

EFFECTS OF WILDFIRE SEVERITY ON SMALL MAMMALS IN NORTHERN ARIZONA PONDEROSA PINE FORESTS

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ABSTRACT

We examined effects of a varied-severity wildfire on the community structure of small mammals and populations of the 2 most abundant species, the deer mouse (*Peromyscus maniculatus*) and the gray-collared chipmunk (*Tamias cinereicollis*), in northern Arizona ponderosa pine (*Pinus ponderosa*) forests. We examined 2 fire severities and compared them to unburned controls. The average number of species captured was similar among the 3 plot types: 2.0 on high-severity plots, 3.0 on moderate-severity plots, and 3.5 on control plots. However, the species composition differed among these types. Specifically, gray-collared chipmunks were not captured on high-severity plots in the first year following the fire. We found no statistically significant difference among treatments for gray-collared chipmunk densities even though they were not captured on high-severity plots ($P = 0.074$). Deer mouse densities on high-severity fire plots were greater than on control plots ($P = 0.028$) and were marginally greater than on moderate-severity plots ($P = 0.051$). We did not find a significant difference between moderate and control plots ($P = 0.25$). Deer mouse densities were strongly correlated with forb ($P = 0.002$) and shrub ($P = 0.038$) cover in a stepwise linear regression (adjusted $R^2 = 0.67$). Based on these results, we suggest that a consideration of fire severity in a structural sense does not provide a clear picture of the impacts of wildfires or prescribed fires on the small mammal community. We propose that the composition of the postfire understory plant community must also be considered.

keywords: Arizona, deer mouse, fire severity, golden-mantled ground squirrel, gray-collared chipmunk, habitat, *Peromyscus maniculatus*, *Pinus ponderosa*, ponderosa pine, *Spermophilus lateralis*, *Tamias cinereicollis*, wildfire effects.

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INTRODUCTION

Many ponderosa pine ecosystems in northern Arizona have undergone drastic changes over the past century. Savage and Swetnam (1990) suggested that intensive grazing in the latter part of the nineteenth century may have disrupted the natural fire regimes by reducing the herbaceous layer. They noted that fire frequency had declined sharply in the Chuska Mountains of northeastern Arizona by 1829. This decline corresponded with major increases in sheep herds pastured in these mountains by the Navajo tribe. Organized fire suppression began several decades later. Average pre-grazing and presettlement fire intervals reported for ponderosa pine forests in north-central Arizona ranged from 2–3 years (Dietrich 1980) to nearly 12 years (Weaver 1951). Several authors have concluded that these wildfires were primarily understory fires of low to moderate severity (Weaver 1951, Cooper 1960, Covington and Moore 1994). This information, however, is a result of dendrochronological studies. These methods may not detect more severe fires because the trees, logs, and stumps from which the data are collected may be consumed in these fires. Shinneman and Baker (1997) suggested that large-scale, crown fires were a major organizing force in ponderosa pine forests in the Black Hills of South Dakota. This argument was based largely on stand structure from an extensive timber survey of the area conducted in 1899. No such

data exist for northern Arizona, and there are no estimates for the presettlement occurrence or size of high-severity wildfire in this area.

In the early decades of the twentieth century, the establishment and survival of ponderosa pine seedlings significantly increased. Reduced competition from grasses and a series of years with higher than average precipitation created nearly optimal conditions for ponderosa pine seedlings (Pearson 1950). The inception of the U.S. Department of Agriculture, Forest Service and organized fire suppression in the decades that followed contributed to the success of these seedlings (Cooper 1960, Covington and Moore 1994). These factors created the nearly continuous, closed-canopy forest that exists today (White 1985, Covington and Moore 1994). Most of these forests are now characterized by higher stem densities, greater fuel loading, and less herbaceous understory than presettlement forests (White 1985, Covington and Moore 1994). These conditions have resulted in an increased size and occurrence of stand-replacing wildfires (Covington and Moore 1994, U.S. Department of the Interior 1995:60–61).

Management and restoration efforts in northern Arizona ponderosa pine forests have focused primarily on prescribed fires and combinations of thinning and prescribed fire. However, there is little information on the effects of these treatments or wildfires on wildlife in these forests. This information would help land

managers weigh the potential effects of these treatments. In addition, information on the effects of wildfire on wildlife would be valuable in determining management strategies following these events.

