CHEATGRASS DYNAMICS FOLLOWING WILDFIRE ON A SAGEBRUSH SEMIDESERT SITE IN CENTRAL UTAH

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ABSTRACT

Most vegetation studies tend to be short term and determine the status of communities at a single instance in time. Year-to-year and longer term environmental fluctuations, together with internally driven changes in vegetation, require studies conducted over periods of decades for a full understanding of the vegetation dynamics. This study considers 11 years of data from plots with burnt/ungrazed, burnt/grazed, and unburnt/grazed histories at an initially relict sagebrush steppe site near Mills in central Utah. Cover data were collected near the end of each growing season using a gimbeled point technique.

All burnt plots showed a complete reduction in sagebrush cover, and increased cover by native perennial bunchgrasses (on a percentage basis). All monitored plots showed considerable increase in cheatgrass (Bromus tectorum) *during the initial 2 years, followed by a 2-year period of rapid reduction in cover. The ensuing 3 years show considerable year-to-year fluctuation in cover by cheatgrass. The final years of monitoring show a negligible presence of cheatgrass. These results deviate from commonly held notions that the presence of cheatgrass dictates irreversible change resulting in continued cheatgrass dominance. All treatment combinations involving burning showed similar, higher magnitudes of cheatgrass cover regardless of livestock grazing or not, negating conclusions derived from other, shorter term studies.*

INTRODUCTION

Numerous papers have recorded the preponderance of cheatgrass (*Bromus tectorum*) in the Great Basin over the past decades (Billings 1990; Klemmedson and Smith 1964; Mack 1981; Morrow and Stahlman 1984; Tisdale and Hironaka 1981). The spread of cheatgrass has been associated with livestock disturbance, although studies of relict areas indicate *Bromus* spp. presence (and sometimes dominance) within undisturbed areas (Hunter 1991; Passey and others 1982; Svejcar and Tausch 1991; Tisdale and others 1965). Similar trends are observed within rangelands that have been rested for a long time (Anderson and Inouye 1989; Daubenmire 1940; Harris 1967).

It has become accepted that fire plays an important role in conversion of sagebrush-steppe to annual grasslands. Livestock grazing may be an important management tool for the maintenance of bunch grasses by preventing the accumulation of fire loads (Young and Evans 1978), thus preventing earlier and more frequent fires that are so detrimental to the native perennials (Hunter 1991; Whisenant 1990).

It is also generally assumed that cheatgrass invasion results in the degradation of sagebrush-steppe to a new stable state, and that only direct intervention is capable of returning this degraded system to a semblance of its former self (Miller and others, in press; Young and Evans 1978). This may be the case, but few long-term studies provide data to validate these notions.

One such long-term study is ongoing at the Idaho National Engineering Laboratory research sites in south-central Idaho. Anderson and Inouye (1989) discuss the establishment of dense stands of cheatgrass since the initial appearance of cheatgrass in 1975. The authors note that cheatgrass establishment occurred in the absence of fire and grazing, and during a period of higher rainfall (1966-75). A subsequent decrease during drier years implies a dependence on rainfall trends.

Secondary succession within abandoned fields first examined by Piemeisel (1951) and Hironaka and Tisdale (1963) showed several stages of development. Cheatgrass dominated sites after a few years, but was followed by dominance by bottlebrush squirreltail (*Sitanian hystrix*), and the gradual spread of perennial herbs and Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) (Hunter 1991).

One of the few studies examining postfire community dynamics within Wyoming big sagebrush-dominated vegetation was conducted in southeastern Idaho (Humphrey 1984). Eight environmentally similar sites were burned at different times over a period of 36 years, resulting in a series of communities representative of vegetation development following fire (Humphrey 1984). Data were collected at one instance in time. Initial establishment after disturbance favored perennial grasses and forbs able to resprout following fire. Cheatgrass showed a peak of approximately 11 percent relative cover 3 years after the fire event. This declined to 1-2 percent by the sixth year following the burn. Shrubs able to resprout became prevalent after 6 years, whereas shrubs establishing from seed required longer intervals.

Few long-term studies have been conducted examining community changes at a single site following fire or the advent of grazing, let alone their interaction. More needs to be known of the community changes occurring following

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Paper presented at the Symposium on Ecology, Management, and Restoration of Intermountain Annual Rangelands, Boise, ID, May 18-22, 1992.

