EC 761

# Effects of Fire
## in the Northern Great Plains

U.S. Fish and Wildlife Service
and Cooperative Extension Service
South Dakota State University
U.S. Department of Agriculture

EC 761

# Effects of Fire in the Northern Great Plains

**Kenneth F. Higgins**
U.S. Fish and Wildlife Service
South Dakota Cooperative Fish and Wildlife Research Unit
South Dakota State University, Box 2206
Brookings, South Dakota 57007

and

**Arnold D. Kruse**
U.S. Fish and Wildlife Service
Northern Prairie Wildlife Research Center
Box 2096
Jamestown, North Dakota 58402

and

**James L. Piehl**
U.S. Fish and Wildlife Service
R.R. 1, Box 76
Fergus Falls, Minnesota 56337

## Preface

This publication is a review of selected literature about prescribed burning in the Northern Great Plains (NGP) for management of wildlife. It also will be useful to other resource managers and researchers and to persons interested in the NGP. It is more "descriptive" than "interpretative."

The publication is a joint effort of the South Dakota State Cooperative Fish and Wildlife Research Unit (SDCFWRU), South Dakota State University, Brookings; the Northern Prairie Wildlife Research Center (NPWRC), Jamestown, N.D.; and the U.S. Fish and Wildlife Service (USFWS), Fergus Falls, Minn. Manuscript typing and library services were shared between SDCFWRU and NPWRC.

This publication (EC 761) is the second of three SDSU Extension circulars on grassland fires. EC 760 is *Prescribed burning guidelines in the Northern Great Plains;* EC 762 is *Annotated bibliography of fire literature relative to northern grasslands in South-Central Canada and North-Central United States* and contains many more citations than presented in this publication. All three circulars may be obtained from either the Wildlife and Fisheries Sciences Department; SDSU Box 2206; ph (605) 688-6121; or from the Ag Communications Bulletin Room; SDSU Box 2231; ph (605) 688-5628; both in Brookings, S.D. 57007.

## Acknowledgements

We thank Mary Brashier, publications editor, and Duane Hanson, graphic designer, Department of Agricultural Communications, SDSU, for editorial assistance and R.F. Gartner, W.D. Svedarsky, C.H. Halvorson, and K.L. Solberg for reviewing the manuscript. SDSU wildlife and range science students (Appendix A) helped with the literature search during coursework in grassland fire ecology in spring 1986.

--Kenneth F. Higgins

## Contents

*EC 761*

# *Effects of Fire*

## *in the Northern Great Plains*

Fire has been used inconsistently to manage native and tame grasslands in the Northern Great Plains (NGP) of the north-central U.S. and south-central Canada, particularly the grasslands found in prairies, plains, agricultural land retirement programs, and moist soil sites.

This has happened for three primary reasons: (1) the reduction of American Indian use of fire after 1875, (2) fire suppression and land use changes that put increasingly more acres under annual tillage since about the same time, and (3) a growing resistance to the use of fire since about 1940, largely due to media overemphasis of its harmful effects (e.g., "Bambi" and "Smokey the Bear").

Little can be done to change the first two factors but there is ample opportunity to change human attitudes about fire.

Attitudes change when the knowledge (or lack of it) changes. We believe that people have been reluctant to include fire in resource management programs in the NGP because of a lack of adequate information about the effects of fire on the soils, plants, and animals in the region.

This document provides information concerning fire effects on the grassland biome of the NGP, with special emphasis on the use of fire for wildlife management. In several instances we have drawn from published literature outside the geographic region, but only to provide a more complete reference for readers and decision makers.

In most instances, we only state or abstract the published findings of others without interpretation, either pro or con. Readers can fit the information into their specific circumstances.

English and scientific names are from *Flora of the Great Plains* by the Great Plains Flora Association and from the *Checklist of vertebrates of the United States, the U.S. territories, and Canada* by R.C. Banks, R.W. McDiarmid, and A.L. Gardner.

### Effect of fire on soil nutrients and minerals

Fire increases or decreases soil nutrient amounts, depending on the intensity and duration of the burn. Two obvious direct effects are volatilization of certain elements and modification of soil particles due to heat.

Volatilization sends carbon, hydrogen, and oxygen, (C, H, and O) into the atmosphere, along with varying amounts of sulfur (S), and phosphorus (P) depending on the composition of the organic matter burned and the degree of combustion (Raison 1979).

Nutrients in mineral form are affected by the changing physical properties of soil particles due to heating and subsequent cooling. When micaceous minerals and clays dehydrate or fracture, the solubility of elements such as P and potassium (K) can increase or decrease (White et al 1973).

Chemical changes at mineral surfaces can be caused by alkaline or alkaline earth compounds from the heated minerals or by organic matter combustion. Solubility of P or K can increase or decrease depending on the chemical compounds formed when the material cools. Rapid heating and cooling may break a mineral apart as it expands or contracts. Fresh unweathered surfaces could release P and K more rapidly than weathered surfaces.

#### Nitrogen (N)

One effect of fire on N is volatilization (DeBell and Ralston 1970; Sharrow and Wright 1977a; Tiedmann and Anderson 1980). Fire intensity, amount of green material, and fuel moisture have been reported to influence the amount of N lost through volatilization (Dunn and DeBano 1977).

Amounts of N lost range from 30 to 33 lb/A (34 to 37 kg/ha) with 2,000 to 3,000 lb/A (2,240 to 3,360 kg/ha) of fuel (Sharrow and Wright 1977b). N decline has also been noted for litter, mor, and A-1 horizons when temperatures exceeded 200 C (White et al 1973).

Although there is ample evidence that N in organic matter is volatilized, some authors report an increase in total soil N (which would include organic N, nitrate, and ammonia) after a fire (Vlamis and Gowans 1961; Vlamis et al 1955; White and Gartner 1975).

Nitrate levels have also risen after a fire (Kramer 1973; Christensen 1976; Sharrow and Wright 1977a; Worcester 1979). Schripsema (1977) found nitrate and ammonia declined in August following a winter burn; total N was also lower on a spring burn.

Researchers have seen an increase in ammonia after burning (White and Gartner 1975; Christensen 1976; Worcester 1979). Schripsema (1977) thought lower levels of ammonia and nitrate may have reflected increases in plant uptake.

The reported increases in all forms of N could be due to stimulation of legumes (Mayland 1967), the washing of charred surface material into the soil (Metz et al 1961), formation of ash which increases growth of nitrifying bacteria (Burns 1952), and increased growth of nitrogen-fixing microorganisms (Isaac and Hopkins 1937). Nitrifying bacteria are protected from heat and recover quickly to produce nitrates from organic matter (Sharrow and Wright 1977a).

Ammonia increases have also been attributed to increases in biological activity after heating (Walker and Thompson 1949; Jenkinson 1966; Simon-Sylvestre 1967). Ammonifying bacteria can withstand heat up to 212 F (100 C), while nitrifiers die at 127- 142 F (53-58 C) (Raison 1979). Certain forms of N increase or decrease, depending on fire intensity.

Heat also intensifies the physiochemical processes which lead to the decomposition of nitrogen-containing organic matter and to the release of ammonia from soil minerals (Aref'yeva and Kolesnikov 1964). Ammonia loss peaks at 482-572 F (250-300 C), which might explain why ammonia could increase while organic N decreases as a result of volatilization at 392 F (200 C) (Raison 1979).

A guide to determine N loss is the appearance of the ash. Up to 392 F (200 C), material is charred. At 392-752 F (200-400 C), grayish ash skeleton becomes apparent. At 752-932 F (400-500 C), the litter and mor become grayish ash while the A-1 horizon becomes reddish or grayish (White et al 1973).

### Phosphorus (P)

P as phosphate is another nutrient released by burning. Schripsema (1977) found the availability of P to vary by site. Others have found availability to increase (Kramer 1973; Smith and Owensby 1973; White and Gartner 1975; Christensen 1976; Raison 1979).

White and Gartner (1975) found an increase in available P only if temperatures did not exceed 392 F (200 C). They also speculated that, as in the case of ammonia, soil moisture and heat determine the extent of the increase in P availability.

### Other Nutrients

Comparatively few studies discuss nutrients other than N and P. Availability of K, calcium (Ca), and magnesium (Mg) may increase after fire (Christensen 1976; Raison 1979). Soluble K will increase in the litter, mor, and A-1 horizon if temperatures do not exceed 392 F (200 C) (White et al 1973). Ohr and Bragg (1985) found that Ca, iron (Fe), and manganese (Mn) decreased. However, they also found that if the plot was burned in consecutive years, then K, copper (Cu), Fe, and zinc (Zn) availability increased. This was attributed to different rates of plant uptake for each nutrient.

### Nutrient loss

Although an overall increase of most cations is well documented, fire can induce losses in some cases.

Losses may be due to surface erosion (Wells et al 1979), movement below the root zone from leaching (Stark 1979), dilution effects of increased runoff (DeBano and Conrad 1978), and losses in fly ash (DeByle 1976).

These findings confirm that actual effects on soil nutrients at any given site will be variable depending on the condition of the vegetation, character of the soil and topography, and climatic factors (Vogl 1974).

### pH

Increases in pH have been attributed to ash accretion (Ahlgren and Ahlgren 1960; Smith 1970). The response depends on the amount of ash and buffering capacity of the soil (Tryon 1948) and is considered negligible in grasslands (Nye 1959).

Timing of the burn and pH level of the existing soil may be important. Vlamis et al (1955) found pH to rise on neutral but not acid soils. Owensby and Wyrill (1973) found a larger increase in pH from winter and midspring burning than after late spring burns. This rise in pH is because mineral substances are released as oxides or carbonates that usually have an alkaline reaction (Schripsema 1977). This is supported by others who have found that ash is dominated by carbonates of alkaline and alkaline earth metals (Youngberg 1953; Daubenmire 1968). Mayland (1967) found pH to be 0.5 higher, and Christensen (1976) found no change at all.

There is also the possibility of pH rising 0.5 to 0.4 but only persisting for 1 or 2 years (Wright and Bailey 1982).

### Nutrient availability

Although nutrient levels are important, nutrient availability must also be high if plants are to benefit. One effect of fire is to make water-soluble cations immediately available for plant uptake (Raison 1979; Wright and Bailey 1982). Raison (1979) found cation exchange capacity to be lowered by presence of oxidized organic matter, but light burns did not affect the exchange system (Scotter 1963).

### Other effects

Increased nutrient levels due to fire have been well documented, but some authors think the effect on soil microbes and residues is more important (Wright and Bailey 1982). Bacterial populations, for example, decrease after a burn but increase three- to tenfold within a month because soil temperatures and nutrients for their growth are more favorable (Jurgensen et al 1979).

Litter removal and a dark surface cause soil temperatures to increase (Sharrow and Wright 1977a). After a fire, higher temperatures shorten the oxidative process and are believed to be the main effect fire has on surface organic matter (Harvey et al 1976).

This coincides with Hulbert (1969), who stated that the major short-term effect of fire is the removal of litter instead of fire-induced nutrient changes. He found that burned and clipped plots responded in the same manner, which suggested that the removal of mulch explained renewed vigor in burned stands.

Increases in available nutrients have often, but not always, been attributed to ash accretion (Tyron 1948; Ahlgren and Ahlgren 1960; Smith 1970).

### Summary

Fire affects pH, cation exchange capacity, organic matter oxidation, and soil organism activity in soils.

These factors, by themselves or acting together, determine availability of soil nutrients and plant benefits from fire. The interaction of these factors needs to be understood so that fire can be a better tool in grassland management for wildlife, livestock, and forage production.

## Effects of fire on upland grasses and forbs

One of the simplest and least expensive practices to improve poor quality grassland is prescribed burning. Selective suppression or promotion of a particular plant species depends primarily upon the date of the fire in relation to the phenology of the particular species.

Usually, those species actively growing when the area is burned are much more susceptible to injury and death than dormant species or those just initiating growth (Anderson et al 1970).

The proper time to burn can be based on physiological stages (e.g., root reserves) or morphological stages (e.g., when buds are exposed). A sequence of fires may be necessary to restore grasslands to proper condition.

Fire severity (which is closely related to fuel amounts and distribution, weather, and moisture content of soil and fuel) is also a major factor affecting fire damage to plants (Wright and Bailey 1982).

Research within the past few decades has shown that fire has been an important natural component of many grassland communities (Daubenmire 1968). Although historical records of fire in the Great Plains are limited (Higgins 1986a), fire suppression since the early 1900s has changed the structure and composition of many plant communities, particularly those subject to frequent fires (Daubenmire 1968; Wells 1970; Bailey and Wroe 1974; Gartner and White 1986; Gartner et al 1986).

Numerous factors affect the response of plants to fire. The biotic and abiotic factors generally recognized are grassland type, fire history, season, fuel and soil moisture conditions, wind speed and direction, air temperature, and time of day of the fire. Because of the complexity and interaction of factors and the lack of data concerning burns in the same community under similar circumstances, results of fire effects are often confusing and misleading.

**Not** burning may have as much effect upon grass production as burning itself. Tomanek (1948) stated that although mulch reduces soil temperatures and evaporation, it also increases filtration rates. Excessive amounts of litter can accumulate under light or no grazing or lack of fire. Buildup of litter usually causes degeneration of grass stands and lower yields.

Burning native prairie in northeastern Iowa increased grass seedstalk production (Ehrenreich and Aikman 1957). Possible causes for this increase were removal of large quantities of litter, stimulation of floral bud induction resulting from the direct heat of the fire, and higher temperatures earlier in spring. The authors concluded that burning resulted in an increased accumulation of carbohydrates due to improved plant growth conditions.

After a burn the soil warms more rapidly in the spring. Removal of the litter permits soil temperatures to average as much as 52 F (11 C) higher than on unburned sites in early spring (Peet et al 1975). Soil temperatures in early spring are inversely related to the amount of litter and duff (Ehrenreich 1959).

Early rising soil temperatures stimulate the increase of certain bacteria that decompose organic matter. This allows warm-season grasses to grow at an optimum rate if moisture is adequate. Most of the fertilizing effects after a fire result from nitrates released by bacteria consuming organic matter, not from nutrients in the ash (Sharrow and Wright 1977a).

On the other hand, Launchbaugh (1973) stated that yield reductions were associated with the removal of dormant growth by burning. The net effect was less soil moisture available for plant growth due to exposure to extreme winter temperatures and increased respiration. He added, however, that when mulch accumulations are excessive, burning will result in greater yields.

Cool- and warm-season species growing together may respond differently to the same fire; seasonal timing is critical (Bragg 1982; Wright and Bailey 1982). Some plants may be actively growing and especially susceptible at the time of the fire while others will be dormant and less susceptible.

Many cool-season plants will be actively growing during spring and fall fires, but most warm-season plants either will be dormant or will have not yet expended a significant amount of stored energy on new growth. In summer, cool-season plants have nearly stopped growth or are dormant. Fire at this time is usually detrimental to warm-season species (Vogl 1974).

Spring burning will reduce species competition. Repeated burning on March 1 resulted in a sharp decrease in the number of Kentucky bluegrass *(Poa pratensis)* plants in Iowa (Ehrenreich 1959). Bluegrass, a cool-season exotic, also decreased sharply by repeated burning in early March (Bailey 1978; Engle and Bultsma 1984). Most native grasses are still dormant at this time when Kentucky bluegrass, beginning to grow, becomes highly susceptible to heat injury from fire. Thus, warm-season native grasses have higher yields because of decreased competition from cool-season invaders such as Kentucky bluegrass.

Native annuals are usually encouraged by burning if the fires occur at the appropriate time (Daubenmire 1968). Many annuals, as well as short-lived perennials, are opportunistic or pioneer species which require the open soil, reduced competition, and full sunlight characteristic of many post-burn sites (Vogl 1974).

Besides creating favorable sites for pioneer species, grassland fires directly affect seed germination and seedling establishment of native annuals. Vareschi (1962) found that soil temperature in many grasslands can reach 140 F (60 C) for several hours after a fire due to solar radiation. This was not detrimental to the seeds of native annuals; soil surface temperatures of even 194 F (90 C) for a few seconds were not harmful to most seeds.

The leaves and stems of annuals are frequently dry while the seeds contained in the inflorescence are still ripening. Fire occurring while the seeds are held aloft usually kills most of them (Daubenmire 1968). Fire also is detrimental to most actively growing annuals. Recurring fires during active growth can eliminate some annual plants (Vogl 1974).

Many perennial species are capable of vegetative reproduction, which gives them a competitive advantage in colonizing open or post-burn sites and aids the species in surviving damage from fire or other catastrophes (Vogl 1974).

The effect of fire on perennial plants varies with stage of development, fire intensity, and relative position of the perennating buds. Some species have perennating buds on above-ground stems where they are easily killed by fire. Others have their buds underground on roots or rhizomes. Buds at or below the soil surface are less susceptible to damage by fire than those above the soil surface. Hot or prolonged fire is detrimental to perennials when high temperatures destroy the perennating buds (Daubenmire 1968).

Perennial plants are also susceptible to fire after food translocation has taken place. Generally, as new foliage reaches maturity, the major portion of the food reserves has been withdrawn from the underground organs. Leaf and stem destruction at this time injures the plant most severely (Aldous 1934).

Although many environmental factors alter the effects of burning, drought conditions are the most limiting to grass production in the NGP (Wright and Bailey 1980; Engle and Bultsma 1984). During a drought, first post-year herbage yields were not increased by burning even though excessive mulch accumulations were removed by fire (Engle and Bultsma 1984).

## Shortgrass prairie

Although shortgrass prairie occurs only in southern Alberta, southeastern Wyoming, northeastern Colorado, and western Kansas within the NGP, we believe it is important enough to include as part of these guidelines. One of the greatest benefits from burning shortgrass prairie is an increase in utilization by livestock (Wright and Bailey 1982).

The primary grass species dominating shortgrass prairies are buffalograss *(Buchloe dactyloides)* and blue grama *(Bouteloua gracilis)*. Fire tolerance of most species in the shortgrass prairie under different moisture regimes appears to be similar to that for buffalograss and blue grama.

Red threeawn *(Aristida longiseta)* and sand dropseed *(Sporobolus cryptandrus)* are usually harmed by fire. However, sand dropseed tolerated fire when winter and spring precipitation was 40% above normal.

During dry years, most species of the shortgrass prairie are harmed by fire. Following a spring wildfire, when soil was dry, the recovery time for a buffalograss-blue grama community was three growing seasons (35%, 62%, and 97% recovery following the first, second, and third growing seasons, respectively) (Wright and Bailey 1982).

Other species harmed by a wildfire during a year of below normal precipitation included slim stem muhly *(Muhlenbergia filiculmis)*, ring muhly *(M. torreyi)*, wolftail *(Lycurus phleoides)*, and galleta *(Hilaria jamesii)*.

In the shortgrass area of southern Alberta, spring burning reduced forage production by 50% in the first year and by 15% in the second year, with recovery completed by the third year. Fall burning was less serious, reducing production by 30% the first year with recovery complete by the end of the second year (Clarke et al 1943).

A wildfire in a western Kansas shortgrass range reduced the basal cover of buffalograss and blue grama grasses by 48% and 67%, respectively. Shortgrass areas with heavy litter were severely damaged by burning, based on basal cover and forage production, compared to lighter damage on areas with less litter (Hopkins et al 1948).

Near Hays, Kan., March burning decreased first-year yields in a buffalograss-blue grama community by 65% and in a western wheatgrass *(Agropyron smithii)* shortgrass type on two locations by 82% and 48% (Launchbaugh 1964). By the third growing season, production differences were no longer significant.

Reduced production following a fire is attributed to (1) partial killing of the forage present, (2) reduction of plant vigor of the remaining forage plants, and (3) reduced moisture penetration associated with reduced ground cover and greater evaporation of soil moisture.

Although grasses are the major plants in shortgrass prairie, many species of forbs occur during years with above normal precipitation. Total forb yields are usually reduced more by spring burns than fall burns. In all cases, however, forb composition will be increased by burning when plants are dormant. Young, actively growing forbs will be severely harmed by fire.

The average basal diameters of bunches of blue grama increased regardless of treatment type. Basal diameter of red threeawn and sand dropseed decreased on plots burned 2 years in succession. Blue grama continued to increase except when burned 2 consecutive springs with a headfire.

Height reduction following fire has often been noted. A spring burn in western Kansas (Launchbaugh 1964) resulted in decreased heights of blue grama, buffalograss, and western wheatgrass. Height of blue grama was less in all burned plots, compared to unburned plots during the first burn year, with apparent recovery after the second year.

Wright and Bailey (1980) concluded that burning during dry years is apt to have negative results on shortgrass range. However, from his work in Texas, Vallentine (1971) suggested that infrequent burning should not harm grasses if done during moist periods. A burn may provide little benefit, however, unless an excessive litter buildup has occurred (Vallentine 1971).

These studies appear to suggest that burning shortgrass sites is undesirable because production and ground cover are both reduced. However, much of the evidence is based largely on wildfires or prescribed burns made without consideration of weather or soil moisture conditions.

### Mixed prairie

The mixed prairie of the NGP is located in eastern Montana, eastern Wyoming, all but the eastern edges of North and South Dakota, southeastern Alberta, and southern Saskatchewan (Wright and Bailey 1980). Annual precipitation varies from 15 to 19 inches (38-48 cm) per year in some mesic areas to less than 15 inches (38 cm) in semiarid regions.

Prescribed burning on mixed prairie in the NGP has become a controversial management technique during the past two decades. Negative attitudes toward burning have limited funding of fire ecology research in most NGP states (Gartner and White 1986) and have limited the use of fire as a possible management tool (White and Currie 1983a).

The effects of fire on native grasslands are indeed varied, but evidence shows that prairie closed to both grazing and fire soon begins to deteriorate (Anderson et al 1970; Kirsch and Kruse 1973; Schacht and Stubbendieck 1985).

Anderson et al (1970) burned upland mixed prairie in the Flint Hills of Kansas in early spring (March 20), midspring (April 10), and late spring (May 1).

Big bluestem increased under mid- and late spring burning, but increased only slightly under early spring or no burning. Sideoats grama *(Bouteloua curtipendula)* herbage remained constant under all burning systems.

Kentucky bluegrass was nearly eliminated from the treatment sites regardless of the time of burning. Buffalograss declined in the late spring burned pastures and was stable in the others. Blue and hairy grama *(B. hirsuta)* were favored by early and midspring burning.

Periodic droughts have a strong influence on recovery of mixed-prairie grasses after a fire (Hopkins et al 1948; Wright and Bailey 1980; White and Currie 1983a).

**Semiarid mixed prairie.** In the more arid regions of the mixed prairie, fire can result in decreased herbage yield (Gartner et al 1978) and critical reductions in litter (Dix 1960).

However, effects differ, primarily with season of burning, pre- and post-burn precipitation, and plant species composition (Clarke et al 1943; Coupland 1973).

