The Impact of Recreational Trails and Grazing on Small Mammals in the Colorado Piedmont

CARRON A. MEANEY, ANNE K. RUGGLES,
NORMAN W. CLIPPINGER, and BRUCE C. LUBOW

Denver Museum of Nature and Science and University of Colorado Museum,
777 Juniper Avenue, Boulder, CO 80304 (CAM)
850 37th Street, Boulder, Colorado, 80302 (AKR)
Department of Environmental, Population, and Organismic Biology,
University of Colorado, Boulder, Colorado, 80309 (NWC)
Department of Fishery and Wildlife Biology, Colorado State University,
1821 Willow Springs Way, Fort Collins, Colorado, 50523 (BCL)

ABSTRACT -- We conducted a three-year study of the impact of recreational trails and grazing on species richness, relative abundance, and species diversity of small mammals at six paired sites with and without trails along South Boulder Creek, Boulder, Colorado. In our analysis, we used a set of alternative models, which we evaluated using Akaiki's Information Criteria (AIC) to compute strength of evidence supporting each alternative and then made all inferences based on weighted averages of these model results. Our data provided strong evidence for an increase (2.0 individuals per 100 trap nights ± 0.51 SE) of deer mice (*Peromyscus maniculatus*) on the grazed sites, but little evidence for effects on relative abundance of other species or on species richness or diversity. Repeated measures ANOVA results for paired trail and non-trail sites showed only weak evidence for a negative effect of trails on species richness, species diversity, and relative abundance. In addition to small mammal trapping, we employed mark-recapture techniques on Preble's meadow jumping mouse (*Zapus hudsonius preblei*), a federally listed threatened subspecies of the meadow jumping mouse, to determine linear population density estimates of this subspecies on the trail and non-trail sides of the creek. Repeated measures ANOVA for these density estimates provided weak evidence for a possible negative trail effect (-11.6 individuals/km ± 9.5 SE) that was greater in males than females. Although the low precision of these estimates makes the results inconclusive, the magnitude of the estimated effect (a 31% lower population density of Preble's meadow jumping mice on sites with trails) highlights the need for careful management and additional research. Our data revealed large natural temporal and spatial variation in these populations that resulted in poor precision of estimated effects of interest.

Key words: grazing, Preble's meadow jumping mouse, recreational impacts, small mammals, trail impacts.

As human populations have increased, there has been an increase in demand for places to engage in outdoor recreational activities (Flather and Cordell 1995). As development proceeds, there is a decrease in available open landscapes and wildlife is restricted to ever-smaller areas. Consequently, there is a strong motivation to optimize the shared use of existing open spaces for both recreation and wildlife. Because of the attraction of the lush vegetation and water, alignment of trails within riparian corridors is common; they often become a preferred recreational corridor, enjoyed by humans for hiking, cycling, bird-watching, jogging, and dog-exercising (Bekoff and Meaney 1997). Little is known about what impact the use of these trails might have on small mammals.

The study of the impact of recreational trails on wildlife is a new and growing field of research with a relative paucity of data. Disturbance by recreationists can affect habitat, populations (abundance), or community interactions and composition (e.g., species richness (Anderson 1995, Gutzwiller 1995)). Trails fragment habitats, increasing the so-called "edge effect". Most studies of wildlife and recreational trails have focused on birds. Very little is known about the effects of trails on small mammal communities. There have been a few studies on the effect of roads on animal crossings. The width of inhospitable habitat presented by a road was the most critical factor in determining road crossings by mammals (Oxley et al. 1974). In some cases, even a 10 m strip of grass, dirt, or gravel inhibited movements of prairie voles (*Microtus ochrogaster*) (Meserve 1971, Cole 1978, Swihart and Slade 1984).

One motivating factor for our study was the presence of Preble's meadow jumping mouse (*Zapus hudsonius preblei*) on City of Boulder open space in Boulder, Colorado. This small mammal subspecies, a Pleistocene relic, occurs only in Colorado and Wyoming, and was listed as threatened under the Endangered Species Act on May 13, 1998 (63 FR 26517). In Colorado, *Z. h. preblei* exhibits an affinity for complex riparian communities with shrub, tree, grass, and forb species (Ryon 1996). Development along riparian areas and associated uplands, predation by both wild and domestic predators, destruction of wetland areas, grazing, and gravel-mining probably all have had a detrimental effect on local populations (Fitzgerald et al. 1994, Ryon 1996). Riparian corridors also were favored for grazing due to availability of cover and water. Small mammal species richness and diversity was higher in exclosures than in adjacent riparian areas in Nevada (Medin and Clary 1989). In northern Colorado, the deer mouse (*Peromyscus maniculatus*) was more abundant in the grazed area whereas the western jumping mouse (*Zapus princeps*) was more abundant in the exclosures (Schulz and Leininger 1991).

The purposes of our study were to learn 1) whether trails have an impact on small mammal species richness, relative abundance, and species diversity; 2) whether trails have a negative effect on linear population density estimates of Preble's meadow jumping mice specifically; and 3) how grazing affects small mammal richness, relative abundance, species diversity, and species composition in a riparian corridor.

