Herpetofauna and Riparian Microhabitat of

Urban and Wildland Reaches Along the Salt River, Arizona

by

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ABSTRACT

Worldwide, riverine floodplains are among the most endangered landscapes. In response to anthropogenic impacts, riverine restoration projects are considerably increasing. However, there is a paucity of information on how riparian rehabilitation activities impact non-avian wildlife communities.

I evaluated herpetofauna abundance, species richness, diversity (i.e., Shannon and Simpson indices), species-specific responses, and riparian microhabitat characteristics along three reaches (i.e., wildland, urban rehabilitated, and urban disturbed) of the Salt River, Arizona. The surrounding uplands of the two urbanized reaches were dominated by the built environment (i.e., Phoenix metropolitan area). I predicted that greater diversity of microhabitat and lower urbanization would promote herpetofauna abundance, richness, and diversity. In 2010, at each reach, I performed herpetofauna visual surveys five times along eight transects (n=24) spanning the riparian zone. I quantified twenty one microhabitat characteristics such as ground substrate, vegetative cover, woody debris, tree stem density, and plant species richness along each transect.

Herpetofauna species richness was the greatest along the wildland reach, and the lowest along the urban disturbed reach. The wildland reach had the greatest diversity indices, and diversity indices of the two urban reaches were similar. Abundance of herpetofauna was approximately six times lower along the urban disturbed reach compared to the two other reaches, which had similar abundances. Principal Component Analysis (PCA) reduced microhabitat variables to five factors, and significant differences among reaches were detected. Vegetation structure complexity, vegetation species richness, as well as densities of *Prosopis* (mesquite), *Salix* (willow), *Populus* (cottonwood), and animal burrows had a positive correlation with at least one of the three

herpetofauna community parameter quantified (i.e., herpetofauna abundance, species richness, and diversity indices), and had a positive correlation with at least one herpetofauna species.

Overall, rehabilitation activities positively influenced herpetofauna abundance and species richness, whereas urbanization negatively influenced herpetofauna diversity indices. Based on herpetofauna/microhabitat correlations established, I developed recommendations regarding microhabitat features that should be created in order to promote herpetofauna when rehabilitating degraded riparian systems. Recommendations are to plant vegetation of different growth habit, provide woody debris, plant *Populus*, *Salix*, and *Prosopis* of various ages and sizes, and to promote small mammal abundance.

I would like to dedicate my thesis to my dear husband.

Without you, this would not have been feasible, thank you.

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INTRODUCTION

Worldwide, riverine floodplains are one of the most endangered landscapes (Tockner and Stanford 2002). For instance, in the United States, 70% of the original riparian forests of the lower 48 states have been converted to other land uses (Turner et al. 1998). Throughout history, floodplains have been a focal point for anthropogenic development. Indeed, density of people decreases with distance from major rivers (Small and Cohen 2004). Developments of great civilizations were permitted by the annual overflow of rivers, which renewed the fertility of soils (Sparks 1995), and allowed large gathering of humans to establish into sedentary settlements, establishing the first "urban societies" (Redman 1999). Metropolitan areas are continually growing. Although only 10% of the global population occurred in urban areas in 1900 (Grimm et al. 2008), in 2008, the global urban population equaled the rural population, and has now exceeded it (United Nations 2008). Unfortunately, metropolitan areas can impact riparian ecosystems through fragmentation (Dynesius and Nilsson 1994; Andersson et al. 2000), altering hydrologic regimes (Larson et al. 2005), and by introducing non-native species (Nilsson and Berggren 2000).

Within the last 200 years, most major river systems have been regulated for human needs (Benke 1990; Dynesius and Nilsson 1994; Tockner and Stanford 2002). Rivers have been modified to facilitate navigation, divert water for municipal and agricultural needs, reduce flooding, impound water, and to produce hydro-electricity (Tockner and Stanford 2002). Impacts may result from these modifications. For instance, shoreline length may decrease in channelized rivers (Tockner and Stanford 2002) and affect habitat availability for endangered communities (Naiman and Decamps 1997), exchange of nutrients and organisms may be inhibited and thus decrease biological productivity (Bayley 1995; Naiman and Decamps 1997), and hydrological

regimes may be significantly altered (Tockner and Stanford 2002). The modified flow regime of regulated rivers often results in fewer flood events (Junk et al. 1989) and can negatively alter adjacent riparian ecosystems (Gore and Shields 1995; Nilsson and Berggren 2000). Indeed, changes in hydrological regimes may cause shifts from native dominated to non-native dominated riparian vegetation (Stromberg et al. 2007; Merritt and Poff 2010).

In response to anthropogenic impacts on those important ecosystems, riverine restoration projects considerably increased over the past several decades (Bernhardt et al. 2005; Follstad Shah et al. 2007). Naiman et al. (1993) suggest that effective riparian corridor management could improve many ecological issues related to land use and environmental quality. Riparian areas should be viewed as a key global resource (Tockner and Stanford 2002) as they are a key landscape feature having substantial regulatory controls on environmental vitality (Naiman et al. 1992). They cover only 1.4% of the global land surface and yet riparian areas and associated floodplains contribute to more than 25% of all terrestrial ecosystem services including water supply, disturbance regulation, waste treatment, and nitrogen removal (Tockner and Stanford 2002). In addition to being highly productive (Tockner and Stanford 2002), riparian corridors are among the most diverse terrestrial habitats on Earth (Naiman et al. 1993), and typically support a significantly different species pool than adjacent upland habitat (Sabo et al. 2005). The mosaic structure, presence of refugia, species assemblages, and broad ranges of environmental settings contribute to the high diversity of riparian areas (Naiman et al. 2005). In the western United States, where less than 1% of the landscape remains covered by riparian vegetation (Knopf et al. 1988; Knopf and Samson 1994), riparian areas provide habitat for more species of breeding birds than any other type of vegetation association (Knopf and Samson 1994). Some sources (Sparks 1995; Naiman

and Decamps 1997) suggest that riparian areas provide critical migration corridors for a variety of wildlife and plant species, and can provide important corridors for wildlife in urban settings (Savard et al. 2000). Modifying riparian areas through restoration or rehabilitation can influence the availability of habitats and microhabitats used by wildlife. However, there is little published information on how these riparian rehabilitation activities impact non-avian wildlife communities (Bateman and Paxton 2010).

Amphibians and reptiles (collectively referred to as herpetofauna) are important vertebrates to monitor when habitats are being altered and rehabilitated because they respond to structural changes in habitat (Pianka 1967; Jones 1981), they can be used as indicators to understand the effects of modifying riparian habitats (Bateman et al. 2008), and are good indicators of riparian ecosystem structure and function (Lowe 1989; Welsh and Hodgson 1997). Furthermore, alterations to habitat may affect herpetofauna to a greater extent than other wildlife such as birds and mammals, as they are less mobile and have smaller home ranges (Burton and Likens 1975a). Within riparian ecosystems, there is a paucity of data documenting the relationships between herpetofauna communities and microhabitat and habitat structure (Jones and Glinski 1985; Szaro et al.1985; Szaro and Belfit 1986). There is also a lack of information regarding the ecology of most herpetofauna (Rosen 2009), herpetofauna responses to habitat degradation (Gibbons et al. 2000), and herpetofauna responses to land use disturbance such as land development (Barrett and Guyer 2008). In fact, herpetofauna responses to urban development are perhaps the least understood of all vertebrate classes (Germaine and Wakeling 2001).

Herpetofauna species occurrence increases with habitat heterogeneity (Burbrink et al. 1998). Indeed, the presence of a range of habitat type benefits herpetofauna species richness by insuring the presence of all habitats necessary to satisfy all life-cycle requirements (Burbrink et al. 1998). Herpetofauna abundance and diversity also vary

with structural diversity, as spatial heterogeneity allows the coexistence of several species (Pianka 1966; Jakle and Gatz 1985). Microhabitat partitioning also plays an important role in the structuring of herpetofauna communities (Barbault and Maury 1981; Vitt et al. 1981; Jones 1988b; Amo et al. 2007).

In urban landscapes, many native bird species have reduced survival and reproduction rates near homes (Hansen et al. 2005), and overall native wildlife species richness tend to decrease with increasing urbanization (McKinney 2008). Urbanization threatens over one-third of the world's amphibian species (Hamer and McDonnell 2008), and typically decreases herpetofauna species richness (McKinney 2008), diversity (Pillsbury and Miller 2008), and abundance (Germaine and Wakeling 2001; Pillsbury and Miller 2008). The decline and extirpation of herpetofauna in urban environments are directly related to habitat loss, fragmentation, isolation, and overall degradation of habitat (Mitchell and Brown 2008). Introduced invasive species, environmental pollution, disease and parasitism, unsustainable use, and global climate change also contribute to the global decline of herpetofauna (Gibbons et al. 2000).

Riparian communities can support extensive herpetofauna abundance and diversity (Brode and Bury 1984; Lowe 1989). For example, approximately 60% of the herpetofauna species present in the Mojave, Sonoran, and Chihuahuan deserts are found in riparian habitats (Lowe 1989). In lowland riparian communities of southeastern Arizona, several species which are typically not considered riparian obligates can be locally restricted to the riparian environment (Jones and Glinski 1985; Jones et al. 1985; Rosen et al. 2005). In arid regions of low elevation, some herpetofauna species rely on the dense vegetation, high humidity, soil moisture, and perennial water found only along the riparian corridor (Rosen 2009). Riparian zones also provide dispersal corridors and habitat islands important to many herpetofauna species, especially in arid climates (Brode

and Bury 1984). Thus, the rehabilitation of riparian ecosystems could be beneficial for herpetofauna, particularly in metropolitan areas of arid regions.

Herpetofauna can be major contributors to biological complexity in terms of species diversity, trophic dynamics, and species interactions within communities (Gibbons 1988). They may also contribute to energy flow and nutrient cycling (Burton and Likens 1975a), necessary functions of ecological systems (Gurevitch et al. 2002). Herpetofauna are an important component to ecosystems food webs and energy flow, as they efficiently transfer energy within an ecosystem (Pough et al. 1998). As ectotherms, they efficiently transform the energy they intake as food and convert it into body tissue or biomass (Pough et al. 1998), creating an abundant energy source for higher trophic levels. Indeed, a study in New Hampshire found that the biomass of salamanders exceeded the biomass of birds during the peak of the breeding season (Burton and Likens 1975a). Furthermore, when compared to birds or mice and shrews, the salamander population provided a large source of higher quality prey (high protein content) for tertiary consumers, (Burton and Likens 1975b). Therefore, promoting herpetofauna abundance when rehabilitating degraded riparian systems would help to reestablish ecosystem functions.

Enhancing diversity is also important when rehabilitating degraded systems as diverse ecosystems are typically more stable, more resistant, and/or more resilient to disturbance (Tilman and Downing 1994). However, refer to Tilman and Downing (1994) for a discussion on species diversity and functional diversity. Menninger and Palmer (2006) state that "insurance" for communities in terms of their function may be provided by high species diversity; with high species diversity, there is a greater chance that some species may be able to compensate for the loss of another by, for instance, sharing the

same ecological function. Thus, promoting herpetofauna diversity would contribute to the insurance of proper ecosystems functioning.

