AN ABSTRACT OF THE THESIS OF

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Title: Ecological Effects of Spring and Fall Prescribed Burning on Basin Big Sagebrush/Idaho Fescue--Bluebunch Wheatgrass Communities.

Abstract approved: J. Boone Kauffman

The vegetation response of spring and fall prescribed fires in basin big sagebrush (*Artemisia tridentata* subsp. *tridentata* Nutt.)/Idaho fescue (*Festuca idahoensis* Elmer)--bluebunch wheatgrass (*Agropyron spicatum* Pursh. (Scribn. & Smith)) communities was measured at the John Day Fossil Beds National Monument in eastern Oregon. Objectives of the study were to quantify fuel loads, environmental conditions, fire behavior and vegetation response corresponding to these two fire treatments.

Pretreatment fuel loads in the experimental units ranged from 5 to 12 Mg/ha, with the fall treatment units averaging 10.5 Mg/ha, and the spring units averaging 6.2 Mg/ha. Both treatments contained large amounts (> 3 Mg/ha) of herbaceous fuels. Moisture content of grass and herb fuels were significantly greater in the spring burned units. This is believed to be largely responsible for the less severe fire behavior observed in the spring burn treatment. Flame lengths averaged 4.2 m in fall burns, compared to a mean of 1.7 m in the spring plots. Similarly, rate of spread was significantly greater in the fall units, averaging 1.6 m/s, compared to 0.2 m/s in the spring treatment. Fireline Intensity was seven times greater, and total energy release was twice was great in the fall burns.

Neither burn treatment resulted in significant mortality of

bluebunch wheatgrass, but fall burning did cause significant mortality of Idaho fescue, where 20% of the population was killed. Fall burning stimulated tillering of bluebunch wheatgrass, as the average basal area increased both one and two years following burning. Average basal area per plant of Idaho fescue was reduced by 23% the first year following fall burning; however plants recovered to 90% of their pre-burn size by the second post-fire year. Spring burning resulted in no significant change in basal area of either species. Fall burning significantly reduced the number of flowering culms on bluebunch wheatgrass plants the first post-fire year (from 36 to 12 per plant); however, by the second post-fire year, number of flowering culms was significantly greater than either pre-burn or control levels (59/plant). Similarly, fall burning of Idaho fescue averaged 60% more flowering culms per plant as adjacent controls (11 compared to 7/plant). Spring burning reduced flowering of both species the first year following burning.

Both burn treatments reduced the frequency of annual grasses, while causing no change in frequency of perennial grasses. Annual forbs increased in abundance following both burn treatments. Fire resulted in replacement of exotic annual grasses with annual forb species. Dominant perennial forbs responded variably in both burn treatments, as well as control plots. Frequency of sagebrush increased significantly in both spring and control experimental units in 1989 (one year after spring burning), while fall burns (two years post-treatment) demonstrated no such increase. Apparently, factors relating to the greater fire severity (e.g. consumption, total energy) in the fall burns reduced the rate and degree of reinvasion by sagebrush in the fall burn plots.

Densities of annual grasses and woody species were significantly reduced by both burn treatments. Cheatgrass (*Bromus tectorum* L.) density before burning averaged 446 and 552/m^2 in fall and spring units, respectively, as compared to 10 and 85/m^2, respectively, the first post-fire year. Big sagebrush was completely eliminated by the fall fire, while spring burning resulted in an 84% decrease in density. Density of western juniper (*Juniperus occidentalis* Hook.) was reduced 100% by both burn treatments.

Species diversity, as measured by the Shannon-Weaver Index (H'), was reduced by fall burning from 2.69 before treatment to 2.53 the first year following burning , but increased to 2.81 second post-fire year. Control plots behaved similarly, although changes were not as great. Changes were most evident in terms of rare species, many of which were not present prior to burning. Spring burning resulted in an no change in species diversity the first year after burning, although, species richness increased from 34 to 41.

Both burn treatments appeared effective at changing stand structure to that of a dominance by native perrenial grasses and forbs. The reduced competition from woody plants has, and presumably will continue to favor surviving herbaceous plants. Overall fire effects appear to fit into land management policy of the National Park Service in regard to maintaining wildlands in a pristine state. Specifically, both spring and fall burning reduced fuel hazard, and increased the relative abundance of native species, indicating that prescribed burning may be an effective land management tool for the National Park Service and others managing similar rangelands.

Ecological Effects of Spring and Fall Prescribed Burning on Basin Big Sagebrush/Idaho Fescue--Bluebunch Wheatgrass Communities

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CHAPTER 1

A Review of the Literature On Prescribed Burning and Fire Ecology in Sagebrush-grass Ecosystems

INTRODUCTION

Barney (1975) defined fire management as the integration of fire-related biological, ecological, physical, and technological information into land management to meet desired objectives. The purpose of this chapter is to review the information currently available concerning the implementation, analysis, and overall success of using prescribed burning as an ecological and managerial tool in similar rangelands.

PRESCRIBED BURNING

Guidelines for burn prescriptions and techniques for sagebrush-dominated ecosystems can be found in Bunting et al. (1987), Clark et al. (1985), Martin et al. (1977), Martin and Dell (1978), Wright et al. (1979), and Wright and Bailey (1982). The environmental parameters associated with prescribed burning are summarized in Schroeder and Buck (1970).

Fire behavior

Numerous publications address the proper documentation, description, and evaluation of prescribed fire (e.g. Albini 1976, Alexander 1982, Martin 1984, Rothermel and Deeming 1981). However, more research is needed to determine the ecological relationships between environmental conditions, fuel characteristics, and fire behavior as they relate to effects on all plant communities. Historically, most studies did not quantitatively document the fire behavior, fuel consumption, and burn conditions (i.e. weather, fuel moisture) during the fire under investigation. As Rothermel and Deeming (1981) noted, the consequence of using non-standard, qualitative descriptors of a fire's behavior or effects severely limits the information because of the problem of correlating results from different studies. Martin (1984) stated that: "without proper documentation the observer is describing the effects of an unknown treatment".

Alexander (1982) reviewed the importance of fire behavior and the parameters commonly used to document fire behavior. Some useful include Fireline Intensity (I_{fl}), Reaction Intensity (I_r), Heat per unit Area (H_a), and total energy release (Rothermel and Deeming, 1981). Fireline Intensity is defined as the rate of heat release per unit length of fire front. Byram (1959) developed a relationship between fuel heat of combustion, fuel consumption, and rate of spread, such that I_{fl} can be estimated from the following equation:

$$I_{fl} = Hwr \qquad [Kw/m]$$

where: H is the fuel low heat of combustion (kJ/kg),

w is the weight of fuel consumed per unit area (kg/m^2),

r is the rate of spread (m/s).

Since net heat of combustion has been shown to vary by only 10% or so from fuel to fuel (Van Wagner 1972), it is generally thought of as a constant; a commonly used value is 18,700 kJ/kg (Albini 1976, Van Wagner 1973). A subtraction from the net heat of combustion to account for the latent heat of evaporation of 24 J per moisture percentage point gives the low heat of combustion value (Van Wagner 1972).

Byram (1959) also developed an empirical relationship between I_{fl} and the length of flames (FL), such that it can easily be estimated using the following formula:

$$I_{fl} = 258FL^{2.17} \qquad [Kw/m]$$

Reaction Intensity is the heat release per unit area within the flaming zone, and has been suggested to be a good measure for studies investigating surface or sub-surface fire effects (Rothermel and Deeming 1981). Reaction Intensity can be estimated if the depth of the flaming zone (D) is measured, from the following equation:

$$I_r = I/D \qquad [kW/m^2]$$

The total energy released in the active flame-front, or heat-per-unit area (H_a), can be calculated if the rate of spread (ROS) of the advancing fire-front is known by using the following:

$$H_a = I/ROS \qquad [KJ/m^2]$$

Additionally, the total energy released by both flaming and secondary combustion that occurs after the passage of the flame-front can be defined as the product of the total mass of fuel consumed times the energy content of that fuel.

Investigations of fire effects in sagebrush/grass ecosystems should include fire behavior/fuel measurements to allow for correlation and comparisons between other fires. These measurements include flame length, flame depth, rate of spread, type of fire (e.g. head, flank, or back), flame height, flame angle, and residence time (Martin 1984).

Temperature

Many of the fire-induced changes in the chemical, physical, and biological components of an ecosystem are a direct result of the degree and duration of heating (Rundel 1982). For example, temperature and duration of heating are believed to be significant factors in bunchgrass mortality (Britton et al. 1983, Champlin 1983, Conrad and Poulton 1966, Nimir and Payne 1978, Wright 1971, Wright and Klemmedson 1965), nutrient dynamics (Christensen and Muller 1975, DeBano and Conrad 1978), and seed mortality/stimulation (Champlin 1983, Christensen and Muller 1975, Patton et al. 1988).

Soil surface temperatures are highly variable, and affected by weather (Mason 1949), firing technique (Byram 1958), kind and amount of fuel (Conrad and Poulton 1966, Hopkins et al. 1948), and vertical distance of fuel from the soil surface (Byram 1958). Bentley and Fenner (1958) found soil surface temperatures to range from 90 - 180°C in grassland fires, whereas DeBano et al. (1977) found that chaparral fires may result in surface temperatures exceeding 700°C. Wright and Klemmedson (1965) noted in a wildfire in a south-central Idaho sagebrush community soil surface temperatures were greater than 400°F (205°C).

Daubenmire (1968) outlined two basic methods for measuring temperature during fire. One technique involves setting out a comparative series of heat-sensitive materials with different reaction points. Fenner and Bentley (1960) constructed pyronometers such as these by painting vertical strips of temperature-sensitive paint onto a thin sheet of mica that is backed by an asbestos fiber, or some other material with a low thermal conductivity. The sensor can then be placed in the soil provide both above and below ground temperature data. The other method for collecting temperature data involves using transducers such as thermistors or thermocouples that correlate temperature with resistance and voltage, respectively. The advantages of thermocouples include their capacity to measure a wide and extreme range of temperatures, with relatively small sensing points (Daubenmire 1968). These devices offer considerably more precision than the pyronometers, as well as the capacity to measure heat duration. However, they are costly, and present difficulty in logistics when conducting a prescribed burn.

Fuels

As fuels can be defined as any material that have the potential to undergo combustion, fuel loads in sagebrush dominated ecosystems are equivalent to total aboveground biomass. Fuel particle size and moisture content are considered to be important variables effecting combustion (Albini 1976). Consequently, fuels are often partitioned into classes based on rates of moisture loss or dry down (e.g. a timelag constant). The rate of moisture loss of organic materials tends to follow a logarithmic rate for which the time required to reach an equilibrium moisture content can be divided into periods (Pyne 1984). The timelag constant of any given fuel particle is the time for it to lose approximately 63% (1 - 1/e) of the difference between its initial moisture content and a new equilibrium moisture content under standard laboratory conditions of 27°C and 20% relative humidity (Byram 1963). In North America, moisture timelag classes used to partition downed woody material based on their diameter are: 1-hr (0-0.64 cm), 10-hr (0.64-2.54 cm), 100-hr (2.54-7.6 cm), and 1000-hr (>7.6 cm)

(Deeming et al. 1977).

Considerable attention has been paid to estimating fuel loads, both as an indicator of fire hazard, and as a means of estimating fuel consumption and fire behavior (Britton et al. 1981, Clark et al. 1985). As stated earlier, fuel consumption is a very important parameter in determining the amount of heat that is generated by combustion processes, and hence ecological effects on soil, vegetation, nutrients, and wildlife.

Although sampling of the biomass of fine fuels (i.e. the herbaceous component) presents little problem, accurate determination of shrub fuel loading by size class utilizing a non-destructive sampling technique can be difficult. Many researchers (see Brown 1982, Fransden 1983) have employed simple and multivariate regression analysis to estimate fuel loads, using such parameters as shrub cover, height, basal diameter, and crown volume.

Harniss and Murray (1976), Rittenhouse and Sneva (1977) and Uresk et al. (1977) measured elliptical crown area, circumference, height, and volume to estimate biomass of sagebrush plants. Rittenhouse and Sneva (1977) used log transformations of these measurements individually and in stepwise multiple regression to estimate photosynthetic and woody biomass of Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis* Nutt.). Coefficients of determination (R^2) values ranged from .72 to .97. However, these methods provide estimations only for total biomass and foliage.

Fransden (1983), used height and canopy area data from Uresk et al. (1977) and Rittenhouse and Sneva (1977) to develop predictive equations for fuel loads partitioned by size class for Wyoming and basin big sagebrush. Martin et al. (1981) similarly give a fuel estimation for mountain big sagebrush (*A. tridentata* ssp. *vasseyana* Nutt.) based on an average height and crown diameter.

Champlin (1983) developed single and step-wise multiple regression estimates of fuel by size class and category (i.e. live or dead), for all three subspecies of big sagebrush. Independent variables included: height, elliptical crown area, number of individuals intersecting the line transect, and percent cover. Estimates of basin big sagebrush gave R^2 values for all fuel size classes and categories ranged from .73 to .84. The required sampling is fairly rapid, simple, and is non-destructive. Consequently it can be employed *in situ* to estimate fuel loads in research units investigating fire effects. However, the fuel size classes used in this study did not completely agree with established categories in regard to standard timelag classes.

The nature of the variability between subspecies and sites necessitate caution when estimating sagebrush fuels. The best information available for similar sagebrush fuels come from recent work by Kauffman and Cummings (unpubl. data) on basin big sagebrush fuels near Bear Creek, Oregon. Equations to predict total fuel, total live fuel, foliage, and woody biomass by standard size-class, gave R^2 values ranging from .82 to .95.

In sagebrush ecosystems, threshold fuel loads for sustained ignition and spread of fire have been reported to range from 674 to 786 kg/ha of herbaceous fuel (Beardall and Sylvester 1976) or a minimum of 20 percent sagebrush canopy cover (Pechanec et al. 1954). Britton and Clark (1985) developed more advanced models that included weather factors, and found a minimum of 20% sagebrush canopy cover and 300kg/ha herbaceous fuel insuring successful prescribed burns under moderate weather conditions (20-27°C, 15-20% relative humidity, and windspeed between 14 and 24 km/hr). These requirements would undoubtedly change with increasing windspeeds, where the herbaceous fuel would no longer be necessary to carry fire from one shrub to another.

FIRE ECOLOGY

Numerous reviews regarding the use and effects of fire in grass and shrublands have been written (Daubenmire 1968, Heady 1975, Wright 1974, and Wright and Bailey 1982). Particular emphasis on the sagebrush-grass steppe can be found in Bunting et al. 1987, Tisdale et al. 1981, and Wright et al. (1979). When predicting fire effects, Daubenmire (1968), Harniss and Murray (1973) and Winward (1985) emphasized the need to distinguish big sagebrush communities at the subspecies level.

Fire history

Fire is a natural component in sagebrush/grass communities. Houston (1973) estimated the average fire frequency for northern Yellowstone National Park (Wyoming big sagebrush) communities were from 39 to 70 years. However, by dating all fires that occurred within a locale (i.e. area based estimation), Houston (1973) theorized that fire frequency within the park was actually 20 - 25 years. Basing their estimates on the life span of horsebrush (*Tetrademia canescens* D.C.), Harniss and Murray (1973) believed that probable frequency for these communities to be around 50 years. In contrast, Martin and Johnson (1979) found mean fire return interval to be only 5-15 years for pine/bitterbrush/sagebrush/grass communities at Lava Beds National Monument in northern California. Here the dominant subspecies was mountain big sagebrush. Although I am aware of no fire history work in basin big sagebrush communities, our understanding of the relative productivity of these areas would support the belief that fire frequencies would be intermediate between mountain and Wyoming big sagebrush communities. Winward (1985) notes that given the wide range of fuel situations, and our understanding of yearly climatic variation in the sagebrush ecosystem, a naturally wide variation in fire frequencies in this system should be expected. He asserted that as we increase our understanding of different plant communities and habitat types we will be able to develop better site-specific burn prescriptions.

FIRE EFFECTS

The effects of fire on vegetation can be divided into short-term mortality for some individual plants, and longer-term modification of the conditions for growth and reproduction for others (Young and Miller 1985). Morphology, phenology, and ecotypic variation have all been considered important factors in determining plant response to fire (Britton and Ralphs 1979, Britton et al. 1983, Wright 1971). Sampling methods that have proven useful in evaluating and monitoring fire effects in sagebrush-grass ranges are summarized in Bunting et al. (1987).

The mechanisms by which plants have adapted to survive recurrent fire are outlined by Gill (1977) and Kauffman (1990). The two most significant adaptations that are viewed to be significant in relation to post-fire succession appear to be vegetative regrowth, or sprouting from undamaged tissues, and the capacity to occupy newly opened sites with new reproduction from seed sources (Kauffman and Sapsis 1989).

Effects on woody species

All of the Great Basin subspecies of big sagebrush do not resprout, and are easily killed by fire (Blaisdell 1953, Pickford 1932). This late seral dominant may remain subordinant in the stand for 15 to 50 years. Blaisdell (1953) found that 12 years after fire, production of mountain big sagebrush was only 10% of that found on control plots. After 30 years, however, the same area had returned to near pre-burn levels of sagebrush cover (Harniss and Murray 1973). Mueggler (1956) found that the seed source for regeneration of big sagebrush largely came from seeds stored in the soil, and that windborne seed is restricted to areas fairly near an unburned seed source. Consequently, large, severe fires will probably reduce available seed, and thereby slow reestablishment of big sagebrush. Bunting (1985) believed that next to seed availability, the most important factor in sagebrush establishment is the pattern and amount of precipitation following a burn. Moist soil during the spring germination period coinciding with good seed crops from the previous year have been hypothesized to result in observed cases of high seedling densities (Bunting 1985). Daubenmire (1975) found that establishment of basin big sagebrush in Washington occurred in the first few years following fire.

