CANYON GRASSLAND VEGETATION CHANGES FOLLOWING THE MALONEY CREEK WILDFIRE

A Thesis Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Range Resources

in the

College of Graduate Studies

by

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May 2004

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ABSTRACT

Mosaic patchworks of native and non-native vegetation are a regular occurrence in western rangelands. Efforts to protect native grasslands involve combating exotic species invasions. Disturbances that may aid or setback an intact native community are often time, site, and type specific. In August 2000, a wildfire burned 11 of 25 previously sampled plots within *Pseudoroegneria spicata-Poa secunda* (bluebunch wheatgrass-Sandberg bluegrass) habitat type. Sampling species coverages using the modified Daubenmire cover class method within the 25 plots of mixed seral stages resumed in 2002. *Pseudoroegneria spicata* had recovered to pre-burn levels by 2003. *Poa secunda*, *Bromus tectorum*, *Centaurea solstitialis*, and *Lupinus sericeus* canopy coverages were significantly greater on burned plots. *Balsamorhiza sagittata* decreased slightly while *Achillea millefolium* and *Phlox colubrina* decreased substantially following the fire. Leguminous forbs, perennial forbs, introduced annual forbs, and native forbs increased on both burned and unburned sites suggesting favorable weather conditions aided in post-fire recovery. Significantly greater Pielou’s evenness, Shannon-Weiner’s, Simpson’s diversity and relatively unchanged richness on burned plots suggested a homogenizing fire effect on species coverages within burned plots. Some coverage change differences between burned and unburned sites when *Bromus tectorum*, *Centaurea solstitialis*, *Balsamorhiza sagittata*, and *Pseudoroegneria spicata* were investigated could be related to changes specific to particular seral communities.

ACKNOWLEDGMENTS

This research involved the cooperation and assistance of many people. I am thoroughly appreciative to all members of my committee, Dr. Stephen Bunting, Dr. Jim Kingery, and Dr. Timothy Prather, whose combined expertise and knowledge was invaluable to the direction, conduction, and completion of this study. I thank Dr. Bunting for his insight and thoroughness but most of all for his good-natured attitude throughout this project. My extreme gratitude also goes to Sandra Robins whose knowledge of canyon grassland species is unsurpassed and to whom I am indebted to for all pre-burn data. Her support, kindness, and friendship were much appreciated throughout this endeavor. I extend special thanks to Lynn Danly of the BLM for helping access the sites and for answering many management questions regarding the study area. I am also grateful to Art Talsma of the Nature Conservancy who provided much appreciated logistical information and plush accommodations during data collection. To Chris Williams, statistical consultant, I thank you for phenomenal patience and expertise.

Many others aided in less technical aspects of this project. To my terrific friend, Cathy Sampselle, whose support, ideas, and general greatness helped me in ways that I only hope she knows – I am truly grateful. Also to all those others: Karen Launchbaugh, Kathy Mallory, Eva Strand, fellow graduate and undergraduate range students I extend many thanks. Lastly, I thank my family, Scott Hauser, and Magill for their unconditional support and consistent faith in my abilities.

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INTRODUCTION

The canyon grassland communities that occupy the steep slopes above the Snake and Salmon Rivers of Idaho now include several aggressive, non-native plant species. The invasion by the exotic plant species is a result of anthropogenic disturbances that include the abandonment of agriculture and over-grazing in the area. While intact patches of mostly native *Pseudoroegneria spicata-Poa secunda* (bluebunch wheatgrass-Sandberg bluegrass) habitat (Tisdale 1979) persist, the encroachment of exotic weeds into these communities reduces their forage value and ecological function (Sheley and Larson 1994). The most abundant and problematic weeds within these grassland communities include *Centaurea solstitialis* (yellow starthistle) and a complex of annual *Bromus* species, the most dominant being *Bromus tectorum* (cheatgrass).

Fires occurring when the vegetation is essentially dormant within this habitat type are not considered detrimental and can stimulate the growth of certain species (Vogl 1974, Wright et al. 1979, Agee and Marnoka 1994, Erichsen-Arychuk et al. 2002). However, the removal of canopy cover and release of nutrients creates opportunities for ruderal and weed species (Vogl 1974, Hobbs and Huenneke 1992, With 2002). Studying the species specific fire effects should provide insight enabling improved post-fire management of canyon grasslands.

