

SMALL MAMMAL SPECIES STATUS SURVEY IN THE PINYON-JUNIPER WOODLANDS OF CITY OF ROCKS NATIONAL RESERVE AND CASTLE ROCKS STATE PARK, IDAHO



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SMALL MAMMAL SPECIES STATUS SURVEY IN THE PINYON-JUNIPER WOODLANDS OF CITY OF ROCKS NATIONAL RESERVE AND CASTLE ROCKS STATE PARK, IDAHO

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Abstract

The area in and around City of Rocks National Reserve in southern Idaho coincides with a unique biogeographic setting where the pinyon-juniper woodland reaches its northern distributional limit, occurs in conjunction with large granite cliffs, and supports a diverse but poorly described mammalian fauna. These mammalian communities include several rare species also at their northern distributional limit that are not found elsewhere in Idaho. Surveys for these species were conducted during the summer and fall seasons of 2005 and 2006. Integrated sampling efforts involving the utilization of small mammal live trapping and motion cameras were used to provide new information on the distribution, abundance, and habitat association of the cliff chipmunk (*Tamias dorsalis*), pinyon mouse (*Peromyscus truei*), canyon mouse (*Peromyscus crinitus*), brush mouse (*Peromyscus boylii*) and the ringtail (*Bassariscus astutus*). Remotely-deployed motion triggered cameras were used to target the ringtail. Cameras were placed in rocky areas where game trails funneled between rock features provided good opportunities to photograph animals. Scent stations were established using commercially available lures. Rodents were surveyed using standard aluminum live traps placed in randomly located plots. Vegetation, distance to important features, and other habitat characteristics were described for sampling locations. No ringtails were detected during the study although a confident sighting was made by knowledgeable park visitors in April 2006. Also, no canyon mice or brush mice were detected during the study. Four cliff chipmunks were captured at 4 sites in 2005, and 90 cliff chipmunks were captured at 19 sites in 2006. No pinyon mice were captured in 2005, but 139 pinyon mice were captured at 13 sites in 2006. Genetic analysis of 6 pinyon mice specimens confirmed field identification and provided confidence in our assumption that false-positive misidentification was minimal for this potentially confusing species. An *a priori* suite of candidate multi-season occupancy models that explicitly incorporate detectability were developed for both of these rodent species using 2006 capture detection histories. Information-theoretic methods of model selection were used to determine best models. For the cliff chipmunk, we selected a model in which the probability of detection was estimated uniformly across all sites, surveys, and seasons to be 0.51. The probability of occupancy was positively influenced by rock cover and site distance to the nearest pinyon-juniper woodland edge, and negatively influenced by increasing grass, forb, and shrub cover. The pinyon mouse exhibited a similar response to these local scale habitat features. Detectability for this species was estimated to be 0.81, and bare ground and distance to the woodland edge positively influenced the probability of occupancy and grass, forb, and shrub cover negatively influenced occupancy. The pinyon mouse exhibited a pattern of local extinction between season 1 and 2 in sites with greater understory vegetation and we present for future investigation the hypothesis that pinyon mice may undergo seasonal occupancy contraction and expansion dynamics in sites with suboptimal habitat characteristics. These results provide the first vouchered confirmation of the pinyon mouse in Cassia County, Idaho, and provide new information on important local-scale habitat characteristics that appear important to these unique species. This study suggests that contiguous stands of pinyon-juniper woodlands near large rock expanses and with an open understory may be “core” habitat for these species that could be conserved or managed for over time. This management conclusion will become even more important if this habitat association can be made for other species of conservation concern through other studies.

Introduction

In 2005, Idaho State Parks and Recreation (IDPR), Idaho Department of Fish and Game (IDFG), and the National Park Service Upper Columbia Basin Network collaborated in the design and implementation of an occupancy status survey of the pinyon mouse (*Peromyscus truei*), canyon mouse (*Peromyscus crinitus*), brush mouse (*Peromyscus boylii*), cliff chipmunk (*Tamias dorsalis*) and ringtail (*Bassariscus astutus*) at City of Rocks National Reserve (CIRO) and Castle Rocks State Park (CRSP) in southern Idaho. Interest in these species stems from the fact that their northern distributional limits are believed to occur in the vicinity of the Reserve, in association with the northern terminus of the distribution of single-leaf pinyon pine (*Pinus monophylla*).

The purpose of this project was to provide new information on the occurrence and habitat association of this unique mammalian fauna associated with the co-occurrence of pinyon-juniper woodland and large granitic rock formations in CIRO and CRSP. The combination of these habitat features is unique in Idaho and has enabled several mammal species to persist or extend their range into the state. Accounts of these species elsewhere in their range indicate a strong association with rocky pinyon-juniper woodlands (Holbrook 1978, Hall 1981, Hoffmeister 1981, Hammond and Yensen 1982, Johnson and Armstrong 1987, Hart 1992, Carraway et al. 1993, Verts and Carraway 1998, Rompolo and Anderson 2004).

Information on the mammalian fauna of the pinyon-juniper woodland in southern Idaho is sparse. Larrison and Johnson (1981) provided unvouchered reports of pinyon mice and cliff chipmunks in the CIRO area as early as 1967, and additional reports of these species in the southern Idaho area have been provided by the University of Idaho (Madison et al. 2003) and Bureau of Land Management (BLM) (Bartels and Niwa 2004). Hoffmeister (1981) did not consider Idaho to be within the range of the pinyon mouse yet Hart (1992) reported that the cliff chipmunk ranged as far north as southern Cassia County and the CIRO area. Gafur et al. (1980) and Hammond and Yensen (1982) provided vouchered documentation of pinyon mice in the juniper woodlands of Owyhee County in southwestern Idaho and to our knowledge this is the only area of the state where the occurrence of the pinyon mouse has been definitively confirmed (Carraway and Verts 2002, E. Yensen, Albertson College, personal communication).

Insufficient information is available to adequately assess the conservation status of these species in the CIRO area. While the state conservation ranking of the cliff chipmunk remains S1, the pinyon mouse ranking was recently changed from S1 to S2 due to a lack of reliable conservation data for the species (R. Dixon, Conservation Data Center, personal communication).

Unconfirmed reports of the brush mouse in southern Idaho have also recently been made (P. Bartels, BLM, personal communication), and confirmation of this species would indicate a significant range extension and a first record for Idaho (Hall 1981).

In March 2003, a dead ringtail was found in CRSP by IDPR. This specimen provided the first record of the species in Idaho. Possible tracks of the species were encountered in the park in February 2005 and a creditable witness observed a ringtail in the “Tahitian Wall” area of CIRO in April 2006. The presence of the species in southern Idaho represents a significant northward range extension (Hall 1981, Poglayen-Neuwall and Toweill 1988). The ringtail was removed

from the state's special status list in 1996 due to a lack of evidence that it actually occurred in the state. Confirmation of an established ringtail population in the CIRO area would warrant returning the species to the special status list.

The fauna of this area has not been well studied but the need for species information has become increasingly important and several emerging issues warrant further study. First, an outbreak of the pinyon ips beetle (*Ips confusus*) has been confirmed in CIRO and a significant number of singleleaf pinyon pines in that area may have been killed from this outbreak (S. Cook, University of Idaho, personal communication). Second, climate change scenarios developed for the Great Basin indicate that altered temperature and precipitation patterns could lead to significant changes in the distribution of pinyon-juniper woodland and consequently among the associated fauna (Wagner et al. 2003). Preliminary results from a resurvey of the Joseph Grinnell/UC-Berkeley mammal surveys originally conducted during the early 20th century in the Yosemite area of California indicate that the pinyon mouse may now occur at higher elevations (J. Patton and J. Perrine, University of California, personal communication). An analogous latitudinal shift may also be occurring in areas of current northern distributional limits such as southern Idaho.

Furthermore, accelerated frequency and intensity of *Ips* outbreaks may also be triggered by climate change (Logan and Powell 2001). A series of historic photos taken at CIRO beginning in 1868 clearly indicate that pinyon-juniper vegetation has increased both in extent and density within the park (Klett et al. 2004, see <http://www.thirdview.org/3v/rephotos/index.html>). This is due at least in part to over-grazing and fire suppression, but also perhaps to climate change and other stochastic events (Rust and Coulter 2000, Baker and Shinneman 2004, Klett et al. 2004, Soulé et al. 2004). This change may have increased susceptibility of the pine to *Ips* infestation. It may have also facilitated a northward range extension of some of the mammal species in question.

Within the context of accelerating environmental change in southern Idaho, it is increasingly important that baseline information be established in reference sites against which future changes can be measured. CIRO and CRSP and the surrounding federal land provide an ideal location for such a purpose and this study has made a significant contribution toward that effort. The NPS Upper Columbia Basin Network Inventory and Monitoring program, which conducts natural resource studies at CIRO, has expressed an interest in developing a long-term monitoring strategy for peripheral species associated with pinyon-juniper woodland in CIRO. This project has provided pilot data to the National Park Service to assist in its efforts to assess the feasibility and sampling design requirements of such a project. In addition, this study will provide important data critical to the IDFG and Idaho Conservation Data Center as it implements its state wildlife conservation strategy. Results from this study may also provide insight to IDPR resource management decision makers related to the competing concerns of pinyon-juniper woodland expansion and potential loss of pinyon-juniper woodlands and associated flora and fauna.

Study Area

CIRO and CRSP are located near the town of Almo in southeastern Idaho (Figure 1). The parks are in the Albion mountain range, approximately 50 miles south of Burley, Idaho. The study area is located in the center of a unique biogeographic setting where the Great Basin desert and Rocky

Mountain ecosystems merge together and the pinyon-juniper woodland reaches its northern distributional limit occurring in conjunction with large granite cliffs. The elevation ranges from 1700 to over 2700 m. Streams in the park are fed by snow pack and springs. Circle Creek is the main water source in the park and consists of the north, center, and south forks starting just below Graham Peak and merging together before entering into the Raft River. Flora communities consist of Douglas fir (*Pseudotsuga menziesii*) and lodgepole pine (*Pinus contorta*) forests in the higher elevations of CIRO, mountain mahogany (*Cercocarpus ledifolius*), aspen (*Populus tremuloides*), and pinyon pine/ Utah juniper (*Juniperus osteosperma*) woodlands in the mid-elevations, and sagebrush-steppe throughout the lower elevations.

The area was settled in the mid to late 1800's and the land was used for agricultural purposes including grazing, dryland crested wheat farming, and mica mining. Grazing continues today but mining and dryland farming no longer occur and most of the areas that were seeded to crested wheat are now heavily dominated by mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*). Fire suppression and cattle grazing may have contributed to the increased extent of the pinyon-juniper coverage (Rust and Coulter 2000). This increase can be seen in historic photos available for CIRO dating back to the late 1860's (Klett et al. 2004). Several species of noxious invasive plants have become established in the study area, particularly in the lower elevations of the park.

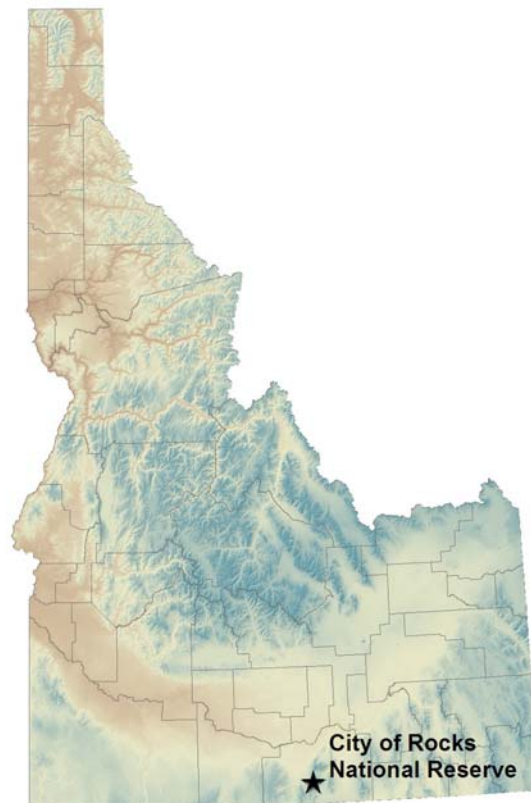


Figure 1. Map of Idaho showing the location of City of Rocks National Reserve. Castle Rocks State Park is adjacent to the Reserve on its northern boundary.

Methods

Two surveying techniques were used over this 2 year, 3 season study. The first method, capture via small mammal live traps, was used to target the cliff chipmunk, pinyon mouse, canyon mouse, and brush mouse. The second method employed motion sensing cameras to target the ringtail and collect information on non-target species as well.

Small Mammal Trapping

A stratified random sampling strategy with repeat surveys was designed in order to enable the probability of occupancy (ψ) and detection (ρ) to be estimated as outlined by MacKenzie et al. (2002, 2006). Strata were adopted simply to ensure that a representative sample was drawn from specific habitat areas within the parks. Specifically, this ensured that areas on or near large rock expanses were adequately sampled. Given the model-based analytical setting for this project, strata were not used to obtain reduced variance estimates, as is frequently done with design-based estimation (Thompson 2002). In 2005, three strata were identified based on habitat associations for these species as reported in recent literature (Fitzgerald et al. 2004, Verts and Carraway 1998, Rompola and Anderson 2004). Using a current land cover map for the reserve, we generated 45 random sample points for each strata; in pinyon-juniper habitat within 30 m of rock, in all other pinyon-juniper habitat, and in non pinyon-juniper habitat within 60 m of pinyon-juniper edge. Expected probabilities of detection and occupancy in optimal habitat areas were based on 1) previous trapping experiences with the pinyon mouse and canyon mouse in the John Day Fossil Beds National Monument in central Oregon, which is also at the northern terminus of these species' distributions, 2) the relative ease with which the pinyon mouse was encountered during a targeted (non-probabilistic sampling) mammal inventory in CIRO in 2003 (Madison et al. 2003), and 3) the frequency of visual sightings of the cliff chipmunk in the Reserve area. We therefore anticipated moderate probabilities of detection and occupancy in optimal habitat areas (i.e. $\rho \approx 0.5$ and $\psi \approx 0.5$) for these species and allocated survey effort in a way that maximized the number of sampling sites and spatial coverage over intensity of trapping effort at individual sites (MacKenzie et al. 2006).

