

EFFECTS OF SEASONAL PRECIPITATION AND HABITAT
ASSOCIATIONS ON THE DEMOGRAPHICS OF MEXICAN SPOTTED
OWL PREY IN THE CANYONLANDS REGION OF SOUTHERN UTAH

by

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ABSTRACT

Most food habit studies of Mexican Spotted Owls (*Strix occidentalis lucida*) have been conducted in forested environments in more southern portions of their range. Through regurgitated pellet analyses, these studies showed the majority of Mexican Spotted Owl prey consumed was comprised of *Cricetid* rodents, specifically woodrats (*Neotoma* spp.) and white-footed mice (*Peromyscus* spp.). In the northernmost portions of their range, Mexican Spotted Owls inhabit rocky canyon habitats within the Colorado Plateau. In the canyonlands region, few studies have investigated the population ecology and habitat associations of the primary prey of spotted owls and no studies have examined the relationships among primary prey demographics and their responses to seasonal precipitation. Given the Mexican Spotted Owls status as a threatened species, increased knowledge of prey species relationships with climate and habitat may assist in future management of spotted owl populations across the canyonlands region. Using a seven-year historic data set collected at three study sites in Grand Staircase – Escalante National Monument from 2001 to 2007 and three years of data collected at five study sites in Capitol Reef National Park from 2013 to 2015, I described the nocturnal small mammal communities, investigated primary prey habitat and microhabitat component associations, and investigated the effects of timing and amount of seasonal precipitation on primary prey abundance and diversity in both study areas. *Cricetid* rodents were the most abundant nocturnal small mammals and potential prey available for spotted owls inhabiting rocky canyon habitats. Microhabitat analyses revealed *Cricetid* rodents partitioned space and resources that minimized interspecific competition enabling coexistence in narrow canyon systems with limited biological resources. Linear mixed-effects modeling indicated winter precipitation was the primary driver of spotted owl primary prey demographics in the canyonlands region.

CHAPTER ONE

INTRODUCTION TO THESIS

The Mexican Spotted Owl (*Strix occidentalis lucida*) is one of three sub species of spotted owl inhabiting North America. Distribution of Mexican Spotted Owls (Figure 1.1) extends from central Mexico into northeastern Utah, with *lucida* geographically and genetically isolated from the Northern (*S.o. caurina*) and California (*S.o. occidentalis*) subspecies (Barrowclough and Gutiérrez 1990). Mexican Spotted Owls (MSO) were first observed in the early 1900's in riparian areas along the Rio Grande River in New Mexico and the San Pedro River in Arizona and were observed as early as 1928 in southern Utah (USDI 2012). However, MSOs have not been observed in those historic sites in recent decades due to suspected habitat alteration (Hayward et al. 1976, USDI 2012). The Northern Spotted Owl was the focus of national attention in the 1980's because of habitat loss due to old growth logging within forests of the Pacific Northwest. This resulted in the U.S. Fish and Wildlife Service (USFWS) listing the Northern Spotted Owl as "threatened" under the Endangered Species Act in 1990 (USDI 2011). Resource managers observed similar reductions in distribution of mature forests in the Southwest U.S. due in part to wildfires and timber-management practices, and the USFWS listed the MSO as threatened in 1993 (USDI 2012).

Rocky canyonland habitats within the Colorado Plateau of Utah (Figure 1.2) were assumed to be the northern most extent of MSO's range because fragmented narrow canyon watersheds, along with arid climate and sparse vegetation communities did not

resemble classical forest habitats used by conspecifics in the Pacific Northwest, Arizona, and New Mexico (Forsman et al. 1984, Ganey and Balda 1989). Initially, researchers surmised MSOs would be strongly associated with forested environments of the Colorado Plateau like conspecifics in the Pacific Northwest (Ganey and Balda 1989; May et al. 2004). However, initial research showed narrow canyon habitats across the Colorado Plateau did support small and isolated populations of MSOs. Kertell (1977) identified six MSO territories in Zion National Park and noted all territories were in narrow, steep-walled canyons that supported relatively thick mixed-conifer vegetation. Wagner et al. (1982) observed MSOs in the 1970's occupying narrow sandstone canyons, with ephemeral water, and riparian vegetation in Capitol Reef National Park. More intensive research in the canyonlands region began in the 1990's with broad surveys across habitat types (Rinkevich and Gutiérrez 1996, Willey 1998), investigations of juvenile dispersal (Willey 1998, Willey and van Riper 2000), home range characteristics (Willey 1998, Willey and van Riper 2007, Bowden 2008), territorial occupancy (Hockenbary 2011), and food habits (Ganey 1992, Rinkevich 1996, Willey 2013). Results of these studies increased the number of known MSO territories from 28 in 1996 to over 200 across Utah in 2010 (Willey and Willey 2010).

Mexican Spotted Owl diet studies have shown *Cricetid* rodents, particularly woodrats (*Neotoma* spp.) and white-footed mice (*Peromyscus* spp.) comprised the greatest proportion of prey recorded from regurgitated pellets analyses (Ganey 1992, Ward and Gutiérrez 1998, Sureda and Morison 1998, Block et al. 2005). Willey (2013) showed woodrats and white-footed mice comprised up to 90% of overall prey biomass in

dissected pellets of MSOs. Prey species population dynamics have been shown to be associated with spotted owl habitat selection, survival rates, and productivity (Forsman 1984, Zabel et al. 1995, Ward and Gutiérrez 1998, USDI 2012). Ward et al. (1998) concluded that availability and distribution of primary prey species was positively correlated to Northern Spotted Owl nest success and habitat selection in the Pacific Northwest. However, other than Sureda and Morrison (1998) and Willey and Willey (2010), little is known about the population ecology of the primary prey of MSOs in canyon environments within the Colorado Plateau. Furthermore, no studies have examined the relationships among *Cricetid* rodent abundance, habitat associations, and variation in annual precipitation in the canyonlands region of southern Utah. Given MSOs' status as a threatened species, increased knowledge of their prey population's demographics and climate and habitat drivers may help explain recent MSO territorial abandonment and recolonization patterns and generate management guidelines concerning future potential site extirpation or recolonization events in the region. With this impetus, I conducted an intensive analysis of relationships among climate, habitat, and prey species of MSOs from field data collected by Willey and Willey (2010) from 2001 – 2007 in Grand Staircase – Escalante National Monument (GSENM) and field investigations I conducted from 2013 – 2015 in Capitol Reef National Park (CARE) and 2015 in GSENM (Figures 1.2 and 1.3).

Objectives of this study were to 1) describe nocturnal rodent communities at three study sites in GSENM and five study sites in CARE; 2) investigate influences of seasonal precipitation and habitat characteristics on woodrat and white-footed mouse abundance;

and 3) investigate if variation in woodrat and white-footed mouse abundance explained MSO site-occupancy patterns at a sample of historic nest sites in the study areas.

Field work was conducted in GSENM and CARE in southern Utah (Figures 1.2 and 1.3). These areas may provide important habitat for subpopulations of MSOs augmented by dispersal from large source populations located in Zion and Canyonlands National parks (Figure 1.2; Willey 1998, Willey and van Riper 2000). Originally comprising 764,856 ha, GSENM was established in September 1996 and was the largest National Monument in the contiguous United States (USDI 2014). However, President Donald Trump signed Presidential Proclamation 9682 modifying the boundaries of GSENM on 4 December 2017. Currently, the Bureau of Land Management is redefining the monuments boundaries which has been significantly reduced (USDI 2018). Comprising 98,711 ha, CARE was established as a National Monument in August 1937, and became a National Park in December 1971 (USDI 2015). GSENM and CARE are managed by the Bureau of Land Management and National Park Service respectively and study areas were within 100 km of each other.



Figure 1.1. Distribution of three subspecies of spotted owl in North America (USDI 2012).

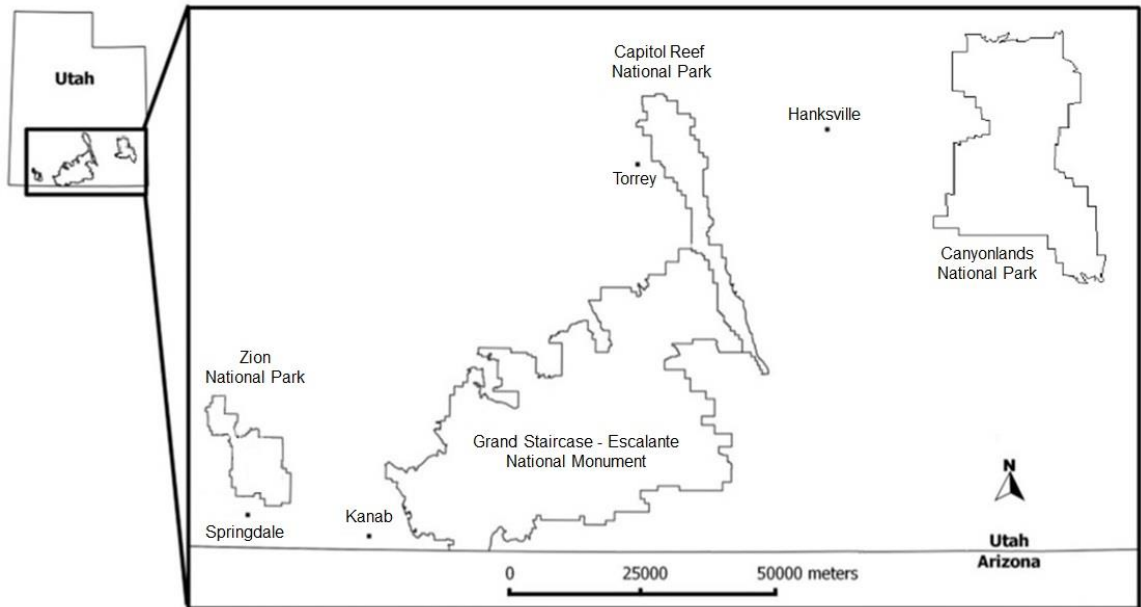


Figure 1.2. Grand Staircase – Escalante National Monument (original boundary) and Capitol Reef National Park study areas relative to two potential source populations of Mexican Spotted Owls in Zion and Canyonlands National Parks, southern Utah. Modified from Hockenbary (2011).

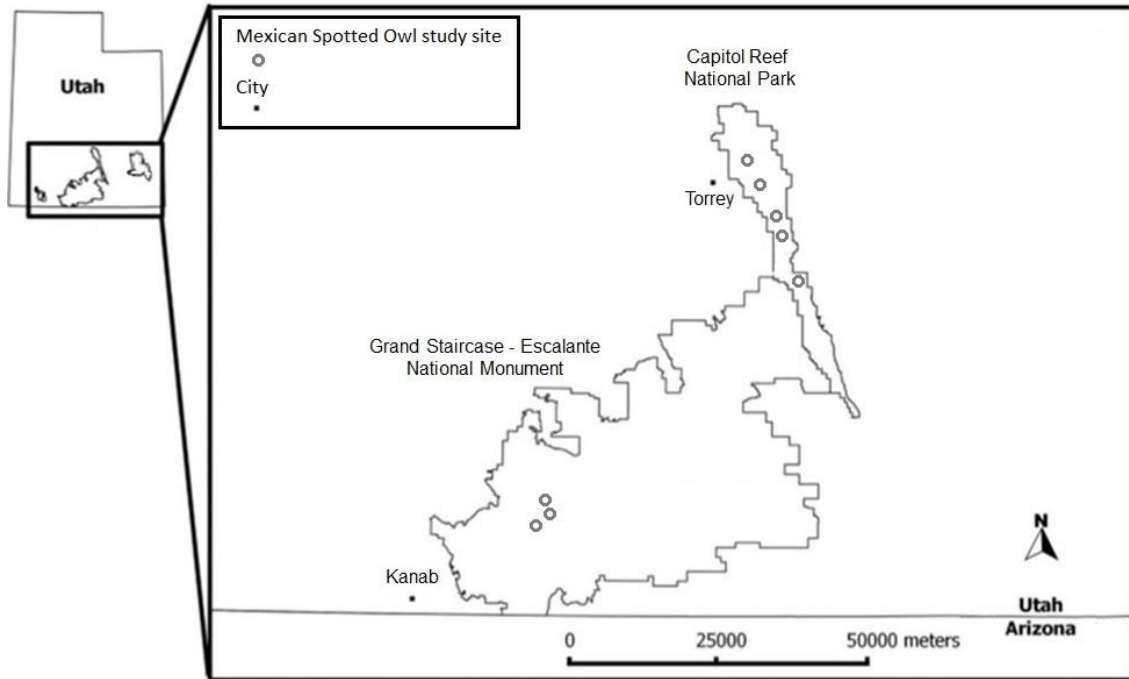


Figure 1.3. Grand Staircase – Escalante National Monument (original boundary) and Capitol Reef National Park Mexican Spotted Owl study areas, southern Utah. Open circles indicate approximate locations of study sites. Modified from Hockenbary (2011).

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CHAPTER TWO

DRIVERS OF SMALL MAMMAL DEMOGRAPHICS IN THE CANYONLANDS
REGION OF SOUTHERN UTAH: INVESTIGATING HABITAT ASSOCIATIONS
AND INFLUENCE OF SEASONAL PRECIPITATIONAbstract

In the northernmost portions of their range, Mexican Spotted Owls (*Strix occidentalis lucida*) inhabit rocky canyon habitats within the Colorado Plateau. Few studies have investigated the population ecology and habitat associations of the primary prey of spotted owls inhabiting rocky canyon environments and no studies have examined the relationships among *Cricetid* rodent demographics and their responses to seasonal precipitation within this region. Given the Mexican Spotted Owls status as a threatened species, increased knowledge of prey species relationships with climate and habitat may assist in future management of spotted owl populations across the canyonlands region. Over ten years of field work and 35,650 trap nights of capture effort across three study sites in Grand Staircase – Escalante National Monument (2001-2007 and 2015) and five study sites in Capitol Reef National Park (2013-2015), *Cricetid* rodents accounted for approximately 90% of all captures indicating they were the most abundant nocturnal small mammal and potential prey available for spotted owls inhabiting rocky canyon habitats. Microhabitat analyses showed *Cricetid* rodents exhibited spatial and resource separation that may have minimized interspecific competition enabling coexistence. *Cricetid* rodents were strongly ($p < 0.001$) associated with greater vegetative cover provided by trees and shrubs and evidence suggested increased vegetation provided by larger trees on moderate (20°) slopes in riparian areas enabled primary prey species to endure periods of drought. Winter precipitation was the primary driver of *Cricetid* rodent community demographics and explained significantly ($p < 0.001$) more variation in *Cricetid* rodent abundance compared to annual and summer precipitation. Results suggest increased primary prey abundance followed greater winter precipitation. This may have implications for potential spotted owl recolonization of abandoned territories. Mexican Spotted Owl management plans should incorporate management strategies for spotted owl primary prey species by protecting narrow canyon habitats with forested riparian corridors from disturbance.

Introduction

In the southwestern U.S., most studies of desert rodent communities have been conducted in the Mojave Desert (Beatley 1969, 1974, Brown 1973), Great Basin Desert (Beatley 1976, Hamilton et al. 2015), Sonoran Desert (Rosenzweig and Winakur 1969, Reichman and Graaff 1975), and Chihuahuan Desert (Guo and Brown 1996, Ernest et al. 2000, Brown and Ernest 2002). These studies recognized precipitation as the primary limiting resource in desert ecosystems and showed timing and amount of precipitation directly influenced primary productivity of desert plant communities which strongly affected desert rodent population dynamics (Beatley 1976, Thibault et al. 2010). Deserts occur on all continents and rainfall in desert ecosystems is discontinuous, highly variable, and unpredictable from year to year (Noy-Meir 1973, Kelt 2011). Annual precipitation in North American desert ecosystems falls primarily during two distinct seasons, summer and winter. North American summer monsoons are random thunderstorm events that can deliver heavy localized rainfall over minutes to hours. Water infiltrates soil poorly during monsoons, plant root uptake is low, and surface runoff and evaporation are high (Noy-Meir 1973, Loik et al. 2004). These random thunderstorms initiate quick pulses of production in drought resistant flora typified by quick phenological reaction times and shallow root systems (Schwinning and Sala 2004, Loik et al. 2004). Summer monsoons are typically short-lived, spatially variable rainfall, followed by long periods of drought. In contrast, winter precipitation is characterized by frontal storm systems which originate over the Pacific Ocean and travel east across the southwestern United States delivering consistent regional precipitation over days to weeks (Ernest et al. 2000, Brown and

Ernest 2002). Consistent winter precipitation delivered by regional rainfall and snow allows water to infiltrate the soil profile more effectively than monsoons, be absorbed by plant roots more efficiently, and is primarily responsible for germination and reproductive success of desert plant communities in the spring (Beatley 1969, 1974, 1976, Noy-Meir 1973, Loik et al. 2004). Consequently, due to the dynamic and unpredictable patterns of precipitation during both winter and summer seasons, desert ecosystems experience pulses of biological resources resulting in periods of high and low species abundance and productivity across trophic levels (Schwinning and Sala 2004).

Studies show increased biological resources associated with seasonal precipitation directly influenced population dynamics of desert plant and rodent communities. Over a 13-year study in the Mohave Desert, Beatley (1969, 1974) observed the rainfall regime varied greatly across individual study sites and from year to year, noting in no two years was the rainfall regime similar. Beatley (1974) concluded timing and quantity of seasonal precipitation, not total annual precipitation, was the critical parameter for desert plant and rodent abundance each season. Beatley (1976) and Thibault et al. (2010) identified significant positive relationships between timing and magnitude of winter precipitation events on successful reproduction of desert primary producers and consumers (i.e., *Heteromyid* rodents). In addition, Thibault et al. (2010) observed peak small mammal abundance and increased shrub density followed three consecutive years of above average winter precipitation, while small mammal abundance reached its minimum following three consecutive years of low annual precipitation < 90 mm (Thibault et al. 2010).

Abundance and availability of primary prey species was directly correlated to habitat selection and reproductive success of both Northern Spotted Owls (*S.o. caurina*) and California Spotted Owls (*S.o. California*) in the Pacific Northwest, as well as *lucida* in the southwest demonstrating the importance of small mammals to the general ecology of all three spotted owl subspecies (Smith et al. 1999, Seamans et al. 2002, and Block et al. 2005). Prey species composition and population dynamics can influence spotted owl habitat selection, survival rates, and productivity (Zabel et al. 1995, Ward and Gutiérrez 1998, USDI 2012). Mexican Spotted Owl (MSO) diet studies have shown *Cricetid* rodents, particularly woodrats (*Neotoma* spp.) and white-footed mice (*Peromyscus* spp.) comprised the greatest percentage of prey recorded from regurgitated pellets analyses (Ganey 1992, Ward and Gutiérrez 1998, Sureda and Morrison 1998, Block et al. 2005). Willey (2013) showed woodrats and white-footed mice comprised up to 90% of overall prey biomass in MSO prey remains in dissected pellets. Within the canyonlands region in southern Utah, Sureda and Morrison (1998) and Willey and Willey (2010) showed *Cricetid* rodents were the most abundant nocturnal small mammals inhabiting narrow slot canyon environments.

Working across nine owl territories in Grand Staircase – Escalante National Monument (GSENM), Willey and Willey (2010) observed MSO occupancy and small mammal abundance were positively correlated with precipitation during the breeding season. They observed low owl fecundity in years with low precipitation and low rodent abundance during May through July and high owl fecundity in years with high precipitation and rodent abundance. In Capitol Reef National Park (CARE), Willey

(1998) documented presence of ten pairs of MSOs within narrow canyon habitats. However, by summer 2014 only one pair of owls was observed and single males were detected at only two other sites suggesting numerous site extirpations had occurred throughout the Park. Willey and Willey (2010) documented a similar loss of nesting territories adjacent to CARE along the Paria River in GSENM, and suggested prolonged drought, linked with declines in woodrat and white-footed mouse populations, may have negatively affected MSO territory occupancy. Sureda and Morrison (1998), Willey (1998, 2013), and Willey and Willey (2010) investigated population ecology of *Cricetid* rodents in canyon environments but no studies have examined effects of seasonal precipitation on woodrat and white-footed mouse population dynamics in this region. Increased understanding of how timing and amount of seasonal precipitation affects desert plant and small mammal communities might help explain recent patterns of territorial extirpations and recolonization by MSOs. Given MSOs threatened status, identification of seasonal precipitation timing and effects on MSOs primary prey species dynamics may help predict and manage potential ecological impacts to owls and their primary prey across the canyonlands region of southern Utah. With this impetus, I conducted an intensive analysis of relationships among climate, habitat, and prey species of MSOs from field data collected by Willey and Willey (2010) from 2001 – 2007 in GSENM, and field investigations I conducted from 2013 – 2015 in CARE and 2015 in GSENM (Figures 1.2, 1.3).

Objectives of this study were to (1) describe the nocturnal rodent community characteristics at three study sites in GSENM and five study sites in CARE, (2) examine

how white-footed mouse and woodrat abundance varied relative to microhabitat variables and seasonal precipitation trends, and 3) investigate if variation in woodrat and white-footed mouse abundance explained MSO site-occupancy patterns at a sample of historic nest sites in the study areas.

Study Areas

Research was conducted at two study areas in the canyonlands region of southern Utah: GSENM and CARE (Figure 1.2). Originally comprising 764,856 ha, GSENM was established in September 1996 and was the largest National Monument in the contiguous United States (USDI 2014). However, President Donald Trump signed Presidential Proclamation 9682 modifying the boundaries of GSENM on 4 December 2017. Currently, the Bureau of Land Management is shrinking the monuments boundaries (USDI 2018). The Paria River study area, within the newly designated Kaiparowits Unit of GSENM, consisted of three study sites; Snake Canyon, Hogeeye Canyon, and Starlight Canyon (Figure 1.3). Paria River study sites were located in the south west corner of the Monument, approximately 45 km east of Kanab Utah. Comprising 98,711 ha, CARE was established as a National Monument in August 1937 and designated a National Park in December 1971 (USDI 2015). CARE included five study sites; Upper Spring Canyon, Cassidy Arch Canyon, Burro Canyon, 5-Mile Canyon, and Bitter Creek Divide Canyon (Figure 1.3). CARE is located approximately 25 km east of Torrey, Utah, and is characterized by numerous sandstone canyons deeply eroded into a 160-km north-south tending geologic monocline (Heil et al. 1995, Willey and van Riper 2015). CARE

study sites were located approximately 100 km northeast of GSENM study sites (Figure 1.3).

Study area elevations ranged from 1,590 to 2,202 m. Willey (2013) described the canyon study sites in GSENM and CARE as part of the high plateaus subsection of the Colorado Plateau physiographic province. Both study areas contained similar topography with maze-like sedimentary canyons eroded into cliff-formations. Canyonlands region of southern Utah is defined as a semiarid desert ecosystem (Noy-Meir 1973). Annual precipitation across study areas averaged 260.7 mm ($n = 15$ years, $SE = 8.1$ mm) (Oak Ridge National Laboratories Distributed Active Archive Center, <https://daac.ornl.gov>) and temperatures ranged from -4.6 to 28.2 °C ($n = 15$ years) (Utah Climate Center, Utah State University, <https://climate.usurf.usu.edu>). Precipitation occurred during two distinct seasons; summer monsoons and winter precipitation.

