Herpetofauna Community Responses to

Saltcedar (*Tamarix* spp.) Biological Control and Riparian Restoration

Along a Mojave Desert Stream, U.S.A.

By

Kent Mosher

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Master of Science

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Graduate Supervisory Committee:

Heather Bateman, Chair
Juliet Stromberg
William Miller

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ABSTRACT

In riparian ecosystems, reptiles and amphibians are good indicators of environmental conditions. Herpetofauna have been linked to specific microhabitat characteristics, microclimates, and water resources in riparian forests. My objective was to relate herpetofauna abundance to changes in riparian habitat along the Virgin River caused by the *Tamarix* biological control agent, *Diorhabda carinulata*, and riparian restoration.

During 2013 and 2014, vegetation and herpetofauna were monitored at 21 riparian locations along the Virgin River via trapping and visual encounter surveys. Study sites were divided into four stand types based on density and percent cover of dominant trees (*Tamarix*, *Prosopis*, *Populus*, and *Salix*) and presence of restoration activities: *Tam*, *Tam-Pros*, *Tam-Pop/Sal*, and Restored *Tam-Pop/Sal*. Restoration activities consisted of mechanical removal of non-native trees, transplanting native trees, and introduction of water flow. All sites were affected by biological control. I predicted that herpetofauna abundance would vary between stand types and that herpetofauna abundance would be greatest in Restored *Tam-Pop/Sal* sites due to increased habitat openness and variation following restoration efforts.

Results from trapping indicated that Restored *Tam-Pop/Sal* sites had three times more total lizard and eight times more *Sceloporus uniformis* captures than other stand types. *Anaxyrus woodhousii* abundance was greatest in *Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites. Visual encounter surveys indicated that herpetofauna abundance was greatest in the Restored *Tam-Pop/Sal* site compared to the adjacent Unrestored *Tam-Pop/Sal* site. Habitat variables were reduced to six components using a principle component analysis and significant differences were detected among stand types. Restored *Tam-Pop/Sal* sites were most similar to *Tam-Pop/Sal* sites. *S. uniformis* were positively associated with large woody debris and high densities of *Populus*, *Salix*, and large diameter *Prosopis*.

Restored *Tam-Pop/Sal* sites likely supported higher abundances of herpetofauna, as these areas exhibited greater habitat heterogeneity. Restoration activities created a mosaic habitat by reducing canopy cover and increasing native tree density and surface water. Natural resource managers should consider implementing additional restoration efforts following biological control when attempting to restore riparian areas dominated by *Tamarix* and other non-native trees.

DEDICATION

To my parents, Vicky and Ralph,

for introducing me to the great outdoors,

and Stuart Wells,

for helping me start an exciting career.

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TABLE OF CONTENTS

APPENDIX Page

LIST OF TABLES

LIST OF FIGURES

Figure Page

Introduction

In riparian ecosystems of the western United States, reptiles and amphibians are good indicators of environmental conditions and habitat quality (Bateman & Paxton 2010). About 60% of herpetofauna species in the Chihuahuan, Great Basin, Mojave, and Sonoran Deserts utilize riparian or wetland habitats and about half of these species are true riparian or wetland obligates (Lowe 1989). Furthermore, many herpetofauna species in riparian areas are linked to specific microhabitat characteristics (Szaro & Belfitt 1986; Vitt et al. 2007; Bateman & Ostoja 2012). For example, Bateman and Ostoja (2012) found that yellow-backed spiny lizards (*Sceloporus uniformis*, formally *S. magister uniformis*; Schulte et al 2006) along the Virgin River were associated with riparian habitats containing native trees, woody debris, and logs; whereas, common side-blotched lizards (*Uta stansburiana*) and long-tailed brush lizards (*Urosaurus graciosus*) were associated with open sites consisting of low canopy or shrub cover. Since herpetofauna are ectothermic and required to maintain optimal body temperatures necessary for physiological and reproductive processes (Huey 1982), behavioral thermoregulation is an important driver of microhabitat selection in reptiles. For instance, both common side-blotched lizards (Goller et al. 2014) and long-tailed brush lizards (Adolph 1990) have been found to select for specific microhabitats that allow for suitable behavioral thermoregulation. In addition, these lizards have also been found to respond to changes in microhabitat structure, temperature, and humidity caused by non-native tree mechanical removal (Bateman et al. 2008a) and biological control (Bateman et al. 2014). Therefore, examining herpetofauna communities in riparian areas can provide important information on how higher-trophic level species respond to changes in riparian habitat.

Saltcedar (*Tamarix* spp.) was initially introduced to the United States in the 19th century as an ornamental species and a solution to erosion issues in the American West (Robinson 1965). Due to changes in stream hydrology (Everitt 1998; Shafroth et al. 2002) and the plant's unique tolerances to drought and saline conditions (Smith et al. 1998), saltcedar spread rapidly and is now the third most abundant riparian tree in the western United States (Friedman et al. 2005). Saltcedar can form dense, monotypic stands (Deloach et al. 1999), which have been linked to a decline in richness and diversity of native plant (Engel-Wilson & Ohmart 1978; Lovich et al. 1994) and wildlife (Anderson et al. 1977; Knutson et al. 2003; Durst et al. 2008) species in riparian areas. As a result, natural resource managers have invested millions of dollars to control this non-native species (Shafroth & Briggs 2008).

There is still much debate surrounding the effects of saltcedar, as studies have also shown that saltcedar habitat can support a wide variety of riparian wildlife. For example, van Riper III et al. (2008) found that bird abundance along the lower Colorado River was highest at sites where saltcedar comprised 40 to 60% of riparian habitat. Furthermore, abundance and richness of arthropods (Ellis et al. 2000) and small mammals (Ellis et al. 1997) were observed to be greater in saltcedar-dominated habitats compared to cottonwood (*Populus spp.*)-dominated habitats in the Middle Rio Grande Valley, New Mexico (these differences were attributed to the presence of arid-adapted wildlife species at saltcedar-dominated sites). Therefore, it is important to determine if saltcedar is playing a functional role in a riparian ecosystem prior to implementing control efforts.

A variety of techniques have been utilized to remove saltcedar, including burning, chemical and mechanical removal, and the introduction of biological control agents. Each method has its own advantages and disadvantages that largely depend on current habitat conditions, site accessibility, and economic constraints (Shafroth et al. 2008; O'Meara et al. 2010). Wildlife responses to saltcedar removal efforts are often complex. For example, the relative abundance of lizard species along the Middle Rio Grande, New Mexico increased following saltcedar and Russian olive (*Elaeagnus angustifolia*) removal in mixed riparian habitats due to increased habitat openness (Bateman et al. 2008a). Whereas, another study in the same area reported a significant decline in mid-story bird densities as removal of saltcedar and Russian olive greatly reduced mid-story habitat (Finch & Hawksworth 2006).

Saltcedar removal in riparian areas can fail or cause negative effects to habitat and wildlife if managers do not fully understand the biological history of the location, the reasons behind the initial saltcedar colonization, and/or the impacts resulting from saltcedar control (Zavaleta et al. 2001; Shafroth et al. 2008). Long-term domination of saltcedar at a site may significantly alter soil chemistry (Yin et al. 2009) and microbial assemblages (Meinhardt & Gehring 2012), potentially preventing or limiting native plant species establishment (Zavaleta et al. 2001; Harm & Heibert 2006). Additionally, changes in stream hydrology that led to saltcedar-domination at a site, may no longer be able to support native vegetation (Stromberg et al. 2007). In these situations, saltcedar or other non-native plant species may recolonize restored areas (Shafroth et al. 2005; Hultine et al. 2010; Ostoja et al. 2014). Even worse, large-scale removal of saltcedar may result in the removal of the only habitat available for riparian wildlife species in the area (Hultine et al. 2010; Paxton et al. 2011).

As such, riparian locations affected by saltcedar often need to be actively managed and require additional restoration efforts to ensure success. Reintroducing historical hydrologic regimes (Décamps et al. 2004) and transplanting native plant species (Shafroth et al. 2005; Bay & Sher 2008) may be necessary to prevent non-native plant species colonization and to promote native plant species growth and establishment. Within the Bosque del Apache National Wildlife Refuge, New Mexico, active revegetation and irrigation of cottonwoods and willow (*Salix* spp.) following saltcedar removal resulted in an increase in the diversity and richness of a variety of wildlife species, including birds, herpetofauna, and small mammals (Taylor & McDaniel 1998).

In this study, I examined if herpetofauna communities responded to changes in habitat structure and riparian plant composition caused by saltcedar biological control and riparian restoration. I was particularly interested in if additional restoration efforts in areas following saltcedar biological control benefitted habitat and herpetofauna. My objectives were to (1) determine how riparian habitats differ in habitat structure and physiognomy; (2) determine if riparian habitat structure and physiognomy affects herpetofauna communities; and (3) determine how restored and unrestored riparian sites differ in herpetofauna communities.

