The Effects of Artificial Water Sources on Small Mammal Communities

by

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ABSTRACT

Modified and artificial water sources can be used as a management tool for game and non-game wildlife species. State, federal, and private agencies allocate significant resources to install and maintain artificial water sources (AWS) annually. Starting in 2007, Barry M. Goldwater Range in southwestern Arizona began renovating AWS. Resource managers were concerned with the effects of AWS on wildlife. I used capture mark recapture methods to sample small mammal communities in the vicinity of five AWS and five paired control sites (treatments) in the surrounding Sonoran desert from October 2011 to May 2012. I measured plant species richness, density, and percent cover in the spring of 2012. A Multi-response Permutation Procedure was used to identify differences in small mammal community abundance, biomass, and species richness by season and treatment. I used Principle Component Analysis to reduce 11 habitat characteristics to five habitat factors. I related rodent occurrence to habitat characteristics using multiple and logistic regression. A total of 370 individual mammals representing three genera and eight species of rodents were captured across 4800 trap nights. Desert pocket mouse (*Chaetodipus penicillatus*) was the most common species in both seasons and treatments. Whereas rodent community abundance, biomass, and richness were similar between seasons, community variables of AWS were greater than CS. Rodent diversity was similar between treatments. Desert pocket mouse abundance and biomass were twice as high at AWS when compared to controls. Biomass of white-throated woodrat (*Neotoma albigula*) was five times greater at AWS. Habitat characteristics were similar between treatments. Neither presence of water nor distance to water explained substantial habitat variation. Occurrence of rodent

species was associated with habitat characteristics. Desert rodent communities are adapted for arid environments (i.e. Heteromyids) and are not dependent on "free water". Higher abundances of desert pocket mouse at AWS were most likely related to increased disturbance and debris and not the presence of water. The results of this study and previous studies suggest that more investigation is needed and that short term studies may not be able to detect interactions (if any) between AWS and desert small mammal communities.

DEDICATION

I would like to dedicate this work to my family, my parents Don and Ginny Barrett, Bruce M. Switalski and my siblings Thad and Myriah.

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Efforts of many people came together making this thesis possible. I would like to thank Dr. Heather Bateman (Arizona State University, Morrison School of Agribusiness and Resource Management, Dr. William Miller, (ASU Morrison School of Agribusiness and Resource Management), and Dr. Eddie Alford (ASU Morrison School of Agribusiness and Resource Management) for their mentorship, assistance on study design, techniques, and analyses. I want to thank Jonathon Quinsey for his time and help in collecting data in the field and Brandy Smith for volunteering her time to assist in data collection. Finally, I would like to thank the Department of the Air Force, Barry M. Goldwater Range and the Army Corps of Engineers, for funding and allowing me to perform this study. I would also like thank Mr. Rick Whittle (Barry M. Goldwater Range) for logistical support. Finally, I am grateful to Dr. Bateman for her unwavering encouragement and support.

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INTRODUCTION

Natural resource managers are concerned with the effects of management practices on habitat and wildlife. Desert ecosystems support a variety of wildlife including: insects (Davidson 1977), mammals (Hoffmeister 1986), birds (Hensley 1954, Naranjo and Raitt 1993), reptiles (Brennan and Holycross 2009), and amphibians (Sullivan and Fernandez 1999). Water is seen as a limiting resource for the distribution of many animal species in arid environments (Roberts 1977, Broyles 1997, Rosenstock et al. 1999); therefore, water has been used as a management tool for game species, livestock (Broyles 1997), and for mitigating the loss of natural water sources from increased aridity, human use, and urbanization (Dolan 2006, Longshore et al. 2009).

Arid ecosystems are punctuated by lower productivity, greater temperatures, unpredictable water resources, and extreme conditions (Polis 1991). These characteristics contribute to arid ecosystems being susceptible to human disturbance (Verstraete and Schwartz 1991, Evans and Belnap 1999). Disturbance is "any relatively discrete event in time that disrupts ecosystem, community, or population structure, and changes resources, substrate availability, or the physical environment" (White and Pickett 1985). Disturbance of natural habitat can lead to loss of biodiversity by changing habitat structure and altering ecosystem function (Clovis and Stacey 2009). Consequently, even management practices such as artificial water sources (AWS) are a concern as they might affect local habitat and wildlife communities.

AWS are human constructed or modified natural water sources (i.e. tinaja) which are maintained. Guzzlers, stock tanks, earthen ponds and other human

constructed water sources can be collectively described as AWS. AWS are commonly used by state and federal agency managers, to supplement or enhance existing natural sources of water. Use of AWS is based on the assumption that increased availability of water would benefit wildlife populations. The use of AWS has been the focus of much debate. Some researchers highlight possible negative effects of AWS which include increased predation (Broyles 1995, Broyles and Cutler 1999), increased competition (Rosenstock et al. 1999), poor water quality, and disease transmission (Schmidt and DeStefano 1996, Swift et al. 2000). In contrast, DeStefano et al. (2000) suggest that AWS have a dispersal effect on predator concentration. Other authors suggest AWS benefit both game and nongame species and suggested negative effects of AWS are not supported (Rosenstock et al. 2004, O'Brien et al. 2006). Rosenstock et al. (2004) notes that studies claiming AWS increases competition and predation use indirect anecdotal observations (i.e. prey remains) or increased sign (e.g. scat) as evidence. Most negative claims have not been directly studied and require additional investigation. Despite this debate, AWS remain a common technique for managing wildlife populations in arid environments (Krausman et al. 2006).

The majority of research on AWS has focused on quail (*Callipepla* spp.), desert bighorn (*Ovis canadensis*), deer (*Odocoileus* spp.), and pronghorn (*Antilocapra americana*), for which the AWS were originally constructed. Despite the frequency of AWS construction, limited research has investigated the effects of AWS on nongame wildlife (i.e. small mammals and birds; Simpson et al. 2011). Recent research investigating vertebrate habitat use (Cutler and Morrison 1998), anthropogenic disturbance (Stacey and Post 2009), wildlife association with and use of AWS

(Burkett and Thompson 1994, O'Brien et al. 2006) has increased our knowledge about the effects of AWS on nongame wildlife. Of the research that has investigated AWS-nongame relationships; Cutler and Morrison (1998) studied multiple taxa at a single pair of AWS sites and Burkett and Thompson (1994) sampled multiple AWS for a limited time (one night). O'Brien et al. (2006) lacked the ability to accurately assess small mammal or rodent occurrence in the vicinity of AWS. Cutler and Morrison (1998) observed no difference in abundance and richness of small mammal between the two sites. In contrast, Burkett and Thompson (1994) documented higher abundances of small mammals but concluded that there was no biological significance or linkage to the presence of water and that elevated abundances were related to increased disturbance and debris in the vicinity of the AWS. By using video surveillance, O'Brien et al. (2006) could only document unknown rodent species visiting AWS on five percent of surveillance days.

