THE POTENTIAL EFFECTIVENESS OF PRESCRIBED BURNING TO RESTORE SHRUB-GRASS OPENINGS IN A DOUGLAS-FIR/STEPPE ECOTONE

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Judy L. Sunderland Ferguson

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Major Professor: Stephen Bunting, Ph.D.

AUTHORIZATION TO SUBMIT

THESIS

This thesis of Judy L. Sunderland Ferguson, submitted for the degree of Master of Science with a major in Rangeland Ecology and Management and titled "The Potential Effectiveness of Using Prescribed Fire to Restore Openings in a Douglas-fir/Shrub-steppe Ecotone," has been reviewed in final form. Permission, as indicated by the signatures and dates given below, is now granted to submit final copies to the College of Graduate Studies for approval.

Major Professor ______________________________ Date__________
Stephen C. Bunting

Committee
Members ______________________________ Date__________
James L. Kingery

______________________________ Date__________
Edward O. Garton

Department
Administrator ______________________________ Date__________
Kendall L. Johnson

Discipline's
College Dean ______________________________ Date__________
Charles R. Hatch

Final Approval and Acceptance by the College of Graduate Studies

______________________________ Date__________
Charles R. Hatch

Abstract

The encroachment of conifers into sagebrush steppe during the Euro-American period has greatly impacted the ecotone between these vegetation types. Long-term decline in antelope bitterbrush (*Purshia tridentata*), mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyan*a), and many other species associated with shrub steppe has been attributed to this encroachment. Prescribed fire has been suggested as a land management alternative to address this encroachment. Study sites were selected to assess the effects of Douglas-fir (*Pseudostuga menziesii*) encroachment and prescribed fire on vascular plant composition and breeding bird distribution in this ecotone. Fire behavior models were utilized to assess the potential of using prescribed fire for restoration within various stages of conifer encroachment. Increasing Douglas-fir canopy coverage is associated with the decline of bitterbrush, mountain big sagebrush, bluebunch wheatgrass (*Agropyron spicatum*), and other shrub steppe plant species. The historical mosaic pattern of sagebrush steppe and forest results in greater bird diversity than continuous forested vegetation. All stages of forest succession were found to be important to birds. Prescribed fire greatly reduced mountain big sagebrush, but this shrub began to recover quickly. Fire reduced the density of bitterbrush, particularly in the more advanced stages of Douglas-fir encroachment where bitterbrush was in a decadent state. Bitterbrush is in such a state of decadence on heavily forested sites that it declined on all plots, whether they were burned or not. This indicates that prescribed fire alone may not adequately restore populations of this species in the more advanced stages of forest development. Bitterbrush recruitment was not significant after the burns, at least in the short term. The immediate effects of fall prescribed fire in the more advanced encroachment forests did not result in differences in plant species richness, but, after spring prescribed fire, there was a significant decrease in both species richness and canopy coverage. This may be an indication that the physiological stage of some forb and grass species in the spring make them more susceptible to fire mortality. There was no significant difference in bird species richness for either spring or fall burns as compared to the unburned sites; some species left the burned areas, but they seem to have been replaced by other species that were attracted to burned sites. Fire models indicate that the optimal time to burn and maintain the shrub component is during the early stages of conifer development.

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Chapter I

Introduction

In some areas of the West, long-term wildfire suppression is believed to have resulted in forest expansion into historic shrub-steppe openings (Arno and Gruell 1983 and 1986, Clark 1991, Lehman 1995, Sindelar 1971). On the east slope of the Continental Divide in southwestern Montana, such expansion is noticeable along the ecotone between Douglas-fir (*Pseudostuga menziesii* var. *glauca*) forests and shrub-grasslands (Gruel *et al.* 1986). Douglas-fir and Rocky Mountain juniper (*Juniperus scopulorum*) are spreading into the mosaic of park-like openings once dominated by shrubs and grasses. In some areas, the mosaic pattern between forest and shrub-steppe that once characterized this ecotone appears to be disappearing under a continuous expanse of trees. The detrimental effect of forest encroachment on two shrub species, bitterbrush (*Purshia tridentata*) and mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*), is causing concern for land managers. The effects of this encroachment upon other attributes of this ecotone, such as plant species richness, small mammal populations, and bird populations are largely unknown or unquantified.

Mountain big sagebrush and bitterbrush are of special concern in this area because of their importance to wildlife. Introducing fire back into this region is considered to be the best option to halt the decline in shrub cover. Since burning does kill sagebrush and can kill bitterbrush, to reintroduce such a disturbance regime without first determining whether the desired result of increasing shrub densities, particularly bitterbrush, can be obtained would be unwise. It may be that too much time has passed for fire to be a practical remedy to restore the shrub-steppe openings in this ecotone, particularly if the bitterbrush component of the under story with its large, heavy seeds is too decadent to regenerate the area. Replanting bitterbrush seedlings to supplement natural reproduction after burning may be important if it can be determined whether restoring openings are advantageous. Whether the effort is worthwhile depends upon the ecological costs of returning fire to current forested sites as well as upon understanding the negative ecological costs of doing nothing.

If burning is a possible solution, either for natural shrub regeneration or as a pre-planting strategy for bitterbrush, the appropriate timing for prescribed burns in order to get the desired results in the shrub component needs to be better understood. This includes the seasonal (spring/fall) timing as well as the best succession stage to burn.

If burning causes differences in the diversity of this ecotone, these differences should be quantifiable in some way. Diversity changes are very difficult to measure over a whole landscape, but particular aspects can be chosen and measured with the understanding that they are meant to represent a sample of the dimensions of the total ecotone. Species richness, the number of different species found in a given area, is widely accepted as one of the most easily understood and most easily quantifiable measures of diversity (Gaston 1996, Verner 1985).

This research has three objectives. One is to use fire behavior models to assess the optimal sites or successional stages to burn in order to decrease tree overstory and release bitterbrush and sagebrush shrubs from an understory position. The second objective is to measure changes in bird distribution across these same five succession stages of tree encroachment. Differences in abundance, species richness, and diversity between successional stages will be compared. In addition, specific habitat variables will be assessed as to their importance for both individual species and guilds of birds. This information can be added to the prescription phase to help determine size and stage to burn to meet management objectives for conservation of shrub-steppe bird species of special concern. The third objective is to measure changes in plant species composition following prescribed fire in areas of advanced encroachment. Vegetation, small mammal abundance, and bird abundance will be measured before and after burn treatments in openings with advanced succession of trees. The hypothesis to be tested is that spring burning, fall burning, and no burning will have different results. The null hypothesis is there will be no differences.

Specific objectives of this research are:

- predict the best succession stage to burn to maintain openings with a bitterbrush and sagebrush component;
- find differences in bird distributions between the succession stages within this ecotone;
- find differences in habitat characteristics for bird species, particularly in size of openings and shrub densities;
- test differences in vegetation, birds and small mammal use of late succession sites before and after spring and fall burns.

Literature Review

A mosaic pattern of trees intermixed with patches of shrubs and grasses is characteristic of the ecotone where Douglas-fir forest meets shrub-steppe vegetation in southwestern Montana. Patten (1963) studied this mosaic to determine what environmental factors caused it or allowed it to continue. He determined that topography, fire, forest succession, and edaphic factors all played a role, but failure of conifer seedlings to establish in the openings was the primary reason for maintenance of the openings. He concluded that this failure to establish was due to lack of available soil moisture.

Organic matter, or lack of it, may play a role in the moisture regime for seedling establishment in shrub-grass openings. It is known that Douglas-fir seedlings are able to establish on mineral soil with organic layers less than five centimeters thick (Ryker 1975). Other research indicates that interior Douglas-fir seedling establishment is enhanced by soil organic matter (Graham *et al.* 1991). Organic matter prolongs soil moisture conditions to aid seedling survival. In soils with little or no organic matter, moisture requirements for seedling establishment must be met by years of favorable precipitation. Sindelar (1971) found that years with increased precipitation in Montana are the crucial factor for tree colonization into shrub-grass openings because increased precipitation allows Douglas-fir seedlings to survive long enough to establish. He believes lack of fire then allows the trees to continue growth and production. Therefore, without periodic fire, increased Douglas-fir seedling establishment occurs during years when increased precipitation coincides with an abundant cone crop. With the passage of time, these conditions result in the eventual loss of historical shrub/grass openings.

Natural fire is a historically important disturbance factor in southwestern Montana (Arno and Gruell 1983, 1986, Clark 1991, Lehman 1995, Sindelar 1971). Fire delays succession by reducing tree over story and by maintaining or creating new openings for grass and shrubs (Arno and Gruell 1986).

Young Douglas-fir trees are not fire resistant. Young trees have thinner bark and lower canopies. Although mature Douglas-fir are susceptible to crown fires due to flammable foliage and low-growing branches which stay on the bole, they are considered to be fire resistant (Smith and Fischer 1997). This is due to protection on the lower bole of thick bark that insulates the cambium from the heat (Fischer and Bradley 1987). Ryan and Reinhardt (1988) found that the probability of postfire mortality for conifers in the Northwest increased with percentage of the crown killed, and decreased with thickness of the bark. On dry sites, it may take forty years for such thick, insulating bark to develop (Fischer and Bradley 1987). Although the intensity of fire in Douglas-fir stands varies with site, season of burn, and fuel load, Stark (1977) found that 60% of the fires burned at an intensity of less than 300°C ($n = 20$). While this fire intensity did not result in accelerated soil nutrient loss, it did reduce up to 60% of residual fuel from the plots. Residual fuels were a combination of litter and green fuels (shrubs and herbs).

Fire can be used to decrease tree density and tree canopy coverage, but its effect on shrub density may also be negative. Two important shrub species in southwestern Montana are mountain big sagebrush and bitterbrush. Fire kills sagebrush outright, but the long-term effects on sagebrush regeneration are variable. Sagebrush is capable of rapidly re-establishing on the site within a few years (Bunting *et al.* 1987, Young and Evans 1978). In other instances, sagebrush regeneration is delayed for many years after burning (Walhof 1997). The outcome depends upon many factors. Four factors listed by Bunting *et al.* (1987) are species of sagebrush, habitat type, site potential, and size of the burn. Fire intensity is also a major factor. Fire intensity depends upon characteristics of fuel loading and fuel moisture when burning takes place. Fire in areas with low or discontinuous fine fuel loading may result in a mosaic of unburned patches interspersed with burned areas. The intact unburned areas provide a nearby seed source. So, even after hot, intense burns, sagebrush can regenerate in relatively short time spans. The main factors affecting the rate of sagebrush recovery to burning are the length of the fire-free intervals and distance to adequate sagebrush seed sources (Young and Evans 1978).

Bitterbrush seeds are too heavy for wind dispersal after a fire. This species' physiological strategies are to tolerate fire by resprouting from dormant adventitous buds or from meristematic tissue (Blaisell and Mueggler 1956, Driscoll 1963) and to produce palatable seeds, which are harvested and cached by rodents and insects. Uneaten seed caches (5.3 to 8.5% of caches) are then able to germinate after fire (Vander Wall 1994, 1995). After germination, rodents graze on sprouting cotelydons (Clements and Young 1996, Vander Wall 1994). Clements and Young (1996) found that deer mice did not consume bitterbrush seedlings so much as destroy the seedlings through digging activity in spring, while searching for the seed cache source.

The ability of bitterbrush to resprout after burning varies (Noste and Bushey 1987). In general, whether burning kills bitterbrush outright with no post-fire re-sprouting seems to depend upon the vigor of the plants before the burn, the severity of the fire, and the amount of moisture in the soil at the time of the burn (Driver 1982). Bunting *et al.* (1984) looked at bitterbrush survival at least three years after burns on 56 sites. After an initial decrease in density following fire, bitterbrush continued to be productive. They determined that, in the long-term, bitterbrush populations require intermittent disturbance to remain vigorous.

Bitterbrush can be effectively regenerated after fire by rodent-cached seeds only if there is a seed source (Clements and Young 1996, Young *et al.* 1993). Bitterbrush blooms and forms seeds only on second-year wood (Bilbrough and Richards 1991). If browsing is heavy, all new growth is removed every year, seed production is curtailed, and no seeds are available for rodents to cache. Big game animals select for new leader growth when browsing (Burrell 1982). Thus, heavy browsing curtails seed production and regeneration potential of bitterbrush (Pflug 1999). Seed production also depends on moisture during the previous year's growing season. Good seed production follows years of abundant rainfall when stem leader growth averages at least 7.5 cm (Nord 1965). Even when seed production is high, litter can impact seedling establishment. Pflug (1999) found litter buildup on undisturbed sites to be an important barrier to natural regeneration of bitterbrush because it limited suitable locations for rodent caching and seed germination. Plant age can also affect seed production. Bitterbrush productivity declines with advancing age, especially once plants reach 70 years (McConnell and Smith 1977).

Prescribed burning is being considered as a means to keep the dynamics of forest succession in some areas at early seral stages. Prescribed fires have been used to bring back the historic characteristics of several original forest types (Smith and Fischer 1997). Fire behavior models have been devised that take into consideration the attributes of each site, such as slope, aspect, overstory vegetation, understory vegetation, and woody debris. These are combined with variables such as wind, temperature and moisture levels to anticipate the behavior of burns (Anderson 1982, Andrews 1986, Andrews and Bradshaw 1990, Andrews and Chase 1989, Burgan and Rothermel 1984). The effects of prescribed burns are important due to the vegetation changes that occur after fires. The effects of vegetational changes on wildlife use and distribution may be positive or negative after burns. The effects may also be of short or long duration. Reinhardt *et al.* (1997) have developed a fire model that considers species of vegetation, type of vegetation, fuel loads, season of burn, and intensity of burn and uses these parameters to predict the effect burns will have on tree density and shrub load and soil exposure.

Since the main goal of prescribed burns for this project is to restore sagebrush steppe habitat, the effects of a mosaic pattern of habitat openings in forest stands should be considered from the standpoint of species richness as well as abundance. The edge created when two habitats meet is often high in species numbers due to differences in height, species composition, or other variables contributed by two vegetative communities (Lopez de Casenave *et al.* 1998). Some wildlife species prefer edges, while some species avoid it.