Forde et al (1984) burned different areas of mixed prairie on the same day in the Wind Cave National Park in South Dakota. In the Red Valley burn, most of the perennial species decreased in percentage of ground cover the year of the burn, but cover rapidly increased during the next 2 years. In the Bison Flat burn, frequency of perennials decreased 25%, but air-dried biomass increased 38%, meaning fewer but larger plants remained after the fire.

Burning at various fuel moisture levels was investigated in two plant communities in Wind Cave National Park in South Dakota. Burning vegetation at 30%, 38%, and 46% fuel moisture had no significant effect on either little bluestem or a mixed grass community. With the exception of a decrease in cool-season species due to burning in late May and early June, no major species alterations were noted (Worcester 1979).

On April 25, 1980, a mixed prairie in the Loess Hills of southern Nebraska was burned with backing fires (Schacht and Stubbendieck 1985). One study tract was dominated by a shortgrass community, but showed remnants of some desirable species of the mixed prairie such as big bluestem, sideoats grama, and little bluestem. The purpose of the burn was to shift species composition to higher yielding, native mixed grasses.

The initial effect of the fire was to greatly suppress the herbage yields of cool-season species. Annual bromes were nearly eliminated, and bluegrasses were damaged to a degree. Yields of both blue grama and sand dropseed were significantly higher on burned plots than on unburned plots.

Sand dropseed is a prolific seed producer and is drought resistant. It is one of the first species to grow on denuded rangeland where soil texture is sandy to silty.

Yield responses for western wheatgrass, blue grama, and threadleaf sedge *(Carex filifolia)* were measured after both spring and fall burning on a mixed prairie in eastern Montana (White and Currie 1983a).

Overall, blue grama responded better under spring burning. Western wheatgrass production was unaffected by spring and fall burning. Threadleaf sedge was found to decrease in production following fall burning. Spring burning resulted in higher total productivity than fall burning.

Redmann (1978) studied plant and soil water status throughout the growing season following an October fire in northern mixed prairie. Lower water potentials in the burned sites resulted in decreased production of western wheatgrass and Junegrass *(Koeleria pyramidata).*

DeJong and MacDonald (1975) also indicated that burning can alter the microclimate, resulting in unfavorable plant and water status.

Gartner et al (1978) conducted burns in western South Dakota to determine the effect of seasonal burning on Japanese brome *(Bromus japonicus).* Winter, late spring, and fall fires significantly reduced this annual grass, while at the same time the yield of western wheatgrass increased after winter and fall burns but declined with late spring burning.

Vegetative changes attributed to wildfire in the timbered breaks of central Montana were observed over a 10-year period (Eichhorn and Watts 1984). Although differences between the five plant associations were noted, some general trends existed:

Burning eliminated non-sprouting woody species such as big sagebrush *(Artemisia tridentata)* and Rocky Mountain juniper *(Juniperus scopulorum).* Sprouting shrubs such as choke cherry *(Prunus virginiana)*, snowberry (*Symphoricarpos* spp), and rose *(Rosa* spp) increased. Forbs peaked 3 to 4 years after the burn and then decreased.

**Mesic mixed prairie.** Wright and Bailey (1982) summarized numerous burning studies conducted in the mesic mixed prairie. Most concerned the effect of fire relative to seasonal changes in plants.

Engle and Bultsma (1984) studied the effect of burning during a period of below-average precipitation at the Samuel H. Ordway Memorial Prairie in north-central South Dakota. Mid-May and mid-June fires reduced Kentucky bluegrass and green needlegrass *(Stipa viridula).* The authors noted a similarity between plant responses following a burn in a mesic mixed prairie during drought to plant responses after burning in semiarid or xeric mixed prairie.

Burns were made on a mesic mixed prairie in Iowa on March 1 (Ehrenreich 1959). Dominant grass species were prairie dropseed (*Sporobulus heterolepis)*, little bluestem *(Andropogon scoparius)*, and big bluestem *(A. gerardii).* Areas were burned 1, 2, and 3 consecutive years.

Vegetation on areas with two burns began growing the earliest, matured earlier, and produced more flower stalks. This was attributed to a decrease in litter and higher soil temperatures. Grass growth began earlier, and the number of native plants which flowered increased. This occurred in the first growing season after a burn, but declined until the third growing season after burning, when both burned and unburned areas appeared very similar.

Some general statements may be made regarding production of mixed grass species in the NGP following prescribed burning.

Big bluestem increased in herbage during all periods of spring burning. Little bluestem seemed productive under burning, but not to the degree of big bluestem. Although sideoats grama yields remained constant, blue grama yields increased after spring burning. Finally, responses of *Stipa* species varied with spring burning.

Prescribed spring burning has increased production of many warm-season grasses in the mixed prairie. These increases vary, depending on rainfall and litter accumulation prior to and after burning (Smith and Owensby 1973).

Time of burning may affect certain species in a variety of ways because of differing phenological characteristics (Anderson et al 1970).

Prescribed burning is a viable management technique for mixed-prairie grassland management but is not recommended under drought conditions in the NGP (White and Currie 1983a; Engle and Bultsma 1984).

### Tallgrass prairie

The tallgrass prairie occurs mainly on the eastern edge of the NGP. Precipitation varies from approximately 18 inches (46 cm) annually in southwestern Manitoba to 30 inches (76 cm) in south-central Minnesota. Glacial till soils are predominant (Wright and Bailey 1982).

Of all the grassland ecosystems in North America the tallgrass prairies seem to benefit most from fire.

Many species, including big bluestem, little bluestem, Indian grass, and switchgrass *(Panicum virgatum)*, increase after burning (Wright and Bailey 1982).

Grasses such as sideoats grama and buffalograss, which are sometimes found in the tallgrass prairie understory, do not seem to be adversely affected by burning (Anderson et al 1970).

Cool-season grasses such as *Bromus*, *Elymus*, and *Poa* do not benefit and actually may be harmed by spring burning. Kentucky bluegrass, an exotic species which did not evolve under fire, can be almost eliminated by spring burning in tallgrass prairie (Wright and Bailey 1982; Svedarsky et al 1986).

Fire helps control woody plants (Bragg and Hulbert 1976) and Eurasian "weeds" in tallgrass prairies, and it enhances the growth of native prairie plants (Pauly 1982). The absence of natural fire or prescribed burning has allowed woody vegetation to increase in many areas of the tallgrass prairie.

Some possible results of burning in tallgrass prairie include litter reduction, suppression and eradication of unwanted species, shifts in species composition, and increases in production and diversity (Vogl 1974).

Heitlinger (1975) found that white sweet clover *(Melilotus alba)* can be controlled on Minnesota native prairie by one of the following three burning strategies: (1) burning annually in early May when the second-year shoots are clearly visible; (2) burning every second year in early July before seed of second-year plants ripens; and (3) burning annually in early September near the beginning of the critical growth period. Sweetclover is a biennial producing vegetative growth in year one and reproductive growth in year two.

Dziadyk and Clambey (1980) compared six plant communities in western Minnesota after a fall wildfire. Post-burn herbage production on five communities was largely attributed to an unusually cool spring following the burn.

Peet et al (1975) found that big bluestem herbage in Wisconsin was higher on burned sites due to more favorable environmental conditions after litter removal. Year-end big bluestem biomass was 0.11 1b/sq ft (531 g/sq m) on the burned site and 0.03 1b/sq ft (173 g/sq m) on the unburned control.

Controlled burning was used in the Cedar Creek Natural History Area in central Minnesota to promote grasses and forbs and reduce northern pin oak *(Quercus ellipsoidalis)* stands. Annual spring burning for 13 years increased the understory forb and grass species and decreased smaller oaks (less than 10 inches, or less than 25 cm dbh). Burned areas averaged 25 species of grasses and forbs, while unburned controls averaged only 13 species (White 1983).

Tester and Marshall (1962) found that burning a tallgrass prairie in Minnesota did not cause any noticeable shifts in vegetation composition, but the density of some undesirable species declined. Curtis and Partch (1948) reported that regardless of the burning schedule, Canada bluegrass *(Poa compressa)* and Kentucky bluegrass declined in abundance in a Wisconsin fire study.

In Missouri, Kucera and Dahlman (1968) observed 39% less root biomass on plots of big bluestem after 6 years of fire exclusion than on plots that had been burned annually for 10 years. They suggested a maximum of 3 years between burnings to avoid stand depletion of big bluestem from litter accumulation.

On the Trelease Prairie near Urbana, Ill., Hadley and Kieckhefer (1963) found that one year without burning resulted in marked decreases in living shoot and flowering stalk production of big bluestem and Indian grass.

They also noted that root biomass increased with burning frequency and Kentucky bluegrass biomass was reduced following late spring burns. However, Hadley and Buccos (1967) found herbage production on the Oakville Prairie in the Red River Valley of North Dakota was comparable on burned and unburned sites.

For North Dakota tallgrass communities, Hadley (1970) said, "continued burning of the upland communities probably will stimulate herbage yields and seed production by most of the grasses, while maintaining the forb and small shrub components. Sustained burning may or may not decrease yields and species diversity in the lowland communities." Burning on uplands produced 22% more biomass but 15% less herbage than on lowland saline sites.

In Wisconsin, Pauly (1982) found that the most successful prairie burns were conducted in late March, April, or early May. Drier sites should be burned earlier in the spring than wet sites due to earlier onset of growth. Spring fires were easier to control because the vegetation was usually packed down by snow, the fire moved more slowly, and flame height was reduced.

At Buena Vista Marsh in central

Wisconsin, little bluestem was stimulated by burning to produce greater germination and seed production (Zedler and Loucks 1969).

Tallgrass prairie vegetation will respond dramatically if prescribed burns are conducted at the proper time of year. Towne and Owensby (1984) and Launchbaugh and Owensby (1978) reported from the Kansas Flint Hills that the closer the time of burning is to the beginning of spring growth, the more favorable the response.

Owensby and Anderson (1967) found early spring burns reduced forage yields but late spring burns increased yield, compared to controls. Towne and Owensby (1984) further suggested that the discrepancies between past studies regarding the effects of fire on herbage yield are due to the differences in time of burning. They maintained that manipulation of the vegetation is possible with fire.

Tallgrass prairie burning reduces mulch cover and increases the number of reproductive grass shoots (Ehrenreich and Aikman 1957; Zelder and Loucks 1969; Hickey and Ensign 1983), and it also results in a more rapid phenological development of young plants and an increase in flower production (Hadley and Keickhefer 1963).

Curtis and Partch (1950) also found big bluestem plants to bloom profusely after burning. Ehrenreich and Aikman (1957) found the number of big bluestem seedstalks to be seven times greater in burned compared to unburned prairie in an Iowa study. Little bluestem and prairie dropseed showed an eightfold increase, and Indian grass had a threefold increase. Canada wildrye *(Elymus canadensis)* was unaffected by burning. The increase in seedstalk numbers corresponded with an increase in total seeds and more noticeably erect flower stalks. Percentage purity and germination was greater for seed harvested from the burned area as opposed to the unburned area, with the exception of Canada wildrye.

Hickey and Ensign (1983) reported burning increased panicle number and increased seed yield 1.6-fold compared to mechanical thatch removal in Kentucky bluegrass fields. Hulbert (1969) increased tiller numbers 1.5 to 2.7 times by mulch removal in undisturbed bluestem prairie in Kansas; however, inflorescences were rare in both mulched and unmulched plots.

Seed production of western ironweed *(Vernonia baldwinii)*, a common tallgrass prairie forb, was 32 to 43% lower (in seeds/plant) on burned sites than on sites where two or more seasons had passed since the last burn (Knapp 1984).

Weaver and Rowland (1952) found that when the mulch layer was removed by hand the flower stalk of big bluestem more than doubled in height. Big bluestem and switchgrass yields also increased after the removal of mulch. Ehrenreich (1959) found that vegetation of burned areas grew and matured earlier and produced more flower stalks than nearby unburned areas, but he pointed out that the greater height and increase in numbers of seedstalks were only temporary. He found little difference in burned and unburned areas after the second growing season.

Peet et al (1975) noted that 5 weeks after a burn, stem density of big bluestem was three times higher in burned areas. After only 3 more weeks there was little difference in stem density between burned and unburned plots in Wisconsin.

Fire affects big bluestem in three ways: (1) direct effect of the heat of the fire on the buds in the plant crown, (2) removal of accumulated litter from previous growth, and (3) the liberation of mineral fertilizers from the ashes (Curtis and Partch 1950).

The most important appears to be the removal of litter. Dark, bare soils warm faster in the spring than those shaded by litter, thereby enhancing seed germination (Hopkins 1954). Hadley and Kieckhefer (1963) attributed the increase in number of flower stalks to many factors but mainly to removal of litter. Curtis and Partch (1950) considered the presence of litter over the crowns to be the most important factor influencing flowering of big bluestem. When litter cover was removed, flower production increased six times and plant height increased by 60%.

Ehrenreich and Aikman (1957) agreed that increases in seedstalk production could be stimulated by the removal of large quantities of litter, but they felt the addition of ash and induced heat stimulation of buds to be important as well. They proposed that the most likely factor increasing seedstalk production was the increased accumulation of carbohydrate material in the plant from improved growth conditions, but they did not clarify what these improved growth conditions might be.

Hulbert (1969) reported soil temperatures on denuded plots in undisturbed bluestem prairie to be 34 to 41 F (1 to 5 C) higher than on mulched plots during the entire season. He concluded that earlier and greater growth and increased tiller numbers on denuded plots were due to higher temperatures and increased light intensity.

Weaver and Rowland (1952) reported that an accumulation of mulch thinned a stand of big bluestem and other tallgrasses to about a third the usual number of stems. They found soil temperatures under the mulch to be about 25 F (14 C) lower than on plots where mulch had been removed. They also found that only 1 to 5% of light penetrated through even the first inch of the normally compacted mulch. Decreased soil temperatures and less light resulted in a 3-week delay in spring growth, and production of flower stalks was delayed in the grasses which remained mulched.

Peet et al (1975) attributed production increases of big bluestem in a field following burning to more favorable environmental conditions for net photosynthesis from the time of leaf emergence through late June because of more light and higher soil temperatures.

In research by Old (1969), grass seed yields increased following burning due to litter removal, removal of competing cool-season plants, and increased nitrification due to increased soil temperatures.

Hardison (1980) reported that fire has been used for 30 years by commercial grass seed producers in the Pacific Northwest to reduce weed seeds and to control insects and several plant diseases.

In summary, fire does have a measurable positive effect on the yield of most tallgrass prairie grasses. Soil moisture is an important determining factor in the yield potential of seed and foliage. Native grasses on upland sites respond favorably, but tallgrass species on moist lowland or saline sites may not be as competitive with other species.

## Effects of fire on some undesirable species

### Cacti

Cacti are relatively fire susceptible (Wright and Bailey 1980). Plains prickly pear cactus *(Opuntia polyacantha)* is adversely affected by repeated burning (Martin 1983).

Bunting et al (1980) used prescribed burning to control the density of eight species of cacti on southern mixed prairie in west Texas. Seven of eight species suffered mortality of 49 to 100% by the fourth year after burning. Mortality was either a direct effect of fire or was a fire-induced interaction with insects, rodents, or disease.

Burning makes all cactus species more attractive to cattle, and the reproductive rate of most species is low. Burning at intervals of 5 to 6 years prevents development of dense stands of prickly pear. Fall burning of prickly pear killed more than 80% of the pads, while spring burns accomplished a 40% pad kill (Dodd et al 1985).

In Alberta, pronghorn antelope (*Antilocapra americana*) are known to readily consume prickly pear cactus after burning removes the spines. Over 50% of the green pads were utilized after a burn (Stelfox and Vriend 1977). The use of fire in prickly pear control along with the response of pronghorns and cattle to eating burned pads may warrant further investigation.

### Kentucky bluegrass

Late-spring fire has been a particularly effective method of controlling Kentucky bluegrass (Hensel 1923).

In many mesic areas of the mixed prairie, prescribed burning has controlled cool-season grasses without reducing herbage yields or cover of warm-season grasses (Kirsch and Kruse 1973; Gartner and Thompson 1973); although Dwyer and Pieper (1967) report total herbage yield was reduced the first year following a burn. Zelder and Loucks (1969) found that growth began earlier and continued to be greater on burned plots.

Schacht and Stubbendieck (1985) indicated that bluegrasses were damaged but not eliminated from burned plots. Bluegrasses appeared to regain their vigor by the second year following the fire, but their herbage yields on burned plots remained lower than on control plots.

Zelder and Loucks (1969) found that the standing crop of Kentucky bluegrass was greater on unburned ridge sites. Bluegrass height was greater on the unburned than on the corresponding burned plots. This response, however, could be expected, since plots were burned after the plants had begun growth (Zelder and Loucks 1969).

Zelder and Loucks (1969) also reported a general trend for spring fires to decrease fruiting of early blooming grasses and to increase fruiting of late blooming grasses. They suggested that burning may damage flower primordia of early blooming grasses. On upland sites, burning reduced Kentucky bluegrass seed production and increased seed production of little bluestem.

The increase in fruiting that results from burning late blooming prairie grasses is well documented by other studies, as is the decrease of Kentucky bluegrass fruiting (Curtis and Partch 1950; Ehrenreich and Aikman 1963).

In summary, Kentucky bluegrass is more susceptible to damage by fire on ridge sites and little affected in depressions. The low fertility and high permeability of the ridge soils seem to make the effect of fire somewhat more devastating than on soils of deeper, heavier texture. It seems reasonable, then, to suggest that consecutive burning for several years running of areas where exposure is high would probably increase desirable species and decrease Kentucky bluegrass.

**Cheat grass** *(Bromus secalinus)*

Fire hazard in a stand of vegetation is increased by the presence of cheat grass. The extremely high flammability of the dry grass permits fires to start and spread with unusual rapidity.

Fire will also enhance establishment and spread of cheat grass (Klemmedson and Smith 1964; Schacht and Stubbendieck 1985; Young et al 1976).

In the Utah foothills, Pickford (1932) found that cheat grass made up less than 1% of the vegetative composition on ungrazed and unburned areas. On ungrazed but burned ranges, cheat grass made up 22%, whereas on unburned but grazed areas it comprised 15%. Cheat grass dominated vegetation under the combination of both burning and grazing, at 38% plant frequency.

Repeated burning every few years or burning in early summer will deplete a stand of perennial grasses and allow annual grasses, primarily cheat grass, to increase sharply (Young et al 1976). Once a sagebrush-grass community is depleted of perennial plant cover, secondary succession goes from Russian thistle *(Salsola iberica)* to mustard *(Sisymbrium* and *Descurainia* spp*)* to cheat grass within 5 years (Wright and Bailey 1982).

Pechanec and Hull (1945) found that burning reduced cheat grass plants, depending on the month of the burn. Early summer burns, at the time of year when climax perennials are easily killed by fire, were only a temporary setback for cheat grass (Wright and Bailey 1982). Therefore, the density of cheat grass increases over time while fewer perennials survive after each fire.

Young et al (1976) reported that after a late July burn there was an 80% or greater reduction in cheat grass and cheat grass seed production. However, in a burn study conducted by Barney and Frischknect (1974), the cover value of cheat grass varied from 12.6% in the 3-year-old burns to 0.9% in the oldest stands. Cheat grass declined in cover the first 22 years after fire, then leveled off and stayed about the same.

Pechanec and Hull (1945) showed that during the year following burning, cheat grass plants were far fewer on burned than on unburned ranges.

These studies give us considerable difference of opinion about the effectiveness of fire as a tool for reducing cheat grass stands.

Time of burning is evidently an important factor determining subsequent cheat grass stand density. Cheat grass was effectively controlled by burning in late spring, just as the seed matured but before it shattered (Stark et al 1946; Plummer et al 1955). Areas burned in early summer had light remnant stands, compared with fall-burned areas (Pechanec and Hull 1945). Their studies near Boise, Idaho, showed that June and July burns reduced plant numbers to 14 and 11 per square foot compared to 41, 45, and 124 plants per square foot, respectively, on August, October, and November burns.

Warg (1938), in disagreement with many other observers, felt that burning was not a satisfactory means of controlling cheat grass. Leopold (1941) agreed, stating, "The more you burn cheat the thicker it grows next year, for the seeds shatter early and harbor in cracks in the ground."

The latter part of Leopold's statement is significant and has been stressed by others as a key to the success of cheat grass in competing with perennials.

Warg (1938) observed that cheat grass was damaged less by heat than were perennial natives. After 5 minutes at 257 F (125 C), germination of cheat grass was 87.25% as compared to 9.87% with the control. After 302 F (150 C) for 5 minutes, cheat grass seed failed to germinate.

Cheat grass fire hazard differs from that of most perennial grasses of the western range. The plant matures early in June and dries out within 1 or 2 weeks after maturing, remaining a hazard until fall.

The high flammability of cheat grass is not only a function of its early maturity and uniform stands, but may be at least partially explained by its low moisture content when mature (Klemmedson and Smith 1964).

Fire is a major cause of disturbance that has enhanced the establishment and spread of cheat grass, but fire can also be used to control the species.

## Effects of fire on shrubs

Fire commonly is used in rangelands to remove shrubs of low forage value (Stoddart et al 1975).

Burning increases range browse availability mainly by reducing shrub crown heights, by the addition of new browse plants through seed germination, and by increasing palatability associated with young growth (Vallentine 1971; Mathews 1984).

However, burns on wildlife range in any one year should be limited in size, since browsers are often unable to fully utilize all of the new sprouts on large continuous burns (Vallentine 1971). The amount of woody plants may actually increase.

Control of wildfire, Bailey (1976) said, resulted in an unprecedented increase in woody plants on grassland, to the advantage of big game populations. Conversely, brush encroachment has always decreased the carrying capacity of rangeland for cattle. There is a need for more controlled burns to maintain grasslands and shrublands (Bailey 1976).

After fall burning there is no regrowth of winter browse for wildlife. Spring burns usually increase sprouting after 4 to 8 weeks, but fall burns promote a taller regrowth the following year. In both spring and fall burns, shrubs are reduced in height and twig diameter, making regrowth more available for animal use (Leege and Hickey 1971).

In general, shrubs contain higher crude protein percentages in fall and winter and lower percentages in spring and summer than do grasses and forbs. The leaves of shrubs contain a higher percentage of crude protein than stems, and the tips of stems contain a higher protein level than the thicker mid and butt sections (Dietz 1972).

Most prescribed fires do not consume living woody material larger than 1/2 inch (1.2 cm) in diameter. Consequently, the proportion of smaller fuels is important in determining the character and behavior of a fire in a shrub stand. Living fuels usually contain large amounts of moisture and hence do not burn well. Burning dead fuels can provide the heat necessary to dry the living fuel to a point where it will ignite and add to the total energy release from a fire (Nord and Countryman 1972).

