

Small Mammal Communities and Responses to Forest Management at Mount Rushmore National Memorial, 2010-2012



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Cover Photo: Volunteer T. Juntti processes a live-captured *Peromyscus* during 2010 field sampling. Photo by R. Gitzen.

INTRODUCTION

Old-growth ponderosa pine (*Pinus ponderosa*) forests are a highly significant resource at Mount Rushmore National Memorial (MORU), making up 66% of the Memorial's area (Symstad and Bynum 2006). This is an important resource regionally: the Memorial supports the second largest area of old-growth left in the Black Hills. Throughout the Black Hills, logging has removed this old-growth component across most of the landscape, while fire suppression during the last century has led to increased tree density and potentially higher risk of large-scale high severity fires and epidemics of mountain pine beetles (Brown and Cook 2006).

Prior to European settlement, forests of the Mount Rushmore area were shaped by frequent low-severity fires, less frequent mixed-severity fires that included patches of low, medium, and high-severity burns, and rare stand-replacement fires (Brown and Sieg 1996, Shinneman and Baker 1997, Brown and Cook 2006, Brown 2006). Due to the high resistance of large ponderosa pine to low or moderate-intensity fires (Agee 1993), natural fires in these forests typically would maintain rather than remove old-growth stands. Like most of the Black Hills, MORU has experienced more than a century of fire suppression with the result of increased surface fuel loadings, tree densities, and vertical fuel continuity (Brown and Cook 2006). As a result, catastrophic fire and mountain pine beetle epidemics became a major threat to the memorial's old-growth forest and to forest wildlife species dependent on large trees and snags within closed-canopy forests (Symstad and Bynum 2006).

In 2002, MORU developed a Fire Management Plan to address strategies for fuel modification and hazardous fuel reduction in an attempt to restore pre-settlement conditions and reduce the risk of memorial-wide high-severity fires. The primary fuel treatment involves thinning small diameter trees, cutting and piling of branches and boles into wood "teepees", and

burning the slash piles in a subsequent year. When conditions permit, more extensive prescribed burns may be conducted. Selected areas of the park were thinned to reduce fuel loads prior to 2010. By 2010, a regional mountain pine beetle epidemic on the surrounding landscape started to kill individual and patches of old growth trees in the park, and MORU prioritized immediate density reductions across a large portion of the park to reduce conditions conducive to spread of the epidemic and reduce fuel loads. One point of criticism based by the park as it implemented immediate large-scale thinnings was the lack of information for understanding how density-reduction alternatives might affect many aspects of the park's ecosystem.

Other than limited inventories (Stebler 1939, Turner 1974, Schmidt et al. 2004), little information has available about the habitat associations and distribution of small mammals in forests of the interior Black Hills, or their response to forest manipulations and natural disturbances in this region. At MORU, the large trees, snags, and logs in old-growth forests likely provide valuable habitat for southern red-backed voles (*Clethrionomys gapperi*), red squirrels (*Tamiasciurus hudsonicus*), and northern flying squirrels (*Glaucomys sabrinus*). Several species of high management concern in the Black Hills and at MORU are partially dependent on this rodent prey base, including the northern goshawk (*Accipiter gentilis*; Kennedy 2003) and American marten (*Martes americana*; Buskirk and Ruggiero 1994). The high abundance and species richness of the small mammal community gives them other critical roles in the MORU ecosystem. By sheer numbers, small mammals play a major role in the forest ecosystem as prey items (e.g. Soutiere 1979), herbivores (Golley et al. 1975), predators on invertebrates, consumers and dispersers of seeds and fungal spores (Cazares et al. 1999), and potential nest predators (e.g., Sieving and Willson 1998). Common species such as the deer mouse (*Peromyscus maniculatus*) may respond to and affect the success of other management

actions in burned areas, such as biocontrol of invasive plants (Ortega et al. 2004). Moreover, high populations of deer mice or white-footed mice (*P. leucopus*) may increase the risk of hantavirus to park visitors and staff (Mills et al. 1999)

In general, small mammals show strong responses to changes in forest structure, including components such as the amount and distribution of large logs, litter depth, and cover of ground-level vegetation, shrubs, and trees (Lovejoy 1975, Martell 1984, Monthey and Soutiere 1985, Nordyke and Buskirk 1991). Changes in these components affect availability of food sources such as fungi, conifer seeds, berries, and invertebrates; alter the ground-level microclimate to which some mature-forest species are sensitive; change structural components used directly for shelter or burrows (e.g., logs, litter layer, arboreal nest sites); and lead to changes in competitive interactions. The small mammal community at MORU contains species (red-backed vole, red squirrel, northern flying squirrel) that might be expected to respond negatively to disturbances. However, in many regions, such species demonstrate some degree of flexibility and tolerance of intermediate levels of disturbance such as that produced by fuels treatments at MORU (e.g., Gitzen and West 2001, Gitzen et al. 2007).

To provide MORU managers with information for consideration in future decision making about the forest resources of the Memorial, we conducted a park-wide study of the small mammal community in 2010 and 2012. Our objectives were: 1) To assess baseline species occurrence of small mammals during late-spring to early summer at the scale of 0.36 ha sampling units; 2) To examine changes in species occurrence between the time before the large-scale park-wide density reduction treatment in 2010 and the second growing season after these treatments (2012). 3) To assess whether these changes varied between sites based on the 2010 treatment action received.

METHODS

Site Selection

To obtain a spatially balanced sample of sites across MORU, we used the Generalized Random Tessellation-Stratified (GRTS) sample of sites selected for long-term vegetation monitoring by the Northern Great Plains Inventory and Monitoring Program (NGPN; Symstad et al. 2011). In this sample draw, 256 sites (Fig. 1) were drawn from a sample frame comprised of approximately 54 x 54 m grid cells covering MORU excluding developed areas, the restricted zones, and selected cliffs that could be reliably identified from spatial data layers. We used a subset of these sites for livetrapping sampling of small mammal species occurrence.

To select sites for small mammal live trapping, we excluded the first 20 sites from the GTRTS sample draw (Appendix 1) as these were either being used for intensive long-term vegetation monitoring by the NGPN or had been excluded by that program as unsampleable due to terrain. We then evaluated sites on the list as trapping commenced, selecting sites for trapping based on their order in the GRTS sample draw, adjacency to other selected sites, terrain, and available field time. We considered sites 21-123 in the GRTS draw for livetrapping. Sites within this range were not selected for livetrapping if a) a trap grid could not be safely installed and checked due to safety issues; b) portions of the trap grid would overlap roads or developed areas; c) the trap grid edge of one site would come within approximately 50 m of the trap grid edge of an adjacent site; or d) available effort for the season ran out before the site could be sampled. The default location of the trap grid (see below) within each site was to have the center grid point overlapping with the GRTS sample point listed in Appendix 1. However, to minimize the number of sites excluded from the sample draw, we allowed grid centers to be shifted up to 30 m

from the UTM center point if that situate the grid on sampleable terrain or shift two adjacent grids far enough apart to both be sampled. In addition, as noted below, our individual trap grids were larger (0.36 ha) than the size of grid cells used in the GRTS sample frame (approximately 0.29 ha). Thus, our use of the GRTS sample-draw sites for the livetrapping study should not be seen as providing an absolute equal probability sample, but as a mechanism for obtaining a well-dispersed sample of randomly selected sites, one linked to a long-term sample draw to facilitate repeated sampling of the small mammal community over time. Based on this approach, we selected and live trapped 68 sites in 2010 and 77 sites in 2012, respectively; the 2012 effort included the 68 2010 sites plus 9 additional sites.

Field sampling

Small mammal live trapping

On each site selected for live trapping, we established a 5 x 5 grid of trap stations, with 15 m between stations, producing 0.36 ha grids. This scale was selected as a scale at which site-level species occupancy for common small mammals could be a sensitive metric of responses to habitat condition. Trap spacing was modified in some cases when access to a gridline was restricted by cliffs or other rocky terrain when such modifications did not significantly change the total size of the area effectively covered by the grid. At each trap station, we placed a single large folding or non-folding Sherman trap on the ground (Fig. 2). Traps were baited with a mix of rolled oats and peanut butter and provided with polyester batting for insulation. Bark, needles, and debris were placed on top of the trap to stabilize it and provide further shelter.