Where forests meet shrub steppe, the juxtaposition of both forage and security cover is important to large, relatively mobile wildlife species, such as mule deer (*Odocoileus hemionus*) and elk (*Cervus canadensis*) (Mackie 1970, Mackie and Dusek 1992). Ecotones where forest and shrub steppe vegetation are adjacent are very productive mule deer and elk habitats (Franklin 1993, Mackie and Dusek 1992, Saunders *et al.* 1991, Walhof 1997). High deer densities are typically associated with complex, diverse and stable habitats where food-cover complexes are closely interspersed (Mackie and Dusek 1992). Mule deer have been found to select seral stages with the highest bitterbrush cover regardless of season and to prefer burned Douglas-fir and ponderosa pine (*Pinus ponderosa*) habitat types (Keay and Peek 1980, Griffith and Peek 1989).

For birds and smaller mammals, two factors need to be considered when manipulating habitat with fire: the amount of edge and the fragmentation of forested habitats that will result. Research has explored both the detrimental and the positive effects of edge to various species of avian and mammalian wildlife (Boulinier *et al.* 1998, Bowers and Matter 1997, Foster and Gaines 1991, Faaborg *et al.*1992, Franklin 1993, Keller and Anderson 1992, Lopez de Casenave *et al.* 1998, Maurer 1992, Robbins et al 1992, Saunders *et al.* 1991, Songer *et al.* 1997). For example, some bird studies have found that the density of avian nests decreased with increasing distance from a forest edge (Gates and Gysel 1978). Increased concentrations of nests along forest edges may result in decreased productivity due to increased cowbird (*Molothrus ater*) parasitism (Brittingham and Temple 1996). Increased fragmentation can have serious effects for bird species that require expanses of forest. Boulinier *et al.* (1998) found in their study in the Eastern United States that habitat fragmentation reduced the number of some forest bird species. Manuwal and Huff (1987) compared bird species richness between young, mature, and old growth Douglas-fir in Washington and found little difference between the age of Douglas-fir stands and bird species richness and abundance in spring. However, species richness, diversity, and abundance were all greater in old growth stands during the winter months.

Small mammal species respond differently to habitat changes as well. While some species rely on forest interiors, other species need openings with grass and/or shrubs in order to survive. Deer mice (*Peromyscus maniculatus*) are significantly associated with forest edges (Sekgororoane and Dilworth 1995). Red-back boreal voles (*Clethrionomys gapperi*) avoid openings in forest canopy caused by timber harvest. Deer mice were found to use forest shrub steppe sites in southwestern Montana with bitterbrush a greater proportion of the time than would be predicted from the availability of the shrub (Douglas 1989). Bitterbrush generally grows in greater densities on more open sites.

Study Area

The study area is located between Butte and Whitehall, Montana, along the east slope of the Continental Divide at latitude: 45°53' and longitude: 112°17'. Soils in this area are fast-draining and sandy. They are made up predominately of decomposed granitic particles with very little organic matter and limited A-horizon development. The area has many boulder outcroppings. Boulders are characterized by eroded depressions that hold water into late spring. Under Douglas-fir canopy, the duff and litter layer lying on top of mineral soil is often substantial. Annual precipitation for this area averages 38 to 64 cm.

Soils under the permanent treatment plots located in historic openings in the Toll Mountain Grazing Allotment (Bureau of Land Management) were the following types: Catgulch, moist-Burtoner-Rock Outcrop complex, Shaboom-Kellygulch-Rock Outcrop complex, and Baxton-Connieo, very bouldery Rock Outcrop complex. In addition to soil types already mentioned for the permanent plots, other soil types found under remaining fifty data collection plots include Ashbray, bouldery-Rock, Rubbleland complex, Clancy-Connieo complex, and Jeffcity-Connieo-Rock Outcrop complex (Strom and McCain unpublished, Keck and Strom unpublished).

The habitat type for the forested areas is Douglas-fir/Idaho fescue (*Festuca idahoensis*) (Pfister *et al.* 1977). Small numbers of lodgepole pine (*Pinus contorta*) and limber pine (*Pinus flexilis*) are present on a few sites. Forested sites are dominated by Douglas-fir and Rocky Mountain juniper. The shrubs encountered in this area are predominantly mountain big sagebrush and bitterbrush with both gray rabbitbrush (*Chrysothamnus nauseosus*) and green rabbitbrush (*Chrysothamnus viscidiflorus*) as important components on some sites. In rocky outcroppings across the landscape, curlleaf mountain-mahogany (*Cercocarpus ledifolius*) is common. Primary grass species are bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue, Sandberg bluegrass (*Poa secunda*), blue grama (*Bouteloua gracilis*), and several species of needlegrasses (*Stipa* spp.). Elk sedge (*Carex geyeri*) is also present in the understory.

The estimated historical mean fire intervals for this research area is taken from work done by Arno and Gruell (1983, 1986). They estimated the mean fire interval for Douglas-fir/Idaho fescue (dry) habitat type for southwestern Montana to be 45 years over the period from 1609 to 1862. Mean fire intervals for sites they surveyed close to this research area were estimated to be even more frequent, averaging every 25 years from 1690 to 1800. They also believe historical burns were extensive in area. They determined that trees were historically confined to rocky outcroppings and other microsites characterized by little vegetation or fuels to carry fire.

Chapter II

Criteria for Ecotone Restoration Based on Fire Model Data

Introduction

The decline of bitterbrush and mountain big sagebrush in southwestern Montana is a growing concern to land management agencies and wildlife biologists. This decline is a direct result of Douglas-fir encroaching into historical shrub steppe openings. Seedling establishment takes place because of years of increased moisture, but it is lack of fire that allows the trees to grow and prosper (Patten 1963, Sindelar 1971). It is believed that returning fire to this ecotone may be necessary in order to restore and maintain the mosaic pattern of forests and shrub openings that historically characterized the landscape.

It is possible to burn any stage of conifer succession from small trees to mature, old growth Douglas-fir, but removal of tree overstory is only part of the solution. A more crucial need is to determine which stage of conifer colonization is the most appropriate one to burn in terms of restoring shrub cover, particularly bitterbrush cover. It is not feasible to burn across all stages of succession and wait for decades to see what results. Reliable estimates of projected results are needed now in order to determine the best stage to burn to prevent further declines in shrub openings and to restore the ecotone to a more balanced mosaic pattern, a pattern with a mixture of forest stands and shrub openings.

In order to identify the best stage or stages to burn, five successional stages were surveyed for trees, shrubs, grass and downed woody debris. By stratifying the research sites across the broad spectrum of the ecotone, a continuum of Douglas-fir succession could be studied. The First Order Fire Effects Model (FOFEM) was used to predict the results of burning on each of the five succession stages (Reinhardt *et al.* 1997). The objective of this research was to find the most appropriate stage/stages to burn to decrease tree canopy while increasing the probability of leaving sufficient shrubs in the understory to regenerate the opening created by the fire.

Methods

In 1997, fifty plots were measured for the following parameters: above-ground herbaceous biomass, shrub and tree densities, and canopy cover of shrubs and trees using the Canfield line-intercept method (1941). Three 60-meter transects were placed parallel to each other within vegetation stands that met specific criteria from early to late succession. Ten replicates were surveyed for each successional class. The successional classes or stages of vegetation development were:

Class 1-grassland openings;
Class 2-shrub steppe openings;
Class 3-young forest;
Class 4-mixed-age forest;
Class 5-mature forest.

Grassland openings were chosen based on grass as the dominant vegetation. Shrubs were allowable, but not in densities so great that they appeared to dominate the site. Live trees could not be present. Since this stage is an early seral stage of the shrub-steppe ecotone with Douglas-fir forest, this category cannot be maintained without disturbance, usually fire, and were the most difficult areas to find. This difficulty underscores the changing pattern of this ecotone to a more forested landscape.

Shrub steppe openings were defined by the presence of shrubs, particularly mountain big sagebrush and antelope bitterbrush. Shrubs dominated the vegetation on these sites. This stage had five or fewer live trees, which were less than 4.5 m tall. This successional stage was still present but difficult to find.

Young forest sites encompassed the first stage of conifer encroachment. The criteria for inclusion on these sites were a majority of trees less than 4.5 m in height. There were no criteria for density of small trees, shrubs, or grass on these sites, and they encompassed a wide array of densities. No more than 10 older age trees was allowable. Considering the widespread influence of tree encroachment on this ecotone, it was surprisingly difficult to find areas where trees were still short enough to fit the criteria for inclusion in this type.

Mixed-age forest sites were defined by a mixture of vegetation. Grasses, shrubs, and various-aged trees were the components on these sites. This later stage of succession is the most abundant stage in the study area.

Mature forest sites were dominated by trees. These sites had almost no living sagebrush or bitterbrush in the understory. The abundance of grass was not a standard for inclusion. In some cases these sites were dominated by large old Douglas-fir and in others by thick stands of mature, but younger, Douglas-fir. Generally, the duff layer on these sites was thick. These sites were difficult to find because they needed to be on soils that could potentially support bitterbrush and sagebrush, but that no longer had these shrubs. There were many forest stands along this ecotone where depleted populations of decadent shrubs survived in the understory despite decades of competition from trees.

Canopy cover was measured using the Canfield (1941) method of line intercept along each of the three 60-meter transects within each macroplot. The amount of line covered by a shrub or tree species was measured and added for each plot. If different species had canopy overlap, each was counted separately. The same species overlapping each other were counted as a single unit.

Above-ground herbaceous biomass was measured using a clip-and-weigh technique with five one-meter square quadrats clipped per macroplot. These quadrats were placed at the center point and at each end of the two outside transects on each plot. Within each quadrat, current-year forbs, current-year grass, and standing dead biomass were separated and weighed. Grasses and forbs were collected separately and weighed green in the field. These were later oven-dried for 48 hours at 84-90° C, until all moisture was removed, and there was no change in weight. The dry weights were used to compute above-ground herbaceous biomass estimates.

Density of shrubs and trees was measured along a one-meter belt along each transect. This resulted in 180 square-meters of density used for the estimate per macroplot. Tree density was measured by species into height size classes: 5-55, 55-150, 150-500, and greater than 500 cm. Shrub density was counted by species into age classes: seedling, young, mature, decadent, or dead.

Downed woody debris counts were completed along 18 m of the center transect of each plot (Brown 1974, Brown *et al.* 1982). The woody debris along the line was counted into one of four categories by diameter size. The three smallest diameter classes (0.0 to 0.63 cm, 0.63 cm to 2.5 cm, 2.5 to 5.1 cm) were counted along the first 3.66 meters of the transect. Eighteen meters were inventoried for the largest woody debris class (greater than 7.5 cm). Debris in this class was identified as to species and soundness.

The First Order Fire Effects Model (FOFEM) was used to estimate the predicted effects of burning each of the inventoried succession classes (Reinhardt *et al.* 1997). The mean, the lower and the upper extremes of the fuel data collected for each class were used as input in FOFEM. The resulting predictions for burn intensity and tree mortality were then compared to the present canopy cover of shrubs and herbaceous biomass to determine the most appropriate stage of succession to burn with adequate fine fuels to carry the fire and with an intact shrub component after burning to either maintain or rejuvenate shrub populations, particularly bitterbrush.

Results

Results from the vegetation surveys will be given first because these are the data used for runs in the burn simulation model and because they quantify site characteristics. Table 2.1 gives the results from shrub density surveys. Table 2.2 shows mean estimates of percent decadence of sagebrush and bitterbrush. Decadence estimates are important because they are correlated to the potential of shrubs to regenerate on the site after fire.

Table 2.1. Mean density of shrubs per hectare across five succession stages with standard error estimates in parentheses.

	Living Bitterbrush	Living Big Sagebrush	Dead Bitterbrush	Dead Big Sagebrush
Grass Openings	39 (29)	3143 (1261)	256 (256)	989 (573)
Shrub Steppe Openings	367 (170)	7120 (1476)	322 (202)	1011 (269)
Young Forests	567 (299)	5089 (1217)	833 (498)	811 (245)
Mixed-Aged Forest	228 (91)	2176 (429)	957 (348)	1430 (174)
Mature Forest	94 (39)	72 (26)	438 (245)	309 (120)

Table 2.2. Mean percentages of total living shrub densities (number plants/ha) that are decadent with standard error estimates in parentheses.

	Sagebrush	Bitterbrush
Grassland Openings	4.9 (1.8)	0.0 (0.0)
Shrub Steppe Openings	18.3 (3.1)	22.4 (6.0)
Young Forests	43.1 (11.0)	17.4 (8.1)
Mixed-Aged Forest	54.6 (9.3)	36.1 (11.4)
Mature Forest	66.7 (33.3)	25.0 (25.0)

Canopy cover estimates for shrubs can be compared with density estimates. Canopy cover is an estimate of stature, which is a good comparison to consider when looking at vigor as opposed to presence. Table 2.3 shows the contrasts across successional stages for shrubs and trees.

Table 2.3. Mean percent canopy cover by species across successional stages with standard error estimates in parentheses.

	Grassland Openings	Shrub Steppe Openings	Young Forests	Mixed-Age Forest	Mature Forest
Mountain Big Sagebrush	2.5 (0.7)	15.4 (1.9)	10.2 (2.7)	3.3 (0.7)	0.01 (0.09)
Bitterbrush	0.1 (0.1)	1.1 (0.6)	0.7 (0.3)	0.1 (0.7)	0.002 (<0.01)
Douglas-fir	0.04 (0.04)	0.06 (0.06)	9.7 (4.2)	28.6 (2.8)	44.1 (3.4)
Rocky Mountain Juniper	0.0 (0.0)	0.06 (0.04)	2.5 (1.3)	3.3 (1.0)	3.7 (1.3)

Density of trees by size class is another variable that is important FOFEM in order to estimate mortality. These data are also important to include when considering burning for restoration of shrub-grass openings. Table 2.4 gives tree density per hectare.

Table 2.4. Mean number of trees per hectare across height classes with standard error estimates in parentheses. (Grassland Opening class had less than one/half tree per hectare and is not included.)