STUDY SITE

South Boulder Creek lies in a broad floodplain, with well-developed riparian vegetation, adjacent grasslands, and numerous wetlands. A number of ditches draw irrigation water from the creek onto the adjacent lowland meadows. Cottonwoods (*Populus* spp.) dominate the tree overstory; smaller trees include alder (*Alnus incana*) and hawthorn (*Crataegus macracantha*). Coyote willow (*Salix exigua*) dominates the shrub understory, and other shrubs include western wild rose (*Rosa woodsii*), chokecherry (*Padus virginiana*), wild plum (*Padus americana*), golden currant (*Ribes aureum*), and western snowberry (*Symphoricarpos occidentalis*). There is a diverse herbaceous community including forbs, grasses, and sedges.

A long stretch of the creek is under open space management and there is a recreational trail on one side of the creek and no trail on the other side (Fig. 1). Grazing occurs in the riparian corridor in the southern half of the study area, where the trail is on the west side of the creek, but does not occur in the riparian corridor in the northern segment, where the trail is on the east side of the creek. Fencing excludes the cattle (*Bos taurus*) from the riparian corridor in the northern segment. Along most of its length through the study area, the creek is about 5 to 8 m wide and fast enough to discourage crossing by small mammals. The hiking trail is 3 m wide, composed of road base and crusher fines. A concrete bicycle trail of similar width parallels the hiking trail for 0.8 km at the north end and was completed within the study area in the spring of 1997.

METHODS

Six study sites were randomly selected from all possible non-overlapping 100-meter sections of this 3.7 km stretch of South Boulder Creek and adjacent trail (Fig. 1). Each site contained two trapping grids, one on the trail side of the creek and the other on the non-trail side of the creek, placed opposite each other except when obstacles such as barns or fawning grounds were present. In such cases the two

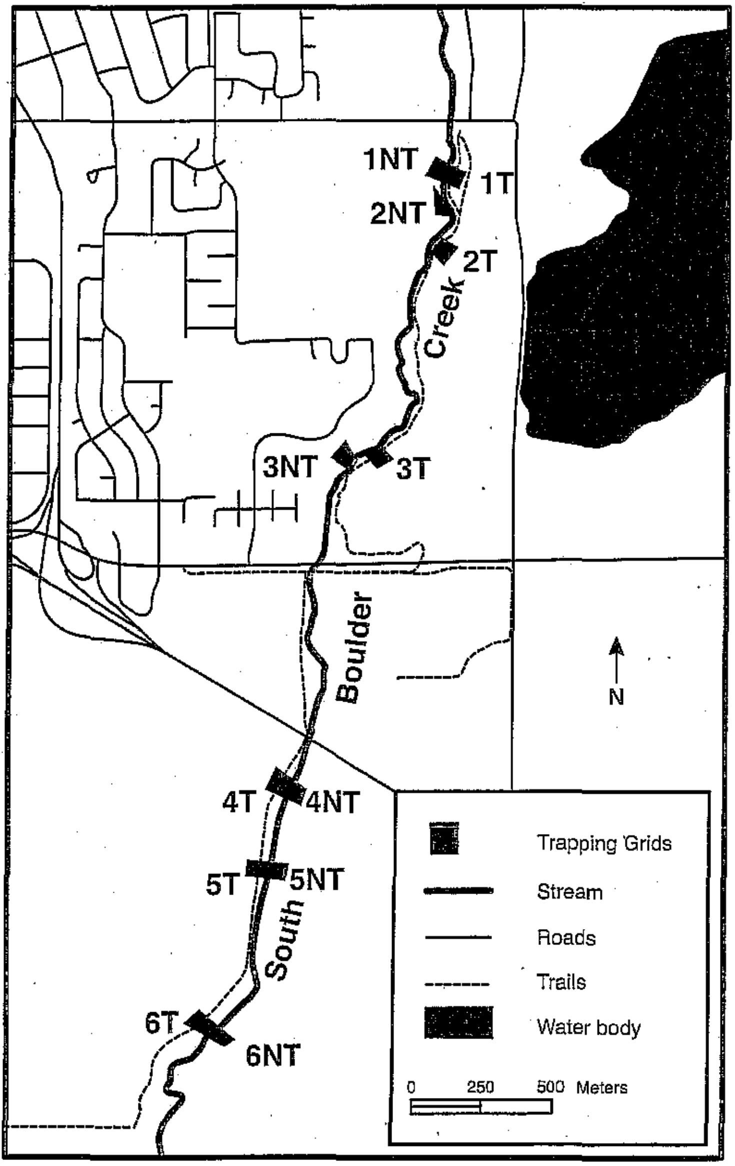

Figure 1. South Boulder Creek study area, showing 12 paired trapping grids on the trail and non-trail side of the creek, Boulder, Colorado. Sites 1 - 3 are non-grazed by the creek. Sites 4 - 6 are grazed up to the creek.

grids were somewhat offset. Sites 1 to 3 were located in the northern segment and Sites 4 to 6 were located in the southern segment.