Mount Rushmore NMEM Small Mammal Study

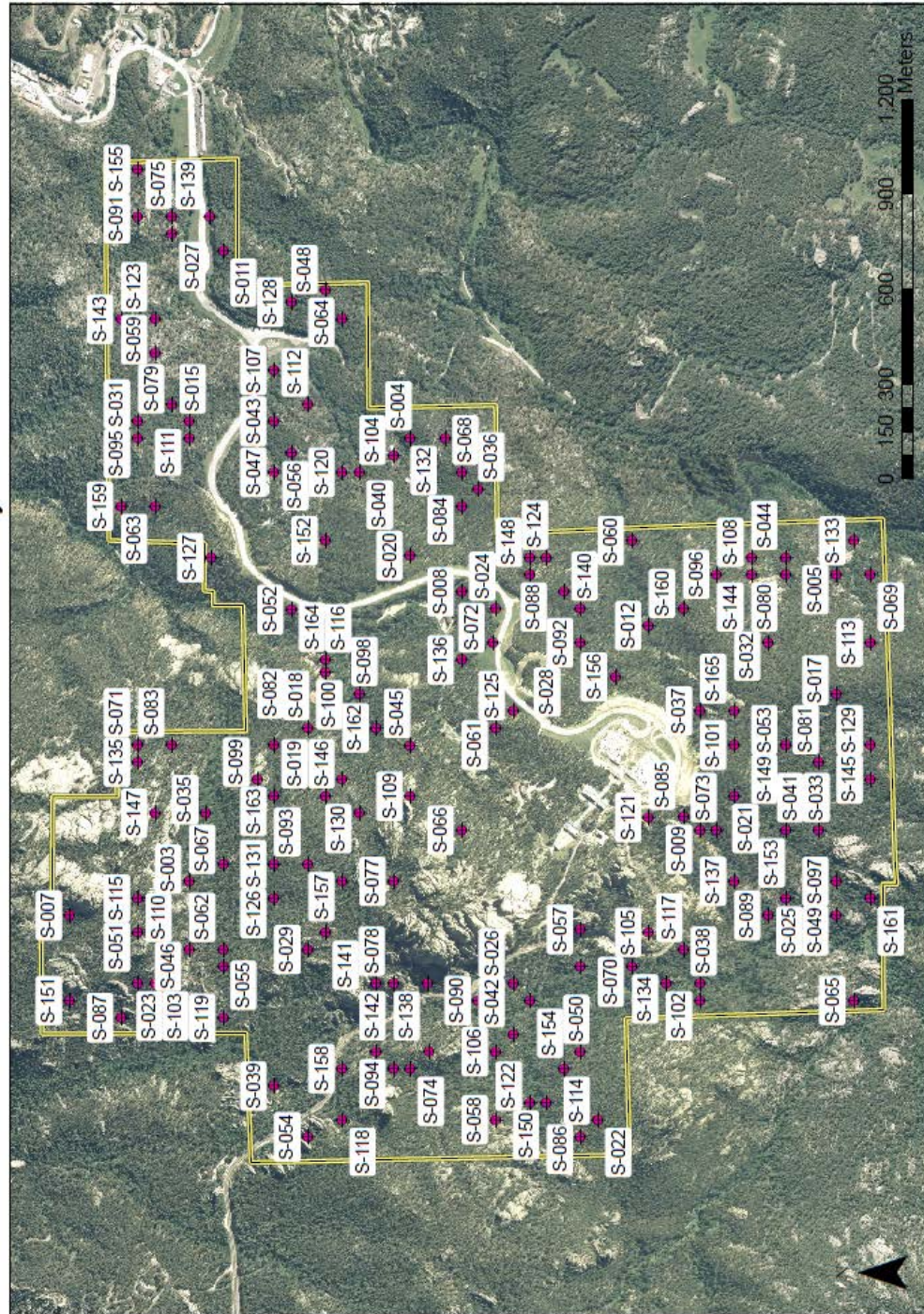


Figure 1. Pool of potential sample sites used for small mammal sampling in this study; sites are from the Northern Great Plains Inventory and Monitoring Network long-term vegetation probability sample for Mount Rushmore National Memorial.



Figure 2. Sherman live trap with side panel opened to permit release of a captured bushy-tailed woodrat at Mount Rushmore during livetrapping sampling.

Each site was sampled for one trap session per year, during late May – July 2010 and/or 2012. Trap sessions consisted of 3- or 4-night trapping intervals: traps were opened on Day 0, then left open and checked once per day in the morning or early afternoon on Trap Days 1, 2, and 3 (and 4 for 4 nights sessions). During initial field work in 2010, 32 sites were sampled for 4 trap nights before we switched to 3 night trap sessions for the remainder of 2010 and all but one site in 2012. In 2012, 4 sites were trapped for a 4th night after heavy hail sprung most traps on the first trap night, while data for only 2 trap nights was obtained for 2 additional sites at the end of the season.

At each trap check, all traps were re-baited and re-set as needed. Captured animals were identified to species (see Appendix 4); sex, mass, and notes on reproductive activity were recorded. For mice, shrews, and voles, we measured tail, ear, and hind foot lengths as an aid to species identification. *Peromyscus* and chipmunks were individually marked with a small numbered eartag; other species were marked with a black marker to aid in identification of recaptures within a trap session. Except for trap mortalities, animals were released after handling at the point of capture.

Data Analysis

Species differentiation based on morphological characteristics is imperfect for *P. maniculatus* vs. *P. leucopus* (Appendix 4), a factor which should be considered when evaluating our relative capture rates of the two species. We provide tallies based on morphological identification of nearly all *Peromyscus* except for a small number in 2010, which were classified as *Peromyscus* spp. due to uncertainty or escape by the individual before measurements. Species differentiation

based on morphology is even more problematic for *Sorex haydeni* vs. *S. cinereus*, and capture totals combine these species into a single group.

Several aspects of the observed data and our sampling protocol shaped the analyses we focus on in this report. We report captures by species or species group, but for interpreting patterns focus on species occurrence at the scale of individual trap grids. The trap grid size in this study was scaled to measure fine-scale habitat occupancy (presence / absence in individual patches), rather than abundance via mark-recapture modeling. The three most common species occurred at nearly all sites in 2010, precluding meaningful examination of species occurrence in relation to pre-2010 forest management activities. However, these changes showed major changes in occurrence between 2010 and 2012, and the most relevant question was whether species showed differential changes in occurrence as a result of 2010 treatment impacts. In addition, a second pattern of regional interest was the potential distribution of meadow jumping mice and other species in relation to hardwood-draw occurrence. Therefore, we focused on examining species changes as a function of 2010 treatment categorization and on examining species occurrence in areas with and without hardwood draws. We classified, based on field examination, each trap grid as to whether it was predominately thinned or not thinned in 2010, whether down woody material had been chipped on the majority of the grid in 2010 or previously and on whether it contained significant hardwood or spruce mesic-draw conditions.

With this analysis, we focus on patch-level occurrence and between-year changes of occurrence for thinned and unthinned patches. Given the scale of treatments at MORU in 2010, it could be argued that declines observed could represent a landscape-scale reduction due to a landscape-scale management action, one causing declines in occurrence both in thinned and untreated portions of the landscape. While we cannot rule out this possibility given the lack of

simultaneous data from control landscapes in the Black Hills, we view it as unlikely. We note that most species we studied have small home ranges and are able to persist even in relatively small forest fragments after severe forest disturbances in some cases (e.g. Gitzen et al. 2007). Although problematic from an experimental design framework, the MORU thinning treatments effectively served as a large-scale natural experiment with major areas of the park grouped into either thinned or unthinned conditions: i.e., areas not treated in 2010 were not small fragments, but significant blocks of the park. In addition, large scale year to year changes due to factors such as those present in the winter of 2011-2010 (dry winter with little snow cover) can easily cause year to year variation in small mammal abundance of the magnitude observed here. Thus, we expect that any signature of the 2010 treatments in causing the decline in occurrence we observed would be measurable as a heterogeneous pattern between thinned and unthinned sites.

A potential source of bias in comparisons among habitats based on raw detections of species is that each trap grid does not perfectly sample the species present at that patch: there are potential false absences due to species being present but never captured during the 3-4 day trap session. To estimate detectability and examine changes of the 3 most frequently detected species (least chipmunk, deer mouse, southern red-backed vole), while accounting for incomplete and variable detectability, we fit a two season Bayesian hierarchical occupancy model separately for each species in WinBUGS, following Kéry and Schaub (2012). In this analysis, the data consists of the site-level detection histories (1 if detected, 0 if not) by trap night and year for the 77 sites. Our model for each species incorporated the following parameters and structure (see also MacKenzie et al. 2006):

- If the species is present at the site in that year, it is detected with probability p . For the independent analyses of each species, $\text{logit } p$ was modeled as a function of an intercept,

fixed-effect adjustments for trap nights 2 and for trapnights 3 and 4, a fixed effect adjustment for 2012 sampling, and a year-specific random site effect. Thus, we incorporated a systematic change in detectability from 2010 to 2012, temporal variation among the 3 or 4 trap nights (nights 3 and 4 assumed to share a common effect) with these nightly effects assumed constant across sites and times, and year-specific heterogeneity among sites.

Note that in the occupancy model estimation, mixing site x year combinations that were sampled for 3 trap nights with those sampled for 4 trap nights does not cause any problems, as the model accounts for the effect of variability in trap nights on the probability of detecting the species at an occupied site. For summarizing estimated detectability, for each year, we calculated an average overall detection probability for 3 trap nights as a derived parameter: i.e., we calculated the probability that a species present at a site would be detected at least once in 3 trap nights, based on the estimated population-wide nightly estimated detection probabilities each year, ignoring the site random effects.