Despite the importance of herpetofauna in ecosystems, wildlife management plans in general are developed with little concern for reptiles and amphibians (Gibbons 1988; Scott and Seigel 2001; Chung-MacCoubrey and Bateman 2006), perhaps because of their limited economic value (Gibbons 1988). Considering the need for information, and since herpetofauna have a valuable ecological role in riparian systems, it would be beneficial to investigate relationships between riparian microhabitat features and herpetofauna abundance and diversity. Such investigations could provide insights on microhabitat promoting herpetofauna abundance and diversity. With this information, I could recommend to restoration practitioners which microhabitat features should be created when rehabilitating degraded riparian systems in order to promote a diverse and abundant herpetofauna assemblage. Furthermore, understanding herpetofauna and microhabitat relationships could aid natural resource managers to make decisions regarding the improvement, and rehabilitation of ecosystems.

I investigated how riparian microhabitat characteristics and herpetofauna abundance, species richness, and diversity indices differed among reaches of a regulated stream in central Arizona. My research objectives were (1) to compare herpetofauna abundance, species richness, and diversity indices, (2) to compare riparian microhabitat characteristics, and (3) to investigate relationships between microhabitat characteristics and herpetofauna abundance, diversity indices, and species richness, among three reaches which differ in terms of urbanization and rehabilitation effort. A reach was considered urbanized if the surrounding uplands were dominated by the built environment. I predicted that reaches with higher diversity of microhabitat and lower level of urbanization would have greater herpetofauna abundance, richness, and diversity indices.

METHODS

Study Site

The Salt River located in Arizona represents a model system to address my research objectives. The Salt River system has been highly altered by anthropogenic activities; at the beginning of the twentieth century several dams and water diversion projects were constructed and significantly modified this once large perennial river (White and Stromberg 2009). Other anthropogenic activities such as sand and gravel mining, channelization for flood control, development of landfills, grazing, off road vehicles, and urbanization further modified the Salt River riparian community. Efforts to reverse these anthropogenic impacts have resulted in some restoration and/or rehabilitation activities along the Salt River.

I selected three reaches which vary in degree of urbanization and extent of rehabilitation efforts (Appendix A). I selected one reach on Tonto National Forest, upstream from the Salt-Verde confluence (lat 33°33'28 N, long 111°36'33 W). This reach is located approximately five miles from the closest city boundary (hereafter, wildland reach; Appendix B, Photograph I). In the past, the riparian vegetation of the area was severely disturbed. To facilitate natural plant recruitment within the riparian community, the U.S. Forest Service (USFS) closed the area in the late 1970s to authorized grazing, and to off road vehicles in the early 1990s (E. Alford 2010, Arizona State University, Mesa, AZ, personal communication). Since the closures, riparian vegetation such as *Populus fremontii* (Fremont cottonwood) and *Salix gooddingii* (Goodding's willow) forest associations, as well as *Prosopis* spp. (Mesquite species) woodlands have reestablished.

I selected two urbanized reaches which flow through the greater Phoenix metropolitan area. One urban reach is located in Phoenix, between 7th Avenue and 7th

Street (lat 33°25’19 N, long 112°04’23 W), and has been rehabilitated within the last 10 years by the Rio Salado Habitat Restoration Project (Rio Salado). Prior to rehabilitation, the area was a dumping site (hereafter, urban rehabilitated reach; Appendix B, Photograph II). As part of Rio Salado, managers removed large amounts of garbage including tons of buried tires, created a 200-foot wide, 15-foot deep low flow channel, and built terraces. Managers also planted southwestern riparian vegetation communities such as *P. fremontii* and *S. gooddingii* forest associations, as well as *Prosopis* spp. woodlands.

The second urban reach is located between State Route 143 and Priest Drive in Tempe, east of the Phoenix Sky Harbor International Airport (lat 33°26’08 N, long 111°58’10 W). This reach has lost most of the riparian vegetation, most likely due to the historic lack of water caused by water diversion. Furthermore, in order to decrease the chances of bird strikes due to the proximity of the airport, streamside vegetation is periodically removed along this reach, which may have further contributed to the lost of the riparian vegetation. Nowadays, regulated water perennially flows through this reach and it represents a highly disturbed urban floodplain (hereafter, urban disturbed reach; Appendix B, Photograph III).

The three reaches are similar in terms of elevation, temperature, and precipitation. Within the last century, the average mean for the months of June, July, and August for the wildland, urban disturbed, and urban rehabilitated reaches are respectively: temperature average 31°C, 31°C, and 30°C; precipitation average 20.6 mm, 18.3 mm, and 17 mm; and elevation average 412 m, 345 m, 324 m.

Sampling Design

I randomly established eight transects at each reach (Appendix C), and had a sample size of 24. Transects started at the water's edge, ran perpendicular to the stream, and spanned one side of the riparian zone (Fig. 1). I spaced transects at least 150 m apart to ensure independence. The length of each transect varied depending on the width of the riparian zone. To keep sampling effort consistent, I established three 10-m wide by 20-m long segments along each transect. Segments were established in a stratified random fashion; where possible, I established one segment on the cobble bar, one segment in the *P. fremontii* and *S. gooddingii* forest association, and one segment in the higher floodplain terraces where *Prosopis* woodland occur.

To quantify herpetofauna occurrence and abundance, I performed visual surveys along transects. This method effectively tracks relative abundance and species richness, and is preferred over visual encountered surveys when used across a gradient such as a riparian zone (Jaeger 1994). Five daytime visual surveys were performed along each transect between June and August 2010. Surveys were conducted in the morning, during times of high diurnal herpetofauna activity, and under similar environmental conditions (i.e., warm, sunny, wind between 0 and 3 on Beaufort scale). To perform the visual surveys, two observers (a field technician and I; "we") walked each segment, side by side, at the same pace. We each surveyed a 5 m width, which is half of the segment. Searches were not constrained by time but we kept sampling intensity constant between surveys. Doan (2003) used a similar technique which was effective to measure abundance and species richness when compared to other sampling methods. We limited visual searches to about 2 m in height above ground, searched debris piles and downed logs, and moved vegetation to flush hidden reptile or amphibian individuals (herpetiles). We recorded species and location of every herpetile observed within each segment. We

identified species based on Brennan and Holycross (2009) and classified them according to Crother (2008). If a herpetile was unidentified at first sight, we tracked and surrounded the individual to get a positive identification. Once identified, we resumed the survey.

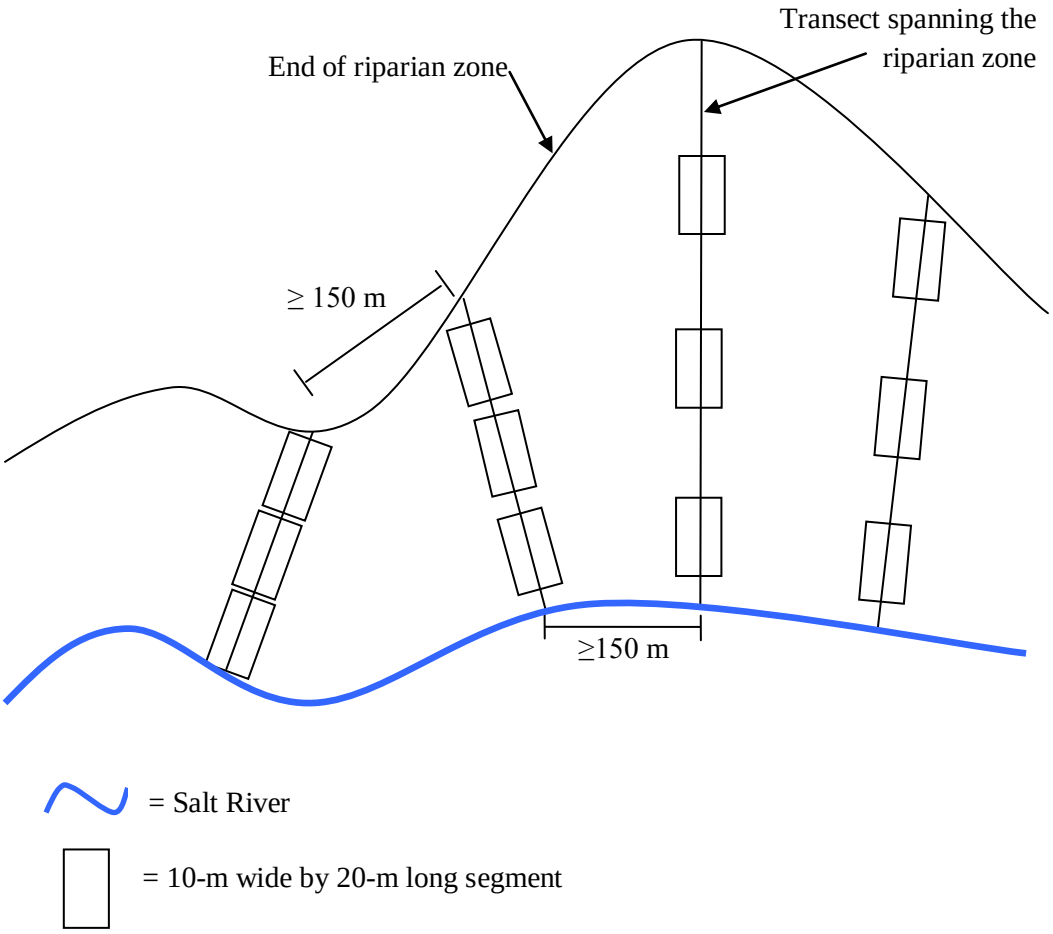

Figure 1. Transects Schematic.
Schematic of transect and segment locations relative to river and riparian zone. Eight transects spaced at least 150 m apart were randomly established at three selected reaches along the Salt River, Arizona. Transects started at the water's edge, ran perpendicular to the stream, and spanned one side of the riparian zone. To keep effort consistent between transects, three 10-m wide by 20-m long segments were established along each transect. Segments were established in a stratified random fashion; when possible, one segment was established on the cobble bar, one segment in the *Populus* and *Salix* forest association, and one segment in the higher floodplain terraces where *Prosopis* woodland occur. Herpetofauna visual surveys and quantification of microhabitat variables were performed within each segment (total of 72 segments).

I quantified 21 microhabitat characteristics along each segment using various methods (Appendix D and E). I used a line-intercept method to measure the areas lacking vegetation, percent cover of an open understory (i.e., areas lacking vegetation up to 1 m in height), percent cover of debris pile, and percent cover of vegetation per growth habit type such as tree, tree shrub, shrub, or subshrub. Growth habit per species (Appendix F) was determined by the United States Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS), Plants Database website (USDA and NRCS 2011). I used a point-intercept method to estimate the proportion of ground cover types such as bare ground, woody ground, and litter ground. I also recorded the depth of litter and woody debris when observed at a point. I counted the number of woody debris along a pre-determined line. I used a convex spherical densiometer to estimate overstory percent cover and I used Daubenmire plots (Daubenmire 1959) to estimate percent cover of herbaceous vegetation. I recorded the stem density of *P. fremontii*, *S. gooddingii*, and *Prosopis* spp. in a plot nested within the segment and counted the number of vegetation species as well as number of burrows within each segment.

Statistical Analyses

Herpetofauna

Herpetofauna minimum abundance (hereafter referred to as abundance) was defined as the greatest number of individuals of each species for a given transect detected during any one of the five visits. Since I did not individually mark herpetiles, this method conservatively estimated abundance and ensured that no individuals were counted twice. Conversely, number of sightings refers to the total number of herpetiles seen during the five rounds of surveys. I summarized species richness per transect by adding all species observed during the five visits for a given transect.

