ECOLOGY OF FIRE IN SHORTGRASS PRAIRIE COMMUNITIES OF THE KIOWA NATIONAL GRASSLAND

Paulette L. Ford*
Rocky Mountain Forest and Range Experiment Station
2205 Columbia SE
Albuquerque, NM 87106

Guy R. McPherson
University of Arizona
School of Renewable Natural Resources
325 Biological Science East Building
Tuscon, AZ 85721

*Corresponding author

ABSTRACT: Although recurrent fires were historically prevalent natural components of grassland ecosystems, the incidence of fire in grasslands has greatly declined since the 1800s. This phenomenon is largely attributed to reduction of fuel from grazing by domestic livestock and active fire suppression. The use of prescribed fire has increased recently, primarily as a management tool to control invasion of woody plants into grasslands and to increase productivity of rangelands. Though there has been considerable descriptive research on the effects of wildfire on plant cover and productivity, the effects of prescribed burns on the shortgrass prairie ecosystem are not well known. Experimental research in the southern Great Plains on the Kiowa National Grassland in northeastern New Mexico, is an example of the kind of research needed to address the effects of season and frequency of fire in shortgrass prairie.

Key words: shortgrass steppe, experimental fire research, fire ecology, animal response to fire

INTRODUCTION

Environmental disturbance creates gaps in communities and allows recolonization of the gaps by individuals of the same or different species. Disturbance by fire plays a role in structuring most wildland plant communities, and the role of fire in community organization and development of grassland ecosystems is widely acknowledged (McPherson 1995). Grassland communities are likely to be differentially influenced by fire due to evolutionary adaptations to particular habitat features and conditions (e.g., precipitation patterns), the current physical and biological environment, and present and past land-use patterns.

Over a century ago, expansion of cultivated areas and removal of available fuel by livestock grazing contributed to post-settlement (i.e., post-1800s) decline in fire frequency, altered fire regimes, and extreme vegetation change in the Great Plains. Reliable historical records of fire frequencies in prairies of the southern Great Plains are not available because there are no trees to carry fire scars from which to estimate fire frequency. However, reconstruction of fire history via examination of charcoal fragments from lake sediment cores indicates that post-settlement patterns of charcoal deposition were highly variable but generally much lower than during presettlement intervals (Umbanhowar 1996). This suggests that settlement decreased the number of fires, and this parallels findings from historical accounts of desert grasslands (Bahre 1991).

In the absence of periodic fires, most grasslands give way to dominance by woody plants. Fire interacts with numerous other factors, including topography, soil, insects, herbivores (rodents, lagomorphs), and herbaceous plants to restrict woody plant establishment in grasslands (Grover and Musick 1990, McPherson 1995, Wright and Bailey 1982). Currently, there is general agreement that fire is necessary (though usually not sufficient) to control the abundance of woody plants and maintain most grasslands (McPherson 1995). The use of fire as a management tool has increased, but the question of how fire affects rangelands has not been fully addressed (McPherson 1995, Steuter and McPherson 1995). As a result, the extent and duration of fire’s impact on grassland communities in the southern Great Plains is largely unknown.

NATURAL HISTORY

Southern Great Plains: Shortgrass Prairie

The southern Great Plains include the eastern third of New Mexico, the northern two-thirds of Texas, and most of Oklahoma. The region can be divided into shortgrass,
mixed, and tallgrass prairies (Figure 1). Within the area, the shortgrass prairie lies west of the 100th meridian (Wright and Bailey 1982). It is estimated that less than 23% of true shortgrass prairie still exists in native vegetation (National Grasslands Management Review [NGMR] 1995). The grassland is semi-arid, with annual precipitation between 38 and 51 cm (15 and 20 inches).

Except for the sandy soils in southeastern New Mexico and the Canadian River country in northern Texas and western Oklahoma, soils are primarily clay loams, silt loams, and sandy loams. A caliche layer is frequently present at 51 to 91 cm (20 to 36 inches) in the fine-textured soils. Most of the area is tableland that is 1200 to 1829 m (4000 to 6000 ft) in elevation (south to north) on the western edge, and it slopes eastward to 915 m (3000 ft) on the edge of Llano Estacado in Texas. Dominant grasses are buffalograss (Buchloe dactyloides (Nutt.) Engelm.) and blue grama (Bouteloua gracilis (H.B.K.) Lag. ex Steud.), with varying amounts of threeawns (Aristida spp. L.), lovegrass (Eragrostis spp. Beauv.), tridens (Tridens spp. Roem. and Schult.), sand dropseed (Sporobolus cryptandrus (Torr.) A. Gray), sideoats grama (Bouteloua curtipendula (Michx.) Torr.), tobasgrass (Hilaria mutica (Buckl.) Benth.), galleta (H. jamesii (Torr.) Benth.), vine-mesquite (Panicum obtusum H.B.K.), bush muhly (Muhlenbergia porteri Scribn.), and Arizona cottontop (Trichachne californica (Benth.) Chase) (Bailey 1995, Wright and Bailey 1982).