Plant age, soil moisture at time of burn, intensity of fire, season of burn, health of the plants, and frequency of droughts all play a part in how fire affects shrubs in the long run. To maintain a healthy shrub community, it is best to burn when the plants you wish to preserve are dormant and soil moisture is good (Wright 1972). Very probably, much of the true prairie would have evolved or would have succeeded into shrub or forest land if fire had been excluded.

Recurring fires generally favor grasses and herbaceous species over woody plants and shrubs (Vogl 1974). Most fire-adapted or fire-tolerant woody species cannot sustain large populations in grasslands subject to intense fires on a frequent basis (Glover 1972).
Food is translocated in most deciduous woody plants prior to the seasonal dormancy period (White 1983), but they do not die back at the end of their growing season as most grasses and forbs do. Regardless of whether the plant is actively growing or dormant, fire will damage living tissue (Vogl 1974; White 1983).

Many woody plants sprout or "sucker" from meristematic buds on underground stems or roots (Anderson and Bailey 1980; Wright and Bailey 1982). The season and frequency of fire can determine the net change, if any, in density and stand of sprouting species. If fire occurs before active growth has begun, increased density from sucker development may result (Anderson and Bailey 1980).

Bitterbrush *(Purshia tridentata)* is a desirable shrub but may not be compatible with fire.

Clark et al (1982) stated that bitterbrush survival after burning in Oregon has been variable. In eastern Idaho, bitterbrush sprouted inversely with burn intensity; in California sprouting was variable, with 5 to 25% sprouting after a fire. In north and central Utah, limited sprouting occurred after wildfires; but in the steppes of Washington and in the western Great Basin, wildfire always killed bitterbrush.

Spring burns are the least detrimental to bitterbrush if soils are wet during or just after a burn (Vallentine 1971; Wright 1972).

Reports on poison ivy *(Toxicodendron rybergii)* are mixed. Fires produce a definite and long-lasting increase in poison ivy, according to Wright (1972). However, Bock and Bock (1984) reported that poison ivy was unaffected by fire.

Smoke from burning poison ivy contains resins that can severely irritate lungs.

Western wild rose *(Rosa woodsii)* is fire tolerant and is considered a desirable forage species which takes 2 to 3 years to recover completely from a fire (Monsen and Davis 1985). Leege and Hickey (1971) and Bock and Bock (1984) reported that wild rose plants sprouted after a burn and remained at the same densities as at pre-burn. Wright and Bailey (1982) reported that *Rosa woodsii* is enhanced by fire.

Raspberry *(Rubus* spp*)* increases after a fire, especially hot burns (Wright 1972; Wright and Bailey 1982).

Choke cherry, serviceberry *(Amelanchier alnifolia)*, and snowberry sprout vigorously following fire (Miller 1963; Wright et al 1979; Pelton 1953; Wright and Bailey 1982).

However, prairie wild rose *(Rosa arkansana)* and western snowberry did not change appreciably after a fire in east-central North Dakota, and fire may reduce the abundance and vigor of silverberry (Kirsch and Kruse 1973; Wright and Bailey 1980).

Annual spring burning is often used to control shrub invasion of Canadian grasslands (Bailey 1976). However, frequency and stem densities of serviceberry and prairie wild rose increased on annually burned areas in Alberta (Anderson and Bailey 1980). Western snowberry and wild raspberry declined in frequency and stem densities on areas burned annually, but no shrubs were eliminated. Stem densities of western snowberry and wild raspberry increased two to five times after single-event fires.

Blackberry *(Rubus* spp*)* can be eliminated with 2 to 3 successive years of burning in late spring (Owensby and Launchbaugh 1976).

Two cool-season fires (spring and fall) consistently reduced densities of *Ribes* spp (Bock and Bock 1984). Peek et al (1979) observed western red currant *(R. cereum)* resprouting on burned sites.

Desirable shrubs such as serviceberry, snowbrush *(Ceanothus velutinus)*, and true mountain mahogany *(Cercocarpus montanus)* are only temporarily set back by fire (Wright and Bailey 1982). In another paper, Wright (1972) reported that serviceberry was severely damaged by fire. Bock and Bock (1984) found that serviceberry was reduced after a fire but increased during post-burn. Stem densities of serviceberry were greater on burned areas in an Idaho ponderosa pine community (Merrill et al 1982).

Merrill et al (1982) also found that seedlings of redstem ceanothus (*Ceanothus sanguineus)* increased following fire in the first year and that stem densities increased until the fourth year. In the first post-burn growing season, total shrub biomass on the burned area was about 50% that of the unburned area. By the third growing season, total shrub biomass exceeded that of the unburned sites. By the fourth season it was 35% more than the unburned area.

Smooth sumac *(Rhus glabra)* is an aggressive sprouter following fire (Wright 1972; Owensby and Launchbaugh 1976). Bragg and Hulbert (1976) found smooth sumac to be a major invader on all Kansas prairie sites, but that the invasion was negligible when sites were regularly burned.

Leadplant *(Amorpha canescens)* is a desirable leguminous shrub that is a prominent sprouter following burns (Wright 1972; Bock and Bock 1984; Towne and Owensby 1984).

In Kansas, a major grassland invader is roughleaf dogwood *(Cornus drummondii)*. Thick stands are reported on unburned plots (Towne and Owensby 1984); but with regular burns, encroachment is negligible (Bragg and Hulbert 1976). With successive burns for 2 or 3 years, dogwood can be substantially reduced (Owensby and Launchbaugh 1976).

Burning in ungrazed Kansas tallgrass prairie had different effects on woody species, but shrub composition in any treatment rarely exceeded 1% of the total vegetation (Towne and Owensby 1984). Plots burned in winter and early and midspring contained significantly higher amounts of woody plants than late spring burned or unburned plots.

White coralberry *(Symphoricarpos albus)*, when regularly burned, had greater stem densities compared with unburned control areas (Merrill et al 1982). Leege and Hickey (1971) and Bragg and Hulbert (1976) reported that white coralberry was held in check by regular burning and that stem densities did not increase.

Owensby and Launchbaugh (1976) indicated that 2 to 3 years of burning in late spring will substantially reduce coralberry *(S. orbiculatus)*.

Anderson and Bailey (1979) said annual burning restricted expansion of western snowberry *(S. occidentalis)* colonies into grasslands, whereas periodic burning enhanced the spread of this species. Western snowberry begins sprouting about 2 weeks after a burn and, by the end of 3 months, usually has a canopy cover greater than on control plots.

Johnson and Strang (1983) found that fire virtually eliminated gray rabbit brush *(Chrysothamnus nauseosus).* Cluff et al (1983) found that salt rabbit brush *(C. n.* var *consimilis)* resprouted in small areas following fire.

Broom snakeweed *(Gutierrezia sarothrae)* is severely damaged by fire (Wright 1972). Although it is easily killed by fire, it will reestablish itself with seedlings following wet winters and springs (Wright and Bailey 1980). Oswald and Covington (1983) found a preponderance of broom snakeweed on severely burned sites, suggesting that the species is fire tolerant.

Soapweed (*Yucca glauca)* can be adversely affected by fire, but in general most *Yucca* species are tolerant of fires and hold their own in various plant communities despite fire (Wright 1980).

Four-wing saltbush *(Atriplex canescens)* and winter fat *(Ceratoides lanata)* are desirable shrubs that resprout vigorously after fire (Wright and Bailey 1980). Greasewood *(Sarcobatus vermiculatus)*, sometimes an undesirable species, is also known to resprout following a burn (Cluff et al 1983).

Big sagebrush *(Artemisia tridentata)* is fire sensitive and is usually controlled by burning (Harniss and Murray 1973; Young and Evans 1974; Peek et al 1979; Cluff et al 1983; Johnson and Strang 1983).

A Wyoming big sagebrush *(A. t.* var. *wyomingensis)* site showed little re-establishment after 15 years post-burn, whereas mountain big sagebrush *(A. t.* var *vaseyana)* began to increase 12 years after the burn (Peek et al 1979).

Owensby and Launchbaugh (1976) reported that spring burning to top-kill plants, in combination with moderate grazing to retard sprouts and seedlings, will drastically reduce the density of sand sagebrush *(A. filifolia).* They urged caution with sandy sites where thick brush occurs, because complete removal by fire and hoof action may open an area up to wind and water erosion.

Wright (1972) stated that sand sagebrush is a non-sprouter whose seedlings come back vigorously following fire.

Burning controls three-tip sagebrush *(A. tripartita)*, black sagebrush *(A. nova)*, and low sagebrush *(A. arbuscula)* when sufficient fuel is available to support a fire (Wright et al 1979). Beetle and Johnson (1982) stated that black sagebrush is a good forage plant that is non-sprouting and fire susceptible and does not need to be controlled.

Silver sagebrush *(A. cana)* is completely top killed with spring and fall burns regardless of fire intensity. Plants with only foliage consumed, however, tended to resprout sooner than those that were completely burned (White and Currie 1983b). Beetle and Johnson (1982) found that dwarf sagebrush spreads extensively by root sprouting when stimulated by burning. Burning intensity acted to retard resprouting rather than to physically change the location of the resprouting point.

In spring, when soil moisture was high and silver sagebrush plants were just becoming physiologically active after winter dormancy, about a third of the plants burned to the stump, and 10% of those not burned so extensively were killed by fire. Considerably higher plant mortality was achieved by burning under dry fall conditions after sagebrush plants had completed their growth and reproductive cycles. After fall burning about three fourths of the plants that were completely burned to the stump died and almost 40% of those only partially burned were killed by fire (White and Currie 1983b).

## Effects of fire on trees

Woodlands in the NGP occur along streams and rivers, in draws, and in isolated localities having favorable moisture. Green ash *(Fraxinus pennsylvanica)*/choke cherry is the most common deciduous woodland habitat type (Hansen et al 1984; Girard 1985).

American elm *(Ulmus americana)* and box elder *(Acer negundo)* are present as minor components of the overstory. Undergrowth is generally dominated by choke cherry, western snowberry, western wild rose, American plum *(Prunus americana)*, and occasionally buffaloberry *(Shepherdia argentea).*

In addition, many deciduous woodlands have been invaded by Kentucky bluegrass, leafy spurge *(Euphorbia esula)*, and Canada thistle (*Cirsium arvense).*

Aspen *(Populus tremuloides)* in the NGP will be either enhanced or inhibited by fire, depending on the frequency of burns. Fire often kills the tops of aspen, but regeneration from root suckers takes place quickly after burning. Frequently, post-burn aspen abundance will exceed that of pre-burn (Anderson and Bailey 1980).

Most deciduous woodland species in the NGP exist at the edge of their ranges. Even on favorable sites, woody plants live under stressful conditions, characterized by extremes of temperature, wind, and precipitation.

Most deciduous trees and shrubs are capable of sprouting from roots, root-collars, or stems (Spurr and Barnes 1980). Many species respond favorably to increases in light intensity following burning. Seeds of most species survive fire; in some cases they are stimulated by heat to germinate (Ahlgren 1974).

In the absence of fire, shrubs and trees may become decadent, and the accumulation of downed woody material increases the fuel load and the likelihood of a hot, lethal fire.

Season of burning has been reported to differentially influence sprouting response of deciduous species (DeByle 1985). These variations in response are probably related to carbohydrate reserves stored in roots. Seasonal periodicity of carbohydrate reserves is known for many deciduous species.

Reserve carbohydrates attain their maximum at the beginning of autumn and diminish slightly through winter. In April and May, root reserves diminish rapidly and are consumed by formation of new branches and roots. Therefore, deciduous plants are most susceptible to serious damage in early to midsummer when carbohydrate levels are lowest.

However, burning in early spring before leaf-out or in autumn or winter when reserves are relatively high should result in a vigorous sprouting response.

Method of burning also influences the degree of survival and sprouting of deciduous species, because rates of spread and intensity will vary. Ferguson (1957) reported that hardwood stems killed by backing fires in loblolly pine *(Pinus taeda)* stands resulted in slightly but consistently fewer sprouts than those killed by headfires. However, on rough fescue *(Festuca scabrella)* prairie, brush up to 1 inch (2.5 cm) in diameter at flame height was top-killed by backing fires, while most brush up to 2 inches (5 cm) in diameter was top-killed by headfires (Wright and Bailey 1982).

In all cases, fuel loads and moisture, topography, and weather influence the degree of top kill of deciduous species.

Limited data are available on the response of native woodlands in the NGP to fire. Shrub densities were not reduced 1 and 2 years after a wildfire burned through a deciduous woodland in southwestern North Dakota, and the fire stimulated vigorous sprouting of many shrub species (Zimmerman 1981).

Other evidence on the response of deciduous species to controlled burning in this region is provided by Gartner and Thompson (1973) from foothills ponderosa pine *(Pinus ponderosa)* in western South Dakota. Burning did not appear to affect the frequencies of shrubs, and some species, such as leadplant and common choke cherry, survived the fire very well.

Bock and Bock (1984) reported that light prescription burns in early spring and late fall in ponderosa pine stands in the southern Black Hills reduced densities of currants *(Ribes* spp*)*, but most other shrubs were unaffected. However, a fall crown fire resulted in an increase in most shrub species, including red raspberry *(Rubus idaeus)*, currant, roses, and western snowberry.

Ecologists have postulated that *Juniperus* species are generally restricted to shallow soils on steep slopes and ridges because the species is fire intolerant (Gartner and White 1986). Unburned areas support interspersions of red cedar *(J. virginiana)* and American elm. In the absence of fire, trees progressively invade and will eventually dominate the tallgrass prairie (Towne and Owensby 1984).

Data from other regions suggest that fire may effectively stimulate reproduction of deciduous species. American elm seedlings established quickly after a spring burn in Kansas (McMurphy and Anderson 1965).

Because of the historical frequency of fires in the NGP and the apparent adaptability of many native plant species to fire, it is likely that fire maintained the integrity of plant communities in this region. However, the paucity of data on the impact of fires on native deciduous woodlands remains a weakness in our understanding of native woodlands ecology.

## General observations of fire effects on certain plant species

The effects of fire on most plant species in northern mixed prairie, particularly those associated with long-term burning, are not well known. Most of the available information has been based on short-term post-fire evaluations, (e.g. Dix 1960; Schripsema 1977; Wright and Bailey 1980; Kirsch and Kruse 1973).

The following are general observations of fire effects on certain plant species that we noted during recent field studies. They are based solely on observation and not empirical data.

Big bluestem, little bluestem, blue grama, Indian grass and switchgrass all increase in abundance with frequent spring (May-June) burns.

Composition and coverage of green needlegrass, needle and thread *(Stipa comata)*, and porcupine grass *(S. spartea)* increased during the first few sequential (May-June) burns but often declined rapidly after a sequence of five or more spring fires on the same area. Spring burning to reduce Kentucky bluegrass will commonly reduce *Stipa* spp at the same time.

Kentucky bluegrass and quackgrass *(Agropyron repens)* apparently decline in abundance after several consecutive spring (May-June) fires. Fires at the time of seedhead emergence appear most effective. Too few observations have been made on fall burns to generalize.

Western wheatgrass increases in abundance after spring, summer, or early fall burns, but considerably more after late summer or early fall fires. Intermediate wheatgrass *(A. intermedium)*, tall wheatgrass *(A. elongatum)*, smooth brome grass *(Bromus inermus)*, Junegrass, and spike oat *(Helictotrichon hookeri)* all responded well to spring fires and particularly to very early spring (March-April) burns.

Basin wild rye *(Elymus cinereus)* was unchanged with a 3-year rotation of May-June burns. Sweetgrass *(Hierochloe odorata)* responded most after August fires, but the sample of observations was small.

No changes to slight decreases occurred after periodic spring fires for white sage *(Artemisia ludoviciana)*, fringed sage *(A. frigida)*, wormwood *(A. absinthium)*, Flodman's thistle *(Cirsium flodmanii)*, western yarrow *(Achillea millefolium)*, prairie coneflower *(Ratibida columnifera)*, northern bedstraw *(Galium boreale)*, western ragweed *(Ambrosia psilastachya)*, stiff sunflower *(Helianthus rigidus)*, and leafy spurge. Occasionally, notable decreases in wormwood have been seen when this species was about 6 inches (15 cm) tall at the time of the burn.

Silver-leaf scurf pea *(Psoralea argophylla)*, lead plant, blue false indigo *(Baptisia australis)*, pasque flower *(Anemone patens)*, many-flowered aster *(Aster falcatus)*, lady slipper *(Cypripedium spp)*, white camas *(Zigadenus elegans)*, wild lily *(Lilium philadelphicum)*, tall gayfeather *(Liatris ligulistylis)*, Maximilian sunflower *(Helianthus maximilianii)*, sweet clover *(Melilotus* spp*)*, purple prairie clover *(Dalea purpurea)*, and harebell *(Campanula rotundifolia)* increased in abundance following spring burns.

Pasque flower bloomed in August and September after a late July or early August fire. Silver-leaf scurf pea showed greater increases after August fires than spring fires, but we have limited observations for August burns.

Alfalfa *(Medicago sativa)* was favored by early spring burns, but substantial declines followed late summer or fall burns.

Dramatic increases in sprouts of western snowberry often occur after a first fire, particularly on areas that have been idle for several years. A sequence of spring fires on the same area will eventually reduce abundance. Significant reduction requires five or more fires in 10 years or less. One or two fires followed by a series of rest years will result in an increase of aerial coverage. Hot burns in late summer to early fall have caused severe root burns on western snowberry plants.

Buffaloberry does not occur in dense patches like western snowberry, nor is it as widely distributed. However, its response to spring fires is very similar. In a few instances, buffaloberry abundance has been greatly reduced with hot fires in early August.

Prairie wild rose, western wild rose, and willows *(Salix* spp*)* apparently survive frequent fires fairly well even though there appears to be a small reduction in plant abundance after repeated fires.

Stems of older plants of Juneberry, hawthorn *(Crataegus* spp*)* and choke cherry are often killed by hot spring fires, but they can survive cool or incomplete burns. Sprouting of new shoots occurs in all three species after either spring or fall burns but is less pronounced after late summer or fall burns. Resprouting has been seen on areas with histories of five or six fires over a period of about 15 years.

## Effects of fire on emergent vegetation in prairie wetlands

Little is known of the environmental effects of fire in prairie wetlands (Kantrud 1986). However, wetlands often become choked with emergent vegetation and are in need of manipulation to increase cover interspersion (Linde 1969).

Vogl (1967) used fire to control woody plants in Wisconsin wetlands, and Truax and Gunther (1951) used fall and winter burns to control undesirable vegetation in Horicon Marsh, Wisconsin. Uhler (1944) stated that several wetland plant species were controlled in Minnesota marshes by burning when the substrate soils were dry.

Fire has been used extensively to open up dense stands of vegetation in marshes (Ward 1942; Uhler 1944; Schlichtemeier 1967; Ward 1968; van der Toorn and Mook 1982; Ball 1984). Burning has also been used to control plant succession and to promote the aquatic plants that produce seeds and roots for waterfowl foods (Grange 1949; Yancey 1964) and to improve the use of wetlands by breeding waterfowl (Evans and Black 1956).

### **Phragmites** *(Phragmites communis)*

Phragmites is an aggressive perennial of little value to waterfowl (Ward 1942), often forming dense stands in northern marshes.

Pratt and Andrews (1981) estimated the dry weight of an above-ground standing crop of phragmites to be as much as 0.23 lb/sq ft (1,118 gm/sq m). Stem densities up to 19 stems/sq ft (200 stems/sq m) have been recorded in a Utah marsh (Cross 1983). Weller (1981) stated that phragmites communities are very productive, having an estimated 9.4 T/A (21 metric tons per hectare) of emergent plant material produced per growing season.

Spring fires are effective in removing stem litter and creating openings in dense stands of phragmites (Ward 1942). Schlichtemeier (1967) observed an 85% decrease in accumulated dead stems after burning over the ice in February. Fires conducted in early spring can initiate an earlier emergence of overwintering buds (Cross 1983) and increase the opportunity for frost to damage new growth (Mook and van der Toorn 1982; Thompson and Shay 1984).

Generally, the thicker shoots emerge first (van der Toorn and Mook 1982; Cross 1983). When these shoots are damaged by fire or frost, plants respond by forming one or more thinner regrowth shoots (Mook and van der Toorn 1982; van der Toorn and Mook 1982; Thompson and Shay 1984).

Higher values for shoot densities, flowering shoot densities, aerial biomass, and carbohydrate reserves were observed after a mid-May burn at the Delta Marsh in Manitoba (Thompson and Shay 1984). These observations support Ward's (1968) statement that spring burns maintain the climax status of mature stands of phragmites.

After a July fire in the Delta Marsh, regrowth attained only half of its normal height and stem densities were reduced (Ward 1968). Thompson and Shay (1984) found that an August burn resulted in lower values for aerial biomass, standing crop of overwinter buds, shoot biomass, and flowering shoot densities the following summer.

The movement of nutrients from shoots to rhizomes for deposition in winter buds begins in mid-June (Mook and van der Toorn 1982). This is also the time when rhizome carbohydrate levels are at a minimum, suggesting that a fire in mid-June could have a more deleterious effect on stands than a burn conducted earlier in the growing season (Thompson and Shay 1984).

These observations suggest that summer burns have potential for thinning dense stands of phragmites.

October burns may enhance vegetative spread during the following growing season through higher below ground biomass and higher carbohydrate reserves (Thompson and Shay 1984). Higher aerial biomass and lower flowering shoot densities were also noted in these burns.

Burning of phragmites stands when the substrate is dry and the humidity of litter and stem bases is low can damage rhizomes. The effects of these intense burns vary from retarding emergence for 1 to 2 months (van der Toorn and Mook 1982) to burning deep into peat layers and destroying the rhizomes, thereby permanently eliminating stands (Ward 1942; Uhler 1944; Cross 1983).

In summary, phragmites can be maintained and even encouraged by using spring and fall burns. It also can be reduced and eliminated by using summer burns. Fire in stands of phragmites with dry substrates in late summer (June through August) combines the effect of burning when carbohydrate reserves are low with the potential of burning deep into organic soils. Such a burn could have a significant impact on the rhizome network of a dense stand of phragmites.

### **Whitetop** *(Scholochloa festucacea)*

Whitetop is a common hydrophyte in the shallow marsh zone of wetlands throughout the prairie pothole region (Stewart and Kantrud 1971).

Seasonal wetlands with flooded stands of whitetop were the preferred brood-rearing habitat of mallards (Talent et al 1982).

Whitetop growing on drier ground is used as nesting cover by waterfowl (Ward 1968). Shallow seasonal wetlands containing stands of whitetop frequently dry up by late summer and are mowed for hay (Diiro 1982; Neckles et al 1985).