Methods

Study Area

I established study sites in riparian areas along the Virgin River from St. George, Utah (UTMs NAD83 274660mE 4107912mN) to Gold Butte in Clark County, Nevada

(738634mE 4050113mN; Fig. 1; Appendix A). Study sites were divided into four riparian stand types based on density and percent cover of dominant woody trees and the presence of restoration activities: saltcedar-dominated stands (*Tam*), saltcedar-mesquite (*Prosopis* spp.) stands (*Tam-Pros*), saltcedar-cottonwood/willow stands (*Tam-Pop/Sal*), and restored saltcedar-cottonwood/willow stands (Restored *Tam-Pop/Sal*). Saltcedar consisted of *T. ramosissima* and related species/hybrids. Common shrub species at study sites included arrowweed (*Pluchea sericea*), seepwillow (*Baccharis* spp.), thornbush (*Lycium cooperi*), and saltbush (*Atriplex* spp.). Water flow in the Virgin River during 2013 and 2014 was perennial at all study sites; however, surface water within study sites varied depending on location. Utah sites had standing surface water with marshy habitat, whereas, all other sites had no surface water or marshes present.

Restored *Tam-Pop/Sal* sites were riparian areas where 50% of saltcedar and Russian olive were mechanically removed, native willow and mesquite stems were transplanted, and water flow was introduced via trenching and redirection of irrigation/stormwater runoff (Trujillo & Dobbs 2012; Edwards 2013). Saltcedar and Russian olive were removed via chainsaws and dragged to the edge of the riparian habitat where it was left in piles, wood chipped, or burned. Cut stumps were sprayed with the herbicide Garlon 3a. Restoration occurred during Winter/Spring of 2012 and 2013 by Utah Division of Wildlife Resources and Santa Clara Fire Department staff. Since Restored *Tam-Pop/Sal* sites were restricted to Utah and were not monitored prior to restoration, I used a space-for-time substitution. I compared a Restored *Tam-Pop/Sal* site to an adjacent Unrestored *Tam-Pop/Sal* site of similar habitat structure and composition as a proxy to pre-restored conditions (Appendix A).

All study sites were affected by biological control. In 2006, natural resource managers released the non-native Northern tamarisk beetle (*Diorhabda carinulata*; Tracy & Robbins 2009) in the city of St. George, Utah to control saltcedar stands along the Virgin River (Bateman et al. 2010). Larvae and adult beetles of this species feed exclusively on the foliage of saltcedar causing defoliation of the plant (Lewis et al. 2003). After several defoliation events, carbohydrate reserves necessary for re-growth become depleted due to lack of photosynthetic tissue available and the saltcedar dies (Hudgeons et al. 2007).

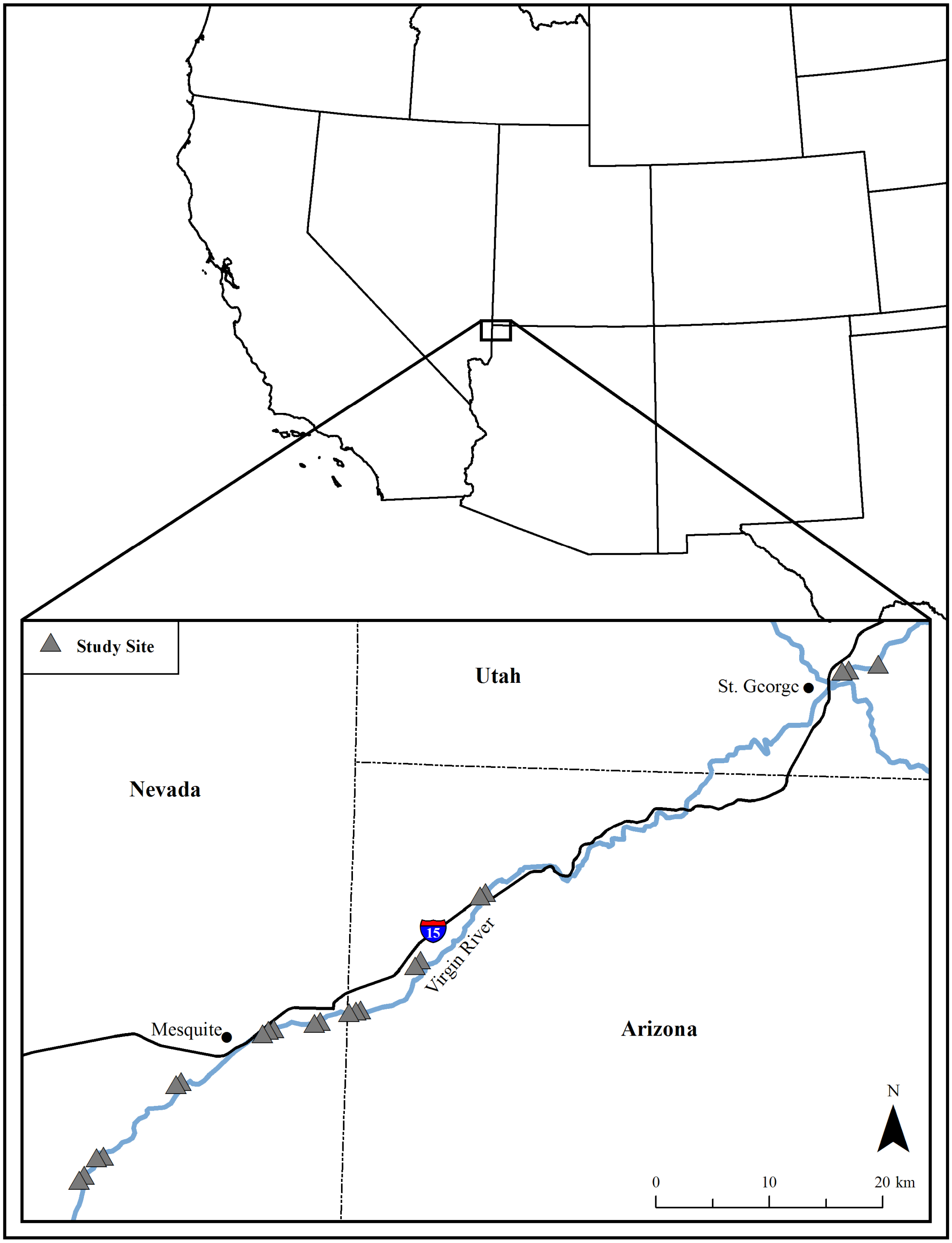

Figure 1. Map of study area along the Virgin River in Utah, Arizona, and Nevada. Study site locations offset to improve visibility on map.

Herpetofauna Sampling

To determine if stand type affected herpetofauna communities, I monitored herpetofauna at 21 sites (eight *Tam*, five *Tam-Pros*, six *Tam-Pop/Sal*, two Restored *Tam-Pop/Sal*) using trap arrays during 2013 and 2014. Trap arrays consisted of four pitfalls (9L) and six funnel traps positioned along three 6m long drift fences oriented at 0, 120, and 240 degrees (Fig. 2; Bateman & Ostoja 2012). Each array was randomly established using ArcGIS 9.3.1 and located at least 25m from habitat edge. I checked traps every 24 hours (when open) during May through July in 2013 and 2014. Captured herpetofauna were classified to species, measured, weighed, sexed, and released near the site of capture. Lizards were marked with a unique toe clip (Waichman 1992). Amphibians and snakes were not marked due to low abundances.

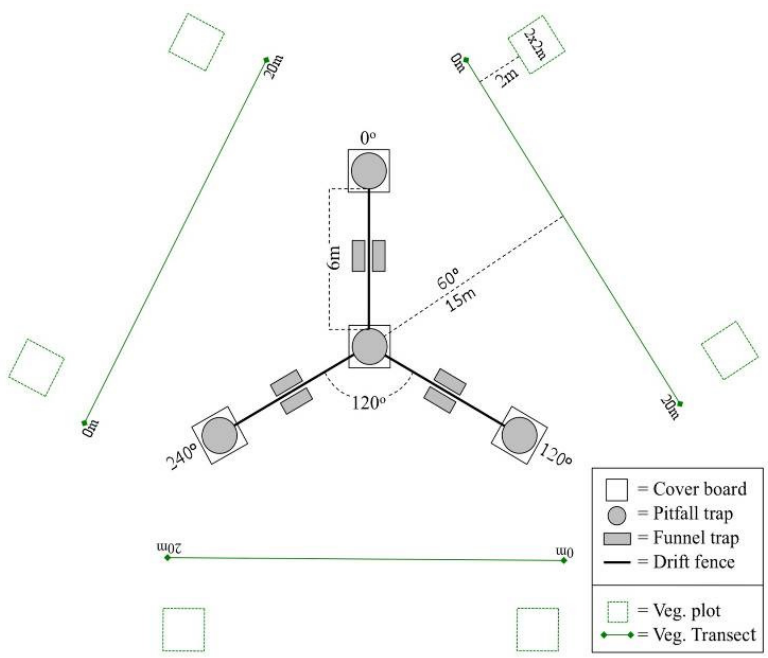

Figure 2. Diagram of trap array and vegetation transects and plots used to monitor herpetofauna and habitat along the Virgin River in Utah, Arizona, and Nevada. Not to scale.