Nine randomly selected vegetation plots were sampled at trap stations on each of the 12 trapping grids (108 plots in all). Plots were 5 m in radius with the trap at the center. A visual estimate of the percent canopy cover of trees, shrubs, grasses, and forbs was made to the nearest 10%. Plant species richness was tabulated in one of three categories: 0 to 1, 2 to 3, and 4 to 5 species for trees and grasses; 0 to 2, 3 to 5, and 6 to 8 species for shrubs, and 1 to 3, 4 to 7, and greater than 7 species for forbs.

Visitor use data were collected during the summer of 1998. We counted the number of people and dogs and recorded the type of activity in which they were engaged during 35 three-hour observation periods. Observation periods were scheduled throughout the summer (17 in June, 10 in July, and 8 in August) and included roughly equal coverage of morning, midday, and evening (14, 10, and 11 observation periods, respectively). Seventeen observation periods occurred in the three northern sites and 18 periods occurred in the three southern sites. The study area was divided into these northern and southern portions by a busy four-lane road and a highway. Dogs were allowed, under voice and sight control, in the northern portion and were prohibited on the southern portion. Activities noted included: walking, jogging, biking, and horseback riding. Observation points coincided with the grid locations and allowed simultaneous viewing of both the trail side and the non-trail side of the creek.

Two types of trapping grids were employed. In 1997, three small mammal trapping grids composed of three parallel transects of 25 traps each (for a total of 75 traps per grid) were used. The study was revised in 1998 and expanded with the addition of nine grids. The new trapping grids contained 72 traps, each laid out as eight trap stations along the creek and nine trap stations perpendicular to the creek. Traps were placed 9 m apart on both axes, thus forming a grid 63 m along the creek by 72 m out from the creek. The 1997 grids either superimposed or overlapped with the 1998 and 1999 grids, and thus were incorporated easily into the revised and expanded design. Grid locations were walked in the field and recorded with a Global Positioning System unit to show their location on the map (Fig. 1). Using Sherman live traps (8 cm x 9 cm x 23 cm), we followed standard field procedures for small mammal trapping and followed the guidelines approved by the Animal Care and Use Committee of the American Society of Mammalogists (1998). Traps were baited with molasses in rolled oats, corn, and filler grains. A ball of polyester batting was placed inside each trap for insulation and bedding.

We trapped the grids twice during each year's field season: four nights in June and four nights in August. Field work was conducted for three years (1997 through 1999) for a total of six trapping periods. In 1997, trapping was conducted in July instead of June. Two sites (four grids, 288 traps) were trapped each week and then we rotated to new sites, thereby covering all six sites over a three-week period. We noted the sex, age, and reproductive status of small mammals and marked them with non-toxic permanent marker in order to distinguish novel and recaptured animals. Traps were washed at the end of each trapping session in a 10 percent bleach solution.

Preble's meadow jumping mice were weighed and marked individually with Passive Integrated Transponder (PIT) tags, inserted under the skin on the back (Schooley et al. 1993). We purchased Destron-Fearing PIT tags with a scanner exciter frequency of 125 kHz and a mini portable reader from Biomark of Boise, Idaho. Mark-recapture was used on the individually PIT-tagged Preble's meadow jumping mice to develop population estimates for this taxon.

There has been a recent and rapid shift in the recommended approach to statistical modeling and reporting of research results that de-emphasizes null hypotheses and significance tests (Cherry 1998, Guthery et al. 2001, Johnson 1999) in favor of comparing multiple biologically plausible models and evaluating the support for each provided by the available data (Anderson et al. 2000, 2001). Consequently, we adopted the framework of information theoretic statistical analysis based on Akaiki's Information Criteria (Burnham and Anderson 1998) and reported estimates of effect sizes, their precision, and measures of the strength of evidence supporting alternative models. We also commented separately on the biological importance of an effect (based on its estimated size) and its precision, which was a consequence of sample size relative to environmental noise.

We conducted analyses of variance (ANOVAs) on plant species richness, plant cover and visitor use data by using PROC GLM (SAS Institute 1989). We fitted six alternative ANOVA models to the plant cover observations with combinations of site, trail, and grazing explanatory covariates. Plant species richness observations were modeled with six alternative multinomial models using combinations of site, trail, and grazing covariates and fitted by maximum likelihood. Visitor use data were modeled by using ANOVA with the addition of an explanatory covariate for time-of-day (morning, afternoon, and evening), which resulted in 12 alternative models.

Population size for Preble's meadow jumping mouse was estimated by using the Robust Design model (Kendall et al. 1997) for capture-recapture studies (Pollock et al. 1990, Seber 1992) in program MARK (White and Burnham 1999). In the most general model considered, we allowed for differences in capture probabilities between sex, time (week and trapping period), and site (trapping grid). Because of the large number of trapping occasions and small population sizes, estimation of separate capture probabilities for each trapping occasion (night) was not possible. Instead, we considered separate capture probabilities by week and session. The possibility that recapture probability differed from capture probability also was examined. As in all closed population studies, permanent migration out of the study areas cannot be distinguished from mortality, only temporary migrations can be measured.

Because all six sites were trapped in a trapping period that lasted three weeks, it was possible for an individual trapped on one grid to be recaptured on an adjacent grid during a subsequent week of trapping. To account for these cross-week trapping events, separate capture probabilities were considered for each pair of trapping grids for which such movements were observed. Thus, an individual was assigned to a home grid based on the most frequent captures, but was assumed to have some reduced probability of capture on adjacent grids. These cross-grid capture probabilities were estimated for each pair of grids with observed exchanges of individuals.