Burning and mowing can increase the yield of whitetop (Smith 1973). Herbage production ranged from 2,744 to 13,436 lb/A (3,080 to 15,080 kg/hectare), with a production estimate of 10,246 lb/A (11,500 kg/ha) for burned areas.

Millar (1973) found that burned stands of whitetop apparently suffered no damage. Kantrud (1986) has suggested that whitetop is a fire-tolerant species. Shallow basins subjected to repeated burning and mowing will form pure stands of whitetop; grazing will eventually eliminate whitetop (Smith 1973).

The removal of litter enhances growth and increases shoot densities of whitetop on burned areas (Diiro 1982). Ward (1968) found that after a spring fire had opened dense stands of phragmites, whitetop growth was stimulated, stem densities increased, and whitetop invaded areas formerly dominated by phragmites. Diiro (1982) observed that whitetop plants grew most rapidly in seasonal wetlands that were burned on June 1.

Spring burns in wetlands that are not flooded after the fire have no significant increase in whitetop production (Diiro 1982). Therefore, spring burning is recommended to manage whitetop stands only in wetlands which will be flooded following a burn (Neckles et al 1985).

Fall burning removes litter and darkens the substrate, causing the soils to warm rapidly the following spring. This enhances shoot growth and increases stem densities (Diiro 1982; Neckles et al 1985). Diiro (1982) found that whitetop plants in fall-burned ponds were taller than plants in control ponds during early spring.

Production of whitetop was greater on fall-burned ponds than in any other burn treatment used (Diiro 1982). Smith (1973) stated that fall burning can increase production up to 55% if the area is flooded the following spring.

Because residual vegetation is removed during a fall burn, the amount of snow trapped in a burned wetland may be reduced. But, as with spring burns, those wetlands that are burned in the fall and receive sufficient runoff the following spring will have the highest production increase (Smith 1973; Diiro 1982; Neckles et al 1985).

**Cattail** *(Typha* spp*)*

Cattail has become a problem in many prairie wetlands because it often forms dominant monotypic stands (Linde et al 1976). These tall, dense monotypic stands are less attractive to wildlife (Kantrud 1986). Fire is often used to increase interspersion in cattail stands (Uhler 1944; Beule 1979; Ball 1984).

Some studies have shown that fire is not an effective means of controlling cattail (Beule 1979; Gorenzel et al 1981). In a Utah marsh burned in September, cattail growth the following summer had higher shoot weights (Smith and Kadlec 1985) and higher protein content (Smith and Kadlec 1984) than cattail from control areas. This would suggest that cattail stands may even be enhanced by fire, depending on the conditions. Nevertheless, under proper conditions, fire can control cattail. Interspersion will improve by burning over the ice. Stem densities were reduced by 70% and no fruiting heads were formed on areas burned over the ice and flooded the following spring (Ball 1984).

Burning or mowing cattail over the ice is less effective in eliminating cattails when the remaining stubble is not flooded the following spring.

Cattail rhizomes are supplied with oxygen during the dormant season by old stems extending above the water surface (Linde et al 1976). Removal of these stems by burning and subsequent flooding of the stubble the following spring will cause anaerobic conditions to develop in the rapidly growing shoots (Ball 1984), causing many shoots to die before emerging above the water surface.

Ball (1984) also concluded that burning over the ice is a practical technique for improving interspersion when water levels are adequate to submerge stubble the following spring. He also noted that backing fires left shorter stubble above the ice, requiring a smaller increase in water levels to flood stubble the following growing season.

Like phragmites, cattail can also be killed by burning when the substrate is dry. Uhler (1944) stated that "root burns," which occurred when the soil was dry 3 to 6 inches (8 to 15 cm) below the surface, provided long-term control of cattail. Beule (1979) noted that occasionally a fire that had burned into the peat layer of a dry marsh would kill cattails.

Linde et al (1976) found that the total non-structural carbohydrate levels in cattail stands at the Horicon Marsh reached a minimum in late June. Therefore, burning in late June or early July in wetlands with dry substrates could be a potentially effective technique for killing cattail. The use and effectiveness of such a burn would depend on the ability to draw the water levels down enough to dry the substrate surface.

**Bulrush** *(Scirpus* spp*)*

A February burn over the ice in Nebraska reduced stem densities of bulrush *(Scirpus* spp*)* by 60% (Schlichtemeier 1967).

The annual production of bulrush species in a Utah marsh was not affected by a burn conducted in September (Smith and Kadlec 1985). However protein levels were higher in hardstem bulrush *(S. acutus)* (Smith and Kadlec 1984) and shoot weights lower in bulrush *(S. lacustris)* (Smith and Kadlec 1985) following the same burn.

**Carex** (*Carex* spp*)*

Sedge communities were maintained with annual burning in Wisconsin wetlands (Thompson 1959). Millar (1973) found no change in sedge stands after repeated burning, suggesting fire tolerance.

**Spikerush** *(Eleocharis* spp*)*

Stands of spikerush *(Eleocharis palustris)* also appeared to change little after repeated burns (Millar 1973).

In summary, wetland grasses and sedges can be enhanced with properly timed, less intense burns. In contrast, a slow moving fire which would burn deep into the organic soil or peat of wetland substrates will have an impact on all hydrophytes (Uhler 1944; Yancey 1964; Millar 1969). Uhler (1944) noted that such a fire (called a "root burn") provided a control of phragmites, cordgrass *(Spartina* spp*)*, cattail, river bulrush *(Scirpus fluviatilis)*, sedges, and other hydrophytes.

The use of a "root burn" fire is limited to marshes that can be completely drawn down or those marshes experiencing severe or periodic drought.

## Effects of fire on insects

Probably the best example of the use of fire to control insects was fall or winter burning on the True Prairie in Kansas to manage cinch bug populations (Hayes 1927).

However, grasshoppers are the principal above-ground insect consumers; therefore, it is no surprise that the effect of fire on grasshopper populations has been studied more than for other species.

Knutson and Campbell (1976) found that early spring burning caused grasshoppers to emerge 3 weeks earlier than normal and grasshoppers were higher in number the second year following an early burn. Midspring burning produced fewer grasshoppers than early burning, and late spring burning produced fewer grasshoppers than mid- or early spring burning.

Nagel (1973) quantitatively measured the effect of a single spring burn on the biomass and density of arthropods in the native True Prairie near Manhattan, Kan. He measured herbivorous, non-herbivorous, and total arthropods both at night and during the day once every 2 weeks from June 6 to August 26 on formerly grazed, burned, and unburned pastures. Grazing was controlled on the burned pasture to prevent overgrazing of the burned areas. Both pastures were on a similar upland range site.

He found that the burned area contained significantly greater numbers of arthropods and greater biomass than the unburned area.

The numbers of non-herbivorous insect species were equal on both areas, but non-herbivorous insect biomass was higher on the burned than on the unburned areas. Greater numbers and less biomass were collected during the day than at night, mostly due to higher numbers of *Diptera* (flies) collected during the day.

Arnett (1960) found that areas burned in late March produced greater grasshopper populations than heavily grazed areas in this same general area (Nagel 1973).

Timing is a major factor in insect fire ecology. Early spring burning results in earlier emergence and higher numbers of grasshoppers than a late spring burn, especially if coupled with heavy grazing pressure (Knutson and Campbell 1976; Arnett 1960).

Cancelado and Yonke (1970) also found greater population differences "from the beginning to the middle of the growing season than later in the year, where they are reduced or are not apparent...." Too much or too little litter decreases grasshopper populations. Late spring burning reduces litter and kills many of the grasshopper nymphs directly.

The general consensus seems to be that late spring burning reduces insect populations more than early spring burning. This is probably due to mortality of newly hatched insect nymphs during the burn, which reduces the population potential of subsequent generations.

Cancelado and Yonke (1970) tested the effect of spring burning on *Hemiptera* (sucking insects) and *Homoptera* (cicadas, hoppers, aphids, and scale insects) at the Tucker Prairie Research Station in east-central Missouri, burning 30 acres (12 ha) of a 145-acre (59 ha) site on March 23, 1968.

Significantly higher numbers of Cicadellidae *(Homoptera)*, Miridae, and Lygaeidae *(Hemiptera)* occurred on the burned area than on the unburned area.

In the Waubun Prairie of northwestern Minnesota, Tester and Marshall (1961) conducted a fall burn (October 28, 1957), early spring burn (March 15, 1958), and a late spring burn (April 11, 1958) on upland sites hayed since 1921. The early spring site burned very hot due to high winds and then was grazed during the study period. The late spring site was not grazed.

Most grasshoppers came from areas having light to moderate amounts of litter. Optimum grasshopper habitat consisted of vegetation that was recovering from burning rather than that freshly burned or long unburned.

There was no significant change in Coleoptera numbers during the first year, but in the second year both the spring and the fall burns contained high numbers of Coleoptera. This was correlated to the sparse litter found on both sites.
Lussenhop (1976) studied the effect of fire on soil arthropods at the Curtis Prairie near Madison, Wis. Burning of re-established tallgrass prairie that had been burned biannually since 1950 was continued for two more burns on one area and discontinued on the other. As a control, a third area was raked to remove the litter. Soil microarthropods were counted about 2 weeks after the burns and again about 7 months after the burns.

First-year readings showed no significant difference in soil microarthropods, but by the fourth year the unburned areas had significantly fewer herbivore and carnivore species than the burned and control (raked) areas.

Lussenhop concluded that the unburned area was less productive in vegetation and roots, therefore soil microarthropods decreased in specific numbers.

Rice (1932) studied the effect of fire on prairie animal communities following spring burning. At least two burns were studied, one of which occurred March 12, 1932, near Seymour, Ill. She found "10 charred spider cocoons/sq m, dead tenebrionid beetles, carabid beetles and cutworm larvae at 3/sq m, and living under a 3 by 8-inch piece of charred wood, 19 chinch bugs, 2 cut-worm larvae, 1 ground beetle, 1 slug and 2 centipedes. Aulax larvae in stems of *Silphium* and *Lactuca*, *Eurosta* and *Lepidoptera* larvae in goldenrod stems were killed."

Mortality was severe, compared to eight living larvae/stem in unburned areas. Ants increased in numbers, and earthworms came to the surface earlier but then decreased with decreasing soil moisture the first month after the burn.
Rice (1932) found that insects (minus ants) were reduced 35%, not all insects were killed outright by the spring burn, those insects not killed by the fire fled to adjacent unburned areas shortly after the burn, soil temperatures were not high enough to kill all animals in hibernation, gall insects suffered very high mortality rates, and ants and other underground organisms were not immediately affected by the fire.

In the southern mixed prairie, fire will top-kill mesquite and leave the stems in a state where wood borers will attack and aerate them to such an extent that they will easily be consumed by a reburn (Wright and Bailey 1982).

Coleopterans and Hemipterans appear to increase following burning (Cancelado and Yonke 1970; Winter 1984). This may also be due to increased productivity following burning (Lussenhop 1976). Insect exoskeletons contain nitrogen which becomes available to plants after fire (Kirchner 1977).

Gall insects suffer high mortalities from fire. However, not all are killed; many flee or escape by hiding under rocks or other objects, and ants actually increased in one study (Rice 1932).

Winter (1984) in a Butte County, Idaho, study of sage sparrows *(Amphispiza belli)* and Brewer's sparrows *(Spizella breweri)* found that a September 5, 1982, headfire burn of sagebrush-grassland at 81 F (27 C) air temperature, 29% relative humidity, and 8.5 ft/second (2.6 m/second) wind had increased flixweed *(Descurainia sophia)* in the burned areas. Lepidopteran larvae were very abundant on the plants.

Sage sparrows spent less time foraging after the burn. The author attributed this decrease primarily to greater arthropod abundance. Greater numbers of arthropods occurred on the burned plots than on unburned areas during July of the year after the fire. This increase consisted largely of larvae.

It seems possible that a shorter foraging time of certain bird species may be a good indicator of greater insect densities. In a study of this same area in Idaho, Petersen and Best (1986) found that, although fire changed the vegetation, it did not affect the composition of nestling diets or food size of the sage or Brewer's sparrow.

In the southeastern U.S., bobwhite quail *(Colinus virginianus)* are called "fire birds" because they can be found at the edges of burns before the fire even stops smoking. They fill their crops in a matter of minutes because dead insects and seeds are so readily abundant (Stoddard 1963).

Fire to control insects and diseases in crop residue is age-old. But the effect of fire on grassland insect populations is not as well understood or documented.

Insects, especially grasshoppers, are an important herbivorous component of grasslands. Nematodes represent an even larger potential with a biomass of at least 10 times greater than above ground invertebrates (Risser et al 1981).

Fire causes an immediate decrease in insect populations (except ants and other underground species), followed by a gradual increase in numbers as the vegetation recovers. The insects eventually reach a population level higher than adjacent areas, then decline to near preburn levels as vegetation and soil litter stabilize.

## Effects of fire on nongame birds

Bird species evolving with fire may show fire-adapted behavior and responses, whereas other species exposed infrequently to fire in their evolutionary histories may be severely inhibited by it (Best 1979).

### Habitat selection

The selection of breeding habitat by birds is based on the recognition of vegetation structure (foliage patterns and density) which fit preconceived notions of "home." Removal or modification of any vegetation, whether by burning or heavy grazing, reduces the diversity of bird species (Lack 1933).

A spring burn (late April) in shrub-grasslands in Illinois did not cause major changes in field sparrow *(Spizella pusilla)* territory configurations. There was no male abandonment after the burn. Best (1979) concluded that burning in March or early April may interfere with the process of site selection and ultimately result in reduced population densities because of vegetation structure alterations.

In sagebrush grasslands in Idaho, male sage sparrows expended significantly more time in territorial maintenance after a fall burn than before, while Brewer's sparrows spent about the same amount of time (Winter 1984).

Spring burns in Wyoming in sagebrush-grasslands initially reduced the breeding pair density of green-tailed towhees *(Pipilo chlorurus)*, vesper sparrows *(Pooecetes gramineus)*, and white-crowned sparrows *(Zonotrichia leucophrys)*, but their breeding densities increased 2 years after the burn (McGee 1976).

Breeding pair densities of lark sparrows *(Chondestes grammacus)* in central Texas were highest in the most recently burned areas (Renwald 1977). They decreased with increased litter buildup and lower grass production, due to large areas being taken over by old, decadent stands of tobosagrass *(Hilaria mutica)*.

Grasshopper sparrows *(Ammodramus savannarum)*, the only birds significantly affected by spring burning treatments in the grasslands of South Dakota, decreased in numbers immediately after fire. Western meadowlarks *(Sturnella neglecta)* generally decreased after the burns, while vesper sparrows increased (Forde et al 1984).

Huber and Steuter (1984) found similar trends in grasshopper sparrows and western meadowlarks after a spring grassland burn. Grasshopper sparrows were not present on the burned plots one month after the fire. Western meadowlarks decreased slightly after the fire, but within 2 months numbers were greater on the burned plots than on the control plots.

On a grassland in Minnesota, Tester and Marshall (1961) found that the presence of bobolinks *(Dolichonyx oryzivorus)*, savannah sparrows *(Passerculus sandwichensis)*, and LeConte's sparrows *(Ammodramus leconteii)* was positively correlated with the amount of litter cover present on the study site.

All three species declined after burns (fall and spring), though the authors believed there were other factors involved.

No bobolinks were present on the burned plots until after one season's litter had accumulated. Savannah sparrows showed similar responses; they required more than 2 years of litter accumulation. LeConte's sparrows appeared to need a moderate amount of litter cover in wet meadow zones. This species was not present until after one season of litter had accumulated.

### Nest site selection

Habitat alteration by fire may change the nesting behavior of some birds. Kirsch et al (1978) stated that nongame birds that nest in upland areas are influenced by the amount and quality of available vegetation.

Winter and Best (1985) found a significant difference in nest placement between pre-burn and post-burn nesting seasons. The year previous to their burn, all sage sparrow nests found had been built within sagebrush canopies. After the burn, 17% of the nests were located in depressions on the ground under small sagebrush plants, and one nest was located in a bluebunch wheatgrass *(Agropyron spicatum)* clump.

Nest placement also differed significantly the first post-burn year. Fifty percent (6) of all early nests were built in sites other than sagebrush canopies, while all late nests (17) were within sagebrush plants. They concluded that the reduction of available sagebrush plants by fire required some of the sage sparrows to use areas other than their preferred habitat to obtain enough cover and concealment for their nests.

### Foraging behavior

The immediate effect of fire on bird populations depends greatly upon the season and intensity of the burn. A relatively cool fire during the dormant season could greatly increase food sources (Wright and Bailey 1982). Birds are adapted to eat particular kinds of food, and the birds' abundance may depend largely on the supply of the appropriate kind of food (Bendell 1974).

Best (1979) found that the major impact of burning on foraging behavior was to make plant foods accessible, particularly grass seed that was unavailable before the burn because of the accumulation of grass litter. After burning, field sparrows were frequently observed picking up seeds from among the ashes.

Other bird species, especially wood thrushes *(Hylocichla mustelina)* but also gray catbirds *(Dumetella carolinensis)* and chipping sparrows *(Spizella passerina)*, fed more frequently on the study area after the burn than in previous years.

McGee (1976) also found an influx of non-breeding birds to the burned areas in his study. He attributed this to the increased availability of plants and insects as food items.

Winter (1984) found sage sparrows spent significantly less time foraging in the post-burn period. Evidently, the foraging efficiency of sage sparrows increased after the fire, whereas Brewer's sparrow foraging efficiency remained unchanged.

Sage sparrows and Brewer's sparrows partitioned food resources by their foraging behavior. Brewer's sparrows foraged more often in the outer foliage of sagebrush than did sage sparrows, but sage sparrows utilized grasses, forbs, and bare ground more often than did Brewer's sparrows.

Fire caused male Brewer's sparrows to fly farther to unburned patches to forage, while the sage sparrows foraged in the burned areas. Differences in their foraging behavior after the fire reduced competition between these species. Winter (1984) also found that in late July burned patches contained more arthropods than unburned areas; with the reduced vegetative cover after burning, there was increased arthropod accessibility.

There is general agreement that fire reduces breeding pair density, alters nest site selection, and changes foraging behavior, at least during the first breeding season after the burn. The duration of the impact depends on the extent of the habitat alteration.

Forde et al (1984) found that two or three breeding seasons were required to increase bird species numbers to pre-burn densities.

The frequency of fire may render a habitat unsuitable for use by a given species, depending on its habitat requirements. When prescribed burning creates a fine-grained mosaic with good interspersion of habitat types and with a maintained edge, the greatest number of species requiring subclimax vegetation will benefit (Best 1979; Winter 1984).

## Effects of fire on upland gamebirds

Upland gamebirds of the NGP include mourning doves *(Zenaida macroura)*, woodcock *(Scolopax minor)*, and galliforms. We exclude wild turkey *(Meleagris gallopavo)*, which is considered a big-game species, and ruffed grouse *(Bonasa umbellus)*, because of its strong association to woodlands.

### Mourning doves

The only research project specifically designed to evaluate the role of fire on dove nesting was done in west Texas (Soutiere and Bolen 1973) in mesquite *(Prosopis* spp)/shortgrass areas. Their major discovery was that, when woody species (mesquite) were removed, doves reverted to ground nesting with at least equal nest success. Otherwise, burning had little impact on dove nesting.

Several researchers (Kirsch and Kruse 1973; Lawrence 1966; Kruse and Piehl 1986) noted doves within their study areas but made no inferences. In Illinois, Edwards and Ellis (1969) observed several doves that flew only 10 to 20 ft (3-6 m) above flames and landed on warm ashes.

### Greater prairie chickens

In Illinois, greater prairie chicken *(Tympanuchus cupido pinnates)* nest densities increased from one nest per 9.3 acres (3.8 ha) to one nest per 6 acres (2.4 ha) for second, third, and fourth year post-burn sites (Westemeier 1973). These increases came after both spring and fall burns, the difference being the selection for cool-season grasses by burning in the fall (August) and the selection for warm-season grasses by burning in the spring (March).

Tester and Marshall (1961; 1962) stated that greater prairie chicken nesting rates in Minnesota would probably be at a minimum in the year following a burn. They suggested a 4-year prairie management schedule of burning, no treatment, grazing, and no treatment again.

Svedarsky (1979) recommended against burning of preferred prairie chicken nest habitats in spring in northwestern Minnesota. He did recommend fall burning of willow lowlands to create better brood habitat.

Anderson (1969) reported male greater prairie chickens used a lek (booming ground) only 1 day after burning.

### Sharp-tailed grouse

Sharp-tailed grouse *(Tympanuchus phasianellus)* in Manitoba appeared to select a burned lek (dancing ground) over an unburned one.

The preference was quite likely due to changes in vegetation structure. The two leks were 525 yards (480 meters) apart (Sexton and Gillespie 1979). Ammann (1957) proposes that fire and lek use by males are related.

Four of five sharp-tailed grouse nests which were active during a spring burn in North Dakota eventually hatched (Kruse and Piehl, 1986). Kirsch and Kruse (1973) found two to three times as many nests on spring burned areas compared to unburned areas in North Dakota.

### Sage grouse

There is a lack of conclusive information that compares burned versus unburned situations in sage grouse *(Centrocercus urophasianus)* management, according to Klebenow and Beall (1978). Braun et al (1977) provide guidelines for managing sage grouse habitat, but they do not mention any effects of fire as a management tool.

Sage grouse habitat suffers in value as a direct result of attempts to convert sagebrush to grasslands (Braun et al 1977). Klebenow and Gray (1968) preferred fire over herbicides for managing sagebrush because fire does not remove all forbs. Seeds of forbs, including sagebrush seeds, are important food for sage grouse.

### Northern bobwhite

Although northern bobwhites *(Colinus virginianus)* occur only incidentally in the NGP, they deserve mentioning.

In Illinois, Ellis et al (1969) compared three management systems for bobwhites: providing food patches (e.g. grains), prescribed burning with share cropping, and burning alone. All burning was done in late winter or early spring. They found the burn-share crop system to be the most productive and efficient, followed by burning alone and the food patch system.

In Nebraska, Erwin and Stasiak (1979) found two bobwhite nests destroyed by early spring prescribed fires. They did not note any successful nests.

In Illinois, Edwards and Ellis (1969) observed four bobwhites flying directly to a burn and landing within a few meters of the flames. They also reported observing a covey of quail flushed by fire and flying about 88 yd (80 m) away from the flames. Since there was no disorganization of the covey in flight or in landing, they surmised that the quail were relatively unafraid of the flames. They concluded that bobwhites respond rapidly to fire by immediately utilizing recently burned sites.

### Other species

Ring-necked pheasants *(Phasianus colchicus)*, gray partridges *(Perdix perdix)*, and woodcocks were only occasionally mentioned in the fire literature.

