Natural Resources and Environmental Issues

Volume 16 *Shrublands: Wildlands and Wildlife Habitats* Article 12

2011

Post-fire Recovery of Wyoming Big Sagebrush Steppe in Central and Southeast Montana

Stephen V. Cooper

Peter Lesica

Greg M. Kudray

Post-fire Recovery of Wyoming Big Sagebrush Steppe in Central and Southeast Montana

Stephen V. Cooper[1], Peter Lesica[2], and Greg M. Kudray[3]

ABSTRACT

Big sagebrush (Artemisia tridentata *Nutt.*) *steppe is a widespread habitat throughout eastern Montana and supports several sagebrush-dependent species including Greater Sage-grouse* (Centrocercus urophasianus). *We sampled 24 burned-unburned paired sites in central and southeastern Montana dominated by Wyoming big sagebrush* (*ssp.* wyomingensis). *Time since fire ranged from 4 to 67 years. Prescribed burns and wildfires typically resulted in the complete mortality of big sagebrush. We found negligible post-fire Wyoming big sagebrush recovery for 17 of the 24 sites and the oldest burn (67 years) was only 8 percent recovered. Perennial grass cover increased 27 percent and 20 percent following prescribed fire and wildfire, respectively; western wheatgrass* (Pascopyrum smithii) *accounted for most of this increase. Annual grass cover increased by 11 percent due primarily to field brome* (Bromus arvensis, *formerly* B. japonicus). *Plant species richness significantly declined in burned plots compared to unburned controls. There was no change after burning in overall sub-shrub or forb cover or the density of Cichorieae forbs that are important for successful Greater Sage-grouse brood rearing. Managers concerned about Greater Sage-grouse and other sage-dependent species should be extremely cautious with prescribed burns and wildfires in this region. Fire will likely eliminate sagebrush habitat, increase weedy annual grass cover and reduce species richness; sagebrush cover could take a century or more to recover to pre-burn conditions.*

INTRODUCTION

Sagebrush steppe is a dominant vegetation type in the Great Basin and Intermountain Region of western North America but it is also important in portions of the Northern Great Plains where agriculture (cereal grains) and mixed-grass prairie now dominate. Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* Beetle and Young) - dominated vegetation is an important component of the semiarid landscapes east of the Rocky Mountains stretching from Wyoming through Montana and western North Dakota to just south of the Canadian border. Physiognomy of Wyoming big sagebrush stands in the Northern Great Plains differs from that of the Intermountain Region by having undergrowth generally dominated by rhizomatous grasses as opposed to tussock-forming grasses. Also influencing stand physiognomy are two notable clines in Wyoming big sagebrush size presumed to reflect available soil moisture; one of increasing plant height from 1) south to north and 2) from lower to higher elevation.

Fire was instrumental in structuring presettlement sagebrush ecosystems, generating a mosaic of stands of different size and various seral stages (West 2000). Conserving native species diversity is expected to require maintaining a comparable mosaic. Greater Sage-grouse (*Centrocercus urophasianus*), for example, require barren habitats for leks, relatively dense stands of medium height for nesting (Klebenow 1969; Wallestad and Pyrah 1974; Aldridge and Brigham 2002), open stands for brood raising (Klebenow 1973; Wallestad 1971), and full-canopied tall stands for wintering (Eng and Schladweiler 1972). Greater Sage-grouse populations apparently can be constrained by the loss of any one of these structural types (Connelly and others 2000; Roscoe 2002). Antelope, Brewer's Sparrow, Sage Sparrow, Sage Thrasher and Sagebrush Vole are also sagebrush dependent at some stage of their life cycles.

Management strategies that promote the conservation of all sagebrush steppe-dependent species are currently being formulated, and prescribed fire has been proposed as a method to control the density of big sagebrush stands (Klebenow 1973; Pyle and Crawford 1996). However, we only have limited knowledge of changes in sagebrush cover, height, associated vegetation and other characteristics as succession proceeds from immediate post-treatment to mature structure. Though considered a climax-dominant species, evidence suggests that big sagebrush fire response varies according to subspecies and may require many years for post-fire re-establishment (Baker 2007). Wyoming big sagebrush, although highly variable in response (Walhof 1997; Wambolt and others 2001; Watts and Wambolt 1996), demonstrates little recovery for the first 30 years (Wambolt and Payne 1986; Eichhorn and Watts 1984; Lesica and others 2007) and generally requires at least 50 years to attain a density equal to that of the unburned control (Baker 2007; Colket 2003); a state-and-transition model developed for ssp. *wyomingensis* within the Columbia Basin is distinctly more optimistic envisioning recovery in approximately 30 years (Hemstrom and others 2002). With the lone exception of the Eichhorn and Watts (1984) study in central Montana's Missouri River Breaks, none of these studies were conducted in a Great Plains environment. The ecological dynamics and habitat characteristics of these sagebrush communities are

In: Wambolt, C.L. et al. comps. 2011. Proceedings – Shrublands: wildlands and wildlife habitats; 2008 June 17-19; Bozeman, MT. NREI, volume XVI. S.J. and Jessie E. Quinney Natural Resources Research Library, Logan, Utah, USA.
[1]Stephen V. Cooper, Montana Natural Heritage Program (Retired), Helena, MT
[2]Peter Lesica, Conservation Biology Research, Missoula, MT
[3]Greg M. Kudray, Montana Natural Heritage Program, Helena.

almost certainly strongly influenced by their age (size) structure. Landscape scale comprehensive management of sagebrush cannot be achieved without understanding how structural and compositional components change with time since disturbance.

We also have little knowledge of how undergrowth vegetation changes with succession. Fire often increases the abundance of grasses (Harness and Murray 1973; Wambolt and Payne 1986). Forbs, especially those in the *Cichorieae* Tribe of the *Asteraceae*, are an important component of sage grouse diet during the summer and are often critical for successful brood rearing (Klebenow and Gray 1967; Peterson 1970; Barnett and Crawford 1994; Drut and others 1994). At what age do stands provide the most succulent forbs for sage grouse brood rearing, the best nesting cover for grouse, the best habitat for Brewer's sparrows and sage thrashers? Fire is reported to have positive effects on grass productivity in other regions of the western U.S. (Pechanec and others 1954; Harniss and Murray 1973; Bunting and others 1998), but there is little information on whether this is true for the Great Plains.