Preble's meadow jumping mouse in Colorado generally is restricted to habitats along streams; consequently, the best population estimates are standardized to linear stream distance. Population size estimates for each site were converted to linear population density estimates of the number of Preble's meadow jumping mice per km of stream reach. We assumed that trapping grids on one side of South Boulder Creek measured the population size along that side of the creek only, and therefore were doubled to yield linear density estimates comparable to other locations where streams are crossed freely by Preble's meadow jumping mouse. The width and swiftness of the creek and the low estimates of trapping probability justify this assumption. Because there are no natural or artificial boundaries on a given side of the creek to ensure geographic closure of populations on each trapping grid (an important assumption of the analytical techniques used), the linear population estimates included individuals drawn from outside the boundaries of the grid. To adjust for this, we used a residency correction factor based on other researchers' radio-telemetry data collected on Preble's meadow jumping mice in nearby Jefferson and Douglas counties (White and Shenk 2001). For our 64 m grids, corrected density was 43.5% of the direct linear population density estimates for Preble's meadow jumping mouse.

We conducted analyses of variance (ANOVAs) on species richness, relative abundance, and species diversity indices, and on the linear population density estimates for Preble's meadow jumping mouse by using PROC GLM (SAS Institute 1989). We defined species richness as the number of species captured on a particular trapping grid during a trapping session. Relative abundance was calculated as the number of individuals captured per 100 trap nights. Species diversity was calculated by the Shannon-Wiener diversity index (H') as described in Zar (1996). Alternative models for each of these response variables were examined that included several temporal and spatial effects likely to occur in our study. Temporal effects considered included categorical variables for season (June versus August), year (1997, 1998, or 1999), and independent effects of each trapping period (periods 1 to 6). We also considered a linear time trend as a continuous covariate. Spatial effects examined were categorical covariates for trail use (trail versus no trail), cattle grazing (present or absent), and individual site (sites 1 to 6). For Preble's meadow jumping mouse linear density estimates, models with sex effect also were considered. We constructed models for individual temporal and spatial covariates separately and for combinations of two or three of these that we considered biologically reasonable. All candidate models had from 1 to 13 degrees of freedom (including intercepts and error terms). In all, we fitted 35 models for the small mammal indices and 70 models in the ANOVA of Preble's meadow jumping mouse density. Analyses for species richness, relative abundance, and species diversity were repeated both with and without inclusion of two exotic species (i.e., the house mouse (*Mus musculus*) and the Norway rat (*Rattus norvegicus*)); however, results were negligibly different, so we reported only results with all species included.

In addition, using the differences between paired trapping sites on opposite sides of the creek, we conducted repeated measures ANOVA to examine trail effects. This pairing of sites was done to increase the precision of our estimated differences. Our results (not shown) confirmed this assumption. Similar pairing of grazed and ungrazed sites was not possible because grazing was always either present or absent on both sides of the creek. Models with the same combinations of explanatory variables (except trail effect) were fitted in the repeated measures ANOVA as in the fully randomized ANOVA models (previous paragraph). In the repeated measures (trail effects) ANOVA, we considered 22 models for species diversity, species richness, and relative abundance indices; with the addition of sex effect, we evaluated 43 models for Preble's meadow jumping mouse density.

For all models of small mammals, vegetation, and visitor use, we computed Akaiki's Information Criteria with small sample bias correction (AIC_c) and used these values to compute relative model weights (which sum to 100%) indicating the relative strength of support in the data for each model (Burnham and Anderson 1998). Relative model weights for all models containing a particular covariate (e.g., sex effect) were summed to assess the total support in the data for the presence of that effect; for example, if sex was a covariate in several models with combined weights of 60%, then the data provide 1.5 = 60/(100-60) times as much support for the alternative of a sex effect versus no effect. Using the ESTIMATE statement in PROC GLM (SAS Institute 1989), we estimated the quantities of interest for each model. Reported estimates and effect sizes (model coefficients) and their standard errors were computed by weighting the individual estimates from each model by these same relative weights. This procedure incorporated an additional component of variance arising from the uncertainty associated with model selection, which is often ignored in traditional statistical analyses.

RESULTS

Overall, estimated plant cover (calculated as the AIC_c model-weighted estimates from all models) was 16.4% (± 4.1) for trees, 7.1% (± 1.8) for shrubs, 64.9%

(± 2.7) for grass, and 22.1% (± 4.0) for forbs. Estimated plant cover for trees, shrubs, grasses, and forbs was generally similar between the trail, non-trail, grazed, and non-grazed sites. We found low support (sum of model weight) for differences between trail and non-trail sites on trees (22.4%), shrubs (25.3%), grass (33.1%), and forbs (40.6%). Support for differences between grazed and non-grazed sites was low for trees (13.2%), shrubs (11.7%), and grass (17.5%), but moderately higher for forbs (64.1%). We estimated this difference as 4.5% (± 4.0) higher forb cover on the grazed sites, however precision is too low to draw definitive conclusions.