Desert rodents are one of the most studied small mammal communities in the Southwest. A large body of research dating from the mid to late 1800's (Mares 1983) has described community structure (Brown and Davidson 1977, Davidson et al. 1980, Brown and Kurzius 1987, Kelt 1996, Brown et al. 2000), reproductive and seasonal activity (Kenagy 1973, Reichman and Van De Graff 1973), and resource utilization (Riechman 1975, Stamp and Ohmart 1978). Several long term studies have focused on desert rodent populations (Heske et al. 1994, Valone and Brown 1995, Valone and Sauter 2005, Thibault et al. 2010). This combined research has detailed the importance of rodent communities in arid environments, such as supporting higher trophic levels by being a major part of the prey base (Lidicker 2007). Rodent communities are affected by many processes (e.g. disturbance,

predation; Valone and Brown 1995) and are sensitive to changes in habitat structure (Whitford 1997, Valone and Sauter 2005). For example, Price (1978) reported differences in habitat selection of Merriam's kangaroo rat (*Dipodomys merriami*) and Arizona pocket mouse (*Perognathus amplus*). With some exceptions (i.e. Sciuridae), Arizona desert rodent communities are composed of granivorous (seed eating), nocturnal rodents of the Heteromyidae family. These "ecological engineers" excavate burrows and build mounds mixing soil and influencing the dispersal of seeds (Hansell 1993, Jones et al. 1994). Some desert rodent species, particularly Merriam's kangaroo rat (*Dipodomys merriami*) are considered keystone species because they influence habitat structure (Brown and Heske 1990). In Arizona, many species of Sonoran Desert rodents are wide spread and sympatric while having differences in habitat selection (Hoffmeister 1986).

Within the Sonoran Desert, Barry M. Goldwater Range (BMGR) is ecologically critical habitat and a significant part (>50 %) of 5000 acres of mainly contiguous and low disturbance Sonoran Desert in the United States (BMGR Integrated Natural Resources Management Plan (INRMP; October 2012). As part of early management (1960's) and the more recent INRMP, multiple AWS were constructed to support Desert bighorn sheep populations and draw Sonoran pronghorn (*Antilocapra americana sonoriensis*) away from military testing ranges (R. Whittle pers. comm.).

Small mammal communities are ideal for investigating habitat change. Sensitivity to environmental changes, wide distribution, and knowledge of ecology make small mammal communities good indicators of change. I chose small mammal communities (specifically rodent communities) to investigate the effects of AWS on

nongame wildlife within the Sonoran Desert. The presence of AWS and undisturbed Sonoran Desert habitat make BMGR an ideal site to investigate effects of AWS on rodent communities. The goal of this study is to investigate the effects of AWS on desert rodent communities and to develop guidance for the management of AWS. I investigated: (1) how rodent species abundance, and richness, and diversity differ between AWS sites and non AWS sites, (2) how vegetation characteristics differ between AWS and non AWS sites, and (3) what vegetation characteristics are good predictors of small mammal species' abundance and occurrence.

STUDY AREA

I conducted this study within the Sauceda Mountains on the Barry M. Goldwater Range-East (BMGR-E), a 424,919 ha military training area in southern Arizona, USA. The study site was located approximately 39 km south of Gila Bend, Arizona and 32 km northeast of Ajo, Arizona in Maricopa County (Figure 1). Study site elevations ranged from 425 m to 730 m (1400 ft. to 2400 ft.). The 100 year average monthly temperature for October to May as measured at Ajo, Arizona was 17.4 °C ranging from 23.3° C in October to a low of 11.7° C in January and increasing to a high of 24.72° C in May. Monthly average temperature difference from 100 year mean was 6° C with differences from the mean ranging 3.9° C to 7.7° C (1981-2012 Desert Research Institute 2012).

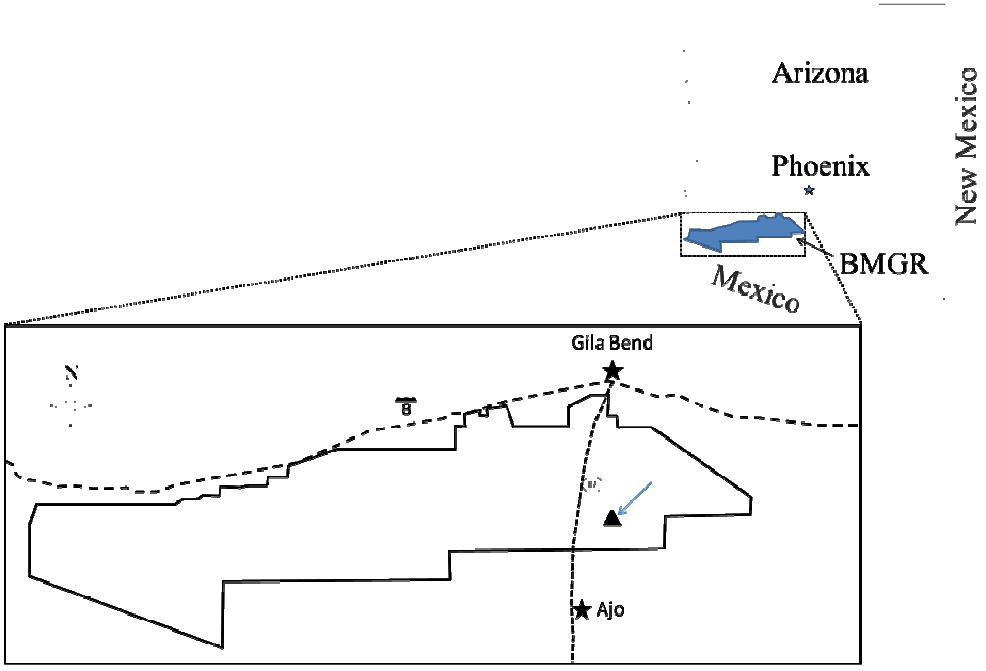


Figure 1. Study Area was located on Barry M, Goldwater Range (BMGR) in Maricopa County, Arizona, USA approximately 39 kilometers south of Gila Bend AZ, USA. The triangle represents the study area and five study sites within the Sauceda Mountains

Topography throughout the study site was characterized by large hills, valleys, and xeroriparian areas (ephemeral wash). The resulting mosaic of vegetation was characteristic of Arizona Upland and Lower Colorado subdivisions of the Sonoran Desert scrub vegetation community (Brown and Lowe 1980). Study sites within these two subdivisions with associated transitional zones and ephemeral washes supported multiple desert plant species. Common plant species present included creosote bush (*Larrea tridentata*), triangular bursage (*Ambrosia deltoidea*), yellow paloverde (*Parkinsonia microphylla*), saguaro cactus (*Carnegiea gigantean*), cholla (*Cylindroptunia* spp.), *Acacia* spp, and ocotillo (*Fouquieria splendens*). Additional species present in xeroriparian areas were thornbush (*Lycium* spp.), velvet mesquite (*Prosopis velutina*), desert ironwood (*Olneya testoa*), and desert honeysuckle (*Anisacanthus thuberi*).

METHODS

Mammal Trapping

To investigate differences in small mammal communities around AWS, I sampled areas adjacent to five AWS and sampled in the surrounding desert (referred to as control sites; CS). I sampled small mammals using capture mark recapture (CMR) trapping methods during winter 2011(October-January) and spring 2012 (February-May). AWS included four human-constructed sites and one modified natural site (tinaja) identified as 583, 584, 579, 580, and 675 respectively (Appendix D). Sites were selected based on presence of water and accessibility. For the purpose of this study, modified natural water sources were considered AWS (Appendix C).

Eight 135 m trapping transects were established at each AWS site. Four transects were randomly located within 50 m of the AWS radiating outward. Four CS transects were randomly placed between 500-700 m from the AWS (n=80 during winter 2011, n=80 during spring 2012). An overall sample size of 160 was the outcome of the sampling design. Transect direction was random and transects between and within treatment did not intersect. To ensure independence (i.e., prevent movement of animals between treatments), distance between transects in different treatments was ≥ 250 m. CS and AWS are defined as individual treatments with presence of AWS being the discriminating factor that separates these two treatment types. Random points were generated using ESRI ArcMAP 10 software. Secondary and tertiary random points were used if sites were deemed unsuitable (e.g., steep slopes) by ground reconnaissance.

