal components analysis (PCA). The number of relevant factors was determined based on scree plots and factors with eigenvalues greater than 1. Factors were interpreted based on the correlation matrix. We compared PCA factor scores with a paired t test. We predicted that factor scores before and after plant removal would differ in treated sites but not in untreated (control) sites.

To relate lizard abundance to changes in vegetation, we correlated species' occurrences and relative abundances with factor scores derived from the PCA. For three species of lizards that were not ubiquitous across regions, we used a logistic regression analysis to determine which factor scores best predicted the presence of species. We used a variable selection procedure that first eliminated variables with p greater than 0.25 (Hosmer & Lemeshow 2000) and then computed a backward elimination logistic regression. We used stepwise regression to identify which factor scores correlated with abundances of species. For the three species that were not ubiquitous, we conducted the regression analysis with abundance data only from sites where each species occurred. We considered sites as independent and our basic sampling unit.

To determine how lizard abundance differed before and after plant removal, we compared lizard abundance using both parametric and nonparametric BACI (before, after, control, impact) type designs. We used repeated measures analysis of variance (version 13.0; SPSS Inc., Chicago, IL, U.S.A.; mixed procedure with first-order autoregressive) with years as the repeated effect to test for effects of region (North, Middle, and South), period (pre- versus post-treatment), treatment (control versus treated site), and period × treatment interaction (test for treatment effect) on lizard abundance. If treatment effect was detected, a second analysis was run to test for period × type interaction (test for differences among the three types of treatment). Number of lizards per site was transformed with a square root transformation to meet assumptions of normality. Pre-treatment years were 2001–2002 and post-treatment years were 2003–2006, except for two sites, which were treated in 2004 and 2005.

We chose to test lizard abundance by also using a nonparametric design with the estimation of period × treatment interaction split into two steps: (1) we computed the difference between number of lizards in each treated site and its respective control site each year and divided it by number of lizards in both sites and then (2) we averaged this metric for all pre-treatment years and all post-treatment years and estimated the interaction by testing the paired differences in time.

The reasons we used this metric to estimate lizard abundance was to handle complications in treated sites and to improve comparability among sites by accounting for temporal variability unrelated to plant removal, which could not be addressed in the repeated measures analysis. Non-native plants were removed from treated sites in 2003, 2004, and 2005 in Middle region. Our metric allowed for comparisons of lizard abundance in treated sites to abundance in control sites on a year-to-year basis.

The second step of the interaction combined pre-treatment years for comparison with post-treatment years. This allowed us to take advantage of the power of a paired design (Zar 1996). This analysis provided a simple framework for interpreting whether plant removal had an effect on lizard abundance. The intent of the two-step process was to determine whether differences between treated and control sites changed between pre- and post-treatment periods.

We used a nonparametric Wilcoxon signed rank procedure to test for significant differences between periods. Values of the metric were non-normally distributed and standard data transformations (i.e., square root and log) were not appropriate because data were both positive and negative as well as greater than and less than 1. A nonparametric Wilcoxon signed rank test is the equivalent and acceptable test to replace a repeated measures analysis or paired *t* test (Hollander & Wolf 1999).

Results

Vegetation

Treatments to remove non-native plants and reduce fuels changed vegetation characteristics of the riparian forest (Tables 1 & 2). Treatments were highly effective in removing saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*) (Table 1). However, treatments redistributed debris from the understory, in the form of dead branches and logs, and increased debris on the ground, in the form of wood chips and small sticks left over from slash piles and chainsaw use (Table 1; Merritt et al. 2006).

Five elements explained 75% of the variation in riparian forest vegetation before and after treatments (PCA; Table 2). The forest was differentiated based on the following: factor 1, understory structure (i.e., areas with surface litter, debris heaps, and thickets of non-native plants); factor 2, understory composition (i.e., park-like understory lacking non-native plants); factor 3, overstory structure (i.e., mature stands of cottonwood [*Populus deltoides*] trees with few shrubs); factor 4, plant cover (i.e., grassy or weedy ground cover); and factor 5, surface cover (i.e., deep layer of surface litter). Vegetation in the riparian forest significantly differed over time (Table 3). After plant removal treated sites had significantly less surface litter and debris heaps and were significantly more park-like with an understory lacking non-native plants (Table 3). Control sites became more grassy and weedy over time (Table 3).