Tree species richness was low, 0 to 1 species at 89.8% (± 2.9) of sites and fewer than four species at all sites. Shrub species richness was low, 0 to 2 species at 96.3% (± 1.8) of sites and fewer than six species at all sites. Grass species richness was intermediate, with 2 to 3 species at 53.7% (± 4.8) of sites and 5 to 6 species at 38.9% (± 4.7) of sites. Intermediate species richness (4 to 7 species) of forbs occurred at 55.1% (± 5.0) of sites and only 5.1% (± 2.2) of sites were more rich. All categories of plant species richness were very similar between the trail and non-trail sides in both the grazed and non-grazed segments of the study site. Model weights supporting each of these differences for trees, shrubs, grass, and forbs ranged from 15.9 to 37.8%. The most supported effect was a possibly higher proportion of trail sites with low species richness (0 to 4 species); this difference was estimated to be 16.1% (± 9.7).

We observed 3958 human visitors at an estimated rate of 29.1 ± 8.3 people/hr and 619 dogs at 6 ± 3.6 dogs/hr. We observed 3944 people and 604 dogs on the trail side and only 14 people and 15 dogs on the non-trail side. Model weights strongly supported differences between trail and non-trail sites for people (100%) and dogs (83.2%). Estimated trail effect was an additional 37.4 ± 3.2 people/hr and 5.6 ± 1.1 dogs/hr. Almost all of the non-trail visitor use occurred at Site 3, where an unofficial trail connecting a nearby neighborhood to the official trail has developed over the years. Our data provided nearly as strong support for differences between the northern and southern segments for people (83.2%) and dogs (94.9%). We estimated that southern sites had 11.0 ± 5.1 fewer people/hr and 5.7 ± 1.4 fewer dogs/hr.

We captured 1132 individual small mammals of 10 identified species over the three years during 14,724 trap nights on the 12 grids (Table 1). One hispid pocket mouse (*Chaetodipus hispidus*) was captured on the non-trail side of the grazed segment; two Mexican woodrats (*Neotoma mexicana*) were captured only on the non-trail side on both grazed and non-grazed segments; a Norway rat was captured only on the trail side in the grazed segment; and four thirteen-lined ground squirrels (*Spermophilus tridecemlineatus*) were captured on the trail and non-trails sides on the non-grazed segment. Otherwise, all species were found on both trail and non-trail sides of the creek and on the grazed and non-grazed segments. More individuals were captured on the non-trail side (593 versus 539). Preble's meadow jumping mice comprised 13.6% of the individual small mammals captured. No

Table 1. Individual small mammal captures and recaptures by species in descending order of frequency from 14,724 trap-nights on South Boulder Creek, Colorado, 1997 - 1999.

Species	Captures	Recaptures
Deer Mouse	469	358
Meadow Vole	336	61
Preble's Meadow Jumping Mouse	154	63
Prairie Vole	108	20
Unidentified Vole	32	0
House Mouse	15	2
Western Harvest Mouse	10	0
Thirteen-lined Ground Squirrel	4	0
Mexican Wood Rat	2	1
Hispid Pocket Mouse	1	1
Norway Rat	1	0
Total	1132	506

Preble's meadow jumping mice were caught in July of 1997, an apparent anomaly, so we analyzed linear population density data from the remaining five periods. Data for the unpaired ANOVA consisted of 102 observations for Preble's meadow jumping mouse (one for each sex, trapping grid, and period after excluding the July 1997 period) and 54 observations for the indices (species richness, relative abundance, and species diversity) where data were not tallied separately by sex. Repeated measures ANOVA's had 51 observations for Preble's meadow jumping mouse density and 26 for the indices.

Overall, estimated species richness (calculated as the AIC_c model-weighted estimates from all models) was 4.0 (± 0.18) species. Temporal effects dominated the variation in species richness and these consisted mostly of variation by month and year; for example, 69% of the model weight supported variation in species richness due to year (Table 2), thus evidence for a year effect is favored 2.2:1 [69/(100-69)] over evidence against it. There was no support for a linear time trend. Evidence for spatial variation was weak. The most strongly supported spatial effect was the trail effect, however, models with no trail effect were favored 2.7:1 over those with an effect (Table 2). The estimated net trail effect on species richness was low

Table 2. Strength of evidence for effects of explanatory variables on small mammal indices and Preble's meadow jumping mouse linear density based on AIC_c model weights for several alternative ANOVA models on South Boulder Creek, Boulder, Colorado, 1997 - 1999.

	Percent Strength of Evidence for Effect[1]						
	Time[5]	Month	Year	Period	Trail	Site	Grazing
Species Richness[2]	1	58	69	13	27	21	18
Species Diversity[3]	2	42	88	9	0	100	0
Relative Abundance[4]							
All Species	0	0	0	100	8	27	37
Prairie Vole	0	0	0	100	10	1	8
Meadow Vole	0	0	0	100	28	1	9
Deer Mouse	21	77	6	0	30	2	98
Preble's Meadow Jumping Mouse	15	10	5	42	27	2	18
Preble's Meadow Jumping Mouse Linear Density	55	62	14	3	15	36	9

[1] Sum of AICc model weights for those models containing a given effect. Individual values range from 0 to 100%, reflecting the percent weight of effect of the particular variable in the overall model. Percentages do not add to 100 across rows because many models contain multiple effects, therefore their weights are included in several sums.
[2] Number of species captured.
[3] Shannon-Wiener index.
[4] Individuals per 100 trap nights.
[5] Linear time trends, i.e., linear changes over the 25 months of the study.