The overall effect of fire on the shrub component is dependent on the fire severity, species present, site characteristics, and ecological condition (Bunting et al. 1987). Presence of rabbitbrush (*Crysothamnus* sp.) or horsebrush is often thought to be an obstacle to using fire to control shrubs because of their capacity to resprout. However, Winward (1985) reported that there are many subspecies of both green rabbitbrush (*C. viscida* (Hook.) Nutt.), and gray rabbitbrush (*C. nauseosis* (Pall.) Britt.), and they vary considerably in their capacity to resprout. He suggested that we need to identify individuals at the subspecies level and begin assembling a record of best management practices according to subspecies. Winward (1985) also suggested that in most situations in the Pacific Northwest, the rabbitbrush are early successional species and are relatively short lived, and probably have a less competitive role than previously ascribed to them.

Under traditional succession theory, fire retards ecological succession in communities that progress toward western juniper (*Juniperus occidentalis* Hook.) dominated climax (Kauffman and Sapsis 1989, Martin and Johnson, 1979). Consequently, fire acts as a significant instrument in limiting juniper abundance. Martin (1979) provided a model for predicting juniper survival by tree height according to firing pattern and weather conditions. For example, with a headfire conducted under fairly extreme conditions (10% relative humidity, 8-19 kph windspeed, 27°C) virtually all trees under 4.7 m in height, and 60% of those over 5 m would be killed. On sites where juniper invasion is evident and burning is feasible, prescribed fire is considered by many (e.g. Frischknecht 1979, Martin 1979) to be the most effective and economic eradication treatment.

Effects on grasses

The effect of fire on the grass component of sagebrush/grass communities is largely influenced by season of burn, size of plant, species, growth form, prior precipitation, and amount of dead material surrounding the individual plant (Wright 1985). Blaisdell (1953), Mueggler and Blaisdell (1958), Conrad and Poulton (1966), Harniss and Murray (1973), Uresk et al. (1976, 1980), Olson et al. (1982) and Champlin (1983) all found some increases in perennial grasses. However, species, parameters, and duration of increase varied greatly between studies. Daubenmire (1970) noted that herbaceous vegetation was not only more productive following fire, but as a result of shrub removal was also more accessible to large herbivores. Daubenmire (1970) also discussed some other ways in which sagebrush removal may affect the community. He postulated that many nutrients leached below the herbaceous root zone would no longer be taken up by the deeper rooted shrubs, and consequently not recycled to the surface.

Qualitative descriptions of damage to perennial bunchgrasses in relation to fire severity, or appearance of the burned vegetation, have been observed by Blaisdell (1953), Conrad and Poulton (1966), and Harniss and Murray (1973), which may account for the variability reported. Other investigations have burned individual plants using a various techniques and equipment (see Britton and Wright 1979).

The first work on burning individual bunchgrasses under controlled conditions was conducted by Wright and Klemmedson (1965), who studied seasonal effects of burning on two size classes of four species of native perennial bunchgrasses. A 55 gallon oil drum was converted into a burn chamber, and pre-weighed shredded paper was used as fuel designed to keep soil surface temperatures at 200 and 400°C, (no thermocouple was used to monitor temperature). Wright and

Klemmedson (1965) found that Sandberg's bluegrass (*Poa sandbergii* Vasey) was essentially unharmed, while squirreltail (*Sitanion hystrix* (Nutt.) Smith), needle-and-thread (*Stipa comata* Trin. & Rupr.) and Thurber's needlegrass (*S. thurburiana* Piper) all suffered some degree of damage one year after burning. Squirreltail suffered no mortality, but July burns reduced basal area of both size classes by 10 to 20%, while burning in August resulted in reductions for only the larger plants. In contrast, needlegrass mortality was greatest after June burning, with no August mortality. Virtually all needle-and-thread plants were killed by burning in June, while only 20% were killed in July, which was not significantly different from the control treatment. Basal area of those not killed by June or July burning was reduced by 79 to 99%, depending on the size of the plant. In general, the larger the individual, the greater the reduction in basal cover.

In a later study (Wright 1971), a propane torch was used instead of pre-weighed paper, and thermocouples were used to monitor the temperature heat treatment. Treatment variables included season of burning (each month for May through September), soil surface temperature (400 and 800°F), and two species of bunchgrasses (needle-and-thread and squirreltail). In general, needle-and-thread was considerably more damaged than squirreltail. Specifically, only the 800°F treatment in July and August caused mortality of squirreltail, whereas both temperature treatments killed needle-and-thread in all months except for 400°F in June. Wright (1971) suggested that the differences in culm morphology and relative accumulation of dead material in proximity to meristematic tissues strongly influence a given species' response to fire. Squirreltail has solid culms and relatively little accumulation of dead material around the crown, and consequently under similar burning conditions results in less heat being transferred to the bud zones than in the denser, fine-stemmed needlegrasses.

Conrad and Poulton (1966) noted the difference in bud zone depth may have been responsible for their observations of significantly greater mortality (27%) of Idaho fescue (*Festuca idahoensis* Elmer) over bluebunch wheatgrass (*Agropyron spicatum* (Pursh.) Scribn. & Smith) (1%) following a July wildfire in northeastern Oregon. They postulated that the dense crowns of Idaho fescue have budding areas at or above the ground surface, therefore are more exposed to heat flux. In contrast, bluebunch wheatgrass had very short rhizomes that produce buds below the ground surface and was therefore more resistant to fire-caused mortality. Britton and Ralphs (1979) hypothesized that coarse-stemmed bunchgrasses such as bluebunch wheatgrass suffer less mortality because the nature of the fuel tends to favor quick combustion. In contrast, fine stemmed bunchgrasses such as Idaho fescue have a greater density of combustible material within the clump, which results in longer duration of burning, and greater heat flux to the growing points (Britton and Ralphs 1979).

In mountain big sagebrush communities in central Oregon and northeastern California, Champlin (1983) found significant increases in basal area of bluebunch wheatgrass and Sandberg's bluegrass two years after burning. In contrast, decreases in basal area of Thurber's needlegrass and Idaho fescue were observed. Spring burning at Lava Beds National Monument resulted in considerably less reduction of Idaho fescue than a fall burn at Crooked River National Grassland. At the National Grassland a 90% reduction of basal area was observed, with little recovery after two years. In contrast, the fall burn stimulated productivity of bluebunch wheatgrass considerably over pre-burn levels throughout the study period. Although these variable findings between seasons of burning are not surprising, the ecological conditions under which these differences manifest themselves are poorly understood.

These findings support the contention of Wright (1971) that the effect of fire on bunchgrass mortality is largely a function of growth form and season of burning. Bunchgrasses with densely clustered culms such as needle-and-thread and Idaho fescue can be severely damaged by fire, especially if it occurs during active growth (June/July). Bluebunch wheatgrass and Squirreltail, on the other hand are generally little effected, regardless of season of burn. This is supported by evidence that bluebunch wheatgrass returns to pre-burn basal area levels within one to three years following fire (Blaisdell 1953, Conrad and Poulton 1966, Champlin 1983, Peek et al. 1979, Uresk et al. 1976).

Burning of individual Idaho fescue plants in eastern Oregon demonstrated that soil moisture had no effect on basal area or yield when burned in either late-August or mid-October (Britton et al. 1983). Although no mortality was recorded, plants burned in late-summer demonstrated significantly greater reductions in basal area and yield than compared to plants burned in the fall (Britton et al. 1983).

Increases in perennial grass density by establishment of new seedlings is poorly understood (Bunting 1985). Only slight increases in perennial grass density been observed within the first five years following fire (Peek et al. 1979, Young and Evans 1978). However because of our lack of knowledge on the population recruitment dynamics of perennial grasses, it is uncertain whether this is a problem. Although observations of increased vigor and seed production following fire have been made (Acker 1988, McShane and Sauer 1985, Patton et al. 1988, Uresk et al. 1976, Young and Miller 1985) , little effect on new plant establishment has been reported (Peek et al.1979, Kuntz 1982). Conversely, numerous bunchgrass seedlings have been observed following fire in central and eastern Oregon mountain big sagebrush communities (J. Kauffman, pers. comm.).

One factor which may limit bunchgrass reestablishment is excessive competition with annual grasses, most commonly the exotic annual cheatgrass (*Bromus tectorum* L.). Harris (1976) ascribed the competitive advantage of cheatgrass over native bunchgrasses to phenological differences in root development between competing plants. Cheatgrass is able to extend its root system throughout the winter months, during which time bluebunch wheatgrass root development ceases. The result is that cheatgrass is able to utilize spring moisture and nutrients before the perennial plant is able to extend its root system, thereby precluding its establishment. However, this competition is density dependent. If fire limits the density of cheatgrass through seed mortality, and encourages seed production of perennial bunchgrasses, this competition may be minimized.

Many exotic annual grasses are well adapted to invade post fire environments, and may well dominate a site if insufficient perennial grasses are present prior to the disturbance. Young and Evans (1978) found that at least 2.5 perennial plants per square meter were necessary to preempt invasion by alien annuals in a big sagebrush\Thurber's needlegrass community near Reno, Nevada. Pechanec and Hull (1945) found that burning earlier in the season (June) reduced cheatgrass germinants roughly ten-fold over burning in late fall (November). However, the effect was short-lived and cheatgrass germinants returned to unburned levels within two years. Even though burning in late spring before seed shatter resulted in 90% mortality of cheatgrass seeds, this timing coincides with a period of high sensitivity to fire damage for perennial grasses (Wright and Klemmedson 1965).

Effects on forbs

Perennial forbs are generally less sensitive to fire than grasses, especially when burned in late summer or fall (Young and Evans 1978, Wright et al. 1979). Pechanec et al. (1954) observed that many forb species respond quickly and favorably in post-fire environments, especially if the plant reproduces by some below-ground organ (e.g. bulbs, corms, rhizomes). Increases have been reported to be generally in regard to productivity, and not density, however, and are strongly influenced by pre-burn forb composition and structure (Bunting 1985). Bunting (1985) emphasized the importance of distinguishing habitat types within the sagebrush/grass vegetation when trying to predict forb response. Perennial forbs that respond favorably include prickly lettuce (*Lactuca serriola* L.) and western yarrow (*Achillea millefolium* L.). Many species of annual forbs are able to pioneer into sites temporarily opened up by fire, including Jim Hill tumblemustard (*Sisymbrium altissimum* L.), and fireweed (*Epilobium* sp.) (Wright 1985).

Effects on community composition and structure

Studies of the effects of fire on the synecological response of sagebrush/grass systems have been limited; usually a given community has been analyzed in terms of examination of individual species or life forms. For instance, Harniss and Murray (1973) examined 30 years of vegetation change in relative cover of shrubs, grasses and forbs. This and other work (e.g. Britton et al. 1981, Humphrey 1984, Olson et al. 1982, Wright et al. 1979) indicate that fire retards the development of sagebrush and juniper dominance, in favor of dominance of herbaceous species.

The effects of fire on community level descriptions such as species richness, heterogeneity, and evenness, however, has received relatively little attention. Humphrey (1984) examined post-fire succession in southeastern Idaho in terms of richness, and diversity, but could find no relationship between stand structure and age since fire. Acker (1988) looked at vegetation change on 5-6 year old stands in terms dissimilarity and trajectory in low-order ordination space, and found that the scale of cover data was more sensitive than frequency data in detecting vegetation change. Communities dominated by both basin and Wyoming big sagebrush demonstrated compositional dynamics that were best explained by weather patterns, but no information regarding the nature of the subject fires was included in the analysis (Acker 1988).

FIRE MANAGEMENT AND PARK SERVICE POLICY

Management of wildlands administered by the National Park Service are under policy mandates that are somewhat unique. The establishment of these lands as parks, or "natural areas" carries with it an entirely different scope of ecosystem management than that of other public lands (Agee and Johnson 1988). The Organic Act of 1916, which established the National Park Service, stated that the mission of the Service is to conserve "the natural and historic objects, and the wildlife therein...for the enjoyment of present and future generations". These two goals, particularly as they relate to fire management, tend to cause conflict, and have spawned a new process-oriented view of wildland management in the National Park Service.

The Leopold report (Leopold et al. 1963) on wildlife management in the Park Service resulted in a major shift in National Park Service policy, with a "vignette of primitive America" becoming the desired condition of our park's wildlands. However, as successful as this new paradigm was for interpreting naturalness, it is still somewhat confined by its static imagery. Recent discussions have emphasized that recreating a "vignette" actually requires the restoration and maintenance of natural processes (Parsons et al. 1986), and thus can be viewed as a "moving picture" (Christensen et al. 1977). Within this framework, fire can be seen as an endogenous ecological process, and consequently a necessary component of wildland management (Parsons et al. 1986).

Currently, policy at John Day Fossil Beds National Monument calls for suppression of all wildfires "until the role of fire is determined and found to be beneficial under specific conditions" (National Park Service 1986). This study has been established to provide such information in regard to the role of natural fire in these systems,and how it can be used to restore ecosystem structure and function.

CONCLUSION

Prior to Euroamerican settlement, fire was a dominant ecological factor that helped shape the landscape of eastern Oregon. Since that time, much of Oregon's big sagebrush country has been radically affected by cultivation, overgrazing, and fire suppression. Accounts and photographs of the presettlement physiognomy indicate that sagebrush densities were historically lower, partially due to intentional burning by native Indians (Shinn 1977). Although fire has often been suggested as a means of stand maintenance or restoration, quantitative data that correlates fire treatments with fire effects is limited.

Objectives of this study are to examine effects of spring and fall prescribed burning on basin big sagebrush communities at John Day Fossil Beds National Monument. The following chapters will focus on characterization of the fuel consumption, environmental conditions and fire behavior associated with the two prescribed fire treatments (Chapter 2), the response of populations of bluebunch wheatgrass and Idaho fescue to these two fire treatments (Chapter 3), and the synecological effects these fires have on the entire plant community (Chapter 4).

CHAPTER 2

A Characterization of the Fire Behavior and Fuel Consumption Associated with Two Prescribed Burns Conducted at Different Seasons

ABSTRACT

Fuel biomass, fuel moisture content, weather conditions at the time of burning, fire behavior characteristics, and fuel reduction (consumption), were measured on experimental plots at John Day Fossil Beds National Monument, Oregon. Two fire treatments were applied; one during the early fall of 1987 (n=4 plots), the other during late spring, 1988 (n=5 plots). Total aboveground biomass (fuel loads) were significantly greater in the fall burn units, averaging 10.6 Mg/ha compared to 6.2 Mg/ha in the spring units. This difference was largely due to more fuel biomass in the living sagebrush component. The was no significant difference between treatments in terms of herbaceous fuels or total fine fuels (grass/herbs plus sagebrush foliage); fall units averaged 3.8 Mg/ha fine fuels compared to 3.0 Mg/ha in spring plots. There were significant differences in fire behavior data associated with the flame dimensions and rate of spread. Fires in the fall treatment plots had greater flame length (4.1 m. compared to 1.7 m. in spring units), rate of spread (1.6 compared to .3 m/s), fireline intensity (6,400 compared to 880 KW/m), and total heat load (18,120 compared to 9,270 KJ/m^2). Reaction intensity and heat-per-unit-area were not significantly different. Mass of fuel consumed was significantly greater in the fall burn units (9.8 compared to 5.2 Mg/ha). However, percent consumption was not significantly different: 92% in fall units compared to 84% in spring units. Data such as these illustrate the fundamental differences in fire behavior and fuel consumption between spring and fall burning, and hence may aid in understanding differences in post-fire vegetation composition, productivity, and succession.

INTRODUCTION

In order to ascertain the effect of a fire on the biotic elements in an ecosystem, quantification of the inherent variability of the fire is of paramount importance. Fires may vary in biomass consumption and flame characteristics (i.e. fire severity) which would in turn cause variable influences on nutrient losses, plant mortality, and post-fire succession. A description of fire behavior in quantitative terms allows for correlations with observed post-fire effects, as well as comparisons between studies (Alexander 1982, Rothermel and Deeming 1980). Generally, experimental fires are either only noted for their occurrence, or described qualitatively with nonstandard descriptors such as "cool" or "hot" (Rothermel and Deeming 1980).

Although many researchers have developed methods for estimating sagebrush biomass (e.g. Brown 1976 and 1982, Fransden 1983, Martin et al. 1981, Rittenhouse and Sneva 1977), none have estimated total aboveground biomass. Similarly, few studies in sagebrush ecosystems have actually quantified fire behavior parameters. Much of the previous work on fire effects in sagebrush ecosystems has been based on after-the-fact observations of the burned vegetation (e.g. Blaisdell 1953, Conrad and Poulton 1966, Uresk et al. 1976, 1980) or on artificial application of fire to individual plants (e.g. Britton et al. 1981, Wright 1971, Wright and Klemmedson 1965). Zschaechner (1985) measured fire and fuel loads, but did not record fuel consumption. Kauffman and Cummings (unpublished data) documented fuel loads, consumption, and fire behavior on an alluvial fan in central Oregon. Unfortunately, this was in a dramatically different sagebrush habitat type of unusually high productivity. Similarly, extensive documentation of fuel, weather and fire characteristics have been accomplished in research of other rangeland systems (Clark et al. 1985, Roberts et al. 1988).

As a consequence of inadequately describing fuel and fire characteristics in quantitative terms, knowledge of the ecological role of fire is limited because of the difficulty in comparing results of different studies (Martin 1984). Alexander (1982) stated that quantitative descriptions of fire behavior are not so useful in their exact description of the energy release process of the fire, but rather in providing information that will enable future researchers/managers the ability to accurately predict the effects of a given fire treatment.

Prediction of fire effects is of concern to wildland resource managers for a variety of reasons. In the case of wildlands under the management of the National Park Service, policy objectives are aimed at restoring all natural processes, including fire, into these systems (Leopold et al. 1963, Parsons et al. 1986). The associated effects of fire in these systems are likely to impinge on all ecological and managerial concerns: vegetation, wildlife, fuel hazard, wilderness, and aesthetics.

This study was established to examine the variable response of fuels and fire behavior in basin big sagebrush (*Artemisia tridentata* subsp. *tridentata* Nutt.)-dominated ecosystems resulting from burning in different seasons: spring and fall. Objectives were to quantify differences in burn treatments based on fuel loads by size class, fuel moisture conditions, weather at time of burn, fire behavior characteristics, and fuel biomass reduction (consumption).