Small mammals occupy an important niche in ponderosa pine ecosystems. Changes in population size and community structure can affect a variety of other ecosystem processes. Populations of predators such as the northern goshawk (*Accipiter gentilis*) (Reynolds et al. 1992) and the Mexican spotted owl (*Strix occidentalis lucida*) (Ward and Block 1995) can exhibit numerical and functional responses to fluctuations in prey populations. These changes could also affect plant succession by altering dispersal pathways for seeds and plant propagules (Vander Wall 1990:179) or mycorrhizal fungi (Maser et al. 1978). Information concerning the effects of fire severity in ponderosa pine forests on small mammals could provide insight into the process of forest succession.

We present preliminary results from a study in which we address questions concerning the effects of wildfire severity on the small mammal community and populations of deer mice and gray-collared chipmunks in northern Arizona ponderosa pine forests. Our initial research hypotheses are that: species richness decreases with increasing fire severity, deer mouse densities increase with increasing fire severity, and chipmunk densities decrease with increasing fire severity. These results reflect the first year of data collection from a continuing study.

STUDY AREA

In May and June 1996, 2 large wildfires burned on the Peaks Ranger District of the Coconino National Forest, in north-central Arizona. The Horseshoe fire was the first of these, covering approximately 3,500 hectares of primarily ponderosa pine forest. The Hochderffer fire began about a month later and burned >6,600 hectares adjoining the Horseshoe fire. The predominant vegetation type in this area was also ponderosa pine. These 2 fires created a complex mosaic characterized by a gradient of fire severity ranging from high-severity crown fires to relatively low-severity understory fires. These fire severities also occurred across a gradient of spatial scales within the study area, ranging from <1 hectare to several hundred hectares in size. We define fire severity as the response of an ecosystem to fire (Simard 1991), and focus on responses of ecosystem flora and fauna. Fire severity reflects both the amount of heat released by the fire and fire behavior (DeBano et al. 1998:61).

The overstory of all plots examined on the Horseshoe-Hochderffer fire complex was exclusively second-growth ponderosa pine. Understory tree species were also primarily ponderosa pine with an occasional one-seed juniper (*Juniperus monosperma*), pinyon pine (*Pinus edulis*), limber pine (*Pinus flexilis*), or trembling aspen (*Populus tremuloides*). Tree stem densities on the plots sampled ranged from 215 stems per hectare on a moderate-severity plot to 1,210 per hect-

are on a high-severity plot. The shrub and herbaceous layer of the controls and moderate-severity plots also included Mogollon ceanothus (*Ceanothus fendleri*), wild rose (*Rosa woodsii*), Oregon grape (*Mahonia repens*), Arizona fescue (*Festuca arizonica*), mountain muhly (*Muhlenbergia montana*), squirrel tail (*Sitanion hystrix*), and mullen (*Verbascum thapsis*). On the high-severity plots the herbaceous layer was dominated by annual forbs such as lamb's quarter (*Chenopodium* spp.), lupine (*Lupinus* spp.), locoweed (*Oxytropis lambertii*), and deer vetch (*Lotus wrightii*). Shrubs such as Mogollon ceanothus, wild rose, and mountain mahogany were also present on the high-severity plots.

METHODS

Since the focus of this study is wildfires, the collection of pretreatment data on stand composition, vegetation, or the small mammal community was impossible. In addition, existing Forest Service databases contain little information that is germane to this study. These databases focus on relatively coarse spatial scales based on subjective stand boundaries, which are often assigned as a result of previous fires or timber harvests. Because they are directed primarily at timber management, these databases contain little or no information on prefire understory vegetation and no information on the prefire small mammal community. Therefore, we were limited to a comparison of post-wildfire areas with unburned areas.

We focused on residual forest structure to assign areas to 2 fire-severity categories. We characterized high-severity areas as crown fires that killed all or most of the trees at the time of the fire or shortly thereafter. In addition, most of the canopy, litter, and coarse woody debris was consumed, as was much of the organic material in the upper layers of the soil. Moderate-severity fires left most of the trees and upper canopy intact but killed most of the shrubs and tree saplings. Although the season of occurrence was different from that of most prescribed fires in ponderosa pine forests, the moderate-severity plots are comparable to prescribed fires in terms of fuel consumption, tree mortality, and resulting forest structure.