Due to non-normality of the data, I used a non-parametric test, the Multi-Response Permutation Procedures (MRPP), to detect significant differences among reaches' abundance and species richness. If significant differences were detected, a non-parametric comparison test, the Multiple Comparisons based on Peritz Closure (hereafter referred to as Peritz comparison test), was performed to determine which reaches differed.

To analyze herpetofauna diversity, I used the Species Diversity and Richness 4.1.2 software (Seaby and Henderson 2006). To compare diversity among reaches, I computed Shannon and Simpson diversity indices, which are commonly used. To detect significant differences among reaches' diversity indices, I performed randomization tests which use a resampling method (Solow 1993). I also computed a diversity ordering index, the Renyi index, to determine if the three herpetofauna communities were comparable in terms of diversity. For instance, community A may have a higher Shannon index than community B, whereas community B may have a higher Simpson index than community A. In order to be comparable, the communities' diversity must rank consistently, regardless of the diversity index used (Tothmeresz 1995). I used the Renyi index as it is one of the most useful methods for ordering communities, and it performs well regardless of how many species are present (Tothmeresz 1995). Hereafter, diversity indices will be referred to as diversity.

Microhabitat

I used a Principal Components Analysis (PCA) to reduce the 21 microhabitat variables to fewer components. I applied a varimax rotation to the principal components to improve their interpretation (McGarigal et al. 2000). To determine how many principal components should be retained, I used the latent root criterion approach; I eliminated

components with eigenvalues less than one from further analysis (McGarigal et al. 2000). I interpreted components based on the weight of microhabitat variables; variables with larger weights have a higher importance in describing the component. Due to non-normality of the generated component scores, I used MRPP and Peritz comparison test to detect microhabitat significant differences among reaches.

Relationships

I computed non-parametric Spearman's rank correlation coefficients to identify relationships between microhabitat components and herpetofauna community parameters (i.e., abundance, species richness, and diversity), and herpetofauna species. I used the Shannon diversity index to compute the correlation between microhabitat and diversity.

RESULTS

Herpetofauna

From June to August 2010, a total of 134 sightings were recorded during five rounds of surveys, including 59 sightings at the wildland reach, 67 sightings at the urban rehabilitated reach, and eight sightings at the urban disturbed reach. A minimum total abundance of 84 herpetiles and eight species were observed along the three reaches (Fig. 2 and Appendix G). Herpetofauna abundance differed among reaches (MRPP, $p<0.001$). With a mean abundance of 0.75 herpetiles per transect, the urban disturbed reach had at least six times fewer herpetiles than the two other reaches (Peritz comparison test, $p<0.001$). The mean abundance of 4.5 and 5.25 herpetiles per transect for the urban rehabilitated and wildland reaches respectively, were similar.

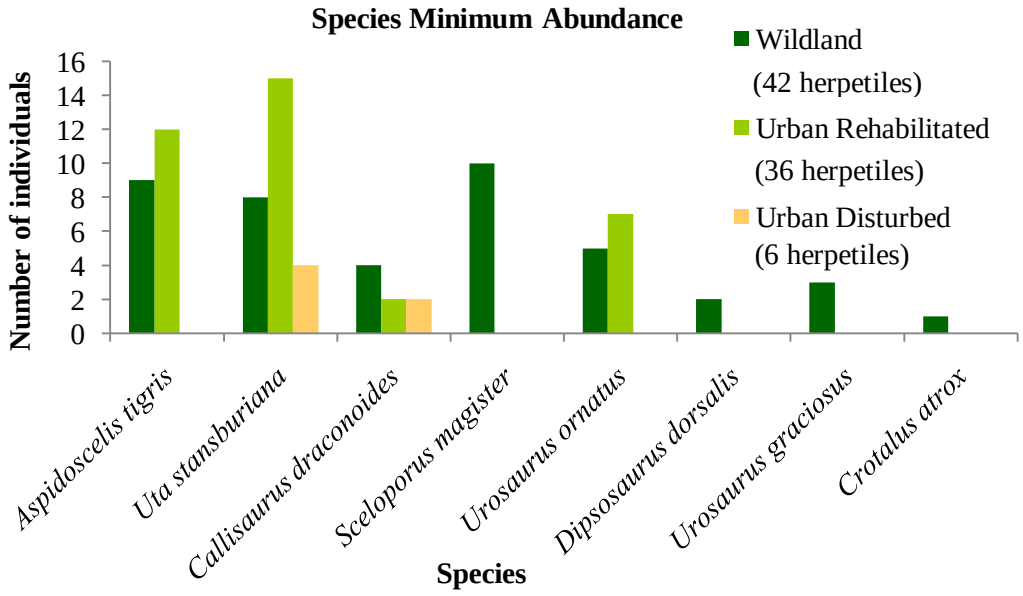

Figure 2. Herpetofauna Abundance per Species per Reach
Minimum number of individuals per species per reach recorded between June and August 2010, during five rounds of visual surveys performed along eight transects established at three reaches along the Salt River, Arizona. Minimum abundance was defined as the greatest number of individuals of each species for a given transect detected during any one of the five rounds of visual survey performed. Total minimum abundance per reach is presented in bracket.

Herpetofauna species richness differed among reaches (MRPP, $p<0.001$), with all reaches being different (Peritz comparison test, $p<0.04$). The wildland reach had the highest species richness with a mean of 3.9 (±0.4 standard error) species per transect, whereas the urban rehabilitated and urban disturbed reaches had a mean of 2.4 (±0.3) and 0.6 (±0.3) species per transect, respectively.

Based on the Renyi index, diversity of the three herpetofauna communities were comparable (Fig. 3). Respectively, for the wildland, urban rehabilitated, and urban disturbed reaches, the Shannon diversity index was 1.887, 1.210, and 0.636, and Simpson diversity index was 6.674, 3.264, and 2.143. Diversity of the wildland reach was significantly greater than diversity of the two urban reaches, which were similar (randomization tests, $p<0.05$).

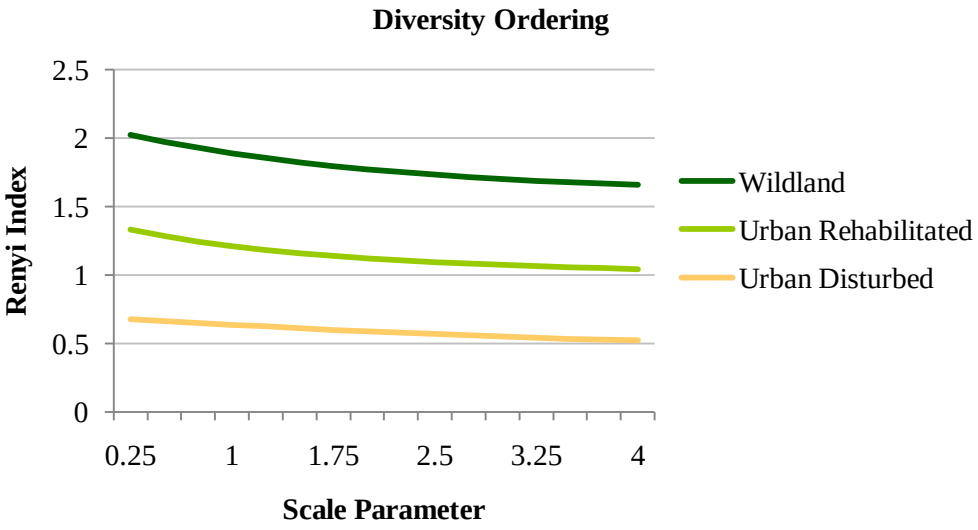

Figure 3. Renyi Index
Diversity ordering index showed that the diversity of three herpetofauna communities located at three reaches selected along the Salt River, Arizona ranked consistently regardless of the diversity index used, thus communities were comparable. The Renyi index was computed with the Species Diversity and Richness 4.1.2 software (Seaby and Henderson 2006) from herpetofauna data collected while performing five rounds of herpetofauna visual surveys between June and August 2010.

Microhabitat

The PCA reduced the 21 microhabitat variables to five components with eigenvalues higher than one. Five components explained 87.3% of the microhabitat variation among reaches. The mean of microhabitat variables (Table 1) influenced the score of each component. I interpreted each component based on microhabitat variables having a high weight for the component (Table 2). I interpreted the components as the following: component 1, complexity of the vegetation structure; component 2, *Populus* and *Salix* occurrence; component 3, density of large *Prosopis* (>7.5 cm diameter) and burrows; component 4, density of small *Prosopis* (<7.5 cm diameter) and vegetation species richness; and component 5, percent cover of subshrub and depth of litter.

There were no significant differences among reaches (MRPP, $p<0.05$) for component 2 (*Populus* and *Salix*; Fig. 4) and component 5 (subshrub and litter depth; Fig. 5). Component 1 (vegetation structure complexity) significantly varied among reaches (MRPP, $p<0.001$), and all reaches differed (Peritz comparison test, $p<0.02$). The wildland reach had the most complex vegetative structure; whereas, the urban disturbed reach had the lowest level of vegetation structure complexity (Fig. 6). Component 3 (large *Prosopis* and burrow) also significantly differed among reaches (MRPP, $p<0.001$). The wildland reach had the highest density of large *Prosopis* and burrows compared to the two urban reaches (Fig. 7), which had similar densities (Peritz comparison test, $p<0.002$). Component 4 (small *Prosopis* and vegetation richness) also significantly varied among reaches (MRPP, $p<0.001$), and all reaches were different (Peritz comparison test $p<0.03$). The urban rehabilitated reach had the highest density of small *Prosopis* and high number of vegetation species; whereas, the urban disturbed reach had the lowest density of small *Prosopis* and low vegetation species richness (Fig. 8).

Table 1. Microhabitat Variables Mean and Standard Error
Mean (±SE) of microhabitat variables used to describe herpetofauna abundance and occurrence. Variables were quantified along eight transects established at three reaches along the Salt River, Arizona (total of 24 transects). Microhabitat variables were quantified within three 10-m wide by 20-m long segments established along each transect (total of 72 segments). Each variable correlation (positive or negative) with one of the five Principal Component Analysis (PCA) components they mainly load on is also presented.