	5-55 cm		55-150 cm		150-500 cm		>500 cm
	PSME	JUSC	PSME	JUSC	PSME	JUSC	PSME
Shrub Steppe Openings	0	22 (15)	0	6 (6)	0	11 (11)	0
Young Forests	607 (507)	101 (31)	1317 (1017)	216 (52)	269 (115)	129 (41)	23 (17)
Mixed-Age Forests	386 (139)	364 (92)	272 (60)	190 (66)	336 (50)	249 (53)	55 (25)
Mature Forests	890 (367)	273 (84)	272 (110)	173 (50)	534 (174)	318 (94)	378 (111)

(PSME = Douglas-fir; JUSC = Rocky Mountain juniper)

Total Douglas-fir density estimates for the young forest stage are 2216 trees per ha with 87% of those trees under 150 cm tall. In the mixed-aged canopy stage, there was a total of Douglas-fir 1049 trees/ha and 63% were less than 150 cm in height. Mature forests had 2074 trees/ha with 56% under 150 cm. Densities for juniper were relatively low across most successional stages. Although there were no measurable Douglas-fir densities for the Shrub Steppe Opening class, there was a mean density of 39 juniper/ha for this successional class with 72% in the two shortest height classes. The young forest class had a total mean juniper density of 446 trees/ha with 71% that were less than 150 cm tall. In the mixed canopy stage, there was a mean total density of 803 trees/ha with 69% less than 150 cm tall. The mature stage had 764 juniper/ha with 58% in the shorter two height classes.

Grass and forb aboveground biomass estimates are important because they are the primary fine fuels that carry the fire across the site (Table 2.5).

Table 2.5. Kiln dry weight in kilograms per hectare of herbaceous aboveground biomass by successional stage with standard error estimates in parentheses.

	Grass	Forbs	Total
Grassland Openings	1349 (205)	438 (433)	1787 (292)
Shrub Steppe Openings	618 (74)	216 (89)	833 (145)
Young Forests	532 (133)	65 (22)	598 (137)
Mixed-Age Forests	296 (28)	92 (58)	388 (74)
Mature Forests	260 (132)	52 (52)	312 (179)

Environmental Conditions Used in FOFEM:

Environmental conditions used for the burning conditions in FOFEM runs were set in two ways. Independent data sets from the inventories collected across the 50 plots were used, when applicable. The information gathered in these inventories has been noted in the previous section of this paper. Some of these data were transposed to different forms for the model runs. The model needed tree inputs in the form of diameter-at-breast-height (dbh) units rather than heights. Also, data used in the model runs were transposed into English units from the metric units that were collected in the field.

Other, more arbitrary, input choices were set by the author based on likely field conditions under most types of negotiated prescribed burning situations. For example, it is unlikely that, under extremely dry conditions in the fall, a high intensity prescribed burn would be given ignition approval. These arbitrary data inputs into FOFEM were identical for all burn runs. They were: fall burns, moderate intensity, dry conditions, typical, natural fuel loads for sites in the Interior West.

Model Results:

Three runs were executed for each set of fine fuel data. These were the mean weights (listed in Table 2.5 but converted to pounds per acre for the model). In addition, the highest yield measured on any plot within a succession class and the lowest yield measured on any plot within a succession class were also used in separate runs to get the range of variation in the burn conditions of the each successional stage. All runs are for fall burns of moderate intensity in natural fuel beds under dry conditions.

Table 2.6 gives fire model results for fuel loads. This table provides the results from using the mean values measured on the ground. Changes when high and low values for fine fuels on the sites can be found in the appendices, but they are similar to the means shown. Shrub opening sites and small tree invasion sites (young forests) are predicted by FOFEM to leave the greatest amounts of shrubs on the sites after burning.

Tree mortality from burns is 98% for young forest sites with postburn densities reduced from 3205 trees/ha to 58 trees/ha. On mixed-age forest sites, moderate intensity fall burns should reduce tree densities from 1635 trees/ha to 148 trees/ha (91% mortality). Most of the surviving trees for the mixed age site are in the larger size classes with only 33% mortality predicted for larger than 4-dbh trees. Probability of mortality for mature forests with greater canopy coverage and little understory vegetation is predicted to be

82%. This is a reduction in tree density from 3235 trees/ha to 570 trees/ha. Average mortality for large trees classes on these mature forest sites is 31%.

Table 2.6. Pre-burn and postburn fuel loads predicted by FOFEM for total fuels and the shrub component across successional stages (results changed from original output of tons/acre to the metric units of metric tons/hectare).

	Grassland Openings	Shrub Steppe Openings	Young Forests	Mixed-Age Forests	Mature Forests
Pre-burn Load (mt/ha)	1.87	2.97	9.35	18.48	41.25
Postburn Load (mt/ha)	0.44	0.22	1.87	3.85	10.34
Pre-burn Shrub (mt/ha)	1.10	2.53	2.20	0.22	0.00
Postburn Shrub (mt/ha)	0.44	0.22	0.88	0.11	0.00

Discussion

With successional advancement, the density of living shrubs doubles for sagebrush from the grass to the shrub stage and then starts to decline rapidly, especially after small tree encroachment begins. Dead sagebrush densities remain stable through the first four stages and declines rapidly under mature trees as canopy closes and long-term soil changes began with the addition of layers of duff and needle. The decline in dead shrubs densities under greatest tree canopy is probably a function of increased time since shrubs were living thus increasing the amount of time for decomposition of shrub skeletons.

Although bitterbrush density is not as abundant for early successional stages as sagebrush, abundance for living bitterbrush increases through the small tree stage. By the time mature canopy forest coverage is reached, bitterbrush has surpassed sagebrush densities in direct density comparison. This may be an asset when considering using fire to recreate openings for bitterbrush.

Since fire kills sagebrush, the vitality of sagebrush shrubs is not as important for this species as is preserving safe sites to reseed the area. Percentages of living sagebrush that are decadent increases with succession to a high of 66.7% under mature tree canopy. The percentage of bitterbrush in the decadent age class is an important consideration because decadent shrubs (low vitality rating) are not as likely to withstand burning (Ayers and Bedunah 1999, Bunting *et al.* 1984). Percentage of decadent bitterbrush is low through the young tree stage.

Mortality of trees is predicted to be the greatest on young forest sites. Most of the considerable load of small trees would be removed by burning these sites. Since young forest sites have the highest densities of bitterbrush, this could be the most appropriate stage of succession to burn within this ecotone in order to reduce tree encroachment while favoring bitterbrush.

Over much of this ecotone, succession has advanced beyond the young tree stage into the mixed-age forest stage, so the predictions for mixed-age forest are particularly important to consider. FOFEM predicts that shrub loads will be reduced by half on these sites. Preburn cover for shrubs on these sites is about 3%. This would mean a decrease to 1.5%, but about half of the shrubs on these sites with more advanced succession are decadent and may not be able to regenerate these sites as well as early successional sites where more vigorous shrubs survive.

Conclusions

The concern for the fate of shrub populations with increasing tree overstory in this ecotone is reinforced by the results from this study. The density of dead shrubs (Table 2.1) for both sagebrush and bitterbrush is highest in the mixed canopy stage. Although dead shrubs appear to decline rapidly under greatest tree canopy coverage, these declines are a function of disintegration of dead shrub skeletons with time. Corresponding with increasing levels of dead shrubs with increasing tree canopy is an increase in percent of living shrubs classed as decadent. Increasing decadence levels under greater tree canopy coverage is especially critical when the small number of living shrubs left on those sites is considered. Differences in the magnitude of change between the bitterbrush and sagebrush through succession changes may be subtle but important. There are much higher densities of sagebrush through most successional stages, but under the heaviest tree overstories, it is bitterbrush that apparently survives the longest.

The most appropriate stage to burn is the young tree invasion stage for several reasons. Burning young forest sites means the trees on them are still small enough to be susceptible to burns of moderate intensity. Moderate burns can remove most of the tree overstory, which is the ultimate necessity if successful restoration is to be achieved. In contrast, it would take higher intensity fires to remove the more mature trees on more advanced succession sites because the fine fuel load, which carries the fire, is so low under greater canopy. In addition to high tree mortality with burning, young forest sites have the highest densities of bitterbrush plants and the least percentage of decadent bitterbrush. Since it takes vigorous plants to resprout or to reseed an area, this is an important consideration.

Chapter III

Effects of Forest Successional Changes on Species Distribution of Breeding Birds

Introduction

In southwestern Montana, forest expansion into shrub-steppe openings is of long-term concern (Arno and Gruell 1986, Gruell *et al.* 1986, Sindelar 1971). Many factors should be considered when deciding whether to use prescribed burning to release shrubs from the understory of trees along this ecotone. These include the effects that burning is likely to have upon the vegetation and the responding effects that vegetation changes will have on wildlife. The significance of shrubs to big game animals is well-documented with research (Burrell 1982, Constan 1972, Griffith and Peek 1989, Keay and Peek 1980, Mackie 1970, Mackie and Dusek 1992, Ward 1971, Welch and Wagstaff 1992).

There is less documentation on the effects of the loss of openings on bird species abundance and diversity. The importance of the size of openings, the structure of vegetation within them, and the distance they are from other important habitat components varies from species to species. Canopy cover changes within forest stands have been found to affect bird density, diversity, and species richness. Ground-understory foragers, hummingbirds and some flycatchers were found to be less abundant in closed-canopy forest stands due to the greater structural array of vegetative structure in the understory of more open canopy stands (Beedy 1981). One study found that, out of a total of 87 species detected across 15 major vegetation types, 15 bird species were more abundant on early post-fire sites than on other cover types (Hutto 1995). Other species of birds prefer closed-canopy forest, so there may be a trade-off when evaluating habitat modification such as prescribed burning or thinning (Apfelbaum and Haney 1981, Bendell 1974, Franzreb 1978, Hagar *et al.* 1996, Hutto and Young 1999, Manuwal and Huff 1987, Sakai and Noon

1991, Sieg 1991, Whitaker and Montevecchi 1999).

This study was designed to compare bird distribution within five distinct succession stages along a shrub-steppe and forest ecotone in Southwestern Montana. Within the ecotone, increases in Douglas-fir (*Psuedostuga menziesii*) and Rocky Mountain Juniper (*Juniperus scopulorum*) cover occur at the expense of grass and shrub abundance. Different bird species may be positively or negatively affected by this change in vegetation. The objective of this research was to determine if habitat associations for breeding birds change with successional changes in shrub steppe and forest communities within this ecotone.

Methods

In 1997, fifty plots were measured for several vegetative parameters: above-ground biomass, shrub and tree densities, and canopy cover of shrubs and trees using the Canfield line-intercept method (1941). Vegetation surveys were carried out along three 60-meter transects placed parallel to each other within vegetation stands that met specific criteria from early to late succession. Ten replicates were surveyed within five different habitat classes for all classes except the grassland openings class. Nine replicates were surveyed for that class. Openings where grass dominated were the most difficult areas to find because disturbance, such as fire, is necessary to keep them at such an early stage of succession.

The five habitat classes or stages of vegetation studied were:

- Class 1-grassland openings;
- Class 2-shrub steppe openings;
- Class 3-young forest;
- Class 4-mixed-age forest;
- Class 5-mature forest.

(See Chapter II methods section for description of these successional classes and for field methods for vegetative biomass and density surveys.)

In addition to herbaceous biomass, density, and canopy cover measurements, a range-finder was used to estimate distances to a series of habitat variables. These data were all used to correlate species with habitat factors. The specific variables measured were: opening size, distance to edge, distance to large trees, distance to deciduous trees, distance to large snags, distance to mature trees, number of snags, and distance to curlleaf mountain-mahogany (*Cercocarpus ledifolius*).

Bird Surveys:

In 1999 breeding bird surveys were completed on forty-nine plots across five successional stages, grassland openings through mature forests. These were carried out in May and June. A point count protocol was used for these surveys (Ralph *et al.* 1995, Verner 1985). This protocol included a ten-minute survey between the hours of dawn and 11:00 a.m. on days with no wind and no heavy rain. No counting was done during the first two minutes to let the birds return to normal behavior. All birds detected over the next eight minutes were noted. Those located in the first three minutes were listed separately from those in the last five minutes. The time period was divided into two segments to guard against counting birds that move frequently but still include birds that may sing at less frequent intervals. Each macroplot was visited three times for a total of 147 counts. During these surveys each species detected was recorded and its location estimated. All birds within 50 m of the plot center were counted separately from birds heard outside this radius. The activity of birds within this radius was recorded. It is the results from these 50-m circles that will be discussed in this paper. This short radius was chosen to be sure that the effects of the habitat would be properly correlated with the bird species for data analyses.

Analyses:

Species richness summarizes the number of different species counted within each succession stage of increasing tree densities. It is an easily understood measure of diversity. When looking at differences across habitat classes, diversity indices can suggest important habitat characteristics for rarer species and are often considered along with evenness of counts across habitats (Magurran 1988, Simpson 1949). For this reason, four diversity statistics were computed with the data collected from the bird surveys. These were species richness, Simpson index, Shannon index (also called Shannon-Weiner index), and Shannon evenness (Magurran 1988).

Bird Guilds:

All species detected within fifty meters of the center of each plot were placed into guilds by using Spearman correlation analysis for all species against all habitat variables. Guilds are often used to aggregate species together that use habitat in specific ways, such as foraging guilds or nesting guilds (Apfelbaum and Haney 1981, Beedy 1981, Eckhardt 1979, Knopf *et al.* 1988, Whitaker and Montevecchi 1999). In order to find whether one or more successional stages were used more than others, each bird species was placed into five guilds by using habitat variables associated with vegetation abundance and type. These were canopy cover of trees and shrubs, density of trees and shrubs, opening size, distance to edge, herbaceous biomass, distance to large trees, distance to deciduous trees, distance to large snags, distance to mature trees, number of snags, and distance to mountain-mahogany.

All of these variables for each site were compared with each bird observed for that site. This resulted in Open-habitat Guild, Closed-tree Canopy Guild, Mature-tree Canopy Guild, Edge-association Guild, and Generalist Guild. Step-wise linear regression was then used for each guild against all habitat variables in order to find which variables explained most of the distribution for that guild.

Results

The following results from collected field data will be provided: vegetation data (tree and shrub canopy-cover and density and herbaceous above-ground biomass), habitat variables collected with a rangefinder, and point counts bird surveys. Then the correlation results when all three of these are combined together will be given.

Vegetation:

Living shrub densities varied somewhat by species with sagebrush reaching highest densities in the shrub stage and bitterbrush in the small tree stage. The loss of shrubs with increasing tree overstory can be seen by the increasing densities of dead shrubs as advanced succession advances (Table 3.1). Table 3.2 compares the total tree densities for Douglas-fir and Rocky Mountain juniper by successional stage and height class.

Table 3.1. Mean density of shrubs per hectare across five successional stages with standard error estimates in parentheses.

