

Short-term Influence of Wildfire on Canyon Grassland Plant Communities and Spalding's Catchfly, a Threatened Plant

Abstract

Fire is a significant component of the grassland ecosystem. It is often critical to the viability of native plant populations, and may promote invasive exotic species. We examined the influence of late season wildfire on Spalding's catchfly, a threatened plant species, and its associated bunchgrass plant community in the canyons adjacent to the Snake River. We used stratified random sampling to assess vigor of the catchfly and changes in plant communities found in burned and unburned areas during the 2 yr of the study. Fire decreased total vegetative cover and total grass cover in the first year after burning, yet total forb cover remained similar to unburned areas. Fire decreased cover of Idaho fescue and increased cover of silky lupine, while cover of bluebunch wheatgrass, exotic grasses, and most other forb species remained similar to unburned areas. Catchfly cover and abundance also remained similar before and after fire. Although burning tended to decrease plant size slightly and decrease the proportion of flowers that matured to seed-filled capsules, fire did not appear to influence levels of flowering, change the number of flowers or capsules produced per stem, or alter the number of seeds per capsule. We determined that catchfly and the plant communities that support it are well adapted to naturally occurring, late season, low to moderate intensity fire.

Introduction

Fire is a significant component of grassland ecosystems, including tallgrass prairies (Hartnett 1991, Towne and Knapp 1996), aspen parklands (Anderson and Bailey 1980), Canadian dry mixed prairie (Erichsen-Arychuk et al. 2002), and grasslands of the intermountain west (Antos et al. 1983, Johnson 1998). By removing aboveground plant litter and biomass, fire can temporarily increase the exposure of the soil surface to sunlight (DiTomaso et al. 1999), resulting in higher soil temperatures (Antos et al. 1983, Defosse and Robberecht 1996). Under some circumstances, burning can increase short-term nutrient availability by speeding decomposition of plant litter (Dudley and Lathja 1993). Grassland fire may volatilize some soil nitrogen and increase leaching or runoff, however these losses are variable, and are typically replaced through greater activity of nitrogen fixing plants and nitrification by soil bacteria (Vogl 1974, Wright and Bailey 1982). Depending on site characteristics, burning can also influence soil moisture availability (DiTomaso et al. 1999). In water limited ecosystems, reduction of plant litter accumulations after fire can result

in increased soil temperatures and evaporation, reduced snow retention, and reduced moisture availability in the upper soil depths (De Jong and MacDonald 1975, Redmann 1978). Competition for water after fire can have strong influences on plant regeneration and community productivity (Defosse and Robberecht 1996), and may affect community composition and species diversity (Wilson and Shay 1990).

Because many ecosystems and sensitive species are adapted to fire (Pickett and White 1985), prevalent policies of fire suppression are changing, and fire is now frequently considered as a management tool. Fire recurring at natural frequencies can be critical to maintain native plant communities, promoting the balance between dominant and subordinate species (McGinley and Tilman 1993). Fire can also be vital to endemic or threatened species that depend on disturbances from fire to persist in plant communities (Bowles et al. 1990). Prescribed fire has been utilized as a tool in restoration efforts to increase rare plant population size (Borchert 1989, Menges 1995), and can be used to increase rare plant densities (Lesica 1999). Under some circumstances, burning may also control certain exotic species in plant communities, for example yellow starthistle (*Centaurea solstitialis*) (DiTomaso et al. 1999, Kyser and DiTomaso 2002), and medusahead (*Taenatherum caput-medusae*) (Miller et al. 1999). Fire

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is often most effective in weed control if areas are burned in repeated years and if it is timed to occur after most vegetative growth but before seed set, or if used in combination with herbicides (DiTomaso et al. 1999, Sheley and Petroff 1999). A primary concern regarding the use of prescribed fire in plant communities is that, as a disturbance, fire also has the potential to increase the susceptibility of plant communities to invasion by species not normally occurring in the plant community (Burke and Grime 1996). For example, fire can promote dominance and seed production in the Eurasian annual cheatgrass (*Bromus tectorum*) (Melgoza et al. 1990, Pyke 1994). In turn, cheatgrass litter enhances fire frequencies, which have resulted in widespread conversion of western US steppe and shrub-steppe to annual grassland (Peters and Bunting 1994).

Historically, western grasslands are estimated to support fire return intervals of ~5-10 yr (Wright and Bailey 1982). Fires probably occurred late in summer, when fuels were driest and lightning storms were most frequent. Many native species in these communities are adapted such that they can tolerate or even respond favorably to this regular disturbance (Wright and Bailey 1982, Tracy and McNaughton 1997). Some sensitive native or rare plants adapted to this disturbance regime may not tolerate prescribed burning in other seasons or at greater frequencies. Alternatively, if fire is completely excluded, populations of various rare species may perform poorly and experience reduced viability (Menges and Dolan 1998, Kaye et al. 2001). Fire can be beneficial to rare plant populations by promoting germination, seedling survival, and recruitment through reduced plant litter (Lesica 1999) and decreased competition from established plants (Maret and Wilson 2000). In some species, fire can also stimulate adult plant growth, flowering, and seed set (Hartnett and Richardson 1989, Pendergrass et al. 1999). Greater seed production can, however, be balanced by mortality of unprotected seeds on the soil surface (Hartnett and Richardson 1989, Hartnett 1991), which may help maintain the genetic diversity of the soil seed bank. These and other factors suggest that a variety of fire frequencies, seasons, and conditions are likely necessary to ensure preservation of biodiversity in grassland systems (Howe 1994).