Understanding vegetation changes following fire is important to the future management of canyon grassland communities. Therefore objectives of this study were 1) To assess the fire effects on native and non-native plant species within the *Pseudoroegneria spicata-Poa secunda* habitat type; 2) To determine whether or not fire increases invasion success of yellow starthistle and other weed species; 3) To determine if plant community composition changes after burning; 4) To investigate community changes within the early-, mid-, or late-seral staged communities with respect to the dominant exotic and native vegetation.

LITERATURE REVIEW

Disturbance ecology

Studying the changes in plant communities that result from fire and human impacts, including fire management, road building, agriculture, and domestic grazing has increased our knowledge and appreciation for ecological processes and interactions. A disturbance is “any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment” (Pickett and White 1985). The type, duration, and intensity of a disturbance varies over time and space. Understanding the vegetative response to natural and anthropogenic disturbances that interact temporally and spatially is crucial to the successful management of an area.

There is increasing agreement among conservationists and ecologists that the restoration of historic disturbance patterns typical for an area is important to the restoration of the landscape as a whole (Hobbs and Huenneke 1992). This idea stems from the belief that ecosystems are adapted to and require particular disturbance regimes to function. Yet, ecosystems that differ compositionally and/or functionally from their historic state are challenging to manage. Understanding changes that occur when historically typical disturbances are reintroduced into these compositionally and/or functionally altered systems is important to our understanding and proper management of these systems.

Northern Idaho canyon grasslands

The canyon grasslands that occupy the often steep slopes of the Snake, Salmon, and Clearwater Rivers constitute approximately 486,000 hectares in Idaho (Tisdale 1979).

Grassland communities are the dominant vegetation in the low and mid elevations while coniferous communities appear above these. Within these arid low elevation habitats, woody shrub vegetation is restricted to drainage areas. The dominant canyon grassland habitat types are *Festuca idahoensis-Koeleria cristata* (Idaho fescue-Junegrass) on the moister northern slopes and *Pseudoroegneria spicata-Poa secunda* on the drier southern slopes (Tisdale 1979, earlier described in Washington by Daubenmire 1970). The *Pseudoroegneria spicata-Poa secunda* grassland communities within the Garden Creek Preserve of the Craig Mountain Wildlife Management Area were studied following a recent late summer wildfire (Figure 1).

While it is believed that these canyon grasslands burned seasonally, the lack of woody vegetation to provide fire scar data and lack of records make it difficult to determine the historical fire frequencies for the area (Weddell 2001). In addition to lightning ignition, fires were historically started in grasslands by Native Americans to increase forage available to themselves and their hunted game the following growing season (Gruell 1985). Whether burning was purposeful and regular in the Craig Mountain grasslands is unknown, but the arid climate and vegetation suggest fires would have been favored in this region (Shinn 1980).

Debate about the historical frequency of fires in canyon grasslands continues, but the more recent exclusion of fire in grassland areas is well-known (Shinn 1980, Johnson 1994). Since 1900, the prevention and suppression of fire has been the objective of federal land management plans. These fire suppression efforts coincided with the introduction of cattle and sheep to the region. There was widespread belief that fires in grasslands and forests would decrease the amount of forage available to domestic grazing animals.

Following 1870, European settlement of the canyon grasslands was extensive and domestic grazing became the most common disturbance on the landscape (Evans 1967). *Pseudoroegneria* dominated grassland habitat may not have evolved with intensive large herbivore grazing, an activity that left perennial bunchgrasses susceptible to damage (Mack and Thompson 1982). In simulated grazing studies in the bunchgrass prairies of southeastern Washington, native and non-native annual species increased in size and number, when *Pseudoroegneria spicata* was clipped severely during the active growing season (Daubenmire 1940). Overgrazing facilitated invasion by the pioneering winter annual, *Centaurea solstitialis* (Maddox and Mayfield 1985, DiTomaso 2000).

Weeds such as *Crupina vulgaris* (common crupina), *Bromus tectorum*, *Centaurea solstitialis*, and many others continue to thrive in the area. Following the removal of perennial rangeland grasses by grazing herbivores, invasion by yellow starthistle is possible. *Centaurea solstitialis* and *Bromus tectorum* together occupy over 250,000 hectares of Pacific Northwest rangelands (Sheley and Larson 1997) and *Centaurea* species spread between an estimated 8-14% per year (Whitson 1999, cited in DiTomaso 2000). Grasslands converted to *Bromus tectorum* and *Centaurea solstitialis* have decreased forage value and ecological function (Sheley and Larson 1994).