In 2005, two Sherman traps (3x3.5x9" folding traps) were placed at each of 3 trap stations in 10 m intervals along a 20 m transect, for a total of 6 traps per site. Sites were trapped for 3 consecutive nights. We attempted to survey 30 sites in each of the 3 strata, although higher than expected travel time between sites and staffing difficulties early in the project resulted in the survey of only 69 sites. Figure 2 shows the location of these survey sites. Traps were baited with a peanut butter-black oil sunflower seed mix and provided with polyester batting to reduce thermal stress. Traps were pre-baited for 3 nights and left open with tongue depressors to allow animals to become accustomed to the traps, increase capture success, and minimize the influence of trap familiarity on detectability. Random azimuths were selected to establish transect direction from the sampling point origin. Traps were placed within 2 m of the transect line and in ways to maximize capture success (under cover, along runways against rock features, etc.). Traps were checked once in early morning, left closed during the day, and reopened in evening. Animals were removed from traps and checked for key morphological characteristics. To identify *Peromyscus spp.*, tail, ear, and hind foot length measurements were collected, and the number of mammae on adult females were noted to aid in detection of canyon mice (Fitzgerald et al. 1994, Verts and Carraway 1998). Study skins provided by the University of Washington Burke

Museum of Natural History were also studied prior to field work. Cliff chipmunks were easily identified based on dorsal pelage (Fitzgerald et al. 1994).

The sampling design was changed for 2006 based on 2005 capture results. The design revision was intended to balance the need to increase detections, thereby improving occupancy estimation and modeling performance, but to remain fast and efficient in order to meet logistical and personnel constraints. Thirty-six sites were sampled from a stratified random sample drawn from 2 (rather than 3) strata (20 sites per strata). Strata 1 consisted of all pinyon-juniper habitat (including *Cercocarpus* classes) ≤ 30 m from rock. Strata 2 included all other pinyon-juniper habitat. Samples were restricted to the north-central portion of CIRO (off the overlook road and along the main road to “Bread Loaves”) and to CRSP. The sampling frame included only areas within 500 m of traversable roads in order to further reduce travel time between sites. Private land was excluded from the sampling frame as well. Figure 2 shows the 2006 sampling frame and the location of 2006 sampling sites. Two extra sites were sampled in season 2, bringing the total number of sites to 38, but these were not included in analyses since data were missing from season 1. The minimum distance between sampling sites was approximately 50 m. Each site was sampled with 2 (rather than 1) 20 m transects that crossed at the 10 m mid-point as shown in Figure 3. Transects were oriented by placing the first transect along a random azimuth and placing the second one perpendicular to the first. Two Sherman traps were placed at trap stations every 5 m along transects for a total of 20 traps per site (effectively a 10 m radius plot). This approach resembled a modification of the trapping web approach described by Parmenter et al. (2003). Traps were pre-baited and left open for 3 nights and operated for 3 consecutive nights as was done in 2005.

A small subset of captured individuals were collected during both years as voucher specimens and sent to the Burke Museum for skull and skin preparation, species confirmation, and curation. *Peromyscus* individuals were selected for collection based on “promising” features suggestive of target species (i.e. tail length > 75 mm for *Peromyscus*). Verts and Carraway (1998), Fitzgerald et al. (1994), and species accounts from the Mammalian Species series were used to identify target species (Hoffmeister 1981, Johnson and Amrstrong 1987, Hart 1992). *Tamias* and *Microtus* can be difficult to identify to species in the field and were collected as well. An additional subset of pinyon mice specimens collected in 2006 were subjected to additional molecular analysis by the Portland State University genetics lab to confirm species identity through sequencing of the cytochrome-b mitochondrial gene

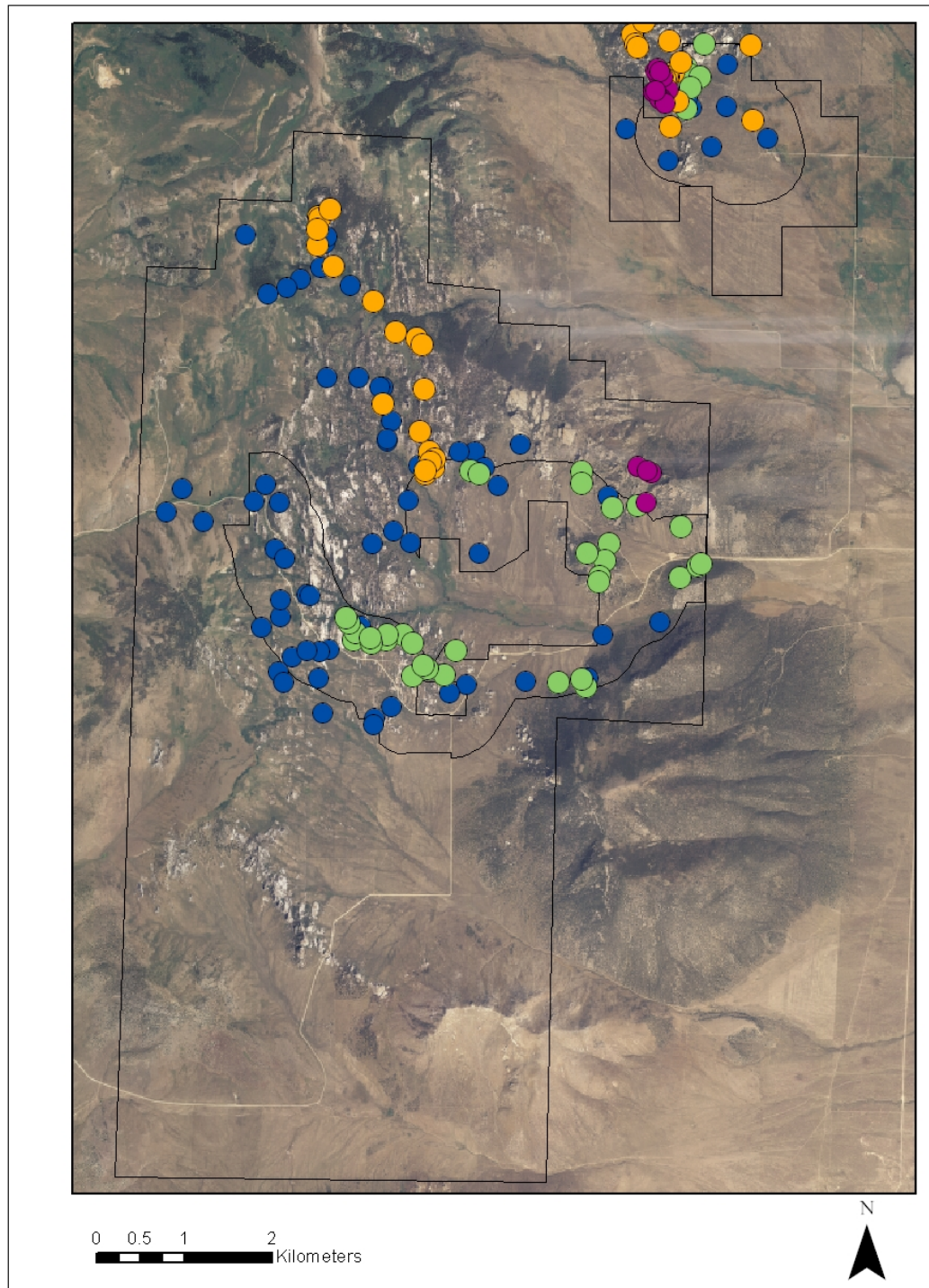


Figure 2. The City of Rocks National Reserve and Castle Rocks State Park study areas, 2006 trap sampling frame, 2005 trapping sites (blue), 2006 trapping sites (green), 2005 camera locations (purple), and 2006 camera locations (orange). The southern sampling frame polygon in City of Rocks included 82 hectares, and the smaller northern sampling frame polygon in Castle Rocks State Park included 22 hectares.

Vegetation and substrate cover was estimated by the point-line intercept method with a 6 mm wide 2.6 m long pole following recommendations by Elzinga et al. (2001). 20 points per transect (1 m interval) were measured, enabling a 2.5% precision for cover estimates. Cover estimates were obtained for rock expanses larger than 5 m² (Rock), bare ground and litter (Bare), grass and forbs (Grass), shrubs, and trees. Shrub height was measured with a ruler painted on the intercept pole and tree height was estimated in 4 broad height classes. Site distance to rock expanse (DistRock) and pinyon-juniper woodland edge (DistEdge) was estimated in the field and checked with 1 m National Agricultural Imagery Project (NAIP) aerial photography (available from the USDA at <http://165.221.201.14/NAIP.html>). We measured the precipitation for each 24 hour period preceding each morning's trap check with a Tru-Check gauge that is accurate to the hundredth inch when measuring up to 0.20 inches placed at the CRSP headquarters. This information was collected in order to account for any effect precipitation may have had on detectability and capture success. We developed sampling covariate matrices for precipitation for each survey and site to include in models estimating ρ (probability of detection). We also developed a sampling covariate matrix that allowed ρ to change after the first capture of a target species in order to account for the influence of "trap happy" or "trap shy" individual rodents on ρ .

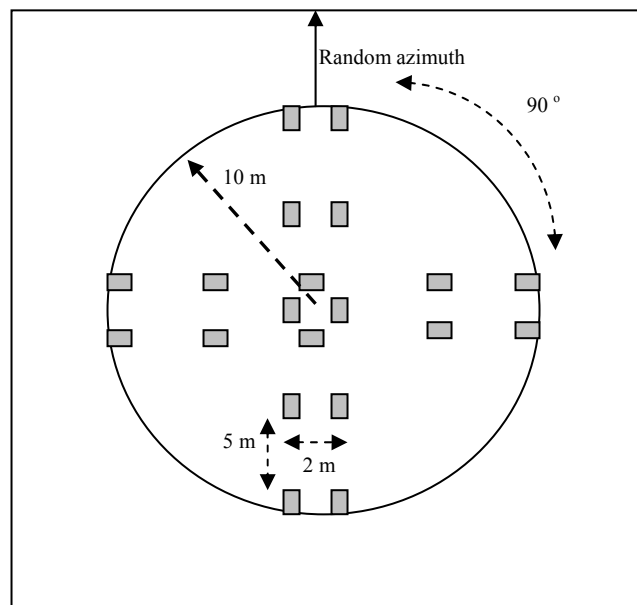


Figure 3. The 10-m radius plot used for small mammal live trapping in 2006. Gray boxes represent Sherman live traps.

Statistical analysis was conducted in the R language and software environment (<http://www.r-project.org/>) and in PRESENCE 2.0 (<http://www.mbr-pwrc.usgs.gov/software.html>). We relied on exploratory data analysis using graphical and summary statistical tools to refine existing hypotheses about habitat features influencing site occupancy (e.g. a null model describing H_0 : target species probability of occupancy is randomly distributed across the study site, versus H_a : target species probability of occupancy varies non-randomly across the study site according to habitat features as described in existing literature) and to develop a suite of candidate occupancy

models. Occupancy models were developed following recommendations by MacKenzie et al. (2002, 2003, 2006) and fit to observed data using PRESENCE.

We used a specific parameterization of the multi-season model available in PRESENCE, described as an “implicit dynamics” model by MacKenzie et al. (2006), that assumes random change in ψ between seasons (probability of “local colonization” and “local extinction” is 0). This parameterization is notated as $\psi(.)\gamma(.)\epsilon=1-\gamma,p(.)$, and the estimate for gamma (γ) represents ψ in season 2, allowing direct estimation of ψ for each season without having to fit separate single-season models. While this approach clearly fails to account for the possibility of serial correlation between seasons (e.g. sites occupied in season 1 have a higher probability of being occupied in season 2) we felt that this was the most appropriate and parsimonious approach given only 2 seasons of data and uncertainty in hypotheses about seasonal dynamics given the inherent fluctuations in rodent populations. We also briefly explored an alternative parameterization of the form $\psi(.)\epsilon(.)p(.)$ for modeling pinyon mouse detection histories that permits season-specific estimates of ψ and explicitly accounts for between-season extinction and colonization probabilities. We present this result as a hypothesis generating exercise but did not pursue this avenue of inquiry in depth, as we are of the opinion that, although promising, additional sampling will be required in order to obtain reliable results.

Model sets subject to consideration included a null model $\psi(.)\gamma(.)\epsilon=1-\gamma,p(.)$ which describes a scenario in which species occupancy probabilities would be randomly distributed across the study area and equally likely to be detected at all sites and surveys within a season, a global model in which all site habitat and sampling covariates hypothetically influencing occupancy and detectability were included, and a series of intermediate “reduced” models with subsets of influential covariates. We reduced the number of covariates included in the global model through consideration of graphical descriptions, tests of significance at the $\alpha=0.10$ level (via a two-sample Welch’s t -test for unequal variance), and Pearson’s correlation coefficients. We assessed model goodness-of-fit following recommendations by MacKenzie and Bailey (2004) and MacKenzie et al. (2006) by obtaining a χ^2 test statistic for observed and expected species detection histories from 10,000 bootstrap resamples from a single-season global model ($\psi(.)p(.)$) and calculating the summary statistic $\hat{c}$. Goodness-of-fit is presumed adequate when $\hat{c}$ is near 1. Model selection relied on Akaike’s Information Criterion corrected for small sample sizes (AIC_c) (Burnham and Anderson 1998, MacKenzie et al. 2006). “Weight of evidence” was assessed with ΔAIC_c , model weights (w_c), consideration of variance estimates, and biological relevance. The focus for this study was on obtaining site-specific estimates of occupancy and site-sample-specific estimates of detectability rather than on the actual effect sizes of individual model covariates (odds ratios). Despite uncertainty in model selection, top models ($\leq 4 AIC_c$ values) provided nearly identical parameter estimates. Model averaging, therefore, was not done for this report but we acknowledge its potential value. We obtained site estimates of ψ , γ , and p directly from the PRESENCE output for the “best” model but obtained accurate 95% confidence intervals for the estimates by transforming site probability estimates to the logit scale, calculating standard errors of each transformed value by multiplying the appropriate variance-covariance matrix by the corresponding vector of covariate values for each site, and exponentiating interval endpoints back to probabilities (D. MacKenzie, Proteus Wildlife Research Consultants, personal communication; T. Weller, U.S. Forest Service, personal communication).

Motion Cameras

Five motion-triggered cameras in the 2005 season and 6 cameras in the 2006 season were employed in targeted, non-random locations. Survey sites began with the location of the 2003 ringtail carcass recovery and 2005 ringtail winter tracks. In 2006, the “Tahitian Wall” was heavily focused on for 1 month in response to a ringtail sighting that same spring. IDFG contributed 4 Bushnell Trailscout Pro 2.1 mega pixel resolution cameras to this project, each with a 90-foot passive infrared motion sensor and automatic night flash. IDPR contributed 2 Stealth Cam 3.1 mega pixel cameras with a 30-foot passive infrared motion sensor and automatic night flash. Baited camera stations were placed in obvious travel corridors along the base of cliffs and through narrow game trails in woodland vegetation.