Lizard Community

During our 7-year study, we recorded 15,246 captures of 8,332 individual lizards representing 11 species (Appendix 2). We report results for relative abundances of the six most common species, which represent 98% of all captures. Other species of lizards were too few to include in analyses; however, all species are represented when summarized as "all lizards."

Lizard relative abundance varied across treated and control sites during the period of study (Fig. 3). Relative abundances and presence/absence of species of lizards were related to vegetation characteristics of the riparian forest. Four lizard species were associated with vegetation characteristics that significantly changed after non-native plant removal (Table 3). Two species, Great Plains skinks (*Plestiodon obsoletus*) and Eastern fence lizards (*Sceloporus consobrinus*), were negatively associated with surface litter, debris heaps, and thickets of non-native plants (factor 1; Tables 4 & 5). Two species, Chihuahuan spotted whiptails (*Aspidoscelis exsanguis*) and Desert grassland whiptails (*A. uniparens*), were positively associated with a park-like understory (factor 2; Tables 4 & 5). Two species, Great Plains skinks and Side-blotched lizards (*Uta stansburiana*), were positively associated with deep surface litter (factor 5; Tables 4 & 5), a variable that did not change after plant removal.

Table 1. Mean and standard error of 15 vegetation variables measured before and after non-native plant removal along the MRG, New Mexico, U.S.A.

Variable	*Pre-Treatment, $\bar{X}$ (SE)*	*Post-Treatment, $\bar{X}$ (SE)*
Control sites		
Bare ground (%)	12.2 (12.2)	14.9 (12.9)
Wood chips ground cover (%)	0	0
Forbs and grass ground cover (%)	17.2 (8.7)	49.9 (32.7)
Litter cover (%)	77.1 (10.1)	67.3 (12.3)
Depth of litter (cm)	4.8 (0.6)	4.6 (0.3)
Woody debris ground cover (%)	7.3 (1.7)	11.3 (3.7)
Number of dead branches, small diameter (2–5 cm)	39.8 (2.6)	40.4 (9.9)
Number of dead branches, long diameter (>5 cm)	8.9 (1.4)	8.2 (1.6)
Number of shrubs	0.7 (0.0)	6.3 (5.7)
Number of exotic trees	1.8 (0.8)	1.1 (0.6)
Average cottonwood diameter (cm)	25.0 (12.8)	21.3 (10.5)
Average Russian olive diameter (cm)	2.0 (1.1)	2.9 (2.1)
Average saltcedar diameter (cm)	5.1 (0.5)	4.9 (3.0)
Canopy cover (%)	87.1 (8.2)	86.5 (5.4)
Cottonwood basal area (m^2/ha)	18.1 (2.7)	21.9 (4.7)
Treated sites		
Bare ground (%)	4.6 (1.8)	8.2 (2.5)
Wood chips ground cover (%)	0	14.7 (5.2)
Forbs and grass ground cover (%)	33.7 (12.3)	54.0 (24.1)
Litter cover (%)	81.9 (3.3)	56.6 (5.0)
Depth of litter (cm)	3.5 (0.3)	3.9 (0.3)
Woody debris ground cover (%)	8.2 (1.3)	11.3 (2.5)
Number of dead branches, small diameter (2–5 cm)	42.5 (5.9)	33.9 (7.4)
Number of dead branches, long diameter (>5 cm)	14.2 (2.4)	10.9 (4.1)
Number of shrubs	2.0 (0.9)	3.7 (2.4)
Number of exotic trees	3.0 (1.0)	0
Average cottonwood diameter (cm)	24.6 (3.2)	24.4 (5.9)
Average Russian olive diameter (cm)	2.6 (1.4)	0
Average saltcedar diameter (cm)	4.4 (0.9)	0
Canopy cover (%)	87.7 (4.9)	81.9 (3.4)
Cottonwood basal area (m^2/ha)	14.1 (0.6)	15.6 (2.1)

Effects on Lizard Abundances

Both parametric and nonparametric analyses revealed a consistent pattern of increase in abundance for two species and no significant decrease in abundance for any species after non-native plants were removed.