Variables	Wildland	Urban Rehabilitated	Urban Disturbed	Correlation
Debris pile (% cover)	5.0 (1.3)	2.7 (1.1)	-	1 (+)
Woody debris per 10 m	13.9 (1.3)	4.0 (2.5)	0.2 (0.1)	1 (+)
Woody depth (cm)	7.8 (2.5)	2.7 (2.6)	0.1 (0.1)	1 (+)
Tree shrub (% cover)	47.8 (4.9)	16.6 (3.7)	0.1 (0.1)	1 (+)
Lack of vegetation (% cover)	41.7 (5.3)	65.7 (5.1)	89.4 (2.0)	1 (-)
Bare ground (% cover)	46.0 (3.5)	67.9 (3.2)	95.6 (1.3)	1 (-)
Woody ground (% cover)	4.4 (0.7)	1.2 (0.8)	0.2 (0.2)	1 (+)
Litter ground (% cover)	49.6 (3.3)	30.8 (2.8)	4.2 (1.4)	1 (+)
Open understory (% cover)	64.4 (3.8)	72.8 (3.8)	89.3 (2.1)	1 (-)
Canopy cover (%)	44.9 (6.6)	18.3 (5.6)	-	1 (+)
Shrub (% cover)	5.6 (1.6)	3.2 (0.9)	-	1 (+)
Herbaceous (% cover)	22.7 (3.3)	15.6 (1.6)	4.0 (0.9)	1 (+)
P. fremontii and *S. gooddingii* stems/100m², size class D&E	0.5 (0.2)	0.5 (0.3)	-	2 (+)
Tree (% cover)	10.0 (4.1)	6.8 (3.7)	-	2 (+)
P. fremontii and *S. gooddingii* stems/100m², size class A,B&C	0.2 (0.1)	0.8 (0.7)	-	2 (+)
Prosopis spp. stems/100m², size class D&E	1.8 (0.6)	0.2 (0.1)	-	3 (+)
Burrows/100m²	4.8 (1.8)	1.0 (0.3)	0.3 (0.2)	3 (+)
Prosopis spp. stems/100m², size class A,B&C	1.4 (0.6)	3.7 (1.0)	0.04(0.04)	4 (+)
Vegetation species richness	11.9 (0.8)	12.0 (0.6)	2.4 (0.5)	4 (+)
Subshrub (% cover)	3.5 (1.0)	13.4 (3.0)	10.5 (2.0)	5 (+)
Litter depth (cm)	2.4 (0.4)	2.9 (1.3)	1.6 (0.8)	5 (+)

Table 2. Weight of Microhabitat Variables on Components
Rotated Principal Component Analysis (PCA) components matrix depicting the weight of each microhabitat variable quantified along three reaches of the Salt River, Arizona, for each PCA component. Interpretation of components was based on variables having a high weight for that component (bolded values).

	PCA Components				
Microhabitat Characteristics	1	2	3	4	5
Debris pile (% cover)	**0.937**	0.073	-0.115	-0.040	0.163
Woody debris per 10 m	**0.929**	0.065	0.254	-0.038	0.099
Woody depth (cm)	**0.893**	0.005	-0.116	-0.184	0.219
Tree shrub (% cover)	**0.839**	0.275	0.313	0.108	-0.161
Lack of vegetation (% cover)	**-0.832**	-0.422	-0.245	-0.171	-0.004
Bare ground (% cover)	**-0.797**	-0.362	-0.270	-0.305	0.166
Woody ground (% cover)	**0.781**	-0.083	0.289	0.032	0.066
Litter ground (% cover)	**0.767**	0.403	0.257	0.327	-0.188
Open understory (% cover)	**-0.765**	-0.184	-0.248	-0.362	-0.276
Canopy cover (%)	**0.753**	0.559	0.219	0.018	-0.148
Shrub (% cover)	**0.696**	-0.068	-0.045	0.189	-0.361
Herbaceous (% cover)	**0.560**	-0.008	0.430	0.536	-0.223
P. fremontii and *S. gooddingii* stems/100m², size class D&E	0.175	**0.950**	-0.043	-0.008	-0.093
Tree (% cover)	0.251	**0.896**	0.011	0.007	-0.073
P. fremontii and *S. gooddingii* stems/100m², size class A,B&C	-0.068	**0.834**	-0.060	0.061	0.075
Prosopis spp. stems/100m², size class D&E	0.204	0.115	**0.921**	0.083	-0.017
Burrows/100m²	0.142	-0.186	**0.902**	0.113	-0.094
Prosopis spp. stems/100m², size class A,B&C	-0.068	-0.009	0.055	**0.895**	0.084
Vegetation species richness	0.598	0.175	0.178	**0.638**	0.003
Subshrub (% cover)	-0.170	-0.156	-0.133	0.182	**0.877**
Litter depth (cm)	0.474	0.052	-0.005	-0.139	**0.805**
Eigenvalue	10.35	2.75	2.40	1.61	1.22
% Variation explained	49.28	13.11	11.44	7.65	5.82

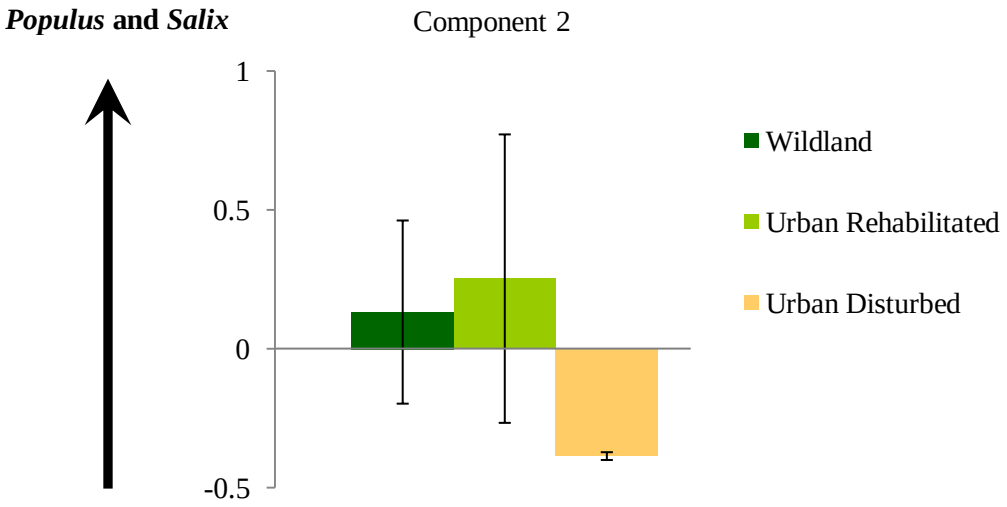

Figure 4. Component 2 Mean Score per Reach
Mean score and standard error of Principal Component Analysis (PCA) component 2 for each of the three reaches selected along the Salt River, Arizona. PCA was performed based on 21 microhabitat variables quantified along eight transects established at each selected reach. A high score indicates a high density and percent cover of *P. fremontii* and *S. gooddingii.*

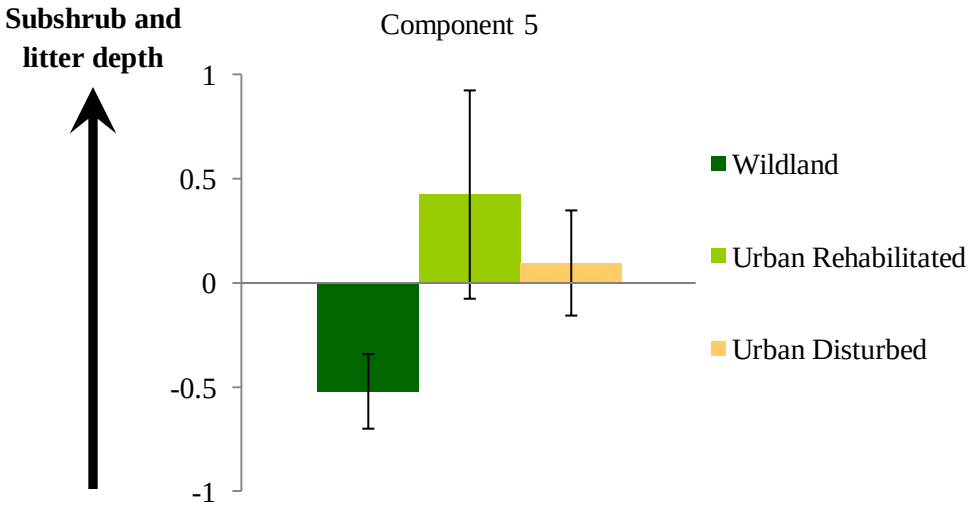

Figure 5. Component 5 Mean Score per Reach
Mean score and standard error of Principal Component Analysis (PCA) component 5 for each of the three reaches selected along the Salt River, Arizona. PCA was performed based on 21 microhabitat variables quantified along eight transects established at each selected reach. A high score indicates a high subshrub percent cover along with deep litter, where litter is present.

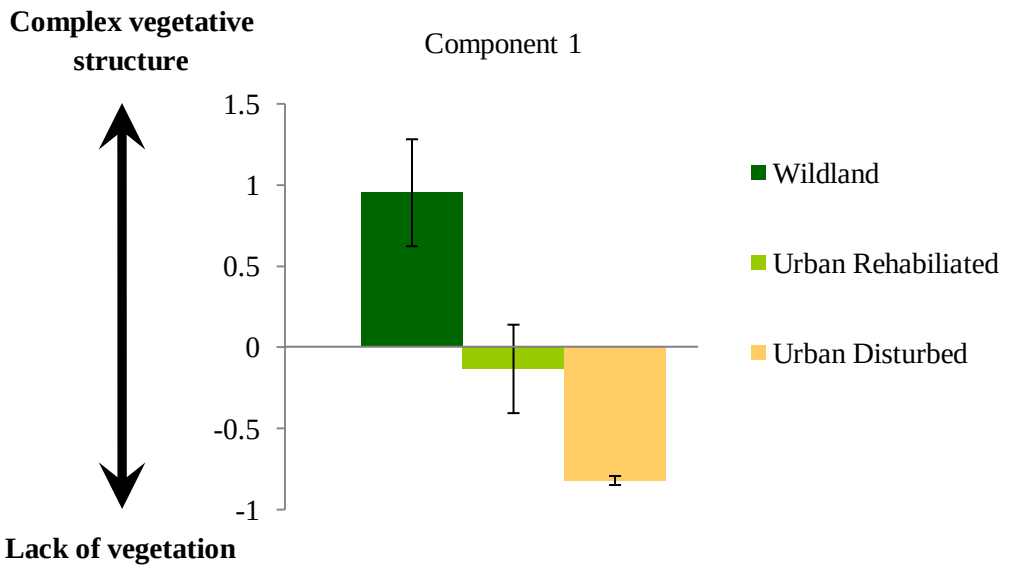

Figure 6. Component 1 Mean Score per Reach
Mean score and standard error of Principal Component Analysis (PCA) component 1 for each of the three reaches selected along the Salt River, Arizona. PCA was performed based on 21 microhabitat variables quantified along eight transects established at each selected reach. A high score represents a highly complex vegetation structure.

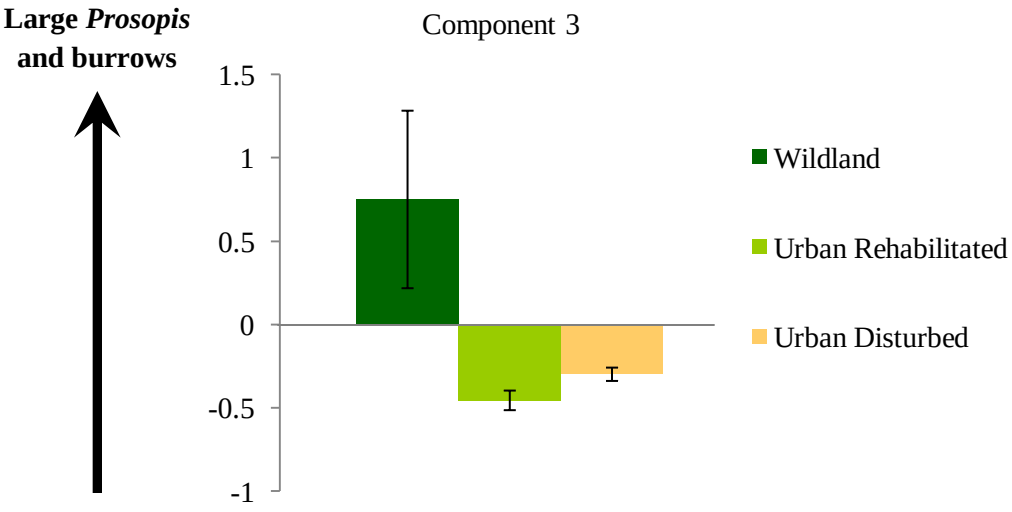

Figure 7. Component 3 Mean Score per Reach
Mean score and standard error of Principal Component Analysis (PCA) component 3 for each of the three reaches selected along the Salt River, Arizona. PCA was performed based on 21 microhabitat variables quantified along eight transects established at each selected reach. A high score indicates high density of large *Prosopis* spp. and high density of burrows.