- The model formulation estimates the initial probability of occupancy in 2010, with the partly unobserved true occupancy state as a Bernoulli random variable. We allowed initial probability of occupancy in 2010 to vary between sites that subsequently were or were not thinned, because thinning focused on sites with high tree densities. Thus, we incorporated a pre-fuels reduction 2010 habitat comparison largely comparing sites with higher densities of trees, subsequently thinned in 2010, vs. those not thinned in 2010, which were largely sites that had received past thinnings or otherwise had lower tree densities.

- Changes in site occupancy between 2010 and 2012 are modeled as a function of two occupancy dynamics parameters.
 - Site-level persistence probability is the probability that a site occupied in 2010 would remain occupied in 2012. Our model allowed persistence probability to vary among sites not thinned, thinned but not chipped, and thinned and chipped in 2010. If one of the management treatments significantly reduced occurrence of any of the 3 focal species, we would expect this to be measured by lower estimated probability of persistence.
 - Site-level colonization probability is the probability that a site unoccupied in 2012 would be occupied in 2010. Because of the low rate of colonizations observed between years, we estimated only 2 colonization probabilities, one for sites not thinned and one for all sites thinned in 2010, regardless of chipping status.
- Site-occupancy probability in 2012 is calculated as derived parameter as a function of initial occupancy probability, colonization probability, and persistence probability. We derived site-occupancy probabilities within the WinBUGS Markov chain Monte Carlo (MCMC) iterations for unthinned, thinned and not chipped, and thinned and chipped site categories, respectively.

All unobserved quantities were given uninformative prior distributions (Appendix 5). We ran 3 MCMC chains for each species, monitoring convergence via diagnostic plots and the Gelman-Rubin statistic. We report mean and 95% credible intervals for each estimated parameter from the saved MCMC sample draws.

RESULTS AND DISCUSSION

General Results

We sampled 77 sites, including 68 sites sampled in both years, and 9 sites added and sampled in 2012 only. This produced a total nominal trap effort of 11,725 trap nights (approximately 5900 in 2010, 5825 in 2012). Of the species targeted by this study, we captured a minimum of 10 species in Sherman live trapping, and visually detected two additional species on or off trap grids (Table 1). All species were detected in both 2010 and 2012 except for shrews and the western harvest mouse, which were livetrapped only in 2010. Although we captured the northern flying squirrel on our live trap grids only in 2012, additional trapping in 2010 as part of a complementary study detected the species (see below).

Of the species we detected, two were not previously confirmed in the Memorial: the northern flying squirrel and the northern pocket gopher. Both species are widespread in appropriate habitat in the Black Hills (R. Gitzen, pers. obs.; Turner 1974), and their occurrence in the Memorial is unsurprising. Confirmation of the occurrence of northern flying squirrels and discussion of habitat management for this species is discussed in a report in preparation by D. Licht, Midwest Region Wildlife Biologist. All detections of northern pocket gophers were detections of burrow mounds. Definite burrow mounds (Fig. 3) were first detected in the powerline corridor (UTMs 13 T 625483.00 m E 4860220.61 m N) southwest of the Highway 244- Highway 16 intersection, and subsequently detected scattered in deeper-soiled drainages scattered throughout the Memorial. In 2010, we attempted to capture northern pocket gophers using modified live traps. Although mounds were active with well-defined tunnels and gophers consistently pushed dirt into traps in repairing their burrow system, we did not capture any

pocket gophers, likely due to either non-optimized trap placement or the shortness of the traps we were using.

Schmidt et al. (2004) also reported captures of shrews in the *S. cinereus* / *S. haydeni* group, but was similarly unable to differentiate the two species. Therefore, neither species has been verified as present at the Memorial. Based on tail length criteria recommended by Higgins (2000), nearly all individuals we captured would be classified as masked shrews, with tail length greater than 36 mm. Two shrews had tail lengths of 36 and 35 mm, respectively. Without molecular confirmation, the species composition of shrews at the Memorial remains unclear. However, based on our results it appears that the Memorial either supports only masked shrews or a mix of masked and prairie shrews.

Table 1. Small mammals detected during field sampling at Mount Rushmore National Memorial, 2010 or 2012.

Common Name	Scientific Name	Detection Method
Masked or Prairie shrew	<i>Sorex cinereus</i> or <i>S. haydeni</i>	Live trap
Red squirrel	<i>Tamiasciurus hudsonicus</i>	Visual
Northern flying squirrel	<i>Glaucomys sabrinus</i>	Live trap
Least chipmunk	<i>Tamias minimus</i>	Live trap
Northern pocket gopher	<i>Thomomys talpoides</i>	Visual (sign)
Meadow jumping mouse	<i>Zapus hudsonius</i>	Live trap
Bushy-tailed woodrat	<i>Neotoma cinerea</i>	Live trap
White-footed mouse	<i>Peromyscus leucopus</i>	Live trap
Deer Mouse	<i>Peromyscus maniculatus</i>	Live trap
<i>Peromyscus</i> spp.	<i>P. leucopus</i> or <i>maniculatus</i> ,	Live trap
Western harvest mouse	<i>Reithrodonomys megalotis</i>	Live trap
Meadow vole	<i>Microtus pennsylvanicus</i>	Live trap
Southern red-backed vole	<i>Myodes gapperi</i>	Live trap

Table 2. Captures of small mammals during livetrapping at Mount Rushmore National Memorial, 2010 and 2012. Total Captures includes recaptures.

Species	Unique Individuals Captured		Total Captures	
	2010	2012	2010	2012
Masked or Hayden's shrew	15	0	15	0
Northern flying squirrel	0	2	0	2
Least chipmunk	232	162	327	241
Meadow jumping mouse	8	1	8	1
Bushy-tailed woodrat	5	2	5	2
White-footed mouse	7	2	7	2
Deer Mouse	130	34	176	44
<i>Peromyscus</i> spp.	6	0	19	0
Western harvest mouse	1	0	1	0
Meadow vole	3	1	3	3
Southern red-backed vole	224	55	238	59



Figure 3. Representative northern pocket gopher mounds detected in Mount Rushmore National Memorial during 2010. Photo by R. Gitzen.



Figure 4. Sample of species captured during 2010-2012 sampling at Mount Rushmore National Memorial. Top row: Masked or prairie shrew (left); meadow jumping mouse (right). Bottom row: southern red-backed vole (left); meadow vole (right). Photos by R. Gitzen.

Species Occurrence

Captures were dominated by 3 widespread species: least chipmunk, deer mouse, and southern red-backed vole. Both capture numbers and observed occurrence (Table 4) dropped greatly between 2010 and 2012. In 2010, the southern red-backed vole was the most frequently captured species, and was detected at least once at 90% of all sites. However, in 2012, it was recorded at only 39% of all sites, and was not recorded in 2012 at 47% of the sites where it was detected in 2010. The deer mouse showed a very similar pattern, with observed occurrence dropping from 72% of sites in 2010 to 27% in 2012, including a "naïve local extinction rate" of 50% (i.e., it was observed in 2012 at only 50% of the sites where observed in 2010). Other infrequently captured species showed similar patterns: we captured shrews at 12 sites in 2010 but recorded no shrew captures in 2012. Meadow jumping mice were recorded at 6 sites in 2010 but only 1 in 2012. The least chipmunk showed the lowest change in naïve occupancy status between years, as it was captured at 81% of sites in 2010 and 79% in 2012.

Of the species we captured, northern flying squirrels and bushy-tailed woodrats likely are not effectively sampled with ground-based Sherman traps. Therefore, the low detection rate (detections at 0-4% of sites sampled; Table 4) likely greatly underestimates the occurrence of these species on our sites due to a trap-method bias. Although some woodrat individuals can fit into Sherman traps (Fig. 2), their body size is such that many individuals are likely to spring the traps before squeezing inside. In addition, their occurrence is likely more localized than most other species we captured, as they were generally caught near selected rock outcrops and cliffs, as would be expected based on their general habitat requirements. Dan Licht (Midwest Region Wildlife Biologist, unpublished report) summarizes a more extensive pilot survey of northern

flying squirrels at the Memorial. In this report, we treat captures of both of these species as incidental.

In contrast, Sherman trapping is a standard approach for sampling meadow voles, white-footed mice, jumping mice, and harvest mice. The low observed occurrence of these species likely is affected much less by detectability biases than by true low occurrence during our sampling years across the Memorial. Both harvest mice and meadow voles are associated with denser herbaceous/grass layers with vegetation densities much higher than occurs at MORU outside of selected mesic draws and road cuts. The single harvest mouse captured was slightly below a roadcut to the southwest of the main MORU visitor's parking lot. Most captures of the meadow vole occurred in sites containing or bordering denser vegetation associated with intermittent drainages. Captures of *Peromyscus* identified as white-footed mice occurred both in hardwood draws and upslope areas. Meadow voles, harvest mice, and white-footed mice are widespread species of no species-specific conservation concern in the region; at the scale of the Black Hills, they occur commonly in suitable habitats. Their occurrence in this study was too low to support any meaningful insights about habitat affinities beyond the consistency with previously reported patterns in the literature.