Transects consisted of 10 Sherman traps (five traps of 8 x 9 x 23 cm and five traps of 8 x 9 x 33 cm) spaced 15 m apart alternating along the transect (*sensu*

Burkett and Thompson 1994, Pearson and Ruggerio 2003, Hopkins and Kennedy 2004). Traps were placed before sundown and baited with apple wafer pellets, a horse treat (Manna Pro, St. Louis, MO). Trap transects were operated for three consecutive nights per site. Polyester or cotton batting was placed in each trap during colder months to reduce exposure and minimize trap mortality. Traps were checked starting 1.5 hours before sunrise each of the three days unless environmental conditions warranted checking traps earlier.

Captured animals were identified to species (Hoffmeister 1986), aged (e.g., juvenile or adult), sexed, and weighed. Tail length and left hind foot length measurements were recorded along with other morphological features to assist in species identification. When species identification was uncertain it was classified to the Genus level (e.g., *Chaetodipus* spp.). Animals were individually marked with a metal ear tag (National Band and Tag Company, New Port, KY) in the left ear and/or with ink marks (Sharpie; two identical marks, one in each ear). Species with ears too small for ear tags (e.g., *Perognathus* spp.) only received ink marks. Captures with damaged ears were marked on the underside and base of the tail. Captured animals were released alive at the site of capture. Animals were handled and processed following Arizona State University (ASU) Institutional Animal Care and Use Committee (IACUC) protocol #09-1051R.

Vegetation Sampling

I quantified habitat characteristics along trapping transect with various techniques (Appendix F). I measured plant species richness, density, and cover to assess how habitat differed between AWS and CS and to relate habitat to the small mammal occurrence. Data were collected in the spring season. Two 4 x 25 m (0.01

ha) macro-plots were randomly located along each trapping transect to quantify plant density and plant species richness (Appendix F; Carpenter 1999). Shrub cover was recorded along the midline of the macro-plot using line intercept sampling techniques. I placed Daubenmire frames (20 x 50 mm) at five m intervals along the line intercept transect to collect herbaceous percent cover data. Grasses and forbs were identified as annual or perennial. Grasses and forbs data were divided into 12 cover classes (Appendix B; Elzinga et al. 2001). I identified shrubs and trees to species when possible or Genus when species could not be determined (Epple and Epple 1995). All measurements were recorded to the nearest cm. Macro-plot and transect locations were recorded using GPS.

Data Analyses

I calculated trap success as a measure of relative rodent abundance (hereafter, abundance) defined as the number of (marked) individuals captured per 100 trap nights. Species richness was summarized for each treatment (i.e. AWS and CS) by summing the total number of species captured on each transects per treatment. Individual biomass was calculated using the mean of biomass for individuals (e.g. if an individual was encountered more than once, mass was averaged). Total biomass was described at the transect-level by summing the individual biomass for all captures per species (here after referred to as biomass). Shannon (H') and Simpsons diversity indices were calculated to examine small mammal diversity between treatments. Because of the lack of data normality, I utilized a non-parametric analysis in the form of Multi-response Permutation Procedures (MRPP; Biondini et al. 1988) to investigate differences in rodent community factors (i.e. abundance, species richness, and biomass) in relation to

treatment. If a significant difference was detected (e.g. between seasons), MRPP was also used for pairwise comparisons. A Sidak correction was utilized to adjust for type I error across multiple tests (Abdi 2007).

I summarized habitat variation between treatments using PCA. Interpretation of habitat variable importance was based on variable weight for each factor. Selection of initial habitat variables for inclusion in the PCA was accomplished by excluding variables weighted less than 0.500. I used eigenvalues and scree plots to discriminate relative importance of factors. Factors with eigenvalues less than one were excluded from the model. PCA factor scores and habitat variables were compared between treatments using a Mann-Whitney Rank Sum Test (U; Table 4). I established relationships between the most common rodent species and habitat characteristics using multiple regression analysis. For less common species, I identified habitat characteristics that predicted occurrence of each species using logistic regression.

RESULTS

Small Mammals

I captured 370 individuals representing three genera and eight species of rodents across 4800 total trap nights. The most common species encountered was the desert pocket mouse (*Chaetodipus penicillatus*) having the highest number of captures in both season and treatment (Figure 2 and 3). Other species encountered were the rock pocket mouse (*C. intermedius*), Bailey's pocket mouse (*C. baileyi*), Merriam's kangaroo rat (*Dipodomys merriami*), white-throated woodrat (*Neotoma albigula*), cactus mouse(*Peromyscus eremicus*), Arizona pocket mouse(*Perognathus amplus*), and Harris's antelope squirrel (*Ammospermophilus harrisii*; Figure 2).

Arizona pocket mouse and Harris's antelope squirrel species were represented by one capture each. Cactus mouse, Arizona pocket mouse, and Harris's antelope squirrel were excluded from analyses due to limited capture but are included in the results of this study for comparison only. The remaining five species represented more than 90 percent of captures and were included in analyses.

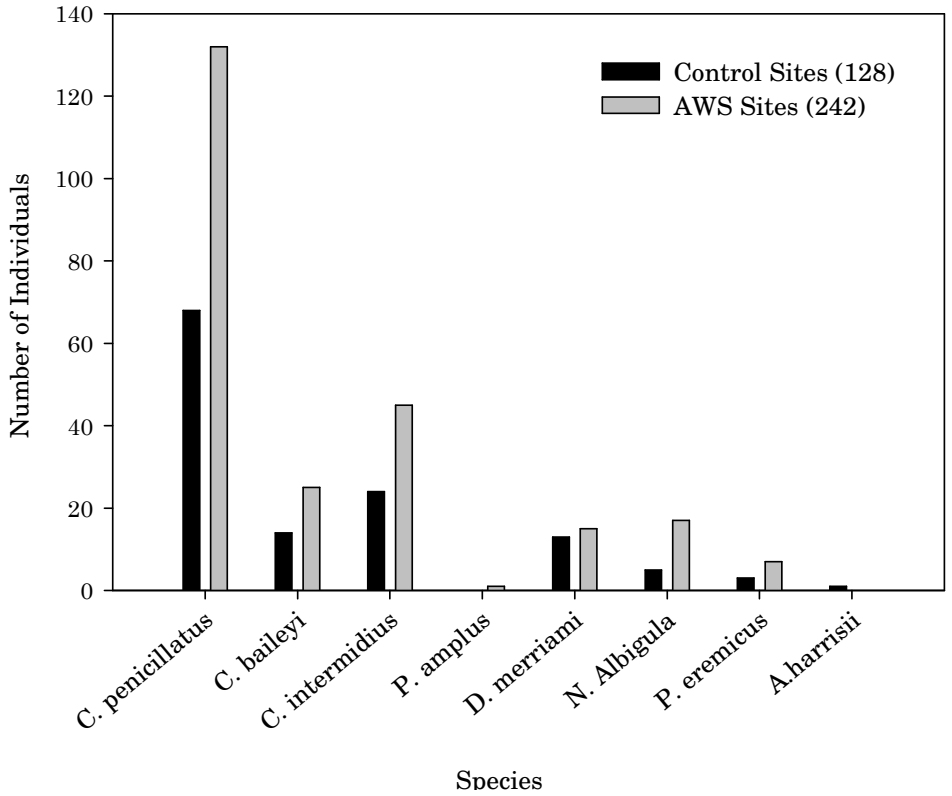


Figure 2. Total number of individuals captured by treatment by species recorded in winter 2011 and Spring 2012, at five Artificial Water Source(s) (AWS) and Control sites (CS) during trapping (4800 trap nights) performed on Barry M. Goldwater Range (BMGR) in Maricopa county, Arizona, USA. Total number of individuals captured per treatment is presented in brackets.