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these disturbances if we are to ably manage our rangelands. This study is aimed at increasing our understanding of community dynamics within burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments.

STUDY SITE

The study site is located near Mills in central Utah (Section 30, R2W, T15S, Salt Lake Meridian), Juab County, on a pediment remnant. Slopes are 1-2 percent east facing, and elevation ranges from 1,617 to 1,622 m (5,305-5,320 ft). For further details see West and Hassan (1985).

METHODS AND MATERIALS

A wildfire swept through the study site shortly after collection of the "pretreatment" data in 1981. Patchy burning of the site resulted in a natural experiment allowing the investigation of changes in burnt and unburnt patches under grazed and ungrazed conditions. Figure 1 summarizes the history of this study. Pretreatment data were collected on four macroplots (20 by 50 m) in 1981. The plots were fenced after the burn, but prior to the second season of data collection. Additional plots (three per treatment) were selected to represent the burnt-grazed and unburnt-grazed treatments. Livestock grazing was initiated after the establishment of watering points prior to the 1982 data collection season. The site may have supported light winter grazing by sheep before the start of the study. A second light wildfire occurred during 1987. One replicate of the burnt-grazed treatment was inadvertently lost due to chaining prior to the 1988 data collection season.

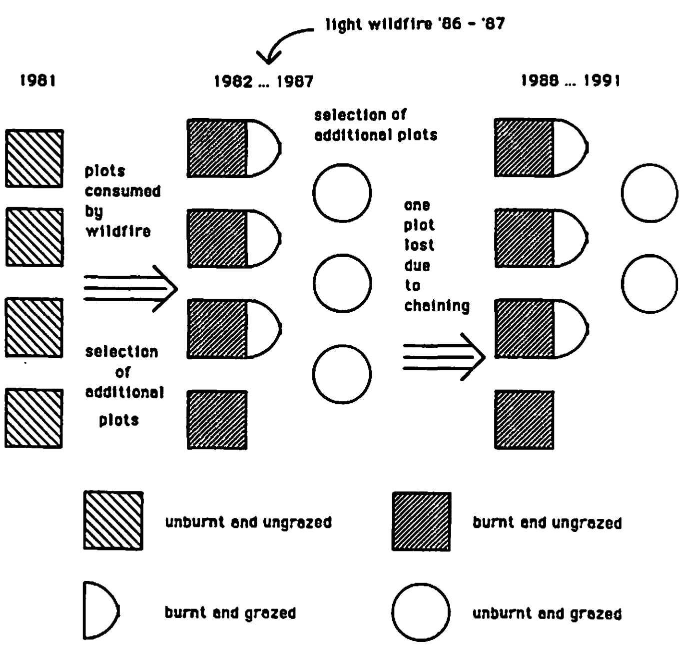

Figure 1—Graphical representation of the experimental design.

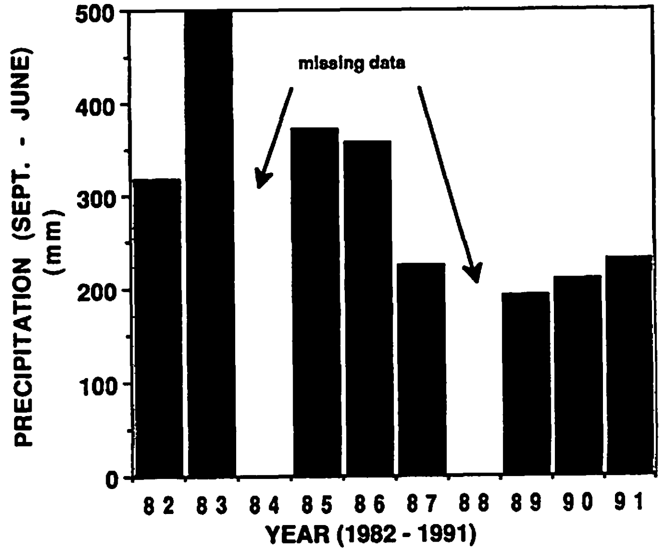

Figure 2—Precipitation records for the Mills study site (September-June; 1981-92).

Data were collected using a gimbeled point technique (Winkworth and Goodall 1962). One hundred points were sampled per macroplot, and interpreted as percentage cover. Data are presented as means with vertical bars representing the range of values for the treatments.