The northerly slopes of Garden Creek Ranch Preserve's canyon grasslands in north-central Idaho support one of the largest known occurrences of

Spalding's catchfly (*Silene spaldingii*), a perennial forb that was listed as threatened by the US Fish and Wildlife Service (USFWS) in 2001. In northwest Montana, burning enhanced recruitment and density of catchfly, probably because fire removed persistent plant litter derived from rough fescue (*Festuca scabrella*) (Lesica 1999). However, fire may have different effects at the preserve, where plant communities tend to have more cover of finer-leaved Idaho fescue (*F. idahoensis*) than Montana sites. In this study, our objectives are to (1) examine the short-term influence of naturally occurring late season fire on the species composition of the canyon grassland communities supporting catchfly, (2) compare these results with those from previous studies of grassland communities in different locations, and (3) investigate the influence of late season fires on catchfly population vigor, as measured by plant size, levels of flowering and reproductive output within populations.

Methods

Study Area

Garden Creek Ranch Preserve (GCRP), a 5,668 ha natural area jointly owned and managed by the USDI Bureau of Land Management and The Nature Conservancy, is located approximately 15 km north of the Hells Canyon National Recreation Area. The preserve consists of several steep, dissected canyons draining into the Snake River, and contains canyon grassland communities, dominated by Idaho fescue and bluebunch wheatgrass (*Pseudoroegneria spicata*) (nomenclature follows the PLANTS database; USDA National Resource Conservation Service 2002). In addition to the catchfly, the preserve contains numerous other rare plant species (Hill and Gray 1998). These canyon grasslands are similar to the historical plant communities found in the rolling hills of the Palouse prairie. They have avoided conversion to dry-land wheat cultivation because of their steep topography, which often rises over 1000 m within ~3.25 km. Soils on the preserve are of the Lickskillet-Rock outcrop association, with a basalt parent material containing deposits of loess and ash (University of Idaho and USDA Soil Conservation Service 1976). Soils are typically shallow and rocky along ridges, with deeper soils on canyon slopes. The preserve experienced livestock grazing of various intensities until the early 1990s.

We conducted the research reported here during summer 2001 and 2002 in Lower Corral Creek and China Garden Creek, two of the three areas that support catchfly at the preserve. Most of China Garden Creek (CGC) burned in lightning-ignited fires and backfires in September 2000, and a portion of Lower Corral Creek (LCC) burned in similar wildfires in October 2001. These were the only large fires recorded since 1976 at the preserve.

In Lewiston, Idaho, at an elevation similar to the lowest portions of the preserve and about 67 river km north, mean January and July temperatures range between -3 -4°C and 14-32°C, with a mean annual precipitation of 320 mm (Western Regional Climate Center 2003). In Cottonwood, Idaho, 40 km east and ~110 m higher in elevation than catchfly-supporting areas of the preserve, average annual precipitation is 570 mm and average annual snow pack is 146 cm (in years with complete records; Western Regional Climate Center 2003). In both areas, precipitation is distributed relatively evenly throughout the year, peaking slightly in late spring. The 2 yr of our study (2001 and 2002) differed climatically. The snow pack in the winter of 2000-2001 was ~70% of normal, while snow pack in the winter of 2001-2002 was ~120% of normal.

Species Description

Spalding's catchfly occurs in British Columbia, northwest Montana, north-central Idaho, and eastern Washington and Oregon. Cultivation and livestock grazing have altered the majority of its grassland habitat. It occurs currently at one site in Canada and 51 sites in the United States (US Fish and Wildlife Service 2001). GCRP is one of only six catchfly occurrences that contain more than 500 plants (US Fish and Wildlife Service 2001). We will refer to each occurrence on a separate site within the preserve as a population or site. Some catchfly-supporting sites also support yellow starthistle or cheatgrass. The relationship of these

species with catchfly was examined in a related study (Menke and Muir 2004).

Each catchfly genet (plant) consists of one to several ramets (stems) that branch from the caudex just below the soil surface. Plants can have ~85 cm long taproots, which may allow access to deep water unavailable to many other plants in the community during summer drought (Kephart and Paladino 1997). Plant tissues die back completely each winter. First year seedlings are small rosettes that occur briefly early in the season and senesce before producing reproductive tissues. Plants of two or more years produce stems 8-25 cm tall, with extremely sticky, glandular-pubescent cauline leaves, and may produce cymes of 1-15 flowers during July and August. Flowers have inconspicuous, pale green to light pink petals, and mature into a single capsule, each containing many 1-2 mm diameter seeds. Catchfly plants depend on outcrossing to avoid substantial reductions in fitness from inbreeding depression (Lesica 1993). Plants appear to be long lived, but may frequently seem to be completely absent because of summer dormancy (Lesica 1997). Germination occurs in the spring or fall, since seeds appear to require approximately 4 wk of cold stratification (Lesica 1993).