Although meadow jumping mice were captured infrequently, their detections are noteworthy because the Black Hills has a "T3- Vulnerable" status (<http://www.natureserve.org>) and because of their putative association with riparian areas such as aspen draws that are of management concern in the region. Of the 6 sites where we detected this species, 5 sites were classified as encompassing significant aspen/birch/spruce draws, as in Starling Basin, while the 6th site bordered mesic intermittent drainages in upper Starling Basin. This association with mesic vegetation is consistent with that reported in the literature (e.g. Higgins et al. 2000).

Although the sparse data we obtained for this species precludes formal estimation of detectability, we note that at most sites where the species was detected, it was detected on only a single trap night out of the 3-4 trap night session. If we assume a per-night detection probability of 0.33 per trap night, the likelihood of detection at an occupied site at least once in 3 trap nights would be $1-(1-0.33)^3 = 0.70$; i.e., we would have a 0.30 probability of not detecting the species once in 3 trap nights at an occupied site. This suggests that our results underestimate occurrence of this species by ~30%. Thus, the species likely occurred but was not detected at a few additional mesic-draw sites we sampled. The strong association of this species with mesic draws was one of the more definitive patterns emerging from this study.

Although mesic conditions with denser vegetation and litter are most suitable for masked shrews (Higgins et al. 2000), it is a widespread species. Our detections of shrews were not strikingly associated with mesic-draw habitat: 4 of the 12 sites where shrews were detected in 2010 were classified as in hardwood/spruce mesic draws, which made up 15 out of 68 sites sampled in 2010. Our data for this group are too sparse to support meaningful examination of shrew habitat occupancy, but we note that shrews are among the most understudied vertebrate groups in the Black Hills. More specialized studies using pitfall traps are needed to better document patterns of occurrence for the shrew community in the region.

Table 3. Site detection status in small mammal live trapping during 2010 and 2012. Sites detected = number of sites with at least one detection of the species or species group by year; number in parentheses is the proportion of sites sampled (68 in 2010, 77 in 2012) where the species was detected. Site Detection History gives the change in detections status for the 68 sites sampled in both 2010 and 2012, designating the number (proportion) of sites where the species was caught in both years, neither year, or only one of the two years.

Species	Sites Detected		Site Detection History			
	2010	2012	Both Years	Neither	2010 Only	2012 Only
Masked or Hayden's shrew	12 (0.18)	0 (0.00)	0 (0.00)	56 (0.82)	11 (0.18)	0 (0.00)
Northern flying squirrel	0 (0.00)	2 (0.03)	0 (0.00)	66 (0.97)	0 (0.00)	2 (0.03)
Least chipmunk	55 (0.81)	61 (0.79)	48 (0.71)	8 (0.12)	7 (0.10)	5 (0.07)
Meadow jumping mouse	6 (0.09)	1 (0.01)	1 (0.01)	61 (0.90)	5 (0.07)	0 (0.00)
Bushy-tailed woodrat	3 (0.04)	2 (0.03)	0 (0.00)	63 (0.93)	3 (0.04)	2 (0.03)
White-footed mouse	7 (0.10)	2 (0.03)	1 (0.01)	60 (0.88)	6 (0.09)	1 (0.01)
Deer Mouse	49 (0.72)	21 (0.27)	15 (0.22)	15 (0.22)	34 (0.50)	4 (0.06)
Western harvest mouse	1 (0.01)	(0.00)	0 (0.00)	67 (0.99)	1 (0.01)	0 (0.00)
Meadow vole	2 (0.03)	1 (0.01)	0 (0.00)	65 (0.96)	2 (0.03)	1 (0.01)
Southern red-backed vole	61 (0.90)	30 (0.39)	25 (0.37)	5 (0.07)	36 (0.53)	2 (0.03)

Changes after 2010 Thinning

Given the large-scale decreases in occurrence of most species from 2010 to 2012, the key question is whether this decrease in occurrence appeared to be driven by year-to-year changes apart from those due to thinning, due to thinning treatments, or to some combination of the two major factors. Of the 77 sites sampled in 2012, the majority portions of 39 trap grids were in areas thinned in 2010, while 38 sites were not thinned in 2010. Counting only sites sampled in both years, 33 of 68 sites were treated in 2010, while 35 were not. Thus, we focused on comparing species detections between sites that were or were not thinned in 2010, breaking the analysis into segments to address multiple questions.

For reasons described earlier, we do not analyze changes in occurrence of meadow voles, bushy-tailed woodrats, or northern flying squirrels, as their occurrence or susceptibility to trapping is problematic for meaningful inference in this study. Similarly, because of uncertainty in morphological differentiation of the two *Peromyscus* species, we pooled them for examination of changes after 2010 fuels-reduction treatments.

Shrews: Shrews were captured at 12 sites in 2010 prior to density reduction treatments but none in 2012. Half of these 12 sites were in areas treated during 2010 fuels-reduction treatments, and half of the sites were in untreated areas. Although the data are too sparse to support meaningful examination of thinning effects, patterns in the data suggest that the decline in shrew detections for 2012 vs. 2010 was at least partially a park-wide change, one presumably driven by random year to year weather variation consistent with changes often noted in forested shrew communities (e.g. Gitzen et al. 2007).

Meadow jumping mouse: This species was detected at 1 site in both sampling years, while at 5 other sites it was observed only in 2010 prior to fuels reduction treatments. Although the site where it maintained observed occurrence was not treated in 2010, the other 5 sites where it was captured only in 2010 included a mix of sites not treated (3 sites) and treated (2 sites) in 2010. Thus, there is no strong pattern of differential loss from areas treated in 2010. This does not mean we can conclude that the fuels treatments had no effect on meadow jumping mice, shrews, or other infrequently captured species: i.e., lack of a detectable effect in a sparse data set does not imply lack of any true impact.

As with shrews, a more formal occupancy analysis accounting for variable detectability could be considered, but given the sparse captures, there would be very high uncertainty in estimates of detection probability and relative occupancy. Given the lack of any strong differential pattern in nightly detection records between the two groups of sites compared, such an analysis would not reveal anything beyond the patterns described above.

Common species – Changes in detection histories: Note that all of these following tallies include only the 68 sites sampled in both years. In 2012, least chipmunks were captured at 30 out of 35 sites not treated in 2010, and 23 out of 33 sites that had been treated in 2010. Of the 7 sites where the species was captured in 2010 but not 2012, 2 were not treated in between the two samples, while 5 were treated. However, of the 5 sites where the species was captured only in 2012, 2 were not treated in 2010, while 3 were.

In 2012, deer mice were captured at 12 out of 35 sites not treated in 2010, and 8 out of 33 sites that had been treated in 2010. Of the 33 sites where the species was captured in 2010 but not 2012, 16 were not treated in between the two samples, while 17 were treated. Of the 4 sites where the species was captured only in 2012, 2 were not treated in 2010, while 2 were.

In 2012, southern red-backed voles were captured at 15 out of 35 sites not treated in 2010, and 12 out of 33 sites that had been treated in 2010. Of the 36 sites where the species was captured in 2010 but not 2012, 18 were not treated in between the two sample periods, while 18 were treated. Of the 2 sites where the species was captured only in 2012, 1 was not treated in 2010, while 1 was.

Collectively, these patterns based on changes in raw detections do not show a dramatic difference between areas treated vs. not treated in 2010. Rather, the reduced detectable occurrence of deer mice and southern red-backed voles in 2012 vs. 2010 was consistent with a park-wide decline. The least chipmunk declined in observed occurrence far less than other species between years, suggesting it was relatively less sensitive to whatever factors were driving the large-scale decline in observed occurrence seen for other species.

Common species – Occupancy modeling:

For southern red-backed voles, occupancy modeling indicated that reduced detections of the species across MORU in 2012 was largely a matter of reduced detectability rather than reduced occurrence. Our model estimates indicated that for a 3 night trap session, the probability of detecting at least one red-backed vole at an occupied site dropped from 0.93 in 2010 to 0.41 in 2012. Thus, the reduced observed occurrence of this species was a sampling artifact: estimated probability of occupancy remained high in 2012.

Initial occupancy showed little variation among thinning categories (Table 4). Persistence at occupied sites was high across unthinned, thinned but not chipped, and thinned and chipped sites. That is, any changes in occurrence between years were not associated with the 2010 forest-management treatments. At the species-occurrence scale, the 2010 treatments did not have a detectable effect on occurrence of this species. We note that by allowing heterogeneity among

sites in detectability, we allowed for systematic differences in detectability among site categories: our analysis allowed correction for any habitat biases in detection probability. The lack of a strong response in occurrence by red-backed voles is noteworthy because we would have expected it to be one of the more likely species to show a negative response, at least in the short term, on sites that received fuels-management treatments.