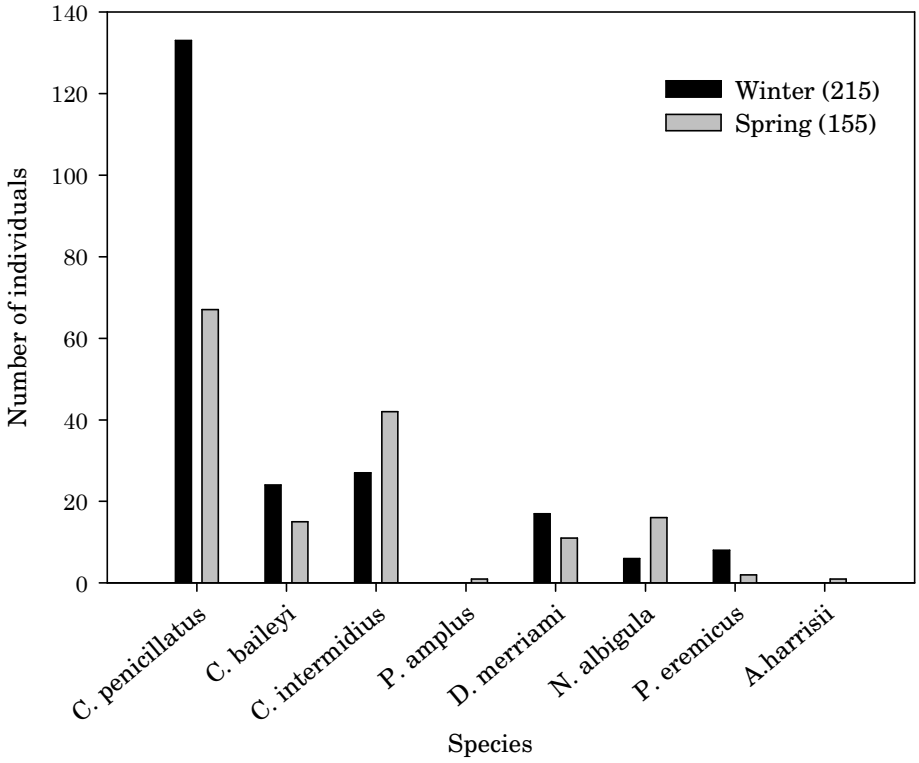


Figure 3. Total number of individuals captured by season by species recorded in winter 2011 and Spring 2012, at five Artificial Water Source(s) (AWS) and Control sites (CS) during trapping (4800 trap nights) performed on Barry M. Goldwater Range (BMGR) in Maricopa county, Arizona, USA. Total number of individuals captured per treatment is presented in brackets.

No significant difference in rodent community abundance, biomass, and richness was detected between seasons (MRPP $p>0.05$; Table 2). However, community abundance was significant at the $p= 0.054$ level. AWS and CS did significantly differ in rodent community abundance, biomass, and richness (MRPP, $p<0.001$). Rodent abundance was almost twice as high at AWS compared to CS (Table 1). Rodent species diversity between treatments was similar. Respectively, AWS and CS Shannon diversity index was 1.362 and 1.377 and Simpson diversity index was 2.859 and 2.971. The same numbers of species were encountered at both treatments. However species richness was significantly different between

treatments (Table 1). No Arizona pocket mouse or Harris's antelope squirrels were captured at CS and AWS, respectively (Figure 2).

Table 1. Mean (±SE) Rodent community variables by treatment during winter 2011 and Spring 2012 trapping, at five artificial water source (AWS) sites and control sites (CS) on Barry M. Goldwater Range in Maricopa County, Arizona, USA. Species richness included all species. Biomass measured in grams. Abundance is individuals captured/100 trap nights. Test statistics reported are for Multi-response Permutation Procedure (MRPP); n=160; α=0.05

Variable	AWS	CS	Statistic	P
Abundance	10.1 (0.6)	5.3 (0.7)	-9.506	<0.001
Biomass	89.6 (7.8)	40.4 (5.2)	-10.196	<0.001
Species Richness	1.6 (0.1)	0.9 (0.1)	-7.967	<0.001

Table 2. Mean (±SE) Rodent community variables by season during winter 2011 and Spring 2012 trapping, at five artificial water source (AWS) sites and control sites (CS) on Barry M. Goldwater Range in Maricopa County, Arizona, USA. Species richness included all species. Biomass measured in grams. Abundance is individuals captured/100 trap nights. Test statistics reported are for Multi-response Permutation Procedure (MRPP); n=160. α=0.05

Variable	Winter	Spring	Statistic	P
Abundance	8.9 (1.0)	6.4 (0.8)	-1.918	0.054
Biomass	63.9 (7.8)	62.4 (11.1)	-0.141	0.306
Species Richness	1.3 (0.1)	1.2 (0.1)	0.450	0.567

Desert pocket mouse abundance significantly differed between AWS and CS treatments (MRPP, $p<0.01$). Biomass of desert pocket mouse at AWS was nearly twice that of CS and biomass of white-throated woodrat was over five times greater at AWS treatments (Figure 4).

Table 3. Mean (±SE) Rodent Abundance (individuals captured/100 trap nights) by treatment by species during winter 2011 and spring 2012 at five artificial water source (AWS) sites and control sites (CS) on Barry M. Goldwater Range in Maricopa County, Arizona. USA. Test statistics reported are for Multi-response Permutation Procedure (MRPP); n=160. (5 tests, α=.05, Sidak correction=0.010)

Family Species	AWS	CS	Statistic	P
Heteromyidae				
Chaetodipus penicillatus	5.5 (0.8)	2.8 (0.5)	-4.143	0.007
Chaetodipus baileyi	1.0 (0.2)	0.6 (0.2)	-2.942	0.023
Chaetodipus intermedius	1.9 (0.4)	1.0 (0.5)	-1.193	0.106
Dipodomys merriami	0.6 (0.2)	0.5 (0.2)	0.348	0.490
Cricetidae				
Neotoma albigula	0.7 (0.2)	0.2 (0.1)	-2.918	0.022