(Tables 2 and 3) and of little biological importance even if precision were adequate to confirm this effect. Model weights (Table 2) and estimated values (Table 4) provided no evidence of a grazing effect on species richness.

Overall, the model-weighted diversity index estimate was 0.4 (± 0.02). Both temporal and spatial effects were supported strongly by the data, with annual variation dominating the temporal component and individual site variation dominating the spatial component (Table 2). The data provided essentially no support for trail or grazing effects on species diversity (Tables 2 to 4).

The model-weighted estimate of overall relative abundance of all species

Table 3. Estimates of small mammal indices and Preble's meadow jumping mouse linear population density at sites with and without trails and net effects of trails based on AIC_c-weighted estimates from multiple ANOVA models for South Boulder Creek, Boulder, Colorado, 1997 - 1999.

	Value for Grids with Trail			Value for Grids without Trail			Trail Effect (Paired[1])			Relative Effect
	Estimate	SE	CV	Estimate	SE	CV	Estimate	SE	CV	
Species Richness[2]	4.0	0.2	5%	4.1	0.21	5%	-0.24	0.17	-72%	-6%
Species Diversity[3]	0.41	0.02	5%	0.41	0.021	5%	0.00	0.02	-570%	-1%
Relative Abundance[4]										
All Species	10.9	0.8	7%	10.9	0.78	7%	-0.25	0.85	-335%	-2%
Prairie Vole	1.4	0.1	10%	1.4	0.13	9%	-0.22	0.17	-81%	-15%
Meadow Vole	3.8	0.3	8%	3.6	0.44	12%	0.43	0.34	79%	12%
Deer Mouse	3.1	0.3	9%	3.3	0.29	9%	-0.32	0.35	-107%	-10%
Preble's Meadow Jumping Mouse	1.0	0.2	18%	1.1	0.19	17%	-0.18	0.16	-88%	-17%

Table 3, continued.

	Value for Grids with Trail			Value for Grids without Trail			Trail Effect (Paired[1])			Relative Effect
	Estimate	SE	CV	Estimate	SE	CV	Estimate	SE	CV	
Preble's Meadow Jumping Mouse Density	37.2	5.6	15%	38.8	5.9	15%	-11.6	9.5	-81%	-31%
Male							-17.1	12.9	-76%	-42%
Female							-6.2	11.3	-183%	-18%

[1] Paired analysis treats trapping grids on opposite sides of the creek as repeated measure.

[2] Number of species captured.

[3] Shannon-Wiener index.

[4] Individuals per 100 trap nights.

Table 4. Estimates of small mammal indices and Preble's meadow jumping mouse linear population density at sites with and without grazing by cattle and net effects of grazing based on AIC_c-weighted estimates from multiple ANOVA models for South Boulder Creek, Boulder, Colorado, 1997 - 1999.

	Value for Grids with Grazing			Value for Grids without Grazing			Grazing Effect (Unpaired[1])			
	Estimate	SE	CV	Estimate	SE	CV	Estimate	SE	CV	Relative Effect
Species Richness[2]	4.1	0.20	5%	4.0	0.18	4%	0.04	0.096	222%	1%
Species Diversity[3]	0.41	0.021	5%	0.41	0.021	5%	0.00	0.000	-252%	0%
Relative Abundance[4]										
All Species	11.5	1.11	10%	10.3	1.21	12%	1.22	1.687	138%	11%
Prairie Vole	1.4	0.14	10%	1.4	0.13	9%	-0.01	0.030	-305%	-1%
Meadow Vole	3.7	0.32	9%	3.7	0.29	8%	-0.03	0.081	-243%	-1%
Deer Mouse	4.2	0.38	9%	2.2	0.33	15%	2.00	0.508	25%	63%
Preble's Meadow Jumping Mouse	1.1	0.18	17%	1.1	0.17	15%	-0.03	0.079	-306%	-2%

Table 4, continued.

	Value for Grids with Grazing			Value for Grids without Grazing			Grazing Effect (Unpaired[1])			Relative Effect
	Estimate	SE	CV	Estimate	SE	CV	Estimate	SE	CV	
Preble's Meadow Jumping Mouse Density	37.9	5.53	15%	38.1	5.46	14%	-0.17	0.897	-519%	0%

[1] Trap grids treated as simple random sample.
[2] Number of species captured.
[3] Shannon-Wiener index.
[4] Individuals per 100 trap nights.

combined was 10.9 (± 0.76) individuals/100 trap-nights. Temporal variation on overall relative abundance was explained exclusively by variation between periods, with no support for time trend, year, or month effects (Table 2). Only weak evidence was found to support trail and grazing effects, which were disfavored relative to models without these effects by margins of 2.7:1 and 1.7:1, respectively (Table 2). The estimated trail effect was small in both biological terms (2% relative abundance reduction for all species) and relative to measurement error (Table 3). Estimated grazing effect was a biologically modest increase in overall relative abundance (11%), which was too imprecise (95% CI = -2.1, 4.5 individuals) to draw definitive conclusions (Table 4).