Erwin and Stasiak (1979) observed 38 ring-necked pheasant nests destroyed in seeded native grassland by early spring prescribed fires in Nebraska. No successful nests were mentioned.

Edwards and Ellis (1969) observed a single "peenting" woodcock which flew from brushy cover and landed within 20 ft (6 m) of flames from a spring prescribed burn. The woodcock then initiated normal courtship behavior, alternating peenting with landing near the flames. They attributed this seemingly unconcerned behavior with fire adaptation.

In summary, recently burned areas appear to be attractive to greater prairie chickens, sharp-tailed grouse, and northern bobwhites. These species also appear to increase in density in burned versus unburned areas.

Mourning doves have not exhibited significant population changes in response to burning. However, they have shown a change in nesting habitat selection, from trees and shrubs in unburned areas to ground nesting in burned areas.

Woodcocks, ring-necked pheasants, and gray partridges have been insufficiently researched to draw any specific conclusions.

We would expect that species that have evolved within the grassland environment would also have become more fire tolerant and perhaps more fire dependent than those that have not.

There is a great void in information which relates fire effects and the life cycles of upland game in the NGP. Even when studies have been made, they have not been replicated, which limits interpretation between populations.

Nevertheless, although we lack complete information on fire and upland game bird relationships, Kirsch and Kruse (1973) believe that, in general, we have enough basic information to use fire as an effective management tool.

## Effects of fire on waterfowl

Only in very recent times have scientists examined if vegetative burning, both natural and prescribed, is harmful or beneficial to waterfowl and shorebirds. The response of waterfowl to burned areas was usually noted only after burns, as remnants of nests or eggs were found.

In most cases birds are affected more by the abrupt habitat change than the fire itself.

Researchers believe fire suppression has greatly reduced the extent of waterfowl nesting habitat because some grassland habitat has reverted to woodlands.

Vogl (1967) stated that areas in Wisconsin that used to produce thousands of ducks are now forested and produce few ducks. Kirsch and Kruse (1973) speculated that the highest populations of prairie nesting ducks in the Dakotas occurred around 1880, after the decimation of big game herds had reduced grazing and before settlement introduced fire suppression to prairie vegetation.

Most information on the response of waterfowl to burning concerns spring burns. The obvious immediate effect of spring fires on upland nesting waterfowl is destruction of nests and their contents by fire (Leedy 1950; Moyle 1964; Erwin and Stasiak 1979).

Adult birds often return to a nest after a fire and try to resume incubation. Bent (1923) observed a northern pintail *(Anas acuta)* incubating a clutch of scorched eggs immediately after a burn. Leedy (1950) found a mallard *(A. platyrhynchos)* nest with scorched eggs plus four that had been laid after a fire. He also found an American black duck *(A. rubripes)* still incubating a nest with twelve scorched eggs. Moyle (1964) mentioned similar cases of hens continuing to incubate eggs damaged by a prairie fire. Fritzell (1975) saw a green-winged teal *(A. carolinensis)* remove one burned egg from a nest and later lay four more.

Kruse and Piehl (1986) found that in North Dakota prairie burns there are sometimes "skips," or areas that remain unburned where active nests are not affected. These "skips," usually dense patches of shrubs with little or no understory vegetation, are also utilized by birds that start nesting after the fire.

Kruse and Piehl (1986) also found 20 clutches of eggs in the unburned vegetation on the burns; 15 hatched. They concluded that burning an area during the nesting season does not necessarily eliminate all ground nesting in the area for that year.

Kirsch and Kruse (1973) compared nesting on similar plots of unburned and burned prairie for several years following burning. They found that 52% of the duck nests on burned grassland habitat were successful, compared to 33% on unburned areas. During the second season after the fire, duck production was greater on the burned plot than on the unburned.

They also noted that the greatest measured change in vegetation after burning was a marked increase in plant variety. Burning also changes the growth form and pattern of nesting cover, which may make it more attractive to nesting waterfowl.

Prescribed burning to improve nesting cover has been practiced mainly in spring. A major concern is the presence of active nests, which can be avoided by fall burns.

Higgins (1986b) compared nesting success of waterfowl on mixed prairie areas burned in spring with those fall burned in North Dakota. Duck nesting success was greater in the fall burn plots than in spring burn plots, when all species were combined and when success was compared during the first few post-burn years.

He found that in the first spring after a fall burn there is little cover available for nesting and the area is sparsely utilized. However, the second year after a fall burn, the available nesting cover is much taller and denser than on spring burn areas, and ducks had greater nesting success on fall burn plots.

Upland waterfowl nesting response was nearly equal between the spring and fall burns after the third post-fire growing season. Higgins (1986b) concluded that duck production can be greater on fall burns than on spring burns, if averaged over 3 or 4 post-burn years.

Areas recently burned are sometimes utilized by nesting waterfowl. In Iowa, Glover (1956) observed blue-winged teal *(A. discors)* initiating nests in May after an April burn, and Messinger (1974) found more duck nests on burned versus control plots but with 1973 nesting success reduced on plots burned April 5, 1973. Keith (1961) found 17 northern pintail nests on bare ground after an April burn, with some very near unburned areas with good cover. No other duck species used this burned area.

Fritzell (1975) found higher rates of nesting success on burned as compared to unburned areas but fewer nests per unit area in Manitoba. He stated that spring burning is more detrimental to early nesters such as mallard and pintail than to later nesting species. He also mentioned that mallards may be particularly susceptible to spring burns due to their preference for heavier cover which often burns.

Fritzell (1975) also concluded that controlled burning is an efficient tool in wildlife management, but indiscriminant annual burning reduced the quality and quantity of waterfowl nesting cover.

Fire can benefit waterfowl in ways other than improved nesting cover. Prescribed burning is used as a marsh management tool to burn out thick growths of cattails and phragmites. This increases the edge cover which improves brood habitat. Marsh burning can also initiate the growth of preferential duck food (Vogl 1967).

Ward (1968) reported that both spring and summer fires are used for marsh management at Delta, Manitoba. The spring fires are set prior to April 20 when mallards and pintails begin nesting. The primary purpose of the spring fires is to create more edge for nesting and brood cover. Summer fires have a greater effect on regrowth and are directed toward lasting changes in the plant community. At Delta, summer burns were used to remove phragmites, because it was seldom utilized by waterfowl, and to enhance the growth of whitetop (Ward 1968).

## Effects of fire on shorebirds

Some research has evaluated the effect of fire on shorebird nesting habitats. Vogl (1973) found in Florida that burned shorelines along wetlands increased use by shorebirds such as common egrets *(Casmerodius albus)* and great blue herons *(Ardea herodias)*. The birds were attracted to the shallow-water fishing ground made available when the fire removed the heavy grass mat that covered the shallow flats on the shorelines.

In Minnesota, Niemi (1978) found killdeer *(Charadrius vociferus)* were attracted to recently burned shorelines.

Kirsch and Kruse (1973) found more upland sandpiper *(Bartramia longicauda)* broods were produced on burned grasslands than on unburned or grazed areas. Kirsch and Higgins (1976) reported that mean production of upland sandpipers was highest on prairie managed by prescribed burning during 2 out of 5 years. They suggested rotational burning at 3-year intervals.

Huber and Steuter (1984) also noted that upland sandpipers made greater use of areas previously burned than of unburned areas. After a May 3 burn in South Dakota, they found 50 upland sandpipers in the burned area in June; the unburned had none. In July the burned area had 24 and the unburned had six.

In summary, prescribed burning is a valuable management tool for upland nesting birds in grassland areas. The optimum timing and frequency of the prescribed burns is still being researched.

Kruse and Piehl (1986) stated that land managers who burn in the spring should consider partial burns if they are concerned about nesting birds. These burns have less impact on total vegetation changes but can result in higher recruitment rates than complete burns.

Higgins (1986b) surmised that annual fall burning would be harmful to wildlife due to the lack of residual nesting cover and suggested that to enhance waterfowl production burning should be done every other year at most.

## Effects of fire on small mammals

Although most research indicates limited direct mortality to rodents, several instances have been reported.

Many nests of the western harvest mouse *(Reithrodontomys megalotis)* in Nebraska were destroyed by fire, and an estimated 205-522 pups were killed over the entire burn (Erwin and Stasiak 1979). Of 41 mice marked in a pre-burn area by Tevis (1956), only 11 were recaptured post-burn. The rest were presumed dead.

After a fire, Chew et al (1958) found carcasses of 28 dusky-footed woodrats *(Neotoma fuscipes)* and four mice of three species. Few of the carcasses had been charred or singed; cause of death was asphyxiation or heat prostration.

Motobu (1978) estimated 51% mortality in mountain beaver *(Aplodontia rufa)* on an area completely burned and only 20% mortality on an area of patchy burn. Few of the surviving animals showed signs of burn injury.

An immediate, indirect cause of mortality from burning is predation. The lack of cover immediately after a fire produces an exposed environment and improves accessibility to avian and mammalian predators (Motobu 1978). Beck and Vogl (1972) suggested that some of the mortality associated with fire may have actually been caused by predation. Post-burn predation may be more restrictive to rodent populations than the burning itself (Lawrence 1966).

The lethal temperature tolerance of rodents is 122-145 F (50-63 C) at 22% relative humidity (Howard et al 1959); however, at 60% relative humidity, the lethal temperature drops to 120 F (49 C) (Lawrence 1966). To escape the heat of a fire many rodents take refuge in unburned islands (Motobu 1978), in rock outcroppings (Howard et al 1959), by running ahead of flames (Erwin and Stasiak 1979), or by taking refuge in burrows (Lawrence 1966; Quinn 1979). Beneath the soil surface, temperatures are reduced (Lawrence 1966) and rodents are able to survive.

Lawrence (1966) demonstrated the necessity for adequate air circulation in the burrow system. He also suggested that animals survive as long as the burrow systems allow vapor pressure below 40 mm Hg.

Fires affect population densities principally by altering habitat. The decrease of vegetative cover results in fewer microhabitats available for use by wildlife, especially rodents. However, with the reduction of ground litter, primary production is enhanced. Within 2 to 4 years after a fire, litter gradually increases again, with a decrease in primary production (Dix 1960; Vogl 1965; McGee 1982). Based upon these habitat changes and the habitat and food preferences of rodents, major shifts in species composition and density should also occur within the first few years after a fire.

The major changes in food availability affect the type of species that will invade after a fire. Removal of the litter layer increases availability of seeds and invertebrates for granivores and omnivores (Ahlgren 1966; Stout et al 1971; Kaufman et al 1983). For the first year, these type of rodents are abundant. Species considered herbivores are limited, especially on complete burns.

As the abundance of seeds decreases, so does the population of granivores. However, by the third year new seed producing vegetation has become established and the seed eating rodent populations increase (Ahlgren 1966; Sims and Buckner 1973).

Depending upon climatic conditions, concealment vegetation will develop after 2-5 years. This allows herbivores and those rodents restricted by lack of cover to recolonize an area and reach populations similar to preburn levels (Gashwiler 1970; Fala 1975; McGee 1976).

Many studies show the rate of capture of deer mice *(Peromyscus maniculatus)* in geographically diverse post-burn habitats is significantly greater than in unburned habitats (Cook 1959; Tester 1965; Ahlgren 1966; Stout et al 1971; Beck and Vogl 1972; Sims and Buckner 1973; McGee 1976; Bock and Bock 1978, 1983).

Deer mouse populations show a positive response to the early stages of secondary succession (Beck and Vogl 1972; Kaufman et al 1983). They prefer xeric habitats with open vegetation and sparse litter cover (Kaufman et al 1983) and are restricted from areas of dense vegetation (Rickard 1960). They are opportunistic omnivores (Johnson 1961), often shifting diets according to the availability of seeds and invertebrates (Williams 1959; McGee 1976). Their food and habitat preferences make them particularly suited to exploit burned areas.

Deer mice will usually invade an area within 2-4 weeks after a fire (Cook 1959; Tevis 1956; Sims and Buckner 1973). This immigration is a response to the availability of a new food source and to the open space in which a home range may be established (Tevis 1956). Many of the colonizing mice are juveniles (Tester 1965; Stout et al 1971; Sims and Buckner 1973). Sadleir (1965) reported that although deer mice are not territorial, adults become intolerant of juveniles and will drive them out during the breeding season.

Within 3 years, deer mouse populations on a burned area will increase greatly over that of an unburned area (Cook 1959; Tevis 1956; McGee 1976; Bock and Bock 1983; Kaufman et al 1983). These increases may be caused by additional immigration or increased reproductive rates in response to favorable environmental conditions (Lawrence 1966; McGee 1976). Deer mice remain the dominant species for 2-4 years until the accumulation of vegetation becomes too dense for optimum habitat (Rickard 1960; McGee 1976).

The western harvest mouse, a granivore, will also inhabit a burn, but tends not to invade until some vegetative cover is established (Cook 1959; Kaufman et al 1983).

If western harvest mice responded favorably only to the availability of seeds, densities should peak early in the first year, as with deer mice. Therefore, habitat deficiencies must be the limiting factor in this species' response (Kaufman et al 1983). Kangaroo rats *(Dipodomys* spp*)* and pocket mice *(Perognathus hispidus)* also utilize burned areas (Bock and Bock 1978; Quinn 1979). Both of these species are also granivores (Johnson 1961).

Ground squirrels *(Spermophilus* spp*)* and chipmunks *(Eutamias* spp*)* are common in burn areas but are limited by the amount of remaining vegetation (Gashwiler 1970; McGee 1976). House mice *(Mus musculus)* also show a preference for habitat created by fire (Cook 1959). Other species may utilize a burned area depending upon the surrounding habitat types and the amount and type of vegetation that becomes established after a burn.

Not all rodent species are positively affected by fire.

Herbivores are generally absent or in low densities after a burn (Fala 1975). Voles *(Microtus* spp*)* are restricted to habitats with dense vegetative cover in which to build runways (Rickard 1960; Sims and Buckner 1973; McGee 1976). Populations of voles are usually low for the first 2-4 years following a fire, until undergrowth accumulations reach that of unburned areas (Cook 1959; McGee 1976). Tester (1965) found red-backed vole *(Clethrionomys gapperi)* densities to be unaltered by fire, but others have found this species to respond like *Microtus* species (Ahlgren 1966; Beck and Vogl 1972; Gashwiler 1970). Jumping mice *(Zapus* spp*)* are also restricted due to lack of food and cover (Sims and Buckner 1973; McGee 1976).

The small mammal response is not considered a direct response to fire but a reaction to fire-altered habitat. Fire alters the composition of rodent species from those associated with the climax community to those considered early successional species (McGee 1982).

There is a predominant shift from chaparral species (Cook 1959; Lawrence 1966) and forest species (Beck and Vogl 1972) to prairie and grassland species. Food and habitat resources are the primary factors influencing the population shifts and fluctuations. Granivores and omnivores that require little cover (deer mice, for example) are favored. As vegetative cover increases on burned areas, other rodent species also invade.

Eventually, litter accumulation, flora, and the rodent community again resemble those of an unburned area.

## Effects of fire on large mammals

Fire and fire-perpetuated environments, such as grasslands, have been of the utmost importance in the evolution of mammals. The lives of many mammal species today are also directly or indirectly affected (Handley 1969).

The potential lethal hazard of fire for large mammals depends on a combination of variables. Fire can be and often is a disaster for animals dwelling in forests or other places where fires are infrequent. But mammals living in environments exposed to frequent fires, as in grasslands, survive because of their adaptations (Handley 1969).

Plains Indians extensively burned the prairies to attract the roaming herds of bison (Higgins 1986a). But Europeans brought with them an ingrained fear of fire. They suppressed fire without any awareness of its part in the maintenance of grassland communities. Reduction of fire in a fire-evolved system promotes the development of advanced successional stages which may not provide the optimum ecological conditions for large mammals adapted to fire (Gruell 1983).

Since fire is a natural part of the environment for many animals, Komarek (1969) hypothesized that these animals lack an innate fear of fire and that some sensing mechanism and behavior patterns certainly must give warning in sufficient time for large mammals to move out of danger. His observation of large mammals showed their relative disregard of fire.

Ivey and Causey (1984) reached a similar conclusion in a study of radio-tagged white-tailed deer *(Odocoileus virginianus)*. Immediate and short-term responses of deer during burning activities showed deer to use streambeds and other moist sites as refuges from fire. Deer were observed feeding to within 65 ft (20 m) of approaching fire with no apparent alarm. At no time were deer observed running in response to fire.

Ivey and Causey (1984) also reported that burning up to 70% of a home range did not cause deer to change their home range.

Natural fires in grasslands usually burn cool near the ground, then progress in a discontinuous front, leaving a mosaic pattern of burned and unburned areas (Handley 1969). This allows large mammals to avoid fire and leaves shelter and mature food sources near burned areas.

### Habitat suitability

Suppression of fire results in gradual changes of ecological conditions with long-term consequences, including deterioration and loss of some important wildlife habitats (Gruell 1983). Wildlife responses to changes in their habitats are largely determined by species requirements and frequency of disturbance (Gruell 1983). In the absence of fire, advanced plant succession has had profound effects on the capability of habitat to support wildlife.

Grass and forb eating species that do not exhibit strong requirements for abundant escape cover, such as pronghorn antelope, bison *(Bison bison)*, and bighorn sheep *(Ovis canadensis)*, are favorably influenced by the increase in the grass component of habitats after fire. We could expect that fires would also have a favorable influence on wintering elk *(Cervus elaphus)*, which are primarily grass foragers in these ecosystems.

In deep snow country, where trees provide critical snow interception and thermal cover, optimum habitat may not be reached for 30 years or more after fire. During early stages of regrowth, diversity apparently improved through development of woody plants on grassland sites. Increased cover seems to have benefited mule deer *(Odocoileus hemionus)* and elk in marginal habitats previously lacking in cover.

But the absence of fire for 50 years or more, with subsequent conifer encroachment, canopy closure, and deterioration of herbs and shrubs, has resulted in deterioration of big game habitat. Loovas (1976) reported that fire suppression in the Black Hills of South Dakota resulted in thickening of pine stands and decreases in secondary stages of plant succession important to mule and white-tailed deer.

Small burns of variable intensity can improve deer habitat by creating temporary openings, improving shrub growth, and generally creating more diversity by changing the age class structure of vegetation (Wallmo 1981).

Prescribed fire has largely replaced herbicides in control and reduction of big sagebrush and stimulation of herbaceous plants (Gruell 1983). Such conversion has enhanced elk spring and winter ranges. Prescribed burning has also improved spring ranges used by mule deer. Some prescribed burns have short-term negative effects on mule deer habitat by removing big sagebrush, an important winter forage.

### Distribution and movements

Large mammals show an attraction to newly burned areas. On areas managed by prescribed burning, animals were seen moving several kilometers into burned areas. Pronghorn antelope were concentrated on burns, using areas they had not been seen in for many years (Klebenow and Beall 1978).

Bison in Wind Cave National Park in South Dakota showed a strong affinity for prescribed burn areas (Forde et al 1984). They fed within the confines of the Red Valley burn in 1981 and 1982 and moved to another area burned by wildfire in 1983. Their continued grazing may be important in delaying the normal progression of plant succession in the Red Valley.

Fire may affect the short- and long-term seasonal use of habitat by altering the distribution and movements of large mammals. Historically, it appears mule deer were largely confined to breaks and rough terrain where shrubs were protected from fires. White-tailed deer frequented riparian bottomlands that were less susceptible to frequent fire.

In Minnesota, Irwin (1975) showed white-tailed deer preferred the periphery and unburned forest in winter and spring and the burn area in summer and fall following a spring burn. Moose *(Alces alces)* selected the periphery of the burn in winter and open parts of the burn from May to September 2 years after the fire.

Observations prior to a May 1965 fire on the Nebraska National Forest indicated white-tailed deer utilized the unburned plantation areas over 80% of the time. Few deer were seen in the burned plantation area. Whitetails in the Sand Hills of Nebraska are essentially inhabitants of the tree-shrub community. Their use of the burned area was about 8% in 1965, and declined to about 5% the following year (Wolfe 1973).

Mule deer, in comparison, showed a very substantial response to the burned area. They are normally considered a deer of the prairie baseline. Observations in 1964 showed that mule deer utilized the prairie only slightly more than evergreen plantations (53% vs. 48%). After the 1965 fire, mule deer made about equal use of the burned and unburned plantation areas. During the same period, numbers of mule deer observed in the prairie declined substantially. By 1966, only about 28% of the mule deer observed were using the burned plantations.

Lowe et al (1978) studied long-term use of habitats by deer and elk after fire, finding deer summer-fall use declined the first year following fire but increased to levels approaching 2.5 times the control through the rest of the 20-year evaluation period. Deer winter-spring use also declined immediately following fire, returned to the control level for several years, and then increased to levels exceeding 20 times that of the control.

Deer winter-spring values reflected the relatively high use in the latter years of the evaluation period. Low winter-spring deer use on all areas except the 20-year-old burn indicated an annual shift to winter range as the summer range became increasingly less suitable.

The 20-year old burn was used more as winter range because it was relatively open and provided easy movement along the edge to and from nearby lower elevations.

Elk summer-fall use declined after fire, then increased to levels nearly three times the level of the control before dropping back at the end of the 20-year period. Elk winter-spring use was higher than the control throughout the entire evaluation period, with the highest recorded post-fire use 7 years after fire.

The relatively low elk summer-fall use 20 years after fire was due to unpredictable shifts in elk population centers, or to the fact that sheep used the 20-year-old burn for a few weeks in late spring and early summer. Elk remained on summer range as long as forage was available or the weather was tolerable. Higher grass production on the burned areas was sufficient to sustain at least seven times the elk use of the control during winter.

The size of a burn will affect habitat use. Klebenow and Beall (1978) found deer ranged 0.25 mile (0.4 km) into a burn, but forage use was concentrated at the edge within a 274-yd (250 m) range inside and outside of the burn.

On recent burns in a grass-forb succession stage, deer did not penetrate the burns (Klebenow and Beall 1978). Most deer sign was concentrated within 109 yd (100 m) or less of the burn edge in unburned woodland. On older burns (over 24 years) in a shrub dominated stage of succession, more deer pellet groups were found within the burn area away from the edge than within 55 yd (50 m) of the edge. Steep and broken topography substituted for tree cover in the older burns.

## Population density and reproduction

Fire adversely affects population densities of animals, principally by altering habitat and not by killing. The greatest number of deer will be produced by keeping the habitat in the early stages of plant succession by methods which include burning (Troester 1970). A patchy burn with about 20% unburned vegetation is most desirable for most wildlife species (Wright 1974). This leaves adequate cover for big game and a winter food supply.

Fire may provide a reproductive advantage for adapted species. Efficient use of a variety of several habitats suggests evolutionary adaptation to fire through genetic diversity.

Exclusion of fire through suppression programs tends to compress genetic diversity and reduce the ability of populations to respond to dramatic environmental changes (Martinka 1976).