	Living Bitterbrush	Living Big Sagebrush	Dead Bitterbrush	Dead Big Sagebrush
Grassland Openings	39 (28.9)	3143 (1261.3)	256 (255.6)	989 (573.7)
Shrub Steppe Openings	367 (170.0)	7120 (1476.3)	322 (202.5)	1011 (269.7)
Young Forests	567 (299.1)	5089 (1217.5)	834 (497.8)	811 (245.0)
Mixed-Age Fprests	207 (98.5)	2176 (429.5)	209 (59.7)	1430 (174.5)
Mature Forests	101 (38.8)	72 (26.1)	445 (236.3)	309 (120.3)

Table 3.2. Mean total tree density per hectare across height categories with standard error estimates in parentheses. [Grassland openings had less than one/half tree per hectare.]

	5-55 cm	55-150 cm	150-500 cm	> 500 cm	Total Trees
Shrub Steppe Openings	22 (15)	6 (6)	0	0	28 (19)
Young Forests	86 (521)	1549 1032)	292 (85)	6 (6)	2708 (1543)
Mixed-Age Forests	810 (215)	462 (101)	585 (75)	55 (25)	1912 (328)
Mature Forests	1146 (357)	329 (76)	843 (196)	904 (586)	3222 (575)

Canopy cover is a measure of plant stature. Canopy cover estimates for both sagebrush and bitterbrush are greatest in shrub steppe openings, followed by young forest encroachment sites (Table 3.3). The loss of canopy cover of shrubs with increasing overstory canopy coincides with increasing dead shrubs in the understory (Table 3.1). Other vegetative components of this ecotone also change with increasing canopy cover of trees. Both forb and grass biomass declined continually with advancing forest succession (Table 3.4).

Table 3.3 Mean percent canopy cover by species across succession stages. Standard error estimates are in parentheses.

	Grassland Openings	Shrub Steppe Openings	Young Forests	Mixed-Age Forests	Mature Forests
Mountain Big Sagebrush	2.512	15.44	10.230 (2.747)	3.283	0.114
	(0.778)	(1.888)		(0.747)	(0.099)
Bitterbrush	0.111	1.099	0.676	0.136	0.002
	(0.106)	(0.605)	(0.293)	(0.690)	(0.002)
Douglas-fir	0.044	0.056	9.711	28.566	44.146
	(0.044)	(0.056)	(4.247)	(2.813)	(3.355)
Rocky Mountain Juniper	0	0.062	2.500	3.306	3.666
		(0.038)	(1.269)	(1.022)	(1.274)

Table 3.4. Dry weight of forbs and grasses in kilograms per hectare by succession stage. Above-ground herbaceous biomass are given with standard error estimates in parentheses.

	Grass	Forbs	Total
Grassland Openings	1349 (205)	438 (433)	1787 (292)
Shrub Steppe Openings	618 (74)	216 (89)	833 (145)
Young Forests	532 (133)	65 (22)	598 (137)
Mixed-Age Forests	296 (28)	92 (58)	388 (74)
Mature Forests	260 (132)	52 (52)	312 (179)

Birds:

Forty-three different species of birds were counted in 147 surveys across 49 sites (Table 3.5a and 3.5b). The species list includes the scientific name for each species and the field code. The list is divided into two sections because only the 28 species detected within 50 meters of the center of each plot will be used for further analysis. (Appendix III gives these results by succession stage.) Diversity estimates, guild and species distribution patterns are all based on these 28 species.

Table 3.5a. Bird species detected within 50 meters of plot center across all successional stages.

Species	*Scientific Name*	Code
AmericanGoldfinch	*Carduelis tristis*	AMGO
American Robin	*Turdus migratorius*	AMRO
Black-capped Chickadee	*Parus atricapillus*	BCCH
Brewer's Sparrow	*Spizella breweri*	BRSP
Brown Creeper	*Certhia americana*	BRCR
Brown-headed Cowbird	*Molothrus ater*	BHCO
Chipping Sparrow	*Spizella passerina*	CHSP
Clark's Nutcracker	*Nucifraga columbiana*	CLNC
Common Nighthawk	*Chordeiles minor*	CONH
Dark-eyed Junco	*Junco hyemalis*	DEJU
Dusky Flycatcher	*Empidonax oberholseri*	DUFC
Evening Grosbeak	*Coccothraustes vespertinus*	EVGR
Hairy Woodpecker	*Picoides pubescens*	HAWP
Hammond's Flycatcher	*Empidonax hammondii*	HAFC
Hermit Thrush	*Catharus guttatus*	HETH
Mountain Bluebird	*Sialia currucoides*	MTBB
Mountain Chickadee	*Parus gambeli*	MTCH
Northern Flicker	*Colaptes auratus*	NOFL
Northern Goshawk	*Accipiter gentilis*	NOGH
Pine Siskin	*Carduelis pinus*	PISI
Red-breasted Nuthatch	*Sitta canadensis*	RBNH
Ruby-crowned Kinglet	*Regulus calendula*	RCKI
Savannah Sparrow	*Passerculus sandwichensis*	SASP
Townsend's Solitaire	*Myadestes townsendi*	TOSO
Vesper Sparrow	*Pooecetes gramineus*	VESP
Warbling Vireo	*Vireo gilvus*	WAVI
Western Meadowlark	*Sturnella neglecta*	WEML
Yellow-rumped Warbler	*Dendroica coronata*	YRWA

Table 3.5b. Bird species detected only outside a 50-meter radius of plot centers for all succession stages. This includes fly-overs.

Species	*Scientific Name*	Code
Common Raven	*Corvus corax*	CORA
Common Snipe	*Gallinago gallinago*	COSN
Grasshopper Sparrow	*Ammodramus savannarum*	GHSP
Great Horned Owl	*Bubo virginianus*	GHOW
Green-tailed Towhee	*Pipilo chlorurus*	GTTO
Mourning Dove	*Zenaida macroura*	MODO
Ovenbird	*Seiurus aurocapillus*	OVBI
Red Crossbill	*Loxia curvirostra*	RECR
Red-tailed Hawk	*Buteo jamaicensis*	RTHA
Rock Wren	*Salpinctes obsoletus*	ROWR
Ruffed Grouse	*Bonasa umbellus*	RUGR
Turkey Vulture	*Cathartes aura*	TUVU
Western Tanager	*Piranga ludovioiana*	WETA
Western Wood-Pewee	*Contopus sordidulus*	WEWP
White-breasted Nuthatch	*Sitta carolinensis*	WBNH

Habitat and Bird Distribution Results

Guild Results:

Most bird species in my study were either common, but not highly associated with specific habitat factors, or so uncommon or restricted in distribution that good habitat analysis results on a species by species would be difficult to find, even if they exist in reality. Grouping birds for better analyses by placing birds into foraging and nesting guilds for habitat distribution analysis is common (Apfelbaum and Haney 1981, Eckhardt 1979, Emlen 1974, Faaborg 1985, Knopf *et al.* 1988, Whitaker and Montevecchi 1999). Since few individual species in these data were correlated with individual habitat factors, I decided to look at structure height of vegetation and groups species according to their use of structure because that is the variable that is the most apparent change with tree invasion into grassland and shrub steppe openings.

In order to place the species of birds encountered during the field surveys into guilds, I used Spearman's correlations. All species positively correlated with the habitat variable 'size of opening' were grouped as species of open habitats into an Open Guild. Birds with positive correlations to openings were: Mountain Bluebird (0.580), Vesper Sparrow (0.600), Brewer's Sparrow (0.412), Western Meadowlark (0.229), and Savannah Sparrow (0.264).

For birds of forested areas, positive correlations with variables at the other extreme from openings, canopy cover and density of trees, were grouped together. Species had to be positively correlated with both variables in order to be placed in this guild. These correlations resulted in a Closed-canopy Forest Guild with six species. These were (correlation for percent canopy of trees listed first and for number of trees per hectare listed second) are Dusky Flycatcher (0.535 and 0.508), Yellow-rumped Warbler (0.445

and 0.305), Hermit Thrush (0.360 and 0.241), Mountain Chickadee (0.419 and 0.387), Hairy Woodpecker (0.228 and 0.363) and Warbling Vireo (0.317 and 0.232). All of these species except Hermit Thrush and Yellow-rumped Warblers were also positively associated with sites with mature trees.

Forested species, not correlated with density of trees, were grouped by correlations to mature trees. Four species were placed in a Mature Tree category. Hammond's Flycatchers were correlated with percent canopy cover of trees (0.237) and with sites with mature trees (0.488). Red-breasted Nuthatches were correlated with mature trees (0.472). They were negatively correlated with distance to large trees (-0.247). Evening grosbeaks were correlated to mature tree sites (0.615). They were also correlated to canopy cover of trees (0.302) and with a negative correlation to distance to large trees (-0.202). Both the Northern Goshawk and the Brown Creeper were positively correlated with mature tree sites (0.302).

The next guild is restricted to those species that were negatively correlated with distance to the edge of a habitat class. The higher the negative number, the stronger the association of the species to the habitat edge. Five species fit into the Edge Guild: Chipping Sparrow (-0.262), Clark's Nutcracker (-0.235), American Robin (-0.411), Dark-eyed Junco (-0.314), and Brown-headed Cowbird (-0.206). Brown-headed Cowbirds were also positively associated with mature trees (0.233). (Evening Grosbeaks were associated with edges (-0.260) as well as mature trees, but they were not placed in this category because of their strong correlation with mature trees.)

The remaining species were placed in a Habitat Generalist category because they did not correlate as a group with any available variable measures. A few of these species had correlations greater than 0.200 for particular variables, and these are noted. Species included in the Habitat Generalist Guild are Black-capped Chickadee (forb biomass 0.239), Northern Flicker, Ruby-crowned Kinglet (percent sagebrush –0.257), Townsend's Solitaire (distance to nearest snags –0.410, number of snags 0.381, percent bitterbrush cover 0.264), Pine Siskin (herbaceous biomass 0.233 and distance to large trees 0.225 and to mountain-mahogany 0.311), Common Nighthawk (bitterbrush canopy cover 0.332 and density of bitterbrush 0.316), and American Goldfinch (distance to snags –0.212). Guild assignments are shown on Table 3.5. Figure 3.1 provides comparisons of each guild and the five habitat successional stages.

Table 3.5. Bird species listed within each guild designation.

Open-habitat Guild	Habitat-generalist Guild	Edge Guild	Mature-tree Guild	Closed-canopy Guild
Brewer's sparrow Mountain bluebird Savannah sparrow Vesper Sparrow Western Meadowlark	American Goldfinch Black-capped Chickadee Common Nighthawk Northern Flicker Pine Siskin Ruby-crowned Kinglet Townsend's Solitaire	American Robin Brown-headed Cowbird Chipping Sparrow Clark's Nutcracker Dark-eyed Junco	Brown Creeper Evening Grosbeak Hammond's Flycatcher Northern Goshawk Red-breasted Nuthatch	Dusky Flycatcher Hairy Woodpecker Hermit Thrush Mountain Chickadee Warbling Vireo Yellow-rumped Warbler

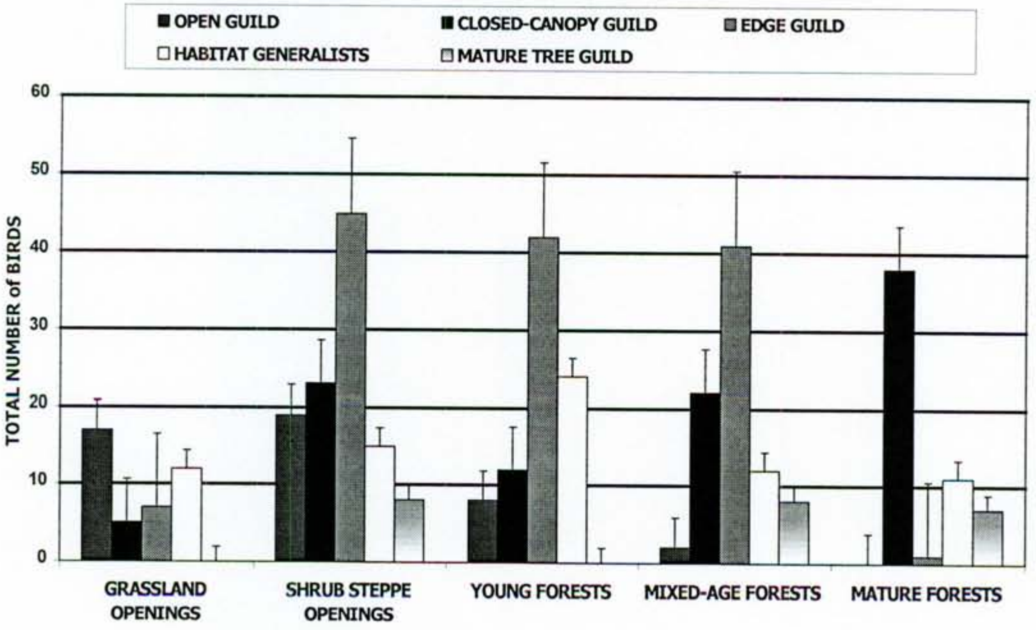

Figure 3.1. Use of the five designated succession stages by five bird guilds with standard error bars.

Result of Guild Regression on Habitat Variables:

The specific importance of habitat variables to bird guilds using this ecotone were examined with linear regression models. Linear regression on the Open Guild identified two habitat variables that explained 63% ($r^2 = 0.397$) of the open guild bird distribution. These two variables were area of opening ($p = 0.049$) and grass yield ($p = <0.001$). This equation was Y = -0.380 + (0.014 x Area Opening) + (0.005 x Dry Grass in kg/ha).

The mean area of openings used by this guild is 37.47 ha with a 95% confidence interval of plus or minus 24.07 ha. The highest density of birds (6.35) was found in three different sized openings (13.84, 22.59, and 54.99 ha).