We also examined relative abundance data for the four most common species captured. Prairie voles constituted an estimated 1.4 (± 0.13), meadow voles (*Microtus pennsylvanicus*) 3.7 (± 0.30), deer mice 3.2 (± 0.25), and jumping mice 1.1 (± 0.16) individuals/100 trap-nights. Both species of voles exhibited very strong temporal variation between sessions, which were not attributable to annual, monthly, or linear time trend effects. Spatial effects were supported only weakly, with a trail effect in meadow voles most prominent, but still disfavored 2.6:1 by models with no effect (Table 2). The estimated trail effects were in opposite directions for the two species of voles and of only modest size (less than 15%) and low precision (Table 3). In contrast, deer mouse relative abundance showed strong evidence for monthly (June versus August) fluctuations and a positive grazing effect, and weak support for a trail effect. The estimated positive effect of grazing on deer mouse relative abundance (Table 4) was statistically unambiguous (95% CI = 1.0, 3.0) and more than explains the increase in overall species relative abundan associated with grazing. Finally, Preble's meadow jumping mouse relative abundance exhibited no strongly supported spatial or temporal variation, although variation effect between periods was supported the most strongly followed by the trail effect (Table 2). In the more precise repeated measures analysis of this trail effect on jumping mouse relative abundance, we estimated a moderately major (-17%) negative response to the trail, but with inadequate precision (95% CI = -0.49, 0.13 individuals/100 trap nights) to support definitive conclusions. Although this confidence interval encompasses the possibility of no trail effect, at its lower end it also encompasses the possibility of a 45% population reduction (Table 3).

Mark-recapture analysis of Preble's meadow jumping mouse populations revealed high capture probabilities for this species (0.353 ± 0.020). We also found evidence of movement between trapping grids with capture probabilities in one case as high as 0.231 (± 0.058) for Preble's meadow jumping mice caught at a second location. Estimates of temporary emigration from trapping grids were high for both summer 0.502 (± 0.378) and winter 0.951 (± 0.033). These results indicated that site fidelity was very low.

We estimated that overall linear density of Preble's meadow jumping mouse was 38.0 (± 5.4) individuals/km. There was moderate support for monthly and linear time trend effects, and weak support for a site effect (Table 2). We also found some support for a sex effect (50% of model weight, not shown in table). We estimated an overall linear density of 40.9 (± 6.6) male individuals/km, 35.1 (± 6.6) females; and 42.6 (± 7.4) individuals/km for both sexes in June and 33.3 (± 6.8) in August. We also estimated a linear decline over the duration of the study of -0.59 individuals/km/month (± 0.67); nearly 40% decline in the starting population over the 25 month duration of our study. Estimated trail effect on Preble's meadow jumping mouse population density was -31% (Table 3). However, measurement error was substantial so strong inference about the presence of a trail effect was not possible without additional data. We also noted that the estimated trail effect was 2.8 times higher for males than females. Grazing had no measurable effect on linear densities of Preble's meadow jumping mouse (Table 4).

DISCUSSION

The patterns of the vegetation between the trail and non-trail side of the creek were similar, such that differences in small mammal indices or Preble's meadow jumping mouse population densities did not appear to be confounded by the vegetation. The vast majority of visitor use was on the trail side of the creek and the non-trail side experienced almost no human traffic, which confirmed the anticipated consequence of the trail. Evidence for a possible negative impact of trails on small mammal richness, relative abundance, and species diversity was weak and only suggestive. Evidence of a trail effect on Preble's meadow jumping mouse population estimates was slightly stronger, and although inconclusive, its magnitude as suggested by our data was potentially high enough to be of concern for this threatened species.

Large temporal and spatial variation, unrelated to the effects we were studying, resulted in low precision of many of the estimated effects. Despite intensive trapping twice per year over three years, our results regarding trail impacts are merely suggestive and not conclusive. Pairing of sites on opposite sides of the creek helped reduce variation and increase precision, as anticipated. Nevertheless, we found that few quantities of interest were easily measurable against the background noise. Using the trail effect on Preble's meadow jumping mouse density as an example, we computed that 521 pairs of trapping grids would be required to obtain even a 50% coefficient of variation for an estimate of a 20% reduction in linear density. This represents over 10 times the amount of effort expended on our study and still resulted in only 64% power to detect a statistically significant effect ($\propto = 0.05$). For similar precision on a 30% population reduction, 232 pairs would be needed.