Romme et al. (1993) and Coles et al. (2009) described the vegetative communities in the canyonlands region and noted the vegetation community type strongly depended on topography, hydrology, and surface geology. Vegetation classifications described a mosaic of plant species assemblages rather than distinct community types. Based upon Romme et al. (1993) and Coles et al. (2009), plant associations common across the study sites were categorized into seven habitat types. Desert Scrub was typified by Blackbrush (*Coleogyne ramosissima*), Rubber Rabbitbrush (*Ericameria nauseosa*), Mormon-tea (*Ephedra viridis*), Sagebrush (*Artemisia* spp.), Saltbrush (*Atriplex* spp.), bunch grasses (*Stipa* spp.), Cacti species (*Opuntia* spp.), and Yucca species (*Yucca* spp.). Desert scrub was most common on the terraced bench-lands above canyon bottoms up to canyon rims

and in wider canyon washes. Mountain Scrub was typified by Singleleaf ash (*Fraxinus anomala*), Squawbush (*Rhus trilobata*), Utah Serviceberry (*Amelanchier utahensis*), Silver Buffaloberry (*Shepherdia argenta*), Mountain-mahogany (*Cercocarpus* spp.), and Globe Mallow (*Sphaeralcea* spp.). Mountain scrub communities were most common in higher elevation canyon systems in CARE but not present in GSENM due to the lower elevation of study sites within that study area. Individual mountain scrub species were present in GSENM but not common enough in the community assemblage to label it a habitat type. Grassland was typified by Indian Ricegrass (*Stipa hymenoides*), Needle-and-thread grass (*Stipa comata*), Brome species (*Bromus* spp.), Gramma species (*Bouteloua* spp.), and Western Wheatgrass (*Elymus smithii*). Grassland was most common on the terraced bench-lands above canyon bottoms and in wider canyon washes. Gambel's oak Woodland was typified by Gambel Oak (*Quercus gambelii*), Utah Serviceberry (*Amelanchier utahensis*), Bunch grasses (*Stipa* spp.), Western Wheatgrass (*Elymus smithii*), and Rubber Rabbitbrush (*Ericameria nauseosa*). Gambel's oak Woodland was most common in canyon bottoms of narrow canyon reaches with intermittent surface water. Pinyon – Juniper Woodland was typified by Two-needle pinyon (*Pinus Edulis*), Utah Juniper (*Juniperus osteosperma*), Bunch grasses (*Stipa* spp.), Silver Buffaloberry (*Shepherdia argenta*), Mormon-tea (*Ephedra viridis*), Sagebrush (*Artemisia* spp.), and Saltbrush (*Atriplex* spp.). Pinyon – Juniper woodland was the most common community type in the study areas and was found on terraced bench-lands above canyon bottoms extending up to the adjacent mesa tops and present in bottomlands across wider canyon washes. Riparian Woodland was typified by Box-elder (*Acer negundo*), Fremont

Cottonwood (*Populus fremontii*), Willow species (*Salix* spp.), Cattail species (*Typha* spp.), and Horsetail species (*Equisetum* spp.). Riparian woodland was most common in canyon bottoms of narrow canyon reaches with intermittent surface water. Lastly, Rock-Talus-Cliff habitat type was typified by areas of no vegetation, boulders, talus slopes, and vertical canyon walls (all sandstone).

Methods

Small Mammal Sampling

Small mammal population data collected in the Paria River of GSENM from 2001 – 2007 (Willey and Willey 2010) were combined with data collected in CARE and GSENM during 2013 – 2015 for analyses of relative abundance and habitat associations. In 2001, small mammal trap grids and line-transects were established in the Paria River watershed of GSENM in three occupied MSO territories located in Hogeye, Snake, and Starlight Canyons (Willey and Willey 2010). Trap grids consisted of 100 Sherman folding live traps (7.6 x 9.5 x 30.5cm, model XLK, H.B. Sherman Traps, Inc., Tallahassee, FL) covering a total of 1-ha arranged in a 100 x100-m square with 10-m spacing between each trap (Figure 2.1). Line-transects consisted of 50 model XLK Sherman traps spanning 500-m, arranged in a line along canyon bottoms with 10-m spacing between trap stations. Sherman live traps were insulated with polyester batting (Jo-Ann Stores, Inc.) and baited with a mixture of rolled oats (Quaker Oats Company, PepsiCo, Inc.) and chunky peanut butter (Western Family, Inc.). Trap grids were arranged to traverse major vegetative communities (see Study Area) within 400-m of

spotted owl nest sites, and sampled habitat from canyon bottoms up to canyon benches. In addition to grids, two line-transects were established 100-m down canyon of trap grids in Snake and Hogeeye canyons. One to three trap sessions ranging from three to six trap nights were conducted on grids and transects during July – September from 2001 to 2007 (Willey and Willey 2010). Traps were baited one hour prior to sunset and checked at sunrise. Captured rodents were measured, weighed, identified to species, aged, sexed, then marked with a unique numbered Monel ear tag (National Band and Tag Co., Newport, KY) before being released at the point of capture. Ear tag number of recaptured individuals was recorded and recaptures were released at point of capture.

Small mammals were captured during summers 2013 – 2015 in CARE. A 100x100-m trap grid (100 traps) and two line-transects (25 trap each), were established in two historic spotted owl territories (Willey 1998) in Upper Spring Canyon and Bitter Creek Divide in 2013 repeating methods of Willey and Willey (2010). In 2014, small mammals were captured from June through August. In addition to trap sites established in 2013, a line-transect (25 traps) was deployed at a historic spotted owl territory (Willey 1998) in Cassidy Arch Canyon where a single MSO was observed roosting in April 2014. Additional line-transects (25 traps each) were established in CARE in 2015 at an occupied spotted owl territory in Five Mile Canyon where MSOs were observed nesting and in Burro Canyon where a pair of MSOs was observed exhibiting territorial behavior. Line-transects (25 traps) were re-established at two historic sites (Willey and Willey 2010) in GSENM in Hogeeye Canyon and an occupied spotted owl territory in Snake Canyon where MSOs were observed nesting. Two to three trapping sessions ranging

from three to five nights were conducted during June to August from 2013 to 2015. Only one trap grid was established in CARE (Upper Spring Canyon) because most owl sites in the Park were in canyons too narrow to support 100x100-m trap grids. Line-transects were the primary method used due to topographic constraints.

Habitat Measurements

Vegetation sampling was conducted at each individual trap location at GSNEM study sites ($n = 450$ trap sites) from 2002 – 2006 and in 2015, and each trap location at CARE study sites ($n = 225$ trap sites) in 2015. Sixteen habitat variables were measured at trap sites to describe the vegetative community, ground cover types, and microhabitat components available to small mammals. Microhabitat components were recorded within 10-m diameter (0.008-ha) circular plots centered on every individual trap. Line-point intercept method (Floyd and Anderson 1987) was used to measure percent grass, shrub, rock, water, tree basal area, litter, cactus cover, and bare ground along a 5-m sampling cord centered at each trap extending toward the next adjacent trap on all trap grids and line transects. Within each 10-m diameter circular trap plot maximum shrub height, inverse canopy cover (percent clear sky) above trap (spherical densitometer), and five meters north of each trap were recorded (Willey and Willey 2010). In addition: number of trees, tree height (m), diameter at breast height (DBH; cm), slope (degree), aspect (degree), and a list of plant species present were recorded.

To investigate potential relationships among *Cricetid* rodent abundance and habitat variables I explored the following research questions: Do *Cricetid* species prefer specific habitat types, unique vegetation species and microhabitats, and what

microhabitat components are associated with *Cricetid* species abundance? I tested the hypotheses that percent vegetation and rock cover are important habitat characteristics that drive *Cricetid* rodent abundance in narrow canyon systems with limited resources. To identify possible microhabitat associations, two Null hypotheses were tested: 1) percent vegetation and rock cover were not correlated with white-footed mouse abundance, 2) percent vegetation and rock cover were not correlated with woodrat abundance.

Climate Data

Precipitation data was obtained from the Oak Ridge National Laboratories Distributed Active Archive Center (<https://daac.ornl.gov>; Thornton et al. 2017). Daymet Version 3 daily precipitation metrics were recorded at 1-km x 1-km spatial resolution and summed for the two distinct rainfall seasons on the Colorado Plateau in southern Utah. Three precipitation response variables were calculated 1) Summer monsoon precipitation (mm; July – September), 2) winter precipitation (mm; October – June), and 3) annual precipitation (mm; October– September; Appendix Table B.1).

Drought indices were obtained via the United States Drought Monitor (USDMD, <http://drought.gov>). Drought indices were calculated by USDMD using National Weather Service Climate Prediction Center soil moisture models (<http://www.cpc.ncep.noaa.gov>). USDMD drought indices were categorized as: no drought (soil moisture above 30%), abnormally dry (soil moisture 21-30%), moderate drought (11-20%), severe drought (6-10%), extreme drought (3-5%), and exceptional drought (0-2%). Measurements were averaged for summer monsoon season at GSENM and CARE study sites. Two drought

predictor variables were calculated 1) no drought at study site (no drought + abnormally dry), and 2) drought at study site (moderate drought + severe drought + extreme drought + exceptional drought (Table B.1).

To investigate the influence of seasonal precipitation and drought on *Cricetid* rodent community dynamics in the canyonlands region, I followed an information-theoretic approach (Anderson and Burnham 2002). I used a long term ten-year data set to quantify the effects of seasonal precipitation and drought on *Cricetid* rodent abundance and diversity. I hypothesized winter precipitation was the primary driver of *Cricetid* rodent abundance and diversity in narrow canyon systems with limited resources within the Colorado Plateau.

Data Analyses

Small Mammals. Number of annual small mammal captures was standardized as a capture-per-unit effort rate (R) to report capture success per 100 trap nights and calculated by $R = (\sum_{IDYR} / \sum_{TN}) * 100$; where \sum_{IDYR} = number of all unique individuals captured per year and \sum_{TN} = number of trap nights per year. Species relative abundance (A) at each study site was calculated by $A = (\sum_{IDS} / \sum_{IDYR})$; where \sum_{IDS} = total number of unique individuals within each species group and \sum_{IDYR} = total number of all individuals captured per year. Number of unique individual *Cricetid* species captured per year was used as an index of abundance to investigate potential effects of seasonal precipitation. Four relative abundance variables were calculated: 1) species richness (Bird and Bildstein, 2007), 2) Primary prey (*Pri.Prey*) = number of *Cricetid* rodents captured

annually, 3) Woodrats (*WR*) = number of woodrats captured annually, and 4) White-footed mice (*WFM*) = number of white-footed mice captured annually. To ensure the estimates of relative abundance were comparable across all years of the study, annual capture effort was standardized to only include *Cricetid* rodents captured in the first four trap nights of each annual capture effort because annual capture efforts were inconsistent due to weather and logistic constraints. Four trap nights was the highest number of trap nights on the first annual sampling session across most of the study sites. For example, one study site in CARE had three trap nights the first trap session due to flashfloods which was excluded from analyses, while others had five trap nights. Standardizing annual capture effort to only include the first trap session avoided violating closure assumptions.

To investigate potential effects of seasonal precipitation on rodent diversity in GSENM and CARE, three diversity variables (Bird and Bildstein 2007) were calculated: 1) Shannon Diversity index ($H' = -\sum P_i \ln(P_i)$), 2) Shannon Evenness ($J' = H'/H \max'$), and 3) Simpson's Diversity index was calculated to account for both richness and Evenness ($D_I = 1 - \sum P_i^2$). Following the methods above, variables were standardized to only include *Cricetid* rodents captured in the first four trap nights of each annual capture effort.

Habitat Associations. Each individual trap location was classified as one of the following seven habitat types available to *Cricetid* rodents (see Study Area) desert scrub, grassland, Gambel's oak woodland, mountain scrub, pinyon – juniper woodland, riparian woodland, and rock-talus-cliff. To report percent occurrence of captured *Cricetid* rodents

in each habitat type, habitat type associations (H) were calculated by $H = (\sum_{IDHT} / \sum_{IDYR}) * 100$; where \sum_{IDHT} = total number of *Cricetid* rodents captured in each habitat type and \sum_{IDYR} = total number of *Cricetid* rodent captures per year. Percent occurrence of the five most common woody plant species (P_{lant}) present at trap locations where *Cricetid* rodents were captured (used trap) was calculated by $P_{lant} = (\sum_{VegSpp} / \sum_{VegTotal}) * 100$; where \sum_{VegSpp} = number of vegetation species present at capture trap and $\sum_{VegTotal}$ = total number of vegetative species present at all trap locations.

Microhabitat Associations. Microhabitat was defined as vegetation and other habitat components that might have been important to the presence or absence of *Cricetid* rodent species (see Habitat Measurements). Microhabitat components at traps where *Cricetid* rodents were captured (used) were compared to traps where they were not captured (unused). Summary statistics were calculated for all components and a Welch's two sample t-test was employed to examine any statistical differences between used and unused traps (Program R, <https://cran.r-project.org>). Microhabitat was described at traps used by eight species of *Cricetid* rodents separated into two species groups. The woodrat group consisted of three species; white-throated woodrat (*Neotoma. albigula*), desert woodrat (*N. lepida*), and bushy-tailed woodrat (*N. cinerea*). The white-footed mouse group consisted of five species; brush mouse (*Peromyscus boylei*), cactus mouse (*P. eremicus*), canyon mouse (*P. crinatus*), deer mouse (*P. maniculatus*), and pinyon mouse (*P. trueii*). A Bonferroni adjustment was used to account for repeated measures of conducting t-tests on 16 microhabitat components in GSENM and 14 components in CARE on multiple *Cricetid* species. Bonferroni correction set significance cut off at ($\alpha /$

n) calculated as $(p \leq \frac{0.05}{16})$ (Ramsey and Schafer 2013). Results of t-tests were accepted as indicative of difference at a value of $p \leq 0.003$ in GSENM and $p \leq 0.004$ in CARE.

Recaptures were excluded from microhabitat analyses to avoid potential biases due to “trap-happy” individuals but did include traps where multiple species of small mammal were captured. Microhabitat associations of traps used by woodrat and white-footed mouse species groups were compared to unused traps in GSENM and CARE. Microhabitat associations of traps used by four species of white-footed mouse were compared to unused traps in GSENM. Species-specific habitat associations were not conducted in CARE.

Seasonal Precipitation. Effects of timing and amount of precipitation on *Cricetid* rodent abundance and diversity across study sites was analyzed by methods similar to Johnston and Beever (2018). Following an information-theoretic approach (Anderson and Burnham 2002), *A priori* predictors and regression models were identified based on hypothesized effects of timing and amount of seasonal precipitation on *Cricetid* rodent abundance and diversity. Potential relationships between *Cricetid* rodent abundance and diversity (i.e., dependent variables) as a function of precipitation and drought (i.e., explanatory variables) were examined using linear mixed-effects model analyses, where both fixed-effects (i.e., explanatory variables) and random-effects (i.e., study site) were incorporated to analyze diversity indices and small mammal abundance for each year of the study. Each model used study site as a random-effect to facilitate analyses of repeated measures. Adding study site as a random-effect enabled independence and accounted for variation due to individual site differences by assigning a random intercept for each study

site. Fixed-effects (predictor variables; i.e., explanatory variables) for models represented winter precipitation, summer precipitation, annual precipitation, one-year precipitation lag effect, and drought index at site (Appendix Tables B.2, B.3). All models included one or two predictors, and models did not include more than one precipitation or drought predictor because of autocorrelation among predictors. Summer precipitation was not modeled with drought because drought values were correlated to summer precipitation. A model set of 17 full models was created for each response variable and model sets between abundance count and diversity response variables had the same model structure (Table B.4).

Prior to modeling, boxplots and univariate models were constructed to examine the need to transform the data. Natural log adjusted covariates for winter precipitation, summer precipitation, and annual precipitation were used. Abundance, diversity, and drought index predictor variables were not adjusted because transformation did not reduce over-dispersion. Analyses were performed in Program R (package *lme4*; Bates et al. 2015; <https://cran.r-project.org>).

Linear mixed-effects models (LMER) were used to analyze diversity indices and generalized linear mixed-effects models (GLMER) with a negative binomial distribution were used for abundance count variables to account for over-dispersion of count data. Support for hypotheses was based on Akaike information criteria for small sample sizes (AIC_c), differences in Akaike's information criterion for small sample sizes (ΔAIC_c), and Akaike weights. In addition, model fit was assessed with marginal and conditional coefficients of determination (R^2) for LMER models using the *rsquared* function

(Program R, package *piecewiseSEM*; Lefcheck 2015; <https://cran.r-project.org>) and following an R script for negative binomial GLMER models from Nakagawa et al. (2017) and Johnston and Beever (2018). Marginal R^2 measured variation explained by fixed effects only (i.e., winter precipitation), while conditional R^2 measured variation explained by fixed and random effects combined (i.e., winter precipitation and study site). A Likelihood Ratio Test was conducted to attain p -values for the fixed-effects in the top three LMER models for each diversity index using the *Anova* function (Program R, package *car*; Fox and Weisberg 2011; <https://cran.r-project.org>) and Type II Wald f -tests with Kenward-Roger estimates for degrees of freedom (Kenward and Roger 1997). To assess the significance of fixed-effects on *Cricetid* rodent abundance in the top three negative binomial GLMER models, the *Anova* function (Program R, package *car*; Fox and Weisberg 2011; <https://cran.r-project.org>) and Type II Wald chi-square tests were employed. If the difference in the Likelihood Ratio Tests was significant ($p \leq 0.05$), it was concluded the fixed-effect was a strong predictor of *Cricetid* rodent abundance and diversity.

Low, average, and high precipitation values of top competing models were used to estimate effect of predictors on *Cricetid* rodent abundance and diversity. For example, the estimate of Shannon's Diversity index in an above average summer precipitation year was assessed for a difference from estimated Shannon's Diversity index in a below average precipitation year. The difference in abundance and diversity in above average, average, and low precipitation years was reported as a fine-scale method to quantify the effects of timing and amount of precipitation on *Cricetid* rodent abundance and diversity.

For precipitation predictors, the lowest two years of winter, summer, and annual precipitation recorded during the study were averaged to represent a below average precipitation year (Table B.5). The greatest two years of winter, summer, and annual precipitation recorded during the study were averaged to represent an above average precipitation year (Table B.5). The averages of summer, winter, and annual precipitation (without high and low years) were used to represent an average precipitation year (Table B.5).

Effects of predictors on diversity (D_e) by LMER models were calculated by $D_e = (\beta_0 + (\beta_1 * \text{precipitation}))$ to get an estimate of diversity in low, average, and high precipitation years where β_0 = model intercept, and β_1 = predictor coefficient. Model estimates were used to graph diversity indices over the complete range of observed precipitation and for the top model from each diversity index. GLMER models that included an additive effect of two predictors (e.g., winter precipitation + drought index) on abundance (A_e) were calculated by $A_e = (\beta_0 + (\beta_1 * \text{precipitation}) + (\beta_2 * \text{drought}))$ to estimate abundance in a low, average, and high winter precipitation year plus drought where; β_0 = the model intercept, β_1 = first predictor coefficient, β_2 = second predictor coefficient. GLMER models that included the interactive effects of two predictors (e.g., winter precipitation * drought index) on abundance (A_e) were calculated by $A_e = (\beta_0 + (\beta_1 * \text{precipitation}) + (\beta_2 * \text{drought}) + (\beta_3 * (\text{precipitation} * \text{drought})))$ to estimate abundance in a low, average, and high winter precipitation year and the interaction of drought where; β_0 = the model intercept, β_1 = first predictor coefficient, β_2 = second predictor coefficient, β_3 = coefficient of the interaction of β_1 and β_2 . Model estimates were used to graph

Cricetid rodent abundance over the complete range of observed precipitation values and for the top model from each species group.

Results

Small Mammal Demographics

GSENM – 2,683 unique individuals representing at least 16 species of small mammals were captured during 29,175 trap nights from 2001 – 2007 and in 2015 (Appendix Table A.1). Overall trap success was 9.2 (95% CI = 4.4 - 13.9; range 3.2 [2002] to 16.9 [2006]). Trap success for recaptures was 23.4 (95% CI = 9.0 – 37.7; range 10.2 [2007] to 46.9 [2006]).

White-footed mice (WFM) were the most frequently captured species group ($n = 2,123$) and represented 79.1% of all captures (Table A.2). Brush mice were the most common species captured (75.1%), followed by American deer mice (15.7%), pinyon mice (4.1%), canyon mice (4%), cactus mice (0.8%), and unidentified *Peromyscus* spp. comprised 0.4% of captures. Woodrats were the second most frequently captured species group ($n = 310$) and represented 11.6% of all captures (Table A.2). White-throated woodrats were the most common species captured (88.7%), followed by desert woodrats (6.8%), bushy-tailed woodrats (2.9%), and unidentified *Neotoma* spp. comprised 1.6% of captures. Other small mammals captured included the long-tailed pocket mouse (*Chaetodipus formosus*), cliff chipmunk (*Tamias dorsalis*), western harvest mouse (*Reithrodontomys megalotis*), least chipmunk (*Tamias minimus*), kangaroo rat (*Dipodomys* spp.), ringtail (*Bassariscus astutus*), rock pocket mouse (*Chaetodipus*

intermedius), and an unidentified species of shrew (*Sorex spp.*) comprised 9.3% of all captures.

CARE – 365 unique individuals representing 12 species of small mammals were captured over 6,475 trap nights from 2013 – 2015 (Table A.3). Overall trap success was 6.2 (95% CI = 2.9 - 9.6; range 4.0 [2014] to 10.1 [2013]). Overall trap success for recaptures was 15.0 (95% CI = 11.5 – 18.6; range 11.3 [2014] to 18.4 [2013]).

WFM were the most frequently captured species group ($n = 195$) and accounted for 53.4% of all captures (Table A.4). Brush mice were the most common species captured (62.1%), followed by American deer mice (19.0%), pinyon mice (9.7%), canyon mice (5.1%), and cactus mice (4.1%). Woodrats were the second most frequently captured species group ($n = 105$) and accounted for 28.8% of all captures (Table A.4). Desert woodrats were the most common species captured (58.1%), followed by bushy-tailed woodrats (36.2%), and white-throated woodrats (5.7%). Other small mammals captured included the cliff chipmunk (*Tamias dorsalis*), western spotted skunk (*Spilogale gracilis*), rock squirrel (*Spermophilus variegatus*), and Great Basin pocket mouse (*Perognathus parvus*) comprised 17.8% of all captures.

Cricetid Rodent Diversity

GSENM – Standardized for capture effort, 1,251 *Cricetid* rodents were captured from 2001 – 2007 and in 2015; range 40 [2015] to 491 [2006]; Table 2.1). Mean annual *Cricetid* rodent species richness was 5.6 species (95% CI = 4.59 – 6.91; range 3 [2007] to 8 [2006]). Mean annual Shannon’s Diversity index was 1.03 (95% CI = 0.76 – 1.31; range 0.502 [2007] to 1.438 [2001]). Mean annual Simpson’s Diversity index was 0.52

(95% CI = 0.37 to 0.66; range 0.259 [2007] to 0.894 [2001]). Mean annual Evenness was 0.60 (95% CI = 0.45 to 0.75; range 0.356 [2006] to 0.894 [2001]).

CARE – Standardized for capture effort, 129 *Cricetid* rodents were captured from 2013 – 2015; range: 24 [2014] to 63 [2013]; Table 2.2). Mean annual *Cricetid* rodent species richness remained stable at six species captured per year. Mean annual Shannon’s Diversity index was 1.49 (95% CI = 0.84 – 2.13; range 1.187 [2013] to 1.641 [2015]). Mean annual Simpson’s Diversity index was 0.74 (95% CI = 0.41 – 1.06; range 0.584 [2013] to 0.819 [2014]). Mean annual Evenness was 0.83 (95% CI = 0.47 – 1.19; range 0.663 [2013] to 0.916 [2015]).

Cricetid Rodent Species Group Microhabitat Associations

GSENM – White-throated woodrats comprised 91.5% of all woodrat captures ($n = 200$) at 199 trap stations. Small sample size prevented analysis of species level microhabitat associations for desert woodrats ($n = 10$), bushy-tailed woodrats ($n = 4$), and three unidentified *Neotoma* species and they were excluded from species specific microhabitat analyses. However, they were included in the pooled woodrat species group analyses. Microhabitat associations are presented for woodrats as a pooled species group. Woodrat capture sites were strongly ($p < 0.001$) associated with relatively large diameter trees, in dispersed low density-stands, and in areas with greater shrub height compared to unused traps (Table 2.3). In addition, woodrat capture sites were strongly ($p < 0.001$) associated with greater canopy cover 5-m north of the capture trap, but were also associated with less percent tree basal area (a measure of stem-density versus canopy cover) compared to unused traps. No differences between used and unused traps were

detected for percent canopy cover over capture trap, shrub cover in trap vicinity, litter cover, water, cactus, or rock cover. However, there was evidence that taller trees, a southern aspect, moderate slopes, less forb cover, and patches of bare ground might have been important habitat components to woodrats (Table 2.3).