However, this pattern needs to be interpreted carefully. Decreased detectability was almost certainly driven by a large-scale drop in abundance between years: there were fewer voles to detect per site in 2012, and thus a lower probability of detection. Thus, changes in the number of sites where we recorded detections between years reflect changes in abundance rather than changes in occurrence at the 0.36 ha scale. Although outside the scope of this report, analyses focusing on abundance rather than occurrence would allow examination of whether the 2010 treatments affected patch-level abundance.

The same pattern was observed for *Peromyscus* occurrence: 3-day estimated detection probability dropped from 0.84 in 2010 to 0.34 in 2012, and therefore the sharp drop in number of sites where the species was observed in 2012 overestimates the real decline in occurrence (Table 4). Again, after accounting for variable and incomplete detection probability, there was not strong evidence that the 2010 forest treatments affected patterns of occurrence for *Peromyscus*.

Least chipmunks showed little change in 3-night average detectability (0.96 in 2010 vs. 0.95 in 2012), unlike red-backed voles or *Peromyscus*. Although mean poster-distribution estimates of persistence and 2012 occurrence were somewhat lower on thinned and thinned and chipped sites vs. sites not treated in 2010, the wide 95% credible intervals indicate that this pattern is inconclusive. Least chipmunks would be expected to occur widely across habitat conditions in the Black Hills. This widespread, flexible species occurs commonly in rocky, open

forests, and we expect this species is more likely to increase in treated areas rather than decrease as time since treatment progresses.

Table 4. Posterior distributions from Bayesian hierarchical 2-season dynamic occupancy model. Results are means (95% credible intervals) from MCMC samples for each species. See text for additional information.

	Least Chipmunk	<i>Peromyscus</i> spp.	Southern red-backed vole
2010 Occupancy probability: Sites not thinned in 2010	0.92 (0.79, 0.99)	0.96 (0.92, 1.00)	0.98 (0.90, 1.00)
2010 Occupancy probability: Sites thinned in 2010	0.88 (0.73, 0.99)	0.94 (0.72, 1.00)	0.98 (0.88, 1.00)
2012 Derived occupancy probability: Sites not thinned in 2010	0.91 (0.79, 0.99)	0.80 (0.40, .00)	0.88 (0.57, 1.00)
2012 Derived occupancy probability: Sites thinned but not chipped in 2010	0.88 (0.74, 0.98)	0.71 (0.23, 1.00)	0.84 (0.39, 1.00)
2012 Derived occupancy probability: Sites thinned and chipped in 2010	0.84 (0.54, 0.98)	0.79 (0.28, 1.00)	0.92 (0.56, 1.00)
Persistence probability from 2010 to 2012: Sites unthinned	0.98 (0.89, 1.00)	0.82 (0.41, 1.00)	0.88 (0.57, 1.00)
Persistence probability from 2010 to 2012: Sites thinned but not chipped	0.98 (0.88, 1.00)	0.74 (0.23, 1.00)	0.85 (0.39, 1.00)
Persistence probability from 2010 to 2012: Sites thinned and chipped	0.93 (0.58, 1.00)	0.83 (0.27, 1.00)	0.93 (0.56, 1.00)
Colonization probability from 2010 to 2012: Sites unthinned	0.17 (0, 0.82)	0.44 (0.00, 1.00)	0.49 (0.00, 1.00)
Colonization probability from 2010 to 2012: Sites thinned	0.14 (0, 0.78)	0.41 (0.00, 1.00)	0.49 (0.00, 1.00)

CONCLUSIONS

We did not detect strong effects of the 2010 fuels management treatments on small mammals, based on comparing patterns in sites that were or were not treated during 2010. We note that this is an observational comparison: in the absence of randomized treatments and supplemental data from other areas in the Black Hills, our study does not provide strong inference about the effects of fuels management on small mammals. However, given the year to year changes frequently seen in small mammal populations, we cannot argue against the null model that the 2010 to 2012 changes in abundance were simply year to year variation rather than a systematic change due to the 2010 fuels treatments. Without a longer period of systematic data collection, we have no basis for inferring whether 2010 was a time of unusually high populations, whether 2012 was a time of unusually low populations, or any other possible combination of random effects. Unlike with many taxa, with small mammals a drop in occurrence of the magnitude observed here between two sampling years, particularly over a time frame as short as 2 years, is of ecological interest but not of any special management concern: the most likely scenario is that with more years of data following 2012, the change would appear as part of random year to year variation. This is a limitation of small mammal studies focusing on very short time periods: year to year variation beyond the factors being examined can wash out any finer scale changes due to factors of direct interest such as forest management.

We do not suggest, however, that potential management impacts on small mammal occurrence and abundance are inconsequential given the already extremely high year to year variation apart from any management effects. A trend in the long-term average abundance of common species may translate into large ecological impacts given the roles these species play in

the ecosystem. Unfortunately, detecting such changes requires expensive repeated monitoring dealing with high spatial and temporal variation.

We emphasize also that our results are specific to the scale at which we exemplified small mammal responses: species occurrence, rather than abundance. It is possible that patch-level occupancy at the scale of 0.36-ha patches is an insensitive metric of species responses, and that changes could occur at the level of abundance or finer-scale demographic patterns rather than species occurrence. Additional analyses of these data should consider capture counts as an additional metric for examining potential effects of the 2010 treatments.

This study did not detect a clear effect of the 2010 fuels reduction treatments on the small mammal community of Mount Rushmore National Memorial. However, in integrating small mammal responses into consideration of future potential ecological responses to fuels management at Mount Rushmore National Memorial, several factors should be considered:

- We suggest that avoiding significant disturbance to mesic aspen/birch/oak/spruce draws patches is an important aspect of maintaining the diverse small mammal community of the Memorial. Jumping mice occurred in hardwood-spruce draws with high herbaceous/grass cover, sites also suitable for shrews, meadow voles, northern flying squirrels, and most other species captured. These sites also frequently have high floral diversity compared to pine-dominated uplands. If continued fuels treatments and unplanned disturbances maintain most of MORU's uplands in less dense, albeit forest, states, the value of mesic draws for supporting some the park's fauna is likely to increase.
- Other structural habitat elements may have a large impact on the occurrence of some small mammal species. In particular, large down logs are a highly important

resource for forest-floor small mammals, particularly species such as southern red-backed voles and the northern flying squirrel. The park should continue to consider the relative habitat value of these structures in considering management activities that reduce the length or persistence of these species. From the standpoint of forest floor small mammals, it could be argued that the most important role an old, large tree plays for these species is after it falls and provides a long-term ground level habitat structure.

- Large-scale changes in the abundance of forest-floor small mammals at MORU, particularly the southern red-backed vole, are not likely to be caused by moderate changes in canopy cover per se, given the near ubiquitous occurrence of this species in 2010, the lack of strongly differential responses in areas treated vs. not treated in 2010, and the frequent flexibility of these species in dealing with partial disturbances that retain significant forest cover. Instead, such changes are more likely to occur if there are major changes in the structural state of the Memorial (e.g. a change to an open woodland state after fire or insect epidemic) or long-term changes in the amount of litter, down logs, and food resources such as fungi. Thus, it is the long-term effects of fuels management in affecting the risk of major disturbance and on site productivity that should be of most concern for small mammals.

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Appendix 1. Locations of Small Mammal Live Trapping Sites at Mount Rushmore National Memorial

Site	UTM-X	UTM-Y	Years Sampled
S-021	624145	4858871	2010, 2012
S-022	623122	4859302	2010, 2012
S-023	623553	4860756	2010, 2012
S-024	624738	4859626	2012
S-025	623822	4858709	2010, 2012
S-026	623553	4859571	2010, 2012
S-027	625922	4860648	2010, 2012
S-029	623661	4860217	2010, 2012
S-030	623338	4860002	2010, 2012
S-031	625330	4860756	2010, 2012
S-032	624630	4858763	2010, 2012
S-033	624038	4858602	2010, 2012
S-034	624145	4860163	2012
S-035	624091	4860540	2010, 2012
S-037	624415	4858979	2010, 2012
S-038	623553	4858979	2010, 2012
S-039	623230	4860325	2010, 2012
S-040	625168	4860056	2010, 2012
S-041	624038	4858709	2010, 2012
S-042	623391	4859571	2010, 2012
S-043	625330	4860325	2010, 2012
S-044	624899	4858709	2010, 2012
S-045	624304	4859894	2012
S-047	625168	4860325	2010, 2012
S-048	625745	4860163	2010, 2012
S-049	623768	4858548	2010, 2012
S-050	623338	4859356	2010, 2012
S-051	623715	4860756	2012
S-052	624734	4860271	2010, 2012
S-053	624307	4858709	2010, 2012
S-054	623068	4860217	2012
S-059	625545	4860702	2010, 2012
S-060	624953	4859194	2012
S-061	624361	4859625	2012