The purpose of this study was to describe and model the change in sagebrush and associated vegetation after fire in the Northern Great Plains of eastern and central Montana. We documented changes in shrub height, cover and size-class distribution and canopy cover of undergrowth herbaceous species by sampling burned and adjacent unburned stands of various post-fire ages.

STUDY AREA

Sampling was conducted over a broad swath of eastern Montana from a westernmost site within the Bighorn Basin to the glaciated plains north of the Missouri River. Most of the sampling occurred in the Tongue and Powder River Basins and in the Missouri River Breaks (figure 1). Elevations ranged from 270 to 1,220 m (890 to 3,990 ft). Annual precipitation varied from 274 mm (10.8 in) in the Missouri River Breaks to 415 mm (16.35 in), on plateaus east of the Powder River (DAYMET model, Thornton and others 1997). Approximately 45 percent of this precipitation occurs in the biologically critical spring quarter (April-June). The continental climate is characterized by strong seasonal (winter to summer) and diurnal temperature fluctuations. All the sampling sites were characterized as flat to gently rolling with no slopes greater than 13 percent and were post-sampling keyed (based on laboratory determination of soil texture) to one of three Natural Resource Conservation Service (NRCS) "Ecologiocal Site" types; Silty (Si, 58 percent of sites); Clayey (Cy, 32 percent); Sandy (Sy, 10 percent). Soil texture ranged from clays to sandy loam with no true "sands" type being represented.

Wyoming big sagebrush was the only big sagebrush subspecies at sampling sites, although silver sagebrush (*A. cana* ssp. *cana*) occurred on nearby stream terraces and sites having a greater percentage of sand in the soil. All the sites were characterized as flat to gently rolling with no slopes greater than 13 percent. Soils texture ranged from clays to sandy loam. Dominant undergrowth grasses were bluebunch wheatgrass (*Pseudoroegneria spicata*) on the driest sites to Idaho fescue (*Festuca idahoensis*) on the most mesic sites; the most common native species were western wheatgrass (*Pascopyrum smithii*), green needlegrass (*Nassella viridula*), Sandberg's bluegrass (*Poa secunda*), blue grama (*Bouteloua gracilis*), and needle-and-thread (*Hesperostipa comata*).

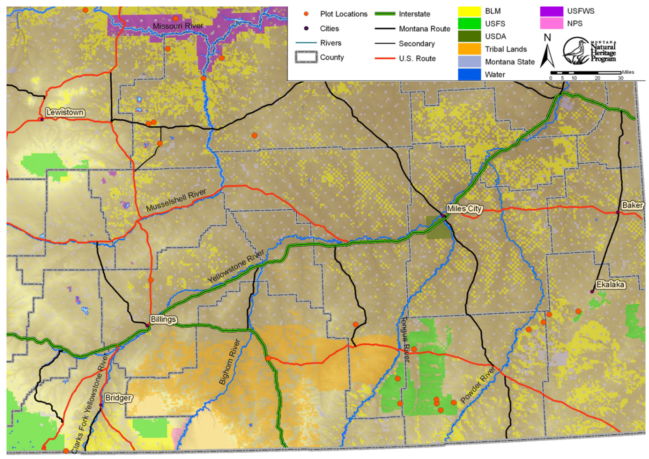

Figure 1—Study area map with sampling sites (plots) indicated.

METHODS

Field Methods

In June and July of 2006 and 2007 we sampled 24 sites dominated by Wyoming big sagebrush in central and southeastern Montana. We used lists of potential sites provided by the Miles City Office of the Bureau of Land Management (BLM), the Ashland Ranger District of the Custer National Forest, and personal communication with their respective staffs to select sampling sites based on age of burn and accessibility. Larry Eichhorn, retired BLM range conservationist from Lewistown, provided information valuable in relocating the original sample sites of his study of post-fire succession in central Montana (Eichhorn and Watts 1984). We focused on federally or state owned lands but fortuitously encountered several privately held sites with cooperative owners.

At each site a 1000 m^2 macroplot (20 m by 50 m) was visually selected to represent prevailing conditions within the burned area and be as close as feasible and preferably on the downwind side of the unburned area. A "control" sample macroplot was established in unburned sagebrush-dominated vegetation as close as possible to the burn. The control was chosen to be as similar as possible to the abiotic setting (slope, aspect, soils) of the burned sample plot. Although the unburned control macroplots are not true controls because of not being randomly assigned prior the fires, nonetheless they function as controls by exemplifying what the burned plot probably would have constituted, had they not burned. With one exception, burned macroplots were located within 25 m or less of the unburned control and always in the same grazing pasture (not separated by fencing). We noted the positions of fence lines and water developments and attempted to locate sampling points as far removed as possible to ensure that grazing pressure was not excessive. However, we had no way of accurately accounting for grazing regimes.

We used line intercept to estimate shrub canopy cover along five evenly-spaced, parallel 20 m transects originating at the 50 m macroplot baseline (Mueller-Dombois and Ellenberg 1974). Circular microplots 3m^2 were established at the 5 and 10 m marks of the five transect lines (10 total) for determining rooted density for all shrub species by four size classes: (1) seedlings (< 10 cm high), (2) juveniles (> 10 cm and stem diameter at ground level < 1 cm), (3) sub-adults (stem 1 to 3 cm diameter), and (4) adults (> 3 cm stem diameter). Age and height were recorded for one sagebrush plant of each size class in alternate microplots (5 total), focusing on specimens exhibiting the least crown damage. Sagebrush plants were cut with a fine-blade saw or sharp pruning shears at ground level (usually required removing accumulated detritus from around mature stems). Annual growth rings were field counted with a 10X or 20X hand lens (Ferguson 1964). It was often necessary to sample sagebrush plants outside the microplots to ensure that we had at least three estimates for each size class. Rotten stems or those damaged by grazing defied even approximate age determination in many plants. We frequently experienced stands where all sub-adult and adult specimens we sampled (10 to 20 specimens) could not be accurately aged; the smaller size classes had significant stem damage as well. The only burn information recorded was ignition source, either wildfire or prescribed burn, and the year of occurrence. We were unable to consistently characterize fire severity, a potentially significant explanatory variable because immediate post-fire conditions were not generally recorded at the time and quite a number of the burns were old enough (20+ years) that potentially telling clues had been obscured.