WFM were the most abundant species group captured throughout the study in GSENM ($n = 1,010$). Small sample size prevented analysis of species level microhabitat associations for cactus mice ($n = 16$) and eight unidentified *Peromyscus* species and they were excluded from microhabitat analyses but they were included in the WFM pooled species group. WFM capture sites were strongly ($p < 0.001$) associated with relatively dense stands of large diameter trees, with greater canopy cover centered at capture traps and 5-m north of the trap. Traps used by WFM also showed greater shrub height and were on more moderate slopes compared to unused traps (Table 2.4). WFM capture traps showed relative close proximity to water and less rock cover compared to unused traps. No differences between used and unused traps were detected with respect to number of trees, a particular aspect, percent shrub cover, tree basal area, or cactus. However, there was evidence that relatively taller trees, greater litter cover, and larger patches of bare ground might have been important habitat components to WFM (Table 2.4).

CARE – Microhabitat analyses were conducted on woodrat and WFM species as pooled species groups. Species-specific microhabitat analyses were not conducted in *CARE* because captures were dominated by two species, desert woodrats and brush mice. Bushy-tailed woodrats, white-throated woodrats, deer and canyon mice had insufficient captures ($n < 20$) to assess these species uniquely and they were excluded from

microhabitat analyses. However, they were included in the pooled species group analyses. No differences were detected in microhabitat components for woodrats between used and unused traps (Table 2.5).

WFM were the most abundant species group captured in CARE and capture sites were strongly ($p < 0.001$) associated with greater shrub height at used traps. In addition, there was evidence that greater canopy cover at capture trap and greater percent shrub cover may have been important habitat components for WFM in CARE (Table 2.6).

Seasonal Precipitation and *Cricetid* Rodent Abundance and Diversity

GSENM – Precipitation varied significantly over the course of the study (Table 2.1, Figure 2.2). From 2000 to 2015, mean winter precipitation was 205.2 mm (± 116.3 SD; range 71 mm [2002] to 514 mm [2005]), mean summer precipitation was 82.1 mm (± 32.7 SD; range 25 mm [2003] to 145 mm [2014]), and mean annual precipitation was 287.3 mm (± 118.0 SD; range 139 mm [2002] to 586 mm [2005]).

Cricetid rodent abundance was greatest and number of captures were highest in 2006, the year following the greatest amount of winter precipitation observed in 2005 (490mm; Table 2.1). However, the high *Cricetid* abundances observed in 2006 was associated with second lowest Simpson's and Shannon's Diversity indices and the lowest Shannon's Evenness index recorded because those captures were dominated by brush mice (73%) and white-throated woodrats (14%). Following two consecutive years (i.e., 2002 – 2003) of lowest winter precipitation recorded (75mm) combined with exceptional drought, diversity indices steadily declined. (Table 2.1, Figure 2.3). Diversity likely declined due to predominance of brush and deer mice, and white-throated woodrats

which together comprised 90% of captures in 2004 (Tables 2.1, A.1). From 2003 to 2004, canyon mouse captures decreased 91% and pinyon mice captures decreased 100%. However, brush mice and white-throated woodrat abundance increased from 2001 through 2006. Brush mice comprised 86% of captures in 2007 resulting in the lowest diversity and Evenness indices recorded during the study (Table 2.1). Overall, the WFM species group exhibited strong annual variability apparently in response to the effects of seasonal variation in precipitation. Following low winter precipitation and exceptional drought during 2002-2003, deer mice abundance increased for one year in 2004, then declined for the remainder of the study. Canyon and pinyon mice abundance varied substantially resulting in years where very few or none of these species were captured. Brush mice and woodrat abundance increased over the duration of the study until 2007, when brush mice abundance decreased 88%, woodrat abundance decreased 100%, and few other *Cricetid* rodents were captured (Tables 2.1, A.1).

CARE – Seasonal precipitation varied significantly over the course of the study (Table 2.2, Figure 2.4). From 2012 to 2015, mean winter precipitation was 143.1 mm (± 52.8 SD; range 92 mm [2014] to 232 mm [2015]), mean summer precipitation was 141.3 mm (± 32.2 SD; range 101 mm [2012] to 210 mm [2013]), and mean annual precipitation was 284.4 mm (± 50.4 SD; range 196 mm [2012] to 363 mm [2015]).

Cricetid rodent diversity indices increased from 2013 to 2015. Shannon's Diversity index and Shannon's Evenness reached their maximum, and Simpson's Diversity was second highest following the largest amount of winter precipitation (208mm) and the lowest drought levels observed in 2015 (Table 2.2, Figure 2.5).

Diversity indices were at their minimum in 2013 (first year of the study) likely in response to the lowest recorded winter precipitation (107mm), combined with severe drought that occurred in 2012 and extended into 2013. *Cricetid* rodent abundance was highest in 2013 and lowest in 2014, and relatively high *Cricetid* rodent abundances observed in 2013 were primarily comprised of brush and pinyon mice (76%) resulting in the lowest diversity indices recorded during the study (Table 2.2). Although species richness remained stable in these communities ($S = 6$), the same six species were not captured each year. Demonstrating a similar trend as WFM in GSENM, the CARE WFM species group exhibited strong responses to the effects of seasonal precipitation through high annual variance in abundance (Table 2.2). Following two consecutive years of the lowest recorded winter precipitation combined with severe drought in 2012 - 2013, WFM abundance declined 90% reaching its minimum in 2014. However, diversity increased in 2014 due to relatively similar abundances of six *Cricetid* rodent species. Pinyon mouse abundance declined 100% from 2013 to 2014 and they were not captured again during the study. Following greater winter precipitation combined with no drought, WFM abundance increased from 2014 to 2015. The CARE woodrat species group, like woodrats in GSENM, exhibited a consistent increase in abundance over the duration of the study (Table 2.2).

Seasonal Precipitation, Drought Predictors, and *Cricetid* Rodent Diversity

Overall, linear mixed-effects model analyses indicated that Shannon's Diversity, Shannon's Evenness, and Simpson's Diversity were most strongly influenced by

precipitation from the preceding year (Figures 2.6, 2.8, 2.10) and as precipitation increased, diversity decreased.

The top-ranked model for the effects of precipitation on Shannon's Diversity index provided some evidence that increased summer precipitation decreased Shannon's Diversity at study sites ($F_{1,24} = 3.62$, $p = 0.07$, $w_i = 0.27$; Table 2.7). Thirteen percent of the conditional variation was explained by the top ranked model. The second top-ranked model ($\Delta AIC_c = 1.35$) provided weak evidence that increased annual precipitation from the preceding year (1-year lag) was associated with a decreased Shannon's Diversity index among study sites ($F_{1,24} = 2.69$, $p = 0.11$, $w_i = 0.14$; Table 2.7). Nine percent of the conditional variation was explained by the model. The third top-ranked model ($\Delta AIC_c = 1.85$) provided weak evidence that increased winter precipitation from the preceding year (1-year lag) decreased Shannon's Diversity across study sites ($F_{1,28} = 2.20$, $p = 0.15$, $w_i = 0.11$; Table 2.7). Eleven percent of the conditional variation was explained by the model. The null model was $\Delta AIC_c > 2$ from the top three models and indicated the models were competitive. However, the null model was only slightly greater than $\Delta AIC_c = 2$ and the top three model Akaike weights summed to 0.52 (Table 2.7) suggesting the fixed-effects were not strong predictors of *Cricetid* rodent diversity.

The top-ranked model for Shannon's Evenness provided some evidence that increased winter precipitation from the preceding year (1-year lag) decreased Shannon's Evenness at study sites ($F_{1,27} = 3.62$, $p = 0.07$, $w_i = 0.21$; Table 2.7). Twenty-seven percent of the conditional variation (winter precipitation and study site) and 11% of the marginal variation (study site only) was explained by the top-ranked model. The second

top-ranked model ($\Delta\text{AIC}_c = 1.06$) provided some evidence that increased annual precipitation decreased Shannon's Evenness index ($F_{1,23} = 3.00$, $p = 0.10$, $w_i = 0.13$; Table 2.7). Twenty-six percent of the conditional variation and 11% of the marginal variation was explained by the model. The third top-ranked Evenness model ($\Delta\text{AIC}_c = 1.10$) provided weak evidence that increased annual precipitation from the preceding year decreased Shannon's Evenness ($F_{1,23} = 2.86$, $p = 0.10$, $w_i = 0.12$; Table 2.7). Twenty-four percent of the conditional variation and eight percent of the marginal variation was explained by the model. The null model for Shannon's Evenness ($\Delta\text{AIC}_c = 1.91$) was within $\Delta\text{AIC}_c < 2$ from the top three models suggesting the models were competitive. However, the null model was only slightly less than $\Delta\text{AIC}_c = 2$ and the top three model Akaike weights summed to 0.46 (Table 2.7) suggesting the fixed-effects were not strong predictors of *Cricetid* rodent evenness.

Linear mixed-effects model analyses showed Simpson's Diversity index exhibited a similar trend as Shannon's Evenness index (Table 2.7, Figures 2.8, 2.10). Like Shannon's Evenness, Simpson's Diversity index was most strongly influenced by precipitation from the preceding year. As winter and annual precipitation from the previous year increased, diversity decreased. Simpson's Diversity index slightly increased as summer precipitation from the previous year increased (Figure 2.10). Simpson's Diversity index LMER model outputs showed a similar trend as Shannon's Evenness (Tables B.7, B.8).

Seasonal Precipitation, Drought Predictors, and *Cricetid* Rodent Abundance

Overall, generalized linear mixed-effects analyses showed primary prey abundance was most strongly influenced by winter precipitation from the preceding year (Figure 2.11). Primary prey abundance increased following increased annual precipitation but the increase was not as strong as the response to winter precipitation. Primary prey abundance decreased as summer precipitation from the preceding year increased.

The top-ranked model for the effects of precipitation on primary prey abundance provided very strong evidence that as winter precipitation from the preceding year (1-year lag) increased, primary prey relative abundances at study sites increased ($\chi^2_1 = 12.1$, $p < 0.001$; Table 2.9, Figure 2.12). The top-ranked model showed the highest Akaike weight 0.56, and explained 26% of the conditional variation (winter precipitation w/1-year lag and study site) and 21% of the marginal variation (study site only). The second top-ranked model ($\Delta AIC_c = 1.77$) provided strong evidence that increased winter precipitation following a preceding year experiencing drought increased primary prey relative abundance at study sites ($\chi^2_1 = 12.6$, $p < 0.001$, $\chi^2_1 = 1.28$, $p < 0.26$; Table 2.9, Figure 2.13). This model had the second highest Akaike weight 0.23, and explained 31% of the conditional variation and 20% of the marginal variation. The next best model was ($\Delta AIC_c = 4.9$) indicating it was not competitive. The two top-ranked primary prey model Akaike weights summed to 0.80, and all top ranked primary prey models included winter precipitation and its additive or interactive effects with drought index at study site with a cumulative Akaike weight of 0.88 which indicated the potential importance of winter

precipitation on primary prey relative abundance (Table 2.9). The best model which included summer precipitation as a predictor was ($\Delta\text{AIC}_c = 6.9$) and the best model which included annual precipitation was ($\Delta\text{AIC}_c = 6.9$). Compared to the null model, the high ΔAIC_c scores and low model weights of summer and annual precipitation models indicated the potentially strong influence of winter precipitation on primary prey abundance (Table B.9).

Overall woodrat abundance was most strongly influenced by the effects of drought at study sites. The top-ranked model for woodrat abundance provided strong evidence that as the drought index at study site increased, woodrat abundance decreased ($\chi^2_1 = 10.5, p = 0.001$; Table 2.9). Woodrat abundance decreased by 1.4 woodrats when the study site was experiencing drought conditions compared to when the study site was not experiencing drought (Table 2.10, Figure 2.14). Seventeen percent of the conditional variation was explained by the model. The second top-ranked model ($\Delta\text{AIC}_c = 1.25$) for woodrat abundance provided some evidence that increased annual precipitation following a preceding year experiencing drought increased woodrat abundance at study sites ($\chi^2_1 = 0.80, p = 0.37, \chi^2_1 = 5.76, p = 0.02$; Table 2.9). As annual precipitation and the interactive effect of no drought increased woodrat abundance slightly decreased. However, as annual precipitation increased following a year in drought conditions, woodrat abundance increased strongly and indicated the importance of increased annual precipitation following a drought year on woodrat abundance (Figure 2.15). Thirty-four percent of the conditional variation was explained by the model. The third top-ranked model ($\Delta\text{AIC}_c = 1.90$) for woodrat abundance provided some evidence that increased annual precipitation

and the additive effect of drought increased woodrat relative abundance at study sites ($\chi^2_1 = 0.97, p = 0.32, \chi^2_1 = 5.77, p = 0.02$; Table 2.9). Twenty-two percent of the conditional variance was explained by the model. The null model ($\Delta\text{AIC}_c = 9.76$) was $\Delta\text{AIC}_c > 2$ indicating models were competitive. Akaike model weights for the top three woodrat models summed to 0.45 indicating fixed-effects were not strong predictors of woodrat abundance.

Generalized linear mixed-effects analyses showed white-footed mouse abundance exhibited a similar trend as primary prey abundance and was most strongly influenced by winter precipitation from the preceding year (Table 2.10). Similar to primary prey abundance, white-footed mouse abundance was strongly influenced by increased winter precipitation following a drought year (Figures 2.13, 2.16). Like primary prey abundance, white-footed mouse abundance increased following increased annual precipitation but the increase was not as strong as the response to winter precipitation. White-footed mouse GLMER model outputs showed a similar trend as primary prey outputs (Table 2.10). The top two WFM model Akaike weights summed to 0.64 and indicated the importance of winter precipitation on WFM relative abundance (Table 2.9).

Discussion

Spanning ten years of field work and 35,650 trap nights of capture effort, *Cricetid* rodents accounted for approximately 90% of all captures and indicated they were the most abundant nocturnal small mammals and potential prey available for MSOs inhabiting the rocky canyon habitats of GSENM and CARE (Tables A.1, A.3). Although

eight species of *Cricetid* rodents were captured, captures were dominated by brush mice and American deer mice across all study sites, and indicated these species may be the most ubiquitous nocturnal small mammals in the region (Tables A.2, A.4). Sureda and Morrison (1998) also documented nocturnal small mammal captures dominated by the WFM species group in a similar study conducted in the Manti – LaSal National Forest, approximately 100 km east of CARE.

Results of microhabitat analyses provided strong support for hypotheses that percent vegetation cover was an important habitat component to the primary prey species of MSO inhabiting rocky canyon habitats, results consistent with other MSO primary prey studies (Sureda and Morrison 1999, Block et al. 2005). However, compared to these studies, strong *Cricetid* rodent associations with tree variables were observed in GSENM and mean values for tree variables were generally higher in CARE. Both woodrat and WFM species groups were strongly associated with moderate slopes ($< 20^\circ$) close to canyon bottoms and often associated riparian woody vegetation (Tables 2.3, 2.4). Similarly, in the forests of northern Arizona, Block et al. (2005) captured *Cricetid* rodents primarily on slopes $< 20^\circ$ indicating the combination of vegetation on moderate sloped canyon habitats was an important habitat characteristic for these species. Furthermore, woodrat and WFM species groups had low captures in grassland and rock-talus-cliff relative to other habitat types indicating associations for cover provided by trees and shrubs (Tables 2.11, 2.12).

Rock cover was not an important habitat component to all primary prey species of MSO inhabiting rocky canyon habitats. Canyon mice were the only species strongly ($p <$

0.001) associated with greater rock cover at capture traps (Table 2.13). Conversely, brush mice were strongly ($p < 0.001$) associated with less rock cover at capture traps (Table 2.14). Mean percent rock cover at traps used by woodrats and pinyon mice was greater at used vs unused traps in both GSENM and CARE but no significant differences were detected (Tables 2.15, 2.3). Importance of rock cover for *Cricetid* rodents, particularly woodrats has been demonstrated by (Armstrong 1979, Block et al. 1995, Mullet and Ward 2010) but Thompson (1982) concluded woodrat foraging and feeding behaviors were primarily associated with habitat structure which provided cover from aerial and visually oriented predators on three sides, as well as from above, and noted cover provided by vegetation was used when available. When vegetation was not available woodrats were restricted to rock outcroppings (Thompson 1982). Cover for woodrats in canyon bottoms may have been provided by large trees explaining their lack of association with rock cover at capture traps. While statistical evidence lacked support for rock cover at the microsite level where *Cricetids* foraged and were captured at night, all *Cricetid* rodents were captured in rocky canyon environments with abundant rock cover above canyon bottoms indicating its potential biological importance at broader spatial scales.

Habitat Associations and *Cricetid* Rodent Coexistence

Microhabitat analyses were useful in understanding the general ecology of different *Cricetid* rodent species inhabiting rocky canyon habitats. Analyses of capture data indicated *Cricetid* species tended to be captured in specific vegetation assemblages suggesting specialized habitat use and spatial separation while foraging. Resource

partitioning observed among *Cricetid* rodents may have minimized interspecific competition and enabled coexistence in narrow canyon environments with limited resources. For example, woodrat capture sites were more evenly distributed across desert scrub (33.5%), pinyon – juniper woodland (30.5%), and riparian woodland (29%) habitat types compared to WFM in GSENM (Table 2.11). White-throated woodrats were frequently captured sympatric with brush mice ($n = 129$) in riparian and desert scrub habitats during high abundances observed in 2005 and 2006. However, resource and spatial separation among woodrat and WFM species coexisting in similar habitat types were observed in GSENM (Table 2.16). Woodrats in GSENM and CARE avoided open areas similar to results of Thompson (1982).

Brush mice were the most frequent primary prey species captured. They were most strongly associated with water and forested riparian areas of all *Cricetid* species (Table 2.14). Brush mice also showed the strongest ($p < 0.001$) associations with the greatest canopy cover, the tallest and largest diameter trees, most moderate slopes, and the least percent rock cover of all WFM. Similar to other studies in desert environments (Honeycutt and Roppe 1981, Hallett 1982, and Ellison and van Riper 1998), brush mice were the most common *Cricetid* rodent captured in both riparian (45%) and Gamble's oak woodland habitat types in GSENM and CARE (7.6%; Table 2.17). Brush mice climb well and often forage in trees (Flinders et al. 2002) and four of the five most common woody vegetative species present at capture sites were trees (Table 2.18).

American deer mice were the second most frequent primary prey species captured. Deer mice were the most widely distributed WFM species across all habitat

types indicating its generalist nature (Table 2.17; Holbrook 1978, Honeycutt et al. 1981, Flinders et. al 2002). Deer and brush mice were often collected sympatric ($n = 95$) in 2005-2006. However, deer mice were captured with greater frequency in desert scrub, grassland, and rock-talus-cliff habitat compared to brush mice which indicated possible resource and spatial separation enhancing coexistence of these two species (Table 2.17). Holbrook (1978) observed evidence of coexistence of these species noting, deer mice primarily foraged on ground level resources while brush mice foraged both on ground and arboreal resources in areas where their distributions overlapped. Overall, high abundances and comparable use of habitat types indicated brush and deer mice were the most successful WFM species in both study areas.

Pinyon mice were captured more frequently than all other WFM species in pinyon – juniper woodland and is consistent with other desert small mammal studies (Table 2.17; Wilson 1968, Armstrong 1979, Honeycutt and Roppe 1981). Trap sites used by pinyon mice had lowest mean canopy cover at capture trap, 5-m north of trap, and lowest mean percent forb cover compared to other WFM species (Table 2.15). Pinyon mice were captured at sites with greater mean percent tree basal area, rock cover, and on steeper slopes compared to brush and deer mice. Greater tree basal area and less canopy cover were common at capture sites in proximity to Utah juniper which provided less canopy cover compared to cottonwood trees. Pinyon mouse associations with pinyon – juniper habitat, common woody vegetation (i.e., Utah juniper and Pinyon pine), and steeper rocky slopes removed from riparian corridors at capture sites supported evidence of spatial separation from brush and deer mice (Tables 2.17, 2.18). Pinyon mouse capture

sites showed evidence of minimal interaction with other *Cricetid* rodents and were rarely captured sympatric to deer ($n = 3$) or canyon mice ($n = 3$), and infrequently with brush mice ($n = 22$). Sympatric captures occurred predominately in riparian woodlands in 2005 and 2006 when overall *Cricetid* abundances were greatest.

Of all *Cricetid* rodents, canyon mice were most frequently captured in desert scrub habitat (Table 2.17). Used trap sites were strongly ($p < 0.001$) associated with the steepest, rocky slopes, least number of trees, least litter cover, furthest from water and riparian areas compared to all other *Cricetid* species (Table 2.13). Similarly, Johnson and Armstrong (1987) and Sureda and Morrison (1999) observed canyon mice occurred exclusively in rocky habitats and noted species distribution was primarily associated with rocky substrates not vegetation assemblages. Canyon mouse capture sites were most often associated with desert scrub habitat, common woody vegetation (i.e., green ephedra and silver buffaloberry), and steep rocky slopes away from riparian corridors, separated from brush and deer mice (Tables 2.17, 2.18). Pinyon and canyon mice were rarely caught sympatric ($n = 3$) suggesting spatial separation and resource partitioning between species. Canyon mice were rarely captured sympatric to deer mice ($n = 3$) and infrequently with brush mice ($n = 27$) occurring predominately in desert scrub habitat in 2005 and 2006 when overall *Cricetid* abundances were greatest.

Seasonal Precipitation and *Cricetid* Rodent Abundance and Diversity

A direct relationship between primary prey abundance and presence or absence of winter precipitation was apparent both in GLMER model results (Figure 2.11) and abundance relative to winter precipitation (Tables 2.1, 2.2). Winter precipitation with a

1-year lag effect was the strongest predictor ($p < 0.001$) of primary prey abundance indicating primary prey species responded primarily to winter precipitation from the previous year. Additionally, increased winter precipitation following a drought year was a strong driver ($p < 0.001$) of primary prey abundance (Figures 2.13, 2.17, 2.18). Similar rodent abundance lag responses to precipitation have been reported by (Beatley 1976, Brown and Heske 1990, Brown and Ernest 2002, and Thibault et al. 2010).

Increased winter precipitation did not increase *Cricetid* rodent diversity in both study areas. Results showed greater winter precipitation had the opposite effect on diversity indices in GSENM. All diversity indices were negatively related to precipitation. Negative relationship between diversity and precipitation is likely a result of two consecutive summers (2002 – 2003) of record low precipitation combined with record high drought. Following these two years, all three diversity indices declined throughout the remainder of the study because the *Cricetid* rodent community declined from eight species to three species (Figure 2.3). Following record high winter precipitation in 2005, small mammal abundance increased dramatically. However, increased abundance was composed primarily of three species; brush mice, deer mice, and white-throated woodrats resulting in lower diversity. LMER analyses showed Shannon's Evenness and Simpson's Diversity index slightly increased as summer precipitation increased (Figures 2.8, 2.10). This is likely a result of the small mammal community becoming more even because captures were dominated by three species. In contrast, diversity indices in CARE increased as winter precipitation increased (Figure 2.5). *Cricetid* species abundances were lower in CARE compared to GSENM because of

less intensive capture effort, however, a more even number of *Cricetid* rodent species were captured which increased diversity. Low sample sizes in CARE likely explains why these results were not detected in the analyses.