METHODS

All research burns were conducted on similar stands of basin big sagebrush/perennial bunchgrass communities at John Day Fossil Beds National Monument, Sheep Rock Unit, in central Oregon (USGS 15′ series "Picture Gorge"; T 11 S, R 26 E, sections 31 and 32). All units were dominated by basin big sagebrush with a grass/herb understory dominated by Idaho fescue (*Festuca idahoensis* Elmer) and bluebunch wheatgrass (*Agropyron spicatum* (Pursh.) Scribn. & Smith). All units occurred on north slopes, ranging from 20 to 65%. Elevations ranged from 700 to 860 meters. Soils were classified in the Simas-Tub association of very stony clay-loam soils, that occur on moderately deep and well drained slopes and receive 25 to 36 cm. of precipitation per year (USDA Soil Conservation Service 1981). Annual precipitation at the Visitor's Center (approx. 2 km. away from the study site) averaged 29 cm. between 1978 and 1988 (National Park Service 1988).

This study was established in a completely randomized design, with four replicates of a fall burn treatment, and five replicates of spring burn treatment. Size of each unit was at least 30 x 50 meters, with the long axis oriented along the direction of slope.

Pre-fire herbaceous and downed woody fuels were measured in each unit by clipping and collecting all material in ten 30 x 60 cm plots located every two meters along a randomly located 20 meter transect. Fuels were partitioned into the following categories: grass/herb live, grass/herb dead, woody debris 0 -.63 cm in diameter (1-hr timelag), woody debris .63 - 2.54 cm in diameter (10-hr timelag), and woody debris >2.54 cm in diameter (100-hr timelag). These woody fuel size classes correspond to the standardized fuel particle size classes used in inventorying forest fuels, where size classes are partitioned according to rates of fuel moisture loss (Deeming et al. 1977).

Sagebrush cover was estimated by line-intercept along five 15 m. transects in each unit (Canfield 1941), and sagebrush density was measured in 1x15 m. belt transects adjacent to the line transect (Appendix 3). The biomass of living sagebrush plants was estimated using regression analysis, using mean shrub area and volume from the five 15 m. transects. The elliptical crown area of all live sagebrush plants intercepting the line transects was measured utilizing the following equation:

$$A = (W_1 * W_2 * \pi)/4$$

where W_1 = longest crown dimension, and W_2 = the longest crown dimension perpendicular to W_1. Shrub volume was calculated by multiplying crown area by maximum shrub height. These values were then averaged for each transect, and used in fuel loading regression equations developed by Kauffman and Cummings (unpublished data). These equations partitioned sagebrush biomass into the following categories: foliage, 1-hr live, 10-hr live, 100-hr live, and attached dead (no size differentiation). Total loads were then calculated by multiplying the mean loads per plant by the density of shrubs measured in the 1x15 m. belt transects.

To quantify biomass of standing dead woody debris, 50 standing dead sagebrush plants were randomly chosen from an adjacent area. These dead plants were then partitioned into the following size classes: 1-hour, 10-hour and 100-hour. Sub-samples of each size class were taken to determine moisture content, and loadings by size class were corrected accordingly to account for fuels on a dry weight basis. From these data, an average dead sagebrush biomass was determined. This mean was then used in conjunction with dead sagebrush density data from the five 1x15 m. belt transects to determine total standing dead sagebrush biomass. Total aboveground biomass was then calculated as the sum of all grass/herb, dead and downed woody, standing live sagebrush, and standing dead sagebrush fuels. Additionally, fine fuels were calculated as those from sagebrush foliage and the grass/herb category. These fine fuels are considered very important in affecting fire behavior and rate of spread due to their very high surface to volume ratio (Albini 1976, Rothermel 1972, Burgan 1987).

Post-fire herbaceous fuels were determined by clipping ten 30 x 60 cm plots along the opposite side of the pre-burn herbaceous fuel transect. Post-fire woody fuels were estimated in each unit by collecting all material in four randomly located 2x2 m. plots. Fuel consumption was then calculated by subtracting post-fire fuel loads from pre-fire loads.

Before and during all burns, weather conditions were monitored every 30 minutes with hand held devices associated with belt weather kits. The conditions monitored included ambient temperature, relative humidity, and in-stand (2 m. above ground) windspeed.

To estimate moisture content, six samples of soil (0-2 cm), grass/herb, 10-hr woody, and sagebrush foliage were collected immediately prior to ignition of each treatment unit. In the laboratory, the samples were weighed, oven dried for 48 hrs. at 70°C, then re-weighed to determine soil and fuel moisture conditions at time of treatment.

Fires were ignited using a drip torch, in a strip-head firing pattern, with strip widths of 5-10 meters. Units were contained using wetline techniques (Martin et al. 1977), with foam-fortified water. Fire behavior data included type of fire (e.g. strip-head), flame length (m.), flame height (m.), flame depth (m.), flame angle (degrees off the horizontal), rate of spread (m/s), and residence time of active flaming combustion (seconds) (Alexander 1982). All of the above behavior characteristics were measured by randomly selecting 5 to 12 areas in each unit for observation. Photographs taken of the observation points were used to verify accuracy of ocular estimates of flame dimensions.

From the measurements of flame length (FL), the rate of energy release per meter width of flame front, or fireline intensity (I_{FL}) was estimated. Based on an empirical relationship between observed flame length and fireline intensity developed by Byram (1959), intensity was calculated using the following formula given in Rothermel and Deeming (1980):

$$I_{FL} = 258*FL^{2.17} \qquad [KW/m]$$

Reaction intensity (I_R), or the rate of energy release per square meter of flaming zone, was calculated by dividing the fireline intensity by flame depth. The total energy released in the active flame front, or heat-per-unit-area (H_a), was calculated by dividing the fireline intensity by the rate of spread. The total heat load released by the fire was calculated as the product of the total fuel consumed and the net heat of combustion of that fuel. There are no values for heat of combustion of sagebrush fuels available in the literature; however, it has been shown to vary by less than 10% between fuel types, so an average value of 18,700 KJ/kg, subject to a reduction of 24 KJ per moisture percentage point, was used in this calculation (Van Wagner 1972). Considering the high quantities of volatile oils and waxes, this energy estimate is probably a conservative one.

Analysis of all variables employed the use of a paired t-test to test for differences between burn treatment means. Tests of significance were conducted at the $p<.05$ level, using a two-tailed probability.

RESULTS AND DISCUSSION

Aboveground biomass of sagebrush dominated ecosystems

Total aboveground biomass ranged from 5 to 12 Mg/ha in the nine experimental units; fall treatment units averaged 10.59 Mg/ha, as compared to 6.23 Mg/ha in the spring treatments (Table 1). Much of this disparity between treatments was due to greater amounts of live sagebrush in the fall units. Mean sagebrush canopy cover was 15% in fall units and 7.5% in spring units. Consequently, all fuel categories associated with the live sagebrush were significantly different between treatments (Table 1). The proportion of the total fuel load consisting of live sagebrush was accordingly relatively higher in the fall units: 48% as compared to 29% in the spring units.

There were no significant differences between treatments in terms of standing dead woody debris, dead and downed woody debris, or total grass/herb fuels (Table 1). As expected, however, the spring units contained significant amounts of live grass/herb fuels, a direct result

Table 1. Aboveground biomass (Mg/ha) for spring and fall prescribed burn units. Numbers in parenthesis are the standard error of the group mean.

FUEL CATEGORY	TREATMENT FALL	TREATMENT SPRING
LIVE SAGEBRUSH		
ATTACHED DEAD*	1.06 (.12)	0.42 (.16)
TOTAL LIVE*	4.11 (.48)	1.67 (.63)
FOLIAGE*	0.84 (.13)	0.32 (.12)
1-HOUR T.L.*	1.16 (.19)	0.44 (.16)
10-HOUR T.L.*	1.35 (.22)	0.52 (.19)
100-HOUR T.L.*	2.18 (.25)	0.89 (.33)
DEAD SAGEBRUSH		
1-HOUR T.L.	0.11 (.02)	0.11 (.02)
10-HOUR T.L.	0.28 (.06)	0.26 (.04)
100-HOUR T.L.	0.50 (.11)	0.46 (.07)
GRASS/HERBS		
LIVE*	0.00 (0)	0.86 (.22)
DEAD	3.01 (.65)	1.81 (.22)
DEAD AND DOWNED		
1-HOUR T.L.	0.52 (.24)	0.31 (.09)
10-HOUR T.L.	0.58 (.21)	0.25 (.13)
100-HOUR T.L.	0.04 (.04)	0.00 (0)
TOTALS		
STANDING LIVE*	4.11 (.48)	1.67 (.63)
STANDING DEAD*	1.96 (.19)	1.26 (.09)
GRASS/HERBS	3.01 (.65)	2.67 (.23)
1-HOUR T.L.*	1.80 (.36)	0.86 (.11)
10-HOUR T.L.*	2.22 (.37)	1.03 (.20)
100-HOUR T.L.*	2.72 (.24)	1.35 (.28)
TOTAL ABOVEGROUND BIOMASS*	10.59 (1.12)	6.23 (.69)

* denotes significant difference between treatments at p<.05.

of the inherent phenological differences that result from burning in different seasons.

Total standing live fuels were significantly greater in the fall treatment units, averaging 4.11 Mg/ha, compared to a mean of 1.67 Mg/ha in spring units. Similarly, total standing dead woody debris (from both live and dead shrubs) was significantly greater in the fall units (1.96 Mg/ha), than in the spring units (1.26 Mg/ha). Both treatments contained relatively abundant amounts of grass/herb fuels; fall treatments averaged 3.01 Mg/ha, spring units averaged 2.67 Mg/ha. It should be noted that these herbaceous fuel loads well exceed the threshold values for fire spread. Beardall and Sylvester (1976) estimated that 674-786 kg/ha is required to sustain fire spread; thus the loads here are roughly four times that required for sustained combustion. Aboveground biomass partitioned into the aforementioned classes is shown in Figure 1.

In both treatments, the fuel loads followed similar size class distributions, with the largest proportion of fuels being from the fine (herb plus sage foliage) category, comprising 37% of the total fuel in fall units and 48% in spring units. The next most abundant size class for both treatments were the 100-hr fuels, compromising from 23 to 28% of the total fuel load. The remaining fuels were comprised of 10-hr and 1-hr fuels, with the 10-hr fuels being slightly more abundant.

Fuel moisture and fire weather

Fall plots were burned under drier fuel conditions in comparison to the spring burns (Table 2). Soils, dead fines, and 10-hr fuels were very dry (3-9% moisture content) in both treatments, and no significant differences of the parameters was apparent between treatments. However, there were significant differences between treatments in regard to live fuels (Table 2). Where no live grass existed during the fall burn, an average of one-third of the grass/herb biomass was live

Figure 1. Aboveground biomass by fuel size-class and burn treatment.

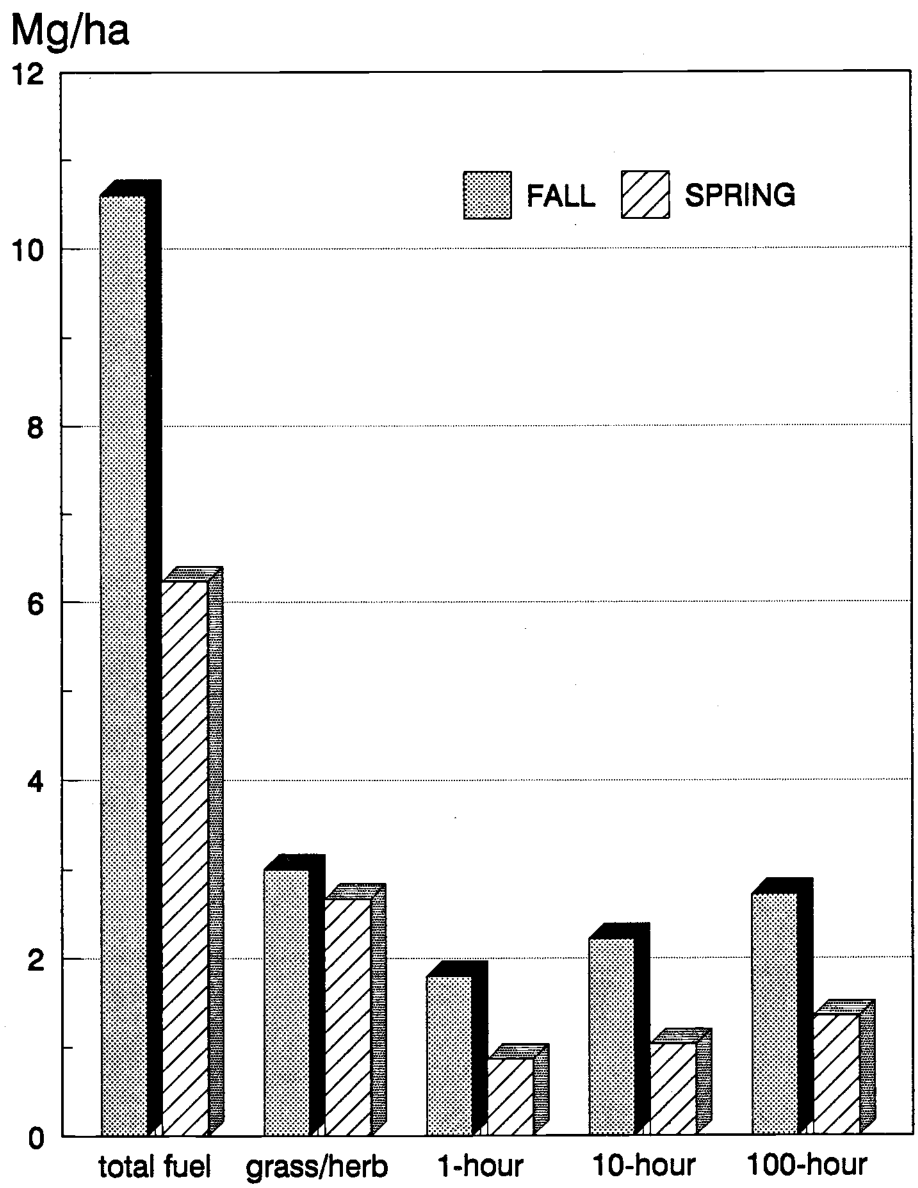


Table 2. Environmental conditions associated with fall and spring prescribed burns. Data include date and time of prescribed burns, ranges of weather conditions, and mean fuel moisture contents, based on 16 samples per treatment. Numbers in parenthesis are the standard error of the group mean.

TREATMENT	**FALL**	**SPRING**
Date	9/25/87	5/24/88
Time of burning	9:35-13:45	12:35-15:26
Temperature (°C)	15-18	23-35
Relative Humidity	41-48%	21-24%
Windspeed (kph)	0-15	0-17
Soil M.C. (%)	2.90 (.20)	3.21 (.41)
Dead grass/herb M.C.*	8.88 (.66)	7.36 (.87)
Live grass M.C.	N/A	142.60 (11.79)
Sagebrush foliage M.C.*	97.19 (6.27)	186.02 (5.12)
10-hr Timelag M.C.	4.59 (.09)	4.99 (.35)

* Denotes a significant difference between treatment means at $p<.05$.
N/A = Not applicable.

in the spring units. These fuels averaged 142% moisture content. Thus, the effective moisture content for the herbaceous fuel layer was significantly higher in the spring burn units than in the fall units. All grass/herb fuels in the fall units had completely cured by the time of burning, averaging 8.9% moisture content. Similarly, foliage of live sagebrush was significantly different between treatments. The spring burn occurred when foliar moisture levels were at an average of 186% compared to only 97% during the fall burn. This difference is most likely due to the change in phenological stages of sagebrush. During spring, when soil moisture is relatively available, high rates of stomatal conductance and photosynthesis occurs in sagebrush (Depuit and Caldwell 1973). These higher moisture levels in live fuels may be responsible for the somewhat lower consumption rates and survival of some sagebrush individuals associated with the spring burn.

Although most fuels were drier in the fall, dead grasses were slightly drier in the spring (7.4%). This is a reflection of the lower relative humidity during the spring burns (Table 2). However, the effect of the lower relative humidity and higher ambient temperatures on fire behavior and consumption was likely overshadowed by the presence of a significant quantity of live fuels at much higher moisture contents in the spring burns.

Fire behavior

All measurements of flame dimensions and dynamics were statistically significantly different in the two treatments (Table 3). Only those parameters based on equal area of flame zone (i.e. reaction intensity and heat-per-unit-area) were not significantly different between treatments.

The fall treatment averaged over 4 m. flame length, compared to less than 2 m. for the spring burn. As I_{FL} varies exponentially with flame length, this difference results in a seven-fold increase in fireline intensity (Table 3). Likewise, the rate of spread of the

Table 3. Fire behavior and fuel consumption for fall and spring prescribed burn treatments. Numbers in parenthesis are the standard error of the group mean.

VARIABLE	TREATMENT FALL	SPRING
Flame length* (m)	4.14 (0.77)	1.74 (0.77)
Intensity (I_{FL})* (KW/m)	6,441 (1,861)	883 (100)
Intensity (I_R) (KW/m^2)	591 (169)	346 (42)
Flame Height* (m)	2.17 (0.31)	1.12 (0.02)
Flame Depth* (m)	10.35 (1.84)	2.56 (0.21)
Rate of Spread* (m/s)	1.57 (0.29)	0.23 (0.03)
Heat per Area (KJ/m^2)	3,253 (812)	3,935 (461)
Total Energy* (KJ/m^2)	18,119 (2,357)	9,267 (1,464)
Residence Time (s)	6.92 (0.91)	11.66 (6.92)
Consumption* (Mg/ha)	9.80 (1.30)	5.23 (0.91)

* significant difference between treatments at $p<.05$

flame front was six times faster in the fall burn. Consequently, in terms of above ground fire effects, the significant difference in the physical nature of the two fires may explain some of the observed differences in vegetation response between the two treatments (see Chapters 3 and 4).

The reaction intensity and heat-per-unit-area were approximately equivalent in both treatments (Table 3). Since the linear advance of the flame front was only 16% in spring burns compared to that for the fall treatment, a given area remained in the active flaming zone for roughly twice as long. This was probably due to the higher moisture content of the live fuels during the spring burn, since average wind/slope factors affecting rate of spread were roughly the same.