To examine the effects of restoration on herpetofauna communities, I conducted visual encounter surveys (VES) at a Restored and Unrestored *Tam-Pop/Sal* site during 2013 and 2014. At each site, I randomly established three transects using ArcGIS 10.0 (Fig. 3). Transects were at least 150m apart, except for one transect at the restored site where permanent water restricted its placement to be about 125m to the nearest transect. Transects varied in length depending on the width of the site. Two, 10x20m plots were randomly placed along each transect (at least 25m from habitat edge) to ensure equal sampling effort between sites. Due to restoration activities and a fire at the Unrestored *Tam-Pop/Sal* site that compromised two transects prior to the 2014 field season, I established all new transects at both sites in 2014. I conducted four VES per field season between 0730 – 1130 hours and under similar weather conditions. Two observers walked side by side at the same pace within each plot. Each observer surveyed a 5m width (half of the plot) and limited their search to no higher than 2m above the ground. Observers searched debris piles and downed logs and moved vegetation to flush hidden herpetofauna. Observed herpetofauna were classified to species.

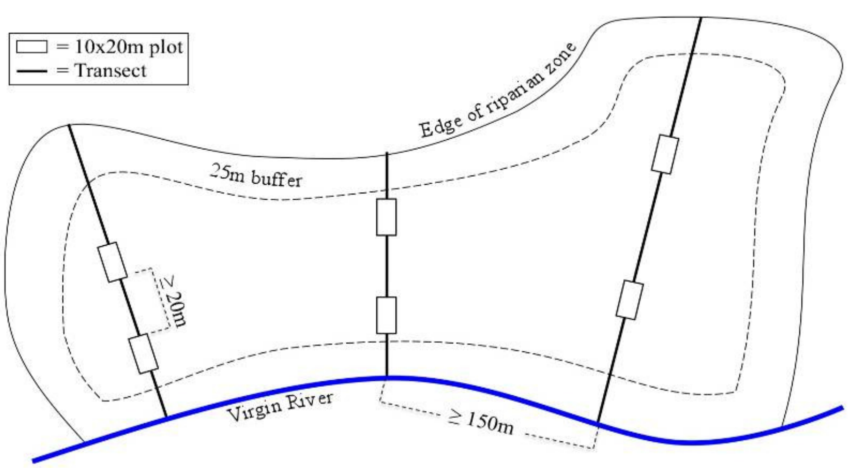

Figure 3. Diagram of visual encounter survey transects and plots at the Restored and Unrestored *Tam-Pop/Sal* sites in St. George, Utah. Not to scale.

Vegetation Sampling

I measured habitat structure and composition at all trapping and VES sites during 2013 and 2014. Vegetation was measured at trapping sites using two randomly selected 20m transects and four 2x2m plots (Fig. 2; Appendix B). Transects were located 15m from the center pitfall trap and orientated at 60, 180, or 300 degrees. Plots were located 2 meters away from both ends of each transect. At one meter intervals along each transect, I recorded ground cover type, depth of litter (if applicable), and woody tree and shrub cover type. At alternating meter intervals along each transect, I recorded the number and size class of woody debris (below 0.5m in height) that crossed each transect. In the 2x2m plots, I recorded the number of stems and size class of each plant species rooted within the plot (these variables were only measured in 2014). Canopy cover (spherical densiometer) and visible light (µmol; LI-COR LI-250A Light Meter) readings were taken in the four cardinal directions from the center of each plot. To calculate percent visible light, I averaged visible light readings in both plots and divided by a control reading (measured in area with no canopy cover). I measured vegetation at VES plots in 2014 along a 20m transect that ran through the center of the plot. The same variables recorded along the trapping transects were measured along the VES transects. Canopy cover and visible light readings were taken in the four cardinal directions at 0, 10, and 20m.

Data Analyses

I defined lizard abundance at trapping sites as the number of uniquely marked individuals captured per 100 trap days. This conservatively estimated abundance as the minimum number of animals per site. Since amphibians were not marked, I defined amphibian abundance as the total number of captures per 100 days. I calculated species richness as the average number of species recorded in a given stand type. Simpson's Diversity and Brillouin Evenness indices were calculated for each stand type using Species Diversity & Richness 4.1.2 Software (Seaby & Henderson 2006). To determine if there was a significant effect of year and stand type on lizard abundances, I used a repeated measures General Linear Model (GLM; SPSS version 22.0). All abundances were log(x+1) transformed to increase data normality.

I defined herpetofauna abundance at VES sites as the greatest number of individuals of each species detected for a given transect during any one of the four surveys conducted each year. This ensured that individuals were not counted twice throughout surveys. I calculated species richness per transect by summing all species observed during surveys. I performed a chi-square analysis on reptile and amphibian abundance to determine if abundances differed between the Restored and Unrestored *Tam-Pop/Sal* site.

I summarized the variation among habitat metrics at trapping sites using a principal component analysis (PCA) with a Varimax rotation (SPSS version 22.0). Habitat metrics that were recorded in both 2013 and 2014 were averaged prior to PCA. The number of relevant components was determined based on a scree plot, components with eigenvalues greater than one, and parsimony (Legendre & Legendre 1998). Components were compared based on the correlation matrix. Component scores were calculated for each site and graphed via a scatterplot to compare habitat variation at *Tam*, *Tam-Pros*, *Tam-Pop/Sal*, and Restored *Tam-Pop/Sal* sites. Components scores were compared among stand types using a one-way ANOVA. I compared VES vegetation measures at the Restored and Unrestored *Tam-Pop/Sal* sites with a *t*-test.

To determine if habitat components from the PCA were good predictors of herpetofauna presence, I ranked multiple-linear regression models using a multiple-model inference approach (Burnham & Anderson 2004). All possible models were considered in this analysis, meaning that all combinations of components were included (sensu Pennington et al. 2010). The “top model” was the model where $\Delta AIC = 0$; however, I also considered all models that had a $\Delta AIC = 2$. Variable weights were calculated to determine the relative importance of each component.

Results

Herpetofauna

During 2013 and 2014 (1,060 trap days), I captured eight species of lizards (656 unique individuals), three species of snakes, and three species of amphibians at 21 trapping sites along the Virgin River (Appendix C). Lizard captures were dominated by tiger whiptails (*Aspidoscelis tigris*), followed by common side-blotched lizards and yellow-backed spiny lizards (Table 1). These three species comprised 95% of unique lizards captured. The tiger whiptail was the only species captured at every site. Total lizard abundance was three times greater at restored *Tam-Pop/Sal* sites; *Tam, Tam-Pros,* and *Tam-Pop/Sal* sites had similar abundances (Fig. 4). Yellow-backed spiny lizards were the only species of lizard that exhibited preference among stand types. Yellow-backed spiny lizards had greatest abundance at restored *Tam-Pop/Sal* sites (Fig. 5). No yellow-backed spiny lizards were captured at *Tam* sites. Lizard diversity, evenness, and richness were greatest at Restored *Tam-Pop/Sal* sites; however, there was no significant difference among stand types (Table 2).

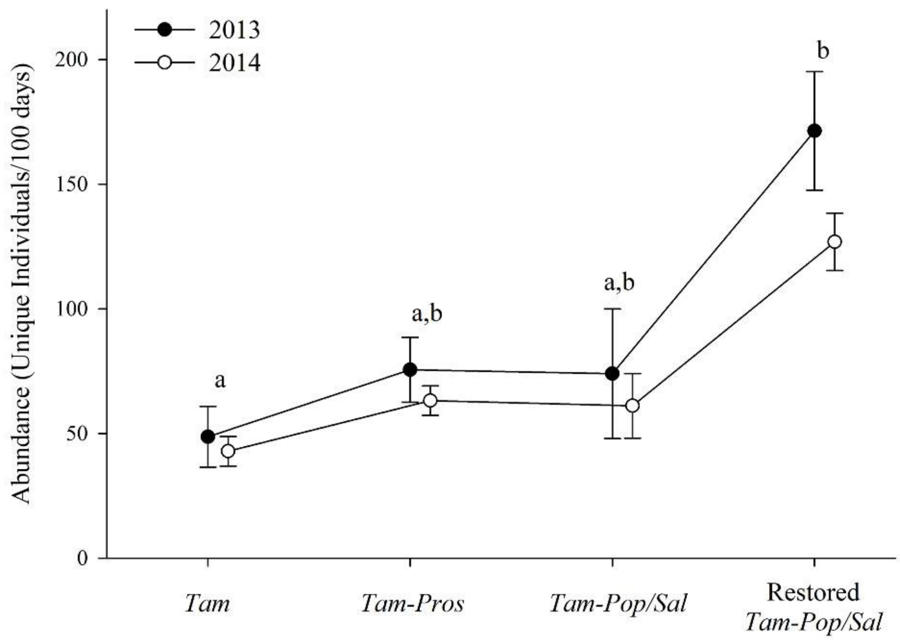

Figure 4. Mean (±SE) number of all lizards (unique individuals) captured per 100 trap days during 2013 and 2014 in *Tam, Tam-Pros, Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Arizona, and Nevada. Letters represent significant difference (based on repeated measures GLM).