A basic vegetation description was conducted at each location within an 8-meter radius circle around the camera. Stem counts were taken for tree species and their associated height classes were recorded (0-2m, 2-10m, >10m). Ground cover was assessed for percent bare ground, herbaceous cover and shrub cover. Percent aerial cover was approximated. Estimates of proximity of the camera to rock, tree, deadfall, and woodland edge were obtained in the field. Cameras were mounted 2-3 feet off the ground on mountain mahogany, pinyon pine or juniper, and focused toward the baited stations. Cameras were left in place a least 1 week before relocation in order to balance the objectives of covering as many locations as possible while still giving adequate time at each locality. Several different commercially available scents and lures were used to attract ringtails to the camera locations.

Results

Small Mammal Trapping

Sixty-nine sites were sampled between 28 August and 13 October 2005. There were 6 traps per site and 3 consecutive trapping events per site. After removing 1 trap night in which a trap was not set, trapping effort totaled 1,241 trap nights. 460 individual rodents were captured representing 6 species: 435 deer mice (*Peromyscus maniculatus*), 9 Great Basin pocket mice (*Perognathus parvus*), 4 least chipmunks (*Tamias minimus*), 4 cliff chipmunks, 3 montane voles (*Microtus montanus*), 1 long-tailed vole (*Microtus longicaudus*), 2 unidentified *Peromyscus* spp., and 2 unidentified rodents from escaped captures. Overall capture success was 37%. Regarding percent of overall 2005 captures, deer mice represented 94%, Great Basin pocket mice 2%, and cliff chipmunks 0.8%. Cliff chipmunks were captured at 4 sites in 2005 for a naïve ψ estimate of 0.05 (e.g. if detection were perfect).

Twenty-five specimens from 2005 were prepared and curated by the Burke Museum (see Table 1). This resulted in confirmation of 13 deer mice, 4 Great Basin pocket mice, 2 cliff chipmunks, 1 long-tailed vole, and 5 juvenile *Peromyscus* spp. that are all likely deer mice (J. Bradley, University of Washington Burke Museum of Natural History, personal communication).

Table 1. Specimens collected during the 2005 small mammal trapping that were prepared and curated by the University of Washington Burke Museum of Natural History.

Burke Museum Catalog Number	Species
78892	<i>Peromyscus maniculatus</i>
78893	<i>Peromyscus maniculatus</i>
78894	<i>Peromyscus</i> spp.
78895	<i>Peromyscus maniculatus</i>
78896	<i>Peromyscus maniculatus</i>
78897	<i>Perognathus parvus</i>
78898	<i>Microtus longicaudus</i>
78899	<i>Peromyscus maniculatus</i>
78900	<i>Peromyscus maniculatus</i>
78901	<i>Perognathus parvus</i>
78902	<i>Peromyscus</i> spp.
78903	<i>Tamias dorsalis</i>
78904	<i>Peromyscus maniculatus</i>
78905	<i>Peromyscus maniculatus</i>
78906	<i>Peromyscus</i> spp.
78907	<i>Peromyscus maniculatus</i>
78908	<i>Tamias dorsalis</i>
78909	<i>Peromyscus</i> spp.
78910	<i>Peromyscus maniculatus</i>
78911	<i>Peromyscus</i> spp.
78912	<i>Perognathus parvus</i>
78913	<i>Perognathus parvus</i>
78914	<i>Peromyscus maniculatus</i>
78915	<i>Peromyscus maniculatus</i>
78916	<i>Peromyscus maniculatus</i>

Thirty-six sites were sampled between 13 June and 20 July 2006 (season 1). There were 20 traps per site, 3 consecutive trapping events per site, and 108 nights surveyed. This yielded a trapping effort of 2,160 trap nights and a total of 921 individual rodents captured. There were 591 deer mice captured at all 36 sites. Deer mice were caught every night at every plot except on 5 occasions (95% of trapping occasions). There were 123 Great Basin pocket mice, 89 pinyon mice, 57 least chipmunks, 23 cliff chipmunks, 19 *Microtus* spp. (3 of those confirmed as montane voles), 13 bushy-tailed woodrats (*Neotoma cinerea*), 4 golden-mantled ground squirrels (*Spermophilus lateralis*), and 1 unidentified *Peromyscus* spp. Figure 4 shows capture count data for the pinyon mouse and cliff chipmunk for season 1.

Overall capture success for season 1 in 2006 was 42%. Deer mice represented 64% of the captures, Great Basin pocket mice represented 13%, cliff chipmunks represented 2%, and the pinyon mouse represented 9%. Overall species richness, excluding the unidentified *Peromyscus* spp. and pooling the *Microtus* spp. into 1 category, was 8 species. There is a potential richness of 9 or 10 species given that several animals captured were not positively identified to the species level due to confusing external morphology. Cliff chipmunks were captured at 8 sites during 2006, season 1, for a naïve ψ estimate of 0.22. Pinyon mice were captured at 13 sites during 2006 season 1 for a naïve ψ estimate of 0.36.

Thirty-eight sites were sampled between 5 September and 27 September 2006 (season 2). There were 114 nights surveyed totaling 2,280 trap nights. This resulted in 850 individual rodents captured. There were 519 deer mice captured, 104 Great Basin pocket mice, 50 pinyon mice captured, 67 cliff chipmunks, 3 golden-mantled ground squirrels, 82 least chipmunks, 12 montane voles, 6 bushy-tailed woodrats, 2 *Peromyscus* spp., 1 Ord's kangaroo rat (*Dipodomys ordii*), and 3 yellow-pine chipmunks (*Tamias amoenus*). Season 2 sampling resulted in the detection of 10 species.

Overall capture success for season 2 in 2006 was 37%. Deer mice represented 61% of the captures, Great Basin pocket mice 12%, pinyon mice 6%, cliff chipmunks 8%, and least and yellow-pine chipmunks represented 10%. Figure 4 shows capture count data for the pinyon mouse and cliff chipmunk for season 2. Cliff chipmunks were captured at 18 sites in 2006 season 2 for a naïve ψ estimate of 0.5. Pinyon mice were captured at 9 sites in 2006 season 2 for a naïve ψ estimate of 0.25. Table 2 presents a comparison of capture success, scaled by trap night, between 2005 and 2006, as a measure of the striking difference between capture success and relative abundance of species between years. Table 3 provides location coordinates for cliff chipmunk and pinyon mice capture sites obtained from 2005 and 2006 sampling efforts.

For the 2006 effort, 34 vouchers of 10 species were confirmed by the Burke Museum (see Table 4). Tissue samples from 6 of the field-identified pinyon mice were confirmed the same through amplification and sequencing of the cytochrome-*b* mitochondrial gene by the Portland State University genetics lab (Jan Zinck, PSU Biology, personal communication).

Three specimens were confirmed as yellow-pine chipmunks by the Burke Museum in season 2 (J. Bradley, personal communication). It is likely that the yellow-pine chipmunk was captured during season 1 in 2006 but no vouchers were obtained for that species. An unknown proportion of captured species identified as least chipmunks are probably yellow-pine chipmunks. These two species are extremely difficult to distinguish from external morphology (Verts and Carraway 1998).

Table 2. Number of captured animals per trap night for the deer mouse (PEMA), Great Basin pocket mouse (PEPA), pinyon mouse (PETR), and the cliff chipmunk (TADO).

Species	2005	2006	
		Season 1	Season 2
PEMA	0.35	0.27	0.22
PEPA	0.007	0.056	0.045
PETR	0.00	0.041	0.022
TADO	0.003	0.01	0.03

Table 3. Universal Transverse Mercator (UTM) location coordinates for locations where the cliff chipmunk (*Tamias dorsalis*) and the pinyon mouse (*Peromyscus truei*) were captured during 2005 and 2006. Coordinates were collected in UTM Zone 12, using the North American Datum (horizontal) of 1983. Estimated GPS accuracy at the time of collection was 3 m.

Tamias dorsalis (2005)			Tamias dorsalis (2006)			Peromyscus truei		
Plot	UTM E	UTM N	Plot	UTM E	UTM N	Plot	UTM E	UTM N
22	275819	4664107	1	279379	4668394	1	279379	4668394
71	276876	4663567	4	276813	4663499	6	277981	4663521
181	275494	4661655	6	277981	4663521	22	278185	4662332
281	275463	4661761	7	275570	4661632	23	278180	4662272
			8	275946	4661666	24	278625	4663143
			10	279181	4668122	26	279113	4662900
			13	275772	4661662	29	277991	4663395
			14	275754	4661603	32	278255	4662515
			15	275584	4661574	33	278047	4662600
			16	275332	4661797	40	278297	4662709
			17	275301	4661865	42	278331	4663115
			18	275412	4661696	47	279310	4662456
			22	278185	4662332	48	279107	4662316
			24	278625	4663143			
			29	277991	4663395			
			33	278047	4662600			
			40	278297	4662709			
			42	278331	4663115			
			47	279310	4662456			
			48	279107	4662316			
			49	277984	4661156			

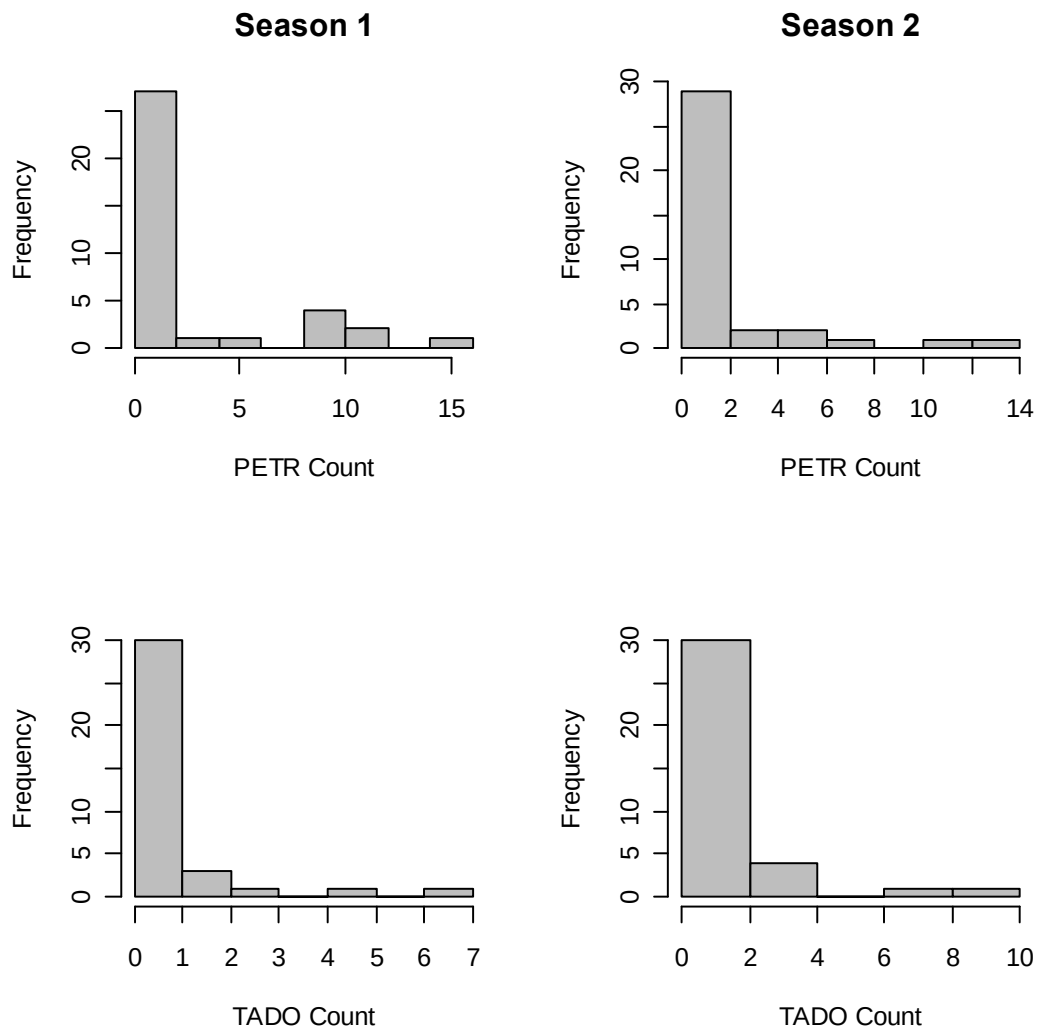


Figure 4. Frequencies of site capture counts for the pinyon mouse, *Peromyscus truei* (PETR) and the cliff chipmunk, *Tamias dorsalis* (TADO). More pinyon mice were captured in season 1, 2006 (89 versus 50) and fewer cliff chipmunks were captured in season 1 (23 versus 49).

Table 4. Specimens collected during the 2006 small mammal trapping that were prepared and curated by the University of Washington Burke Museum of Natural History. Tail, foot, and ear lengths are reported here because of their importance in identification of *Peromyscus truei*, which was only collected in 2006.

Burke Museum Catalog Number	Species	Length (mm)		
		Tail	Foot	Ear
79645	<i>Peromyscus truei</i>	71	22	24
79646	<i>Peromyscus truei</i>	98	23	27
79647	<i>Peromyscus truei</i>	93	23	27
79648	<i>Tamias minimus</i>	85	29	14
79649	<i>Tamias amoenus</i>	96	32	20
79650	<i>Peromyscus maniculatus</i>	61	20	16
79651	<i>Peromyscus maniculatus</i>	68	19	15
79652	<i>Spermophilus lateralis</i>	90	40	18
79653	<i>Microtus montanus</i>	60	22	13
79654	<i>Tamias amoenus</i>	94	31	17
79655	<i>Microtus montanus</i>	55	20	13
79656	<i>Tamias amoenus</i>	98	33	17
79657	<i>Tamias dorsalis</i>	95	33	21
79658	<i>Neotoma cinerea</i>	126	44	29
79659	<i>Peromyscus truei</i>	82	22	24
79660	<i>Peromyscus maniculatus</i>	73	20	18
79661	<i>Peromyscus truei</i>	89	23	25
79662	<i>Peromyscus maniculatus</i>	60	19	16
79663	<i>Peromyscus maniculatus</i>	55	19	14
79664	<i>Microtus montanus</i>	58	20	13
79665	<i>Perognathus parvus</i>	97	24	7
79666	<i>Peromyscus truei</i>	88	23	25
79667	<i>Perognathus parvus</i>	93	22	8
79668	<i>Perognathus parvus</i>	70	22	7
79669	<i>Peromyscus maniculatus</i>	79	21	20
79670	<i>Perognathus parvus</i>	95	23	7
79671	<i>Tamias dorsalis</i>	92	33	22
79672	<i>Dipodomys ordii</i>	141	38	12
79673	<i>Peromyscus maniculatus</i>	73	21	18
79674	<i>Peromyscus truei</i>	90	25	28
79675	<i>Peromyscus maniculatus</i>	68	20	18
79676	<i>Peromyscus maniculatus</i>	62	19	16
79677	<i>Peromyscus truei</i>	90	23	25
79678	<i>Tamias minimus</i>	94	29	14

The pattern of relative abundance and detection at sites between seasons differed between the cliff chipmunk and the pinyon mouse. For the cliff chipmunk, all but one site with captures in season one had captures in season 2, and an additional 9 sites gained detections in season 2 (“local colonization”). For the pinyon mouse, 4 sites with detections in season 1 had no detections in season 2 (“local extinction”) and no additional colonization was detected.