We detected significant effects of region and plant removal treatment for some species from the repeated measures analysis (Table 6). Desert grassland whiptails, New Mexico whiptails (*Aspidoscelis neomexicana*), and Eastern fence lizards did not occur at every site resulting in differences among regions. We detected a weak treatment effect for New Mexico whiptails and Eastern fence lizards; both species increased in abundance after plant removal in treated sites (Fig. 4). We ran a second analysis to test for differences among the three types of treatment for Eastern fence lizards and found no evidence that types differed (numerator $df = 2$, denominator $df = 41.0$, $f = 0.8$, $p = 0.455$).

Similarly, we detected treatment effects on lizard abundances from the Wilcoxon signed rank test (Table 7). New Mexico whiptails significantly increased, and Eastern fence lizards showed a weak increase in treated sites compared to control sites after plant removal (Fig. 5).

Discussion

Riparian forests along the MRG are currently the focus of rehabilitation efforts that aim to reduce wildfire occurrences and the extent and distribution of such non-native plants as saltcedar (*Tamarix* spp.) and Russian olive (*Elaeagnus angustifolia*), as well as reestablish native bottomland species. As part of understanding how the eradication of non-native plants can alter habitat and affect wildlife, our study showed that some species of lizards responded positively to non-native plant removal and none of the lizard species that were studied significantly decreased in abundance or disappeared from sites when non-native plants were removed.

Although the effects of non-native plant removal on reptiles have not been well studied, our results tend to agree with the small pool of studies on this topic that have been conducted. For example, Pilliod et al. (2006)

Table 2. Five factors best explain the variance among 12 sites before and after non-native plant removal along the MRG, New Mexico, U.S.A.

Vegetation Variables	*Factor 1*	*Factor 2*	*Factor 3*	*Factor 4*	*Factor 5*
Bare ground (%)	−0.532	−0.082	−0.262	0.441	0.489
Wood chips ground cover (%)	−0.388	0.175	**0.590**	−0.436	−0.237
Forbs and grass ground cover (%)	−0.593	0.097	0.176	**0.529**	0.004
Litter cover (%)	**0.738**	−0.293	−0.304	−0.346	−0.091
Depth of litter (cm)	0.189	0.197	0.068	−0.424	**0.768**
Woody debris ground cover (%)	0.475	**0.797**	−0.036	0.114	0.073
Number of dead branches, small diameter (2–5 cm)	**0.697**	0.468	−0.361	−0.006	−0.086
Number of dead branches, long diameter (>5 cm)	**0.601**	0.582	−0.248	0.164	−0.102
Number of shrubs	−0.457	0.138	**−0.606**	0.000	−0.029
Number of exotic trees	**0.672**	**−0.548**	0.051	0.276	−0.115
Average Cottonwood diameter (cm)	0.315	0.475	0.368	**0.566**	−0.079
Average Russian olive diameter (cm)	**0.606**	−0.476	0.135	0.192	−0.211
Average saltcedar diameter (cm)	**0.696**	−0.332	−0.052	0.076	0.391
Canopy cover (%)	0.373	−0.218	**0.565**	0.280	0.292
Cottonwood basal area (sq m/ha)	0.281	0.318	**0.582**	−0.252	0.053
Cumulative variance explained (%)	28.5	44.4	57.2	67.8	75.9

Correlation matrix shows major variables (in bold) that influence factor scores.

Table 3. Differences in vegetation characteristics before and after removing non-native plants along the MRG, New Mexico, U.S.A. (from a PCA).

		Treated Sites		*Control Sites*	
Variables		*t*	*p*	*t*	*p*
Factor 1	Debris heaps, thickets of non-native plants	5.79	**<0.001**	0.35	0.759
Factor 2	Park-like understory	−2.35	**0.046**	−1.56	0.258
Factor 3	Mature cottonwood stand	−1.40	0.199	0.07	0.952
Factor 4	Grassy, weedy	0.90	0.394	−6.44	**0.023**
Factor 5	Deep surface litter	−0.31	0.764	−0.02	0.983

Paired *t* tests show major changes (in bold) in factor scores.