We estimated canopy cover (Daubenmire 1959) of all herbaceous species, bare soil, gravel, rocks, litter, lichens, mosses, and basal vegetation in the ten microplots using 13 cover classes (T^- = >0, ≤0.1; T = >0.1, ≤1; P = >1, ≤5; 1 = >5, ≤15; 2 = >15, ≤25, 3 = >25, ≤35; 4 = >35, ≤45; 5 = >45, ≤55; 6 = >55, ≤65; 7 = >65, ≤75; 8 = >75, ≤85; 9 = >85, ≤95; F = >95, ≤100 percent). We also recorded the number of plants of each species of the Cichorieae tribe of the Asteraceae in each of the ten microplots. Taxonomic nomenclature follows the Plants Database (USDA Natural Resources Conservation Service 2008)

Data Analysis

We characterized Wyoming big sagebrush recovery as the ratio of mean canopy cover or height for the burned macroplot to the corresponding values from the unburned control macroplot. Sagebrush stand height was the mean of the size class with the greatest canopy cover. Rate of recovery for sagebrush was calculated as the percent recovery for either canopy cover or height divided by the number of years since burning. Species richness was measured by the number of vascular plant species recorded in the line intercepts and ten microplots.

The relative aridity of a site, as measured by precipitation and potential evapotranspiration, was hypothesized to affect recruitment and other aspects of stand recovery. Slope and aspect are the primary determinants of potential evapotranspiration; these two variables along with latitude have been integrated into a "heat loading" index by McCune and Keon (2002). Average site annual precipitation was estimated by DAYMET a statistical model that integrates elevation, other aspects of local terrain, and geographic position with weather station data for the past 20 years (Thornton and others 1997).

The Wilcoxon signed-rank test (statistic noted as ***T***) was used to evaluate the differences between burned and unburned control macroplots for Wyoming big sagebrush canopy cover, total shrub cover and cover of perennial grasses, annual grasses, forbs and selected individual species; our data was congruent with the assumptions of this non-parametric test. Linear regression analysis was used to model the recovery of sagebrush height and canopy cover and herbaceous cover with time since fire; sagebrush recovery regression lines were forced through the origin to reflect biological realities. Regression analysis was also used to test the association between sagebrush recovery rate and the abiotic site factors of precipitation, heat load index and soil texture.

RESULTS

Wyoming Big Sagebrush and Shrub Recovery

The mean time since fire is 22 ± 16 (16 = 1 std. dev.) years, ranging from 4 to 67 years (median of 16 years) for the 24 sampled paired macroplots (control and burned). Fire resulted in virtually a complete loss of shrub canopy cover as revealed by examination of recently burned macroplots (≤10 years, $N = 6$); five of the six plots had no shrub canopy cover and one had < 2 percent. Wyoming big sagebrush is the dominant shrub on the control macroplots with an average cover of 20 ± 8 percent, whereas on burned plots cover of this species averaged 0.81 ± 1.8 percent (difference highly significant, Wilcoxon's $T = 0$, $P = << 0.01$). Total shrub cover of control plots (21 ± 8 percent) averaged only slightly more than the cover of Wyoming big sagebrush alone, with the additional species most commonly including silver sage, rubber rabbitbrush (*Ericameria nauseousa*), green rabbitbrush (*Chrysothamnus viscidiflorus*) and spineless horsebrush (*Tetradymia canescens*). The difference in total shrub cover between control and burned plots (1.2 ± 2.1 percent) was highly significant ($N = 24$, Wilcoxon's $T = 0$, $P = << .01$). Silver sagebrush was the only shrub even approaching Wyoming big sagebrush in cover and that occurred on only one sandy loam site. The average height of the dominant cohort of Wyoming big sagebrush in control plots was 61±11 cm.

A linear model resulted in the best fit for the recovery of Wyoming big sagebrush canopy cover with age since fire explaining 29 percent of the variation in cover (t = 2.81, $P = 0.010$; figure 2). Results for total shrub cover recovery were similar. The mean recovery rate for Wyoming big sagebrush canopy cover was 0.16 percent / year ± 0.45 for the first 70 years. Height recovery of the dominant Wyoming big sagebrush cohort was best fit by a highly significant ($t = 4.81$, $P = <0.001$) exponential function (figure 3) in which time since fire explained 55 percent of the variation and the extrapolated time of recovery is approximately 68 years. We did not stratify the dataset according to mode of ignition because there were only six prescribed burn sites and four of these showed no recovery in big sagebrush cover; in addition fire effects, based on stump height, were the same between the two ignition modes. Of the 24 burned macroplots the seedling class was represented in only 4 macroplots, the juvenile class in 5 macroplots, the subadult class in 6 macroplots, and the adult class in 4 macroplots.

Heat load varied by only 12 percent between the most extreme of study sites, an expected result in rolling plains where the steepest slope study site slope was only 13 percent. Variation in understory herbaceous vegetation among the control plots was more likely due to the 50 percent difference in annual precipitation (10.8 to 16.3 inch [274 to 414 mm]) and soil texture. A linear regression model incorporating the heat load index and mean annual site precipitation explained 30 percent of the variation in the rate of Wyoming big sagebrush canopy recovery ($P = 0.064$). However, neither annual precipitation ($P = 0.827$) nor the heat index load ($P = 0.54$) alone were significantly related to canopy recovery rate.

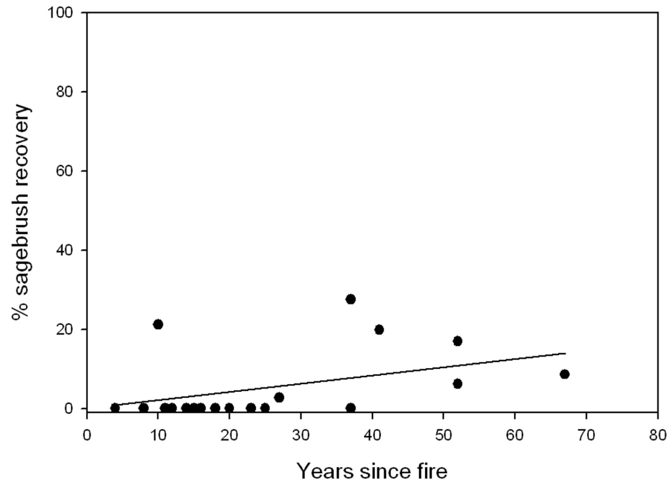

Figure 2—Linear regression model depicting Wyoming big sagebrush percent canopy recovery and time since fire for 24 sites (both wildfire and prescribed fire); regression model constrained to pass through the origin.