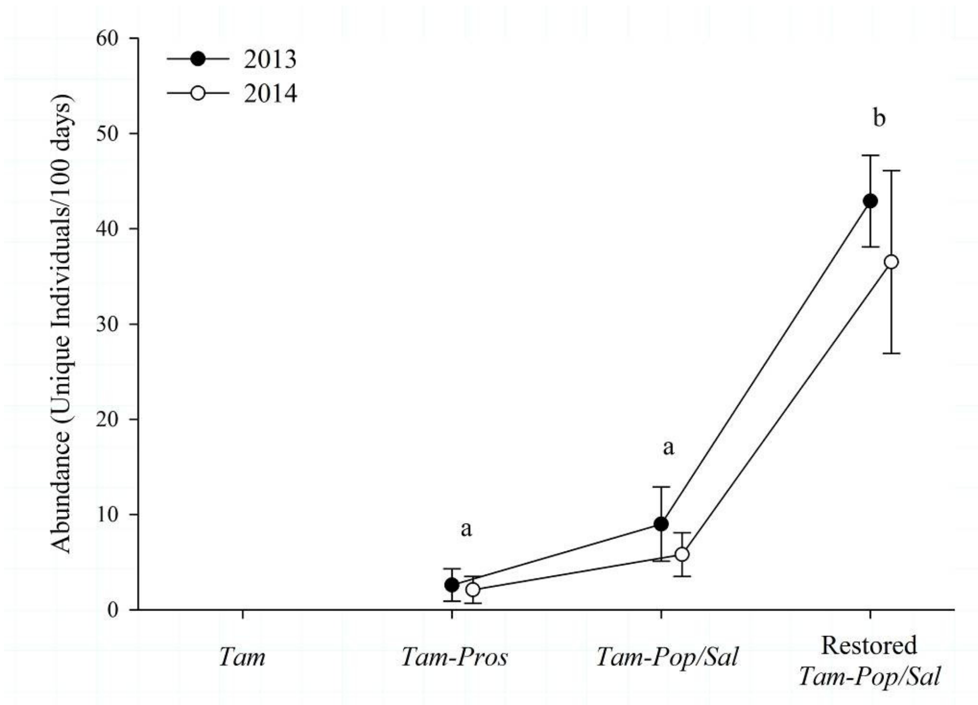

Figure 5. Mean (±SE) number of yellow-backed spiny lizards (unique individuals) captured per 100 trap days during 2013 and 2014 in *Tam, Tam-Pros, Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Arizona, and Nevada. Letters represent significant difference (based on repeated measures GLM).

Table 1. Mean (±SE) number of lizards (unique individuals) and amphibians (total captures) captured per 100 trap days during 2013 and 2014 in *Tam, Tam-Pros, Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Arizona, and Nevada. Species and class abundances were log(x+1) transformed to increase data normality. Species that were restricted to specific geographic locations (i.e., *A. velox*) or had low abundances were not included in analyses. Significant (P<0.05) terms in bold.

	2013				2014						
Family	*Tam*	*Tam-Pros*	*Tam-Pop/Sal*	Restored *Tam-Pop/Sal*	*Tam*	*Tam-Pros*	*Tam-Pop/Sal*	Restored *Tam-Pop/Sal*	Stand Type	Year	Interaction
Species	n=8	n=5	n=6	n=2	n=8	n=5	n=6	n=2	*F(P)*	*F(P)*	*F(P)*
Tiiedae											
Aspidoscelis tigris	35.0 (7.2)	50.1 (6.2)	46.9 (13.4)	61.9 (4.8)	28.2 (3.4)	52.2 (5.6)	39.2 (7.7)	67.3 (1.9)	2.70 (0.08)	0.02 (0.88)	0.12 (0.95)
Aspidoscelis velox	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	4.8 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-	-	-
Phrynosomatidae											
Calisaurus draconoides	0.5 (0.5)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-	-	-
Sceloporus uniformis	0.0 (0.0)	2.6 (1.7)	9.0 (3.9)	42.9 (4.8)	0.0 (0.)	2.1 (1.4)	5.8 (2.3)	36.5 (9.6)	**13.07 (0.00)**	0.41 (0.53)	0.10 (0.96)
Urosaurus graciosis	2.2 (0.9)	6.3 (5.2)	0.0 (0.0)	0.0 (0.0)	0.4 (0.4)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	1.55 (0.24)	3.37 (0.08)	1.40 (0.28)
Urosaurus ornatus	0.0 (0.0)	0.8 (0.8)	0.0 (0.0)	0.0 (0.0)	1.3 (0.9)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-	-	-
Uta stansburiana	9.9 (5.6)	15.8 (7.7)	18.2 (13.3)	61.9 (14.3)	12.5 (4.9)	8.3 (4.4)	16.1 (6.3)	23.1 (0.0)	1.07 (0.39)	0.21 (0.66)	1.45 (0.26)
Eublepharidae											
Coleonyx variegatus	1.0 (1.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.4 (04)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-	-	-
Bufonidae											
Anaxyrus microscaphus	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	2.4 (2.4)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	-	-	-
Anaxyrus woodhousii	11.8 (3.1)	7.2 (4.7)	43.8 (6.3)	42.9 (23.8)	3.5 (1.7)	13.1 (12.3)	8.6 (4.5)	115.4 (100.0)	**4.06 (0.02)**	**4.44 (0.05)**	2.98 (0.06)
Hylidae											
Pseudacris regilla	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	66.7 (42.9)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	19.2 (11.5)	-	-	-
All Lizards	48.7 (12.2)	75.6 (13.0)	74.0 (26.0)	171.4 (23.8)	42.9 (6.0)	63.2 (5.9)	61.1 (13.0)	126.9 (11.5)	**3.78 (0.03)**	0.99 (0.33)	0.242 (0.87)
All Amphibians	11.8 (3.1)	7.2 (4.7)	43.8 (6.3)	111.9 (64.3)	3.4 (1.7)	13.1 (12.3)	8.6 (4.5)	134.7 (111.5)	-	-	-

Table 2. Summary values for diversity measures of lizards captured at trap arrays in *Tam, Tam-Pros, Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Arizona, and Nevada.

Diversity Measure	*Tam*	*Tam-Pros*	*Tam-Pop/Sal*	Restored *Tam-Pop/Sal*	*F (P)*
Simpson's D	1.735	1.788	1.997	3.004	2.000 (0.152)
Brillouin E	0.549	0.502	0.626	0.814	0.511 (0.680)
Richness	2.875	2.600	2.500	4.000	0.880 (0.471)

Woodhouse's toads (*Anaxyrus woodhousii*) comprised 84% of total amphibians captured and were the only amphibian species captured at all stand types. Woodhouse's toad abundance was greatest at *Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites, but was significantly different between years (Fig. 6). Pacific tree frogs (*Pseudacris regilla*) were restricted to restored *Tam-Pop/Sal* sites, so no further analyses were conducted.

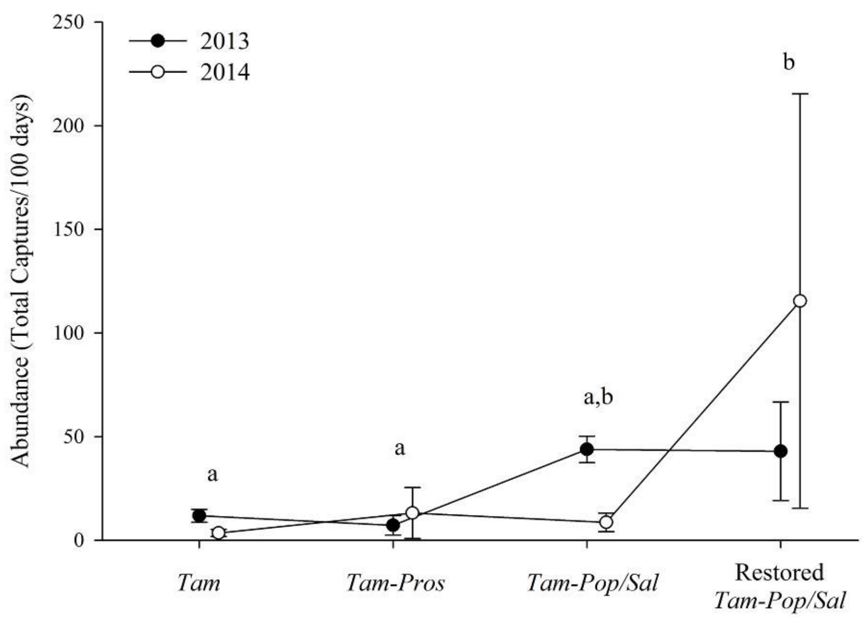

Figure 6. Mean (±SE) number of Woodhouse's toads (total captures) captured per 100 trap days during 2013 and 2014 in *Tam, Tam-Pros, Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Arizona, and Nevada. Letters represent significant difference (based on repeated measures GLM). There was a significant difference in abundance between 2013 and 2014

Visual observations along transects at the Restored and Unrestored *Tam-Pop/Sal* site indicated that reptile and amphibian abundance was greatest at the restored site (Table 3; Table 4). Abundances for individual species were too low to run analyses on the species-level. There was no significant difference in reptile and amphibian richness between sites (Table 4).