Site	UTM-X	UTM-Y	Years Sampled
S-062	623661	4860486	2010, 2012
S-063	625061	4860702	2010, 2012
S-064	625653	4860109	2010, 2012
S-067	623930	4860486	2010, 2012
S-068	625168	4859732	2010, 2012
S-069	624845	4858440	2010, 2012
S-070	623607	4859194	2010, 2012
S-073	624038	4858925	2010, 2012
S-074	623284	4859894	2010, 2012
S-076	624792	4859409	2010, 2012
S-079	625384	4860648	2010, 2012
S-081	624253	4858602	2010, 2012
S-084	625061	4859732	2010, 2012
S-085	624080	4859032	2010, 2012
S-087	623445	4860809	2010, 2012
S-088	624845	4859517	2010, 2012
S-089	623768	4858763	2010, 2012
S-090	623499	4859679	2010, 2012
S-091	625976	4860756	2010, 2012
S-092	624630	4859356	2010, 2012
S-093	623930	4860217	2010, 2012
S-096	624845	4858925	2010, 2012
S-097	623876	4858548	2010, 2012
S-098	624468	4860056	2010, 2012
S-099	624199	4860379	2012
S-100	624536	4860163	2012
S-101	624307	4858871	2010, 2012
S-104	625222	4859948	2010, 2012
S-105	623715	4859140	2010, 2012
S-106	623338	4859625	2010, 2012
S-107	625492	4860325	2010, 2012
S-108	624899	4858817	2010, 2012
S-110	623661	4860648	2010, 2012
S-111	625276	4860594	2010, 2012
S-112	625384	4860217	2010, 2012
S-113	624630	4858440	2010, 2012
S-114	623284	4859409	2010, 2012

Site	UTM-X	UTM-Y	Years Sampled
S-117	623661	4859032	2010, 2012
S-118	623122	4860109	2010, 2012
S-119	623445	4860486	2010, 2012
S-121	624078	4859140	2010, 2012
S-122	623176	4859517	2010, 2012
S-123	625653	4860702	2010, 2012

Appendix 2. Total unique individuals captured by site for infrequently captured species at Mount Rushmore, 2010-2012.

Only sites with captures of at least one of these species are listed; captures at all other sites from Appendix 1 and Appendix 3 were zero for these species.

Species codes: SOSP = Shrew species; GLSA = Northern flying squirrel; ZAHU = Meadow Jumping mouse; PELE = White-footed mouse; PESP = Peromyscus spp.; REME = Western harvest mouse; MIPE = Meadow vole.

Site	SOSP	GLSA	ZAHU	NECI	PELE	PESP	REME	MIPE
S-021	0	0	0	0	1	1	0	0
S-022	1	0	0	0	0	0	0	0
S-025	2	0	2	0	2	0	0	0
S-026	0	0	0	0	0	1	0	0
S-027	0	0	0	2	0	0	0	0
S-033	1	0	0	0	0	0	0	0
S-035	0	0	1	0	0	0	0	1
S-038	1	0	3	0	0	0	0	0
S-042	1	0	0	0	0	0	0	0
S-047	1	0	1	2	0	0	0	0
S-049	0	0	0	0	1	0	0	0
S-053	0	0	0	0	0	1	0	0
S-059	0	0	0	1	0	0	0	0
S-069	0	0	0	1	2	0	0	0
S-070	0	0	1	0	1	0	0	0
S-073	1	0	0	0	0	0	0	0
S-074	0	0	0	0	0	1	0	0
S-089	0	0	0	0	1	0	0	0
S-090	0	1	1	0	0	1	0	0
S-096	1	0	0	0	1	0	0	0
S-098	0	0	0	0	0	0	0	2
S-106	0	0	0	0	0	0	0	1
S-107	1	0	0	0	0	0	0	0
S-111	0	0	0	1	0	0	0	0
S-114	1	1	0	0	0	0	0	0
S-117	2	0	0	0	1	0	0	0
S-121	0	0	0	0	0	1	1	0
S-122	2	0	0	0	0	0	0	0

Appendix 3. Total unique individuals captured by site for frequently captured species at Mount Rushmore, 2010-2012.

NA = site not sampled in 2010.

	Southern red-backed vole				Deer Mouse				Least Chipmunk		
Site	2010	2012	Total		2010	2012	Total		2010	2012	Total
S-021	3	1	4		5	0	5		9	3	12
S-022	4	0	4		0	0	0		2	0	2
S-023	3	0	3		0	0	0		3	1	4
S-024	NA	0	0		NA	0	0		NA	0	0
S-025	5	0	5		1	0	1		3	1	4
S-026	4	0	4		1	0	1		0	0	0
S-027	0	0	0		4	1	5		6	5	11
S-029	3	3	6		3	0	3		1	1	2
S-030	2	0	2		2	1	3		1	2	3
S-031	5	0	5		2	0	2		2	4	6
S-032	2	1	3		4	0	4		0	2	2
S-033	2	2	4		2	3	5		1	0	1
S-034	NA	0	0		NA	0	0		NA	1	1
S-035	3	3	6		2	3	5		1	0	1
S-037	8	0	8		4	1	5		7	2	9
S-038	3	2	5		0	0	0		0	1	1
S-039	3	2	5		1	0	1		1	2	3
S-040	4	0	4		0	0	0		2	0	2
S-041	4	0	4		2	0	2		2	3	5
S-042	1	0	1		0	0	0		2	0	2
S-043	3	0	3		1	0	1		1	1	2
S-044	4	0	4		0	0	0		6	1	7

	Southern red-backed vole				Deer Mouse				Least Chipmunk		
S-045	NA	1	1		NA	0	0		NA	0	0
S-047	5	1	6		3	2	5		9	5	14
S-048	0	1	1		2	1	3		7	1	8
S-049	0	0	0		4	0	4		0	4	4
S-050	6	2	8		0	0	0		0	0	0
S-051	NA	0	0		NA	0	0		NA	2	2
S-052	6	0	6		5	0	5		2	3	5
S-053	1	4	5		0	1	1		1	1	2
S-054	NA	0	0		NA	3	3		NA	4	4
S-059	1	0	1		5	0	5		15	9	24
S-060	NA	1	1		NA	0	0		NA	7	7
S-061	NA	1	1		NA	1	1		NA	3	3
S-062	5	0	5		1	2	3		0	0	0
S-063	1	0	1		2	2	4		8	3	11
S-064	4	0	4		0	0	0		5	1	6
S-067	9	3	12		1	2	3		0	2	2
S-068	4	2	6		0	0	0		5	2	7
S-069	9	0	9		1	0	1		1	2	3
S-070	3	1	4		1	0	1		4	1	5
S-073	3	1	4		0	0	0		2	0	2
S-074	1	0	1		1	0	1		4	2	6
S-076	4	0	4		2	2	4		7	3	10
S-079	1	1	2		1	0	1		7	7	14
S-081	1	1	2		3	0	3		0	0	0
S-084	4	0	4		0	0	0		11	0	11
S-085	0	2	2		8	0	8		1	0	1
S-087	2	0	2		0	1	1		2	3	5
S-088	1	0	1		6	0	6		6	1	7
S-089	3	0	3		3	0	3		3	2	5

	Southern red-backed vole				Deer Mouse				Least Chipmunk		
S-090	3	1	4		5	1	6		0	0	0
S-091	0	0	0		3	1	4		5	3	8
S-092	2	0	2		4	0	4		8	4	12
S-093	6	2	8		0	1	1		2	0	2
S-096	7	0	7		3	0	3		5	4	9
S-097	7	1	8		3	0	3		1	0	1
S-098	4	0	4		0	0	0		0	2	2
S-099	NA	0	0		NA	0	0		NA	1	1
S-100	NA	0	0		NA	0	0		NA	3	3
S-101	2	1	3		6	0	6		2	3	5
S-104	7	0	7		1	0	1		7	5	12
S-105	4	0	4		3	0	3		6	1	7
S-106	1	1	2		1	0	1		0	0	0
S-107	6	0	6		2	0	2		1	3	4
S-108	0	0	0		2	0	2		5	5	10
S-110	0	0	0		0	1	1		1	2	3
S-111	1	2	3		2	0	2		9	5	14
S-112	6	0	6		0	1	1		6	1	7
S-113	1	0	1		3	0	3		4	5	9
S-114	3	1	4		0	0	0		0	0	0
S-117	4	0	4		2	0	2		2	1	3
S-118	6	8	14		0	0	0		5	1	6
S-119	5	0	5		1	0	1		3	3	6
S-121	2	0	2		3	3	6		3	1	4
S-122	5	2	7		0	0	0		0	0	0
S-123	2	0	2		2	0	2		7	11	18

Appendix 4. Keys/Descriptions to Small Mammals at Mount Rushmore

The following keys and descriptions were used in during this study to guide identification of species based on morphological characteristics and based on likelihood of occurrence at MORU. The information is a consolidation of records and information from the National Park Service NPSpecies database circa 2009 and from the following references: Hall (1981), Higgins et al. (2002), Jones et al. (1985), Kays and Wilson (2002), Long (1965), Nagosen (2002), Schwartz and Schwartz (2001), and Turner (1974). The keys/descriptions are provided for internal use only, as an unpublished synthesis of published information.