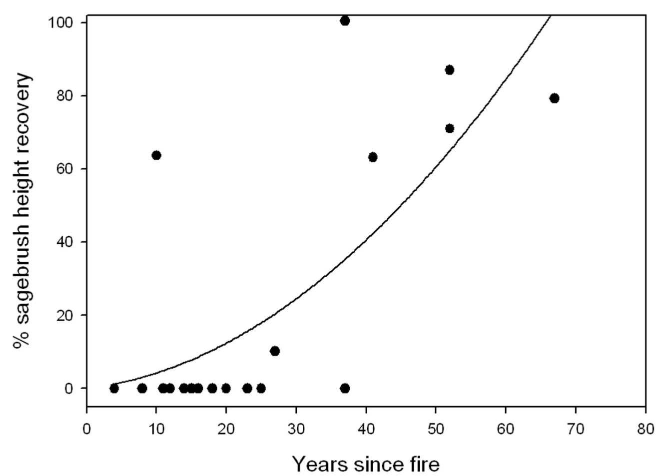

Figure 3—Exponential model depicting canopy height of Wyoming big sagebrush dominant cohort since fire for 24 sites; regression model constrained to pass through the origin.

Herbaceous Recovery

The important perennial graminoids in order of declining constancy were western wheatgrass, Sandberg bluegrass, blue grama, prairie junegrass (*Koeleria macrantha*), green needlegrass, needle-and-thread, bluebunch wheatgrass and sun sedge (*Carex inops* ssp. *heliophila*). Driest sites had relatively low canopy cover of bluebunch wheatgrass (mean = 28 percent cover), while undergrowth on the most mesic sites was dominated by Idaho fescue and western wheatgrass (mean = 74 percent cover). Grazing history probably also accounts for some of the variation in understory vegetation. Mean perennial grass canopy cover on control macroplots was 40 percent, of which approximately half was western wheatgrass. Burned macroplots had an average of 61 percent perennial grass cover, 39 percent of that cover is western wheatgrass. The differences in perennial grass cover between control and burn macroplots was highly significant (*N* = 24, Wilcoxon's ***T*** = 40, *P* = <0.01). Western wheatgrass had a mean of 39 percent and 22 percent cover in burned and control plots respectively (*N* = 22, Wilcoxon's ***T*** = 19, *P* = < 0.01). The other important vegetatively reproducing (sod-forming) graminoid, blue grama, exhibited 10 percent average canopy cover on burn plots and only 4 percent on controls, but due to highly variable cover this difference was not significant (*N* = 21,Wilcoxon's ***T*** = 73, *P* = >0.05).

Annual grasses had an average canopy cover of 19 percent and 9 percent in burned and control macroplots respectively (*N* = 24, Wilcoxon's ***T*** = 58, *P* = <0.01). Common species included the introduced brome grasses field brome (*Bromus arvensis*, formerly Japanese brome, *B. japonicus*) and cheatgrass (*Bromus tectorum*). However, the native sixweeks fescue (*Vulpia octoflora* = *Festuca octoflora*) also had appreciable constancy, although its cover was negligible.

There was no statistical difference in average canopy cover of forbs between burned (8.3 percent) and control (8.0 percent) macroplots (*N* = 24, Wilcoxon's ***T*** = 89, *P* = >0.05). Though forb canopy cover was consistently less than graminoid cover, it did range as high as 27 percent due to an unusual post-fire increase in the non-native corn speedwll (*Veronica arvensis*) on one productive high-elevation site. The most common forbs were the non-natives pale madwort (*Alyssum alyssoides*), field cottonrose (*Logfia arvensis*, formerly *Filago arvensis*), herb Sophia (*Descurainia sophia*), littlepod false flax (*Camelina microcarpa*), yellow salsify (*Tragopogon dubius*), common dandelion (*Taraxacum officinale*), and the natives wooly plantain (*Plantago patagonica*), tiny trumpet (*Collomia linearis*), rough false pennyroyal (*Hedeoma hispida*), spiny phlox (*Phlox hoodii*), American vetch (*Vicia americana*) and scarlet globemallow (*Sphaeralcea coccinea*).

The most commonly occurring forbs of the *Cichorieae* tribe were the non-natives common dandelion and yellow salsify. The natives, prairie-dandelion (*Nothocalais troximoides*) and pale agoseris (*Agoseris glauca*), were found in only one and two plot pairs, respectively. Due to their extremely low densities native species were lumped with non-natives for analysis. The mean density of members of the *Cichorieae* tribe was 1.8 ±2.8 plants / m^2 for the burned macroplots and 1.3 ± 1.8 plants / m^2 for the unburned controls; this difference was not significant (*N* = 23, Wilcoxon's ***T*** = 132, *P* = >>0.05).

Species Richness

Species richness, the average number of species per macroplot, was 32 ± 5 for control plots and 26 ± 6 for burned plots and significantly different between treatments (*N* = 24, Wilcoxon's ***T*** = 26, *P* = <<0.01). Extreme values ranged from 19 to 44 and 12 to 40 for control and burned plots, respectively. An insignificant 0.7 percent (R^2 = 0.007) of the difference in species richness between burned and control macroplots was explained with time since burning (*t* = 0.383, *P* = 0.71).

DISCUSSION

Sagebrush and Shrubs

Observation of both recently burned stands and those of considerable post-burn age (> 20 years) indicate that Wyoming big sagebrush mortality was virtually complete. There was no measured canopy recovery for Wyoming big sagebrush in 17 of the 24 sites. Our linear model of canopy recovery based on 24 sample pairs indicates an exceedingly slow recovery rate. The highest sagebrush canopy recovery rate in our study, 0.72 percent / year still implies (assuming a linear model) full recovery would require about 138 years. An exponential function with recovery rate increasing with time is probably more realistic because it accounts for increasing propagule pressure Lesica and others 2007). Even so, the oldest burn, 67 years, was only 8 percent recovered and recovery on the most moisture-stressed sites as well as sites with the greatest precipitation and most mesophytic vegetation composition registered no recovery within 14 years. Even on an older (27 years), and ostensibly cooler prescribed burn, recovery was only 3 percent. The only site (Little Bighorn Battlefield National Monument) without domestic stock use within the recovery period (and for more than five decades prior to burning) recorded no shrub canopy recovery after 25 years.