We compared small mammal communities among these 2 wildfire severities and unburned controls. We selected 4 high-severity plots based on size and lack of planned salvage logging. We chose 4 moderate-severity plots based on size and location within the complex in relation to the high-severity plots. For comparison, we selected 4 control plots around the Horseshoe-Hochderffer fire complex that were not burned during the 1996 fires and showed no signs of recent wildfire or prescribed fire.

Small Mammal Trapping

We sampled small mammal populations during 4 trapping sessions between 23 June–12 August 1997. During each trapping session we selected and sampled 3 plots: 1 high-severity, 1 moderate, and 1 control. Each of the 12 plots was trapped once during the sum-

mer of 1997. On each plot, we established a 4-hectare trapping grid, which was the largest size that would fit within patches and maintain an approximate 100-meter minimum buffer. Traps were placed at stations with 20-meter spacing. One large ($76 \times 89 \times 228$ millimeters) Sherman live trap was placed at each trap station. In addition, one extra large ($101 \times 114 \times 381$ millimeters) Sherman live trap was placed at 36 trap stations with 40-meter spacing within the grid for larger species such as golden-mantled ground squirrels (*Spermophilus lateralis*). Each grid contained 157 traps.

We pre-baited the traps for 1 day prior to trapping session. The doors were left closed with a small amount of bait in front of the door to allow the animals to adjust to their presence. After 24 hours, we opened the doors and inserted a small amount of cotton batting and bait. We then checked the traps twice daily for 5–7 days. In an effort to reduce potential mortality, we concluded each trapping session when recaptured animals accounted for 90% of total captures after ≥ 5 days. There were 10,277 trap nights and 8,409 trap days during the study for all 12 plots. On control plots there were 3,349 trap nights (751–1,063 per plot) and 2,733 trap days (611–881 per plot). On the moderate-severity plots there were 3,407 trap nights (747–973 per plot) and 2,810 trap days (576–838 per plot). On high-severity plots there were 3,521 trap nights (757–1,072 per plot) and 2,866 trap days (608–863 per plot).

For each small mammal captured we attached a #1 Monel tag with a unique number in each ear for identification. We then recorded the animal number, location of capture, species, sex, age according to pelage condition, and reproductive status (adapted from Linsdale and Tevis 1951).

Habitat Data

We collected data on habitat variables from randomly selected cells from each grid. A cell was a 20×20 -meter area delineated by 4 adjacent trapping stations. From a sample of 20 cells per grid, we recorded the species and diameter at breast height (DBH) for all snags and trees ≥ 2 meters in height. In addition we recorded the length and diameter in centimeters at midpoint of all coarse woody debris ≥ 2 meters in length and ≥ 10 centimeters in diameter.

We sampled understory vegetation using a method adapted from Daubenmire (1959). In a subsample of 4 cells per grid we established 2 parallel, randomly located 12-meter transects, 5 meters apart. At 1-meter intervals along each transect we placed a 0.2-square meter quadrat; 24 quadrats per grid cell were sampled. In each quadrat, we recorded the percent canopy cover of each species of grass, forb, or shrub and the percent cover of litter, bare soil, rock, and plant basal area. In addition, we recorded the vegetation height and litter depth at the center of each quadrat. For these analyses, we used life-form categories such as forbs, graminoids, and shrubs.

Table 1. Number of small mammal individuals captured from 23 June–12 August 1997 on high-severity and moderate-severity wildfire plots, and unburned control plots from the Horseshoe-Hochderffer fire complex, Coconino County, AZ.

Species	Plot Type			Total
	High	Moderate	Control	
Deer mice	119	83	51	253
Gray-collared chipmunks	0	9	25	34
Golden-mantled ground squirrels	1	3	5	9
Silky pocket mice	2	4	1	7
Mexican voles	1	5	0	6
Mexican woodrats	2	0	1	3
Brush mice	0	0	1	1
Total Individuals	125	104	84	313

Analysis

We calculated average number of species captured for the 4 plots in each treatment. We did not calculate evenness values or other indices because they assume that all species have equal detection probabilities. This assumption does not hold for mark-recapture studies. In addition, no statistical test was performed on species richness data because this index does not represent species that cannot be captured with these methods.