Annual abundance among *Cricetid* species varied significantly and all *Cricetid* rodents showed significant relationships with precipitation. This could be an indicator of mechanisms to adapt to low precipitation and drought conditions. For example, pinyon and canyon mice exhibited the greatest fluctuations in annual abundance of all WFM species in both study areas. In GSENM, following record low precipitation combined with exceptional drought in 2002 to 2003, canyon mouse abundance decreased 91%, and pinyon and cactus mouse abundance decreased 100% in 2004 (Table A.1). Similarly, following two years of low precipitation and drought conditions in CARE in 2012 to 2013, pinyon mouse abundance decreased 100% in 2014 and were not captured the following year in 2015. Rosenzweig and Winakur (1969) observed some desert rodents were at times present but inactive and untrappable, noting cactus mice estivated and questioned which other species might exhibit this behavior. This study provided possible evidence that both pinyon and canyon mice might exhibit this adaptive strategy following consecutive years of low precipitation and extreme drought.

In contrast, brush mice and white-throated woodrats in GSENM and desert woodrats and bushy-tailed woodrats in CARE increased in abundance during periods of low precipitation and drought (Tables 2.1, 2.2). Potential mechanisms which enabled increased abundances could be investigated by examining the microhabitat features these species were associated with. Brush mice were most associated with forested riparian

woodlands in canyon bottoms of all WFM. Similarly, woodrats were strongly associated with large trees common in riparian areas suggesting these species could endure drought conditions due to cooler micro-climates and associated vegetation common in forested riparian areas. In a temperature influenced diet study, Dearing et al. (2008) found temperature significantly affected total food intake and altered diet selection of white-throated woodrats. Woodrats acclimated to cooler temperatures (20°C) had food intakes 1.9-2.2 times greater than woodrats acclimated to higher temperatures (28°C). These results further indicate the importance of cooler micro-climates provided by forested riparian areas. While this study did not investigate vegetation responses to precipitation, it is probable vegetation assemblages in riparian corridors were able to tolerate drought conditions more effectively than exposed vegetation assemblages above canyon bottoms where pinyon, canyon, and cactus mice were most abundant. Vegetation communities on exposed bench lands likely experienced the effects of drought through lower soil moisture and greater surface temperatures more severely than sheltered vegetation assemblages common in riparian corridors where soil moisture was likely higher and the micro-climate likely cooler, enabling greater abundances of riparian associated *Cricetid* species during periods of drought. Ernest et al. (2000) identified temporal variation in vegetation responses of seasonal precipitation noting up to three-year lag responses on plant growth and seasonal plant cover on different vegetation communities and suggested rodent responses to precipitation could be mediated by the species composition of the vegetation community. Interestingly, brush mice and woodrats in GSENM increased in abundance for three years (i.e., 2004 – 2006) then exhibited sharp declines in abundance

in 2007 (Table A.1). This might be evidence of the three-year lag effect observed by Ernest et al. (2000) on riparian vegetation following extreme drought. In a similar long-term small mammal study in Arizona, Thibault et al. (2010) did not observe a dramatic increase in plant abundance following a record precipitation event in 2006 and noted a surprising degree of complexity in the response of primary producers. Record precipitation was observed during this study in 2005 and was likely beneficial to all vegetation in the region. However, how this precipitation was used by primary producers and mechanisms responsible for declines in abundance of all *Cricetid* rodents in 2007 remain unclear. This study did not investigate the effects of predation on *Cricetid* rodent abundance by MSOs, competitive interactions with other desert rodent species (i.e., long-tailed pocket mouse, western harvest mouse, chipmunk), or competition with non-rodent species for seeds and insects (i.e., birds). Influences of predation and interspecific competition on *Cricetid* abundance would be expected during limited biological resources associated with low precipitation and drought conditions.

Another possible mechanism enabling increased woodrat and brush mouse abundance during periods of drought might be different foraging strategies and different diets among *Cricetid* species. For example, woodrats feed primarily on vegetation and all water needs are met by ingestion of green vegetation (Lee 1963, Thompson 1982). Interestingly, green ephedra (*Ephedra viridis*) was the third most common woody plant species at woodrat capture in GSENM and it was the second most common woody plant species at woodrat capture sites in CARE indicating it might be an important food source for woodrats during drought conditions (Tables 2.19, 2.20). Brush mice also feed on

vegetation, berries, acorns and seeds with greater frequency compared to deer mice (Flinders et al. 2002). Green ephedra was the seventh most common vegetative species at brush mouse capture sites ($n = 97$) again suggesting this might be an important food source during drought conditions. Furthermore, in a small mammal food habit study in Colorado, Haufler and Nagy (1984) showed arthropods comprised 96% of the diet of 43 deer mice while eight bushy-tailed woodrats were found to be strictly herbivorous with a diet comprised primarily of woody vegetation (85%) and forbs (14%). Even in years of above average seed and vegetation production, deer mice primarily consumed arthropods while bushy-tailed woodrats primarily consumed woody vegetation and suggested rodent species appeared to avoid competition for food by selecting different food items as the main component of their diets (Haufler and Nagy 1984). Canyon and pinyon mice are omnivorous and eat seeds, insects, and green vegetation in amounts depending on availability and insects comprised a significant portion of the diet when seeds were rare (Johnson and Armstrong 1987, Flinders et al. 2002). While seed production and insect abundance was not measured during this study, possible declines of arthropods and seed production in upland habitats during drought conditions combined with possible estivation / inactivity strategy could be a signal on why these species declined during the exceptional drought conditions observed during this study.

Cricetid Rodent Abundance and Mexican Spotted Owl Site Occupancy

This study was primarily focused on assessing the distribution, habitat associations, and responses of MSO primary prey species to seasonal precipitation. However, through observations and MSO occupancy surveys conducted from 2013 –

2015 in CARE, it is possible to evaluate MSO responses to increased primary prey abundance in a sample of historic MSO territories. For example, small mammal trapping was conducted at five historic MSO territories (Willey 1998) in CARE from 2013 – 2015 and only one of these territories, Five-Mile Canyon, was continuously occupied by MSO. At the beginning of this study, Bitter Creek Divide (BCD) was selected as a small mammal trap site because it was historically one of the most active MSO territories in CARE (Willey, pers comm.) but was abandoned prior to the start of the study. BCD was trapped in 2013 – 2015 and consistently had high abundances of bushy-tailed woodrats and the highest abundances of brush mice of all CARE study sites (Table A.4). Furthermore, woodrat abundance increased each year of the study. BCD was surveyed for MSO in 2013 and 2014 by vocal mimicry with no detections. Following the greatest amount of winter precipitation (208 mm) and highest woodrat abundance recorded in 2015, BCD was re-occupied by a single male MSO in 2015, and a nesting pair was observed the following summer in 2016. BCD was one of the narrowest slot canyons of all study sites in CARE, similar to Five-Mile Canyon. BCD and Five-Mile were narrow slot canyons with abundant riparian vegetation and trees which provided cooler microclimates for both owls and primary prey. The biological significance of an abandoned territory being re-occupied by a pair of MSOs following increased winter precipitation and increased primary prey is an important indicator and could be used as an example of future recolonization events in the region.

Management Considerations

Results of this study showed *Cricetid* rodents were the most abundant nocturnal small mammals and potential prey available to MSO inhabiting rocky canyon habitats of GSENM and CARE within the Colorado Plateau. Therefore, MSO management plans are encouraged to incorporate management strategies for MSO primary prey species, specifically woodrats and brush mice. Habitat use patterns for *Cricetid* rodents vary at broader spatial scales in different desert ecosystems. Many of the primary prey habitat associations observed in this study were consistent with other studies (Sureda and Morrison 1998, Ellison and van Riper 1998, Flinders et al. 2002, Block et al. 2005). However, strong associations with tree variables were observed in GSENM and CARE. Therefore, the following recommendations for both MSO and their primary prey focus specifically on rocky canyon habitats within GSENM and CARE in southern Utah. This study provided strong evidence riparian corridors enabled primary prey species to endure extended periods of low precipitation and drought. Woodrats, brush mice, and deer mice exhibited higher abundances on approximately 20° forested slopes near riparian areas, even in periods of extreme drought. Furthermore, successful MSO nest sites were similarly located in narrow forested canyons with dense riparian vegetation providing sheltered cooler micro-climates. In addition to providing resources to primary prey, forested riparian areas provide MSOs with thermal refuge during extreme temperatures, escape cover from predators, as well as roost and nursery areas for juveniles. Therefore, I recommend management efforts focus on protecting forested riparian corridors within GSENM and CARE, specifically riparian corridors with large diameter trees and tall

shrubs from disturbance. For example, Willey and Willey (2010) suggested limiting cattle grazing within canyons with known MSO territories due to negative impacts of cattle on riparian vegetation. Furthermore, woodrats and brush mice were frequently captured in desert scrub and pinyon juniper habitat types over the duration of this study. In addition to protecting riparian corridors, I recommend management plans focus on maintaining a mixture of vegetation types within canyon environments. Ward and Block (1995) and Sureda and Morison (1998) suggested this may provide a buffer against the effects of small mammal cycles in a particular vegetation type.

Results of this study provided strong evidence that timing and amount of seasonal precipitation affected the primary prey species of MSO. Seasonal precipitation could be used as a metric to determine possible effects on MSO populations. For example, two consecutive years of above average winter precipitation (> 200 mm) might be used to indicate increases in primary prey abundance and result in increased MSO recolonization at extirpated nest sites. Conversely, two to three consecutive years of below average winter precipitation (< 100 mm) could be used as an indicator of low primary prey abundance, shifts in *Cricetid* community diversity, and possible extirpation events of MSO territories in the region.

Tables

Table 2.1. Summary of *Cricetid* rodent abundance, diversity, and seasonal precipitation in Grand Staircase - Escalante National Monument, Utah. Summers 2001 - 2007 and 2015.

Year	WR^1	WFM^2	S^3	H'^4	J^5	D_I^6	AP^7	WP^8	SP^9	DI^{10}
2001	7	40	5	1.438	0.894	0.735	332.7	274.0	58.7	0
2002	1	49	6	1.327	0.741	0.676	143.3	75.0	68.3	1
2003	3	95	6	1.333	0.744	0.665	185.0	158.7	26.3	1
2004	18	176	6	1.029	0.574	0.555	274.3	172.0	102.3	1
2005	29	212	6	0.831	0.464	0.404	565.3	490.3	75.0	0
2006	62	429	8	0.740	0.356	0.346	276.7	191.0	84.7	0
2007	0	90	3	0.502	0.457	0.259	281.6	180.3	101.3	1
2015	6	34	6	1.079	0.602	0.503	320.3	195.0	125.3	1

¹ No. woodrats captured (adjusted for effort)

² No. white-footed mice captured (adjusted for effort).

³ Species richness.

⁴ Shannon's diversity index.

⁵ Shannon's evenness.

⁶ Simpson's diversity index.

⁷ Annual precipitation (mm).

⁸ Winter precipitation (mm).

⁹ Summer precipitation (mm).

¹⁰ Drought Index. (0) = no drought + abnormally dry, (1) drought = moderate drought + severe drought, extreme drought, exceptional drought.

Table 2.2. Summary of *Cricetid* rodent abundance, diversity, and seasonal precipitation in Capitol Reef National Park, Utah. Summers 2013 - 2015.

Year	WR^1	WFM^2	S^3	H^4	J^5	D_1^6	AP^7	WP^8	SP^9	DI^{10}
2013	9	54	6	1.187	0.663	0.584	321.0	107.0	214.0	1
2014	10	5	6	1.632	0.911	0.819	257.6	111.4	146.2	1
2015	24	18	6	1.641	0.916	0.806	330.6	208.0	122.6	0

¹ No. woodrats captured (adjusted for effort)

² No. white-footed mice captured (adjusted for effort).

³ Species richness.

⁴ Shannon's diversity index.

⁵ Shannon's evenness.

⁶ Simpson's diversity index.

⁷ Annual precipitation (mm).

⁸ Winter precipitation (mm).

⁹ Summer precipitation (mm).

¹⁰ Drought Index. (0) = no drought + abnormally dry, (1) drought = moderate drought + severe drought, extreme drought, exceptional drought.

Table 2.3. Woodrat microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 200 individual woodrats captured in Grand Staircase-Escalante National Monument, Utah, Summers 2002-2007 and 2015.

Variable	Used traps ($n = 199$)			Unused traps ($n = 1,434$)			
	Mean	SE	95% CI	Welch's t -test	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	55.0	2.7	50.0, 60.5	($t_{261} = -2.56, p = 0.01$)	62.5	62.5	60.5, 64.6
Inverse canopy cover north (%) ²	55.4	2.7	50.0, 60.7	($t_{251} = -3.99, p < 0.001$)*	66.9	1.0	65.0, 68.9
Tree diameter (cm)	24.2	1.5	21.2, 27.2	($t_{250} = -6.12, p < 0.001$)*	14.2	0.5	13.2, 15.3
Tree height (m)	7.1	0.5	6.1, 8.2	($t_{269} = 2.9, p = 0.004$)	5.5	0.2	5.1, 5.9
Number trees	2.4	0.2	2.0, 2.9	($t_{732} = -5.41, p < 0.001$)*	4.3	0.2	3.8, 4.8
Shrub height (cm)	124.5	5.2	114.3, 134.7	($t_{263} = 6.02, p < 0.001$)*	91.1	2.0	87.2, 95.1
Aspect (degree)	188.6	6.4	176.0, 201.1	($t_{233} = -2.83, p = 0.005$)	207.7	2.4	203.2, 212.4
Slope (degree)	19.2	1.1	17.0, 21.5	($t_{302} = -2.66, p = 0.008$)	22.6	0.6	21.5, 23.8
% Forb/grass cover ³	13.7	1.6	10.5, 16.9	($t_{287} = -2.74, p = 0.006$)	18.6	0.8	17.2, 20.1
% Shrub cover	9.4	1.2	6.9, 11.8	($t_{257} = -0.09, p = 0.9$)	9.5	0.5	8.6, 10.4
% Litter cover	16.0	1.7	12.7, 19.3	($t_{255} = 0.29, p = 0.8$)	15.5	0.6	14.2, 16.7
% Tree ⁴	2.6	0.7	1.2, 4.0	($t_{460} = -5.43, p < 0.001$)*	7.6	0.5	6.5, 8.6
% Water	0.3	0.2	0.0, 0.8	($t_{292} = -0.69, p = 0.5$)	0.5	0.1	0.3, 0.7
% Cactus	0.1	0.1	0.0, 0.2	($t_{995} = -1.09, p = 0.3$)	0.2	0.1	0.0, 0.3
% Bare (no cover)	32.4	2.2	28.1, 36.7	($t_{249} = 2.68, p = 0.007$)	26.3	0.8	24.8, 27.8
% Rock cover (no veg cover)	25.6	2.3	20.9, 30.2	($t_{250} = 1.43, p = 0.2$)	22.0	0.8	20.3, 23.6

* Used traps different than unused traps. Bonferroni correction $p \leq 0.003$.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.4. White-footed mouse microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 1,010 individual white-footed mice captured in Grand Staircase-Escalante National Monument, Utah. Summers 2002-2007 and 2015.

Variable	Used traps ($n = 848$)			Unused traps ($n = 785$)		
	Mean	SE	95% CI	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	55.1	1.4	52.4, 57.8	68.6	1.3	65.9, 71.2
Inverse canopy cover north (%) ²	60.5	1.3	57.9, 63.0	71.0	1.3	68.4, 73.5
Tree diameter (cm)	18.2	0.7	16.7, 19.6	12.5	0.7	11.1, 13.9
Tree height (m)	6.1	0.2	5.6, 6.6	5.3	0.3	4.6, 5.9
Number trees	4.2	0.3	3.6, 4.8	4.0	0.3	3.4, 4.6
Shrub height (cm)	117.3	2.7	112.0, 122.7	71.3	2.4	66.7, 76.0
Aspect (degree)	202.3	3.2	196.0, 208.6	208.8	3.0	202.9, 214.7
Slope (degree)	19.8	0.7	18.5, 21.1	24.8	0.8	23.1, 26.4
% Forb/grass cover ³	18.3	1.0	16.4, 20.2	17.7	1.0	15.8, 19.7
% Shrub cover	8.8	0.6	7.6, 10.0	10.2	0.6	8.9, 11.4
% Litter cover	17.1	0.9	15.4, 18.8	13.9	0.8	12.3, 15.4
% Tree ⁴	6.4	0.7	5.1, 7.7	7.6	0.7	6.2, 9.0
% Water	0.8	0.2	0.4, 1.2	0.1	0.1	0.0, 0.3
% Cactus cover	0.2	0.1	0.0, 0.3	0.1	0.1	0.0, 0.2
% Bare (no cover)	28.5	1.0	26.5, 30.5	25.4	1.0	23.4, 27.4
% Rock cover (no veg cover)	20.0	1.0	17.9, 22.0	25.0	1.2	22.7, 27.4

* Used traps different than unused traps. Bonferroni correction $p \leq 0.003$.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.5. Woodrat microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 41 individual woodrats captured in Capitol Reef National Park, Utah. Summer 2015.

Variable	Used traps (n = 38)			Unused traps (n = 187)		
	Mean	SE	95% CI	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	73.8	6.3	61.0, 86.5	77.6	2.7	72.2, 83.0
Inverse canopy cover north (%) ²	74.8	6.7	61.3, 88.3	80.5	2.6	75.4, 85.6
Tree diameter (cm)	5.4	1.2	2.9, 7.9	5.9	1.0	3.9, 7.8
Tree height (m)	2.8	0.4	2.0, 3.6	2.0	0.2	1.7, 2.3
Number trees	2.2	0.6	1.0, 3.4	1.6	0.2	1.3, 1.9
Shrub height (cm)	184.2	15.9	151.9, 216.5	178.5	7.7	163.3, 193.7
Aspect (degree)	123.6	13.0	97.3, 149.8	131.4	5.5	120.6, 142.2
Slope (degree)	18.9	2.3	14.2, 23.7	22.8	1.1	20.6, 25.1
% Forb/grass cover ³	16.2	3.9	8.4, 24.0	22.4	1.8	18.8, 25.9
% Shrub cover	24.0	5.3	13.3, 34.7	13.5	1.7	10.1, 17.0
% Litter cover	2.5	1.0	0.4, 4.6	2.1	0.5	1.1, 3.2
% Tree ⁴	9.3	3.5	2.1, 16.4	8.6	1.4	5.9, 11.4
% Bare (no cover)	23.7	4.5	14.6, 32.7	29.8	2.0	25.8, 33.8
% Rock cover (no veg cover)	24.4	4.5	15.3, 33.5	23.4	1.9	19.6, 27.2

* Used traps not different than unused traps.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.6. White-footed mouse microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 53 individual white-footed mice captured in Capitol Reef National Park, Utah. Summer 2015.

Variable	Used traps (n = 44)			Unused traps (n = 181)		
	Mean	SE	95% CI	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	69.7	6.4	56.9, 82.6	78.8	2.7	73.4, 84.1
Inverse canopy cover north (%) ²	64.4	6.7	50.9, 78.0	83.3	2.5	78.4, 88.1
Tree diameter (cm)	7.2	2.1	2.9, 11.4	5.5	0.9	3.7, 7.3
Tree height (m)	2.6	0.4	1.7, 3.5	2.0	0.2	1.7, 2.4
Number trees	1.9	0.3	1.2, 2.5	1.7	0.2	1.3, 2.1
Shrub height (cm)	235.1	20.8	193.1, 277.1	165.9	6.6	152.8, 179.0
Aspect (degree)	122.5	11.9	98.4, 146.6	131.9	5.6	120.9, 142.9
Slope (degree)	20.9	2.3	16.2, 25.6	22.4	1.1	20.2, 24.7
% Forb/grass cover ³	18.4	3.4	11.5, 25.3	22.1	1.9	18.4, 25.7
% Shrub cover	25.9	5.0	15.9, 36.0	12.7	1.7	9.4, 16.1
% Litter cover	1.5	0.6	0.2, 2.8	2.4	0.6	1.2, 3.5
% Tree ⁴	8.1	2.6	2.8, 13.3	8.9	1.5	6.0, 11.8
% Bare (no cover)	27.6	4.4	18.8, 36.4	29.1	2.0	25.0, 33.1
% Rock cover (no veg cover)	18.5	4.2	10.1, 26.9	24.8	1.9	21.0, 28.7

* Used traps different than unused traps. Bonferroni correction $p \leq 0.004$.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.7. Differences in Akaike's information criterion for small sample sizes (ΔAIC_c), model weights (w_i), variation explained (R^2_c = conditional, R^2_m = marginal), log-likelihood (L), number of model parameters (k), and F -test results for three top-ranked models and the null model in analyses of *Cricetid* rodent diversity index in Grand Staircase - Escalante National Monument and Capitol Reef National Park, Utah, 2001 - 2007 and 2013 - 2015.

Response ¹	Model ²	ΔAIC_c	w_i	R^2_c	R^2_m	L	k	Type II Wald F -test
H'	SP	0.00	0.272	0.126	0.126	-8.8	4	$F_{1,24} = 3.62, p = 0.07$
H'	AP.lag	1.35	0.138	0.087	0.087	-9.5	4	$F_{1,24} = 2.69, p = 0.11$
H'	WP.lag	1.85	0.108	0.110	0.077	-9.7	4	$F_{1,28} = 2.20, p = 0.15$
H'	Null	2.19	0.091	0.000	0.000	-10.9	4	
J	WP.lag	0.00	0.214	0.265	0.113	6.1	4	$F_{1,27} = 3.62, p = 0.07$
J	AP	1.06	0.126	0.260	0.113	5.6	4	$F_{1,23} = 3.00, p = 0.10$
J	AP.lag	1.10	0.124	0.240	0.780	5.5	4	$F_{1,23} = 2.86, p = 0.10$
J	Null	1.91	0.054	0.144	0.000	4.1	4	
D ₁	WP.lag	0.00	0.257	0.240	0.110	4.6	4	$F_{1,28} = 3.46, p = 0.07$
D ₁	DI(s).lag	1.71	0.117	0.161	0.661	3.8	4	$F_{1,28} = 1.93, p = 0.18$
D ₁	AP.lag	2.17	0.087	0.169	0.052	3.6	4	$F_{1,23} = 1.80, p = 0.19$
D ₁	Null	1.87	0.101	0.166	0.000	2.7	4	

¹ H' = Shannon's diversity index, J = Shannon's evenness, D₁ = Simpson's diversity index.

² SP = summer precipitation, AP = annual precipitation, AP.lag = annual precipitation w/1-year lag effect, WP.lag = winter precipitation w/1-year lag effect, DI(s).lag = drought index at site w/1-year lag effect.

Table 2.8. Effect estimates from linear mixed-effects models. Intercept of model (β_0), standard error of intercept (SE β_0), coefficient of predictor (β_1), standard error of coefficient (SE β_1), 95% confidence interval for effect estimate (95%CI β_1), and estimates of diversity in low precipitation year (Low), average year (Avg), and high precipitation year (High) in analyses of *Peromyscus* rodent diversity in Grand Staircase - Escalante National Monument, Utah, 2001 - 2007 and 2015 and Capitol Reef National Park, Utah, 2013 - 2015.

Response ¹	Model ²	β_0	SE β_0	β_1	SE β_1	95% CI (β_1)	Low ³	Avg	High
H'	SP	2.028	0.529	-0.248	0.117	-0.485, -0.011	1.221	0.972	0.882
H'	AP.lag	2.501	0.922	-0.285	0.166	-0.620, 0.050	1.085	0.925	0.693
H'	WP.lag	1.823	0.586	-0.180	0.116	-0.415, 0.054	1.044	0.885	0.706
J	WP.lag	1.377	0.340	-0.139	0.067	-0.275, -0.001	0.779	0.656	0.519
J	AP	1.683	0.565	-0.176	0.100	-0.379, 0.031	0.809	0.710	0.567
J	AP.lag	1.602	0.530	-0.165	0.095	-0.357, 0.031	0.784	0.692	0.558
D ₁	WP.lag	1.300	0.360	-0.145	0.071	-0.290, -0.000	0.674	0.546	0.401
D ₁	DI(s).lag	0.481	0.070	0.128	0.082	-0.036, 0.296	0.481	0.545	0.608
D ₁	AP.lag	1.347	0.570	-0.137	0.102	-0.342, 0.345	0.665	0.588	0.476

¹ H' = Shannon's diversity index, J = Shannon's evenness, D₁ = Simpson's diversity index.