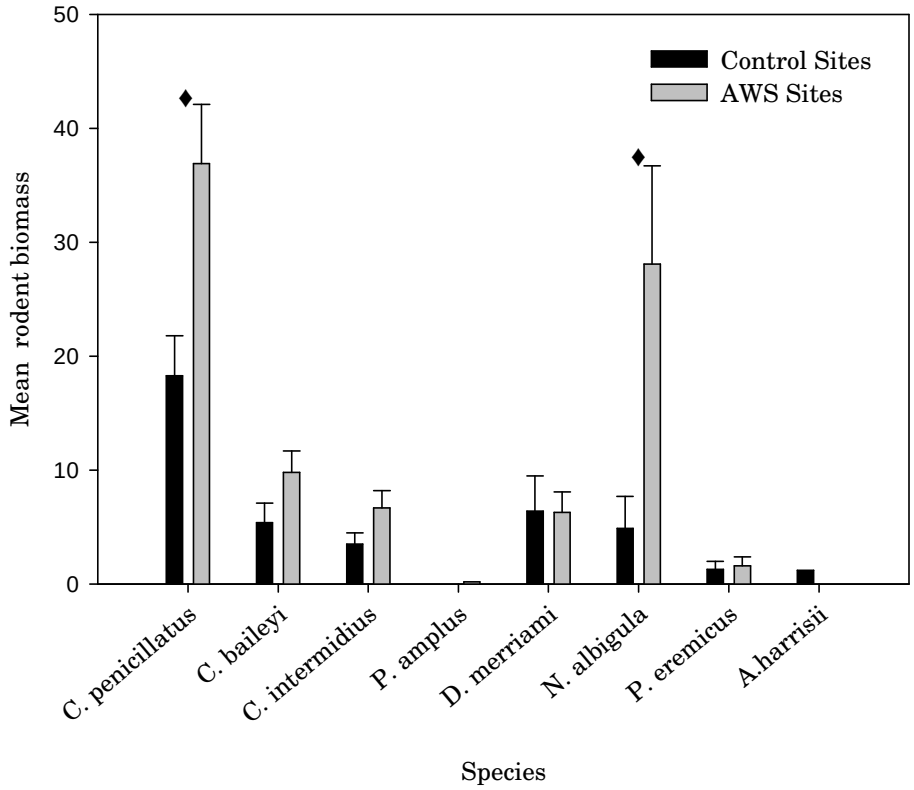


Figure 4. Total mean (±SE) rodent biomass by species by treatment winter 2011 and spring 2012 at Artificial Water Source (AWS) sites and Control sites (CS) on Barry M. Goldwater Range in Maricopa County, Arizona, USA. Biomass measured in grams. Test statistics reported are for Multi-response Permutation Procedure (MRPP); n=160. (5 tests, α=.05, Sidak correction=0.010), ♦=Significant

Habitat

Eleven habitat variables were reduced to five factors with eigenvalues greater than one (Table 4 and Appendix A). These five factors accounted for 91.1% of habitat variation between the two treatments (Appendix A). Habitat variables associated with presence of water (i.e. distance to AWS or distance to wash) did not explain a large percentage of variation and were not included in the final PCA. I assigned each factor a description based upon habitat variables that loaded high (>0.500) for each factor: factor 1, ground cover; factor 2, shrub cover; factor 3, tree overstory; factor 4, cactus density; and factor 5, shrub density (Appendix A). Treatments were similar in habitat characteristics and PCA factor scores (Table 4).

Small Mammals and Habitat Relationships

Of the five species of rodents included in habitat analyses, three species showed significant habitat relationships (Table 5). Bailey's pocket mouse occurrence was positively related to areas with higher cactus density. Merriam's kangaroo rat occurrence was negatively related to areas with high tree density and cover with high shrub density. Rock pocket mouse occurrence was negatively influenced by greater herbaceous ground cover.

Table 4 Mean (±SE) of habitat variables and Principal Component Analysis (PCA) factors measured on line transects and macro-plots at five artificial water sources (AWS) and control sites (CS) in spring 2012 on Barry M. Goldwater Range in Maricopa County, Arizona, USA. Test statistics reported Mann-Whitney Rank Sum Test (U); n = 80.

Habitat Variables	AWS	CS	Statistic (U)	P
Bare ground (% cover)	93.0 (0.5)	93.0 (0.7)	764.0	0.733
Herbaceous (% cover)	7.4 (0.4)	7.5 (0.6)	764.5	0.736
Forbs (% cover)	6.8 (0.5)	6.6 (0.7)	728.5	0.494
L. tridentata (% cover)	6.6 (1.1)	6.4 (0.8)	773.0	0.798
Shrub (% cover)	13.0 (1.1)	12.4 (1.0)	777.0	0.829
L. tridentate density/ha.	296.3 (34.7)	351.2 (41.7)	720.0	0.443
Tree density/ha.	190.0 (37.1)	142.5 (21.1)	761.0	0.708
Tree (% cover)	6.3 (1.2)	4.8 (0.8)	754.0	0.658
C. leptocaulis density/ha.	48.7 (15.6)	123.6 (44.8)	742.5	0.515
Cacti density/ha.	293.8 (34.0)	436.2 (71.7)	661.5	0.183
Shrub density/ha.	2025.0 (175.1)	2005.0 (184.6)	780.0	0.851
Factor Scores				
Ground Cover (F1)	-0.022 (0.13)	0.022 (0.18)	772.0	0.791
Shrub Cover (F2)	-0.003 (0.17)	0.003 (0.14)	753.0	0.655
Tree Overstory (F3)	0.176 (0.02)	-0.176 (0.11)	687.0	0.279
Cactus Density (F4)	-0.215 (0.08)	0.215 (0.20)	619.0	0.082
Shrub Density (F5)	0.007 (0.15)	-0.007 (0.16)	792.0	0.942

Table 5. Occurrence of selected common rodents species predicted by habitat characteristics from Principal Component Analysis (PCA) (Factors, F) using multiple linear regression (n=33) and logistic regression (n=80). Rodents were captured at five artificial water source(s) (AWS) and control sites (CS) in spring 2012 on the Barry M. Goldwater range, Maricopa County, Arizona, USA. Multiple linear regression used only habitat components of transects where the species were present. C = Correlation.

Species	C	Habitat	Statistic	P
Chaetodipus penicillatus[a]				NS
Chaetodipus baileyi[b]	+	Cactus Density (F4)	5.028	0.024 (82.5%)
Chaetodipus intermedius[b]	-	Ground Cover (F1)	7.064	0.008 (66.3%)
Dipodomys merriami[b]	-	Tree Overstory (F3)	12.201	0.002 (91.3%)
	-	Shrub Density (F5)		
Neotoma albigula[b]	+	Cactus Density (F4)	1.634	0.201 (86.3%)

[a] Multiple Regression
[b]Logistic Regression

DISCUSSION

On the Barry M. Goldwater Range, rodent abundance and biomass was higher at AWS compared to the surrounding Sonoran Desert. The rodent community was similar, between seasons; however diversity and species richness was greater at AWS compared to CS. For example, community biomass at AWS was two times greater than CS. The rodent community was dominated by generalist species, such as the desert pocket mouse, which was the most frequently captured species in my study. Within Sonora, Mexico the desert pocket mouse was one of the most abundant species of pocket mouse (Mantooth and Best 2005). In my study, the desert pocket mouse and white-throated woodrat had the greatest influence on community biomass. Rodent species occurrence was significantly related to habitat characteristics for most species.

Rodent communities can be influenced by local habitat characteristics and environmental conditions. For example, in Arizona, Reichman and Van De Graaff

(1973) found a correlation between rodent mass and species activity during winter for five Heteromyid rodents. Smaller heteromyids (i.e. rock pocket mouse) had low to no activity during winter months, whereas larger heteromyids (i.e. Merriam's kangaroo rat) were active throughout the winter. In California, Kenagy (1973) trapped the little pocket mouse (*Perognathus longimembris*) throughout winter in and observed the species to be active at -10° C. Activity of some rodents can also differ by location. For example, Desert pocket mouse in California is active all year; in Arizona activity is reduced during colder months. In my study, it is possible that low captures were due to cold conditions. However, species with low captures were excluded from analyses to avoid outliers. To reduce environmental influences on my trapping, nights with full moons were avoided when possible. To increase the range of species captured, I used two sizes of traps. Transects were used in lieu of a grid design to increase the area sampled by trapping. Trapping each location on four occasions (2 repetitions per season) for three consecutive nights each trapping session under similar conditions should reduce the effect of stochastic events (i.e. precipitation).

Habitat Characteristics

In general, habitat characteristics at AWS and CS were similar. The difference in habitat characteristics related to creosote bush and cactus densities could be the result of clearing when an AWS was initially installed or renovated as shrubs and cacti are regularly cleared or trans-located prior to AWS installation (R. Whittle, Barry M. Goldwater Range, per. comm.; Arizona Game and Fish Department 2008).