We estimated densities for deer mice and gray-collared chipmunks from each plot using program CAPTURE (Otis et al. 1978, White et al. 1982, Rexstad and Burnham 1992). We compared density estimates across trapping sessions using a multiple response permutation procedure (MRPP) in the statistical package BLOSSOM (Slauson et al. 1994) at a 0.05 level of significance. This Euclidean distance-based analysis is analogous to a 1-way analysis of variance and is well suited to small sample sizes and data that are non-normally distributed. Density estimates were then compared for trapping session across all plot types using MRPP at a 0.05 level of significance to determine whether the data could be lumped over the 4 trapping sessions. We then compared these estimates for plot type across all trapping sessions also using MRPP at a 0.05 level of significance and used Bonferroni corrections to examine differences between individual plot types (Neter et al. 1996:736–738). The lack of asymptotic normality in these data prevented the exploration of a session by treatment interaction in traditional parametric analyses.

We used a stepwise multiple linear regression procedure to examine relationships between habitat variables and deer mouse densities (Neter et al. 1996:217–388). We used a square-root transformation of deer mouse densities in this analysis to meet the assumptions of multiple linear regression. Predictor variables in this analysis were the following habitat variables: percent cover of forbs, grasses, and shrubs; herbaceous vegetation height; volume of coarse woody debris; and stem density of trees.

RESULTS

During the summer of 1997 we captured 313 individuals from 7 different species (Table 1). The deer

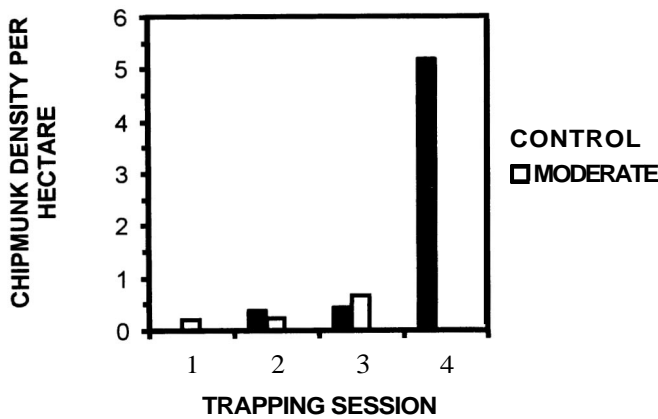


Fig. 1. Gray-collared chipmunk densities calculated using Program CAPTURE for 23 June–12 August 1997. These numbers represent high-severity and moderate-severity wildfire plots and unburned control plots from the Horseshoe-Hochderffer fire complex, Coconino County, AZ. Missing bars indicate that no chipmunks were captured on that grid.

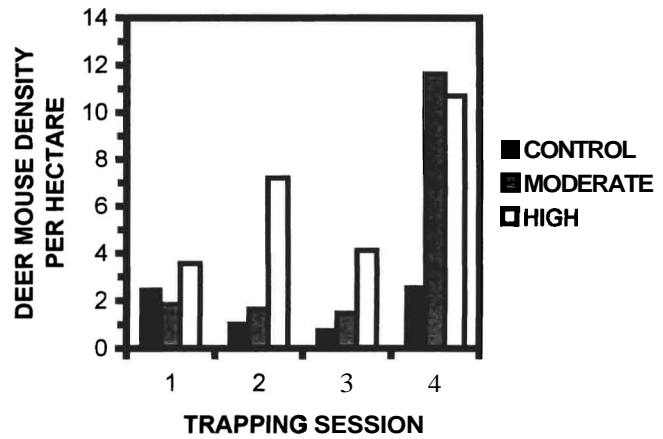


Fig. 2. Deer mouse densities calculated using Program CAPTURE for 23 June–12 August 1997. These numbers represent high-severity and moderate-severity wildfire plots and unburned control plots located from the Horseshoe-Hochderffer fire complex, Coconino County, AZ.

mouse was by far the most **common** species captured on the study area, accounting for 253 individuals. The gray-collared chipmunk was the second most **common** species with 34 individuals. Average number of species captured was **similar** between the 3 plot types: 2.5 species on all high-severity plots, 3.0 on moderate plots, and 3.5 on control plots. However, the species composition differed somewhat. For instance, gray-collared chipmunks were not captured on any of the 4 high-severity plots.