² SP = summer precipitation, AP = annual precipitation, AP.lag = annual precipitation w/1-year lag effect, WP.lag = winter precipitation w/1-year lag effect, DI(s).lag = drought index at site w/1-year lag effect.

³ See Table B.5 for Low, Avg, High predictor year values.

Table 2.9. Differences in Akaike's information criterion for small sample sizes (ΔAIC_c), model weights (w_i), variation explained ($R^2_c =$ conditional, $R^2_m =$ marginal), log-likelihood (L), number of model parameters (k), and Chi-squared test results for top-ranked models and the null, in analyses of *Citricetid* rodent abundance in Grand Staircase - Escalante National Monument, Utah, 2001 - 2007 & 2015 and Capitol Reef National Park, Utah, 2013 - 2015.

Response ¹	Model ²	ΔAIC_c	w_i	R^2_c	R^2_m	L	k	Type II Wald Chi-squared tests*
Pri.prey	WP.lag	0.00	0.56	0.26	0.21	-138.4	4	($\chi^2_1 = 12.1, p < 0.001$)
Pri.prey	WP.lag + DI(s).lag	1.77	0.23	0.31	0.20	-137.9	5	($\chi^2_1 = 12.6, p < 0.001$) ($\chi^2_1 = 1.28, p = 0.26$)
Pri.prey	Null	9.4	<0.01	0.23	0.00	-144.1	4	
WR	DI(s)	0.00	0.36	0.17	0.17	-78.7	4	($\chi^2_1 = 10.5, p = 0.001$)
WR	AP * DI(s)	1.25	0.19	0.34	0.34	-76.3	6	($\chi^2_1 = 0.80, p < 0.37$) ($\chi^2_1 = 5.76, p = 0.02$)
WR	AP + DI(s)	1.90	0.14	0.22	0.22	-78.2	5	($\chi^2_1 = 0.97, p < 0.32$) ($\chi^2_1 = 5.77, p = 0.02$)
WR	Null	6.87	0.01	0.00	0.00	-83.2	4	
WFM	WP.lag	0.00	0.48	0.24	0.14	-135.7	4	($\chi^2_1 = 8.09, p = 0.004$)
WFM	WP.lag + DI(s).lag	2.20	0.16	0.29	0.13	-135.4	5	($\chi^2_1 = 8.23, p = 0.004$) ($\chi^2_1 = 0.78, p = 0.38$)
WFM	Null	6.0	0.02	0.32	0.00	-139.7	4	

¹ Pri.prey = No. primary prey, WR = No. woodrats, WFM = No. white-footed mice.

² WP.lag = winter precipitation w/1-year lag effect, DI(s).lag = drought index at site w/1-year lag effect, AP.lag = annual precipitation w/1-year lag effect, AP = annual precipitation, DI(s) = drought index at site.

* Chi-squared test result for each fixed-effect in the model respectively.

Table 2.10. Effect estimates from top Generalized Linear Mixed-Effects Models. Intercept of model (β_0), standard error of intercept (SE β_0), coefficient of predictors (β_1 , β_2), standard error of coefficients (SE β_1 , β_2), 95% confidence interval for effect estimates (95%CI β_1 , β_2), coefficient of interaction (β_3), standard error of interaction (SE β_3) and estimates of *Cricetid* rodent abundance in low precipitation year (Low), average year (Avg), and high precipitation year (High) in analyses of *Cricetid* rodent abundance in Grand Staircase - Escalante National Monument, Utah, 2001 - 2007 and 2015 and Capitol Reef National Park, Utah, 2013 - 2015.

Response ¹	Model ²	β_0	SE (β_0)	β_1	SE (β_1)	95% CI (β_1)	β_2	SE (β_2)	95% CI (β_2)	β_3	SE (β_3)	95% CI (β_3)	Low	Avg	High
Pri.prey	WP.lag	-0.949	1.270	0.898	0.258	(0.391, 1.412)							2.926	3.716	4.610
Pri.prey	WP.lag + DI(s).lag	-2.383	1.759	1.101	0.310	(0.481, 1.707)	0.452	0.399	(-0.343, 1.233)				2.372	3.569	4.891
WR	DI(s)	2.319	0.311	-1.359	0.421	(-2.208, -0.546)							2.319		0.960
WR	AP * DI(s)	5.199	5.529	-0.491	0.939	(-2.270, 1.491)	-18.62	8.707	(-36.14, -1.528)	3.088	1.535	(0.073, 6.178)	-0.413 ^a		3.076 ^b
WR	AP + DI(s)	-2.665	5.055	0.857	0.871	(-0.796, 2.649)	-1.130	0.471	(-2.078, -0.219)				0.496 ^a		1.647 ^b
WFM	WP.lag	-1.059	1.468	0.864	0.304	(0.268, 1.466)							2.673	3.434	4.295
WFM	WP.lag + DI(s).lag	-2.321	2.035	1.039	0.362	(0.308, 1.742)	0.403	0.457	(-0.512, 1.295)				2.164	3.280	4.516

¹ Pri.prey = No. primary prey, WR = No. woodrats, WFM = No. white-footed mice.

² WP.lag = winter precipitation w/1-year lag effect, DI(s).lag - drought index at site w/1-year lag effect, AP = annual precipitation, DI(s) = drought index at site

³ See Table B.5 for Low, Avg, High predictor year values.

^a Estimated woodrat abundance in low annual precipitation year combined with drought effect at site

^b Estimated woodrat abundance in high annual precipitation year combined with drought effect at site.

Table 2.11. Percent occurrence of woodrat and white-footed mouse species groups in used and unused traps relative to six habitat types in Grand Staircase - Escalante National Monument, Utah. Summers 2002 - 2007 and 2015.

Habitat type ¹	Woodrats (<i>n</i> = 200)		White-footed mice (<i>n</i> = 1,010)	
	Used (<i>n</i> = 199)	Unused (<i>n</i> = 1,434)	Used (<i>n</i> = 848)	Unused (<i>n</i> = 785)
Riparian woodland	28.93	39.05	39.98	35.67
Desert scrub	33.50	34.94	33.17	34.90
Pinyon - Juniper woodland	30.46	14.30	15.30	17.71
Gamble's oak woodland	2.03	4.95	6.91	3.82
Grassland	0.51	4.67	2.37	5.61
Rock - talus - cliff	4.57	2.09	2.27	2.29

¹ See study area

Table 2.12. Percent occurrence of woodrat and white-footed mouse species in used and unused traps relative to seven habitat types in Capitol Reef National Park, Utah. Summer 2015.

Habitat type ¹	Woodrats (<i>n</i> = 41)		White-footed mice (<i>n</i> = 53)	
	Used (<i>n</i> = 38)	Unused (<i>n</i> = 187)	Used (<i>n</i> = 44)	Unused (<i>n</i> = 181)
Riparian woodland	36.59	16.58	33.96	16.57
Desert scrub	29.27	48.13	43.40	45.86
Pinyon - Juniper woodland	9.76	21.39	13.21	20.44
Mountain scrub	12.20	10.16	5.66	11.60
Gamble's oak woodland	4.88	1.07	3.77	1.66
Grassland	4.88	2.14	0.00	2.76
Rock - talus - cliff	2.44	0.53	0.00	1.10

¹ See study area

Table 2.13. Canyon mouse microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 62 canyon mice captured in Grand Staircase-Escalante National Monument, Utah, summers 2002-2007 and 2015.

Variable	Used traps ($n = 62$)			Unused traps ($n = 1,571$)		
	Mean	SE	95% CI	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	58.1	5.4	47.3, 68.8	61.8	1.0	59.8, 63.7
Inverse canopy cover north (%) ²	63.3	5.2	53.0, 73.7	65.6	0.9	63.7, 67.5
Tree diameter (cm)	17.2	2.8	11.7, 22.7	15.4	0.5	14.3, 16.4
Tree height (m)	4.2	0.6	2.9, 5.5	5.8	0.2	5.4, 6.2
Number trees	1.4	0.3	0.7, 2.0	4.2	0.2	3.7, 4.6
Shrub height (cm)	101.0	6.9	87.2, 114.8	95.0	2.0	91.1, 98.8
Aspect (degree)	205.6	7.2	191.2, 220.0	205.4	2.3	201.0, 109.9
Slope (degree)	28.8	3.3	22.1, 35.5	22.0	0.5	20.9, 23.0
% Forb/grass cover ³	12.6	2.8	7.0, 18.2	18.2	0.7	16.9, 19.6
% Shrub cover	8.8	1.8	5.3, 12.3	9.5	0.5	8.6, 10.4
% Litter cover	5.6	1.5	2.7, 8.5	15.9	0.6	14.7, 17.1
% Tree ⁴	5.3	2.3	0.7, 9.9	7.0	0.5	6.0, 8.0
% Water	0.0	0.0	0.0, 0.0	0.5	0.1	0.3, 0.7
% Cactus	0.0	0.0	0.0, 0.1	0.1	0.1	0.0, 0.3
% Bare (no cover)	30.8	3.8	23.3, 38.3	26.9	0.7	25.4, 28.3
% Rock cover (no veg cover)	40.0	4.7	27.7, 46.3	21.8	0.8	20.3, 23.4

* Used traps different than unused traps with Bonferroni correction of $p \leq 0.003$.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.14. Brush mouse microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 672 brush mice captured in Grand Staircase-Escalante National Monument, Utah, Summers 2002-2007 and 2015.

Variable	Used traps ($n = 672$)			Unused traps ($n = 961$)			
	Mean	SE	95% CI	Welch's t -test	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	52.1	1.5	49.1, 55.1	($t_{1321} = -8.26, p < 0.001$)*	68.3	1.2	65.9, 70.7
Inverse canopy cover north (%) ²	57.6	1.5	54.7, 60.4	($t_{1407} = -7.18, p < 0.001$)*	71.1	1.2	68.7, 73.4
Tree diameter (cm)	19.7	0.8	18.0, 21.3	($t_{1364} = 6.80, p < 0.001$)*	12.5	0.6	11.2, 13.8
Tree height (m)	6.5	0.3	6.0, 7.0	($t_{1602} = 3.53, p < 0.001$)*	5.2	0.3	4.6, 5.7
Number trees	4.1	0.3	3.5, 4.8	($t_{1457} = 0.22, p = 0.8$)	4.0	0.3	3.5, 4.6
Shrub height (cm)	122.9	3.1	116.9, 129.0	($t_{1313} = 12.43, p < 0.001$)*	75.8	2.2	71.4, 80.2
Aspect (degree)	201.7	3.7	194.5, 209.0	($t_{1324} = -1.37, p = 0.2$)	208.0	2.7	202.7, 213.4
Slope (degree)	18.1	0.7	16.7, 19.4	($t_{1603} = -6.89, p < 0.001$)*	25.1	0.8	23.6, 26.6
% Forb/grass cover ³	19.2	1.1	17.1, 21.4	($t_{1412} = 1.43, p = 0.2$)	17.2	0.9	15.5, 18.9
% Shrub cover	9.0	0.7	7.6, 10.5	($t_{1333} = -0.85, p = 0.4$)	9.8	0.5	8.7, 10.9
% Litter cover	18.3	1.0	16.3, 20.3	($t_{1291} = 3.84, p < 0.001$)*	13.6	0.7	12.2, 15.0
% Tree ⁴	6.2	0.7	4.8, 7.6	($t_{1511} = -1.30, p = 0.2$)	7.5	0.7	6.2, 8.8
% Water	1.0	0.2	0.5, 1.4	($t_{801} = 3.25, p = 0.001$)*	0.2	0.1	0.0, 0.3
% Cactus	0.2	0.1	0.0, 0.4	($t_{1046} = 0.47, p = 0.6$)	0.1	0.1	0.0, 0.2
% Bare (no cover)	28.4	1.1	26.1, 30.6	($t_{1422} = 1.56, p = 0.1$)	26.1	0.9	24.2, 27.8
% Rock cover (no veg cover)	17.8	1.1	15.7, 20.0	($t_{1588} = -5.03, p < 0.001$)*	25.6	1.1	23.4, 27.8

* Used traps different than unused traps. Bonferroni correction of $p \leq 0.003$.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.15. Pinyon mouse microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 65 pinyon mice captured in Grand Staircase-Escalante National Monument, Utah, summers 2002-2007 and 2015.

Variable	Used traps ($n = 65$)			Unused traps ($n = 1,568$)		
	Mean	SE	95% CI	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	65.8	4.6	56.6, 75.0	61.4	1.0	59.5, 63.4
Inverse canopy cover north (%) ²	71.2	4.5	62.2, 80.2	65.3	1.0	63.4, 67.2
Tree diameter (cm)	15.4	2.2	11.0, 19.7	15.5	0.5	14.4, 16.5
Tree height (m)	5.2	0.7	3.9, 6.6	5.7	0.2	5.3, 6.1
Number trees	3.0	0.5	1.9, 4.1	4.1	0.2	3.7, 4.6
Shrub height (cm)	105.4	9.7	86.1, 124.7	94.8	1.9	91.0, 98.6
Aspect (degree)	189.7	10.5	168.6, 210.8	106.1	2.3	201.7, 210.5
Slope (degree)	21.2	2.5	16.1, 26.3	22.3	0.5	21.2, 23.3
% Forb/grass cover ³	10.9	3.0	4.9, 17.0	18.3	0.7	16.9, 19.7
% Shrub cover	9.9	1.8	6.3, 13.4	9.4	0.5	8.6, 10.3
% Litter cover	12.2	2.6	7.0, 17.4	15.7	0.6	14.4, 16.9
% Tree ⁴	9.1	3.2	2.8, 15.4	6.9	0.5	5.9, 7.8
% Water	0.3	0.3	0.0, 0.9	0.5	0.1	0.3, 0.7
% Cactus	0.3	0.3	0.0, 0.10	0.1	0.1	0.0, 0.2
% Bare (no cover)	29.8	3.8	22.16, 37.5	26.9	0.7	25.4, 28.3
% Rock cover (no veg cover)	27.4	4.7	18.0, 36.9	22.2	0.8	20.6, 23.8

* Used traps not different than unused traps.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.16. Woodrat and white-footed mouse species group microhabitat comparison. Microhabitat components measured within 10-m diameter circular plots centered on 199 traps used by woodrats and 848 traps used by white-footed mice captured in Grand Staircase-Escalante National Monument, Utah, summers 2002-2007 and 2015.

Variable	Woodrats ($n = 200$)			White-footed mice ($n = 1,010$)		
	Mean	SE	95% CI	Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	55.2	2.7	50.0, 60.5	55.1	1.4	52.4, 57.8
Inverse canopy cover north (%) ²	55.4	2.7	50.0, 60.7	60.5	1.3	57.9, 63.0
Tree diameter (cm)	24.2	1.5	21.2, 27.2	18.2	0.7	16.7, 19.6
Tree height (m)	7.1	0.5	6.1, 8.2	6.1	0.2	5.6, 6.6
Number trees	2.4	0.2	2.0, 2.9	4.2	0.3	3.6, 4.8
Shrub height (cm)	124.5	5.2	114.3, 134.7	117.3	2.7	112.0, 122.7
Aspect (degree)	188.6	6.4	176.0, 201.1	202.3	3.2	196.0, 208.6
Slope (degree)	19.2	1.1	17.0, 21.5	19.8	0.7	18.5, 21.1
% Forb/grass cover ³	13.7	1.6	10.5, 16.9	18.3	1.0	16.4, 20.2
% Shrub cover	9.4	1.2	6.9, 11.8	8.8	0.6	7.6, 10.0
% Litter cover	16.0	1.7	12.7, 19.3	17.1	0.9	15.4, 18.8
% Tree ⁴	2.6	0.7	1.2, 4.0	6.4	0.7	5.1, 7.7
% Water	0.3	0.2	0.0, 0.8	0.8	0.2	0.4, 1.2
% Cactus	0.1	0.1	0.0, 0.2	0.2	0.1	0.0, 0.3
% Bare (no cover)	32.4	2.2	28.1, 36.7	28.5	1.0	26.5, 30.5
% Rock cover (no veg cover)	25.6	2.3	20.9, 30.2	20.0	1.0	17.9, 22.0

* Used traps different than unused traps. Bonferroni correction $p \leq 0.003$.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table 2.17. Percent occurrence of white-footed mouse species relative to six habitat types in Grand Staircase - Escalante National Monument, Utah. Summers 2002 - 2007 and 2015.

Habitat type ¹	Brush mouse (<i>n</i> = 672)	Deer mouse (<i>n</i> = 194)	Pinyon mouse (<i>n</i> = 65)	Canyon mouse (<i>n</i> = 62)
Riparian woodland	45.00	39.20	20.00	12.90
Desert scrub	30.00	33.50	32.30	71.00
Pinyon - Juniper woodland	13.80	13.40	37.00	9.70
Gamble's oak woodland	7.60	6.20	3.10	1.60
Grassland	1.80	4.10	4.60	1.60
Rock - talus - cliff	1.80	3.60	3.00	3.20

¹ See study area

Table 2.18. Five most common woody plant species present within 10-m diameter circular plots centered on traps used by white-footed mouse species in Grand Staircase-Escalante National Monument, Utah. Summers 2002-2007 and 2015.

Brush mouse (<i>n</i> = 672)	No. ¹	% ²
Sage (<i>Artemisia</i> spp.)	243	18.7
Gambel's oak (<i>Quercus gambelii</i>)	191	14.7
Utah juniper (<i>Juniperus osteosperma</i>)	124	9.5
Willow (<i>Salix</i> spp.)	121	9.3
Fremont cottonwood (<i>Populus fremontii</i>)	107	8.2
<hr/>		
Deer mouse (<i>n</i> = 197)		
Sage (<i>Artemisia</i> spp.)	56	13.7
Gambel's oak (<i>Quercus gambelii</i>)	50	12.2
Utah juniper (<i>Juniperus osteosperma</i>)	40	9.8
Fremont cottonwood (<i>Populus fremontii</i>)	36	8.8
Willow (<i>Salix</i> spp.)	34	8.3
<hr/>		
Pinyon mouse (<i>n</i> = 65)		
Sage (<i>Artemisia</i> spp.)	38	24.2
Utah juniper (<i>Juniperus osteosperma</i>)	19	12.1
Pinyon pine (<i>Pinus edulis</i>)	18	11.5
Green ephedra (<i>Ephedra viridis</i>)	15	9.6
Silver Buffaloberry (<i>Shepherdia argenta</i>)	11	7.0
<hr/>		
Canyon mouse (<i>n</i> = 62)		
Green ephedra (<i>Ephedra viridis</i>)	30	33.3
Silver buffaloberry (<i>Shepherdia argenta</i>)	16	17.8
Pinyon pine (<i>Pinus edulis</i>)	9	10.0
Squawbush (<i>Rhus trilobata</i>)	7	7.8
Gambel's oak (<i>Quercus gambelii</i>)	4	4.4

¹ Number occurrences of woody plant species at used traps.

² Relative abundance of woody plant species at used traps.

Table 2.19. Five most common woody plant species present within 10-m diameter circular plots centered on traps used by woodrat and white-footed mouse species in Grand Staircase-Escalante National Monument, Utah. Summers 2002-2007 and 2015.

	No. ¹	% ²
Woodrats (<i>n</i> = 200)		
Sage (<i>Artemisia spp.</i>)	108	23.1
Utah juniper (<i>Juniperus osteosperma</i>)	62	13.3
Green ephedra (<i>Ephedra viridis</i>)	57	12.2
Gambel's oak (<i>Quercus gambelii</i>)	35	7.5
Pinyon pine (<i>Pinus edulis</i>)	36	7.7
White-footed mice (<i>n</i> = 1,010)		
Sage (<i>Artemisia spp.</i>)	267	16.7
Gambel's oak (<i>Quercus gambelii</i>)	227	14.2
Willow (<i>Salix spp.</i>)	136	8.5
Pinyon pine (<i>Pinus edulis</i>)	133	8.3
Green ephedra (<i>Ephedra viridis</i>)	129	8.1

¹ Number occurrences of woody plant species at used traps.

² Relative abundance of woody plant species at used traps.

Table 2.20. Five most common woody plant species present within 10-m diameter circular plots centered on traps used by woodrat and white-footed mouse species in Capitol Reef National Park, Utah. Summer 2015.

Woodrats ($n = 41$)	No. ¹	% ²
Utah service berry (<i>Amelanchier utahensis</i>)	19	16.1
Green ephedra (<i>Ephedra viridis</i>)	17	14.4
Hackberry (<i>Celtis reticulata</i>)	15	12.7
Squawbush (<i>Rhus trilobata</i>)	14	11.9
Singleleaf ash (<i>Fraxinus anomala</i>)	12	10.2
<hr/>		
White-footed mice ($n = 53$)		
Utah service berry (<i>Amelanchier utahensis</i>)	19	13.9
Squawbush (<i>Rhus trilobata</i>)	18	13.1
Green ephedra (<i>Ephedra viridis</i>)	16	11.7
Hackberry (<i>Celtis reticulata</i>)	15	10.9
Silver buffaloberry (<i>Shepherdia argenta</i>)	14	10.2

¹ Number occurrences of woody plant species at used traps.

² Relative abundance of woody plant species at used traps.

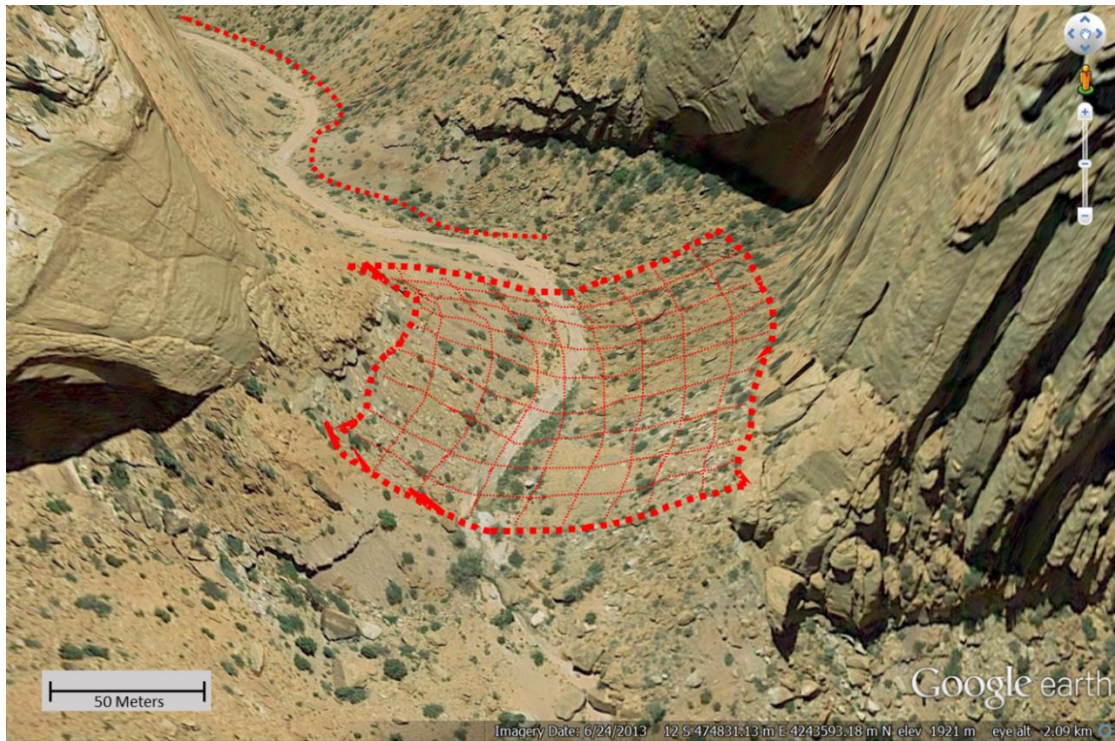
Figures

Figure 2.1. Example of trapping grid and line-transect employed at study sites in Grand Staircase – Escalante National monument and Capitol Reef National Park, Utah, 2001 - 2007 and 2013 - 2015. Image depicts trap grid and line-transect employed in Upper Spring Canyon in Capitol Reef National park, Utah. Summers 2013 – 2015.