Because of the low capture rate for target species in 2005, we did not pursue exploration of habitat characteristics or occupancy models with 2005 data. Table 5 presents summary statistics for each of the habitat characteristics measured at sites in 2006. In 2006, cliff chipmunks were detected at sites with significantly higher rock cover in season 1 than in sites without detections and at sites with higher rock and lower grass/forb and shrub cover in season 2 ($P < 0.05$, Welch’s *t*-test). Figures 5, 6, and 7 show box plots comparing the median and quartiles of mean cover and distances to woodland edge and rock expanse for sites with and without cliff chipmunk detections. Distance to woodland edge and tree cover was marginally greater in sites with cliff chipmunk detections during season 2 ($P < 0.10$). Pinyon mice were captured in sites with significantly higher bare ground cover and at greater distances to woodland edge than sites without detections during both seasons ($P < 0.05$; Figures 7, 8, and 9). Likewise, pinyon mice capture sites also had significantly lower shrub and grass understory during both seasons ($P < 0.05$).

Table 5. Range and mean values for each habitat variable measured at 2006 sampling sites.

Habitat Variable	Min	Mean	Max
Distance to Rock (m)	0	41	300
Distance to Edge (m)	0	90	225
Rock Cover (%)	0.00	0.17	0.87
Bare/Litter Cover (%)	0.02	0.34	0.80
Grass/Forb Cover (%)	0.00	0.40	0.77
Shrub Cover (%)	0.00	0.19	0.65
Tree Cover (%)	0.15	0.44	0.80
Shrub Height (cm)	0.0	65.7	196.0
Tree Height Class (4 classes)	2	3	4

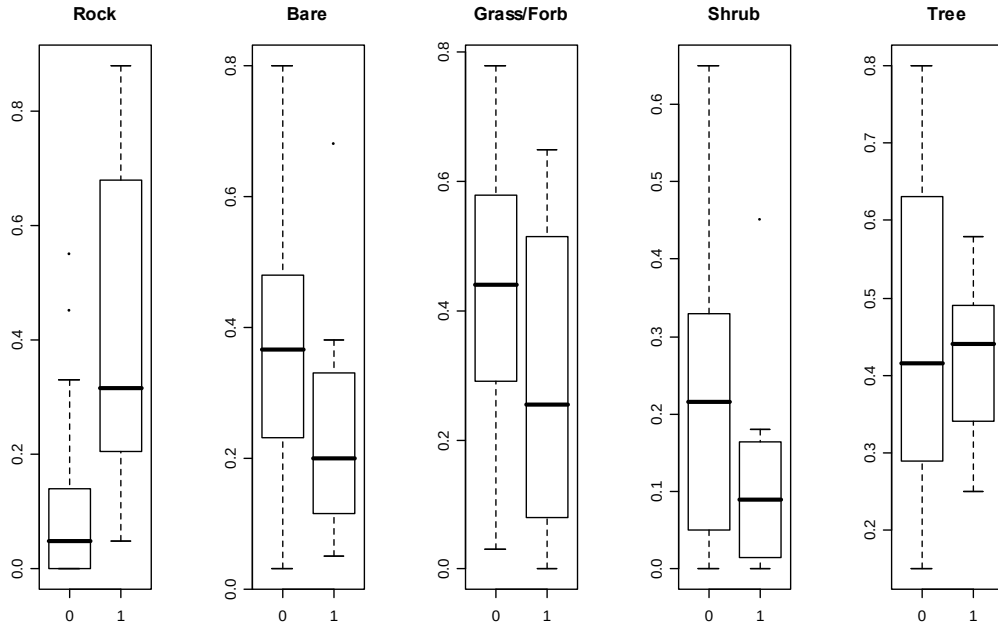


Figure 5. Box and whisker plots comparing median and quartiles for the average amount of cover in sites with and without cliff chipmunk detections (1=detected) during season 1.

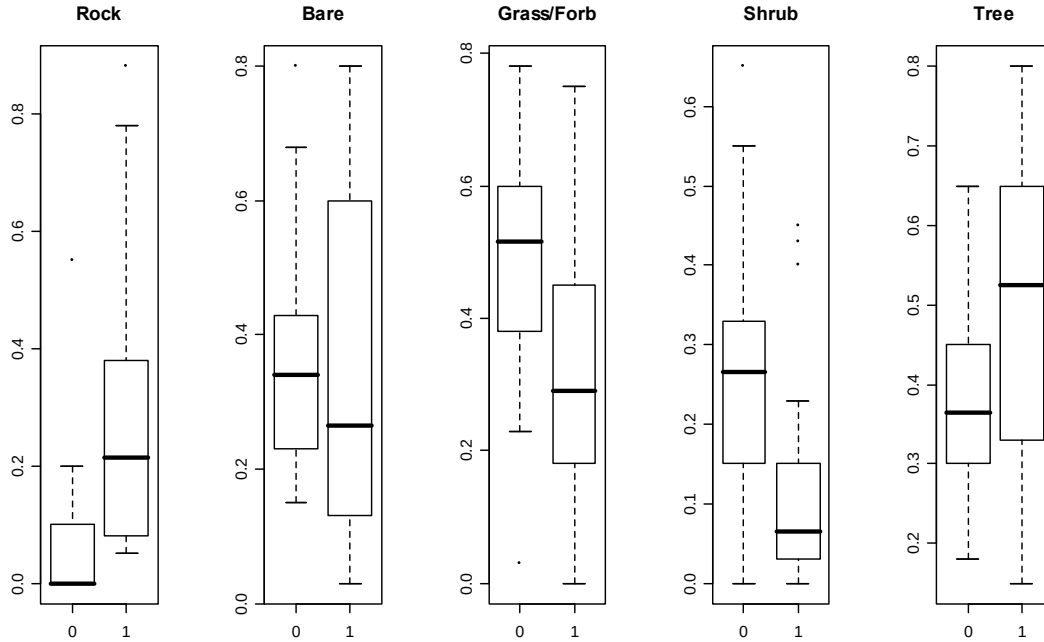


Figure 6. Box and whisker plots comparing median and quartiles for the average amount of cover in sites with and without cliff chipmunk detections (1=detected) during season 2.

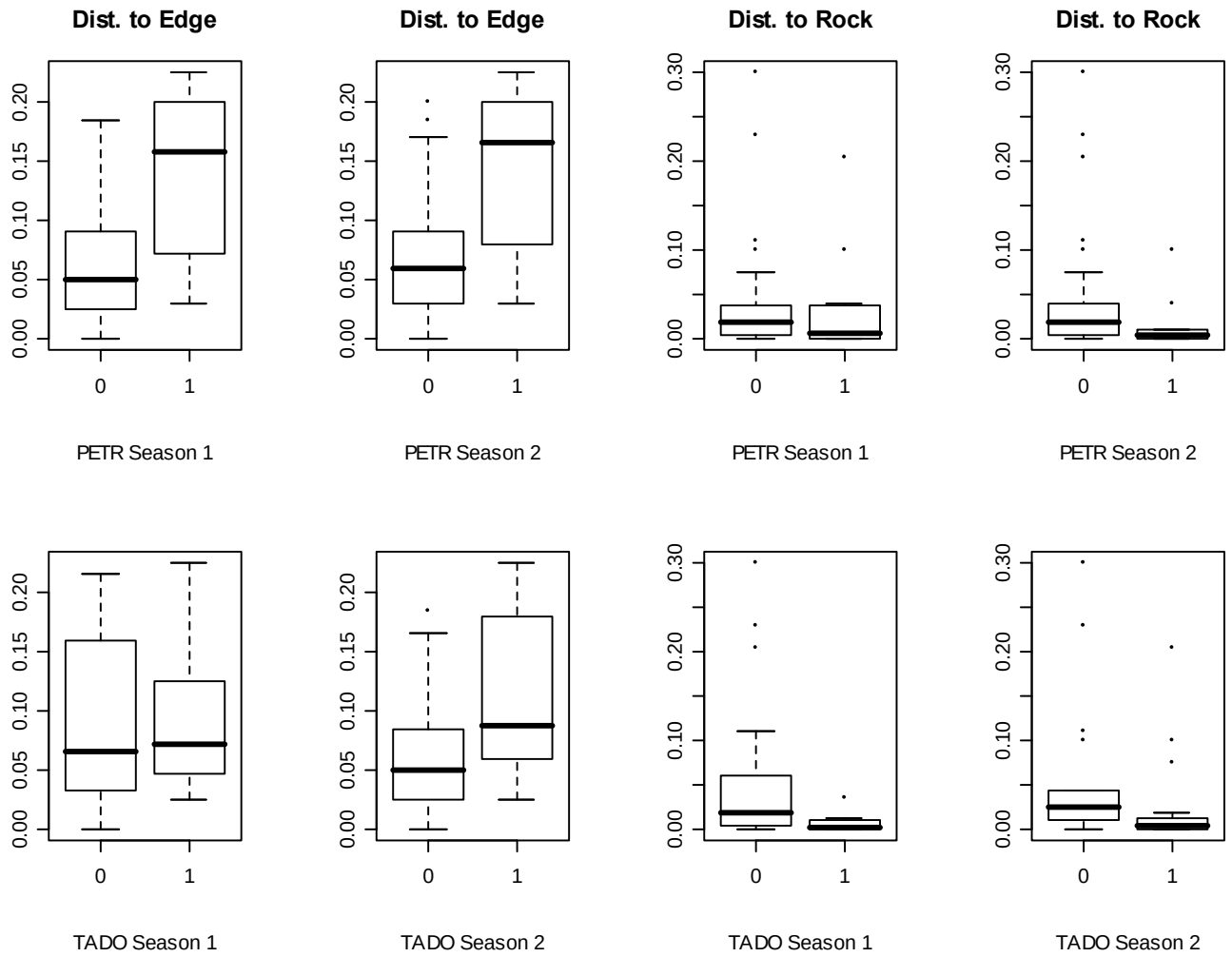


Figure 7. Box and whisker plots comparing median and quartiles for the average distance to woodland edge and rock expanse ($> 5\text{m}^2$) for sites with and without pinyon mouse (PETR) and cliff chipmunk (TADO) captures.

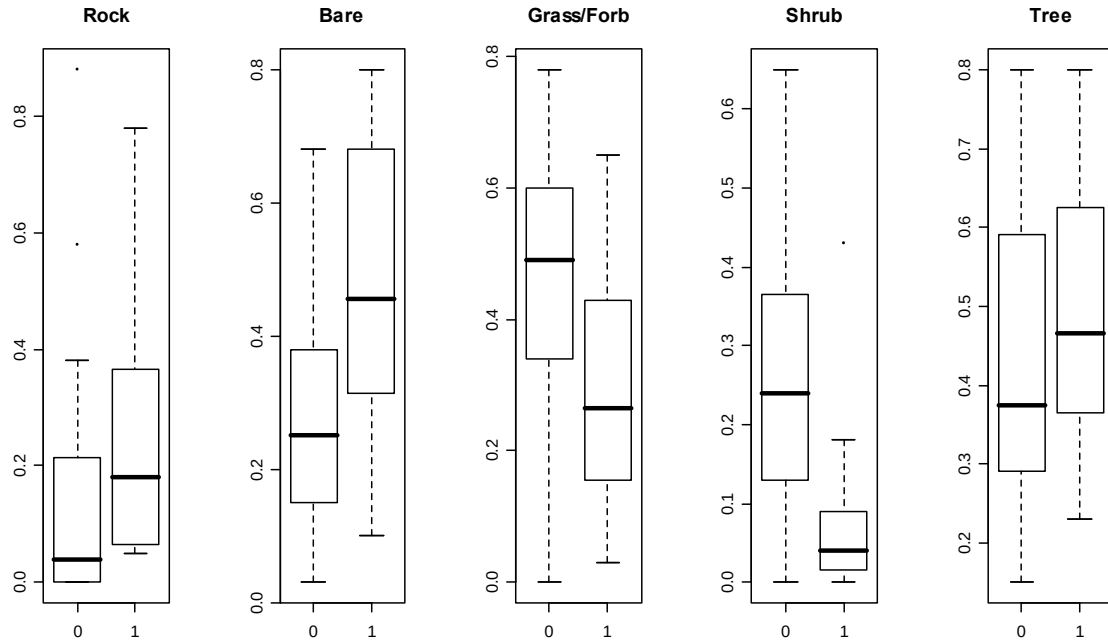


Figure 8. Box and whisker plots comparing median and quartiles for the average amount of cover in sites with and without pinyon mouse detections (1=detected) during season 1.

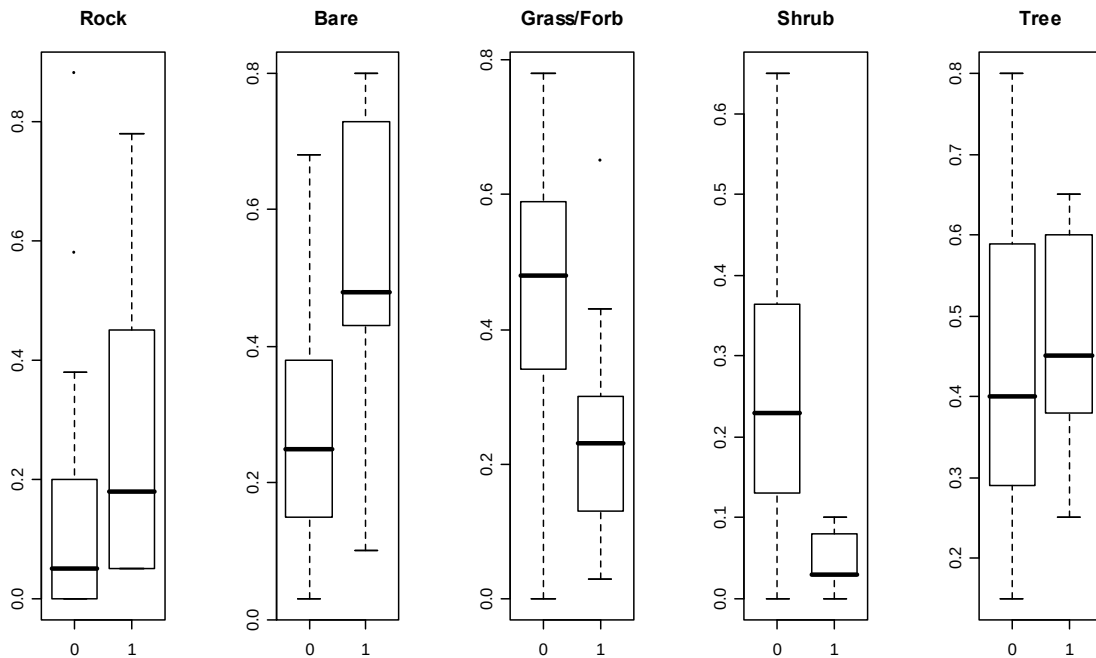


Figure 9. Box and whisker plots comparing median and quartiles for the average amount of cover in sites with and without pinyon mouse detections (1=detected) during season 2.