PRECIPITATON

Figure 2 indicates that the years 1987 to 1991 were considerably drier than the preceding years. Nineteen eighty-three appears to have been the wettest year according to existing data.

PERCENTAGE LIVE PLANT COVER

No clear differences in total percentage live plant cover occurred between treatments (fig. 3), although the burnt and ungrazed (BU) treatment generally shows a higher percentage live plant cover than the burnt and grazed (BG) and unburnt and grazed (UG) treatments.

GROWTH FORMS

Four growth forms are considered in this section. Two natural groups are shrubs and perennial grasses. Half-shrubs represent a less distinct group. Since forbs comprised a small proportion of the plant community in this study, perennial and annual forbs are considered as a whole within the forb class. Cheatgrass represents an annual grass functional group, but is considered as an individual species.

Figure 4 (tree and shrub cover) shows only slight fluctuations in shrub cover within the UG treatment. Shrubs show no recovery in either the BU or BG treatments following the 1981 wildfire.

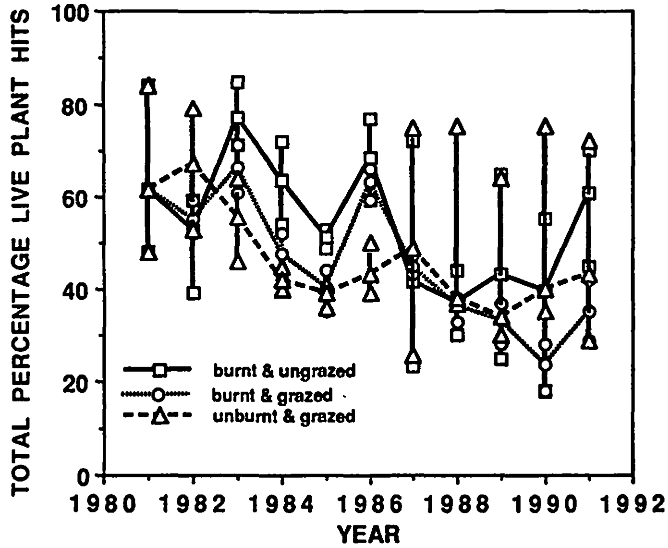

Figure 3—Total percentage live plant hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

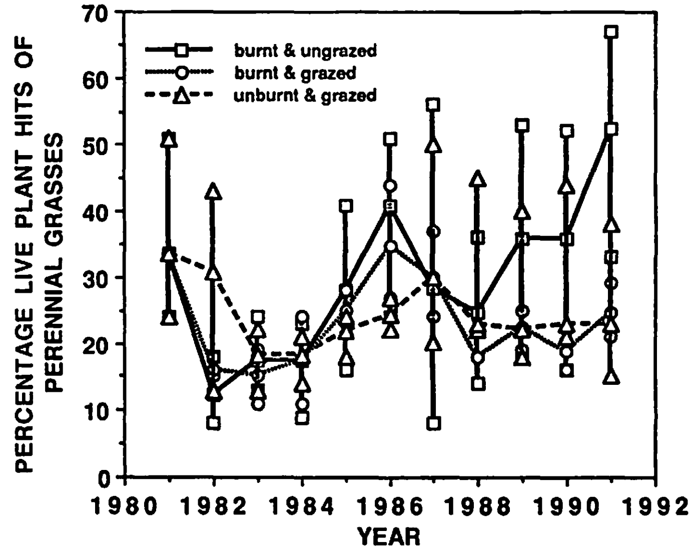

Figure 5—Percentage live perennial grass hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

Percentage cover by perennial grasses (fig. 5) shows considerable covariation. A large difference in mean percentage cover by perennial grasses between the BU and the BG and UG treatments developed from 1989 to present.

Percentage cover by forbs (fig. 6) fluctuates considerably on an annual basis within its range (approximately 0-8 percent live plant hits). No single treatment appears to have a significantly different percentage cover by forbs. During 1982 the UG treatment showed the highest percentage live plant cover. During the final 2 years of study, the BG treatment showed relatively higher percentage forb cover than the BU and UG treatments.