Table 3. Number of unique individuals observed during visual encounter surveys along transects at the Restored and Unrestored *Tam-Pop/Sal* site in St. George, Utah during four sampling occasions.

Site	Species	2013	2014	Total
Restored *Tam-Pop/Sal*	*Uta stansburiana*	6	2	8
	Sceloporus uniformis	3	1	4
	Unknown lizard	1	0	1
	Anaxyrus woodhousii	0	33	33
	Pseudacris regilla	34	181	215
	Lithobates catesbeianus	0	4	4
	Total Reptiles	10	3	13
	Total Amphibians	34	218	253
Unrestored *Tam-Pop/Sal*	*Uta stansburiana*	0	2	2
	Anaxyrus woodhousii	12	15	27
	Psuedacris regilla	1	0	1
	Lithobates catesbeianus	6	0	6
	Total Reptiles	0	2	2
	Total Amphibians	19	15	34

Table 4. Results of chi-square analyses on reptile and amphibian abundances at the Restored and Unrestored *Tam-Pop/Sal* site. Significant (P<0.05) terms in bold.

	Restored *Tam-Pop/Sal* (Observed)	Unrestored *Tam-Pop/Sal* (Observed)	$X_{0.05,1}$ (3.841)
Reptiles	13	2	**8.067**
Amphibians	253	34	**166.168**
Richness	5	4	0.111

Habitat

Habitat structure and physiognomy varied among stand types at trapping sites (Table 5). Six components explained 80% of the variation across trapping sites based on habitat structure and composition (Appendix D). Habitat was differentiated based on the following characteristics: (C1) overstory and mesquite/saltcedar cover, (C2) large woody debris and cottonwood/willow cover, (C3) cottonwood/willow density, (C4) small woody debris, (C5) small diameter saltcedar/large diameter mesquite density, and (C6) arrowweed/large diameter saltcedar density. C1 scores were highest at *Tam* sites (i.e., high canopy cover/low light, high saltcedar/low mesquite cover, and high litter ground cover) and lowest at *Tam-Pros* sites (i.e., low canopy cover/high light, low saltcedar/high mesquite cover, and high bare ground; Table 6). C2 scores were highest at *Tam-Pop/Sal* sites (i.e., high cottonwood/willow cover, deep litter, and high large woody debris) and were lowest at *Tam* sites (i.e., low cottonwood/willow cover, shallow litter, and low large woody debris; Table 6). C3 scores were highest at Restored *Tam-Pop/Sal* sites (i.e., high cottonwood/willow density) and lowest at *Tam* sites (i.e., low cottonwood/willow density); however, there was no significant difference among *Tam, Tam-Pros,* and *Tam-Pop/Sal* sites (Table 6). Overall, *Tam* sites consisted of high saltcedar cover, high canopy cover, and high shade (Fig. 7). *Tam-Pros* sites had high mesquite cover with open canopies and high bare ground. *Tam-Pop/Sal* sites exhibited greater variation in habitat structure, but typically had high cottonwood/willow cover, large woody debris, and intermediate levels of canopy cover. Restored *Tam-Pop/Sal* sites were more similar to *Tam-Pop/Sal* sites, but exhibited habitat characteristics of all stand types.

Table 5. Mean (±SE) of habitat variables measured during 2013 and 2014 at trap arrays in *Tam, Tam-Pros, Tam-Pop/Sal,* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Nevada, and Arizona. Density measures were not recorded in 2013.

Variable	*Tam* n=8	*Tam-Pros* n=5	*Tam-Pop/Sal* n=6	Restored *Tam-Pop/Sal* n=2
Bare Ground (%)	23.4 (5.2)	46.0 (6.4)	26.5 (7.2)	25.0 (6.3)
Woody Debris Ground Cover (%)	7.8 (2.7)	9.5 (5.3)	13.1 (2.8)	10 (3.8)
Litter Ground Cover (%)	68.8 (6.1)	44.5 (6.3)	60.4 (6.7)	65.0 (10.)
Litter Depth (cm)	1.3 (0.1)	1.1 (0.2)	2.8 (0.7)	1.4 (0.3)
Tamarix Cover (%)	90.4 (3.6)	44.4 (4.9)	51.2 (9.9)	35.6 (12.0)
Density of *Tamarix* (stems/10m^2)	55.6 (8.0)	56.5 (23.2)	23.8 (15.3)	4.4 (4.4)
Populus/Salix Cover (%)	0.0 (0.0)	0.0 (0.0)	27.0 (11.5)	14.5 (12.0)
Density of *Populus/Salix* (stems/10m^2)	0.0 (0.0)	0.0 (0.0)	1.6 (0.7)	17.5 (7.2)
Prosopis Cover (%)	0.0 (0.0)	19.6 (2.7)	4.4 (2.0)	3.9 (3.9)
Density of *Prosopis* (stems/10m^2)	0.8 (0.8)	0.3 (0.3)	0.1 (0.1)	0.3 (0.3)
Elaeangus Cover (%)	0.0 (0.0)	0.0 (0.0)	0.7 (0.7)	8.3 (8.3)
Density of *Elaeangus* (stems/10m^2)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
Pluchea Cover (%)	0.0 (0.0)	32.1 (13.3)	17.4 (6.4)	0.8 (0.8)
Density of *Pluchea* (stems/10m^2)	4.4 (4.4)	110.8 (33.9)	46.5 (19.7)	0.8 (0.8)
Baccharis Cover (%)	0.0 (0.0)	3.3 (3.3)	4.5 (1.5)	4.5 (2.0)
Density of *Baccharis* (stems/10m^2)	0.0 (0.0)	0.5 (0.5)	3.8 (2.7)	14.4 (4.1)
Atriplex Cover (%)	5.4 (3.5)	0.3 (0.3)	0.0 (0.0)	24.0 (22.5)
Density of *Atriplex* (stems/10m^2)	2.0 (2.0)	0.1 (0.1)	0.0 (0.0)	3.1 (3.1)
Lycium Cover (%)	0.0 (0.0)	4.8 (4.8)	0.0 (0.0)	0.0 (0.0)
Density of *Lycium* (stems/10m^2)	0.0 (0.0)	5.0 (5.0)	0.0 (0.0)	0.0 (0.0)
Number of Dead Branches, Sm. Diam. (1.0-2.5cm)/10m	25.4 (4.2)	26.0 (6.5)	36.3 (5.7)	29.0 (1.8)
Number of Dead Branches, Lg. Diam. (>2.5cm)/10m	4.2 (0.8)	2.5 (0.9)	8.5 (3.7)	3.8 (0.5)
Canopy Cover (%)	72.0 (5.3)	42.9 (10.1)	66.6 (7.0)	60.3 (12.7)
Visible Light (%)	28.5 (5.5)	56.9 (8.9)	44.7 (8.3)	44.5 (9.5)

Table 6. Mean (±SE) of component scores at trap arrays in *Tam, Tam-Pros, Tam-Pop/Sal,* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Arizona, and Nevada. To determine if there was a difference in component scores among stand types, a one-way ANOVA with Tukey's HSD was conducted. Significant (P<0.05) terms in bold.