In preparation for model fitting, we calculated Pearson's correlation coefficients for each cover and distance covariate. All coefficients (r) were $< \pm 0.65$ except shrub cover * distance to edge ($r = -0.77$), rock cover * grass cover ($r = -0.70$), and shrub cover * grass cover ($r = 0.70$). Because of the importance of each of these covariates as unique measures of site habitat, we opted to include each of these in modeling exercises. Modeling proceeded by fitting a single-season global model containing parameters for significant site covariates for the estimation of ψ (occupancy) and a single parameter for the "first capture" sampling covariate for ρ (detectability) that allowed site detection probabilities to change after first capture of a target species. Model fit was assessed for the cliff chipmunk using season 2 detection histories since this contained all but 1 of the 19 total capture sites for this species. Model fit was assessed for the pinyon mouse using season 1 detection histories since this contained all 13 capture sites. Model fit was determined to be adequate. P-values from the χ^2 goodness-of-fit test were 0.77 and 0.28, respectively, for the cliff chipmunk and pinyon mouse and $\hat{c}$ for each species was 0.58 and 1.13, respectively. We evaluated sampling covariates for detectability at this stage with single season models as well and concluded that the precipitation covariate was not appropriate or necessary to consider in subsequent steps.

Table 6 presents parameter estimates from the null model $\psi(.)\gamma(.)\epsilon=1-\gamma,p(.))$, which provides an overall estimate of the proportion of area occupied within the sampling frame for each season, as well as the probability of detection. Naïve ψ is the simple proportion of total sites surveyed in which the target species was detected. Because ρ is high for the pinyon mouse, the incorporation of detectability provides little additional information and adjusted estimates are nearly identical (rounding gives an appearance of exact equality). Adjusted estimates are higher than naïve estimates for the cliff chipmunk because of the lower overall probability of detection for this species during our survey period.

Table 7 presents the suite of ranked candidate models for the cliff chipmunk. In this case, γ should be interpreted as the probability of occupancy for season 2. Based on ΔAIC_c and w_c , the top model, $\psi(\text{rock}),\gamma(\text{rock,shrub,distedge}),\epsilon=1-\gamma,p()$ has considerable support. Figures 10 and 11 show plots of site-specific estimates for ψ and γ and 95% confidence intervals for these estimates. These intervals are quite wide and standard errors ranged from 0.05 to 0.23 for ψ and from 0.03 to 0.31 for γ . Nonetheless, a map of sites coded by ψ estimates and observed detections (Figure 12) suggests that the model provided reasonable estimates and was relatively successful in describing the observed pattern of occupancy. Figure 13 shows a scatterplot of sites, coded by observed detections, by γ and shrub cover. Sites with cliff chipmunk captures and a high probability of occupancy are clearly associated with low shrub cover. This top model estimated ρ to be 0.51, nearly the same as that provided by the null model.

Table 6. Parameter estimates from the fit of the null model $\psi(\cdot)\gamma(\cdot)\epsilon=1-\gamma,p(\cdot)$ to 2 seasons of detection history data from 2006 sampling in CIRO and CRSP. TADO = *Tamias dorsalis* and PETR = *Peromyscus truei*.

Species	Naïve ψ_1	Naïve ψ_2	ψ	SE_ψ	γ	SE_γ	ρ	SE_ρ
TADO	0.22	0.50	0.24	0.07	0.55	0.09	0.52	0.06
PETR	0.36	0.25	0.36	0.08	0.25	0.07	0.81	0.05

Table 7. Ranked model set for the cliff chipmunk, fit to 2 seasons of detection history data from 2006 sampling in CIRO and CRSP. k represents the number of model parameters. Site occupancy covariates include rock cover (rock), grass cover (grass), shrub cover (shrub), distance to edge (distedge), distance to rock (distrock). Covariates for detectability include night following first capture of the target species (firstcapture), and a two parameter covariate for season-specific detectability.

Model	k	AIC _c	Δ AIC _c	w _c
psi(rock),gam(rock,shrub,distedge),eps=1-gam,p()	7	179.92	0.00	0.79
psi(rock),gam(shrub,distedge),eps=1-gam,p()	6	184.41	4.49	0.03
psi(rock),gam(distedge),eps=1-gam,p()	5	186.02	6.10	0.08
psi(rock),gam(grass,shrub),eps=1-gam,p()	6	186.03	6.11	0.02
psi(rock),gam(rock,grass,shrub,distedge,distrock),eps=1-gam,p()	9	186.50	6.58	0.04
psi(rock),gam(grass,shrub,distedge),eps=1-gam,p()	7	187.01	7.09	0.04
psi(rock),gam(grass,shrub,distedge,distrock),eps=1-gam,p()	8	190.00	10.08	0.01
psi(.),gam(.),eps=1-gam,p()	3	195.82	15.90	0.00
psi(.),gam(.),eps=1-gam,p(firstcapture)	4	197.98	18.06	0.00
psi(.),gam(.),eps=1-gam,p(season)	4	198.11	18.19	0.00
psi(.),gam(.),eps=1-gam,p(season,firstcapture)	5	200.23	20.31	0.00

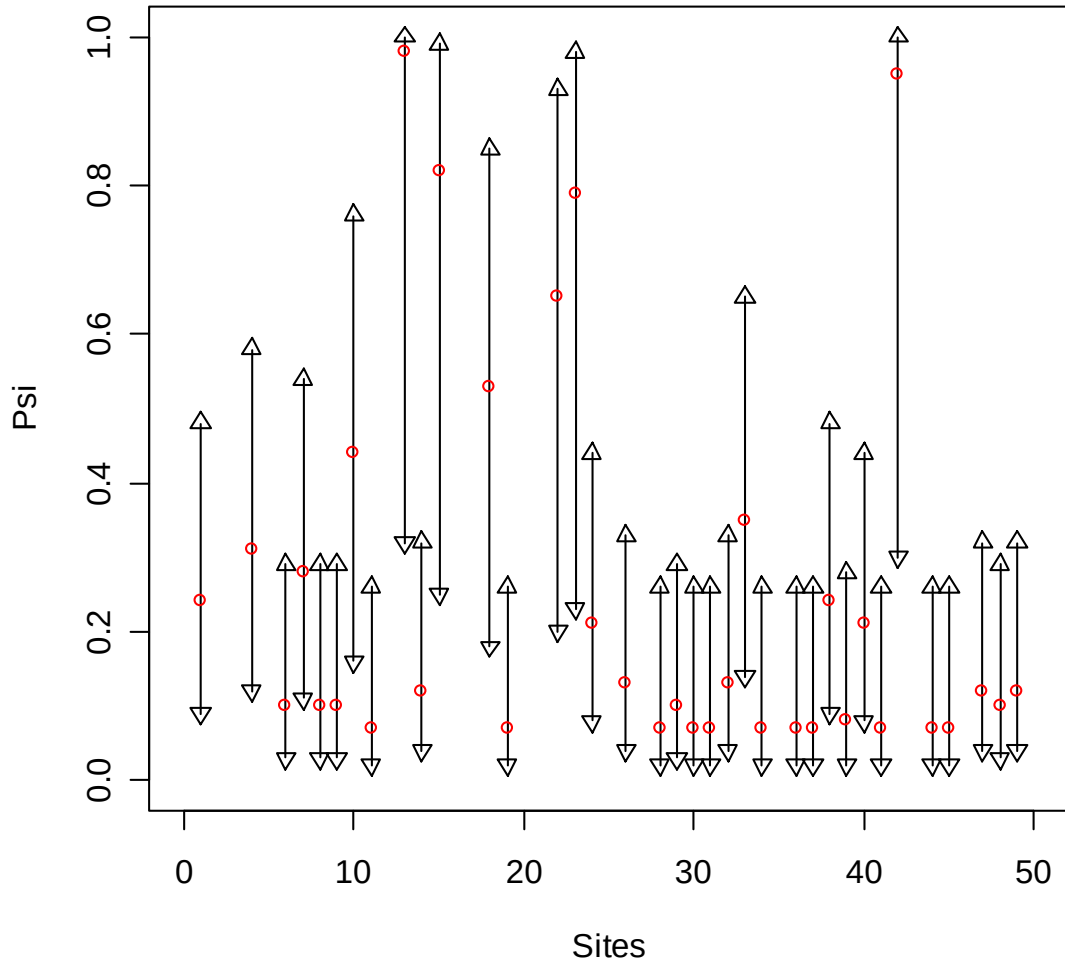


Figure 10. Site estimates of ψ (red circles) and 95% confidence intervals (black lines) and end points (triangles) from the fit of the top model $\psi(\text{rock}), \gamma(\text{rock}, \text{shrub}, \text{distedge}), \varepsilon = 1 - \gamma, p()$ to the observed 2006 detection histories of the cliff chipmunk in CIRO and CRSP.

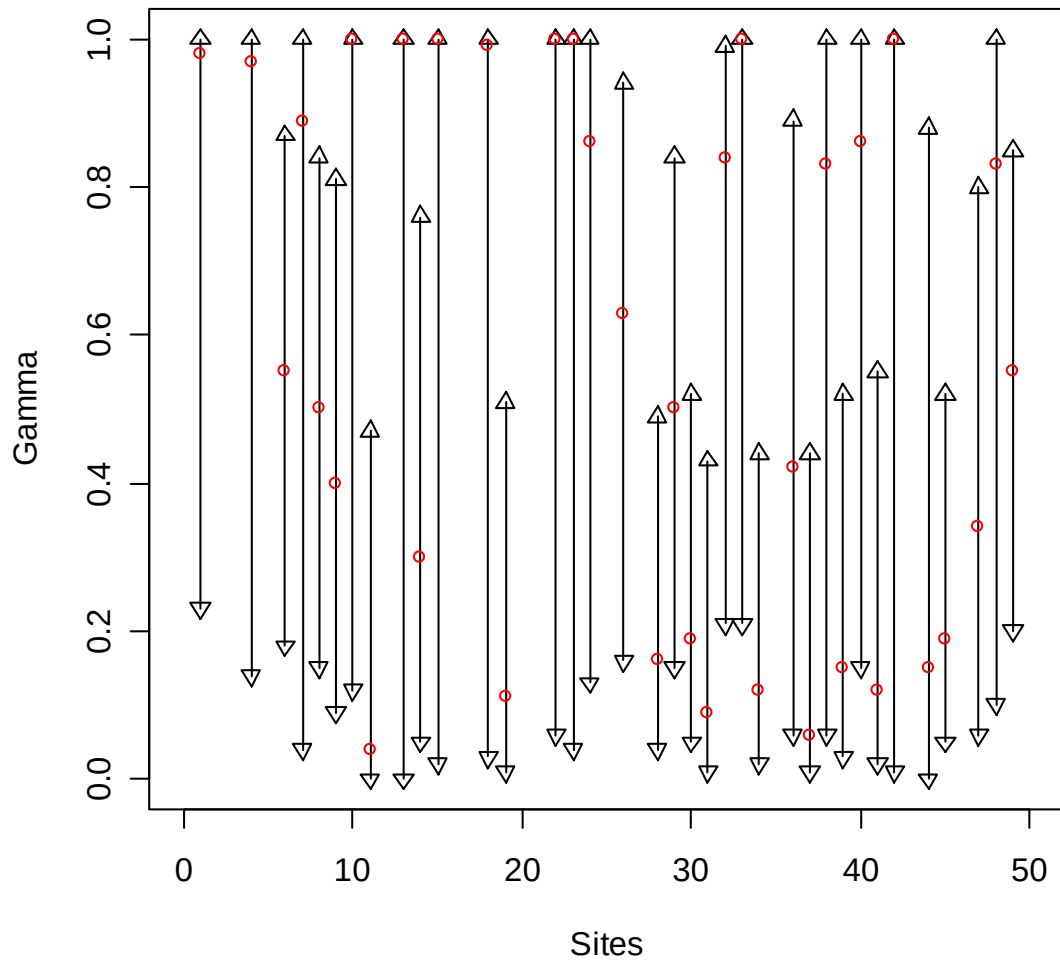


Figure 11. Site estimates of γ (red circles), the probability of occupancy for season 2, and 95% confidence intervals (black lines) and end points (triangles) from the fit of the top model $\psi(\text{rock}), \gamma(\text{rock}, \text{shrub}, \text{distedge}), \varepsilon = 1 - \gamma, p()$ to the observed 2006 detection histories of the cliff chipmunk in CIRO and CRSP.