Current habitat relationships of wintering elk reflect both adaptability and responsiveness to the spectrum of vegetation change associated with a fire program, particularly at an intermediate stage in post-fire faunal succession. Wintering elk populations responded to fire by expanding population levels, but at a rate less than biological potential. Expansions correlated directly with improving forage conditions. Mule deer population levels seemed favored by extensive shrub fields of early post-fire successional stages (Martinka 1976).

Fire stimulated the production of browse, which resulted in an increase in deer populations (Bendell 1974). An area opened by burning produced heavier deer. Does had a higher frequency of ovulation and more fawns at heel, and they wintered in better condition (Bendell 1974).

The increased nutritional quality of burned grasslands provides good summer range capable of carrying deer in good condition through the breeding season, a necessary requirement for maximum herd productivity. White-tailed deer on poor range showed ovulation rates 67% of those attained by deer on good range (Julander et al 1961).

A comparison of wildlife production on burned and unburned grassland on the Woodworth Study Area of North Dakota (Kirsch and Kruse 1973) found no white-tailed deer fawns on an unburned 124-acre (50 ha) plot, compared to four fawns each during the second growing season on burned plots of 135 and 121 acres (55 ha and 49 ha).

Vogl and Beck (1970) determined the summer density of white-tailed deer on a burned area 8 years after a major fire to be 2.4 times greater than on the unburned control area.

Ten years after fire, if there is no further burning, tree crowns close in, reduce browse supply, and result in a lowered carrying capacity and a deer population too large to be supported by the reduced food supply (Leopold et al 1947).

Fires, in general, increase the diversity of wildlife species as well as the population densities on most vegetation types, with some exceptions. An increased abundance of one species may reduce the number of other large mammals through interspecific competition (Bendell 1974). Mule deer, moose, and bighorn sheep abundance in Banff and Jasper national parks, Canada, declined after fires which encouraged grassland and shrubland habitat favorable to elk. The elk outcompeted the other species for food and shelter.

## Parasites and disease

After a fire, infestations of external and internal parasites may be lower, a benefit to large mammals.

Drew et al (1985) found prescribed spring burning in central Alberta reduced but did not eliminate the number of winter tick *(Dermacentor albicuptus)* larvae available in autumn.

The degree of tick control is dependent upon the habitat type being burned, weather conditions prior to the burn, and the fuel load on the burn site. The majority of ticks are found in the elevated foliage of shrubs in the spring. Hot, intense burning of the shrub layer during spring melt and leaf-out was the most effective in reducing the number of engorged female ticks. Autumn burns would reduce tick numbers in the larval stage, provided a slow, hot fire is maintained to ensure adequate burning of the duff layer. A decrease in the amount of winter forage available to ungulates would be a factor to consider in the use of autumn burning.

Seip and Bunnell (1985) found higher counts of lungworm larvae in feces from Stone's sheep *(Ovis canadensis)* that used alpine winter ranges in February than in feces from sheep using burned, subalpine range. In May, sheep on unburned range that had wintered on the alpine meadows had higher lungworm levels than sheep that had wintered on the burned, subalpine range.

## Forage and nutrition

The objective of burning has been to improve availability and palatability of forage by killing aerial stems and stimulating crown growth (Willms et al 1980). Deer displayed greatest preference for forage from the burned treatment and least preference for forage from the control in the spring following fall burning. The regenerating brush sprouts and seedlings following fire offer deer a palatable and nutritious diet (Dasmann et al 1968).

Ordinarily, after large burns the food supply exceeds demand, and large areas away from suitable cover receive little browsing pressure. In areas of light browsing the brush will rapidly grow back into dense stands. Lotan and Brown (1985) found small burns may concentrate ungulates and inhibit regeneration in browse species such as aspen.

Fire affects plant communities primarily through the nutritional content, quantity, and availability of forage. Hobbs and Spowart (1984) tested the hypothesis that prescribed burning improves the nutritional quality of the diets of mule deer and mountain sheep.

Prescribed burning increased the protein concentration and in vitro digestible organic matter (IVDOM) in winter but not spring diets of mountain sheep and mule deer feeding in grassland and mountain shrub communities.

Effects of burning on diet crude protein persisted for 2 years in both communities. Treatment effects on diet IVDOM lasted for 2 years in the mountain shrub area but were absent during the second year in grassland, possibly due to the less intense nature of fire in grassland which allowed quicker return to pre-burn conditions.

Hobbs and Spowart (1984) concluded fire substantially improved the winter diets of mountain sheep and mule deer in grassland and mountain shrub communities but caused only small changes in the quality of individual forages. Inferences based on forage studies alone may severely underestimate improvements in ungulate nutrition following burning.

Burning of big sagebrush and bluebunch wheatgrass increases bighorn sheep forage and decreases mule deer forage. The sheep prefer the grass in winter while mule deer prefer the sage. Thus, sheep competition is reduced (Peek et al 1979).

Hobbs and Swift (1985) found fire reduced range supplies of dry matter, metabolizable energy, and nitrogen in forages consumed by mule deer, primarily because of the large decrease in the standing crop of shrubs following burning.

Range food supply for mountain sheep was less strongly affected. Metabolizable energy and nitrogen remained the same, while dry matter declined following burning. Estimates of carrying capacity reflected these differences. Unburned areas could support more deer than burned areas, but burning had no effect on carrying capacity of mountain sheep. Burns tended to have more forage with high nutrient concentrations but less forage overall. Unburned habitat is superior to burned areas for supporting high densities of mule deer on a relatively low plane of nutrition.

Burning becomes a productive treatment when management objectives specify supporting fewer animals at higher diet quality levels.

**Forage quantity and availability**

Observed shifts in habitat preference or avoidance following fire are probably related to changes in food availability (Dills 1970; Lowe et al 1978).

Burning reduced litter and standing dead herbage, which increased the amount of green forage ungulates could find and consume (Hobbs and Spowart 1984).

Understory production decreased the first post-burn year in the Jackson Hole area, then increased to levels well above those on the unburned sites in the second and third post-burn years. On one site, second-year production of willow-herb *(Epilobium angustifolium)*, a species palatable to elk, was double that prior to burning (Lotan and Brown 1985).

Forbs, particularly annuals, were abundant 4 years following a burn. Up through 16 years there were significantly more forbs than in unburned sites. Only a 24-year-old burn had significantly more forbs, indicating this may be about as long a change could be expected.

Grasses appeared to respond later; 24-, 45- and 115-year old burns had the most grass basal area (Klebenow 1985). This would be beneficial for species such as elk.

Wydeven and Dahlgren (1983) found graminoids to be the major forage class eaten by elk in spring and summer. Forbs were the most important forage class consumed in fall and winter, along with some graminoids.

Controlled burning of aspen provides more browse for deer. Following a spring burn, aspen stem densities had increased from a few hundred per acre prior to the fire to greater than 25,000/A (10,000/ha) due to root sprouting. Prior to treatment, aspen was too tall for ungulates to reach. Two years after the burn a large supply of aspen was at a height that could be utilized (Gordon 1976). These burns appeared to inexpensively provide not only an increased food supply but also increased cover.

Fire can affect forage species utilized. Following a burn in Alberta, pronghorn antelope showed a higher use of spineless, burned cactus, a forage item usually sparsely consumed (Stelfox and Vriend 1977).

In summary, fire creates vegetative diversity and therefore enhances wildlife habitat. Optimum benefits occur where fire creates a mosaic pattern of burned and unburned vegetation which provides new growth of nutritional forages, seasonal habitats, and maintenance of vegetation in early stages of succession. Improved habitat and forage increases the carrying capacity of habitats for large mammals.

## Effects of burning on livestock

Early settlers of the Flint Hills region of Kansas discovered that cattle selected forage from burned range more readily than from unburned range. This discovery lead to the observation that steers gained weight faster by grazing on burned range.

The practice of grazing burned range changed when permanent fences were installed. Restricted movement of livestock, coupled with burning too frequently, caused changes in botanical composition of the forage and reduced livestock gains. This change in botanical composition is now recognized as a factor influencing range condition (Anderson et al 1970).

Eventually, the settlers linked the decreased production of forage and livestock to improper timing of the burns. Fire then became a management tool to maintain quality forage and increase livestock production (Anderson et al 1970; Rains et al 1975; Woolfolk et al 1973; Launchbaugh and Owensby 1978).

Although the settlers knew that increased livestock production could be obtained by proper burning of the range, it is unlikely that they completely understood the reasons. The "why" has since been researched (Arnold and Hill 1972; Ellis et al 1976; Goatcher and Church 1970) and documented as a function of palatability of the plant and preference by the animal.

The use of fire to increase livestock production is based on a recognition that forage growing after burning becomes more palatable and is preferred by livestock. A strong positive correlation between protein content and preference by cattle and sheep was illustrated by Leigh (1961).

The concepts of preference and palatability are very much interrelated and together help explain the concept of forage selection and why livestock congregate on burned sites.

### Influences of burning on production

Investigations of livestock growth performance on burned range date from before the 1940s. Most work has been done in Kansas, followed by Florida, Georgia, and Louisiana.

There is agreement in the literature that grazing burned range versus unburned range will increase weight gains or enhance the factors that would tend to lead to increased weight gains in livestock.

Improved weight gains of livestock have occurred when fertilization is combined with burning treatments (Woolfolk et al 1973). The combination produced greater weight gains than did burned-only treatments. However, current costs of agricultural fertilizers outdistance the benefit of increased production, precluding their use.

Gains in beef production on burned versus unburned range can be attributed to changes in diet selectivity and improved forage quality, according to studies on the Edward Plateau in Texas (McGinty et al 1983).

Grass contributed a higher percentage of the diet in burned than unburned paddocks during late spring and early summer. The pattern was reversed for late summer and fall, indicating the forb component contributed a greater percentage of the diet in unburned paddocks during early and late summer. The availability of forbs was found to be less in burned paddocks as a result of the fire.

Browse structure was about even for both treatments after midspring, although diets of the steers in the control paddocks had a higher browse content in early to midspring. Live plant material intake was greater in burned paddocks in midspring because forage became more accessible to livestock after the dead plant material was burned up. The significantly increased intake of green plant material dropped off after the June sampling date.

Mineral (ash) intake in steer diets from the burned paddocks was greater in the late summer and fall. The increased ash content of the diet corresponded with more use of prickly pear cactus by the steers. Fraps and Cory (1940) found that prickly pear is high in soluble ash, possibly explaining the high ash results reported by McGinty et al (1983).

Crude protein in the steer diets on the control paddocks was greater than on the burned paddocks in the fall. Clipped plots did, however, show increased protein in the burned paddocks, but this was not reflected in the steer diets because forbs were reduced by the burn (McGinty et al 1983).

Digestibility was generally higher over the grazing season for the burned paddocks, mostly due to the increased green plant intake in the spring and prickly pear consumption in the fall (McGinty et al 1983). Forage digestibility is important because more pounds of beef can be produced from easily digestible forage than from the same quantity of less digestible forage.

Burning increased cattle production and their preference for grasses, especially weeping lovegrass (Klett et al 1971). They also found winter burns to increase forage yields 14% and utilization by cattle 53%. Burning also more than doubled crude protein, from 3.6% on untreated plots to 7.6% on unfertilized burned plots.

Fertilizer appeared to have no effect on crude protein. When it was applied to burned and unburned areas, the increase of crude protein was the same as burning alone. Allen et al (1976) also found crude protein to increase with burning but that nitrogen fertilizer had no effect.

Allen et al (1976) found certain chemical composition changes in plants after burning. Dry matter, which usually increases during the growing season, was reduced. Ether extract increased with burning. Crude fiber decreased but was increased with fertilizer application. Nitrogen free extract was decreased by nitrogen fertilizer but increased as a result of burning. Ash increased with fertilizer while fire produced little effect. Cell wall constituents increased with age, but burning lowered these constituents and improved forage quality. Neither burning nor fertilizer had any significant effect on hemicellulose, which declined with maturity. Lignin, a compound that increases as digestibility decreases, was reduced by burning but increased with fertilizer.

Heifers in the burned paddocks gained weight in June through September, while controls gained in June, July, and September. During August the heifer gains were significantly lower on controls than on burned paddocks. During the 155-day grazing period the average daily gains of the burned treatments were significantly higher than on the control treatments (McGinty et al 1983).

Hilmon and Hughes (1965) reported cattle gains of 15-27 lb/A (17- 30 kg/ha) after burning forested range in Georgia and Florida. Greater palatability and production of forage were cited as the factors influencing these improved gains.

Greene (1929) reported 18 lb/A (20 kg/ha) gains on burned bluestem pasture in Mississippi with improved gains peaking 60 to 90 days post-burn. The increased live plant material was indicated to be the major cause of the improvements in the livestock gains.

Kirk and Hodges (1970) reported annual winter burning of half the range (in study pastures in Florida each year) increased the weaning crop percentage gain per calf from 9-12 lb/A (10-13 kg/ha) and gain per cow from 180-233 lb (82-106 kg) of body weight.

With burning, improved gains can be expected for steers, breeding heifers, cows, and calves. When to burn is important in achieving the desired gains and maintaining range condition for annual repeated livestock gain (Duvall and Whitaker 1964; Anderson et al 1970; Woolfolk et al 1973, 1975; Launchbaugh and Owensby 1978).

The added weight gain by the cows during the grazing season from improved forage quality or quantity could make a difference in the profit or loss statement at year's end. As McGinty et al (1983), Hilmon and Hughes (1965), and Kirk and Hodges (1970) have reported, the benefits from burning are achieved when increased forage quality is converted to meat and fiber by the animal. White and Currie (1983a) recorded increased quantity of forage which could lead to either increased livestock productivity through individual performance, or group performance through increased grazing capacity.

### Timing of burns

Timing of burns for improved gains in livestock was explored in Kansas by Anderson et al (1970); Woolfolk et al (1975); and Launchbaugh and Owensby (1978). Mid- to late spring burns provided maximum benefit to livestock. Steers had significant weight gains early in the growing season following burning (Anderson et al 1970).

Work done by Smith and Young (1959) on bluestem pastures in Kansas indicated midspring burning increased protein and mineral fractions within the plant. Halls et al (1952) reported increased phosphorous and protein content in forages on coastal plains forests with midspring burning.

Anderson et al (1970) and White and Currie (1983a) found that burning in spring is the best time to improve the quality of the forage for livestock. Appropriate stocking maximizes benefits from prescribed burning.

The decision to burn should be based on anticipated forage needs and on the forage species that dominate the pastures (White and Currie 1983a,b). Improved livestock gain is no real net gain at all if range condition is compromised.

Fire can rejuvenate a pasture by increasing the numbers of seedstalks and density of desired plants. Also, cattle find these burned pastures more desirable because plants are more palatable. Ranchers like this because the nutritive value of the plants is increased and cattle gain faster.

### Burning and management

Managing burned pastures, whether on season-long or rotational grazing treatments, requires management to maintain range condition or, if possible, improve condition and distribution of the livestock.

Duvall and Whitaker (1964) set up a rotation burning system for managing longleaf pine-bluestem ranges in Louisiana. The research was conducted over 6 years where each third of the unit was burned every 3 years. The other two thirds were "naturally deferred" (avoided) by the cattle for up to 2 years.

Cattle began grazing the burned subunit within 1-4 weeks, depending on regrowth of the forage. Grazing was heavy until late summer, and little selectivity was documented. The unburned subunits were used moderately in early spring, with declining use during late spring and summer. Utilization was equal in burned and deferred subunits during late August.

When fall flowering grasses reached the late boot stage the cattle selected the seedheads in great quantity until they began to shatter out. Cattle congregated once again on the burned subunit after the seedheads became dry and unpalatable. During the winter months, cattle grazed intermittently on the burned subunit, but did not remain for extended periods (Duvall and Whitaker 1964).

Cattle gained more weight with a rotation burning system throughout the grazing season. Cattle on unburned longleaf pine-bluestem pastures in Florida rarely gained weight before calves were weaned.

Cows nursing calves on the three-pasture 3-year rotation burning system were 57 lb (27 kg)/head heavier when calves were weaned in August than in April. After weaning, these cows put on an additional 9 lb (36 kg) of body weight (Duvall and Whitaker 1964).

Ethridge et al (1985) studied the economic feasibility of burning tobosagrass *(Hilaria mutica)* in Texas. Burns were conducted from 1968 to 1976 on seven sites on rolling plains throughout Texas. The estimated increase in tobosagrass production resulted in a $89/A ($36/ha) increase in livestock sales over a 5-year period.

The authors concluded that the added estimated potential returns from burning must be compared to the added cost of burning.

They also stated that the main environmental variable that restrained grass production was lack of rainfall during the growing season. Wright (1969) maintains that this problem can be avoided by burning in late March when soil moisture can be more adequately assessed (Ethridge et al 1985).

Costs of burning could include fire break construction, labor, retardant cost, liability and risk factors, and other costs, depending on each situation. The economic feasibility could vary with time among ranches and among pastures within ranches (Ethridge et al 1985). Each manager must determine the cost compared to the gain on an individual basis.

In summary, the literature on prescribed burning and its effects on livestock production present enough favorable results to justify the use of fire in range management over much of the NGP.

### Summary

Among the numerous fire publications, reports, bibliographies, and burning plans that we reviewed, only a small percentage provided information from well-designed research studies. Most of the literature was descriptive in nature rather than quantitative, and most of the research information was from short-term studies.

Much of the fire-effects literature specific to the NGP has been concentrated on soils, upland plants, and wildlife, particularly birds.

Topics greatly lacking in fire effects research and literature include insects, water quality, emergent aquatic plants, trees, big game, forage crops, and livestock range. These "empty spots" in fire-effects literature are in contrast to the published materials from forest and grassland areas in other parts of the U.S. and Canada where fire research has received greater emphasis in the past.

We would like to stress a very important point: the results of burning effects from different but similar plant communities in other parts of the country are not totally adaptable to the NGP. For example, a tallgrass prairie site in the 40-inch precipitation zone of Illinois will respond to fire much differently than a tall-grass prairie site in the 16-inch precipitation zone of southern Canada.

Much remains to be learned about the effects of fire on the abiotic and biotic components of the NGP. Burning for management and research should stress seasonality, frequency, intensity, and the interaction of these variables.

Fire research needs should also include better design of experiments and pre- and post-fire evaluations including but not limited to the following quantifiable parameters: soil moisture, fuel moisture, fuel amounts (loads) and distribution, soil temperature, weather measurements, fire intensity and behavior, costs and labor effectiveness, public acceptance, and particularly, long-term evaluations of post-burn effects on the flora and fauna (both domestic and native species).

Our intent has been to provide a descriptive review of fire effects on the grassland biome of the NGP with special emphasis on the use of fire for wildlife management.

Because our interpretation of the literature may differ from yours, we encourage you to study the orginal research (see EC 762 for additional references) before making your own interpretations.

## Literature cited

Ahlgren, C.E. 1966. Small mammals and reforestation following prescribed burning. J For 64:614-618.

Ahlgren, C.E. 1974. Effects of fires on temperate forests: North-central United States. Pp 195-223 IN T.T. Kozlowski and C.E. Ahlgren (eds), Fire and ecosystems. Academic Press, New York.

Ahlgren, I.F., and C.E. Ahlgren. 1960. Ecological effects of forest fires. Bot Rev 26:483-533.

Aldous, A.E. 1934. Effects of burning on Kansas bluestem pastures. Kansas AES Tech Bull 38, 65 pp.

Allen L.J., L.H. Harbers, R.R. Schalles, C.E. Owensby, and E.F. Smith. 1976. Range burning and fertilizing related to nutritive value of bluestem grasses. J Range Manage 29:306-308.

Ammann, G.A. 1957. Prairie grouse of Michigan. Michigan Dept Conserv Tech Bull, 200 pp.

Anderson, H.G., and A.W. Bailey. 1980. Effects of annual burning on grassland in the aspen parkland of east-central Alberta. Can J Bot 58:985-996.

Anderson, K.L., E.F. Smith, and C.E. Owensby. 1970. Burning bluestem range. J Range Manage 23:81-92.

Anderson, M.L., and A.W. Bailey. 1979. Effect of fire on a *Symphoricarpos occidentalis* shrub community in central Alberta. Can J Bot 57:2819-2823.

Anderson R.K. 1969. Prairie chicken responses to changing booming-ground cover type and height. J Wildl Manage 33:636-643.

Arefyeva, Z.N., and B.P. Kolesnikov. 1964. Dynamics of ammonia and nitrate nitrogen in forest soils of the Transurals at high and low temperatures. Sov Soil Sci 3:246-260.

Arnett, W.H. 1960. Responses of acridid populations to range land practices and range sites. Unpbl PhD dissertation. Kansas State Univ., Manhattan, 114 pp.

Arnold, G.W., and J.L. Hill. 1972. Chemical factors affecting selection of food plants by ruminants. Pp 72-99 IN J.B. Harborne (ed), Phytochemical ecology. Academic Press, London, LTD, 272 pp.

Bailey, A.W. 1976. Alberta's range land resources. Rangemans J. 3:44-46.

Bailey, A.W. 1978. Effects of fire on the mixed prairie vegetation. Proc. Prairie Prescribed Burning Symposium and Workshop, U.S. For Serv, Bur Land Manage, and USFWS, Jamestown, ND, 5 pp.

Bailey, A.W., and R.W. Wroe. 1974. Aspen invasion in a portion of the Alberta Parklands. J Range Manage 27:263-266.

Ball, J.P. 1984. Habitat selection and optimal foraging by mallards: a field experiment. MS thesis, Univ of Guelph, Ontario, 44 pp.

Banks, R.C., R.W. McDiarmid, and A.L. Gardner. 1987. Checklist of vertebrates of the United States, the U.S. Territories, and Canada. USFWS Resour Publ 166, 79 pp.

Barney, M.A., and N.C. Frischkneckt. 1974. Vegetation changes following fire in the pinyon-juniper type of west-central Utah. J Range Manage 27:91-96.

Beck, A.M., and R.J. Vogl. 1972. Effects of spring burning on rodent populations in a brush prairie savanna. J Mamm 53:336- 346.

Beetle, A.A., and K.L. Johnson. 1982. Sagebrush in

Wyoming. Wyoming AES Bull B-779, 68 pp.

Bendell, J.F. 1974. Effects of fire on birds and mammals. Pp 73-138 IN T.T. Kozlowski and C.E. Ahlgren (eds), Fire and ecosystems, Academic Press, New York.

Bent, A.C. 1923. Life histories of North American wildfowl. U.S. Nat Mus Bull, 126 pp.

Best, L.B. 1979. Effects of fire on a field sparrow population. Am Midl Nat 101:434-442.

Beule, J.D. 1979. Control and management of cattails in southeastern Wisconsin wetlands. Wisconsin Dept Nat Resour Tech Bull 112, 41 pp.