Forbs can be abundant during wet years but they are seldom a major component of the shortgrass prairie. Common forbs include annual broomweed (Xanthocephalum dracunculoides DC.), false mesquite (Hoffmanseggia densiflora Benth.), western ragweed (Ambrosia psilostachya DC.), horsetail conyza (Erigeron canadensis L.), warty euphorbia (Euphorbia spathulata Lam.), silver-leaf night shade (Solanum elaeagnifolium Ca.), many system evax (Evax multicaulis DC.), woolly plantago (Plantago purshii Roem. and Schult.), dozedaisy (Aphanostephus spp. DC.), goosefoot (Chenopodium spp. L.), croton (Croton spp. L.), summer cypress (Kochia scoparia L. Schrad.), and globemallow (Sphaeralcea spp. St. Hil.) (Wright and Bailey 1982).

**Figure 1.** Natural vegetation of Great Plains grasslands (Wright and Bailey 1980). Modified from Kuchler (1965) and Rowe (1972). DMG = Dry mixed grassland; F = Forest; FG = Fescue grassland; GF = Grassland forest; MG = Mixed grassland; MG-J = Mixed grassland-juniper-oak; MG-M = Mixed grassland-mesquite; SG = shortgrassland; TG = Tall grassland; * Kiowa study site.
Dominant woody plants are honey mesquite, sand shinnery oak (Quercus havardii Rydb.), sand sagebrush (Artemisia filifolia Torr.), perennial broomweed (Gutierrezia sarothrae (Pursh) Britt. & Rusby), yucca (Yucca spp. L.), and fourwing saltbush (Atriplex canescens (Pursh.) Nutt.). Cactus (Opuntia spp. Miller) can also be abundant, particularly prickly pear (Opuntia polyacantha Haw.), brownspine pricklypear (O. phaeacantha Engelm.), walkingstick cholla (O. arborescens Engelm.), and tasajillo (O. leptocaulis DC.) (Wright and Bailey 1982).

**Ecosystem Response to Fire**

Considerable descriptive research (e.g., Launsebaugh 1964, Dwyer and Pieper 1967) was conducted to address fire effects on shortgrass prairie vegetation before 1980 (reviewed by Ford and McPherson 1996). These descriptive studies indicate that fire leads to decreased herbaceous production for one to three years, and herbaceous response is influenced strongly by precipitation. Fires also contribute to reductions in woody plant cover and increases in density and diversity of herbaceous dicots. In general, plant species in semi-arid grasslands are more strongly influenced by fire season and frequency than fire behavior (Steuter and McPherson 1995).

There is a lack of research that directly addresses the effects of fire on animal communities in shortgrass prairie. Arthropods and mammals play important roles in ecosystem functioning of shortgrass prairie, serving as decomposers, pollinators, herbivores, predators, or prey. They cycle nutrients and form valuable links among trophic levels. Numerous studies in other ecosystems have indicated animal species, populations, and communities respond differentially to disturbance by fire, due in part to the fact that fire can have both direct and indirect effects. Direct effects are acute but ephemeral (i.e., fire-induced mortality). Indirect effects (i.e., alterations in habitat) are long-lasting and usually more important. Therefore, grassland fires may directly or indirectly elicit changes in population or community structure, and the magnitude of these changes depends on the vagility, life history and trophic level of the animal, and the timing, extent, and intensity of the fire (Ford and McPherson 1996).

Grassland burning elicits a diverse array of responses by arthropods. The degree of modification of arthropod populations by fire, the direction of change, and whether the effects are acute or chronic vary with several factors including fire characteristics, arthropod species, timing of the burn relative to phenological stage of arthropod development, influence of the fire on predator/prey and parasite/host ratios, post-burn weather, and the direction and degree of habitat restructuring. Responses of arthropods to season and frequency of fire also appear to vary between species (Warren et al. 1987).

The reaction of mammals to fire is a function of size and vagility. Most small mammals escape fires by hiding in burrows or rock crevices (Howard et al. 1959, Heinselman 1973). The most common cause of death for small mammals during fire is a combination of heat effects and asphyxiation. However, studies cited by Bendell (1974) indicate that soil provides insulation from fire for burrowing animals (Krampl et al. 1983). Other causes of death include physiological stress as mammals overexert themselves to escape, trampling as large mammals stampede, and predation as small mammals flee from fire (Kaufman et al. 1990).