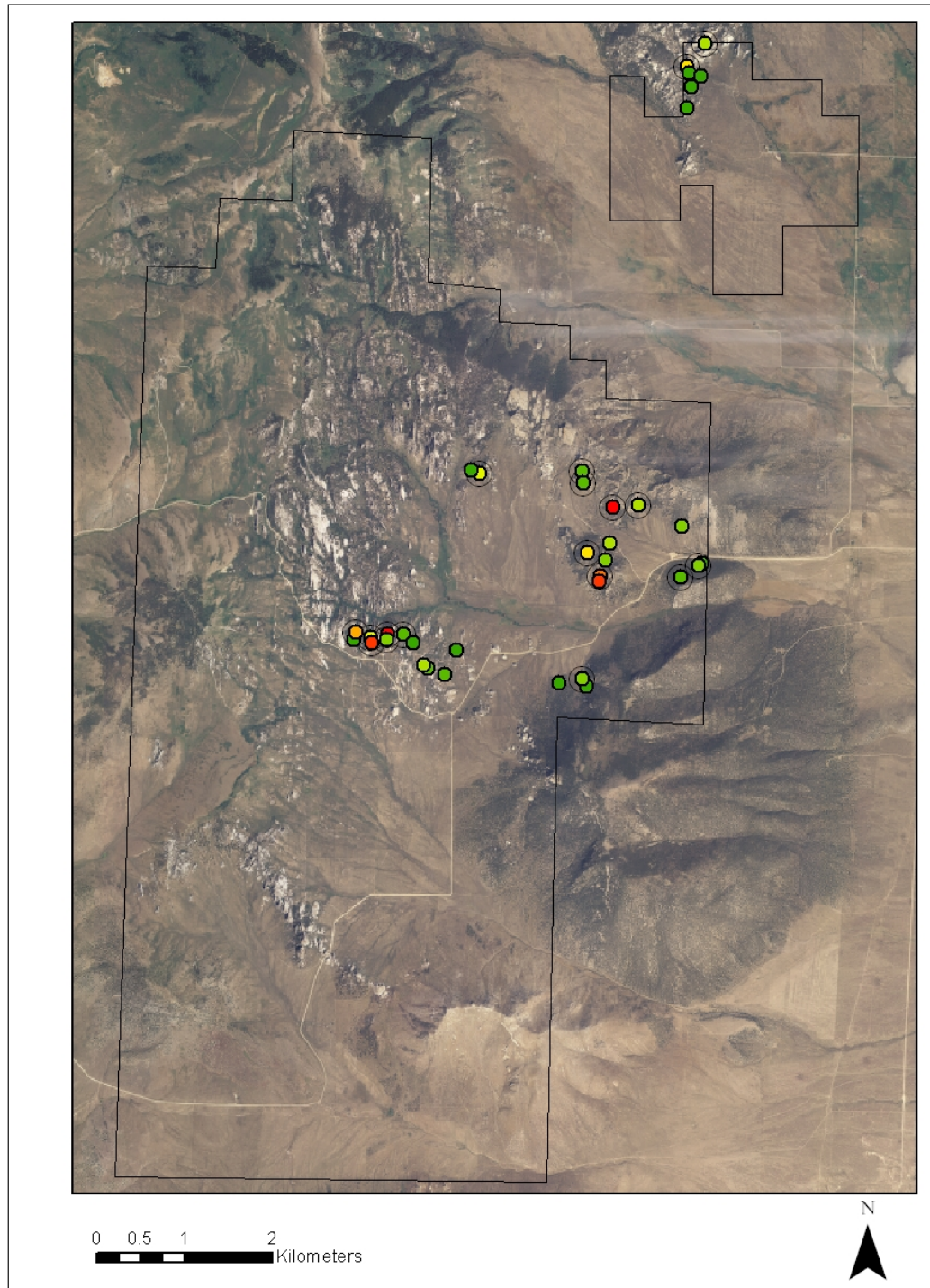


Figure 12. A map of the study area showing 2006 sample sites color coded by estimated ψ values for the cliff chipmunk. “Hot” colors indicate a high probability of occupancy. Sites where cliff chipmunks were actually captured in 2006 are enclosed in a black circle.

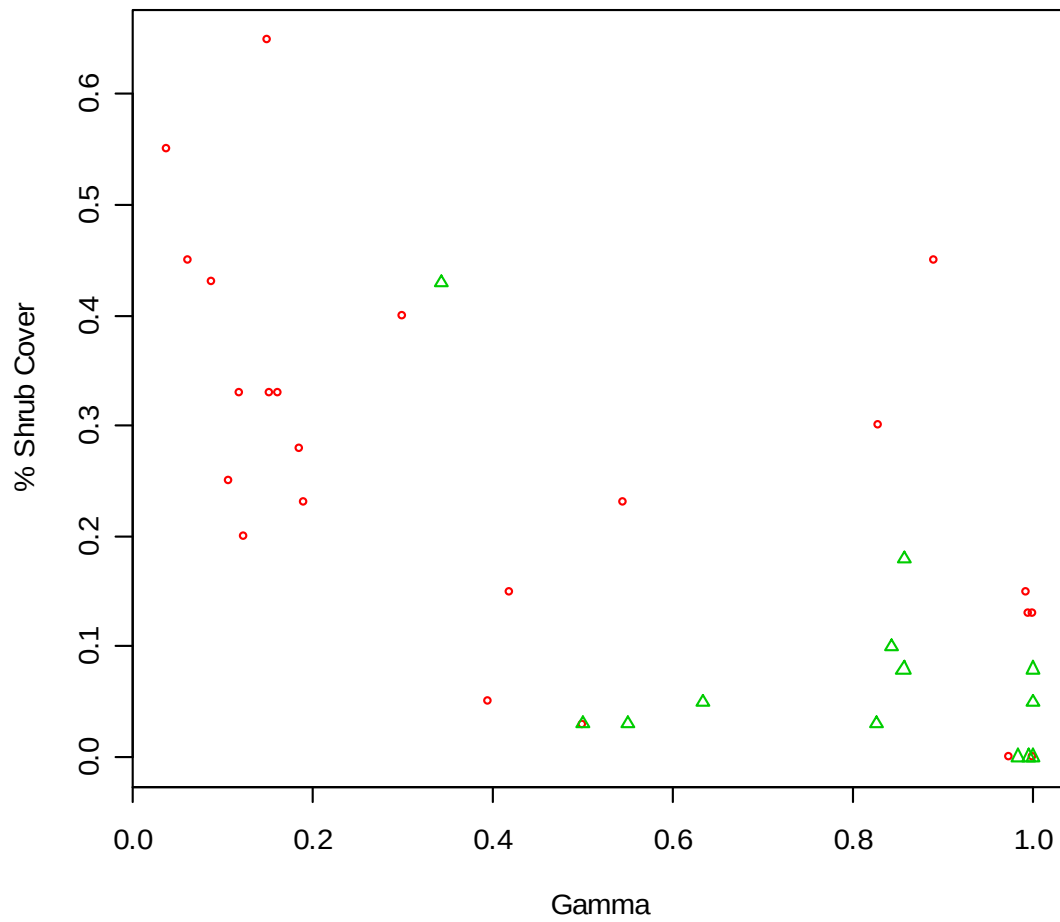


Figure 13. Scatterplot of γ , the probability of occupancy in season 2, by increasing shrub cover for the cliff chipmunk. Sites where cliff chipmunks were actually captured in season 2 are represented by green triangles.

Table 8 presents the suite of ranked candidate models for the pinyon mouse. As with the cliff chipmunk, γ should be interpreted as the probability of occupancy for season 2. Based on ΔAIC_c and w_c , the top model, $\psi(\text{bare}, \text{distedge}), \gamma(\text{shrub}, \text{distedge}), \epsilon=1-\gamma, p()$, has considerable support although the second model, at 2.99 AIC_c values from the top model, has a modest level of empirical support as well. Estimates for ρ remain unchanged for either model at 0.81. Figures 14 and 15 show plots of site-specific estimates for ψ and γ and 95% confidence intervals for these estimates. These intervals are considerably narrower than those obtained for the cliff chipmunk, and standard errors ranged from 0.03 to 0.26 for ψ and from 0.0002 to 0.22 for γ . A map of sites coded by ψ estimates and observed detections (Figure 16) indicates an improvement in model performance over the top model for the cliff chipmunk. Figure 17 shows a scatterplot of sites, coded by observed detections, by ψ and bare ground. Sites with pinyon mouse captures and a high probability of occupancy in season 1 are clearly associated with a high percentage of bare ground.

Table 8. Ranked model set for the pinyon mouse, fit to 2 seasons of detection history data from 2006 sampling in CIRO and CRSP. k represents the number of model parameters. Site occupancy covariates include bare cover (bare), grass cover (grass), shrub cover (shrub), distance to edge (distedge), distance to rock (distrock). Covariates for detectability include night following first capture of the target species (firstcapture), and a two parameter covariate for season-specific detectability.

Model	k	AIC_c	ΔAIC_c	w_c
$\psi(\text{bare}, \text{distedge}), \text{gam}(\text{shrub}, \text{distedge}), \epsilon=1-\text{gam}, p()$	7	131.90	0.00	0.79
$\psi(\text{bare}, \text{shrub}, \text{distedge}), \text{gam}(\text{bare}, \text{shrub}, \text{distedge}), \epsilon=1-\text{gam}, p()$	9	134.89	2.99	0.18
$\psi(\text{bare}, \text{shrub}, \text{distedge}), \text{gam}(\text{bare}, \text{grass}, \text{shrub}, \text{distedge}), \epsilon=1-\text{gam}, p()$	10	138.77	6.87	0.03
$\psi(\text{bare}, \text{grass}, \text{shrub}, \text{distedge}), \text{gam}(\text{bare}, \text{grass}, \text{shrub}, \text{distedge}), \epsilon=1-\text{gam}, p()$	11	142.92	11.02	0.00
$\psi(\text{distedge}), \text{gam}(\text{distedge}), \epsilon=1-\text{gam}, p()$	5	146.35	14.45	0.00
$\psi(.), \text{gam}(.), \epsilon=1-\text{gam}, p()$	3	156.64	24.74	0.00
$\psi(.), \text{gam}(.), \epsilon=1-\text{gam}, p(\text{firstcapture})$	4	159.14	27.24	0.00
$\psi(.), \text{gam}(.), \epsilon=1-\text{gam}, p(\text{season})$	4	159.18	27.28	0.00
$\psi(.), \text{gam}(.), \epsilon=1-\text{gam}, p(\text{season}, \text{firstcapture})$	5	161.84	29.94	0.00

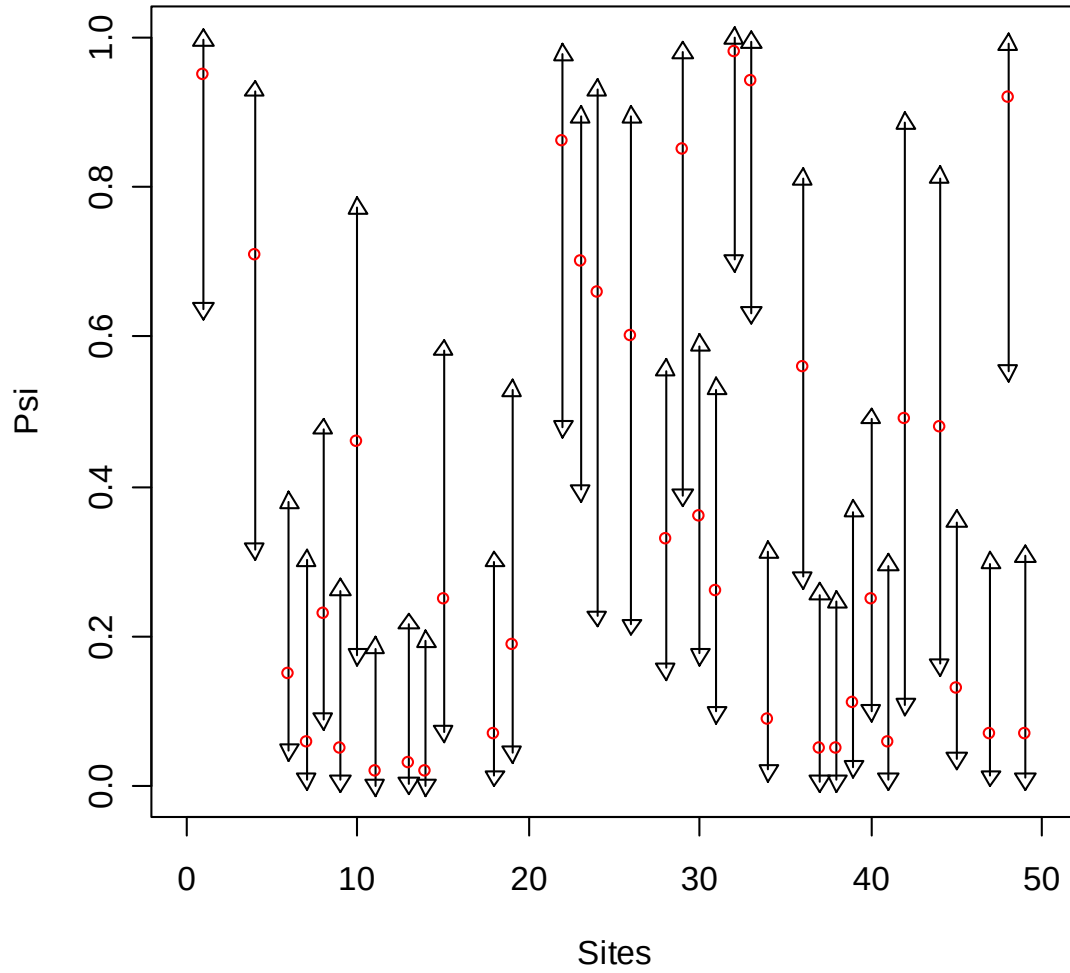


Figure 14. Site estimates of ψ (red circles) and 95% confidence intervals (black lines) and end points (triangles) from the fit of the top model $\psi(\text{bare}, \text{distedge}), \gamma(\text{shrub}, \text{distedge}), \varepsilon = 1 - \gamma, p()$ to the observed 2006 detection histories of the pinyon mouse in CIRO and CRSP.