Field Key and Handling Instructions for Mice

I. Tail much longer than $\frac{1}{2}$ the body length (could be slightly shorter, equal to, or slightly or much longer than the body). Not noticeably stocky.

A. External cheek pouches absent.

1. Tail longer than body. Very large hind feet relative to body (HF ~28-31 mm). Visually, immediately obvious that it is not one of the other mice present at MORU. Ear small for body size compared to *Peromyscus*. Coarse, darker dorsal fur; possibly orange tinge along the flank between dorsal darker fur and ventral white fur. Powerful jumper.

ZAHU: *Zapus hudsonius*. Meadow jumping mouse.

2. Tail <, =, or slightly > body length. Hind feet ~10-25 mm, obviously not a ZAHU, general *Peromyscus*-like body morphology.
 - a) Upper incisors conspicuously grooved lengthwise, almost making the two teeth look like four if not seen clearly. Adult (animal reproductively active/ has adult pelage) mass usually < ~15 g. Dorsal adult pelage of confirmed MORU species has a reddish tinge compared to *Peromyscus*.
 - (1) Adults not tiny, 10-12 g, adult total length >120 mm, tail longer, > $\frac{1}{2}$ total length (i.e., longer than body), tail bicolored but with wider dorsal stripe; adult pelage may have slight reddish tinge; dorsal midline vaguely or not darker than flanks.

REME: *Reithrodontomys megalotis*. Western harvest mouse.

After seeing many *Peromyscus*, you likely usually will recognize this as something different; adults usually are the size of juvenile *Peromyscus*. But there is overlap; check incisors of any moderately small mouse. Captured 1 in 2010.

- (2) Adults very small; tail relatively short (<body length length), adult total length usually <120 mm, tail distinctly bicolored with narrow dorsal stripe <1/4 diameter of tail, dorsal midline darker, adult pelage has no reddish tinge

REMO: *Reithrodontomys montanus*. Plains harvest mouse.

Not likely to be present. SAVE ANIMAL FOR CONFIRMATION.

- b) Upper incisors not grooved. Adults usually 15-25 g.
 - (1) Annulations of scales easily visible on mostly naked tail, ears nearly naked, ventral pelage gray or darker.

MUMU: *Mus musculus*. House mouse.

SAVE ANIMAL FOR CONFIRMATION. Probably present in MORU but likely to be captured only rarely if at all in MORU native habitats.

I.A.2.b) continued (Mice; tail $>> \frac{1}{2}$ body length, no cheek pouches, front incisors not grooved)

(2) Tail "scales" mostly/completely covered with fur (and noticeable "pencil" hairs at tip of tail), ears prominent but furred, ventral pelage white

(a) Tail sharply bicolored. Adult tail usually < 70 mm, hind foot usually < 20 mm, summer pelage relatively dark but less dull, may have richer brown tint

**PEMA: *Peromyscus maniculatus*. Deer mouse.
Collect tail-hair samples. Ear tag.**

(b) Tail bicolor less distinct. Adult tail often > 70 mm, adult hind foot usually > 20 mm, pelage somewhat duller, adults larger on average (non-pregnant adults > 25 g are more likely to be this species)

**PELE: *Peromyscus leucopus*. White-footed mouse.
Collect tail-hair samples. Ear tag.**

(c) Tail bicolor not clearly in one of the above categories; not confident in ability to tell whether it is PELE or PEMA.

**PESP: *Peromyscus sp.* Unidentified *Peromyscus*.
Collect tail-hair samples. Ear tag.**

B. External cheek pouches present, often filled with seeds. Auditory bullae at back of skull are large; gives head a little different appearance than other mice and can be felt when holding animal behind neck. Immediately obvious based on pelage and morphology that it is not a *Peromyscus*.

PEPO: *Perognathus sp.* Pocket mouse.

SAVE ANIMAL FOR CONFIRMATION. Remotely possible but unlikely to be captured

II. Tail $\leq \frac{1}{2}$ as long as body; tail < 44 mm long; much stouter and heavy bodied compared to other mice; adults typically weight 35-40 g; long claws for a mouse. Pelage may be similar to other mice of MORU, but body structure is very obviously different.

ONLE. *Onychomys leucogaster*. Northern grasshopper mouse.

SAVE ANIMAL FOR CONFIRMATION. This is a grassland species: widespread in the Plains, possible in Black Hills foothills, very unlikely at MORU

Field Key and Handling Instructions for Voles

I. Dorsal stripe reddish/reddish brown contrasting with gray sides. Adults similar to PEMA in body size; non-pregnant adult mass typically 15-25 g. Six plantar tubercles (six pads on sole of hind foot).

MYGA: *Myodes* [formerly *Clethrionomys*] *gapperi*. Southern red-backed vole.

Notes: Low density but widespread at MORU. Usually the red band is very distinct, but may not be in a few animals, mainly very young or soaking wet animals. Some *Microtus* may have enough of a brown or slight reddish tint to make you second-guess what they are. But generally the difference between these genera is not subtle.

II. No distinctive reddish dorsal band (although reddish tinge dorsally possible for MILO). Five or six plantar tubercles. Adults equal to or much larger than MYGA (adult mass 30-50+ g)

A. Tail usually <50% of body length, tail <~50 mm.

1. Tail extends 2x or slightly >2x further than hindlegs when both are extended straight back. Body fur relatively fine rather than coarse. Dorsal fur slightly grizzled but without salt and pepper appearance. Belly fur generally grayish to whitish, possibly silver tinge. Adult females have 4 or 5 pairs of teats. Usually 6 plantar tubercles on soles but some may have 5.

MIPE. *Microtus pennsylvanicus*. Meadow vole.

2. Tail extends ~1.5x beyond hind legs when both are extended straight back. Dorsal fur coarse, shaggy, possibly with noticeable salt-and-pepper appearance. Underparts usually buff to ochraceous/yellowish tinge or brownish gray. Name means "yellowish belly." Three pair of teats. Usually 5 plantar tubercles but some may have 6.

MIOC. *Microtus ochrogaster*. Prairie vole.

SAVE any suspected MIOC for verification.

B. Tail usually >50% of body length, >50 mm, tail extends more than 2x further than hindlegs when both are extended straight back. Fur not noticeably coarse/shaggy. Possibly diffuse reddish dorsal tinge, but not a band as in MYGA. Underfur grayish white. 6 plantar tubercles. 8

MILO: *Microtus longicaudus*. Long-tailed vole

SAVE for verification.

C. Tail damaged or inconclusive; escaped before measurement; uncertain of species.

MISP. *Microtus* sp. Unknown *Microtus*.

SAVE for examination.

Microtus notes: There is overlap in characteristics of these 3 species, and subtle pelage differences are a judgment call. Collection of specimens and examination of teeth is necessary to ID many individuals. Only MIPE has been documented at MORU. However, both MILO and MIOC have been documented in the central Black Hills and Jewel Cave NM. Suitable habitat is mostly absent at MORU

Key to Voles: Dental Characters

(Intended only for dead specimens for which skulls are not prepared)

- I. Most loops of enamel on upper molars rounded. Third upper molar lacks distinct posterior heel; first lower molar has four projections on lingual side. [Doublecheck accuracy of this criteria for Black Hills species]. Molars rooted in adults.

MYGA: *Myodes* [formerly *Clethrionomys*] *gapperi*. Southern red-backed vole

- II. Most loops of enamel on upper molars sharply angled. Third upper molar has distinct posterior heel; first lower molar has five projections on lingual side. [Doublecheck this for Black Hills species.] Molars not rooted in adults, ever-growing.

- A. M2 (second upper molar) with four islands of dentine surrounded by enamel (four closed triangles) and no posterior loop.

1. M3 with 2 closed triangles

MIOC. *Microtus ochrogaster*. Prairie vole.

2. M3 with 3 closed triangles

MILO: *Microtus longicaudus*. Long-tailed vole.

- B. M2 has five islands of dentine surrounded by enamel; distinct posterior loop that often appears as a fifth triangle

MIPE. *Microtus pennsylvanicus*. Meadow vole.

Shrews

There are 1-3 shrew species considered possible at MORU. Distinguishing these species reliably with live specimens is problematic. Re-examination of shrew taxonomy and biogeography occurs frequently, and generally depends on voucher specimens. Moreover, more than with any other mammal taxa, there is a (small) possibility of capturing species not expected to be present and not known from the Black Hills – and a high possibility that an unexpected species would look nearly the same as expected species.

Field Key to Shrews

I. Tail <40% of body length

CRPA: *Cryptotis parva*. Least shrew. Tail 15-18 mm. 4-6 g. Not considered possible but documented at Wind Cave and Pennington Cty, so within the realm of possibility. **Save any shrew with relative tail length much shorter than other shrews.**

II. Tail >40% of body length

A. Average-sized small shrew; 3-6 g. Hind foot usually >11 mm. Tail probably distinctly bicolored.

1. Tail <36 mm. No black tuft at tip of tail. Paler than SOCI; has tricolor pattern laterally. **Save any very pale/gray shrew.**

SOHA: *Sorex haydeni*. Hayden's (Prairie) shrew.