Small Mammals and Habitat Characteristics

Species habitat relationships from this study were consistent with findings of other research. The desert pocket mouse is considered a habitat generalist with an affinity for sandy soils and creosote bush. Some researchers have related desert pocket mouse occurrence with various plant species and general habitat characteristics (i.e. sandy soil and large bushes; Price 1984, Mantooth and Best 2005). Desert pocket mouse habitat use has shown to change in response to moonlight and temperature (Meyer and Valone 1999). In this study, habitat models for desert pocket mouse occurrence were inconclusive.

Merriam's kangaroo rat occurrence had a significant negative relationship to high shrub and tree density. This kangaroo rat prefers open habitat with few shrubs and trees (Rosenzweig and Winakur 1969, Cutler and Morrison 1998). In California, Stevens and Tello (2009) found that Merriam's kangaroo rat was positively correlated with creosote bush and open habitats. Along the Virgin River in Nevada and Arizona, Bateman and Ostoja (2012) found Merriam kangaroo rat to be negatively associated with riparian habitat consisting of shady saltcedar (*Tamarix* spp.) thickets.

In my study, Rock pocket mouse occurrence was negatively associated with higher amounts of herbaceous ground cover and low amounts of bare ground. This finding was consistent with other descriptions of habitat use, with rock pocket mouse preferring rocky soils, bare ground, and areas with limited herbaceous growth (Hoover et al. 1977, Hoffmeister 1986). Bailey's pocket mouse and white-throated woodrat occurrence were positively related to higher densities of cactus. However, only Bailey's pocket mouse relationship with cactus density was

significant. Brown et al. (1972) found that desert woodrat (*Neotoma lepida*) density was dependent on the presence of teddy bear cholla (*Cylindroptunia bigelovii*).

Small Mammals and Presence to Water

Rodent occurrence was not explained by water availability (distance to water) or distance to ephemeral water sources (e.g. arroyo or wash). Neither presence of a water source nor distance to water effectively explained variation in habitat characteristics. This is not unexpected as desert rodents, particularly the family Heteromyidae, have physiological adaptations (e.g., specialized kidneys, concentrated urine) and behavioral adaptations (e.g., torpor, burrowing and nocturnal activity) to minimize water loss and metabolize water from food (Kenagy 1973, MacMillen and Hinds 1983, Franks 1988). Similarly, white-throated woodrats are well equipped for survival in arid habitats by adaptations such as nocturnal, diurnal activity patterns in dens, and feeding on succulent fruits (i.e. cactus; Macedo and Mares 1988).

Small Mammals and Disturbance

Although not directly quantified during this study, soil disturbance and greater amounts of structure (i.e. construction debris, above ground tanks, rain collectors) were observed at AWS compared to the surrounding desert. Burrowing species of rodents (i.e. Merriam's kangaroo rat) favor disturbed soils with better burrowing conditions (Schmidly et al. 1988). Burkett and Thompson (1994) suggested a relationship between soils and higher rodent abundance around AWS. Breck and Jenkins (1997) suggested that Merriam's kangaroo rat were associated with sandy or loose soils because burrow and mound construction could have a lower energetic cost in disturbed soil. Burkett and Thompson (1994) suggested that debris

and structure in the vicinity of AWS provided additional habitat for rodent species as a possible explanation for higher abundances in the vicinity of AWS.

Conclusion

Significant differences were detected in desert pocket mouse abundance and biomass among AWS and CS treatments. White-throated woodrat showed only significant difference in total mean biomass between treatments. No other species exhibited significant difference in relation to the presence of AWS. It is important to note that other studies investigating small mammal communities in the vicinity of AWS document mixed responses. Some researchers found that rodent abundances were higher at AWS when compared to areas without (Burkett and Thompson 1994); whereas, Cutler and Morrison (1998) observed no difference in abundance at AWS. Current research has had mixed results pertaining to the effects of AWS on rodent and small mammal communities. Earlier research has shown that desert rodent communities are well adapted to arid environments and are not dependent on "free water". Additional research is needed to further investigate the effects of AWS on rodent communities and should be expanded to include other wildlife taxa. Attention is needed to determine if the presence of water or changes in habitat possibly influence abundance of certain species (i.e. desert pocket mouse) more than others. Long term studies of wildlife populations and communities before and after the installation of AWS are needed to fully explore the effect of AWS wildlife to include small mammals

Management Implications

The use of water as a management tool for endangered or game species is popular and has increased in recent history. Even with debate about its effectiveness as a management tool state, federal, and private agencies have made significant allocation of resources to install and maintain AWS. In 1995, BLM estimated that 5.65 million dollars was needed to maintain and develop water projects in the west for desert bighorn sheep (Broyles 1997a). Arizona spends around one million dollars a year to on AWS (Broyles 1997a). Wildland-urban interface is growing and in the last 20 years has raised concerns about water resources, habitat loss, and invasive species (Theobald and Romme 2007). Resource managers must make informed decisions to direct activities to maintain current management or promote alternatives to current policies along with allocating resources. The results of this study will help state and federal agencies and their resource managers make informed management decisions.

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APPENDIX A

ROTATED PRINCIPAL COMPONENT ANALYSIS

Appendix A. Rotated principal component analysis (PCA) of habitat characteristics quantified along mammal trapping transects located at five artificial water sources (AWS) and control sites (CS) on Barry M. Goldwater Range in Maricopa County, Arizona, USA. Selections of initial habitat variables were selected for inclusion in the PCA by factor weight (> 0.500). Interpretation of factors was based on variables having a high weight for each factor (bolded values)

	Factor				
Habitat Characteristics	F1	F2	F3	F4	F5
Bare ground (% cover)	**-0.984**	0.003	-0.090	-0.041	0.114
Herbaceous (% cover)	**0.984**	-0.001	0.089	0.040	-0.114
Forbs (% cover)	**0.982**	-0.017	0.094	-0.025	-0.043
L. tridentate (% cover)	0.003	**0.912**	-0.123	-0.106	-0.253
Shrub (% cover)	-0.122	**0.890**	0.030	-0.091	0.318
L. tridentate density/ha.	0.196	**0.613**	-0.473	-0.224	-0.197
Tree density/ha.	0.155	-0.066	**0.892**	0.111	-0.055
Tree (% cover)	0.112	-0.093	**0.883**	0.045	0.108
C. leptocaulis density/ha.	0.003	-0.044	0.124	**0.947**	-0.175
Cacti density/ha.	0.053	-0.241	0.071	**0.897**	0.253
Shrub density/ha.	-0.187	-0.035	0.064	0.027	**0.958**
Eigenvalue	3.03	2.07	1.86	1.79	1.26
% Variation explained	27.55	18.85	16.94	16.28	11.45
Cumulative % Variation	27.55	46.40	63.34	79.62	91.07

APPENDIX B

MODIFIED DAUBENMIRE COVER CLASSES

Appendix B. Modified Daubenmire cover classes used for herbaceous and bare ground percent cover measurements.