Potential fire effects

Combinations of natural and human-induced disturbances on the landscape are becoming increasingly common in rangeland systems. Understanding the effect of combined historical and recent disturbances is essential to making sound resource management decisions.

Wildfires, which characteristically create early-seral conditions by removing vegetation canopy and releasing a pulse of nutrients, increase the susceptibility of a native plant community to invasion by ruderal species (Vogl 1974, Hobbs and Huenneke 1992, With 2002). Yet some suggest that fire suppression has contributed to overall increases in available nitrogen and has facilitated the invasion of *Centaurea* species into nitrogen limited rangeland systems (LeJune and Seastedt 2001). The burning effect on *Bromus tectorum* is also important, as it is well known to increase following fire, increasing fine fuel availability that can increase fire frequencies for the area (Knapp 1996, Vitousek et al. 1996).

Research suggests that communities with *Pseudoroegneria spicata* and *Poa secunda* cover greater than 10-15% may be more resistant to invasion by *Centaurea solstitialis* (Robins 1999). Others suggest that when the functional diversity (combinations of annual and perennial grasses with early and late season annual and perennial forbs) of a plant community increases, its susceptibility to *Centaurea solstitialis* invasion decreases (Dukes 2001). For this reason, studying the post-fire response of the native vegetation is valuable in predicting increases or decreases in weed coverage and community invasibility.

Numerous studies have investigated the response of bunchgrasses to fire. *Pseudoroegneria spicata* showed increased growth and reproductive productivity following an August wildfire in *Artemesia* (sagebrush)-*Pseudoroegneria spicata* habitat of southeastern Washington (Uresk et al. 1976, 1980). Similarly, both *Pseudoroegneria spicata* and *Poa secunda* increased the year following a late June wildfire in ungrazed grasslands of western Montana (Antos et al. 1983). Nevertheless, native plants do not always increase following fire, particularly when subjected to stressful post-fire conditions. Six years following a fire in California, burned bunchgrasses had just 30% the basal area of unburned counterparts, a condition partially attributable to the continued domestic grazing in the area (Countryman and Cornelius 1957). *Pseudoroegneria spicata* decreased in *Stipa-Bouteloua-Pseudoroegneria* habitats for three growing seasons following a wildfire in dry mixed prairies of Alberta, Canada suffering post burn drought conditions (precipitation up to 31% below normal) (Erichsen-Arychuk et al. 2002).

Late summer wildfires still may not result in increased coverage of *Pseudoroegneria spicata* even in areas free of domestic livestock grazing. Daubenmire (1975) found that canopy cover and frequency of *Pseudoroegneria spicata* declined, while canopy cover of *Poa secunda* doubled after steppe vegetation of southwestern Washington burned in July. Community composition following this burn recovered 12 years after the fire. Following a July fire in the *Artemesia tridentata*/*Pseudoroegneria spicata* rangelands of northeastern Oregon, basal diameter measurements of *Pseudoroegneria spicata* were 52% of those measured in the un-burned areas (Conrad and Poulton 1966). Following an August burn in montane grasslands of the Selway-Bitterrot Wilderness of Idaho, *Pseudoroegneria spicata* production did not increase even four years following the fire (Merrill et al. 1980).

To understand the plant community's response to fire, attention must also focus on the response of annual grasses, and annual and perennial forbs whose presence increases the functional diversity of an area. Productivity of annual grasses and forbs increased for four years following the August wildfire in xeric *Pinus ponderosa* sites of Idaho (Merrill et al. 1980). Similarly, after a July burn in the grasslands of southeastern Washington, the cover of perennial forbs was greater on burned areas when compared to unburned areas (Daubenmire 1975). Also, forb production did not significantly decrease following fire, even under post-burn drought conditions (Erichsen-Arychuk et al. 2002).

Site characteristics, climate regimes, and post-burn conditions vary and influence the results of fire effects studies. Most studies on wildfire effects do not have pre-fire composition data. Burned and unburned reference sites are established after the fact in nearby sites deemed the same. Two years of pre-burn vegetation composition data exists for those plots that burned in 2000 and domestic grazing has not occurred in the area for more than 10 years. The pre-burn sampling and removal of confounding domestic grazing effects allows for pre- and post-burn differences to be more directly linked to the effects of the fire.

