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Relative Abundance of Small Mammals in Four Successional Stages of Spruce-Fir Forest in Idaho

Abstract

This paper discusses the relative abundance of four small mammal species in successional stages (1-10, 11-39, 40-79, and 80+ years following disturbance) of spruce-fir forest in Idaho. Species evaluated include *Peromyscus maniculatus*, *Eutamias ruficaudus*, *Zapus princeps*, and *Clethrionomys gapperi*. For the specific study sites we investigated, intraspecific comparisons indicated *P. maniculatus* increased with succession; *E. ruficaudus* abundance varied but was generally most common in mid-successional stages; *Z. princeps* preferred willow-alder thickets within mid-successional stages; and *C. gapperi* was most abundant in the mature forest.

Introduction

In many parts of the Pacific Northwest, the logging industry is the dominant land management activity. Thomas *et al.* (1976) pointed out that the weakest link in the process of contrasting alternatives for long term management of forests, was the inability to predict effects on wildlife populations. Since timber management influences wildlife, data on species and population sizes from different forest types, using different forest management procedures are necessary to achieve the needed predictability.

Tevis (1956), Gashwiler (1959, 1970), Hooven and Black (1976), Ahlgren (1966), Stock (1976), and Ramirez and Hornocker (1981) have compared small mammal populations of forested areas with recently logged areas. Less is known, however, about small mammal abundance in relations to forest succession, particularly in the spruce-fir (*Abies* spp.-*Picea* spp.) complex (Hooven 1969). The objective of this study was to investigate the relative abundance of small mammal species in successional stages of spruce-fir forest.

Study Area and Methods

Study areas were located within the Pierce Ranger District of the Clearwater National Forest in Idaho. USDA-Forest Service records of known-aged timber stands were consulted and study sites within two areas (I and II) were selected, successionally representing a range from clear-cut to climax in the spruce-fir association. In each area, one stand of each age class (1-10, 11-39, 40-79, and 80+ years following disturbance) was selected.

Study sites were from 708 to 1539 m above sea level. On the 1-10-year-old sites, Fireweed (*Epilobium agustifolium*), was the dominant forb, with Elderberry (*Sambucus* spp.) and Snowbrush (*Ceanothus velutinus*) the dominant shrubs. In the 11-39 and the 40-79-year-old sites, Heart-leaved Arnica (*Arnica cordifolia*) and Twin-flower (*Linnaea borealis*) were dominant forbs, and Scouler Willow (*Salix scouleriana*) and Honey-suckle (*Lonicera utahensis*) were dominant shrubs in the 11-39-year-old sites. The dominant shrub in the 40-79-year-old sites was Huckleberry (*Vaccinium membranaceum*). Wild Ginger (*Asarum caudatum*) was the dominant forb in the 80+-year-old sites and Huckleberry (*V. membranaceum*) the dominant shrub. In all successional stages, Grand Fir (*Abies grandis*) was the dominant tree species.

Each of the eight study sites contained one small mammal trapping grid measuring 100 by 100 m (1 ha) with permanently marked trap stations at 10-m intervals. Two Sherman traps, baited with rolled oats, were placed at each trap station. In each of the two areas, approximately two weeks were spent trapping the four study sites. Each grid was trapped for seven consecutive days during August of 1978 and 1979. Grids were checked in morning, afternoon, and evening. Traps containing animals were reset immediately. Data recorded for animals included species, age class, sex, trap location, reproductive condition, and weight. Each animal was toe-clipped for identification then released.

Using a GSK statistical analyses described by Grizzle *et al.* (1969), an X^2 test was used to compare the proportions of animals caught in various stages of succession. For each species, we first tested the null hypothesis that the three main effects (stand age, area, and year) were independent. Since in all instances the three-way test for independence, was significant ($P < 0.05$), we then tested two effects for independence, with the remaining effect fixed at a given level. Generally, the stand age by year within a given area test for independence was nonsignificant ($P > 0.05$); we therefore combined the data for each year within a given area. Thus, because there appeared to be some heterogeneity between area I and II, mammal abundance trends within the four successional stages were analyzed separately for each area, with year treated as replicate.