Bock, C.E., and J.H. Bock. 1978. Response of birds, small mammals and vegetation to burning sacaton grasslands in southeastern Arizona. J Range Manage 31:296-300.

Bock, C.E., and J.H. Bock. 1983. Responses of birds and deer mice to prescribed burning in ponderosa pine. J Wildl Manage 47:836-840.

Bock, J.H., and C.E. Bock. 1984. Effects of fire on woody vegetation in the pine-grassland ecotone of the southern Black Hills. Am Midl Nat 112:35-42.

Bragg, T.B. 1982. Seasonal variations in fuel and fuel consumption by fires in a bluestem prairie. Ecology 63:7-11.

Bragg, T.B., and L.C. Hulbert. 1976. Woody plant invasion of unburned Kansas bluestem prairie. J Range Manage 29:19-24.

Braun, C.E., T. Britt, and R.O. Wallestad. 1977. Guidelines for maintenance of sage grouse habitats. Wildl Soc Bull 5:99-106.

Bunting, S.C., H.A. Wright, and L.F. Neuenschwander. 1980. Long-term effects of fire on cactus in the southern mixed prairie of Texas. J Range Manage 33:85-88.

Burns, P.Y. 1952. Effects of fire on forest soils in the Pine Barren region of New Jersey. Yale Univ School of Forestry Bull 57, 50 pp.

Cancelado, R., and T.R. Yonke. 1970. Effect of prairie burning on insect populations. J Kans Entomol Soc 43:274-281.

Chew, R.M., B.B. Butterworth, and R. Grechman. 1958. Effects of fire on the small mammal populations of chaparral. J Mamm 40:235.

Christensen, N.L. 1976. Short-term effects of mowing and burning on soil nutrients in Big Meadows, Shenandoah National Park. J Range Manage 29:508-509.

Clark, R.G., C.M. Britton, and F.A. Sneva. 1982. Mortality of bitterbrush after burning and clipping in eastern Oregon. J Range Manage 35:711-714.

Clarke, S.E., E.W. Tisdale, and N.A. Skoglund. 1943. Effects of climate and grazing on shortgrass prairie vegetation. Canada Dept Agric Tech Bull 46, 53 pp.

Cluff, G.J., J.A. Young, and R.A. Evans. 1983. Edaphic factors influencing the control of Wyoming big sagebrush and seedling establishment of crested wheatgrass. J Range Manage 36:786-792.

Cook, S.F., Jr. 1959. Effect of fire on a population of small rodents. Ecology 40:102-108.

Coupland, R.T. 1973. Producers: I. Dynamics of above-ground standing crop. Matador Project, Canadian IBP Prog Tech Rep 27, Saskatoon, Saskatchewan, 159 pp.

Cross, D.H. 1983. Wildlife habitat improvement by control of *Phragmites communis* with fire and herbicide. MS thesis, CSU, Fort Collins, 81 pp.

Curtis, J.T., and M.L. Partch. 1948. Effect of fire on the competition between blue grass and certain prairie plants. Am Mid Nat 39:437-443.

Curtis, J.T., and M.L. Partch. 1950. Some factors affecting flower production in *Andropogon gerardi*. Ecology 31:488-489.

Dasmann, W., R. Hubbard, W.G. MacGregor, and A.E. Smith. 1968. Evaluation of the wildlife results from fuel breaks, browseways, and type conversion. Proc, Tall Timbers Fire Ecol Conf 7:179- 193.

Daubenmire, R. 1968. Ecology of fire in grasslands. Pp 209-266 IN J.B. Cragg (ed), Advances in ecological research, vol 5, Academic Press, New York.

DeBano, L.F., and C.E. Conrad. 1978. Effect of fire on nutrients in a chaparral ecosystem. Ecology 59:489-497.

DeBell, D.S., and C.W. Ralston. 1970. Release of nitrogen by burning light forest fuels. Proc, Soil Sci Soc Amer 34:936-938.

DeByle, N.V. 1976. Soil fertility as affected by broadcast burning following clearcutting in Northern Rocky Mountain larch/fir forests. Proc, Tall Timbers Fire Ecol Conf 14:447-464.

DeByle, N.V. 1985. Managing wildlife habitat with fire in the aspen ecosystem. IN J.E. Lotan and J.K. Brown (eds), Fire's effects on wildlife habitat, symposium proceedings. U.S. For Serv Gen Tech Rep INT-186, 96 pp.

DeJong, E., and K.B. MacDonald. 1975. Soil moisture regime under native grassland. Geoderma 14:207-221.

Dietz, D.R. 1972. Nutritive value of shrubs. Pp 289-310 IN C.M McKell, J.P. Blaisde, and J.R.

Goodin(eds), Wildland shrubs, their biology and utilization. U.S. For Serv, Intermt For Range Exp Stn Gen Tech Rep INT-1, 494 pp.

Diiro, B.W. 1982. Effects of burning and mowing on seasonal whitetop ponds in southern Manitoba. MS thesis, Iowa State Univ, Ames, 48 pp.

Dills, G.G. 1970. Effects of prescribed burning on deer browse. J Wildl Manage 34:540-545.

Dix, R.L. 1960. Effects of burning on the mulch structure and species composition of grasslands in western North Dakota. Ecology 41:49-56.

Dodd, J.L., J.D. Rodgers, and B.S. Mihlbachler. 1985. Effects of fall and spring burning on available forage production, soil water balance, and cactus mortality in high density cactus- rangelands of eastern Wyoming. Abstracts, 38th ann mtg, Soc Range Manage, Feb 11-15, 1985, Salt Lake City, UT, p 56.

Drew, M.L., W.M. Samuel, G.M. Lukiwski, and J.N. Willman. 1985. Evaluation of burning for control of winter ticks *(Dermacentor albicuptus)* in central Alberta. J Wildl Diseases 21:313-315.

Dunn, P.H., and L.F. DeBano. 1977. Fire's effect on the biological properties of chaparral soils. Pp 75-84 IN Proc, Int Symp, Environmental Consequences of Fire and Fuel Management in Mediterranean Climate Ecosystems (Forests and Scrublands), U.S. For Serv Gen Tech Rep WO-3.

Duvall, V.L., and L.B. Whitaker. 1964. Rotation burning: a forage management system for longleaf pine-bluestem ranges. J Range Manage 17:222-226.

Dwyer, D.D., and R.D. Pieper. 1967. Fire effects on blue grama-pinyon-juniper rangeland in New Mexico. J Range Manage 20:359- 362.

Dziadyk, B., and G.K. Clambey. 1980. Vegetation studies in the prairie-forest transition region: IV, Effect of burning on net production of a western Minnesota tall grass prairie. Proc ND Acad Sci 34:21.

Edwards, W.R., and J.A. Ellis. 1969. Responses of three avian species to burning. Wilson Bull 81:338-339.

Ehrenreich, J.H. 1959. Effect of burning and clipping on growth of native prairie in Iowa. J Range Manage 12:133-137.

Ehrenreich, J.H., and J.M. Aikman. 1957. Effect of burning on seedstalk production of native prairie grasses. Iowa Acad Sci 64:205-212.

Ehrenreich, J.H., and J.M. Aikman. 1963. Ecological study of the effect of certain management practices on native prairie in Iowa. Ecol Monogr 33:113-130.

Eichhorn, L.C., and C.R. Watts. 1984. Plant succession on burns in the river breaks of central Montana. Proc, MT Acad Sci 43:21- 34.

Ellis, J.A., W.R. Edwards, and K.P. Thomas. 1969. Responses of bobwhites to management in Illinois. J Wildl Manage 33:749-762.

Ellis, J.E., J.A. Wiens, C.F. Rodel, and J.C. Anway. 1976. Conceptual model of diet selection as an ecosystem process. J Theor Bio 60:93-108.

Engle, D.M., and P.M. Bultsma. 1984. Burning of northern mixed prairie during drought. J Range Manage 37:398-401.

Erwin, W.J., and R.H. Stasiak. 1979. Vertebrate mortality during the burning of a reestablished prairie in Nebraska. Am Midl Nat 101:247-249.

Ethridge, D.E., R.G. Sudderth, and H.A. Wright. 1985. Economic returns from burning tobasagrass in the Texas rolling plains. J Range Manage 38:362-365.

Evans, C.D., and K.E. Black. 1956. Duck production studies on the prairie potholes of South Dakota. USFWS Spec Sci Rep, Wildl 32, 59 pp.

Fala, R.A. 1975. Effects of prescribed burning on small mammal populations in a mixed-oak clearcut. J For 73:586-587.

Ferguson, E.R. 1957. Stem-kill and sprouting following prescribed fires in a pine-hardwood stand in Texas. J For 55:426-429.

Forde, J.D., N.F. Sloan, and D.A. Shown. 1984. Grassland habitat management using prescribed burning in Wind Cave National Park, South Dakota. Prairie Nat 16:97-110.

Fraps, G.S., and V.L. Cory. 1940. Composition and utilization of range vegetation of Sutton and Edwards counties. Texas AES Bull 586.

Fritzell, E.K. 1975. Effects of agricultural burning on nesting waterfowl. Can Field-Nat 89:21-27.

Gartner, F.R., and W.W. Thompson. 1973. Fire in the Black Hills forest-grass ecotone. Proc, Tall Timbers Fire Ecol Conf 12:37-68.

Gartner, F.R., and E.M. White. 1986. Fire in the northern Great Plains and its use in management. Pp 13-21 IN Proc, Prescribed Fire and Smoke Management Symposium, Feb 13, 1986 (Kissiminee, Fla), Soc Range Manage, Denver.

Gartner, F.R., J.R. Lindsey, and E.M. White. 1986. Vegetation responses to spring burning in western South Dakota. Proc, 9th N.A. Prairie Conf 9:143-146.

Gartner, F.R., R.I. Butterfield, W.W. Thompson, and L.R. Roath. 1978. Prescribed burning of range ecosystems in South Dakota. Proc, First Int Range Congr 1:687-690.

Gashwiler, J.S. 1970. Plant and mammal changes on a clearcut in west-central Oregon. Ecology 51:1018-1026.

Girard, M.M. 1985. Native wood land ecology and habitat type classification of southwestern North Dakota. PhD thesis, North Dakota State Univ, Fargo, 331 pp.

Glover, F.A. 1956. Nesting and production of the blue-winged teal in northwest Iowa. J Wildl Manage 20:28-46.

Glover, P.E. 1972. Nature, wildlife, and the habitat with a discussion on fires and other influences. Proc, Tall Timbers Fire Ecol Conf 11:319-336.

Goatcher, W.D., and D.C. Church. 1970. Review of some nutritional aspects of the sense of taste. J Anim Sci 31:973-981.

Gordon, F.A. 1976. Spring burning in an aspen-conifer stand of moose habitat, West Boulder River, Montana. Proc, Tall Timbers Fire Ecol Conf 14:501-538.

Gorenzel, W.P., R.A. Ryder, and C.E. Braun. 1981. American coot response to habitat change on a Colorado marsh. Southwest Nat 26:59-65.

Grange, W.B. 1949. Way to game abundance. Charles Scribner's & Sons, New York, 365 pp.

Great Plains Flora Association. 1986. Flora of the Great Plains. Uni Press of Kansas, Lawrence.

Greene, S.W. 1929. Stockman's interest in protecting forest and range from fire. For Long Proc 33:338-344.

Gruell, G.E. 1983. Fire and vegetative trend in the northern Rockies: interpretations from 1871-1982 photographs. U.S. For Serv Gen Tech Rep INT-158, 117 pp.

Hadley, E.B. 1970. Net productivity and burning responses of native eastern North Dakota prairie communities. Am Midl Nat 84:121-135.

Hadley, E.B., and R.P. Buccos. 1967. Plant community composition and net primary production within native eastern North Dakota prairie. Am Midl Nat 77:116.127.

Hadley, E.B., and B.J. Kieckhefer. 1963. Productivity of two prairie grasses in relation to fire frequency. Ecology 44:389- 395.

Halls, L.K., B.L. Southwell, and F.E. Knox. 1952. Burning and grazing in coastal plain forest. Georgia AES Bull 51, 33 pp.

Handley, C.O., Jr. 1969. Fire and mammals. Proc, Tall Timbers Fire Ecol Conf 9:151-159.

Hansen, P.L., G.R. Hoffmann, and A.J. Bjugstad. 1984. Vegetation of Theodore Roosevelt National Park, North Dakota: a habitat type classification. USDA For Serv, Rocky Mt For and Range Exp Stn, Gen Tech Rep RM-113, 35 pp.

Hardison, J.R. 1980. Role of fire for disease control in grass seed production. Plant Disease 64:641-645.

Harniss, R.O., and R.B. Murray. 1973. Thirty years of vegetal change following burning of sagebrush-grass range. J Range Manage 26:322-325.

Harvey, A.E., M.F. Jurgensen, and M.J. Larsen. 1976. Intensive fiber utilization and prescribed fire: effects on the microbial ecology of forests. U.S. For Serv, Intermt For and Range Exp Stn, Gen Tech Rep INT-28, 46 pp.

Hayes, W.P. 1927. Prairie insects. Ecology 8:238-250.

Heitlinger, M.E. 1975. Burning a protected tallgrass prairie to suppress sweetclover, *Melilotus alba* Desr. Pp 123-130 IN M.K. Wali (ed), Prairie: a multiple view, U of North Dakota Press, Grand Forks.

Hensel, R.L. 1923. Recent studies on the effect of burning on grassland vegetation. Ecology 4:183-188.

Hickey, V.G., and R.D. Ensign. 1983. Kentucky bluegrass seed production characteristics as affected by residue management. Agron J 75:107-110.

Higgins, K.F. 1986a. Interpretation and compendium of historical fire accounts in the Northern Great Plains. U.S. Fish & Wildlife Serv, Resouurc Publ 161. 39 pp.

Higgins, K.F. 1986b. Comparison of burn season effects on nesting birds in North Dakota mixed-grass prairie. Prairie Nat 18:219-228.

Hilmon, J.B., and R.H. Hughes. 1965. Fire and forage in the wiregrass type. J Range Manage 18:251-254.

Hobbs, N.T., and R.A. Spowart. 1984. Effects of prescribed fire on nutrition of mountain sheep and mule deer during winter and spring. J Wildl Manage 48:551-560.

Hobbs, N.T., and D.M. Swift. 1985. Estimates of habitat carrying capacity incorporating explicit nutritional constraints. J Wildl Manage 49:814-822.

Hopkins, H.F., W. Albertson, and A. Riegel. 1948. Some effects of burning upon a prairie in west central Kansas. Trans, Kansas Acad Sci 51:131-141.

Hopkins, H.H. 1954. Effects of mulch upon certain factors of grassland environment. J Range Manage 7:255-258.

Gartner, F.R., R.I. Butterfield, W.W. Thompson, and L.R. Roath. 1978. Prescribed burning of range ecosystems in South Dakota. Proc, First Int Range Congr 1:687-690.

Gashwiler, J.S. 1970. Plant and mammal changes on a clearcut in west-central Oregon. Ecology 51:1018-1026.

Girard, M.M. 1985. Native wood land ecology and habitat type classification of southwestern North Dakota. PhD thesis, North Dakota State Univ, Fargo, 331 pp.

Glover, F.A. 1956. Nesting and production of the blue-winged teal in northwest Iowa. J Wildl Manage 20:28-46.

Glover, P.E. 1972. Nature, wildlife, and the habitat with a discussion on fires and other influences. Proc, Tall Timbers Fire Ecol Conf 11:319-336.

Goatcher, W.D., and D.C. Church. 1970. Review of some nutritional aspects of the sense of taste. J Anim Sci 31:973-981.

Gordon, F.A. 1976. Spring burning in an aspen-conifer stand of moose habitat, West Boulder River, Montana. Proc, Tall Timbers Fire Ecol Conf 14:501-538.

Gorenzel, W.P., R.A. Ryder, and C.E. Braun. 1981. American coot response to habitat change on a Colorado marsh. Southwest Nat 26:59-65.

Grange, W.B. 1949. Way to game abundance. Charles Scribner's & Sons, New York, 365 pp.

Great Plains Flora Association. 1986. Flora of the Great Plains. Uni Press of Kansas, Lawrence.

Greene, S.W. 1929. Stockman's interest in protecting forest and range from fire. For Long Proc 33:338-344.

Gruell, G.E. 1983. Fire and vegetative trend in the northern Rockies: interpretations from 1871-1982 photographs. U.S. For Serv Gen Tech Rep INT-158, 117 pp.

Hadley, E.B. 1970. Net productivity and burning responses of native eastern North Dakota prairie communities. Am Midl Nat 84:121-135.

Hadley, E.B., and R.P. Buccos. 1967. Plant community composition and net primary production within native eastern North Dakota prairie. Am Midl Nat 77:116.127.

Hadley, E.B., and B.J. Kieckhefer. 1963. Productivity of two prairie grasses in relation to fire frequency. Ecology 44:389- 395.

Halls, L.K., B.L. Southwell, and F.E. Knox. 1952. Burning and grazing in coastal plain forest. Georgia AES Bull 51, 33 pp.

Handley, C.O., Jr. 1969. Fire and mammals. Proc, Tall Timbers Fire Ecol Conf 9:151-159.

Hansen, P.L., G.R. Hoffmann, and A.J. Bjugstad. 1984. Vegetation of Theodore Roosevelt National Park, North Dakota: a habitat type classification. USDA For Serv, Rocky Mt For and Range Exp Stn, Gen Tech Rep RM-113, 35 pp.

Hardison, J.R. 1980. Role of fire for disease control in grass seed production. Plant Disease 64:641-645.

Harniss, R.O., and R.B. Murray. 1973. Thirty years of vegetal change following burning of sagebrush-grass range. J Range Manage 26:322-325.

Harvey, A.E., M.F. Jurgensen, and M.J. Larsen. 1976. Intensive fiber utilization and prescribed fire: effects on the microbial ecology of forests. U.S. For Serv, Intermt For and Range Exp Stn, Gen Tech Rep INT-28, 46 pp.

Hayes, W.P. 1927. Prairie insects. Ecology 8:238-250.

Heitlinger, M.E. 1975. Burning a protected tallgrass prairie to suppress sweetclover, *Melilotus alba* Desr. Pp 123-130 IN M.K. Wali (ed), Prairie: a multiple view, U of North Dakota Press, Grand Forks.

Hensel, R.L. 1923. Recent studies on the effect of burning on grassland vegetation. Ecology 4:183-188.

Hickey, V.G., and R.D. Ensign. 1983. Kentucky bluegrass seed production characteristics as affected by residue management. Agron J 75:107-110.

Higgins, K.F. 1986a. Interpretation and compendium of historical fire accounts in the Northern Great Plains. U.S. Fish & Wildlife Serv, Resouurc Publ 161. 39 pp.

Higgins, K.F. 1986b. Comparison of burn season effects on nesting birds in North Dakota mixed-grass prairie. Prairie Nat 18:219-228.

Hilmon, J.B., and R.H. Hughes. 1965. Fire and forage in the wiregrass type. J Range Manage 18:251-254.

Hobbs, N.T., and R.A. Spowart. 1984. Effects of prescribed fire on nutrition of mountain sheep and mule deer during winter and spring. J Wildl Manage 48:551-560.

Hobbs, N.T., and D.M. Swift. 1985. Estimates of habitat carrying capacity incorporating explicit nutritional constraints. J Wildl Manage 49:814-822.

Hopkins, H.F., W. Albertson, and A. Riegel. 1948. Some effects of burning upon a prairie in west central Kansas. Trans, Kansas Acad Sci 51:131-141.

Hopkins, H.H. 1954. Effects of mulch upon certain factors of grassland environment. J Range Manage 7:255-258.

the Breckland avifauna. J Anim Ecol 2:239-262.

Launchbaugh, J.L. 1964. Effects of early spring burning on yields of native vegetation. J Range Manage 17:5-6.

Launchbaugh, J.L. 1973. Effect of fire on shortgrass and mixed prairie species. Proc, Tall Timbers Fire Ecol Conf 12:129-151.

Launchbaugh, J.L., and C.E. Owensby. 1978. Kansas rangelands: their management based on a half century of research. Kansas AES Bull 622, 56 pp.

Lawrence, G.E. 1966. Ecology of vertebrate animals in relation to chaparral fire in the Sierra Nevada foothills. Ecology 47:278-291.

Leedy, D.L. 1950. Ducks continue to nest after brush fire at Castalia, Ohio. Auk 67:234.

Leege, T.A., and W.O. Hickey. 1971. Sprouting of northern Idaho shrubs after prescribed burning. J Wildl Manage 35:508-515.

Leigh, J.H. 1961. Relative palatability of various varieties of weeping lovegrass *(Eragrostis curvula* (Schrad) Nees). J Brit Grassl Soc 16:135-140.

Leopold, A. 1941. Cheat takes over. Land 1:310-313.

Leopold, A., L.K. Sowls, and D.L. Spencer. 1947. Survey of over-populated deer ranges in the United States. J Wildl Manage 11:162-177.

Linde, A.F. 1969. Techniques for wetland management. Wisconsin Dept Nat Resour Rep 45, 156 pp.

Linde, A.F., T. Janisch, and D. Smith. 1976. Cattail: the significance of its growth, phenology and carbohydrate storage to its control and management. Wisconsin Dept Nat Resour Tech Bull 94, 27 pp.

Loovas, A.L. 1976. Introduction of prescribed burning to Wind Cave National Park. Wildl Soc Bull 4:69-73.

Lotan, J., and J.K. Brown. 1985. Proc, Fire's effects on wildlife habitat symposium, U.S. For Serv Gen Tech Rep INT-186, 96 pp.

Lowe, P.O., P.F. Ffolliot, J.H. Dieterick, and D.R. Patton. 1978. Determining potential wildlife benefits from wildfire in Arizona ponderosa pine forests. U.S. For Serv, Rocky Mtn For and Range Exp Stn Gen Tech Rep RM-52.

Lussenhop, J. 1976. Soil arthropod response to prairie burning. Ecology 57:88-98.

Martin, S.C. 1983. Responses of semidesert grasses and shrubs to fall burning. J Range Manage 36:604-610.

Martinka, C.J. 1976. Fire and elk in Glacier National Park. Proc, Tall Timbers Fire Ecol Conf 14:377-389.

Mathews, E.E. 1984. Fire on Montana's rangelands: a successful pilot program. Western Wildlands 10:16-19.

Mayland, H.F. 1967. Nitrogen availability on fall-burned oak-mountain mahogany chaparral. J Range Manage 20:33-35.

McGee, J.M. 1976. Some effects of fire suppression and prescribed burning on birds and small mammals in sagebrush. PhD thesis, Univ of Wyoming, Laramie, 145 pp.

McGee, J.M. 1982. Small mammal populations in an unburned and early fire successional sagebrush community. J Range Manage 35:177-180.