Grassland fires that temporarily remove food and cover (litter and standing dead vegetation) may be detrimental to small rodents immediately after fire (Daubenmire 1968, Kaufman et al. 1990). However, repopulation of such areas is reported to be nearly complete within six months after the fire (Cook 1959). Mice and rodent populations often increase after fire in response to increased availability of forb seeds and insects (Lyon et al. 1978). In addition, burned areas often support more diverse animal populations than comparable unburned sites. This may be a result of habitat diversity (Beck and Vogt 1972, Wirtz 1977). Omnivores and carnivores are attracted to burns by increased plant diversity and associated small mammal populations (Gruell 1980). Levels of animal parasites are often lower in burned habitats (Bendell 1974).

Kaufman et al. (1990) suggest that most effects of fire on small mammals in grasslands are not neutral, but are instead either fire-positive or fire-negative. Fire-negative mammals include species that forage on invertebrates in the litter layer, species that live in relatively dense vegetation and eat plant foliage, and species that use, at least partially, aboveground nests of plant debris.

Fire-positive mammals include species that use ambulatory locomotion in microhabitats with a relatively open herbaceous layer, feed on seeds and/or insects, and use saltatorial locomotion (Kaufman et al. 1990). They exhibit an increase in populations and habitat use after fire because of an increased availability of forb seeds, insects, newly greening vegetation, the creation of open areas in otherwise dense habitat, and an eventual increase in forb cover. Increases may occur immediately or gradually as the areas begin to revegetate and habitat diversity increases.

No studies have focused on the issue of seasonal effects of fire on small mammals. Since most of the effects of season on population responses will undoubtedly be more subtle than general fire-negative and fire-positive responses, studies of differences in effects of grassland fires on small mammal populations will require intensive, replicated studies (Kaufman et al. 1990). Numerous studies have examined the response of small mammals in
spring and autumn or spring and winter burn plots (Bock and Bock 1978, 1983; Bock et al. 1976; Tester and Marshall 1961); however these analyses focused on only the general effects of fire on small mammals, and no effects of season were evident (Kauffman et al. 1990).

Research

Ongoing research on the Kiowa National Grassland in northeastern New Mexico (Figure 1), uses a long-term (18 years) experimental framework to analyze the effects of season and frequency of fire on vegetation, small mammal, and arthropod communities in shortgrass prairie. Most previous research on the effects of fire in shortgrass prairie has not employed the experimental approach, but instead, has relied on study designs that are largely descriptive in nature.

Descriptive research is suitable for identifying patterns, but is considerably less useful for determining underlying mechanisms. This type of research has limited predictive power and, consequently, limited utility to managers (McPherson 1997, Weltzin and McPherson 1995). Manipulative field-based experimental research will help disentangle important driving variables because of strong correlations between factors under investigation (Gurevitch and Collins 1994). Identification of underlying mechanisms of change in community structure will enable researchers to predict community response to changes in fire or climate with a level of certainty useful to management (McPherson and Weltzin in press).

Preliminary analyses of short-term data from the Kiowa study have indicated that the shortgrass prairie ecosystem recovers relatively quickly from disturbance by fire. For example, vegetation cover and arthropod and mammal species richness on 2-ha experimental units treated with dormant-season fire recovered in approximately two months and did not significantly differ from untreated units. Some examples of research needs that can be addressed with long-term data from the Kiowa study are: the evaluation of the population responses of arthropods and mammals to prairie restoration using prescribed fire; identification of plant and animal species that are fire-dependent, neutral, or exhibit positive or negative responses to fire; evaluation of length of time after fire before positive or negative responses are produced; evaluation of the use of prescribed fire to benefit sensitive, threatened, or endangered plant and animal species; and determination of whether fire suppression or differences in season and frequency of prescribed burns will continue to contribute to population declines of some species (Ford and McPherson 1996).

Conclusions

The effects of fire on animal community structure in grasslands are related to trophic relationships and plant community structure. Conceivably, the effects of fire on arthropods will carry over to birds and small rodents that rely on arthropods as their prey base. This in turn will affect larger mammals and raptors. These relationships change rapidly as vegetation establishes and grows in recently-burned areas. Thus, community structure is likely to be temporally dynamic (Ford and McPherson 1996).

Change is the normal course of events for most ecological systems (Connell and Sousa 1983), and management of ecosystems is challenging in part because we seek to understand and manage areas that change (Christensen et al. 1996). Knowledge of plant and animal responses to fire timing and frequency may allow scientists and resource managers to predict the effects of prescribed burns on ecosystems.

Acknowledgments

We thank the Kiowa/Rita Blanca National Grasslands Ranger District Office and the Clayton Livestock Research Center for supplying support and housing for this research, and the anonymous reviewer.

Literature Cited