Field Methods

During summer 2001 and 2002, we sampled plant communities at all sites within LCC and CGC that supported catchfly (Table 1). At each site, we positioned a 0.5 x 4 m plot in each patch of catchfly plants. Because the distribution of catchfly plants within individual sites was patchy in some cases and continuous in others, the number of plots per site varied. Plots were placed using a random azimuth from patch center rotated clockwise (if necessary) until it intercepted a catchfly plant; plots were centered on that plant and oriented downslope.

TABLE 1. Distribution and fire history of catchfly sites within two study areas at Garden Creek Ranch Preserve. Lower Corral Creek (LCC) burned in fall of 2001, after our sampling was completed for the year. In LCC, half of one site burned while the other half did not, and we treat each half as a separate site in 2002. China Garden Creek burned in fall of 2000, prior to our sampling.

	Lower Corral Creek (LCC)	China Garden Creek (CGC)
2001	20 sites (unburned)	5 sites (1 yr postfire)
2002	8.5 sites (unburned) 11.5 sites (1 yr post-fire)	5 sites (2 yr postfire)

In each plot, we recorded density of catchfly stems and measured percent cover of bare ground, plant litter, and all plant species using cover classes approximating an arcsine square root transformation (1 = <1%, 2 = 1-5%, 3 = 6-25%, 4 = 26-50%, 5 = 51-75%, 6 = 76-95%, 7 = 96-99%, 8 = 100%). In 2001, since data were taken over several weeks (June 15-July 22), we re-read plant cover on a subsample of plots every 7-10 days to determine whether cover estimates should be adjusted for seasonal drying. No correction was required. In 2002, cover data were taken over one week (15 June-22 June). Because of the timing of our sampling, estimates of species richness may not reflect the diversity of earlier-flowering forbs, thus we do not report differences in species richness between burned and unburned sites. In each plot we also took data on slope, aspect, elevation, and topographic position (ridge, upper slope, mid slope, bench, and lower slope), so that we could compare the abiotic environment between sites.

We counted catchfly stems and plants at most sites in 2001. At a randomly selected subset of four sites in CGC and nine sites in LCC in 2002, we recorded abundance of catchfly plants and stems so that we could detect changes in abundance after fire. At all sites, in June-early July 2001, we marked 30 stems (when not looking for changes in abundance, we frequently measured stem rather than plant characteristics since it could be done more quickly) selected at random distances along a restricted-random origin and azimuth transect placed before catchfly was flowering. In populations of < 30 stems, we sampled all stems. We then revisited these stems after flowering began in August, and evaluated the percent of flowering (reproductive) stems per population. Stems that had been grazed were tabulated separately and excluded from this analysis.

In 2002, we re-evaluated the percentage of flowering in the same populations using similar transects. We also measured additional plant vigor parameters in 2002, including stems/plant, stem height, flower number/stem, and mature capsule number/stem. We also counted the number of seeds/capsule in a subset of five plants per site (as per US Fish and Wildlife Service permit TE056322-0). Because half of one LCC population burned in 2001 and half remained unburned, we included each respective part in the appropriate group, treating them as separate populations in 2002.

Analysis

For all analyses of the plant community, we converted species cover classes to their midpoints. We used data from LCC only, since we had pre and postfire data for this study area. Although there were multiple plots within large sites, we used each plot as an individual sample. This method was appropriate since the plant community varied within sites. Since sites varied in size and plot number, aggregating plots to the site level would have tended to enhance species richness in larger sites. Because plots within sites were not independent sample units, we interpret the statistical results descriptively rather than as formal hypothesis tests.

Because our sampling spanned 2 yr, we had to examine variation between years in unburned areas before we could assess the fire-induced response in plant communities. Using data from 16 plots in unburned sites (which remained unburned both years), we compared the overall (multivariate) community composition between 2001 and 2002 with Multi-Response Permutation Procedures (MRPP) (Mielke 1984) in PC-ORD (McCune and Mefford 1999) using a Sørensen distance and data relativized by species totals. MRPP is a non-parametric multivariate procedure to test for a null hypothesis of no difference between groups, and uses Monte-Carlo (randomization-based) tests to evaluate statistical significance. We also compared cover of each individual species and basic cover types (forbs, shrubs, dominant, and subordinate grasses) in 2002 to that in 2001, using non-parametric Mann-Whitney U tests in SPSS (SPSS 1998; used for all univariate analyses). Cover of Idaho fescue and bluebunch wheatgrass was summed as dominant grass cover, with the remaining grass cover consisting of subordinate species.