McGinty, A., F.E. Smeins, and L.B. Merrill. 1983. Influence of spring burning on cattle diets and performance on the Edwards Plateau. J Range Manage 36:175-179.

McMurphy, W.E., and K.L. Anderson. 1965. Burning Flint Hills range. J Range Manage 18:265-269.

Merrill, E.H., H.F. Mayland, and J.M. Peek. 1982. Shrub responses after fire in an Idaho ponderosa pine community. J Wildl Manage 46:496-502.

Messinger, R.D. 1974. Effects of controlled burning on waterfowl nesting habitat in northwest Iowa. MS thesis, Iowa State Univ, Ames, 49 pp.

Metz, L.J., T. Lotti, and R.A. Klawitter. 1961. Some effects of prescribed burning on Coastal Plain forest soils. U.S. For Serv, Southeastern For Exp Stn, Res Pap SE-133, 10 pp.

Millar, J.B. 1969. Observations on the ecology of wetland vegetation. Pages 49-56 IN Saskatoon Wetland Seminar, Can Wildl Serv Rep Ser 6.

Millar, J.B. 1973. Vegetation changes in shallow marsh wetlands under improving moisture regime. Can J Bot 51:1443-1457.

Miller, H.A. 1963. Use of fire in wildlife management. Proc, Tall Timbers Fire Ecol Conf 2:19-30.

Monsen, S.B., and J.N. Davis. 1985. Progress in the improvement of selected western North American rosaceous shrubs. Abstracts, 38th ann mtg, Soc for Range Manage, Feb 11-15, 1985. Salt Lake City, UT, p 52.

Mook, J.H., and J. van der Toorn. 1982. Influence of environmental factors and management on stands of *Phragmites australis*, II. Effects on yield and its relationship with shoot density. J Applied Ecol 19:501-517.

Motobu, D.A. 1978. Effects of controlled slash burning on the mountain beaver *(Aplodontia rufa rufa)*. Northwest Sci 52:92-99.

Moyle, J.B. (ed). 1964. Ducks and land use in Minnesota. Minn Dep Conserv Tech Bull 8, 140 pp.

Nagel, H.G. 1973. Effect of spring prairie burning on herbivorous and non-herbivorous arthropod populations. J Kans Entomol Soc 46:485-496.

Neckles, H.A., J.W. Nelson, and R.L. Pederson. 1985. Management of whitetop *(Scolochloa festucacea)* marshes for livestock forage and wildlife. Delta Waterfowl Wetlands Res Stn Tech Bull 1, Portage la Prairie, Manitoba, 12 pp.

Niemi, G.J. 1978. Breeding birds of burned and unburned areas in northern Minnesota. Loon 50:73-83.

Nord, E.C., and C.L. Countryman. 1972. Fire relations. Pp 88-97 IN C.M. McKell, J.P. Blaisdell, and J.R. Gordon (eds), Wildland shrubs, their biology and utilization, U.S. For Ser, Intermt For and Range Exp Stn Gen Tech Rep INT-1, 494 pp.

Nye, P.H. 1959. Some effects of natural vegetation on the soils of West Africa and their development under cultivation. UNESCO Abidjan Symposium, Tropical Soils and Vegetation, pp 59-63.

Ohr, K.M., and T.B. Bragg. 1985. Effects of fire on nutrient and energy concentration of five prairie grass species. Prairie Nat 17:113-126.

Old, S.M. 1969. Microclimate, fire and plant production in an Illinois prairie. Ecol Monogr 38:355-383.

Oswald, B.P., and W.W. Covington. 1983. Changes in understory production following a wildlife in southwestern ponderosa pine. J Range Manage 36:507-509.

Owensby, C.E., and K.L. Anderson 1967. Yield responses to time of burning in the Kansas Flint Hills. J Range Manage 20:12-17.

Owensby, C.E., and J.L. Launchbaugh. 1976. Controlling weeds and brush on rangeland. Kansas CES Bull L-465.

Owensby, C.E., and J.B. Wyrill, III. 1973. Effects of range burn ing on Kansas Flint Hills soil. J Range Manage 26:185-188.

Pauly, W.R. 1982. How to manage small prairie fires. Bull, Dane County Environmental Council and Dane Co. Highway and Transportation Dept, Madison, Wisc, 23 pp.

Pechanec, J.F., and A.C. Hull, Jr. 1945. Spring forage lost through cheatgrass fires. Nat Wool Grower 35(4):13.

Peek, J.M., R.A. Riggs, and J.L. Lauer. 1979. Evaluation of fall burning on bighorn sheep winter range. J Range Manage 32:430-432.

Peet, M., R. Anderson, and M.S. Adams. 1975. Effect of fire on big bluestem production. Am Midl Nat 94:15-26.

Pelton, J. 1953. Studies on the life history of *Symphoricarpos occidentalis* in Minnesota. Ecol Monogr 23:17-19.

Petersen, K.L., and L.B. Best. 1986. Diets of nestling sage sparrows and Brewer's sparrows in an Idaho sagebrush community. J Field Ornith 57:283-294.

Pickford, G.D. 1932. Influence of continued heavy grazing and of promiscuous burning on spring-fall ranges in Utah. Ecology 13:159-171.

Plummer, A.P., A.C. Hull, Jr., G. Stewart, and J.H. Robertson. 1955. Seeding rangelands in Utah, Nevada, southern Idaho, and western Wyoming. U.S. For Serv Agric Handb 71, 73 pp.

Pratt, D.C., and M.J. Andrews. 1981. Research in biomass/special energy crop production in wetlands. Pp 71-81 IN B. Richardson (ed), Selected proc, Midwest Conf on Wetland Values and Management, Freshwater Soc, Navarro, Minn.

Quinn, R.D. 1979. Effects of fire on small mammals in chaparral. Calif-Nev Wildl Trans, pp 125-133.

Rains, J.R., C.E. Owensby, and K.E. Kemp. 1975. Effects of nitrogen fertilization, burning, and grazing on reserve constituents of big bluestem. J Range Manage 28:358-362.

Raison, R.J. 1979. Modifications of the soil environment by vegetation fires, with particular reference to nitrogen transformations: a review. Plant and Soil 51:73-108.

Redmann, R.E. 1978. Plant and soil water potentials following fire in a northern mixed grassland. J Range Manage 31:443-445.

Renwald, J.D. 1977. Effect of fire on lark sparrow nesting densities. J Range Manage 30:283-285.

Rice, L. 1932. Effect of fire on prairie animal communities. Ecology 13:392-401.

Rickard, W.H. 1960. Distribution of small mammals in relation to the climax vegetation mosaic in eastern Washington and northern Idaho. Ecology 41:99-106.

Risser, P.G., E.C. Birney, H.D. Blocker, S.W. May, W.J. Parton, and J.A. Wiens. 1981. True prairie ecosystem. Hutchinson Ross Publishing Co, Stroudsburg, Penn, 557 pp.

Sadleir, R.M.F.S. 1965. Relationship between agonistic behaviour and population changes in the deer mouse, *Peromyscus maniculatus*

(Wagner). J Animal Ecol 34:331-352.

Schacht, W., and J. Stubbendieck. 1985. Prescribed burning in the Loess Hill mixed prairie of southern Nebraska. J Range Manage 38:47-51.

Schlichtemeier, G. 1967. Marsh burning for waterfowl. Proc, Tall Timber Fire Ecol Conf 6:40-46.

Schripsema, J.R. 1977. Ecological changes on pine-grassland burned in spring, late spring, and winter. MS thesis, South Dakota State Univ, Brookings, 99 pp.

Scotter, G.W. 1963. Effects of forest fires on soil properties in northern Saskatchewan. For Chron 39:412-421.

Seip, D.R., and F.L. Bunnell. 1985. Nutrition of Stone's sheep on burned and unburned ranges. J Wildl Manage 49:397-405.

Sexton, D.A., and M.M. Gillespie. 1979. Effects of fire on the location of a sharp-tailed grouse arena. Can Field-Nat 93:74-76.

Sharrow, S.H., and H.A. Wright. 1977a. Effects of fire, ash, litter on soil nitrate, temperature, moisture and tobasagrass production in the Rolling Plains. J Range Manage 30:266-270.

Sharrow, S.H., and H.A. Wright. 1977b. Proper burning intervals for tobasagrass in west Texas based on nitrogen dynamics. J Range Manage 30:343-346.

Simon-Sylvestre, G. 1967. Initial results of a survey on the effects of steam sterilization of soils. Ann Agron 18:243-266.

Sims, H.P., and C.H. Buckner. 1973. Effect of clear cutting and burning of *Pinus banksiana* forest on the population of small mammals in southeastern Manitoba. Am Midl Nat 90:228-231.

Smith, A.L. 1973. Production and nutrient status of whitetop. J Range Manage 26:117-120.

Smith, D.W. 1970. Concentrations of soil nutrients before and after fire. Can J Soil Sci 50:17-29.

Smith, E.F., and C.E. Owensby. 1973. Effects of fire on true prairie grasslands. Proc, Tall Timbers Fire Ecol Conf 12:9-22.

Smith, E.F., and V.A. Young. 1959. Effect of burning on the chemical composition of little bluestem. J Range Manage 12:139- 140.

Smith, L.M., and J.A. Kadlec. 1984. Effects of prescribed burning on nutritive quality of marsh plants in Utah. J Wildl Manage 48:285-288.

Smith, L.M., and J.A. Kadlec. 1985. Fire and herbivory in a Great Salt Lake marsh. Ecology 66:259-265.

Soutiere, E.C., and E.G. Bolen. 1973. Role of fire in mourning dove nesting ecology. Proc, Tall Timbers Fire Ecol Conf 12:277- 288.

Spurr, S.H., and B.V. Barnes. 1980. Forest ecology, third ed. John Wiley & Sons, New York, 687 pp.

Stark, N.M. 1979. Nutrient losses from timber harvesting in a larch/Douglas fir forest. U.S. For Serv, Intermt For and Range Exp Stn Res Paper INT-231, 41 pp.

Stark, R.H., J.L. Toevs, and A.L. Hafenrichter. 1946. Grasses and cultural methods of reseeding abandoned farm lands in southern Idaho. Idaho AES Bull 267.

Stelfox, J.G., and H.G. Vriend. 1977. Prairie fires and pronghorn use of cactus. Can Field-Nat 91:282-285.

Stewart, R.E., and H.A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. USFWS Resour Publ 92, 57 pp

Stoddard, H.L., Sr. 1963. Bird habitat and fire. Proc, Tall Timbers Fire Ecol Conf 2:163-175.

Stoddart, L.A., A.D. Smith, and T.W. Box. 1975. Range management, third ed. McGraw-Hill Book Co, New York, N.Y. 532 pp.

Stout, J., A.L. Farris, and V.L. Wright. 1971. Small mammal populations of an area in northern Idaho severely burned in 1967. Northwest Sci 45:219-226.

Svedarsky, W.D. 1979. Spring and summer ecology of female greater prairie chickens in northwestern Minnesota. PhD dissertation, Univ of North Dakota, Grand Forks, 166 pp.

Svedarsky, W.D., P.E. Buckley, and T.A. Fiero. 1986. Effects of 13 years of annual burning on an aspen-prairie ecotone in north western Minnesota. Pp 118-122 IN G.K. Clambey, W.C. Whitman, and R.H. Pemble (eds), Proc, 9th NA Prairie Conf, Moorhead, Minnesota.

Talent, L.G., G.L. Krapu, and R.L. Jarvis. 1982. Habitat use by mallard broods in south central North Dakota. J Wildl Manage 46:629-635.

Tester, J.R. 1965. Effects of a con trolled burn on small mammals in a Minnesota oak-savanna. Am Midl Nat 74:240-243.

Tester, J.R., and W.H. Marshall. 1961. Study of certain plant and animal interrelations on a native prairie in northwestern Minnesota. Univ of Minn Mus Nat Hist Occas Pap 8, 51 pp.

Tester, J.R., and W.H. Marshall. 1962. Minnesota prairie management techniques and their wildlife implications. Trans, NA Wildl Nat Res Conf 27:267-287.

Tevis, L. 1956. Effects of slash burning on forest mice. J Wildl Manage 29:405-409.

Thompson, D.J., and J.M. Shay. 1984. Effects of fire on *Phragmites australis* in the Delta Marsh, Manitoba. Can J Bot 63:1864-1869.

Thompson, D.Q. 1959. Biological investigations of the Upper Fox River. Wisconsin Conserv Dept Spec Wildl Rep 2, 41 pp.

Tiedmann, A.R., and T.D. Anderson. 1980. Combustion losses of sulfur from native plant materials and forest litter. Pp 22-227 IN Proc. 6th Conf on Fire and Forest Meterology, April 22-24, 1980, Soc Amer For, Washington, D.C, 6:220-227.

Tomanek, G.W. 1948. Pasture types of western Kansas in relation to the intensity of utilization in past years. Trans, Kans Acad Sci 51:171-196.

Towne, G., and C. Owensby. 1984. Long-term effects of annual burning at different dates in ungrazed Kansas tallgrass prairie. J Range Manage 37:392-397.

Truax, W.C., and L.F. Gunther. 1951. Effectiveness of game management techniques employed on Horizon Marsh. Trans, NA Wildl Conf 16:326-330.

Troester, H.G. 1970. Managed prairie burning for wildlife. North Dakota Outdoors 32:7-9.

Tryon, E.H. 1948. Effect of charcoal on certain physical, chemical, and biological properties of forest soils. Ecol Monogr 18:81-115.

Uhler, F.M. 1944. Control of undesirable plants in waterfowl habitats. Trans, NA Wildl Conf 9:295-303.

Vallentine, J.F. 1971. Range development and improvements, 2nd ed. Brigham Young Univ Press, Provo, Utah, 545 pp.

van der Toorn, J., and J.H. Mook. 1982. Influence of environmental factors and management on stands of *Phragmites australis:* I. Effects of burning, frost and insect damage on shoot densities and shoot size. J Applied Ecol 19:477-499.

Vareschi, V. 1962. La quema como factor ecologico en los llanos. Soc Venez Cienc Nat Bol 101:9-26.

Vlamis, J., and K.D. Gowans. 1961. Availability of nitrogen, phosphorus, and sulfur after brush burning. J Range Manage 14:38-40.

Vlamis, J., H.H. Biswell, and A.M. Schultz. 1955. Effects of prescribed burning on soil fertility in second growth young ponderosa pine. J For 11:905-909.

Vogl, R.J. 1965. Effects of spring burning on yields of brush prairie savanna. J Range Manage 18:202-205.

Vogl, R.J. 1967. Controlled burning for wildlife in Wisconsin. Proc, Tall Timbers Fire Ecol Conf 6:47-96.

Vogl, R.J. 1973. Effects of fire on the plants and animals of a Florida wetland. Am Midl Nat 89:334-347.

Vogl, R.J. 1974. Effects of fires on grasslands. Pp 139-194 IN T.T. Kozolowski and C.E. Ahlgren (eds), Fire and ecosystems, Academic Press, New York.

Vogl, R.J., and A.M. Beck. 1970. Response of white-tailed deer to a Wisconsin wildfire. Am Midl Nat 84:270-273.

Walker, T.W., and R. Thompson. 1949. Some observations on the chemical changes effected by the steam sterilization of glasshouse soils. J Hort Sci 25:19-35.

Wallmo, O.C. 1981. Mule and black-tailed deer of North America. Univ of Nebraska Press, Lincoln, 605 pp.

Ward, E. 1942. Phragmites management. Trans, NA Wildl Conf 7:294-298.

Ward, P. 1968. Fire in relation to waterfowl habitat of the Delta marshes. Proc, Tall Timber Fire Ecol Conf 8:254-267.

Warg, S.A. 1938. Life history and economic studies on *Bromus tectorum.* Unpubl MS thesis, Univ of Montana, Missoula, 38 pp.

Weaver, J.E., and N.W. Rowland. 1952. Effects of excessive natural mulch on development, yield, and structure of native grassland. Bot Gaz 114:1-19.

Weller, M.W. 1981. Freshwater marshes: ecology and wildlife management. Univ of Minnesota Press, Minneapolis, 146 pp.

Wells, C.G., R.E. Campbell, L.F. DeBano, C.E. Lewis, R.L. Fredickson, E.C. Franklin, R.C. Froelich, and P.H. Dunn. 1979. Effects of fire on soils: a state-of-knowledge review. U.S. For Serv Gen Tech Rep WO-7, 34 pp.

Wells, P.F. 1970. Postglacial vegetational history of the Great Plains. Science 167:1574-1582.

Westemeier, R.L. 1973. Prescribed burning in grassland managment for prairie chickens in Illinois. Proc, Tall Timbers Fire Ecol Conf 12:317-338.

White, A.S. 1983. Effects of thirteen years of annual prescribed burning on a *Quercus ellipsoidalis* community in Minnesota. Ecology 64:1081-1085.

White, E.M., and F.R. Gartner. 1975. Immediate effects of prairie fire on the soil nitrate, ammonium, available phosphorous and total N content. Proc, SD Acad Sci 54:188-193.

White, E.M., W.W. Thompson, and F.R. Gartner. 1973. Heat effects on nutrient release from soils under ponderosa pine. J Range Manage 26:22-24.

White, R.S., and P.O. Currie. 1983a. Prescribed burning in the northern Great Plains: yield and cover responses of three forage species in the mixed grass prairie. J Range Manage 36:179-183.

White, R.S., and P.O. Currie. 1983b. Effects of prescribed burning on silver sagebrush. J Range Manage 36:611-613.

Williams, O. 1959. Food habits of the deer mouse. J Mamm 40:415-419.

Willms, W., A.W. Bailey, A. McLean, and R. Tucker. 1980. Effects of fall grazing or burning bluebunch wheatgrass range on forage selection by deer and cattle in spring. Can J Anim Sci 80:113- 122.

Winter, B.M. 1984. Effects of prescribed burning on avian foraging ecology and arthoropod abundance in sagebrush-grassland. MS thesis, Iowa State Univ, Ames, 82 pp.

Winter, B.M. and L.B. Best. 1985. Effect of prescribed burning on placement of sage sparrow nests. Condor 87:294-299.

Wolfe, C.W. 1973. Effects of fire on a sand hills grassland environment. Proc, Tall Timbers Fire Ecol Conf 12:241-255.

Woolfolk, J.S., C.E. Owensby, L.H. Harbers, R.R. Schalles, L.J. Allen, and E.F. Smith. 1973. Response of yearling steers to burning, fertilization, and intensive early stocking of bluestem pasture. Kansas AES Bull 568:10-11.

Woolfolk, J.S., E.F. Smith, R.B. Schalles, B.E. Brent, L.H. Harbers, and C.E. Owensby. 1975. Effects of nitrogen fertilization and late-spring burning of bluestem range on diet and performance of steers. J Range Manage 28:190-193.

Worcester, L.L. 1979. Effects of prescribed burning at different fuel moisture levels on vegetation and soils of grasslands in Wind Cave National Park. MS thesis, South Dakota State Univ, Brookings, 101 pp.

Wright, H.A. 1969. Effect of spring burning on tobasagrass. J Range Manage 2:425-427.

Wright, H.A. 1972. Shrub response to fire. IN Wildland shrubs, their biology and utilization. An Int Symp, Utah State Univ, Logan, U.S. For Serv, Intermt For and Range Exp Stn Gen Tech Rep INT-1, 494 pp.

Wright, H.A. 1974. Range burning. J Range Manage 27:5-11.

Wright, H.A. 1980. Role and use of fire in the semidesert grass-shrub type. U.S. For Serv, Intermt For and Range Exp Stn Gen Tech Rep INT-85, 24 pp.

Wright, H.A., and A.W. Bailey. 1980. Fire ecology and prescribed burning in the Great Plains--a research review. U.S. For Serv Gen Tech Rep INT-77, 60 pp.

Wright, H.A., and A.W. Bailey. 1982. Fire ecology: United States and southern Canada. John Wiley and Sons, Inc, New York, 501 pp.

Wright, H.A., L.F. Neuenschwander, and C.M. Britton. 1979. Role and use of fire in sagebrush-grass and pinyon-juniper plant communities. U.S. For Serv, Intermt For and Range Exp Stn Gen Tech Rep INT-58, 48 pp.

Wydeven, A.P., and R.B. Dahlgren. 1983. Food habits of elk in the northern Great Plains. J Wildl Manage 47:916-923.

Yancey, R.K. 1964. Matches and marshes. Pp 619-626 IN J.P. Linduska (ed), Waterfowl tomorrow, U.S. Government Printing Office, Washington, D.C.

Young, J.A., R.A. Evans, and R.A. Weaver. 1976. Estimating potential downy brome competition after wildfires. J Range Manage 29:332-325.

Young, J.A., and R.A. Evans. 1974. Population dynamics of green rabbitbrush in disturbed big sagebrush communities. J Range Manage 27:127-132.

Youngberg, C.T. 1953. Slash burning and soil organic matter maintenance. J For 51:202-203.

Zelder, J., and O.L. Loucks. 1969. Differential burning response of *Poa pratensis* fields and *Andropogon scoparius* prairies in central Wisconsin. Am Midl Nat 81:341-352.

Zimmerman, G.M. 1981. Effects of fire upon selected plant communities in the Little Missouri Badlands. MS thesis, North Dakota State Univ, Fargo, 60 pp.

## APPENDIX A. Topics on which students assisted on literature search, 1986.

**Waterfowl - ducks/shorebirds**
Chuck Dieter
**Passerine birds**
Mark Lesinski
**Upland game birds**
Steve Riley
**Trees/forest**
Scott Larson
**Small rodents**
Tom Easterly
**Insects**
Ken Fischer
**Large mammals**
Steve Hirtzel
**Wetlands**
Brian Wangler
**Soil nutrients and minerals**
Scott Laudenslager
**Native brush and shrubs**
David George
**Noxious weeds, reproductive shoots**
Lawrence Kreil
**Plant height; density and nutrition**
Chad Hansen
**Range livestock productivity**
Kevin Peterson
**Exotic grass species**
Jeff Loof
**Tall grass prairie productivity**
Roger Knapp
**Mid (mixed) grass prairie productivity**
Mike Cox
**Short grass prairie productivity**
Jim McKee
**Species composition changes**
Kevin King
**Africa**
Asse'tou Kanoute'

Issued in furtherance of Cooperative Extension work, Acts of May 8 and June 30, 1914, in cooperation with the USDA. Larry Tidemann, Director of Extension, Associate Dean, College of Agriculture & Biological Sciences, South Dakota State University, Brookings. Educational programs and materials offered without regard for race, color, creed, religion, national origin, ancestry, citizenship, age, gender, sexual orientation, disability, or Vietnam Era Veteran status.
EC 761: 750 reprinted at $1.67 each from RREA funds July 2000. (ES520)