Of note is the complete absence from all treatments of half-shrubs (fig. 7) during the initial years of the study. During the final 7 years, half-shrubs show considerable

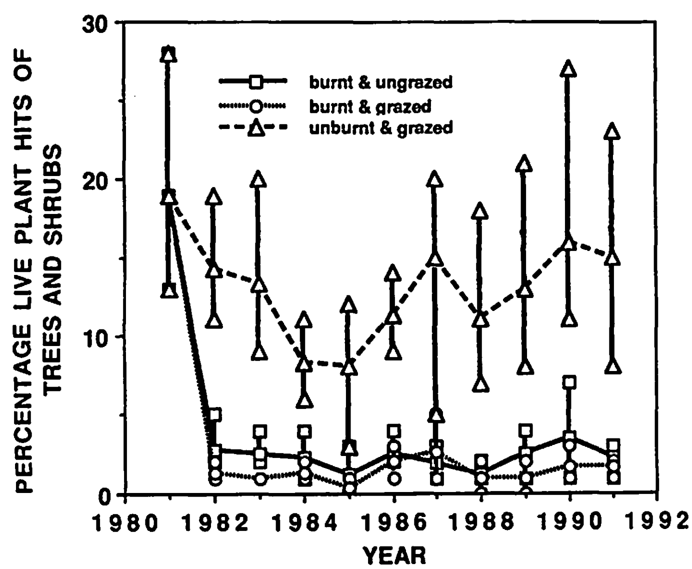

Figure 4—Percentage live tree and shrub hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

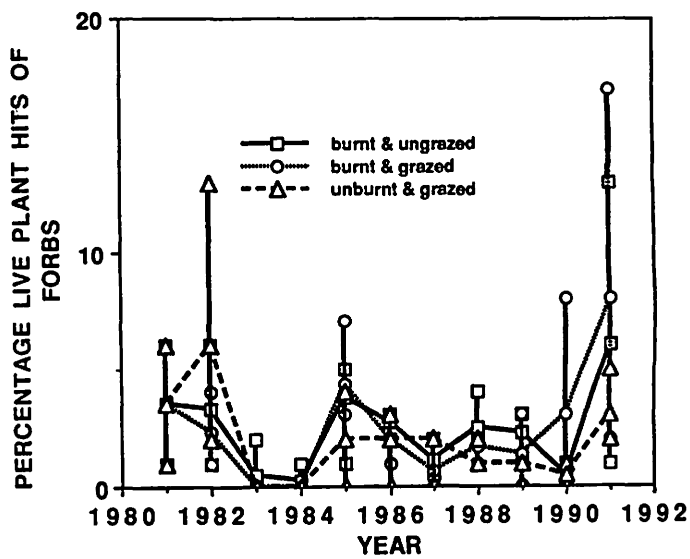

Figure 6—Percentage live forb hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

annual variation within the range of occurrence (approximately 0-3 percent live plant hits) in percentage cover for all treatments. The UG treatment shows higher abundance of half-shrubs during 1990 and 1991 than the other treatment combinations.

INDIVIDUAL SPECIES

Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) (fig. 8) is the predominant shrub within the study site, and thus follows the interpretation for the shrub functional group closely. Sagebrush cover is maintained within the UG treatment (approximately 4-9 percent live plant hits), but completely absent within the BG and BU treatments.

Bearded bluebunch wheatgrass (*Agropyron spicatum*) (fig. 9) is the dominant perennial grass, and thus follows the interpretation for the perennial grass functional group most closely. The only distinct difference between the long-term trends for the different treatments occurs during the latter 3 years of this study. The percentage cover of bearded bluebunch wheatgrass for the BU treatment increases to a peak (approximately 30 percent live plant hits) in 1991, whereas cover for the BG and UG treatments remains constant (approximately 13 percent live plant hits).

The percentages of live cover for Indian ricegrass (*Oryzopsis hymenoides*) (fig. 10) for the BU and BG treatments covary and are of the same magnitude during the initial 9 years of the study, but both are higher than for the UG treatment. The UG treatment generally shows the lowest percentage live plant cover of all treatment combinations from 1983 to present. The percentage cover by Indian ricegrass within the BG treatment shows an increase (approximately 7-14 percent live plant hits) from 1989 to present, whereas cover within the BU and UG treatments appears steady (approximately 5 and 3 percent live plant hits respectively) within the same time period.