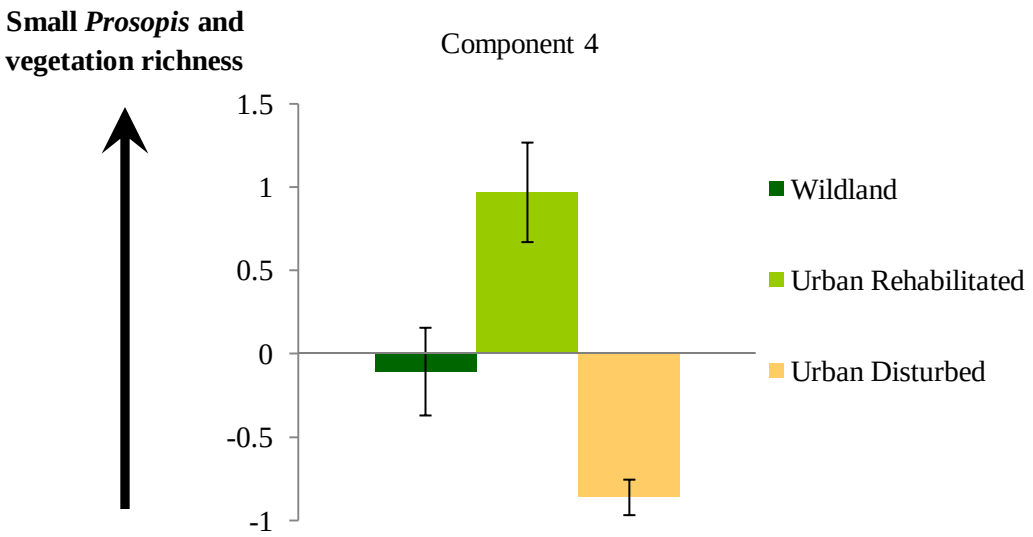

Figure 8. Component 4 Mean Score per Reach
Mean score and standard error of Principal Component Analysis (PCA) component 4 for each of the three reaches selected along the Salt River, Arizona. PCA was performed based on 21 microhabitat variables quantified along eight transects established at each selected reach. A high score indicates a high density of small *Prosopis* spp. and presence of numerous vegetation species.

Relationships

Some microhabitat variables were good predictors of herpetofauna community parameters (Table 3). For instance, herpetofauna abundance, diversity, and species richness were positively correlated with a high complexity of vegetation structure, as well as high density of small *Prosopis* spp., and high vegetation species richness. Furthermore, herpetofauna diversity, and species richness were positively associated with a high density of large *Prosopis* spp., and high density of burrows. Species richness was also positively influenced by high percent cover and density of *P. fremontii* and *S. gooddingii.*

Table 3. Spearman Correlation Coefficients for Herpetofauna Community Parameters and Microhabitat

Spearman correlation coefficients (*r*) between microhabitat (Principal Component Analysis (PCA) components) and herpetofauna community parameters (i.e., abundance, species richness, and diversity) quantified along three reaches of the Salt River, Arizona. Shannon diversity index was used as the diversity measure. Values in bold highlights significant correlations at *p* less than 0.1.

		PCA Components				
		1	2	3	4	5
Abundance	*r* =	**0.72**	0.26	0.26	**0.55**	0.03
	p=	**0.00007**	0.22	0.22	**0.005**	0.89
Species Richness	*r* =	**0.73**	**0.35**	**0.35**	**0.44**	- 0.12
	p=	**0.00005**	**0.09**	**0.09**	**0.03**	0.57
Diversity	*r* =	**0.74**	0.32	**0.39**	**0.37**	- 0.18
	p=	**0.00004**	0.13	**0.06**	**0.07**	0.41

Species-specific abundances were correlated with microhabitat (Table 4). *Aspidoscelis tigris* (Tiger Whiptail) and *Uta stansburiana* (Common Side-blotched Lizard) were present in higher numbers when the environment had a complex vegetation structure as well as a high density of small *Prosopis* spp. and high vegetation species

richness. *Urosaurus ornatus* (Ornate Tree Lizard) were more abundant where vegetation structure was complex and where there was a high density and percent cover of *Populus* and *Salix*. *Callisaurus draconoides* (Zebra-tailed Lizard) were more abundant where there was a high density and percent cover of *Populus* and *Salix*. *Sceloporus magister* (Desert Spiny Lizard) were more abundant where vegetation structure was complex and where there was a high density of large *Prosopis* and burrows. However, high percent cover of subshrub and litter depth negatively influenced the abundance of *S. magister*.

Table 4. Spearman Correlation Coefficients for Herpetofauna Species and Microhabitat Spearman correlation coefficients (*r*) between microhabitat (Principal Component Analysis (PCA) components) quantified and herpetofauna species observed along three reaches of the Salt River, Arizona. Values in bold highlights significant correlations at *p* less than 0.1

		PCA Components				
		1	2	3	4	5
Aspidoscelis tigris	*r* =	**0.69**	0.02	- 0.006	**0.55**	- 0.04
	p=	**0.0002**	0.91	0.98	**0.005**	0.84
Callisaurus draconoides	*r* =	- 0.01	**0.52**	0.33	0.16	0.32
	p=	0.96	**0.009**	0.11	0.47	0.12
Crotalus atrox	*r* =	0.16	0.20	0.29	0.29	0.02
	p=	0.44	0.36	0.18	0.18	0.94
Dipsosaurus dorsalis	*r* =	0.22	- 0.16	0.26	0.14	- 0.16
	p=	0.29	0.44	0.23	0.53	0.44
Sceloporus magister	*r* =	**0.55**	0.03	**0.58**	0.16	**- 0.43**
	p=	**0.005**	0.89	**0.003**	0.46	**0.04**
Urosaurus graciosus	*r* =	0.22	- 0.17	0.26	0.14	- 0.16
	p=	0.29	0.44	0.23	0.53	0.44
Urosaurus ornatus	*r* =	**0.39**	**0.40**	0.07	- 0.01	- 0.10
	p=	**0.06**	**0.06**	0.74	0.96	0.64
Uta stansburiana	*r* =	**0.44**	0.10	- 0.05	**0.55**	0.27
	p=	**0.03**	0.63	0.82	**0.005**	0.20

DISCUSSION

Along the Salt River, Arizona, herpetofauna communities differed in riparian areas varying in degree of urbanization and extent of rehabilitation effort. For instance, abundance was approximately 6 times lower at the urban disturbed reach compared to the two other reaches, suggesting that rehabilitation positively influenced herpetofauna abundance. However, the diversity indices of the two urban reaches were lower than the diversity of the wildland reach, suggesting that urbanization and/or disturbance negatively impacts herpetofauna diversity. In addition, it suggests that full restoration of the herpetofauna community at the urban rehabilitated reach has yet to be achieved. Nonetheless, the urban rehabilitated reach had greater species richness than the urban disturbed reach, suggesting that rehabilitation of degraded riparian community positively influences species richness within urban settings.

Most species observed at the reaches were detected during the surveys. Species encountered outside of the surveys were documented (Appendix H) but excluded from analyses. Increasing the number of visits could increase the chance to detect more species. However, none of the segments were directly located adjacent to water, therefore seeing *Trachemys scripta* (Pond Slider), *Kinosternon sonoriense* (Sonora Mud Turtle), or *Lithobates catesbeianus* (American Bullfrog) within a segment would be unlikely. In order to have a better representation of all herpetofauna species present at each reach, as many sampling methodologies as possible should be used (Hutchens and DePerno 2009). Conversely, since I used only one technique, I was able to perform five visits and recorded an adequate representation of diurnal lizards present at each reach.

Microhabitat

Microhabitat differed among the three riparian areas. Compared to the other reaches, the wildland reach had a highly complex vegetation structure, and harbored large *Prosopis*. These microhabitat characteristics may reflect some recovery conditions of the reach from prior disturbance. Through time, the vegetation structure diversified and *Prosopis* matured. The urban rehabilitated reach had a "medium" complexity of vegetation structure, and high density of small *Prosopis*. The dominance of small *Prosopis* at the reach perhaps reflects the recent rehabilitation activities. The urban disturbed reach lacked overall vegetation, reflecting its high level of disturbance.

Although *Populus* and *Salix* occurrence did not significantly differ among reaches, when looking at the means of *Populus* and *Salix* stems per 100 m^2 and percent cover (Table 1), it is evident that the urban disturbed reach did not support any *P. fremontii* and *S. gooddingii* within the segments. It is possible that no significant differences were detected among reaches due to high variability within the reaches.

Microhabitat and Herpetofauna

I established correlations between herpetofauna community parameters and microhabitat, which were consistent with other studies. For instance, I detected positive correlations between all three herpetofauna community parameters and vegetation structure complexity (Table 3), which concur with several other studies (Pianka 1966; Jones 1981; Jakle and Gatz 1985). I also detected positive correlations between herpetofauna community parameters and occurrence of *Populus, Salix,* and *Prosopis* (Table 3). *Populus* and *Salix* forest associations may support a considerable diversity and abundance of herpetofauna (Rosen 2009), and resources provided by *Prosopis* woodlands are extensively used by herpetofauna (Rosen 2009). Of the five microhabitat components

generated, the complexity of the vegetation structure could have the greatest impact on herpetofauna, as it positively influenced all three herpetofauna community parameters and four species abundances.

Of the five species having a correlation with microhabitat (Table 4), only *C. draconoides* did not correlate with the complexity of vegetation structure. Indeed, *C. draconoides* inhabits areas where plant growth is scarce and where there are open spaces for running (Stebbins 2003). I observed three arboreal lizards during my study: *U. gracious*, *U. ornatus*, and *S. magister*. Two of them correlated with trees; *U. ornatus* was positively correlated with *Populus* and *Salix* component, and *S. magister* was positively correlated with large *Prosopis* and burrow component. These three arboreal lizards may occur in close association in riparian communities of central Arizona (Vitt et al. 1981). The high number of *U. ornatus* at the urban rehabilitated reach (Fig. 2) may be explained by the lack of *S. magister* and *U. gracious*, leaving all the microhabitat available for *U. ornatus* to occupy. I did not detect any correlation between microhabitat and three herpetofauna species (Table 4), perhaps because they were present in such small numbers (Fig. 2).

Vegetation structure diversity is often positively correlated with herpetofauna diversity (Pianka 1966; Jones 1981; Jakle and Gatz 1985), and abundance (Pianka 1966; Jones 1981; Jakle and Gatz 1985; Germano and Lawhead 1986). However, despite differences in vegetation structure complexity among all reaches, diversity was similar at the two urban reaches, and abundance was similar between the urban rehabilitated and wildland reaches. These similarities in diversity and abundance could potentially be further explained by urbanization.