Results and Discussion

Four species were captured in sufficient numbers during both years for analysis of relative abundance trends (Table 1). These include deer mice (*Peromyscus maniculatus*), red-tailed chipmunks (*Eutamias ruficaudus*), western jumping mice (*Zapus princeps*), and red-backed voles (*Clethrionomys gapperi*). With few exceptions, more individuals of all four species were captured in each successional stage in 1979.

In area I there was an increase in the total number of animals captured as the years since clear-cutting increased. Deer mice were the major contributors to this abundance trend. In area II this trend was not as apparent primarily because of the relatively high number of animals captured in the 1-10-year-old successional stage. Although in a different forest type, Pearson (1959) also found the relative abundance of the total small mammal fauna generally to increase from early seral stages to mature oak forests. He noted that between the 16- and 46-year-old stages, grassland degenerated and the small mammal population decreased abruptly in abundance. When the forest matured, mammal abundance increased.

On the sites we investigated, the abundance of deer mice generally increased as

TABLE 1. Number of animals captured (1400 trap nights/successional stage/year) within two areas of the Clearwater National Forest in Idaho. Each area contained four study sites, successionally representing a range from clear-cut to mature forest. Data were gathered in August 1978 and 1979.

		Years since clear cutting							
		Area I				Area II			
		1-10	11-39	40-79	80+	1-10	11-39	40-79	80+
1978	<i>Eutamias ruficaudus</i>	0	18	28	6	12	11	27	12
	<i>Peromyscus maniculatus</i>	20	11	22	54	14	8	9	19
	<i>Zapus princeps</i>	12	18	21	12	3	0	0	3
	<i>Clethrionomys gapperi</i>	0	1	14	14	1	2	0	6
	Total	32	48	85	86	30	21	36	40
1979	<i>Eutamias ruficaudus</i>	7	33	42	7	40	18	14	18
	<i>Peromyscus maniculatus</i>	42	68	69	127	42	31	44	84
	<i>Zapus princeps</i>	9	24	25	22	5	1	0	1
	<i>Clethrionomys gapperi</i>	5	4	34	27	0	4	3	4
	Total	63	129	170	183	87	54	61	107

succession proceeded to climax. In both areas, significantly ($P < 0.05$) more deer mice were captured in the 80+-year-old successional stage than in the 1-10-year-old successional stage. Other authors (Hooven 1969, 1973; Hooven and Black 1976; Stock 1976; Martell and Radvanyi 1977; Ramirez and Hornocker 1981) suggested that early seral stages are most attractive to deer mice, while Harris (1968) and Petticrew and Sadlier (1974) reported little difference in population densities between successional stages.

We were surprised to find deer mice more abundant in the oldest successional stages and suggest additional comparisons should be done, including determination of factors that might influence the distribution of deer mice in spruce-fir forests. Factors thought to influence the distribution of deer mice include food and cover (Hooven and Black 1976, Martell and Radvanyi 1977, Ramirez and Hornocker 1981); intraspecific behavioral interactions and predators (Stock 1976, Sullivan 1979); and long term evolutionary selection (Hooper 1942, Foster 1959).

We then compared the total number of red-tailed chipmunks captured in mid-successional stages (11-39 plus 40-79-year-old sites) with the total number caught in early and late successional stages. In area I, there were significantly ($P < 0.05$) more chipmunks in mid-successional stages than in the early and late successional stages for both 1978 and 1979. The relatively high chipmunk abundance for mid-successional stages may be related to possible preferences for Alder (*Alnus* spp.) thickets (Hoffman 1960). Such thickets were particularly common in mid-seral stages within area I. In area II, this trend was also evident for 1978; however, in 1979, more chipmunks were caught in early and late successional stages than in mid-successional stages. This phenomenon was primarily due to the over three-fold increase in the number of chipmunks caught in the 1-10-year-old sites.