Bluegrass (*Poa* spp.) (fig. 11), bottlebrush squirreltail (*Sitanian hystrix*) (fig. 12) and needlegrass (*Stipa* spp.) (fig. 13) generally constitute less than 5 percent live plant cover and are characterized by a lack of distinct long-term trends. During the latter 5 years of the study, the highest abundances of bluegrass occurred within the UG treatment, the lowest within the BG treatment, whereas the BU treatment appears intermediate. Percentage live plant

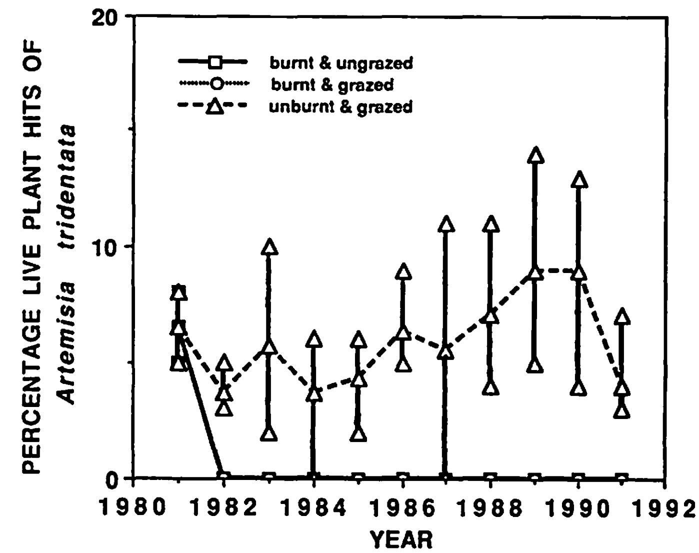

Figure 8—Percentage live Wyoming big sagebrush hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

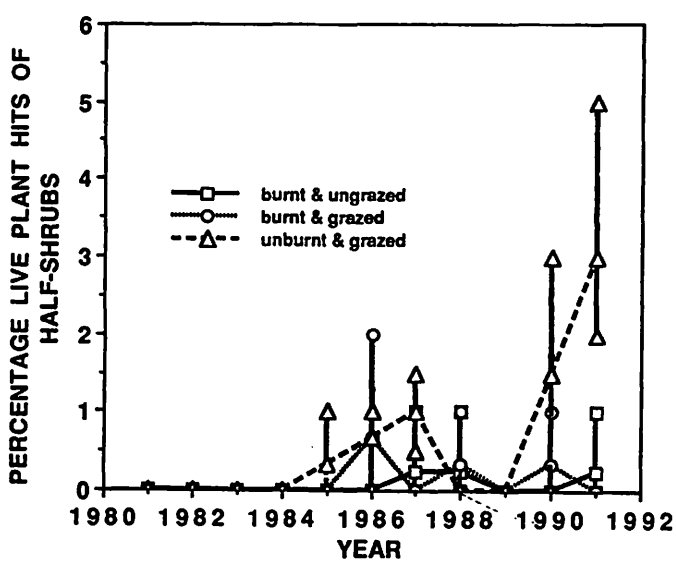

Figure 7—Percentage live half-shrub hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

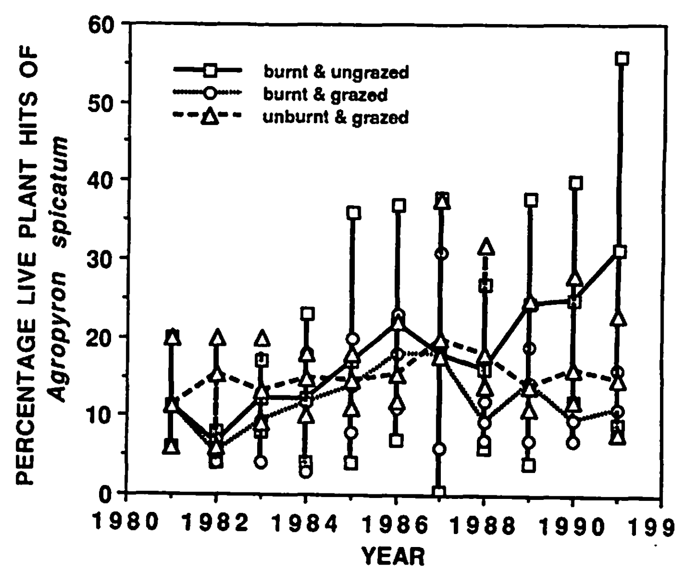

Figure 9—Percentage live bearded bluebunch wheatgrass hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

cover is consistently less in the BU treatment for needlegrass during the last 7 years. No distinct differences exist between treatments for bottlebrush squirreltail.