Wildland versus Urban

Typically, the net effect of urbanization is a loss of herpetofauna species richness (McKinney 2008), which concurs with the lower diversity and species richness found at the urban reaches compared to the wildland reach. However, urbanization may result in hyper-abundance of some species (Germaine and Wakeling 2001). Some species can survive or even thrive in urban settings (urbanophiles), whereas others (urbanophobes) may become extirpated (Rodda and Tyrrell 2008). Urbanophiles have attributes such as wide niche breadth (Rodda and Tyrrell 2008), which is also a characteristic of generalist species. Similar species such as *A. tigris, U. stansburiana,* and *C. draconoides* were observed at both urban reaches (Fig. 2, and *A. tigris* was observed outside the segments at the urban disturbed). The absence of *U. ornatus* from the urban disturbed reach (Fig. 2) could be explained by the lack of vertical structure. *Aspidoscelis tigris, U. stansburiana*, and *U. ornatus*, which are habitat generalists (Brennan and Holycross 2009), could perhaps be urbanophiles. The occurrence of *C. draconoides* at the urban reaches could be explained by the presence of open spaces (Stebbins 2003). The three potential urbanophiles were present in high numbers at the urban rehabilitated reach (Fig. 2). These species' occurrences may have contributed to the abundance similarities between the urban rehabilitated and wildland reaches, despite the reaches' differences in vegetation structure complexity.

In order to be a successful urban species, both animals and humans must tolerate each other's close physical proximity (Rodda and Tyrrell 2008). Thus, species such as *Crotalus atrox* (Western Diamond-backed Rattlesnake) could be absent from urban areas since humans typically don't tolerate dangerous species. Other species such as *S. magister*, *Dipsosaurus dorsalis* (Desert Iguana), and *Urosaurus gracious* (Long-tailed Brush Lizard), which were absent from the urban reaches (Fig. 2), could perhaps be

urbanophobes, and may not be able to thrive in urban systems. Unfortunately, there is a paucity of studies separating herpetofauna into urbanophiles and urbanophobes (Rodda and Tyrrell 2008). This lack of information makes it difficult to conclude if some species are absent from the urban reaches because they can't survive in urban settings, or if there are other explanations for their absence. In my study, physical factors such as habitat isolation, soil compaction, and habitat patch size, and biotic factors such as predator and prey densities could be limiting the presence of some species at the urban reaches.

Connectivity or dispersal barriers may influence species occurrence of the urban reaches. Indeed, habitat patches created in urban environments may be colonized naturally, but isolation, dispersal barriers, and a lack of source populations limit the colonization process (Scott et al. 2001). Szaro and Belfit (1986) suggest that the lack of riparian herpetofauna to a newly formed riparian zone may result from an isolation factor. Furthermore, Burbrink et al. (1998) found that species richness is inversely correlated with distance to core areas and Jones et al. (1985) found that isolation was of primary importance to predict species richness. Looking at the landscape surrounding the urban reaches with, for instance, a Geographic Information Systems (GIS), could provide more information on the potential connectivity of the area. Szaro and Belfit (1986) suggest that introduction of herpetofauna might be necessary, particularly in isolated "island" situations such as isolated urban riparian areas. Indeed, the riparian area of the urban rehabilitated reach is isolated in the sense that upstream and downstream riparian communities are highly disturbed. However, connectivity and dispersal barriers may not be the main limiting factor for herpetofauna diversity of the urban rehabilitated reach; one *Lampropeltis getula* (Common Kingsnake) was observed at the urban rehabilitated reach, which may suggest that species typically not observed in urban settings are able to colonize the urban rehabilitated reach.

Soil compaction can play a role in the occurrence of herpetofauna species (Garden et al. 2007) and may contribute to the low diversity of the urban reaches. Burrowing species would unlikely inhabit or persist in areas with hardened soils where burrowing is difficult (Garden et al. 2007). When compared to the wildland reach, it is possible that the urban reaches have more compacted soils and perhaps have a lower density of burrowing wildlife, as evidenced by the lower density of burrows. Highly disturbed areas and areas with low vegetation cover, such as the urban disturbed reach, often have compacted soils (Garden et al. 2007). It is also possible that compacted soils resulted from rehabilitation activities such as the creation of terraces and mechanical disturbances (i.e., heavy equipment use) at the urban rehabilitated reach. Thus, the urban reaches may not be suitable for species using burrows such as *D. dorsalis*, *S. magister*, and *U. gracious* (Stebbins 2003), which were found only at the wildland reach (Fig. 2).

Habitat patch size could influence herpetofauna community along the reaches. For instance, Hokit and Branch (2003) looked at scrub habitat and found a positive relationship between patch size and *S. woodi* (Florida Scrub Lizard) abundance, recruitment, and survivorship. Conversely, Burbrink et al. (1998) and Jones et al. (1985) concluded that the width or size of the riparian area doesn't significantly affect herpetofauna species richness. Results from my research suggest that the size of the area didn't have an impact on herpetofauna abundance, as the wildland reach is much larger than the urban rehabilitated reach, and they had similar abundances. However, habitat patch size may have influenced herpetofauna diversity. It is possible that smaller areas have less habitat diversity, and habitat heterogeneity positively impact herpetofauna species richness (Burbrink et al. 1998).

Trophic interactions play a role in occurrence and abundance of herpetofauna and may have influenced my study results. For instance, predator densities may impact the

herpetofauna assemblage (Hawlena and Bouskila 2006). Indeed, the predation hypothesis may play a role in the number of herpetofauna species present (Pianka 1967). The predation theory suggests that local animal diversity is positively related to abundance of predators as well as predators' efficiency at preventing monopolization of resources by a single species (Paine 1966). It is possible that there is a low density and diversity of predators at the urban reaches compared to the wildland reach. This perhaps contributed to the fact that few herpetofauna species dominated the urban reaches. However, I observed predators such as coyotes, road runners, and raptors at all three reaches. Furthermore, differences in availability of prey items such as insects could also influence herpetofauna. Indeed, if food is a limiting factor, food resources could regulate lizard communities (Barbault and Maury 1981). Wenninger and Fagan (2000) investigated wolf spider-habitat associations along the Salt River, Arizona. They compared a site where sufficient water flowed in the riverbed, upstream from Granite Reef Dam (site nearby the wildland reach in my study), to a site downstream from the dam which had less flowing water (similar to the urban reaches in my study). They found a higher abundance of wolf spiders and a higher diversity of prey items at the site nearby the wildland reach, although abundance of prey items did not differ between sites. This suggests that there is a higher diversity of prey items at the wildland reach compared to the urban reaches, which could contribute to the more diverse herpetofauna community of the wildland reach.

Rehabilitation Activities

Herpetofauna abundance and species richness may be higher at the urban rehabilitated reach compared to the urban disturbed reach because of the differences in amount of vegetation present (Fig. 4, 6 and 8). This observation concurs with Attum et al. (2006)

which found that desert lizard species richness and abundance were higher at sites having higher vegetation percent cover and height. Thus, this suggests that rehabilitation activities such as planting vegetation positively influenced herpetofauna abundance and species richness.

However, the fairly recent rehabilitation activities of the urban rehabilitated reach may be a factor limiting herpetofauna diversity. For instance, it is possible that the site is too "young" and herpetofauna may not have had enough time to colonize the newly hospitable area. Ecological time theory, or time required for dispersal of species into newly suitable areas, is of paramount importance for herpetofauna occurrence (Pianka 1967). The area may be saturated ecologically, meaning that all resources are being used, perhaps as reflected by the high herpetofauna abundance of the urban rehabilitated reach, but it may be unsaturated in total number of species the area may support (Pianka 1967). Indeed, ecological time is an important determinant for diversity in cases where dispersal is impeded by pronounced barriers (Pianka 1967), perhaps such as urban landscapes. Furthermore, time for the urban rehabilitated reach to mature and diversify has been limited, and maybe some key elements are missing. For instance, Queheillalt and Morrison (2006) found that abundant herpetofauna species present in their restored site were generalist species; whereas, habitat specialists were exclusively present in the comparison site. Queheillalt and Morrison (2006) suggested that the restored site was not yet suitable for habitat specialists. In my study, a positive correlation was detected between *S. magister* and large *Prosopis* (Table 4), which were both lacking at the urban reaches (Fig. 2 and 7). With time, the *Prosopis* of the urban rehabilitated reach will mature, get larger, and maybe then *S. magister* will occur at the urban rehabilitated reach. Moreover, with time, the overall area will further diversify (i.e., trees will mature, some will die and become snags or logs, branches will fall and create more woody debris, etc.).

Thus, time could create a higher diversity of microhabitat, which could positively impact herpetofauna community diversity (Vitt et al. 1981; Jones 1988b; Amo et al. 2007).

Conclusion

Overall, rehabilitation activities appear to positively influence herpetofauna abundance, and species richness; whereas, urbanization appears to negatively influence herpetofauna diversity. However, further investigations pertaining to aspects previously mentioned such as connectivity, soil compaction, size of area, and predator and prey densities could provide insights on other components potentially influencing herpetofauna assemblages along the Salt River, Arizona.

Future Research

Future research could evaluate specific microhabitat characteristics. Szaro and Belfit (1986) suggest that there is a loss of information resulting from the summarization process. Previous studies have found strong herpetofauna species-specific substrate preferences (Jones and Glinski 1985; Warren and Schwalbe 1985; Jones 1988a; Jones 1988b) and found that various substrate may influence herpetofauna's behavior (Huey et al. 1989; Sabo 2003; Rosen 2009). Furthermore, plant volume diversity is highly correlated with lizard diversity (Pianka 1966; Pianka 1967). Moreover, plant species specific data such as plant taxa and floristics could potentially explain most variation in herpetofauna species abundance (Szaro and Belfit 1986), perhaps due to the fact that different plant species provide different ambient temperature, different prey availability, and different camouflage opportunity (Valentine et al. 2007). Thus, exploring relationships between herpetofauna and single microhabitat variables could provide more detailed information regarding specific components influencing herpetofauna. However,

Garden et al. (2007) found that in isolated and highly fragmented lowland remnant vegetation patches within an urban setting, habitat structural complexity is more important than floristics to describe occurrence of native reptiles.

Other research, such as long-term monitoring of the urban rehabilitated reach, could provide insights on processes regulating herpetofauna community assemblages. For instance, long-term monitoring could perhaps elucidate if time is a main limiting factor driving diversity in the rehabilitated area, as it may take a long time for ecosystems to fully recover from disturbance and thus regain their original wildlife species composition. Conversely, long-term monitoring of the urban rehabilitated reach could conclude that rehabilitation alone is not sufficient, and that other methods, perhaps translocation, may be necessary for the full restoration of the ecosystem components such as the herpetofauna community. Long-term monitoring studies of rehabilitated areas are necessary and highly valuable to further understand the approaches used in restoration. For instance, long-term monitoring of the urban rehabilitated reach could provide insights to the "field of dream" approach, which is commonly used in stream restoration (Hilderbrand et al. 2005).

Future research addressing the use of supplemental sediments by burrowing species could be beneficial to habitat rehabilitation projects. The addition of burrowing substrate could potentially counter the negative effects of soil compaction and increase the use of an area by burrowing species.