The significant difference in total energy released by the two treatments (Table 3), reveals a shortcoming of using heat-per-unit-area to assess fire effects. Whereas the two treatments appear equivalent in terms of H_a, it reflects only the heat flux realized in the active flame front. That is, this value differs from the total energy released in the same way as the amount of fuel consumed in the active combustion zone differs from the total fuel consumed. Photographic evidence revealed that considerable standing biomass remained immediately after the passage of the fire front. However, the next day much less standing biomass remained. Fuels consumed after the passage of the flame front continue to release heat, and may be extremely important in regard to the effect of sustained high temperatures on living tissues. This total energy released by both flaming and glowing combustion averaged over 18,100 KJ/m^2 in the fall burn units, as compared to 9,267 kJ/m^2 in the spring burns. This difference may manifest itself in terms of effects on plants, as well as below ground heat flux, and consequently may prove to be a better parameter for correlation with fire effects.

Fuel consumption

Total fuel consumption was significantly different between the two treatments (Table 4). Total consumption was almost twice as great in the fall (9.80 Mg/ha) as in the spring (5.23 Mg/ha), but as noted earlier, most of this difference was due to live sagebrush fuels. Post-fire fuel loads (i.e. residual fuels) were not significantly different, with the spring units averaging slighter more residual biomass (Table 4).

Fuel consumption of the fine and 10-hr fuels was not significantly different between treatments (Table 4). Consumption of both 10-hr and 100-hr fuels was significantly greater in the in the fall units. Greater fuel loads and lower fuel moistures of live sagebrush fuels in these size classes both contributed to this difference in consumption. A graphic representation of fuel consumption by fuel size class is given in Figure 2.

Consumption expressed as a total of pre-fire fuel loads reveals that both treatments consumed similar percentages of the fuels present (Table 4). In terms of total biomass, the fall treatment averaged 92.5 % consumption, as compared to 84% in the spring burns. Fine fuels and 100-hr fuels were consumed at similar levels in both treatments, ranging from 92 and 96%. The 1 and 10-hr fuels were consumed at lower percentages in the spring units.

The potential misrepresentation of fire effects through reporting only consumption in terms of percent of pre-fire fuel loads is apparent in Figure 3. In situations where pre-fire fuels are significantly different between treatments, it is misleading to only report that biomass consumption varied by less than 10%. Consequently, it seems appropriate that both gross consumption and consumption as a percent of pre-fire fuel loads be expressed when evaluating fire effects.

Figure 3 reveals an anomaly associated with the consumption of the 10-hr fuel size-class. While all other fuel categories

Table 4. Residual fuel loads and fuel consumption (Mg/ha) for fall and spring prescribed burns. Numbers in parenthesis are the standard error of the group means.

RESIDUAL FUEL LOADS:

FUEL CATEGORY	TREATMENT FALL	SPRING
FINE FUELS	0.23 (.16)	0.23 (.03)
1-HR T.L.	0.15 (.04)	0.20 (.05)
10-HR T.L.	0.32 (.02)	0.49 (.07)
100-HR T.L.	0.09 (.04)	0.08 (.03)
TOTAL BIOMASS	0.79 (.18)	1.00 (.22)

FUEL CONSUMPTION:

FUEL CATEGORY	FALL	SPRING
FINE FUELS	3.64 (.72)	2.76 (.27)
percent	94.54	92.31
1-HR T.L.	1.65 (.40)	0.66 (.16)
percent	91.66	76.74
10-HR T.L.*	1.90 (.39)	0.54 (.27)
percent	85.59	52.43
100-HR T.L.*	2.63 (.28)	1.27 (.31)
percent	96.69	94.07
TOTAL BIOMASS*	9.80 (1.30)	5.23 (.91)
percent	92.54	83.95

* Denotes a significant difference between treatment means (p<.05).

Figure 2. Fuel consumption (Mg/ha) by size class and burn treatment.

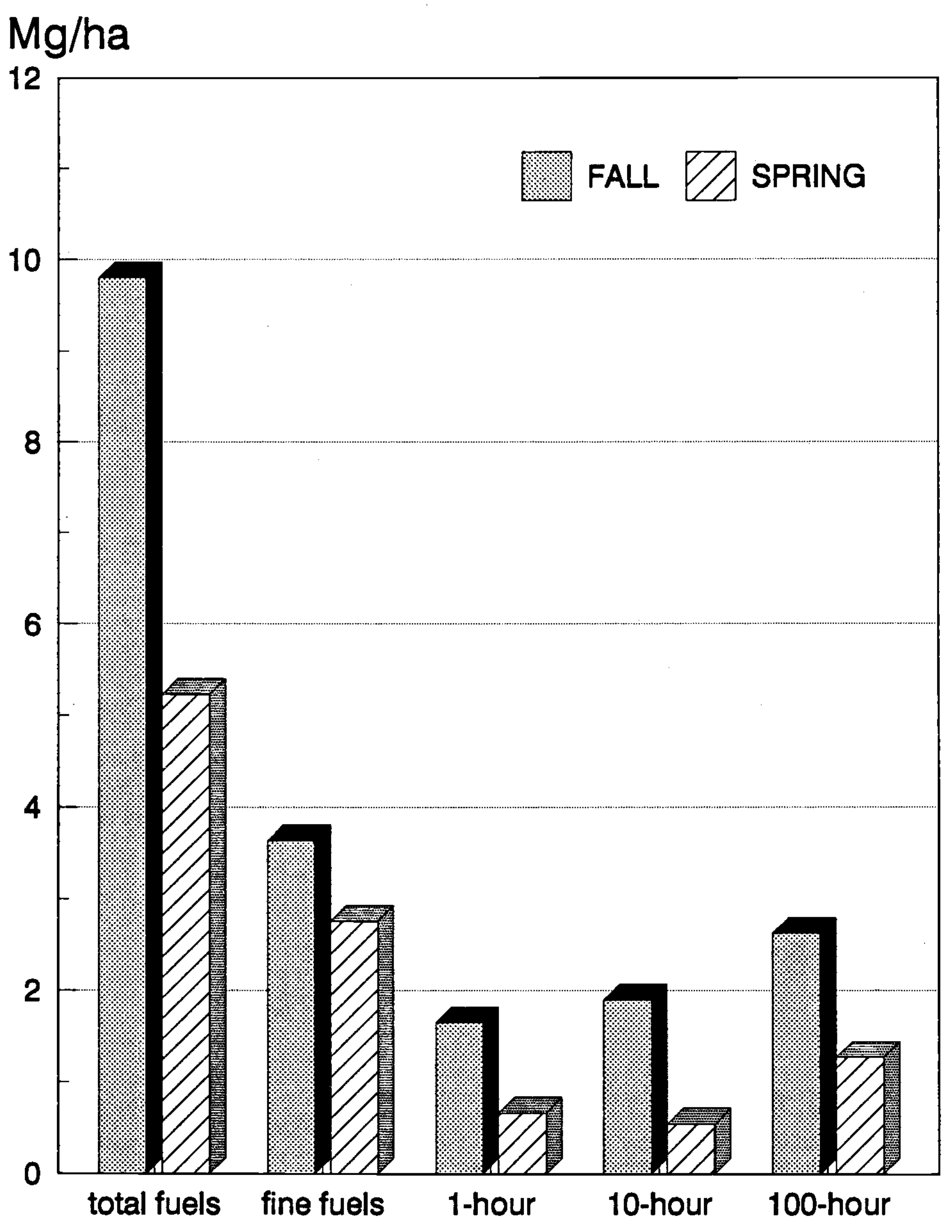
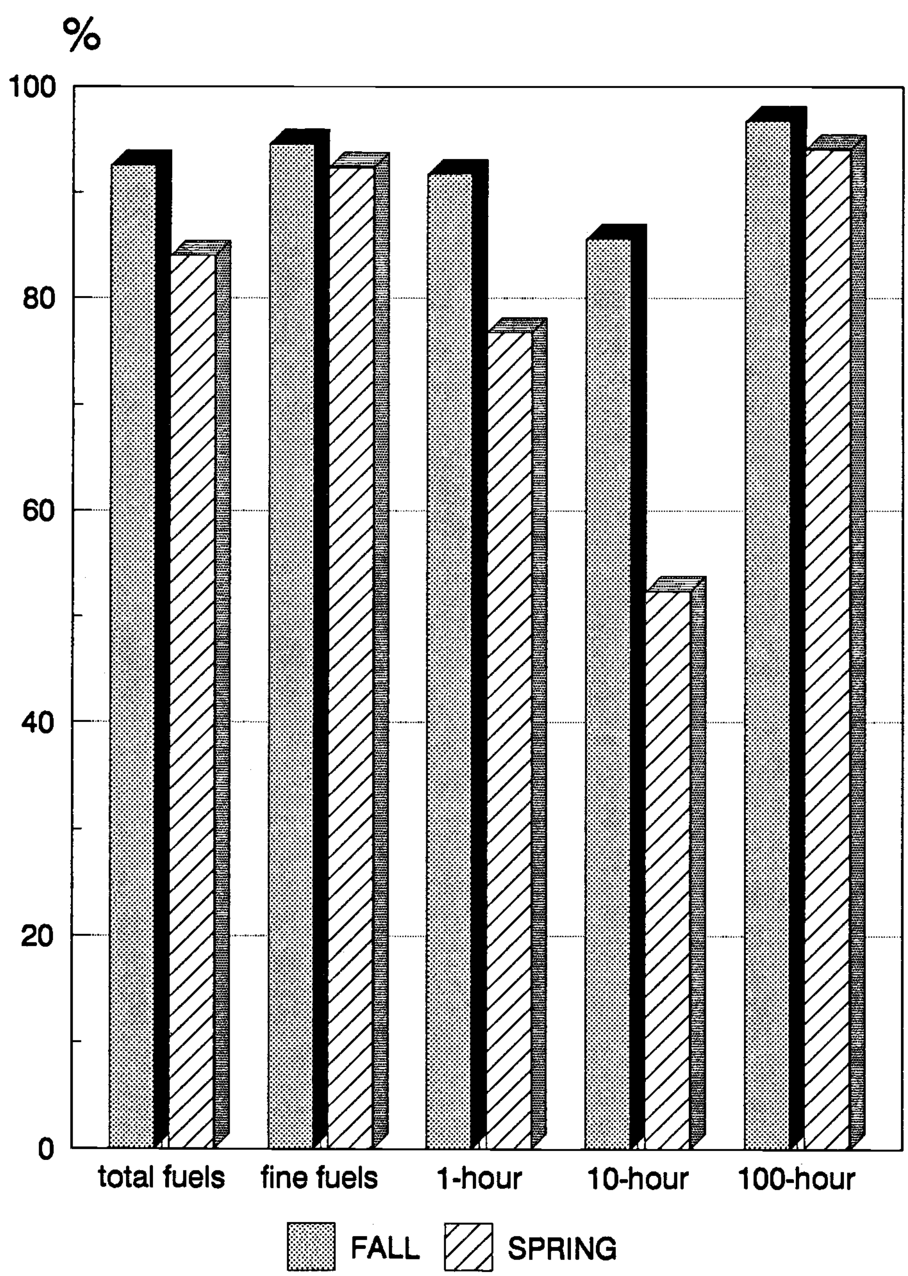


Figure 3. Fuel consumption (%) by fuel size-class in fall and burn treatment.



demonstrated rather similar consumption percentage rates, there was a striking difference between treatments with respect to 10-hr fuel consumption. Where almost 85% of the 10-hr fuels were consumed by the fall fire, only 52% of the 10-hr fuels were consumed by the spring burn. It seems likely that higher moisture content of live 10-hr fuels in the spring units contributed to the lower consumption recorded.

CONCLUSION

In this paper I have attempted to adequately describe pre-fire fuel loads, weather and fuel moisture characteristics at the time of the burns, and the fire behavior and fuel consumption associated with two experimental burns conducted on basin big sagebrush dominated ecosystems in eastern Oregon.

The fire behavior characteristic selected for correlation must have a logical relationship to the effect being studied. Understanding post-fire effects on the plant community in general, with particular concern for the native perennial grasses, was the predominant objectives of this study. Therefore, I believe that the treatment documentation presented here will provide information necessary to understand fire effects on the vegetation components of basin big sagebrush dominated ecosystems.

Historically, fire was a dominant ecological process affecting the sagebrush/perennial bunchgrass communities of the western United States (Daubenmire 1968, Kauffman and Sapsis 1989). However, fire is as variable in space and time as any other natural physical or climatic disturbance factor present in an ecosystem. Developing quantitative data such as those presented here is an important first step in understanding the variability of fire in this ecosystem, as well as developing a scale of understanding of ecological effects wrought by the myriad of potential fires that may sweep through these landscapes.

The two subject fires in this study provide a good contrast of fire behavior and fuel consumption, supporting the contention of the inherent variability of fire in space and time. Thus, they provide a basis for understanding the natural role of fire, as well as its potential uses as a land management tool.

CHAPTER 3

Response of Idaho Fescue and Bluebunch Wheatgrass to Spring and Fall Prescribed Burning

ABSTRACT

Effects of spring and fall prescribed burning on populations of Idaho fescue (*Festuca idahoensis* Elmer) and bluebunch wheatgrass (*Agropyron spicatum* (Pursh.) Scribn. & Smith) were investigated through measurement of plant density, basal cover, mean basal area, mean number of flowering culms, and mean plant height. A total of 403 bluebunch wheatgrass plants, and 389 Idaho fescue plants, were tagged and measured in a basin big sagebrush (*Artemisia tridentata* subsp. *tridentata* Nutt.) dominated community. Burn treatments resulted in 84-93% of aboveground biomass consumption, and 87-100% mortality of sagebrush plants. Control and fall burned plants were measured before and for two years after burning; spring burned plants were measured before and one year after burning. Fire induced mortality of bluebunch wheatgrass was approximately 5% for both burn treatments. Approximately 20% of Idaho fescue plants were killed by fall burning, while spring burning resulted in only 3.5% mortality. No significant changes were detected in plant density or basal cover of either species, although fall burning reduced cover of Idaho fescue from 10% to 6% the first year following burning. The mean basal area of bluebunch wheatgrass significantly increased by 36% two years after fall burning. At this time, these plants were significantly larger than those in adjacent controls. Spring burning did not significantly change plant size one year after burning. Fall burning of Idaho fescue initially reduced mean basal area 23% the first year after burning; however, by the second year plants had recovered to 90% of their pre-burn size. Spring burning resulted in no significant change in basal area of fescue. Fall burning reduced number and height of flowering culms on both species the first year after burning. However, significant increases were measured in comparison to both pre-burn and unburned controls the second year. Spring burning of bluebunch wheatgrass caused no significant change in numbers of flowering culms.

Idaho fescue plants demonstrated significantly more flowering culms and greater plant height the first year after spring burning when compared to pre-burn levels. However, flowering culms were significantly fewer and shorter than control plants. Fall burning appeared to stimulate growth and reproductive effort of bluebunch wheatgrass, while causing moderate decreases in growth of Idaho fescue. Spring burning appears to have little effect on populations of either species.

INTRODUCTION

Given the tremendous variation in environmental factors such as weather, fuel mass, or topography, fires can result in dramatically different levels of plant survival and regrowth. Additionally, many biotic factors have been suggested to contribute this variability, including species, season of burn, size of plant, growth form, and amount of dead material surrounding the plant crown (Wright 1985). The descriptive parameter used to assess vegetation change is also an important criterion for determining perceived changes resulting from fire (Bunting 1985). For instance, although a given species may respond favorably in terms of change in basal area, it may decrease in terms of plant density or annual productivity.

Many autecological studies of perennial grasses have examined fire effects related to wildfires (e.g. Conrad and Poulton 1966, Countryman and Cornelius 1957, Houston 1973, Uresk et al. 1976, 1980). Unfortunately, these post-hoc studies have been largely based on the appearance of the burned vegetation, and thus failed to give a quantitative description of the fire that was responsible for the observed effects. Other studies have artificially implemented fire by burning individual plants inside a 55-gallon drum "barrel burner", using shredded paper (e.g. Wright and Klemmedson 1965) or a propane torch (e.g. Britton et al. 1983) to ignite the plant.

Studies using prescribed fire over the entire landscape have generally been restricted to severely damaged systems that were in very poor range condition (Blaisdell 1953) or limited to few observations (Zschaechner 1985). Investigations of prescribed burning as an ecological agent in basin big sagebrush (*Artemisia tridentata* subsp. *tridentata* Nutt.) dominated systems are lacking. Since much of the observed variability of fire effects can be attributed to site conditions that manifest themselves as different big sagebrush subspecies, it is important to note the subspecies when investigating fire effects (Winward 1985).

Idaho fescue is generally considered to be very sensitive to fire (Blaisdell 1953, Champlin 1983, Conrad and Poulton 1966, Wright 1971), with the degree of damage largely a function of season of burn (Britton et al. 1983, Conrad and Poulton 1966, Champlin 1983, Wright et al. 1979). For example, mid-summer burns of a mountain big sagebrush site in eastern Oregon resulted in a 50% reduction in basal area, as compared to 34% reduction in basal area following fall burning (Britton et al. 1983). Similarly, a mid-july wildfire in northeastern Oregon resulted in a 50% reduction in basal area and 27% mortality (Conrad and Poulton 1966). Effects of spring burning of Idaho fescue are less clear; burning in mid-May caused a 48% reduction in basal area and 30% mortality (Britton and Sneva 1981), although others have reported few negative effects (Blaisdell et al. 1982). Reproductive potential of Idaho fescue has been reported to increase the first few years following fire in both mountain and basin big sagebrush sites; increases in flowering culms recorded on a Wyoming big sagebrush site were slower, not appearing until five years after burning (Patton et al. 1988).

In contrast, bluebunch wheatgrass (*Agropyron spicatum* (Pursh.) Scribn. & Smith) has been shown to be slightly affected by burning, with reductions evident only the first year following burning (Uresk et al. 1980). Season of burn has been shown to be an important factor affecting successional development; fall burns on a mountain big sagebrush dominated site appeared to have little or no effect on bluebunch wheatgrass, compared to greater mortality and increased reduction in basal area from burning in mid-May (Britton and Sneva 1981). These findings are in contrast to Champlin (1983), where basal area of bluebunch wheatgrass increased one and two years following both spring and fall burns in mountain big sagebrush dominated sites in northern California and central Oregon. Zschaechner (1985) reported from zero to 80% mortality, and from 25 to 75% reduction in basal area of surviving plants from late summer prescribed fires in mountain big sagebrush sites in Nevada. Reproductive effort has been reported to initially decrease (McShane and Sauer 1980, Patton et. al. 1988), with a positive response generally observed by the second year after burning (Patton et. al. 1988, Uresk et. al. 1980).