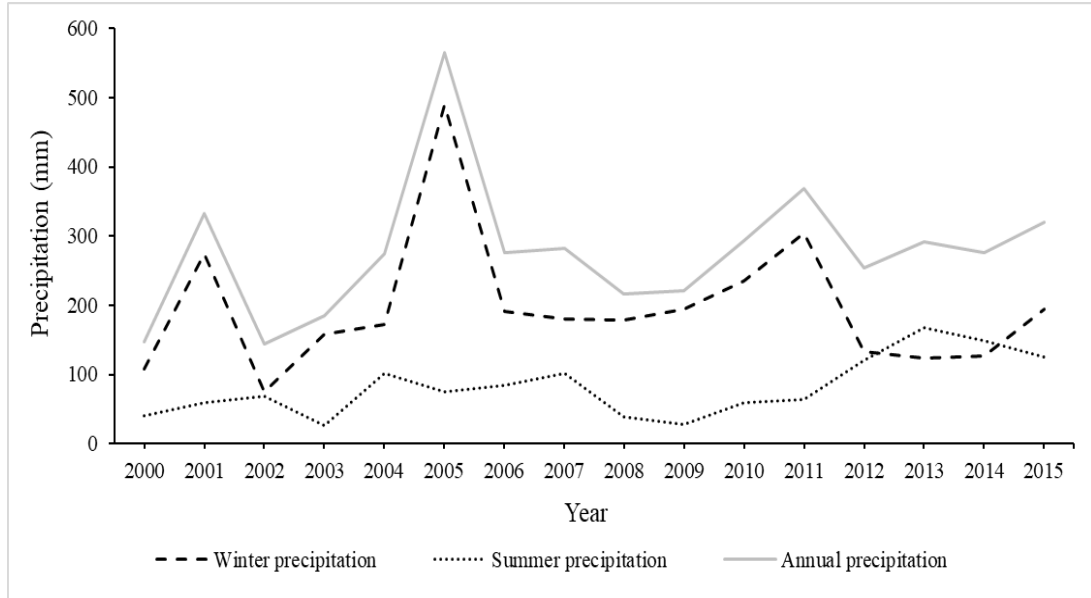


Figure 2.2. Seasonal precipitation trends over 15 years in Grand Staircase – Escalante National Monument, Utah. 2000 – 2015.

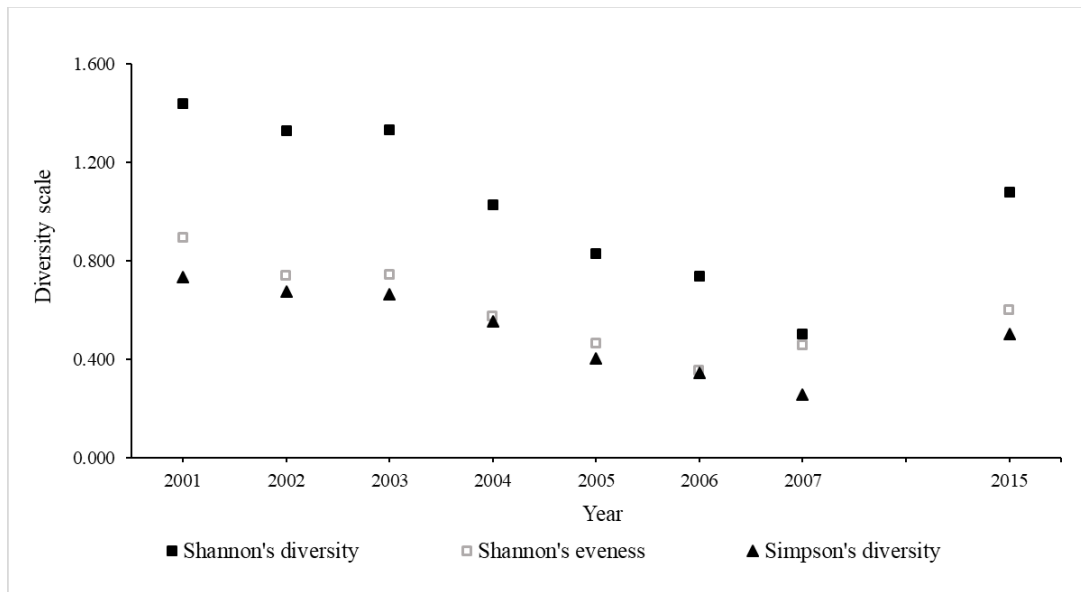


Figure 2.3. *Cricetid* rodent Diversity indices in Grand Staircase – Escalante National Monument, Utah. Summers 2001 – 2007 and 2015.

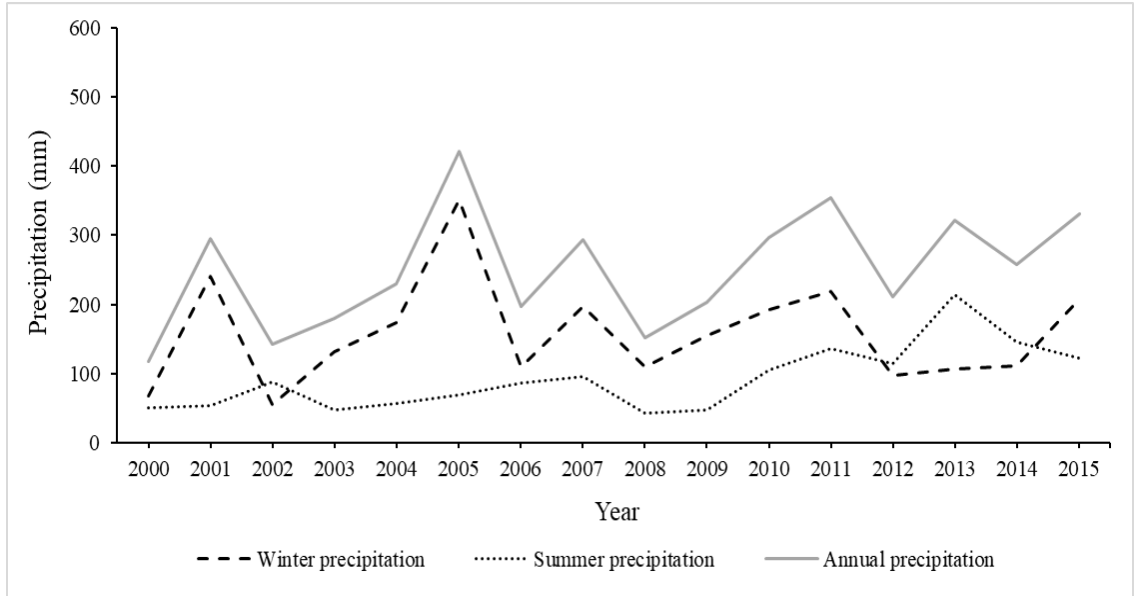


Figure 2.4. Seasonal precipitation trends over 15 years in Capitol Reef National Park, Utah. 2000 – 2015.

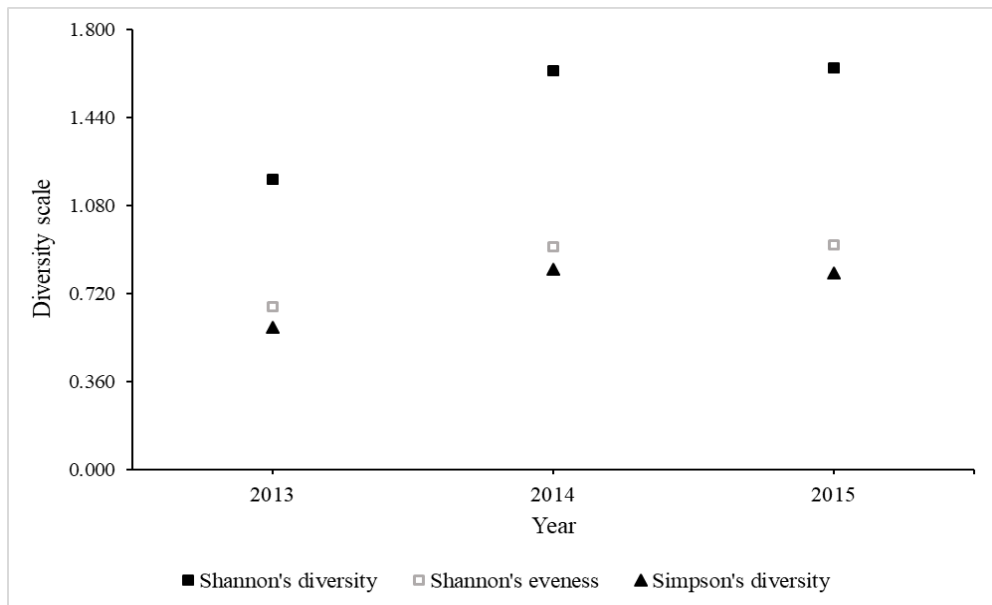


Figure 2.5. *Cricetid* rodent diversity indices in Capitol Reef National Park, Utah. Summers 2013 – 2015.

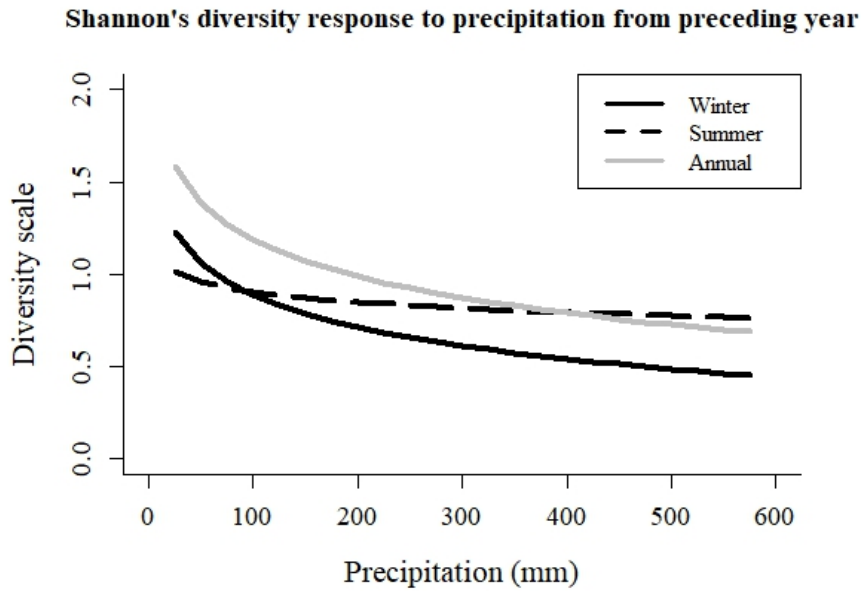


Figure 2.6. Shannon's diversity index relative to precipitation from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

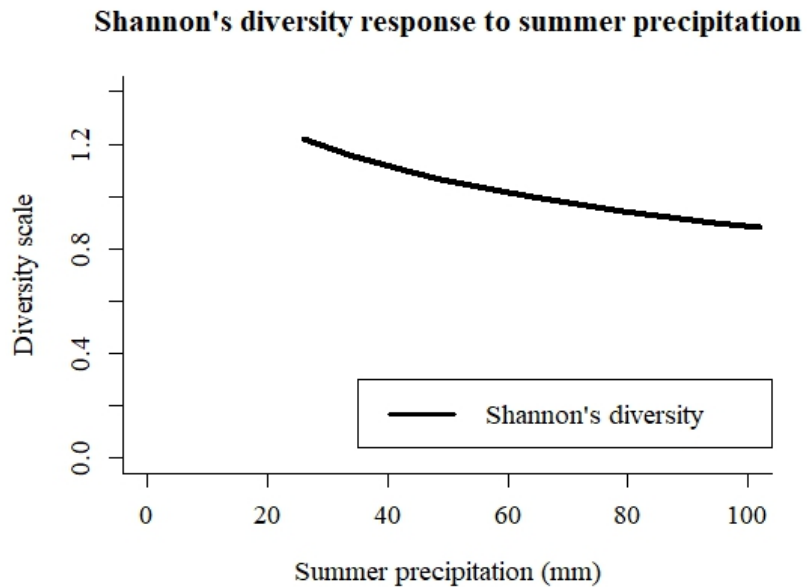


Figure 2.7. Shannon's diversity index relative to summer precipitation in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

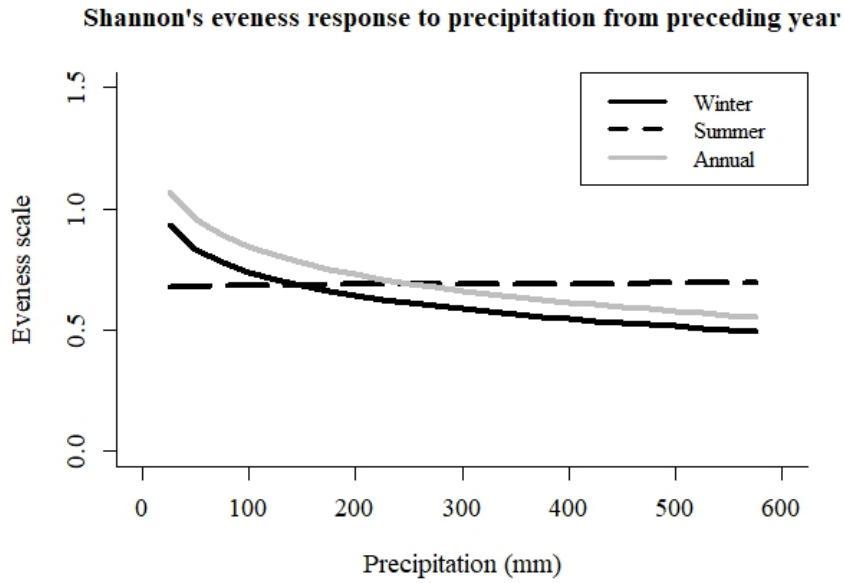


Figure 2.8. Shannon's Evenness relative to precipitation from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

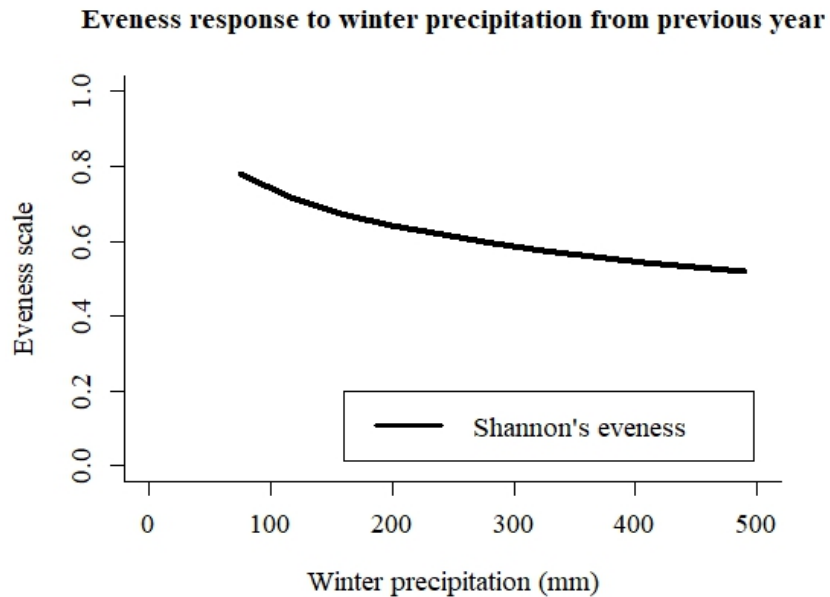


Figure 2.9. Shannon's Evenness relative to winter precipitation from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

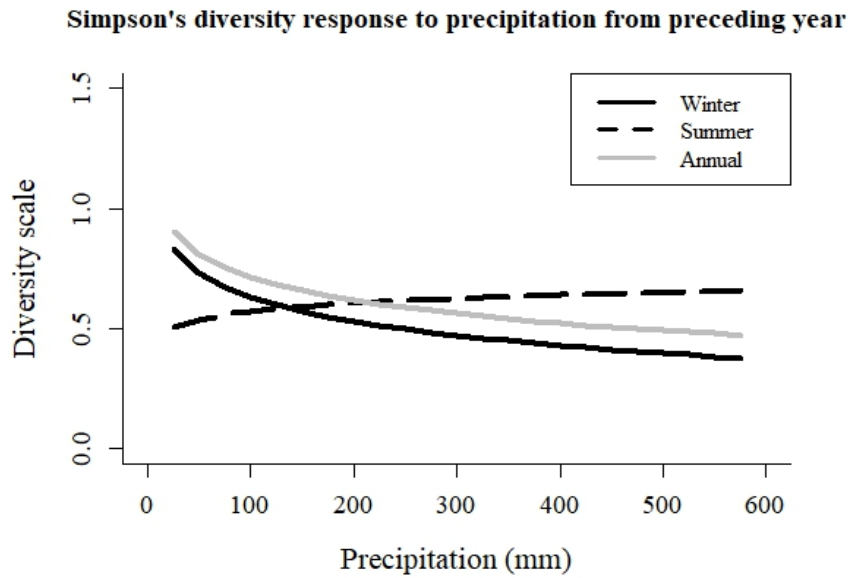


Figure 2.10. Simpson's diversity index relative to precipitation from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

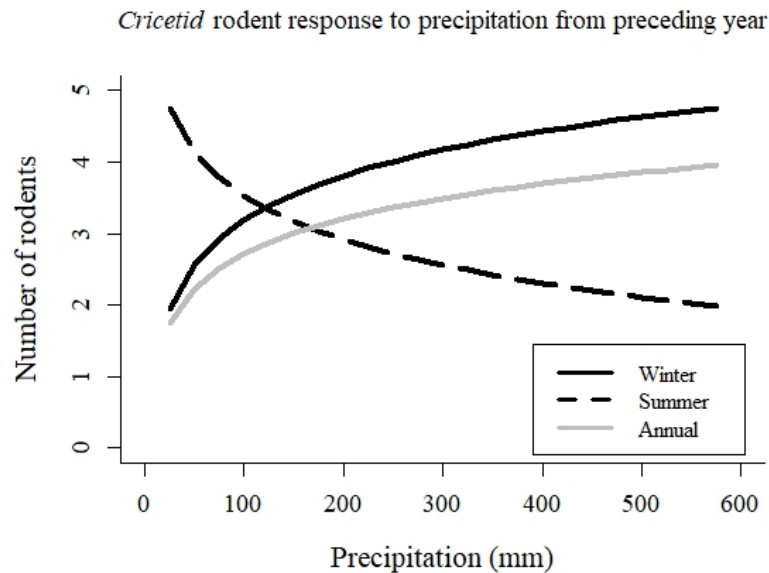


Figure 2.11. Primary prey species abundance relative to precipitation from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

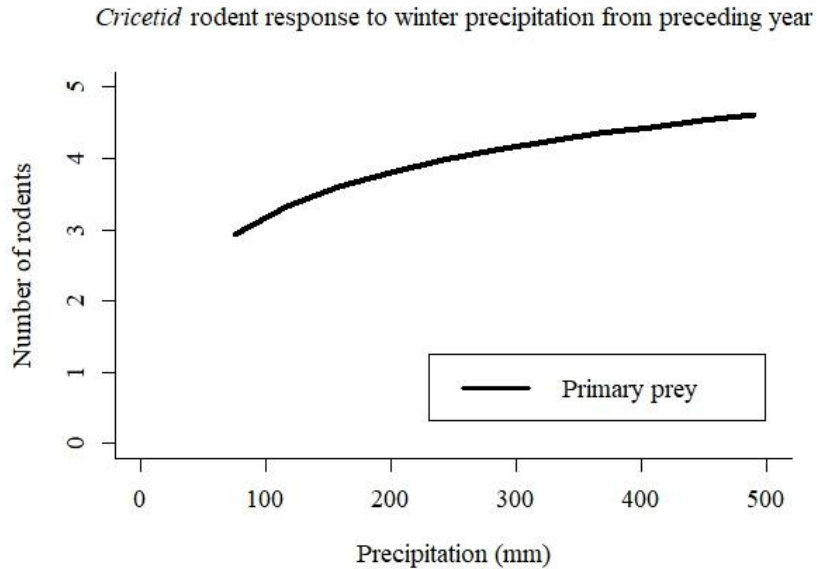


Figure 2.12. Primary prey species abundance relative to winter precipitation from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

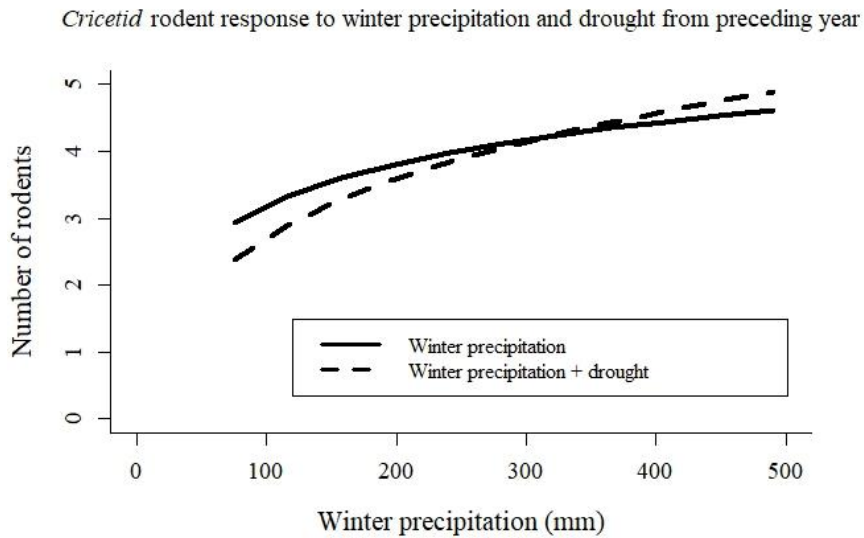


Figure 2.13. Primary prey species abundance relative to winter precipitation and drought index from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

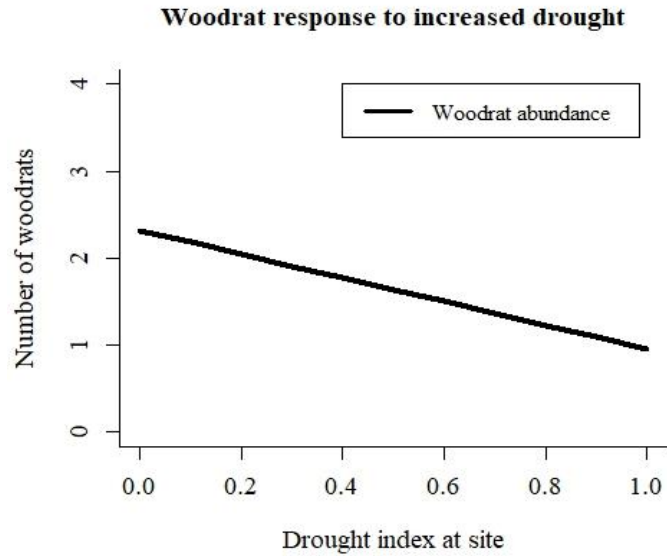


Figure 2.14. Woodrat abundance relative to increased drought in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

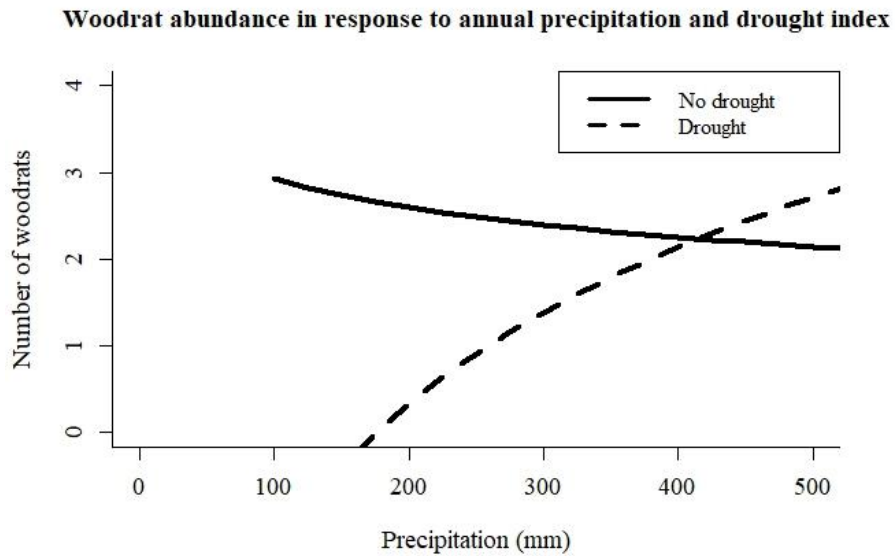


Figure 2.15. Woodrat abundance relative to winter precipitation and drought index in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

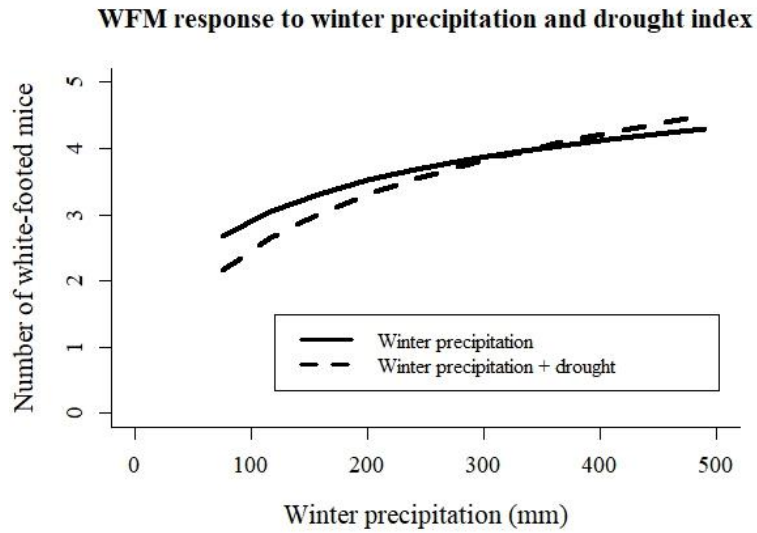


Figure 2.16. White-footed mouse abundance relative to winter precipitation and drought index from the preceding year in Grand Staircase – Escalante National Monument and Capitol Reef National Park, Utah. 2001 – 2007 and 2013 – 2015.