Gray-collared chipmunk densities ranged from 0–5.20 individuals per hectare for all 3 plot types (Figure 1), but were not significantly different for sampling period ($P = 0.67$) or plot type ($P = 0.074$). Deer mouse densities ranged from 0.77–11.49 individuals per hectare for the 3 treatments (Figure 2). When all 12 plots were analyzed together, we did not find significant differences for sampling period ($P = 0.15$) or plot type ($P = 0.12$).

Deer mouse densities on 3 of the 4 moderate-severity plots ranged from 1.47–1.81 individuals per hectare, whereas the estimate for the fourth plot was 11.50. Mean percent forb cover on this moderate plot was 40% compared to 6.94% for the other 3 plots. Average percent forb cover for the high-severity plots was 34%. On this basis, we excluded this moderate plot and reanalyzed the data. We did not find a significant difference for sampling period ($P = 0.78$) without this plot, but there was a significant difference for plot type ($P = 0.005$).

Based on the second analysis, we computed Bonferroni multiple comparisons for the plot type. Deer mouse densities were significantly greater on high-severity plots than control plots ($P = 0.028$). In addition, the difference between high-severity plots and moderate-severity plots approached statistical significance ($P = 0.051$). Moderate-severity plots were not significantly different from controls ($P = 0.25$).

Square-root transformed deer mouse densities were strongly correlated with percent cover of forbs ($P = 0.002$; Figure 3) and shrubs ($P = 0.038$; Figure

4). The adjusted R^2 for this regression was 0.67. However, any conclusions based on this analysis should consider that it is based on a restricted sample size.

DISCUSSION

The ponderosa pine small mammal community is not particularly diverse when compared with other vegetation types. Both higher and lower elevation systems tend to have a wider variety of species, so we did not expect major differences in species richness. In addition, species richness in this case reflects only the number of species that **might** be caught using our sampling methods and not total species present. Other species such as tassel-eared squirrels (*Sciurus aberti*) and Botta's pocket gophers (*Thomomys bottae*) were

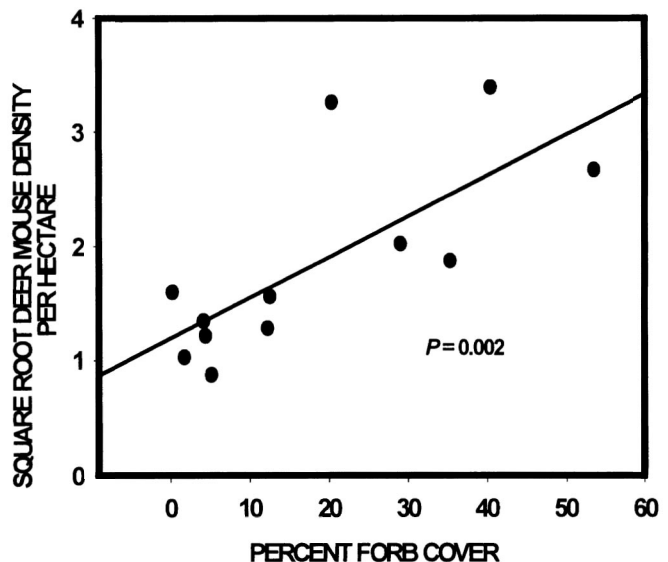


Fig. 3. Scatterplot of the relationship between percent forb cover and square root transformed deer mouse densities (per hectare) from the Horseshoe-Hochderffer fire complex, Coconino County, AZ.

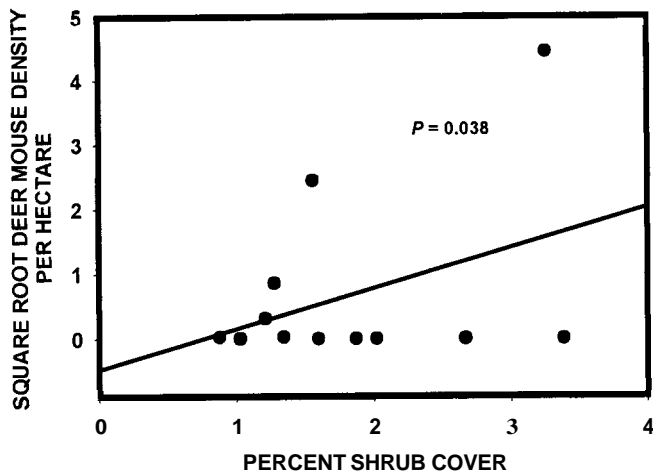


Fig. 4. Scatterplot of the relationship between percent shrub cover and square root transformed deer mouse densities (per hectare) from the Horseshoe-Hochdetffer fire complex, Coconino County, AZ.

observed on the study area but were not captured. What is useful is a comparison of components.