The overall objective of this study was to quantify the adaptations and ecological response of Idaho fescue and bluebunch wheatgrass to spring and fall burns in a basin big sagebrush-dominated ecosystem. To address this objective the following questions were posed: (1) does mortality differ between spring and fall burns?; (2) does mortality differ between species?; (3) does fire-enhanced reproductive effort exist for these two species, and if so what effect does season of burn have on it?; and (4) what effect does season of burn have on vegetative regrowth?. Population parameters measured included mortality, density, basal cover, mean basal area per plant; reproductive effort was examined by measuring the height and number of flowering culms on individual bunchgrasses. From these data both the effects of varying fire treatments and the adaptive mechanisms that allow for persistence in the face of variable fires will be addressed.

METHODS

The study area was located at John Day Fossil Beds National Monument, Sheep Rock Unit, approx. 10 km west of Dayville, OR. Elevation ranges from 700 to 850 m, and average annual precipitation is approx. 30 cm. Soils are moderately deep, very stony clay-loams of the Simas-Tub association (U.S. Soil Conservation Service, 1980). The area is dominated by basin big sagebrush in the overstory, and Idaho fescue and bluebunch wheatgrass in the understory. Dominant forb species included western yarrow (*Achillea millefolium* L.) and threadstalk milkvetch (*Astragalus fillipes* Torr.). Species richness prior to burning ranged from 27 to 34. Slopes were all north exposures, ranging from 20 to 65%.

The study was established in a completely randomized design. Data were collected from 10 experimental units burned on September 25, 1987 (fall treatment), and five experimental units burned on May 24, 1988 (spring treatment). Each unit was at least 30 x 50 m, with the long axis oriented with the slope. A detailed description of the fuel loads, fire behavior, and fuel consumption associated with these two treatments can be found in Chapter 2. Three similar units were not burned to serve as controls.

In each unit 8-32 individual bunchgrasses of both species were tagged with a flagpin, and numbered with an aluminum tag. The location of each tagged individual was then was then mapped to aid in finding the plants after burning. Each tagged individual was measured for its longest basal dimension (w_1), and the longest basal dimension perpendicular to the first axis (w_2), from which elliptical basal area was calculated using the following formula:

$$Area = (w_1 * w_2 * \pi)/4$$

Each plant was also measured for its maximum height (including flowering culms), and number of flowering culms.

Density of the two species was measured along three transects in each unit. Each transect had ten 30x60 cm microplots located every meter along its length (Appendix 3). Basal cover was calculated by multiplying the mean density per microplot by the mean basal area of plants in that unit, then dividing by the microplot area.

All measurements were conducted on fall and control units in July and August, 1987, May and June, 1988, and June and July, 1989. Spring units were measured in April and May, 1988, and June and July, 1989.

Analysis was accomplished utilizing Analysis of Variance techniques for a completely randomized design (Peterson 1985) to test for differences between years within the fall and control treatment units. If the F-test was significant, a multiple comparison test (Scheffe') was used to distinguish which years were different from one another. A t-test used to compare the two years of spring burn data. Since the two burn treatments were applied in different years, tests between treatments within years were restricted to comparisons between burn treatments and control using a t-test. All analyses were conducted using a $p<0.10$ significance level.

RESULTS

Mortality

First year mortality of tagged bluebunch wheatgrass plants was 5.2% in fall units, and 4.4% in spring units (Table 1). In contrast, the fall burn resulted in 20.1% mortality of tagged Idaho fescue, as compared to only 3.5% from spring burning. No significant mortality (0-0.7%) of either species was measured in the control units.

Density

Although there were low levels of fire-induced mortality of tagged plants, density of bluebunch wheatgrass in permanent transects remained relatively constant in all treatment units throughout the study period (Table 2). The was no significant change in plant density in either fall or control units between 1987 and 1988; however both control and spring units showed increases of approximately 14%, while fall units showed a similar size decrease in density between 1988 and 1989.

Density of Idaho fescue was more variable, both between treatments, and over time (Table 2). Fall units averaged approximately 17 plants per square meter, compared to less than $4/m^2$ in the spring and control units. Although tests between years within treatments revealed no significant differences over time, all three treatments recorded minor (<10%) density reductions the first year

Table 1. Mortality (expressed as a percent of pre-treatment numbers) of *Agropyron spicatum* (AGSP) and *Festuca idahoensis* (FEID) by burn treatment. Numbers in parenthesis are the standard error of group means.

TREATMENT	SPECIES **AGSP**	**FEID**
CONTROL	0^a (0)	0.7^a (0.7)
FALL	5.2^b (2.1)	20.1^b (4.1)
SPRING	4.4^b (1.8)	3.5^a (2.6)

Different superscripted letters denote a significant difference in mortality in different treatments ($p<.10$).

Table 2. Density (numbers per m^2) and basal cover (%) of *Agropyron spicatum* (AGSP) and *Festuca idahoensis* (FEID) by treatment and year. Numbers in parenthesis are the standard error of the mean.

	TREATMENT		
	FALL	SPRING	CONTROL
AGSP:			
Density '87	2.222 (1.281)	ND	1.852 (2.553)
Density '88	2.241 (1.352)	3.297 (1.223)	1.852 (2.986)
Density '89	1.908 (.912)	3.815 (1.345)	2.0 9 (2.226)
% change			
year 1	0.86	15.71	0
year 2	-14.13	ND	13.37
Cover '87	3.7 (1.9)	ND	3.2 (1.0)
Cover '88	4.2 (2.6)	4.5 (1.6)	2.9 (0.4)
Cover '89	4.6 (1.8)	4.4 (1.2)	3.4 (0.2)
% change			
year 1	13.51	-2.27	-10.35
year 2	24.32	ND	6.3
FEID:			
Density '87	18.798^{a} (6.139)	ND	1.669 (.609)
Density '88	17.298^{a} (5.915)	3.185 (.902)	1.519 (.618)
Density '89	17.285^{a} (5.300)	2.926 (.841)	1.173 (.627)
% change			
year 1	-4.85	-8.85	-9.88
year 2	-8.75	ND	-42.29
Cover '87	10.2^{a} (2.8)	ND	1.0 (0.4)
Cover '88	5.7^{a} (2.6)	1.2 (0.4)	0.9 (0.5)
Cover '89	6.8^{a} (2.3)	1.4 (0.7)	0.9 (0.4)
% change			
year 1	-44.12	16.67	-10.00
year 2	-33.33	ND	0

Different superscripted letters denote a significant difference between burn treatment means and control ($p<.10$).
ND = No Data.

following burning.

Basal Cover

Basal cover of bluebunch wheatgrass increased in the fall burn units from 3.7% before burning, to 4.6% two years later (Table 2). Over the same period, control units increased from 3.2% to 3.4%. Spring units averaged 4.5% before burning in 1988, and 4.4% the following year. All comparisons, both between treatments, and over time, revealed no significant differences.

Changes in cover of Idaho fescue were considerably more variable than bluebunch wheatgrass, both between treatments and between years. Reflecting the greater abundance of fescue in the fall units, pre-burn cover in the fall units was an order of magnitude greater than either the spring or control plots (approx. 10% as compared to approx. 1%). These differences remained significant throughout the study period. Tests between years revealed no significant changes; however, fall units decreased from 10.2% cover, to 5.7% in 1988, with a positive response to 6.8% in 1989. The spring treatment resulted in a modest increase in cover, going from 1.2% before burning in 1988, to 1.4% in 1989. Basal cover of Idaho fescue did not significantly change over the three years.

Basal Area

Mean basal area of bluebunch wheatgrass increased from 199 cm^2 to 215 cm^2 the first year following fall burning, and to 271 cm^2 the second year (Table 3). This amounts to a net increase of 36% over pre-burn levels. Spring burning resulted in a decrease from 161 cm^2 prior to burning to 149 cm^2 the following year, or a net decrease in basal area of 7.6%. Control plants were somewhat variable, averaging 189, 154, and 172 cm^2 in 1987, 1988, and 1989, respectively. Tests within treatments between years revealed significant regrowth over pre-burn levels for the fall units both years after burning, while neither spring or control units changed significantly (Table 3). Tests within

Table 3. Sample size and mean basal area (cm^2) of *Agropyron spicatum* (AGSP) and *Festuca idahoensis* (FEID) by treatment and year. Numbers in parenthesis are the standard error of group means.

		YEAR		
SPP/TRMT	**n**	**1987**	**1988**	**1989**
AGSP:				
FALL	175	198.5^{a} (17.1)	214.7^{ab1} (19.0)	271.2^{b1} (23.1)
SPRING	133	ND	161.2 (32.0)	148.9 (28.4)
CONTROL	95	189.3 (27.9)	153.6 (17.0)	171.8 (19.1)
FEID:				
FALL	195	60.9 (5.5)	46.9 (5.3)	54.8 (6.2)
SPRING	138	ND	40.5 (6.2)	45.6 (6.7)
CONTROL	56	66.1 (8.0)	56.8 (8.9)	48.6 (7.1)

Means with different superscripted letters denote a significant difference between years within treatments; means with different superscripted numbers denote differences between burn treatment and control within years ($p<.10$).
ND = No Data.

years between treatments showed no significant difference in mean basal area of bluebunch wheatgrass between either burn treatment and control prior to burning, but both years following burning the fall treatment was significantly greater than control (Table 3). Prior to burning, fall unit wheatgrass plants averaged approx. 9 cm^2 larger basal area than adjacent control plants; two years later the difference had grown to approx. 100 cm^2.

Prior to burning, mean basal area of Idaho fescue was not significantly different between burn treatments and control, although both burn treatments averaged smaller plants (Table 3). Tests in subsequent years revealed no significant differences between burn treatments and controls. Although tests within treatments over time revealed no statistical significance, fall burning of fescue resulted in an initial decrease of 23% (from 60.9 cm^2 before burning to 46.9 cm^2 a year later). A moderate recovery to a net decrease of 10% from pre-burn levels(mean = 54.8 cm^2) was recorded the second year. Spring burning resulted in a 12.5% increase in mean basal area one year after burning, whereas control plants followed a steady trend of decreasing size throughout the study period.

Reproductive Vigor

Prior to burning, neither species exhibited a significant difference in either number of flowering culms or maximum plant height between burn treatments and the controls (Table 4). In 1987, bluebunch wheatgrass averaged 33 flowering culms per plant in the fall units, compared to 46 in the controls. Tests over time revealed flowering culms on bluebunch wheatgrass declining the first year following fall burning from 33/plant to less than 12/plant, a decrease of 65%. However, by the second year, wheatgrass plants averaged greater than 58 culms/plant, a 75% increase over pre-burn levels. Maximum plant height followed a similar trend, with a significant decrease the first year followed by a significant increase the second year (Table 4).

Table 4. Mean number and height (cm.) of flowering culms on *Agropyron spicatum* (AGSP) and *Festuca idahoensis* (FEID) by treatment and year. Numbers in parenthesis are the standard error of group means.

SPP/TRMT/VARIABLE	YEAR 1987	1988	1989
AGSP:			
FALL			
number	33.6^{a} (3.2)	11.6^{b1} (1.9)	58.7^{c1} (5.5)
height	67.1^{a} (1.4)	59.5^{b1} (1.4)	83.1^{c1} (1.5)
SPRING			
number	ND	17.1 (4.8)	16.7 (3.8)
height	ND	64.8^{a} (1.4)	59.6^{b1} (1.7)
CONTROL			
number	46.0^{a} (7.9)	28.1^{b} (5.1)	22.6^{c} (4.4)
height	70.0^{a} (1.9)	63.8^{b} (1.5)	73.4^{a} (2.5)
FEID:			
FALL			
number	17.7^{a} (1.94)	0.0^{b} (0.0)	11.1^{c1} (1.9)
height	42.7^{a} (1.3)	14.7^{b1} (0.7)	36.8^{a} (2.2)
SPRING			
number	ND	0.0^{a} (0.0)	2.3^{b1} (0.7)
height	ND	19.4^{a} (0.5)	26.2^{b1} (1.6)
CONTROL			
number	22.4^{a} (4.1)	0.0^{b} (0.0)	6.9^{c} (1.5)
height	39.6^{a} (2.4)	18.3^{b} (0.7)	39.0^{a} (2.9)

Means with different superscripted letters denote a significant difference between years within treatments; burn means with superscripted numbers denote a difference between control within years ($p<.10$).
ND = No Data.

For bluebunch wheatgrass, spring burning resulted in no significant change in number of flowering culms, but a significant decrease in maximum height (64.8 cm. before burning and 59.6 cm. the first post-fire year). Bluebunch wheatgrass plants in control units demonstrated significant reductions in number of flowering culms in both 1988 and 1989. Plant height decreased significantly the first year (1988), but recovered to 1987 levels the second year (Table 4). Although the number of culms were not significantly different in spring and control units the first post-fire year, mean height was significantly less for the spring burned plants.

There was no significant difference in the number of flowering culms on Idaho fescue plants in fall and control units prior to burning in 1987 (18 and 22 culms/plant, respectively) (Table 4). In 1988, no flowering occurred on any fescue plants in any treatment, thus fall burned plants recorded significant reductions in both number of flowering culms and plant height the first post-fire year. Although number of Idaho fescue culms had not recovered to pre-burn levels, height in 1989 was not significantly different than that prior to burning. The mean number of culms on fall burned fescue plants in 1989 was 11/plant, compared to 18 before burning; however fall plants had significantly more flowering culms than control plants in that year (7/plant).

Spring burning of Idaho fescue resulted in a significant increase in both mean number of flowering culms and maximum plant height. Flowering culms went from zero to a mean of 2.3/plant the year after burning; height went from 19 cm. to 26 cm. during the same period. One year after burning, however, both of these parameters were significantly less than plants in controls. Control plants demonstrated significant reductions in both culms and height between 1987 and 1988, with full recovery of height in 1989, but only moderate response of flowering culms, averaging 7 culms/plant in 1989, compared to 22/plant in 1987.

DISCUSSION

These findings are similar to other reports regarding fall burning of bluebunch wheatgrass, and further document this species resistance to fire-induced mortality. The spring burn, however, showed significantly lower mortality rates than that previously reported by Britton and Sneva (1981), who found 50% mortality following burning at a similar season to this study (mid-May). Bluebunch wheatgrass plants killed by the prescribed burns were smaller plants than the mean size of the population. The mean basal area of plants that were killed averaged 123 cm^2 and 110 cm^2 for fall and spring burns, respectively, compared to 198 and 180 cm^2 for all plants (Table 3).

The mortality of Idaho fescue reported here is among the lowest recorded for late season burns (i.e. late summer-early fall). Studies from the literature suggest that late summer appears to be the most sensitive period for this species (see Conrad and Poulton 1966). In contrast, lower mortality has been demonstrated following October burning after the plants have become dormant (Britton et al. 1983). The fall burn in this study was intermediate both in timing and reported mortality to that of Britton et al. 1983; however, the phenological stage of plants in the fall burn appeared more similar to their October treatment (1983). The spring burn resulted in similar findings to other work that showed Idaho fescue relatively undamaged following spring burning (Blaisdell et al. 1982). Like bluebunch wheatgrass, spring burning tended to kill smaller than average size plants (mean = 19 cm^2), but plants killed by the fall burn were larger (mean = 58 cm^2).

It is likely that factors associated with the two fire treatments are largely responsible for the differential mortality of Idaho fescue reported. The fuel and fire behavior characteristics associated with the two burns are summarized in Appendix 1. The fall burn resulted in roughly 7 times the fireline intensity, and twice the total energy as the spring burn.

Additionally, the presence of significant amounts of moisture in the herbaceous fuel during the spring burn likely dampened the heat flux into the root crown of Idaho fescue due to the thermal conductivity of water. Although the plants were at a relatively sensitive phenological stage, the physical characteristics of the fire were insufficient to cause significant mortality. Most individuals killed by the fall fire were located near sagebrush plants, and the greater consumption of these fuels was likely responsible for releasing sufficient heat to result in mortality. Plant morphology (and it's secondary effects on fuels) is the likely mechanism between the differential sensitivity to fire-induced mortality witnessed between the two species (See Conrad and Poulton 1966, Wright 1971). The fine-stemmed fescue transmits more heat flux to the bud zones than the coarse-stemmed wheatgrass. During severe fire behavior associated with the fall treatment, this morphological difference resulted in the higher mortality of Idaho fescue.

The data on plant density indicate that neither burn treatment resulted in significant losses in plant numbers. The plot data were not sensitive enough to detect the mortality measured on the tagged plants. In addition, the poor correlation of density with mortality (e.g. density of bluebunch wheatgrass increasing 15% following a spring burn that killed 5% of the population) is also a result of changes in stand structure detected by the sampling methodology. Since recruitment of either species was extremely rare throughout the study, changes in density could have been due to mortality, fire splitting one individual into two or more new "individuals", or sampling error. The mortality in control units was limited to one fescue plant that was dug up by a rodent.

Changes in basal cover and mean basal area give the greatest insight into fire effects on populations of these two species. Bluebunch wheatgrass appeared to respond well to fire, regardless of season. This is largely due to two adaptations that allow for expansion into areas opened up by the fire: tillering and fire-enhanced flowering. The vegetative growth resulting from tillering by new bud zones in the root crown was evident on most bluebunch wheatgrass plants that were burned. Fall burning stimulated increases in plant cover and mean basal area the first year following burning, with more dramatic increases occurring the second year. Spring burning resulted in no significant change in either cover or basal area over time, although minor decreases were noted (Tables 2 and 3). Plants in controls increased slightly over the same period in terms of cover and basal area, but the increases were not significant. In the years 1988 and 1989, there were no significant differences between bluebunch wheatgrass cover and basal area between spring and control treatments. These results are consistent with other studies that indicate moderate reductions in plant size following spring burning (e.g. Britton and Sneva 1981), however the duration of this suppression remains for future investigation.