Implications for Practice

Based on the relationships generated from my research project, I developed the following recommendations regarding which microhabitat features should be created when rehabilitating southwestern degraded riparian systems to benefit herpetofauna. Since

structural diversity is very important, I would suggest planting several different vegetation species of various growth habits such as trees, tree shrubs, shrubs, and subshrubs. This would provide a diversity of vegetation layers as well as a higher microhabitat diversity and greater diversity of resources such as shelter and food. I would also suggest leaving woody debris on site, as woody debris can provide suitable basking locations, habitat for prey species, and nesting and refuge niches for herpetofauna (Garden et al. 2007), in addition to contributing to structural diversity. Furthermore, if there is a lack of woody debris in an area being rehabilitated, it could be beneficial to introduce woody debris. I would also suggest planting *Populus, Salix*, and *Prosopis*, as they provide important resources for herpetofauna in southwestern systems (Rosen 2009). Furthermore, instead of having a forest composed principally of a monotypic age class, I would suggest planting trees of various ages and sizes, which could help to provide a more diverse habitat early in the rehabilitation process. Finally, promoting small mammal abundance could be beneficial, as small mammals would increase burrow density, which was positively correlated with herpetofauna species richness and diversity.

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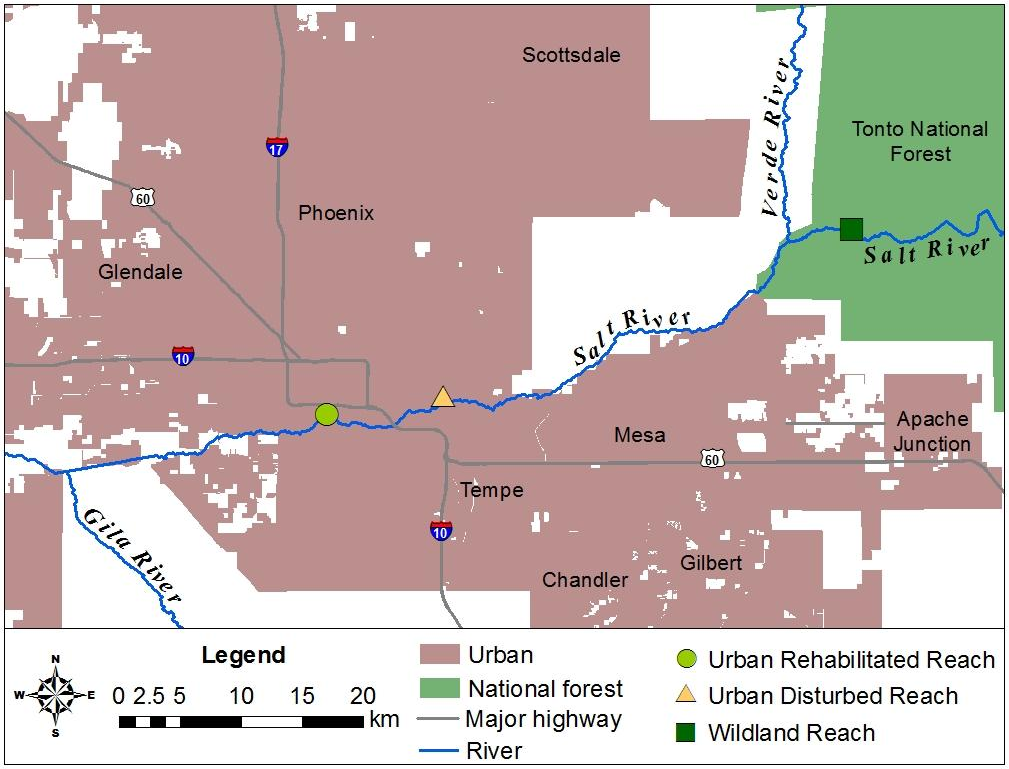
APPENDIX A

MAP OF STUDY AREA



Appendix A. Map of the study area depicting location of three selected reaches which differ in terms of rehabilitation effort and urbanization along the Salt River, Arizona.

APPENDIX B

PHOTOGRAPHS OF THE THREE REACHES

Photograph I. Photo representing the wildland reach, one of three reaches selected to perform herpetofauna visual surveys and to quantify microhabitat along the Salt River, Arizona. This reach is located approximately five miles from closest city boundary and has recovered from previous disturbance.

Photograph II. Photo representing the urban rehabilitated reach, one of three reaches selected to perform herpetofauna visual surveys and to quantify microhabitat along the Salt River, Arizona. This reach is located in an urban setting. Within the last ten years, rehabilitation activities such as garbage removal, terraces and low flow channel creation, as well as native southwestern riparian vegetation planting occurred.

Photograph III. Photo representing the urban disturbed reach, one of three reaches selected to perform herpetofauna visual surveys and to quantify microhabitat along the Salt River, Arizona. This highly disturbed reach is located in an urban setting and has lost most of the riparian vegetation.

APPENDIX C

TRANSECTS LOCATION

Appendix C. Start and end location of 24 transects established at three reaches along the Salt River, Arizona. Transects were used to collect data on herpetofauna occurrence and abundance as well as riparian microhabitat. Locations are presented in Universal Transverse Mercator (UTM), North American Datum of 1983 (NAD83).

Wildland Reach

Transect	Start			End		
1	12 S	443514 N	3713327 E	12 S	443416 N	3713297 E
2	12 S	443618 N	3713199 E	12 S	443470 N	3713141 E
3	12 S	443673 N	3713057 E	12 S	443561 N	3713018 E
4	12 S	443678 N	3712908 E	12 S	443596 N	3712871 E
5	12 S	443708 N	3712758 E	12 S	443653 N	3712732 E
6	12 S	443347 N	3713509 E	12 S	443227 N	3713318 E
7	12 S	443192 N	3713537 E	12 S	443128 N	3713453 E
8	12 S	442941 N	3713461 E	12 S	443006 N	3713350 E

Urban Rehabilitated

Transect	Start			End		
1	12 S	399876 N	3698626 E	12 S	399850 N	3698676 E
2	12 S	400034 N	3698638 E	12 S	400044 N	3698714 E
3	12 S	400313 N	3698586 E	12 S	400314 N	3698653 E
4	12 S	400470 N	3698551 E	12 S	400492 N	3698626 E
5	12 S	400909 N	3698419 E	12 S	400913 N	3698354 E
6	12 S	401122 N	3698426 E	12 S	401124 N	3698372 E
7	12 S	400594 N	3698466 E	12 S	400587 N	3698374 E
8	12 S	400745 N	3698444 E	12 S	400741 N	3698360 E

Urban Disturbed

Transect	Start			End		
1	12 S	409335 N	3699857 E	12 S	409316 N	3699978 E
2	12 S	409489 N	3699847 E	12 S	409509 N	3699770 E
3	12 S	409633 N	3699900 E	12 S	409652 N	3699819 E
4	12 S	409779 N	3699944 E	12 S	409800 N	3699850 E
5	12 S	409925 N	3699979 E	12 S	409948 N	3699873 E
6	12 S	410077 N	3700007 E	12 S	410074 N	3700154 E
7	12 S	410230 N	3700009 E	12 S	410232 N	3700143 E
8	12 S	410385 N	3699996 E	12 S	410394 N	3700122 E

APPENDIX D

MICROHABITAT METHODS AND VARIABLES

Appendix D. Description of methods used to quantify twenty one microhabitat variables used to describe herpetofauna abundance and occurrence. Variables were quantified along eight transects established at three reaches along the Salt River, Arizona (total of 24 transects). Data were collected within three 10-m wide by 20-m long segments established along each transect (total of 72 segments). Refer to Appendix E for segment schematic showing pre-determined locations used to quantify microhabitat variables.

Method	**Method Description**
Line-intercept	I used the line-intercept method to measure % cover of various variables. The raw data was collected in meters, to the closest 0.05 m. I recorded the start and end of each variable described below along three pre-determined lines totaling 40 m per segment; 120 m per transect. To calculate the % cover reported, I added all the meters for one variable along the whole transect and converted the total to a percentage; variable # of meters/120 m*100.
Variables	**Variables Description**
Debris pile (% cover)	I defined a debris pile as being an accumulation of debris (litter and/or woody) that was at least 10cm in height and 20cm in length.
Subshrub (% cover)	This variable quantified the % cover of vegetation qualified as having a "subshrub" or "subshrub shrub" growth habit as defined per USDA and NRCS 2011.
Shrub (% cover)	This variable quantified the % cover of vegetation qualified as having a "shrub" or "shrub tree" growth habit as defined per USDA and NRCS 2011.
Tree shrub (% cover)	This variable quantified the % cover of vegetation qualified as having a "tree shrub" growth habit as defined per USDA and NRCS 2011.
Tree (% cover)	This variable quantified the % cover of vegetation qualified as having a "tree" growth habit as defined per USDA and NRCS 2011.
Lack of vegetation (% cover)	This variable quantified the % cover of lack of vegetation. "Lack of vegetation" was defined as an area lacking vegetation such as tree, tree shrub, shrub and subshrub from the ground, up to the sky. However, if herbaceous vegetation and/or forbs were present, it was still recorded as "lack of vegetation".

Open understory (% cover)	This variable quantified the % cover of an open understory. "Open understory" was defined as an area free of vegetation such as tree, tree shrub, shrub and subshrub from the ground, up to 1 m in height. However, if herbaceous vegetation and/or forbs were present, it was still recorded as "open understory".

Method	**Method Description**
Point intercept	I used a point intercept method to determine % cover of ground cover type and average depth of litter and woody, where litter or woody were present. I recorded the ground cover type at every meter along the 20-m line used for the line-intercept for a total of 20 points per segment; 60 points per transect. To calculate the % cover per ground cover type, I added the number of points for each ground cover type along the whole transect and convert to %; ground cover type # of points/60 points *100. I recorded the litter and woody depth in cm, to the closest 0.5 cm, at each point where litter or woody were present. I calculated the average litter and woody depth by adding all litter or woody depth recorded for a whole transect and divided by the number of points where litter or woody were present.
Variables	**Variables Description**
Bare ground	Bare ground cover type was defined as a lack of litter and/or woody debris.
Litter ground	Litter ground cover type was defined as vegetation debris such as various non-woody plant material and/or small twigs having a diameter less than 1cm at the point location.
Woody ground	Woody ground cover type was defined as woody vegetation debris having a diameter greater than 1cm at the point location.
Litter depth (cm)	This variable represents the average depth of litter, where litter was present. Refer to "litter ground" variable for the definition of litter.
Woody depth (cm)	This variable represents the average depth of woody, where woody was present. Refer to "woody ground" variable for the definition of woody.

Method	Method Description
Daubenmire plots	Daubenmire plots (Daubenmire 1959) were used to quantify the vegetation qualified as herbaceous. I quantified four Daubenmire plots per segment; 12 per transect. I set the Daubenmire plots at pre-determined locations along the 20-m line used for the line-intercept method. They were located at point 0, 5, 10, and 15 m, on the upstream side of the line. I used the Daubenmire scale (Daubenmire 1959) to determine class cover. Daubenmire scale class cover are: not present, less than 1%, 1-5%, 5-25%, 25-50%, 50-75%, 75-95%, and 95-100%. To determine the herbaceous % cover for a transect, I added all the middle values from the class cover scale and divided by 12.
Variable	**Variable Description**
Herbaceous (% cover)	The herbaceous variable includes all grasses and forbs.