We did not find statistically significant differences in the densities of gray-collared chipmunks among plot types despite their absence from one of the plot types. This lack of statistical significance likely results from a combination of the small sample size and the ratio of the size of the gray-collared chipmunk home range to the size of the trapping grids. A retrospective power analysis for MRPP is unavailable at this point (B.S. Cade, U.S. Geological Survey, Mid-Continent Ecological Science Center, personal communication). We can assume that the power is low given this restricted sample size. This analysis nevertheless provides an improvement over the power of traditional statistical techniques such as analysis of variance in the case of small sample sizes or violation of normality assumptions (R. King, U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Biometrics Unit, personal communication). There are no estimates of home range size for this species based on telemetry studies in the literature. However, the mean maximum distance moved for all individuals between captures is often used as an index of the diameter of the average home range (Otis et al. 1978:67, White et al. 1982:120). In our study, this value was 122.64 meters (SE = 39.10 meters), suggesting that on average a 4-hectare trapping grid would cover only about 1.5 gray-collared chipmunk home ranges. While a 4-hectare grid might appear to be too small to estimate gray-collared chipmunk densities with any confidence, this species' absence from the high-severity plots probably has some biological explanation. Gashwiler (1959) postulated that chipmunks might be more dependent on live trees than are many other small mammals. Campbell et al. (1977) noted that gray-collared chipmunks were less common on both moderately and severely burned areas than on unburned areas from a 290-hectare wildfire in a northern Arizona ponderosa pine forest. Krefting and Ahlgren (1974) also recorded a decline in eastern chipmunk (*T. striatus*) numbers in

a jack pine (*P. banksiana*) forest after a wildfire in Minnesota.

Deer mice often increase in density following high-severity wildfire (Tester 1965, Beck and Vogl 1972, Campbell et al. 1977, Lowe et al. 1978), as was the case in this study. Campbell et al. (1977) found that deer mice were also more abundant on moderate-severity plots than on unburned plots in a northern Arizona ponderosa pine forest. Our findings suggest that there is high variability in these moderate-severity areas for both the understory plant community and the small mammal community. We suggest that this variability is a result of the amount of heat transferred to the soil by the fire. A moderate-to-heavy duff layer could produce a relatively hot fire that, under the right wind conditions, would not reach the canopy. Increased heat transference could have the effect of a higher severity fire on the understory plant community while retaining much of the prefire overstory structure. No data are available on the actual amount of heat transferred to the soil at this site.

MANAGEMENT IMPLICATIONS

These results suggest that a structural view of fire severity may not be an adequate predictor of fire effects on small mammal population density and community structure. Residual forest structure is important, as is the composition of the herbaceous layer. We suggest that both structure and composition must be considered in management practices involving wildfire rehabilitation or prescribed fire. One common practice in the recovery of wildfires is to seed burned areas with grasses to prevent erosion. We noted a strong correlation between deer mouse densities and forb and shrub cover. This vegetation includes nitrogen-fixing plants such as lupine, locoweed, and Mogollon ceanothus, which are beneficial in postfire succession. Because deer mice function as dispersers of plant propagules and soil microbes, it may prove valuable to include these plant species in seed mixes with sterile varieties of exotic grasses or native grasses for use in erosion-prone areas. In addition, current measures of the success of prescribed fires, such as the amount of coarse woody debris left after the fire, may be less important for some species of small mammals than the resulting understory plant community.

We recommend further experimental or pseudo-experimental study of both the effects of wildfire on the small mammal community and the role of small mammals in forest succession following wildfire. Management practices involving wildfire rehabilitation and prescribed fire must consider possible effects, both positive and negative, on this important species assemblage.

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