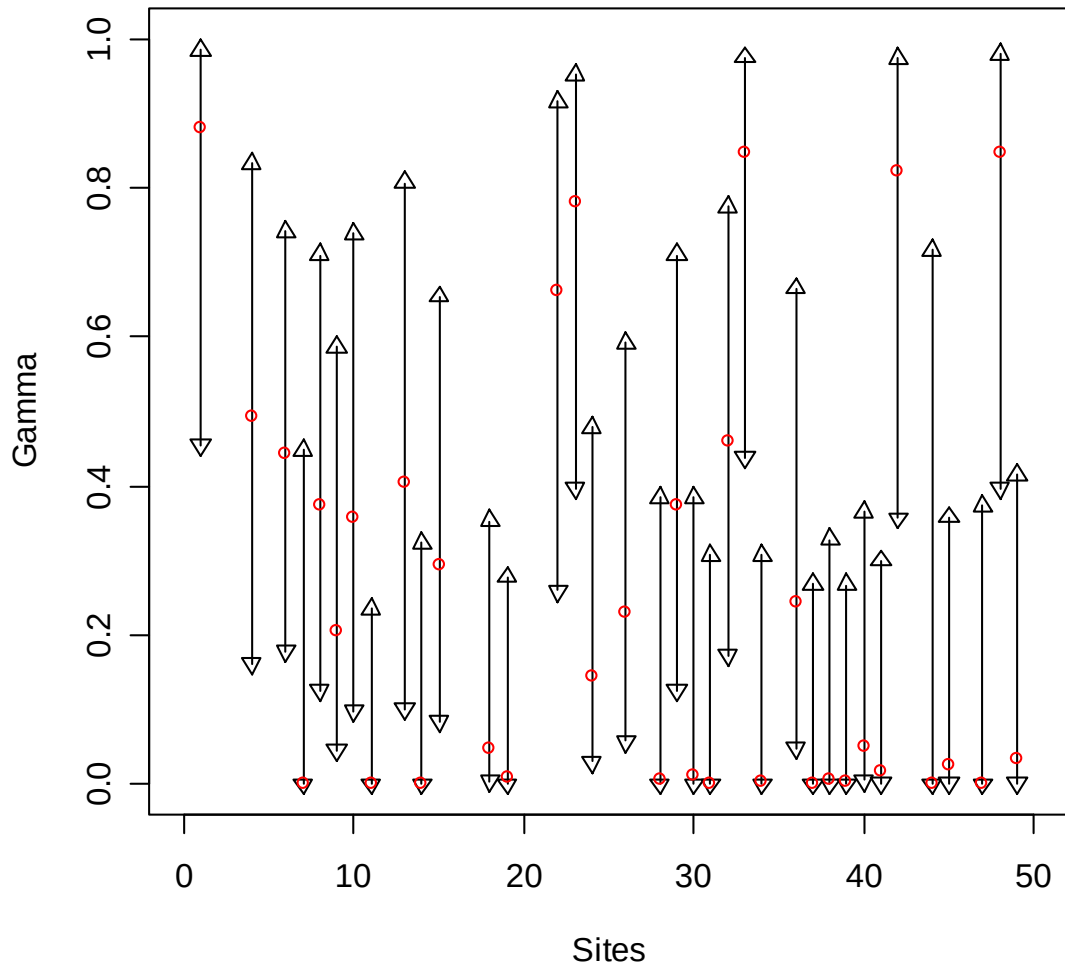


Figure 15. Site estimates of γ (red circles) and 95% confidence intervals (black lines) and end points (triangles) from the fit of the top model $\psi(\text{bare}, \text{distedge}), \gamma(\text{shrub}, \text{distedge}), \varepsilon = 1 - \gamma, p()$ to the observed 2006 detection histories of the pinyon mouse in CIRO and CRSP.

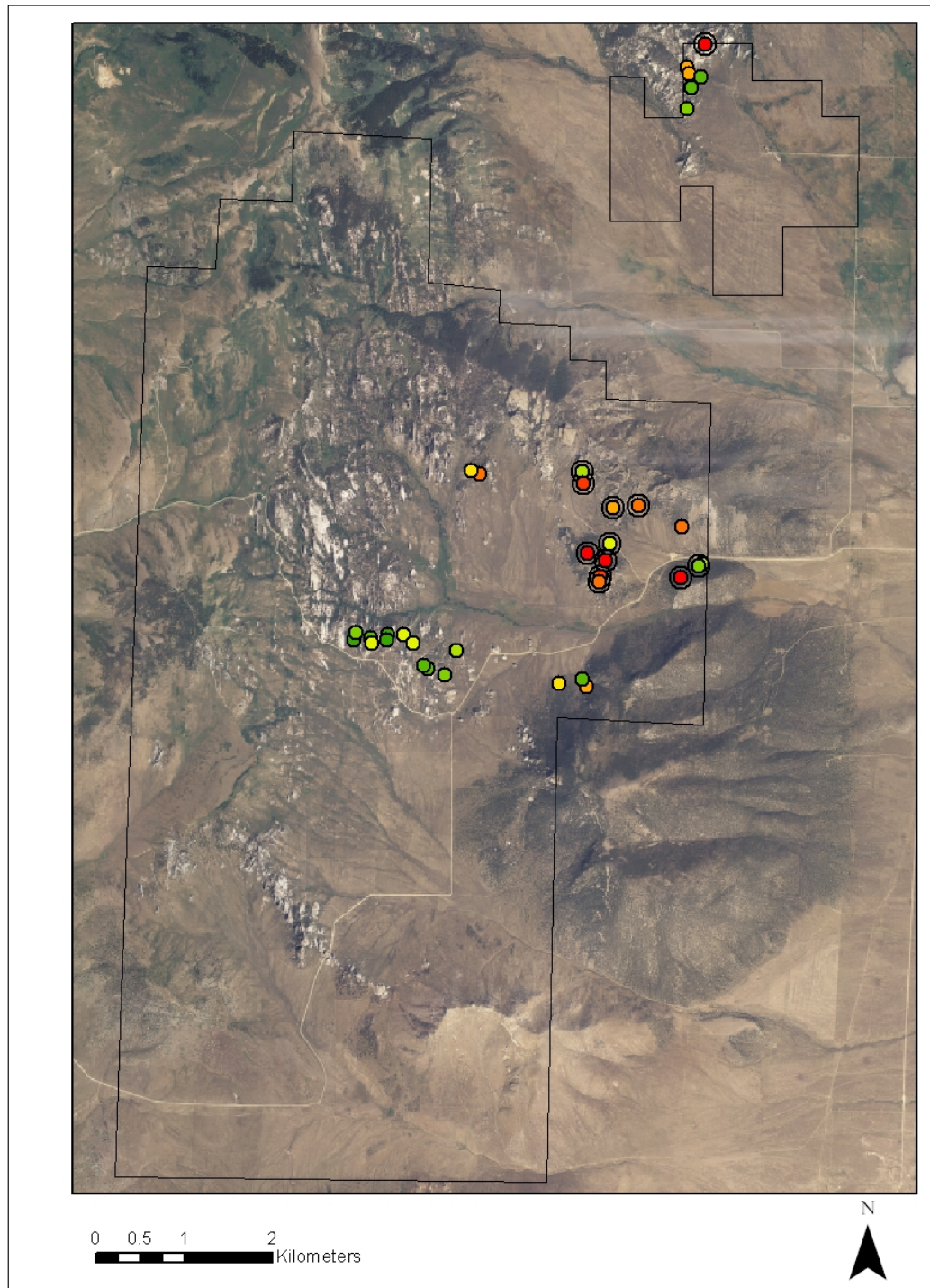


Figure 16. A map of the study area showing 2006 sample sites color coded by estimated ψ values for the pinyon mouse, where “hot” colors indicate a high probability of occupancy. Sites where pinyon mice were actually captured in 2006 are enclosed in a black circle.

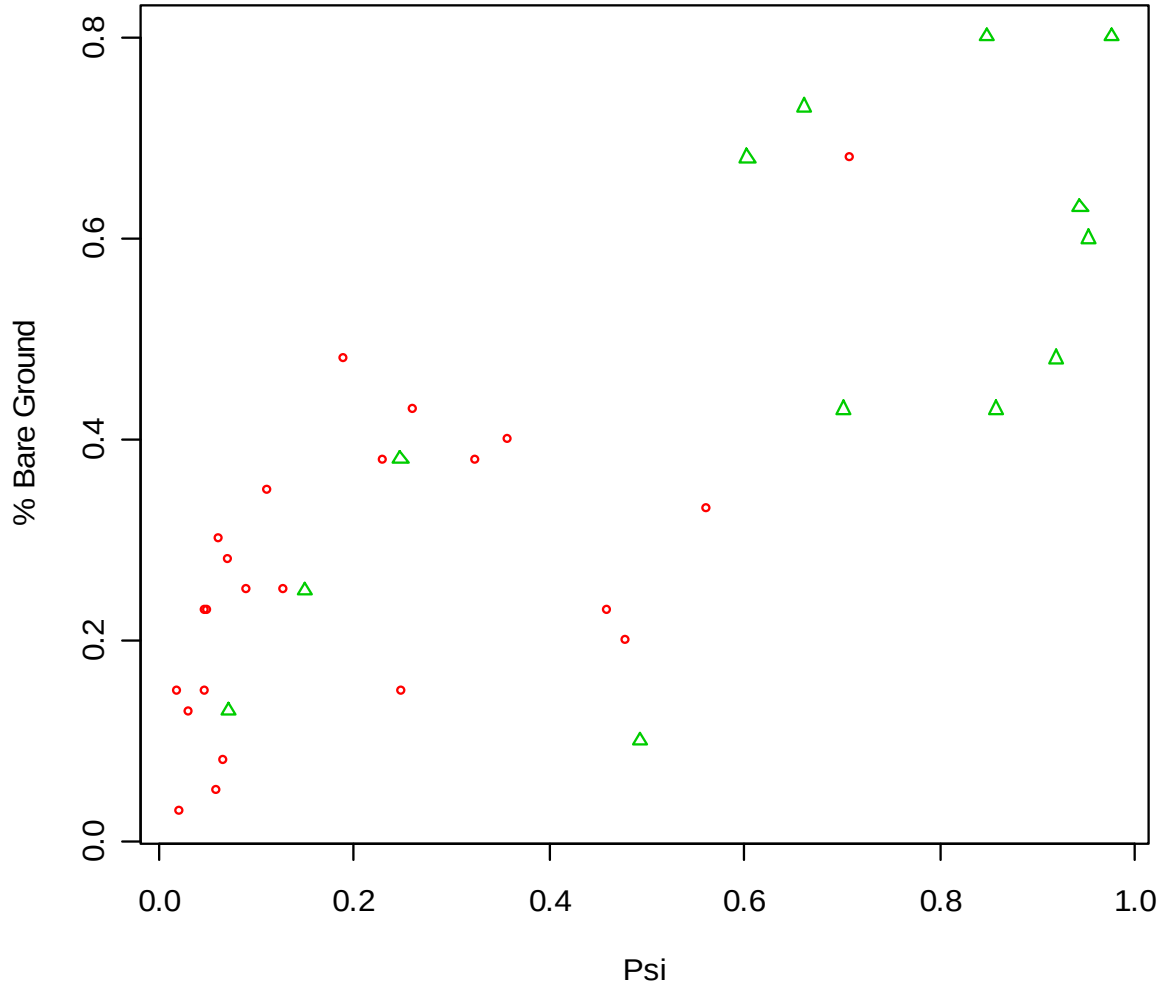


Figure 17. Scatterplot of ψ , the probability of occupancy in season 1, by increasing bare ground cover for the pinyon mouse. Sites where pinyon mice were actually captured in season 1 are represented by green triangles.

As an hypothesis generating exercise, we fit a 2-season model with an alternative parameterization of the form $\psi(.)\epsilon(.)p(.)$ to the 2006 pinyon mouse detection histories and the same suite of habitat covariates as were used for the previously described model selection procedure. The 2 top models were $\psi(\text{bare}, \text{shrub}, \text{distedge})\epsilon(.)p(.)$ and $\psi(\text{bare}, \text{shrub})\epsilon(\text{grass})p(.)$, separated by only 1.75 AIC_c values. The second model provided site estimates of extinction probability (ϵ) as a function of increasing grass and forb cover. Figure 18 presents site estimates of ϵ plotted against grass/forb cover, coded by 2006 observed occupancy results. There is an apparently strong positive relationship between these variables, and the clustering of capture sites along the lower ends of the axes provides additional compelling evidence of this relationship. The fact that four of the top six capture sites with highest extinction probabilities actually experienced “extinction” is also noteworthy.

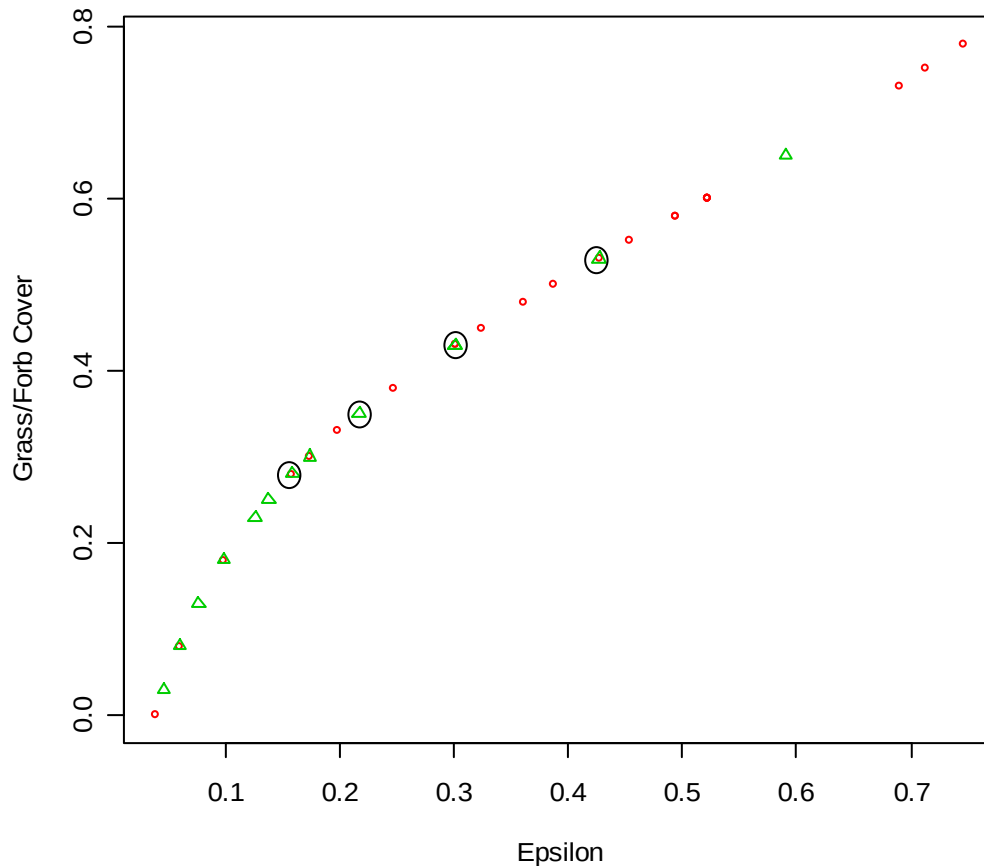


Figure 18. Scatterplot of ϵ , the probability of extinction between seasons 1 and 2, plotted along increasing grass/forb cover for the pinyon mouse. Sites where pinyon mice were actually captured in 2006 are represented by green triangles. The 4 sites where “extinction” occurred (e.g. detected in season 1 but not detected in season 2) are circled in black.

Motion Cameras

In 2005, 16 camera locations were used in 367 survey nights. In 2006, the purchase of the 6 additional cameras and a more concentrated effort resulted in 45 camera locations with 824 survey nights. There were 61 camera locations with a total of 1,191 survey nights for the 2-year study. Some species caught on photos were unidentifiable due to low resolution or poor lighting conditions.

In 2005, cameras were set up near the location where the dead ringtail carcass and possible ringtail tracks were discovered. All 6 cameras were set up in April 2006 at Tahitian Wall near the early April ringtail sighting and were left in the area for just over 1 month. No ringtails were detected on camera during this study. However, 7 cliff chipmunks, another target species, were photographed in 3 separate locations. A number of other species of interest were also captured on motion camera providing useful species presence data for CIRO and CRSP.