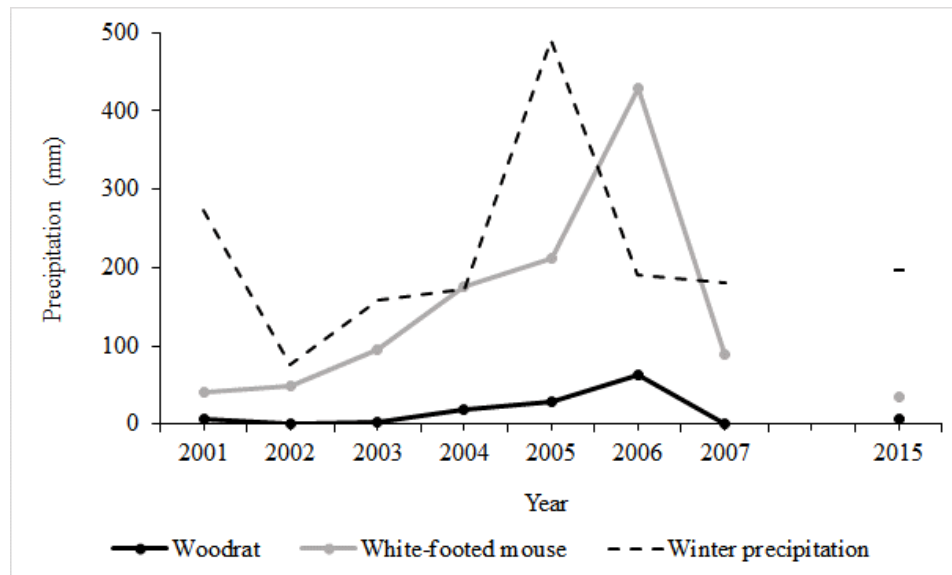


Figure 2.17. Primary prey captures relative to winter precipitation in Grand Staircase – Escalante National Monument, Utah. 2001 – 2007 and 2015.

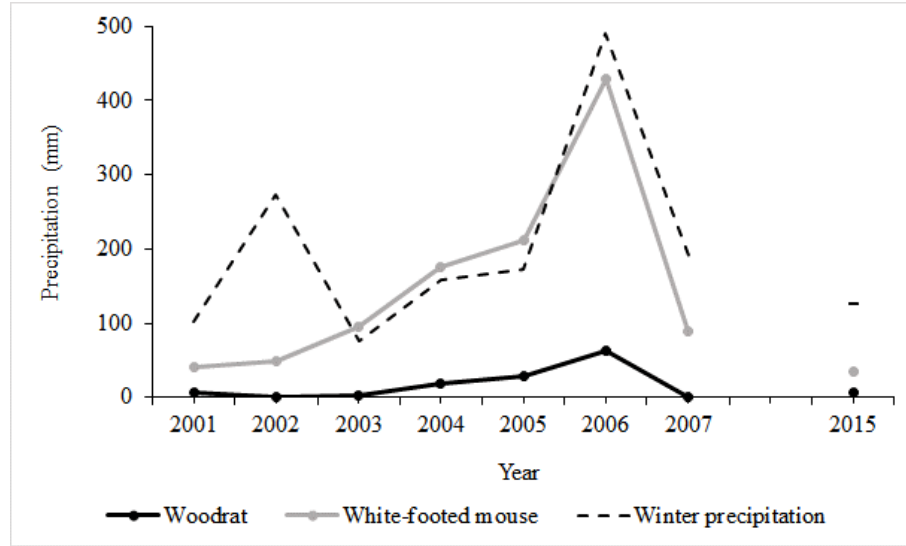


Figure 2.18. Primary prey captures relative to winter precipitation with a one-year lag effect in Grand Staircase – Escalante National Monument, Utah. 2001 – 2007 and 2015.

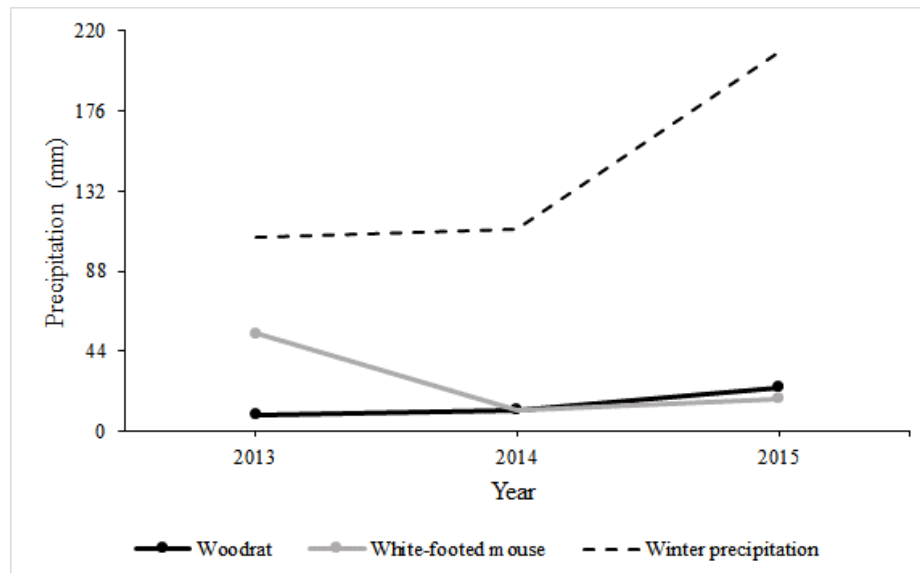


Figure 2.19. Primary prey captures relative to winter precipitation in Capitol Reef National Park, Utah. 2013 – 2015.

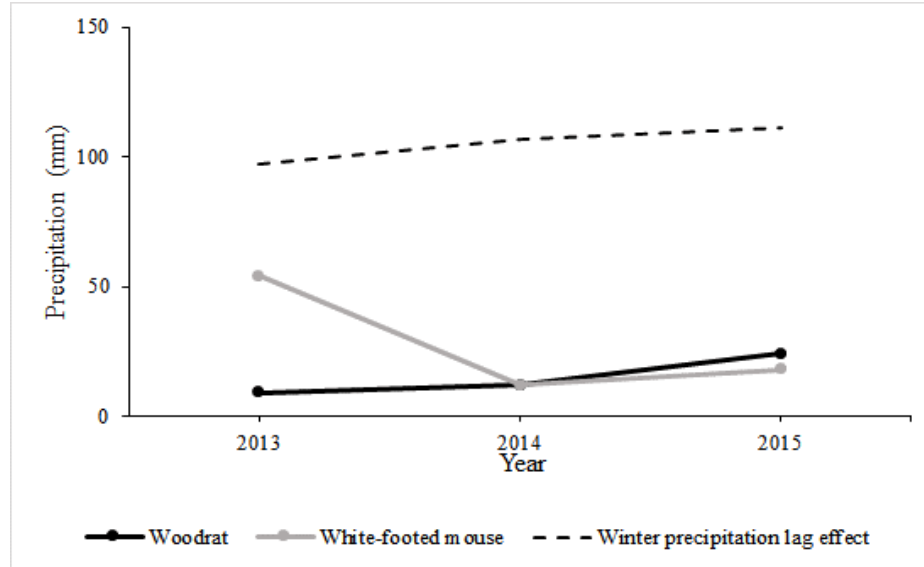


Figure 2.20. Primary prey captures relative to winter precipitation with a one-year lag effect in Capitol Reef National Park, Utah, 2013 – 2015.

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CHAPTER THREE

GENERAL CONCLUSIONS

The objectives of this thesis were to investigate the influence of microhabitat and the effects of seasonal precipitation on the population ecology of the primary prey species of Mexican Spotted Owls (MSO) inhabiting rocky canyon habitats within the canyonlands region of southern Utah. Using a seven-year historic dataset and three years of newly collected data, the nocturnal small mammal communities at three study sites in Grand Staircase – Escalante National Monument (GSENM) and five study sites in Capitol Reef National Park (CARE) were described and habitat associations among these species were investigated. *Cricetid* rodents accounted for approximately 90% of all captures indicating they were the most abundant nocturnal small mammals and potential prey available for MSO in the canyonlands region. Brush mice (*Peromyscus boylei*) and American deer mice (*P. maniculatus*) were the most ubiquitous nocturnal small mammals observed in the region.

Results of habitat analyses provided strong support for the hypothesis that increased vegetation cover was an important habitat component for primary prey species of MSO. Evidence also suggested vegetation cover provided by trees and shrubs in riparian areas could act as a buffer to variation in rainfall and enable primary prey to endure periods of drought. However, results did not support hypotheses that percent rock cover was an important habitat component for all primary prey species at the microsite scale (i.e., vicinity of captures). For example, canyon mice (*P. crinitus*) were

strongly associated with greater percent rock cover and brush mice (*P. boyleyi*) were strongly associated with less rock cover at capture sites. Vegetation cover of large trees may have provided an alternate cover component in the vicinity where *Cricetid* rodents foraged and could have explained lack of association with overhead rock cover at capture sites. Microhabitat analyses also indicated that *Cricetid* rodents exhibited spatial and resource separation among species that may have minimized interspecific competition and enabled coexistence in rocky canyon environments with limited resources.

This study provided strong evidence supporting the hypothesis that timing and amount of winter precipitation was a primary driver of *Cricetid* rodent community demographics within the canyonlands region during this study. Winter precipitation explained significantly more variation in *Cricetid* rodent abundance across all study sites compared to annual and summer precipitation. However, results did not support the hypothesis that increased winter precipitation increased diversity in both study areas. Increased winter precipitation following two consecutive years of low winter precipitation combined with exceptional drought appeared to be associated with declines in the *Cricetid* community from eight species to three species and reduced community diversity in GSENM. In CARE, *Cricetid* abundance and diversity increased following greater winter precipitation, however these results were not detected in models due to low sample sizes.

This study provided evidence that increased winter precipitation resulted in increased primary prey abundances which could lead to potential MSO recolonization of abandoned territories. In CARE, following the greatest amount of winter precipitation

(208 mm) and highest woodrat abundances recorded (i.e., 2015), Bitter Creek Divide Canyon was re-occupied by a single male MSO in 2015, and a nesting pair was observed the following summer in 2016.

Further Recommendations

This long-term data set provides options for numerous Post hoc analyses, which were outside the scope of this thesis. Further examination of the small mammal capture data could provide more insight into *Cricetid* rodent population and community dynamics as well as identify further mechanisms of small mammal abundance and variance over time such as a two or three year lag effect observed by Ernest et al. (2000). Subtracting the outlier year 2005 from the data set to further quantify winter precipitation and *Cricetid* rodent abundance relationships is an option which may further identify small mammal responses to seasonal precipitation. Examining the vegetation data to see if there was a vegetation response to the record precipitation observed in 2005 might provide evidence of response of primary producers to extreme precipitation. The data set was standardized to only include the first four days of capture effort each year to ensure abundances were comparable across years. The original data set could be used to conduct mark-recapture studies to further examine *Cricetid* rodent species community dynamics and densities over time.

APPENDICES

APPENDIX A

SMALL MAMMAL COMMUNITIES IN GSENM AND CARE

Table A.1. Small mammals captured in Grand Staircase - Escalante National Monument, Utah. Summers 2001 - 2007 and 2015.

Species	No. individuals year							Relative abundance ¹ Rank ²			
	2001	2002	2003	2004	2005	2006	2007	2015	Total		
American deer mouse (<i>Peromyscus maniculatus</i>)	25	23	45	108	71	44	9	9	334	12	2
Brush mouse (<i>P. boyleyi</i>)	8	39	131	180	474	643	77	42	1,594	59	1
Bushy-tailed woodrat (<i>Neotoma cinerea</i>)	0	0	0	0	5	2	0	2	9	0	11
Cactus mouse (<i>P. eremicus</i>)	6	3	2	0	3	2	0	0	16	1	10
Canyon mouse (<i>P. crinitus</i>)	0	2	22	2	7	46	0	5	84	3	6
Cliff chipmunk (<i>Tamias dorsalis</i>)	15	38	2	2	0	16	2	0	75	3	7
Desert woodrat (<i>N. lepida</i>)	1	0	0	12	3	0	0	5	21	1	9
<i>Dipodomys</i> spp.	0	0	0	0	0	1	1	0	2	0	16
Least chipmunk (<i>T. minimus</i>)	0	0	3	0	0	0	0	0	3	0	14
Long-tailed pocket mouse (<i>Chaetodipus formosa</i>)	0	14	12	20	29	26	5	0	106	4	4
<i>Neotoma</i> spp.	0	1	0	3	1	0	0	0	5	0	13
<i>Peromyscus</i> spp.	1	0	0	1	0	6	0	0	8	0	12
Pinyon mouse (<i>P. trueii</i>)	9	19	19	0	20	16	4	0	87	3	5
Ringtail (<i>Bassaris astutus</i>)	0	0	0	2	0	0	0	0	2	0	15
Rock pocket mouse (<i>Chaetodipus intermedius</i>)	0	1	0	0	0	0	0	0	1	0	18
Shrew (<i>Sorex</i> spp.)	0	0	0	0	0	1	0	0	1	0	17
Western harvest mouse (<i>Reithrodontomys megalotis</i>)	0	0	1	0	13	46	0	0	60	2	8
White-throated woodrat (<i>N. albigula</i>)	9	2	8	10	121	123	0	2	275	10	3
Total captures	74	142	245	340	747	972	98	65	2,683		
Total recaptures	90	496	746	789	1,781	2,697	198	247	7,044		
# species	8	10	10	10	11	13	6	6	16		
# trap nights	625	4,500	6,000	3,750	6,000	5,750	1,950	600	29,175		

¹ Relative abundance, number of each species captured as a percentage of the total number of small mammals captured.

² Rank depicts order of the most abundant small mammal species captured from 1 - 18.

Table A.2. Small mammals captured at each study site in Grand Staircase - Escalante National Monument, Utah. Summers 2001 - 2007 and 2015.

Species ¹	Hogeye Grid		Hogeye Tran		Snake Grid		Snake Tran		Starlight Grid	
	No. ²	% ³	No.	%	No.	%	No.	%	No.	%
American deer mouse	56	8.7	76	16.9	73	14.3	44	10.8	85	12.7
Brush mouse	321	49.8	279	62.1	346	67.7	314	76.8	335	50.0
Bushy-tailed woodrat	0	0.0	3	0.7	6	1.2	0	0.0	0	0.0
Cactus mouse	6	0.9	3	0.7	2	0.4	0	0.0	5	0.7
Canyon mouse	54	8.4	4	0.9	10	2.0	8	2.0	8	1.2
Cliff chipmunk	28	4.3	9	2.0	30	5.9	5	1.2	3	0.4
Desert woodrat	5	0.8	8	1.8	0	0.0	5	1.2	3	0.4
Kangaroo spp.	0	0.0	0	0.0	0	0.0	0	0.0	2	0.3
Least chipmunk	1	0.2	0	0.0	2	0.4	0	0.0	0	0.0
Long-tailed pocket mouse	37	5.7	0	0.0	0	0.0	0	0.0	69	10.3
Neotoma spp.	4	0.6	1	0.2	0	0.0	0	0.0	0	0.0
Peromyscus spp.	2	0.3	1	0.2	0	0.0	0	0.0	5	0.7
Pinyon mouse	29	4.5	5	1.1	7	1.4	5	1.2	41	6.1
Ringtail	0	0.0	2	0.4	0	0.0	0	0.0	0	0.0
Rock pocket mouse	0	0.0	0	0.0	0	0.0	0	0.0	1	0.1
Shrew spp.	1	0.2	0	0.0	0	0.0	0	0.0	0	0.0
Western harvest mouse	19	3.0	9	2.0	13	2.5	12	2.9	7	1.0
White-throated woodrat	81	12.6	49	10.9	22	4.3	17	4.2	106	15.8
Total captures	644		449		511		410		670	

¹ See Table 2.1 for scientific names.

² Number of individuals.

³ Relative abundance, total number of each species divided by total captures.

Table A.3. Small mammals captured in Capitol Reef National Park, Utah. Summers 2013 - 2015.

Species	No. individuals/year				Relative	
	2013	2014	2015	Total	abundance ¹	Rank ²
American deer mouse (<i>Peromyscus maniculatus</i>)	10	8	19	37	10.1	5
Brush mouse (<i>P. boylei</i>)	66	21	34	121	33.2	1
Bushy-tailed woodrat (<i>Neotoma cinerea</i>)	2	15	21	38	10.4	4
Cactus mouse (<i>P. eremicus</i>)	0	8	0	8	2.2	8
Canyon mouse (<i>P. crinitus</i>)	1	0	9	10	2.7	7
Cliff chipmunk (<i>Tamias dorsalis</i>)	37	11	8	56	15.3	3
Desert woodrat (<i>N. lepida</i>)	15	23	23	61	16.7	2
Great Basin pocket mouse (<i>Perognathus parvus</i>)	0	0	1	1	0.3	12
Pinyon mouse (<i>P. trueii</i>)	19	0	0	19	5.2	6
Rock Squirrel (<i>Spermophilus variegatus</i>)	1	2	0	3	0.8	11
Western spotted skunk (<i>Spilogale gracilis</i>)	0	2	3	5	1.4	10
Western white-throated woodrat (<i>N. albigula</i>)	0	1	5	6	1.6	9
Total captures	151	91	123	365		
Total recaptures	276	258	416	950		
# species	8	9	9	12		
# trap nights	1,500	2,275	2,700	6,475		

¹ Relative abundance, number of each species captured as a percentage of the total number of small mammals captured.

² Rank depicts order of the most abundant small mammal species captured from 1 - 12.

Table A.4. Small mammals captured at each study site in Capitol Reef National Park, Utah. Summers 2013 - 2015.

Species ¹	Bitter Creek		Ups Grid		Ups Tran		Cassidy		Burro		5 - Mile	
	No. ²	% ³	No.	%	No.	%	No.	%	No.	%	No.	%
American deer mouse	16	12.3	6	5.9	1	1.9	5	10.9	7	33.3	2	14.3
Brush mouse	72	55.4	22	21.6	8	15.4	9	19.6	5	23.8	5	35.7
Bushy-tailed woodrat	31	23.8	0	0.0	0	0.0	2	4.3	5	23.8	0	0.0
Cactus mouse	0	0.0	1	1.0	0	0.0	7	15.2	0	0.0	0	0.0
Canyon mouse	0	0.0	0	0.0	1	1.9	5	10.9	1	4.8	3	21.4
Cliff chipmunk	8	6.2	30	29.4	14	26.9	2	4.3	2	9.5	0	0.0
Desert woodrat	2	1.5	24	23.5	20	38.5	11	23.9	1	4.8	3	21.4
Great Basin pocket mouse	0	0.0	0	0.0	0	0.0	1	2.2	0	0.0	0	0.0
Pinyon mouse	1	0.8	13	12.7	5	9.6	0	0.0	0	0.0	0	0.0
Rock Squirrel	0	0.0	3	2.9	0	0.0	0	0.0	0	0.0	0	0.0
Western spotted skunk	0	0.0	3	2.9	2	3.8	0	0.0	0	0.0	0	0.0
White-throated woodrat	0	0.0	0	0.0	1	1.9	4	8.7	0	0.0	1	7.1
Total captures	130		102		52		46		21		14	

¹ See Table 2.2 for scientific names.² Number of individuals.³ Relative abundance, total number of each species divided by total captures.

Table A.5. American deer mouse microhabitat associations. Microhabitat components measured within 10-m diameter circular plots centered on traps used and unused by 194 deer mice captured in Grand Staircase-Escalante National Monument, Utah, summers 2002-2007 and 2015.