Cover Class	Percent Cover	Percent Cover Midpoint
1	<1	0.5
2	1-5	3.0
3	6-15	10.5
4	16-25	20.5
5	26-35	30.5
6	36-45	40.5
7	46-55	50.5
8	56-65	60.5
9	66-75	70.5
10	76-85	80.5
11	86-95	90.5
12	>95	98.0

APPENDIX C

ARTIFICIAL WATER SOURCE LOCATIONS

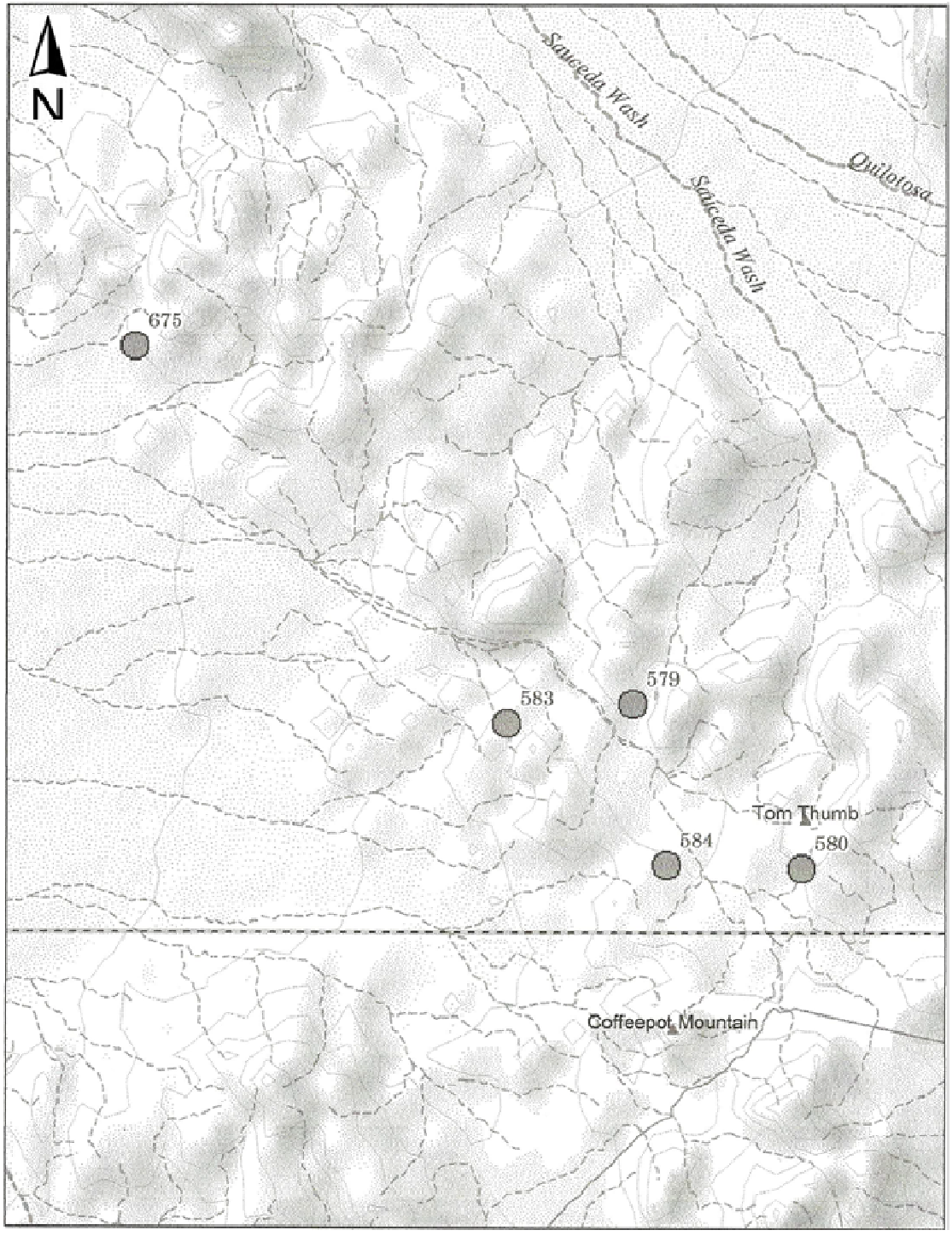
Appendix C. Locations and attributes for five artificial water sources on Barry M. Goldwater Range in Maricopa County, AZ, USA. UTMs are in NAD83 datum.

Site #	UTM	Installed/Renovated	Type
579	12S 346472 3603849	1961/1995 minor	Artificial
580	12S 350926 3599343	1961/2008 major	Artificial
583	12S 343051 3603364	1961/2008 major	Artificial
584	12S 347326 3599470	1961	Artificial
675	12S 333159 3613740	1965	Modified Tinaja

APPENDIX D

MAP OF ARTIFICIAL WATER SOURCE LOCATIONS

Appendix D. Map of artificial water source (AWS) locations on Barry M, Goldwater Range (BMGR) in Maricopa County, Arizona, USA approximately 39 kilometers south of Gila Bend AZ, USA. Circle represents location of AWS. Produced using Delorme Topo USA 8 software. Distance between 583 and 579 is approximately 3.5 km. Map represents an area of approximately 800 square km.



APPENDIX E

HABITAT METHODS AND VARIABLES

Appendix E. Description of methods used to quantify habitat variables used to describe rodent abundance and occurrence. Variables were quantified along 80 transects established at five artificial water source (AWS) and control sites (CS), on Barry M. Goldwater Range (BMGR) in Maricopa County, Arizona, USA. Data were collected within two 4 x 25 m macro-plots established along each trapping transects.

Method	Method Description
Line-Intercept	I used the line-intercept method to measure % cover of various variables and plant species. The raw data was collected in meters, to the closest cm. I recorded the start and end of each variable (intercept), described below, along the midline of a macro-plot totaling 25 m per macro-plot; 50 m per trapping transect. To calculate the % cover, I added all the total line intercept for one variable along the whole transect and converted the total to a percentage; total intercept in meters/25 m*100 . Then averaged percentages from both macro plots.
Variables	*Variable Description*
Shrub (% cover)	Variable quantified % cover for vegetation having a growth form of shrub, sub shrub, or shrub tree growth habit as defined by USDA and NRCS 2011
Tree (% cover)	Variable quantified % cover for vegetation having a growth form of tree shrub or tree as defined by USDA and NRCS 2011 excluding Cactaceae and ocotillo.
Cactus (% cover)	Variable quantified % cover for vegetation belonging to the Cactaceae family (Epple and Epple 1995) and including ocotillo

Method	Method Description
Daubenmire plot	Daubenmire plots (Daubenmire 1959) were used to quantify the vegetation qualified as herbaceous. I quantified 5 Daubenmire plots per macro-plot; 10 per trapping transect. I set the Daubenmire plots along the 25-m line used for the line-intercept method. They were located at point 0, 5, 10, 15, 20 and 25 m, on left side of the line when facing the line from the starting point. I used the a12 class modified Daubenmire scale to determine class cover (Appendix D). To determine