Stand Type	C1 Overstory *Pros/Tam* Cover	C2 Lg. Woody Debris *Pop/Sal* Cover	C3 *Pop/Sal* Density	C4 Sm. Woody Debris	C5 Sm. *Tam*/ Lg. *Pros* Density	C6 *Pluchea*/ Lg. *Tam* Density
Tam	0.757 (0.208)	-0.466 (0.041)	-0.307 (0.044)	-0.096 (0.382)	0.278 (0.160)	-0.465 (0.166)
Tam-Pros	-1.159 (0.251)	-0.377 (0.160)	-0.234 (0.015)	-0.127 (0.552)	-0.132 (0.837)	0.477 (0.697)
Tam-Pop/Sal	-0.391 (0.399)	0.987 (0.605)	-0.083 (0.095)	0.235 (0.414)	-0.043 (0.281)	0.328 (0.404)
Restored *Tam-Pop/Sal*	-0.014 (0.337)	-0.155 (0.176)	2.060 (2.242)	-0.003 (0.216)	-0.653 (0.556)	-0.317 (0.268)
One-Way ANOVA	**F = 7.368** **P = 0.002**	**F = 4.056** **P = 0.024**	**F = 5.187** **P = 0.010**	F = 0.141 P = 0.934	F = 0.483 P = 0.699	F = 1.292 P = 0.309

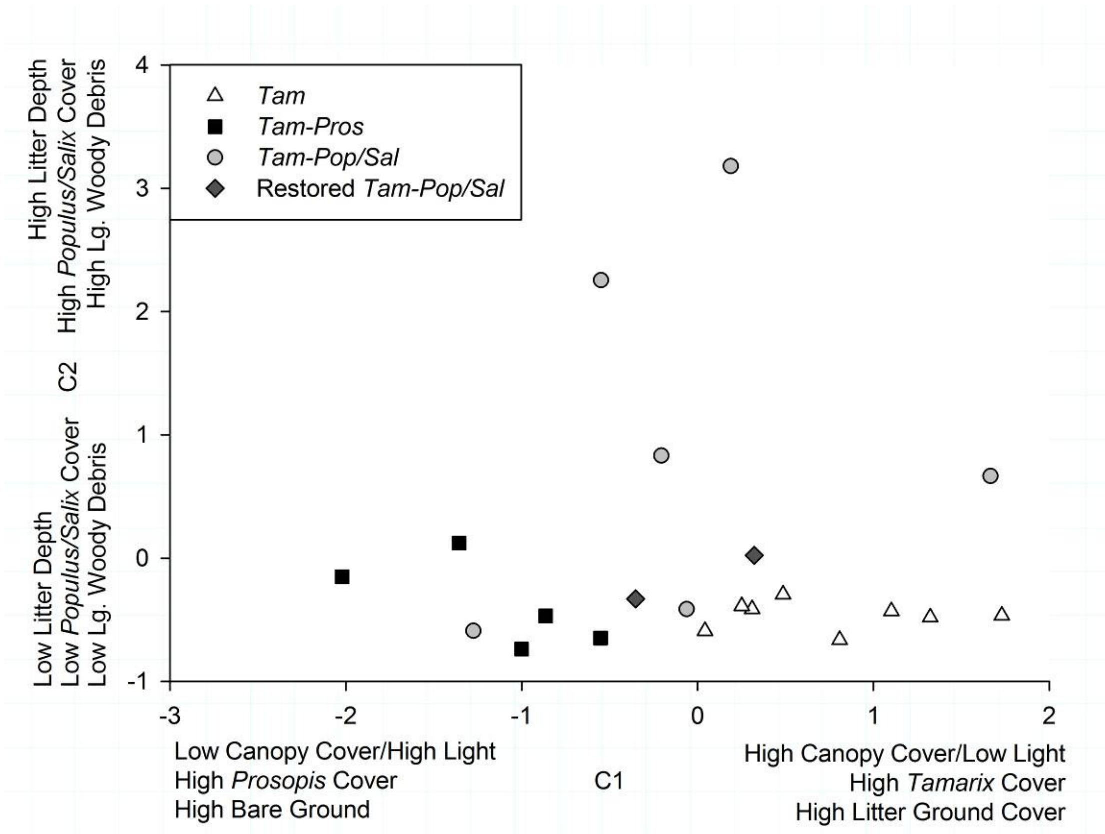

Figure 7. Habitat physiognomy values (C1 and C2) derived from PCA at trap arrays in *Tam, Tam-Pros, Tam-Pop/Sal,* and Restored *Tam-Pop/Sal* sites along the Virgin River in Utah, Nevada, and Arizona.

Habitat structure and physiognomy also varied between the Restored and Unrestored *Tam-Pop/Sal* site during VES (Table 7). Canopy cover was significantly lower at the Restored *Tam-Pop/Sal* site. Although not significant, the Restored *Tam-Pop/Sal* site had lower saltcedar cover and a higher proportion of bare ground and woody debris than the Unrestored *Tam-Pop/Sal* site (Table 7). Photographs taken at the restored site showed that there was a large increase in the abundance of secondary species, particularly kochia (*Bassia scoparia*), following restoration activities (Appendix E).

Table 7. Mean (±SE) of habitat variables measured along transects at the Restored and Unrestored *Tam-Pop/Sal* site in St. George, Utah. To determine if there was a difference in habitat variables between sites, an independent *t*-test was conducted. Significant (P<0.05) terms in bold.

Variables	Unrestored *Tam-Pop/Sal* n = 3	Restored *Tam-Pop/Sal* n = 3	*T (P)*
Bare Ground (%)	37.5 (2.5)	53.3 (7.3)	2.06 (0.108)
Woody Debris Ground Cover (%)	5.8 (2.2)	7.5 (2.9)	0.46 (0.670)
Litter Ground Cover (%)	56.7 (4.4)	39.2 (6.7)	-2.19 (0.094)
Litter Depth (cm)	0.7 (0.1)	1.1 (0.2)	1.69 (0.116)
Proportion of *Tamarix*	0.9 (0.1)	0.6 (0.1)	-1.96 (0.121)
Proportion of *Populus/Salix*	0.1 (0.1)	0.2 (0.1)	0.76 (0.492)
Proportion of *Elaeangus*	0.1 (0.1)	0.1 (0.1)	-0.59 (0.590)
Proportion of *Fraxinus*	0.0 (0.0)	0.03 (0.03)	1.00 (0.374)
Proportion of *Baccharis*	0.1 (0.0)	0.0 (0.0)	-1.96 (0.121)
Proportion of *Kochia*	0.2 (0.2)	0.4 (0.3)	0.82 (0.457)
Number of dead branches, sm. diam. (1.0-2.5 cm)	27.5 (3.8)	34.5 (1.9)	1.67 (0.171)
Number of dead branches, lg. diam. (>2.5 cm)	3.3 (1.2)	9.8 (3.8)	1.65 (0.174)
Canopy cover (%)	81.8 (3.9) n = 6	57.7 (8.9) n = 6	**-2.51 (0.031)**
Visible light (%)	27.3 (4.8) n = 6	43.7 (8.6) n = 6	1.66 (0.127)

Herpetofauna and Habitat Relationships

The four most common species of herpetofauna (3 lizards, 1 amphibian) were included in habitat analyses, as rare species were too few to examine herpetofauna/habitat relationships. Yellow-backed spiny lizards had the only conclusive habitat models (Table 8). Yellow-backed spiny lizards were found to be positively associated with areas having high densities of cottonwood/willow and large diameter mesquite and negatively associated with areas having high densities of small diameter saltcedar (C3 and C5; Table 8). Additional models and variable weights indicated that yellow-backed spiny lizards might also be positively associated with high cottonwood/willow cover, large woody debris, and deep litter (C2; Table 8). Although not conclusive, common side-blotched lizards appear to be positively associated with areas having high densities of large diameter saltcedar and negatively associated with areas having high small diameter woody debris cover and high densities of arrowweed (C4 and C6; Table 8).

Table 8. Abundance of the four most common species of herpetofauna (1 toad, 3 lizards) as predicted by habitat characteristics (components) from ranked multiple-linear regression models using a multiple-model inference approach. Species abundances were log(x+1) transformed to increase data normality. Components with high variable weights in bold.

Species	Top Model (+/-)	Variable AICc Weights					
		C1	C2	C3	C4	C5	C6
Anaxyrus woodhousii	- C4 (Sm. Woody Debris)	0.30	0.30	0.32	0.33	0.31	0.31
Aspidoscelis tigris	- C1 (Overstory, *Pros/Tam* Cover)	0.31	0.30	0.30	0.30	0.31	0.30
Sceloporus uniformis	- C5 (Sm. *Tam/* Lg. *Pros* Density)	0.33	**0.43**	**0.51**	0.26	**0.52**	0.28
Uta stansburiana	- C6 (*Pluchea*/Lg. *Tam* Density)	0.32	0.29	0.32	**0.36**	0.31	**0.38**

Discussion

Along the Virgin River, many species of herpetofauna utilized *Tam*, *Tam-Pros*, *Tam-Pop/Sal*, and Restored *Tam-Pop/Sal* habitats. Results from trapping indicated that herpetofauna communities were dominated by generalist species (i.e., tiger whiptails, common side-blotched lizards, and Woodhouse's toads) with abundances greatest at Restored *Tam-Pop/Sal* sites. Although Restored *Tam-Pop/Sal* sites were limited in number and restricted to St. George, Utah, the space-for-time substitution also indicated that reptile and amphibian abundances were greatest at Restored *Tam-Pop/Sal* sites compared to an adjacent Unrestored *Tam-Pop/Sal* site. Restored *Tam-Pop/Sal* sites likely supported higher abundances of herpetofauna, as these riparian areas exhibited greater habitat heterogeneity. Restoration activities created a more mosaic habitat by reducing canopy cover and increasing native tree density and surface water availability.