We then selected a set of 16 burned plots as similar in prefire (2001) community composition as possible to the unburned plots in 2001 by calculating a Sørensen distance matrix in PC-ORD, then editing it so that unburned plots were rows and all candidate prefire plots were columns. For each candidate plot, we averaged the Sørensen distance to each unburned plot, then selected the 16 candidate prefire plots with the smallest average distance. To evaluate success of this selection in terms of prefire similarity of the selected plots and the unburned plots, we compared the overall plant community with MRPP, using a

Sørensen distance and data relativized by species totals. We also compared cover of each individual species and the previously mentioned basic cover types between the selected plots and the unburned plots with Mann-Whitney tests.

To evaluate responses of species cover and basic cover types to fire, we compared cover between 2001 (prefire) and 2002 (postfire) in the 16 selected burned plots with Mann-Whitney tests. Next, to evaluate the difference between changes related to year-to-year variation and changes related to fire, we calculated the change in species and basic cover types between 2001 and 2002 in burned and unburned areas, then compared these changes between unburned plots and burned plots with Mann-Whitney tests. We included only the species that were present in both burned and unburned areas, and with average cover $> 0.025\%$ for ≥ 1 yr during the study. To evaluate changes in overall (multivariate) community composition between 2001 and 2002 in burned areas, and to compare 2002 composition in burned and unburned areas, we used MRPP, a Sørensen distance and data relativized by species totals.

To assess changes in catchfly genet abundance with and without fire, we report year-to-year changes in abundance for unburned and burned sites in LCC and for CGC sites that had burned the year before sampling began (to compare 1-yr versus 2-yr postfire). Changes in abundance may relate to several factors, including breaking or inducing of dormancy, recruitment, mortality, yearly climatic differences, whole-plant herbivory, or sampling error.

We compared levels of flowering in 2001 to those measured in 2002 in 12 LCC sites that burned and the 9 that remained unburned using paired t-tests within each treatment. In the half-burned LCC site, we used the same prefire flowering data for both halves, and hereafter analyze each half as a separate site. We also compared 2001 and 2002 flowering in five CGC sites with paired t-tests.

We made multivariate comparisons of vigor measures between burned and unburned LCC sites sampled in 2002 using MRPP and a Euclidean (Pythagorean) distance measure. Before the analysis, we calculated the percentage of flowers that matured to seed-bearing capsules as the number of filled capsules per stem divided by the number of flowers per stem, multiplied by 100. Prior to MRPP analysis, all data were adjusted to their

standard deviate, such that each transformed value represented the number of standard deviations by which it differed from the variable mean.

To assess whether our comparisons of burned and unburned vigor reflect environmental differences that were not caused by fire, we used univariate one-way analysis of variance (ANOVA) to compare abiotic environmental variables (aggregated across plots within burned and unburned sites) between burned and unburned sites. We calculated potential annual direct incident radiation (PDIR) and heat load based on slope, aspect, and latitude (McCune and Keon 2002). When examining aspect as an individual variable, we treated it as continuous past 360° (i.e. $10^\circ = 370^\circ$). With this treatment, 1° was similar to 360° , which is more conducive to describing similarities in light and warmth regimes within the narrow NW to NE range of catchfly habitat.

We also compared individual population vigor variables measured in 2002 (stems/plant, stem height, flowers and capsules/stem, seeds/capsule, and percentage of flowers matured to seed-bearing capsules) between burned and unburned LCC sites using one-way ANOVA. The number of seeds per capsule was natural log transformed prior to analysis to equalize variance. We did not include the five CGC sites in the ANOVA since there were no prefire data or any unburned CGC sites for comparison. We do, however, report data from CGC sites since they represent the only available sample of catchfly vigor the second season after fire. We did not test differences in vigor between study areas; CGC populations tended to support fewer plants and had greater variation in vigor parameters than LCC populations. We did not calculate flower to seed-filled capsule ratios for a CGC site that produced no flowers. The level of significance for all tests was set at $P=0.05$.

Results

Plant community response

In summer 2001 (prefire), overall community composition was similar in unburned-site plots and the selected plots in sites that would burn in fall 2001. Unburned-site plots tended to have less cover of Idaho fescue, less dominant grass cover, and less total grass cover than plots that would burn in fall 2001. The relative composition of communities in unburned areas did not change

TABLE 2. Mean percent cover of plant community components in 2001 and 2002 in 16 areas that burned between those years, and 16 areas that remained unburned. Within the burned or unburned column, cover values that changed significantly between years are followed by different superscript letters. Differences in the yearly changes between burned and unburned areas were also compared; *P*-values reflect statistical significance of differences in the magnitude or direction of year-to-year changes between burned and unburned areas.

Component	Unburned		Burned		<i>P</i>
	2001	2002	2001	2002	
Dominant Grasses	41.2 ^a	57.2 ^b	56.9 ^c	30.4 ^b	<0.001
Subordinate Grasses	4.97	1.13	7.19	8.41	0.316
All Grasses	46.1	58.3	64.1 ^a	38.8 ^b	<0.001
Forbs	24.6 ^a	41.2 ^b	14.8 ^a	41.5 ^b	0.417
Shrubs	6.41	8.75	1.41	1.72	0.261
Total cover	77.2 ^a	108.2 ^b	80.2	82.0	0.029

significantly between 2001 and 2002. The postfire (2002) community composition of burned areas differed significantly from their prefire (2001) composition, and also differed from the community composition of unburned areas in 2002 (both $P < 0.003$).