In contrast, Idaho fescue responded less negatively to spring burning. Whereas fall burns resulted in a 44% reduction in cover and a 23% decrease in mean basal area the first post-fire year, plants in spring burn plots had an increase of 17% in cover and 12% in mean basal area one year after burning. However, these comparisons between spring and fall burns are somewhat misleading in that they occurred in different years. However, the increases in basal area of fescue after spring burning occurred over the same period that control plants demonstrated reductions in basal area, suggesting a treatment effect on this species. Clearly, of all species-treatment combinations, fall burning of Idaho fescue was the most damaging; however, the positive response during the second year after burning indicates that the population was resilient to fire disturbance. Data from future years will further our understanding of the autecological effects of both treatments in regard to population dynamics and stand structure of these two species.

Reproductive vigor of bluebunch wheatgrass was reduced by fall burning the first year after burning, but there were significant increases in flowering the second year. Flowering of control plants during the same period declined (Table 4), indicating significant treatment effects in regard to this adaptation. Spring burning apparently had little effect on flowering the first post-fire year, other than a height reduction of flowering culms.

Idaho fescue followed a similar trend in that fall burning resulted in significant decreases in flowering the first year after burning, however it seems unlikely that this was related to the fire, as plants in controls responded similarly. The response the second post-fire year was a significant increase in flowering over comparable plants in control units; but unlike wheatgrass, remained below pre-burn levels. Spring burning resulted in a significant reduction of culm numbers and plant height the first year after burning, as compared to control plants.

The future effects of fire and the resultant fire-enhanced flowering on the population dynamics of the perennial bunchgrasses is beyond the scope of this study. However, both ecological and physical changes brought about by the fire have created many unoccupied sites for new plant establishment. Spring burns, having only one year to recover in this study, may behave similarly to the fall burns and exhibit increased reproductive effort in 1990 (or later).

Since the two burn treatments in this study were implemented in different years, it was of little ecological value to compare them within a given year. However, it appears that not only did treatments respond differently over time, so did the effects of the treatments at any one year. Clearly, there exists a year effect; probably a manifestation of weather patterns that could explain some of this variability. In particular, precipitation amounts received in the previous year have been shown to dramatically influence vegetation in eastern Oregon (Sneva 1982). A summary of precipitation over the course of this study is given in Appendix 2. For instance, 1988 was an abnormally dry year, receiving only 77% of normal precipitation for the period of Oct. 1987 to Sept. 1988 (National Park Service 1989). The degree of this drought is considerably more pronounced if one looks at the 12 month period prior to the time of sampling in that year (August 1987 to July 1988), where only 60% of normal precipitation occurred. It could be postulated that not only was the reductions in plant vigor in controls a consequence of lack of available moisture, so were the decreases on plants in burned treatments. For example, reductions in flowering the first year Idaho fescue plants in fall units appears to be the result of a year effect (e.g. drought), and not related to the fire treatment itself.

In contrast, 1989 was moderately wet year receiving 114% of normal precipitation (Appendix 2). However, the three month period prior to sampling (March-May, 1989) was excessively wet, with over half the yearly total recorded during this time. Fall burned plants of both species recorded significant increases in basal area between 1988 and 1989, as did bluebunch wheatgrass plants in controls. Idaho fescue plants in controls exhibited no significant change in basal area. In contrast to the fall burn, spring burning resulted in no significant changes in basal area of either species, thus confounding interpretation of fire treatments during this period.

Consequently, with such interactions between treatments and time, more research is needed to definitively explain the causes of observed effects. For example, we are not sure whether the increase in basal area of Idaho fescue following spring burning as compared to decreases following fall burning can be attributed to differences between treatments (e.g. biomass consumption or energy release) or to environmental differences between 1988 and 1989. However, the results presented here permit a few generalizations:

1. Fall burning stimulated growth of bluebunch wheatgrass both the first and second year after burning.
2. Fall burning of Idaho fescue resulted in moderate (20%) mortality and reductions in plant size the first post-fire year with recovery of both size and reproductive vigor by the second post-fire year.
3. Fall burning induced increased reproductive effort for both species by the second post-fire year.
4. Spring burning had little effect on plant size of either species the first year after burning, although flowering was suppressed on Idaho fescue plants.
5. Proximity to sagebrush plants appears to be an important factor relating to plant mortality.
6. Independent of fire effects, reproductive effort is extremely variable from year to year for both species. Other parameters such as density, cover and mean basal area demonstrated no significant variability over time.

CONCLUSION

Initial post-fire changes in population structure of Idaho fescue and bluebunch wheatgrass indicate that both species are well adapted to surviving spring and fall fires. Mortality of both species was isolated to areas near fuel accumulations (e.g. under a sagebrush canopy), and surviving individuals exhibited enhanced vegetative growth from axillary buds located in the root crown (i.e. tillering). The fall burn stimulated growth of bluebunch wheatgrass while reducing the size of Idaho fescue plants, however this reduction appears to be temporary. The spring burn caused little change for either species.

Although reports in the literature suggest that Idaho fescue is very sensitive to fire, the ecotypes of this species at John Day Fossil Beds National Monument are apparently more resilient than previously believed. Fall burns of severe fire behavior, did not result in widespread damage to the population. No significant changes in density, cover, and mean basal area of Idaho fescue were measured. By the second post-fire year, increases of all stand parameters were recorded, including significantly greater flowering.

The ecological response of these two species to fire aids in the explanation of this ecosystem's dynamic response to fire disturbances. These two bunchgrasses comprise the dominant herbaceous component in the basin big sagebrush-dominated ecosystems of the John Day Fossil Beds National Monument, and are consequently critical components of the structure and function of the systems. The functional role of fire in the sagebrush region is to generally favor the herbaceous component over woody species. This paradigm is further substantiated by this study. The precise effects of fire on these species appear to be variable, and that the variability in historical fires (e.g. timing, consumption, energy) emulated in this study, appear largely responsible for this variability. It is clear that the variable response was a function of not only the fire itself, but of other variables such as weather.

From a management standpoint, burning in either season appears to fit into National Park Service policy of restoring native vegetation composition and structure. However, the lesser fire hazard associated with conducting spring burns may favor this season for conducting management-level prescribed fires.

CHAPTER 4

Response of Basin Big Sagebrush/Idaho Fescue-Bluebunch Wheatgrass Communities to Spring and Fall Prescribed Burning

ABSTRACT

The vegetation response to spring and fall prescribed burning was investigated in basin big sagebrush (*Artemisia tridentata* ssp. *tridentata* Nutt.)/Idaho fescue (*Festuca idahoensis* Elmer)-bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith) dominated communities at John Day Fossil Beds National Monument, OR. Fall-burned and control (no burn) communities were sampled before burning in 1987, and for two years following; spring-burned communities were sampled prior to burning in 1988, and one year later. Percent frequency and density of annual grasses were significantly reduced by both burn treatments. However, frequency and density of perennial grasses remained unchanged. In addition, density and cover of sagebrush and western juniper (*Juniperus occidentalis* Hook.) was reduced by 84-100%. An abundance of sagebrush seeds germinated in 1989 in both spring and control units following an abnormally wet spring, causing significant increases in percent frequency. However, no significant increase was detected in fall units. Results from greenhouse studies indicated that both burn treatments caused significant reductions in soil-stored seed populations of cheatgrass (*Bromus tectorum* L.), with reductions of 90 and 98% for fall and spring burning, respectively. Species richness in fall and control units decreased from 33 and 27 respectively, before burning in 1987, to 31 and 25 respectively,in the first post-fire year; however richness rose in both fall and control units to highest levels (37 and 30 respectively) in 1989. Species diversity indices (Shannon-Weaver Measure and Simpson's Diversity Index) followed a similar pattern in fall and control units. Spring units increased in richness (from 34 to 41), and decreased in evenness, resulting in no change in species diversity indices between 1988 and 1989. Presumably, these changes were a result of an interaction of fire effects and environmental factors, specifically precipitation in the previous year. Fire induced changes in community composition and structure toward a

dominance of herbaceous vegetation, particularly native perennial bunchgrasses such Idaho fescue and bluebunch wheatgrass. These changes indicate that fire may be a useful land management tool to achieve National Park Service policy objectives in regard to restoration of native vegetation composition and structure.

INTRODUCTION

It is accepted that fire played an important role in determining the vegetation composition and structure in the big sagebrush/perennial bunchgrass communities of the western United States prior to Euroamerican settlement (Daubenmire 1968, West 1979, Wright et al. 1979). Secondary succession following fire is influenced by many factors, including surviving species and by the proximity and mechanisms of invading species, degree of disturbance, and the various labile site factors (Gill 1978, West 1979). It is likely that at John Day Fossil Beds National Monument, historical grazing by domestic livestock, fire suppression, and invasion of alien species, have dramatically altered community composition and structure (Kauffman and Sapsis 1989). This study investigates the community response of this ecosystem to spring and fall prescribed burning, and the potential of using fire as a land management tool to achieve desired management goals pertinent to Park Service policy (Agee and Johnson 1988, Leopold et al. 1963).

Most investigations of fire effects in sagebrush/bunchgrass ecosystems have focused on response of particular species (e.g. Chapter 3, Britton et al. 1983, Conrad and Poulton 1966, Harniss and Murray 1973, Uresk et al. 1980, Wright 1971). These and other studies demonstrated that fire alters plant community composition late-seral dominance by woody species (i.e. sagebrush and juniper), to an early-seral vegetation structure dominated by herbaceous species. However, the exact nature and dynamics of secondary succession following fire is not clearly understood due to the highly variable response of specific populations. Further, variations in habitat type, ecological status, site factors, and burn variables (e.g. season) make comparisons between studies problematic (Bunting et al. 1987). For example, although a wildfire on a *A. tridentata* ssp. *tridentata* habitat type in eastern Washington reduced basal area of bluebunch wheatgrass the first year after burning(Uresk et al. 1980), prescribed burning a similar habitat type in eastern Oregon increased basal area the first post-fire year (Chapter 3).

Recent work has attempted to apply multivariate analyses to understand the influence of fire on vegetation change (Acker 1988, Humphrey 1984). Similarly, wildfire has recently received considerable attention as a factor of vegetation dynamics contributing to the maintenance of some vegetation types (Wright and Bailey 1982, Sousa 1984, White 1987), leading to the concept of "fire types".

The overall objective of this research was to quantify the community level response of basin big sagebrush dominated communities to spring and fall prescribed burning. Specifically, experiments were designed to answer the following questions: (1) what is the vegetation response to spring and fall burning in terms of changes in percent frequency?; (2) how do these changes compare to similar unburned control sites?; (3) what are the responses of annual grasses, dominant perennial grasses, and woody species in terms of plant density?; (4) how do the two burn treatments compare in terms of reducing soil-stored seed populations of annual grasses in adjacent degraded areas; (5) how is species diversity within plant communities affected by the burn treatments?; and (6) how do the sites resemble one another between treatments and years, in terms of abundance of co-occurring species?

METHODS

The study area was located on the Sheep Rock Unit of the John Day Fossil Beds National Monument (U.S.G.S. 15' series "Picture Gorge"; T 11 S, R 26 E, sections 32 and 32), approx. 10 km. west of Dayville, OR. Elevation ranges from 700 to 850 m, with annual precipitation at the nearby Visitors Center averaging 29 cm. Soils are moderately deep, very stony clay-loams of the Simas-Tub association (U.S. Soil Conservation Service, 1980). Aspects are north exposure, and range from 20 to 65% slope. Basin big sagebrush dominates the overstory, and Idaho fescue and bluebunch wheatgrass dominate the understory. Abundant perennial forbs include Yarrow (*Achillea millefolium* L.) and threadstalk milkvetch (*Astragalus fillipes* Torr.). Alien annual grasses, largely cheatgrass (*Bromus tectorum* L.), dominate areas not occupied by perennials. The study area has not been grazed by domestic livestock since the establishment of the National Monument in October, 1975.

The study was established in an unbalanced completely randomized design, with 7 replicate units of a fall burn treatment, 5 replicates of a spring burn treatment, and 3 controls (no burn) replicates. The fall burn treatment was conducted on Sept. 25, 1987; the spring burn was conducted on May 24, 1988. A detailed description of the fuel loads, weather, consumption, and fire behavior associated with the two prescribed burns can be found in Chapter 2. A summary of the fire treatment variables is given in Appendix 1.

Fall and control units were sampled in the summer of 1987 during June - August. Sampling in 1988 included spring units in April - May, and fall and control units in June. Sampling in 1989 of all units was done in June - July.

Vegetation sampling and plot layout was similar the that found in Bunting et. al. (1987), and used in a previous inventory of the monument (Youtie and Winward 1977). A typical plot layout in a depicted in Appendix 3. Plant community composition and relative abundance was measured in terms of relative frequency using 30x60 cm. quadrats as samples for presence/absence. Species frequency in each unit was measured along three 10 m transects by locating a quadrat every meter, (i.e. n=10 samples for each transect and 30 samples per unit). The transects were established systematically from a random origin in the northeast quadrant of each transect, with parallel transects 7 m. apart oriented upslope. Plant identification and nomenclature follows that of Hitchcock and Cronquist (1973), and Winward (1980).

Density of perennial grasses was also measured in the frequency plots. Individual Idaho fescue and bluebunch wheatgrass plants were counted in each 30x60 cm plot. Density of annual grass species was measured in a nested 5x5 cm plot.

Density of woody species was measured in five 1x15 m. belt transects in each treatment unit. Three of these transects were located adjacent to the frequency transects, extending an additional five m. upslope; two were oriented across slope, one at each end of the upslope transects. (Appendix 3).

The effect of burning on soil seed banks was investigated by collecting all material down to 2.54 cm depth of mineral soil in five 20x20 cm plots in paired burned and unburned areas of both burn treatments. The areas chosen for this study were adjacent flatlands in poor ecological status, where herbaceous cover was dominated by cheatgrass. The samples were then stratified at 0°C for 2 months, then planted in germination flats with equal amounts of vermiculite, and grown in the greenhouse under optimal conditions. The flats were sampled weekly for successful germinants, and after 6 weeks, totalled.

A Chi-square test with Yates' correction factor (Mueller-Dombois and Ellenburg 1974) was used to determine significant differences between post-burn years (observed frequencies), and pre-burn years (expected frequencies). When a species' frequency was zero, a value of 0.5 was used to avoid dividing by zero. No comparisons of frequencies between burned units and controls within a given year were possible due to unbalanced sampling between treatments.

The density data was analyzed with Analysis of Variance techniques for a completely randomized design (Peterson 1985), to test for changes over time. If the F-test was significant, a multiple comparison test (Scheffe') was used to detect which years were different from one another. Since the fires were implemented in different years it was not meaningful to directly compare fall and spring treatments within any given year. The greenhouse trials were analyzed comparing burned and unburned density using a Mann-Whitney rank sum test of independent pairs. All tests of significant difference were conducted at $p<.10$, with the exception of the greenhouse trials, which used $p<.05$.

The multivariate analyses of the communities used the species-relative frequency data for each treatment-by-year combination using transects as samples. Each treatment-year was measured for the following community composition parameters: (1) Richness; (2) Shannon-Weaver Measure (H'); (3) Simpson's Diversity Index (SDI); and (4) Pielou's Evenness Measure (J') (Peet 1974). Resemblance between the treatment-by-year groups was analyzed using Percent Similarity. This measure quantifies the degree of overlap of species co-occurring in two samples (Goodall 1978). These measures were calculated using the microcomputer based Analysis of Information and Diversity (AID1 and AIDN) programs developed at Oregon State University (W. S. Overton, B. G. Smith, and C. D. McIntyre, unpubl. paper, 1987).

RESULTS

Frequency

Alpha codes corresponding to species, are given with their common names in Appendix 4. A complete summary of the percent frequency of all species encountered in each treatment-by-year group is given in Appendix 5. A total of 59 species were recorded in the frequency data. However, relatively few demonstrated significant changes over time. Eight species recorded significant changes in the fall units, as compared to nine species in the spring units (Table 1). The control demonstrated relative stability in species frequency, with only 4 species displaying significant changes over time.

Neither burn treatment resulted in significant changes in frequency of perennial grasses. Fall burning reduced frequency of Idaho fescue from 58% before burning to 46% the first post-fire year, with recovery to 50% in 1989 (Appendix 5). Bluebunch wheatgrass declined slightly the first year following fall burning (from 23 to 21%), but recovered to above pre-burn levels in 1989. Spring burning of Idaho fescue resulted in a drop in frequency from 28% before burning to 23% in the first post-fire year, while bluebunch wheatgrass remained unchanged at 31% over the same period (Appendix 5).

In the first post-fire year, both burn treatments resulted in a significant decrease in frequency of cheatgrass, with the fall burn decrease somewhat greater (a net reduction of 34% in the fall units compared to 16% in the spring units) (Table 1). However, the reduction in the distribution of cheatgrass in fall burn treatments appeared to be temporary, since it returned to pre-burn levels by the second post-fire year. The fall burn also caused a significant decrease in rattlesnake brome (*Bromus brizaeformis* Fisch. & Mey), both years after treatment. The fall treatment resulted in a significant increase of Mariposa Lily (*Calachortus macrocarpus* Dougl.) was observed by the second post-fire year, increasing from 2% before burning in 1987, to 20% in 1989. A number of annual forbs increased significantly 2 years after fall burning, including whitlow-grass (*Draba verna* L.), threadleaf phacelia (*Phacelia linearis* (Pursh) Holz.), and Jim-Hill tumblemustard (*Sisymbrium altissimum* L.), while jagged chickweed (*Holosteum umbellatum* L.) initially decreased in the first post-fire year (1988), and recovered to above pre-burn levels the second year (Table 1).

In the spring burn treatment plots, softchess (*Bromus mollis* L.) was reduced as well as cheatgrass the first post-fire year. Among perennial forbs, yarrow increased from 46% frequency before burning in 1988, to 67% in 1989, while large fruit lomatium (*Lomatium macrocarpum*

Table 1. Mean plant frequency (%) of species with significant responses over time, by treatment and year.