Percentage live cover of cheatgrass (fig. 14) is the most dynamic of all those presented within this paper. All treatments show a dramatic increase in cover following wildfire and cattle disturbance, with a maximum during the second year following the initial wildfire. The covariation of percentage cover is similar for all three treatments. The magnitudes for the BU and BG treatments are similar (approximately 50 percent live plant hits) and considerably larger than that for the UG treatment (approximately 25 percent live plant hits). All treatments show a negligible percentage cover by cheatgrass during the latter 2 years of investigation (1990 and 1991). The rate of decline of percentage cover following 1983 is lower than the rate of increase to the year of peak cover.

Russian thistle (*Salsola kali*) was first detected (approximately 1 percent live plant hits) in 1989 within the BG treatment and increased to approximately 13 percent during the final year of investigation (fig. 15). Russian thistle was noted within the BU treatment in 1990 and increased to approximately 6 percent in 1991. Russian thistle was first monitored (approximately 1 percent live plant hits) within the UG treatment in 1991.

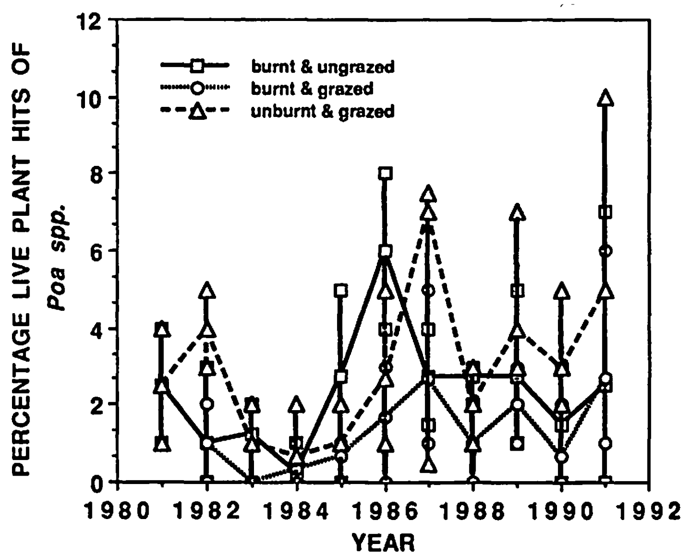

Figure 11—Percentage live bluegrass hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

DISCUSSION

The observed trend of cheatgrass establishment, invasion, and die-back may be unusual, or it may reflect the lack of long-term studies monitoring community change in sagebrush semidesert. The last 4 years of this study have shown lower than average rainfall. This may have played a part in the decrease of cheatgrass, since establishment of cheatgrass has been linked to high rainfall years. The reestablishment of perennial grasses is due to the continued presence of vegetatively produced culms following the 1981 wildfire. The lack of a serious wildfire since 1981 (perhaps due to a low fire load as a result of the drought years and heavy grazing) may have been important in the maintenance of the perennial grasses. These factors all need to be taken into account when interpreting data such as these.

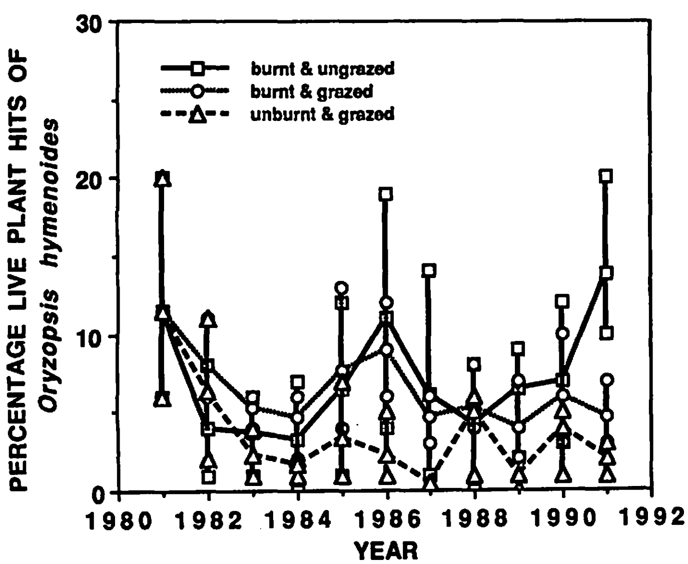

Figure 10—Percentage live Indian ricegrass hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