In areas remaining unburned for the entire study, absolute total vegetative cover, forb cover, and dominant grass cover increased between 2001 and 2002 (Table 2), although total grass cover, subordinate grass cover, and shrub cover remained similar between years. In contrast, in burned areas (~1 yr after fire), total cover remained similar between years, and total grass cover decreased with fire, driven by decreases in dominant grasses. Forb cover increased with fire while shrubs, which had relatively little prefire cover in burned areas, maintained similar cover after fire. The comparison of year-to-year changes between burned and unburned sites suggests that burning may have diminished growth in total cover, primarily through loss of dominant grass cover. Fire did not appear to influence significantly the dynamics of forb cover and subordinate grass cover.

Changes in individual species suggest that fire reduced cover of Idaho fescue, but did not change cover of bluebunch wheatgrass (Table 3). Silky lupine (*Lupinus sericeus*) increased in burned areas while remaining relatively constant in unburned

areas. Cover of the exotic annual forb thymeleaf sandwort (*Arenaria serpyllifolia*) and the native forb lambstongue ragwort (*Senecio integerrimus*) increased in burned areas from low (or zero) prefire levels. Cover of catchfly and most exotic annual grasses did not change in burned or unburned areas. Cover by cheatgrass and yellow starthistle, exotic species of particular concern at GCRP, was minimal in our plots, did not increase significantly after fire.

Catchfly Population Response

Our pre- and postfire observations of catchfly genet abundance within a subset of LCC sites suggest that catchfly did not experience fire-related mortality (Table 4). Numbers of adult plants increased in burned LCC sites, which may reflect fire-stimulated breaking of dormancy; abundance increased in only one of the four unburned LCC sites. Plant abundance 1-2 yr postfire in CGC was nearly constant across 2001 and 2002; few, if any new adult plants resulted from recruitment the first year after fire. We found few seedlings in either year of the study, and did not observe substantial flushes of germination the first or second season after fire.

Within LCC and CGC, levels of flowering in burned and unburned sites were similar between years (Figure 1). Overall population vigor in burned and unburned LCC catchfly sites did differ significantly ($P = 0.041$). The abiotic environment did differ slightly between burned and unburned LCC sites, with unburned sites tending to occur on warmer aspects (mean difference: 25.6°) with greater predicted annual heat loads than unburned sites (both $P < 0.01$).

Neither the number of flowers or capsules per catchfly stem differed significantly between burned and unburned sites the first season after fire (Figure 2). Stem heights and percentages of flowers successfully matured to seed-filled capsules were significantly smaller in burned compared to unburned sites in LCC. The mean number of stems per plant (data not shown) and seeds per capsule did not differ significantly between burned and unburned sites the first season after fire. Stem height in CGC, 2 yr after fire, was intermediate between unburned and 1 yr postfire sites in LCC. Numbers of seeds per capsule at CGC did not differ from numbers in burned or unburned LCC sites.

TABLE 3. Mean percent species cover in 2001 and 2002 in 16 areas burned between those years, and 16 areas that remained unburned. Within the burned or unburned column, cover values that changed significantly between years are followed by different superscript letters. Differences in the yearly changes between burned and unburned areas were also compared; *P*-values reflect statistical significance of differences between burned and unburned areas in magnitude and direction of year-to-year changes. t = mean species cover < 0.025%.