FALL BURNED UNITS:

SPECIES	YEAR 1987	1988	1989
grasses:			
Bromus brizaeformis	22	2*	4*
Bromus tectorum	87	54*	84
perennial forbs:			
Calachortus macrocarpus	2	3	20*
Tragopogon dubious	9	12	3*
annual forbs:			
Draba verna	0	0	59*
Conyza canadensis	0	2	15*
Epilobium minutum	4	3	14*
Hollosteum umbellatum	45	11*	55*
Phacelia linearis	0	1	9*
Sisymbrium altissimum	1	1	27*

SPRING BURNED UNITS:

SPECIES	YEAR 1988	1989
grasses:		
Bromus mollis	27	3
Bromus tectorum	89	73
perennial forbs:		
Achillea millefolium	46	67
Lithophragma parviflora	24	0
Lomatium macrocarpum	15	1
Microseris troximoides	20	4
annual forbs:		
Holosteum umbellatum	33	72
Lactuca serriola	9	24
shrubs/trees:		
Artemisia tridentata	13	50

Table 1 (cont.)

CONTROL UNITS:

SPECIES	**YEAR** 1987	1988	1989
grasses:			
Bromus brizaeformis	10	0*	0*
perennial forbs:			
Achillea millefolium	2	17*	39*
Lomatium triternatum	0	17*	1
shrubs/trees:			
Artemisia tridentata	2	9	33*

*denotes a significant difference between expected frequency (1987), in fall and control units. All spring frequencies show significant differences between 1988 and 1989, ($p<.10$).

(Nutt.) Coult. & Rose), small-flowered fringecup (*Lithophragma parviflora* (Hook.) Nutt.), and false agoseris (*Microseris troximoides* Gray) declined (Table 1). Basin big sagebrush (owing to considerable numbers of new germinants) demonstrated a significant increase in plant frequency, going from 13% before burning, to 50% the first year after.

Rattlesnake grass, present in controls at a frequency of 10% in 1987, was absent in both 1988 and 1989. Yarrow demonstrated significant increases in controls throughout the study, which contrasts to the initial depression in the fall units measured the first post-fire year (Appendix 3). Nine-leaf lomatium (*Lomatium triternatum* (Pursh) Coult. & Rose), was significantly more abundant in control units in 1988 than either 1987 or 1989. Similar to the spring units, sagebrush frequency increased significantly from below 10% in 1987 and 1988 to 33% in 1989, due to new seedling germinants.

Density

The fall burn reduced the density of all annual grasses. For example, density of softchess and cheatgrass was 82 and $446/m^2$ respectively, in 1987, and 10 and $43/m^2$ respectively, in 1988 (Table 2). Both these species remained significantly less abundant the second post-fire year. However, density of cheatgrass was considerably higher ($169/m^2$) the second post-fire year. No perennial grasses had any significant change in density through time in any treatment.

Significant reductions in the density of softchess and cheatgrass were also measured in spring burn treatment units. Density of cheatgrass was $552/m^2$ and $85/m^2$ in 1988 and 1989 respectively, while softchess was reduced from $37/m^2$ to zero in 1989 (Table 2). Like the fall treatment, the spring burn resulted in no significant change in perennial grass density.

Both burn treatments resulted in high levels of mortality for the woody species (e.g. sagebrush, western juniper, shadscale (*Atriplex*

Table 2. Density of grasses (/m^2) by treatment and year. Numbers in parenthesis are the standard error of the mean.

TRMT/SPECIES CODE	YEAR* 1987	1988	1989
FALL:			
annuals			
BRBR	23 (13)	0 (0)	2.0 (2.0)
BRMO	82^a (28)	10^b (8)	0^b (0)
BRTE	446^a (75)	43^b (12)	169^b (53)
FEMI	1.9 (1.9)	11 (7)	1.9 (1.9)
perennials			
AGSP	2.2 (1.2)	2.2 (1.4)	1.9 (0.9)
FEID	19 (7)	17 (6)	17 (5)
SPRING:			
annuals			
BRBR	--**	3.0 (3.0)	0 (0)
BRMO	--	37^a (16)	0^b (0)
BRTE	--	552^a (169)	85^b (26)
FEMI	--	0 (0)	0 (0)
perennials			
AGSP	--	3.3 (1.2)	3.8 (1.3)
FEID	--	3.1 (0.9)	2.9 (0.8)
CONTROL:			
annuals			
BRBR	13 (9)	0 (0)	0 (0)
BRMO	160^a (87)	0^b (0)	0^b (0)
BRTE	524 (130)	662 (114)	476 (101)
FEMI	0 (0)	0 (0)	0 (0)
perennials			
AGSP	1.8 (2.5)	1.8 (3.0)	2.1 (2.2)
FEID	1.7 (0.6)	1.5 (0.6)	1.2 (0.6)

Different superscripted letters denote a significant difference between years ($p<.10$).

*1987 is pre-fire year for fall treatment; 1988 is pre-fire year for spring treatment.

**no data collected

confertifolia (Torr. & Frem.) Wats.), and snakeweed (*Gutierrezia sarothrae* (Pursh) Britt. & Rusby). These woody species were completely eliminated by the fall fire. Spring burning resulted in lower mortality rates: sagebrush was reduced by 84% (from 987/ha before burning to 133/ha one year after) and juniper by 100% (369/ha prior to burning) by the spring burn. Snakeweed numbers were reduced from 400/ha prior to burning, to 112/ha the first post-fire year, a decrease of 72%. In both burn treatments, density of rabbitbrush remained unchanged over time (Table 3).

Density of annual grasses in control units was relatively stable, with the exception of softchess, which disappeared after 1987 (Table 2). The perennial grasses and woody species remained relatively constant in control units over the three years, with no significant changes detected. Western juniper, however, did increase from 733/ha in 1988, to 1040 in 1989, an increase of 41%, due to a number of new seedlings that established.

Community Composition Parameters

The measure of species richness (s), or number of species observed in the frequency plots, decreased from 33 to 31 in the first post-fire year in the fall units, while increasing to 37 the second post-fire year (Table 4). The spring burn resulted in a pronounced increase from 34 species present prior to burning, to 41 in the first post-fire year. Control plots followed a similar trend, showing a modest depression between from 27 to 25, with a more sizable increase measured in 1989 ($s = 30$).

Species diversity (the Shannon-Weaver measure H') followed the same trends as richness in the fall and control units, with an initial decrease in heterogeneity between 1987 and 1988, followed by an increase in 1989 to over 1987 levels. Fall treatments went from an H' of 2.69 before burning, initially dropping to 2.53 in 1988, and increasing to 2.88 in 1989 (Table 4.) Although richness was increased

Table 3. Density of woody species (per hectare) by treatment and year. Numbers in parenthesis are the standard error of the mean.

TRMT/SPECIES CODE	YEAR* 1987	1988	1989
FALL:			
ARTR	3033[a] (2102)	0[b] (0)	0[b] (0)
ATCO	133 (348)	0 (0)	0 (0)
CHVI	33 (33)	33 (33)	33 (33)
GUSA	500 (988)	0 (0)	0 (0)
JUOC	456 (1108)	0 (0)	0 (0)
SPRING:			
ARTR	ND	987 (998)	133 (291)
ATCO	ND	0 (0)	0 (0)
CHVI	ND	240 (505)	240 (505)
GUSA	ND	400 (648)	112 (140)
JUOC	ND	369 (1048)	0 (0)
CONTROL:			
ARTR	1334 (298)	1334 (298)	1334 (298)
ATCO	667 (1564)	667 (1564)	667 (1564)
CHVI	44 (44)	44 (44)	44 (44)
GUSA	889 (545)	1121 (980)	946 (730)
JUOC	733 (1236)	733 (1236)	1040 (1127)

Different superscripted numbers denote significant differences between years ($p<.1$).

* 1987 is pre-burn year for fall treatment; 1988 is pre-burn year for spring treatment.
ND = No Data

Table 4. Community composition parameters by treatment and year. Parameters include species richness (s), Shannon-Weaver's Information Measure (H'), Simpson's Diversity Index (SDI), and Pielou's Evenness Measure (J').

	COMMUNITY PARAMETER			
TRMT/YEAR	s	H'	SDI	J'
FALL 1987	33	2.69	.90	.77
FALL 1988	31	2.53	.88	.74
FALL 1989	37	2.81	.92	.78
SPRING 1988	34	2.88	.92	.82
SPRING 1989	41	2.88	.92	.78
CONTROL 1987	27	2.61	.88	.79
CONTROL 1988	25	2.46	.87	.78
CONTROL 1989	30	2.64	.90	.78

in spring treatments, the spring units remained unchanged in regard to H'. For species diversity the control units followed the same pattern as the fall treatment units. There was a slight depression recorded between 1987 and 1988 (s = 27 and 25, respectively), with a more sizeable increase followed an identical pattern as H' in all treatment-by-year groups, with the range going from lowest diversity (SDI = 0.87) in control 1988 units, to highest diversity (SDI = 0.92) in fall 1989, spring 1988, and spring 1989 groups.

Changes in community evenness was similar to that of the two heterogeneity measures for the fall and control units, indicating that the changes in species diversity reflect both changes in richness and equitability (evenness). Thus, the decrease in fall treatment species diversity between 1987 and 1988 is not only due to fewer species in 1988, but also a decrease in evenness. Similarly, the increase in diversity recorded in fall units between 1988 and 1989 is apparently due to increases in both richness and evenness. The spring units, however, show an decrease in evenness in the first post-fire year (from J' = 0.82 to 0.78), indicating that, independent of species richness, the species present became less evenly distributed in terms of relative abundance (Table 4). Consequently, the increases in species diversity recorded in the spring burns must be attributed to the richness component, not the equitability component.

The Percent Similarity (PS) measure is a comparison of how the treatment-by-year groups compare with one another, based on the entire species-abundance assemblage (Goodall 1978). Based on the similarity means, no one treatment x year combination is more than 79% like any other group (Table 5). Relatively high dissimilarity existed both between treatments within years (e.g. fall and control 1989 had a similarity of 67%), and between years within treatments (fall 1988 and fall 1989 PS = 68%). The greatest similarity measured was between control units in 1988 and 1989 (79%), and the least similarity recorded

Table 5. Percent Similarity between treatment-by-year groups. Parameter is based on the relative abundance of coexisting species.

	FALL			SPRING		CONTROL		
	'87	'88	'89	'88	'89	'87	'88	'89
FALL:								
1987	100							
1988	78.4	100						
1989	72.5	68.1	100					
SPRING:								
1988	72.0	67.8	70.4	100				
1989	62.1	55.6	69.4	73.6	100			
CONTROL:								
1987	73.2	67.5	61.5	63.2	60.6	100		
1988	67.3	64.7	62.8	69.2	66.9	78.7	100	
1989	67.4	60.9	67.4	71.4	76.2	68.6	78.8	100

was between fall 1988 and spring 1989 (56%). (Table 5).

Greenhouse trials

Both burn treatments resulted in significant reductions in viable soil-seed populations. The fall burn resulted in a reduction of cheatgrass germinants from 4600 to 475/m^2, which amounted to a 90% reduction in viable seeds. The spring burn was even more successful at reducing cheatgrass seeds, with a reduction from 4025/m^2 in unburned samples to 100/m^2 in burned samples recorded, or 98% (Table 6). There was also significant reduction in density of two annual forbs (whitlow-grass, and jagged chickweed). The spring burn resulted in a significant reduction of fillaree (*Erodium cicutarium* (L.) L'Her) (Table 6).

DISCUSSION

The interpretation of the frequency data provided a good explanation how abundant species (e.g. cheatgrass, Idaho fescue, yarrow) responded to burning. However, frequency was not particularly robust in detecting significant changes in the less common, or rare species. Many species not present in the community prior to burning, were present either one or two post-fire years. Annual forbs appearing after fall burning include whitetop (*Cardaria draba* (L.) Hand.), horseweed (*Conyza canadensis* (L.) Cronq.), thyme-leaf spurge (*Euphorbia serpyllifolia* Pers.), pepperweed (*Lepidium perfoliatum* (Donn.) Howell), thread-leaf phacelia, miner's lettuce (*Montia perfoliata* (Donn.) Howell), white plectritis *macrocera* T. & G.) and lambsquarter (*Chenopodium album* L.). Similarly, although present in small numbers in 1987, fireweed (*Epilobium minutum* Lindl.) and Jim-Hill tumblemustard dramatically increased in 1989. Although there were 13 species found in post-fire years not initially present prior to burning the fall units, only 3 were found to be statistically significant.

Table 6. Greenhouse trials of burned and unburned soil and duff samples, by burn treatment. Data are mean number of germinants, by species, in paired burned and unburned samples (.04m^2), from fall and spring prescribed burns. Numbers in parenthesis are the standard errors of the means.

TRMT/SPECIES	TREATMENT	
	BURNED	**UNBURNED**
FALL:		
*Bromus tectorum**	19 (9)	184 (28)
*Draba verna**	8.2 (4.3)	95 (12)
Erodium cicutarium	0.6 (0.4)	3.8 (1.9)
*Holosteum umbellatum**	5.4 (4.7)	151 (26)
Taraxacum officinale	0.2 (0.2)	1.4 (0.6)
SPRING:		
*Bromus tectorum**	4.0 (3.0)	161 (50)
Draba verna	12.8 (5.2)	67 (24)
*Erodium cicutarium**	4.2 (2.6)	20.2 (3.9)
Holosteum umbellatum	0 (0)	2.0 (0.9)

* denotes a significant difference between burned and unburned treatments ($p<.05$).

Similarly, 14 new species appeared after spring burning, but they were all so rare that the tests were unable to establish significance. While these changes are not statistically significant, the magnitude of composition change is of ecological importance.

Consequently, these findings substantiate the likelihood for type II errors (i.e. declaring changes as not being statistically significant, when, in fact, they are) when analyzing the frequency of rare or uncommon species for vegetation change (see Smith et al. 1986, and Whysong and Brady 1987). Changes in sampling methods, using both more and larger frequency plots would likely result in a more sensitive analysis (Smith et al. 1986).

Fires resulted in conditions such that sites formerly occupied by annual grasses were occupied by pioneering annual forbs. The fall burn had very little effect on perennial grasses, but resulted in increases in some perennial forbs and decreases in others. For example yarrow, and Mariposa lily increased, while low pusseytoes (*Antennaria dimorpha* (Nutt.) T. & G.) and false agoseris declined. Most perennial forbs showed little change after fall burning (Appendix 5).

Vegetation response was similar on spring burn sites; decreases in annual grass frequency was accompanied by increases in annual forb frequency. Perennial grasses remained unaffected, while the perennial forb component responded variably. Yarrow and prickly lettuce demonstrated the greatest increases in the first post-fire year (approx. 20%), while many other species, including wooly groundsel (*Senecio canus* Hook.) and mullein (*Verbascum thapsus* L.) increased in lesser amounts (5 and 6% frequency) in 1989. Perennial forbs that decreased in abundance in spring burn plots included small-flowered fringecup, large-fruit lomatium, and false agoseris. It is possible that these changes are due to differences in sampling date, and hence species detectability. For example, early-season plants such as fringecup and whitlow-grass showed significantly higher numbers when sampled earlier (e.g. April 1988), as opposed to later in the season (June 1989).

Examination of the frequency data in the control units indicates species responses to other environmental, physical, and climatic effects besides the burn treatments (e.g. year effects). For example, although fall burning reduced rattlesnake brome from 22% in 1987 to 2% in 1988, this species also declined from 10% to absent in controls over the same period. Similarly, controls followed the same increasing trend in frequency of yarrow as that measured in both burn treatments. However, for some species the trend in controls was opposite that in the burned areas (e.g. *Sisymbrium*), clearly indicating a treatment effect. Caution of comparisons with control data should be taken, however, since frequency was based on only 90 sample quadrats. Thus changes that appear dramatic, such as large-fruit lomatium going from absent in 1987, to 17% in 1988, back to absent in 1989, may be a result of year effects, differences in sampling date, or sample error.

Large increases for many species in 1989 over previous years in controls (as well as burned treatments), is probably due to the abnormally high precipitation recorded in the previous year. While 1987 was average, and 1988 was some 40% below average, 1989 was 15% above normal for the previous 12 month period prior to sampling (Appendix 2). The spring of 1989 was particularly wet; the period of March through May of 1989 received 16.5 cm. of rain, or twice as much as normal for that period.

Patterns of sagebrush reinvasion following disturbance, and the effects of fire severity on reinvasion was apparent. Both spring and control units record significant increases in sagebrush abundance in 1989 (Table 3). This was due to the presence of numerous seedlings in these plots. The occurrence of sagebrush seedlings was much lower in the fall treatment (Appendix 5). The fall burn eliminated all sagebrush, while in the spring burn some individuals survived. New germinants in the spring burn may have originated from seeds produced by these survivors. However, most post-fire regeneration of big sagebrush has been reported to likely result from seeds stored in the soil (Mueggler 1956). Therefore, it seems also likely that differences in fire behavior between the two burns are responsible for greater soil-stored seed mortality in the fall burn. Specifically, the total heat load was twice as great in the fall burn (Appendix 1), and probably resulted in significant death of these seed populations.

Reestablishment of sagebrush following disturbance can be quite variable (Blaisdell 1953, Daubenmire 1975, Kuntz 1982, Young and Evans 1978), and dependent on the infrequent occurrence of conditions favorable for seedling establishment (Daubenmire 1975). This likely includes the timing and amount of precipitation following fire (Britton and Clark 1985). In addition to climatic and environmental conditions, it is apparent that the greater severity of the fall fire resulted in slower reinvasion of sagebrush compared to areas burned in the spring.

The density data clearly demonstrates the effect of the fire treatments on stand structure. Both treatments significantly reduced annual grass density, did not affect perennial grass density, and significantly decreased density of woody species. The overall structure of the community has been altered to that of dominance of herbaceous vegetation, particularly native perennial bunchgrasses (see Ch. 3). This was somewhat more pronounced in the fall burned units, as reflected in the frequency data. Survival of 16% of the sagebrush after the spring burn compared to complete elimination in fall treatments was largely a function of differences in burn variables (See Ch. 2). Specifically, flame length was less than half as long in the spring treatment, a result from lower fuel loads and higher live fuel moisture conditions associated with this treatment (Appendix 1).

Of particular interest is the total absence of western juniper in any post-burn unit. Both fires were effective in retarding succession to juniper woodland that is common in areas of prolonged fire absence (Kauffman and Sapsis 1989, Martin 1979). In unburned areas this species appeared to respond to the above-normal spring precipitation recorded in 1989 through a greater than 40% increase in density. This increase represents a high level of establishment of western juniper seedlings recorded in controls. No changes in rabbitbrush density were observed in any treatment.