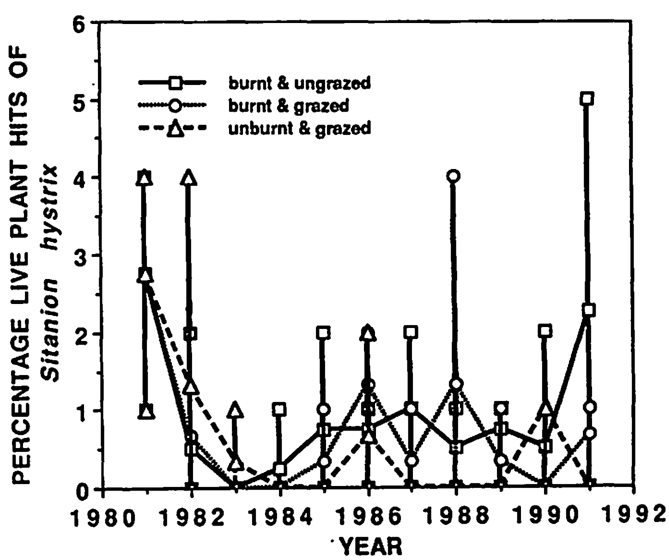

Figure 12—Percentage live bottlebrush squirreltail hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

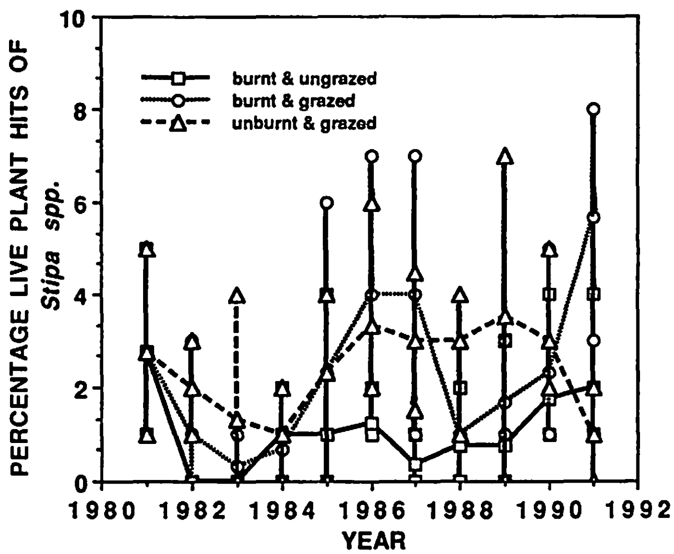

Figure 13—Percentage live needlegrass hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

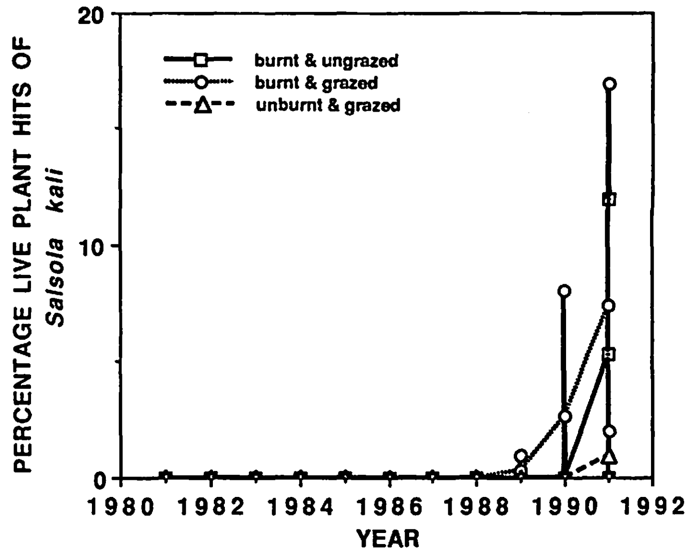

Figure 15—Percentage live Russian thistle hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