In general, little is known regarding the effects of logging on red-tailed chipmunks. More has been reported on the congeneric Townsend chipmunk (*Eutamias townsendii*) (Gashwiler 1959, 1970; Ahlgren 1966; Hooven and Black 1976). Generally, chipmunks prefer timber and shrubby-brush areas (Tevis 1956; Hooven and Black 1976). Following cutting, population densities were reported to decline; but three to ten years later,

densities were two to four times greater than virgin forests (Tevis 1956; Gashwiler 1959, 1970).

Few chipmunks were captured in the mature forest. Possibly the presence of the red squirrel (*Tamiasciurus hudsonicus*) has an influence on chipmunk distribution. Broadbooks (1970a) observed pine squirrels (*Tamiasciurus douglasi*) chasing yellow pine chipmunks (*Eutamias amoenus*) from the vicinity of squirrel food caches. Elsewhere, however, Broadbooks (1970b) suggests that whereas the two sciurids may overlap in food and habitat, competition is reduced by each specializing in certain plant foods—the pine squirrel in fresh pine and fir cones, and the chipmunk in seeds and flowering plants.

For both years, we found western jumping mice significantly ($P < 0.05$) more common in the mid-successional stages of area I. These mid-successional stages contained the greatest concentration of Willow (*Salix* spp.)-Alder thickets, reportedly a preferred habitat type (Hoffman 1960; Whitaker 1963; Brown 1967). In area II, jumping mice were more common in the early and late successional stages. In this area, however, the sample size was small in all successional stages, and the results from this group of successional stages are inconclusive, as were most of the references we consulted on *Zapus* and forest succession (Rickard 1960; Gashwiler 1970; Hooven and Black 1976; Ramirez and Hornocker 1981). It is noteworthy that Willow-Alder thickets were not as common in the area II sites as they were in area I.

Though a few investigators (Brown 1967; Clark 1971) have studied preferences of jumping mice for different habitat types, studies in all successional stages are lacking. In comparing the relative abundance of jumping mice in cut and uncut forests, Stock (1976) found jumping mice to be more closely correlated with the local availability of suitable habitat than with overall cover density. He found suitable habitat occurred on both cut and uncut sites. Jumping mice are generally considered inhabitants of grass-bordered streams and meadows (Davis 1934; Hamilton 1935; Whitaker 1963; Stock 1976). Hoffman (1960) and Brown (1967) captured them in moist aspen groves, willow-alder thickets, and damp forb-sedge meadows. Getz (1961) found meadow jumping mice (*Zapus hudsonius*) also inhabiting moist situations and suggested that the type of vegetation did not influence the local distribution. Whitaker (1963), however, believed soil moisture had no effect on jumping mice, except indirectly as it influenced vegetation. Our results generally concur with the view that jumping mice are most common in forest stages containing willow-alder thickets.

When the number of captures of red-backed voles for the 40-70 and 80+-year-old sites are grouped and compared with the number of voles captured in the 1-10 and 11-39-year-old sites, significantly more ($P < 0.05$) animals were found in the oldest two successional stages. Most consider red-backed voles to prefer cool, damp, virgin forests (Tevis 1956; Hoffman 1960; Rickard 1960; Morris 1969; Stock 1976; Ramirez and Hornocker 1981). Stock (1976) found that while red-backed voles favor habitat providing dense overstory, the voles did not necessarily prefer a high percentage of ground cover. He often found clearcut sites to contain high percentages of ground cover. He often found clearcut sites to contain as much or more ground cover as forests, yet clearcut sites still did not produce any voles. Hooven and Black (1976) showed that removal of timber and reduction of vegetative cover reduced habitat suitability. Others may be removed, enough slash debris remain to provide a closed canopy at ground level suggest that logging does not affect red-backed voles, and that even though the canopy

(Tevis 1966; Gashwiler 1959; Ahlgren 1966). When management plans call for logging and slash burning, the habitat is significantly reduced to cause a decline in population level. Whereas Ahlgren (1966) reported population levels began to achieve pre-logging levels three years following cutting and burning, in our study this did not occur until approximately 40 years.

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