Investigating the response of native vegetation and exotic annual weeds to wildfire in the canyon grasslands of Garden Creek will add to the current understanding of weed, fire, and vegetation ecology. Results from this study should allow managers to make more informed decisions about whether fire suppression in this area is warranted. It will also allow managers to anticipate changes occurring post-burn, and help them to determine weed control measures and/or restoration efforts that may be useful in complementing or combating post-fire vegetation changes.

METHODS

Study area

The Craig Mountain Wildlife Area (CMWA) is roughly 40 km south of Lewiston in both Nez Perce and Lewis counties. Within the CMWA, is the Garden Creek Nature Preserve that occupies approximately 4761 ha of Craig Mountain. The Nature Conservancy and the Bureau of Land Management jointly manage the preserve where the study sites were located (Figure 1). This area receives an average of 30 cm of annual precipitation, primarily in the winter and spring months as rain and snow (WRCC 2004). Maximum summer temperatures average 31.7 ^{o} C and winter lows average 3.9 ^{o} C in this region. Studies were conducted in the *Pseudoroegneria spicata/Poa secunda* habitat type (Tisdale 1986) occurring on the southern aspects of two watersheds, China Creek and Corral Creek.

The study plots were previously sampled (Robins 2001) and were reinvestigated following the Maloney Creek fire that burned approximately 30,155 ha from August 15 to 24, 2000, including a large portion of the Garden Creek Nature Preserve (USDI 2000, Inside Idaho 2000). There were 25 study plots of early-, mid-, and late-seral stages. All plots are located on southeast to southwest aspects with slopes of 16-52% and at elevations of 796-1046 m. Eleven of these plots burned in the Maloney Creek fire. Two of the six early-seral stage, six of the eleven mid-seral stage, and three of the eight late-seral stage plots burned in 2000. The portion of the Maloney Creek fire that burned these plots within the China Creek watershed was a backing fire that burned August 20, 2000. This fire began at the southern most point of the fire line along the ridge north of China Garden Creek and was described as burning with low severity (Hawkins, pers. com.). Humidity levels varied from 27-33%, the average temperature was 23.6 ^{o} C, and wind speeds ranged from 0-16.7 km/h with gusts up to 35.2 km/h during the day this portion of the Maloney Creek fire burned (Weather Underground 2004).

Sampling methods

Vegetation was sampled throughout late May and early June to parallel the earlier study of these plots and to ensure that comparisons could be made between the four years of data collection. UTM coordinates, percent slope, aspect, and elevation were recorded for all 25 plots (Appendix III). The 25x25-m study plots were sampled using a cover class method and a 20x50-cm quadrat. Canopy cover for the native and exotic vegetation within the plots was estimated visually (Appendix I). Using cover class estimates established by Daubenmire (1959) and revised by Bailey and Poulton (1968), vegetation canopy estimates of 0-1% were assigned to and recorded as cover class 1, canopy estimates of 1-5% = 2, 5-25% = 3, 25-50% = 4, 50-75% = 5, 75-95% = 6, and 95-100% = 7. In addition to vegetation canopy cover, estimates of the percentage of bare soil, rock (>2cm), litter, and cryptogams, were obtained for each quadrat (Appendix IV). To determine quadrat placement within the study plot, a 25-m transect was extended from the middle of the bottom edge of the study plot and to the upper edge above. Random numbers between 1 and 25 were generated to determine placement of five 10-m transects within the plot. Even numbers established 10-m transects to the right of center as one looks at the plot from below, while odd numbers dictated 10-m transects to the left of the plot center. Quadrat cover estimates of the contained vegetation were made at each 1-m increment. A total of 1250 quadrats were analyzed.

Analysis methods

There were nine dominant forb and grass species and four groups (perennial forbs, leguminous forbs, introduced annual forbs, and native annual forbs) analyzed (Appendix II). Infrequently occurring grasses were left out of the analysis. To determine the significant differences between burned and unburned plots by species and groups over time, the mixed model analysis (PROC MIXED) in SAS was employed (Littell et al.1996). The compound symmetric, autoregressive order one, and unstructured covariance patterns were evaluated and chosen based on Akaike's Information Criterion (AIC). All individual species and species group coverages were log transformed and covariance structure was assigned on a species by species basis according to the lowest AIC obtained. Significance was evaluated for the given year using the slice command in SAS. The overall p-value reported represents the change in treatment differences over time (time*treatment interaction).