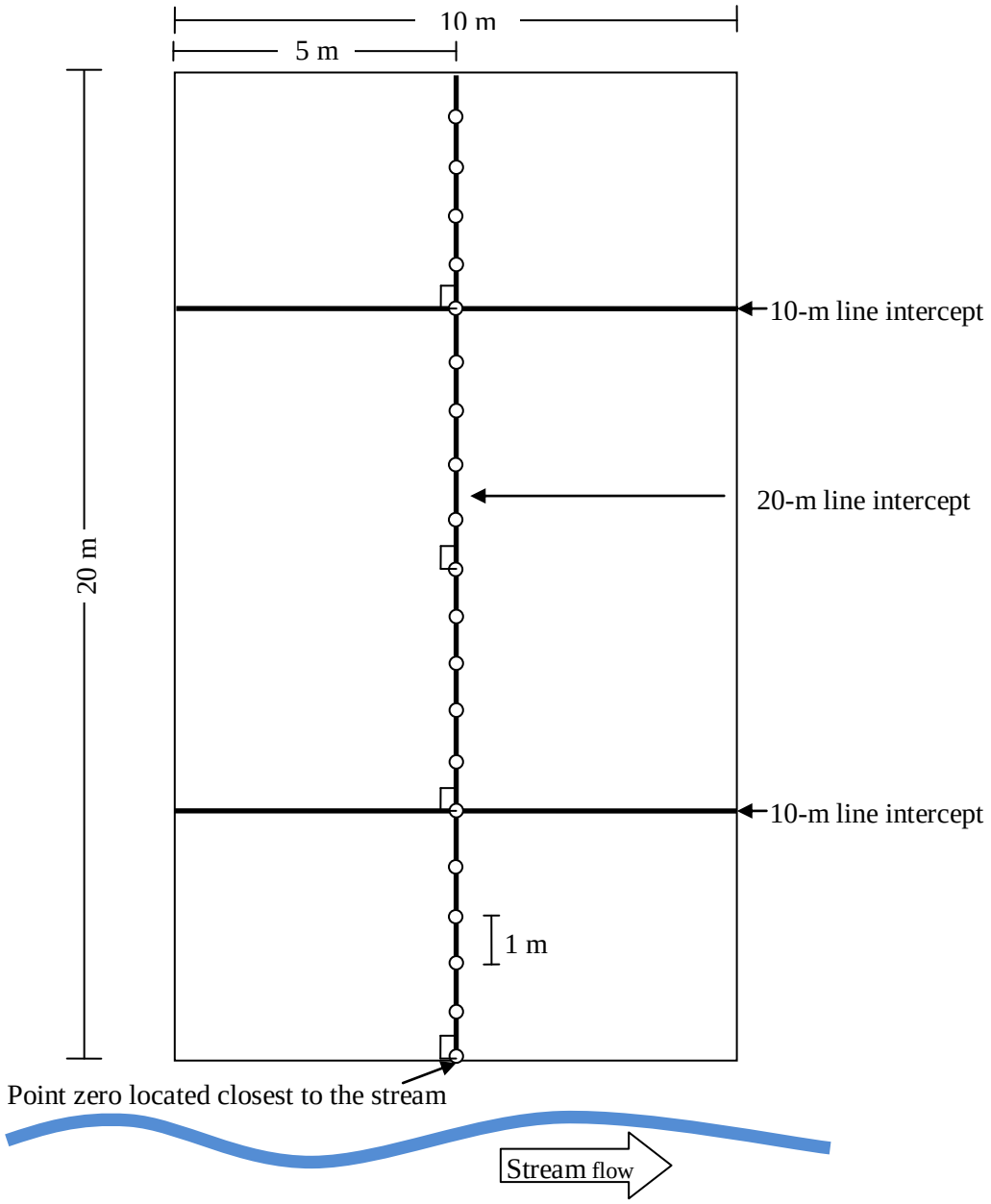
Method	Method Description
Convex spherical densiometer	A convex spherical densiometer was used to quantify overstory % cover. I took readings at two pre-determined points per segment; six per transect. The pre-determined points were located at 5 and 15 m along the 20-m line used for the line-intercept. I took four readings at each point; two readings facing each direction of the 20-m line and two readings facing each direction of the 10-m lines. The average of the four readings for a point results in the overstory % cover for that point. To determine the overstory % cover for the whole transect, I added the six averaged reading for that transect and divided the total by 6.
Variable	**Variable Description**
Overstory (% cover)	I took the readings with the densiometer at breast height. Thus, everything above waste height that was seen in the densiometer was recorded as overstory % cover.

Method	Method Description
Plot stem count	I counted all stems present within a pre-determined 5 by 20 m plot nested within the segment. The plot was half of the segment, the half located on the upstream side of the 20-m line used for the line-intercept. I counted every stems that had more than half of the stem located inside the plot. I calculated the stem density for each transect by adding all stems per size class counted along the three segments and divided by three. The density is expressed as the number of stems/100 m².
Variables	**Variables Description**
Prosopis spp. stem density per 100 m² per size class	I counted all *Prosopis* spp. stems present within the plot. *Prosopis* spp. stems were divided into five size classes. I recorded the diameter at 20 cm off the ground. Size classes are: A = height less than 1 m; B = height greater than 1 m and diameter less than 2.5 cm; C = height greater than 1 m and diameter between 2.5 and 7.5 cm; D = height greater than 1 m and diameter between 7.5 and 30 cm; and E = height greater than 1 m and diameter greater than 30 cm.
P. fremontii stem density per 100 m² per size class	I counted all *P. fremontii* stems present within the plot. *P. fremontii* stems were divided into five size classes. I recorded the diameter at breast height (approximately 1.3 m off the ground). Size classes are: A = height less than 1 m; B = height greater than 1 m and diameter less than 5 cm; C = height greater than 1 m and diameter between 5 and 10 cm; D = height greater than 1 m and diameter between 10 and 30 cm; and E = height greater than 1 m and diameter greater than 30 cm.
S. gooddingii stem density per 100 m² per size class	I counted all *S. gooddingii* stems present within the plot. *S. gooddingii* stems were divided into five size classes. I recorded the diameter at breast height (approximately 1.3 m off the ground). Size classes are: A = height less than 1 m; B = height greater than 1 m and diameter less than 5 cm; C = height greater than 1 m and diameter between 5 and 10 cm; D = height greater than 1 m and diameter between 10 and 30 cm; and E = height greater than 1 m and diameter greater than 30 cm.

Method	**Method Description**
Various	Described below
Variables	**Variables Description**
Burrow density per 100 m^2	All burrows encountered within each segment were counted. For each transect, I calculated the density of burrows per 100 m^2 by adding the number of burrows encountered in each segment divided by six.
Woody debris per 10m	Woody debris was defined as all dead woody plant material with a diameter greater than 1cm, and located from the ground, up to 0.5 m in height. Thus, woody debris also included dead branches still attached to the plant. I recorded the number of woody debris along the 20-m line used for the line-intercept, along a total of 10 m per segment; 30 m per transect. I counted woody debris along 1m in length, 1cm in width and 0.5 m in height, at every other meter, starting at point zero. For each transect, I calculated the average number of woody debris per 10 m by adding all woody debris counted in each segment and divided the total by three.
Vegetation species richness	I recorded all vegetation species for growth habit such as tree, tree shrub, shrub, subshrub, and subshrub forbs present within each segment. Vegetation species richness reported for the transect is the total number of species present within all three segments.

APPENDIX E

SEGMENT SCHEMATIC



Appendix E. Segment (10-m wide by 20-m long) schematic depicting pre-determined locations to quantify microhabitat variables used to describe herpetofauna abundance and occurrence. Eight transects were established at three reaches along the Salt River, Arizona (total of 24 transects). Three 10-m wide by 20-m long segments were established along each transect (total of 72 segments). Appendix D describes microhabitat variables quantified within each segment along with method used. Daubenmire plot locations (⊏), lines used to perform the line-intercept (——), and points used for the point-intercept (○) are depicted.

APPENDIX F

GROWTH HABIT PER SPECIES

Appendix F. List of vegetation species observed along eight transects established at three reaches along the Salt River, Arizona (total of 24 transects). Transects consisted of three 10-m wide by 20-m long segments (total of 72 segments). Growth habit according to USDA and NRCS (2011).

Genus	**Species**	**Growth Habit**
Cassia	*covesii*	subshrub forb
Machaeranthera	*pinnatifida*	subshrub forb
Sphaeralcea	*ambigua*	subshrub forb
Stephanomeria	*pauciflora*	subshrub forb
Ambrosia	*ambrosioides*	subshrub shrub
Ambrosia	*eriocentra*	subshrub shrub
Atriplex	*lentiformis*	subshrub shrub
Bebbia	*juncea*	subshrub shrub
Encelia	*farinosa*	subshrub shrub
Hymenoclea	*monogyra*	subshrub shrub
Trixis	*californica*	subshrub shrub
Hymenoclea	*salsola*	subshrub
Isocoma	*acradenia*	subshrub
Atriplex	*canescens*	shrub
Baccharis	*salicifolia*	shrub
Baccharis	*sarothroides*	shrub
Hyptis	*emoryi*	shrub
Justicia	*californica*	shrub
Larrea	*tridentata*	shrub
Lycium	spp.	shrub
Opuntia	*acanthocarpa*	shrub
Opuntia	*engelmannii*	shrub
Pluchea	*sericea*	shrub
Simmondsia	*chinensis*	shrub
Celtis	*ehrenbergiana*	shrub tree
Acacia	*constricta*	tree shrub
Acacia	*farnesiana*	tree shrub
Acacia	*greggii*	tree shrub
Chilopsis	*linearis*	tree shrub
Parkinsonia	*aculeate*	tree shrub
Parkinsonia	*floridum*	tree shrub
Parkinsonia	*microphyllum*	tree shrub
Prosopis	spp.	tree shrub
Tamarix	*pentandra*	tree shrub
Vitex	*agnus castus*	tree shrub
Ziziphus	*obtusifolia*	tree shrub
Carnegiea	*gigantea*	tree
Populus	*fremontii*	tree
Salix	*gooddingii*	tree

APPENDIX G

LIST OF HERPETOFAUNA SPECIES

Appendix G. Family, common name, and scientific name of eight herpetofauna species observed while performing visual surveys along eight transects established at three reaches along the Salt River, Arizona (total of 24 transects). Surveys were performed within three 10-m wide by 20-m long segments established along each transect (total of 72 segments). Herpetofauna visual surveys were performed five times along each segment between June and August 2010.

Family	Common Name	Scientific Name
Iguanidae	Desert Iguana	*Dipsosaurus dorsalis*
Phrynosomatinae	Zebra-tailed Lizard	*Callisaurus draconoides*
Phrynosomatinae	Desert Spiny Lizard	*Sceloporus magister*
Phrynosomatinae	Long-tailed Brush Lizard	*Urosaurus graciosus*
Phrynosomatinae	Ornate Tree Lizard	*Urosaurus ornatus*
Phrynosomatinae	Common Side-blotched Lizard	*Uta stansburiana*
Teiidae	Tiger Whiptail	*Aspidoscelis tigris*
Viperidae	Western Diamond-backed Rattlesnake	*Crotalus atrox*

APPENDIX H

HERPETOFAUNA SPECIES ENCOUNTERED OUTSIDE SURVEYS

Appendix H. Common name and scientific name of herpetofauna species observed at the selected reaches along the Salt River, Arizona but not encountered while performing herpetofauna visual surveys.

Reach	Common Name	Scientific Name
Wildland	Sonora Mud Turtle	*Kinosternon sonoriense*
	Coachwhip	*Coluber flagellum*
	Common Kingsnake (observed skin only)	*Lampropeltis getula*
Urban Rehabilitated	American Bullfrog	*Lithobates catesbeianus*
	Pond Slider	*Trachemys scripta*
	Common Kingsnake	*Lampropeltis getula*
Urban Disturbed	Tiger Whiptail	*Aspidoscelis tigris*
	American Bullfrog	*Lithobates catesbeianus*