Species	Unburned		Burned		<i>P</i>	
	2001	2002	2001	2002		
Grasses						
Rattlesnake brome	<i>Bromus briziformis</i>	t	0.28	0.03	0.09	0.44
Japanese brome	<i>B. japonicus</i>	0.19	0.03	0.47	0.47	0.93
Cheatgrass	<i>B. tectorum</i>	0.03	t	t	0.03	0.16
Idaho fescue	<i>Festuca idahoensis</i>	26.5	38.1	42.7 ^a	8.3 ^b	<0.01
Junegrass	<i>Koeleria macrantha</i>	4.75	0.78	6.69	7.81	0.14
Blucbunch wheatgrass	<i>Pseudoroegneria spicata</i>	14.7	19.0	14.2	12.1	0.22
Forbs						
Common yarrow	<i>Achillea millefolium</i>	2.03	3.25	1.72	2.16	0.77
Thymeleaf sandwort	<i>Arenaria serpyllifolia</i>	t	t	t ^c	4.34 ^b	-
Twin arnica	<i>Arnica sororia</i>	t ^a	2.22 ^b	t ^c	2.00 ^b	0.57
Arrowleaf balsamroot	<i>Balsamorhiza sagittata</i>	3.94	5.38	3.38	6.31	0.54
Red besseya	<i>Besseya rubra</i>	t	0.06	t	0.06	1.00
Paintbrush	<i>Castilleja</i> sp.	t	t	0.03	1.28	-
Field chickweed	<i>Cerastium arvense</i>	4.22	11.2	0.19	0.06	0.08
Yellow starthistle	<i>Centaurea solstitialis</i>	t	t	t	0.19	-
Erigeron	<i>Erigeron</i> sp.	0.44	0.47	0.47	0.19	0.75
Whitestem fraseria	<i>Frasera albicaulis</i>	1.56	2.50	2.38	4.00	0.65
Stickywilly	<i>Galium aparine</i>	t	0.06	t	0.25	0.59
Old man's whiskers	<i>Geum triflorum</i>	1.34	1.38	0.38	0.38	0.70
Houndstongue hawkweed	<i>Hieracium cynoglossoides</i>	0.78	1.28	0.88	2.28	0.95
Common St. Johnswort	<i>Hypericum perforatum</i>	0.19	0.03	0.03	0.03	0.55
Western stone seed	<i>Lithospermum ruderale</i>	0.75	2.34	0.25	0.66	1.00
Silky lupine	<i>Lupinus sericeus</i>	4.69	5.53	3.25 ^a	13.7 ^b	0.02
Gairdner's yampah	<i>Perideridia gairdneri</i>	t	0.06	t	0.09	0.63
Stickystem Penstemon	<i>Penstemon glandulosus</i>	1.13	0.63	0.19	0.03	0.41
Snake River phlox	<i>Phlox colubrina</i>	t	0.03	t	0.09	0.29
Slender cinquefoil	<i>Potentilla gracilis</i>	0.03	0.41	0.22	0.75	0.70
Lambstoungue ragwort	<i>Senecio integerrimus</i>	0.03	t	t ^a	1.19 ^b	0.02
Spalding's catchfly	<i>Silene spaldingii</i>	1.28	1.13	1.44	1.28	0.98
Yellow salsify	<i>Tragopogon dubius</i>	t	0.03	0.03	0.03	1.00
Meadow deathcamas	<i>Zigadenus venenosus</i>	t	0.28	t	0.03	0.14
Shrubs						
Rose	<i>Rosa</i> spp.	0.19	2.06	0.09	1.16	0.91
Common snowberry	<i>Symphoricarpos albus</i>	6.22	6.69	1.31	0.56	0.14

Discussion

Total vegetative cover in catchfly habitat increased significantly in unburned, but not burned areas, between the 2 yr (Table 2). Under the water-limited conditions during summer at GCRP, recovery of the plant community and catchfly in burned areas may have been limited, perhaps by reduced moisture availability resulting from litter loss (De Jong and MacDonald 1975, Redmann 1978). It

does not appear that differences in plant community or catchfly dynamics after fire were related to aspect or heat load between burned and unburned sites. Burned sites were on cooler aspects, which, in theory, should be less water-limited and more optimal for plant growth and reproduction than the warmer aspects of unburned sites. However, increases in plant cover between years were typically smaller on burned than unburned sites, and most catchfly vigor measures did not differ

TABLE 4. Catchfly abundance (number of plants per site) in 2001 and 2002, and changes between years.

Study area (Fire history)	Site #	Catchfly Abundance		
		2001	2002	Change
Lower Corral Creek (Burned between 2001 and 2002)	56	19	27	8
	57-1	11	12	1
	57-2	12	13	1
	75	19	30	11
	86	2	22	20
Lower Corral Creek (Unburned)	80	16	16	0
	223	35	38	3
	238	59	40	-19
	271	4	4	0
China Garden Creek (Burned in 2000)	128-2	6	7	1
	128-1	2	2	0
	144	7	7	0
	128-w	2	2	0

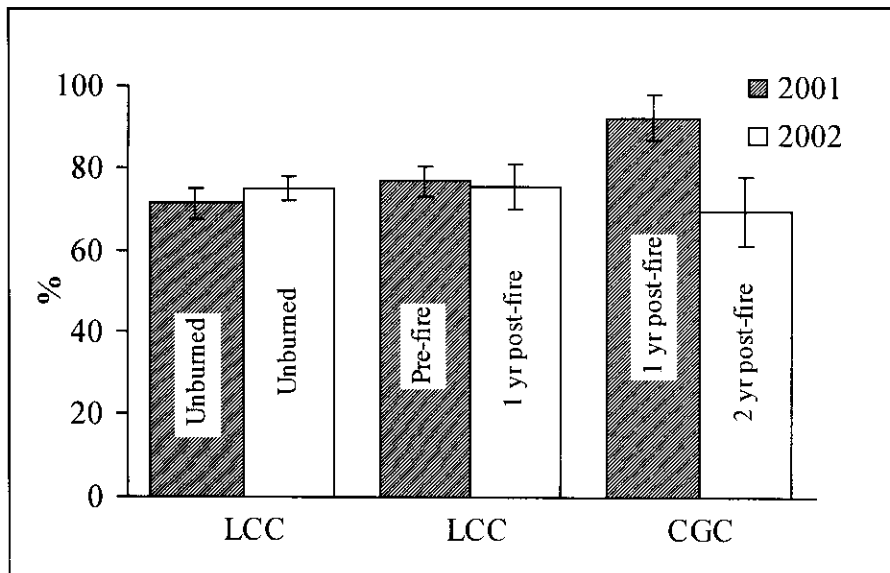


Figure 1. Percent flowering of catchfly stems (means \pm SE) in 2001 and 2002 for 9 unburned and 12 burned Lower Corral Creek sites, and 5 China Garden Creek sites. Differences between years within groups were compared with paired t-tests, and were not statistically significant.

between these site types. Additional research at GCRP (Menke and Muir 2004) also found no correlation between aspect or heat load and catchfly vigor measurements.