Other studies have also found that Wyoming big sagebrush recovers slowly following fire. In the only other comparable study within our sampling area, Eichhorn and Watts (1984) found no re-establishment of Wyoming big sagebrush in 14 years following wildfires in the Missouri River Breaks and vicinity. In southwest Montana, Lesica and others (2007) documented almost no Wyoming big sagebrush canopy recovery in six wildfire burn plots, the most being 3 percent in 23 years. In southeastern Idaho (8.5 inch precipitation zone) Colket (2003) found, measuring density not cover, that 3 of 17 plots attained full recovery in 53 years and that by 92 years 16 of the 17 plots reached full density. As Baker (2007) observes attaining full density is not equivalent to recovery of canopy cover, which undoubtedly would require additional decades for shrubs to mature. However, some researchers have reported shorter recovery times. Wambolt and others (2001) reported a 72 percent recovery of Wyoming big sagebrush after 32 years in one burn and 96 percent recovery after only nine years in another in southwest Montana, and Watt and Wambolt (1996) documented 76 percent recovery within 30 years in another southwestern Montana study. It should be noted these studies documented recovery following cool-season, prescribed fires. A state-and-transition model (Hemstrom and others 2002) derived for the interior Columbia Basin (ca. 8 inch annual precipitation) indicates that approximately 30 years are required to produce an open Wyoming big sagebrush / bunchgrass community starting with a post-burn bunchgrass-dominated habitat.

The average height of the dominant Wyoming big sagebrush cohort in control plots is 61 ± 11 cm, which agrees well with our southwestern Montana (Lesica and others 2007) measures of this subspecies (61 ± 6 cm). Only 4 of 24 burned macroplots even had a mature size/age class represented and the average height was 50 ± 6 cm; on one burned macroplot several individuals had attained full height recovery in approximately 38 years. Removing the zero values for height recovery from figure 3 would obviously shorten the time expected for full recovery and would more closely model what would be expected in the rate of height growth of individual plants once established on a site. However, for the model to be realistic on a stand basis the zero values should be included.

A nearby seed source is generally regarded as promoting faster stand recovery (Blaisdell 1953; Gruell 1980) because the seed bank of *A. tridentata* is negligible (Akinsoji 1988; Meyer 1992; Young and Evans 1989). Even though we located burn sample plots downwind from and as close as practicable to control plots, we still observed slow recovery rates. Wambolt and Payne (1986) found that close proximity to seed source still resulted in no Wyoming big sagebrush re-establishment six years post-burn.

We hypothesized that stands on areas of higher precipitation and/or with a lower heat load index would have a higher rate of recovery, similar to results from Johnson and Payne (1968). However, we were unable to detect any biotic or abiotic variables associated with Wyoming big sagebrush recovery across our study area. A model with age since fire, heat load index and precipitation explained 30 percent of the canopy recovery variation however, almost all of this explained variation was attributable to using age as a covariate.

More than 80 percent of burned macroplots lacked any representation of a seedling class. Seedling production was virtually nil, even in 2007, a year with abundant spring moisture that should have favored at least seedling germination, if not survival. We questioned this lack of seedlings as perhaps anomalous or a consequence of inadequate sampling. Therefore, in addition to visual scrutiny of the 10 microplots, we conducted extensive searches of adjacent terrain and uniformly failed to detect seedlings there as well. We encountered one instance of multiple seedling germination within a microplot (5 seedlings) out of 480 microplots examined. In general, the control macroplots had all maturity classes represented, however more than 50 percent of the stands did not have a seedling class present. The considerable difficulty Wyoming big sagebrush exhibits in site recolonization might be expected given that it occupies the driest sites with the most poorly developed soils (Morris and others 1976; Barker and McKell 1983).

With the exception of the mostly missing seedling class, nearly all the unburned control plots were uneven-aged (had multiple size/maturity classes represented), indicating recruitment generally occurs in pulses. Three control plots were somewhat anomalous in that only an adult class was present. Two of these three plots had approximate stem ages indicating that there had been no recruitment in more than 25 and 40 years. Although the adult class of these two stands was not even-aged, their age structure suggests episodic reproduction at some point in time.

Graminoids

The large (21 percent) increase in perennial grass cover following fire showed no consistent diminution with time since fire, probably because the Wyoming big sagebrush canopy failed to make significant recovery even after more than 60 years. Possibly even greater increases in perennial grass cover may have been negated by post-fire livestock grazing when the grass becomes more accessible after shrub canopy elimination (Pechanec and others 1954; Harniss and Murray 1973; Bunting and others 1998). In our unburned control plots perennial grass cover, an index of long-term grazing intensity, was not associated with proportional changes in grass cover following fire. This implies that post-fire grazing has not had a large impact on fire-induced changes. The major contributor to the significant increase in post-fire perennial grass cover was the rhizomatous western wheatgrass. None of the other perennial grass species showed a consistent significant fire response although several species showed strong responses in one or two plots.

Average annual grass canopy cover for both burned (19 percent) and control (9 percent) macroplots (N = 23, P = 0.010) was comprised almost entirely of the non-native field brome, which exhibited no significant diminution of cover with time since fire. Its cover ranged from zero to 69 percent in burned plots. Field brome is usually regarded as a weed on rangelands because it competes with native perennials for water and nutrients (Stubbendieck and others 1985; Gartner and others 1976). Fire is noted (Gartner and others 1986; Whisenant 1990) to reduce field brome population density for one or two years post-burn primarily as a consequence of litter reduction (critical for seed germination and establishment). We found no research that followed this species post-burn course of succession for more than two years. We hypothesize that the observed field brome response was due to exploitation of space, water and nutrients following sagebrush mortality.