A mountain lion (*Puma concolor*) was photographed in 2005, indicating that CIRO continues to provide adequate habitat, at least for migration, for this species. Although no presumptions of habitat use other than occasional presence in the park can be determined by the photo, this

information adds supporting data to the mountain lion survey completed in the CIRO area in the late 80's and early 90's (Laundre et. al 1993). This study determined lion density in CIRO for 1989-1990 to include 1 adult male, a female with 3 kittens, a female with 2 kittens, and an additional individual of unknown sex. Tracks and eyewitness observations of a mountain lion by CIRO staff during the 2005 and 2006 season and the photograph in 2005 conclude that the species continues to use the area.

Bobcat (*Lynx rufus*) photographs in 2005 included an adult in July and 2 kittens in October. The 2 bobcat kitten photos in 2005 were taken 16 days apart using a pass through the Castle Rocks ridge line indicating that this rock pass may be used regularly. An unclear photo of a mink (*Mustela vison*) was confirmed by IDFG staff. CIRO falls in the mink's range but its status in the park is currently listed as unknown on the wildlife checklist due to lack of verification. This photo is the first indication of their existence in CIRO and continuing efforts will be needed to provide more concrete data to confirm their presence. Spotted skunks (*Spilogale putorius*) were commonly photographed in several locations during the study. Another interesting finding is the frequency of cottontails (*Sylvilagus nuttalli*) captured on top of rock features. The cameras were also sensitive enough to capture an admiral butterfly (*Limenitis weidemeyerii*). Table 9 gives a complete list of the species captured on the motion cameras for the 2005 season.

Table 9. Animals detected using motion cameras in 2005.

Species	Number of detections
<i>Mammals</i>	
Bobcat	3
Cliff chipmunk	1
Cottontail	2
Cougar	1
Golden-mantled ground squirrel	1
Mink	1
Mule deer	1
Rabbit	1
Spotted skunk	3
Unknown chipmunk	1
Woodrat	1
<i>Birds</i>	
Dark-eyed junco	1
Northern flicker	1
Robin	1
Unknown bird	1
<i>Other</i>	
Admiral butterfly	1
Unknown animal	2

In 2006, 1 adult and 1 kitten bobcat were detected in the Castle Rocks Interagency Area. This finding along with those from the 2005 survey indicates that the reserve has sufficient habitat to support bobcat reproduction. In May 2006, a red fox (*Vulpes vulpes*) was observed by one of us (R. Hirnyck) in Indian Grove in CIRO during field operations. Red foxes are considered probably present but have never been confirmed in the park. In late September 2006, red fox

urine was used to bait a camera in that area and 2 photos of a red fox were obtained on the motion camera, confirming its presence in the area.

Thirty bighorn sheep (*Ovis canadensis*) were released by IDFG in 2004 in the US Forest Service land surrounding CIRO. Over the next 2 years all collared bighorn sheep either moved to the Jim Sage Mountains or were found dead due to predation by mountain lion (M. Todd, IDFG, personal communication). It was thought that the reintroduction was unsuccessful until a bighorn sheep was photographed by one of our cameras in July 2006. Several bighorn sheep have been observed in the area since then and their feces have been documented numerous times, resulting in a designation change from unknown to present on the CIRO and CRSP mammal species list. Table 10 gives a complete list of the species photographed on the motion cameras for the 2006 season.

Table 10. Animals detected using motion cameras in 2006.

Species	Number of detections
<i>Mammals</i>	
Bighorn sheep	1
Bobcat	4
Cliff chipmunk	6
Cottontail	1
Golden-mantled ground squirrel	11
Mule deer	35
Red fox	2
Unknown bat	1
Unknown chipmunk	6
Unknown rodent	21
Unknown squirrel	9
Weasel	3
Woodrat	2
Yellow-bellied marmot	7
<i>Birds</i>	
Green-tailed towhee	1
Northern flicker	6
Robin	38
Unknown bird	8
<i>Other</i>	
Unknown animal	6

Rocky outcrops were targeted extensively in the camera study due to the ringtail association with rocky habitat documented in the literature. The vegetation most commonly associated with rock outcrops is open woodlands and shrubland dominated by a combination of pinyon pine, Utah juniper, and mountain mahogany. Most sites had a mixture of these species with varying height classes and appeared to be dominated by pinyon pine.

Discussion

Small Mammal Trapping

This study provides the first vouchered confirmation of the presence of pinyon mice in Cassia County, and we believe this to be the only location outside of the area in Owyhee County reported by Gafur et al. (1980) and Hammond and Yensen (1982) where pinyon mice are known to occur in the state (Carraway and Verts 2002). Capture results varied widely between 2005 and 2006. Overall capture success was similar and moderate for each of the three seasons, indicating that traps were not “swamped” and many traps remained available to catch new individuals and species throughout each survey. However, the species composition and the proportion of captures represented by dominant species were very different between 2005 and 2006 and relatively similar within 2006. Trap effort was increased three-fold in 2006, and this certainly accounts for some of the differences between years. However, given the inherently cyclic nature of rodent populations in general, and among *Peromyscus* in particular, it is also likely that the trapping session conducted in 2005 coincided with a low point in the population cycles of many of these species (Krebs and Myers 1974). Given the greater breadth of habitat types sampled in 2005, the species composition of capture results was surprisingly low. Also of interest is the apparent synchrony in the population cycles of several rodent species over the course of the study, which as a group may have been out of phase with the local deer mouse population. The number of captures per trap night for deer mice decreased nearly 60% between 2005 and season 2 of 2006 but increased 6 fold for the Great Basin pocket mouse and 10 fold for the cliff chipmunk. Although no information was obtained for the pinyon mouse in 2005, an apparently large upswing in abundance also occurred between fall 2005 and fall 2006 for this species as well. Synchronous population cycling has been reported within rodent communities and mechanisms driving these cycles appear varied and difficult to discern, but can include episodic masting events in locally important food plants and weather patterns, both of which are plausible explanations for the rodent community cycles in the CIRO/CRSP area (Brown and Heske 1990, Brady and Slade 2004, Elias et al. 2004). Given the narrow temporal window of this study, the appearance of synchrony among species other than the deer mouse could also simply be coincidental, and evidence from available literature also suggests that long-term asynchrony can also occur within rodent communities as a result of inherent stochasticity (Brown and Heske 1990). Additional study into the population cycling of the rodent community at CIRO/CRSP is warranted. Pinyon pine masting and the dynamics associated with environmental change for populations at the northern periphery of their range make this avenue of inquiry particularly important. Both pinyon mice and cliff chipmunks reportedly rely heavily on tree seeds, and the annual variability in pinyon pine and juniper cone crops in the CIRO/CRSP area could be a particularly important driver of population cycles for these species (Smartt 1978, Hart 1992, Hall and Morrison 1997, VanderWall 1997).

Detectability of target species and the estimation of p should be influenced by local abundance as well. We investigated this by fitting models with season-specific parameters for p but these consistently ranked at least 2 AIC values lower than a null model with a single parameter for p . The inclusion of parameters to account for the potential influence of trap happy or trap shy individuals on detectability was also of negligible effect. Estimates of p did consistently increase for both species after first capture but only by very small amounts ($\leq 2\%$). In the interest of parsimony, particularly given the challenges of fitting models with many parameters to small

sample datasets, we did not keep these sampling covariates in subsequent candidate models. However, given their biological relevance and the apparently low cost of their retention (e.g. within 2-3 AICc values, similar variance), one could conclude differently. Estimates of ρ were relatively high and very consistent across candidate models for both species, providing an additional measure of confidence in these estimates. Because ρ was so high, the gain in information over the simple proportion of sites with captures (naïve ψ) for an overall estimate of the proportion of area occupied was relatively small. Without the inclusion of site-specific habitat variables, ψ seems a rather uninformative number in this case.

Although uncertainty and imprecision in site estimates from top models was quite high, model selection from the full suite of candidate models, as well as supporting information from graphical tools and *t*-tests, consistently identified the relative importance of habitat characteristics to site occupancy for both the cliff chipmunk and the pinyon mouse. Neither species were widely distributed throughout the study area. Examination of maps and scatterplots reveal that both species occur in relatively small patches of what may be considered “core” habitat. The lack of support for null models indicates that site occupancy was not distributed randomly across the study area. Both species appear to respond similarly, in terms of occupancy, to shrub cover and distance to the pinyon-juniper woodland edge. Best models support the conclusion that sites with lower shrub cover and at greater distances to woodland edge are more likely to be occupied. Grass and forb cover was lower in occupied sites, although the contribution of this information to models was modest. For the cliff chipmunk, the presence of rock cover was also very important, and for the pinyon mouse, open bare ground and litter was of additional importance. Tree cover was not as significant a factor as might have been expected given the preponderance of literature that highlights the association between these species with pinyon and juniper trees. We suspect that the lack of “effect” of juniper on occupancy was an artifact of the 2006 sampling frame, which targeted only areas classified as pinyon-juniper woodland on a park vegetation map. This essentially enabled our study to focus on additional habitat features within the pinyon-juniper woodland, particularly those microhabitat features underneath the woodland canopy. And while the general association between these two target species and pinyon-juniper and rock is well established in the literature, the additional information on woodland understory characteristics provided by our study is new and we are aware of only one other study from Wyoming that addressed some of the same microhabitat features as our study (Rompola and Anderson 2004). Rompola and Anderson (2004) reported somewhat inconclusive and contradictory results. They found that cliff chipmunks were captured in locations with lower grass cover but also lower bare ground cover, and with taller shrubs. They found that pinyon mice were captured in sites with greater canopy cover, lower forb cover, and greater distances to woodland edge. However, they also reported a negative association between pinyon mouse capture sites and proximity to rock, which contradicts much of the available literature on this species (e.g. Verts and Carraway 1998). Pinyon mice capture sites in our study were generally found in areas with greater rock cover and in closer proximity to rock features, although this difference was not statistically significant. The design of their study differed from ours, however, in that they compared habitat characteristics between sites with captures and randomly selected sites where no trapping was conducted. They did not account for detectability and the possibility that random sites may have also been occupied by target species. We consider our approach, following a framework developed and described by MacKenzie et al.

(2006), to be a more robust and direct approach for drawing conclusions about the habitat associations of these rodents.

Through analysis of these data we have identified an additional line of inquiry for future studies concerning the possibility of seasonal contraction and expansion of the proportion of occupied sites. We hypothesize that the reduction in site occupancy by the pinyon mouse in season 2 may be explained by the presence of “suboptimal” habitat features. Understory vegetation cover was greater in sites unoccupied in season 2 but previously occupied in season 1. Exploratory model selection with the alternative parameterization for explicit dynamics specifically indicated that increasing grass/forb cover was associated with elevated extinction probabilities (ϵ). A phenomenon of seasonal occupancy contraction could be present in the pinyon mouse population of CIRO/CRSP. Changes in abundance (and therefore detectability and/or occupancy) in peripheral sites could be caused by dispersal of juveniles or males. Male pinyon mice reportedly have larger home ranges and may utilize habitat differently than females (Scheibe and O’Farrell 1995, Hall and Morrison 1997, Ribble and Stanley 1998). A “source-sink” dynamic could also be involved, in which mortality rates are higher in peripheral sites. Abundance is a potentially confounding and somewhat complicated issue related to interpretation of occupancy estimates. It may well be that peripheral sites not only experience higher rates of extinction but also present a lower probability of detection, which could cause between-season dynamics to be overestimated. Additional sampling information will be required to address this issue in the modeling process. Habitat covariates could be included in models of detectability, and direct inclusion of site capture count data as an indication of abundance may also be possible (MacKenzie et al. 2006). We observed an opposite trajectory in occupancy for the cliff chipmunk. No strong hypothesis to account for this pattern is immediately apparent. Because detectability of the chipmunk was considerably lower than that of the pinyon mouse (by 30%) and uncertainty in estimates is correspondingly higher, we are reluctant to speculate on the possible processes that might be driving this pattern (if the pattern truly exists at all).

Motion Cameras

Although no ringtails were photographed during the study, photographs obtained for other species will be useful. We confirmed the presence of several species previously unconfirmed for the two parks. Evidence of use and breeding in or near the park by previously confirmed species but for which little information beyond simple presence was known is also very informative to park staff. This is particularly true for the cats and bighorn sheep.

Motion cameras can provide a fairly affordable, time efficient, and non-intrusive method of sampling for species presence throughout the park. Although the motion cameras did not capture the targeted ringtail, the cliff chipmunk, another target species in the survey, was captured at 7 different locations. Cameras are able to indiscriminately capture a diversity of species in their natural habitat in a non-invasive manner while providing interesting and useful information to resource managers on species presence and distribution in the park.

Management Implications and Future Recommendations

Results from small mammal trapping provide compelling evidence of a qualitative difference between occupied and unoccupied sites for both the cliff chipmunk and pinyon mouse. The

discovery that both species are more likely to occupy (or use) interior portions of rocky pinyon-juniper woodland with open understory is significant. We suspect that a generalization could be made from this, given the consistency within existing literature, and that other facultative or obligate pinyon-juniper woodland vertebrate species of conservation concern such as the pinyon jay (*Gymnorhinus cyanocephalus*), Virginia's warbler (*Vermivora virginiae*), and juniper titmouse (*Baeolophus ridgwayi*), may also be served well by maintenance of this type of habitat. Additional research on this topic is warranted, particularly with regard to understory microhabitat characteristics. Future research should focus on gaining added precision and verification of results provided by studies such as ours, with the goal to be able to provide resource managers enough information to confidently develop management prescriptions for pinyon-juniper woodlands. For example, mechanical removal of understory vegetation, prescribed burning, and protection of contiguous stands from fragmentation (e.g. direct roads around rather than through large stands) are options that might meet an objective to maintain the kind of habitat described by this study. And while the perception of pinyon-juniper woodland expansion into shrub-steppe and other habitats has prompted some resource managers to try and eliminate this habitat in some areas, the dependence of a unique vertebrate fauna on this woodland habitat should reinforce the importance of protecting this habitat in other areas.

The reported sighting of a ringtail by a knowledgeable park visitor in April 2006 leads us to believe that the species probably does occur in the area, and that it is only a matter of time and persistence until a definitive encounter is made. We recommend that some level of effort continue to be made to provide documentation. Some reconsideration of lures and bait, camera equipment, trigger settings, and camera positioning is recommended. The use of track plates may be helpful, and once a reliable method of detection for ringtails in the area can be identified, it may become possible to establish a quantitative baseline measure of occupancy against which future changes can be detected in a manner similar to what was done for the rodents included in this study.

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