Using PC-ORD software, the following diversity indices: richness, Pielou’s evenness, Simpson’s, and Shannon-Weiner’s were obtained for all four years of data collection (McCune and Grace 2002). Diversity calculations included all plots and all species coverages without relativizations or downweighting transformations. Richness values reflect the number of species per study plot while Pielou’s evenness measured the equitability of coverage for each species within a study plot. The same log transformations, AIC criteria, and slice command were used in the PROC MIXED SAS procedure described above to obtain overall time*treatment p-values and treatment p-values for all sampling years.

Community changes within the early-, mid-, and late-seral plots were also analyzed using the SAS PROC MIXED model. This analysis was done for those species that are primarily used to classify the seral stage within the *Pseudoroegneria spicata/Poa secunda* habitat type (the area's most abundant annual and perennial species, *Bromus tectorum, Centaurea solstitialis, Pseudoroegneria spicata, Poa secunda,* and *Balsamorhiza sagittata*). Overall p-values reflect the time, treatment, and block (seral stage) interaction (time*treatment*block). P-values relating to treatment differences were also calculated for each given year within each seral stage. The three graphs representing the early-, mid-, and late-seral communities were then compared to evaluate how they related to those changes in species coverages that were apparent when all seral stages were incorporated.

RESULTS AND DISCUSSION

Background

The Maloney Creek fire followed 45 days of below average precipitation, but immediately following the fire in September of 2000, precipitation was above average (Figure 2). For the first post-burn sampling year (2002), April and May precipitation was below normal and the second sampling season (2003) followed months when precipitation amounts were considerably greater than the long-term average. This precipitation pattern undoubtedly influenced the results reported for this study.

Results regarding species coverage changes with respect to time and burning treatment are presented in graphical format. P-values at each year represent differences in plot locations prior to 2002, but represent differences attributable to burning for 2002 and 2003. For many of the species investigated, there are differences evident prior to burning. These differences may reflect the greater distance between plots, the higher degree of agricultural abandonment, and/or the greater proportion of early seral staged communities for those plots within the unburned area.

Native and introduced grass species

The dominant perennial grass within the study area, *Pseudoroegneria spicata*, showed significantly less coverage within the burned plots for 2002, the first sampling season following the fire ($p = 0.035$) (Figure 3). Burned and unburned plots were not significantly different by year 2003, and burned plot coverage by 2003 was greater than either pre-burn sampling. Very similar short-lived declines in *Pseudoroegneria spicata* following burning have been reported (West and Hassan 1985). Daubenmire (1975) suggests *Pseudoroegneria* *spicata* recovery to pre-burn canopy coverage may take as long as 12 years, yet he notes that reduced canopy cover is the result of more erect growth, as basal coverage had not changed. Others have found *Pseudoroegneria spicata* to increase slightly the first few years following fire (Harniss and Murray 1973, Uresk et al. 1980, Antos et al. 1983). Still other research suggests that severity of the burn (Johnson 1998) or topographic position of the vegetation (Erichsen-Arychuk 2002) may dictate conflicting post fire responses in *Pseudoroegneria spicata*.

Poa secunda cover within the burned plots was significantly higher for both years following the fire (2002 $p = 0.045$, 2003 $p = <0.0001$) (Figure 4). Daubenmire (1975) and Antos et al. (1983) found *Poa secunda* coverage doubled on burned sites. Daubenmire suggests the reduced light competition from *Pseudoroegneria spicata* facilitated this increase. Others have also noted no change or increases in *Poa secunda* following burning (Wright and Klemmedson 1965, Harniss and Murray 1973, and Agee and Maruoka 1994).

In the pre-burn sampling, *Bromus tectorum* coverage was found to be significantly lower on the burned sites when burned and unburned plots were compared (2000 $p = 0.002$) (Figure 5). However, by 2003 burned and unburned plots were not significantly different. *Bromus tectorum* increases following fire are well documented in the literature (Conrad and Poulton 1966, Merrill et al. 1980, West and Hassan 1985, Knapp 1996). However, these findings are contrary to those of Daubenmire (1975) where decreases in all *Bromus* species (including *tectorum*) within the *Pseudoroegneria spicata-Poa secunda* habitat were recorded for all post burn samplings that spanned 12 years.