Forbs

Our results suggest that forbs are generally well-adapted to these fire-prone communities because there was no difference in cover between burn (8 percent) and control (8 percent) macroplots (paired-sample t = .132, P = 0.896). However, some plots did exhibit dramatic increases or decreases in forb cover, usually due to the unique responses of one or two species. For example, on both the youngest (4 years) and oldest (67 years) burns lesser spikemoss (*Selaginella densa*) had 0 to 2 percent cover in burn plots compared to 41 percent and 22 percent cover, respectively, on the control macroplots. On two plot pairs a positive response to burning was displayed by the annual non-natives field cottonrose (<1 percent to 24 percent) and corn speedwell (<1 percent to 23 percent). The rather stochastic nature of these responses is emphasized by the fact that field cottonrose cover was minor (1.5 percent) in the burned macroplot where corn speedwell cover was so high. It is noteworthy that these large differences in forb cover are due to annuals, not to native perennials, which register hardly any change. Similar results have been reported for prescribed burns in sagebrush steppe by Peek and others (1979), who found forb frequency was not affected three years post-burn, and also by the Harniss and Murray (1973) report of stable forb cover for 30 years following fire in eastern Idaho. Wildfire did not produce any change in canopy cover of forbs in south-central or southwest Montana (Hoffman 1996; Fraas and others 1992).

Forbs of the *Cichorieae* Tribe of the *Asteraceae* Family have been determined to comprise an important component of Greater Sage-Grouse summer diet and are often crucial for successful brood rearing (Klebenow and Gray 1967; Peterson 1970; Barnett and Crawford 1944; Drut and others 1994). An increase in forbs can be expected with the fire-induced reduction in the cover of shrubs and grasses (Klebenow 1973; Glenn-Lewin and others 1990). We combined the relatively rare occurrences (<40 plants / 1,440 m^2) of native *Cichorieae*, prairie-dandelion and pale agoseris, with the much more abundant non-native *Cichorieae* densities, but found no evidence for a fire-driven change. Since non-native *Cichorieae* are invasive and increase with disturbance (Hobbs and Huenneke 1992; Kotanen and others 1998) their lack of response was unexpected. Lesica and others (2005) also found no change with fire for non-native Cichorieae in southwest Montana.

MANAGEMENT IMPLICATIONS

Most available evidence indicates that Wyoming big sagebrush canopy recovery rate is slow. Our data from central and southeastern Montana suggest that recovery (attaining the canopy cover exhibited by control plots) will require much more than 100 years. No cases of "rapid" Wyoming big sagebrush recovery were observed within the study area. This observation is corroborated by limited data from southwestern Montana (Lesica and others 2007). The time to fully recover Wyoming big sagebrush on an extensive burn could be very considerable. Our results support the contention that, although fire is an important natural disturbance in sagebrush steppe, it could not have occurred as often as suggested in the past (see Baker 2007 for a review). Our results accord with Baker's (2007) interpretation indicating that fire rotations are about 100 to 240 years for Wyoming big sagebrush and that sagebrush steppe belongs to fire regime V (long rotation, stand replacement). We found no evidence that soil texture, precipitation, slope, or were associated with the rate of canopy recovery in our study. Thus, managers cannot presume that stands of Wyoming big sagebrush on more mesic sites will necessarily exhibit faster recovery.

Our results are pertinent for managing domestic stock within the study area sagebrush steppe. We observed an average increase in perennial grass cover of 27 percent and 20 percent followed prescribed fire and wildfire, respectively. We have no evidence that this amplified cover will be diminished until sagebrush canopy cover becomes substantial at some future time, possibly at least a century after burning. Domestic livestock will benefit from this increased perennial grass cover and from increased accessibility to the herbaceous component due to shrub canopy removal. However, the 11 percent increase in annual grasses is due almost wholly to field brome which is considered by some a noxious weed (Stubbendieck and others 1985) because it competes with native perennials and has a brief window of grazing availability as it rapidly matures and loses nutrient content, digestibility and palatability (Stubbendieck and others 1985). Although various studies (see Stubbendieck and others 1985) indicate it declines with time on a site we have no indication this is the case. Burning sagebrush stands infested with field brome may result in a long-term increase in this undesirable species.

These results also have implications for protecting sagebrush-dependent wildlife. Land managers should be aware of Wyoming big sagebrush post-fire response. Greater Sage-grouse are dependent on some mixture of open- and closed-canopy sagebrush habitats to complete their life cycle (Connelly and others 2000). Wyoming big sagebrush recovery takes so long that managers considering prescriptive burns need to have a long-term view of the landscape before eliminating a sagebrush habitat that will probably not be sufficiently dense to constitute appropriate habitat for at least 50 years and perhaps much longer. Even greater concern should attend wildfire management in sagebrush habitats, if for no other reason than the resulting burns usually are more extensive (than prescribed burns) and hence result in yet greater recovery times. Greater

Sage-Grouse should be benefited by increased perennial grass cover resulting from fire (Wallestad and Pyrah 1974; Aldridge and Brigham 2002). Although success of Greater Sage-grouse brood rearing is dependent on available forbs, especially those of the Cichorieae, both native and exotic (Connelly and others 2000), we found no predictable increase in forb cover, including those of the Cichorieae, with fire.

Meyer's (1992) thorough account of the ecology of sagebrush recruitment notes aerial seeding (fall or winter), should be considered a potentially cost-effective approach for sagebrush establishment; however, this approach is subject to stochastic effects (and failure) beyond the manager's control as documented by Blew and others (2004). On less extensive burns nucleating with snow harvesting sites (creating favorable microsites) shows potential for enhancing emergence and recruitment (Meyer 1992).

CONCLUSION

Wyoming big sagebrush recovery from both prescribed fire and wildfire was extremely slow within the study area, likely requiring well over 100 years to reach pre-burn sagebrush cover. Results were similar across all environmental conditions, even on relatively mesic sites. Perennial and annual grass cover increased after burning, but the annual grass increase consisted almost entirely of field (Japanese) brome, a non-native that is considered a weed with negative habitat and livestock value. Forbs, most especially those of the Cichorieae tribe of the Asteraceae family, are important for Greater Sage-grouse brood rearing; however, we found no predictable change of this component with fire. Plant species richness was lower in burned plots. Resource managers concerned about Greater Sage-grouse and other sage-dependent species should carefully consider the long-term ramifications of any form of burning on Wyoming big sagebrush habitat in eastern Montana.