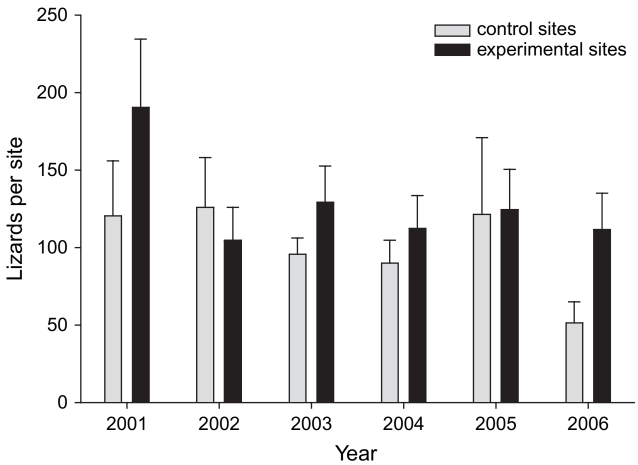

Figure 3. Mean and standard error of individual lizards captured in 12 sites along the MRG, New Mexico, U.S.A. Lizard abundance is the number of individual lizards captured per year. Non-native plants were removed from seven experimental sites in 2003, one site in 2004, and one site in 2005. Non-native plants remained intact in control sites.

suggested that some reptiles may benefit when shrubs, ground cover, and litter are removed. A study in the southeastern United States monitored herpetofaunal species in restored sites (i.e., treated to remove hardwoods) and reference sites of longleaf pine (*Pinus palustris*) in Florida (Litt et al. 2001). Six-lined racerunners (*A. sexlineata*) and Eastern fence lizards had higher capture rates in prescribed burn sites compared to other types of treatments or control sites. More specific to arid and semiarid climates, several studies showed that reptile densities and diversity are lower in saltcedar habitats than in native vegetation habitats in the southwestern United States (Jakle & Gatz 1985; Jones 1988; Lovich & DeGouvenain 1998) and in Australia (Griffin et al. 1989).

For our study, four of six common species of lizards were either positively associated with habitat characteristic of treated sites or negatively associated with habitat characteristic of untreated sites. Lizard abundances may have increased in treated riparian forest along the MRG because treatments created a more open habitat with reduced densities of non-native trees and debris heaps. Open habitat is generally not a characteristic associated

Table 4. Presence of three species of lizards as predicted by vegetation characteristics from a logistic regression. Lizards were monitored in sites with non-native plants removed and in untreated control sites along the MRG, New Mexico, U.S.A.

Species	*Positive or Negative Correlation*	*Vegetation Characteristics*	*Significance*
Desert grassland whiptail	+	Park-like understory	$p = 0.001$ (87.5%)
	−	Mature cottonwood stand	
	−	Grassy, weedy cover	
Side-blotched lizard	−	Mature cottonwood stand	$p = 0.004$ (95.8%)
	+	Grassy, weedy cover	
	+	Deep surface litter	
Eastern fence lizard	−	Debris heaps, thickets of non-native plants	$p = 0.059$ (75.0%)

Classification accuracies of models are in given parentheses.

Table 5. Lizard abundance as it relates to vegetation characteristics from a linear regression. Lizards were monitored in sites with non-native plants removed and in untreated control sites along the MRG, New Mexico, U.S.A.

Species	*Positive or Negative Correlation*	*Vegetation Characteristics*	*Significance, r^2*
Chihuahuan spotted whiptail	+	Park-like understory	$p = 0.024$, $r^2 = 0.210$
Desert grassland whiptail			Not significant
New Mexico whiptail			Not significant
Great Plains skink	+	Mature cottonwood stand	$p = 0.007$, $r^2 = 0.337$
	+	Deep surface litter	
Side-blotched lizard	−	Debris heaps, thickets of non-native plants	$p = 0.003$, $r^2 = 0.502$
	−	Mature cottonwood stand	
	+	Deep surface litter	
Eastern fence lizard	−	Debris heaps, thickets of non-native plants	$p = 0.028$, $r^2 = 0.379$
	+	Grassy, weedy	

Table 6. Results from repeated measures analysis of variance model testing for effects of region (north, middle, south), period (pre- versus post-treatment), treatment (control versus treated sites), and period × treatment interaction of square root transformed lizard abundance along the MRG, New Mexico, U.S.A.