The variation in density estimates of Preble's meadow jumping mouse across sites and periods might be due to biological phenomenon such as patchiness of food resources in space and time, social behavior, degree of territoriality, size of home range, pre- and post-hibernation movement, or to factors related to sampling. We do not know to what extent any of these factors played a role. Our estimate of a linear decline over time in Preble's meadow jumping mouse population estimates was of a large enough magnitude to be notable, although too imprecise to draw definitive conclusions about the size or even presence of a decline. This decline might reflect a natural cycle or it could simply be a spurious result due to inadequate data for this highly variable population. Meadow jumping mouse populations were thought to fluctuate widely (Tester et al. 1993) and individuals disappeared and reappeared on trapping areas in another study (Blair 1940). However, in a related study, we determined that sampling error explained much of the random temporal and spatial variation in our density estimates, due in large part to the correction factor for open boundary effects (unpublished data). Our cross-grid capture probabilities and emigration rate estimates confirmed that lack of geographic closure was a severe problem. Thus, future studies should strive to reduce error associated with a correction factor by running trapping grids or transects that extend a greater length (i.e. greater than 64 m) along the riparian corridor, thereby reducing the error associated with lack of geographic closure. This design might also help to reduce the between-site spatial variation by effectively averaging over more habitat types.

The deer mouse was more abundant in the southern segment of the creek. This observation might be explained by management differences that occurred in grazing patterns and recreational use between the northern and southern segments. In the northern segment, fencing protected the riparian corridor from grazing, whereas the cattle were not fenced out in the southern segment. The northern segment was grazed from December to February at the rate of 1 to 2 Animal Unit Months per 0.4 ha, and the southern segment was grazed from December to mid-May at the rate of 1 to 2 Animal Unit Months per 0.4 ha. The amount of forage removed was the same in both areas but the southern segment had more spring grazing, was grazed longer, and the grazing occurred in the riparian corridor. The northern segment experienced a higher trail usage and dogs were allowed (they were not allowed on the southern segment). Development was closer in the northern segment, whereas the southern segment had a greater extent of agricultural use on adjacent lands. Habitat disturbances benefitted the quintessential generalist deer mouse (Armstrong 1977). Adapted to exploit disturbances, the deer mouse was tolerant of the reduction in vegetation associated with grazing, and studies have found that the deer mouse was more abundant under grazed conditions (Lusby et al. 1971, Moulton 1981, Moulton et al. 1981, Schulz and Leininger 1991). Although a number of differences in management occurred between the northern and southern segments, we suggest that grazing might have a greater effect on small mammal community composition than does the trail. From the trail effect analysis, we know that trails do not have an effect on species richness, species diversity, or relative abundance of small mammals. Proximity to urbanization, however, has been shown to have a negative effect on rodent abundance, including deer mice (Bock et al. 2002).

The limited grazing (December through February) on the study area appeared not to have an impact on the linear population estimates of Preble's meadow jumping mouse. The western jumping mouse also was captured in both grazed and ungrazed habitats in Nevada (Medin and Clary 1989). Habitat requirements of Preble's meadow jumping mouse might be more similar to those of the meadow vole than the deer mouse, as suggested by evidence that the meadow vole can exclude the meadow jumping mouse (Boonstra and Hoyle 1986). The ecological relationships between the deer mouse and meadow jumping mouse are not known, although the deer mouse (and meadow vole) is common on small mammal trapping grids where the meadow jumping mouse is found in Colorado.

The presence of dogs and higher visitor use in the northern segment might be counterbalanced by the lack of grazing in the riparian corridor. Additionally, along the northern segment, the trail weaved in and out from the creek and was bordered by a rail fence for portions of its length. Along its entire length in the north, the trail was fenced (10 strand high-tensile smooth wire) from the adjacent wet meadows. In each case the fencing discouraged movement of people and pets from the trail. Although these factors were confounded, and precise allocation of impacts and benefits was not possible, we do suggest that particular habitat factors as well as trail management and design can go a long way to offset disturbance.

The potential negative effect on Preble's meadow jumping mouse of trails might be offset with well-developed vegetation. As new trails are developed, there is much potential for variation in alignment, width, surfacing, maintenance, adjacent plant communities, geomorphology, and landscape context. We suggest that care should be taken in the alignment of trails to ensure that well-developed riparian vegetation can be maintained to the greatest extent possible in the vicinity of trails close to creeks. Furthermore, it might be beneficial to jumping mice and other wildlife to weave trails out of the riparian corridor as much as possible.

ACKNOWLEDGMENTS

We wish to thank Cary Richardson, Todd Kipfer, Lauren Whittemore, Erik Butler, and others of the City of Boulder Open Space Department for their assistance with logistics, discussions, and GIS support. The Denver Museum of Nature and Science kindly prepared the final figure. Collin Ahrens, Lauren Whittemore, Alison Deans, Melissa Ryder, Barbara Spagnuolo, Suki Camena, Brian and Talus McMahon, Julian Turner, Stoney Petit, Nan Hampton, Christine Ruggles, Kara Csibrik, and numerous other interns and volunteers were a great help in the field. Thanks to Cheri Jones, Ron Beane, Tanya Shenk, Carl Bock, Rob Schorr, and Mark Bakeman for discussions about the design of the project. Marc Bekoff, Mark Bakeman, and Dave Armstrong kindly reviewed an earlier draft of the manuscript. A very special thank you to our funding sources: the Greenways Program of the City of Boulder Transportation Department, City of Boulder Open Space, Environmental Protection Agency (Region 8), University of Colorado Undergraduate Research Opportunity Program, and the Fish and Wildlife Foundation.

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Received: 26 March 2001 *Accepted: 13 July 2002*