Another important variable for Open Guild bird species is grass biomass (dry weight). Mean grass biomass for all sites was 276.1 kg/ha (SE 31.7kg/ha). This overall mean does not correspond to plots with the highest densities of Open Guild bird observations. The grass biomass on sites with the highest density of birds was 535 kg/ha. This number fits more with the mean from Grassland Openings than with any other succession stage, including Shrub Steppe Openings.Grass biomass and opening size explain almost 65% of the distribution of the Open Guild, but only three species in this guild, Mountain Bluebird, Savannah Sparrow and Brewer's Sparrow, were strongly correlated with grass biomass, the other two species were significantly correlated with the shrub layer. Shrub density was significant for both Vesper Sparrow ($p = 0.010$) and Western Meadowlark ($p = 0.056$). Vesper Sparrows were also strongly correlated with a particular shrub, sagebrush ($p = 0.039$), but Western Meadowlarks were not ($p = 0.501$). The last variable discarded before the linear regression model concluded was shrub density ($p = 0.232$). Since two of the five species in this guild have an association with the amount of shrub density, this may be an important variable. Mean shrub density for the distribution of birds in the Open Guild was 6145 shrubs per ha (95% confidence limit of 3393 to 8898 shrubs per ha). The highest numbers of birds were observed at shrub densities of 4020 shrubs per ha.

The Edge Guild distribution was explained by only one variable and that was a negative correlation with distance to nearest change habitat structure. This variable explained 39% ($R^2= 0.152$) of the distribution for this grouping ($p = 0.008$). Usually, edge-associated species are not of concern, but two species in this guild are listed as declining or needing study, Clark's Nutcracker and Chipping Sparrow. The equation for this guild is $Y = 3.493 + (-0.017 \times \text{Distance to Change in Height Structure})$.

Closed-canopy Forest Guild regression model consists of two habitat variables. These are a negative correlation with distance from snags ($p = 0.060$) and a positive correlation with canopy cover of trees ($p = < 0.001$). These two variables account for 68% of the distribution pattern for this guild ($R^2 = 0.488$). Mean tree canopy cover for this guild's distribution is 34% (95% Confidence Interval of 24% to 42%). The regression equation for this guild is $Y = 1.272 + (-0.005 \times \text{Distance to Nearest Snag}) + (0.063 \times \text{Tree Canopy})$.Mean distance to snags for this group is 140 meters (95% CI of 98 to 182 meters). The highest density of Closed-canopy birds was observed at a mean distance of only 58 meters from the four nearest snags. A mean of 2.74 snags per site (95% CI of 1.9 to 3.5) were also associated with the birds in this guild.

Mature Tree Guild birds were found on sites that averaged 56.6% canopy cover (95% CI of 35.0 to 56.6). This canopy is a function of the larger stature trees associated with sites where the species in this guild were observed. The average number of large trees per site was 2.4 (95% CI of 0.5 to 4.2). When linear regression was run on this guild and all habitat variables, the variable 'Mature Tree' explained 100% of the distribution of this species ($R^2 = 1.0$). [Equation: $Y = 1.27 \times \text{Number of Mature Trees}$.] This is probably a function of this variable being the most important one found when using Spearman's rank order analysis to place birds into this guild. The greatest density of birds were observed on sites with six mature trees. (Six was the maximum number of mature trees measured on any plots.) When the 'Mature Tree' variable was deleted from the independent variable list and the linear regression model for this guild was run again, two variables explained 44% of the distribution ($R^2 = 0.19$), canopy cover of trees ($p = 0.006$) and a negative correlation with nearest deciduous tree location ($p = 0.029$). (The only deciduous trees that were encountered and measured for this study were aspen (*Populus tremuloides*)). The greatest density of birds for this guild was found at about 169 meters from these trees. The average distance from deciduous trees for sites where species forming this guild were observed was 719 meters (95% CI of 237 to 1201). The regression equation for this guild, after removing mature trees as a variable, was: $Y = 0.533 + (-0.001 \times \text{Nearest Aspen}) + (0.064 \times \text{Tree Canopy})$.

Relative Abundance ProportionsAnother way to consider species distribution is with relative abundance proportions across succession stages. Relative abundance proportions for species use of the five habitat stages are given on Table 3.6. Relative abundance calculations are used as a measure of clustering across habitat types to determine the proportion of times a given species was observed in each succession stage. Table 3.6 indicates that either more species spend some time and/or a few species spent more time in mixed and mature forest succession stages. The hypothesis that birds were found equally in each succession stage (habitat) was rejected ($p = 0.012$). The differences in proportion of use were significant between Shrub Steppe and Mixed-age Forest stages ($p = 0.014$), between Shrub Steppe and Mature Forest ($p = 0.024$), between Young Forest and Mixed-age Forest ($p = 0.027$), and between Young Forest and Mature Forest ($p = 0.057$). On mature forest sites, the total was substantially increased by the inclusion of three species that were only found on plots within that habitat class. Mixed-age Forest and Grassland Opening sites each had two exclusive species while small tree sites had only one, and shrub sites had no species exclusive to themselves.

Table 3.6. Relative abundance proportions of 28 species detected across five habitat successional classes. Proportion of all detections for a given species within each succession stage of tree development.

Species	TOTAL#	GRASSLAND OPENING	SHRUB STEPPE	YOUNG FOREST	MIXED-AGE FOREST	MATURE FOREST
American Goldfinch	4	0.25	0.00	0.00	0.00	0.75
American Robin	4	0.25	0.25	0.00	0.50	0.00
Black-capped Chickadee	4	0.25	0.00	0.25	0.25	0.25
Brewer's Sparrow	3	0.33	0.67	0.00	0.00	0.00
Brown Creeper	1	0.00	0.00	0.00	1.00	0.00
Brown-headed Cowbird	8	0.00	0.00	0.13	0.88	0.00
Chipping Sparrow	33	0.09	0.09	0.46	0.36	0.00
Clark's Nutcracker	9	0.00	0.00	0.00	1.00	0.00
Common Nighthawk	2	0.00	0.00	1.00	0.00	0.00
Dark-eyed Junco	20	0.10	0.05	0.25	0.55	0.05
Dusky Flycatcher	26	0.04	0.00	0.12	0.35	0.50
Evening Grosbeak	8	0.00	0.00	0.00	0.63	0.38
Hairy Woodpecker	6	0.00	0.00	0.00	0.50	0.50
Hammond's Flycatcher	2	0.00	0.00	0.00	0.50	0.50
Hermit Thrush	3	0.00	0.00	0.00	0.00	1.00
Mountain Bluebird	19	0.47	0.21	0.21	0.11	0.00
Mountain Chickadee	19	0.16	0.00	0.00	0.37	0.47
Northern Flicker	10	0.00	0.00	0.20	0.80	0.00
Northern Goshawk	1	0.00	0.00	0.00	0.00	1.00
Pine Siskin	14	0.43	0.07	0.36	0.00	0.14
Red-breasted Nuthatch	3	0.00	0.00	0.00	0.33	0.67
Ruby-crowned Kinglet	9	0.11	0.22	0.11	0.33	0.22
Savannah Sparrow	1	1.00	0.00	0.00	0.00	0.00
Townsend's Solitaire	6	0.17	0.33	0.00	0.00	0.50
Vesper Sparrow	16	0.31	0.69	0.00	0.00	0.00
Warbling Vireo	3	0.00	0.00	0.00	0.00	1.00
Western Meadowlark	1	1.00	0.00	0.00	0.00	0.00
Yellow-rumped Warbler	15	0.07	0.07	0.20	0.20	0.47

In 147 bird surveys across five habitat classes, three species were encountered in every habitat category. These were Ruby-crowned Kinglet, Dark-eyed Juncos and Yellow-rumped Warblers. Seven species were only detected in one habitat class. These were Clark's Nutcracker, Common Nighthawk, Hermit Thrush, Northern Goshawk, Savannah Sparrow, Warbling Vireo, and Western Meadowlark. If these species can be confirmed by further research to be limited in association to a single habitat, the ecological consequences of losing this habitat type would be extremely important.

There were a few species that had a large enough number of total observations to individually assess the specific habitat variables each species seemed to be using. Spearman's correlations results were obtained by correlating all of the habitat factors against all of the birds by species. Most of these correlations were not significant, but three species of birds had correlations that were very close to or above 50% for specific habitat factors. The species with distinct correlations were Dusky Flycatcher, Mountain Bluebird, and Vesper Sparrow.

The presence of Dusky Flycatchers (n = 26) was positively correlated with increased canopy cover for trees (0.513), with increased density of trees (0.503), and with later succession stages (0.515). The presence of these flycatchers was negatively associated with size of opening (-0.574) and with distance to trees (-0.554). They also were negatively associated with distance to nearest curlleaf mountain-mahogany (-0.491). These correlations suggest an affinity for sites with forests and/or curlleaf mountain-mahogany vegetation.

Mountain Bluebird (n =19) presence was positively correlated with size of opening (0.523) and with the vegetation found in openings (grass = 0.497 and forbs 0.450). Snags were included in this regression. There was no correlation (0.084) with distance to large snags.

Vesper Sparrow (n =16) presence was positively correlated with size of opening (0.551) and with distance to large trees (0.519). These two variables seem to emphasize this species preference for large openings. The correlation with vegetation is a negative association with trees since their presence was negatively correlated with increasing succession stage (-0.520). In addition, this species was emphatically negatively correlated with trees both with increasing amount of canopy cover (-0.531) and increasing tree density (-0.549).

Species Richness and Diversity:

Species richness counts for each succession stage, and diversity calculations for each succession stage are given in Table 3.7. Species richness estimates are simply the number of individual species counted. The other indices are measures of diversity and evenness. They give estimates of the probability that two individuals chosen at random will be a different species. Zero would indicate no diversity, or that all individuals are the same species, and one would indicate the highest possible diversity, or that every individual is from a different species.

Table 3.7. Four species diversity measures for breeding birds across habitat successional stages. *The complement of Simpson's Index of species diversity was calculated with the formula $1\text{-}D=1-\Sigma(p_i)^2$ (Simpson 1949). ** $H'=\Sigma(p_I)(\log_2 p_I)$ which measures the uncertainty in a community of species. [A community with only one species has no uncertainty, and $H'=0$.] *** $E=H'/\ln S$ which measures the evenness across habitats.

	Grass Openings	Shrub Steppe Openings	Young Forests	Mixed-Age Forests	Mature Forests
Species Richness	16	10	11	17	16
Simpson's Index*	0.880	0.793	0.819	0.910	0.886
Shannon Function**	3.481	2.778	2.932	3.688	3.538
Shannon Evenness***	1.256	1.206	1.223	1.302	1.276

Discussion

Habitat relationships between species of birds and their distribution are complex. Some species are generalist while others restrict their use to specific types of habitat. The difficulty in assessing important habitat characteristics can be seen by looking across the relative proportion use of the five succession stages in Table 3.6. The majority of species detected apparently use more than one stage. Four of the seven species detected in only one habitat type can be seen to have been single encounters. It can be argued that single observations should be removed from the data set, but the significance, ecologically, of single encounters is often very important. Of the four species observed only once during the counts, three are listed as species of special concern by various bird monitoring research projects (Hutto and Young 1999, Paige and Ritter 1999, Robbins *et al.* 1992, Saab and Rich 1997). These are Northern Goshawk, Brown Creeper, and Western Meadowlark.

The strong correlations of Mountain Bluebirds with forage biomass over succession stage and number and distance to snags is interesting. If Mountain Bluebirds prefer openings with higher grass and forb production, whatever the succession stage, perhaps there are higher populations of insect on sites with greater herbaceous biomass. Other research finds an association between Mountain Bluebirds and post-burn sites (Hutto and Young 1999). Since they are cavity users, it is usually assumed that the dead snags left from burns are the main attraction for them. Perhaps increased herbaceous biomass is also attracting them to burned sites.

The Simpson and Shannon diversity indices for these five succession stages are all high. This indicates that all habitat stages have high value for a variety of species in spite of confining bird detections to the area within a fifty-meter radius of the plot center (0.78

ha). Since these high indices may be a product of the interspersion of the five types across the landscape in this ecotone, the restoration of historical patterns of openings and forest seral stages may be extremely important.

The Breeding Bird Survey (BBS) has been tracking trends in bird abundance for more than thirty years. Researchers have been concerned about forest fragmentation effects on bird populations (Robbins *et al.* 1992). There is also concern for non-forest interior species whose populations are in long-term declines in the West. Of the five species placed together in the Open Guild, three have populations that are considered to be in long-term decline, Brewer's Sparrow, Vesper Sparrow, and Western Meadowlark (Paige and Ritter 1999, Saab and Rich 1997). Hutto and Young (1999) consider Brewer's Sparrows to be one of the most severely declining songbirds. Although I included both shrub density and shrub canopy cover in the regression analysis, I found significant correlations only to grass biomass and increasing distance to tree line. This may be because the total census for these sparrows was so low (n = 3). Walcheck (1970) found Brewer's Sparrows were by far the most abundant species in sagebrush-grassland in Montana. Other research on the effects of using herbicides to kill sagebrush found breeding pairs of Brewer's Sparrows declined by 54% on plots with total kill of sagebrush (Best 1972). Open areas with sagebrush are important to this species because they nest in shrubs. Prescribed burns that decrease tree encroachment may help ease the long-term decline in Brewer's Sparrow abundance if sagebrush returns to the burned areas.

Vesper Sparrows (n =16) were strongly associated with total forb and grass biomass. Their distribution pattern was not random for either sagebrush or bitterbrush. They were positively associated with sagebrush and negatively associated with bitterbrush. The negative association with bitterbrush may possibly be explained by more by site characteristics for areas that have healthy bitterbrush stands rather than by the shrub species itself. These sites tend to have less grass and more mineral soil exposed. There was no significant change in numbers of breeding pairs of this sparrow in research that used spray to kill sagebrush (Best 1972). This may be because Vesper Sparrows nest on the ground and the spray used did not affect grass canopy. This study was of two year duration and spraying did not affect the overhead structure of sagebrush in that short time period, the skeletons of dead sagebrush still stood as a form of overstory.

The third species of concern, Western Meadowlark, shows significant correlations with sagebrush canopy and shrub density, but it was one of the species detected in only one habitat type, Grassland Openings. These sites were characterized by having an important grass component even when shrubs are present.

Conclusions

This research found that all five habitat succession stages that make up the ecotonal variations in this region of Montana are extremely important to the diversity of bird life. Diversity indexes indicate that all five of the habitat classifications investigated (grassland, shrub steppe, young forest, mixed-age trees with some under story, and mature forest areas with little under story) have high bird species diversity. This is convincing evidence that prescribed burns could be a valuable tool to increase habitat for land birds in this area whether the burn is able to regenerate bitterbrush or not. Two striking associations are apparent. First, bird species that were found to be significantly associated with areas of substantial grass and sagebrush are also species of concern in the West. Second, bitterbrush is apparently not as important as sagebrush, grass, and openings in general to most bird species that use this ecotone. Only Townsend's Solitaire and Common Nighthawks showed a correlation with areas of higher than normal bitterbrush cover.