Both burn treatments were successful at significantly reducing the number of viable cheatgrass seeds in litter and soil layers (Table 6). These findings are similar to that of Olson et. al. (1982) and Young et. al. (1976). However, survival of as little as 2% following spring burning still leaves an average of 100 viable caryopses per m^2. These reductions are unlikely to be sufficient to initiate change given that perennial plants are all but absent from these areas. A density of 2.5 perennial plants/m^2 is considered a requirement to preempt invasion by cheatgrass (Young and Evans 1978). The higher mortality of cheatgrass seeds in spring burned samples is similar to that reported by Pechanec and Hull (1945).

The analyses of community composition indicate that the greatest changes in community structure came from changes in the rare species. As evident in the frequency analysis, species richness increased following both burn treatments, although the increase is both quicker (occurred the first pos-fire year), and to a greater degree, following the spring burn (Table 4). The similar response of control units however, preclude inference of direct treatment effects on increasing alpha diversity.

Both heterogeneity measures, combining both the richness and evenness components, were similar among treatments, although change in H' is somewhat more pronounced than in SDI, given its greater weighting of rare species (Peek 1974, Whittaker 1965). In general, the heterogeneity measures followed the same pattern as richness, with diversity decreasing from 1987 to 1988, and increasing to the highest level in 1989. This occurred in both fall and control units and thus, was likely to be a result of year effects (i.e. precipitation) as opposed to fire-mediated effects. However, the spring units behaved somewhat differently, exhibiting no change in heterogeneity despite a pronounced change in species richness. This was a result the reduction in equitability in spring units shown in the reduction of J' between 1988 and 1989. Although there were more species present, species diversity did not change as these species are were less evenly distributed in relative abundance. Specifically, a great number of species in the spring 1989 plots were extremely rare, occurring in less than 2% of the quadrats, effectively reducing species evenness.

The species diversity of these sites are substantially higher than those reported for sagebrush stands in southeastern Idaho that were burned from 2 to 36 years previously (Humphrey 1984). This is apparently due to considerably greater evenness in this study, since the number of species observed was actually less. Five and six years after fire Acker (1988), reported that frequency data did not detect vegetation change in post-fire environments. Presumably, factors relating to on-site survival and establishment of new species in the near-term scale of this study, make frequency an adequate measure for interpreting immediate changes in composition resulting from fire.

The resemblance measure between the treatment-by-year groups indicate that there is a high degree heterogeneity in stand structure present throughout the landscape, and even stands in close proximity to one another were no greater than 80% similar. No pattern of similarity could be established, however, treatment differences tended to outweigh yearly differences. For example, fall units in 1988 were more similar to the fall 1987 units prior to burning (78%) than they were to the fall 1989 units (68%). This could mean that dramatic changes in post-fire succession occur in the second post-fire year, since the controls actually increased similarity between 1988 and 1989. Unfortunately, direct comparisons of burn treatments within years would be spurious due to different years of treatment application.

CONCLUSION

The effects of spring and fall prescribed burning on community composition and structure of the basin big sagebrush dominated communities at John Day Fossil Beds National Monument, shift these systems toward dominance by herbaceous species, predominantly perennial bunchgrasses. Annual grasses were reduced, and to some degree were replaced by annual forbs. Woody species were effectively reduced by both burn treatments, although the reduction was somewhat greater following fall burning. Forb abundance generally increased following both burn treatments, although similar trends were evident in controls. The first year following fall burning resulted in a decrease in species diversity, but by the second post-fire year diversity had increased to over pre-burn levels. Spring burning caused an increase in species richness in the first post-fire year, but a reduction in equitability resulted in no change in diversity measures. Control units demonstrated similar trends between years, indicating a strong year effect on vegetation structure and composition. Diversity patterns in both burned and unburned units followed patterns of precipitation, with greater moisture related to higher levels of species diversity.

Although the scope of this research only covers the initial (1-2 year) period following burning, the research has been designed for long-term monitoring analysis of community succession resulting from spring and fall burns. Thus the findings reported here are only the foundation for understanding the role fire plays in shaping the basin big sagebrush/Idaho fescue-bluebunch wheatgrass communities at the John Day Fossil Beds National Monument.

The change in community structure toward dominance by native perennial grasses initiated by both fire treatments indicate that fire may be an excellent land management tool for National Park Service personnel to achieve desired objectives in regard to vegetation on the monument. The autecological response detailed in Ch. 3., and the reduction of fire hazard documented in Ch. 2. provide additional information relevant to the use of prescribed fire as a vegetation management tool. Together with this analysis of community response, I believe the National Park Service now has the required information to plan and implement management level prescribed burns as part of its vegetation management programs.

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APPENDICES

Appendix 1. Summary of fire treatment characteristics. Data are from two experimental burns conducted at John Day Fossil Beds National Monument.

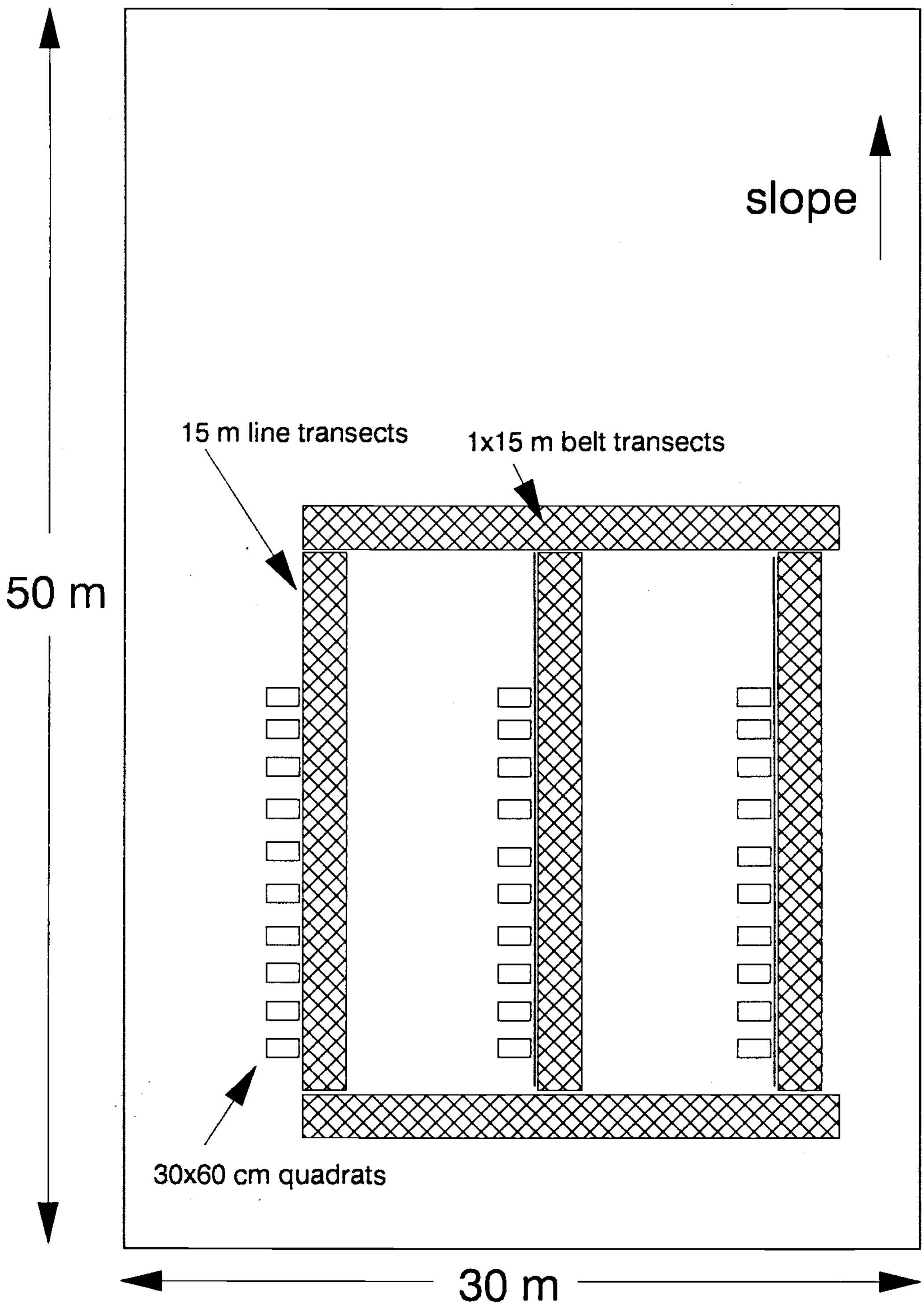
VARIABLE	FALL BURN	SPRING BURN
DATE	9/25/87	5/24/88
TOTAL FUEL LOAD (Mg/ha)	10.6	6.2
FINE FUEL LOAD (Mg/ha)	3.9	3.0
GRASS/HERB MOISTURE (%)	8.9	142.6
SAGEBRUSH FOLIAR MOISTURE (%)	97.2	186.0
FUEL CONSUMPTION (Mg/ha)	9.8	5.2
FUEL CONSUMPTION (%)	92.5	84.0
FLAME LENGTH (m)	4.1	1.7
FIRELINE INTENSITY (kW/M)	6,441	883
RATE OF SPREAD (m/s)	1.6	0.2
TOTAL HEAT LOAD (kJ/m^2)	18,119	9,267

Appendix 2. Precipitation (cm.) recorded at the John Day Fossil Beds National Monument Visitors Center, 1986-1989. Values for both the weather calendar year (Oct.-Sept.) and the 12 month period immediately prior to that year's sampling, and deviation (%) from average yearly precipitation, are presented.

	PERIOD		
	1986-87	1987-88	1988-89
Weather year	28.85	22.60	31.93
% deviation from mean*	-1.2	-22.6	+9.3
Prior 12 months from sampling date	28.16	17.40	33.43
% deviation from mean	-3.6	-40.4	+14.4

* mean annual precipitation for the period 1977-1986 was 29.21 cm.

Appendix 3. Typical plot layout for vegetation sampling.



Appendix 4. Scientific names, alpha codes used for abbreviation, and common names of species measured.

Scientific Name	Alpha Code	Common Name
grasses		
Agropyron spicatum	AGSP	bluebunch wheatgrass
Bromus brizaeformis	BRBR	rattlesnake grass
Bromus mollis	BRMO	softchess
Bromus tectorum	BRTE	cheatgrass
Festuca idahoensis	FEID	Idaho fescue
Festuca microstachys	FEMI	Nuttall's fescue
Koeleria cristata	KOCR	junegrass
Poa bulbosa	POBU	bulbous bluegrass
Poa pratensis	POPR	Kentucky bluegrass
Poa sandbergii	POSA	Sandberg's bluegrass
Sitanion hystrix	SIHY	squirreltail
Stipa comata	STCO	neddle and thread
perennial forbs		
Achillea millefolium	ACMI	western yarrow
Antennaria dimorpha	ANDI	low pussytoes
Astragalus filipes	ASFI	threadstalk milkvetch
Astragalus purshii	ASPU	woolypod milkvetch
Calachortus macrocarpus	CAMA	Mariposa lily
Cirsium vulgare	CIVU	bull thistle
Crepis occidentalis	CROC	western hawksbeard
Erigeron filifolius	ERFI	thread-leaf fleabane
Erigeron linearis	ERLI	line-leaf fleabane
Erigeron pumilus	ERPU	shaggy fleabane
Eriogonum strictum	ERST	strict buckwheat
Lactuca serriola	LASE	prickly lettuce
Lithophragma parviflora	LIPA	small-fld. fringecup
Lomatium macrocarpum	LOMA	large-fruit lomatium
Lomatium triternatum	LOTR	nine-leaf lomatium
Microseris troximoides	MITR	false agoseris
Phacelia hastata	PHMA	silverleaf phacelia
Sedum lanceolatum	SELA	lanceleaved sedum
Senecio canus	SECA	wooly groundsel
Sphaeralcea munroana	SPMU	Munro's globemallow
Tragopogon dubious	TRDU	yellow salsify
Verbascum thapsus	VETH	flannel mullein

Appendix 4 (cont.).

Species	Alpha code	Common Name
annual forbs		
Alysum allysoides	ALAL	pale alyssum
Amsinckia tessellata	AMTE	fiddleneck
Blepharipappus scaber	BLSC	blepharipappus
Cardaria draba	CADR	whitetop
Cerastium viscosum	CEVI	sticky chickweed
Chenopodium album	CHAL	lambsquarter
Collinsia parviflora	COPA	small-fld. blue-eyed mary
Conyza canadensis	COCA	horseweed
Descurainia pinnata	DEPI	tanseymustard
Draba verna	DRVE	whitlow-grass
Epilobium minutum	EPMI	small-fld. willow-herb
Euphorbia serpyllifolia	EUSE	thyme-leaf spurge
Galium aparine	GAAP	bedstraw
Helianthus anuus	HEAN	sunflower
Holosteum umbellatum	HOUM	jagged chickweed
Lagophylla ramossisima	LARA	slender rabbitleaf
Lepidium perfoliatum	LEPE	clasping pepperweed
Montia perfoliata	MOPE	miner's lettuce
Phacelia linearis	PHLI	thread-leaf phacelia
Plectritis macrocera	PLMA	white plectritis
Sisymbrium altissimum	SIAL	Jim-Hill tumblemustard
Taraxacum officinale	TAOF	dandelion
shrubs/trees		
Artemisia tridentata	ARTR	big sagebrush
Atriplex confertifolia	ATCO	shadscale
Chrysothamnus viscidiflorus	CHVI	green rabbitbrush
Gutierrezia sarothrae	GUSA	broom snakeweed
Juniperus occidentalis	JUOC	western juniper

Appendix 5. Plant community composition as expressed by plant frequency (%), by treatment and year. A "T" denotes trace, or less than 2% frequency.

FALL BURNED UNITS:

	YEAR		
Species code	**1987**	**1988**	**1989**
annual grasses			
BRBR	22	2	4
BRMO	24	11	11
BRTE	87	54	84
FEMI	6	4	7
perennial grasses			
AGSP	23	21	24
FEID	58	46	50
KOCR	3	7	2
POPR	T	T	T
POSA	82	85	83
SIHY	T	0	0
STCO	0	0	T
annual forbs			
ALAL	T	4	0
CADR	0	0	T
CEVI	0	0	2
CHAL	0	0	T
COCA	0	2	15
DEPI	0	0	3
DRVE	0	0	59
EPMI	4	3	13
HOUM	45	11	55
LARA	T	0	0
LEPE	0	0	7
MOPE	0	T	T
PHLI	0	T	9
PLMA	0	0	3
SIAL	T	T	27
perennial forbs			
ACMI	38	29	50
ANDI	13	4	4

Appendix 5 (cont.).

FALL BURNED UNITS:

Species code	YEAR 1987	1988	1989
perennial forbs (cont.)			
ASFI	13	16	12
CAMA	2	3	20
ERFI	T	T	0
ERLI	9	7	10
ERPU	2	T	3
ERST	0	T	T
LASE	7	T	3
LOMA	2	4	3
LOTR	T	T	0
MITR	12	5	7
SECA	0	0	2
SELA	7	T	3
SPMU	0	0	2
TRDU	9	12	3
shrubs/trees			
ARTR	5	T	T
ATCO	T	0	0
CHVI	T	0	0
GUSA	5	0	0
JUOC	4	0	0

SPRING BURNED UNITS:

Species code	YEAR 1988	1989
annual grasses		
BRBR	3	T
BRMO	27	3
BRTE	89	73
FEMI	3	3
perennial grasses		
AGSP	31	31
FEID	28	23
POBU	T	2
POPR	T	0
POSA	79	80

Appendix 5 (cont.).

SPRING BURNED UNITS (cont.):

Species code	YEAR 1988	YEAR 1989
annual forbs		
ALAL	3	0
AMTE	2	T
BLSC	0	2
CADR	0	2
CEVI	2	4
COPA	T	0
COCA	0	3
DRVE	35	36
EPMI	9	13
EUSE	0	3
GAAP	3	2
HEAN	0	T
HOUM	33	72
LEPE	0	3
PHLI	0	3
PLMA	9	5
SIAL	0	5
perennial forbs		
ACMI	46	67
ANDI	3	2
ASFI	5	4
ASPU	0	T
CAMA	3	T
CIVU	0	2
CROC	10	9
ERLI	T	0
ERPU	0	2
ERST	T	0
LASE	9	24
LIPA	24	0
LOMA	15	7
LOTR	15	11
MITR	20	4

Appendix 5 (cont.).

SPRING UNITS (cont):

	YEAR	
Species code	**1987**	**1988**
perennial forbs (cont.)		
PHHA	0	T
SELA	2	3
SECA	0	5
TRDU	15	9
VETH	0	6
shrubs/trees		
ARTR	13	50
GUSA	3	11

CONTROL UNITS:

		YEAR	
Species code	**1987**	**1988**	**1989**
annual grasses			
BRBR	10	0	0
BRMO	4	0	2
BRTE	97	100	97
FEMI	3	2	8
perennial grasses			
AGSP	27	20	26
FEID	19	17	13
KOCR	9	6	6
POPR	T	0	T
POSA	82	84	60
annual forbs			
ALAL	4	0	0
CEVI	0	3	6
DRVE	0	13	24
EPMI	16	13	34
HOUM	34	43	50
LARA	0	0	T
LEPE	0	0	2

Appendix 5 (cont.).

CONTROL UNITS (cont.):

Species code	YEAR 1987	1988	1989
annual forbs (cont.)			
PHLI	0	2	2
PLMA	0	10	6
SIAL	6	0	0
perennial forbs			
ACMI	12	17	39
ANDI	8	T	2
ASFI	7	8	10
CROC	0	0	T
ERFI	14	7	4
LASE	11	4	8
LOMA	6	T	0
LOTR	0	17	T
MITR	4	3	T
SELA	13	0	T
SECA	T	0	0
TRDU	6	13	19
shrubs/trees			
ARTR	2	9	33
ATCO	2	T	T
GUSA	6	T	2
JUOC	2	2	2