It appears that fire shifted (at least temporarily) the competitive balance within plant communities in catchfly habitat at GCRP. Dominant grass

cover decreased, and some subordinate species (grasses and forbs) increased to occupy more area in burned areas (Tables 2 and 3). In a Minnesota old-field community, McGinley and Tilman (1993) found similar results. They suggest burning may have liberated above and belowground resources from use by dominant species.

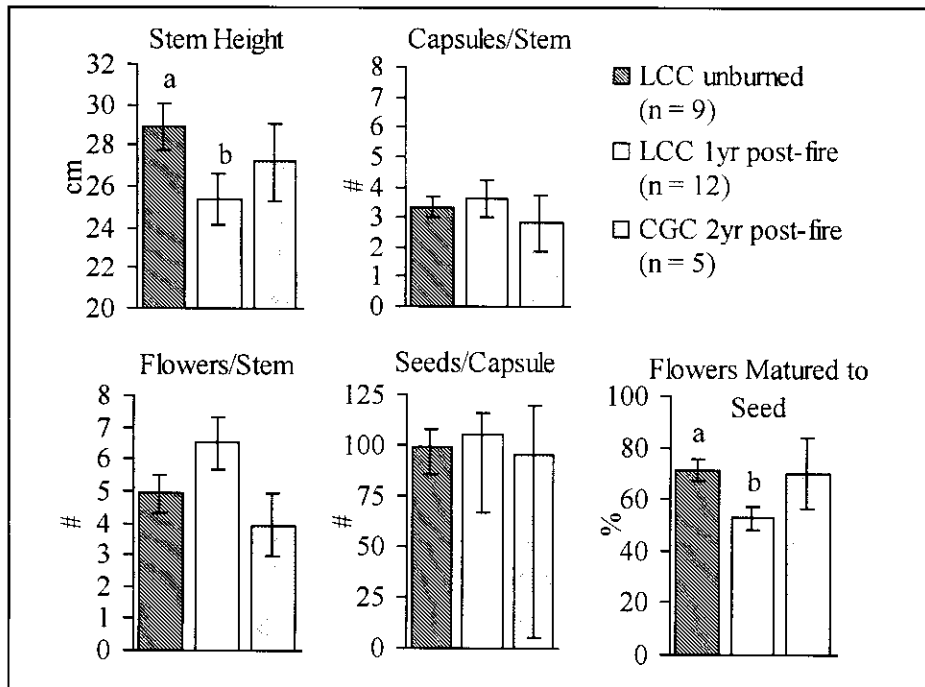


Figure 2. Catchfly vigor components (means \pm SE) measured in 2002 and averaged across sites. Number of seeds/capsule was natural log transformed prior to analysis; back-transformed median and quartiles are shown. Vigor measures for burned and unburned Lower Corral Creek sites were compared with one-way ANOVA and post-hoc LSD tests. Different letters on bars within a graph indicate $P < 0.05$ for the comparison (China Garden Creek sites were not included in ANOVAs).

Idaho fescue showed the largest decreases after fire, while cover of bluebunch wheatgrass did not change significantly (Tables 2 and 3). Our results are similar to postfire findings for both of these species in northeastern Oregon grasslands (Johnson 1998), for Idaho fescue in western Washington prairies (Tveten and Fonda 1999), and for bluebunch wheatgrass in a western Montana grassland (Antos et al. 1983). Wheatgrasses tend to accumulate less fine litter around their meristems, making them more fire tolerant than fescues (Wright and Bailey 1982). Differences may also reflect greater drought tolerance in bluebunch wheatgrass than Idaho fescue (Tisdale and Hironaka 1981, Defosse and Robberecht 1996), as fire generally promotes drought-like conditions.

A primary concern regarding burning, prescribed or natural, at GCRP, is that it may promote the spread and growth of subordinate grasses (particularly in exotic species), which may capitalize on newly released resources before they are sequestered by dominant bunchgrasses. Greater

dominance by exotic annuals would likely be detrimental to sensitive plant species at the preserve, including catchfly. Cheatgrass, well known to increase in dominance after burning (Melgoza et al. 1990), did not change significantly from its minimal prefire cover in our burned catchfly sites (Table 3). The most common exotic annual grass coexisting with catchfly at GCRP is Japanese brome (*Bromus japonicus*). It often decreases with fire, as it depends on litter for seedling germination and establishment (Whisenant 1990). We, however, did not observe such a decrease (Table 3), and in adjacent, comparable community types in eastern Oregon, Japanese brome remained constant or increased after fire (Johnson 1998). The other subordinate, but native and perennial, grass common in catchfly habitat is prairie Junegrass (*Koeleria macrantha*), which often increases after fires (Wilson and Shay 1990, Erichsen-Arychuk et al. 2002). Junegrass may be fire tolerant since it is smaller and accumulates less litter than fescues or wheatgrasses (Wright and Bailey 1982).