In addition to overall diversity, several species seem to be dependent on distinct succession stages in this ecotone. This is especially true for the five species making up the Open Guild and perhaps for two of the species associated with greater forest canopy. Since these species occupy the whole spectrum of habitat stages, I believe that it is critical to hold onto mature stands of trees, while using fire as a tool to open up sites with high young tree density over a period of decades. Where it is still possible to delineate some of these old sagebrush-grass openings, they should be burned even if tree encroachment is far advanced. Historical data indicates a fire interval of approximately 45 years (Arno and Gruell 1983, 1986). This would be a good threshold to aim for if climatic factors allow encroachment to persist. Although the mean size of openings is probably not as crucial as simply increasing the number of openings, to be effective, it will be necessary to burn large enough areas that shrub steppe species such as sparrows will be induced to live there. A wide range of openings, from 14 to 55 ha, were found to support high densities of open habitat birds.

Chapter IV

Changes in Habitat Variables with Spring and Fall Burns

Introduction

Douglas-fir and Rocky Mountain juniper have been increasing in area in Southwestern Montana for decades (Arno and Gruell 1986, Sindelar 1971). Because of perceived changes in the shrub component due to this long-term encroachment, research was begun in 1996 to chronicle the effects of returning fire, one of the major historical disturbance factors to the area. Lack of fire due to heavy grazing early in the 20th Century and more than fifty years of active fire suppression has long been held to be the primary cause of tree encroachment (Arno and Gruell 1983, 1986). Sindelar (1971) believes that pulses of forest expansion occur after years of higher than normal spring rainfall. He felt that fire suppression was not a causal factor in tree invasion in his study area, but that fire suppression was important because it allowed newly established tree seedlings to persist. If burning is recommended as a method of restoring openings in this ecotone, the probability of success needs to be determined for the later stages of forest succession that are beginning to be predominant over the landscape.

In 1996, twelve macroplots were established in the Toll Mountain area of Jefferson County, Montana, to study the long-term effects of prescribed burning on various habitat factors. Six of the twelve plots were set aside as controls with no treatments prescribed. The remaining six plots were divided into two groups. Three of these plots were assigned for fall prescribed burns and three plots for spring prescribed burns. These twelve plots were permanently marked so that long-term monitoring could be carried out within them. The objectives of this research are to determine the effects of prescribed fire on late seral tree encroachment sites. Specific outcomes for differences before and after fire for vegetation, small mammals and birds will be tested for significance against the hypothesis that fire causes no change in density and/distribution of these.

Methods

Before burn treatments were initiated, three 60-meter permanent transects were fixed thirty meters apart and parallel to each other within each of the twelve study plots. These transects were marked with re-bar and witness posts. Baseline data were collected in 1996 along these transects on vegetation characteristics and on small mammals. Bird survey data were collected from the center of each 60 x 60-meter plots. The plots received their treatments in the fall of 1997 and the spring of 1998. The plots were re-surveyed in the summer of 1998, and some habitat variables were re-surveyed in 1999. The methods used for these surveys will be discussed here.

Vegetation:

In order to measure vegetation changes at several structural levels, four types of vegetation surveys were carried out within each macroplot. In addition, photo points were established and used in all years. Canopy cover estimates were measured using 20 x 50 cm quadrat (Daubenmire 1958). The quadrat was placed and read at ten six-meter intervals along each of the three transects within each macroplot. The canopy coverage of each vascular plant with leaf area occurring over the qudrat was included. All individuals of a given species that intersected over the quadrat were estimated as a single unit. Each one of these estimates was assigned to one of seven categories. This is one more category than originally recommended (Daubenmire 1958). The mid-point of each category was used to obtain average canopy cover per macroplot for each species. Plants did not need to be rooted within the quadrat to have coverage over it counted. The frame was painted in a checkerboard orange and white pattern to give visual reference at 5, 25, 50, 75, and 95% of its area. The quadrat was placed consecutively along each 60-meter transect at six-meter intervals for a total of ten frames per transect, thirty frames per macroplot. This information was also translated to percent frequency for each species encountered.

Cover Class	Range of Coverage	Midpoint
1	0-1%	0.5%
2	1-5%	2.5%
3	5-25%	15.0%
4	25-50%	37.5%
5	50-75%	62.5%
6	75-95%	85.0%
7	95-100%	97.5%

The quadrat was also used to obtain ground cover estimates. Ground cover categories include basal vegetation cover, bareground, rock, cryptogramic crust, and litter estimates. For ground cover estimates, the plants had to be rooted within the quadrat in order to be counted. Ground cover estimates sum to 100% for each quadrat. (Shrub line intercept and density methods were given in Chapter II and will not be repeated here.)

Small Mammals:

Deer mice have an effect upon bitterbrush seedling establishment due to their propensity for caching and eating bitterbrush seeds (Vander Wall 1995). They are believed to be instrumental in moving and expanding the range of bitterbrush establishment. This is important because, unlike sagebrush seeds, bitterbrush seeds are too heavy for wind-blown dispersal.

If natural regeneration is to take place, there needs to be a healthy population not only of bitterbrush, but also of deer mice and chipmunks facilitate regeneration. The range of variability in deer mice populations can be and usually is enormous. In order to estimate the difference, if any, in the distribution of deer mice both before and after burning, trapping grids were set out in 1996 along each transect on each macroplot. A set of control and treatment macroplots was always trapped on the same nights to offset weather differences. Sherman live-traps were placed at 10-meter intervals along each transect and baited with rolled oats and peanut butter. Trapped mice were weighed, sexed, aged, marked, and released. The traps were reset and re-measured the next day. This same procedure was followed again in 1998 after the prescribed burn treatments to check for any short-term changes in distribution. These trapping efforts were also used as a measure of species richness for the twelve macroplots. Collecting a small mammal species list for the study sites was important because it was originally hypothesized that lack of seed-caching rodents could be a factor in the decline in bitterbrush.

Birds:

The effects of habitat fragmentation on bird species are mixed in the research literature. The effects of re-introducing fire into what has become a less fragmented landscape with respect to the forested vegetation are therefore of major concern to biologists for this reason. Because of the small burn size, it was difficult to limit the scope of the bird surveys to only the area burned. Therefore, bird surveys were strictly divided into two segments: those within a 50-m radius of the center of the macroplot and those outside that radius. Only birds heard or seen within 50 meters of plot centers are used for analysis.

Bird surveys were carried out in late May and early June in 1996, 1998, and 1999. A series of three surveys were completed in each year. The surveys consisted of standing in the center of each macroplot between the hours of dawn and 11:00 a.m. on days with no wind and no heavy rain for a total of eight minutes. During this time all species detected were recorded and their location estimated. The time period was divided into two segments to guard against counting birds that move frequently but still include birds that sing at less frequent intervals. These time periods were of three and five minute duration.

Prescribed Burns:

Objectives set for the burns were to reduce fuel loading over the area from 8 to 12 metric tons per ha down to 2 to 3 metric tons per ha, to retain soil protection by retaining at least 60% ground cover (litter, basal vegetation, and rocks), to reduce sapling component by 70 to 100%, to retain 80 to 90% greater than 20 cm in diameter, and to create openings that are between 2 and 12 ha in size. The overall purpose of the prescribed burns was to move plant communities into early succession stages for ecosystem maintenance and to reduce the risk of a major wildland fire by reducing accumulated fuels.

Fall Burns:

The fall treatment plots were ignited September 9, 1997. Relative humidity was 38% with calm winds and 1000 hour fuel moisture at 14%. Average temperature was 17° C with a maximum temperature of 29° C. Flame length was measured at 1.2 to 1.8 m with an 2.4 m maximum. Rate of spread was 300 to 400 m per hour with good smoke dispersion and a 183 m spotting potential. Mop-up continued until September 11. Final mop-up was mainly due to deep duff and punky logs on part of the area. Because of the absence of fire lines on 50% of the area risk of escape was considered to be moderate.

Spring Burns:

The spring burns were ignited on different days and will be explained individually. There were still patches of snow along the edges of some units when these fires were ignited. When ignited, 1000-hour fuel moisture was 25%. Unit B4 and B6 were ignited April 20, 1998. Relative humidity was 37%. Average temperature was 6 ° C with a high of 16° C. Flame length was 0.6 to 1.5 m with variable spread, good to drifty smoke column, and low spotting potential. The next day units B4A and B5 were ignited. The average air temperature for that day was 6° C with a high of 18° C. The fire on site B4A had flame lengths of 0.6 to 1.2 m with poor spread, drifty smoke and low spotting potential. Unit 5 burned with 0.6 to 3 m flames at 60 m per hour, good smoke column, and moderate spotting potential. Erratic winds made Unit 5 more of a concern. By April 26, all visible smokes were out on Unit 5. By the following day all of the spring burns were considered to be out.

Fire-caused Tree Mortality:

Tree mortality was significantly greater on two of the three fall burns compared to the spring burns. One fall burned plot had an estimated mortality of 87%. Scorch heights varied on fall burns from 0.8 to 15 m with a mean scorch height of 3.99 m and standard error of 2.53 (n=13). There was no tree mortality caused by fire on the three spring burn research plots. Loss of tree canopy and density within the surveyed area of these plots were solely from the prefire thinning. [It should be noted that, although the burns carried out on this research project were of three different sizes, they could all be considered to be relatively small varying in size from 5 to 0.5 ha.]

Results

Vegetation:

In 1996, before the burn treatments, mean total tree canopy cover for all the plots was 37.9%. The results of this research, therefore, should be considered most applicable to encroachment sites with a similar canopy cover of trees. By species, 82.5% of this canopy cover was composed of Douglas-fir. The remaining 17.5% was composed of Rocky Mountain juniper.

After burn treatments, mean canopy cover for Douglas-fir decreased on fall burn plots to 14.0% in 1998 and remained virtually the same in 1999 (14.5%). Post-burn spring burn canopy cover for Douglas-fir was down to 25.1% in 1998 (25.58 in 1999). Table 4.1 and Figure 4.1 show the results for canopy cover for these three years. The only significant difference between Douglas-fir canopy cover before and after prescribed burns was for the fall burn treatments (p=0.056).

Table 4.1. Mean percent canopy cover for Douglas-fir before and after prescribed burns. Standard error bars given in parentheses. *Pre-burn year.

	1996*	1997	1998
Fall Burns	33.30 (6.79)	14.02 (4.31)	14.50 (3.79)
Spring Burns	32.60 (0.96)	25.13 (6.97)	25.56 (8.00)
Fall Controls	27.81 (7.71)	31.24 (9.17)	32.78 (4.17)
Spring Controls	32.02 (2.52)	30.91 (5.44)	30.91 (3.63)

Total mean tree density before burning was 2824 trees per ha. Douglas-fir comprised 55.8% of the pre-burn density; Rocky Mountain Juniper made up the remaining 44.2%. The changes in tree density with both fall and spring burns was substantial (Table 4.2). Part of the change is due to pre-burn thinning of small trees. There was no actual mortality within the spring burn transects. Analysis of variance (ANOVA) indicates a significant difference in density of Douglas-fir after the fall burns ($p = 0.034$). There is not a significant difference between the two postburn years ($p = 0.902$). There is also a significant difference between the postburn fall treatment plots for Douglas-fir when they are compared to the controls ($p = 0.009$).

Spring burns did not have the same effect on Douglas-fir density as fall burns. There was no significant difference between mean Douglas-fir densities on the spring-burned plots before and after one ($p = 0.206$) or two post-burn years ($p = 0.135$). The difference in density between post-burn spring treatment and post-burn control plots is moderately significant ($p = 0.066$). This difference in density is from pre-burn thinning of small diameter trees on the burn plots rather than to fire mortality.

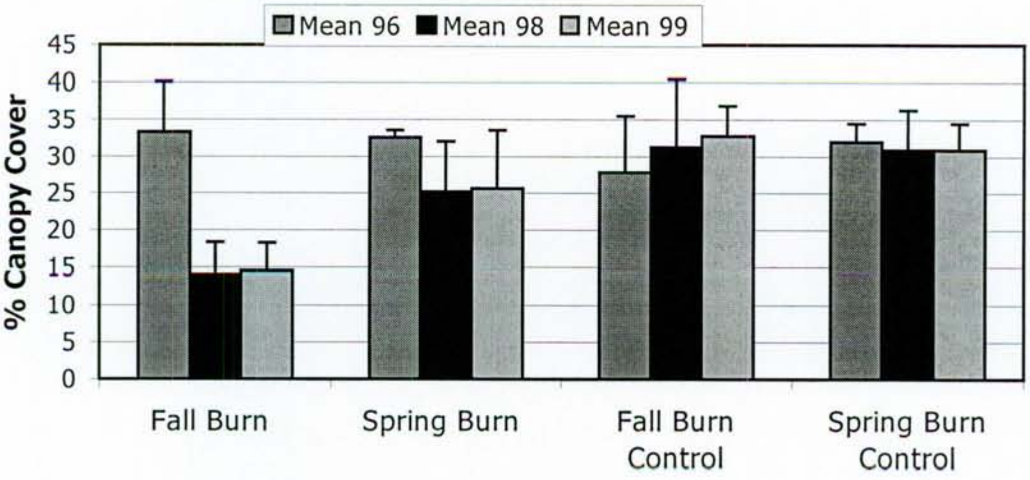

Figure 4.1. Changes in mean percent canopy cover for Douglas-fir before (1996) and for one and two years after burn treatments (1998 and 1999). Standard error bars are included.

Table 4.2. Mean tree density (#/ha) by species before (1996) and after burns. Standard error estimates for these means are in parentheses.

	Douglas-fir Density 1996	Douglas-fir Density 1998	Douglas-fir Density 1999	Juniper Density 1996	Juniper Density 1998	Juniper Density 1999
Fall Burns	2259 (691)	74 (37)	222 (169)	1685 (249)	130 (49)	389 (309)
Spring Burns	1241 (630)	278 (111)	630 (152)	1389 (338)	463 (148)	1111 (285)
Fall Controls	852 (312)	926 (324)	704 (241)	889 (116)	1204 (383)	944 (326)
Spring Controls	2111 (1124)	1407 (629)	1000 (402)	870 (321)	852 (277)	667 (222)

The total pre-burn mean percent canopy cover for bitterbrush was 0.08% of the study area. Sagebrush canopy covered approximately 1.67% of the area. Total shrub canopy was 1.75%.