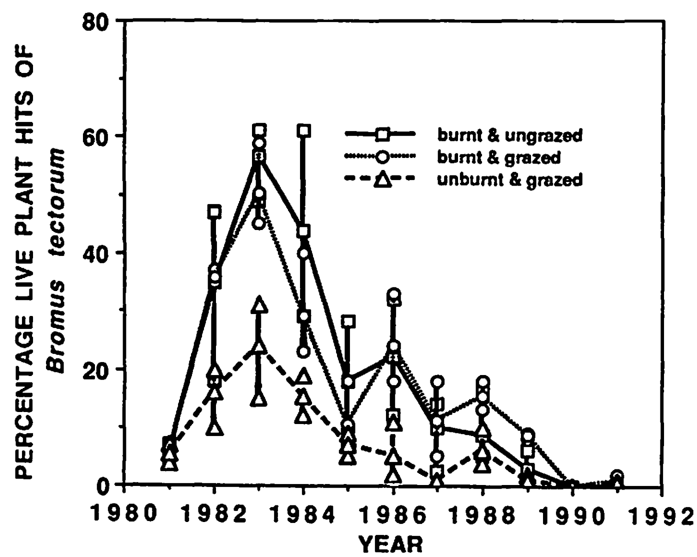

Figure 14—Percentage live cheatgrass hits for burnt and ungrazed, burnt and grazed, and unburnt and grazed treatments (1981-91).

The similar trends and large magnitudes of cheatgrass percentage cover for the BU and BG treatments relative to the UG treatment inexorably link cheatgrass impact with fire events. The larger magnitudes of cheatgrass within treatments encompassing a burn is understandable within the context of fire being regarded as an acute disturbance (total removal of foliage at one instance in time), whereas grazing is a chronic disturbance (partial removal of foliage over a longer time period). The decline of cheatgrass for all treatment combinations in the face of heavy grazing is of considerable interest in view of presently held notions. It is interesting to note that the decline of cheatgrass within the UG treatment (chronic disturbance only) to minimum levels preceded that for the treatments encompassing a burn (acute disturbance).

The lack of clear difference between the BU and BG treatments signifies that cattle grazing does not have a clear impact on vegetation dynamics following a burn. However, this must be interpreted within the context of the community—for example, the long-term trends of perennial grasses and sagebrush also need to be considered.

Let us consider some of the changes that have occurred within other growth forms and individual species.

The removal of sagebrush within the treatments encompassing a burn is normal. Lack of regeneration may be due to unsuitable conditions for germination and persistence of seedlings. Sagebrush establishment may depend on the co-occurrence of specific requirements, resulting in a cohort effect. The slight decline in sagebrush within the UG treatment may be ascribed to natural mortality, or partial die-back in individual plots as a result of cattle disturbance.

The apparent difference in percentage cover of perennial grasses (primarily bearded bluebunch wheatgrass) between the BU and other treatments (BG and UG) may be for several reasons. The higher cover by perennial grasses may simply reflect a differential consumption or an accumulation of litter (although care was taken to differentiate between present year's and previous years' growth); or reflect differential reestablishment (primarily vegetative). The establishment of bottlebrush squirreltail (a shorter lived perennial grass) appears similarly enhanced by wildfire, but remains relatively unaffected by grazing. Clearly, the lack of grazing following the wildfire appeared beneficial in terms of establishing higher cover by perennial grasses.

Forbs are difficult to monitor on a cover basis within the sagebrush-steppe system because of their naturally low cover. The relatively low cover by forbs within the BU treatment for the last 2 years may be indicative of a condition commonly observed within vegetation exclosures—a trend toward complete lack of shorter lived plants (Tueller and Tower 1979). The establishment of Russian thistle within some of the macroplots may herald future changes.

Some caution should be taken when interpreting results from the macroplots to larger spatial scales. It should be remembered that the macroplots are embedded within a larger burnt and heavily grazed system. It is likely that this has influenced vegetation dynamics within the macroplots studied. For example, there may have been a mass effect (Hatton and Carpenter 1986) that resulted in a higher presence of cheatgrass within the UG treatment than would have occurred within a complete system that remained unburnt. Past history of a site may only become apparent many years after the occurrence or initiation (in the case of grazing) of a disturbance event. This is illustrated by the fact that several figures show divergence of long-term trends between treatments within the latter years of the study.

This paper illustrates some of the problems involved with initiating and maintaining long-term studies. It is often luck that guides the ecologist to the right place at the right time (in this case, immediately prior to a wildfire). Considerable foresight is required to envisage beneficial experimental designs. Interpretation of results is often difficult because of the many perspectives possible (individual species, populations, communities, growth forms). This means that the objectives of a study need to be clearly stated. Maintenance of long-term studies is aided if the method of data collection is efficient in time and personnel (two persons for 1 day per year, in this case).

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