In our study, however, cover by this species did not change significantly after fire (Table 3). Fire did not appear to stimulate gains in subordinate grasses cover that could threaten catchfly populations on the preserve.

In burned catchfly habitat, forb cover was the only cover type to increase significantly from its prefire levels; total grass cover decreased and shrub cover remained constant (Table 2). Forbs in burned and unburned sites appeared to respond to the greater precipitation in 2002 as compared to 2001. Although total forb cover increased about 10% more between years in burned sites than in unburned sites, the between year change was not significantly different. Other postfire studies in grassland systems found greater forb cover after fire (Antos et al. 1983), while others found forb cover remained constant after fire (Erichsen-Arychuck et al. 2002). In northeastern Oregon plant communities similar to those at GCRP, postfire cover of arrowleaf balsamroot (*Balsamorhiza sagittata*) and field chickweed (*Cerastium arvense*) decreased, whitestem fraseria (*Frasera albicaulis*) increased, and twin arnica (*Arnica sororia*) responded variably (Johnson 1998). Cover of Snake River phlox (*Phlox colubrina*) and stickystem penstemon (*Penstemon glandulosus*) tended to remain constant (Johnson 1998). At GCRP, changes between 2001 and 2002 in cover of all of these and most other forb species did not differ significantly between burned and unburned sites (Table 3). In general, forbs appear to have recovered completely in the first growing season after fire.

Silky lupine (*Lupinus sericeus*) showed the strongest positive response to fire in catchfly communities (Table 3). A similar positive postfire response was reported for lupine in Yellowstone National Park (Tracy and McNaughton 1997). Research in Montana found no change in its cover after fire (Antos et al. 1983). Various lupine species have been found to increase in abundance or cover after fire (Grigore and Tramer 1996, DiTomaso et al. 1999). The changes in lupine abundance after fire (Johnson 1998) may relate to fire timing (Tveten and Fonda 1999), community composition, or the amount of resources freed from dominant species. In grasslands that may be nitrogen deficient following fire, the ability to fix nitrogen is probably advantageous, and may promote lupines.

How did these community changes, particularly the decreases in dominant grass cover, and

postfire environmental conditions, influence catchfly populations? Several measures of catchfly vigor remained constant in postfire conditions, suggesting that the species is well adapted to fire, as were most other forb species in the community. Similar to Lesica (1999) we saw no significant differences in percentages of catchfly flowering (Figure 1), or in the mean number of flowers per stem (Figure 2) after fire. Burned and unburned areas were also similar in seed production (numbers of seeds per capsule) and in numbers of capsules per stem. In contrast to research in Montana, we did not observe greater catchfly numbers resulting from enhanced recruitment 1-2 yr postfire, which may relate to differences in associated grass species and characteristics of their litter.

Burning may have stimulated mature plants to break dormancy and emerge the season after fire; in our sites, the number of mature catchfly plants in burned sites increased in the first year after burning (Table 4). These results are similar to those in Montana catchfly populations, where burning reduced or shifted the proportion of dormant catchfly plants in populations (Lesica 1999). The proportions of dormant plants varied widely between years in Montana, which is also likely the case at GCRP. The causes and mechanism for such changes in dormancy are unknown.

Two aspects of catchfly vigor decreased with burning: stem height and the proportion of flowers matured to seed filled capsules (Figure 2). In other grassland systems, research suggests fire may reduce forb growth either via damage to tissues or by increased belowground competition from recovering perennial grasses after fire (Hartnett 1991). Since the fires at GCRP occurred after catchfly plants had set seed (and above-ground tissues do not over-winter), we suspect the fire merely topkilled already senescing plants. Due catchfly's deep taproot, it is unclear how important greater belowground competition after fire would be for mature catchfly plants. Further research comparing competition, seed production, and viability in areas with differing fire histories might clarify the influence of fire on this species.

Overall, catchfly and other plant species in associated communities appear to tolerate fire and postfire conditions. Losses in dominant grass cover and gains in native forb cover enhance diversity within plant communities, and such changes may be beneficial for sensitive plant species, including

catchfly, at the preserve. Burning removed no native taxa from the community, nor did it stimulate widespread growth of exotic species (at least in catchfly habitat the first year after fire). However, because exotic species are present at the preserve, and co-exist with catchfly at many sites, we suggest caution with the use of fire to restore or maintain these communities. Catchfly habitat includes some of the most intact plant communities at the preserve. Disturbance could promote spread of exotics from areas of greater weed density into catchfly habitat. Additional research is needed to investigate the influence of burning season, fire frequency, and fire severity on catchfly and the associated community at the preserve.

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