The 1998 post-burn canopy cover estimates for bitterbrush were zero for fall treatment plots and 0.33% for the fall controls. By 1999, the mean cover for bitterbrush was still zero for fall treatment plots and 0.23% for controls. Compared to pre-burn bitterbrush canopy estimate of 0.11% for fall treatment plots the decrease is 100% but it is too small for significance. Bitterbrush canopy coverage for the pre-burn control for fall plots did not change significantly (0.17%).

Sagebrush canopy covered approximately 1.9% of the fall burn plots before the burns. The first year post-burn coverage was down to 0.29%, a statistically significant decline ($p = 0.014$), and the second year post-burn, it was still less than 1% canopy (0.42%). Spring burns did not cause a change in sagebrush canopy ($p = 0.579$) with treatment, probably because two of the three burn areas had no sagebrush intersecting the 180 m of transect lines either before or after the burns.

Pre-burn shrub densities were low over all the plots, reinforcing the concern for the loss of this habitat component. Mean total shrub density for all pre-burn plots was 1794 shrubs/ha. Of this density, 14% (or 255 plants/ha) was bitterbrush; the remainder was mountain big sagebrush. Post-burn shrub densities were significantly down on fall burn plots (p = 0.0002). Mean total shrub density for all control plots post-burn was 2180 shrubs/ha. By 1999, fall post-burn and spring post-burn plots have a mean density of 686 and 147 shrubs/ha, respectively.

Sagebrush density on fall burn plots decreased from a pre-burn mean of 1423 plants/ha to a low in 1998 of 222 plants/ha (p = 0.001). By 1999, the sagebrush component on fall burn plots was already on the increase with mean densities up to 611 plants/ha. Although this is still a significant decrease over pre-burn sagebrush densities (p = 0.004), this is a three-fold increase from the first post-burn year densities and mostly due to sagebrush regeneration (Table 4.3).

Ground cover surveys included five categories (Table 4.4). These were ground cover, percent canopy cover by species, seedling counts, and frequency counts.

Table 4.3. Mean density estimates for shrubs by species in plants per hectare with standard error in parentheses. *1996 is pre-burn.

	Fall Burns	Fall Controls	Spring Burns	Spring Controls
Mountain Big Sagebrush *1996	1423 (125)	3043 (1125)	519 (412)	1056 (686)
Mountain Big Sagebrush 1998	222 (85)	2759 (955)	19 (19)	1128 (679)
Mountain Big Sagebrush 1999	611 (64)	2574 (877)	73 (37)	869 (544)
Bitterbrush *1996	313 (127)	210 (163)	167 (116)	444 (255)
Bitterbrush 1998	56 (56)	333 (147)	37 (19)	370 (236)
Bitterbrush 1999	74 (74)	241 (130)	73 (37)	444 (255)

Table 4.4. Total mean ground cover for all sample plots before burning (1996) and after burning (1998). Standard error estimates are given in parentheses.

	Cryptogramic Crust (%)	Basal Vegetation (%)	Bare Ground (%)	Rock (%)	Litter (%)
Fall Burns 1996	5.1(1.0)	13.0 (5.8)	5.2 (2.7)	3.4 (2.0)	73.3 (6.7)
Fall Burns 1998	4.8 (1.1)	6.4 (1.5)	8.7 (3.2)	2.8 (1.8)	77.4 (2.0)
Fall Controls 1996	6.6 (1.3)	14.4 (4.1)	10.2 (5.1)	6.7 (2.3)	62.1 (10.6)
Fall Controls 1998	21.2 (3.7)	7.9 (1.7)	8.9 (5.5)	3.4 (0.7)	58.6 (6.9)
Spring Burns 1996	1.4 (0.8)	3.3 (1.3)	6.3 (2.0)	7.0 (5.2)	82.0 (3.7)
Spring Burns 1998	7.2 (2.2)	4.8 (1.1)	6.9 (1.4)	10.1 (5.7)	71.0 (5.1)
Spring Controls 1996	3.8 (2.7)	4.5 (1.1)	9.8 (2.8)	5.0 (2.7)	76.9 (7.4)
Spring Controls 1998	15.9 (4.8)	5.3 (0.8)	5.6(2.0)	3.8 (2.0)	69.4 (9.5)

Species richness increased by two with fall burning (Table 4.5). This represents a net loss of two species and a gain of four species on the fall burn plots. Species richness counts for the fall controls increased by three species during the same time period. Species richness on spring burns had a net loss of one species (Table 4.6). It is interesting that, for the same period on the controls, there was a net gain of 10 species.

Excluding the changes in trees canopy coverage with burning that have already been discussed, all other species were considered to see if there were significant changes in understory vegetation spring or fall burns. Before the prescribed burns, there was no significant difference between pre-burn fall treatment plots and their controls (p=0.301) nor between pre-burn spring treatment plots and their controls (p=0.487) for all other species. After fall burns, there was a moderately significant difference in pre- and post-fall burn understory species by canopy coverage (p = 0.088). Interestingly, there was no significant difference between fall post-burn and their controls by canopy cover (p = 0.109). When spring treatment plots were measured before and after the treatment, there was a significant difference between pre- and post- spring burn canopy covers (p =0.026) between 1996 and 1998. Even on the control plots, there was a moderately significant difference between the spring controls (p = 0.066) for the same two time periods.

Three new species that appeared on fall burns are species that are associated with disturbance: plaintain *(Plantago lanceolata)*, dandelion (*Taraxacum officinale*) and arabis (*Arabis* sp.). The other new species was blue grama (*Bouteloua gracilis*). Ten understory species increased with fall burning and eight decreased. Most of the changes were inconsequential, but bluebunch wheatgrass (*Agropyron spicatum*) decreased by about 3% and Idaho fescue(*Festuca idahoensis*) decreased by over 5% with fall burning. On the fall controls, *Agropyron spicatum* increased during the same period while Idaho fescue decreased by nearly 6% even more than with burning. There was 1.5% increase in needlegrasses (*Stipa* spp.) and appearance of blue grama (*Bouteloua gracilis*) (0.17%) with fall burning.

On the spring burn treatment plots, thirteen species decreased and eight species increased. Bluebunch wheatgrass decreased by 1.2% with spring burns and *Festuca idahoensis* decreased by nearly 4%. On the control plots, bluebunch wheatgrass increased slightly (0.61%) and Idaho fescue decreased by about 1%.

Table 4.5. Canopy cover estimates by species for plants encountered on the fall burn plots and the controls before and after burning with standard error estimates in parentheses. Species richness counts are included.

Species	Fall Burn Pre-burn	Fall Burn Post-burn	Fall Control Pre-burn	Fall Control Post-burn
Agropyron spicatum	7.59 (0.34)	4.19 (0.28)	3.94 (0.04)	7.41 (0.30)
Bouteloua gracilis	0	0.17 (0.05)	0	0.01(<0.01)
Festuca idahoensis	14.76 (1.97)	8.96 (0.27)	24.72 (0.92)	18.97 (0.37)
Poa secunda	1.82 (0.55)	1.63 (0.31)	0.88 (0.17)	0.59 (0.09)
Stipa spp.	0.01 (<0.01)	1.52 (0.40)	0	0.44 (0.14)
Total grass	24.18 (2.34)	16.47 (1.32)	29.55 (0.87)	27.43 (0.90)
Achillea millefolium	0.10 (<0.01)	0.78 (0.05)	0.33 (0.05)	0.39 (0.10)
Agoseris glauca	0.06 (0.01)	0	0.33 (0.11)	0.09 (0.03)
Allium spp.	0.02 (<0.01)	0.05 (0.01)	0	0.04 (0.01)
Antennaria spp.	3.14 (0.11)	1.62 (0.01)	1.98 (0.16)	1.93 (0.22)
Arabis sp.	0	0.01 (<0.01)	0	0
Arenaria spp.	0.33 (0.06)	0.16 (0.03)	0.12 (0.02)	0.91 (0.06)
Eriogonium spp.	0.01 (<0.01)	0.50 (0.16)	0.46 (0.13)	0.21 (0.05)
Lomatium spp.	0.02 (<0.01)	0	0.01 (<0.01)	0.03 (0.01)
Oxytropis lambertii	0.13 (0.02)	0.20 (0.06)	0	0
Plantago lanceolata	0	0.17 (0.05)	0	0
Taraxacum officinale	0	0.03 (0.01)	0	0
Zigadenus venenosus	0.06 (<0.01)	0.01 (<0.01)	0	0.06 (0.01)
Total forbs	3.86 (0.09)	3.53 (0.39)	3.24 (0.28)	3.65 (0.47)
Artemisia tridentata	3.07 (0.09)	0.94 (0.30)	3.06 (0.48)	3.64 (0.59)
Purshia tridentata	0	0	1.11 (0.19)	0
Total shrubs	3.07 (0.09)	0.94 (0.30)	4.17 (0.39)	3.64 (0.59)
Juniperus scopulorum	3.27 (<0.01)	0.94 (0.30)	5.33 (0.71)	4.14 (0.23)
Pseudostuga menziesii	26.23 (0.56)	17.64 (1.30)	23.75 (2.91)	30.47 (3.28)
Total trees*	29.49 (0.56)	18.58 (1.60)	29.08 (2.33)	34.61 (3.51)
Species Richness	16	18	13	16

*These canopy estimates are from 20 x50 cm quadrat cover estimates, therefore they do not directly equal those collected from the line-intercept reported earlier (Table 4.1).

Table 4.6. Mean canopy cover estimates by species for plants found on spring burn plots and controls before and after burning. Species richness counts are included.

Species	Spring Burn Pre-burn	Spring Burn Post-burn	Spring Control Pre-burn	Spring Control Post-burn
Agropyron spicatum	5.36 (0.82)	4.08 (0.91)	4.41 (0.42)	5.02 (0.16)
Bouteloua gracilis	0	0.01 (<0.01)	0.42 (0.13)	0.19 (0.06)
Festuca idahoensis	13.10 (1.48)	9.59 (0.51)	15.47 (0.69)	14.37 (1.24)
Poa secunda	1.22 (0.30)	0.0 (<0.01)	1.27 (0.18)	0.98 (0.11)
Stipa spp.	0.17 (0.05)	0.03 (0.01)	0.01 (<0.01)	0.17 (0.05)
Carex geyeri	0	0	0.01 (<0.01)	0.03 (0.01)
Total grass	19.85 (2.04)	13.72 (1.40)	21.57 (1.00)	20.75 (1.36)
Achillea millefolium	0.27 (0.08)	0.59 (0.09)	0.21 (0.07)	0.03 (0.01)
Agoseris glauca	0.03 (0.01)	0.05 (0.01)	0	0
Allium spp.	0.06 (0.01)	0.28 (0.06)	0.03 (0.01)	0.17 (0.05)
Antennaria spp.	2.41 (0.58)	0.77 (0.20)	1.32 (0.22)	0.59 (0.17)
Arabis sp.	0	0	0	0.01 (<0.01)
Arenaria spp.	0.74 (0.14)	0.80 (0.07)	0.51 (0.12)	0.51 (0.09)
Artemisia frigida	0.33 (0.11)	0.03 (0.01)	0	0.01 (<0.01)
Delphinium sp.	0	0	0	0.17 (0.05)
Descurainia pinnata	0	0	0.01 (<0.01)	0
Erigonium spp.	0.53 (0.17)	0.04 (0.01)	0.37 (0.06)	0.03 (0.01)
Erigeron sp.	0	0	0	0.06 (0.01)
Geum triflorum	0.03 (0.01)	0.00	0	0
Lomatium spp.	0.18 (0.04)	0.01 (<0.01)	0.01 (<0.01)	0
Oxytropis lambertii	0.01 (<0.01)	0.17 (0.05)	0	0.03 (0.01)
Plantago lanceolata	0	0.03 (0.01)	0	0
Taraxacum officinale	0	0.00	0	0.03 (0.01)
Viola adunca	0.03 (0.01)	0.00	0	0
Zigadenus venenosus	0	0.01 (<0.01)	0	0
Total forbs	4.59 (0.82)	2.77 (0.51)	2.44 (0.25)	1.65 (0.43)
Artemisia tridentata	0.58 (0.18)	0	1.56 (0.49)	0.17 (0.05)
Chrysothamnus viscidiflorus	0.03 (0.01)	0.17 (0.05)	0	0
Purshia tridentata	0.01 (<0.01)	0	0.17).05)	0.03 (0.01)
Total shrubs	0.62 (0.18)	0.17 (0.05)	1.73 (0.47)	0.20 (0.06)
Juniperus scopulorum	3.72 (0.49)	1.08 (0.34)	2.71 (0.54)	6.28 (1.24)
Pseudostuga menziesii	32.60 (0.30)	20.47 (1.70)	32.02 (0.80)	27.83 (1.80)
Total trees*	36.32 (0.75)	21.55 2.05)	34.72 (0.74)	34.12 (3.04)
Species Richness	20	19	11	21

*These canopy estimates are from 25 x50 cm quadrat cover estimates, therefore they do not directly equal those collected from the line-intercept reported earlier (Table 4.1).

Small Mammals

Four small mammal species were identified from live trapping grids on the research plots. Deer mice were the only species trapped in large enough numbers to analyze. Besides deer mice, boreal redback vole (*Clethrionomys gapperi*), yellow pine chipmunk (*Eutamias amoenus*) and least chipmunk (*Eutamias minimus*) were also identified. In 1996, trapping efficiency for deer mice was 78.8% over 504 trap nights. The efficiency for the same year for both chipmunk species together was less than 1% (0.99%) and for the boreal redback voles the trapping efficiency was even less at 0.20%. The high percentage of traps occupied by deer mice in 1996 is a result of extremely high densities of deer mice in the research area that year.

Trapping efficiency for deer mice declined substantially in 1998 to 24.8% for the same number of trap nights as 1996 (504). In 1998, there was an increase in trapping efficiency for the same two species of chipmunks, 3.57%. There was also an increase for boreal redback voles to 0.79%. These increases are probably a function of increased access to empty traps due to the overall decline in deer mice numbers.