Habitat Characteristics and Yearly Differences

Habitat significantly differed among stand types along the Virgin River. Although multiple factors likely contributed to habitat conditions at sites, tree composition was important in predicting habitat structure. For example, *Tam* sites had higher canopy cover and lower solar radiation than *Tam-Pros*, *Tam-Pop/Sal*, and Restored *Tam-Pop/Sal* due to high saltcedar cover, whereas *Tam-Pop/Sal* sites had higher litter depth and large woody debris due to high cottonwood/willow cover. Restored *Tam-Pop/Sal* sites were most similar to *Tam-Pop/Sal* sites; however, restoration efforts increased variation in overstory cover and tree species composition. There were minimal differences in habitat metrics within stand types between 2013 and 2014; however, Restored *Tam-Pop/Sal* sites will likely change over time as transplanted willow and mesquite stems establish and mature.

In the short term, Restored *Tam-Pop/Sal* sites have exhibited observable changes in non-woody vegetation composition since restoration efforts were first implemented in 2012. Site photos indicate that non-woody plant density, particularly kochia, has greatly increased following restoration efforts (Appendix E). This response is likely due to increased soil disturbance and solar radiation from mechanical removal of saltcedar and Russian olive, as kochia is a common successional plant in disturbed areas (DiTomaso et al. 2013). Kochia has been found to limit growth of certain crop species (Wicks et al. 1997; Casey 2009), but no information is available on the effects of kochia on young riparian tree species. Active management of these sites may be necessary if kochia's chemical and physical properties are limiting growth and establishment of transplanted willow and mesquite stems.

Herpetofauna and Habitat Relationships

Although herpetofauna diversity and richness did not significantly differ in riparian habitats along the Virgin River, some reptile and amphibian species were found to prefer certain stand types. Yellow-backed spiny lizards were most abundant at Restored *Tam-Pop/Sal* sites, followed by *Tam-Pop/Sal* and *Tam-Pros* sites, and did not occur at *Tam* sites. Habitat analyses indicated that these lizards were positively associated with areas having high densities of cottonwood/willow and large diameter mesquite trees, as well as large diameter woody debris and deep litter (Table 8; Bateman & Ostoja 2012). The similar desert spiny lizard (*Sceloporus magister*) has been found to forage primarily on large diameter trees within riparian areas of central Arizona (Vitt et al. 1981). Since saltcedar has a shrublike growth form and does not typically reach large diameters like cottonwood, willow, and mesquite, it may provide less suitable foraging habitat for semi-arboreal lizards like desert spiny lizards and yellow-backed spiny lizards (Bateman & Ostoja 2012). This difference may explain why yellow-backed spiny lizards were not captured at *Tam* sites.

Woodhouse's toad abundance was greatest at *Tam-Pop/Sal* and Restored *Tam-Pop/Sal* sites; however, habitat analyses did not find strong relationships between toad abundances and habitat metrics. During my study, amphibian abundance was dependent upon water availability and was typically highest immediately following monsoon rains. Along the Middle Rio Grande, New Mexico, Bateman et al. (2008b) found that Woodhouse's toad and Great Plains toad (*Anaxyrus cognatus*) abundances increased during flooding as high flow events created temporary pools. As such, Woodhouse's toad abundance was likely higher at *Tam-Pop/Sal* and Restored *Tam-Pop/Sal* due to greater water availability and moister soils at these sites. In particular, Restored *Tam-Pop/Sal* sites had permanent water due to water enhancement projects during restoration. Pacific tree frogs were only found at upstream sites (Utah) as the species was recently reintroduced to St. George, Utah and has likely not established in the lower Virgin River Basin (Utah Division of Wildlife Resources 2009).

Overall, Restored *Tam-Pop/Sal* sites likely supported higher abundances of herpetofauna as habitat heterogeneity within the area increased after restoration efforts. Habitat comparisons indicated that Restored *Tam-Pop/Sal* sites had intermediate-levels of vegetation structure and physiognomy when compared to *Tam, Tam-Pros, and Tam-Pop/Sal* sites. It is likely that restoration efforts created a more open and mosaic habitat that is able to support a wider range of herpetofauna species. These findings are consistent with Bateman et al. (2008), who found that lizard abundance was higher following non-native tree removal along the Middle Rio Grande, New Mexico due to an increase in habitat variation and openness.

Effects of Biological Control on Herpetofauna

Since 2009, total lizard abundance along the Virgin River has steadily declined (Bateman et al. 2014). This trend continued in 2013 and 2014, in which lizard abundance was lowest. Overall, total lizard abundance was about 48% lower in *Tam* sites and 66% lower in mixed sites (*Tam-Pros* and *Tam-Pop/Sal*) compared to pre-biological control conditions (Bateman & Ostoja 2012). Lizard abundance has likely decreased due to defoliation caused by saltcedar biological control as habitats have become hotter and dryer (Bateman et al 2013; Bateman et al. 2014). Study sites have also had about a 10% decrease in canopy cover as well. Surprisingly, common side-blotched lizards have seen about a 50% decrease in abundance along the Virgin River, despite the species' affinity for open areas comprised of low canopy and shrub cover (Nielson & Bateman 2013; Bateman & Ostoja 2012). Since my habitat analyses for common side-blotched did not find strong associations with habitat openness and tree cover, it is likely that changes in microclimate caused by defoliation have made habitats less suitable for common side-blotched lizards, either due to decreased thermal variability at study sites or increased temperatures that exceed this species' thermal maximum (Goller et al. 2014).

Management Implications

Biological control can be an effective, although highly variable, method for controlling saltcedar with areas seeing 0 to 80% mortality after multiple defoliation events (Bean et al. 2013). I have observed saltcedar mortality at my study sites along the Virgin River; however, in many cases, most of these trees are still standing. Mechanical removal of saltcedar would decrease saltcedar density, as well as reduce canopy cover in these areas. Decreased saltcedar density and canopy cover in conjunction with high flow events would likely benefit native plant species, such as cottonwoods that depend on high solar radiation and flooding to germinate and grow (Braatne et al. 1996). In addition, saltcedar flammability also increases with density of dead woody fuel (Racher et al. 2001). Although biological control can increase saltcedar fire mortality (Drus et al. 2014), it also increases fire risk and cottonwoods and willows are less successful than saltcedar at tolerating frequent and high intensity fires (Ellis 2001). As such, mechanical removal of saltcedar may prevent further loss of habitat.

Native tree species have also not established or increased in abundance at study sites following biological control. Arrowweed, a native woody shrub, is the main secondary species establishing at study sites, with some sites exhibiting a 2.5-fold increase in arrowweed density. Unfortunately, arrowweed may not provide high quality habitat for wildlife due to its low structural diversity (Ostoja et al. 2014). Additional restoration activities in these areas, such as mechanical removal of saltcedar and transplanting of native woody species, may be necessary to promote growth of desirable plant species and improve habitat quality. Reintroducing native trees would not only improve foraging habitat for yellow-backed spiny lizard, but also increase canopy cover as well. Increased canopy cover would help reduce high site temperatures caused by saltcedar defoliation and would likely increase herpetofauna activity (Bateman et al. 2014).

Conclusions

In the short-term, riparian restoration following saltcedar biological control appears to have improved habitat quality for a variety of herpetofauna species along the Virgin River. Although biological control may decrease lizard abundance by altering riparian microclimate, incorporating restoration activities to reduce saltcedar densities and promote native tree growth can mitigate effects. As indicated by secondary plant growth, sites will likely need to be actively managed to prevent establishment by undesirable plant species that do not provide high quality habitat. Additional research is necessary to understand the long-term effects of riparian restoration on habitat and wildlife communities following saltcedar biological control. Habitat and herpetofauna responses following saltcedar control and restoration efforts will likely vary depending on site differences (i.e., vegetation composition, water availability, etc.; Taylor & McDaniel 2004). Therefore, natural resource managers should consider how saltcedar biological control and restoration efforts alter ecosystem functions and affect wildlife communities prior to management activities. Since saltcedar can dominate areas influenced by livestock grazing (Hughes 1993), future research should also explore how biological control and grazing interact to affect saltcedar.

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APPENDIX A

LOCATIONS OF STUDY SITES

Appendix A. Site name, stand type, and geographic location of trap array and VES sites.