ACKNOWLEDGEMENTS

The following were instrumental in locating sampling sites, Louise deMontigny and Eric Lepisto (Bureau of Land Management, Miles City Field Office), Jeff Dibendetto (ecologist, Custer National Forest, Billings), Michael Stops (Chief Ranger, Little Bighorn Battlefield National Monument), Tom Stivers (wildlife biologist, Montana Fish Wildlife and Parks, Lewistown) and Bill Berg, Bob Skinner and Joann Dullum (Charles M. Russell National Wildlife Refuge, Lewistown). We extend our sincere appreciation to field assistants Lisa Wilson and David Salazar. Larry Eichhorn (retired BLM range conservationist, Lewistown) was generous with his time, helping us relocate a number of his original sagebrush succession study sites and dispensing informed commentary on our study. Soils analyses were performed at the facilities of Oregon State University in Corvallis, OR. We thank the Bureau of Land Management for financial support of this project.

REFERENCES

Akinsoji, A. 1988. Postfire vegetation dynamics in a sagebrush steppe in southeastern Idaho, USA. Vegetatio 78: 151-155.

Aldridge, C.L., Brigham, R.M. 2002. Sage grouse nesting and brood habitat use in southern Canada. Journal of Wildlife Management. 66: 433-444.

Antos, J.A., McCune, B., Bara, C. 1983. The effect of fire on an ungrazed western Montana grassland. American Midland Naturalist. 110: 354-364.

Bailey, R.G. 1995. Descriptions of the ecoregions of the United States. Second Edition Revised and Expanded. USDA, Forest Service. Misc. Publ. 1391. 108 p.

Baker, W.L. 2007. Fire and restoration of sagebrush ecosystems. Wildlife Society Bulletin. 34(1): 177-185.

Barker, J.R., McKell, C.M. 1983. Habitat differences between basin and Wyoming big sagebrush in contiguous populations. Journal of Range Management. 36: 450-454.

Barnett, J.K., Crawford, J.A. 1994. Pre-laying nutrition of sage grouse hens in Oregon. Journal of Range Management. 47:114-118.

Blaisdell, J.P. 1953. Ecological effects of planned burning of sagebrush-grass range on the upper Snake River Plains. Washington, DC. USDA Technical Bulletin 1075. 39 p.

Blew, R., Forman, A., Hafla, J., Pellant, M., Jones, R., White, G., Sands, A., Klar, T. 2004. Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) recovery following fire and artificial seeding. 2 p. Available: http://abstracts.co.allenpress.com/pweb/esa2004/document/?ID=37567 [December 28, 2007].

Bunting S.C., Robberecht, R., Defossé, G.E. 1998. Length and timing of grazing on postburn productivity of two bunchgrasses in an Idaho experimental range. International Journal of Wildland Fire. 8: 15-20.

Clifton, N.A. 1981. Response to prescribed fire in a Wyoming big sagebrush / bluebunch wheatgrass habitat type. Moscow: University of Idaho; 39 p. MS Thesis.

Colket, E.C. 2003. Long-term vegetation dynamics and post-fire establishment patterns of sagebrush steppe. Moscow: University of Idaho; 144 p. MS Thesis.

Connelly, J.W., Schroeder, M.A., Sands, A.R.; Braun, C.E. 2000. Guidelines to manage sage grouse populations and habitat. Wildlife Society Bulletin. 28: 967-985.

Daubenmire, R. 1959. A canopy-coverage method of vegetational analysis. Northwest Science. 33: 43-64.

Drut, M.S., Pyle, W.H., Crawford, J.A. 1994. Diets and food selection of sage grouse chicks in Oregon. Journal of Range Management. 47: 90-93.

Eichhorn, L.C., Watts, C.R. 1984. Plant succession on burns in the river breaks of central Montana. Proceedings of the Montana Academy of Science. 43: 21-34.

Eng, R.L., Schladweiller, P. 1972. Sage grouse winter movements and habitat use in central Montana. Journal of Wildlife Management. 36: 141-146.

Ferguson, C.W. 1964. Annual rings in big sagebrush. Papers of the Laboratory of Tree-Ring Research No. 1. Tucson: University of Arizona Press. 95 p.

Fraas, W.W., Wambolt, C.L., Frisina, M.R 1992. Prescribed fire effects on a bitterbrush-mountain big sagebrush-bluebunch wheatgrass community. Proceedings of a symposium; 1992: Ogden,UT. Gen. Tech. Rep. INT-289. Ogden, UT: USDA, Forest Service, Intermountain Forest Range Experiment Station. 232 p.

Gartner, F.R., White, E.M., Butterfield, R.I. 1986. Mechanical treatment and burning for high quality range forage. Beef Report: Cattle 86-29. Brookings, SD: South Dakota State University, Department of Animal and Range Sciences and Plant Science; Agriculture Experiment Station. 140 p.

Gartner, F.R., Roath, L.R., White, E.M. 1976. Advantages and disadvantages of prescribed burning. Proceedings of a symposium; 1976; Logan, UT. Use of prescribed burning in western woodland and range ecosystems: Utah State University: 11-15 p.