Bromus japonicus (Japanese brome) another abundant exotic grass in the area showed less of an increase on burned plots as compared to unburned plots (Figure 6). Following the Maloney Creek fire, burned plots had significantly less *Bromus japonicus* cover than did unburned plots (2002 p = 0.017). Daubenmire (1975) reported decreases in *Bromus japonicus* for the second, forth, and twelfth post-burn sampling following a July wildfire in a *Pseudoroegneria spicata-Poa secunda* grassland of southwestern Washington (Daubenmire 1975). These species share many phenological and developmental characteristics (Hulbert 1955). Yet, *Bromus japonicus* matures later than *Bromus tectorum* (Hulbert 1955, Robocher 1973). The increased competition from *Bromus tectorum* coupled with slightly delayed *Bromus japonicus* development may partially explain our findings.

Native and introduced forb species

The dominant forb occurring in the study area is *Balsamorhiza sagittata* (arrowleaf balsamroot). Although this species is considered to be undamaged by fire (Wright et al. 1979), our results show decreasing *Balsamorhiza sagittata* coverage following the fire on burned plots (Figure 7). Although there were no significant differences between burned and unburned plots at any given sampling year, the opposing burned and unburned line trajectories for years 2002 and 2003 dictate significant differences for the time by treatment interaction (p = 0.008). *Balsamorhiza sagittata* increased following an August fire in a *Pinus ponderosa* understory community (Merrill et al. 1980) and increased following a severe burn in a *Pseudoroegneria spicata -Poa secunda* grassland (Johnson 1998). The observed decreasing trend in *Balsamorhiza sagittata* on burned sites is puzzling since winter precipitation post-burn was above average. Other perennial forbs increased on burned sites and competition with these forb species may have limited *Balsamorhiza sagittata*.

The only leguminous forb investigated independently, *Lupinus sericeus* (silky lupine), responded favorably to burning (Figure 8). The burned plots contained significantly more lupine than did the unburned plots by 2003 (p = <0.0001). This difference reflects not only an increase in lupine cover on the burned plots but also a decline in lupine on the unburned plots. While *Lupinus* species are considered fire adapted (Wright et al. 1979), the magnitude of the observed increases in *Lupinus sericeus* surpasses those reported in other studies (Antos et al. 1983, Johnson 1998).

In the third year following the fire, *Phlox colubrina* (Snake River phlox), declined in the burned plots (2003 p = 0.024) (Figure 9). For all sampling seasons prior to 2003, there were no statistically significant differences between the burned and unburned plots. While *Phlox colubrina* decreased in both the burned and unburned plots in 2003, the decline in the burned plots was pronounced. This *Phlox* species is not addressed in the literature. However, *Phlox longifolia* decreased in abundance (Wrobleski and Kauffman 2003) and cover (West and Hassan 1985) when sagebrush-grasslands were burned.

Centaurea solstitialis, an abundant exotic forb in this area, was of particular interest in this study as disturbances have been reported to facilitate the invasion process (D'Antonio 2000). The change over time for *Centaurea solstitialis* was significant (p = 0.038) (Figure 10). This is attributable to the change in slope of the line for unburned plots between 2002 and 2003. Another study involving fire and *Centaurea solstitialis* reports declines for this exotic the first year following three consecutive years of early July prescribed fire (Kyser and DiTomaso 2002). However, after discontinuing burning, *Centaurea solstitialis* cover increased from 2.4% to 38%. The timing of the fire studied by Kyser and DiTomaso (2002) came before seed production in *Centaurea solsitialis*, which was not the case for our study.

Achillea millefolium (western yarrow) on burned sites did not increase as reported by others (Merrill et al. 1980, Antos et al. 1983). The plots that burned in late summer 2000 had significantly higher yarrow cover than did the unburned plots (1999 p = 0.002, 2000 p = 0.014) (Figure 11). After the fire, however, the burned and unburned plots showed no significant differences. The decline in *Achillea millefolium* cover following fire is similar to the results of Daubenmire (1975).

There were no significant differences for the leguminous forb group between burned and unburned plots for any given year (Figure 12). Yet burned plots showed an increasing trend in cover following the fire. In a previous study, Wrobleski and Kauffman (2003), found *Lomatium* and *Astragalus* species abundance was unchanged or slightly increased following a fall prescribed burn. However, there were different species within these discussed genera comprising the leguminous forb group in our study (Appendix II).

In the pre-burn sampling, introduced annual forbs on the unburned plots showed significantly higher coverage than did the plots that were burned (2000 p = 0.018) (Figure 13). However by 2003, the burned plots showed significantly more introduced annual forb cover than in the unburned plots (2003 p = 0.003). Interestingly, both the burned and unburned plots showed an increase in the 2002 sampling season that may be a result of the above average precipitation for March and June of 2002 and March, April, and May of 2003 (Figure 2).