2. Tail >36 mm; slightly larger than SOHA; black tuft at tail tip; body has less tricolor laterally.

SOCI: *Sorex cinereus*. Masked (common) shrew. (No obvious mask present.) **Save for examination.**

Hayden's shrew formerly was considered a subspecies of *cinereus*. These characters may allow preliminary ID but expert examination required. Not certain of whether pelage characters are useful. Based on Turner (1974), just *haydeni* or maybe a mix of both species seems likely for the Black Hills. One or both has been captured at MORU. We will save some suspected specimens of each or both, depending on what we capture.

B. Very small; average weight about 2.5 g. Hind foot usually 10-11 mm. Tail indistinctly bicolored.

SONA: *Sorex nanus*. Dwarf shrew. Very small. One notch down in size from other shrews considered possible at MORU. Tail may be indistinctly bicolored. Listed as "unconfirmed" at MORU and WICA; considered by Higgins et al. to occur around but mostly not within the Black Hills. Small enough that it may not spring Sherman traps. **Save any shrew noticeably small compared to SOCI/SOHA.**

Lab Key to shrews: Vaguely Possible Species

- III. Tail <40% of body length. Four unicuspid teeth in each jaw.

***Cryptotis parva*. Least shrew.** Tail 15-18 mm. 4-6 g. Not considered possible but documented at Wind Cave and Pennington Cty, so within the realm of possibility.

- IV. Tail >40% of body length. Five unicuspid teeth in each jaw (3-5 visible from side view).

- A. When skull viewed laterally, 3 unicuspid teeth visible; U3 compressed anteroposteriorly, disklike; HF usually <10 mm.

***Sorex hoyi*. Pygmy shrew.** Mass ~3 g. Not known west of Missouri in SD, but found in northern Rockies and in Rockies of SE WY/NE CO. Very remote chance of occurrence.

- B. When skull viewed laterally, 4-5 unicuspid teeth visible; U3 not as above; HF > 10 mm

1. 3rd unicuspid tooth larger than 4th unicuspid

- a) First upper incisor lacks a medial tine.

***Sorex merriami*. Merriam's shrew.** Usually pale. 3-7 g. In SD known only from one county in NW with sagebrush, but can occur in dry forests.

- b) First upper incisor with a medial tine

***Sorex cinereus*/ *Sorex haydeni* (also *S. preblei*).** Tail may be distinctly bicolored. Distinguishing any of these species requires examination of a museum expert.

Masked (Common) shrew. (Has no obvious mask). 3-5 g. Tail typically greater than 36 mm; slightly larger and darker than Hayden's shrew with a black tuft at tip of tail; less of a tricolor pattern laterally. **Hayden's (Prairie) shrew** formerly was considered a subspecies of *cinereus*. Tail <36 mm, no black tuft at tip of tail, slightly smaller, paler; tricolor pattern laterally. These characters may allow preliminary ID but expert examination required. Based on Turner (1974), just *haydeni* or maybe a mix of both species seems likely for the Black Hills.

Preble's shrew is not considered present in SD or eastern Wyoming. 2-4 g. But present in eastern Montana; new range extensions occur regularly; very remote chance of occurrence.

2. 3rd unicuspid tooth smaller than 4th unicuspid

- a) Tail usually <40 mm, HF usually <11, braincase flattened

***Sorex nanus*. Dwarf shrew.** Very small. Mass 2.5 g, total length 85-94 mm. One notch down in size from other shrews considered possible at MORU. Tail may be indistinctly bicolored. Listed as "unconfirmed" at

MORU and WICA, so; considered by Higgins et al. to occur around but mostly not within the Black Hills.

- b) Tail usually >40 mm, HF usually >11, braincase convex viewed laterally

***Sorex vagrans*. Vagrant shrew.** 3-6 g. Not considered possible; not known from SD but widely distributed in western NA, historical record near Sundance; remote chance of occurrence.

Appendix 5. WinBUGS Code for Fitting Occupancy Models Summarized in the Main Text

```

sink("model.txt")

cat("

model{

#Prior distributions

B0.lp~dnorm(0,0.1)

B2.lp~dnorm(0,0.1)

B3.lp~dnorm(0,0.1)

B12.lp~dnorm(0,0.1)

sigma.p.site~dunif(0,5)

tau.p.site<-1/(sigma.p.site*sigma.p.site)

B0.psi~dnorm(0,0.1)

B1.psi~dnorm(0,0.1)

B0.phi~dnorm(0,0.1)

B1.phi~dnorm(0,0.1)

B2.phi~dnorm(0,0.1)

B0.gamma~dnorm(0,0.1)

B1.gamma~dnorm(0,0.1)


#Generate site random effects in logit p

for(s in 1:nsites){

lp.site.10[s]~dnorm(0,tau.p.site)

lp.site.12[s]~dnorm(0,tau.p.site)}

```

```

#Likelihood

#Occupancy and dynamics
for(i in 1:nsites){

z[i,1]~dbern(psi10[i])

psi10[i]<-1/(1+exp(-psi.lp[i]))

psi.lp[i]<-B0.psi+B1.psi*Thinned[i]


muz[i,2]<-z[i,1]*phi[i]+(1-z[i,1])*gamma[i]

z[i,2]~dbern(muz[i,2])

phi[i]<-1/(1+exp(-phi.lp[i]))

phi.lp[i]<-B0.phi+B1.phi*Thinned[i]+B2.phi*Chipped[i]

gamma[i]<-1/(1+exp(-gamma.lp[i]))

gamma.lp[i]<-B0.gamma+B1.gamma*Thinned[i]


#Observation process

for(j in 1:nsurveys[i,1]){

p10[i,j] <-1/(1+exp(-p10.lp[i,j]))

p10.lp[i,j]<-B0.lp+time2[j]*B2.lp+time3[j]*B3.lp+time4[j]*B3.lp+lp.site.10[i]

mu10[i,j]<-z[i,1]*p10[i,j]

Y10[i,j]~dbern(mu10[i,j])


for(j in 1:nsurveys[i,2]){

p12[i,j] <-1/(1+exp(-p12.lp[i,j]))

p12.lp[i,j]<-B0.lp+time2[j]*B2.lp+time3[j]*B3.lp+time4[j]*B3.lp+B12.lp+lp.site.12[i]

mu12[i,j]<-z[i,2]*p12[i,j]

```

```
Y12[i,j]~dbern(mu12[i,j])
```

```
}
```

```
#Summary Parameters
```

```
for(l in 1:3){
```

```
  p10.average.lp[l]<-B0.lp+ time2[l]*B2.lp+time3[l]*B3.lp
```

```
  p10.average [l]<-1/(1+exp(-p10.average.lp[l]))
```

```
  p12.average.lp[l]<-B0.lp+ time2[l]*B2.lp+time3[l]*B3.lp+B12.lp
```

```
  p12.average [l]<-1/(1+exp(-p12.average.lp[l]))
```

```
}
```

```
p10.average.overall<-1-(1-p10.average[1])*(1-p10.average[2])*(1-p10.average[1])
```

```
p12.average.overall<-1-(1-p12.average[1])*(1-p12.average[2])*(1-p12.average[1])
```

```
psi10.unthinned.lp<-B0.psi
```

```
psi10.thinned.lp<-B0.psi+B1.psi
```

```
psi10.unthinned<-1/(1+exp(-psi10.unthinned.lp))
```

```
psi10.thinned<- 1/(1+exp(-psi10.thinned.lp))
```

```
phi.unthinned.lp<-B0.phi
```

```
phi.thinned.lp<-B0.phi+B1.phi
```

```
phi.thinnedchipped.lp<-B0.phi+B1.phi+B2.phi
```

```
phi.unthinned<-1/(1+exp(-phi.unthinned.lp))
```

```
phi.thinned<- 1/(1+exp(-phi.thinned.lp))
```

```
phi.thinnedchipped<- 1/(1+exp(-phi.thinnedchipped.lp))
```

```
gamma.unthinned.lp<-B0.gamma
```

```
gamma.thinned.lp<-B0.gamma+B1.gamma
```

```
gamma.unthinned<-1/(1+exp(-gamma.unthinned.lp))
```

```
gamma.thinned<- 1/(1+exp(-gamma.thinned.lp))
```

```
psi12.unthinned <-psi10.unthinned*phi.unthinned+(1-psi10.unthinned)*gamma.unthinned
```

```
psi12.thinned <-psi10.thinned*phi.thinned+(1-psi10.thinned)*gamma.thinned
```

```
psi12.thinnedchipped <-psi10.thinned*phi.thinnedchipped+(1-psi10.thinned)*gamma.thinned
```

```
}
```

```
",fill=TRUE)
```

```
sink()
```