Species	*Source*	*Numerator df*	*Denominator df*	*f*	*p*	*Interpretation*
Chihuahuan spotted whiptail	Region	2	87.9	1.8	0.225	
	Period	1	58.1	0.2	0.648	
	Treatment	1	8.0	0.4	0.522	
	Period × treatment	1	58.1	0.4	0.537	
Desert grassland whiptail	Region	2	8.0	5.2	0.035	Rare in middle region
	Period	1	56.2	0.3	0.571	
	Treatment	1	8.0	2.7	0.141	
	Period × treatment	1	56.2	1.4	0.235	
New Mexico whiptail	Region	2	8.5	0.9	0.460	
	Period	1	57.4	0.2	0.658	
	Treatment	1	8.6	0.5	0.513	
	Period × treatment	1	57.4	1.7	0.199	Weak treatment effect
Great Plains skink	Region	2	10.7	8.1	0.007	Most abundant in middle region
	Period	1	62.6	0.1	0.795	
	Treatment	1	11.3	2.4	0.148	
	Period × treatment	1	62.6	0.8	0.374	
Eastern fence lizard	Region	2	8.3	6.5	0.021	Rare in north region
	Period	1	55.8	0.7	0.416	
	Treatment	1	8.3	0.0	0.874	
	Period × treatment	1	55.8	3.2	0.080	Weak treatment effect
All lizard species*	Region	2	8.1	0.6	0.554	
	Period	1	59.2	0.5	0.467	
	Treatment	1	8.3	0.6	0.453	
	Period × treatment	1	59.2	1.5	0.229	

*A total of 11 species of lizards were sampled.

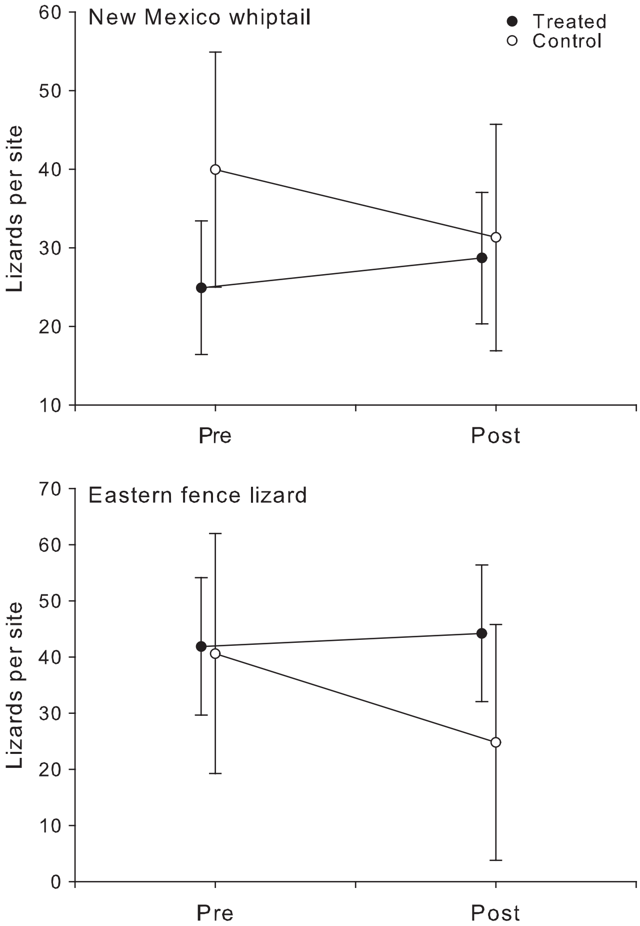

Figure 4. Mean and standard error of numbers of individual New Mexico whiptails (*Aspidoscelis neomexicana*; top panel) and Eastern fence lizards (*Sceloporus consobrinus*; bottom panel) in control sites (open circles) and treated sites (black circles) before and after non-native plant removal along the MRG, New Mexico, U.S.A.

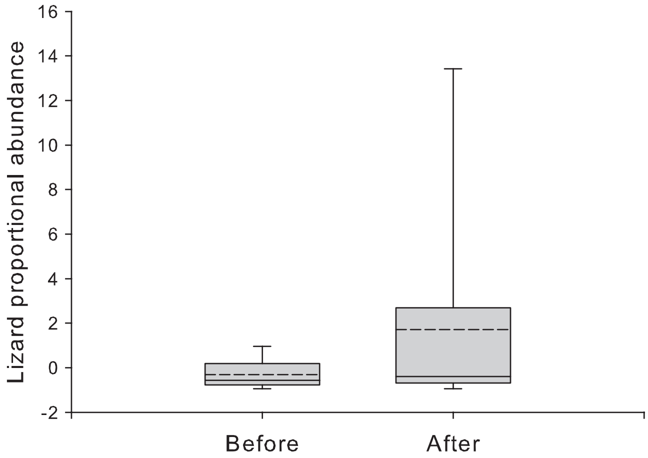

Figure 5. New Mexico whiptails (*Aspidoscelis neomexicana*) significantly increased in experimental sites after non-native plant removal along the MRG, New Mexico, U.S.A. Box plot shows abundance metric (number of lizards in treated sites minus number of lizards in control sites divided by number of lizards in both sites) before and after non-native plant removal. Means are dashed lines and positive values indicate abundance of lizards is greater in treated sites than in control sites.