The amount of native annual forb coverage increased in the post-burn sampling seasons for both the burned and unburned sites (Figure 14). By the third post-burn sampling year, the native annual forb coverage was significantly higher on the burned plots (2003 p = 0.025). These findings suggest a longer-lived increase than that reported by Merrill et al. (1980), who found annual forbs to be significantly more productive on burned sites for just the first post-burn season.

The less dominant perennial forb group showed no significant differences between burned and unburned plots yet both burned and unburned plots increased over time (Figure 15). These findings are not surprising as most forbs are described as unharmed by fall burning (Wright et al. 1979). Other studies involving August fires have reported increases in perennial forb production (Harniss and Murray 1973, Merrill et al. 1980).

Diversity indices

Disturbances and their relation to the diversity of communities have been discussed at length in the literature (Huston 1979, Pickett and White 1985, Hobbs and Huenneke 1992, Prieur-Richard and Lavorel 2000). Much of this information focuses on multiple disturbances occurring over a temporal scale much greater than this study. Diversity as it potentially relates to the invasibility of communities is also interesting with regard to this study. There are inconsistencies in the literature evaluating invisibility and diversity relationships. Some experimental research suggests that increased species richness and/or diversity is coupled with decreased invasibility (Knops et al. 1999, van Ruijven et al. 2003). Other research suggests that an increase in species richness or diversity does not necessarily dictate decreased community invasibility (Symstad 2000, Foster et al. 2002).

Species richness values were not significantly different for any given year between the burned and unburned plots (Figure 16). Kyser and DiTomaso (2002) found a slight, short-lived increase in species richness in a *Centaurea solstitialis* invaded California grassland burned for three consecutive years. Harrison et al. (2003) reported that the number of native and exotic species increased on all sites within chapparal grasslands of California following an October wildfire, but that the increase of exotic species was greater on sites dominated by exotics and the increase of native species was greater on sites dominated by native species. However, these results conflict with a study by Biondini et al. (1989) who reported decreased species richness following a fall fire in a South Dakota mixed prairie.

Pielou's evenness of the species was not significantly different prior to the fire, but the burned plots showed significantly greater evenness after the burn (nearly significant 2002 $p = 0.053$, 2003 $p = 0.001$). Burned plots showed greater homogeneity for species coverage than unburned plots did.

Simpson's index of species dominance was similar between plots before the fire. After burning, however Simpson's (D) diversity was significantly different between burned and unburned plots for both post-burn sampling years (2002 $p = 0.023$, 2003 $p = 0.002$). Similar results were found when Shannon-Weiner's (H') diversity was investigated. Prior to the fire burned plots had lower Shannon-Weiner index values than the unburned plots (2000 $p = 0.014$) (Figure 18). Yet after the fire, burned plots had Shannon-Weiner values significantly greater than those of the unburned plots (2002 $p = 0.010$, 2003 $p = 0.001$). These results show longer-lived increases than those of Kyser and DiTomaso (2002) who found Shannon-Weiner values to be greater on burned sites the first and second years following burning but lower for burned sites by the third post-burn sampling.

The increased evenness coupled with relatively unchanged species richness suggests that the increased Simpson's and Shannon-Weiner's diversity reflect increasing homogeneity for species coverage, not the addition of species within the burned plots.

Management for diverse communities even if not creating less invasible communities is fundamental to the preservation of overall biodiversity of the area. While this fire did not substantially increase species richness, it was instrumental in decreasing species dominance while increasing the coverage of rare species.

Community changes with respect to seral stage

Centaurea solstitialis coverages within the early-seral plots show a decreasing trend for both the burned and unburned plots (Figure 21). However, the coverage of this species within the mid- and late-seral plots appears to be increasing on the burned plots, a trend that is not mimicked on the unburned plots. Within the late-seral stage the trends are different for the burned and unburned plots, an increasing trend within the burned plots from 1999-2003 while coverage decreases on unburned plots between 2002 and 2003. With respect to the changes seen for Figure 10, the increase on the burned plots from 2002 to 2003 is the result of increases in *Centaurea solstitialis* cover in the mid- and late-seral plots. The decrease on the unburned plots between 2002 and 2003 however is likely due to decreases in *Centaurea solstitialis* coverage within the late-seral plots (Figure 21, late). The increased precipitation for 2003 and increased competition from other annual and perennial forbs that increased between 2002 and 2003 may partially explain the decrease in *Centaurea solstitialis* cover.