The Lincoln-Peterson model using Chapman's modification for variance determination was used to estimate mice abundance on mice trapped before and after the prescribed burns. Density estimates for 1996 were very high, a mean value of 89 mice per ha and standard error of 24.57. This is much higher than literature considers normal density. By 1998, the mean density estimate was still 16 mice per ha with standard error of 4.97. The extreme change in population abundance is not unusual for rodents, but it makes it difficult to clearly define differences among treatments. First, treatment plots were compared to their controls before and after burn treatments. These paired plots were always trapped as a unit for this reason. The only significant change in deer mice abundance after burning was between the fall burns and their controls (p = 0.031). In order to try to remove as much variation caused by population changes between years, the proportional change (λ = number postburn/number preburn) for each plot before and after treatment was calculated. Wilcoxon sign rank tests were done on the resulting proportion of change. The proportion of the drop in deermice numbers on fall burn plots did not significantly differ when compared against spring treatments and controls (p = 0.11).

Birds

Pre-burn analyses of species between proposed treatment plots show no difference in bird abundance. Spearman's rank correlations for all treatments were above 50% for all pretreatment plot combinations (Table 4.9). After treatment the correlation between birds counted on the burned plots and their controls decreases. In order to assess what this decrease means, other assessment methods were used. Table 4.7 and 4.8 shows changes in number of birds detected on burn treatments and controls over a three year period.

Table 4.7. Mean number of birds noted over a three-year period on fall burn plots. The data for 1996 is before the burns.

SPECIES	Fall Burn 1996	Fall Burn 1998	Fall Burn 1999	Fall Control 1996	Fall Control 1998	Fall Control 1999
American Goldfinch	0.00	0.33	0.00	0.00	0.00	0.00
Black-backed Woodpecker	0.00	0.00	0.33	0.00	0.00	0.00
Brown-headed Cowbird	0.13	0.00	0.00	0.00	0.00	0.00
Chipping Sparrow	0.40	0.00	1.67	0.53	0.00	1.33
Common Nighthawk	0.00	0.00	0.00	0.07	0.00	0.00
Dark-eyed Junco	1.13	0.50	0.33	0.87	1.17	1.00
Dusky Flycatcher	0.73	0.00	0.00	0.53	0.50	0.33
Evening Grosbeak	0.00	0.00	0.67	0.00	0.00	0.00
Grasshopper Sparrow	0.13	0.00	0.00	0.00	0.00	0.00
Hairy Woodpecker	0.00	0.00	0.67	0.00	0.00	0.00
Hammond's Flycatcher	0.00	0.00	0.00	0.07	0.00	0.00
Mountain Bluebird	0.07	0.00	0.00	0.13	0.00	0.00
Mountain Chickadee	0.13	0.17	0.00	0.20	0.33	1.33
Northern Goshawk	0.00	0.00	0.33	0.00	0.00	0.00
Pine Siskin	1.47	3.33	0.33	0.73	1.67	0.00
Red Crossbill	0.00	0.17	0.00	0.33	0.17	0.00
Red-breasted Nuthatch	0.33	0.33	0.67	0.00	0.00	0.00
Ruby-crowned Kinglet	0.13	0.00	0.33	0.07	0.00	0.00
Townsend's Solitaire	0.00	0.17	1.00	0.07	0.00	0.00
Townsend's Warbler	0.00	0.00	0.00	0.27	0.00	0.00
Western Bluebird	0.13	0.00	0.00	0.27	0.00	0.00
Western Tanager	0.20	0.00	0.67	0.40	0.00	0.00
Yellow-rumped Warbler	0.33	0.00	1.67	0.53	0.00	1.00
Species Richness	13	7	12	15	5	5

Table 4.8. Mean number of birds noted over a three-year period on spring burn plots. The data for 1996 is before the burn.

SPECIES	Spring Burn 1996	Spring Burn 1998	Spring Burn 1999	Spring Control 1996	Spring Control 1998	Spring Control 1999
American Goldfinch	0.00	0.00	0.00	0.00	0.17	0.00
American Robin	0.00	0.00	0.33	0.00	0.00	0.00
Brown Creeper	0.00	0.00	0.00	0.00	0.00	0.33
Brown-headed Cowbird	0.00	0.00	0.00	0.20	0.00	0.00
Chipping Sparrow	0.73	0.17	0.00	0.60	0.17	0.67
Clark's Nutcracker	0.00	0.00	3.00	0.00	0.00	0.00
Dark-eyed Junco	0.67	0.00	0.00	0.73	0.33	1.67
Dusky Flycatcher	0.33	0.00	1.00	0.40	0.00	2.00
Hairy Woodpecker	0.00	0.00	1.67	0.00	0.00	0.33
Hammond's Flycatcher	0.13	0.00	0.00	0.07	0.00	0.00
Mountain Bluebird	0.13	0.00	0.00	0.20	0.50	0.00
Mountain Chickadee	0.20	0.00	1.00	0.00	0.00	0.33
Northern Goshawk	0.00	0.00	0.33	0.00	0.00	0.00
Orange-crowned Warbler	0.00	0.00	0.00	0.07	0.67	0.00
Pine Siskin	1.13	2.83	0.00	1.53	0.00	0.00
Red Crossbill	0.00	0.67	0.00	0.00	0.00	0.00
Red-breasted Nuthatch	0.40	0.17	0.67	0.07	0.00	0.00
Rock Wren	0.13	0.00	0.00	0.07	0.00	0.00
Ruby-crowned Kinglet	0.07	0.17	0.00	0.40	0.00	0.67
Townsend's Solitaire	0.00	0.33	0.33	0.00	0.17	0.00
Townsend's Warbler	0.07	0.00	0.00	0.20	0.00	0.00
Western Bluebird	0.20	0.00	0.00	0.13	0.00	0.00
Western Tanager	0.33	0.00	0.00	0.13	0.00	0.00
Western Wood Pewee	0.07	0.00	0.00	0.13	0.33	0.00
Yellow-rumped Warbler	0.53	0.50	0.67	0.40	0.50	0.00
Species Richness	15	7	9	16	8	7

Table 4.9. Spearman correlation matrix for bird species and abundance among research plots and control plots before prescribed burn treatments.

	Pre-burn 1996	Post-burn 1998	Post-burn 1999
Fall Controls	0.613	0.590	0.277
Spring Controls	0.670	0.369	0.244

Statistical analyses using the student t-test also showed no difference in mean pre-burn spring bird abundance. There was no significant difference in bird species or relative abundance between the proposed fall burn plots and their controls ($p = 0.808$), between proposed fall burn plots and proposed spring burn plots ($p = 0.825$), between the proposed spring burn plots and their controls (p = 0.802), or between the fall control plots and spring control plots (p = 0.809). After the burns, there was still no significant differences between birds on fall burn plots and their controls in 1998 (p = 0.560) or in 1999 (p = 0.140) or between spring burns and their controls in 1998 (p = 0.894) or in 1999 (p = 0.478). There were also no differences between mean bird abundance on type of burn treatment either post-treatment year, spring and fall burn plots in 1998 (p = 0.890) or in 1999 (p = 0.938).

Mean species richness numbers do change in years after burn treatments plots, but they also change on the controls (Table 4.10). Wilcoxon signed rank tests on species richness numbers before and after burn treatments showed no differences between plots in any year. There is a moderately significant difference when the Wilcoxon sign rank test analyzes the same plot through the three years for which data was collected. This includes both changes between 1996 and 1998 and changes between 1996 and 1999 (p = 0.066). There was no significant change between the two post-burn bird counts years (0.180).

Table 4.10. Species richness by treatment and year. These are total number of different species encountered across all replicates within each treatment group (n=3 blocks per treatment).

	Pre-burn 1996	Post-burn 1998	Post-burn 1999
Fall Burns	13	8	12
Fall Controls	14	5	5
Spring Burns	15	7	9
Spring Controls	16	7	7

Discussion

Changes in tree densities and canopy covers are significant for the fall burns. Density estimates showed no mortality on any of the spring burns. Canopy cover includes all tree canopy hanging over the transect line. One of the spring burns probably had a significant decrease in Douglas-fir by canopy. It was characterized as much hotter burn than the other spring burns by the fire crew, probably due to the greater fine fuel load on this plot. If intensity is hot enough, I think spring burns can be as effective as fall for removing tree overstory. In this study, fall burns resulted in the greatest reduction of tree canopy in these old openings where total pre-burn canopy cover averaged approximately 35%.

Reduction of bitterbrush density and canopy coverage was severe with all burns. I believe that one of the reasons this species did not resprout at all was because the plants were decadent and full of dead wood. Bunting *et al.* (1984) found resprouting of decadent plants to be very low. One reason may be that the abundance of dead wood holds the fire over the plant long enough to heat the soil enough to kill meristematic nodes. Another may be that decadent plants do not have enough stored resources to sprout. Nord (1965) considered the amount of moisture that a soil is able to hold of primary importance to the ability of bitterbrush to resprout. His studies found that coarse, sandy soils were not able to hold as much moisture as finer-textured soils so that fires on sandy soils were generally more lethal to bitterbrush because they were dry.

Although fire killed most of the sagebrush plants, sagebrush seedlings quickly re-established on the burned sites. It appears that there is very little sagebrush remaining in the understory under heavy tree canopy, but enough remains even at 35% canopy cover of trees to adequately reseed the burned areas.

It seems significant that there were changes in understory species with both fall and spring burns, but that spring burns had more of species decrease than did with fall burns. Perhaps the hotter fall burns were less detrimental to understory vegetation because the plants were in a state of dormancy; and, although the spring burns were less intense, the vegetation may have been more vulnerable at that time of year. Idaho fescue decreased in coverage during the time period whether it was burned in spring, fall, or not at all.

Ground cover estimates for cryptogamic crust did not change on fall burns and actually increased on spring burns. Bare ground did not change with either treatment in the first year after the burns. Burning caused extensive needle fall even when the trees lived through the fire.

Fall burns were the only treatment with significant differences in deer mice abundance. The significant decrease in these mice on the fall burns is probably a function of reduced vegetation coverage. This can be an asset because any bitterbrush seeds these mice have stored prior to the fall burn may have a greater chance of germinating if the density of mice decreases post-burn (Vander Wall 1994).

Halvorson (1982) did not find that deer mice density decreased on fall burns. He found that the effect of a hard burn was to eliminate most of the organic mantle and all small mammals, except deer mice. On his study area in western Montana, deer mice were the only species present for two years after severe burns and comprised 80-90% of small mammal numbers for five years post-burn. He felt that reduced tree canopy favored deer mice over redback voles. Other research found that deer mice use the edge of forest openings (Sekgororoane and Dilworth 1995). They found the highest relative abundance of deer mice within ten meters of forest and none were captured beyond ten meters into the openings.

Halvorson (1982) found sharp increases in deer mice numbers to be associated with heavy conifer seed crops the previous fall. He also found that deer mice numbers varied inversely with redback vole numbers, except after heavy conifer seed crop years, and that burn intensity influenced species composition as well.

Bird numbers and species richness did not differ between treatments. This could be a result of the small size of the burns. Species richness decreased overall after the burn, but decreased less on the fall burn plots. This is due to the increase in woodpecker use of these burns and of the increase in use by Northern Goshawks. At least one adult and two juvenile Northern Goshawks used the largest fall burn in 1999. The juveniles stayed in the border of unburned trees while the adult hunted through the burn. Three of the species detected before the burns are rare in the research area, Western Bluebird and Orange-crowned Warbler, and Grasshopper Sparrow. Grasshopper Sparrow may return as sagebrush canopy coverage increases.

Conclusions

Small fall burns (0.5 to 5 ha) removed enough tree overstory to make replanting of bitterbrush success at least feasible. The lack of change in estimates for bare ground may be partially due to needle fall from dying trees. The affects of this litter on planting success may be detrimental if broadcast seeding is chosen over nursery stock.

Although fall burns significantly decreased tree over story and shrub density, and deer mice density, they had surprisingly little effect upon bird distribution or canopy cover of understory species. In fact, ten understory species increased slightly on the fall burn plots. Spring burns did have significant changes in understory vegetation. Thirteen species decreased on the spring burn plots. This may be an important issue when determining the best season to burn.

Although the effect of fall burns resulted in a significant decline in deer mice use, this may be a result of the low overall populations allowing the remaining population to be more selective than they might be able to be when populations are high. In years with high numbers of mice, there may be very little distinction between burn treatments, particularly since the size of burns were relatively small.

Fall burns showed an increase in mountain big sagebrush density within two years after the fires. The reduction in tree over story with fall burns, coupled with the decrease in deer mice abundance on fall burns, makes fall a good choice for natural bitterbrush regeneration because rodent-cached bitterbrush seeds may survive to resprout. Pre-burn inventory needs to make sure there are seed-producing bitterbrush on site before burning if bitterbrush recruitment from rodent caching is to be potentially successful.

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APPENDIX I: BIRD COUNTS BY HABITAT STAGE

A total of twenty-eight species of birds were detected within 50-meters of each plot center. This is a listing of the number of each species counted in each habitat or stage of tree encroachment for each species.

SPECIES	GRASSLAND OPENING	SHRUB STEPPE OPENING	YOUNG FOREST	MIXED-AGE FOREST	MATURE FOREST
American Goldfinch	1	0	0	0	3
American Robin	1	1	0	2	0
Black-capped Chickadee	1	0	1	1	1
Brewer's Sparrow	1	2	0	0	0
Brown Creeper	0	0	0	1	0
Brown-headed Cowbird	0	0	1	7	0
Chipping Sparrow	3	3	15	12	0
Clark's Nutcracker	0	0	0	9	0
Common Nighthawk	0	0	2	0	0
Dark-eyed Junco	2	1	5	11	1
Dusky Flycatcher	1	0	3	9	13
Evening Grosbeak	0	0	0	5	3
Hairy Woodpecker	0	0	0	3	3
Hammond's Flycatcher	0	0	0	1	1
Hermit Thrush	0	0	0	0	3
Mountain Bluebird	9	4	4	2	0
Mountain Chickadee	3	0	0	7	9
Northern Flicker	0	0	2	8	0
Northern Goshawk	0	0	0	0	1
Pine Siskin	6	1	5	0	2
Red-breasted Nuthatch	0	0	0	1	2
Ruby-crowned Kinglet	1	2	1	3	2
Savannah Sparrow	1	0	0	0	0
Townsend's Solitaire	1	2	0	0	3
Vesper Sparrow	5	11	0	0	0
Warbling Vireo	0	0	0	0	3
Western Meadowlark	1	0	0	0	0
Yellow-rumped Warbler	1	1	3	3	7
Total Number	38	28	42	85	57
Species Richness	16	10	11	17	16