with riparian areas overrun by saltcedar and other non-native plants, and it may simply be that the lizards we studied prefer mosaic habitats punctuated by areas open to direct sunlight—characteristics that are typically associated with bottomland habitat found along southwestern streams not affected by impoundment. Our results support this theory as the patchy cottonwood overstory did not change due to plant removal, yet the understory vegetation did. Perhaps, removing non-native plants in the understory allows more opportunities for heliothermic lizards to bask in areas where light does penetrate the cottonwood canopy (sensu Jakel & Gatz 1985).

Large annual variation in relative abundances of lizards and complications in completing treatments made it difficult to discern how the method used to remove non-native plants affected the lizard community. Accounting for population fluctuations that occur as a result of varying precipitation and subsequent abundance of arthropods is a challenge inherent to population studies of insectivorous lizards (Ballinger 1977; Maury 1995; Smith 1996). Typical of semiarid, desert ecosystems, there was great fluctuation in precipitation during our 7-year study: the summers 2002 and 2003 were among the driest on record and winter 2005 and summer 2006 among the wettest on record (NOAA 2006). By using both parametric and nonparametric analyses that incorporated a metric of abundance to compare changes in treated sites to control sites, we were able to account for annual variation in lizard abundance and complications in completing treatments during our study, allowing us to identify an effect of non-native plant removal on abundance of lizards following treatments.

These results strengthen the validity of using lizards as indicators of changes in riparian habitat, particularly in relation to changes caused by non-native plant removal. Lizards occur in high numbers and are readily captured using pitfall or funnel traps (Heyer et al. 1994). Moreover,

Table 7. Results from a Wilcoxon signed rank test showing paired differences in lizard abundance metric in sites before and after non-native plant removal along the MRG, New Mexico, U.S.A.

Species	*T+*	*p*	*Interpretation*
Chihuahuan spotted whiptail	27	0.250	
Desert grassland whiptail	—	—	
New Mexico whiptail	40	0.040	Treatment effect
Great Plains skink	10	1.000	
Eastern fence lizard	19	0.094	Weak treatment effect
All lizard species*	30	0.426	

Abundance metric is number of lizards in treated sites minus number in control sites, divided by number in both sites.
*A total of 11 species of lizards were sampled.

lizards respond to structural changes to their habitat (Pianka 1967). An increase in abundance of lizards following the removal of non-native plants (or any other type of rehabilitation strategy) could be a sound indication that the treatment strategy has been effective and is headed in the right direction. In our study, treatments reduced the extent and distribution of the understory component as well as the amount of debris, mirroring some of the habitat changes that typically occur when bottomland surfaces experience flooding. Even though most of the lizards captured during this study were not riparian obligates (i.e., captured species were more typical of such upland habitats as juniper-grassland [*Juniperus* spp.–*Gramma* spp.] and mesquite-creosote bush shrubland [*Prosopis* spp.–*Larrea tridentate*; Degenhardt et al. 1996]), their positive response to artificial treatment actions that replicate some of the consequences of such natural disturbances as flooding is not surprising. Although many of the captured species are typically associated with open, sparse vegetation (Christiansen et al. 1971), some lizard species are abundant in river habitats due to higher levels of insect food resources (Warren & Schwalbe 1985; Sabo & Power 2002).

Conclusions

Overall, removing non-native plants and associated woody debris in riparian forests along the MRG is necessary to reduce the risk of fire and the potential for loss of native cottonwoods (*Populus deltoides*). The spatial and temporal scale and experimental design of our study allowed us to investigate the effects of non-native plant and woody debris removal on relative abundances of lizards. These treatments seemed to be beneficial, or at least be non-damaging, to the lizard community of the MRG. For two common species of lizards (New Mexico whiptail [*Aspidoscelis neomexicana*] and Eastern fence lizard [*Sceloporus consobrinus*]), removal of non-native plants and woody debris seemed to increase relative abundance. Because relative abundance of lizards did not differ among the three types of removal treatments, we recommend to land managers that the monetary investment in prescribed burning may not necessarily result in an increase in abundance of lizards any more than physical removal of non-native plants and subsequent herbicide application.