The coverage increases for *Bromus tectorum* on the burned and unburned plots occur within all seral stages, yet these increases are greater within the burned plots (Figure 22). However, a decreasing trend is evident between 2002 and 2003 for the unburned plots in

Figure 5. This trend seems to be driven by the decrease in *Bromus tectorum* cover within the late-seral plots for this period (Figure 22, late).

Within the burned early-seral plots, the coverage of *Pseudoroegneria spicata* shows an increasing trend from 2002 to 2003 (Figure 20). The mid- and late-seral plots also show recovery from the post-burn coverage decreases by the 2003 sampling season. The overall recovery that is seen for Figure 3 can be attributed to the slight increases from 2002 to 2003 within all seral stages but the contribution to recovery seems to be greatest from the early-seral plots.

Balsamorhiza sagittata coverage within the early-seral plots decreases slightly on both the burned and unburned plots (Figure 23). Within the mid-seral plots, *Balsamorhiza sagittata* coverage is stable on the burned plots and appears to be increasing slightly over time for the unburned plots. For the late-seral plots, the coverage of *Balsamorhiza sagittata* shows no real change in either the burned or unburned areas. This suggests that the slight decreases following the fire seen in Figure 7 are a result on decreased *Balsamorhiza sagittata* coverage within the early seral plots.

The increased coverage of *Poa secunda* on the burned plots evident in Figure 4 was not restricted to any one seral stage. Increases in *Poa secunda* coverage for the burned plots are seen in the early-, mid-, and late-seral plots (Figure 24).

If the increased coverage of *Pseudoroegneria spicata* and *Poa secunda* were to continue within the early seral stage communities, this could result in increased future flower production by these species. This increased recruitment potential could possibly facilitate the advancement of succession with respect to *Pseudoroegneria spicata* and *Poa secunda*. While this type of community change is intriguing and has been suggested as possible by some restoration ecologists (Sheley and Krueger-Mangold 2003, Choi 2004). The data to support this idea of possibly advancing succession through designing particular disturbances does not currently exist.

Likewise, if the trends for increasing cover of *Centaurea solstitialis* and *Bromus tectorum* continue in the late seral plots, these communities may suffer a set back with respect to successional development as it relates to these species and their potential for increased recruitment. Others have documented this type of successional change. Lesica and Cooper (1999) found that burned late-seral communities more resembled unburned mid-seral communities in sandhill vegetation of southwestern Montana. Without knowing the future recruitment of these dominant annual and perennial grassland species, the successional change within these communities as it relates to fire is speculative. However, further study into this area is intriguing.

CONCLUSIONS

Understanding the effects of disturbances such as fire on community composition and dynamics provides managers insight into community dynamics that allows them to make more informed decisions. While many disturbances are uncontrollable, data as presented here is valuable in preparing management for potential desirable or undesirable outcomes within both the native and exotic communities. With an understanding about the vegetation composition changes that occur post-fire, managers can anticipate those areas where weed and native species coverage increases or decreases may occur. With fire data specific to the canyon grasslands of Garden Creek and its grassland species, the post-fire management and restoration efforts within this area are improved.

This research indicates that the Maloney Creek fire did not selectively enhance only the native plant species in the area. *Poa secunda*, *Lupinus sericeus*, *Bromus tectorum*, and *Centaurea solstitialis* all increased on burned plots. *Pseudoroegneria spicata* recovered within 3 years following the fire. Perennial forbs, annual native, and non-native forbs increased on both burned and unburned plots suggesting favorable post-burn weather conditions may have played a role in vegetation recovery. These findings suggest that the reintroduction of fire or discontinuing fire suppression efforts in this area may benefit native plants yet will not select against exotic species.

Post-burn community changes with respect to seral stage suggest that *Centaurea solstitialis* coverage increases were concentrated within the mid- and late-seral communities. However, *Bromus tectorum* coverage increases occurred throughout all seral stages. Interestingly, the coverage of these species decreased within the unburned late-seral plots. *Pseudoroegneria spicata* and *Poa secunda* coverage increased within the burned early-seral plots. These findings suggest that fire may eventually alter the successional pathway of communities. Some restoration ecologists believe that communities may be restored through designing disturbances and facilitating successional change (Sheley and Krueger-Mangold 2003). This type of restoration is innovative and intriguing, however the data to support this idea of altering succession through particular disturbances is lacking. Additional research in this area is valuable and appealing for future restoration and vegetation management.

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