Glenn-Lewin, D.C., Johnson, L.A., Jurik, T.W., Kosek, A., Leoscheke, M., Rosburg, T. 1990. Fire in central North American grasslands: vegetative reproduction, seed germination and seedling establishment. *In* S.L. Collins and L.L. Wallace editors, Fire in central North American grasslands. Norman: University of Oklahoma Press. 28-45 p.
Gruell, G.E. 1980. Fire's influence on wildlife habitat on the Bridger-Teton National Forest, Wyoming, Volume II- changes and causes, management implications. Ogden, UT: Res. Pap. INT-252. USDA, Forest Service, Intermountain Forest and Range Experiment Station. 35 p.
Harniss, R.O., Murray, R.B. 1973. 30 years of vegetal change following burning of sagebrush-grass range. Journal of Range Management. 26: 322-325.
Hemstrom, M.A., Wisdom, M.J., Hann, W.J., Rowland, M.M., Wales, B.C., Gravenmier, R.A. 2002. Sagebrush-steppe vegetation dynamics and restoration potential in the Interior Columbia Basin. Conservation Biology. 16: 1243-1255.
Hobbs, R.J., Huenneke, L.F. 1992. Disturbance, diversity and invasion: implications for conservation. Conservation Biology. 6: 324-337.
Hoffman, T.L. 1996. An ecological investigation of mountain big sagebrush in the Gardiner Basin. Bozeman: Montana State University; 84 p. M S Thesis.
Humphrey, L.D. 1984. Patterns and mechanisms of plant succession after fire on *Artemisia*-grass sites in southeastern Idaho. Vegetatio. 57: 91-101.
Johnson, J.R., Payne, G.F. 1968. Sagebrush reinvasion as affected by some environmental influences. Journal of Range Management. 21: 209-213.
Klebenow, D.A. 1969. Sage grouse nesting and brood habitat in Idaho. Journal of Wildlife Management. 33: 649-661.
Klebenow, D.A. 1973. The habitat requirements of sage grouse and the role of fire in management. Proceedings of the Tall Timbers Fire Ecology Conference. 12: 305-315.
Klebenow, D.A., Gray, G.M. 1967. Food habits of juvenile sage grouse. Journal of Range Management. 21: 80-83.
Kotanen, P.M., Bergelson, J., Hazlett, D.L. 1998. Habitats of native and exotic plants in Colorado shortgrass steppe: a comparative approach. Canadian Journal of Botany. 76: 664-672.
Lesica, P., Cooper, S.V., Kudray, G. 2005. Big sagebrush shrub-steppe postfire succession in southwest Montana. Helena, MT; Montana Natural Heritage Program. Report to USDI, Bureau of Land Management, Dillon Field Office. 29 p. plus appendices.
Lesica, P., Cooper, S.V., Kudray, G. 2007. Recovery of big sagebrush following fire in southwest Montana. Rangeland Ecology & Management. 60: 261-269.
McCune, B., Keon, D. 2002. Equations for potential annual direct incident radiation and heat load. Journal of Vegetation Science. 13: 603-606.
Meyer, S.E. 1992. Germination and establishment ecology of big sagebrush: Implications for community restoration. Proceedings of a symposium; 1992 May18-21; Boise, ID. Gen. Tech. Rep. INT-GTR-313. Ogden, UT: Forest Service, Intermountain Research Station. 416 p.
Morris, M.S., Kelsey, R.G., Griggs, D. 1976. The geographic and ecological distribution of big sagebrush and other woody *Artemisias* in Montana. Proceedings of the Montana Academy of Sciences. 36: 56-79.
Mueller-Dombois, D., Ellenberg, H. 1974. Aims and methods of vegetation ecology. New York: John Wiley & Sons. 547 p.
Nesser, J.A., Ford, G.L., Maynard, C.L., Page-Dumroese, D.S. 1997. Ecological units of the Northern Region: subsections. General Technical Report INT-GTR-369. Ogden, UT: USDA, Forest Service, Intermountain Research Station. 88 p.
Pechanec, J., Stewart, F.G., Blaisdell, J.P. 1944. Sagebrush burning-good and bad. USDA Farmer's Bulletin 148. 34 p.
Peek, J.M., Riggs, R.A., Lauer, J.L. 1979. Evaluation of fall burning on bighorn sheep winter range. Journal of Range Management. 32: 430-432.
Peterson, J.G. 1970. The food habits and summer distribution of juvenile sage grouse in central Montana. Journal of Wildlife Management. 34: 147-155.
Pyle, W.H., Crawford, J.A. 1996. Availability of foods for sage grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management. 49: 320-324.
Roscoe, J.W. 2002. Sage grouse movements in southwestern Montana. Intermountain Journal of Science. 8: 94-104.
Stubbendieck, J., Nichols, J.T., Roberts, K.K. 1985. Nebraska range and pasture grasses (including grass-like plants). E.C. 85-170. Lincoln, NE: University of Nebraska, Department of Agriculture, Cooperative Extension Service. 75 p.
Thornton, P.E., Running, S.W., White, M.A. 1997. Generating surfaces of daily meteorological variables over large regions of complex terrain. Journal of Hydrology. 190: 214-251.
USDA Natural Resource Conservation Service. 2008. Plants Database [http://plants.usda.gove/index.html.]
Walhof, K.S. 1997. A comparison of burned and unburned big sagebrush communities in southwest Montana. Bozeman: Montana State University; 74 p. MS Thesis.
Wallestad, R. 1971. Summer movements and habitat use by sage grouse broods in central Montana. Journal of Wildlife Management. 35:129-135.
Wallestad, R., Pyrah D. 1974. Movement and nesting of sage grouse hens in central Montana. Journal of Wildlife Management. 38: 630-637.
Wambolt, C.L., Payne, G.F. 1986. An 18-year comparison of control methods for Wyoming big sagebrush in southwestern Montana. Journal of Range Management. 39: 314-319.
Wambolt, C.L., Walhof, K.S., M.R. Frisina. 2001. Recovery of big sagebrush communities after burning in south-western Montana. Journal of Environmental Management. 61: 243-252.
Watts, M.J., Wambolt, C.L. 1996. Long-term recovery of Wyoming big sagebrush after four treatments. Journal of Environmental Management. 46: 95-102.
Welch, B.L., Criddle, C. 2003. Countering misinformation concerning big sagebrush. Research Paper RMRS-RP-40. Ogden, UT: USDA, Forest Service, Rocky Mountain Research Station. 28 p.
West, N.E. 2000. Synecology and disturbance regimes of sagebrush steppe ecosystems. Pages 15-26 *in* Entwistle, P.G., DeBolt, A.M., Kaltenecker, J.H., Steenhof, K. compilers. Proceedings of a symposium; 1999 June23-25; Boise, ID. Publication No. BLM/ID/PT-001001+1150. 145 p.
West, N.E., Hassan, M.A. 1985. Recovery of sagebrush-grass vegetation following wildfire. Journal of Range Management. 38: 131-134.
Whisenant, S.G. 1990. Postfire population dynamics of *Bromus japonicus*. American Midland Naturalist. 123: 301-308.
Winward, A.H. 1991. A renewed commitment to management of sagebrush grasslands. In Research in rangeland management. Corvallis, OR: Oregon State University Agricultural Experiment Station Special Report 880. 7 p.
Winward, A.H. 2004. Sagebrush of Colorado: Taxonomy, distribution, ecology and management. Denver, CO: Colorado Division of Wildlife, Department of Natural Resources. 45 p.
Wright, H.A., Neuenschwander, L.F., Britton, C.M. 1979. The role and use of fire in sagebrush-grass and pinyon-juniper plant communities: A state-of-the-art review. Gen. Tech. Rep. INT-58. Ogden, UT: USDA, Forest Service, Intermountain Forest and Range Experiment Station. 48 p.
Young, J.A., Evans, R.A. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. Weed Science. 37: 201-206.