We recommend comparing the response of several taxa to non-native plant removal over years and locations. For example, similar research along the MRG showed that birds exhibited a mixed response to treatments to remove non-native plants. Some of the bird species that used mid-story habitats responded negatively to treatments, likely because these birds built nests and foraged in abundant mid-story habitat provided by non-native plants (Finch et al. 2006). In contrast, toads (*Bufo* spp.) appear to respond more strongly to hydrologic variables as ground-water elevation than they do to the removal of non-native plants (Bateman et al. in press).

Regardless, monitoring for at least several years following the termination of restoration activities should be a priority that will improve our knowledge of how both the removal of undesirable plants as well as the planting native shrubs will affect wildlife communities. In a study in California, vertebrates were monitored 4–5 years after native shrubs were planted to restore a riparian floodplain (Queheillalt & Morrison 2006). Riparian specialists such as amphibians, warblers, and woodrats were only present in reference sites with a full compliment of native riparian plants, suggesting that understory and canopy habitats had not yet developed sufficiently at restored sites to support these species. Therefore, replanting with native vegetation to restore understory and canopy habitats may be beneficial; however, it may take a decade or more to realize those benefits to wildlife.

Implications for Practice

- Use species of lizards as indicators to better understand the effects of various non-native plant and debris manipulations (between conducting little or no non-native plant removal and wholesale removal) on the quality of riparian habitat.
- Effects of removing non-native plants and woody debris in riparian forests are likely species specific; therefore, it is important to monitor the response of several taxa over a period of years and locations.
- Monitoring should include control sites with no restoration activities so that researchers can account for responses not associated with treatments.

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Appendix 1. Condition of control and experimental arrays from June to mid-September 2000–2006 along the MRG, New Mexico, U.S.A.

	Year					
Site	*2001*	*2002*	*2003*	*2004*	*2005*	*2006*
1	PT	PT	PT	PT	PT	PT
2	PT	PT	T3	T3	T3	T3
3	PT	PT	T1	T1	T1	//
4	PT	PT	T1	T1	T1	T1
5	PT	PT	T1	T1	T1*	T1
6	PT	PT	PT	PT	T2*	T2
7	PT	PT	PT	T3	T3	T3
8	/	PT	PT	PT	PT	PT
9	PT	PT	PT	PT	PT	PT
10	PT	PT	T3	T3	T3	—
11	PT	PT	T2	T2	T2	T2
12	PT	PT	T1	T1	T1	T1

PT, pre-treatment condition; T1, treated with mechanical removal of non-native plants; T2, methods from T1 plus burning of slash piles; T3, methods from T1 plus replanting with native shrubs; "/," data unavailable because previously monitored site 8 burned by wildfire and replacement was selected in 2002; "//," data unavailable because burned by wildfire; —, data unavailable because impacted by flash flooding.

Notes: Site numbers listed here corresponding to the following RMRS sites: 1, 2, 3, and 4 are NO1, NO2, NO3, and NO4; 5, 6, 7, and 8 are MI1, MI2, MI3, and MI7; 9, 10, 11, and 12 are SO1, SO2, SO3, and SO4. Year 2000 was excluded because preliminary data were collected from a shorted season.

*Traps inundated with flood waters and traps opened 13 June 2005 in site 6 and 24 June 2005 in site 5.

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Appendix 2. Species of lizards captured June to September from 2000 to 2006 along the MRG, New Mexico, U.S.A.

Scientific Name	*Common Name*
Aspidoscelis exsanguis	Chihuahuan spotted whiptail
A. inornata	Little striped whiptail
A. neomexicana	New Mexico whiptail
A. tesselata	Checkered whiptail
A. tigris	Western whiptail
A. uniparens	Desert grassland whiptail
Plestiodon obsoletus	Great Plains skink
Phrynosoma cornutum	Texas horned lizard
Sceloporus (magister) bimaculosus[a]	Desert spiny lizard
Sceloporus (undulatus) consobrinus[b]	Eastern fence lizard
Uta stansburiana	Side-blotched lizard

[a] Name change based on Schulte et al. (2006).
[b] Name change based on Leache and Reeder (2002).