Variable	Used traps (<i>n</i> = 194)			Welch's <i>t</i> -test*	Unused traps (<i>n</i> = 1,439)		
	Mean	SE	95% CI		Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	61.2	2.9	55.5, 66.8	(<i>t</i> ₂₁₈ = -0.16, <i>p</i> = 0.9)	61.7	1.0	58.6, 63.7
Inverse canopy cover north (%) ²	64.9	2.7	59.5, 70.3	(<i>t</i> ₂₄₄ = -0.23, <i>p</i> = 0.8)	65.6	1.0	63.7, 67.5
Tree diameter (cm)	15.6	1.5	12.6, 18.7	(<i>t</i> ₂₄₅ = 0.12, <i>p</i> = 0.9)	15.4	0.5	14.3, 16.5
Tree height (m)	6.0	0.6	4.8, 7.2	(<i>t</i> ₂₄₄ = 0.46, <i>p</i> = 0.6)	5.7	0.2	5.3, 6.1
Number trees	5.8	1.0	3.9, 7.7	(<i>t</i> ₂₁₂ = 2.00, <i>p</i> = 0.05)	3.9	0.2	3.4, 4.3
Shrub height (cm)	105.9	6.0	94.1, 117.8	(<i>t</i> ₂₃₈ = 1.9, <i>p</i> = 0.06)	93.8	2.0	89.8, 97.7
Aspect (degree)	199.9	7.1	186.0, 213.9	(<i>t</i> ₂₃₆ = -0.84, <i>p</i> = 0.4)	206.2	2.3	201.6, 210.7
Slope (degree)	21.0	1.4	18.2, 23.9	(<i>t</i> ₂₅₆ = -0.86, <i>p</i> = 0.4)	22.4	0.6	21.2, 23.5
% Forb/grass cover ³	20.3	2.1	16.2, 24.4	(<i>t</i> ₂₄₄ = 1.17, <i>p</i> = 0.2)	17.7	0.7	16.3, 19.2
% Shrub cover	7.1	1.2	4.7, 9.4	(<i>t</i> ₂₅₈ = -2.12, <i>p</i> = 0.03)	9.8	0.5	8.9, 10.7
% Litter cover	15.5	1.7	12.2, 18.8	(<i>t</i> ₂₅₁ = 0.00, <i>p</i> = 1.0)	15.5	0.6	14.3, 16.8
% Tree ⁴	6.2	1.3	3.7, 8.7	(<i>t</i> ₂₆₅ = -0.64, <i>p</i> = 0.5)	7.1	0.5	6.0, 8.1
% Water	0.2	0.2	0.0, 0.6	(<i>t</i> ₃₃₀ = -1.35, <i>p</i> = 0.2)	0.5	0.1	0.3, 0.8
% Cactus	0.1	0.1	0.0, 0.3	(<i>t</i> ₃₄₆ = -0.25, <i>p</i> = 0.8)	0.1	0.1	0.0, 0.3
% Bare (no cover)	29.8	2.3	25.3, 34.3	(<i>t</i> ₂₃₈ = 1.33, <i>p</i> = 0.2)	26.6	0.8	25.1, 28.1
% Rock cover (no veg cover)	20.8	2.3	16.3, 25.3	(<i>t</i> ₂₅₀ = -0.76, <i>p</i> = 0.4)	22.6	0.8	20.9, 24.3

* Used traps not different than unused traps.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

Table A.6. Woodrat and white-footed mouse species group microhabitat comparison. Microhabitat components measured within 10-m diameter circular plots centered on traps used by woodrats and white-footed mice captured in Capitol Reef National Park, Utah, summer 2015.

Variable	Woodrats (<i>n</i> = 41)			Welch's <i>t</i> -test*	White-footed mouse (<i>n</i> = 53)		
	Mean	SE	95% CI		Mean	SE	95% CI
Inverse canopy cover trap (%) ¹	73.8	6.3	61.0, 86.5	(<i>t</i> ₈₀ = 0.45, <i>p</i> = 0.7)	69.7	6.4	56.9, 82.6
Inverse canopy cover north (%) ²	74.8	6.7	61.3, 88.3	(<i>t</i> ₈₀ = 1.10, <i>p</i> = 0.3)	64.4	6.7	50.9, 78.0
Tree diameter (cm)	5.4	1.2	2.9, 7.9	(<i>t</i> ₆₈ = -0.72, <i>p</i> = 0.5)	7.2	2.1	2.9, 11.4
Tree height (m)	2.8	0.4	2.0, 3.6	(<i>t</i> ₈₀ = 0.29, <i>p</i> = 0.8)	2.6	0.4	1.7, 3.5
Number trees	2.2	0.6	1.0, 3.4	(<i>t</i> ₅₉ = 0.47, <i>p</i> = 0.6)	1.9	0.3	1.2, 2.5
Shrub height (cm)	184.2	15.9	151.9, 216.5	(<i>t</i> ₇₇ = -1.9, <i>p</i> = 0.06)	235.1	20.8	193.1, 277.1
Aspect (degree)	123.6	13.0	97.3, 149.8	(<i>t</i> ₇₈ = 0.06, <i>p</i> = 1.0)	122.5	11.9	98.4, 146.6
Slope (degree)	18.9	2.3	14.2, 23.7	(<i>t</i> ₈₀ = -0.58, <i>p</i> = 0.6)	20.9	2.3	16.2, 25.6
% Forb/grass cover ³	16.2	3.9	8.4, 24.0	(<i>t</i> ₇₇ = -0.41, <i>p</i> = 0.7)	18.4	3.4	11.5, 25.3
% Shrub cover	24.0	5.3	13.3, 34.7	(<i>t</i> ₇₉ = -0.27, <i>p</i> = 0.8)	25.9	5.0	15.9, 36.0
% Litter cover	2.5	1.0	0.4, 4.6	(<i>t</i> ₆₃ = 0.76, <i>p</i> = 0.4)	1.5	0.6	0.2, 2.8
% Tree ⁴	9.3	3.5	2.1, 16.4	(<i>t</i> ₇₀ = 0.27, <i>p</i> = 0.8)	8.1	2.6	2.8, 13.3
% Bare (no cover)	23.7	4.5	14.6, 32.7	(<i>t</i> ₇₉ = -0.64, <i>p</i> = 0.5)	27.6	4.4	18.8, 36.4
% Rock cover (no veg cover)	24.4	4.5	15.3, 33.5	(<i>t</i> ₇₈ = 0.97, <i>p</i> = 0.3)	18.5	4.2	10.1, 26.9

* Used traps not different than unused traps.

¹ Percent open sky. Lower number = higher canopy cover.

² Canopy cover measurement 5-m north of trap location.

³ Percent vegetative cover (cm) along 5-m line transect.

⁴ Tree basal area (cm) along 5-m transect.

APPENDIX B

SEASONAL PRECIPITATION ANALYSES

Table B.1. Precipitation and climate data for Grand Staircase - Escalante National Monument and Capitol Reef National Park study sites, Utah. 2000 - 2015.

Year	Area	Site	Winter (mm)	Summer (mm)	Annual (mm)	DI ¹
2000	GSENM	hogeye	103	40	143	0
2001	GSENM	snake	264	58	322	0
2001	GSENM	starlight	290	62	352	0
2001	GSENM	hogeye	268	56	324	0
2002	GSENM	hogeye	72	67	139	1
2002	GSENM	snake	71	68	139	1
2002	GSENM	starlight	82	70	152	1
2003	GSENM	hogeye	150	25	175	1
2003	GSENM	snake	153	27	180	1
2003	GSENM	starlight	173	27	200	1
2004	GSENM	hogeye	170	103	273	1
2004	GSENM	snake	168	102	270	1
2004	GSENM	starlight	178	102	280	1
2005	GSENM	hogeye	487	76	563	0
2005	GSENM	snake	470	77	547	0
2005	GSENM	starlight	514	72	586	0
2006	GSENM	hogeye	193	83	276	0
2006	GSENM	snake	177	84	261	0
2006	GSENM	starlight	203	87	290	0
2007	GSENM	hogeye	177	97	274	1
2007	GSENM	snake	169	110	279	1
2007	GSENM	starlight	195	97	292	1
2012	CARE	ups	104	124	228	1
2012	CARE	bcd	95	101	196	1
2013	CARE	bcd	107	199	306	1
2013	CARE	ups	100	210	310	1
2014	CARE	bcd	104	150	254	0
2014	CARE	ups	118	144	262	0
2014	CARE	5.mile	127	167	294	0
2014	CARE	burro	116	150	266	0
2014	CARE	cassidy	92	120	212	0
2014	GSENM	hogeye	126	142	268	1
2014	GSENM	snake	123	145	268	1
2015	CARE	5.mile	232	131	363	0
2015	CARE	bcd	224	106	330	0
2015	CARE	burro	220	130	350	0
2015	CARE	cassidy	175	117	292	0
2015	GSENM	hogeye	192	122	314	1
2015	GSENM	snake	194	118	312	1
2015	CARE	ups	189	129	318	0

¹ DI - Drought Index at study site (0) = no drought + abnormally dry, (1) = moderate drought + severe drought + extreme drought + exceptional drought.

Table B.2. Response variables used for mixed - effects analyses of timing and magnitude of seasonal precipitation on *Cricetid* rodent abundance and diversity in Grand Staircase-Escalante National Monument and Capitol Reef National Park, Utah. 2001 - 2007 & 2013 - 2015.

Response variable	Definition
H'	Shannon's Diversity Index
J	Shannon's Evenness Index
D ₁	Simpson's Diversity Index
Pri.Prey	Count of <i>Cricetid</i> rodents (adjusted for capture effort).
WR	Count of woodrats (adjusted for capture effort).
WFM	Count of white-footed mice (adjusted for capture effort).

Table B.3. Precipitation and drought predictors tested with mixed - effects analyses of timing and magnitude of seasonal precipitation on *Cricetid* rodent abundance and diversity in Grand Staircase-Escalante National Monument and Capitol Reef National Park, Utah. 2001 - 2007 & 2013 - 2015.

Predictor	Term	Unit	Definition
Annual precipitation	AP	mm	Total annual precipitation (October - September) log transformed.
Winter precipitation	WP	mm	Total winter precipitation (October - June) log transformed.
Summer precipitation	SP	mm	Total summer precipitation (July - September) log transformed.
Annual precipitation lag	AP.lag	mm	Annual precipitation from previous year, log transformed.
Winter precipitation lag	WP.lag	mm	Winter precipitation from previous year, log transformed.
Summer precipitation lag	SP.lag	mm	Summer precipitation from previous year, log transformed.
Drought index at site	DI(s)	(0-1)	(0) = no drought + abnormally dry, (1) = moderate + severe + extreme + exceptional.
Drought index at site lag	DI(s).lag	(0-1)	(0) = no drought + abnormally dry, (1) = moderate + severe + extreme + exceptional.

Table B.4. *A priori* candidate model set for mixed - effects analyses of timing and magnitude of seasonal precipitation on Cricetid rodent abundance and diversity in Grand Staircase-Escalante National Monument and Capitol Reef National Park, Utah. 2001 - 2007 & 2013 - 2015.

Model Term	Model*
Response variable	~ Site (NULL)
Response variable	~ Site + WP
Response variable	~ Site + WP.lag
Response variable	~ Site + SP
Response variable	~ Site + SP.lag
Response variable	~ Site + AP
Response variable	~ Site + AP.lag
Response variable	~ Site + DI(s)
Response variable	~ Site + DI(s).lag
Response variable	~ Site + WP + DI(s)
Response variable	~ Site + WP.lag + DI(s).lag
Response variable	~ Site + AP + DI(s)
Response variable	~ Site + AP.lag + DI(s).lag
Response variable	~ Site + WP * DI(s)
Response variable	~ Site + WP.lag * DI(s)
Response variable	~ Site + AP * DI(s)
Response variable	~ Site + AP.lag * DI(s).lag

* See Table B.3 for model predictor definitions and units.

Table B.5. Low, average, and high precipitation values used to estimate effects of predictors from mixed - effects analyses of timing and magnitude of seasonal precipitation on *Cricetid* rodent abundance and diversity in Grand Staircase-Escalante National Monument and Capitol Reef National Park, Utah. 2001 - 2007 & 2013 - 2015.

	WP	SP	AP	DI(s)
Low	75	26	143	0
Average	181	71	250	
High	490	102	565	1

WP = Winter precipitation (mm), SP = Summer precipitation (mm), AP = Annual precipitation (mm), DI(s) = Drought index at site.

Table B.6. Summary of linear mixed-effect models. Models are ranked by differences in Akaike's information criterion for small sample sizes (ΔAIC_c) for analyses of Shannon's diversity in Grand Staircase - Escalante National Monument, Utah. 2001 - 2007 & 2015 and Capitol Reef National Park, Utah. 2013 - 2015.

Model	<i>k</i>	AIC	AIC _c	ΔAIC_c	<i>exp</i>	<i>wi</i>	Model
4	4	25.626	27.23	0.00	1.00	0.272	H' ~ site + Summer precip
7	4	26.978	28.58	1.35	0.51	0.138	H' ~ site + Annual precip w lag effect
3	4	27.479	29.08	1.85	0.40	0.108	H' ~ site + Winter precip w lag effect
6	4	27.672	29.27	2.05	0.36	0.098	H' ~ site + Annual precip
1	4	27.813	29.41	2.19	0.33	0.091	H' ~ site (Null)
9	4	29.026	30.63	3.40	0.18	0.050	H' ~ site + drought index at site w lag effect
5	4	29.330	30.93	3.70	0.16	0.043	H' ~ site + Summer precip w lag effect
2	4	29.552	31.15	3.93	0.14	0.038	H' ~ site + Winter precip
8	4	29.781	31.38	4.15	0.13	0.034	H' ~ site + drought index at site
12	5	28.906	31.41	4.18	0.12	0.034	H' ~ site + Annual precip + drought index at site
13	5	28.976	31.48	4.25	0.12	0.033	H' ~ site + Annual precip + drought index at site w lag effects
11	5	29.373	31.87	4.65	0.10	0.027	H' ~ site + Winter precip + drought index at site w lag effects
16	6	30.191	33.84	6.62	0.04	0.010	H' ~ site + Annual precip * drought index at site
10	5	31.510	34.01	6.78	0.03	0.009	H' ~ site + Winter precip + drought index at site
15	6	30.869	34.52	7.29	0.03	0.007	H' ~ site + Winter precip * drought index at site w lag effects
17	6	30.921	34.57	7.35	0.03	0.007	H' ~ site + Annual precip * drought index at site w lag effects
14	6	33.494	37.15	9.92	0.01	0.002	H' ~ site + Winter precip * drought index at site

Table B.7. Summary of linear mixed-effect models. Models are ranked by differences in Akaike's information criterion for small sample sizes (ΔAIC_c) for analyses of Shannon's evenness in Grand Staircase - Escalante National Monument, Utah. 2001 - 2007 & 2015 and Capitol Reef National Park, Utah. 2013 - 2015.

Model	k	AIC	AIC_c	ΔAIC_c	exp	wi	Model
20	4	-4.173	-2.57	0.00	1.00	0.214	J ~ site + Winter precip w lag effect
23	4	-3.116	-1.52	1.06	0.59	0.126	J ~ site + Annual precip
24	4	-3.072	-1.47	1.10	0.58	0.124	J ~ site + Annual precip w lag effect
28	5	-3.303	-0.80	1.77	0.41	0.088	J ~ site + Winter precip + drought index at site w lag effects
18	4	-2.268	-0.67	1.91	0.39	0.083	J ~ site (Null)
19	4	-2.224	-0.62	1.95	0.38	0.081	J ~ site + Winter precip
25	4	-1.229	0.37	2.94	0.23	0.049	J ~ site + drought index at site
21	4	-1.107	0.49	3.07	0.22	0.046	J ~ site + Summer precip
26	4	-0.759	0.84	3.41	0.18	0.039	J ~ site + drought index at site w lag effect
22	4	-0.274	1.33	3.90	0.14	0.030	J ~ site + Summer precip w lag effect
30	5	-1.148	1.35	3.93	0.14	0.030	J ~ site + Annual precip + drought index at site w lag effects
29	5	-1.117	1.38	3.96	0.14	0.030	J ~ site + Annual precip + drought index at site
27	5	-0.229	2.27	4.84	0.09	0.019	J ~ site + Winter precip + drought index at site
32	6	-1.321	2.33	4.90	0.09	0.018	J ~ site + Winter precip * drought index at site w lag effects
34	6	-0.055	3.60	6.17	0.05	0.010	J ~ site + Annual precip * drought index at site w lag effects
33	6	0.224	3.88	6.45	0.04	0.009	J ~ site + Annual precip * drought index at site
31	6	1.672	5.32	7.90	0.02	0.004	J ~ site + Winter precip * drought index at site

Table B.8. Summary of linear mixed-effect models. Models are ranked by differences in Akaike's information criterion for small sample sizes (ΔAIC_c) for analyses of Simpson's diversity in Grand Staircase - Escalante National Monument, Utah. 2001 - 2007 & 2015 and Capitol Reef National Park, Utah. 2013 - 2015.

Model	k	AIC	AIC_c	ΔAIC_c	exp	wi	Model
37	4	-1.274	0.33	0.00	1.00	0.257	D1 ~ site + Winter precip w lag effect
43	4	0.298	1.90	1.57	0.46	0.117	D1 ~ site + drought index at site w lag effect
35	4	0.599	2.20	1.87	0.39	0.101	D1 ~ site (Null)
41	4	0.899	2.50	2.17	0.34	0.087	D1 ~ site + Annual precip w lag effect
40	4	1.211	2.81	2.49	0.29	0.074	D1 ~ site + Annual precip
45	5	0.700	3.20	2.87	0.24	0.061	D1 ~ site + Winter precip + drought index at site w lag effects
36	4	1.825	3.43	3.10	0.21	0.055	D1 ~ site + Winter precip
38	4	2.106	3.71	3.38	0.18	0.047	D1 ~ site + Summer precip
39	4	2.207	3.81	3.48	0.18	0.045	D1 ~ site + Summer precip w lag effect
42	4	2.560	4.16	3.83	0.15	0.038	D1 ~ site + drought index at site
47	5	1.974	4.47	4.15	0.13	0.032	D1 ~ site + Annual precip + drought index at site w lag effects
46	5	2.339	4.84	4.51	0.10	0.027	D1 ~ site + Annual precip + drought index at site
44	5	2.732	5.23	4.91	0.09	0.022	D1 ~ site + Winter precip + drought index at site
49	6	2.176	5.83	5.50	0.06	0.016	D1 ~ site + Winter precip * drought index at site w lag effects
51	6	3.583	7.24	6.91	0.03	0.008	D1 ~ site + Annual precip * drought index at site w lag effects
50	6	3.912	7.56	7.24	0.03	0.007	D1 ~ site + Annual precip * drought index at site
48	6	4.731	8.38	8.06	0.02	0.005	D1 ~ site + Winter precip * drought index at site

Table B.9. Summary of generalized linear mixed-effect models. Models are ranked by differences in Akaike's information criterion for small sample sizes (ΔAIC_c) for analyses of primary prey abundance in Grand Staircase - Escalante National Monument, Utah. 2001 - 2007 & 2015 and Capitol Reef National Park, Utah. 2013 - 2015.

Model	K	AIC	AIC _c	ΔAIC_c	exp	wi	Model
54	4	284.850	286.450	0.00	1.00	0.563	Pri.prey ~ site + Winter precip w lag effect
62	5	285.716	288.216	1.77	0.41	0.233	Pri.prey ~ site + Winter precip + drought index at site w lag effects
66	6	287.673	291.325	4.88	0.09	0.049	Pri.prey ~ site + Winter precip * drought index at site w lag effects
59	4	290.303	291.903	5.45	0.07	0.037	Pri.prey ~ site + Drought index at site
56	4	291.716	293.316	6.87	0.03	0.018	Pri.prey ~ site + Summer precip w lag effect
58	4	291.763	293.363	6.91	0.03	0.018	Pri.prey ~ site + Annual precip w lag effect
67	6	289.904	293.556	7.11	0.03	0.016	Pri.prey ~ site + Annual precip * drought index at site
63	5	291.771	294.271	7.82	0.02	0.011	Pri.prey ~ site + Annual precip + drought index at site
53	4	292.749	294.349	7.90	0.02	0.011	Pri.prey ~ site + Winter precip
61	5	292.139	294.639	8.19	0.02	0.009	Pri.prey ~ site + Winter precip + drought index at site
57	4	293.154	294.754	8.30	0.02	0.009	Pri.prey ~ site + Annual precip
60	4	293.771	295.371	8.92	0.01	0.007	Pri.prey ~ site + Drought index at site w lag effect
68	6	291.958	295.610	9.16	0.01	0.006	Pri.prey ~ site + Annual precip * drought index at site w lag effects
52	4	294.238	295.838	9.39	0.01	0.005	Pri.prey ~ site (Null)
64	5	293.759	296.259	9.81	0.01	0.004	Pri.prey ~ site + Annual precip + drought index at site w lag effects
65	6	293.551	297.203	10.75	0.00	0.003	Pri.prey ~ site + Winter precip * drought index at site
55	4	296.216	297.816	11.37	0.00	0.002	Pri.prey ~ site + Summer precip

Table B.10. Summary of generalized linear mixed-effect models. Models are ranked by differences in Akaike's information criterion for small sample sizes (ΔAIC_c) for analyses of woodrat abundance in Grand Staircase - Escalante National Monument, Utah. 2001 - 2007 & 2015 and Capitol Reef National Park, Utah. 2013 - 2015.

Model	K	AIC	AIC _c	ΔAIC_c	exp	wi	Model
76	4	165.477	167.077	0.00	1.00	0.355	WR ~ site + Drought index at site
84	6	164.674	168.326	1.25	0.54	0.190	WR ~ site + Annual precip * drought index at site
80	5	166.477	168.977	1.90	0.39	0.137	WR ~ site + Annual precip + drought index at site
78	5	167.476	169.976	2.90	0.23	0.083	WR ~ site + Winter precip + drought index at site
74	4	169.744	171.344	4.27	0.12	0.042	WR ~ site + Annual precip
71	4	169.988	171.588	4.51	0.10	0.037	WR ~ site + Winter precip w lag effect
75	4	170.015	171.615	4.54	0.10	0.037	WR ~ site + Annual precip w lag effect
79	5	169.652	172.152	5.08	0.08	0.028	WR ~ site + Winter precip + drought index at site w lag effects
70	4	170.782	172.382	5.30	0.07	0.025	WR ~ site + Winter precip
82	6	169.369	173.021	5.94	0.05	0.018	WR ~ site + Winter precip * drought index at site
81	5	171.380	173.880	6.80	0.03	0.012	WR ~ site + Annual precip + drought index at site w lag effects
69	4	172.363	173.963	6.89	0.03	0.011	WR ~ site (Null)
83	6	171.618	175.271	8.19	0.02	0.006	WR ~ site + Winter precip * drought index at site w lag effects
77	4	173.890	175.490	8.41	0.01	0.005	WR ~ site + Drought index at site w lag effect
72	4	174.066	175.666	8.59	0.01	0.005	WR ~ site + Summer precip
73	4	174.320	175.920	8.84	0.01	0.004	WR ~ site + Summer precip w lag effect
85	6	173.360	177.012	9.94	0.01	0.002	WR ~ site + Annual precip * drought index at site w lag effects

Table B.11. Summary of generalized linear mixed-effect models. Models are ranked by differences in Akaike's information criterion for small sample sizes (ΔAIC_c) for analyses of white-footed mouse abundance in Grand Staircase - Escalante National Monument, Utah, 2001 - 2007 & 2015 and Capitol Reef National Park, Utah, 2013 - 2015.

Model	K	AIC	AIC _c	ΔAIC_c	exp	w _i	Model
88	4	279.431	281.031	0.00	1.00	0.477	WFM ~ site + Winter precip w lag effect
96	5	280.734	283.234	2.20	0.33	0.159	WFM ~ site + Winter precip + drought index at site w lag effects
90	4	282.551	284.151	3.12	0.21	0.100	WFM ~ site + Summer precip w lag effect
93	4	284.356	285.956	4.93	0.09	0.041	WFM ~ site + Drought index at site
100	6	282.708	286.360	5.33	0.07	0.033	WFM ~ site + Winter precip * drought index at site w lag effects
92	4	284.867	286.467	5.44	0.07	0.031	WFM ~ site + Annual precip w lag effect
91	4	285.161	286.761	5.73	0.06	0.027	WFM ~ site + Annual precip
86	4	285.415	287.015	5.98	0.05	0.024	WFM ~ site (Null)
87	4	285.476	287.076	6.05	0.05	0.023	WFM ~ site + Winter precip
94	4	285.771	287.371	6.34	0.04	0.020	WFM ~ site + Drought index at site w lag effect
97	5	285.815	288.315	7.28	0.03	0.012	WFM ~ site + Annual precip + drought index at site
102	6	284.805	288.457	7.43	0.02	0.012	WFM ~ site + Annual precip * drought index at site w lag effects
101	6	284.930	288.582	7.55	0.02	0.011	WFM ~ site + Annual precip * drought index at site
95	5	286.230	288.730	7.70	0.02	0.010	WFM ~ site + Winter precip + drought index at site
89	4	287.375	288.975	7.94	0.02	0.009	WFM ~ site + Summer precip
98	5	286.817	289.317	8.29	0.02	0.008	WFM ~ site + Annual precip + drought index at site w lag effects
99	6	287.954	291.606	10.58	0.01	0.002	WFM ~ site + Winter precip * drought index at site

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