Site	Array	Stand Type	Zone	Datum	Easting	Northing
Seegmiller	1	*Tam-Pop/Sal*	12 S	NAD 83	274625	4107894
Riverside	1	Restored *Tam-Pop/Sal*	12 S	NAD 83	271619	4107481
Riverside	2	Restored *Tam-Pop/Sal*	12 S	NAD 83	271442	4107303
Littlefield	1	*Tam-Pop/Sal*	12 S	NAD 83	239617	4087585
Littlefield	2	*Tam-Pop/Sal*	12 S	NAD 83	239527	4087439
Big Bend	1	*Tam-Pros*	12 S	NAD 83	233971	4081452
Big Bend	2	*Tam-Pros*	12 S	NAD 83	233801	4081292
Mesquite	1	*Tam*	11 S	NAD 83	764162	4077107
Mesquite	2	*Tam*	11 S	NAD 83	763642	4077072
Mesquite	3	*Tam*	11 S	NAD 83	763428	4077062
Mesquite	1	*Tam-Pop/Sal*	11 S	NAD 83	760694	4075899
Mesquite	2	*Tam-Pop/Sal*	11 S	NAD 83	760420	4075755
Bunkerville	1	*Tam-Pros*	11 S	NAD 83	756519	4075058
Bunkerville	2	*Tam*	11 S	NAD 83	756138	4074890
Bunkerville	2	*Tam-Pop/Sal*	11 S	NAD 83	756178	4074562
Toquap	1	*Tam*	11 S	NAD 83	748917	4069716
Toquap	2	*Tam*	11 S	NAD 83	748797	4069854
Gold Butte	1	*Tam*	11 S	NAD 83	742331	4062936
Gold Butte	2	*Tam*	11 S	NAD 83	742103	4062743
Gold Butte	1	*Tam-Pros*	11 S	NAD 83	740744	4061172
Gold Butte	2	*Tam-Pros*	11 S	NAD 83	740638	4060710
Riverside (15 acres)	VES	Restored *Tam-Pop/Sal*	12S	NAD 83	271710	4107537
Seegmiller (12 acres)	VES	*Tam-Pop/Sal*	12S	NAD 83	274660	4107912

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APPENDIX B

DESCRIPTIONS OF HABITAT VARIABLES

Appendix B. Description of habitat variables recorded at trap arrays and VES sites.

Method/Variable	**Description**
Point-intercept	**Recorded at 1m intervals along 20m transect.**
Ground cover	Bare ground: exposed soil Litter: non-woody debris (i.e., leaves) and woody debris (i.e., sticks) less than 1cm in diameter Woody debris: woody debris greater than 1cm in diameter, located 0-0.5m above the ground
Litter depth (cm)	Depth of litter when present (see ground cover type).
Tree and shrub cover	Woody plant cover located directly above point, classified to species.
Line-intercept	**Recorded at alternating 1m intervals along 20m transect**
Woody debris count	Small diameter: number of dead branches 1.0-2.5cm in diameter Large diameter: number of dead branches greater than 2.5cm
Plot Stem Counts	**Stems of woody plants were counted in 2x2m plots located at both ends of transect. Only stems rooted within the plots were counted.**
Shrubs	Number of stems per woody shrub species.
Trees	Number of stems per size class per woody tree species. Size classes were further condensed in PCA based on natural breaks in the data.
Tree size classes *Tamarix* spp. *Prosopis* spp.	 A: less than 1m tall B: greater than 1m tall, less than 2.5cm in diameter C: greater than 1m tall, 2.5-7.5cm in diameter D: greater than 1m tall, greater than 7.5cm in diameter
Populus fremontii *Salix* spp. *Elaeagnus angustifolia*	A: less than 1m tall B: greater than 1m tall, less than 5cm in diameter C: greater than 1m tall, 5-10cm in diameter D: greater than 1m tall, 10-30cm in diameter E: greater than 1m tall, greater than 30cm in diameter

Various	**Percent canopy cover and visible light were recorded in 2x2m plots (trap arrays) and at 0, 10, and 20m along 20m transect (VES).**
Canopy Cover (%)	Percent cover determined using a spherical densiometer. Readings were taken in the four cardinal directions from the center of each plot/point and averaged.
Visible Light (%)	Percent visible light detected using LI-COR LI-250A Light Meter. Readings were taken in the four cardinal directions from the center of each plot/point, averaged, and divided by a control reading (measured in area with no canopy cover).

APPENDIX C

HERPETOFAUNA SPECIES LIST

Appendix C. List of herpetofauna species captured or observed at trap arrays and visual encounter survey sites. Total captures represents the total number of captures (individuals may have been captured multiple times) recorded for each species at trap arrays during 2013 and 2014.

Lizard		Total Captures
Common side-blotched lizard	*Uta stansburiana*	212
Long-tailed brush lizard	*Urosaurus graciosus*	13
Ornate tree lizard	*Urosaurus ornatus*	6
Plateau striped whiptail	*Aspidoscelis velox*	2
Tiger whiptail	*Aspidoscelis tigris*	728
Western banded gecko	*Coleonyx variegatus*	3
Yellow-backed spiny lizard	*Sceloporus uniformis*	120
Zebra-tailed lizard	*Callisaurus draconoides*	1
Snake		
California kingsnake	*Lampropeltis getula californiae*	4
Coachwhip	*Coluber flagellum*	2
Western threadsnake	*Rena humilis*	2
Frog/Toad		
American bullfrog	*Lithobates catesbeiana*	-
Arizona toad	*Anaxyrus microscaphus*	1
Pacific tree frog	*Pseudacris regilla*	38
Woodhouse's toad	*Anaxyrus woodhousii*	210

APPENDIX D

PCA COMPONENT MATRIX

Appendix D. Variation among habitat metrics at trapping sites was summarized using a principal component analysis (PCA) with Varimax rotation. The number of relevant components was determined based on a scree plot, components with eigenvalues greater than one, and parsimony (Legendre and Legendre 1998). Habitat variables with low counts across sites (i.e., Proportion/Density of *Elaeangus*) were not included in the PCA. Although Component Seven had an eigenvalue greater than one, it consisted of only one variable and explained <8% of the variation. Thus, it was not included in habitat comparisons.

Variable	Component						
	1	2	3	4	5	6	7
Canopy Cover (%)	**0.822**	0.229	0.064	0.061	-0.133	-0.310	-0.018
Prosopis Cover (%)	**-0.803**	-0.056	-0.134	-0.054	-0.072	0.145	-0.161
Visible Light (%)	**-0.778**	-0.180	-0.002	0.089	0.205	0.228	-0.289
Bare Ground (%)	**-0.746**	-0.268	-0.084	0.090	-0.358	-0.165	0.385
Tamarix Cover (%)	**0.691**	-0.457	-0.379	0.056	0.268	-0.107	0.040
Litter Ground Cover (%)	**0.684**	0.135	0.121	-0.457	0.377	-0.004	-0.313
Litter Depth (cm)	0.067	**0.895**	0.001	0.140	0.094	-0.108	0.025
Populus/Salix Cover (%)	0.215	**0.835**	0.210	0.117	0.001	0.207	-0.083
Number of Dead Branches, Lg. Diam. (>2.5cm)/10m	0.088	**0.818**	-0.107	0.282	-0.046	-0.191	-0.055
Density of *Populus/Salix* stems, Sm. Diam. (≤10cm)/10m^2	0.056	0.052	**0.991**	0.011	0.011	-0.005	-0.036
Density of *Populus/Salix* stems, Lg. Diam. (>10cm)/10m^2	0.072	0.006	**0.986**	-0.051	-0.016	-0.013	-0.030
Number of Dead Branches, Sm. Diam. (1.0-2.5cm)/10m	-0.079	0.245	0.039	**0.866**	0.143	-0.143	-0.069
Woody Ground Cover (%)	0.066	0.264	-0.087	**0.803**	-0.075	0.349	-0.122
Density of *Prosopis* stems, Lg. Diam. (>7.5cm)/10m^2	-0.209	-0.201	-0.068	-0.013	**-0.849**	0.171	-0.007
Density of *Tamarix* stems, Sm. Diam. (≤2.5cm)/10m^2	-0.227	-0.440	-0.191	0.100	**0.708**	0.071	0.249
Density of *Pluchea* stems/10m^2	-0.359	-0.232	-0.134	0.125	-0.113	**0.827**	-0.104
Density of *Tamarix* stems, Lg. Diam. (>2.5cm)/10m^2	0.469	-0.237	-0.371	0.043	0.097	**-0.622**	-0.250
Density of *Prosopis* stems, Sm. Diam. (≤7.5cm)/10m^2	0.119	-0.066	-0.045	-0.131	0.118	0.005	**0.911**
Variance Explained (%)	22.018	16.879	13.362	9.938	9.459	8.589	7.531
Cumulative Variance Explained (%)	22.018	38.896	52.258	62.196	71.654	80.243	87.774

APPENDIX E

PHOTOGRAPHS OF RESTORED *TAM-POP/SAL* SITE

Appendix E1. Photo plot at Restored *Tam-Pop/Sal* site showing pre-restoration conditions (Winter 2012; top), immediately following restoration (Spring 2012; center), and roughly two years after restoration (Summer 2014; bottom). Top two photos provided by Christian Edwards (Utah Division of Wildlife Resources).

Appendix E2. Photo of trap array at Restored *Tam-Pop/Sal* site one year post-restoration (March 2013; top) and two years post-restoration (June 2014; bottom). In bottom photo, kochia was cleared away from drift fence before picture was taken.