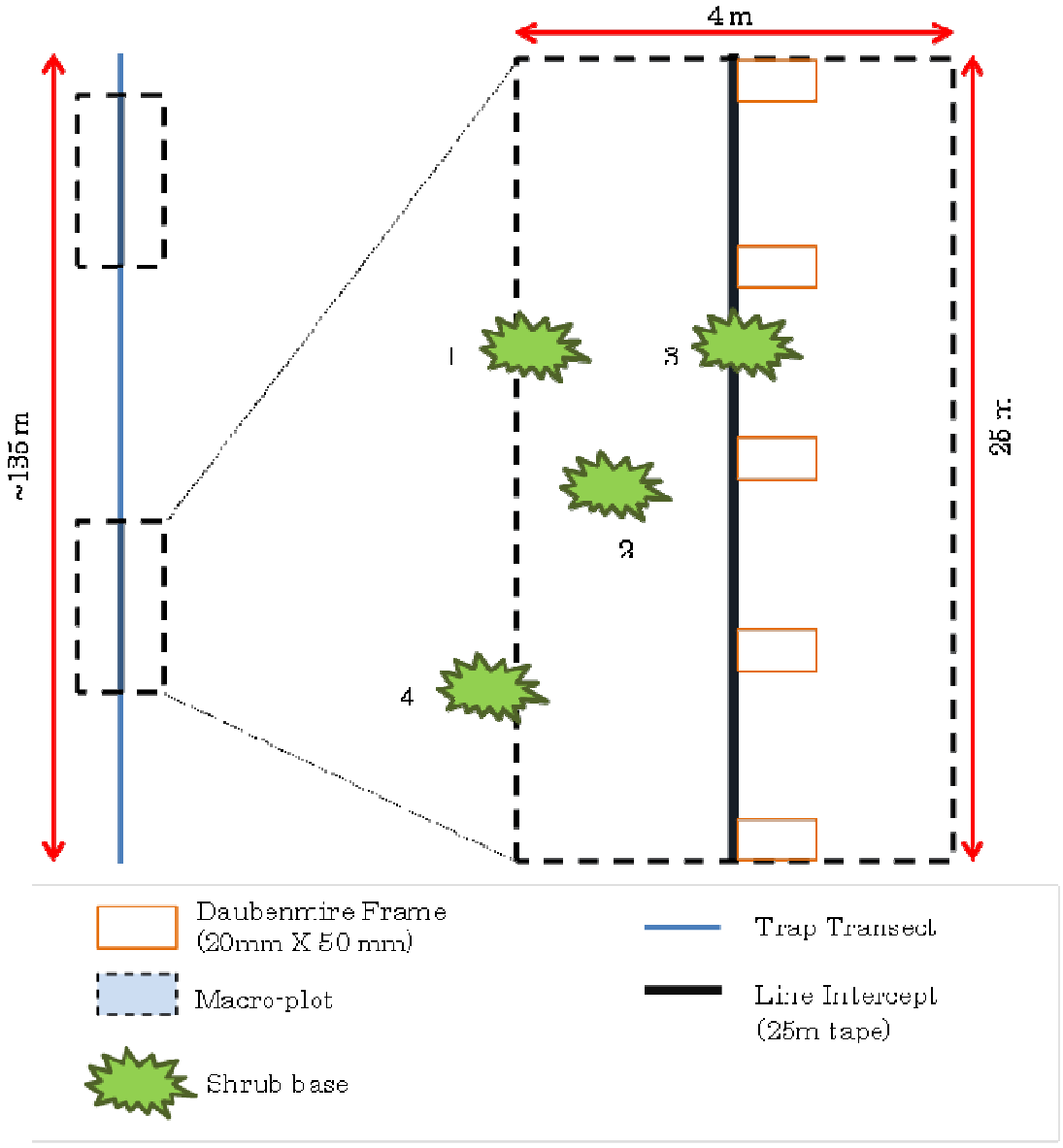
	the herbaceous % cover for a macro-plot, I added all the middle values from the class cover scale for each macro-plot and divided by 6. I then added the mean % cover values from each macro-plot and divided by 2.
Variable	*Variable Description*
Herbaceous (% cover)	Herbaceous variable included all forbs and grasses. Grasses and forbs were identified as either annual or perennial.

Method	Method Description
Macro-plot	I used a macro-plot method to quantify density/ha of various variables and plant species. Raw data was collected from a 4 m by 25 m macro-plot (2 m on either side of transect) centered on a trapping transect (2 per trapping transect). I recorded counts for every species within the macro-plot. To determine mean density I added up total number of each variable in the macro-plots and multiplied by 100 to estimate density /ha. I then added the estimated density from both macro-plots and divided by 2 to get a mean density/ha.
Variable	*Variable Description*
Shrub density/ha	Variable quantified density/ha for vegetation having a growth form of shrub, sub shrub, or shrub tree growth habit as defined by USDA and NRCS 2011 excluding cacti and ocotillo.
Tree density/ha	Variable quantified density/ha for vegetation having a growth form of tree shrub or tree as defined by USDA and NRCS 2011 excluding cacti.
Cactus density/ha	Variable quantified density/ha for vegetation belonging to the Cactaceae family (Epple and Epple 1995) and including ocotillo

APPENDIX F

HABITAT SAMPLING DIAGRAM

Appendix F. Habitat Sampling Diagram depicting placement of sampling methods relative to trapping transects. Two macro-plots were randomly located along trapping transects. Quantification of habitat was performed within each macro-plot. Shrub and tree cover estimates were performed along the midline of macro-plot using 25m tape by line intercept methods. Herbaceous cover data was collected using six Daubenmire frames placed every five m along the midline of macro-plot. Plant richness and density estimates were performed inside the macro-plot. Plants whose base was ≥ 50 percent within the macro-plot were considered in the macro-plot (1-3). Plants with bases < 50 percent in the plot were considered outside the plot.



APPENDIX G

GENERAL MORPHOLOCIAL CHARACTERISTICS

Appendix G. Mean (±SE) characteristics of species captured in Sherman traps on Barry M. Goldwater Range in Maricopa County, Arizona, USA. Study conducted during winter 2011 and spring 2012. Some individuals were not measured for all morphometrics (number measured in each category given by N). Other species encountered were in traps or seen in the field. Blank N fields represent ubiquitous nature of the species at the time or place of sighting (e.g. multiple sighting).

Family Species	N	Body Mass (g)	Hind foot Length (mm)	Head Body Length(mm)	Tail Length (mm)
Heteromyidae					
Dipodomys merriami	38/37/32/37	36.6 (0.9)	35.6 (0.3)	88.7 (1.1)	143.1 (1.2)
C. penicillatus	101/102/89/102	22.5 (0.3)	24.5 (0.1)	73.6 (0.6)	109.7 (1.5)
C.intermedius	77/79/68/79	12.8 (0.2)	19.3 (0.2)	63.7 (0.8)	9.8 (1.6)
C.baileyi	39/39/36/38	31.5 (0.7)	26.7 (0.1)	81.2 (0.7)	117.0 (2.4)
Cricetidae					
Peromyscus eremicus	9/10/10/10	18.0 (0.8)	19.0 (0.5)	71.6 (1.5)	102.0 (4.3)
Perognathus amplus	1	12.0	20.0	62.0	82.0
Neotoma albigula	18/21/20/21	146.9 (9.4)	30.8 (0.3)	144.8 (4.1)	146.6 (3.2)
Sciuridae					
Ammospermophilus harrisii	1	99.0	36.0	120.0	80.0
Other Species Encountered					
Sylvilagus audubonii	1	120.0	46.0	>140.0	25.0
Spilogale gracilis	1				
Crotalus cerastes	1				
Crotalus molossus	1				
Crotalus atrox	1				
Trimorphodon biscutatus	1				
Phainopepla nitens					
Wilsonia pusilla					

APPENDIX H

PICTURES OF ARTIFICIAL WATER SOURCES AND SURROUNDING AREA

Appendix H. Pictures: 1. Artificial water source guzzler for 584. 2. Artificial water source 675; modified tinaja configuration. 3. Renovated artificial water source 583 after renovation (2008). 4. Artificial water source 584 with above ground tank and rain collection apron (1961 design). 5. Typical wash habitat 675. 6. Elevated view of AWS 579 with rain collection apron, sunken tank, and tub guzzler. 7. *Chaetodipus intermedius* after release. 8. *Neotoma albigula* with ear tag. 9. Jonathon Quinsey, Field technician on the job.

Picture 1.



Picture 2.



Picture 3.



Picture 4.



Picture 5.



Picture 6.



Picture 7.



Picture 8.



Picture 9.

